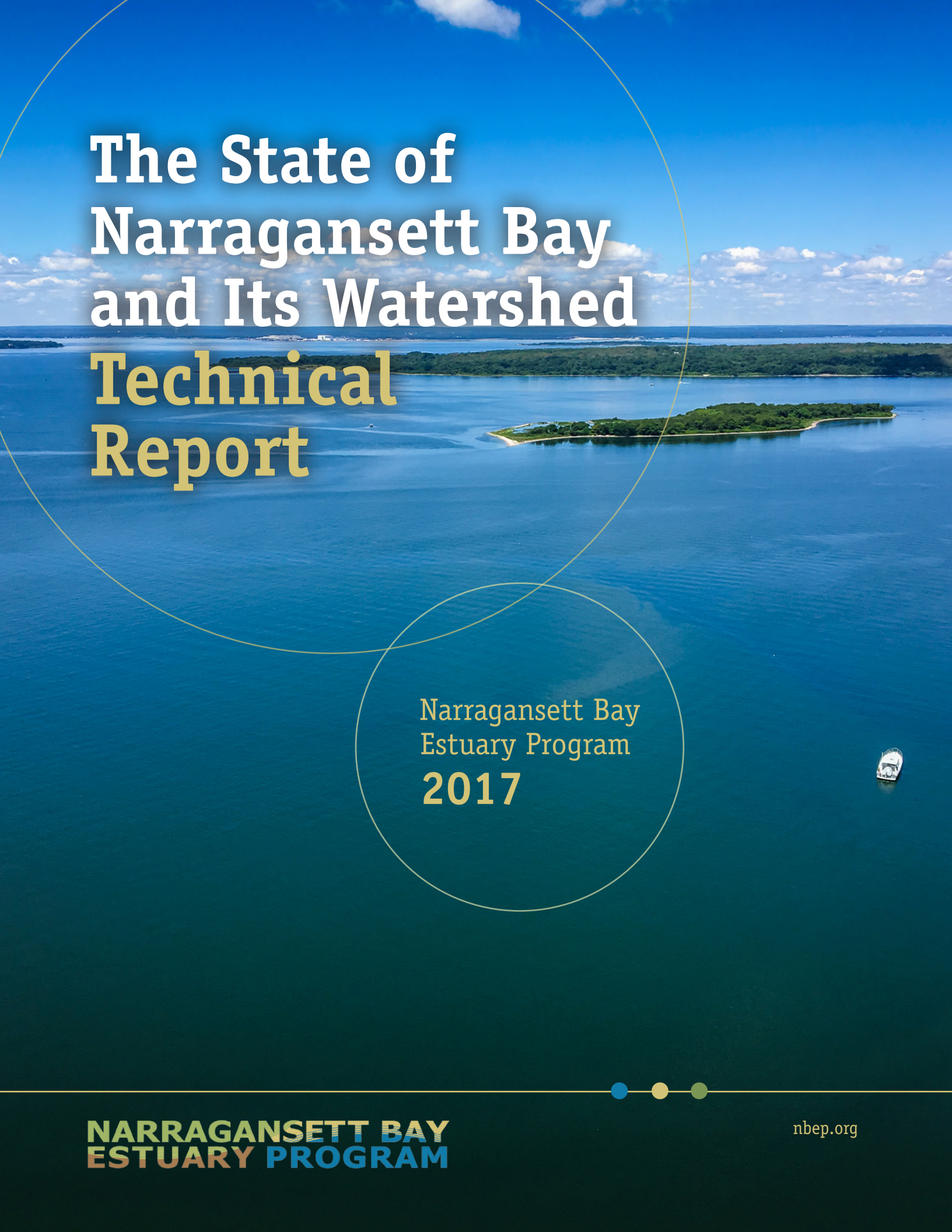


The State of Narragansett Bay and Its Watershed

Technical Report

Narragansett Bay Estuary Program

2017

NARRAGANSETT BAY ESTUARY PROGRAM

nbep.org

The *State of Narragansett Bay and Its Watershed* Technical Report
Narragansett Bay Estuary Program, 2017

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Cover photo: Aerial view of Narragansett Bay from Portsmouth, RI (Ayla Fox)
Back cover photo: Blackstone River Gorge, Blackstone, MA (Ayla Fox)



State of Narragansett Bay and Its Watershed
2017 Technical Report

EXECUTIVE SUMMARY

Please use the following citation: Narragansett Bay Estuary Program. 2017. State of Narragansett Bay and Its Watershed (Executive Summary, pages 3-6). Technical Report. Providence, RI.

Photo: Casey Farm, Saunderstown, RI (Ayla Fox)

NARRAGANSETT BAY ESTUARY PROGRAM

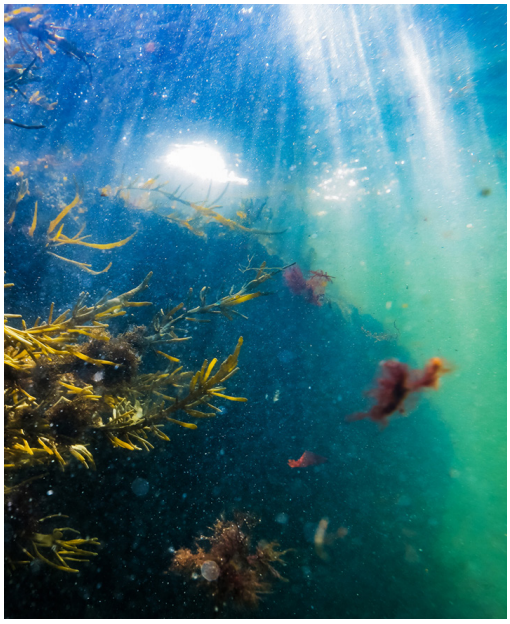
Today's environmental status of Narragansett Bay (Bay) and the entire Narragansett Bay Watershed (Watershed) is the result of hundreds of years of human activities and environmental variability. Numerous organizations have worked together for decades to reverse trends of declining environmental conditions and to understand new threats such as those associated with climate change. Beginning in 2014, the Narragansett Bay Estuary Program brought together more than 50 practitioners from universities, organizations, and agencies in Massachusetts and Rhode Island to collaborate on the 2017 *State of Narragansett Bay and Its Watershed* report. This report presents and tracks 24 indicators in order to

- evaluate key stressors to the Bay and Watershed,
- assess the chemical, physical, and biological conditions,
- describe past and recent trends,
- look ahead to potential future changes, and
- identify data and research needed to advance our understanding of these changes.

The data and analyses presented in this report revealed many noteworthy patterns and trends, some favorable and others of concern. Major themes, reflecting the many interactions and relationships among the 24 indicators, included the following:

1. ***The water in the Bay is cleaner*** because of restrictions on harmful chemicals and reductions in nutrient and pathogen pollution from wastewater treatment facilities and combined sewer overflows. Legacy contaminants such as metals and PCBs have declined, but some—notably mercury—still pose human health risks. Nitrogen and phosphorus loadings have dropped by approximately half in the last fifteen years, and pathogen loading has declined due to capture and treatment of combined sewer overflows. However, non-point source pollution remains a challenge throughout the Watershed. An increasingly problematic source of pollution is runoff of pollutant-laden stormwater from roads, buildings, and other impervious surfaces into waterways. In addition, nearly 40 percent of the Watershed's residents are served by septic systems and cesspools, a source of nutrient, pathogen, and chemical pollution to groundwater and surface waters.

2. ***The Bay and Watershed ecosystems are changing*** and producing unanswered questions. Some changes could be responses to water quality improvements through nutrient reduction, such as apparent improvements in seafloor habitat seen in data from 1988 to 2008, and through pathogen reduction, such as an increase in areas open to shellfishing since 2005. However, existing data lack the temporal and spatial resolution needed to discern trends in these and other important indicators of ecosystem condition and public health, such as dissolved oxygen, phytoplankton, freshwater fish, and beach closures. To date, certain locations and parameters in the Bay and Watershed have been monitored more thoroughly than others, and as monitoring continues and potentially increases (see #5 below) it is expected that a more comprehensive understanding of the ecosystem's responses to reduced nutrient and pathogen discharges will be realized.



Underwater scene at Beavertail State Park, Jamestown, RI (Ayla Fox)

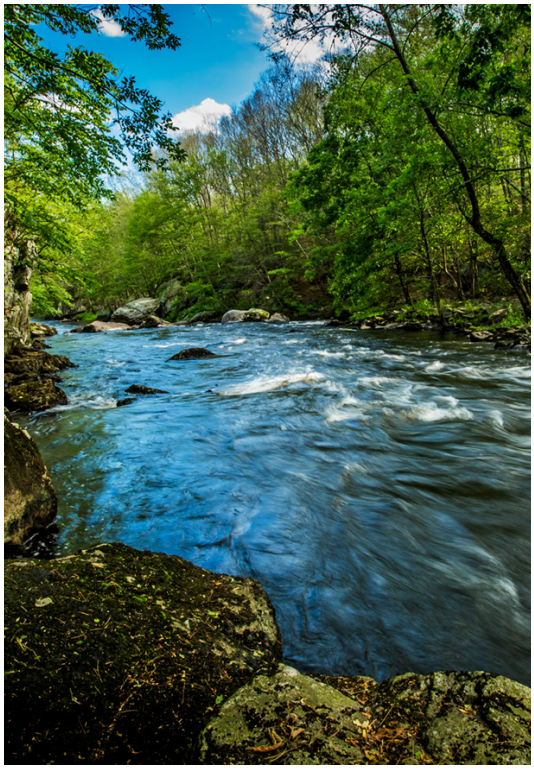
3. ***Conditions improve with distance from urban areas*** in the Bay and Watershed. Pollution of the Bay's water and sediments generally improves from north to south, as does water quality as indicated by dissolved oxygen and chlorophyll levels. While fifteen percent of the Watershed's land has been protected as natural open space, forest lands have rapidly been converted to urban areas. As people move outward from urban centers, they demand more infrastructure, building on currently unprotected natural lands, which fundamentally changes the landscape and reduces protection of habitat, water quality, and human wellbeing.

4. ***Stressors associated with climate change are increasing rapidly.*** Air and water temperatures are warming, the intensity and seasonality of precipitation are changing, and

sea level is rising. These stressors are already causing ecological responses such as altering the estuarine fish community, with warm-water species increasing and cold-water species declining. Impacts of climate change on the cities, towns, and ecosystem of Narragansett Bay are projected to intensify, such as increased flooding and erosion of coastal properties, loss of salt marshes, and potentially more beach closures due to pathogens. Adaptation will require well-informed action by local communities.

5. ***More research and monitoring are needed to understand the rapid, major changes occurring in the Bay and Watershed in order to enable well-informed adaptation and mitigation.*** For many key indicators of stress and condition, such as onsite wastewater treatment systems and emerging contaminants, filling the data gaps will require new approaches and/or a long-term financial commitment.

By fostering a common understanding of historical trends and the present status, this report supports decision-making to protect, maintain and restore the essential attributes and functions of the Bay and Watershed, and to sustain ecosystem services that provide benefits for people.



Blackstone River Gorge, Blackstone, MA (Ayla Fox)

Selected Findings for All Indicators

Stressor Indicators

CLIMATE CHANGE STRESSORS

- **Air temperature** increased approximately 1.5°C (2.7°F) from 1960 to 2015, while the increase in Bay water temperature was slightly greater at 1.6°C (2.9°F). Climate projections suggest that air temperature in the region will increase another 2 to 6°C (5 to 10°F) by 2100.
- Annual **precipitation** has increased 0.4 to 0.7 inches per decade since 1895, and the amount falling during intense storms has increased 71 percent since 1965. Local projections include greater frequency, volume, and intensity and changing seasonality of precipitation.
- **Sea level** in Rhode Island rose nine inches as measured at the tide gauge in Newport from 1930 to 2015 and 6.6 inches at Providence from 1938 to 2015. NOAA projects that sea level at Newport could rise as much as 3.4 feet by 2050 and eleven feet by 2100, considering factors such as rapid melting of land-based polar ice.

LANDSCAPE STRESSORS

- The Watershed's **population** increased by eight percent from 1990 to 2010. In 2014, the population of 1.9 million was split approximately equally between Massachusetts and Rhode Island. Recent population growth was concentrated in suburban and lesser-developed areas, where people use more land per capita.
- From 2001 to 2011, areas of urban **land use** increased by 8.5 percent, and forest lands decreased by four percent. In fact, most of the land conversion was from forest to urban. In 2011, coastal subwatersheds were 65 to 85 percent urban, while headwater subwatersheds were 70 percent forest.
- **Impervious cover** such as pavement and buildings constituted fourteen percent of the Watershed's landscape. Thirty-six of the 52 subwatersheds in the Watershed had more than the detrimental threshold of ten percent impervious cover.
- On average, 37 **wastewater treatment** facilities discharge approximately 203 million gallons of treated wastewater per day into the rivers and Bay. Thirty-eight percent of the Watershed's residents are served by septic systems and cesspools, with eight percent living in areas with high densities of these onsite systems.

- Over the last fifteen years, **nutrient loading** from wastewater treatment facilities decreased by 55 percent for total nitrogen and 45 percent for total phosphorus.

CHEMICAL STRESSORS

- Although **legacy contaminants** such as metals, PCBs, and pesticides have decreased dramatically in the last 50 years, sediments in some areas still have high concentrations and mercury concentrations in fish still pose a significant threat to human health.
- **Emerging contaminants** of concern from personal care products, pharmaceuticals, and industrial practices have appeared recently, and their concentrations tend to be highest in the Upper Bay.

Indicators of Ecosystem Condition

BAY ECOSYSTEM INDICATORS

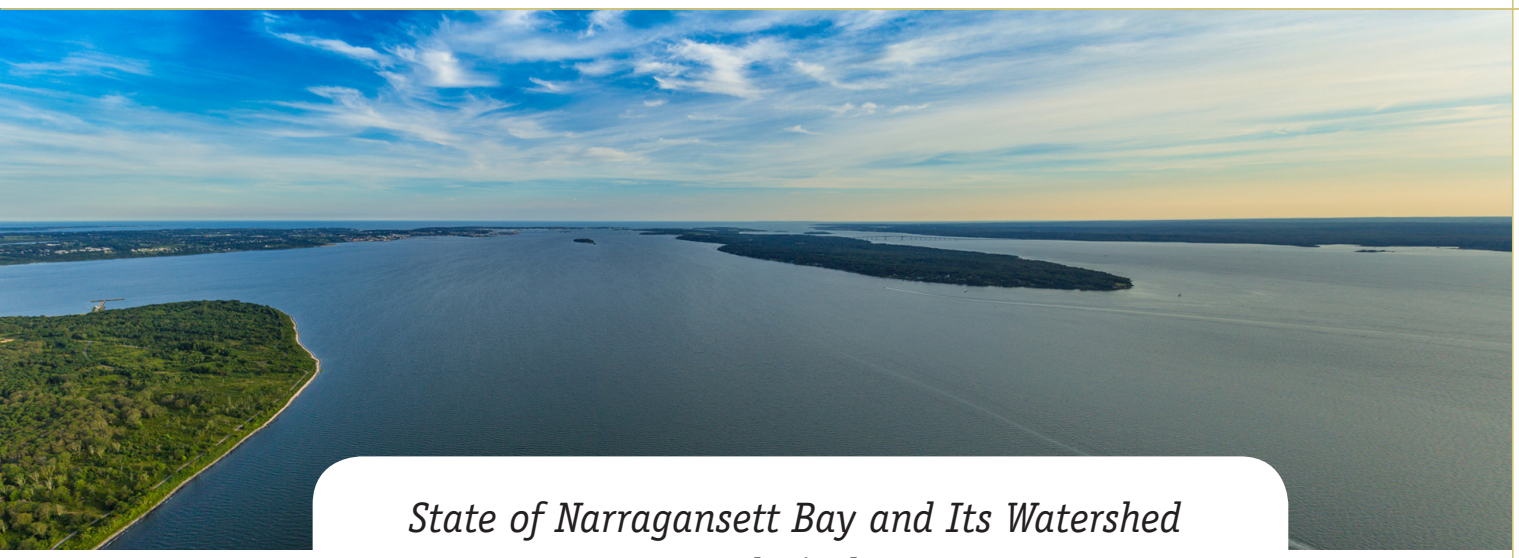
- In 2012, 513 acres of **seagrasses** were documented in the Bay–37 percent more than in 2006. Acreage estimates from 2016 indicate a small decline of eight percent for a total of 479 acres.
- Recent analysis of aerial photographs found 3,321 acres of **salt marsh** in the Bay. Thirteen to 87 percent of Rhode Island's marshes may be lost under one- and five-foot projections of future sea level rise.
- The quality of **benthic habitats** improved between 1988 and 2008, as indicated by an increase in small, tube-building crustaceans that are sensitive to pollution.
- **Estuarine fish communities** have changed since the 1980s with warm-water species such as scup and butterfish displacing cool-water resident species such as winter flounder and red hake, partly due to increasing temperatures.
- From 2005 to 2015, summer **dissolved oxygen** concentrations in bottom waters of the Upper Bay were highly variable–generally higher in dry years and lower in wet years.
- In late winter and summer, **chlorophyll** concentrations consistently decreased from the Upper Bay to the Lower Bay. From 2001 to 2015, summer Upper Bay chlorophyll concentrations were variable–generally lower in dry years and higher in wet years.
- Data collected in 2014 showed that **water clarity** was greatest in the Lower Bay and declined into the urbanized Upper Bay. Water clarity in the Upper Bay varied greatly in recent years with no discernible trends.

WATERSHED ECOSYSTEM INDICATORS

- Half of the estuarine waters and more than half of the streams and rivers assessed by the states had healthy **water quality conditions for aquatic life**. Conversely, the other half of estuarine waters and 20 percent of the rivers assessed for aquatic life were impacted by nutrient enrichment and low oxygen levels.
- Of the 78 sites assessed for **stream invertebrates** between 2002 and 2014, all but one site had good habitat conditions, as expected in riffle habitats. However, nineteen percent of sites with suitable habitat had poor macroinvertebrate health, likely due to water quality impacts.
- The Upper Blackstone River had **freshwater fish communities** with the highest proportions of fluvial species and cold- and coolwater species. More than fourteen percent of the Watershed's landscape was identified as areas with streams where brook trout had once been present, naturally.
- As of 2015, 171,244 acres of natural lands had been protected as **open space**, representing over fifteen percent of the Watershed's land. More than 185,000 acres of natural lands with high ecological integrity remained unprotected.

Indicators of Public Health

- Eighty-five percent of estuarine waters and nearly 80 percent of the total acreage of lakes and ponds assessed by the states had suitable **water quality conditions for recreation** such as swimming and boating. Over 60 percent of freshwater streams were determined to be unsuitable for recreational use because of pathogens.
- Of the 37 licensed **marine beaches** in Narragansett Bay, fourteen were considered of High Concern. Prior to 2009, most closures of High Concern beaches occurred in the Upper Bay, in particular during wet summer seasons. After 2010, beach closures continued but there is no evidence that events were increasing or that rainfall was the major trigger.
- After a decline of over 2,500 acres of suitable **shellfishing areas** for direct human consumption occurred between 1995 and 2005, a recovery began in the Upper Estuary. From 1995 to 2005, 1,600 acres were upgraded to conditionally approved. In 2015, 63 percent of the Bay was open (without restrictions), while thirteen percent of the Bay's area was conditionally approved. By 2017, 3,711 acres were opened without restrictions in the Upper Bay.



State of Narragansett Bay and Its Watershed
2017 Technical Report

INTRODUCTION

Please use the following citation: Narragansett Bay Estuary Program. 2017. State of Narragansett Bay and Its Watershed (Introduction, pages 7-22). Technical Report. Providence, RI.

NARRAGANSETT BAY ESTUARY PROGRAM

Photo: Aerial view of Narragansett Bay looking south over Prudence Island (left) and Jamestown, RI (right) (Ayla Fox)

Narragansett Bay is one of the largest estuaries in New England, covering approximately 196 square miles. It is about two-thirds the size of New York City, or two times the size of Boston, and receives freshwater primarily from the Blackstone and Pawtuxet Rivers in the west and the Taunton River in the east. The Narragansett Bay Watershed is densely populated, containing over one hundred cities and towns (Figure 1; Appendix) with a total population of 1.95 million residents. While the majority of the Bay itself and its approximately 560 miles of coastline are situated in the state of Rhode Island, 60 percent of the 1,705-square-mile Watershed is in Massachusetts.

Narragansett Bay and the Watershed produce many benefits (or ecosystem services) for people. The ecosystem provides recreational opportunities, food, storm and flood protection, natural beauty, and important cultural significance to the region. The estuarine and fresh waters, forests, and wetlands also support tourism, transportation, and other economic activities, as they have for centuries. Other important ecosystem services include natural processes that filter and clean water and air, support wildlife, and sequester carbon (mitigating global climate change). The region supports a broad range of scientific and educational activities, and its lands and waters are highly valued by residents and visitors alike.

The capacity to provide those benefits depends, in large part, on natural ecological processes, which can be affected by many types of human activities. The most significant human influences likely began in the 1800s when the Blackstone River Valley became the birthplace of the American Industrial Revolution. The resulting development completely reconfigured the northern portion of the Watershed through damming of rivers, construction of mills and canals, and urbanization (Pastore 2014). Today, the Watershed's three major population centers are located in Providence, Rhode Island, and in Fall River and Worcester, Massachusetts, but the population has spread far beyond the urban cores. In the past 40 years, management and regulatory actions have focused on improving water quality through numerous local, state, and federal mechanisms, including upgrades to wastewater treatment technologies and improvements to stormwater infrastructure, supported by years of water quality monitoring by agencies, researchers, citizens, and other environmental stewards. Related actions have sought to manage development within the Watershed, preserve open space, and manage land use to reduce and mitigate pollution sources. Although nutrients and other contaminants discharged from point sources have been reduced significantly (Chintala et al. 2015), pollution from non-point sources, the legacy of all previous human activities, the emergence of new stressors (e.g., pharmaceuticals), and the effects of rapidly accelerating climate change continue to affect ecosystem condition and public health throughout the Watershed.



Conimicut Lighthouse, Warwick, RI (Ayla Fox)

The Narragansett Bay Estuary Program was launched in 1985 with federal financial support as a pilot program for estuarine research. In 1988, the National Estuary Program selected Narragansett Bay for inclusion in the national network, charging it with goals informed by the federal Clean Water Act to restore and maintain the "chemical, physical, and biological integrity of the estuary, including restoration and maintenance of water quality, a balanced indigenous population of shellfish, fish, and wildlife, and recreational activities in the estuary." One of the hallmarks of the National Estuary Program is its role in bringing an ecosystem approach and sound science to environmental management. Under the umbrella of the Narragansett Bay Estuary Program, the US Environmental Protection Agency, Rhode

Island Department of Environmental Management, Massachusetts Executive Office of Environmental Affairs (now known as the Massachusetts Executive Office of Energy and Environmental Affairs), numerous other state agencies, and conservation-oriented partners launched a research and planning program that generated scores of peer-reviewed studies to characterize the state of the Bay and Watershed. The studies served as the foundation for the first Comprehensive Conservation and Management Plan (CCMP) published in 1992. Since then, the Estuary Program has reported on status and trends at approximately five-year intervals. The information was used to produce an updated CCMP in 2012 with new or revised actions for implementation.

Since 2014, the Estuary Program has been focusing on the bi-state elements of the Bay and Watershed, bringing together practitioners from universities, organizations, and agencies in Massachusetts and Rhode Island to collaborate on gathering data and information for this 2017 *State of Narragansett Bay and Its Watershed* report. The Estuary Program worked with partners and organized topical technical workgroups to assemble data, develop and validate methods, and write technical content for the report. A Science Advisory Committee was convened to approve indicators, data sources, and methodological approaches developed by the workgroups. This process enabled the Estuary Program to integrate the most recent, comparable, and compatible data across state lines, and to report results seamlessly, using an array of state-of-the-art tools to analyze data and interpret and present results. For many topics, the Estuary Program and collaborators created unique datasets and new methodologies. Not only did this effort review and synthesize the most recent information as of 2015, it also worked with a community dedicated to maintaining high-quality data, bringing research to bear on management needs, and communicating results widely across different audiences. It is the purpose of this report to reflect these broad goals.

More specifically, this report seeks to advance an understanding of changing conditions, so that future decisions affecting the processes and functions of the region are better informed and reflect the many benefits provided by these systems. As in all estuaries and coastal watersheds across New England, scientists and other practitioners are challenged with understanding and managing the legacy of historical ecosystem stressors and the potentially profound future impacts due to climate change and a growing population. The Bay and Watershed of the future will not look like the Bay and Watershed of the present. By creating a common understanding of the Narragansett Bay Watershed's status and trends, this report can inform prospective decision-making of Estuary Program partners to maintain the region's essential attributes and functions that make it a place where people can and want to live, visit, and enjoy.

While this status and trends assessment encountered some notable information gaps as identified in the indicator chapters, and social science aspects of the indicators—the relationship of people, society, and economy to a changing ecosystem—are only peripherally addressed, this technical report represents the most current and comprehensive environmental synthesis of the Narragansett Bay and Watershed to appear in a single volume and to benefit from broad multidisciplinary collaboration.



Lincoln Woods State Park Beach, Lincoln, Rhode Island (Ayla Fox)

Geographic Setting

The geographic scope of this report is Narragansett Bay and its Watershed (Figure 1), which span the border of Rhode Island and Massachusetts. The Bay is a phytoplankton-based temperate estuary positioned east of Long Island Sound, with Rhode Island Sound and the Atlantic Ocean to its south. The mouth of Narragansett Bay is defined by Point Judith in the west and Sakonnet Point in the east. Within the Bay, the East and West Passage are differentiated by Conanicut Island (Jamestown) and Prudence Island. In the most easterly portion of the Bay are Mount Hope Bay and the Sakonnet River, the latter of which is separated from the East Passage by Aquidneck Island. The East and West Passages and the Sakonnet River are important navigational channels to the cities of Providence, Rhode Island, and Fall River, Massachusetts. The deepest part of the Bay is 194 feet in the East Passage near Fort Adams Park and Newport Harbor, while the average depth of the Bay is 26 feet. (NGDC 2011).

The Watershed extends northwest to Worcester, Massachusetts, and the headwaters of the Blackstone River and east to Brockton, Massachusetts, and the headwaters of the Taunton River. Together, the Blackstone and Taunton Rivers deliver 67 percent of the freshwater flow to the Bay (Ries 1990), including the effluent from nineteen wastewater treatment facilities that discharge to those rivers.



Aerial photo of Narragansett Bay facing southeast overlooking Jamestown Verrazzano Bridge (Ayla Fox)

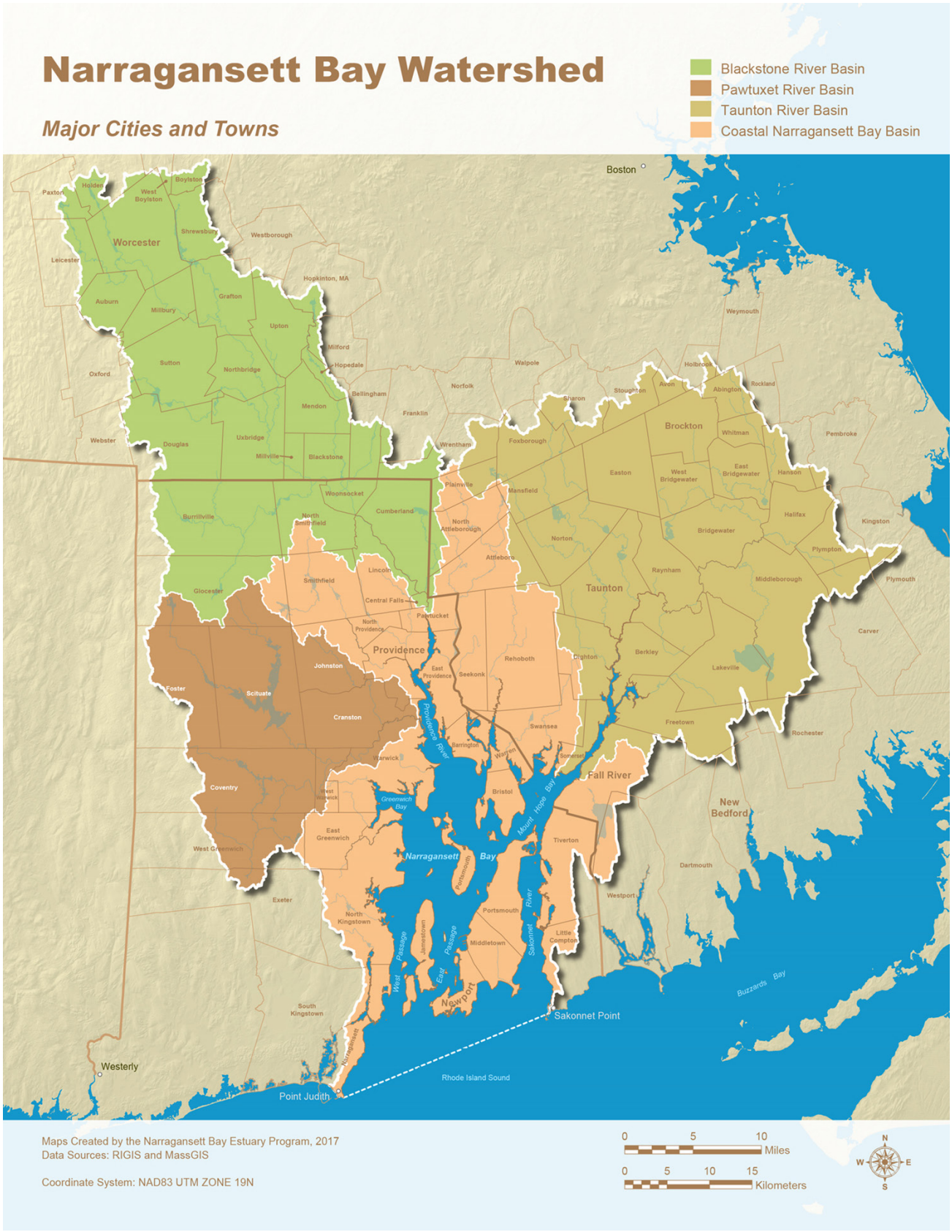


Figure 1. Towns and cities in the Narragansett Bay Watershed

Reporting Status and Trends: The Estuary Program's Approach and Organizing Considerations

Stressor-Condition Framework

This report describes the status and past trends for 24 indicators, identifies the linkages between and among these indicators, and discusses how trends may change through time in response to changing stressors and management actions. The 24 indicators described in this report were selected based on extensive consultation with Estuary Program partners and data availability following the Quality Assurance Project Plan (NBEP 2015).

The Estuary Program categorizes the 24 indicators as Stressor Indicators or Condition Indicators (Figure 2). The Stressor Indicators provide information about key climate change, landscape, and chemical processes and characteristics that cause changes in the Bay and Watershed. The Condition Indicators provide information about the status of the ecosystem and public health. Collectively, the indicators give insight into the ability of the Bay and Watershed to provide benefits to people such as freshwater and marine fisheries, clean water, and outdoor recreation.

Importantly, the indicator results are useful to resource managers, advocates, and the public to enable well-informed adaptation to climate change, land use planning, conservation and restoration, and infrastructure improvements. This report also identifies research and data needs to enhance understanding of the complex interaction of processes that influence the status and trends of the Bay and Watershed.

An important stressor not included among the 24 indicators is ocean and coastal acidification. Ocean and coastal acidification is a significant concern because of its potential impacts on ocean and estuarine food webs and on economically important organisms such as shellfish (Gledhill et al. 2015). The Estuary Program did not dedicate a complete chapter of this report to ocean and coastal acidification because few data were available for Narragansett Bay and currently there is considerable uncertainty in predicting responses to acidification in the estuarine environment. However, a discussion of this important issue is included in the Introduction to Climate Change Stressors section.

The Stressor-Condition framework used by the Estuary Program aligns with the Biological Condition Gradient (BCG) framework, a conceptual, scientific

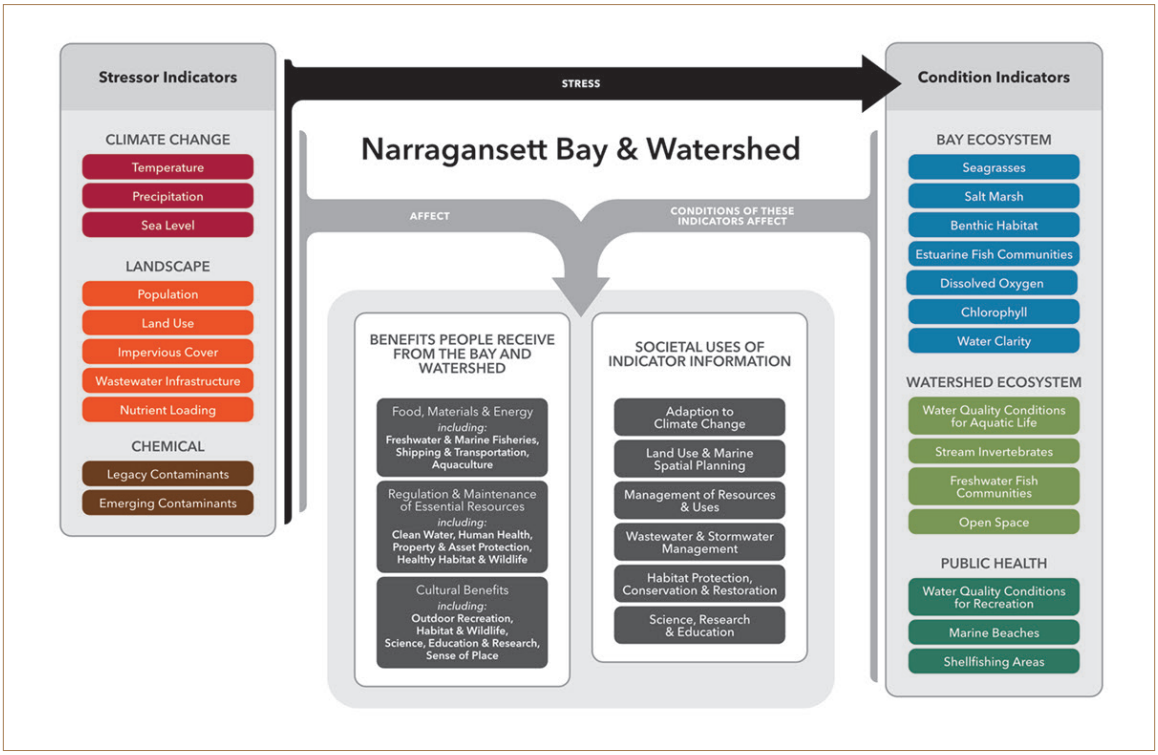


Figure 2. The 24 indicators described in this report are categorized as Stressor Indicators (left) or Condition Indicators (right). All of the indicators provide information about benefits that people receive from the Bay and its Watershed (center left), and the indicator information can be used in decision-making (center right).

framework for interpreting biological response to increasing effects of stressors on aquatic ecosystems (USEPA 2016) (Figure 3).

Changes in ecological condition affect ecosystem functions and the societal benefits that humans receive from the ecosystem (Bradley and Yee 2015; Figure 2). These changes may prompt management actions by federal, state, or municipal government. The Stressor-Condition framework and BCG can be used to help determine the appropriate management actions and to better anticipate outcomes. After management actions are implemented, Condition Indicators can be monitored to track the effects of the actions on the ecosystem.

Stressor-Condition Relationships

While this report contains a separate chapter for each of the 24 indicators, the indicators are not separate or independent, but relate to each other in complex and important ways. Various combinations of natural and human-induced factors exert pressures on the Bay and Watershed ecosystems and cause changes in ecological state (Bradley and Yee 2015). Most of the Condition Indicators are influenced by multiple Stressor Indicators, some Stressor Indicators influence other Stressor Indicators, and similarly, Condition Indicators influence other Condition Indicators.

To reflect these important relationships, each indicator chapter begins with a conceptual graphic that illustrates major connections among indicators. As an example, Figure 4 shows the Benthic Habitat Indicator stressed by Climate Change, Landscape, and Chemical Stressors, and influenced by Estuarine Fish Communities (via predation) and Dissolved Oxygen (via hypoxia). This report did not seek to quantify causal relationships between Stressor Indicators and Condition Indicators, but instead uses the graphics to represent overall relationships. To the extent possible based on previous studies, the chapters include quantitative and qualitative information about the relationships.

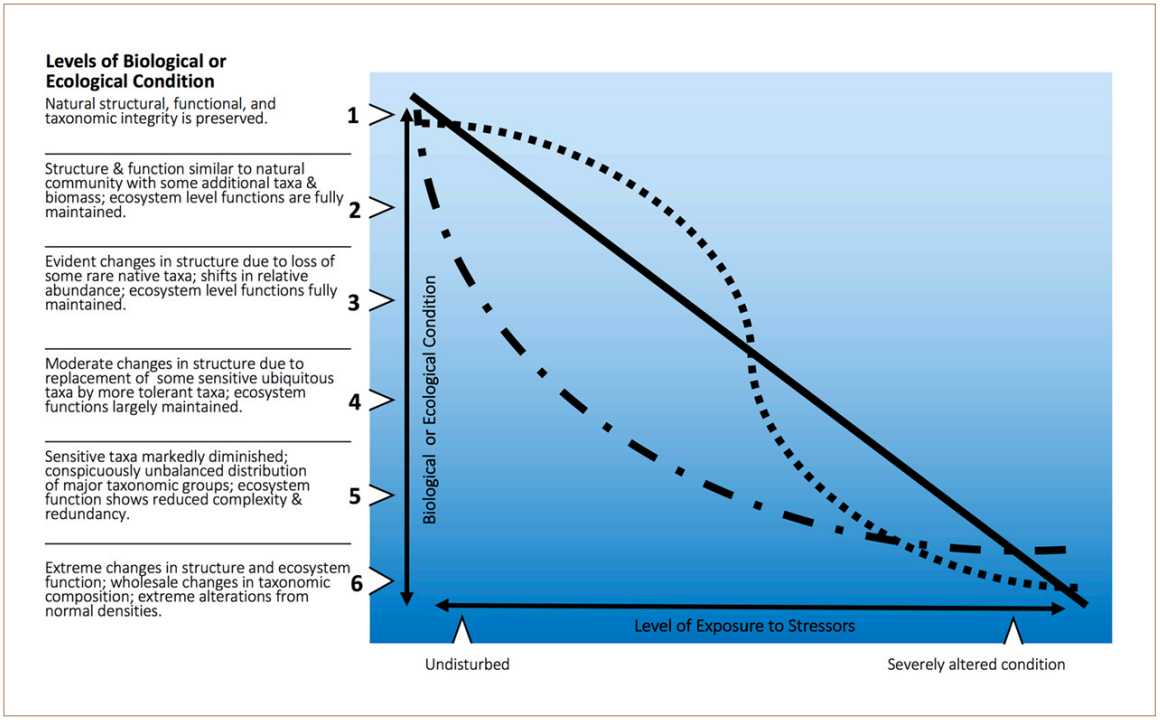


Figure 3. A conceptual illustration based on the Biological Condition Gradient (BCG) framework, showing potential relationships between biological or ecological condition (vertical axis) and a gradient of stress ranging from undisturbed or baseline conditions to severely altered conditions (horizontal axis). In this generalization, multiple lines represent different potential relationships between stressors and ecological condition. Adapted from Shumchenia et al. 2016 and Cicchetti et al. 2017.

Collaborative Effort

The Estuary Program collaborated with partners from federal and state agencies, universities, and non-profit organizations who supported the development of indicators, the validation of methods, the preparation and sharing of data, and the interpretation of results, and they provided substantial and insightful comments during the development of this report. This collaboration included extensive engagement of partners associated with state agencies in Massachusetts and Rhode Island. An interdisciplinary Science Advisory Committee was formed to provide for the exchange and analysis of existing and emerging scientific issues and to support the development of conceptual frameworks integrating physical, chemical, biological, geological, ecological, economic, and policy dimensions. Topical bi-state and inter-institutional expert workgroups were formed to focus on indicators for seagrass, salt marsh, water clarity, dissolved oxygen, and chlorophyll. Additionally, a GIS workgroup collaborated to define the best geospatial data analysis approaches for the indicators. These collaborative efforts improved the Estuary Program's overall understanding of the 24 indicators and built opportunities for partnerships and future interactions.

Data Sources and Sampling Frequency

DATA AVAILABILITY AND VARIABILITY

Stressor Indicators, Ecosystem Condition Indicators, and Indicators of Public Health are each described by one or more metrics. For example, temperature—a Climate Change Stressor Indicator—is described by air, freshwater, and estuarine water temperatures. The highest-quality data sources for each indicator were identified by input from Estuary Program partners. The Estuary Program did not collect new data for this report but did generate results from new analyses of existing data (NBEP 2015). The Estuary Program considered technically reviewed data from multiple sources, including state and federal agencies, nonprofit organizations, and academic and other institutions, prioritizing data collected specifically for monitoring and assessments. The Estuary Program sought data that conformed to quality assurance and quality control standards.

In many cases, the existing data were specific to the Bay or Watershed, but in some cases national, regional, or state-level data were distilled to the Bay or Watershed level. Significant challenges in obtaining and integrating different data sources arose from the fact that both the Bay and the Watershed span

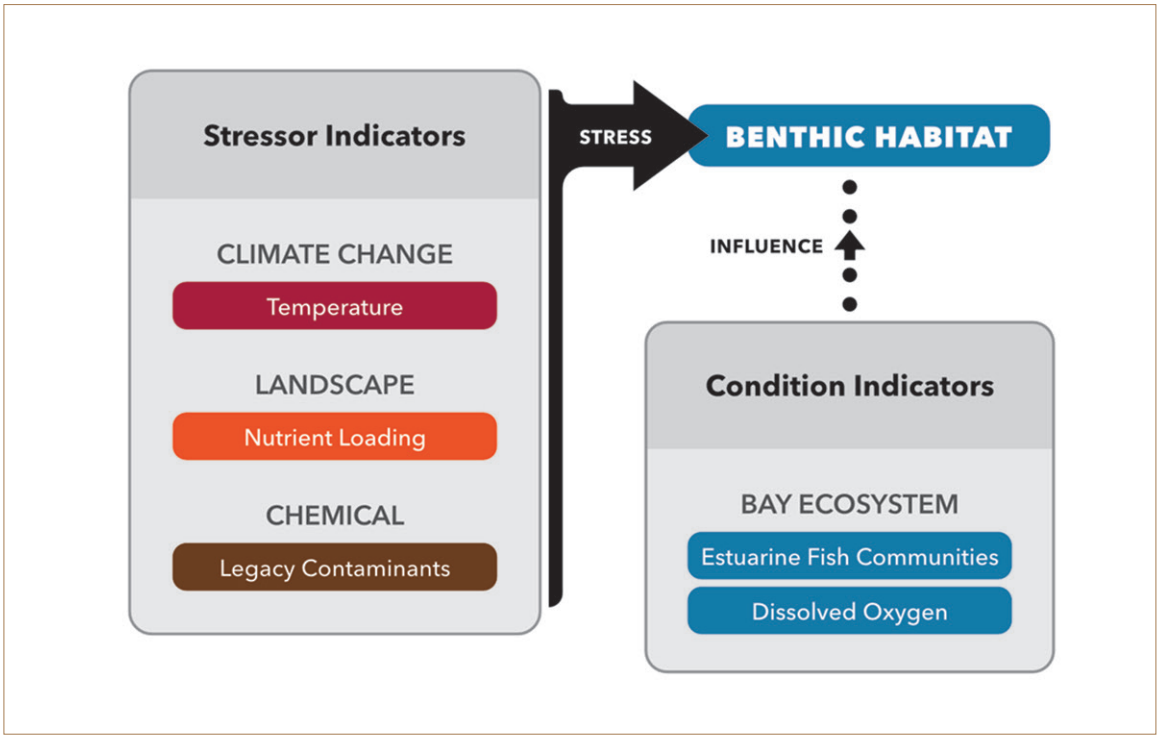


Figure 4. Example of conceptual graphic conveying important Stressor-Condition relationships. This example shows Stressor Indicators (left) affecting the Benthic Habitat Indicator of Bay Ecosystem Condition. Other Condition Indicators, in this case Estuarine Fish Communities and Dissolved Oxygen, also influence the focal indicator.

the state boundary of Massachusetts and Rhode Island. Some existing datasets were collected and reported according to municipal or state boundaries, and at varying time scales. The Estuary Program prioritized data that were consistent across state boundaries, even if the individual states had more finely resolved, or more accurate, data. The Estuary Program and partners conducted analyses taking into consideration the inherent limitations derived from the datasets and among sources. These considerations are described on a case-by-case basis within each indicator chapter.

DATA FROM MONITORING EFFORTS

Many of the datasets came from monitoring programs conducted by government, non-government, and academic entities. Data derived from monitoring programs reflect the financial, technical, and/or time constraints of each particular program. Monitoring strategies may be developed to detect seasonal environmental changes or longer-term changes, and those goals typically determine the frequency of sampling. Basic differences in the technology and effort required to monitor certain features also translate to significant differences in sampling frequency. In practice, this means that some data, such as temperature, could be recorded nearly continuously (for example, every 30 seconds), whereas other data such as seagrass extent may be collected at five-year intervals. In some cases, monitoring data had been collected over a period of many years at a frequency of once to a few times per year at multiple sites throughout the Bay or Watershed. Depending on the spatial and temporal resolution, these data were used to develop either a broad characterization of indicator condition, such as for freshwater fish communities, or in conjunction with other data to identify trends, such as for water clarity.

DATA FROM STATE AGENCIES

State agencies of Massachusetts and Rhode Island provided data for numerous indicators including: wastewater infrastructure, stream invertebrates, fish communities, water quality conditions, open space, marine beaches, and shellfishing areas. Some of these data had already been analyzed, interpreted, and reported to meet regulatory requirements. The frequency of data collection by the state agencies differed among states and programs, and the Estuary Program worked closely with the states to ensure that interpretations for this report were consistent and comparable. Some data from state agencies required a crosswalk to make certain the data were consistent before analysis, such as the data for the Marine Beaches and Shellfishing Areas indicators.

DATA FROM OTHER STATE, NATIONAL, AND REGIONAL EFFORTS

For some indicators, the Estuary Program chose to use geographical data from national or regional efforts when data covered the entire area of the Watershed and the resolution of the data was appropriate. These datasets were used when state-level data were not comparable and could not be reconciled. Additionally, the use of national datasets facilitates future assessments because most of the datasets are updated routinely using the same methods. When a national dataset did not provide the necessary resolution and state-level data were available for more accurate results, the Estuary Program merged state datasets from Massachusetts and Rhode Island.

Spatial Analysis

The Estuary Program and partners used spatial analysis tools and developed models within ArcGIS Desktop (Esri 2016) to conduct analyses at multiple spatial scales: (1) to summarize data at the level of information required to measure each metric, (2) to produce maps to support data interpretations, and (3) to export results as tabular data to create graphs and tables, to use as inputs for other analyses, and ultimately to interpret results. For some indicators, multiple datasets in varying formats and scales were reconciled into seamless datasets that included information only for areas within the boundaries of the Bay and its Watershed. Spatial data used for this report met minimum criteria of comparability, consistency, accuracy, and suitability for reconciling across datasets, at the Bay and Watershed scale (NBEP 2015). For its analyses, the Estuary Program used a hierarchy of spatial scales for the Bay and another for the Watershed (Table 1). Terminology and definitions for each level of the hierarchies are discussed below.

For reference while reading the indicator chapters, the Appendix provides detailed lists of the names and sizes of individual geographic units at each level of the hierarchies, such as River Basins, watersheds, and subwatersheds.

HIERARCHY OF BAY SCALES

Narragansett Bay: This scale is the broadest geographical scale for the Bay. It encompasses all estuarine waters, including tidal rivers emptying into the Bay, and it extends south to the boundary between Point Judith and Sakonnet Point.

> Bay Regions: This scale divides Narragansett Bay into five Regions: the Upper Estuary (those waters in the Upper Bay into which the major rivers and most of the Watershed drain), the East Passage, the West Passage, the Sakonnet River, and the Mouth of Bay (which is influenced by open ocean waters and includes the Narrow River).

>>Bay Sections: Bay Regions are further subdivided into eleven Bay Sections: Providence River Estuary; Upper Narragansett Bay; Greenwich Bay; Warren, Palmer, and Barrington Rivers; Taunton River; Mount Hope Bay; East Passage; West Passage; Sakonnet River; Mouth of Narragansett Bay; and Narrow River.

>>>Bay Segments: Bay Sections are further subdivided into Bay Segments, which include features such as coves, embayments, and tidal rivers. Bay Segments are the finest geographical scale for the Bay. For the most part, the boundaries of the Bay Segments correspond to delineations by researchers studying the Bay[1]. The Estuary Program compiled all available information and used the most common delineations to define Bay Segments. The upper boundaries of the estuarine waters correspond to the limits of tidal waters defined by: (1) the presence of existing dams (as of 2016) and (2) the extent of estuarine waters as defined and delineated by the states to assess water quality and shellfishing areas.

HIERARCHY OF WATERSHED SCALES

Narragansett Bay Watershed: This scale is the broadest geographical scale for the Watershed, encompassing all lands that drain into the major rivers and their tributaries, and coastal areas surrounding and draining directly to the Bay. Nested within the Watershed are two different types of geographic areas referenced in this report: (1) River Basins and (2) watersheds and subwatersheds defined by the US Geological Survey, each of which is assigned a ten- or twelve-digit Hydrologic Unit Code (HUC10 or HUC12).

>River Basins: The Estuary Program defined four River Basins representing major drainage areas within the Narragansett Bay Watershed: Blackstone River, Taunton River, Pawtuxet River, and Coastal Narragansett Bay.

>>Watershed Planning Areas (WPAs): These 42 areas were defined by Massachusetts and Rhode Island to develop watershed management plans. WPAs are nested within the River Basins, and some WPAs have the same boundaries as HUC12 subwatersheds (see below). The Estuary Program calculated results focused on WPAs for some indicators relevant to resource management and planning.

>HUC10 watersheds: A total of eleven HUC10 watersheds, defined by the USGS, together form the larger Narragansett Bay Watershed.

>>HUC12 subwatersheds: Nested within the HUC10 watersheds are a total of 52 HUC12 subwatersheds, the finest geographic scale used for reporting of Watershed indicators.

In this report, the capitalized term "Watershed" refers to the entire Narragansett Bay Watershed, and the lowercase term "watershed" refers to HUC10 watersheds. The Estuary Program analyzed some indicators at the scale of individual cities and towns within the Watershed. While those results are not presented in this report, they are available on request.

Historical Data

The rich history of Narragansett Bay and its Watershed includes thousands of years of habitation by native peoples, colonial influxes of Europeans, the Industrial Age, and decades of suburbanization. In addition to analyzing current datasets, the Estuary Program sought to characterize historical data for each indicator to the extent practicable. Such a characterization provides important context about conditions that have contributed to the present status and potential future changes. The amount and time periods of available historical data varied among indicators (Table 2). Collection of ecological data, for example, generally did not begin until the 1950s and 1960s, whereas longer historical records exist for indicators such as population and air temperature. Table 2 highlights data gaps and suggests where the current understanding of trends may be more robust or more limited based on the types of historical data available.

[1] The full list of sources used by the Estuary Program is available upon request. Briefly, the Estuary Program used Hicks 1959, Kramer and Nixon 1978, Pilson 1985, Kramer et al. 2010, and NBNERR 2009. For tidal water boundaries, the Estuary Program used Massachusetts and Rhode Island Shellfish Management Areas and Integrated Water Quality Monitoring and Assessment Reports.

Limitations and Future Considerations

Some indicators presented in this report have been well studied with ample information available to describe status and trends, whereas other indicators are not well understood. Therefore, some indicators are still under development by the Estuary Program. Work with scientists, managers and practitioners to advance particular indicators and to better understand stressor-response relationships will continue. Although the selected indicators are quite comprehensive, it may be determined in the future that additional indicators should be added to this framework, perhaps in response to new problems that may emerge or monitoring approaches that become more refined. For example, indicators for ocean and coastal acidification, cyanobacteria, and fish passage may be added in future reports. Similarly, assessment of some indicators may be discontinued if the science to understand the complex interactions does not adequately advance.

Recognizing the library of information assembled, the status and trends documented for this report still reflect important limitations such as irregular and changing methods, inconsistent funding for data collection, and changes to sampling locations and frequencies.

VALUE OF THE INDICATORS

Despite these challenges, there are many indicators with very clear histories and trajectories. Several indicators show improving ecological condition–the result of decades of work to improve water quality–demonstrating that indicators are sensitive to management action. Other indicators may not detect changes in response to specific management actions, such as the Climate Change Stressor Indicators or the Population Indicator, but are important to include because of their influence on the trajectories of other indicators. A fundamental challenge addressed in this report is understanding the long-term change in each indicator and presenting considerations to resource managers and the public for which degrading trends can be or should be slowed or reversed, and improving trends facilitated if necessary.

Table 1. Spatial scales of indicator results included in the chapters of this report. Asterisks indicate additional spatial scales at which the Estuary Program conducted analyses; those results are available by request.

Indicators	Spatial Scales									
	Narragansett Bay				Narragansett Bay Watershed					
	Bay	Regions	Sections	Segments	Watershed	HUC10 watersheds	HUC12 subwatersheds	River Basins	Watershed Planning Areas	Municipalities
Stressor Indicators										
Climate Change										
Temperature	X				X					
Precipitation					X					
Sea Level	X									
Landscape										
Population					X	X	X			*
Land Use					X	X	X			
Impervious Cover					X	X	X	X	*	*
Wastewater Infrastructure					X			*	X	
Nutrient Loading	X				X			X		
Chemical										
Legacy Contaminants	X									
Emerging Contaminants	X									
Condition Indicators										
Bay Ecosystem										
Seagrasses	X		X	X						
Salt Marsh	X		X	X						
Benthic Habitat	X									
Estuarine Fish Communities	X									
Dissolved Oxygen	X	X	X							
Chlorophyll	X	X	X							
Water Clarity	X	X	X							

Table 1 continued.

Indicators	Spatial Scales									
	Narragansett Bay				Narragansett Bay Watershed					
	Bay	Regions	Sections	Segments	Watershed	HUC10 watersheds	HUC12 subwatersheds	River Basins	Watershed Planning Areas	Municipalities
Watershed Ecosystem										
Water Quality Conditions for Aquatic Life	X				X			X		
Stream Invertebrates					X	X	*	X		
Freshwater Fish Communities					X	X	X	*		
Open Space					X	*	*	X	X	*
Public Health										
Water Quality Conditions for Recreation	X				X			X		
Marine Beaches	X	X								
Shellfishing Areas	X	X								

Table 2. Timeline of datasets analyzed for the indicators in this report.

Indicators	Pre-Colonial	Colonial			Industrial				Suburbanization					Present						
	Before 1650	1700	1750	1800	1850	1890	1930	1940	1945	1960	1970	1975	1980	1985	1990	1995	2000	2005	2010	2015
Stressor Indicators																				
Climate Change																				
Temperature					1895–2015															
Precipitation					1895–2015															
Sea Level							1938–2017													
Landscape																				
Population					1850–2000															
															1990–2010					
Land Use					1850–2000															
																	2001–2011			
Impervious Cover																		2005–2011		
Wastewater Infrastructure					1850–2000															2015
Nutrient Loading													1982–1983				2000 – 2004		2007 – 2010	2013 – 2015
Chemical																				
Legacy Contaminants					1705–1998															
							1935–2015													
Emerging Contaminants								1940–2015												
Condition Indicators																				
Bay Ecosystem																				
Seagrasses					1840–1994													2006	2012	
Salt Marsh															1990–2010					
Benthic Habitat													1988						2008	
Estuarine Fish Communities											1959–2012									

Table 2 continued.

Indicators	Pre-Colonial	Colonial			Industrial				Suburbanization					Present						
	Before 1650	1700	1750	1800	1850	1890	1930	1940	1945	1960	1970	1975	1980	1985	1990	1995	2000	2005	2010	2015
Dissolved Oxygen																	2001–2015			
Chlorophyll											1972–2014									
Water Clarity											1972–2014									
Watershed Ecosystem																				
Water Quality Conditions for Aquatic Life																				2014
Stream Invertebrates																	2002–2014			
Freshwater Fish Communities																1992–2014				
Open Space																				2015
Public Health																				
Water Quality Conditions for Recreation																				2014
Marine Beaches																	2000–2015			
Shellfishing Areas																1995–2015				

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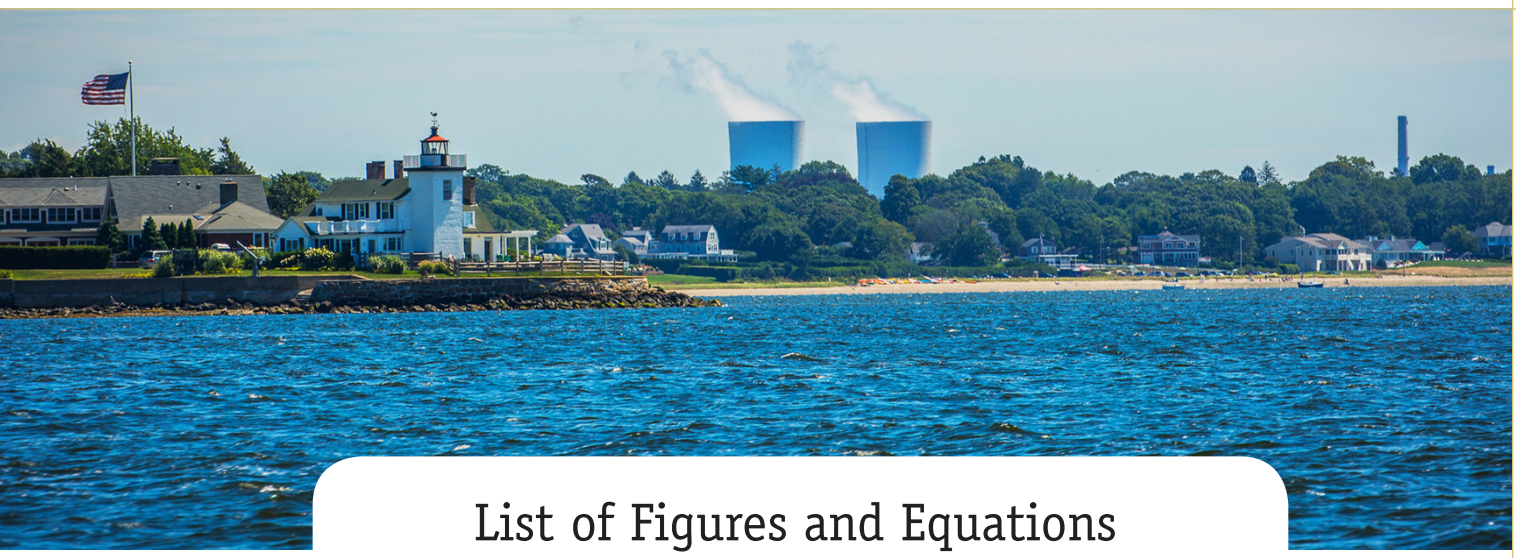
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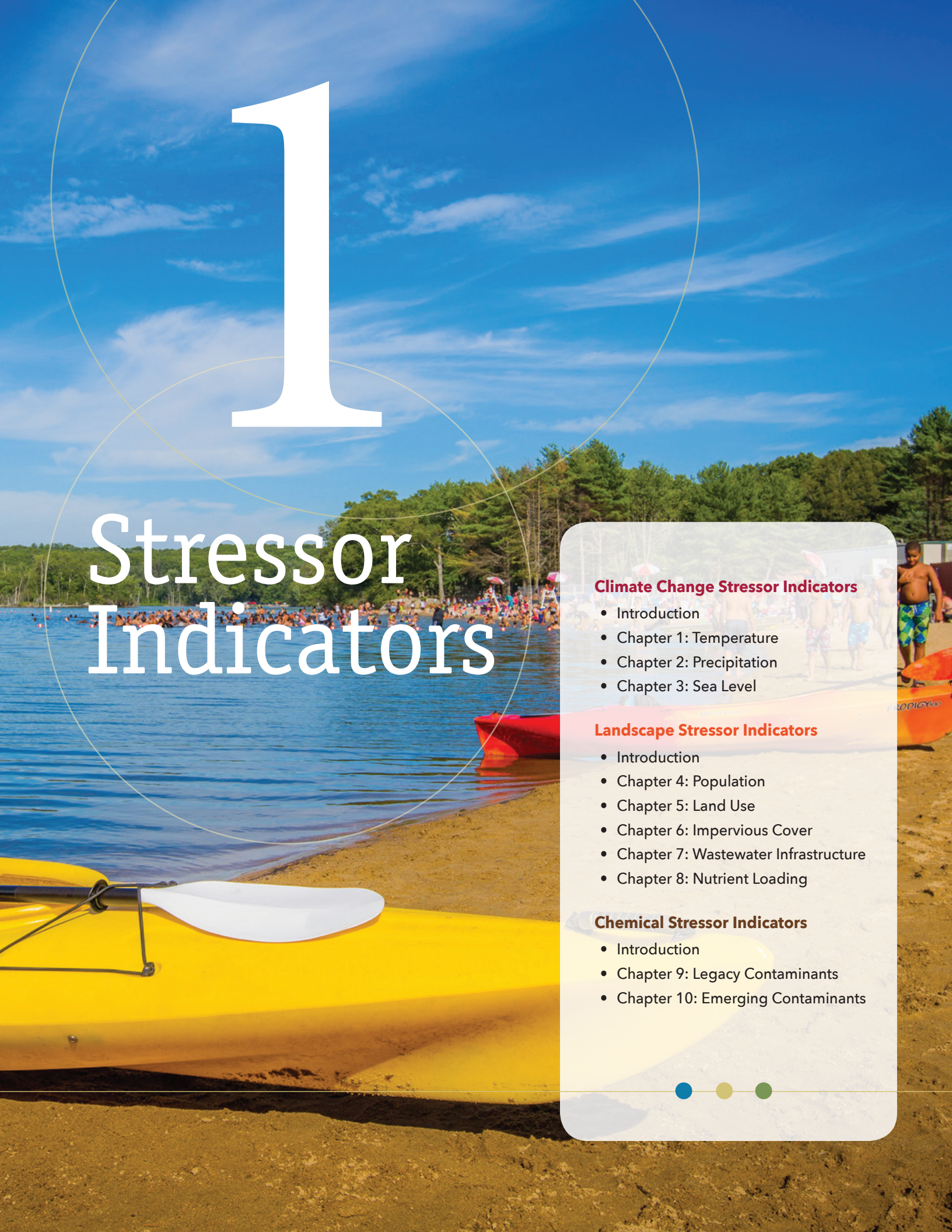
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1 Stressor Indicators

Climate Change Stressor Indicators

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Landscape Stressor Indicators

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Chemical Stressor Indicators

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State of Narragansett Bay and Its Watershed
2017 Technical Report

Climate Change Stressor Indicators

INTRODUCTION

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Photo: Napatree Point Conservation Area, Westerly, RI (Ayla Fox)

The *State of Narragansett Bay and Its Watershed* focuses on three climate change stressor indicators—temperature, precipitation, and sea level—because of their ubiquitous impacts on other stressor, ecosystem condition, and public health indicators. Based on datasets spanning nearly a century or more, analysis of these climate change indicators reveals long-term trends that are useful in understanding past environmental changes and projecting future changes in the Bay and its Watershed.

Climate science involves the examination of long-term average weather patterns resulting from the interactions between solar electromagnetic radiation and terrestrial, oceanic, and atmospheric systems on Earth. Solar electromagnetic radiation is 99.97 percent of the energy supporting Earth's systems (Taylor 2006). Short wavelength electromagnetic radiation emitted from the sun permeates the atmosphere, where 30 percent is reflected back into space, 20 percent is absorbed by the atmosphere, and 50 percent reaches the Earth's surface and is absorbed. When heat energy is reemitted from the earth's surface as long wavelength infrared radiation, it is trapped by greenhouse gases (GHG) in the atmosphere. Greenhouse gases are dispersed through the atmosphere, but together they function like a heat trapping layer analogous to the glass in a greenhouse, known as the greenhouse effect. The greenhouse effect is a natural process that keeps the Earth habitable, but accelerated GHG emissions amplify the greenhouse effect.

During the combustion of hydrocarbons stored in fossil fuels (coal, petroleum, natural gas), carbon dioxide (CO_2) and nitrous oxide (N_2O) are released into the atmosphere. These gases, along with methane (CH_4), are the major greenhouse gases in the atmosphere. When these gases are emitted through human activity (such as operating cars or creating fuel to power society), they are referred to as anthropogenic GHG. Although biogeochemical cycles, particularly the carbon cycle, marginally offset the GHG output, 40 percent of the CO_2 emitted since 1750 remains in the atmosphere, half of which was emitted within the past 40 years (IPCC 2014). While CO_2 levels have been higher in the past, this large addition of GHGs in such a short period of time is increasing the amount of heat energy trapped by the atmosphere, altering Earth's energy budget so the input of energy exceeds the output. The increased rate of change is unprecedented (IPCC 2014).

The alteration of Earth's energy budget is causing average global temperatures to increase, and it is altering global atmospheric and oceanic circulation patterns (Taylor 2006). The atmosphere has circulation cells that are determined by air temperature and the rotation of the Earth. Similarly, the world's oceans are connected through the global conveyer belt, an interoceanic network of currents influenced by salinity and thermal gradients (Taylor 2006). These well-established atmospheric and oceanic circulation patterns rely on the physical principle that a cold medium is denser than its warmer counterpart. This relationship drives the rising and falling of molecules through the atmospheric and oceanic columns. Therefore, increasing the atmospheric temperature alters the movement of the circulation cells and heats up the surface ocean, altering oceanic circulation. Interactions between oceanic and atmospheric circulation patterns determine the climate patterns around the world, and future patterns can be projected through climate modeling (Taylor 2006).

Climate models are a series of mathematical algorithms using established physical principles about the interrelationships of environmental factors and the combined influence of these factors on global climate over time. These models require a variety of environmental data inputs including, but not limited to, temperature, CO_2 concentrations, precipitation, and radiation levels. Depending on the type of model, the resulting projections may include atmospheric and oceanic circulation changes, sea level fluctuations, temperature fluctuations, precipitation pattern shifts, or biome shifts. The rate of anthropogenic GHG emissions is one of the most heavily considered environmental factors in climate modeling. The 2014 Intergovernmental Panel on Climate Change (IPCC) report presents a series of climate projections based on varying GHG addition rates (Figure 1) and examples of resulting environmental changes (Figures 2 and 3).

Figures 2 and 3 show IPCC model results for global temperature, sea level rise, and precipitation, which are all heavily impacted by anthropogenic GHG emissions to the atmosphere. These three categories correspond to the three climate change stressor indicators selected by the Narragansett Bay Estuary Program for reporting in the *State of Narragansett Bay and Its Watershed*. The IPCC climate models predict an increase in all three categories, which will have cascading effects on natural processes. The most immediate impact of GHG emissions is the increase in atmospheric temperature (Figure 2; see "Temperature" chapter). Atmospheric and oceanic circulation patterns rely on a consistent differential heating pattern in order to maintain the established cycling. When this differential heating process is interrupted through increased atmospheric temperature, shifts may occur in oceanic heat absorption, sea level rise, and precipitation patterns. The world's oceans

absorb more than 90 percent of the atmospheric heat resulting from anthropogenic emissions (IPCC 2014). Sea level will rise as temperature increases due to melting of glaciers and ice sheets as well as the thermal expansion of the water itself (Figure 2; see "Sea Level" chapter). Thermal expansion occurs when the water molecules heat up, move faster, and increase the volume of space occupied by the molecules. A warmer atmosphere holds more moisture, which has cascading impacts on circulation of the global atmosphere, according to the National Climate Assessment (Melillo et al. 2014). Current climate modeling projections show increasing global precipitation as a result of the warmer atmosphere (Figure 3; see "Precipitation" chapter).

The Narragansett Bay Watershed's dense population and low elevation present public health challenges and threats to local infrastructure. Excessive heat exposure is considered to be a public health threat, and it is likely to become a more prominent issue as the ambient air temperature increases due to climate change. In urban areas, impervious surfaces absorb a large amount of heat energy, further raising local temperature, which will be compounded by rising global temperatures. Community health research in New England found that the number of emergency room visits and deaths increased even when the heat index (a measure of how hot it feels when temperature and relative humidity are factored together) is below the current local guideline criteria of 100°F used to issue heat advisories (Wellenius et al. 2017).

Local infrastructure is threatened due to the risk of nuisance and catastrophic flooding from sea level rise and storm surges. This flooding is of specific concern for the low-lying communities that fringe the Bay. Wastewater treatment facilities are particularly threatened because they tend to be located along rivers and coastlines, putting them at substantial risk for overflow events that can introduce large volumes of inorganic and organic pollution–including

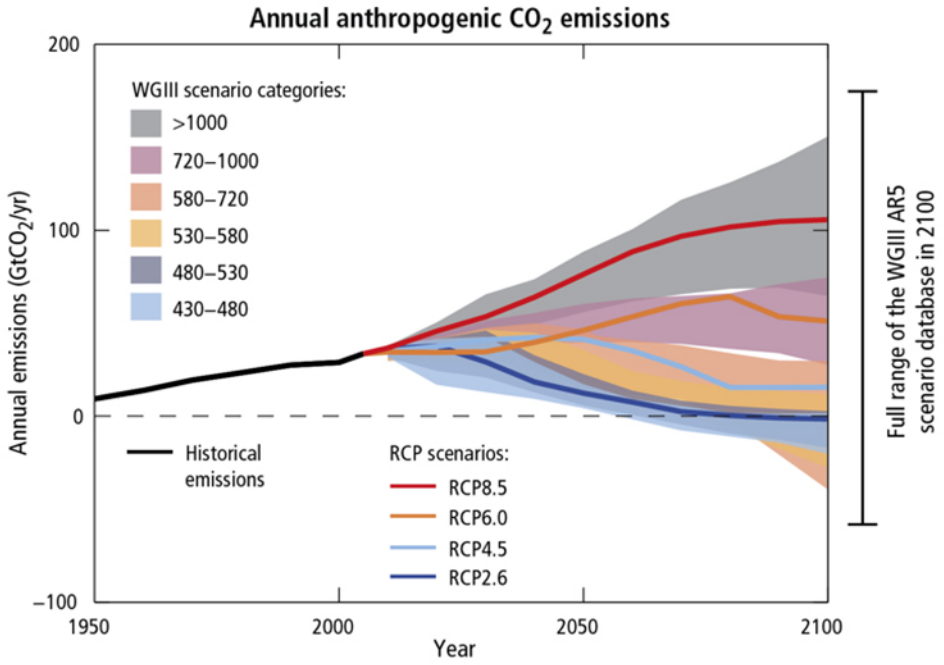


Figure 1. Summary of the GHG emission rates displayed as Representative Concentration Pathways (RCPs) that were produced through the WGIII climate model. The WGIII model considers anthropogenic emissions based on factors such as population size, economic activity, land use patterns, technology, and climate policy and was used to develop four possible emission projections for 2100 represented by RCPs. The RCPs produced through this model include RCP2.6 (stringent emission mitigation efforts), RCP4.5 to 6.0 (intermediate emission mitigation efforts), and finally RCP8.5 (exaggerated GHG emission rates). For context, if current emission trends are not adjusted, the effects are projected along the RCP6.0 and RCP8.5 trend lines. The four emission projections are shown in this graphic with correlating color shading, which indicates the atmospheric $CO_{2\text{-eq}}$ concentration in parts per million (ppm) with 95 percent confidence intervals. Source: IPCC (2014)

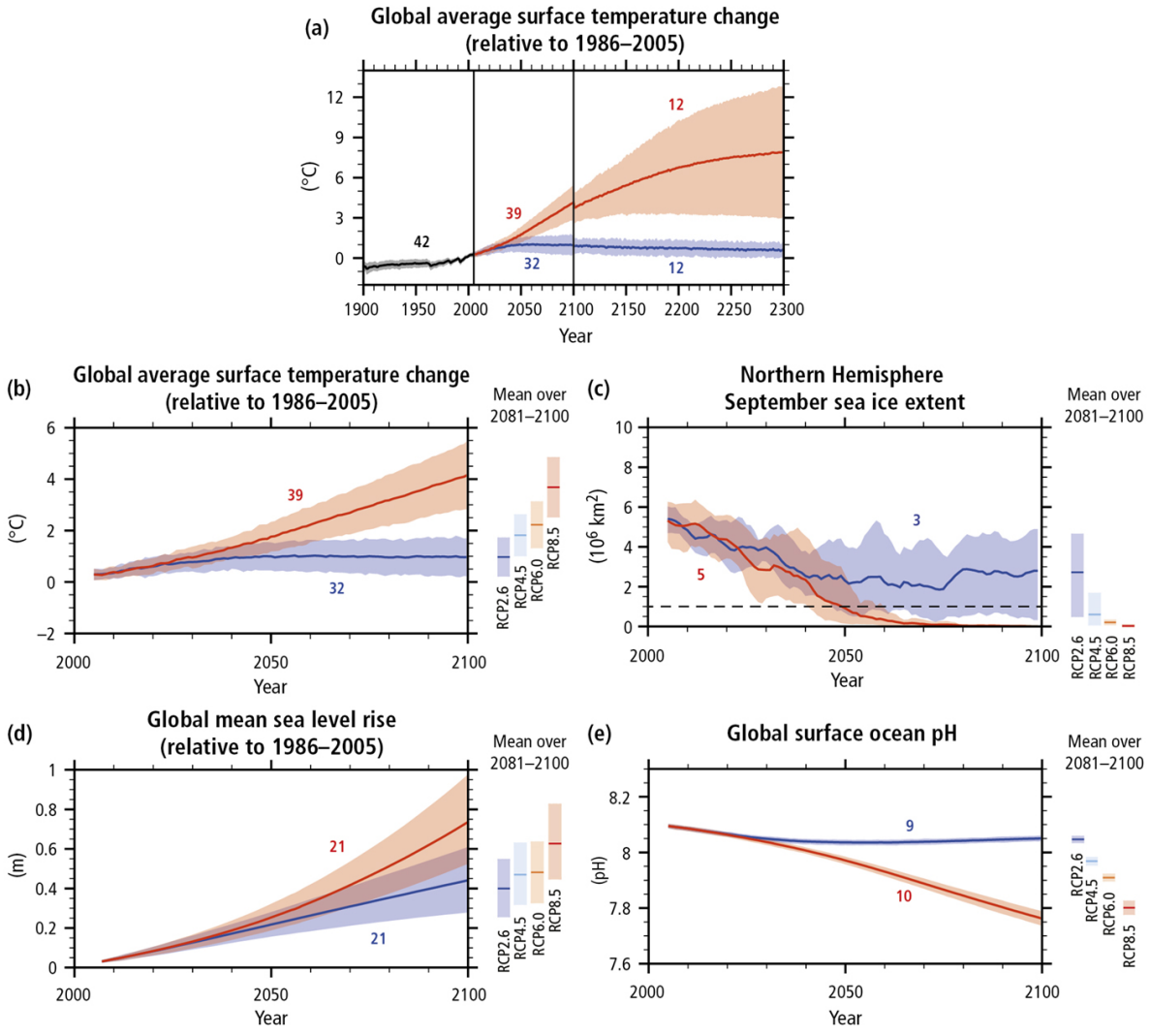


Figure 2. (a) Summary of the change in global mean temperature between 1900 and 2300 according to projections produced by the Coupled Model Intercomparison Project Phase 5 (CMIP5). The solid line indicates the multi-model mean, and the shading represents the 5 to 95 percent range across the distribution of models. Red shading represents scenarios in which GHG emissions are exaggerated or slightly mitigated. Blue shading represents scenarios in which GHG emissions are moderately to severely mitigated (see Figure 1). The number of CMIP5 models considered in each mean are included near the lines/shading. (b) Same as (a) but for the 2006 to 2100 period (relative to 1986 to 2005). (c) Change in northern hemisphere sea-ice extent in September (five-year running mean). The dashed line represents nearly ice-free conditions. (d) Change in global mean sea level. (e) Change in ocean surface pH. Source: IPCC (2014)

pathogens–into the local waterways, resulting in cascading impacts throughout the ecosystem (Woodard & Curran et al. 2017; MAOEEA 2011). Sea level is projected to rise as much as eleven feet (3.4 meters) at Newport, Rhode Island, by 2100 according to the National Oceanic and Atmospheric Administration's extreme scenario for climate change, accounting for rapid melting of Greenland and Antarctica ice sheets, thermal expansion of warming ocean waters, and other regional geological and oceanographic factors (Sweet et al. 2017; see "Sea Level" chapter).

The Narragansett Bay Watershed is exceptionally susceptible to the impacts of rising temperatures and increased precipitation as a result climate change. Air temperature is anticipated to increase by 2 to 6°C (5 to 10°F) by 2100, depending on the GHG emission scenario, resulting in warmer fresh and estuarine water temperatures (Bradley et al. 2016; see "Temperature" chapter). Increases in precipitation and changes in precipitation patterns will result. In the Narragansett Bay Watershed, projected increases in precipitation are one to three inches (2.5 to 7.6 centimeters) per decade, along with an increase in the frequency and intensity of storms. Additionally, precipitation events will cluster, meaning long periods of intense rain will be followed by intense periods of drought and drought-like conditions (Bradley et al. 2016; see "Precipitation" chapter).

Acidification in Narragansett Bay

In addition to its effects as a greenhouse gas, carbon dioxide is also causing changes to ocean chemistry in ways that are likely to affect the Narragansett Bay ecosystem. The oceans have absorbed between a quarter and a third of anthropogenic CO_2 emissions during the industrial age (Le Quéré et al. 2012). The resulting change in ocean pH, termed ocean acidification, is expected to continue under the emissions scenarios considered by the IPCC (Figure 2e) and has become a significant concern because of its potential impacts on ocean food webs and on economically important organisms such as shellfish in the coastal environment (Gledhill et al. 2015).

When CO_2 dissolves in seawater, it forms carbonic acid, which quickly splits into hydrogen and bicarbonate ions (Pilson 2013). Bicarbonate can then dissociate further into hydrogen and carbonate ions. Collectively, these components and reactions are known as the seawater carbonate system and constitute the largest carbon pool in the biosphere. A conceptual summary of the reactions is as follows:

1. carbonic acid ←→ bicarbonate + hydrogen
2. bicarbonate ←→ carbonate + hydrogen
3. carbonate + calcium ←→ calcium carbonate (e.g., aragonite)

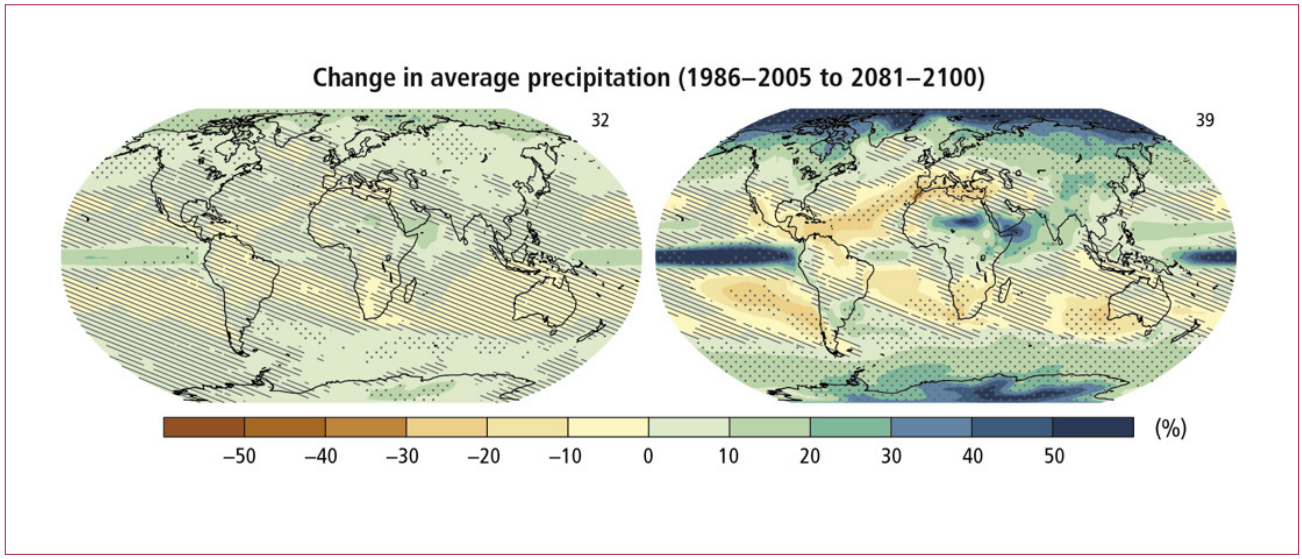


Figure 3. Coupled Model Intercomparison Project Phase 5 (CMIP5) multi-model mean projections for the change in annual mean precipitation, in percentages, during the 2081 to 2100 period under the RCP2.6 (left) and RCP8.5 (right) scenarios. Changes are shown relative to the 1986 to 2005 period. The number of CMIP5 models used to calculate the multi-model mean is indicated in the upper right corner of each panel. Source: IPCC (2014).

Acidity is determined by the concentration of hydrogen ions, including those involved in these reactions. Surplus hydrogen ions are naturally buffered when the bicarbonate side of reaction 2 is favored (the leftward reaction), but this buffering consumes carbonate ions and therefore pushes reaction 3 to the left (i.e., dissolution of calcium carbonate). An important biological consequence is that the change in pH and the reduction in carbonate ion availability—often expressed as "aragonite saturation state"—make the formation and maintenance of shells and other calcareous structures by bivalves and other calcifying organisms more energetically expensive or impossible under acidifying conditions.

These are basic chemical features of the marine environment that are observable and non-controversial, but the biological community's effect on and response to these changes are much more complicated. For example, certain types of marine plants and algae may be harmed by lower pH (i.e., higher acidity), but they may also benefit from the increased levels of CO_2 in the water, which they require for photosynthesis. There are many clear cases of extreme biological sensitivity to acidification among economically important coastal organisms such as clams, oysters, and scallops, but the biological responses of many other species are variable and difficult to predict (Kroeker et al. 2010, 2013). Thus, although species composition and relative abundance may change in the future, both the details and the ecosystem-level consequences (e.g., food production) are unpredictable (Grear et al. 2017).

The decrease in pH in the open ocean during the industrial age has been on the order of 0.1 to 0.2 pH units per century (Caldeira and Wickett 2003), which translates to more than a 25 percent increase in the concentration of hydrogen ions. pH is represented on a logarithmic scale, so small changes of a couple of tenths in pH are large in terms of concentration. The low diurnal and seasonal variations of pH in the open ocean have made that signal relatively easy to detect. In addition, the ocean is extremely important to the global carbon cycle, so scientists have been taking highly precise measurements of the carbonate system for many decades. However, the greater variability of pH in the coastal environment means that any existing long-term trends would likely require a larger number of samples to detect, creating a unique challenge for coastal biogeochemists. Current best practices for handling and analyzing samples from the open ocean are expensive and therefore may not be feasible for the high-frequency and spatially extensive sampling that would be necessary to detect decadal and spatial trends in the coastal environment.

In places like Narragansett Bay, the chemical environment is much more variable in time and space than in laboratory experiments or in the open ocean. Measurements of pH in Narragansett Bay have shown daily and seasonal fluctuations, but longer-term trends have not been detected. Daily and seasonal trends are expected in estuaries because pH is determined primarily by the amount of CO_2 that enters the water column. The two key sources for this CO_2 are absorption from the atmosphere and respiration by marine organisms. This respiratory contribution includes CO_2 from the very same microbial decay processes that consume dissolved oxygen and cause concern about hypoxia. Conversely, when photosynthetic organisms (e.g., phytoplankton) produce oxygen, they consume CO_2, creating daily and seasonal patterns of rising and falling pH that often mirror trends seen in oxygen measurements. A pH probe placed in the water column of Narragansett Bay, for example, will indicate rising pH (lower acidity) to levels sometimes exceeding 8.0 during the day when photosynthesis exceeds respiration, and then declining pH to 7.8–7.9 at night when photosynthesis stops (Hofmann et al. 2011). The same occurs on a seasonal basis, when net consumption of CO_2 through photosynthesis exceeds net respiration during early and mid-summer months (Dore et al. 2009, Gledhill et al. 2015). Like hypoxia, the development of low pH in bottom waters, sometimes to pH levels below 7.5, strongly depends on stratification (Wallace et al. 2014). These daily and seasonal dynamics are much less dramatic in the open ocean, where long-term declines in pH through the industrial age stand out clearly against a less noisy background of pH levels near 8.1 (Caldeira and Wickett 2003).

Changes in the pH of Rhode Island's coastal waters are indeed expected, due to the influx of poorly buffered ocean water to the region, expected increases in poorly buffered freshwater runoff from the watershed, and ongoing increases in atmospheric CO_2 concentration (Gledhill et al. 2015). Laboratory studies have already shown that lower pH (higher acidity) reduces the survival of larval finfish and shellfish (e.g., Talmage and Gobler 2010, Baumann et al. 2012), but field studies that help to explain and predict effects on these and other marine organisms within the Bay environment are lacking. A comprehensive program to study acidification processes and responses in the Bay is clearly needed, although it will be difficult to detect trends and perhaps to understand the processes contributing to any observed pH trends. Observations of reduced pH may arise from the atmospheric changes in CO_2 (see above), or through the production of respiratory CO_2 that occurs during low-oxygen events (hypoxia)

brought on by the discharge of excess nutrients (see "Dissolved Oxygen" chapter). The co-occurrence of hypoxia and reduced pH may be a larger threat to aquatic life than currently recognized (Gobler et al. 2014, Wallace et al. 2014). Reducing point-source nutrients through management actions (see "Nutrient Loading" chapter) may have the dual benefit of reducing hypoxic events as well as relieving the nutrient-driven component of acidification.

The Estuary Program did not dedicate a complete chapter of this report to address the important issues of acidification because of limited data and the uncertainty in predicting responses to acidification in the estuarine environment; however, it is clear that continued research is necessary. A number of researchers, many of whom interact through the Northeast Coastal Acidification Network, are conducting research in ecosystems of the Northeast, including Narragansett Bay, measuring carbonate parameters and working on guidelines for efficient methods of observation that will address some of the mystery and expense involved in monitoring coastal acidification.

Conclusion

As detailed in the chapters that follow, global projections of climate change are clearly reflected on a local scale in the Narragansett Bay Watershed. Air temperature and water temperature (both freshwater and marine) are currently increasing and are projected to continue their upward trends. More extreme precipitation events are occurring in the Northeast, coupled with changes in the types of precipitation (less snow, more rain). Tropical storms (e.g., hurricanes) are projected to be more intense. Changes in the timing or volume of river discharge are likely to alter buffering against the acidifying effects of organic loading, eutrophication, and absorption of atmospheric carbon dioxide. With atmospheric and oceanic warming, thermal expansion of the ocean, and melting ice, the rate of relative sea level rise is projected to increase dramatically, with rates expected to be especially high along the mid-Atlantic and northeastern U.S. coast. In response to the changing climate, there will be a corresponding alteration in the species composition, structure, and function of Narragansett Bay Watershed ecosystems, coastal and inland flooding will intensify, and sectors of the regional economy will be impacted (e.g., fisheries, agriculture, tourism), among other projected changes to the environment and society.

Acknowledgments

This introduction to climate change stressors was written by Courtney Schmidt, Staff Scientist, and Miranda Rayner, Scientific Program Intern, with Narragansett Bay Estuary Program. The coastal ocean acidification section was written by Jason Grear with the United States Environmental Protection Agency, Atlantic Ecology Division.

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State of Narragansett Bay and Its Watershed
2017 Technical Report

Climate Change Stressor Indicators

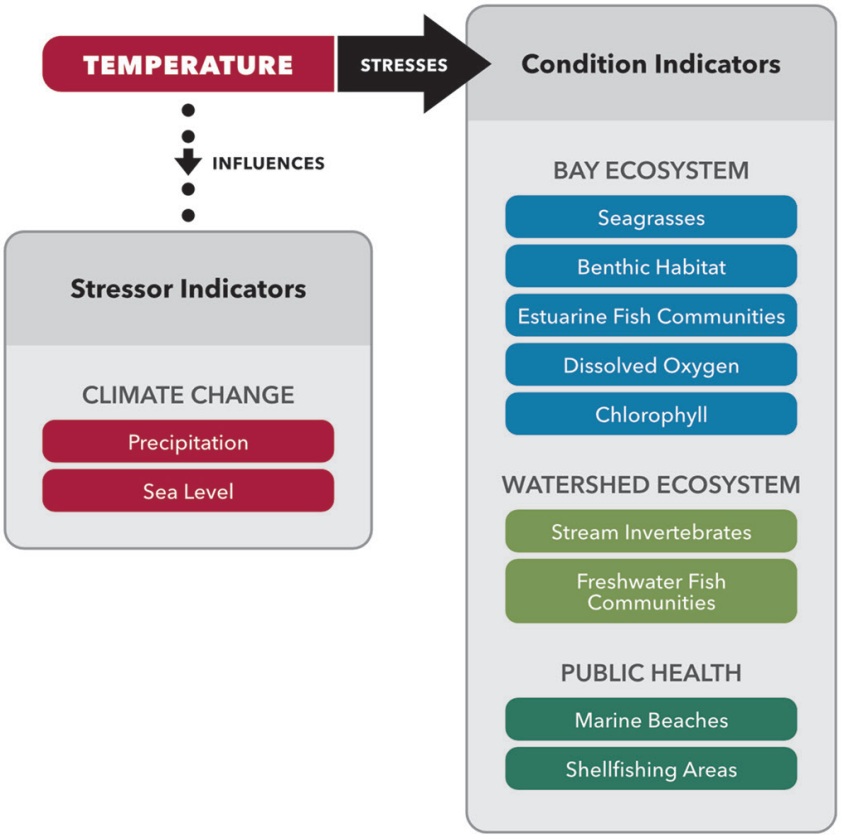
CHAPTER 1: TEMPERATURE

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NARRAGANSETT BAY ESTUARY PROGRAM

Overview



BACKGROUND

- Warming air, river, and estuarine water temperatures will increasingly affect biological communities (such as phytoplankton [chlorophyll], seagrasses, fish, and stream invertebrates), physical conditions (such as dissolved oxygen), and public health (pathogen contamination of marine beaches and shellfishing areas).

KEY FINDINGS

- ***Trends:*** From 1960 to 2015, air temperature in Rhode Island increased 0.031°C (0.06°F) per year, and in Massachusetts it increased 0.023°C (0.04°F) per year. This led to a total warming of 1.3 to 1.7°C (2.3 to 3.1°F) from 1960 to 2015 across the Narragansett Bay Watershed. Over the same time period, estuarine water temperature increased 0.028 to 0.031°C (0.05 to 0.06°F) per year, for a total warming of 1.5 to 1.7 °C (2.7 to 3.1°F).
- ***Projections:*** Climate projections suggest that average air temperature will increase approximately 2 to 6°C (5 to 10°F) by 2100 depending on the greenhouse gas emission rate. Local studies show that for every 1°C (1.8°F) increase in air temperature, the water temperature in rivers increases 0.6 to 0.8°C (1.1 to 1.4°F).

Introduction

Temperature strongly influences public health and environmental conditions on global and local scales, making it an important indicator of climate. It is also an easily measured parameter, and many organizations routinely monitor air and water temperatures in the Narragansett Bay Watershed, such as the National Oceanic and Atmospheric Administration (NOAA), the University of Rhode Island's Graduate School of Oceanography (GSO), and the United States Geological Survey (USGS).

Local air and water temperature have been measured and trends studied for over a century. In an analysis of air temperature data from the Providence station of the National Weather Bureau, Pilson (2008) found an overall increase of 0.94°C (1.7°F) between 1905 and 2005. The Narragansett Bay Estuary Program's 2009 status and trends report stated that the Bay's annual average surface water temperature increased by 1°C (1.8°F) from 1959 to 2007 (NBEP 2009). Other studies that reported water temperature records from the URI-GSO fish trawl and phytoplankton surveys, the NOAA tide gauge at Newport, and the Brenton Reef lightship (no longer active) found a similar increase of 0.91 to 1.2°C (1.6 to 2.2°F) between 1960 and 2005, with the majority of the warming occurring since 1970 (Nixon et al. 2009, Smith et al. 2010). The trends observed in the Narragansett Bay Watershed are consistent with those in other coastal locations (e.g., Woods Hole, Massachusetts; Figure 1), and local researchers (Nixon et al. 2004, Oviatt 2004, Pilson 2008) suggest that the rate has increased starting in the 1960s or 1970s.

Increases in river water temperature appear to be driven primarily by rising air temperatures (Morrill et al. 2005, Seekell and Pace 2011). In Massachusetts, Morrill and colleagues (2005) analyzed water temperature changes in three rivers and concluded that the rivers' temperatures increased 0.6 to 0.8°C (1.1 to 1.4°F) for every 1°C (1.8°F) increase in air temperature. This implies that river temperatures will likely increase 2 to 3°C (3.6 to 5.4°F) if air temperature increases 3 to 5°C (5.4 to 9.0°F) over the next century (Morrill et al. 2005). Data from the Hudson River in New York support the conclusions of Morrill and colleagues, as the river has warmed 0.95°C (1.7°F) from 1946 to 2008, driven by warming air temperatures (Seekell and Pace 2011).

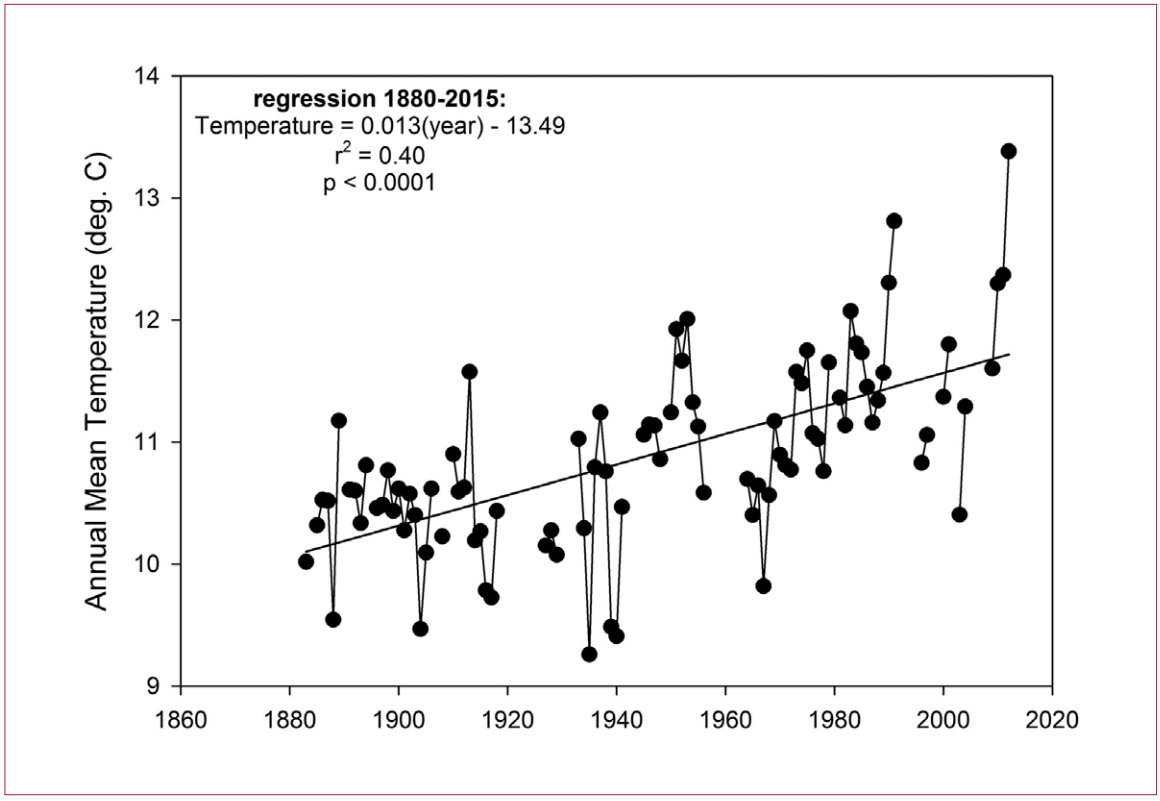


Figure 1. Annual average water temperatures at Woods Hole, Massachusetts, from 1880 to 2015. Black line is a linear regression for the entire time period, 1880 to 2015. Data provided by J. Manning (NOAA).

Urban development throughout the Narragansett Bay Watershed is projected to magnify the impact of rising temperatures on human and environmental health (USEPA 2016). Urbanization has been shown to raise water and air temperatures through an increase in impervious surfaces (heat island effect and heated runoff) and the regulation and impoundment of freshwater (Mohan et al. 2013). While this accelerated warming from urban landscapes is occurring year-round, wintertime air and water temperatures are warming the most rapidly (e.g., Fulweiler et al. 2015).

The trends in air and water temperatures, globally and within the Narragansett Bay Watershed, are significant because they have many effects on ecological processes. For example, warmer winters have changed the timing of the winter-spring phytoplankton bloom, which then affects food supply for zooplankton and other species (Oviatt 2004, Nixon et al. 2009, Smith et al. 2010; see "Chlorophyll" chapter). Many of the indicators presented in other chapters of this report are influenced by warming air and water temperatures, including biological (phytoplankton, benthic habitat, seagrasses, fish communities, stream invertebrates), physical (dissolved oxygen), and public health (bacterial levels at marine beaches and in shellfish harvest areas) indicators.

In this chapter, the Narragansett Bay Estuary Program analyzes temperature trends in light of their impacts on ecosystem condition and public health, and reviews climate change projections for Narragansett Bay and its Watershed.

Methods

To analyze air and water temperature trends in the Narragansett Bay Watershed, the Estuary Program used datasets obtained from several sources: NOAA's National Climate Data Center (NOAA 2016) for air temperature, Fulweiler et al. (2015) for estuarine water temperature, and the USGS and the Narragansett Bay Commission for riverine water temperature (Table 1).

The NOAA air temperature datasets spanned a period of 110 years (1895 to 2015) for Providence, Rhode Island, and 66 years (1949 to 2015) for Worcester, Massachusetts. The Estuary Program analyzed the complete datasets using linear regressions to quantify the trends. Linear regressions were also performed on the data from 1960 and later because other studies have suggested that temperatures began to rise more quickly after 1960 (Nixon et al. 2004, Oviatt 2004, Pilson 2008). That year was also, coincidently, when estuarine water temperature records started to be collected continuously in Narragansett Bay (Fulweiler et al. 2015), and focusing on the years after 1960 enabled comparison of air and estuarine water temperatures.

For estuarine water temperature, the Estuary Program used data for a 50-year period (1960 to 2010) that had been compiled by Fulweiler et al. (2015) from three sources: the University of Rhode Island Graduate School of Oceanography (GSO) phytoplankton surveys, the GSO fish trawl surveys, and the NOAA tide station at Newport, Rhode Island.

Table 1. Sources of temperature data, including length of record and sampling frequency.

Data Source	Length of Record	Sampling Frequency
Air Temperature		
NOAA Climate at a Glance (RI)	1895–2015 (110 years)	Daily
NOAA Climate at a Glance (MA)	1949–2015 (66 years)	Daily
Estuarine Water Temperature		
GSO phytoplankton survey	1960–2010 (50 years)	Weekly
GSO fish trawl survey	1960–2010 (50 years)	Weekly
NOAA tide station - Newport	1960–2010 (50 years)	Daily until 1996; Every 15 minutes from 1996 to 2010
River/Stream Water Temperature		
USGS – Millville, MA	1969–1981 and 2011–2014	Daily
USGS – Scituate Reservoir Region	2001–2006 and 2010–2013	Daily
Narragansett Bay Commission – RI and MA rivers	2007–2015	Bi-weekly

Because continuous multi-decade datasets for river and stream water temperatures in the Narragansett Bay Watershed were not available, the Estuary Program used three datasets that covered shorter time periods. Data from the USGS station in Millville, Massachusetts, covered two periods: 1969 to 1981 and 2011 to 2014. USGS data for twelve rivers and streams in the Scituate Reservoir region of Rhode Island included 2001 to 2006 and 2010 to 2013, and the Narragansett Bay Commission collected data in eleven rivers and streams discharging to Narragansett Bay during 2007 to 2015. The Estuary Program did not analyze trends in the river and stream datasets because the length of the records was not sufficient.

Status, Trends, and Projections

STATUS AND TRENDS

Air Temperature

Air temperature in Massachusetts increased by 1°C (1.8°F) from 1949 to 2015 at a rate of 0.14°C (0.3°F) per decade (Figure 2). During the period from 1960 to 2015, air temperature increased at a faster rate of 0.23°C (0.4°F) per decade for a total increase of 1.4°C (2.5°F) (Figure 2).

Air temperature in Rhode Island increased by 1.9°C (3.4°F) over the entire period at an average rate of 0.16°C (0.3°F) per decade (Figure 3). During the period from 1960 to 2015, air temperature increased at a faster rate of 0.31°C (0.6°F) per decade for a total increase in air temperature of 1.6°C (3.0°F).

Estuarine and Freshwater Temperature

In Narragansett Bay, the annual average temperature of estuarine waters, as measured in the three different datasets, increased a total of 1.4 to 1.6°C (2.5 to 2.9°F) from 1960 to 2010 with an average of 0.3°C (0.5°F) per decade (Figure 4).

River and stream water temperatures measured from 2002 to 2013 by the USGS in Rhode Island averaged 11°C (52°F) (Figure 5), while temperatures measured by the USGS at Millville, Massachusetts, from 1969 to 2014 and by the Narragansett Bay Commission from 2007 to 2015 at multiple sites in the Narragansett Bay Watershed each averaged 13°C (55°F) (Figures 6 and 7). The Estuary Program did not calculate trends for the river and stream data because the lengths of records were not sufficient. However, it is noted that water temperature at the USGS station in Millville appears to have declined (Figure 6). Given the large data gap between 1981 and 2011, and the limited number of data points, it cannot be stated with any certainty that this is a trend.

PROJECTIONS

On a regional scale, the University of Massachusetts Climate System Research Center produced climate model projections using two emission rates to project potential changes in air temperature for Rhode Island and Massachusetts (Bradley et al. 2016). Because the temperature projections for Rhode Island and Massachusetts were comparable, only the Rhode Island projection is presented here. Under the lower emission scenario, temperature would increase by 2.0 to 3.0°C (3.6 to 5.4°F) by 2100 (Figure 8); however, the lower emission scenario is possible only if substantial efforts are made to mitigate greenhouse gas emission rates during future development so that emission rates are comparable to emission rates in 2000 (Bradley et al. 2016). Current greenhouse gas emissions are tracking along the higher emission rate scenario, which is projected to cause a local temperature increase of 4.0 to 6.1°C (7.2 to 11°F) by 2100 (Figure 8) (Bradley et al. 2016).

The higher projected temperature increase by 2100 is expected to create a climate in New England like that of present-day Virginia. The warmest recent winters in southern New England may be representative of the coldest winters experienced in 2100, and the warmest recent summers may be cool in comparison to the summers experienced in 2100 (Bradley et al. 2016). By 2100, the average winter temperature in Rhode Island is projected to approach 10°C (50°F), and the summer mean temperature is projected to be just below 32°C (90°F) (Figure 8) (Bradley et al. 2016). For comparison, the average maximum temperature from 1950 to 2005 in winter was 4°C (39°F) and in summer was 26°C (79°F) (USGS 2016).

Marine and freshwater temperatures are coupled with air temperature, and projected increases in air temperature carry implications for marine and freshwater systems. According to Sheaman and Lentz (2009), average seawater temperatures near the coast tend to follow, with a time lag, trends in air temperature. Coastal seawater temperatures are affected primarily by the sea-air interface and by currents that flow along the continental shelf, meaning that local waters are highly influenced by regional air temperature shifts and by the warming of currents flowing from the Arctic (Shearman and Lentz 2009). Freshwater systems are affected similarly by regional air temperature shifts (Morrill et al. 2005, Seekell and Pace 2011). Based on the trend from 1960 to 2012 in the estuarine water temperature

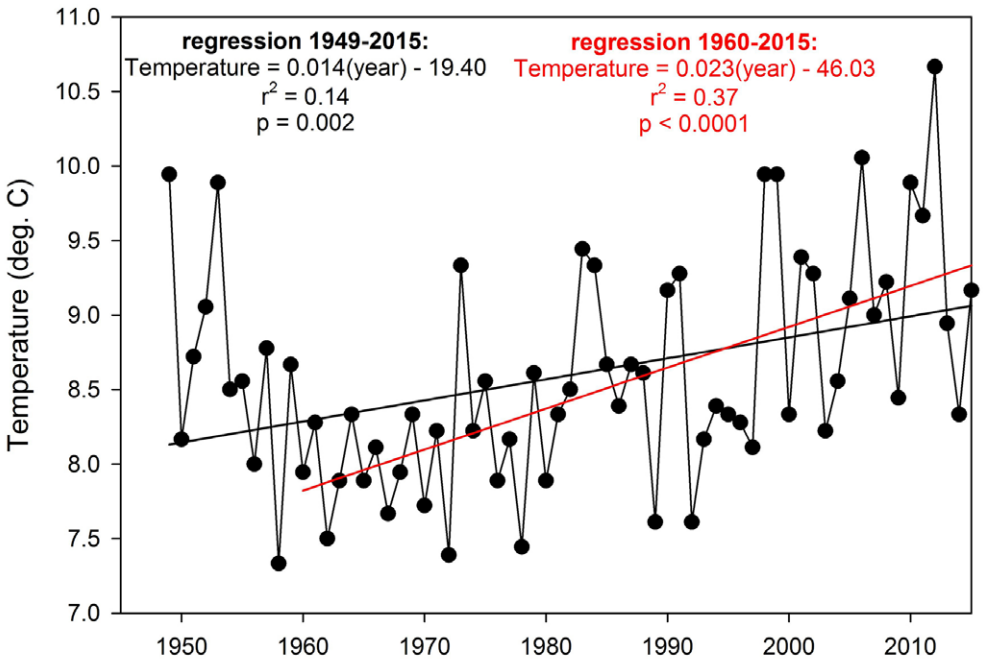


Figure 2. Annual mean air temperatures at Worcester, Massachusetts, from 1949 to 2015. Black line is regression for the entire dataset, and red line is regression for dataset from 1960 to 2015. Source: NOAA (2016)

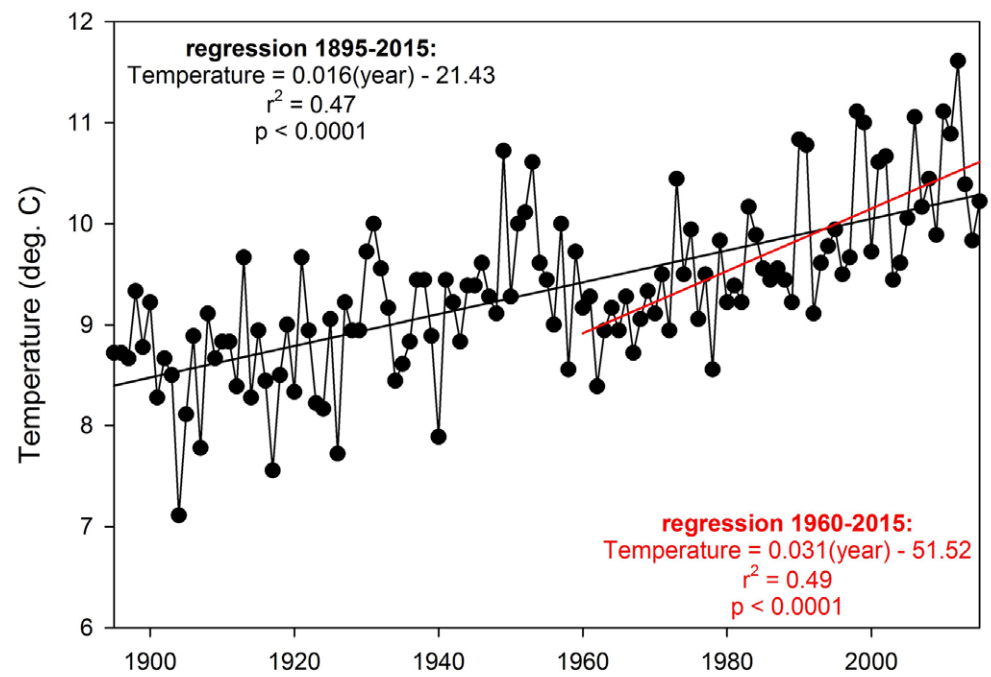


Figure 3. Annual mean air temperatures at Warwick, Rhode Island, from 1895 to 2015. Black line is regression for the entire dataset, and red line is regression for dataset from 1960 to 2015. Source: NOAA (2016)

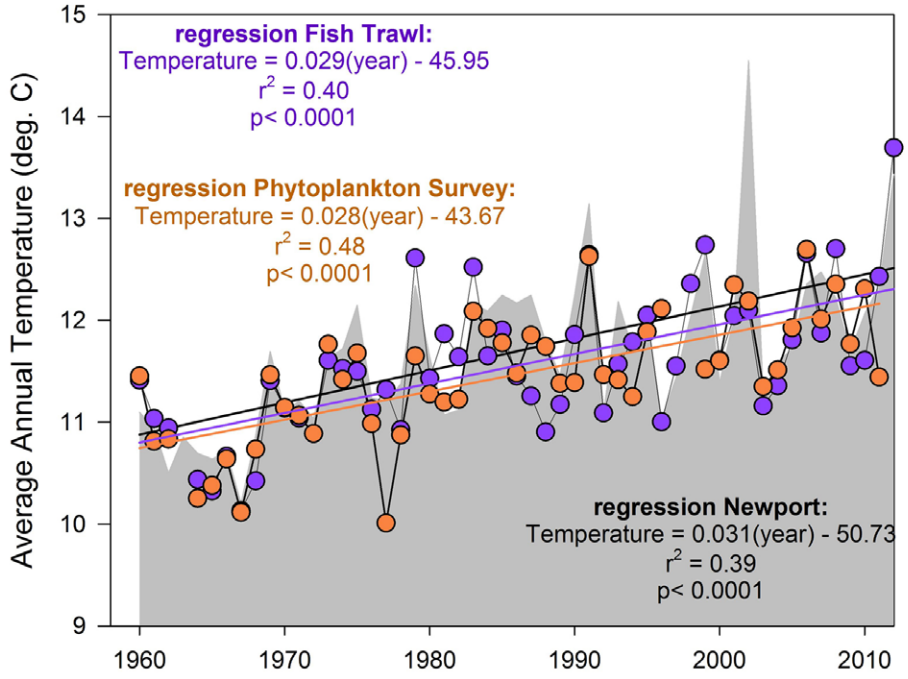


Figure 4. Annual mean surface water temperatures in Narragansett Bay from 1960 to 2010, including datasets from three sources: URI GSO phytoplankton survey (orange circles and line), URI GSO fish survey (purple circles and line), and NOAA station at Newport (gray shaded area and black line). Figure based on Fulweiler et al. (2015).

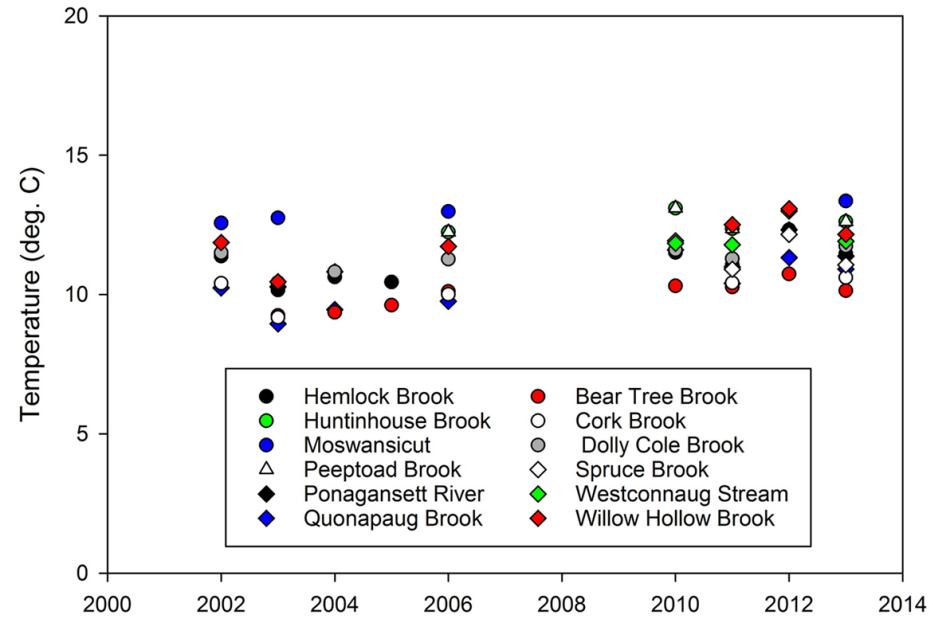


Figure 5. Annual mean river/stream water temperatures in the Scituate Reservoir region (Rhode Island) from 2002 to 2006 and 2010 to 2013. Source: USGS

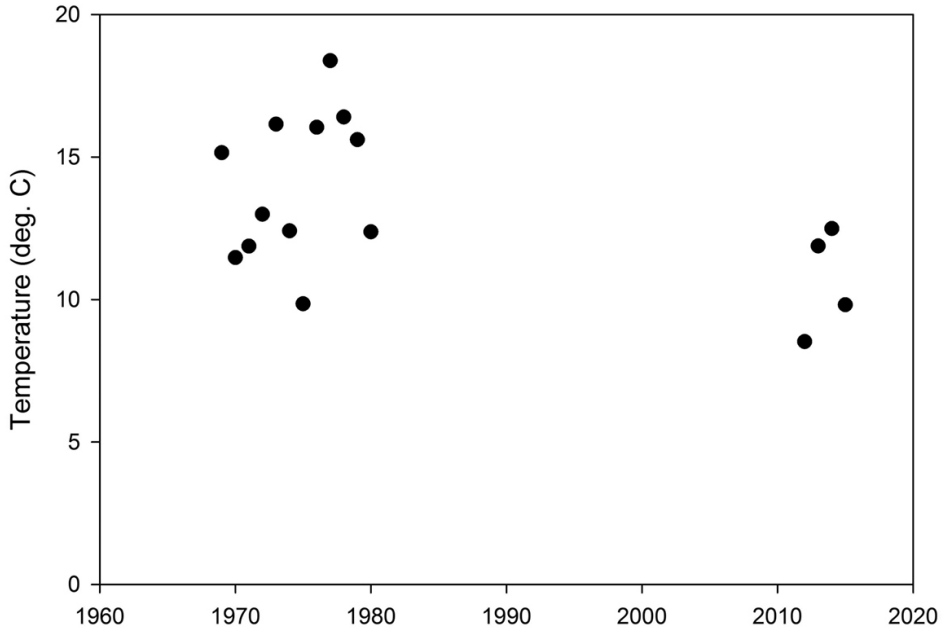


Figure 6. Annual average river/stream water temperatures at Millville, Massachusetts, from 1969 to 1981 and 2011 to 2014. Source: USGS

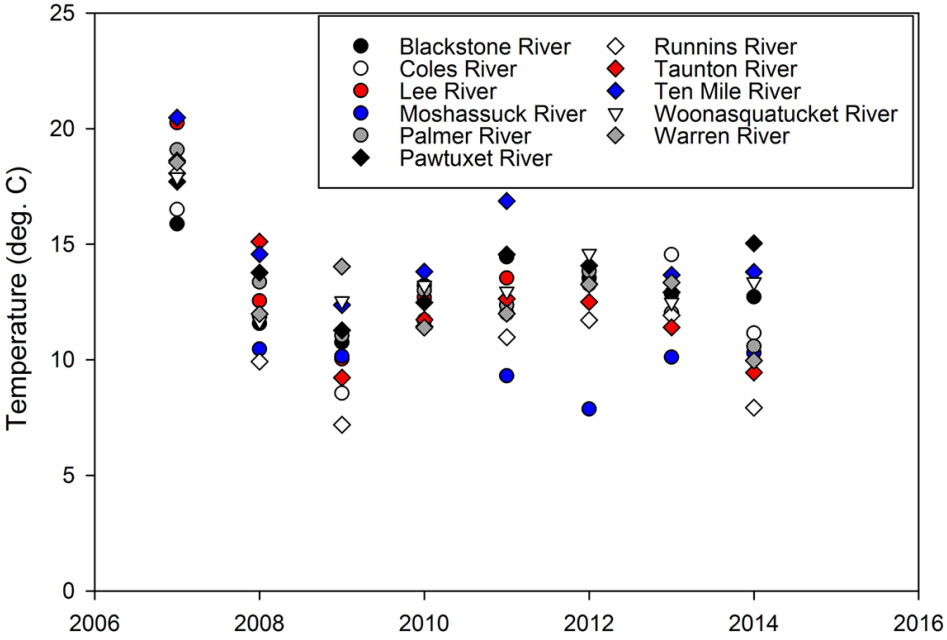


Figure 7. Annual mean river/stream water temperatures from 2007 to 2014 in eleven rivers and streams discharging to Narragansett Bay. Source: Narragansett Bay Commission

dataset for Narragansett Bay, the water temperature in the Bay may increase at a rate of 0.028 to 0.031°C (0.05 to 0.06°F) per year for a total of 2.7 to 3.2°C (5 to 6°F) over the next 100 years (Figure 4). However, that increase may be an underestimate, as it does not account for the accelerated rate of temperature rise that may occur (Seekell and Pace 2011, Melillo et al. 2014, Bradley et al. 2016).

Discussion

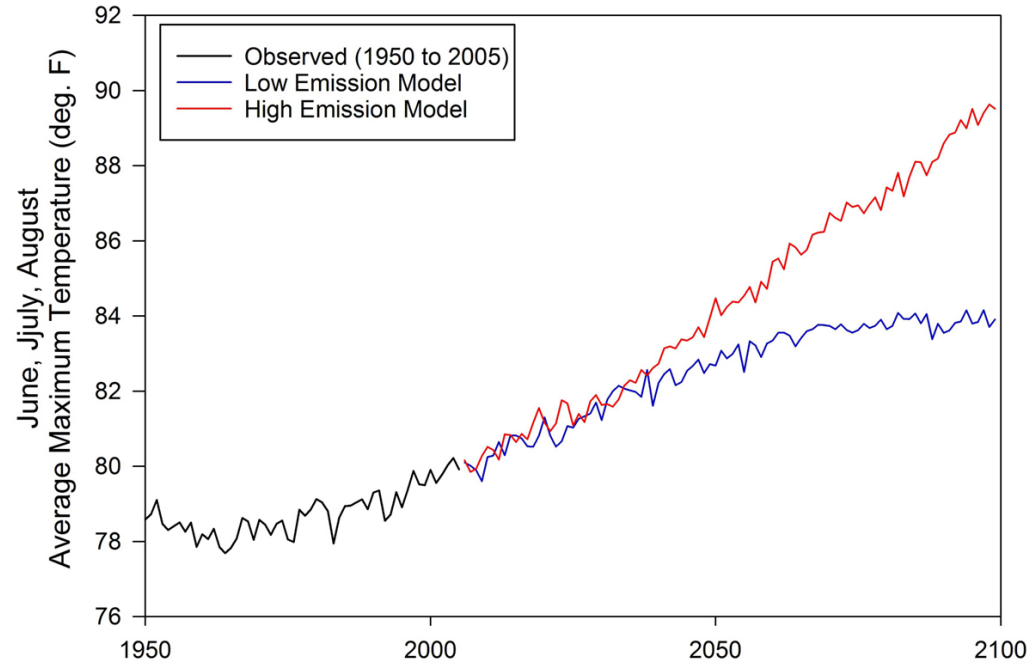
Narragansett Bay and its Watershed have experienced substantial increases in air and water temperatures in recent decades, and projected future warming will likely lead to greater impacts on human health and ecological processes. Regional climate change projections suggest an increase in air temperature of 2.8 to 5.6°C (5.0 to 10.0°F) by 2100 from temperatures experienced in 1950 (Figure 8; Bradley et al. 2016). Estuarine water temperatures in Narragansett Bay showed trends similar to those observed at Woods Hole, Massachusetts (Figure 1). The 1960s were a colder-than-normal decade, so an increase from that point forward is expected. Air and estuarine water temperature increased from 1960 to 2015 at a rate of 0.23 to 0.31°C (0.4 to 0.6°F) per decade. If the temperature trends continue, air and water will warm by 2.3 to 3.1°C (4 to 6°F) over the next 100 years (Figures 2 and 3). The current climate projections from Bradley and colleagues (2016) are 1 to 2°C (3 to 5°F) higher than these current trajectories. Greenhouse gas emission rates are driving the projected increases, and if emission rates continue to be high, the Narragansett Bay Watershed may experience temperature increases closer to the 6°C (11°F) mark.

In the Narragansett Bay Watershed, rising air and water temperatures are affecting biological communities, physical conditions, and bacterial levels at beaches and in shellfishing areas (see chapters on "Chlorophyll," "Seagrass," "Estuarine Fish Communities," "Freshwater Fish Communities," "Stream Invertebrates," "Dissolved Oxygen," "Marine Beaches," and "Shellfishing Areas"). In biological communities, species often rely on environmental cues, specifically temperature, to regulate lifecycle functions from reproduction to feeding. Increases in temperature affect the timing of specific functions, which in turn can affect the entire ecosystem. For example, as waters have warmed and days have become cloudier (Pilson 1991, Borkman 2002), the winter-spring diatom bloom has been delayed and diminished because of grazing pressure (brought on by warmer temperatures) or decreased light (from cloudier days) (Oviatt 2004, Nixon et al. 2009; see "Chlorophyll" chapter). The lack of a winter-spring bloom has decreased the amount of food for benthic organisms, diminishing the connection between the water column and benthos in Narragansett Bay. Similarly, the freshwater systems within the Narragansett Bay Watershed are experiencing a decrease in invertebrate populations, with temperature changes identified as a contributing factor (see "Stream Invertebrates" chapter).

Increasing water temperatures are related to a shift in fish communities within Narragansett Bay (Oviatt et al. 2003, Oviatt 2004; see "Estuarine Fish Communities" and "Freshwater Fish Communities" chapters). Currently, Narragansett Bay is a boundary area between northern and southern marine/estuarine species. The warming trends are shifting this boundary northward, changing the species that are present in the Bay (Nixon et al. 2009, Smith et al. 2010). The trends could be reflected similarly in terrestrial and freshwater aquatic systems (Matthews et al. 2004, Brooks 2009). For estuarine fish, multiple studies have documented a decrease in cold-water demersal species (e.g., winter flounder) and an increase in warm-water pelagic species (e.g., butterfish) (e.g., Oviatt 2004, Collie et al. 2008). Similarly, as air and river/stream temperatures warm, environmental managers are concerned that cold-water fish habitat in freshwater streams, now common in the Narragansett Bay Watershed, may decline. Cold-water habitats could be replaced by warm-water habitats, decreasing the amount of habitat suitable for brook trout and altering the ecosystem as cold-water species give way to warm-water species.

Seagrasses in Narragansett Bay can become stressed if water temperatures exceed 25 to 30°C (72 to 86°F) for extended periods, potentially leading to decreases in important habitat (Short and Neckles 1999, Bintz et al. 2003; see "Seagrasses" chapter). Warmer waters can also shift seagrass reproduction strategy from root and rhizome growth to exclusively seed propagation, which increases the vulnerability of the population.

Warmer water contains less dissolved oxygen, and when dissolved oxygen levels decline significantly then hypoxic conditions can result. Fish, crabs, and other mobile species can leave the hypoxic area, but sessile species cannot. Low oxygen levels also affect the sediment-water interface, changing the types of species that can thrive there (Diaz 2001; see "Dissolved Oxygen" and "Benthic Habitat" chapters). The possibility of climate change leading to more widespread hypoxia is a major concern for coastal environmental managers throughout the world (Rabalais et al. 2014).



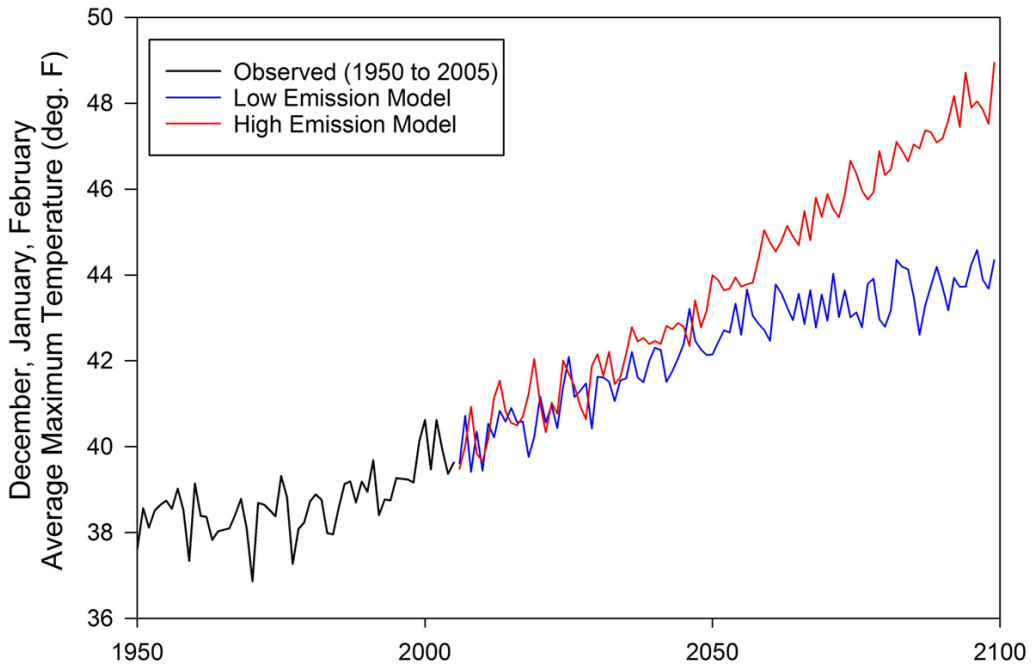


Figure 8. Seasonal air temperature projections for Rhode Island from 1950 to 2100. Results are from a USGS climate model (Bradley et al. 2016, USGS 2016). Black line represents air temperature data collected in Rhode Island from 1950 to 2005. Red line represents projected temperatures based on high greenhouse gas emission rates that are comparable to present-day emission rates. Blue line represents projected temperatures based on a lower emission rate that would be a result of reducing present-day emission rates through the introduction of energy-efficient technologies (Bradley et al. 2016, USGS 2016). Source: USGS (2016)

Projected increases in water temperature will affect pathogens in Narragansett Bay. State agencies monitor marine beaches and commercial shellfishing areas for pathogens that are harmful to humans, using the bacteria Enterococci and fecal coliform as pathogenic indicators (see "Marine Beaches" and "Shellfishing Areas" chapters). Beaches are closed for all recreational activity when bacterial counts exceed established values (MADPH 2016, RIDOH 2016). Shellfishing areas tend to be closed based on precipitation amount, which is correlated with pathogen loading (RIDEM 2015). Because warmer water promotes bacterial growth, monitored beaches may experience more closures, and shellfishing areas may be closed outside of heavy rainfall events due to high background bacterial counts.

Environmental management and research groups are tracking the impacts of changing temperature trends on the biological and physical indicators of the Narragansett Bay Watershed. The rising temperatures are projected to have a variety of impacts on the integrity of the Bay and its Watershed, and many of these impacts are not yet well understood. Further research into these issues will help regulatory agencies implement management practices as effectively as possible.

Data Gaps and Research Needs

- Continuous temperature data in rivers and streams is a major data gap. A sustained river/stream temperature monitoring network in the Narragansett Bay Watershed needs to be established to include long-term monitoring stations in key locations. Lacking that, a conversion factor between air temperature and freshwater temperature (e.g., Morrill et al. 2005) would estimate changes in stream temperature from existing air temperature data.
- An assessment is needed to determine whether the existing network of fixed sites for collecting water temperature data continuously in the Bay provides adequate information for tracking the long-term changes. Data collection in embayments is a recognized gap. Prior monitoring strategies recommended building capacity to periodically assess water quality conditions in such areas. Rising temperatures are likely to affect shallow and highly urbanized basins before the rest of Narragansett Bay (Oczkowski et al. 2015).
- Further analysis of trends in air and water temperature datasets is needed. While the Estuary Program and other researchers (e.g., Nixon et al. 2004, Oviatt 2004, Pilson 2008) used 1960 as a start date for regressions, the perception of a recent acceleration in warming may be a consequence of using that date. An analysis of the datasets could determine whether this trend is meaningful and potentially identify other significant trends, such as seasonal variability (e.g., freeze/thaw cycles).
- Experiments using mesocosms are needed to determine how the Narragansett Bay ecosystem may be affected by climate change (e.g., Bintz et al. 2003). Latitudinal gradient studies would be beneficial to predict future ecosystem and species shifts resulting from changing temperature trends (e.g., Crosby et al. 2017).

Acknowledgments

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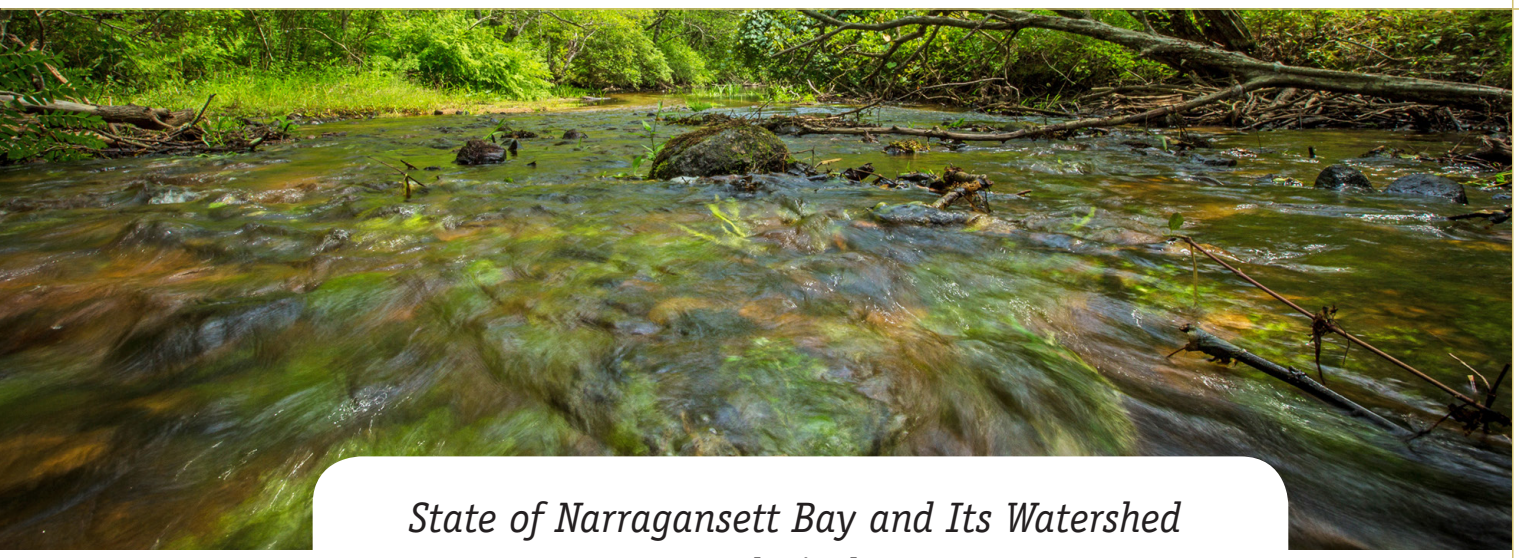
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State of Narragansett Bay and Its Watershed
2017 Technical Report

Climate Change Stressor Indicators

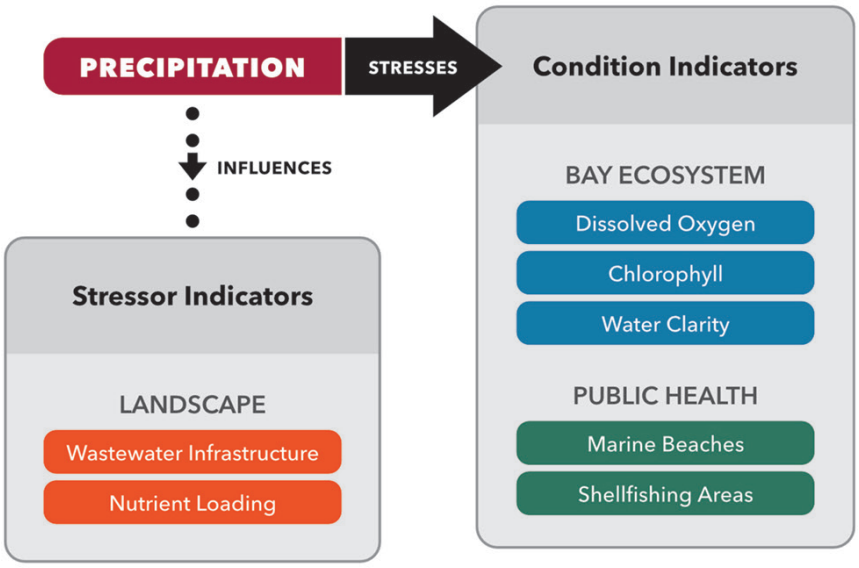
CHAPTER 2: PRECIPITATION

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Photo: Upper Wood River in Exeter, RI (Ayla Fox)

NARRAGANSETT BAY ESTUARY PROGRAM

Overview



BACKGROUND

- Changes in global and regional patterns of precipitation stress ecosystem condition indicators such as dissolved oxygen, chlorophyll, and water clarity, as well as public health indicators including marine beach and shellfishing area closures. Precipitation also influences other stressor indicators, such as wastewater infrastructure and nutrient loading.

KEY FINDINGS

- ***Trends:*** Regionally, average annual precipitation as rainfall has increased 0.4 to 0.7 inches per decade since 1895, and the amount of annual precipitation falling during intense storms has increased 71 percent since 1965.
- ***Projections:*** Local projections include increased volume of annual precipitation, greater frequency and intensity of precipitation events, and changing seasonality with increased winter precipitation (as rain) and limited summer precipitation, prompting drought or drought-like conditions.

Introduction

Changes in precipitation patterns affect many aspects of ecological and public health conditions in the Narragansett Bay region. Analyzing changes in precipitation is essential to understand recent changes in the Bay ecosystem and to anticipate future changes, enabling more effective climate change adaptation practices.

The National Climate Assessment reported increases in the average annual volume of precipitation in the Northeast, severity of tropical cyclones in the North Atlantic, and frequency of extreme precipitation events (Melillo et al. 2014). Several regional analyses show an increase of 0.1 inch (0.25 centimeter) per year, or one inch (2.54 centimeters) per decade, in annual precipitation from 1905 to 2005 (Pilson 2008, Smith et al. 2010, Heffner et al. 2012, UNH 2016). An increased intensity of precipitation events has become the norm, with a 71 percent increase in the annual amount of precipitation falling in very heavy events (defined as the heaviest one percent of all daily events) from 1958 to 2012 in Rhode Island and Massachusetts (Melillo et al. 2014, Bradley et al. 2016). Increased precipitation can potentially affect organic and inorganic pollution levels from urban flooding and wastewater treatment discharge. Although there is a trend of increasing annual precipitation, shifts in seasonality of precipitation lead to longer periods of drought and drought-like conditions in summer (MAOEEA 2011).

In this chapter, the Narragansett Bay Estuary Program analyzes the status, recent trends, and future projections of precipitation patterns in the Narragansett Bay Watershed. The chapter also discusses effects of changing precipitation patterns on physical conditions of the Bay, including water clarity, dissolved oxygen, and chlorophyll, as well as the impacts on human health, such as beach or shellfishing area closures.

Methods

The Estuary Program analyzed data from two NOAA stations: Providence, Rhode Island (1895 to 2015) and Worcester, Massachusetts (1948 to 2015). Precipitation data and Palmer Drought Severity Index (PDSI) data were obtained from NOAA's Climate at a Glance database (NOAA 2016). PDSI data are available only for states, not cities (e.g., Providence and Worcester), and therefore the Estuary Program used the data available for Rhode Island and Massachusetts from 1895 to 2015. This source was selected because of the long time period and completeness of the dataset.

The Palmer Drought Severity Index is a measurement of the relative dryness of an area on a standardized scale of −10 to +10 calculated from temperature and precipitation data. This parameter assumes the precipitation is readily available to be evaporated or transpired by plants, and it does not account for snow or ice, which experience delayed runoff (Dai et al. 2016).

Status, Trends and Projections

STATUS AND TRENDS

The status of rainfall in the Narragansett Bay Watershed is based on data from 2015. In 2015, both Providence and Worcester received about 40 inches of precipitation. The long-term annual averages for the entire datasets—1895 to 2015 for Providence and 1948 to 2015 for Worcester—were 45.9 inches (116.6 centimeters) and 46.9 inches (119.1 centimeters), respectively (Figures 1 and 2).

The data show that although precipitation patterns have been variable, the annual average precipitation has increased steadily. Providence's annual average precipitation increased 0.40 inch (1.02 centimeters) per decade since 1895 in a trend that was statistically significant (p=0.016) (Figure 1). Worcester's average precipitation increased 0.73 inch (1.85 centimeters) per decade since 1948 in a trend that was not statistically significant (Figure 2). The largest periods of drought were in the 1940s and 1960s (Figures 3 and 4). In the 1970s, precipitation in Providence and Worcester shifted from being generally below to generally above the long-term average (Figures 1 and 2). This pattern has continued despite some recent years of below-average precipitation.

Despite the recent years of droughts and drought-like conditions, the Palmer Drought Severity Index agrees with the precipitation data, showing that wetter years have become more common. The Palmer Drought Severity Index has increased 0.03 per decade in Rhode Island and 0.07 per decade in Massachusetts (Figures 3 and 4), although these increases are not statistically significant.

PROJECTIONS

The University of Massachusetts Climate System Research Center produced climate model projections for Rhode Island and Massachusetts, as well as other states in New England and the Midwest (Bradley et al. 2016). The precipitation projections were based on two greenhouse gas emission rates, similar to the IPCC Special Report on Emissions Scenarios model (IPCC 2014). The lower emission

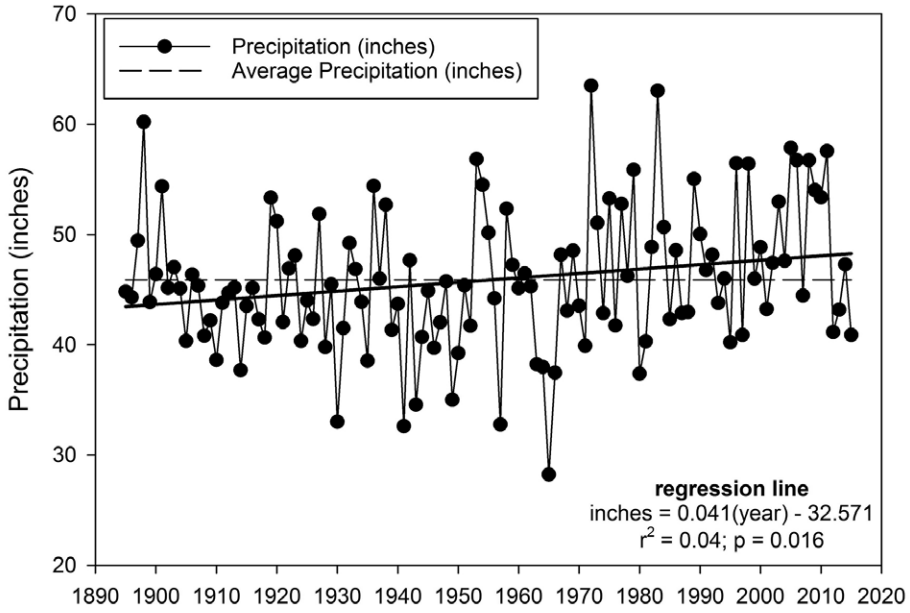


Figure 1. Annual precipitation at Providence, Rhode Island. Dotted line is the average annual precipitation over the entire dataset. Black line represents a statistically significant increase in precipitation. Source: National Oceanic and Atmospheric Administration (NOAA 2016)

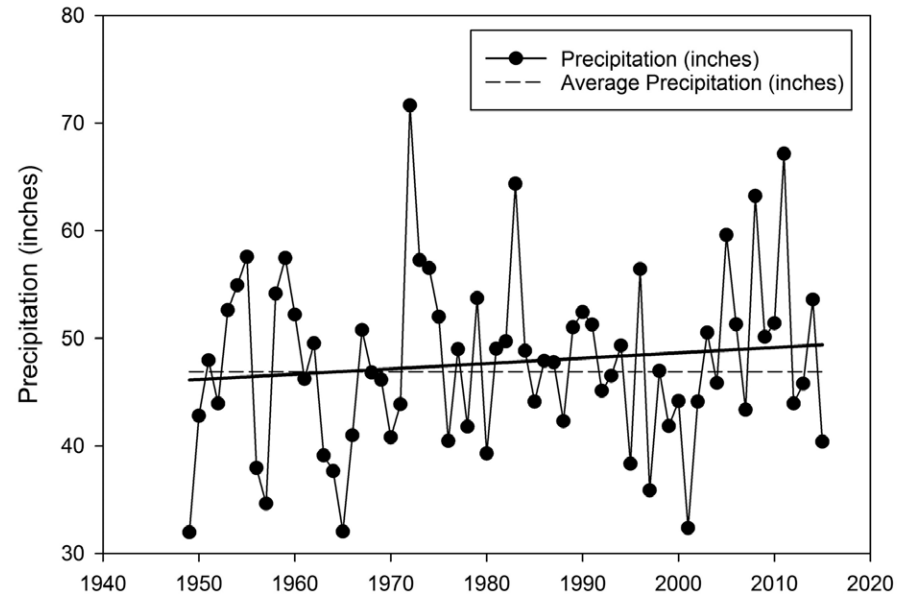


Figure 2. Annual precipitation at Worcester, Massachusetts. Dotted line is the average annual precipitation over the entire dataset. Black line is a guide to indicate an apparent increasing trend, but the trend is not statistically significant. Source: National Oceanic and Atmospheric Administration (NOAA 2016)

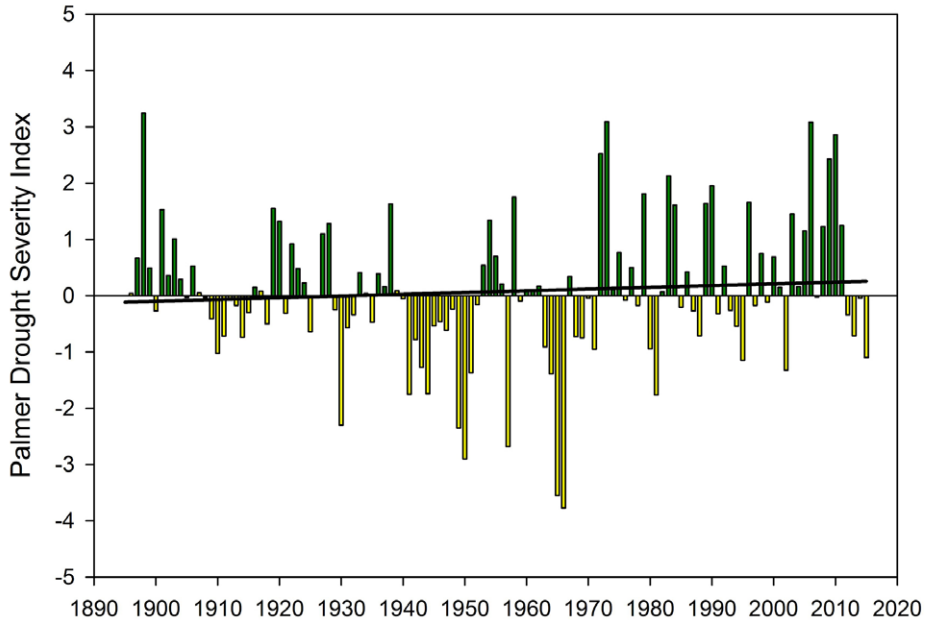


Figure 3. Annual Palmer Drought Severity Index (PDSI) for Rhode Island. Yellow bars (negative values) indicate drier years. Green bars (positive values) indicate wetter years. Black line is a guide to indicate an apparent increasing trend, but the trend is not statistically significant. Source: National Oceanic and Atmospheric Administration (NOAA 2016)

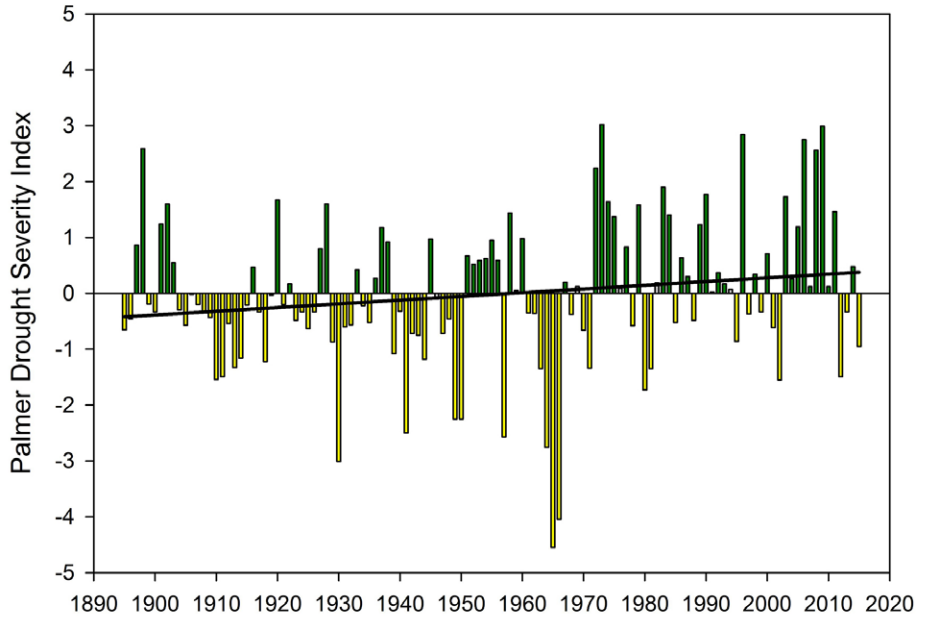


Figure 4. Annual Palmer Drought Severity Index for Massachusetts. Yellow bars (negative values) indicate drier years. Green bars (positive values) indicate wetter years. Black line is a guide to indicate an apparent increasing trend, but the trend is not statistically significant. Source: National Oceanic and Atmospheric Administration (NOAA 2016)

rate scenario will occur only if there are substantial efforts to mitigate greenhouse gas emission rates during future economic development, while the higher emission rate is comparable to current greenhouse gas emissions (Bradley et al. 2016). Both emission rates resulted in similar findings regarding precipitation. The projections are similar for Rhode Island and Massachusetts, so they are included in one graph. The projections show that by 2100, precipitation will range from eleven to fifteen inches (28 to 38 centimeters) per winter season (Figure 5), which is an increase of zero to three inches (zero to 7.6 centimeters) from the 1950 to 2005 data. The overall amount of winter precipitation is projected to increase, but warming temperatures will cause more rain and less snowfall (Figure 6). These changing precipitation patterns and rising temperatures will create a climate in New England in 2100 that is similar to present-day Virginia (Bradley et al. 2016).

Discussion

Current climate models project that total volume of precipitation will increase by zero to three inches (zero to 7.6 centimeters) per decade locally (Figure 5) and that seasonality of precipitation will continue to change, leading to cascading negative impacts on the overall health of Narragansett Bay and its Watershed (Bradley et al. 2016). Rhode Island and Massachusetts have received at least 30 inches (76.2 centimeters) of rain annually since 1980, and the annual amount of precipitation has increased between 0.5 and one inch (1.3 to 2.5 centimeters) per decade (Figures 1 and 2; Pilson 2008, Vallee and Giuliano 2014, UNH 2016). The local increase in precipitation is due to large, slow-moving storm systems, multiple events in a short period of time, and the increase in frequency and intensity of rain events (Vallee and Giuliano 2014). The projected increase in intensity and frequency of precipitation events in places with aging infrastructure, urbanization, and poor soil quality is expected to increase the likelihood of flooding (see "Impervious Cover" chapter). This will increase pressure on local wastewater treatment facilities, causing more overflow events and introducing excess nutrients and pathogens into the rivers and the downstream estuarine waters (Melillo et al. 2014, USEPA 2016; see "Wastewater Infrastructure" chapter). This increased runoff and overflow in the Narragansett Bay Watershed is expected to affect many aspects of the health of the Watershed.

Intense flooding and overflow events, such as the severe 2010 event that dramatically affected Rhode Island, and less intense events will continue to alter the physical, biological, and human health conditions within the Narragansett Bay Watershed. Water quality is degraded when a large volume of precipitation is introduced rapidly, creating turbidity within the water column. The increased inputs of nutrients, pollutants, pathogens, and sediment decrease the clarity of the water column (see "Water Clarity" chapter). Poor water clarity impacts important habitats, such as seagrasses (see "Seagrass" chapter), and is an overall indicator of poor water quality.

Increased precipitation also affects chlorophyll concentrations and primary production by phytoplankton. Primary production occurs predominantly at the surface of the water column where nutrients (i.e., from precipitation, runoff, point-source discharge, and remineralization), sunlight, and dissolved oxygen are readily available. As the increased volume of precipitation introduces additional organic and inorganic nutrients to the Bay, it is expected that primary production may increase in surface waters, prompting eutrophication (see "Chlorophyll" chapter). As the primary producers die and settle to the bottom, this flux of organic matter decomposes, potentially resulting in decreased dissolved oxygen in the water column and negatively affecting benthic and other estuarine communities.

Another influence on dissolved oxygen levels is water column stratification. Freshwater from precipitation and runoff rests on top of the denser, saltier water. If no mixing occurs (either wind, tidal, or storm-induced), the bottom waters will become isolated and depleted of oxygen as organic matter decomposition and respiration of organisms consume oxygen. Without mixing, or advection of new water to the region, dissolved oxygen cannot be replenished (see "Dissolved Oxygen" chapter). The increase in precipitation and intense precipitation events may lead to more stratification of the water column in Narragansett Bay, depleting dissolved oxygen in the bottom water; river runoff may be a stronger climate-related driver of increased stratification than increases in temperature (Codiga 2012).

The physical and biological impacts of increased precipitation often prompt local beach and shellfishing closures due to the decreased water quality. Increased precipitation often brings pathogens, indicated by fecal coliform or *E. coli* bacteria, to rivers and estuaries. Precipitation amounts and correlated bacteria counts have been used to temporarily close monitored marine beaches and areas approved for shellfishing (see "Marine Beaches" and "Shellfishing Areas" chapters). Both marine beaches and shellfishing areas are opened as soon as the threat to human health passes.

While the impacts from increased flooding and overflow events are the most obvious results of

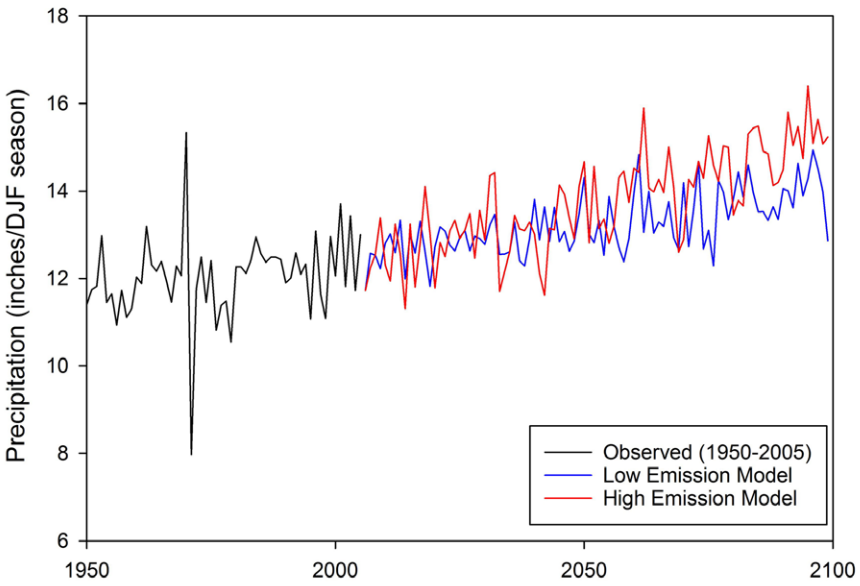


Figure 5: Climate model projection of the winter (December, January, and February) total precipitation in Rhode Island or Massachusetts to 2100. The black line represents precipitation data collected 1950 and 2005. The red line is the precipitation projection under a scenario of high greenhouse gas emission rates that are comparable to present-day emissions. The blue line is the precipitation projection under a scenario of lower emission rates that would be a result of reducing present-day emission rates through the introduction of energy-efficient technologies. Source: United States Geological Survey (USGS 2016)

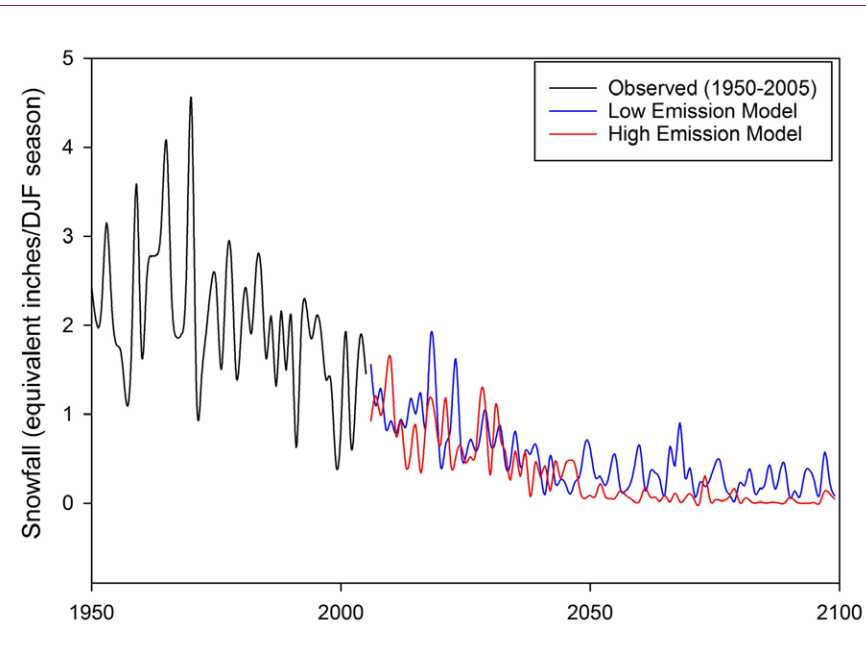


Figure 6: Climate model projection of the winter (December, January, and February) annual snowfall in Rhode Island or Massachusetts to 2100. The black line represents precipitation data collected 1950 and 2005. The red line is the precipitation projection under a scenario of high greenhouse gas emission rates that are comparable to present-day emissions. The blue line is the precipitation projection under a scenario of lower emission rates that would be a result of reducing present-day emission rates through the introduction of energy-efficient technologies. Source: United States Geological Survey (USGS 2016)

changing precipitation patterns witnessed in the region, this is only one aspect of the changing patterns. Although the total volume of precipitation is projected to further increase, when and how the precipitation falls is also changing. According to the Massachusetts Executive Office of Energy and Environmental Affairs (MAEOEEA), winter precipitation is expected to increase faster than precipitation during other seasons, often falling as rain rather than snow (Figures 5 and 6; MAOEEA 2011). This increase of rain events in the winter months will impact local snowpack, spring snowmelts, and peak streamflow (Trenberth 2011, MAOEEA 2011, RI EC4 STAB 2016, USEPA 2016). Historically, snowfall was immobilized on land, and warmer spring temperatures would reintroduce the now-melted snow to waterways, increasing river base flow. In comparison, when rain is delivered to a watershed, the precipitation is less delayed on its journey into the waterways. This near-immediate introduction increases base flow of streams in the winter, rather than in the spring thaw. Due to this change, low-flow periods are expected to shift earlier in the spring, extending the summer dry season. Therefore, the projected alteration of water delivery to the Narragansett Bay Watershed will alter the timing and intensity of local low-flow periods (MAOEEA 2011).

The projected drought and drought-like conditions are expected to increase the water temperatures in small rivers and streams, degrade reservoir and groundwater reserves, and magnify the impact of environmental pollutants (RI EC4 STAB 2016). During times of reduced water availability, withdrawal of water will be felt more intensely, especially if the fresh groundwater is not replaced through precipitation. With greatly reduced groundwater replacement, saltwater intrusion can occur (USEPA 2016). This can occur most commonly around estuaries like Narragansett Bay (USEPA 2016).

Periods of drought allow pollutants to build up on land. When precipitation does fall, the first flush (first pulse of water over land) contains a very high concentration of pollutants delivered to receiving waters. The high concentration of organic and inorganic pollutants associated with the first flush has the potential to magnify the negative impacts on turbidity, water clarity and quality, chlorophyll, dissolved oxygen, and human health within the Narragansett Bay Watershed (Wright et al. 1992). In order to prevent the impacts from an event such as the first flush, or any large flooding event, the Narragansett Bay Commission and other organizations are conducting local mitigation efforts by creating a combined sewer overflow abatement program in Providence and surrounding towns to prevent approximately one billion gallons of untreated runoff and wastewater from entering Narragansett Bay annually as a result of flooding and overflow (NBC 2017). The success of the first two phases of this project has allowed the Rhode Island Department of Health to investigate opening an urban beach on the Providence River estuary and adjusting the precipitation thresholds that trigger a shellfish area closure (RIDOH 2015; see "Shellfishing Areas" chapter). The city of Fall River is also implementing a combined sewer overflow abatement program.

Overall, the changing precipitation patterns as a result of global climate change will likely alter the physical and biological conditions of the Bay and indirectly impact human health. The changing trends will pose new challenges for the health of the Bay and biodiversity, but with a proactive and innovative perspective people can work to mitigate the impacts in order to preserve the integrity of the Narragansett Bay Watershed.

Data Gaps and Research Needs

- The existing network of stream gages in the Narragansett Bay Watershed should be assessed to ascertain key gaps, and data records should be analyzed to characterize variability in rainfall across the Watershed and identify where additional rain gages may be needed (e.g., the Pawtuxet River watershed). Sustained funding for the network is critical to ensure adequate hydrologic data is available to support management decision-making.
- Extreme precipitation and drought in the Narragansett Bay Watershed need to be further investigated using a combination of approaches, such as the Palmer Drought Severity Index, the Crop Moisture Index, and Cornell University's effort to analyze frequency and intensity of precipitation (Cornell University 2016). The results of these efforts will detail the impacts that extreme precipitation and drought have on water resources management and water quality.
- The frequency, amount, seasonality (freeze/thaw cycles), and type (rain, snow) of precipitation influence physical, chemical, and biological processes within the Narragansett Bay Watershed. The impacts of climate change on precipitation need to be further explored using downscaling of climate models or other methods. Results of these efforts will increase knowledge of how sensitive habitats will change, and how to plan for the resiliency of infrastructure.

Acknowledgments

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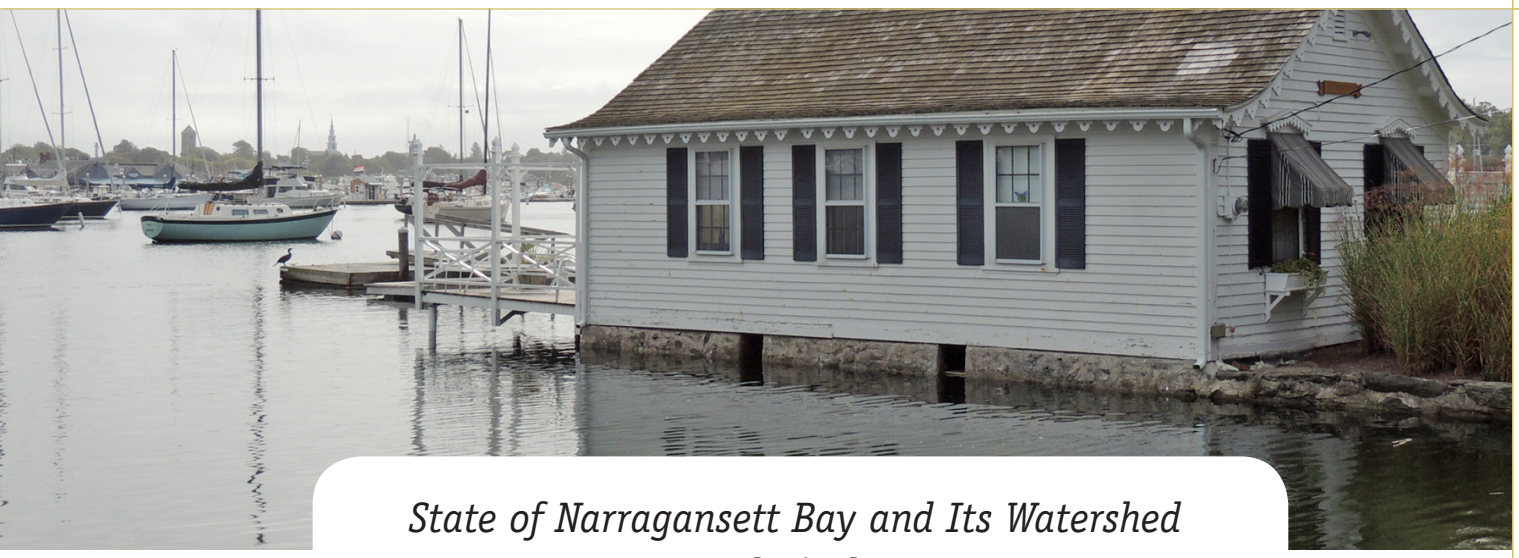
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State of Narragansett Bay and Its Watershed
2017 Technical Report

Climate Change Stressor Indicators

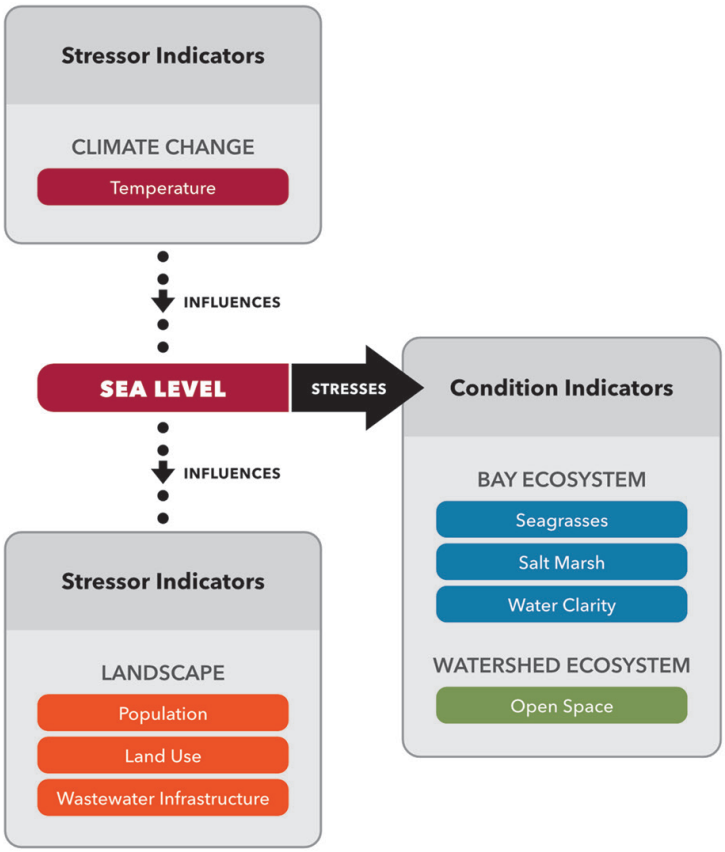
CHAPTER 3: SEA LEVEL

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Photo: High tide in the Barrington River, Barrington, RI (James Tobey)

NARRAGANSETT BAY ESTUARY PROGRAM

Overview



BACKGROUND

- In recent decades, sea level has been rising at an increasingly rapid pace in Narragansett Bay. The change in sea level affects ecosystem condition, including salt marshes and seagrasses, and it influences other stressors including population, land use, and wastewater infrastructure. Temperature influences the rate of sea level rise.

KEY FINDINGS

- ***Status and Trends:*** Sea level rose nine inches at the tide gauge in Newport from 1930 to 2015 and 6.6 inches at Providence from 1938 to 2015. Three nuisance flooding events occurred at Providence in 2016.

- ***Projections:*** NOAA projects that sea level could rise as much as eleven feet at Newport by 2100. Along Rhode Island's coastline in Narragansett Bay, approximately seventeen square miles of land and 3,765 buildings would be inundated under a seven-foot sea level rise scenario. Over 10,000 people live today within this area along Rhode Island's coastal areas of Narragansett Bay. Narragansett Bay salt marshes are converting to unvegetated tidal flats and open water, and the rate of this marsh loss is projected to increase with rising rates of sea level. The frequency of nuisance flooding in the Northeast region is projected to increase 25-fold by the year 2045 due to sea level rise.

Introduction

Sea level is the height of the sea with respect to a standardized reference elevation, or datum. Sea levels are rising at an accelerating rate along most of the world's coastlines (IPCC 2013). At the global scale, sea level is rising because of two main processes: (1) seawater is becoming warmer, causing the seawater to expand in volume, and (2) glaciers and the polar ice sheets are melting, causing more water to enter the ocean. The driving force behind those processes is the emissions of carbon dioxide and other greenhouse gases into the atmosphere, where they trap heat and cause the air and ocean to become warmer (see introduction to "Climate Change Stressor Indicators" section in this report). The global change in sea level is termed "eustatic or absolute sea level rise." As there is a time lag in sea level rise in response to climate change, sea levels are expected to continue rising long into the future regardless of steps taken to curb global greenhouse gas emissions (IPCC 2007). Other factors, such as changes in ocean circulation patterns, also have important influences on sea level.

In addition to eustatic sea level rise, other processes can augment or reduce the amount of sea level rise that occurs in a particular place. For example, vertical movement of landmasses—whether they are subsiding or uplifting due to geological processes—can affect the amount of sea level rise observed on a local or regional scale. The combination of eustatic sea level rise with subsidence- or uplift-related changes in sea level is termed "relative sea level rise" (Rovere et al. 2016). The New England coast is subsiding (sinking) slowly, and this effectively increases the local rate of sea level rise. At the Newport, Rhode Island, tide gauge, the land along the coastline moved downward at a rate of 0.35 inch (8.8 millimeters) per decade between 1930 and 2006 (NOAA 2013).

Changes in ocean circulation can affect sea level on a regional scale and are a source of uncertainty in regional projections of sea level rise. The Intergovernmental Panel on Climate Change (IPCC) predicts that ocean circulation in the North Atlantic Ocean is likely to slow down by 2100 because of shifts in salinity and temperature of the Arctic Ocean and North Atlantic Ocean (IPCC 2007 and Yin et al. 2009). As a result, sea level is rising faster along the coast from Cape Hatteras, North Carolina, to the Canadian Maritime Provinces compared to other parts of the world (Sallenger et al. 2012).

The Rhode Island Coastal Resources Management Council (CRMC) formally adopted a set of sea level rise projections published in 2012 by the National Oceanic and Atmospheric Administration (NOAA) for Newport (Parris et al. 2012). Based on those projections (Sea Level Rise Curves), relative sea level at Newport was projected to increase by a maximum of approximately one foot (0.3 meter) by 2035, two feet (0.61 meter) by 2050, and 6.6 feet (two meters) by 2100. However, as discussed later in this chapter (see Projected Sea Level Rise), NOAA recently increased its projections quite dramatically (Sweet et al. 2017), based in large part on projected increases in melting of the Greenland and Antarctic ice sheets.

There is increasing evidence in the scientific literature indicating that the rate of ice sheet disintegration and melting may be much greater than previously anticipated. Beyond an unknown threshold of greenhouse gas emissions, the collapse of the polar ice sheets, especially in Greenland and Antarctica, will be inevitable and irreversible (Overpeck et al. 2006, DeConto and Pollard 2016, Hansen et al. 2016).

In this chapter, the Estuary Program presents the most recent research and findings from numerous sources that have calculated sea level trends and future projections at global, regional, and local scales, and are relevant to Narragansett Bay. The Estuary Program focused on providing a comprehensive summary of the literature currently available, including studies that have estimated impacts on habitat, people, and infrastructure. In addition, the Estuary Program calculated the number of buildings and total population along Rhode Island's coastline in Narragansett Bay that may be affected under different sea level rise scenarios. The many factors that cause sea level rise are dynamic and have their own drivers and stressors, and new research and updated projections regarding sea level rise, and emerging research and updated projections regarding sea level rise and nuisance flooding are expected to continue.

Methods

STATUS AND TRENDS OF SEA LEVEL RISE

NOAA maintains a network of tide gauge stations for monitoring water levels and rates of sea level rise. Most stations provide readings every six minutes, making it possible to monitor real-time tide elevations and inundation threats. Sea level is measured in relation to a tidal datum, or height, which is a standard elevation defined for each station (Table 1). NOAA uses monthly mean sea level data from the tide gauge measurements to characterize linear trends, average seasonal cycles, and interannual variations.

Long-term datasets from NOAA stations can be used to analyze relative mean sea level rise trends and to develop projections for future sea level rise. From NOAA Tides and Currents, the Estuary Program gathered information about water levels and relative mean sea level trends for two tide gauge stations in Narragansett Bay, both in Rhode Island: Newport (station #8452660) and Providence (station #8454000). At Newport and Providence, NOAA has monitored mean sea levels since 1930 and 1938 respectively. While there is a tide gauge station in Fall River, Massachusetts, NOAA does not provide sea level trends for that station because the record of data collection is too short (tide gauge station was established in 1955). Table 1 summarizes the types of information available for each tide gauge station and provides definitions used throughout this chapter.

PROJECTIONS OF SEA LEVEL RISE

In this chapter, the Estuary Program summarized the most current sea level rise projections relevant to Narragansett Bay with the understanding that the science will continue to develop and projections will be modified over time. The primary source for projections affecting Narragansett Bay was the recent NOAA report on global mean sea level (GMSL) rise and regional relative mean sea level rise (Sweet et al. 2017). The data for projections were obtained directly from the database that accompanies the NOAA technical report (Sweet et al. 2017).

Visualization of Sea Level Rise Scenarios

Government agencies have developed sea level rise geospatial models and user-friendly online tools that are helpful for visualizing and evaluating the potential impacts of sea level rise (e.g., https://www.climate.gov/maps-data/dataset/sea-level-rise-map-viewer). Because of data limitations at the Bay scale, including incomparability of methods and varying spatial and time scales among models, the Estuary Program did not attempt to reconcile data from different national and state sea level rise data sources, rather, the Estuary Program built upon information from two well-established sources, STORMTOOLS and the Sea Level Affecting Marshes Model (SLAMM), both developed and completed for the statewide coastal areas of Rhode Island.

Sea Level Rise Impacts on the Landscape

The University of Rhode Island developed STORMTOOLS for the CRMC to aid state and local adaptation planning efforts statewide. The STORMTOOLS maps have very high resolution for the Rhode Island sections of Narragansett Bay. This mapping tool allows the user to identify coastal areas of Rhode

Table 1. Definitions of selected terms used in this chapter.

Term	Definition
Tidal Datum	A tidal datum is a standard elevation defined by a certain phase of the tide. Tidal datums are used as references to measure local water levels and should not be extended into areas having differing oceanographic characteristics without substantiating measurements.
National Tidal Datum Epoch (NTDE)	An epoch is a nineteen-year tidal cycle used to calculate datums. The present National Tidal Datum Epoch (NTDE) is 1983 through 2001.
Station Datum (STND)	A fixed base elevation at a tide station to which all water level measurements refer. It is unique to each station.
Mean Sea Level (MSL)	The arithmetic mean of hourly heights observed over the National Tidal Datum Epoch. It is the reference datum for calculating sea level trends.
Mean Higher High Water (MHHW)	The average of the higher high water height of each tidal day observed over the National Tidal Datum Epoch. It is the reference datum for determining frequency and duration of nuisance flooding. • Newport tide gauge: MHHW is 5.74 feet (1.75 meters). • Providence tide gauge: MHHW is 8.33 feet (2.5 meters).
Mean Lower Low Water (MLLW)	The average of all the lower low water heights observed over the National Tidal Datum Epoch. • Newport tide gauge: MLLW is 1.89 feet (0.58 meters). • Providence tide gauge: MLLW is 3.49 feet (1.06 meters).

Source: NOAA Tidal Datums Website (2013); NOAA Datums for Newport, RI, and Providence, RI, (2017).

Island that would be affected by different elevations of sea level rise (one to seven feet; 0.3 to 2.1 meters) with or without storm surge (predictions for 25-, 50-, and 100-year return interval storms) at a 95-percent confidence level.

While data on sea level rise scenarios derived from STORMTOOLS have been used widely in Rhode Island to analyze impacts on people and infrastructure at the state and local level, by the same group that developed STORMTOOLS and other state and local partners, the Estuary Program focused only on the coastline around Narragansett Bay. Readily available data from these other efforts were in tabular form, typically covered the entire state or towns rather than only the Narragansett Bay Watershed, and were not geospatial. Thus, the Estuary Program performed a geospatial analysis that covered only the portions of Rhode Island that lie in the Narragansett Bay Watershed, excluding areas of the state and portions of towns that fall outside of the Watershed. For this reason, the results of this cursory analysis represent only areas in Rhode Island and not the portions of the Narragansett Bay Watershed in Massachusetts, such as Mount Hope Bay.

The Estuary Program used the Derived Inundations Surfaces data available in RIGIS, which are the data outputs of sea level rise scenarios from STORMTOOLS, to calculate vulnerable coastal areas in the Rhode Island portion of Narragansett Bay under sea level rise scenarios of one, three, and seven feet. The following analyses were performed by the Estuary Program to calculate the number of buildings and the total number of residents (based on Census 2010 data) that would be exposed to inundation under each sea level rise scenario.

- **Number of Buildings (Rhode Island Portion of Watershed Only):** The Estuary Program performed spatial analyses of Rhode Island e911 Exposure Assessment data similar to those conducted by the developers of STORMTOOLS. The e911 geo-spatial dataset contains locations for all buildings in the state with known street addresses; this dataset represents buildings that have been identified from 2006 through 2017.
- **Total Population Residing in the Watershed (Rhode Island Portion of Watershed Only):** The estimated number of residents affected by the sea level rise scenarios was calculated using population data from dasymetric analysis (see "Population" chapter), which identifies population distribution on a finer spatial scale than towns or census blocks. This approach enabled the Estuary Program to identify more accurately where people live within the Watershed boundaries, as opposed to state, municipal, or census block boundaries. (Information about impacts of sea level rise scenarios at the state and municipal level are available from the State of Rhode Island, Department of Administration, Division of Planning.)

Sea Level Rise Impacts on Estuarine Habitats

The CRMC produced a report in 2015 that assessed the vulnerability of Rhode Island's salt marshes to sea level rise based on results from the Sea Level Affecting Marshes Model (SLAMM) (CRMC 2015). The report included a series of maps depicting areas of salt marsh, parcel by parcel, that were projected to persist or be lost under scenarios of one, three, or five feet (0.3, 0.9, or 1.5 meters) of sea level rise. These three scenarios for sea level rise were chosen because CRMC had adopted in 2008 projections of three to five feet of sea level rise by 2100 based on the latest peer-reviewed science and research (CRCM 2015).

Similarly, the Massachusetts Office of Coastal Zone Management (MA CZM) recently completed modeling using SLAMM for salt marsh habitat along the Massachusetts coastline of Narragansett Bay, including Mount Hope Bay (Marc Carullo, presentation at the Narragansett Bay Estuary Program's Steering Committee Meeting, June 2017). Massachusetts used scenarios of 0.8, 2.3, 4.5, and 7.1 feet (0.25, 0.7, 1.37, and 2.16 meters) of sea level rise, which were based on National Climate Assessment (Parris et al. 2012) projected scenarios for sea level rise by 2100; 4.5 feet was the intermediate-high projection.

Comparing the SLAMM modeling conducted by both states, the main similarities and differences include the following: 1) Both states used National Wetland Inventory (NWI) data to classify wetlands in the model (see "Salt Marsh" chapter). 2) Both mapped and calculated areas of potential losses and gains of salt marsh, such as through conversion from upland or other vegetated wetland to salt marsh in response to sea level rise. 3) The Massachusetts SLAMM results also include projected changes in wetland classes, such as changes from infrequently flooded marsh to regularly flooded marsh. 4) The Rhode Island SLAMM report includes results from two modes for projecting marsh migration: (a) including all undeveloped and developed (e.g., roads) lands as potential marsh migration corridors or (b) including only undeveloped lands; modeling in Massachusetts included only the latter (CRMC 2015 and Marc Carullo, proceedings for Estuary Program Steering Committee Meeting, June 2017).

Data, results, and the final report of the Massachusetts SLAMM modeling effort were not available at the time this chapter was completed.

STATUS OF NUISANCE FLOODING

Nuisance flooding is defined as a water level that exceeds the National Weather Service's local threshold for minor flooding impacts established for emergency preparedness (Sweet et al. 2014, 2015, 2016); it is also defined as flooding that causes public inconveniences.

Nuisance flooding is measured based on NOAA tide gauges (tidesandcurrents.noaa.gov). At the Providence tide gauge station, an event of nuisance flooding is defined as occurring when the tide elevation is 2.16 feet (0.66 meter) above Mean Higher High Water (MHHW). This is the level at which buildings and infrastructure will be flooded. The Providence tide gauge is the only station in Narragansett Bay with an established threshold for nuisance flooding. Nuisance flooding may occur at different water levels in other areas of the Narragansett Bay Watershed.

