



The School for Marine Science and Technology

Framework for Formulating the Mt. Hope Bay Natural Laboratory: A Synthesis and Summary

Rodney Rountree, Dave Borkman, Wendell Brown, Yalin Fan, Lou Goodman, Brian Howes, Brian Rothschild, Miles Sundermeyer, and Jefferson Turner

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Chapter 1. Introduction (Rodney Rountree)

Context: Our estuaries have been impacted by human activity for several centuries. These impacts relate to shore structures, channels, damming, nutrient introductions, sedimentation, toxics introduction, thermal modification, and fishing. But it has been only in the last few decades that public awareness of these various impacts has reached a point of regulating these impacts. As a result, many regulations have been promulgated to control or mitigate the mix of effects of human activity. Given that many aspects of human activity are difficult to modify, it is important to develop public policy that correctly link observations of effect (i.e., “the bay is dying,” “the fish are disappearing,” “the jellyfish have taken over”) with their causes. Without understanding the effects and their causes, we are likely to focus attention on the issues that are simple rather than those that are most harmful. Only by carefully linking cause and effect can we develop appropriate, cost effective, focused remedies to correcting, modifying, or mitigating the effects.

The problem is not simple. A multiplicity of human impacts or “causes” interact in a complex and non-linear way. These multiple causes, in turn, interact with a similar multiplicity of natural changes in the environment (for example, the recent global warming trends). It has become apparent that environmental issues can no longer be addressed by narrow studies that take into account only one or two causal variables. We can no longer study the issues with disjoint theory and observation. Rather, we need to develop a more hands-on approach combining theory and observation in a laboratory setting. Traditionally it is in the laboratory

setting where observations are made, data are collected, and experiments are formulated.

It is from this vantage point that we are working to create a natural laboratory in Mt. Hope Bay. The natural laboratory is intended to be a total systems analysis of the biological and physical dynamics of Mt. Hope Bay. As such, we will be viewing the chemistry, biology, and the physics of Mt. Hope Bay in terms of its internal and external dynamics. This approach will provide a unique understanding of the factors that influence change in Mt. Hope Bay. For example, from an internal dynamics point of view, we will be examining how nutrients, sedimentation, and local thermal inputs contribute to modifying the oxygen concentration and light availability in the water column. In turn, from an external point of view, we will be examining how the Taunton and Lee's river watershed and Narragansett Bay, itself, affect the nutrients, sedimentation, and thermal structure of Mt. Hope Bay. In a similar manner, to consider the effects on both recruitment and the adult stock of fish, we will need to consider the fish stocks not only just in Mt. Hope Bay but also throughout their entire range of activity. The idea of the natural laboratory is to not only address issues focused in Mt. Hope Bay, but to use Mt. Hope Bay to construct new and unique approaches to addressing the complex of human and natural factors that influence the estuarine environment in general and to contribute to improving public policy on estuarine management.

What is the MHBNL: The Mt. Hope Bay Natural Laboratory (MHBNL) program is a 5-year interdisciplinary program to examine the temporal and spatial

variability of the Mt. Hope Bay ecosystem. Emphasis is on the relative contributions of naturally occurring to anthropogenic factors in inducing changes to the bay's ecosystem. An overriding theme of the MHBNL is to quantify how various processes come together to produce the ecosystem and how that ecosystem evolves, both from naturally occurring as well as anthropogenic factors. It is a "laboratory" in that a variety of tools will be used for observation, experimentation and hypothesis testing. These tools are: numerical modeling, usage of existing data and results of ongoing monitoring programs, new *in situ* observations, and data assimilation, the latter being the "glue" which brings together all of the other tools. The Natural Laboratory approach will allow individual sources of the Bay's variability to be isolated and quantified in their importance to environmental, habitat, and fish population changes.

Location: Mt. Hope Bay is an ideal site for a natural laboratory. It is inland and enclosed, has well defined locations of inflow and outflow, and has had a large amount of observations made on its fish populations as well as a limited amount of measurements on its physical environment and habitats. Because of its limited size and access to land it is relatively easy to perform *in situ* deployments of instrumentation, for example for the monitoring of: (1) the tide, (2) river runoff, and (3) meteorological forcing, and (4) inflow and outflow through connecting passages. Note that exchange with coastal marine waters occurs only through the passages connecting the Sakonnet River and Narragansett Bay. Mt. Hope Bay is also the sink for the second largest watershed in Massachusetts with relatively little research on the impact of this watershed on the

bay. Mt. Hope Bay is less well studied than other Massachusetts estuaries because of its geography. It is located in the northeast corner of Narragansett Bay and lies partially within both Massachusetts and Rhode Island. The boundary between the two states bisects Mt. Hope Bay, effectively isolating it from other Massachusetts waters. Thus Rhode Island monitoring programs typically only sample the lower MHB, while Massachusetts programs tend to overlook MHB entirely. This happenstance of geography, together with recent public concern over the decline in fish stocks and habitat quality in MHB, has resulted in a growing need for improved monitoring of the Bay’s environment and biological communities.

Phases: Development of the MHBNL involves four temporally overlapping phases (Figure 1.1): 1) review and synthesis of research to date on the

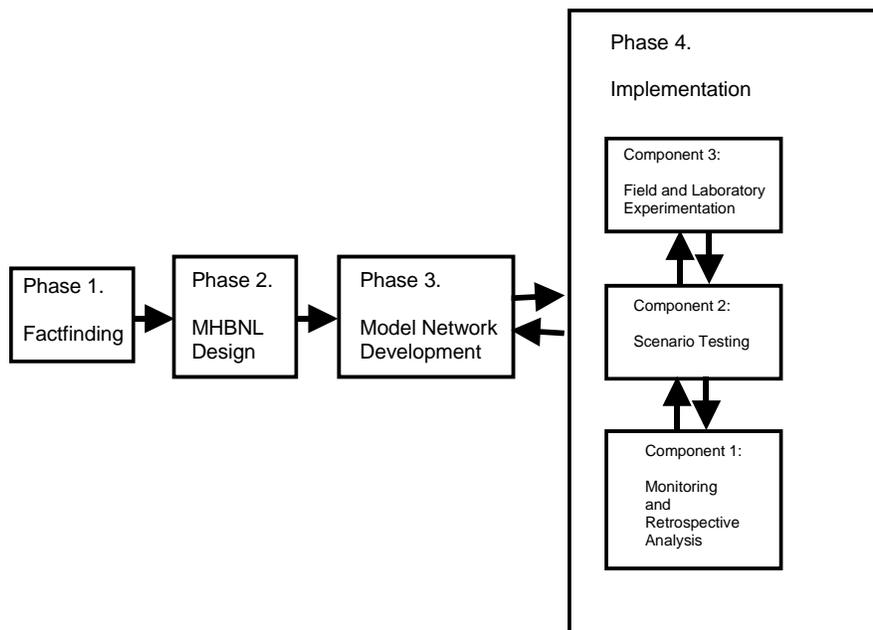


Figure 1.1. Schematic of the Mt. Hope Bay Natural Laboratory operations (see text). Arrows represent information flow and feedback loops.

Mt. Hope Bay ecosystem; 2) design of the natural laboratory structure; 3) development and validation of the model network; and 4) model network implementation and application to scenario testing. This report constitutes the completion of phase 1 and serves as the foundation for the program planning in Phase 2. Phase 2 will involve the planning and design of the MHBNL. Information contained in the Phase 1 report, together with retrospective analysis of data identified in the report, and comparison with previous estuarine modeling programs will be used to design the MHBNL program. During this phase we will identify the most promising existing models for incorporation into the MHBNL, as well as the appropriate observational variables and model input and outputs. In the third phase of the program, a network of integrated models of physical and biological processes operating in MHB will be developed. These models will be implemented and applied to scenario testing during the fourth phase. When integrated these models will allow the prediction of the impact of annual, seasonal, and episodic events on Mt. Hope Bay. The implementation itself will involve three iterative components (Figure 1.1), including monitoring, scenario testing, and experimentation. A carefully constructed monitoring program of the physical, biological and geochemical environment (e.g., spatial and temporal trends in salinity, temperature, DO, turbidity, nutrient loading, pollution, habitats and of the plankton, fish and invertebrate populations) is the critical foundation for the implementation. The availability of quantitative physical and biological data on the appropriate scales is essential for the success of the model scenario testing. The heart of the project is the use of model scenario testing to predict the

effects of the various natural and man-made impacts on the MHB system, while the process oriented and hypothesis driven field and laboratory experimentation will be used for model and scenario testing validation. The three components—monitoring, scenario testing, and experimentation—will form a dynamic feedback loop to constantly improve the model performance, and allow us to improve our understanding of how the MHB system functions. The model scenario testing will dictate the types of field and laboratory testing that are conducted, and the results of the field and laboratory testing will be used to provide input into, and perhaps suggest model modifications for, the scenarios.

Sources of variation in MHB: An overriding goal of the MHBNL is to determine how natural resources such as fish stocks and habitats are impacted by temporal changes in the environment and the biological community structure. Fish population dynamics are influenced by numerous natural and anthropogenic factors. Changes in interspecies interactions, environment quality, habitats, and fishing pressure all shape changes in populations. In addition, Mt. Hope Bay is not a closed system, and factors operating in adjacent areas can have a profound impact on its environment, fauna and flora. In particular, fish stocks in Mt. Hope Bay are not isolated, but rather in most cases represent small components of regional populations. Interactions between Mt. Hope Bay, Narragansett Bay and broader regional populations need to be examined. Although there are many potential sources of temporal variation that can affect Mt. Hope Bay fish stocks and faunal abundance patterns, we have identified at least six major categories in this report (Figure 1.2). These categories are not mutually exclusive and in fact

are strongly interrelated. These sources of variation can be grouped into biological and environment sources as described below.

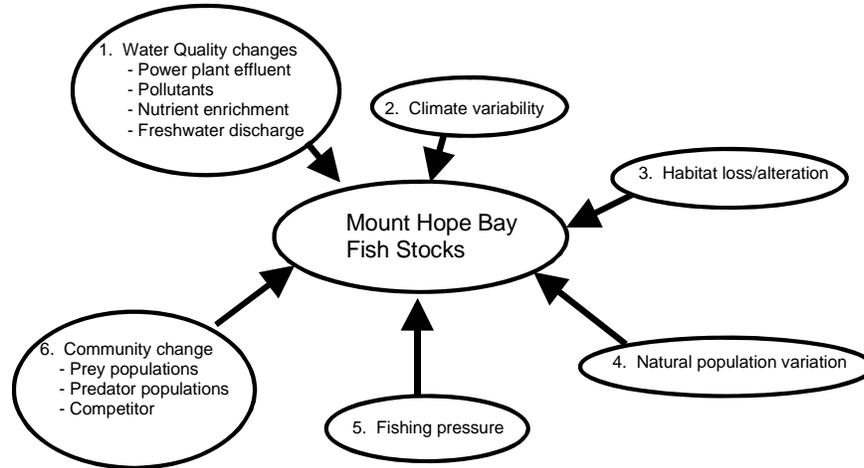


Figure 1.2. Diagram of selected sources of variation affecting Mt. Hope Bay fish stocks over annual, decadal and longer-term time scales.

1. Water Quality: Water quality can be affected by numerous factors; of particular concern in Mt. Hope Bay are Nutrient enrichment, environmental contaminants (pollution), and temperature changes resulting from the discharge of heated power plant effluent. Of these, nutrient enrichment and pollution are likely to be the most important, though the impact of the increase of water temperature has not been completely quantified. Mt. Hope Bay is well known to be seriously nutrient enriched. In effect this means that the bay has been over “fertilized” which can lead to eutrophication and result in profound changes in Mt. Hope Bay habitat quality and community structure. In severe cases, this can lead to sediment and habitat type changes in the system. For example, nutrient enrichment is often implicated in the demise of eelgrass habitat and their replacement with macroalgae beds and/or anaerobic mud flats (e.g., Short et

al.1996). Similarly, other types of pollution, such as heavy metal accumulation in sediments and faunal tissues can directly impact habitat quality, fish mortality, and availability of fishery resources. Warming of Mt. Hope Bay waters can have several types of impacts on the system: 1) warming can result in local avoidance of habitats by some species or specific life stages of a species, 2) can increase local mortality of some species or life stages that can not avoid effected areas, 3) contrastingly, it could attract and/or increase survival of other species depending on their thermal preferenda and tolerances. In addition, any differences in seasonal warming patterns may affect seasonal migration behaviors of fauna and ontogenetic migrations of fauna among habitats, resulting in changes in trophic linkages among habitats. Just as importantly, small changes in temperature, well within a species' thermal tolerance, may still strongly affect its habitat use and migration patterns due to energetic effects (i.e., the cost of acclimation, Rountree 1992, Rountree and Able 1993, Craig and Crowder 2000).

2. Climate Variability: Mt. Hope Bay water temperature is strongly affected by natural variation and cycles in regional and global climate change. Local, short-term and long-term changes in climate can have a strong impact on the MHB environment and community structure. Warming of MHB waters can have similar effects to those described above for the effects of heated power plant effluent, and in fact the relative contribution of natural climate variability and power plant effects on temperature patterns in MHB can be difficult to determine. Finally, annual variation in the seasonal pattern of freshwater discharge from rivers feeding into Mt. Hope Bay can also impact the environment, habitats, and

fauna in numerous ways. Most obvious would be changes in the salinity distribution in the bay, which can affect fauna distribution and habitat use patterns. Also of importance is the timing and magnitude of the discharge as larval ingress into the Bay can be strongly influenced by these factors. More subtly, changes in water volume of the discharge may alter the distribution and aerial coverage of shoreline and tidal flood plain habitats.

3. Habitat Loss/Change: Perhaps one of the most significant factors affecting Mt. Hope Bay fauna is that of habitat loss and change. It has long been recognized that specific habitats vary in their potential contribution to estuarine fauna populations, and especially for fish stocks, however in recent years it is becoming increasingly clear that habitats are linked to varying degrees by nekton movement patterns (e.g. Deegan et al. 2000). Habitat use by estuarine nekton varies among life stages, size classes within life stages, among seasons, and between day and tide cycles (Weinstein and Kreeger 2000). Because estuarine habitats are strongly linked by these processes, they cannot be considered in isolation. Quantification of the relative links among habitats, and their relative contribution to faunal production should be an important component of population models in order to fully explain population changes. Besides the obvious impact of loss of essential fish habitats (EFH), changes in the relative cover of Mt. Hope Bay habitats, and linkages among them, will likely have a strong impact on faunal abundances and local fish production.

4. Natural Population Variation. This arises from intrinsic mechanisms and long term population cycling. Intra-specific processes such as cannibalism and resource competition can also sometimes be important.

5. Fishing Pressure: Although fishing can be considered as a special case of a predator-prey relationship, we consider it separately because of its overriding interest to the program and to distinguish “anthropogenic” from “natural” sources. Recreational and commercial fishing likely have a profound influence on the Mt. Hope Bay system and its faunal constituents. Just as for natural predator-prey sources of community change, both direct and indirect effects can be of significance. The obvious direct effect is mortality associated with the direct harvest of a species of interest. Several types of indirect effects include: the removal of a predator, prey or resource competitor for a species of interest through harvesting; cascading effects caused by removing one component of a food web and resulting in complex changes in species relationships; and habitat disturbance resulting from the fishing activity. More subtly, fishing activity in other geographic areas can also impact transient species that use Mt. Hope Bay. An extreme example of this might be the removal of adult winter flounder along the coast before they can return to the MHB to spawn.

6. Community Change: The changes in community structure, i.e. changes in relative species abundance, distributions, and food webs, can have large impact on individual species population dynamics. These result in changes in the direct and indirect interspecific interactions. Most important of these are predator-prey interactions, but competitive interactions and indirect interactions such as “trophic

cascading” can sometimes be important. Unfortunately, all these types of interactions are often highly complex and difficult to quantify. Food habits studies provide perhaps the best source of data to define predator-prey relationships. Direct predator-prey relationships are the easiest to identify, but it is often very difficult to determine the strength of the relationship. This can often be supplemented or enhanced by stable isotope studies that can help determine the strength of trophic relationships among species. Changes in a prey species abundance for example, might result in changes in a predator species if the predator derives a significant component of its trophic resources from the species and it cannot switch to an alternate prey.

Report Contents: Extensive summaries of research to date on the Mt. Hope Bay physical, chemical, and biological environment have been performed by PG&E, Applied Science Associates (ASA), Marine Research, Inc. (MRI), various state and federal agencies, as well as by a number of academic researchers. This report will not duplicate these summaries but rather will highlight their important points and synthesize research to date with the objective of identifying gaps in our knowledge base related to the sources of variation identified above. The report is organized into seven chapters, including this introduction. The second chapter provides a general description of the MHB and summarizes important sources of environmental variability on various scales. The third chapter summarizes what is known of different habitat types in MHB and discusses potential sources of variation that can impact habitat quality. The major focus of this chapter is on nutrient loading to the Bay and how that affects

habitat quality. The fourth chapter summarizes what is known about the plankton community of MHB and its temporal patterns, including the phytoplankton, zooplankton and ichthyoplankton components. In the fifth chapter we summarize what is known about the nekton community in MHB, emphasizing fishery species. However, rather than repeat extensive summaries of data on the fish community available in previous reports, we concentrate on the life history and ecology of winter flounder. We feel that identification of key data gaps for winter flounder, and eventual modeling of winter flounder population dynamics and ecological interactions will provide a solid foundation for modeling of other fishes. The sixth chapter reviews models that have been specifically applied to the MHB system, but does not attempt to review model applications to other estuaries (that will be one focus of the MHBNL planning stage). Finally, Chapter seven summarizes the report findings and provides an analysis of the major data gaps for the MHB system in the context of understanding the six sources of variation described in the introduction.

Chapter 2. Physical Environment (Yalin Fan & Wendell Brown)

A. Introduction

Mt. Hope Bay (the Bay) is situated the northeast corner of Narragansett Bay (Figure 2.1), lying within both Rhode Island to the south and west and Massachusetts to the north and east (Figure 2.2). The Bay adjoins the East Passage of Narragansett Bay to the southwest where the Mt. Hope Bridge crosses over from Aquidneck Island to Bristol Point. The Taunton River discharges into the Bay from the north, along with the smaller Kickamuit, Cole, Lee, and Quequechan Rivers. (The Quequechan River starts from the Watuppa ponds, runs under dozens of city streets of Fall River, MA, and discharges into the Taunton River. Maybe because the Quequechan River is covered, its channel connecting to the Taunton River is not shown on any map.) The Sakonnet River is really an embayment that "originates" between Tiverton and Aquidneck Island and connects southern Mt. Hope Bay to Rhode Island Sound to the south. The Bay is seven miles (11.2 km) in length along its north-south axis (Kauffman and Adams 1981), covers an area of 35.2 km² (13.6 mi²) (Kauffman and Adams 1981) and has a volume of 201.7 million m³ (53.3 billion gal) at mean low water (MLW) (Chinman and Nixon 1985). The Mt. Hope Bay depth (Figure 2.1), which averages 5.7 m at MLW (Chinman and Nixon 1985), increases steadily from the relatively flat northern half of the Bay to the south (Kauffman and Adams 1981). A deep (about 10 m) channel connects Mt. Hope Bay to Narragansett Bay. In some parts of the channel its depth is greater than 24 m (NOAA 1992). Two dredged channels, maintained at a depth of approximately 10.7 m, connect the

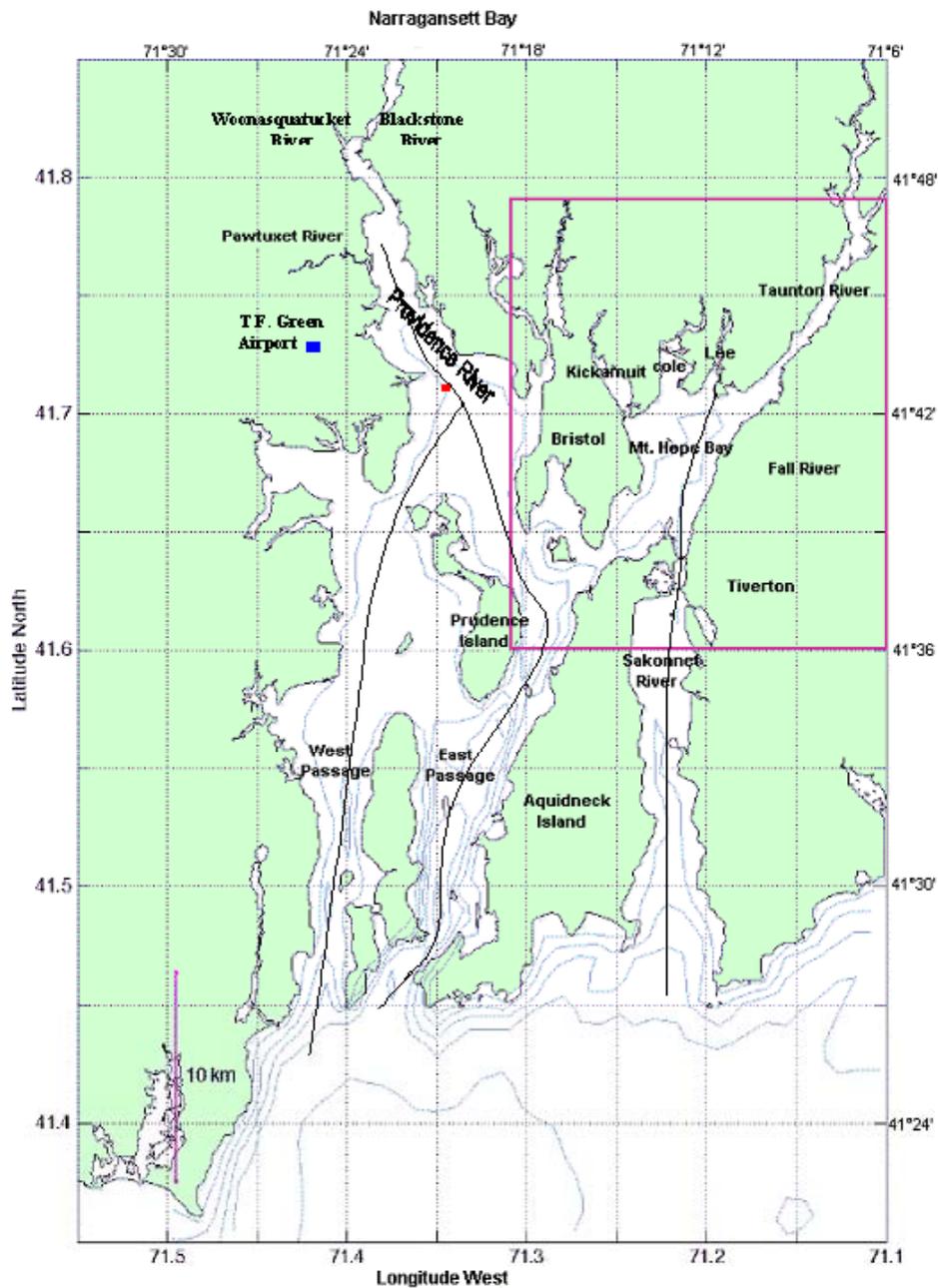


Figure 2.1. The bathymetry of the Narragansett Bay region, including Mt. Hope Bay, is defined by the contour lines for 5, 10, 15, 20, 30 and 40 m. The red box outlines the Mt. Hope Bay/Taunton River region shown in Figure 2.2. The Hicks (1959c) water property measurement transects and the Weisberg (1976) moored current meter location (red square in Providence River) are also shown.

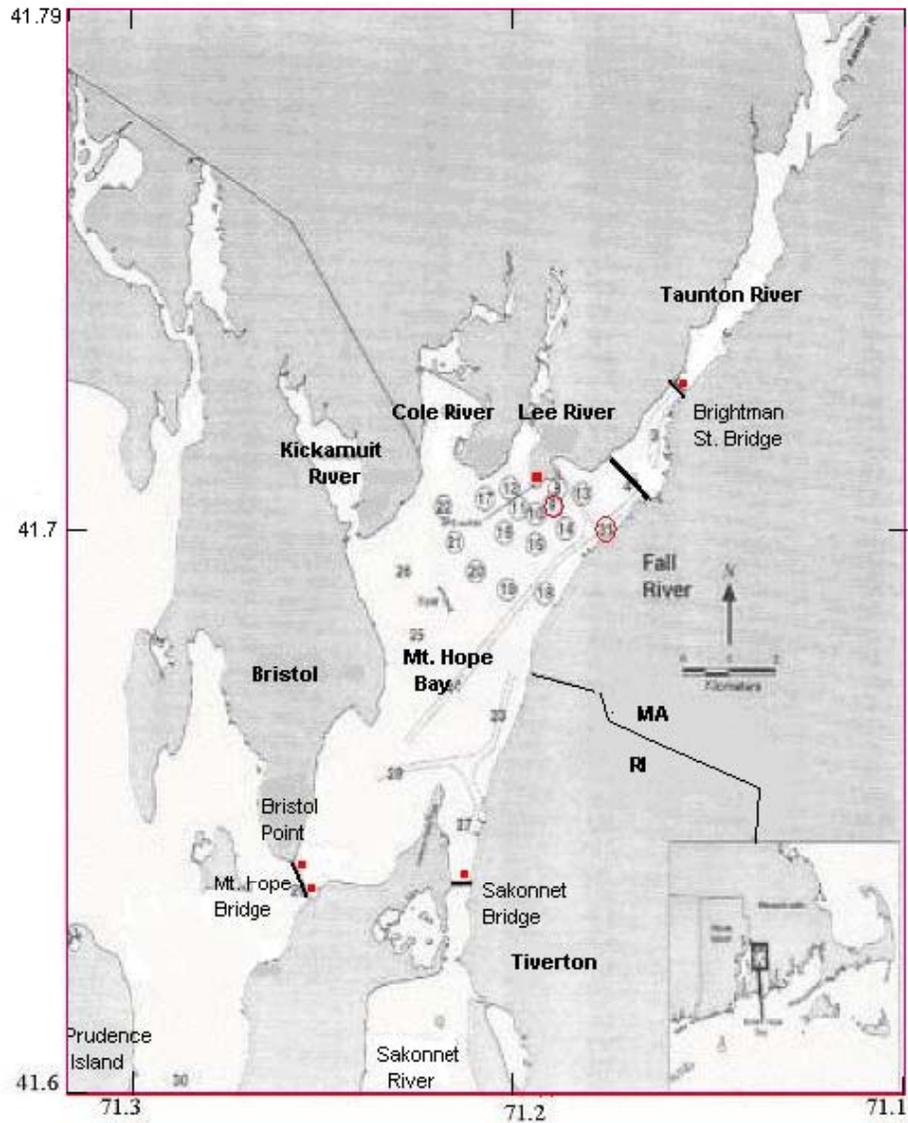


Figure 2.2. Mt. Hope Bay with distribution of the 31 ASA thermistor strings during the February 1999 thermal mapping study. Circles indicate the locations of moorings with bottom sensors (Swanson et al. 1999). The Spaulding and White (1990) moored current meter locations (red squares) are shown.

Bay to the Taunton River at Fall River. The Sakonnet Passage has a minimum channel depth of 7.5 m (McMaster 1960).

In recent years questions have been raised concerning the effect of the 1600MW fossil fuel-fired electrical generating facility at Brayton Point,

Massachusetts on the Mt. Hope Bay (Figure 2.2) ecosystem. The plant was built in the mid-1970s and has since been expanded starting in the mid-1980s. With the expansion, the amount of cooling water drawn from the Bay on the east side of Brayton Point has increased to the almost five million cubic meters per day. The amount of heat returned to the Bay through a channel and venturi system that discharges directly to the south of Brayton Point has also increased. The plant is operated so that the increase in the temperature between the intake and the discharge water is about 8°C (Sen 1996). This temperature rise is significantly less than the maxima permitted for summer (12.2°C) and winter (16.7°C) (Swanson et al. 1999). Nevertheless, the increased heat load to the Bay has been implicated in the decline of winter flounder (e.g., Gibson 1996a, 1996b).

One of the objectives of this paper is to place the Brayton Point thermal discharge in the context of natural temperature variability. In what follows, we describe what is known about the variability of the physical properties, including temperature, of Mt. Hope Bay, Narragansett Bay proper and the coastal ocean to the south. We consider variability of the properties at tidal (12 and 24 hours), weather band (2-10 days), and seasonal (1-3 months) periodicities, and typical annual (1 year) and interannual (multiple years) frequencies. These will be discussed in terms of estuarine processes related to river flow and stratification. The weather band variability of Bay properties is due predominantly to local and remote meteorological forcing. Thus we begin by reviewing the forcing. Fortuitously, the National Ocean Service (NOS) division of the National Oceanic and Atmospheric Administration (NOAA) recently began operational

measurements of many of the meteorological and oceanographic variables at several sites in the Narragansett Bay region.

The NOAA/NOS Physical Oceanographic Real-Time System (PORTS) (http://www.co-ops.nos.noaa.gov/d_ports.html) is designed to support safe and cost-efficient navigation by providing shipmasters and pilots with accurate real-time information. In addition to the measurement systems, PORTS includes a centralized data acquisition and dissemination system that provides both real-time and historical water levels, water temperature, and other meteorological data (i.e., air temperature, wind, rainfall, dew point) and barometric pressure data. It also provides real time current data at some of these stations. In some locations PORTS employs numerical circulation models to provide nowcasts and predictions of some of these variables.

The locations of the Narragansett Bay region PORTS stations are presented in Figure 2.3. The available observation products at the different PORTS stations are indicated in Table 2.1. (See Table 2.2 for detailed Narragansett Bay PORTS products information). Generally, P-M-Young sensors are used to measure wind variations, and an RDI, Inc., ADCP, with a Paroscientific pressure sensor, is used to measure currents and sea level at PORTS sites.

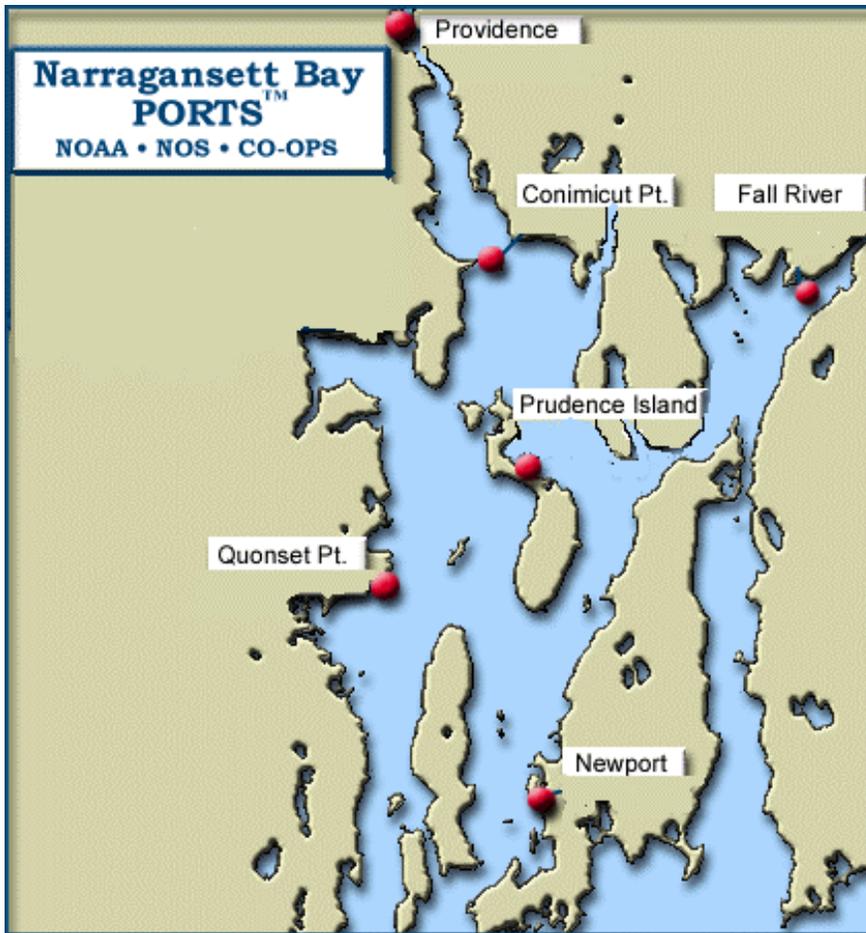


Figure 2.3. The NOAA PORTS sites in the Narragansett Bay region.

Table 2.1. Narragansett Bay PORTS station data products.

| Location | Lat (N) | Lon (W) | SL (m) | Current (Knots) | Water T (°C) | Air T (°C) | Wind (m/s, °) | BP (mbar) |
|-------------------------------|----------|----------|--------|-----------------|--------------|------------|---------------|-----------|
| Providence | 41°48.4' | 71°24.1' | X | X | X | X | X | X |
| Conimicut Light | 41°43.0' | 71°20.6' | X | | X | X | X | X |
| Fall River | 41°42.3' | 71°9.8' | X | X | X | X | | X |
| Prudence Island (Potter Cove) | 41°38.1' | 71°20.4' | | | X | X | X | X |
| Quonset Point | 41°35.1' | 71°24.5' | X | X | X | X | X | X |
| Newport | 41°30.3' | 71°19.6' | X | | X | X | X | X |

Table 2.2. Available historical and real-time PORTS information for the Narragansett Bay area. Status indicates which versions of the different variables are available online, including historical data (H), real time data (RT), and information (X).

| Station | Begin Time | End Time | Data Type | Status |
|---------------------------------|------------------|-------------------|----------------------|--------|
| Providence | 10/6/1999 19:00 | 12/19/2001 17:00 | Wind | H RT |
| | 9/22/1999 18:00 | 12/19/2001 17:00 | Air Temperature | H RT |
| | 9/25/1995 22:00 | 12/19/2001 17:00 | Water Temperature | H RT |
| | 5/17/1999 1:00 | 12/19/2001 17:00 | Barometric Pressure | H RT |
| | 8/24/1999 19:00 | 1/11/2001 9:00 | Water Conductivity | |
| | | | Acc Harmonic Const | X |
| | 1/1/1996 0:00 | 12/19/2001 17:54 | 6-Minute Water Level | H RT |
| | 1/1/1996 0:00 | 10/31/2001 23:00 | Hourly Height | H |
| | 1/1/1996 2:18 | 10/31/2001 23:54 | Acc High/Low | H |
| | 6/1/1938 0:00 | 10/31/2001 23:54 | Acc Monthly Mean | |
| | 1/1/1939 0:00 | 12/31/1992 23:54 | Acc Annual Mean | |
| | | | Acc Station Datum | X |
| Conimicut Light | 10/17/1999 11:00 | 12/19/2001 16:00 | Wind | H RT |
| | 10/17/1999 11:00 | 12/19/2001 16:00 | Air Temperature | H RT |
| | 10/17/1999 11:00 | 12/19/2001 16:00 | Water Temperature | H RT |
| | 10/17/1999 11:00 | 12/19/2001 16:00 | Barometric Pressure | H RT |
| | 11/29/1999 12:00 | 11/29/1999 12:54 | Dew Point | |
| | 11/29/1999 12:00 | 10/11/2001 13:54 | Rainfall | |
| | 11/1/1999 0:00 | 11/29/1999 23:00 | Acc Harmonic Const | X |
| | 10/17/1999 8:00 | 12/19/2001 16:54 | 6-Minute Water Level | H RT |
| | 10/17/1999 14:00 | 10/31/2001 23:00 | Acc Hourly Height | H |
| | 10/17/1999 14:00 | 10/31/2001 23:54 | Acc High/Low | H |
| | 10/18/1999 0:00 | 10/31/1999 0:00 | Acc Daily Mean | |
| | 11/1/1999 0:00 | 10/31/2001 23:54 | Acc Monthly Mean | |
| | | Acc Station Datum | X | |
| Fall River | 10/17/1999 11:00 | 12/19/2001 16:00 | Air Temperature | H RT |
| | 12/31/2000 1:00 | 12/31/2000 1:00 | Air Temperature | H RT |
| | 10/17/1999 11:00 | 12/19/2001 16:00 | Water Temperature | H RT |
| | 10/17/1999 11:00 | 12/19/2001 16:00 | Barometric Pressure | H RT |
| | 2/29/2000 19:00 | 6/18/2001 19:00 | Water Conductivity | |
| | 12/14/2000 3:00 | 12/14/2000 3:54 | Solar Radiation | |
| | 11/1/1999 0:00 | 11/29/1999 23:00 | Acc Harmonic Const | X |
| | 10/17/1999 7:00 | 12/19/2001 16:54 | 6-Minute Water Level | H RT |
| | 10/17/1999 7:00 | 10/31/2001 23:00 | Acc Hourly Height | H |
| | 10/17/1999 7:00 | 10/31/2001 23:54 | Acc High/Low | H |
| | 10/1/1977 0:00 | 10/31/2001 23:54 | Acc Monthly Mean | H |
| | | | Acc Station Datum | X |
| | | Published BM | X | |
| Providence Island (Potter Cove) | 10/17/1999 11:00 | 12/19/2001 15:54 | Wind | H RT |
| | 10/17/1999 11:00 | 12/19/2001 15:00 | Air Temperature | H RT |
| | 3/1/2000 16:00 | 12/19/2001 15:00 | Barometric Pressure | H RT |
| | 9/19/2001 21:00 | 9/19/2001 21:00 | Water Conductivity | |

| | | | | | | |
|----------------|------------|-------|------------|--------------|----------------------|------|
| | 0/17/1999 | 11:00 | 12/19/2001 | 16:00 | Wind | H RT |
| | 10/17/1999 | 11:00 | 12/19/2001 | 16:00 | Air Temperature | H RT |
| | 10/17/1999 | 11:00 | 12/19/2001 | 16:00 | Water Temperature | H RT |
| | 10/17/1999 | 11:00 | 12/19/2001 | 16:00 | Barometric Pressure | H RT |
| | 2/29/2000 | 16:00 | 7/21/2000 | 21:00 | Water Conductivity | |
| | 3/1/2000 | 0:00 | 3/29/2000 | 23:00 | Acc Harmonic Const | X |
| | 10/17/1999 | 8:00 | 12/19/2001 | 13:30 | 6-Minute Water Level | H RT |
| | 10/17/1999 | 15:00 | 10/31/2001 | 23:00 | Acc Hourly Height | H |
| | 10/17/1999 | 14:18 | 10/31/2001 | 23:54 | Acc High/Low | H |
| | 3/1/2000 | 0:00 | 10/31/2001 | 23:54 | Acc Monthly Mean | H |
| | | | | | Acc Station Datum | X |
| Newport | 10/13/1999 | 20:00 | 12/19/2001 | 17:00 | Wind | H RT |
| | 8/20/1999 | 18:00 | 12/19/2001 | 17:00 | Air Temperature | H RT |
| | 9/12/1995 | 17:00 | 12/19/2001 | 17:00 | Water Temperature | H RT |
| | 2/6/1996 | 5:00 | 12/19/2001 | 17:00 | Barometric Pressure | H RT |
| | 2/26/1999 | 11:00 | 12/19/2001 | 17:00 | Water Conductivity | |
| | 12/31/1999 | 22:00 | 12/31/1999 | 22:45 | Rainfall | |
| | | | | | Acc Harmonic Const | X |
| | 1/1/1996 | 0:00 | 12/19/2001 | 17:54 | 6-Minute Water Level | H RT |
| | 11/8/1997 | 10:00 | 11/8/1997 | 10:18 | Pressure 6-Minute | |
| | 9/10/1930 | 5:00 | 12/31/1993 | 23:00 | Acc Hourly Height | H |
| | 1/1/1994 | 0:00 | 10/31/2001 | 23:00 | Acc Hourly Height | H |
| | 8/23/1976 | 5:36 | 12/31/1993 | 23:54 | Acc High/Low | H |
| | 1/1/1994 | 2:42 | 10/31/2001 | 23:54 | Acc High/Low | H |
| | 10/1/1930 | 0:00 | 10/31/2001 | 23:54 | Acc Monthly Mean | |
| | 1/1/1931 | 0:00 | 12/31/1993 | 23:54 | Acc Annual Mean | |
| | | | | | Acc Station Datum | X |
| | | | | Published BM | X | |

B. Natural Variability of the Narragansett Bay/Mt. Hope Bay Region

1. Meteorology

Weisberg (1976) notes that the effects of wind can penetrate throughout the entire water column of a partially mixed estuary like Narragansett Bay and Mt. Hope Bay and thus can be of equal (or even greater) importance than the tides or gravitational convection in influencing the circulation.

The intensity and preferred direction of Narragansett Bay winds vary considerably with season (see Figure 2.4). Wind speeds average 5 m/s (from the

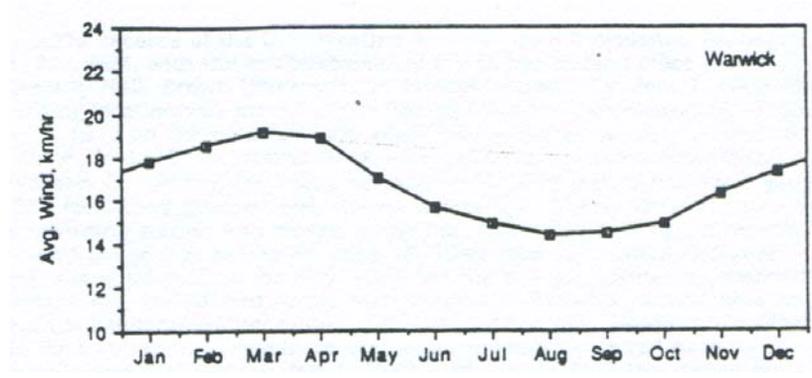


Figure 2.4. Annual cycle of monthly mean wind speeds at T.F. Green State Airport in Warwick, RI. The data are averaged over the years 1964 to 1987, inclusive (Pilson 1991).

northwest) during the winter and spring and 4 m/s (from the southwest and west) during summer and fall (Spaulding and White 1990). For example, during a typical January/February the vector-averaged wind direction (Figure 2.5) is

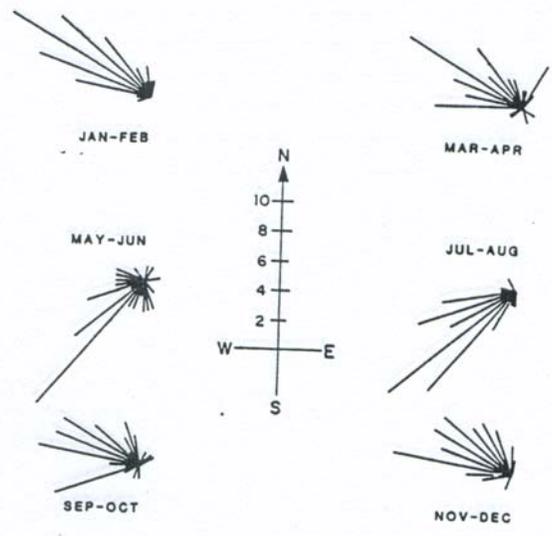


Figure 2.5. Annual cycle of bimonthly circular frequency histograms of wind vectors at T.F. Green State Airport in Warwick, RI. These histograms were formed by vector-averaging winds in 10° sectors. The scale indicates the number of years (in the 1964 to 1986 interval of observations) that the vector-averaged wind blew from the indicated direction in that particular bimonthly period. (Reprinted from Pilson 1991.)

confined to a sector between 10°E of north and 10°S of west, with the most common wind direction being slightly W of NW. During the summer months of May to August the wind typically blows from the southwest.

How representative of Narragansett Bay are the Green Airport winds? Pilson (1991) did point out that, while Green Airport winds are representative of the regional winds, the orientation of the coastline has a considerable influence on local wind conditions. Thus, the response of the water movements to local winds in different locations within the bay may vary considerably. In addition, further work is needed to define the wind-driven part between the exchange of bay and offshore waters.

The bay-wide structure of the Narragansett Bay winds can be seen more clearly in the Figure 2.6 comparison of the PORTS daily wind vector time series between 31 October 2000 and 31 October 2001. We can see that winds mainly come from the north in winter and from the south in summer. Sea breeze is a very common phenomenon along the coastal area. Because the ocean warms up and cools down more slowly than the land, the land is warmer than the ocean in the daytime, and cooler than the ocean at night. Due to the pressure difference, the wind will usually blow from the ocean to the land in the daytime and from the land to the ocean at night. In the summertime the phenomenon of the sea breeze is particularly important. It is commonly observed that the wind may blow from the north or northwest in the morning, but that sometime after midday the sea breeze sets in and blows up Narragansett Bay.

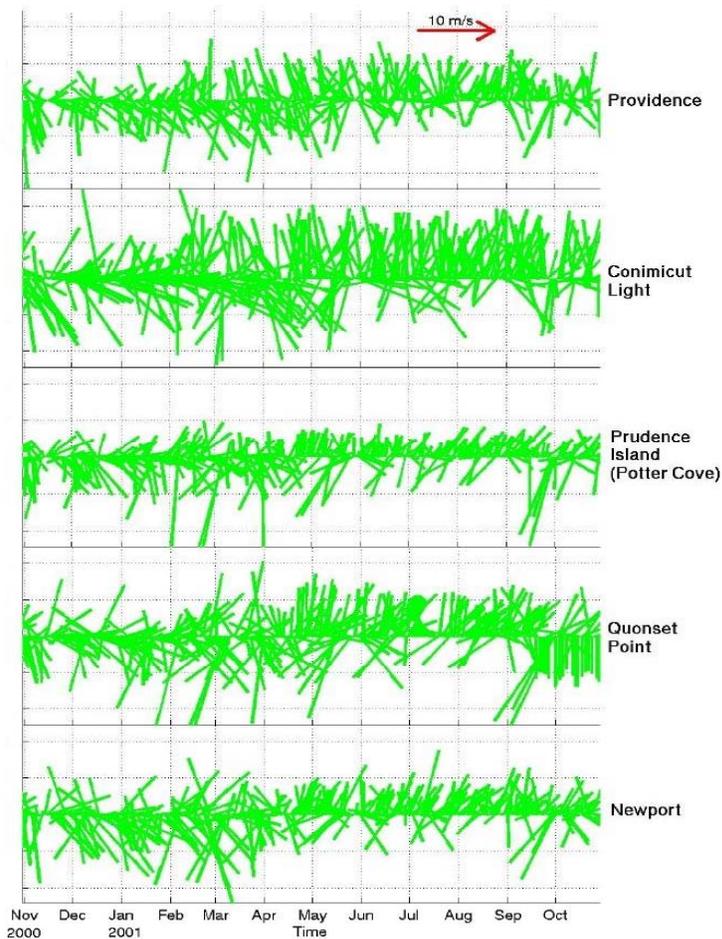


Figure 2.6. Daily wind vectors from PORTS stations at Providence, Conimicut Light, Prudence Island, Quonset Point and Newport between 31 October 2000 and 31 October 2001. The actual wind observations are hourly at Providence, Conimicut Light, Quonset Point, and Newport; every six minutes at Prudence Island.

2. Fresh Water Inflow

The U.S. Geological Survey (USGS) Water Resources Division provides data on surface water, ground water and water quality data for the Massachusetts-Rhode Island District. The short-term variability of some river discharges in the region is measured. Every four hours, 15-minute data for river stage, discharge, water temperature, specific conductance, and precipitation is available from

approximately 65 stream gauges in Massachusetts and Rhode Island

(<http://ma.water.usgs.gov/>).

Discharge information for four important rivers in the Narragansett Bay area have been obtained from the USGS archive for the years 1930 to 1999. (See Table 2.3 for the station information.) The Taunton, Blackstone, Woonasquatucket, and Pawtuxet River (Figure 2.1) records for September 1999 to September 2000 are presented in Figure 2.7. We can see from these plots that the

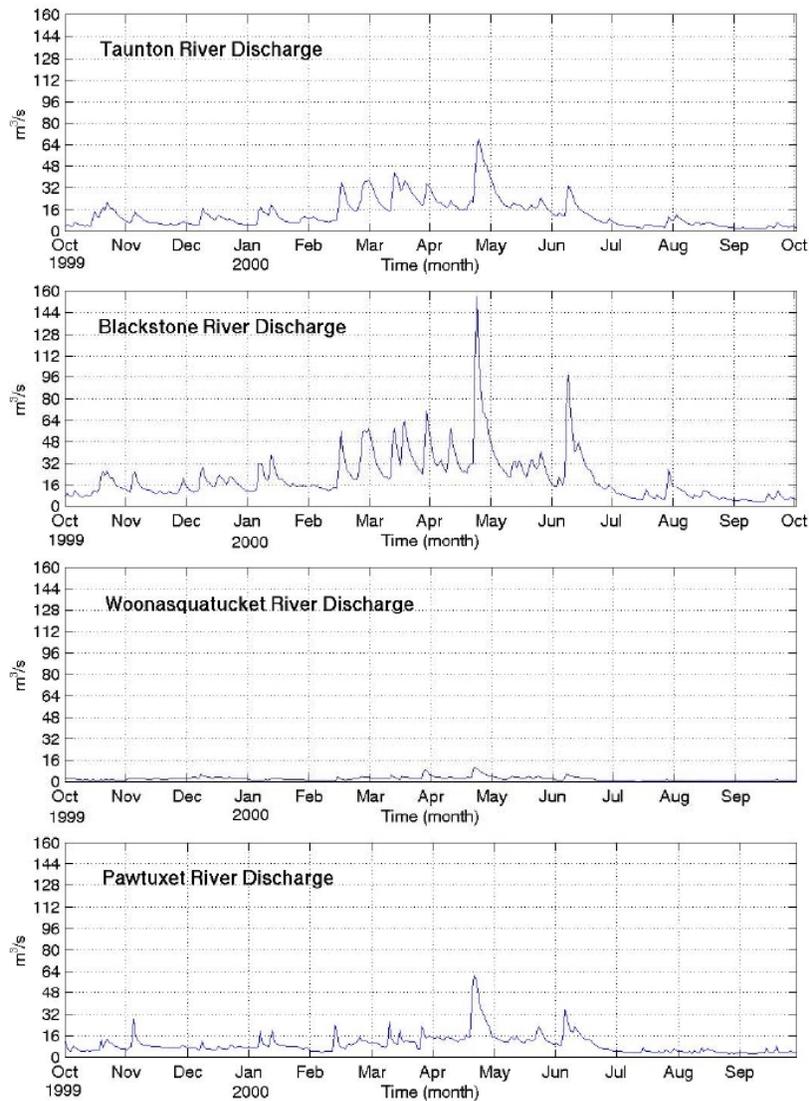


Figure 2.7. Taunton, Blackstone, Woonasquatucket and Pawtuxet River discharge records from 30 September 1999 to 30 September 2000. (See Table 2 for details of data record information.)

highest discharges in all rivers are in April, as the result of spring runoff and snowmelt. Low discharges for the year are in July, August, and September, when high evapotranspiration rates limit the amount of precipitation that becomes available for runoff.