The Estuary Program used NOAA's Inundation Analysis Tool (IA) (NOAA 2013) to determine how many times the threshold for nuisance flooding was exceeded at the Providence tide gauge in 2016. The IA tool allows the user to set a specific water level, such as the minor or moderate flooding threshold, and a specific date range. For each inundation event, the IA tool plots the duration of inundation versus elevation above MHHW.

Status, Trends and Projections

TRENDS OF SEA LEVEL RISE

The long-term historical rate of sea level rise at the Newport tide gauge, averaged over the 1930 to 2015 period, was approximately 1.1 inches (2.7 centimeters) per decade. Sea level rose a total of more than nine inches (0.23 meter) over the 85-year period (Figure 1). At that rate, sea level would rise 10.7 inches (0.27 meter) over 100 years.

At the Providence tide gauge, based on monthly mean sea level data from 1938 to 2015, sea level increased by approximately 0.9 inch (2.2 centimeters) per decade for a total of 6.6 inches (16.9 centimeters) over the 76-year period. At that rate, sea level would rise 8.9 inches (22.6 centimeters) over 100 years (Figure 2).

The rates at which sea level would rise based on those trends do not factor in accelerated melting of polar ice sheets, among other factors that would make sea levels rise at a faster rate. Table 2 summarizes sea level trends and rates reported by different sources at varying geographical and temporal scales based on a literature review by the Estuary Program as of January 2017. Recent rates for both relative sea level rise (for Newport and Providence) and global sea level rise appear to have accelerated over the past several decades, when compared to the longer-term rates. However, caution is advisable when citing short-term datasets (less than 30 years) because of inherently large regression errors and anomalous oceanographic events such as the slowdown in the

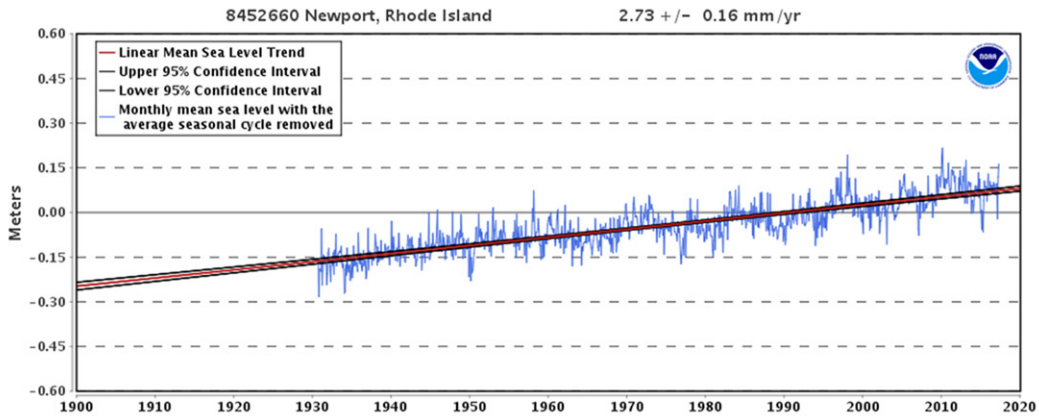


Figure 1. Historical sea level rise trend at Newport, Rhode Island, tide gauge station from 1930 to 2015. Source: NOAA Tides and Currents – Mean Sea Level Trend for Newport, RI #8452660

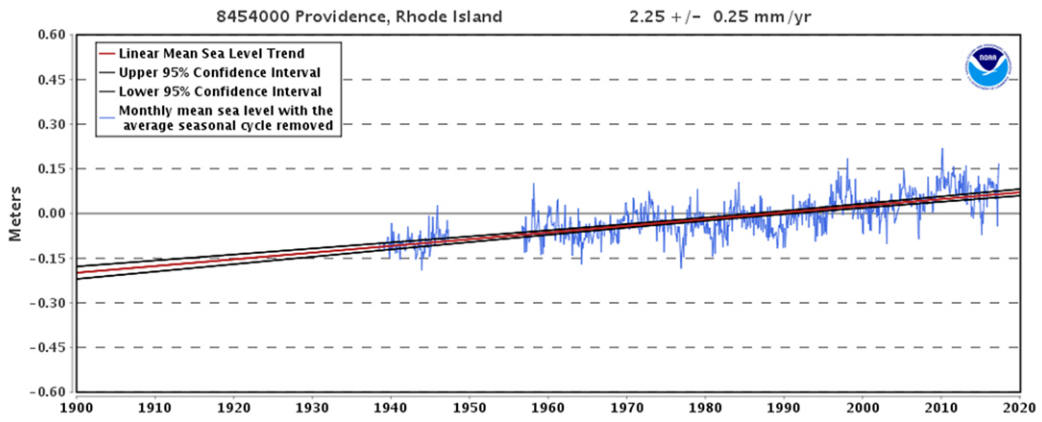


Figure 2. Historical sea level rise trend at Providence, Rhode Island, tide gauge station from 1938 to 2015. Source: NOAA Tides and Currents – Mean Sea Level Trend for Providence, RI # 8454000

Table 2. Sea level trends by geographical scale and source. Note that some of the data presented are relative sea level rise, and others are global or eustatic sea level rise.

Spatial Scale	Source	Time Scale for Calculating Sea Level Trends By Sources													Rate of Trends	
		1900	1910	1920	1930	1940	1950	1960	1970	1980	1990	2000	2010	2020	mm/ yr	inch/ decade
Newport	***NOAA (2015)***				***1930-2015***										***2.72***	***1.1***
Providence	***NOAA (2015)***					***1938-2015***									***2.22***	***0.9***
Newport	PSMSL[1]										1985-2014				4.2	1.7
Newport	Carey et al[1]									1984-2011					4.1	1.6
Global	IPCC[2]								1901-2010						1.7	0.7
Global	IPCC[2]										1993-2010				3.2	1.3
Global	Satellite altimetry[3]										1993-2016				3.4	1.3
Global	Hay et al[4]								1901-1990						1.2	0.5
Regional	Engelhart et al[4]								1901-1990						1.8	0.7

Sources:

1. Permanent Service for Mean Sea Level (PSMSL) collects and analyses relative sea level data from a global network of tide gauges. For Newport, the mean annual rate of sea level rise trend from the last thirty years (1985–2014) was approximately 1.7 inches (4.3 centimeters) per decade. The PSMSL analysis used the NOAA (2015) data from Newport but included only a portion of the complete record. Carey et al. (2017a) also calculated relative sea level rise using a portion of the NOAA (2015) data for Newport (1984–2011) with similar results.
2. The Intergovernmental Panel on Climate Change (IPCC) estimated a global or eustatic mean rate of sea level rise of 0.7 inches (1.8 centimeters) per decade between 1901 and 2010 for a total sea level rise of 7.3 inches (0.19 meter) (IPCC 2013). Between 1993 and 2010, the rate was higher at 1.3 inches per decade (IPCC 2013).
3. Using satellite altimetry, the Global Mean Sea Level (GMSL) trend (1993–2016) generated from the Integrated Multi-Mission Ocean Altimeter Data for Climate Research (GMSL dataset) was approximately 1.3 inches (3.3 centimeters) per decade (Nerem et al. 2010).
4. An analysis of global sea level rise that corrected for spatial bias in the tide gauge records calculated a rate of 0.5 inches (1.3 centimeters) per decade from 1901 to 1990 (Hay et al. 2015). Along the U.S. East Coast, the rate was 0.7 inches (1.8 centimeters) per decade during the same period (Engelhart et al. 2009).

Atlantic Meridional Overturning Circulation in 2009 and 2010, which caused a temporary surge in sea level rise (Goddard et al. 2015). Globally, proxy and instrumental sea level data indicate a transition in the late nineteenth century and the early twentieth century from relatively low mean rates of sea level rise over the previous two millennia to higher rates of rise (IPCC 2013). It is likely that the rate of global mean sea level rise has continued to increase since the early twentieth century, and other studies are consistent with the NOAA sea level trends.

PROJECTED SEA LEVEL RISE

Two decades ago, the IPCC estimated that global mean sea level would rise between 0.6 foot (0.18 meter) and 1.9 feet (0.58 meter) by 2100. A decade later, estimates were higher, ranging from 1.6 feet (0.49 meter) to 6.6 feet (two meters) by 2100 (Rahmstorf et al. 2007, Horton et al. 2008, Pfeffer et al. 2008, Allison et al. 2009, Richardson et al. 2009). These projections are based on various scenarios of greenhouse gas emissions and the effects on these emissions on global temperature.

In January 2017, NOAA released revised projections for global sea level rise scenarios. The extreme scenario for global mean sea level (GMSL) rise from 2000 to 2100 is 8.2 feet (2.5 meters) (Sweet and colleagues 2017), which is 1.6 feet (0.5 meters) higher than the upper-bound scenario published by NOAA in 2012 (Parris et al. 2012). The revised projections incorporated the growing evidence of accelerated ice loss from Antarctica and Greenland, and other factors. Six GMSL rise scenarios are shown in Table 3.

Along the U.S. Atlantic coast from Virginia northward, including Narragansett Bay, relative sea level rise is projected to be faster than GMSL rise. Sweet and colleagues (2017) estimated that under the Intermediate-High, High, and Extreme scenarios, the Northeast region would experience relative sea level rise that exceeds GMSL rise by one to three feet (0.3 to 1.0 meter) by 2100.

For the Newport tide gauge station, NOAA has provided projections of relative sea level rise to 2100 (Sweet et al. 2017), including three sub-scenarios (Low, Medium, High) for each scenario. Under the Extreme scenario of GMSL rise (8.2 feet; 2.5 meters), the relative sea level rise at Newport under the Medium sub-scenario (50th percentile) is 0.92 foot (0.28 meter) from 2000 to 2020 and 11.15 feet (3.40 meters) from 2000 to 2100 (Table 4). Figure 3 shows the relative sea level change at the Newport tide gauge, in each of the six GMSL rise scenarios (Low to Extreme) as well as the relative sea level rise caused by vertical land movement (VML) only.

Table 3. NOAA's revised projections for global mean sea level (GMSL) rise from 2000 to 2100.

GMSL Scenarios	GMSL Rise (2000–2100)	
Low	0.3 meters	1.0 feet
Intermediate-Low	0.5	1.6
Intermediate	1.0	3.3
Intermediate-High	1.5	4.9
High	2.0	6.6
Extreme	2.5	8.2

Source: Sweet et al. (2017)

Table 4. NOAA's sub-scenarios of projected relative sea level rise (in feet) at Newport, Rhode Island, under the Extreme scenario of 8.2 feet (2.5 meters) of global mean sea level rise by 2100.

Sub-scenarios(1)		2000-2020	2000-2050	2000-2100
Low	17th percentile	0.45	2.2	8.6
Medium	50th percentile	0.92	3.41	11.15
High(2)	83rd percentile	1.02	3.71	11.68

(1) For each GSML rise scenario, there is a low, medium, and high sub-scenario, corresponding to the 17th, 50th, and 83rd percentile of the climate-related sea level projections consistent with the GMSL scenario.
(2) Shown in Figure 4 for the Sweet et al. (2017) extreme scenario.
Source: Sweet et al. (2017) and the database that accompanies that NOAA technical report

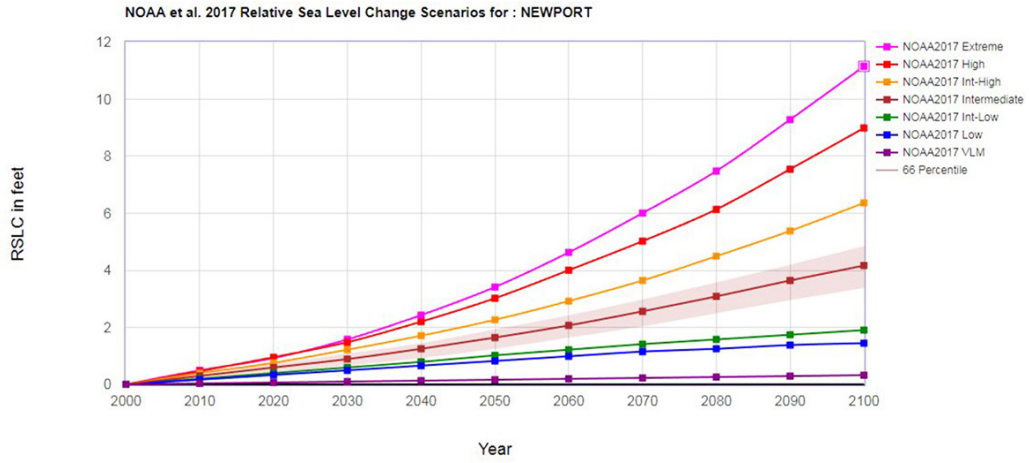


Figure 3. NOAA projections for relative sea level rise through 2100 at the Newport, Rhode Island, tide gauge station for the Intermediate GMSL scenario (1 meter) under six different climate change scenarios. 66 Percentile Confidence Range for the Intermediate Scenario is shown. Vertical Land Movement (VML): 0.00322 feet/year, as shown in the lowest curve. All values expressed in feet. Lines shown are the result of interpolation between values plotted. (Revised July 18th, 2017, http://www.corpsclimate.us/ccaceslcurves.cfm) Data Sources: Sweet et al. (2017). Charts: U.S. Army Corps of Engineers Sea Level Change Curve Calculator.

Table 5. Estimates of the land area, number of buildings, and population along Rhode Island's coastline in Narragansett Bay that will be affected by sea level rise scenarios of one, three, and seven feet above MHHW.

Sea Level Rise Scenarios	SLR 1 ft	SLR 3 ft	SLR 7 ft
Area in Square Miles (Acres)[1]	3.2 (2,025)	7.2 (4,623)	17.2 (10,956)
Number of Buildings[2]	14	283	3,765
Population[3]	267	1,667	10,274

[1] Also includes a few areas in Massachusetts around the Palmer River, inherent from STORMTOOLS
[2] Location of buildings from e911 database as of February 2017
[3] Based on 2010 population from dasymetric analysis (see "Population" chapter)
Data: RIGIS. 2015 NACCS-Derived Inundation Surfaces for Rhode Island Incorporating the Effects of Both Storm Surge and Tide and Rhode Island E 9-1-1 Uniform Telephone System.

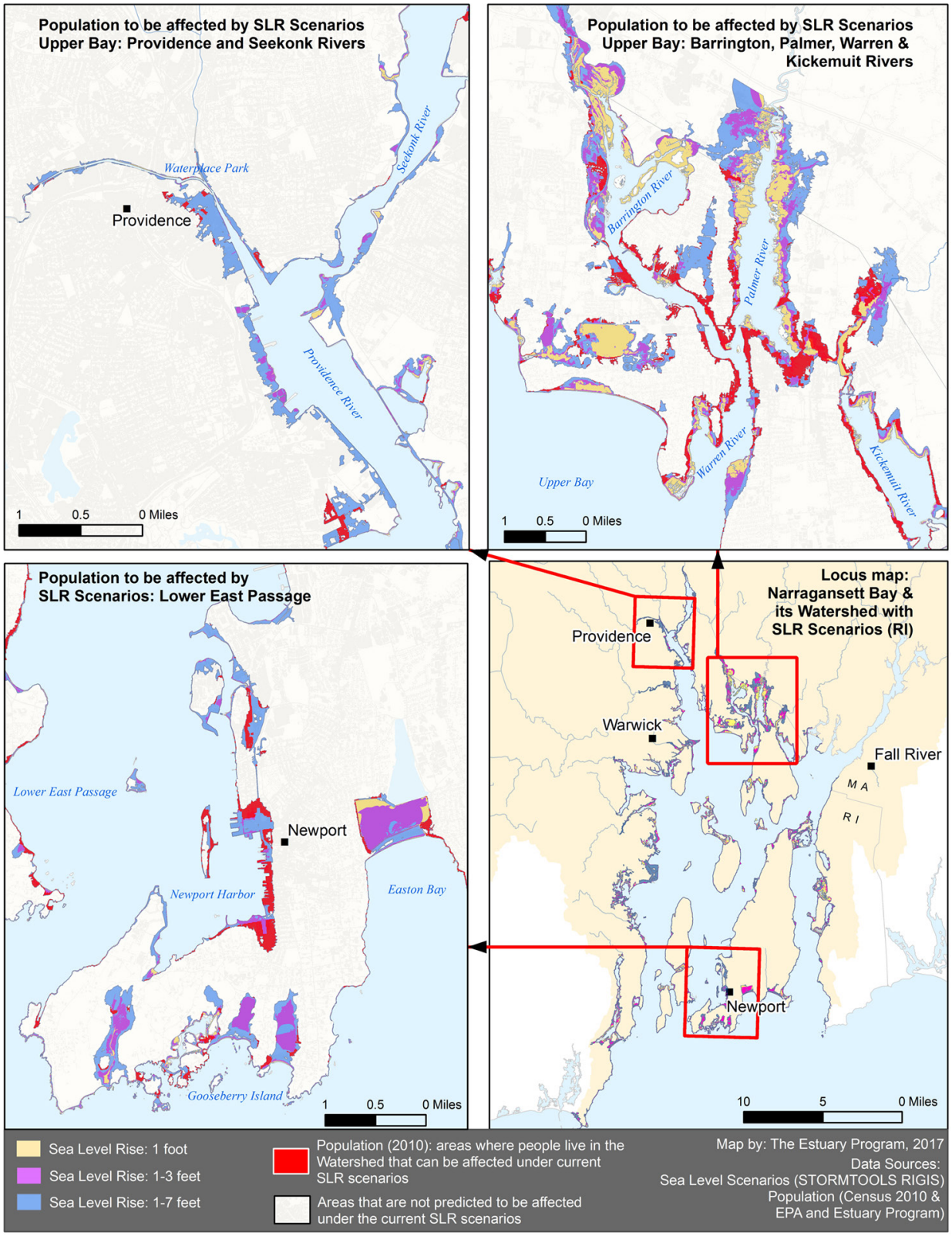


Figure 4. Map of select focus areas in the Rhode Island portion of the Narragansett Bay Watershed showing areas predicted to be inundated by one, three, or seven feet of sea level rise based on STORMTOOLS (developed for Rhode Island only). Red indicates areas where people live and that are projected to be inundated with seven feet of sea level rise. A total of 10,274 people (see Table 5) live in those areas, based on dasymetric analysis (see "Population" chapter).

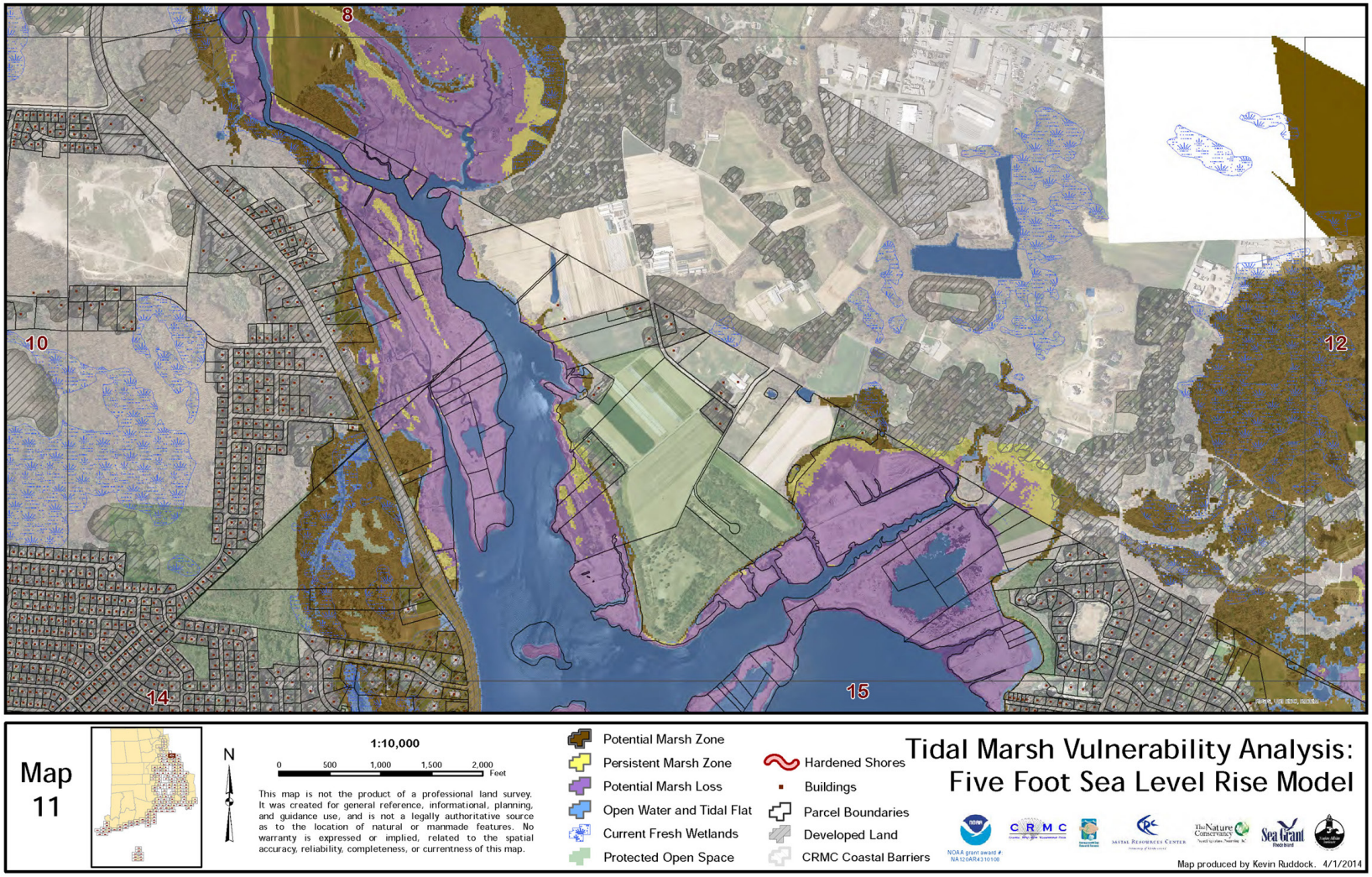


Figure 5. SLAMM model results for Barrington, Rhode Island, showing potential areas of salt marsh losses and salt marsh migration with five feet of sea level rise. Source: CRMC (2015)

Visualization of Sea Level Rise Scenarios

Sea Level Rise Impacts on the Landscape

The estimated land area and number of buildings in the Rhode Island part of Narragansett Bay that would be flooded under three different sea level rise scenarios, not considering storm surge, are shown in Table 5. The number of buildings includes current structures of all types, such as residential, educational, and commercial. A map of areas and population distribution affected by sea level rise scenarios is presented in Figure 4. Across the coastal areas of Rhode Island along Narragansett Bay, the estimated impacts of sea level rise in the three scenarios chosen for this analysis should be considered underestimates if development and population continue to increase within the land areas projected to be exposed to permanent flooding.

Sea Level Rise Impacts on Estuarine Habitats

Results from SLAMM for Rhode Island suggest that under scenarios of one, three, or five feet of sea level rise, the state could lose 13, 52, or 87 percent of its existing salt marshes, respectively. SLAMM results also indicate locations where marshes may, hypothetically, migrate landward in response to sea level rise. However, given that rates of sea level rise are projected to accelerate, perhaps quite dramatically, it is likely that the process of landward migration by marshes could be impeded by rapid sea level rise, similar to the marsh platform being unable to keep pace with sea level and becoming submerged. In addition, development such as seawalls, bulkheads, and roads in urban and suburban areas that are common in the Narragansett Bay Watershed, along with relatively steep upland slopes in some areas, will impede landward marsh migration. Field research and modeling are needed to better understand the landward marsh migration process under regimes of accelerated sea level rise, while also pursuing efforts to preserve upland areas that could serve as suitable migration corridors assuming that salt marshes will maintain some capacity to move landward. The SLAMM model is a useful tool for identifying these corridors where upland slopes are slight and free of obstructions. Figure 5 is an example of SLAMM results for Barrington, Rhode Island, including potential areas of salt marsh losses and salt marsh migration in a scenario of five feet of sea level rise.

As discussed above in the Methods section, SLAMM results for Massachusetts were not available at the time this chapter was completed, but a final report and data summaries were in production.

STATUS OF NUISANCE FLOODING

At the Providence tide gauge station, NOAA's Inundation Analysis Tool (IA) showed a total of three nuisance tidal inundation events in 2016 (Figure 6). The observed nuisance flood events ranged from 2.3 to 2.6 feet (0.7 to 0.8 meter) above MHHW. Tidal inundation events such as those observed in 2016 are expected to become increasingly common, consistent with the view that today's flood tide will become tomorrow's high tide.

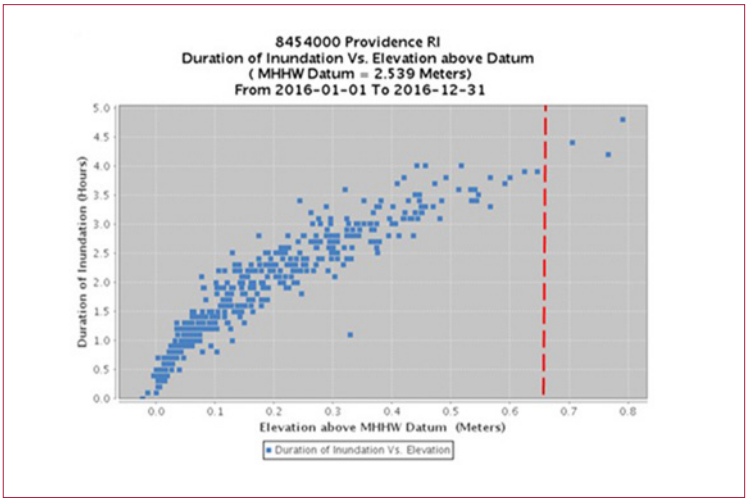


Figure 6. Elevation and duration of inundation events in 2016 at Providence tide gauge station. Red dashed line indicates the threshold for nuisance flooding as defined for this station by Sweet et al. (2016). Source: NOAA (2016)

Discussion

Tide gauge data show that sea level in Narragansett Bay has risen approximately an inch per decade since 1930, and the pace appears to have accelerated in recent decades (Table 2; PSML 2014, Carey et al. 2017a). Over the last 85 years, sea level at Providence has increased by nine inches, and further increases of up to eleven feet are projected for Narragansett Bay by the end of the century under an extreme scenario of accelerated sea level rise (Figure 3; Sweet et al. 2017). That projection is approximately five feet higher than what was projected in 2012 (Figure 7; Parris et al. 2012).

It is documented that sea level rise is accelerating globally and that from Cape Hatteras, North Carolina, to the Canadian Maritime Provinces the rate of relative sea level rise is greater than the global average rate of sea level rise. NOAA projects that by 2100 sea level rise in the northeastern United States could be eleven to twenty inches greater than the 8.2-foot rise projected for global mean sea level (Sweet et al. 2017).

Approximately three, seven, or seventeen square miles of land in Rhode Island around Narragansett Bay would be flooded permanently with sea level rise of one, three, or seven feet, respectively (Table 5). With the potential for nearly 11,000 acres of permanently flooded land in low-lying areas of the Rhode Island coast, not even considering scenarios with storm surge, there are very significant socio-economic and ecological implications given the extensive urbanization and development along the coastline. The Rhode Island Statewide Planning Program has begun to integrate sea level rise and climate change into its comprehensive planning process by using STORMTOOLS to calculate socioeconomic indicators for different sea level rise scenarios (Rhode Island Statewide Planning 2015).

If population continues to grow as expected in these coastal areas, leading to an increase in urban development (see "Population" and "Land Use" chapters), the impacts of sea level rise and nuisance flooding will be exacerbated. Based on geospatial analysis and calculations by the Estuary Program for areas of Rhode Island in the Narragansett Bay Watershed, approximately 283 buildings would be permanently flooded when the sea rises three feet, and 3,765 buildings would be flooded with seven feet of sea level rise (Table 5). Similarly, between 267 and 10,274 people who currently reside in these coastal areas of Narragansett Bay will be affected by sea levels rising one to seven feet, respectively (Figure 4). Those magnitudes of sea level rise are projected to occur by approximately 2050 and 2080, respectively, under the extreme sea level rise scenario (Sweet et al. 2017). Because these calculations do not include the Massachusetts portion of Narragansett Bay, except for some areas around the Palmer River, the impacts for the entire Narragansett Bay are expected to be even greater than these estimates.

Roads, stormwater and wastewater infrastructure, residential homes, commercial enterprises, beaches, tourism, water quality (surface and groundwater), and habitats are already being impacted by tidal inundation, especially during extreme high tides, causing nuisance flooding in low-lying areas around the Bay. With higher sea levels, local flooding thresholds can be reached more easily during average high tides (Dahl et al. 2017). Nuisance flooding, even with little or no storm surge, is increasingly common along the East and Gulf Coasts of the United States (Sweet et al. 2014). The frequency of nuisance flooding in the Northeast region is projected to increase 25-fold by the year 2045 due to sea level rise (Dahl et al. 2017).

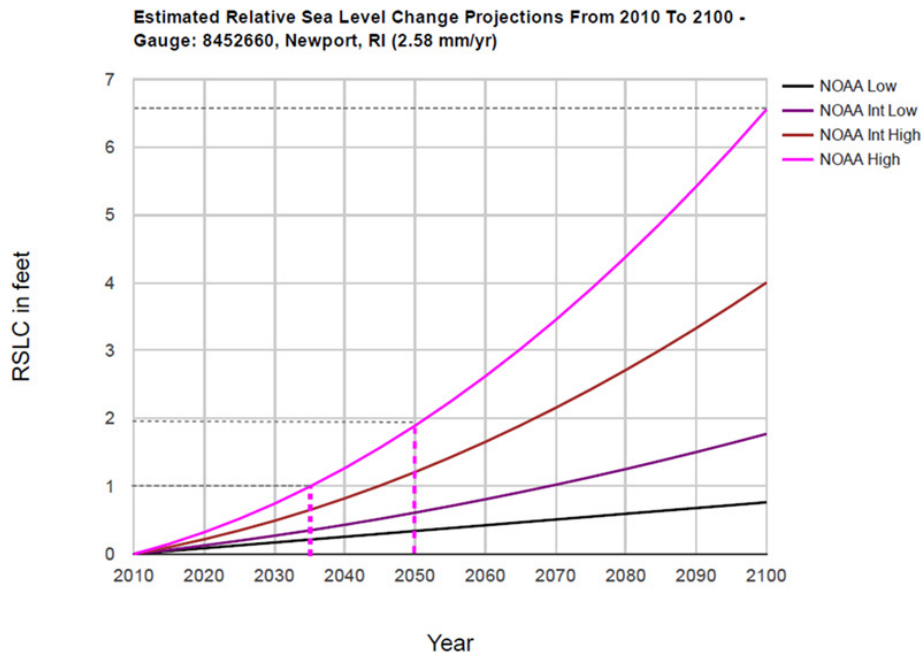


Figure 7. NOAA projections for relative sea level rise from 2010 to 2100 at Newport, Rhode Island, under a range of climate change scenarios. Projections published in 2012 (Parris et al. 2012). See Figure 3 for updated NOAA projections published in 2017. Source: USACE Sea Level Rise Curves

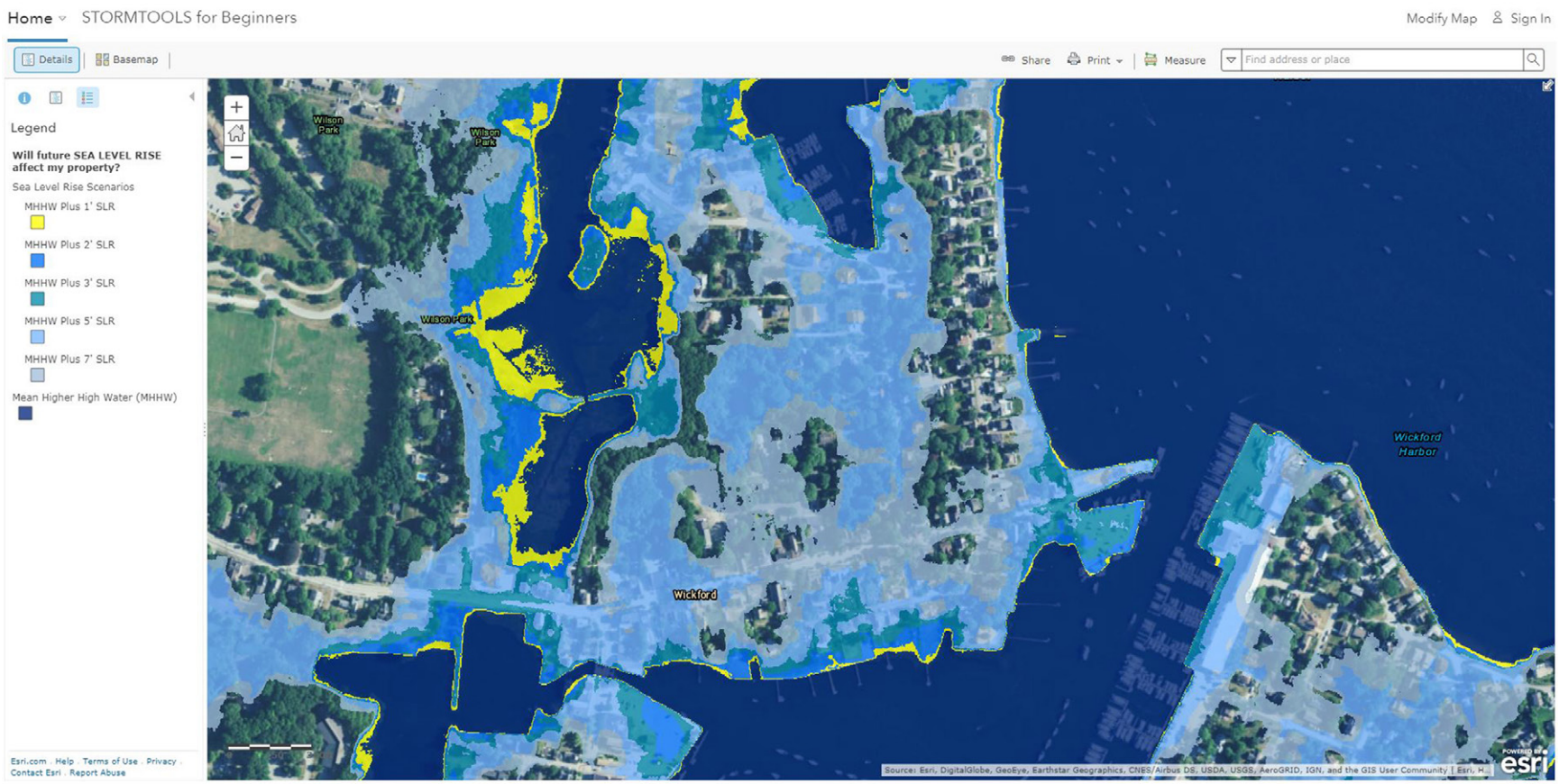


Figure 8. Example of STORMTOOLS viewer for a section of Narragansett Bay's coastal area (Wickford Harbor) Source: STORMTOOLS for Beginners

Many of the buildings subject to flooding are served by onsite wastewater treatment systems, and the combination of elevated groundwater levels driven by sea level rise and overland flooding will put those systems at high risk of failing (Walter et al. 2016). Areas with failed septic systems pose a high risk of water quality degradation along the coast (see "Wastewater Infrastructure" chapter). Likewise, wastewater treatment facilities (WWTF) are also threatened by sea level rise. The Rhode Island Department of Environmental Management (RIDEM) completed a vulnerability study for the major municipal WWTFs, which treat approximately 120 million gallons of sewage per day. With sea level rise of one to five feet, two-thirds of the fifteen coastal WWTFs in the state will experience flooding, compromising water quality and raising public health concerns (RIDEM and Woodard & Curran 2017). Of the facilities located in the Narragansett Bay Watershed, five facilities will be predominantly inundated, four will be partially inundated, and five will not be affected by sea level rise of five feet.

Coastal and estuarine natural habitats are also threatened by higher sea levels. Of the approximately 3,320 acres of salt marsh in Narragansett Bay, a significant proportion may be submerged or lost if sea level rises one to five feet (CRMC 2015; see "Salt Marsh" chapter). Frequent flooding and conversion of some Narragansett Bay salt marshes to mudflats and open water is already occurring, as these coastal wetlands cannot gain sufficient elevation to keep up with sea level rise (CRMC 2015, Raposa et al. 2017). This trend of salt marshes being stressed by sea level rise, among other factors, and becoming wetter has been reported elsewhere in southern New England (Smith 2015, Carey et al. 2017b, Watson et al. 2017). The CRMC, based on SLAMM modeling, expects that total statewide losses of existing coastal wetlands may be 13 percent, 52 percent, or 87 percent under one, three, or five feet of sea level rise, respectively. The SLAMM study also suggested, hypothetically, that if wetlands were able to migrate landward onto currently developed and undeveloped areas the amount of coastal wetlands could increase (CRMC 2015). However, while some coastal areas are currently protected as open space and might potentially be suitable for wetland migration, very few significant natural lands remain unpreserved along the Bay's shoreline (see "Open Space" chapter). The CRMC's results cover Rhode Island only; results for Massachusetts are forthcoming from the Massachusetts Office of Coastal Zone Management.

The loss of coastal wetlands would mean a loss of the protection they provide to the coast as an important natural barrier to storm surge. There is evidence that reefs and wetlands help protect coastlines by reducing wave energy (Shepard et al. 2011, Ferrario et al. 2014, Narayan et al. 2016). A recent study estimated that during Hurricane Sandy the presence of wetlands reduced statewide property damages by $300,000 in Rhode Island and $6,300,000 in Massachusetts, and that on average where wetlands were present they reduced damages by more than ten percent (Narayan et al. 2016).

Seagrass habitats are also threatened by higher sea levels, especially in Narragansett Bay where there is some evidence of slow recovery (see "Seagrasses" chapter). Sea level rise is expected to change the tidal regime and water depth of Narragansett Bay, affecting the distribution of seagrasses (Short and Neckles 1999, Saunders et al. 2013, USEPA 2016). When sea level rises by 1.6 feet, which is projected to occur by 2030 (Sweet et al. 2017), the increase in water depth could reduce seagrass growth in existing seagrass beds by 30 to 40 percent (Short and Neckles 1999), impairing further restoration and recovery efforts, although seagrass can migrate to shallower areas—if conditions are appropriate—as upland areas are submerged (see "Seagrasses" chapter).

The research, findings, predictions, and implications of sea level rise should be considered in planning actions, as many of the Estuary Program's partners are already doing. The states of Rhode Island and Massachusetts are actively planning for future sea level rise. The Rhode Island CRMC accounts for sea level rise in Section 145 of the Rhode Island Coastal Resource Management Plan, and many tools—such as STORMTOOLS (Figure 8) and SLAMM—are available for planners, businesses, and homeowners to understand the future effects of sea level rise.

Data Gaps and Research Needs

- The STORMTOOLS model should be expanded to include the Massachusetts portion of Narragansett Bay to identify and evaluate high-risk areas.
- An analysis of the potential impacts of sea level rise on groundwater, drinking water supplies, floodplains, and individual wastewater treatment systems is needed (Walter et al. 2016).
- Data and research are needed to evaluate the effects of sea level rise on other ecological systems at the landscape and seascape level, such as the impacts on bird, mammal, and amphibian migration and breeding habitat, submerged aquatic vegetation, freshwater wetlands (palustrine and lacustrine), estuarine beaches and shores, shellfish habitat, and fish passage habitat (diadromous and anadromous fish).

- A sea level rise trend analysis is needed for Mount Hope Bay using data from the Fall River tide gauge, which NOAA has operated since 1955. This analysis is especially important because of the low elevations of the Taunton River watershed.
- Enhanced bathymetry data would improve the resolution of the hydrodynamic models that are used to predict flooding potential from sea level rise and storm surge.

Acknowledgments

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State of Narragansett Bay and Its Watershed
2017 Technical Report

Landscape Stressor Indicators

INTRODUCTION

Please use the following citation: Narragansett Bay Estuary Program. 2017. State of Narragansett Bay and Its Watershed (Landscape Stressors Introduction, pages 88-89). Technical Report. Providence, RI.

Photo: Aerial view of the Providence River, RI (Ayla Fox)

NARRAGANSETT BAY ESTUARY PROGRAM

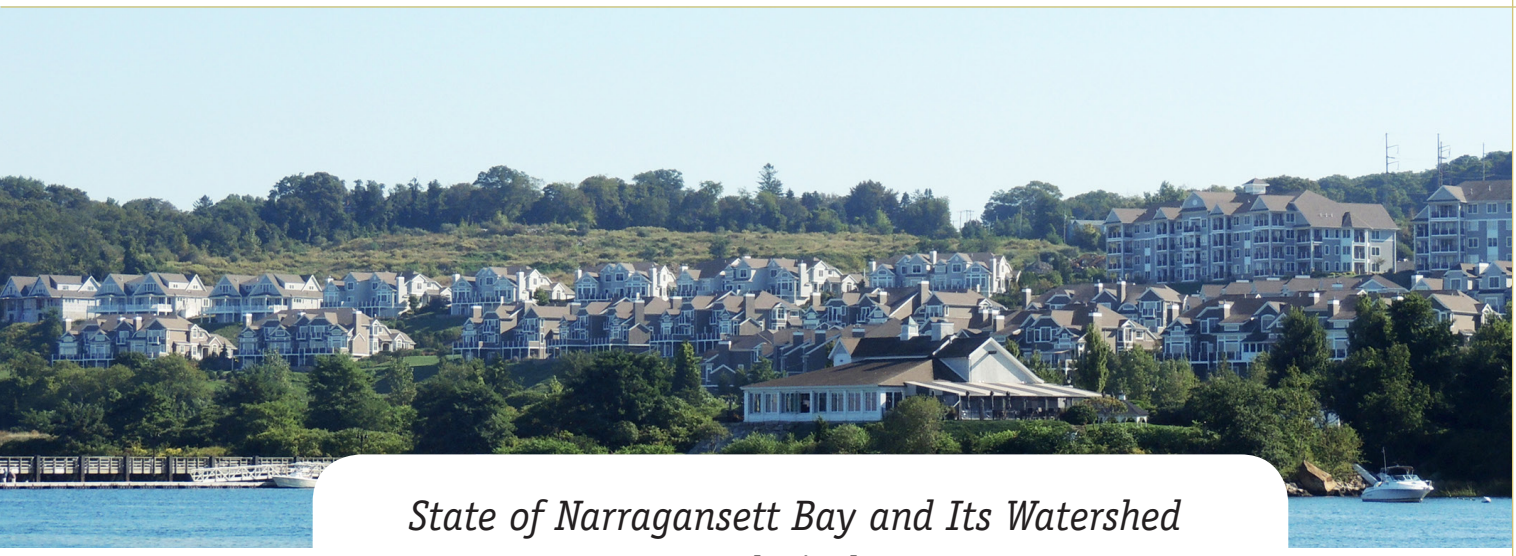
People are a fundamental driving force on the landscape. Human population growth increases the demand for larger or new infrastructure, which leads to land conversion (primarily from forest land to urban land), with construction of new roads, buildings, and other utilities and amenities. These changes in the landscape reshape the Watershed's functions by altering natural hydrological paths, strongly influencing the freshwater and estuarine ecosystems of the Watershed and the Bay. The increase and spread of the human population in the Bay's Watershed and the conversion of forest land to urban land has primarily altered the conveyance of rainfall as stormwater and decreased the connectivity of natural habitats, diminishing the Watershed's capacity to balance the effects of runoff and groundwater replenishment. The synergic impacts of these alterations include increasing flooding (or, conversely, worsening drought conditions), or exacerbating water quality issues through the transportation of contaminants, which, consequently, diminish the benefits of the Watershed's natural resources to public health and aquatic life. Therefore, the shift of the population from crowded urban areas to more rural areas is an important driver of landscape changes, where less people are using more land to settle. Today, previously developed lands provide the infrastructure that a high-density population demand, whereas new development, without proper planning, can create new stressors to the surrounding landscape.

The Narragansett Bay Watershed has three major population centers: Providence, Rhode Island, and Fall River and Worcester, Massachusetts. The locations of these cities near major water sources facilitated the industrial expansion from the mid-1700s to the early 1900s. Population tripled during the industrial era (1850 to 1900) and then doubled over the next century (1900 to 2000). It was during this time that infrastructure such as wastewater treatment facilities, buildings, dams, and roads expanded dramatically to accommodate the growing needs of the population and their livelihoods. These historic changes created the foundation for the development that is currently reshaping the Watershed. The concurrent increases in impervious cover (i.e. roads, parking lots, driveways) and residential and industrial discharges of nutrients, pathogens and other pollutants stressed the Bay and Watershed. In recent years, development has sprawled into suburban and rural areas at a faster rate than the population has grown. At the same time, pollution from wastewater treatment facilities has declined due to improved treatment technologies, improving water quality. However, challenges remain in understanding the extent of pollution from individual onsite systems (septic systems and cesspools), affecting water quality in freshwaters and the estuary

The Narragansett Bay Estuary Program uses five indicators to determine the extent of potential impacts, or changes in the landscape that stress the Bay and Watershed: population, land use, impervious cover, wastewater infrastructure, and nutrient loading. The status and trends of these indicators inform the Estuary Program on where people are living, how they are using the land, and how their waste is being conveyed—all of which affect the condition of the ecosystem and its ability to provide benefits for people and wildlife.



Photo: Lincoln Woods State Park Beach, Lincoln, RI (Ayla Fox)



State of Narragansett Bay and Its Watershed
2017 Technical Report

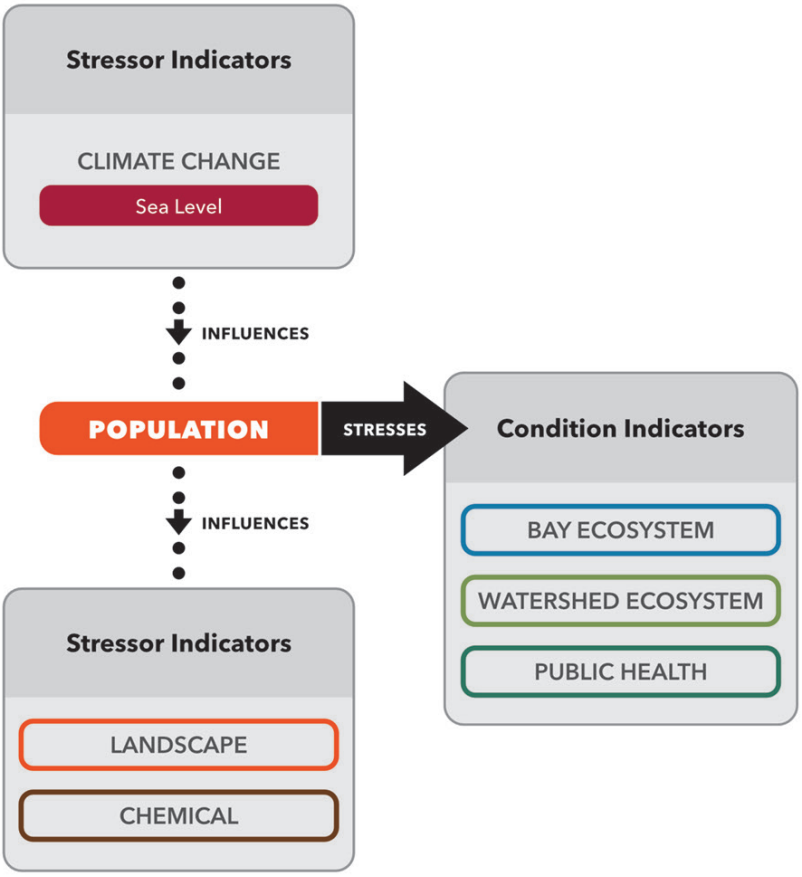
Landscape Stressor Indicators

CHAPTER 4: POPULATION

Please use the following citation: Narragansett Bay Estuary Program. 2017. State of Narragansett Bay and Its Watershed (Chapter 4, Population, pages 90-107). Technical Report. Providence, RI.

Photo: Development along the Sakonnet River, Tiverton, RI (Eivy Monroy)

Overview



BACKGROUND

- Population growth influences many stressors on Narragansett Bay and its Watershed, including all landscape and chemical stressors discussed in other chapters of this report. In numerous ways, population growth affects the condition of the Bay ecosystem, Watershed ecosystem, and human health through loss of natural lands, increased pollutant loadings, declining water quality, changes in biotic communities, and other impacts.

KEY FINDINGS

- ***Status:*** In 2014, an estimated 1.9 million people lived in the Narragansett Bay Watershed with approximately half in Massachusetts and half in Rhode Island. Most of the population was concentrated in three urban areas: Providence, Rhode Island, and Worcester

and Fall River, Massachusetts. Twenty percent of the Watershed population resided in coastal lands that drain directly into Narragansett Bay.

- ***Trends:*** The population of the entire Narragansett Bay Watershed more than tripled between 1850 and 1900, and then it more than doubled between 1900 and 2000. Between 1990 and 2010, the population increased by eight percent. This recent population growth was concentrated in the suburban and lesser-developed areas of the Watershed, primarily in the watersheds of the Taunton River and Pawtuxet River; in contrast, coastal areas experienced a slight population decline of two percent. The amount of developed land per capita was much higher in rural and suburban areas than in urban areas and increased over time, showing intensifying sprawl.

Introduction

It is well established that the way human society uses and protects the land within a watershed has critical implications for freshwater streams, estuarine waters, and associated habitats. Human population growth is one of the fundamental driving forces in land conversion, bringing increases in infrastructure, transportation, and commerce (Meyer and Turner 1992, August et al. 2002). Human populations are often concentrated in coastal regions for economic and recreational resources, and population density can have direct effects in the watershed, such as altering runoff patterns, decreasing the amount of habitat, and introducing exotic species (Niemi et al. 2007). Higher population densities are associated with higher amounts of impervious surfaces and infrastructure, such as water and sewer lines, utility corridors, and roads, all influencing the natural movement of water and directly contributing to point and nonpoint pollution.

To examine changes in land use in relation to population density, the amount of developed land per capita can be calculated (Tu et al. 2006). This metric indicates the extent and rate of sprawl, a term used to describe the spread of development into rural or undeveloped areas despite population levels staying level or even declining. Urban areas with high population density generally have low amounts of developed land per capita, as people in urbanized areas tend to occupy less developed land. Conversely, rural or suburban areas may have lower population density but a higher amount of developed land per capita, meaning that each person effectively occupies more developed land than in urban areas (Tu et al. 2006).

Because population growth is linked intrinsically to land use change, the Narragansett Bay Estuary program analyzed spatial patterns of land development over time in relation to population growth. In some parts of the Narragansett Bay Watershed, the amount of developed land has increased at a faster rate than the population has grown. The movement of population into rural areas has the greatest potential for increased habitat fragmentation, loss of forested lands, and increases in pavement and other impervious land cover. For this chapter, the Estuary Program and partners developed an indicator to examine the total population and where people live across the Narragansett Bay Watershed, often referred to as "population distribution". For this indicator, a dasymetric model was used to identify population density, which made it possible to quantify total population living in the Watershed and to calculate changes spatially, from 1990 through 2010. The changes of total population and the spatial distribution of these changes allows for the examination of where people have moved to or from, thereby influencing changes across the landscape. The Estuary Program mapped and computed the total population that dwelled within the boundaries of the Watershed as of 2010 and used total population from 1990 and 2000 to calculate recent changes. Results of the population distribution analysis were integrated into other chapters of this report, reinforcing the fact that people alter the landscape (for example, see "Land Use" and "Open Space" chapters), contribute to pathogen and nutrient loadings (see "Wastewater Infrastructure" chapter), and are affected by climate change (see "Sea Level" chapter).

Methods

The methods for analyzing population were developed by the US Environmental Protection Agency (EPA) Office of Research and Development (ORD), National Health and Environmental Effects Research Laboratory, Atlantic Ecology Division in collaboration with the Narragansett Bay Estuary Program and other partners. Through a workgroup of GIS specialists, the data and data processing for this indicator were examined and selected. The following were mapped and calculated at the Watershed and other geospatial scales (HUC10 watersheds and HUC12 subwatersheds; see the Appendix for definitions and lists):

- Total population as projected by US Census Bureau for 2014
- Total population for 1990, 2000, and 2010 by means of dasymetric analysis
- Total population changes as gross and net percent changes
- Developed land per capita (acres of land per one hundred people) for 1990, 2000, and 2010

Population (as projected by the Census Bureau for 2014)

US Census Bureau's projections for 2014 were used to estimate recent population sizes in the Narragansett Bay Watershed. The Census Bureau provides projections of population using a cohort-component method and assumptions about demographic components of change (future trends in births, deaths, and net international migration). The Census Bureau's block group data for Massachusetts and Rhode Island were used to calculate the 2014 population using area weighting models that aggregate data from block groups within the perimeter of the Watershed (Esri 2016). The last US Census was completed in 2010. To provide a more recent estimate of population, the Estuary Program decided to use the latest projections that were available (2014).

Dasymetric Analysis

To analyze temporal and spatial trends in population across the Watershed, a dasymetric model and mapping technique were used to map population density relative to residential land use types (Sleeter and Gould 2007, USGS 2015). Dasymetric modeling and mapping of human population density are geospatial techniques used to increase the spatial resolution of population data by incorporating related ancillary data layers such as land use and land cover (USGS 2015). Tracking spatial trends in population density assists in the detection of the phenomenon of sprawl. The dasymetric mapping tool used Census Bureau data from 1990, 2000, and 2010, and corresponding years for land use data (Table 1). The dasymetric model was run using datasets obtained from Massachusetts and Rhode Island state-level datasets because they provide a finer spatial resolution than the National Land Cover Database, and the years in which the data were mapped correspond more closely to the Census Bureau data.

The dasymetric model provided results for population density as the number of people within an area of ten meters by ten meters. For the purpose of mapping and reporting, density units were converted to number of people per acre of land. From the population density, dasymetric total population estimates were determined for each state, clipped to the Narragansett Bay Watershed boundary, and then summarized at the Watershed, HUC10 watershed, and HUC12 subwatershed scales at the three time-steps of 1990, 2000, and 2010. For changes across these time-steps, gross change (totals) and net percent change (percent of population change between two time-steps) were calculated for each watershed scale.

Table 1. Data sources used for dasymetric analysis (US Census and state land use data).

Census Bureau Year	Massachusetts (MassGIS)	Rhode Island (RIGIS)
1990	MA 1985	RI 1988
2000	MA 1999	RI 2003–04
2010	MA 2005	RI 2011

Heat Map of Population Change

In addition, a kernel density analysis was performed to highlight the changes (increases and decreases) in population from 1990 to 2010. A kernel density analysis takes known quantities of some phenomenon (e.g., change in population) and spreads them across the landscape based on the quantity that is measured at each location and the spatial relationship of the locations of the measured quantities (Esri 2016). A heat map was generated using the dasymetric population estimates by subtracting 2010 from 1990 population (both datasets were in GIS raster/cell format). One dataset with the new values was used to generate points that represent the population gross change in that particular raster (box) for the entire surface of Narragansett Bay Watershed. Any points in a waterbody larger than one acre were removed (Esri 2016). Using the remaining points, a kernel density raster at a cell size of one hundred meters (328 feet) was created, with a search radius of one thousand meters (3,281 feet). This geospatial analysis provides a visual illustration of where population changes were more substantial, indicating the movement of people over one decade across the Watershed lands.

Developed Land Per Capita

To evaluate how population has driven sprawl, the amount of developed land per capita was calculated, using the dasymetric total population estimates and data on urban land use (Vogelmann et al. 2001, Homer et al. 2007, Homer et al. 2015). This metric was calculated as developed acres of HUC10 watersheds and HUC12 subwatersheds per one hundred people. A high value of the metric indicated that population in the Watershed had driven sprawl, whereas a low value indicated that even though population density might be high, it was concentrated in smaller unit area.

Historical Trends

Historical changes in population that encompass eras of industrialization and suburbanization are presented based on research by Vadeboncoeur and colleagues (2010).

Status and Trends

STATUS OF POPULATION

The estimated population in the Narragansett Bay Watershed based on the 2014 census projections was 1,962,003 people (Table 2). Slightly more than half (51 percent) of the population lived in Massachusetts and 49 percent in Rhode Island. Sixty percent of the Watershed's total area is in Massachusetts.

The 2014 estimates were similar to the 2010 census results, despite the data and methodological differences. Population based on dasymetric modeling estimated a total of 1,949,764 people in the Watershed—just over 12,000 fewer than the 2014 projections (Tables 2 and 3). The 2010 census population density (Figure 1), coupled with total population within each of the eleven HUC10 watersheds (Table 3), illustrates the range and distribution.

The watershed (HUC10) with the largest population was the coastal Narragansett Bay watershed, which includes Providence and much of Fall River, with 384,963 people. It accounted for 20 percent of the entire population of the Narragansett Bay Watershed, while the least-populated watershed was the Palmer River, with only two percent of the total population (Table 3; Figure 1).

At the subwatershed (HUC12) scale, the largest population was concentrated in Worcester (Tatnuck Brook-Blackstone River subwatershed) with 139,119 people (Table 4). In coastal areas of Narragansett Bay, the Seekonk River-Providence River and Woonasquatucket River subwatersheds also were heavily populated (Table 4). The least-populated HUC12 subwatersheds included the Upper West Passage, which encompasses the industrial areas of Quonset Point and the sparsely populated western portion of Prudence Island, the Big River, the Barden and Scituate Reservoirs, and the Chepachet River (Table 4).

Associating the nested HUC12 subwatersheds within the HUC10 watersheds assists with the detection of where population may be considered a stressor indicator even in rural, less-developed areas of the Watershed. In the least-populated HUC10 watershed, the Palmer River, 52 percent of the population was concentrated in the Barrington River-Warren River subwatershed. The five subwatersheds with the greatest percent of the Watershed's population (Table 4) correspond to the areas with the highest concentrations of people living in the Watershed, which have also been determined as having the greatest percentage of the HUC10's population (Table 5).

The complete results of Tables 4 and 5 for all the HUC12 subwatersheds are available upon request.

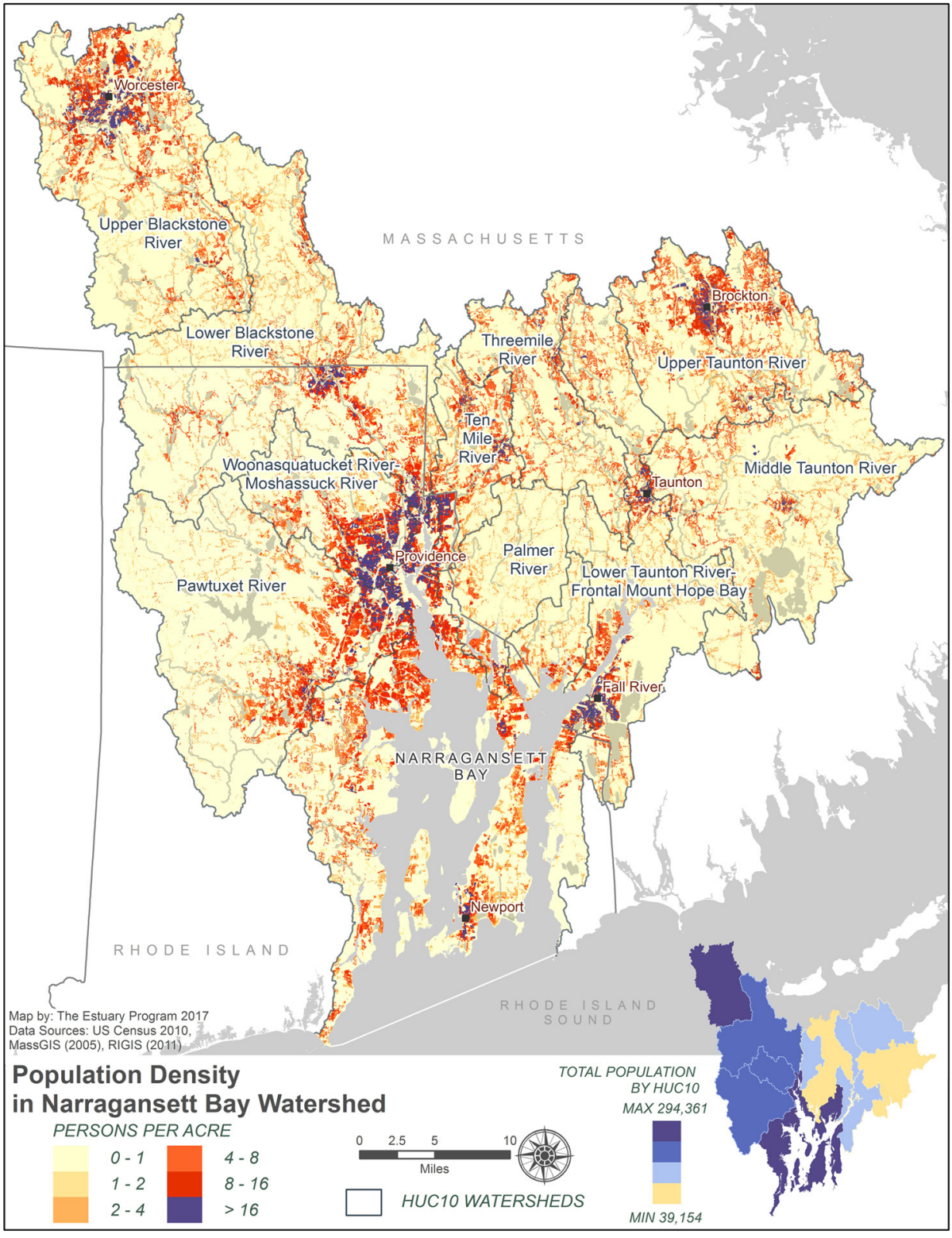


Figure 1. Dasymetric model representing 2010 population density and distribution in the Narragansett Bay Watershed. Inset map: Total population for each watershed (HUC10)

Table 2. Estimate of the total population in the Massachusetts and Rhode Island sections of the Narragansett Bay Watershed based on US Census Bureau Block Projections for 2014.

State within the Watershed	Estimated Population	Percentage of Bay Watershed Population
Massachusetts section of Watershed	1,009,890	51
Rhode Island section of Watershed	952,113	49
Total Narragansett Bay Watershed	**1,962,003**	

Table 3. Total population in watersheds (HUC10) of Narragansett Bay and percentage of the total Watershed's population based on dasymetric modeling of Census 2010.

Watershed (HUC10) Name[1]	Population	Percentage of Watershed's Population
Narragansett Bay	384,963	20
Upper Blackstone River	294,361	15
Woonasquatucket River-Moshassuck River	208,973	11
Pawtuxet River	208,714	11
Lower Blackstone River	203,717	10
Upper Taunton River	198,325	10
Lower Taunton River-Frontal Mount Hope Bay	159,335	8
Ten Mile River	106,129	5
Middle Taunton River	81,228	4
Threemile River	64,865	3
Palmer River	39,154	2
Total Narragansett Bay Watershed	**1,949,764**	

[1] Watersheds (HUC10) are listed from highest to lowest population. See the Appendix for HUC10 code numbers and map.

Table 4. Total population in the five most-populated and five least-populated subwatersheds (HUC12) in the Narragansett Bay Watershed and percentage of the total Watershed's population based on dasymetric modeling of Census 2010.

Subwatershed (HUC12) Name[1]	Population	Percentage of Watershed's Population
	Highest population	
Tatnuck Brook-Blackstone River	139,119	7
Seekonk River-Providence River	125,740	6
Woonasquatucket River	125,517	6
Matfield River	108,987	6
Ten Mile River	106,129	5
	Lowest population	
Chepachet River	5,421	0.3
Scituate Reservoir	4,850	0.2
Barden Reservoir-Ponaganset River	3,751	0.2
Big River	2,586	0.1
Upper West Passage	1,946	0.1

[1] Subwatersheds (HUC12) are listed from highest to lowest population. See the Appendix for HUC12 code numbers and map.

Table 5. Name and population of the subwatershed (HUC12) having the greatest percentage of each HUC10 watershed's total population.

Watershed (HUC10) Name	*Nested HUC12 subwatersheds[1] Name and Population*		Percentage of HUC10 watershed's population[2]
Lower Blackstone River	*Peters River-Blackstone River*	*78,469*	39
Lower Taunton River-Frontal Mount Hope Bay	*Taunton River-Frontal Mount Hope Bay*	*56,217*	35
Middle Taunton River	*Cotley River-Taunton River*	*40,847*	50
Narragansett Bay	*Seekonk River-Providence River*	*125,740*	33
Palmer River	*Barrington River-Warren River*	*20,343*	52
Pawtuxet River	*Pawtuxet River[3]*	*94,955*	45
Ten Mile River	*Ten Mile River[4]*	*106,129*	100
Threemile River	*Threemile River*	*37,730*	58
Upper Blackstone River	*Tatnuck Brook-Blackstone River*	*139,119*	47
Upper Taunton River	*Matfield River*	*108,987*	55
Woonasquatucket River-Moshassuck River	*Woonasquatucket River*	*125,517*	60

[1] HUC12 subwatershed nested in HUC10 watershed, containing the greatest percentage of the HUC10's population.
[2] Total HUC12 population divided by the total HUC10 population.
[3] USGS defines the Pawtuxet River HUC12 subwatershed as the area draining to the river emptying to the Bay; it is nested within the Pawtuxet River HUC10 watershed.
[4] USGS defines the Ten Mile River HUC10 watershed and HUC12 subwatershed as the same drainage area. Therefore, the HUC12 subwatershed contains one hundred percent of the HUC10 population.

POPULATION CHANGES AND TRENDS

Changes in population based on dasymetric modeling performed for each time-step of census data (1990, 2000, and 2010) revealed that population across the Watershed increased by 141,812 people (gross change) in two decades, an eight percent increase. However, the majority of that increase in population (six percent) occurred between the 1990 and 2000 census (Table 6).

In recent trends between 1990 and 2000, the HUC10 watersheds that experienced the largest increases in population included the Upper and Lower Blackstone River and the Woonasquatucket River-Moshassuck River (Table 6). The Middle Taunton River and the Threemile River HUC10 watersheds experienced the largest percentage increases in population–20 and 22 percent respectively–over the two decades. Conversely, the currently most-populated watershed–the Narragansett Bay HUC10 watershed, which includes the lands directly adjacent to and entirely surrounding Narragansett Bay–experienced a population decline of 6,382 people, representing a two percent decrease from 1990 to 2000 (Table 6). Within that HUC10 watershed, the loss of population can be seen in more detail at the subwatershed (HUC12) scale. Two subwatersheds on Aquidneck Island had the greatest percentage decreases at -10 percent and -15 percent (Table 7), while the Seekonk River-Providence River subwatershed had the largest gross population decline, losing 3,339 people (-3 percent change) (Table 8).

Of the 52 HUC12 subwatersheds in the Narragansett Bay Watershed, 42 had increases in population between 1990 and 2010, and the remaining ten experienced a decline in population. Among all the subwatersheds (HUC12) around Narragansett Bay, the largest percent increases in population between 1990 and 2010 occurred in the West River (38 percent) and Big River (30 percent) subwatersheds (Table 7; Figure 2). The heat map (Figure 2) shows the areas with the largest gross population increases and declines. The largest population increase at the HUC12 scale was in the Woonasquatucket River subwatershed, where a total of 14,947 people likely moved into its suburban areas. The second largest increase occurred in the Cotley River-Taunton River subwatershed, where population increased by 10,782 (Table 8). The increases in the Taunton River Basin can be seen in Taunton and Bridgewater in Figure 2.

Of the 52 HUC12 subwatersheds, eleven added more than 5,000 people between 1990 and 2010, and the largest increase was nearly 15,000 people (Table 8). Conversely, six subwatersheds decreased

Table 6. Total population in 1990, 2000, and 2010, and gross and percent change between decades, in HUC10 watersheds and the entire Narragansett Bay Watershed.

Watershed (HUC10) Name[1]	Population			Gross Change (% Change)		% Change	
	1990	2000	2010	**1990–2010**		1990–2000	2000–2010
Middle Taunton River	63,434	73,440	81,228	17,794	(22)	14	10
Threemile River	51,944	61,263	64,865	12,921	(20)	15	6
Lower Blackstone River	179,030	193,587	203,717	24,687	(12)	8	5
Upper Blackstone River	265,811	276,790	294,361	28,550	(10)	4	6
Woonasquatucket River-Moshassuck River	189,104	204,918	208,973	19,869	(10)	8	2
Lower Taunton River-Frontal Mount Hope Bay	148,386	159,970	159,335	10,949	(7)	7	0
Pawtuxet River	195,829	204,065	208,714	12,885	(6)	4	2
Palmer River	36,744	38,641	39,154	2,410	(6)	5	1
Upper Taunton River	186,402	195,473	198,325	11,923	(6)	5	1
Ten Mile River	99,922	104,984	106,129	6,207	(6)	5	1
Narragansett Bay	391,345	394,658	384,963	-6,382	(-2)	1	-3
Total Narragansett Bay Watershed	*1,807,952*	*1,907,790*	*1,949,764*	*141,974*	*(8)*	*6*	*2*

[1] Watersheds (HUC10) are listed from highest to lowest by percent change in population. See the Appendix for HUC10 code numbers and map.

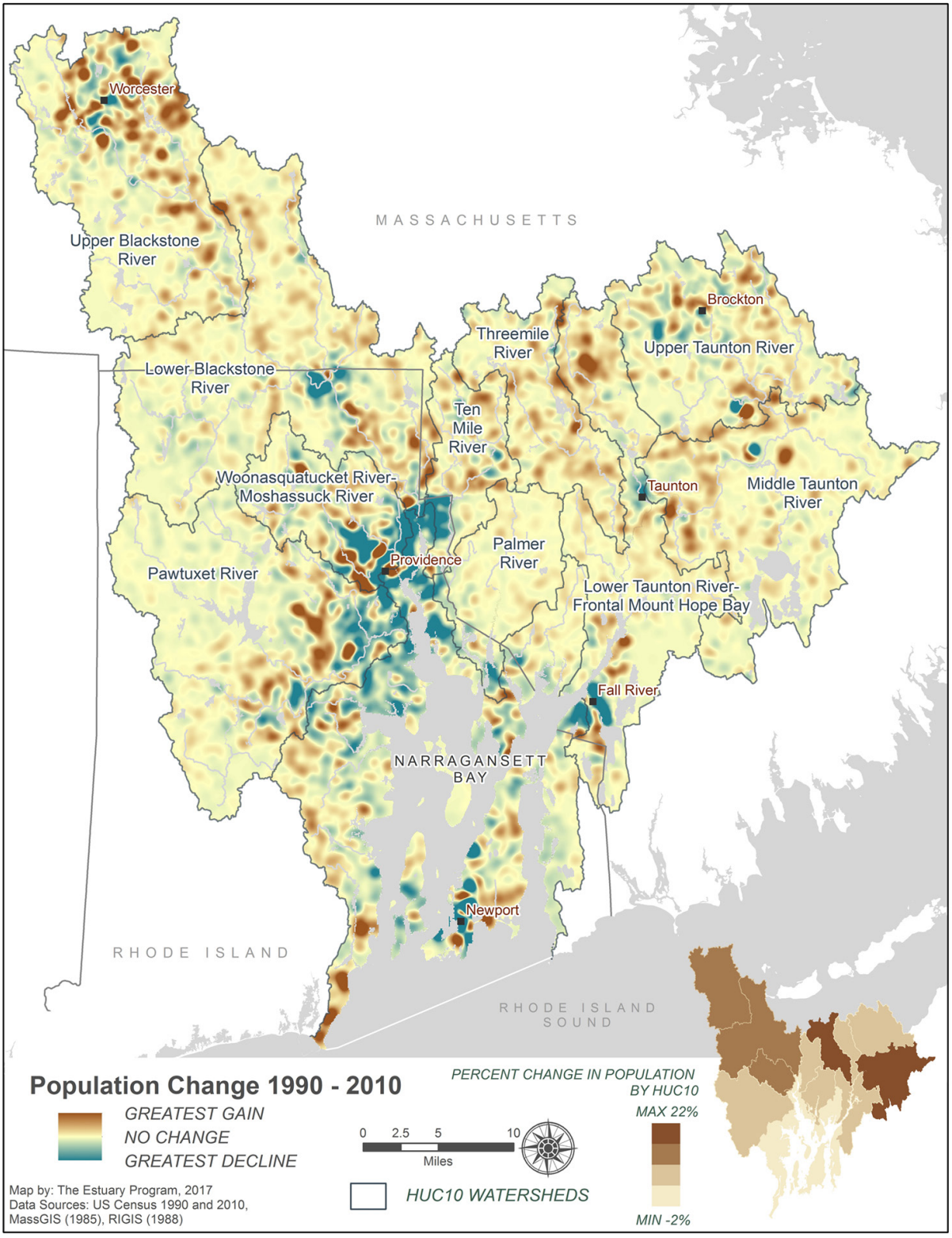


Figure 2. Heat map of population change from 1990 to 2010 in watersheds of Narragansett Bay. Inset map: Percent change in population for each watershed (HUC10)

Table 7. Subwatersheds (HUC12) with the largest increases (over 20 percent) and decreases (over 3 percent) from 1990 to 2010 (net percentage change).

Subwatershed (HUC12) Name[1]	Total Population			% Change		
	1990	2000	2010	1990–2000	2000–2010	**1990–2010**
Largest % Increases (1990–2010)						
West River	8,385	10,451	13,568	20	23	38
Big River	1,815	2,300	2,586	21	11	30
Cotley River-Taunton River	30,064	36,548	40,847	18	11	26
Winnetuxet River-Taunton River	11,844	13,767	15,686	14	12	24
Wading River	21,105	25,092	27,135	16	8	22
Assonet River	6,726	7,997	8,525	16	6	21
Abbott Run	19,427	22,246	24,350	13	9	20
Largest % Decreases (1990–2010)						
Seekonk River-Providence River	129,079	129,353	125,740	0.2	-3	-3
Greenwich Bay	46,798	47,363	45,484	1	-4	-3
Quequechan River	51,680	50,918	49,550	-1	-3	-4
Upper East Passage	17,900	17,367	17,077	-3	-2	-5
Upper Narragansett Bay	21,971	21,680	20,748	-1	-4	-6
Aquidneck Island-Frontal Atlantic Ocean	20,094	19,792	18,218	-2	-9	-10
Lower East Passage	25,248	23,546	21,933	-7	-7	-15

[1] See the Appendix for HUC12 code numbers and map.

Table 8. Subwatersheds (HUC12) of the Narragansett Bay Watershed with the largest increases and decreases in gross total population from 1990 to 2010.

Subwatershed (HUC12) Name	Population		Gross Total Change
	1990	2010	2000–2010
Increased by more than 5,000 people:			
Woonasquatucket River	110,571	125,517	14,946
Cotley River-Taunton River	30,064	40,847	10,783
Quinsigamond River	56,249	66,774	10,525
Mill River (Taunton)	37,775	45,043	7,268
Threemile River	30,839	37,730	6,891
Tatnuck Brook-Blackstone River	132,318	139,119	6,801
Peters River-Blackstone River	71,800	78,469	6,669
Ten Mile River	99,922	106,129	6,207
Wading River	21,105	27,135	6,030
Singletary Brook-Blackstone River	31,740	37,183	5,443
West River	8,385	13,568	5,183
Decreased by more than 1,000 people:			
Upper Narragansett Bay	21,971	20,748	-1,223
Greenwich Bay	46,798	45,484	-1,314
Aquidneck Island-Frontal Atlantic Ocean	20,094	18,218	-1,876
Quequechan River	51,680	49,550	-2,130
Lower East Passage	25,248	21,933	-3,315
Seekonk River-Providence River	129,079	125,740	-3,339

in population by more than 1,000 people. These gains and losses are illustrated in Figure 2. The complete results of Tables 7 and 8 for all the HUC12 subwatersheds are available upon request.

DEVELOPED LAND PER CAPITA

The amount of developed land per capita is a measure of the development patterns in the HUC10 watersheds or HUC12 subwatersheds (Table 9). Densely populated urban areas have lower amounts of developed land per capita, whereas suburban or rural areas, especially those with sprawl, have higher amounts. The subwatersheds (HUC12) with the highest and lowest amounts of developed land per capita are listed in Table 10.

The three least populated watersheds (HUC10) across the Narragansett Bay Watershed (Table 3) have also ranked the highest for developed lands per capita (Table 9), identifying areas where fewer people use more acres of land in the Watershed. In contrast, in urban areas (see "Land Use" chapter) where population density is highest (Figure 1), more people use less land per capita, and the land is primarily already developed.

The HUC10 watersheds with the largest amounts of developed land per capita were the Middle Taunton River and the Palmer River watersheds, which in 2010 had nearly 40 acres of developed land per one hundred people (Table 9). In contrast, the Woonasquatucket River-Moshassuck River watershed had only twelve acres of developed land per one hundred people (Table 9).

Although the Middle Taunton watershed was one of the least-developed watersheds (Table 3), it ranked as having the most developed land per capita, and the amount increased consistently over two decades (Table 9). The increase seems driven by population growth since this watershed also experienced the greatest percent increase in population (Tables 3 and 6).

Comparing subwatersheds (HUC12) with the highest and lowest developed land per capita, it is important to highlight that some residential areas have mixed land uses. The Upper West Passage, the least populated of the subwatersheds, had the highest amount of developed land per capita across the Watershed from 1990 to 2010 (Table 10), as a result of the Quonset Point industrial park. In contrast, the densely populated Seekonk River-Providence River subwatershed had only ten acres of developed land per one hundred people, equivalent to 0.1 acre per person (Table 10).

Most important is tracking population increases in areas of the Narragansett Bay Watershed that have more potential for sprawl. For example, the population of the Big River subwatershed increased

Table 9. Developed land per capita in 1990, 2000, and 2010 in the HUC10 watersheds of the Narragansett Bay Watershed. Sorted from highest to lowest values of developed land per capita in 2010.

Watershed (HUC10) Name[1]	Developed Acres per 100 people		
	1990	2000	2010
Middle Taunton River	34	38	39
Palmer River	31	35	38
Threemile River	30	32	33
Lower Blackstone River	21	24	26
Lower Taunton River-Frontal Mount Hope Bay	17	19	21
Pawtuxet River	18	19	21
Narragansett Bay	18	19	20
Upper Taunton River	17	19	19
Upper Blackstone River	16	18	19
Ten Mile River	17	18	18
Woonasquatucket River-Moshassuck River	17	11	12
Total Narragansett Bay Watershed	**18**	**20**	**21**

[1] See the Appendix for HUC10 code numbers and map.

by 30 percent, causing more land to be developed. The number of acres of developed lands per one hundred people increased from 76 to 82 acres (Table 10), promoting sprawl in suburban and more rural areas.

The complete results of Table 10 for all the HUC12 subwatersheds are available upon request.

HISTORICAL TRENDS

Between 1850 and 1900, the population of the entire Narragansett Bay Watershed more than tripled (Table 11; Vadeboncoeur et al. 2010). Most of the population growth during that period was concentrated in the coastal areas of the upper sections of the Bay, including Providence and Fall River, where population quadrupled (Table 11). Between 1900 and 2000, the population in the Watershed more than doubled.

Between 1850 and 1950, the subwatersheds containing the greatest percentages of the Watershed's total population were the upper Bay (ranging between 32 and 40 percent) and the upper Blackstone River (18 to 20 percent) (Table 12; Vadeboncoeur et al. 2010).

Discussion

Population growth is an important indicator of underlying pressure and a driver of many related stressors (see the chapters in this report on landscape stressors and chemical stressors). In the Narragansett Bay Watershed, trends in population growth can be analyzed back to the 1850s. Vadeboncoeur and colleagues (2010) calculated that the Watershed's population increased from 260,660 people in 1850 to 1,936,117 people in 2000 (Figure 3). Population estimates for 1990 and 2000 by the Estuary Program were comparable to those by Vadeboncoeur and colleagues (2010), which validates both methods. The Estuary Program used methods that integrated population and land use to depict spatial distribution of population (where people live) within the boundaries of the Watershed at a finer resolution than the analysis by Vadeboncoeur and colleagues (2010). The latter determined total population within municipal boundaries, including population outside the Watershed boundaries, without depicting geographically where people lived. Thus, new geospatial demographic methods such as dasymetric modeling and mapping allow for spatial enhancement of the Census Bureau data, providing improved population estimates and trend analyses in population growth

Table 10. Subwatersheds (HUC12) with the highest and lowest amounts of developed land per capita between 1990 and 2010. Sorted by 2010 values of developed land per capita.

HUC12 subwatersheds	Developed Acres per 100 people		
Name	1990	2000	2010
More than 50 developed acres per 100 people:			
Upper West Passage	112	101	100
Big River	76	83	82
Barden Reservoir-Ponaganset River	55	57	67
Scituate Reservoir	43	49	54
Assonet River	42	48	54
Winnetuxet River-Taunton River	35	45	51
Moswansicut Pond-Huntinghouse Brook	41	46	51
Less than 15 developed acres per 100 people:			
Pawtuxet River	13	14	14
Matfield River	12	13	14
Quequechan River	10	11	13
Woonasquatucket River	12	11	12
Moshassuck River	11	11	11
Tatnuck Brook-Blackstone River	11	11	11
Seekonk River-Providence River	10	10	10

Table 11. Historical rates of population growth in the entire Narragansett Bay Watershed and in subwatersheds within it (Vadeboncoeur et al. 2010).

	Industrialization		Suburbanization	Century
Subwatersheds	**1850–1900**	**1900–1950**	**1950–2000**	**1900–2000**
Blackstone River above Millville	3.4	1.8	1.1	2.0
Blackstone R. Millville to Manville	2.3	1.8	1.2	2.1
Pawtuxet R. above Pettaconsett	1.4	1.6	3.4	5.4
Taunton R. above Bridgewater	2.9	1.5	2.0	3.1
Taunton R. Bridgewater to Taunton	2.7	1.3	1.6	2.1
Taunton R. below Taunton	1.4	2.2	2.8	6.2
Small watersheds	4.1	2.0	1.6	3.1
Upper Bay	4.0	1.6	1.0	1.6
Lower Bay	1.9	2.1	1.6	3.5
Total Narragansett Bay Watershed	**3.2**	**1.7**	**1.4**	**2.3**

Table 12. Historical percentage of population in subwatersheds of the Narragansett Bay Watershed (Vadeboncoeur et al. 2010).

Subwatersheds	**1850**	**1900**	**1950**
Blackstone River above Millville	18	19	20
Blackstone R. Millville to Manville	6	5	5
Pawtuxet R. above Pettaconsett	6	3	2
Taunton R. above Bridgewater	12	10	9
Taunton R. Bridgewater to Taunton	5	4	3
Taunton R. below Taunton	4	2	2
Small watersheds	10	13	15
Upper Bay	32	40	38
Lower Bay	8	4	5
Total Narragansett Bay Watershed	**260,660**	**832,438**	**1,417,816**

across and within the boundaries of the Narragansett Bay Watershed. This was particularly important and useful for conducting analyses for this chapter at different watershed scales.

Even though 60 percent of the Watershed is in Massachusetts, its population of 1.9 million was nearly evenly divided between Massachusetts and Rhode Island. Overall, the historical centers of urban and industrial development (Providence, Worcester, and Fall River) continued to dominate as population centers (Figure 1). While 20 percent of the 1.9 million people in the Watershed lived within the areas immediately draining the estuarine waters of the Bay (Table 3), development patterns in the last 20 years or so show losses in these urban cores, especially Providence and Fall River, and gains in adjacent suburban areas (Figure 2). On average, 2,200 people moved from each of these coastal subwatersheds, and the percent decline ranged from three to fifteen percent of the 1990 population (Tables 7 and 8). In contrast, population growth was more evident in suburban areas, as people moved where there was more potential for new development, promoting sprawl across the Watershed. This pattern was localized in many subwatersheds of the Taunton River and the rural, forested areas (see "Land Use" chapter) of the Pawtuxet River and Blackstone River subwatersheds, where population increased between 20 and 38 (Table 7, Figure 2).

Consequently, population growth seems to have driven land development in the lesser-developed areas. The least-populated watersheds and subwatersheds (Tables 3 and 4) had the most developed land per capita (Tables 9 and 10). In some of these less-populated areas, population growth was substantial over the last two decades (Tables 6 and 7). People moving to the lesser-developed areas used more acreage per person, promoting landscape changes through habitat fragmentation (see "Land Use" and "Open Space" chapters) and increases in impervious cover and wastewater infrastructure (see "Impervious Cover" and "Wastewater Infrastructure" chapters). These landscape changes also alter the hydrological regime of the Watershed and contribute to pollutants loadings to the receiving waters (see "Nutrient Loadings" chapter). Landscape changes can also affect wildlife and human uses of the Bay and fresh waters in the Watershed (see "Water Quality Conditions for Aquatic Life," "Water Quality Conditions for Recreation," "Marine Beaches," and "Shellfishing Areas" chapters). In addition, people living near the coast of Narragansett Bay are vulnerable to the effects of climate change as sea level rise and storm surge are imminent threats (see "Sea Level" chapter).

In addition to the results reported above, the Estuary Program explored data by the SILVIS Lab (2014) on the number of household units per square kilometer in the Narragansett Bay Watershed in 1940, 1970, and

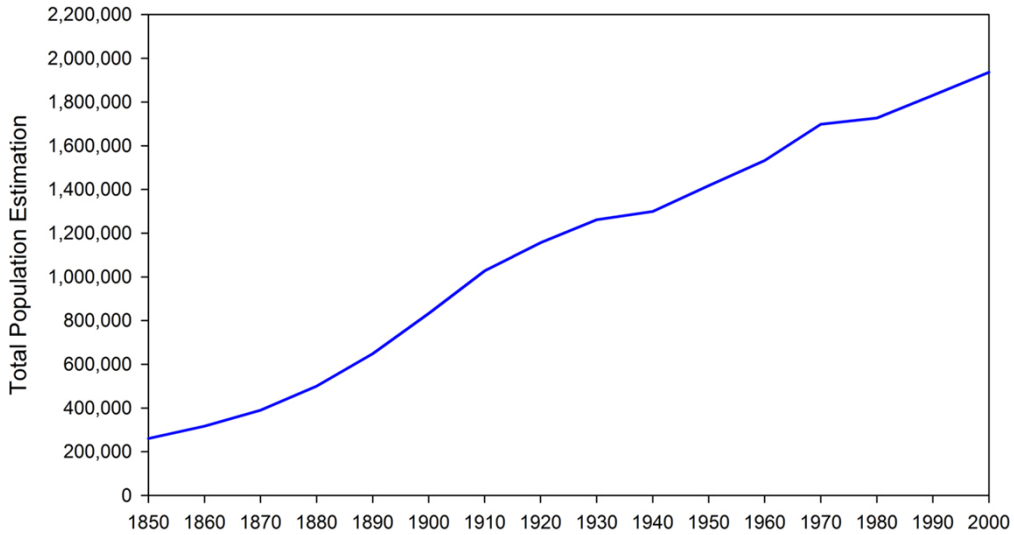


Figure 3. Estimated total population from 1850 to 2000 in Narragansett Bay Watershed as calculated by Vadeboncoeur and colleagues (2010).

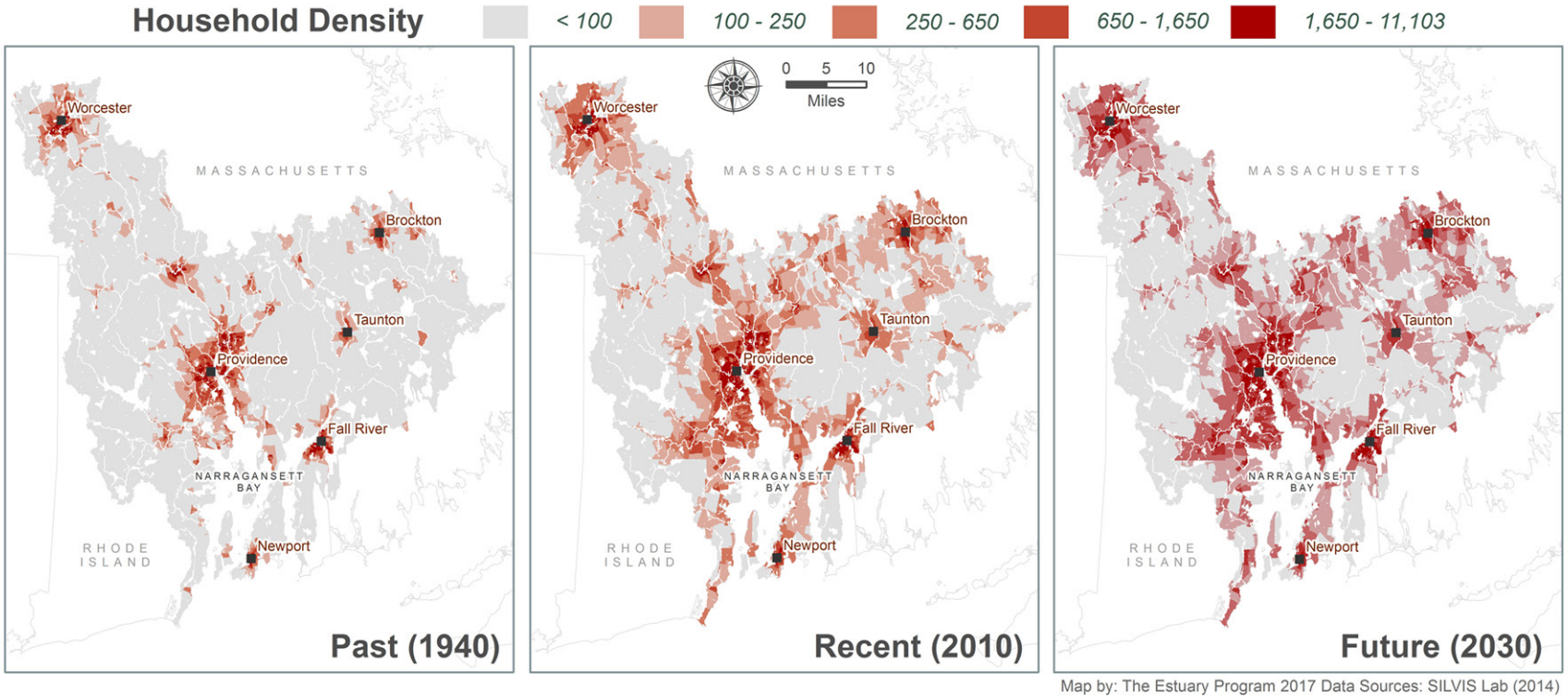


Figure 4. Changes in household density across the Narragansett Bay Watershed from 1940 to 2010, and as projected for 2030 (SILVIS Lab 2014).

2000, as well as projections through 2030. A preliminary analysis showed that the average number of household units per square kilometer increased 60 percent between 1940 and 2000, and it is expected to increase approximately twelve percent from 2000 to 2030 (Table 13).

Figure 4 illustrates patterns of development by comparing estimated household densities in 1940, 2010, and 2030. These maps depict household density data from the SILVIS Lab, clipped to the boundaries of the Watershed, which can be used as a proxy for population changes driving land development. Consistent with the results from dasymetric analysis and the heat map (Figure 2), these data showed population growth around the historical urban corridors, expanding toward the suburban and more rural areas of the Watershed, where land use has changed through the loss of forest lands to urbanization (see "Land Use" chapter).

Projections through 2030 for growth in household density illustrate continued growth, or sprawl, in these adjacent areas (Figure 4). Additional work is needed to further explore and analyze these data at different geographical scales, including at the municipal level, to identify more precisely the expected population and land development changes.

Unless effective land management and regional planning practices are implemented as populated areas expand, these trends will likely have a direct impact on the condition of Narragansett Bay and its Watershed. Impacts include changing physical conditions such as decreases in water clarity, dissolved oxygen concentrations, and water quality, which in turn affect biological parameters such as chlorophyll, as well as public health issues through beach and shellfishing closures. As future Census Bureau reports become available, the geospatial analyses conducted here should be updated and trend monitoring continued to inform land use and planning decisions at local and watershed scales.

Data Gaps and Research Needs

There are no critical data gaps or research needs, assuming that detailed US Census Bureau data continue to be collected each decade and that funds are made available to conduct geospatial analyses. This research is needed to provide a more comprehensive understanding of trends and to provide context for other indicators of stressors and conditions in Narragansett Bay and its Watershed. More robust data analyses should be performed to interrelate total population changes with developed area per capita and housing density, two factors that are linked to the effects of population on other landscape and chemical indicators.

Table 13. Estimated and projected household density in the Narragansett Bay Watershed (SILVIS Lab 2014).

Number of household units per square km[1] (per acre)			
1940	**1970**	**2000**	**2030**
570	754	914	1,030
(2.3)	(3.1)	(3.7)	(4.2)

[1] Original units from data source

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This chapter was written by Eivy Monroy (Watershed and GIS Specialist with the Narragansett Bay Estuary Program), Anne Kuhn (US Environmental Protection Agency, Office of Research and Development, Atlantic Ecology Division, Narragansett, RI), and Michael Charpentier (GIS Analyst with SRA International, Inc., A CSRA Company). Assistance in the development of this chapter was provided by Peter August (University of Rhode Island, Department of Natural Resources Science) and Paul Jordan (Rhode Island Department of Environmental Management). Cartography by Julia Twichell, GIS Analyst with the Estuary Program. The information in this document has been subjected to Environmental Protection Agency review and approved for publication. Mention of trade names or commercial products does not constitute endorsement or recommendation for use.

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State of Narragansett Bay and Its Watershed
2017 Technical Report

Landscape Stressor Indicators

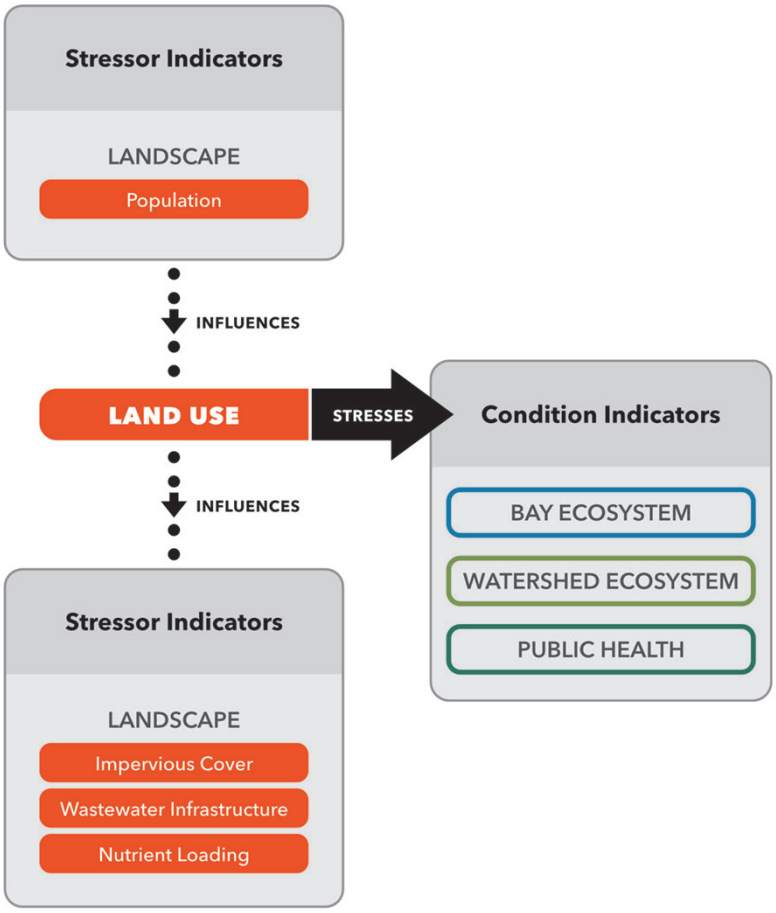
CHAPTER 5: LAND USE

Please use the following citation: Narragansett Bay Estuary Program. 2017. State of Narragansett Bay and Its Watershed (Chapter 5, Land Use, pages 108-134). Technical Report. Providence, RI.

NARRAGANSETT BAY ESTUARY PROGRAM

Photo: Aerial view of Bristol, RI with Fall River and Somerset, MA in the distance (Ayla Fox)

Overview



BACKGROUND

- Changes in land use in the Narragansett Bay Watershed, especially the conversion of natural lands to urban areas, affects hydrologic functions, alters the delivery of nutrients to rivers and the Bay, affects terrestrial, aquatic and estuarine wildlife and habitat conditions, and contributes to increased pathogens in recreational and shellfishing waters. Land use changes that reduce natural lands are an indicator of habitat fragmentation, diminishing habitat value as well as water quality and quantity.

KEY FINDINGS

- ***Status:*** As of 2011, coastal subwatersheds (HUC12) were the most urbanized in the Watershed, ranging from 65 to 85 percent urban lands. Conversely, 70 percent of land in less-developed headwater subwatersheds was classified as forest.

- ***Trends:*** From 2001 to 2011, forest lands decreased in the Bay's Watershed by four percent, and urban lands increased by 8.5 percent, encroaching into rural areas. Watersheds (HUC10) of the Taunton River Basin experienced dramatic changes. Forest lands of the Upper-Taunton River and Ten Mile River decreased by nine percent, and the Middle-Taunton River had an 18 percent increase in urban lands as forest lands were lost to new developed areas.

Introduction

Land development and some types of land uses in the Narragansett Bay Watershed are stressors on water quality, water quantity, freshwater and estuarine habitats, and human health. The conversion of natural lands such as forests and wetlands to human-dominated uses can exert considerable influence on runoff quality and quantity, and contribute to increases in water pollution (Tu and Xia 2006).

Sources of water pollution are generally grouped into two categories: point sources and non-point sources. Over the past few decades, point source pollution, including from domestic and industrial wastewater discharges (see "Wastewater Infrastructure" chapter), has been greatly reduced through management actions and changes in industrial uses, as local economies shift from manufacturing-based sectors to service-based economies (USEPA 2008). Conversion of a natural land cover such as forests to an urban or developed land use can significantly increase non-point source stressors as well as the flow patterns of streams after rain events. Non-point source inputs are influenced by land use alteration (impervious land, agriculture, golf courses, residential and commercial development), riparian buffer degradation, sediment from poorly managed construction sites, stormwater runoff, road salt, atmospheric deposition of nitrogen, failing septic systems, and other factors. Human population growth is a fundamental driving force in land conversion. As the population grows, the infrastructure to support homes, transportation, and commerce increases (Meyer and Turner 1992, August et al. 2002).

Several studies have provided comprehensive historical analyses of watershed stressors and responses for Narragansett Bay from 1850 to 2000 (Nixon 1997, Hamburg et al. 2008, Vadeboncoeur et al. 2010, Pastore 2011). Others have assessed trends in historical land use and changes in impervious surface cover at the state level in Rhode Island and Massachusetts for large portions of the Narragansett Bay Watershed but not the entire Watershed (Novak and Wang 2004, Rhode Island Statewide Planning Program 2006, Stone 2007, Zhou and Wang 2007, Tu et al. 2007, Blumstein and Thompson 2015). Those studies highlighted the conversion of forest and agricultural lands to residential, commercial, and industrial developed lands as significant trends within the Watershed and the surrounding parts of Rhode Island and Massachusetts. The most recent assessment of land use change by the Massachusetts Audubon Society reported that approximately 38,000 acres of forest or other undeveloped land were converted to development in Massachusetts between 2005 and 2013 (Mass Audubon 2014).

The conversion rate of natural land cover to developed land has outpaced the population growth rate in this region (see "Population" chapter) over the last few decades. In addition, recent changes in land use have not been distributed uniformly across the Narragansett Bay Watershed; they have varied temporally and spatially as the population has moved from the urbanized centers to the more suburban and rural parts of the Watershed. This chapter presents an analysis of land use change in the Watershed, focusing on the changes in area (acreage) of forest lands and urban lands in the decade from 2001 to 2011. The chapter also discusses historical changes in land use and the rates of change since the industrial revolution.

Methods

The methods for analyzing land use as an indicator of environmental conditions in the Narragansett Bay Watershed were developed by the US Environmental Protection Agency (EPA) Office of Research and Development (ORD), National Health and Environmental Effects Research Laboratory, Atlantic Ecology Division in collaboration with the Narragansett Bay Estuary Program and other partners. A work group of

GIS specialists examined and selected the data and data processing methods used for this indicator.

NATIONAL LAND COVER DATABASE (NLCD)

The National Land Cover Database (NLCD) was used to classify land use at a resolution of 30 meters (Homer et al. 2015). For this analysis, the NLCD's sixteen classes of land use were aggregated into seven land use categories based on the seven classes of the Anderson Level I classification scheme (Anderson et al. 1976) (see Table 14 in Extended Methods section of this chapter). Land use data were analyzed for the Narragansett Bay Watershed using an array of geospatial tools (Esri 2016, ArcGIS Desktop platform).

Using NLCD data from 2011, the Estuary Program calculated the status of the seven land use categories, including the total acreage in each category and the percentage of the Narragansett Bay Watershed in each category. This chapter focuses on two of the land use classes: urban lands and forest lands. Data for 2001, 2006, and 2011 were analyzed at three spatial scales: the Narragansett Bay Watershed, watersheds (HUC10), and subwatersheds (HUC12) (see the Appendix for definitions, lists, and maps).

The Estuary Program focused on two of the seven land use categories (urban lands and forest lands) for three reasons: (1) these two categories cover the majority (74 percent) of the Watershed, (2) an increase in urban and a decrease in forest lands can indicate that the Watershed is changing to a more disturbed condition, and (3) preliminary analysis of other land use types revealed that changes were not as large compared to urban and forest lands.

Land use data from NLCD 2001 (2011 Edition), 2006 (2011 Edition), and 2011 were utilized for the change analysis. NLCD land use datasets for years prior to 2001 are incompatible for comparison with the more recent datasets. The 2001, 2006, and 2011 NLDC datasets all have a sixteen-class land cover classification scheme and are based primarily on a decision-tree classification of circa 2001, 2006, and 2011 Landsat satellite data, respectively. The Multi-Resolution Land Characteristics Consortium cautions against using NLCD data in watersheds on a scale of less than tens of square kilometers (USGS 2012). However, a multiple-extents accuracy assessment suggested that NLCD data may be accurate for spatial extents as small as ten square kilometers, particularly for predominant land use classes or those with unique spectral signatures (Hollister et al. 2004). The smallest HUC12 subwatershed within the Narragansett Bay Watershed is 21 square kilometers, and thus the NLCD data were used with confidence at HUC10 watershed and HUC12 subwatershed scales.

Change analyses were based on total gross change in acreage and net percent change within each geographic scale. Total gross change represents the change in acreage by category, and net percent change is the change of area in percentage between two specific dates (Loveland et al. 2002, Sohl et al. 2004). Gross change and net percent change were calculated as follows:

$$Gross\ change = Acreage\ in\ 2011 - Acreage\ in\ 2001$$

$$(Net)Percent\ change = \frac{Acreage\ of\ 2011 - Acreage\ of\ 2001}{Acreage\ of\ 2001} x100$$

STATE-LEVEL LAND USE DATA

In addition, data with finer spatial resolution were available at the state level in both Massachusetts and Rhode Island (Massachusetts: 1.0-acre resolution; Rhode Island: 0.5-acre resolution), and those datasets offered the advantages of increased spatial resolution and interpretation of land use classification. However, using those data required matching or a "crosswalk" of land use classifications across state boundaries (Table 1; Tables 14, 15, and 16 in Extended Methods section of this chapter), both spatially and temporally. Because land use data are not consistent methodologically across states and years within each state, and it is unknown when the states would update their land use data, the Estuary Program decided it was most appropriate to use the NLCD data for tracking long-term trends across the Watershed. Results using the state land use data crosswalk are presented in this chapter only to compare between state and national land use datasets (Table 1; Tables 14, 15, and 16 in Extended Methods section of this chapter). For the state temporal crosswalk, the most recent data for Massachusetts were from 2005, while Rhode Island's most recent data were from 2011, making it necessary to match data from earlier years (2003–2004) for Rhode Island.

The Estuary Program compared land use categories from NLCD 2011 and bi-state crosswalk data (Massachusetts 2005 and Rhode Island 2003–2004) within the Watershed. There is close agreement between these two datasets, which supports the use of the National Land Cover Database data for the status and change analyses, despite the difference in geographical resolution, temporal scales, and methodology.

Table 1. Types of state land use classes compared to Anderson Level I classification scheme.

Anderson Level I Categories	Bi-state Crosswalk of Land Use Types[1]
Urban Land	high, medium, low residential; commercial, industrial, transportation, cemeteries, wastewater treatment facilities, waste disposal, landfills, commercial water-based facilities, airports, railroads, urban parks, zoos, golf courses
Agricultural Land	pasture, hay fields, orchards, concentrated animal feeding operations, cropland, nurseries
Forest Land	deciduous, evergreen, mixed
Brushland	shrub and brush areas undergoing reforestation
Wetland[2]	*Rhode Island:* non-forest wetlands; *Massachusetts:* forest and non-forest wetlands, and saltwater wetlands
Barren	beaches, sandy areas other than beaches, rock outcrops, gravel, mining pits
Water	reservoirs, lakes and ponds

[1] This is an overarching summary of Land Use classes by each state as listed in Tables 15 and 16 in Extended Methods section of this chapter.

[2] Differences between NLCD and across states for classifying wetlands is noteworthy since this chapter focuses on changes of forest lands and not wetlands. For instance, NLCD classifies forested wetland as "wetland", while Rhode Island classifies forested wetland as "forest land" for the 2004-2005 land use classification (RIGIS 2007).

Table 2. Land use in the Narragansett Bay Watershed[1] based on NLCD (2011) and state (2003–2005) data.

Land Use Category	National Land Cover Dataset 2011[2]		Crosswalk State Data 2003–2005	
	Acres	Percent	Acres	Percent
Urban or Built-up[3]	379,804	34.7	413,455	37.9
Forest Land[3]	424,642	38.8	429,234	39.3
Agricultural Land	68,358	6.2	51,095	4.7
Brushland	10,711	1.0	6,895	0.6
Barren	8,713	0.8	11,008	1.0
Wetland	164,895	15.1	135,115	12.4
Water	36,668	3.4	44,164	4.0

[1] Total area of Narragansett Bay Watershed is 1,091,112 acres.

[2] Data source used for change analysis of urban and forest lands.

[3] Land use categories used for change analysis.

Urban lands and forest lands constituted 35 percent and 39 percent of the Narragansett Bay Watershed, respectively, based on the most recent 2011 National Land Cover Database (Table 2). These results are very similar to the results based on the bi-state crosswalk data for Rhode Island (2003-2004) and Massachusetts (2005), which showed urban lands as 38 percent and forest land as 39 percent of the Watershed. The remaining 27 percent of the Watershed (based on NLCD) were a combination of land uses including agriculture, brushland, barren land, wetlands, and water (Table 2).

HISTORICAL TRENDS

Based on previous research by Vadeboncoeur and colleagues (2010), the Narragansett Bay Estuary Program calculated historical changes in urban and forest land that encompassed eras of industrialization and suburbanization.

Because Vadeboncoeur and colleagues (2010) calculated and provided data on historical land cover by percent of urban and forest and inland water[1] by subwatershed[2], as opposed to by total area, the Estuary Program analyzed each of these land cover types as percent point change (see Tables 17 and 18 in Extended Methods section of this chapter).

COMPARING METHODS ACROSS INDICATORS: LAND USE (FOREST LANDS) AND OPEN SPACE

In the "Open Space" chapter of this report, the Estuary Program developed a methodology, differing from the analysis used in this chapter, to analyze open space lands classified as protected natural lands and unprotected natural lands. The open space chapter did not use the NLCD data, and instead used the Conservation Assessment and Prioritization System (CAPS), which ranks natural lands, including forest and wetlands among other natural features in the landscape, from lowest to highest ecological integrity (Index of Ecological Integrity 0.01 to 1). Although the results using CAPS were similar to those based on NLCD–differing by one percent in total acreage in the Watershed–the results presented in this chapter and those in the "Open Space" chapter should be interpreted separately due to the methodological differences (Table 3).

[1] Inland waters accounted for greater than five percent of total cover in all subwatersheds (Vadeboncoeur et al. 2010).

[2] These subwatersheds are not comparable with any of the Estuary Program's geographical scales. Vadeboncoeur and colleagues (2010) defined the subwatersheds for their study by the municipal boundaries of the drainage areas.

Table 3. Comparison of forest lands data (NLCD) from this chapter with natural lands data from the "Open Space" chapter.

Estuary Program Method	Area (Acres)	Percent
Total NLCD (Forest Lands and Wetlands)	589,470	54
Forest Lands (NLCD 2011)(1)	424,642	39
Wetlands (NLCD 2011)	164,828	15
Index of Ecological Integrity (CAPS 0.01-1)(2)	600,140	55

- (1) Data used to analyze extent of forest land in this chapter.
- (2) Conservation Assessment and Prioritization System (CAPS) data used to analyze natural lands in "Open Space" chapter.

Status and Trends

As of 2011, the Narragansett Bay Watershed still had more forest lands than urban lands (Table 2).

STATUS OF URBAN LANDS

The urban land area of the Narragansett Bay Watershed totaled 379,804 acres in 2011, representing 35 percent of the Watershed (Figure 1; Table 2). In HUC10 watersheds, urban land ranged from a low of 22 percent (25,300 acres) in the Middle Taunton River watershed to a high of 55 percent (19,667 acres) in the Ten Mile River watershed (Table 8).

Of the 52 HUC12 subwatersheds, fifteen subwatersheds had more than 50 percent of their area classified as urban, while only four subwatersheds had ten percent or less of their land classified as urban (Table 4). All five of the HUC12 subwatersheds with the highest percentages of urban land were adjacent to and drain directly to Narragansett Bay (Seekonk River-Providence River, Greenwich Bay, Upper Narragansett Bay, Pawtuxet River, and the Aquidneck Island-Frontal Atlantic Ocean), and they correspond closely with the subwatersheds with the lowest percentages of forest land (Table 4).

STATUS OF FOREST LANDS

There were 424,642 acres of forest land in the Narragansett Bay Watershed in 2011, constituting 39 percent of the Watershed (Figure 2; Table 2). The HUC10 watershed with the lowest percentage of forest land was the Ten Mile River, which had only 24 percent (8,461 acres) forest. The watershed with the highest percentage was the Lower Blackstone River with 55 percent (94,731 acres) (Table 9).

Of the 52 HUC12 subwatersheds, twelve had more than 50 percent forest. The five subwatersheds with the highest percentages of forest land were the Barden Reservoir-Ponaganset River, Clear River,

Table 4. Subwatersheds (HUC12) in the Narragansett Bay Watershed with the highest and lowest percentages of urban lands.

Subwatershed (HUC12) Name	Urban Lands (NLCD 2011)	
	Acreage	Percent
Urban lands ≥50 percent of the subwatershed:		
Seekonk River-Providence River	5,978	85.5
Greenwich Bay	4,210	75.9
Upper Narragansett Bay	3,687	75.7
Pawtuxet River	2,751	75.7
Aquidneck Island-Frontal Atlantic Ocean	1,544	71.3
Moshassuck River	6,419	65.8
Tatnuck Brook-Blackstone River	1,706	63.3
South Branch Pawtuxet River	3,037	62.2
Lower East Passage	1,594	61.8
Matfield River	2,975	60.3
Barrington River-Warren River	14,182	59.3
Quinsigamond River	5,585	57.0
Pocassett River	10,347	56.9
Ten Mile River	2,110	55.2
Upper East Passage	6,236	50.8
Urban lands ≤10 percent of the subwatershed:		
Clear River	9,008	10.2
Big River	3,025	9.4
Scituate Reservoir	4,423	8.7
Barden Reservoir-Ponaganset River	14,459	7.3

(1) Total area and maps of the subwatershed (HUC12) are provided in the Appendix.

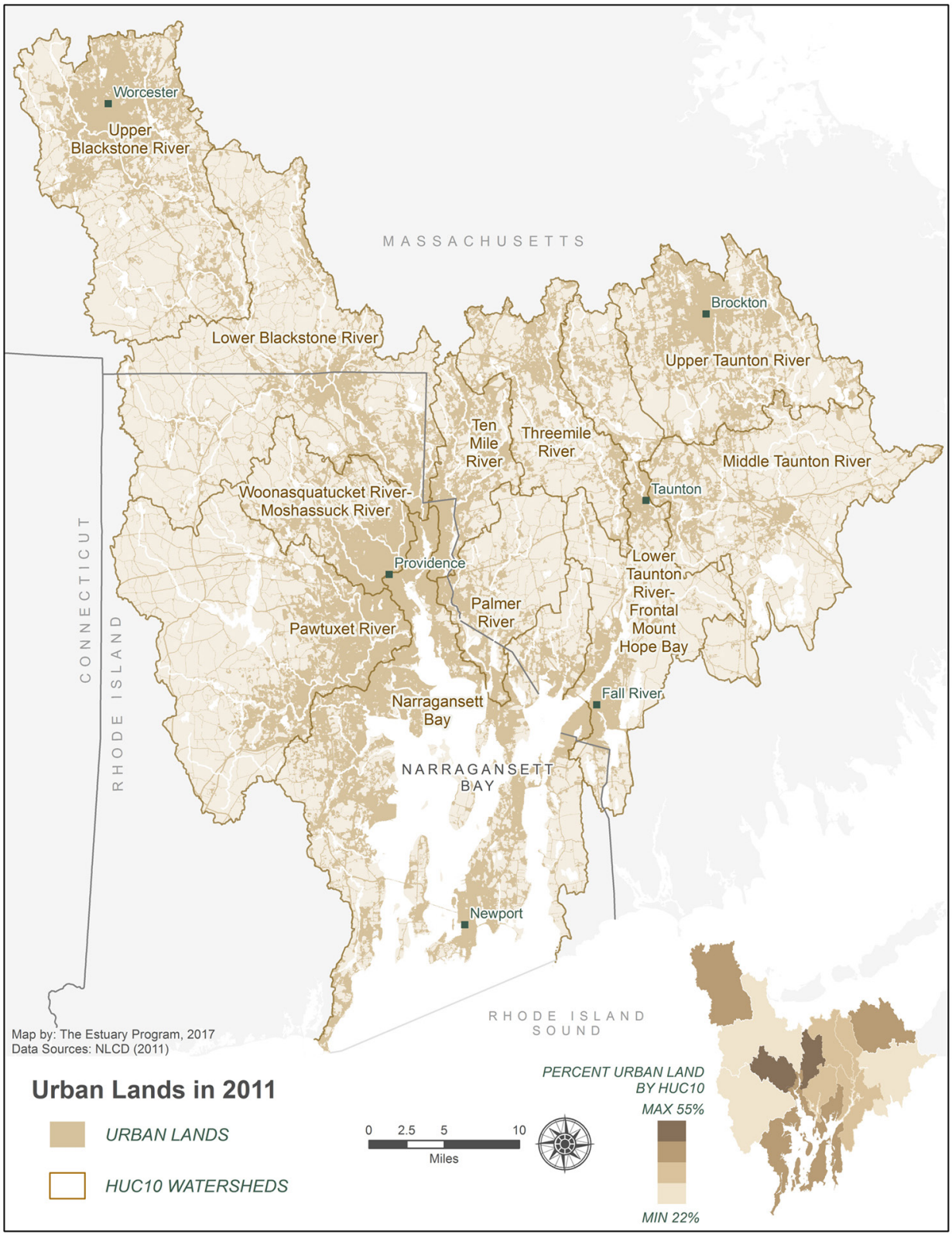


Figure 1. Urban lands (NLCD 2011) in the Narragansett Bay Watershed. Inset map: Percent of urban lands for each watershed (HUC10).



Figure 2. Forest lands (NLCD 2011) in the Narragansett Bay Watershed. Inset map: Percent of forest lands for each watershed (HUC10).

Table 5. Subwatersheds (HUC12) in the Narragansett Bay Watershed with the highest and lowest percentages of forest lands.

Subwatershed (HUC12)[1] Name	Forest Lands (NLCD 2011) Acreage	Percent
Forest lands ≥50 percent of the subwatershed:		
Barden Reservoir-Ponaganset River	7,956	71.9
Clear River	380	70.8
Big River	14,195	70.3
Chepachet River	12,014	66.4
Headwaters South Branch Pawtuxet River	15,142	66.1
Moswansicut Pond-Huntinghouse Brook	1,575	65.9
Scituate Reservoir	12,760	63.2
West River	10,218	63.1
Mumford River	9,094	60.4
Branch River	20,558	60.0
Assonet River	11,536	54.9
Emerson Brook-Blackstone River	11,887	54.2
Forest lands ≤10 percent of the subwatershed:		
Upper Narragansett Bay	14,970	8.5
Aquidneck Island-Frontal Atlantic Ocean	11,005	6.4
Seekonk River-Providence River	12,699	5.6

[1] Total area and maps of the subwatershed (HUC12) are provided the Appendix.

Table 6. Total area (acres) and percentage of forest and urban lands in the Narragansett Bay Watershed[1] in 2001, 2006, and 2011 (NLCD).

Land Use Category	2001		2006		2011	
	Acreage	Percent	Acreage	Percent	Acreage	Percent
Forest	443,800	40.6	430,793	39.4	424,642	38.8
Urban	350,369	32.0	371,836	34.0	379,804	34.7

[1] Total area of Narragansett Bay Watershed is 1,091,112 acres.

Table 7. Total gross change (acres) and percent change of forest and urban lands in the Narragansett Bay Watershed over five-year periods—2001 to 2006 and 2006 to 2011—and the full ten-year period of 2001 to 2011 (NLCD).

Land Use Category	2001 to 2006		2006 to 2011		2001 to 2011	
	Acreage	Percent Change	Acreage	Percent Change	Acreage	Percent Change
Forest	-13,007	-2.9	-6,151	-1.4	-19,158	-4.3
Urban	21,467	6.2	7,968	2.1	29,435	8.5

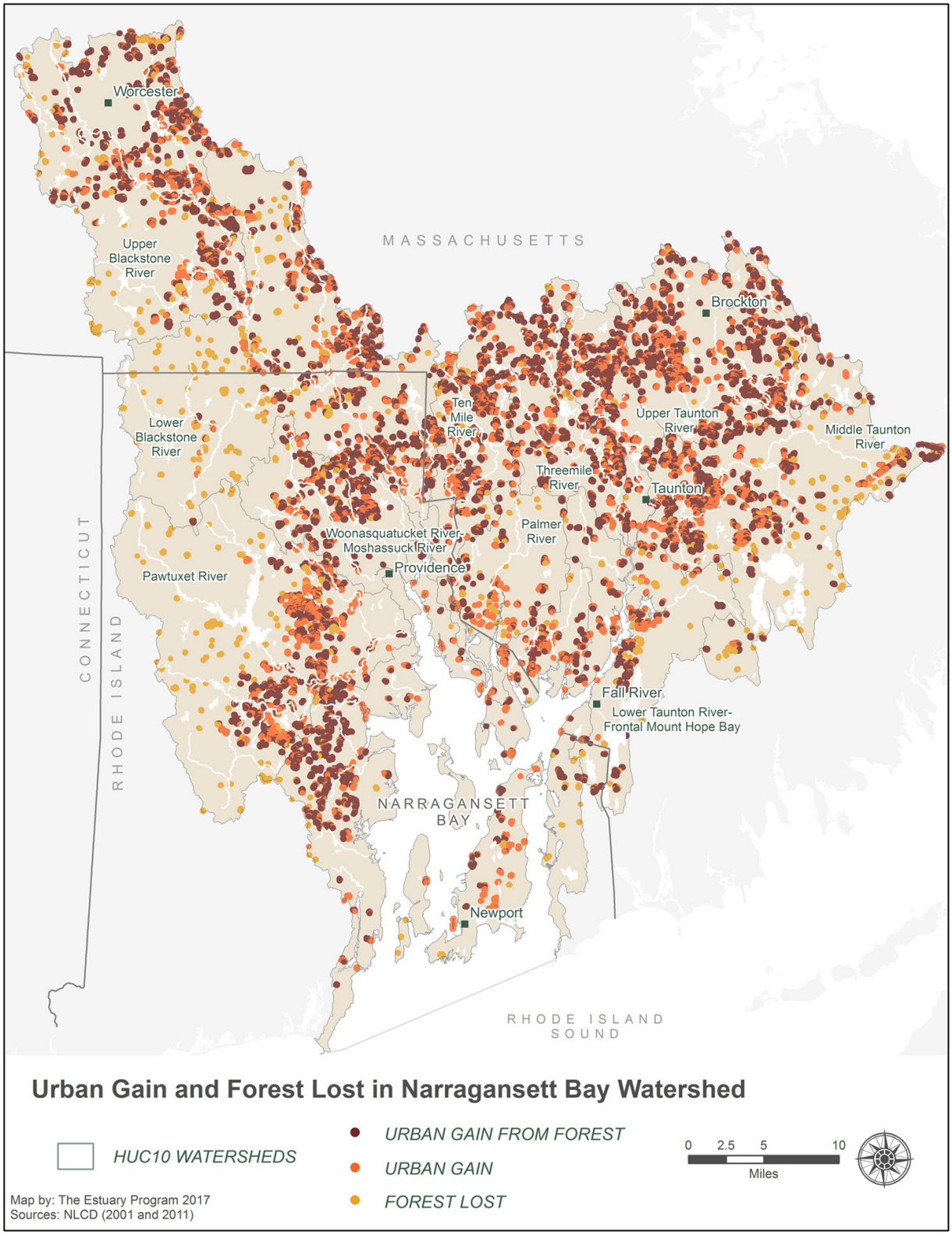


Figure 3. Changes from 2001 to 2011 in forest and urban lands.

Table 8. Total area and percent of urban lands in 2001, 2006, and 2011 in Narragansett Bay's HUC10 watersheds.

HUC10 watersheds[1]	2001		2006		2011	
Name	Percent	Acreage	Percent	Acreage	Percent	Acreage
Lower Blackstone River	21.2	36,508	22.8	39,224	23.5	40,463
Lower Taunton River-Frontal Mount Hope Bay	27.3	27,586	29.0	29,258	29.9	30,177
Middle Taunton River	18.7	21,472	21.3	24,441	22.0	25,300
Narragansett Bay	48.6	76,481	50.1	78,403	50.4	79,051
Palmer River	27.9	12,161	29.1	12,658	29.4	12,807
Pawtuxet River	25.6	37,971	27.2	40,396	27.4	40,662
Ten Mile River	51.0	18,160	54.5	19,410	55.2	19,667
Threemile River	31.7	17,283	34.6	18,862	36.4	19,885
Upper Blackstone River	34.3	45,102	36.4	47,805	37.3	49,078
Upper Taunton River	39.7	34,945	42.7	37,592	44.0	38,693
Woonasquatucket River-Moshassuck River	48.1	22,841	50.4	23,938	50.9	24,174

[1] Total area and maps of the watersheds (HUC10) are provided in the Appendix.

Table 9. Total area and percent of forest lands in 2001, 2006, and 2011 in Narragansett Bay's HUC10 watersheds.

HUC10 watersheds	2001		2006		2011	
Name	Percent	Acreage	Percent	Acreage	Percent	Acreage
Lower Blackstone River	56.6	97,469	55.6	95,750	55.0	94,731
Lower Taunton River-Frontal Mount Hope Bay	38.6	39,026	37.7	38,033	37.0	37,368
Middle Taunton River	38.8	44,603	37.2	42,769	36.6	42,108
Narragansett Bay	25.0	39,395	24.3	38,115	24.1	37,736
Palmer River	31.1	13,534	30.6	13,317	30.4	13,242
Pawtuxet River	52.7	78,202	51.9	76,989	51.5	76,482
Ten Mile River	26.2	9,332	24.3	8,651	23.7	8,461
Threemile River	38.0	20,746	36.2	19,741	34.9	19,070
Upper Blackstone River	45.2	59,415	43.9	57,660	43.1	56,617
Upper Taunton River	28.4	24,968	26.5	23,330	25.7	22,620
Woonasquatucket River-Moshassuck River	36.4	17,291	35.0	16,612	34.5	16,378

[1] Total area and maps of the watersheds (HUC10) are provided in the Appendix.

Table 10. Percent change and total gross change (acres) of urban lands from 2001 through 2011 in Narragansett Bay HUC10 watersheds.

HUC10 watersheds Name	2001–2006		2006–2011		2001–2011	
	Percent	Acreage	Percent	Acreage	Percent	Acreage
Middle Taunton River	13.8	2,968	3.5	859	17.8	3,826
Threemile River	9.1	1,578	5.4	1,023	15.1	2,601
Lower Blackstone River	7.4	2,714	3.2	1,239	10.8	3,953
Upper Taunton River	7.6	2,645	2.9	1,101	10.7	3,746
Upper Taunton River	7.6	2,645	2.9	1,101	10.7	3,746
Lower Taunton River-Frontal Mount Hope Bay	6.1	1,671	3.1	919	9.4	2,590
Upper Blackstone River	6.0	2,703	2.7	1,272	8.8	3,975
Upper Taunton River	7.6	2,645	2.9	1,101	10.7	3,746
Ten Mile River	6.9	1,250	1.3	257	8.3	1,507
Pawtuxet River	6.4	2,424	0.7	266	7.1	2,690
Woonasquatucket River-Moshassuck River	4.8	1,096	0.9	235	5.8	1,332

(1) Total area and maps of the watersheds (HUC10) are provided in the Appendix.

Table 11. Percent change and total gross change (acres) of forest lands from 2001 through 2011 in Narragansett Bay's HUC10 watersheds. Sorted from highest to lowest percent loss of forest lands.

HUC10 watersheds Name	Change 2001–2006		2006–2011		2001–2011	
	Percent	Acreage	Percent	Acreage	Percent	Acreage
Upper Taunton River	-7	-1,637	-3	-710	-9	-2,347
Ten Mile River	-7	-681	-2	-190	-9	-871
Threemile River	-5	-1,004	-3	-671	-8	-1,675
Middle Taunton River	-4	-1,834	-2	-661	-6	-2,494
Woonasquatucket River-Moshassuck River	-4	-678	-1	-234	-5	-912
Upper Blackstone River	-3	-1,754	-2	-1,043	-5	-2,797
Lower Taunton River-Frontal Mount Hope Bay	-3	-992	-2	-664	-4	-1,657
Narragansett Bay	-3	-1,280	-1	-379	-4	-1,659
Lower Blackstone River	-2	-1,718	-1	-1,018	-3	-2,736
Pawtuxet River	-2	-1,212	-1	-507	-2	-1,719
Palmer River	-2	-217	-1	-74	-2	-291

(1) Total area and maps of the watersheds (HUC10) are provided in the Appendix.

Big River, Chepachet River, and Headwaters South Branch Pawtuxet River (Table 5). The three HUC12 subwatersheds with the lowest percentages of forest land were the Seekonk River-Providence River, Aquidneck Island-Frontal Atlantic Ocean, and Upper Narragansett Bay (Table 5). The complete results of Tables 4 and 5 for all the HUC12 subwatersheds are available upon request.

TRENDS – CHANGES IN URBAN AND FOREST LANDS

In the Narragansett Bay Watershed, the amount of land classified as urban increased from 350,369 acres in 2001 to 379,804 acres in 2011. The increase of 29,435 acres represented a change of 8.5 percent (Tables 6 and 7). During the same time period, the Watershed lost 19,158 acres of forest land, a decline of 4.3 percent (Table 7). Figure 3 shows where these changes in forest lands and urban lands occurred. For both urban and forest lands, the rates of change were greatest from 2001 to 2006 (Table 7), when forest land declined by three percent and urban land increased by six percent.

All eleven HUC10 watersheds experienced increases in urban land and concomitant decreases in forest land from 2001 to 2011 (Tables 8 through 11). The Lower Blackstone River watershed had the largest percentage of forest lands, while the Ten Mile River watershed had the largest percentage of urban lands–over 55 percent–in each of the three years (2001, 2006, 2011) (Tables 8 and 9).

The HUC10 watersheds experiencing the largest net percentage increases of urban land were the Middle Taunton River and Threemile River watersheds (Table 10; Figure 3), while the percentage losses of forest land were largest in the Upper Taunton River, Ten Mile River, and Threemile River watersheds (Table 11; Figure 3).

At the finest scale of HUC12 subwatersheds, all but one (Barden River-Ponaganset River) of the 52 subwatersheds had increases in the amount of urban land.

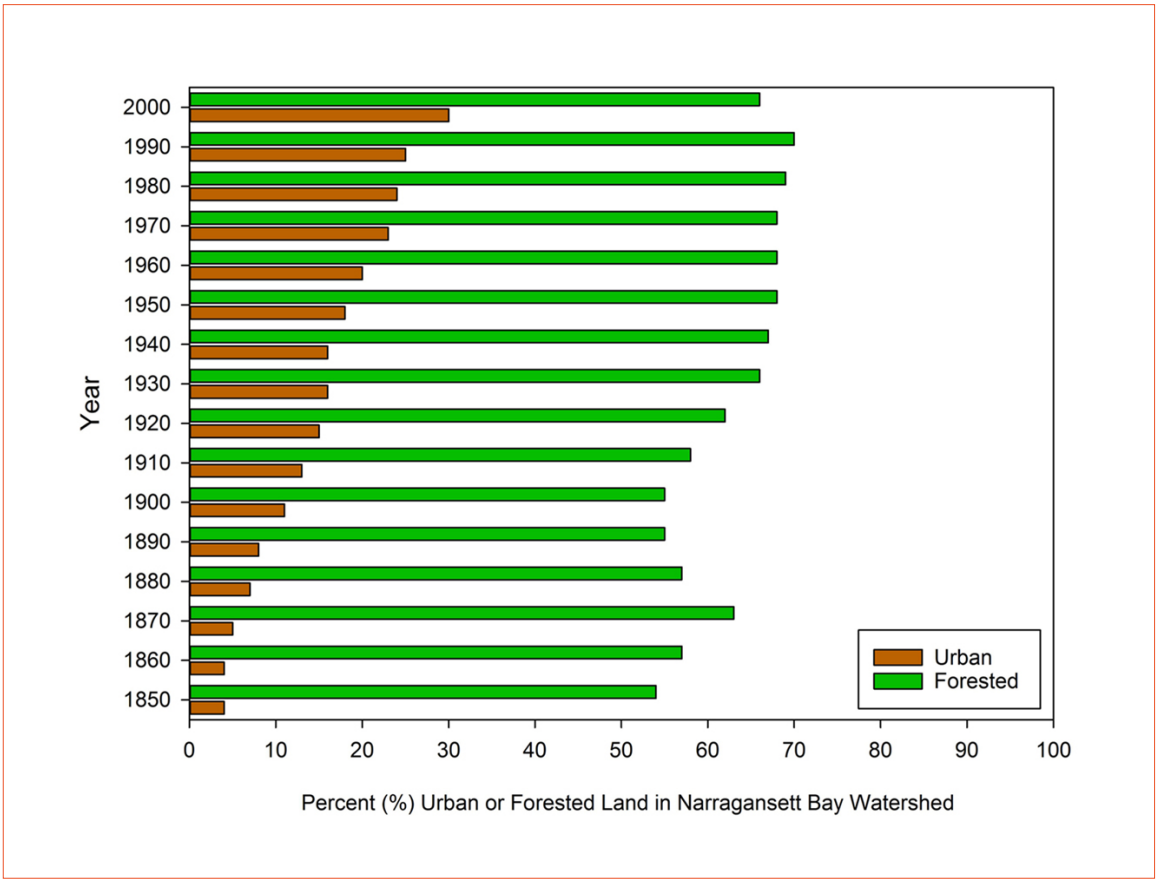


Figure 4. Historical changes in percentage of Narragansett Bay Watershed classified as forest (green) or urban (brown). Based on Vadeboncoeur and colleagues (2010).

All subwatersheds had decreases in forest land from 2001 to 2011. The subwatersheds with the largest gains in urban lands also experienced the largest losses in forest lands. Of the top 25 subwatersheds ranked by the largest increase in acres of urban land, 21 of those subwatersheds were also ranked for the largest loss of forest land. These results are available upon request.

HISTORICAL TRENDS

Between 1850 and 2000, the percentage of urban lands in the Narragansett Bay Watershed increased eightfold, doubling every fifty years on average. The greatest increase occurred between 1950 and 2000, when urban lands increased from 18 to 30 percent of the entire Watershed (Figure 4). The Blackstone River (above Millville) and the Upper Bay subwatersheds had threefold increases in percentage of urban lands. However, the small watersheds had the greatest percentage increase by a factor of four (Table 12). Some of the recent changes in the Taunton River Basin seem to have started in the 1950s, as the percentage of urban land in the Taunton River (below Taunton) subwatershed increased fourfold from 1950 through 2000. In the small watersheds, the amount of urban land increased by a factor of seventeen from two percent in 1850 to 33 percent in 2000 (Table 12).

The rates at which forest lands, including inland waters, changed over time were not more than twofold across time periods and across the Watershed or subwatersheds. However, Narragansett Bay Watershed lost most of the forest in the Taunton River subwatersheds between 1950 and 2000, and during the same period these subwatersheds had the greatest percent increase of urban lands (Tables 12 and 13). The increase of forest lands could be attributed to losses in agricultural lands during the industrial revolution, while forest was recovering. Later, during the suburbanization era, forest lands in the Pawtuxet River subwatershed increased by a factor of two; the Scituate Reservoir is an important

Table 12. Percent[1] change[2] and factor of percentage change[3] in urban lands in Narragansett Bay Watershed and subwatersheds as defined by Vadeboncoeur and colleagues (2010).

Subwatersheds	Industrialization 1850–1900		Industrialization 1900–1950		Suburbanization 1950–2000		1850–2000	
	%	*Factor*	%	*Factor*	%	*Factor*	%	*Factor*
Blackstone R. above Millville	9	*3*	9	*2*	6	*1*	24	*9*
Blackstone R. Millville to Manville	3	*2*	4	*2*	1	*1*	8	*5*
Pawtuxet R. above Pettaconsett	1	*2*	1	*2*	5	*3*	7	*8*
Taunton R. above Bridgewater	6	*2*	4	*1*	17	*2*	27	*10*
Taunton R. Bridgewater to Taunton	9	*2*	5	*1*	19	*2*	33	*7*
Taunton R. below Taunton	1	*1*	5	*2*	25	*4*	31	*11*
Small watersheds	7	*4*	10	*2*	14	*2*	31	*17*
Upper Bay	24	*3*	19	*2*	6	*1*	49	*7*
Lower Bay	5	*2*	11	*2*	15	*2*	31	*7*
Total Narragansett Bay Watershed	**7**	**2**	**7**	**2**	**12**	**2**	**26**	**8**

[1] Percent of urban lands for each decade by subwatershed from Vadeboncoeur et al. (2010) are presented in the Extended Methods of this chapter (Table 17).
[2] Result of subtracting the percent of urban lands by Watershed or subwatersheds, within the time period. For example, between 1850 (3 percent) and 1900 (12 percent), urban lands had a nine-percentage point change.
[3] Result of dividing the percent of urban land of the most recent year by the percent of the year to compare by Watershed or subwatersheds. For example, between 1850 (3 percent) and 1900 (9 percent), percent of urban lands increased by a factor of three.

drinking water source in Rhode Island and protecting its surrounding lands has been a priority.

There were substantial methodological differences between the analysis of NLCD by the Estuary Program and partners, and the analysis by Vadeboncoeur and colleagues (2010). The differences included data, spatial and temporal resolution, and the definition of boundaries for the Watershed and subwatersheds. While the percentages of urban lands in the Watershed for 2000 were similar between the two studies (Table 6; Figure 2), the percent of forest lands were not in agreement. This can be attributed mainly to the fact that Vadeboncoeur and colleagues (2010) included inland water bodies and perhaps wetlands, whereas the Estuary Program's analysis included only forest.

Discussion

Land use in the Narragansett Bay Watershed is subject to conversion, and these changes influence the Watershed's hydrologic functions. Changes of natural habitat such as wetlands and forests to urban lands have impacted how water is delivered to rivers and lakes, to groundwater, and ultimately to the Bay. Measuring the total area of land use change over time highlights the conversion of forest and other natural lands to residential, commercial, and industrial developed lands (Figure 3). It is important to highlight that acreage of forest lands reported in this chapter does not account for wetlands. However, no substantial changes in wetland extent between 2001 and 2011 were detected in preliminary analysis by the Estuary Program and partners. Wetlands are protected under federal, state, and local laws, which may explain the lack of detectable changes.

Table 13. Percent[1] change[2] and factor of percentage change [3] in forest lands in Narragansett Bay Watershed and subwatersheds as defined by Vadeboncoeur and colleagues (2010).

Sub-watersheds	Industrialization 1850–1900		Industrialization 1900–1950		Suburbanization 1950–2000		1850–2000	
	%	*Factor*	%	*Factor*	%	*Factor*	%	*Factor*
Blackstone R. above Millville	0	*1*	21	*2*	4	*1*	25	*2*
Blackstone R. Millville to Manville	9	*1*	11	*1*	5	*1*	25	*1*
Pawtuxet R. above Pettaconsett	9	*1*	25	*1*	4	*1*	38	*2*
Taunton R. above Bridgewater	0	*1*	7	*1*	-10	*1*	-3	*1*
Taunton R. Bridgewater to Taunton	-9	*1*	12	*1*	-13	*1*	-10	*1*
Taunton R. below Taunton	-2	*1*	13	*1*	-17	*1*	-6	*1*
Small watersheds	-5	*1*	15	*1*	-4	*1*	6	*1*
Upper Bay	-10	*1*	-3	*1*	2	*1*	-11	*1*
Lower Bay	17	*2*	13	*1*	1	*1*	31	*2*
Total Narragansett Bay Watershed	**1**	**1**	**13**	**1**	**-2**	**1**	**12**	**1**

[1] Percent of forest lands for each decade by subwatershed from Vadeboncoeur et al. (2010) are presented in the Extended Methods section of this chapter (Table 18).
[2] Result of subtracting the percent of urban lands by Watershed or subwatersheds, within the time-period. For example, in the Upper Bay between 1850 (52 percent) and 1900 (42 percent), there was a 10-percentage point decline in forest lands.
[3] Result of dividing the percent of urban land of the most recent year by the percent of the year to compare by Watershed or subwatersheds. For example, in the Lower Bay from 1850 (27 percent) to 1900 (44 percent), forest land nearly doubled.

The declining trend of forest lands in recent decades and historically points to the dramatic transformation of the Watershed's landscape. The losses weaken the protection that forest lands offer for estuarine and inland water quality, habitat, and human health. While efforts to preserve forested and other natural areas throughout the Watershed have been successful at local, state, regional, and national levels, this should be continued with urgency. For example, most of the forest lands in the Pawtuxet River watershed surround one of the most important drinking water sources in Rhode Island, the Scituate Reservoir and the upper reaches of the Watershed (Figure 2), where most of the forest lands are currently protected as open space; however, nearly thirty percent of ecologically significant natural lands, including unfragmented forests, in this watershed remain unprotected (see "Open Space" chapter).

Land use change analysis by the Estuary Program and EPA (ORD) show that most of the changes in the decade of 2001 to 2011 occurred in areas draining to the Taunton River and the Ten Mile River (Tables 10, 11, and 12). These areas had the largest increases of urban lands as well as the largest losses of forest lands, indicating that urban sprawl occurred and expanded from the urban corridors. These changes in land use are consistent with changes in population distribution, as more people settled in suburban areas (see "Population" chapter). In the Taunton River subwatersheds, these changes began to be more evident from 1950 through 2000 (Tables 12 and 13; Tables 17 and 18 in Extended Methods section of this chapter).

Changes in land use can impact water quality, water quantity, freshwater and estuarine habitats, and human health. The conversion of natural lands to developed lands affects these resources as changes in population demand new urban infrastructure in the form of impervious cover and wastewater infrastructure (see "Impervious Cover" and "Wastewater Infrastructure" chapters). Urban sprawl typically results in loss of forest lands, as shown in this analysis. Sprawl has contributed to habitat fragmentation with smaller areas remaining to protect the Watershed's natural resources (see "Open Space" chapter). In the Taunton River Basin, which had substantial declines in forest lands, natural lands protected as open space represent sixteen percent of the Basin, but nearly ten percent of the most ecologically significant natural lands for watershed protection remain unpreserved (see "Open Space" chapter).

Data Gaps and Research Needs

- Data from the NOAA Coastal Change Analysis Program (C-CAP) should be utilized to improve the spatial and classification accuracy of land cover classes and change analysis for the Watershed.
- Further data analysis to correlate land use and other attributes of the landscape with water quality and habitat conditions is needed to improve understanding of such relationships.
- Additional research is needed to provide better tools for estimating the value of ecosystem services provided by forest lands in the Watershed. Examples of these ecosystem services are water quality protection for both surface and groundwater, wildlife habitat conservation, climate change adaptation, and stormwater mitigation.

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Extended Methods

Table 14. National Land Cover Database sixteen classes and definitions aggregated into seven classes based on Anderson Level I.

Code	Description of 2001, 2006, and 2011 NLCD land cover classes	Aggregated Category
11	Open Water - All areas of open water, generally with less than 25% cover or vegetation or soil.	Water
21	Developed, Open Space - Includes areas with a mixture of some constructed materials, but mostly vegetation in the form of lawn grasses. Impervious surfaces account for less than 20 percent of total cover. These areas most commonly include large-lot single-family housing units, parks, golf courses, and vegetation planted in developed settings for recreation, erosion control, or aesthetic purposes.	Urban or Built Up
22	Developed, Low Intensity -Includes areas with a mixture of constructed materials and vegetation. Impervious surfaces account for 20-49 percent of total cover. These areas most commonly include single-family housing units.	Urban or Built Up
23	Developed, Medium Intensity - Includes areas with a mixture of constructed materials and vegetation. Impervious surfaces account for 50-79 percent of the total cover. These areas most commonly include single-family housing units.	Urban or Built Up
24	Developed, High Intensity - Includes highly developed areas where people reside or work in high numbers. Examples include apartment complexes, row houses and commercial/industrial. Impervious surfaces account for 80 to100 percent of the total cover.	Urban or Built Up
31	Barren Land (Rock/Sand/Clay) - Barren areas of bedrock, desert pavement, scarps, talus, slides, volcanic material, glacial debris, sand dunes, strip mines, gravel pits and other accumulations of earthen material. Generally, vegetation accounts for less than 15% of total cover.	Barren Land
41	Deciduous Forest - Areas dominated by trees generally greater than 5 meters tall, and greater than 20% of total vegetation cover. More than 75 percent of the tree species shed foliage simultaneously in response to seasonal change.	Forest Land
42	Evergreen Forest - Areas dominated by trees generally greater than 5 meters tall, and greater than 20% of total vegetation cover. More than 75 percent of the tree species maintain their leaves all year. Canopy is never without green foliage.	Forest Land
43	Mixed Forest - Areas dominated by trees generally greater than 5 meters tall, and greater than 20% of total vegetation cover. Neither deciduous nor evergreen species are greater than 75 percent of total tree cover.	Forest Land
52	Shrub/Scrub - Areas dominated by shrubs; less than 5 meters tall with shrub canopy typically greater than 20% of total vegetation. This class includes true shrubs, young trees in an early successional stage or trees stunted from environmental conditions.	Brushland
71	Grassland/Herbaceous - Areas dominated by gramanoid or herbaceous vegetation, generally greater than 80% of total vegetation. These areas are not subject to intensive management such as tilling, but can be utilized for grazing.	Agricultural land
72	Sedge/Herbaceous - Alaska only areas dominated by sedges and forbs, generally greater than 80% of total vegetation. This type can occur with significant other grasses or other grass like plants, and includes sedge tundra, and sedge tussock tundra.	Agricultural land

Table 14 continued

Code	Description of 2001, 2006, and 2011 NLCD land cover classes	Aggregated Category
81	Pasture/Hay - Areas of grasses, legumes, or grass-legume mixtures planted for livestock grazing or the production of seed or hay crops, typically on a perennial cycle. Pasture/hay vegetation accounts for greater than 20 percent of total vegetation.	Agricultural land
82	Cultivated Crops - Areas used for the production of annual crops, such as corn, soybeans, vegetables, tobacco, and cotton, and also perennial woody crops such as orchards and vineyards. Crop vegetation accounts for greater than 20 percent of total vegetation. This class also includes all land being actively tilled.	Agricultural land
90	Woody Wetlands - Areas where forest or shrub land vegetation accounts for greater than 20 percent of vegetative cover and the soil or substrate is periodically saturated with or covered with water.	Wetland
95	Emergent Herbaceous Wetlands - Areas where perennial herbaceous vegetation accounts for greater than 80 percent of vegetative cover and the soil or substrate is periodically saturated with or covered with water.	Wetland

Table 15. Rhode Island land use categories (2003–2004) state "crosswalk" based on Anderson Level 1.

LCLU	Description	Aggregated Category
111	High Density Residential (<1/8 acre lots)	Urban or Built Up Land
112	Medium High Density Residential (1/4 to 1/8 acre lots)	Urban or Built Up Land
113	Medium Density Residential (1 to 1/4 acre lots)	Urban or Built Up Land
114	Medium Low Density Residential (1 to 2 acre lots)	Urban or Built Up Land
115	Low Density Residential (>2 acre lots)	Urban or Built Up Land
120	Commercial (sale of products and services)	Urban or Built Up Land
130	Industrial (manufacturing, design, assembly, etc.)	Urban or Built Up Land
141	Roads (divided highways >200' plus related faci	Urban or Built Up Land
142	Airports (and associated facilities)	Urban or Built Up Land
143	Railroads (and associated facilities)	Urban or Built Up Land
144	Water and Sewage Treatment	Urban or Built Up Land
145	Waste Disposal (landfills, junkyards, etc.)	Urban or Built Up Land
146	Power Lines (100' or more width)	Urban or Built Up Land
147	Other Transportation (terminals, docks, etc.)	Urban or Built Up Land
151	Commercial/Residential Mixed	Urban or Built Up Land
152	Commercial/Industrial Mixed	Urban or Built Up Land
161	Developed Recreation (all recreation)	Urban or Built Up Land
162	Vacant Land	Urban or Built Up Land
163	Cemeteries	Urban or Built Up Land
170	Institutional (schools, hospitals, churches, etc.)	Urban or Built Up Land
210	Pasture (agricultural not suitable for tillage)	Agricultural Land
220	Cropland (tillable)	Agricultural Land
230	Orchards, Groves, Nurseries	Agricultural Land
240	Confined Feeding Operations	Agricultural Land
250	Idle Agriculture (abandoned fields and orchards)	Agricultural Land

Table 15 continued

LCLU	Description	Aggregated Category
300	Brushland (shrub and brush areas, reforestation)	Brushland
410	Deciduous Forest (>80% hardwood)	Forest Land
420	Softwood Forest (>80% softwood)	Forest Land
430	Mixed Forest	Forest Land
500	Water	Water
600	Wetland	Wetland
710	Beaches	Barren Land
720	Sandy Areas (not beaches)	Barren Land
730	Rock Outcrops	Barren Land
740	Mines, Quarries and Gravel Pits	Barren Land
750	Transitional Areas (urban open)	Barren Land
760	Mixed Barren Areas	Barren Land

Table 16. Massachusetts land use categories (2005) state "crosswalk" based on Anderson Level 1.

LUCODE	Description	Aggregated Category
1	Cropland	Agricultural Land
2	Pasture	Agricultural Land
3	Forest	Forest Land
4	Non-Forested Wetland	Wetland
5	Mining	Barren Land
6	Open Land	Urban or Built Up Land
7	Participation Recreation	Urban or Built Up Land
8	Spectator Recreation	Urban or Built Up Land
9	Water-Based Recreation	Urban or Built Up Land
10	Multi-Family Residential	Urban or Built Up Land
11	High Density Residential	Urban or Built Up Land
12	Medium Density Residential	Urban or Built Up Land
13	Low Density Residential	Urban or Built Up Land
14	Saltwater Wetland	Wetland
15	Commercial	Urban or Built Up Land

Table 16 continued

LUCODE	Description	Aggregated Category
16	Industrial	Urban or Built Up Land
17	Transitional	Barren Land
18	Transportation	Urban or Built Up Land
19	Waste Disposal	Urban or Built Up Land
20	Water	Water
23	Cranberry Bog	Agricultural Land
24	Powerline/Utility	Urban or Built Up Land
25	Saltwater Sandy Beach	Barren Land
26	Golf Course	Urban or Built Up Land
29	Marina	Urban or Built Up Land
31	Urban Public/Institutional	Urban or Built Up Land
34	Cemetery	Urban or Built Up Land
35	Orchard	Agricultural Land
36	Nursery	Agricultural Land
37	Forested Wetland	Wetland
38	Very Low Density Residential	Urban or Built Up Land
39	Junkyard	Urban or Built Up Land
40	Brushland/Successional	Brushland

Table 17. Percent of urban lands in subwatersheds of the Narragansett Bay Watershed (Vadeboncoeur et al. 2010)

Sub-watersheds	1850	1860	1870	1880	1890	1900	1910	1920	1930	1940	1950	1960	1970	1980	1990	2000
Blackstone R. above Millville	3	4	6	7	9	12	14	17	19	19	21	21	21	21	22	27
Blackstone R. Millville to Manville	2	2	3	3	4	5	6	7	8	8	9	9	9	10	10	10
Pawtuxet R. above Pettaconsett	1	2	2	2	2	2	2	2	3	3	3	4	6	6	7	8
Taunton R. above Bridgewater	3	4	4	5	6	9	11	12	12	12	13	16	21	24	25	30
Taunton R. Bridgewater to Taunton	6	8	10	11	13	15	17	18	18	18	20	21	24	25	28	39
Taunton R. below Taunton	3	3	4	4	4	4	5	6	7	8	9	12	17	20	22	34
Small watersheds	2	3	5	5	7	9	12	14	17	17	19	21	24	25	27	33
Upper Bay	8	10	12	18	24	32	39	43	47	48	51	51	52	52	52	57
Lower Bay	5	6	6	8	9	10	12	12	12	14	21	28	29	31	34	36
Total Watershed	**4**	**4**	**5**	**7**	**8**	**11**	**13**	**15**	**16**	**16**	**18**	**20**	**23**	**24**	**25**	**30**

Table 18. Percent of forest (includes inland waterbodies) in subwatersheds of the Narragansett Bay Watershed (Vadeboncoeur et al. 2010).

Sub-watersheds	1850	1860	1870	1880	1890	1900	1910	1920	1930	1940	1950	1960	1970	1980	1990	2000
Blackstone R. above Millville	41	52	61	48	41	41	47	52	59	61	62	65	67	70	71	66
Blackstone R. Millville to Manville	60	60	72	67	70	69	71	74	77	78	80	81	83	84	85	85
Pawtuxet R. above Pettaconsett	50	51	51	55	57	59	66	74	81	83	84	86	86	88	89	88
Taunton R. above Bridgewater	70	71	77	69	69	70	71	73	75	77	77	76	73	72	72	67
Taunton R. Bridgewater to Taunton	65	63	83	69	58	56	59	62	66	67	68	68	67	67	66	55
Taunton R. below Taunton	69	73	70	67	65	67	71	75	79	80	80	79	76	74	74	63
Small watersheds	55	58	66	55	51	50	54	58	61	64	65	65	64	66	66	61
Upper Bay	52	55	58	51	47	42	38	38	38	39	39	41	42	44	46	41
Lower Bay	27	29	33	38	41	44	47	52	57	60	57	54	56	59	60	58
Total Watershed	**54**	**57**	**63**	**57**	**55**	**55**	**58**	**62**	**66**	**67**	**68**	**68**	**68**	**69**	**70**	**66**



State of Narragansett Bay and Its Watershed
2017 Technical Report

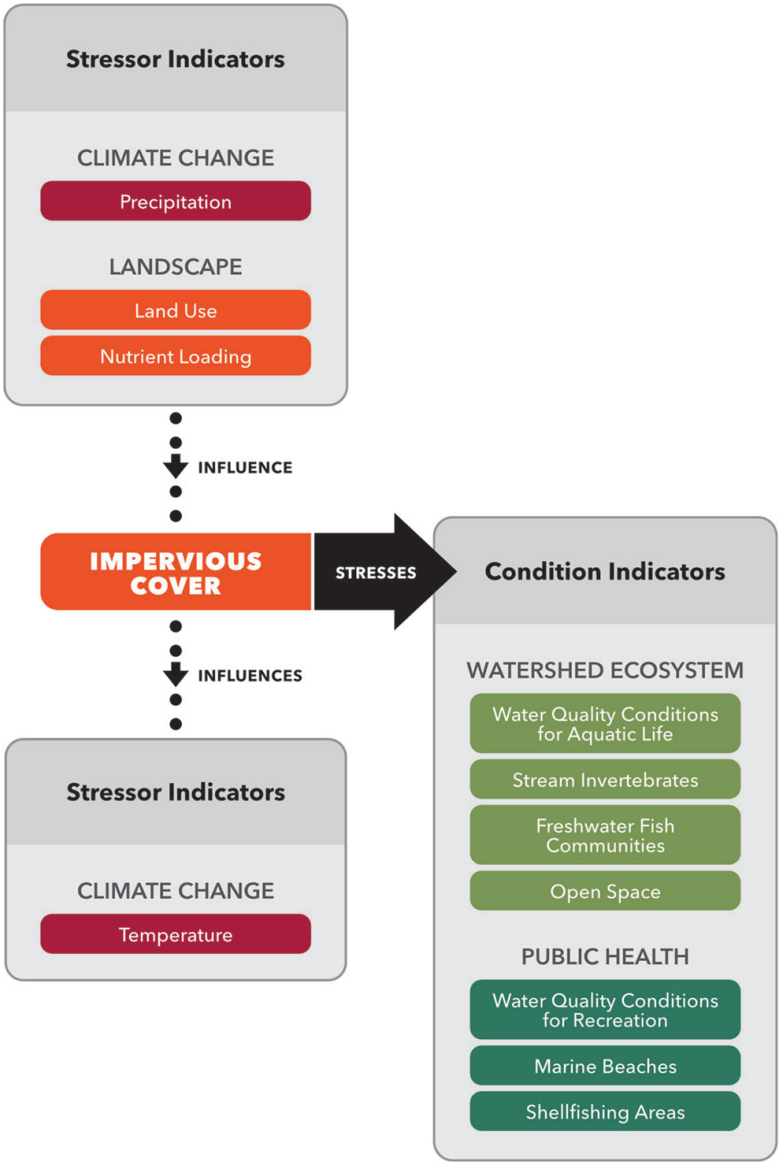
Landscape Stressor Indicators

CHAPTER 6: IMPERVIOUS COVER

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Photo: Quonset Point, North Kingstown, RI (Ayla Fox)

Overview



BACKGROUND

- Pavement, buildings, and other forms of impervious cover are ubiquitous across the Narragansett Bay Watershed. Precipitation falling on impervious areas flows as storm-water into streams and rivers, rather than infiltrating through the soil where nutrients, pathogens, and other pollutants can be removed naturally before they enter the Bay. In addition, impervious cover makes the water warmer. Increases in impervious cover are frequently associated with loss of natural open space.

- As a result, the amount of impervious cover is an important indicator of stress on many aspects of ecosystem condition (water quality conditions for aquatic life, stream invertebrates, fish communities, and open space) and public health (water quality conditions for recreation, marine beaches, and shellfishing areas). It also influences other stressors (temperature, precipitation, land use, nutrient loading).

KEY FINDINGS

- ***Status:*** Based on data from 2011 (Rhode Island) and 2005 (Massachusetts), fourteen percent of the Narragansett Bay Watershed was impervious cover, and 20 percent of the impervious cover was located in the immediate coastal lands that drain to the Bay. Thirty-six of the 52 subwatersheds in the Watershed had more than the detrimental threshold of ten percent impervious cover, and those impacted subwatersheds represent 66 percent of the Narragansett Bay Watershed.

Introduction

The term impervious cover refers to buildings, pavement, and other unnatural, solid surfaces in a watershed that cause water and pollutants to flow aboveground, rather than being absorbed and retained by soils or vegetation. By promoting greater runoff from the land into streams and waterbodies, impervious cover results in multiple stressors to a watershed, such as increased pollutant loads, altered stream flow, decreased stream bank stability, decreased groundwater recharge, increased peak flows, and increased water temperatures. These stressors have direct impacts on aquatic habitat and associated species.

The amount of impervious cover in a watershed is an integrative, comprehensive, and measurable indicator of the impacts of urban development on ecosystems and water resources (Schueler 1994 and 2003, Allan 2004, Wickham et al. 2016). Impacts in four broad categories include changes in hydrologic, physical, water quality, and biological conditions (Schueler 2003). Impervious cover can influence many hydrologic aspects of streams by shortening the time to flood peaks and causing increases in bankfull discharges and surface runoff volume. Impervious cover is considered a key environmental indicator because of its impacts on aquatic systems and role in increasing the transport and concentration of pollutants (Leopold 1968, Arnold and Gibbons 1996).

There are numerous studies suggesting that watershed degradation increases substantially when impervious cover exceeds a threshold of ten percent of the watershed's land area. Schueler (1994, 2003) was among the first to establish, and later refine (Schueler et al. 2009), a conceptual model to apply impervious surface as an index of environmental disturbance. According to the conceptual model, threshold ranges of impervious cover are associated with different degrees of stream quality as reflected in aquatic life: sensitive (1–10 percent impervious cover), impacted (11–25 percent impervious cover), non-supporting (26–60 percent impervious cover), and urban drainage (greater than 60 percent impervious cover).

In Maryland, King and colleagues (2011) documented declines in diversity of macroinvertebrates at very low thresholds of less than two percent impervious cover. The states of Connecticut and Maine presently use impervious cover thresholds as a mechanism to identify stream segments that are deemed impaired under the Clean Water Act. Connecticut uses a twelve percent impervious cover threshold to identify waters impaired for aquatic life, and Maine uses thresholds of five, nine, and fifteen percent for different stream classifications (Wickham et al. 2014). These threshold levels appear to agree with the ranges provided by Schueler and colleagues (2009), as the impervious cover model clearly identifies the wide variability in stream quality that can occur, especially at lower percentages of impervious cover.

Armstrong and colleagues (2011) analyzed relationships between fish communities and anthropogenic factors, including impervious cover, across catchment

areas within Massachusetts, where 60 percent of the Narragansett Bay Watershed is located. The results indicated that a one-percent increase in impervious cover was associated with a 3.7 percent decrease in the relative abundance of fluvial fish, a 5.4 percent decrease in fluvial fish species richness, and an 8.7 percent decrease in brook trout relative abundance (see "Freshwater Fish Communities" chapter).

For this chapter, the Narragansett Bay Estuary Program analyzed the extent of impervious cover in the Narragansett Bay Watershed as a whole and at three spatial scales within the Watershed to identify: 1) River Basins that have the most impervious cover, 2) the distribution of impervious cover among watersheds (HUC10) and subwatersheds (HUC12) across the Narragansett Bay Watershed (drainage areas within the Watershed that are more intensively developed), and 3) subwatersheds (HUC12) categorized as sensitive (less than ten percent impervious cover) or impacted (equal or greater than ten percent impervious cover). Subwatersheds categorized as sensitive are important to protect by not exceeding the ten percent threshold, whereas an impacted subwatershed is where impervious cover might be already causing detrimental effects of water quality and habitat, and where restoration efforts can be targeted (see the Appendix for description and maps of the different geographical scales presented in this chapter).

Methods

The Estuary Program analyzed and calculated the status of impervious cover in the Narragansett Bay Watershed as the percentage of land area that is covered by impervious surfaces by means of geospatial analysis and cross-tabulation (i.e., calculation at different geographical scales) (Esri, 2016). Impervious cover datasets were obtained from Rhode Island (RIGIS) and Massachusetts (MassGIS) and were reconciled into a seamless dataset by the Rhode Island Department of Environmental Management. The RIGIS data were based on imagery captured in 2011 with a spatial resolution of two feet (0.61 meter). The MassGIS data included imagery acquired in 2005 with a spatial resolution of 3.28 feet (one meter). While the differing dates of the imagery could not be reconciled, the differing spatial resolutions were addressed through the creation of a seamless dataset that had a resolution of 25 feet (7.62 meters). This coarser resolution was deemed appropriate for an analysis at the watershed and subwatershed scale.

Data on impervious surfaces were also available through the National Land Cover Database (NLCD) for different years, and thus useful for a change analysis. However, the resolution of the national dataset was coarser than the data at watershed level used by the Estuary Program (30-meter vs. 7.62-meter). An exploratory analysis was performed using NLCD at the different time-steps, and while it proved to be valuable to identify relative changes of impervious cover, it also resulted on large percent-error when compared with the finer resolution data, as total percentage of impervious cover was overestimated at the watershed scale. However, the resolution of the national dataset was in most cases thirty meters (98.4 feet). The high-resolution datasets provided by RIGIS and MassGIS included smaller cell areas, or pixels, and the seamless dataset with a resolution of 25 feet more accurately reflected the extent of impervious cover at the watershed and subwatershed scales. Further, studies in Massachusetts and Rhode Island concluded that the NLCD underestimated impervious cover in low-development areas and overestimated it in highly urbanized areas (Weiskel et al. 2010, Smucker et al. 2016).

Using geospatial analysis tools (Esri 2016), the Estuary Program used the seamless impervious cover data developed at the Watershed scale by RIDEM to calculate the total area and percentage of impervious cover within the Watershed (overall percent), River Basins, HUC10 watersheds, and HUC12 subwatersheds. For this report, it was not possible to determine trends of changes in impervious cover, as the methods and spatial resolution used to map impervious cover in past studies were not comparable with the most recent datasets, such as the limitations from the NLCD datasets as noted above. However, the Estuary Program is coordinating with researchers at USEPA to examine trends in impervious cover for future reports. Impervious cover thresholds similar to those offered by Schueler and colleagues (2009) were used to relate impervious cover to stream quality within the Narragansett Bay Watershed. Specifically, HUC12 subwatersheds with less than ten percent impervious cover were categorized as sensitive, and those with ten percent or more impervious cover as impacted.

The Estuary Program's analysis focused on total impervious cover, rather than effective impervious cover. Effective impervious cover, also termed connected impervious cover, is defined as gray stormwater infrastructure or continuous paved surfaces that are directly connected to stream channels, thus conveying runoff effectively to streams. Analyzing effective impervious cover would have required imagery with higher spatial resolution than was available, as well as site-specific data on locations of gray (e.g., catch basins) or green (e.g., grass swales) stormwater infrastructure. In addition, it was

not possible to identify impervious areas draining to stormwater best management practices (BMPs) designed to meet recharge and runoff volume criteria, and thus hydrologically disconnecting the impervious cover from stream channels, because data were not readily available or standardized for analyses at the watershed scale.

Status

Based on seamless impervious cover data at the Watershed scale, fourteen percent of the Narragansett Bay Watershed is covered by buildings, roads, parking lots, and other hard surfaces (Figure 1), as there were an estimated 155,274 acres of impervious cover in the 1,091,120-acre Watershed (Table 1). The Pawtuxet River, Taunton River, and Blackstone River Basins all had twelve percent impervious cover. In contrast, the coastal Narragansett Bay Basin had 20 percent impervious cover (Table 1). Total area by River Basins, watersheds (HUC10) and subwatersheds (HUC12) can be found in the Appendix of this report.

The Estuary Program also calculated the extent and percentage of impervious cover in the HUC10 watersheds and their nested HUC12 subwatersheds to examine the spatial distribution of the fourteen percent total impervious cover across the Narragansett Bay Watershed. The analysis identified areas where impervious cover was concentrated within the Watershed and categorized subwatersheds (HUC12) as sensitive or impacted.

Among the HUC10 watersheds, the Middle Taunton River watershed had the lowest proportion of developed land with an impervious coverage of only nine percent, and the Lower Blackstone River watershed was the second least-developed watershed with just over ten percent impervious cover. The two HUC10 watersheds with the highest percentages of impervious cover were the Ten Mile River and Woonasquatucket River-Moshassuck River watersheds with 24 and 23 percent impervious cover respectively (Table 2).

The Narragansett Bay HUC10 watershed,[1] which encompasses lands along the immediate coastline that discharge directly to the Bay, was third highest with 20 percent impervious cover. However, it had by far the highest total acreage of impervious cover of all the watersheds. Its 31,100 acres of impervious cover represented 20 percent of all impervious cover in Narragansett Bay's entire Watershed (Table 2). The Upper Blackstone River watershed was second in terms of acreage with 20,238 acres of impervious cover, representing thirteen percent of all impervious cover in Narragansett Bay's entire Watershed.

The three HUC12 subwatersheds with the least impervious cover–3.2 percent–were all located in the upper reaches of the Pawtuxet River HUC10 watershed, also referred to as the Pawtuxet River Basin.[2] In contrast, the Pawtuxet River HUC12 subwatershed, located in the lower reaches where the Pawtuxet River enters Narragansett Bay, had 36 percent impervious cover–the second highest among all 52 subwatersheds. That and three other subwatersheds that drain directly to the Providence River estuary ranked as the most impacted with over 30 percent impervious cover; the Seekonk River subwatershed was highest with 47 percent of its area covered by pavement, roads, and rooftops (Table 3; Figure 1).

Of the 52 HUC12 subwatersheds that are nested within the Narragansett Bay Watershed, 36 subwatersheds (66 percent) were categorized as impacted because they exceeded the ten percent impervious cover threshold, and sixteen subwatersheds (34 percent) were deemed sensitive as they were below that threshold (Table 4; Figure 1). Figure 1 illustrates the variation in impervious cover among HUC12 subwatersheds according to percentage thresholds.

Discussion

The amount of impervious cover in Narragansett Bay's subwatersheds (HUC12) demonstrates the considerable extent of developed land around the Bay, reflecting both the historically known urban

[1] Definitions and total area of Narragansett Bay Watershed, River Basins and watersheds (HUC10) and subwatersheds (HUC12) can be found in the Appendix of this report.

[2] The term "Pawtuxet River HUC10 watershed," designated by the U.S. Geological Survey, refers to all land drained by the Pawtuxet River. It is the same as the "Pawtuxet River Basin," a term used by the Estuary Program. There is also a smaller "Pawtuxet River HUC 12 subwatershed" that is designated by the U.S. Geological Survey; it is nested within the Pawtuxet River HUC10 watershed and covers a drainage area along Bay's immediate coastline.

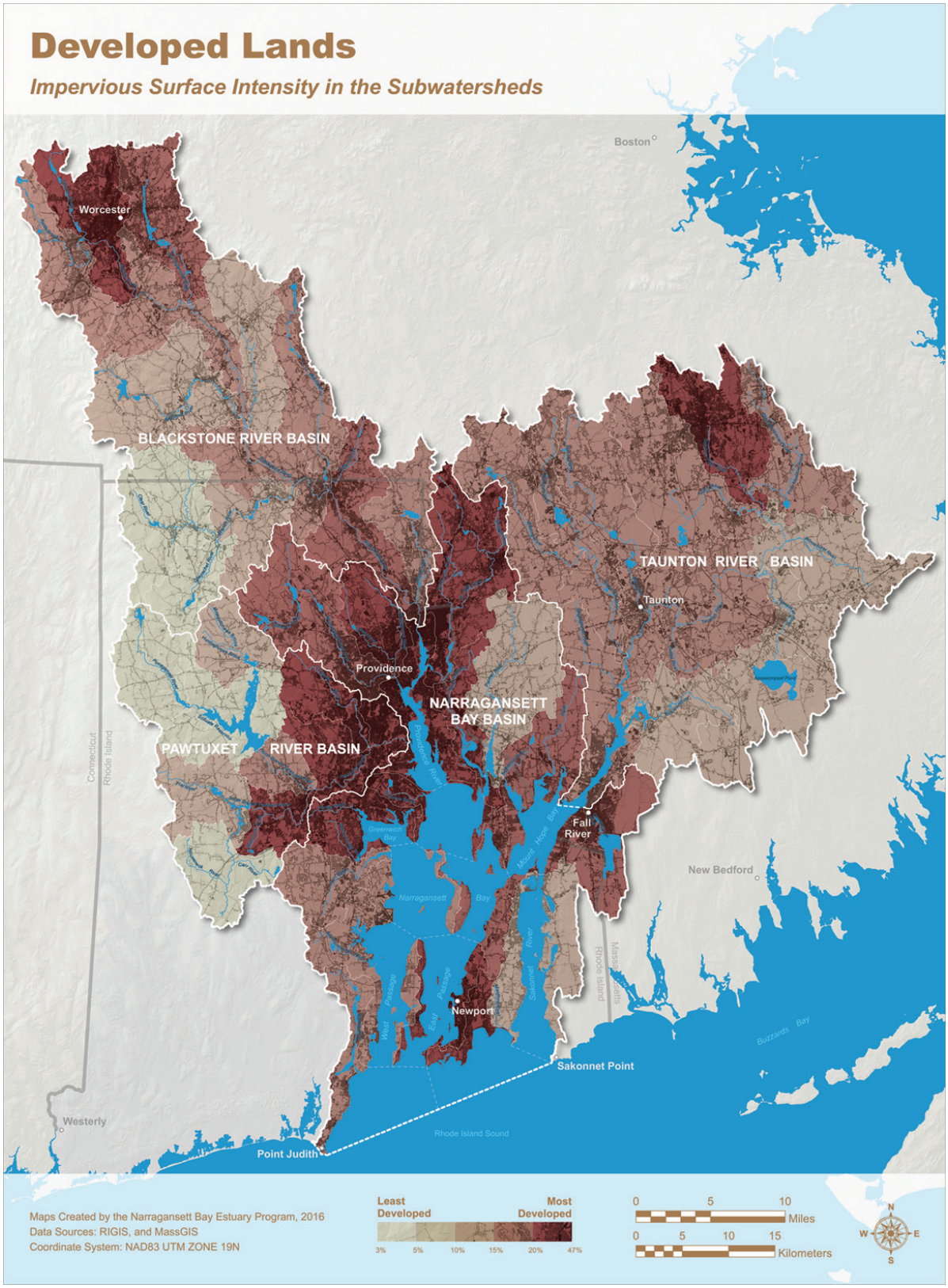


Figure 1. Extent of impervious cover in the Narragansett Bay Watershed and percentage of impervious cover in subwatersheds (HUC12). White lines indicate boundaries of River Basins. For each subwatershed, light brown to dark red shading indicates percentage of its area that is covered with impervious surface.

Table 1. Extent and percentage of impervious cover in the Narragansett Bay Watershed and in River Basins, listed from highest to lowest percentage.

River and Coastal Basins in Narragansett Bay Watershed	Impervious Cover	
	Extent (Acres)	Percentage of Watershed Area
Coastal Narragansett Bay Basin	58,956	20
Blackstone River Basin	37,586	12
Taunton River Basin	41,105	12
Pawtuxet River Basin	17,627	12
Total in the Watershed:	**155,274**	**14**

Table 2. Extent and percentage of impervious cover in the HUC10 watersheds that are nested within the Narragansett Bay Watershed, listed from highest to lowest percentage.

HUC10 Watershed Name	Impervious Cover		
	Extent (Acres)	Percentage of HUC10 Watershed Area[1]	Percentage of Narragansett Bay Watershed's Impervious Cover[2]
Ten Mile River	8,394	24	5
Woonasquatucket River-Moshassuck River	11,085	23	7
Narragansett Bay	31,100	20	20
Upper Taunton River	14,167	16	9
Upper Blackstone River	20,238	15	13
Threemile River	6,919	13	5
Lower Taunton River-Frontal Mount Hope Bay	12,546	13	8
Palmer River	5,201	12	3
Pawtuxet River	17,627	12	11
Lower Blackstone River	17,346	10	11
Middle Taunton River	10,624	9	7

[1] Area of impervious cover within the HUC10 watershed relative to the total area of the HUC10 watershed: [(Acres of impervious cover in HUC10 watershed/Acres of HUC10 watershed)*100]

[2] Area of impervious cover within the HUC10 watershed relative to the total area of impervious cover (not total land area) in the Narragansett Bay Watershed: [(Acres of impervious cover in HUC10 watershed/Acres of impervious cover in Narragansett Bay Watershed)*100]

Table 3. Extent and percentage of impervious cover in the HUC12 subwatersheds that are nested within the Narragansett Bay Watershed, listed from lowest to highest percentage impervious cover.

HUC12 Subwatershed Name	Impervious Cover	
	Extent (Acres)	Percentage
Less than 5 Percent Impervious		
Big River	580	3.2
Scituate Reservoir	801	3.2
Barden Reservoir-Ponaganset River	683	3.2
Clear River	1,385	4.8
Chepachet River	669	4.9
5.1 to 10 Percent Impervious		
Headwaters South Branch Pawtuxet River	1,005	5.6
Assawompsett Pond	1,784	5.7
Moswansicut Pond-Huntinghouse Brook	824	5.8
West River	1,546	6.5
Assonet River	1,495	6.9
Mumford River	2,711	7.5
Branch River	1,313	7.7
Palmer River	2,590	8.0
Sakonnet River	1,924	8.6
Winnetuxet River-Taunton River	2,729	8.6
Nemasket River	1,275	9.5
10.1 to 20 Percent Impervious		
Mill River – Blackstone	2,429	11.2
Wading River	3,213	11.5
Mill River	3,330	11.5
North Branch Pawtuxet River	1,048	11.6
Satucket River	2,626	11.7
Emerson Brook-Blackstone River	2,584	11.8
Singletary Brook-Blackstone River	3,088	12.1
Kettle Brook	2,540	12.2
Abbott Run	2,294	12.5
Hunt River	1,962	12.6
Cotley River-Taunton River	4,837	12.6
Town River	5,043	13.0
Pettaquamscutt River-Frontal Atlantic Ocean	1,432	13.2
Lower West Passage	2,318	13.5
Threemile River	3,706	13.9
Upper West Passage	723	14.0
Taunton River-Frontal Mount Hope Bay	4,569	14.9
Upper East Passage	1,538	16.2
Quequechan River	3,152	16.4
Mount Hope Bay	4,525	17.5
Peters River-Blackstone River	5,127	19.1
Quinsigamond River	4,688	19.3
Woonasquatucket River	6,459	19.7
Greater than 20 Percent Impervious		
South Branch Pawtuxet River	2,458	23.1
Ten Mile River	8,394	23.6
Matfield River	6,497	24.3
Pocassett River	3,196	24.3
Barrington River-Warren River	2,611	24.3
Aquidneck Island-Frontal Atlantic Ocean	1,629	27.4
Lower East Passage	1,988	28.5
Tatnuck Brook-Blackstone River	7,211	29.0
Greenwich Bay	3,972	29.7
Upper Narragansett Bay	1,945	30.3
Moshassuck River	4,627	31.3
Pawtuxet River	7,033	36.0
Seekonk River-Providence River	7,143	47.2

centers and the recent evidence of sprawl outside the urban corridors (see "Land Use" chapter). The urbanized areas around Providence, Worcester, Brockton, Fall River, and Newport contained the vast majority of the impervious cover in the Narragansett Bay Watershed (Figure 1). The urbanized areas around Providence, Worcester, Brockton, Fall River, and Newport contained the vast majority of the impervious cover in the Narragansett Bay Watershed (Figure 1). Impervious cover exceeded the ten percent ecological threshold in 36 of the 52 subwatersheds, which together contained 131,933 acres of impervious cover or 85 percent of all the impervious cover in the Narragansett Bay Watershed (Table 5). Those 36 subwatersheds constituted 66 percent of the total Narragansett Bay Watershed area (722,357 acres), meaning 85 percent of the impervious cover was concentrated in 66 percent of the Watershed (Figure 1; Table 4).

Using the impervious cover model prepared by Schueler and colleagues (2009), 28 subwatersheds were deemed impacted (11–25 percent impervious cover), and eight were deemed non-supporting (greater than 26 percent impervious cover). Non-supporting streams are defined as no longer supporting hydrological, habitat, water quality, and biological diversity functions (Schueler et al. 2009).

Among the sixteen HUC12 subwatersheds deemed sensitive, the five with the lowest percentages of impervious cover–all of which had less than five percent impervious cover–were located in the western portion of the Narragansett Bay Watershed. The three lowest were located in the Pawtuxet River Basin (including two near the Scituate Reservoir), and the two next lowest were in the southwestern portion of the Blackstone River Basin (Figure 1; Tables 3 and 4). The other eleven sensitive subwatersheds, which had between five and ten percent impervious cover, were distributed throughout the upper reaches of the Narragansett Bay Watershed were distributed throughout the upper reaches of the Narragansett Bay Watershed, where not only the most of forested areas in the Watershed are concentrated, but also where a large extent of these natural lands including those with high ecological integrity remained unprotected (see "Land Use" and "Open Space" chapters). Two HUC12 subwatersheds that stand in stark contrast to other urbanized areas in the Coastal Narragansett Bay Basin were the Palmer River and the Sakonnet River subwatersheds, which had only eight percent and 8.6 percent impervious cover, respectively, compared to more than 20 percent in their neighboring subwatersheds (Figure 1; Table 3). Importantly, the sixteen HUC12 subwatersheds in which impervious cover was below ten percent

Table 4. HUC12 subwatersheds with total area and percent within impervious cover thresholds.

Impervious Cover Thresholds	Within thresholds				
	Number of HUC12 Subwatersheds	Total Area of the HUC12 Subwatersheds (Acres) [1]	Percentage of the Narragansett Bay Watershed [2]	Total Area of Impervious Cover (Acres) [3]	Percentage of Total Impervious Cover [4]
Sensitive	*16*	*368,755*	*34*	*23,314*	*15*
0 to 5%	5	106,740	10	4,118	3
5.1 to 10%	11	262,014	24	19,196	12
Impacted	*36*	*722,357*	*66*	*131,933*	*85*
10.1 to 15%	17	379,968	35	47,743	31
15.1 to 20%	6	138,491	13	25,487	16
20.1% or more	13	203,898	19	58,703	38
Total in the Watershed	**52**	**1,091,112**		**155,247**	

[1] Sum of total acreage of all HUC12 subwatersheds that are within each threshold group
[2] Relative to the total acreage of Narragansett Bay Watershed: [(Total acreage of HUC12 subwatersheds within threshold group/Total acreage of Narragansett Bay Watershed)*100]
[3] Sum of acreage of impervious cover of all HUC12 subwatersheds that are within each threshold group
[4] Relative to the total acreage of impervious cover of Narragansett Bay Watershed: [(Total acreage impervious of all HUC12 subwatersheds within threshold group/Total acreage of impervious cover in Narragansett Bay Watershed)*100]

(sensitive subwatersheds) encompassed 368,755 acres, representing 34 percent of the Narragansett Bay Watershed (Table 4), and should be examined for continued protection through open space and stream corridor protection (see "Open Space" chapter). Eleven of the HUC12 subwatersheds categorized as sensitive had more than 50 percent forested land, while not surprisingly the thirteen HUC12 subwatersheds with over 20 percent impervious cover were the most urbanized, each having more than 50 percent urban lands (see "Land Use" chapter).

While this chapter does not present trends of changes in impervious cover over time, the creation of impervious surfaces is irreversible in most cases, and typically the percentage of impervious cover in a watershed only increases over time. However, the effects of increased impervious cover can be mitigated to some degree by retrofitting urban drainage infrastructure with storage areas to contain stormwater (Novotny et al. 2005). Numerous management efforts have been undertaken in the Narragansett Bay Watershed to implement green infrastructure through stormwater best management practices, but these efforts are not widely inventoried, monitored, or assessed for their effectiveness. The Estuary Program is currently engaged with partners at the EPA Atlantic Ecology Division Laboratory to improve the assessment of impervious cover by using more robust methods with data that were not readily available when this report was being developed, including an assessment of trends at higher spatial resolution across multiple years. In addition, this analysis can shed light on findings by Armstrong and colleagues (2011) that each one-percent increase of impervious cover affects important fish communities in the freshwaters of the Watershed. Preliminary results by the Estuary Program and USEPA indicated that the average of percent relative abundance for fluvial fish across HUC10 watersheds ranged from nine to 65 percent in the Upper Blackstone River and Palmer River HUC10 watersheds, respectively, and approximately 640 square miles across the Narragansett Bay Watershed encompass catchment areas of importance to further protect or restore brook trout habitat (see "Freshwater Fish Communities" chapter).

As discussed above, not all areas of impervious cover are directly connected hydrologically to surface waterbodies through storm drains or other drainage infrastructure (Weiskel et al. 2010). Effective impervious area–also termed connected impervious area–has a greater effect on water quality than impervious areas separated from waterbodies (Schuler 2004). Figure 2 shows an example of impervious cover that is directly connected to the Bay. The development of a functional relationship between total impervious cover and effective impervious cover is a topic of active research (Weiskel et al. 2010). Wickham and colleagues (2014, 2016) discussed important differences among the total impervious cover in a watershed, the connected or effective impervious cover, and the impervious cover in the vicinity of surface waters.

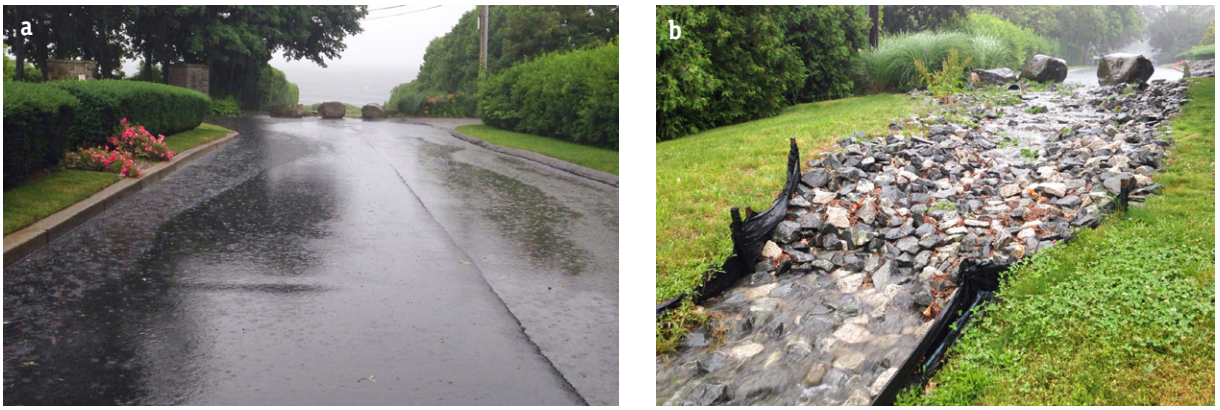


Figure 2. Example of effective impervious cover, also termed connected impervious cover, carrying stormwater directly to Narragansett Bay. (a) Road pavement draining directly to the Bay via (b) rock riprap, which is designed primarily to prevent erosion. While the riprap may slow down water velocity, it does not increase infiltration of stormwater and still effectively conveys runoff to the Bay. Photo credit: Eivy Monroy

When considering the results presented in this chapter, it should be assumed that a fraction of the total impervious cover has been disconnected hydrologically or hydraulically through the implementation of stormwater best management practices or gray infrastructure, such as areas regulated as Municipal Separate Storm Sewer Systems (better known as MS4s). For stormwater best management practices that have been implemented, there is a dearth of research and information related to their actual effectiveness in recharging and capturing stormwater runoff.

Refinement of the impervious cover indicator for the Narragansett Bay Watershed would require further analysis of effective (connected) versus disconnected impervious cover and proximity of impervious cover to surface waterbodies. Gathering an inventory of stormwater infrastructure, the locations of best management practices, and drainage areas to which they are connected is essential. Better management of stormwater runoff from impervious surfaces will help municipalities become more resilient to the more intense storms and rainfall that are projected with climate change (see "Precipitation" chapter).

Data Gaps and Research Needs

- Data on sites where stormwater best management practices have been installed are not readily available. To address this important data gap, information could be compiled from state and local permitting records. Mechanisms to capture the data moving forward need to be developed. The data should include location, drainage area being captured, type of treatment provided, and effectiveness of treatment.
- Research is needed to examine hydrological regimes and runoff to major rivers and streams at appropriate subwatershed scales to evaluate the relationship between percent impervious cover and various water quality and habitat indicators, such as water temperature, water quality for aquatic life, stream invertebrates, fish communities, and all public health indicators. Likewise, spatial data on impervious cover, in conjunction with other indicators such as land use, should be investigated as a proxy to estimate nutrient loadings from non-point sources at varying watershed scales.

Acknowledgments

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State of Narragansett Bay and Its Watershed
2017 Technical Report

Landscape Stressor Indicators

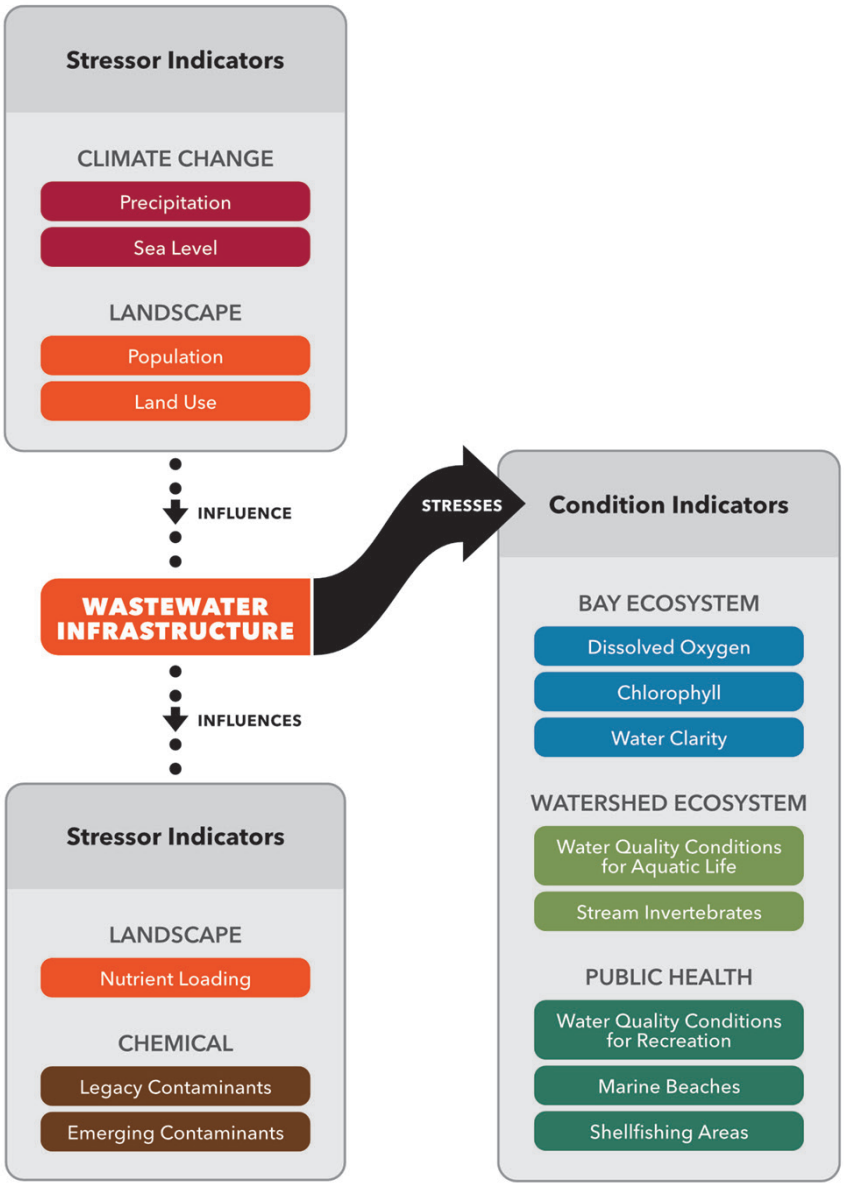
CHAPTER 7: WASTEWATER INFRASTRUCTURE

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Photo: Advanced Onsite Wastewater Treatment System (Estuary Program)

NARRAGANSETT BAY ESTUARY PROGRAM

Overview



BACKGROUND

- Wastewater, if not properly conveyed or treated, is a significant source of pollution to surface water and groundwater. Domestic sewage contains pathogens, nutrients, and other contaminants, such as pharmaceuticals, that can affect ecological resources and human health. Sewer facilities provide the most effective treatment, while individual or community onsite wastewater treatment systems (OWTS) provide more limited treatment. There are still places in the Narragansett Bay Watershed where sewage is discharged to cesspools, which for the most part provide no treatment.

KEY FINDINGS

- ***Status:***

  Wastewater Treatment Facilities (WWTF): Thirty-seven wastewater treatment facilities in the Narragansett Bay Watershed discharged approximately 203 million gallons of treated wastewater per day, based on monthly averages from 2013 to 2015. These plants served approximately 62 percent of the population in the Watershed.

  Onsite Systems (OWTS and cesspools): Thirty-eight percent of the population was served by onsite systems, including septic systems (both conventional and advanced) or cesspools. Eight percent of the population resided in areas with high density of onsite systems. While representing only three percent of the total Watershed area, these areas posed potential concern for water quality impacts. Most areas with high density of onsite systems areas were located along the coastal areas of the Bay, primarily around the Sakonnet River and the West Passage, and areas of the Upper Estuary such as the Bristol and Kickemuit Rivers, Buckeye Brook, and Mount Hope Bay.

Introduction

The population in the Narragansett Bay Watershed produces over 200 million gallons of treated wastewater daily. Regulations governing the discharge of wastewater require treatment prior to disposal to prevent adverse effects to public health and the environment. The release of raw or poorly treated wastewater degrades water quality and results in risks to public health. These releases may occur due to overflows or operational failures of either public or individual wastewater systems. When these situations occur, bacterial and viral pathogens found in raw sewage–in addition to other pollutants–can render waterways and beaches unsafe for recreation and can result in areas being closed to shellfishing. Even with effective treatment, wastewater discharges may be associated with impacts to water quality. In addition, excess nutrients discharged by wastewater infrastructure can cause phytoplankton and algae bloom, reducing light penetration. These bloom die, sink, and are respired by bacteria, contributing to higher biological oxygen demand (BOD), reducing oxygen levels in bottom waters, and harming biota that cannot migrate out of the low-oxygen zone. Additionally, emerging contaminants such as pharmaceuticals and personal care products may pose potential risks to human health and to aquatic organisms. The risks posed by pharmaceuticals and personal care products discharged through wastewater infrastructure are not yet well understood, and currently it is an area of significant research.

Historically, population growth and industrialization led to significant negative impacts on the Bay due to the discharge of raw sewage and industrial wastewaters. Governments in the Narragansett Bay Watershed responded by expanding wastewater collection systems and building treatment works that are among the oldest in the country, including Woonsocket (1897) and Providence (1901) (Rhode Island Division of Planning and CRMC 2016). The proportion of the Upper Bay's population served by sewers increased from fourteen percent in 1880 to 50 percent in 1900, and to 83 percent in 1950, according to Estuary Program calculations of sewered population using data from Vadeboncoeur and colleagues (2010). In the Lower Bay, which includes the islands and coastal areas bordering the Sakonnet River, four percent of the population was sewered in 1890 compared to 31 percent by 1950; these estimates are consistent with the rate of population growth and the increase in urban land use in that period (see "Population" and "Land Use" chapters).

Despite improvements in treatment, the discharge of wastewater was recognized as a major pollution problem during the twentieth century. In the early part of the century, anecdotal evidence suggested that 260 pipes located throughout the Bay transported raw sewage directly to receiving waters (Schumann 2015). A report on water pollution prepared by the Rhode Island Department of Health in 1946 concluded that population growth and increased industrial activity had extended pollution

farther down the Bay than ever before and caused "extensive damage to natural resources". Most notably, the wastewater treatment facility (WWTF) at Field's Point was overwhelmed in 1946, and poorly treated sewage flowed directly into the Bay, resulting in extremely high fecal coliform bacteria concentrations (Shea 1946). By 1970, Field's Point poured 65 million gallons of untreated or inadequately treated sewage into the Bay daily due to lack of flow capacity for a growing sewered population. During that decade, the Rhode Island Department of Environmental Management (RIDEM) closed shellfishing areas in the Upper Bay frequently to protect public health (Schumann 2015).

Recognition of the severity of water pollution problems led to passage of federal and state laws aimed at protecting and restoring water quality. The federal Clean Water Act, passed in 1972, spurred major investment in public wastewater infrastructure, including construction of new WWTFs and the upgrade of existing plants to achieve higher levels of treatment. By the 1980s, industrial pretreatment programs were developed and dramatically reduced the release of toxics into the Bay (see "Legacy Contaminants" chapter).

Contemporary WWTFs provide primary, secondary, or tertiary treatment, followed by disinfection using chlorine or ultraviolet light to ensure effluent permit limitations are being met. The modernization of wastewater treatment, along with operator licensing, improved the performance of WWTFs and drastically reduced the amount of pollution (pathogens, nutrients, biological oxygen demand, toxics) discharged into the Narragansett Bay Watershed.

In some locations, the public wastewater infrastructure can result in combined sewer overflows (CSOs). CSOs result from sewage and stormwater runoff being directed into the same collection system. During rain events, the capacity of the collection system can be exceeded, resulting in combined sewer overflows discharging untreated or partially treated wastewater. The resulting nutrient, pathogen, and contaminant loadings contribute to water pollution problems (see "Nutrient Loading," "Marine Beaches," and "Shellfishing Areas" chapters). In the Narragansett Bay Watershed, combined sewer systems exist in the wastewater systems of the Narragansett Bay Commission (NBC), City of Newport, City of Taunton, City of Worcester, and City of Fall River. CSO abatement actions have been taken within all of these systems. The largest effort involves the NBC, which has completed two phases of a three-phase CSO abatement program (NBC 2017). Infrastructure enhancements have included, but are not limited to, construction of a massive storage tunnel (65 million gallons per day), sewer interceptors, and sewer separation. The NBC's CSO Abatement Project also includes infrastructure modification and improvement in Pawtucket, Central Falls, and East Providence.

Various types of sanitary wastewater infrastructure serve the population in the Narragansett Bay Watershed. For its analysis, the Estuary Program considered two main categories:

1. **Wastewater treatment facilities (WWTFs)** that serve sewered areas, and
2. **Onsite systems** (cesspools and types of onsite wastewater treatment systems [OWTS]: conventional septic systems, advanced septic systems, and small package plants).

Outside of sewered areas, dwellings are required to treat sewage either with septic systems or via locally sited package treatment plants. Under certain circumstances, operation of these systems can result in water pollution problems. Residential septic systems have been linked to nitrogen loading in coastal bays via transport through groundwater (e.g., Valiela et al. 1992). Outdated cesspools and poorly designed or maintained septic systems are known to deliver fecal pathogens and other pollutants in wastewater to groundwater and surface waters. Areas with a high density of septic systems have been shown to contribute more to pathogen loading in waterways compared to low-density areas (Sowah et al. 2017).

While conventional septic systems deliver approximately 24 pounds (11 kilograms) of nitrogen per year to the groundwater, advanced septic systems are designed to reduce nitrogen contributions by 50 to 75 percent through nitrification and denitrification processes that remove nitrogen from wastewater before it enters a leach field (Lancellotti 2016). Within the Narragansett Bay Watershed, advanced septic systems to reduce nitrogen pollutant loadings are required by Rhode Island in the Narrow River subwatershed and other coastal areas of concern in the state (RIDEM 2009). Failing septic systems and cesspools remain a management concern in many areas of the Watershed, as well as in certain locations of the Bay such as Greenwich Bay, northern Portsmouth Harbor, and Wickford Harbor. Pollutant loadings from onsite systems may cause or contribute to water quality degradation that is generally of localized impact. Understanding the effects of onsite systems can be challenging as the geomorphological characteristics and land use in the watershed can alter the groundwater flow and the ultimate delivery of nitrogen to Bay waters (Nowicki and Gold 2008).

One of the most studied areas in the Bay with respect to groundwater inputs is Greenwich Bay, where Urish and Gomez (2004) estimated that 60 to 70 percent of the freshwater inputs from the land into this watershed were derived from groundwater recharge, and most of the nutrients–approximately 65 to 75 percent of loadings entering Greenwich Bay–were from onsite wastewater treatment systems. Likewise, high densities of septic systems have been linked to bacterial pollution in adjacent waters in developed watersheds (Sowah et al. 2017).

In this chapter, the Narragansett Bay Estuary Program presents a watershed-scale analysis of wastewater infrastructure in the Narragansett Bay Watershed. The chapter discusses the coverage and influence of wastewater infrastructure in relation to human population and the potential effects on environmental condition. The chapter does not address the direct discharge of industrial wastewaters, except in the calculation of the average daily flow discharge from WWTFs because it includes the flow of all treated sewage. For its analysis, the Estuary Program developed a set of metrics to quantify: 1) sewered areas served by WWTFs, 2) non-sewered developed areas that were assumed to be served by onsite systems, and 3) total population, as of 2010, that resided within each of the service areas. The analysis used building density as a proxy to estimate the density of onsite systems and to identify areas that may merit higher management concern. In this chapter, the term "onsite systems" refers to both cesspools and OWTS.

Methods

The Estuary Program analyzed wastewater infrastructure throughout the Narragansett Bay Watershed to identify areas served by (a) sewer systems or (b) onsite systems. The total population served by each of those two categories of systems was estimated for the entire Narragansett Bay Watershed, as well as for each of the 42 watershed planning areas (WPAs). In collaboration with representatives from the Massachusetts Department of Environmental Protection (MassDEP) and the Rhode Island Department of Environmental Management (RIDEM), the Estuary Program gathered and reconciled data at the state level to analyze the extent of wastewater infrastructure (Table 1). In addition, partners from EPA assisted in the analysis of population estimates for each of the service areas. Using the data sources and methods listed in Tables 1 and 2, the Estuary Program calculated the following metrics:

- Areas (in acres) served by **WWTFs**
- Areas (in acres) served by **onsite systems**
- Areas (in acres) classified as **high-density areas** served by onsite systems
- Total population and population density (number of people per acre) served by WWTFs versus onsite systems (for all unsewered areas and high-density unsewered areas)

Because of the multiple datasets compiled for this analysis, from different sources, at varying time and geographical frames, these are among the caveats that need to be acknowledged upfront in this section. The array of geospatial analyses performed to quantify the metrics presented above should be considered as a snapshot of current conditions, even though land use is in constant change across the Watershed (see "Population" and "Land Use" chapters), and so are the management plans and implementation of wastewater systems, generally at the local level. Changes in population is the primary factor, as people move to or from existing developed areas or drive new development for dwellings and other types of buildings that require any type of sewage treatment, whether by connecting to existing sewer lines or by adding a new OWTS. In addition, existing buildings that were served by an onsite system are also connecting to sewer lines. All this information is not readily available at the Watershed scale. Consequently, the results in this chapter are spatially oriented because despite the limitations in the data, the Estuary Program was able to reconcile them by geospatial means, but not within a temporal timeframe. The details of data caveats, limitations in the methodology, and assumptions of the results are presented in Tables 1 and 2.

For its analysis, the Estuary Program compiled all the information presented in Tables 1 and 2, reconciled datasets across the state boundaries and agencies, and utilized a watershed approach using the Watershed Planning Areas. However, the analysis included simplifying assumptions that may not have reflected actual conditions and therefore represent limitations in the analysis. In addition, there were inherent constraints of the data sources. For example, a significant number of buildings with access to sewer lines in the Greenwich Bay subwatershed were not yet connected, but this fact was not reflected in the datasets. Also, the state datasets may not have included recent extensions of sewer collection systems.

Table 1. Readily available data at the state level for wastewater infrastructure in the Narragansett Bay Watershed.

Type of wastewater infrastructure information	Data availability and information provided by state agencies to the Estuary Program	
	Massachusetts	Rhode Island
Location of sewered areas served by wastewater treatment plant facilities (WWTFs)	MassDEP estimates of municipal sewered areas (December 2016)[1].	RIDEM sewer service areas (September 2015)[2].
Location of individual onsite systems: conventional septic systems	No statewide coverage. (Data on systems treating less than 10,000 gallons per day maintained by municipal boards of health and outside the scope of this report.)	Location information from RIDEM (October 2015) including plat and lot numbers and limited street addresses. Emergency 911 data used to match 46 percent of the sites. Data included systems that were approved, installed, and under construction; however, some properties may have been connected to the sewer system.
Location of individual onsite systems: advanced septic systems	Data not readily available.	RIDEM data included 64 percent of all advanced septic systems within portion of the state in the Watershed, as estimated by the Estuary Program (October 2015).
Cesspools	Data not readily available.	Data not readily available.
Areas served by wastewater treatment systems that discharge to groundwater over 10,000 gallons per day	MassDEP digitized parcels served by permitted treatment plants (2016).	RIDEM data on total flow by location is available, but areas of service were not available (October 2015).
Average daily flow discharged by WWTFs[3]	Discharge Monitoring Report (DMR) from EPA (between 2013 and 2015).	Discharge Monitoring Report (DMR) from RIDEM (between 2013 and 2015).
Population served by WWTFs[3]	WWTF permit by EPA (timeframe varies)	WWTF permits and RIDEM Wastewater Treatment Facility web page

[1] While MassDEP utilized the most of up-to-date sources of information to create these datasets, by municipality, the data only depicted the approximate coverage of service areas by WWTFs; because reports were produced at varying timeframes, these areas should be considered as present conditions as of 2016, when MassDEP provided data to the Estuary Program.
[2] Data developed by RIDEM represented all areas in the State expected to be, or currently defined as, sewer service areas. These areas do not depict the exact extent of buildings that are served by WWTFs; at the sewered-area scale, it is important to note that there are still buildings within the sewer service areas that are not connected to the sewer lines and continued to rely on OWTS. This level of analysis was not conducted to develop this indicator given the limitations of the data, and scale, where the results of this analysis are at the Watershed and Watershed Planning Areas scales.
[3] Data were downloaded from https://cfpub.epa.gov/dmr/ or provided by RIDEM for WWTFs in Rhode Island.

Table 2. Data sources, processing methods, and assumptions for calculating area, average flow, and population served by wastewater infrastructure in the Narragansett Bay Watershed.

Type of Information	Data Processing	Assumptions
Sewer Service areas by:		
a) *Wastewater treatment facilities (WWTF) discharging to surface waters*	Integrated data from both states. Merged MassDEP and RIDEM sewer service areas. MassDEP manually digitized sewer services areas for each one of the municipalities within Narragansett Bay Watershed, based on information from municipal reports, among other sources, to provide data for this chapter.	• Assumed all buildings within the sewered areas were connected to the sewer lines using a buffer area around existing or planned sewer lines. • Assumed that state data represented current information on extent of sewered areas, as of 2016 for areas in MA, and 2015 for RI.[1] • Assumed service for entire municipality in Massachusetts, when the spatial extent was unknown but the information provided by MassDEP indicated that the municipality had a service area for greater than 80 percent of buildings (e.g., Fall River and Somerset).
b) *Wastewater treatment facilities (WWTF) discharging to groundwater*	Where information was available, included parcel areas as part of the sewered areas for systems with groundwater discharge over 10,000 gallons per day (only for Massachusetts).	As mapped by MassDEP, these systems were considered as WWTF given that these systems provide greater treatment than an advanced septic system.
Location of buildings served by onsite systems	Rhode Island: Building structure point locations, using E911 sites[2], outside of sewered areas used to estimate unsewered buildings. Massachusetts: Eliminated polygons of buildings (corresponding to approximate area of garages or sheds) less than 50 square meters (538 square feet), using building structures[3], and converted each polygon as point data.	• Assumed each point corresponds to a building that generates domestic sewage. • Assumed each point discharges to an onsite system, either individual or shared OWTS or a cesspool; that buildings have not been connected to the sewer system. • Estuary Program did not distinguish between buildings that may be discharging to an OWTS versus a cesspool due to the lack of data at the watershed scale; grouped them as "onsite systems."
Density of estimated buildings served by onsite systems.	Point data for unsewered buildings were used to calculate density of points (number of buildings per hectare [2.5 acres]) within a defined distance (250 meters; 820 feet), also referred as neighborhood areas, using the Kernel Density Analysis tool in ArcGIS. Areas were classified from low density (ten percent higher than the rest of the Watershed) to high density (90 percent higher).	• Assumed that 250-meter (820-foot) search radius corresponded to average distance between houses in high density residential areas. • Assumed that areas with a high density of onsite systems were accurately characterized by using the 90 percent threshold. • Because it is impossible to reconcile the array of datasets that have been created at different timeframes, within one temporal scale, the results of this analysis should be considered at face value and a snapshot of present conditions as of 2016.

Table 2 continued.

Type of Information	Data Processing	Assumptions
Population (Census 2010) served by WWTFs	• Population distribution data (dasymetric analysis in "Population" chapter) were intersected with sewered areas to calculate population (residents)[4] and percentage served by WWTFs. • Population within sewered areas were further intersected with Watershed Planning Areas (WPAs) to calculate total resident population within each of the 42 WPAs. • Total population and population density were computed.	• Assumed that number of residents based on census data accurately reflected population served within the sewered areas. However, census data do not include people who reside outside the area but work, recreate, worship, or engage in other activities there. Therefore, the Estuary Program's analysis focuses on residential areas served by WWTFs.
Population served by WWTFs[5]	Estimates of population served by each WWTF were available for 90 percent of all WWTFs discharging to surface waters.	• Assumed that this included population served at other types of land use buildings (institutional, commercial, industrial, etc.) in addition to residential. • The population is calculated as the total served by WWTFs that are located within River Basins, in other words what that total population generates as sewage per capita, but it does not represent population across sewered areas by River Basins, since service areas span over different River Basins; and the data was not available by geospatial means, hence the difference with the previous method.
Population (Census 2010) served by onsite systems	Same as above[4], except the distribution of the population was intersected with the areas representing density of onsite systems. Total population and population density residing in areas with a high density of onsite systems were calculated accordingly.	• Assumed distribution of population within the areas of a high density of onsite systems intersected with areas where people reside. • Assumed that buildings served by onsite systems were moderately to highly populated within areas of a high density of onsite systems.
Population by watershed planning area (WPA) for sewered areas, onsite system areas, and areas with a high density of onsite system.	Intersected sewered areas, onsite system areas, and population distribution with total area (acres) of WPA. Total population and population density were computed for each of the 42 WPAs.	• Assumed population distribution represents total sewered population within a WPA, even when WWTF discharge or outfalls are outside the Watershed or WPAs. • Assumed that WPAs with a large percentage of areas with a high density of onsite systems pose risk to surface and groundwater based on literature.

Table 2 continued.

Type of Information	Data Processing	Assumptions
Estimated daily average flow from WWTFs in millions of gallons per day (MGD)	Total average daily flow, from 2013 to 2015, for each WWTF that discharged directly to the Bay or rivers in the watershed. WWTFs discharging into Rhode Island Sound were not included.	• These average daily flows were calculated by the Estuary Program and were not the same as the *permitted average daily flow* for each WWTF. • Assumed WWTF average daily flow was representative of both wet and dry years between 2013 and 2015. Available data used from the monitoring periods; not all WWTFs had data for all years or seasonal flow.
Exclusion	Removed lakes and ponds from the above data, in order to create a "wall" which forces the Kernel Analysis to not take account the area within these waterbodies, limiting the geospatial calculations to areas that do have buildings.	• Other areas outside the areas served by WWTFs and onsite systems were assumed to be undeveloped with no buildings or residents within them (i.e., open space areas without buildings); therefore, the calculation of total area does not take into account other areas that do generate sewage.

[1] The temporal context of these datasets represents what the Estuary Program knew as of 2015 for Rhode Island and 2016 for Massachusetts, but it does not reflect the exact extent of sewered areas for these two years. Data on sewered areas are not gathered or updated frequently by the state agencies. For example, MassDEP created data, town by town, relying on the most recent and available information at the local level, which included updated reports or old reports that have not been updated. However, sewered areas for towns within the Narragansett Bay Watershed were the first to be created for Massachusetts, a starting point for the agency. Metadata presenting sources and the process of this data development is in progress.

[2] Downloaded E911 sites from RIGIS in 2016; point data represents the locations of all known buildings with an address in the state of Rhode Island. Because data have been updated since, the original GIS data or metadata for this version are not available online as they were replaced with the updated version of the data. The updated version's E911 sites, definitions, attributes, or metadata may no longer be applicable to the data used for this analysis.

[3] Downloaded Building Structures from MassGIS in 2016; polygon data represents roofprints of all known buildings in the state of Massachusetts. Because data have been updated since, the original GIS data or metadata for this version are not available online as they were replaced with the updated version of the data. The updated version's Building Structures, definition, attributes, or metadata may no longer be applicable to the data used for this analysis.

[4] Population by means of dasymetric analysis is different than total population represented as Census Blocks. Dasymetric analysis distributes people across residential areas (by means of land use) into equal fine scale areas (total people for each cell of 30 meters by 30 meters), whereas Census Blocks are defined areas by US Census Bureau that are not uniform and quite bigger in area, and given a total population regardless of where residential areas are located.

[5] Total population served by each individual WWTF was extracted from NPDES permits, which may factor in people utilizing sewered buildings other than households, therefore it should not be directly compared with the total population by means of Census data (2010) and dasymetric analysis.

The results of the analyses can be considered at the following timeframes:

- Sewered and unsewered areas are representative of 2015 conditions.
- Sewered and unsewered population are representative of 2010. Data on population distribution using Census 2010 was integrated within each of the two service areas.
- Average daily discharge flow from WWTFs to surface waters is representative of 2013 through 2015.

Status and Trends

Historically, sewer systems began to be developed in urbanizing areas in the late 1800s. By the turn of the nineteenth century, the sewered population in the Narraganset Bay Watershed had increased thirteenfold (Figure 1). During the twentieth century, 55 to 65 percent of the Watershed's total population was served by sewer systems (Figure 2; Vadeboncoeur et al. 2010).

Based upon the 2010 census data, it is estimated that sewers served 62 percent of the population in the Narragansett Bay Watershed, with the total sewered area constituting 24 percent of the total Watershed area (Table 3; Figure 1). The sewered portion of the Narragansett Bay Watershed is served by a total of 37 WWTFs. Onsite systems served an estimated 38 percent of the Watershed's population, based on the 2010 census, and the area served by onsite systems encompasses 44 percent of the total area of the Watershed (Table 3; Figure 4).

WASTEWATER TREATMENT FACILITIES (WWTFs)

Based on calculations of actual discharge flow from WWTFs from 2013 through 2015, the 37 WWTFs in the Narragansett Bay Watershed discharged an estimated 203 million gallons of treated sewage per day directly into the Bay and into rivers of the Watershed (Table 4). Twelve of the fifteen WWTFs located within the Coastal Narragansett Bay Basin discharge more than half of the daily discharge directly to Narragansett Bay–103 million gallons per day–not including nine million gallons per day from three WWTFs, discharging to the Ten Mile River and Woonasquatucket River. The next largest volumes were discharged into the Blackstone River (40 million gallons per day) and the Taunton River (30 million gallons per day) (Table 4).

The twelve WWTFs that discharged directly to Narragansett Bay had approximately the same average combined daily flow (103 million gallons) as

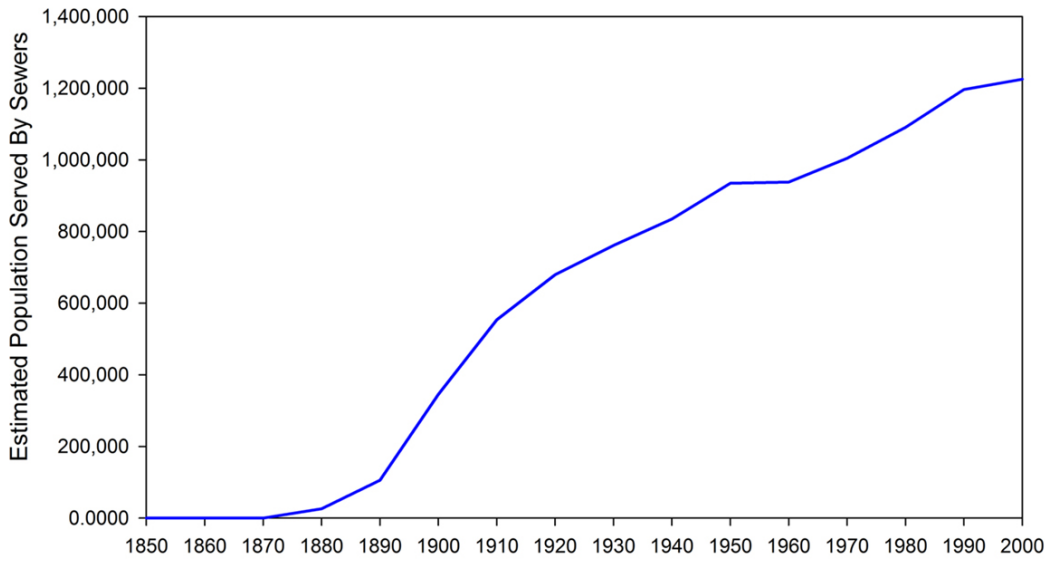


Figure 1. Total population and percentage of population in the Narragansett Bay Watershed served by sewers from 1880 to 2010. Sewered population data from 1880 to 2000 from Vadeboncoeur and colleagues (2010).

Table 3. Areas and population served by sewers and onsite systems in the Narragansett Bay Watershed.

Areas Served by Wastewater Infrastructure	Service Areas		Population[1]		Population Density (People/Acre)
	Acres	Percent of Watershed[2]	Total	Percent[3]	
Wastewater treatment facilities	262,566	24	1,213,034	62	4.62
Onsite systems[4]	483,888	44	731,185	38	1.51
High-density onsite systems[5]	*31,762*	*3*	*146,363*	*8*	*4.61*

[1] Based on Census 2010 by means of dasymetric analysis (see "Population" chapter).
[2] Of the total area of Narragansett Bay Watershed; the remaining 32 percent not listed here included undeveloped areas and open freshwaters.
[3] Of total population in the Watershed.
[4] Including all areas with buildings assumed to be served by onsite systems.
[5] Part of the total area with onsite systems; see text for definition.

Table 4. Average daily flow in millions of gallons per day (MGD) discharged by WWTFs into Narragansett Bay or rivers in the Narragansett Bay Watershed, and estimated population served. Text in *italics* represent the totals for WWTFs discharging into waters in the riverine system.

Number of WWTFs	WWTFs are located within these River Basins:	Receiving Waters:	Average Daily Flow (MGD)[1]	Population Served[2]
12	Coastal Narragansett Bay Basin	Discharging directly to Narragansett Bay	103	609,364
25		Discharging to rivers and tributaries	100	735,008
10	Blackstone River Basin	*Blackstone River*	*40*	*294,935*
3	Taunton River Basin	*Taunton River*	*30*	*206,559*
9	Pawtuxet River Basin	*Pawtuxet River*	*21*	*165,000*
2	Coastal Narragansett Bay Basin	*Ten Mile River*	*7*	*54,514*
1	Coastal Narragansett Bay Basin	*Woonasquatucket River*	*2*	*14,000*
37	**Total in Narragansett Bay Watershed**	**Total to Narragansett Bay**	**203**	**1,344,372[3]**

Sorted by Average Daily Flow from the largest to smallest discharge flow.
[1] Based on data from 37 WWTFs between 2013 and 2015.
[2] Estimated based on 34 WWTF permits; three WWTFs did not have population estimates.
[3] Difference of the total population using dasymetric analysis and the population as estimated by the WWTF permits was 10 percent; this percent is likely larger given the lack of data as mentioned in (2).

the remaining 25 WWTFs that discharged to rivers (100 million gallons) (Table 4). WWTFs discharging to rivers served 735,008 residents, which was 17 percent more than the 609,364 residents served by WWTFs discharging to the Bay (Table 4).

Of all 37 WWTFs, the two with the highest average daily flows were the Field's Point WWTF, which discharged 41.1 million gallons per day into the Providence River Estuary, and the Upper Blackstone WWTF, which is adjacent to Worcester and discharged 29.1 million gallons per day into the Blackstone River (Figure 1; Table 5). These two WWTFs combined serve 35 percent of the total sewered population in the Narragansett Bay Watershed (Figure 1; Table 5).

Of the 42 watershed planning areas (WPAs) in the Narragansett Bay Watershed, ten had 75 percent or more of their populations served by sewer systems (Table 6; Figure 2). In the Providence-Seekonk Rivers WPA, Moshassuck River WPA, and Tatnuck Brook-Blackstone River WPA, over 95 percent of the population within each WPA was connected to sewer systems (Table 6).

Table 5. Average daily flow and population served by WWTFs discharging over 10 million gallons per day to Narragansett Bay and its Watershed.

Wastewater Treatment Facility	Receiving Waterbody	Average Daily Flow (MGD)	Population Served[1]
Field's Point	Narragansett Bay[2]	41.1	226,000
Upper Blackstone	Blackstone River	29.1	207,535
Fall River	Narragansett Bay	20.8	92,164
Bucklin Point	Narragansett Bay[2]	19.2	120,000
Brockton	Taunton River	15.2	126,256
Cranston	Pawtuxet River	11.5	73,200

[1] Based on estimates by individual WWTFs from DMRs; see Tables 1 and 2 for data source and methods for calculation.
[2] The Field's Point and Bucklin Point WWTFs are both located in the Coastal Narragansett Bay Basin. They discharge to the estuary of the Providence River and Seekonk River, respectively.

Table 6. Watershed planning areas (WPAs) in which 75 percent or more of the population was served by sewer systems.

Watershed Planning Areas[1]	Percent of Population Sewered[2]	Total Sewered Population (2010)[3]	Percent of WPA Area Sewered[4]
Providence-Seekonk Rivers	99	126,335	98
Moshassuck River	96	80,397	81
Tatnuck Brook-Blackstone River	96	133,874	71
Barrington-Palmer-Warren Rivers	88	17,864	12
Greenwich Bay	84	38,254	77
Buckeye Brook	84	14,289	79
Stafford Pond	83	41,311	27
Woonasquatucket River	70	98,613	33
Matfield River	76	82,487	47
Mount Hope Bay	75	27,314	53

[1] See the Appendix for WPA code numbers and map.
[2] Percent of the total population in the WPA.
[3] Based on population estimates from dasymetric analysis; see "Population" chapter.
[4] Percent of the WPA area not including lakes or ponds.

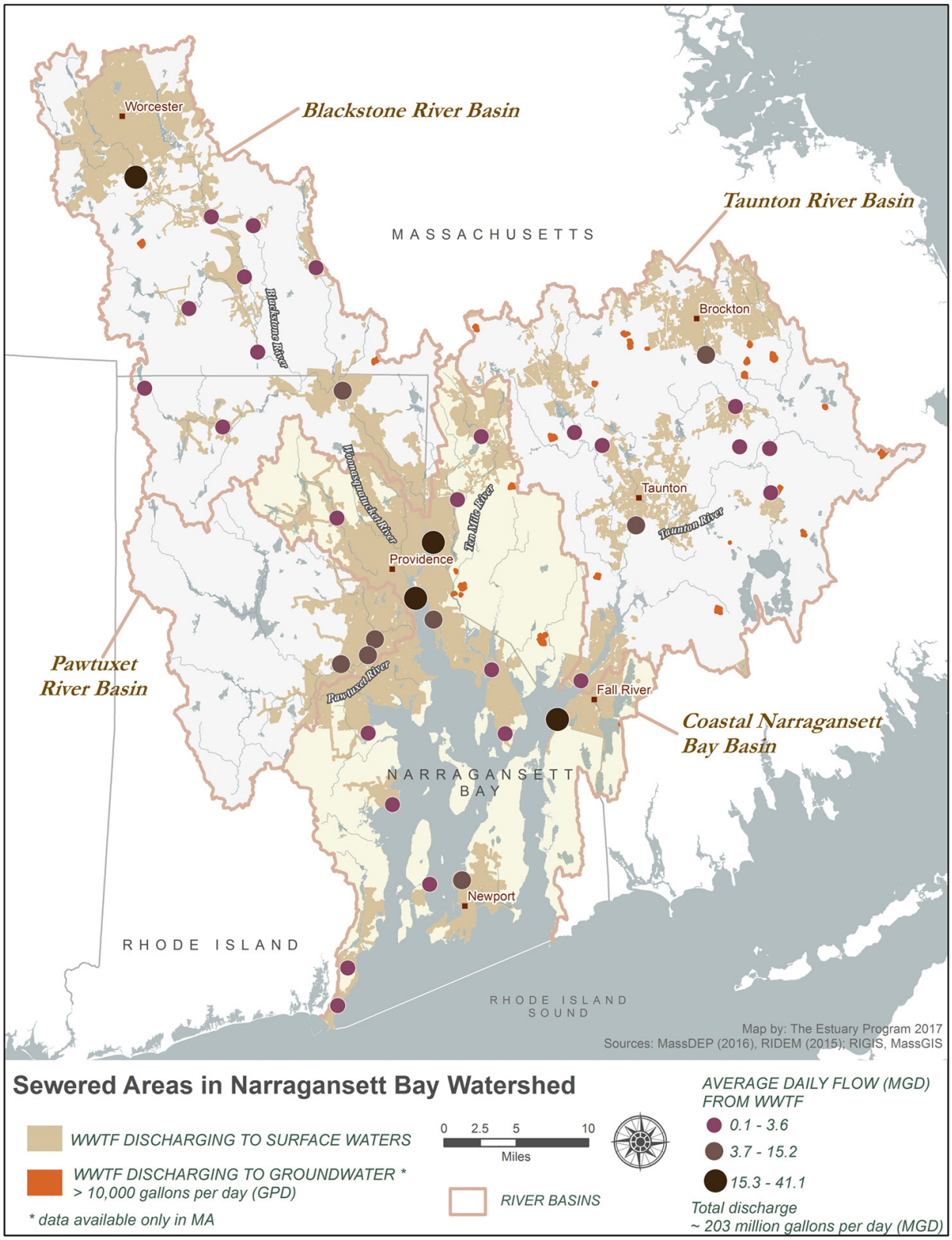


Figure 2. Extent of sewer service areas served by wastewater treatment facilities (WWTFs) in the Narragansett Bay Watershed.

It is important to emphasize that the total sewered population from Table 3 represents the distribution of people residing within the boundaries of the Watershed and does not consider people who commute to or otherwise use amenities from commercial, industrial, institutional, or other non-residential buildings. In contrast, the results in Table 4 and 5 represent the estimates of population served by each WWTF as presented in their permits. The estimates of population served for each WWTF may include people who reside outside the Watershed but who are part of the Watershed's transitory population; geospatial data do not exist or are not readily available to characterize this component of the WWTF estimates of population served. Yet, the difference of total population estimates was only ten percent.

ONSITE SYSTEMS (OWTS AND CESSPOOLS)

In the Narragansett Bay Watershed, an estimated 731,185 people from the 2010 census–38 percent of the population–were served by onsite systems (Table 3). Three percent of the Watershed's acreage was identified as areas with high densities of onsite systems, and eight percent of the Watershed's population resided in those areas (Table 3). Twenty percent of the total area serviced by onsite systems was considered high density (Table 7). The Estuary Program defined high-density areas of onsite systems as those having building densities at least 90 percent higher than the average building density across the Watershed (Figure 3).

There were ten WPAs in which areas with high density of onsite systems constituted five percent or more of the total WPA area (Table 8). On Aquidneck

Table 7. Watershed planning areas (WPAs) in which at least 20 percent of the population served by onsite systems lived in areas with high density of onsite systems.

Watershed Planning Areas (WPAs)[1]	Population residing within areas served by onsite systems[2]			
	Total [3]	Areas with high density of onsite systems		
		Of the total[4]	Percentage of the total[5]	Population density[6] (People per acre)
Buckeye Brook	2,780	2,063	74	6.4
Aquidneck Island	22,216	12,601	57	4.3
Sakonnet East	11,952	6,356	53	5.1
Prudence Island	216	112	52	1.0
West Bay	12,979	5,544	43	4.2
Pawtuxet River	65,441	26,495	40	5.1
Jamestown	3,464	1,357	39	3.3
Bristol-Kickemuit River	20,034	7,082	35	5.2
Hunt River	16,280	5,125	31	4.8
Stafford Pond	8,126	2,351	29	3.5
Matfield River	26,380	6,845	26	4.6
Greenwich Bay	7,236	1,799	25	4.6
Annaquatucket River	4,460	1,083	24	3.6
Barrington-Palmer-Warren Rivers	21,032	4,538	22	4.5
Total in Narragansett Bay Watershed	**731,185**	**146,363**	**20**	**4.61**

[1] See the Appendix for WPA code numbers and map.
[2] Calculated population distribution using dasymetric analysis (see "Population" chapter) within onsite systems areas.
[3] Population within areas served by onsite systems.
[4] Population within areas of high-density onsite systems.
[5] Formula: ((Population within areas of high-density onsite systems)/(Total population within areas served by onsite systems)*100).
[6] Formula: (Population within areas of high-density onsite systems)/(Acres of areas with high-density of onsite systems).

Island, for example, 2,949 acres of the WPA had high-density onsite systems, representing twelve percent of the total watershed planning area. Population density ranged from 3.3 to 6.4 people per acre in areas of high-density onsite systems (Table 8). Many of the areas with high density of onsite systems were located in coastal areas of Narragansett Bay, including the Sakonnet River, Mount Hope Bay, and the West Passage (Figure 3).

In addition, because data on advanced septic systems were available only for Rhode Island, the Estuary Program quantified the number of advanced septic systems by means of geospatial analysis. Not all entries in the RIDEM database were mappable. There were 2,035 advanced septic systems in the Rhode Island portion of the Watershed that were mapped and identified spatially, these included systems that had been installed, approved, or under construction by 2015 (see inset map in Figure 3). These advanced OWTS are designed to efficiently remove nitrogen loadings before outflow enters the underlying soils, thus reducing impacts associated to high levels of nitrogen in estuarine waters by means of groundwater flow, either directly draining to the estuary, or into receiving freshwaters, that also flow into the Bay. However, the advanced OWTS implemented in Rhode Island are not designed to further remove other contaminants found in conventional sewage such phosphorus or pathogens that are associated to impacts in freshwaters, affecting aquatic life or human health. The complete results for the 42 WPAs are available upon request. Data were not available to determine temporal trends for population served by onsite systems.

Discussion

The total population in the Narragansett Bay Watershed has increased 85 percent since 1850 and 25 percent since 1950, driving urbanization and generating increasing amounts of household and industrial sewage. Since 1910, more than 50 percent of the people residing in the Watershed have been served by sewer systems and wastewater treatment systems (Figure 2). In the Estuary Program's analysis, 62 percent of the nearly 1,950,000 people residing in the Watershed based on the 2010 census were served by sewer systems. The remaining 38 percent of the population was served by various types of onsite systems. Sewered areas encompassed approximately 24 percent of the Watershed area, and 44 percent of the area was served by onsite systems. The remaining 32 percent of the Watershed area was not developed.

Table 8. Watershed planning areas (WPAs) in which areas of high-density onsite systems represented at least five percent of the total WPA area, and population served.

Watershed Planning Areas[1]	Areas with high density of onsite systems		Population residing within areas of high-density onsite systems	
	Acres[2]	Percent[3]	Percent population of total WPA [4]	Population density[5] (People/acre)
Aquidneck Island	2,949	12	22	4.3
West Bay	1,324	9	30	4.2
Buckeye Brook	324	7	12	6.4
Sakonnet – East	1,252	7	16	5.1
Jamestown	415	7	25	3.3
Hunt River	1,071	7	31	4.8
Annaquatucket River	304	6	24	3.6
Pawtuxet River	5,207	6	14	5.1
Matfield River	1,498	6	6	4.6
Bristol-Kickemuit River	1,360	5	16	5.2

[1] See the Appendix for WPA code numbers and map.

[2] Total acreage of areas with high-density of onsite systems within WPA.

[3] Percent of WPA acreage with high-density of onsite systems.

[4] Formula: (Total population within areas of high density onsite systems)/(Total population of WPA).

[5] Formula: (Total population within areas of high density onsite systems)/(Acres with high-density onsite systems per WPA).

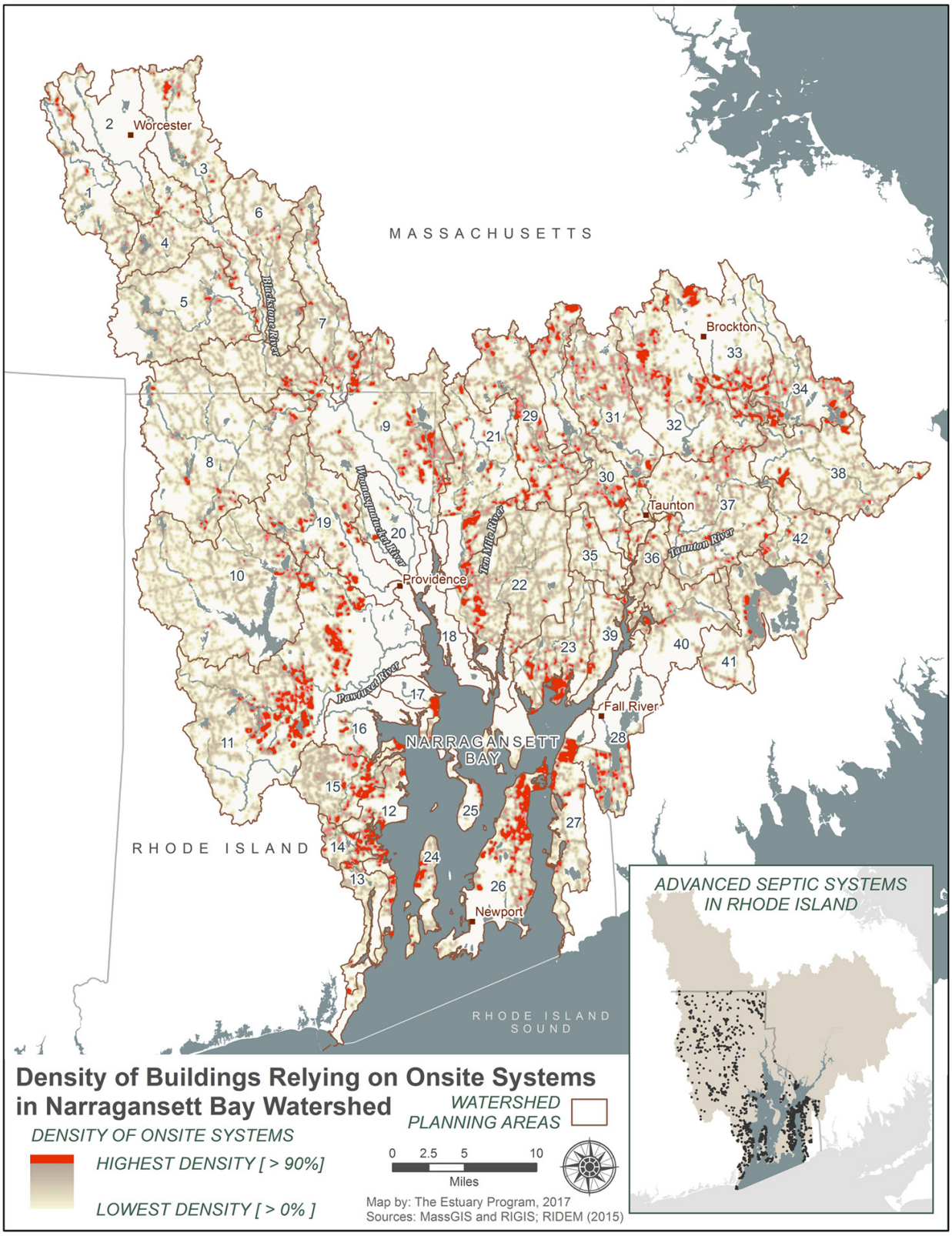


Figure 3. Density of buildings that are estimated to be served by onsite systems. Densities are represented by areas from lowest to highest density of onsite systems. Inset map shows the location of advanced septic systems in the Rhode Island portion of the Watershed. See the Appendix for a complete list of WPA names and ID codes.

Looking ahead, the performance of public wastewater infrastructure will require a commitment to proper ongoing maintenance, repair, and replacement, including mechanisms to provide adequate funding for the needed work. Wastewater systems face increasing threats to their operations due to rising sea level and flooding. WWTFs are sited in low-lying areas to take advantage of gravity in conveying wastewater to the facility, which makes them vulnerable to the impacts of sea level rise. A recent RIDEM study examined the potential risks to nineteen municipal/quasi-state WWTFs in Rhode Island and projected that six of them located on the coast would be predominantly inundated under one to five feet of sea level rise (Woodard and Curran 2017; see "Sea Level" chapter).

This chapter identified areas with high density of onsite systems, which have higher potential for water quality impacts. The greatest risks are associated with cesspools and improperly functioning septic systems. These areas of high-density onsite systems may be important to target for enhanced management and research.

A comprehensive and quantitative assessment of onsite systems has not been done to distinguish between OWTS as conventional and advanced septic systems, and cesspools, for impacts on Bay and Watershed water quality. The challenge lies in the limitations in readily available information on the location and type of OWTS and cesspools in Massachusetts and Rhode Island, and the scale, varying between at the local and state level respectively. However, the density analysis in this chapter points to areas of higher potential for water quality impacts from the onsite systems, particularly from septic systems that may be providing limited treatment of contaminants before being discharged to the groundwater and cesspools that are not designed to treat sewage and are only serving as collection systems.

Sowah and colleagues (2017), studying in Georgia, correlated areas of high-density septic systems with increased pathogen loadings in waterbodies. This type of analysis could also provide a surrogate for nitrogen loadings, primarily for OWTS that are conventional systems. At the scale of watershed planning areas, additional planning efforts could provide a more comprehensive understanding of the total flow discharged through OWTS to specific areas of concern such as surface freshwaters or coastal waters.

Advanced septic systems that efficiently remove nitrogen are required in Rhode Island in certain areas associated with nitrogen-enriched waters. In the Rhode Island portion of the Narragansett Bay Watershed, the Estuary Program calculated that nearly 1,500 advanced systems had been constructed, over 250 systems had been approved but not yet constructed, and over 250 systems were under construction as of November 2015 (RIDEM personal communication 2015). The advanced systems that had been installed were clustered primarily in coastal areas in the towns of Portsmouth, Jamestown, Tiverton, and North Kingstown, mainly within areas of high-density onsite systems. Researchers at the University of Rhode Island recently examined the performance of advanced nitrogen-removing OWTS in Rhode Island and found that these systems are capable of lowering total nitrogen in the effluent to meet regulatory standards (Amador et al. 2017). This type of study—including conventional and advanced septic systems, package plants, and cesspools, and encompassing the range of OWTS densities and soil/geological/groundwater conditions within the Watershed—would be especially valuable for quantifying pollutant loading (e.g., nutrients, pathogens, contaminants) to the Bay and Watershed.

These types of local analyses in the Watershed are valuable for understanding the approximate nitrogen loadings from both conventional and advanced OWTS at different watershed scales. The data and methods used by the Estuary Program complement these studies by characterizing areas and population served by OWTS and cesspools. The nitrogen loading from OWTS (conventional and advanced) in the Narragansett Bay Watershed could be estimated assuming attenuation factors. Better data on locations of onsite systems, including the types of systems, would make it possible to further evaluate the impacts, including those associated with other pollutants in wastewater such as phosphorus, which can adversely impact aquatic life in freshwaters. Under adverse conditions such as improper design, groundwater depth, proximity to shoreline of surface waters, and other geological characteristics, OWTS can contribute to excess phosphorus loadings.

Data Gaps and Research Needs

- To improve data quality, a more systematic means of periodically updating public sewer service information should be developed, and the information should be made easily accessible and shareable. It should include data on buildings and population that have been connected to the sewer systems over time.
- There is a need to improve the capacity to compile data from state (Rhode Island) and local (Massachusetts) records to map the

locations and types of onsite wastewater treatment systems (OWTS), including traditional and advanced systems and cesspools. Data should include buildings that have converted from cesspools to conventional or advanced septic systems, or from conventional to advanced septic systems. This information would allow for further analyses related to water quality and climate change vulnerabilities.

- To address the above data gaps, one option that can be standard and trackable, for both sewered areas and onsite systems, is to include in the parcel data an attribute or attributes that define the type of sewage treatment, across all towns within the Watershed, when parcel data are updated.
- There is a need to integrate other readily available data such as soils, natural buffers, streams, and land use, among others, to identify whether groundwater at areas where onsite systems are estimated to be located, based on the preliminary results in this chapter, is likely at higher or lower risk of sewage contamination due to soil properties, proximity to resources of concern, or other constraints. A study similar to the one by Sowah and colleagues (2017) should be replicated in the Narragansett Bay Watershed to develop more robust mapping and information related to high-density onsite systems and their effects on water quality for aquatic life and human health. The Estuary Program has already advanced in this research need, by engaging soil scientists in both Massachusetts and Rhode Island with the US Department of Agricultural, Natural Resources Conservation Service (USDA-NRCS) to start compiling soil data and properties to develop a suitability map for the Watershed.
- There is limited data analysis on groundwater across the Watershed, except for areas in Greenwich Bay. This is an outstanding data need that is imperative for understanding groundwater direction, flow, and attenuation, and other factors that can provide a more complete picture of the risks of sewage contamination to surface waters or the Bay, via onsite systems, whether septic systems or cesspools. Alternatively, or while methods are developed for groundwater monitoring, other approaches can be undertaken, such as coordinating with partners at the University of Connecticut to follow their methods to start gathering information about groundwater inputs to the Bay, and consequently assess the impacts of onsite systems to public health (due to pathogen loadings, primarily) and habitat (due to increase of nitrogen or phosphorus loadings to freshwaters and the estuary).
- Additional data on the performance of advanced treatment OWTS should be collected. Analysis of data should be completed to evaluate whether advanced systems are achieving expected treatment efficiencies during actual use.
- Improved field studies and models to estimate nutrient and pathogen loadings from onsite systems are needed to quantify and evaluate the impacts on streams and embayments, such as Greenwich Bay.

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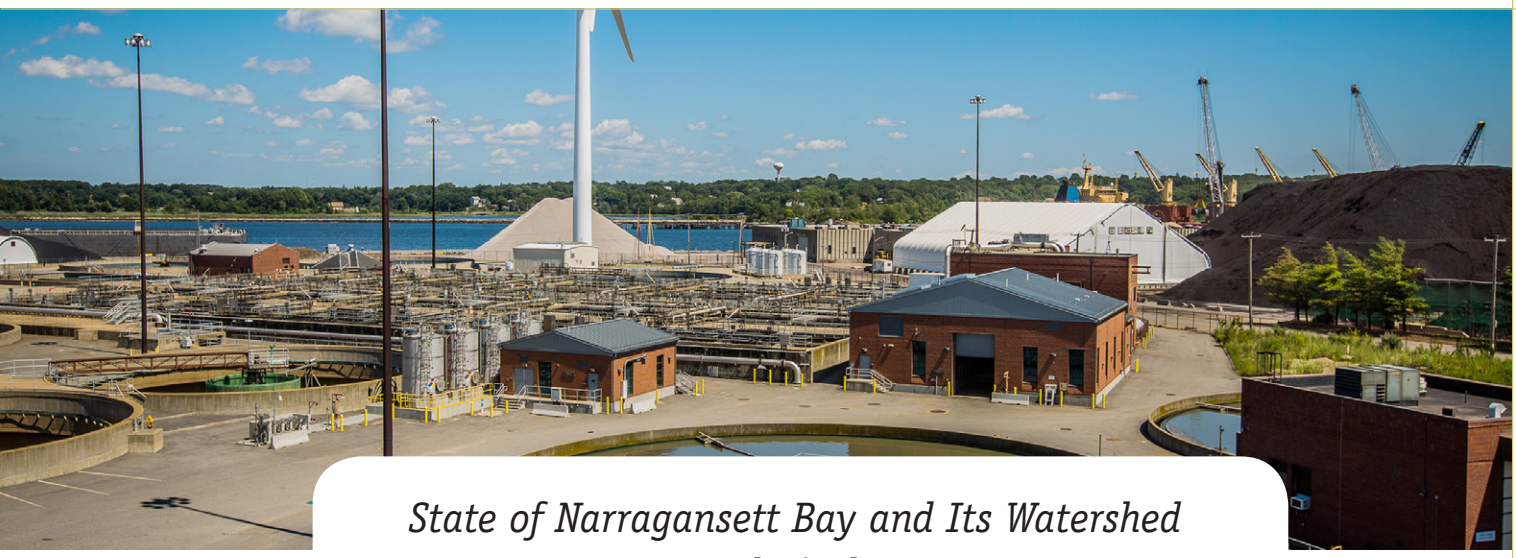
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State of Narragansett Bay and Its Watershed
2017 Technical Report

Landscape Stressor Indicators

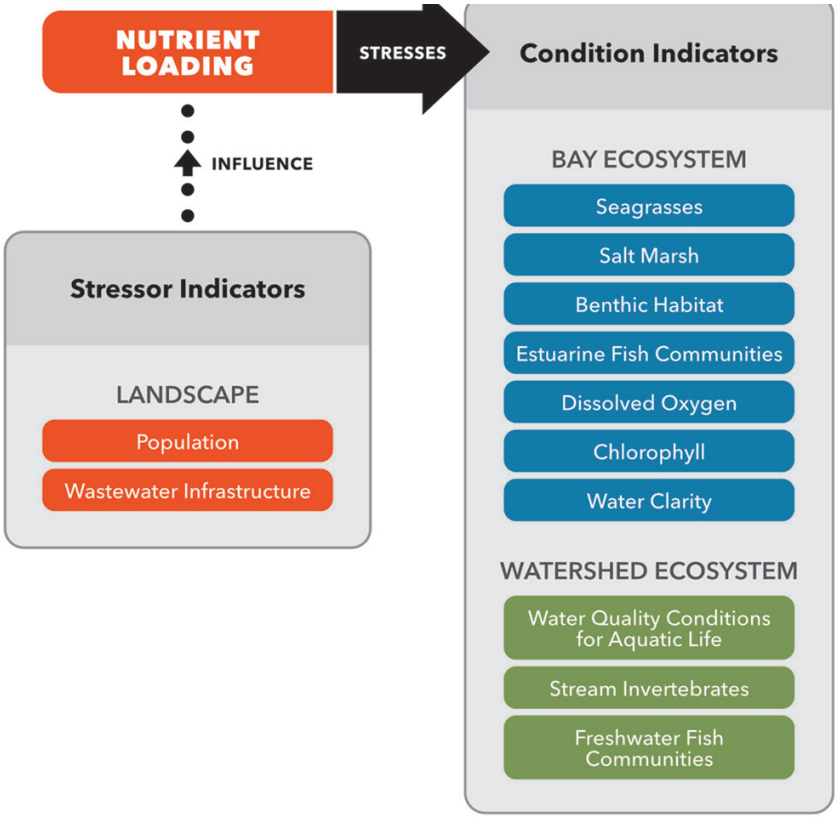
CHAPTER 8: NUTRIENT LOADING

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NARRAGANSETT BAY ESTUARY PROGRAM

Photo: Narragansett Bay Commission, Field's Point, Providence, RI (Ayla Fox)

Overview



BACKGROUND

- Nutrient loading refers to the input of nutrients into the ecosystem from numerous anthropogenic and non-anthropogenic sources. Population growth and development, particularly wastewater infrastructure, have increased the amount of nutrient loading to the Narragansett Bay Watershed. Excessive nutrient loading is a stressor to freshwater and estuarine ecosystems, affecting primary production of phytoplankton, macroalgae, and freshwater and estuarine plants. It also indirectly affects higher levels in the food web (e.g., stream invertebrates, benthic biota, and estuarine and freshwater fish communities) and biogeochemical and physical processes within the water column (e.g., dissolved oxygen, water clarity, and water quality).

KEY FINDINGS

- ***Status:*** During 2013 to 2015, a total of 5,569x10^3 pounds per year of nitrogen and 762x10^3 pounds per year of phosphorus were discharged from 37 wastewater treatment facilities located throughout the Watershed. Riverine loading totaled 4,688x10^3 pounds

per year of nitrogen and 712x10^3 pounds per year of phosphorus, which included the impacts of 2,633x10^3 pounds of nitrogen and 196x10^3 pounds of phosphorus from 25 wastewater treatment facilities discharging to the rivers during the same time period.

- ***Trends:*** Over the last fifteen years, management policies and significant investments in wastewater facilities have reduced the amount of nitrogen and phosphorus loadings to the Watershed. A comparison of nutrient budgets from 2000–2004 and 2013–2015 revealed a 55 percent decrease in wastewater treatment facility loadings throughout the Watershed in total nitrogen and a 45 percent decrease in total phosphorus. This same comparison showed a 62 percent decline in total nitrogen and a 78 percent decline in total phosphorus loadings from the rivers. Over the last 30 years, since 1982–1983, total nitrogen and phosphorus loadings to the Watershed have decreased 55 percent and 57 percent, respectively.

Introduction

Sufficient nutrient levels in freshwater and estuarine ecosystems are essential for primary producers such as aquatic plants, macroalgae, and phytoplankton to photosynthesize and maintain the base of the food web. Nitrogen (N) and phosphorus (P) are limiting nutrients, meaning that the availability of these nutrients in a waterbody regulates the amount of primary production that occurs. Nitrogen is typically limiting in estuarine waters, and phosphorus in fresh waters. An overabundance of these nutrients can lead to an unusually large increase in primary production within the ecosystem (see "Chlorophyll" chapter), otherwise known as eutrophication (Nixon 1995). Eutrophication can lead to negative impacts including reduction in light penetration from excessive macroalgae and phytoplankton blooms, and reduction of dissolved oxygen levels through increased respiration and decomposition. The reduced light penetration can cause a loss of seagrass, and organisms depending on seagrasses may be negatively impacted. If dissolved oxygen levels decline to extremely low levels, marine or aquatic organisms die or leave the area (see "Dissolved Oxygen" chapter). Trends in nutrient loading are an important indicator of ecosystem condition, given the potential for cascading negative impacts on Bay and Watershed ecosystems.

Human influence on the Narragansett Bay Watershed is responsible for the environmental issues associated with excessive nutrient loading. During the pre-colonial period prior to 1650, clear water, strong tidal mixing, shallow depths, and relatively nutrient-rich areas near coastal shelf waters influenced nutrient levels and productivity within Narragansett Bay (Nixon et al. 2008). Land-based nutrient export accounted for just seventeen percent of the total nitrogen inputs and one percent of the phosphorus inputs to Narragansett Bay (Nixon et al. 2008). The majority of nutrients came from offshore waters that circulated into the Bay. That nutrient balance remained largely unchanged until the late 1800s, when indoor plumbing and centralized sewer systems were introduced. Previous methods of dry waste disposal had largely retained nitrogen and phosphorus in the soil, regardless of population numbers (Nixon et al. 2008). An unintended consequence of the introduction of running water to the region was the release of untreated human and animal waste (and nutrients) through a main sewer directly into the rivers and Bay (see "Wastewater Infrastructure" chapter; Nixon et al. 2005, Hamburg et al. 2008, Nixon et al. 2008).

By the 1880s, construction of sewers began, and by 1889, wastewater from 54,000 people—50 percent of Providence's population—was collected into the sewer system. The system collected both stormwater and wastewater, conveying them directly into rivers and the Bay (Nixon et al. 2008). In addition to Providence, other shoreline communities throughout the Watershed constructed treatment facilities to manage the waste produced by their populations (Nixon et al. 2008, Vadeboncoeur et al. 2010). The percentage of the upper Bay's population served by sewers increased steadily until the 1950s, and since then the population connected to sewers has remained steady at roughly 83 percent (see "Wastewater Infrastructure" chapter; Vadeboncoeur et al. 2010). Around the lower Bay, 31 percent of the population was served by sewers by 1950 (see "Wastewater Infrastructure" chapter; Vadeboncoeur

et al. 2010). The increasing populations in Bay communities quickly overwhelmed the initial capacities of the treatment facilities, particularly Field's Point (Shea 1946), decreasing water quality in receiving waters.

The passage of the Clean Water Act in 1972 authorized the Environmental Protection Agency (EPA) to implement the National Pollutant Discharge Elimination System (NPDES). The NPDES was designed, in part, to regulate major point source discharges into the nation's waters. Wastewater treatment facilities (WWTFs) were required to obtain permits to discharge effluent and upgrade facilities with primary and secondary treatment to tertiary or other advanced treatments. As a result of the NPDES, all WWTFs must demonstrate permit compliance by measuring the level of pollutants in their discharge and submitting Discharge Monitoring Reports (DMRs) to the EPA or a state discharge-permit regulating agency to remain in operation.

Research funded by the Narragansett Bay Estuary Program during the 1980s found seasonal low-oxygen conditions in bottom waters of the Providence and Seekonk Rivers along with Upper Narragansett Bay, and these low-oxygen conditions were believed to be caused by both nutrient loading and biochemical oxygen demand (BOD) from WWTFs (Doering et al. 1989, Pilson and Hunt 1989, Doering and Oviatt 1990). Prior to the 1990s, WWTF improvements reduced BOD and Total Suspended Solids loads to receiving waters, and since that time state agencies and EPA began efforts to determine and set appropriate nutrient reductions for WWTFs (Angelo Liberti, Rhode Island Department of Environmental Management, personal communication).

Efforts to set appropriate nitrogen permit limits were accelerated after a 2003 fish kill in Greenwich Bay. A severe rainstorm flushed nutrients into the Bay, and combined with physical forcing (wind and tide) prompted eutrophic conditions, culminating in a severe hypoxic/anoxic event and fish kill (RIDEM 2003). In response, Rhode Island enacted a 2004 statute resulting in a nutrient management plan to further reduce summer (May through October) point source nitrogen loadings to Upper Narragansett Bay. The objective of the management plan was to reduce summer nitrogen loadings by 50 percent relative to 1995/1996 loading levels (reported at 15,000 pounds per day or $5{,}475 \times 10^3$ pounds per year) from eleven Rhode Island wastewater treatment facilities that discharge either directly to Narragansett Bay or the rivers (e.g., the Blackstone River) or embayments (e.g., Greenwich Bay) that impact the Upper Narragansett Bay (RIDEM 2005). Comparable reductions were completed at six Massachusetts WWTFs on the Blackstone and Ten Miles Rivers. Some WWTFs completed upgrades in a stepwise fashion.

The 50 percent reduction goal was first met in 2012 with a 72 percent reduction in the summer inputs of total nitrogen loadings (equivalent to a 64 percent reduction in annual WWTF loads) from the eleven Rhode Island targeted wastewater treatment facilities (RIDEM 2016). The permits issued by RIDEM for Rhode Island facilities and by EPA for Massachusetts facilities required WWTFs in both states to improve wastewater treatment plant technologies and to achieve designated limits on nitrogen and phosphorus. The effects of this nutrient reduction on the Narragansett Bay ecosystem are beginning to be explored (see "Dissolved Oxygen" and "Chlorophyll" chapters, other Bay and Watershed Condition chapters, and this chapter).

In this chapter, the Narragansett Bay Estuary Program reports on the present status and trends of nitrogen and phosphorus loading from wastewater treatment facilities and six major rivers in the Narragansett Bay Watershed. The chapter provides information to relate this environmental stressor to the physical (dissolved oxygen concentration, water clarity, and water quality) and biological (seagrasses, salt marsh, benthic habitats, estuarine and freshwater fish communities, and stream invertebrates) indicators presented in other chapters of this report.

Methods

The Estuary Program focused its analysis on two key pieces of a Watershed-wide nutrient budget: wastewater treatment facility (WWTF) and riverine loadings. Recent data on 37 WWTFs and six tributary rivers were used to develop a nutrient budget, which was then compared to prior nutrient budgets. Total nitrogen and total phosphorus were used for all loading calculations. Where total nitrogen data were unavailable, total nitrogen was calculated using the sum of total Kjeldahl nitrogen, nitrite, and nitrate concentrations. Where total phosphate data were unavailable, orthophosphate was converted to total phosphate using a conversion factor determined by Nixon and colleagues (2008) and updated by Krumholz (2012).

The Methods, Status and Trends, and Discussion sections in this chapter are organized using the following structure:

- Wastewater Treatment Facility Loadings
- Riverine Loadings
- Nutrient Budget

WASTEWATER TREATMENT FACILITY LOADINGS

Wastewater treatment facility loadings were calculated by multiplying monthly average concentration by flow and converting the units to pounds per day for each facility (Equation 1). Data were then converted to pounds per month by multiplying the daily load by the number of days in that particular month. Monthly loads were summed to achieve yearly loads. Some facilities did not have data for every month of every year. In these cases, the available monthly loads were multiplied by the ratio of twelve divided by the number of months with data available, per EPA guidance (Equation 2; Dave Pincumbe, USEPA Region 1, personal communication). All figures and tables use the units of thousands ($x10^3$) of pounds per year for ease of reading. The Rhode Island Department of Environmental Management's Office of Water Resources (RIDEM) provided data for all WWTFs in Rhode Island. The EPA's Discharge Monitoring Report Pollutant Loading Tool was utilized to acquire data for WWTFs in Massachusetts. In addition, EPA, Massachusetts Department of Environmental Protection, and individual facilities provided data.

Not all WWTFS are required to monitor phosphorus (or orthophosphate). These facilities discharge directly to estuarine waters, which are not phosphorus limited. For those WWTFs, total phosphorus was calculated using population-served estimates and the average phosphorus load per person (Nixon et al. 2008, Krumholz 2012).

RIVERINE LOADING

The river nutrient loadings do not distinguish among the various nutrient sources (wastewater, runoff, atmospheric deposition, on-site wastewater facilities, etc.) and were calculated using river flow data from the United States Geological Survey (USGS) and nutrient data from the Narragansett Bay Commission (NBC) Nutrient Monitoring Program. As in the budget analysis completed by Nixon and colleagues (Nixon et al. 1995, 2008) and Krumholz (2012), the Estuary Program's analysis included six rivers: Blackstone, Pawtuxet, Moshassuck, Ten Mile, Woonasquatucket, and Taunton. Survey gages were chosen based on their proximity to the mouth of the rivers to capture representative flow to the Bay. Similarly, the Narragansett Bay Commission nutrient monitoring stations were chosen based on their proximity to the mouth of their respective rivers (Table 1). The Taunton River did not have a gage close to the mouth of the river, and therefore the Estuary Program used a gage located upstream from the mouth, the same gage as previous budgets (Nixon et al. 1995, 2008; Krumholz 2012). The Pawtuxet River and the Woonasquatucket River also did not have gages close to the river mouth, nor were they close to the NBC sample stations; however, in both cases, flow gages and sample stations were on the main stem of each river and were considered representative. Daily river flow was converted from cubic feet per second to millions of gallons per day and then multiplied by a ratio to account for the ungaged flow area of each river. This ratio was calculated by dividing the total area of each river basin by the area each gage measured (Table 2).

$$\left[concentration\left(\frac{mg}{L}\right)\right] \times [flow(MGD)] \times 8.34 = loading\left(\frac{lbs}{day}\right)$$

Equation 1. Conversion from concentration to loading estimate. 8.34 is a conversion factor that converts mg/L x millions gallons/day to lbs/day.

$$\sum average\ monthly\ loadings\left(\frac{lbs}{month}\right) \times \left(\frac{12}{number\ of\ months\ with\ data\ available}\right)$$

Equation 2. Scaling method used for WWTF loadings when data were not available for all months in all years.

Table 1. List of gages and stations used for river nutrient budget.

River	Narragansett Bay Commission Station	USGS Gage #	USGS Gage Name
Blackstone	Blackstone @ Slater Mill	01113895	Blackstone River @ Roosevelt St
Ten Mile	Ten Mile @ Omega Pond Outlet	01109403	Ten Mile River @ Pawtucket Avenue
Moshassuck	Moshassuck @ Mill St	01114000	Moshassuck River @ Providence
Woonasquatucket	Woonasquatucket @ Valley St	01114500	Woonasquatucket @ Centerdale
Pawtuxet	Pawtuxet @ Terminal Falls	01116500	Pawtuxet River @ Cranston
Taunton	Taunton River	01108000	Taunton River @ Bridgewater, MA

Table 2. Gaged areas and total drainage areas for rivers in the Narragansett Bay Watershed. Gaged and total drainage areas were determined by USGS for the gage stations used (Table 1). Unmeasured river drainage area (categorized under Coastal Narragansett Bay Basin) calculated by subtracting Total Drainage Area (1,464.2 square miles) of the named rivers from the total Basin area (1,704.9 square miles, Appendix). River Basin definitions and information are provided in the Appendix. N/A means not applicable.

Basin Name River Name	Gaged Area (sq. miles)	Total Drainage Area (sq. miles)	Ratio (gaged area/total drainage area)
Coastal Narragansett Bay Basin[1]			
Ten Mile River	53.1	54	1.02
Moshassuck River	23.1	23.6	1.02
Woonasquatucket River	38.3	50	1.31
Unmeasured Rivers	N/A	240.7	N/A
Blackstone River Basin			
Blackstone River	474	540	1.14
Taunton River Basin			
Taunton River	261	565	2.16
Pawtuxet River Basin			
Pawtuxet River	200	231.6	1.16
Total	**1,049.5**	**1,704.9**	

[1] The Coastal Narragansett Bay Basin includes both the Bay and some small, tributary rivers.

The Narragansett Bay Commission provided total nitrogen data from May 2013 through December 2015 and orthophosphate data from the calendar years 2013 to 2015. Nutrient data were collected monthly. Similar to the approach in Krumholz (2012), orthophosphate values were converted to total phosphorus values using a ratio of total phosphorus to inorganic phosphorus unique to each river. Beale's estimator was used to calculate yearly average loadings because it accounts for situations in which there is an abundance of flow data and a scarcity of nutrient concentration data (Dolan et al. 1981). This is the same estimator used by Nixon and colleagues (1995, 2008) and Krumholz (2012). Data are presented in thousand pounds per year for ease of reading.

The USGS collects primarily monthly nutrient monitoring data at certain gages, and those data are freely available. In the Narragansett Bay Watershed, the USGS maintains records for the Blackstone and Pawtuxet Rivers (Savoie et al. 2017). No data were available for 2013 to 2015 for the other four rivers listed above (Moshassuck, Ten Mile, Woonasquatucket, and Taunton Rivers). The Narragansett Bay Commission collects data that cover more of the Watershed. Additionally, the Commission data were used in the most recent budget estimates (Krumholz 2012). Therefore, even though the USGS does maintain nutrient data, the Estuary Program opted to proceed with the Narragansett Bay Commission data.

Previous budgets (Nixon et al. 1995, 2008; Krumholz 2012) included a component for rivers in which flow and/or nutrients had not been measured. Based on an analysis by the USGS (Ries 1990), Nixon and colleagues calculated that the unmeasured rivers accounted for 169.9 square miles of coastal drainage (Nixon et al. 1995, 2008). The Estuary Program calculated a similar unmeasured river area by subtracting the named rivers' total drainage areas from the total River Basin area (Table 2; Appendix). Since these areas are mostly in the coastal zone, the Estuary Program categorized the unmeasured rivers element as part of the Coastal Narragansett Bay Basin. The Estuary Program used total N and total P yields to calculate loading from the unmeasured rivers from Fulweiler and Nixon (2005) following Nixon and colleagues (2008).

NUTRIENT BUDGET

The Estuary Program updated river and wastewater treatment facility loadings with publicly available data for 2013 to 2015, and the remaining components (urban runoff, atmospheric deposition, and groundwater loading) were assumed to be unchanged and were carried over from the 2007 to 2010 budget (Krumholz 2012). Urban runoff was updated by Nixon and colleagues (2008) and Krumholz (2012). All previous budgets have defined urban runoff using four land use categories: residential, commercial, industrial, and highway. Urban runoff was calculated as loading from these categories directly to Narragansett Bay. Atmospheric deposition remained the same between all three previous budgets (Nixon et al. 1995, 2008; Krumholz 2012), using the same data and time period. The only budget to include groundwater nutrient loading was Krumholz (2012), with estimates based on a study conducted in Greenwich Bay (Urish and Gomez 2004). Nixon and colleagues (1995, 2008) believed that groundwater nutrient loading was not significant to the Bay proper, as groundwater flow was not considered to be significant (Nowicki and Gold 2008).

For information about the methods used to calculate the budget categories for the Bay and the Rivers, including assumptions and limitations, see the Wastewater Treatment Facility Loadings and Riverine Loadings sections above.

The Estuary Program recognizes that the calculations in its analysis contain inherent error. For ease of understanding, and comparison across budgets, an error analysis was not included in this chapter; this information is available upon request. Krumholz (2012) also conducted an error analysis and reported how error was determined. Additional information about methods for the development of the nutrient loadings budget is also available upon request from the Estuary Program.

Status and Trends

WASTEWATER TREATMENT FACILITY LOADING

Status and Trends of Nitrogen Loading

The current status of nitrogen loading was calculated for 37 wastewater treatment facilities (WWTFs) in the Narragansett Bay Watershed that discharged $5{,}562 \times 10^3$ pounds per year based upon the 2013 to 2015 nutrient budget. The fifteen WWTFs discharging in the Coastal Narragansett Bay Basin accounted for 58 percent of the total nitrogen loading in the Watershed (Table 3; Figure 1). Of these fifteen facilities, twelve discharge directly to Narragansett Bay, which includes the Providence and Seekonk Rivers along with Mount Hope Bay and Greenwich Bay (53 percent of total loadings), and the remaining discharge to the Ten Mile and Woonasquatucket Rivers. The nine facilities discharging in the Taunton

River Basin have the second largest nitrogen loading with nineteen percent of the total loading. The Blackstone River Basin has the third largest nitrogen loading with ten facilities contributing fourteen percent. The Pawtuxet River Basin only has three WWTFs and has eleven percent of the total loading.

Trends in WWTF loading in the Narragansett Bay Watershed from 2000–2004 to 2013–2015 were examined by comparing the loadings from the Estuary Program budget to the Nixon and colleagues (2008) and Krumholz (2012) budgets. The Estuary Program could not compare WWTF loadings to the Nixon and colleagues budget from 1982–1983 (Nixon et al. 1995) because that budget did not include a list of individual WWTF loadings. A total of 28 WWTFs were included in this trend analysis, as the earlier budgets did not assess all 37 facilities in the Watershed (Table 4). Figures 1 and 2 illustrate the range of percent reductions at the WWTFs and the Basins from the 2000–2004 budget to the 2013–2015 budget.

For some facilities, changes in reported effluent loading may be due to variations in calculation methods, frequency of nitrogen data collection, or changes in WWTF flows. Nitrogen reductions have been required from those WWTFs that significantly contribute to nutrient-related impairments, but not all facilities required to reduce nitrogen have done so.

Many nitrogen upgrades to WWTFs have been completed or are scheduled. Those investments have resulted in reductions at various WWTFs in the Watershed (Table 4). The facilities with the most significant reductions based on the nitrogen budget analysis are listed in Table 5.

Status and Trends of Phosphorus Loading

In the 2013–2015 nutrient budget, total phosphorus loading from the 37 WWTFs in the Watershed was 762×10^3 pounds per year (Table 6). Excess phosphorus is primarily a concern in fresh waters, and WWTF upgrades to reduce phosphorus have been limited to those that discharge to tributary rivers.

Table 3. Nitrogen loadings and percentage of watershed loading originating from each of the 37 wastewater treatment facilities in the Narragansett Bay Watershed based on the Estuary Program's 2013 to 2015 nutrient budget. Italics indicate WWTFs located in Massachusetts. River Basin names are shaded.

Discharge Basin Receiving Waters Name of WWTF	WWTF Total Nitrogen Loading ($x10^3$ lbs/year)	Percent of Watershed Loading
Coastal Narragansett Bay Basin[1]	**3,158**	**58%**
Narragansett Bay	**2,936**	**53%**
Fall River	*1,010*	*18%*
Field's Point	727	13%
Bucklin Point	339	6%
Bristol	286	5%
Newport	156	3%
South Kingstown	134	2%
East Providence	124	2%
Warren	58	1%
Quonset Point	36	1%
East Greenwich	35	1%
Scarborough	25	0.5%
Jamestown	6	0.1%
Ten Mile River	**170**	**3%**
Attleboro	*106*	*2%*
North Attleborough	*64*	*1%*
Woonasquatucket River	**52**	**1%**
Smithfield	52	1%

Table 3 continued

Discharge Basin Receiving Waters Name of WWTF	WWTF Total Nitrogen Loading ($x10^3$ lbs/year)	Percent of Watershed Loading
Taunton River Basin	**1076**	**19%**
Taunton River		
Taunton[2]	*301*	*5%*
Brockton	*235*	*4%*
Mansfield	*156*	*3%*
Middleborough	*155*	*3%*
Somerset	*140*	*3%*
Bridgewater	*76*	*1%*
MCI-Bridgewater	*12*	*0.2%*
Wheaton	*5*	*0.1%*
White Oak Island Trust	*3*	*0.1%*
Blackstone River Basin	**737**	**13%**
Blackstone River		
Upper Blackstone (Worcester)	*417*	*7%*
Woonsocket	129	2%
Grafton	*93*	*2%*
Hopedale	*28*	*1%*
Burrillville	27	0.5%
Northbridge	*19*	*0.3%*
Uxbridge	*13*	*0.2%*
Douglas	*5*	*0.1%*
Upton	*5*	*0.1%*
Zambarano	1	0.02%
Pawtuxet River Basin	**591**	**11%**
Pawtuxet River		
Cranston	241	4%
West Warwick	231	4%
Warwick	119	2%
Total (37 WWTFs)	**5,569**	**100%**

(1) The Coastal Narragansett Bay Basin includes both the Bay and some tributary rivers. The WWTFs that discharge directly to the Bay are listed under the heading "Narragansett Bay" as the receiving waters.
(2) The Estuary Program was unable to obtain data for the Taunton WWTF for 2013 and 2014. Therefore, the loading value presented here is only for the year 2015.

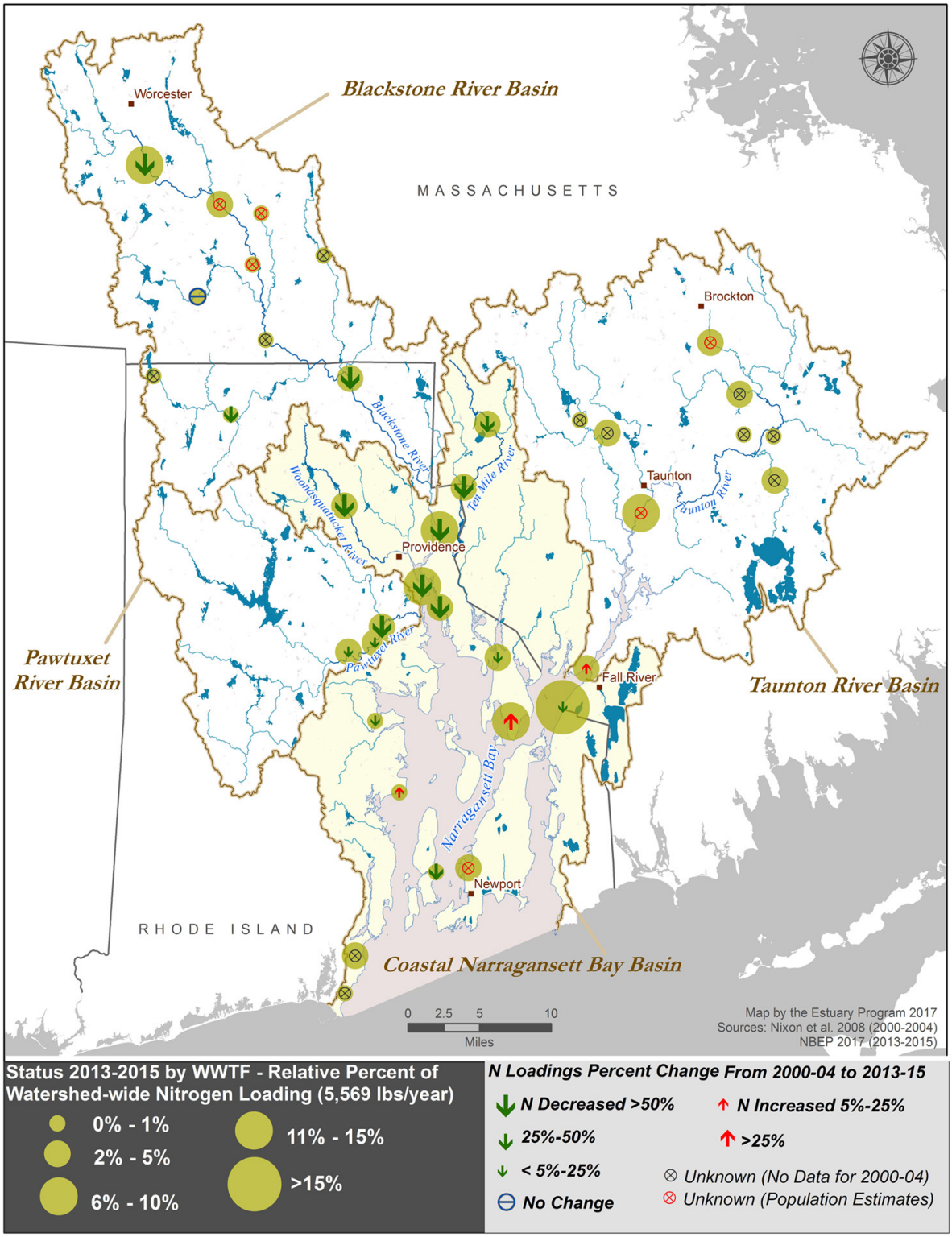


Figure 1. Nitrogen loadings from the 37 wastewater treatment facilities in the Narragansett Bay Watershed. Symbols represent individual wastewater treatment facilities (WWTFs).

Table 4. WWTF nitrogen loadings from 2000–2004, 2007–2010, and 2013–2015, including only those WWTFs assessed by Nixon and colleagues (2008) and Krumholz (2012). Italics indicate WWTFs located in Massachusetts. River Basin names are shaded. Dashes (-) indicate data not available or not reported. Loadings with an asterisk (*) were calculated using population estimates.

	WWTF Total Nitrogen Loading ($x10^3$ lbs/year)		
Discharge Basin	**Nixon et al. (2008)[3]**	**Krumholz (2012)[3]**	**This Study**
Receiving Waters			
Name of WWTF	**2000–2004**	**2007–2010**	**2013–2015**
Coastal Narragansett Bay Basin[1]	**5,766**	**4,793**	**2,999**
Narragansett Bay	**5,253**	**4,420**	**2,777**
Fall River	*1,056*	*1,023**	*1,010*
Field's Point	1,993	1,956	727
Bucklin Point	1,149	582	339
Bristol	209	193	286
Newport	400*	323*	156
East Providence	302	232	124
Warren	69	57	58
Quonset Point	29	22	36
East Greenwich	37	27	35
Jamestown	9	5	6
Ten Mile River	**379**	**328**	**170**
Attleboro	*263*	*236*	*106*
North Attleborough	*116*	*92*	*64*
Woonasquatucket River	**134**	**45**	**52**
Smithfield	134	45	52
Taunton River Basin	**1,514**	**1,509**	**669**
Taunton River			
Brockton	*1,130**	*1,125**	*235*
Somerset	*129*	*255**	*140*
Taunton	*255**	*129**	*301*
Blackstone River Basin	**2,936**	**988**	**723**
Blackstone River			
Upper Blackstone (Worcester)	*2,076*	*511*	*417*
Woonsocket	576	154	129
Grafton	*85**	*101**	*93*
Millbury[2]	*72**	*75**	-
Hopedale	-	-	*28*
Burrillville	40	43	27
Northbridge	*79**	*94**	*19*
Douglas	*5*	*6**	*5*
Upton	*3**	*4**	*5*
Pawtuxet River Basin	**979**	**778**	**591**
Pawtuxet River			
Cranston	582	385	241
West Warwick	252	247	231
Warwick	145	146	119
Total (28 WWTFs)	**11,195**	**8,068**	**4,989**

[1] The Coastal Narragansett Bay Basin includes both the Bay and some tributary rivers. The WWTFs that discharge directly to the Bay are listed under the heading "Narragansett Bay" as the receiving waters.
[2] This facility was decommissioned in 2005 and connected to Worcester's Upper Blackstone facility.
[3] For some of the population estimates from these authors, actual data may exist. However, the Estuary Program chose to report the authors' data as they published them.

The WWTFs discharging directly to Narragansett Bay accounted for 74 percent of total phosphorous loading. The Taunton River had the second highest total phosphorus loading, and the Blackstone River had the third (Table 6).

Trends in WWTF loading in the Narragansett Bay Watershed from 2000–2004 to 2013–2015 were examined by comparing the loadings from the Estuary Program budget to the Nixon and colleagues (2008) and Krumholz (2012) budgets. The Estuary Program could not compare WWTF loadings to the Nixon and colleagues budget from 1982–1983 (Nixon et al. 1995) because that budget did not include a list of individual WWTF loadings. A total of 28 WWTFs were included in this trend analysis, as the earlier budgets did not assess all 37 facilities in the Watershed (Table 7).

Since phosphorus monitoring from the WWTFs discharging to Narragansett Bay is not included in many permits, only the three largest of its ten facilities report total phosphorus. Thus, loading estimates for seven WWTFs were calculated using population estimates, and the WWTF budget may not be accurately reporting the total phosphorus loading for Narragansett Bay (Table 7). The estimated reduction and percent reduction were calculated for each of the River Basins (Figure 3).

RIVERINE LOADING

Trends in River Nitrogen and Phosphorus Budgets

The river budgets showed a decrease in total nitrogen and total phosphorus over time in each of the six rivers and the unmeasured rivers when comparing

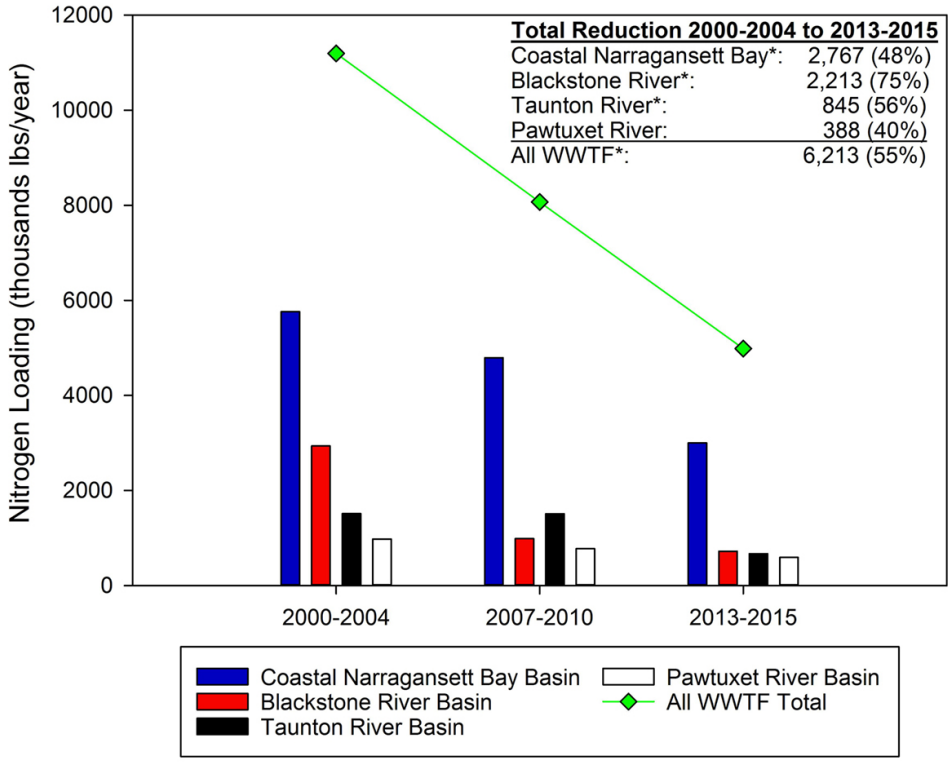


Figure 2. Comparison of WWTF nitrogen loading budgets, including only those WWTFs assessed by Nixon and colleagues (2008) and Krumholz (2012) (Table 4). "All WWTF Total" is the total of all WWTF loadings discharged to the Watershed. Asterisks (*) indicate Basins for which loadings were calculated using population estimation. Monitoring and estimation data were combined to create Watershed-wide loadings. Reduction units are in thousands ($x10^3$) of pounds per year.

Table 5. WWTFs with more than 50 percent reduction in nitrogen loading from the 2000–2004 budget (Nixon et al. 2008) to the 2013–2015 budget. Facilities for which Nixon and colleagues (2008) used population estimates and the Estuary Program used Discharge Monitoring Report data were not included. Sorted from largest to smallest decrease in pounds per year. Italics indicate WWTFs located in Massachusetts.

WWTF	Change in Nitrogen Loading ($x10^3$ lbs/year) 2000–2004 to 2013–2015	Percent Reduction 2000–2004 to 2013–2015
Upper Blackstone (Worcester)	*-1,659*	*80%*
Field's Point	-1,266	64%
Bucklin Point	-810	70%
Woonsocket	-447	78%
Cranston	-341	59%
Attleboro	*-157*	*60%*
East Providence	-178	59%
Smithfield	-82	61%

Table 6. Total phosphorus loading and percent of watershed loading for 37 WWTFs in the Narragansett Bay Watershed from the 2013–2015 budget. Italics indicate WWTFs located in Massachusetts. River Basin names are shaded. Loadings with an asterisk (*) were calculated using population estimates.

Discharge Basin Receiving Waters Name of WWTF	WWTF Total Phosphorus Loading ($x10^3$ lbs/year)	Percent of Watershed Loading (%)
Coastal Narragansett Bay Basin[1]	**570**	**74.9%**
Narragansett Bay	**566**	**74.3%**
Field's Point	171	22.7%
Bucklin Point	137	18.2%
Fall River	*96**	*12.7%*
Newport	45*	6.0%
South Kingstown	32*	4.2%
East Providence	26	3.4%
Bristol	22*	2.9%
Quonset Point	11*	1.5%
Warren	9*	1.2%
Scarborough	8*	1.0%
East Greenwich	7*	0.9%
Jamestown	2*	0.3%
Ten Mile River	**3**	**0.5%**
North Attleborough	*2*	*0.3%*
Attleboro	*1*	*0.2%*
Woonasquatucket River	**1**	**0.1%**
Smithfield	1	0.1%

Table 6 continued

Discharge Basin Receiving Waters Name of WWTF	WWTF Total Phosphorus Loading ($x10^3$ lbs/year)	Percent of Watershed Loading (%)
Taunton River Basin	**92**	**12%**
Taunton River		
Taunton[2]	*57*	*7.5%*
Somerset	*14**	*1.8%*
MCI-Bridgewater	*8*	*1.0%*
Brockton	*6*	*0.8%*
Bridgewater	*5*	*0.6%*
Mansfield	*1*	*0.1%*
Middleborough	*1*	*0.1%*
Wheaton	*0.1*	*0.01%*
White Oak Island Trust	*0.05*	*0.01%*
Blackstone River Basin	**58**	**7.6%**
Blackstone River		
Woonsocket	25	3.3%
Upper Blackstone (Worcester)	*19*	*2.5%*
Hopedale	*6**	*0.8%*
Grafton	*4*	*0.6%*
Northbridge	*1*	*0.1%*
Burrillville	1	0.1%
Uxbridge	*1*	*0.1%*
Douglas	*0.5*	*0.1%*
Zambarano	0.2	0.02%
Upton	*0.1*	*0.01%*
Pawtuxet River Basin	**41**	**5.4%**
Pawtuxet River		
Cranston	23	3.0%
West Warwick	11	1.5%
Warwick	7	0.9%
Total (37 WWTFS)	**762**	**100%**

(1) The Coastal Narragansett Bay Basin includes both the Bay and some tributary rivers. The WWTFs that discharge directly to the Bay are listed under the heading "Narragansett Bay" as the receiving waters.
(2) The Estuary Program was unable to obtain data for the Taunton WWTF for 2013 and 2014. Therefore, the loading value presented here is only for the year 2015.

Table 7. WWTF phosphorus loading from 2000–2004, 2007–2010, and 2013–2015. Dashes (-) indicate data not available or not reported. Italics indicate WWTFs located in Massachusetts. River Basin names are shaded. Loadings with an asterisk (*) were calculated using population estimates.

Discharge Basin Receiving Waters Name of WWTF	WWTF Total Phosphorus Loading ($x10^3$ lbs/year) Nixon et al. (2008)[3] 2000–2004	Krumholz (2012)[3] 2007–2010	This Study 2013–2015
Coastal Narragansett Bay Basin[1]	**598**	**622**	**530**
Narragansett Bay	**551**	**618**	**526**
Field's Point	160	211	171
Bucklin Point	148	207	137
Fall River	*79**	*76**	*96**
Newport	32*	39*	45*
East Providence	51	34	26
Bristol	12	12	22*
Quonset Point	5*	7	11*
Warren	7	3	9*
East Greenwich	38	28	7*
Jamestown	19*	1	2*
Ten Mile River	**26**	**3**	**3**
North Attleborough	*15*	*2*	*2*
Attleboro	*11*	*1*	*1*
Woonasquatucket River	21	1	**1**
Smithfield	21	1	1
Taunton River Basin	**85**	**85**	**39**
Taunton River			
Taunton	*19**	*19**	*57*
Somerset	*11**	*11**	*14**
Brockton	*55**	*55**	*6*
Blackstone River Basin	**328**	**147**	**57**
Blackstone River			
Woonsocket	143	37	25
Upper Blackstone (Worcester)	*149*	*71*	*19*
Hopedale	*1**	*1**	*6**
Grafton	*8**	*9**	*4*
Northbridge	*9**	*11**	*1*
Burrillville	2	1	1
Douglas	*1**	*1**	*0.5*
Millbury[2]	*15**	*16**	-
Upton	*0**	*0**	*0.1*
Pawtuxet River Basin	**212**	**72**	**41**
Pawtuxet River			
Cranston	86	28	23
West Warwick	90	30	11
Warwick	36	14	7
Total (28 WWTFs)	**1,223**	**926**	**667**

[1] The Coastal Narragansett Bay Basin includes both the Bay and some tributary rivers. The WWTFs that discharge directly to the Bay are listed under the heading "Narragansett Bay" as the receiving waters.
[2] This facility was decommissioned in 2005 and connected to Worcester's Upper Blackstone facility.
[3] For some of the population estimates from these authors, actual data may exist. However, the Estuary Program chose to report the authors' data as they published them.

the four budget estimates: 1982–1983 (Nixon et al. 1995), 2000–2004 (Nixon et al. 2008), 2007–2010 (Krumholz 2012), and 2013–2015 (Tables 8 and 9).

River loadings were calculated for all rivers that had wastewater treatment facility outfalls within the Watershed (Figure 1). The Estuary Program included the Moshassuck River, which had no wastewater treatment facility outfalls discharging to it but is a relatively large river in terms of discharge to Narragansett Bay (Nixon et al. 1995). Tables 8 and 9 also include an "unmeasured rivers" component, which captures all other smaller rivers for which nutrient and/or flow data were not available. The drainage area of the "unmeasured rivers" falls within the Coastal Narragansett Bay Basin.

The methods for calculating the unmeasured rivers element has changed over the last 30 years. In the 1982–1983 budget, Nixon and colleagues (1995) relied on research by the USGS to assess the drainage area of the unmeasured rivers and then calculated the gaged area to total drainage area ratio (as in Table 2). The resulting loading was weighted towards heavily anthropogenically influenced regions. When Nixon and colleagues (2008) revisited this term in the 2000–2004 budget, they determined that much of the unmeasured flow was located in suburban and rural areas, and they used nutrient yields from the Pawcatuck River, a suburban river in the southwest portion of Rhode Island (Fulweiler and Nixon 2005), to calculate the unmeasured river loading. This reduced the estimate from their earlier work substantially (Tables 8 and 9).

NUTRIENT BUDGET

In these budgets, components that directly discharge into Narragansett Bay (WWTF loading, atmospheric deposition, run-off, and groundwater)

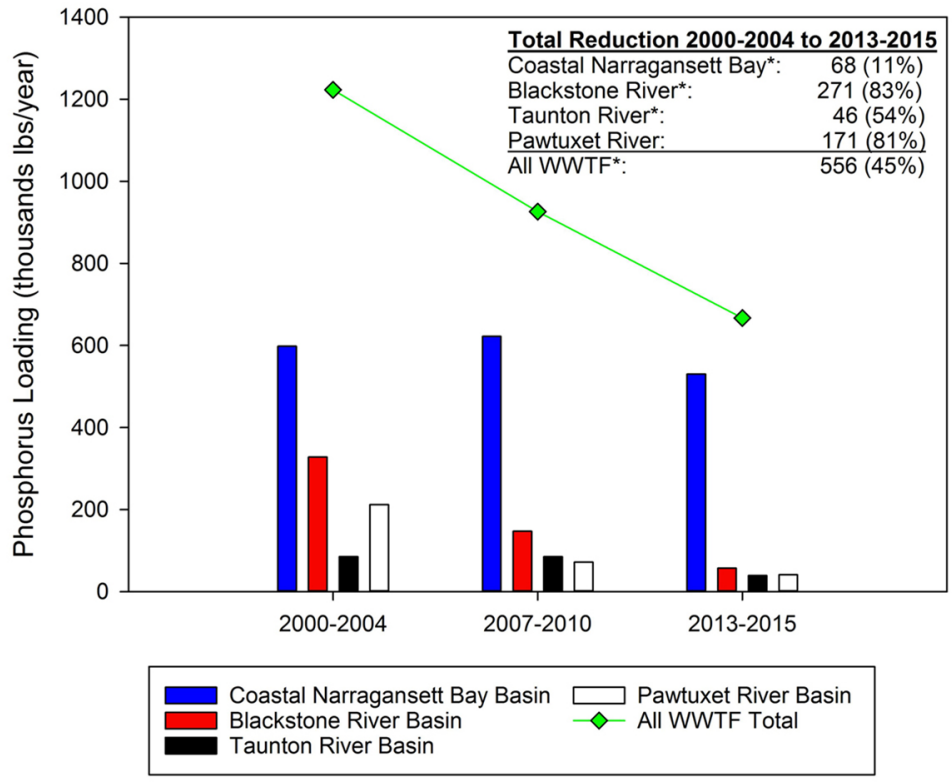


Figure 3. Comparison of WWTF phosphorus loading budgets, including only those WWTFs assessed by Nixon and colleagues (2008) and Krumholz (2012) (Table 7). "All WWTF Total" is the total of all WWTF loadings discharged to the Watershed. Asterisks (*) indicate Basins for which loadings were calculated using population estimation. Monitoring and estimation data were combined to create Watershed-wide loading. Reduction units are in thousands ($x10^3$) of pounds per year.

are listed separately. The river component includes sources that discharge to the rivers (WWTF loading, atmospheric deposition, run-off, and groundwater). The WWTF loading totals and river loading totals are derived from Tables 4, 7, 8, and 9.

For the total nitrogen budget–including air, river, WWTF discharge, run-off, and groundwater–a clear decrease in total nitrogen was observed over time when comparing all four budget estimates: 1982–1983, 2000–2004 (Nixon et al. 1995 and 2008), 2007–2010 (Krumholz 2012), and 2013–2015 (Table 10).

Similarly, a decrease in total phosphorus was observed over time when comparing the four budget estimates: 1982–1983, 2000–2004 (Nixon et al. 1995 and 2008), 2007–2010 (Krumholz 2012), and 2013–2015 (Table 11).

Discussion

WASTEWATER TREATMENT FACILITY LOADING

Since 2000–2004, wastewater treatment facility loading has decreased 55 percent in nitrogen and 45 percent in phosphorus across the Watershed (Tables 4 and 7; Figures 2 and 3). The majority of the decrease in nitrogen loading was due to compliance with the goal of reducing loading from eleven Rhode Island WWTFs by 50 percent of 1995/1996 loading levels (RIDEM 2005). Six of those WWTFs are listed in Table 5 as having greater than 50 percent reductions in the last fifteen years. Additionally, Massachusetts has made similar reductions at WWTFs along the Blackstone and Ten Mile Rivers, and since 2000 those WWTFs have also contributed to the large decline in nitrogen loading (Table 5). Phosphorus has a

Table 8. River nitrogen budget comparison. Dashes (-) indicate data not available or not reported.

Discharge Basin Rivers	Total Nitrogen ($x10^3$ lbs/year) Nixon et al. (1995) 1982–1983	Nixon et al. (2008) 2000–2004	Krumholz (2012) 2007–2010	Estuary Program 2013–2015
Coastal Narragansett Bay Basin	**2,680**	**1,308**	**2,775**	**876**
Ten Mile River	-	434	443	313
Woonasquatucket River	308	265	176	110
Moshassuck River	154	147	83	50
Unmeasured Rivers[1]	2,218	462	2,073	403
Blackstone River Basin	**4,035**	**3,038**	**2,610**	**1,536**
Blackstone River	4,035	3,038	2,610	1,536
Taunton River Basin	**3,604**	**4,220**	**1,161**	**1,925**
Taunton River[2]	3,604[3]	4,220[3]	1,161[4]	1,925
Pawtuxet River Basin	**1,971**	**1,826**	**1,133**	**756**
Pawtuxet River	1,971	1,826	1,133	756
Total River Nitrogen Loading	**12,290**	**10,421**	**7,678**	**4,688**

[1] The Unmeasured Rivers loading value for 2013–2015 was calculated following the method of Nixon et al. 2008. Krumholz (2012) followed the method of Nixon et al. (1995), which used larger nitrogen yields (for full discussion, see Nixon et al. 2008). See text.

[2] The Taunton River loading value for 2013–2015 was calculated following the method of Nixon et al. (2008). Krumholz (2012) reduced the drainage area of the Taunton River based on the locations of the USGS Taunton River gage at Bridgewater and the location of the NBC nutrient monitoring station (see Table 1), reducing the overall loading. He added the ungaged drainage area into the "unmeasured rivers" element of the budget (Krumholz 2012).

[3] The increase between 1982–1983 and 2000–2004 is based solely on a land-use model, and it is believed that actual loadings from the Taunton River decreased during this time, based on the 20 to 25 percent decreases observed in the other rivers (Nixon et al. 2008).

[4] Krumholz (2012) did not calculate his Taunton River loading in the same manner as previous or the current budgets. The Estuary Program estimated the Taunton River loading from Krumholz (2012) as if he had used the same method as the previous budgets, which would be $2,508x10^3$ pounds per year. See text.

significant impact on freshwater ecosystems, and the majority of load reductions have been focused in facilities discharging to freshwater. The majority of phosphorus declines were realized on the Blackstone (83 percent) and Pawtuxet Rivers (81 percent) (Table 7; Figure 3).

Four individual WWTFs had increases in nitrogen loading between 2000-2004 and 2013-2015 (Table 4). The largest facility to experience an increase was Bristol, which represented five percent of the total nitrogen loading to the Watershed. Year-to-year variations are expected due to changes in flow or influent loadings, and increases in loading will occur due to population and economic growth. The facilities are designed to manage these changes while still complying with their permits. Only one facility for which the Estuary Program had monitoring data for 2000-2004 and 2013-2015 showed an increase in phosphorus loading. The facility discharges directly to the Bay, where phosphorus is not thought to be of great concern.

The wastewater treatment facility loadings were calculated two ways throughout the years: by population estimation and by monitoring data. Loadings that were calculated by population estimates were noted in the tables (3, 4, 6, and 7) and removed from the analysis presented in Table 5. However, those data were not removed from the trends analysis presented above or from the Watershed-wide total nutrient budget. Removing those facilities from the Watershed-wide trends analysis showed a 45 percent decrease in nitrogen loading and a 56 percent decrease in phosphorus loading. These loading reductions are still quite significant over the last fifteen years.

Table 9. River phosphorus budget comparison. Dashes (-) indicate data not available or not reported.

Discharge Basin Rivers	Total Phosphorus ($x10^3$ lbs/year) Nixon et al. (1995) 1982–1983	Nixon et al. (2008) 2000–2004	Krumholz (2012) 2007–2010	Estuary Program 2013–2015
Coastal Narragansett Bay Basin	**296**	**108**	**214**	**58**
Ten Mile River	-	53	18	1
Woonasquatucket River	26	21	7	3
Moshassuck River	13	9	1	1
Unmeasured Rivers[1]	257	25	188	53
Blackstone River Basin	**350**	**255**	**354**	**116**
Blackstone River	350	255	354	116
Taunton River Basin	**350**	**350**	**37**	**73**
Taunton River[2]	350[3]	350[3]	37[4]	73
Pawtuxet River Basin	**403**	**238**	**108**	**30**
Pawtuxet River	403	238	108	30
Total River Phosphorus Loading	**1,399**	**951**	**712**	**205**

[1]The Unmeasured Rivers loading value for 2013–2015 was calculated following the method of Nixon et al. 2008. Krumholz (2012) followed the method of Nixon et al. (1995), which used larger phosphorus yields (for full discussion, see Nixon et al. 2008). See text.

[2] The Taunton River loading value for 2013–2015 was calculated following the method of Nixon et al. (2008). Krumholz (2012) reduced the drainage area of the Taunton River based on the locations of the USGS Taunton River gage at Bridgewater and the location of the NBC nutrient monitoring station (see Table 1), reducing the overall loading. He added the ungaged drainage area into the "unmeasured rivers" element of the budget (Krumholz 2012).

[3] The Taunton River loading values from Nixon et al. (2008) were based on a land-use model, which showed no change in total phosphorus from the previous budget (Nixon et al. 2008).

[4] Krumholz (2012) did not calculate his Taunton River loading in the same manner as previous or the current budgets. The Estuary Program estimated the Taunton River loading from Krumholz (2012) as if he had used the same method as the previous budgets, which would be $80x10^3$ pounds per year. See text.

Table 10. Nitrogen budget comparison with previous budgets. Components with an asterisk (*) were carried over to the 2013–2015 budget from the 2007–2010 budget. Dashes (-) indicate data not available or not reported.

Source	Annual Total Nitrogen Loading ($x10^3$ lbs/year)			
	Nixon et al. 1995 1982–1983[1]	Nixon et al. 2008 2000–2004[2]	Krumholz 2012 2007–2010[3]	Estuary Program 2013–2015[4]
	Direct Discharge to the Bay			
Atmospheric Deposition*	924	924	924	924
WWTF Discharge[5]	5,636	5,253	4,420	2,777
Urban Run-Off *	1,140	1,140	1,910	1,910
Groundwater Discharge*	-	-	123	123
Rivers[5]	12,290	10,421	7,678	4,668
Total	**20,020**	**17,738**	**15,030**	**9,019**

[1] 1982–1983 budget numbers from Tables 16 and 19 in Nixon et al. (1995).
[2] 2000–2004 budget numbers from Table 5.15 in Nixon et al. (2008).
[3] 2007–2010 budget numbers from Table 3-1 in Krumholz (2012).
[4] 2013–2015 budget numbers from Tables 4, 7, 8, and 9.
[5] The term "WWTF Discharge" only includes WWTFs that discharge directly to the Bay. WWTFs discharging to rivers are included in the Rivers term along with other sources to rivers (run-off, groundwater, atmospheric deposition).

Table 11. Phosphorus budget comparison with previous budgets. Components with an asterisk (*) were carried over to the 2013–2015 budget from the 2007–2010 budget. Dashes (-) indicate data not available or not reported.

Source	Annual Total Phosphorus Loading ($x10^3$ lbs/year)			
	Nixon et al. 1995 1982–1983[1]	Nixon et al. 2008 2000–2004[2]	Krumholz 2012 2007–2010[3]	Estuary Program 2013–2015[4]
	Direct Discharge to the Bay			
Atmospheric Deposition*	9	9	9	9
WWTF Discharge[5]	924	551	618	526
Urban Run-Off *	264	264	383	383
Groundwater Discharge*	-	-	-	-
Rivers[5]	1,399	951	712	205
Total	**2,596**	**1,775**	**1,722**	**1,123**

[1] 1982–1983 budget numbers from Tables 16 and 19 in Nixon et al. (1995).
[2] 2000–2004 budget numbers from Table 5.15 in Nixon et al. (2008).
[3] 2007–2010 budget numbers from Table 3-1 in Krumholz (2012).
[4] 2013–2015 budget numbers from Tables 4, 7, 8, and 9.
[5] The term "WWTF Discharge" only includes WWTFs that discharge directly to the Bay. WWTFs discharging to rivers are included in the Rivers term along with other sources to rivers (run-off, groundwater, atmospheric deposition).

RIVERINE LOADING

River loading for both nitrogen and phosphorus declined across the Watershed. For nitrogen, loadings declined 61 percent since 1982–1983, and for phosphorus, loadings declined 85 percent since 1982–1983. These large declines may be due to a combination of factors: (1) changes in how the Taunton River and unmeasured rivers elements were calculated, (2) reductions in base flow at the USGS river gages, and/or (3) load reductions from the WWTFs that discharge to the rivers.

The Taunton River loadings were calculated differently between Nixon and colleagues (1995, 2008) and Krumholz (2012). The Nixon and colleagues (1995, 2008) budgets include the drainage area downstream of the gage at Bridgewater to the mouth of the Taunton River in Somerset (Figure 1), and the Estuary Program followed this method (Table 2). Krumholz (2012) took a different tactic and assigned the large portion of the Taunton that is ungaged to the unmeasured rivers category (see below). This reassignment makes it difficult to directly compare Taunton River loadings across years, and in fact Tables 8 and 9 report an increase in nitrogen and phosphorus loadings between 2007–2010 and 2013–2015. While Tables 8 and 9 report the findings as they were presented in Krumholz (2012), the Estuary Program calculated a rough estimate of what the loadings from Krumholz (2012) would be if the area downstream of the gage were included. The nitrogen and phosphorus loadings would increase to $2{,}508 \times 10^3$ pounds per year and 80×10^3 pounds per year, respectively. Using this method, loadings from 2007–2010 to 2013–2015 showed a decrease of 23 percent and nine percent, respectively.

All three previous budgets have calculated the unmeasured components differently. The differences between the Nixon and colleagues' budgets (1995, 2008) has been explained earlier (see Status and Trends for Riverine Loading). In his 2007–2010 budget, Krumholz (2012) reduced the drainage area of the Taunton River and added about 250 square miles to the unmeasured rivers total drainage area. Based on estimations by Nixon and colleagues (1995), he calculated approximately $2{,}000 \times 10^3$ pounds per year in unmeasured river nitrogen loading and 188×10^3 pounds per year in phosphorus loading. These estimates far exceeded the update by Nixon and colleagues (2008). To create a direct comparison between Nixon and colleagues (2008) and Krumholz (2012), the Estuary Program added approximately 250 square miles to the unmeasured river drainage area used by Nixon and colleagues (2008) and multiplied it by the nutrient yields from Fulweiler and Nixon (2005). The Estuary Program calculated 888×10^3 pounds per year of nitrogen loading and 119×10^3 pounds per year of phosphorus loading for 2000-2004. The new unmeasured rivers loading from the 2000-2004 budget showed a decrease from the 1982–1983 loadings but was still less than the 2007–2010 loadings.

The Estuary Program returned to Nixon and colleagues (2008) for their unmeasured river component. They calculated the unmeasured area by subtracting out the drainage areas of the named rivers from the total drainage area in the Watershed (Table 2; Appendix). They also used the same nutrient yield ratios. Additionally, the Estuary Program included the same drainage area of the Taunton River as the budget by Nixon and colleagues (2008). This method resulted in unmeasured rivers and Taunton River components that were comparable to the 2000–2004 budget (Tables 8 and 9).

River flow has been quite variable since the first budget was completed using 1982 to 1983 data. Blackstone River flow has declined about 38 percent since the early 1980s (Figure 4). This decline was not steady, and the 2000–2004 and 2013–2015 periods were quite similar (Figure 4). This was also true for the Taunton River (data not shown). Therefore, while a 30-year comparison between budgets may have more error due to changes in river flow, the 2000–2004 and 2013–2015 periods are a more direct comparison.

From 2000–2004 to 2013–2015, total nitrogen river loading decreased by 55 percent and total phosphorus decreased by 78 percent. Wastewater treatment facility load reductions should account for the majority of the observed decreases. On a Watershed-wide scale, loading from WWTFs discharging to rivers decreased by 49 percent for nitrogen and 82 percent for phosphorus since 2000–2004 (Tables 4 and 7). With the loadings from population estimates removed, nitrogen decreased by 65 percent and phosphorus by 85 percent, which more than accounts for the reductions noted in river loadings.

The Estuary Program's analysis provided an interesting insight into variations in loadings from nonpoint sources. Loading from the Moshassuck River declined 67 percent in nitrogen and approximately 90 percent in phosphorus since 2000–2004. No treatment facilities discharge to the Moshassuck River, so those point-source load reductions were not a factor in the load declines observed for this river. In addition, the average river flow at the USGS gage was similar for 2000–2004 and 2013–2015: 23.2 MGD and 21.1 MGD, respectively, for a difference of nine percent. Therefore, some other factors were influencing nutrient loading reductions in the Moshassuck River.

These factors could include best management practices (BMPs) designed to decrease nutrient loading to the rivers, or a reduction of loading from sources discharging to the river. However, it is important to recognize that for rivers with WWTFs, in addition to improvements to the facilities, other factors are also contributing to the estimated temporal trend of declining nutrient loadings.

NUTRIENT BUDGETS

Analysis of nutrient budgets for the Narragansett Bay Watershed spanning from 1982–1983 to 2013–2015 revealed a general trend of declining nutrient loading—55 percent for total nitrogen and 57 percent for total phosphorus (Tables 10 and 11). As mentioned previously, the atmospheric deposition, urban runoff, and groundwater components only apply to deposition directly on Narragansett Bay, including the Providence and Seekonk River as well as Greenwich Bay and Mount Hope Bay. Atmospheric deposition has been held steady throughout all budgets due to a lack of new data. Given changes in air pollution standards, it is expected that atmospheric deposition of NO_x has decreased, but without updated data, the term was not changed in these studies. Also, although urban runoff was recalculated after the two budgets by Nixon and colleagues (1995, 2008), it was held steady to the Krumholz budget in the most recent calculations. This assumption therefore does not reflect changes in urban runoff due to changing precipitation patterns, recent land use change, and stormwater infrastructure improvements (see "Precipitation" and "Land Use" chapters).

As noted above, the Estuary Program combined the WWTF loading data that were calculated using population estimates and monitoring data. For the general nutrient budget, this combination only affects the "WWTF Discharge Directly to the Bay" element (Table 12). Data estimated based on population can be removed from all but the 1982–1983 loadings. This shows no change from the 1982–1983 to 2013–2015 nitrogen budgets, and a minimal change in the percent reduction from 2000–2004 to 2013–2015 nitrogen loading budgets. For phosphorus, the change is slightly more pronounced due to the number of WWTF loadings that needed to be estimated, resulting in a 64 percent decrease between the 1982–1983 and 2013–2015 total phosphorus budgets, which is seven percent greater than shown above (57 percent).

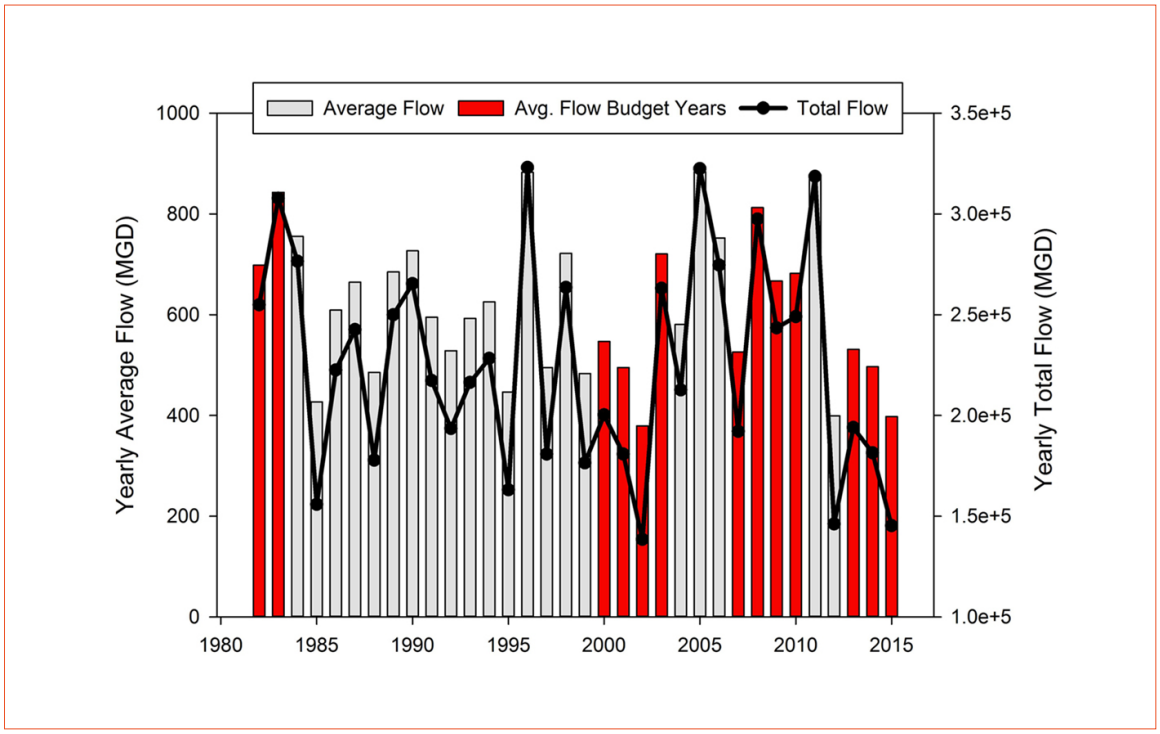


Figure 4. Yearly average flow and total flow in millions of gallons per day (MGD) for the Blackstone River from USGS gage at Woonsocket. Red bars are yearly average flow from years presented in the nutrient budgets (see Tables 8 through 11).

GENERAL CONCLUSIONS

Total nitrogen and total phosphorus loadings have declined over the past 30 years (Tables 10 and 11). Focusing on the wastewater treatment facilities as the major contributor of these reductions, there was a 57 percent decrease in total nitrogen loading and a 45 percent decrease in total phosphorus loading between the 2000–2004 and 2013–2015 budgets (Tables 4 and 7; Figures 2 and 3). Of course, as WWTF reductions approach their reduction limits, other loading sources become relatively larger contributors to nutrient loading (Tables 10 and 11). The reduction in nutrient loading is expected to have positive impacts on dissolved oxygen (DO) concentration in the water column, chlorophyll production, aquatic life, and overall environmental quality of the Bay, although the nature of these impacts is not yet fully understood (see Data Gaps and Research Needs section below and "Dissolved Oxygen" and "Chlorophyll" chapters).

The nutrient loading contributions specifically from the eleven Rhode Island wastewater treatment facilities that were the focus of the 2004 nutrient reduction statute (RIDEM 2005) have been analyzed extensively, providing a robust dataset behind the observed reductions in nutrients to the Watershed as a whole. These Rhode Island facilities and six facilities in Massachusetts along the Blackstone and Ten Mile Rivers all affect areas with low DO events in the Upper Narragansett Bay, including Greenwich Bay. Further, the USGS recently released a report analyzing nutrients (total nitrogen, nitrite + nitrate [NO_2+NO_3], and total phosphorus) loading on the Blackstone, Pawtuxet, Branch, and Pawcatuck Rivers for the last 37 years (1978–2015) (Savoie et al. 2017). The USGS utilized gages located further upstream on the rivers than were used for the Estuary Program's analysis (Savoie et al. 2017). The Blackstone River experienced a seventeen percent decrease in NO_2+NO_3 load from 1978 to 2015, a 46 percent decrease in total nitrogen load, and a 69 percent decrease in total phosphorus load (Savoie et al. 2017). On the Pawtuxet, total nitrogen loads decreased by 25 percent over the same time period, NO_2+NO_3 loads increased 80 percent, and total phosphorus loads decreased by 76 percent. The changes in loading on the rivers was most likely due to upgrades at WWTFs, and declines in other sources that discharge to these rivers (Savoie et al. 2017). The nutrient reduction findings presented by USGS support the nutrient budget findings for the Blackstone and Pawtuxet Rivers presented in this chapter. Using the 1982–1983 budget as a baseline (Nixon et al. 1995) and comparing to the 2013–2015 budget, the Blackstone River and the Pawtuxet River experienced similarly significant total nitrogen and total phosphorus loading reductions from the WWTF budget and the river budget (Tables 8 and 9).

The downward trend in total nutrient loading will likely improve the physical and biological quality of ecosystems in the Narragansett Bay Watershed. The largest threat that nutrient loading (nitrogen and phosphorous) poses for a watershed is the potential for eutrophication, which can include a series of trophic interactions that adversely impact the overall water quality of a watershed. Excessive nutrient loading in a watershed may result in depleted DO concentrations in the water column, excess chlorophyll production, and adverse impacts to aquatic life (see "Dissolved Oxygen," "Chlorophyll," "Benthic Habitat," and "Water Quality Conditions for Aquatic Life" chapters). The low DO concentration in the water results in a dead-zone where marine and aquatic life cannot survive. This can prompt a die-off of fauna that are unable to escape to more

Table 12. Wastewater discharge directly to the Bay with population-estimated data removed. All units are thousand (x10^3) pounds per year.

	Nixon et al. 1995 1982–1983[1]	Nixon et al. 2008 2000–2004	Krumholz 2012 2007–2010	Estuary Program 2013–2015
		Nitrogen		
WWTF Discharge	5,636	4,853	3,397	2,777
		Phosphorus		
WWTF Discharge	924	1,666	1,649	334

[1] The 1982–1983 budget cannot be altered because Nixon and colleagues (1995) did not indicate if they used population estimates for any data.

oxygen-rich waters, as experienced in Greenwich Bay in 2003 and frequently in other urban estuaries throughout the globe (Diaz and Rosenburg 2008). Water column stratification due to temperature or freshwater delivery to the Bay and predominant wind patterns can often exacerbate the effects of eutrophication (see "Dissolved Oxygen" and "Chlorophyll" chapters; Diaz and Rosenburg 2008). However, when nutrient loading from WWTFs and nonpoint sources are minimized through management efforts, the threat of eutrophication is decreased. This could mean reductions in nuisance macroalgae blooms or excessive chlorophyll (phytoplankton) levels and thus reduce the magnitude of DO concentration fluctuation, and the improvement of water clarity, supporting high quality and diverse aquatic and marine ecosystems throughout a watershed.

Data Gaps and Research Needs

- A monitoring strategy is needed to address data gaps in the information required to ascertain the ecosystem response to nutrient reductions. It would be expected to include additional monitoring of biological and water quality parameters, such as benthic species and phytoplankton species composition and productivity–two ecosystem components that are expected to be responsive to the changes in nutrient loading. Data should be suitable to validate relevant water quality and ecosystem models.
- Data used to estimate the contribution to nutrient budgets from nonpoint sources need to be refined. The data should include atmospheric deposition, stormwater contributions, agriculture, and other nonpoint sources.
- Continued development and validation of a water quality/ecosystem model for Narragansett Bay is needed to provide an additional tool for evaluating nutrient dynamics. Such models need to be linked with validated hydrodynamic modeling and may also need to be appropriately applied to sub-regions of the Bay, particularly embayments.
- Groundwater inputs of nutrients to estuarine and surface fresh waters in the Watershed continue to be a major data gap.
- An assessment should be conducted to determine whether there is a need to standardize monitoring of total nitrogen and total phosphorus concentrations year-round at wastewater treatment facilities.
- Further refinement of nutrient budgets is needed to provide insight into differences among seasonal load changes (winter, summer, and spring) at different scales aligned with potential ecosystem impacts, such as limiting the productivity of the Bay.

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State of Narragansett Bay and Its Watershed
2017 Technical Report

Chemical Stressor Indicators

INTRODUCTION

Please use the following citation: Narragansett Bay Estuary Program. 2017. State of Narragansett Bay and Its Watershed (Chemical Stressors Introduction, pages 190-191). Technical Report. Providence, RI.

Photo: South Branch of the Pawtuxet River, Coventry, RI (Ayla Fox)

NARRAGANSETT BAY ESTUARY PROGRAM

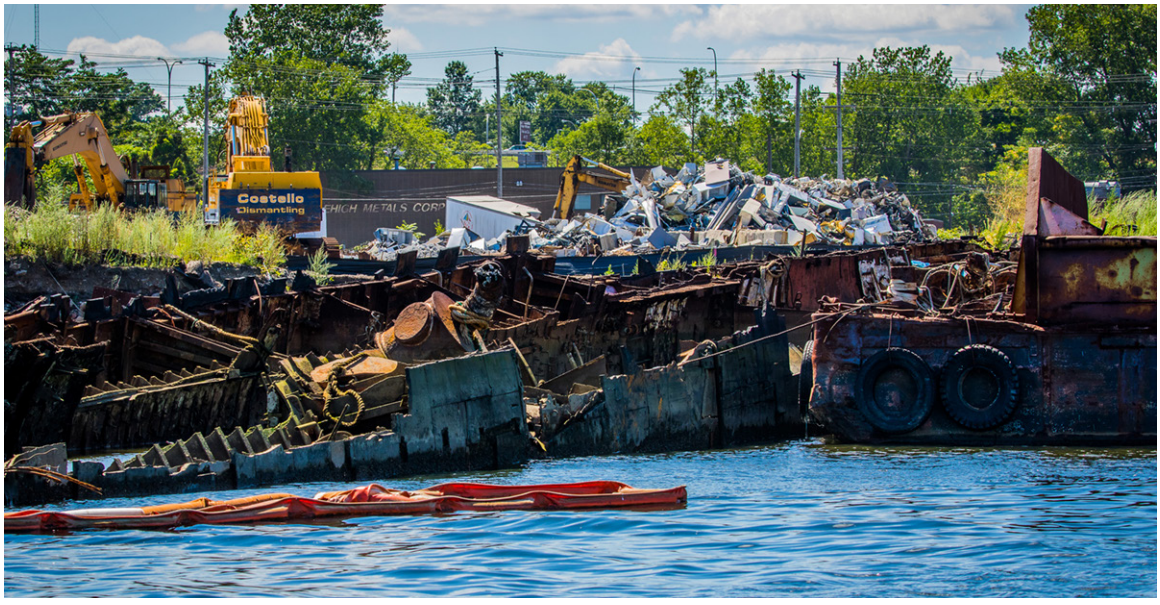
Chemical contaminants have stressed Narragansett Bay and its Watershed since the Industrial Revolution, and new contaminants have become increasingly problematic with continued growth of the human population around the Bay. This section focuses on two indicators of chemical stressors: legacy contaminants and emerging contaminants. Legacy contaminants are substances such as metals, PCBs, and pesticides that have been recognized and regulated as pollutants for many years. Although they may no longer be used, they persist in the environment for decades after their release, and their concentrations are still measurable in sediment surface samples, sediment cores, and fin and shellfish in the Bay. In contrast, emerging contaminants are chemicals that are only now starting to be evaluated for their ecological significance and risks for public health or aquatic life. They tend to be from personal care products and pharmaceuticals, or associated with industrial practices, and they have no regulatory standards associated with them.

Legacy and emerging chemical stressors negatively affect estuarine and freshwater fish communities, benthic habitats, stream invertebrates, water quality, shellfishing areas, and other aspects of the Bay and Watershed. Many contaminants, particularly metals and PCBs, can biomagnify through the food web, meaning that organisms higher on the food chain build up higher concentrations of these contaminants. Human health risks do exist, and the states of Rhode Island and Massachusetts provide guidance on how much fin and shellfish to consume to reduce the risk of exposure, especially to mercury.



Population and wastewater infrastructure influence the amounts and locations of chemical contaminants in the Bay and Watershed. Both legacy and emerging contaminants are concentrated near urban centers, although the major sources of these contaminants differ. Climate change may affect the impacts of chemical stressors through temperature and precipitation.

In this section, the Estuary Program explores the spatial and temporal trends of legacy and emerging contaminants. These changes are discussed in the context of historical trends and climate change, when possible.



Photos: Narragansett Bay Commission Lab, Providence, RI (top); Providence River Shoreline, Providence, RI (above). Photos by Ayla Fox.



State of Narragansett Bay and Its Watershed
2017 Technical Report

Chemical Stressor Indicators

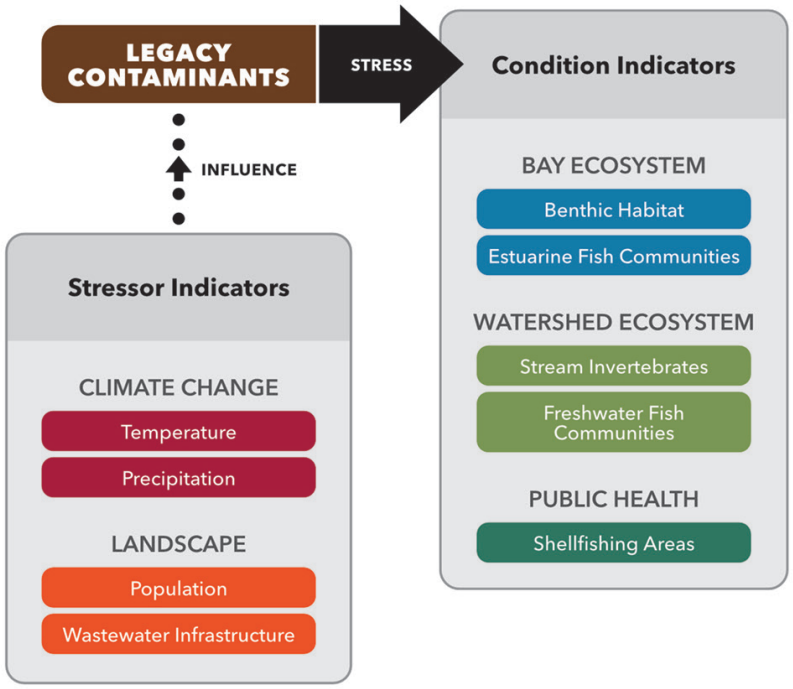
CHAPTER 9: LEGACY CONTAMINANTS

Please use the following citation: Narragansett Bay Estuary Program. 2017. State of Narragansett Bay and Its Watershed (Chapter 9, Legacy Contaminants, pages 192-210). Technical Report. Providence, RI.

Photo: Mill on the Blackstone River (Estuary Program)

NARRAGANSETT BAY ESTUARY PROGRAM

Overview



BACKGROUND

- Exposure to metals, polychlorinated biphenyls (PCBs), and pesticides causes a variety of human health issues (especially through consumption of contaminated fish and shellfish) and reduces environmental quality particularly in benthic habitats. Because of their long-term effect, metals, PCBs, and pesticides are referred to as legacy contaminants and are considered chemical stressors. Industrial manufacturing processes are major sources of these contaminants, with transport to the ecosystem by atmospheric deposition, river runoff, and wastewater discharges. While the Estuary Program recognizes the impact of metals and PCBs on freshwater ecosystems, this chapter focuses on the estuarine portions of the Narragansett Bay Watershed.

KEY FINDINGS

- ***Status:*** Although significant reductions have been made, the Seekonk River, Providence River and Taunton River sediments and some Upper Bay sediments have high concentrations of many contaminants (particularly mercury) and may still pose a human health risk through the bioaccumulation of these contaminants in locally harvested seafood.
- ***Trends:*** Generally, legacy contaminant concentrations have decreased dramatically in the last 40 to 50 years due to intense regulation and removal programs (pre-treatment and upgrades at wastewater treatment facilities) instituted in the 1970s. Sediment cores show similar patterns–rapid increases in deposition in the late 1800s and early 1900s, followed by rapid declines after the 1950s.

Introduction

Metals, polychlorinated biphenyls (PCBs), and pesticides are released into the environment through numerous sources including industrial practices, incomplete combustion, and insect control practices. Atmospheric deposition of these contaminants involves the release of small particles into the atmosphere that fall onto nearby surfaces or are captured in precipitation (rain, snow) and delivered to the ground. Many of these contaminants do not dissolve in water and instead collect in sediments, and they can be resuspended when those sediments are disturbed. When organisms come into contact with these contaminants, through ingestion or resuspension, the pollutants can bind to their fatty tissues. These contaminants then bioaccumulate–meaning that they are retained in the tissue of the organisms and passed up the food web to the apex predators, a process known as biomagnification. Additionally, the half-life of many of these compounds in the environment is quite long, making them persist for decades beyond when they were initially released (Nixon 1995). Because of their long-term effect on human and environmental quality, metals, PCBs, and pesticides are referred to as legacy contaminants and are considered chemical stressors.

Legacy contaminants are strongly associated with manufacturing practices globally and are concentrated near urban centers. They decrease with distance from the discharge points, although pesticides tend to be more ubiquitously distributed (Valiela 2006). The major sources of these contaminants to estuaries are atmospheric deposition, river runoff, and wastewater discharges (Nixon 1995, Valiela 2006). Prior to upgrades to wastewater treatment facilities (which reduced the amount of metals and contaminants in wastewater) and the creation of pre-treatment programs (which prevent the contaminants from reaching the facilities), industrial wastewater was a significant source of contaminants (Nixon 1995). Even though contamination tends to be concentrated near urban centers, evidence of metals, PCBs, and pesticides have been found in remote areas, such as the waters off Antarctica (Valiela 2006).

Legacy contaminants behave differently in water, sediments, or animal tissue, and how these contaminants react to salinity or temperature or affinity to particles affects how they impact the food web. Metals such as mercury that undergo methylation (the process by which a methyl group binds to a metal) tend to be lipophilic (combining with or dissolving in lipids and fats) and are more likely to enter animal cells and be stored in fatty tissue. On the other hand, lead sulfides and phosphates are permanent sinks of lead in soils and sediments and are not readily brought into the food chain (Valiela 2006). PCBs tend to resist degradation, attach to particles (including sediments), and are lipophilic, giving them a long-term persistence in the environment and food web. Similar behavior is observed for many pesticides. All metals, PCBs, and pesticides that are lipophilic tend to biomagnify and increase tissue concentration in organisms further up the food chain.

A variety of human health issues can result from legacy contaminants depending on level of exposure and how the chemical contacts the body (inhalation, ingestion, manual handling). Acute exposure tends to have short-term effects, mostly gastrointestinal and irritations (lung, skin). However, chronic exposure has caused genetic mutations, cancer, neurotoxicity, and endocrine disruption (Tchounwou et al. 2012, OSHA 2016). Mercury and lead, in particular, are known to cause developmental issues in unborn babies and small children. Many organizations and state health agencies have worked to educate the public on how they can reduce their exposure to heavy metals and PCBs in their diets and environment (e.g., Rhode Island Department of Health and Massachusetts Department of Public Health).

Narragansett Bay was one of the first estuaries in the United States to be intensively polluted by metals discharged to waters and released to the atmosphere from fuel combustion and industrial practices (Nixon 1995, Nixon and Fulweiler 2012). Starting in the late 1700s, Fall River and greater Providence were home to large industries of cotton textile and woolen production, machinery production, jewelry makers, and metals finisher manufacturers (Rhode Island Historical Preservation Commission 1981). The majority of these industries were originally located within the Narragansett Bay Watershed near major streams and rivers that provided both power and convenient discharge. As environmental regulations were not in place or were limited at this time, millions of pounds of various legacy pollutants made their way into Narragansett Bay. The metals-based manufacturing and subsequent pollution in Narragansett Bay became most prevalent in the mid-1800s (Corbin 1989, Nixon 1995).

Metals and PCBs have historically been used extensively in industrial and mill operations, particularly during the 1800s and early 1900s. Cadmium, copper, and lead were all emitted during the smelting process, removing the metals from ore (Nixon 1995). These emissions were usually atmospheric, but some may have been discharged directly as waste into receiving waters. Mercury was used in the

textile industry, as were chromate (chromium) and other chemicals used to produce dye. Lead was also emitted heavily by coal combustion and automobile emissions, prior to the reduction of lead in gasoline in the 1970s. PCBs had many industrial uses, including in insulating material, fire-resistant material, and coolant fluid (Valiela 2006). Production of PCBs was banned in the 1970s.

Manufacturing changed with time, from machine shops to textile processors and jewelry makers. Each shift in industry or produced goods changed the legacy contaminants that were emitted. Concurrently, the amount of impervious surface increased, particularly after the automobile was introduced in the early 1900s (Nixon 1995). The increased amount of impervious surface created easy avenues for metals to discharge directly into the Bay (see "Impervious Cover" chapter). Around this same time, wastewater treatment included chemical precipitation, creating contaminant-laden sludge. This sludge was initially dumped into receiving waters or incinerated (Nixon 1995). These processes released the legacy contaminants into the Bay's ecosystem. It was not until later in the twentieth century that pre-treatment programs were enacted to reduce the amount of legacy contaminants coming into the treatment facilities (see below). Records of atmospheric emissions and sediment cores show similar patterns—rapid increases in deposition in the late 1800s and early 1900s, followed by rapid declines after the 1950s (Bricker 1993, Nixon 1995).

Most of the practices that delivered heavy metals, PCBs, and pesticides to the Narragansett Bay Watershed are waning. Industrial and mill operations are no longer the main economic resource in Rhode Island or Massachusetts. Some of the chemicals have been banned, and alternatives used, or the use of those chemicals has fallen out of favor (such as chromate dyes). While new emissions and discharges are lower than they were in the past, these legacy contaminants still affect the environment.

Researchers at the University of Rhode Island during the 1980s and 1990s analyzed the depositional history of metals and PCBs in sediment (Corbin 1989, Nixon 1991a Latimer and Quinn 1996, Hartmann et al. 2004a and 2004b). These assessments found a north-south gradient in Narragansett Bay. Concentrations in surface sediments were greatest in the north near Providence and decreased southward in the Bay. Nixon (1991b) found that metal inputs to Narragansett Bay were greatest during wet years, owing to atmospheric deposition and overland flow delivering metals to the Bay. Latimer and Quinn (1996) found that the sediments in the Providence River Estuary accumulated PCBs at a greater rate than those in the rest of the Bay, a pattern later confirmed by Hartmann and colleagues (2004a and 2004b). They established that the major contaminant source to Narragansett Bay was the Providence River Estuary, and that the Taunton River appeared to have little impact on the western sections of the Bay (Hartmann et al. 2004a, 2004b).

In the early 1990s, Jeon and Oviatt (1991) reviewed the biological effects of pollutants on organisms within Narragansett Bay and compared them to field and laboratory experiments. This review was part of the same body of work described above. They, too, found that benthic diversity had a north-south gradient (see "Benthic Habitat" chapter) and that pollution was a driving factor in the gradient. Their review showed that toxic pollutants have declined in Narragansett Bay, particularly in the 1980s as noted by Nixon (1991b), and levels of contaminants in the Providence River Estuary were below proposed FDA alert levels (which were never formally adopted; Bender et al. 1989).

More recently, research has focused on surficial sediment. Murray and colleagues (2007) conducted an assessment of metals concentrations in surface sediments, finding that a strong spatial gradient still existed. The legacy contaminants measured included chromium, copper, mercury, nickel, lead, and zinc. Using copper as an example, the surface sediment pattern essentially showed an exponential decrease in sediment contamination from north to south in the Bay and that, in the upper reaches of the Bay, contaminant levels often exceeded sediment quality guidelines (SQG) otherwise known as Effects Range Median (ERM) and Effects Range Low (ERL) (Long et al. 1995) (Figure 1). This detailed mapping effort also pinpointed localized contaminant hotspots. The sediments in the Seekonk River and Providence River were highly contaminated with copper, those in the Upper Bay and Greenwich Bay were moderately contaminated, and those in the lower portions of the Bay were relatively clean (Figure 2).

Similarly, Murray and colleagues (2007) examined mercury concentrations in surface sediments, also finding a strong spatial gradient (Figure 3). The surface sediment pattern for mercury is similar to copper with an exponential decrease in sediment contamination from north to south in the Bay with additional localized highly contaminated hotspots in Greenwich Bay and Bristol Harbor (Figure 4). Nixon and Fulweiler (2012) reported on the findings by Murray and colleagues noting the strong spatial gradients in the upper portions of the Bay. They also noted that while the north-south gradient is clear,

there is a need to further examine the distribution of legacy contaminants from the Taunton River and Mount Hope Bay.

Murray and colleagues then worked with researchers from Roger Williams University and conducted additional sediment grab samples for mercury throughout Narragansett Bay (Taylor et al. 2012). The potential areas of high mercury exposure included the upper reaches of the Bay, including the Taunton River, Seekonk River, and Providence River (Figure 5). The results of the spatial analysis of land use and mercury levels demonstrated that the percent of total mercury that was methylmercury was significantly related to population density. This organic form of mercury is much more bioavailable and toxic than inorganic mercury. In addition, this study showed an association between higher levels of mercury in surface sediments and higher levels of mercury in the tissues of finfish harvested in the same areas (Taylor et al. 2012).

The bioavailability of other legacy contaminants (e.g., copper, lead, and cadmium) warrants further investigation. Studies of divalent trace metals that can produce toxicity in the marine environment indicate that they are often bound up as insoluble sulfides in anoxic sediments (Di Toro et al. 1990, 1991) and are only bioavailable when the total molar concentration of these metals exceeds that of the sulfide in the sediment. As an indicator of bioavailability, the Estuary Program's partners used the molar ratio of simultaneously extracted metals (SEM) to acid volatile sulfide (AVS) concentrations found in sediments by dissolving surface sediment samples in strong acid for the divalent metals cadmium, copper, nickel, zinc, and lead (Di Toro et al. 1990, 1991). Ratios higher than 1.0 indicate potential bioavailability because the concentration of metals exceeds the concentration of sulfides. The excess metals are not bound as insoluble metal sulfide compounds, are likely to be found in solution, and are therefore potentially bioavailable and toxic. Using this method

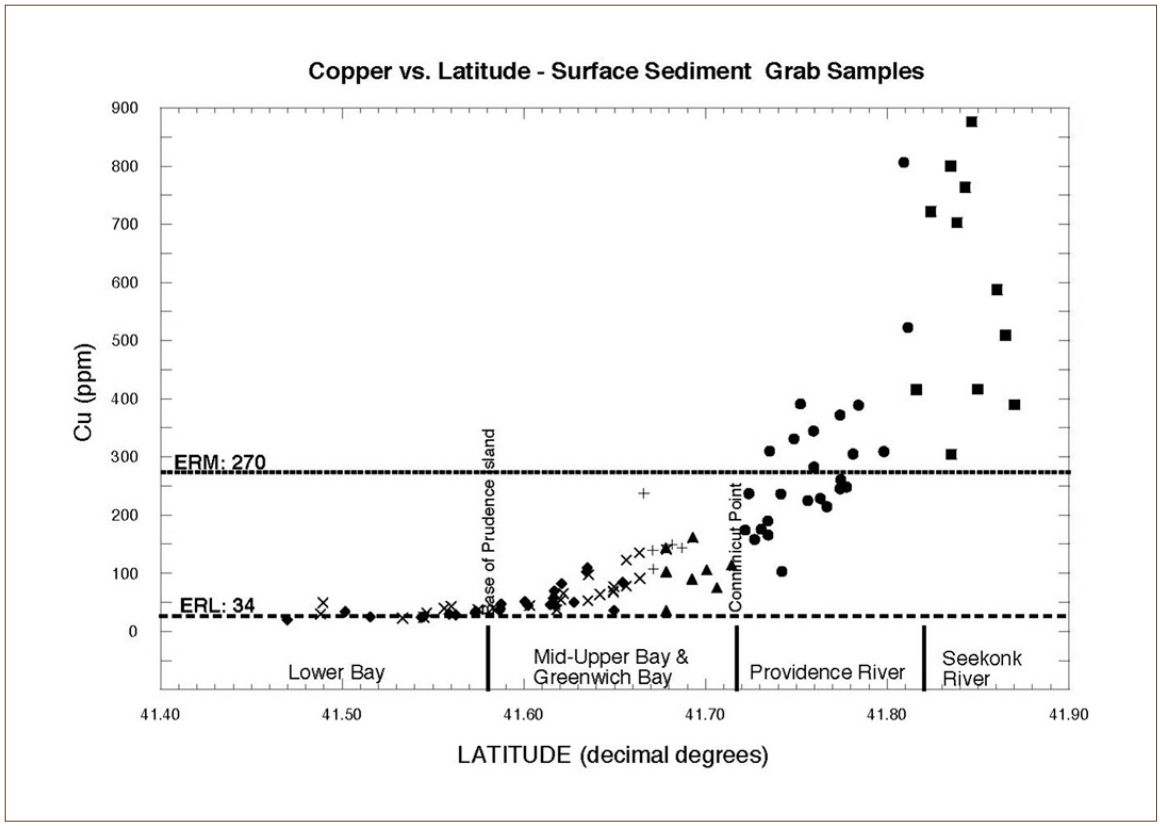


Figure 1. The gradient of surface sediment copper concentrations in parts per million (ppm) dry sediment. Symbols indicate the location of the sample: squares were from the Seekonk River, circles from the Providence River, triangles from the Upper Bay, diamonds from the West Passage, x's from the East Passage, and plus signs from Greenwich Bay. The gradient shows an exponential north-to-south decrease in contamination using the sediment quality guidelines (SQGs) ERM and ERL (Long et al. 1995). Source: Murray et al. 2007

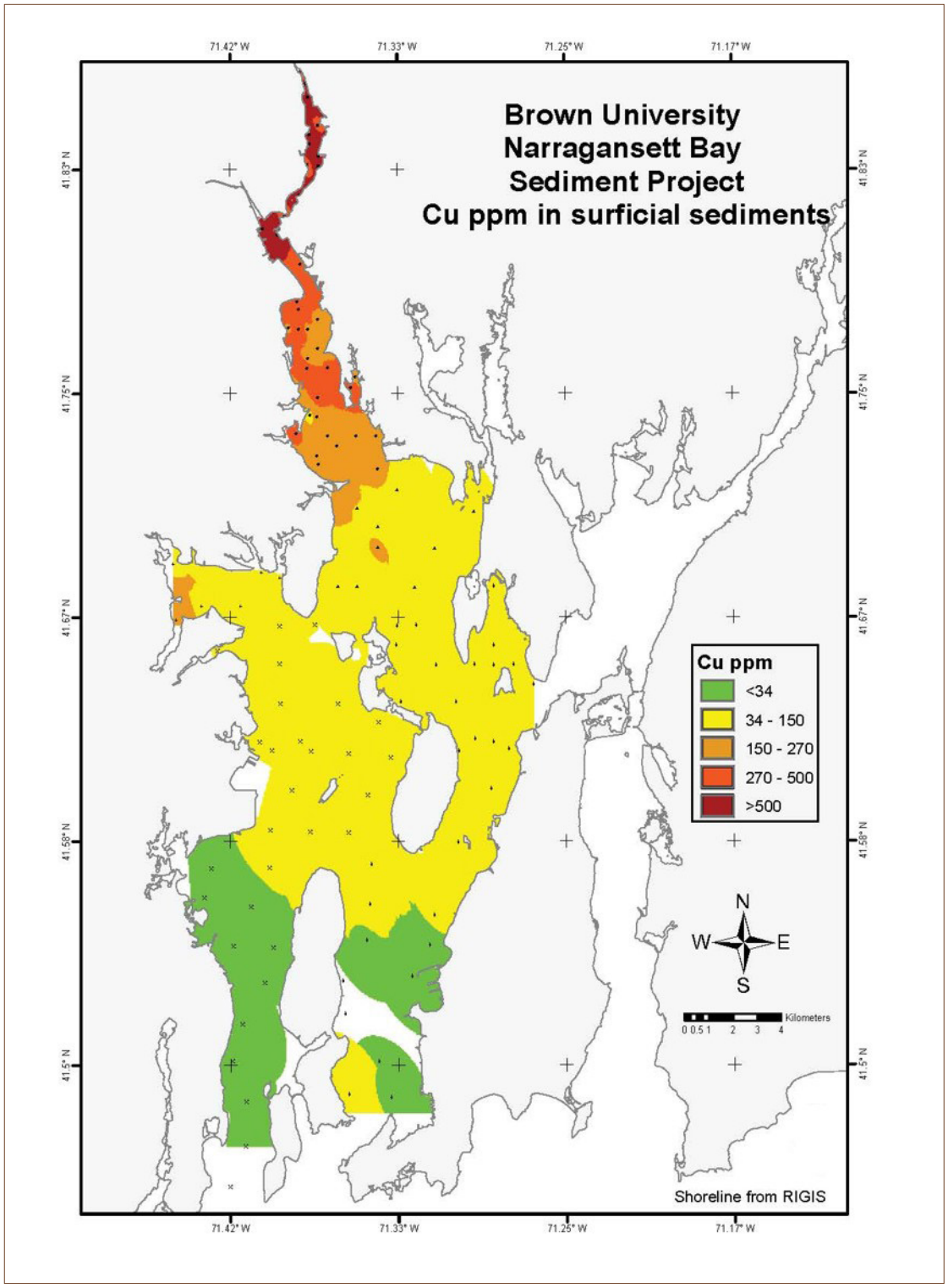


Figure 2. A map of surface sediment copper concentrations in parts per million (ppm) dry sediment. Symbols indicate the location of the sample: squares were from the Seekonk River, circles from the Providence River, triangles from the Upper Bay, diamonds from the West Passage, x's from the East Passage, and plus signs from Greenwich Bay. The map reveals regions of particular concern with red color for values above the ERM level (270 ppm), orange and yellow colors for values between the ERM and ERL level, and green color for values below the ERL (34 ppm). Source: Murray et al. 2007

developed by Di Toro and colleagues, studies done in Narragansett Bay of the ratio of these trace metals to sulfide indicated that few sites were likely to have excess metals that would be bioavailable and could impact the biota, particularly near contaminant sources (Figure 6; Laliberti and King unpublished data). However, if contaminated and sulfide-rich sediments are disturbed either during major storms, or by dredging activities, then the sulfides can oxidize and release toxic metals to the water column producing biological impacts. The potential biological impacts of such disturbance events would need to be monitored.

A critically important study was completed recently by Taylor and Williamson (2017) on mercury contamination and its transfer up the food web to targeted fish species that could be consumed by humans (black sea bass, bluefish, scup, striped bass, summer flounder, tautog, and winter flounder). This study provided strong evidence that about half of the anglers and their families harvesting coastal fish from the Narragansett Bay system and adjacent waters experienced mercury exposures that exceeded the USEPA reference dose and were at risk of impacts from mercury neurotoxicity. Taylor and colleagues (2012) and Taylor and Williamson (2017) have clearly shown that mercury found in contaminated sediment is bioavailable to marine organisms, and it bioaccumulates and biomagnifies sufficiently to pose a human health risk.

Pesticides have been used extensively since the mid-1900s for control of disease-carrying insects and for insect pest control in both agricultural and developed areas. In Narragansett Bay, using a core from Apponaug Cove in Greenwich Bay, Hartmann and colleagues (2005) determined that DDT and chlordane were present in sediments starting in the late 1940s to early 1950s. DDT concentrations in sediment cores peaked in the early 1970s, around the time of their ban in 1972, and chlordane appeared to be declining (Hartmann et al. 2005). Chlordane was banned in 1988, twelve years before the sediment cores were collected. Sedimentation rates and bioturbation made it difficult for Hartmann

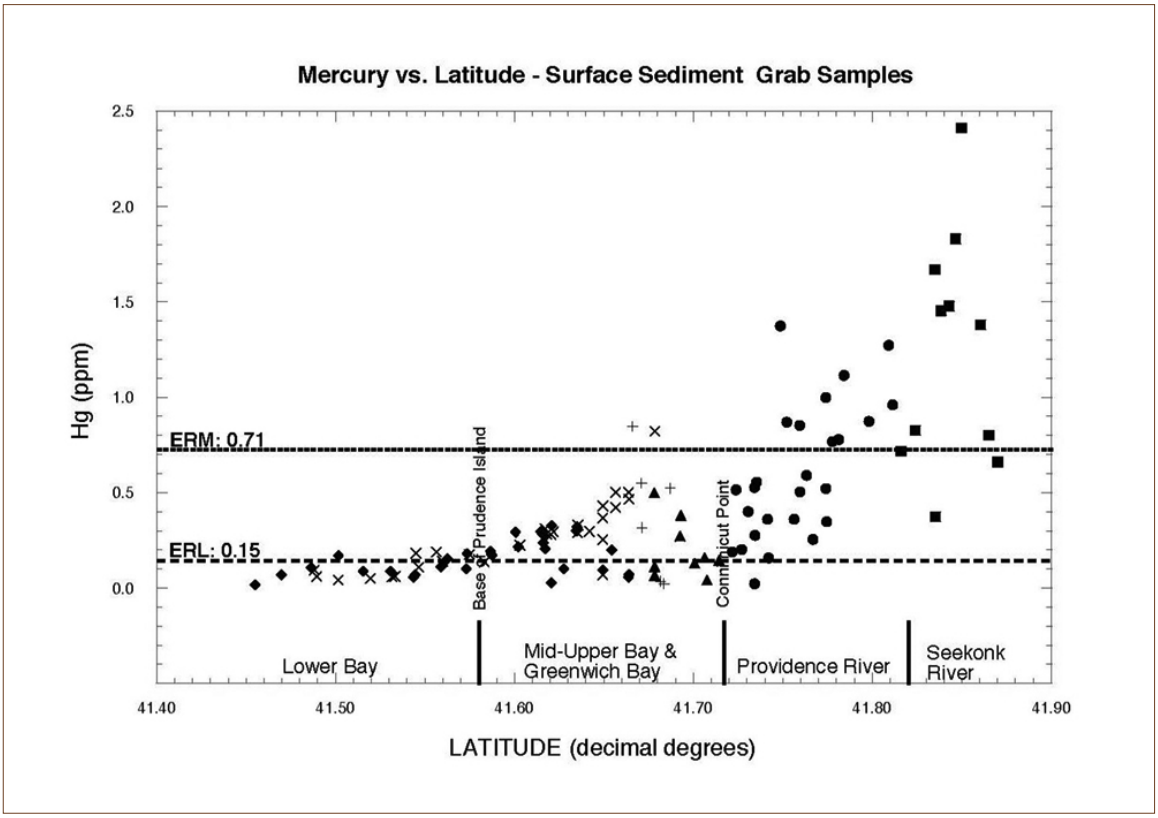


Figure 3. The gradient of surface sediment mercury concentrations in parts per million (ppm) dry sediment. Symbols indicate the location of the sample: squares were from the Seekonk River, circles from the Providence River, triangles from the Upper Bay, diamonds from the West Passage, x's from the East Passage, and plus signs from Greenwich Bay. The gradient shows an exponential north-to-south decrease in contamination using the sediment quality guidelines (SQGs) ERM and ERL (Long et al. 1995). Source: Murray et al. 2007

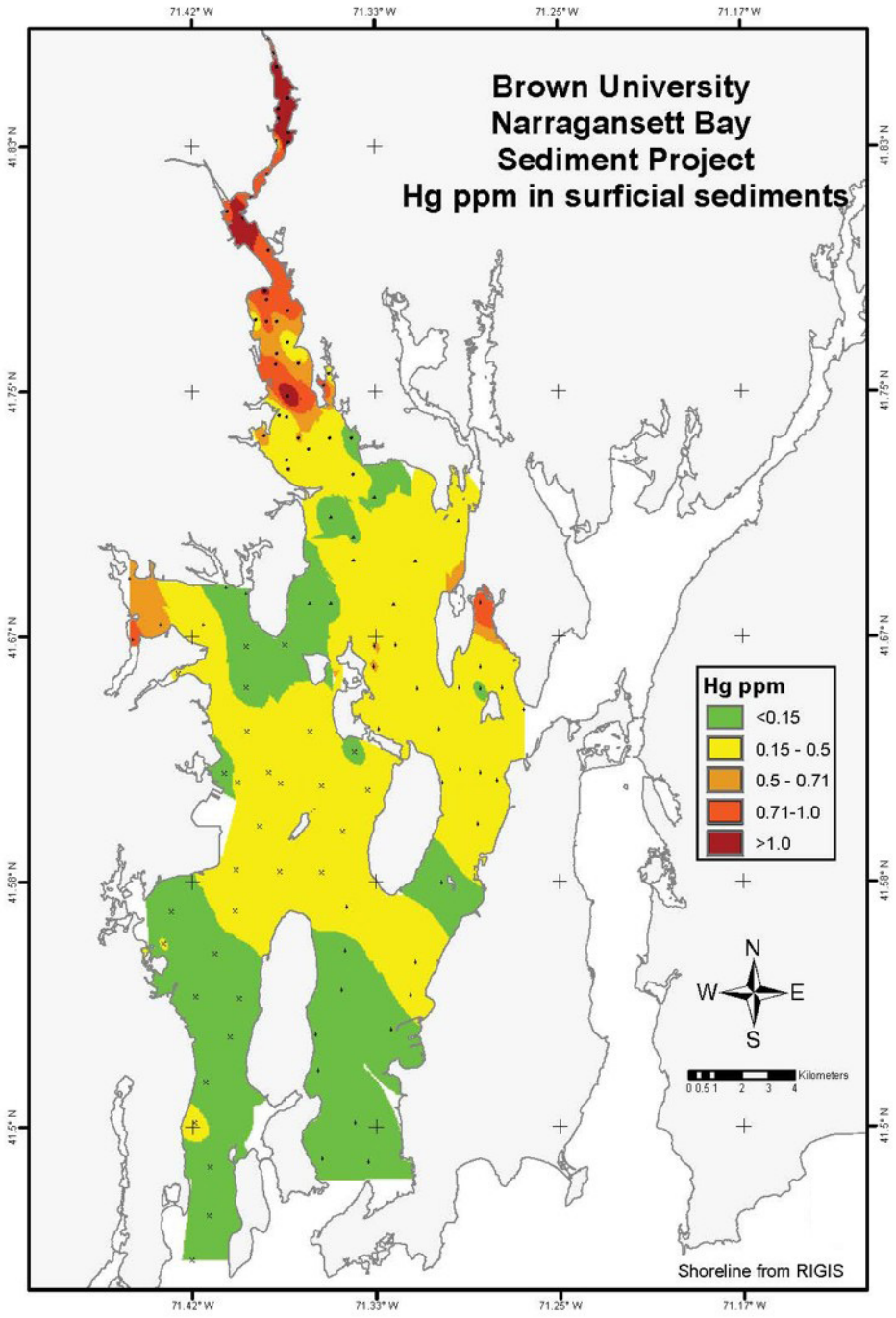


Figure 4. A map of surface sediment mercury concentrations in parts per million (ppm) dry sediment. Symbols indicate the location of the sample: squares were from the Seekonk River, circles from the Providence River, triangles from the Upper Bay, diamonds from the West Passage, x's from the East Passage, and plus signs from Greenwich Bay. The map reveals regions of particular concern with red color for values above the ERM level (0.71 ppm), orange and yellow colors for values between the ERM and ERL level, and green color for values below the ERL (0.15 ppm). Source: Murray et al. 2007

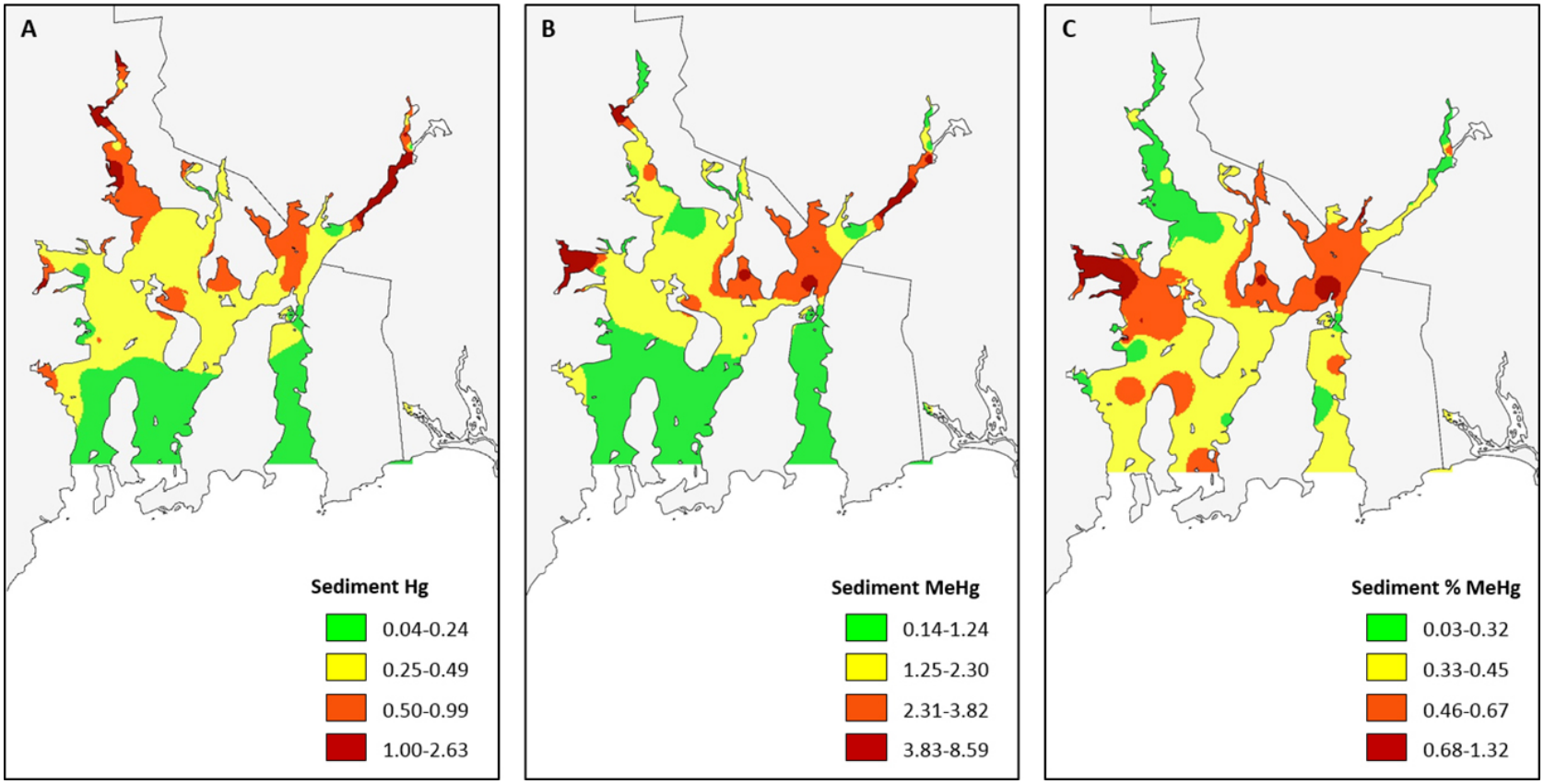


Figure 5. (A) Total surface sediment mercury (Hg) concentrations in parts per million (ppm) dry weight. Note that the ERM = 0.71 ppm and the ERL= 0.15 ppm, so most of the Bay was highly to moderately contaminated with mercury. (B) The surface sediment concentration of methylmercury in ppm dry weight. (C) The percent of total mercury that was methylmercury. Areas with the dark shades (high percent) were areas of particular concern. Source: Taylor et al. 2012

and colleagues (2005) to determine if chlordane peaked near its ban date, much like DDT.

It should also be noted that the Estuary Program decided not to utilize data and studies of metals from water column samples. In the 1980s, researchers at the University of Rhode Island compiled (Kester et al. 1987) and analyzed (Bender et al. 1989) trace metal concentration data from Narragansett Bay waters and determined that concentrations were higher in the upper Bay and decreased down bay. In general, even the highest concentrations did not exceed water quality criteria, and the waters of the Bay were not likely to produce biological impacts. More recent research (Kozelka and Bruland 1998) confirmed the north-south gradient for copper, zinc, cadmium, and lead–metals that can be toxic when present in high concentrations in the environment. This work also showed that these metals in the "dissolved" form were generally bound to other materials (e.g., organic matter) and were not particularly bioavailable and unlikely to cause toxicity. The limited studies of metals in the water column suggest that the waters of Narragansett Bay in recent decades are relatively clean and unlikely to cause adverse effects. However, studies of legacy contaminants from water samples are expensive and produce data that is noisy and difficult to interpret in terms of status and trends. On the other hand, studies of legacy contaminants in sediments (surface samples and dated sediment cores) and organisms (e.g., mussel tissue) are integrated over longer time frames, and are therefore less variable and better suited for determining environmental status and trends.

In this chapter, the Narragansett Bay Estuary Program reports on the available data concerning legacy contaminants in Narragansett Bay. These data include results of sediment core and mussel tissue analysis for key metals (cadmium, chromium, copper, lead, and mercury), PCBs, and pesticides. While the Estuary Program recognizes the impact of metals and PCBs on freshwater ecosystems, this chapter focuses on the estuarine portions of the Watershed. The results presented in the Status and Trends section of this chapter will be discussed in the context of previous work (above) and how legacy contaminant bioavailability may be altered by climate change.

Methods

To examine legacy contaminants, the Narragansett Bay Estuary Program worked with its partners to analyze recent data in Narragansett Bay involving dated sediment cores and tissue from blue mussels (*Mytilus edulis*).

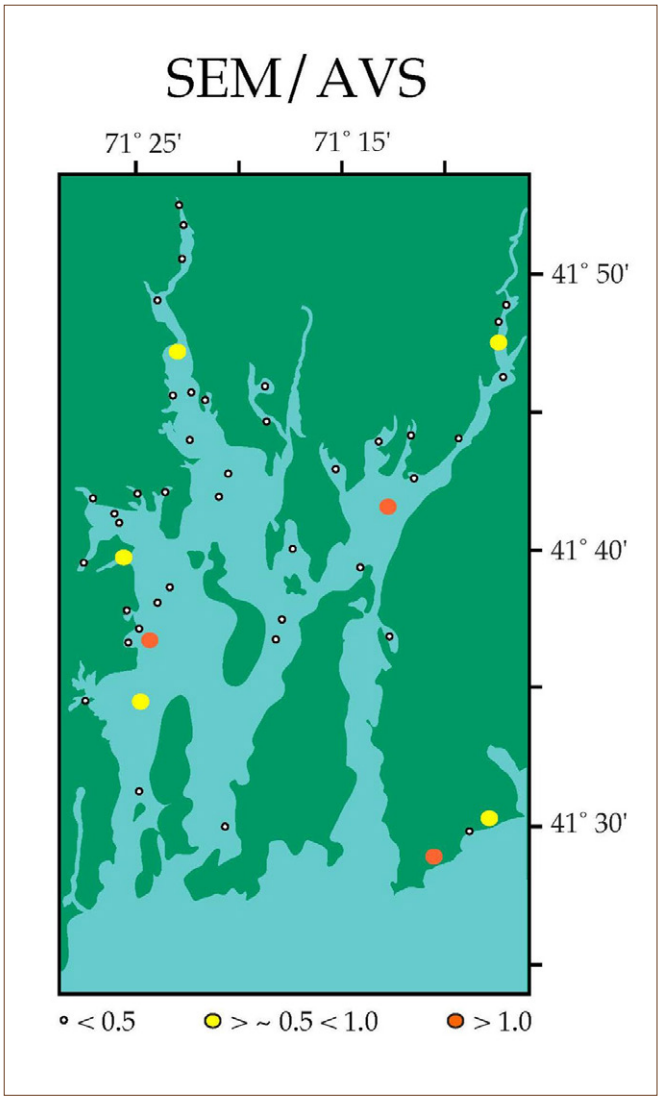


Figure 6. The molar ratio of simultaneously extracted metals (SEM) to acid volatile sulfide (AVS) for cadmium, copper, lead, nickel, and zinc (following Di Toro et al. 1990, 1991). Values lower than 1.0 indicate that metals are bound up as insoluble metal sulfide compounds and are not likely to be bioavailable and toxic. Conversely, values higher than 1.0 indicate the potential for bioavailability and toxicity. Source: Laliberte and King unpublished data

DATED SEDIMENT CORES

Several dated sediment cores in various sections of Narragansett Bay were analyzed to provide an historical analysis of legacy contaminant concentrations. A dated sediment core from the Seekonk River (Corbin 1989, King unpublished data) was used to measure metal concentrations for specific legacy contaminants (copper, lead, cadmium, and chromium) and the age model used is based on dating done by

Cs137, Pb210, pollen stratigraphy, and radiocarbon dating. ERM values for these metals were compared to metal concentration results from the 1770s to modern-day results. A high-resolution record from a dated sediment core near the Field's Point sewage treatment plant in the Providence River was examined for trends in trace metals (copper and lead) and PCBs (Cantwell and King unpublished data). The age model was based on Cs137 and Pb210 from the 1940s to 2015, and a change in sediment type near the base that indicated dredging in 1941. Lastly, a dated sediment core at Field's Point in the Providence River was examined for chlorinated pesticides and results, from every three to four years from 1941 to 2015, were summed for trans-chlordane, cis-chlordane, and 4,4'-DDE (Cantwell and King unpublished data).