The Taunton River is the major source of fresh water to the Mt. Hope Bay, with a mean annual discharge of $29.7 \text{ m}^3/\text{s}$ (7,846 gal/s) (Ries 1990). The Cole River provides an additional annual mean discharge of $0.81 \text{ m}^3/\text{s}$ (214 gal/s), and additional discharges of lesser volumes are provided from ungauged areas adjacent to Mt. Hope Bay (Ries 1990).

The Taunton River exerts a significant effect on the Mt. Hope Bay System, both through the discharge of its nitrogen load and through its effect upon the salinity distribution and water column density field within the estuary. Freshwater discharge from the Taunton River helps to create the vertical density stratification of Mt. Hope Bay, primarily due to salinity. In order to assess potential inter-annual variations in the effect of the Taunton River on water column stratification, annual discharge measurements collected at the Taunton Gauge by the USGS from 1929-1999 were obtained. However, there are data missing for some periods (see Table 2.3 for details).

Table 2.3. Station information for Taunton, Blackstone, Woonasquatucket and Pawtuxet Rivers USGS gauge stations.

| Site | Latitude | Longitude | Gauge Information | Near Real-Time Discharge | Daily Mean Stream Flow |
|------------------------|------------|-------------|--|--------------------------|--|
| Taunton River | 41°56'02" | 70°57'25" | Water-stage recorder. Datum of gauge is 2.93 m above sea level. Prior to Oct. 1996, at sites 12.20m apart about 122.00 m upstream: Oct. 1929 to Sept. 30, 1931, inverted non-recording gauge with zero of gauge at 3.06 m; 1931/10/01 to 1934/06/08, non-recording gauge; 1934/06/09 to 1976/04/23 and 1985/04/19 to May 1988, water stage recorders at present datum. | X | <p>Period 1 1929/10/01 – 1976/04/23</p> <p>Period 2 1985/04/19 – 1988/05/31</p> <p>Period 3 1996/10/01 – 2000/09/30</p> |
| Blackstone River | 42°00'22" | 71° 30' 13" | Water-stage recorder. Datum of gage is 32.76 m above sea level. | X | 1929/02/22 – 2000/09/30 |
| Woonasqua-tucket River | 41° 51'32" | 71 °29'16" | Water-stage recorder. Elevation of gage is 28.98 m above sea level, from topographic map. Satellite gage-height telemeter at station. | X | 1941/07/09 – 2000/09/30 |
| Pawtuxet River | 41°45'03" | 71°26'44" | Water-stage recorder. Datum of gauge is 2.44 m above sea level. | X | 1939/12/06 – 2000/09/30 |

Since there is no data gap for Blackstone River discharge during this period, cross- correlation between these two rivers' discharge records was calculated (Figure 2.8). Daily discharge time series from 1 October 1929 to 23 April 1976 for these two rivers were used to calculate the cross-correlation. We can see from Figure 2.8 that these two rivers are highly correlated at zero lag. So we estimated the missing Taunton River yearly discharge from Blackstone River yearly discharge based on the cross- correlation function (Figure 2.8). We can see that the Taunton River exhibits a large degree of inter-annual variation in discharge.

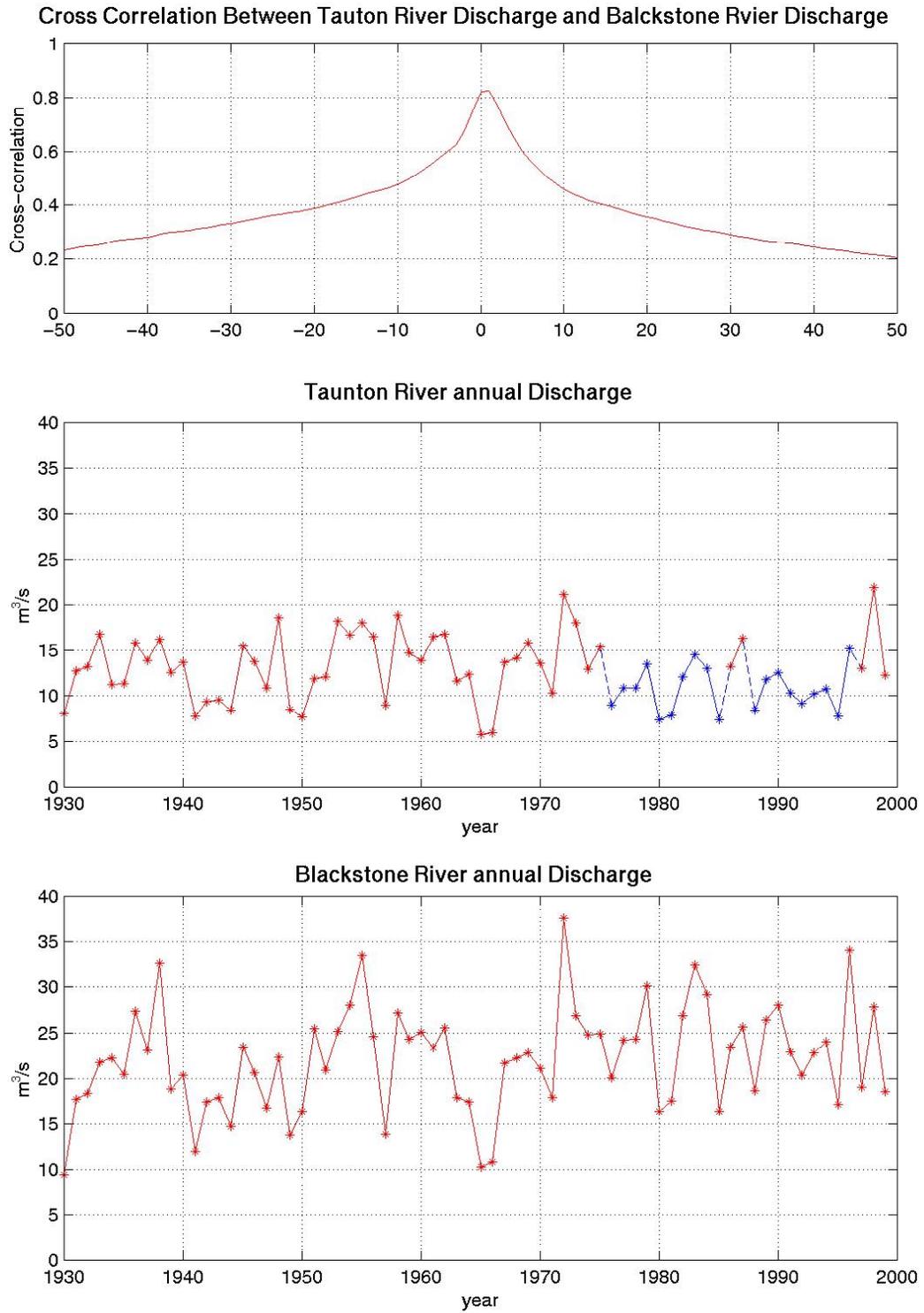


Figure 2.8. Upper panel: Cross-correlation between Taunton River discharge and Blackstone River discharge; middle panel: Taunton River annual discharge (the red line with stars showing measurement, and the blue line with stars showing estimated results); bottom panel: Blackstone River annual discharge.

3. Tidal Sea Levels and Currents

Sea Level: Tidal variations affect the variability of physical properties throughout the Narragansett Bay system, including Mt. Hope Bay, in important ways. The principal tidal sea level variations occur at semidiurnal periods, with the M_2 constituent surface elevation amplitudes being the largest (Figure 2.9). This is clear from the table of harmonic constants for the 5 PORTS sites in Table 2.4.

As the cotidal chart for sea level (Figure 2.10) shows, the M_2 tidal amplitude increases by about 20% (10 cm) from the entrance to the upper reaches of both Narragansett and Mt. Hope Bays, while the corresponding phase differences are small (< 3%). Thus the tides occur nearly simultaneously throughout the bays. The same is true for the other important semidiurnal (S_2 , N_2), diurnal (O_1 , K_1) and higher harmonic (M_4) constituents.

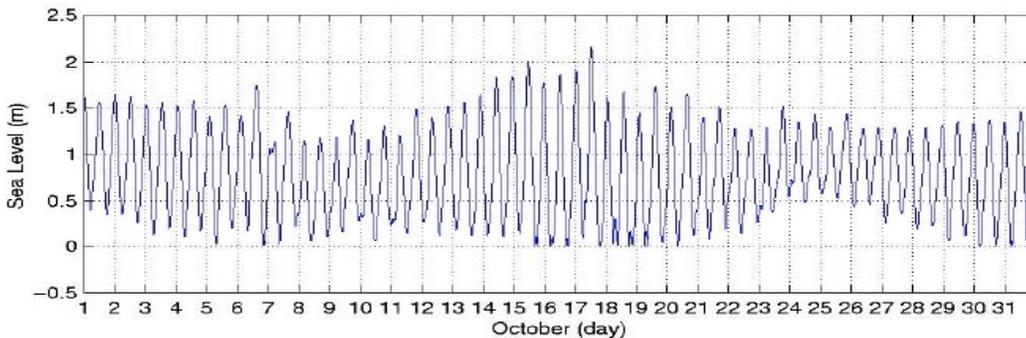


Figure 2.9. Sea level record at the Fall River PORTS station for October 2001.

Table 2.4. Tidal harmonic constants for the important tidal constituents at several sites in the Narragansett Bay system, including Providence, Conimicut Light, Fall River, Quonset Point, and Newport. Tidal phases are in Greenwich epoch degrees.

| Name | Providence | | Conimicut Light | | Fall River | | Quonset Point | | Newport | |
|------|------------|-------|-----------------|-------|------------|-------|---------------|-------|---------|-------|
| | Amp (m) | G (°) | Amp (m) | G (°) | Amp (m) | G(°) | Amp (m) | G(°) | Amp (m) | G(°) |
| M2 | 0.643 | 9.5 | 0.589 | 7.8 | 0.607 | 9.3 | 0.527 | 3.5 | 0.518 | 2.2 |
| S2 | 0.138 | 33.7 | 0.127 | 29.3 | 0.128 | 32 | 0.119 | 24.8 | 0.11 | 24.3 |
| N2 | 0.152 | 354.6 | 0.133 | 344.8 | 0.138 | 347.3 | 0.126 | 345.2 | 0.123 | 346.2 |
| K1 | 0.073 | 169.4 | 0.056 | 182.3 | 0.058 | 186.8 | 0.059 | 167.5 | 0.065 | 166.6 |
| M4 | 0.103 | 62.1 | 0.09 | 62.1 | 0.099 | 69.1 | 0.07 | 43.6 | 0.055 | 36 |
| O1 | 0.056 | 202.2 | 0.047 | 203.1 | 0.046 | 200.1 | 0.056 | 199.5 | 0.052 | 200.4 |
| M6 | 0.027 | 312.7 | 0.015 | 304.7 | 0.019 | 337.4 | 0.006 | 275.2 | 0.005 | 221.7 |
| S4 | 0.014 | 23.8 | 0.015 | 22.6 | 0.015 | 29 | 0.01 | 19.5 | 0.006 | 356.9 |
| NU2 | 0.027 | 353 | 0.026 | 347.9 | 0.027 | 350.3 | 0.024 | 347.6 | 0.023 | 348.4 |
| MU2 | 0.031 | 358.6 | 0.014 | 203.7 | 0.015 | 204.3 | 0.013 | 199.3 | 0.025 | 346.1 |
| 2N2 | 0.022 | 343.2 | 0.018 | 321.9 | 0.018 | 325.4 | 0.017 | 326.9 | 0.018 | 327.6 |
| SA | 0.06 | 131.8 | 0 | 0 | 0 | 0 | 0 | 0 | 0.061 | 140.5 |
| Q1 | 0.016 | 179.9 | 0.009 | 213.4 | 0.009 | 206.7 | 0.011 | 215.3 | 0.015 | 177.6 |
| T2 | 0.013 | 14.1 | 0.008 | 28.4 | 0.008 | 31.1 | 0.007 | 24 | 0.009 | 7.5 |
| 2Q1 | 0.001 | 234.6 | 0.001 | 223.7 | 0.001 | 213.3 | 0.001 | 231.2 | 0.003 | 229.7 |
| P1 | 0.025 | 182.3 | 0.018 | 183.8 | 0.019 | 187.8 | 0.02 | 169.9 | 0.023 | 181.7 |
| M3 | 0.012 | 68.3 | 0 | 0 | 0 | 0 | 0 | 0 | 0.009 | 52 |
| L2 | 0.012 | 321.4 | 0.017 | 30.8 | 0.017 | 31.3 | 0.015 | 21.8 | 0.009 | 316 |
| K2 | 0.038 | 29.7 | 0.035 | 31 | 0.035 | 33.8 | 0.032 | 26.6 | 0.032 | 28.4 |

The M_2 tides typically account for 90% of the sea level variance in Mt. Hope Bay (Spaulding and White 1990). The tidal sea level fluctuations measured at Fall River (Figure 2.9) indicate a mean tidal range of approximately 1.34 m for Mt. Hope Bay. This corresponds to a mean Mt. Hope Bay tidal prism of approximately 37.3 million m^3 . Of course, the tidal prism varies with the stage of the tides in the spring-neap cycle. For example, the maximum tidal range is about 1.68 m for spring conditions, and the minimum is about 1.0 m for neap conditions (Spaulding and White 1990). The corresponding spring tide tidal prism is estimated to be 59.5 million m^3 , while the neap tidal prism is about 15.8 million m^3 (Chinman and Nixon 1985).

Transport: Based on tidal prism estimates, we estimate that the average tidal transport into (or out of) Mt. Hope Bay during a half-tidal period (6.21 hours) is approximately 6.01 million m^3 /hour. The spring tide average tidal transport is about 9.58 million m^3 /hour, while the neap average tidal transport is about 2.54 million m^3 /hour.

Assuming that the tidal transports through the Mt. Hope Bridge and Sakonnet Passages are proportional to their respective cross-sectional areas (est. 10,000 m^3 and 5000 m^3 , respectively, or 2:1), the tidal average transport through the Mt. Hope Bay Bridge is 4.0 million m^3 /hour; 2.0million m^3 /hour through the Sakonnet Passage section.

Currents: The estimated section-averaged tidal current, based on the transports above, is 11.1 cm/s through both the Mt. Hope Bridge and the Sakonnet Passage sections. By contrast, Spaulding and White (1990) estimated

corresponding section-averaged tidal currents of 20.3 cm/s and 4.6 cm/s (or a 5:1 ratio), using current measurements from single moorings only. The obvious differences can only be explained with further study. These Mt. Hope Bay tidal currents are of course part of the larger Narragansett Bay system.

It is not surprising that M_2 tidal currents dominate tidal currents in the Narragansett Bay system (Figure 2.10). Throughout much of Narragansett Bay the flood/ebb tidal current maxima occur approximately midway between high and low water, while slack waters occur approximately at high and low water (Hicks 1959c). (Alternately, tidal currents lead surface elevation by

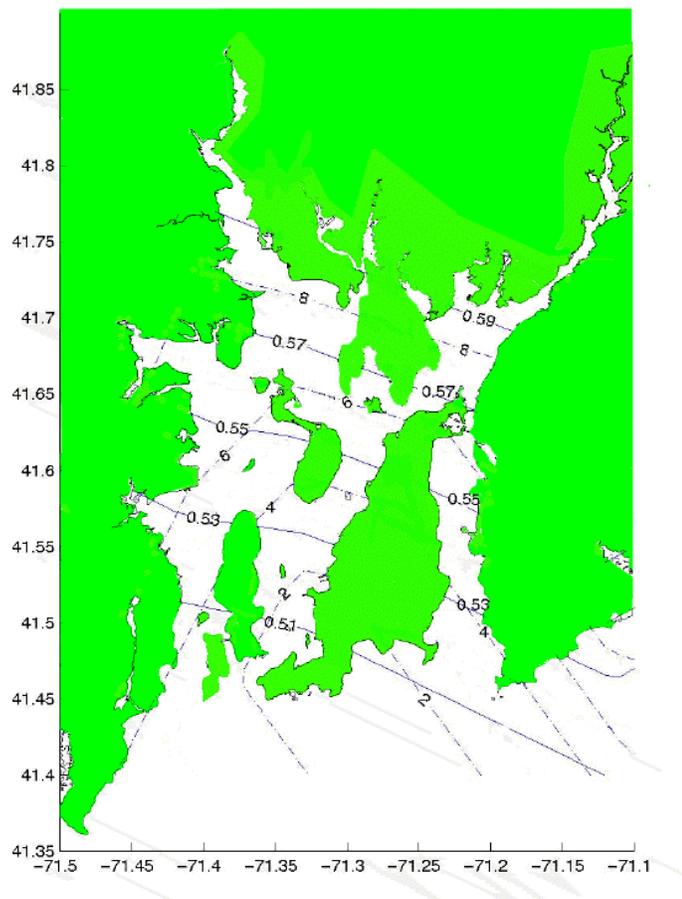


Figure 2.10. Co-amplitude (solid – m) and co-phase (dash – °Greenwich) lines for M_2 (12.42 hours) semidiurnal tide.

approximately 90° (NOAA/NOS 1993 cited in USGen 2001).) This phase relation between tidal currents and sea level is consistent with an approximate standing wave pattern for the M_2 tides in the Narragansett Bay system.

In Mt. Hope Bay, the M_2 tides typically account for 80-90% of the current variance (Spaulding and White 1990). Tidal currents are typically 10-25 cm/s in Mt. Hope Bay, but can reach 2 m/s in the narrow Sakonnet Passage connecting Mt. Hope Bay to the Sakonnet River (Figure 2.11).

4. Temperatures

Short-term temperature measurements in Mt. Hope Bay (Figure 2.12) reveal temporal variability on a number of time scales, including tidal and weather band. (Note both the spring/neap variations and what is probably wind-driven 2-10 day weather band variability in the temperature record.) The usual explanation for such variability is that lateral currents advect quasi-stationary horizontal temperature gradients past the temperature sensors. We explore this conceptual model by first reviewing the annual cycle (i.e., seasonal evolution) in the lateral temperature structure in the Narragansett Bay/Mt. Hope Bay region.

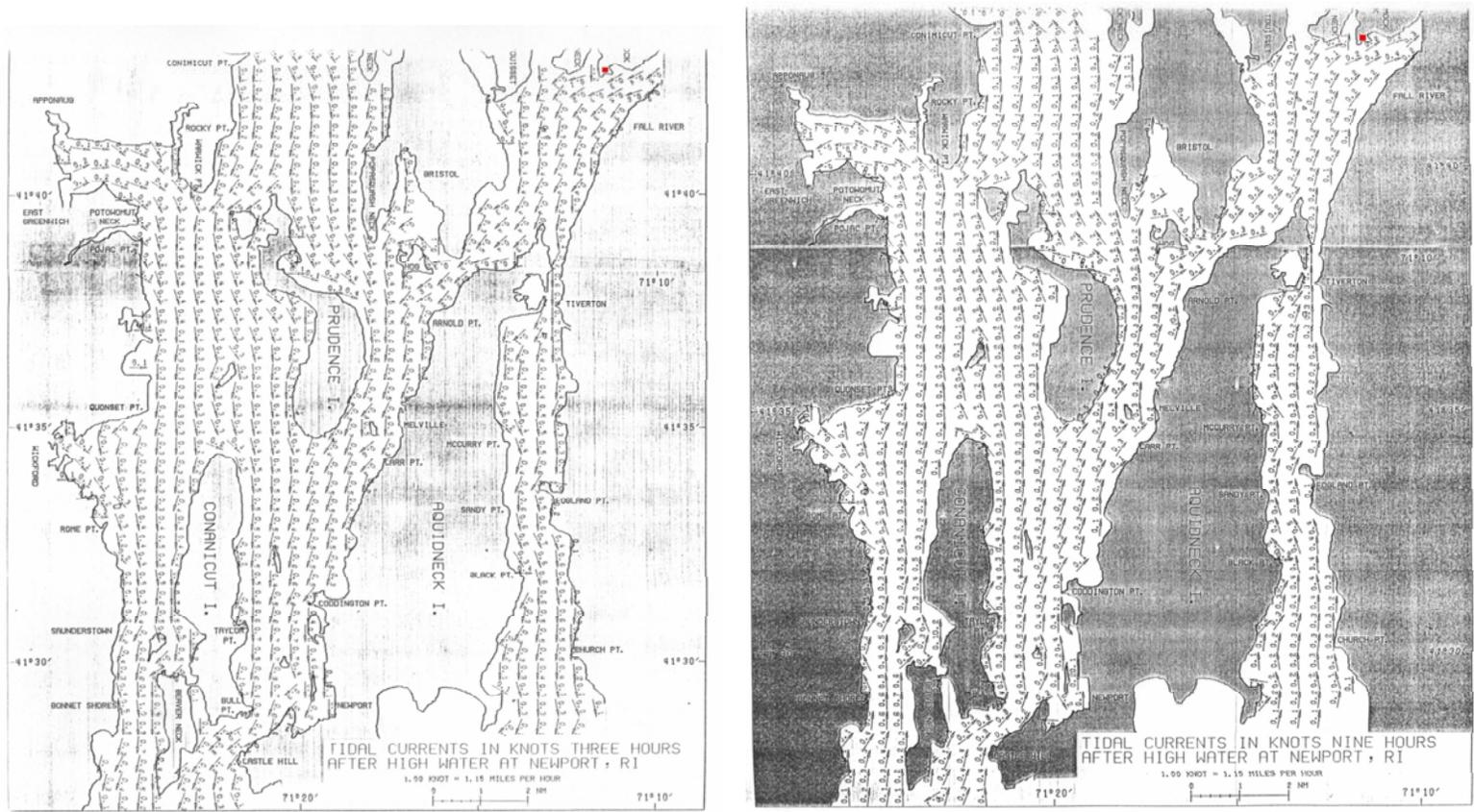


Figure 2.11. A) Ebb tidal currents (knots) in the Narragansett Bay system three hours after high water at Newport, Rhode Island. B) Flood tidal currents (knots) nine hours after high water at Newport, Rhode Island. The red square on the map indicates the location of the Brayton Point Power Plant. (Adapted from Spaulding and Swanson 1984.)

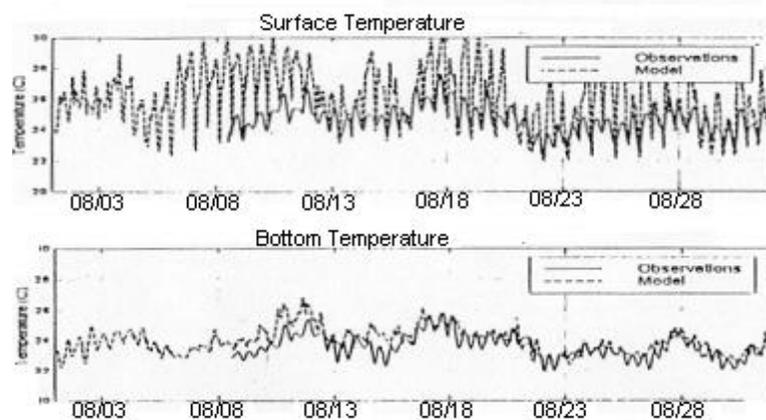


Figure 2.12. Observed (solid) surface (upper panel) and bottom (lower panel) temperature series at Brayton Point mooring station (Station 9 in Figure 2.2) for August 1997. Model temperatures are the dashed lines. (From Spaulding et al. 1999a.)

a. Annual cycle

The annual cycles in the temperature fields of Narragansett and Mt Hope Bays are defined in terms of our presentation of the Hicks (1959c) surface and bottom temperature distributions along the Providence River-East Passage and Mt. Hope Bay-Sakonnet sections (Figure 2.13). (See Figure 2.1 for transect locations.) Both surface and bottom water temperatures in Narragansett Bay are highest in August and lowest in February (Hicks 1959c). The vertically averaged version of these data show that temperatures in the lower part of the Narragansett Bay estuary are strongly influenced by the adjacent coastal ocean water (Figure 2.14). For example, during the winter Narragansett Bay surface temperatures generally increase from the cold waters in the upper bay to those of the slightly milder coastal ocean. The converse occurs during the summer.

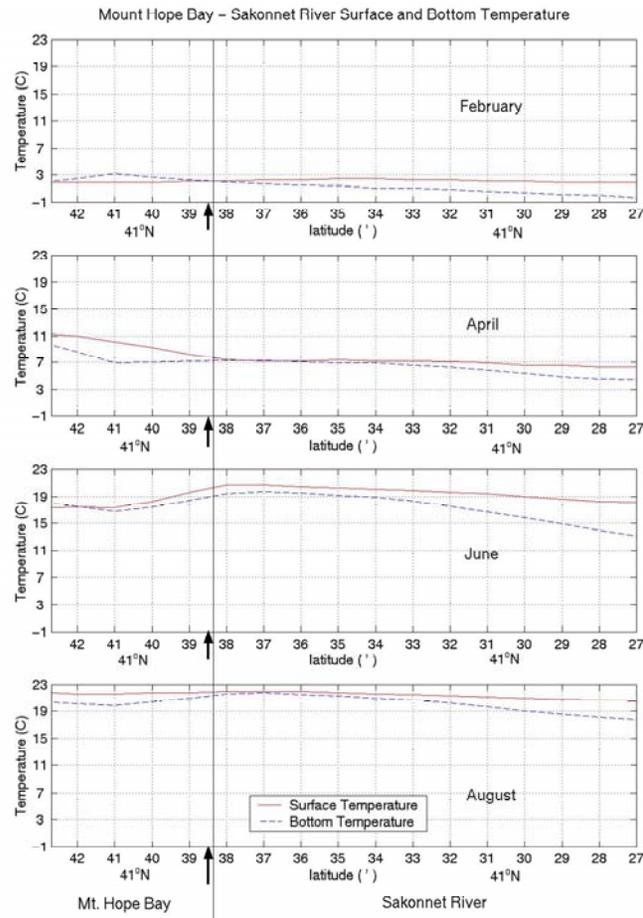
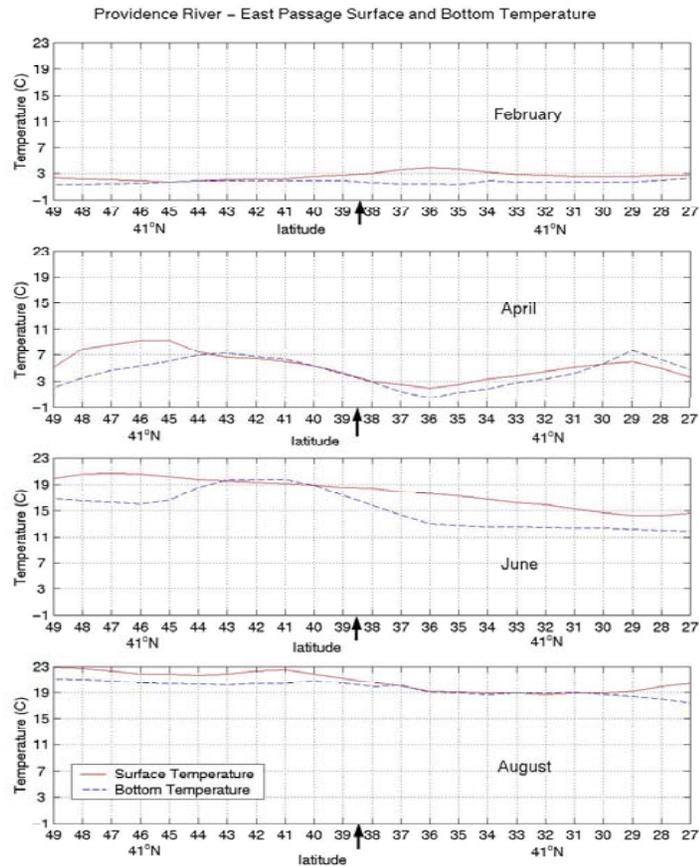


Figure 2.13. Surface (solid) and bottom (dashed) temperature (°C) distributions along A) axes of Providence River-East Passage channels and B) axes of Mt. Hope Bay-Sakonnet River channels (see Figure 2.1) at ±1hour of slack water before ebb during February, April, June, and August, 1956, respectively. The locations are given in minutes of latitude relative to 41°N. The arrow on the panels shows the latitude at which Narragansett Bay connects to Mt. Hope Bay. (Data from Hicks 1959c.)

These results are explained by the fact that the shallower waters of the upper Narragansett Bay lose (and gain) heat more quickly and extensively in the winter (and summer) than do the waters of the deeper coastal ocean. Thus during early spring, Narragansett Bay warms more quickly than do surface temperatures in the ocean. The greatest increase in Narragansett Bay temperatures occurs between April and June. Maximum Narragansett Bay (and ocean) temperatures are reached in mid-August (see Figure 2.15 for horizontal distributions). By late October, Narragansett Bay becomes colder than the ocean.

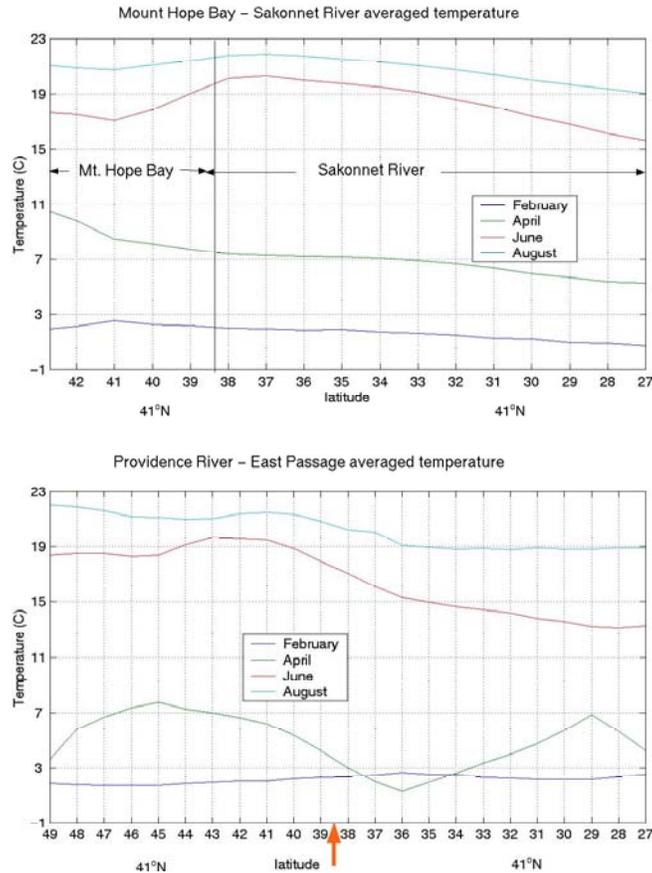


Figure 2.14. Water column average temperature (°C) distribution at ±1hour of slack water before ebb along axes of the (above) Mt. Hope Bay-Sakonnet River (below) Providence River-East Passage channels (Figure 2.1) during February, April, June, and August, 1956, respectively. The locations are given in minutes of latitude relative to 41°N. The orange arrow on the lower panel shows the latitude at which Narragansett Bay connects to Mt. Hope Bay. (Data from Hicks 1959c.)

Mt. Hope Bay is generally warmer than much of Narragansett Bay through the year except in late spring, when Mt. Hope Bay temperatures become lower than the upper Narragansett Bay (Figure 2.13). However, even then, Mt. Hope Bay is warmer than the Narragansett Bay-wide average temperature (Figure 2.14).

b. Tidal variability

The short-term time series measurements of August 1997 surface and bottom temperatures at the Brayton Point Power Station in Mt. Hope Bay exhibit tidal variability throughout the water column (Figure 2.12). This is because during the flood tidal current into Mt. Hope Bay (Figure 2.11), colder water from Narragansett Bay is advected into Mt. Hope Bay and past the temperature sensor. Hence the measured temperature decreases. During ebb tide (out of Mt. Hope Bay), the currents advect the warmer shallow water from the Taunton River and probably some of the warm water discharged from the power station (Figure 2.11) past the sensor. The value of the tidal current in Mt. Hope Bay determines the distance over which Narragansett Bay water is displaced into and out of the Bay (and hence the magnitude of the temperature change).

We can get a sense of the August temperature gradients by studying the Hicks (1959c) pre-power-plant 1956 measurements presented in Figures 2.13-2.15. We note that (1) the surface temperature in Mt. Hope Bay is nearly spatially uniform at about 21.7°C (71°F); and (2) the bottom temperature decreases from 21.7°C (71°F) in the south to about 20°C (68°F) in the middle of the Bay and then increases northward up the Taunton River. Because these temperatures were measured near high water (i.e., within an hour of ebb), this cooler water was

probably advected into Mt. Hope Bay by the flood tidal current (i.e., a 0.23°C decrease at Brayton Point). On the ebb in this situation, we would expect a 0.57°C increase at Brayton Point. But the measurement in Figure 2.12 shows a 2°C range between ebb and flood; this increase in temperature range during a tidal cycle could be the result of the addition of heat to the Bay by the power plant.

c. Weather-band variability

Note the presence of significant non-tidal variability in the Figure 2.12 temperature measurements—some of which could be due to wind forcing. If no upwelling or mixing existed in the Bay, the water in the Bay would be always stratified, with warmer water at surface in the summertime and colder water at the surface in wintertime. But of course this is not always true. From the Hicks (1959c) surface and bottom temperature distributions data (Figure 2.13), we can see surface temperature records and bottom temperature records cross with each other in any season, which indicates that wind-induced upwelling and mixing redistributed water over the entire water column. Wind can also induce lateral mixing, and the relatively uniform temperature distribution in the whole Bay in February may indicate a very thorough lateral mixing caused by wind.

d. Interannual variability

Hicks (1959c) estimated the temperature variation from year to year by comparing the surface and bottom temperatures from measurements made in August-September 1951, August-September 1952, and August 1956, respectively. He found that the average (the literature did not indicate what kind of average he referred to) surface temperature in late summer of 1956 was 1.11°C cooler than

that in 1951; and the bottom water temperature was 0.72°C cooler in 1956 than in 1951. He also found that the 1956 surface temperature was 2.61°C cooler than in 1952, and that the 1956 bottom temperature was 0.5°C warmer than in 1952.

Hicks (1959c) also concluded from the data that the inter-year late summer temperature differences decreased from about 2.22°C at the entrance to the West and East Passages to about 0.39°C at the mouth of the Providence River. Thus it would appear that temperature variations from year to year are due to changes in the coastal waters and/or direct meteorological effects rather than to differences in temperatures of the river contribution.

e. Recent measurements

The above temperature measurements were made before the Brayton Point Power Station was built. More recent Narragansett Bay-wide temperature distributions have been measured and studied. For example, a field study was conducted in Mt. Hope Bay and the lower Taunton River by Spaulding and White (1990) to describe the circulation in response to tide, wind and density-induced forcing. Surface and bottom measurements were made at three stations: Mt. Hope Bridge, Brightman St. Bridge, and the entrance to the Sakonnet River. At any given time of year, the water column at these specific locations within Mt. Hope Bay was found to be relatively well mixed (Spaulding and White 1990). These three sites represent the perimeter of Mt. Hope Bay, including the mouth of the Taunton River (Brightman St. Bridge) and the two regions of exchange between Mt. Hope Bay and greater Narragansett Bay (Mt. Hope Bridge and Sakonnet River). Assuming enhanced velocities at these locations due to river flow and

tidal exchange processes, it is reasonable to assume that these locations may be well mixed while stratification may persist within the interior of Mt. Hope Bay.

Recently, the thermal effluent discharged from the Brayton Point Power Station has been tracked, described and modeled. In particular, the relationship between Brayton Point thermal effluent and Mt. Hope Bay temperature has been investigated by a remote sensing group at Brown University. Mustard et al. (1999) analyzed the seasonal variability of surface temperatures in the Narragansett Bay region from a composite of 14 thermal infrared satellite images

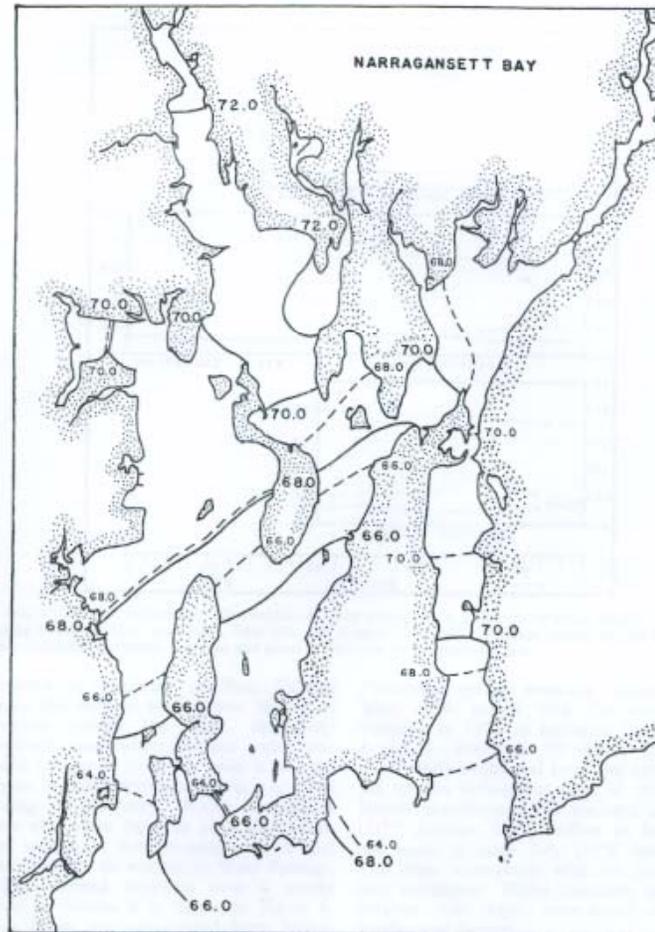


Figure 2.15. Horizontal distributions of surface (solid) and bottom (dashed) temperature (°F) distributions at ±1 hour of slack water before ebb for 6-10 August 1956. (Data from Hicks 1959c.)

(Landsat TM Band 6) with a spatial resolution of 120 m. Using the technique of unsupervised classification, they demonstrated that surface temperatures in Mt. Hope Bay were warmer than surface temperatures in other regions of Narragansett Bay that have comparable surface-to-volume ratios and tidal exchange (e.g., Upper Narragansett Bay). The temperature difference was found to be seasonal, peaking during late summer and autumn, with an average surface temperature difference between Mt. Hope Bay and Upper Narragansett Bay on the order of 0.8°C. Comparison of seasonal temperature cycles between the two regions suggests that surface temperatures within Mt. Hope Bay generally decrease with distance from the BPPS (Figures 2.16, 2.17).

The remote sensing characterizations of Mt. Hope Bay temperature are limited by the nature of remote sensing infrared instrumentation, which detects the temperature of only the upper few millimeters (the "sea skin temperature") of

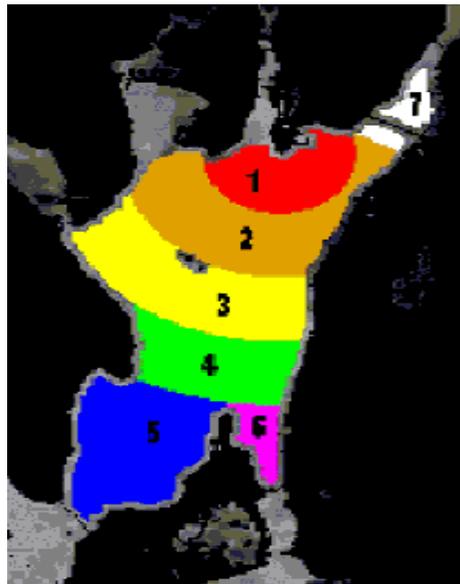


Figure 2.16. Study areas within Mt. Hope Bay. Segments 1-4 defined by 1.4-km radius from Brayton Point Power Station (Carney 1997).

a body of water. Correlation of this surface skin temperature with sub-surface temperatures is expected to decrease with water column depth. In a study comparing remotely sensed Mt. Hope Bay sea skin temperatures to in situ temperatures from depth, the correlation between surface and 1m temperatures was 0.8, while correlation between the surface and 4 m fell to 0.38 (Dave 1998). Although some studies have suggested that sea skin temperatures can be characteristic of bulk water temperatures under certain conditions (e.g., Schneider and Mauser, 1996), the presence of a positively buoyant heated discharge can result in a surface-trapped plume, with surface temperatures significantly higher than the majority of the water column. This would indicate that caution is in order prior to making assumptions regarding well-mixed conditions in Mt. Hope Bay. Given the possible effects of thermal effluent on Mt. Hope Bay organisms, especially demersal fishes, characterization of the thermal plume over depth is a necessary component of a full ecosystem analysis.

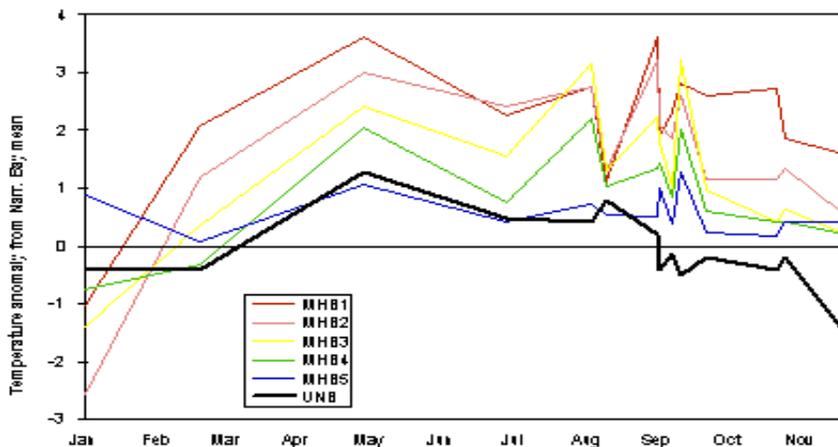


Figure 2.17. Seasonal temperature signals of Mt. Hope Bay sections (see Figure 2.15) distributions derived from Landsat (TM Band 6) satellite infrared images with a spatial resolution of 120 m. The upper Narragansett Bay temperatures are given for comparison. Note that the main body of Mt. Hope Bay (sections 1-4) is warmer than upper Narragansett Bay from February through December (Carney 1997).

Also, during February 1999, ASA measured temperatures directly with thermistor chains deployed throughout Mt. Hope Bay at the sites indicated in

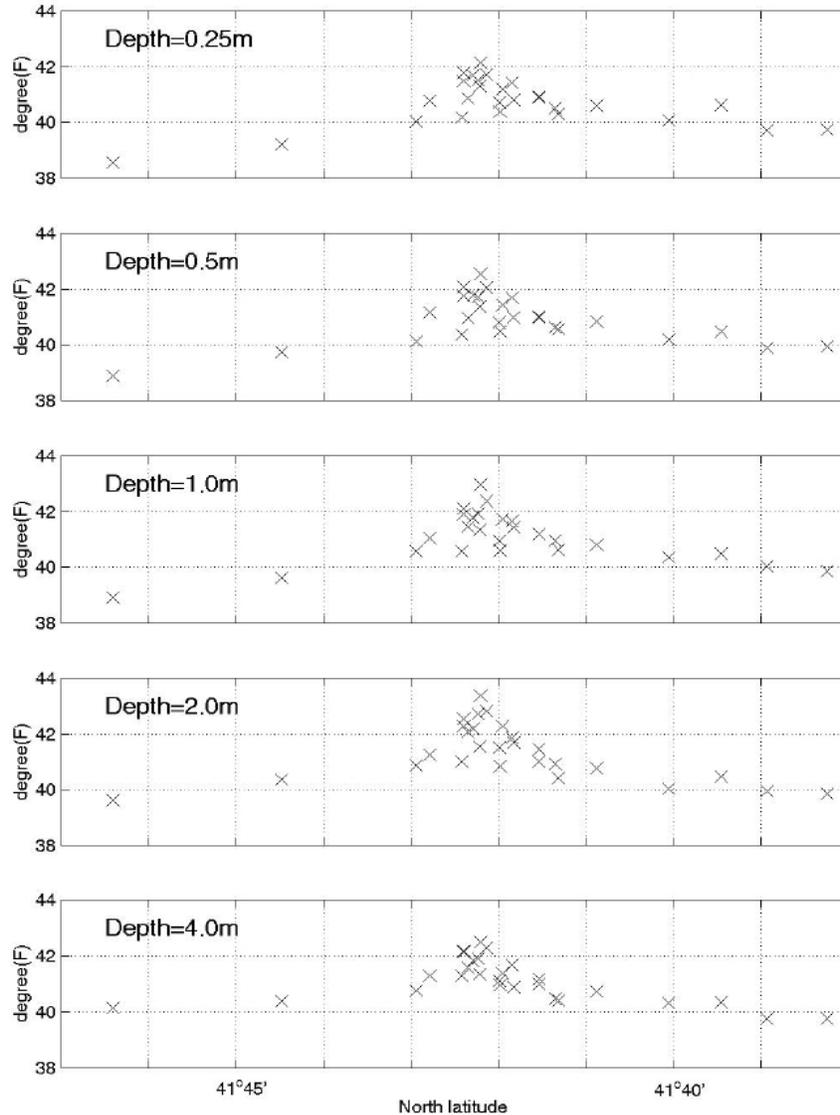


Figure 2.18. Series mean temperatures at 5 different depths at different stations (Figure 2.2) in Mt. Hope Bay during February 1999. The Brayton Point Power Plant cooling water outlet is located at about 41° 43' (Swanson et al. 1999).

Figure 2.2 (Swanson et al. 1999). The mean temperatures—plotted as a function of latitude in Figure 2.18—show a clear signature of the power plant cooling water.

5. Salinity

Short-term salinity measurements in Mt. Hope Bay (Figure 2.19) reveal temporal variability on a number of time scales including tidal, weather-band, and river discharge-related variability. The usual explanation for tidal and weather-band variability is that lateral currents advect quasi-stationary horizontal salinity gradients past the conductivity (i.e., salinity) sensors. We explore this conceptual model by first reviewing the annual cycle (i.e., seasonal evolution) in the lateral salinity structure in the Narragansett Bay/Mt. Hope Bay region.

a. Annual cycle: Salinity and vertical stability

The seasonal variation in the along-channel surface and bottom salinity distributions (Figure 2.20) in Mt. Hope Bay and Narragansett Bay are derived from the Hicks (1959c) measurements. (See Figure 2.1 for the transect locations). These distributions reflect the tidal mixing of coastal ocean waters from the south, with river inflows to the heads of Mt. Hope Bay and Narragansett Bay. The significant seasonality of river discharges (see Figure 2.7) is reflected in the minimum salinities at the heads of these respective branches (Figure 2.20).

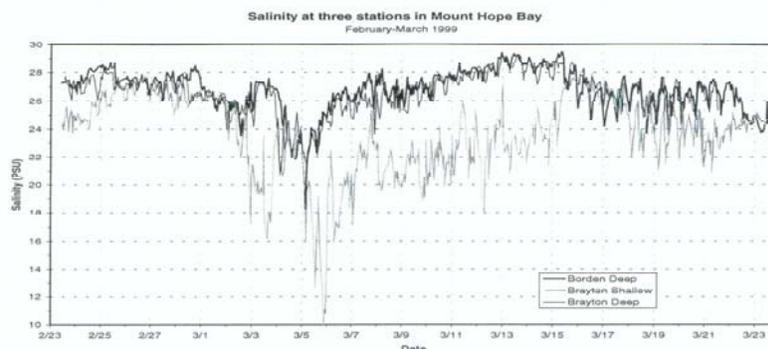


Figure 2.19. Salinity records in Mt. Hope Bay during February-March 1999; near-surface and bottom at Brayton Point (station 9; Figure 2.2) and near-bottom at the Borden Flats (station 31; Figure 2.2) (from Swanson et al. 1999).