MUSSEL DATA

Two approaches were utilized to analyze metals and PCB concentrations in blue mussel tissue. First, using an earlier USEPA study by Phelps and Galloway (1979) as a model, the Narragansett Bay Commission (NBC) designed a study, unpublished, to measure legacy contaminants. Blue mussels were collected from Fort Getty, Jamestown (Lower West Passage) during the fall seasons of 2008, 2009, and 2012. As a control, a portion of each set of mussels collected were immediately put on ice and frozen and brought to the NBC laboratory for analysis. Remaining mussels were then deployed in cages at Conimicut Point (Upper Bay), in approximately the same location as used by Phelps and Galloway (1979), for a four-week time period. Mussels were then collected and analyzed for metals concentrations.

Second, a research project evaluated the use of indigenous mussel populations as sentinel organisms for indicating levels of pollutants in coastal marine waters (Goldberg et al. 1978). The National Oceanic and Atmospheric Administration (NOAA) adapted this research effort and created the Mussel Watch program. This program has sites throughout the United States and collects blue and ribbed mussels as well as surface sediment samples to analyze concentration levels of 25 different heavy metals and 98 types of PCBs. Mussel Watch data for concentration of metals and PCBs in Narragansett Bay were collected yearly from 1986 to 1999, and then every other year until 2011. The Estuary Program analyzed Mussel Watch data from three sites within Narragansett Bay–Dutch Island (Lower West Passage), Dyer Island (Middle East Passage), and Patience Island (Upper West Passage).

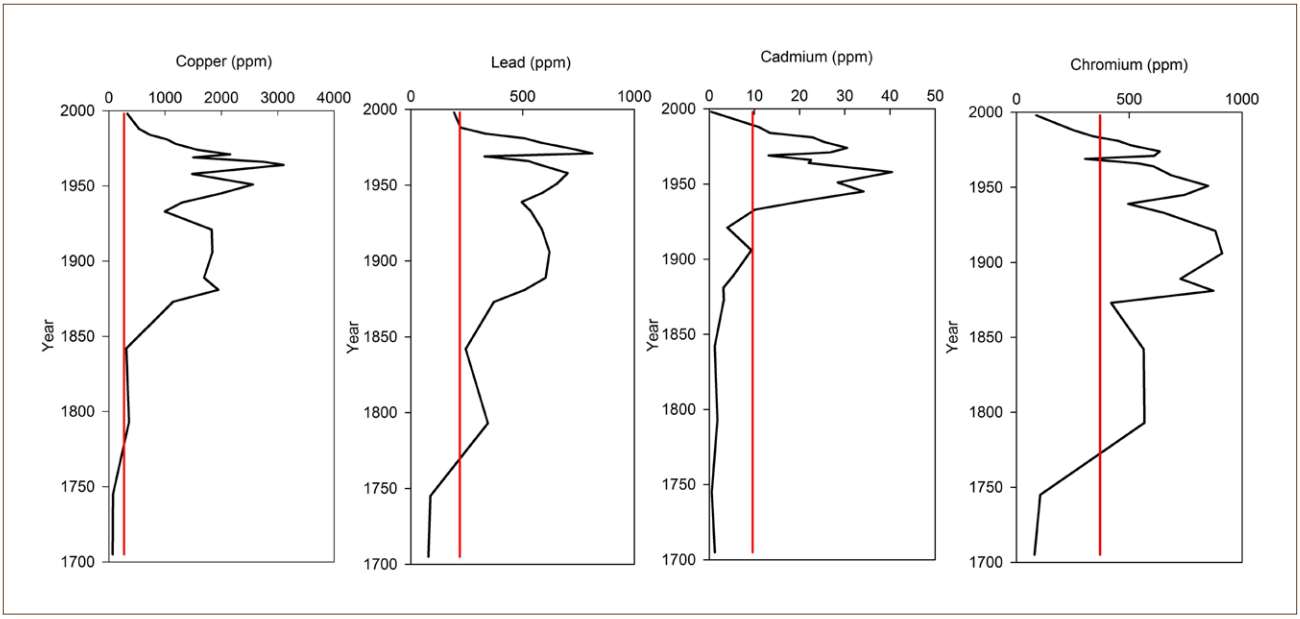


Figure 7. Metal concentrations (dry weight ppm) from a dated sediment core from the Seekonk River (Corbin 1989, King unpublished data). The age model is based on Cs137, Pb210, pollen stratigraphy and radiocarbon dating. The red lines show the ERM values for the respective metals (Long et al. 1995).

Both approaches analyzed the same metals: cadmium, chromium, copper, and lead. Total PCB concentration trends were also analyzed in the Mussel Watch program. All metals data from the Mussel Watch program were based on one sample per metal, so no standard deviations could be calculated. PCB data included the total concentrations of all congeners tested in the Mussel Watch program.

Status and Trends

DATED SEDIMENT CORES

A dated sediment core from the Seekonk River showed the 300-year history of trace metal inputs for copper, lead, cadmium, and chromium (Figure 7; Corbin 1989, King unpublished data). This record reflected the rise during the industrial age in the Providence area followed by the decline in response to strict environmental regulations (Clean Air and Clean Water Acts) in the 1970s. After 1990, contaminant levels decreased below Sediment Quality Guideline (SQG) levels that would likely be associated with biological impacts. These trends have been described by Corbin (1989) and Nixon (1995) as reflecting industrial usage and improvements in environmental regulation. These sediments were contaminated based on regulatory standards, but levels have fallen in recent decades to levels at or below SQG levels at which biological impacts are likely to be observed (ERM).

A very high-resolution record from a core near the Field's Point sewage treatment plant showed trends in trace metals and PCBs that revealed improvements achieved by environmental regulation (Figure 8; Cantwell and King unpublished data). At this site, SQGs were achieved around 2000 for lead and only recently, in 2015, for copper and PCBs.

Several chlorinated pesticides measured in the Field's Point core provide insights into their long-term use and persistence (Figure 9; Cantwell and King unpublished data). Here, summed concentrations of chlordane and ∑DDT rapidly increased from 1940 to maximum values in the mid-1950s, reflecting their high volume of usage. Declines in the late 1950s onward reflect limitations placed on the use of DDT as concerns over its usage increased, with an eventual ban in 1972. Chlordane usage continued until its ban, as seen by the decline in concentration in 1988. Measurable concentrations were present at the surface of the core, likely from land-based

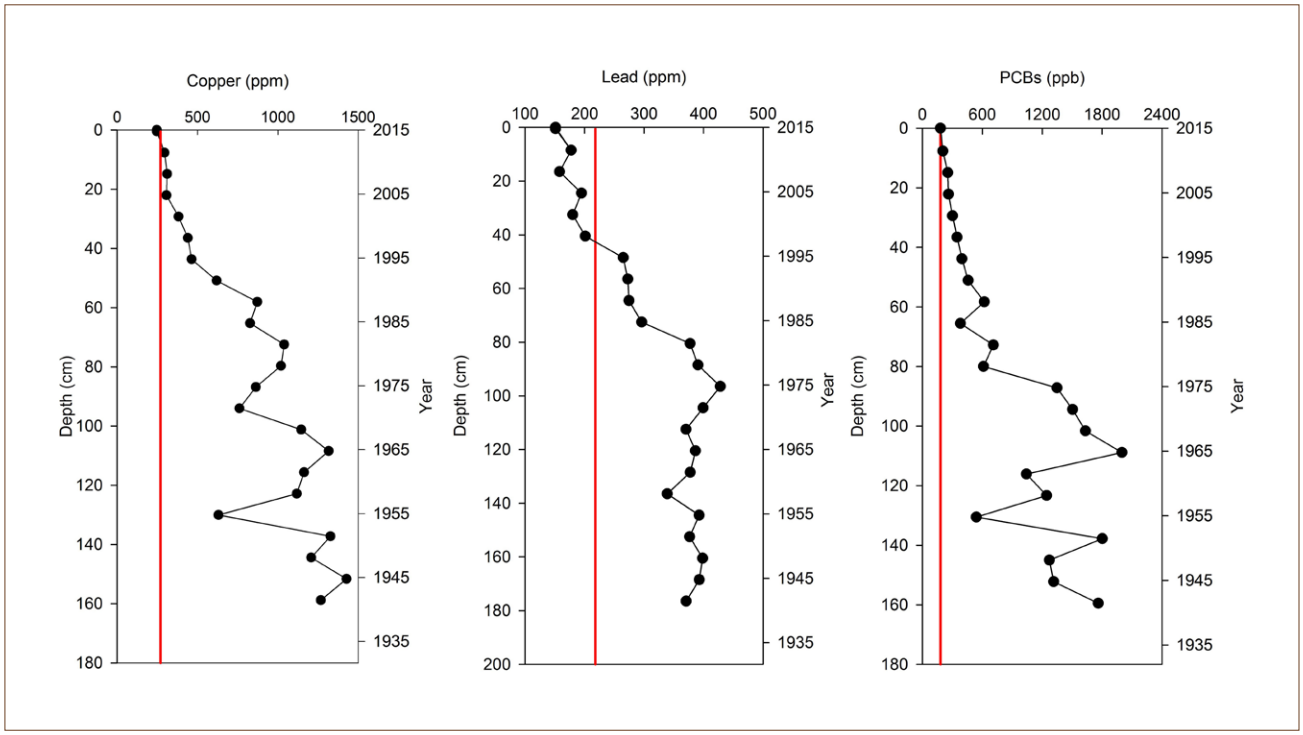


Figure 8. Metals concentrations (ppm) and PCBs concentrations (ppb) from a dated sediment core from Field's Point in the Providence River (Cantwell and King unpublished data). The age model is based on Cs137, Pb210, and a change in sediment type near the base that indicates dredging in 1941. The red lines indicate the ERM values for the legacy contaminants (Long et al. 1995).

residuals of both pesticides continuing to enter the Bay, testifying to their persistence.

MUSSEL TISSUE

The following results were developed by the Narragansett Bay Commission (NBC) from an updated study by Phelps and Galloway (1979). Metals concentrations measured in mussels deployed at Conimicut Point (Upper Bay) in each of the three more recent NBC study years were compared to samples collected in 1976 (Phelps and Galloway 1979) and analyzed for statistical differences using ANOVAs (Figure 10). The 2008, 2009, and 2012 concentrations in mussel tissue for all metals analyzed in all three study years were significantly lower when compared to the 1976 samples, with the exception of chromium in 2009. These data suggest a trend of decreasing metals concentrations in the Upper Bay in response to water quality improvements over the 30-year time interval. Nickel showed the greatest percent reduction with an average decrease of 88 percent, while copper had the least decrease of fourteen percent. It is also interesting to note that the more recent concentrations of cadmium, nickel, and zinc in the study mussels from Conimicut Point were lower than those from Jamestown North in 1976, a site further down the Bay.

Current (2008, 2009, 2011, and 2012) concentrations of metals and total PCBs from NBC and the NOAA Mussel Watch data are included in Table 1. Cadmium, chromium, mercury, and lead were among the metals with the lowest concentrations in mussels and copper had the highest.

Of the five metals analyzed during the study period (1976 to 2012), cadmium and lead had a decreasing trend (Figure 11). Between 1976 and 1986, metal concentrations decreased dramatically except copper (Figure 11). Mercury concentrations appear to have decreased at most sites within the last decade (Figure 12). No sites appear to have consistently increasing concentrations in any of the metals.

Total PCB concentration in mussel tissue was highest at both Dutch and Dyer Islands in the late 1980s followed by a decline (Figure 13). Concentrations at Patience Island may reflect a decreasing trend since the mid- to late 1990s.

Discussion

Radiometrically dated sediment cores from areas of the Bay with high sedimentation rates and limited mixing by deep bioturbation can provide useful information on both the status and trends in environmental quality over the time frames of several decades to hundreds of years. Records from the upper reaches of the Bay and from coves can typically provide records of environmental quality with a resolution of a few years. This approach provides very useful information on the impacts of historical land use and industrial changes within the Bay system and of environmental regulatory actions and infrastructure upgrades.

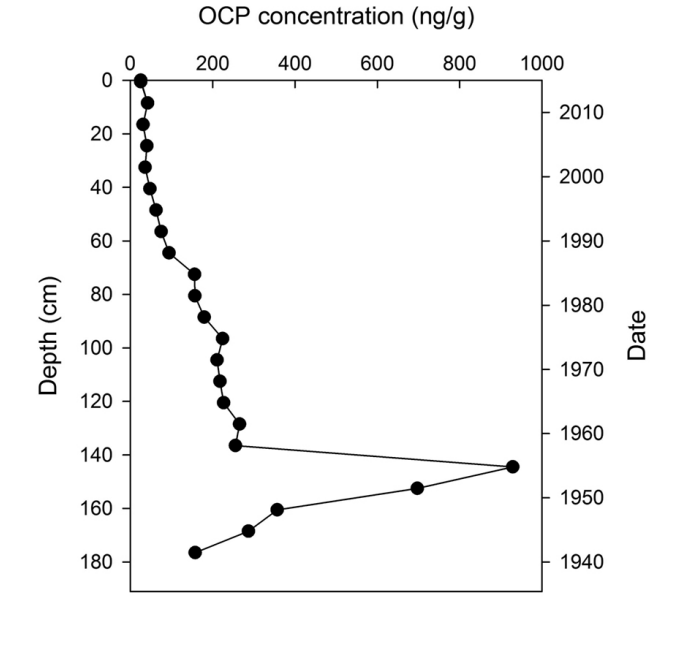


Figure 9. Summed legacy contaminant concentrations (ng/g, ppb) for organochlorinated pesticides (OCP; trans-chlordane, cis-chlordane, and 4,4′-DDE) from a dated sediment core (1941 to 2015) from Field's Point in the Providence River (Cantwell and King unpublished data).

Dated sediment cores showed a consistent pattern of legacy contamination—an increase in metals and PCB concentrations from the start of the Industrial Revolution through the early to mid-1900s, and then a decline as manufacturing practices shifted (Figures 7 and 8). Currently, metals and PCB concentrations appear to be at or near their respective ERMs (Figures 7 and 8), indicating that Bay sediments at and near the surface are getting cleaner. These findings are similar to previous findings (Nixon 1995, Hartmann et al. 2004 and 2005, Nixon and Fulweiler 2012).

In general, the mussel data from NBC's study and NOAA's Mussel Watch program indicate that metal and PCB concentrations were decreasing or remaining similar throughout the study period (1976 to 2012) (Figures 10 through 13). This trend was

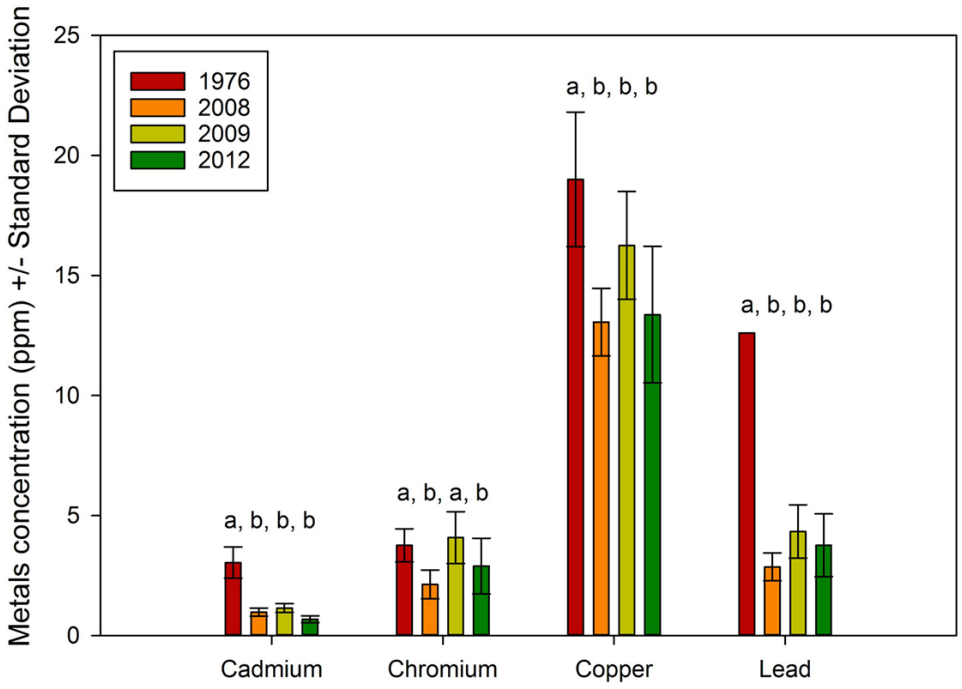


Figure 10. Comparison of metals concentrations in Conimicut Point mussels from 2008, 2009, and 2012 to results from the mussels placed in the Conimicut Point in 1976 (Phelps and Galloway 1979). Error bars were not included on lead for Phelps and Galloway data since a range (10.8 to 15.5) was given in the paper and not a standard deviation. The letters indicate ANOVA tests of significance on the average metal concentrations: different letters indicate significance levels greater than 0.05; similar letters indicate no significant difference between time periods.

Table 1. Mussel tissue average concentration for metals and total PCBs for each sample site from 2008 to 2012. NBC data (Conimicut Point 2008, 2009, 2012) include standard deviations of mussel tissue samples. Patience, Dutch, and Dyer Islands sites from 2011 are from NOAA Mussel Watch.

	Conimicut Point	Conimicut Point	Patience Island	Dutch Island	Dyer Island	Conimicut Point
Year Collected	2008	2009	2011	2011	2011	2012
Metals (ppm)						
Cadmium (Cd)	0.98±0.17	1.15±0.18	0.74	0.75	0.74	0.68±0.14
Chromium (Cr)	2.13±0.60	4.08±1.08	0.7	0.8	3.7	2.90±1.16
Copper (Cu)	13.06±1.41	16.26±2.24	10	13	10	13.37±2.84
Lead (Pb)	2.86±0.58	4.33±1.11	1.49	1.67	1.4	3.76±1.31
Mercury (Hg)	N/A	N/A	0.12	0.08	0.08	N/A
Total PCBs (ppb)	N/A	N/A	3	12	5	N/A

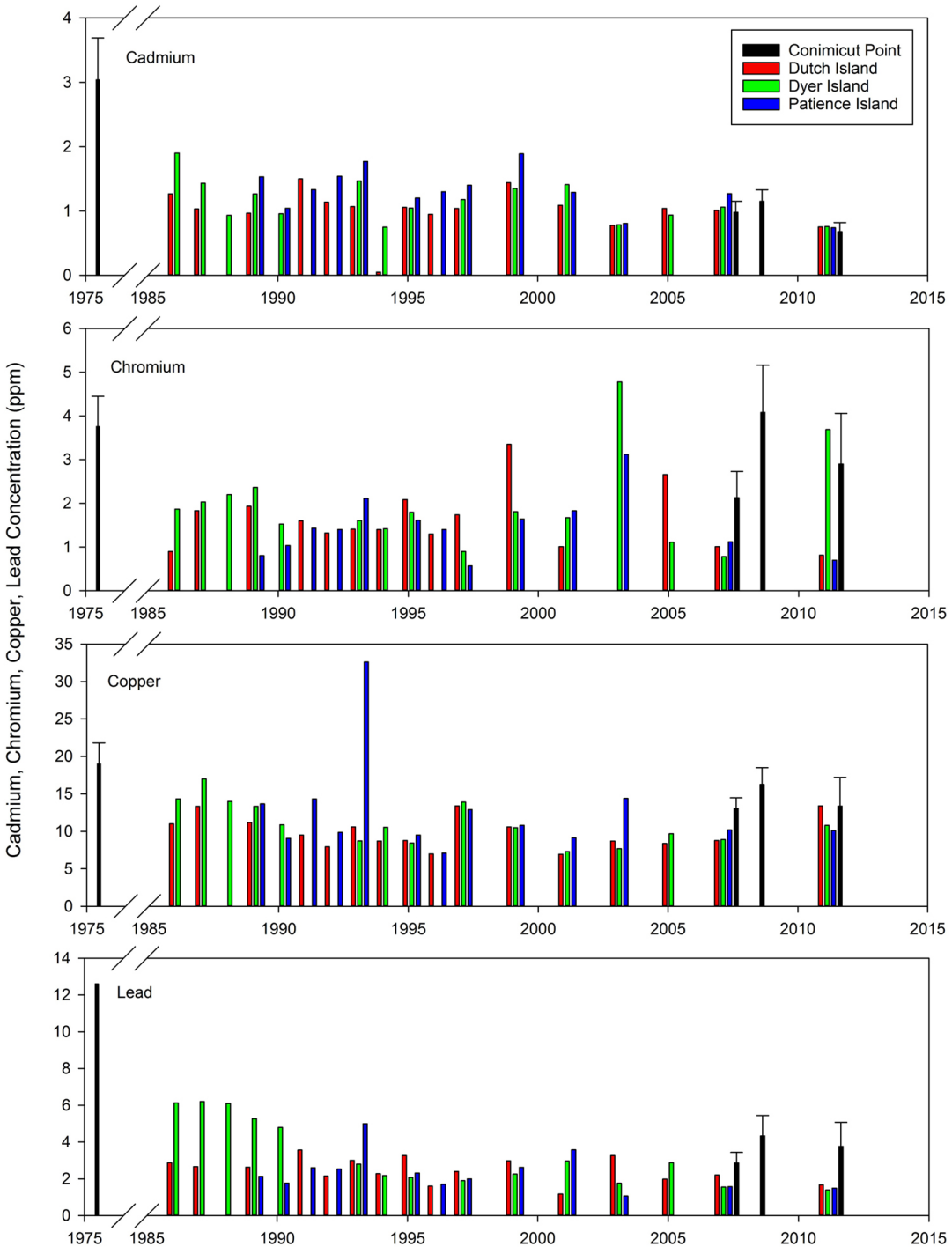


Figure 11. Concentrations (ppm) of cadmium, chromium, copper, and lead in mussel tissue. Conimicut Point values are from NBC's study comparing Phelps and Galloway data from 1976 to data collected in 2008, 2009, and 2012 (see text). Patience, Dutch, and Dyer Islands sites are from NOAA Mussel Watch.

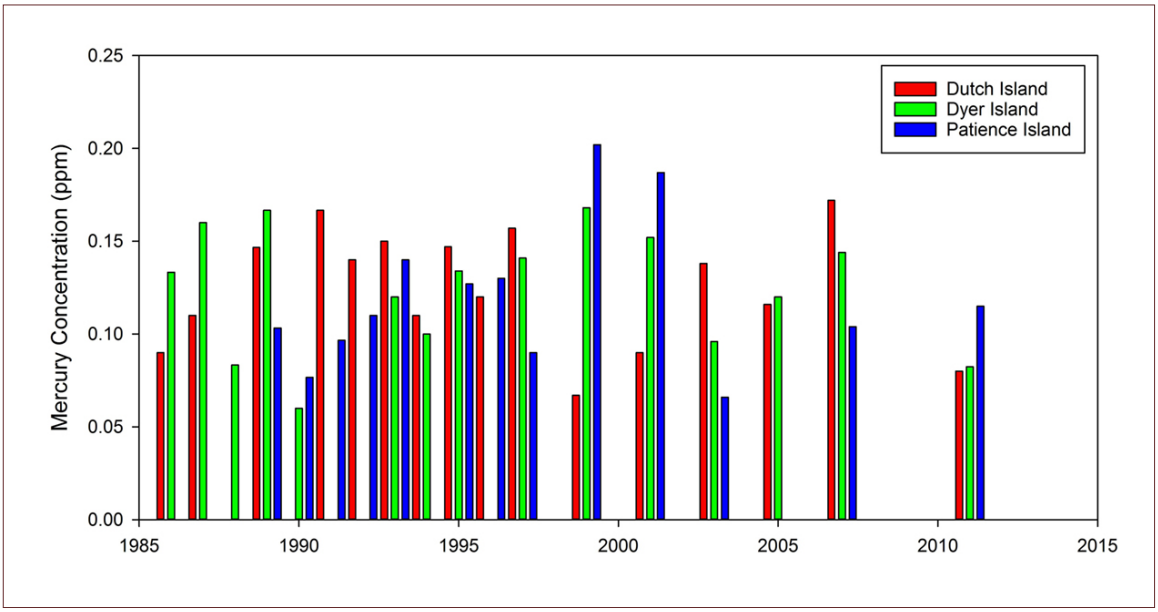


Figure 12. Mercury concentrations (ppm) in mussel tissue from NOAA Mussel Watch.

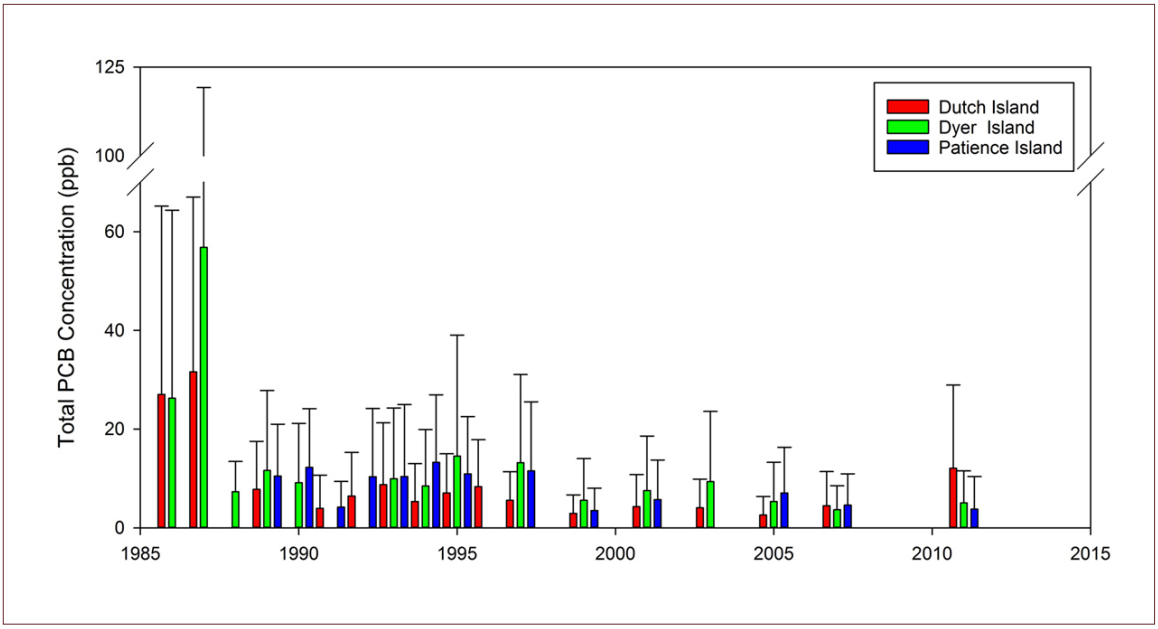


Figure 13. Total PCB concentrations (ppb) in mussel tissue from NOAA Mussel Watch.

expected since the loadings of these contaminants have been decreasing through the late twentieth and early twenty-first centuries (Nixon 1995, Nixon and Fulweiler 2012). These conclusions also compare well with a National Estuarine Research Reserve Mussel Watch trend analysis using mussel tissue collected from sites throughout the US (Lauenstein and Cantillo 2002, O'Connor and Lauenstein 2005).

Influent loadings of both copper and chromium to the Field's Point wastewater treatment facility have decreased by more than 98 and 97 percent, respectively, since 1981 (NBC 2015), although the decreases observed in the mussel data were not as great. There are several reasons these differences may exist. While both copper and chromium were used extensively in industries prevalent in Rhode Island for decades, the decrease in these types of manufacturers has reduced copper and chromium loading into the Field's Point plant greatly. Copper, however, still has a more prevalent presence within the waters of the Bay from sources such as copper piping used for water distribution and anti-fouling methods used in the marine industry.

An important point of distinction is the total mercury findings from the Mussel Watch program (Figure 12) compared to the total mercury findings from Murray and colleagues and Taylor and colleagues (Figures 3 through 5; Murray et al. 2007, Taylor et al. 2012). Mercury levels have generally declined since 1985 in the Mussel Watch data (Figure 12). However, in surficial sediment, mercury levels are elevated in most of the Bay with the mercury commonly in the bioavailable form of methylmercury (Taylor et al. 2012). Mercury values have decreased in sediments but still tend to considerably exceed ERM values in the upper reaches of the Bay (Cantwell et al. 2007). Therefore, mercury contamination poses an ongoing significant human health risk in the Narragansett Bay system (Taylor and Williamson 2017). Evaluations of other legacy contaminants that tend to be found in more available forms, taking a similar approach to that of Taylor and colleagues (2012) and Taylor and Williamson (2017), are warranted. ERM values can provide a useful indicator for focusing these studies.

Two aspects of climate change–temperature and precipitation–may have effects on legacy contaminants. Increased temperature may enhance volatility and partitioning of persistent organic pollutants (such as PCBs) from the water to the atmosphere, which could reduce exposure to aquatic biota (Noyes et al. 2009). In terms of the effect of temperature on metabolism, the literature presents several possibilities. Increases in temperature can increase metabolism, which raises the concentration of contaminants within an organism (Schiedek et al. 2007, Sokolova and Lannig 2008, Noyes et al. 2009). Temperature increases may also increase toxicity of certain contaminants, which may affect biomagnification (Noyes et al. 2009). Degradation of contaminants may increase with increasing temperature, which makes the contaminants unavailable to biota (Whitehead et al. 2009).

Precipitation is another aspect of climate change that may change the availability of metals and PCBs. Erosion and runoff may allow more bioavailable metals to reach the water column. Winter/spring precipitation is expected to increase in the Narragansett Bay Watershed by approximately 30 percent by the year 2100, and the frequency of extreme precipitation events is also expected to increase significantly (see "Precipitation" chapter). The combination of these factors is expected to increase the remobilization of contaminated sediments in the upper reaches of the Bay. Under changing hydrology conditions, increases in methylmercury production have been noted (Whitehead et al. 2009). Precipitation can alter salinity regimes in coastal waters, and then affect the chemical itself through oxidation state changes, or enhanced/reduced toxicity based on salinity levels (Noyes et al. 2009). Combined with temperature changes, the interactive effects of metals, PCBs, and pesticides are very complex and need to be understood for the specific location or organisms of interest.

In general, the water and sediment data from Narragansett Bay indicate that the water is relatively clean with respect to legacy contaminants throughout the Bay system (Bender et al. 1989), whereas contaminated sediments are found in surface sediments in the upper reaches of the Bay system (Corbin 1989, Murray et al. 2007, Cantwell et al. 2007, Taylor et al. 2012) and highly contaminated sediments are found at depth in sediment cores in the coves and upper reaches of the Bay system (Corbin 1989, Cantwell et al. 2007, King et al. 2008). The general spatial pattern of legacy contaminants can be described as clean water and contaminated sediments in the upper reaches of the Bay system, and clean water and relatively clean sediments in the mid to lower Bay. Temporal trends indicate that most legacy contaminants have decreased significantly from maximum levels in the 1950 to 1970 interval to values that are either at or below ERM values today.

Data Gaps and Research Needs

- The concentration of legacy contaminants, including mercury, in estuarine and freshwater fish and shellfish is a data gap. More studies using an approach similar to that used by Taylor et al. (2012) and Taylor and Williamson (2017) for mercury are needed to determine the human health risk posed by the uptake of legacy contaminants by fish and other human-consumed biota (e.g., shellfish). Future work would be to expand the state monitoring programs to include estuarine and near-shore fish (i.e., Taylor's work) to create a holistic assessment of mercury in commercially and recreationally important species throughout the Bay. Other legacy contaminants that need to be assessed include, at a minimum, PCBs, pesticides, and cadmium.
- The concentration of legacy contaminants in river sediments within the Narragansett Bay Watershed is a data gap that can contribute to delays in pursuing riverine restoration actions. Studies like Cantwell et al. (2014) need to be conducted to assess the amount of contaminants in the sediments and water column before and after dam removals.
- Brayton Power Plant maintained metals-monitoring data in quahogs (*Mercenaria mercenaria*) that could be incorporated into the status and trends analyses. Given Brayton Power Plant's shut down, it is unlikely this monitoring program will continue. Adding a Mussel Watch monitoring station to Mount Hope Bay would be useful in tracking legacy contaminants in that region.
- These results are framed around a north-to-south gradient, with the study sites reflecting that preference. However, sediment contaminant maps have pinpointed localized hotspots throughout the Bay—such as near the East Greenwich Wastewater Treatment Facility in Greenwich Bay—that warrant further research (Figures 2 and 4).
- The climate change section of this chapter showed that there is little knowledge of how these legacy contaminants will behave under a changing climate. While release into the environment is decreasing, these contaminants may still pose health risks due to relic deposits in sediments. Understanding how climate change will affect mobility and toxicity of these contaminants both directly and indirectly is important to inform human and environmental risk assessments.

Acknowledgments

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State of Narragansett Bay and Its Watershed
2017 Technical Report

Chemical Stressor Indicators

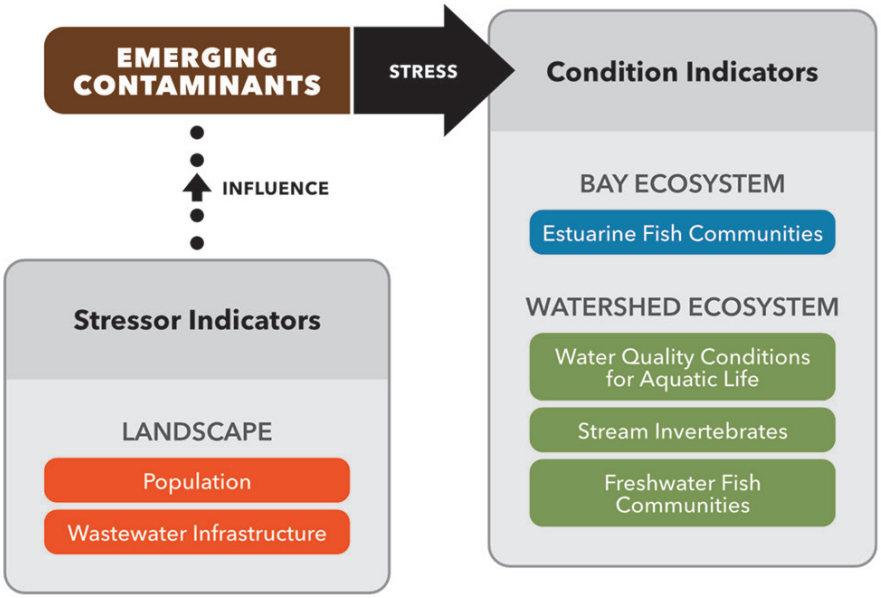
CHAPTER 10: EMERGING CONTAMINANTS

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NARRAGANSETT BAY ESTUARY PROGRAM

Overview



BACKGROUND

- The term "chemical contaminants of emerging concern" (CECs) refers to chemicals with unknown ecological effects and no associated regulatory standards. Many CECs are associated with personal care products, pharmaceuticals or industrial chemicals and have been identified as being present at low levels in natural waters such as Narragansett Bay. CECs are usually found in highest concentrations near the outfalls of wastewater treatment facilities.

KEY FINDINGS

- ***Trends:*** Sediment cores from Narragansett Bay show the recent appearance of CECs, contrasting with legacy contaminants that have declined following enactment of strict regulatory standards. CECs and many other pollutants generally decrease from north to south in Narragansett Bay because the human population and wastewater treatment facilities are concentrated in the Upper Bay.
- ***Indicator in development:*** More research is needed to identify the key CECs in Narragansett Bay and to assess their behavior, fate, and potential to impart adverse effects.

Introduction

The behavior and fate of chemical contaminants of emerging concern (CECs) in aquatic systems are not well understood. The term "chemical contaminants of emerging concern" refers to chemicals that have been identified as being present in natural waters, have no regulatory standards associated with them, and are only now starting to be evaluated for their ecological significance and risks for public health or aquatic life (e.g. estuarine and freshwater fish communities, stream invertebrates) and water quality. CECs tend to be from personal care products, pharmaceuticals, and industrial chemicals. With hundreds of new chemicals entering the commercial marketplace annually, it is likely that there are numerous CECs in the environment that have not yet been identified as such. Since new chemicals are entering the environment faster than they can be thoroughly evaluated, detailed chemical assessments often focus on highly produced compounds with properties that exhibit potential for persistence, bioaccumulation, and toxicity. Thus, many chemicals enter commercial use with limited information, resulting in their unregulated and unmonitored presence in the environment.

Many CECs are present at extremely low concentrations, making detection and assessment of their effects challenging. Consequently, knowledge about the exposure risk and their potential impacts on aquatic life and human health is limited. This is especially true for most CECs in coastal ecosystems such as Narragansett Bay.

There are numerous classes of CECs. Commonly encountered CECs are found in personal care products such as soaps, cosmetics, and detergents containing various additives: antimicrobials such as triclosan; UV blockers in sunscreens such as oxybenzone; DEET, a pesticide for human use; and fragrances such as synthetic musks. Pharmaceuticals (both over-the-counter and prescribed formulations) span a broad range of classes, including but not limited to antidepressants, antihypertensives, antibiotics, painkillers, and synthetic hormones. Specifically, there are concerns about pharmaceuticals remaining biologically active after entering the environment, since they are designed to impart therapeutic effects to humans and animals at low levels (Daughton and Ternes 1999). Finally, industrial CECs likely comprise the largest and most diverse assemblage of chemicals. These include but are not limited to flame retardants such as organophosphate esters, synthetic additives to plastics such as phthalates, bisphenol A (commonly referred to as BPA), benzotriazoles, and poly- and perfluoroalkyl substances (PFASs).

Many of the CECs associated with industrial usage, such as plastic additives, are non-polar and highly hydrophobic with low aqueous solubility–important factors controlling their environmental behavior. This results in their partitioning to organic particles during the wastewater treatment process and sorption to particles present in the waters of Narragansett Bay. This generally results in rapid and relatively efficient removal from the water column and sequestration in sediments. In contrast, other CECs such as pharmaceuticals and many personal care products are polar and more soluble in water, and thus they remain largely in the dissolved phase of the water column upon entering Narragansett Bay. Although CECs have been detected in tissues and their potential to cause adverse effects is well known, studies on the presence of such CECs in estuarine biota and their organismal effects are severely lacking (Prichard and Granek 2016).

Sources of CECs entering Narragansett Bay vary both in magnitude and type of releases. Point source inputs such as wastewater treatment facility effluents are the primary contributor due to their continuous and high-volume discharge to Narragansett Bay–approximately 200 million gallons per day (Cantwell et al. 2016a; see "Wastewater Infrastructure" chapter). These facilities were never designed or intended to treat or remove CECs from the wastewater. Influent streams to these facilities are diverse and originate from residential dwellings, commercial businesses, industrial operations, and health care facilities and, when combined, account for many of the CECs present in Narragansett Bay and other urban estuaries. In Narragansett Bay, most wastewater treatment facility effluent, approximately 90 percent, is discharged directly into upper portions of Narragansett Bay including Greenwich Bay, Mount Hope Bay, the Providence River Estuary, and the major rivers feeding the Bay, while the remainder (approximately ten percent) is discharged to Mid and Lower Bay locations (Cantwell et al. 2017; see "Nutrient Loading" and "Wastewater Infrastructure" chapters). Non-point sources of CECs may include surface runoff and residential or onsite wastewater treatment systems, as evidenced by several recent studies (Oppenheimer et al. 2012, Phillips et al. 2015, Subedi et al. 2015, James et al. 2016), but those sources likely contribute a very small portion of the total loading of CECs to the Bay.

The quantity and detail of information available on CECs in Narragansett Bay, like most other estuaries, is limited. Potential ecological effects of many of these compounds are either unknown or not well documented. Some of the earliest research on CECs in estuaries was conducted in Narragansett Bay on CECs such as benzotriazoles and triclosan.

Consequently, they have received more attention, and there is more data on their potential for adverse effects. Jungclaus and colleagues (1978) and Lopez-Avila and Hites (1980) identified numerous chemicals entering the Bay via industrial discharge to the Pawtuxet River in Cranston, Rhode Island. Some of those compounds were benzotriazoles, which comprise a broad class of chemicals used as UV stabilizers in plastics and as corrosion inhibitors for metals. Due to their long-term discharge, high concentrations, and local high-volume production, benzotriazoles are arguably one of the first studied CECs in Narragansett Bay. Subsequent research in Narragansett Bay focused mainly on benzotriazoles and examined their occurrence in clams (Pruell et al. 1984), sediment-binding mechanisms (Reddy et al. 2000), and depositional history (Hartmann et al. 2005). Recently, Cantwell and colleagues (2015) identified long-term trends and persistence of individual benzotriazoles and confirmed the presence of several anti-corrosive benzotriazoles not previously reported in Narragansett Bay.

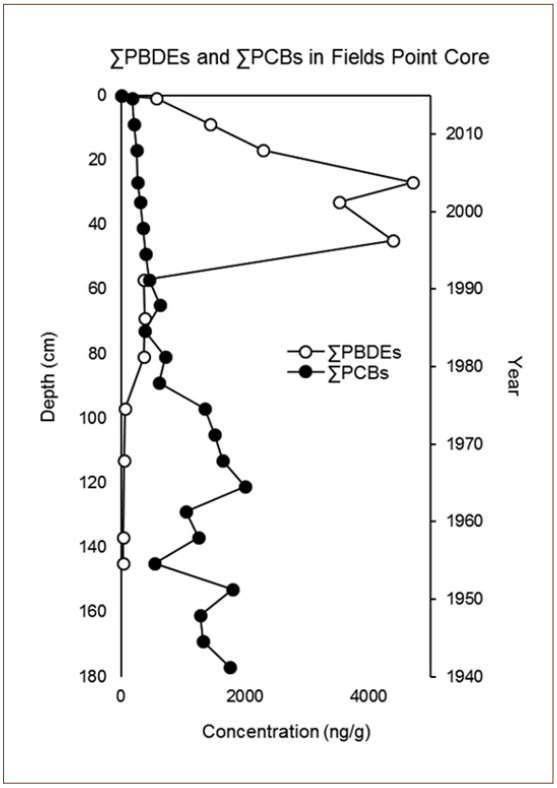


Figure 1. Concentration of PBDEs (polybrominated diphenyl ethers, flame retardants) and PCBs (polychlorinated biphenyls) from a sediment core at Field's Point, Narragansett Bay. Source: Cantwell (unpublished data)

Lopez-Avila and Hites (1980) also identified triclosan, a highly used antimicrobial CEC added to many personal care products such as soaps, detergents, and cosmetics. Cantwell and colleagues (2010) measured triclosan in Narragansett Bay and other urban estuaries, documenting its accumulation and persistence in sediments. Sacks and Lohmann (2012) and Perron and colleagues (2013) measured triclosan and methyl-triclosan using passive sampling technology. Finally, Katz and colleagues (2013) identified sources and modeled the spatial distribution of triclosan in the surface water and sediments of Greenwich Bay.

There is limited research detailing water column concentrations of other CECs in Narragansett Bay. Polybrominated diphenyl ethers (PBDEs) are a class of highly produced flame retardants used in many products such as furniture, foam, and plastics, and they were measured in the water column of Narragansett Bay by both Sacks and Lohmann (2011) and Perron and colleagues (2013). Both studies reported very low concentrations of triclosan and flame retardants in the waters of Narragansett Bay. Historical trends of contaminants measured in sediment cores from Narragansett Bay show the recent appearance of PBDEs, contrasting with legacy contaminants such as PCBs that show sustained decline due to enactment of strict environmental regulatory standards (Figure 1; see "Legacy Contaminants" chapter).

Other CECs measured include alkylphenols (Sacks and Lohmann 2011), which are components of many personal care products such as soaps, detergents, and pesticides. Recently, numerous PFASs have been found at sites throughout the Narragansett Bay Watershed (Zhang et al. 2016). Pharmaceutical compounds have also come under investigation for their presence and potential for effects in estuaries. Pharmaceuticals enter wastewater streams following consumer use and enter the environment following wastewater treatment processing. Removal efficiency of pharmaceuticals by these facilities is highly variable due to their mainly being present in the dissolved phase. Cantwell and colleagues (2016a) reported elevated concentrations of dissolved and particulate pharmaceuticals along with partitioning coefficients for a number of highly prescribed drugs entering Narragansett Bay from riverine inputs. A recent yearlong study showed the spatial distribution of numerous classes of pharmaceuticals present throughout Narragansett Bay (Cantwell et al. 2017).

In this chapter, the Narragansett Bay Estuary Program reports on the available data concerning emerging contaminants in Narragansett Bay. While a number of CECs have been detected, the existing information is

not sufficient to develop specific metrics for ongoing indicator reporting. Rather, this chapter identifies the need to prioritize and continue measuring concentrations of emerging contaminants throughout Narragansett Bay. Efforts are presently under way to refine and validate indicators that will be effective in identifying wastewater-associated contaminants in urban estuaries such as Narragansett Bay (Cantwell et al. 2016b).

Status, Trends, and Discussion

The paucity of research to date on CECs in Narragansett Bay means that it is not yet possible to conduct an accurate and comprehensive examination of the magnitude and extent of contamination over time and space. In an attempt to address the lack of information, one approach is to use a suitable representative proxy or CEC itself as a marker to elucidate the behavior, fate, and transport of CECs. However, interpretation and transferability of this type of approach is limited to CECs that have the same sources and similar physico-chemical characteristics.

An example of this approach is demonstrated with the spatial and temporal trends of triclosan, for which there is the greatest amount of recent sediment data among CECs in Narragansett Bay. In Figure 2, a sediment core from the upper Providence River Estuary shows concentrations of triclosan from its inception point in 1963–when it was patented and first produced–to the surface of the core in 2007 (Cantwell et al. 2010). The influences of local production and use are clear, showing high levels well into the 1980s. Trends show the response to termination of local production in 1985, as well as the continued presence due to its widespread use in personal care products. Data from a Greenwich Bay core show lower levels, reflecting releases from the local wastewater treatment facility, which discharges approximately one million gallons per day of effluent (Katz et al. 2013). Finally, a core in the Taunton River is data limited but shows that discharges from wastewater treatment facilities in the Taunton River watershed are a continuous source for low levels of triclosan to this sub-embayment of Narragansett Bay (Cantwell, unpublished data).

Spatial distributions of triclosan from 2010 to 2013 illustrate the importance of point source discharges and transport processes throughout Narragansett Bay and their influence on contaminant concentrations baywide in contrast to levels observed in isolated embayments and the coastal salt ponds. Proximity and magnitude of wastewater treatment facility discharges are important factors regarding CEC distribution. Areas in the Upper Bay, such as the Providence River Estuary, where wastewater treatment facilities heavily impact the receiving waters, have relatively high levels of triclosan. Locations such as the Narrow River, with no wastewater treatment facilities, have very little measurable triclosan present in the surficial sediments. Residual levels observed are suspected to be from submarine groundwater inputs, emanating from on-site residential waste treatment systems. However, there are no local studies to confirm this.

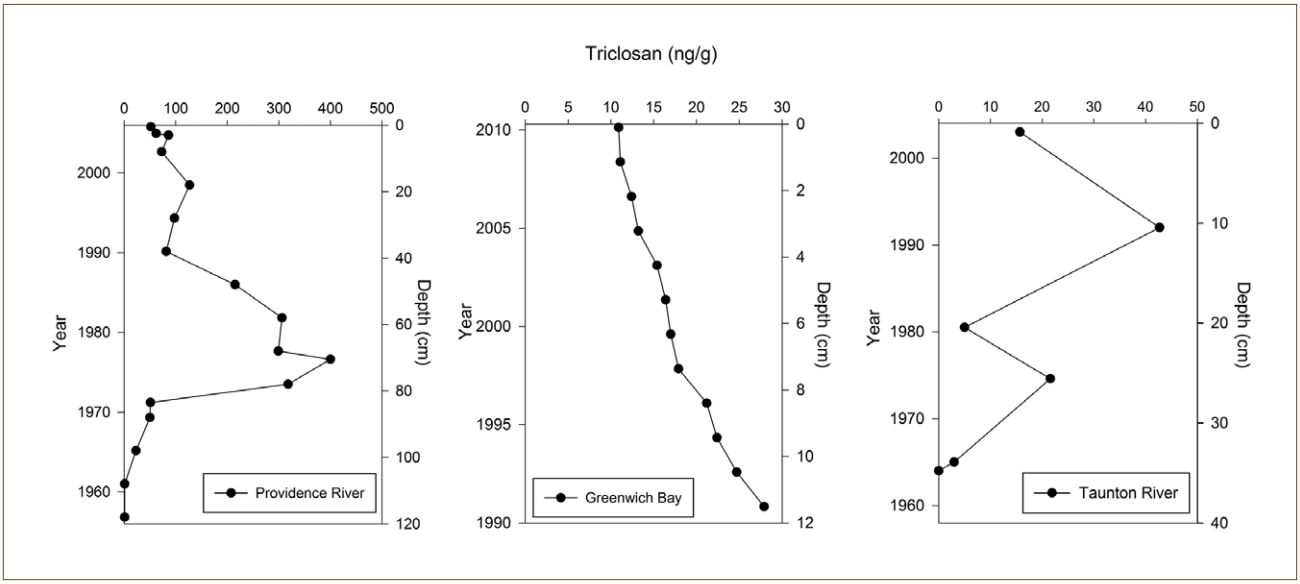


Figure 2. Triclosan concentrations in sediment cores from the Providence River and Greenwich Bay (Cantwell et al. 2010), and the Taunton River (Cantwell, unpublished data).

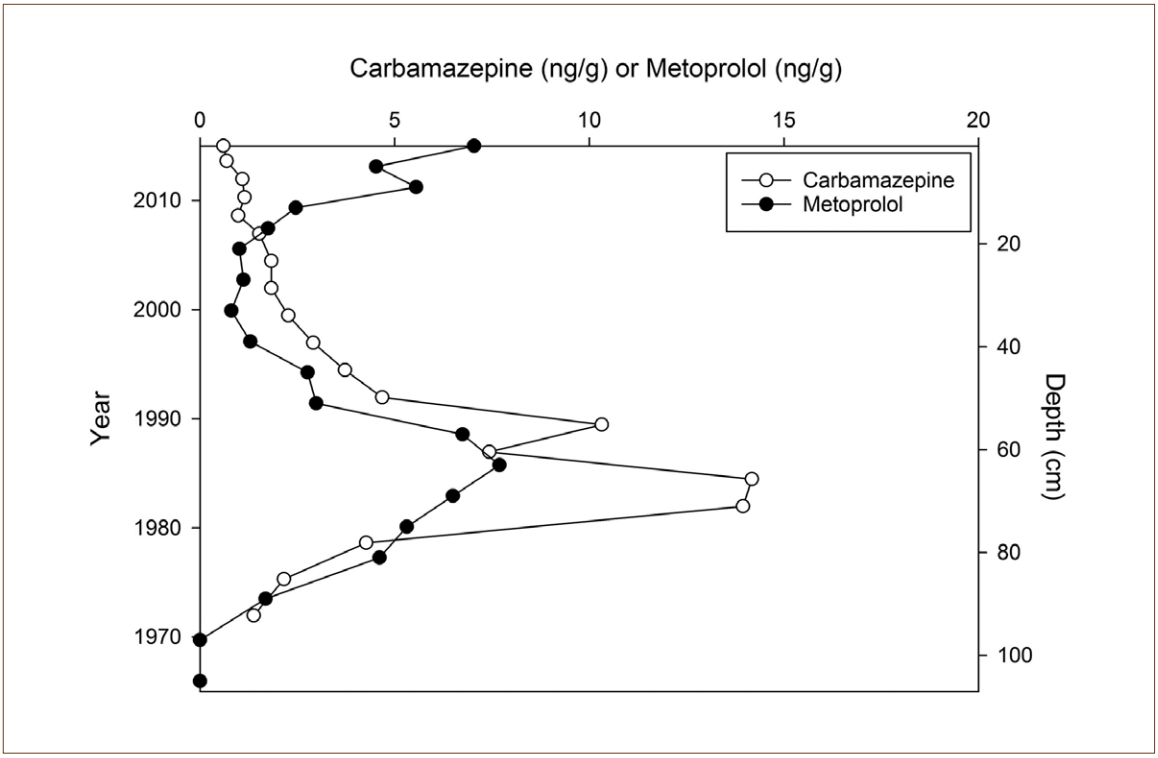


Figure 3. Concentrations of the pharmaceuticals carbamazepine and metoprolol in a sediment core from Field's Point, Narragansett Bay. Source: Cantwell (unpublished data)

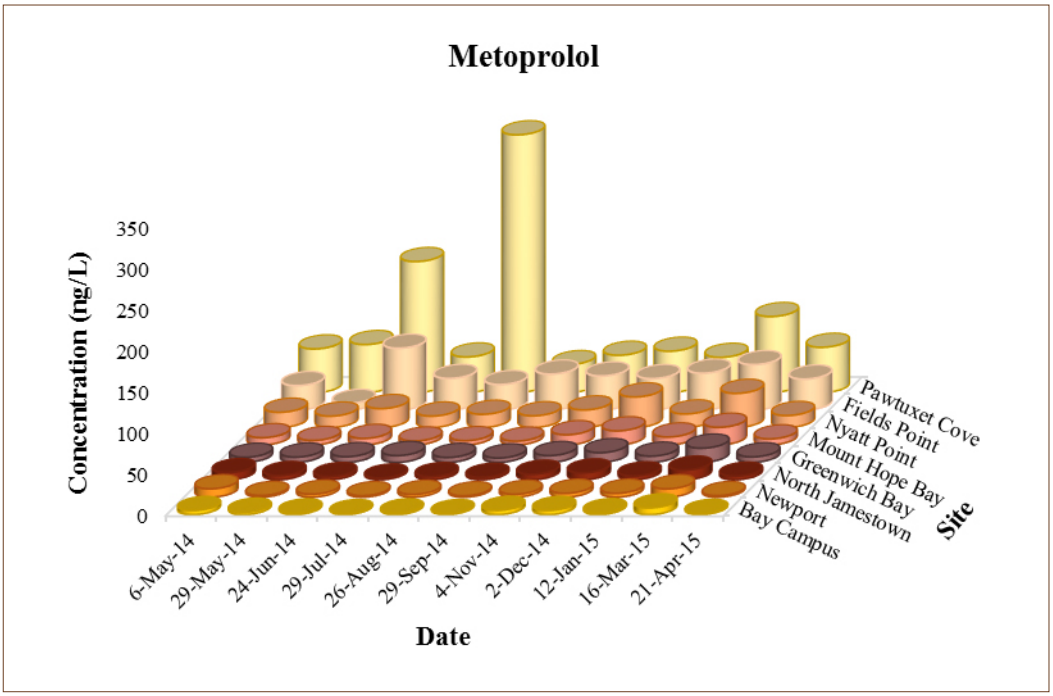


Figure 4. Spatial and temporal concentrations of metoprolol throughout Narragansett Bay. Source: Cantwell et al. (2017)

Pharmaceutical compounds in Narragansett Bay originate from the same point source inputs as many other CECs: wastewater treatment facilities. The pharmaceuticals present in aquatic systems are residues that enter sanitary wastewater treatment systems following therapeutic use or disposal via toilet flushing. They are assumed to remain bioactive and consequently have the potential to affect aquatic organisms. This may also be the case for metabolites of numerous pharmaceuticals. The physico-chemical characteristics of pharmaceuticals are quite different from most other CECs by design, in order to provide their intended therapeutic effects efficiently. Their sorption to particles is limited by characteristics such as high solubility, resulting in little of their total mass discharged being removed to sediments (Cantwell et al. 2016a). Measurements of several highly prescribed pharmaceuticals in a sediment core taken from the Providence River Estuary show that, overall, the concentrations in the sediments are very low (Figure 3). In the case of metoprolol, a beta-blocking antihypertensive drug, levels in the sediment remain below ten nanograms per gram. In contrast, dissolved water column concentrations on average are an order of magnitude greater, confirming their partitioning behavior in estuarine waters (Cantwell et al. 2016a). Another drug, carbamazepine, which is used to treat seizures and other disorders, exhibits similar behavior (Figure 3).

A consequence of pharmaceuticals with high aqueous solubility is that they remain largely in the dissolved phase and are still bioactive and likely bioavailable. If so, this exposure raises the potential concern that some organisms may be bioaccumulating some of these compounds. In locations such as the upper Providence River Estuary where large volumes of wastewater treatment effluents continuously enter Narragansett Bay waters, many of these pharmaceuticals may pose a risk due to the sustained, elevated levels present in the receiving waters.

In fact, a recent study (Cantwell et al. 2017) conducted over the course of a year showed elevated levels of numerous pharmaceuticals in the water column of Narragansett Bay. Various pharmaceuticals, such as metoprolol, were present at all sites and sampling periods, confirming their widespread spatial and temporal distribution (Figure 4). The study also showed many pharmaceuticals (e.g., sulfamethoxazole, an antibiotic) exhibiting a strong negative correlation with salinity, indicating that they are entering at the head of the Bay from freshwater sources such as wastewater treatment facilities (Figure 5).

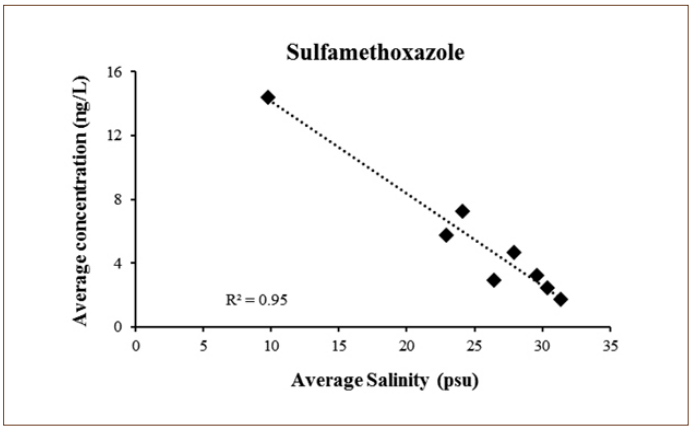


Figure 5. Correlation between sulfamethoxazole concentrations and salinity in Narragansett Bay. Source: Cantwell et al. (2017)

Data Gaps and Research Needs

- Continued research is needed to better understand the potential exposure and assess the likelihood of ecological and human health risks resulting from existing and newly identified contaminants of emerging concern (CECs). This includes research into the fate and transport of CECs in the environment.
- An assessment should be performed to identify key CECs prior to further investment in initiating a monitoring program. Any monitoring program will need to adapt to changes in the use of CECs. For example, as compounds are banned or phased out from use, compounds that may replace them should be considered for inclusion in monitoring.
- For CECs that are highly soluble and remain in the dissolved phase in the water column for extended periods of time, it would be beneficial to have an improved understanding of the hydrodynamic processes within Narragansett Bay. This information along with eco-toxicity and bioaccumulation data, the direct measurement of CECs, and the use of spatial models will help to identify potential locations of concern as well as ascertain the transport, behavior, and ultimately the fate of CECs within Narragansett Bay.

Acknowledgments

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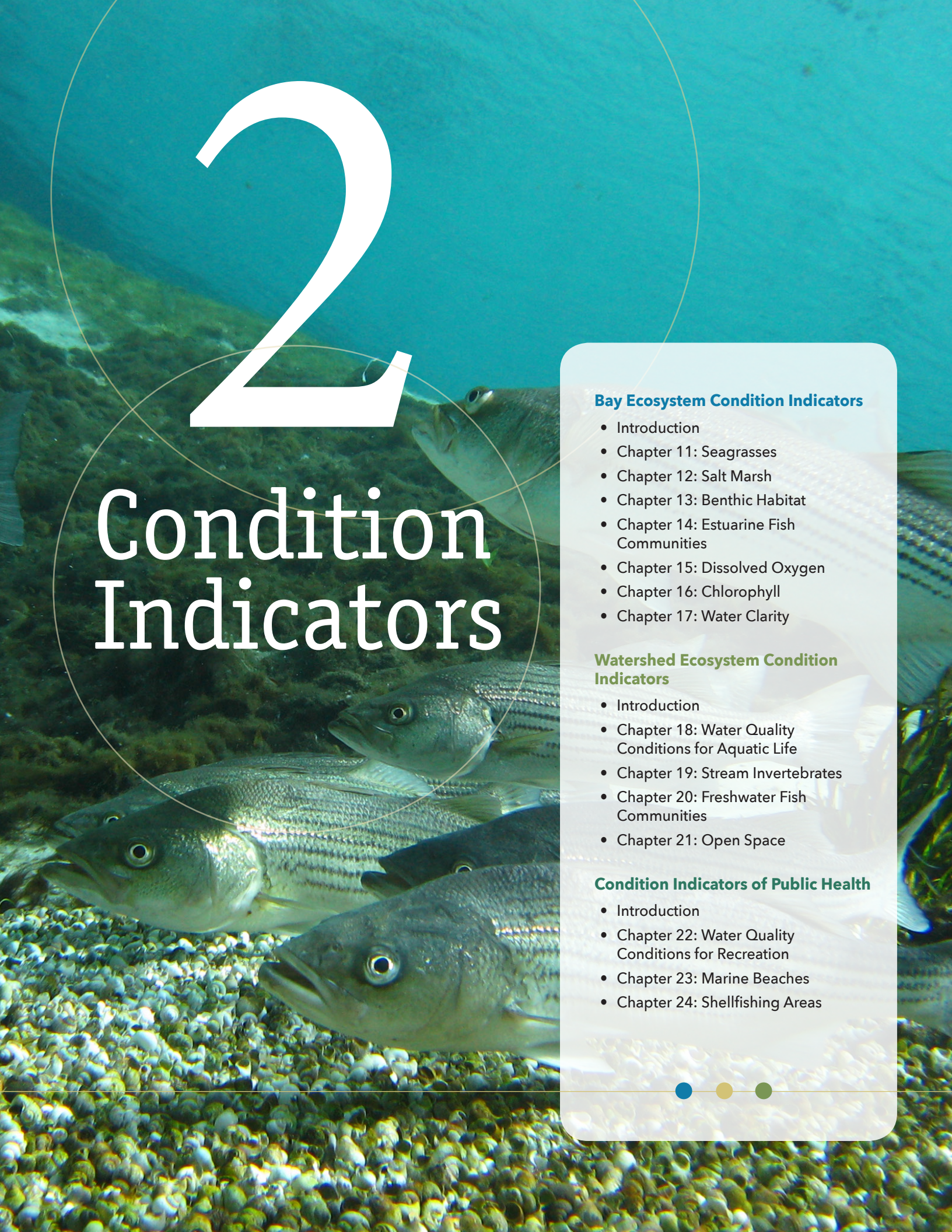
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2 Condition Indicators

Bay Ecosystem Condition Indicators

- Introduction
- Chapter 11: Seagrasses
- Chapter 12: Salt Marsh
- Chapter 13: Benthic Habitat
- Chapter 14: Estuarine Fish Communities
- Chapter 15: Dissolved Oxygen
- Chapter 16: Chlorophyll
- Chapter 17: Water Clarity

Watershed Ecosystem Condition Indicators

- Introduction
- Chapter 18: Water Quality Conditions for Aquatic Life
- Chapter 19: Stream Invertebrates
- Chapter 20: Freshwater Fish Communities
- Chapter 21: Open Space

Condition Indicators of Public Health

- Introduction
- Chapter 22: Water Quality Conditions for Recreation
- Chapter 23: Marine Beaches
- Chapter 24: Shellfishing Areas



State of Narragansett Bay and Its Watershed
2017 Technical Report

Bay Ecosystem Condition Indicators

INTRODUCTION

Please use the following citation: Narragansett Bay Estuary Program. 2017. State of Narragansett Bay and Its Watershed (Bay Ecosystem Introduction, pages 220-221). Technical Report. Providence, RI.

Photo: Salt Marsh in North Kingstown Town Beach, RI (Ayla Fox)

Located between Long Island Sound and Cape Cod, Narragansett Bay is a partially well-mixed estuary with a salinity range of 29 to 32 practical salinity units and a daily freshwater input of approximately 2,000 million gallons (Pilson 1985, Swanson and Spaulding 2008). This section focuses on seven important indicators of the Bay ecosystem's condition: seagrasses, salt marsh, benthic habitat, estuarine fish communities, dissolved oxygen, chlorophyll, and water clarity. These indicators represent the major habitats, culturally and economically important animal groups, and physical characteristics of the Bay.

The Bay ecosystem is affected by the climate change, landscape, and chemical stressors discussed in other sections of this report. Nutrient loading (a landscape stressor) and climate change have, arguably, the largest impacts on the Bay ecosystem. Excessive nutrient loading can drive eutrophication causing a suite of negative impacts such as hypoxia, destabilized salt marshes, decreased water clarity and seagrass extent, and deposits of excessive amounts of organic matter onto benthic habitats. Increased temperature, precipitation, and sea levels as a result of climate change can reduce diversity within the habitats, change species composition, change the amounts of primary production (thus affecting dissolved oxygen levels and water clarity), and alter habitats through flooding.

The chapters in this section present findings about past trends and recent status of the seven Bay ecosystem indicators. For some of these indicators, the data reveal clear trends, while for others the available data are limited and contain high amounts of variability that make it difficult to discern long-term trends. Moving forward, the indicators presented here can be used to assess how the Bay is responding to management actions and climate change.



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Photos: Beavertail State Park, Jamestown, RI (top), and Fish on Ice, Narragansett, RI (above). Photos by Ayla Fox.



State of Narragansett Bay and Its Watershed
2017 Technical Report

Bay Ecosystem Condition Indicators

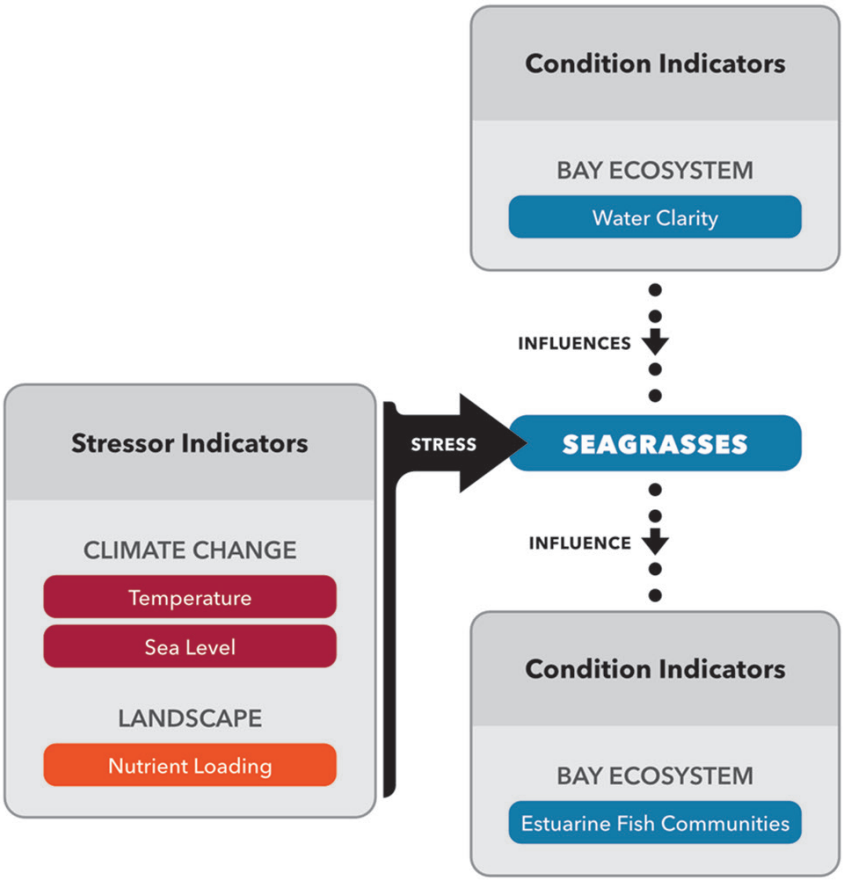
CHAPTER 11: SEAGRASSES

Please use the following citation: Narragansett Bay Estuary Program. 2017. State of Narragansett Bay and Its Watershed (Chapter 11, Seagrasses, pages 222-234). Technical Report. Providence, RI.

Photo: Ninigret Pond, Charlestown, RI (Ayla Fox)

NARRAGANSETT BAY ESTUARY PROGRAM

Overview



BACKGROUND

- Seagrasses are aquatic vascular flowering plants that stabilize sediments and provide vital habitat and nursery grounds for fish and shellfish. Because they require good water quality to thrive, seagrasses serve as good indicators of ecosystem health. Temperature, sea level, and nutrient loading are important stressors, and water clarity is an important influence on seagrass condition.

KEY FINDINGS

- ***Status:*** A survey in 2012 mapped 513 acres of seagrass in Narragansett Bay. Of that total acreage, 29 acres were found in Greenwich Bay, where seagrass had not been documented since the 1990s. A survey conducted in 2016 mapped 479 acres of seagrass in Narragansett Bay, according to initial analysis of the data.

- ***Trends:*** Prior to the 1930s, seagrasses were prevalent throughout Narragansett Bay, including the Providence River Estuary and Mount Hope Bay. A marked decline related to increased nutrient loading, disease, and physical removal or disturbance occurred, and now seagrasses are found predominantly in the Lower Bay. Between 2006 and 2012, seagrass increased by 37 percent (132 acres) in areas of Narragansett Bay that were mapped in both years. Seagrass acreage decreased by seven percent from 2012 to 2016, but the acreage in 2016 (479 acres) was still greater than the 2006 acreage (357 acres). Under climate change, warmer temperatures and sea level rise may become increasingly important stressors that impair seagrass growth and survival.

Introduction

Two types of seagrasses are found in Narragansett Bay: eelgrass (*Zostera marina*) and widgeon grass (*Ruppia maritima*). Eelgrass is a predominantly estuarine species, while widgeon grass thrives in lower salinity waters (Kantrud 1994). Eelgrass is taller than widgeon grass, and the two species have been observed intermixing within seagrass beds in Narragansett Bay (Peg Pelletier, USEPA-Atlantic Ecology Division, Narragansett, Rhode Island, personal communication). To date, most seagrass research in Narragansett Bay has focused on eelgrass, and additional research on widgeon grass would be useful for a more comprehensive understanding of seagrass dynamics.

Because seagrasses require abundant light, they are restricted to shallow areas with clear water. The slope of the substrate and the amount of light that can penetrate the water determine the greatest distance that seagrasses can grow from shore (Dennison and Alberte 1985, Mann 2000). Seagrasses in the temperate zones flower between 50 and 70°F (10 and 20°C). They live in areas with low nutrient input, as high nutrient levels tend to favor nuisance macroalgae, phytoplankton, and epiphytic growth that shade seagrasses and reduce their growth (Mann 2000). Seagrasses are perennial plants, but in the shallowest areas (less than approximately 3.3 feet [1 meter] depth), they may be considered functional annuals because the plants are often killed by ice scouring, freezing, and other seasonal stresses (Costa 1988).

Prior to the 1930s, the total extent of seagrass acreage in the estuarine waters of the Bay was vast—encompassing almost all sections of the Bay that were less than 10 to 12 feet deep, including the Providence River Estuary and Mount Hope Bay (Doherty 1995). Seagrasses then declined markedly, and now they are found only in the Lower Bay (Bradley et al. 2007, 2013). The decline was caused by stressors such as nutrient enrichment and physical disturbances (e.g., dredging, removal through boating or other activities, and storms), as well as by a seagrass disease outbreak in the 1930s that caused extensive losses along the Atlantic coast (Costa 1988, Short et al. 1993, Doherty 1995, Kopp et al. 1995).

Seagrass beds are highly productive and help to create complex habitats for a variety of other species that live in, on, or above the seabed, and they help to maintain the physical, chemical, and biological integrity of the ecosystem (e.g., Thayer et al. 1975, Collette and Klein-MacPhee 2002, Liu and Nepf 2016). In southern New England, seagrass beds provide nursery grounds, refuge, and feeding grounds for many commercially important and iconic organisms, such as bay scallops, flounder, striped bass, tautog, and seahorses (e.g., Heck et al. 1989). Additionally, seagrasses bind and stabilize sediment by slowing water currents and causing sediment to drop out of the water column (Liu and Nepf 2016). This provides food for animals that feed on the bottom and creates clearer water, increasing the amount of light reaching the seagrass blades (Orth 1977).

The productivity of seagrass beds makes them potentially valuable candidates for long-term carbon storage to mitigate the impacts of climate change (known as blue carbon). Seagrasses store or sequester carbon through both primary production and accumulation in the sediment (Lavery et al. 2013, Greiner et al. 2013). Data on organic carbon content of living seagrasses and sedimentary accumulation in seagrass meadows worldwide show a significant amount of storage capacity—roughly 4.6 to 9.3 billion tons (4.2 to 8.4 petagrams) of carbon (Fourqurean et al. 2012). The amount of organic carbon stored per unit area of seagrass is similar to that of forests

worldwide (Fourqurean et al. 2012). Protection and conservation of seagrass beds enhances global and regional resilience to climate change.

The ecological and societal value of seagrasses makes it critical to adequately monitor trends in the extent and condition of seagrass beds. Seagrasses are considered "coastal canaries" because the loss of seagrass often indicates ecosystem degradation and loss of ecosystem services, which can result in habitat regime shifts (Orth et al. 2006, Costello and Kenworthy 2011).

In this chapter, the Narragansett Bay Estuary Program reports on the extent (in acres) of seagrasses based on mapping data collected in 2006 and 2012. Mapping data were also collected in 2016, and initial results became available in late June 2017. The chapter discusses findings about seagrass acreage in the context of other recent and historical data, and explores how present and future stressors such as nutrient loading, warmer temperatures, and sea level rise may affect seagrasses.

Methods

In the past twenty years, several surveys have mapped seagrasses in Narragansett Bay and nearby waterbodies (Table 1). In 1996, the Narragansett Bay Estuary Program and partners commissioned aerial photography to map seagrass habitat (Huber 1999). The 1996 survey documented approximately 100 acres of seagrass in Narragansett Bay. Starting in 2006, the Rhode Island Eelgrass Task Force (Task Force) continued and refined these efforts by developing a set of mapping and monitoring protocols (Raposa and Bradley 2009). The Task Force is composed of researchers from the University of Rhode Island, state agencies, and non-profit organizations. Seagrass surveys using the Task Force protocols were conducted in 2006 and 2012. Mapping efforts were repeated in 2016.

Technological and methodological differences between the 1990s surveys and the 2000s surveys make statistical comparisons and analysis of change between them problematic (Bradley et al. 2007). For that reason, this report focuses on data from the 2006 and 2012 surveys (Bradley et al. 2007, 2013).

The Task Force developed a three-tiered system (Raposa and Bradley 2009) for monitoring and mapping seagrasses, based on the work of Neckles and colleagues (2012). In Tier 1, mapping is performed based on aerial photography with seagrass signatures digitized by a GIS technician; fieldwork is then conducted to augment and ground-truth the mapping data. The Task Force recommended conducting Tier 1 mapping and ground-truthing every three to five years. The 2006 and 2012 mapping efforts were Tier 1 mapping assessments conducted in Narragansett Bay with the methodology and results summarized by Bradley and colleagues (2007, 2013). While data from the other two tiers in the three-tiered system are not included in this chapter, the methods are important to summarize as background for the Data Gaps and Research Needs section. In Tier 2, a subset of seagrass beds is monitored annually for percent cover and other metrics of eelgrass condition. Currently, Tier 2 monitoring is conducted only at one seagrass bed in Narragansett Bay, at the southern end of Prudence Island. In Tier 3, biomass and other metrics are monitored repeatedly over multiple time scales within individual sites, following the SeagrassNet protocol (Short et al. 2002). Although Tier 3 monitoring occurred at two sites (Fort Getty in Jamestown and T-Wharf on Prudence Island) from 2005 to 2013, it is currently suspended due to lack of funding.

Table 1. Inventory of seagrass surveys using aerial photography in Narragansett Bay since 1996.

Year Photographs Taken	Orthophotograph Resolution	Citation
1996	1:12,000	Huber 1999
2006	1:5,000	Bradley et al. 2007
2012	1:5,000	Bradley et al. 2013
2016[1]	1:5,000	Bradley et al. 2017

[1] Detailed analysis of the 2016 data has not been completed as of this writing (June 2017)

The Estuary Program analyzed seagrass status in the 2006 and 2012 surveys, and assessed persistence of seagrass beds between the two surveys. Mapping was conducted at a 1:1500 scale and ground-truthing focused on new beds or areas of gain/loss (Bradley et al. 2013). Areas of seagrass present in both surveys were considered persistent, while other areas were classified as either gains or losses of seagrass acreage. No formal error analysis has been conducted but is planned for future survey datasets.

Because this report focuses on only two years of data, differences in seagrass coverage are discussed as changes, not as trends.

To examine historical changes that occurred prior to the 2006 survey, the Estuary Program conducted a presence-and-absence analysis of seagrass based on a comprehensive review of historical documents and oral history ranging from 1848 to 1994 (Doherty 1995, Kopp et al. 1995) and a comparison of those

Table 2. Changes in seagrass acreage in Narragansett Bay between 2006 and 2012. Acreage values were rounded to the nearest whole number. Data are reported for sections (bold) and segments (plain) of Narragansett Bay (see Introduction and Appendix for definitions of geographic areas). Persistence is the number of acres that were consistent between the two years of record. N/A means not applicable because 2006 acreage was zero or unknown. N/D means no data were collected. Mapping was conducted at a 1:1500 scale and ground-truthing focused on new beds or areas of gain/loss (Bradley et al. 2013).

Narragansett Bay Sections	Total Acreage		Persistence
	2006	2012	(acres)
Providence River Estuary	**0**	**0**	**N/A**
Warren, Palmer, and Barrington Rivers	**0**	**0**	**N/A**
Taunton River	**0**	**0**	**N/A**
Upper Narragansett Bay	**0**	**0**	**N/A**
Mount Hope Bay	**0**	**0**	**N/A**
Greenwich Bay[1]	**0**	**29**	**N/A**
Apponaug Cove	0	1	N/A
East Greenwich Bay	0	3	N/A
West Greenwich Bay	0	25	N/A
West Passage	**55**	**91**	**44**
Upper West Passage	3	2	1
Wickford Harbor	5	7	5
Middle West Passage	15	29	12
Lower West Passage	32	53	25
East Passage	**210**	**258**	**183**
Middle East Passage	89	112	77
Newport Harbor (Brenton Cove)	4	6	3
Lower East Passage	118	140	103
Sakonnet River	**31**	**52**	**26**
Lower Sakonnet River	31	52	26
Narrow River (Pettaquamscutt River)	**N/D**	**24**	**N/A**
Mouth of Narragansett Bay	**61**	**59**	**48**
Gooseberry (Island) Bay	35	21	28
Mouth of Narragansett Bay	26	38	20
TOTAL	**357**	**513**	**301**

[1] Widgeon grass was mapped in Greenwich Bay in 2012; no eelgrass or widgeon grass was documented in Greenwich Bay in the 2006 mapping survey.

Table 3: Comparison of historical presence of seagrass (1840 to 1994) to recent presence (2006 and 2012). Data are reported for sections (bold) and segments (plain) of Narragansett Bay. Green cells indicate presence of seagrass documented in Doherty (1995), Kopp et al. (1995), Bradley et al. (2007), or Bradley et al. (2013). Light blue cells preceding green cells indicate likely presence of seagrass based on Doherty (1995) and Kopp and colleagues (1995). "Unknown" indicates no evidence of seagrass presence; this does not imply absence, just no evidence of presence. For the 2006 and 2012 data, a blank cell indicates no seagrass found, and the numbers inside the green cells correspond to acreage in Table 2. Thick vertical line indicates the separation between historical data and the recent data. Information from Huber (1999) was not included because that report only calculated Bay-wide acreage, not acreage in specific areas of the Bay.

Narragansett Bay Sections	1840–1899	1900–1939	1940–1979	1980–1994	2006	2012
Providence River Estuary						
Upper Providence River			*Unknown*	*Unknown*		
Bullock Cove				*Unknown*		
Middle Providence River						
Lower Providence River						
Warren, Palmer, and Barrington Rivers						
Warren River			*Unknown*	*Unknown*		
Barrington River	*Unknown*			*Unknown*		
Palmer River						
Taunton River						
Taunton River (all segments)	*Unknown*	*Unknown*	*Unknown*	*Unknown*		
Upper Narragansett Bay						
Upper Bay						
Mount Hope Bay						
Lower Cole River			*Unknown*	*Unknown*		
Lee River			*Unknown*	*Unknown*		
Upper Cole River			*Unknown*	*Unknown*		
Mount Hope Bay	*Unknown*			*Unknown*		
Kickemuit River				*Unknown*		
Greenwich Bay						
Greenwich Cove			*Unknown*	*Unknown*		
East Greenwich Bay						3
Buttonwoods Cove			*Unknown*	*Unknown*		
Warwick Cove			*Unknown*	*Unknown*		
West Greenwich Bay						25
Apponaug Cove			*Unknown*	*Unknown*		1
West Passage						
Potowomut River						
Wickford Harbor					5	7
Middle West Passage					15	29
Bissel Cove				*Unknown*		
Dutch Harbor			*Unknown*	*Unknown*		
Quonset (Allen) Harbor						
Upper West Passage					3	2
Lower West Passage					32	53

Table 3 continued.

Narragansett Bay Sections	1840–1899	1900–1939	1940–1979	1980–1994	2006	2012
East Passage						
Upper East Passage						
Lower East Passage					118	140
Newport Harbor (Brenton Cove)					4	6
Middle East Passage					89	112
Potter Cove				*Unknown*		
Bristol Harbor						
Sakonnet River						
Nannaquaket Pond				*Unknown*		
The Cove						
Lower Sakonnet River	*Unknown*				31	52
Upper Sakonnet River	*Unknown*					
Narrow River (Pettaquamscutt River)						
Narrow River (all segments)	*Unknown*	*Unknown*	*Unknown*	*Unknown*	*Unknown*	24
Mouth of Narragansett Bay						
Gooseberry (Island) Bay	*Unknown*				35	21
Mouth of Narragansett Bay					26	38
Sections of Narragansett Bay with Known Presence of Seagrass	9	9	9	8	4	6

findings with more recent Task Force assessments (Bradley et al. 2007, 2013). The U.S. Coast and Geodetic Survey performed extensive surveys from 1832 to 1948 that noted seagrass locations. Other records relating to seagrass distributions were found in archives, herbariums, and reports. Oral interviews were also conducted to obtain information on past or present eelgrass locations. The Estuary Program performed a geospatial analysis of the historical data and developed a presence/absence analysis for sections of the Bay (Doherty 1995). The historical analysis did not attempt to quantify seagrass acreage, only presence or absence.

Status and Trends

In 2012, 513 acres of seagrass were mapped in Narragansett Bay, compared to 357 acres in 2006 (Table 2). In Greenwich Bay, where no seagrass (widgeon grass or eelgrass) had been mapped in 2006, 29 acres of widgeon grass were mapped in 2012. The Narrow River (Pettaquamscutt River), which was not covered in the 2006 survey, had 24 acres of seagrass when it was mapped in 2012. Excluding Greenwich Bay and the Narrow River (Pettaquamscutt River), the 2012 survey found that the other previously mapped areas of Narragansett Bay gained 103 acres of seagrass between 2006 and 2012 for an increase of 29 percent. This increase occurred primarily in the Sakonnet River (21-acre gain), the East Passage (48-acre gain), and the West Passage (36-acre gain) (Table 2). When the 29 acres of widgeon grass in Greenwich Bay are included, the areas of Narragansett Bay that were mapped in both 2006 and 2012 (i.e., not including the Narrow River [Pettaquamscutt River]) gained 132 acres of seagrass for an increase of 37 percent.

The seagrass beds in Narragansett Bay showed strong persistence between 2006 and 2012 (Table 2). Almost 85 percent of the seagrass beds mapped in 2006–301 of the total 357 acres–were also mapped in 2012, indicating that the center of the beds was seemingly stable in the six years between surveys.

The historical analysis of seagrass coverage in Narragansett Bay showed that seagrasses were widespread throughout the Bay until the middle of the twentieth century and then gradually disappeared from the Upper Bay (Table 3). In the twentieth century, seagrass was consistently documented as

present in eight or nine sections of the Bay. In the surveys conducted in 2006 and 2012, however, it was present in only four and six sections, respectively–all in the Lower Bay.

Discussion

Seagrass was more prevalent throughout Narragansett Bay before the 1940s, particularly in the northern sections of the Bay including Fox Point in the Upper Providence River, Mount Hope Bay, Hundred Acre Cove in the Barrington River, Potter Cove on Prudence Island, Greenwich Bay, and other locations in the northern reaches of the Bay (Doherty 1995, Kopp et al. 1995, Barrett et al. 2006, Nixon et al. 2008, Pesch et al. 2012, Chintala et al. 2015; Table 3). From 1840 to 1940, seagrass was noted in many sections of the Bay currently devoid of seagrass. Cicchetti (in prep.) concluded that almost 90 percent of seagrass acreage in the Providence River Estuary and Upper Narragansett Bay has been lost since the 1900s. The losses occurred in pulses associated with multiple factors, such as nutrient enrichment and physical removal during dredging and filling activities. From the 1930s through the 1960s, dramatic declines in seagrass acreage were reported (Doherty 1995, Kopp et al. 1995, Short et al. 1996). These declines were most likely due to increased nutrient input from a burgeoning population, punctuated by severe losses from an epidemic of seagrass wasting disease in the 1930s and two major hurricanes in 1938 and 1954 (Costa 1988, Short et al. 1993, Doherty 1995, Kopp et al. 1995; see "Wastewater Infrastructure" and "Nutrient Loading" chapters).

From 2006 to 2012, Narragansett Bay showed substantial gains in seagrass acreage, although the gains do not reflect a recovery to pre-1940s condition (Table 3). While a direct comparison with data from the 1996 survey (Huber 1999) is not possible because of methodological differences, there does seem to have been a substantial increase between 1996 and 2012. The 1996 survey found approximately 100 acres, compared to 513 acres in the 2012 survey. The difference is so great that it probably outweighs any methodological differences between the two datasets, leading researchers to believe that seagrass extent in Narragansett Bay did, in fact, increase over that time period, even if the magnitude of the increase is unclear (Mike Bradley, University of Rhode Island, personal communication). This view is supported by the observation that some seagrass study sites (e.g., Fort Getty and T-Wharf) did have increases in seagrass extent during the same time period (Bradley et al. 2007).

The sudden appearance of widgeon grass in Greenwich Bay in 2012 is noteworthy. Available information from 1996 and 2006 indicates that widgeon grass and eelgrass were not present in Greenwich Bay in those years, or else any seagrass was not visible in the aerial photographs and/or any seagrass beds were too small to be mapped. However, the historical analysis showed that seagrass had been present previously in East and West Greenwich Bay through 1994 (Table 3). Widgeon grass can tolerate fresher and warmer water than eelgrass (Kantrud 1994) and is prevalent in the Southwest Coastal Ponds and Briggs Marsh (located on the southeast side of the Sakonnet River, just outside the Narragansett Bay Watershed). It is unknown why seagrass apparently disappeared from Greenwich Bay for decades, although it is suspected that high nitrogen loading in Greenwich Bay from septic systems and a wastewater treatment facility–coupled with macroalgal and phytoplankton blooms (reducing water clarity), and poor circulation–may have contributed to the dramatic seagrass declines (Deacutis 2008).

While many factors affect seagrasses–such as disease, storms, ice scouring, and dredging–three key stressors are especially important with respect to present-day and future status and trends: nutrient loading, temperature, and sea level rise.

In the past, degradation of water quality appears to have been the main cause of seagrass loss (Costa 1988, Valiela et al. 1992, Hauxwell et al. 2003). Increased phytoplankton productivity, epiphyte growth, and turbidity (due to nutrient enrichment) are often invoked as the reasons for light limitation leading to seagrass decline (Kemp et al. 1983, Duarte 1995, Taylor et al. 1999, Pryor et al. 2007, Chintala et al. 2015). The recent gains in seagrass acreage in Narragansett Bay likely stemmed from improved water quality. A reduction in nutrient loading from local wastewater treatment facilities (see "Nutrient Loading" chapter) likely reduced epiphyte coverage on seagrass leaves, phytoplankton blooms, and macroalgae growth, improving water clarity (see "Water Clarity" chapter). Improved water clarity allows light to penetrate to greater depths, allowing seagrass beds to flourish and expand into deeper waters.

Second, warming waters can affect the spread of seagrass diseases, stress the plants, and influence how they reproduce. As waters warm, diseases such as wasting disease may spread more quickly. A combination of other climate impacts and anthropogenic factors can also exacerbate wasting disease outbreaks (Short et al. 1993, Doherty 1995). To date, wasting disease has not been observed since the

early 1930s, even though temperature is warming in Narragansett Bay (Fulweiler et al. 2015; see "Temperature" chapter).

In the temperate zone, seagrasses can reproduce in two ways: by extending new shoots and rhizomes, or through seed propagation. Warming waters may promote seed propagation instead of rhizome and shoot growth, particularly at high temperatures near or above 77 to 86°F (25 to 30°C) (Phillips et al. 1983, Short and Neckles 1999, Bintz et al. 2003). While seed germination can promote expansion of seagrass beds into new areas, if conditions are such that seed germination is restricted or a seed bank cannot be established (Harwell and Orth 2002), then seagrass may suffer and decline. Surface waters in the main channel of Narragansett Bay (Fox Island, West Passage) did not have sustained temperatures above 77 to 86°F (25 to 30°C) during the summer months (June, July, August, September), and seagrass acreage increased as temperatures increased (Figure 1). Many interacting factors will influence the future status of seagrass, with temperature an important factor. Seagrass may reach a tipping point and start to decline when sustained summer water temperatures are above 77 to 86°F (25 to 30°C). Estuarine water temperatures have risen approximately 3.6°F (2°C) over the last 50 years (see "Temperature" chapter), and if the trend continues water temperatures will reach that tipping point.

Finally, sea level rise is expected to change the tidal regime and water depth of Narragansett Bay, affecting the distribution of seagrasses (Short and Neckles 1999, Saunders et al. 2013, USEPA 2016). Increased tidal range would increase water depth, depending on local geomorphology. With increased water depth, light penetration may become limiting in places where seagrass currently grows, leading to decreases in seagrass productivity and changes in seagrass condition. Short and Neckles (1999) estimated that a 19.7-inch (50-centimeter) increase in water depth would reduce seagrass growth by 30 to 40 percent, and Saunders and colleagues (2013) predicted seagrass habitat in Moreton Bay, Australia, would decline 17 percent by 2100 if sea level rises

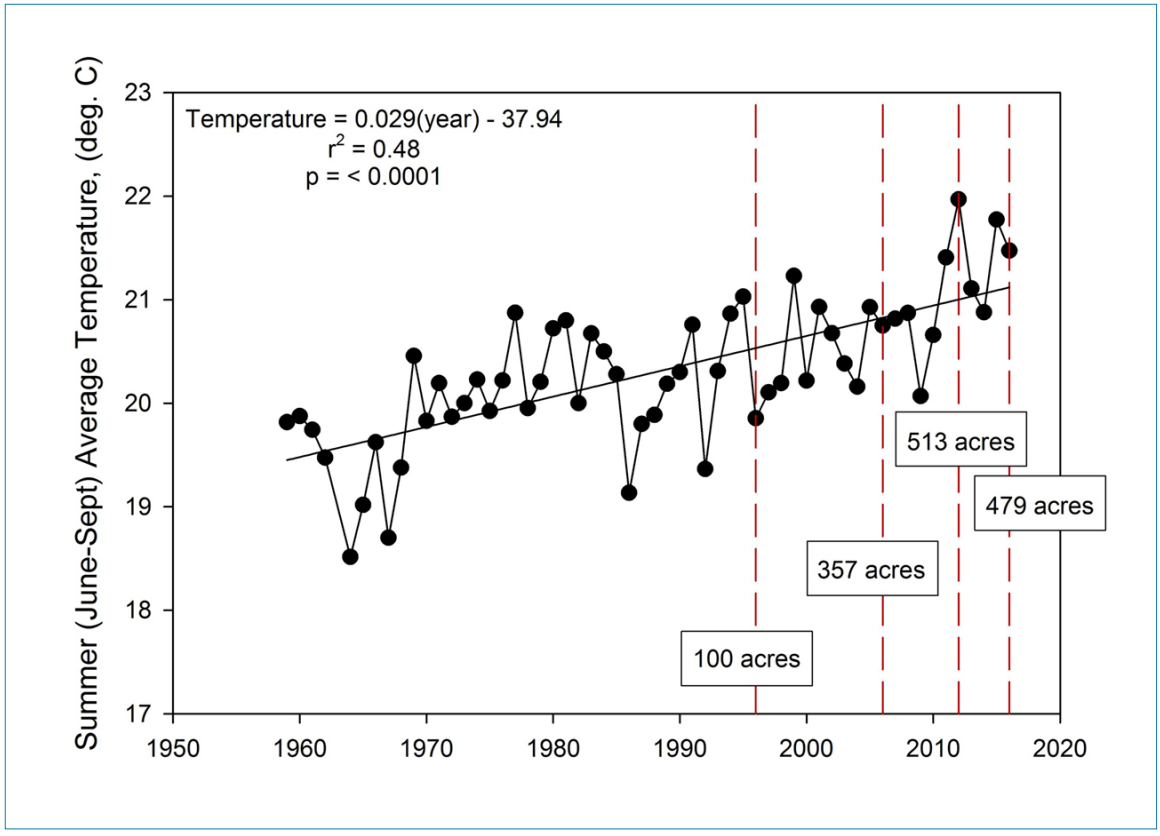


Figure 1. Summer (June, July, August, September) average water temperature from 1959 to 2016. Data are from the GSO Fish Trawl Survey at Fox Island, located in the West Passage of Narragansett Bay. Red vertical dotted lines show the years (1996, 2006, 2012, 2016) of the most recent seagrass mapping efforts with the total number of seagrass acres mapped.

3.6 feet (1.1 meters). In a process potentially offsetting some losses caused by deeper water, seagrass beds can expand landward with sea level rise, if they are not blocked by coastal development or hardening. Saunders and colleagues (2013) predicted that if impervious surfaces could be removed from newly inundated areas of land, loss of seagrass in Moreton Bay could be reduced to 5 percent. The predicted effect of sea level rise on seagrass in Narragansett Bay has not been modeled in a similar fashion. The outcome would depend on many factors, such as site-specific differences in the slope and sediment characteristics of the Bay's seabed, as well as the amount and locations of shoreline hardening around the Bay and the resulting scope for landward expansion of seagrass. Current sea level rise predictions for Narragansett Bay range up to a maximum of 11 feet (3.4 meters) by 2100 (see "Sea Level" chapter). That amount of sea level rise could be detrimental to seagrass beds in Narragansett Bay based on the findings from other parts of the world.

These stressors and others will affect not only the extent but also the condition of seagrass beds. If the extent shrinks or the condition deteriorates, seagrass habitats and the larger ecosystem they support will also deteriorate. The capacity for seagrass beds to store blue carbon would also decrease, and stored organic carbon could even be released as sediments destabilize. However, between 2006 and 2012 seagrass acreage increased by 37 percent in areas of Narragansett Bay that were mapped in both years, primarily in the Sakonnet River and in the East and West Passages (Table 2). In addition, in 2012 there was a newly mapped 29-acre bed of widgeon grass in Greenwich Bay, an area where seagrass had not been documented since the 1990s.

In June 2017, as this chapter was being finalized, Bradley and colleagues (2017) released a report on the 2016 seagrass mapping effort. Using the same methodology described above, they mapped 479 acres of seagrass in Narragansett Bay, with the majority located in the East Passage (Table 4). While that represents a slight decline (7 percent) since 2012, when 513 acres were mapped, the acreage in 2016 was still greater than in 1996 and 2006 (Figure 1), pointing to a potential recovery of seagrass habitat, happening at a time of major nutrient reductions (see "Nutrient Loading" chapter).

Data Gaps and Research Needs

- The Rhode Island Eelgrass Task Force's recommendations for a three-tiered approach to seagrass mapping and monitoring (Raposa and Bradley 2009) need to be implemented in order to conduct seagrass analysis more

Table 4. Seagrass acreage for sections of Narragansett Bay (see Introduction and Appendix for definitions of geographic areas). Acreage values were rounded to the nearest whole number. N/D means no data were collected.

Narragansett Bay Sections	Total Acreage		
	2006	2012	2016
Providence River Estuary	0	0	0
Warren, Palmer, and Barrington Rivers	0	0	0
Taunton River	0	0	0
Upper Narragansett Bay	0	0	0
Mount Hope Bay	0	0	0
Greenwich Bay[(1)]	0	29	25
West Passage	55	91	65
East Passage	210	258	237
Sakonnet River	31	52	44
Narrow River (Pettaquamscutt River)	N/D	24	44
Mouth of Narragansett Bay	61	59	64
TOTAL	357	513	479

[(1)] Widgeon grass was mapped in Greenwich Bay in 2012; no eelgrass or widgeon grass was documented in Greenwich Bay in the 2006 mapping survey.

systematically, including more refined methods to examine extent and condition.

- Warming temperatures, changes in precipitation patterns, and sea level rise can all affect how seagrass beds survive from year to year. Research is needed to fully understand how Narragansett Bay's seagrass beds will respond.
- A better understanding is needed of the life history traits of eelgrass and widgeon grass in Narragansett Bay. More knowledge of the life history traits will aid in conservation and restoration of seagrass beds to maintain or increase acreage or condition of the beds. Of particular interest is widgeon grass, as it is far less studied than eelgrass. Extensive mesocosm experiments on the response of eelgrass to nutrients, temperature, and other interactive factors have been conducted in Rhode Island (e.g., Bintz et al. 2003, Taylor et al. 1999). These types of studies should be pursued for widgeon grass, as well as for seagrass communities composed of both eelgrass and widgeon grass.

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State of Narragansett Bay and Its Watershed
2017 Technical Report

Bay Ecosystem Condition Indicators

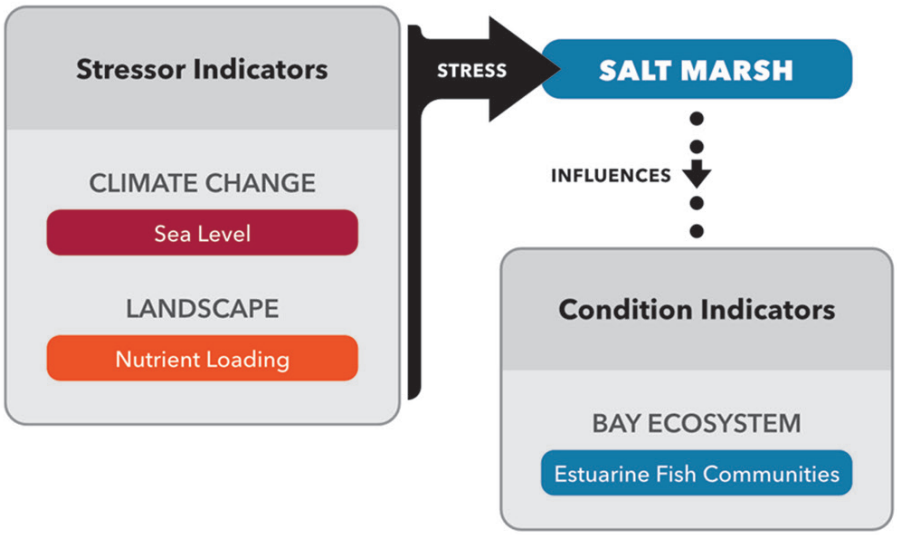
CHAPTER 12: SALT MARSH

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NARRAGANSETT BAY ESTUARY PROGRAM

Overview



BACKGROUND

- Salt marshes provide nursery grounds for fish and wildlife, protect inland areas from storms and flooding, and filter pollution from stormwater runoff. Sea level rise and excessive nutrient loading stress salt marshes, reducing their extent and condition, which then affects the estuarine fish community.

KEY FINDINGS

- ***Status:*** Narragansett Bay had a total of 3,321 acres of salt marsh, according to analysis of National Wetlands Inventory images taken in 1995, 2008, and 2010. More than one-third of the salt marsh acreage was in the Warren, Palmer, and Barrington Rivers.
- ***Trends:*** Between the 1800s and the 1970s, over 50 percent of Narragansett Bay's salt marshes were lost due to development and other stressors, such as excessive nutrient loading. It is estimated that an additional 13 to 87 percent of remaining salt marshes may be converted to open waters under one- and five-foot future projections of sea level rise, respectively.

Introduction

Salt marshes are intertidal ecosystems located between land and open salt water (salinity greater than 30 parts per thousand) or brackish water (salinity from 0.5 to 30 parts per thousand), occurring along protected shorelines and embayments. In this chapter, the term salt marsh is used as a general term encompassing both salt and brackish marshes. New England salt marshes are characterized by distinct vegetation zonation, related primarily to the tolerance of plants to the frequency and duration of tidal flooding (e.g., Niering and Warren 1980, Bertness 1991, Mann 2000). The low marsh is flooded by daily tides (i.e., regularly flooded) and is dominated by tall-form smooth cordgrass (*Spartina alterniflora*), which often lives in a narrow band along creek banks, ditches, and bayfront margins. The high marsh is flooded by high tides less frequently (i.e., irregularly flooded) and is generally dominated by a mosaic of vegetation, including salt hay (*Spartina patens*), black rush (*Juncus gerardii*), spike grass (*Distichlis spicata*), and short-form smooth cordgrass, among others. The high marsh may also contain sparsely vegetated shallow depressions, or pannes. Along the upland border of the salt marsh, where flooding occurs only during the highest of tides and storm surges, high-tide bush (*Iva frutescens*) and common reed (*Phragmites australis*) are often encountered. Marsh plants facilitate sediment accretion (where possible) by decreasing water movement. As they grow, die, and partly decay, they contribute organic matter to the marsh sediments, forming an organic soil known as peat, which helps stabilize the marsh substrate. In addition to marsh vegetation, other features such as tidal creeks, ditches, and pools define the marsh landscape and provide important habitat for fish species, most notably common mummichog (*Fundulus heteroclitus*), sticklebacks (*Gasterosteus aculeatus*, *Apeltes quadracus*), and silversides (*Menidia menidia*) (e.g., Raposa 2008).

Sea level rise affects salt marshes in several ways (Reed 1995). In southern New England, salt marshes form when smooth cordgrass colonizes intertidal flats, and then the aboveground vegetation facilitates the trapping of sediment, while roots and rhizomes grow, die, and decay below the marsh surface. These processes—sediment accumulation on the wetland surface and the build-up of peat (belowground live and decaying plant material and sediment)—contribute to the vertical growth of a marsh, ultimately resulting in conversion of low marsh areas to high marsh (those marsh areas of higher elevation). However, the increase in salt marsh elevation must keep pace with the rise in sea level, or the marshes will become submerged and too wet to support marsh plants. For three to four centuries (e.g., Redfield 1972, Orson et al. 1987), salt marshes in southern New England have been responding to sea level rise by accumulating peat and gradually increasing in elevation. However, recent evidence shows that some Narragansett Bay salt marshes are getting wetter, with high marsh vegetation being replaced by the more flood-tolerant smooth cordgrass or by open water (Donnelly and Bertness 2001, Raposa et al. 2017b). If no obstacles such as bulkheads or roadways exist, the high marsh migrates into upland areas as sea level rises. However, if obstacles exist, or the gradient is too steep, high marsh plants cannot migrate, and the marsh area becomes dominated by low marsh plants and/or open water. The marshes of Narragansett Bay may not be keeping pace with rates of sea level rise that appear to have accelerated in recent decades and are anticipated to increase even more dramatically as global temperatures continue to rise (see "Sea Level" chapter).

Researchers have attempted to establish salt marsh loss rates for Narragansett Bay. Bromberg and Bertness (2005) estimated that the state of Rhode Island lost 53 percent of its salt marsh area between 1832 and 1988, while the state of Massachusetts lost 41 percent of its salt marsh area between 1832 and 1999. Losses were most significant in the earlier parts of those time periods due to urbanization and the Industrial Revolution. Another wave of significant loss occurred during the late 1940s to 1950s with the suburbanization of Greenwich Bay, the Upper East and West Passages, and the Mount Hope Bay region.

After the 1950s, salt marsh loss slowed but did not come to a halt. From the 1950s to 1990s, roughly ten percent—306 acres—of the salt marsh area throughout Narragansett Bay was lost, mainly as a result of filling or in response to tidal restriction and ditching, which caused changes from herbaceous marsh vegetation to shrub or forested wetland (Table 1; Tiner et al. 2004). Using aerial photographs dating from 1972 to 2011, investigators studied 36 salt marshes throughout Narragansett Bay, Little Narragansett Bay, and the Southwest Coastal Ponds and reported a seventeen percent loss of vegetated salt marsh (Watson et al. 2017). The loss occurred through erosion of marsh shorelines, expansion of creeks, and conversion of vegetated marsh to ponds or unvegetated wet areas on the marsh surface. Similar findings have been reported for other areas of southern New England (Smith 2009). Although the data sources in Table 1 seem to show an increase in Narragansett Bay's salt marsh acreage from 1996 and 2010, that apparent increase may be attributable to better photointerpretation methods and an updated imagery database with finer resolution, not an actual increase in marsh

area (Tiner et al. 2014). Multiple drivers and stressors have been implicated in recent losses of vegetated marsh such as sea level rise, increasing temperature, nutrient enrichment, and crab herbivory on *Spartina* (Gedan et al. 2011a, Deegan et al. 2012, Bertness et al. 2014, Roman 2017).

Salt marshes have long been recognized for providing multiple benefits: shoreline stabilization, flood mitigation, filtration of nutrients and pollutants, carbon sequestration, nursery grounds for commercially and recreationally important species, and habitat for fish and wildlife (Roman et al. 2000, Valiela and Cole 2002, Gedan et al. 2011b, Mcleod et al. 2011). The first European settlers used salt marshes for pastures and hay production, and as marine access points (Gedan et al. 2009). With industrialization, salt marshes were filled for development, landfills, or disposal sites for dredge material; tidally restricted with dams or road crossings; polluted by runoff; and ditched for mosquito control (Gedan et al. 2009). These alterations to the salt marsh landscape led to their decline, resulting in the loss of approximately half of the salt marshes in Rhode Island and Massachusetts, and even larger losses in the upper Narragansett Bay region (Bromberg and Bertness 2005, Cicchetti in prep).

Recent estimates of salt marsh area throughout Narragansett Bay range from 2,660 to 4,000 acres, regardless of sampling year (Table 1). The variability is likely due to differences in the resolution of the aerial images, differences in salt marsh classification, and variability among the photointerpreters. Tiner and colleagues (2004) reduced some of the variability in their comparison of 1950s and 1990s aerial photographs by using the same classification and photointerpretation methods. Assessing and documenting temporal changes in the area of salt and brackish marshes should continue to be pursued as it is an important indicator of marsh status throughout the Bay.

In this chapter, the Narragansett Bay Estuary Program analyzes the total area of salt marsh in Narragansett Bay as well as the area of salt marsh within sections and segments of the Bay.

Methods

The Narragansett Bay Estuary Program compiled salt marsh data from the US Fish and Wildlife Service National Wetlands Inventory (NWI) dataset for Rhode Island and Massachusetts. This data source was selected because the data collection methods used by the US Fish and Wildlife Service were consistent between both states. The NWI used 2008 imagery for Massachusetts (except for the upper Taunton River imagery from 1995) and 2010 imagery for Rhode Island. The original NWI data were processed by the Estuary Program using ArcGIS and filtered for two

Table 1. Estimates of salt marsh acreage. Study areas included either all of Narragansett Bay (encompassing the areas in both Rhode Island and Massachusetts), only the Rhode Island areas of the Bay, or Rhode Island (including the Southwest Coastal Ponds and other areas outside the Estuary Program's study area).

Year	Study Area	Salt Marsh Area (acres)	Source
1951/2	Rhode Island areas of Bay only	2,966	Tiner et al. 2004
1974	Rhode Island areas of Bay only	3,668	Halvorson and Gardiner 1976
1988	Rhode Island areas of Bay only	4,000	Bromberg and Bertness 2005
1996	Rhode Island areas of Bay only	2,660	Tiner et al. 2004
2010	Rhode Island[1]	3,776	Tiner et al. 2014
2010	All of Narragansett Bay[2]	3,320	RI CRMC 2015
1995/2008/2010	All of Narragansett Bay	3,321	This Chapter

[1]The Estuary Program could not separate the Southwest Coastal Ponds or areas outside of the Estuary Program's study area from the Tiner and colleagues (2014) acreage estimate.
[2] For the RI CRMC (2015) acreage estimate listed here, the Estuary Program excluded the towns of Charlestown, New Shoreham, and Westerly, along with the parts of South Kingstown and Little Compton outside the Narragansett Bay Watershed.

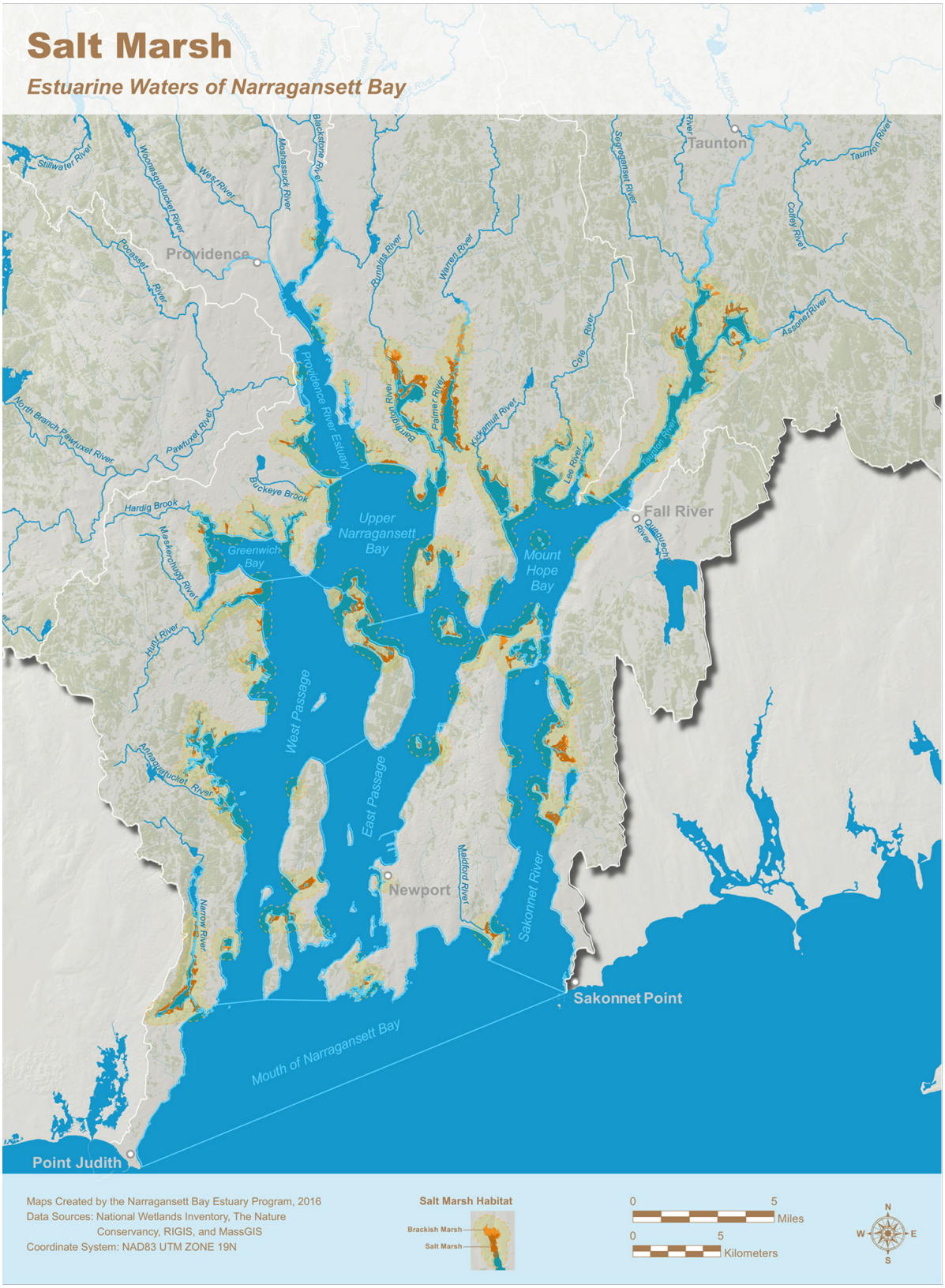


Figure 1. Marsh area in the estuarine waters of Narragansett Bay. Dark orange is salt marsh, while light orange is brackish marsh. Light blue text and lines indicate sections of Narragansett Bay used in the Estuary Program's analysis of salt marsh area (see Table 2).

specific categories: salt marsh and brackish marsh. The salt marsh and brackish marsh categories were defined by the land cover codes used in the Sea Level Affecting Marshes Model (SLAMM; RICRMC 2015), and wetland areas were analyzed at a scale of 1:24,000 or smaller. Salt marsh and brackish marsh areas were calculated for each section and segment of Narragansett Bay's estuarine waters (see Introduction and Appendix).

Three different research efforts have used the same NWI dataset to estimate salt marsh acreage in all of Rhode Island or in Narragansett Bay (including Rhode Island and Massachusetts): Tiner and colleagues (2014), RI CRMC (2015), and this chapter (Table 1). For the entire state of Rhode Island, Tiner and colleagues (2014) estimated a total of 3,776 acres of salt marsh. For Narragansett Bay, the RI CRMC (2015) estimated 3,320 acres. RI CRMC (2015) also produced an estimate that included the Southwest Coastal Ponds and areas outside of the Narragansett Bay Watershed for a total of 4,067 acres. The areas outside the Narragansett Bay Watershed were Charlestown, New Shoreham, Westerly, and a percentage of each South Kingstown and Little Compton (see Appendix). The RI CRMC (2015) acreage estimate for the Narragansett Bay Watershed is nearly the same as that reported by the Estuary Program in this chapter.

State-level datasets–including Rhode Island's Ecological Communities assessment (Enser et al. 2011) and the Massachusetts Department of Environmental Protection's wetlands dataset–were all more detailed than the NWI dataset and may have captured smaller, fringing marshes that the NWI wetlands inventory did not. However, those datasets are not comparable across state lines, so a cohesive watershed analysis was not possible. If both states adopted the same assessment method, then perhaps these more detailed data could be used for future salt marsh trend analyses instead of the NWI data.

Status

The National Wetlands Inventory data from 1995, 2008, and 2010 showed that Narragansett Bay had an estimated 3,321 acres of salt marsh (Figure 1; Table 2). Approximately one-third (27 percent) of salt marsh area was concentrated within the Warren, Palmer, and Barrington Rivers (902 acres). Other concentrations of marsh area occurred in the West Passage (551 acres), the Sakonnet River (441), and the Taunton River Estuary (369 acres).

Discussion

Salt marshes provide numerous ecosystem services, including critical habitat and nursery grounds, shore stabilization, and filtration of stormwater runoff, making them excellent indicators of ecosystem condition. Using aerial imagery from 1995, 2008, and 2010, the Estuary Program estimated that the Narragansett Bay Watershed had 3,321 acres of salt marsh. Since the 1800s, the Watershed has lost 50 to 60 percent of its marshes due mainly to development and other stressors (Bromberg and Bertness 2005). The most prevalent stressor today is climate change, particularly sea level rise, which threatens the extent and condition of the remaining salt marshes in the region (Watson et al. 2017).

Climate change is a major stressor on salt marsh form and function, and it may interact with other stressors commonly associated with anthropogenic impacts on salt marshes (Crain et al. 2008 and 2009, Gedan et al. 2011a and b, Oczkowski et al. 2016, USEPA 2016, Roman 2017, Watson et al. 2017). In the past, salt marshes in Narragansett Bay have accreted sediment and peat on pace with sea level rise, but that has changed in the last 30 years (Carey et al. 2017) as increases in marsh surface elevation are lagging behind relative sea level rise (Raposa et al. 2017a). As previously noted, marshes are getting wetter, as evidenced by conversion of high marsh vegetation to the more flood-tolerant salt marsh cordgrass and by the increase in open water on marsh platforms (Raposa et al. 2017b). Moreover, although the build-up of marsh peat is an important process contributing to the vertical growth of marshes, peat is decomposing at a greater rate than in the past, which is assumed to be related to increasing water temperature and precipitation (Charles and Dukes 2009, Carey et al. 2017).

In order to quantify how salt marsh extent will change with rising seas, both Rhode Island and Massachusetts have applied the Sea Level Affecting Marshes Model (SLAMM). Findings from this model have been published in a planning document for Rhode Island municipalities (RICRMC 2015) and are forthcoming for Massachusetts. Results from SLAMM suggest that Rhode Island could lose 13, 52, or 87 percent of its existing salt marshes under one-, three-, and five-foot sea level rise scenarios, respectively. SLAMM results also indicate locations where marshes may, hypothetically, migrate landward in response to sea level rise. However, given that rates of sea level rise are projected to accelerate, perhaps quite dramatically, it is likely that the process of marsh landward migration could be impeded by rapid sea level rise,

Table 2. Marsh area (acres) in Narragansett Bay. Data are reported for sections (bold) and segments (plain) of Narragansett Bay (see Introduction and Appendix for definitions of sections and segments). Marsh categories were defined by the land cover codes used in the Sea Level Affecting Marshes Model (SLAMM; RICRMC 2015). The Estuary Program's analysis included both salt marsh and brackish marsh, and in this chapter the results and discussion of marsh acreage refer to numbers in the "total" column. Wetland areas were analyzed at a scale of 1:24,000, or smaller if detected; therefore, some smaller fringing marshes may not be included in the estimate of marsh area.

Narragansett Bay Sections	Marsh Area (acres)		
	Salt Marsh	Brackish Marsh	Total
Providence River Estuary	**72**	**16**	**88**
Upper Providence River	1	0	1
Watchemoket Cove	3	2	5
Bullock Cove	3	2	5
Seekonk River	4	0	4
Middle Providence River	11	1	12
Lower Providence River	50	11	61
Warren, Palmer, & Barrington Rivers	**705**	**197**	**902**
Warren River	32	15	47
Barrington River	300	77	377
Palmer River	373	105	478
Taunton River Estuary	**325**	**44**	**369**
Assonet River	7	0	7
Upper Taunton River	31	39	70
Lower Taunton River	52	5	57
Broad Cove	83	0	83
Assonet River & Bay	152	0	152
Upper Narragansett Bay	**173**	**23**	**196**
Buckeye Brook	5	20	25
Upper Bay	168	3	171
Mount Hope Bay	**225**	**50**	**275**
Lower Cole River	16	0	16
Lee River	28	7	35
Upper Cole River	44	4	48
Mount Hope Bay	53	39	92
Kickemuit River	84	0	84
Greenwich Bay	**82**	**11**	**93**
Greenwich Cove	0	1	1
East Greenwich Bay	2	0	2
Buttonwoods Cove	15	2	17
Warwick Cove	21	1	22
West Greenwich Bay	21	5	26
Apponaug Cove	23	1	24
Greenwich Cove	0	1	1

similar to the marsh platform being unable to keep pace with sea level and becoming submerged. In addition, development such as seawalls, bulkheads, and roads in urban and suburban areas that are common in the Narragansett Bay Watershed, along with relatively steep upland slopes in some areas, will impede landward marsh migration. Field research and modeling are needed to better understand the landward marsh migration process under regimes of accelerated sea level rise, while also pursuing efforts to preserve upland areas that could serve as suitable migration corridors, assuming that salt marshes will maintain some capacity to move landward. The SLAMM model is a useful tool for identifying these corridors where upland slopes are slight and free of obstructions.

In addition to sea level rise, nutrient loading and herbivory have an influence on salt marsh elevation. With nutrient enrichment, marsh peat can decompose at a high rate and eventually collapse, no longer serving the important role of maintaining marsh elevation (Watson et al. 2014). Further, under high nutrient regimes, marsh plants tend to allocate more biomass to the aboveground portion of the plant than belowground, resulting in decreased peat formation (e.g., Wigand et al. 2014). There is also evidence that the purple marsh crab (*Sesarma reticulatum*) population is increasing in Rhode Island's salt marshes, and their grazing on cordgrass along creek banks is an important factor contributing to vegetation dieback and marsh loss (Bertness et al. 2014).

Table 2 continued

Narragansett Bay Sections	Marsh Area (acres)		
	Salt Marsh	Brackish Marsh	Total
West Passage	**505**	**46**	**551**
Wesquage Pond	4	21	25
Bissel Cove	11	0	11
Quonset (Allen) Harbor	14	6	20
Potowomut River	58	1	59
Wickford Harbor	66	7	73
Dutch Harbor	115	4	119
Upper West Passage	117	4	121
Middle West Passage	120	3	123
East Passage	**144**	**11**	**155**
Lower East Passage	1	0	1
Newport Harbor (Brenton Cove)	2	0	2
Middle East Passage	13	4	17
Potter Cove	19	0	19
Bristol Harbor	21	7	28
Nag Pond	38	0	38
Upper East Passage	50	0	50
Sakonnet River	**406**	**35**	**441**
Nannaquaket Pond	6	0	6
The Cove	45	4	49
Lower Sakonnet River	52	0	52
Nonquit Pond	75	5	80
Upper Sakonnet River	228	26	254
Narrow River (Pettaquamscutt River)	**232**	**1**	**233**
Mouth of Narragansett Bay	**15**	**3**	**18**
Gooseberry (Island) Bay	15	3	18
Total Narragansett Bay Watershed	**2,884**	**437**	**3,321**

In response to sea level rise, increasing temperatures leading to peat decomposition and collapse, high nutrient loads, low sediment supply (a natural characteristic of New England watersheds; Roman et al. 2000), increasing herbivory on marsh plants, and other factors, salt marshes are drowning, and the outlook for long-term sustainability is grim given projections of increased sea level rise and global temperatures. A positive factor to consider is that nutrient loading to Narragansett Bay from wastewater treatment facilities is declining (RIDEM 2005; see "Nutrient Loading" chapter), thus reducing the impact of this stressor on marshes. In addition, at a few marsh sites in Rhode Island, efforts are under way to add sediment to the surface of marshes (known as thin-layer deposition) to increase their elevation relative to sea level. The combination of reduced nutrient enrichment and restoration efforts may serve to improve the resiliency of salt marshes in Narragansett Bay.

Data Gaps and Research Needs

- The multi-parameter Rhode Island Salt Marsh Monitoring Strategy (Raposa et al. 2015) needs to be fully implemented, including refining methods, in order to document status and trends in salt marsh extent in Narragansett Bay, and changes in marsh cover types (after Watson et al. 2017). This information is needed to assess the effects of sea level rise and other stressors on the long-term sustainability of marshes.
- Research and monitoring is needed to evaluate methods that will facilitate salt marsh resilience to sea level rise (e.g., thin layer deposition; preservation of upland to allow for migration). A cost-benefit analysis coupled with multi-year monitoring could be used to help determine the best methods to improve long-term sustainability.
- The existing SLAMM maps (RICRMC 2015) identify areas where marshes could migrate landward. Field research and modeling are needed to better understand the process of landward marsh migration under regimes of accelerated rates of sea level rise.
- Sea level rise is a major factor contributing to the recent trend of Narragansett Bay's marshes tending toward submergence, but there are also many other factors interacting with sea level rise (e.g., nutrients, grazing, sediment supply, increasing temperature, increasing carbon dioxide). Additional empirical research and modeling are required to understand the complexity of these interactions so that effective adaptation strategies can be implemented.

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State of Narragansett Bay and Its Watershed
2017 Technical Report

Bay Ecosystem Condition Indicators

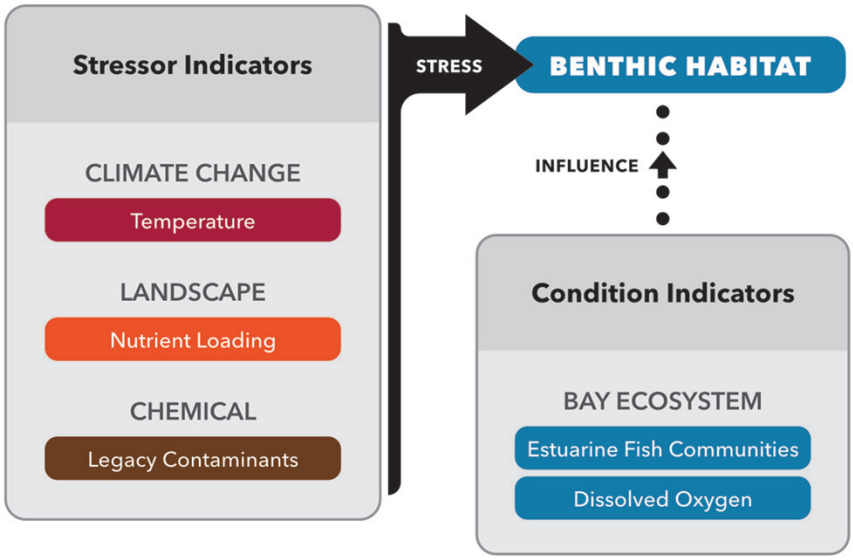
CHAPTER 13: BENTHIC HABITAT

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Photo: Ninigret Pond, Charlestown, RI (Ayla Fox)

NARRAGANSETT BAY ESTUARY PROGRAM

Overview



BACKGROUND

- The seafloor is a mosaic of various sediment types and other characteristics. These characteristics create habitats that support different groups of shellfish and fish, and a wide range of other species. The biota living in benthic habitats cycle nutrients, regulate water quality, and add to the overall productivity of the ecosystem.
- Warming water temperatures, nutrient loading, legacy contaminants in the sediments, and low levels of dissolved oxygen can affect the abundance and types of species living in benthic habitats. In turn, those species serve as an important food source for estuarine fish and other organisms.

KEY FINDINGS*

- ***Status:*** As of 2008, beds of small, tube-building crustaceans called *Ampelisca* were the dominant benthic habitat in the upper and middle sections of Narragansett Bay. *Ampelisca* amphipod species are the hallmark of a recovering benthic ecosystem but not a fully recovered benthic ecosystem. *Ampelisca* indicate an intermediate step on the path to lower organic loadings and improved water quality.
- ***Trends:*** Benthic habitat quality improved between 1988 and 2008, as indicated by an increase in pollution-sensitive species such as *Ampelisca*. Considering the large reductions in nutrient inputs to the Bay since 2008, future assessments of benthic habitats may find more oxygenated sediments and a different suite of benthic species.

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Introduction

Benthic habitats, or biotopes, incorporate descriptors of the abiotic environment (e.g., sediment type) and the associated assemblage of species living there (Connor et al. 2004, Davies et al. 2004, Costello 2009). Benthic organisms, such as burrowing worms and small crustaceans, live within the mud or sand, or are attached to the bottom, and many do not move large distances in their adult phase. This makes them especially good indicators of water and sediment quality.

In particular, monitoring benthic habitats is a useful way to identify excessive nutrient inputs from wastewater treatment discharge, improperly functioning septic systems, runoff of fertilizers, and other sources (see "Nutrient Loading" chapter). The seafloor receives a near-constant supply of organic matter that rains down from the water column or is produced from other bottom-dwelling organisms. This organic matter is usually derived from sources such as phytoplankton, zooplankton, macroalgae, and fish and shellfish excrement, together called organic loading. Excessive nutrients entering the water as pollution from human sources can amplify the amount of organic matter reaching the bottom. Nutrient pollution is problematic because high levels of organic matter can lead to low levels of dissolved oxygen, when decomposition occurs (see "Dissolved Oxygen" chapter). Pollution-tolerant species such as small polychaetes favor areas with high organic loading, while pollution-sensitive species such as deep-burrowing worms and shrimp tend to thrive in areas with lower organic loading. As a result, the types of benthic species found in an area indicate whether the area has good or poor water quality. In this chapter, the presence of small, pollution-sensitive crustaceans called *Ampelisca* is used as an important indicator of improving water quality.

Benthic habitats are effective integrators, reflecting cumulative stressors such as eutrophication and hypoxia (Pearson and Rosenberg 1978, Valente et al. 1992, Germano et al. 2011). The structure of surface sediment and the composition, or successional stage, of benthic communities are linked to the degree of organic loading (Rosenberg 2001) and readily indicate recent (weeks to months) and longer-term water quality changes in Narragansett Bay (Cicchetti et al. 2006, Shumchenia and King 2010a). Coastal marine habitats comprise mosaics at the landscape scale (Bostrom et al. 2011) and are ecologically meaningful units for conservation and management purposes (Salomidi et al. 2012). Habitat mosaics are interrelated and functionally connected such that a change to one habitat may affect others, as well as the entire ecosystem (Bostrom et al. 2011). The composition of a habitat mosaic and how it changes over time may indicate degradation or recovery of an ecosystem (Dunning et al. 1992, Wiens et al. 1993, Pittman et al. 2007). Thus, historical and recent data that describe habitat mosaics can be used to assess the effects of human alterations and multiple stressors on Narragansett Bay (e.g., Cicchetti and Greening 2011).

Abiotic components of benthic habitats (e.g., surficial geology, sediment organic content, contaminants) in Narragansett Bay have been described Bay-wide by a few different efforts since the 1950s. McMaster (1960) measured sediment grain size throughout the Bay in a systematic survey and produced the first comprehensive surficial sediment maps of the Bay. (McMaster [1984] also mapped subsurface geology of the Bay in subsequent work.) In the 2000s, investigators measured sediment grain size, organic content, and the concentration of metals throughout the Bay, with the highest density of sampling in the Upper Bay (Murray et al. 2007). Also in the 2000s, two projects partnered to map the Bay bottom using new technologies in addition to traditional grab samples and sediment cores. Acoustic bathymetry and backscatter information were collected, as well as sediment profile imagery, along with the traditional bottom samples. The BayMap project concentrated on habitats deeper than fifteen feet, while the MapCoast project focused on waters shallower than fifteen feet (King et al. 2007). The data were used to map the geomorphic features of the Bay bottom (Figure 1a) and the texture of the surface sediment (Figure 1b). The maps provided the context for understanding how benthic communities and other estuarine fauna (e.g., eelgrass, fish, and shellfish) are related to benthic habitats.

Benthic habitats in Narragansett Bay were first qualitatively described in the context of understanding commercially important species such as shellfish and demersal fish. The first quantitative study of benthic habitats was not conducted until the 1950s. Phelps (1958) found that sediment grain size and percent sediment organic matter were the most important factors influencing benthic species distribution in the Bay. Since the 1950s, there have been numerous directed studies of benthic habitats (i.e., to determine the effects of human actions such as dredging, wastewater discharges, and other development), but no consistent Bay-wide monitoring effort (see Frithsen 1990 for a detailed accounting of Narragansett Bay benthic studies). Analyses of benthic data between the 1950s and around 1980 show a

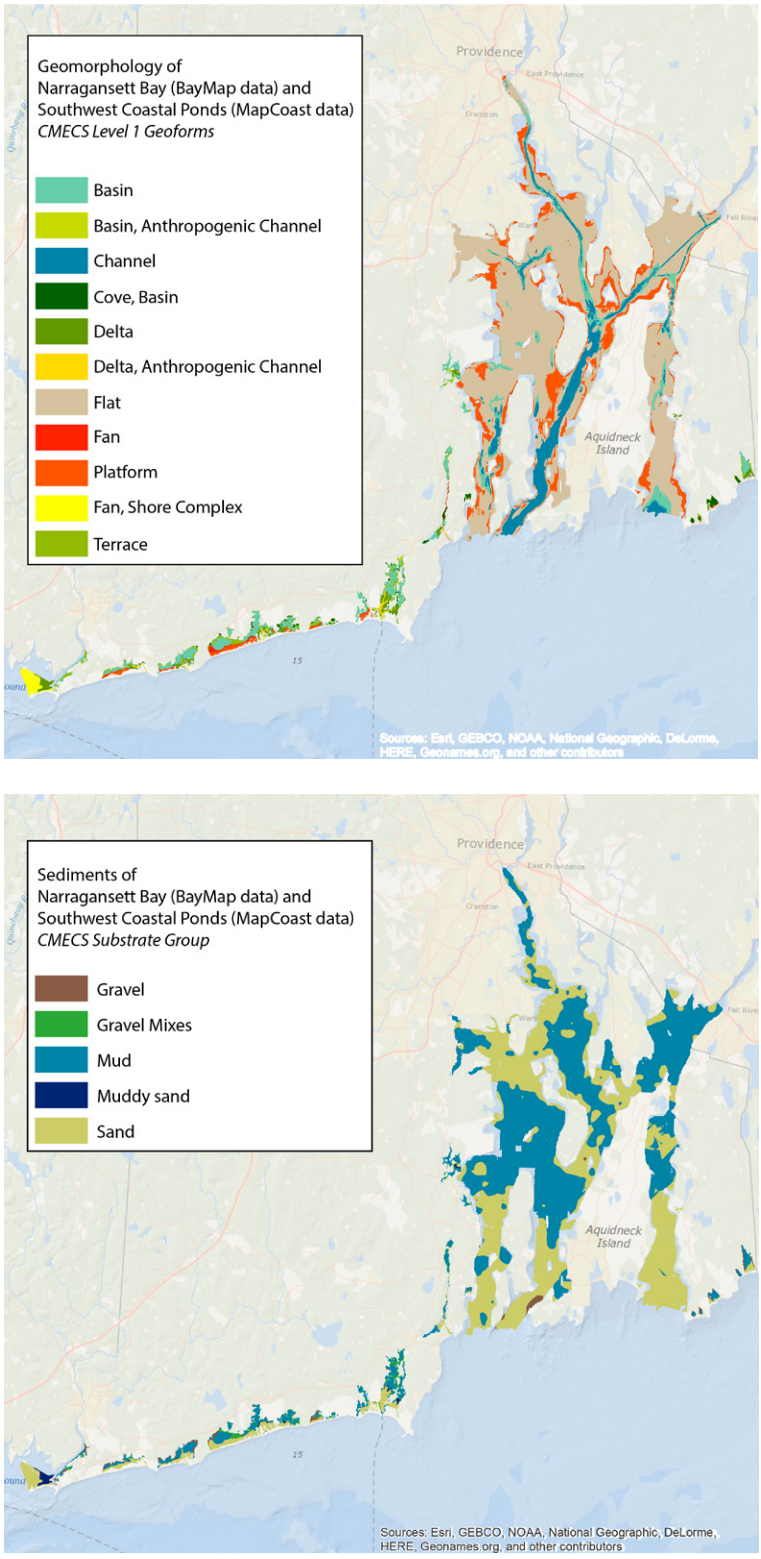


Figure 1. Maps of abiotic (nonliving) benthic habitats in Narragansett Bay and Southwest Coastal Ponds. Top: Geomorphology of the Bay bottom, as derived from bathymetry surveys and supporting sediment type. Bottom: Sediment type, derived from benthic samples and grain size analyses.

gradient of increasing benthic species abundance and diversity from the Providence River Estuary to Conimicut Point, likely due to the decreasing gradient of organic loadings, sewage effluent, incidences of hypoxia and anoxia, and phytoplankton blooms (Frithsen 1990). South of Conimicut Point, there was little spatial variation in benthic species abundance and diversity within this same time period; this pattern was attributed to lower exposure to stressors (Frithsen 1990). More recent efforts to monitor benthic habitats are summarized in Table 1. Each survey has spatial and/or temporal limitations, and only a few can provide data than can support a consistent trend assessment over more than ten years.

The Marine Ecosystems Research Laboratory at the University of Rhode Island (Table 1) described a recent spatial trend in benthic habitat quality that corresponded to the decreasing gradient in nutrients and contaminants from the Upper to the Lower Bay (Calabretta and Oviatt 2008). The United States Environmental Protection Agency (EPA) produced a report that described trends in the benthic community in the Upper Bay (i.e., Providence River, Greenwich Bay, Mount Hope Bay) from the 1950s to 2015 based on a variety of studies, including those listed in Table 1, except for the benthic video sled (Chintala et al. 2015). The EPA trends report found that overall benthic diversity had declined since the 1950s and that there had been a shift from pollution-sensitive to pollution-tolerant species at many of the locations sampled (Chintala et al. 2015). The report also linked those patterns with stressors including nutrients, legacy contaminants, and changing phytoplankton communities (Chintala et al. 2015). Pelletier et al. (2017) discussed a subset of the findings from the EPA trends report for data collected in Greenwich Bay in 2004. These authors found that Greenwich Bay was primarily impacted by eutrophication-related stressors, that the sediments were enriched with organic carbon, and that the resulting benthic community had a low number of species. The National Coastal Condition Assessment 2010 did not report estuary-specific trends, but the national benthic index showed a statistically significant increase of 67 percent of the total coastal area rated "good" in 2010 from 50 percent rated "good" in 2005 and 2006 (US EPA 2015). The benthic video sled deployed by the Narragansett Bay Commission is unique when compared to the other surveys in Table 1, as it primarily sampled epifauna (i.e., organisms living on the sediment surface). The benthic video survey has been conducted only in portions of the Providence River Estuary, but it has been attempted at roughly monthly intervals–a high sampling frequency.

Sediment profile imaging (Table 1) is a rapid reconnaissance technique that delivers clear images of benthic habitats regardless of water column turbidity (Germano et al. 2011). In 1988, sediment profile imaging was used in the first comprehensive survey of benthic habitat quality in Narragansett Bay

Table 1. Recent efforts to monitor benthic habitat and/or community composition in Narragansett Bay.

Benthic monitoring strategy	Organizing group	Spatial extent	Temporal extent	Future sampling
Benthic macrofauna community composition and abundance	Marine Ecosystems Research Laboratory, URI-GSO	4 stations in Upper Bay	4 stations 2000, 2001, 2002, 2004; 1 station 2005–2010	None planned
Trends report	EPA	Upper Bay	Trends analysis of data 1950s–2015	None planned
National Coastal Condition Report	EPA	Bay-wide (some stations permanent, others changing)	Summer 2005/6, 2010, 2015	2020
Benthic video sled	Narragansett Bay Commission	3 transects in Providence River Estuary	2–6 times per year, 2014–present	Attempted monthly
Sediment profile imagery	URI	Bay-wide	1988, 2008	None planned

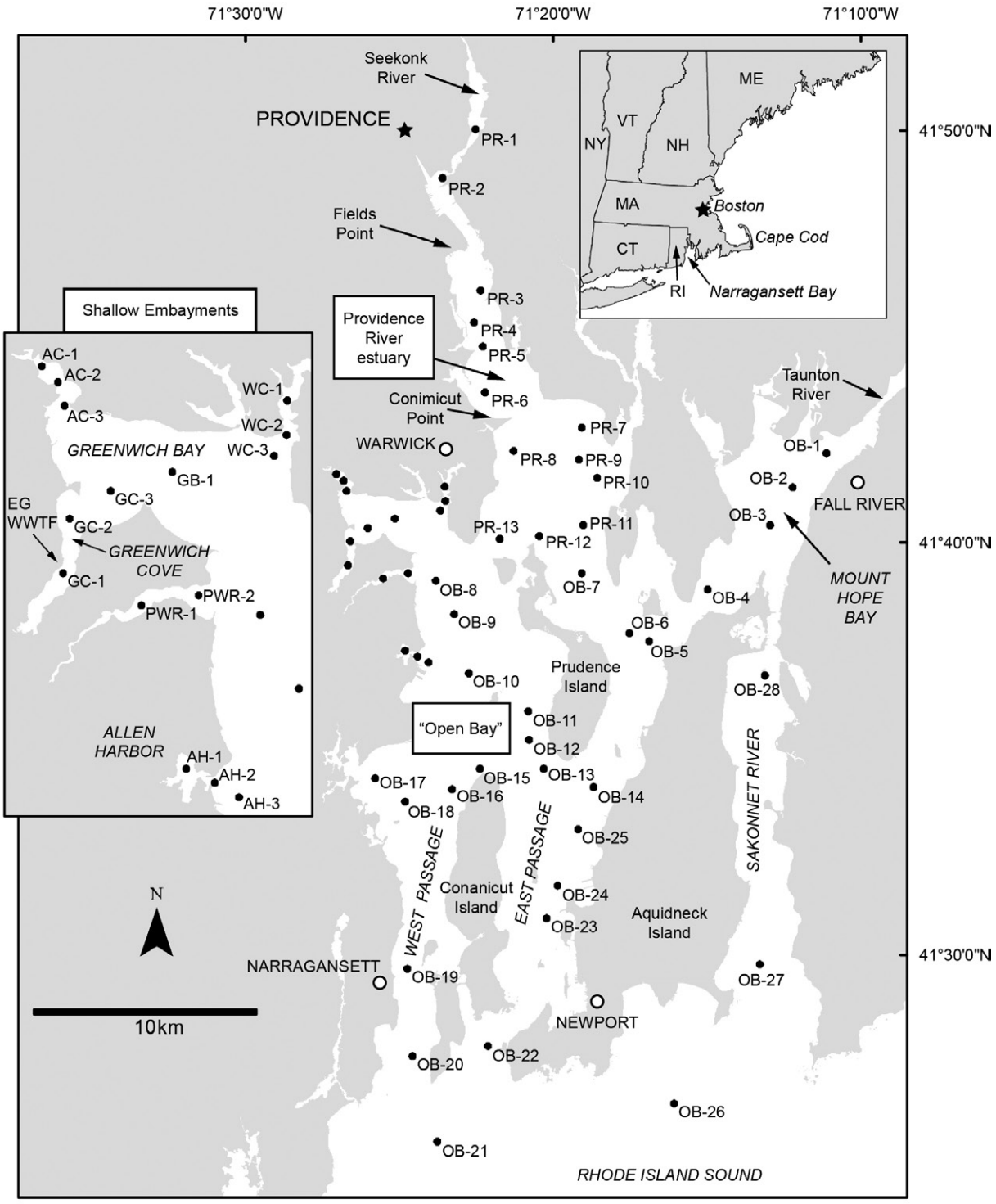


Figure 2. Locations within Narragansett Bay where sediment profile images were taken in 1988 and 2008. Three sub-regions used in the 1988 analyses are highlighted for comparison: Providence River Estuary stations (PR-); Open Bay (OB-); Shallow Embayments (inset). Note locations of Fields Point and East Greenwich (EG) Waste Water Treatment Facilities (WWTF).

(Figure 2) during a study of organic enrichment from wastewater treatment facility discharges (Valente et al. 1992). The 1988 imaging study provided the first *in situ* snapshot of benthic processes in Narragansett Bay soft sediments. Researchers were surprised by the finding that a substantial proportion of the Bay bottom had been exposed to high levels of organic deposition and low concentrations of dissolved oxygen (Granger et al. 2000). Many of the sites identified as having excessive organic enrichment and degraded benthic habitat were located in the Providence River Estuary or Shallow Embayments sub-regions of the Bay, near wastewater treatment facility outfalls, or in coves and other spatially constricted areas that received effluent (refer to Figure 2; Valente et al. 1992). These sites were sampled again in 2008 using the sediment profile imaging technique, providing the main data source supporting a status and trends assessment for Bay-wide benthic habitat quality (to be discussed in detail in Methods; Shumchenia et al. 2016).

Like most estuaries and coasts globally, there has been much human intervention and human-mediated change in Narragansett Bay and its Watershed since the 1980s. With increasing human population in the Watershed, land use has changed and impervious surfaces increased, wastewater treatment facilities have been improved, bottom-water hypoxic events continue, and warmer water temperatures are linked to decreased rates of primary production with food web implications. Given these and other trends over the past several decades, it is important to evaluate and monitor the response of benthic communities, which serve as excellent indicators of Narragansett Bay ecosystem quality.

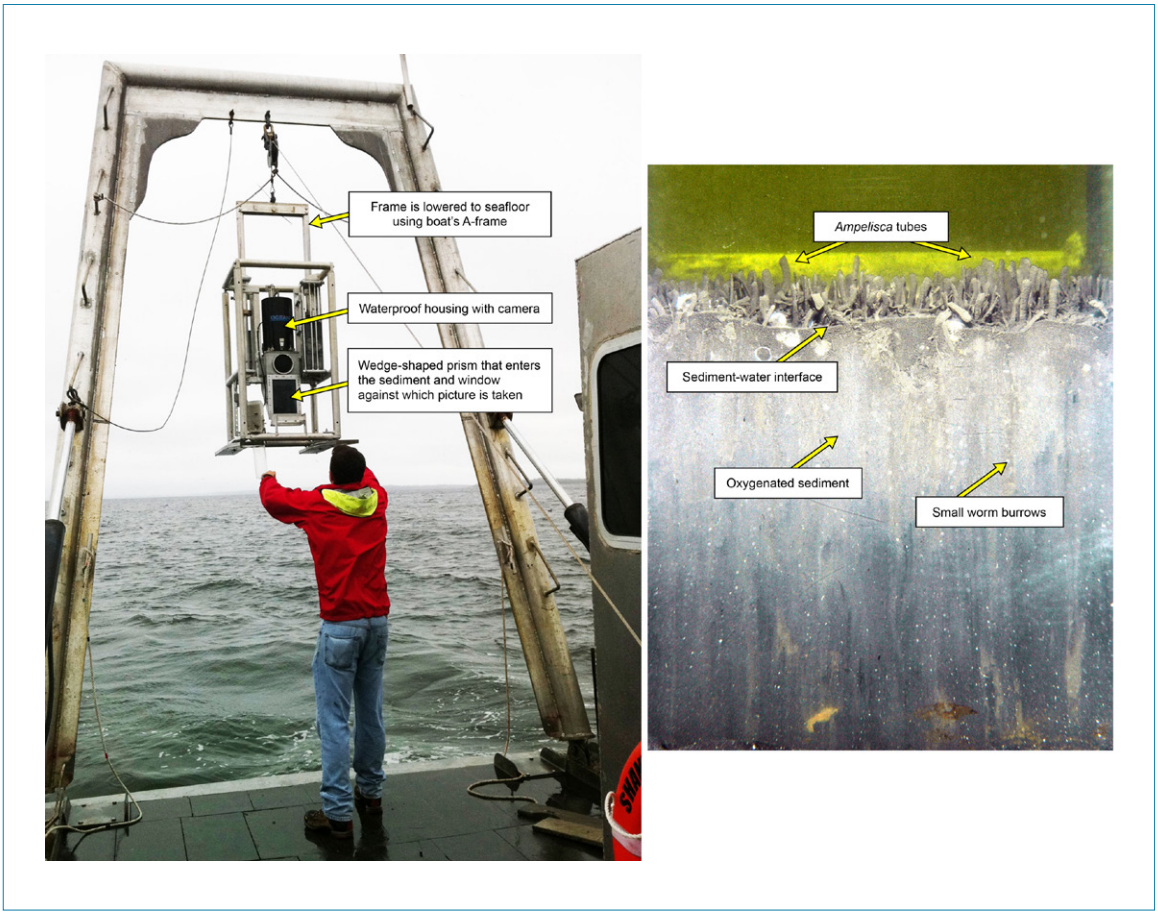


Figure 3. Sediment profile imagery equipment (left) consists of a frame that holds a waterproof housing containing a camera and a wedge-shaped prism with a window that enters the sediment and takes pictures of the sediment-water interface. A sediment-profile image (right) is about 5.9 inches (15 centimeters) across and shows organisms and evidence of organisms living on the surface and below the sediment. The image on the right was taken at station PR-9 in August 2008 (lower Providence River Estuary).

Methods

Researchers assessed benthic habitat diversity in 1988 and 2008 at 52 sites throughout the Bay using sediment profile imagery (Figure 2; Shumchenia et al. 2016). Sediment profile imagery is a rapid and efficient technique that collects "optical cores" of marine and estuarine sediments (Germano et al. 2011). A large frame is lowered from a boat, and once on the bottom, a wedge-shaped prism slices through the surface sediment, triggering a camera in a watertight housing to take pictures of the sediment-water interface and up to 7.9 inches (20 centimeters) below—the most biologically active zone of the sediment column (Figure 3; Shumchenia et al. 2016). This study aimed to characterize how higher temperatures and changes in the magnitude of anthropogenic nutrient inputs may have influenced benthic organic enrichment and habitat quality between 1988 and 2008.

To understand these stressor-response relationships, the researchers defined eight general habitats from sediment and biota descriptors that were present in both 1988 and 2008 (Table 2).

Using only the subset of stations that could be directly compared between years, the proportion of images for which the habitat changed between 1988 and 2008 was calculated for the entire Bay and for each Bay region (Providence River Estuary, Shallow Embayments, Open Bay; see Figure 2). For habitats that changed, the resulting 2008 habitat was noted. The habitat mosaic, or the spatial arrangement of different habitats, was described for each survey year Bay-wide and within Bay regions.

While the sediment profile imaging data reported here provided a whole-estuary characterization of benthic habitat quality change over time, there

Table 2. Descriptions of the eight habitats based on observed sediment type and biota.

Sediment category	Dominant biota	Other biota	Habitat
Organic-rich mud; mud; sandy mud; sand	*Ampelisca* spp.	Small tube-building fauna; shallow-burrowing fauna; *Beggiatoa* spp.; *Mulinia lateralis*	*Ampelisca* beds occasionally with other small tube-building and shallow-burrowing fauna on substrates ranging from organic-rich muds to sand. *Beggiatoa* spp. or *Mulinia lateralis* may be present.
Organic-rich mud	None dominant	*Beggiatoa* spp.; tube-building polychaetes; shallow-burrowing fauna	Organic-rich mud with tolerant species such as *Beggiatoa* spp., tube-building polychaetes, and shallow-burrowing fauna.
Mud with shell hash	Burrowing fauna	Tube-building polychaetes; deep-burrowing fauna	Burrowing fauna on mud with shell hash. Tube-building polychaetes or deep-burrowing fauna may be present.
Mud	Burrowing fauna	Tube-building polychaetes; *Chaetopterus*; deep-burrowing fauna	Burrowing fauna on mud. Tube-building polychaetes, larger tube-builders such as *Chaetopterus*, or deep-burrowing fauna may be present.
Sandy mud	None dominant	Burrowing fauna; tube-building fauna	Burrowing and tube-building fauna on sandy mud.
Mud	*Crepidula fornicata*	Mobile crustaceans; mobile gastropods; *Beggiatoa* spp.	*Crepidula* bed on mud. Mobile crustaceans, gastropods or *Beggiatoa* spp. may be present.
Very coarse sand with shell hash	None dominant	Rafting macroalgae; *Crepidula fornicata*	Very coarse sands with shell hash. Rafting macroalgae or *Crepidula* beds may be present.
Hard sand	None dominant	Epibenthic sponges; rafting macroalgae; attached macroalgae; mobile gastropods	Hard sands with epibenthic sponges, rafting or attached macroalgae, and/or mobile gastropods.

are some important caveats. First, benthic habitat quality was measured at just two points in time over a twenty-year span. Because benthic habitat quality was not measured continuously between 1988 and 2008, it is not possible to determine if and what changes occurred in each year between 1988 and 2008, nor can it be projected with a high degree of certainty what future assessments will find. Other limitations related to the technology and approach are discussed in Data Gaps and Research Needs.

This approach provided a Bay-wide quantitative analysis of the status and trends in benthic habitat quality over several decades. However, there is currently no established restoration target for benthic habitat quality in Narragansett Bay. The Discussion section describes anticipated changes in the benthic habitat mosaic under improving water quality conditions. With future work, these descriptions could form the basis of a restoration target.

Status and Trends

Based on the sediment profile imagery study, benthic habitat quality improved between 1988 and 2008 (Shumchenia et al. 2016). Habitats dominated by pollution-sensitive *Ampelisca* spp. tubiculous amphipods increased more than fivefold between 1988 and 2008 and expanded into the more urban, anthropogenically stressed Providence River Estuary (Figure 4c and d, Figure 5). In 1988, conditions did not favor widespread *Ampelisca* beds; stations with high organic loading and surface sediment indicated poor water quality conditions (Valente et al. 1992). Between 1988 and 2008, conditions theoretically became increasingly favorable for *Ampelisca* beds, as management strategies to reduce organic loadings and improve water quality were initiated, and changes in climate were noted (e.g., increase in temperature). In 2008, there was an increase in the proportion of *Ampelisca* beds Bay-wide, especially in areas where organic loading was known to be previously high. Water quality monitoring programs continue to record hypoxic events (see "Dissolved Oxygen" chapter and http://www.dem.ri.gov/bart), but it is possible that hypoxia occurs now over a smaller area, with less frequency and/or intensity than previous events (the first Bay-wide dissolved oxygen monitoring program did not begin until 1999; Prell et al. 2004).

Overall, the sediment profile imagery technique showed great promise to provide information rapidly that indicates benthic habitat quality. For example, the researchers noticed examples of both apparent habitat fidelity (Figure 4 a, b) and marked change (Figure 4 c, d). For sites showing a stark change in sediment type between years, apparent habitat change could actually reflect high degrees of small-scale benthic heterogeneity or a shift in sediment transport dynamics in the intervening twenty years. Of the 38 stations that could be reliably compared between years, habitat change was observed at 24 stations (63 percent). The Providence River Estuary saw the highest degree of benthic habitat change (eight out of ten stations; 80 percent), followed by Shallow Embayments (six out of ten stations; 60 percent) and Open Bay (ten out of eighteen stations; 55 percent).

The benthic habitat mosaic of 2008 is likely to represent improved conditions for the protection and growth of other organisms. When the critical boundaries of organic enrichment were in the more-shallow, protected (constricted) regions of the Bay, as in 2008, robust *Ampelisca* beds were likely to have served as structural habitat and provided food for other organisms like crustaceans and juvenile fish. When the critical boundaries of organic enrichment existed in deeper, less-protected waters, as in 1988, fewer *Ampelisca* beds were observed. With future warming and decreasing anthropogenic nutrient inputs, *Ampelisca* beds should be monitored more frequently as potential indicators of patterns in organic enrichment and important habitats.

Discussion

It has been suggested that *Ampelisca* spp. are organic enrichment opportunists (McCall 1977). There is also debate as to whether ampeliscids serve as indicators of impending hypoxia (Levin et al. 2009) or of improving conditions (Diaz et al. 2008, Rhoads and Germano 1986). A study of hypoxia in Greenwich Bay demonstrated that ampeliscids colonized degraded habitats soon after the resumption of normal oxygen conditions and were reliable indicators of improving water quality (Shumchenia and King 2010). Dense *Ampelisca* spp. communities in areas with high organic input and good water quality have been previously observed within Narragansett Bay and in Boston Harbor (Stickney and Stringer 1957, Diaz et al. 2008). The cessation of primary sewage discharges to Boston Harbor in the early 1990s appears to have set the stage for the observed widespread increases in *Ampelisca* spp. throughout the harbor. Prior to 1992, organic loading was high but water quality may have been too poor to allow *Ampelisca* spp. to thrive (Diaz et al. 2008). A decade later, declines in *Ampelisca* spp. tubes were associated with the reductions in organic loadings to the harbor and the eventual depletion of sediment organic inventories (i.e., surface sediment total organic carbon) (Diaz et al. 2008). This pattern

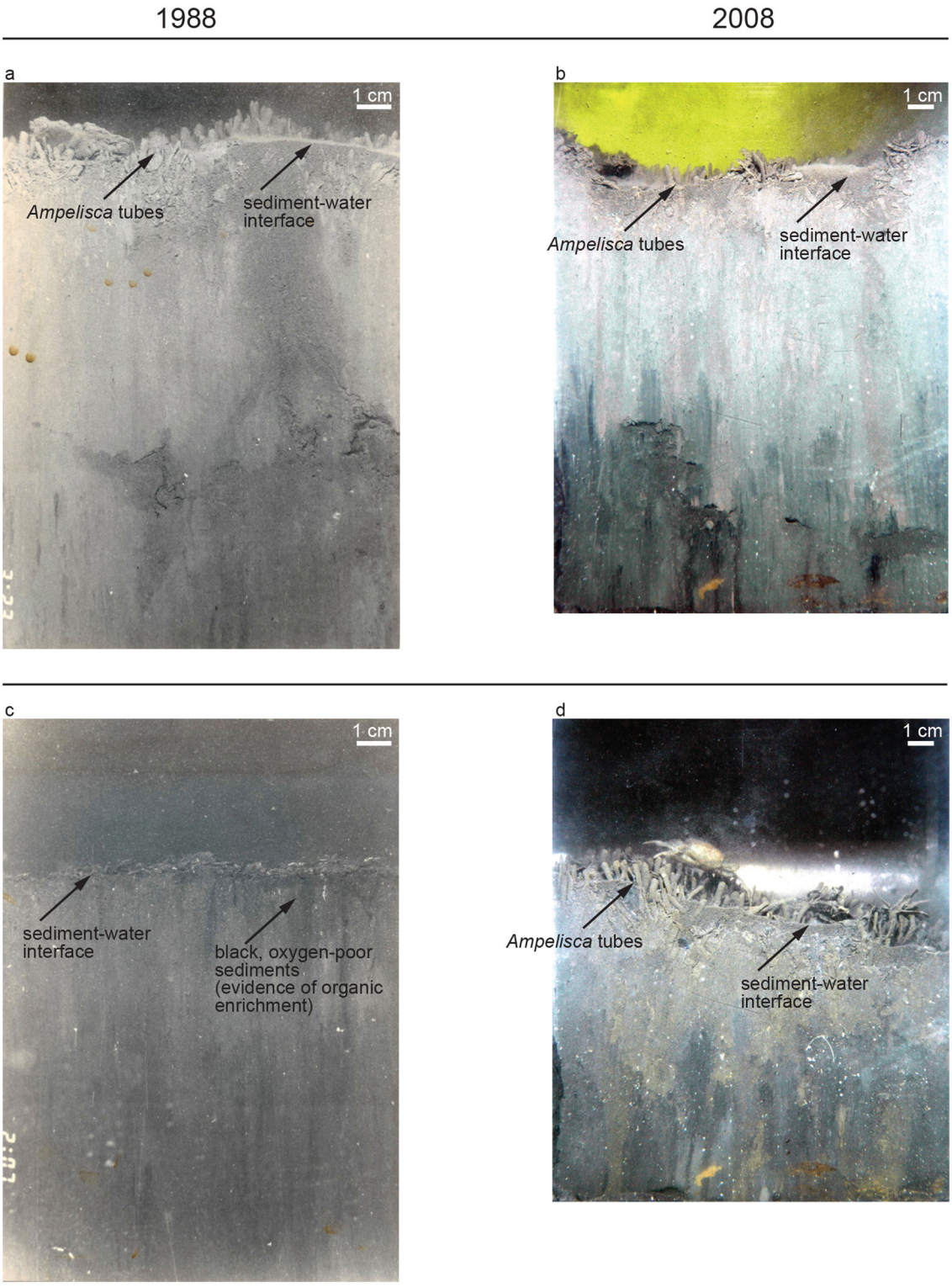


Figure 4. Upper: Example images of a site within Narragansett Bay (OB-2) where benthic habitat appeared to remain the same as *Ampelisca* beds between (a) 1988 and (b) 2008. Lower: Example images of a site (PR-3) where benthic habitat appeared to change from (c) organic-rich muds with pollution-tolerant species in 1988 to (d) *Ampelisca* beds in 2008.

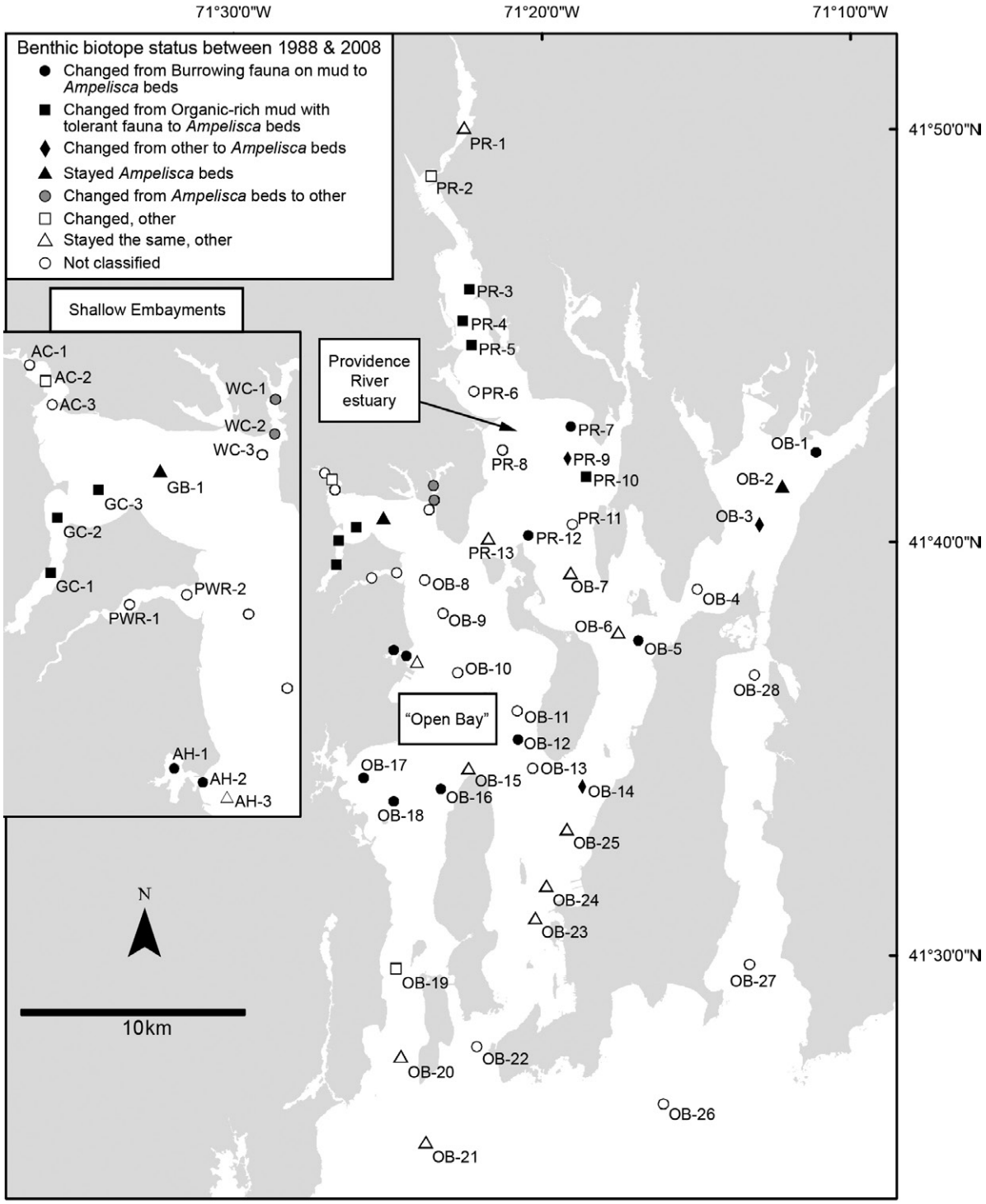


Figure 5. Benthic habitat change between the 1988 and 2008 surveys in Narragansett Bay was primarily from burrowing fauna on mud or organic-rich muds with pollution-tolerant species to *Ampelisca* beds.

indicates that *Ampelisca* spp. are the hallmark of a recovering benthic ecosystem but not a fully recovered benthic ecosystem. *Ampelisca* are an intermediate step on the path to lower organic loadings and improved water quality. A benthic ecosystem with lower organic loading will likely have different dominant species and few or no *Ampelisca* spp. However, reductions in organic matter and lower numbers of *Ampelisca* spp. in Boston Harbor apparently have coincided with positive changes in fish populations, as the recreational fishing community has made note of recent increases in winter flounder populations there (Powers 2015). Narragansett Bay could be exhibiting a pattern similar to the Boston Harbor example. Unfortunately, benthic habitats were not continuously monitored between 1988 and 2008, and it cannot be determined exactly where Narragansett Bay is located along the trajectory of highly degraded to restored habitat. However, the decrease in habitats showing organic enrichment and increase in *Ampelisca* beds suggests benthic habitats are improving.

Assuming that organic loadings in Narragansett Bay continue to decline, future surveys could find any of the following responses in benthic habitat quality.

1. No change: Similar extent of *Ampelisca* beds with good or improving water quality
2. Improving: Habitats reflecting more "mature" benthic communities–as were observed in the lower Bay in 2008, such as burrowing and tube-building fauna on sandy mud–increase under good or improving water quality conditions
3. Declining: Novel but more depauperate habitats emerge (such as with stress-tolerant species), paradoxically with good water quality

Data Gaps and Research Needs

- The sites characterized in 1988 and 2008 should be revisited every five years using sediment profile imagery to quantify benthic habitat type, conspicuous species, and sediment oxygen penetration to link benthic habitat quality with water column conditions.
- The sediment profile imaging technique used in this analysis may not adequately represent the presence of shellfish such as quahogs, soft-shell clams, and blue mussels, or larger fauna such as mantis shrimp and lobster. There is a need to coordinate benthic monitoring efforts in the upper Bay–such as any future sediment profile imagery surveys, the Narragansett Bay Commission's benthic video work, and the RIDEM's fish habitat projects–to provide a more complete assessment of benthic habitats.
- There is a need for future assessments of benthic habitat quality to incorporate measurements of benthic biogeochemistry, and for future benthic biogeochemistry studies to take a habitat-based approach.

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State of Narragansett Bay and Its Watershed
2017 Technical Report

Bay Ecosystem Condition Indicators

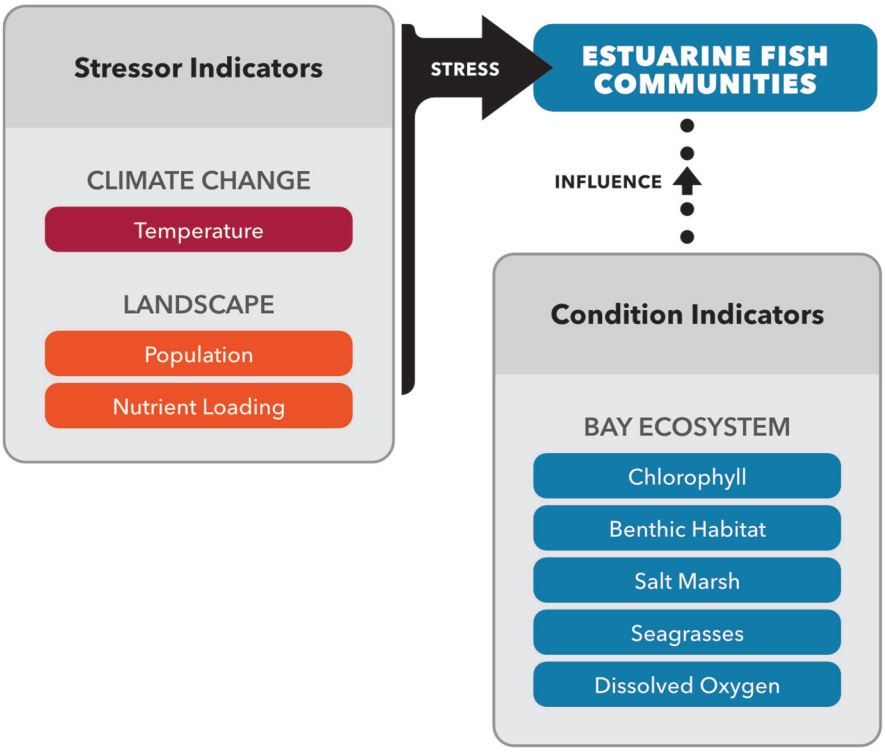
CHAPTER 14:
ESTUARINE FISH COMMUNITIES

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Photo: Striped Bass, Rhode Island Sound (Mike Laptew, www.mikelaptewfineartimages.com)

NARRAGANSETT BAY ESTUARY PROGRAM

Overview



BACKGROUND

- Estuarine fish are important parts of Narragansett Bay's food web, provide sustenance to humans, and serve as critical links between the estuary and the open ocean. Two ways of measuring estuarine fish communities are by the ratio of pelagic (live in the water column) to demersal (live near the bottom) species (the P:D ratio) and by the abundance of species that prefer warm or cold water.
- Many factors can cause changes in estuarine fish communities, such as fishing pressure from the human population, nutrient loading, contaminants, chlorophyll, dissolved oxygen, and loss of benthic, salt marsh, and seagrass habitats. However, warming of coastal waters–which has been documented over the last 60 years in Narragansett Bay and is projected to continue into the future–is an increasingly important stressor. Because their composition responds to warming waters and other stressors, estuarine fish communities are considered to be excellent indicators of ecosystem change.

KEY FINDINGS

- ***Status:*** Opportunistic seasonal migrant species such as scup (*Stenotomus chrysops*) and butterfish (*Peprilus triacanthus*) have displaced the demersal resident species in the Bay such as winter flounder (*Pseudopleuronectes americanus*) and red hake (*Urophycis*

chuss), partly due to increasing temperatures. These changes have altered the character of the Narragansett Bay fish community.

- ***Trends:*** Compared to the period before the 1980s, demersal species have become less numerous, warm-water species are more abundant, and cold-water species such as winter flounder (*Pseudopleuronectes americanus*) and American lobster (*Homarus americanus*) may have reduced rates of survival. As water temperatures in Narragansett Bay and the open ocean continue to warm, fish communities will be composed of fewer cold-water species and more warm-water species, reflecting a change from a typical southern New England temperate estuarine community to one that is tending toward a Mid-Atlantic estuarine community.

Introduction

Fish populations in Narragansett Bay are a vital ecological and economic resource. Fish are harvested commercially and recreationally, and they are important links in the estuarine food chain. Some fish in Narragansett Bay, such as menhaden (*Brevoortia tyrannus*), are consumed by birds, invertebrates, and larger fish, while others are top predators, such as striped bass (*Morone saxatilis*). Fish that spawn in rivers and migrate to the sea (e.g., herring species) represent the transport of nutrients between the upland portions of the Narragansett Bay Watershed and the open ocean. The composition of fish populations, therefore, helps determine what ecosystem services are provided in an estuarine ecosystem and in what proportions.

In general, fish populations fluctuate on interannual to multidecadal scales, and are influenced by a number of natural and anthropogenic factors. Relative to oceanic fish populations, those in estuaries are affected by more intense anthropogenic stressors that may alter their composition. For example, accessibility to fisheries, exposure to pollution, habitat loss, and introduced species are all stressors that are higher in Narragansett Bay and other estuaries than in the broader Northwest Atlantic Ocean (Collie et al. 2008). In addition, water temperatures in Narragansett Bay have warmed faster than the surrounding Northwest Atlantic Ocean surface waters (Smith et al. 2010, Fulweiler et al. 2015). This trend is expected to continue, or even intensify, as model forecasts indicate that the Northwest Atlantic Ocean could warm three times as fast as the global average (Saba et al. 2016). Because many fish species in Narragansett Bay already live at the edge of their thermal tolerances (Smith et al. 2010), warming waters are an important stressor to consider (see "Temperature" chapter).

Rising water temperatures are a direct impact from climate change on the Narragansett Bay ecosystem. Warmer waters should favor warm-water species over cold-water species. Fish will move to more favorable habitats, and scientists are already seeing examples of this at the scale of the Northeast Shelf Large Marine Ecosystem. A recent assessment of climate vulnerability of 82 northeastern US fish species found that more than 50 percent exhibit very high or high potential for a change in distribution due to their biological sensitivity and exposure to factors associated with a changing climate (Hare et al. 2016). Nye and colleagues (2009) documented changes in abundance, pole-ward shifts, and shifts to deeper water for several fish species in the Northwest Atlantic between 1968 and 2007. In addition, cold-water species may be unable to reproduce successfully in warmer waters, further reducing their numbers (Jeffries 1994, Jeffries and Johnson 1974). Locally, the average abundance of warm-water species caught in Narragansett Bay and Long Island Sound trawls between 1987 and 2000 increased, although this increase is likely not exclusively due to temperature (Wood et al. 2009).

Indirect impacts of climate change will also affect Narragansett Bay fish communities. For example, warmer conditions increase the predation rates of sand shrimp (*Crangon septemspinosa*) on juvenile winter flounder, perhaps leading to extreme local depletion (Taylor and Collie 2003). Warming waters are also disrupting the magnitude and timing of the winter-spring phytoplankton bloom in Narragansett Bay (Li and Smayda 1998, Oviatt et al. 2002; see "Chlorophyll" chapter). Because phytoplankton are the major primary producers in Narragansett Bay, changes in primary productivity represent an indirect impact of climate change that will have implications for the entire food web, including fish. Notably,

decreased phytoplankton biomass leads to lower quantities and quality of organic matter delivery to benthic invertebrates (Keller et al. 1999), altering the composition of benthic communities (see "Benthic Habitat" chapter). Changes in benthic communities over the past twenty years, resulting partly from changes in primary productivity, have the potential to impact fish species that rely on the benthos as a food source (Shumchenia et al. 2016).

Phytoplankton productivity and benthic productivity are also affected by the delivery of nutrients to the Bay. Nutrient input is another process affected by climate change that could represent an indirect impact on fish communities. In the nineteenth and twentieth centuries, nutrient inputs to the Bay rose as human population increased (Nixon and Fulweiler 2012). Increases in total precipitation and increases in extreme precipitation events due to climate change have facilitated the delivery of terrestrial-derived nutrients to the Bay (Smith et al. 2010). Only recently has nutrient pollution of the Bay decreased markedly through management actions such as improvements in wastewater treatment facilities (see "Nutrient Loading" chapter).

Other factors that have direct impacts on fish community composition in Narragansett Bay include harvesting and habitat loss. Both of these factors must be considered when interpreting trends in the context of climate change. From analysis of the earliest available quantitative fisheries data in the late 1800s, it was apparent that both habitat loss and harvesting had already altered fish community composition (Oviatt et al. 2003). Anadromous species, bay scallop (*Argopecten irradians*), and oyster (*Crassostrea virginica*) have all declined or nearly disappeared from the Narragansett Bay estuarine community as a result of habitat loss and exploitation, and overall fish abundance decreased 81 percent between 1898 and 1999 (Oviatt et al. 2003). In the case of bay scallop, habitat loss was due to nutrient enrichment. Beds of eelgrass (*Zostera marina*), a vital habitat for many fish and shellfish species including bay scallop, were lost due to nutrient inputs and shading from excessive phytoplankton blooms (Oviatt et al. 2003; see "Seagrasses" chapter).

Trends in the estuarine fish community detected in the available data likely resulted from a complex and cumulative combination of natural and anthropogenic stressors—one that has changed over time. Because the Bay has been fished intensively for hundreds of years, there were potentially significant changes to habitats and fish communities prior to the onset of the quantitative monitoring programs that began in the mid-twentieth century at the University of Rhode Island Graduate School of Oceanography (GSO) and at the Rhode Island Department of Environmental Management (RIDEM). Therefore, the fish communities characterized in the 1960s likely represented an already-shifted baseline (Oviatt et al. 2003). These two trawl surveys constitute the most continuous and comprehensive datasets on Narragansett Bay fish communities. The GSO trawl, conducted since 1959, is one of the longest continuous records of fish and invertebrate relative abundance in the world (Taylor and Collie 2000).

In this chapter, the Narragansett Bay Estuary Program reports results from two methods of measuring estuarine fish communities: (1) the ratio of pelagic (live in the water column) to demersal (live near the bottom) species (the P:D ratio) and (2) the abundance of species that prefer warm or cold water. Datasets from the GSO and RIDEM fish trawl surveys provide information to support these measures since 1959 and 1983, respectively. In addition to fish, this chapter includes findings related to lobster (*Homarus americanus*). This chapter discusses these characteristics of the estuarine fish community in the context of multiple stressors including temperature, chlorophyll, nutrient loading, and other condition indicators such as seagrasses and benthic habitats.

Methods

Given the existing body of previous research characterizing trends in Narragansett Bay's fish communities, the Narragansett Bay Estuary Program sought to distill the indicators already in use by GSO and RIDEM fish monitoring programs: the P:D ratio, the abundance of key cold-water species, and the abundance of key warm-water species.

Data sources for these indicators include annual mean abundance of fish species at all fixed and random Narragansett Bay stations from the RIDEM fish trawl between 1983 and 2012 (data provided by RIDEM Marine Fisheries) and annual mean abundance per trawl at the Fox Island (Middle-West Passage) station from the GSO fish trawl between 1959 and 2012 (obtained from www.gso.uri.edu/fishtrawl16)[1] (Figure 1). Characteristics of each survey and major differences between them are important to note when considering the datasets (Table 1). A comparison by the Rhode Island Marine Fisheries Institute determined that the two surveys displayed similar trends for most species, but rates

[1] At the time that the Estuary Program made data requests for this report in 2014, data were compiled by these entities through 2012.

of catchability are different between trawls (RIMFI 2015). For example, it is apparent from comparing the tow duration and speed in Table 1 that the GSO trawl samples along a slightly longer transect (1.15 miles, 1.85 kilometers) than the RIDEM trawl (0.96 mile, 1.54 kilometers).

The list of the 25 most-abundant species caught in the GSO trawl (Collie et al. 2008) was used as a starting point for this analysis. Five invertebrate species on this list (*Cancer irroratus, Cancer borealis, Libinia emarginata, Asterias forbesi,* and *Ovalipes ocellatus*) were not reported in the RIDEM data obtained by the Narragansett Bay Estuary Program (Table 2). Therefore, each trawl dataset was reduced to species in common, for a total of 21 species, and each species was categorized as having either pelagic or demersal habitat (Table 2).

To examine potential indirect impacts of climate change on the estuarine fish community, the P:D ratio was calculated for each year of available data for each trawl. Following Collie and colleagues (2008), the P:D ratio was calculated as the sum of individuals defined as pelagic species divided by the sum of individuals defined as demersal species. To examine potential direct impacts of climate change on the estuarine fish community, the abundance of four warm-water species (butterfish, longfin inshore squid, scup, summer flounder; Table 2; based on Collie et al. [2008]) were plotted over time with *in situ* sea surface temperature recorded during the

Table 1. Characteristics of the two Narragansett Bay fish trawl surveys discussed in this report: the University of Rhode Island Graduate School of Oceanography (GSO) fish trawl and the Rhode Island Department of Environmental Management (RIDEM) fish trawl.

	GSO Fish Trawl	RIDEM Fish Trawl (Narragansett Bay stations only)
Temporal Coverage and Sampling Frequency	1959–present, weekly	• 1979–present: Stratified random stations, seasonal • 1988–present: Fixed stations, seasonal • 1990–present: Fixed stations, monthly
Number of Stations	2*	• Monthly, fixed: 12 • Seasonal, fixed: 12 • Seasonal, random: 14
Station Locations	Fox Island station (Middle West Passage) Whale Rock station (Mouth of Narragansett Bay)	Narragansett Bay including Mount Hope Bay and Sakonnet River, excluding Greenwich Bay and Providence River estuary
Trawl Methods	• 3-inch net mesh, 2-inch cod-end mesh • Steel otterboards • Tow duration: 30 minutes • Towing speed: 2 knots	• Otter trawl with 4.5-inch net mesh in wings, 3-inch mesh in main body, 2-inch mesh at cod end • Currently Thyboron Type II trawl doors (wooden Tomkiewicz flat otter trawl doors until 2012) • Tow duration: 20 minutes • Tow speed: 2.5 knots
Units Reported	Number of individuals (by species) per tow	Number of individuals (by species) per tow
Additional Variables Recorded	• Surface and bottom water temperature (since 1959) • Surface and bottom salinity (since 2006) • Surface and bottom dissolved oxygen (since 2006)	• Wind speed and direction • Sea condition • Air temperature • Cloud cover • Surface and bottom water temperature • Net performance metrics

*Data from only one station—Fox Island—were included in the analyses reported in this chapter. The other station is more representative of a Rhode Island Sound (open ocean) fish community.

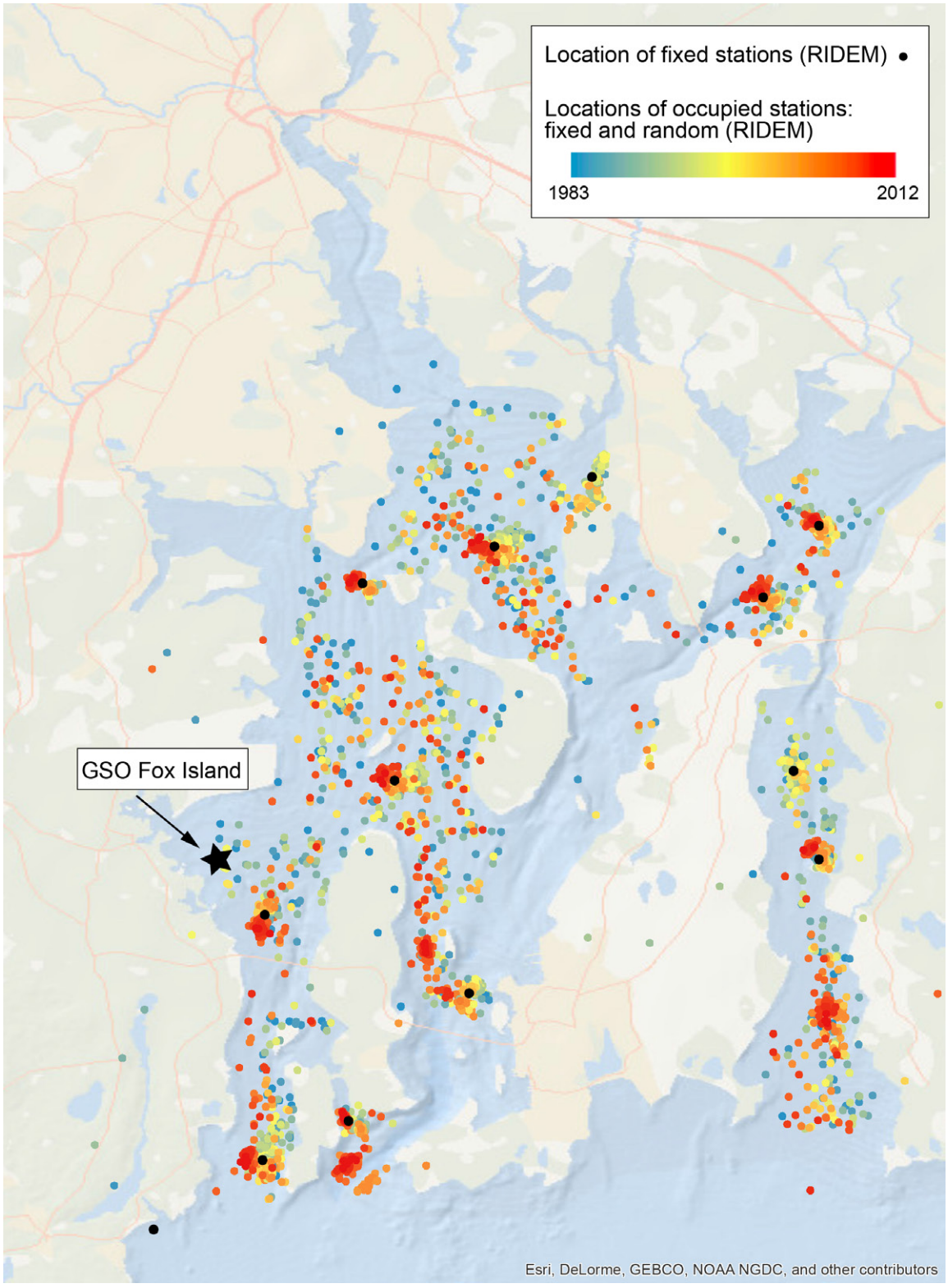


Figure 1. Locations of fish trawl stations from the University of Rhode Island Graduate School of Oceanography (GSO) and the Rhode Island Department of Environmental Management (RIDEM) fish trawl surveys that were used in the Narragansett Bay Estuary Program analysis of estuarine fish communities.

GSO trawls and bottom water temperature recorded during the RIDEM trawls. These four species are of interest because they are some of the most abundant species in the GSO trawl (see Table 2). The abundance of winter flounder and American lobster, both cold-water demersal species, were also plotted over time. Winter flounder and American lobster are of particular interest because historically they were abundant in the Bay (Collie et al. 2008) but have shown sharp declines recently.

The P:D ratio, species abundances, and water temperatures were plotted separately for each GSO and RIDEM survey, without applying any correction factors. Because the RIDEM data are summarized as annual mean abundance across all Narragansett Bay sites and the GSO data are summarized as annual mean abundance per trawl, the magnitudes of the abundances should not be directly compared between survey programs.

Status and Trends

Previous analyses from both surveys indicated that the abundance of demersal fish species had declined relative to pelagic species (Oviatt et al. 2003, Oviatt 2004, Collie et al. 2008). The P:D ratio for Narragansett Bay was variable over time but generally increased between 1959 and 2012, suggesting

Table 2. Fish species caught in regular monitoring of Narragansett Bay by the University of Rhode Island Graduate School of Oceanography (1959 to 2012) and the Rhode Island Department of Environmental Management (1983 to 2012). Species are listed in order of greatest to least mean abundance per tow (1959 to 2005) from the GSO trawl.

Species Common Name	Species Scientific Name	Habitat
Scup	*Stenotomus chrysops*	Demersal
Winter flounder	*Pseudopleuronectes americanus*	Demersal
Butterfish	*Peprilus triacanthus*	Pelagic
Silver hake	*Merluccius bilinearis*	Demersal
Longfin inshore squid	*Loligo pealeii*	Pelagic
American lobster	*Homarus americanus*	Demersal
Little skate	*Leucoraja erinacea*	Demersal
Windowpane	*Scophthalmus aquosus*	Demersal
Red hake	*Urophycis chuss*	Demersal
Atlantic herring	*Clupea harengus*	Pelagic
Northern sea robin	*Prionotus carolinus*	Demersal
Longhorned sculpin	*Myoxocephalus octodecemspinosus*	Demersal
Fourspot flounder	*Paralichthys oblongus*	Demersal
River herring*	*Alosa* spp.	Pelagic
Cunner	*Tautogolabrus adspersus*	Demersal
Summer flounder	*Paralichthys dentatus*	Demersal
Horseshoe crab	*Limulus polyphemus*	Demersal
Bluefish	*Pomatomus saltatrix*	Pelagic
Striped searobin	*Prionotus evolans*	Demersal
Tautog	*Tautoga onitus*	Demersal
Weakfish	*Cynoscion regalis*	Pelagic

* includes alewife (*Alosa pseudoharengus*) and blueback herring (*Alosa aestivalis*)

that increasing temperatures, eutrophication, and harvesting were continuing to affect fish community composition (Figure 2). The longer time period of data provided by the GSO survey revealed the major shift in fish community composition that occurred in the 1980s (Figure 2; Collie et al. 2008). While P:D ratios varied within the two periods—before (1959 to 1980) and after (1981 to 2012) the shift—the average magnitude of the ratio differed greatly between the two periods. There also appears to have been a declining trend beginning in the mid-1990s. In the RIDEM survey, from which data were provided since 1983, P:D ratios were variable, but with an increasing trend for the duration of the data series. The RIDEM trawl data had substantially higher P:D ratios than the GSO trawl data. This may be attributable to differences between trawl methods noted in Table 1, including the greater number of stations over a wider area represented by the RIDEM data.

The abundance of warm-water species increased slightly since 1959, according to the results of both trawl surveys (Figures 3a and 3c; Collie et al. 2008). Both datasets showed large increases in the variability in their respective metrics since the beginning of each time series, suggesting that along with increasing temperatures (Figures 3b and 3d), other factors were influencing fish species abundance.

American lobster and winter flounder abundance data showed different trends (Figure 4). Lobster abundance was low from the 1960s through the 1980s, increased in the mid- to late 1990s, and then abruptly declined from the early 2000s to 2012. This recent decline corresponds with trends reported in stock assessment reports (ASMFC 2015) and studies of American lobster nurseries in Narragansett Bay (Wahle et al. 2015). Winter flounder, on the other hand, were most abundant in the earliest segment of the time series and declined since the 1980s (Figure 4; Figure 5, left panel). This decline is consistent with previous findings that winter flounder abundances have decreased 90 percent since 1980 in Narragansett Bay (Oviatt 2004), linked to warmer winter water temperatures (Jefferies 1994, Jefferies and Johnson 1974, Taylor and Collie 2003). Declines in winter flounder likely released lobster and other decapods from predation pressure, which in turn led to increases in their abundance during this same time period (Figure 5, center panel; Oviatt et al. 2003).

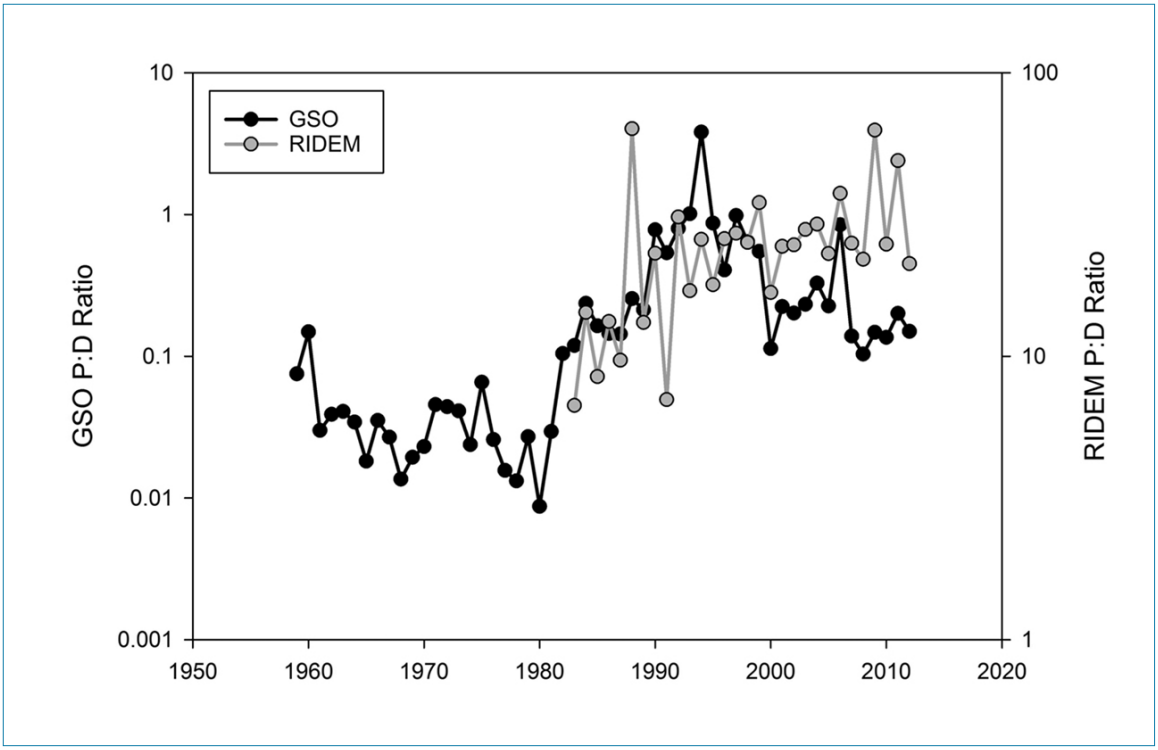


Figure 2. Narragansett Bay pelagic-to-demersal (P:D) ratio for the fish community calculated from abundance-per-tow data for GSO (black line) and mean annual abundance data for RIDEM (grey line) trawl surveys. GSO = University of Rhode Island Graduate School of Oceanography. RIDEM = Rhode Island Department of Environmental Management. Note the difference in scales between the GSO and RIDEM P:D ratios.

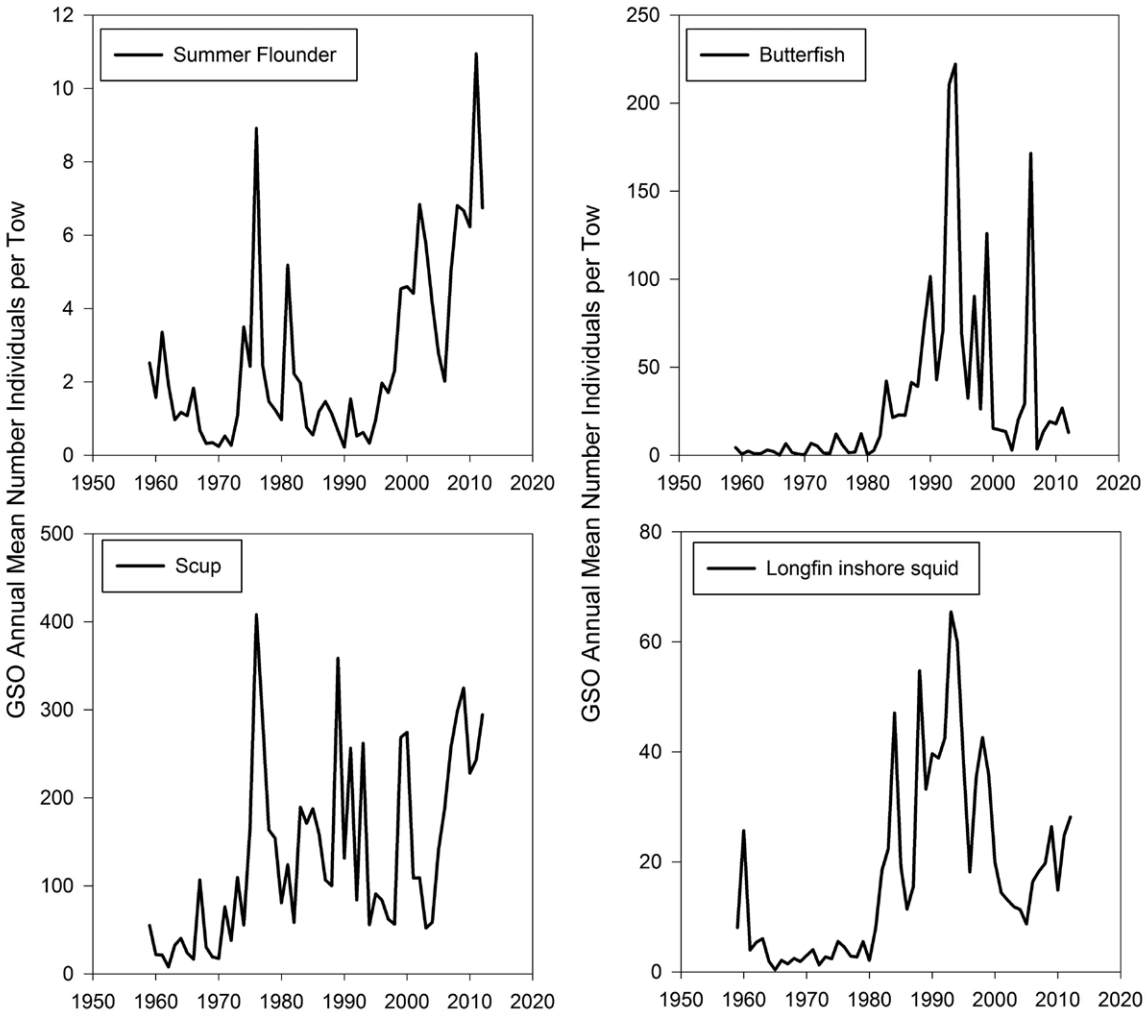


Figure 3a. Mean annual number of individuals per tow for four warm-water fish species caught at the Fox Island (Middle West Passage) station of the University of Rhode Island Graduate School of Oceanography (GSO) trawl survey. *Clockwise from top left:* summer flounder, butterfish, longfin inshore squid, and scup.

As the warming trend continued into the 2000s, and warm-water species that were previously summer migrants–including scup and black sea bass–have remained in the Bay longer each year and taken lobsters and other decapods as prey, the abundance of those decapods has declined. The combination of increased predation from summer migrants and the effects of warmer temperatures on survival and reproduction rates has likely been responsible for the observed low abundance of American lobster in the Bay today (Figure 5).

Discussion

The ratio of pelagic to demersal fish species abundance–the P:D ratio–has been used as an indicator of changing fish communities in Narragansett Bay and elsewhere (Collie et al. 2008). Increases in the P:D ratio of fish communities have been correlated with nutrient enrichment (de Leiva Moreno et al. 2000) and fishing pressure on demersal species (Rochet and Trenkel 2003). However, Collie and colleagues (2008) found that the P:D ratio from 1959 to 2005 in Narragansett Bay was correlated with warming temperatures. The temperature increase was correlated with declines in winter flounder, red hake, and silver hake–demersal species that reside in Narragansett Bay in the winter months–and increases in butterfish and scup–warm-water species that migrate into the Bay in summer. These results suggest that opportunistic seasonal migrant species have displaced the demersal resident species in the Bay, partly due to increasing temperatures. All of these changes have altered the character of the Narragansett Bay fish community from a typical southern New England temperate estuarine community to one that is tending toward a Mid-Atlantic estuarine community (Collie et al. 2008).

In recent years, lobster abundance has declined markedly, likely due to a combination of factors. American lobster is emblematic of a species subject to cumulative stressors, exacerbated by climate change. The southern New England lobster stock was reported to be in a depleted condition in 2006 (ASMFC 2015). At that time, managers developed a "rebuilding timeline," but assessments in 2009 showed continued declines. To address the decline, managers implemented a ten percent reduction in exploitation of the stock (ASMFC 2015). In 2012, further management actions were taken to reduce

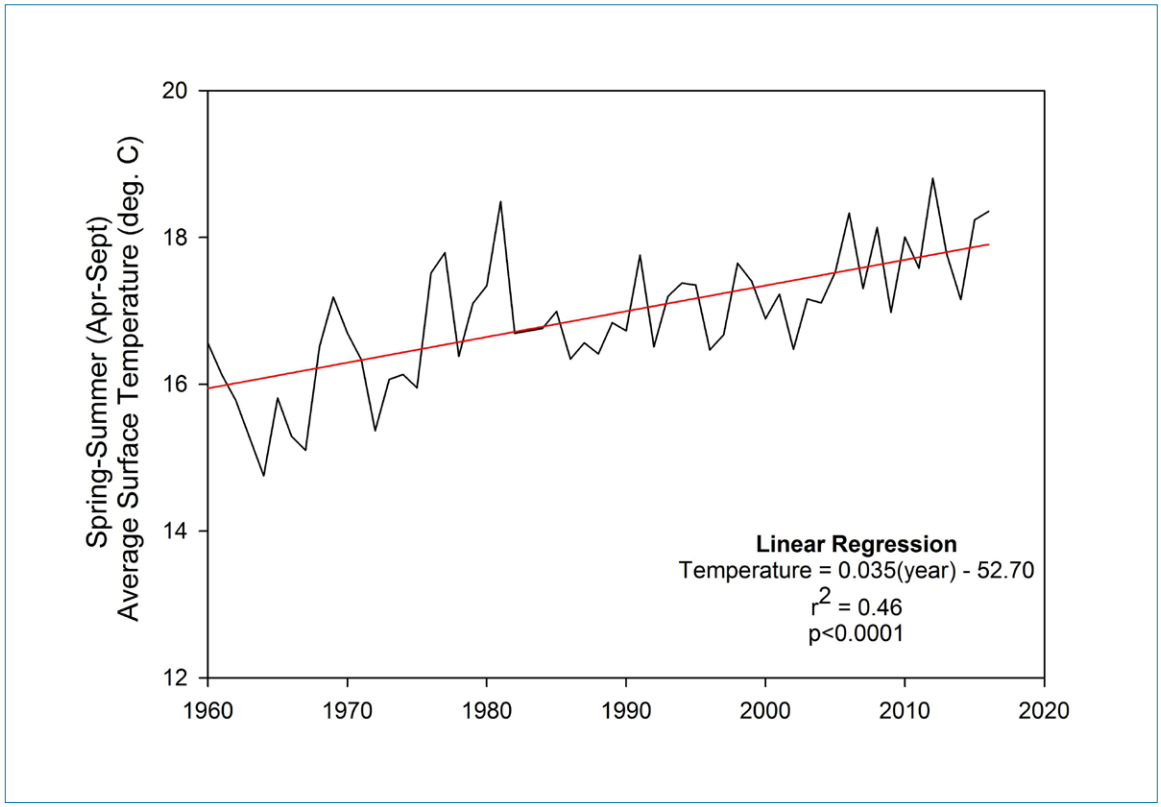


Figure 3b. *In situ* mean spring-summer sea surface temperature from the GSO trawl survey (https://www.gso.uri.edu/fishtrawl16/).

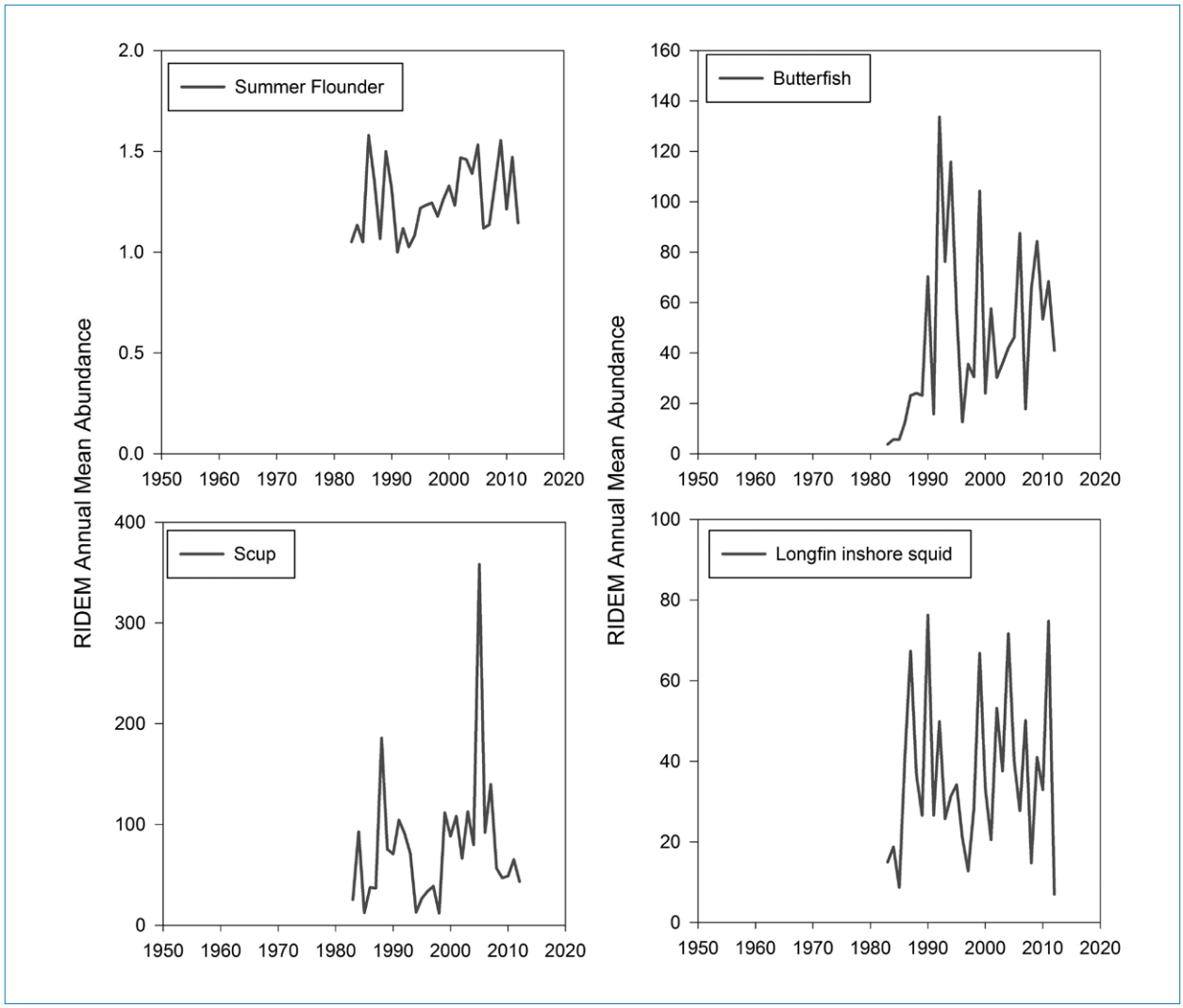


Figure 3c. Annual mean abundance (mean number of individuals caught across all trawls per year) for four warm-water fish species caught in the Rhode Island Department of Environmental Management (RIDEM) trawl survey. *Clockwise from top left: summer flounder, butterfish, longfin inshore squid, and scup.*

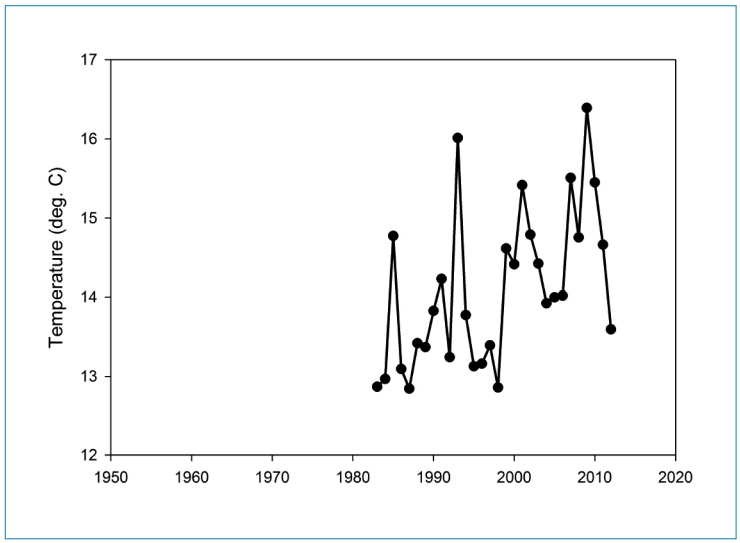


Figure 3d. *In situ* mean annual bottom-water temperature from the RIDEM trawl survey.

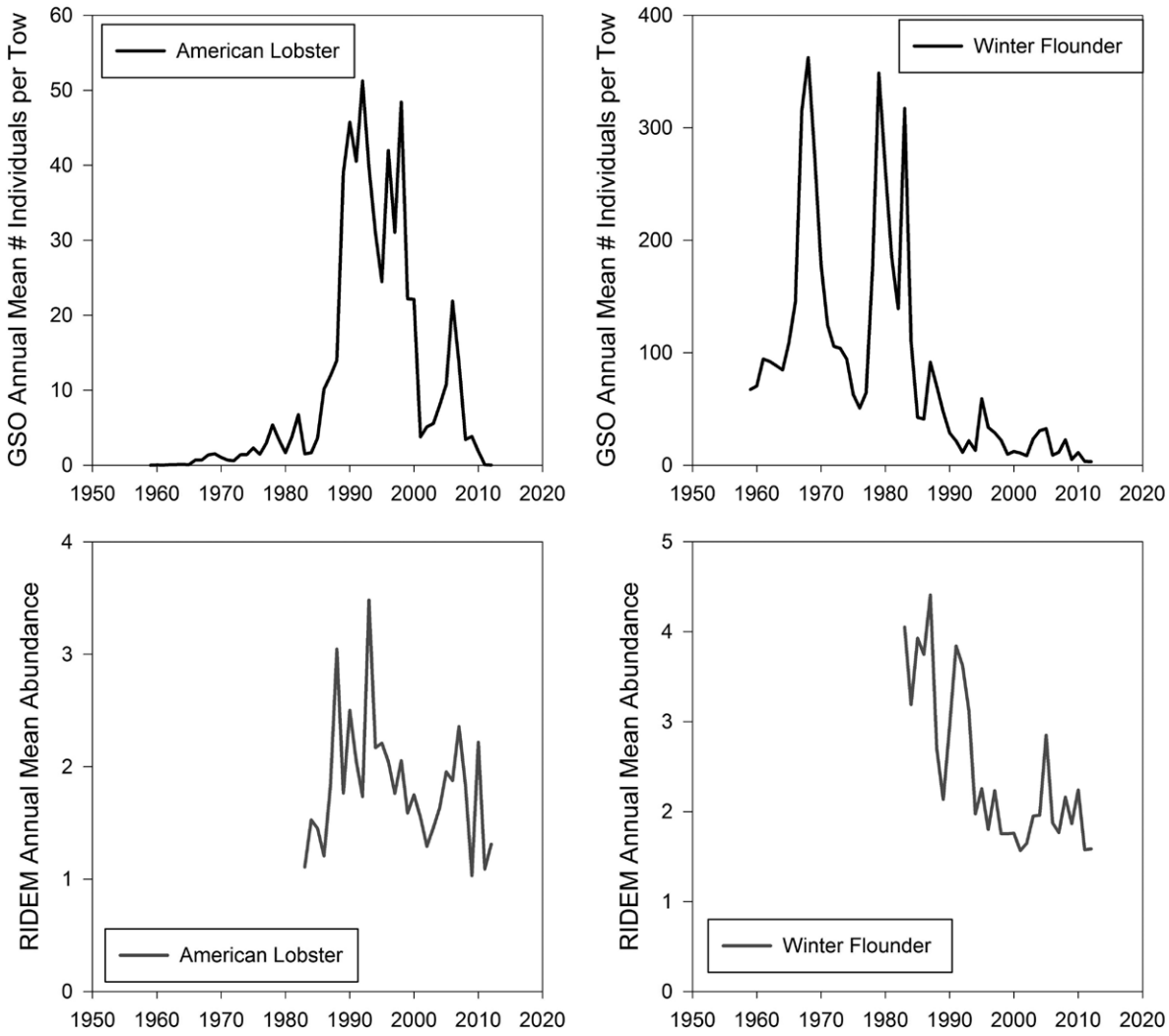


Figure 4. Mean annual number of individuals per tow (GSO trawl, top row) and mean annual abundance (RIDEM trawl, bottom row) for American lobster (left column) and winter flounder (right column). GSO = University of Rhode Island Graduate School of Oceanography. RIDEM = Rhode Island Department of Environmental Management.

traps allocated inshore by 50 percent and offshore by 25 percent over five years. However, the lobster stock reached a record low in 2013, and the inshore portion was classified as "collapsed" (ASMFC 2015). As the inshore population has declined, fishing pressure has increased on the offshore population (ASMFC 2015). It is important to note that although this stock is depleted, overfishing is not occurring–indicating that there are ecological factors are at play, such as recruitment failure potentially due to increasing temperatures. Because the inshore lobster population (including nursery habitats in Narragansett Bay) is critical to sustaining both the inshore and offshore populations, measures were taken to reduce fishing pressure in both areas (ASMFC 2015). As previously noted, juvenile or young-of-the-year American lobsters have declined in abundance throughout Narragansett Bay based on a comparison of findings from 1990 and 2011/2012 (Wahle et al. 2015). Juvenile lobsters, once occurring throughout the mid and lower Bay, are now mostly restricted to deeper waters at the mouth of the Bay and offshore (Wahle et al. 2015). The optimal temperature range throughout the American lobster life history is 12 to 18°C (53.6 to 64.4°F). With portions of the inshore southern New England lobster stock recently experiencing prolonged periods above 20°C (68°F), these habitats have become extremely stressful to lobster and are likely contributing to the lack of recruitment (Fogarty et al. 2007, ASMFC 2015).

Shell disease, another stressor for the southern New England lobster population, is not directly linked to increased water temperatures, but shell disease susceptibility does appear to be associated with environmental stressors in general such as temperature, salinity, and oxygen conditions (ASMFC 2015). Studying shell disease in Massachusetts, Glenn and Pugh (2006) reported a much higher incidence in Buzzards Bay lobsters, compared to other regions of the state, and suggested warm waters as a prime factor contributing to shell disease. Prevalence of shell disease in southern New England has increased since the late 1990s and is a contributing factor to the status of the stock observed today.

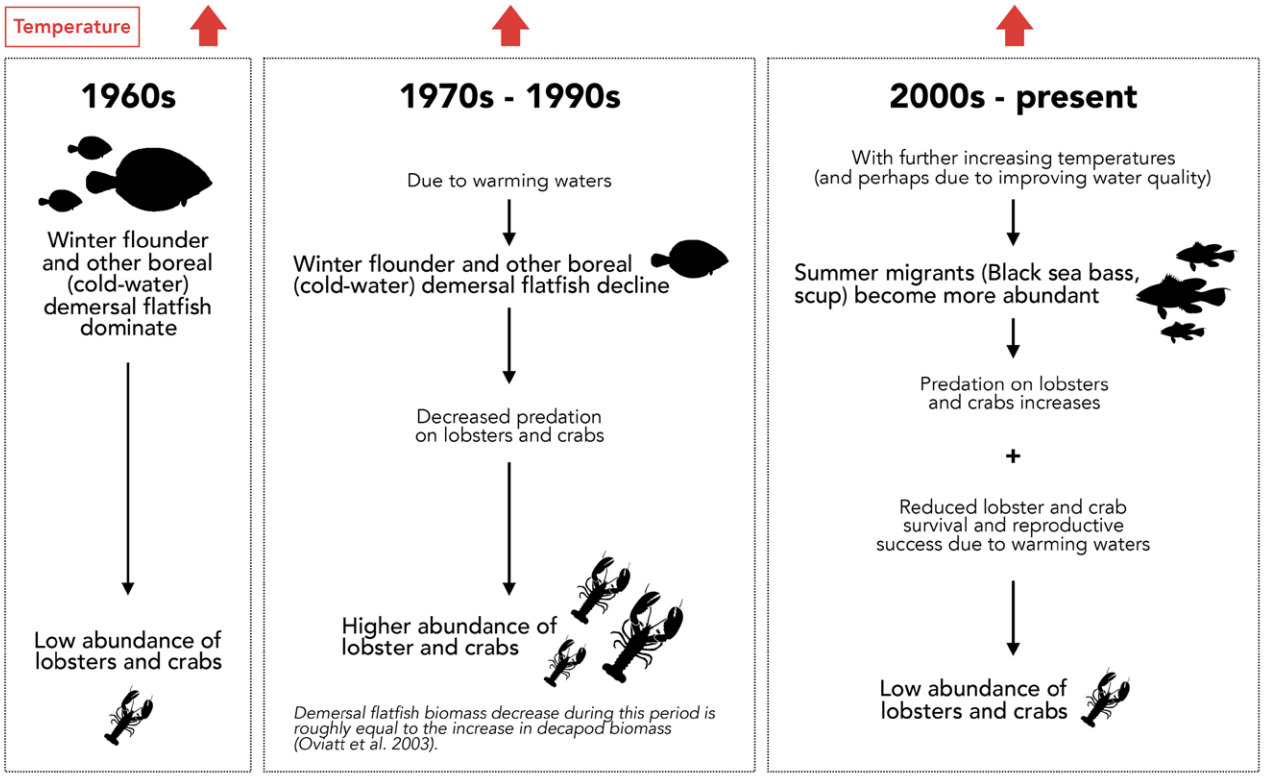


Figure 5. A timeline and description of changes to the Narragansett Bay fish community due to changing abundances of cold-water demersal species such as winter flounder, lobsters, and crabs, and summer migrants like black sea bass and scup, all linked with increases in temperature and shifts in predation.

Winter flounder face a similar combination of stressors: population effects from fishing mortality and the effects of warming waters (Oviatt et al. 2003, Collie et al. 2008, Smith et al. 2010). Fisheries yield for winter flounder is estimated to have declined by a factor of more than six between 1900 and 1999 (Oviatt et al. 2003). Winter flounder lay their eggs in winter, when predators such as sand shrimp are either absent or dormant (Oviatt 2004). As winter water temperatures have increased over the past several decades, sand shrimp have remained active during winter and can therefore prey on winter flounder larvae (Whitehouse 1994, Taylor and Collie 2003). In addition, water temperatures in winter may have increased beyond the optimal temperature for winter flounder egg health (Keller and Klein-MacPhee 2000). All of these factors contribute to low winter flounder reproductive success and present some possible explanations for the observed declines.

An "unnatural experiment" for helping to determine the potential effects of increased temperatures of fish stocks has taken place in Mount Hope Bay since the establishment of heated-water discharge from the Brayton Point Power Station (BPPS) in Somerset, Massachusetts, in 1963 (Rountree and MacDonald 2006). In 1984, a drastic decline in winter flounder populations was observed in Mount Hope Bay. To determine if temperature increase associated with the BPPS was influencing winter flounder, windowpane, hogchoker, tautog, and scup abundance, DeAlteris and colleagues (2006) compared trends within Mount Hope Bay with trends throughout Narragansett Bay. They found that there were no differences in abundance trends for these species between Mount Hope Bay and Narragansett Bay, and suggested that large-scale factors such as overfishing, climate change, and increased predator abundance were more important drivers of fish community change. It is the interaction of warming with one or more of these factors (e.g., increased predator abundance/activity) that is predicted to lead to further declines in cold-water species such as winter flounder in Narragansett Bay (Smith et al. 2010).

Oviatt et al. (2003) suggested that warming waters, an increase in available estuarine space from fish populations reduced by harvesting, and temperature stress have all facilitated the observed increase in warm-water species in Narragansett Bay. Aside from changes to the ecology of the Bay, these shifts will likely cause a change in fishing practices as "new" species establish in Bay waters. The results of fish population research, like those described in this chapter, are just beginning to inform the adaptation of new policies and fishery management schemes to climate change. For example, the Gulf of Maine Research Institute has developed the report *Preparing for Emerging Fisheries: An Overview of Mid-Atlantic Stocks on the Move* (Hudson and Peros 2013). The report discusses trends and management options for warm-water species that are increasingly common in the Gulf of Maine (as well as Narragansett Bay) including butterfish, black sea bass, summer flounder, squid, and scup. Recommendations from the report include allowing fishermen flexibility to take advantage of new fishing opportunities in an effort to offset losses that may be occurring in existing fisheries with decreasing quotas.

Data Gaps and Research Needs

- Analyses are needed to better characterize the comparability of the GSO and RIDEM trawl data over time, including an examination of the timing and effects of any gear changes.
- There is a need to convene experts to advise on other approach(es) to use in the future to characterize changes in estuarine fish communities, including consideration of different or additional focal species, and different or additional metrics, such as a weighted-mean preferred temperature metric (e.g., Collie et al. 2008).
- Data on estuarine fish communities in the Upper Bay, including the Providence River Estuary and Greenwich Bay, were not included in this analysis. Existing data on those areas need to be compiled and analyzed to provide a more complete understanding of Bay-wide trends.
- This chapter only analyzed the RIDEM and GSO datasets through 2012. Data collected since 2012 need to be analyzed to identify more recent changes in the estuarine fish community.

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State of Narragansett Bay and Its Watershed
2017 Technical Report

Bay Ecosystem Condition Indicators

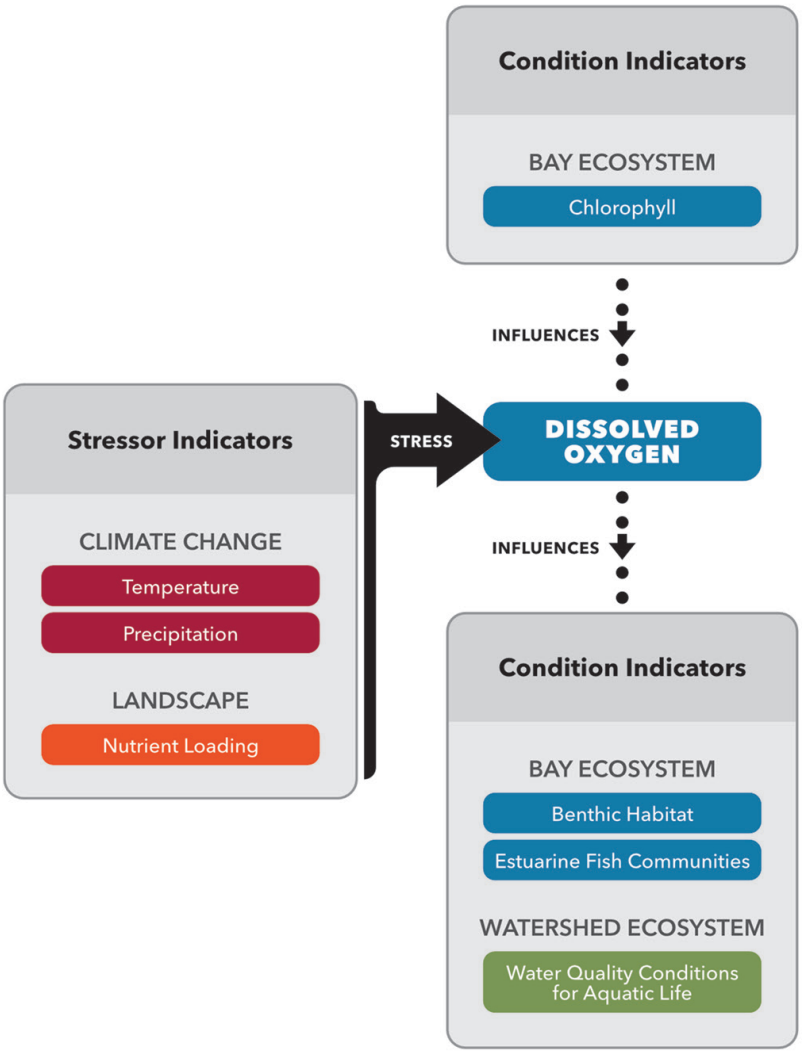
CHAPTER 15: DISSOLVED OXYGEN

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NARRAGANSETT BAY ESTUARY PROGRAM

Photo: Bay Water Quality Monitoring by the Narragansett Bay Commission (Ayla Fox)

Overview



BACKGROUND

- Dissolved oxygen is essential for the survival of many types of marine life, and thus it serves as an important indicator of ecosystem condition. Dissolved oxygen concentrations are influenced directly by oxygen generation (photosynthesis) and consumption by living organisms (in particular, bacteria). In addition, dissolved oxygen is influenced by many other factors such as water temperature, exchange with the atmosphere as driven by winds, and exchange with nearby waters as governed by currents and mixing (processes shaped by precipitation-driven riverine inputs that promote distinct water layers, and by tidal conditions). Excessive nutrient loading can lead to depletion of dissolved oxygen by stimulating the growth of large amounts of phytoplankton (indicated by chlorophyll concentrations) and macroalgae. As the phytoplankton and macroalgae decay, bacterial

consumption can deplete dissolved oxygen, stressing organisms, especially those that are unable to move out of the oxygen-depleted area. Both Rhode Island and Massachusetts assess estuarine waters for impacts by nutrient enrichment and/or oxygen depletion to protect aquatic life.

KEY FINDINGS

- ***Status:*** In 2013 to 2015, the lowest bottom water oxygen concentrations were recorded in the Providence and Seekonk Rivers, as well as in Greenwich Bay, with concentrations elsewhere generally increasing throughout the bay southward and eastward. Bay-wide, hypoxia was variable year to year, with low oxygen concentrations in 2013 when summer precipitation and river flow were higher than typical, and relatively high oxygen concentrations in 2014 and 2015 which were drier than typical. Observed hypoxia characteristics from 2013 to 2015 were typical of conditions seen since focused measurement programs began, in the early 2000s.
- ***Trends:*** Data from 2005 to 2015 revealed high variability from year to year, linked to spring/summer precipitation and river flow. Relative to these interannual changes, long-term trends were not sufficiently pronounced to be identified well by the measurements.

Introduction

Dissolved oxygen (DO) in the water column is critical to sustain water quality and marine life in Narragansett Bay. The concentration of dissolved oxygen depends on a complex mix of biological, chemical, and physical processes. Dissolved oxygen sources are mostly in the upper water column, whereas sinks are throughout the water column and concentrated below the pycnocline (where water density increases rapidly with depth). Sources of dissolved oxygen include mixing with the atmosphere, photosynthesis, and exchange with more-oxygenated waters through circulation and mixing. Sinks of dissolved oxygen include respiration within the water column and sediments, and exchange with less oxygenated waters through circulation and mixing.

Hypoxia is defined as low levels of dissolved oxygen concentrations that stress or kill marine organisms (USEPA 2000, Miller et al. 2002, Saarman et al. 2008). Hypoxia occurs when oxygen levels are depleted and not readily replenished. According to the United States Environmental Protection Agency (USEPA 2000), unacceptable chronic impacts to aquatic life occur at dissolved oxygen concentrations below 4.8 mg/l, while unacceptable acute impacts to adult and juvenile species occur at levels below 2.3 mg/l for at least 24 hours. Between these two values the intensity and duration of hypoxia must be examined to determine habitat suitability. Because larvae are more sensitive to dissolved oxygen than juveniles or adults, the Rhode Island Department of Environmental Management (RIDEM) adopted criteria calculated from USEPA (2000) guidance to protect larvae from a 24-hour exposure of less than 2.9 mg/l in bottom waters during the recruitment season (RIDEM 2010). Both Massachusetts and Rhode Island assess estuarine waters for impacts from nutrient enrichment and depleted oxygen concentrations. Of the state-assessed estuarine waters, approximately 50 percent were identified as impacted by nutrient enrichment and/or depleted oxygen levels (see "Water Quality Conditions for Aquatic Life" chapter). This translates to 37 percent of all estuarine waters in the Rhode Island and Massachusetts portions of Narragansett Bay. These areas are located in the northern sections of the Bay (the Upper West Passage, Greenwich Bay, Upper Narragansett Bay, Providence River Estuary, Mount Hope Bay, and the Taunton River).

Hypoxia is influenced by many factors, including excessive nutrient loading, respiration/decomposition, and stratification. Excessive nutrient loading causes phytoplankton or macroalgae to bloom, which, when the phytoplankton or macroalgae consume oxygen (respiration) and when they die and sink into deep waters and are decomposed,

depletes the water of oxygen. If sub-pycnocline/bottom water cannot mix with surface water, or gets trapped in silled or small embayments due to lack of circulation or mixing, consumption of oxygen can exceed oxygen introduced to the area (from the atmosphere, photosynthesis, or circulation) and the waters can become oxygen depleted. As respiration and decomposition continue, oxygen levels are driven down (Bergondo et al. 2005, Deacutis et al. 2006, Saarman et al. 2008, Codiga et al. 2009). During the summer months, warm waters support high productivity and respiration rates. In addition, the Bay is often density stratified, meaning there are distinct layers with relatively warm and lower salinity surface water overlying colder and saltier deep water (Codiga 2012). Wet weather tends to increase freshwater discharge to an estuary, which increases nutrient loading and further enhances stratification due to the density differences of water, leading to decreased mixing. This results in increased likelihood of hypoxic events following rainstorms, as stratification can isolate the bottom waters from sources of oxygen near the surface. Biochemical reactions and respiration in both the water column and the sediments remove oxygen from the waters. This oxygen demand, coupled with warm waters and density stratification, increases the risk of hypoxic conditions in the summer months. The warming climate may also lead to higher hypoxia risk in coastal regions, as warmer water holds less oxygen and contributes to increases in respiration rates (USEPA 2016).

Hypoxic conditions have been widely studied in estuaries that experience eutrophication, with notable examples including the Gulf of Mexico, Chesapeake Bay, and Long Island Sound (Diaz 2001, Bricker et al. 2007, USEPA 2012). Coastal hypoxia is a global issue, and it may result in changes to the food web that have lasting socioeconomic and environmental effects (Bricker et al. 2007, USEPA 2012, Rabalais et al. 2014). Benthic habitats can be affected (see "Benthic Habitat" chapter), and migration by mobile organisms out of affected areas commonly occurs due to hypoxia. Many of these changes can also be linked to changes in nutrient inputs (Rabalais et al. 2014), which may alter the phytoplankton community and shift trophic interactions (Turner et al. 1998).

By the mid to late 1980s, researchers established that bottom waters in the Providence-Seekonk River Estuary and the upper portions of Narragansett Bay experienced periods of low oxygen during the summer months (e.g., Pilson and Hunt 1989, Doering et al. 1990). The low dissolved oxygen contributed to changes in the benthic community (Germano and Rhoads 1989; see "Benthic Habitat" chapter) and resulted in periodic fish kills. During the 1990s, managers and researchers strived to implement management actions to mitigate hypoxic events, such as reducing nitrogen loading (specifically ammonium, and later, total nitrogen) from wastewater treatment facilities (see "Wastewater Infrastructure" and "Nutrient Loading" chapters). In 2003, a large fish kill, caused by severe hypoxia, occurred in Greenwich Bay, prompting public outcry and accelerated plans to reduce point source nutrient inputs to Narragansett Bay (RIDEM 2003). Following the fish kill, managers also sought to expand the Narragansett Bay Fixed Site Monitoring Network to include thirteen sites measuring dissolved oxygen and other properties.

This chapter analyzes bottom water dissolved oxygen concentrations in Narragansett Bay using two separate methods–a Hypoxia Index and Spatial Surveys–to assess the duration, intensity, and spatial extent of hypoxic events in the Bay. The analysis using the Hypoxia Index focuses on the growing season (May to October) from 2001 to 2015, while the analysis using the Spatial Surveys targets the summer season (June through September) from 2005 to 2015.

Methods

To analyze dissolved oxygen concentrations and hypoxia, the Narragansett Bay Estuary Program employed two methods:

- ***Hypoxia Index:*** The Hypoxia Index, using data from the Narragansett Bay Fixed Site Monitoring Network (NBFSMN), combines measures of the amount or magnitude that bottom water dissolved oxygen concentrations fell below a fixed threshold, and how long they stayed below the threshold. The Hypoxia Index has been used to identify the areas of the Bay that experience the most severe hypoxia (Codiga 2008, Codiga et al. 2009). It provides a tool to consistently characterize hypoxia across the bi-state estuarine waters of the Bay.
- ***Spatial Surveys:*** The Estuary Program analyzed the spatial distribution of bottom water dissolved oxygen concentration using data collected by the Day Trippers, which is a collaborative sampling effort involving several organizations (Prell et al. 2015, 2016).

These efforts have two separate methodologies (detailed below) and generate results which may not be consistent with each other. The Status and Trends section of this chapter is divided into two parts to separately report the results for each method. Status was based on the most recent data – 2013 to 2015

for the Hypoxia Index and 2015 for the Spatial Survey. Trends were determined using the entire dataset for both metrics. Additional methods are available upon request.

HYPOXIA INDEX

The Hypoxia Index utilizes summer bottom water dissolved oxygen data from the Narragansett Bay Fixed Site Monitoring Network (NBFSMN) to quantify spatial and temporal changes in dissolved oxygen. Sensors located 0.5 to 1 meter (1.6 to 3.3 feet) above the seafloor collected the dissolved oxygen records for bottom waters every 15 minutes in units of mg/l (or mg l-1) (NBFSMN 2016). Data from ten of the fixed stations were analyzed individually for 2001 to 2015 (Table 1; Figure 1) using the seasonal period May 15 through October 14, building on work done previously (Codiga 2008, Codiga et al. 2009). May through October is the WWTFs subject to advanced nitrogen removal are required to do so (see "Nutrient Loading" chapter). From 2001 to 2005, a relatively small number of sites were monitored, mostly in the Providence River Estuary and Upper Bay. From 2006 to 2015, most or all of the ten sites were sampled (Table 2).

To identify and quantify individual hypoxic events, the Hypoxia Index applies an algorithm, known as a moving window trigger, to each time series (Codiga 2008, Codiga et al. 2009). The start (or end) of an event is identified when the concentration falls below (or rises above) the 2.9 mg/l threshold for longer than a "trigger duration" to avoid mistakenly identifying short-duration excursions below (or above) the threshold as starting (or ending) an event. Due to tidal processes, such short-duration excursions are common, lasting less than about six hours or one-half the dominant tidal period (M2, lunar semidiurnal), so the algorithm used a trigger duration of nine hours (Codiga 2008, Codiga et al. 2009). An individual event is identified when a start and end are detected with sub-threshold concentration for a duration of at least 24 hours between them. The algorithm determines the event deficit-duration, which is the area below the threshold in the time series of observed concentrations (Figure 2). The deficit-duration is a reflection of both the intensity of the event (amount or magnitude by which concentration falls below the threshold during the event) and its duration (time that the concentration is lower than the threshold).

Deficit-duration is measured in units of mg/l day (mg l-1 day). As an example of the meaning of the deficit-duration units, consider that if an individual hypoxic event has an oxygen concentration that falls below the threshold by an amount of two mg/l, and remains there for a duration of three days, it has a deficit-duration of six mg/l day.

To characterize hypoxic conditions, three fixed oxygen thresholds were used, as motivated by RIDEM water quality standards: 4.8 mg/l, 2.9 mg/l, and 1.4 mg/l; however, only the 2.9 mg/l threshold results are presented in this chapter to focus on acute hypoxia (see Codiga et al. 2009 and Codiga 2016 for the complete analysis). The Hypoxia Index differs from the regulatory metric used by RIDEM to evaluate the different allowable exposure periods associated with each DO threshold/criteria (SAIC 2006). The Hypoxia Index events can start or end at any time during the day, while the RIDEM metric

Table 1. Acronyms for each site name and site group (shaded) and water depth relative to mean low water. Sensors are located 0.5 to 1 meter (1.6 to 3.3 feet) above the bottom.

Site Group	Site	Acronym	Water Depth (m)
PRUB	Bullock's Reach	BR	6
PRUB	Conimicut Point	CP	7
PRUB	North Prudence	NP	11
UWP	Mount View	MV	7
UWP	Quonset Point	QP	7
UEP	Poppasquash Point	PP	8
UEP	T-Wharf	TW	6
GRBY	Greenwich Bay	GB	3
GRBY	Sally Rock	SR	4
MH	Mount Hope Bay	MH	5

determines exceedances based on daily, midnight-to-midnight periods and incorporates a nonlinear biological response function. Detailed comparison of results from the Hypoxia Index and RIDEM metrics, applied to the same measurement records, revealed that they lead to qualitatively similar findings (Codiga et al. 2009). However, it should be emphasized that the Hypoxia Index is not a regulatory metric and, while useful to characterize hypoxic conditions, should not be used to assess compliance of Narragansett Bay waters with dissolved oxygen criteria. Massachusetts does not have an equivalent regulatory metric to RIDEM.

The Estuary Program computed multi-site average annual Hypoxia Index results, using the fixed sites grouped by geographic location throughout Narragansett Bay (Table 1; Figure 1):

- Providence River and Upper Bay group (PRUB)
- Upper West Passage group (UWP)
- Upper East Passage group (UEP)
- Greenwich Bay group (GRBY)
- Mount Hope group (MH; one site).

The Hypoxia Index, referred to as the cumulative deficit, is determined for each individual site and each year as the sum of all deficit-durations for all individual hypoxic events (2.9 mg/l threshold) that occur there during the mid-May to mid-October period (Figure 2). Periods of missing observations (gaps), when field sensors were not operating or data did not meet quality standards, typically comprise about 10 to 40 percent of the mid-May to mid-October period in a given year at a given site (Table 2). Due to these gaps, the moving window trigger algorithm gives a lower bound for the seasonal Hypoxia Index. The gap contribution, and thus an upper bound for the Hypoxia Index, is estimated (Codiga 2016) using the time period(s) of the gap(s), together with event statistics (specifically, the probability and severity of events, for the time of year of the gap and the particular site) from all other years at the site. The seasonal Hypoxia Index is the mean of the upper and lower bounds and its uncertainty due to gaps is half the difference between the upper and lower bound. The multi-site Hypoxia Index is computed for groups of sites as the mean of the Hypoxia Index results of individual sites in the group; they are presented with uncertainties that are the mean, of the sites in the group, of the gap-based uncertainties for individual sites.

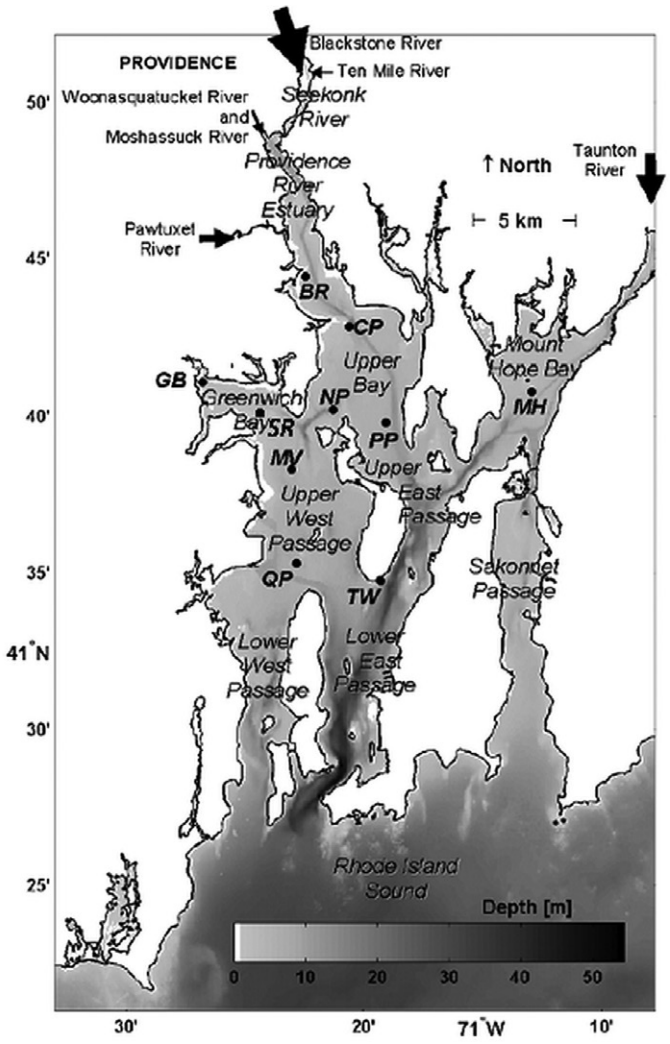


Figure 1. Site locations for the Hypoxia Index. Arrows indicate river discharge to the Bay, with size of the arrow representing relative discharge volume (from Codiga 2016).

SPATIAL SURVEYS

The Spatial Surveys used in this analysis were initiated in 2005 using Sea-Bird 19 Plus SEACAT profilers. Three boat groups sampled 77 sites covering 150 square kilometers (58 square miles) in the Providence River Estuary, Greenwich Bay, and the East and West Passages of Narragansett Bay (Figure 3). The surveys focused on the warm summer months in the morning hours irrespective of tidal phase. Tidal excursions in Narragansett Bay are small compared to the scale of the surveys, so no attempts were made to correct for tidal phase. The data were calibrated and organized to produce an internally consistent and documented, interpolated (0.5 meter) dataset of temperature,

Table 2. Percent of time during mid-May to mid-October analysis period with valid near-bottom dissolved oxygen values at each of the ten sites in each of the fifteen years. Asterisks (*) indicate records with less than 55 percent valid values, which were not included in the analysis. Blanks indicate sites that were not sampled in a given year.

	BR	CP	NP	MV	QP	PP	TW	GB	SR	MH
2001	100		62							
2002	75		81							
2003	79	65	73				80	63		
2004	86		82	65		64	88	99		
2005	89	76	91	66	*	*	90	83		60
2006	79	79	72	94	78	98	89	98		98
2007	66	94	78	93	55	93	92	93		93
2008	96	96	89	93	97	92	86	88	96	83
2009	100	100	80	83	80	87	88	96	81	87
2010	85	77	78	82	88	82	97	96	83	84
2011	85	90	90	91	85	83	72	100	91	73
2012	98	87	96	96	74	96	100	99	94	96
2013	99	91	98	80	90	92	98	92	93	100
2014	94	96	93	92	96	85	100	100	96	96
2015	77	81	86	91	82	91	100	100	79	86

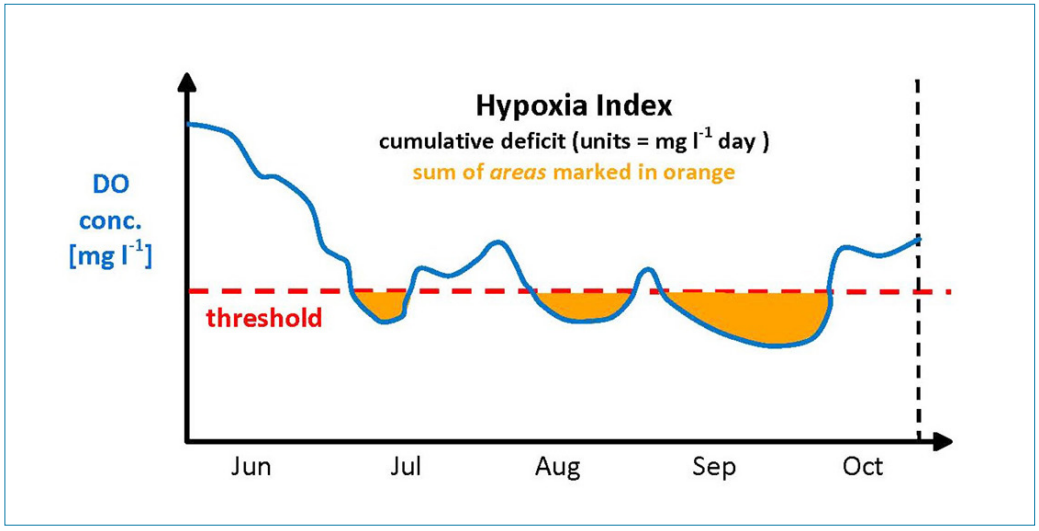


Figure 2. Schematic showing computation of the Hypoxia Index, relative to a given threshold (red dashed horizontal line), from the time series of near-bottom dissolved oxygen measurements (blue line) at a single mooring site. The index is the total area beneath the threshold level. It is the season-cumulative deficit-duration of all individual hypoxic events (this schematic shows three events) during the mid-May to mid-October analysis period.

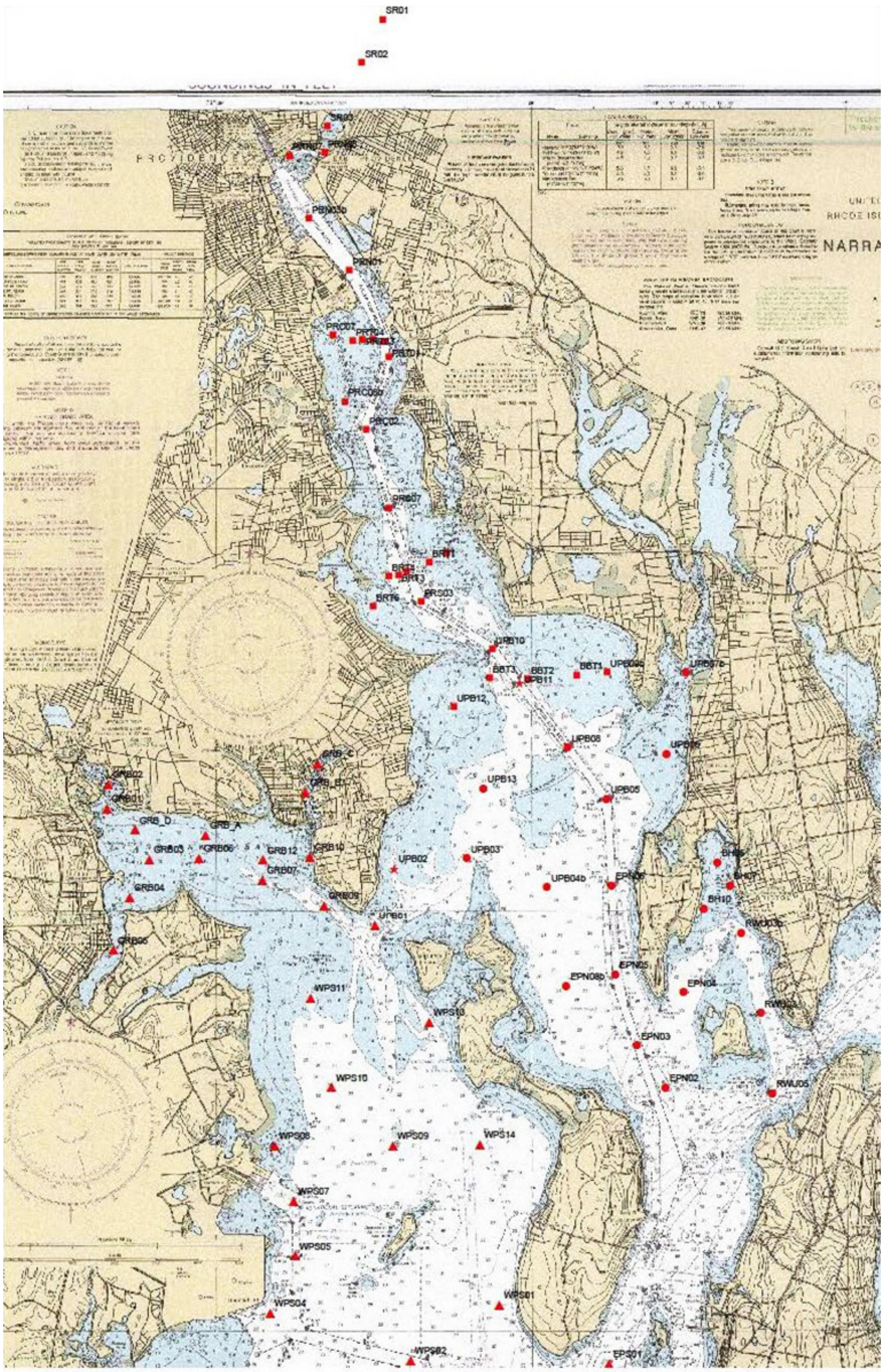


Figure 3. Locations of spatial survey sites (from Prell et al. 2015).

salinity, dissolved oxygen, density, and chlorophyll (see "Chlorophyll" chapter). The dataset has 58 surveys at 77 sites over eleven years (2005 to 2015) and contains 4,154 individual profiles. Results were interpolated using GIS to determine percentage of the survey area that had bottom water dissolved oxygen concentrations below the thresholds (Prell et al. 2015, 2016).

Determination of Wet and Dry Years

Interannual variations in oxygen concentrations are correlated with freshwater runoff into Narragansett Bay more strongly than other factors (Codiga et al. 2009). Accordingly, the Estuary Program examined differences in dissolved oxygen concentrations and the Hypoxia Index between dry years (low river flow and low precipitation) and wet years (high river flow and high precipitation) to assess the impact of freshwater flux and its associated nutrient flux. The impact of freshwater runoff was assessed using the daily discharge from the Blackstone, Pawtuxet, and Taunton Rivers (as measured by United States Geological Survey gages) for 2005 to 2015 along with the long-term data of Ries (1990) to calculate the total freshwater flux to the Bay during the summer (June, July, August, September; abbreviated JJAS). The long-term median from 2005 to 2015 was calculated and compared with median flow for individual years. When individual medians were greater than the dataset median (2000 to 2015) they were considered wet years. Dry years were those with individual medians less than the dataset median (Table 3; Prell et al. 2016).

Status and Trends

HYPOXIA INDEX

The Estuary Program reports the status of hypoxia in Narragansett Bay based on conditions in the most recent years (2013 to 2015) using Hypoxia Index results from individual sites (Figure 4) and multi-site average Hypoxia Index results (Figure 5). Trends are analyzed using the entire dataset from 2001 to 2015.

The individual site results from 2013 to 2015 reveal two dominant aspects that are generally applicable to other years. The first aspect is pronounced interannual variability. The Hypoxia Index was higher in 2013 than in 2014 or 2015 at all sites, and substantially so at most sites (Figure 4) indicating lower overall dissolved oxygen concentrations in 2013 than in 2014 or 2015. This interannual variability is notably far larger than uncertainties in the Index values due to sampling gaps. The higher 2013 Index values over those of 2014 and 2015 were generally similar at all sites, indicating they result from a process with Bay-wide influence. Precipitation is a major factor responsible for this variability, as hypoxia was more severe in 2013, which had wet spring and summer conditions, compared to the drier years of 2014 and 2015 (Table 3).

The second dominant characteristic was a down-Bay gradient, in which the Hypoxia Index generally decreased from north to south (Figures 4 and 5). This was a persistent feature, with very few exceptions, in any given year and was more pronounced in wet

Table 3. River flux departure during June, July, August, and September. Dry years occur when the annual summer departure is less than the long-term median; wet years occur when the departure is greater than the long-term median. Wet years are shaded, and "wet" is bold text.

YEAR	River Flux Departure m^3/s JJAS	Wet/Dry Designation
2005	-10	DRY
2006	26	**WET**
2007	-13	DRY
2008	-4	DRY
2009	44	**WET**
2010	-17	DRY
2011	25	**WET**
2012	-5	DRY
2013	23	**WET**
2014	-17	DRY
2015	-8	DRY

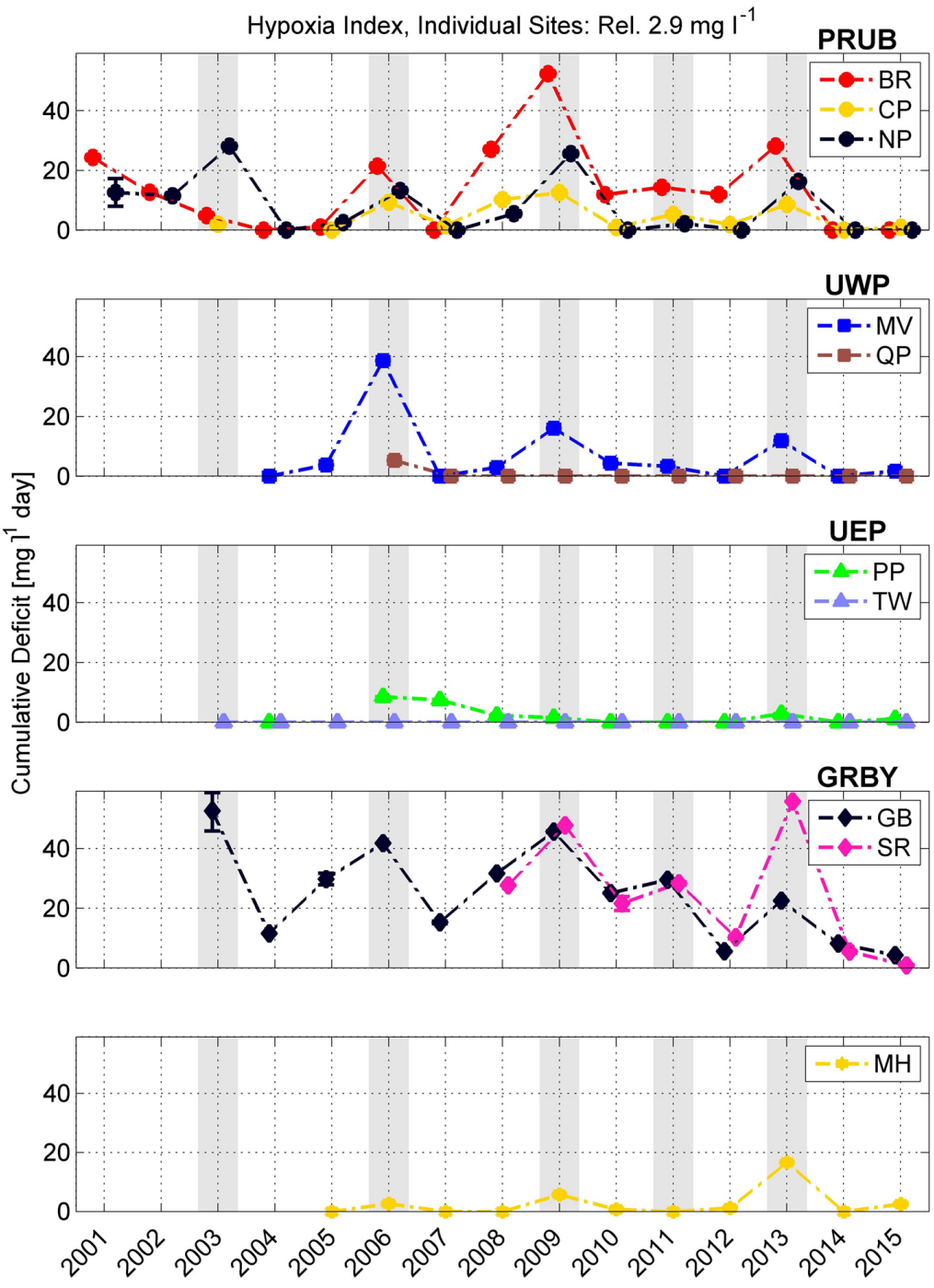


Figure 4. Hypoxia Index for individual sites within each site group, relative to the 2.9 mg/l threshold. Years with higher than typical river runoff during June to September (wet years; Table 3) are marked by gray vertical bars. To improve clarity by reducing overlap of lines and symbols, symbols from each site are systematically offset a small distance horizontally relative to those of other sites in the frame. In most cases the symbols are larger than the error bars, which indicate uncertainties due to gaps in the time series because of sensor malfunction or data not meeting quality standards.

years. The Hypoxia Index decreased from north to south from the Providence River (PRUB) down through the Upper West Passage (UWP) (Figures 4 and 5). Bottom water dissolved oxygen increased with distance from Providence.

Using 2.9 mg/l as a dissolved oxygen threshold, the cumulative deficit results for the Hypoxia Index were typically less than 30 mg/l day with a maximum of about 60 mg/l day (Figure 4). During the drier years of 2014 and 2015, the Index was effectively zero everywhere except in Greenwich Bay (GRBY). During the wet year of 2013, the Index included elevated results for GRBY, PRUB, Mount Hope Bay (MH), and the Mount View site (MV) in UWP, while the Index at other sites in the Bay was zero or very near to zero (oxygen concentrations remained above the 2.9 mg/l threshold).

To detect longer-term trends, the Estuary Program analyzed the results from all years (2001 to 2015), using the multi-site mean Hypoxia Index values for the 2.9 mg/l threshold. However, trends extending over multiple years were difficult to discern due to the relatively large interannual variability. This fundamental characteristic of the dataset sharply limited the confidence that can be placed in conclusions regarding trends (Figure 5).

This trend assessment confirmed the pattern that higher Hypoxia Index values occurred in wetter conditions (Figures 4 and 5). The years with higher than typical runoff (2003, 2006, 2009, 2011, and 2013) appeared as local maxima in the Hypoxia Index, generally at all sites. However, there was no strong evidence for a long-term temporal trend over the fifteen-year period, due to the interannual variability. This was true whether examining all years together, wet years alone, or dry years alone.

SPATIAL SURVEYS

The Spatial Surveys data for the summers of 2005 to 2015 were used as a point of comparison to the Hypoxia Index. The summer of 2015 was characterized by relatively low freshwater flux, high salinity, and a low percent area of hypoxia (Tables 3 and 4; Figures 6c and 6d). To illustrate the spatial extent of hypoxic waters, maps of bottom water and minimum dissolved oxygen on August 12 of 2015, a dry year with a minimal area of hypoxia waters, were compared to August 4 of 2009, a wet year with a large area of hypoxic waters (Figure 6). In general, the spatial pattern remained similar throughout the surveys: hypoxic waters typically occurred in the Seekonk and Providence Rivers and Greenwich

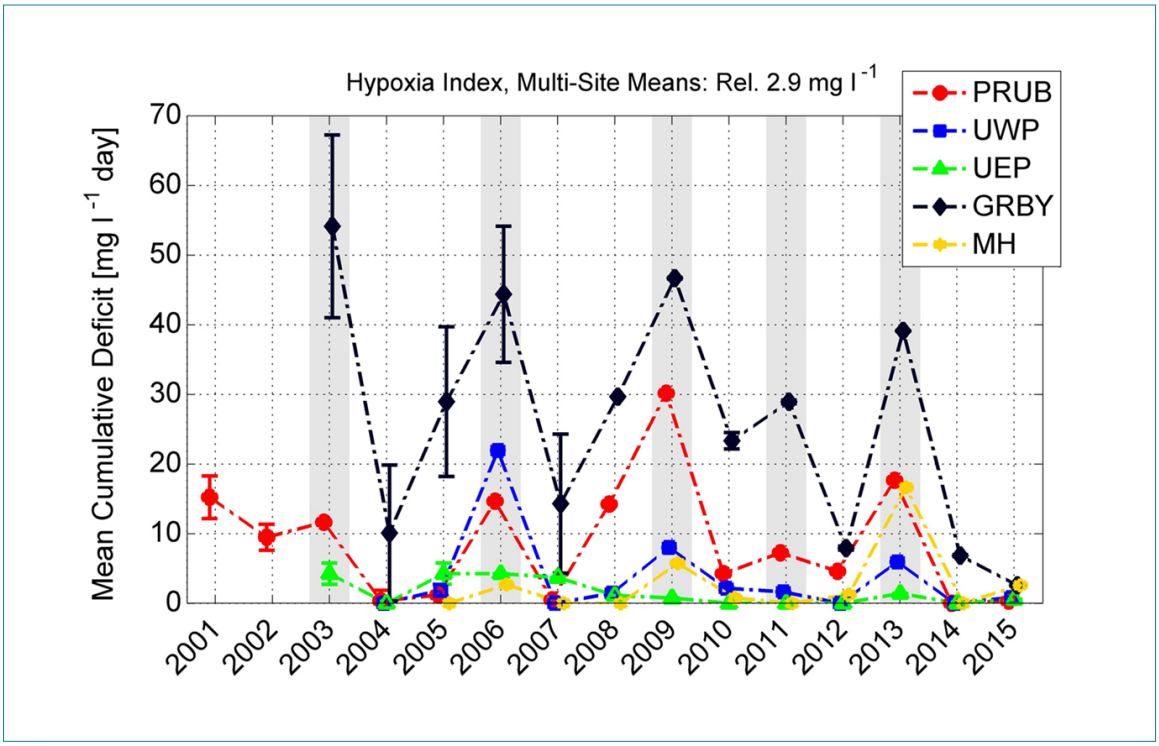


Figure 5. Hypoxia Index averaged over all sites within each site group, relative to threshold 24-hour 2.9 mg/l day. Error bars indicate the mean uncertainty, among the stations in the group, due to gaps in the time series.

Bay, expanding to the Upper and Middle West Passage during more extreme wet-weather events. These maps are useful for showing spatial trends in dissolved oxygen concentrations, but it should be noted that these maps represent a single-day survey event and do not depict the duration of the hypoxic event; duration is an important component for defining a hypoxic event.

The lowest dissolved oxygen values typically occurred at the bottom, but during intervals of high river flux and stratification the minimum levels of dissolved oxygen were often measured near the pycnocline, which was typically at a depth of four to six meters (thirteen to 20 feet). This pattern was especially evident in the dredged shipping channel in the Providence River and the eastern Upper Bay (Figures 6a and 6b). During intervals of infrequent hypoxia, the bottom and minimum maps were virtually identical as no mid-water minimum developed (Figures 6c and 6d). On the maps, areas of higher dissolved oxygen surrounded by lower dissolved oxygen typically indicate shallow sites that were within the mixed layer and thus more highly oxygenated. This pattern was evident on the margins of the Providence River Estuary north of Conimicut Point, where a number of shallow sites did not change from bottom to minimum maps (Figures 6a and 6b). During intervals of high hypoxia, the low dissolved oxygen extended over the Upper Bay and down the West Passage to Quonset Point. The East Passage, south of Poppasquash Point, remained largely oxygenated (Figures 6a and 6b).

Table 4. Summary of the average bottom dissolved oxygen and the average minimum dissolved oxygen for 2015 compared to the long-term (2005–2015) averages for different areas of the Bay. All units are mg/l.

Year	Seekonk and Providence Rivers		Upper Bay		Greenwich Bay		Lower Bay	
	Average	Min.	Average	Min.	Average	Min.	Average	Min.
2015	3.6	2.1	4.5	3.0	4.0	2.5	5.0	3.5
Long-term Average (2005–2015)	3.4	0.9	4.1	1.3	3.9	0.5	4.8	2.5

Table 5. Summary statistics for all summer (June, July, August, September: JJAS) surveys during each survey year. Wet years, based on river flux (m^3/s; see Methods), are shaded. DO is dissolved oxygen.

Year	River Flux Departure m^3/s JJAS	Average Surface Salinity $^0/_{00}$	Average Surface Temp. ^0C	% Area with Bottom DO < 2.9 mg/l JJAS	Average % DO Surface Saturation
2005	-10	28.0	23.2	5.3	101
2006	26	23.7	22.0	27.1	96
2007	-13	27.7	21.1	2.9	91
2008	-4	26.5	23.1	17.6	91
2009	44	24.3	21.6	29.8	101
2010	-17	27.6	24.2	17.4	105
2011	25	25.3	23.6	13.7	107
2012	-5	27.6	23.7	12.2	97
2013	23	23.9	23.0	18.0	107
2014	-17	27.9	23.6	3.7	101
2015	-8	28.4	23.4	5.6	88

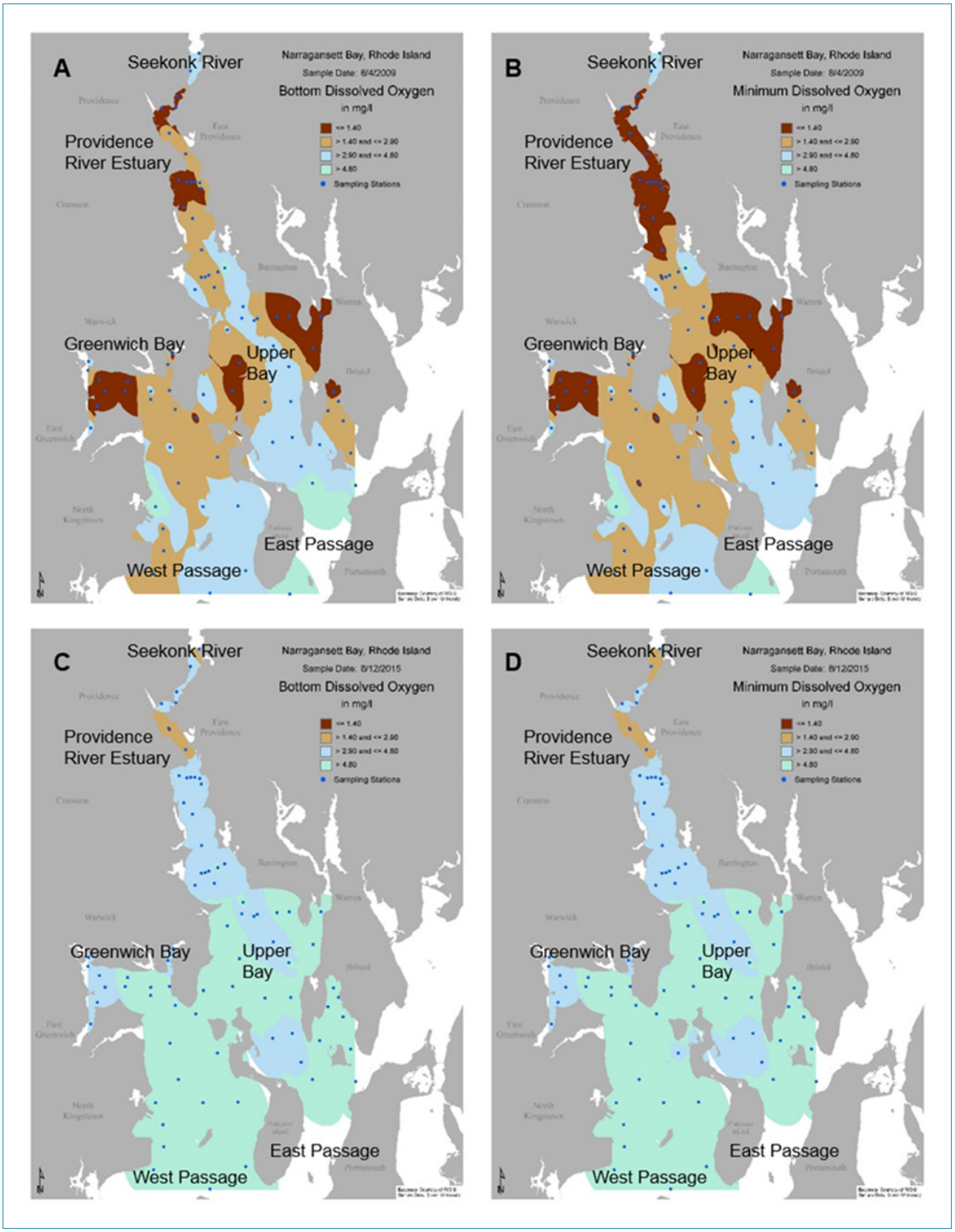


Figure 6. The spatial extent of bottom water dissolved oxygen (left column: 6a, 6c) and minimum dissolved oxygen (right column: 6b, 6d), comparing a highly hypoxic survey (top row: 6a, 6b; August 4, 2009) and a minimally hypoxic survey (bottom row: 6c, 6d; August 12, 2015). Dissolved oxygen categories correspond to the three thresholds (4.8, 2.9, and 1.4 mg/l) used by RIDEM to identify chronic and acute hypoxia.

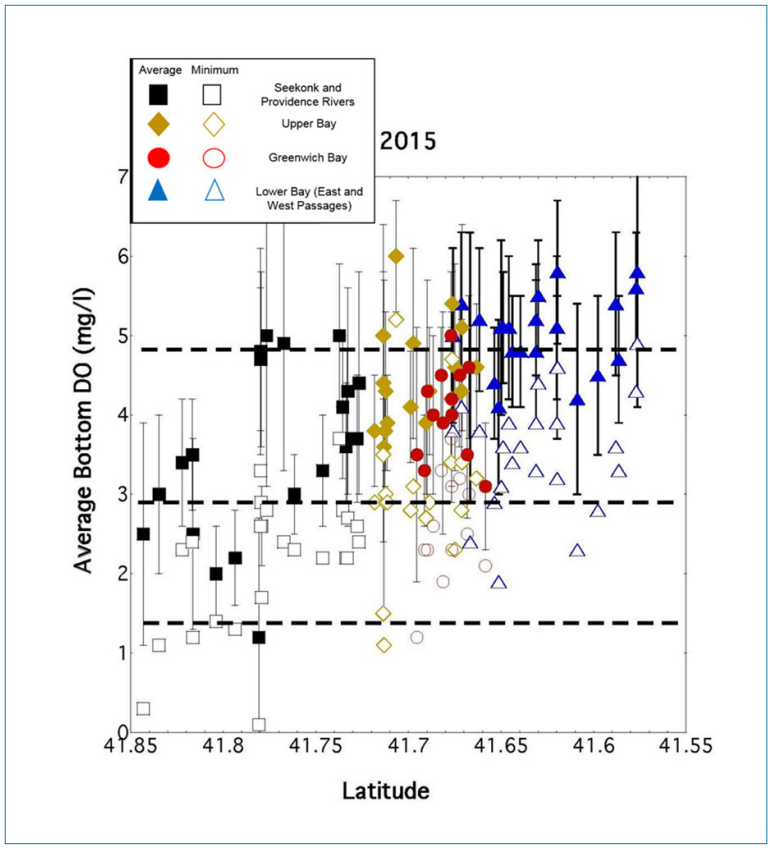


Figure 7. The distribution of average (solid symbols) and minimum (open symbols) dissolved oxygen concentrations in bottom water for sites in each region of Narragansett Bay in 2015 surveys. Sites are shown in the graph from north (left) to south (right). The irregular north-to-south gradient in dissolved oxygen is attributable to differences between East and West Passages, shallow sites, and low dissolved oxygen in coves and embayments at different latitudes. Dashed lines indicate the three thresholds (4.8, 2.9, and 1.4 mg/l) used by EPA and RIDEM to identify chronic and acute hypoxia for waters below a pycnocline. However, EPA and RIDEM have established different allowable exposure periods associated with each DO threshold, while the spatial surveys results are instantaneous measurements. Bars indicate standard deviation for mean dissolved oxygen.

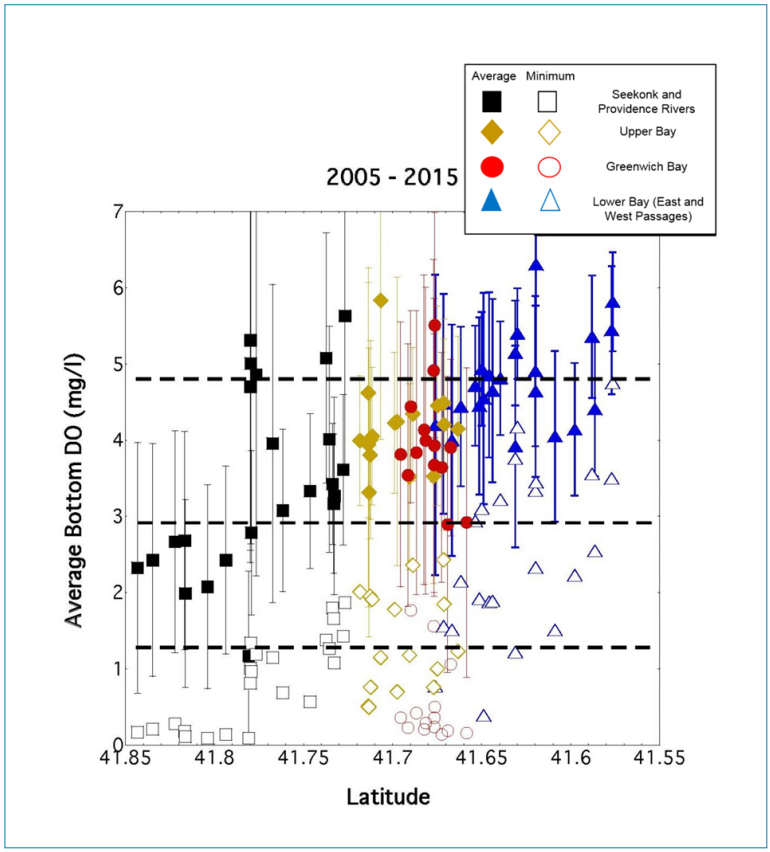


Figure 8. The distribution of average (solid symbols) and minimum (open symbols) dissolved oxygen concentrations in bottom water for sites in each region of Narragansett Bay averaged over all surveys (2005 to 2015). Shown as in Figure 7.

The average bottom dissolved oxygen for different parts of the Bay in 2015 was slightly higher (about 0.2 mg/l) than the 2005 to 2015 average (Table 4; Figures 7 and 8), and the percent area covered by hypoxic bottom water (5.6 percent) during the summer (July, August, September: JAS) was among the lowest observed during the past decade (Table 5; Figure 9).

The minimum bottom water dissolved oxygen observed at each site and the mean bottom water dissolved oxygen values for each site and its standard deviation over all surveys revealed bottom conditions throughout the Bay (Figure 8). Of the 77 sites, eleven sites (14 percent) had mean bottom water dissolved oxygen of less than 2.9 mg/l, and 67 sites (87 percent) had a minimum observed dissolved oxygen of less than 2.9 mg/l.

Although the average dissolved oxygen values (Figures 7 and 8) show large-scale spatial gradients in the Bay, they do not capture the temporal variability. To document these patterns, the Estuary Program mapped the dissolved oxygen in bottom waters and calculated the area of each dissolved oxygen class (Figure 9). The percent area of hypoxic bottom water (less than 2.9 mg/l) during the summer (JAS) exhibited significant intra- and interannual

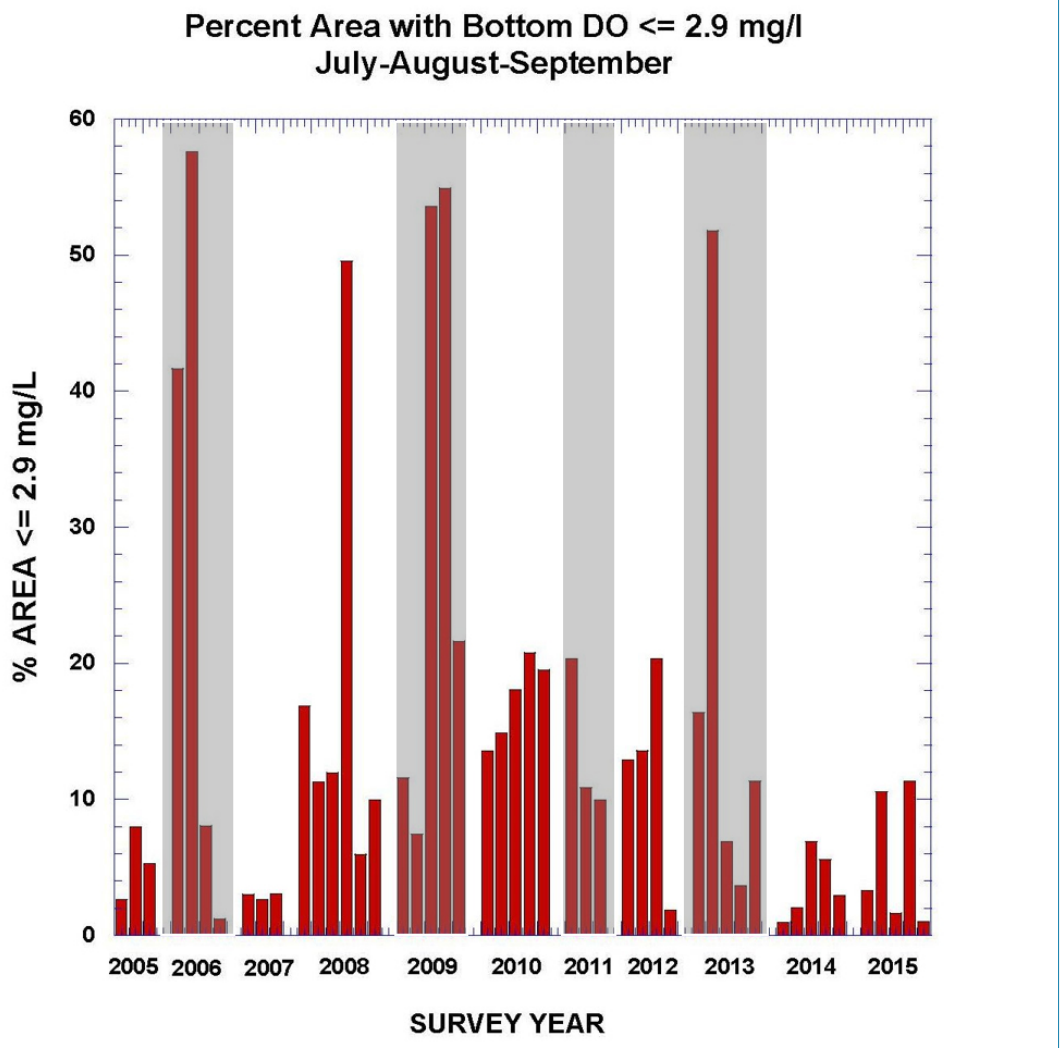


Figure 9. Percent of the survey area (150 square kilometers; 58 square miles) considered hypoxic (dissolved oxygen less than 2.9 mg/l) from 2005 to 2015. Each bar represents a survey. Only July, August, and September (JAS) surveys are shown.

variability. Typically, less impairment occurred in June (not shown), which tended to have lower water temperatures. The area of hypoxic bottom waters ranged from only a few percent to over 40 percent in sporadic extremes in 2006, 2008, 2009, and 2013, all wet years except for 2008.

Discussion

COMPARISON OF METHODS

The Hypoxia Index and the Spatial Surveys are different but complementary methods, using different data and different temporal scales. The Hypoxia Index uses an array of temporal data collected every fifteen minutes from May to October, but it does not have high spatial resolution, given that there are just ten fixed sites used by the Index. Conversely, the Spatial Surveys have more spatial resolution (both horizontally and vertically) with 77 sites, but it focuses on only five or six survey days each summer with a single dissolved oxygen profile measure at each station, providing low temporal resolution. The Estuary Program conducted an initial analysis comparing these methods in the Providence River Estuary. The Hypoxia Index results for two sites (Bullocks Reach, BR; Conimicut Point, CP) had a strong correlation to the spatial extent of hypoxia in the entire survey area covered by the Spatial Surveys (Figure 10). These results suggest that bottom water DO concentrations at the BR station is a good predictor of the extent of hypoxia in the entire Upper Bay. There is a need to expand this analysis and complete correlations for other sites in the Bay to assess whether similar correlations exist for stations that experience less frequent hypoxia, and to assess the strength of correlations in a manner that captures interannual variability.

The Hypoxia Index and the Spatial Surveys reveal areas where low levels of dissolved oxygen in bottom water tends to be a problem, including the Providence-Seekonk River Estuary, Upper Bay, and Greenwich Bay, with sporadic events in the Upper West Passage, and Mount Hope Bay fixed sites (Deacutis et al. 2006, Melrose et al. 2007, Codiga et al. 2009, Prell et al. 2016). These data also show that the Providence River Estuary periodically suffers from hypoxic episodes where dissolved oxygen concentrations fell below the 2.9 mg/l threshold, although the minimum dissolved oxygen concentration was not always at the bottom of the water column (Deacutis et al. 2006, Melrose et al. 2007). In the Upper Bay, the minimum dissolved oxygen value was often observed near the pycnocline (Deacutis et al. 2006, Melrose et al. 2007, Prell et al. 2016). The most current year (2015) shows increased bottom water dissolved oxygen over the long-term averages for the Spatial Surveys (Table 4), and the Hypoxia Index was similar to other dry years (e.g. 2004, 2007, and 2012) (Figure 5).

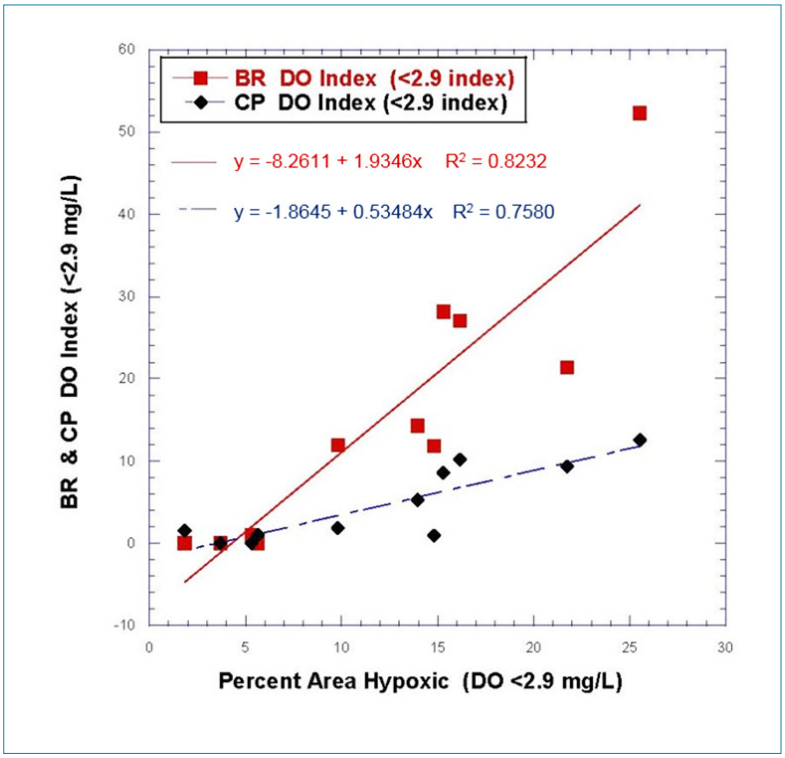


Figure 10. Correlation of Hypoxia Index (DO Index) for Bullock's Reach (BR; red squares and lines) and Conimicut Point (CP; black diamonds and blue line) with spatial survey (percent area hypoxic) at the 2.9 mg/l threshold for 2005 to 2015.

The analyses presented in this chapter confirmed the north-to-south, down-Bay gradient in pollution established in prior research (Pilson and Hunt 1989, Doering et al. 1990, Bergondo et al. 2005, Deacutis et al. 2006, Melrose et al. 2007). With the exception of certain embayments (e.g., Greenwich Bay and Mount Hope Bay), there is a down-Bay increase in dissolved oxygen concentration or decrease in the Hypoxic Index starting in the Providence River Estuary and the Seekonk River (Figures 4, 5, and 6). Similarly, the Spatial Surveys showed a down-Bay gradient in which the long-term mean bottom water dissolved oxygen concentration ranged from 1.2 mg/l in the Providence River Estuary to 6.3 mg/l in the shallow West Passage (Figure 8). The decrease of bottom water hypoxia with distance down the Bay follows the gradient of anthropogenic inputs to Narragansett Bay (Oviatt et al. 2002, Murray et al. 2007, Oviatt 2008).

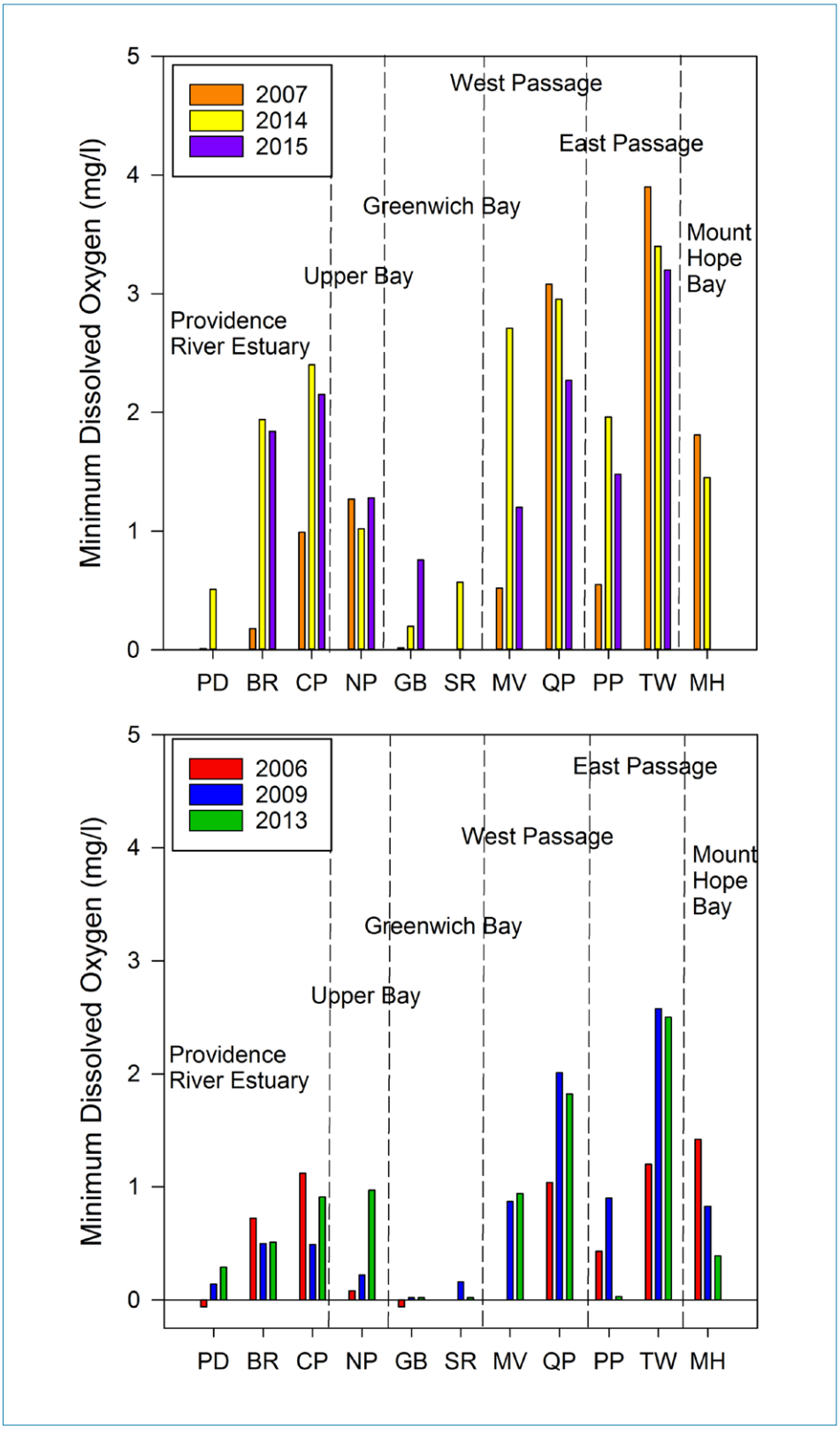


Figure 11. Minimum bottom water dissolved oxygen values for all Narragansett Bay Fixed Site Monitoring Network sites for the dry years of 2007, 2014, and 2015 (top) and for the wet years of 2006, 2009, and 2013 (bottom). See Table 1 for acronyms. PD is Phillipsdale Dock in the Seekonk River. Wet and dry year determination included in Methods section (Table 3). The sites are presented by Narragansett Bay sections.

HYPOXIA IN DRY YEARS VS. WET YEARS

Both the Hypoxia Index and the Spatial Surveys showed high interannual variability (Figures 4 and 9), which appears to be linked with the amount of precipitation in any given summer (June through September; Tables 3 and 5). Freshwater flux was quantitatively defined (see Methods) and wet (2006, 2009, 2011, 2013) and dry (2005, 2007, 2008, 2010, 2012, 2014, 2015) years established (Tables 3 and 5).

The Hypoxia Index showed that hypoxic events tended to occur in the Providence River-Upper Bay (PRUB), Upper West Passage (UWP), and Greenwich Bay (GRBY) during wet years, with Greenwich Bay showing significant hypoxic events in dry years as well (Figures 4 and 5). Hypoxia during dry years may also occur in smaller embayments not included in this analysis, such as the Seekonk River, north of the Providence River Estuary (PD in Figure 11). These areas tend to be tidally restricted and shallower, which could encourage more hypoxia to develop.

The area of hypoxic bottom waters in the Spatial Surveys also appeared to track with the freshwater flux with wet years having a higher area of hypoxic waters (Table 5; Figure 9). The greater area of hypoxic waters in 2010 was likely related to higher nutrient loading from the failed Pawtuxet River wastewater treatment facilities (due to catastrophic April 2010 flooding event), while the cause of the higher area of hypoxic waters in 2008 was unclear (Table 5). The area of hypoxic bottom water approximately doubled between the dry (8 percent) and wet (19 percent) years, while the departure from mean river flux changed from -10.6 m^3/s to 29.5 m^3/s (Table 5).

An additional comparison analyzed the minimum bottom water dissolved oxygen in wet years (2006, 2009, and 2013) and dry years (2007, 2014, and 2015) (Figure 11). The minimum dissolved oxygen value for each NBFSMN site was the lowest single fifteen-minute value for the entire season. Overall, minimums generally follow a north-south gradient, with higher minimums in the southern sections of the Bay (TW). Minimum DO concentrations were generally higher in drier years. Finally, the minimum DO concentrations for each station do fall below the 2.9 mg/l threshold, indicating some degree of low oxygen during the summer season.

The mechanism by which wetter conditions make hypoxia more severe is not fully understood. Higher-than-average runoff does contribute to stronger stratification, which can enhance hypoxia by impeding re-aeration through reduced vertical mixing (Codiga 2012). Wet years are also thought to increase nutrient loads (see "Precipitation" chapter) and chlorophyll concentrations (see "Chlorophyll" chapter) that drive hypoxia. The relative importance of these processes, and possible influence of differing rates or patterns of circulation during wet and dry years, is a topic that requires additional research.

HYPOXIA AND WASTEWATER TREATMENT FACILITY NUTRIENT REDUCTIONS

An important context for examining trends in hypoxia is that wastewater treatment facilities have been upgrading over the past fifteen years to reduce nutrient pollutant loadings, with a 50 percent reduction in nitrogen loadings (relative to 1995/96 loadings) achieved in the summer of 2012 (see "Nutrient Loading" chapter). Indirectly, these reductions are intended to decrease the amount of hypoxia experienced in the Providence River Estuary and Upper Bay. However, given the multiple factors influencing formation of hypoxia and high interannual variability in the monitoring data, it is too soon to determine how the Bay is responding to the reductions. The Estuary Program is encouraged that the recent results of the Hypoxia Index and Spatial Surveys reflect weakening hypoxia, as results for 2014 and 2015 were the lowest on record. However, those were dry years, and earlier dry years (2004 and 2007; Figures 4, 5 and 9) were comparable to 2014 and 2015. Significant time lags may occur between nitrogen reduction and the biological response of the ecosystem (e.g., organic matter production, decomposition, and oxygen depletion). Additional monitoring is necessary before the response of Bay hypoxia to nutrient reductions can be determined. In particular, much will be learned from future years that are wetter than average. Wet years will best reveal any changes brought by wastewater treatment facility upgrades, which included extensive combined sewer overflow (CSO) abatement efforts (NBC 2017; see "Precipitation" chapter).

HYPOXIA AND CLIMATE CHANGE

Dissolved oxygen levels in Narragansett Bay will be influenced by a changing climate. Warming temperatures are an expected consequence of climate change (see "Temperature" chapter) and are expected to result in higher rates of primary production and respiration. Warmer temperatures also contribute to the stratification of the water column, which prevents mixing of bottom water with the surface (USEPA 2016). Biochemical reactions and respiration in the water column and sediments would continue to decrease the dissolved oxygen concentrations in bottom water, and less vertical mixing

could exacerbate hypoxic conditions. This would increase the consumption of dissolved oxygen and reduce dissolved oxygen concentrations towards hypoxic conditions. Additionally, water temperature dictates how much oxygen is available, with warmer water holding less dissolved oxygen than cooler water. As temperatures rise, DO concentrations will be affected regardless of the impacts of temperature on primary production and stratification although at the projected summer temperatures, the change in DO concentration will be very small compared to biological processes (USEPA 2016).

Another predicted effect of climate change is the increased frequency of intense storms (RI EC4 STAB 2016, USEPA 2016; see "Precipitation" chapter). These storms can lead to higher pulses of river runoff, causing more nutrient loading and salinity-induced stratification. The addition of nutrients and stratification can set up the same hypoxic conditions noted with warming temperatures. Analysis of stratification concluded that climate-driven changes will be due to increases of river runoff more than increases in temperature (Codiga 2012).

River runoff in late spring/early summer sets up the severity of hypoxia for the remainder of the summer in Narragansett Bay (Codiga 2012). If river runoff pulses occur in late spring/early summer, hypoxic events could increase throughout the summer, even if no further pulses occur. This type of situation occurred in 2013, when June was very wet and the remainder of the summer was very dry, but summer bottom water dissolved oxygen concentrations were very low. This same phenomena may have also happened during years with higher than average river flow and higher than average stratification, such as 2003 (year of the Greenwich Bay fish kill), 2006, and 2009 (Codiga et al. 2009). There is a relationship between stratification and hypoxia, but it has limits (Codiga 2012), possibly because strong stratification is due to high runoff. The higher runoff may also reduce the flushing time, decreasing the time nutrients stay in the system. Further investigation into these processes is needed.

SUMMARY

The Hypoxia Index and Spatial Surveys show that hypoxia was low for 2014 to 2015. A down-Bay increase was evident in bottom water dissolved oxygen concentrations, with the lowest concentrations occurring in the Providence-Seekonk River Estuaries and the highest at the southern end of Narragansett Bay. The down-Bay gradient is a persistent annual feature, regardless of the hypoxia levels in any given year. Both methods show a strong pattern of interannual variability in dissolved oxygen dynamics that likely stems from the amount of freshwater delivery to the Bay. Therefore, in dry years like 2014 and 2015, bottom water dissolved oxygen concentrations would be higher than in wet years like 2013, resulting in more severe hypoxia in wet years. The Hypoxia Index and Spatial Surveys complement each other, even though the approaches have different data, resolution, and timescales. The wet/dry analysis conducted with both approaches is probably the most important step to learn how dissolved oxygen concentrations are responding to changes in weather/climate. The findings offer hints at how dissolved oxygen is responding to management actions, such as nutrient reductions.

Data Gaps and Research Needs

- A major gap with the Narragansett Bay Fixed Site Monitoring Network and spatial survey is the lack of resource commitment (e.g., funding and personnel) to continue these field monitoring and data processing efforts. The NBFSMN and spatial survey require constant equipment maintenance and costly upgrades. Additionally, gaps in the NBFSMN for dissolved oxygen exist for portions of Mount Hope Bay, the Sakonnet River, and the Lower East Passage where there are no monitoring stations.
- High interannual variability limits the discernment of temporal trends in available datasets. Additional data synthesis studies or longer-term monitoring are needed to further explore the different temporal and spatial scales of dissolved oxygen variability and their relationships to other forcing factors (e.g., seasonal rainfall or temperature) and the physical structure of the water column.
- The Phillipsdale site, which has unique circulation patterns and is proximal to a major freshwater source (the Blackstone River), was not analyzed for the Hypoxia Index or the Chlorophyll Bloom Index (see "Chlorophyll" chapter). In light of nutrient reductions and changes to the dissolved oxygen and chlorophyll concentrations in other sections of the Bay, the Phillipsdale data need to be analyzed to see how this upper section of the Seekonk River is changing.
- The combination of dissolved oxygen data and hydrodynamic modeling efforts can provide a better understanding of how hydrodynamic properties of the Bay are influenced by physical forces, such as wind, precipitation, and river flow, and how dissolved oxygen levels respond. Models should be used to better understand the connection between benthic conditions and overlying dissolved oxygen conditions.

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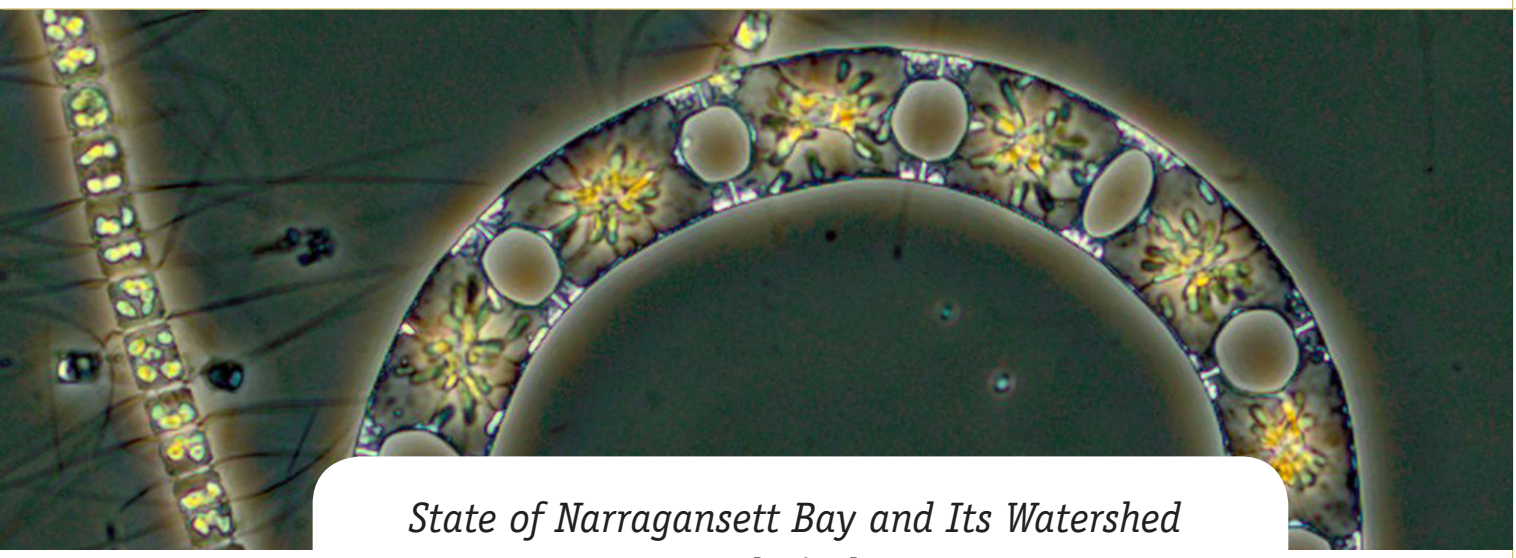
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2017 Technical Report

Bay Ecosystem Condition Indicators

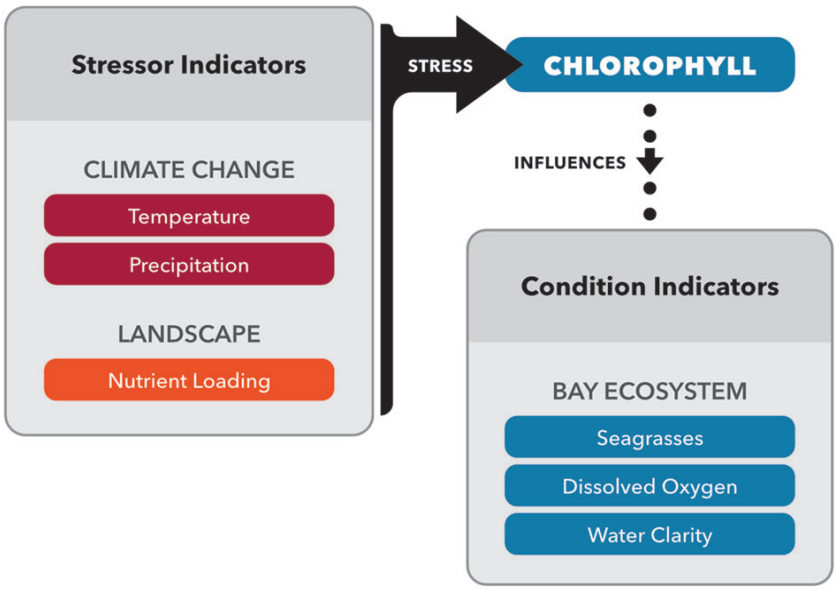
CHAPTER 16: CHLOROPHYLL

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NARRAGANSETT BAY ESTUARY PROGRAM

Photo: Eucampia *spp., 200X (Narragansett Bay Commission)*

Overview



BACKGROUND

- Chlorophyll is the main photosynthetic pigment in phytoplankton. The concentration of chlorophyll indicates the amount of phytoplankton in the water and is used widely as an indicator of water quality. Because phytoplankton growth varies with the amount of nutrients available, chlorophyll concentrations are influenced by nutrient loading from wastewater treatment facilities and by precipitation, which carries nutrients from land into the Bay. The amount of phytoplankton in the water influences dissolved oxygen concentrations, mostly causing decreases through respiration and decomposition but also causing increases through photosynthesis under some conditions. Increases in phytoplankton biomass cause declines in water clarity, reducing light available to seagrass habitat. Warmer temperatures encourage grazing by zooplankton on phytoplankton, particularly in the winter-spring, reducing the amount of chlorophyll.

KEY FINDINGS

- ***Status:*** During 2013 to 2015, Narragansett Bay's summer chlorophyll concentrations were variable and declined along a north-to-south gradient. In 2013, chlorophyll concentrations were relatively high throughout the Bay, and in 2014 and 2015, chlorophyll concentrations were among the lowest in recent years, except in some locations, such as Greenwich Bay and Mount Hope Bay.
- ***Trends:*** During winter-spring and summer, chlorophyll concentrations consistently decrease from the Upper Bay to the Lower Bay. Recent summer data (2001 to 2015) collected throughout the Bay did not show any clear trends over time, but the

concentrations varied strongly from year to year, potentially in relation to the amount of summer precipitation. Measurements collected from 1972 to 2015 at a site in the southern portion of the Bay showed that average annual chlorophyll concentration declined by 57 percent. Nutrients discharged into the Bay from wastewater treatment facilities have been reduced in recent years, and the potential effects of this reduction on chlorophyll concentrations continue to be assessed.

Introduction

Chlorophyll *a* (hereafter, chlorophyll) is the main photosynthetic pigment in phytoplankton. It is easily measurable, making it useful as a proxy for phytoplankton biomass, which can be defined as the amount of organic carbon being made available through primary production for use by higher trophic levels. Methods of directly measuring primary production are available and routinely used (Oviatt et al. 2002, 2016); however, the measurements are difficult to conduct, and methods are not always comparable (Oczkowski et al. 2016). The concentration of chlorophyll serves as a proxy for those measurements and is used widely as an indicator of water quality to help identify negative effects of excess nutrient loading. The National Oceanic and Atmospheric Administration and the United States Environmental Protection Agency both use a chlorophyll concentration of 20 micrograms per liter (µg/l) as a threshold to designate elevated concentration levels. Concentrations above 20 µg/l are considered indicative of nutrient enrichment and poor condition (Bricker et al. 2007, USEPA 2012).

Chlorophyll concentrations vary with changes in light, nutrient inputs, grazing, temperature, and water circulation and mixing. Phytoplankton need to be in the euphotic zone of the water column (shallow layer of the water column, which receives the most light) in order to photosynthesize, and they require a certain level of nutrients within the euphotic zone. Primary production and chlorophyll concentrations by extension have been found to be positively related to *in situ* total nitrogen concentrations (Oviatt 2008). If no nutrients are added to the system, or phytoplankton use the nutrients faster than can be replenished, chlorophyll production will be limited by nutrient availability. This is especially apparent during the summer in temperate estuaries (Nixon and Pilson 1983, Howarth 1988).

In Narragansett Bay, chlorophyll concentrations vary widely throughout the year. Historically, Narragansett Bay experienced two periods of chlorophyll blooms—one in the winter-spring and the other in summer (Li and Smayda 1998). However, more recently this classic pattern has not always been apparent, as winter-spring blooms have become intermittent (Oviatt et al. 2002), leaving the summer months with the dominant chlorophyll concentrations in some years. In addition to seasonal and yearly changes, Narragansett Bay has a spatial gradient in which chlorophyll generally decreases from north to south (NBEP 2009). Chlorophyll concentrations are typically highest in the Seekonk River (greater than 60 µg/L in the growing season), and high concentrations have also been documented in Greenwich Bay and the lower Taunton River (NBEP 2009). Chlorophyll concentrations decrease southerly through the East and West Passages. In the Lower Bay, typical concentrations at the University of Rhode Island Graduate School of Oceanography (GSO) sampling site at Fox Island (Middle West Passage) and at the GSO Dock (Lower West Passage) are less than 10 µg/l.

The northern parts of the Bay have high chlorophyll concentrations because of nutrient loading from wastewater treatment facilities (WWTFs) and rivers. High chlorophyll concentrations, typically due to phytoplankton blooms, are linked with high amounts of organic matter being delivered to the benthic habitat. The respiration of this organic matter, coupled with physical forcing, can reduce the amount of dissolved oxygen available in the water column, negatively affecting marine life (see "Dissolved Oxygen" chapter). Riverine loading changes with precipitation, affecting the amount of nutrients delivered to the upper reaches of the Bay. More precipitation could exacerbate organic matter deposition and hypoxia, while less could mitigate the effects (Harding et al. 2014; see "Dissolved Oxygen" and "Precipitation" chapters). Recently, wastewater treatment facility nutrient loadings in the Upper Bay have been reduced by over 50 percent from levels observed 20 years ago (RIDEM 2015; see "Nutrient Loading" chapter). The question of whether

ecosystem changes have occurred in response to meeting this nutrient reduction goal is only starting to be explored.

In addition to nutrient reductions, climate change and grazing pressure by zooplankton play roles in determining chlorophyll concentrations. Annual average chlorophyll concentrations in the Mid and Lower East and West Passages have been declining since the 1970s (Li and Smayda 1998, 2001; Oviatt et al. 2002; NBEP 2009), resulting in a reduction in the amount of organic matter being delivered to benthic habitats (Oviatt et al. 2002, Nixon et al. 2009). The chlorophyll reductions are thought to be the result of increased grazing pressure as well as regional atmospheric and climate patterns (Keller et al. 1999, Nixon et al. 2009, Borkman and Smayda 2009). Intense grazing by zooplankton may reduce phytoplankton levels throughout the year and reduce the magnitude of winter-spring blooms (Keller et al. 1999, Oviatt et al. 2002, Nixon et al. 2009). Grazing pressure is increased with warmer winters, a likely effect of climate change (Fulweiler et al. 2015; see "Temperature" chapter). Warmer winters may also increase cloud cover, decreasing the amount of light available to stimulate phytoplankton blooms. Regional climate patterns, such as the North Atlantic Oscillation, may affect bloom dynamics as well. Years with a low oscillation index tend to be colder, allowing for a winter-spring bloom to develop, while years with a high oscillation index tend to have warmer winters with more pronounced summer blooms (Borkman and Smayda 2009). These changes are explored further in the Discussion below.

This chapter focuses on the response of chlorophyll concentration to two stressors–precipitation (which helps to carry nutrients from the Narragansett Bay Watershed into the Bay) and nutrient loadings (from wastewater treatment facilities)–and the potential influence of chlorophyll concentration on dissolved oxygen concentrations. This information is essential to understand how primary production may respond to future changes in nutrient loading and precipitation regimes. Additionally, this chapter provides a framework to help develop chlorophyll concentration as an indicator of Narragansett Bay's condition. The selection of the most appropriate method to track and report on chlorophyll over time in the Bay remains a work in progress, and the preliminary assessments presented in this chapter provide a foundation for future work.

Methods

The Narragansett Bay Estuary Program used a combination of three methods to evaluate the status and trends of chlorophyll concentrations in the Bay:

- ***Grab Samples:*** The Estuary Program analyzed chlorophyll concentration data from grab (filtered surface water) samples collected at sites throughout the Providence River Estuary and the East and West Passages by multiple partners (see below).
- ***Chlorophyll Bloom Index:*** The Estuary Program calculated a Chlorophyll Bloom Index using the Narragansett Bay Fixed Site Monitoring Network (NBFSMN) near-surface time series

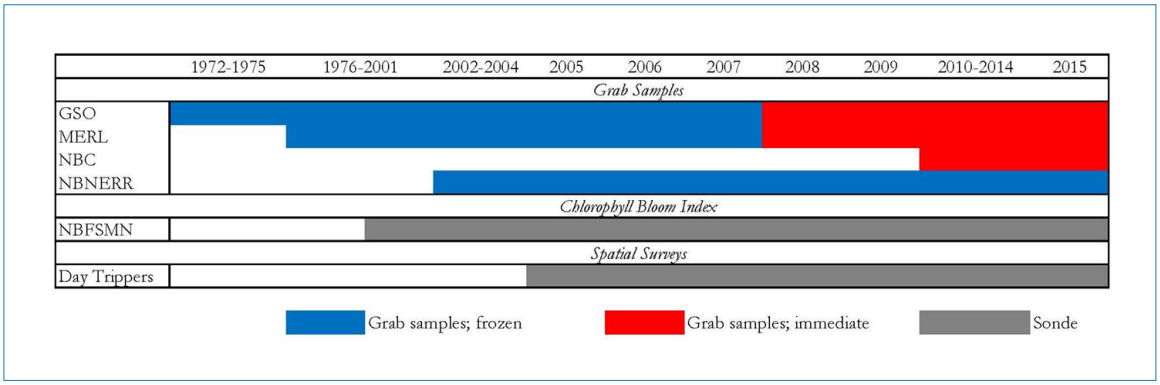


Figure 1. All data analyzed for assessment of chlorophyll status and trends. Descriptions of site locations are provided in the text. Blue indicates samples that were frozen prior to analysis. Red indicates samples that were immediately extracted and analyzed. Additional information about methods is available upon request. *GSO: Phytoplankton Survey at the University of Rhode Island's Graduate School of Oceanography. MERL: Marine Ecosystem Research Laboratory at the University of Rhode Island's Graduate School of Oceanography. NBC: Narragansett Bay Commission. NBNERR: Narragansett Bay National Estuarine Research Reserve. NBFSMN: Narragansett Bay Fixed Site Monitoring Network.*

of *in situ* fluorescence. The Chlorophyll Bloom Index is a modification of the Hypoxia Index (see "Dissolved Oxygen" chapter), substituting surplus-duration for deficit-duration. The Chlorophyll Bloom Index was developed for the State of Narragansett Bay and Its Watershed, and while the results of the first iteration are presented here, the Index remains under development.

- ***Spatial Surveys:*** The Estuary Program analyzed the spatial distribution of surface chlorophyll using data collected by the Day Trippers, which is a collaborative sampling effort involving several organizations (Prell et al. 2015, 2016).

These methods are detailed below. Figure 1 shows the data sources and the temporal coverage of the three types of data that were analyzed. The Status and Trends section of this chapter is divided into three parts to report separately the results for each method.

GRAB SAMPLES

The Narragansett Bay Estuary Program obtained chlorophyll concentration grab sample data from the Narragansett Bay Commission (NBC); Narragansett Bay National Estuarine Research Reserve (NBNERR); the Marine Ecosystem Research Laboratory (MERL) at the University of Rhode Island's Graduate School of Oceanography (GSO); the Phytoplankton Survey (2003 to 2015) at GSO; and the NarrBay portal for GSO data collected between 1972 and 1997 (Figure 1). The sample sites were Phillipsdale Dock, India Point Park, Bullock's Reach, and Conimicut Point in the Providence River Estuary (NBC), Fox Island in the Middle West Passage (GSO), the dock at the University of Rhode Island's Graduate School of Oceanography (MERL) in the Lower West Passage, and T-Wharf on Prudence Island in the Middle East Passage (NBNERR).

The grab samples were surface water collected and filtered for the express purpose of measuring chlorophyll concentration. The water was filtered in the lab or field to collect particles, including phytoplankton, on a glass-fiber filter. The filters were then either preserved and frozen or immediately placed in an acetone solution to extract the chlorophyll pigments. Pigment concentrations were determined in the laboratory with a bench-top fluorometer. Samples were collected weekly to monthly at each site depending on weather. GSO and MERL had the most consistently collected samples at weekly intervals, rarely missing a week throughout their datasets.

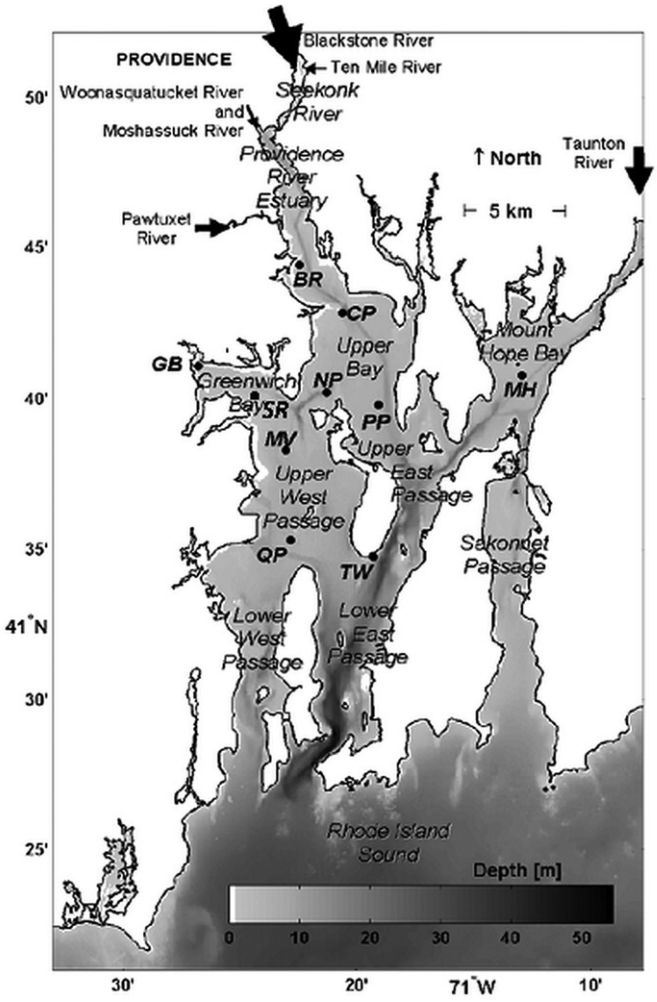


Figure 2. Site locations, bathymetry, and river sources for the NBFSMN data used for the Chlorophyll Bloom Index (from Codiga 2016).

All data were converted to units of µg/l for ease of comparison. Additionally, GSO data from post-2008 were corrected for a methods change (Graff and Rynearson 2011, Fulweiler and Heiss 2014). Prior to 2008, samples from GSO were frozen prior to analysis, and then after 2008 samples were immediately extracted and analyzed. Graff and Rynearson (2011) found that freezing prior to analysis reduced chlorophyll concentrations by 50 percent. Based on their work, Graff and Rynearson (2011) provided a correction to the GSO data, which was applied by the Estuary Program. Data from other sources were either immediately extracted (NBC) or frozen until analysis with no correction available (NBNERR and MERL). All figures in this chapter use blue to indicate samples that were frozen and red for samples that were immediately extracted. GSO samples that

were corrected to reflect the methods change are represented in purple and can be compared to one another.

The GSO and MERL data are summarized by yearly average and maximum concentrations and by summer (July–September) average and maximum concentrations, while NBC and NBNERR data are summarized by summer average and maximum concentrations. The summer average and maximum concentrations are presented for comparison across all types of data, including the Chlorophyll Bloom Index and Spatial Surveys. Therefore, the summer grab sample data focus on the years from 2002 to 2015, while the yearly data are presented for the entire dataset available (1972–2015 for GSO and 1976–2015 for MERL).

CHLOROPHYLL BLOOM INDEX

The Estuary Program, with Dan Codiga (independent contractor), developed a new metric, the Chlorophyll Bloom Index, to quantify phytoplankton blooms based on time series measurements of chlorophyll. Summer surface chlorophyll concentration (as fluorescence) data from the Narragansett Bay Fixed Site Monitoring Network (NBFSMN) for 2001 to 2015 were analyzed to understand the spatial and temporal changes since the Network's inception. Chlorophyll data were collected every fifteen minutes using fluorescence sensors located one meter below the surface and were reported in units of µg/L. Information about the NBFSMN, including monitoring site locations and depths, is available online (NBFSMN 2016). The quality control and quality assurance protocols used are described in Rhode Island Department of Environmental Management's Quality Assurance Project Plan (RIDEM 2014).

Data from ten NBFSMN sites were analyzed individually (Figure 2; Table 1) using data from the mid-spring to mid-autumn seasonal period from May 15 through October 14. From 2001 to 2005, a relatively small number of sites were monitored, mostly in the Providence River Estuary and Upper Bay. From 2006 to 2015, most of the ten sites were sampled (Table 2).

To identify and quantify individual phytoplankton bloom events, the Estuary Program applied a moving window trigger algorithm (Codiga 2008, Codiga et al. 2009). The algorithm determined the start and end time for each individual event (phytoplankton bloom) and its surplus-duration (µg/l day), which is represented by the area in the time series above the threshold (Figure 3). The surplus-duration of an event increases as a result of chlorophyll concentration reaching higher values during the event and/or as a result of the event having a longer duration. Thus, the surplus-duration is a reflection of both the intensity of the event (concentration difference from the threshold, during the event) and its duration (time that the concentration is higher than the

Table 1. Chlorophyll thresholds for each site, computed as the 80th percentile of all years' data between May 15 and October 14 from that site. Acronyms for site names correspond to acronyms on map in Figure 2. For analysis, sites were grouped by geographic location. Site groups were Providence River-Upper Bay (PRUB), Upper West Passage (UWP), Upper East Passage (UEP), and Greenwich Bay (GRBY).

Site Group	Site	Acronym	Threshold [µg/l]
PRUB	Bullock's Reach	BR	26.1
PRUB	Conimicut Point	CP	19.6
PRUB	North Prudence	NP	15.0
UWP	Mount View	MV	13.1
UWP	Quonset Point	QP	9.3
UEP	Poppasquash Point	PP	12.8
UEP	T-Wharf	TW	5.0
GRBY	Greenwich Bay	GB	28.7
GRBY	Sally Rock	SR	17.9
MH	Mount Hope	MH	14.6

threshold). As an example of the meaning of the surplus-duration units, consider that if an individual event has chlorophyll concentration that rises above the threshold by an amount of two µg/l and remains there for a duration of three days, it has a surplus duration of six µg/l day.

The thresholds used for chlorophyll were site-specific and computed as the 80th percentile of all chlorophyll values recorded in all sampled years at a given site (Table 1). Numeric water quality criteria for chlorophyll concentration have not been established for Rhode Island or Massachusetts. Therefore, the threshold used for the bloom index does not represent an acceptable chlorophyll condition, merely the level that 80 percent of the data are below at a given site. The thresholds for both the Greenwich Bay and Bullock's Reach stations were calculated to be above the 20 µg/l threshold that federal agencies consider poor condition (Table 1; Bricker et al. 2007, USEPA 2012). The annual Chlorophyll Bloom Index was determined for each individual site as the sum over all individual bloom events that occurred during the mid-May to mid-October period (Figure 3). The annual Chlorophyll Bloom Index is referred to as the cumulative surplus because it is the cumulative surplus-duration, or sum of the surplus-durations of all individual bloom events during the May-October period.

Periods of missing observations (gaps), when field sensors were not operating or data did not meet

Table 2. Percentage of time during the mid-May to mid-October analysis period for which valid near-surface chlorophyll values were available at each of the ten sites in each of the fifteen years. Acronyms for site names correspond to acronyms on map in Figure 2 and Table 1. Asterisk (*) indicates site with less than 55 percent valid values; that site was not included in analysis for that year. Blank cells indicate sites that were not sampled in a given year. Numbers in this table may differ from those in the similar table in the "Dissolved Oxygen" chapter because although located on the same mooring, the chlorophyll and oxygen sensors are independent, and their data quality can be influenced by different factors.

	BR	CP	NP	MV	QP	PP	TW	GB	SR	MH
2001	100		69							
2002	83		71							
2003	55	62	69					57		
2004	87		86	66		56	82	76		
2005	89	76	80	67	*	57	89	79		68
2006	80	79	61	99	61	88		97		98
2007	59	94	93	63	81	88		89		92
2008	95	96	83	100	97	97		82	100	83
2009	98	65	85	92	83	55		92	91	88
2010	85	87	91	84	89	86	96	84	83	84
2011	85	90	90	91	91	90	75	91	91	84
2012	95	96	96	87	96	95	100	97	80	95
2013	96	100	86	98	100	97	95	87	84	100
2014	92	96	96	91	83	96	96	96	78	84
2015	76	91	89	91	91	91	98	93	86	91

quality standards, typically included about ten to 40 percent of the mid-May to mid-October period in a given year at a given site (Table 2). Due to these gaps, the moving window trigger algorithm gives a lower bound for the seasonal Chlorophyll Bloom Index. The gap contribution, and thus an upper bound for the Chlorophyll Bloom Index, is estimated (Codiga 2016) using the time period(s) of the gap(s), together with event statistics (specifically, the probability and severity of events, for the time of year of the gap and the particular site) from all other years at the site. The seasonal Chlorophyll Bloom Index is the mean of the upper and lower bounds, and its uncertainty due to gaps is half the difference between the upper and lower bound. The multi-site Chlorophyll Bloom Index is computed for groups of sites as the mean of the Chlorophyll Bloom Index results of individual sites in the group (Table 1); it is presented with uncertainties that are the mean, of the sites in the group, of the gap-based uncertainties for individual sites.

This approach to examine events where chlorophyll concentrations were above the 80th percentile has limitations, and it may not be the optimal way to describe these events. In future research, the Estuary Program plans to examine these events further, analyze the validity of this method, update the results as necessary, and explore other metrics for analyzing chlorophyll blooms if needed.

SPATIAL SURVEYS

The Estuary Program, with Warren Prell (Brown University) under contract, analyzed the spatial distribution of surface chlorophyll using data collected by the Day Trippers. The Spatial Surveys began in 2005 (Prell et al. 2015, Prell et al. 2016) and used a Sea-Bird Electronics SBE 19 Plus SeaCAT profiler equipped with a Self-Contained Underwater Fluorescence Apparatus. This device emits light at 460 nm to excite particles of chlorophyll in the water, which then emit fluorescent light back at a signature wavelength of 685 nm (Turner Designs 2004). The surveys focused on the warm summer months (June, July, August, and September) near neap tides when the risk of hypoxia is believed to be greatest (see "Dissolved Oxygen" chapter). Because dissolved oxygen exhibits little diurnal variability below the pycnocline (the depth at which water density rapidly increases), the surveys were conducted in the morning hours irrespective of tidal phase so that photosynthesis was not maximum. Tidal excursions in the Bay are small compared to the scale of the survey, so no attempts were made to correct for tidal phase. The Spatial Surveys dataset included 58 surveys at approximately 77 sites over eleven years (2005 to 2015) during the four summer months, and it contained 4,154 individual profiles throughout the Providence River Estuary, Upper Bay, and Upper West and East Passages (Figure 4; see Prell et al. 2015, Prell et al. 2016 for details on survey

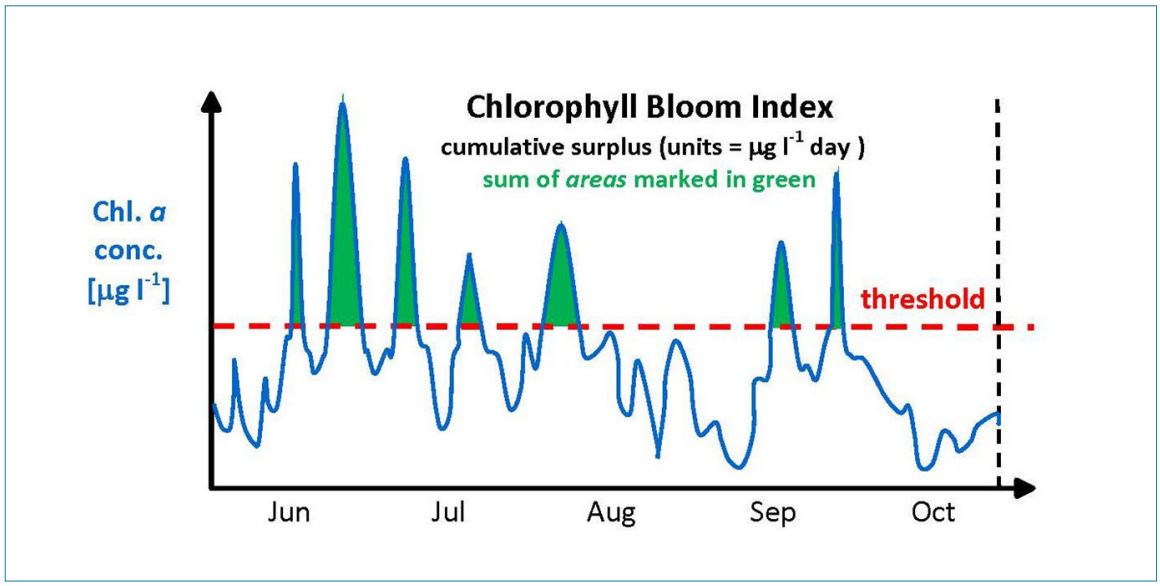


Figure 3. Schematic showing computation of the Chlorophyll Bloom Index, relative to a given threshold (red dashed horizontal line), from the time series of near-surface fluorescence-based chlorophyll measurements (blue line) at a single mooring site. The index is the total area above the threshold level and below the observations curve. It is the season-cumulative surplus-duration of all chlorophyll bloom events during the mid-May to mid-October analysis period.

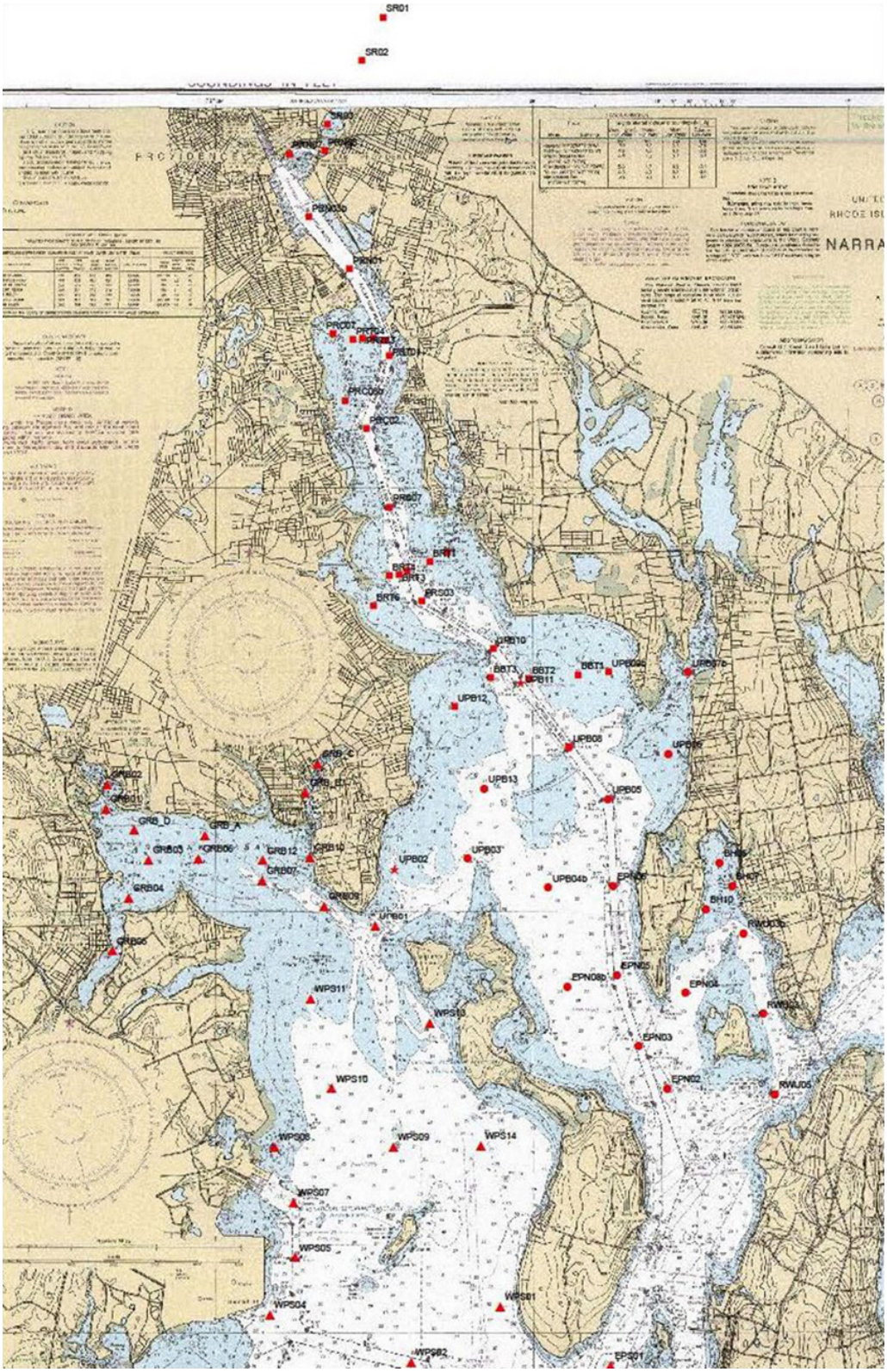


Figure 4. Locations of Spatial Surveys sites. Coordinates, water depths, and key to site names are available in Prell et al. 2015 and 2016.

sites and sampling dates). Profiles at the stations are measured five to seven days per year. Comparison of the 2005 data to the 2015 data revealed that the chlorophyll measurements were systematically offset between sondes and were adjusted to generate internally consistent data among the surveys (Prell et al. 2016).

The concentrations of chlorophyll varied widely across the Bay within individual surveys. As numeric water quality criteria for chlorophyll have not been established, the Estuary Program determined useful mapping intervals by examining the probability distribution of all surface measurements (Figure 5). Survey chlorophyll measurements were interpolated at one meter increments; therefore surface chlorophyll concentrations are those which were interpolated at one meter depth. In the probability distribution of all data, five outliers (were greater than 140 µg/l) identified that exceeded sonde specifications and calibrations and were eliminated from further calculations. The probability plot revealed

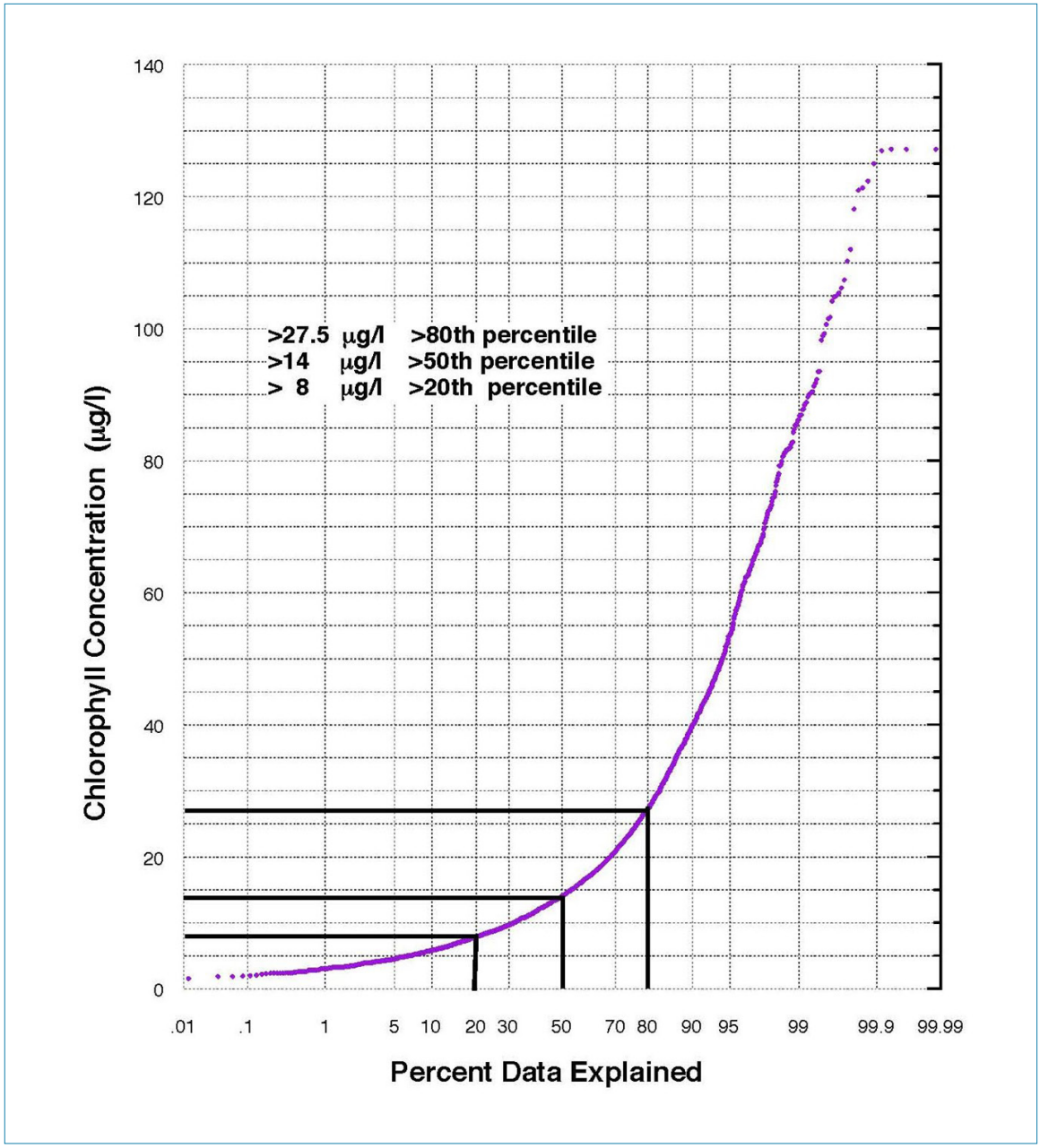


Figure 5. Probability distribution of chlorophyll concentrations (µg/l) from all surveys (2005 to 2015) at depth of one meter or less.

that approximately 20 percent of the data exceeded 27.5 µg/l (80th percentile), 50 percent exceeded 14 µg/l (50th percentile) and 80 percent exceed 8 µg/l (20th percentile). The 80th percentile (see Chlorophyll Bloom Index in Methods above) was selected to represent high chlorophyll bloom events for comparison to other water properties. The 20th and 50th percentiles were chosen to provide proportional divisions and were used in mapping surface chlorophyll. Maps were created using GIS interpolation of the results of the 20th, 50th, and 80th percentile analyses (Prell et al. 2016).

Determination of Wet and Dry Years

The Estuary Program examined differences in chlorophyll concentrations between dry years (low river flow and low precipitation) and wet years (high river flow and high precipitation) to assess the impact of freshwater flux and its associated nutrient flux. Data from the United States Geological Survey (USGS) on daily flow discharges from the Blackstone, Pawtuxet, and Taunton Rivers were combined with long-term data from Ries (1990) to calculate the total freshwater flux to the Bay during the summer (June, July, August, September). The long-term median from 2005 to 2015 was calculated and compared with median flow for individual years. When individual medians were greater than the dataset median (2005 to 2015), they were considered wet years. Dry years were those with medians less than the dataset median (Table 3; Prell et al. 2016).

Status and Trends

The Estuary Program examined the data for evidence of elevated chlorophyll concentrations. The results presented for this indicator are preliminary. Additional work is needed to determine the method or suite of methods that are most appropriate for use by the Estuary Program.

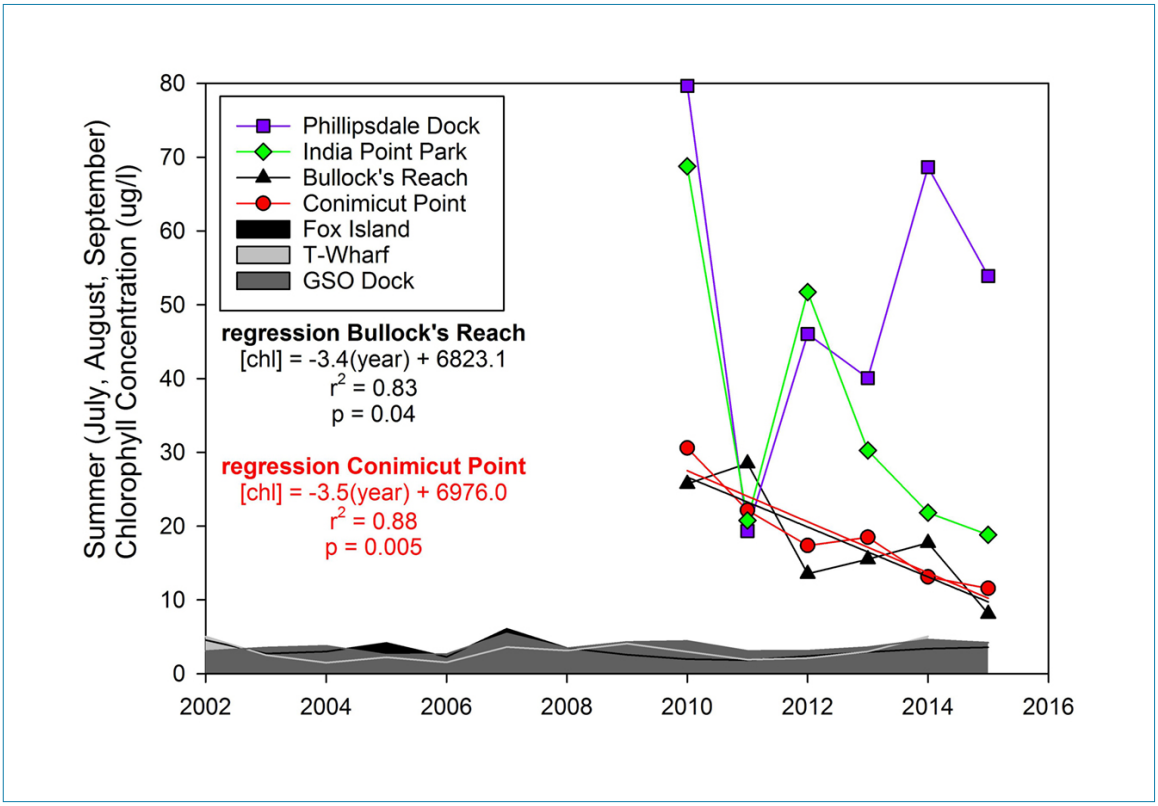


Figure 6. Summer (July–September) chlorophyll concentrations from grab (filtered water) samples from 2002 to 2015. Not all sites had data available for all years. Sites are listed from north to south in the Bay. Bullock's Reach (black line) and Conimicut Point (red line) were the only sites that had significant reductions in chlorophyll over the time period.

Table 3. River flux departure during June, July, August, and September (JJAS). Dry years were those in which the annual summer departure was less than the long-term median; wet years were those in which the departure was greater than the long-term median. Wet years are shaded, and "wet" is bold text.

YEAR	River Flux Departure m^3/s JJAS	Wet/Dry Designation
2005	-10	DRY
2006	26	**WET**
2007	-13	DRY
2008	-4	DRY
2009	44	**WET**
2010	-17	DRY
2011	25	**WET**
2012	-5	DRY
2013	23	**WET**
2014	-17	DRY
2015	-8	DRY

Table 4. Yearly and summer (July–September) average and maximum chlorophyll concentrations (µg/l) in Bay sections for 2014/2015.

Narragansett Bay	Yearly Mean	Yearly Maximum	Number of Samples	Summer Mean	Summer Maximum	Number of Samples
			Providence River Estuary			
Phillipsdale Dock	31.0	262.1	19	53.9	262.1	7
India Point Park	12.7	44.7	20	18.8	44.7	6
Bullock's Reach	11.5	33.0	23	17.7	25.2	7
Conimicut Point	12.8	40.7	20	11.6	28.3	6
			West Passage			
Fox Island	2.4	8.9	51	3.6	8.9	13
GSO Dock[(1)]	3.7	5.9	37[(1)]	4.2	5.9	12
			East Passage			
T-Wharf[(2)]	3.3	6.0	24	5.7	6.0	6

[(1)] Received data through September 2015.
[(2)] 2015 data unavailable; 2014 data used to represent status.

GRAB SAMPLES

The dataset generated by grab samples is limited by the number of samples collected through the year but still contributes to understanding of chlorophyll concentrations in Narragansett Bay. For 2014/2015, mean summer (July to September) concentrations ranged from 3.6 to 53.9 µg/l for all sites, and maximum concentrations ranged from 5.9 to 262.1 µg/l (Table 4). Yearly mean surface water chlorophyll concentrations ranged from 2.4 to 31.0 µg/l, and maximum concentrations ranged from 5.9 to 262.0 µg/l (Table 4). Given the limited sampling frequency at most stations, these data may not be fully representative of conditions at stations known to experience high variability.

The grab sample data showed higher concentrations of chlorophyll in the Providence River Estuary than in the West Passage or East Passage (Table 4; Figure 6). This down-Bay gradient is consistent with other chlorophyll data sources and visible in other metrics, such as nutrients and bottom water dissolved oxygen (Oviatt et al. 2002; see "Dissolved Oxygen" chapter).

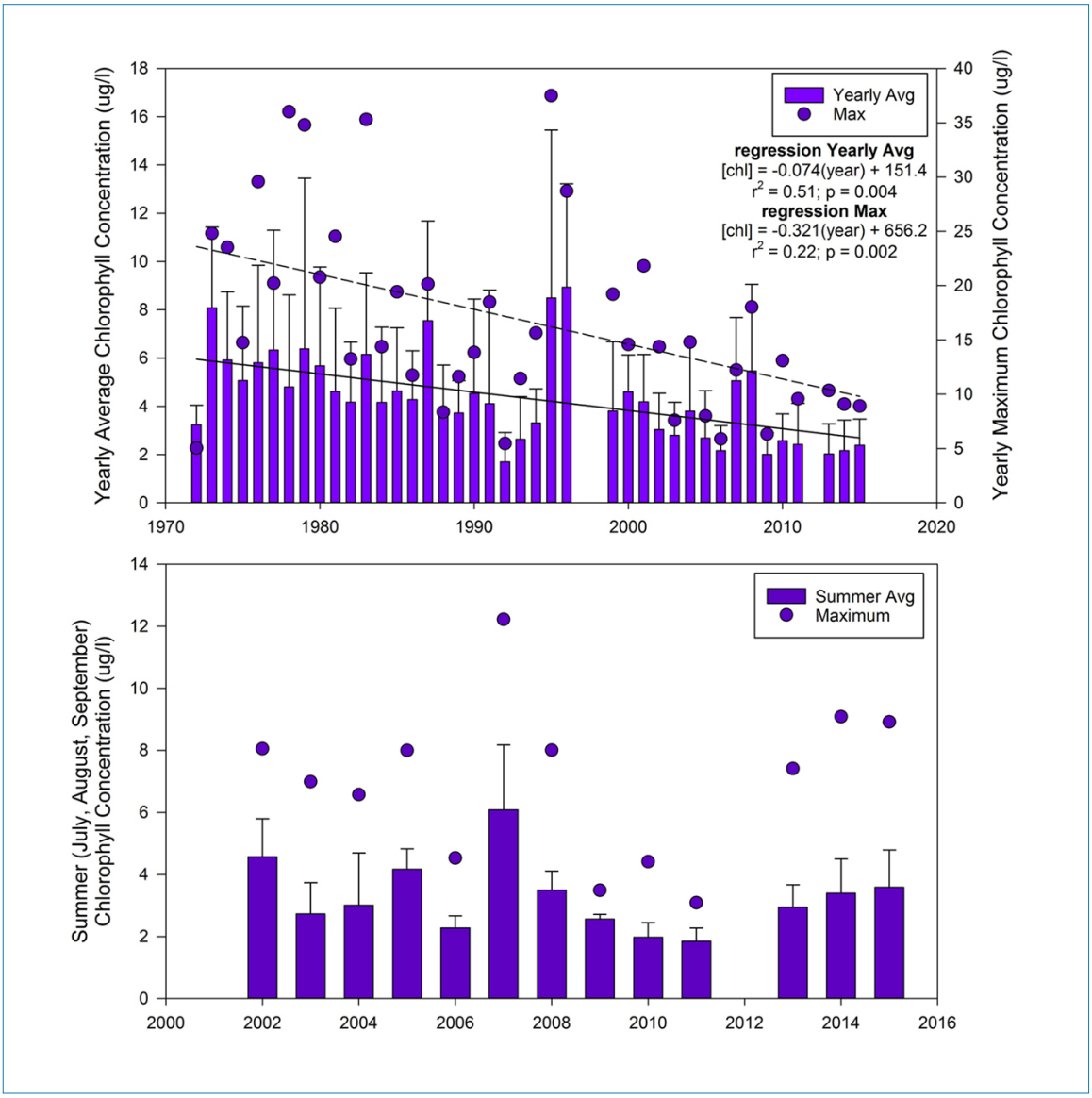


Figure 7. Fox Island (GSO) chlorophyll concentrations (µg/l). Top: Yearly mean (with standard deviations) and yearly maximum surface chlorophyll concentrations (circles). Linear regressions were run for yearly mean (solid line) and maximum (dotted line) concentrations, and both were found to be statistically significant. Bottom: Summer (July–September) mean (with standard deviations) and maximum surface chlorophyll concentrations for 2002 to 2015. A linear regression was run and not found to be statistically significant.

From 1972 to 2015, the yearly mean chlorophyll concentration decreased by 57 percent at Fox Island (Figure 7). Trends in the MERL dataset were analyzed by each method type: frozen or immediately extracted chlorophyll samples. Neither the frozen yearly mean concentration data from 1976 to 2008 nor the immediately extracted data from 2009 to 2015 had any statistically significant changes (Figure 8). Yearly maximum chlorophyll concentrations significantly declined at both Fox Island (1972 to 2015; GSO) and the GSO Dock (1976 to 2008; MERL). Summer maximum concentrations did not appear to change between 2002 and 2015 for these West Passage sites (Figures 7 and 8). In the Providence

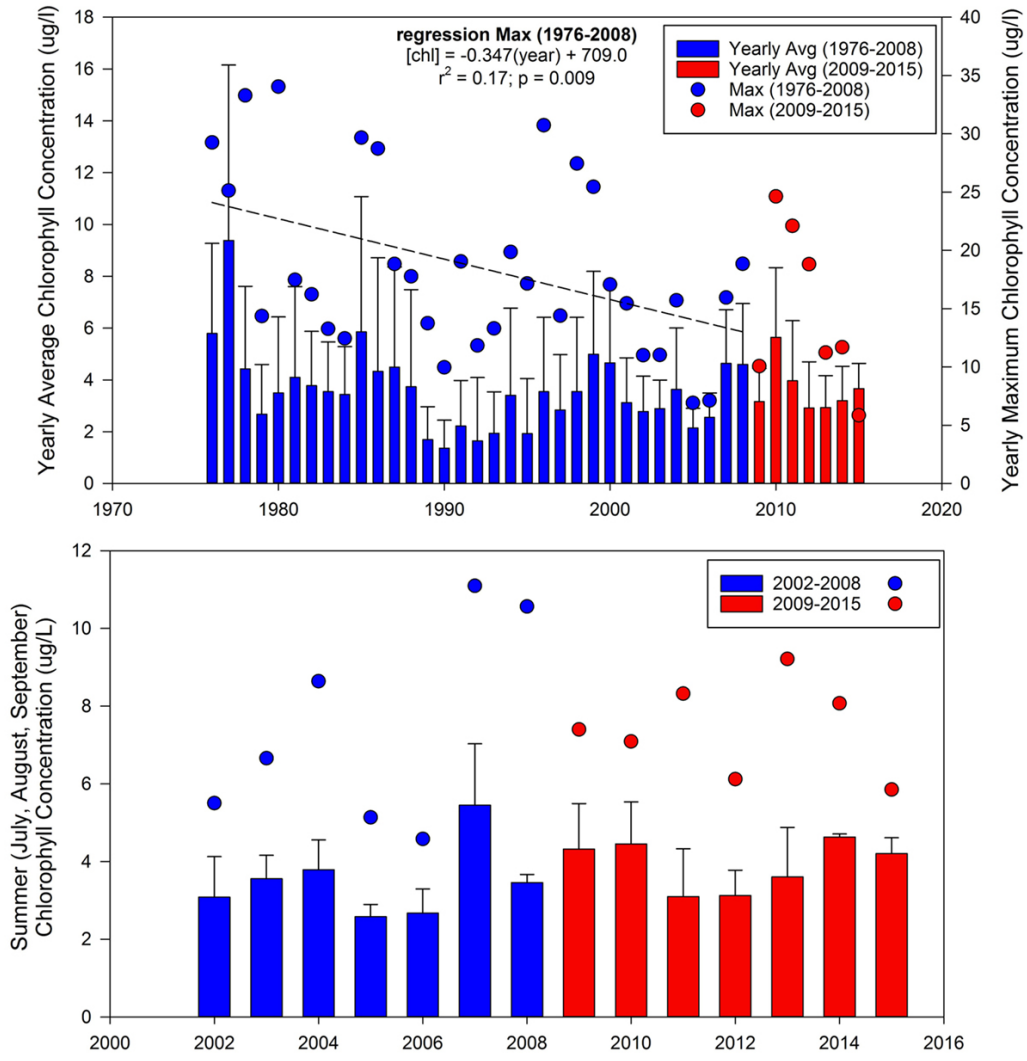


Figure 8. GSO Dock (MERL) chlorophyll concentrations (µg/l). Top: Yearly mean (with standard deviations) and yearly maximum surface chlorophyll concentrations (circles). A linear regression was run for yearly maximum concentrations (1976 to 2008; frozen samples) and found to be statistically significant (dotted line). Bottom: Summer (July–September) mean (with standard deviations) and maximum surface chlorophyll concentrations for 2002 to 2015. A linear regression was run and not found to be statistically significant. Blue indicates that samples were frozen prior to analysis. Red indicates that samples were immediately extracted and analyzed. No correction factor was available for these data (Oviatt, University of Rhode Island, personal communication).

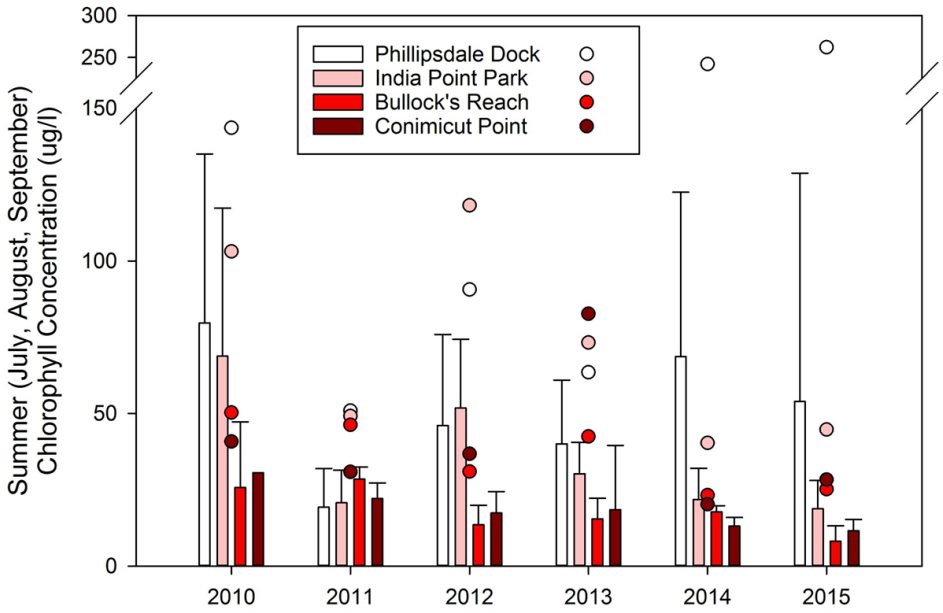


Figure 9. Mean summer (July-September) chlorophyll concentrations (µg/l) (with standard deviations) and maximum surface chlorophyll concentration (circles) from 2010 to 2015. Data were from the Narragansett Bay Commission (NBC) for the Providence River Estuary. Sites are listed from north (white) to south (dark red). All samples were immediately extracted and analyzed.

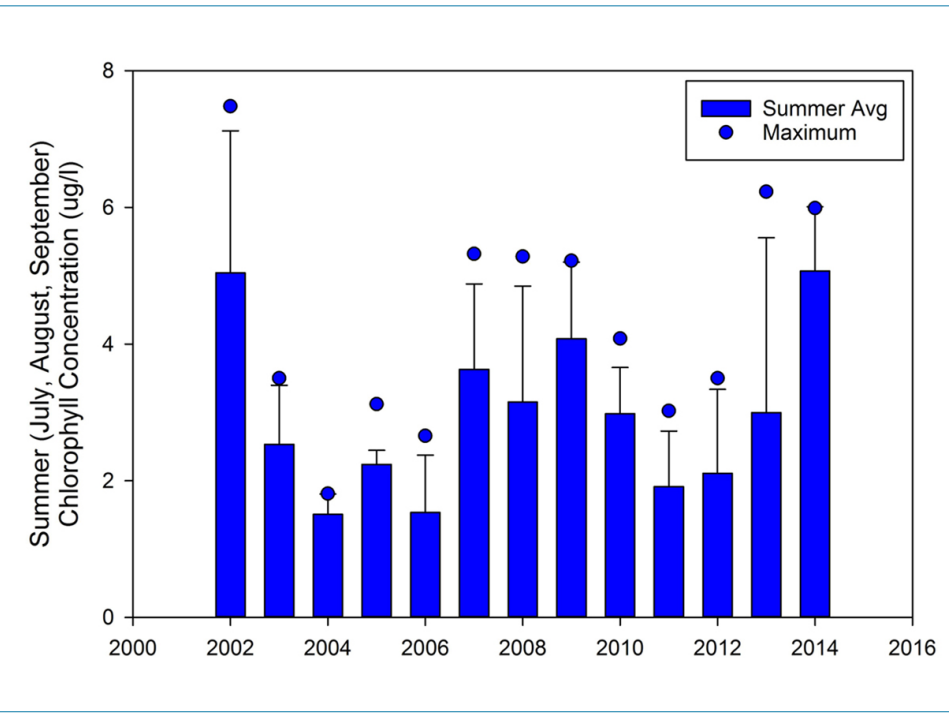


Figure 10. Mean summer (July-September) surface chlorophyll concentrations (µg/l) with standard deviations (bars) and maximum surface chlorophyll concentration (circles) from 2002 to 2014. Data are from the Narragansett Bay National Estuarine Research Reserve (NBNERR) for T-Wharf in the East Passage. All samples were frozen prior to extraction.

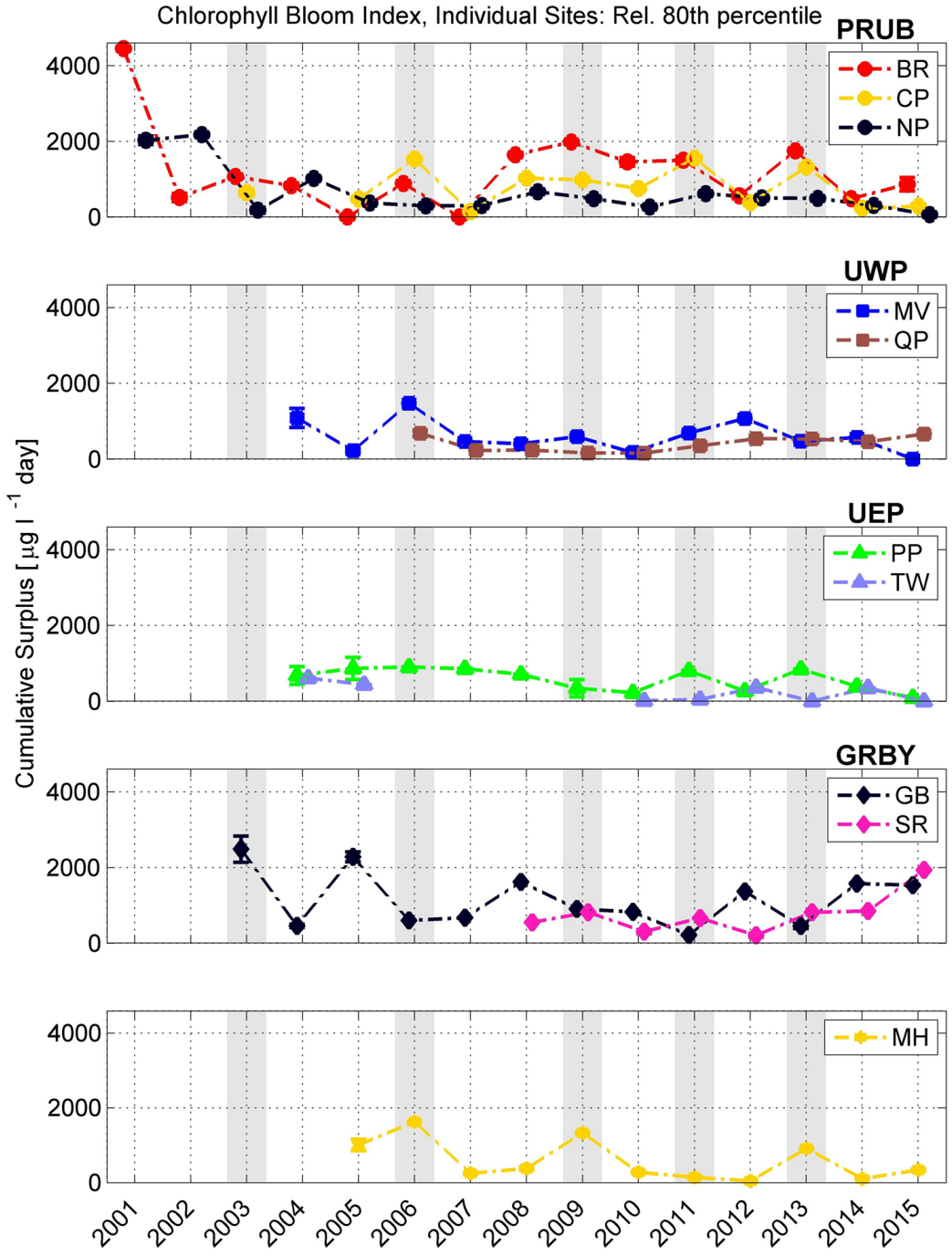


Figure 11. Chlorophyll Bloom Index for individual sites, relative to site-specific thresholds. Gray vertical shading indicates wet years (see Methods). To improve clarity by reducing overlap of lines and symbols, symbols from each site are systematically offset a small distance horizontally relative to those of other sites in the frame. Uncertainties are due to gaps in the time series due to sensor malfunction or data not meeting quality standards. In most cases, the uncertainties are smaller than the symbol size.

River Estuary, the Phillipsdale Dock (NBC) summer maximum decreased from 2010 to 2011 and then increased until 2015 (Figure 9). The variability of chlorophyll concentrations in this area was most likely due to both the tidally restricted and unique circulation patterns of the Seekonk River and the degraded nature of this portion of the Bay. At India Point, summer maximum peaked in 2012/2013 and then decreased until 2014 (Figure 9). Summer maximums at both Bullock's Reach and Conimicut Point increased from 2012 to 2013, followed by a decrease to 2014/2015 (Figure 9). A linear regression was run on all sites, and Bullock's Reach and Conimicut Point were found to have significant declines (Figure 6). Given high interannual variability, continued and more frequent sampling is warranted to assess trends over time. The T-Wharf data appear to be variable, and no definitive trends can be determined (Figure 10).