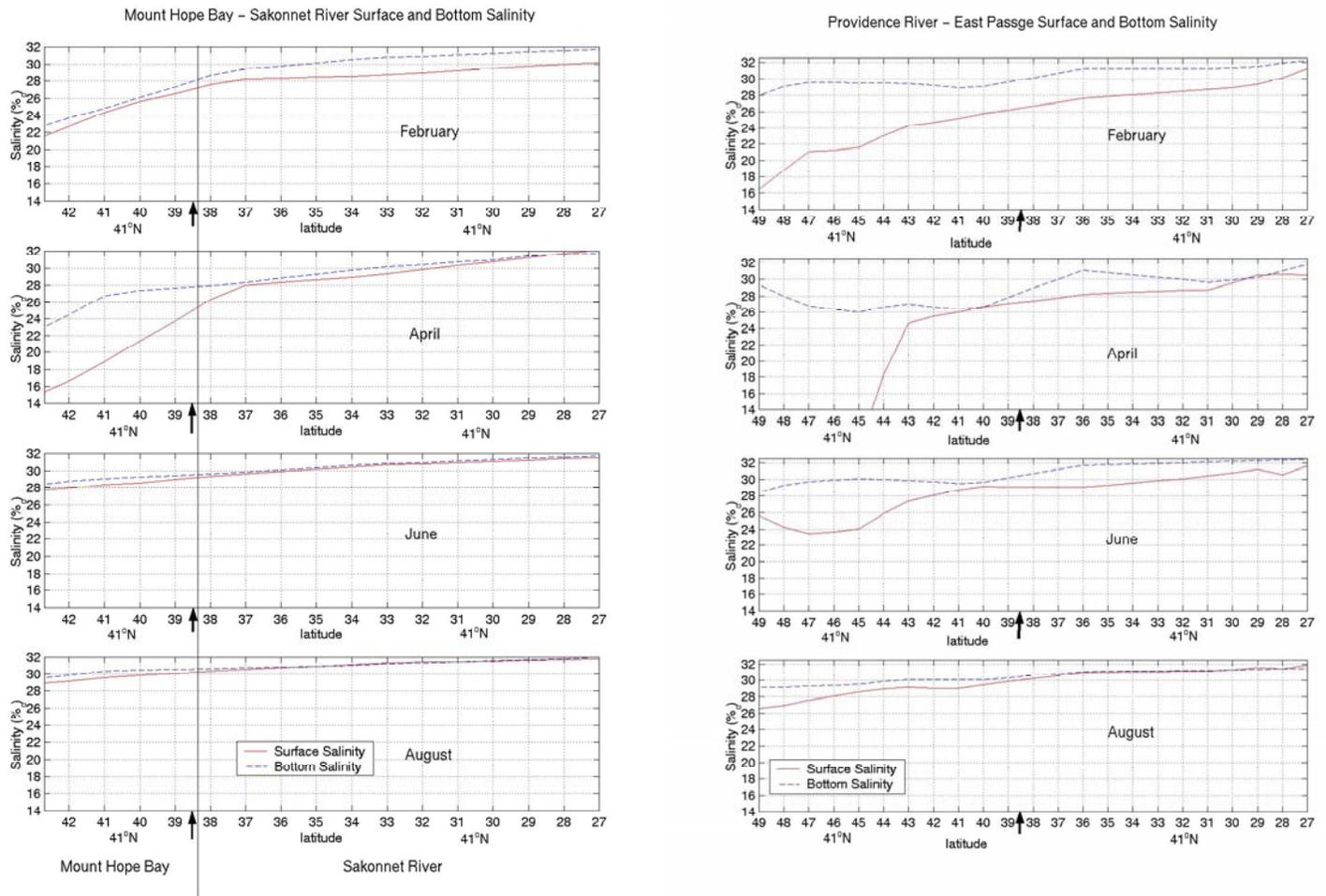


Figure 2.20. Surface and bottom salinity (‰) distribution at slack before ebb ± 1 hour along A) axes of Mt. Hope Bay-Sakonnet River channel and B) axes of the Providence River-East Passage channel (see Figure 2.1) during February, April, June, and August, 1956. The locations are given in minutes of latitude relative to 41°N. The arrow on the panels shows the latitude at which Narragansett Bay connects to Mt. Hope Bay. (Data from Hicks 1959c.)

Because most of the river discharge is in spring, the average water salinity in the Narragansett Bay system is lowest in April and highest in August.

In the Narragansett Bay/Mt. Hope Bay system (like other partially mixed estuaries), the more saline (and denser) coastal ocean waters from the south tend to underlie the less saline (and less dense) surface waters from the north. The actual details of the lateral and vertical distributions of salinity in the Narragansett Bay system during a particular season are strongly influenced by the ratio of the

volumes of freshwater inflow and tidal current inflow (i.e., the tidal prism) (e.g., see Figure 2.22 for an August salinity distribution). The highest river discharges during the year occur in March and April as the result of the relatively large precipitation and snowmelt. Thus the salinities at the head(s) of the Narragansett Bay system during spring are usually much lower than at other times during the year (see Figure 2.21). The relatively low river discharges during summer result in the salinity maxima during August.

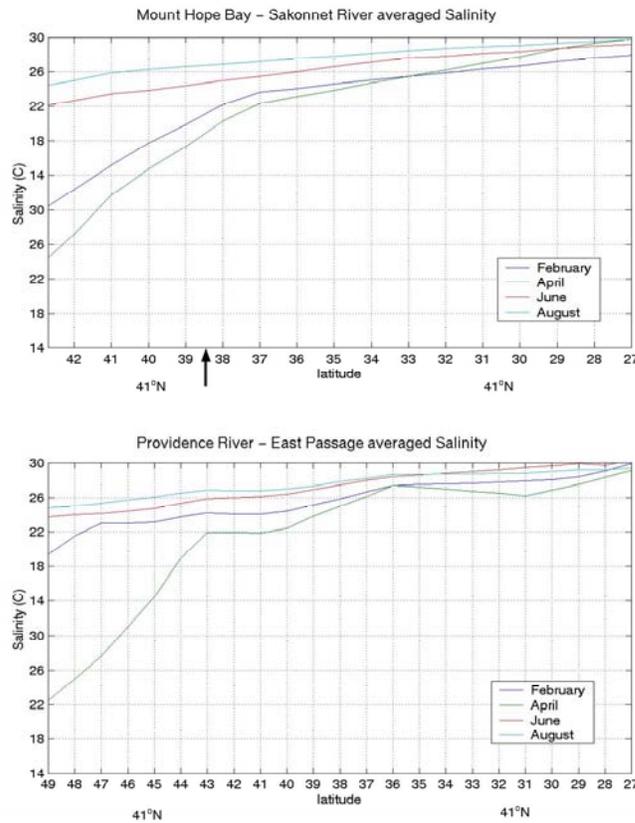


Figure 2.21. Water column average salinity (‰) distribution at slack before ebb ± 1 hour along axes of channels (Figure 2.1) during February, April, June, and August, 1956. The locations are given in minutes of latitude relative to 41°N. The arrow on the upper panel shows the latitude at which Narragansett Bay connects to Mt. Hope Bay. (Data from Hicks 1959c.)

The salinity distributions dominate density structure, particularly the vertical gradients and hence stability, in the Narragansett Bay system, as shown by both the Hicks (1959c) and Weisberg (1976) measurements. The strength of this vertical stability of the water column can be expressed in terms of the buoyancy (or Brunt-Vaisala) frequency. It can be seen in Figure 2.23 that the stability of the water column generally increases from the ocean entrances to Narragansett Bay to the respective heads in the Providence and Taunton Rivers. This increase is more obvious in the Providence River-East Passage section for all seasons because the Blackstone River discharges about three times more fresh water into the Providence River than the Taunton River discharges into Mt. Hope Bay, especially in April.

b. Tidal variability

The Swanson et al. (1999) salinity time series measurements in Mt. Hope Bay during February and March 1999 (Figure 2.18) exhibit tidal variability throughout the water column. Given the typical along-channel salinity gradients (Figures 2.19 and 2.20) and tidal current magnitudes (Figure 2.10), we would expect salinity fluctuations of about 5.8 psu. Of course, estimated values vary

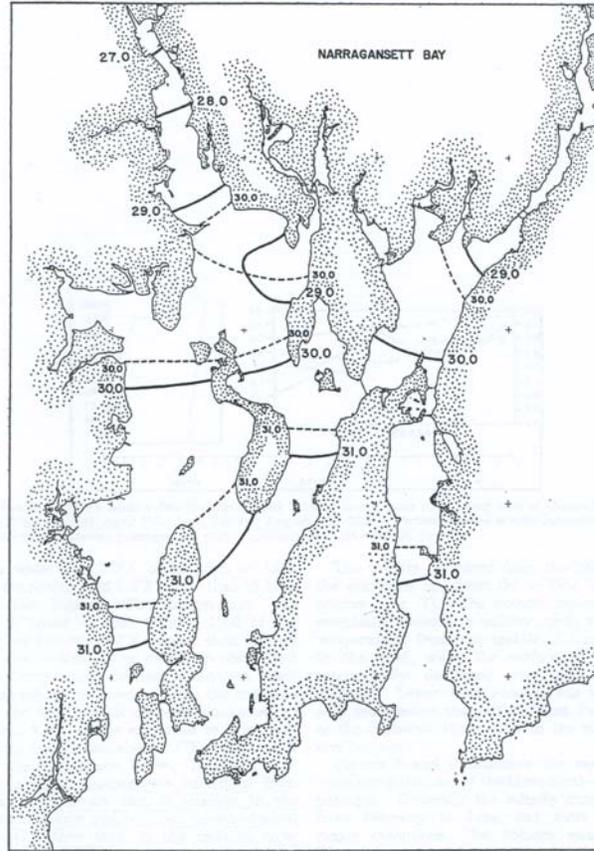


Figure 2.22. Horizontal salinity (‰) distribution at slack before ebb ± 1 hour during cruise 19 (6-10 August 1956). Surface – solid lines and large numerals); bottom – dashed lines and small numerals. (Reprinted from Hicks 1959c.)

with the stage in the spring-neap cycle. These estimates compare with the observed tidal fluctuation in salinity magnitudes of 2-10 psu, clearly seen shown in the observations.

Besides advecting horizontal salinity gradients, tidal currents in narrow passages can also cause vertical mixing. The latter process might explain the very small vertical differences in salinity in the June and August (Figure 2.20, left panel) Sakonnet River sector of the Mt. Hope Bay/Sakonnet River salinity

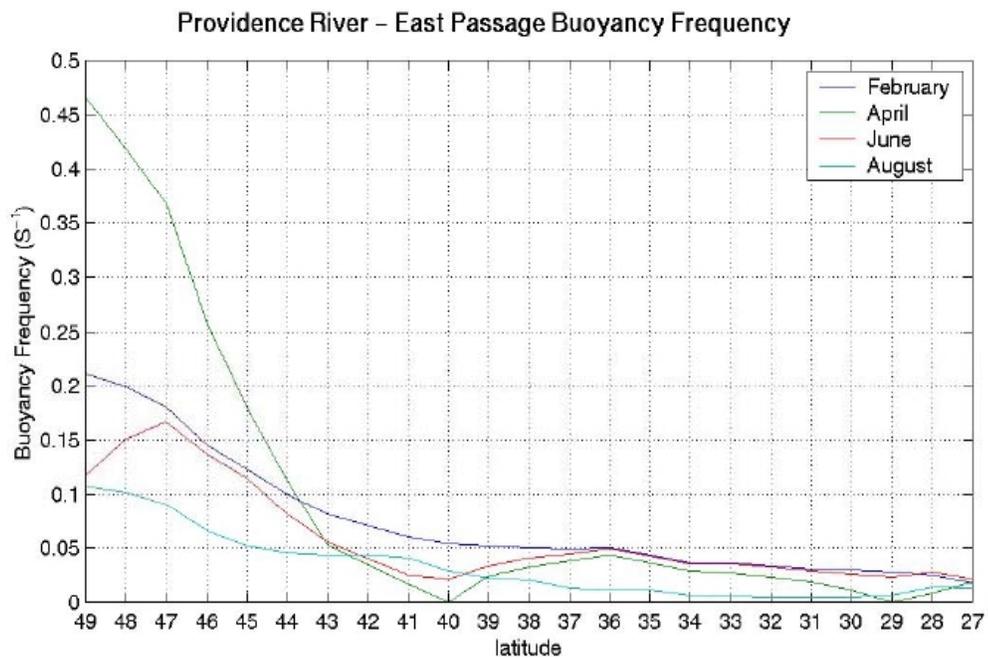
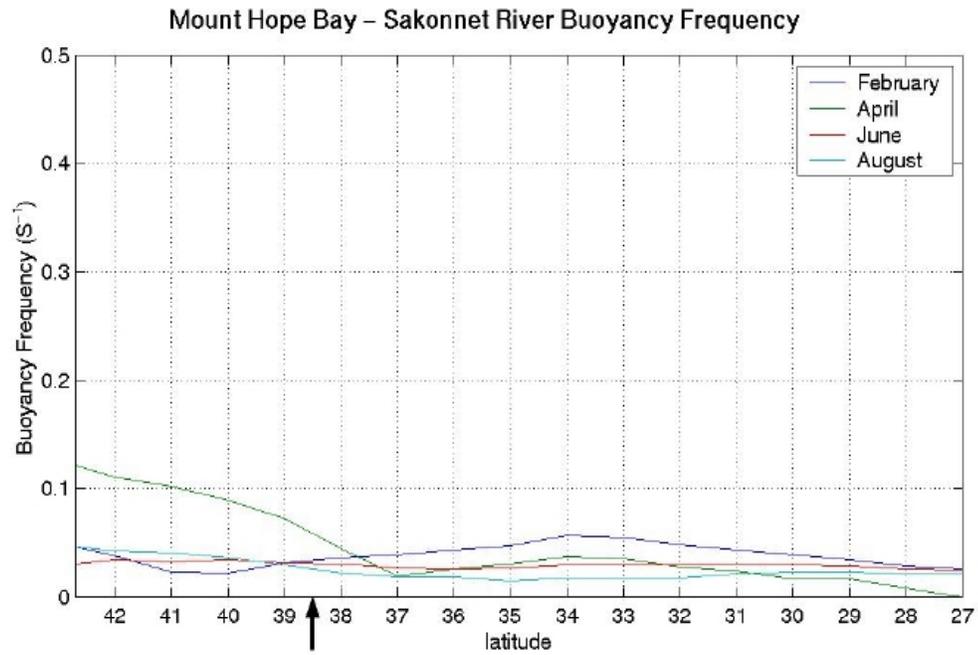


Figure 2.23. Water column vertical stability distribution in terms of buoyancy frequency at slack before ebb ± 1 hour along axes of channels (Figure 2.1) during February, April, June, and August, 1956. The arrow on the upper panel shows the latitude at which Narragansett Bay connects to Mr. Hope Bay. (Data from Hicks, 1959c.)

distribution. It may be that the strong alternating flood and ebb tidal currents in the Sakonnet Passage may homogenize the water that resides there.

c. Weather band

During February-March 1999, surface salinity varied from 28 to 10 ppt (Figure 2.19), with the lower values probably due to a major precipitation and/or snow melt runoff event. During the same time, bottom salinity records decreased from about 29 to 20 ppt. The surface salinity record, however, was more complex than the near-bottom record, probably because of the vertical mixing effects of wind and perhaps tidal currents.

d. Interannual variability

Salinity variations from year to year appear quite small in comparison with longitudinal or seasonal variations. A comparison of Hicks (1959c) average data indicates that the late summer 1956 bottom water was 0.2‰ less saline than that in 1951. However, the 1956 surface and bottom salinities were greater than their 1952 counterparts by 0.7‰ and 0.2‰, respectively.

6. Nontidal Currents

Weisberg (1976) examined the effects of wind, atmospheric pressure and river inflow variability on the nontidal flow in the Providence River section of Narragansett Bay. The study focused on a 51-day (18 October - 9 December 1972) Geodyne model 850 current record 2 m above the bottom (the mean low water depth is 12.5 m) near the entrance to the Providence River (Figure 2.1). He reported along-channel current mean and variance of 11.7cm/s (landward) and

166.9 cm/s², respectively. The semidiurnal tides contributed 45% of the total variance, while 7% was due to higher tidal harmonics. The rest, 48%, resided at subtidal frequencies and was due to nontidal forcing. Weisberg (1976) found that 97% of the along-channel nontidal current variance in the most energetic portion of the spectrum (4-5 day periods) was coherent with, and lagged by about 4 hours, the Green Airport wind velocity component in the direction of maximum fetch, which is 335° to the north.

Spaulding and White (1990) conducted a field study between December 1985 and January 1987 in Mt. Hope Bay and the lower Taunton River to determine the response of the circulation to tide, wind and density induced forcing. Wind data were obtained at hourly intervals from the National Weather Service Station at T.F. Green Airport, Warwick, RI. Surface and bottom current meters were deployed (typical duration of 60 days) at the Mt. Hope Bridge, at the Brightman St. Bridge in Fall River, and at the entrance to the Sakonnet River (at Hummocks near the Sakonnet River Bridge) (Figure 2.2). The hydrography showed that the water column is well mixed throughout Mt. Hope Bay.

Spaulding and White (1990) summarized and averaged the peak coherences found in the wind-frequency ranges (30-hour low-pass filtered) for seven Brightman and eleven Mt. Hope meter records. They concluded from the results that the overall effect of wind excitation on Mt. Hope Bay currents is very small.

Spaulding and White's (1990) conclusion was obviously inconsistent with Weisberg's (1976) conclusion concerning wind (measured at T.F. Green Airport) forcing of currents. There are several possible reasons for this inconsistency:

1) Providence River is much closer to T.F. Green Airport than is Fall River.

According to Pilson (1991), local wind conditions can vary significantly from location to location in Narragansett Bay. Thus it is not surprising that winds measured at T.F. Green airport might be statistically unrelated to water movements at Brightman St. Bridge, which is located in a narrow upstream section of the Taunton River in Fall River.

2) The two sets of observations were made in different seasons.

The Weisberg (1976) analysis was based on measurements done in autumn, while Spaulding and White's (1990) analysis was based on results averaged over the whole year. Climatological records (Figures 2.4 and 2.5) show that winds differ considerably over the year. It is also clear from Figure 2.7 that Taunton River discharge is relatively large and highly variable on the 2-10 day weatherband time scales between March and June. Hence the Taunton River discharge variability could dominate the flow variability, thus causing reduced wind/current coherence on an annual time scale.

3) Current measurements in the two studies were at different relative depths.

The Weisberg (1976) currents were 2 m off the bottom in 12.5 m of water, while Spaulding and White (1990) averaged their surface, mid-depth and bottom current measurements.

We conclude that there is a lack of a comprehensive study of wind-forced currents in Mt. Hope and Narragansett Bays. Thus we recommend that we conduct a

comprehensive statistical analysis of the PORTS winds and currents during the early stages of the design phase of the Mt. Hope Bay Natural Laboratory.

Spaulding and White (1990) note that the long-term average flow of the Taunton River in Fall River (near Brightman St. Bridge) is consistent with a classical stratified estuarine circulation pattern: that the surface flow is down-river and bottom flow is upriver. They also found that the long term flows through the Sakonnet Passage and Mt. Hope Bridge sections were consistently out of Mt. Hope Bay. According to their measurements, the outflow through the Mt. Hope Bridge section was approximately $1000 \text{ m}^3/\text{s}$, and the outflow through the Sakonnet River Bridge section is was approximately $150 \text{ m}^3/\text{s}$, which means the outflow through East Passage is more than 7 times the outflow through the Sakonnet River. The averaging period for these results is not clear from the literature, but these data could imply that there must be residual inflow at other places along the Mt. Hope Bridge and Sakonnet River sections which were not measured at the deployment time. Those inflows could be possibly associated with eddy currents associated with the topography there.

C. Summary of Results

This report outlines a basic understanding of many of the important aspects of the physical environment of Narragansett/Mt. Hope Bay, including sea level, currents, temperatures and salinities (densities) and how they respond to tidal, wind and river discharge forcing.

The tides in Narragansett Bay are dominated by the M_2 semidiurnal constituent and have approximate standing wave characteristics. The M_2 constituent is the largest, accounting for 90% of sea level variance and 80-90% of current variance in Mt. Hope Bay. The tidal prism in Mt. Hope Bay accounts for 1/5 of the water volume in the Bay. The tidal average transport through Mt. Hope Bay is two times that through the Sakonnet Passage section.

Blackstone River discharge accounts for about half of the fresh water inflow in the Narragansett Bay region. Taunton River discharge accounts for more than 90% of fresh water inflow to Mt. Hope Bay. All the rivers in the Narragansett Bay region have the highest discharge in March and April and lowest discharge in July to September.

Both surface and bottom water temperatures in Narragansett Bay are highest in August and lowest in February. Mt. Hope Bay is generally warmer than much of the Narragansett Bay area during the spring, summer and fall. Recent studies and measurements show clear signatures of the power plant's influence on the Bay's temperature field.

Salinity distributions in Narragansett Bay have typical river-to-ocean salinity gradients. The average salinity in the Narragansett Bay system is lowest in April and highest in August. The actual details of the lateral and vertical distributions of salinity in the Narragansett Bay system during a particular season are strongly influenced by the ratio of the volumes of freshwater inflow and tidal current inflow.

D. Recommendations for Future Work

There are still, however, several important aspects that need further work if we are going to build a model-based Mt. Hope Bay Natural Laboratory. There is a need to:

- **Analyze the available long-term PORTS time series measurements of the basic physical variables and obtain additional new records from “critical locations.”**

Most measurements to date have been short-term. ASA has done some detailed thermistor chain measurements in Mt. Hope Bay (31 stations with sensors at several levels from surface to bottom in the water column), but for only about one month. These temperature measurements (and some corresponding salinities) are not very well analyzed. Only visual analysis was done to show the heat-loading effects from the Brayton Point Station plume. No time series analysis results are available from their results. Also, we can obtain only some tidal frequency band variations from analyzing these one-month-long records. In order to determine weather band, seasonal, and interannual variations in temperature, we need much longer observation records. Fortunately, there is a growing archive of PORTS observations. An analysis of these data could provide important insight into the long-term variability in the bays and thus a physical context for evaluating the effects of different kinds of anthropogenic influences. We have already obtained the PORTS Fall River time series measurements of water temperature, air

temperature and sea level for October 1999 to date. Table 2.2 details these and the other available time series data.

- **Obtain a modern system-wide description of the annual evolution of hydrographic properties and hence physical stability structure of the bays.**

The only systematic measurements of the annual evolution of the structure of the water properties in the Narragansett/Mt. Hope Bays system are those data from 1957 described by Hicks (1959c).

- **Clearly describe the evolution of the tidal, wind-driven and density-driven circulation patterns in the bays.**

The variability in the wind-driven and density circulation determine how the pollution and nutrients are transported in the bays over time scales much longer than the daily tidal time scales. There is a need to know how these patterns differ with (a) strong and weak wind-forcing; (b) spring and neap tidal stage; (c) large and small river discharges; (d) season. Answers to these questions are an important part of the definition of the natural habitats that support the ecosystems in the bays.

Chapter 3. Habitats and Habitat Quality (Brian Howes & Miles Sundermeyer)

A. Habitats

Research aimed at the identification of fish nursery habitats has become increasingly important to the management of both marine fishes and the coastal habitats themselves. The importance of this type of research has recently been emphasized by the adoption of Essential Fish Habitat (EFH) provisions by the U.S. Congress as part of the reauthorization of the Magnuson-Stevens Fishery Conservation and Management Act (Oct. 1996). Essential Fish Habitats are defined as "those waters and substrate necessary for fish for spawning, feeding or growth to maturity." The reauthorization of the Magnuson-Stevens Fishery Conservation and Management Act in 1996, therefore, has led to an explosion in research aimed at identifying fish habitats. The most rapid advances in this area have arguably been made towards defining habitats necessary for juvenile fish growth in estuarine habitats; less emphasis has been placed on identification of spawning habitats (Kneib 1997, Able 1999, Deegan et al. 2001).

Mt. Hope Bay is composed of many types of contiguous and trophically interconnected habitats. Classification of habitat types is somewhat subjective and dependent on individual perspectives, and can be based on geology, geography, dominant flora, or sediment types. Important broad habitat types include the open bay, salt marshes, freshwater marshes and rivers. The open bay habitats can be delineated into water column, benthos, shoreline and shoal habitats, each of which can further be delineated by sediment type, flora and

hydrography (e.g., mud flats, sand flats, eelgrass and macroalgae beds). Salt marshes can be divided into tidal zones, salinity zones and dominant plant type.

Ongoing studies are currently beginning to map major habit type distributions in Narragansett Bay (Huber 1999, RIDEM 2001). The estimated total acreage of 14 broad habitat types for Narragansett Bay based on a preliminary summary of 1996 mapping data is provided in Table 3.1. However, other than eelgrass beds subtidal habitats are poorly represented because they are not well accessed with aerial photography. Aerial maps exist for Mt. Hope Bay, but were unavailable for our review and have not been specifically summarized to date.

Table 3.1. Summary of estuarine and marine habitat acreages for Narragansett Bay in 1996. (Adapted from Huber 1999.)

| Habitat Type | Area in Acres |
|------------------------------------|----------------------|
| Open water | 124,259 |
| High scrub-shrub marsh | 159 |
| High salt marsh | 2,709 |
| Pannes and pools | 46 |
| Low salt marsh | 443 |
| Brackish marsh | 428 |
| Stream beds | <4 |
| Dunes | 43 |
| Beaches | 1,450 |
| Rocky shores | 573 |
| Tidal flats | 569 |
| Eelgrass beds | 100 |
| Artificial jetties and breakwaters | 23 |
| Oyster reefs | 9 |

Eelgrass habitat has long been known to be one of the most important fish nursery habitats (e.g., Heck and Thoman 1984), but is arguably the most endangered. Once extensive eel grass beds in estuaries along the entire east coast

of the U.S. were decimated by a fungal blight in the 1930s which may have eliminated as much as 90% of the habitat (Thayer and Fonseca 1984, Short et al. 1993). Unfortunately, little recovery has occurred in most areas. Worse, in the last decade the remaining eelgrass beds on the East Coast have suffered further dramatic declines due to nutrient loading effects (e.g., Valiela et al. 1992, Short et al. 1996). Narragansett and Mt. Hope Bays have not been spared these impacts. Less than 100 acres of eelgrass beds remain in Narragansett Bay today (Figures 3.1 and 3.2). As evidence for the nutrient loading impact, all of the once-extensive beds in upper Narragansett Bay (Figures 3.1 and 3.2), including the entire Mt. Hope Bay, have been lost (Rines 2001 cited in PG&E 2001). Beds in the lower bays have also suffered serious declines. Eelgrass habitat has been largely replaced by open mud flats and macroalgae fields. The functional value of these habitats compared to eelgrass is unclear, though macroalgae may serve as a suitable replacement for some fishes such as tautog (i.e., *Ulva*; Sogard and Able 1991, 1992). The long-term impact of fragmentation and loss of eelgrass habitat to regional fisheries is unknown. Because of the importance of this habitat, several restoration projects have been conducted, are ongoing, or are planned for the future in Narragansett Bay. However, the long-term success of these efforts is doubtful unless the underlying cause of eelgrass loss is addressed.

Saltmarshes (including brackish marshes) are thought to provide major trophic support and nursery habitats for our fishery species (e.g., Kneib 1997, Able 1999, Deegan et al. 2001, Weinstein and Keeger 2001). But they too are greatly endangered in the Narragansett and Mt. Hope Bay systems. In general, it

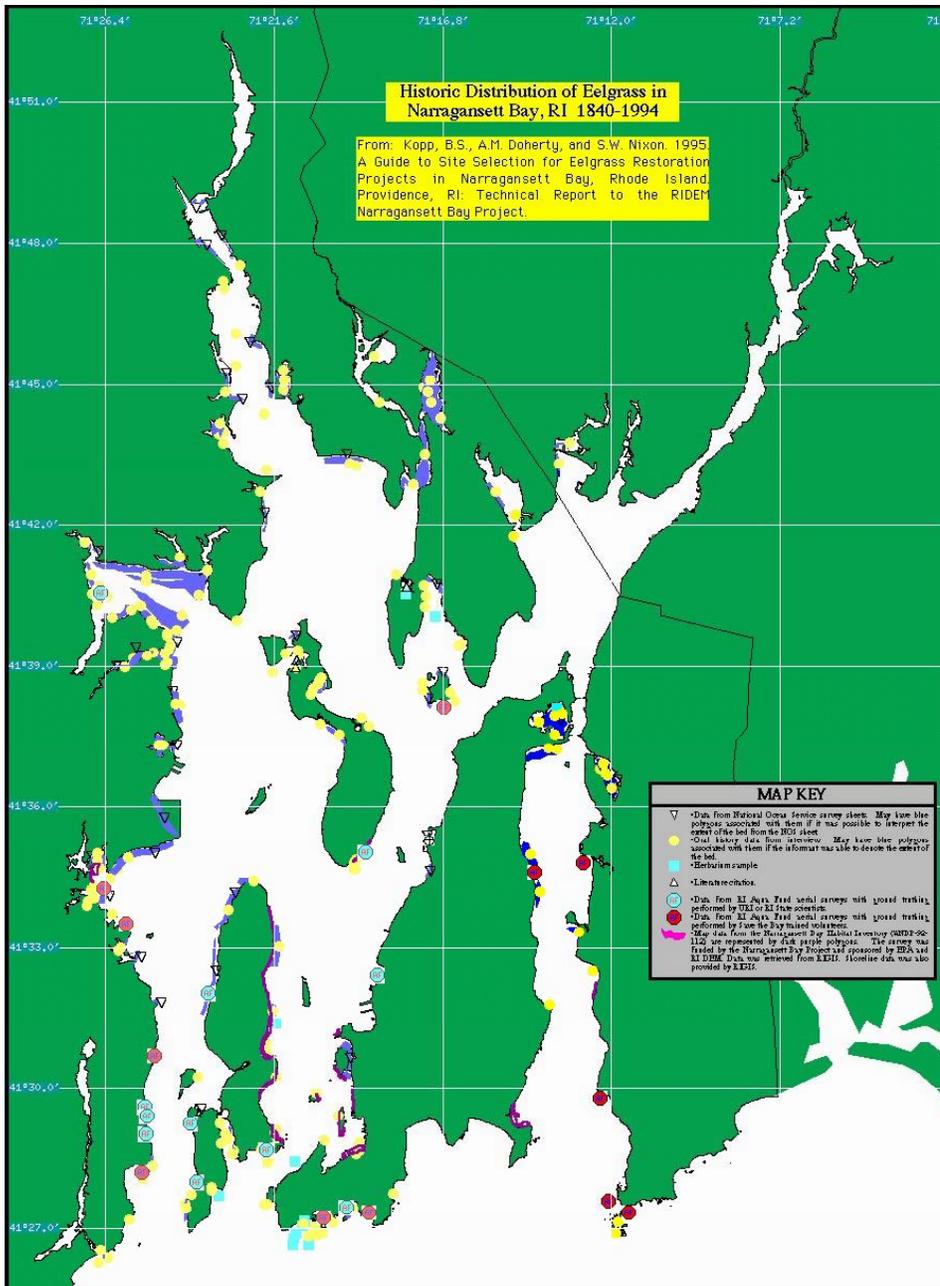


Figure 3.1. Historical distribution of eelgrass beds in Narragansett and Mt. Hope Bays. (Reprinted from Kopp et al. 1995.)

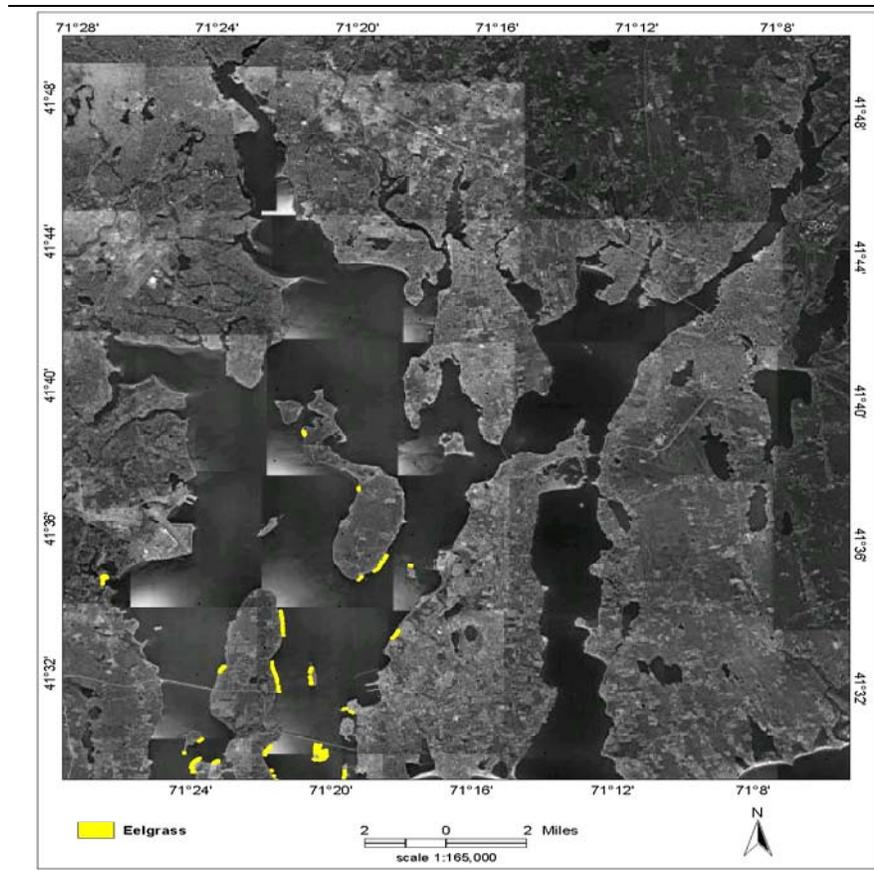


Figure 3.2. Eelgrass distribution in Mt. Hope Bay and upper Narragansett Bay. Eelgrass, shown in yellow, is absent from Mt. Hope Bay and is confined to shallow marginal areas within the lower western portions of Narragansett Bay. This system-wide distribution is typical of estuaries receiving significant nutrient inputs to the upper tributaries. Mt. Hope Bay has high turbidity and periodic low dissolved oxygen during summer, conditions not supportive of eelgrass beds. (Data provided by RIDEM—the Rhode Island Department of Natural Resources and Environmental Management.)

is thought that over half of the wetlands in the U.S. have been lost since colonial times (Dahl 1990, cited by the Massachusetts Coastal Zone Management Office (<http://www.state.ma.us/czm/walossd.htm>)). In Rhode Island, 10% of the coastal wetlands were filled for development between 1955 and 1964 alone. In Narragansett Bay, as much as 70% of the remaining wetlands have restricted tidal flows and 60% are subjected to some amount of filling and dumping activity. In addition, drainage patterns of about 50% of the remaining marshes have been

modified through mosquito ditching. To make matters worse, native plants are being replaced by the foreign invasive *Phragmites* in as much as 1/3 of the remaining marshes. Almost half of the brackish marshes in Narragansett Bay are now dominated by *Phragmites*. *Phragmites* replacement of native salt marsh plants is occurring at an alarming rate throughout the Mid-Atlantic Bight and New England region and has been the focus of much concern in the last decade. The degree of wetlands and salt marsh habitat loss and modification patterns in Mt. Hope Bay are unknown. Although modification of tidal flood and drainage patterns in salt marshes may not seem to be of major importance, when one considers that most marsh nekton access the marsh's production through tidal movements (Rountree 1992, Deegan et al. 2000), any modifications to water flow patterns take on greater significance. In fact, tidal marsh creeks are thought to be a major access route to the saltmarsh for nekton (Rountree 1992, Rountree and Able 1992a, 1993, 1997), yet they are often subjected to development and modification. The consequences have not been addressed. Change in saltmarsh habitat coverage over the last four decades in Narragansett Bay is currently being quantified by the Narragansett Bay Estuary Program and their collaborators (RIDEM 2001), but is not currently known.

Other habitats - Little is known of the importance of other habitat types in Narragansett or Mt. Hope Bay; however, Save the Bay has summarized the coverage area of some of the other types of habitats (Table 3.1). There are over 46 acres of marsh pannes and pools in Narragansett Bay. The latter is an important habitat for resident marsh nekton, and may be an important wintering

ground for some species (Smith and Able 1994, Smith 1995). This habitat is subject to adverse effects from marsh modification, especially through mosquito ditching. Tidal flats make up over 560 acres of Narragansett Bay. The importance of tidal flats has not been well documented, but it is known that tidal flats are important foraging grounds for winter flounder (Tyler 1971a, Wells et al. 1973). Shoreline beaches are a major habitat type in Narragansett Bay (1,450 acres), and are mostly dominated by irregularly flooded sand (47%) and regularly flooded sand and/or cobble (48%). Rocky shores make up another 570 acres. Information on the importance of these habitats to winter flounder and other fishes is limited. Sediment types such as mud, shell, silt, clay, sand, pebble, etc., also constitute important benthic habitat types for the open bay system but have not been mapped or quantified to date.

Conspicuously lacking in the summary (Table 3.1) is an identification of macroalgae beds that have largely replaced eelgrass beds in many areas. Although macroalgae may not be the optimal habitat for many species, recent studies suggest it is an important habitat for some species (Sogard and Able 1991, 1992). A recent survey of Narragansett and Mt. Hope Bays concluded that macroalgae is the dominant vegetative cover and is an important nursery habitat for tautog (Dorf and Powell 1997). A survey of macroalgae habitat types and distribution in Narragansett Bay recently concluded that macroalgae habitats are extremely spatially variable and suggested that attempts to monitor macroalgae habitats should maximize their spatial coverage (Harlin et al. 1996). An indication of the effect of nutrient loading and pollution on benthic habitats

dominated by shellfish is indicated by an examination of the closed shellfish areas for Narragansett and Mt. Hope Bays (Figure 3.3). Note that shellfish beds in the upper reaches of Narragansett Bay and most of Mt. Hope Bay are largely closed, suggesting potentially strong impacts on habitat quality.

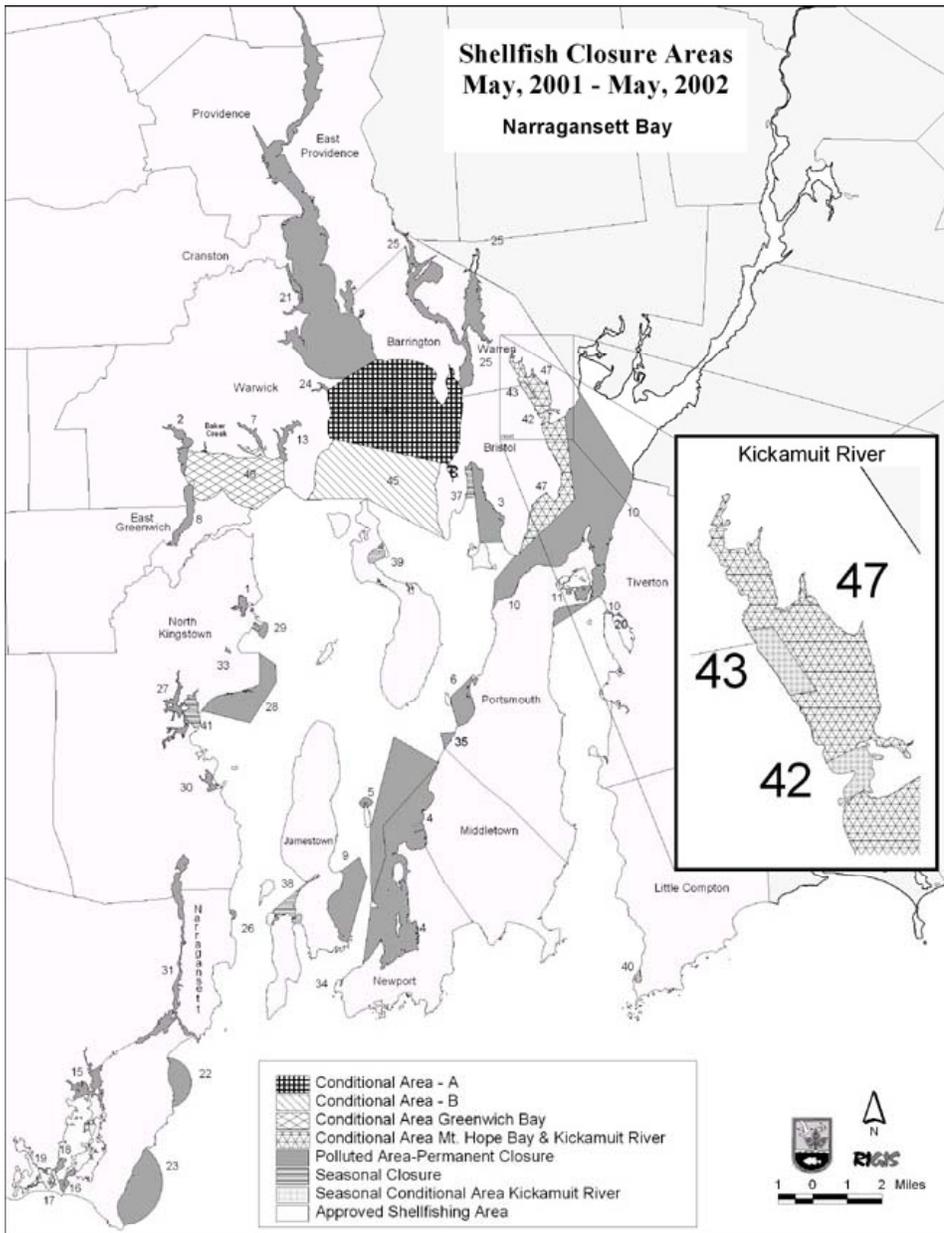


Figure 3.3. Evidence of the impact of nutrient loading and pollution on habitats and ecosystems depicted by closure of shellfish areas. (From www.state.ri.us/dem/maps/static/shellnar.jpg.)

Habitat quality/suitability – A great deal of interest in quantifying habitat quality or suitability has been generated in the last two decades (Able 1999), partly because it represents a tangible unit for resource managers. The functions of habitats are varied, but each habitat can support fish and invertebrate secondary production in several ways: 1) direct use as a nursery (growth), spawning, predator refuge, environment refuge, and/or foraging ground, and 2) as the ultimate source of primary production supporting secondary production in another habitat or ecosystem (habitat linkage). Although many estuarine species appear to be prevalent in many types of habitats, each habitat is likely to contribute differently to the success of a given species (Figure 3.4). Attributes

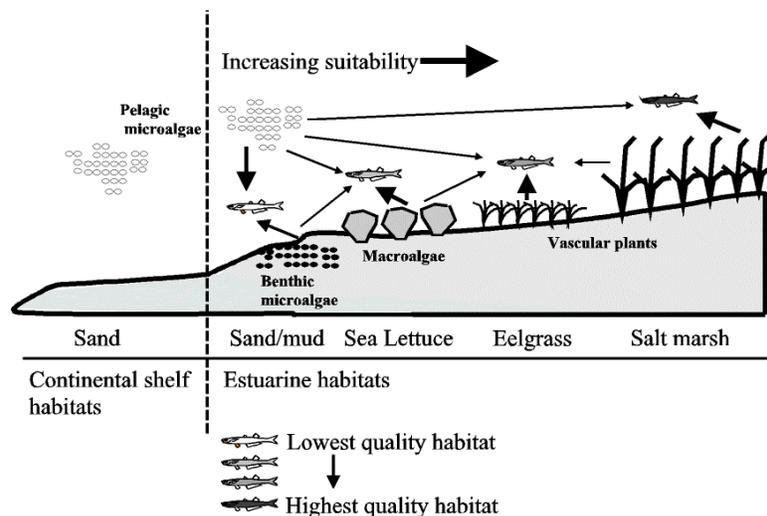


Figure 3.4. Although a species can be distributed across many types of habitats, variation in the habitat quality as measured by mortality, growth, fecundity, spawning success, etc., likely vary strongly among the habitats, such that some can be considered of higher "quality" than others.

such as growth, food availability, mortality, predator refuge capacity, spawning, and other factors can be used to quantify habitat quality. Although species may

be capable of switching from a "higher quality" habitat to an alternate "lower quality" habitat, the consequence of habitat modification and loss is theoretically that of a reduced carrying capacity of the estuary, and hence a reduced population size. It is important to note that habitat quality can, and often does differ

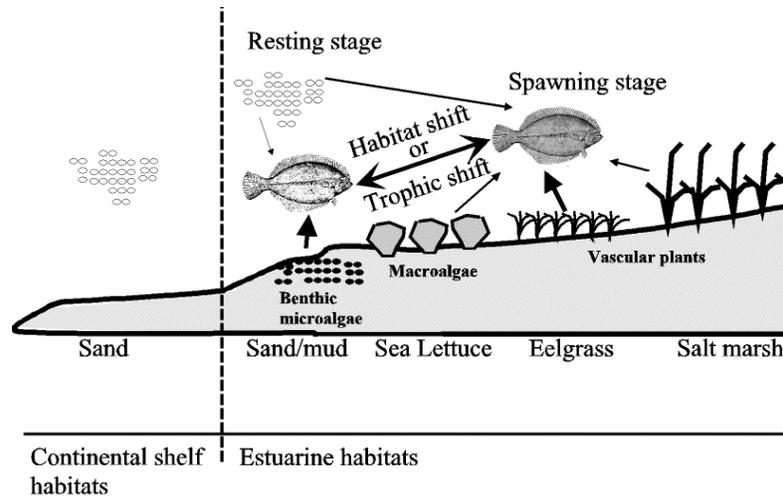


Figure 3.5. Habitat quality or suitability can shift among reproductive stages so that habitats optimal for spawning may not be optimal for other periods.

temporally (Figures 3.5 and 3.6). For example, habitats important during spawning may be different from those most important during resting and development stage (Figure 3.5). Similarly, habitat quality may shift among life stages, season, and even tidal and diel stages (Figure 3.6).

The importance of habitat linkages is crucial to understanding habitat quality and suitability for a given species, as often species may depend on habitats they do not directly utilize. For example, it is currently thought that the major pathway through which production originating in saltmarshes is transferred to open estuarine and coastal habitats is through trophic relay (Kneib 1997, Deegan et al. 2001). Trophic relay can be accomplished in many ways, but basically it is the

Patterns of temporal shifts in marsh habitat use

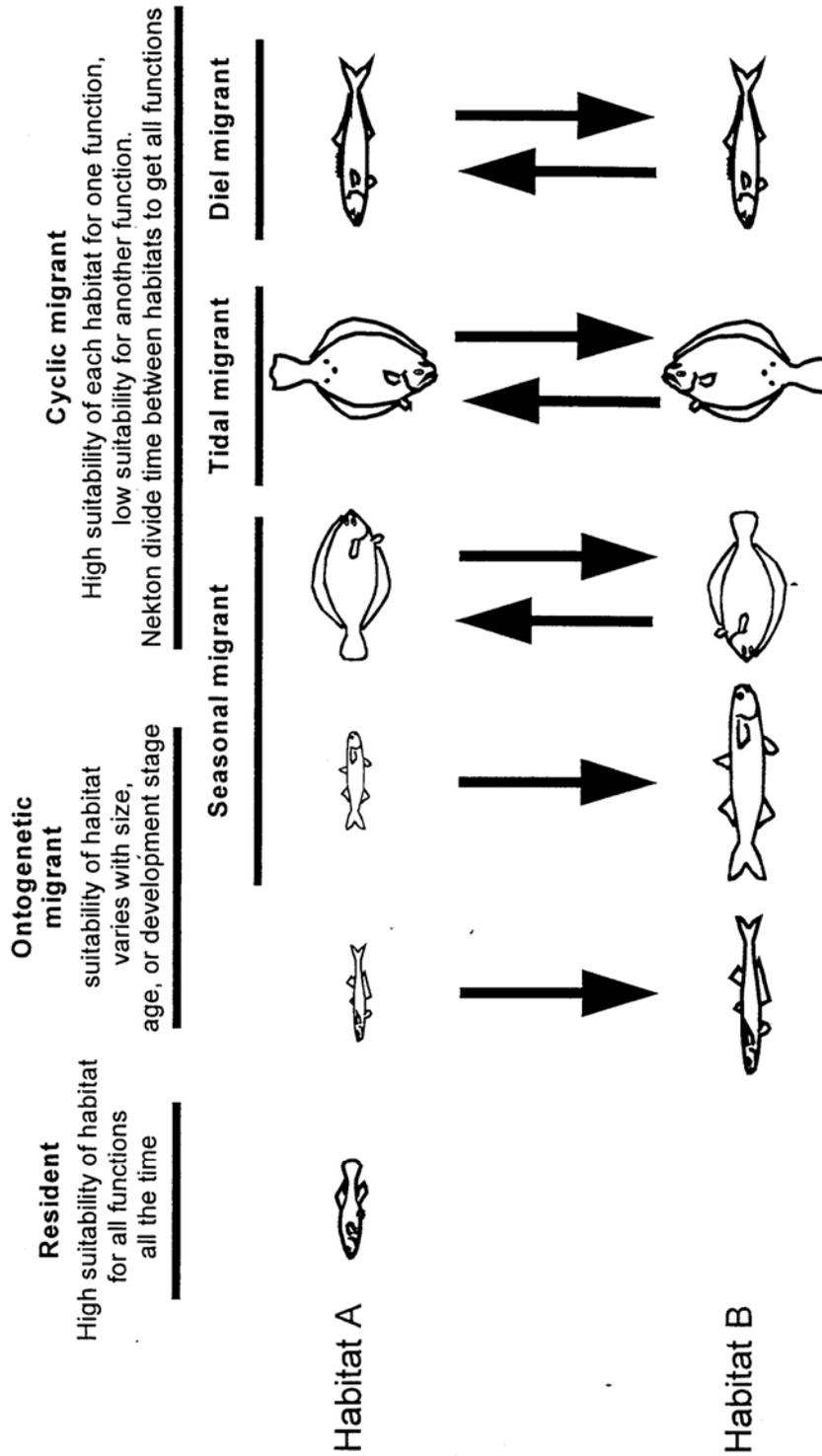


Figure 3.6. Estuarine habitats are trophically linked by nekton movements in a variety of ways, including ontogenetic, seasonal, tidal and diel migrations. (Adapted from Rountree 1992; permission pending.)

transfer of materials incorporated in living tissues through the movements of nekton between habitats. The most widely cited mechanism is through the emigration of nursery species from the saltmarsh (Figure 3.7). Materials

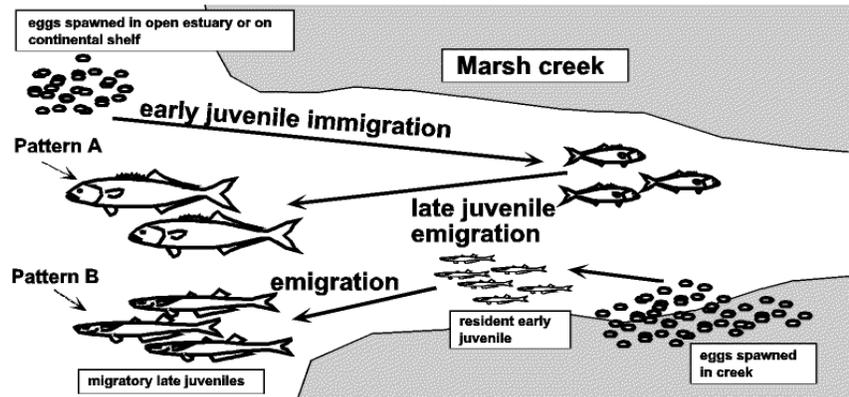


Figure 3.7. Primary and secondary production in saltmarshes and other shallow estuarine habitats supports secondary production in other habitats, sometimes far removed from them, through migration of nursery species. (Reprinted from Deegan et al. 2000; permission pend.)

incorporated into body tissues during growth are thus exported from the saltmarsh into coastal waters (e.g., Rountree 1992, Deegan et al. 2001). Less well known is the export of materials through tidal and diel foraging movements of nekton (Figure 3.8, Rountree 1992, Deegan et al. 2001). These processes result in the trophic linkage of habitats through a “chain of migration” (Figure 3.9), where primary production within important estuarine habitats such as saltmarshes and submerged aquatic vegetation (e.g., eelgrass and macroalgae) contributes to the secondary production of habitats that can be well removed both spatially and temporally. Loss of saltmarsh habitats, therefore, can have unforeseen impacts on nekton populations using other estuarine habitats.