CHLOROPHYLL BLOOM INDEX

The Chlorophyll Bloom Index results are, by definition, a measure of bloom events relative to spatially varying thresholds. They consist of individual-site values (Figure 11) and multi-site means for geographic sections of the Bay (Figure 12). The Estuary Program used data from 2013 to 2015 to represent status and the entire dataset to determine trends.

Results of the Chlorophyll Bloom Index, while preliminary, show a persistent decreasing down-Bay gradient. Geographic variations in the site-specific thresholds are representative of the long-term mean chlorophyll levels (Table 1). The highest threshold values (about 15 to 29 µg/L) occurred in Greenwich Bay and the Providence River Estuary. Intermediate values (about 13 to 15 µg/L) occurred at the Mount Hope Bay site and the northern sites in the Upper West Passage (MV) and Upper East Passage (PP). The lowest values (about 5 to 9 µg/L) occurred at the southern sites in the upper West Passage (QP) and upper East Passage (TW).

The results from 2013 to 2015 showed pronounced interannual and inter-site variability without discernible coherent temporal or spatial structure or patterns (Figures 11 and 12). Index values were largest for Greenwich Bay, ranging from about 700 to 2000 µg/l day, and the Providence River Estuary and Upper Bay, ranging from about 0 to 1800 µg/l day. As noted above, those two sites had the largest threshold values as well. The Upper West Passage, Upper East Passage, and Mount Hope Bay values

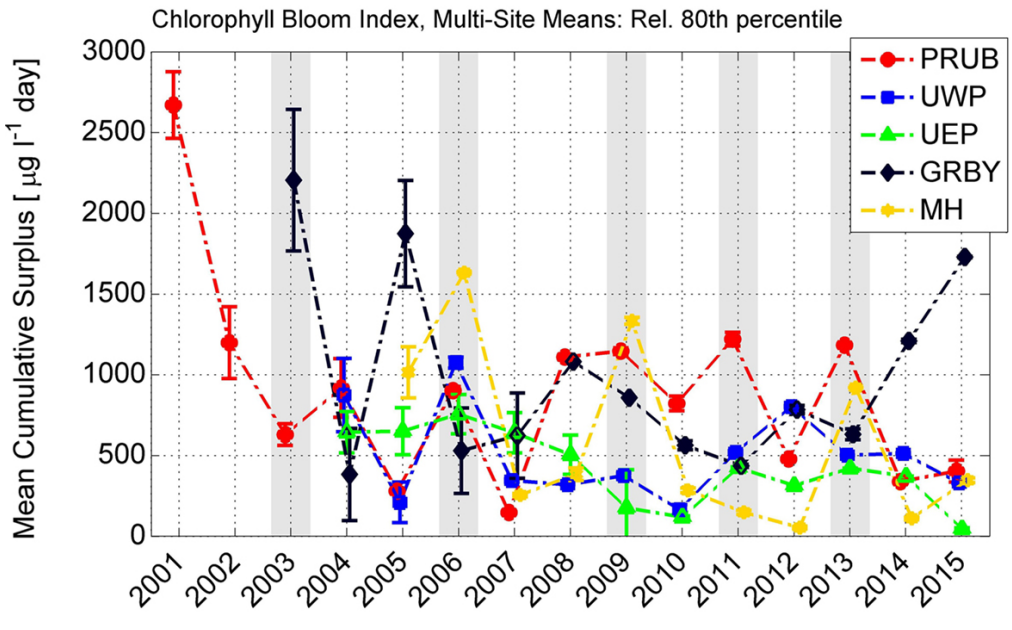


Figure 12. Chlorophyll Bloom Index averaged over multiple sites, relative to site-specific 80th percentile thresholds (Table 1). Shown as in Figure 11. The Mount Hope (MH) curve is a single site, repeated from Figure 11.

were not substantially dissimilar in magnitude from each other, each in the range of about 0 to 500 µg/l day.

The Index data for 2013 to 2015 included one year that was wetter than average (2013) and two that were relatively dry (2014 and 2015) (Table 3). There was no strong or spatially consistent relationship between those conditions and the Index during those three years. In the Providence River Estuary, Upper Bay, and Mount Hope Bay, the Index was higher during the wet year. Greenwich Bay was higher in the two drier years, and in the other Bay sections the Index did not differ strongly among wet and dry years. The main pattern for chlorophyll during 2013 to 2015 was a north-to-south gradient with higher values in the Providence River Estuary, Upper Bay, and Greenwich Bay than in the southern Bay sections.

The Estuary Program examined long-term trends by considering all years (from eight to fifteen years, depending on the site; Figures 11 and 12). Long-term trends were not prominent or easily identified. This was in part due to the interannual variability, which was not spatially coherent and was larger than long-term changes that may be present. Some evidence exists that relatively higher Index values occurred prior to 2005 at multiple sites compared to subsequent years (Figure 11). However, because the records from 2005 and earlier included substantially fewer sites, and the uncertainties were larger due to longer missing data gaps in those years (Table 2), there was little support to conclude that a long-term shift occurred. There does appear to be a recent (after 2013) short-term decline in the Index for the Providence River-Upper Bay, with Bullock's Reach and Conimicut Point mostly responsible (Figures 11 and 12).

The Mount Hope Bay record is most suggestive, albeit weakly, of a possible long-term change. The peak values, generally in the wetter years, decreased consistently from 2005 to 2015. The values during drier years also generally decreased throughout the record, with the exception of the final two years. The magnitude of this potential trend is at most comparable in magnitude to the range of interannual variability.

SPATIAL SURVEYS

In the Spatial Surveys, chlorophyll data were collected infrequently–only five to six days during the summer months. However, similar to the grab samples, the Spatial Surveys data can contribute to understanding of chlorophyll in Narragansett Bay. The 2015 Spatial Surveys had relatively low freshwater flux and high salinity, and showed that mean and maximum chlorophyll concentrations and their variability decreased from north to south in the Bay with the exception of local embayments and coves, which had higher concentrations than their latitude would suggest (Figure 13). The Seekonk River and Providence River sections had the highest average and maxima (24.1 and 46.5 µg/l) followed by Greenwich Bay (18.6 and 32.7 µg/l), the Upper Bay (9.5 and 16.4 µg/l), and the Lower Bay (9.5 and 20.3 µg/l). The Upper and Lower Bay were virtually the same in 2015. Using the 80th percentile as the threshold, bloom conditions occurred in the Providence and Seekonk River section (both average and maxima) and Greenwich Bay maxima. The maxima in the Lower Bay were typically lower than the average for the Seekonk-Providence River and Greenwich Bay areas. The high maxima in the Lower Bay were in Bristol Harbor.

The long-term (2005 to 2015) latitudinal gradients are illustrated by the mean and maximum chlorophyll concentration for each site (Figure 14). The overall gradient was similar to the 2015 results (Figure 13) but with higher maxima, which reflected extreme bloom years such as 2009. Comparison of 2015 with the long-term averages and maxima (Table 5; Figures 13 and 14) showed that the 2015 chlorophyll concentrations were lower with the largest changes occurring in the Providence River Estuary and Upper Bay sections. The average chlorophyll concentration in the Upper Bay decreased over 40 percent from the long-term average, and the average minima decreased over 70 percent. The average maxima were lower by half, except in the Lower Bay (Table 5). Overall, the recent chlorophyll concentrations were much lower than the long-term averages and reflected a very dry year as well as reduced nutrient loading from wastewater treatment facilities (see Discussion below).

Although the 2015 means and all-survey means (Figures 13 and 14) show large-scale gradients in these sections of Bay, they do not capture the temporal variability and trends of chlorophyll. To evaluate these patterns, the Estuary Program mapped the surface chlorophyll concentrations and calculated the area of each chlorophyll class for all surveys. The total area covered by the Spatial Surveys was approximately 150 square kilometers (58 square miles) in the northeastern sections of the Bay (Figure 4). The percentage of that area that had chlorophyll >14 µg/l (50th percentile) and the percentage with >27.5 µg/l (80th percentile) in each survey are shown in Figure 15 (A and B, respectively).

The percentage area plots exhibit significant intra- and interannual variability as well as distinct chlorophyll events (i.e., blooms) (Figure 15). The

percentage area of the Spatial Surveys with >27.5 µg/l ranged from less than one percent to 45 percent, while the percentage area with >14 µg/l ranged from two percent to 97 percent. With the exception of six bloom events–in which more than 25 percent of the area was above 27.5 µg/l (Figure 15, B)–the area with surface chlorophyll >27.5 µg/l ranged from about five to six percent in 2005 to 2007, increased in 2008 and 2009 to about fifteen percent, and then decreased to less than five percent in 2015. Both the intra- and interannual variability in chlorophyll appeared to be linked to precipitation and river discharge. However, further evaluation of the conditions that existed at the time(s) the Spatial Surveys were conducted is needed. The longer-term decrease was largely coincident with drier years and decreased wastewater treatment facility nutrient discharges to the Bay. This longer-term trend in decreasing chlorophyll from 2009 parallels the changes in hypoxic area of the Spatial Surveys (see "Dissolved Oxygen" chapter).

To illustrate the variability in chlorophyll concentrations between wet and dry years, the Estuary Program also compared maps of surface chlorophyll that are representative of an extreme bloom event from a wet year (August 4, 2009) with a more recent

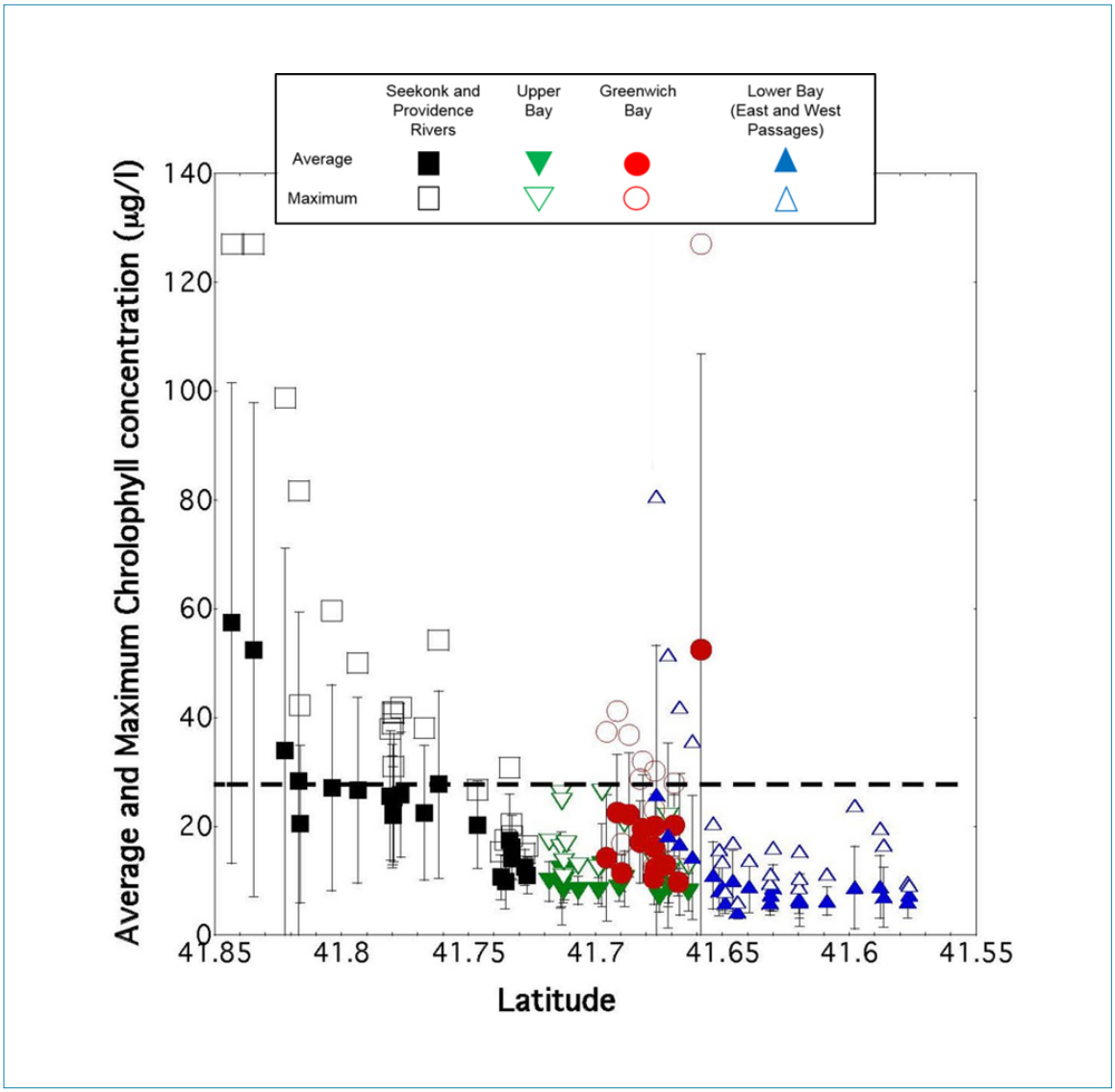


Figure 13. Average surface and maximum chlorophyll concentrations (regardless of depth) (µg/l) in 2015, collected June through September, for the Providence River Estuary (black squares), Upper Bay (green triangles), Greenwich Bay (red circles), and Lower Bay (blue triangles). Black dashed line is the 80th percentile: 27.5 µg/l. See Figure 4 for the spatial distribution of the sample sites.

Table 5. Comparison of 2015 and long-term (2005 to 2015) average surface chlorophyll and average maximum chlorophyll concentrations for sections of the Bay. All units are µg/l. See Prell and colleagues (2016) for original data.

Year	Seekonk and Providence Rivers		Upper Bay		Greenwich Bay		Lower Bay	
	Avg	*Max*	*Avg*	*Max*	*Avg*	*Max*	*Avg*	*Max*
2015	24.1	46.5	9.6	16.4	18.6	32.7	9.5	20.3
2005–2015	29.1	107.0	16.7	57.0	22	80.0	11.6	33.8
% Decrease	17	57	43	71	15	59	18	40

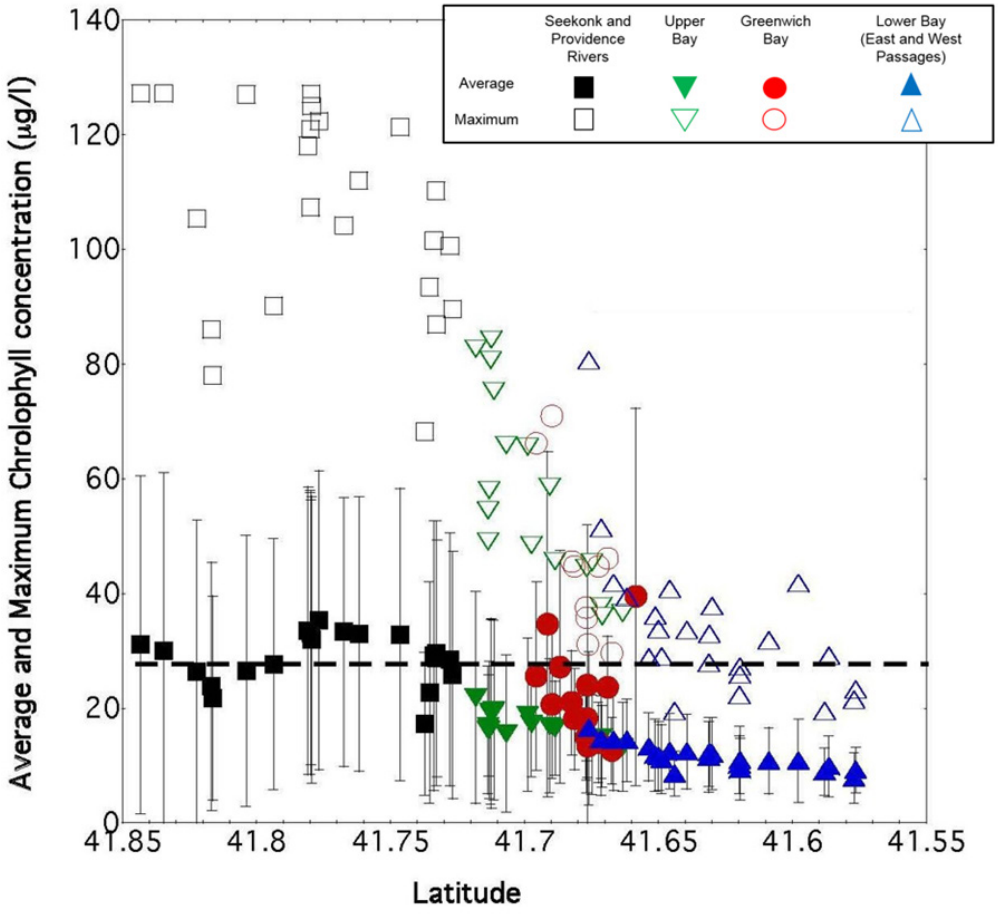


Figure 14. Average and maximum chlorophyll concentrations (µg/l) for the Providence River Estuary (black squares), Upper Bay (green triangles), Greenwich Bay (red circles), and Lower Bay (blue triangles) for all surveys from 2005 to 2015. Black dashed line is the 80th percentile: 27.5 µg/l. See Figure 4 for the spatial distribution of the sample sites.

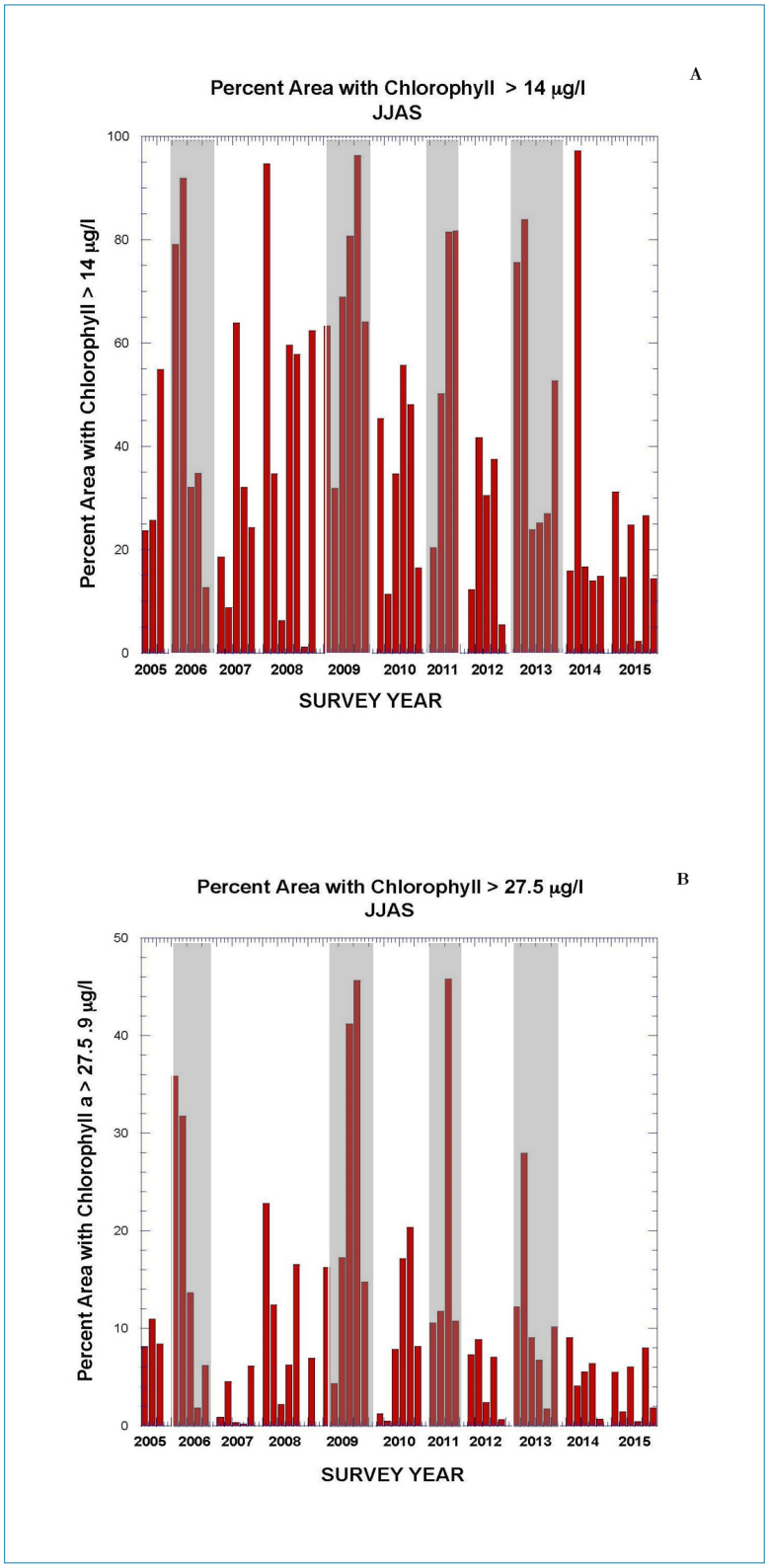


Figure 15. Percent of survey area (150 square kilometers; 58 square miles) with chlorophyll >14 µg/l (50th percentile) (A) and >27.5 µg/l (80th percentile) (B) in June, July, August, and September (JJAS) for all survey years from 2005 to 2015. Each bar represents a single survey conducted during a sample year. Shading designates wet years. See Prell and colleagues (2016) for original data.

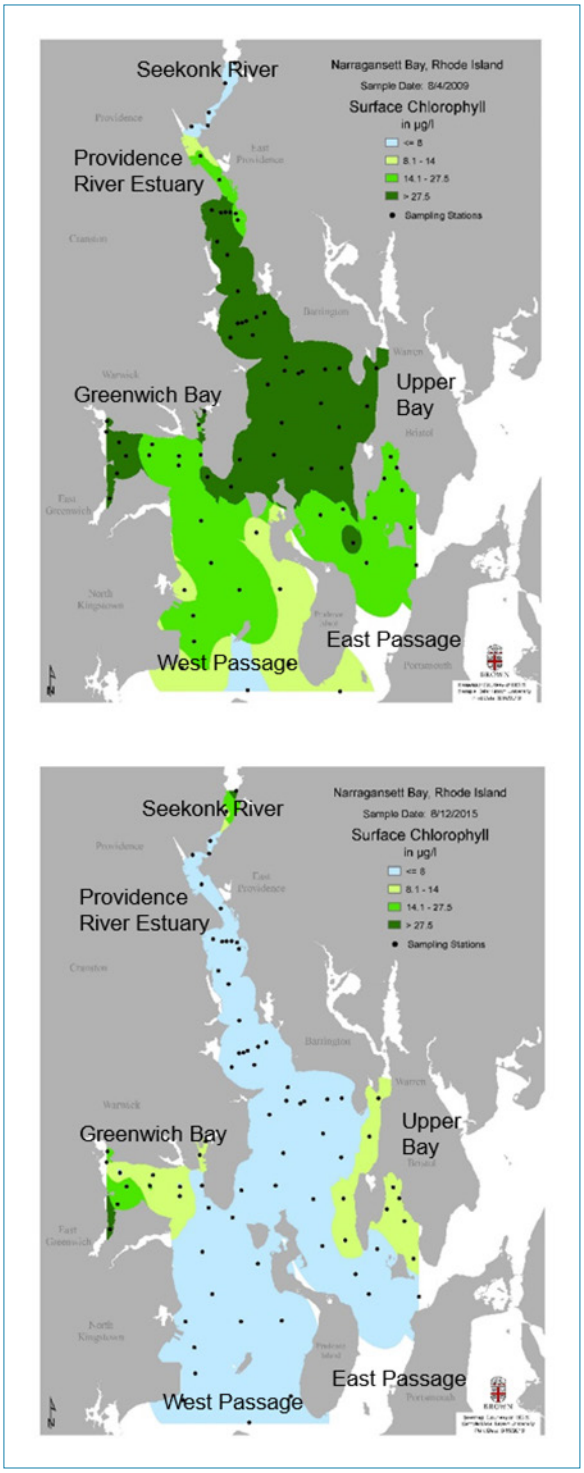


Figure 16. Comparison of surface chlorophyll concentration (µg/l) in a wet year (top: August 4, 2009) and a dry year (bottom: August 12, 2015).

low-chlorophyll event during a dry year (August 12, 2015) (Figure 16). During the 2009 extreme bloom event, chlorophyll was highest in the Providence River Estuary, Greenwich Bay, and the Upper Bay, but greater-than-average chlorophyll extended down toward the East and West Passages. Lower chlorophyll was observed in the uppermost portions of the Providence River Estuary and the Seekonk River, which may have reflected the lower salinity. During the 2015 event, chlorophyll was lower overall across the Bay with high concentrations in Greenwich Bay.

Discussion

COMPARISON OF THE THREE METHODS

Analysis of all data revealed a clear spatial gradient in surface chlorophyll from high chlorophyll in the northern sections of the Bay to lower chlorophyll in the southern sections of the Bay (Tables 1 and 4; Figures 6, 12, 13, and 14). During bloom events, as noted from the Spatial Surveys and the Chlorophyll Bloom Index, the high chlorophyll extended from the Providence River Estuary and Greenwich Bay down to the West and East Passages (Figures 11 and 16). Grab sample results from 2015 showed greatly decreased surface water chlorophyll compared to the long-term averages (Figure 6; Table 5), and both the area of high chlorophyll waters (Spatial Surveys; Figure 15) and the Chlorophyll Bloom Index declined (Figure 12), particularly since 2010. Bloom events were highly variable, leading to inter- and interannual variations in chlorophyll concentration throughout the Bay.

Given the patchiness or small-scale spatial variability relative to temperature, salinity and dissolved oxygen, and the differences in sampling frequency and spatial distribution of sample stations, it is not unexpected that the detection of temporal trends differed among three methods. The Spatial Surveys, in the northern sections of the Bay, suggest a decline in the percentage area of summer chlorophyll concentrations >14.5 µg/L starting in 2009 (Figure 15, top). Only results from the Bullock's Reach and Conimicut Point grab samples showed a significant decline over their sample periods (2010 to 2015) (Figure 6). The Chlorophyll Bloom Index showed no consistent declines, except for a recent short-term decline, after 2013, noted at Bullock's Reach and Conimicut Point (Figure 11).

In addition to sampling frequency, the differing results can be attributed to where and how the data were collected. The chlorophyll time series measurements at the ten NBFSMN sites used in the Chlorophyll Bloom Index were collected mostly in relatively deeper areas of the Bay and relatively far from coastlines, while the Spatial Surveys data included more sample sites closer to the shore. The Spatial Surveys provide some additional information in certain shallow waters (e.g., Greenwich Bay), but data from coves and embayments in other areas of the Bay are needed. This may explain, in part, why the Spatial Surveys may yield different results from the Chlorophyll Bloom Index. Therefore, the Spatial Surveys provide a strong complementary metric to the Chlorophyll Bloom Index.

The three methods appear to be complementary, as the monitoring of bloom-like conditions requires the spatial and temporal resolutions found in all three methods. The Chlorophyll Bloom Index, using NBFSMN data, provides high temporal resolution (data collected at fifteen-minute intervals, May to October), but low spatial resolution (just ten stations throughout the Bay). The Spatial Surveys have extremely high spatial resolution (77 stations), including shallow areas and embayments, but limited temporal resolution (five to seven sample dates between June and September). The grab samples have good temporal resolution (weekly or monthly samples throughout the year), but poor spatial resolution.

HOW STRESSORS IMPACT CHLOROPHYLL CONCENTRATION

Certain stressors impact chlorophyll concentrations at different times of the year. During summer, precipitation levels (as they refer to river nutrient loading, see "Precipitation" chapter) play a role in the interannual variability of chlorophyll levels, particularly in the northern sections of Bay. Reduction in wastewater treatment facility nutrient loading may also influence chlorophyll levels during summer. Interactions between the two stressors–temperature and wastewater treatment facility nutrient loadings–affect winter-spring bloom dynamics.

Summer Chlorophyll Levels: Summer Precipitation

Both the Chlorophyll Bloom Index and the Spatial Surveys showed areas of high chlorophyll concentration in wet years. In the Chlorophyll Bloom Index, there is evidence indicating that higher Index values tend to occur during wet years, particularly for the Mount Hope Bay site and to some extent in the Providence River Estuary and Upper Bay sections, but

Table 6. Summary statistics for chlorophyll concentrations and water properties for each spatial survey year. JJAS: June, July, August, September; JAS: July, August, September. Wet years are shaded.

YEAR	River Flux Departure m^3/s JJAS	Average Surface Salinity $^0/_{00}$	Average Surface Temp. °C	% Area with Bottom DO < 2.9 mg/l JAS	% Area with [Chl]. >27.5 µg/l JJAS	% Area with [Chl]. >14 µg/l JJAS
2005	-10	28.0	23.2	5.3	9.2	34.8
2006	26	23.7	22.0	27.1	13.4	50.1
2007	-13	27.7	21.1	2.9	2.2	29.6
2008	-4	26.5	23.1	17.6	7.4	45.3
2009	44	24.3	21.6	29.8	24.6	67.5
2010	-17	27.6	24.2	17.4	10.8	35.3
2011	25	25.3	23.6	13.7	22.8	58.4
2012	-5	27.6	23.7	12.2	4.8	25.5
2013	23	23.9	23.0	18.0	11.1	48.1
2014	-17	27.9	23.6	3.7	5.2	31.8
2015	-8	28.4	23.4	5.6	3.6	19.0

there are many exceptions (Figures 11 and 12). Similarly, in the Spatial Surveys the area of high chlorophyll concentrations in surface water correlated with the freshwater flux in wet years (Figures 17 and 18). The average area of high chlorophyll concentrations in surface water (>27.5 µg/l) increased threefold between the dry and wet years (from approximately six percent to approximately eighteen percent), while the area of higher-than-average (>14 µg/l) chlorophyll increased from 31 percent to 56 percent (Table 6). Although further analysis and interpretation of trends are required, the relationship between chlorophyll blooms and freshwater flux is evident.

Regarding the Spatial Surveys data and the relationship between hypoxia and chlorophyll, there is a strong correlation between the area of high surface chlorophyll concentrations and areas of bottom water hypoxia (Table 6, Figure 18). This relationship with hypoxia is less evident in analysis of NBFSMN chlorophyll and dissolved oxygen time series (Codiga et al. 2009).

Summer Chlorophyll Levels: Reductions to Wastewater Treatment Facility Loading

Rhode Island set a summer (May to October) nitrogen reduction goal of 50 percent (relative to 1995/1996 loadings) for eleven of its wastewater treatment facilities (WWTFs) impacting the Providence River, Seekonk River, and Upper Narragansett Bay (see "Nutrient Loading" chapter). This Rhode Island goal was met for the first time during the summer of 2012. Massachusetts WWTFs impacting these waters also achieved significant reductions in this time period. In summer 2013, the first full season following this

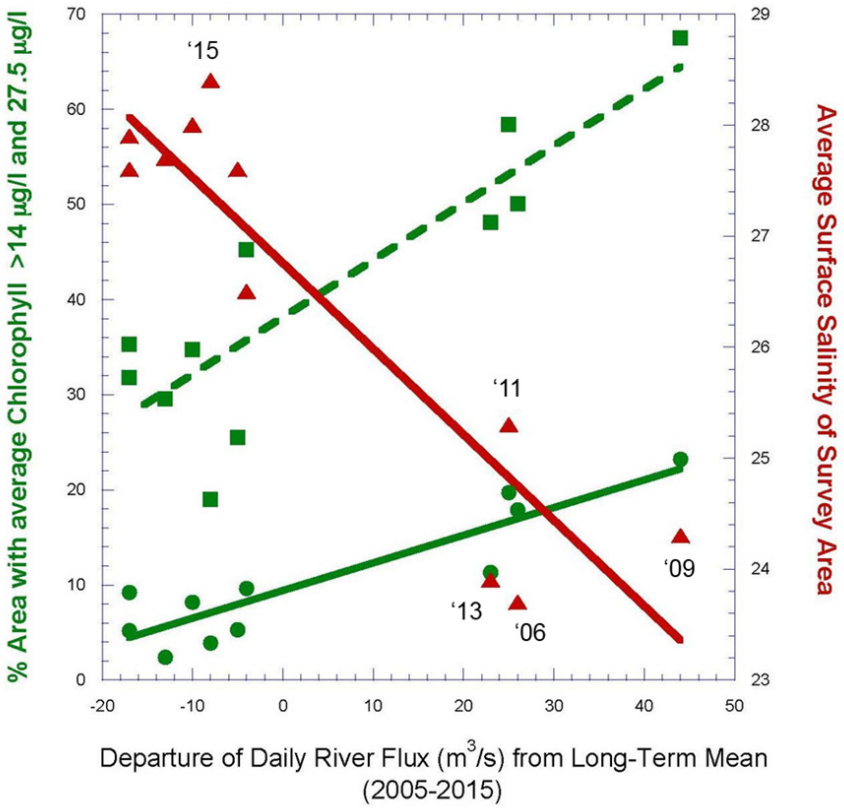


Figure 17. Percentage of the survey area with average chlorophyll >14 µg/l (50th percentile) (filled squares) and >27.5 µg/l (80th percentile) (filled circles) as a function of the departure of daily river flux (m^3/s) from the long-term mean (2005 to 2015). Regressions were run for each: Chl >27.5 = 0.3x + 9.4; r^2 = 0.81; Chl>14 = 0.6x + 38.1; r^2 = 0.75. Red triangles represent average surface salinity of survey area. The numbers above and below the red triangles indicate the wet years: 2006, 2009, 2011, and 2013. Status year (2015) is also indicated.

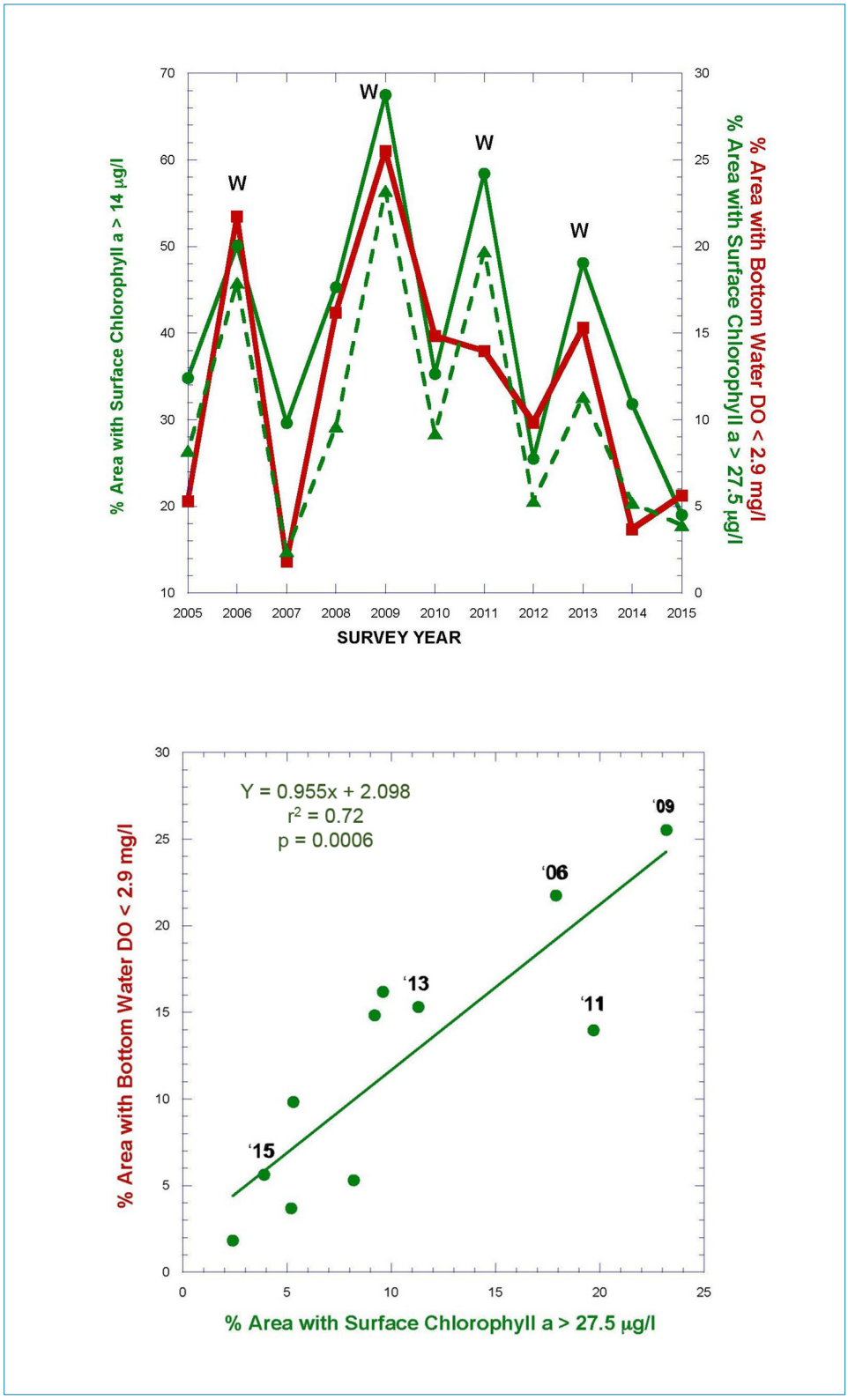


Figure 18. Top: Percentage of survey area with surface chlorophyll concentrations >14 µg/l (50th percentile) (green circles) and >27.5 µg/l (80th percentile) (green triangles) as a function of survey year. Red squares indicate percentage of area with bottom water dissolved oxygen (DO) concentrations <2.9 mg/L. "W" indicates wet years. Bottom: Percentage of survey area with bottom water DO concentrations <2.9 mg/l as a function of percentage of area with surface chlorophyll concentrations >27.5 µg/l. Status year (2015) and wet years are indicated.

achievement, the Chlorophyll Bloom Index and Spatial Survey results were relatively high and not strongly distinguishable from earlier wet years that also had higher values (e.g., 2006) (Figures 12 and 15). There was a decline in the Index and bloom area from the Spatial Surveys in 2014 and 2015, both dry years, but again these levels were noted in dry years previous to achievement of the nutrient loading reduction goal (e.g., 2007). The grab samples also do not reveal a consistent declining trend in chlorophyll in recent years. As previously noted, there is considerable year-to-year variability in the chlorophyll concentration data at all sample stations in the Providence River Estuary, Seekonk River, and Upper Narragansett Bay, related to precipitation and other factors. Continued monitoring is required before the impact of nutrient reductions on chlorophyll concentration can be determined with statistical certainty.

In 2008, the Narragansett Bay Commission opened a large tunnel designed to capture heavy precipitation events that normally would flood the Field's Point wastewater treatment facility, causing minimally treated or untreated sewage to flow into the Bay (NBC 2017). The reduction of nutrient loading due to the capture of combined sewer overflows (CSO) to this tunnel could also impact chlorophyll concentrations. The use of the tunnel is triggered by large precipitation events, particularly those in very wet years. Since the tunnel opened, there have been three wet years: 2009, 2010, and 2013. During those years, all three methods showed chlorophyll concentrations similar to concentrations from the wet year 2006, before the tunnel opened. An analysis is needed to evaluate the relationship between individual events when the tunnel was used and Bay chlorophyll concentrations.

Winter-Spring Bloom Considerations: Temperature and Nutrients

The Massachusetts and Rhode Island WWTFs, which are required to reduce nitrogen loading 50 percent from 1995/96 loadings during the summer (see above and "Nutrient Loading" chapter), were also required to operate nitrogen removal processes during the remainder of the year. The Rhode Island Department of Environmental Management (RIDEM) evaluated the January, February, and March loads

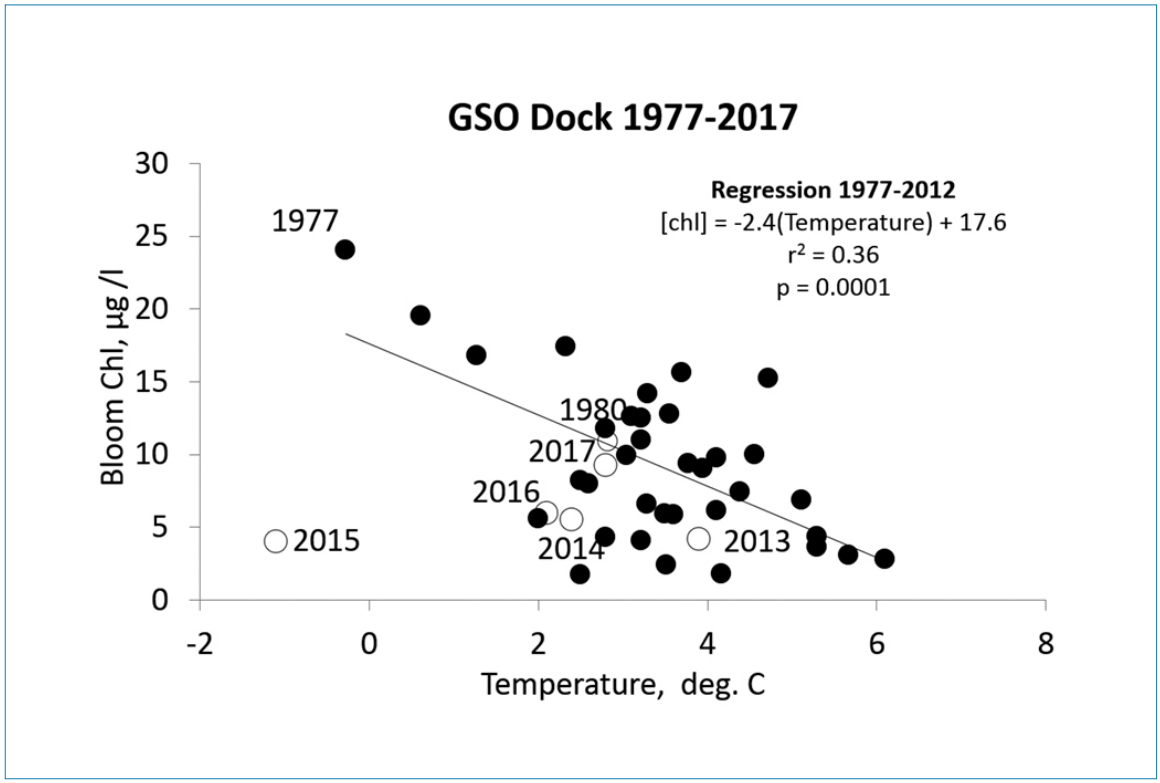


Figure 19. Winter-spring chlorophyll (mean three weeks of the bloom) plotted versus water temperature during the month of the bloom from 1977 to 2017 at the GSO Dock (weekly samples). Closed circles represent data between 1980 (pre-nitrogen reduction conditions) and 2013 (after the 50 percent nitrogen reduction goal was met). Open circles depict 1980, and 2013 through 2017. (see text).

from nine of the eleven WWTFs and from three large WWTFs in Massachusetts and determined that the winter of 2012 (January, February, and March) nitrogen loads were 33 percent lower than the loads seen in the same months in 2011 (Travers, RIDEM, personal communication). Thus, winter bloom chlorophyll data were examined before, during, and after the winter nutrient reduction. These data were combined with Bay-wide nutrient survey data (Oviatt, University of Rhode Island, personal communication) to ascertain changes in the strength of the winter bloom in conjunction with decreases in nitrogen.

The winter-spring diatom bloom has always been highly variable in Narragansett Bay. Usually, low water temperatures during cold winters reduce copepod grazing, allowing the biomass of the bloom to develop (Keller et al. 1999, Oviatt et al. 2002). Cold winters typically occurred during negative North Atlantic Oscillation (NAO) periods, with intense and widespread winter-spring blooms in the mid-twentieth century (Pratt 1965). By contrast, in warm winters, no bloom or a bloom of reduced temporal and spatial intensity may occur. Warm winters typically occurred during positive NAO periods in the late twentieth century (Keller et al. 1999, Oviatt et al. 2002) but with climate change have been more prevalent during both positive and negative NAO periods. There have been some recent cold winters during negative NAO periods with a return of winter-spring blooms in 2008, 2010, and 2011. In data from the GSO Dock, usually but not always the bloom fails to develop when water temperatures remain at 3°C or higher (Figure 19). At the warmer winter temperatures, a mesocosm experiment has shown increased biomass and grazing activity by zooplankton (Keller et al. 1999). Field data in Narragansett Bay also show a greater biomass of zooplankton in warm winters. Of course, many other factors may cause the failure of the bloom, such as a change in water mass, storm mixing, low light conditions, and low nutrient conditions.

During winter, WWTFs in Rhode Island and Massachusetts are required to continue to operate their nitrogen reduction infrastructure, but they do not have to add chemicals that aid in the nitrogen removal process during the winter period. Winter nitrogen reductions vary based on many factors, including maintenance procedures at the plant and temperature. Cold temperatures limit the WWTF biological treatment process, making it less effective

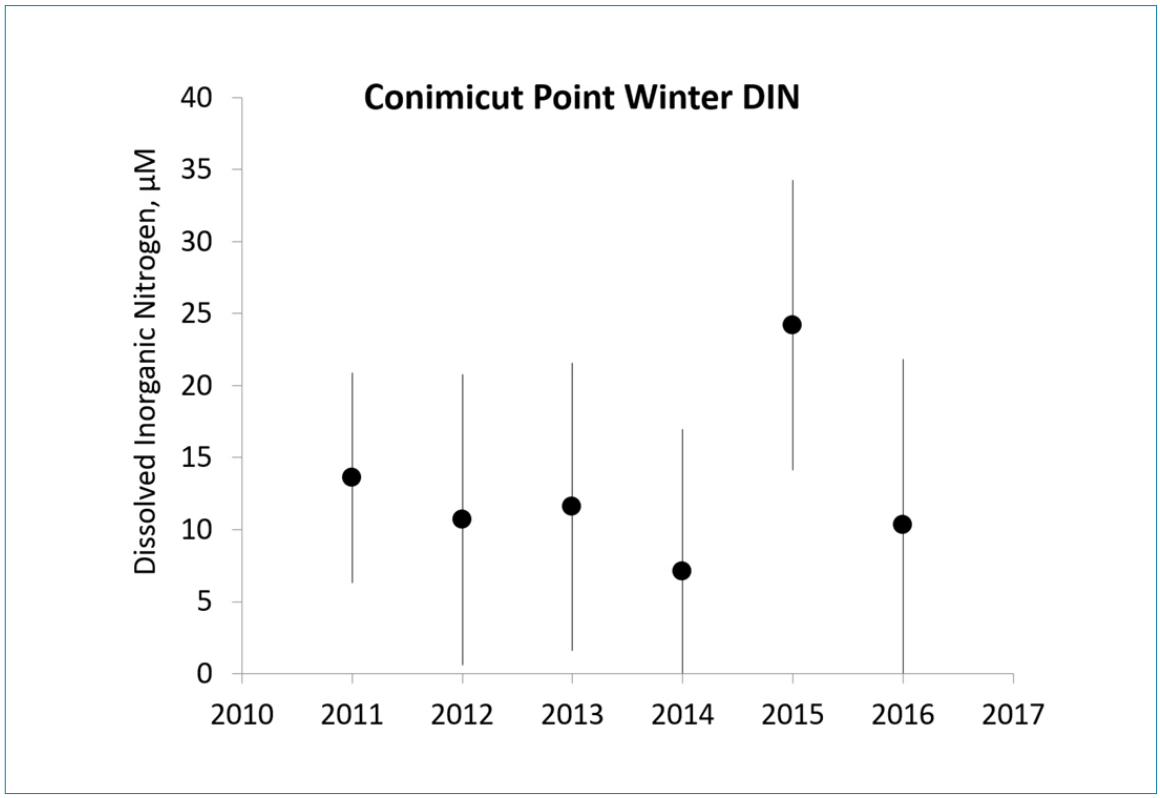


Figure 20. Conimicut Point winter dissolved inorganic nitrogen (DIN) concentration mean plus/minus one standard deviation from before nutrient reduction 2011 to 2012 and after 2013 to 2016.

in winter and releasing more nutrients during the period of the winter-spring bloom. During winter 2014, after improvements of the WWTFs, dissolved inorganic nitrogen at Conimicut Point reached the lowest value. Nitrogen increased at Conimicut Point in 2015, although no bloom occurred in cold, low-nitrogen 2014 or surprisingly, cold, higher-nitrogen 2015 (Figures 19 and 20). The quantity of nitrogen released by the WWTFs to the Upper Bay was essentially unchanged between 2014 and 2015, suggesting that the lack of a bloom (i.e., lack of nitrogen uptake by phytoplankton), and not the WWTFs, may have been responsible for the increase in dissolved inorganic nitrogen (DIN) measured at Conimicut Point between 2014 and 2015.

Stations from the northern Bay to the southern Bay show significant decreases in average winter chlorophyll after nutrient reduction using data from the Narragansett Bay Fixed Site Monitoring Network and Bay-wide surveys conducted by the Oviatt group in 1980 (Figure 21). Bay stations (Conimicut Point, T-Wharf on the southern end of Prudence Island, and the GSO dock) show significantly reduced chlorophyll concentrations both during and after nutrient reduction compared to before. Greenwich Bay, where nitrogen reduction has been less effective due to additional nutrient sources other than the WWTF, stands out in the Bay gradient as still having very intense blooms. Except for Greenwich Bay, very large blooms have been absent in the Bay since nutrient reduction was initiated. The effects of many variables including warmer winters and now reduced nitrogen have contributed to variable blooms.

CONCLUSION

In general, all three analytical approaches showed a north-to-south gradient of chlorophyll concentrations in Narragansett Bay, with the highest concentrations

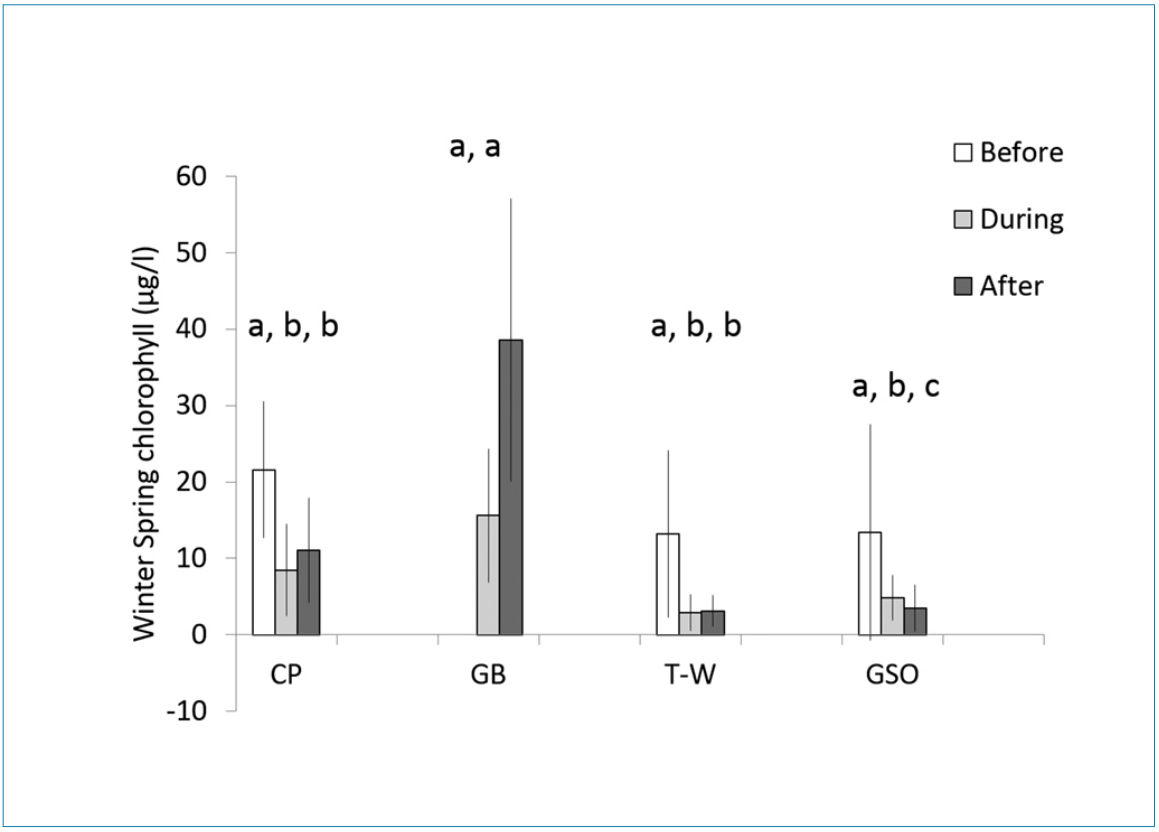


Figure 21. Impact of nitrogen reduction on the winter-spring bloom in Narragansett Bay. Winter-spring mean chlorophyll during January, February and March from before nitrogen reduction (1980) (no data for Greenwich Bay, GB); during reduction at Conimicut Point (CP, 2009 to 2012), GB (2007 to 2010 and 2012), T-Wharf (T-W, 2011), and GSO (2005 to 2012); and after reduction at all stations (2013 to 2016). The mean values include plus/minus one standard deviation of the means. The letters indicate ANOVA tests of significance on the daily values from January through March; different letters indicate significance levels greater than 0.0001; similar letters indicate no significant difference between time periods.

occurring in the Providence River Estuary (Figures 6, 12, and 15). Greenwich Bay also had high chlorophyll concentrations, which likely stem from local forcing factors—reduced circulation and localized non-point pollution sources. This spatial gradient is reflected in the threshold levels (80th percentile) relative to which the Chlorophyll Bloom Index and Spatial Surveys were defined (Figures 12 and 15). The winter-spring chlorophyll concentrations also show this same north-to-south gradient after nutrient reductions (Figure 21).

All approaches showed significant interannual variability and no discernible longer-term trends in chlorophyll levels from 2001 to present. Some shorter-term reductions were noted in recent years, but these are not interpreted as trends. The Chlorophyll Bloom Index, the Spatial Surveys, and the Providence River Estuary grab samples did show interannual variability that can be linked with precipitation. During wet years (more precipitation than the median for 2005 to 2015), high chlorophyll concentrations (above the 80th percentile) were more frequent than in dry years (precipitation less than the median for 2005 to 2015).

Since wastewater treatment facility nutrient reductions were achieved in the summer of 2012, there has been much interest in how Narragansett Bay will respond. The summer grab samples show a decline in chlorophyll concentration since 2010 at Bullock's Reach and Conimicut Point (Figure 6), and the Spatial Surveys show overall declines in chlorophyll blooms (concentrations > 27.5 µg/l) since 2011 (Figure 15). The Chlorophyll Bloom Index did not show similar results. During the winter-spring, chlorophyll concentrations have been significantly reduced during the nutrient reduction period and after nutrient reduction goals were met as compared to 1980.

The complementary datasets can be used to tease apart the patterns noted here to further develop a reliable metric for chlorophyll. The grab samples have increased in spatial coverage from the mid-lower Bay to include the Providence River Estuary, but they have a medium temporal resolution, as they are collected weekly to monthly all year long. The NBFSMN data used in the Chlorophyll Bloom Index have high temporal resolution (data collected every fifteen minutes) but limited spatial resolution. The opposite is true of the Spatial Surveys, which have high spatial resolution and low temporal resolution (only collected five or six times per year). Further analysis of correlations among these datasets may expand the utility of each dataset to reveal connections among surface chlorophyll, nutrient loading, and bottom water dissolved oxygen concentrations, or the connection between precipitation, temperature, and chlorophyll. This information will then improve understanding of how chlorophyll will change in Narragansett Bay under future climate regimes and anthropogenic nutrient-loading scenarios.

Data Gaps and Research Needs

- Collection of additional chlorophyll data is needed in order to be representative of all major sub-regions of Narragansett Bay and improve the spatial resolution of existing datasets.
- High interannual variability makes it difficult to detect temporal trends in existing datasets. Synthesis studies are needed to further explore the different temporal and spatial scales of chlorophyll variability and their relationships to other influencing factors (e.g., sunlight, pH, and temperature) as well as the physical structure of the water column.
- Further analysis of the Chlorophyll Bloom Index is needed, including whether the 80th percentile fully encompasses the definition of a bloom, or if a second percentile should be added (such as the 20th percentile). Additionally, all three methods show high variability, and a sensitivity analysis should be done to reduce this variability.
- Analysis of changes in phytoplankton species composition and abundance over time is needed to understand how species composition impacts chlorophyll concentration trends. The results will also inform any monitoring or analysis for phytoplankton nuisance or harmful algal blooms. Species composition has been studied before (Windecker 2010), and the GSO Phytoplankton Survey and NBC continue to record species-specific information.
- Controlled mesocosm studies should be done to evaluate the response of the benthic community to increased water clarity and decreased phytoplankton production (i.e., decreased input of organic matter to the benthos). This would address how the ecosystem is responding to nutrient reductions and inform a discussion regarding an appropriate balance of nutrient levels and ecosystem response.

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Professor of Oceanography Emeritus, and David Murray, Lecturer, with Brown University. A section on the winter-spring bloom was written by Candace Oviatt, Director, and Heather Stoffel of the University of Rhode Island Marine Ecosystem Research Laboratory. The work described in the "winter-spring bloom" section was funded by the Coastal Hypoxia Program of NOAA grants NA05NOS4781201 and NA11NOS4780043 and contracts from the Rhode Island Department of Environmental Management, Office of Water. Angelo Liberti, Sue Kiernan, Heidi Travers, and David Borkman provided helpful comments.

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The threshold and duration for the Chlorophyll Bloom Index were developed by a chlorophyll subcommittee composed of Candace Oviatt (University of Rhode Island's Graduate School of Oceanography), Warren Prell and David Murray (Brown University), Dave Borkman and Susan Kiernan (Rhode Island Department of Environmental Management), and Heather Stoffel (University of Rhode Island and Rhode Island Department of Environmental Management).

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State of Narragansett Bay and Its Watershed
2017 Technical Report

Bay Ecosystem Condition Indicators

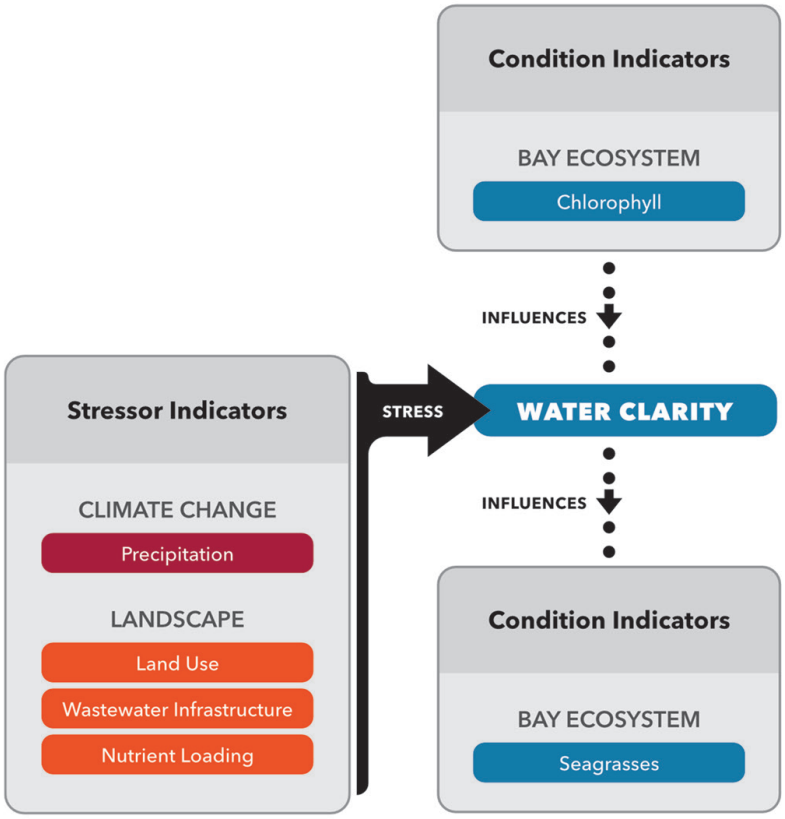
CHAPTER 17: WATER CLARITY

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Photo: Beavertail State Park, Jamestown, RI (Ayla Fox)

NARRAGANSETT BAY ESTUARY PROGRAM

Overview



BACKGROUND

- Water clarity is an indicator of how much light penetrates through the water to support growth of seagrasses, phytoplankton (chlorophyll), and macroalgae, which in turn provide food for fish, shellfish, and other animals. Presently, major stressors to water clarity include wastewater discharges and runoff of precipitation from land, which add sediment and nutrients to the water column and encourage excessive growth of phytoplankton and macroalgae, shading the water column and decreasing water clarity. Land use affects the runoff of precipitation and the amount of nutrient loading.

KEY FINDINGS

- ***Status:*** Data collected in 2014 showed that water clarity was greatest in Lower Narragansett Bay and declined into the Upper Bay. Overall, the water clarity was highest during the winter and decreased through the spring and summer.
- ***Trends:*** From 1972 to 1997, water clarity improved steadily at Fox Island in the Lower Bay, especially in the summer months, but data from 2004 to 2014 did not show any improvement. In the Bay's urbanized northern sections, water clarity data collected in recent years showed strong interannual variability with no discernible trends.

Introduction

Clarity is a water quality indicator used to measure how deep light can penetrate through the water column. Light is an important driver of photosynthesis and primary production. The amount of light available for photosynthesis is influenced by the concentration of suspended sediments, organic material, microorganisms, macroalgae, and phytoplankton present in the water column, collectively affecting the turbidity or clarity of the water (Vant 1990, Smith et al. 2006). Water clarity can fluctuate over the course of a year due to many factors, such as flooding, drought, seasonal winds, temperature, and pollution. For example, rainstorms carry sediment from land into Narragansett Bay, whereas drought reduces the delivery of sediment (Balch et al. 2016, Michigan Sea Grant 2016, Chesapeake Bay 2016, USEPA 2016).

In estuaries, light can become the limiting factor for primary production. Seagrasses and microphytobenthos (small phytoplankton that live on the sediment surface) are less likely to occur in light-limited waters (Morrison et al. 2006, Smith et al. 2006). Light levels and light-penetration depth may also alter the types of phytoplankton present in an estuary or change the production rates of the resident phytoplankton (Borkman and Smayda 2016).

Measurements of water clarity are useful for detecting anthropogenic impacts on the Bay from dredging, erosion, changes in land use, eutrophication, and other factors (Vant 1990, Hoyer et al. 2002, Smith et al. 2006). Precipitation that falls on land and then runs off into rivers carries sediment and other particles into coastal waters, increasing turbidity (USEPA 2016). Dredging and wastewater treatment facility effluent can also increase the total concentration of suspended solids in the water. Additionally, large inputs of nutrients from wastewater and nonpoint source runoff can stimulate excessive growth of epiphytic and free-floating macroalgae and phytoplankton, and this eutrophication can, in turn, cloud the water. When reductions occur in nutrient and sediment inputs, water clarity and water quality typically increases.

During the pre-colonial period to the age of industrialization (approximately 1650 to 1850), water clarity was presumably high in Narragansett Bay. Clear waters support seagrasses and oysters, both of which were found in the Providence River Estuary portion of the Bay (Nixon et al. 2008; see "Seagrasses" chapter). However, land clearing and deforestation led to a significant sediment input to rivers and the Bay (Foster et al. 1992, Roman et al. 2000, Nixon et al. 2008). While reforestation began around 1860 and has continued to the present, water clarity declined during the Industrial Revolution (Nixon et al. 2008). The advent of centralized wastewater collection and treatment, along with dredging of the shipping channel in the Providence River Estuary, added nutrients and particles to the water column (see "Nutrient Loading" chapter). These stressors contributed to the decline of water clarity and seagrasses throughout the Bay (see "Seagrasses" chapter).

Water clarity has been measured consistently since 1972 at one station (Fox Island) in the West Passage, using a Secchi disk. From 1972 to 1996, data from this station showed an increase in water clarity, attributed to reduced nutrient and suspended solid loading from improved wastewater treatment processing and a decline in phytoplankton and macroalgae (Borkman and Smayda 1998, 2016). In recent years, additional measurements of water clarity have been made throughout the Bay, particularly since the expansion of the Narragansett Bay Fixed Site Monitoring Network in 2005. By 2007, the Narragansett Bay Commission, Narragansett Bay National Estuarine Research Reserve, and University of Rhode Island (URI) Graduate School of Oceanography were taking water clarity measurements routinely along the length of the Bay from the northern end of the Providence River Estuary to the Lower West Passage.

In this chapter, the Narragansett Bay Estuary Program reports on status and trends of water clarity using all available data for Narragansett Bay from 1972 to 2014. The chapter also discusses how the new findings fit with the historical condition of water clarity and examines how the key stressors–precipitation, land use, and nutrient loading–may affect water clarity in the future.

Methods

Water clarity data used in this analysis were collected using two different methods: (1) Secchi disk readings and (2) underwater light-meter measurements of photosynthetically active radiation (PAR). Commonly used in fresh and estuarine waters, the Secchi disk is lowered through the water column by a rope or chain, and the depth at which the disk or disk definition is no longer visible is taken as a measure of the transparency of the water. Secchi disk readings do not provide an exact measure of water clarity, as there can be errors and subjectivity. In contrast, underwater light meters make it possible to precisely quantify the PAR available at a particular depth in the water column. Both types of measurements can be converted to light extinction coefficients, *k,* to provide a standard metric of water clarity (Figure 1). Clear water has a low *k*, and turbid water has a high *k*.

To compile the available data, the Estuary Program worked with many partners, including the Narragansett Bay Commission (NBC), Narragansett Bay National Estuarine Research Reserve (NBNERR), and University of Rhode Island's Marine Ecosystem Research Laboratory, which maintains the Narragansett Bay Fixed Site Monitoring Network (NBFSMN). Additionally, the Estuary Program accessed data through the University of Rhode Island's Graduate School of Oceanography (URI-GSO) phytoplankton surveys and NarrBay, an online data portal. Figure 2 summarizes the sampling methods and temporal coverage of each of the datasets. For this report, the Estuary Program analyzed data from 1972 to 2014. Spatial coverage of the datasets increased after 2007.

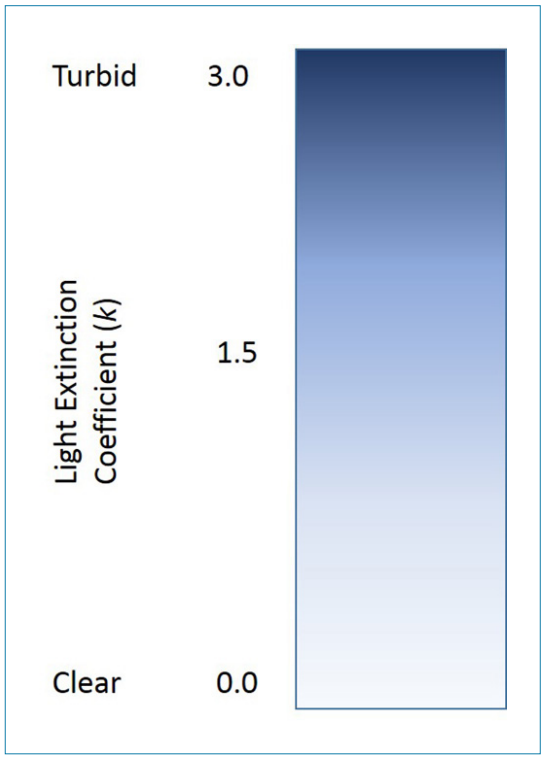


Figure 1. Light extinction coefficient (*k*) and how it relates to water clarity. As *k* decreases (becomes closer to zero), water becomes clearer (water clarity improves), and as *k* increases, water becomes more turbid (water clarity declines).

The Estuary Program used PAR data when available; otherwise, Secchi data were converted to *k*. To perform the conversion, the Estuary Program considered three potential approaches used or suggested by its partners: (1) an equation from Poole and Atkins (1929), (2) an equation from Cole (1989), and (3) a Narragansett Bay-specific equation based on NBC data for the Providence River Estuary. The Poole and Atkins (1929) equation performs best with Secchi data from naturally low-turbidity waters, such as those in the mid to lower sections of Narragansett Bay. The Cole (1989) equation was derived from data in San Francisco, a naturally turbid environment much like the Providence River Estuary and Upper Bay. A Narragansett Bay-specific equation has the potential to account for turbidity, like the Cole (1989) equation, and offers the advantage of being based on data collected in the Bay. Because of the differences in turbidity levels across the Bay, the Estuary Program chose to use two equations, one for the Mid and Lower Bay and one for the Upper Bay (Table 1).

To convert Secchi data from the Mid and Lower Bay, the Estuary Program used the equation derived from Poole and Atkins (1929):

$$k \text{ (per m)} = 1.7(\text{Secchi depth})^{-1}$$

For data from the Upper Bay, including Mount Hope Bay, the Estuary Program used a Narragansett Bay-specific equation derived from NBC's data:

$$k \text{ (per m)} = 1.178(\text{Secchi depth})^{-0.623}$$

Converting Secchi depth to *k* unavoidably introduces error through both the choice of equation and the results computation. For that reason, the Estuary Program used PAR *k* whenever possible to reduce error. In cases where conversion was necessary, the equation choice and computation had approximately 20 percent error. The Estuary Program aims to reduce this error in the future, either by focusing its

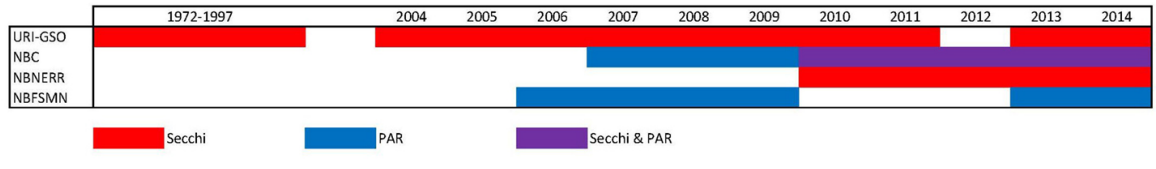


Figure 2. Temporal coverage of datasets obtained from all sources. Sampling methods were Secchi disk (red), photosynthetic active radiation (PAR, blue), or both (purple). *URI-GSO: University of Rhode Island Graduate School of Oceanography. NBC: Narragansett Bay Commission. NBNERR: Narragansett Bay National Estuarine Research Reserve. NBFSMN: Narragansett Bay Fixed Site Monitoring Network.*

efforts solely on PAR *k* or Secchi depth, or by deriving an improved equation when more data become available. Further information about the methods and decision-making process regarding Secchi depth conversion is available upon request.

To express *k* values as depth, the Estuary Program used the appropriate equation for the sample site and solved for Secchi depth. The conversion of *k* to depth also introduces an error of approximately 20 percent, and therefore the depths are provided only as a guide for readers unfamiliar with *k* values, not as an absolute measurement of water clarity.

The Estuary Program performed statistical analyses when appropriate. To analyze differences in water clarity between groups, the Estuary Program used a one-way analysis of variance (ANOVA) combined with a Holm-Sidak post-hoc test. The Holm-Sidak test is recommended as a conservative test to determine the means that are significantly different from each other. The Estuary Program performed linear regressions on multi-year data from individual sample sites when applicable. In all analyses, a p-value of 0.05 was used to determine significance.

To determine the recent status of water clarity, the Estuary Program used data from 2014, which was the most recent year for which a complete dataset was available (Figure 2). The Estuary Program examined the data by season from India Point in Providence (Upper Providence River) to Fox Island (Middle West Passage) and T-Wharf (Middle East Passage). Bullock's Reach and Conimicut Point are located in the Providence River Estuary. The seasonal analysis used Secchi-converted *k* data because it was the dataset available year-round, and the analysis included only those stations for which year-round data were available.

In addition, the Estuary Program examined differences in water clarity between dry years (low river flow and low precipitation) and wet years (high river flow and high precipitation) before and after nutrient reduction upgrades occurred (see "Nutrient Loading" chapter). This calculation was done the same way using the same data for multiple indicators: dissolved oxygen, chlorophyll, marine beaches, and water clarity. For more information, see the "Dissolved Oxygen" chapter. In short, the Estuary Program compared summer median river flow for individual years against the median for approximately fifteen years. When individual medians were greater than the dataset median, they were considered wet years. Dry years were defined as those years with individual medians less than the dataset median.

Status and Trends

Water clarity in 2014 was greater in the Mid to Lower Bay than in the Upper Bay, particularly in spring and summer (Figure 3) based on Secchi depth converted to *k*. On average for the entire year, *k* decreased from 0.8 in the north to 0.5 in the south, translating to an average clarity depth of 2 meters (6.6 feet) in the Upper Providence River to 3.5 meters (11.5 feet) in the East Passage.

Seasonally, regardless of station, water was clearest in winter and more turbid in summer ($F = 5.270$; $p = 0.016$). The greatest spatial differences in the Bay occurred in summer, when *k* declined from 0.9 in the north to 0.7 in the south, translating to an average

Table 1: Locations in the Upper Bay and the Mid to Lower Bay for which PAR or Secchi depth data were analyzed.

Bay Section	Station Name	Data-Collection Entity
	Upper Bay	
Providence River Estuary	India Point Park, Bullock's Reach, Conimicut Point	NBC, NBFSMN (Conimicut Point only)
Mount Hope Bay	Mount Hope Bay	NBFSMN
Upper Bay	North Prudence	NBFSMN
	Mid to Lower Bay	
East Passage	T-Wharf	NBNERR
West Passage	Fox Island, GSO Dock	URI-GSO (Fox Island), NBFSMN (GSO Dock)

NBC: Narragansett Bay Commission. NBFSMN: Narragansett Bay Fixed Site Monitoring Network. NBNERR: Narragansett Bay National Estuarine Research Reserve. URI-GSO: University of Rhode Island – Graduate School of Oceanography.

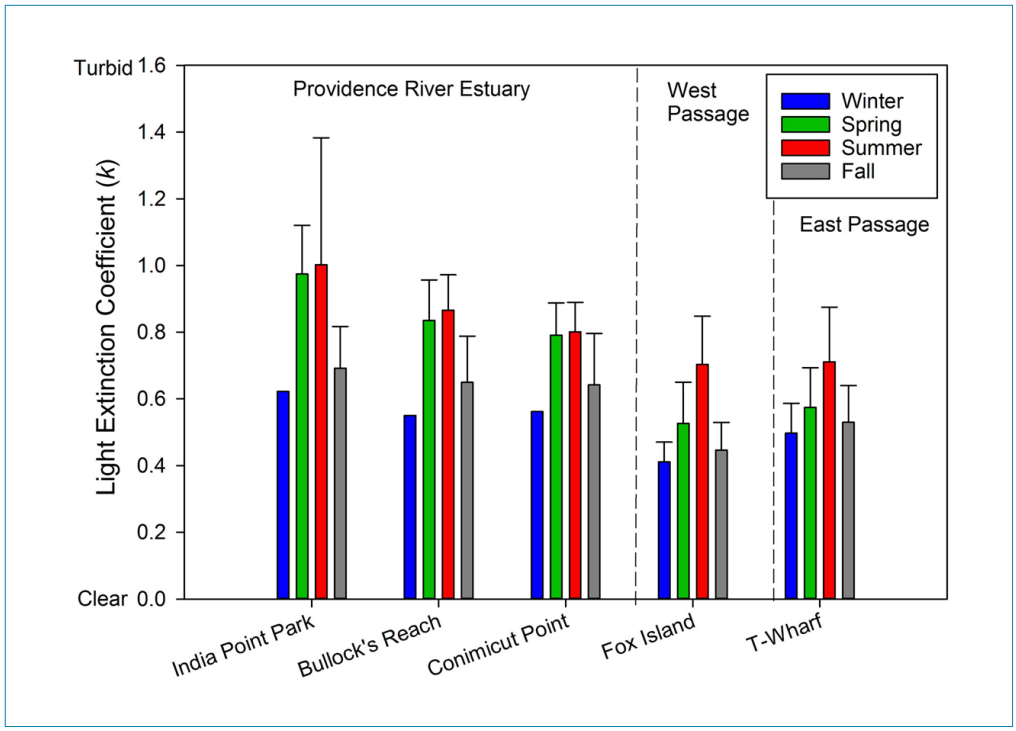


Figure 3. Light extinction coefficients (*k*), derived from Secchi depth, by season in 2014. Stations are listed according to geographic position from Upper Bay (left) to Lower Bay (right). Secchi depth data were converted to *k* (see Methods), introducing an error of approximately 20 percent. *Seasons: winter (January, February, March), spring (April, May, June), summer (July, August, September), fall (October, November, December). Sample sizes: winter, n > 8 (except Providence River Eestuary, n = 1); spring, n > 7; summer, n > 10; fall, n > 8. Error bars are standard deviations.*

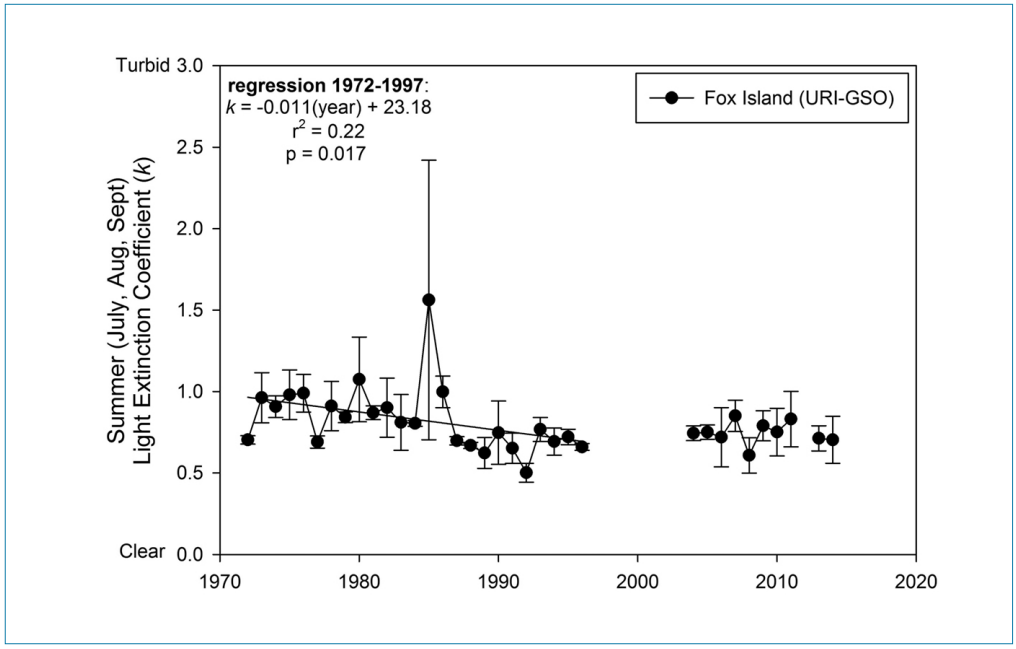


Figure 4. Summer averaged light extinction coefficients (*k*), calculated from Secchi depth, at Fox Island. Solid line represents the linear regression for 1972 to 1997 data. Data were collected weekly. Error bars are standard deviations. Sample size (n) is greater than ten for all years. Secchi depth data were converted to *k* (see Methods), introducing an error of approximately 20 percent.

clarity depth of 1.7 meters (5.6 feet) at India Point Park in the Upper Providence River to 2.7 meters (8.9 feet) at Fox Island in the West Passage (Figure 3). Summer light extinction coefficients were significantly higher at India Point Park than in the Lower Bay ($F = 4.141$, $p = 0.02$). In fall and winter, the Upper Bay and the Mid to Lower Bay had similar water clarity (Figure 3).

The Estuary Program focused its in-depth analysis of water clarity trends on data collected during summertime. Most of the water clarity data had been collected during summer, along with data on other water quality parameters such as chlorophyll concentrations and dissolved oxygen levels. Managers have focused on this period of the year because it is when lower clarity has a greater potential to adversely affect the aquatic ecosystem. Additionally, in 2012 eleven Rhode Island wastewater treatment facilities achieved a 50 percent reduction in nutrient loading (see "Nutrient Loading" chapter), which was designed to reduce hypoxia and may have positive effects on water clarity, and the analysis looked for such effects. From 1972 to 1997, summer light extinction coefficients (k) derived from Secchi depth declined by about 20 percent at Fox Island in the West Passage (Figure 4), indicating an improvement in water clarity. However, the more recent data from 2004 to 2014 did not show any improvement. Data for 1998 to 2003 and 2012 were unavailable.

While the Fox Island dataset covers the longest time period, the Estuary Program also analyzed approximately eight years of summertime Secchi (for Fox Island and T-Wharf) and PAR (for all stations in the Providence River Estuary) data for stations located along a north-south transect. Figure 5 shows the same stations as Figure 3, but presents only summer data. In addition, Figure 5 shows data before and after 2012, when a 50 percent reduction was achieved in nitrogen loading from Bay wastewater

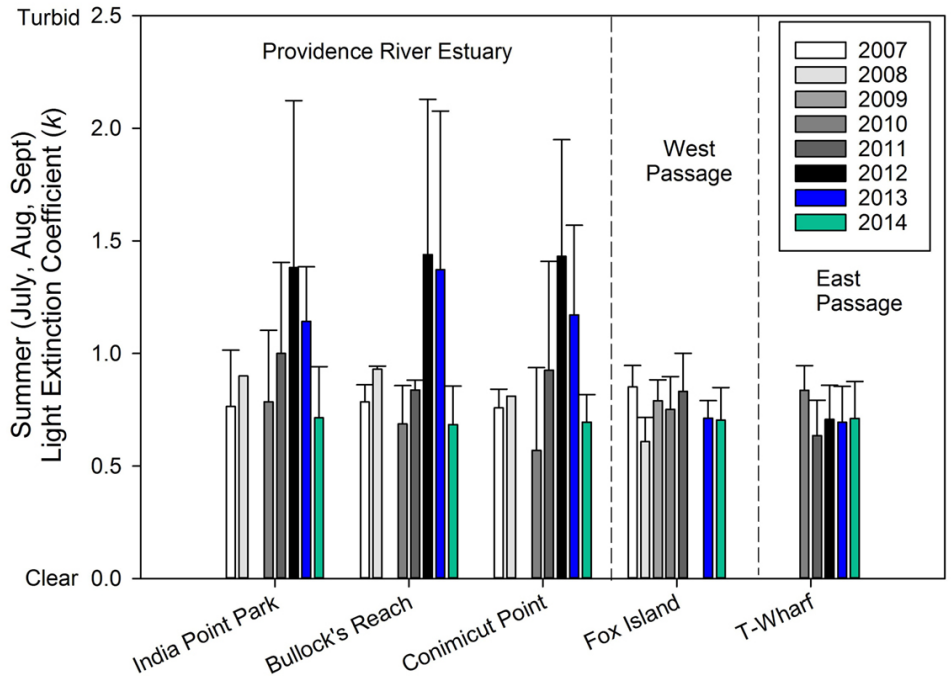


Figure 5. Summer (July, August, September) light extinction coefficients (k) for 2007 to 2014 based on PAR data for the Providence River Estuary and Secchi depth data for the West and East Passages. Stations are listed according to geographic position from Providence (left) to the Lower Bay (right). Gray bars (2007 to 2012) indicate the time period before the 50 percent nitrogen-reduction goal was met (see text). Colored bars (2013 and 2014) indicate the time period after the nitrogen reduction goal was met. Error bars are standard deviations. Secchi depth data were converted to k (see Methods), introducing an error of approximately 20 percent. *Sample sizes: For the Providence River Estuary stations (India Point Park, Bullock's Reach, Conimicut Point), $n > 2$ for 2007 to 2011, except 2008 at India Point Park and Conimicut Point where $n = 1$, and $n > 10$ for 2012 to 2014. For Fox Island and T-Wharf, $n > 8$ for all years.*

treatment facilities (see "Nutrient Loading" chapter). Because of the limited number of values and the interannual variability, it is not possible to discern meaningful trends. In the Providence River Estuary, water clarity was at its lowest in 2012 and 2013, and then it improved in 2014. The two stations located in the East and West Passages had less variability than the stations in the Providence River Estuary.

Discussion

Water clarity data in Narragansett Bay reflect a north-to-south gradient and are characterized by interannual variability. In 2014, the stations farthest south in the Bay—Fox Island and T-Wharf—on average reported greater clarity throughout the year when compared to all other stations (Figure 3). All stations also reflected an annual pattern of greater clarity in the winter followed by declining clarity in the spring and summer (Figure 3). This is expected given the biological activity in the Bay, including increased growth of phytoplankton during the warmer seasons.

Fox Island water clarity data showed an improvement from 1972 to 1997, and then remained steady until present (Figure 4). The temporal increase identified in the summer water clarity was similar to the increase in the annual average water clarity reported by Borkman and Smayda (1998) using the same Fox Island data. They linked the improvements in water clarity to a reduction of suspended solids from wastewater discharge during the 1980s (Borkman and Smayda 1998, 2016). During the same time period, chlorophyll concentrations declined (Nixon et al. 2009, Oviatt et al. 2015, see "Chlorophyll" chapter), contributing to the improvement in water clarity as well.

There is interest in the response of the Bay to reduced nutrient pollutant loadings achieved in recent years. While planned reductions in nutrient loadings from certain wastewater treatment facilities were achieved in 2012, the infrastructure improvements were phased in over several years. The Providence River Estuary data revealed a decline in clarity from 2009 through 2012, followed by improved summer water clarity during 2013 to 2014 (Figure 5). In 2014, water clarity across the Bay appeared to be similar with little difference reported between the Providence River Estuary and the East and West Passages (Figure 5). However, clarity conditions similar to 2014 had also been recorded prior to 2010, indicating additional data collection is needed before drawing any conclusions about a lasting change or trend in clarity being associated with the nutrient reductions.

In addition to nutrient loading, precipitation and associated stormwater runoff introduce suspended sediment to the Bay, contributing to declines in water clarity, particularly in the Providence River Estuary. As a preliminary inquiry, water clarity from wet and dry summers, using one year from before and one year from after nitrogen-reduction occurred, were examined for stations along a north-south transect (Figure 6). Wet summers had reduced water clarity than dry summers, most notably in the Providence River Estuary. A significant improvement in water clarity was evident in the dry years before (2007) and after (2014) the 50 percent nitrogen-reduction ($F = 8.692$, $p = 0.0150$). Because of the strong connection between nutrient loading and water clarity (Borkman and Smayda 1998, 2006), the Estuary Program also expected to see an improvement in the pre- and post-reduction wet years (2006 and 2013). However, it was not possible to test that hypothesis because of limited data availability for 2006. Further data collection and analysis in a future wet year could make it possible to determine how precipitation and nutrient loading interact to affect water clarity.

In 2008, the Narragansett Bay Commission opened a large tunnel designed to capture heavy precipitation events that normally would flood the Field's Point wastewater treatment facility, causing minimally treated or untreated sewage to flow into the Bay (NBC 2017; see "Precipitation" chapter). The reduction of particulates entering the Bay due to the capture of combined sewer overflows (CSO) to this tunnel could also impact water clarity. The majority of the data analyzed in the Providence River Estuary were collected after the tunnel opened, hindering the analysis of the impact the tunnel had on water clarity. To truly assess the impact of the tunnel on water clarity, the days the tunnel was used and water clarity measurements taken around those days would need to be correlated.

Climate change may also affect water clarity. Increased rainfall and more intense rainfall events will likely deliver more sediment, nutrients, dissolved organic matter, and other particles through urban runoff or nonpoint sources, potentially decreasing water clarity (Balch et al. 2016, USEPA 2016), perhaps influencing primary production. Warming waters in response to climate change will also affect water clarity as phytoplankton species are expected to change (Smayda et al. 2004, Nixon et al. 2009). New species in the Bay may have different tolerances for increased water clarity or nutrient levels than the current suite of phytoplankton that occurs in the Bay.

Water clarity has improved throughout the Bay since 2012, and in the lower portions of the Bay

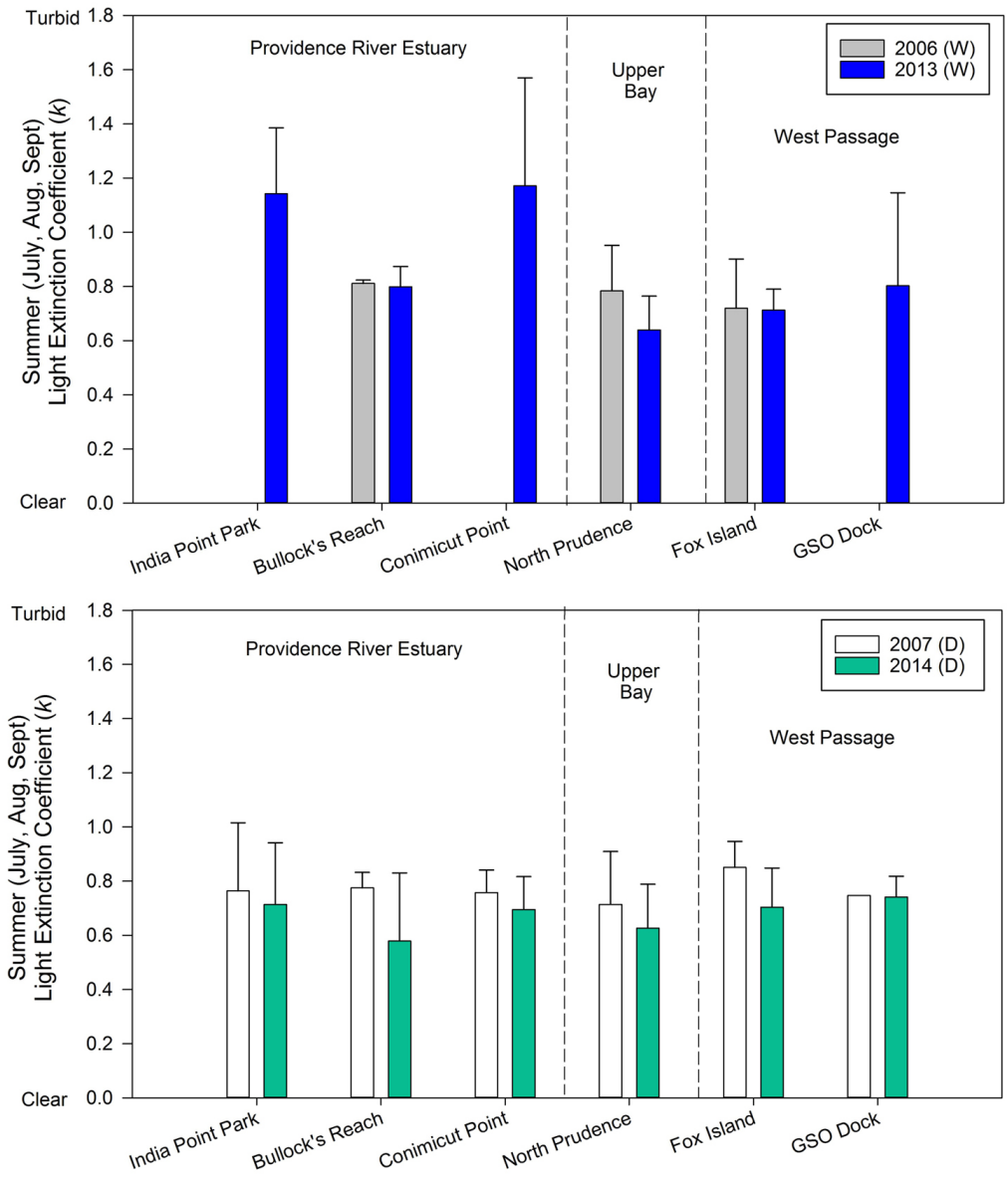


Figure 6. Summer light extinction coefficients (*k*) in wet years (top: 2006 and 2013) and dry years (bottom: 2007 and 2014) based on PAR data and Secchi depth data (Fox Island only). Stations are listed according to geographic position from Providence (left) to the Lower Bay (right). Error bars are standard deviations. Secchi depth data were converted to *k* (see methods), introducing an error of approximately 20 percent. *Sample sizes: For 2006, n > 2 for Bullock's Reach and North Prudence; Fox Island n = 14. For 2013, n > 5 for Bullock's Reach, North Prudence, and GSO Dock; n = 11 for India Point Park and Conimicut Point; n = 14 for Fox Island. For 2007, n > 4 for all stations but Fox Island (n = 10) and GSO Dock (n = 1). For 2014, n > 3 for Bullock's Reach, North Prudence, and GSO Dock; n > 11 for India Point Park, Conimicut Point, and Fox Island.*

over the last 30 years. The continuation of improved conditions is dependent upon nutrient loading, stormwater management, land use practices, and changing precipitation patterns associated with climate change. Point-source nutrient loading has declined in the Providence River Estuary, while precipitation (and river flow) will deliver sediment and non-point source nutrients to the Bay, making precipitation as stormwater runoff a very important stressor affecting water clarity. While precipitation itself cannot be controlled by management actions, improvements in how runoff is captured and treated (e.g., the CSO tunnel) are possible and could benefit water clarity. The benefits could be realized in better water quality conditions for seagrass habitat, as well as increased benthic primary production, enhancing nutrient recycling in the shallow parts of the Bay and improving the overall environmental condition of the Bay.

Data Gaps and Research Needs

- There are gaps in the availability of clarity data for portions of the Bay, especially the embayments. Devising a plan to achieve more consistent methods, greater frequency of sampling, and better spatial coverage throughout the Bay is appropriate.
- In devising a sampling plan, attention should be paid to the appropriate sampling intervals in order to reduce variability in the datasets and to enhance the ability to detect change. Accordingly, it would be valuable to conduct a careful analysis of the various datasets and/or a field study to determine an optimal sampling frequency to detect changes in water clarity.
- The Estuary Program compared *k* values for both Secchi depth and PAR to maximize the use of available data. Ideally, one monitoring method—either Secchi depth or PAR—would be used throughout the Bay. However, the Estuary Program will continue to evaluate the comparison between Secchi depth and PAR using data collected in Narragansett Bay. Comparison of *k* values from the two monitoring methods would facilitate accurate use of *k* as a water clarity metric throughout the Bay.
- Improving the spatial resolution of coastal water clarity measurements based on satellite remote sensing would reduce the need to take field measurements and would allow for a Bay-wide assessment, including embayments.
- An event-based study of water clarity is needed to determine how closely total suspended solid loading is related to storm events, and how to manage those loads.

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State of Narragansett Bay and Its Watershed
2017 Technical Report

Watershed Ecosystem Condition Indicators

INTRODUCTION

Please use the following citation: Narragansett Bay Estuary Program. 2017. State of Narragansett Bay and Its Watershed (Watershed Ecosystem Introduction, pages 339-340). Technical Report. Providence, RI.

NARRAGANSETT BAY ESTUARY PROGRAM

Photo: Moswansicut Reservoir, Scituate, RI (Northern Rhode Island Conservation District)

The Narragansett Bay Watershed comprises four River Basins: the Blackstone River Basin, the Pawtuxet River Basin, the Coastal Narragansett Bay Basin, and the Taunton River Basin. These four basins differ in topography, hydrology, and land use histories, leading to differences in ecosystem composition and quality. The Blackstone River Basin has the highest elevation, with many streams and rivers fragmented by dams from the industrial era. The Pawtuxet River Basin has moderate elevations and is highly forested in its upper reaches to protect critical water supplies. The Coastal Narragansett Bay Basin is highly developed and represents those small rivers and streams that discharge directly to the Bay. Finally, the Taunton River Basin has low-lying lands and few dams, which enables salt and fresh water to mix in a 20-mile-long unobstructed estuary.

Natural lands serve as buffers that protect habitat for stream invertebrates and freshwater fish, as well as water quality. However, the Narragansett Bay Watershed is ubiquitously stressed by urbanization, which increases impervious cover, changes land use, increases nutrient loadings, degrades habitats, and alters hydrology, as discussed in other chapters of this report. Climate change is exacerbating these issues through changing species community structure, increasing water temperatures, and increased stormwater runoff.

This section focuses on four indicators of the condition of the Narragansett Bay Watershed ecosystem: stream invertebrates, water quality conditions for aquatic life, freshwater fish communities, and open space. These indicators represent key habitats, culturally and economically important animal groups, and physical characteristics of the Watershed. The indicators provide information on the quality of the Watershed's ecosystem and point to areas that require restoration or further protection. The data show that ecosystem condition varies greatly from place to place in the Watershed and that opportunities exist to better protect natural lands around the Bay.

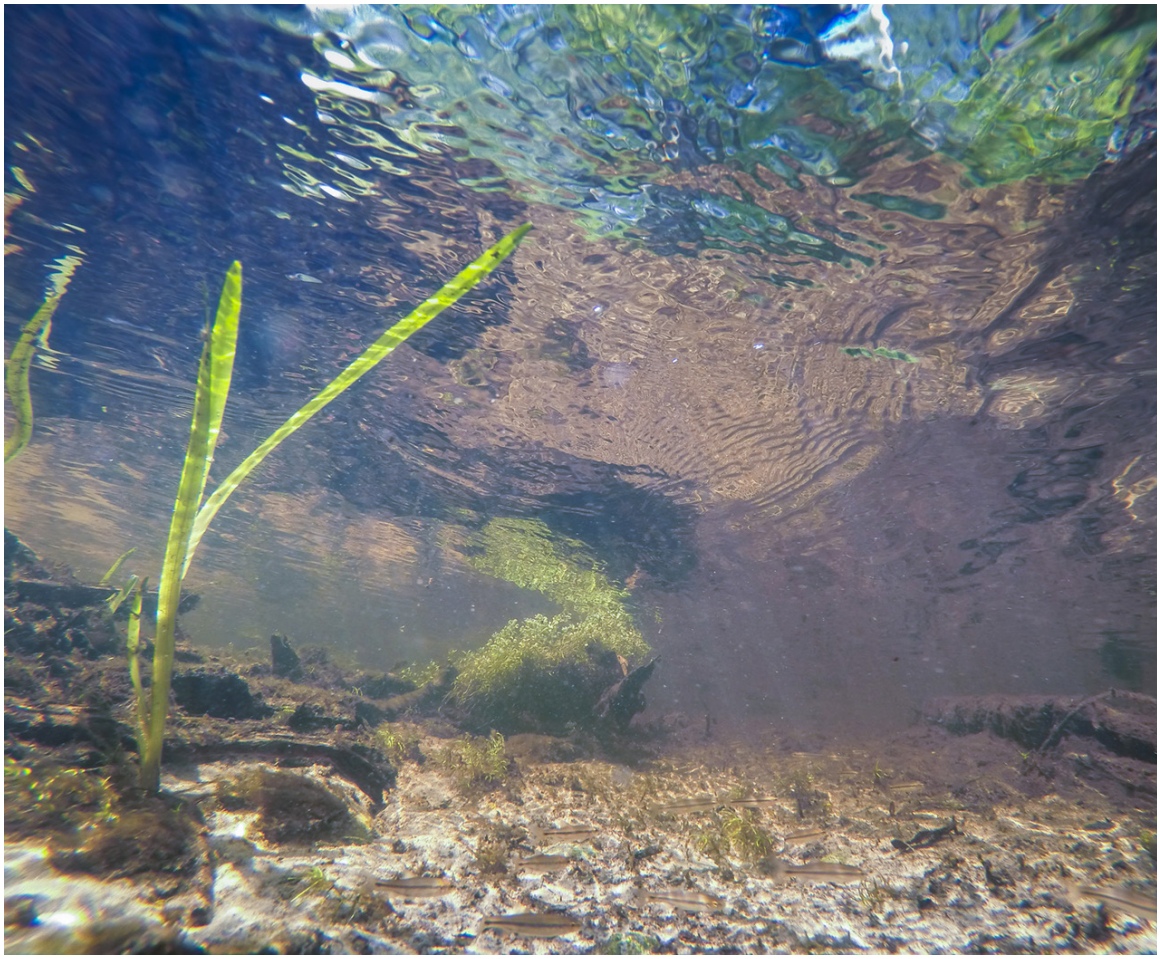


Photo: Upper Wood River, Exeter, RI (Ayla Fox)



State of Narragansett Bay and Its Watershed
2017 Technical Report

Watershed Ecosystem Condition Indicators

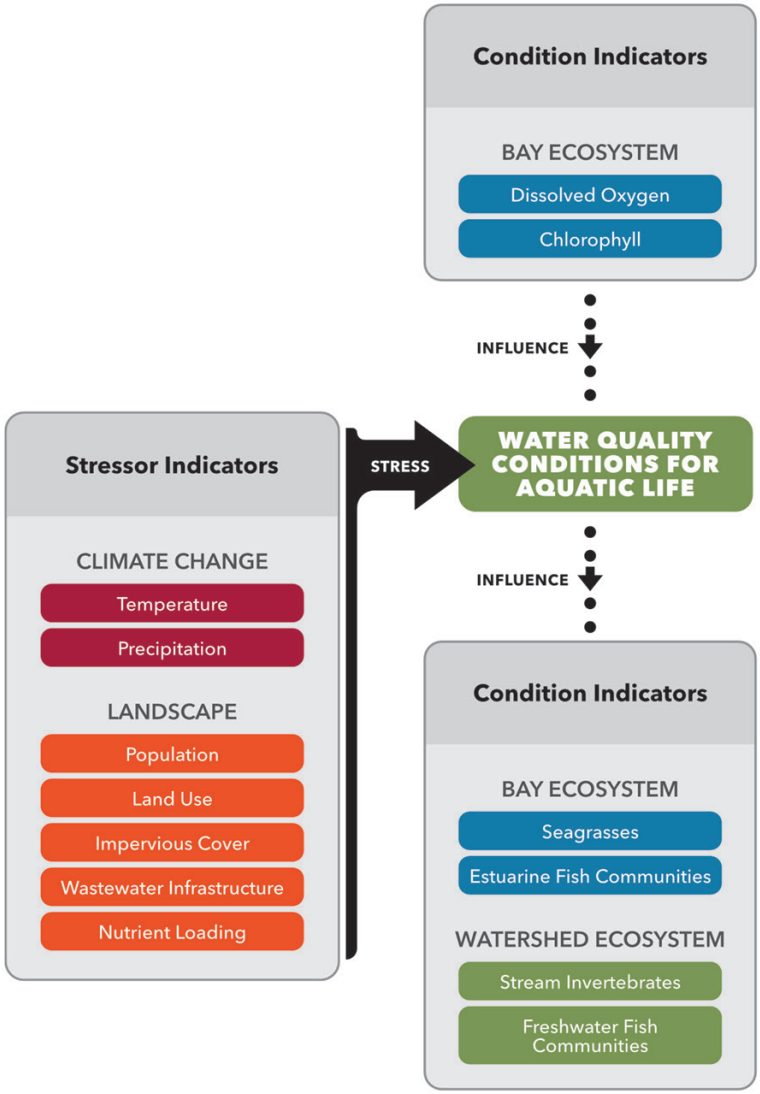
CHAPTER 18: WATER QUALITY CONDITIONS FOR AQUATIC LIFE

Please use the following citation: Narragansett Bay Estuary Program. 2017. State of Narragansett Bay and Its Watershed (Chapter 18, Water Quality Conditions for Aquatic Life, pages 341-361). Technical Report. Providence, RI.

NARRAGANSETT BAY ESTUARY PROGRAM

Photo: Pond in Richmond, RI (Ayla Fox)

Overview



BACKGROUND

- Acceptable levels of nutrients and dissolved oxygen are necessary to support thriving aquatic communities. However, excess nutrients combined with other factors can be detrimental to aquatic systems by prompting algal blooms and subsequent low-oxygen conditions. Sources of nutrients to both estuarine and freshwaters include wastewater treatment plant discharges, onsite wastewater treatment systems, fertilizer use, and pet and livestock waste, among others. These sources are exacerbated by increased population, urbanization, and heavy precipitation.

KEY FINDINGS

- ***Status***

Estuarine Waters: Half of the estuarine waters assessed by the states for aquatic life use were determined to be in healthy condition for aquatic life, with the remainder impacted by nutrient enrichment and depleted oxygen. Impacted waters were all in the Upper Estuary, including Mount Hope Bay.

Streams and Rivers: Over half of the streams and rivers assessed by the states for aquatic life were identified as suitable for fish, and other wildlife. Most of these waters with acceptable conditions were found in the Pawtuxet River Basin. Nearly 20 percent of the streams were impacted by excess nutrients and oxygen depletion, including the entire Blackstone River, while aquatic life use in the remaining waters (26 percent) were impacted by other parameters such as non-native aquatic plants, toxicity, and chlorophyll, among others.

Lakes and Ponds: One-quarter of the total acreage of lakes and ponds assessed by the states for aquatic life use were acceptable for aquatic life use. One-quarter were impacted by excess nutrients and oxygen depletion, and half were impacted by other parameters. Over 50 percent of the lakes and ponds in the Coastal Narragansett Bay Basin were impacted by nutrient enrichment and low oxygen levels.

Nearly 40 percent of the freshwaters in the Narragansett Bay Watershed and 27 percent of estuarine waters in the Bay were not assessed by the states for aquatic life use, including the entire Sakonnet River.

Introduction

Water quality is critical for aquatic life; one of the goals of the Clean Water Act is to achieve water quality that provides for the protection and propagation of fish and wildlife. Acceptable levels of nutrients (e.g., nitrogen and phosphorus) and dissolved oxygen are critical factors that support thriving aquatic communities. Nutrients are key drivers of the primary production that supports aquatic ecosystem food webs. However, excess nutrients can lead to degradation of aquatic ecosystems. High nutrient loads can intensify primary productivity, often resulting in algal blooms and proliferating aquatic plant growth and decomposition. When the plants die and decompose, oxygen in the water column can be depleted to critically low levels for fish and other organisms. Oxygen levels can also be influenced by physical parameters such as density stratification (temperature and salinity), and mixing and transport of low-oxygen waters (see "Dissolved Oxygen" chapter).

Urbanization of the Narragansett Bay Watershed has exacerbated the transport of nutrients from sources such as fertilizers, wastewater, atmospheric deposition, and pet and livestock waste to receiving waters, particularly during heavy precipitation events (see "Nutrient Loading" chapter). In the upper portion of the Bay and its major tributary rivers, discharges from wastewater treatment facilities remain a major source of nutrients. Historically, the waters in the Watershed and the Bay were impacted by nutrient inputs from agricultural practices (Pastore 2014). In much of the Watershed, stormwater runoff is the primary means by which nutrients are transported to waterbodies (see "Impervious Cover" chapter). Greater amounts of impervious surfaces increase the risk of nutrient delivery to surface waters via stormwater runoff, often discharging directly into adjacent streams or estuaries (Nowicki and Gold 2008). High impervious cover and developed land uses have been correlated with higher nutrient concentrations in streams and estuaries (Lee et al. 2012, Clapcott et al. 2012, Sanger et al. 2015).

Additionally, nutrients discharged to groundwater from onsite wastewater treatment systems (septic systems) and cesspools are well-known sources of pollution to streams and coastal waters. Nowicki and Gold (2008) estimated that groundwater input to the Bay is five to ten percent of the total freshwater input, and they identified that localized groundwater seepage to numerous coves and embayments along the Bay's shoreline (e.g., Greenwich Bay) had significant inputs of nitrogen from groundwater. Other studies in smaller, non-urban watersheds of the region suggest that groundwater discharge can supply more than 80 percent of the nitrogen inputs into shallow estuarine embayments not receiving wastewater treatment plant discharges (Bokuniewicz 1980, Capone and Bautista 1985, Lee and Olsen 1985, Valiela et al. 1990, Gobler and Boniello 2003).

Climate change will likely increase the influence of precipitation and temperature on water quality conditions (see "Precipitation" and "Temperature" chapters). Rising temperatures are expected to alter nutrient cycling and oxygen dissolution, which may affect aquatic ecosystem function (Kundzewicz and Krysanova 2010). Precipitation and flooding associated with more frequent and intense storms may amplify nutrient transport to receiving waters. Analyzing the extent and distribution of waters within the Narragansett Bay Watershed that either support aquatic life or are impacted by excess nutrients or depleted oxygen informs an understanding of water quality conditions for aquatic life, as well as long-term efforts to improve water quality.

In accordance with the federal Clean Water Act (CWA), Rhode Island and Massachusetts have identified and designated waters according to specific uses and periodically publish these results in Integrated Reports submitted to the US Environmental Protection Agency (EPA) (MassDEP 2015, RIDEM 2015). All waterbodies that are tracked by environmental agencies are designated for aquatic life use to fulfill obligations under the Clean Water Act; however, the states require that sufficient data and information are available on parameters that can affect aquatic life use in order to decide whether water quality is suitable for fish, macroinvertebrates, or other aquatic life. Where such information is lacking, these waterbodies cannot be assessed for this use. Measurement of nutrients and dissolved oxygen, in conjunction with numerous other parameters and indicators, can be used to identify waterbodies for inclusion in the List of Impaired Waters (Section 303(d) of the CWA) that is used to prioritize waterbodies for development of restoration plans known as Total Maximum Daily Loads (TMDLs) (MassDEP 2015 and RIDEM 2015). While both states use different water quality standards for nutrients and dissolved oxygen, based on narrative and numeric criteria (Extended Methods, Table 8), they lead to similar determinations as to whether conditions are acceptable for aquatic life use.

In this chapter, the Estuary Program focuses on the water quality assessment by the states, based on attainment status ("fully supporting," "not supporting," or "unknown"), and the cause of impairment for aquatic life with an emphasis on nutrient enrichment and low dissolved oxygen. To determine water quality conditions for aquatic life, the states also test many other parameters, such as toxins, that are unrelated to nutrients and dissolved oxygen, but such parameters were not the focus of this chapter. The goal of measuring water quality conditions for specific parameters causing impairment due to nutrients and depleted oxygen was to provide a quantitative and geographical overview of areas that may be contributing to sources of human-caused nutrient enrichment (e.g., proximity to conventional onsite wastewater treatment systems or cesspools) (see "Wastewater Infrastructure" chapter)and therefore creating conditions for eutrophication and/or hypoxia (see "Nutrient Loading" and "Dissolved Oxygen" chapters).

To measure this indicator, the Estuary Program integrated information about the extent of state water quality assessments within the broader hydrological context of the Bay and Watershed in both Massachusetts and Rhode Island, which functions as a reference for understanding the status of water quality conditions for aquatic life use. Only a proportion of all waters that define and shape the Watershed and the Bay are assessed by the states for aquatic life use.

It is important to acknowledge that while the results from this analysis are consistent with the Integrated Reports from both states (MassDEP 2015 and RIDEM 2015), the water quality conditions for this indicator as defined by the Estuary Program reflect that (a) the extent of acceptable waters is equivalent to the extent of state-assessed waters where conditions are "fully supporting" of aquatic life use, (b) the extent of state-assessed waters that are impacted for aquatic life use differs from states' reporting of impaired waters or the 303(d) list by reporting only on "not supporting" due to nutrient enrichment and depleted oxygen, and (c) waters identified as unknown were not assessed for aquatic life use by the states.

By knowing the total measurement and geographic extent of waters that are assessed or not for aquatic life use, these results can address three key issues: (1) waters with acceptable conditions for aquatic life use should be protected; (2) waters that were

identified as impacted due to nutrient enrichment and low oxygen levels can pinpoint potential sources of contamination and stressors that can also be associated with natural or anthropogenic hotspots (McClain et al. 2003) from sewage contamination, fertilization, and other sources; and (3) waters that are unknown can be identified as data gaps within the state assessments to further evaluate other state-assessed waters for aquatic life use.

Methods

EXTENT OF STATE-ASSESSED WATERS

The Estuary Program calculated the total extent of fresh and estuarine waters within the Bay and Watershed to compare with the extent of state water quality assessed waters. The states do not assess every stream segment, pond, and lake in the Watershed or all the areas in the Bay. Waters assessed by the states for a variety of uses represent only a portion of the entire hydrological network in the Watershed and Bay waters because of the states' criteria and capacities for implementing waterbody assessments.

For the purposes of this chapter, the total extent of freshwaters is defined by the National Hydrography Dataset (NHD). The Estuary Program calculated total extents using (1) NHD flowlines for streams and rivers (excluding other segments such as canals), and (2) NHD waterbodies greater than one acre for lakes, ponds, and reservoirs (excluding other categories such as wetlands). The extent of Narragansett Bay estuarine waters was defined by the Estuary Program (see this report's Introduction and Appendix). These data were also used to calculate total extents of fresh and estuarine waters by state and by River Basin.

The Estuary Program used NHD data at a high resolution (1:24,000) to best define a baseline of all the waters in the Watershed that have been delineated by the US Geological Survey (USGS). It is important to note that a portion of NHD waters may not meet states' criteria to be tested as part of the Integrated Reports. For example, the NHD data used by the Estuary Program do not differentiate intermittent versus perennial streams, a limitation that might be considered by the states.

As a reference for this chapter, Table 1 summarizes information about the extent of waters mapped in the high-resolution NHD, and total state-assessed waters (shown by state), as shown in Figure 1.

The states assessed 37 percent of NHD stream miles, 82 percent of NHD ponds and lakes greater than one acre, and 78 percent of the Bay as defined by the Estuary Program (Table 1; Figure 1). While more than three-quarters of the estuarine waters were assessed by the states, 43.5 square miles of waters in Narragansett Bay between Aquidneck Island and the Rhode Island Sound were not assessed (Figure 1). These metrics situate this analysis within the broader hydrological context of the Bay and Watershed. However, they are not presented as a target for state assessments, as the states have differing criteria and capacities for water quality assessment implementation.

Table 1. Extent of all state-assessed waters in Narragansett Bay and its Watershed. Total areas for National Hydrography Dataset (NHD) and Bay waters are included for reference.

Within Narragansett Bay and its Watershed	Estuarine Waters (sq. miles and percent)	Stream and Rivers (miles and percent)	Lakes and Ponds (acres and percent)
Total extent of all waters [1]	197	3,578	40,910
Total state-assessed waters	156	1,337 [2]	33,795
MassDEP	10	451 [2]	20,586
RIDEM	146	886 [2]	13,210
Percent of all waters that are state-assessed [3]	78 percent	37 percent	82 percent

[1]Based on National Hydrography Dataset (NHD); the area of the Bay as defined by the Estuary Program.
[2]An estimated four miles of MassDEP-assessed segments of Runnins River and Blackstone River were removed from analysis where the features both crossed outside of the state border and were duplicated in RIDEM GIS features.
[3] Total state-assessed waters divided by all waters in the Bay and the Watershed. These percentages represent the extent of waters in the Bay and in the Watershed that were assessed by the state agencies (MassDEP and RIDEM).

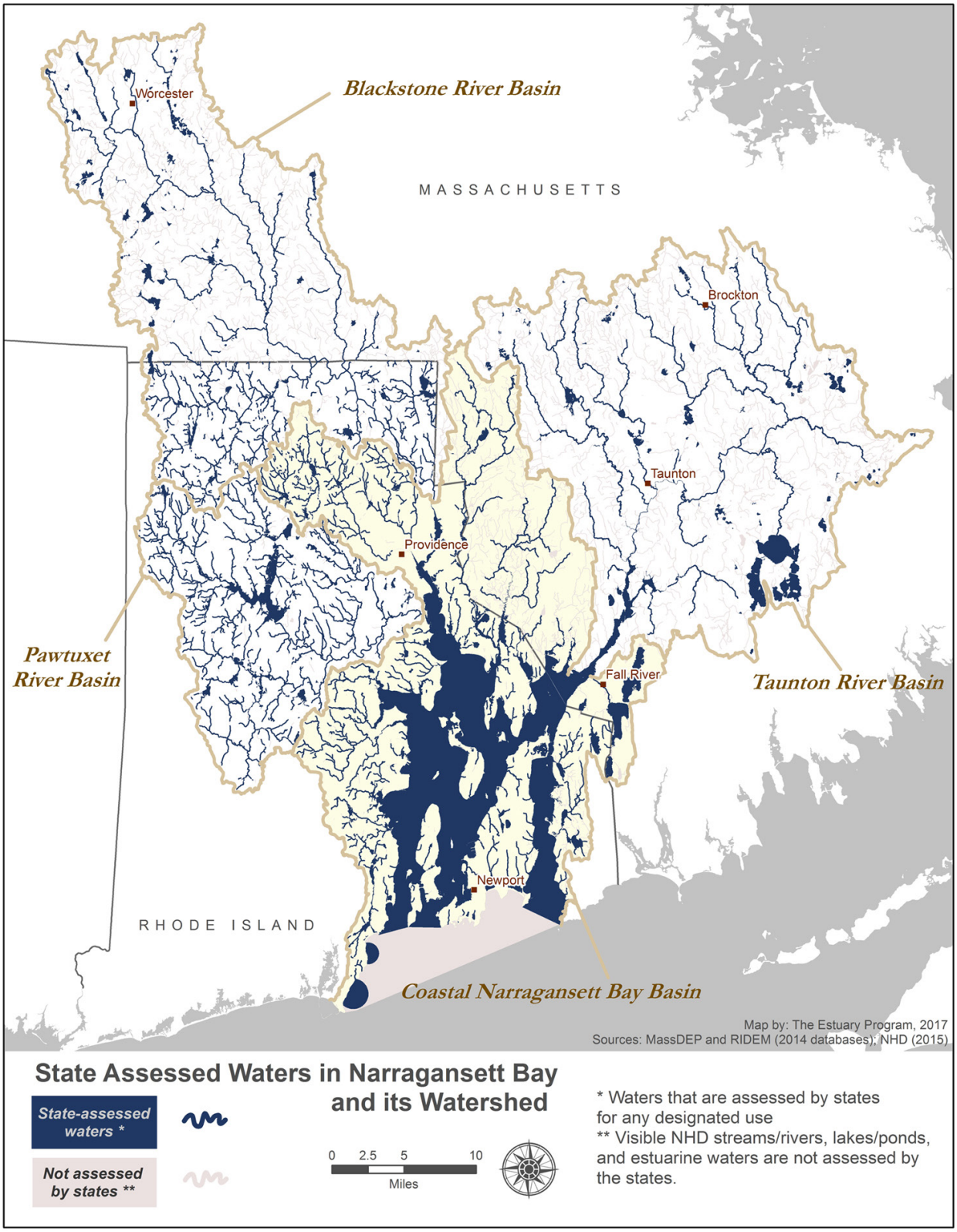


Figure 1. The extent of freshwaters and estuarine waters in Narragansett Bay and its Watershed and the proportion of waters that have been assessed by MassDEP and RIDEM. Additional high-resolution National Hydrography Dataset (NHD) waters show all waters that shape the Bay and its Watershed.

A combined analysis of state water quality assessments in Massachusetts and Rhode Island must be informed by important differences between the states. Nearly two-thirds of the state of Rhode Island (63 percent) falls within the boundaries of the Narragansett Bay Watershed compared to only 13 percent of the state of Massachusetts; however, Massachusetts comprises 60 percent of the Watershed area. The Pawtuxet River Basin is completely within Rhode Island, and the Taunton River Basin is completely within Massachusetts. Both the Coastal Narragansett Bay Basin and the Blackstone River Basin span the Rhode Island-Massachusetts border (Figure 1).

Massachusetts has a much larger network of streams and rivers under its jurisdiction, and as a result Rhode Island's assessments reflect the capacity of its programs to analyze a greater number of streams and rivers per unit area compared to Massachusetts. Due to the larger size of Massachusetts, many waterbodies are not yet assessed for aquatic life use within the Massachusetts portion of the Watershed.

EXTENT OF STATE-ASSESSED WATERS FOR AQUATIC LIFE USE

The Estuary Program calculated total extent of "state-assessed waters" using the states' 2014 Assessment Databases. Figure 2 shows the categorization of waters used by the Estuary Program. Extents were categorized by state and by River Basin. Many waterways and waterbodies were assessed for other designated uses (e.g., recreational use; see "Water Quality Conditions for Recreation" chapter) and not for aquatic life use. While all waterbodies assigned unique waterbody identification and tracked in the states databases are designated for aquatic life use, there are many waters that have not been assessed due to insufficient data or information available to determine water quality conditions for aquatic life use. Those waters were included in the Estuary Program's total extent of state-assessed waters but not in the total extent of state-assessed waters for aquatic life use, thus representing unknown conditions for aquatic life use.

WATER QUALITY CONDITIONS FOR AQUATIC LIFE

Waters assessed for aquatic life use by the states have sufficient data and information on an array of parameters or indicators to determine water quality conditions, which are categorized based on state water quality standards as: (1) "fully supporting," meaning acceptable for aquatic life use, or (2) "not supporting" aquatic life use due to impacts by one or more parameters (Table 2). The states have multiple parameters that can be used to assess waterbodies

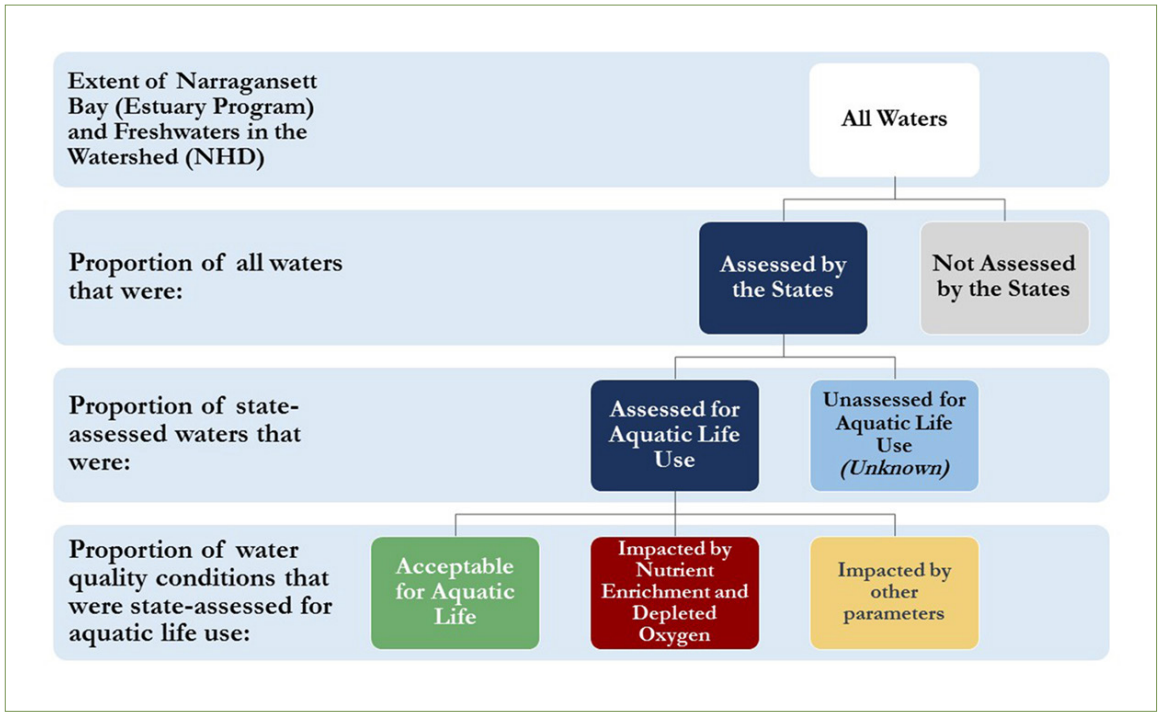


Figure 2. Conceptual diagram of the Estuary Program's methodology representing the breakdown of state-assessed waters for measuring water quality conditions for aquatic life use. Box colors represent the symbology in the maps (Figures 1, 4, and 6).

as acceptable or impaired for aquatic life use. When one or multiple causes of impairment are identified by the state agencies, the waterbody is identified as not supporting aquatic life use and may be added to the List of Impaired Waters (CWA Section 303[d]) and prioritized for development of TMDLs (RIDEM 2016). The state reports reflect assessments derived from data generated in multiple years (within a timeframe of five years). Not all waters are monitored within the same year, as both MassDEP and RIDEM conduct testing and assessments on a rotating-Basin basis. Dates for Basin assessments were not readily available when this indicator was developed; however, data can be available upon request, and the Estuary Program will do so to incorporate that information in future reports.

The Estuary Program's methods for reporting water quality conditions were designed in coordination with Rhode Island and Massachusetts state agencies to reconcile the differences between state water quality assessments. State monitoring efforts focus on biological indicators, physical habitat, conventional parameters, and other indicators that may be useful to define potential stressors or sources of aquatic life impairment (RIDEM 2014, MassDEP 2016). For this report, the Estuary Program focused on two causes of impairment that are tested by the states and that are important indicators of a waterbody's ability to support aquatic life: nutrients and dissolved oxygen. These conventional parameters are part of a subset of several factors used by the states to determine if water quality supports aquatic life use (Table 2) (MassDEP 2012 and 2016, RIDEM 2014).

The Estuary Program defined categories of conditions that are comparable across states by relying on the determinations made by the states rather than raw data from the different parameters tested for nutrient enrichment or dissolved oxygen (Extended Methods, Table 8). The raw data were not used because the parameters used by each state to determine nutrient enrichment or dissolved oxygen were different (Table 2) and the timeframe when data were collected varied between states. From the categorical determinations made by the states, the Estuary Program calculated the extent of state-assessed waters for aquatic life use as (1) miles of streams and rivers, (2) acres of lakes and ponds, and (3) square miles of estuarine waters.

These metrics were calculated for the entire Narragansett Bay and its Watershed and for freshwaters within individual River Basins, and reported as the percentage of state-assessed waters for aquatic life that were acceptable, impacted by excess nutrients and/or depleted oxygen, and impacted by other parameters for aquatic life as follows:

Table 2. List of parameters that are assessed by MassDEP and RIDEM for aquatic life to determine whether waterbodies are impacted by nutrient enrichment and/or depleted oxygen levels.

Massachusetts	Rhode Island
A total of 60 parameters[1] are tested across the state waters for *Fish, other Aquatic Life and Wildlife*	A total of 36 parameters[1] are tested across the state waters for *Fish and Wildlife Habitat*
The Estuary Program identified waterbodies as impacted by nutrient enrichment and depleted oxygen based on the following causes of impairment:	
✓ Total Phosphorus for freshwaters ✓ Total Nitrogen for estuarine waters ✓ Dissolved oxygen for freshwaters and estuarine waters ✓ Dissolve oxygen saturation for freshwaters ✓ Organic Enrichment (Sewage) Biological Indicators ✓ Nutrient/Eutrophication Biological Indicators	✓ Total Phosphorus for freshwaters ✓ Total Nitrogen for estuarine waters ✓ Dissolved oxygen for freshwaters and estuarine waters

Source: MassDEP and RIDEM – database for the 2014 Integrated Report.
(1) Not all parameters are tested on all waters.

- **Acceptable for Aquatic Life Use:** Waters that fully support aquatic life. This is equivalent to a "fully supporting" determination by the states. These are waters that, because of good water quality conditions, can be targeted for further protection. This assessment may be based on multiple parameters (Table 2).

  Acceptable waters are generally not tested for all parameters or indicators that may affect aquatic life use. However, the data available from the states for this analysis did not have information about which parameters were tested in the "fully supporting" waterbodies. As a result, it cannot be assumed that acceptable water quality conditions signify acceptable nutrient concentrations or dissolved oxygen levels that can support aquatic life. Therefore, by definition, this category-condition cannot be understood as directly complementary to the following two categories for impacted waters. On the other hand, the Estuary Program did compare relative extents given that these categories additively make up all waters that are state-assessed for aquatic life use.

- **Impacted by Nutrient Enrichment and Depleted Oxygen:** Waters that are "not supporting" of aquatic life due to impact by nutrient enrichment or depleted oxygen. These are waterbodies that, because of adverse conditions, can be targeted for restoration and to identify potential sources of nutrient enrichment that may be detrimental to aquatic life.

  While Table 2 shows all parameters where nutrients and depleted oxygen were the cause for not supporting aquatic life use (used for this analysis), the states may rely on multiple indicators (parameters) as evidence of nutrient enrichment, but are not listed as the primary cause or identified as such. Nutrients may be identified as a cause of impairment when (1) there is evidence of high productivity from multiple indicators such as pH, dissolved oxygen, and others, and (2) nutrient concentrations exceed the states' criteria (Extended Methods, Table 8). For example, nutrient enrichment is not considered problematic to aquatic life if nuisance primary producers or certain physico-chemical parameters that indicate high primary productivity (e.g., fluctuations of dissolved oxygen) are absent, even if nutrient concentrations exceed their recommended criteria (MassDEP 2012).

  Furthermore, note that the results of waters "impacted by nutrient enrichment and depleted oxygen" differs from the reporting by the states for impaired waters. The states test numerous physico-chemical parameters, toxicity, and other pollutants, and they monitor biological indicators that, in addition to nutrients and oxygen, can also be the cause of impairment for aquatic life use. Multiple parameters can impact the same waters (e.g., river segments). The Estuary Program did not analyze these other parameters comprehensively as they were not the focus of this chapter but were included within the "impacted by other parameters" category-condition.

- **Impacted by Other Parameters:** Waters that were "not supporting" of aquatic life due to parameters other than nutrient enrichment or oxygen depletion. Assessments for aquatic life use in Massachusetts include 60 parameters, and those in Rhode Island include 36 parameters (Table 2). Not all parameters are tested in all waters.

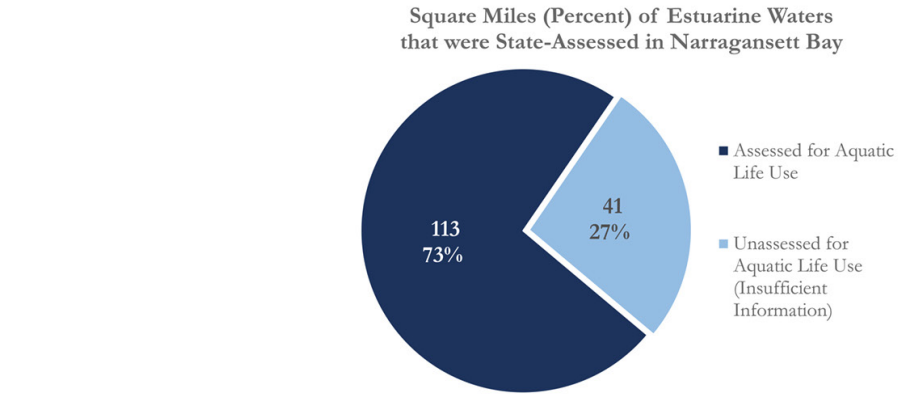
  Note that the extent of waters impacted by nutrient enrichment and dissolved oxygen, and other parameters only, differ from the reporting by the states for impaired waters. In this chapter, the "not supporting" waters for aquatic life use are broken down into these two category-conditions.

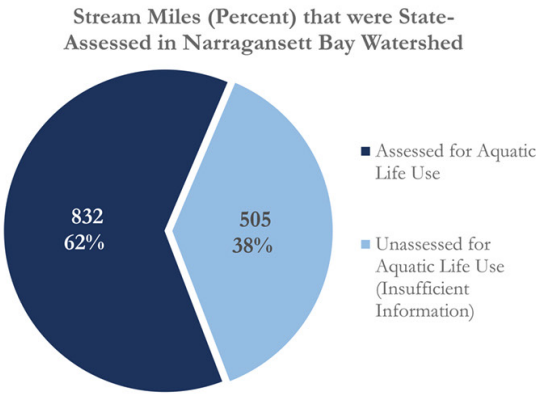
The Estuary Program used the Assessment Database for 2014 from each state to extract, synthesize, compute, and analyze the status of each water quality condition (MassDEP 2015, RIDEM 2015). From the Assessment Databases, data were selected and synthesized by River Basin, designated water use (aquatic life use), attainment (fully supporting and not supporting), and cause of impairment (nutrient enrichment, depleted oxygen or other parameters; see Table 2). An array of geospatial analysis (Esri 2016) was performed to calculate the total extent of streams and river segments and areas of fresh and estuarine waterbodies in the entire Narragansett Bay, its Watershed, and River Basins within the Watershed.

Status

THE EXTENT OF STATE-ASSESSED WATERS FOR AQUATIC LIFE

The Estuary Program found that 73 percent (113 square miles out of 156 square miles) of state-assessed estuarine waters were assessed for aquatic life use, compared to 62 percent of state-assessed streams and rivers and 58 percent of state-assessed lakes and ponds (Table 1, Figure 3).





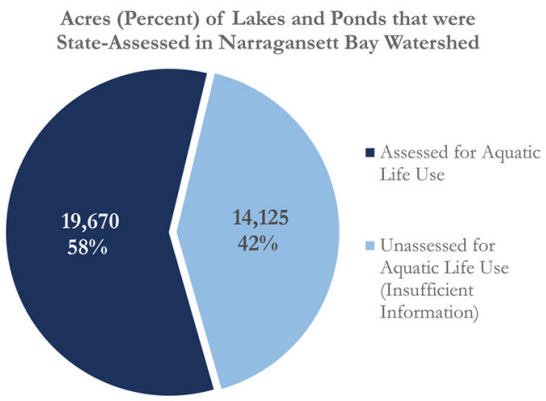


Figure 3. Proportion of state-assessed waters (not all waters, Table 1) that were either assessed or not assessed (due to insufficient data) for aquatic life use in Narragansett Bay and its Watershed.

Areas in the Bay that were not assessed for aquatic life but were assessed by the states for other uses include: the Sakonnet River, which constitutes ten percent (20.5 square miles) of Narragansett Bay's estuarine waters; Newport Harbor; Wickford Harbor; Barrington, Bristol, Kickemuit, and Warren Rivers; Potowomut River; Assonet River and Palmer River in Massachusetts; and coastal waters where two main wastewater treatment facility outfalls are located in the towns of South Kingstown and Narragansett, Rhode Island (Figure 1 and 4).

Of the total state-assessed waters by River Basin, Table 3 shows the total extent of freshwaters that were assessed and unassessed (conditions unknown) for aquatic life use (Figure 4).

The Pawtuxet River and Blackstone River Basins contained the greatest extent (over five thousand miles) and percentage (over seventy percent) of the total state-assessed waters that were assessed for aquatic life use (Table 3; Figure 3). In the Taunton River Basin and the Coastal Narragansett Basin respectively, half of streams and rivers (52 and 50 percent) and less than half of lakes and ponds acreage (39 and 35 percent) remained not assessed for aquatic life use (Table 4; Figure 3).

Table 3. Total (miles) of state-assessed streams and rivers that were either assessed or not assessed for aquatic life use in the Narragansett Bay Watershed's River Basins.

Streams and Rivers	Blackstone River	Pawtuxet River	Coastal Narragansett Bay	Taunton River
Assessed for Aquatic Life Use(1)	252	245	217	118
Unassessed for Aquatic Life Use(2)	98	85	213	108
Total State-Assessed Waters	*351*	*330*	*430*	*226*

(1) This is the total extent of waterbodies that have been assessed for aquatic life use by the states, which is further broken down into water quality conditions.
(2) This is the total extent of waterbodies that have been assessed by the states for other uses (e.g., recreational use) but not for aquatic life use, and thus conditions were unknown for aquatic life use.

Table 4. Total (acres) of state-assessed lakes and ponds that were either assessed or not assessed for aquatic life use in the Narragansett Bay Watershed's River Basins.

Lakes and Ponds	Blackstone River	Pawtuxet River	Coastal Narragansett Bay	Taunton River
Assessed for Aquatic Life Use(1)	5,041	5,707	3,672	5,251
Unassessed for Aquatic Life Use(2)	2,792	1,191	4,331	5,811
Total State-Assessed Waters	*7,833*	*6,897*	*8,003*	*11,062*

(1) This is the total extent of waterbodies that have been assessed for aquatic life use by the states, which is further broken down into water quality conditions.
(2) This is the total extent of waterbodies that have been assessed by the states for other uses (e.g., recreational use) but not for aquatic life use, and thus conditions were unknown for aquatic life use.

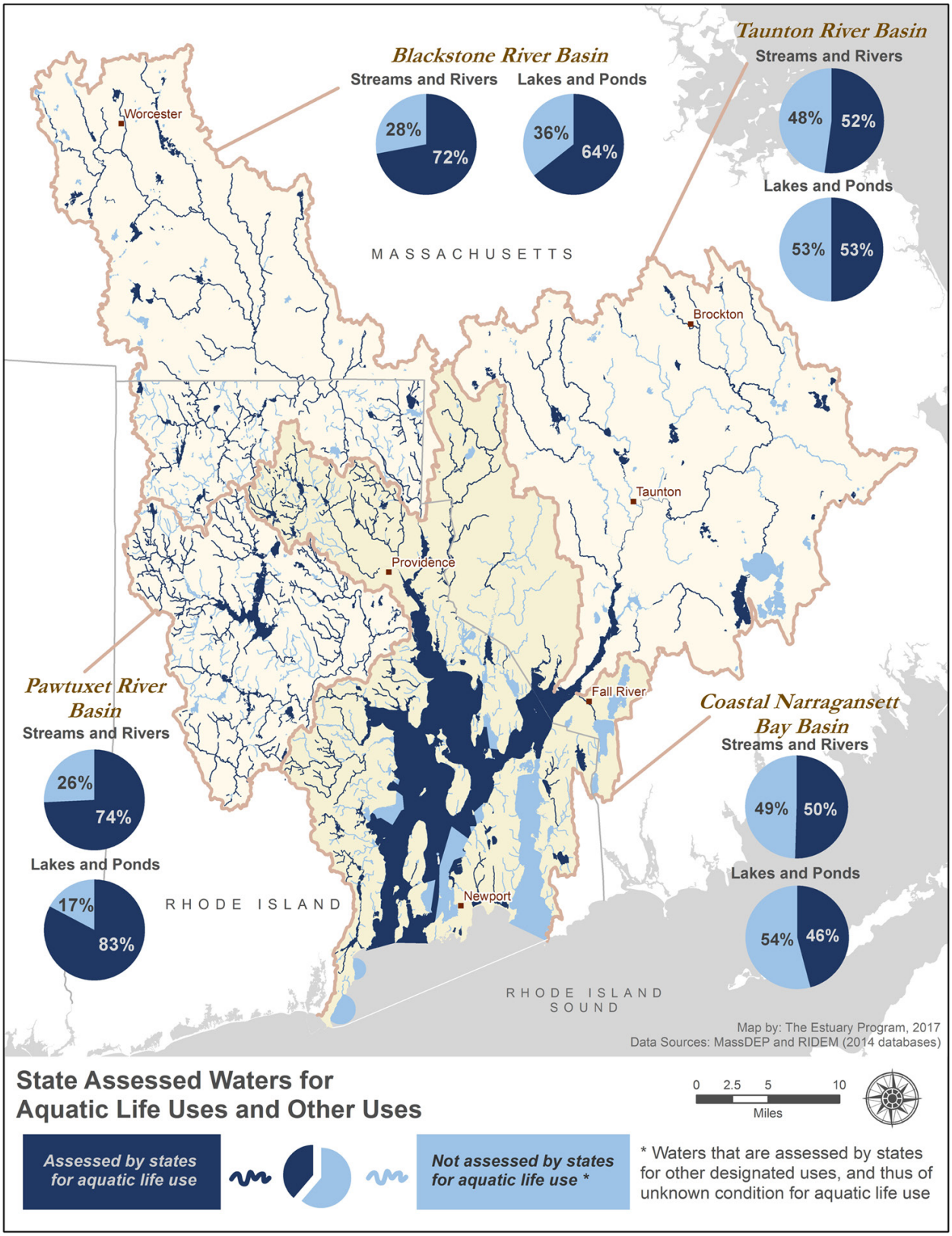


Figure 4. The extent and proportion of state-assessed waters that were either assessed or not assessed for aquatic life use in Narragansett Bay and its Watershed. For the waters assessed by the states for aquatic life use, the Estuary Program further calculated proportions with different water quality conditions (Table 6; Figure 6).

WATER QUALITY CONDITIONS FOR AQUATIC LIFE USE

Narragansett Bay and its Watershed

Of the 113 square miles of estuarine waters assessed by the states for aquatic life use (Figure 3), less than half were identified as acceptable for aquatic life use (49 percent) (Table 5; Figure 5). These areas were largely in the East and West Passages. Areas characterized as impacted by nutrient enrichment and/or depleted oxygen (50 percent) included most of what the Estuary Program defines as the Upper Bay, where tidal flushing is reduced. Impacted areas in the Upper Bay included the Providence River Estuary, a large part of Mount Hope Bay, and Greenwich Bay (Figure 6).

Of the 832 stream miles that were assessed by the states for aquatic life use (Figure 3), greater extents of streams and rivers (54 percent) were acceptable for aquatic life use than impacted by either nutrient enrichment/depleted oxygen or other parameters (19 and 26 percent, respectively) (Table 5; Figures 4 and 5). In contrast, of the 19,670 acres of ponds and lakes that were assessed by the states for aquatic life use (Figure 2), only one-quarter were acceptable for aquatic life, while one-quarter were impacted by nutrient enrichment or depleted oxygen and half by other parameters (Table 5; Figures 5 and 6).

River Basins

The Pawtuxet River Basin had the greatest relative extent of streams and rivers of all the state-assessed waters that are acceptable for aquatic life (77 percent) and the least relative extent (6 percent) impacted by nutrient enrichment and/or oxygen depletion. The remaining 17 percent of streams and rivers in the Pawtuxet River Basin were impacted by other parameters (Table 6, Figure 6).

Roughly half of the streams and rivers in the Coastal Narragansett Bay Basin and the Taunton River Basin (47 and 56 percent, respectively) were acceptable for aquatic life, with only 36 percent acceptable in the Blackstone River Basin. The Blackstone River Basin had the highest relative extent of waters impacted by nutrient enrichment or oxygen depletion, and most of these were within the Blackstone River and its network of tributaries (Table 6; Figure 6).

As with streams and rivers, the Pawtuxet River Basin also had the greatest extent of acceptable ponds and lakes (70 percent) (Table 7). In the Blackstone River, Taunton River, and Coastal Narragansett Bay Basins, ponds and lakes were significantly impacted by both nutrient enrichment and oxygen depletion and by other parameters.

Lakes and ponds seemed to be impacted primarily by parameters other than nutrient enrichment and/or depleted oxygen (Table 7). Even with these results, the Coastal Narragansett Bay Basin supported the least relative extent of ponds and lakes identified as acceptable for aquatic life use (8 percent) and greater than half of ponds and lakes (54 percent) impacted by nutrient enrichment and/or oxygen depletion (Figure 6). Meanwhile, roughly a quarter of lakes and ponds in the Blackstone River and Taunton River Basins were impacted by nutrient enrichment and oxygen depletion.

Extents of freshwaters impacted by nutrient enrichment and oxygen depletion were often smaller than extents impacted by other parameters (Tables 6 and 7; Figures 5 and 6). This is largely because (1)

Table 5. Extent of estuarine waters, streams and rivers, and lakes and ponds in Narragansett Bay and its Watershed by water quality conditions for aquatic life use.

Water Quality Conditions in Narragansett Bay and its Watershed	Estuarine Waters (Square miles)	Stream and Rivers (Miles)	Lakes and Ponds (Acres)
Total state-assessed waters for aquatic life use in Narragansett Bay Watershed	*113*	*832*	*19,670*
Acceptable for aquatic life use	55	450	5,056
Impacted by nutrient enrichment/depleted oxygen	57	162	4,853
Impacted by other parameters[1]	0.1	220	9,761

[1] Waters identified as impacted for aquatic life use for parameters other than nutrients and/or dissolved oxygen.



Figure 5. Proportion of the state-assessed waters for aquatic life use in Narragansett Bay and its Watershed that were acceptable, impacted by nutrient enrichment and/or depleted oxygen, or impacted by other parameters for aquatic life use.

Table 6. Total (miles) and percent of streams and rivers in the Narragansett Bay Watershed's River Basins by water quality conditions for aquatic life use.

Streams and Rivers by River Basins	Extent in miles (percent of state-assessed waters for aquatic life use)			
	Blackstone River	Pawtuxet River	Coastal Narragansett Bay	Taunton River
Total state-assessed for aquatic life use in Narragansett Bay Watershed	*252*	*245*	*217*	*118*
Acceptable for aquatic life use	92 (36)	190 (77)	102 (47)	66 (56)
Impacted by nutrient enrichment/depleted oxygen	78 (31)	14 (6)	43 (20)	27 (23)
Impacted by other parameters[1]	82 (33)	41(17)	72 (33)	25 (21)

[1] Waters identified as impacted for aquatic life use for parameters other than nutrient enrichment/depleted oxygen.

Table 7. Total (acres) and percent of lakes and ponds in the Narragansett Bay Watershed's River Basins by water quality conditions for aquatic life use.

Lakes and Ponds by River Basins	Extent in acres (percent of state-assessed waters for aquatic life use)			
	Blackstone River	Pawtuxet River	Coastal Narragansett Bay	Taunton River
Total state-assessed waterbodies for aquatic life use [1]	*5,041*	*5,707*	*3,672*	*5,251*
Acceptable for aquatic life use	729 (14)	4,022 (70)	305 (8)	- (0)
Impacted by nutrient enrichment/depleted oxygen	1,137 (23)	433 (8)	1,972 (54)	1,311 (25)
Impacted by other parameters[1]	3,175 (63)	1,252 (22)	1,395 (38)	3,940 (27)

[1] Waters identified as impacted for aquatic life use for parameters other than nutrient enrichment/depleted oxygen.

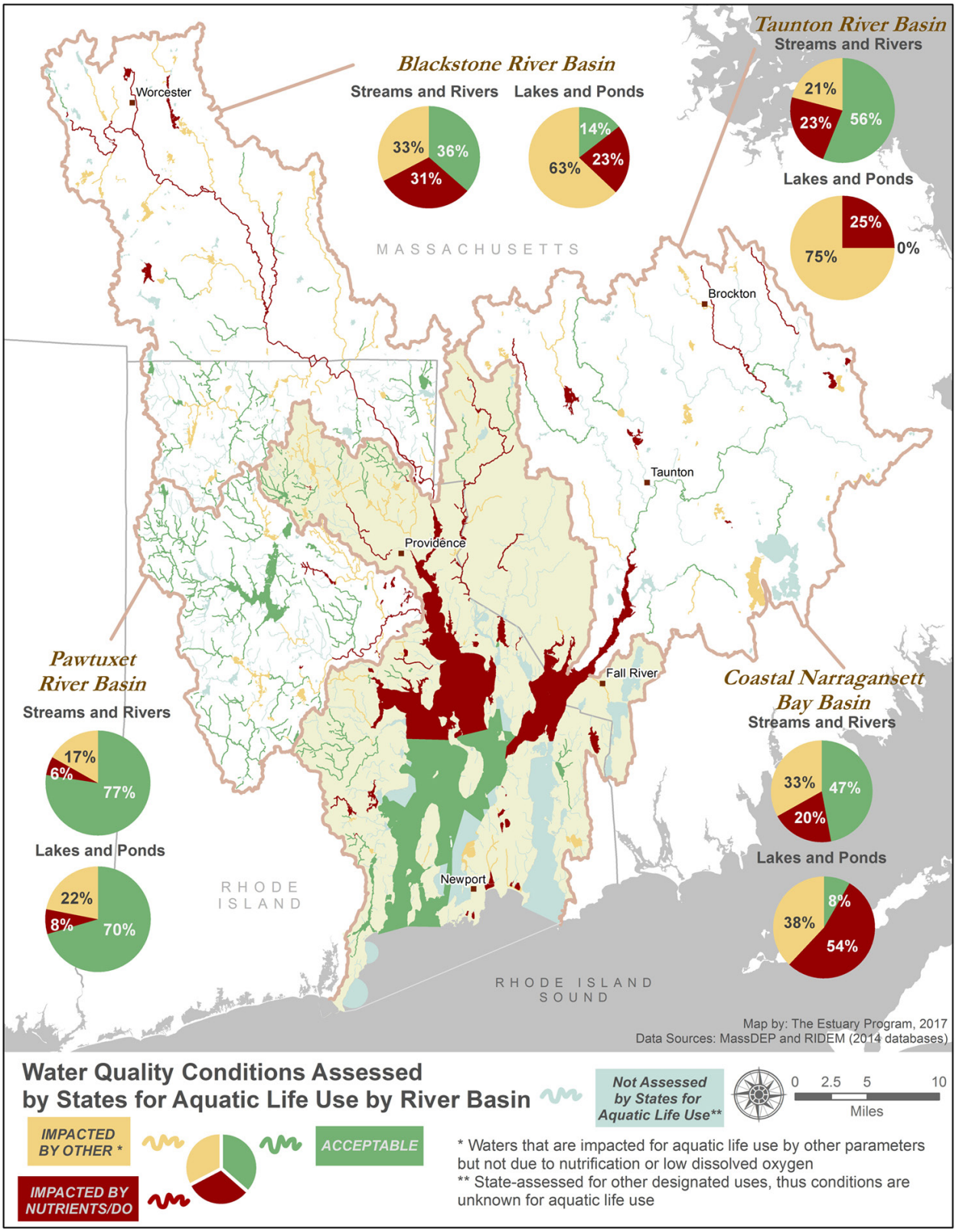


Figure 6. The extent and proportion of water quality conditions for aquatic life use the Narragansett Bay Watershed's River Basins. For each River Basin, pie charts show the percentage of fresh waters in each category-condition for recreational use (acceptable, impacted by nutrients or depleted oxygen, or impacted by other parameters).

water quality conditions for aquatic life have a large number of parameters tested in addition to the conventional parameters, and not in all waters (Table 2; MassDEP 2014 and RIDEM 2014 databases); (2) there may also be waters that have not yet been tested for nutrients concentrations or oxygen levels, reflecting a data gap in the state assessments; or (3) even if these waters were impacted by other parameters, and were also tested for nutrients and/or dissolved oxygen, the states did not identify these two parameters to be the cause of impairment to aquatic life, either because (3a) they did not exceed or meet the state criteria for nutrients and dissolved oxygen (Extended Methods, Table 8) or (3b) nutrient enrichment was not identified as problematic for the functions of the aquatic ecosystem.

Discussion

Clean streams, rivers, lakes, ponds, and estuarine waters in Narragansett Bay and its Watershed are essential to supporting aquatic communities. In the estuarine waters of the Bay, of the waters that have been assessed by the states for aquatic life use, the East and West Passages (spanning from the mouth of the Bay to the northern tip of Prudence Island) were the only sections of the Bay found to have acceptable water quality conditions for aquatic life use based on any of the parameters tested by states (Figure 6). Within the Narragansett Bay Watershed, the Pawtuxet River Basin was characterized by the greatest relative extent of acceptable streams and rivers as well as ponds and lakes, whereas the Taunton River, Blackstone River, and Coastal Narragansett Bay Basins had relatively small extents of acceptable fresh waters (Figure 6).

Evaluation of overall water quality conditions across Narragansett Bay and its Watershed illustrates that, at a minimum, large sections of the Bay and many of the major rivers flowing into the Bay have been determined as impacted for aquatic life use. The Estuary Program found that one-third of all state-assessed estuarine waters in Narragansett Bay suffered from excess nutrients and/or depleted oxygen. These waters occurred in the Upper Estuary (Providence River and Seekonk River Estuary, Greenwich Bay, Taunton River and Mount Hope Bay, and other small embayments), where nutrient concentrations were high and with other factors cause nutrient enrichment that can affect aquatic life. These findings align with evidence of point source nutrient loading from wastewater treatment plants (see "Nutrient Loading" chapter) and depleted levels of dissolved oxygen in the Bay (see "Dissolved Oxygen" chapter).

In the Narragansett Bay Watershed, only nineteen percent of streams and rivers and 25 percent of lakes and ponds of the state-assessed waters for aquatic life use were identified as impacted by excess nutrients and/or depleted oxygen. The Pawtuxet River Basin had the lowest percentage of fresh waters impacted (less than ten percent of the state-assessed waters in the River Basin), whereas the Blackstone River Basin contained the greatest percentage of streams and rivers impacted by nutrient enrichment/depleted oxygen (31 percent). In fact, the entire 52-mile length of the Blackstone River was identified as impacted by nutrient enrichment/depleted oxygen. In addition, many of the tributaries in the upper reaches of the Blackstone River Basin that were also assessed for aquatic life use were impacted by other parameters (Figure 6).

Recent substantial reductions in nutrient loadings have been realized with a reduction of more than 50 percent achieved by major wastewater treatment facilities (WWTF) in the Watershed (see "Nutrient Loading" chapter), and the impacts of these reductions are still being assessed by the states and researchers. It is noteworthy to emphasize that while the Estuary Program used the database for the 2014 state assessments, the assessments may not reflect data available after 2013. When the states present assessment results in the Integrated Reports, the data and time period being assessed lag behind the year of the assessment cycle. Because it takes more than a year to conduct the assessments, the states generally use data from the five prior years. For example, the Rhode Island 2014 water quality assessments utilized data that was primarily from 2008 to 2013. There has not been adequate time to realize any meaningful response to the nutrient reductions, and moreover, the state assessment data are more suitable to track longer-term changes in water quality conditions. The process is not designed for, nor does it support, the identification of short-term changes (RIDEM, personal communication, June 2017). For example, notwithstanding the significant nutrient loading reductions achieved in wastewater treatment facilities in the Upper Bay (e.g., Field's Point WWTF), this area was designated as impaired for aquatic life due to dissolved oxygen. Once a waterbody has been listed as impaired, it will be delisted only when there is sufficient new water quality data to do so, and delistings require EPA approval. In the case of the Upper Bay, there is a need for multiple years of data to assess the inherent interannual variability and to identify more persistent trends. Recent data collected by RIDEM will not support a delisting as the conditions have not yet returned to full compliance with the dissolved oxygen water quality standards

(Extended Methods, Table 8), and while preliminary results (see "Dissolved Oxygen" chapter) showed that recent years in the Upper Bay had less duration of hypoxia, these improvements in dissolved oxygen concentrations represent only partial progress toward meeting water quality criteria. While RIDEM expects that conditions have improved, the Upper Bay area designated as impaired for aquatic life due to dissolved oxygen needs continued monitoring to properly characterize the water quality response and remains the target of management attention. RIDEM was expecting that a better assessment of change in the Upper Bay due to nutrient reductions relative to dissolved oxygen would occur with the 2016 assessment report, which was nearing completion at the time this chapter was written (RIDEM, personal communication, June 2017).

Similarly, the Blackstone River stem is impacted due to nutrients and dissolved oxygen, but upgrades in the Upper Blackstone WWTF have resulted in substantial nutrient loading reductions to the river. It is important to remember that the periodic assessments by the states are based on earlier data and may not reflect recent changes. For example, RIDEM was undertaking monitoring in the Blackstone River in summer 2017 to assess whether a delisting for dissolved oxygen would be appropriate (RIDEM, personal communication, June 2017).

It may be expected that persistent nutrient and dissolved oxygen impairments to water quality could have negative impacts on stream invertebrates. However, the Estuary Program does not recommend comparing the results from the stream invertebrate analysis (see "Stream Invertebrates" chapter) to the aquatic life use assessment. Stream invertebrate samples are collected from wadeable streams, while the majority of river miles assessed for aquatic life are non-wadeable (RIDEM, personal communication, June 2017).

In practice, water quality assessments allow states to identify impacted waters in order to develop restoration plans and, conversely, to determine waters with good water quality that can be protected from degradation through conservation efforts. Waterbody characterizations can be used strategically to identify potential protection, restoration, and mitigation projects, including the implementation of stormwater best management practices, low-impact development practices, protection of riparian buffers and floodplains, and the preservation of all natural and ecologically significant lands (see "Open Space" chapter).

While significant progress has been made to address point source nutrient loadings, non-point sources of nutrients, including discharges of stormwater runoff, require additional management practices to achieve effective pollution control. Furthermore, pressures exerted by increasing urbanization, changing land use practices, and altered precipitation and temperature regimes associated with climate change will continue to pose challenges for protecting and improving water quality conditions for aquatic life.

Data Gaps and Research Needs

- Bi-state coordination between MassDEP and RIDEM could improve and streamline water quality sampling of specific streams/rivers/lakes that share state boundaries in order to provide data that can reflect the most current water quality conditions of individual state-assessed waterbodies. However, limitations by the states and the nature of the assessments, including those discussed in this chapter, should be considered.
- Coordination between the Estuary Program and state partners is needed to share data and facilitate the tracking of this indicator by linking the time of sampling (year, season) and assessment for each water body or area of water. In addition, new listings and delistings of water impairments need to be tracked as they occur between water quality assessment cycles, with the goal of quantifying changes over time. These changes can shed light on improvements or declines in water quality for aquatic life due to increased or reduced nutrient loadings.
- Many different entities, particularly watershed NGOs and universities, monitor and routinely collect data on nutrients and dissolved oxygen parameters at varying frequencies (e.g., monthly) and scales (e.g., Taunton River watershed). Further evaluation is needed to determine whether water quality data from these efforts could be reconciled, combined, and standardized with the state datasets to improve temporal and spatial coverage for this indicator.
- Research is needed to understand how landscape stressors (e.g., impervious cover, land use) and climate change stressors (e.g., precipitation, temperature) relate to increases in nutrient enrichment in waterbodies that can result in eutrophication and hypoxia events, harmful to aquatic life, in fresh waters of the Watershed and estuarine waters of the Bay. This should be explored on a variety of scales from larger watersheds to individual catchment areas.

- There is a need to develop or utilize available tools to allow evaluation of the efficacy of stormwater management practices, including retrofitting of existing infrastructure, at appropriate scales (e.g., sub-Basin). This includes practices designed to treat/retain nitrogen and phosphorus loadings as well as those designed to address peak flows, as precipitation exacerbates the impacts of nutrient enrichment.
- While cyanobacteria blooms are primarily a public health issue, monitoring cyanobacteria blooms in fresh waters and other harmful algal blooms in marine waters is needed. Data on harmful algal blooms, including an inventory of waterbodies with a history of blooms, frequency of events, and other parameters measured during these events, are needed to augment understanding of (a) the causes and consequences of blooms, (b) the dynamics of bloom suppression, and (c) whether nutrient enrichment, oxygen depletion, low stream flows, water levels or flushing, or high water temperatures, or a combination of those factors, can result in or be used to predict blooms.

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Table 8. MassDEP and RIDEM's narrative and numeric criteria for nutrients and dissolved oxygen parameters used to assess aquatic life use and that may cause "not supporting" conditions.

Parameters/Causes	Massachusetts (MassDEP)[1]	Rhode Island (RIDEM)[2]
Nutrients	**Streams:** total phosphorus should not exceed 50 ug/l (0.05 mg/l) **Lakes and ponds:** total phosphorus should not exceed 25 ug/L (0.025 mg/l) **Estuaries:** a seasonal average (n>3 samples) of the total nitrogen concentration data collected during an ebb tide will be screened against the MEP critical indicator threshold of >0.5 mg/l for waters where eelgrass habitat has not been documented.	There are no numeric criteria for nutrients in rivers or estuarine waters, **nutrient enrichment is considered an impairment when**: Evaluations of persistent, potentially severe eutrophication and/or low DO, where **total phosphorus** listed as the suspected cause in **freshwater rivers** and **total nitrogen** listed as the suspected cause in **estuaries**. See Table 1 of RIDEM Water Quality Regulations[3]. **Lakes:** **Average Total Phosphorus** shall not exceed 0.025 mg/l in any lake/pond; in tributaries ***at the point where they enter such bodies of water*** shall not cause exceedance of this phosphorus criteria, except as naturally occurs, or different value for phosphorus is necessary to prevent cultural eutrophication. Also, lakes and ponds that exhibit persistent elevated chlorophyll *a* and/or recurring cyanobacteria blooms, with seasonal average TP less than 25 ug/l, may be assessed as impaired for nutrients, except as naturally occurs.
Dissolved Oxygen	**Cold water fishery:** ≥6.0 mg/l **Warm water fishery:** ≥5.0 mg/l Based on the national criteria [3] daily minima, **(1.0 mg/L less than the 7-day mean), DO is considered an impairment when:** **Stream and Rivers:** frequent (typically >10%), prolonged and/or consecutive or not-consecutive low DO measurements, from criteria. **Lakes:** if the criterion is not meet at all depths for >10% of the lake surface during the summer season. **Estuaries:** frequent (typically >10%), prolonged and/or consecutive or not-consecutive low DO measurements, from criteria.	**Freshwater** DO criteria are based upon cold water and warm water fish habitat. **Cold water fish habitat:** DO content of not less than 75% saturation, based on a daily average, and an instantaneous minimum DO concentration of at least 5 mg/l, except as naturally occurs. **Warm water fish habitat:** DO content of not less than 60% saturation, based on a daily average, and an instantaneous minimum DO concentration of at least 5.0 mg/l, except as naturally occurs. **Estuaries** with a DO concentration above an instantaneous value of 4.8 mg/l shall be considered protective of Aquatic Life Uses. When instantaneous DO values fall below 4.8 mg/l, the waters shall not be: 1) Less than 2.9 mg/l for more than 24 consecutive hours during the recruitment season; nor 2) Less than 1.4 mg/l for more than 1 hour more than twice during the recruitment season; nor 3) Shall they exceed the cumulative DO exposure presented in Table 3.A. of RIDEM Water Quality Regulations[3].

[1] MassDEP CALM 2016.
[2] RIDEM CALM 2014.
[3] RIDEM Water Quality Regulations, 2010.
[4] Quality Criteria for Water (USEPA, 1986).



State of Narragansett Bay and Its Watershed
2017 Technical Report

Watershed Ecosystem Condition Indicators

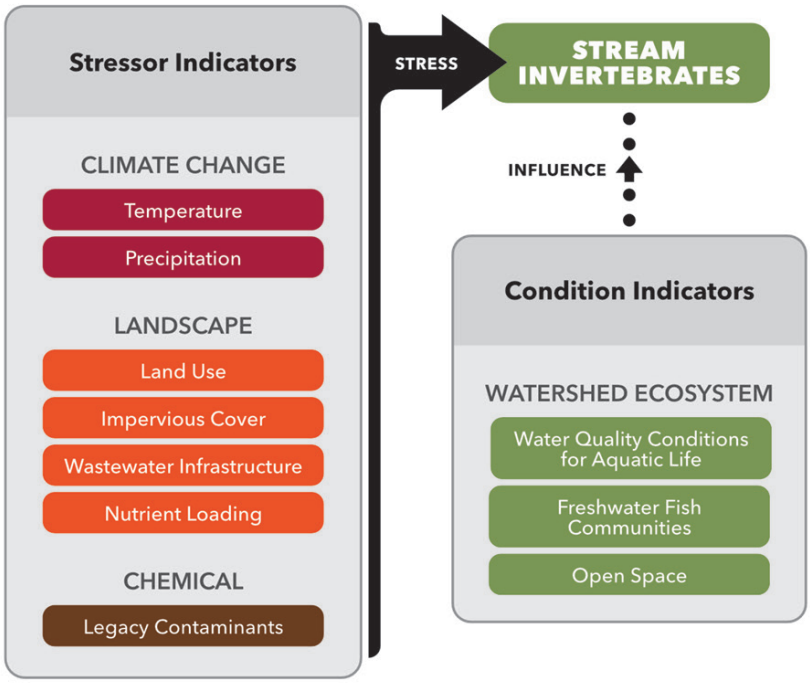
CHAPTER 19: STREAM INVERTEBRATES

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Photo: Flat River, Exeter, RI (Ayla Fox)

NARRAGANSETT BAY ESTUARY PROGRAM

Overview



BACKGROUND

- Insects, worms, snails, mussels, and crayfish—known collectively as stream benthic macroinvertebrates—are excellent indicators of water quality conditions. They rely on intact habitat and clean water, and state agencies monitor these invertebrates in wadeable streams throughout the Narragansett Bay Watershed. The assemblage of stream invertebrates that live at a particular site reflects the effects of both short-term and cumulative stressors such as stormwater discharges, increased water temperatures, excess nutrients, sedimentation, and other physical alterations of stream habitat, all driven by changes in climate and land use.

KEY FINDINGS

- ***Status***

  Streams with Good Macroinvertebrate Health and Good Habitat Quality: Of the 78 sites assessed for stream invertebrates between 2002 and 2014, 62 sites (79 percent) were classified as having both good macroinvertebrate health and good habitat quality. The Pawtuxet River Basin appeared to have the healthiest stream conditions with nineteen of its 21 sites (90 percent) in this classification. However, the other three River Basins also had 67 to 80 percent of their sites in this classification.

Streams with Poor Macroinvertebrate Health and Good Habitat Quality: Nineteen percent of sites in the Watershed had poor macroinvertebrate health but good habitat quality. The Pawtuxet River Basin had 10 percent of its sites in this classification, and the other three River Basins had between 20 and 27 percent. Forty percent of all the sites in the Watershed with poor macroinvertebrate health were located in the Blackstone River Basin.

Streams with Poor Macroinvertebrate Health and Poor Habitat Quality: A single site was classified as having both poor macroinvertebrate health and poor habitat quality, in the Taunton River Basin.

Introduction

Stream benthic macroinvertebrates are a large group of animals (insects, worms, snails, mussels, and crayfish) that dwell in fresh water. In a comprehensive review, Wallace and Webster (1996) reported that stream invertebrates serve as prey for fish, amphibians, mammals, birds, waterfowl, and other wildlife; influence nutrient cycles, primary productivity, and decomposition of organic matter (e.g., fallen leaves); and serve as important indicators of stream condition. Some freshwater invertebrates such as crayfish, mussels, worms, and snails spend their entire life in the water, while insects such as dragonflies, damselflies, mayflies, and beetles spend only their early life stages in streams before emerging from the water to fly. Invertebrates depend on healthy stream habitats, sufficient amounts of water, and adequate water quality. When stream habitat, water quality, or water quantity begin to degrade, the macroinvertebrate community is affected. Accordingly, invertebrates in freshwater streams are monitored to indicate ecological condition (Plafkin et al. 1989, Barbour et al. 1999).

Stream macroinvertebrates are impacted by stormwater runoff, water pollution, changes in water temperature and flow, erosion, sedimentation, and habitat degradation (Thorp and Covich 1991). Landscape stressors that influence the status of stream invertebrates include increased impervious cover (Morse et al. 2003), land use such as urbanization (Moore and Palmer 2005), and nutrient loading (Yuan 2010). Urbanization adversely affects stream ecosystems, and evidence is mounting that even low levels of urbanization can cause significant changes in stream biology, physical habitat, and chemistry (Morse et al. 2003, Yuan 2010, Coles et al. 2014). Large increases in impervious cover, together with increases in storm drains and channelization, can result in increases in the velocity and amount of water flowing in streams. An analysis of the relationship between stream insects and impervious cover in Maine streams found that increasing impervious cover (greater than six percent) was related to a decline in the taxonomic richness of the stream insect community (Morse et al. 2003). Of the 52 subwatersheds (HUC12) in the Narragansett Bay Watershed, 44 subwatersheds had over six percent impervious cover (see "Impervious Cover" chapter).

Land use changes, contributing increased pollutants such as sediment, nutrients from fertilizers, and other contaminants, have been linked to numerous types of changes in streams, such as (1) hydrology, including the amount, movement, and distribution of water, (2) physical habitat, the actual structure of the stream that is home for organisms (biota), such as invertebrates and fish, and (3) chemistry. While the mechanisms for these effects are understood, the response of streams to the multiple stressors associated with urbanization are complex. To support management, further investigation and analysis are needed to better characterize the effects of various stressors on stream conditions and to better understand the range of response in wadeable streams to urbanization in the Narragansett Bay Watershed (USGS 2004, Coles et al. 2014). Climate change also affects stream invertebrates through changes in precipitation patterns and rising temperatures (Poff et al. 2010, Fengqing 2013 et al.).

Aquatic organisms that dwell in riverine systems respond to environmental and biological interactions,

influencing the species assemblage of fish (see "Freshwater Fish Communities" chapter) and invertebrates (Annear et al. 2004). Since macroinvertebrates live months and even years in a stream, they encounter a wide range of stressors throughout the seasons. The macroinvertebrate communities assimilate the effects of different pollution sources over time, providing an aggregate or cumulative measure of stressors (Plafkin et al. 1989). If a stream macroinvertebrate community lacks pollution-sensitive species or is dominated by pollution-tolerant species, it is generally indicative of poor water quality conditions and stream habitat disturbance. Macroinvertebrate communities sampled from degraded waters usually have low species richness and abundance. Therefore, sampling the resident communities provides a stable representation of constantly fluctuating environmental conditions (Rosenberg and Resh 1996). Quantifying macroinvertebrate diversity and abundance can serve as an indicator to characterize the general conditions of a watershed. An example of different macroinvertebrate communities indicating various stream conditions is shown in Figure 1.

Because benthic macroinvertebrate communities reflect water quality and aquatic habitat disturbance over time and are relatively easy to collect and identify, biomonitoring is a cost-effective screening tool to monitor for general stressors (Barbour et al. 1999). In fact, stream macroinvertebrates are the most widely used biological assemblage in monitoring conducted by state water resource agencies (USEPA, undated). In New England, state agencies have used various methods of biomonitoring to evaluate rivers and streams (Shelton and Blocksom 2004). The Massachusetts Department of Environmental Protection (MassDEP) began collecting macroinvertebrate, habitat, and physical-chemical data from wadeable streams (streams shallow enough for the analyst to safely walk into with chest waders) beginning in 1983, while the Rhode Island Department of Environmental Management (RIDEM) began a wadeable

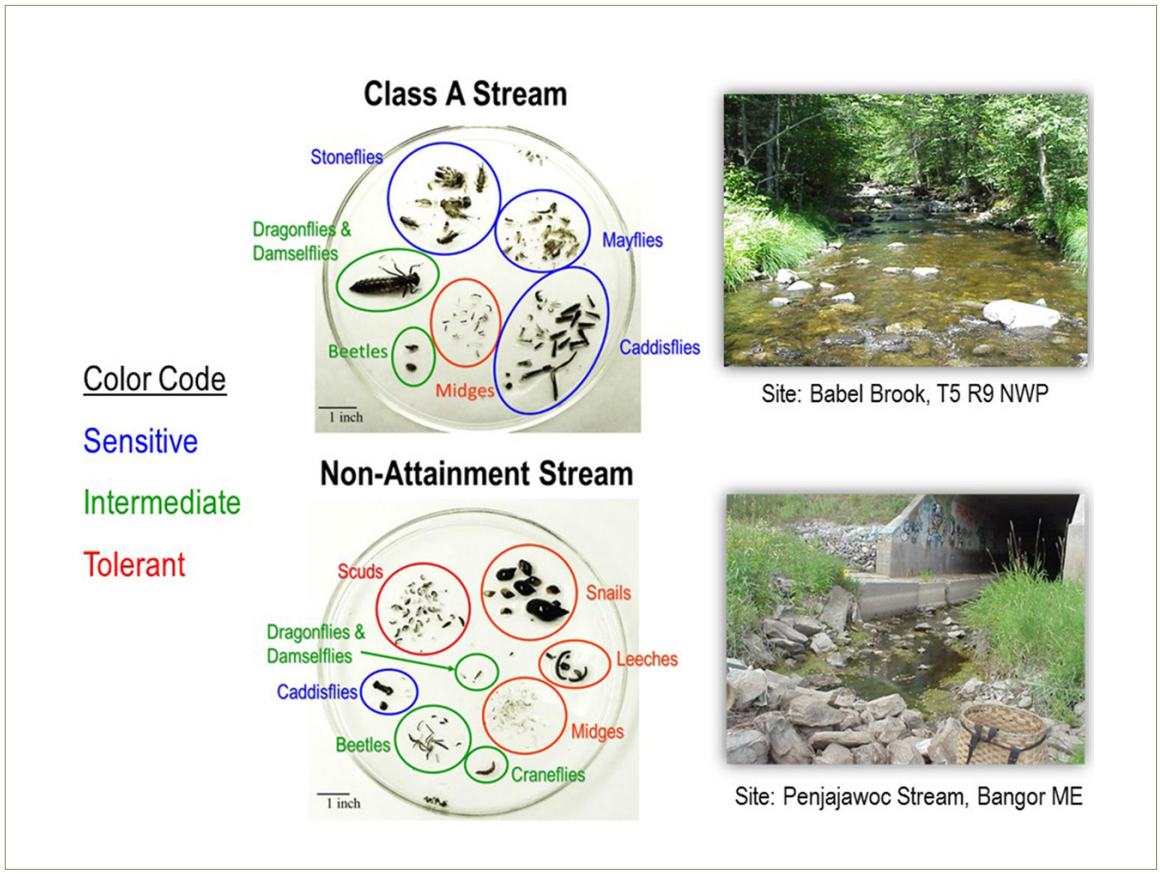


Figure 1. Example of stream macroinvertebrate communities as indicators of water quality conditions and stream habitat disturbance. Credit: Tom Danielson, Maine DEP.

stream macroinvertebrate sampling program in 1991 in cooperation with Roger Williams University (RIDEM 2005).

Sampling protocols have evolved over the years, but wadeable stream methodologies are generally based on the EPA's Rapid Bioassessment Protocols (Barbour et al. 1999) to measure local habitat features (e.g., physical structure, flow regime), water quality parameters, and macroinvertebrate communities (Nuzzo 2003, ESS Group, Inc. 2014). The quality of in-stream habitat and surrounding buffer habitats are major factors influencing the health and diversity of stream invertebrates (Plafkin et al. 1989). Each state agency's quality assurance and quality control documents describe their specific methods, as does the Consolidated Assessment and Listing Methodology for the Preparation of the Integrated Water Quality Monitoring and Assessment report (MassDEP 2016, RIDEM 2014).

In this chapter, the Narragansett Bay Estuary Program reports on the number of sites in the Narragansett Bay Watershed that were characterized as either "good" or "poor" based on macroinvertebrate health and habitat quality by the state agencies, MassDEP and RIDEM. The Estuary Program and partners simplified the state classification-categories for benthic macroinvertebrate health and habitat quality and then combined macroinvertebrate health and habitat quality into a general indicator of stream condition in the Narragansett Bay Watershed. Because the Estuary Program relied on data from state agencies in Massachusetts and Rhode Island, the categorization scheme used in this chapter was developed through a bi-state effort involving MassDEP and RIDEM. Massachusetts and Rhode Island use different metrics in their respective indices of biological integrity (IBIs) (Nuzzo 2003, ESS Group, Inc. 2014), but the resulting classification categories are similar. For example, the numeric thresholds used by RIDEM correspond to the same qualitative descriptors used by MassDEP. It is important to highlight that the individual and collective metrics used by each state for their IBIs were not the focus of this indicator.

Methods

Habitat and invertebrate data are collected from streams and used by both Massachusetts and Rhode Island to assess stream conditions relative to state water quality criteria. Using a net, invertebrate samples are collected from stream sites and then brought to the lab for taxonomic identification and enumeration. Habitat quality is evaluated visually following the Rapid Bioassessment Protocol, scoring ten parameters that describe in-stream habitats, stream channel morphology (shape), and structure of the stream banks and buffers (Barbour et al. 1999). This information is supplemented with data on additional physical characteristics of the site observed during monitoring: surrounding land use; presence or absence of dams; local stream erosion; potential non-point source pollution; stream width, depth and flow; inorganic and organic substrate types; and presence of odors, oils and deposits (Plafkin et al. 1989), as observed in the field and relying on best professional judgment.

The states calculate stream macroinvertebrate health according to each state's Index of Biotic Integrity (IBI) data and habitat quality based on state scores, and this information is used as part of each state's Integrated Water Quality Monitoring and Assessment Report (MassDEP 2012 and 2016, RIDEM 2014). The Narragansett Bay Estuary Program summarized the data from state agencies to categorize stream macroinvertebrate health and habitat quality.

Habitat measurements used in the methods in this chapter are not the results reported under the official state "Fish & Wildlife Habitat" in the 303(d) state assessment reports. Each state uses several types of data (habitat, invertebrates, and multiple water quality parameters) to assess if a waterbody is meeting its "Fish & Wildlife Habitat" use. However, the metrics used by each state vary. Therefore, the category used by the Estuary Program to determine habitat quality condition should not be compared against the determinations made by the states for aquatic life use.

It is also important to keep in mind that site selection for monitoring followed the states' protocols for sampling stream macroinvertebrates. In general, the streams are characterized by riffles, and in Rhode Island, these sites are within the Southern New England Coastal Plains and Hills ecoregion (Griffith et al. 2009, Figure 2). Riffles are characterized by shallow depths and streambeds composed of cobbles and other coarser materials over which the water flows, whereas pools are deeper areas of streams characterized by finer materials such as silt. Riffles are considered to contribute to healthy aquatic environments because the turbulent flow leads to high concentrations of dissolved oxygen.

This indicator quantifies the number of sites by stream condition level for sites in the Massachusetts and Rhode Island portions of the Narragansett Bay Watershed.

STREAM MACROINVERTEBRATE HEALTH

Data collected through sampling the macroinvertebrate assemblage allows for the calculation of total taxa, percent of dominant taxa, and other metrics that can be used to help distinguish overall stream condition. Based on earlier work, both MassDEP and RIDEM have selected certain metrics that are combined to form an Index of Biotic Integrity (IBI). The IBIs allow an invertebrate sample to be scored using a scale in which higher scores represent better stream conditions. Although Rhode Island and Massachusetts use differing metrics and equations, both states have threshold values that they use to categorize the invertebrate sample results. Rhode Island refers to the categorization as a Biological Condition Category, while Massachusetts uses a Rapid Bioassessment Protocol Status (Table 1).

For this indicator's metric of good or poor macroinvertebrate health, the Estuary Program combined the state data using thresholds for stream macroinvertebrate health because 1) it was determined by both state agencies that the descriptors used by MassDEP to classify the IBI results are comparable with the thresholds used by RIDEM, 2) it was not within the scope of this chapter to reconcile metrics from different IBIs, and 3) a classification-category can provide a useful picture of the stream conditions characterized by macroinvertebrate health and habitat quality, as discussed below. For results of metrics and IBIs as developed by each state, the reader should contact MassDEP and/or RIDEM.

Stream macroinvertebrate data available for this indicator were collected from 2002 to 2014 in Rhode Island and from 2006 to 2009 in Massachusetts. Both state agencies have monitored different basins each year and over time sampled throughout the Watershed, with a somewhat higher density of sampling sites located in Rhode Island. The implications of the temporal differences in the data are discussed further below.

In 2002, RIDEM expanded its biological monitoring program to begin collecting data using EPA's Rapid Bioassessment Protocol identifying macroinvertebrates at a higher taxonomic resolution (ESS 2014). In 2004, the monitoring station selection process was expanded to incorporate a rotating basin approach. More recently, RIDEM began moving toward development of a new assessment method using a biotic index to compare stations against a reference condition model, known as the Multi-Metric Biological Condition Index (abbreviated as MBCI in tables). The MCBI is a tool to classify stream health relative to the reference condition (Tetra Tech 2011) for stations located in the Southern New England Coastal Plains and Hills ecoregion (Griffith et al. 2009; Figure 2).