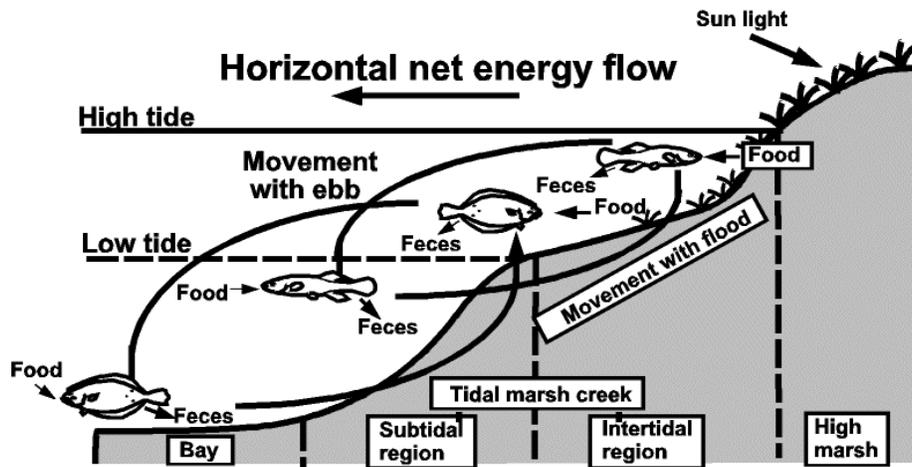


Figure 3.8. Saltmarshes and other shallow tidal habitats support secondary production in deeper subtidal estuarine habits through tidal and diel foraging movements of nekton (reprinted from Deegan et al. 2000; permission pending).

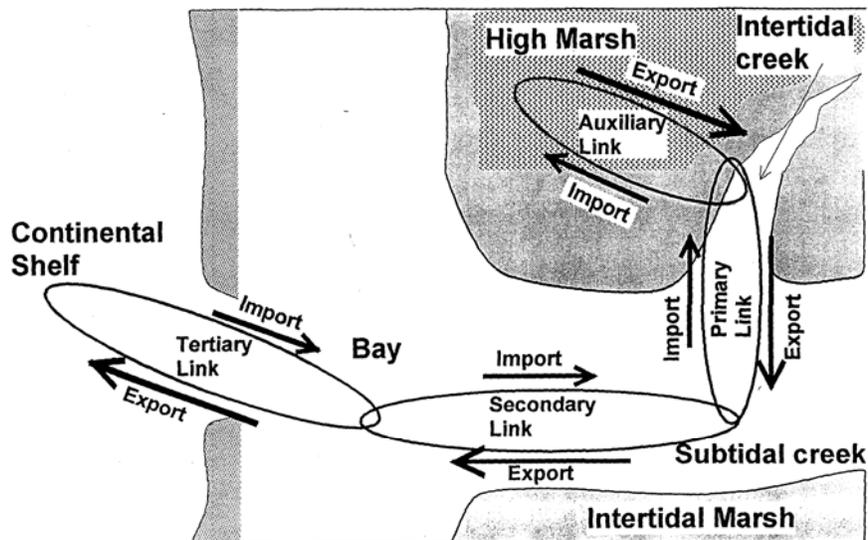


Figure 3.9. Saltmarsh and other shallow estuaries support open bay and coastal marine ecosystems through a chain of migration of nekton species resulting in the trophic relay of energy and materials. (Adapted from Deegan et al. 2000; permission pending.)

B. Watershed Inputs and Nutrient Related Habitat Quality

Mt. Hope Bay is one of the largest estuarine systems in Massachusetts and a major tributary system to Narragansett Bay. Like estuaries throughout the U.S., Mt. Hope Bay has become nutrient enriched as its surrounding watershed has become increasingly developed by the growth in regional population. At present about 1/3 of the total watershed area has been developed. The shift from forest to urban and residential development has enhanced nutrient inputs through wastewater, fertilizers and runoff.

The primary mechanism for watershed nitrogen to enter Mt. Hope Bay is through surface fresh water inflows. Mt. Hope Bay receives direct freshwater discharges from the Cole River, Lee River, Quequechan River and Taunton River systems. Of these, the Taunton River has the largest watershed and discharge. The Taunton River is the second largest river in Massachusetts and has a number of tributary river systems which contribute to its flow. In addition, there are two direct discharges of treated wastewater to the Bay (23 MGD) and five discharges directly to surface water tributaries to the Taunton River (30 MGD).

The primary nutrient related to the habitat quality and ecosystem functioning of Mt. Hope Bay is nitrogen. At present, the lower estuary appears to be receiving nitrogen inputs beyond its capacity to assimilate them without water quality declines. During summer the Bay periodically shows phytoplankton blooms ($>30 \mu\text{g chlorophyll-a L}^{-1}$) and low bottom water dissolved oxygen ($<4 \text{ mg L}^{-1}$), indicative of eutrophic conditions. To assess the relationship of these parameters to the Bay's habitat quality requires analysis of the spatial and

temporal extent of these key parameters relative to the animal and plant communities that have historically versus currently occupied this system.

A powerful approach to evaluating the key parameters which control the habitat quality of Mt. Hope Bay is through eutrophication or water quality modeling. A properly parameterized and validated eutrophication model could then be used to identify: (1) the nutrient sources controlling water quality, both within and external to the Bay; (2) the critical factors and physical conditions which control bottom water oxygen levels; (3) the relationship of oxygen conditions to organic matter production within the Bay versus entering the Bay from the watershed or via adjacent marine waters; (4) the areas where additional field data collection is needed; and (5) the potential for improvements in the health of the Bay through reduction of nitrogen sources or other key variables.

1. Key indicators of embayment "health"

The major ecological issue relating to habitat quality within Mt. Hope Bay is nutrient enrichment or eutrophication of Bay waters. Since eutrophication is the response to nutrients, the key indicators are (a) nitrogen concentrations, (b) chlorophyll-a (phytoplankton response), (c) light penetration (controls distribution of submerged aquatic vegetation–SAV), and (d) bottom water dissolved oxygen (primary ecological structuring parameter). Given the seasonal cycle of biological activity, the summer is the critical period for evaluating system health, and it is generally the period of annual minimum water quality. In addition, key ecological indicators include eelgrass and macroalgae distributions and dominant

benthic animals, such as *Ampelisca*, *Mediomastus* and *Nucula*. A primary variable for predicting interannual changes in key indicators relates to volumetric discharge from the Taunton River and the frequency and duration of watercolumn stratification.

2. Watershed nitrogen loading analysis

The contributing land area to Mt. Hope Bay represents the second largest watershed in the State of Massachusetts (Figure 3.10). The watershed covers

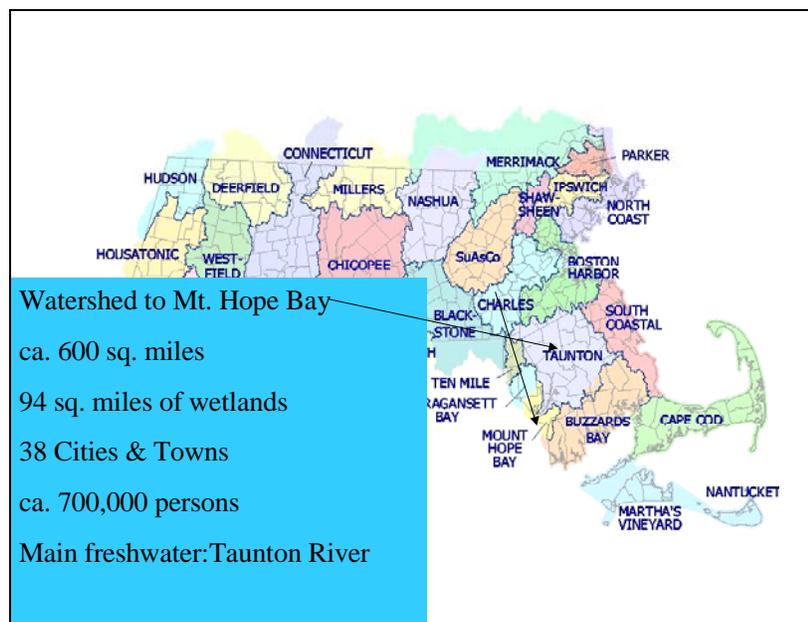


Figure 3.10. Land area contributing freshwater and nutrients to Mt. Hope Bay. The Bay's watershed is the second largest in Massachusetts. (Adapted from MassGIS and the Massachusetts Watershed Initiative.)

about 600 square miles and includes 700,000 people distributed among 38 municipalities. There are 94 square miles of wetlands and 24 major stream and

river systems discharging to the Taunton River or directly to Mt. Hope Bay. The major freshwater discharge to the Bay is the Taunton River, which has estuarine waters in its lower reaches (Figure 3.11). The lower portion of the Taunton River is a major tributary system to Mt. Hope Bay Estuary. It is estimated that about 70% of the freshwater from the Taunton River discharges to Narragansett Bay at the Mt. Hope Bridge (Hicks 1959c).

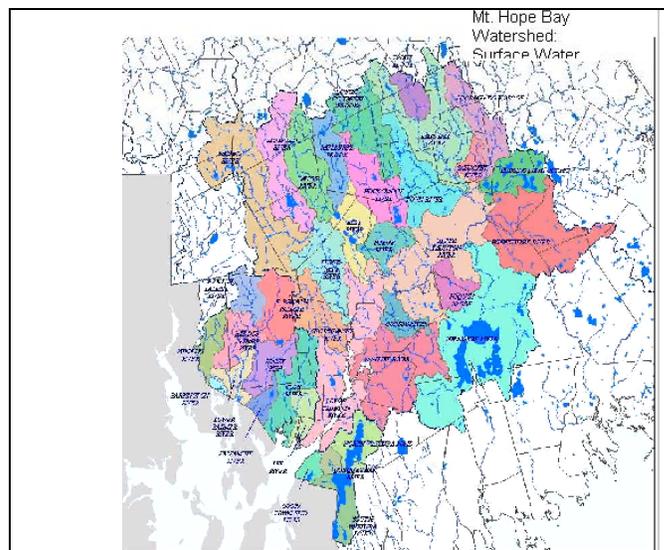


Figure 3.11. Map of the watershed contributing to Mt. Hope Bay via direct discharge or indirectly through the Taunton River. Major surface water sub-watersheds and freshwater streams and rivers are shown. (Map adapted from MassGIS; <http://www.state.ma.us/mgis/>.)

The two major sources of nutrients to Mt. Hope Bay are from its watershed and the marine waters of Narragansett Bay. The magnitude of these inputs can be gauged (1) for the watershed, through land-use analysis and nitrogen load modeling for inputs transported by freshwater and (2) for the marine

boundary inputs, through water quality modeling based upon nutrient levels and hydrodynamics of the Bay. While transported by tidal flows rather than freshwaters, the “marine” organic matter and nutrient inputs are also primarily derived from the surrounding watershed (in this case to Narragansett Bay). A significant factor in the magnitude of these tidally transported nutrients to the total nutrient balance of Mt. Hope Bay, depends upon the extent to which tidal waters originate in the nutrient rich Providence River and are transported by tidal flows into Mt. Hope Bay (at the Mt. Hope Bridge) versus originate from the less nutrient enriched waters of the Sakonnet River. Quantifying these “marine” nutrient inputs would fill a key data gap that needs to be addressed in the creation of a nutrient balance/eutrophication model for the Mt. Hope Bay System.

3. Temporal trends in population and watershed nitrogen loads

The watershed to Mt. Hope Bay is functionally divided into an upper and lower region. The upper watershed discharges freshwater and nutrients to the freshwaters of the Taunton River System (ca. 20 tributary rivers plus the Taunton River) that discharges through the Taunton River the Bay (Figure 3.12). The lower watershed contributes its freshwater and nutrient loads directly to the estuarine waters of the lower Taunton River and Mt. Hope Bay (Figure 3.13).

There has been no comprehensive watershed nitrogen loading evaluation conducted for either the Taunton River System or Mt. Hope Bay. As part of the current review, we have begun this analysis. To determine the load of nitrogen from the watershed to Mt. Hope Bay it is necessary to determine the sources of

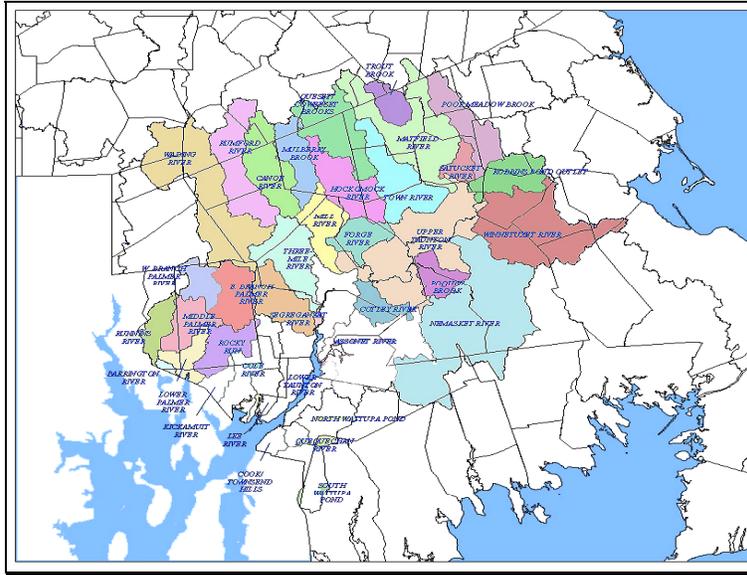


Figure 3.12. Upper watershed to Mt. Hope Bay showing sub-watersheds which contribute nutrients to the Bay via surface water discharges to the Tauton River. The Tauton River is the major freshwater source discharging to the Bay and the major conduit for the transport of nutrients from the upper watershed. (Map adapted from MassGIS.)

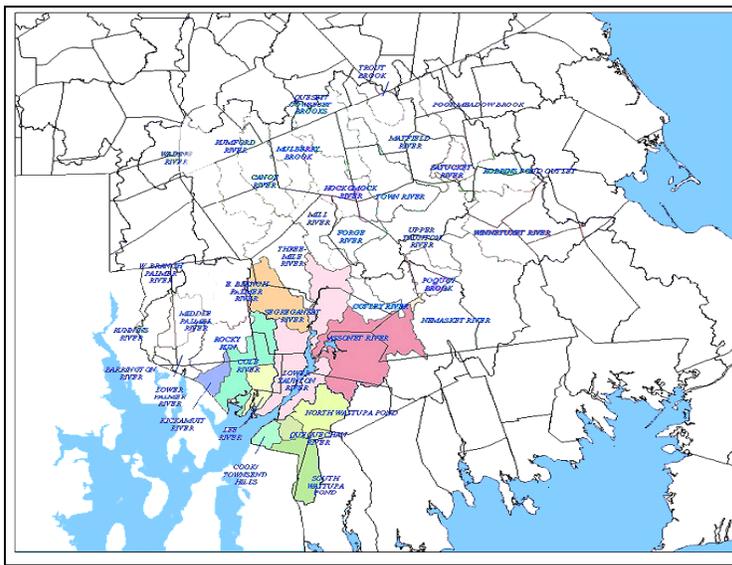


Figure 3.13. Lower watershed to Mt. Hope Bay showing sub-watersheds which contribute nutrients directly to the Bay via small tributary streams or direct groundwater discharges. (Map adapted from MassGIS.)

nitrogen, their magnitude and their spatial distribution. Each type of land-use has an associated nitrogen loading generally determined on an area or unit basis. If the land area is within the lower watershed, this nitrogen load will generally be transported to Mt. Hope Bay without loss or attenuation through natural processes of deposition or denitrification. In contrast, nitrogen transported from the upper watershed is generally attenuated during transport, with the magnitude of the attenuation being related to the specific characteristics of the surface water system through which it moves. Nitrogen attenuation within complex watersheds (e.g., containing diverse surface water systems) can exceed 60% of transported nitrogen. At this time it is not possible to determine the level of transport versus attenuation, since field data collection is required. However, determining the magnitude of attenuation of upper watershed nitrogen prior to entering the estuary is essential to eutrophication modeling and determination of future (and past) trends in habitat quality. This represents a major data gap.

At present, it is possible to gauge the recent temporal trend in nitrogen loading to Mt. Hope Bay based upon our initial watershed land-use analysis. This analysis examined both changes in population within the watershed over the past 40 years and the current pattern of specific land-uses. These data yield insight into both the magnitude of increases in nitrogen loading associated with population growth within the watershed and the extent to which future growth might occur. The major sources of land-use and population data were MassGIS and SRPEDD.

While a full population trend analysis is underway, a partial analysis was conducted which focused on the lower watershed (Figure 3.13). Unlike some areas in New England, the lower Mt. Hope Bay watershed has had a significant increase in population since 1960, with some areas growing 60%-80% (Figure 3.14). Some regions, particularly in urban areas, experienced small population declines (<10%); however, throughout the bulk of the watershed, population increased by 31%-60%. At present, it appears that an estimated average population growth of 30% for the lower watershed over the past four decades is conservative.

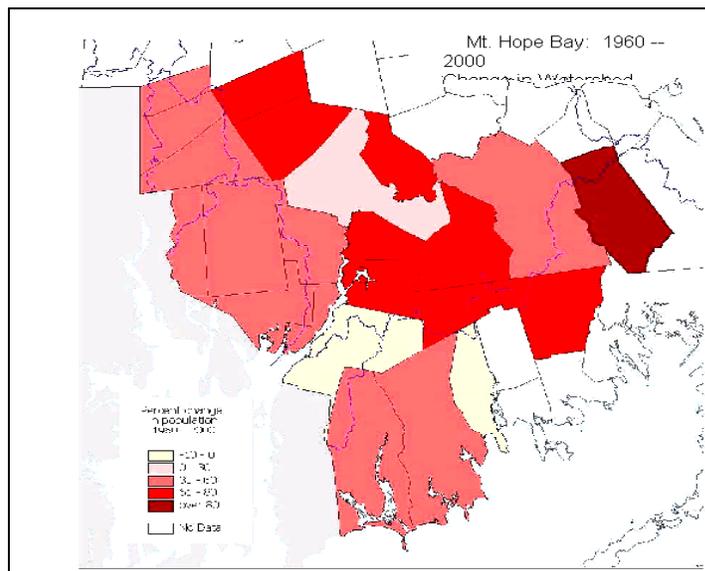


Figure 3.14. Population growth in the Mt. Hope Bay region between 1960 and 2000. Regional population has generally grown 30%-60% over this interval, while the urban (sewered) area has experienced a slight population decline. (Data from MassGIS and SRPEDD.)

The increase in the watershed population almost certainly represents an increase in nitrogen loading to Mt. Hope Bay. However, the level of increase is not proportional, since much of the population has their wastewater processed by municipal facilities. These facilities have been upgrading their performance over the past decade, which has reduced the per capita nitrogen discharge to the Bay (Save the Bay 1997). However, as tertiary treatment is not in effect in the major facilities, it is likely that the nitrogen load through wastewater has and continues to increase. A full watershed nitrogen loading analysis will need to account for present and potential future changes due to population growth (increase) versus improvements in nitrogen removal by wastewater facilities (decrease).

At present there are seven major municipal wastewater treatment facilities that contribute nitrogen, organic matter and freshwater to Mt. Hope Bay. Two facilities (Fall River and Somerset) discharge their treated effluent directly to estuarine waters, while five facilities discharge directly to tributary rivers to the Taunton River, which then flows to the Bay (Table 3.2). Together these seven discharges contribute 1 metric ton of nitrogen and 52 million gallons of treated effluent per day. However, the wastewater treatment facilities discharging directly to the Bay account for most of the wastewater nitrogen input. It should be noted that this evaluation is based upon 1997 data, the most recent year for which synthesis data was available, and additional recent improvements have been made to the Fall River facility which would have to be included into any future modeling effort.

Table 3.2. Discharges from municipal wastewater treatment facilities during 1997. (Data from Save the Bay 1997.)

| WWTF | Flow MGD | Nitrogen mg L⁻¹ | BOD Mg L⁻¹ |
|---------------------------------------|---------------------|---------------------------------------|----------------------------------|
| Taunton River watershed: | | | |
| Bridgewater | 0.74 | 5.69 | 121 |
| Brockton | 19.1 | 5.31 | 98.9 |
| Mansfield | 2.04 | 0.67* | 16.46 |
| Middleborough | 1.27 | 1.76 | 33.75 |
| Taunton | 6.4 | 7.84 | 190 |
| Total = | 10,782 MGY** | 26.2 MT*** | 261 MT |
| Mt. Hope Bay: | | | |
| Fall River | 20.0 | 106 | 277 |
| Somerset | 2.79 | NA | 224 |
| Total = | 8,322 MGY | 352 MT | 495 MT |
| * Data from 1994 | | | |
| ** MGY = millions of gallons per year | | | |
| *** MT = metric tons | | | |

Inventory of current land-use within the upper and lower watersheds to Mt. Hope Bay was conducted with assistance from MassGIS (D. Pahlavan personal communication). The land-uses—as mapped from aerial photographs and processed by MassGIS—were partitioned by sub-watershed, and the number and/or area of each land-use type was determined. The data were then composited by upper or lower watershed region and the total number or area of each land-use type determined.

The upper watershed (115,283 hectares) is more than 3 times the area of the lower watershed (35,285 hectares, Table 3.3). However, the existing land-uses in both regions are virtually the same (Figures 3.15 and 3.16). Although it might seem that the lower watershed has a large urban area (Fall River), the upper watershed contains a number of urban centers as well. Residential and

Table 3.3. Land-use within the upper and lower watersheds to Mt. Hope Bay (data provided from MassGIS).

| Mt. Hope Bay Land-use | | |
|-----------------------|-------------------------|-------------------------|
| Land-Use | Upper Watershed (Acres) | Lower Watershed (Acres) |
| Residential | 65771 | 18249 |
| Commercial | 13526 | 4651 |
| Agriculture | 15875 | 5434 |
| Open Space | 18146 | 5892 |
| Forest | 148068 | 46089 |
| Aquatic | 21633 | 6466 |
| Other | 1845 | 408 |
| Impermeable Area (ha) | 23980 | 8578 |
| | 9705 | 3471 |
| Total Area | 284865 | 87189 |
| Area (ha) | 115283 | 35285 |

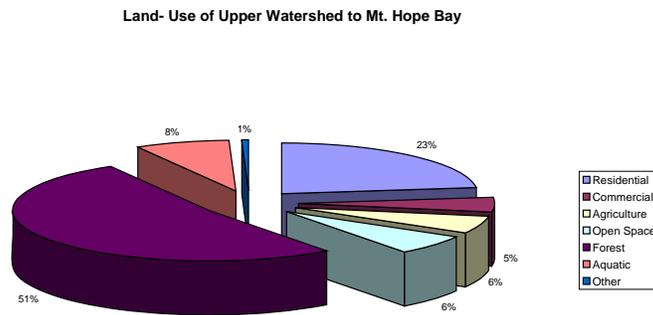


Figure 3.15. Distribution of land-uses within the upper watershed to Mt. Hope Bay (see Figure 3.12). Total land area is 115,300 hectares. (Data from MassGIS.)

commercial areas represent 28% and 26% of the upper and lower watershed areas respectively, with forest dominating both at 52% and 53%, respectively. The large remaining areas of forest in both upper and lower watersheds indicates the potential for continuing development and population growth. Based upon other

Land-Use of Lower Watershed to Mt. Hope Bay

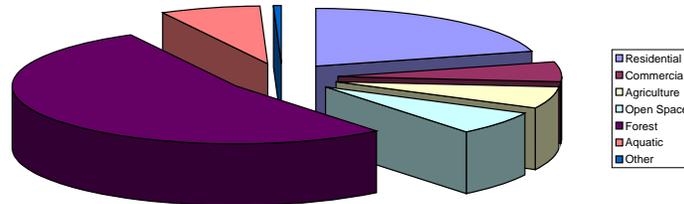


Figure 3.16. Distribution of land-uses within the lower watershed to Mt. Hope Bay (see Figure 3.13). Total land area is 35,300 hectares. (Data from MassGIS.)

Massachusetts land-use analyses, the future population of the watershed to Mt. Hope Bay could likely double over present levels. This would result in a potential doubling of present nitrogen loads, unless parallel improvements in wastewater treatment are instituted. Even if a fraction of this increase occurred, it would almost certainly have significant implications to the ecological health of Mt. Hope Bay.

4. Taunton River flow

The Taunton River is the primary surface water discharge to Mt. Hope Bay. The Cole and Lee Rivers also discharge directly to the Bay but are less than 4% of the Taunton River discharge. The Taunton River integrates numerous tributary streams and rivers throughout its 1153 km² contributing area, and it exerts a significant effect on the Mt. Hope Bay System both through the discharge of its nitrogen load and through its effect upon the salinity distribution and water column density field within the estuary. Freshwater discharge from the Taunton

River helps to create the vertical density stratification of Mt. Hope Bay, primarily due to salinity. Since bottom water oxygen within the Bay is related to water column stratification, the Taunton River plays a role in controlling this primary habitat quality parameter. In addition, the Taunton River gathers nitrogen from ca. 600 square miles of watershed and discharges it to the Bay waters. The overall result is that the river discharge serves both to increase the sensitivity of the Bay to nitrogen inputs (due to stratification) and to be a major contributor of nitrogen to the Mt. Hope Bay Estuarine System.

In order to assess potential seasonal and inter-annual variations in the effect of the Taunton River on water column stratification (and hence the sensitivity of the Bay to nitrogen loading), discharge measurements collected at the Taunton Gauge by the USGS from 1980-1999 were analyzed. As the gauge is located in the upper watershed, it does not capture the full Taunton River flow, but it does give important insight into the patterns of discharge.

The Taunton River exhibits a large degree of inter-annual variation in discharge. From the 1880-1999 data series, consecutive year shifts of 2 fold were common (Figure 3.17). It is not possible, at this time, to determine the effect of these large year-to-year changes in freshwater discharge to Mt. Hope Bay. At the lower river flows, however, it is likely that stratification of Bay waters may have been either less frequent or for shorter duration. This would likely result in higher habitat quality and possibly higher productivity of benthic animals in low flow versus high flow years. Understanding the relationship of freshwater discharge to bottom water oxygen levels is critical to determining inter-annual variations in

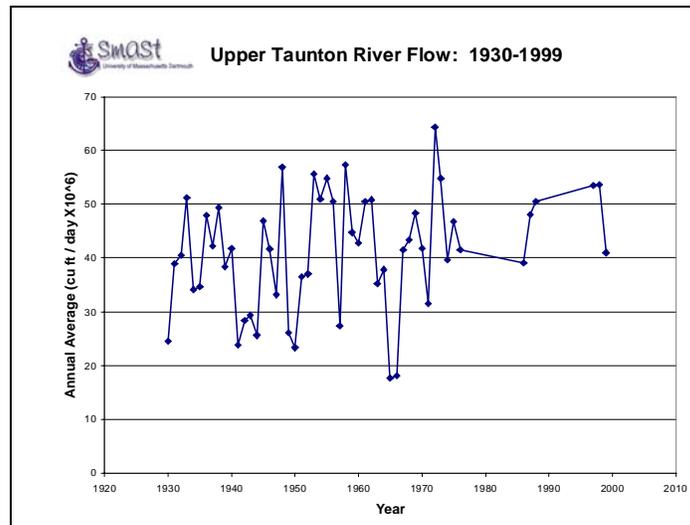


Figure 3.17. Long-term record (1930-1999) of flow in the upper region of the Taunton River as recorded by USGS. Note the 3-fold variation over the record and the frequent 2-fold variation in flow in consecutive years.

habitat quality. Determining this linkage would fill a major data gap and a critical gap in our understanding of the primary controls of annual variations in the health of sub-tidal regions of Mt. Hope Bay.

The Taunton River also exhibits a strong seasonal variation in discharge (Figure 3.18). The peak months of discharge occur in spring (March, April) and have been about 8 times higher than the summer minima (July-September). It appears that even during the low flow months, discharge (and temperature) can be sufficient to cause stratification of the Bay and consequent low bottom water oxygen conditions (see below). While the discharge measurements do not include all of the freshwater flow via the Taunton River to the Bay, it is still worth noting that the minimum discharge at the USGS gauge is 80-90 MGD compared to the total wastewater flows of ca. 50 MGD. It appears that a significant fraction of the freshwater budget of Mt. Hope Bay is associated with treated wastewater effluent.

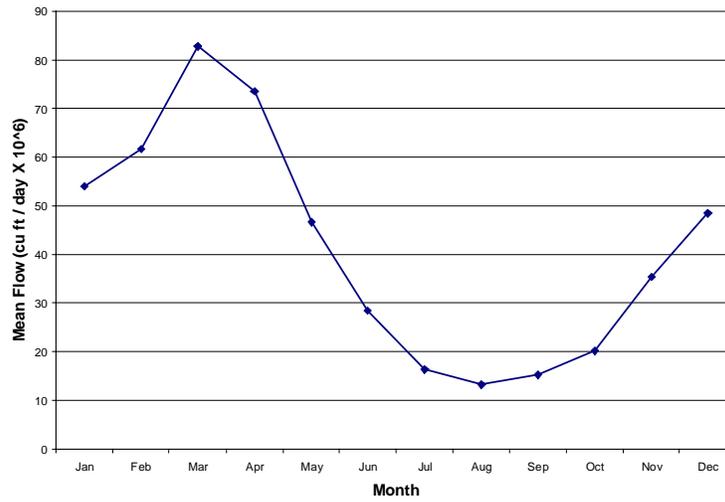


Figure 3.18. Average annual flow in the upper region of the Taunton River as recorded by USGS from 1930-1999 (see Figure 3.17). Note the strong annual cycle resulting from the annual distribution of rainfall and evapotranspiration within the watershed.

5. *Assessment of nitrogen-related water quality indicators within Mt. Hope Bay*

a. *Historic values*

While there has been an extensive amount of work on the fisheries, benthic animals, oxygen and chlorophyll levels within Mt. Hope Bay, there has been much less data collected on key nutrient species. In addition, chlorophyll-a data from 1985 to present is also sparse.

A variety of water quality indicators have been used to evaluate the “health” of coastal embayments. The specific water quality indicators selected change with the local ecological issue being addressed. The major ecological issue relating to habitat quality within Mt. Hope Bay is nutrient enrichment or eutrophication of Bay waters. Since eutrophication is the response to nutrients, the key indicators are (a) nitrogen concentrations, (b) chlorophyll a

(phytoplankton response), (c) light penetration (controls SAV distribution), and (d) bottom water dissolved oxygen (primary ecological structuring parameter). Given the seasonal cycle of biological activity, the summer is the critical period for evaluating system health, and it is generally the period of annual minimum water quality.

Historical information on these key water quality indicators is limited. Chlorophyll-a levels have been monitored in Mt. Hope Bay relative to the power facility discharge (MRI 1999), but annual data collection was relatively low after 1985. More recently, bottom water dissolved oxygen has been surveyed, particularly by MCZM and RIDEM, but these data are being finalized. Other indicators have been relatively undersampled. The most critical of these relates to water column nutrient concentrations. At present, we have found virtually no information on the nitrogen and phosphorus levels within Mt. Hope Bay or the lower Taunton River. There is some limited data on inorganic species (ammonium, nitrate, ortho-phosphate), but little on organic forms. Since most of the nitrogen and phosphorus is in organic form, these data must be collected both for assessment purposes and for the construction of water quality and eutrophication models. While the historical data set is not complete, it is sufficient to conclude that Mt. Hope Bay has been eutrophic for at least the past 3 decades. Chlorophyll-a levels in the early 1970's were generally over $10 \mu\text{g L}^{-1}$ and over $20 \mu\text{g L}^{-1}$ for much of the spring and fall.

b. Present conditions

A variety of ongoing studies in Mt. Hope Bay are sampling water quality indicators (MCZM, RIDEM, EMPACT). Data from these studies support the contention that Mt. Hope Bay is currently exhibiting eutrophic conditions. In part, this results from the fact that Mt. Hope Bay can develop long periods (weeks to months) of water column stratification. This increases the Bay's sensitivity to nitrogen inputs by preventing ventilation of bottom waters. In organic matter rich systems like Mt. Hope Bay, ventilation is necessary to oxygenate bottom waters which otherwise become oxygen depleted due to high rates of respiration.

During the summer of 2001, the water column at a continuous sampling station in the western region of Mt. Hope Bay (SMAST Mooring) indicated periodic phytoplankton blooms (chlorophyll-a over $10 \mu\text{g L}^{-1}$) and bottom water oxygen depletion (Figure 3.19). These data suggest that the ability of the Mt. Hope Bay System to assimilate nutrients without water quality decline has been exceeded.

Oxygen depletion was not uniform throughout the water column, but found primarily within the bottom waters (Figure 3.20). It appears that, for weeks (Figure 3.21) to months (data not shown), bottom water oxygen levels were rarely at atmospheric equilibrium. Oxygen depletion results from the uptake of dissolved oxygen in heterotrophic respiration in sediments and water column.

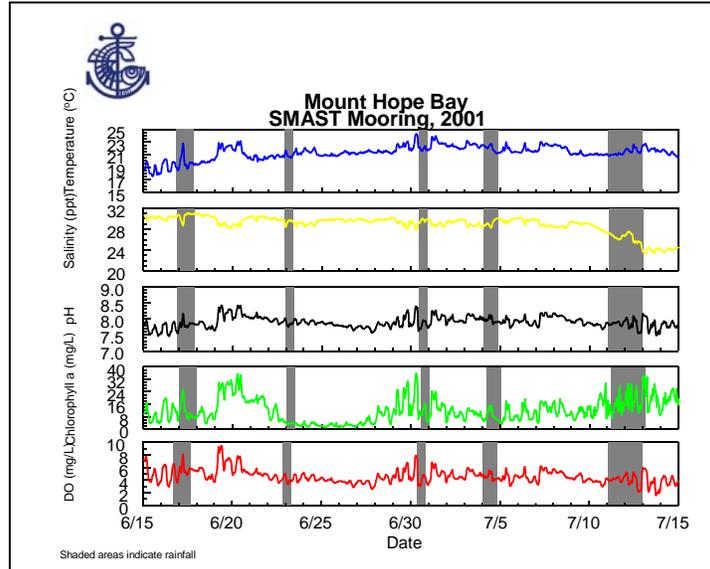


Figure 3.19. Key water quality data collected during mid-summer 2001 at the SMAST mooring near the channel at mid-Bay. (Data collection in collaboration with Narragansett Bay Commission and MCZM under EPA EMPACT Program.)

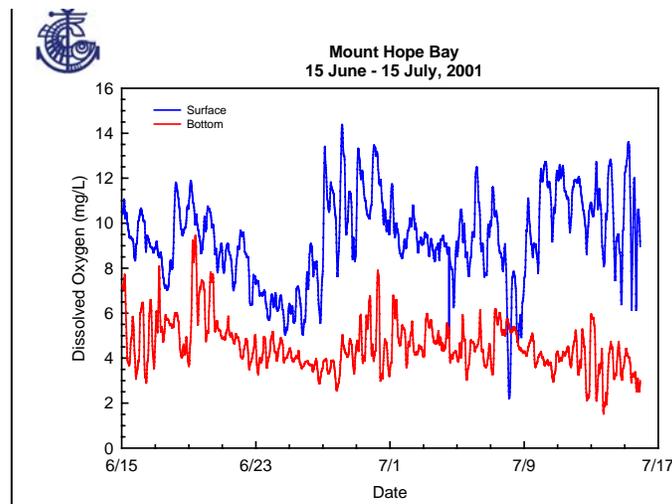


Figure 3.20. SMAST mooring in Mt. Hope Bay during mid-summer 2001. The oxygen levels at the surface are generally above and in the bottom water generally below atmospheric equilibration. This is indicative of eutrophic conditions, where ecosystem oxygen consumption is sufficient to exceed oxygen production through photosynthesis and ventilation. (Data collection in collaboration with Narragansett Bay Commission and MCZM under EPA EMPACT Program.)

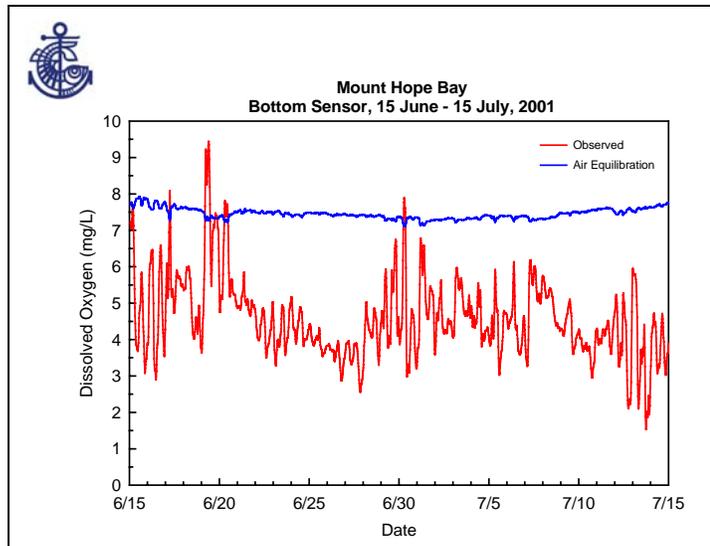


Figure 3.21. Bottom water dissolved oxygen levels and associated levels if at atmospheric equilibration for data interval in Figure 3.20.

Stratification of the water column isolates the bottom waters from reaeration from the atmosphere. If the waters are sufficiently turbid or deep that light does not penetrate below the pycnocline, then photosynthetic oxygen production is also eliminated. Both of these conditions are met within Mt. Hope Bay. The result is that organic matter degradation within the bottom waters and sediments consumes available oxygen and levels decline. Under long periods of stratification or where organic matter pools are large due to nutrient enrichment, large oxygen depletions can occur. Large oxygen depletions (to concentrations of $<4 \text{ mg L}^{-1}$) are stressful to benthic animals and fish, and their communities tend to shift to more tolerant forms.

The proximate cause of low dissolved oxygen within Bay bottom waters is the long periods of stratification that can occur in summer (Figure 3.22). The ultimate cause is the organic matter, produced from nitrogen inputs from the watershed and marine boundary, which supports oxygen consumptive processes.

At present, these factors are producing summertime oxygen levels that should be stressful to many marine organisms.

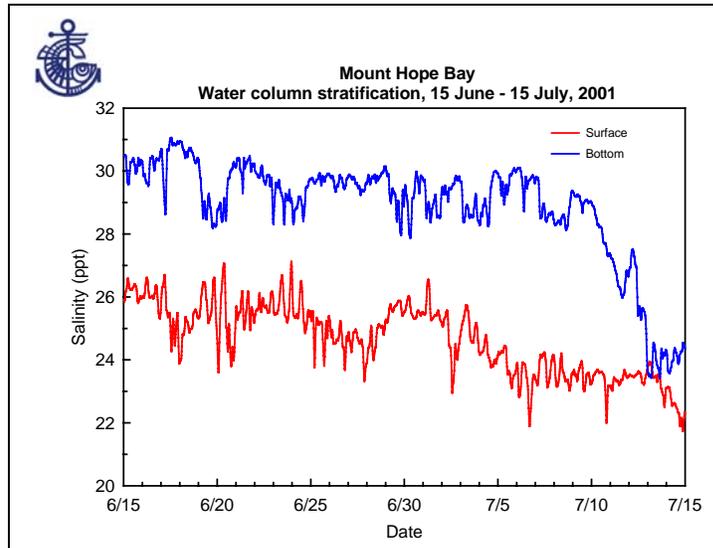


Figure 3.22. Salinity of surface and bottom waters at the SMAST mooring during mid-summer 2001. Note the strong salinity stratification, which is consistent with the low dissolved oxygen observed during this interval. (Data collection in collaboration with Narragansett Bay Commission and MCZM under EPA EMPACT Program.)

Available information supports a conceptual model of nutrient and organic matter cycling within Mt. Hope Bay that is driven by inorganic nutrients entering from the watershed primarily via the Taunton, Cole and Lee Rivers, and from the lower watershed wastewater discharges. These nutrients support high rates of phytoplankton production (MRI 1999), which likely exceed $600\text{-}800\text{ g C m}^{-2}\text{ y}^{-1}$ (calculated from MRI data). These rates are indicative of nutrient-rich estuaries and coastal upwelling areas (Whittaker 1975). It appears that the high apparent rates of respiration within Mt. Hope Bay Estuary likely result from in situ production and decomposition. However, it appears from the inorganic N and P

data collected in 1997-98 (MRI 1999) that even these high rates of production are insufficient to consume all of the available nutrients within the Bay waters, and the system remains nutrient-replete, possibly year-round. The nutrient levels within the Bay are consistent with observations that primary production within this system is generally light-limited (i.e., there is insufficient light to support photosynthesis by 1-2 meters depth). It appears that Mt. Hope Bay is likely a net contributor of inorganic nutrients and organic matter to greater Narragansett Bay.

c. Benthic animal communities

Benthic animals (animals living in the bottom sediments) are good indicators of system health. These animals are resident within the Bay and tend to integrate, over time, the environmental conditions in which they live. For this reason, benthic animals are typically used as bioindicators of system stability or stress (nutrient, oil, organic contamination, etc.). Benthic animal communities are being monitored by MRI as part of the Brayton Point Program. Samples are collected near the Brayton Point Power Station and near mid-Bay, off Spar Island (Figure 3.23).

We have used the Spar Island sampling station (station F) and the station 1+ km from the power plant (station C) to evaluate the general conditions within the central region of Mt. Hope Bay. At present these stations are dominated by *Ampelisca abdita*, and *Mediomastus ambiseta*, with lesser numbers of *Nucula annulata*. These species are indicative of an organic-matter-rich (somewhat stressed) environment, but do not indicate the highest level of stress (as, for

example, would *Capitella*). These species are typical of areas of organic enrichment such as Boston Harbor or at distance from sewage outfalls. They do represent a food source for demersal fish and crustaceans.

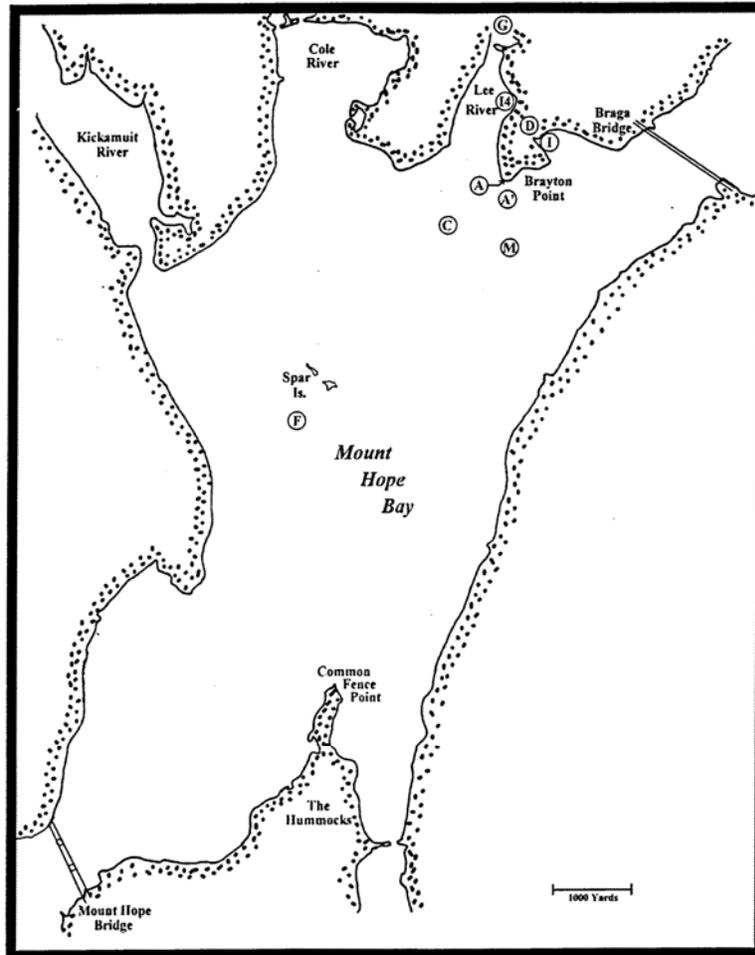


Figure 3.23. Location of benthic sampling (MRI 1999).

Overall, the Spar Island site tended to support between 5,500 and 36,900 total animals per square meter distributed among 17-36 species (Table 3.4). These data indicate a Phase II community as defined by Rhoads and Germano

Table 3.4. Benthic infaunal species and numbers within upper and mid-Mt. Hope Bay at Stations C, F, and I#4, March 1997-February 1998 (adapted from MRI 1999).

| Date | C | | F | | I#4 | | Total Indiv. | Mean Indiv. |
|----------------|-------|--------|-------|--------|-------|--------|-----------------|----------------|
| | Spec. | Indiv. | Spec. | Indiv. | Spec. | Indiv. | | |
| March 10 | 16 | 5250 | 26 | 11350 | 28 | 59925 | 76525 | 25508 |
| April 3 | 20 | 3300 | 24 | 5450 | 23 | 23300 | 32050 | 10683 |
| April 23 | 17 | 4650 | 26 | 15325 | 28 | 39725 | 59700 | 19900 |
| May 17 | 14 | 8850 | 33 | 19050 | 23 | 29400 | 57300 | 19100 |
| June 4 | 23 | 10175 | 29 | 13375 | 19 | 10550 | 34100 | 11367 |
| June 24 | 17 | 7275 | 26 | 29850 | 28 | 25625 | 62750 | 20917 |
| July 16 | 18 | 4175 | 18 | 6100 | 27 | 14775 | 25250 | 8350 |
| August 5 | 18 | 3075 | 27 | 15700 | 23 | 19300 | 38075 | 12692 |
| August 26 | 18 | 3800 | 17 | 10625 | 19 | 9900 | 24325 | 8108 |
| September 17 | 24 | 9475 | 17 | 11625 | 35 | 106550 | 127650 | 42550 |
| October 10 | 29 | 9800 | 19 | 22975 | 39 | 119150 | 151925 | 50624 |
| October 29 | 21 | 4575 | 22 | 17300 | 33 | 136600 | 158475 | 52825 |
| November 20 | 26 | 11650 | 26 | 10650 | 41 | 151775 | 174075 | 58025 |
| December 10 | 24 | 9000 | 24 | 9000 | 42 | 236750 | 254750 | 84917 |
| December 24 | 31 | 19175 | 32 | 34325 | 42 | 135050 | 188550 | 62850 |
| January 15 | 28 | 23750 | 30 | 35850 | 43 | 118425 | 178025 | 59342 |
| February 3 | 21 | 10100 | 36 | 28250 | 41 | 177925 | 216275 | 72092 |
| February 27 | 21 | 4900 | 28 | 24950 | 43 | 155000 | 184850 | 61617 |
| Yearly Average | 21 | | 26 | | 32 | | 113591 | 37859 |
| Indiv. Range | | 3300- | | 5450- | | 9900- | 24325- | 8108- |
| | | 23750 | | 35850 | | 236750 | 216275 | 84916 |

(1982). Phase II communities are represented by shallow burrowers which are typically deposit feeders (rather than filter feeders) and sediments which have limited oxygen penetration and a redox discontinuity layer near the surface. The species tend to be relatively short lived. Given the trends in watershed inputs, it is likely that these communities are transitional over the long term, and are moving towards smaller, shorter-lived species.