In Massachusetts, MassDEP shifted in 2010 from targeted sampling in each watershed (each with its own reference, or "least disturbed," site). The Massachusetts Probabilistic Monitoring and Assessment Program (MAP2) uses data from randomly selected sites to generate an unbiased assessment of water quality conditions throughout the state (MAP2 2017). In conjunction with this approach, the agency in 2011 developed a Reference Site Network using statewide probabilistic sampling of basins on a rotating basis (Nuzzo, personal correspondence). From these data, the agency intends to develop metrics to distinguish between healthy and unhealthy aquatic ecosystems, rather than selecting a reference site for each watershed each time sampling is conducted. However, the most recent data from the five-year cycle of probabilistic analysis between 2010 and 2015 were not available at the time this chapter was written.

Table 1. The Narragansett Bay Estuary Program's categories of Good and Poor Macroinvertebrate Health, derived from and reconciled with each state's Index of Biotic Integrity categories.

Macroinvertebrate Health Level	MassDEP Rapid Bioassessment Protocol Status	RIDEM IBI Thresholds
Good Macroinvertebrate Health	Not Impacted	>85
	Slightly Impacted	56–85
Poor Macroinvertebrate Health	Moderately Impacted	36–55
	Severely Impacted	<35

HABITAT QUALITY

Both MassDEP and RIDEM use the same habitat assessment protocol to observe and quantify each of the ten habitat parameters at each site, as listed below, to report a total habitat score, on a scale of 0 to 20 or 0 to 10, depending on the parameter (Nuzzo 2003, ESS 2014):

1. Instream Cover (Fish)
2. Epifaunal Substrate
3. Embeddedness
4. Channel Alteration
5. Sediment Deposition
6. Frequency of Riffles/Velocity-Depth
7. Channel Flow Status
8. Bank Vegetative Protection
9. Bank Stability
10. Riparian Vegetative Zone Width

These scores can be grouped into four categories (from optimal to poor habitat condition) to describe the general habitat conditions at a sampling station (Table 2). Using those scores, the Estuary Program categorized habitat quality as Good or Poor (Table 2).

STREAM CONDITION LEVEL COMBINING MACROINVERTEBRATE HEALTH AND HABITAT QUALITY

The Estuary Program and partners simplified the state classification-categories for benthic macroinvertebrate health and habitat quality (Tables 1 and 2) and then combined macroinvertebrate health and habitat quality into a general indicator of stream condition in the Narragansett Bay Watershed. As shown in Table 3, the Estuary Program's indicator has three categories:

- Good Macroinvertebrate Health and Good Habitat Quality
- Poor Macroinvertebrate Health and Good Habitat Quality
- Poor Macroinvertebrate Health and Poor Habitat Quality

Macroinvertebrate data for Rhode Island were from 2002 to 2014, and those for Massachusetts were from 2006 to 2009. To identify sites from the statewide macroinvertebrate monitoring programs that are located within Narragansett Bay Watershed, the latitude and longitude for each sampling site were mapped, and associated with River Basins, HUC10 watersheds, and HUC12 subwatersheds, by means of geospatial analysis (Esri 2016; see the Appendix for definitions, lists and maps of River Basins, HUC10 watersheds, and HUC12 subwatersheds). Because macroinvertebrate monitoring follows the timeline of each state's strategy for water quality assessment, using a basin-based rotation approach, the majority of the sampling sites were not monitored more than once within the data collection period. Thus, a trend analysis was not possible due to lack of temporal data for all sites. Because the data were collected over a period of twelve years and many streams were

Table 2. The Narragansett Bay Estuary Program's categories of Good and Poor Habitat Quality, derived from the Rapid Bioassessment Protocol (USEPA) utilized by Massachusetts and Rhode Island.

Habitat Quality Levels	State Habitat Condition Categories	State Habitat Total Score Thresholds
Good Habitat Quality	Optimal Habitat(1)	>150
	Suboptimal Habitat(2)	101–150
Poor Habitat Quality	Marginal Habitat(3)	51–100
	Poor Habitat(4)	<51

(1) Optimal Habitat scores range between 16 to 20, or 9 to 10, depending on the scale that corresponds to each parameter.
(2) Suboptimal Habitat scores range between 11 to 15, or 6 to 8, depending on the scale that corresponds to each parameter.
(3) Marginal Habitat scores range between 6 to 10 or 3 to 5, depending on the scale that corresponds to each parameter.
(4) Poor Habitat scores range between 0 to 5, or 0 to 2, depending on the scale that corresponds to each parameter.

Table 3. The Estuary Program's three stream condition levels used to categorize monitoring sites: Good Macroinvertebrate Health and Good Habitat Quality (blue); Poor Macroinvertebrate Health and Good Habitat Quality (green); Poor Macroinvertebrate Health and Poor Habitat Quality (orange). The colors correspond to sites in each category on the map in Figure 2.

	Macroinvertebrate Health	
Habitat Quality	**GOOD**	**POOR**
GOOD	Good Macroinvertebrate Health and Good Habitat Quality	Poor Macroinvertebrate Health and Good Habitat Quality
POOR	*	Poor Macroinvertebrate Health and Poor Habitat Quality

* The combination of poor habitat and good invertebrate health is generally not encountered and therefore was not included in the Estuary Program's categorization scheme.

Table 4. Number and density of stream macroinvertebrate sample sites and total stream miles in the River Basins of the Narragansett Bay Watershed.

River Basins	Number of Sites (Percent of All Sites in the Watershed)	Total Stream [1] Miles in the River Basin	River Basin Area (square miles)	Density of Sites in Basin (sites per stream mile)
Blackstone River	30 (38%)	1,137.0	474.6	0.03
Coastal Narragansett Bay	15 (19%)	827.4	468.9	0.02
Pawtuxet River	21 (27%)	443.6	231.9	0.05
Taunton River	12 (15%)	1,170.2	529.5	0.01
Total Narragansett Bay Watershed	**78**	**3,578.3**	**1,704.9**	**0.02**

[1] National Hydrography Dataset

sampled five to ten years ago, the results from this analysis should be considered as a view of stream macroinvertebrate conditions over that timeframe.

The Estuary Program quantified the number of sites for each of the three stream condition levels (Table 3) and summarized them by River Basins and HUC10 watersheds, including the percentage of sites in each stream condition level. This information characterizes the general stream conditions only in areas of the Narragansett Bay Watershed where stream macroinvertebrates communities have been used as an indicator of water quality; the information about stream conditions and water quality does not apply to other areas of the Watershed.

To provide a measure of how thoroughly the data represent each River Basin, the density of monitoring sites was calculated. The total stream miles were calculated to determine the density of sites per stream mile within each River Basin, which ranged from 0.01 to 0.05 sites per stream mile (Table 4). The results indicated that fewer data were available for streams in the Taunton River Basin and for coastal streams draining directly into Narragansett Bay. Methods for assessing stream invertebrates in Rhode Island are not applied to many of the coastal freshwater streams, as they do not present the appropriate conditions for sampling in riffle habitats, where flowing water tumbles over rocks and cobblestones in the stream. Many coastal streams have sandy, mucky bottoms and are in regions with low elevations that do not have riffle habitat. It is important to acknowledge that site density results are based on all stream miles in the Watershed, and not only the stream segments that are suitable for monitoring stream macroinvertebrates. Future work is needed to quantify and evaluate the length of the stream and the drainage area of which each site is representative.

Status

As discussed above, the dataset developed for this report covered a period from 2002 to 2014. The results provide a general representation of stream conditions for this period that is presumed to be reasonably indicative of the status of those wadeable streams that were monitored. MassDEP and RIDEM had a total of 78 sites in the Narragansett Bay Watershed where they monitored stream macroinvertebrates. The Blackstone River Basin had the highest (38 percent) number of sites and the Taunton River Basin had the lowest (15 percent), as a percentage of all sites monitored in the Narragansett Bay Watershed (Table 3). The density of sites in each River Basin showed that while the Blackstone River and Taunton River Basins have comparable areas and total number of stream miles, the density of monitoring sites in the Blackstone River Basin was three times that of the Taunton River Basin (Table 4).

Of the 78 stream sites in the Narragansett Bay Watershed, 62 sites had Good Macroinvertebrate Health and Good Habitat Quality (79 percent of classified stream sites), and fifteen sites (nineteen percent) had Poor Macroinvertebrate Health and Good Habitat Quality (Table 5; Figure 2). Only one site in the Watershed, in the Taunton River Basin, had Poor Macroinvertebrate Health and Poor Habitat Quality (Figure 2, partially obscured near Brockton).

At the River Basin and HUC10 watershed scales, the Pawtuxet River Basin and the Blackstone River Basins had 90 and 80 percent of their sites, respectively, showing good stream condition levels based on both macroinvertebrate health and habitat quality.

Overall, the high number and percentage of sites that had healthy stream conditions were expected from these results. Riffles are good habitats for stream macroinvertebrates, and most wadeable stream sampling occurs in small headwater streams where water quality and surrounding habitat are in good condition, thus leading to good stream condition. However, sites with poor macroinvertebrate health but good habitat may have factors such as water pollution affecting stream macroinvertebrates, causing species tolerant of degraded water quality conditions to dwell at these sites at the time of monitoring.

To compare stream condition among River Basins based on stream macroinvertebrates and habitat, the results for each stream condition level were summarized by the number of sites in each River Basin (Table 6). Most of the 62 sites with Good Macroinvertebrate Health and Good Habitat Quality were located in the Blackstone River Basin (39 percent) and the Pawtuxet River Basin (31 percent). Of the fifteen sites in the Watershed that had Poor Macroinvertebrate Health and Good Habitat Quality, 40 percent were in the Blackstone River Basin, followed by 27 percent in the Coastal Narragansett Bay Basin (Table 6).

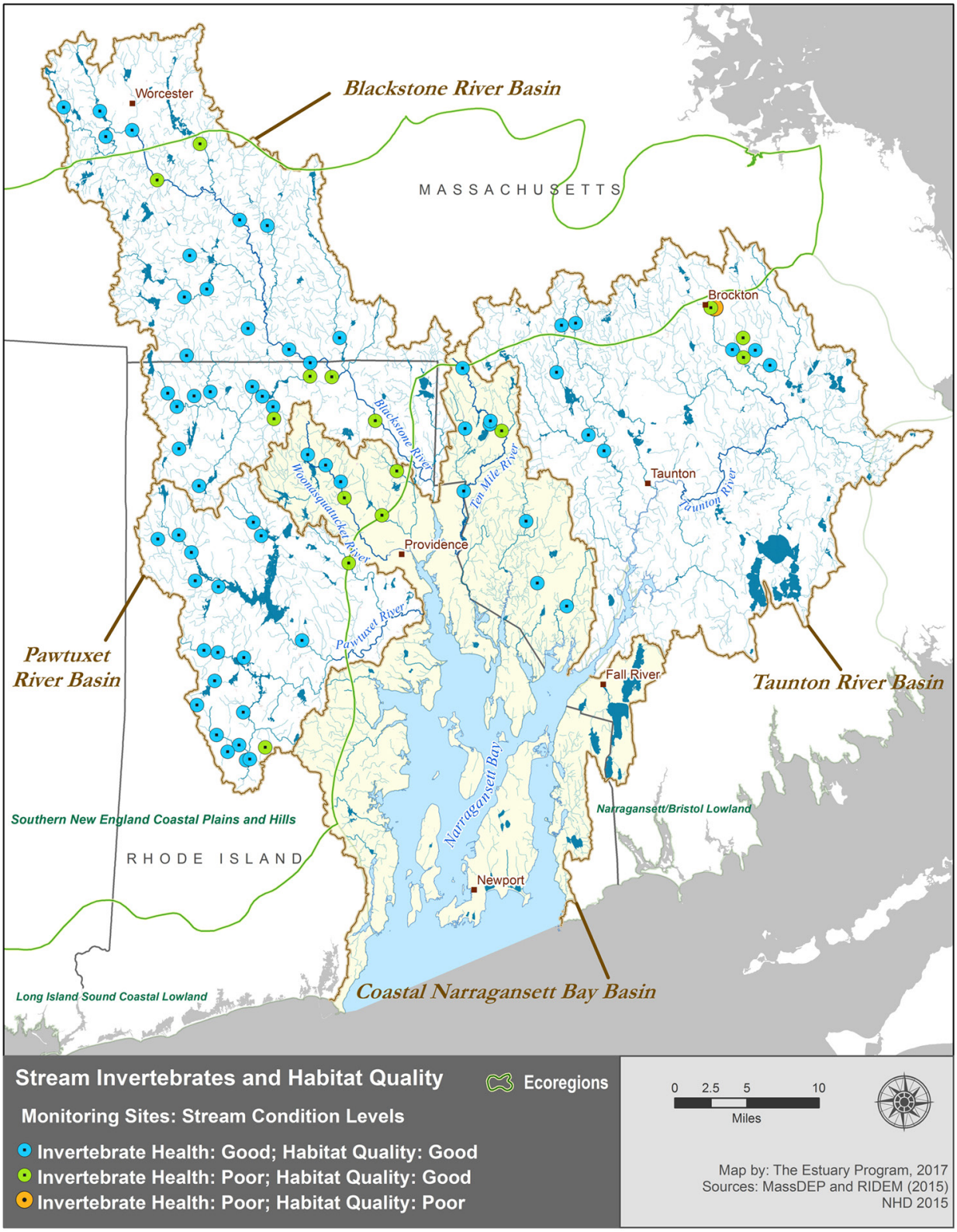


Figure 2. Stream condition levels in the Narragansett Bay Watershed based on stream macroinvertebrate monitoring.

Discussion

For the period from 2002 to 2014, available data on stream condition levels in the Narragansett Bay Watershed, including many headwater streams, reflected a positive finding in that a majority of the streams sampled indicated good ecological conditions for macroinvertebrates. However, the results should be interpreted with caution to avoid extrapolating this characterization of conditions to all streams across the Watershed. The findings of this indicator do not necessarily apply to all wadeable streams in the Narragansett Bay Watershed, but only those with appropriate riffle habitat. In addition, data were not available from many portions of coastal streams because the data collection methods are not suitable as these streams tend to have lower gradients, slow-moving water, silty bottoms, and deeper water depths. Because many of these lowland streams lack the appropriate riffle habitat required for the assessment, coastal streams are poorly represented in the assessment.

Therefore, the results are specifically reflective of the type of small, shallow, wadeable streams that were sampled in the Watershed by the state agencies, MassDEP and RIDEM. Given land use patterns in the Watershed, sampling locations in upstream parts of the Watershed would in general be expected to be located in less-developed areas and to support better habitat and water quality. Based on these criteria and factors that influence stream invertebrate sampling and also the results as presented, it is important to note that the Watershed characteristics vary among River Basins, topographically and hydrologically, not to mention the different characteristics driven by human uses in the landscape. The Blackstone River Basin is characterized by higher elevations and fragmented streams and rivers, primarily by dams, whereas the Taunton River Basin is generally characterized by low-lying lands, and where estuarine waters meet freshwaters in approximately 20 miles of an unobstructed Taunton River. Coastal areas tend to be more developed, and the Pawtuxet River Basin is characterized by moderate elevations with a forested landscape, mainly in the upper reaches of the watershed.

Despite the disparate and low number of sampling sites within and across River Basins (Table 5), the

Table 5. Percentage and number of sites in each stream condition level by River Basin (italics) and HUC10 watershed, and overall within Narragansett Bay Watershed (bold).

River Basins / **HUC10 Watershed Name**	**Stream Condition Level**			
	Good Macroinvertebrate Health and Good Habitat Quality	Poor Macroinvertebrate Health and Good Habitat Quality	Poor Macroinvertebrate Health and Poor Habitat Quality	Total Number of Sites
Blackstone River Basin	*80% (24 sites)*	*20% (6 sites)*		*30*
Upper Blackstone River	8	2		*10*
Lower Blackstone River	16	4		*20*
Coastal Narragansett Bay Basin	*73% (11 sites)*	*27% (4 sites)*		*15*
Ten Mile River	5	1		*6*
Palmer River	2			*2*
Narragansett Bay	1			*1*
Woonasquatucket-Moshassuck Rivers	3	3		*6*
Pawtuxet River Basin	*90% (19 sites)*	*10% (2 sites)*		*21*
Pawtuxet River	19	2		*21*
Taunton River Basin	*67% (8 sites)*	*25% (3 sites)*	*8% (1 site)*	*12*
Upper Taunton River	3	3	1	*7*
Lower Taunton & Frontal Mt. Hope Bay	1			*1*
Three Mile River	4			*4*
Total Narragansett Bay Watershed	**79% (62 sites)**	**19% (15 sites)**	**2% (1 site)**	**78**

findings by River Basin indicate, in terms of the geography and topography of the Watershed, that riffle habitat is primarily found in the upper reaches of the Watershed. These sites provide suitable conditions for healthy stream macroinvertebrates, whereas the lowlands, such as the coastal areas (Table 5 and Figure 2), are less likely to provide the geomorphological conditions for riffles or habitat conditions have been deteriorated by development pressures. In the Taunton River, it is assumed there are fewer monitoring sites because of its flattened watershed, thus providing fewer sites with suitable riffle habitat, or sampling effort is simply small compared with other River Basins.

Within the Narragansett Bay Watershed, the Pawtuxet River and the Blackstone River Basins have 65 percent of the sampling sites in the Watershed, reflecting that they contain most of the riffle habitat within the entire Watershed, and correspondingly they contain most of the sites with good macroinvertebrate health and good habitat quality (Tables 5 and 6). These sites and others throughout the Watershed with both good macroinvertebrate health and good habitat quality should continue to be routinely tracked and targeted for enhanced watershed protection if warranted. While the Blackstone River Basin is represented by good macroinvertebrate health and good habitat quality, it also contains 20 percent of the sampled sites classified as poor macroinvertebrate health and good habitat quality. As previously noted, this may indicate water quality problems such as nutrient enrichment or other pollution issues (Plafkin et al. 1989). Macroinvertebrate health may improve at these sites following implementation of appropriate best management practices aimed at water quality improvement.

MassDEP and RIDEM have found that water quality degradation is more likely to be occurring in the densely developed watersheds. Results from landscape stressor indicators developed by the Estuary Program show that watersheds and subwatersheds (HUC10 and HUC12 respectively; see the Appendix for definitions, lists, and maps) within the Coastal Narragansett Bay Basin are highly populated (see "Population" chapter) and intensively urbanized (see "Land Use" chapter) with the highest percentages of impervious cover (see "Impervious Cover" chapter). While all but one site had good habitat quality, and fifteen sites were found as having poor macroinvertebrate health, the effects of urbanization and other landscape, chemical, and climatic stressors may be contributing to the degradation of water quality.

Efforts by other partners are producing new information about stream invertebrates and other biological indicators in the Narragansett Bay Watershed. For example, EPA's Atlantic Ecology Division (AED) monitored macroinvertebrates in streams at 105 sites across the Watershed in 2012 and 2013. The sites encompassed an impervious cover gradient in which fifteen reference sites had less than one percent impervious cover and 90 sites had varying percentages of cover (Anne Kuhn, personal communication; a report is in preparation). Future collaborative partnerships should be sought to continue using stream macroinvertebrates as biological indicators to augment the understanding of and track changes in stream conditions in the Narragansett Bay Watershed.

Table 6. Percentage of all sites in the Narragansett Bay Watershed in each stream condition level that were located in each River Basin.

River Basins	Percent of Sites by Stream Condition Level		
	Good Macroinvertebrate Health	Poor Macroinvertebrate Health	Poor Macroinvertebrate Health
	Good Habitat Quality	Good Habitat Quality	Poor Habitat Quality
Blackstone River	39	40	
Coastal Narragansett Bay	18	27	
Pawtuxet River	31	13	
Taunton River	13	20	100(1)

(1) A monitoring site in the Taunton River Basin was the only one in the Narragansett Bay Watershed with both poor macroinvertebrate health and poor habitat quality.

Data Gaps and Research Needs

- Existing macroinvertebrate sampling protocols are not appropriate for all rivers and streams in the Watershed. To address coastal streams left unassessed, a multi-year effort of data collection and evaluation is needed and should be conducted at a regional scale to sample a sufficient number of locations in the lowland ecoregion streams. The data should be used to develop a robust biotic index for use in the lowland ecoregions for which the current rapid bioassessment protocol is not appropriate.
- Further analysis of existing data is needed to evaluate how well the existing monitoring strategies represent the conditions of the wadeable rivers and streams throughout the entire Watershed.
- Characterization of stream segments (by calculating stream miles) and drainage area (by defining the contributing catchment area to the site) is needed to study the influences of landscape stressors and other factors on stream conditions. The characterization should focus on sites where macroinvertebrate health was poor but habitat conditions were good. The findings could be used to help identify and amelioriate potential threats at sites with good macroinvertebrate health and good habitat quality that need protection.

Acknowledgments

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State of Narragansett Bay and Its Watershed
2017 Technical Report

Watershed Ecosystem Condition Indicators

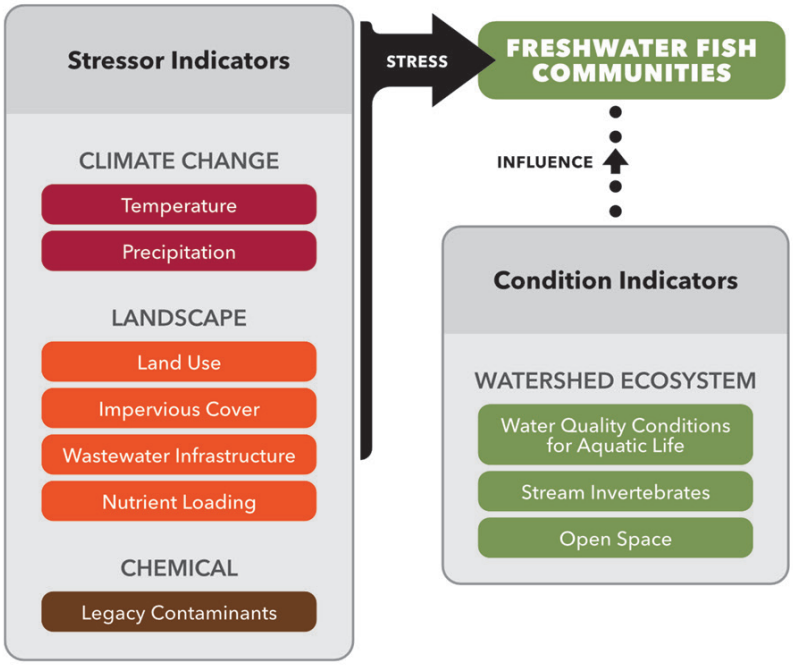
CHAPTER 20:
FRESHWATER FISH COMMUNITIES

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Photo: Brook trout, North Kingstown, RI (Ayla Fox)

NARRAGANSETT BAY ESTUARY PROGRAM

Overview



BACKGROUND

- Freshwater fish communities are indicators of habitat and water quality in streams and rivers, and brook trout, in particular, is an important indicator species. Climate change and land use, including impervious cover, are warming the water and altering flow patterns. These changes affect fish species that require cold or cool water, as well as fluvial species, which require free-flowing water. Excessive nutrient loadings, including those from wastewater infrastructure, and legacy contaminants can also stress freshwater fish communities.

KEY FINDINGS

- ***Status***

  Fish Communities: Based on data collected from 2002 to 2014 in the Narragansett Bay Watershed, the Upper Blackstone River had the highest relative abundance of cold- and coolwater fish species (67 percent) and fluvial fish (65 percent). The Palmer River had the lowest with eleven percent cold-coolwater species and nine percent fluvial species.

  Brook Trout Habitat: More than fourteen percent (640 square miles) of the Narragansett Bay Watershed's landscape was identified as brook trout habitat patches based on past observations of brook trout presence. The Pawtuxet River watershed had the greatest percentage (nearly 30 percent), and the Lower Blackstone River watershed had the greatest area (188 square miles). No brook trout habitat patches were identified in the Upper Taunton and Ten Mile River watersheds.

Introduction

FISH COMMUNITIES

Freshwater fish are ecologically important in stream ecosystems and provide significant value to humans. Ecosystem services supported by fish include the regulation of food web dynamics and recycling of nutrients, linkages within aquatic systems and to terrestrial ecosystems, food production, recreational activities, and information services, including indicators of ecosystem stress and resilience (Holmlund and Hammer 1999). As indicators of stress, fish assemblages may provide evidence of impairment that differs from the signals provided by algae or benthic invertebrates (Carlisle et al. 2008). Accordingly, the assessment of freshwater fish communities is important for resource managers.

Historically, the streams and rivers of southern New England supported moderately diverse and abundant assemblages of native fish communities. Enser and Lundgren (2005) noted that in Rhode Island the upstream reaches of streams and rivers are relatively steep and cold, with narrow, shallow streambeds characterized by well-defined riffles and pools, and mixed bedrock, boulder, and cobble substrate. Characteristic fish include brook trout (*Salvelinus fontinalis*), longnose dace (*Rhinichthys cataractae*), and blacknose dace (*Rhinichthys atratulus*). Downstream reaches of streams and rivers tend to be wider and sluggish, with sand and silt substrate, and native fish including pumpkinseed (*Lepomis gibbosus*), chain pickerel (*Esox niger*), and yellow perch (*Perca flavescens*).

Stream and river habitats are affected by stressors including climate change and land use, both of which can alter flow and increase water temperatures (Bain and Meixler 2000, Kashiwagi and Richards 2009, Kanno and Vokoun 2010, Bain 2011, Meixler 2011, Beauchene et al. 2014). Among the land use-associated stressors affecting streams and rivers are nutrient enrichment (see "Wastewater Infrastructure," "Nutrient Loadings," and "Water Quality for Aquatic Life Use" chapters), toxic chemicals (see "Legacy Contaminants" chapter), and road salts. Hydrologic alterations from dams, channel modifications, shoreline stabilization, and other in-stream activities can alter flow, causing shifts in fish species composition and abundance (Carlisle et al. 2010, Poff and Zimmerman 2010). Armstrong and colleagues (2011) analyzed relationships between fish communities and anthropogenic factors in catchment areas within Massachusetts, where 60 percent of the Narragansett Bay Watershed is located. They found that no more than four fluvial fish species were expected in streams where catchment areas had more than 50 percent impervious cover and flow alterations from groundwater withdrawal exceeded 50 percent of the August median flow. In addition, they report that a one-percent increase in impervious cover was associated with a 3.7-percent decrease in the relative abundance of fluvial fish, a 5.4-percent decrease in fluvial fish species richness, and an 8.7-percent decrease in brook trout relative abundance.

Freshwater fish assemblages are known to change in characteristic ways in response to stressors, and those changes can be used as indicators of stream and river health and integrity. Managers may use an index of biotic integrity (IBI) or a multi-metric index (MMI) to quantify aspects of the fish assemblage that are expected to respond to stressors (Whittier et al. 2007, Stoddard et al. 2008). In both approaches, metrics are calculated and then combined into a final score, which can then be related to a class of impairment. Rhode Island has not developed an index approach for fish communities but does have information on the sensitivity of fish communities to temperature and flow (RIDEM 2012, Rashleigh et al. 2013). Massachusetts and Connecticut have applied multi-metric indices to understand relationships between fish assemblage characteristics and anthropogenic factors, including temperature, flow, and other environmental, land use, and physical watershed characteristics (Kanno et al. 2010, Kanno and Vokoun 2010, Armstrong et al. 2011, Bassar et al. 2016). The approaches taken by these states differ quite significantly. To provide consistency for the entire bi-state Narragansett Bay Watershed, the Narragansett Bay Estuary Program and partners evaluated two metrics for freshwater stream and river fish communities: percent relative abundance of cold-coolwater fish and percent relative abundance of fluvial fish. Fluvial fish require flowing water for some or all of their life cycle. Bi-state coordination could result in the development of a multi-metric index for freshwater fish in the Narragansett Bay Watershed.

BROOK TROUT HABITAT

The eastern brook trout, in the salmon family, has received special attention in the region due to its economic, recreational, ecological, and cultural value. Brook trout primarily eat insects, and they spawn in coldwater streams in the fall. Adults then migrate to deeper waters to overwinter, and eggs hatch in the spring. Sea-run brook trout have been documented in New England, but it is unclear whether any remain in the Narragansett Bay Watershed. In pre-colonial

times, brook trout were present in nearly every coldwater stream and river in the eastern United States (Eastern Brook Trout Joint Venture, undated). As land clearing took place for agriculture and timber harvesting, the amount of sediment entering streams increased, and wild brook trout began to disappear (Eastern Brook Trout Joint Venture, undated). The Narragansett Bay Watershed's main streams and rivers, such as the Blackstone River, were channelized and dammed, reducing brook trout habitat through altered hydrology and pollution from industrial activities. Many of these threats to brook trout persist today.

Brook trout are known to be highly sensitive to multiple stressors, including increased temperatures (see "Temperature" chapter), land use (see "Land Use" chapter), and flow alteration. DeWeber and Wagner (2015) found a negative response of eastern brook trout to increasing temperature and agricultural land cover. Brook trout are found in perennial streams with temperatures between 34 and 72°F (1 and 22°C) (Xu et al. 2010). Bassar and colleagues (2016) demonstrated that declines in brook trout in western Massachusetts were related to changes in temperature and stream flow.

Brook trout are particularly important ecological indicators for the region, as the Narragansett Bay Watershed is near the southern limit of their thermal tolerance, and climate change may have a significant effect on their distribution and abundance. In addition to brook trout, other fish species common in the Watershed rely on coldwater streams and riparian habitats to meet one or more of their life history requirements (Massachusetts Division of Fisheries and Wildlife 2017). Thus, protecting these habitats from current and future anthropogenic and climate change stressors is critical. A decline in the brook trout population indicates negative changes in the habitat and overall ecosystem integrity.

In Massachusetts, the Division of Fisheries and Wildlife maintains a list of waters that are identified as Coldwater Fishery Resources. In addition, the state environmental agencies, Massachusetts Department of Environmental Protection (MassDEP) and Rhode Island Department of Environmental Management (RIDEM), have included the assessment of coldwater streams within their Integrated Reports (MassDEP 2016, RIDEM 2014).

At the regional level, brook trout status has been assessed across its range from Maine to Georgia by the Eastern Brook Trout Joint Venture (EBTJV), a partnership among state and federal agencies, regional and local governments, businesses, conservation organizations, academia, scientific societies, and private citizens working toward protecting, restoring, and enhancing brook trout populations and their habitats across their native range. Hudy and colleagues (2008) and the EBTJV (undated) described an array of analyses comparing current distributions to the native range, as defined by MacCrimmon and Campbell (1969). In Connecticut and Rhode Island, researchers concluded that remaining populations are small and fragmented (EBTJV undated). The Estuary Program used data from the EBTJV as a proxy for habitat conditions for coldwater fish communities, calculating the extent of EBTJV-defined catchment areas, or those areas draining to streams and rivers where brook trout were observed (EBTJV undated).

For this chapter, the Estuary Program analyzed 1) the relative abundance of fluvial fish and cold-coolwater fish, and 2) the extent of brook trout patch areas as a proxy for habitat. These indicators of freshwater fish communities are considered under development, and the results discussed in this chapter are exploratory, laying the groundwork for future investigation and analysis. The information presented in this chapter represents a coordinated effort between the Estuary Program, EPA, and the states of Rhode Island and Massachusetts.

Methods

FISH COMMUNITIES

The Narragansett Bay Estuary Program and the US Environmental Protection Agency (Atlantic Ecology Division) gathered, diagnosed, and reconciled fish sampling data from Massachusetts and Rhode Island for freshwater streams and rivers in the Narragansett Bay Watershed. Other data sources and local studies were considered, but issues related to inconsistencies, geographical coverage, or availability made them unsuitable for this analysis. National datasets such as the National Aquatic Resources Survey were also available, but only a limited number of sampling sites were located in the freshwater streams in the Watershed.

The fish data from Massachusetts and Rhode Island were assumed to be comparable in terms of methods, as both states used single-pass backpack shocking in the upstream direction, sampled all habitats, covered similar lengths of stream (at least 100 meters), and had similar sampling schedules (typically late spring to fall).

Data were obtained from the RIDEM Division of Fish and Wildlife (DFW) (Libby 2004, Libby 2013) and the Massachusetts Division of Fish and Wildlife (MDFW

2014). For Rhode Island, data were collected from April 29, 1992, to September 30, 2009, as part of a multi-year statewide survey of fish communities in rivers and streams; survey work continued on a less frequent basis after 2002. For Massachusetts, sample dates ranged from September 21, 1994, to October 14, 2014. Field data collected by both states included species identifications and abundance of each species.

The Estuary Program and partners calculated two metrics that are used by researchers and resource managers in Massachusetts and Connecticut: (a) the percent relative abundance of cold-coolwater fish and (b) the percent relative abundance of fluvial fish (Table 1). This analytical approach was also supported by RIDEM. Higher values of the metrics indicate a fish population consistent with cold-coolwater and/or fluvial fish dominance. Even though the "cool" water fish designation is not a categorization that RIDEM has used, the Estuary Program included it because measures of temperature tolerance are used in other New England states (e.g., Connecticut: Kanno et al. 2010; Maine: Yoder et al. 2015), and temperature tolerance can be a useful indicator of effects of climate change (Beauchene et al. 2014).

In this analysis, coldwater and coolwater fish were combined into a single group termed cold-coolwater fish, and fluvial dependents and specialists were combined into a single group termed fluvial fish (Table 1). Cold-coolwater designations for species were obtained from Kanno and colleagues (2010, Appendix 1). Fluvial designations for species were derived from Armstrong and colleagues (2011), where fluvial dependents and specialists are fish species that require flowing water for some or all, respectively, of their life cycle. In some cases, Kanno and colleagues (2010) presented different information regarding fluvial designations of fish species. However, fluvial fish designations used by Armstrong and colleagues (2011) for the fish assemblages study in Massachusetts were followed since 60 percent of the Watershed is in Massachusetts.

Table 1. Freshwater fish species of the Narragansett Bay Watershed and their requirements for cold/coolwater temperatures, flowing water (fluvial), or both.

Common Name	Scientific Name	Cold-coolwater Fish	Fluvial Fish[5]
American Brook Lamprey[1]	*Lampetra appendix*	Cold	Specialist
Blacknose Dace[2]	*Rhinichthys atratulus*	Cool	Specialist
Brook Trout	*Salvelinus fontinalis*	Cold	Specialist
Common Shiner	*Luxilus cornutus*	Cool	Dependent
Creek Chub[2]	*Semotilus atromaculatus*	Cool	Specialist
Creek Chubsucker	*Erimyzon oblongus*		Specialist
Fallfish	*Semotilus corporalis*	Cool	Specialist
Fourspine Stickleback[3]	*Apeltes quadracus*	Cool	Dependent
Longnose Dace	*Rhinichthys cataractae*	Cool	Specialist
Rock Bass[4]	*Ambloplites rupestris*	Cool	
Smallmouth Bass[4]	*Micropterus dolomieu*	Cool	
Tessellated Darter	*Etheostoma olmstedi*	Cool	Specialist
White Perch	*Morone americana*	Cool	
White Sucker[2]	*Catostomus commersoni*	Cool	Dependent
Yellow Perch	*Perca flavescens*	Cool	

[1] Brook lamprey was not categorized by Armstrong et al. (2011) but was considered a fluvial specialist by Kanno (2010). However, RIDEM's DFW has found this species in the warm waters of the Blackstone River.
[2] These species are considered tolerant to pollution (Halliwell et al. 1999), so they may be present in areas of low quality despite their specialized habitat requirements. Kanno et al. (2010) developed a metric that omitted blacknose dace. However, Armstrong et al. (2011) demonstrated strong relationships between flow and dace, so the Estuary Program included this species. Also, Kanno et al. (2010) included a metric for white sucker that is expected to be higher in more degraded areas.
[3] The dataset included fourspine stickleback, which is primarily a saltwater or brackish-water fish but is often found in freshwater. The species is very rare in the Narragansett Bay Watershed and did not strongly affect results.
[4] These species are considered nonnative. Some assessment approaches exclude nonnative species, as their presence may not be desirable. The abundance of these species was extremely low and did not strongly affect results.
[5] Fluvial dependents and specialists are fish species that require flowing water for some or all, respectively, of their life cycle. In analyses, the two categories are combined into a single group termed fluvial fish.

The metrics were calculated as follows:

$$\textit{Percent Relative Abundance: } \frac{\textit{Abundance of Cold-coolwater Fish}}{\textit{Total Fish Abundance}} \times 100$$

$$\text{Percent Relative Abundance: } \frac{\textit{Abundance of Fluvial Fish}}{\textit{Total Fish Abundance}} \times 100$$

Here, abundance was defined as the number of individuals, and calculated as ratios.

Geospatial analysis tools (Esri 2016) were used to associate the latitude and longitude reported by the states for each sample site. Within each HUC10 watershed, the average percent relative abundance for each group–cold-coolwater fish and fluvial fish–was calculated.

Regarding the fish data and the metrics utilized to develop this indicator, it is important to acknowledge the following assumptions and caveats, which were considered and addressed for proper data analysis:

- In accordance with standard practice, the Estuary Program and partners adjusted the compiled dataset by removing stocked salmonids–rainbow trout (*Oncorhynchus mykiss*), Atlantic salmon (*Salmo salar*), and brown trout (*Salmo trutta*)–because their presence is due exclusively to stocking (Kanno and Volkun 2010, Bellucci et al. 2011, Libby 2013). Species predominantly associated with marine habitats such as mummichog (*Fundulus heteroclitus*) and anadromous fishes such as alewife (*Alosa pseudoharengus*), blueback herring (*Alosa aestivalis*), shad (*Alosa sapidissima, Alosa mediocris*), striped bass (*Morone saxatilis*), and sea lamprey (*Petromyzon marinus*) were also removed (Meixler 2011, Armstrong et al. 2011). The abundances of these species were very low in the samples, so their inclusion would not likely have changed the results. Any hybrids or specimens not identified to species were also removed from the analysis.
- The two groups of freshwater communities as defined and measured for this report may not be exhaustive of all fish species that have been observed in rivers of the Watershed.
- Because Armstrong and colleagues (2011) reported relative abundance as counts per hour, these methods are not directly comparable.
- Metrics are calculated as ratios (see formulas above), so adjusting abundance to counts per hour, rather than number of individuals, would not change the results.
- The range of years used for this analysis was limited to 2002 through 2014 to provide consistency with other reports and to enhance comparability. Due to limited repeat sampling in the dataset across the Watershed, it was not possible to conduct a trend analysis to identify changes in the fish communities over time. The distribution of sampling sites across years and throughout the Watershed and HUC10 watersheds was sparse (Table 2).
- A preliminary analysis using the full dataset versus this shorter period identified that there were no substantial differences in the results. However, fish data collection for the Woonasquatucket-Mossashuck Rivers (HUC10) watershed took place between 1992 and 2001, and thus outside the period of data analysis; the results of this timeframe prior 2002 are presented to complement the findings (Table 2).

Table 2. Number of sites for fish data in HUC10 watersheds in Narragansett Bay Watershed, from 1992 to 2015.

HUC10 Watersheds	Number of Fish Sampling Sites per Year By HUC10 Watersheds																								
Name	1992	1993	1994	1995	1996	1997	1998	1999	2000	2001	1992-2001	2002	2003	2004	2005	2006	2007	2008	2009	2010	2011	2012	2013	2014	2002-2014	Total 1992-2014
Lower Blackstone River	5	5	47	6		2	5	2	4	32	108	10	3	6	3	10	8	11	6	3	7		3		70	178
Narragansett Bay	1		1			12			20	24	58	17	2	9	1	3	2	5	12	2			1		54	112
Upper Blackstone River							4		1	32	37				2	7	13	13	10	2	6		9	7	69	106
Pawtuxet River			1		37	28		1		1	68	1	6	1		2		2							12	80
Middle Taunton River											0	9	3		12	4	1	3	6		4	1	6	6	55	55
Lower Taunton River-Frontal Mount Hope Bay			2							1	3	8			6	3	4	5	4	2	2		1	2	37	40
Palmer River										2	2	2	2		1	5	2		5	4	1		8	2	32	34
Ten Mile River			1	1					3	11	16	8			2	1	1	3			3				18	34
Upper Taunton River											0	4				4	2					1	4	4	19	19
Woonasquatucket River-Moshassuck River				17							17														0	17
Threemile River											0				6	5	1							4	16	16
Total Narragansett Bay Watershed	6	5	52	24	37	42	9	3	28	103	309	59	16	16	33	44	34	42	43	13	23	2	32	25	382	691

BROOK TROUT HABITAT

The Estuary Program measured the extent of catchment areas draining to streams and rivers of the Narragansett Bay Watershed, where brook trout was observed, based on a salmonid catchment assessment by the Eastern Brook Trout Joint Venture (EBTJV). In 2015, the EBTJV performed a geographic habitat patch analysis in coordination with states and other partners. The EBTJV used brook trout observations, at sampling sites, to define the catchment areas for each site. The habitat patches were used for this analysis as a landscape feature, which can be interpreted as a proxy for coldwater species habitat. The streams within these habitat patches either once supported or are still supporting brook trout populations. EBTJV data were used for two primary reasons: 1) the EBTJV's salmonid assessment of catchment areas was developed at a regional scale, and thus the data representing habitat patch layers covered the entire Narragansett Bay Watershed seamlessly, and 2) the fish data used by the EBTJV as inputs for the model were from monitoring efforts conducted by the state agencies and other partners in streams within the Watershed.

For this indicator, the total extent of brook trout habitat patches was calculated for the Narragansett Bay Watershed and for HUC10 watersheds. It was calculated as the percent of patch or catchment areas in which brook trout had been observed at least once.

Because it used the derived data from the EBTJV's salmonid assessment, the Estuary Program's analysis includes the following limitations and assumptions:

- **Specific information about the data, such as data sources and data collection dates, was not readily available:** The EBTJV methodology for the salmonid catchment assessment and habitat patch layers was described in Coombs and Nislow (2015). In short, the salmonid catchment assessment and habitat patch layers were created using algorithms in the ArcGIS platform. The geospatial analysis used fish data from different Joint Venture partners from each state to define areas where brook trout was observed at least once over the period of two decades, depending on the monitoring efforts by each state. The specific collection dates for the data used in the analysis were not available.
- **Brook trout habitat patches might not represent current conditions:** While it is expected that sampling of brook trout from decades ago might not represent today's conditions, the preliminary results from this analysis can be interpreted as a retrospective view of conditions of brook trout habitat. Consequently, the habitat patch areas do not represent an indication of good or poor habitat, simply a measurement of areas in the Watershed that could have provided, at a given time, healthy aquatic habitat conditions for brook trout.
- **Catchment areas as defined by the EBTJV include all land uses within them:** Because habitat patch areas encompassed the entire extent of a catchment area of sampling sites in streams and rivers where brook trout were observed, the catchment areas include natural buffers, as well as other land use types within them. It is assumed that current floodplains, freshwater wetlands, and overall riparian buffers along the streams within the brook trout habitat patches are ecologically significant for protecting coldwater habitat. For future work, these catchment areas can be useful to further characterize land uses that might be affecting, positively or negatively, the receiving streams and rivers within these areas. Results from characterization could be used to target management actions and research for restoration and conservation of brook trout habitat.
- **Catchment areas representing stocked brook trout were removed:** The EBTJV analysis distinguished catchment areas by brook trout species, including whether they were stocked or not. Because this information was available and the Estuary Program's analysis focused on species that occur naturally in the ecosystem, catchment areas for stocked brook trout were eliminated from the analysis.
- **The total acreage of brook trout habitat patches was calculated, as opposed to normalizing for stream length:** To calculate extent of brook trout habitat patches within the Narragansett Bay Watershed, the Estuary Program performed geospatial analyses to determine total acreage of habitat patches in HUC10 watersheds and HUC12 subwatersheds. The total acreage is also reported as percent of each watershed or subwatershed that contains EBTJV's brook trout habitat patches. While the EBTJV defined the habitat patches by the contributing area to the sampling site of streams and rivers where brook trout was observed, other streams and rivers that are also part of the hydrological network might not have supported brook trout habitat. Therefore, without these data, the Estuary Program did not attempt to normalize results by stream length in the catchment area.

For more information related to the algorithm developed by the EBJTV and visualization of the brook trout catchment and habitat patch layers, see the EBJTV report and web application.

Status

FISH COMMUNITIES

In the Narragansett Bay Watershed, the Blackstone River HUC10 watershed had the highest percent relative abundance of both cold-coolwater and fluvial fish, whereas the Palmer River had the lowest metric values. The percent relative abundance of cold-coolwater species ranged from eleven percent in the Palmer River to 67 percent in the Upper Blackstone River (Table 3). Similarly, relative abundance of fluvial species ranged from nine percent in the Palmer River to 65 percent in the Upper Blackstone River (Table 3). The two metrics were highly correlated (r=0.91), although the data were quite variable.

For most of the HUC10 watersheds, the average percent of cold-coolwater fish and the average percent of fluvial fish were similar, as was the variability, perhaps suggesting that these two metrics can be used interchangeably. The majority of HUC10 watersheds had mean and median values of less than 50 percent for both metrics. In some HUC10 watersheds, the medians were quite different from the means (diamonds), since these watersheds contained a mix of brook trout sites and non-brook trout assemblages. Ranges for both metrics was 100 percent for some watersheds.

BROOK TROUT HABITAT

Overall, catchment areas with brook trout habitat encompassed 14.5 percent of the Narragansett Bay Watershed (Table 4). This percentage represents the total extent of drainage areas to streams and rivers where brook trout was observed at least once at sampling sites, and not only the extent of natural habitat such as riparian zones or other natural buffers.

At the HUC10 watershed scale, the Pawtuxet River watershed had the highest proportion of brook trout habitat patches at nearly 30 percent of the watershed's area. The Palmer River had the lowest, with 1.4 percent of its area having supported brook trout habitat at a given time. There were no brook trout habitat patch areas identified in the Upper Taunton River or the Ten Mile River watersheds (Table 4; Figure 2).

At the HUC12 subwatershed scale, 60 percent of the Hunt River subwatershed, which is nested within the Narragansett Bay HUC10 watershed, was identified as brook trout habitat patches. Table 5 shows other subwatersheds with more than 30 percent of their area identified as brook trout habitat patch areas.

Table 3. Average percent relative abundance for cold-coolwater fish and fluvial fish in the Narragansett Bay HUC10 Watersheds based on data collected between 2002 and 2014. SD = standard deviation.

HUC10 Watershed Name	No. Samples[1]	Average Cold-coolwater Fish % (SD)	Average Fluvial Fish % (SD)
Upper Blackstone River	69	67 (36)	65 (38)
Lower Blackstone River	70	50 (38)	46 (41)
Upper Taunton River	19	22 (28)	22 (28)
Middle Taunton River	55	32 (32)	30 (32)
Threemile River	16	28 (34)	29 (34)
Ten Mile River	18	20 (30)	18 (30)
Woonasquatucket River-Moshassuck River[2]	0	-	-
Pawtuxet River[3]	12	25 (17)	18 (33)
Palmer River	32	11 (22)	9 (17)
Lower Taunton River-Frontal Mount Hope Bay	37	15 (37)	16 (23)
Narragansett Bay	54	31 (37)	24 (36)

[1] Fish samples that were collected between 2002 and 2014.
[2] Fish sampling in the Woonasquatucket River-Mossashuck River watershed took place in 1995 only; a total of 17 fish samples were collected in that year (Table 2). Those data were not included in this analysis, which covered 2002 to 2014.
[3] Most of the fish sampling in the Pawtuxet River watershed took place in 1996 and 1997; a total of 37 and 28 samples of fish data were collected in those years, respectively (Table 2). Those data were not included in this analysis, which covered 2002 to 2014.

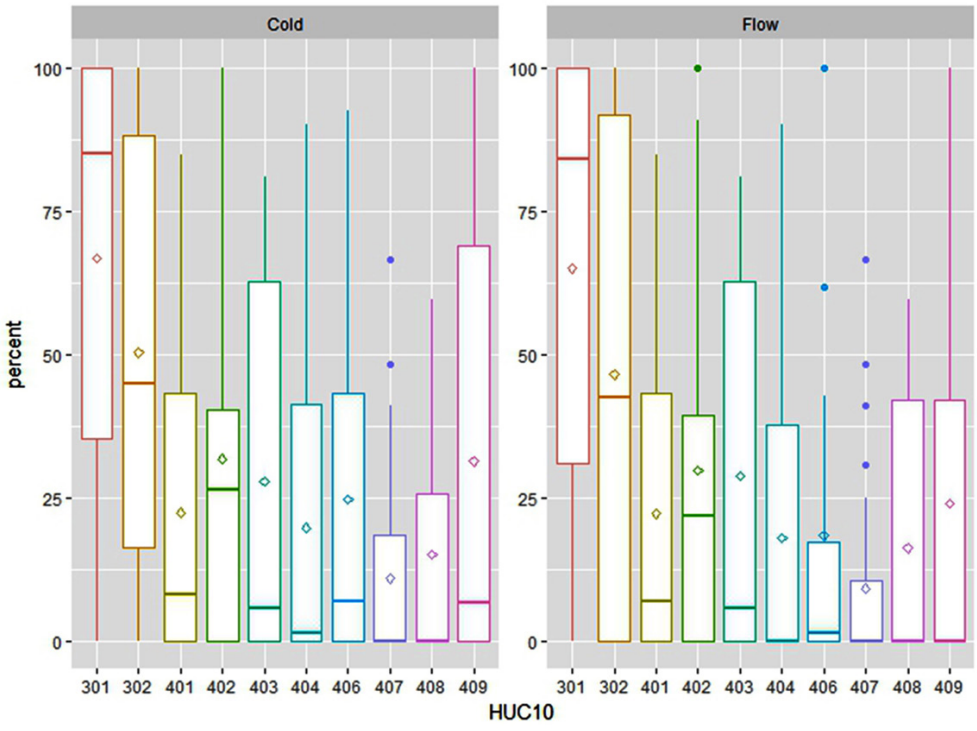


Figure 1. Percent relative abundance of cold-coolwater fish and fluvial fish within HUC10 watersheds in the Narragansett Bay Watershed. HUCs are represented by the last 3 digits of their USGS-assigned code number; see Appendix for HUC numbers and names. Horizontal lines indicate medians. Diamonds indicate means. Boxes represent the 25th percentile to 75th percentile. Whiskers indicate the range, and solid points are outliers.

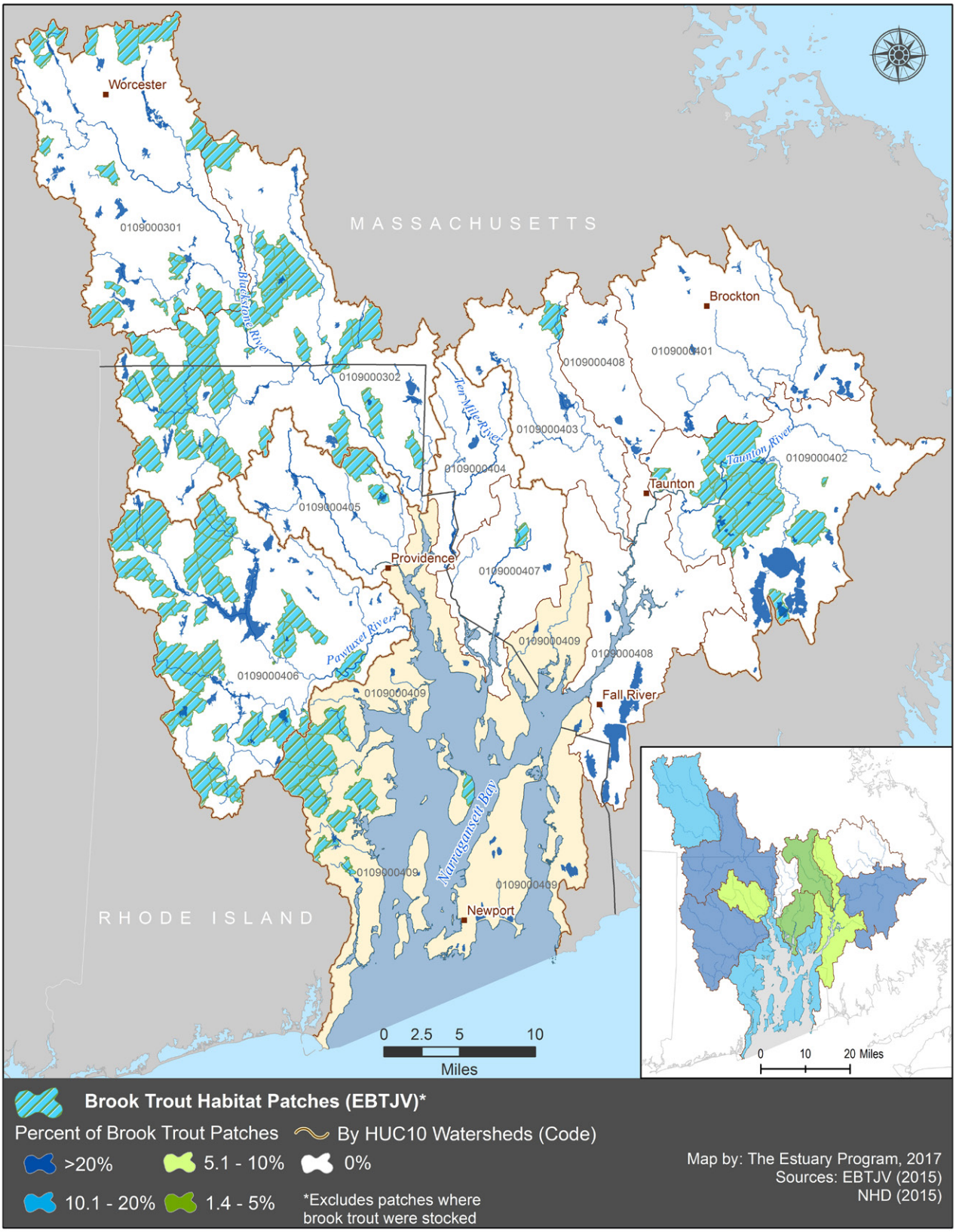


Figure 2. Percentage of brook trout habitat patch area relative to total HUC10 watershed area (blue to green on the locus map) and locations of brook trout habitat patches (blue-and-green striped areas) in HUC10 watersheds in the Narragansett Bay Watershed.

Table 4. Extent of brook trout habitat patch areas within HUC10 watersheds in the Narragansett Bay Watershed. Sorted by highest to lowest percentage of brook trout habitat.

HUC10 Watersheds Name	Brook Trout Habitat Patches within Watershed	
	Percent	Square miles
Pawtuxet River	29.3	175.7
Lower Blackstone River	27.0	188.0
Middle Taunton River	21.3	99.2
Upper Blackstone River	12.4	65.8
Narragansett Bay	10.6	66.4
Woonasquatucket River-Moshassuck River	6.2	11.9
Lower Taunton River-Frontal Mount Hope Bay	5.9	24.1
Threemile River	3.2	7.0
Palmer River	1.4	2.5
Upper Taunton River	0.0	0.0
Ten Mile River	0.0	0.0
Total Narragansett Bay Watershed	14.5	640.6

Table 5. HUC12 subwatersheds with brook trout habitat patch areas covering at least 30 percent of their area. Sorted by highest to lowest percentage of brook trout habitat.

HUC12 Subwatersheds Name	Part of HUC10 Watershed Name	Brook Trout Habitat Patches within HUC12 Subwatersheds	
		Percent	Square miles
Hunt River	*Narragansett Bay*	60	14.5
Cotley River-Taunton River	*Middle Taunton River*	53	32.0
Headwaters South Branch Pawtuxet River	*Pawtuxet River*	52	14.4
Clear River	*Lower Blackstone River*	47	21.2
Barden Reservoir-Ponaganset River	*Pawtuxet River*	44	14.4
Moswansicut Pond-Huntinghouse Brook	*Upper Blackstone River*	41	9.1
South Branch Pawtuxet River	*Lower Taunton River-Frontal Mount Hope Bay*	40	6.6
West River	*Woonasquatucket River-Moshassuck River*	34	12.5
Branch River	*Lower Blackstone River*	31	8.1

Complete results for all HUC12 watersheds are available upon request.

Discussion

FISH COMMUNITIES

Across the HUC10 watersheds, the spatial patterns of the results for abundance of cold-coolwater fish and fluvial fish were not surprising. Both measures were higher in upper reaches of the Narragansett Bay Watershed, likely due to both elevation and development patterns, primarily in the Upper and Lower Blackstone River watersheds (Table 3). Although the results presented here suggest that the streams of the Pawtuxet River watershed did not have a high average of percent relative abundance for cold-coolwater fish and fluvial fish based on data collected between 2002 and 2014, they had higher percentages if fish sampling data from all years were considered, since most of the sampling in that watershed took place outside the period used for this analysis (Table 6): 68 sites that recorded fish species in Table 1 were sampled between 1992 and 2002, whereas twelve sites had samples from 2002 to 2014 (Table 2). Enser and Lundgren (2005) noted that in Rhode Island the upper reaches of streams and rivers are fairly steep and cold, whereas lower stream and river reaches may be warmer and slower due to reduced elevation gradients. Although the absence of cold-coolwater and fluvial species may be due to stressors (e.g., urban development) or environmental setting (e.g., coastal streams), or a combination of the two, the presence of these species is indicative of good environmental health, and their distribution within the study area provides a benchmark for future assessment.

The large range of values (0 to 100 percent) observed for both the cold-coolwater fish and fluvial fish metrics indicates that they are useful and sensitive for the study area (Whittier et al. 2007) for objectives related to freshwater fish management, including brook trout. At one extreme, sites that support only brook trout would show 100 percent for both metrics. A threshold may be selected to separate out predominantly brook trout sites (e.g., >90 percent fluvial fish), which would be similar to other states (e.g., NHDES 2007, Kanno et al. 2010). Cold and coolwater streams are recognized by Rhode Island and Massachusetts as valuable habitats and managed accordingly. At the other extreme, some sites showed zero percent of both cold-coolwater fish and fluvial fish. A threshold could also be set to separate sites with few or no fluvial fish (e.g., <10 percent fluvial fish), which may be located near the coast.

The high correlation (0.91) of the cold-coolwater and fluvial metrics of relative abundance was expected because in many cases the species that make up these metrics are both coolwater and fluvial (Table 1).

Table 6. Average of percent relative abundance for cold-coolwater fish and fluvial fish by HUC10 in the Narragansett Bay Watershed, accounting for all years of data (1992–2014).

HUC-10 watersheds Name	Total Number of Samples from 1992–2014	Total Average of Cold-Cool%	Total Average of Fluvial Fish%
Lower Blackstone River	178	56	52
Lower Taunton River-Frontal Mount Hope Bay	40	17	16
Middle Taunton River	55	32	30
Narragansett Bay	112	31	23
Palmer River	34	11	9
Pawtuxet River	80	46	37
Ten Mile River	34	24	11
Threemile River	16	28	29
Upper Blackstone River	106	69	67
Upper Taunton River	19	22	22
Woonasquatucket River-Moshassuck River	17	44	43
Total Narragansett Bay Watershed	*691*	*43*	*38*

The state of New Hampshire in its IBI development used a cutoff point of 0.8 Spearman correlation coefficient as a threshold for strong correlation (NHDES 2007), so both metrics would likely not be used together in an index. Similarly, the percent of warm-water species would be the complement of the cold-coolwater species. While the streams in the Narragansett Bay Watershed can be expected to support both cold-coolwater and warm-water fisheries, impairment due to stressors can shift the composition toward warm-water fisheries (e.g., Flanagan et al. 1999).

Some additional metrics could be considered for analysis and included as part of an IBI that should be developed to support assessment of fish communities across the Watershed. Trophic and tolerance metrics are often used in biological assessments, although Bain (2010) found that aspects of fish species tolerance were less responsive than habitat in the Quinebaug River in Massachusetts. Coles and colleagues (1995) found that cyprinid species were sensitive to urbanization in New England coastal streams, and Kanno and colleagues (2010) also included cyprinid species in their MMI. Introduced species may also be indicators for impaired streams, particularly those species in the family Centrarchidae (e.g., Bain and Meixler 2000, Bain 2010). These metrics can be related to the Biological Condition Gradient framework, and it may be useful to relate the fluvial fish metric to the land use stressor (i.e., percent of urban lands, impervious cover, forested lands).

BROOK TROUT HABITAT

The results are retrospective in nature, given the time period when data were collected. Conditions of the contributing areas could have changed over the last ten or more years–due to land use changes, habitat fragmentation, climate change, and degraded water quality–which constrains the results to being only an estimation of brook trout habitat. While these results offer only a preliminary view of catchment areas in which streams supported brook trout habitat at the time when data were collected and analyzed, the information could be used to reassess catchment areas in the future and define new or existing areas where conditions are suitable for coldwater species.

EBTJV habitat patches across the watersheds indicated remaining areas that should be considered for protection of coldwater streams that can support brook trout and other cold-coolwater species. The Pawtuxet River and Lower Blackstone River watersheds, and the Hunt River subwatershed, with the greatest percent of brook trout habitat patches, can be targeted to characterize catchment areas and investigate potential conservation and restoration practices for brook trout and other cold-coolwater species. One of the potential applications of comparing coldwater streams as identified by the states with these habitat patches and fish data from monitoring is to assist managers and partners in locating specific areas to evaluate water quality and habitat conditions and to prioritize research and management efforts. A preliminary analysis by the Estuary Program determined that 230 miles of coldwater streams are within the brook trout habitat patches within the Watershed, using the Massachusetts Division of Fisheries and Wildlife's Coldwater Fishery Resources and RIDEM's coldwater fishery streams (RIDEM 2010, RhodeMap 2013).

Although brook trout have been studied within the Wood-Pawcatuck watershed (Tefft 2013), the Estuary Program is not aware of brook trout distributions previously being considered at the scale of the Narragansett Bay Watershed. And, while the Narragansett Bay Watershed is not the primary core habitat area for brook trout in the New England region, this analysis showed that some subwatersheds in the Pawtuxet River and Lower Blackstone River watersheds should be further examined for protection.

Climate change is one of the most critical factors that threatens brook trout and other cold-coolwater species as water temperatures continue warming (see "Temperature" chapter), exacerbated by population growth driving more development across the Watershed (see "Population" and "Impervious Cover" chapters), where runoff carries heated stormwater from pavement and other surfaces to the receiving waters. In addition, many streams no longer have natural vegetated buffers (see "Land Use" chapter), eliminating canopy cover that keeps waters cool and reducing the capacity of groundwater replenishment, which provides baseflow when stream flows are at their lowest, impacting fluvial fish and cold-coolwater fish.

As freshwater fish dwell in these streams and rivers, move up and downstream, and rely on other aquatic organisms for their survival, information about changes in fish assemblages, abundance, and diversity sheds light on conditions and stressors on stream and river habitats. While there is currently limited information on freshwater fish as indicators of watershed ecosystems in the Narragansett Bay Watershed, further development of this indicator could reveal stressors impacting other resources in the Watershed and ultimately the Bay, which is imperative as these stressors are in continual change.

Data Gaps and Research Needs

- Further development of freshwater fish communities as an indicator for status and trends reporting will require an expanded effort to collect fish community data. Evaluation of the resources to support the desired level of fish data collection across the Watershed is an appropriate next step.
- Targeted collection of data on brook trout is needed to better refine brook trout habitat and clarify coldwater stream designations and support the integration and update of the Eastern Brook Trout Joint Venture Salmonid Catchment Assessment and Habitat Patch Layers model.
- Additional data for freshwater habitats that were not considered here, but may have ecological significance to maintain healthy habitat for fish, should be gathered, created, defined, and analyzed, including intermittent streams, freshwater reaches of tidal rivers, wetlands, and riparian areas. Specialized methods for collection of fishes in these habitats may need to be identified or developed.
- Development of an indicator related to stream connectivity should be explored. It could reflect stream continuity in miles open, partially open, and obstructed for freshwater fish and other aquatic life communities, following other efforts already started in the Watershed, such as those led by the US Army Engineer Research and Development Center (Foran et al. under review). See also the stream continuity asssessment in the Tauton River Watershed (Mass Audubon et al. 2017), the assessement of dams, bridges and culverts in the Wood-Pawcatuck Watershed (Fuss and O'Neill 2016), and the efforts by North Atlantic Aquatic Connectivity Collaborative (NAACC) to collect standarized data on stream barriers.
- Provided data collection can be expanded, bi-state efforts and approaches to refine the freshwater fish indicator could involve the development of an IBI or MMI for the Narragansett Bay Watershed. These resulting metrics can be related to the Biological Condition Gradient framework, as has been done in Connecticut (Stamp and Gerritsen 2013).
- Future data analysis should explore and quantify the relationships between freshwater fish metrics and stressors at appropriate scales (e.g., site, watershed, catchment areas). Armstrong and colleagues (2011) quantified the effects on fluvial fish abundance in response to alterations on stream flow and impervious cover, among other anthropogenic stressors.

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State of Narragansett Bay and Its Watershed
2017 Technical Report

Watershed Ecosystem Condition Indicators

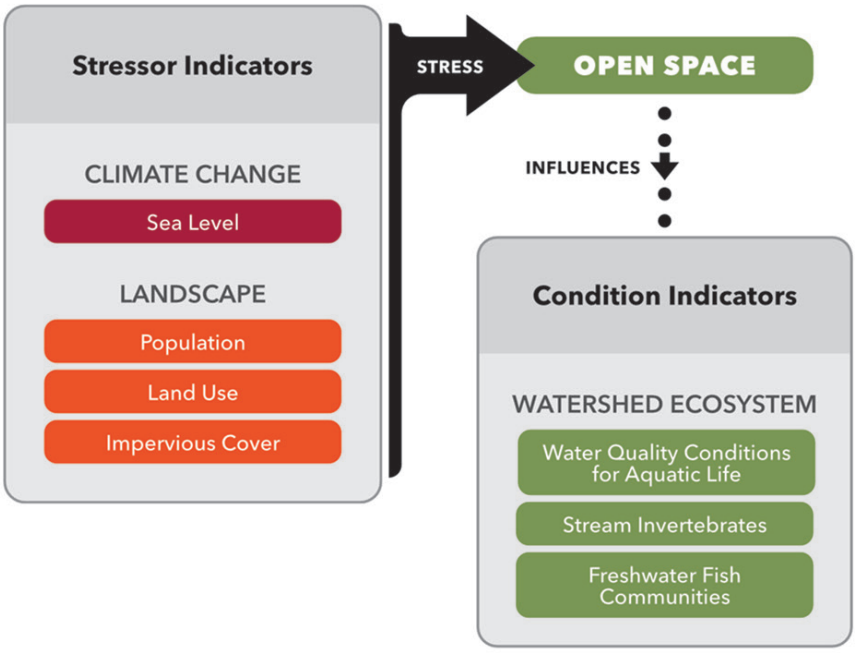
CHAPTER 21: OPEN SPACE

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Photo: Blackstone River Valley, Uxbridge, MA (Estuary Program)

NARRAGANSETT BAY ESTUARY PROGRAM

Overview



BACKGROUND

- Changes in land use in the Narragansett Bay Watershed, especially the conversion of natural lands to urban areas, affects hydrologic functions, alters the delivery of nutrients to rivers and the Bay, affects terrestrial, aquatic and estuarine habitat conditions, and contributes to increased pathogens in recreational and shellfishing waters. Land use changes that reduce natural lands are an indicator of habitat fragmentation and diminishing habitat value.

KEY FINDINGS

- ***Status***

  Natural Lands: As of 2015, 171,244 acres of natural open space lands–such as forests, wetlands, and pasture not for tillage–had been protected, representing over fifteen percent of the Narragansett Bay Watershed. Conversely, 41 percent of the natural lands in the Watershed remained unprotected.

  Ecologically Significant Natural Lands: Of the 268,794 acres of lands with high ecological integrity in the Watershed, 185,233 acres (seventeen percent of the Watershed) remained unprotected. These important lands serve as natural buffers and habitat

corridors, and enhance adjacent natural lands. However, because they are unprotected, they could be targeted for future development. The Mount Hope and Segreganset River Watershed Planning Areas in the Taunton River Basin had the highest ratios of unprotected to protected ecologically significant natural lands (20:1 and 18:1, respectively). These lands provide opportunities for enhancing open space protection and improving the resiliency of the Watershed to landscape, climatic, and chemical stressors.

Introduction

Natural open space lands are undeveloped lands that are either protected or unprotected from future development. In this chapter, protected open space lands are defined as lands subject to agreements by federal, state, municipal, and/or private entities that provide some level of protection from future development. The conversion from natural open space lands to developed lands is often part of the trend toward suburban sprawl (see "Population" and "Land Use" chapters). Conversion of natural lands usually results in habitat fragmentation by landscape barriers (created by roads and the subdivision of undeveloped parcels) or stream barriers (such as culverts, dams, and bridges). This development brings increased impervious cover that alters the hydrological regime of the Watershed and strips the natural vegetation that serves as both canopy and buffer to retain and slow stormwater runoff (see "Land Use" and "Impervious Cover" chapters). In addition, the loss of natural lands reduces the resiliency of the landscape to the impacts of climate change (see "Precipitation," "Temperature," and "Sea Level" chapters).

Open space lands provide a myriad of environmental amenities to people and the ecosystem. Natural lands such as forests, wetlands, riparian buffers, and some types of agricultural lands play important roles by providing wildlife habitat and protecting water quality (see "Stream Invertebrates" and "Freshwater Fish Communities" chapters). Open space lands also offer recreational, cultural, and aesthetic amenities.

The Narragansett Bay Estuary Program examined the extent of natural lands, both in protected and unprotected status, and the ecological significance of open space lands in both categories.

PROTECTED LANDS AS OPEN SPACE

Policies in both Massachusetts and Rhode Island have strongly encouraged the protection of open space. In Rhode Island, the planning document *Ocean State Outdoors: Rhode Island's Comprehensive Outdoor Recreation Plan* envisioned a statewide system of connected green spaces and greenways to form a network of critical natural and cultural resources, recreation facilities, public spaces, community and urban forests, and public and private open spaces (Rhode Island Statewide Planning Program and RIDEM 2009). Among the reasons to preserve open space are protecting water quality and quantity, wetlands and floodplains, islands and coastal areas, forests, agricultural lands, fish and wildlife, and cultural and scenic resources. *The Massachusetts Statewide Comprehensive Outdoor Recreation Plan* reported that land preservation efforts by state agencies and private conservation groups between 1999 and 2005 resulted in the statewide protection of 109,863 acres (MassEOEEA 2012). This plan also referenced Massachusetts Audubon Society's (Mass Audubon) findings that 22 acres of land were lost to development each day during that same period, causing over 30,000 acres of forestland and 10,000 acres of agricultural land to be converted from open space to development.

Studies in Massachusetts have examined fiscal and policy implications of open space preservation. According to the Trust for Public Land (2013), every dollar invested in land conservation returns four dollars in economic value of natural goods and services. When researchers with the Harvard Forest project analyzed four plausible scenarios for future land use, they found that recent trends in the loss of forests to development will undermine significant land conservation gains, and they recommended an increased pace of land conservation with a focus on threatened, undeveloped lands of high resource

value (Thompson et al. 2014). Mass Audubon published the fifth *Losing Ground* report (Lautzenheiser et al. 2014), citing that approximately 1.1 million acres of land in Massachusetts, representing 21 percent of the state, are developed, and over 1.25 million acres–25 percent of the state–are permanently protected. *Losing Ground* found that a majority of core habitat could be preserved if there were a strategic focus on protecting those lands; over 45 percent of core habitat (540,000 acres) already has been permanently protected (Woolsey et al. 2010, Lautzenheiser et al. 2014).

For the first part of this chapter, the Estuary Program calculated the extent of open space lands within the boundaries of the Narragansett Bay Watershed using three overarching land use categories: *(1) natural lands, (2) agricultural lands, and (3) recreational or other lands* (Extended Methods, Table 8 and Figure 4). This assessment focused on the natural lands that are protected as open space, since these areas provide the most benefits to habitat and water quality at the watershed level. The definition of natural lands by means of state land-use classification encompassed forest lands, wetlands, brushland, and other lands (Extended Methods, Table 8). These areas may be providing ecosystem benefits and services as riparian areas and floodplains, such as decreasing erosion, improving water quality, and maintaining river courses (Sweeney et al. 2004). The analysis of the extent of open space by land use categories reflects conditions as of 2015.

NATURAL LANDS AS ECOLOGICALLY SIGNIFICANT

Numerous initiatives have monitored the protection of open space and developed models for prioritizing unprotected lands of ecological significance at the state and regional level. Massachusetts has advanced a project known as BioMap2 to help guide land conservation, protection, and stewardship (Woolsey et al. 2010). In Massachusetts, 40 percent of lands classified in BioMap2 as Core Habitat and Critical Natural Landscape had not been protected as of 2010. These protected and unprotected lands encompassed a wide diversity of natural communities and intact ecosystems that support intact ecological processes, maintain connectivity among habitats, and enhance ecological resiliency.

One of the main inputs to develop BioMap2 was a spatial model of ecological integrity developed by researchers at the University of Massachusetts Amherst. The model, known as the Conservation Assessment and Prioritization System or CAPS, is an ecosystem-based approach for assessing the ecological integrity of lands and waters through landscape characterization and metrics (UMass Amherst website 2011). The overall purpose of CAPS was to delineate the relative wildlife habitat and biodiversity value of any point in the landscape. Areas with high ecological integrity, including forests, shrublands, coastal uplands, coastal wetlands, freshwater wetlands, and aquatic habitat, as defined by the data used to develop CAPS, can enhance and support watershed protection (IEI Maps website 2011).

Other efforts that rely on the robustness of the results from CAPS include the Mapping and Prioritizing Parcels for Resilience (MAPPR) tool (2.0) developed by Mass Audubon and their partners (Mass Audubon 2017). This online platform identifies and prioritizes unprotected areas at the parcel level that offer the greatest benefit for habitat quality, water resources protection, and climate change resilience. One of the key modeling features of the MAPPR tool is Critical Linkages (2012), which is derived from the CAPS model, and highlights important locations to maintain habitat connectivity and continuity. Applications of the MAPPR tool include identifying and targeting areas adjacent to existing open space that can be connected through land protection for a larger ecological benefit.

In Rhode Island, the 2015 Rhode Island Wildlife Action Plan similarly addressed strategic land preservation through a partnership between the Rhode Island Department of Environmental Management and the Rhode Island Chapter of The Nature Conservancy with project and technical assistance from the University of Rhode Island and the Rhode Island Natural History Survey (RIDEM 2015). The Wildlife Action Plan identified land acquisition as a critical component of wildlife habitat conservation and developed a land-preservation mapping tool called Conservation Opportunity Areas. These areas include large unfragmented core natural areas, corridors (rivers, wetlands, and other undeveloped lands that connect core areas), and sites (key habitats that contain high biodiversity value and are highly vulnerable, natural heritage areas, and ecological lands units) (RIDEM 2015). The Ecological Land Units (ELU) are used to identify areas for land protection for biodiversity in the context of climate change. Similar to BioMap2, the purpose of Conservation Opportunity Areas is to identify priority landscapes that can be more resilient in the long-term, as larger, high-quality, diverse, and connected natural areas provide better ecological and environmental benefits, and can increase effectiveness of conservation actions at a broader watershed scale.

The second part of this chapter focuses on protected and unprotected *natural lands* and *ecologically significant natural lands* based on CAPS; ecologically significant natural lands are used as a proxy for high-quality lands for habitat and water resources protection, two of the main components for watershed protection. The Estuary Program selected CAPS for the assessment of natural lands primarily because CAPS data were seamless across the Watershed, using a regional scale dataset that extends across the Northeast. Moreover, CAPS's definition of natural lands, based on an ecological community classification scheme (e.g., forest, coastal beach, shrub swamp, salt marsh, bog, pond), was consistent with the Estuary Program's definition by means of land use categories. In addition, CAPS data rank natural lands based on their ecological integrity from high to low, and thus it was possible to identify high-quality lands across the Watershed by means of Index of Ecological Integrity scores; in this chapter, these lands are referred to as "ecologically significant natural lands." CAPS data were also complementary in the analysis of this indicator to quantify unprotected and protected natural lands as open space.

In summary, this chapter reports on (1) the extent of open space areas as of 2015 by land use categories, focusing on the "natural lands" at a finer watershed scale, and (2) the extent of natural lands and ecologically significant natural lands in 2010 that were protected and unprotected. The results of this indicator should be considered at the Watershed scale. While this chapter is not meant for site-specific assessment such as targeting specific parcels for open space protection, it can assist with the identification of areas where conservation actions can take place and be further assessed at the appropriate scale.

Methods

The Estuary Program analyzed open space protection in the Narragansett Bay Watershed, using an array of geospatial analysis tools in ArcGIS Model Builder (Esri, 2016; see Extended Methods, Figures 4 and 5), in two ways:

1. By identifying land areas that were protected as open space by means of land use classification.
    - In this chapter, "open space" represents all areas that have some level of protection, such as conservation areas, recreational areas, and agricultural lands, as defined by the states (RIDEM and MassEOEEA).
    - A crosswalk of land use classes at the state level for both Massachusetts (2005) and Rhode Island (2011) was conducted to create three overarching land use categories: natural, agricultural, and recreational/other lands (Extended Methods, Table 8).
    - Land use areas by category integrated with open space areas define the natural, agricultural, recreational, or other lands that are protected (Extended Methods, Table 8 and Figure 4)
2. By identifying natural lands and ecologically significant natural lands that were protected and unprotected as open space, by means of the Conservation Assessment and Prioritization System (CAPS) (Extended Methods, Figure 5).
    - In this chapter, "natural lands" are defined as all protected and unprotected areas that are undeveloped and are not agricultural lands, by means of CAPS. Researchers at UMass Amherst map CAPS by defining undeveloped ("natural") land based on an ecological community classification scheme (e.g., forest, coastal beach, shrub swamp, salt marsh, bog, pond). This differs from the category of "protected natural lands" by means of land use because its extent only represents what is protected as open space, whereas CAPS represents the entire extent of natural lands in the Watershed, and hence it is appropriate to quantify the extent of what remains as unprotected. In addition, CAPS data provide information that allows the selection of lands that have high ecological integrity, for a watershed-scale analysis.
    - CAPS ranks the ecological value of points in the landscape using an Index of Ecological Integrity (IEI). For this analysis, "ecologically significant natural lands" were defined as areas with CAPS IEI scores that were higher than the average of all IEI scores across the Watershed.

The Estuary Program identified lands as "protected" open space and "unprotected" by means of an integrated analysis (a geospatial technique that integrates two features in the landscape with diverse and dissimilar values to one with a common scale of values) as follows:

- **Protected:** Open space data, as of 2015, from Massachusetts and Rhode Island were integrated with the land use categories to quantify the extent of "natural, agricultural and

recreational/other" open space areas (Extended Methods, Figure 4).

In addition, since CAPS also defines "natural lands," these data were used to quantify natural lands and ecologically significant natural lands that are within the open space boundaries, hence protected. The former was expected to be similar with the natural lands by means of state land use crosswalk; the latter was expected to identify that most of the existing protected natural lands as open space have high ecologically integrity (Extended Methods, Figure 5).

- **Unprotected:** CAPS data, as of 2010, that encompassed the entire region of the Watershed, were used to identify "natural lands" (encompassing all areas mapped by CAPS) and a subset as "ecologically significant natural lands," that were outside of the open space boundaries, hence unprotected (Extended Methods, Figure 5).

OPEN SPACE BY LAND USE CATEGORIES

Both the Massachusetts Executive Office of Environmental and Energy Affairs (EOEEA) and the Rhode Island Department of Environmental Management (RIDEM) maintain detailed inventories of open space. In this analysis, the Estuary Program used the protected lands data contained in those inventories as of June 2015.

The Massachusetts open space dataset, Protected and Recreational Open Space (MassGIS), contained the boundaries of conservation and outdoor recreational lands owned by federal, state, county, municipal, and nonprofit entities. In Rhode Island, RIDEM maintains two separate open space datasets. The first database, Conservation Lands: State of Rhode Island, includes approximate boundaries of conservation and recreation lands owned by the State through fee title, conservation easement, or deed restriction. The second database, Conservation Lands: Municipal and NGO, includes conservation lands protected from future development by organizations other than the State, such as municipalities and nonprofit organizations. In addition to permanently protected lands, a number of parcels were identified as "Conservation Intent," such as local parks, recreation areas, or some lands associated with cluster subdivisions. Not all of these lands are protected in perpetuity through legally enforceable restrictions, although nearly all have at least some level of protection.

The Estuary Program brought together and reconciled the Massachusetts and Rhode Island datasets of open space (as of 2015) and land use (MassGIS 2005 and RIGIS 2011), for which a crosswalk data analysis was conducted to reduced state land use classes to three overarching land use categories. Thus, the resulting categories of open space are as follows (Extended Methods, Table 8 and Figure 4):

- Protected natural lands (including forest, forested and non-forested wetlands, and pasture not suitable for tillage)
- Protected agricultural lands (including cropland, orchards, nurseries)
- Protected recreational and other lands (including golf courses, beaches, cemeteries)

The focus of this part of the indicator is on "protected natural lands" as open space. Other open space land categories also provide many recreational, agricultural, and cultural values but have more limited functionality for watershed protection.

An array of geospatial analysis tools was used to calculate the following at the scales of the entire Narragansett Bay Watershed and of River Basins:

- Total extent: acreage of protected natural, agricultural, and recreational/other lands.
- Percent of protected lands by category, within watershed units of each scale.

In addition, total extent and percent of protected natural lands was calculated at the Watershed Planning Areas scale. The various steps in the model developed by the Estuary Program to run these analyses are described in the Extended Methods (Figure 4).

NATURAL LANDS AS ECOLOGICALLY SIGNIFICANT LANDS, PROTECTED AND UNPROTECTED

CAPS data represent areas that are undeveloped and not agricultural, thus natural lands. To quantify the extent of lands in the Narragansett Bay Watershed that were ecologically significant to wildlife habitat, the Estuary Program used the Conservation Assessment and Prioritization System (CAPS) as a proxy for high-quality lands for watershed protection. The researchers who developed CAPS defined ecological integrity as the ability of an area to support biodiversity and ecosystem processes necessary to sustain biodiversity over the long term. The CAPS model can be run at different spatial scales and can show the importance of areas based on an index of ecological integrity. For example, the model can be used to identify the top ten percent or 50 percent of land most likely to provide the greatest ecological value.

Although a CAPS assessment for Massachusetts had been completed in 2011 (McGarigal 2011), the Estuary Program instead chose to use data from a 2010 CAPS assessment that covered the northeastern US region because it enabled seamless data analysis for both the Massachusetts and Rhode Island portions of the Watershed. The 2010 regional analysis was part of a larger sustainable landscapes project funded by the North Atlantic Landscape Conservation Cooperative (NALCC) (McGarigal 2017).

The regional CAPS model divided the entire region into cells of 30 by 30 meters (98.4 by 98.4 feet) and calculated an index of ecological integrity (IEI) for each cell. Based on the IEI score, the model identified areas ranging from intact ecosystems to more disturbed undeveloped and non-agricultural lands that were limited by ecological threats or barriers, such as proximity to roads, development, dams/ditches, and pollution, among others. The IEI scores range from 0.01 to 1 with a high score indicating maximum ecological integrity (typically occurring in forested areas isolated from roads and development) and a low score indicating minimum integrity (such as small natural areas surrounded by heavily developed lands).

To determine the highest priority lands for watershed protection, the Estuary Program identified "ecologically significant natural lands" in the Watershed. "Ecologically significant natural lands" were defined, based on expert recommendations, as areas with IEI scores above 0.49, which was the average score of all cell values across the Watershed. This criterion was selected because although areas with below-average scores can be important for habitat conservation and water resources protection, they tend to be surrounded by developed and fragmented areas, and offer less significant ecological benefits at the Watershed scale.

Using the CAPS data and IEI scores for the entire Narragansett Bay Watershed, the Estuary Program calculated for the full extent in acreage of natural lands:

- Total area (full range of IEI scores) including lakes and ponds
- Total land area (full range of IEI scores) not including lakes and ponds[1]
- Total land area as open space, thus protected (where natural lands intersected open space areas)
- Total land area that remains unprotected (natural lands outside the open space boundaries)

In addition, at a finer watershed scale, to determine natural lands that are ecologically significant as "protected" and "unprotected," the Estuary Program performed the following analyses for the Watershed Planning Areas nested in the River Basins[2] (Extended Methods, Figure 5):

- **Protected:** based on state inventories of open space, where ecological significant natural lands intersected open space areas.
- **Unprotected:** the remaining ecologically significant natural lands outside the open space boundaries.

The Estuary Program's analysis focused on watershed planning areas (WPAs) because they can be used as a geographical framework for protecting important natural lands. To identify WPAs with potentially the most opportunities to preserve more natural lands as open space, the Estuary Program calculated for each WPA:

- Ratio of unprotected to protected ecologically significant natural lands

  High values of this ratio indicate that a large proportion of the ecologically significant lands in the WPA have not been protected.
- Ratio of protected to unprotected ecologically significant lands

  High values of this ratio indicate that a large proportion of the ecologically significant lands in the WPA have been already protected.

Finally, in order to validate the data and methodology, the "protected natural lands" identified by means of the crosswalk of the states' land use categories were compared to the area in the CAPS analysis (IEI scores 0.01 to 1), which are expected to have similar extent. In addition, the total area of "protected" ecologically

[1] Lakes and ponds have varying IEI scores depending on the ecological integrity of the surrounding lands. The Estuary Program removed surface waterbodies from the subsequent analyses to maintain consistency with the states' definition of open space. Ponds and lakes are generally not included within the boundaries of open space areas, as delineated by the states. In addition, waterbodies are already protected under the federal law (Clean Water Act).

[2] The Estuary Program also conducted analyses at several other watershed scales including HUC10 watersheds and HUC12 subwatersheds, and at the municipal level. While this chapter reports only the results for WPAs, Basins, and the entire Watershed, the complete results for other geographical scales are available upon request.

significant natural lands by means of CAPS and by land use categories, were compared to identify whether existing protected natural lands have high ecological integrity.

Because of the differing data sources, methodologies, and metrics used across indicators in the *State of Narragansett Bay and Its Watershed*, the natural lands represented in this chapter are not comparable by definition with other indicators, such as forest lands (see "Land Use" chapter). However, an exploratory analysis by the Estuary Program determined that the CAPS data and the National Land Cover Dataset (NLCD) are comparable. When total area of forest lands and wetlands was calculated using CAPS or NLCD, the results differed by only one percent.

Status

PROTECTED OPEN SPACE AREAS BY LAND USE CATEGORIES

Open space areas that are protected include different land use types, encompassing natural, recreational, or other lands (such as cultural). The entire Narragansett Bay Watershed contains over 1,090,000 acres (over 1,700 square miles) of land of which 199,705 acres (312 square miles, 18.3 percent of the Watershed) were protected as of 2015 (Table 1; Figure 1). Of those protected lands, 171,244 acres (268 square miles) were categorized as protected natural lands, representing 15.7 percent of the entire Watershed. Only one percent of the total area of the Watershed were agricultural lands, and nearly two percent were recreational and other lands, including cemeteries and other historical and cultural facilities (Table 1; Figure 1).

While the Pawtuxet River Basin had the largest percentage of protected natural lands compared with other River Basins (Table 1), most of the protected natural lands across the Watershed were in the Taunton River Basin (54,374 acres, 32 percent of natural lands in the Narragansett Bay Watershed).

Because less than three percent of the open space are the combination of agricultural, recreation, or other land use types, these results are not presented at the finer scale of Watershed Planning Areas, while most of the open space were protected "natural lands."

Of the 42 Watershed Planning Areas (WPAs) in the Watershed, seven WPAs had over 25 percent of their area covered with protected natural lands, five of which are in the coastal areas of Narragansett Bay, encompassing eight percent of the total protected

Table 1. Lands in the Watershed and by River Basins, that are protected as open space, by land use category.

River Basins[1]	Total Open Space[2]		*Protected Natural Lands*[3]		*Protected Agricultural*[3]		*Protected Recreational and Other*[3]	
	Area (Acres)	Percentage of Area	Area (Acres)	Percentage of Area	Area (Acres)	Percentage of Area	Area (Acres)	Percentage of Area
Pawtuxet River	33,262	22.4	30,932	20.8	1,010	0.7	1,320	0.9
Taunton River	62,301	18.4	54,374	16.0	2,803	0.8	5,125	1.5
Coastal Narragansett Bay	53,790	17.9	41,600	13.9	4,615	1.5	7,576	2.5
Blackstone River	50,304	16.6	44,293	14.6	1,566	0.5	4,445	1.5
Total Narragansett Bay Watershed	**199,705**	**18.3**	**171,244**	**15.7**	**9,994**	**0.9**	**18,468**	**1.7**

[1] Total area and maps of the River Basins are provided in the Appendix.
[2] Encompasses all lands that are defined as open space: protected natural lands, agricultural, and recreational/other (Extended Methods, Figure 4).
[3] These categories are based on bi-state crosswalk of land use classes (Extended Methods, Table 8).

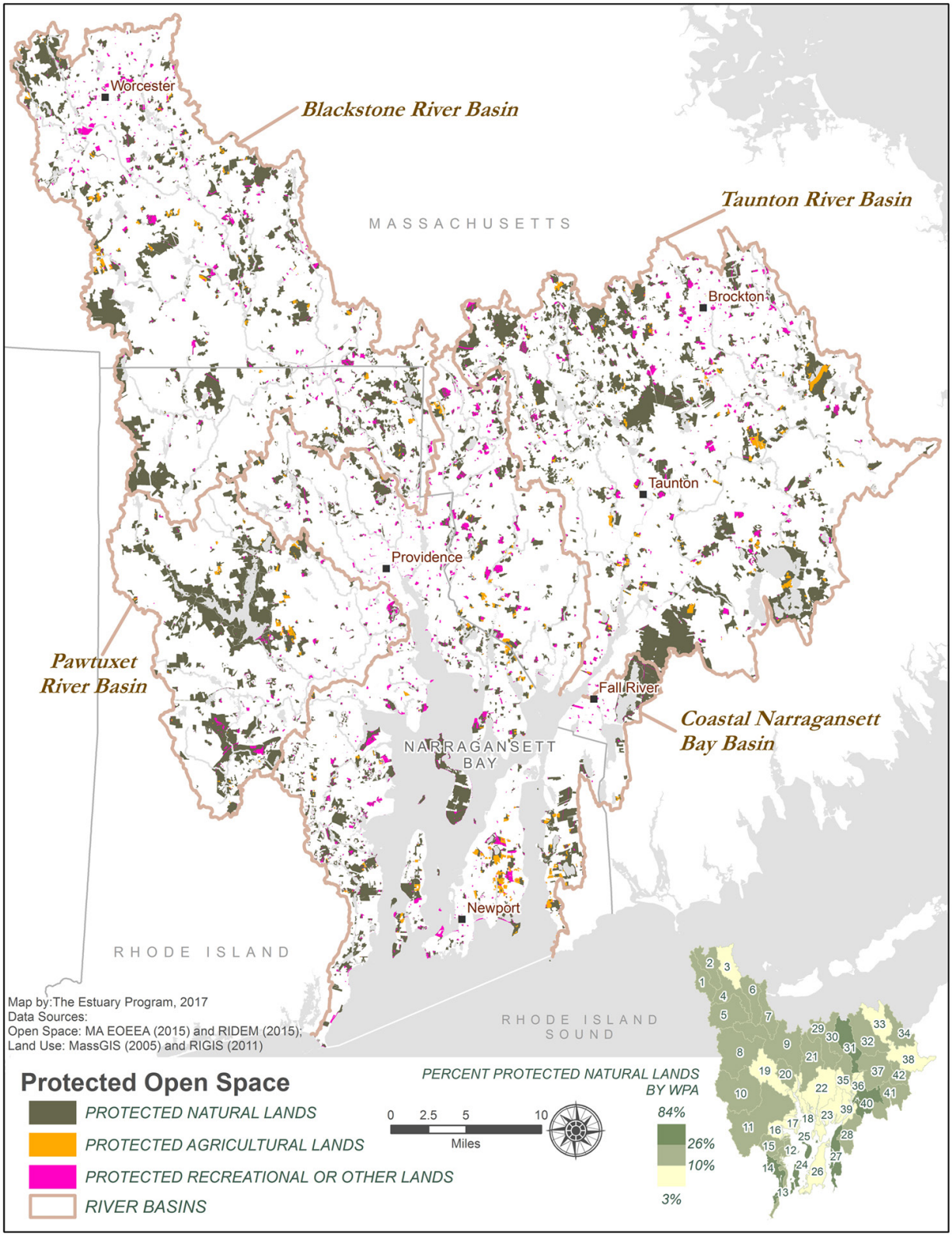


Figure 1. Open space in the Narragansett Bay Watershed based on land use categories. Background color of the Watershed Planning Areas indicates the percentage of natural lands protected as open space.

Table 2. Watershed Planning Areas in which protected natural lands constitute more than 25 percent of the total area.

Basin[1]	Watershed Planning Areas (WPAs)[1]	Protected Natural Lands	
		Total Acres	Percent of WPA[2]
	Greater than 25 percent of the WPA		
Coastal Narragansett Bay	Prudence Island	3,313	84.1
Taunton River	Assonet River	7,483	34.3
Coastal Narragansett Bay	Jamestown	1,821	30.1
Coastal Narragansett Bay	Sakonnet – East	5,272	29.8
Coastal Narragansett Bay	Narrow River	2,527	29.6
Coastal Narragansett Bay	Annaquatucket River	1,243	26.5
Taunton River	Mill River	7,518	26.1
	Less than 10 percent of WPA area		
Coastal Narragansett Bay	Aquidneck Island	2,304	9.4
Taunton River	Matfield River	2,452	9.2
Coastal Narragansett Bay	Bristol - Kickemuit River	2,257	9.0
Coastal Narragansett Bay	Barrington - Palmer - Warren Rivers	3,703	8.5
Coastal Narragansett Bay	Woonasquatucket River	2,752	8.4
Taunton River	Winnetuxet River-Taunton River	2,353	7.4
Blackstone River	Quinsigamond River	1,647	6.8
Coastal Narragansett Bay	Greenwich Bay	739	5.5
Coastal Narragansett Bay	Buckeye Brook	222	4.9
Taunton River	Taunton River	400	4.7
Taunton River	Segreganset River	413	4.4
Coastal Narragansett Bay	Providence - Seekonk Rivers	625	4.0
Taunton River	Mount Hope Bay	433	3.4

[1] Total area and maps of the Basins and the Watershed Planning Areas (WPAs) are provided in the Appendix.
[2] Relative to the total area of each WPA.

lands in the Watershed (Table 2; Figure 1). With 84 percent, Prudence Island had by far the highest proportion of protected natural lands. Conversely, thirteen WPAs had less than ten percent of their areas as protected natural lands, as expected in the most urbanized and developed areas in the Watershed (Table 2; Figure 1). These WPAs include cities and towns such as Providence and Newport in Rhode Island and Fall River, Worcester, and Brockton in Massachusetts.

PROTECTED AND UNPROTECTED NATURAL LANDS BY ECOLOGICAL INTEGRITY

Natural Lands

Because CAPS encompass all the natural lands in the Watershed, as defined by researchers at UMass Amherst, integrating open space lands and CAPS allowed the quantification of those natural lands that are both protected and unprotected. Thus, 55 percent of the Watershed is covered by natural lands and four percent of the Watershed is composed of fresh waterbodies including lakes, ponds, and reservoirs (Table 3). Nearly 150,000 acres of the CAPS natural lands are protected as open space amounting to fourteen percent of the total extent of natural lands in the Watershed (Table 3). This result is similar to the protected natural lands result of 15.7 percent in Table 1 using the bi-state crosswalk of land use classes, as expected. However, the 1.7 percentage point difference between the two results may be attributed to spatial, temporal, and methodological differences.

It is noteworthy that natural lands as defined by CAPS that are protected within the Watershed have an average IEI score of 0.56, indicating that existing protected areas as open space have overall high-quality lands for ecological integrity, above average across the Watershed (IEI greater than 0.49). However, this differs from the results presented thereafter because every cell (30 meters by 30 meters) representing a portion of natural land as ecologically significant has a IEI score greater than 0.49, whereas cells representing overall protected natural lands can have IEI scores lower or higher than 0.49.

ECOLOGICALLY SIGNIFICANT NATURAL LANDS

Extent of Protected and Unprotected Areas

Natural areas that offer substantial ecological benefits and provide ecosystem services to their surrounding lands, the Watershed, receiving waters, and ultimately the Bay are those that can be found as unfragmented portions of land, isolated from the disturbance of development, and can provide habitat connectivity more efficiently. These areas are considered "ecologically significant natural lands." Of the total area of 600,140 acres of CAPS natural lands (Table 3; Figure 2), 268,794 acres were identified as ecologically significant natural lands, which are areas (cells of 30 meters by 30 meters) whose CAPS IEI scores were above average, thus greater than 0.49 (Table 4; Figure 3). Twenty-five percent of the total area of the Watershed was composed of ecologically significant natural areas, largely concentrated in the

Table 3. Natural lands within the Narragansett Bay Watershed based on CAPS Index of Ecological Integrity (IEI).

Natural Lands (CAPS)	Narragansett Bay Watershed	
	Area (Acres)[1]	Percentage
Areas including IEI 0.01 to 1[2]		
Natural Lands (including lakes and ponds)	640,591	59
Natural Lands (excluding lakes and ponds)	600,140	55
Protected Natural Lands[3]	149,989	14
Unprotected Natural Lands[4]	450,151	41

[1] Total area and maps of the Watershed are provided in the Appendix.
[2] Encompasses all the areas from CAPS within the boundaries of the Watershed.
[3] Encompasses all the areas from CAPS that are within the open space areas.
[4] Encompasses all the areas from CAPS that are outside of the open space areas.

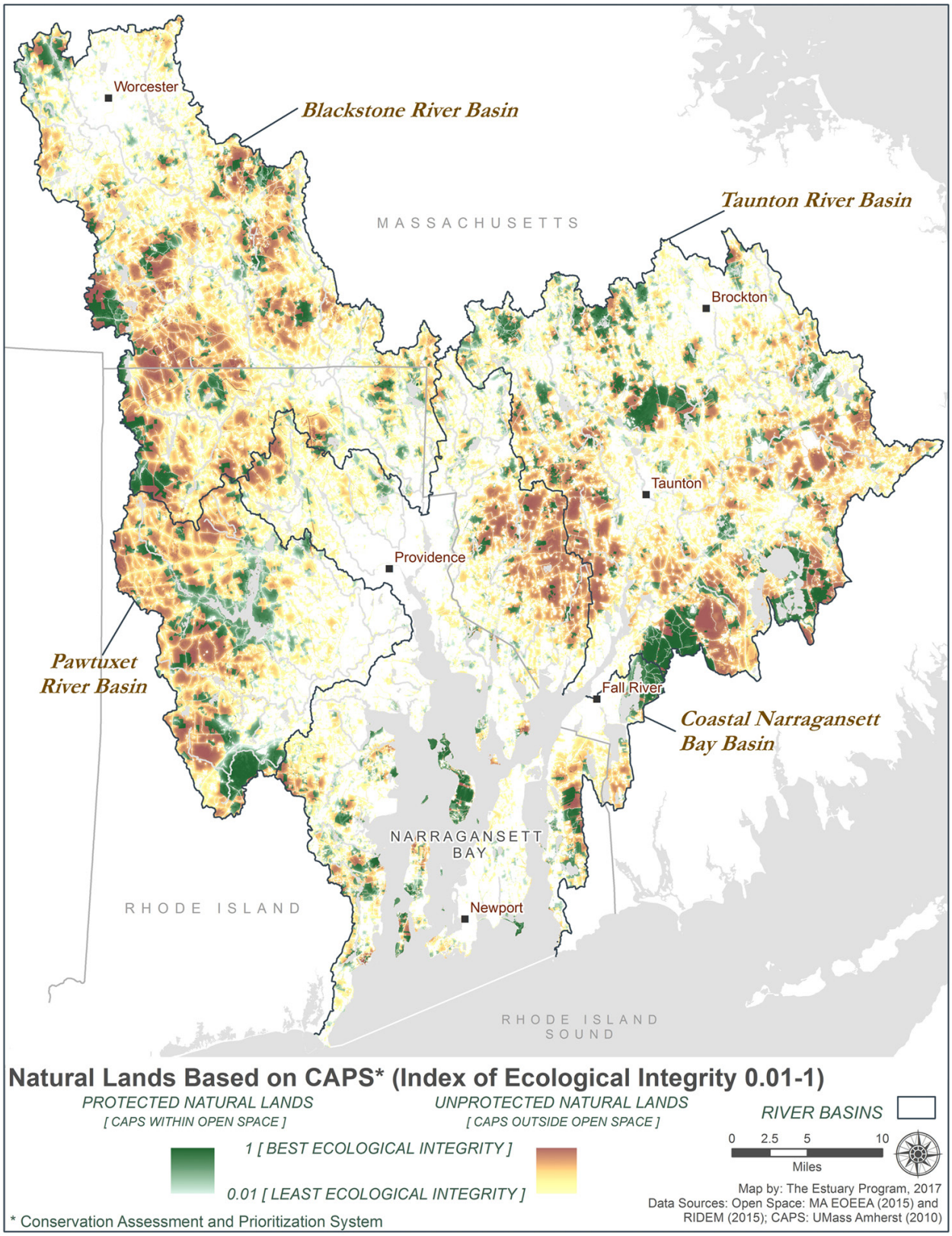


Figure 2. Natural lands in Narragansett Bay Watershed based on CAPS (IEI scores 0.01-1). Unprotected Natural Lands are represented with light brown (highest ecological integrity) to yellow (lowest ecological integrity). Protected Natural Lands are represented with dark green (highest ecological integrity) to light green (lowest ecological integrity).

upper reaches of the Watershed (Figure 3). Only eight percent of the total Watershed area consisted of protected ecologically significant natural lands, while seventeen percent was unprotected ecologically significant natural lands (Table 4).

Because open space lands are often identified in the context of watershed management plans and programs, the Estuary Program focused in presenting the results of these indicators for the 42 Watershed Planning Areas (WPAs) in the Narragansett Bay Watershed.

Nearly ten percent of the Pawtuxet River and Taunton River Basins were ecologically significant natural lands that were protected as open space (Table 5). However, between eighteen and 22 percent of the Pawtuxet, Taunton, and Blackstone River Basins were unprotected ecologically significant natural lands that could be potentially be protected for habitat and water resources (Table 5). As illustrated in Figure 3, a large proportion of these unprotected areas are adjacent to existing open space areas and could represent opportunities to create large, connected areas of protected lands, which are important for supporting the ecological condition of Narragansett Bay and its Watershed.

Ratio of Unprotected to Protected Areas

The ratio of unprotected to protected ecologically significant lands provides a mechanism to identify potential areas for enhanced open space preservation. Of the 42 Watershed Planning Areas (WPA) in the Narragansett Bay Watershed, the acreage of unprotected ecologically significant natural lands is at least double the acreage of protected ecologically significant lands in 23 WPAs (Table 6). Both the Mount Hope Bay and Segreganset River WPAs have the largest ratio, where unprotected ecologically significant lands are nearly 20 and eighteen times the acreage of the protected lands, respectively. In the Scituate Reservoir WPA, significant efforts have been made to protect natural lands, and there may be opportunities to continue these efforts, as the ratio of unprotected to protected lands there is 2.6 (Table 6).

Table 4. Protected and unprotected ecologically significant natural lands in the Narragansett Bay Watershed.

Ecologically Significant Natural Lands (IEI greater than 0.49)	Narragansett Bay Watershed	
	Area (Acres)	Percentage
Ecologically Significant Natural Lands	*268,794*	*25*
Protected	83,561	8
Unprotected	*185,233*	17

Table 5. Total area of protected and unprotected ecologically significant natural lands based on CAPS Index of Ecological Integrity in the River Basins within the Narragansett Bay Watershed.

River Basins [1]	Ecologically Significant Natural Lands			
	Protected		Unprotected	
	Area (Acres)	Percentage of Area	Area (Acres)	Percentage of Area
Pawtuxet River	15,814.3	10.7	33,008	22.2
Blackstone River	21,161.5	7.0	58,509	19.3
Taunton River	32,121.5	9.5	59,625	17.6
Coastal Narragansett Bay	14,447.2	4.8	34,056	11.3

[1] Total area and maps of the Basins are provided in the Appendix.

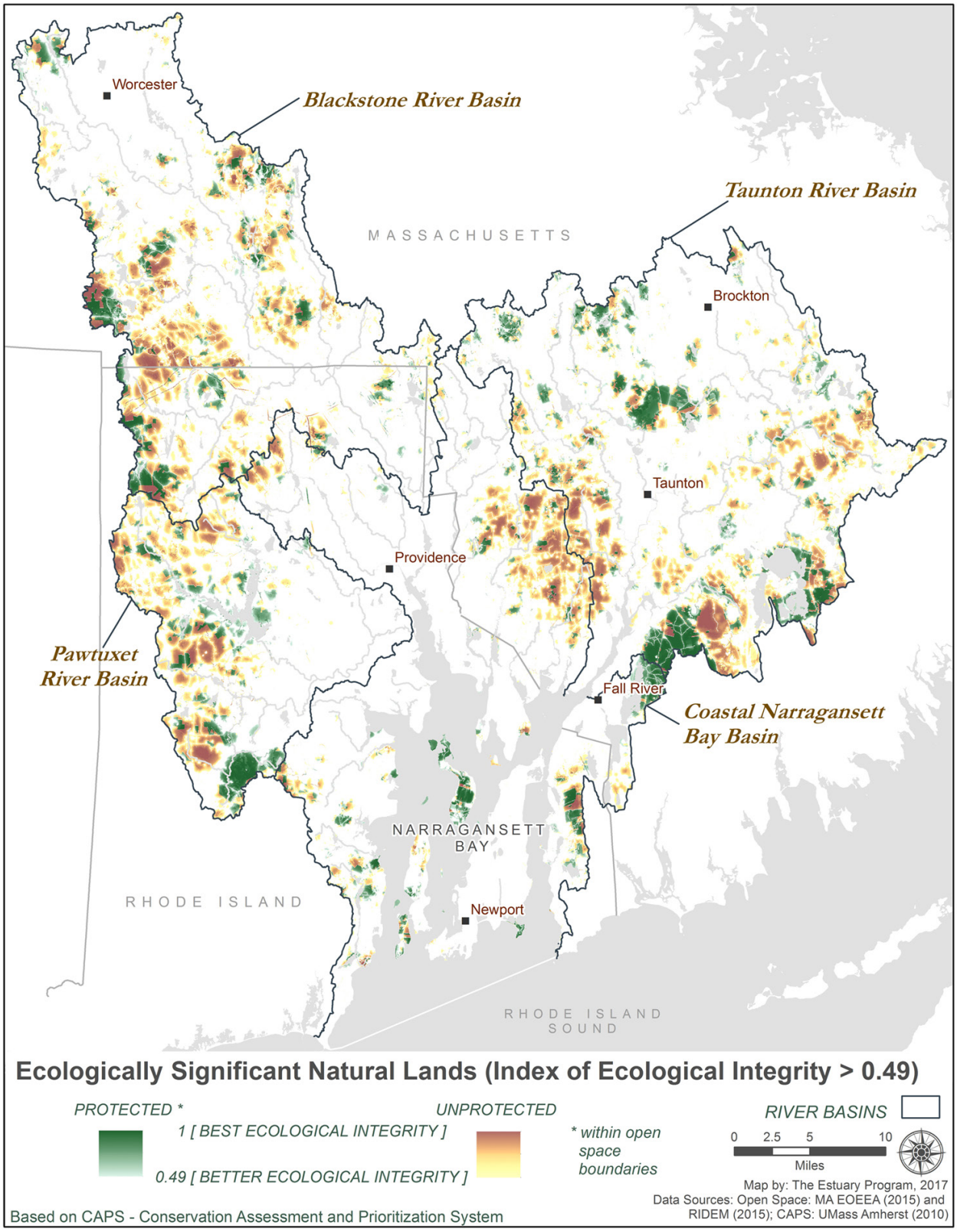


Figure 3. Ecologically Significant Natural Lands in Narragansett Bay Watershed based on CAPS with IEI greater than 0.49. Protected lands intersect open space areas, whereas Unprotected are outside of those areas.

Table 6. Ratio of unprotected to protected ecologically significant natural lands within the Watershed Planning Areas of the Narragansett Bay Watershed. This table does not include WPAs with ratios less than 1:1[1]. Sorted by highest to lowest ratio.

River Basin[2]	WPA[2][3]	Ratio Unprotected to Protected	Unprotected		Protected	
			Acres	Percent	Acres	Percent
	Ratio greater than 5:1					
Taunton River	Mount Hope Bay	20.8	1,958	15.3	94	0.7
Taunton River	Segreganset River	18.1	5,745	61.1	318	3.4
Coastal Narragansett Bay	Barrington - Palmer - Warren Rivers	9.1	14,471	33.4	1,597	3.7
Coastal Narragansett Bay	Bristol - Kickemuit River	8.7	4,869	19.4	560	2.2
Taunton River	Winnetuxet River-Taunton River	6.9	9,925	31.4	1,428	4.5
Coastal Narragansett Bay	Woonasquatucket River	6.6	3,112	9.5	474	1.4
Taunton River	Taunton River	6.3	839	9.9	133	1.6
	Ratio greater than 2:1					
Blackstone River	Quinsigamond River	4.8	550	2.3	115	0.5
Blackstone River	Lower Blackstone River	4.2	9,531	14.2	2,281	3.4
Coastal Narragansett Bay	Hunt River	3.7	2,062	13.2	557	3.6
Blackstone River	Kettle Brook	3.1	2,487	11.9	809	3.9
Taunton River	Nemasket River	3.0	3,271	24.4	1,075	8.0
Blackstone River	Branch River	2.9	20,236	33.9	6,866	11.5
Blackstone River	Mill River - Blackstone River	2.7	4,007	18.4	1,508	6.9
Pawtuxet River	Scituate Reservoir-Pawtuxet	2.6	19,795	33.0	7,605	12.7
Blackstone River	Mumford River	2.6	10,363	28.8	3,992	11.1
Taunton River	Cotley River-Taunton River	2.6	4,017	10.4	1,571	4.1
Blackstone River	West River	2.5	8,564	36.1	3,418	14.4
Taunton River	Threemile River	2.5	3,864	14.5	1,565	5.9
Blackstone River	Singletary Brook-Blackstone River	2.4	1,793	7.0	744	2.9
Coastal Narragansett Bay	Ten Mile River	2.2	1,755	4.9	781	2.2
Taunton River	Assawompsett Pond	2.1	9,933	31.6	4,660	14.8
Taunton River	Matfield River	2.1	976	3.6	472	1.8
	Ratio less than 2:1					
Pawtuxet River	Pawtuxet River	1.6	13,212	15.0	8,209	9.3
Taunton River	Wading River	1.5	4,295	15.4	2,783	10.0
Coastal Narragansett Bay	Sakonnet - East	1.4	2,494	14.1	1,838	10.4
Coastal Narragansett Bay	Narrow River	1.2	1,103	12.9	890	10.4
Taunton River	Satucket River	1.2	1,833	8.2	1,521	6.8
	Ratio 1:1					
Taunton River	Assonet River	1.0	6,562	30.1	6,426	29.5
Coastal Narragansett Bay	Annaquatucket River	1.0	6,426	29.5	6,562	30.08
Coastal Narragansett Bay	Jamestown	1.0	131	2.8	131	2.8

(1) The remaining WPAs have more protected than unprotected ecologically significant natural lands (Table 7).
(2) Total area and maps of the Basins and Watershed Planning Areas are provided in the Appendix.
(3) The Buckeye Brook WPA had only one acre of land identified as ecologically significant, and it was unprotected. Thus, a ratio was not applicable, and that WPA is not included in the table.

Ratio of Protected to Unprotected Areas

Alternatively, the ratio of protected to unprotected ecologically significant natural lands represents the successful efforts to protect a great proportion of the existing lands with high ecological integrity within the WPA. Of the 42 WPAs, eight WPAs had more protected than unprotected ecologically significant lands (Table 7). Prudence Island had six times greater acreage of protected ecologically significant natural lands than unprotected, as most of Prudence Island is protected as open space (85 percent) (Table 2). Some WPAs had very small areas of ecologically significant natural lands to begin with, and most of them have already been protected. While these WPAs had a small extent of ecologically significant lands, it remains important from a localized perspective to pursue continued protection, as there may be other areas that are important to protect for water quality and wildlife habitat but that cannot be identified at the resolution and scale of the analysis presented in this chapter, or within the criteria selected for this indicator. The WPAs of Providence River and Seekonk River are examples of places where site-specific efforts would be important because while this is one of the most developed and populated areas in the Watershed, their drainage area is also directly connected to the Bay. Thus, localized efforts to protect remaining natural lands can provide some environmental benefits such as mitigation of water quality pollution and erosion, and restoration of habitats.

Table 7. Ratio of protected to unprotected ecologically significant natural lands within the Watershed Planning Areas of the Narragansett Bay Watershed. This table does not include WPAs with ratios less than 1:1[1]. Sorted by highest to lowest ratio.

River Basin[2]	WPA[2][3]	Ratio of Protected to Unprotected	Protected		Unprotected	
			Acres	Percent	Acres	Percent
Coastal Narragansett Bay	Prudence Island	6.2	2,362	60.0	380	9.6
Coastal Narragansett Bay	Greenwich Bay	2.8	113	0.8	40	0.3
Coastal Narragansett Bay	Stafford Pond	1.8	2,650	13.8	1,477	7.7
Coastal Narragansett Bay	Aquidneck Island	1.8	290	1.2	164	0.7
Taunton River	Town River	1.7	5,478	14.1	3,205	8.3
Coastal Narragansett Bay	Providence - Seekonk Rivers	1.6	10	0.1	6	0.0
Coastal Narragansett Bay	Moshassuck River	1.5	258	1.7	171	1.2
Blackstone River	Tatnuck Brook-Blackstone River	1.5	1,429	5.8	976	3.9
Taunton River	Mill River	1.3	4,460	15.5	3,339	11.6
Coastal Narragansett Bay	West Bay	1.2	1,081	7.4	936	6.4

(1) The remaining WPAs have more unprotected than protected ecologically significant natural lands (Table 6).
(2) Total area and maps of the Basins and Watershed Planning Areas are provided in the Appendix.
(3) The Buckeye Brook WPA had only one acre of land identified as ecologically significant, and it was unprotected. Thus, a ratio was not applicable, and that WPA is not included in the table.

Discussion

Open space protection has been successful throughout the Narragansett Bay Watershed due in large part to the numerous entities involved in land conservation. Federal and state environmental agencies, cities and towns, land trusts, non-profit organizations, and thousands of private landowners have actively and collaboratively sought to preserve lands. Open space protection provides for the preservation of ground and surface water quality and quantity, ecosystem integrity, recreational and agricultural uses, and cultural, scenic, and historic values.

The protection of natural lands is critical to protect those features such as forests, riparian zones, and vegetated areas that protect water resources and the ecosystem. These undeveloped areas provide many benefits and services to the environment and the community such as nutrient management, water filtration, soil retention, connectivity of plant and wildlife habitat, air purification, climate regulation, carbon sequestration, protection from floods and droughts, and biodiversity. Federal, state, and local entities protect areas of natural lands through regulation (e.g., wetlands) and incentives. However, extensive natural areas of high ecological value are threatened by fragmentation and conversion to other land uses such as residential and commercial development (see "Land Use" chapter). While the findings that 15.7 percent of the Narragansett Bay Watershed is categorized as protected natural lands (Table 1; Figure 1) and 25 percent of the Watershed is covered by ecologically significant natural lands (Table 4; Figure 3) are important, it is equally important–if not more so–to recognize that seventeen percent of the Watershed–185,233 acres–is unprotected ecologically significant natural lands. The willingness of citizens to support the funding of open space conservation has been well established in Massachusetts and Rhode Island, and the results presented in this report demonstrate that continued support is necessary.

Because many open space efforts are addressed by local land trusts, the examination of open space in the WPAs is important to better link open space and water quality protection. The analysis of the ratio of unprotected to protected ecologically significant lands within WPAs highlights various priority areas that can target open space preservation efforts. For example, 61 percent of the Segreganset River WPA, in the Taunton River Basin, is unprotected ecologically significant natural land, eighteen times greater than the acreage of the lands that are protected (Table 6). In addition to the percent of unprotected ecologically significant natural lands in a WPA, it is important to view the acreage or extent of unprotected significant natural lands within each WPA when identifying important areas to focus protection. The Branch River and Scituate Reservoir WPAs have substantial areas of unprotected significant lands (Table 6). The ratios of unprotected to protected ecologically significant natural lands provide insight into areas that are unbalanced in terms of open space protection, and similarly, the spatial extent of unprotected ecologically significant natural lands is important to consider. While this chapter focused on those important lands that can provide larger environmental benefits at the Narragansett Bay Watershed scale, such as within the Taunton River or Blackstone River Basins, other lands, at finer spatial scales, also need to be examined for their contribution to protected open space. This examination can be conducted with a combination of regional, state, and watershed efforts that can assist with the tools, data, and information for better decision making, starting with the fact that 41 percent of the Watershed comprises natural lands that remain at risk of development (Table 3; Figure 2). These unprotected natural lands, whether large or small in extent, provide opportunities to enhance natural land protection.

Another mechanism to analyze the same data is to identify those WPAs where there is a high watershed percentage or a large number of acres of unprotected ecologically significant natural lands (Table 6). The Branch River WPA stands out in terms of both total acres and percent unprotected. While significant areas of natural lands have been protected around the Scituate Reservoir, this planning area contains a large percentage and extent of unprotected ecologically significant lands. It is important to note that WPAs with apparent opportunities to improve open space protection can be identified in Table 6, but it must be stated that diligent efforts for additional open space protection are needed in all WPAs in the Narragansett Bay Watershed, and local efforts could use these results to identity important target areas for preservation, and then validate and/or implement actions at the site scale.

Data Gaps and Research Needs

- Geospatial tools should be used to identify unprotected open space parcels adjacent to currently protected open space parcels. Protecting these natural areas would augment habitat connectivity, increase natural buffers to receiving waters, and improve the resilience of the ecosystem to land use stressors and climate change.

- In addition to CAPS, other tools are useful for open space decision making. Critical Linkages (2012) identifies locations in the landscape that can provide greater ecological benefits to increase connectivity and continuity of habitats. Mass Audubon's Mapping and Prioritizing Parcels for Resilience tool identifies priority parcels for open space protection based on habitat quality, climate change resilience, parcel size, and adjacency to existing protected parcels. Use of such tools should be pursued to assist with planning efforts in the Narragansett Bay Watershed.
- Further analyses of riparian buffer protection and restoration opportunities should be developed at a range of watershed scales, including Watershed Planning Areas.
- Further refinement via a parcel-based analysis is needed to more specifically identify unprotected lands that may provide restoration opportunities such as areas for salt marsh migration as sea levels rise.
- Spatial analyses of open space changes conducted at intervals of a decade or less, with a focus on protected ecologically significant natural lands, are necessary to track advances and spatial trends in conservation in the Narragansett Bay Watershed.

Acknowledgments

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Extended Methods

Table 8. Crosswalk of all state land use classes to the Estuary Program's three land use categories.

Land Use Classes (MA and RI)[3]	Land Use Category for Open Space Analysis
Brushland (shrub and brush areas, reforestation)	Natural
Brushland/Successional	Natural
Deciduous Forest (>80% hardwood)	Natural
Forest	Natural
Forested Wetland	Natural
Mixed Barren Areas	Natural
Mixed Forest	Natural
Non-Forested Wetland	Natural
Open Land	Natural
Pasture (agricultural not suitable for tillage)	Natural
Rock Outcrops	Natural
Saltwater Wetland	Natural
Sandy Areas (not beaches)	Natural
Softwood Forest (>80% softwood)	Natural
Water	Natural
Wetland	Natural
Confined Feeding Operations	Agricultural
Cranberry Bog	Agricultural
Cropland	Agricultural
Cropland (tillable)	Agricultural
Idle Agriculture (abandoned fields and orchards)	Agricultural
Nursery	Agricultural
Orchard	Agricultural
Orchards, Groves, Nurseries	Agricultural
Pasture	Agricultural
Beaches	Recreational
Developed Recreation (all recreation)	Recreational
Golf Course	Recreational
Participation Recreation	Recreational
Saltwater Sandy Beach	Recreational
Spectator Recreation	Recreational
Water-Based Recreation	Recreational
Airports (and associated facilities)	Other

[3] This is inclusive of all state lands use classes. It does not necessarily indicate that all these classes were found within the open space boundaries. Based on methods and tools used for analysis, when land use and open space areas were integrated, and thus protected, areas outside the open space boundaries were removed, by means of intersection (Figure 4).

Table 8 continued.

Land Use Classes (MA and RI)[3]	Land Use Category for Open Space Analysis
Cemeteries	Other
Cemetery	Other
Commercial	Other
Commercial (sale of products and services)	Other
Commercial/Industrial Mixed	Other
High Density Residential	Other
High Density Residential (<1/8 acre lots)	Other
Industrial	Other
Industrial (manufacturing, design, assembly, etc.)	Other
Institutional (schools, hospitals, churches, etc.)	Other
Junkyard	Other
Low Density Residential	Other
Low Density Residential (>2 acre lots)	Other
Marina	Other
Medium Density Residential	Other
Medium Density Residential (1 to 1/4 acre lots)	Other
Medium High Density Residential (1/4 to 1/8 acre lots)	Other
Medium Low Density Residential (1 to 2 acre lots)	Other
Mines, Quarries and Gravel Pits	Other
Mining	Other
Multi-Family Residential	Other
Other Transportation (terminals, docks, etc.)	Other
Power Lines (100' or more width)	Other
Powerline/Utility	Other
Railroads (and associated facilities)	Other
Roads (divided highways >200' plus related facilities)	Other
Transitional	Other
Transitional Areas (urban open)	Other
Transportation	Other
Urban Public/Institutional	Other
Vacant Land	Other
Very Low Density Residential	Other
Waste Disposal	Other
Waste Disposal (landfills, junkyards, etc.)	Other
Water and Sewage Treatment	Other
Commercial/Residential Mixed	Other

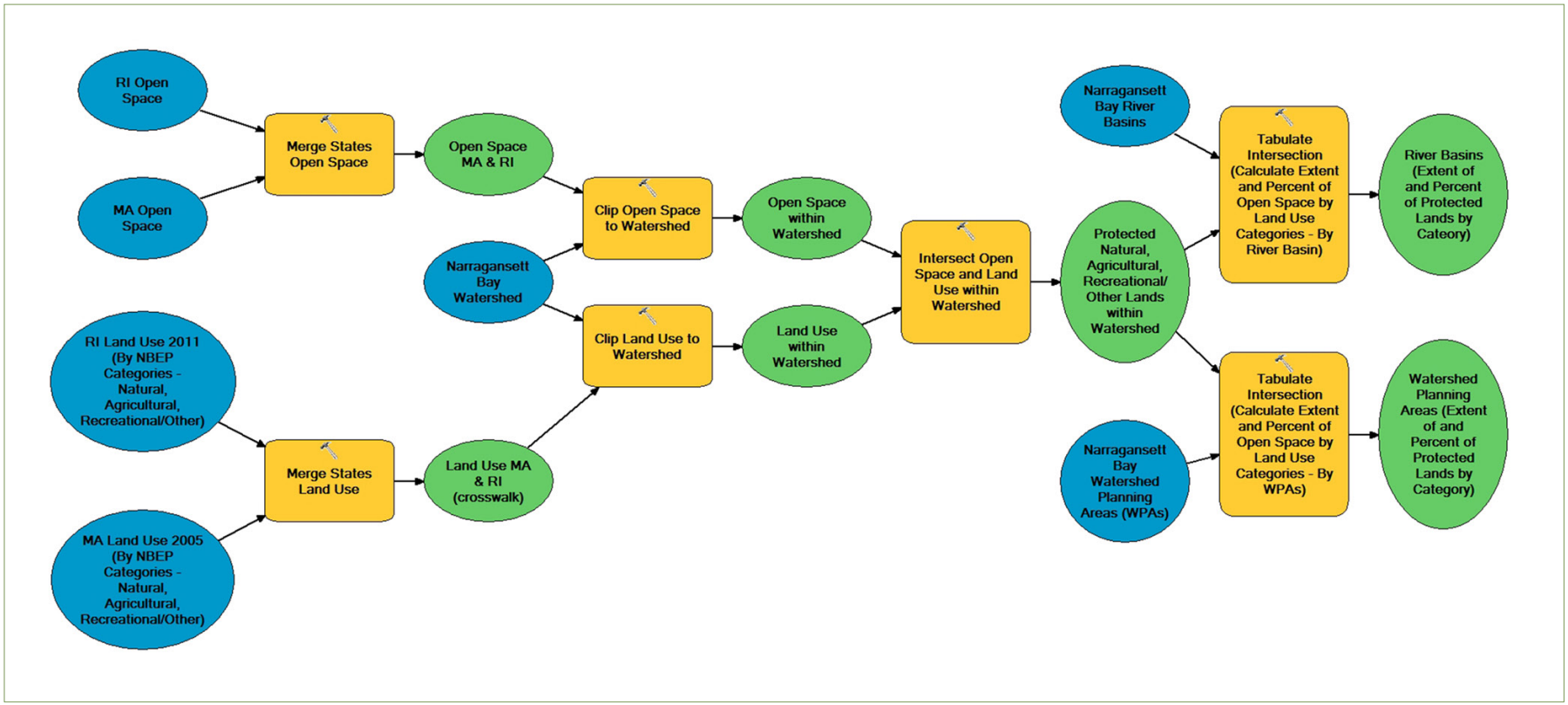


Figure 4. Model Builder (Esri 2016) developed by the Estuary Program, showing input and output data, and tools, used to create and calculate the extent of Protected Lands by Land Use Category within Narragansett Bay Watershed, River Basins, and Watershed Planning Areas.

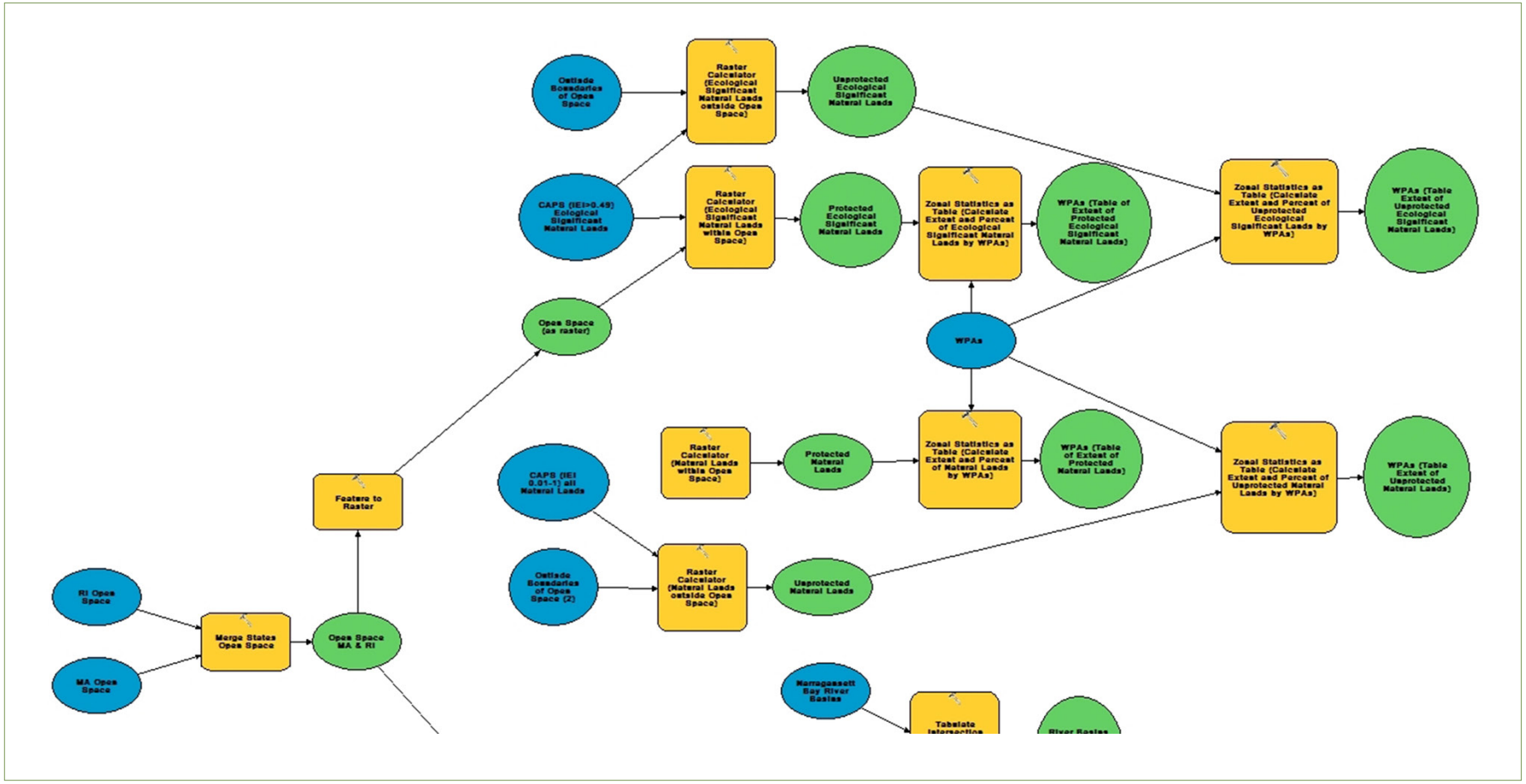


Figure 5. Continuation (Figure 4) of Model Builder (Esri 2016) developed by the Estuary Program, showing input and output data, and tools, used to create and calculate the extent of Protected and Unprotected Natural Lands and Ecologically Significant Natural Lands in Narragansett Bay Watershed by Watershed Planning Areas.



State of Narragansett Bay and Its Watershed
2017 Technical Report

Condition Indicators of Public Health

INTRODUCTION

Please use the following citation: Narragansett Bay Estuary Program. 2017. State of Narragansett Bay and Its Watershed (Public Health Introduction, pages 415-416). Technical Report. Providence, RI.

Photo: Upper Providence River, RI (Ayla Fox)

Protecting the public from harmful pathogens is a paramount objective of environmental management efforts in the Narragansett Bay Watershed. High pathogen levels in waters used for recreation, including swimming and boating, and shellfishing create unsafe conditions for public health, as people use the Bay and Watershed for these activities. Pathogen contamination includes the presence of *Escherichia coli*, a type of fecal coliform, general fecal coliform, and Enterococci bacteria. These three pathogens are monitored and regulated by federal and state departments of environmental management and health. Unsafe levels of pathogens stem from untreated human and animal feces entering the water from failing septic systems, cesspools, combined sewer overflows (CSOs) and other sources of nonpoint pollution.

Increased pathogen contamination results in beach closures, impairment of waters designated for recreational activities, and acreage reduction of approved shellfishing areas designated for direct human consumption. While wastewater treatment facilities have been successfully reducing or eliminating pathogen loads to receiving waters, especially in the last 15 years, individual failing septic systems and cesspools are considered a major culprit for pathogen contamination. However, little is known about the extent or severity of impacts on freshwaters and estuarine waters from failing septic systems and cesspools. Stressors discussed in other sections of this report can exacerbate pathogen contamination, particularly precipitation and resulting stormwater runoff from impervious cover, agricultural lands and other open fields with wildlife and pet feces, primarily. The greater intensity and volume of precipitation associated with climate change is expected to increase pathogen loading, and warmer water temperatures will encourage more pathogen growth. Efforts to reduce pathogen contamination in the Watershed include engineered retention systems, green infrastructure, pet waste management, and construction of tunnels to store CSO discharges for later treatment. These efforts have resulted in improvements to the overall water quality in the last decade.

The Narragansett Bay Estuary Program uses three indicators to monitor public health conditions in the Bay and Watershed: marine beach closures, shellfishing area closures, and water quality monitoring for recreation. The analysis results reflect the efforts by the state agencies to collect, assess, and determine whether waters are suitable for recreational use or shellfishing. Accordingly, the following chapters discuss status and trends and explore how management actions affect spatial and temporal changes in these indicators that are relevant to the daily lives of the population. This section concludes the report by coming full circle. People are the ultimate driver of water quality conditions. Through understanding how stressors can affect the wellbeing of people in the Bay and Watershed, we can also understand how the Bay and Watershed have been substantially been shaped and resources used by people over thousands of years.



Photo: Bonnet Shores Beach Club, Narragansett, RI (Ayla Fox)



State of Narragansett Bay and Its Watershed
2017 Technical Report

Condition Indicators of Public Health

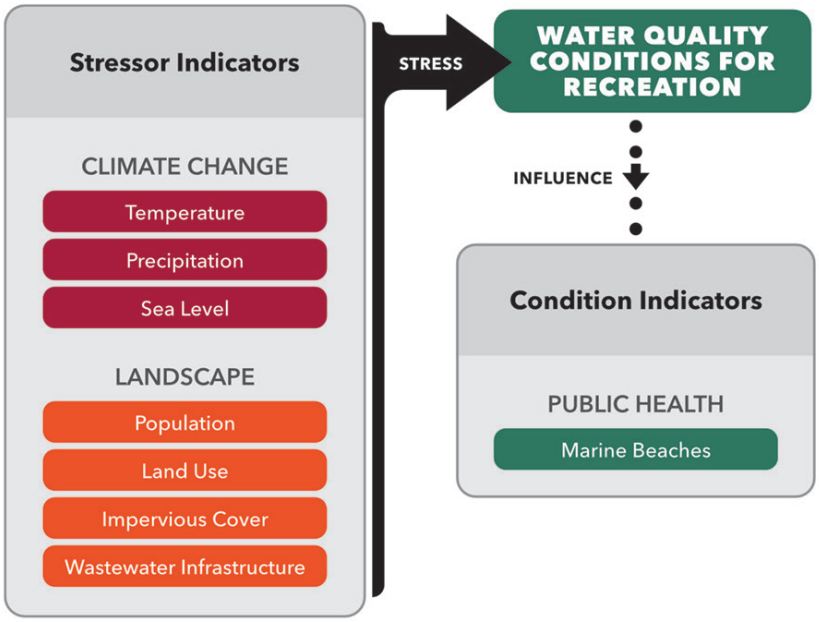
CHAPTER 22: WATER QUALITY CONDITIONS FOR RECREATION

Please use the following citation: Narragansett Bay Estuary Program. 2017. State of Narragansett Bay and Its Watershed (Chapter 22, Water Quality Conditions for Recreation, pages 417-436). Technical Report. Providence, RI.

Photo: Save The Bay Swim from Newport to Jamestown, RI (Save The Bay)

NARRAGANSETT BAY ESTUARY PROGRAM

Overview



BACKGROUND

- Recreational activities in estuarine and freshwaters can be hindered by pathogen pollution from stormwater runoff and the discharge of untreated or poorly treated wastewater, among other sources. Monitoring by the states reveals that the concentrations of pathogens in surface waters are typically higher following precipitation events.

KEY FINDINGS

- ***Status***

  Estuarine Waters: Most of the estuarine waters of Narragansett Bay (85 percent) were suitable for recreational use. Some of the exceptions included estuarine waters in the Providence River, Palmer River, Newport Harbor, lower Mount Hope Bay, and some embayments in Greenwich Bay.

  Streams and Rivers: Over 60 percent of the freshwater streams and rivers in the Narragansett Bay Watershed were not suitable for recreational uses such as swimming and boating. In the Taunton River Basin and the Coastal Narragansett Bay Basin, more than 80 percent of streams and rivers were unacceptable for recreational use due to pathogens, whereas 67 percent in the Pawtuxet River Basin were acceptable.

Lakes and Ponds: Of the total acreage of lakes and ponds in the Watershed assessed for recreational use, 80 percent was acceptable. In both the Pawtuxet River Basin and the Coastal Narragansett Bay Basin, over 95 percent of waters in lakes and ponds were acceptable for recreational use. Pathogens were the cause of less than four percent of lakes and ponds in the Watershed being classified as unacceptable for recreational use. Parameters other than pathogens made nearly 45 percent of the waterbodies in the Massachusetts portion of the Watershed unacceptable for recreational use.

Over 40 percent of the fresh waters in the Watershed were not assessed by the states for recreational use. The Pawtuxet River Basin had the lowest percentage of unassessed waters for recreational use, while the Taunton River Basin had the highest percentage with over 60 percent unassessed.

Introduction

Streams, rivers, lakes, and ponds in the Narragansett Bay Watershed and the estuarine waters of the Bay itself support a variety of recreational activities such as fishing, swimming, and boating. These activities provide significant economic and cultural value to the region.

Fecal coliforms, *Escherichia coli,* and Enterococci are considered primary bacterial indicators for the presence of human pathogens in waters. Exposure to harmful microorganisms through recreation such as swimming and boating can cause health impacts such as gastroenteritis and sore throats, or even meningitis or encephalitis (Cabelli 1983 and 1989, USEPA 1986, Haile 1996, Pruss 1998). Fecal pathogens are implicated as the leading cause of water quality impairment in the country (USEPA 2016).

Various sources have been linked with elevated pathogens in water bodies, such as failing septic systems, storm runoff, agricultural activities, wildlife and waterfowl, and discharges from wastewater treatment systems particularly when combined sewer overflows (CSOs) are triggered. During precipitation events, impervious surfaces facilitate the transport of pathogens in runoff to receiving waters: freshwater streams and rivers, lakes and ponds, and ultimately the Bay. Researchers have found that impervious cover has been positively correlated, and natural land cover negatively correlated, to higher bacterial counts in streams, estuaries, and lakes (Holland et al. 2004, Dolah et al. 2007, Didonato et al. 2009, Crim et al. 2012, Haack et al. 2013, Sanger et al. 2015).

Untreated sewage was historically a major source of pollution across Narragansett Bay. In pre-colonial times and prior to industrialization, lower population densities around Narragansett Bay resulted in lower anthropogenic pathogen input. This changed when industrialization brought a threefold increase in the Watershed's population, primarily along coastal areas of the Upper Estuary (see "Population" chapter; see the Appendix for Bay Regions). As a consequence, wastewater became the major source of pollution across Narragansett Bay. Even as sewered areas began to increase in the early 1900s, pipes funneled untreated sewage directly into Bay waters (Schumann 2015), discharging human pathogen loads from household waste, in addition to industrial waste (see "Wastewater Infrastructure" chapter). In the mid-1900s, inadequately treated sewage and raw sewage continued to be the main source of pollution in the upper Narragansett Bay (Shea 1946).

Today, wastewater treatment plants have been upgraded significantly, resulting in reduced pathogen loadings. However, wastewater systems may release pathogens when they experience sewer system overflows (SSOs) resulting from breaks in pipes or other operational problems. Onsite (individual) wastewater treatment systems (OWTS) and cesspools may also contribute to pathogen water pollution problems through failures or systems that are not properly designed, sited, and maintained (Habteselassie et al. 2011, Humphrey et al. 2011, Sowah et al 2014 and 2017, Schneeberger et al. 2015). Adverse impacts from onsite systems, particularly cesspools or failing OWTS, generally would be expected to be localized to subembayments, river

segments, or receiving ponds and lakes, depending on groundwater flow direction and the type of underlying soils, among other factors (see "Wastewater Infrastructure" chapter).

Discharge of non-point source stormwater runoff occurs throughout the Bay and its Watershed and is known or suspected to contribute to the water pollution that impairs recreational uses of surface waters. Combined sewer overflows (CSOs) are a major point source of pathogens; however, CSOs affect only certain portions of Narragansett Bay–waters in the Providence River Estuary, Newport Harbor, and Fall River, where their outfalls are located. Significant investment in CSO abatement has reduced, but not entirely eliminated, the volume of untreated CSO events.

Climate change will likely increase the influence of precipitation and temperature on water quality conditions (see "Precipitation" and "Temperature" chapters). Heavy precipitation and flooding associated with more frequent and intense storms may amplify pathogen transport to receiving waters. Moreover, increasing water temperatures may allow pathogens to survive for longer periods of time. Analyzing the extent and distribution of waters within the Narragansett Bay Watershed that are impacted by pathogens informs an understanding of public health risks, as well as long-term efforts to improve water quality.

In accordance with the federal Clean Water Act (CWA), Rhode Island and Massachusetts have identified and designated waters according to specific uses and periodically publish these results in Integrated Reports submitted to the Environmental Protection Agency (EPA) (MassDEP 2015, RIDEM 2015). Measurement of pathogens, in conjunction with other factors, can inform state and local management decisions to limit recreational activities including closing beaches to swimming (MassDEP 2015, RIDEM 2015). In this chapter, the Narragansett Bay Estuary Program utilized and reconciled data from the water quality assessments, compiled by both states, for both primary and secondary contact recreational use. Primary contact refers to recreational activities in which humans have direct contact with the water, such as swimming. Secondary contact refers to recreational activities occurring on the water where there is indirect contact, such as canoeing or kayaking. While both states use different pathogen criteria, they lead to similar determinations as to whether conditions are acceptable for recreational use.

RIDEM assesses non-designated bathing beach waters in Rhode Island. RIDEM's numeric criteria for fecal coliforms, Enterococci, and *Escheria coli* in non-designated bathing beach waters are summarized in the Extended Methods (Table 8) section at the end of this chapter. It should be noted that the Rhode Island Department of Health (RIDOH) also assesses water quality conditions at public marine and freshwater beaches during the summer season (Memorial Day to Labor Day). These assessments are used to daily notify the public whether a beach is open or closed for swimming due to high concentrations of pathogens, and are not analyzed within this chapter (see "Marine Beaches" chapter). In Massachusetts, MassDEP water quality conditions criteria vary for primary contact use testing at public bathing beaches during the bathing and non-bathing season as well as for secondary contact use testing, as discussed in the Extended Methods section (Table 9). For primary contact recreational use, Enterococci is tested in both fresh and estuarine waters, while for secondary contact recreational use, it is tested only in estuarine waters. *Escherichia coli* is tested only in fresh waters, for both primary and secondary contact recreational use. Moreover, marine beaches are monitored frequently during the summer season by the Massachusetts Department of Health (see "Marine Beaches" chapter), and in collaboration with local officials and entities, freshwater beaches are also tested for pathogen contamination.

In this chapter, the Estuary Program's report focuses on the water quality assessments by the states, which include attainment of "fully supporting," "not supporting," or "unknown" status and causes of impairment for recreational use, largely due to pathogens as the primary indicator of public health. To measure this indicator, the Estuary Program integrated information about: (a) the extent of state water quality assessments within the broader hydrological context of the Bay and Watershed in both Massachusetts and Rhode Island, and (b) the status of water quality conditions for recreational uses in Narragansett Bay and its Watershed, as assessed by the states. Only a proportion of all waters that define and shape the Watershed and the Bay are assessed by the states, and some of these waters have sufficient information to be assessed for recreational use, primarily based on testing for pathogens.

It is important to acknowledge that while the results from this analysis are consistent with the Integrated Reports from both states (MassDEP 2015 and RIDEM 2015), the water quality conditions for this indicator as defined by the Estuary Program reflect that (a) the extent of acceptable waters is equivalent to the

extent of state-assessed waters where conditions are "fully supporting" of recreational use, (b) the extent of state-assessed waters that are impacted for recreational use differs from states' reporting of impaired waters or the 303(d) list by reporting only on "not supporting" due to pathogens (or other parameters) for recreational use, and (c) waters identified as unknown were not assessed for recreational use by the states.

By measuring the total extent of waters that are assessed or not for recreational use, across the geographical scope of the Watershed and River Basins, these results can shed light on three issues: waters with acceptable conditions for recreational use should be protected; waters that were identified as impacted for recreational due to pathogens, primarily, can pinpoint potential sources of contamination and stressors; and waters that are unknown can be identified as data gaps within the state assessments to further evaluate other state-assessed waters for recreational use.

Methods

EXTENT OF STATE-ASSESSED WATERS

The Estuary Program calculated the total extent of fresh and estuarine waters within the Bay and Watershed to compare with the extent of state water quality assessed waters. The states do not assess every stream segment or pond or lake in the Watershed, nor all the areas in the Bay. Waters assessed by the states for a variety of uses are only a portion of the entire hydrological network in the Watershed and Bay waters due to states' criteria for implementing water body assessments, as well as the states' differing capacities.

For the purposes of this chapter, the total extent of freshwaters is defined by the National Hydrography Dataset (NHD). The Estuary Program calculated total extents using: (1) NHD flowlines for streams and rivers and excluding other segments (i.e., canals), and (2) NHD waterbodies greater than one acre for lakes, ponds, and reservoirs and excluding other categories (i.e., wetlands). The extent of Narragansett Bay estuarine waters was defined by the Estuary Program (see this report's Introduction and Appendix). These data were also used to calculate total extents of fresh and estuarine waters by River Basin using an array of geospatial tools (Esri 2016).

The Estuary Program used NHD data at a high resolution (1:24,000) to best define a baseline of all the waters in the Watershed that have been delineated by the US Geological Survey (USGS). It is important to note that a portion of NHD waters may not meet states' criteria to be tested as part of the Integrated Reports. For example, the NHD data used by the Estuary Program do not differentiate intermittent versus perennial streams, a limitation that might be considered by the states.

As a reference for this chapter, Table 1 summarizes information about the extent of waters mapped in the high-resolution NHD, and total state-assessed waters (for all uses), as shown in Figure 1.

Table 1. Extent of state-assessed waters in Narragansett Bay and its Watershed. Total areas for National Hydrography Dataset (NHD) and Bay waters are included for reference.

Within Narragansett Bay and its Watershed	Estuarine Waters (sq. miles and percent)	Stream and Rivers (miles and percent)	Lakes and Ponds (acres and percent)
Total extent of all waters [1]	197	3,578	40,910
Total state-assessed waters	156	1,337 [2]	33,795
MassDEP	10	451 [2]	20,586
RIDEM	146	886 [2]	13,210
Percent of all waters that are state-assessed [3]	78 percent	37 percent	82 percent

[1] Based on National Hydrography Dataset (NHD); the area of the Bay as defined by the Estuary Program.
[2] An estimated four miles of MassDEP-assessed segments of Runnins River and Blackstone River were removed from analysis where the features both crossed outside of the state border and were duplicated in RIDEM GIS features.
[3] Total state-assessed divided by all waters in the Bay and the Watershed. These percentages represent the extent of waters in the Bay and in the Watershed that are assessed by the state agencies (MassDEP and RIDEM).

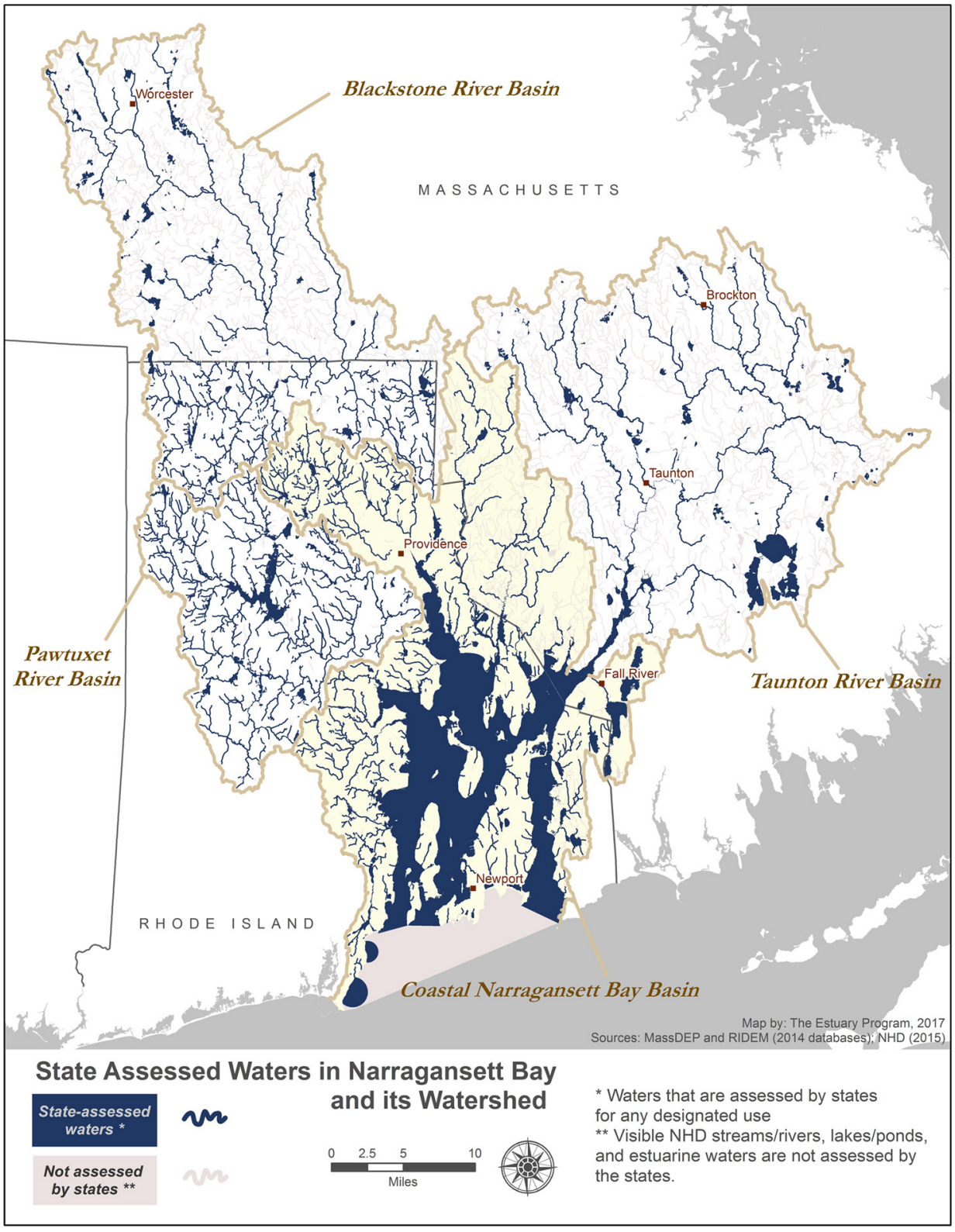


Figure 1. The extent of freshwaters and estuarine waters in Narragansett Bay and its Watershed, and the proportion of waters that have been assessed by MassDEP and RIDEM. Additional high-resolution National Hydrography Dataset (NHD) waters show all waters that shape the Bay and its Watershed.

Across the Watershed, states assess 37 percent of NHD stream miles, 82 percent of NHD ponds and lakes greater than one acre, and 78 percent of the Bay as defined by the Estuary Program (Table 1; Figure 1). While most of the estuarine waters are assessed by the states (79 percent), 43.5 square miles of waters in Narragansett Bay between Aquidneck Island and the Rhode Island Sound are not assessed at all (Figure 1). These metrics situate this analysis within the broader hydrological context of the Bay and Watershed. However, they are not presented as a target for state assessments, as states' have differing criteria and capacities for water quality assessment implementation.

A combined analysis of state water quality assessments in Massachusetts and Rhode Island must be informed by important differences between the states. A large part of Rhode Island (63 percent) falls within Narragansett Bay Watershed boundaries compared to only 13 percent of Massachusetts; however, Massachusetts comprises 60 percent of the Watershed area. The Pawtuxet River Basin is completely within Rhode Island, and the Taunton River Basin is completely within Massachusetts. Both the Coastal Narragansett Bay Basin and the Blackstone River Basin span the Rhode Island-Massachusetts border (Figure 1).

Additionally, Massachusetts has a much larger network of streams and rivers under its jurisdiction, and as a result Rhode Island's assessments reflect the capacity of its programs to analyze a greater number of streams and rivers per unit area compared to Massachusetts. Due to the larger size of Massachusetts, many water bodies are not yet assessed for recreational use within the Massachusetts portion of the Bay's watershed. Most of the assessed waters by MassDEP are at public bathing beaches for primary contact use. MassDEP assesses very few non-bathing waters for secondary use in the Watershed and in the estuarine waters of Mount Hope Bay (Figure 1).

EXTENT OF STATE-ASSESSED WATERS FOR RECREATIONAL USE

The Estuary Program calculated total extent of "state-assessed waters" using the states' 2014 Assessment Databases. Extents were also broken down by River Basin. The Estuary Program also calculated the subset of all "state-assessed" waters that are tested to determine water quality conditions for swimming and boating recreational uses (Figure 2). Waters tested for recreational use consist of either (1) "fully supporting," i.e., acceptable, for recreational use, or (2) "not supporting" recreational use due to impacts by one or more parameters. Many waterways and waterbodies are assessed for other

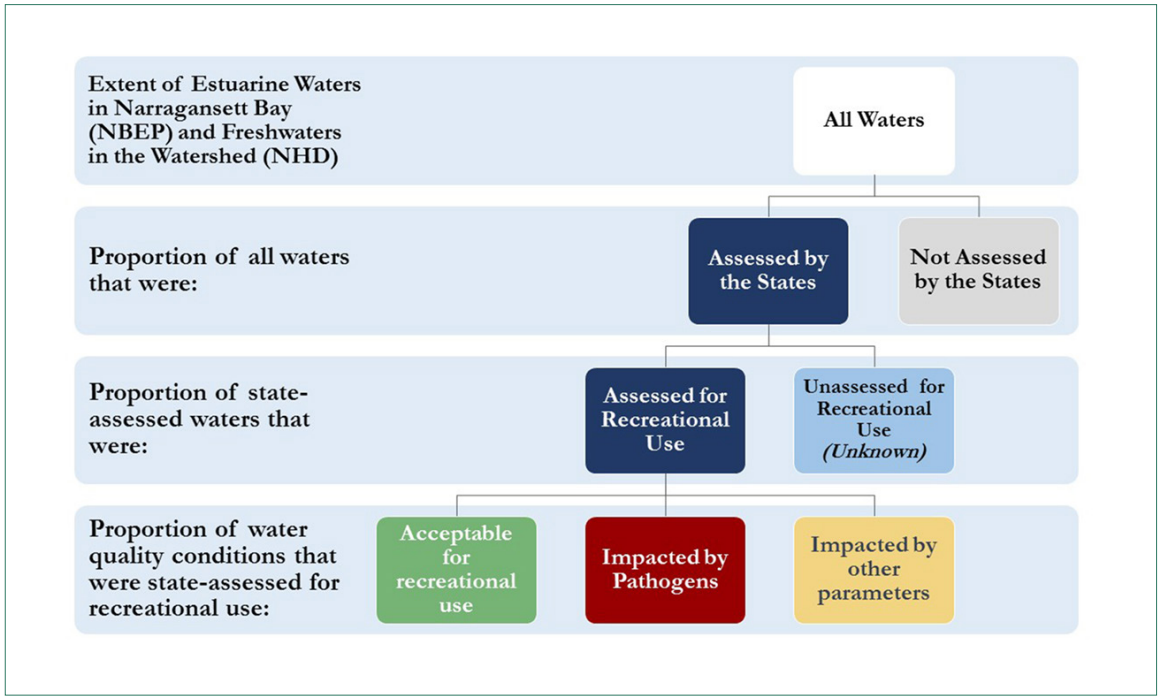


Figure 2. Conceptual diagram of the Estuary Program's methodology representing the breakdown of state-assessed waters for measuring water quality conditions for recreational use.

designated uses (i.e., aquatic wildlife and habitat; see "Water Quality Conditions for Aquatic Life" chapter) and not for recreational use. The Estuary Program defines these waters to be "not assessed" for recreational use although they are included in the total extent of "state-assessed" waters (Figure 2). The extent of state waters that are "not assessed" for recreational use was calculated to better understand the geographical data gaps in state assessments across the Watershed. However, the strategy to determine which waters should be tested and therefore assessed for recreational use may differ among states; waterbodies for which insufficient data or information are available regarding parameters that can affect primary and secondary contact recreational use, primarily pathogens, are not assessed and thus conditions are unknown.

WATER QUALITY CONDITIONS FOR RECREATIONAL USE

Within the portion of waters that states assess for recreational use, the states identify when water quality conditions are "fully supporting" or "not supporting" based on state water quality standards (Extended Methods, Tables 8 and 9). The states have multiple parameters, including pathogens, that can be used to assess waterbodies as primary or secondary contact recreational waters. When one or multiple causes of impairment are identified by the state agencies, the waterbody does not support recreational use and may be added to the List of Impaired Waters (CWA Section 303(d)) and prioritized for development of restoration plans known as Total Maximum Daily Loads (TMDLs) (RIDEM 2016). The state reports reflect assessments derived from data generated in multiple years (within a timeframe of five years). Not all waters are monitored within the same year, as both MassDEP and RIDEM conduct testing and assessments on a rotating-Basin basis. Dates for Basin assessments were not readily available, but the states can provide this information for later reports.

The Estuary Program's methods for reporting water quality conditions were designed in coordination with Rhode Island and Massachusetts state agencies to reconcile the differences between states water quality assessments. Rhode Island uses the same criteria for primary and secondary contact use assessments (Extended Methods, Table 8). In Massachusetts, most public bathing beaches are tested for primary contact use, while non-bathing waters are tested for secondary contact use, meaning that they are not designated for swimming but can be used for recreational boating. Massachusetts tests a larger variety of parameters in state assessments of water quality conditions for primary and secondary recreational use (Extended Methods, Table 9). The Estuary Program analyzed the extent of waters that are supporting of recreational use, or not supporting, focused on parameters related to pathogen impacts, because pathogens are a public health indicator assessed similarly by both states (Table 2).

For the purpose of this indicator, the Estuary Program combined the Massachusetts assessments of primary and secondary recreational use so that results could be comparable to Rhode Island. In most cases, Massachusetts determinations of primary and secondary contact recreation were the same. For the limited instances where attainments for primary and

Table 2. List of parameters that are assessed by MassDEP and RIDEM for primary and secondary contact recreation to determine whether waterbodies are impacted by pathogens.

Massachusetts	Rhode Island
A total of 29 parameters [1] are tested across the state waters for *Primary and Secondary Contact Recreation* as assessed by MassDEP	A total of three parameters[1] are tested across the state waters for *Primary and Secondary Contact Recreation* as assessed by RIDEM
For waterbodies that are impacted by pathogens, the Estuary Program used the following causes of impairment: ✓ Enterococci for fresh waters and estuarine waters ✓ Fecal coliform for fresh waters and estuarine waters ✓ *Escherichia coli* for fresh waters	

Source: MassDEP and RIDEM – database for the 2014 Integrated Reports.
[1] Not all parameters are tested on all waters.

secondary uses were different for the same stream segment or waterbody, the Estuary Program used the primary contact use determination from Massachusetts.

The Estuary Program defined categories of conditions that are comparable across states by relying on the determinations made by the states rather than raw data for bacteria counts, which could not be used given the inherent differences among the state programs and the timeframe when data were collected. From these determinations made by the states, the Estuary Program calculated the extent of state-assessed waters for recreational use as:

1. miles of streams and rivers,
2. acres of lakes and ponds, and
3. square miles of estuarine waters.

Using an array of geospatial tools (Esri 2016) to integrate and reconcile bi-state data, these metrics were calculated at the entire Narragansett Bay and its Watershed and for freshwaters within individual River Basins, and reported as the percentage of state-assessed waters that are acceptable, impacted by pathogens, and impacted by other parameters for recreational use.

- **Acceptable for Recreational Use:** Waters that fully support recreational use for swimming and boating. This is equivalent to a "fully supporting" determination by the states. These are waters that, because of good water quality conditions, can be targeted for further protection.
- **Impacted by Pathogens:** Waters that are "not supporting" of recreational use, due to impact by pathogens. These are waterbodies that, because of adverse pathogen concentrations (bacteria counts), can be targeted for restoration and can assist to identify potential sources of pathogen contamination (e.g., proximity to areas of high density of onsite systems; see "Wastewater Infrastructure" chapter). Note that this differs from the reporting by the states for impaired waters.

  While the focus of this chapter is on pathogens as a primary indicator of public health, the remaining extent of other waters that are "not supporting" for recreational use due to other parameters was also calculated to provide a complete picture of water quality impacts.
- **Impacted by Other Parameters:** Waters that are "not supporting" of recreational use, due to other parameters and not by pathogens. Rhode Island does not test for parameters other than pathogens when determining conditions for primary and secondary contact recreational use. Massachusetts tests up to 29 parameters across the state's waters in addition to pathogens (Table 2). Note that waters impacted by pathogens can also be impacted by other parameters.

In this chapter, the "not supporting" waters for recreational use are broken down into these two category conditions. While the condition-categories defined in this chapter to measure this indicator (acceptable for recreational use, impacted by pathogens, or impacted by other parameters) are derived from the state assessments, the results presented differ from the five reporting categories of assessed waters by the states for the Integrated List of Waters.[1] Accordingly, the results from this chapter and the state reports should not be compared.

The Estuary Program used the Assessment Database for 2014 from each state to extract, synthesize, compute, and analyze the status of each water quality condition. From the Assessment Databases, which were used to develop the Integrated Reports of 2015 (MassDEP 2015, RIDEM 2015), data were selected and synthesized by watershed, designated water use (primary and secondary recreational use), attainment (fully supporting and not supporting), and cause of impairment (pathogens and other parameters; see Table 2).

Status

THE EXTENT OF STATE-ASSESSED WATERS

The Estuary Program quantified that 95 percent (146 square miles out of 154 square miles) of state-assessed estuarine waters were assessed for recreational use, compared to 59 percent of state-assessed streams and rivers and 48 percent of state-assessed lakes and ponds (Figures 3 and 4).

Estuarine waters that were not assessed for recreational use because of insufficient data or no data at all, but were assessed by the states for other uses (e.g., aquatic life), include the Taunton River, Cole River, and Palmer River, all within Massachusetts (Figures 1 and 3).

[1] For example, a waterbody can be listed as Category 5, which means it was impaired for one or more uses. Waterbodies in Category 5 constitute the 303(d) list (MassDEP 2015, RIDEM 2015).

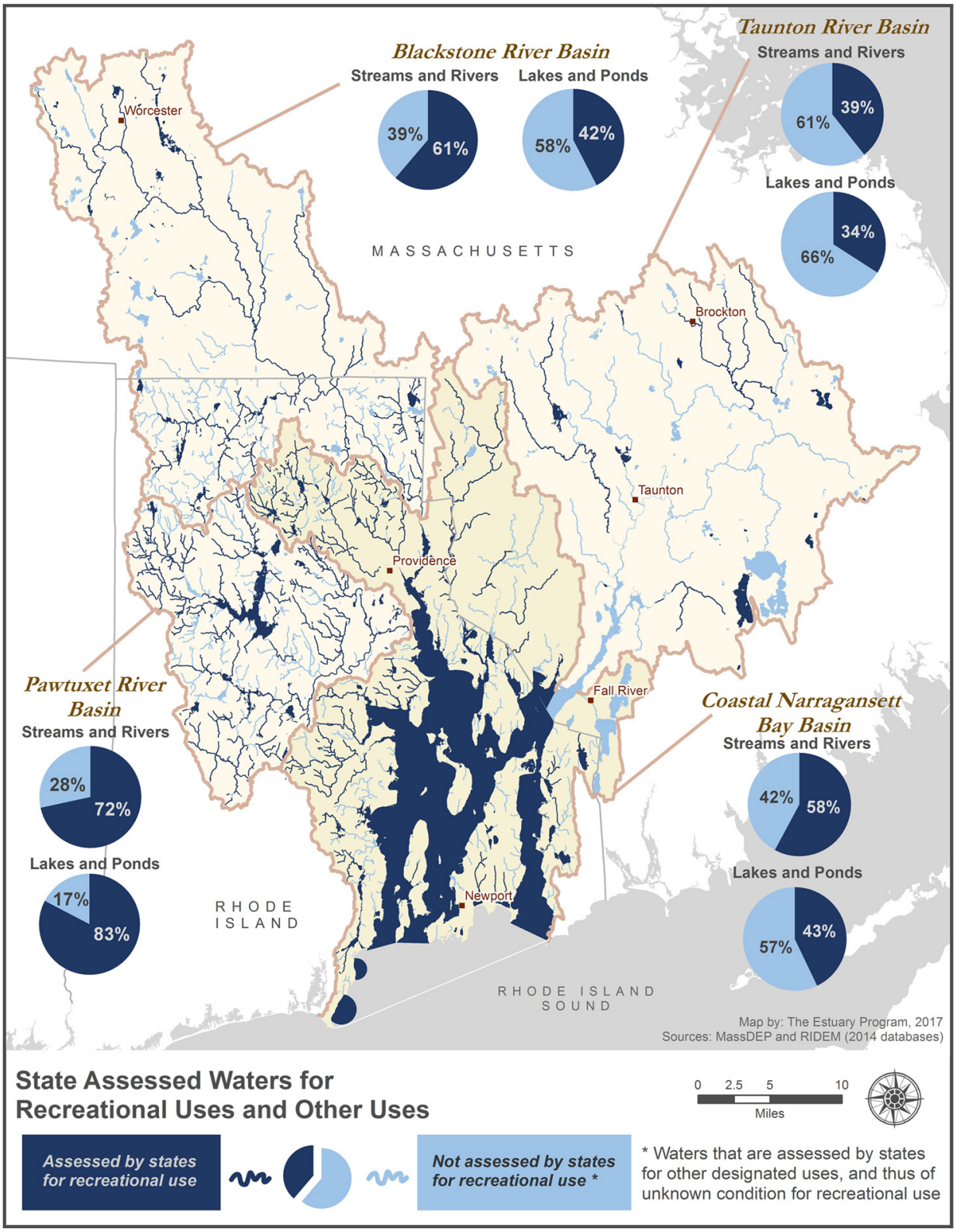


Figure 3. The extent and proportion of state-assessed waters that were either assessed or not assessed for recreational use in Narragansett Bay and its Watershed. For the waters assessed by the states for recreational use, the Estuary Program further calculated proportions with different water quality conditions (Table 5; Figure 5).

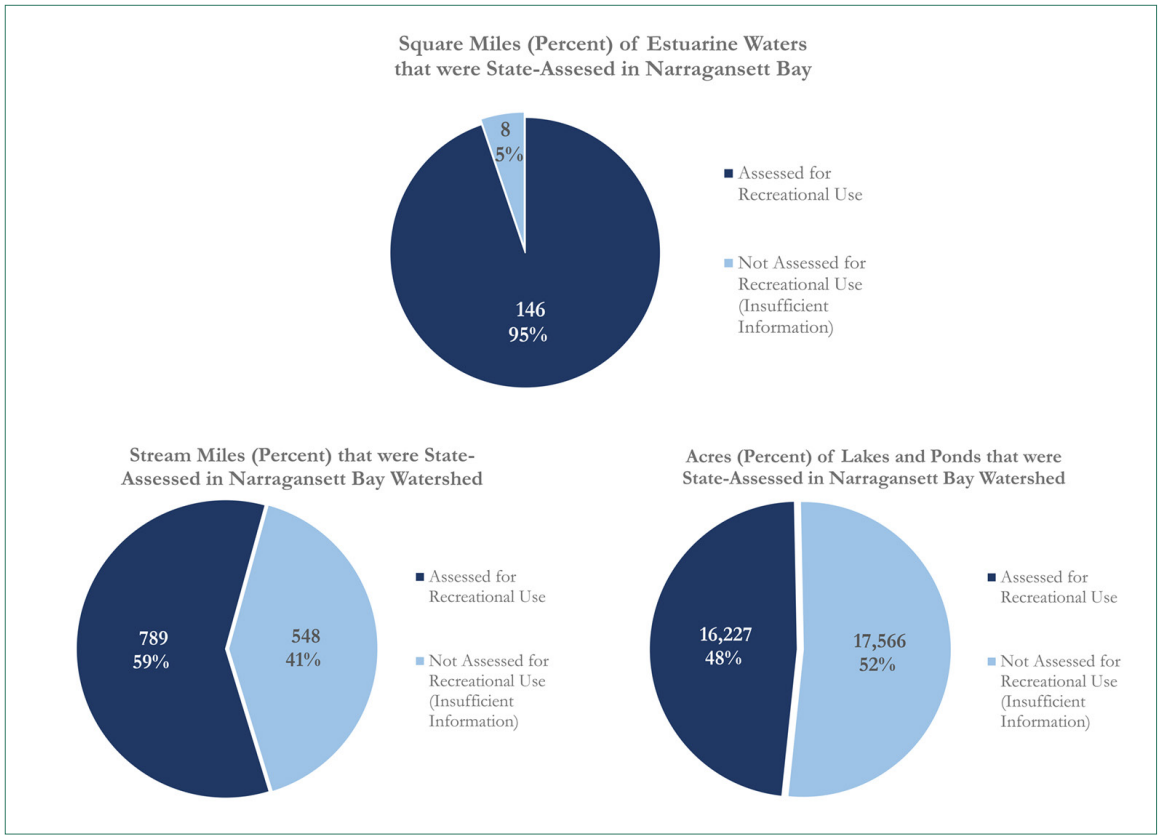


Figure 4. Proportion of state-assessed waters (not all waters) that were either assessed or not assessed (due to insufficient data) for recreational use in Narragansett Bay and its Watershed.

Table 3. Total (miles) of state-assessed streams and rivers that were either assessed or not assessed for recreational use in the Narragansett Bay Watershed's River Basins.

Streams and Rivers By River Basins	Blackstone River	Pawtuxet River	Coastal Narragansett Bay	Taunton River
Assessed for Recreational Use[1]	215	236	249	89
Unassessed for Recreational Use[2]	136	93.6	181.4	137.3
Total State-Assessed Waters	351	330	430	226

[1] This is the total extent of waterbodies that have been assessed for recreational use by the states, which is further broken down into water quality conditions.
[2] This is the total extent of waterbodies that have been assessed for other designated uses (e.g., aquatic life use) but have not been assessed by the states for recreational use, and thus conditions are unknown.

Table 4. Total (acres) of state-assessed lakes and ponds that were either assessed or not assessed for recreational use in the Narragansett Bay Watershed's River Basins.

Lakes and Ponds By River Basins	Blackstone River	Pawtuxet River	Coastal Narragansett Bay	Taunton River
Assessed for Recreational Use[1]	3,322	5,701	3,432	3,772
Unassessed for Recreational Use[2]	4,511	1,196	4,569	7,290
Total State-Assessed Waters	7,833	6,897	8,003	11,062

[1] This is the total extent of waterbodies that have been assessed for recreational use by the states, which is further broken down into water quality conditions.
[2] This is the total extent of waterbodies that have been assessed for other designated uses (e.g., aquatic life use) but have not been assessed by the states for recreational use, and thus conditions are unknown.

For the total state-assessed waters in each River Basin, Tables 3 and 4 shows the total extent of fresh waters that were assessed or unassessed for recreational use (Figure 3).

The Pawtuxet River Basin had the highest percentage of total state-assessed waters that were assessed for recreational use (72 percent of streams and rivers, and 83 percent of lakes and ponds). This is likely reflective of Rhode Island's capacity as a smaller state to assess more waterbodies per unit area, as the Pawtuxet River Basin is the only Basin completely within Rhode Island (Tables 3 and 4; Figure 3). A majority (61 percent) of streams and rivers and 66 percent of lakes and ponds in the Taunton River Basin remained not assessed for recreational use. In the Coastal Narragansett Bay Basin and Blackstone River Basin, nearly half of streams and rivers (42 percent and 39 percent, respectively) (Table 3) and greater than half of ponds and lakes (57 percent and 58 percent, respectively) remained not assessed (Table 4).

WATER QUALITY CONDITIONS FOR RECREATIONAL USE

Narragansett Bay and its Watershed

Of the 146 square miles of estuarine waters assessed by the states for recreational use (Figures 3 and 4), the large majority of Narragansett Bay's estuarine waters were identified as acceptable for recreational use (89 percent) (Table 5; Figure 5). Areas characterized as impacted by pathogens (11 percent) included the upper portion of the Providence River Estuary, parts of Mount Hope Bay, and some areas with high-density development (e.g., Newport, Barrington/Warren) (Figure 5).

Of the 789 stream miles that are assessed by the states for recreational use (Figure 4), more were identified as impacted by pathogens (61 percent) than acceptable for recreational use (34 percent) (Table 5; Figure 5). Parameters other than pathogens, including nuisance aquatic vegetation, are also impacting recreational use along the upper reaches of the Watershed in some tributaries of the Blackstone River, in Massachusetts (Figure 5).

River Basins

The Pawtuxet River Basin had the highest percentage (of freshwaters assessed by the states for recreational use within this Basin) of streams and rivers that were suitable for swimming and boating (67 percent), and the lowest percentage of streams and rivers impacted by pathogens (33 percent), compared with the other River Basins. Meanwhile, the Taunton River Basin was characterized by the lowest percentage of waters identified as acceptable for recreational use such as swimming and boating (18 percent) and the highest percentage of pathogen-impacted streams (82 percent), followed by the Coastal Narragansett Bay Basin (80 percent impacted) (Table 6; Figure 5).

Of the 16,227 acres of ponds and lakes that are assessed by the states for recreational use (Figure 4), most ponds and lakes (78 percent) were acceptable for recreational use, and very few were impacted by pathogens (2 percent) (Table 7). The Pawtuxet River Basin supported the greatest extent of acceptable ponds and lakes, amounting to nearly all waterbodies assessed for recreational use (96 percent) (Table 7). The Blackstone River Basin supported the least relative extent of ponds and lakes identified as acceptable for recreation (50 percent). Notably, in Massachusetts where additional parameters are assessed, between 40 and 50 percent of the waters

Table 5. Extent and percent of estuarine waters, streams and rivers, and lakes and ponds in Narragansett Bay and its Watershed by water quality conditions for recreational use.

Water Quality Conditions in Narragansett Bay and its Watershed	Estuarine Waters Square miles (Percent)	Stream and Rivers Miles (Percent)	Lakes and Ponds Acres (Percent)
Total state-assessed for recreational use	*146*	*789*	*16,227*
Acceptable for recreational use	130.4 (89)	272 (34)	12,733 (78)
Impacted by pathogens	15.4 (11)	481(61)	333 (2)
Impacted by other parameters[1]	0	36 (5)	3,161.4 (19)

[1]Waters identified as impacted for recreational use for parameters other than pathogens.

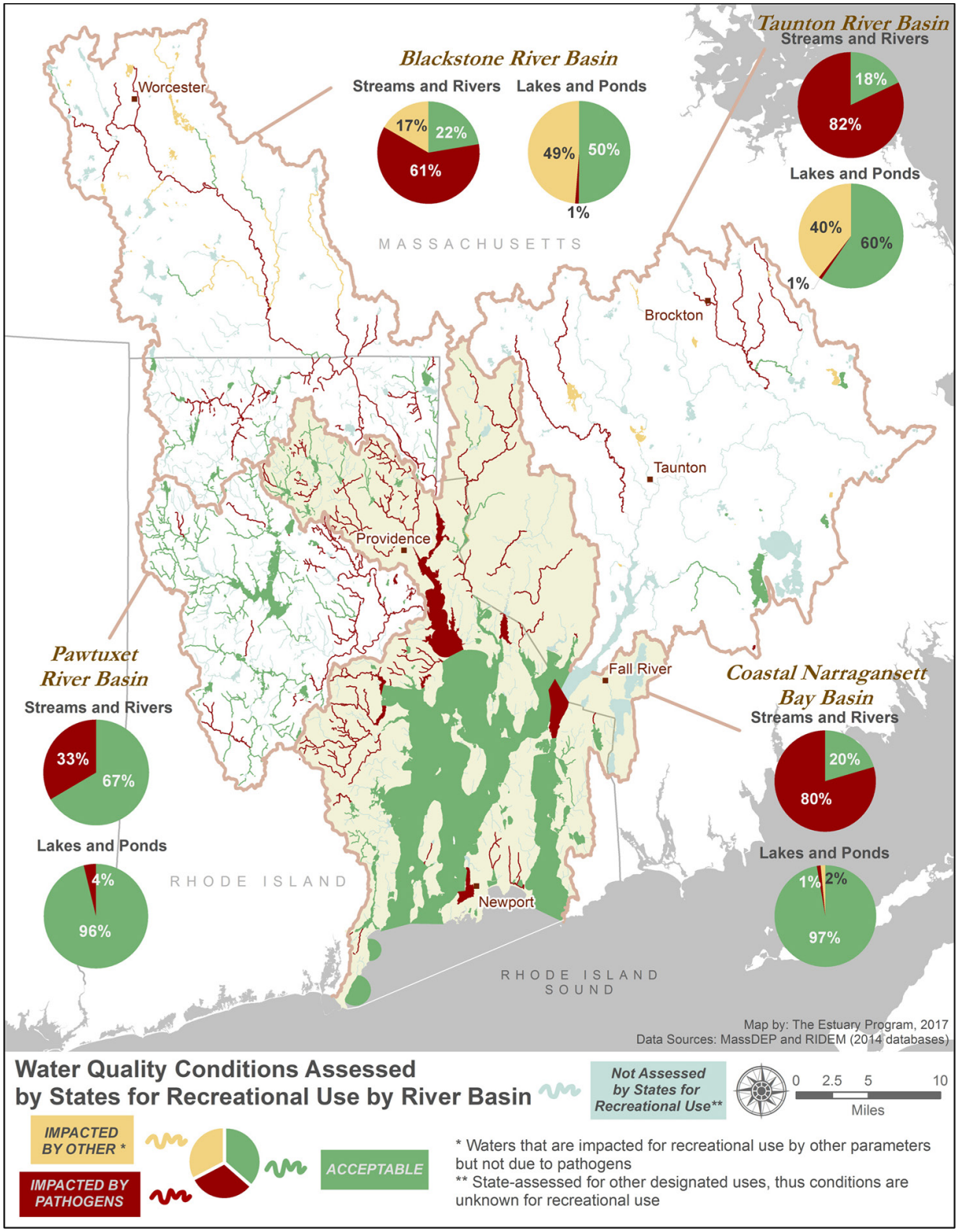


Figure 5. The extent and proportion of water quality conditions for recreational use that were assessed by the states in the Narragansett Bay Watershed's River Basins. For each River Basin, pie charts show the percentage of fresh waters in each category-condition for recreational use (acceptable, impacted by pathogens, or impacted by other parameters).