It appears that over the past 20 years there have been no dramatic shifts in the benthic animal community at mid-Bay. Total animals has remained nearly constant 1978-1992 and 1998 (Figure 3.24). Similarly, specific species, such as

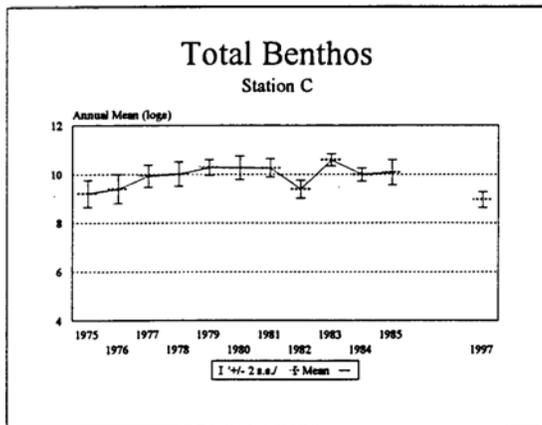
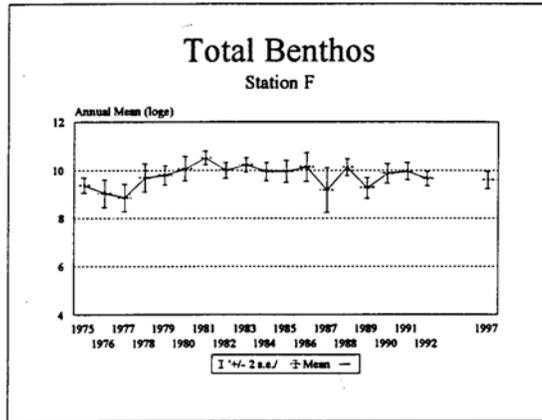


Figure 3.24. Annual variations in median densities of total benthos at stations F and C and annual log mean densities plus and minus 2 S.E. (MRI 1999).

Ampelisca and *Nucula* (Figure 3.25), have varied in numbers, but with no consistent trend. While *Mediomastus* may be experiencing a decline in numbers at present, it is still above the 1975 level (Figure 3.25). Therefore, it appears that

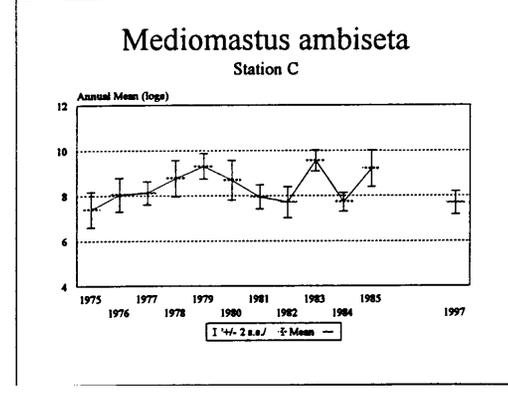
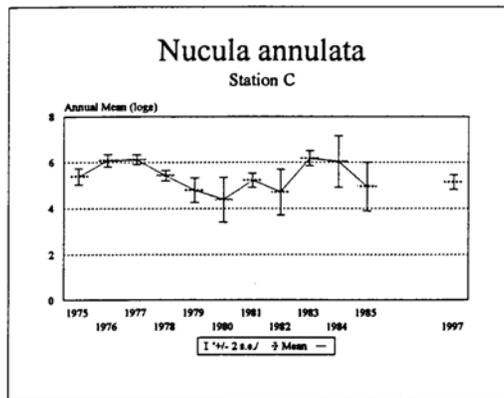
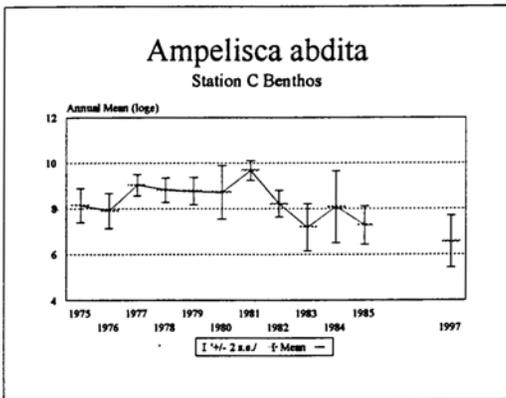
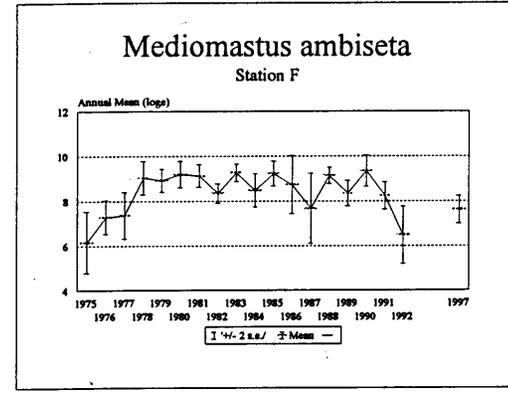
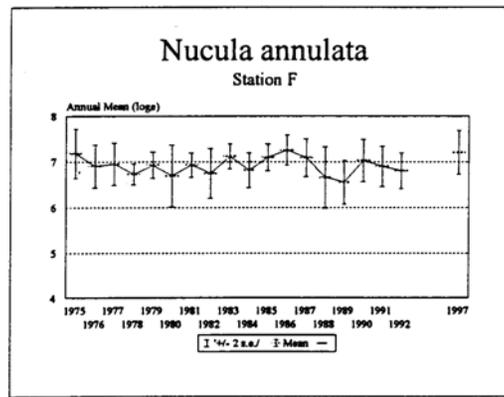
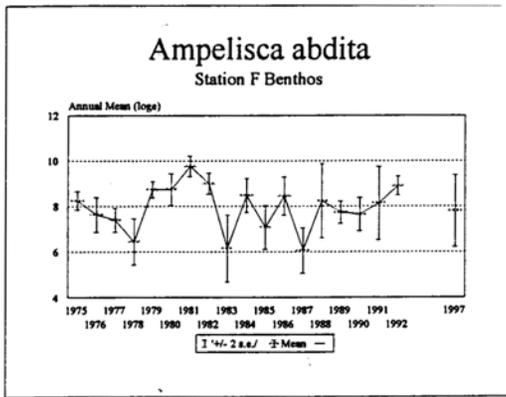


Figure 3.25. Annual variations in median densities of *Ampelisca abdita* (left panels), *Nucula annulata* (middle) and *Mediomastus ambiseta* (right) at stations F and C and annual log mean densities plus and minus 2 S.E. (MRI 1999).

conditions have been relatively “stable” in regard to benthic populations over the past two decades. Stability, however, does not mean that environmental conditions are of high quality, only that sufficient changes have not occurred to alter the community structure in regard to either species or numbers of individuals. This finding is consistent with the limited water quality data that suggests that the present nutrient enriched conditions have existed for several decades.

6. Mt. Hope Bay Nutrients

There has been remarkably little work on nutrients in Mt. Hope Bay. This contrasts greatly with knowledge of the Rhode Island portion of Narragansett Bay, which, due to decades of research efforts by scientists and students of the Graduate School of Oceanography at the University of Rhode Island, is one of the most-intensively studied estuaries in the world.

A draft report that was part of an effort to develop an "Action Plan for the Taunton River Watershed" (compiled by the Urban Harbors Institute, University of Massachusetts Boston) contains data on nutrients in Mt. Hope Bay (total nitrogen, ammonia, total phosphorus). In addition, Boucher (1991) studied total dissolved and particulate nitrogen and phosphorus in the Taunton River estuary. Together these sets of nutrient data indicate that nutrient levels in Mt. Hope Bay are high, with river flow and groundwater as important factors in nutrient variability. Boucher found decreasing nutrient concentrations moving downstream from the Taunton River to Mt. Hope Bay.

Pilson and Hunt (1989) presented data on a large variety of water quality parameters, including nutrients from samples throughout Narragansett and Mt. Hope Bays from cruises in October and November, 1985, and April and May, 1986. Included were data on dissolved inorganic nitrogen (ammonia, nitrate, nitrite) dissolved organic nitrogen, particulate nitrogen, dissolved silicate and phosphate, and total dissolved and particulate phosphorus. Pilson and Hunt found that dissolved organic nitrogen was the most abundant form of nitrogen. The more easily utilized dissolved inorganic forms such as nitrate, nitrite and ammonia declined in spring, presumably due to phytoplankton uptake. This decrease was proportionately greater for nitrogen than phosphorus, because the water column had N:P ratios typically less than the Redfield ratios for marine plankton. This suggests that Narragansett Bay is closer to being limited by nitrogen than by phosphorus. There was a decrease in nutrient levels moving from the Providence River and Mt. Hope Bay down to the mouth of Narragansett Bay. Measured inputs of nutrients from rivers and sewage outfalls were sufficient to replace the total mass of water-column nutrients of Narragansett Bay in 40-125 days, depending on season.

Marine Research, Inc., has analyzed dissolved inorganic nutrients at several sites in Mt. Hope Bay at least once per month from 1972 to 1985 (inclusive), and again from March 1997 through February 1998 (see MRI 1999). Ammonia concentration at the open-water Mt. Hope Bay stations followed the expected pattern of lowest concentration (1-2 μMol) in late winter/early spring increasing to greatest concentration (to near 10 μMol) in late summer/early

autumn. Annual mean ammonia concentration was between 4.0 μMol (1973, 1974, 1976) and ca. 7.5 μMol (in 1979) during 1972 to 1985 monitoring. The 1997/98 annual mean ammonium concentration at the same stations, and analyzed with the same laboratory protocol, was ca. 12.5 μMol , a dramatic (ca. 2-fold) and statistically significant increase over 1972-1985 ammonia concentration observations. No explanation for the elevated 1997/98 ammonia concentration is apparent (MRI 1999). Mt. Hope Bay nitrate concentration displayed the expected pattern of a winter maximum (near 13 μM November through January) and summer minimum (near 1 μM or less in July). Comparison of mean 1972-1985 nitrate concentration (ca. 5 to 6 μM) to apparently elevated 1997-98 mean annual nitrate concentration (ca. 12 μM) yielded no statistically significant difference (MRI 1999). Mt. Hope Bay nitrate concentration in April to August 1997, however, was higher than the 1972-85 mean values for those months. Dissolved inorganic phosphate concentration in Mt. Hope Bay had an annual pattern of late winter/spring minima (near 1 μM or less) increasing to an autumn maximum (near 4 μM in October). Mean annual concentration figures of ca. 1.5 μM (1985) to <5 μMol (1980) were not significantly different from the 1997/98 mean annual value of ca. 2.1 μM (MRI 1999). Dissolved silicon concentration, important for diatom growth, was not reported in MRI (1999). Because of the 1986-1996 gap in nutrient sampling, whether the observed 1997/98 differences in Mt. Hope Bay ammonia and nitrate concentration are symptomatic of a long-term trend, or are anomalous to 1997/98, is not known.

Chapter 4. Plankton (David Borkman & Jefferson Turner)

Introduction

The high fecundity of marine fish and the lack of correlation between size of spawning stock and resultant year-class strength imply a high rate of mortality for each year class of fish (Hjort 1926). Most such mortality occurs during the pelagic larval stage, particularly after yolk absorption, when young fish have to begin feeding on their own. This prompted Hjort (1914) to propose that there is a “critical period” in the survival of first-feeding fish larvae, and that most mortality was due to starvation and transport to unfavorable areas (“larval drift”).

Additional mortality of fish larvae comes from predation by larger fish (Hunter 1981) and invertebrates (Turner et al. 1985). Thus, factors affecting survival of fish larvae are intimately related to fish population dynamics, including stock and recruitment, year-class strength, and population declines (May 1974). Hjort (1914) proposed starvation as the primary cause of larval fish mortality.

Subsequent studies have confirmed the importance of sufficient amounts and types of food to larval fish survival (Lasker 1975, 1981), expanding fishery science to include “fisheries oceanography,” which recognizes that fish, particularly larvae, are subject to biological and oceanographic variations in the food webs that support them.

A review of the feeding habits of 76 species of fish identified from 40 published studies (Turner 1984) revealed that most fish larvae feed on plankton until they are large enough to begin feeding on smaller fish. Thus, attempts to understand the abundance fluctuations of any species of fish in any ecosystem

must include knowledge about the plankton that support the larvae of the fish species of interest. Gut content studies reveal that larvae of the winter flounder (*Pseudopleuronectes americanus*) in New England estuaries feed primarily on small plankton such as nauplius larvae of copepods and ciliate protozoans known as tintinnids (Pearcy 1962), phytoplankton and calanoid copepods (Keller and Klein-MacPhee 2000, Shaheen et al. 2001). Because winter flounder larvae are demersal, living close to the bottom, they also feed upon harpacticoid copepods, which generally inhabit the waters in, or just above the sediments (Pearcy 1962). Young-of-the-year or older juveniles of winter flounder feed upon harpacticoid and calanoid copepods, gastropod larvae, amphipods and polychaetes (Franz and Tanacredi 1992, Stehlik and Meise 2000, Duffy-Anderson and Able 2001, Meng et al. 2001). Because winter flounder larvae feed primarily upon zooplankton and zooplankton feed primarily on phytoplankton, understanding winter flounder requires knowledge of the plankton ecology of their habitat. Thus, a review of Mt. Hope Bay and Narragansett Bay plankton ecology will be presented below.

Phytoplankton

A summary of Narragansett Bay plankton studies may serve as useful background for comparison with Mt. Hope Bay plankton studies. Narragansett Bay is a phytoplankton-based estuary in that the photosynthetic base of the food web is microalgal rather than macrophytic. Phytoplankton in lower Narragansett Bay has been studied intensively for over three decades (Smayda 1957, 1973, 1983, 1984, 1998, Pratt 1959, 1965, 1966, Martin 1965, 1970, Durbin et al. 1975,

Hitchcock and Smayda 1983, Karentz and Smayda 1984, 1998, and references therein). The distinctive phytoplankton feature of the bay is a winter-spring diatom bloom initiating in December-February, and persisting through April. This is in contrast to the typical ephemeral spring bloom observed in other coastal waters. Smayda (1957) attributed the winter-spring bloom in Narragansett Bay to the shallow holomictic water column, which retains phytoplankton in the euphotic zone. Initiation of this bloom in some years in December-January during the annual zooplankton minimum has led to the conclusion that the bloom is prompted by release of copepod grazing pressure (Pratt 1959, Martin 1970). However, Deason (1980) concluded that the effect of grazing on bloom initiation is usually minimal. In other years, bloom initiation appears primarily in response to increasing light after the December-January minimum (Hitchcock and Smayda 1983). The varying explanations for Narragansett Bay winter-spring bloom initiation reflect the multi-factorial and annually variable nature of Narragansett Bay plankton dynamics (Smayda 1998).

Annual variations in the magnitude of the winter-spring bloom have been related to nutrients available prior to bloom initiation, and collapse of the bloom in late spring often co-occurs with reductions in nutrient levels. Inorganic nitrogen appears to be of primary importance, although trace metals may be important for some phytoplankton species (Smayda 1973). There is often a smaller late summer-early fall phytoplankton bloom in some years.

The phytoplankton assemblage may comprise a total of hundreds of species but is typically dominated by only a few at any given time (Karentz and

Smayda 1984). The diatom *Skeletonema costatum* is present and often abundant year-round, and is an important component of both winter-spring and summer-fall blooms. Other winter-spring bloom species include the diatoms *Detonula confervacea* and *Thalassiosira nordenskioldii*, and summer-fall bloom taxa include the diatoms *Leptocylindrus danicus* and *Rhizosolenia fragilissima* (now *Dactyliosolen fragilissima*) and the microflagellate *Olisthodiscus luteus* (= *Heterosigma akashiwo*). Other erratically abundant taxa include the diatoms *Asterionella glacialis* (= *A. japonica*, now *Asterionellopsis glacialis*), *Thalassionema nitzschoides*, various species of the genus *Chaetoceros*, and various dinoflagellates such as *Katodinium rotundatum* (now *Heterocapsa rotundata*) and *Prorocentrum redfieldii* (Karentz and Smayda 1984).

The succession of various species is complicated, and highly variable from year to year (Karentz and Smayda 1984). Seasonal occurrence of many species over a 22-year period of weekly samples appeared strongly related to temperature and light intensity, but for other species these relationships were complex and/or obscure.

Due to essentially weekly sampling since 1959 by Smayda and the URI Graduate School of Oceanography, the phytoplankton of Narragansett Bay is almost certainly better known than for any other coastal waters of the world, but there can still be surprises. In summer of 1985 there was a huge, unprecedented bloom in Narragansett Bay and Long Island, New York and New Jersey coastal waters of a previously undescribed picoplankter *Aureococcus anorexefferens* (Sieburth et al. 1988, Cosper et al. 1989, Smayda and Fofonoff 1989, Smayda and

Villareal 1989a, 1989b, and references therein). This organism formed what has become known as the "brown tide," and it had deleterious effects on zooplankton, mussels, scallops, benthic larval abundance, anchovy fecundity, and kelp beds. Despite the fact that phytoplankton had been studied in Narragansett Bay for decades, *Aureococcus* had never been seen there before. The causes of this bloom are subject to debate, but its unprecedented occurrence is apparently not. The implication is that even in intensively sampled coastal waters where it might be thought that the envelope of variability for phytoplankton events is long established, unprecedented and unpredictable blooms are always a possibility.

Mt. Hope Bay Phytoplankton

Mt. Hope Bay phytoplankton abundance has been assessed as part of BPPS's monitoring program since 1972. Sampling at a minimum of three stations (near BPPS intake, mouth of Lee River and south of Spar Island) was carried out at least once monthly (but at up to a weekly frequency) from February of 1972 to December 1985. Samples were collected at surface and near-bottom depths (or, from 1972 to 1980, at 100%, 50%, 25%, 10% and 1% light level depths) for phytoplankton species identification and enumeration as well as for chlorophyll and nutrient concentration determinations. The results of the February 1972 to February 1979 phytoplankton monitoring effort (along with the results of zooplankton and nutrient studies) have been summarized by Toner (1981). Following 1985, phytoplankton abundance (and the proxy measure of chlorophyll concentration), zooplankton abundance, and dissolved inorganic nutrient

concentration were not routinely monitored. A one-year resumption of the monitoring effort occurred from March 1997 to February 1998. This one-year period of sampling was undertaken to “determine if any gross changes in [plankton] population, chlorophyll-a, nutrients or productivity have occurred since the earlier [pre-1985] studies” (MRI 1999).

Toner (1981) presented results of plankton sampling at two stations in Mt. Hope Bay (near Brayton Point intake and south of Spar Island) where phytoplankton and zooplankton were sampled monthly from 1972-1979. Phytoplankton were preserved in 0.5% gluteraldehyde. Zooplankton were collected via pump sampling ca. 250 L through approximately 50- and 64-mm-mesh nets. In addition, nutrients and hydrographic and meteorological parameters were sampled. Toner found that the phytoplankton consisted of some 230 taxa, primarily dominated by diatoms. Warm-water (>18°C) taxa included the diatoms *Rhizosolenia delicatula* and *R. setigera*, and the flagellates *Calycomonas ovalis* and *Olisthodiscus luteus*. Cold-water taxa (<10°C) included the diatoms *Skeletonema costatum*, *Thalassionema nitzschooides*, *Thalassiosira nordenskioldii*, *Asterionella formosa*, *Coethron criophilum*, *Detonula confervacea*, and the dinoflagellates *Heterocapsa triquetra* and *Peridinium trochoideum*. Increased winds appeared correlated with increases of phytoplankton abundance, presumably by resuspension of benthic nutrients into the shallow water column. Nutrient levels were significantly inversely correlated with phytoplankton abundance, presumably due to utilization. The zooplankton was dominated by copepods and copepod nauplii, with dominance by *Acartia clausi* (= *A.*

hudsonica) in winter and spring, and *A. tonsa* in summer and fall. Gastropod veligers were also abundant from May through November.

A total of 73 phytoplankton species or other taxonomic categories were identified in a more recent (1997–1998) survey of Mt. Hope Bay phytoplankton conducted by MRI (1999). The identified phytoplankton (see Table 4.1) are typical of north temperate estuarine phytoplankton assemblages, with an increased epiphyton component (i.e., epiphytic and ptychopelagic pennate diatoms), as expected in a shallow estuary like Mt. Hope Bay. The diversity of Mt. Hope Bay phytoplankton as identified by the MRI (1999) surveys is lower than that of the west passage of lower Narragansett Bay, where ca. 300 species of phytoplankton have been identified (Hargraves 1988). This difference likely reflects the different lengths of the study periods and different levels of taxonomic expertise applied in the two studies rather than an actual difference in phytoplankton diversity in the two regions. Earlier work (1972 to 1979) in Mt. Hope Bay identified 230 species and categories of phytoplankton (Toner 1981), which is similar to the phytoplankton species diversity of lower Narragansett Bay as listed by Hargraves (1988), suggesting that either 1997–1998 was a year of lowered Mt. Hope Bay phytoplankton diversity or there was a lesser degree of taxonomic refinement applied to the 1997–1998 MRI phytoplankton analyses.

The Mt. Hope Bay phytoplankton community is diatom-dominated (Toner 1981, MRI 1999) but has a greater proportion of euglenoids, chlorophytes and cryptophytes than has been observed in lower Narragansett Bay (Smayda 1957, Pratt 1959). As might be expected, the proportion of freshwater forms

(*Asterionella formosa*, *Tabellaria fenestrata*, etc.) in Mt. Hope Bay is greater than in lower Narragansett Bay. Dominant Mt. Hope Bay phytoplankton observed by MRI (1999) are listed in Table 4.1. The four stations (I – at the BPPS intake, C –

Table 4.1. Mt. Hope Bay phytoplankton identified in MRI 1997-1998 surveys (MRI 1999). Note Toner (1981) refers to 230 species or categories of phytoplankton in Mt. Hope Bay in 1972-1979.

Bacillariophyceae-planktonic

Asterionella formosa
Biddulphia spp.
Chaetoceros spp.
Coscinodiscus spp.
Cyclotella glomerata
Cyclotella meneghiniana
Cyclotella spp.
Ditylum brightwellii
Eucampia zodiacus
Hemiaulus sinensis
Leptocylindrus danicus
Rhizosolenia spp.
Skeletonema costatum
Thalassiosira spp.
 Unidentified diatom

Chlorophyceae

Pyramimonas spp.
Tetraselmis spp.

Dinophyceae

Amphidinium spp.
Ceratium spp.
Dinophysis lachmanni
Dinophysis spp.
Glenodinium lenticula
Gymnodinium nelsoni
Gymnodinium spp.

Bacillariophyceae (epiphytic, marine)

Acnantes spp.
Amphiprora spp.
Amphora spp.
Cylindrotheca closterium
Cylindrotheca spp.
Diploneis spp.
Licmophora spp.

Bacillariophyceae (epiphytic, brackish)

Brebessonia boeckii
Cocconeis spp.
Melosira spp.
Navicula spp.
Nitzschia spp.
Pleurosigma spp.
Surirella spp.

Bacillariophyceae (epiphytic, fresh water)

Cymbella spp.
Diatoma spp.
Eunotia spp.
Fragillaria spp.
Gomphonema spp.
Meridon circulare
Synedra spp.
Tabellaria fenestrata
Tabellaria spp.

Gyrodinium spp.
Heterocapsa triquetra
Katodinium spp.
Peridinium spp.
Polykrikos kofoidi
Prorocentrum micans
Prorocentrum spp.
Unid. naked dinophyceae

Desmidaceae

Crucigenia quadrata
Scenedesmus spp.

Chrysophyceae

Dinobryon sertularia
Dinobryon spp.
Pseudopedinella pyriforme
Unid. flagellate

Cryptophyceae

Chilomonas spp.
Chroomonas spp.
Chroomonas/Hemiselmis sp.
Cryptomonas pseudobaltica
Cryptomonas spp.
Cryptomonas testacea

Euglenophyceae

Euglena spp.
Eutreptia viridis

Cyanophyceae

Anacystis spp.
Oscillatoria spp.
Spirulina spp.

Other Categories

Closterium spp.
Pediastrum spp.
Unknown pigmented cell

mouth of Lee River, F – south of Spar Island, A' – south of Brayton Point) surveyed by MRI in 1997–1998 were similar in phytoplankton abundance and community composition. No significant difference in total phytoplankton abundance (as tested by fixed-effects ANOVA of log transformed data; $p < 0.05$) was found among the four stations, indicating no gross differences in the spatial distribution of Mt. Hope Bay phytoplankton.

Total phytoplankton abundance (as cells ml^{-1}) appears to have shown no change between 1972-1985 and 1997-98. However, long-term changes in Mt. Hope Bay community composition still may have occurred, but the 1986 to 1996 gap in available phytoplankton data complicates analysis. Preliminary analyses indicate that Mt. Hope Bay dinoflagellate abundance may be on the increase (see MRI 1999). Comparison of 1972-1985 versus 1997-1998 dinoflagellate annual abundance patterns shows elevated dinoflagellate abundance, relative to the 1972-1985 pattern, that lasted from mid-summer (July) through the winter months and into spring (April) (Figure 4.1). Much of the 1997-98 elevated dinoflagellate abundance appears to be due to a bloom of the potentially inimical dinoflagellate *Prorocentrum* spp. that reached ca. 7,400 cells ml^{-1} at station F (south of Spar Island) in January 1998 (MRI 1999). This *Prorocentrum* bloom was ca. 37-fold greater than the 1972 to 1985 January total dinoflagellate abundance of near 150 to 250 cells ml^{-1} . The dinoflagellate *Prorocentrum minimum*, common in Narragansett Bay and presumably in Mt. Hope Bay, has been found to form blooms in response to high nitrogen loading in Japanese and southeastern U.S.

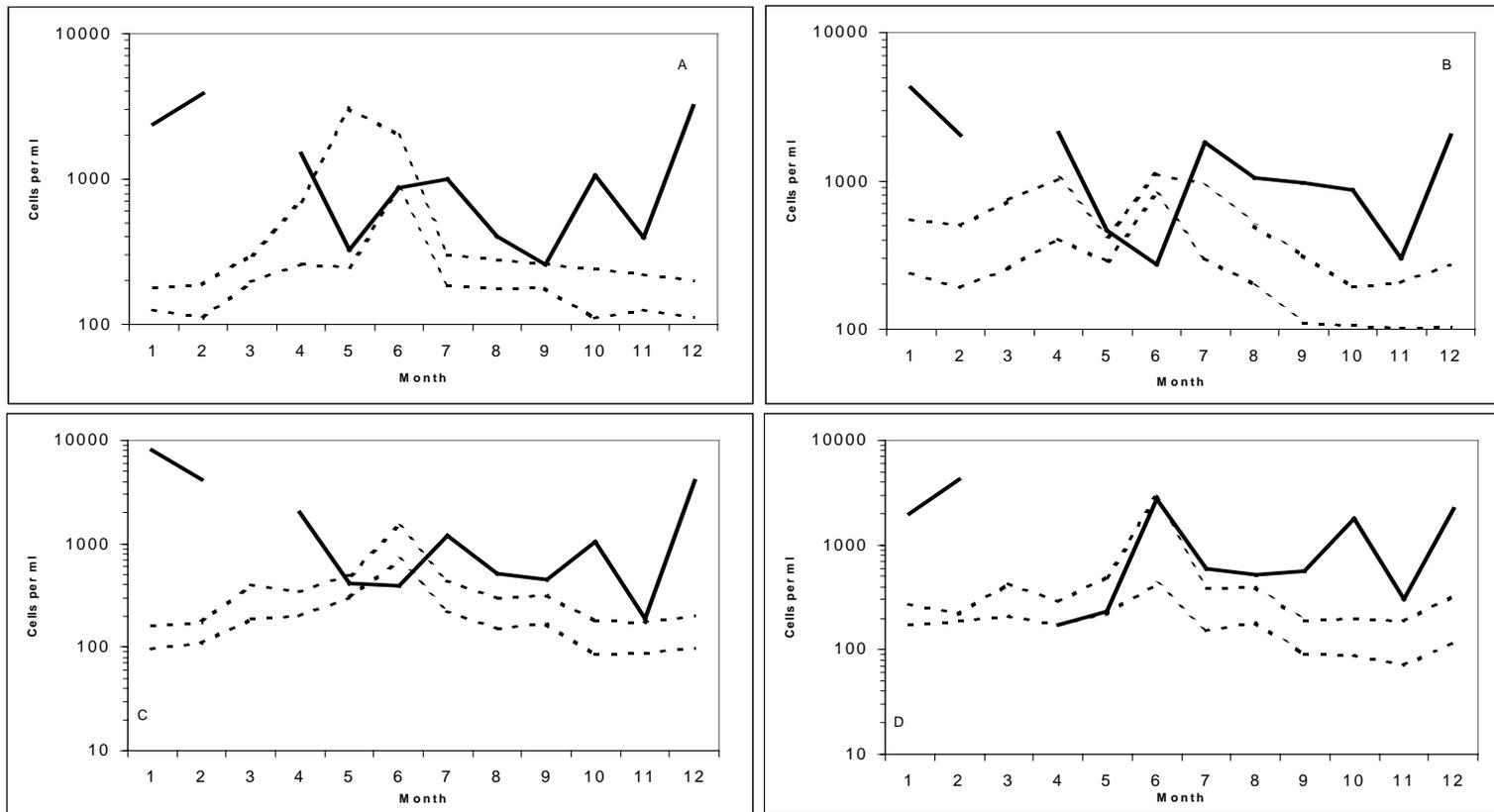


Figure 4.1. 1997-1998 dinoflagellate abundance (heavy line) compared to 1972-1985 mean ± 1 standard error dinoflagellate abundance (dashed line) at four Mt. Hope Bay monitoring stations. Panel A shows pattern at Mt. Hope Bay station 'I', panel B shows station 'C', panel C shows station 'F' and panel D shows station 'A' 1997-1998 and 1972-1985 mean dinoflagellate abundance annual cycles. (From data in MRI 1999.)

coastal waters (Burkholder 1998). Given the potential food- web-altering consequences of dinoflagellates (Turner and Tester 1997, Turner et al. 1998), discerning whether elevated dinoflagellate abundance, especially *Proro-centrum* abundance, is a regularly recurring feature of recent (post-1985) Mt. Hope Bay phytoplankton successional cycles, or was an anomalous feature of the 1997-1998 phytoplankton cycle, may be an important task of future Mt. Hope Bay research.

Chlorophyll concentration in 1997-1998 appears to be decreased relative to that seen in 1972-1983 (Figure 4.2, MRI 1999). This late-1990's decline in Mt. Hope Bay chlorophyll concentration is consistent with declines in lower

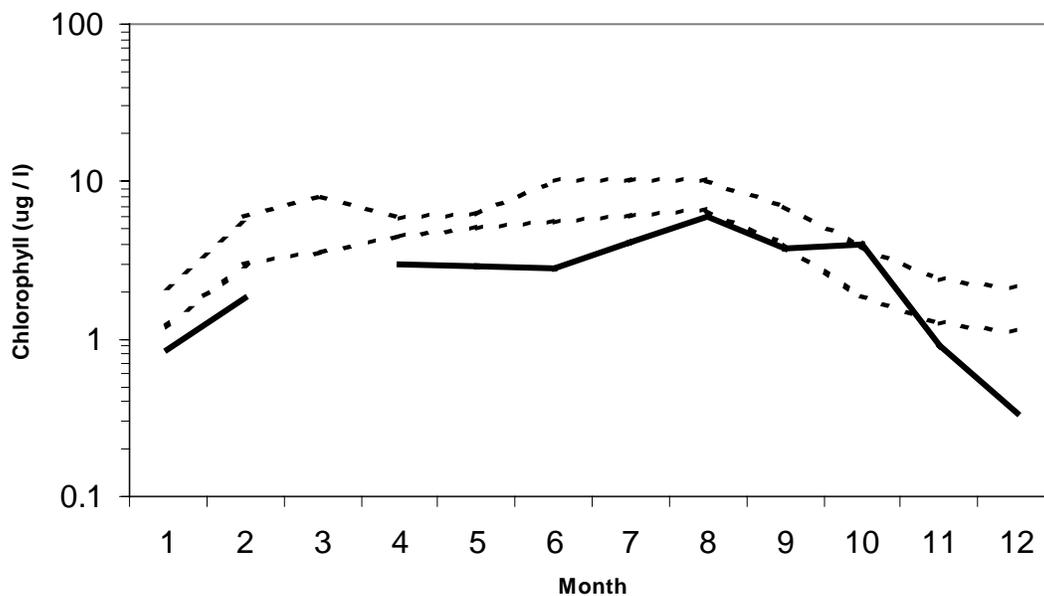


Figure 4.2. Monthly change in Mt. Hope Bay chlorophyll concentration compared between 1972-1985 (dashed line is mean \pm 1 SE) and the 1997-1998 sample year (solid line). (Adapted from Figure P-23, MRI 1999.)

Narragansett Bay chlorophyll concentration (Li and Smayda 1998). A main feature of the recent decline in lower Narragansett Bay chlorophyll has been a decrease in the size and relative contribution of the winter-spring bloom to the annual chlorophyll cycle (Keller et al. 1999b). Mt. Hope Bay chlorophyll in

1997-98 showed declines in the winter-spring as well as in the summer months relative to the pattern seen in 1972-1983. For example, the 4-station (stations I, C, F, A', all depths) mean chlorophyll concentration in February 1998 was ca. 2.4 $\mu\text{g l}^{-1}$ compared to a 1973 to 1983 mean February chlorophyll value of near 4 $\mu\text{g l}^{-1}$; while December 1997 chlorophyll (4 station mean) was ca. 0.4 $\mu\text{g l}^{-1}$ compared to a 1972-83 mean December value of near 1.75 $\mu\text{g l}^{-1}$ (MRI 1999). Summer 1997 declines in chlorophyll, relative to the mean 1972 to 1983 levels, were of similar magnitude, with a June 1997 mean (4 stations, all depths) chlorophyll level of near 4 $\mu\text{g l}^{-1}$ compared to a 1972 to 1983 mean June level of near 7 $\mu\text{g l}^{-1}$ (MRI 1999; see their Figure P-23). As with phytoplankton abundance and community composition, identification of any trends in Mt. Hope Bay chlorophyll concentration, and identification of mechanisms causing such trends remain unknown.

In lower Narragansett Bay recent increases in winter zooplankton abundance and feeding activity, partially in response to warming winter water temperature, have been linked to declines in winter-spring chlorophyll concentration (Keller et al. 1999b). It is worth noting that early winter 1997-1998 Mt. Hope Bay zooplankton abundance was elevated relative to the 1972-1985 mean winter levels (Figure 4.3, MRI 1999). For example, January 1998 total zooplankton abundance at station I (BPPS Intake) was near 300,000 animals m^{-3} compared to a 1972-985 mean January level of near 20,000 animals m^{-3} (Figure 4.3). Most of this increase appears to be due to elevated abundance of *Acartia*

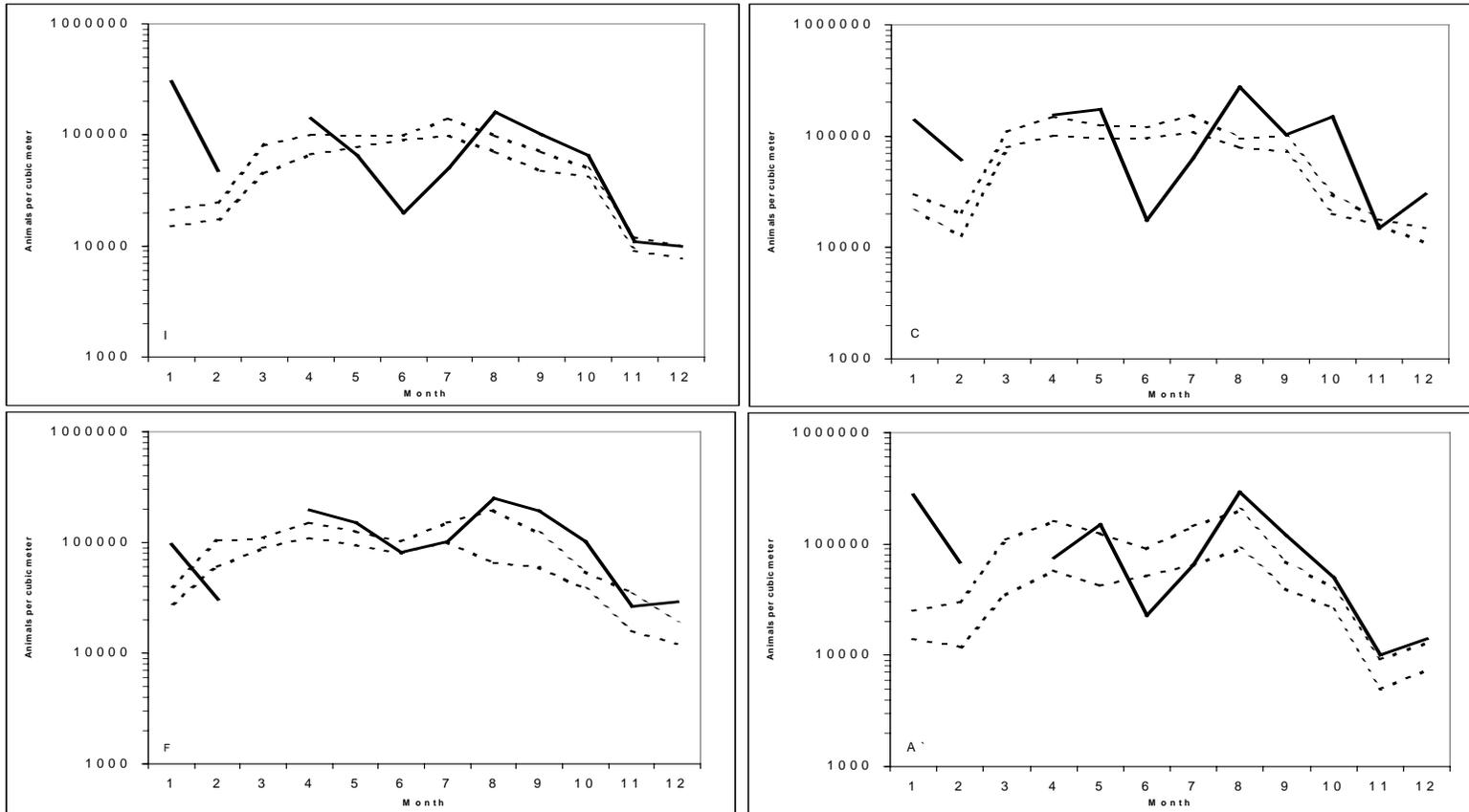


Figure 4.3. 1997-1998 zooplankton abundance (heavy line) compared to 1972-1985 mean ± 1 standard error zooplankton abundance (dashed line) at four Mt. Hope Bay monitoring stations. Panel A shows pattern at Mt. Hope Bay station 'I', panel B shows station 'C', panel C shows station 'F' and panel D shows station 'A' 1997-1998 and 1972-1985 mean zooplankton abundance annual cycles. (Adapted from data in MRI 1999.)

hudsonica, the dominant winter copepod in Mt. Hope Bay, which had a January 1998 abundance of 25,830 animals m⁻³ at Station I compared to a 1972 to 1985 mean abundance of ca. 700 animals m⁻³ at Station I (Figure 4.4, MRI 1999). The Mt. Hope Bay observation of 1997-98 increased winter-spring zooplankton abundance and concomitant declines in winter-spring chlorophyll concentration are consistent with observations, and inferred mechanisms, of Keller et al. (1999b) in lower Narragansett Bay. A secondary effect of increased winter-spring zooplankton abundance and grazing pressure may be selection, via differential grazing, of dinoflagellates (such as *Prorocentrum* spp. as seen in Mt. Hope Bay in January 1998) rather than the typical winter diatom-dominated flora.

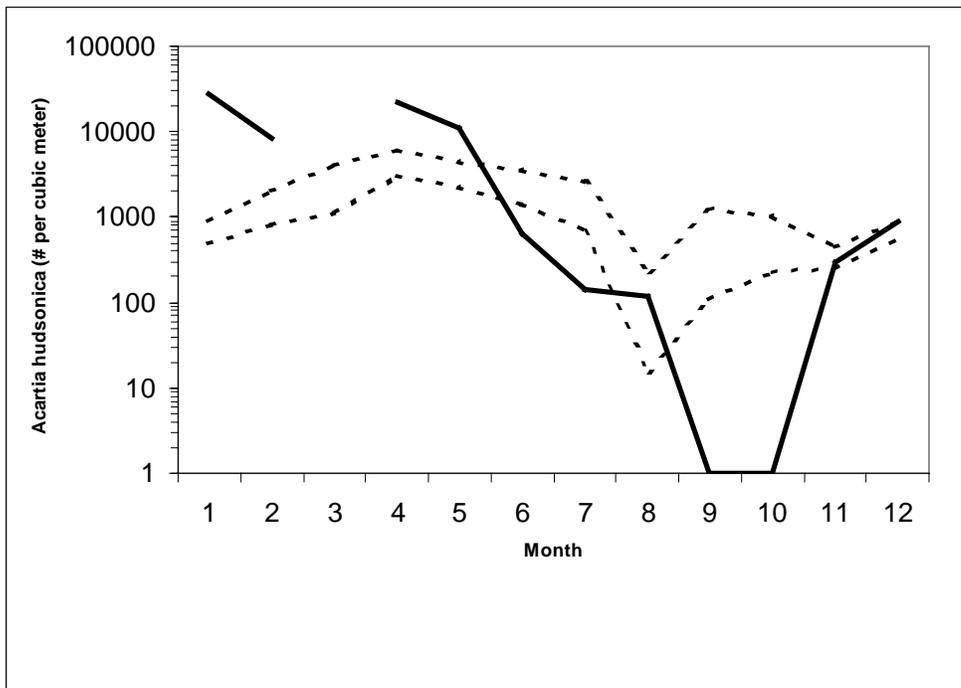


Figure 4.4. Seasonal pattern of *Acartia hudsonica* abundance in Mt. Hope Bay compared between the 1972-1985 (dashed line is mean \pm 1 S.E.) and the 1997-1998 sampling year (solid line). (Adapted from Figure Z-25 in MRI 1999.)

Zooplankton

The well-studied zooplankton of lower Narragansett Bay provide a basis for the limited number of Mt. Hope Bay zooplankton studies. The metazoan zooplankton of Narragansett Bay are dominated by seasonally alternating, congeneric species of copepods (Jeffries 1962, 1967). *Acartia hudsonica* is abundant from late fall to early summer, and *Acartia tonsa* is dominant in summer and fall (Frolander 1955, Martin 1965, Hulsizer 1976, Durbin and Durbin 1981, 1989). The seasonality of occurrence or absence in the plankton is due to alternating periods of population diapause by resting fertilized eggs of these copepods (Zillioux and Gonzalez 1972, Sullivan and McManus 1986). Other important metazoan zooplankters include pulses of various meroplanktonic larvae of benthic invertebrates, and the ctenophore *Mnemiopsis leidyi*. This ctenophore can become extremely abundant and ecologically important in Narragansett Bay in summer and fall (Kremer 1975a, 1975b, 1976a, 1976b, 1977, 1979, Kremer and Nixon 1976, Deason 1982, Deason and Smayda 1982). *Mnemiopsis* bloom abundance and bloom duration may be increasing in Narragansett Bay in response to warming winter water temperatures (Sullivan et al. 2001).

The potential population production rates of the *Acartia* congeners have been investigated by Ann and Ted Durbin, and their collaborators, using a combination of field and laboratory studies (Durbin and Durbin 1978, 1981, 1989, 1992a, 1992b, Durbin et al. 1983, 1992). Potential production rates were highly temperature-dependent. Even though the summer population of *A. tonsa* had a smaller biomass than that of *A. hudsonica* in winter, temperature effects on

reproductive rates allowed the summer *A. tonsa* population to potentially double in less than a day, whereas this could require over nine days for *A. hudsonica* in winter. From studies to estimate if copepods were growing at maximal rates in the bay, the Durbins found that *Acartia tonsa* fecundity and body weight closely tracked variations in natural phytoplankton food supply, and that fecundity and body weight could always be increased by offering surplus food. This indicated that *A. tonsa* production in the bay was continuously food-limited in summer (Durbin et al. 1983).

This food limitation of maximum growth and production rates appeared due to quality rather than quantity of food. Small-sized cells that may have been inefficiently captured and/or nutritionally inadequate dominated the natural phytoplankton in summer. During the winter-spring period, the Durbins found different results for *Acartia hudsonica* (Durbin et al. 1992, Durbin and Durbin 1992a, 1992b). Egg production rates were correlated with temperature, but not with concentration of natural phytoplankton. When additional cultured phytoplankton was offered as food, egg production rates did not increase. This suggests that food was sufficient in quality and quantity to allow actual copepod production rates to approach maximal potential rates. Mild food limitation of *Acartia hudsonica* developed in May-June, but this was more in terms of body weight than egg production. This was partly due to changing age groups during this period, as autumn-hatched (senescent) over-wintering adults were being replaced by newly hatched, winter-spring-hatched adults.

Higher-trophic-level predators such as ctenophores may impact relations between phytoplankton and zooplankton. Deason and Smayda (1982) investigated ctenophore-zooplankton-phytoplankton interactions during six summer-fall seasons (1972-77) in lower Narragansett Bay. In 4 of these years, summer increases in the ctenophore *Mnemiopsis leidyi* accompanied rapid declines in copepods and other zooplankton, and increases in phytoplankton. The summer phytoplankton blooms were typically dominated by *Skeletonema costatum*, the abundance of which was correlated with that of ctenophores. Termination of such blooms coincided with declining ctenophores and increasing copepods in two years. Thus, it appears that summer phytoplankton dynamics are typically affected by, and possibly regulated by ctenophore predation on the copepods that are major grazers of the phytoplankton.

There have been several major studies of tintinnids (thecate ciliate protozoans) in Narragansett Bay (Hargraves 1981, Verity 1984, 1985, 1986a, 1986b, 1987). These are important since tintinnids and other microzooplanktonic protists have long been suspected to be major grazers of the smaller nanoplankton (see Pierce and Turner 1992 for a review). Hargraves (1981) described seasonal patterns of tintinnid abundance and occurrence from weekly samples in Narragansett Bay from March through November, 1976. There were bimodal peaks in abundance, with a maximum in late May-early June, a precipitous decline in late June, a lesser peak in late summer, and a gradual decline in October. There were 33 tintinnid species recorded from 8 genera, with the genus *Tintinnopsis* accounting for 21 species and 90% of total abundance. Tintinnid

abundance was not obviously correlated with salinity, temperature, or nanoplankton chlorophyll abundance.

Verity (1986a) used tintinnid cultures established from Narragansett Bay to determine that clearance, ingestion, respiration, excretion and growth rates for two species of *Tintinnopsis* increased with increasing temperature. Applying these laboratory-determined values to field estimates of tintinnid and phytoplankton abundance and production, Verity determined that the tintinnid population ingested a carbon equivalent of 16-26% of total annual net phytoplankton primary production, and 32-52% of <10 mm nanoplankton production. Tintinnid nitrogen excretion was sufficient to support 11-18% of net primary production.

Verity (1986a) compared photosynthetic nanoplankton growth rates and total microzooplankton (10-202 mm) grazing rates using in situ incubations over the annual cycle in Narragansett Bay. He found that nanoplankton growth rates were linearly related to temperature, and microzooplankton grazing rates were linearly related to nanoplankton production. Verity estimated that the total microzooplankton assemblage grazed an annual mean of 62% of nanoplankton production. In the same incubations, Verity (1986b) found that maximum tintinnid species growth rates were regulated by temperature, but that net tintinnid community growth rates were influenced primarily by food quality and availability. Tintinnid growth rates were suppressed by blooms of small solitary centric diatoms and the toxic flagellate *Olisthodiscus luteus*. Excluding these blooms, tintinnid growth rates increased asymptotically with nanoplankton

biomass and production rates. Tintinnid species exhibited maximum growth rates that exceeded 1-2 doublings d^{-1} . thus, high abundance (up to 10^5 cells l^{-1} - Verity 1987) and rapid growth of tintinnids suggests that they are important grazers of nanoplankton in Narragansett Bay, and presumably Mt. Hope Bay as well.

Mt. Hope Bay Zooplankton

Mt. Hope Bay zooplankton have been sampled by MRI personnel at monthly or greater intervals, at three stations in Mt. Hope Bay (stn. F – south of Spar Island, stn. C – mouth of Lee River, stn. I – BPPS intake) from 1972 to 1985, and from April 1997 to February 1998. The results of the 1972 to 1979 portion of this monitoring were summarized by Toner (1981), who found that Mt. Hope Bay zooplankton were dominated by copepods and copepod nauplii, with dominance by *Acartia clausi* (= *A. hudsonica*) in winter and spring, and *A. tonsa* in summer and fall. Gastropod veligers were also abundant from May through November, reaching 10,000 m^{-3} in August - September. Dominant Mt. Hope Bay zooplankton observed in 1997-98 sampling (MRI 1999) are listed in Table 4.2. Total zooplankton density fluctuated, on average, from a minimum of near 3,000 to 5,000 animals m^{-3} in winter to peaks near 100,000 animals m^{-3} in late spring or summer. Fluctuations in zooplankton abundance were closely linked, with a significant positive correlation, leading Toner (1981) to suggest that zooplankton in Mt. Hope Bay are food-limited and under ‘bottom-up’ control. Similar conclusions have been drawn for lower Narragansett Bay zooplankton (Durbin et al. 1983).

Table 4.2. Mt. Hope Bay zooplankton identified in 1997-1998 MRI surveys (MRI 1999).

| | | |
|-------------------------------|--------------------------|----------------------------|
| <u>Protista</u> | <u>Polychaeta</u> | unid. cyclopoid |
| Codenella spp. | Chaetogaster spp. | unid. parasitic |
| Favella spp. | unid. polychaete | copepod |
| Halteria spp. | larvae | |
| Tintinnid | <u>Arthropoda</u> | <u>Branchiopoda</u> |
| (unidentified) | <u>Crustacea</u> | Evadne normanni |
| | <u>Copepoda</u> | Podon leuckarti |
| <u>Platyhelminthes</u> | Acartia hudsonica | Podon polyphemoides |
| <u>Turbellaria</u> | Acartia tonsa | unid. cladocera |
| Turbellaria | Calanoid copepodite | unid. conchostracan |
| (unidentified) | Centropages hamatus | |
| | Centropages spp. | <u>Cirripedia</u> |
| <u>Coelenterata</u> | Centropages typicus | Balanus cyprinus |
| Coelenterate | Cyclopoid copepodite | larvae |
| (unidentified) | Eurytemora | Balanus nauplii |
| | americana | |
| | Eurytemora herdmani | <u>Malacostraca</u> |
| <u>Aschelminthes</u> | Eurytemora spp. | Crab zoea |
| <u>Rotifera</u> | Harpacticoid | Cumacean |
| Keratella cochlearis | copepodite | (unidentified) |
| Keratella heimale | Microsetella | Gammarid amphipod |
| Keratella serrulata | norvegica | Mysid (unidentified) |
| Keratella spp. | Copepod nauplii | Mysid larvae |
| Synchaeta spp. | (unid.) | |
| Trichocerca similis | Oithona nana | <u>Xiphosura</u> |
| Trichocerca spp. | Oithona similis | Polyphemus larvae |
| Unidentified rotifer | Oithona spinirostris | |
| | Oithona spp. | <u>Chordata</u> |
| <u>Nematoda</u> | Paracalanus | Tunicate larvae |
| Nematode | crassirostris | |
| (unidentified) | Paracalanus spp. | |
| | Pseudocalanus | |
| <u>Mollusca</u> | minutus | |
| <u>Gastropoda</u> | Pseudocalanus spp. | |
| Gastropod larvae | Pseudodiaptomus | |
| | coronatus | |
| <u>Bivalvia</u> | Pseudodiaptomus spp. | |
| Arca transversa | Saphirella spp. | |
| Mya arenaria | Temora longicornis | |
| Mytilus edulis | Temora spp. | |
| unid. bivalve larvae | Tortanus spp. | |
| | unid. harpacticoid | |
| <u>Annelida</u> | unid. calanoid | |

Comparison of 1972 to 1985 zooplankton patterns to those of April 1997 through February 98 suggest that in 1997-1998, Mt. Hope Bay zooplankton abundance was increased relative to the long-term (1972-1985) pattern (MRI 1999). Marine Research, Inc.'s 1999 study revealed that 1997/98 *Acartia hudsonica* abundance exceeded the 1972-1985 mean abundance levels in April, May, December 1997 and in January and February of 1998. The elevated winter 1997/1998 *A. hudsonica* abundance was reflected in the December 1997 through February 1998 total zooplankton abundance, which were higher than the corresponding 1972-1985 mean values. Comparison of 1997/98 total zooplankton abundance to that of 1981 to 1985 showed that 1997-98 total zooplankton abundance was greater than that of 1981 through 1983, but not significantly different from that of 1984 and 1985 (MRI 1999). In contrast to the apparent increase in copepod abundance, 1997/98 abundance of polychaetes, nauplii and gastropods displayed no change relative to the 1972 to 1985 period. As mentioned in the phytoplankton section, recent (1997/98) increases in winter zooplankton abundance are consistent with observations made by Keller et al. (1999b) in lower Narragansett Bay.

Ichthyoplankton

There have been three major studies of ichthyoplankton in Narragansett and Mt. Hope Bays prior to 1989-90, as summarized by Durbin and Durbin (1990). In 1957-58 Herman collected fish eggs and larvae at 4 stations in the East Passage and West Passage of lower Narragansett Bay, the northern entrance to the

Sakonnet River and in lower Mt. Hope Bay. These data were presented in a Master's thesis from the University of Rhode Island (Herman 1958) and later published (Herman 1963).

From 1972-73 there was an extensive survey of ichthyoplankton at 160 stations throughout Narragansett and Mt. Hope Bays, performed by Marine Research, Inc. of Falmouth, Massachusetts. These data were presented by MRI (1974) and later published by Bourne and Govoni (1988). Additional ichthyoplankton monitoring by Marine Research, Inc. has continued in Mt. Hope Bay since 1972. These data were presented in quarterly progress reports to Narragansett Electric Power Company, and were summarized by Durbin and Durbin (1990). The following summary is abstracted from Durbin and Durbin (1990).

The 1972-73 survey reported 42 species of fish eggs and/or larvae with maximum abundance in June-July and minimum abundance in September-January. The more abundant species were, in order, anchovies, menhaden, winter flounder, tautog, cunner, sand lance, weakfish, windowpane, silversides, mackerel, seaboard goby, and fourbeard rockling. Sand lance and winter flounder were winter-spring spawners with peak larval abundance in April. Fourbeard rockling and mackerel were spring-early summer spawners with peak larval abundance in May-June. Late spring-early summer spawners included bay anchovy, tautog, cunner, silversides, weakfish, windowpane, and seaboard goby. Atlantic menhaden had an extended spawning period with greatest larval numbers in spring and fall. There were three main groups of species in terms of

geographic areas of abundance. Of the 12 most abundant species, 8 (menhaden, anchovy, winter flounder, tautog, weakfish, windowpane, silversides, seaboard goby) were most abundant in upper Narragansett/Mt. Hope Bays. Sand lance, rockling and mackerel were most abundant in the lower East Passage, and cunner was abundant in both the upper and lower bay. Thus, the upper and most anthropogenically impacted regions of Narragansett and Mt. Hope Bays were also the major spawning and nursery areas for fish.

Herman (1958, 1963) recorded 34 taxa from his 1957-58 study, all of which were later found by the 1972-73 Marine Research, Inc. study, but the ranking of abundant species was different for the two studies. Herman found that the 12 most abundant species were (in order) sculpin, silversides, cunner, anchovy, scup, sand lance, searobin, butterfish, winter flounder, tautog, northern pipefish, and radiated shanny. Of these, only half were among the most abundant species in the Marine Research study. This was mainly due to relative abundances of 5 species (winter flounder, silversides, goby, sand lance, and sculpin) that hatch from demersal eggs. These discrepancies may be due to differences in sampling gear. Herman's nets for larval collections had a coarse mesh (>1 mm) and a bridle in front that might have contributed to larval avoidance, whereas Marine Research used 505 mm-mesh bongo nets (which do not have a bridle in front). However, Herman also used a Clarke-Bumpus sampler with 366-mm mesh to collect eggs, particularly near-bottom. This may have explained Herman's disproportionate collections of larger and/or demersal-spawning taxa relative to Marine Research, which collected eggs and larvae of primarily pelagic-spawning species.

Continuation of collection by Marine Research from 1972-86 reveals that total numbers of larvae have fluctuated approximately 3-fold over this period. There may have also been major changes in the populations of certain species during this period. Since 1972 there has been a precipitous decline in abundance of larvae of Atlantic menhaden, with replacement by bay anchovy (*Anchoa mitchilli*) as the dominant species in Mt. Hope Bay. Sand lance have also fluctuated in abundance, increasing from low numbers in the early to mid-1970's with a peak in 1979, followed by low abundance in 1985-86. Winter flounder larvae also have varied by 5-6 fold, with apparent 4-5 year periodicity of cycles, but do not exhibit a statistically significant decline (Figure 4.5, Collie and Delong 2001). Thus, although the total abundance of fish larvae of all species may be somewhat stable, larval abundance of certain species is apparently not.

Larval Winter Flounder

Geometric Means and 95% Confidence Limits

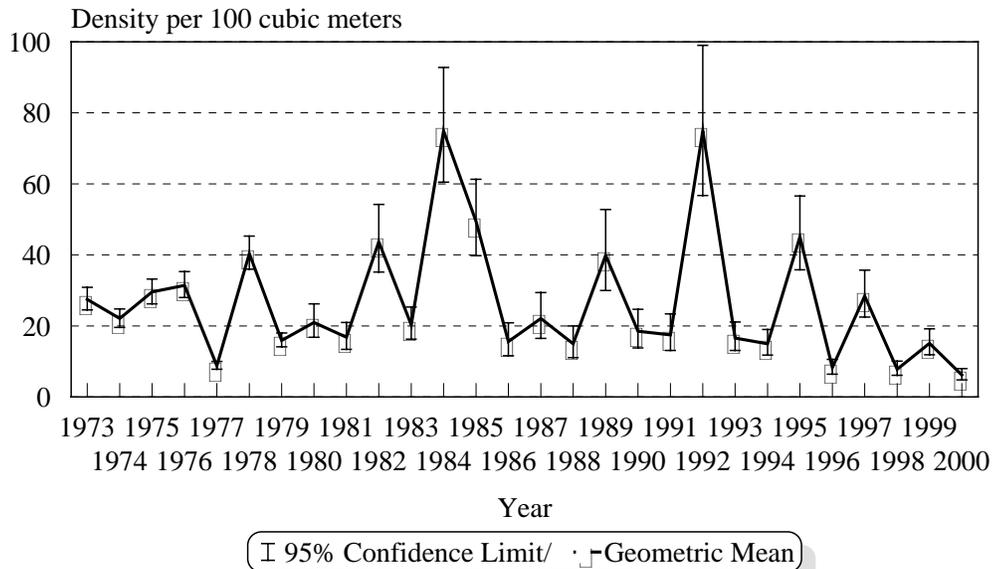


Figure 4.1. Abundance of larval winter flounder collected at the BPPS from 1973-2000. (Reprinted from USGen 2001).

The most recent survey of ichthyoplankton in Narragansett Bay (Keller et al. 1999a) was done in 1989-1990, partly to examine changes in the bay since the last bay wide survey of 1972-73. It used similar sampling protocols to the previous study. The taxonomic composition of eggs and larvae and patterns of seasonal occurrence in 1989-90 were similar to 1972-73. However, abundances of eggs and larvae were lower in 1990 than in 1972-73. Bay anchovy, tautog, and cunner accounted for 86% of the eggs and 87% of the larvae in 1990, compared to only 55% of the eggs and 51% of the larvae in 1972-73. Much of this difference was due to a precipitous decline in abundances of larval menhaden. Winter flounder larvae were significantly less abundant in 1990 than 1973, particularly in the upper bay in Greenwich Bay and the Providence River. Keller et al. (1999a) concluded that there had been a general down-bay shift between the surveys in labrid, anchovy, fourbeard rockling and menhaden eggs, and in anchovy, tautog, cunner, winter flounder and menhaden larvae. This shift was thought to be due to degradation of the pelagic habitat in the upper bay, and the overall decline of ichthyoplankton in Narragansett Bay may have resulted from a combination of habitat loss, overfishing and pollution.

In addition to the adult fish data outlined above, ichthyoplankton abundance has been monitored in Mt. Hope Bay to assess the potential impact of BPPS operations on ichthyoplankton. Fish may be affected by Brayton Point operations in at least three ways: entrainment, impingement, and indirect effects (habitat loss, migration blockage, Q_{10} effects, etc.). Entrainment is the inadvertent capture and passage of fish eggs and larval fish through the plant cooling water

system while impingement is the trapping of fish on the screened water intake. Impingement effects are limited to fish larger than the intake screen mesh (need size?), and are not expected to affect ichthyoplankton. Entrainment and impingement are quantified as part of BPPS monitoring program, with field estimates of numbers of fish entrained and impinged being made by MRI personnel. Impingement was quantified three times per week from 1972 to 1996, and has been monitored daily from 1997 to the present. Entrainment has been estimated under various levels of effort from 1972 to 1985, and 1993 to 2001. MRI has estimated larval fish abundance in Mt. Hope Bay through monthly (or more frequent) surveys in the Bay from June 1972 to 1992. Since 1993, these larval fish surveys have been done only in February to mid-May (coincident with peak winter flounder larvae abundance). Abundance and species composition of larval fish entrained (drawn in with cooling water) by BPPS have been monitored by MRI personnel from June 1972 through the end of 1985, and from 1993 to the present (2001). Bourne and Govoni (1988) have reported on the abundance and distribution of larval fish in Narragansett and Mt. Hope Bays in 1972-1973. A comparison of 1989-90 Narragansett Bay larval fish abundance, distribution and seasonal abundance to the 1972-73 observations of Bourne and Govoni (1988) has been made by Keller et al. (1999a).

Summary

Extensive plankton work in Narragansett Bay provides sound basis for expected plankton processes in Mt. Hope Bay, but site-specific factors affecting

Mt. Hope Bay spatial and temporal plankton distribution need to be identified. MRI phytoplankton, zooplankton and ichthyoplankton data will allow Mt. Hope Bay plankton abundance patterns to be examined for 1972 to 1985. The unfortunate data gap from 1986 to 1996 will prevent more recent analyses. Resumption of phytoplankton and zooplankton sampling (for one year, 1997/1998) in the 1990's revealed several provocative results that may be indicative of recent changes in Mt. Hope. Namely:

1. Apparent increase (ca. 2-fold) in mean ammonia concentration in 1997/98 relative to 1972-85 mean levels.
2. Apparent increase in winter dinoflagellates (especially *Prorocentrum*) in 1997/98 compared to 1972 to 1985 mean pattern.
3. Apparent increase in 1997/98 winter zooplankton relative to 1972-85 mean pattern.
4. Apparent decline in fish egg and larval fish abundance in 1990 survey relative to 1972/73 survey.

Chapter 5. Nekton (Dave Borkman, Jefferson Turner and Rodney Rountree)

Introduction

Except for the fishes, the nektonic community of Mt. Hope Bay has not been well described. Relatively little is known of the distribution and temporal abundance patterns of important decapod crustaceans such as the shrimps *Crangon septemspinosa*, *Paleomonetes intermedius*, *P. pugio*, and *P. vulgaris*, the crabs *Callinectes sapidus*, *Ovalipes ocellatus*, *Libinia* spp., *Cancer* spp., and the hermit crabs *Pagurus* spp., despite their great importance to the ecosystem. The lack of information on *Crangon septemspinosa* is particularly important as there is credible evidence that it is an important predator of winter flounder and other larval fishes (see discussion under the winter flounder section below). More information is available on the fishes, but attempts to investigate temporal and spatial abundance patterns has been limited to a few economically important species susceptible to trawl sampling.

There are approximately 30 years (1971 to present) of monthly to twice-monthly trawl data from various locations in Mt. Hope Bay (source: MRI, October 2001 presentation by M. Scherer) that provide information on the Bay's fish abundance and community composition. A standardized trawl has been made at least once per month at 5 locations (Spar Island, Cole River, Lee River, near BPPS intake, and a 'crossleg' transect south of BPPS) from late 1971 to the present (Figure 5.1). An additional monthly trawl, a Wilcox trawl with a smaller mesh (125 mm), has been implemented since 1993 to census the abundance of smaller fish, especially small winter flounder.

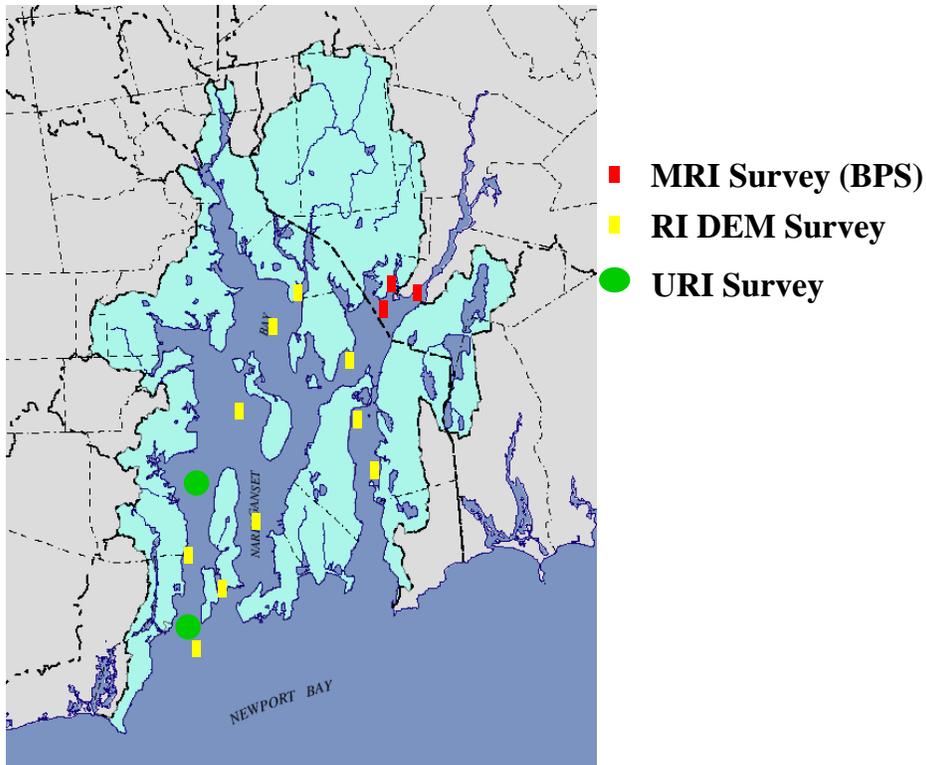


Figure 5.1. Locations of MRI, RIDEM and GSO/URI standard trawl sampling.

Additional sources of Mt. Hope Bay area fish abundance data include the weekly fish trawls taken near the mouth of the West Passage and Rhode Island Sound by GSO personnel since 1959 through the present (Figure 5.1, Jeffries and Johnson 1974, Jeffries and Terceiro 1985, Jeffries et al. 1989). The RIDEM has done a census of juvenile fishes in Narragansett Bay, including one station in Mt. Hope Bay, in the summer/early autumn of 1988 through 1996 (Figure 5.1, Meng and Powell 1999). The Narragansett Bay fish community was surveyed for a single year (June 1971 to May 1972) by Oviatt and Nixon (1973). NMFS provides species and stock-specific catch data for the southern New England area (for example, see Nitschke et al. 2000).

Fishery trends in Mt. Hope and Narragansett Bays

The fish community of lower Narragansett Bay has been monitored via weekly trawls since 1959 and has been more extensively analyzed (Jeffries and Johnson 1974, Jeffries and Terceiro 1985, Jeffries et al. 1988). Additionally, the Rhode Island Department of Fish and Wildlife (RIDFW) has recently summarized RI landings and stock assessment data for several commercially important species (DeAlteris et al. 2000). The above analyses all indicate a recent (post ca. 1983 to 1999) trend of declining abundance of demersal fish and increasing abundance of pelagic fish species in Narragansett Bay. For example, RI landings of demersal winter flounder show dramatic post-1983 declines (discussed in detail below), while RI landings of pelagic longfin squid (*Loligo pealei*) has displayed a nearly 4-fold 1983 to 1999 increase (DeAlteris et al. 2000; see Figure 11 of DeAlteris et al. 2000). This demersal-pelagic dominance swap has also been observed in fishery-independent surveys in lower Narragansett Bay and RI Sound URI/GSO (Figure 5.2, Jeffries and Terceiro 1985).

Jeffries and Terceiro (1985) attributed much of this demersal/pelagic swap to increased availability of food resources due to declines in the abundance of resident and formerly abundant winter flounder. The underutilized food resource is hypothesized to now be used by transient fish (such as scup, butterfish and squid) such that there is a negative correlation between abundance of resident vs. migrant fish in Narragansett Bay. This statistically significant correlation was found by Jeffries and Terceiro (1985) between abundance of resident winter flounder, on the one hand, and migrant ocean pout ($r = -0.55$), red hake ($r = -$

0.47), and silver hake ($r = -0.62$), on the other. This relationship between abundances of transient and Narragansett Bay resident fish has been summarized as a power curve (Jeffries and Terceiro 1985; Figure 5.3).

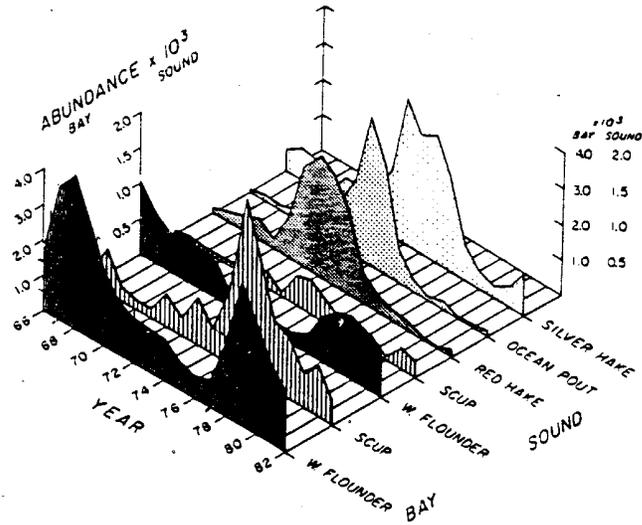


Figure 5.2. Comparison of annual totals of mean monthly trawl catches for demersal fishes in Narragansett Bay and Rhode Island Sound from 1966-1982. (Reprinted from Jeffries and Terceiro 1985). Note different scales among y-axes.

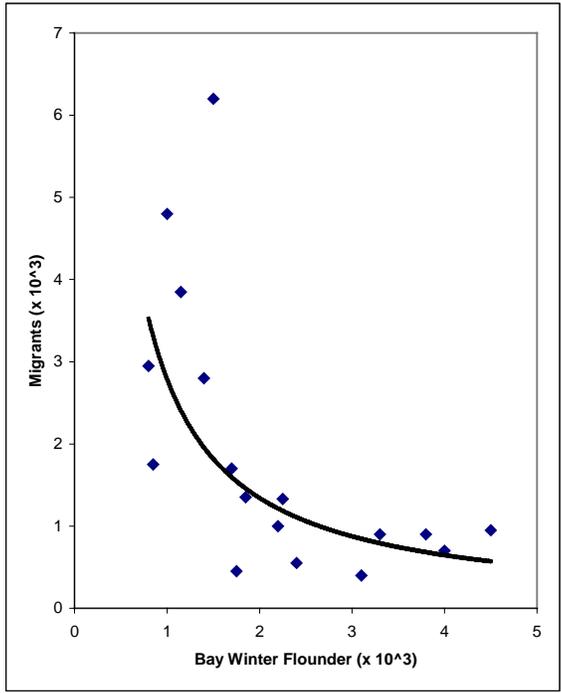


Figure 5.3. Power curve for relation of total annual abundance of Narragansett Bay winter flounder and total migrant species abundance. (After Figure 6 of Jeffries and Terceiro 1985.)

More recent assessment of RI and Narragansett Bay fish abundance confirms the trends and pattern, initially identified by Jeffries and Terceiro (1985), of declining numbers of demersal fish and increasing abundance of pelagic species. For example, RIDFW trawl data has shown that Narragansett Bay pelagic species abundance has increased ca. 3-fold, while demersal species abundance has decreased ca. 3-fold between 1980 and 1999 (DeAlteris et al. 2000). During this period, overall fish abundance (biomass) appears to have remained relatively constant, suggesting that Narragansett Bay fish carrying capacity has remained stable, but the partitioning of fish biomass into demersal and pelagic species has changed (DeAlteris et al. 2000). Mt. Hope Bay trawl data (from MRI surveys) are available to determine whether or not similar changes in the proportions of Mt. Hope Bay demersal and pelagic fishes have occurred over the same period.

Trends in Rhode Island landings and stock assessment of commercially important fishery species—namely winter flounder, summer flounder, striped bass, and longfin squid, as well as quahog, oyster and lobster—have recently been reviewed (DeAlteris et al. 2000). Rhode Island winter flounder catch data compiled by DeAlteris et al. (2000) has been presented above. The overall pattern of Rhode Islands fisheries presented by DeAlteris et al. (2000) is consistent with that reported by Jeffries and Terceiro (1985): an overall 1980-1999 decline in the abundance of demersal species and a simultaneous pattern of increase in pelagic species. For example, between 1980 and 1999 the

Narragansett Bay longfin squid index of abundance (based on URI/GSO and RIDEM trawls) has increased ca. 4- to 5-fold, and the Narragansett Bay striped bass index of abundance (based on MRI trawls and recreational CPUE) has increased ca. 4- to 5-fold, while the winter flounder abundance index (based on URI/GSO and RIDEM trawls) has declined ca. 6-fold (DeAlteris et al. 2000). Jeffries and Terceiro (1985) have indicated that the long-term shift from demersal to pelagic species may be the result of subtle temperature-related changes in Narragansett Bay. What roles overfishing and habitat degradation play in this shift is not well known. However, it is important to realize that this phenomenon is not restricted to Narragansett Bay, but rather has occurred on a broad geographic scale throughout the northwest Atlantic fisheries and is generally thought to be a direct result of overfishing (Sinclair and Murawski 1997). For example, over the same 30-year time period, U.S. groundfish stocks declined strongly while those of elasmobranchs and pelagic fishes increased dramatically (Figure 5.4).

Long-term (1972-1997) abundance indices of several Mt. Hope Bay fishes have been compiled from monthly standard trawl surveys (work of Marine Research, Inc., summarized in NEPC and ASA 1998). Eight species (windowpane, winter flounder, scup, weakfish, butterfish, bay anchovy, little skate, hogchoker) account for most (88% in 1997) of the standard trawl catch in the open waters of Mt. Hope Bay (see NEPC 1996 for chart of trawl locations). Trends in these fish abundance indices are summarized here. Window-pane (*Scophthalmus aquosus*) abundance was highly positively correlated with that of

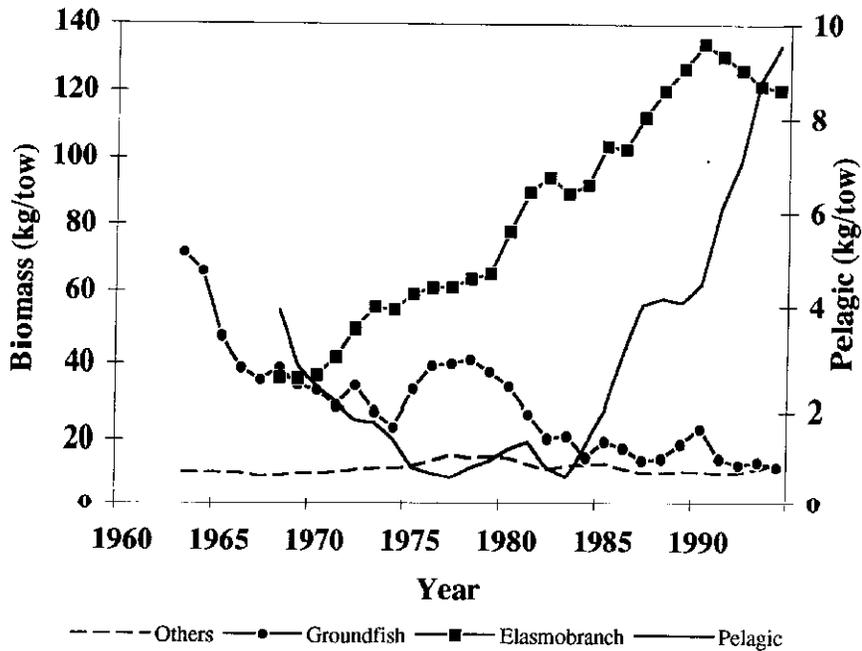


Figure 5.4. Shift in dominance of groundfish species to elasmobranch and pelagic fishes observed in the NW Atlantic. (Reprinted from Sinclair and Murawski 1997.)

winter flounder, and displayed a decline from near 11 fish per tow in 1979 to near 0.07 fish per tow in 1991 and 1992. Since the early 1990's, Mt. Hope Bay windowpane abundance, unlike that of winter flounder, has increased slightly to ca. 0.7 to 0.9 fish per tow in 1996 and 1997. Comparison of Mt. Hope Bay windowpane abundance indices to those of various sites in Connecticut (DEP-Niantic River), lower Narragansett Bay Rhode Island (URI/GSO trawl survey), and Massachusetts (Massachusetts Division of Marine Fisheries and Plymouth Nuclear Power Station) generally revealed similar patterns and trends in windowpane abundance, indicative of regional synchrony in this species' abundance patterns. Another flatfish, the hogchoker, has shown post-1985 declines in Mt. Hope Bay. The hogchoker is a small flatfish (usually <6 inches) for which there is no commercial or recreational fishery, so fishing mortality is

not likely to play a role in the abundance cycles of this fish. The Mt. Hope Bay hogchoker abundance index declined from near 1 to 2 fish per tow in 1972-1985 to a level of 0.1 fish per tow in 1986-1997 (NEPC and ASA 1998). In the late 1990's, modest gains in Mt. Hope Bay hogchoker abundance were observed, with abundance indices of 0.35 (in 1997) to 0.45 (in 1996) fish per tow. As with windowpane, patterns of Mt. Hope Bay hogchoker abundance were similar to those of lower Narragansett Bay (URI/GSO trawl survey) and Connecticut estuaries (CT DEP).

Mt. Hope Bay tautog abundance displayed a decline over 1972 to 1997, with a sharp decline post-1985 (see Figure F-14 of NEPC and ASA 1998). For the standardized MRI trawl, the tautog abundance index declined from 1 to 2 fish per trawl in 1986 to <0.1 fish per trawl in 1997. In 1997, for the first time in the 1972 to 1997 time series, no tautog were caught at the long-term fixed trawl location in Mt. Hope Bay. The ca. 10-fold decline in the Mt. Hope Bay standard trawl tautog catch represents a recent (post-1985) decline in this species that has also been observed in lower Narragansett Bay (URI/GSO trawl data) and in Connecticut estuaries (see NEPC and ASA 1998). However, Dorf and Powell (1997) recently reported that Mt. Hope Bay is an important spawning area for tautog and that macroalgae is an important nursery habitat for the species.

Scup are Narragansett Bay summer migrants, present in Mt. Hope Bay in May through October. Unlike winter flounder, windowpane, hogchoker and tautog, there have been no recent abrupt declines in Mt. Hope Bay scup abundance. The Mt. Hope Bay scup abundance index has remained near 5 to 10

fish per tow—with peaks of 20+ fish per tow in 1976 and 1980—from 1972 to ca. 1992, with a slight decline evident in the late 1990's (NEPC and ASA 1998). The recent decline in Mt. Hope Bay scup is not consistent with the inverse pattern of winter flounder and scup abundance seen in the 1970's to 1980's in Narragansett Bay (Jeffries and Terceiro 1985). Butterfish also showed an inverse pattern of abundance to that of winter flounder in lower Narragansett Bay (Jeffries and Terceiro 1985). This inverse pattern was not seen in Mt. Hope Bay, however, where butterfish abundance indices have shown no trend over the period from 1972 to 1997 (NEPC and ASA 1998). The abundance of little skate displayed a gradual increase in Mt. Hope Bay (NEP and ASA 1998). Although never abundant (usually <0.5 fish per tow, except ca. 1 fish per tow in 1992) in Mt. Hope Bay, little skate numbers there have risen significantly (sign test $p=0.046$, NEPC and ASA 1998) from an index near 0-0.1 fish per trawl in the 1970's, to 0.1-0.9 fish per trawl in the 1990's. This increase is consistent with observed increases in lower Narragansett Bay (URI/GSO trawls) little skate abundance (NEPC and ASA 1998).

Winter Flounder

Because of their former abundance and recent precipitous decline, winter flounder have been a focus of fishery research in Mt. Hope and Narragansett Bays. Historically, winter flounder (*Pseudopleuronectes americanus*) have been a dominant component of the Narragansett Bay fish community, comprising >50% of the fish in lower Narragansett Bay (Oviatt and Nixon 1973, Jeffries and

Johnson 1974). The winter flounder breeds in winter and early spring, spawning between January and May in southern New England (Bigelow and Schroeder 1953). In the post-yolk-sac stages of development, larval winter flounder first feed on diatoms (Bigelow and Schroeder 1953), and in later stages on a diet of crustaceans, especially isopods (Bigelow and Schroeder 1953) and benthic invertebrates (Jeffries and Johnson 1974). The first year of winter flounder life, during which a length of ca. 100 mm is attained, is complicated by the physiological and behavioral changes associated with the metamorphosis from symmetrical planktonic larvae to dorso-ventrally flattened benthic juveniles. These changes, especially in the first three months of development, have been described by Jearld et al. (1992).

There is a tendency for the fry to be concentrated in nearshore areas (bays, estuaries) and move offshore as they grow older (Bigelow and Schroeder 1953, Pearcy 1962). Winter flounder tend to migrate into estuarine embayments in winter (spawning season), followed by migration to deeper offshore (RI Sound) waters in summer (Nitschke et al. 2000). There is evidence that some RI winter flounder migrate easterly as they move offshore, with RI-tagged fish being recovered in Vineyard and Nantucket Sounds, but most tagged RI winter flounder have been recovered in Narragansett Bay and RI Sound (Powell 1989). Natal fidelity, or homing instinct, the return to the same spawning locations, has been reported for winter flounder by Bigelow and Schroeder (1953). In mark and recapture studies, up to 90% of fish tagged in bays and estuaries of New England and Long Island were later recaptured in the same bay or estuary (Perlmutter

1947, Bigelow and Schroeder 1953). Saila (1961), in a 2-year mark/recapture study, found that winter flounder from the south coast of Rhode Island dispersed as far as 54 miles from the original tag location in summer, but returned to nearshore bays and salt ponds in winter. The 20-fathom isobath has been suggested as a limit to the seaward distribution of winter flounder (Saila 1961). Winter flounder migrations of tens of miles, such as a Block Island to Georges Bank migration, have also been documented (Bigelow and Schroeder 1953). A recent tag-recapture study in the Mid-Atlantic Bight also found high fidelity of winter flounder to specific estuarine systems (Phelan 1992). A similar pattern of overall natal fidelity mixed with occasional migration has recently been documented, via genetic analysis, for another common coastal fish (weakfish, *Cynoscion regalis*) metapopulation (Thorrold et al. 2001). The life history of winter flounder has been reviewed by Grimes et al. (1989), while Pereira et al. (1999) have recently reviewed the habitat requirements of various winter flounder life stages.

Narragansett Bay winter flounder appear to be part of the southern New England/Mid-Atlantic population group that is one of three (north of Cape Cod, south of Cape Cod, Georges Bank) separately recognized populations groups (Nitschke et al. 2000). Dramatic declines in winter flounder abundance have been reported in all three areas. Rhode Island winter flounder catch followed a pattern of elevated abundance in the 1970's and early 1980's followed by a drastic decline beginning in ca. 1982 (DeAlteris et al. 2000). During this period, RI

winter flounder landings declined from a peak of 9.3 million pounds in 1981 to a low of 1.3 million pounds in 1998 (DeAlteris et al. 2000; see Figure 5.5).

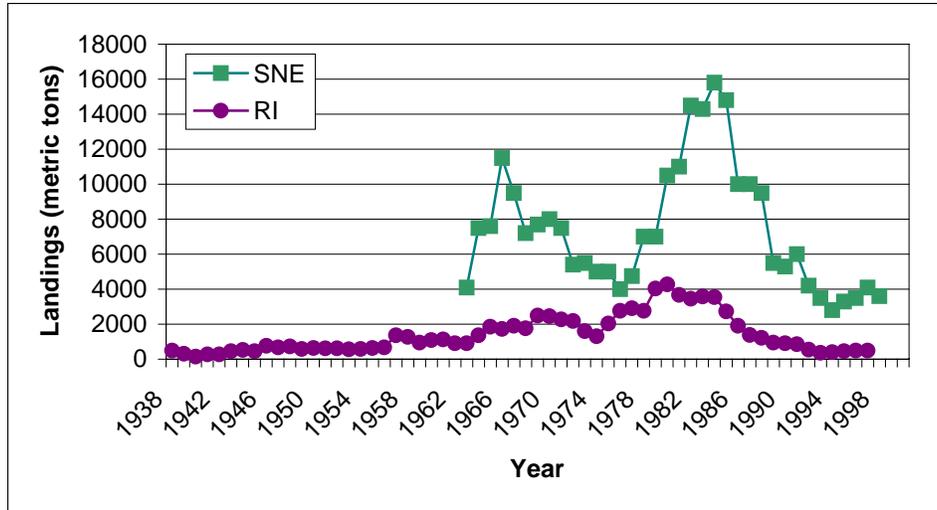


Figure 5.5. Rhode Island (RI) and southern New England/Mid-Atlantic (SNE) winter flounder landings time series. (Rhode Island data after DeAlteris et al. 2000; southern New England data after NMFS data summarized by Nitschke et al. 2000.)

Very little recovery in RI winter flounder abundance or catch is apparent in the data (to 1998) presented by DeAlteris et al. (2000). NMFS data indicate that Gulf of Maine total landings follow a similar pattern, having declined from ca. 3,500-5,000 metric tons per year in 1980-1982 to <100 metric tons per year in 1995-1998 (Nitschke et al. 2000; see Figure 5.6). Similarly, a pattern of decline was also seen in the Georges Bank population, with winter flounder commercial landings falling from sustained levels of 2,000-4,000 metric tons per year in the 1970's and early 1980's to levels of <2,000 metric tons per year in 1990-1998 (Nitschke et al. 2000; Figure 5.7).

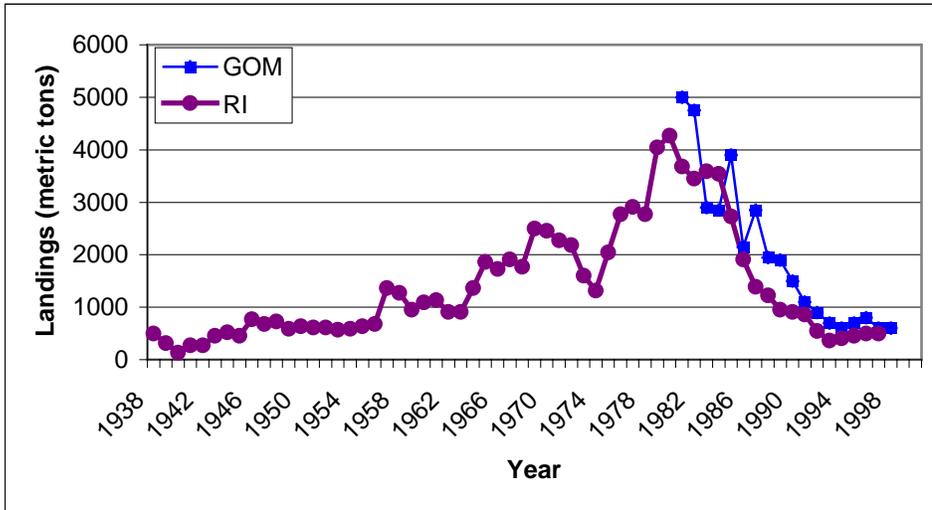


Figure 5.6. Rhode Island (RI) and Gulf of Maine (GOM) winter flounder landings time series. (Rhode Island landing data after DeAlteris et al. 2000; GOM landings after NMGS data summarized by Nitschke et al. 2000.)

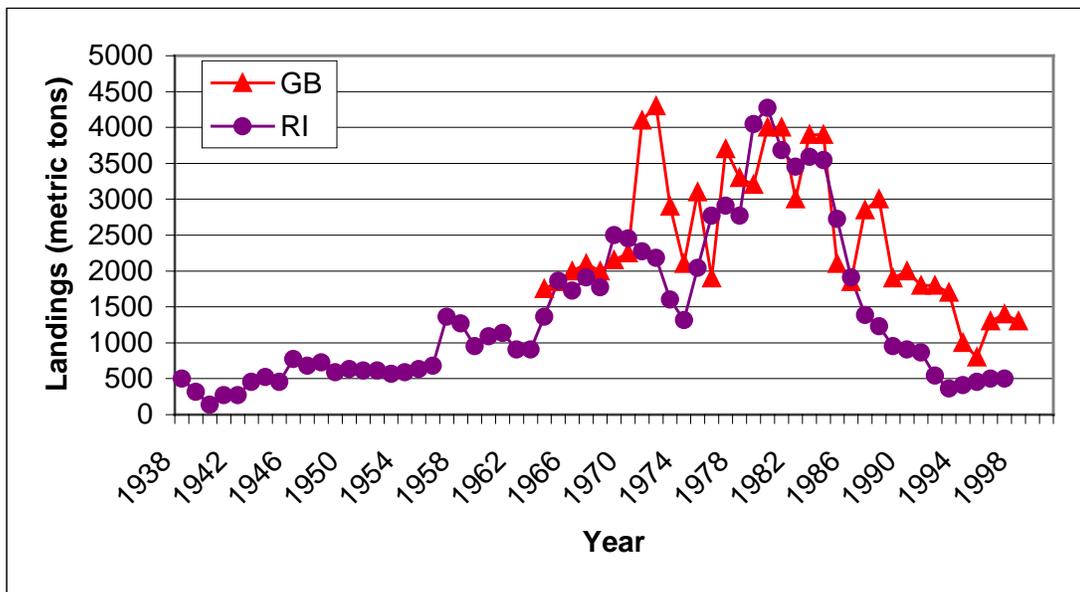


Figure 5.7. Rhode Island (RI) and Georges Bank (GB) winter flounder landings time series. Rhode Island data after DeAlteris et al. 2000; Georges Bank data after NMFS data summarized by Nitschke et al. 2000.)

For both Gulf of Maine and Georges Bank winter flounder, overfishing has been implicated as the cause for recent population declines (Nitschke et al. 2000). In the southern New England/Mid-Atlantic group, total landings declined from a peak of ca. 12,000 to 15,000 metric tons per year in 1980 to 1985 to levels of 5,000 metric tons per year or less in 1993 through 1998 (Nitschke et al. 2000; see Figure 5.8). In the southern New England/Mid-Atlantic group, fishing mortality

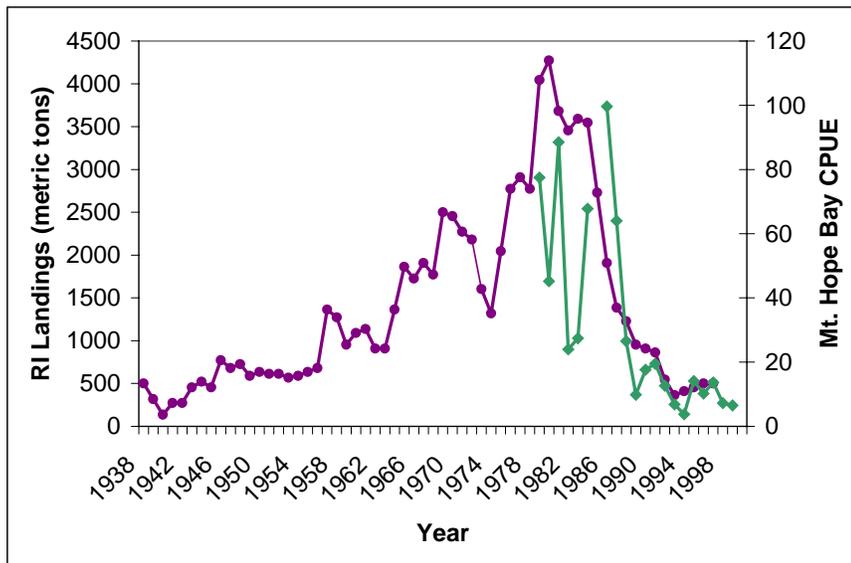


Figure 5.8. Rhode Island (RI) winter flounder landings (filled circles, after DeAlteris et al. 2000) and Mt. Hope Bay winter flounder catch per unit effort (CPUE; filled diamonds). (CPUE data is based on MRI trawl surveys as analyzed and presented by T. Englert at the October 2001 Mt. Hope Bay Workshop.)

was extremely high in 1981-1993, resulting in population declines (Nitschke et al. 2000). Since 1993, reductions in fishing mortality have led to rebuilding of winter flounder biomass, from a low estimate of 8,000 metric tons in 1992 to an estimated 22,300 metric tons in the southern New England group in 1998 (Nitschke et al. 2000; Figure 5.8). The recent (post-1981) decline in RI winter flounder landings, as well as the late 1970's to early 1980's increase, appears to be consistent with the population fluctuations experienced in neighboring (Gulf of

Maine, Georges Bank) population groups. A single pattern of flounder abundance appears to be applicable to all of southern New England; similar patterns of population fall and rise are observed in lower Narragansett Bay, Providence River (Manchester Street Electric Plant), Mt. Hope Bay (BPPS), and offshore populations (Jeffries et al. 1989). However, recent signs of stock rebuilding in winter flounder stocks, including in Rhode Island Sound, have not been mirrored in Narragansett and Mt. Hope Bays, where populations remain depressed (DeLong and Collier 2001).

From mid-1971 to the present, Mt. Hope Bay winter flounder abundance has been assessed via monthly to twice-monthly trawls carried out by MRI. The raw catch numbers have been compiled into an annual catch per unit effort (CPUE) index of Mt. Hope Bay winter flounder abundance for 1979 through 1999 (T. Englert, personal communication). This Mt. Hope Bay winter flounder CPUE index displayed a pattern of elevated CPUE in the early to mid-1980's followed by a precipitous decline in the late-1980's that is similar to the pattern of RI winter flounder landings during the same time period (Figure 5.8).

Narragansett Bay winter flounder population fluctuations have been linked to fluctuations in winter water temperature (Jeffries and Johnson 1974, Jeffries and Terceiro 1985, Keller and Klein-MacPhee 2000). Air temperatures along the Atlantic coast of the United States have displayed a recent increase (Ford 1996), and Narragansett Bay winter water temperatures have increased by 2-3°C between ca. 1960 and 1996 (Cook et al. 1998, Keller et al. 1999a, Keller and Klein-MacPhee 2000). Winter flounder abundance has been statistically linked to

variations in winter water temperature. However, these variations, on the order of $\pm 2\text{-}3^{\circ}\text{C}$, are well within the physiological thermal tolerance of winter flounder, suggesting that some secondary, rather than direct, temperature effects are the controlling mechanism (Jeffries and Johnson 1974, Jeffries and Terceiro 1985, Jeffries et al. 1989). Mortality at early life-stages (i.e., larvae through young-of-year) is likely the main source of variation in determination of fish year-class strength (Houde 1987). This mortality appears to be dependent on food availability at time of first feeding, oceanographic conditions (i.e., larval retention mechanisms, etc.) and predation (Cowan and Houde 1990). Predation on winter flounder larvae at metamorphosis, which would be expected to be elevated in warmer winters due to increased activity of benthic predators, has been suggested as a potential mechanism behind winter flounder declines in Narragansett Bay (Jeffries and Terceiro 1985).

This hypothesis has been tested in a series of controlled Narragansett Bay (MERL) mesocosm studies (Keller and Klein-MacPhee 2000). The MERL-simulated physical and biological marine environment is similar to that of lower Narragansett Bay (see Sullivan and McManus 1986 and references therein). In their study, Keller and Klein-MacPhee manipulated the temperature of the mesocosms to compare winter flounder development and survival rates in February and April of 1997 at cooler than mean (ca. 1.86°C mean temperature) and at warmer than mean (ca. 5.11°C) water temperatures. There was experimental replication, with three cool treatment tanks and three warm treatment tanks. Temperatures were not held fixed at these levels, but tracked the

vernal warming of Narragansett Bay, with the warm treatment being raised ca. 0.7°C warmer than the Bay and the cool treatment being cooled ca. 2.5°C cooler than the Bay. Cold tanks ranged from near 0°C in early February to near 2.5°C in early April, while warm tanks warmed from 3°C in early February to near 6°C in early April (see Figure 1 of Keller and Klein-MacPhee 2000).

Several significant differences were found between the two experimental treatments manipulated by Keller and Klein-MacPhee (2000). Phytoplankton biomass was significantly higher in warm tanks, although diatoms (especially *Skeletonema costatum* and *Chaetoceros* spp.) were more abundant in the cold tanks. Abundance of copepod nauplii, copepodites and adults (mainly *Acartia hudsonica*) was higher in the warm treatment. Flounder hatching success was significantly higher in cold (91 to 94 %) than warm (65 to 82 %) treatments, and hatching-size larvae in cold tanks were significantly larger than those in warm tanks. However, growth rates of those larvae that did hatch were higher (0.016 to 0.018 day⁻¹) in warm versus cold (0.011 to 0.013 day⁻¹) treatments. Thus, there was greater hatching success and larger first-hatch larvae in the cold treatment, but greater rates of larval growth in the warm tank. Greatest larval survival was found in the cold treatment with lowest food availability, suggesting that winter flounder larval survival is dependent on larval mortality (predation) rather than food availability (Keller and Klein-MacPhee 2000). The sand shrimp *Crangon septemspinosa* was present in both treatments, but more active in the warm treatment (in which *Crangon* was seen actively swimming and feeding in the benthos) than in the cold treatment (in which *Crangon* was inactive and buried in

the sediment). There was a highly significant positive correlation between benthic predator activity and larval flounder mortality (see Figure 10 of Keller and Klein-MacPhee 2000), leading Keller and Klein-MacPhee to suggest that warm winter benthic predation may be a primary mechanism behind recent winter flounder year class failure and the decline in Narragansett Bay winter flounder.

Several lines of evidence support Keller and Klein-MacPhee's conclusion. Winter flounder larvae that developed in cool temperatures (2°C) had higher RNA and protein content, indicative of good health, than did those raised in warm (7°C) temperatures (Buckley et al. 1990). Water temperature also significantly affects winter flounder age at metamorphosis and size at metamorphosis, with fish raised at warmer temperatures (10°C) metamorphosing ca. 30 days earlier and being ca. 15% longer than those reared at cool temperature (5°C; Chambers and Legett 1992). The relation between winter flounder juvenile body size and predation has been examined by Whitting (1993). Predation of recently settled larvae of a similar demersal fish (plaice, *Pleuronectes platessa*) by *Crangon* has also been documented in the Wadden Sea (Van der Veer and Bergman 1987). *Crangon* can be much more abundant in Narragansett Bay (to 80 animals m⁻²; Thornton-Whitehouse 1994) than the low abundance (ca. 5 m⁻²) used in Keller and Klein-MacPhee's experiment, suggesting that the impact of *Crangon*-related winter flounder mortality in Narragansett Bay may be even greater than that observed in the above-described mesocosm experiments.