Table 6. Total (miles) and percent of streams and rivers in the Narragansett Bay Watershed's River Basins by water quality conditions for recreational use.

Streams and Rivers (Miles) by Basins	Blackstone River	Pawtuxet River	Coastal Narragansett Bay	Taunton River
Total state-assessed streams and rivers for recreational use[1]	*351*	*330*	*430*	*226*
Acceptable for recreational use	48 (22 percent)	157 (67 percent)	51 (20 percent)	16 (18 percent)
Impacted by pathogens[2]	131 (61 percent)	79 (33 percent)	198 (80 percent)	73 (82 percent)
Impacted by other parameters[1]	36 (17 percent)	0	0	0

Table 7. Total (acres) and percent of lakes and ponds in the Narragansett Bay Watershed's River Basins by water quality conditions for recreational use.

Lakes and Ponds (Acres) by Basins	Blackstone River	Pawtuxet River	Coastal Narragansett Bay	Taunton River
Total state-assessed waterbodies for recreational use [1]	*3,322*	*5,701*	*3,432*	*3,772*
Acceptable for recreational use	1,665 (50 percent)	5,484 (96 percent)	3,339 (97percent)	2,245 (60 percent)
Impacted by pathogens	38 (1 percent)	217 (4 percent)	42 (1 percent)	36 (1 percent)
Impacted by other parameters[1]	1,619 (49 percent)	0	51 (2 percent)	1,491 (40 percent)

[1] Waters identified as impacted for recreational use for parameters other than pathogens.

assessed for recreational use in the Taunton River and Blackstone River Basins are impacted for parameters other than pathogens (Figure 5).

Discussion

There are sufficient data to assess water quality conditions for recreation in most of Narragansett Bay and a significant portion of its Watershed. Overall, estuarine waters in Narragansett Bay largely support recreational uses, with 89 percent of assessed waters found to be acceptable for swimming and boating. The Providence River Estuary, the Palmer River, several coves within Greenwich Bay, and Newport Harbor, all in Rhode Island, are all areas that were exceptions to good water quality and were found to be impacted by pathogens for swimming and other recreational uses (Figure 4). Additionally, water quality conditions for recreational use remain unknown in the northeastern portions of Mount Hope Bay near Fall River and the Taunton River estuary, where waterbodies have not been assessed by Massachusetts, due to insufficient information to determine whether waters are suitable for swimming and/or boating. However, the southeastern portion of Mount Hope Bay, located in Rhode Island, was identified as being impacted by pathogens.

The heavily urbanized sections of Narragansett Bay have historically been affected by pathogens from the CSOs in Fall River and the greater Providence area. The City of Fall River and the Narragansett Bay Commission (NBC), serving the Providence region, have both invested significantly in CSO abatement during the last fifteen or more years. NBC has completed two phases of improvements including construction of a tunnel built in Providence that stores 65 million gallons of stormwater and untreated wastewater. Fall River also built a tunnel that can store 38 million gallons. Untreated wastewater and stormwater are diverted to these tunnels during large rain events and later pumped to their respective treatment plants. As a result, the volume of combined sewage that flows untreated into Narragansett Bay has been significantly reduced. NBC is continuing with a third phase of improvements planned over the next two decades. It is anticipated that pathogen levels in these urbanized areas of the Bay will decrease. The water quality improvements achieved in the Upper Bay region have spurred the City of East Providence and partners to explore establishing a public beach at Sabin Point Park adjacent to the Providence River.

The Pawtuxet River Basin supported the largest extent of fresh waters found to be acceptable for swimming and boating. Excluding its mainstem, this Basin is mostly vegetated and sparsely developed, which are characteristics that typically support low pathogen loading (Didonato et al. 2009, Crim et al. 2012; see "Impervious Cover" and "Land Use" chapters). Vegetation and infiltration control runoff into receiving waters, filtering pathogens from localized sources, such as failing septic systems, pet wastes, and wildlife. The efforts to protect lands (see "Open Space" chapter) around surface waters as natural buffers can be evidenced in the results for the Pawtuxet River Basin.

Conversely, the coastal areas around the Bay—which comprise the Taunton River Basin and Coastal Narragansett Bay Basin—contained the highest relative proportion of impacted rivers and streams (Table 5). This is likely because dense urban development and impervious cover generate higher volumes of stormwater that carries pathogens into receiving waters (see "Impervious Cover" chapter). Generally, few ponds and lakes were impacted by pathogens across all River Basins (Table 6). However, the large percentages of ponds and lakes remaining not assessed for recreational use (52 percent) may have skewed this result (Table 3; Figure 2).

It is noteworthy to mention that while the Estuary Program used the database for the 2014 state assessments, the assessments may not reflect data available after 2013. When the states present assessments results in the Integrated Reports, the data and time period being assessed lag behind the year of the assessment cycle. Because it takes more than a year to do the assessments, the states generally use data from the five prior years. For the Rhode Island's 2014 water quality assessments, the data reviewed were primarily from 2008 to 2013.

Recreational use in Narragansett Bay and surrounding watersheds depends upon good water quality. Climate change will likely pose increasing risks to public health by amplifying pathogen transport to Narragansett Bay and its Watershed via more frequent and intense storms and increased water temperatures (see "Precipitation" and "Temperature" chapters). It is important to the protection of public health to ensure monitoring efforts are both sustained and expanded to support a comprehensive assessment. To date, the large proportion of state-assessed waters remaining not assessed for recreational use reflects an important data gap. Identifying impairment triggers management responses

that may include implementation of pollution control actions and the development of TMDLs. These management responses can be used to reduce pressures on polluted waters, allowing natural systems to recover and build resilience. Above all, reducing public health risks from pathogens and promoting the economic and cultural benefits arising from recreational uses of the Bay by residents and visitors alike is a primary objective of the states to protect water quality for human use, and for the long-term management of Narragansett Bay and its Watershed.

Data Gaps and Research Needs

- Data gaps exist with respect to assessing the recreational use of waters in the Taunton River and Blackstone River Basins in Massachusetts and the Coastal Narragansett Bay Basin in Rhode Island. Monitoring efforts need to be expanded to address these gaps.
- Additional research into the fate and transport of pathogens discharged into the ground from onsite wastewater systems is a need. Research should focus on those subwatersheds or drainage areas in which onsite wastewater treatment systems, including cesspools, are known to or suspected of contributing to pathogen pollution problems.

Acknowledgments

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Extended Methods

Numeric criteria used by the state agencies to assess water quality for recreational use:

Table 8. RIDEM's numeric criteria for fully supporting recreational/swimming of non-designated bathing beach waters.

Numerical Criteria	Freshwaters	Estuarine Waters
Fecal coliform[1]	The geometric mean of 200 MPN/100 ml is met and not more than 10 percent of the total samples taken exceed 400 MPN/100 ml	The geometric mean of 50 MPN/100 ml is met and not more than 10 percent of the total samples taken exceed 400 MPN/100 ml
Enterococci	The geometric mean of 54 colonies per 100 ml is met.	The geometric mean 35 colonies per 100 ml for salt waters is met.
Escherichia coli	The geometric mean of 126 colonies per 100 ml is met.	

Source: RIDEM 2014

[1] Most Probable Number (MPN) of coliform per 100 ml.

Table 9. MassDEP's numeric criteria for recreational/swimming during the bathing and non- bathing season for waters at public beaches for primary contact use or waters designated as secondary contact use.

Numerical Criteria	Freshwaters	Estuarine Waters
Enterococci for Primary Contact Recreational Use		
Public bathing beaches	The geometric mean of the five most recent samples taken during the same bathing season shall not exceed 33 colonies/100 ml and no single Enterococci sample taken during the bathing season shall exceed 61 colonies/100 ml.	No single Enterococci sample taken during the bathing season shall exceed 104 colonies/100 ml and the geometric mean of the five most recent Enterococci samples taken within the same bathing season shall not exceed 35 colonies/100 ml.
During the non-bathing season, for waters at public bathing beaches	The geometric mean of all Enterococci samples taken within the most recent six months shall not exceed 33 colonies/100 ml, typically based on a minimum of five samples, and no single sample shall exceed 61 colonies/100 ml.(1)	No single Enterococci sample shall exceed 104 colonies/100 ml and the geometric mean of all samples taken within the most recent six months, typically a minimum of five samples, shall not exceed 35 colonies/100 ml.(1)
Enterococci for Secondary Contact Use		
		The geometric mean of all Enterococci samples taken within the most recent six months shall not exceed 175 colonies/100 ml, typically based on the five most recent samples, and 10 percent of such samples shall not exceed 350 colonies/100 ml.
Escherichia coli for Primary Contact Recreational Use		
Public bathing beaches as defined by MA DPH	The geometric mean of the five most recent *E. coli* samples taken within during the same bathing season shall not exceed 126 colonies/100 ml and no single sample taken during the bathing season shall exceed 235 colonies/100 ml (1)	
During the non-bathing season, for waters at public bathing beaches	The geometric mean of all *E. coli* samples taken within the most recent six months shall not exceed 126 colonies/100 ml, typically based on a minimum of five samples, and no single sample shall exceed 235 colonies/100 ml.(1)	
Escherichia coli for Secondary Contact Recreational Use		
	The geometric mean of all *E. coli* samples taken within the most recent six months shall not exceed 630 *E. coli*/100 ml, typically based on a minimum of five samples, and 10 percent of such samples shall not exceed 1260 *E. coli*/100 ml.(1)	

Source: MassDEP 2012

(1) This criterion may be applied on a seasonal basis at the discretion of the Massachusetts Department of Public Health.



State of Narragansett Bay and Its Watershed
2017 Technical Report

Condition Indicators of Public Health

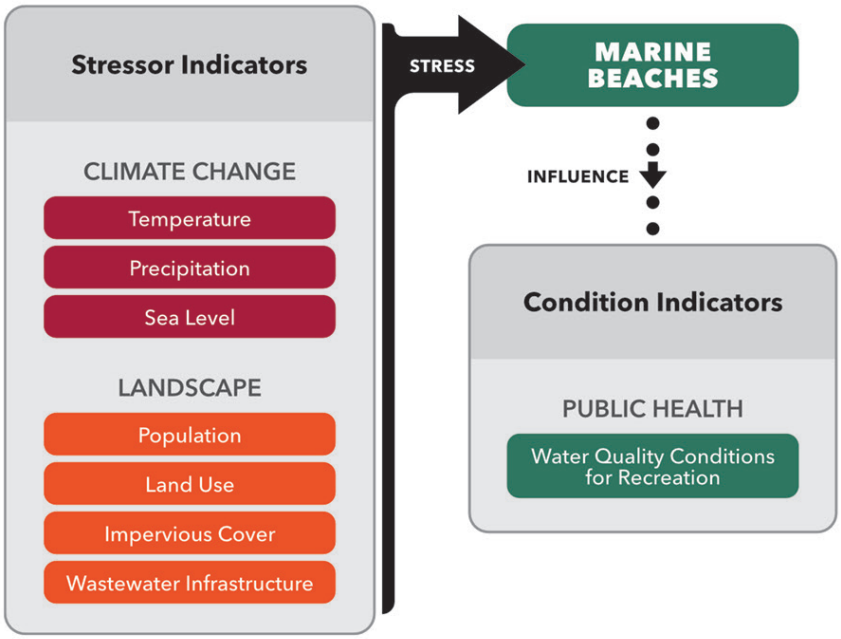
CHAPTER 23: MARINE BEACHES

Please use the following citation: Narragansett Bay Estuary Program. 2017. State of Narragansett Bay and Its Watershed (Chapter 23, Marine Beaches, pages 437-460). Technical Report. Providence, RI.

Photo: North Kingstown Town Beach, RI (Ayla Fox)

NARRAGANSETT BAY ESTUARY PROGRAM

Overview



BACKGROUND

- Marine beaches are an important part of Narragansett Bay's recreational appeal, and beach closures caused by poor water quality reduce the quality of life for residents and visitors alike. Beach closures occur when water tests show high counts of bacteria that indicate contamination from wastewater and/or stormwater. Growth of the human population and changes in land use, especially more impervious cover, can lead to declines in water quality and increases in beach closures. Climate change stressors including increased rainfall, warmer water temperatures, and sea level rise are likely to exacerbate conditions leading to beach closures.
- This indicator, developed in collaboration with state health departments, classified the 37 marine beaches in Narragansett Bay as High Concern or Low Concern, referring to beaches at high or low risk for pathogen contamination, based on the level of use, monitoring frequency, and history of closure rates.

KEY FINDINGS

- ***Status***

  High Concern Beaches: In 2015, there were 38 beach closure events at the fourteen High Concern beaches, and 42 percent of the events occurred in the Upper Estuary.

Low Concern Beaches: In 2015, there were six beach closure events at the 23 Low Concern beaches, and 48 percent of the events occurred in the Upper Estuary.

- ***Trends***

High Concern Beaches: From 2000 to 2015, the High Concern marine beaches in the Upper Estuary had the highest percentage of closure events (over 40 percent) compared with the other Bay Regions. Prior to 2009, closure events at High Concern beaches intensified during wet seasons, as expected. After 2009, however, precipitation did not strongly correlate with the number of closure events. In addition, there is not a clear upward or downward trend of beach closures over the sixteen years of beach monitoring.

Low Concern Beaches: While spikes of beach closure events across the Bay seem to have been linked to wet seasons, there was no pattern at specific beaches or in Regions of the Bay. From 2000 to 2015, the Low Concern marine beaches in the East Passage had a higher percentage of closure events (42 percent) compared with the other Bay Regions.

Introduction

Marine beaches provide significant economic, cultural, recreational, and aesthetic value. Beach waters are susceptible to contamination with harmful microorganisms that can cause health impacts such as gastroenteritis and sore throats, or even meningitis or encephalitis (Cabelli 1983, USEPA 1986, Haile 1996, Pruss 1998). State departments of health, supported by the federal Beaches Environmental Assessment and Coastal Health Act of 2000 (BEACH Act), conduct microbiological monitoring. The goal of the BEACH Act, administered by the United States Environmental Protection Agency (USEPA), is to reduce risks of illness in coastal waters and the Great Lakes by improving beach testing and availability of information to the public. USEPA annually awards grants to eligible states, territories, and Tribal nations to develop and implement beach water quality monitoring and notification programs for recreational beaches.

As most microbiological pathogens are difficult to measure directly, the fecal indicator bacteria Enterococci (typically found in the feces of warm-blooded animals and humans) serve as a proxy for pathogens in beach water monitoring. In Narragansett Bay, the Rhode Island Department of Health (RIDOH) and the Massachusetts Department of Public Health (MDPH), with BEACH Act support, monitor 37 public marine beaches for Enterococci.

Sources of microbial pathogens include discharges of raw sewage from combined sewer overflows (CSOs), failing septic systems, cesspools, and wild and domestic animals. High bacterial counts are driven by watershed conditions at local and regional scales. Precipitation and impervious cover contribute to the delivery of wastewater pathogens via stormwater runoff and/or groundwater directly into Narragansett Bay and tributaries. Changes in land use have been shown to influence the number of beach closures; urbanization near beaches can negatively affect beach microbial water quality, whereas natural lands such as forests and wetlands may reduce the number of beach closures (Wu and Jackson 2016).

Increasingly, pathogenic loads are being reduced through management practices. Engineered retention systems, green infrastructure, pet waste management, and upgrades to CSO facilities have been implemented in the Narragansett Bay Watershed. CSO abatement programs are being implemented in the two largest cities in the Bay to increase storage capacity of both sewage and rainwater from stormwater runoff during heavy rain events, with a holding capacity of 65 million gallons in Providence, Rhode Island, and 38 million gallons in Fall River, Massachusetts.

The Narragansett Bay Commission (NBC) presented results of the CSO project's phase I improvements, from pre-phase I (2004 to October 2008) through

post-phase I (November 2008 to 2014), showing that 50 percent of the CSO volume was captured and treated annually and that there was between 42 and 51 percent reduction of fecal coliform bacteria loads, post-improvements (NBC, presentation by Pamela Reitsma 2015, NBC 2016), discharging to the Lower Providence and Upper Providence River (see map of Narragansett Bay Segments and Sections in the Appendix). However, as of 2015, Enterococci results post-phase I showed that standards had not met the primary contact criteria for designated beaches in the Upper Providence River. In addition, no effects had been shown in the Lower Providence River, since the Enterococci bacteria standards were met an nearly equal number of years, pre- and post-phase I. Although CSO project improvements had a stronger effect on fecal coliform levels than Enterococci bacteria in the Providence River Estuary, reducing CSO discharges into receiving waters is likely to have had at least some impact on reducing pathogen loading at urban beaches in the northern sections of the Bay.

Based on observations of positive changes in the upper Providence River Estuary, the RIDOH launched the Urban Beach Initiative in 2010 to investigate the possibility of re-opening Sabin Point, Rosa Larisa, and Gaspee Point beaches to swimming and other recreational uses. These beaches have been subject to long-term closure due to nearby sources of pathogens and high counts of Enterococci.

For this chapter, the Narragansett Bay Estuary Program and partners analyzed the status (as of 2015) and trends (2000 to 2015) in beach closure events at licensed marine beaches categorized as High or Low Concern beaches. High Concern beaches are more frequently monitored than Low Concern beaches because of the higher likelihood of pathogen contamination. This indicator measures summer beach closure events at the Bay Region scale, as opposed to the local (individual beach) scale, to compare Bay Regions that have been adversely impacted by pathogens. Using the total number of beach closure events and a scale of zero to 100 percent (normalized closure events by Bay Region), the Estuary Program compared the percent of closures between the Bay Regions. In addition, the Estuary Program plotted total seasonal rainfall against total beach closure events in Narragansett Bay and in Bay Regions to explore the effects of rainfall on beach closures.

Methods

The Narragansett Bay Estuary Program collaborated with RIDOH and MDPH to examine beach closure days (Rhode Island) and postings (Massachusetts), sampling locations, and level of concern that dictates frequency of monitoring for all the licensed public marine beaches within Narragansett Bay. The water quality data for public marine beaches in Rhode Island included results from 2000 through 2003 (tested for *Escherichia coli*) and 2004 through 2015 (tested for Enterococci). Data for public marine beaches in Massachusetts included results from 2000 through 2015 (tested for Enterococci).

To analyze how marine beaches have been indicative of water quality for human uses in the Bay, the Estuary Program: (1) standardized and reconciled beach closure data from Massachusetts and Rhode Island for consistency across the Bay; (2) grouped all licensed public marine beaches in Narragansett Bay into two defined beach categories, High Concern and Low Concern, based on current monitoring frequency and history of closure events; and (3) conducted geospatial analysis for the five Regions of the Bay to summarize and calculate:

- total of beach closure events at High Concern beaches and
- total of beach closure events at Low Concern beaches.

The Estuary Program examined the percentage of total beach closure events in Narragansett Bay across the five Bay Regions (Upper Estuary, East Passage, West Passage, Sakonnet Rover, and Mouth of the Bay) in order to compare changes over time among and within Bay Regions.

The analysis conducted for this chapter did not focus on changes in closures of individual public beaches primarily because preliminary analysis by the Estuary Program and partners have shown that those changes are more likely due to localized impacts. The analysis also did not focus on changes at the scale of the entire Bay because changes in water quality at marine beaches, based on beach closure data, are not representative of the Bay as a whole. Rather, the analysis quantified the percentage of beach closure events in Regions of the Bay because the results can provide a better understanding of watershed-level effects on water quality for human use of marine beaches.

WATER QUALITY MONITORING AT PUBLIC MARINE BEACHES

Water quality at public marine beaches is sampled by the state departments of health during the summer season (Memorial Day through Labor Day) and analyzed using either the Modified Enterococci Method 1600 plate method or Enterolert®, a defined substrate method, to estimate counts of viable Enterococci. For beaches in Rhode Island, the single sample standard is 60 cfu per 100 ml (colony forming units per 100 milliliters) of salt water. Prior to 2015, however, the standard was 104 cfu per 100 ml for all marine beaches (RIDOH 2016). In Massachusetts, the state health department has adopted and continues to use the standard for Enterococci in marine waters at 104 cfu per 100 ml for a single sample and 35 cfu per 100 ml for the geometric mean, which is calculated based on the last five non-rain-impacted samples over a 30-day period (MDPH 2016). At the majority of Massachusetts beaches, water quality is considered unsafe for swimming when two samples collected on consecutive days exceed the water quality standard. Beaches with a history of multi-day elevated bacteria levels are still required to post warnings after a single exceedance. For Rhode Island beaches, exceeding the standard is a trigger for beach closure consideration. Beach closures in Rhode Island take additional factors into account, including history of contamination, precipitation, flushing rates, and any additional evidence of contamination (i.e., water quality data from stormwater outfalls).

There are limitations in the assessment of water quality at marine beaches. Management actions to close beaches are often delayed due to the 24 hours of laboratory analysis needed for the approved analytical methods used to measure Enterococci bacteria. This delay means that closures are asynchronous with adverse conditions. The conditions at many beaches change significantly in a single tidal cycle, often making the bacterial count obsolete before results are available. New technical solutions are being tested and faster methods may be available soon. For instance, Rhode Island is investigating a qPCR (quantitative polymerase chain reaction) method that amplifies and measures fecal indicator bacterial DNA in water samples. This method could reduce the time between sampling and the availability of results to as little as six hours. The results from this investigation will provide information on the acceptability of this method and the inherent constraints, such as costs and logistics.

BEACH CLOSURE EVENTS

The Estuary Program standardized the beach closure data from both Massachusetts and Rhode Island into "beach closure events," for public marine beaches within Narragansett Bay (Figure 1). The method for standardizing beach closure events was defined by the Estuary Program in coordination with partners at the state departments of health. Regardless of the duration of a beach closure (e.g., one day or consecutive days in one week), beach closure events were considered equal for the purposes of this analysis. This was necessary given that beach closure data from both states are not consistent in the way the departments of health track and report beach closures. RIDOH quantifies the number of days, whereas MDPH quantifies the number of postings, which can be one day or multiple consecutive days (MDPH 2016, RIDOH 2016).

Length of closure is often dependent on logistical factors related to sampling and lab analysis. It is of note that closure events most likely co-vary to some extent with the frequency of sampling at a given beach location. Thus, a closure event was defined as follows: (1) One beach may have been closed for one day, and another beach for a week, but each case was attributed as a single event. (2) If a beach re-opened one day after closure, and then closed again three days later, the second closing, no matter how many days, was referred to as a separate closure event. (3) While some public beaches are monitored more frequently than others according to state-assigned tiers relating to degree of risk for contamination (Table 1), if a closure is posted, water sampling will continue regardless of the tier until bacterial concentration meets the department's criteria for swimming, based on each state's department of health protocols.

HIGH AND LOW CONCERN BEACHES

Direct comparison of available Massachusetts and Rhode Island beach data poses a challenge because monitoring frequency is variable across beaches and between the states. Monitoring frequency may be driven by multiple objectives and is generally greater for the most at-risk locations. For this analysis encompassing sixteen years of data, normalization to sampling frequency was neither practical nor supportable.

To reconcile beaches in both states within a unified context of relative health concerns, the Estuary Program used a classification system derived

Table 1. Estuary Program classification for marine beaches analogous to tier classification by MDPH and RIDOH.

	Rhode Island		Massachusetts	
State Tier	Monitoring Frequency*	Estuary Program Classification	Monitoring Frequency**	Estuary Program Classification
1	Twice per week	High Concern	More than once per week***	High Concern
2	Twice per month	Low Concern	Once per week	High or Low Concern, determined by historical pattern of beach closures
3	Once per month	Low Concern	Every two weeks or less often	Low Concern

* RIDOH Tier Classification as of 2015
** MDPH Tier Classification as of 2015
*** There were no marine beaches in Narragansett Bay classified as Tier 1 using MDPH criteria.

Table 2. High and Low Concern marine (licensed) beaches by Bay Region in Narragansett Bay. Italics indicate Massachusetts marine beaches.

Bay Region	High Concern	Low Concern
Upper Estuary	BARRINGTON TOWN BEACH BRISTOL TOWN BEACH CITY PARK BEACH CONIMICUT POINT BEACH GODDARD MEMORIAL STATE PARK OAKLAND BEACH *PIERCE BEACH* WARREN TOWN BEACH	*CEDAR COVE* *COLES RIVER CLUB* *LEESIDE* *SANDY BEACH* *SWANSEA TOWN BEACH*
Mouth of the Bay	ATLANTIC BEACH CLUB BEACH EASTON'S BEACH SCARBOROUGH NORTH SCARBOROUGH SOUTH	CAMP GROSVENOR DUNES CLUB GOOSEBERRY BEACH HAZARDS BEACH NARRAGANSETT TOWN BEACH SACHUEST BEACH SPOUTING ROCK BEACH ASSOCIATION
Sakonnet River	PEABODYS BEACH THIRD BEACH	FOGLAND BEACH GRINELLS BEACH SANDY POINT BEACH
East Passage		CAMP ST. DOROTHY KING PARK SWIM AREA FORT ADAMS STATE PARK MACKEREL COVE BEACH
West Passage		BONNET SHORES BEACH CLUB NORTH KINGSTOWN TOWN BEACH PLUM BEACH CLUB SAUNDERSTOWN YACHT CLUB

from current Rhode Island and Massachusetts tier classifications. Both states have three tiers of beach classifications with Tier 1 being the highest concern and Tier 3 lowest, although they use different criteria for classification (Table 1). From those rankings, the Estuary Program consolidated all public marine beaches within Narragansett Bay into two groups–High Concern and Low Concern–based on 2015 monitoring frequency as a proxy for degree of risk and an analysis of beach closure history (Table 1).

Tier designation is the primary factor governing frequency of sampling; however, each state has made occasional changes in risk-based tier assignments. In Massachusetts, tier classification by beach was available for each year between 2000 and 2015, while tier classifications for each year were not readily available for each beach in Rhode Island. Thus, one caveat for this analysis is that the frequency of monitoring at any given beach may have changed over time, limiting the comparability of beach closure within beach categories over time. In general, however, the frequency of routine sampling has been consistent at each beach from one year to the next, making comparisons between years reasonable.

To address this inconsistency issue, the Estuary Program calculated the mean of closure events over the sixteen years of beach monitoring (total beach closures divided by years monitored) and compared the means to the 2015 tier designations. The combination of 2015 tier assignments and mean closure events allowed for more rigorous classification of High and Low Concern beaches.

Thus, the Estuary Program's classification scheme developed with RIDOH and MDPH (Table 1) used the following criteria:

- **High Concern Beaches:** Frequently monitored. Frequent closure events. Mean closure events per year greater than 1.5.
- **Low Concern:** Infrequently monitored. Fewer closure events over time. Mean closure events per year less than 1.5.

Based on each criterion, each beach was categorized as a High Concern or Low Concern beach. In Rhode Island[1], High Concern beaches were verified to have historically high closure frequency (more than 1.5 closure events per year). All Massachusetts beaches except Pierce Beach were excluded from the High Concern category due to low mean closure history.

Of the 37 monitored marine beaches in Narragansett Bay, fourteen beaches were classified as High Concern (13 RI, 1 MA) and 23 as Low Concern (18 RI, 5 MA) (Figure 1, Table 2).

DATA ANALYSIS

The total of beach closure events was calculated for marine beaches within Narragansett Bay and by Bay Regions, for High and Low Concern beaches (Figure 1), and plotted against seasonal rainfall over the sixteen years of data. The mean of beach closure events over time was calculated to quantify the number of years in which the total number of beach closures exceeded the average.

To compare beach closure events between Bay Regions, the Estuary Program choose to normalize the total of closure events by the number of beaches within each Bay Region that are High or Low Concern. By normalizing beach closure events, each Bay Region can be compared to other Bay Regions equally on a scale of 0 to 100 or as the percentage of normalized beach closure events; the latter is presented in this chapter.

The use of normalization reduces a biased representation. As an example, the Upper Estuary has the greatest number of High Concern beaches, and without normalization it would appear to be more affected by pathogen pollution than the other Bay Regions. The Upper Estuary has four times as many beaches as the Sakonnet River, and if the percent of closure events (after normalization) in each of these Bay Regions was 50 percent in any given year, this indicates that both Regions had equivalent impacts on the marine beaches in their respective Region. After normalization, if the Upper Estuary has a higher percentage of beach closure events, this would indicate that this Bay Region has been impacted more than the other Regions by pathogen contamination in that given year or across the years.

It is important to emphasize that this approach provides a comparison of the total of beach closure events on a Bay Region scale as opposed to a local scale, and therefore the results will not reflect trends of closures at individual marine beaches. Information

[1] For public marine beaches in Rhode Island, the Estuary Program estimated monitoring history based on the assumption that sampling began in 2002, when the number of licensed beaches in Rhode Island more than doubled. After 2002, the number of marine beaches monitored became relatively stable (±2 between 2002 and 2015).

on individual beaches is collected by the respective state health departments, MADPH and RIDOH. In addition, the Estuary Program performed Pearson's product-moment correlation analysis to determine trends of total beach closure events, across years, in Narragansett Bay and by Bay Regions, for High and Low Concern marine beaches.

WET AND DRY WEATHER

Total seasonal rainfall was plotted to explore temporal patterns of beach closure events and precipitation. Precipitation data were obtained from T.F Green Airport, Rhode Island, for the period between Memorial Day and Labor Day in each year from 2000 to 2015.

The Estuary Program calculated total rainfall in inches for each season. Wet years were defined as those with total rainfall over fourteen inches during the summer season. That definition was selected to align as closely as possible with wet and dry years as designated in other chapters of this report (see "Dissolved Oxygen" chapter). However, since the definition used in other chapters was based on monthly averages of streamflow, and not daily precipitation, some years differ as wet and dry across chapters.

Status and Trends

HIGH CONCERN BEACHES

Status

During the 2015 beach season, a dry season in terms of rainfall, High Concern beaches (Figure 1) were closed for a total of 38 distinct events across Narragansett Bay (Table 3; Figure 2). For High Concern beaches, closure events in 2015 were the sixth highest annual total during the period from 2000 to 2015. Overall, the Upper Estuary had the greatest percent of beach closure events (42 percent) when compared with the other two Bay Regions where marine beaches were impacted by unsafe pathogen levels. Closure events were normalized by the number of beaches in each Bay Region.

Trends

Analysis of High Concern beaches from 2000 to 2015 suggested that higher numbers of beach closure events corresponded with higher total precipitation through 2009, but not after 2009 (Figure 2). The highest numbers of beach closure events in Narragansett Bay occurred in 2006 and 2009, two wet years with 66 and 64 closures respectively. These wet years saw consistently high total closure events among High Concern beaches across all three Bay Regions (Figures 3, 4, and 5). The Upper Estuary had the greatest percentage of beach closure events in 2006 compared with the other two Bay Regions. In 2009, a wet year, the percentage of beach closure events was nearly even among the three Bay Regions (Figure 6), but the Sakonnet River had the highest percentage (38 percent).

In addition, even though beach closure events are shown to be increasing, from 2000 to 2015, there is no significant trend ($p \geq 0.005$) in closure events in Narragansett Bay (Figure 2) and within Bay Regions (Figures 3 and 5), except for the Sakonnet River Region ($p=0.004$) (Figure 4).

Using the average of total beach closure events over sixteen years of data, the Estuary Program used a yearly comparison to identify when beach closures were above average. Total beach closure events were above average (mean=32.5) in five of the sixteen years, three of which were wet seasons, including 2009 (Figure 2). Similarly, by grouping marine beaches based on their location, by Bay Regions, the patterns are different from or similar to

Table 3. Total and percent of 2015 normalized beach closure events at High Concern marine beaches by Bay Region.

Bay Region	High Concern Beaches in 2015			
	Total Beach Closure Events	Number of Beaches	Normalized Beach Closure Events	Percent of Beach Closure Events
Upper Estuary	25	8	3.13	42
Sakonnet River	4	2	2.00	27
Mouth of the Bay	9	4	2.25	31

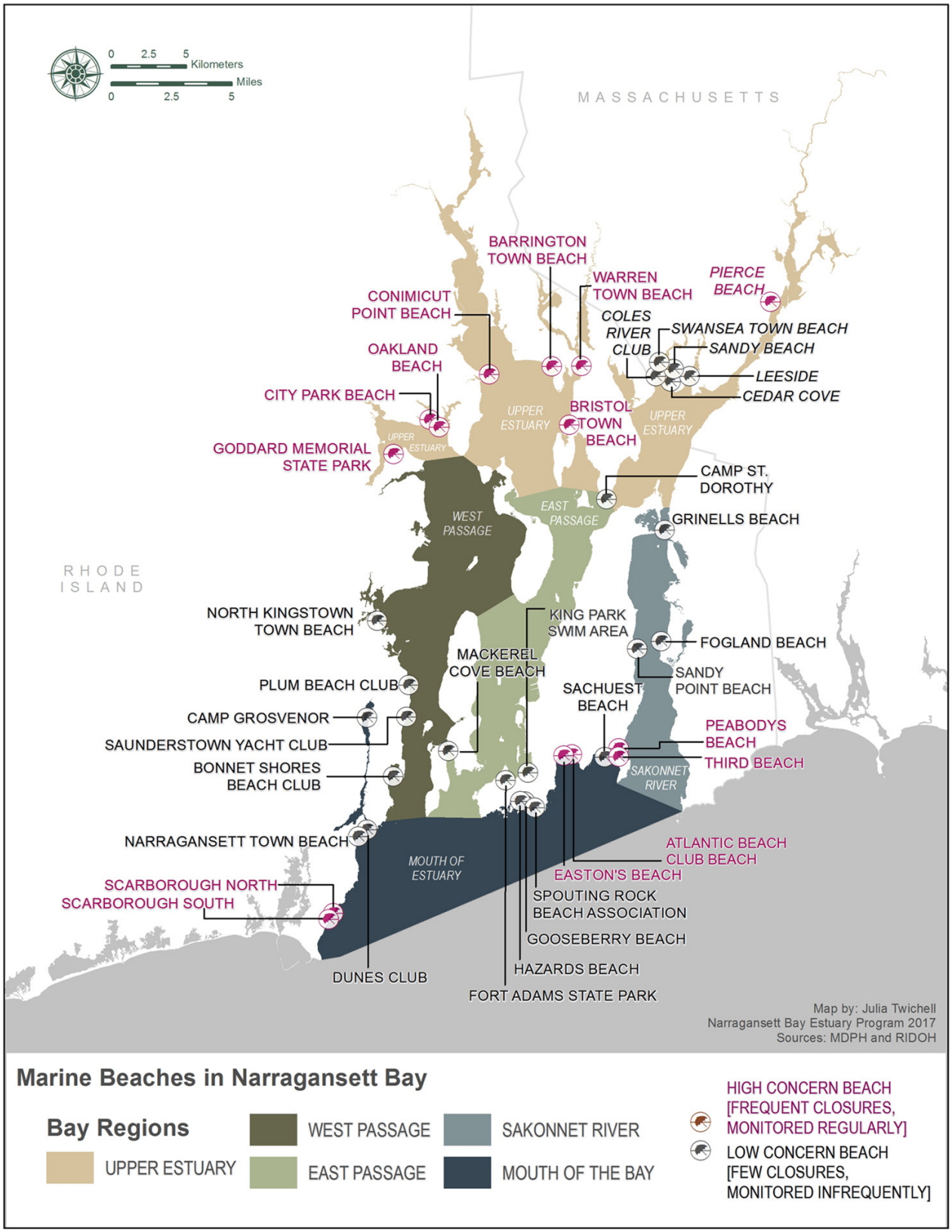


Figure 1. Locations of High and Low Concern marine (licensed) beaches in Narragansett Bay.

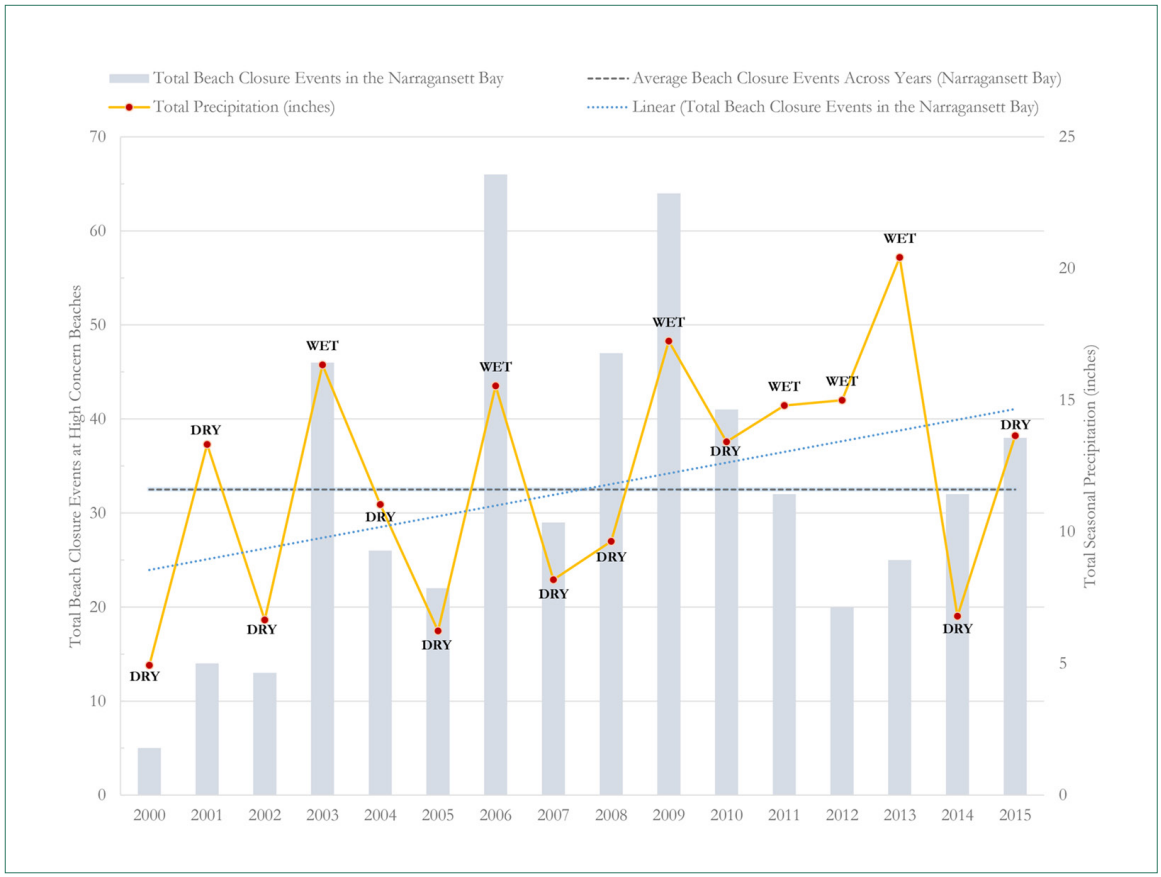


Figure 2. Total beach closure events (and trend line) from 2000 to 2015 in Narragansett Bay at High Concern beaches, and total seasonal rainfall in inches. Horizontal dashed line indicates average number of beach closure events across all years. Wet years were defined as years with total rainfall greater than fourteen inches during the summer season. The regression of total beach closure events was not significant ($p \geq 0.05$, $R^2 = 0.099$).

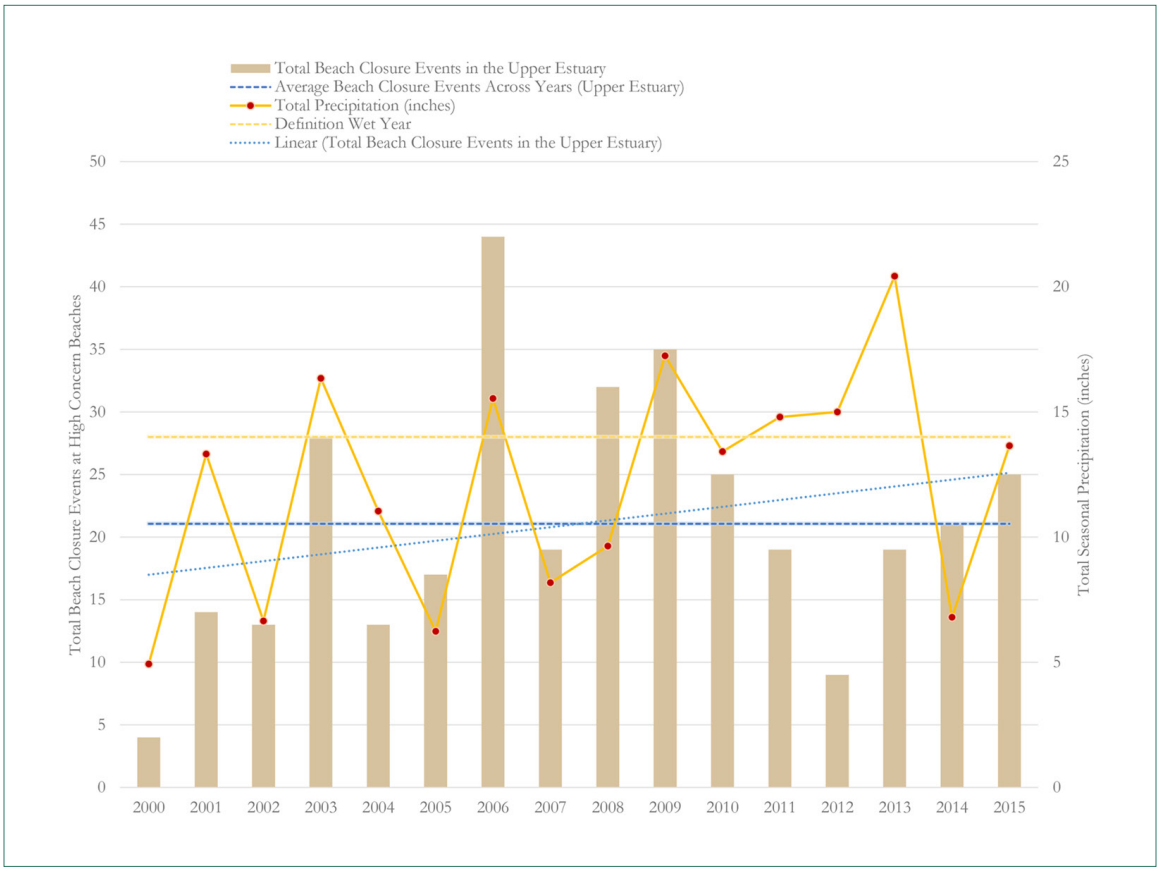


Figure 3. Total beach closure events (and trend line) from 2000 to 2015 in the Upper Estuary at High Concern beaches, and total seasonal rainfall in inches. Wet years were defined as years with total rainfall greater than fourteen inches during the summer season. The regression of total beach closure events was not significant ($p \geq 0.05$, $R^2=0.064$). *Color of bars corresponds to color of Bay Region on map in Figure 1.*

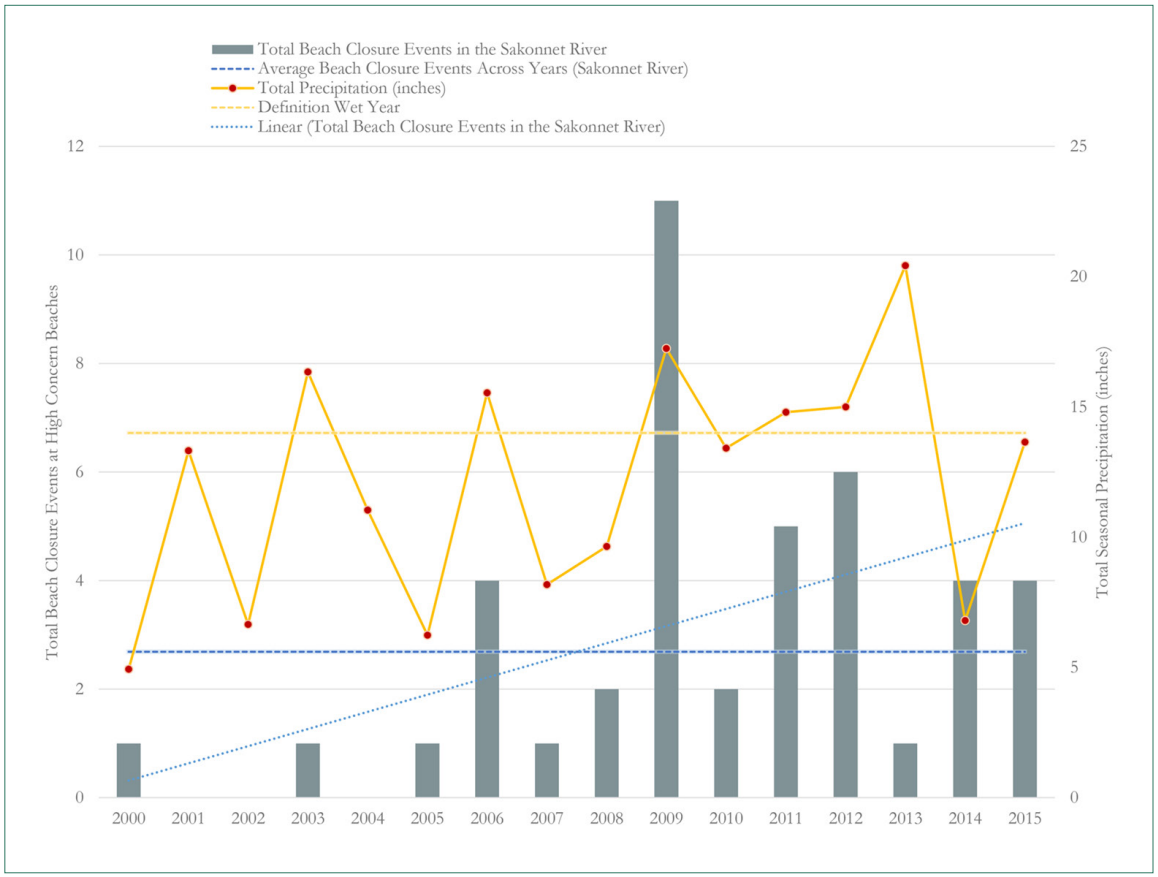


Figure 4. Total beach closure events (and trend line) from 2000 to 2015 in the Sakonnet River at High Concern beaches and total seasonal rainfall in inches. Wet years were defined as years with total rainfall greater than fourteen inches during the summer season. The regression of total beach closure events was statistically significant at $p=0.004$, $R^2=0.267$). *Color of bars corresponds to color of Bay Region on map in Figure 1.*

the Bay. In the Upper Estuary and Sakonnet River, the total beach closure events surpassed the average (mean=21 and mean=3, respectively) in six of the sixteen years (Figures 3 and 4), and the Mount of the Bay (mean=9) in eight of the sixteen-year record (Figure 5). For the Upper Estuary and Mouth of the Bay, three of these spikes occurred in wet seasons (2003, 2006, 2009), the same pattern as for the entire Narragansett Bay (Figures 2, 3 and 5). After 2009, wet years do not appear to correspond to above average total beach closure events. While most of the spikes for the Sakonnet River were during dry seasons, the only wet year was 2009 (Figure 4).

In comparing beach closure events between Bay Regions, the percentage of closure events by Bay Region is based on normalized beach closure events. In eight of the sixteen years of beach monitoring, the Upper Estuary had the highest percentage of beach closures compared with the other two Bay Regions. This is not including 2001 and 2002, when all beach closure events occurred in the Upper Estuary. Two of the eight years were wet seasons (2006 and 2013). In these years, 46 and 58 percent of beach closure events were in the Upper Estuary, indicating that pathogen contamination had a greater impact on overall High Concern beaches of this Bay Region when compared with the other two Bay Regions. During the remaining five years of beach monitoring, the Sakonnet River had the highest percentage of beach closure events in 2009, 2011, and 2012 (all wet seasons), and the Mouth of the Bay had the highest percentage in 2003 (wet) and 2004 (dry) (Figure 6). Even though total beach closure events by Bay Region have been normalized, the Upper Estuary had the highest percent of closure events across years, among the three Bay Regions with High Concern beaches.

Furthermore, the analysis of percent closure events by Bay Region showed that 2015 is a representative year of overall closure events, as percentages across years and in 2015 alone for each Bay Region (Tables 3 and 4) are similar.

Table 4. Total and average of beach closure events at High Concern marine beaches in Narragansett Bay and Bay Regions and percent of normalized beach closure events by Bay Region from 2000 to 2015.

Bay Region (Number of Beaches)	Total (Average) beach closure events across years	Normalized beach closure events across years (normalized by number of beaches)	Percent of beach closure events across years by Bay Region (normalized)
Upper Estuary (8)	337 (21.1)	42.1	43
Sakonnet River (2)	43 (2.7)	21.5	22
Mouth of the Bay (4)	140 (8.7)	35.0	35
Narragansett Bay (14)	**520 (32.5)**		

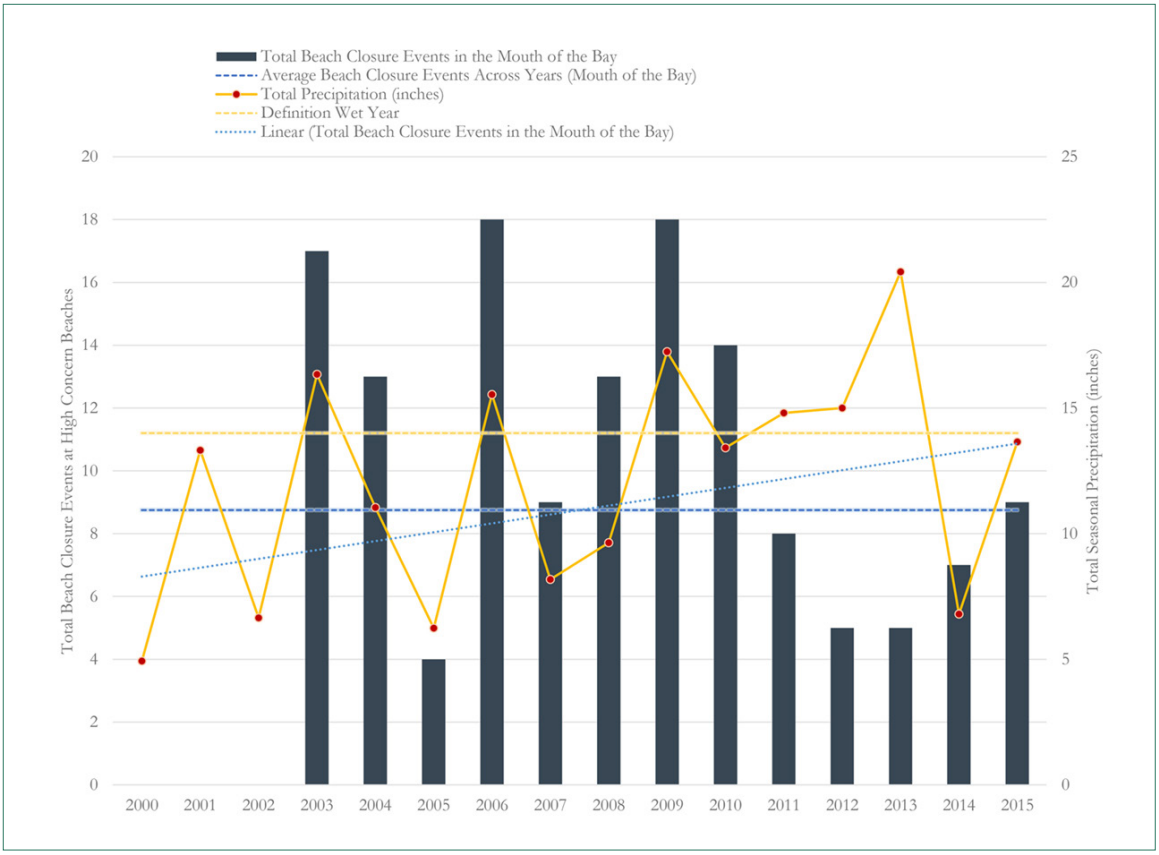


Figure 5. Total beach closure events (and trend line) in the Mouth of the Bay at High Concern beaches and total seasonal rainfall in inches. Wet years were defined as years with total rainfall greater than fourteen inches during the summer season. The regression of total beach closure events was not significant ($p \geq 0.05$, $R^2=0.046$). *Color of bars corresponds to color of Bay Region on map in Figure 1.*

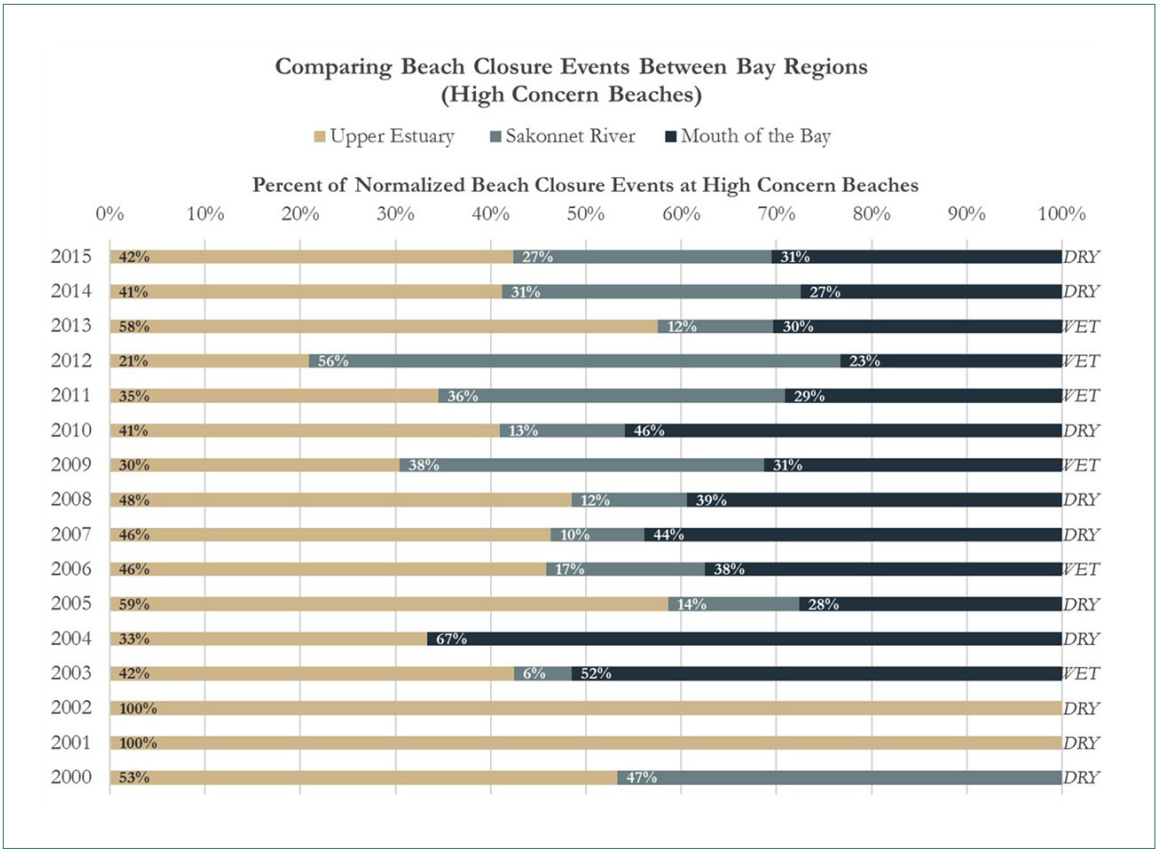


Figure 6. Percent of normalized beach closure events by Bay Region at High Concern beaches. Wet years were defined as years with total rainfall greater than fourteen inches during the summer season.

LOW CONCERN BEACHES

Status

At the Low Concern beaches (Table 2; Figure 1), only six beach closure events occurred in 2015. The Upper Estuary had the highest percent of normalized closure events (48 percent), followed by the West Passage (40 percent), compared with the other Bay Regions; however, there were no beach closure events at the Low Concern beaches in the Sakonnet River or East Passage (Table 5; Figure 7).

Trends

For Low Concern beaches in Narragansett Bay, total beach closure events appeared to follow precipitation patterns in most years (Figure 7). Greater than twelve beach closure events occurred in Narragansett Bay across Low Concern beaches during each of the wet years, at the same time exceeding average of closure event across the sixteen years of beach monitoring (mean=9.9) (Table 6; Figure 7). However, as is true for High Concern beaches, more work will be needed to pinpoint what factors drive closures among Low Concern beaches given the variability of the data across years, the factors that determine the number of closure events, and the inherent limitations of the data.

The magnitude of total beach closure events at Low Concern beaches was much lower compared to High Concern beaches (Figures 2-5, 7). The maximum total beach closure events observed in the beach monitoring record for Low Concern beaches never exceeded ten closure events per Bay Region. However, High and Low Concern categories should not be compared directly as High Concern beaches are more frequently monitored.

The trend of beach closure events across the sixteen years of water quality monitoring at marine beaches is not significant at $p \geq 0.05$ (Figure 7).

Among Bay Regions and across years of beach water quality monitoring, the percent of normalized beach closures was the highest in the East Passage (42 percent). Next highest was the West Passage with 19 percent. These percentages should be used cautiously since factors such as monitoring frequency during a single season or across seasons is expected to be highly variable at Low Concern beaches.

On the other hand, the total of beach closure events at Low Concern beaches within each Bay Region exceeded the average (Table 6) in at least seven of the sixteen years of data record (Figure 7). For example, in the Upper Estuary the average of beach closure events across years, from 2000 to 2015, is 2.4 (Table 6). This can be compared with the total of beach closure events each year in Figure 7, and in 2006 this total doubled the average.

Nonetheless, the percent of normalized beach closure events in Low Concern beaches by Bay Region shows that no single Bay Region stands out as the most impacted by pathogen contamination, over the sixteen years of water quality monitoring during the beach seasons (Figure 8). However, the East Passage had the highest percentage of (normalized) beach closures from 2000 to 2004, compared with the other Bay Regions, while the Sakonnet River generally had the lowest (except in 2013) (Figure 8). The Upper Estuary had the greatest percentage of Low Concern beach closure events, compared with the other Bay Regions, in 2010, 2011, and 2015 (two wet years and one dry year), and this pattern does not match the highest closure years for High Concern beaches in the Upper Estuary.

Table 5. Total and percent of 2015 normalized beach closure events at Low Concern marine beaches by Bay Region.

Bay Region	Low Concern Beaches in 2015			
	Total Beach Closure Events	Number of Beaches	Normalized Beach Closure Events	Percent of the Bay's Beach Closure Events
Upper Estuary	3	5	0.6	48.3
East Passage	0	4	0.0	0
West Passage	2	4	0.5	40.2
Sakonnet River	0	3	0.0	0
Mouth of the Bay	1	7	0.1	11.5

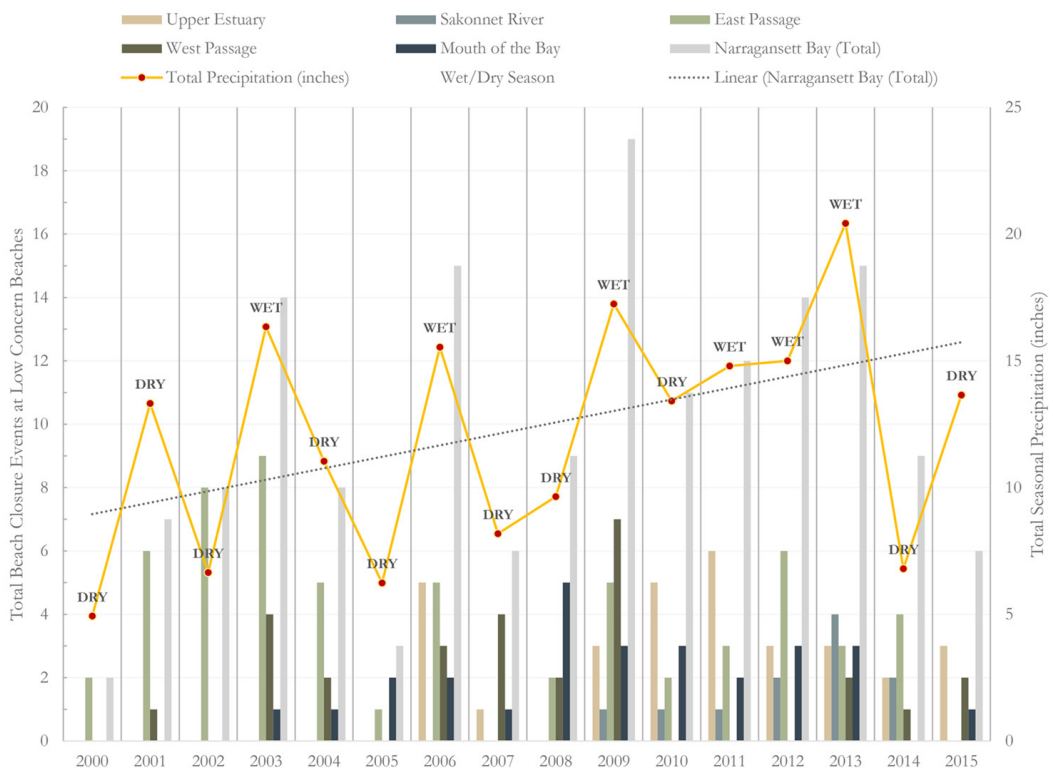


Figure 7. Total beach closure events (and trend line) in Narragansett Bay at Low Concern beaches, by Bay Region, and total seasonal rainfall in inches. Wet years were defined as years with total rainfall greater than fourteen inches during the summer season. The regression of total beach closure events for Low Concern beaches across Narragansett Bay was not significant ($p \geq 0.05$, $R^2=0.134$). *The colors of the bars for the Bay Regions correspond to areas shown on map in Figure 1.*

Table 6. Total and average of beach closure events at Low Concern marine beaches in Narragansett Bay and Bay Regions and percent of normalized beach closure events by Bay Region from 2000 to 2015.

Bay Region (Number of Beaches)	Total (Average) beach closure events across years [1]	Normalized beach closure events across years by number of beaches	Percent of beach closure events across years by Bay Region (normalized)
Upper Estuary (5)	31(2.4)	6.2	17
East Passage (4)	61(3.8)	15.3	42
West Passage (4)	28(1.8)	7.0	19
Sakonnet River (3)	11(0.7)	3.7	10
Mouth of the Bay (7)	27(1.7)	3.9	11
Narragansett Bay (14)	**158(9.9)**		

[1] For the Upper Estuary, the average is based on data from 2003 through 2015 because data were not available prior to 2003. For the other four Bay Regions, the average of beach closure events encompassed the entire period from 2000 to 2015.

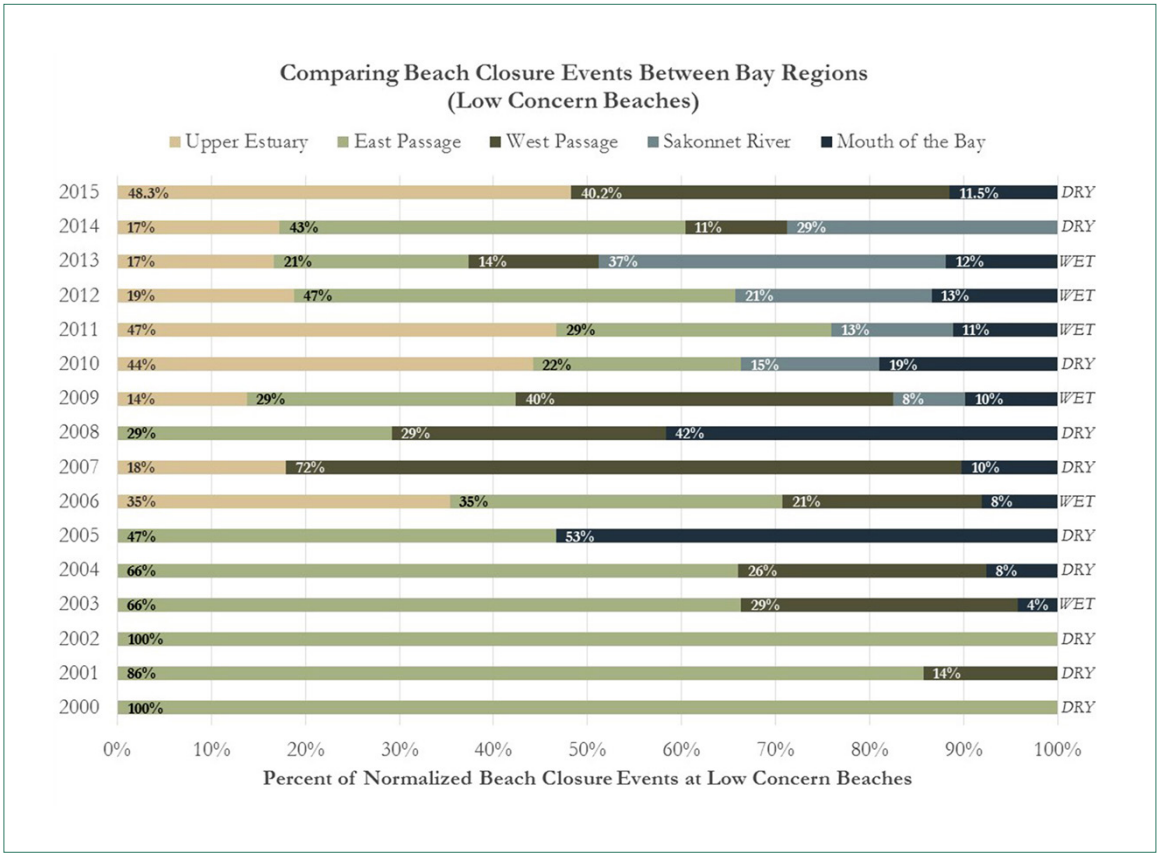


Figure 8. Percent of normalized beach closure events by Bay Region at High Concern beaches. Wet years were defined as years with total rainfall greater than fourteen inches during the summer season.

Discussion

Regional analysis of sixteen years of marine beach closure data in Massachusetts and Rhode Island revealed a striking record of numerous beach closure events concentrated among the eight High Concern beaches in the Upper Estuary (Tables 3 and 4; Figure 3), a Bay Region with historically high pathogen loading. In fact, the Upper Estuary had the highest percentage of normalized beach closure events in 2015 and across years of beach water quality monitoring—more than 40 percent higher than the other Bay Regions with High Concern Beaches (Tables 3 and 4; Figure 6).

Historical patterns in beach closure event frequency at High Concern beaches indicate that closure events may correspond to seasonal precipitation (Figures 2 through 5). During heavy rainfall, stormwater runoff can become contaminated by multiple sources including animal feces (wild and pet) and untreated or poorly treated sewage (failing septic systems, cesspools, and CSOs). This runoff can discharge to waterways, bringing harmful pathogens to beaches. Results suggest that fewer beach closure events occur during dry seasons, perhaps driven by localized and transient factors such as subsurface transport of pathogen-contaminated groundwater (Lipp at al. 2001) or contamination directly from humans, wildlife, or pets. As evidenced in the wet seasons of 2006 and 2009, rainfall exceeded the capacity of the ecosystem to absorb or capture runoff, resulting in a high frequency of beach closure events. However, after 2009 the frequency of beach closure events did not appear to spike during wet seasons such as 2011, 2012, and even 2013 (the wettest season in this record) (Figures 2 through 5).

The Estuary Program performed an exploratory Pearson's product-moment correlation analysis of High Concern beaches based on observations that beach closure events were higher in wet years across the Bay and Bay Regions prior to 2009. These observations suggested that precipitation was a driver of beach closure events prior to 2009. No such pattern was observed after 2009. In the analysis of the entire period from 2003 to 2015, which included all years with reliable monitoring frequency, precipitation was not correlated with mean beach closure events ($r=0.323$, $p=0.223$, $N=13$). However, between 2003 and 2008 precipitation was positively correlated to mean beach closure events ($r=0.828$, $p=0.006$, $N=7$), and between 2009 and 2015 no correlation was observed ($r=-0.375$, $p=0.400$, $N=6$). This preliminary analysis suggests a strong positive relationship between closure events and seasonal precipitation prior to 2009, and no relationship after 2009. Following further development of this beach health indicator, a robust statistical analysis that also accounts for rainfall variability will be necessary to further test the validity of this observation (see Data Gaps and Research Needs).

The weakened response to precipitation among High Concern beaches after 2009 was perhaps related to reduced loads of harmful pathogens to those beaches. Watershed stressors such as impervious cover and wastewater infrastructure that exacerbate pathogen transport to receiving waters during rain events can be mitigated by local and regional management actions (e.g., stormwater retention technology, wastewater system upgrades). However, additional data analysis will be needed to determine the effects of management actions on beach closure events as well as on actual pathogen loadings in Narragansett Bay waters (see Data Gaps and Research Needs).

In addition, another exploratory analysis was conducted for High Concern beaches, where beach closure events showed an increasing trend between 2000 and 2008, and a declining trend thereafter. The relationship of total beach closure events over the sixteen years of beach water quality monitoring, shows a significant evidence that the number of events increased from 2000 to 2008, across Narragansett Bay ($r=0.685$, $p=0.035$) and the Upper Estuary Region ($r=0.668$, $p=0.041$), but not in the other Bay Regions ($p<0.05$). The declining trend, between 2009 and 2015, was not significant at $p<0.05$, for High Concern beaches across the Bay or beaches within the Bay Regions (Figure 9 through 12).

Following these findings (pre- and post-2009) and to complement the results in this chapter, the Estuary Program conducted an exploratory analysis to evaluate the apparent upward trend in closures from 2000 to 2008 and the apparent decline thereafter at High Concern beaches. The relationship of total beach closure events over the sixteen years of beach water quality monitoring shows significant evidence that the number of events increased from 2000 to 2008 across Narragansett Bay ($r=0.685$, $p=0.035$) and the Upper Estuary Region ($r=0.668$, $p=0.041$), but not in the other Bay Regions ($p\geq0.05$). The declining trend, between 2009 and 2015, was not significant at $p\geq0.05$ for High Concern beaches across the Bay or beaches within the Bay Regions (Figures 9–12).

Therefore, the apparent reduction or loss in association of closure events with total seasonal precipitation after 2009 is perhaps a positive sign with respect to source reduction. The combined effect of localized management actions to date appears to have reduced the source loads associated with rainfall. Nevertheless, as reflected in the results of

this analysis, trends of beach closure events should not be explored across the entire Bay, but instead by areas such as the Bay Regions. In addition, it should also be recognized that new baselines for trend analysis should be defined based on the preliminary results pre- and post-2009.

While management actions can mitigate localized stressors, the hydrodynamic characteristics of a beach can also have a strong impact on water quality. Translating or interpolating the outcomes of localized management actions from one location to another part of the Bay that are not geographically, hydrologically, or hydraulically connected can be misleading. Beachfronts that are exposed and well-flushed, like those in the Mouth of the Bay, are less likely to have bacterial contamination (Coakley et al. 2016). Beaches in or near the Mouth of the Bay have greater wave action and water circulation and experience fewer beach closures than those in the Upper Estuary (Table 4). However, High Concern beaches near the Mouth of the Bay continue to see closures despite the benefits of greater circulation. Beaches in enclosed embayments of the Upper Estuary (e.g., Greenwich Bay) with reduced circulation may experience higher closure events comparatively even if pathogen inputs were reduced.

On the landscape spectrum, watershed stressors such as impervious cover and wastewater infrastructure that exacerbate pathogen transport to receiving waters during rain events can be mitigated by local and regional management actions (e.g., stormwater retention technology, wastewater system through CSOs improvements, onsite wastewater system upgrades). However, additional data analysis will be needed to determine the effects of management actions on beach closure events as well as on actual pathogen loadings in Narragansett Bay waters (see Data Gaps and Research Needs).

The Town of Bristol, Rhode Island, has set an example to demonstrate water quality improvements at public beaches. In 2013, the Town completed restoration and implementation of stormwater best management practices (BMPs). Pre-BMPs, the total number of closure days at Bristol Town Beach were linked directly to rainfall events. The number of beach closures declined post-BMPs, despite an increasing trend in precipitation, from an average of eight days per season (metric used by RIDOH) before restoration efforts to zero days following the restoration. These efforts have had ancillary benefits such as improvement of water quality at shellfish beds immediately offshore (USEPA 2015).

Low Concern beaches were characterized by fewer closure events than High Concern beaches (Tables 4 and 6; Figures 2–5, 7). This is in part because Low Concern beaches are monitored less frequently and also because Low Concern beaches are at a lower risk for pathogen contamination. Total closure events at Low Concern beaches closely followed seasonal rainfall almost all years, both across Narragansett Bay and by Bay Region, which could indicate that pathogen contamination is triggered by precipitation, and the magnitude or the frequency of precipitation events.

A majority of the beaches in Narragansett Bay have closed at least once in the past sixteen years, suggesting that beach closures may be difficult to fully eliminate in a highly developed watershed like Narragansett Bay. While reductions in stormwater sources have importantly reduced pathogen loadings associated with rain events, particularly in the Upper Bay, residual and intermittent sources remain. Often these are associated with localized factors and climatic, pollution-related events.

Nevertheless, recent observations made by the Estuary Program and partners indicate that efforts to mitigate contaminated stormwater runoff through sewer improvements, green infrastructure, waste management initiatives, and other BMPs have had positive effects and have contributed to supporting the vital role that beaches play in supporting quality of life, tourism, and the economy.

Regarding climate change stressors, marine beaches are likely to be susceptible. More frequent and intense storms may increase the supply of contaminated stormwater runoff to beaches, particularly if heavy rainfall events exceed the capacity of existing gray and green infrastructure (see "Precipitation" chapter). Additionally, warmer temperatures increase bacterial growth, which may be an additional impact of climate change on beach water quality (Michalak 2016; see "Temperature" chapter). Increased pathogen loads and warmer conditions will likely impact beach closures.

In addition, harmful algal blooms (including microalgae and cyanobacteria) have increasingly garnered attention. Cyanobacteria blooms are more common in freshwater systems but also occur in salt water (Paerl et al. 2011). The toxins potentially associated with bloom events can pose risks to public health and aesthetic enjoyment. More frequent and intense storms expected as a result of climate change may increase nutrient loading from contaminated stormwater runoff, creating conditions favorable to harmful algal blooms (see "Water Quality Conditions for Aquatic Life" chapter).

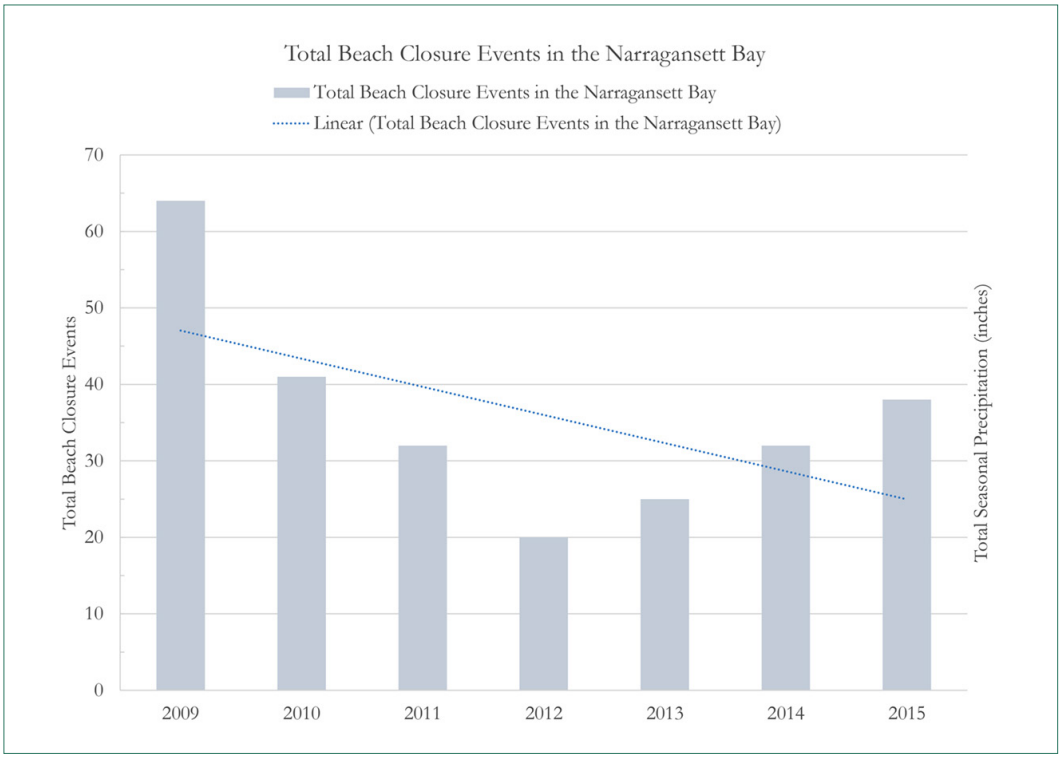


Figure 9. Total beach closure events (and trend line) in Narragansett Bay at High Concern beaches from 2009 to 2015. The regression of total beach closure events was not significant at $p < 0.05$ ($R^2=0.310$)

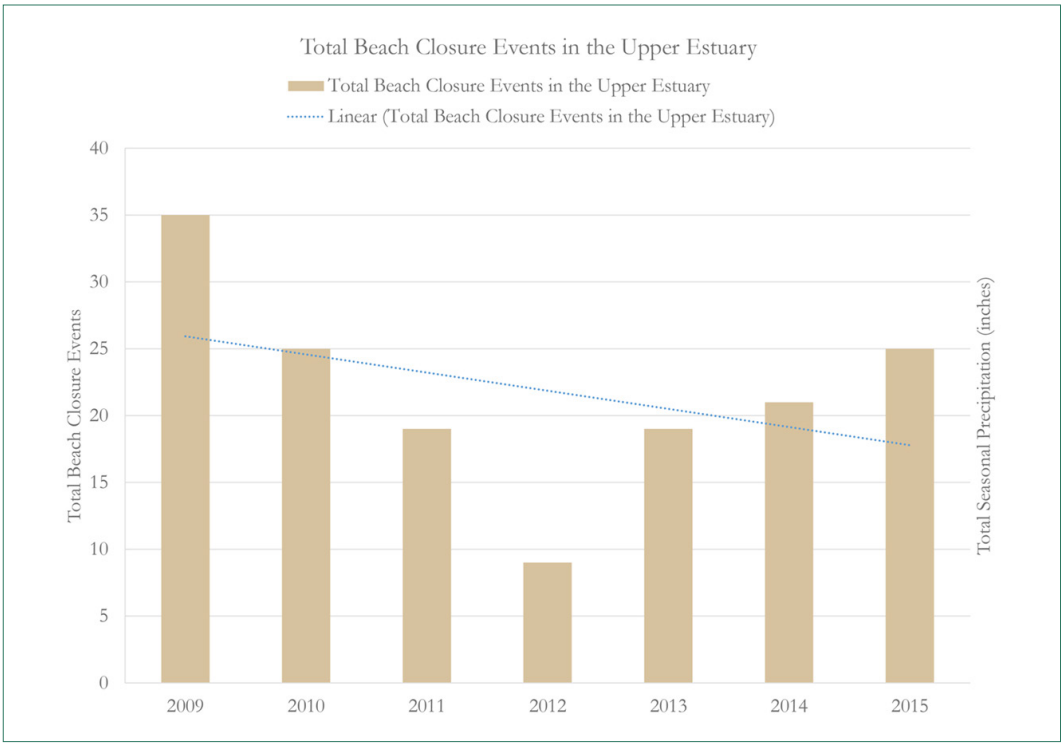


Figure 10. Total beach closure events (and trend line) in Upper Estuary at High Concern beaches from 2009 to 2015. The regression of total beach closure events was not significant at $p < 0.05$ ($R^2=0.138$). *Color of bars corresponds to color of Bay Region on map in Figure 1.*

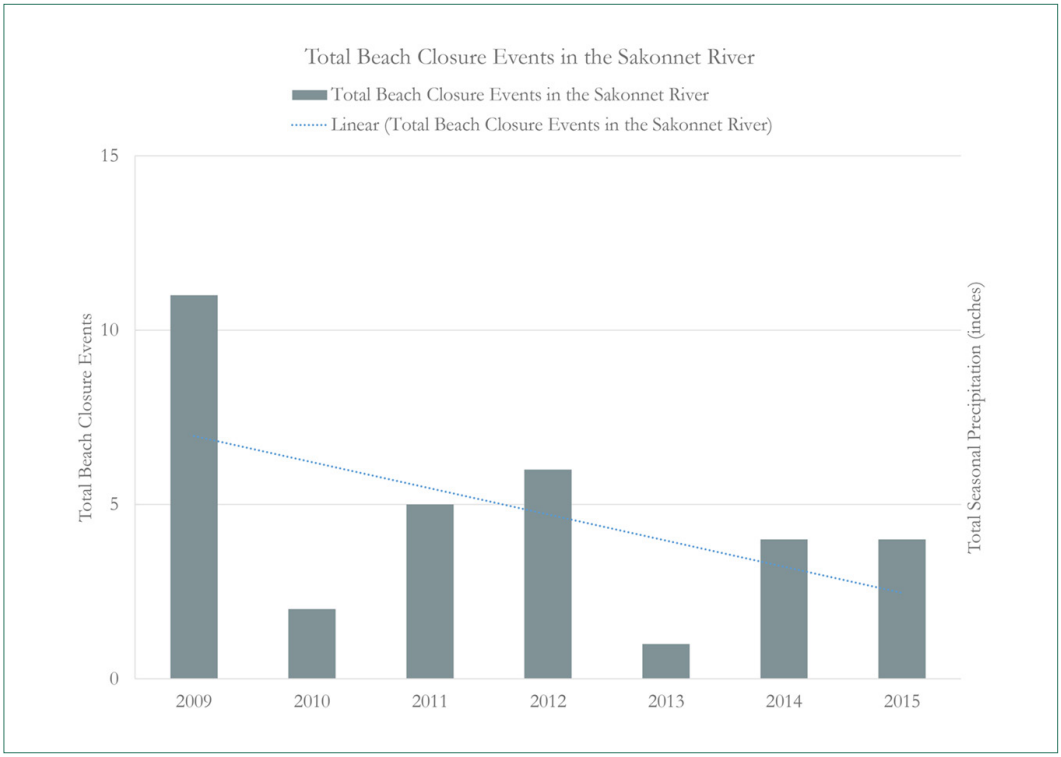


Figure 11. Total beach closure events (and trend line) in the Sakonnet River at High Concern beaches from 2009 to 2015. The regression of total beach closure events was not significant at $p < 0.05$ ($R^2=0.248$). *Color of bars corresponds to color of Bay Region on map in Figure 1.*

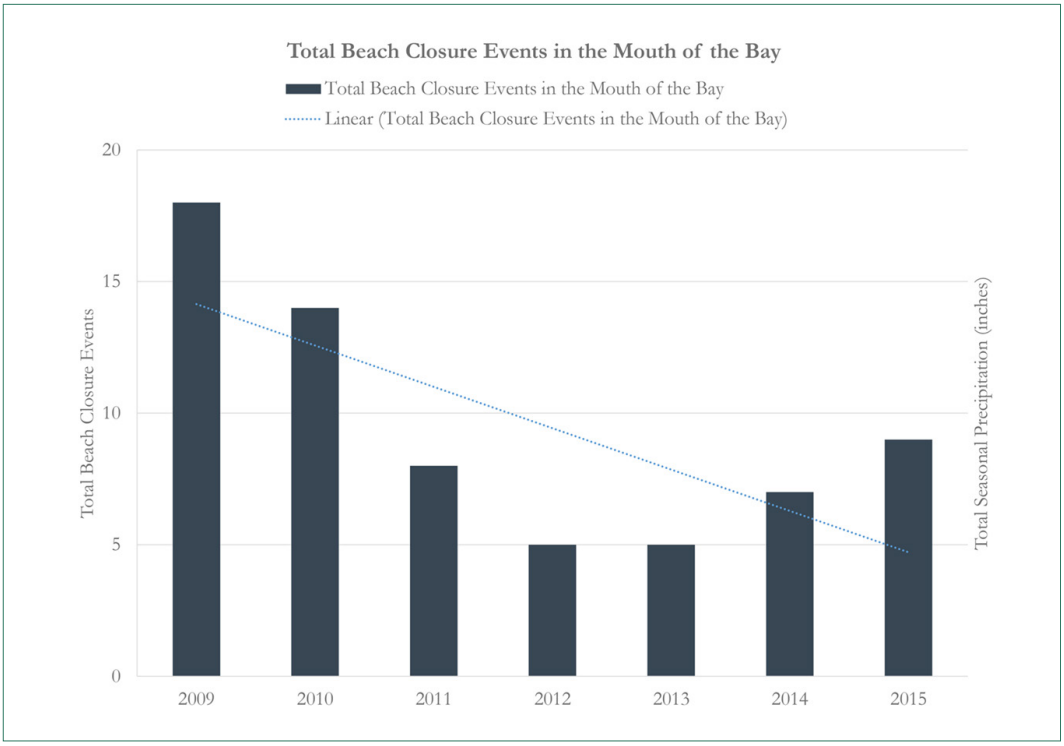


Figure 12. Total beach closure events (and trend line) in the Mouth of the Bay at High Concern beaches from 2009 to 2015. The regression of total beach closure events was not significant at $p < 0.05$ ($R^2=0.488$). *Color of bars corresponds to color of Bay Region on map in Figure 1.*

Climate change may also physically alter the structure of coastlines through sea level rise, storms, storm surge, nuisance flooding, and erosion (see "Sea Level" chapter). These changes in the coastline may contribute to higher levels of pathogen contamination as stormwater and wastewater infrastructure located along the coastline will likely be burdened by higher sea levels. Many beaches are increasingly squeezed between rising seas and expanding coastal development.

Data Gaps and Research Needs

- The beach indicator should be refined by the development of other metrics. One option to explore is the development of a bi-state dataset that uses bacterial counts normalized by monitoring frequency (number of samples per season per beach) for the period of 2000 to the present to develop a more consistent and sensitive metric. Further analysis using bacteria counts associated with sampling dates will allow for cross-comparison between years with differing monitoring frequency and regulatory stringency. A protocol is needed to evaluate bacterial counts in the context of sampling frequency. Furthermore, the results of future analyses should be compared to current findings to corroborate the preliminary trends noted in this report.
- Further work is needed to develop appropriate metrics for freshwater beaches in the Narragansett Bay Watershed. Data are limited and were not reviewed for this report.
- As recent preliminary trends indicate a weakening relationship between rainfall and beach closure events, it will be important to continue to evaluate beach closures in wet years. With an indicator based on bacterial counts, the Estuary Program anticipates that a robust statistical analysis could address temporal trends and relationships with precipitation. Additional factors that influence microbial contamination and its persistance at beaches can be used to develop predictive models on a beach-specific basis. These include wind direction and speed, water temperature, wave height, changes in wastewater infrastructure and land use (Wu and Jackson 2016), and patterns in human use.
- For High Concern beaches, development of models to support management is of interest. With appropriate input data and validation, predictive models can drive better management to reduce exposure to high-risk conditions. Unlike current microbiological analyses which typically characterize water quality on the previous day, models can predict when a beach should be closed (i.e., at the times when adverse conditions result in high levels of enteric microbes).
- Detailed analyses of existing management actions such as CSO abatement projects, stormwater infrastructure improvements, and waste management initiatives based on bacterial counts and sampling history as metrics are likely to be useful in informing BMPs. Improvements at specific beaches are likely related to localized management actions. Pinpointing successful management strategies that target sources of contamination will be beneficial from economic, social, and public health perspectives.
- While continuing to build on the information gained through both state beach monitoring programs, it will also be imperative to relate beach assessments to other programs that evaluate microbial contamination in the Bay's waters. These include assessments of long-term and comprehensive water quality characterizations of the Bay's waters to meet standards for recreational uses, including primary and secondary contact, as well as designations of shellfishing areas.

Acknowledgments

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State of Narragansett Bay and Its Watershed
2017 Technical Report

Condition Indicators of Public Health

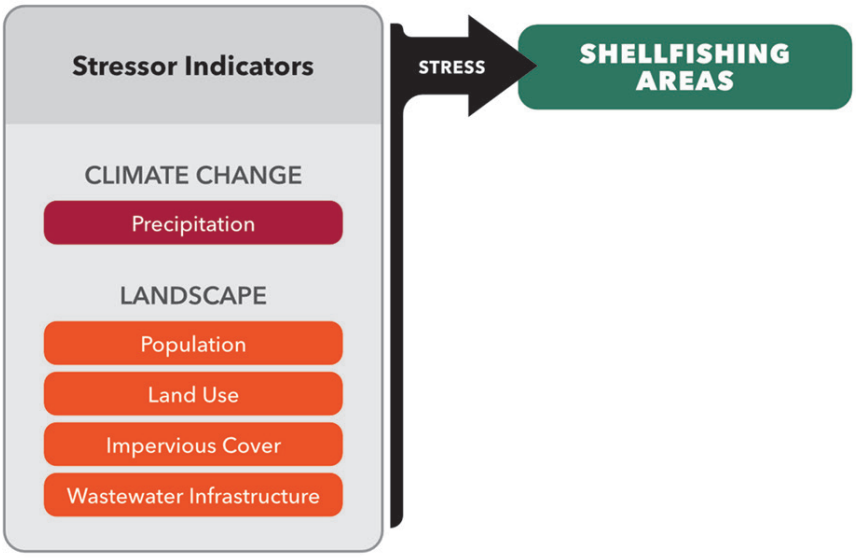
CHAPTER 24: SHELLFISHING AREAS

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Photo: Shellfishing in the Upper Narragansett Bay, south of Nayatt Point Lighthouse and Conimicut Point, RI (Eivy Monroy)

NARRAGANSETT BAY ESTUARY PROGRAM

Overview



BACKGROUND

- Harvesting of shellfish plays an important role in the economy and culture of Narragansett Bay. To protect public health from contaminated shellfish, primarily due to harmful pathogens, state agencies regulate where shellfish can and cannot be harvested for direct human consumption. The status of shellfishing areas serves as an indicator of public health conditions in the Bay. Contamination enters the Bay primarily in discharges from wastewater treatment infrastructure and in runoff of precipitation from land.

KEY FINDINGS

- ***Status:*** In 2015, 63 percent of Narragansett Bay was classified as approved for shellfishing for direct human consumption (although one-third of the approved area may not be suitable for shellfishing because of water depth, commercial navigation channels, or other reasons). Thirteen percent of the Bay was classified as conditionally approved, and 24 percent as prohibited; shellfishing growing areas with these two classifications were located primarily in the Upper Estuary.

- ***Trends:*** Between 1995 and 2005, the total area approved for shellfishing declined three percent from 82,386 acres to 79,753 acres, while prohibited areas increased. Conversely, prohibited areas decreased in the subsequent ten years from 31,667 acres in 2005 to 30,542 acres in 2015, while conditionally approved areas increased. In 2017, 3,711 acres in the Upper Estuary at the mouth of the Providence River were upgraded from conditionally approved to approved. Most of the positive changes in the last decade have occurred in the Upper Estuary, where water quality historically was poor.

Introduction

State agencies in Massachusetts and Rhode Island administer shellfish management programs that assess water quality conditions and regulate shellfish harvesting for commercial and recreational purposes throughout Narragansett Bay. Each of the states classifies shellfishing areas based on water quality levels of fecal coliform bacteria and other management factors. The classifications are designed to protect public health and ensure the safe consumption of shellfish. Approved waters are areas where harvesting is allowed on a regular basis, conditionally approved waters allow shellfish harvest under certain conditions, and in prohibited waters shellfish harvest for human consumption is not allowed at any time (CRC 2014).

The importance of good water quality for shellfishing in Narragansett Bay was recognized over a century ago, when water pollution was a primary cause of the collapse of a vibrant oyster industry. At its height in the early 1900s, the local oyster industry employed over a thousand people and had an annual harvest of 1.4 million bushels (Schumann 2015). Total acreage leased to private growers peaked in 1912 at 20,846 acres, representing seventeen percent of the estuarine waters of Narragansett Bay. Oyster landings in Rhode Island collapsed by the 1950s (Oviatt et al. 2003).

In 1910, an investigation into contamination in oyster beds found traces of sewage and fecal coliform bacteria in many upper Bay oyster beds, a result of the four hundred percent increase from 1850 to 1900 in human population in coastal areas around the Bay (see "Population" chapter; Vadeboncoeur et al. 2010). Following a nationwide typhoid epidemic, caused in part by the consumption of polluted shellfish, the National Shellfish Sanitation Program was established in 1925. This national program created standards for shellfish intended for consumption and required states to sample all shellfish waters and to close areas that did not meet standards. In 1926, Massachusetts closed the waters of the Taunton River and the northern portion of Mount Hope Bay. Between 1937 and 1985, thousands of bushels of quahogs and oysters were relayed or transported from the Taunton River to various towns in Rhode Island and Massachusetts for depuration or cleansing before harvest for human consumption.

Shea (1946) outlined Rhode Island's early water classifications for shellfish areas. Waters suitable for the cultivation of market shellfish were primarily in the West and East Passages, Sakonnet River, lower Mount Hope Bay, Palmer River, Barrington River, Kickemuit River, and parts of Bristol Harbor and Greenwich Bay. Areas that were suitable for culture of seed oysters were identified in Mount Hope Bay and the lower Providence River. Waters in the upper Providence River, north of Field's Point, were defined as grossly polluted. By the 1970s, waters in the Providence River were closed half of the time for shellfishing. As a result of the growing population in the upper Bay and the limited flow capacity of the Field's Point wastewater treatment facility, untreated and partially treated sewage were discharged directly to the upper Bay, raising public health concerns regarding the consumption of shellfish from those waters (Schumann 2015).

Massachusetts regulates the portions of Mount Hope Bay and the Palmer River that lie within its state boundary (MADMF 2013), representing approximately five percent of Narragansett Bay. The Rhode Island Department of Environmental Management (RIDEM) Office of Water Resources and the Division of Fish and Wildlife Marine Fisheries Section determine shellfish harvest restrictions for the remaining 95 percent of Narragansett Bay (RIDEM 2015). All approved and conditionally approved areas are sampled for fecal coliform and harmful algal blooms six to twelve times per year. There are more than 110 fixed stations in the approved growing areas, with nearly 2,000 samples collected annually by the two states.

Estuarine waters for shellfish harvesting are designated by the states pursuant to the Federal Clean Water Act. Class SA waters are designated for shellfishing for direct human consumption; they may be approved or conditionally approved. Class SB waters are suitable for shellfish harvesting for controlled relay and depuration (RIDEM 2010, MassDEP 2012). The maximum level of fecal coliform bacteria allowed in Class SA waters is a geometric mean of 14 organisms per 100 mL, and the maximum for Class SB waters is 88 organisms per 100 mL (MassDEP 2012). Shellfish classifications are also based on sanitary surveys and shoreline surveys conducted by the state shellfish programs.

In this chapter, the Narragansett Bay Estuary Program examines the status of shellfishing areas as of 2015, and identifies trends from 1995 to 2015 in the areas of Narragansett Bay that are classified as approved, conditionally approved, or prohibited for the harvest of shellfish for direct human consumption.

Methods

The Estuary Program coordinated bi-state meetings with representatives from the Massachusetts Division of Marine Fisheries (MADMF) and the Rhode Island Department of Environmental Management (RIDEM) to examine various approaches for using shellfishing areas as a proxy indicator of public health and water quality. Through these discussions, it was determined that shellfish growing areas (area and percent change) can be measured consistently between the two states to track changes (improvements or declines) in water quality in the Bay, as the state data are consistent in purpose and the classification systems are comparable. For this analysis, the Estuary Program defined three categories derived from the shellfish growing area classification systems used by Massachusetts and Rhode Island:

- **Approved:** Growing areas where shellfish harvesting is allowed for direct human consumption all year round (some exceptions may apply)
- **Conditionally Approved:** Growing areas where shellfish harvesting is allowed for direct human consumption with some restrictions, depending on each state's shellfish program criteria
- **Prohibited:** Waters where shellfish harvesting is not allowed for direct human consumption

For the purposes of this chapter, shellfish growing areas used for controlled relay and depuration in Massachusetts and Rhode Island are considered prohibited. The prohibited category also includes small portions of the Bay in Rhode Island that are classified as unassessed or where shellfishing is prohibited for other reasons not related to monitored water quality. Examples include areas located within marinas, near discharges from wastewater treatment facilities, in waters impacted by actual or potential sources of deleterious substances, or in waters where pollution impacts are not predictable. Harvesting of shellfish in certain approved waters may be restricted by RIDEM's Division of Fish and Wildlife Marine Fisheries Section for shellfish management purposes, even when the area is not classified as prohibited.

Data on shellfishing areas in Massachusetts were available for 2005 through 2015, and the Rhode Island dataset covered 1995 through 2015. For this analysis, although data for Massachusetts were not available prior to 2005, all of its waters within the Bay were closed before that year, so the Estuary Program included all Massachusetts waters in the prohibited classification category from 1995 through 2005. Following direction from the Massachusetts Division of Marine Fisheries, the Estuary Program updated the most current Massachusetts shellfishing growing areas to reflect annual changes and status in Mount Hope Bay, which had been closed to shellfishing from the 1980s to 2009. Because only slight changes occurred from one year to the next, the Estuary Program analyzed the data at five-year intervals to identify changes in acreage of shellfish growing areas. The analysis was conducted for the entire Narragansett Bay and for five regions within the Bay: (1) the Upper Estuary (including Mount Hope Bay, Greenwich Bay and other estuarine waters north of Prudence Island and Aquidneck Island), (2) West Passage, (3) East Passage, (4) Sakonnet River, and (5) the Mouth of the Bay including the Narrow River.

The Estuary Program calculated metrics at five-year intervals between 1995 and 2015 for Narragansett Bay and for regions within the Bay. The metrics for each of the three classification categories–approved, conditionally approved, and prohibited–included the following:

1. Total extent (acres) and percentage of the Bay or region
2. Gross change: total change in area (acres)
3. Percentage net change: change in acreage relative to the previous year

Data from Rhode Island and Massachusetts were reconciled to obtain a seamless shellfish growing area dataset at different time steps. Data provided as paper and digital maps were converted into GIS using heads-up digitizing and geo-referencing to a common coordinate system. The Estuary Program used geospatial tools in the GIS platform (Esri 2016) to cross-tabulate the area and percentage of each category of shellfish growing area by Bay Region at each time step.

It is important to note that while every effort was made to ensure accuracy in digitizing paper and digital maps, the data are imperfect, and it was estimated that any changes of one percent or less between years were likely attributable to digitization error, and therefore they were not interpreted as actual changes. This error was computed by the total area of the Bay and each Bay Region across years. The difference of areas was inherent from the secondary sources due to inconsistency of coastline boundaries from year to year. Other limitations and sources of error for the analysis are derived from the data sources.

Status and Trends

Status

In 2015, approximately 63 percent of the Bay's waters were classified as approved shellfish growing areas, and thirteen percent were conditionally approved. In 24 percent of the Bay, shellfish harvesting for direct human consumption was prohibited (Table 1; Figure 1).

In 2015, the Upper Estuary had by far the lowest percentage (six percent) and acreage (2,249 acres) of approved shellfish growing area, compared to the other four Bay regions (Table 2; Figure 1). The other regions had 68 to 95 percent of their area classified as approved. Conditionally approved areas accounted for 44 percent of the Upper Estuary, in contrast to the other regions, which had one percent or less of their area classified as conditionally approved. Half of the Upper Estuary and 31 percent of the East Passage were classified as prohibited (Table 2; Figure 1).

Looking at the Bay as a whole, most of the prohibited areas (58 percent) and nearly all conditionally approved areas (97 percent) were in the Upper Estuary. Approved areas for shellfishing without conditions were located mainly in the West Passage (29 percent) and Mouth of the Bay (35 percent) (Table 3). However, it should be noted that 35 percent of the approved area of Narragansett Bay consisted of

Table 1. Area (acres) and percentage of shellfish growing areas in Narragansett Bay by category in 2015.

Categories of Shellfish Growing Areas	Total Acreage	Percent of Narragansett Bay
Approved	79,253	63
Conditionally Approved	16,297	13
Prohibited	30,542	24
Narragansett Bay	**126,092**	

Table 2. Area (acres) and percentage of shellfish growing areas by category in Bay regions in 2015.

Bay Region	Area (acres)(1)	Approved		Conditionally Approved		Prohibited	
		Acreage	Percent	Acreage	Percent	Acreage	Percent
Upper Estuary	35,803	2,249	6	15,755	44	17,799	50
West Passage	25,364	22,760	90	387	1	2,217	9
East Passage	20,519	14,029	68	105	0.5	6,385	31
Sakonnet River	13,090	12,429	95	0	0	661	5
Mouth of the Bay	31,052	27,784	90	0	0	3,268	10

(1) Due to inconsistencies between data sources at different time steps, the total of the regional acreages listed here does not equal the total Bay acreage in Table 1 (0.2 percent error).

Table 3. Distribution of shellfishing categories among regions of Narragansett Bay in 2015. The table shows the proportion of each category that is located in each region relative to the total acreage of that category across the entire Bay. [1]

Bay Region	Approved (percentage)	Conditionally Approved (percentage)	Prohibited (percentage)
Upper Estuary	3	97	58
West Passage	29	2	7
East Passage	18	1	21
Sakonnet River	16	0	2
Mouth of the Bay	35	0	11
Narragansett Bay	**100**	**100**	**100**

(1) ((Total acreage of a category in region/Total acreage of that category in Narragansett Bay)*100). See also Table 1.

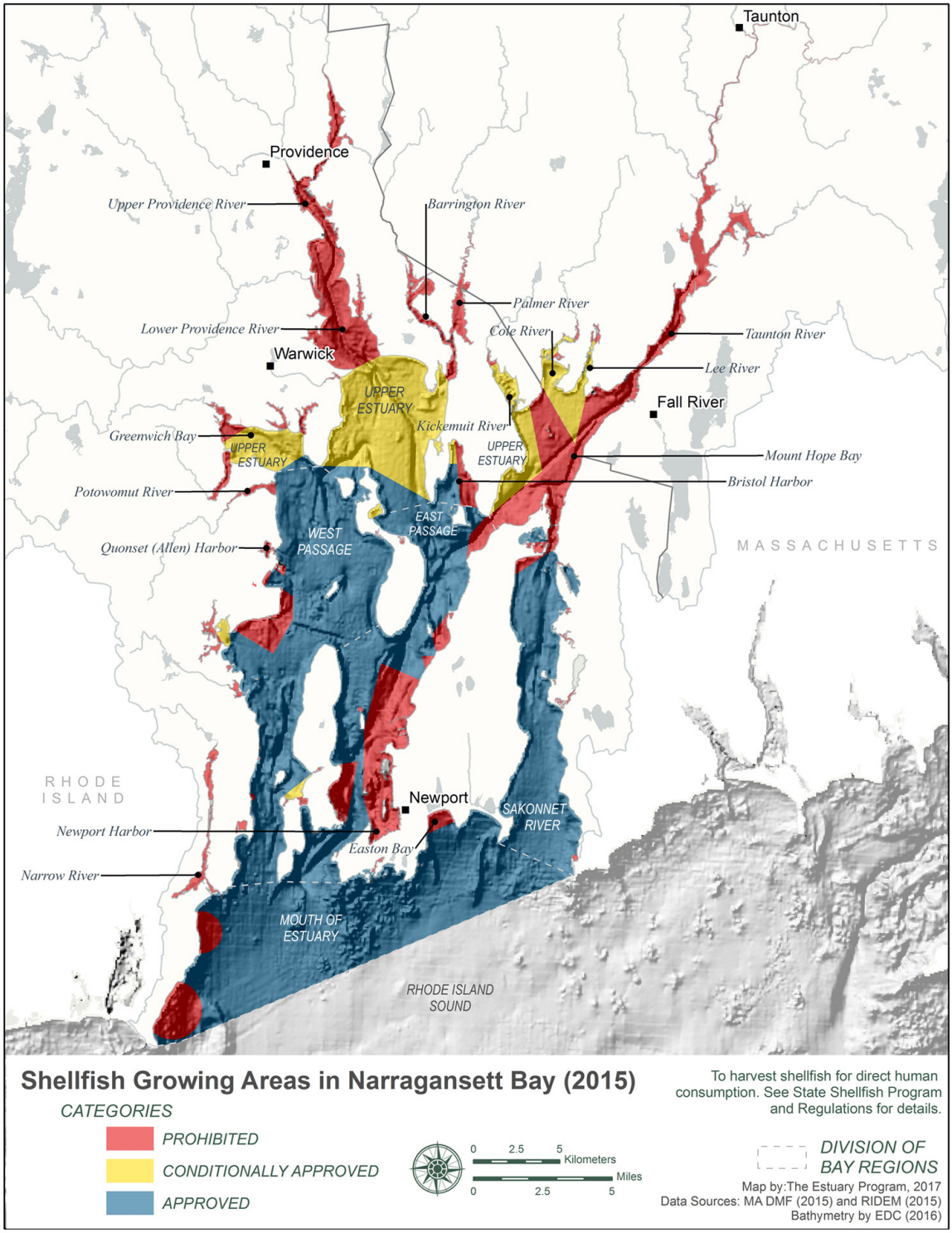


Figure 1. Shellfish growing areas in Narragansett Bay by category in 2015. Labels in capital letters indicate the Bay Regions used for data analysis and reporting. Classification changes that took place in Rhode Island in 2017 in some areas of the Bay are not reflected on this map; see RI Shellfish Harvest Restrictions for most recent updates.

Table 4. Area (acres) and percentage of shellfish growing areas by category in Narragansett Bay at five-year intervals from 1995 to 2015.

Category	Approved		Conditionally Approved		Prohibited	
Year	Acreage	Percent	Acreage	Percent	Acreage	Percent
1995	82,386	66	14,807	12	27,903	22
2000	80,856	65	14,322	11	29,938	24
2005	79,753	63	14,612	12	31,667	25
2010	79,180	63	15,175	12	31,578	25
2015	79,253	63	16,297	13	30,542	24

Table 5. Percent net change of shellfish growing areas by category in Narragansett Bay at five-year intervals from 1995 to 2015.

Category	Approved	Conditionally Approved	Prohibited
Time-step	Percent Change	Percent Change	Percent Change
1995–2000	-1.9	-3.3	7.3
2000–2005	-1.4	2.0	5.8
2005–2010	-0.7	3.8	-0.3
2010–2015	0.1	7.4	-3.3

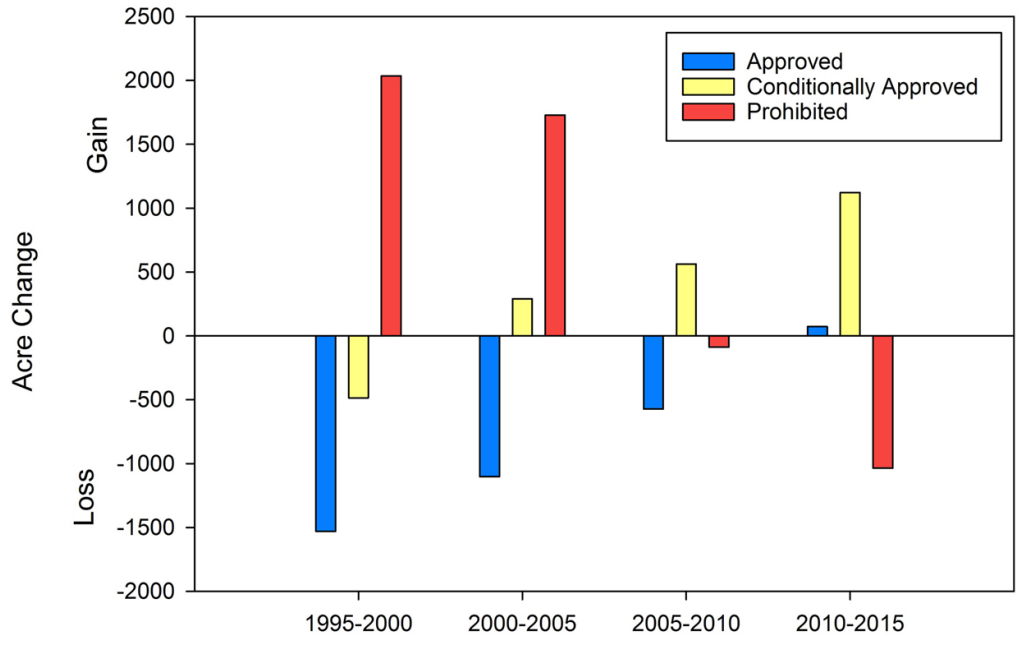


Figure 2. Acreage change in shellfish growing areas by category in Narragansett Bay at five-year intervals.

areas where little or no shellfishing occurs, including some areas within these two regions. Those areas included deeper waters (south of Aquidneck Island and north of the line between Point Judith and Sakonnet Point), regulated navigation channels, and other precautionary areas. For this analysis, these areas were included as approved–and included in the total approved area of 79,253 acres–because data were not readily available to distinguish areas of the Bay that had good water quality but were not suitable for harvest for other reasons.

Trends for Narragansett Bay

The Estuary Program calculated the acreages and percentages of the three categories of shellfishing growing areas in the entire Narragansett Bay at five-year intervals between 1995 and 2015 (Table 4; Figure 2). The relative amount of each shellfishing growing area by category in the Bay changed only slightly between time-steps (Table 4). Figure 2 shows changes in the total acreage in each category, and Table 5 shows the net percent change.

The approved shellfishing areas varied only slightly within each time-step when calculating acreage across the Bay. However, there were reductions in prohibited areas from 2005 to 2015 and increases in conditionally approved areas (Table 5). Table 6 shows where these changes occurred in Bay regions.

Trends in Regions of Narragansett Bay

The Estuary Program also calculated the amount of change within each of the five Bay regions at five-year intervals from 1995 to 2015 (Table 6). Of note, the

Table 6. Changes in shellfish growing areas (acreage and percent net change) by category in Narragansett Bay regions at five-year intervals.

Bay Regions	Category of Shellfish Growing Area					
	Approved		Conditionally Approved		Prohibited	
Five-year Interval	Acres	*Percent*	Acres	*Percent*	Acres	*Percent*
Upper Estuary						
1995–2000	0	*0*	-488	*-3.4*	518	*2.8*
2000–2005	32	*1.5*	264	*1.9*	281	*1.5*
2005–2010	-2	*-0.1*	623	*4.4*	-650	*-3.3*
2010–2015	24	*1.1*	1099	*7.5*	-1095	*-5.8*
West Passage						
1995–2000	0	*0*	2	*0.4*	0	*0*
2000–2005	-238	*-1.0*	23	*5.4*	322	*21.7*
2005–2010	-238	*-1.0*	-61	*-13.5*	429	*23.8*
2010–2015	-116	*-0.5*	-2	*-0.4*	-16	*-0.7*
East Passage						
1995–2000	-1,530	*-10*	0	*0*	1,517	*29.8*
2000–2005	162	*1.2*	8	*8.7*	-53	*-0.8*
2005–2010	0	*0*	0	*0*	0	*0*
2010–2015	162	*1.2*	0	*0*	-164	*-2.5*
Sakonnet River						
1995–2000	0	*0*	0	*0*	0	*0*
2000–2005	0	*0*	-5	*-17*	47	*9.7*
2005–2010	-301	*-2.4*	0	*-0.2*	99	*18.5*
2010–2015	-3	*0*	-25	*-100*	27	*4.2*
Mouth of the Bay						
1995–2000	0	*0*	0	*0*	0	*0*
2000–2005	-1,059	*-3.7*	0	*0*	1,132	*53.8*
2005–2010	-32	*-0.1*	0	*0*	32	*1.0*
2010–2015	3	*0*	0	*0*	0	*0*

Upper Estuary from 2010 to 2015 had an increase of approximately 1,100 acres in conditionally approved areas, as prohibited areas decreased by approximately the same amount (Table 6; Figure 3).

Discussion

The most noteworthy changes in shellfishing areas in Narragansett Bay occurred in the Upper Estuary over the last decade. Acreage of conditionally approved growing areas began to increase in 2000 and continued this trend through 2015, while prohibited areas declined at a comparable rate from 2005 through 2015 (Figure 3). The greatest percent change in the Upper Estuary occurred between 2010 and 2015 when conditionally approved areas gained 7.5 percent and prohibited areas declined by 5.8 percent; this change was twice as large and opposite the change that occurred from 1995 to 2000, when conditionally approved areas declined 3.4 percent and prohibited areas increased 2.8 percent (Table 6). These changes in the Upper Estuary suggest an improvement in water quality conditions. Ninety-seven percent of the 16,297 acres of conditionally approved areas in the entire Narragansett Bay occurred in the Upper Estuary (Table 3).

While the shift from prohibited areas to conditional areas indicated improvement, the Upper Estuary still had a very limited area—only six percent—that was fully approved without conditions as of 2015 (Table 2). At that time, the Upper Estuary's approved areas were limited primarily to waters at the northern portions of Prudence Island (Figure 1). However, as this chapter was being finalized in spring 2017, RIDEM announced that restrictions for portions of the upper Bay were lifted for the first time since 1947, with 3,712 acres of conditionally approved areas upgraded to approved status. This area encompassed upper Bay waters between Warwick Point and Colt State Park in Bristol, north of Prudence Island. This is an extremely positive indicator of Narragansett Bay water quality improvements, considering that this area of the upper Bay had not had a significant increase in waters with an approved status in decades (Table 4; Figure 3).

Of the 30,542 acres where shellfish harvesting was prohibited in Narragansett Bay in 2015, almost 60 percent (17,799 acres) were in the Upper Estuary (Table 2 and 3), where water quality impairments were likely due to loadings from Providence, Fall River, and surrounding urban areas (Figure 1). It is important to note that the prohibited areas in the Estuary Program's analysis included waters with a restricted classification by MADMF, at the mouth of the Taunton River. An upgrade from prohibited to restricted classification indicates water quality improvements in this area (MADMF 2013), but it is still in the classification of not suitable for harvesting shellfish for human consumption. Shellfish in these areas may be allowed for removal to grow out in approved areas.

While the majority of prohibited areas (17,799 acres) were located in the Upper Estuary, large prohibited areas also occurred in other Bay regions. These classifications were not related exclusively to documented water quality concerns. Some of them resulted from National Shellfish Sanitation Program prohibitions on shellfishing in waters near marinas, wastewater treatment facility outfalls, and other regulated areas that are potential sources of contamination by pathogens and other pollutants.

In addition, the East Passage included 6,385 acres of prohibited areas, and much of that acreage was located in and around Newport Harbor (Table 2; Figure 1), an area where shellfishing was prohibited for safety reasons because of the high concentration of commercial, industrial, and military activities. The largest percent change of acreage in the East Passage occurred between 1995 and 2000 with a 30 percent increase in prohibited areas for shellfishing (Table 6). Similarly, a prohibited area was established in the West Passage for safety reasons around the Quonset Business Park (Figure 1). Other prohibited areas in the West Passage were based on water quality concerns, including smaller enclosed areas such as the Potowomut River in Greenwich Bay and Sheffield Cove in Jamestown. Most the changes to prohibited areas in the West Passage region occurred between 2000 and 2010, as prohibited areas increased over 20 percent within each five-year interval (Table 6). Water quality issues were the reason for the prohibited areas in the Sakonnet River, including the northern section near Tiverton and Portsmouth, as well as in small coves and embayments; the greatest change occurred from 2005 to 2010 when prohibited areas increased by 18.5 percent (Table 6). In the Mouth of the Bay, prohibited waters have been located in the Narrow River, at two safety zones around sewer outfalls on the western shore, and by 2005 Easton's Beach in Newport also was downgraded to prohibited status, along with an increase of area of the two safety zones.

Urban landscapes and combined sewer overflows are prime contributors of pathogen loadings to the Bay (see "Wastewater Infrastructure" chapter), but there are many sources to consider, including stormwater runoff, septic systems, and waterfowl, among others. In many instances, stormwater from rainfall triggers temporary closures. This varies by growing area and

by criteria set forth by the state shellfish programs (RIDEM 2015, MADMF personal communication 2017). Within a suburban coastal embayment in nearby Buzzards Bay, investigators found significant loading of fecal coliform from storm drains and small streams during rain events (Weiskel et al. 1996). Others report that septic systems can be a source of pathogens to coastal waters (e.g., Lipp et al. 2001), although Weiskel and colleagues (1996) found that the bacteria were substantially attenuated during transport to coastal waters via groundwater. As progress is made to address fecal coliform contamination from point sources and nonpoint sources, additional upgrades from prohibited to conditionally approved shellfish growing areas can be expected. While the oyster fishery appears to be returning and quahog landings have fluctuated historically (Oviatt et al. 2003), these industries are important to the Bay's economy (Schumann 2015), and opening prohibited areas or upgrading conditionally approved areas to approved would be beneficial to the fishery.

Improvements in water quality in the Upper Estuary, including the Providence River estuary and Mount Hope Bay, started to be evident from a substantial increase of conditionally approved areas and decline of prohibited acreage from 2005 through 2015, most likely due to upgrades in the wastewater treatment plant facilities and combined sewer overflow (CSO) abatement efforts. Tunnels for temporary storage of stormwater and untreated waste, installed by the Narragansett Bay Commission and the City of Fall River, have significantly reduced combined sewer overflows. In recent years, RIDEM's Office of Water Resources has conducted targeted post-storm monitoring in the upper Bay to evaluate water quality improvements resulting from the completion of two phases of the Narragansett Bay Commission's (NBC) combined sewer overflow abatement project. Water quality data from NBC's monitoring program show an overall decrease of 24 percent in fecal coliform concentrations in the lower Providence River at all stations downstream of the CSOs post-Phase I (NBC 2016). An agreement between RIDEM and the United States Food and Drug Administration allows the state to re-open the upper Bay's conditionally approved areas to shellfish harvesting as soon as post-storm monitoring data demonstrate that it is safe to do so.

Though not well documented, considerable improvements of water quality in Mount Hope Bay have also been attributed to CSO abatement. Most

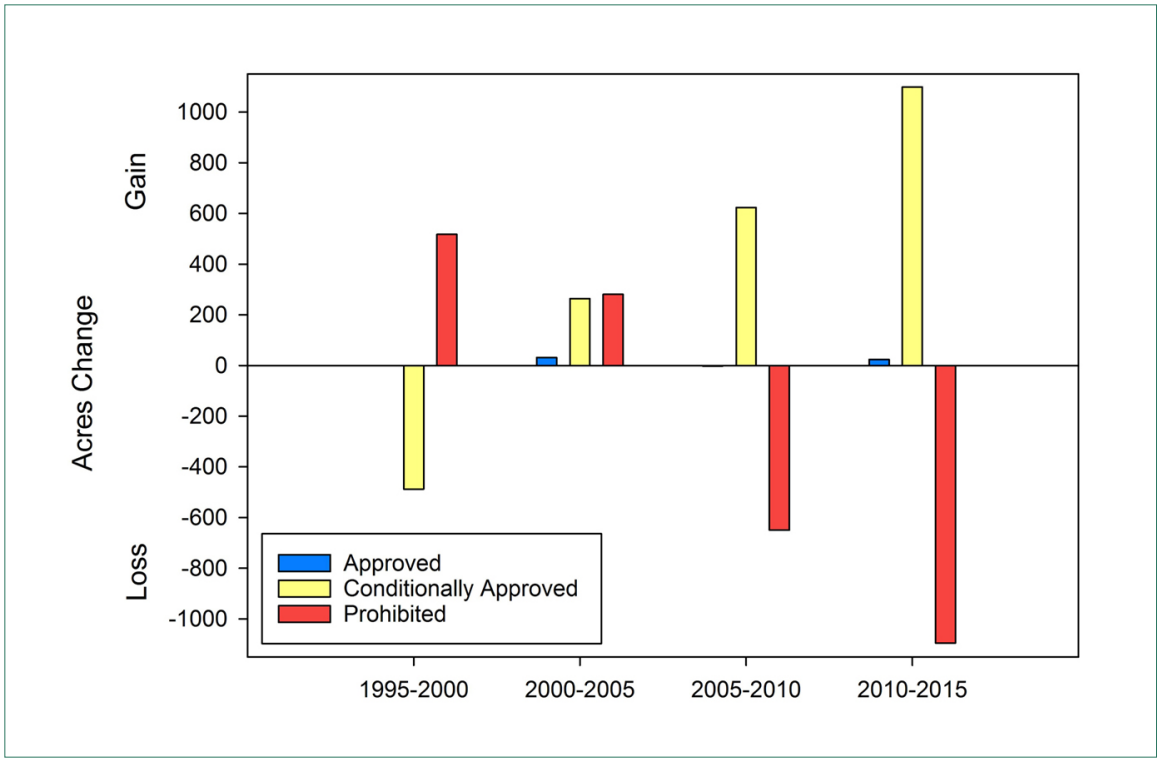


Figure 3. Acreage change in shellfish growing areas by category in the Upper Estuary region at five-year intervals. Bars represent the magnitude of acreage gain or loss by time-steps.

of the upgrades of prohibited areas to conditionally approved areas between 2005 and 2015 occurred in waters of Mount Hope Bay.

Overall, the major improvements in stormwater and sewer infrastructure, which were conducted to reduce pathogen loadings to the Bay, have been linked to upgrades in classification of shellfish growing areas in the Upper Estuary and also to changes in wet weather criteria (e.g., magnitude of precipitation events) as a major factor used to determine the status of conditionally approved areas. However, quantitative analyses to correlate factors that trigger closure of conditionally approved areas and water quality improvements such as reduced pathogen loadings resulting from infrastructure improvements have not been conducted or published to date.

Changes of classification for the shellfish growing areas remain driven by water quality and shoreline survey data, as well as management actions. While the shellfish indicator used by the Estuary Program in this chapter measures areas defined by three classification categories for growing areas, changes in classification within the conditional approved status can also imply water quality improvements, as areas can be upgraded within the same classification, but with different criteria. In addition to the 2017 newly approved areas in the upper Bay in Rhode Island, two conditionally approved areas—Conimicut Triangle and Conditional Area A (Figure 4)—were merged. Previously, these areas would be closed after 0.5 inches and 0.8 inches of rain, respectively (RIDEM 2017; see RI Shellfish Harvest Restrictions online map). Now this area of approximately of 5,980 acres will be closed after 1.2 inches of rain—a significantly higher threshold. Even though the shellfish growing area status has been kept as conditionally approved, the change in wet weather criteria is expected to result in an increased number of days when the waters can be opened for harvesting shellfish (RIDEM 2017). No changes in shellfish classification have occurred in Mount Hope Bay or Palmer River since 2015 (MADMF personal communication 2017).

RIDEM also indicated that the lower portion of the Providence River holds potential as a new conditionally approved area, perhaps implemented when the NBC completes Phase III of the CSO abatement project (RIDEM 2017). Likewise, these improvements could influence marine beaches and recreational uses of the Bay (see "Marine Beaches" and "Water Quality Conditions for Recreation" chapters), which are also evaluated using pathogen concentrations as a metric and have similar stressors as the shellfishing areas.



Figure 4. Shellfishing near Conimicut Light in the Upper Estuary. The most recent changes in the Upper Estuary, as of 2017, occurred south of this location, where 5,980 acres of shellfishing growing areas were merged with a higher rainfall threshold, and 3,711 acres were upgraded from conditionally approved to approved status for shellfishing as mandated by RIDEM. Photo credit: Eivy Monroy.

In addition to high concentrations of pathogens, there are other reasons for which areas of the Bay may be closed to shellfishing. Approved and conditionally approved shellfish growing areas can be closed on an emergency basis due to harmful algal blooms (HAB). In October 2016, a shellfish closure was triggered in Narragansett Bay by a harmful algal bloom of the phytoplankton *Pseudo-nitzschia*spp. RIDEM conducted an intensive program of phytoplankton sampling and shellfish meats collection. Approximately 176 plankton samples and 150 shellfish samples were tested for domoic acid, the toxin produced by *Pseudo-nitzschia* cells that causes amnesic shellfish poisoning. A similar closure, lasting three weeks, occurred in March 2017. In response to the state's first *Pseudo-nitzschia* bloom in October 2016, RIDEM in coordination with the RIDOH is revising its Harmful Algal Bloom Monitoring and Contingency Response Plan to include more frequent routine phytoplankton monitoring to detect HAB blooms. When blooms are detected, the revised plan calls for more intensive monitoring of phytoplankton and shellfish to track spatial extent and intensity of the bloom and the presence of biotoxins, among other provisions (RIDEM personal communication 2017). In addition, Rhode Island Sea Grant is funding research to understand this concerning phenomenon.

Data Gaps and Research Needs

- Conditionally approved areas are monitored frequently, but fewer data are available for prohibited areas. Additional sampling in certain areas may be needed to better document progress of these areas toward water quality improvement goals.
- Synthesis of existing data and development of site-specific models would improve understanding of relationships among land use, point and non-point sources, and bacterial concentrations in receiving waters.
- Recent changes showing a decline of prohibited areas and an increase of approved and conditionally approved areas in the Upper Estuary have been attributed to wastewater and other infrastructure improvements to capture and treat combined sewage and stormwater overflows, and other pollution control efforts.
- Refinement of this indicator using pathogen data could provide a metric more sensitive to water quality improvements, such as by discerning partial progress toward water quality goals.
- Post-storm monitoring of the Mount Hope Bay and Kickemuit River estuary is needed to evaluate water quality improvements associated with construction of the Fall River CSO control structures, and to revise rainfall thresholds that trigger closures of these conditionally approved waters.

Acknowledgments

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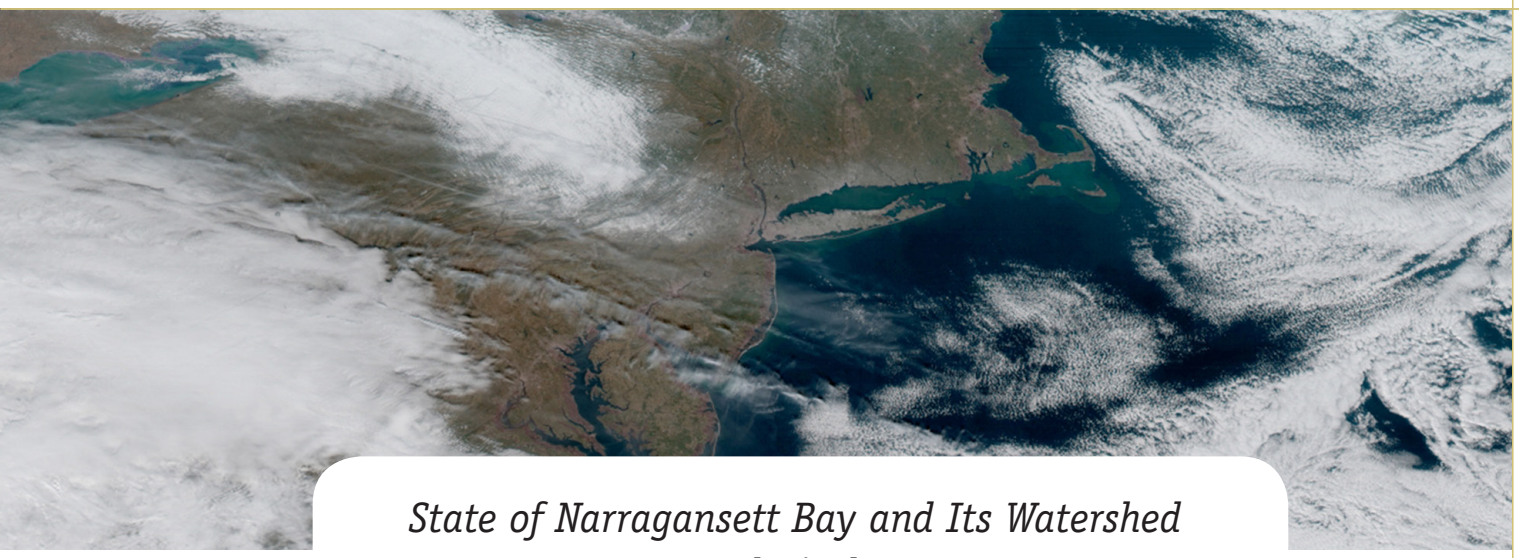
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State of Narragansett Bay and Its Watershed
2017 Technical Report

Appendix

A GUIDE TO THE GEOGRAPHICAL UNITS REFERENCED IN THE TECHNICAL REPORT

Please use the following citation: Narragansett Bay Estuary Program. 2017. State of Narragansett Bay and Its Watershed (Appendix, pages 474-495). Technical Report. Providence, RI.

Photo: Narragansett Bay from Space (NASA)

NARRAGANSETT BAY ESTUARY PROGRAM

Table A-1. Total Area of Narragansett Bay and Its Watershed. The Narragansett Bay Estuary Program defines the outer boundary of Narragansett Bay as a line between Point Judith in the west and Sakonnet Point in the east. The upper boundaries of the Bay's estuarine waters correspond to the limits of tidal waters defined by: 1) the presence of existing dams and 2) the extent of estuarine waters as defined and delineated by the states to assess water quality and shellfishing areas. The Narragansett Bay Watershed is the land area that drains into the Bay. The Watershed area reported in this table is based on HUC10 watershed boundaries delineated by the U.S. Geological Survey (Table A-3). In this report, the capitalized term "Watershed" refers to the entire Narragansett Bay Watershed, and the lowercase term "watershed" refers to HUC10 watersheds.

	Area			Miles of streams[1]
	Acres	Square Miles	Square Kilometers	
Narragansett Bay	125,399	196	507.5	
Narragansett Bay Watershed	1,091,112	1,705	4,416	3,578

(1) Total miles of streams was calculated using the U.S. Geological Survey National Hydrography Dataset (NHD) High Resolution. Within NHD Flowline-codes, the Estuary Program excluded features that were not related to natural streams or rivers such as coastline, pipeline, underground conduit, and canal/ditch. The processing dates of the data by NHD range from 2004 through 2015, where features were created, validated updated, or corrected from previous versions. Metadata description: "The National Hydrography Dataset (NHD) is a feature-based database that interconnects and uniquely identifies the stream segments or reaches that make up the nation's surface water drainage system. This high-resolution NHD, generally developed at 1:24,000/1:12,000 scale. NHD provides a nationally consistent framework for addressing and analysis, water-related information linked to reach addresses by one organization (national, state, local) can be shared with other organizations and easily integrated into many different types of applications to the benefit of all."

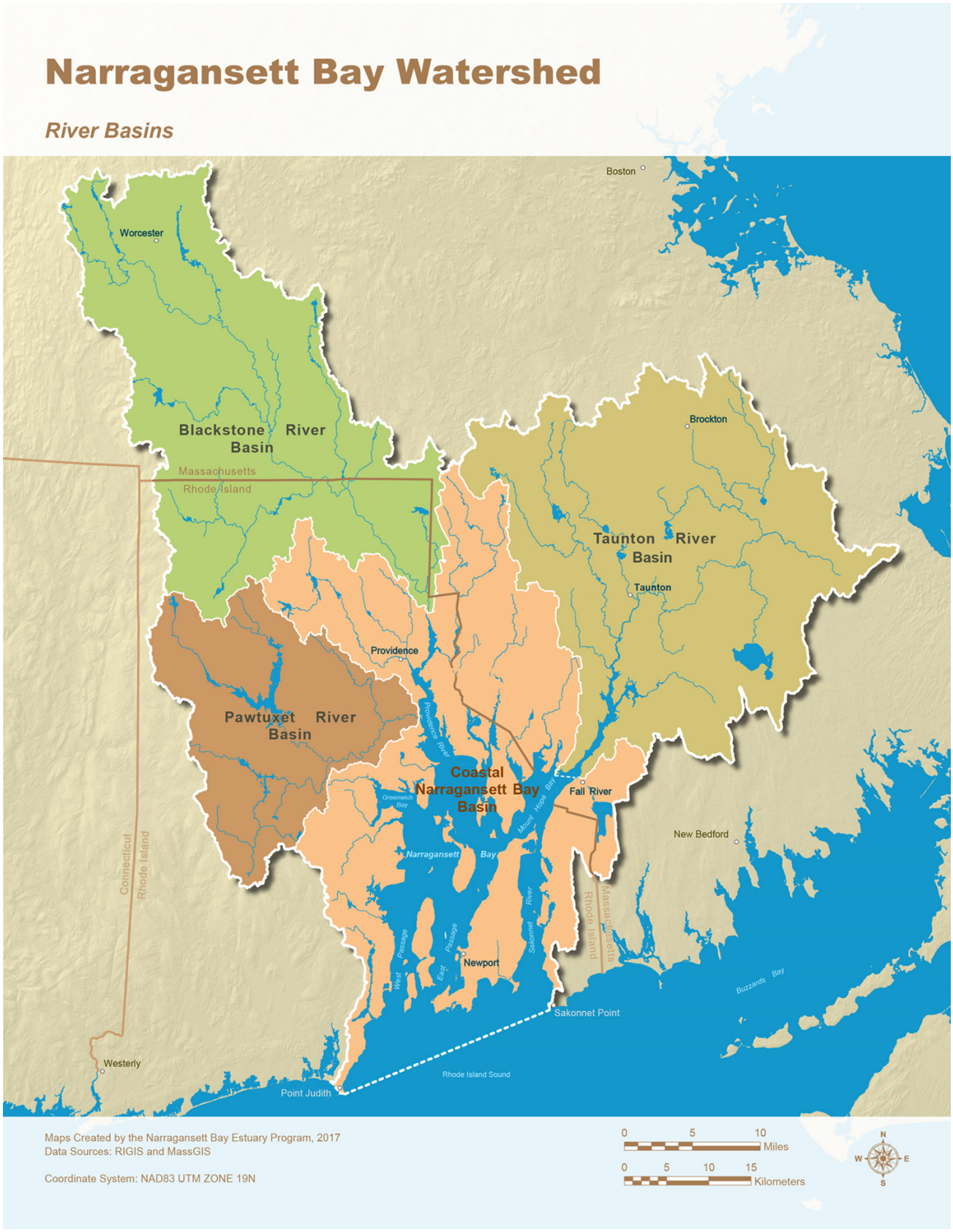


Figure A-1. Narragansett Bay Watershed - River Basins

Table A-2. Narragansett Bay Watershed – River Basins. The Estuary Program for its data analysis and reporting defined four River Basins within the Narragansett Bay Watershed. Three of the River Basins are defined as the land areas drained by the three major rivers and their tributaries. The fourth River Basin is the Coastal Narragansett Bay Basin, which encompasses all land that drains directly into Narragansett Bay, rather than into the three major rivers. The River Basins are not part of the U.S. Geological Survey Hydrologic Unit Code hierarchical framework (Tables A-3 and A-4) and were delineated instead based on Watershed Planning Areas (WPAs) (Table A-5).

River Basin Name[1]	Acres	Square Miles	Square Kilometers	Total Stream Miles
Blackstone River	303,734	475	1,229	1,137
Pawtuxet River	148,404	232	601	444
Taunton River	338,871	530	1,371	1,170
Coastal Narragansett Bay	300,124	469	1,215	827

[1] Because River Basins are defined by the delineation of Watershed Planning Areas (WPAs) (Table A-5), a small section of the Coastal Narragansett Bay Basin lies outside the extent of Narragansett Bay Watershed as defined for Table A-1. Therefore, the total Watershed area (1,705 square miles) listed in Table A-1 differs slightly from the total of the River Basin areas (1,706 square miles) listed in Table A-2. This discrepancy is due to the delineation by RIDEM of the Narrow River WPA, which differs from the delineation by USGS.

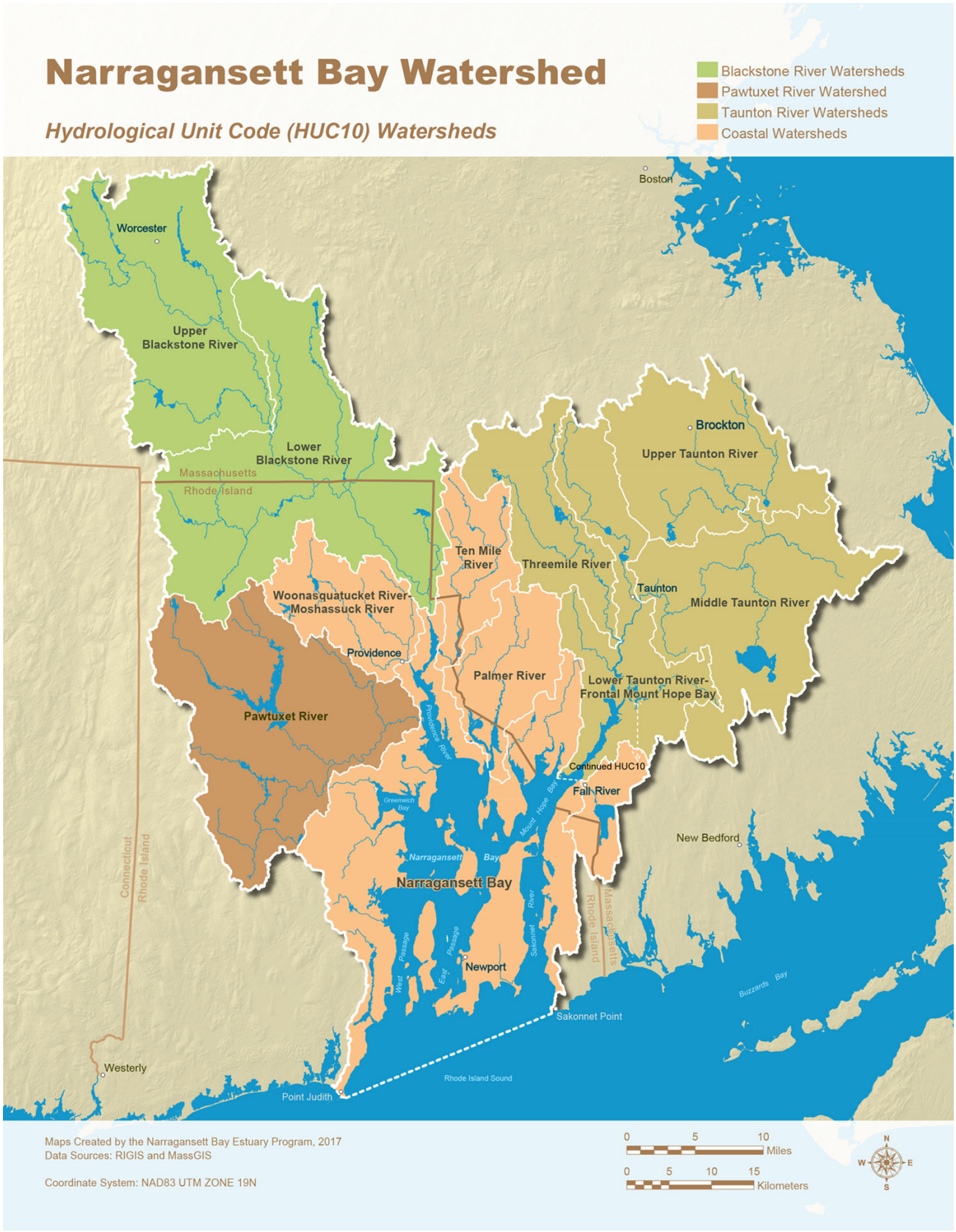


Figure A-2. Narragansett Bay Watershed - Hydrological Unit Code (HUC10) Watersheds

Table A-3. Narragansett Bay Watershed – Hydrological Unit Code (HUC10) Watersheds. Established by the U.S. Geological Survey, Hydrologic Unit Codes (HUCs) are a standardized hierarchical classification framework in which watersheds have ten-digit codes (HUC10s) and the subwatersheds nested within watersheds have twelve-digit codes (HUC12s). This table provides information about the eleven HUC10 watersheds that together form the Narragansett Bay Watershed.

HUC10 Watershed Name	10-digit Hydrologic Unit Code	Acres	Square Miles	Square Kilometers	Total Stream Miles
Upper Blackstone River	0109000301	131,421	205	532	516
Lower Blackstone River	0109000302	172,310	269	697	621
Upper Taunton River	0109000401	88,025	138	356	305
Middle Taunton River	0109000402	114,955	180	465	392
Threemile River	0109000403	54,568	85	221	205
Ten Mile River	0109000404	35,622	56	144	127
Woonasquatucket River-Moshassuck River	0109000405	47,489	74	192	149
Pawtuxet River[1]	0109000406	148,404	232	601	444
Palmer River	0109000407	43,299	68	175	172
Lower Taunton River-Frontal Mount Hope Bay[2]	0109000408	100,593	157	407	298
Narragansett Bay	0109000409	154,426	241	625	350

[1] The Pawtuxet River HUC10 watershed comprises the same area as the Pawtuxet River Basin (Table A-2).

[2] Not all of this HUC10 watershed is included in the Taunton River Basin (Table A-2), as River Basins are delineated based on Watershed Planning Areas (WPAs) (Table A-5) rather than HUC10 watersheds.

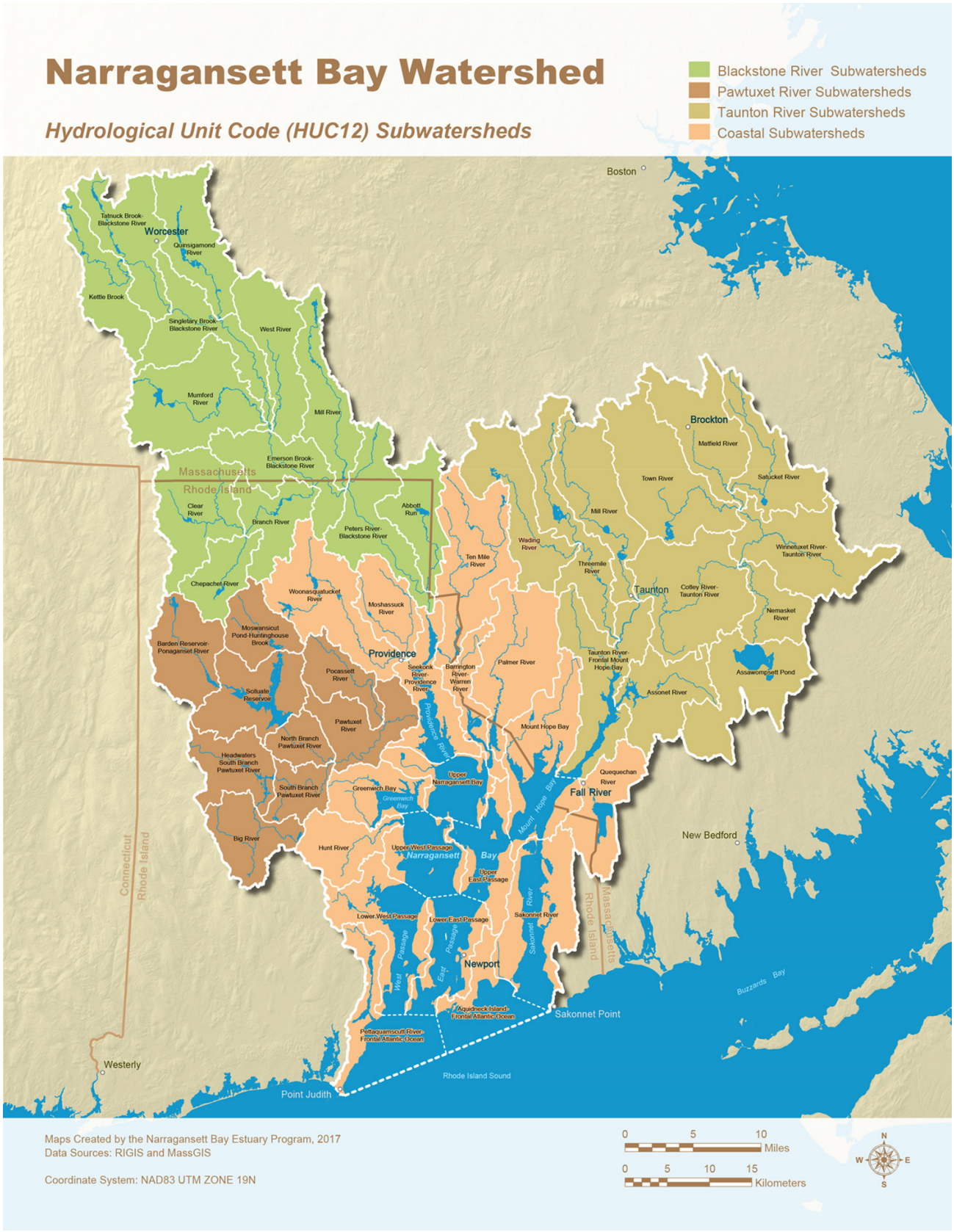


Figure A-3. Narragansett Bay Watershed - Hydrological Unit Code (HUC12) Subwatersheds

Table A-4. Narragansett Bay Watershed – Hydrological Unit Code (HUC12) Subwatersheds. Established by the U.S. Geological Survey, Hydrologic Unit Codes (HUCs) are a standardized hierarchical classification framework in which watersheds have ten-digit codes (HUC10s) and the subwatersheds nested within watersheds have twelve-digit codes (HUC12s). This table provides information about the 52 HUC12 subwatersheds within the Narragansett Bay Watershed.

HUC12 Subwatershed Name	12-digit Hydrologic Unit Code	Acres	Square Miles	Square Kilometers	Total Stream Miles
Kettle Brook	010900030101	20,866	32.6	84.4	100.6
Tatnuck Brook-Blackstone River	010900030102	24,827	38.8	100.5	70.4
Quinsigamond River	010900030103	24,274	37.9	98.2	84.8
Mumford River	010900030104	35,969	56.2	145.6	141.3
Singletary Brook-Blackstone River	010900030105	25,485	39.8	103.1	119.2
West River	010900030201	23,712	37.1	96.0	100.1
Clear River	010900030202	29,035	45.4	117.5	105.8
Chepachet River	010900030203	13,697	21.4	55.4	43.9
Branch River	010900030204	17,025	26.6	68.9	63.7
Mill River	010900030205	21,741	34.0	88.0	76.1
Emerson Brook-Blackstone River	010900030206	21,934	34.3	88.8	77.8
Abbott Run	010900030207	18,317	28.6	74.1	71.7
Peters River-Blackstone River	010900030208	26,848	41.9	108.6	81.7
Satucket River	010900040101	22,421	35.0	90.7	78.2
Matfield River	010900040102	26,766	41.8	108.3	85.2
Town River	010900040103	38,838	60.7	157.2	141.3
Winnetuxet River-Taunton River	010900040201	31,583	49.3	127.8	137.9
Assawompsett Pond	010900040202	31,479	49.2	127.4	86.4
Nemasket River	010900040203	13,416	21.0	54.3	45.7
Cotley River-Taunton River	010900040204	38,477	60.1	155.7	122.1
Wading River	010900040301	27,921	43.6	113.0	95.4
Threemile River	010900040302	26,647	41.6	107.8	109.8
Ten Mile River[1]	010900040401	35,622	55.7	144.2	126.6
Moshassuck River	010900040501	14,758	23.1	59.7	41.5
Woonasquatucket River	010900040502	32,730	51.1	132.5	107.6
Big River	010900040601	18,144	28.4	73.4	43.8
Headwaters South Branch Pawtuxet River	010900040602	17,855	27.9	72.3	53.2
South Branch Pawtuxet River	010900040603	10,648	16.6	43.1	27.8
Moswansicut Pond-Huntinghouse Brook	010900040604	14,196	22.2	57.5	43.8
Barden Reservoir-Ponaganset River	010900040605	21,068	32.9	85.3	76.0
Scituate Reservoir	010900040606	24,796	38.7	100.3	82.1
North Branch Pawtuxet River	010900040607	9,011	14.1	36.5	31.2
Pocassett River	010900040608	13,158	20.6	53.2	39.8
Pawtuxet River[2]	010900040609	19,528	30.5	79.0	46.0
Palmer River	010900040701	32,575	50.9	131.8	135.9
Barrington River-Warren River	010900040702	10,724	16.8	43.4	35.8

Table A-4 continued.

HUC12 Subwatershed Name	12-digit Hydrologic Unit Code	Acres	Square Miles	Square Kilometers	Total Stream Miles
Mill River	010900040801	28,841	45.1	116.7	122.5
Assonet River	010900040802	21,819	34.1	88.3	55.4
Quequechan River	010900040803	19,272	30.1	78.0	29.7
Taunton River-Frontal Mount Hope Bay	010900040804	30,661	47.9	124.1	90.3
Seekonk River-Providence River	010900040901	15,135	23.6	61.2	15.6
Upper Narragansett Bay	010900040902	6,418	10.0	26.0	14.4
Greenwich Bay	010900040903	13,394	20.9	54.2	30.7
Hunt River	010900040904	15,630	24.4	63.3	44.6
Mount Hope Bay	010900040905	25,864	40.4	104.7	73.3
Upper West Passage	010900040906	5,171	8.1	20.9	6.7
Upper East Passage	010900040907	9,503	14.8	38.5	16.4
Lower West Passage	010900040908	17,152	26.8	69.4	31.2
Lower East Passage	010900040909	6,982	10.9	28.3	5.6
Sakonnet River	010900040910	22,387	35.0	90.6	80.0
Aquidneck Island-Frontal Atlantic Ocean	010900040911	5,936	9.3	24.0	9.9
Pettaquamscutt River-Frontal Atlantic Ocean	010900040912	10,854	17.0	43.9	21.8

(1) The Ten Mile River HUC12 subwatershed comprises the same area as the Ten Mile River HUC10 watershed (Table A-3).
(2) Because HUC12 subwatersheds are nested in the HUC10 watersheds, this subwatershed is defined as the immediate area draining to the Pawtuxet River, emptying into the Bay.

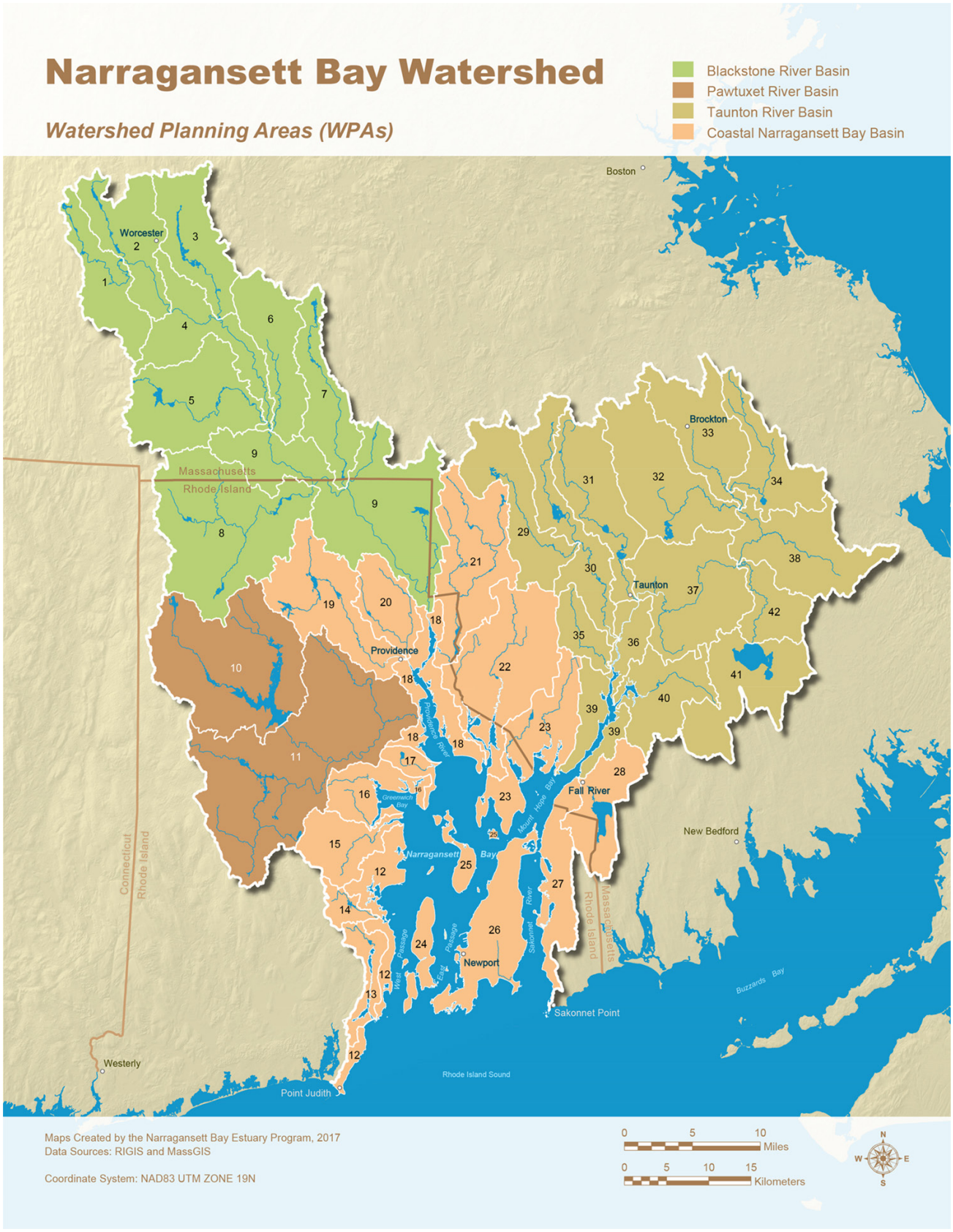


Figure A-4. Narragansett Bay Watershed - Watershed Planning Areas (WPAs)

Table A-5. Narragansett Bay Watershed – Watershed Planning Areas (WPAs). The Rhode Island Department of Environmental Management (RIDEM)[1] and Massachusetts Department of Environmental Protection (MassDEP)[2] defined WPAs for the purpose of developing watershed management plans. There are 42 WPAs within the Narragansett Bay Watershed, and the WPAs are nested within River Basins (Table A-2). Each WPA encompasses one or more HUC12 subwatersheds, but some WPAs are not consistent with HUC12 drainage boundaries (e.g., Aquidneck Island). RIDEM and MassDEP have developed approaches to support watershed management plans that consider the geographical location, in some cases at varying scales, and the work conducted by different partners within the Watershed. In this report, indicator results are reported for WPAs to provide information that supports these efforts.[3]

River Basin	Watershed Planning Area [4] Name	ID Index	Area of the Watershed Planning Area[1]			Total Stream Miles
			Acres	Square Miles	Square Kilometers	
Blackstone River	Kettle Brook	1	20,866	32.6	84.4	100.6
Blackstone River	Tatnuck Brook-Blackstone River	2	24,827	38.8	100.5	70.4
Blackstone River	Quinsigamond River	3	24,274	37.9	98.2	84.8
Blackstone River	Singletary Brook-Blackstone River	4	25,485	39.8	103.1	119.2
Blackstone River	Mumford River	5	35,969	56.2	145.6	141.3
Blackstone River	West River	6	23,712	37.1	96.0	100.1
Blackstone River	Mill River - Blackstone River	7	21,741	34.0	88.0	76.1
Blackstone River	Branch River	8	59,758	93.4	241.8	213.3
Blackstone River	Lower Blackstone River	9	67,103	104.8	271.6	231.3
Pawtuxet River	Scituate Reservoir-Pawtuxet	10	60,060	93.8	243.1	201.8
Pawtuxet River	Pawtuxet River	11	88,345	138.0	357.5	241.8
Coastal Narragansett Bay	West Bay	12	14,581	22.8	59.0	24.9
Coastal Narragansett Bay	Narrow River	13	8,522	13.3	34.5	19.0
Coastal Narragansett Bay	Annaquatucket River	14	4,694	7.3	19.0	9.3
Coastal Narragansett Bay	Hunt River	15	15,648	24.4	63.3	44.6
Coastal Narragansett Bay	Greenwich Bay	16	13,410	21.0	54.3	30.9
Coastal Narragansett Bay	Buckeye Brook	17	4,500	7.0	18.2	10.4
Coastal Narragansett Bay	Providence - Seekonk Rivers	18	15,809	24.7	64.0	16.9
Coastal Narragansett Bay	Woonasquatucket River	19	32,730	51.1	132.5	107.6
Coastal Narragansett Bay	Moshassuck River	20	14,758	23.1	59.7	41.5
Coastal Narragansett Bay	Ten Mile River	21	35,622	55.7	144.2	126.6
Coastal Narragansett Bay	Barrington - Palmer - Warren Rivers	22	43,314	67.7	175.3	171.7
Coastal Narragansett Bay	Bristol - Kickemuit River	23	25,071	39.2	101.5	73.3
Coastal Narragansett Bay	Jamestown	24	6,049	9.5	24.5	4.6
Coastal Narragansett Bay	Prudence Island	25	3,940	6.2	15.9	5.2
Coastal Narragansett Bay	Aquidneck Island	26	24,529	38.3	99.3	44.6
Coastal Narragansett Bay	Sakonnet - East	27	17,676	27.6	71.5	66.3
Coastal Narragansett Bay	Stafford Pond	28	19,272	30.1	78.0	29.7
Taunton River	Wading River	29	27,921	43.6	113.0	95.4
Taunton River	Threemile River	30	26,647	41.6	107.8	109.8
Taunton River	Mill River	31	28,841	45.1	116.7	122.5

Table A-5 continued.

River Basin	Watershed Planning Area [4] Name	ID Index	Area of the Watershed Planning Area[1]			Total Stream Miles
			Acres	Square Miles	Square Kilometers	
Taunton River	Town River	32	38,838	60.7	157.2	141.3
Taunton River	Matfield River	33	26,766	41.8	108.3	85.2
Taunton River	Satucket River	34	22,421	35.0	90.7	78.2
Taunton River	Segreganset River	35	9,399	14.7	38.0	35.9
Taunton River	Taunton River	36	8,503	13.3	34.4	23.7
Taunton River	Cotley River-Taunton River	37	38,477	60.1	155.7	122.1
Taunton River	Winnetuxet River-Taunton River	38	31,583	49.3	127.8	137.9
Taunton River	Mount Hope Bay	39	12,762	19.9	51.6	30.7
Taunton River	Assonet River	40	21,819	34.1	88.3	55.4
Taunton River	Assawompsett Pond	41	31,479	49.2	127.4	86.4
Taunton River	Nemasket River	42	13,416	21.0	54.3	45.7

[1] RIDEM has defined Watershed Planning Areas for the purpose of developing watershed management plans.
[2] MassDEP and Geosyntec Consultants, Inc., developed a Massachusetts Watershed-Based Plans online tool to assist users in the process of preparing these plans at varying geographical scales that correspond to the drainage area of the water body of their interest. Available at: http://prj.geosyntec.com/MassDEPWBP
[3] The Estuary Program merged RIDEM's WPAs and HUC12 subwatersheds in Massachusetts (this scale is comparable with RIDEM's definition of WPAs); accordingly, based on early input by partners across the Watershed, the Estuary Program's WPAs were also defined by grouping or dividing HUC12s in order to delineate watersheds of interest by watershed groups, such as the Segreganset River watershed, that are not defined by USGS as Hydrological Unit or it is not one of the River Basins.
[4] Because a small section of the Narrow River (WPA) is outside the extent of Narragansett Bay Watershed as defined for Table A-1, the total Watershed area listed in Table A-1 differs slightly from the total of the WPA areas listed in Table A-5. This discrepancy is due to the delineation by RIDEM of the Narrow River (WPA), which differs from the delineation by USGS.

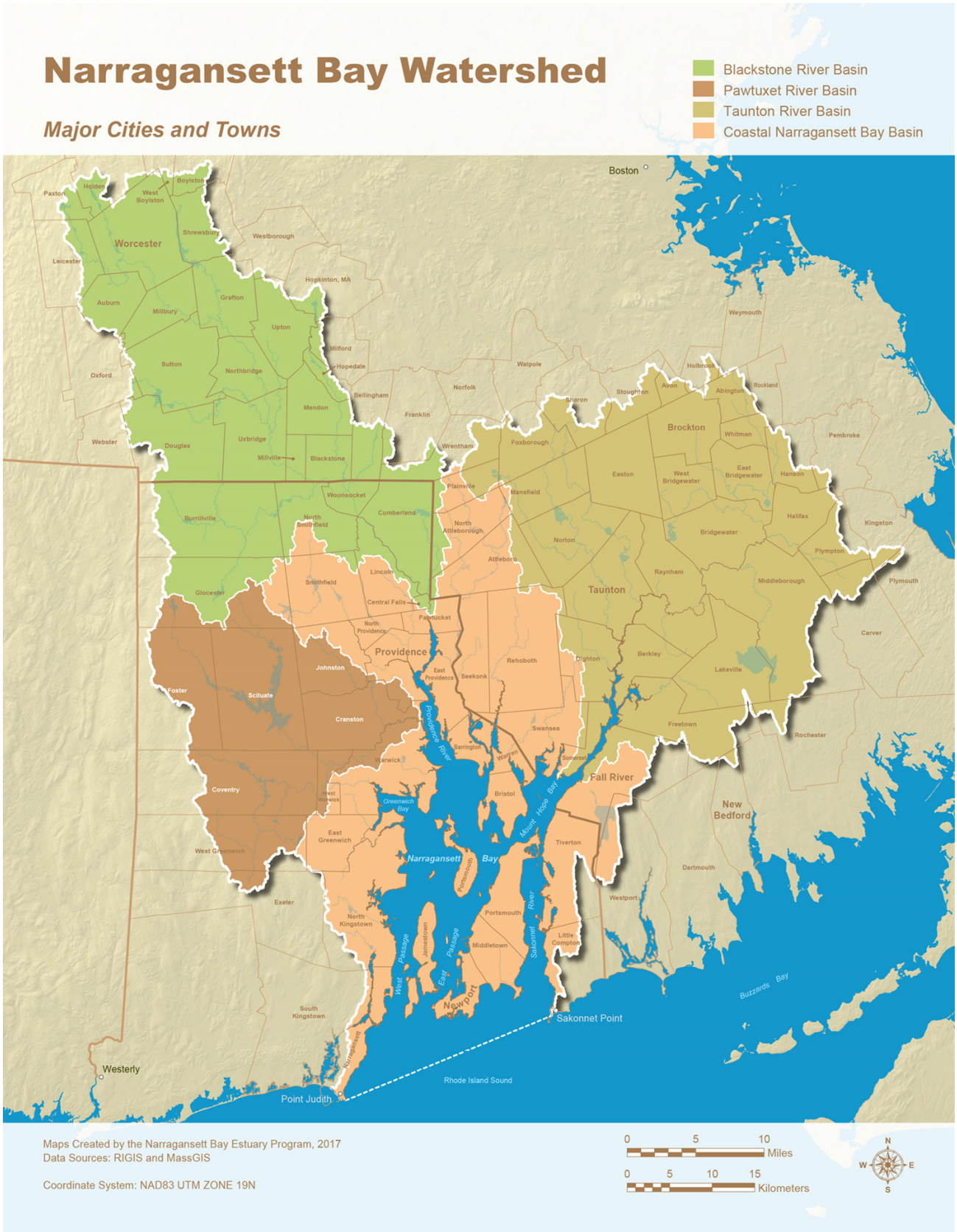


Figure A-5. Narragansett Bay Watershed - Major Cities and Towns

Table A-6. State breakdown of the Narragansett Bay Watershed.

State	Area of the Narragansett Bay Watershed by State[1]			Percent of the Narragansett Bay Watershed within each state	Percent of total state land area within the Narragansett Bay Watershed
	Acres	Square Miles	Square Kilometers		
Massachusetts	655,483	1,024	2,653	60	12.6
Rhode Island	434,853	679	1,760	40	62.5

[1] The total of the states' areas listed in this table differs slightly from the total area of the Narragansett Bay Watershed as listed in Table A-1. The discrepancy is due to differences between available datasets for municipal boundaries and those for boundaries of HUCs and the coastline.

Table A-7. Narragansett Bay Watershed – Major Cities and Towns. The Estuary Program included municipalities partially or completely within the Narragansett Bay Watershed.

State	Name of Municipality	Total Area of the Municipality within the Watershed[1]			Percent of the Municipality within the Watershed
		Acres	Square Miles	Square Kilometers	
Massachusetts	Attleboro	17,802	27.8	72.0	100
Massachusetts	Blackstone	7,308	11.4	29.6	100
Massachusetts	Bridgewater	18,178	28.4	73.6	100
Massachusetts	East Bridgewater	11,233	17.6	45.5	100
Massachusetts	Easton	18,709	29.2	75.7	100
Massachusetts	Mansfield	13,089	20.5	53.0	100
Massachusetts	Millville	3,177	5.0	12.9	100
Massachusetts	North Attleborough	12,422	19.4	50.3	100
Massachusetts	Northbridge	11,569	18.1	46.8	100
Massachusetts	Norton	18,725	29.3	75.8	100
Massachusetts	Plainville	7,347	11.5	29.7	100
Massachusetts	Uxbridge	19,196	30.0	77.7	100
Massachusetts	West Bridgewater	10,029	15.7	40.6	100
Massachusetts	Dighton	14,150	22.1	57.3	100
Massachusetts	Somerset	5,152	8.1	20.8	100
Massachusetts	Seekonk	11,925	18.6	48.3	100
Massachusetts	Berkley	10,560	16.5	42.7	100
Massachusetts	Swansea	14,761	23.1	59.7	99.9
Massachusetts	Raynham	13,268	20.7	53.7	99.9
Massachusetts	Rehoboth	30,320	47.4	122.7	99.8
Massachusetts	Taunton	30,903	48.3	125.1	99.8
Massachusetts	Millbury	10,485	16.4	42.4	99.6
Massachusetts	Lakeville	22,966	35.9	92.9	99.4
Massachusetts	Worcester	24,451	38.2	99.0	99.3
Massachusetts	Brockton	13,688	21.4	55.4	99.2
Massachusetts	Mendon	11,346	17.7	45.9	98.3
Massachusetts	Whitman	4,365	6.8	17.7	98.2
Massachusetts	Halifax	10,905	17.0	44.1	98.2
Massachusetts	Sutton	21,404	33.4	86.6	98.1
Massachusetts	Upton	13,513	21.1	54.7	96.8
Massachusetts	Auburn	9,828	15.4	39.8	93.3
Massachusetts	Grafton	13,934	21.8	56.4	93.3
Massachusetts	Abington	5,648	8.8	22.9	86.8
Massachusetts	Douglas	21,110	33.0	85.4	86.3
Massachusetts	Freetown	19,537	30.5	79.1	86.2
Massachusetts	Hopedale	2,730	4.3	11.0	80.2
Massachusetts	Foxborough	10,644	16.6	43.1	79.8
Massachusetts	Avon	2,247	3.5	9.1	77.4

Table A-7 continued.

State	Name of Municipality	Total Area of the Municipality within the Watershed[1]			Percent of the Municipality within the Watershed
		Acres	Square Miles	Square Kilometers	
Massachusetts	Middleborough	35,118	54.9	142.1	76.0
Massachusetts	Plympton	7,042	11.0	28.5	73.0
Massachusetts	Fall River	17,871	27.9	72.3	72.4
Massachusetts	Shrewsbury	8,789	13.7	35.6	63.2
Massachusetts	Hanson	5,857	9.2	23.7	58.2
Massachusetts	Wrentham	8,417	13.2	34.1	58.1
Massachusetts	Stoughton	5,463	8.5	22.1	51.9
Massachusetts	Bellingham	5,814	9.1	23.5	48.0
Massachusetts	Sharon	5,533	8.6	22.4	35.4
Massachusetts	Leicester	5,175	8.1	20.9	32.8
Massachusetts	Paxton	2,399	3.7	9.7	24.2
Massachusetts	Holbrook	1,086	1.7	4.4	22.9
Massachusetts	Boylston	2,593	4.1	10.5	20.5
Massachusetts	Holden	4,273	6.7	17.3	18.4
Massachusetts	Carver	4,191	6.5	17.0	16.5
Massachusetts	Westport	4,894	7.6	19.8	14.9
Massachusetts	Milford	1,363	2.1	5.5	14.2
Massachusetts	Hopkinton	2,245	3.5	9.1	12.6
Massachusetts	Kingston	1,310	2.0	5.3	10.8
Massachusetts	Franklin	1,600	2.5	6.5	9.3
Massachusetts	Rochester	1,982	3.1	8.0	8.6
Massachusetts	West Boylston	705	1.1	2.9	8.0
Massachusetts	Pembroke	874	1.4	3.5	5.8
Massachusetts	Oxford	845	1.3	3.4	4.8
Massachusetts	New Bedford	511	0.8	2.1	4.0
Massachusetts	Westborough	282	0.4	1.1	2.1
Massachusetts	Weymouth	141	0.2	0.6	1.3
Massachusetts	Webster	82	0.13	0.33	0.87
Massachusetts	Rockland	52	0.08	0.21	0.81
Massachusetts	Plymouth	304	0.47	1.23	0.47
Massachusetts	Walpole	28	0.04	0.12	0.21
Massachusetts	Norfolk	15	0.02	0.06	0.15
Massachusetts	Dartmouth	5	0.01	0.02	0.01
Rhode Island	Bristol	6,265	9.8	25.4	100
Rhode Island	Barrington	5,386	8.4	21.8	100
Rhode Island	Newport	5,045	7.9	20.4	100
Rhode Island	Portsmouth	14,826	23.2	60.0	100
Rhode Island	Jamestown	6,061	9.5	24.5	100
Rhode Island	Middletown	8,399	13.1	34.0	100

Table A-7 continued.

State	Name of Municipality	Total Area of the Municipality within the Watershed[1]			Percent of the Municipality within the Watershed
		Acres	Square Miles	Square Kilometers	
Rhode Island	Pawtucket	5,663	8.8	22.9	100
Rhode Island	Central Falls	825	1.3	3.3	100
Rhode Island	Cumberland	18,081	28.3	73.2	100
Rhode Island	Johnston	15,577	24.3	63.0	100
Rhode Island	Lincoln	12,144	19.0	49.1	100
Rhode Island	North Providence	3,709	5.8	15.0	100
Rhode Island	North Smithfield	15,932	24.9	64.5	100
Rhode Island	Scituate	35,091	54.8	142.0	100
Rhode Island	Smithfield	17,675	27.6	71.5	100
Rhode Island	West Warwick	5,180	8.1	21.0	100
Rhode Island	Woonsocket	5,050	7.9	20.4	100
Rhode Island	Warren	3,967	6.2	16.1	100
Rhode Island	Warwick	22,816	35.7	92.3	100
Rhode Island	Cranston	18,498	28.9	74.9	100
Rhode Island	East Providence	8,748	13.7	35.4	100
Rhode Island	Providence	11,986	18.7	48.5	99.8
Rhode Island	East Greenwich	10,376	16.2	42.0	99.4
Rhode Island	North Kingstown	24,342	38.0	98.5	86.5
Rhode Island	Burrillville	30,965	48.4	125.3	84.9
Rhode Island	Glocester	30,154	47.1	122.0	82.9
Rhode Island	Tiverton	14,836	23.2	60.0	77.7
Rhode Island	Narragansett	6,707	10.5	27.1	75.9
Rhode Island	Coventry	26,411	41.3	106.9	66.0
Rhode Island	Foster	20,369	31.8	82.4	61.3
Rhode Island	West Greenwich	15,507	24.2	62.8	47.3
Rhode Island	Little Compton	4,605	7.2	18.6	33.4
Rhode Island	Exeter	2,200	3.4	8.9	5.9
Rhode Island	South Kingstown	1,456	2.3	5.9	3.8

[1] The total of the towns' areas listed in this table differs slightly from the total area of the Narragansett Bay Watershed as listed in Table A-1. The discrepancy is due to differences between available datasets for municipal boundaries and those for boundaries of HUCs and the coastline.



Figure A-6. Narragansett Bay - Bay Regions

Table A-8. Narragansett Bay – Bay Regions. The Estuary Program for its data analysis and reporting defined a hierarchy of areas of estuarine waters within Narragansett Bay. The Bay Regions listed in this table are the largest areas. As listed in Table A-9, Estuarine Sections are nested within Bay Regions, and Estuarine Segments are nested within Sections.

Bay Region	Acres	Square Miles	Square Kilometers
Upper Estuary	36,044.2	56.3	145.9
East Passage	20,521.2	32.1	83.0
West Passage	25,368.2	39.6	102.7
Sakonnet River	13,292.3	20.8	53.8
Mouth of the Bay[1]	31,048.6	48.5	125.6

[1] While the boundaries of the estuarine waters are consistent across the three different scales, the area of the Mouth of Bay Region listed in this table includes additional area (at the line between the Bay and Rhode Island Sound) to encompass the full extent of shellfishing management areas. Thus, the discrepancy in total area of the Bay Regions listed above and the total Bay area listed in Table A-1.



Figure A-7. Narragansett Bay - Estuarine Sections and Segments

Table A-9. Narragansett Bay – Estuarine Sections and Segments. Estuarine Sections are nested within Bay Regions (Table A-8). Estuarine Segments are smaller areas such as coves nested within the Estuarine Sections.

Bay Region	Estuarine Section	Estuarine Segment	Acres	Square Miles	Square Kilometers
Upper Estuary	Providence River Estuary	Bullock Cove	149.4	0.23	0.60
Upper Estuary	Providence River Estuary	Lower Providence River	2338.6	3.7	9.5
Upper Estuary	Providence River Estuary	Middle Providence River	2080.0	3.3	8.4
Upper Estuary	Providence River Estuary	Omega Pond	32.0	0.05	0.13
Upper Estuary	Providence River Estuary	Pawtuxet River	21.9	0.03	0.09
Upper Estuary	Providence River Estuary	Seekonk River	674.4	1.1	2.7
Upper Estuary	Providence River Estuary	Upper Providence River	685.1	1.1	2.8
Upper Estuary	Providence River Estuary	Watchemoket Cove	73.8	0.12	0.30
Upper Estuary	Providence River Estuary	Waterplace Park	52.2	0.08	0.21
Upper Estuary	Upper Narragansett Bay	Buckeye Brook	22.8	0.04	0.09
Upper Estuary	Upper Narragansett Bay	Mill Gut	59.2	0.09	0.24
Upper Estuary	Upper Narragansett Bay	Upper Bay	10997.5	17.2	44.5
Upper Estuary	Greenwich Bay	Apponaug Cove	142.7	0.22	0.58
Upper Estuary	Greenwich Bay	Buttonwoods Cove	129.4	0.20	0.52
Upper Estuary	Greenwich Bay	East Greenwich Bay	1623.5	2.5	6.6
Upper Estuary	Greenwich Bay	Greenwich Cove	198.3	0.31	0.80
Upper Estuary	Greenwich Bay	Warwick Cove	150.3	0.23	0.61
Upper Estuary	Greenwich Bay	West Greenwich Bay	1006.3	1.6	4.1
Upper Estuary	Warren, Palmer & Barrington Rivers	Barrington River	665.9	1.0	2.7
Upper Estuary	Warren, Palmer & Barrington Rivers	Palmer River	622.8	0.97	2.52
Upper Estuary	Warren, Palmer & Barrington Rivers	Warren River	224.3	0.35	0.91
Upper Estuary	East Passage	Bristol Harbor	1382.3	2.2	5.6
Upper Estuary	Taunton River	Assonet River	83.5	0.13	0.34
Upper Estuary	Taunton River	Assonet River and Bay	428.8	0.67	1.74
Upper Estuary	Taunton River	Broad Cove	80.6	0.13	0.33
Upper Estuary	Taunton River	Lower Taunton River	1714.7	2.7	6.9
Upper Estuary	Taunton River	Upper Taunton River	819.7	1.3	3.3
Upper Estuary	Mount Hope Bay	Kickemuit River	551.3	0.86	2.23
Upper Estuary	Mount Hope Bay	Lower Cole River	575.8	0.90	2.33
Upper Estuary	Mount Hope Bay	Lee River	336.9	0.53	1.36
Upper Estuary	Mount Hope Bay	Mount Hope Bay	7898.3	12.3	32.0
Upper Estuary	Mount Hope Bay	Upper Cole River	221.7	0.35	0.90
East Passage	East Passage	Lower East Passage	5921.9	9.3	24.0
East Passage	East Passage	Middle East Passage	9433.7	14.7	38.2
East Passage	East Passage	Nag Pond	17.4	0.03	0.07
East Passage	East Passage	Newport Harbor (Brenton Cove)	544.9	0.85	2.21
East Passage	East Passage	Potter Cove	105.4	0.16	0.43
East Passage	East Passage	Upper East Passage	4497.9	7.0	18.2

Table A-9 continued.

Bay Region	Estuarine Section	Estuarine Segment	Acres	Square Miles	Square Kilometers
West Passage	West Passage	Bissel Cove	82.7	0.13	0.33
West Passage	West Passage	Dutch Harbor	407.6	0.64	1.65
West Passage	West Passage	Lower West Passage	4894.2	7.7	19.8
West Passage	West Passage	Middle West Passage	15109.3	23.6	61.1
West Passage	West Passage	Potowomut River	199.6	0.31	0.81
West Passage	West Passage	Quonset (Allen) Harbor	94.3	0.15	0.38
West Passage	West Passage	Upper West Passage	4062.9	6.4	16.4
West Passage	West Passage	Wesquage Pond	73.1	0.11	0.30
West Passage	West Passage	Wickford Harbor	444.5	0.69	1.80
Sakonnet River	Sakonnet River	Nonquit Pond	252.5	0.39	1.02
Sakonnet River	Sakonnet River	Lower Sakonnet River	7703.6	12.0	31.2
Sakonnet River	Sakonnet River	Nannaquaket Pond	216.9	0.34	0.88
Sakonnet River	Sakonnet River	The Cove	350.2	0.55	1.42
Sakonnet River	Sakonnet River	Upper Sakonnet River	4769.3	7.5	19.3
Mouth of the Bay	Mouth of Narragansett Bay	Easton Bay	521.6	0.82	2.11
Mouth of the Bay	Mouth of Narragansett Bay	Gooseberry (Island) Bay	385.9	0.60	1.56
Mouth of the Bay	Mouth of Narragansett Bay	Mouth of Narragansett Bay	27981.0	43.7	113.2
Mouth of the Bay	Mouth of Narragansett Bay	Sachuest Bay	666.1	1.0	2.7
Mouth of the Bay	Narrow River	Narrow River (Pettaquamscutt River)	618.5	0.97	2.50



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