At all life stages, winter flounder are a prey species for many common Mt. Hope Bay fish and invertebrates. Larval winter flounder are preyed upon by

small medusae of *Sarsia tubulosa* (Pearcy 1962). The mud anemone (*Ceriantheopsis americana*) has been suggested as a predator on winter flounder larvae in Narragansett Bay (Klein-MacPhee et al. 1993). Summer flounder were found to prey heavily on juvenile winter flounder in a New Jersey estuary, and in fact preferred winter flounder over alternative prey (Manderson et al. 1999). The importance of habitat type was underscored by the finding that predation was lower in eelgrass than in macroalgae. The striped searobin was also found to be a significant predator of juvenile winter flounder (15-70 mm TL) in New Jersey estuaries (Manderson et al. 2000). Summer flounder and searobin are both common in Mt. Hope Bay, with the former accounting for 6% of the catch during 2000 (USGen 2001). Bluefish, *Pomatomus saltatrix*, common in Mt. Hope Bay in summer, prey upon juvenile winter flounder according to Howe et al. (1976), who also suggest that birds (especially gulls and cormorants) are an important source of juvenile winter flounder mortality. The seven-spine shrimp (*Crangon septemspinosa*, mentioned above in conjunction with the work of Keller and Klein-MacPhee 2000) was found to prey on young-of-the-year winter flounder of 10 to 20 mm length (Witting and Able 1995). A compilation of known fish predators of juvenile winter flounder, compiled by Pereira et al. (1999), included summer flounder (*Paralichthys dentatus*), windowpane (*Scophthalmus aquosa*), and sea robin (*Prionotus evolans*). Changing relative abundance of different fishes (i.e., increases in searobin abundance; Manderson et al. 1999) may lead to increased predation of young winter flounder. According to a summary by Pereira et al. (1999), adult winter flounder are known to be prey items for striped

bass (*Morone saxatilis*), spiny dogfish (*Squalus acanthus*), goosefish (*Lophius americanus*), oyster toadfish (*Opsanus tau*), and sea raven (*Hemitripterus americanus*). Like juveniles, adult winter flounder are preyed upon by birds, including cormorants, blue herons and osprey (Pearcy 1962). Harbor seals (*Phoca vitulina concolor*) are also known to prey upon adult winter flounder, and Harbor seals are often observed in Mt. Hope Bay in winter (Post 1998). However, flounder (as a group of five species) made up less than 10% of the harbor seal diet in a study by Payne and Selzer (1989).

The decline in Narragansett Bay and Mt. Hope Bay winter flounder abundance has recently been examined by Collie and DeLong (2001), who analyzed time series of winter flounder abundance at several locations in Narragansett and Mt. Hope Bays to identify potential mechanisms for winter flounder decline. Previous work (Keller and Klein-MacPhee 2000) had identified the potential for climate-related increases in predation and mortality at the egg-larvae stages of winter flounder in Narragansett Bay. In their analyses of 1973-1999 MRI Mt. Hope Bay winter flounder data and 1979-1999 RIDEM data, Collie and DeLong (2001) identified two "bottlenecks," or periods of increased mortality, that are related to total winter flounder mortality: the egg-larvae period and the age-1 fall to age-2 spring period. The correlation coefficient of age-1 fall to age-2 spring mortality with total mortality ($r = 0.80$) was much larger than that of egg-larvae period ($r = 0.37$), or any other period examined, indicating that the age-1 fall to age-2 spring period is the period to which much of the decline in Mt. Hope Bay winter flounder abundance can be attributed (Collie and DeLong 2001).

Further, mortality rate at various stages (age-1 spring to age-1 fall, juveniles June to October) displayed significant positive correlation with bottom temperature, suggesting temperature-dependent mortality of Narragansett Bay winter flounder (see Collie and DeLong 2001, DeLong et al. 2001). Temperature-dependent juvenile winter flounder mortality may be either a direct physiological response or an indirect response to other temperature-related changes such as availability of prey (DeLong et al. 2001). Juvenile winter flounder prefer warmer water than do adults (Casterlin and Reynolds 1982) and are known to feed in shallow, nearshore areas such as tidal creeks (Rountree and Able 1992a,b, 1993) and intertidal mud flats (Tyler 1971, Wells et al. 1973). The distribution of preferred juvenile winter flounder habitat (shallow waters) in Mt. Hope Bay is shown in Figure 5.9. Note that this area is also the most impacted by BPPS heated water effluent (see Chapter 2 herein). Such shallow regions are also expected to be more susceptible to the habitat-degrading effects of such factors as warming, development, and eutrophication than are the deeper regions of Mt. Hope Bay. Loss of suitable habitat in these shallow regions used by age-1 to age-2 winter flounder may be a mechanism contributing to elevated mortality at this developmental period and the overall decline in winter flounder in Narragansett and Mt. Hope Bays.

Other findings of Collie and DeLong's (2001) analyses include a recent (post-1988) trend of declining adult female winter flounder in Narragansett Bay relative to their abundance in Rhode Island Sound (although this analysis is based on a single station in each area; see Figure 5.1), suggesting a relative depletion of Narragansett Bay winter flounder spawning stock. Spatial-temporal analyses of

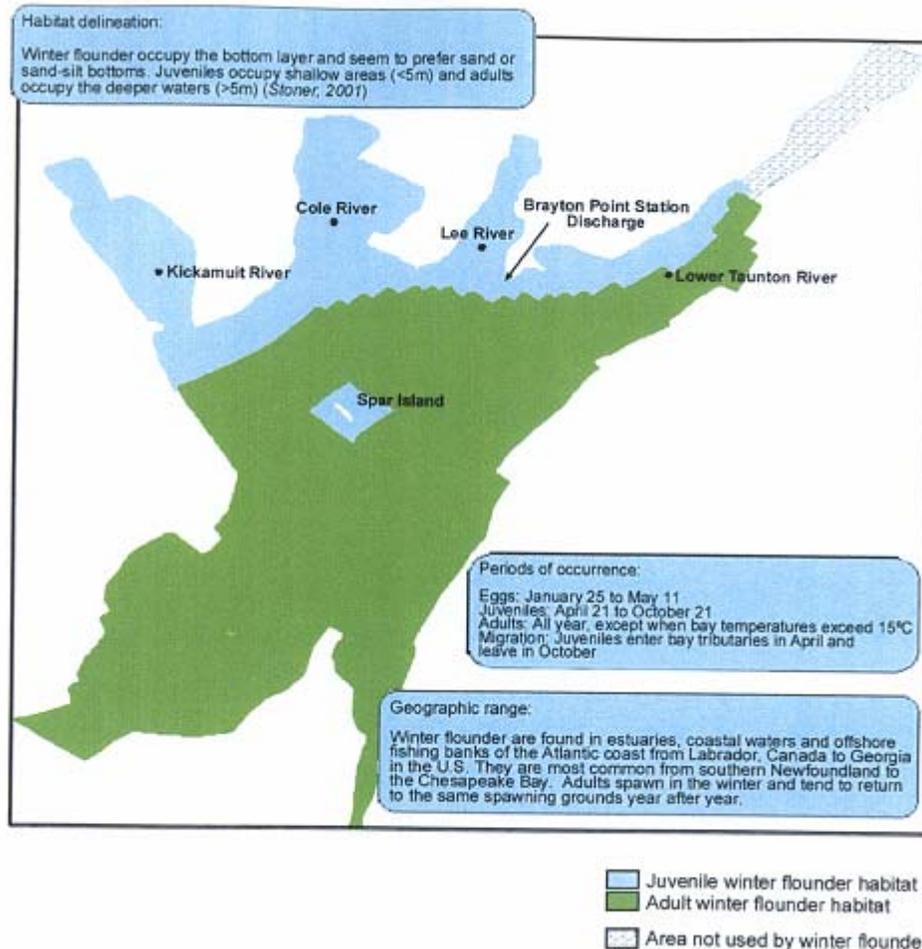


Figure 5.9. Habitat delineation of juvenile winter flounder habitat as determined by the BBPP (USGen 2001).

available winter flounder data (Collie and DeLong 2001) indicate that the habitat suitability of Mt. Hope Bay ("sector 3" of their analyses) has increased for large and small winter flounder during the spring, but that fall habitat suitability has decreased.

Water temperature is a critical component of winter flounder habitat (Percy 1962, Sogard 1992, Pereira et al. 1999, Stoner et al. 2001), and autumn has been identified as the season in which Mt. Hope Bay water temperature is most affected by anthropogenic thermal modification (Carney 1997, Mustard et al.

1999). While anthropogenic temperature modification of Mt. Hope Bay (on the order of 0.8°C; Mustard et al. 1999) is within the physiological range of winter flounder (see Table 5.1), subtle temperature-dependent changes in habitat (e.g., changes in food availability), behavior, etc., may be influencing winter flounder mortality at key developmental stages such as the age-1 fall to age-2 spring period identified by Collie and DeLong (2001). Sediment type has also been identified as of critical importance to juvenile winter flounder (Phelan et al. 2001). Small juveniles prefer fine grain sediments, while larger juveniles prefer larger grain sediments due to size-related changes to burrowing ability. However, prey availability was found to override sediment type preference under experimental conditions.

Winter Flounder and Habitat Change

Watershed development and related nutrient loading often result in degraded estuarine water quality and degraded fish habitat, with an ultimate expression as reduced fish stocks (Polgar et al. 1985, Valiela et al. 1992). Assessments of Rhode Island coastal water habitat quality on winter flounder growth rate have indicated that dissolved oxygen concentration and benthic prey abundance and type are important determinants of winter flounder distribution (Meng and Powell 1999, Meng et al. 2000, Meng et al. 2001). Similar results were observed in Mid-Atlantic Bight estuaries (Bejda et al. 1992). Additionally, recent warm Narragansett Bay winter water temperatures may represent a degradation of the winter flounder's ideal spawning habitat (Jeffries 1994, Keller

Table 5.1. Adult winter flounder thermal tolerance data.

| Exposure Temp(C) | Acclimation Temp(C) | Exposure Time | Response | Source |
|-------------------------|----------------------------|----------------------|--------------------------|---------------------------|
| <u>Heat exposure:</u> | | | | |
| 19 | 4 - | - | upper lethal T | McCracken 1963 |
| 12-15 | - | - | preferred temperature | McCracken 1963 |
| 13.5 | - - | - | preferred temperature | Reynolds 1977 |
| 18.5 | - - | - | preferred T (yearlings) | Casterlin & Reynolds 1982 |
| 22 | 5 | 4-64m | 'low' mortality | Itzckowitz & Schubel 1983 |
| 22 | 7 - | - | critical thermal maximum | Duffy and Luders 1978 |
| 22.5 | 17-23 | - | burial and inactivity | Olla et al. 1969 |
| 23.7 | 14 | - | critical thermal maximum | Duffy & Luders 1978 |
| 24.5 | 14 | - | avoidance behavior | Duffy & Luders 1978 |
| 24.5 | 22 | - | avoidance behavior | Duffy & Luders 1978 |
| 24.9 | 20 | - | avoidance behavior | Duffy & Luders 1978 |
| 26-32 | 4-23 | - | critical thermal maximum | Everich & Gonzalez 1977 |
| 26.5 | 20 | - | upper lethal temperature | McCracken 1963 |
| 27 | 21 | - | critical thermal maximum | Duffy & Luders 1978 |
| 27.5 | 24 | - | avoidance behavior | Duffy & Luders 1978 |
| 28 | 22-28 | 48h | upper lethal temperature | Hoff & Westman 1966 |
| 28-30 | 5 | 4m | 100% mortality | Itzckowitz & Schubel 1983 |
| 29.1 | 28 | - | critical thermal | Duffy & Luders 1978 |
| 30.6-31.4 | 20 | 1h | upper lethal temperature | LI Light Co. 1977 |
| <u>Cold exposure:</u> | | | | |
| 1.0 | 21 | 48h | lower lethal temperature | Hoff & Westman 1966 |
| 5.4 | 28 | 48h | lower lethal temperature | Hoff & Westman 1966 |

and Klein-MacPhee 2000). First settling winter flounder (ca. 25 mm long or less) prefer the fine-grained, high-organic-content sediments typical of depositional areas where there is increased availability of suitable prey items (calanoid and harpacticoid copepods and small spionid polychaetes; see Meng and Powell

1999, Phelen et al. 2001, Stoner et al. 2001). As the juvenile fish grow to ca. 25 to 55 mm length, their diet still includes these same prey items, but the dietary proportion of larger polychaetes increases. In upper Narragansett Bay, the polychaete *Polydora cornuta* seems to be an especially important food for ca. 30mm winter flounder: when present, *P. cornuta* comprised a mean of 70% (by volume) of young winter flounder stomach contents (Meng et al. 2001). Delta N-15 analyses of Narragansett Bay young winter flounder (ca. 30 mm length) show that they feed 1-2 trophic levels above primary producers, suggesting a macroalgae-harpacticoid copepod-polychaete food chain (Meng and Powell 1999). With attainment of ca. 55 mm length, the winter flounder diet shifts from copepods to larger crustaceans like amphipods and mysids and includes larger prey items such as large polychaetes (Stoner et al. 2001). The trend towards the proportion of polychaetes in the diet increasing with winter flounder age continues until adulthood; adult winter flounder have been placed in a benthivore guild (Garrison and Link 2000), with a diet composed of 43% (by volume) polychaetes, followed in volume by amphipods and isopods. Similarly, in a review of Cape Hatteras to Nova Scotia winter flounder stomach contents, Langton and Bowman (1981) found an adult winter flounder diet dominated by polychaetes. Changes in abundance or species composition of the Mt. Hope Bay benthic community, which in turn may affect winter flounder feeding, have not been noted between the mid 1970's and 1997/1998 at four open-bay monitoring locations (MRI 1999).

Winter flounder prefer relatively open sand/silt barren bottom, but may prefer such areas that are adjacent to eelgrass (*Zostera marina*) beds that provide a refuge from predators (Baltz et al. 1993, Meng and Powell 1999). Declines in the extent of coverage of Narragansett Bay eelgrass beds are well documented. Eelgrass beds, which were historically present throughout Narragansett Bay, including the upper bay in Greenwich Cove and the mouth of the Palmer River, are now only found in the lower bay, i.e., south of Jamestown (RIDEM 2000; compare Figures 3.1 and 3.2 herein). Recent reports indicate that there are no eelgrass beds remaining in Mt. Hope Bay (Rines 2001 cited in USGen 2001). Loss of this potentially important micro-habitat may be a factor in recent winter flounder declines. Additionally, profusion of macroalgal mats (especially *Ulva lactuca*), enhanced by N-loading in the upper reaches of Narragansett Bay, may represent a significant degradation in winter flounder habitat. In the Raritan/Navesink estuary (northern New Jersey), *Ulva* cover of up to ca. 20 g wet wt. m⁻² increased the probability of finding 55 mm or greater winter flounder, while the fish avoided *Ulva* densities of >20 g wet wt. m⁻² (Stoner et al. 2001). The heavy *Ulva* cover observed by Stoner et al. (2001) may degrade winter flounder habitat by impeding motility, limiting access to food items in/on the sediment and, when the *Ulva* mat dies and decomposes, leading to reduced dissolved oxygen concentration. These field findings are consistent with the experimental work of Timmons (1995), who found that young winter flounder avoided high concentrations of *Ulva*.

The feeding strategies of winter flounder—i.e., association with shallow, typically nutrient-enriched and depositional areas of estuaries—place them in jeopardy when estuaries become over-enriched (eutrophic). The same features of the estuarine habitat that provide an organically enriched, depositional sediment where deposit-feeding polychaetes thrive, also make that habitat susceptible, when subject to excessive nutrient loading, to low dissolved oxygen concentration and heavy macroalgal cover. For example, in a comparison of winter flounder growth rate in Connecticut, Miese et al. (1999) found that winter flounder growth at the relatively polluted New Haven Harbor was greater than that at the relatively clean Connecticut River estuary. Similarly, Meng et al. (2001) found that winter flounder growth at three sites along the estuarine gradient in Narragansett Bay—from near Providence (estuarine) south to Jamestown (coastal water)—was highest at the mid-bay site. Growth rate was initially greatest at the estuarine site (Gaspee Point, near Providence) but declined in response to decreasing dissolved oxygen (as low as 2.8 mg L^{-1}) concentration, which eventually reduced benthic prey availability. At the lower bay site (Sheffield Cove, Jamestown, RI), oxygen concentration was not growth-inhibiting at 19 mg L^{-1} (winter flounder growth declines at ca. $2.5 \text{ mg O}_2 \text{ L}^{-1}$; Bejeda 1992), but food availability may have reduced growth rate. The mid-bay site (Prudence Island, RI), with intermediate dissolved oxygen concentration (ca. $18 \text{ mg O}_2 \text{ L}^{-1}$) and greatest prey availability, sustained highest winter flounder growth rates (Meng et al. 2001). Areas of upper Narragansett Bay, such as near Gaspee Point, previously identified as primary juvenile winter flounder habitat areas (Meng and Powell 1999), may now be

suffering habitat degradation through the dissolved-oxygen-decreasing effects of eutrophication (Granger et al. 2000). Mt. Hope Bay is thought to be a secondary juvenile winter flounder habitat to Narragansett Bay (Meng and Powell 1999). If eutrophication-induced declines in Mt. Hope Bay bottom dissolved oxygen concentration are reaching the growth-limiting levels (ca. 2 to 3 mg O₂ L⁻¹) observed at Gaspee Pt. (Meng et al. 2001), they represent a significant degradation of winter flounder habitat. Diel pattern in dissolved oxygen concentration observed in Mt. Hope Bay (Howes, unpublished data) may also represent habitat degradation, as winter flounder have been shown to grow more slowly when exposed to diel fluctuation in DO (Bejda et al. 1992).

Summary

It is clear that much is known about the population dynamics and life history of winter flounder in general, and specifically about Mt. Hope Bay populations. However, important gaps in our knowledge are also evident. Important phases in the life history of winter flounder, illustrated in Figure 5.10, can be used to summarize data gaps for the species. Numbers in the following sections refer to critical phases of the movements and behavior of adult winter flounder as illustrated in Figure 5.10a.

1) Migration into the estuary/estuarine fidelity: Once of the most crucial questions is how Mt. Hope Bay contributes to the regional stocks. We've seen that population trends for Mt. Hope Bay mirror those for Narragansett Bay, Rhode Island Sound, and in fact, for the entire NW Atlantic to a large degree. In

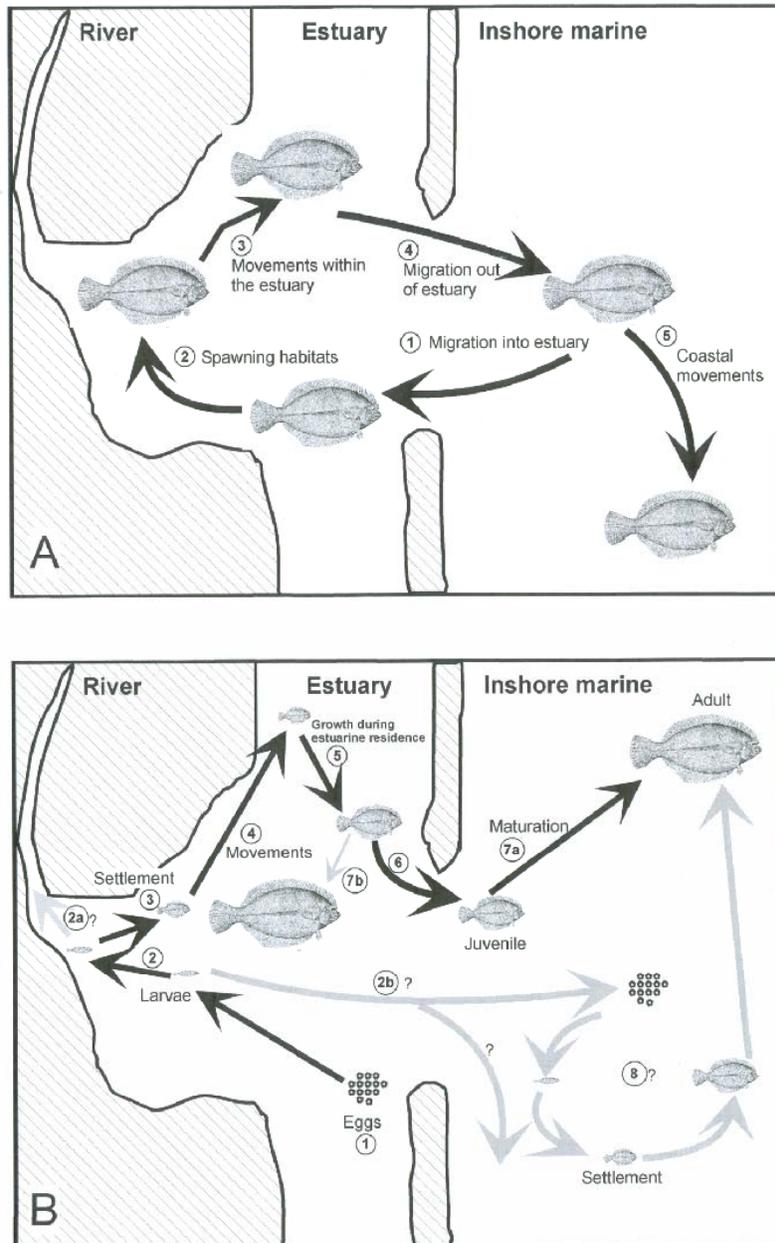


Figure 5.10. Schematic of the life history pattern of winter flounder. Movement patterns of adults are shown separately at the top for clarity. (See text for explanation.)

order to tease apart local from regional sources of temporal trends, we need to determine to what extent adults return to the same estuary each year. As we have reviewed, there is mounting evidence that winter flounder exhibit a high degree of

fidelity to particular estuaries. What percent of surviving adults return to the estuary? Just as importantly, we need to define what constitutes the “home” estuary. Is it Mt. Hope Bay or the greater Narragansett Bay—i.e., what is the spatial scale on which homing operates? These are critical questions because they in effect define population boundaries for the modeling efforts. In addition, it would be important to quantify sources of adult mortality for Mt. Hope Bay relative to Narragansett Bay, the Sakonnet River and Rhode Island Sound. What is the fishing mortality on MHB winter flounder while they summer in Rhode Island Sound? Another important issue is the mechanism(s) regulating the timing of immigration into the estuary from coastal waters. Although we have good data on the general seasonal migration pattern, identification of environmental triggers for migration would greatly enhance modeling efforts and allow us to understand annual variations.

2) Spawning habitats: Identification of specific spawning areas/habitats is currently uncertain, but there is evidence for spatially distinct sites in Rhode Island estuaries (Crawford and Carey 1985). The location of distinct spawning sites in MHB would be of great importance to modeling the impact of various natural and anthropogenic factors on spawning success. Protection of the habitat quality of these sites would of course be of special concern. Although it is known that spawning occurs at night (Stoner et al. 1999), a more precise determination of the timing of spawning events would enhance efforts to model egg dispersal though the bay. Is spawning synchronized with tidal and/or lunar cycles?

3) Estuarine movements: Quantification of the movement patterns of winter flounder within MHB over daily, weekly and seasonal time scales would also be useful. Limited information from conventional tagging studies suggests relatively little movement, but telemetry studies would be required to elucidate finer-scale movement patterns. Of particular importance is quantification of movements of adults among sections of greater Narragansett Bay, including passage into Mt. Hope Bay via the East Passage and Sakonnet River. Since individual winter flounder spawn repeatedly over a two-month period (Stoner et al. 1999), it is possible that individuals may spawn in many different locations throughout the Mt. Hope Bay. The relative success of eggs spawned in various locations would be an important model input (see below).

4) Migration out of the estuary: A better understanding of factors affecting emigration is also important. Estuarine mortality can be estimated by comparing emigration and immigration numbers, assuming the percentage of adults that remain in the estuary is known. And again, identification of the environmental trigger(s) for emigration would be useful to the modeling efforts. The condition of emigrating individuals might also be used to provide an indication of survival likelihood. The geography of Mt. Hope Bay (the East Passage is less than 800 m wide and the Sakonnet River passage less than 100 m wide) makes it feasible to conduct telemetry studies to quantify these parameters.

5) Coastal movements: The movements of winter flounder within coastal waters are generally known, but studies that identify the natal source of winter flounder fish stocks taken in different coastal areas would certainly be valuable.

Conventional tagging and otolith chemical composition methods—e.g., as has been done for weakfish (Thorrold et al. 2001)—could provide that information. This would provide quantification of fishing mortality specific to Mt. Hope Bay populations as well as other estuarine populations.

Numbers in the following sections refer to critical phases in the early life history of winter flounder as illustrated in Figure 5.10b.

1) Egg dispersal and hatching success: Winter flounder eggs are demersal and stick together and are most common in shallow shoal waters of Narragansett and Mt. Hope Bays (Bourne and Govoni 1988). This suggests limited dispersal; however, since it takes about 2-3 weeks for hatching, significant dispersal from spawning sites is possible. The degree of dispersal from the spawning site is unknown and obviously a function of hydrography and temperature. The impact of dispersal on hatching success is also unknown. Are spawning sites chosen to enhance egg-hatching success? As discussed under the adult spawning phase, identification of optimal spawning habitats is an important factor for consideration in the MHBNL program. Identification of the environmental characteristics of spawning sites is particularly important. The fate of eggs once spawned is likely to be an important modeling focus. The duration of the incubation period is highly variable and dependent on temperature, occurring in as few as 11 and many as 63 days (see review in Able and Fahay 1998). Further quantification of the effect of temperature and duration of incubation on hatching success would be helpful. An examination of the relationship between spawning and riverine discharge may also be of interest, as

peak egg densities in Mt. Hope Bay (Figure 4.6) occur at the onset of the period of peak river discharge in March (Figure 3.19).

2. Larval dispersal: Dispersal of larval winter flounder is highly amenable to modeling efforts and is strongly influenced by tidal, gravitational, and wind-driven circulation patterns (Bourne and Govoni 1988). Because duration of the larval stage is highly physiologically plastic in winter flounder (Chambers and Leggett 1987), larvae can remain in the plankton for extended periods if suitable settlement habitats are not encountered. The timing of peak abundance of winter flounder is known to be highly variable between years (Able and Fahay 1998) and is likely related to annual variations in water temperature. Matching of winter flounder larval peak abundance and planktonic food resources is an important unknown in Mt. Hope Bay. If spawning is delayed and larval abundances are mismatched with plankton food resources, greater mortality can be expected; hence efforts to incorporate plankton population dynamics into winter flounder models are clearly warranted. The influence of water quality parameters (temperature, salinity, turbidity, light, pollutants, etc.) on larval behavior is poorly understood and can have a strong impact on larval retention and dispersal patterns. Larval winter flounder may use tidal stream transport mechanisms to maintain position in the estuary, and/or to move into favorable upper estuarine and shallow tidal habitats. In addition, environmental parameters can have a strong effect on larval survival by affecting their energetics. Physiological responses to even small changes in temperature and salinity can have an important energetic cost to larval fishes, and given that mortality of larval

fishes is thought to be strongly a function of starvation, any increased demand on energy reserves is of concern. Loss of Mt. Hope Bay larval populations to upstream dispersal (3a) and downstream transport out of the estuary (3b) are important input and output parameters for modeling efforts. Downstream transport of winter flounder larvae is possibly significant, as loss rates of 3% per day have been estimated for other estuaries (Pearcy 1962).

3) Settlement: Metamorphosis and settlement to the bottom is an important bottleneck in the life history of winter flounder and other flatfishes (Able and Fahay 1998, Pereira et al. 1999). During this stage winter flounder must adapt to a new suite of predators and prey. Predation by benthic crustaceans such as *Crangon septemspinosa* is an important source of mortality (Witting 1995, Witting and Able 1995). Identification of settlement sites and quantification of settlement site suitability should be high priorities. Beam trawl sampling is preferable to seine and trawl sampling for this type of study. The timing of settlement is also important, as it affects predator and prey availability. As mentioned above, winter flounder appear to be able to delay settlement for considerable periods if suitable habitats are not found.

4) Post-settlement movements: Winter flounder are thought to settle in limited areas and then disperse throughout the estuary. Growing evidence suggests that early settlement stages prefer very shallow tidal waters and develop tidal migration behaviors as they grow. We know little about the movements and habitat use patterns of early post-settlement winter flounder in Mt. Hope Bay. Movement patterns of older juveniles have been examined in some detail over

short time scales; however, seasonal movements are less well studied.

Environmental triggers for these movements are currently unknown.

5) Estuarine growth: Estuarine growth of juvenile winter flounder has been relatively well studied (see reviews in Able and Fahay 1998 and Pereira et al. 1999). However, quantification of growth and mortality rates of juveniles specific to Mt. Hope Bay habitats would be helpful. In particular, the effect of nutrient enrichment on growth is unknown. The impact of predation by summer flounder, searobins, bluefish and other piscivores during this phase require closer examination.

6) Estuarine emigration: Juvenile winter flounder do not always remain resident within the estuaries, but in fact a large component can migrate into coastal marine waters by January of the second winter (age-1; Able and Fahay 1998). Because there is evidence (Collie and DeLong 2001) of an increase in the mortality of winter flounder in Mt. Hope Bay at this age, and in greater Narragansett Bay at age-1 to age-2, a closer look at juvenile movements is warranted. Quantification of the percent of resident versus migrating individuals at each life stage is needed. A comparison of growth and mortality rates for coastal and estuarine juveniles may provide an important indication of sources of population changes for the species. Again, as with other migratory stages, identification of important environmental and biotic triggers for migration is also needed.

7) Coastal residence/growth: Growth and movements of juveniles within the coastal waters should be compared to those in estuarine waters.

Habitat use patterns are unknown. Is there a difference in mortality and contribution to adult stocks from juveniles that migrate from the estuary at different times versus those that remain resident in the estuary until the onset of maturity?

Chapter 6. Modeling Efforts (Miles A. Sundermeyer & Yalin Fan)

1. Introduction

A number of hydrodynamic, water quality, and fish modeling studies of Mt. Hope Bay have been conducted to assess the impact of the Brayton Point Power Station (BPPS) thermal plume on local fish populations. To date, much of the hydrodynamic modeling has been performed by Applied Science Associates (ASA), using a three-dimensional boundary-fitted hydrodynamic model, WQMAP (Water Quality Mapping and Analysis Program). In addition, a second embedded hydrodynamic model, CORMIX (the Cornell Mixing Zone Expert System) has been used to simulate the near-field dynamics of the BPPS thermal plume. Results from the above hydrodynamics models have been used to force box models of water quality and dissolved oxygen as well as a fish metapopulation model, the RAMAS GIS/Metapop model. The ultimate purpose of these modeling studies has been to address the following questions:

1) What is the overall impact of BBPS on the hydrodynamics and property distributions (temperature, salinity, DO, BOD, and optionally carbonaceous and nitrogenous biochemical oxygen demand, ammonia, nitrate, and organic nitrogen) of Mt. Hope Bay?

Simulations have been run for pre- and post BPPS conditions, i.e., with and without the thermal plume present. The difference between solutions for these two cases represents the impact of the power station. According to available

reports, to date this analysis has been done for temperature, salinity, and horizontal velocity.

2) *What is the rate of impingement of fish eggs and larvae (i.e., how many are drawn into the cooling intakes) as a function of flow rate and position in the estuary?*

This analysis was suggested by N. Fennessey (memo to W.L. Bridges, dated June 11, 1997). It was suggested that eggs and larvae could be represented by Lagrangian particles in the model. Using an initially uniform distribution of particles in the model, and tracking particle positions for a many tidal cycles, entrainment rates into the BPPS intake could be assessed as a function of location in the estuary. An additional question raised earlier by N. Fennessey (memo to G. Szal, Feb 7, 1997) was whether discharge of dead, entrained larvae could cause a local rise in BOD near the outfall? Finally, a January 1998 memo to EPA (memo from A.H. Aitken, January, 1998) cited evidence that the assumption of 100% larval mortality in water circulating through the power station was likely an overestimate; a 1997 study indicated that survivability of larvae may be 50% or more.

3) *How does the extent of the plume (in terms of area and volume encompassed) vary by season and as a function of cooling water discharge rate and temperature?*

The impact of the plume is assessed in terms of the percent area and volume of Mt. Hope Bay that experiences (a) a rise in temperature greater than or equal to 0.8°C, or (b) an absolute temperature greater than or equal to some critical temperature. The latter critical temperature is species-specific, and has been a subject of ongoing debate (see, e.g., Meetings Notes of TAC Fisheries Subcommittee, November 5, 1997). In a memo to J. Parr from July 1998, it was further suggested that a “mixing zone” be defined in terms of whether the area influenced by the plume interferes with normal migration of fish, i.e., whether it impairs the passage and free movement of migratory species. Related issues are the temporal rate of change in temperature, and the extent to which fish can detect gradients.

4) *What would be the largest possible size/extent that the thermal plume might attain in the “worst-case” scenario?*

Numerical simulations were run by Applied Science Associates, using background conditions from summer 1994, which was characterized by cooler than average temperatures, and from summer 1996, which was characterized by warmer than average temperatures in Mt. Hope Bay. In these runs, they describe the area and volume that experienced a 0.8°C rise in temperature with the plant operating at its maximum allowable discharge temperature of 95°F. The 1994 simulation yielded the largest plume in terms of absolute temperature (percent

area with temperature $>26^{\circ}\text{C}$), while the 1996 simulation yielded the largest in terms of rise in temperature ($>0.8^{\circ}\text{C}$).

5) *Scenario testing for different discharge rates and temperatures.*

Additional simulations were run for summer 1994 and 1996 conditions, but with varying discharge temperatures (number of cooling cells active) and with actual loading based on 1996 power plant data.

History of Hydrodynamic Modeling in Mt. Hope Bay

Early modeling studies of Mt. Hope Bay were conducted by ASA (Huang and Spaulding 1995a) in the context of a combined sewage outfall (CSO) plume study. These studies used an earlier version of ASA's three-dimensional hydrodynamic and pollutant transport model to simulate the dispersal of a passive tracer released on the eastern side of Mt. Hope Bay, specifically, River View and City Pier. ASA validated the model results using field data from a dye-release experiment (also by ASA) conducted during storm conditions from September 22-25, 1990 (Turner et al. 1990).

The present-day ASA hydrodynamic model, WQMAP (described in detail in Spaulding et al. 1999b) is a three-dimensional hydrodynamic model combined with three separate water quality or pollutant transport and fate models. The model solves the conservation of momentum, mass, salt and energy equations on a spherical, non-orthogonal, boundary-conforming, sigma-coordinate grid (Figure

6.1). The grid is staggered in the horizontal, and non-staggered in the vertical. The model employs a split mode solution methodology, with the exterior mode solved semi-implicitly, and the internal mode solved explicitly by finite difference, except for vertical diffusion term, which is also handled implicitly. Eddy viscosities are specified either by the user, or through a simple turbulent kinetic energy model. The WQMAP model is described in further detail in Appendix A.

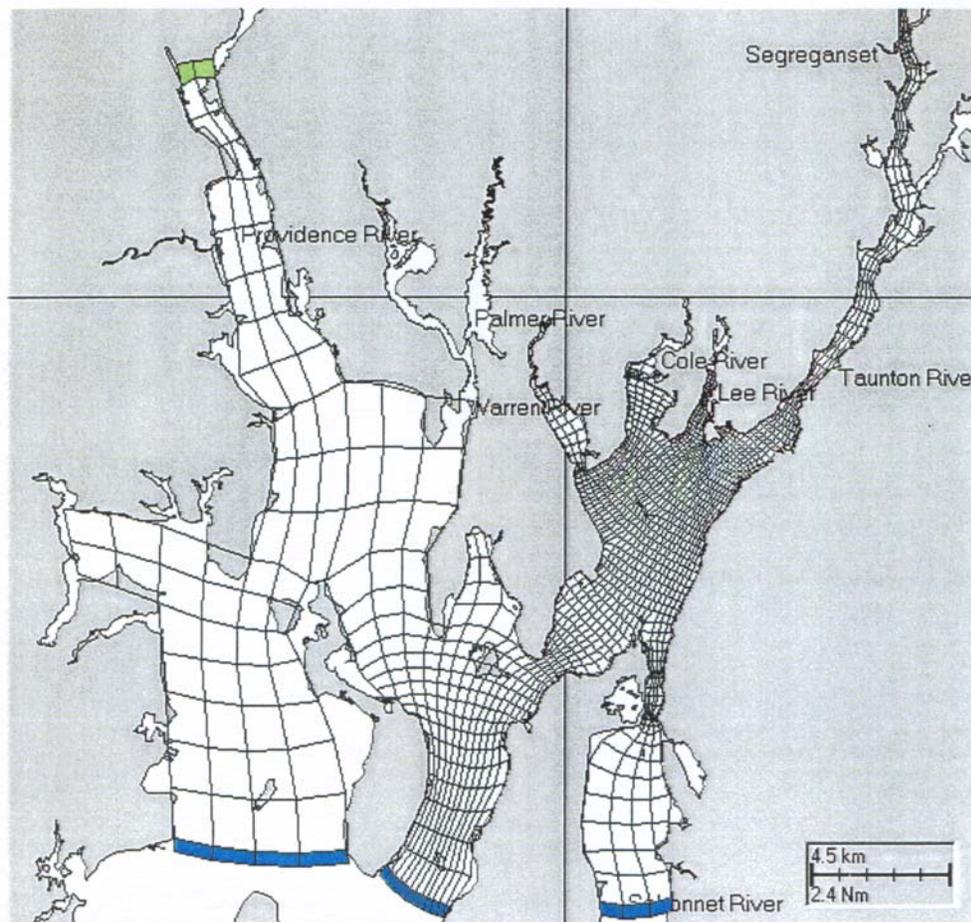


Figure 6.1. Model grid used for simulations in Mt. Hope Bay. The gridded area extends into substantial portions of Narragansett Bay to more accurately reflect conditions at the entrance to Mt. Hope Bay.

WQMAP Model Applications

In the context of the present study, WQMAP was used to hindcast the three-dimensional circulation and thermal dynamics of Mt. Hope Bay, which are subject to the discharge from the Brayton Point Station once-through cooling system. In these simulations, the model grid of 3300 cells extended from mid Narragansett Bay and Sakonnet River north to the upper reaches of the Providence and Taunton Rivers (Figure 6.1). Grid size in Mt. Hope Bay ranged from 50-100 m in the vicinity of the power plant to 200-300 m in other parts of the bay, and to greater than 1 km in parts of Narragansett Bay. Eleven layers were used to resolve the vertical structure.

The model was driven by presumed tidal sea level, temperatures and salinities at the southern boundaries, river flows scaled from gauge data at the northern boundaries, and meteorological forcing (winds, air temperature, solar radiation) from the weather station at T.F. Green State Airport in Warwick, RI. Cooling water intake and discharge conditions were based on Brayton Point Station measurements.

The model was initially calibrated and verified with field data acquired during summer 1996, spring 1997, summer 1997, and winter 1999. Model hindcasts of velocities, temperature, and salinity were found to be in good statistical agreement with the data in terms of relative error, root mean square error, linear regression analysis, and error coefficient of variation. The model was further optimized to the thermistor data acquired near the Brayton Point Station discharge (at approximately 2km) for both summer and winter conditions.

Once the model was calibrated and verified, it was used to simulate conditions in Mt. Hope Bay subjected to a series of historical and hypothetical plant loads. The purpose of the hindcast simulations was to determine the size of the thermal plume generated by the plant during previous years for the purpose of assessing potential effects on fish habitat. A series of scenario-testing simulations were also run using reduced power plant load conditions in order to evaluate potential benefits of new cooling technologies and operations.

CORMIX Model Applications

The near-field dynamics of the BPPS thermal plume were simulated using a second embedded numerical model, CORMIX (Cornell Mixing Zone Expert System)—a commercially available, USEPA-approved, near-field model that simulates temperature distributions within a discharge plume. CORMIX is a length-scale-based model that simulates the dilution of effluent from a submerged or surface discharge (Jirka et al. 1996). The CORMIX model was used to determine the extent of the Brayton Point Station's cooling water discharge plume in the near-field region (within approximately 500 meters = 1,640 ft) of the BPPS venturi. CORMIX3 (Jones et al. 1996)—a subcomponent of CORMIX—was used to estimate near-field surface plume temperatures. The results of CORMIX3 were used to determine biological effects from the thermal plume.

First, the CORMIX model was used to calculate a set of non-dimensional parameters based on ambient conditions, effluent data, and discharge geometry. The discharge flow was then classified according to a variety of defined flow

regimes. Next, CORMIX3 was used to determine the location of the BPPS plume centerline, plume width and depth, and centerline concentration and attenuation as the plume moved through the receiving water body. From these results, the volume and exposure time as a function of temperature rise and plume velocities were calculated.

To ensure consistent results in the application of the CORMIX and CORMIX3 components of the model to Brayton Point Station, CORMIX3 was assessed for sensitivity to variations in certain model input parameters. This was accomplished by running a series of simulations for various discharge flows and discharge temperatures under both summer and winter conditions. Model results were then compared in terms of the resultant temperature and velocity in the near-field region of the discharge venturi.

Combined WQMAP and CORMIX Model Results

CORMIX near-field results were integrated with larger-scale WQMAP results in order to determine the impacts of the Brayton Point Station thermal discharge in terms of the volume of water affected by the thermal plume. Results generally showed modest temperature increases (1°C) over significant portions of the bay (up to 62%), with higher temperature increases (5°C) in the very near field (less than 0.002% of the Bay). Estimates of the exposure times of near-field biota were also estimated with CORMIX. Exposure times were generally less than 15 minutes for temperature increases of 3°C, and less than 3 minutes for increases of 5°C. Exposure times in summer were significantly less than in

winter, because the plume-induced increase in water temperature was generally less in summer.

The blended WQMAP/CORMIX outputs were used to assess the extent of the temperature plume during flood and ebb tides. Results suggest that within a given season, the volume of the plume does not vary significantly over a tidal cycle. Furthermore, for a given discharge, the volume of water in the Bay that undergoes a given increase in temperature is greater in winter than in summer. This is because discharge temperatures are higher, relative to the ambient temperature of the bay, in winter than in summer. For a given discharge, the percent of water in Mt. Hope bay that exceeds a given threshold temperature value is greater in summer than in winter. Again, this is due to the higher ambient water temperatures in summer compared to winter.

Other WQMAP Model Applications in the Narragansett/Mt. Hope Bay Region

WQMAP was also applied to the Providence River/upper bay as part of a combined sewer overflow (CSO) facilities planning effort for the Narragansett Bay Commission. Here model results were compared using both dry and wet weather conditions, and the model was tuned to provide the best overall comparison with observations.

WQMAP has also been used to address the issue of total maximum daily loading (TMDL) for nutrient reduction. The hydrodynamic model was configured to run in a three-dimensional mode, with prognostic calculation of

density-induced flow. Forcing included tidal sea level, winds, river flows, and density distributions.

Finally, the WQMAP hydrodynamic and pollutant transport model has been used in dredging impact studies in the Providence River and upper Narragansett Bay. Again, the three-dimensional, time-dependent, boundary-fitted calculation used 11 layers to simulate the vertical structure of currents that resulted from tidal sea level, wind, density and river flow forcing.

2. Model Validation and Verification

The WQMAP hydrodynamic model of ASA is calibrated by adjusting model parameters to optimize the match between model predictions and observed data. Adjustment parameters include bottom and surface friction, bathymetric resolution, vertical and horizontal diffusivities, and parameterization of surface heat transfer rates. For the water quality portion of the model (i.e., DO), parameters include reaeration rate, deoxygenation rates, nitrogen mineralization and nitrification rates, photosynthesis and perspiration rates.

Model results are evaluated both qualitatively and quantitatively (ASA 1997b). Qualitative evaluations generally rely on visual comparisons of data and model results presented as time series, spatial maps, and power spectra.

Quantitative comparisons are based on the following statistical measures applied to any number of model/observable variables:

- a) Relative mean absolute error

$$e = \frac{|\bar{x} - \bar{c}|}{\bar{x}}$$

b) Relative mean error

$$e = \frac{\bar{x} - \bar{c}}{\bar{x}}$$

c) Mean relative error

$$e = \frac{\overline{|x - c|}}{\bar{x}}$$

d) Root mean square (rms) error

$$r = \frac{\sqrt{\sum (x_i - c_i)^2}}{N}$$

e) Linear regression: slope and intercept (including significance tests), r^2 , standard error

f) Comparisons of means based on t-test

g) Coherence analysis

where x_i are the observed values, c_i are the model-predicted values, and the overbar denotes the mean. Finally, model parameter sensitivity is examined to determine which model parameters are most important.

An EPA report (Martin et al. 1990) provides some guidelines as to the acceptable values of the above statistical measures (see Table 6.1). Note that the above qualitative and quantitative criteria are that they focus on comparing model results to data collected during specific time periods and specific locations in the bay. However, they do not address the larger issue of what is the inherent variability of the system, and hence what is the uncertainty associated with model

results (see: Meeting Notes from TAC Fisheries Subcommittee, November 5, 1997; and Comments by Christian Krahforst, December, 1997).

Table 6.1. EPA model calibration guidelines (McCutcheon et al. 1990).

| Analysis | Property | Value |
|--|-----------------------------------|-------|
| Relative Error | Hydrodynamic (velocity) | <30% |
| | Transport (salinity, temperature) | <25% |
| | Dissolved Oxygen | <15% |
| Coefficient of Variation (std/mean) | Hydrodynamic (velocity) | <10% |
| | Transport (salinity, temperature) | <45% |
| | Dissolved Oxygen | <1.7% |
| Correlation Coefficients | Hydrodynamic (velocity) | <0.94 |
| | Transport (salinity, temperature) | <0.84 |
| | Dissolved Oxygen | <0.80 |

Model Performance

WQMAP has been used to simulate Mt. Hope Bay conditions for a variety of BPPS discharge scenarios. Model validation was generally performed using an array of temperature strings situated around BPPS, with each station containing sensors at 0.25, 0.5, 1, 2, and 4 m below the surface. Deeper stations had an additional sensor at 6 m. Two current meter stations were also used, one at BPPS and one at Borden Flats. Water temperature and tidal height from an NOAA/NOS station at Newport were used to force the model's open boundary, while river flow rates for the Taunton and Three Mile River were obtained from USGS.

Analysis presented in Spaulding et al. (1999a) for a one-month simulation in winter 1996 shows that while the model generally captures heating and cooling trends due to atmospheric forcing, it underpredicts temperatures in Mt. Hope Bay during the coolest periods. Spaulding et al. (1999a) further find that (1) the relative mean error (*rme*) varies from 0.00 to 0.22, averaging 0.07; (2) the root mean square error (*rms* error) varies from 0.40 to 1.75 °C; (3) the correlation coefficient (*r*) ranges from 0.40 to 0.97, averaging 0.80; and (4) the *ecv* ranges from 0.09 to 0.39, averaging 0.17. Plan view maps of *rme* and *rms* temperature error across the Bay (Figure 6.2) show that in general the highest *rme*

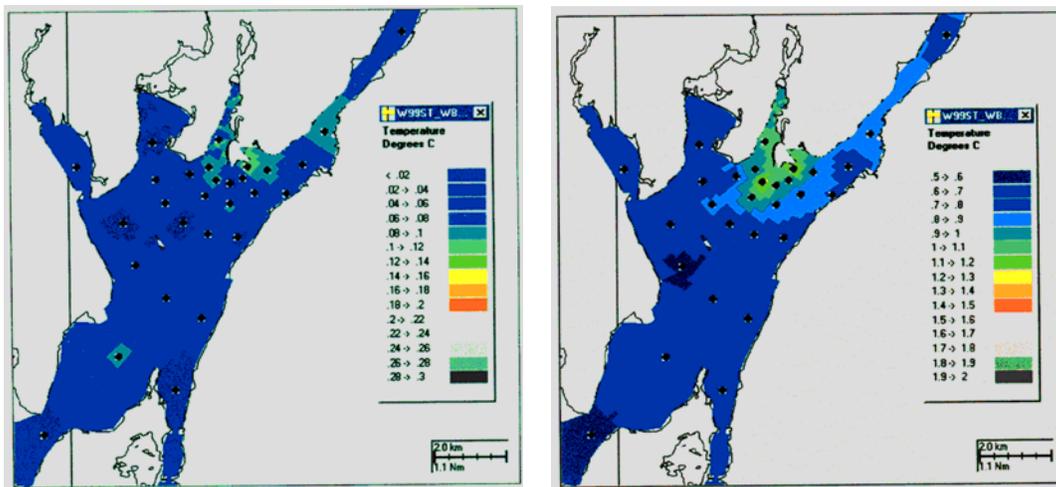


Figure 6.2. ASA WQMAP model error for Mt. Hope Bay simulations: (left panel) relative mean error (*rme*) and (right panel) root mean square error (*rms*) between model predictions and observations for the surface (0.25 m) thermister locations. (From Spaulding et al. 1999b; permission pending).

temperatures occur in the lower Taunton River, the Lee River near the discharge, and just south of Mt. Hope. Similarly, the highest *rms* error temperatures error occur in the Taunton River and near BPPS. Thus the modeling results satisfy the EPA guidelines (McCutcheon et al. 1990). The latter was also true for salinity,

sea surface elevation, and currents. Model-data comparisons were poorest for currents (C. Turner, memo dated May 26, 1998).

Model-data comparisons for winter 1997, and summer 1996 and 1997 calibration runs yielded similar results. Again temperature, salinity, sea surface elevation and currents were compared to standards described in McCutcheon et al.(1990). Model-data comparison statistics for these three sets of runs are described in detail in Swanson et al. (1998).

The criteria outlined by McCutcheon et al. (1990) for model validation and verification provide a variety of measures of performance based on mean properties of the system and at specific locations in the study region. However, it is important to note that the use of only one or two of these criteria alone, and/or the presence of spatial variations in these statistics, may make it difficult to assess model accuracy. For example, in an October 1997 memo to EPA from BBPS, ASA argued—on the basis of relative error statistics of current speed, salinity, and temperature at the surface and bottom at two stations—that the ranges of these statistics were “well within the guidance” set forth by EPA, and that “using this measure, the model is clearly calibrated.” Note, however, that the relative error is a measure only of whether the model reproduces mean property values, and says nothing about how well it simulates variations in time. In short, the complete suite of statistics outlined by EPA, both qualitative and quantitative, are required to give a complete assessment of model performance. As discussed above, a report by Spaulding et al. (1999a) further shows that even when multiple statistics are used together, spatial variations in these statistics may indicate good model

performance in some parts of the domain, while still leaving poor performance in other parts.

3. Water / Habitat Quality Modeling and Changes in Fish Abundance

Beyond the hydrodynamic modeling efforts described in the previous sections, there has been a strong consensus among state and federal agency biologists that the ASA model should be linked to both a water quality model and a fish population model in order to assess effect of BPPS's on issues such as nutrient and dissolved oxygen (DO) levels, benthic habitat, phyto- and zoo-plankton community structure, natural mortality and fishing impacts, habitat modification/avoidance, entrainment, and impingement (e.g., Meeting Notes TAC Fisheries Subcommittee, November, 1997). However, there is also much debate as to whether sufficient information is available to constrain such models of Mt. Hope Bay, particularly since it is an open system with respect to fish populations, and since much of the life history of the local fish populations occurs outside the bay (e.g., Review of Mt. Hope Bay Modeling Proposal by Steve Cadrin, July, 1998, unpublished memo; Coutant et al. 1998).

A review by Coutant et al. (1998) further emphasized that from a modeling perspective uncertainties in a few key variables limit the accuracy of predicted impacts. For example, the following variables are considered the most important for controlling predictions in assessments to estimate the population-level consequences of entrainment and impingement:

1. Size and geographic extent of source adult population
2. Size of source population of early life stages from which losses occur
3. Number of early life stages killed by entrainment and impingement (especially the probability of through-plant survival)
4. Natural mortality rates (especially those for early life stages) used to extrapolate losses of early life stages to equivalent adults, production foregone, or reproductive potential or used to simulate population trajectories or estimate risk of extinction
5. Density-dependent effects

Coutant et al.(1998) further concluded that since all but one or two of these variables (numbers 1 and 3) are extremely difficult to constrain from field data, the expectation or hope that better or different models per se will lead to better predictions of population-level effects is not realistic. The above considerations suggest that a more thorough examination of whether our understanding of the Mt. Hope Bay ecosystem is model- or data-limited is thus advisable in the early developmental stages of the proposed Mt. Hope Bay Natural Laboratory.

Water Quality/DO Modeling

The water quality/DO component of ASA's WQMAP is based on the EPA WASP5 model (Ambrose et al. 1992). It is intended to be able to run at any of six levels of complexity ranging from a simple balance between biochemical oxygen demand and dissolved oxygen (BOD-DO) to increasing levels of nutrient,

phytoplankton, BOD and DO representation, up to full eutrophication kinetics including the benthos. The different levels of complexity are as follows:

1. Streeter-Phelps, a simple BOD-DO balance;
2. Modified Streeter-Phelps, BOD-DO balance with BOD compartmentalized into carbonaceous and nitrogenous BOD (CBOD and NBOD);
3. Full linear DO balance, which adds the effects of photosynthesis, respiration, and more complex nitrogen kinetics;
4. Simple eutrophication kinetics, including phytoplankton kinetics and more complex nutrient interactions;
5. Intermediate eutrophication kinetics, with full nutrient cycles including nonlinear feedbacks;
6. Intermediate eutrophication kinetics with benthos, which includes the simulation of all state variables in the benthic segments.

The DO balance in the WQMAP eutrophication model is determined by five state variables: phytoplankton carbon, ammonia, nitrate, carbonaceous biochemical oxygen demand, and DO. Sources and sinks include reaeration and phytoplankton growth (both of which are sources), phytoplankton respiration, oxidation of carbonaceous material, (including sediment demand) and nitrification (all of which are sinks); plus transport terms.

Calibration of the DO model is described in detail in Swanson et al. (1999). The initial model process rates are based on values obtained from the literature and then adjusted once the hydrodynamic model has been fully calibrated. Calibration of the DO model is based on how well the model reproduces (a) BOD and nitrification process distributions and (2) constituents affected by other processes. The DO balance is calibrated after the BOD, nitrification, and photosynthesis sub-models are calibrated. In addition, quantitative evaluations of DO model-data comparisons (based on the error statistics similar to those used in the hydrodynamic model described above) are compared to EPA-recommended levels.

Preliminary ASA-model simulations of dissolved oxygen (DO) in Mt. Hope Bay are summarized in Isaji and Rines (1998). According to Swanson et al. (1998), initial model runs "successfully" simulated mean DO conditions for summer 1996 but did not achieve the dynamic range of DO seen in the field data. Hence the WASP5 model was replaced by a much simpler box model which employed the same set of governing equations but could not resolve spatial distributions of WASP5. The replacement model included reaeration, BOD oxidation, nitrification, sediment oxygen demand, phytoplankton growth and respiration (a single daily net value), horizontal exchange, and vertical exchange. Three stations were chosen to evaluate how DO can be expected to vary with temperature. Rather than attempting to simulate time series of DO at the specified stations, the runs focused on determining how each term in the DO balance effects the variability observed at each station. In this way, the relative

importance of temperature at each station could be compared to that of the other variables, and the impact of different levels of thermal discharge could be determined.

Eutrophication Modeling of Mt. Hope Bay

The purpose of eutrophication or water quality modeling is to provide a tool for evaluating the linkages to watershed nutrient loading. A properly parameterized and validated eutrophication model can be used to identify: (1) the nutrient sources controlling water quality, both within and external to the Bay; (2) the critical factors and physical conditions which control bottom water oxygen levels; (3) the relationship of oxygen conditions to organic matter production within the Bay versus that entering the Bay from the watershed or via adjacent marine waters; (4) areas where additional field data collection is needed; and (5) the potential for improvements in the health of Mt. Hope Bay through reduction of nitrogen sources or other key parameters.

Typical eutrophication models consist of eight water quality variables: (1) ammonia (NH_3); (2) nitrate and nitrite (NO_2 and NO_3); (3) inorganic phosphorus (OPO_4); (4) organic nitrogen (ON); (5) organic phosphorus (OP); (6) phytoplankton (PHYT); (7) carbonaceous biochemical oxygen demand (CBOD); and (8) dissolved oxygen (DO). In addition, a prerequisite for any water quality or eutrophication model is a good hydrodynamic model that provides information on 1) river discharges (spring and early summer), 2) water exchange between the bay and surrounding area, 3) seasonal development of stratification and sediment

resuspension and 4) tidal- and wind-induced mixing. Benthic and sediment resuspension processes may also be incorporated into water quality models via a benthic layer and/or a sediment pool on the bed of the estuary. Such biological/chemical models are useful for simulating basic transformation processes including photosynthesis, respiration, nitrification, denitrification, sediment suspension, and nutrient release to overlying waters from bottom sediments. As a result, water quality models are useful for understanding the critical parameters controlling eutrophication within a specific embayment and for evaluating options for system management. In addition, eutrophication models can be linked to fish population dynamics through a zooplankton component, which may have application to the Mt. Hope Bay Natural Laboratory Program. The most practical approach for configuring an eutrophication model in a new bay system is to begin with the simplest possible dynamics in the model, and then proceed to incrementally increasing levels of complexity by stages.

To date, eutrophication modeling within Mt. Hope Bay has focused upon simulating dissolved oxygen conditions. Preliminary simulations of dissolved oxygen (DO) in Mt. Hope Bay have been performed by ASA using the EPA WASP5 model (Isaji and Rines 1998; see also Ambrose et al. 1992). However, as reported by Swanson et al. (1998), while initial runs showed that the model was able to simulate mean DO conditions for summer 1996, it could not achieve the dynamic range of DO seen the field data. It was thus proposed by ASA that the model be replaced by a much simpler box model which uses the same set of governing equations as WASP5, but which does not contain any information on

spatial distributions. The proposed model included reaeration, BOD oxidation, nitrification, sediment oxygen demand, phytoplankton growth and respiration (a single daily net value), horizontal exchange, and vertical exchange. Three stations were chosen to evaluate how DO can be expected to vary with temperature. Rather than attempting to simulate time series of DO at the specified stations, the runs focused on determining how each term in the DO balance effects the variability observed at each station. In this way, the relative importance of temperature at each station could be compared to that of the other variables, and the impact of different levels of thermal discharge could be determined. Reports describing the results from the above box model simulations have not yet been obtained by SMAST and have not yet been reviewed at this time.

To adequately understand the role of various inputs of nutrients and freshwater to Mt. Hope Bay relative to habitat health and stress to animal communities, eutrophication modeling needs to be undertaken. In its simplest form, the model needs to integrate watershed inputs with respiration and regeneration within the Bay, inputs from greater Narragansett Bay, and distribution of water quality as determined by hydrodynamics. After this is accomplished, the addition of primary production, dissolved oxygen or animal (fish) components will be addressed.

Fish Modeling

As discussed in Chapters 4 and 5, dissolved oxygen concentration and benthic prey abundance and type play an important role in determining distributions of key fish species in Mt. Hope Bay, including winter flounder. Furthermore, in a January 1999 workshop held at the URI Coastal Institute to discuss factors that may have affected winter flounder in Narragansett Bay, it was concluded that stressors for that species also include natural variability, pollution, habitat change, power plant effects, predators, regime shifts in community structure, and fishing mortality (Collie and DeLong 2001). Table 6.2 provides a list of detailed factors and the life stages that are believed to be affected.

Table 6.2. Factors potentially affecting the survival of winter flounder in Narragansett Bay (from Collie and DeLong 2001).

| Stressors | Spawning | Egg | Larvae | 0-Group | Juveniles | Adults |
|--|-----------------|------------|---------------|----------------|------------------|---------------|
| Natural Variability | | | | | | |
| Temperature | x | x | x | X | x | |
| Precipitation/runoff | | | | X | | |
| Pollution | | | | | | |
| Organics | x | ? | | X | | |
| Metals | ? | ? | | X | | |
| Sewage | ? | ? | ? | X | | |
| Chlorine | | | ? | X | ? | |
| Habitat Loss | ? | ? | | X | ? | |
| Power Plant | | | | | | |
| Entrainment | | | x | | | |
| Heating | x | x | x | X | x | x |
| Predators | x | x | x | X | x | x |
| Regime Shift in Community Structure | | | | | | x |
| Fishery | | | | | x | x |

Notes: x: Scientific evidence to support

Although discussions during the aforementioned workshop focused on winter flounder, it is not unreasonable to assume that similar factors affect other species as well, although the exact details may differ. Indeed, the simplest explanation for the decline in fish populations in Mt. Hope Bay is that the large change in coolant flow at BPPS has modified environmental conditions in MHB to the detriment of the fish populations living there (Coutant et al. 1998). Fluctuations in Narragansett Bay winter flounder abundances have been linked to fluctuations in winter water temperatures; the available data suggest a temperature- and/or oxygen-mediated effect, although causality cannot be demonstrated based on available monitoring data (Coutant et al. 1998). In addition, as discussed in Chapter 4 and 5, increases in winter zooplankton abundance and feeding activity, and corresponding declines in winter-spring chlorophyll concentration in lower Narragansett Bay, occur partially in response to warming (winter) water temperatures. However, as also discussed in Chapter 5, the variations (on the order of $\pm 2-3$ °C) are well within the physiological thermal tolerance of winter flounder, suggesting secondary rather than direct temperature effects.

Other correlations between the decline of fish stocks and habitat changes in Mt. Hope Bay are also evident. For example, it is known that winter flounder prefer relatively open sand/silt barren bottom, as well as such areas that are adjacent to eelgrass beds. Furthermore, as discussed in Chapter 5, there has been a recorded decline in the extent of coverage of eelgrass beds in Narragansett Bay. Again, causality cannot be demonstrated. However, the coincidence of these

trends suggests that the decline of eelgrass beds (which may be due directly or indirectly to the thermal effects of BBPS discharge, or to changes in stratification, circulation, or nutrient and/or DO levels) may be contributing to the decline in the populations of winter flounder and other fish species in the bay.

To date, there have been two attempts to assess the effects of BPPS on fish populations in Mt. Hope Bay. The first, by Collie and DeLong (2001), used empirically based correlation models to determine whether timing and/or magnitude of winter flounder migrations have changed due to BPPS, to look for time trends and density-dependence in mortality rates, and to look for changes in spatial distributions of winter flounder within the Bay. The latter was used to infer changes in habitat suitability, under the assumption that mobile organisms will distribute themselves in accordance with the relative suitabilities of the regions in which they reside. This study confirmed that catches of larger female winter flounder in Narragansett Bay have declined more than they have in areas immediately outside the bay, and that one of the key factors in this decline may be higher mortality rates prior to recruitment (age-1 fall to age-2 spring). Furthermore, spatial analysis suggested that changes in habitat quality within Narragansett Bay may have affected the population.

The second attempt to assess the effects of fisheries, power plant operations, and water quality on fish populations was proposed by Lawler, Matusky & Skelly Engineers, LLP (see Swanson et al. 1998). This study used the RAMAS GIS/Metapop model developed by Applied Mathematics to predict the effects of changes in fishing pressure and BPPS plant operations on selected fish

populations within Mt. Hope Bay. To date, the model has been calibrated using available historical data on catch-per-unit-effort (CPUE) of winter flounder in Mt. Hope Bay. Once calibrated, the model was used to forecast changes in stock abundances that are likely over the next 30 years to result from different management, fishing mortality, and power plant operation scenarios.

In the context of Mt. Hope Bay fish populations, the RAMAS GIS/Metapop model was run as a single-box model in order to simulate trends in selected fish species within the Bay. For input, the model utilizes demographic data such as recruitment, growth rates, fecundity, and natural mortality, as well as anthropogenic influences such as fishing mortality and power plant entrainment and impingement loss. The model integrates these competing factors according to their respective rates until the overall biotic potential of the species in question is balanced by the carrying capacity of the system. The carrying capacity in turn is set by environmental as well as biological factors such as maximum critical temperatures, avoidance temperatures, minimum DO levels, and density dependent mortality. For Mt. Hope Bay, species such as winter flounder are modeled as an isolated population (i.e., metapopulation), possibly linked to a second meta-population representing those fish that migrate in and out of the bay and are hence subject to additional fishing pressures outside the bay.

Historical simulations using the RAMAS model indicated that trends in population abundance of winter flounder during the period 1958-1999 were very similar, regardless of whether the model was run with the plant operating or not. This suggests that the plant's intake and discharge have had negligible effects on

winter founder population trends over the past 40 years. Simulations projecting population levels of the same species, but for the next 30 years, further suggest that proposed targeted fishing restrictions will lead to population recovery approaching historical levels (i.e., pre-BPPS) over approximately the next 10 years. The model predicts that this recovery level would be diminished by 13.9 % in the case where BPPS continues to operate under the MOA II conditions, compared to levels that would obtain if the plant were absent. Similarly, the recovery is predicted to be reduced by 10.4% under the new enhanced multi-mode operation scenario proposed by BPPS.

In addition to the above modeling studies, as discussed in Chapter 5, there are also exist a number of data sources that have hitherto been underutilized. There are approximately 30 years (1971 to present) of monthly to twice-monthly trawl data from various locations in Mt. Hope Bay that provide information on fish abundance and community composition. However, these do not appear to have been thoroughly analyzed in terms of comprehensive examinations of long-term trends in fish abundance and community structure. Dissolved oxygen data are also available from both the MRI data and the DEM juvenile beach seine survey; these would be very useful in calibrating and evaluating future DO modeling studies. Additional data sources, such as the database of chlorine discharges from wastewater treatment plants compiled by Collie and DeLong (2001), could also provide valuable input to the models.

Numerous other data sets also exist for nearby regions, including lower Narragansett Bay; unfortunately, available empirical information from other study

areas cannot always be extrapolated directly to Mt. Hope Bay, and hence expected effects must be inferred. In particular, there is little information in the literature on chronic effects, although available data suggests that significant effects ranging from growth, reproduction, and various behavioral responses can result from temperatures that are sub-lethal (Coutant et al. 1998).

4. Summary of Modeling and Future Directions for the Mt. Hope Bay Natural Laboratory

Given the mixed successes of previous numerical modeling efforts in Mt. Hope Bay, the following questions need to be considered and addressed:

- What level of simulation do we attempt to pursue?
- What kinds of data are currently available for simulation and/or data simulation?
- What kinds of new data are expected from the future field measurements?

With these questions in mind, the following models appear to be viable candidates for Mt. Hope Bay Natural Laboratory modeling studies at SMAST:

- 1) ASA non-orthogonal coordinate model;
- 2) Finite-Volume Coastal Ocean Model (FVCOM–Dr. C. Chen, SMAST);
- 3) RMA11 (3-d finite element water quality model–Resource Management Associates and the Army Corps of Engineers);

- 4) ECOM-si (updated Princeton model);
- 5) POM (old version of the Princeton model);
- 6) Dartmouth Element Model.

Each of these models has strengths and weaknesses that will need to be considered during the next phase of the project. The question of how extensive a biological model should be incorporated as part of the Mt. Hope Bay Natural Laboratory depends on the ultimate goal of the modeling efforts. If the goal is to understand fish decline over the last decade, a water quality model is likely a better choice than a general ecosystem model (such as the PZND model). In either case, however, a good hydrodynamic model is a prerequisite in order to provide information on 1) river discharges (spring and early summer), 2) water exchange between the bay and surrounding area, 3) seasonal development of stratification and sediment resuspension and 4) tidal- and wind-induced mixing.

Chapter 7. Summary and Conclusions (Rodney Rountree)

A great deal of research has been conducted on the physical and biological environment of Mt. Hope Bay and the adjacent Narragansett Bay. However, limited work has concentrated on the problem of main interest to the Mt. Hope Bay Natural Laboratory project, namely, quantification of the sources of temporal variation of the Mt. Hope Bay environment and fish stocks. Biological trends in Mt. Hope Bay appear, for the most part, to mirror trends in greater Narragansett Bay and the surrounding Rhode Island Sound geographical regions, although some trends appear to be specific to Mt. Hope Bay, itself. For example, temporal trends in habitat loss and in winter flounder population dynamics both mirror trends in Narragansett Bay and in the northeast geographic region.

Physical Environment

Understanding the physical environment of Mt. Hope Bay is the foundation of all other components of the Mt. Hope Bay Natural Laboratory Program and is critical to its success. Temporal variation in nutrients and in plankton, crustacean, and fish populations cannot be fully understood without a detailed understanding of the physical environment, its sources of variation, and its interaction with the biological community. As is evident in the review in Chapter 2, there is a significant amount of environmental data for MHB that can be used to build the program; however, gaps in our knowledge need to be

addressed. Priority observational data needs for the physical environment include the following:

- 1) *Long-term monitoring of spatial and temporal patterns in water properties*, including temperature, salinity, DO, and turbidity, is needed to provide adequate input into physical models of the Bay. Previous work has been short-term and cannot be used to fully characterize seasonal and interannual variations. Just as importantly, long-term observational data is critical to understanding the importance of episodic events and their interactions with physical processes on various temporal and spatial scales. Such data would allow for the detailed characterization of water mass boundaries in the Bay, and the identification of stratification boundaries, upwelling and downwelling zones, mixing zones, and sediment resuspension zones.
- 2) *Tidal and wind-driven water circulation patterns* need to be observed on a fine-scale grid throughout the bay and over time scales sufficient to characterize monthly and seasonal variations and interactions with seasonal and episodic weather patterns. Of special interest are circulation interactions between the open bay and the shallow subtidal and intertidal estuarine fringe areas that function as the primary nursery grounds for many estuarine fishes including winter flounder. Attempts to model fish egg, larvae and plankton movements through the bay will be ineffective without this effort. It is critical that the tidal and non-tidal water exchange patterns between Mt. Hope Bay and the

adjacent Sakonnet River and Narragansett Bay be well understood in order to model input and output of nutrients, plankton, and fish eggs and larvae for the Bay.

- 3) Similarly, *data on the freshwater discharge from the Taunton, Lee, Cole, Kickamuit, and Quequechan Rivers* into Mt. Hope Bay need to be obtained over a sufficient period to allow modeling of seasonal, interannual, and episodic patterns. From data presented in Chapters 2 and 3, it is clear that the seasonal timing and magnitude of peak discharge varies greatly among years. Interannual and weather-related changes in discharge patterns may be especially important to understanding larval fish movement patterns, as well as Mt. Hope Bay hydrography changes.
- 4) *Observational data collection and modeling of the thermal discharge plume* from the Bryton Point Power Plant can be improved. Of special interest is the interaction of the plume with shallow subtidal and intertidal waters along the Bay fringe, since these are the primary nursery habitats and foraging grounds for winter flounder and many other fishes and invertebrates of concern.

Habitats and Habitat Quality

The historic, present, and future changes in habitat distribution and habitat quality in Mt. Hope Bay, and the causes of those changes, are poorly understood at present. Current evidence suggests that dramatic changes in habitat distribution

and type have occurred over the last century and accelerated in recent decades. In addition to their impacts on the fisheries resources, many of these changes endanger the habitats themselves, which are in their own right a valuable resource in need of protection. Critically important habitats such as saltmarshes and eelgrass beds have been especially hard hit. Although available habitat mapping data have not been analyzed specifically for Mt. Hope Bay, it is thought that as much as 70% of the saltmarshes in Narragansett Bay have been modified by human activity. Eelgrass beds, once considered a vital habitat for winter flounder, have vanished from Mt. Hope Bay and are restricted mainly to the lower Narragansett Bay. Detailed data on habitat types based on emergent vegetation and shoreline characteristics are available from several sources but have not been summarized specifically for Mt. Hope Bay. A comparison between present vs. predicted future habitat distributions should be an important component of the MHBNL program. In addition, detailed sediment type mapping should be conducted throughout the Bay.

Besides the basic step of mapping habitat type distributions in the bay, an understanding of factors affecting habitat quality is of great importance. Nutrient loading into Mt. Hope Bay is probably one of the most significant factors regulating the Bay's habitat quality. Quantitative data is currently insufficient to model nutrients in the system; however, the conclusion that the Bay is likely nutrient-enriched and eutrophic seems justified.

Because excessive nutrient loading is clearly having a significant impact on habitat quality in Mt. Hope Bay and Narragansett Bay, it is critical that

eutrophication models be used to determine the effect of increasing development on habitat quality. A comprehensive program to monitor nutrient inputs and outflows from Mt. Hope Bay and the Taunton River system is needed. An evaluation of the relationship between freshwater discharge and the dissolved oxygen levels of the Bay's bottom waters would be one important component of such a program. Quantitative observations and subsequent modeling of the spatial and temporal patterns of low dissolved oxygen concentrations in Mt. Hope Bay bottom waters are particularly important to assess habitat quality.

Significant data on the benthic community and production in Mt. Hope Bay is available, but linkages between pelagic and benthic trophic systems and interactions with nutrient availability and environmental conditions need to be examined. Environmental modeling of the Bay's hydrography and identification of vertical mixing zones and sediment re-suspension zones are important to this effort. Emphasis on the production of prey species and environmental factors regulating their spatial and temporal distribution patterns would be especially useful for the study of winter flounder post-settlement larvae and juveniles. A determination as to whether winter flounder larval settlement is matched with benthic prey abundance peaks is an important consideration. Monitoring of the distribution patterns of important biotic integrity indicator species such as *Ampelisca*, *Mediomastus* and *Nucula* is also an effective tool for assessment of habitat quality and should be done on a broader spatial scale along the estuarine gradient.

Plankton

Quantification of plankton dynamics in Mt. Hope Bay is essential for understanding larval fish survival and growth patterns in the Bay. Extensive plankton work in Narragansett Bay provides a sound basis for expected plankton processes in Mt. Hope Bay, but site-specific factors affecting Mt. Hope Bay spatial and temporal plankton distribution need to be identified. Currently, only a basic knowledge of species compositions is available for Mt. Hope Bay itself. Plankton dynamics are dependent on hydrography and nutrient availability. In addition, because phytoplankton and zooplankton are important components of crustacean and fish food webs, especially for the larvae, observational and modeling data on plankton abundance and distribution problems are important to understanding fish population dynamics. Better seasonal and spatial coverage of phytoplankton, zooplankton, and ichthyoplankton abundances are needed to allow analysis of match and mismatch between fishes and their food resources. In particular, correspondence between plankton/zooplankton blooms and the timing of spawning and larval fish immigration into the estuary is of considerable importance. For example, if winter flounder spawning and/or egg hatching occurs out of phase with zooplankton prey peak abundance patterns, excessive larval mortality can occur. Interannual variation in predator and prey matching is an important contributor to annual variations in larval survival. A well-performing model of Mt. Hope Bay water circulation and tidal flow patterns--one which incorporates water quality parameters such as temperature and nutrient availability--is an important preliminary to modeling of phytoplankton,

zooplankton, and ichthyoplankton distribution and dispersal patterns. Behavioral responses of zooplankton and ichthyoplankton to water quality parameters should also be incorporated into the models. This is especially important for the ichthyoplankton, for which starvation is of prime importance. Short of starvation, lesser effects of variability in food availability may be important in tandem with physiological responses to environmental conditions. For example, a poorly fed larval fish may be further weakened if forced to expend its energy reserves in response to even small changes in water density (e.g., for osmoregulation and buoyancy control), temperature and dissolved oxygen (e.g., for metabolism, thermoregulation, etc.).

Nekton

Although we have identified a strong foundation of biological data on fishes for Mt. Hope Bay, data on other nektonic components of the system are lacking. Monitoring of key decapod crustaceans in the Bay is vital to understanding predator-prey relationships and food habits of the fish stocks. At the minimum, data on the population dynamics of the shrimp *Crangon septemspinosa* should be sought, as it is known to contribute to larval settlement and juvenile winter flounder mortality. Historical monitoring programs for the fishes provide a strong foundation for the MHBNL, and should be continued, but some important data gaps are evident here, too. One of the most important data gaps is the complete lack of food habitat data needed to determine trophic linkages and predator-prey relationships among fishes. Piscivory is likely an

important component of natural mortality in the system and should be included in the MHBNL modeling efforts.

Habitat use patterns by various life stages of winter flounder need to be determined. Some effort has been directed towards identification of winter flounder habitats, but further efforts are required to more fully identify and characterize important spawning, larval settlement, juvenile nursery, age-1 nursery, age-2 nursery, and adult habitats. Identification of settlement habitats/areas and post-settlement habitat use patterns is especially important, as the settlement period is generally considered the most significant mortality bottleneck for flatfishes. The exchange of winter flounder of all life history stages among Mt. Hope Bay, Narragansett Bay, Sakonnet River, and Rhode Island Sound needs to be quantified. The contribution of Mt. Hope Bay-spawned winter flounder to local and regional recreational and commercial landings is unknown. What percentage of spawning adult winter flounder in the Bay are removed by the fishery? How rapidly can such removal be compensated by immigration of adults/juveniles from other areas?

In this report we identify at least eight key factors that should be considered in the the MHBNL assessment of fish stocks: 1) estuarine residence/fidelity; 2) contribution of Mt. Hope Bay to both recreational and commercial fishery stocks (i.e., the percentage of harvested fish that are derived from the Bay, and conversely, the percentage of Mt. Hope Bay stocks that are harvested); 3) emigration and immigration timing, and identification of environmental migration triggers; 4) spawning, including habitat identification,

timing, triggers and behavior; 5) larval: mortality, tidal transport behavior, dispersal patterns, intra- and extra-bay sources (i.e., component of larvae derived from spawning within Mt. Hope Bay versus larval influx through the East Passage and the Sakonnet River passage), and energetic response to physiological demands under different environmental conditions; 6) estuarine movements, including daily home range and exchange between MHB and adjacent estuarine areas; 7) estuarine growth in different habitat types and under various environmental conditions; and 8) relative contribution of estuarine and coastal marine nursery areas. Although these factors are discussed in detail for winter flounder, most also apply to other species.

Models

A number of hydrodynamic, water quality, and fish modeling studies of Mt. Hope Bay have been conducted with the principal goal of assessing the impact of the Brayton Point Power Station (BPPS) thermal plume on local fish populations. Modeling efforts in the Bay have been restricted to a few efforts with specific objectives. No attempts have been made to incorporate predator-prey interactions into fish stock models. Although the fish models include a consideration of temperature and dissolved oxygen tolerances, they do not attempt to incorporate behavior effects such as selective tidal-stream transport, energetics, and behavioral thermoregulation on fish movements and habitat-specific use patterns. More subtly, they do not currently incorporate ontogenetic and seasonal shifts in temperature tolerances.

Sources of variation

As described in the introduction, one of the major goals of the MHBNL is to determine how natural resources such as fish stocks are impacted by sources of temporal variation. Our review of research in Mt. Hope Bay suggests much more work is needed before these natural and anthropogenic sources of population variation can be modeled.

Water Quality: We have only limited data on factors affecting water quality in Mt. Hope Bay. The effects of the BBPS on the Bay's water temperature has received the most attention, and is perhaps best understood. Although modeling of the spatial resolution and variation in the thermal plume can be further enhanced, an analysis of the interaction between the thermal plume and the shallow nursery habitats for winter flounder is perhaps most in need of improvement. Quantification of the interaction of the heated discharge with daily and tidal cycles in temperatures, and of the seasonal changes in these patterns, is perhaps of greater importance to understanding habitat suitability for winter flounder than is the knowledge of absolute warming effects. Similarly, if the seasonal rate of change in water temperature is an important trigger for spring and fall migrations of juvenile and adult winter flounder, then annual variation in seasonal rates of change can have important consequences for fish stocks. Nutrient enrichment of the system is probably one of the most important sources of variation to be considered by the MHBNL program. It impacts habitat quality and plankton dynamics. Loss of habitat, changes in sedimentation, changes in

dissolved oxygen levels, and ultimately changes in trophic structure associated with eutrophication can have profound impacts on community structure and fish populations in the Mt. Hope Bay.

Climate variability: Interannual variability, particularly of air temperature, wind patterns and rainfall, likely has an important impact on plankton population dynamics, including fish larvae. Of these, climate effects on water temperatures have been most thoroughly examined. However, interactions between annual temperature changes and community structure need clarification. For example, credible evidence has been presented for Narragansett Bay that suggests that winter flounder population fluctuations result not from direct temperature effects, but indirect effects where temperature mediates the mortality of winter flounder from *Crangon septemspinosa* predation. In warm years *C. septemspinosa* is more abundant and more active, resulting in increased predation on post-settlement winter flounder larvae and early juveniles. This illustrates the importance of incorporating predator-prey as well as environmental factors into fish population modeling efforts as part of the MHBNL program.

Interannual changes in rainfall result in interannual variation in freshwater input into Mt. Hope Bay. Correlations between freshwater discharge volume/timing and the timing of winter flounder spawning, larval hatching, and plankton and zooplankton blooms should be investigated. Once the relationship between freshwater discharge and low dissolved oxygen levels in the Bay's bottom waters is resolved, then annual variation in the occurrence of hypoxic waters can be better modeled and related to habitat suitability. Interannual

variability in wind patterns may also result in annual differences in water circulation patterns, and/or in the timing of stratification events. In short, climate variability among years is likely to be a major factor controlling interannual variability in Mt. Hope Bay environment and community structure.

Habitat loss/change: Knowledge of habitat distribution, loss and change in Mt. Hope Bay is probably our weakest link. It is imperative that better information on habitat type distribution and coverage be obtained in order to understand past and future changes. Programs to map habitat type distribution and change in Mt. Hope Bay, from its connections with Narragansett Bay and the Sakonnet River to the tidal freshwater reaches of the Taunton, Lee, Cole, Kickamuit and Quequechan Rivers, should be a high priority. This includes mapping of vegetation types, tidal pools, tidal creeks, shoreline types, and sediment types. The availability of habitat maps upon which environmental parameters can be overlain to determine habitat suitability would provide a significant improvement in current modeling efforts. Fortunately, excellent raw data on emergent vegetation and shoreline habitat types, although not yet summarized specifically for Mt. Hope Bay, is available from various state and federal programs. Additional efforts to map subtidal habitat types, particularly those based on sediment type and grain size, should be considered.

Fishing pressure: It is clear from our review that fish population trends for Mt. Hope Bay reflect regional patterns that are likely to be driven by fishing pressure, though the contribution of habitat loss and habitat quality change are also likely to be important on a regional scale. However, a local effect is

suggested by the apparent failure of the Mt. Hope Bay and greater Narragansett Bay winter flounder stocks to recover as rapidly as other Rhode Island stocks and those of Georges Bank and the Gulf of Maine.

One of the most important issues to address is whether winter flounder return to spawn in their respective natal estuaries each year, or whether substantial mixing occurs between the spawning populations of Mt. Hope and Narragansett Bay. Better data on the relative contribution of recreational and commercial fishing to intra-bay and extra-bay mortality of Mt. Hope Bay winter flounder is needed to help clarify this issue. The slow recovery in Mt. Hope Bay could result from excessive removal of the Bay's relatively small spawning stocks during their residence in coastal waters or during their migration to and from the coastal waters from the Bay.

Community Change: Although significant data resources are available from standard trawl and seine surveys in Mt. Hope Bay, only limited analysis of fish community change has been attempted. In particular, little effort has been directed towards relating changes in predator populations to winter flounder population dynamics. Known significant fish predators of winter flounder include summer flounder, striped searobin and bluefish. An assessment of their interrelationships is needed. Similarly, as mentioned previously, the shrimp *Crangon septemspinus* has been implicated as a potential population regulator of winter flounder in Narragansett Bay and elsewhere, but has not yet been examined in Mt. Hope Bay. Unfortunately, historical data on its abundance patterns in Mt. Hope Bay do not appear to be available. Predator-prey dynamics

and multispecies interactions are perhaps best quantified through food habit data, but such data appear to be lacking for Mt. Hope Bay. Initiation of food habit studies, particularly for winter flounder and its predators, would likely be an important component of the MHBNL program.

State of knowledge of MHB

Mt. Hope Bay has been subjected to significant environmental and biological change over the last three decades. The Bay has followed a regional warming trend due to climate warming; this has been enhanced by additional warming resulting from thermal discharge of the Brayton Point Power Station. At the same time, the Bay has suffered strong declines in important estuarine habitats such as its saltmarshes and eelgrass beds. In fact, historically important eelgrass beds probably completely disappeared from the Bay sometime during the last three or four decades. Water quality has also declined sharply due to increasing nutrient loading resulting from human population increases and development in the Bay's watershed. Nutrient enrichment in turn modifies habitat availability and quality through eutrophication processes. One impact of this has been the increased prevalence of hypoxic bottom waters in the Bay during the summer months. Finally, some Mt. Hope Bay fish populations, such as winter flounder and tautog, have declined dramatically since the early 1980s, while others have increased. However, each of these patterns appears to result from processes operating on a broader geographic scale, though the magnitude of the effects in Mt. Hope Bay may be influenced by processes specific to the Bay itself.

Although there are many unknown details of concern, there are a few general areas in which our lack of knowledge is most limiting of our understanding of Mt. Hope Bay as an ecosystem. To address these deficiencies, we recommend the following research goals: 1) quantification of the distribution and percent cover of specific habitat types, historic changes in distribution and coverage, and trophic linkages among them; 2) identification of essential habitats for each life stage of key fishes and invertebrates; 3) quantification of nutrient loading to the bay and its impact on dissolved oxygen levels and other attributes of habitat quality; 4) acquisition of food habits data are needed to examine intra- and inter-specific interactions and their impact on community structure; 5) understanding of the exchange of fishes, through tidal, daily, ontogenetic and seasonal movements, between Mt. Hope Bay, its rivers, and Narragansett Bay; 6) quantification of the fidelity of individual winter flounder to Mt. Hope Bay, and of the contribution of within-bay and extra-bay sources of mortality to the population specific to MHB; 7) linking of multispecies population models to environmental processes such as habitat suitability, larval dispersal, and predator and prey population dynamics; and 8) more broadly, integration of a network of models of the Mt. Hope Bay environment, including temperature regimes, dissolved oxygen distribution levels, stratification and water circulation patterns, nitrogen loading and eutrophication, nutrients, phytoplankton, zooplankton and ichthyoplankton retention and dispersal patterns, and fish population dynamics.

This review has found that there are significant data resources available on the environment and fish stocks of Mt. Hope Bay that can serve as a strong

foundation for the Mt. Hope Bay Natural Laboratory. The report also identifies key data gaps that can be addressed to enhance the program. However, although some data gaps are qualified as "important" or "critical," we do not attempt herein to prioritize study areas for inclusion in the Mt. Hope Bay Natural Laboratory. These data gaps will be further identified and prioritized in the planning phase of the program. It is not expected that the MHBNL will directly address all of these areas. Instead, we will focus on areas that we feel represent the strongest regulators of the Mt. Hope Bay environment and community structure and which are most suitable for our modeling efforts.

Mt. Hope Bay Natural Laboratory Design

The MHBNL is envisioned as a total systems study of Mt. Hope Bay. Its purpose is to explain and forecast the interactions among natural and anthropogenic and internal and external dynamics on the Mt. Hope Bay ecosystem. The resulting program will provide scientists, resource managers, and the public with a better understanding of how Mt. Hope Bay and similar estuarine ecosystems function and how they can be managed in the face of competing multiusers.

The Mt. Hope Bay Natural Laboratory will be constructed in a modular format. Currently, we anticipate the modeling to include at least 6 interacting modules: 1) physical-chemical-geological environment (including habitat), 2) phytoplankton, 3) zooplankton, 4) benthos, 5) decapod crustaceans and 6) fish. Modeling of these modules will commence once sufficient observational data has

been obtained for proper parameterization. Efforts will initially emphasize nowcasting, then move into hindcasting of historical trends, and culminate in forecasting and scenario testing. The next report will provide a comprehensive description of the Mt. Hope Bay Natural Laboratory design.

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Appendix A. ASA's WQMAP

WQMAP (Water Quality Mapping and Analysis Program) is a proprietary modeling system developed by Applied Science Associates, Inc. and the University of Rhode Island for water quality mapping and analysis. WQMAP is advertised as an integrated system for modeling the circulation and water quality of estuarine and coastal waters (Spaulding et al. 1999b). The system has a suite of integrated environmental models, including a boundary-conforming grid generation model, a three-dimensional hydrodynamic model, and a set of pollutant transport and fate models (single- and multiple-constituent and WASP5 eutrophication kinetics). All operate on a boundary-conforming grid system and are supported by an embedded geographic information system and environmental data management tools. WQMAP runs on a Windows-based personal computer. Color graphics and animation are used to display model prediction. The system is structured to facilitate application to any geographic area.

WQMAP's hydrodynamic model solves the three-dimensional conservation of water mass, momentum, salt and energy equations on a spherical, non-orthogonal, boundary conforming grid. Relevant equations are presented at the end of this section. The model may be applied to both fresh and salt water systems, and can simulate the effects of tide, river flow, air temperature; solar radiation and wind induced environmental forcing on circulation and water properties including temperature and salinity. Bottom stress is determined from a quadratic law. If sufficient resolution near the bottom boundary is provided, the drag coefficient C_D may be estimated so that velocities match the logarithmic law

of the wall. In those cases where bottom boundary layer is not well resolved, C_D may be specified as a constant, typically between 0.002-0.003, or it may be calibrated based on observations. An environmental heat transfer sub-model at the water surface contains an explicit balance of short-wave solar radiation, long-wave atmospheric radiation, long-wave radiation emitted from the water surface, convective (sensible) heat transfer and evaporative (latent) heat transfer between water and air.

There are three separate models within the WQMAP pollutant transport model system, all using the same grid as the hydrodynamic model. The first is a single constituent transport model, which includes first order reaction terms. This model is suitable for a single constituent contaminant that settles or decays, like suspended sediment and fecal coliform bacteria. The second is a multi-constituent transport and fate model with a reaction matrix that can be specified by the user. This can be used to custom design a multi-component water quality model system. The third is a multi-constituent eutrophication model (e.g. nitrogen, phosphorous, dissolved oxygen) that incorporates EPA WASP5 kinetic rate equations. The user can set the parameters of the rate equations via the user interface or select default values.

WQMAP Governing Equations

The WQMAP solves the following set of equations, which govern the conservation of mass, momentum, salt, energy, dissolved constituents, turbulent

kinetic energy, and turbulent dissipation. Implicit here are the assumptions of hydrostatics and the Boussinesq approximation.

By employing the following operators:

$$\psi_{x'} = \frac{\partial \psi}{\partial x'}, \psi_{y'} = \frac{\partial \psi}{\partial y'}, \psi_{\gamma} = \frac{\partial \psi}{\partial \gamma}, \psi_{t'} = \frac{\partial \psi}{\partial t'}$$

(1)

differential operators in the (x, y, z, t) system can be expressed in terms of derivatives in the (x', y', γ, t') by

$$\frac{\partial}{\partial x} = \frac{\partial}{\partial x'} - \frac{\partial \psi_{x'}}{\partial \gamma} \frac{\partial}{\partial \psi_{\gamma}}, \frac{\partial}{\partial y} = \frac{\partial}{\partial y'} - \frac{\partial \psi_{y'}}{\partial \gamma} \frac{\partial}{\partial \psi_{\gamma}}, \frac{\partial}{\partial z} = \frac{\partial}{\partial \gamma} \frac{1}{\psi_{\gamma}}, \frac{\partial}{\partial t} = \frac{\partial}{\partial t'} - \frac{\partial \psi_{t'}}{\partial \psi_{\gamma}}$$

(2)

conservation equations in (x', y', γ, t') can be written as follow (all primes will be fopped for notational convenience).

Conservation of water mass:

$$\frac{\partial \psi_{\gamma}}{\partial t} + \frac{\partial u \psi_{\gamma}}{\partial x} + \frac{\partial v \psi_{\gamma}}{\partial y} + \frac{\partial \omega \psi_{\gamma}}{\partial \gamma} = 0$$

(3)

conservation of momentum - x and y

$$\begin{aligned} \frac{\partial u \psi_{\gamma}}{\partial t} + \frac{\partial u^2 \psi_{\gamma}}{\partial x} + \frac{\partial uv \psi_{\gamma}}{\partial y} + \frac{\partial u \omega \psi_{\gamma}}{\partial \gamma} - \psi_{\gamma} f_x \\ = \frac{\partial}{\partial \gamma} \left(\frac{K_x}{\psi_{\gamma}} \frac{\partial u}{\partial \gamma} \right) - \frac{\psi_{\gamma}}{\rho_0} \left[\rho g \frac{\partial \eta}{\partial x} - g q D \frac{\partial \bar{\rho}_{\gamma}}{\partial x} + (\rho - \bar{\rho}_{\gamma}) g \frac{\partial q D}{\partial x} \right] + \psi_{\gamma} F_x \end{aligned}$$

(4)

$$\frac{\partial v\psi_\gamma}{\partial t} + \frac{\partial uv\psi_\gamma}{\partial x} + \frac{\partial v^2\psi_\gamma}{\partial y} + \frac{\partial v\omega\psi_\gamma}{\partial \gamma} + \psi_\gamma fu$$

$$= \frac{\partial}{\partial \gamma} \left(\frac{K_v}{\psi_\gamma} \frac{\partial v}{\partial \gamma} \right) - \frac{\psi_\gamma}{\rho_0} \left[\rho g \frac{\partial \eta}{\partial y} - gqD \frac{\partial \bar{\rho}_\gamma}{\partial y} + (\rho - \bar{\rho}_\gamma)g \frac{\partial qD}{\partial y} \right] + \psi_\gamma F_v$$

(5)

conservation of salt and energy

$$\frac{\partial S\psi_\gamma}{\partial t} + \frac{\partial uS\psi_\gamma}{\partial x} + \frac{\partial vS\psi_\gamma}{\partial y} + \frac{\partial \omega S\psi_\gamma}{\partial \gamma} = \frac{\partial}{\partial \gamma} \left(\frac{N_v}{\psi_\gamma} \frac{\partial S}{\partial \gamma} \right) + \psi_\gamma F_s$$

(6)

$$\frac{\partial T\psi_\gamma}{\partial t} + \frac{\partial uT\psi_\gamma}{\partial x} + \frac{\partial vT\psi_\gamma}{\partial y} + \frac{\partial \omega T\psi_\gamma}{\partial \gamma} = \frac{\partial}{\partial \gamma} \left(\frac{N_v}{\psi_\gamma} \frac{\partial T}{\partial \gamma} \right) + \psi_\gamma F_T$$

(7)

conservation of pollutant constituent

$$\frac{\partial C\psi_\gamma}{\partial t} + \frac{\partial uC\psi_\gamma}{\partial x} + \frac{\partial vC\psi_\gamma}{\partial y} + \frac{\partial \omega C\psi_\gamma}{\partial \gamma} + \frac{\partial}{\partial \gamma} \left(\frac{N_v}{\psi_\gamma} \frac{\partial C}{\partial \gamma} \right) + \psi_\gamma (F_C + S + RC)$$

(8)

κ - ε turbulent closure

$$\frac{\partial k\psi_\gamma}{\partial t} + \frac{\partial uk\psi_\gamma}{\partial x} + \frac{\partial vk\psi_\gamma}{\partial y} + \frac{\partial \omega k\psi_\gamma}{\partial \gamma} = \frac{\partial}{\partial \gamma} \left(\frac{K_v}{\psi_\gamma \sigma_k} \frac{\partial k}{\partial \gamma} \right) + \psi_\gamma (P + G) - \varepsilon\psi_\gamma + \psi_\gamma F_k$$

(9)

$$\frac{\partial \varepsilon\psi_\gamma}{\partial t} + \frac{\partial u\varepsilon\psi_\gamma}{\partial x} + \frac{\partial v\varepsilon\psi_\gamma}{\partial y} + \frac{\partial \omega\varepsilon\psi_\gamma}{\partial \gamma} = \frac{\partial}{\partial \gamma} \left(\frac{K_v}{\psi_\gamma \sigma_\varepsilon} \frac{\partial \varepsilon}{\partial \gamma} \right) + \psi_\gamma \left[C_1 \frac{\varepsilon}{k} (P + C_3 G) - C_2 \frac{\varepsilon^2}{k} \right] + \psi_\gamma F_\varepsilon$$

(10)

where

$$q = \gamma - a \frac{D}{D_r} \sin 2\pi\gamma$$

(11)

and ρ_γ is defined as a vertically averaged density at a depth of γ

$$\bar{\rho}_\gamma = \frac{1}{\eta - z} \int_z^\eta \rho dz = -\frac{1}{q} \int_\gamma^\eta \rho \frac{\partial q}{\partial \gamma} d\gamma$$

(12)

t = time; S = salinity; T = temperature; C represents concentration of a dissolved constituent; ρ = density; ρ_0 = mean density; g = gravity; (u, v) = components of velocity in the (x, y) directions, respectively; (K_v, N_v) = vertical eddy viscosity and diffusivity, respectively; f = Coriolis parameter ($2\Omega \sin \theta$), in which Ω is the earth's rotation rate and θ is the latitude; P is turbulent production; G is the buoyancy generation; \mathbf{R} is a reaction matrix; and \mathbf{S} represents sources and sinks of constituent material which are detailed in Thomann and Mueller (1988). The following recommended empirical constants (Laulder and Spalding 1974; Rodi 1980) are adopted: $C_\mu = 0.09$, $\sigma_t = 0.7$, $\sigma_k = 1.0$, $\sigma_\varepsilon = 1.3$, $C_{1\varepsilon} = 1.43$, $C_{2\varepsilon} = 1.92$, $C_{3\varepsilon} = 0.2$.

The vertical eddy viscosity and diffusivity are expressed as

$$K_v = f_1 C_\mu \frac{k^2}{\varepsilon}; \quad N_v = f_2 \frac{K_v}{\sigma_t}$$

(13)

where f_1 and f_2 = empirical functions (Munk and Anderson 1948) dependent on Richardson number R to account for the buoyant effect

$$f_1 = (1 + 10R)^{-0.5}; \quad f_2 = (1 + 3.33R)^{-1.5}; \quad R = -\frac{g}{\rho} \frac{\frac{\partial \rho}{\partial z}}{\left(\frac{\partial u}{\partial z}\right)^2 + \left(\frac{\partial v}{\partial z}\right)^2}$$

(14)

the horizontal diffusion terms (Mellor and Blumberg 1985) are expressed as

$$\psi_\gamma F_x = \frac{\partial}{\partial x} \left(2\psi_\gamma K_h \frac{\partial u}{\partial x} \right) + \frac{\partial}{\partial y} \left[\psi_\gamma K_h \left(\frac{\partial v}{\partial x} + \frac{\partial u}{\partial y} \right) \right]$$

(15)

$$\psi_\gamma F_y = \frac{\partial}{\partial x} \left[\left(\psi_\gamma K_h \left(\frac{\partial u}{\partial y} + \frac{\partial v}{\partial x} \right) \right) \right] + \frac{\partial}{\partial y} \left(2\psi_\gamma K_h \frac{\partial v}{\partial y} \right); \quad \psi_\gamma F_{q_i} = \frac{\partial}{\partial x} \left(\psi_\gamma K_h \frac{\partial q_i}{\partial x} \right) + \frac{\partial}{\partial y} \left(\psi_\gamma K_h \frac{\partial q_i}{\partial y} \right)$$

(16)

where q_i represents S, T, C, κ , and ε ; and (K_h, N_h) = horizontal eddy viscosity and diffusivity, respectively.

An equation of state for seawater relating salinity and temperature to density (Eckert 1958) closes the system. The vertical velocity ω in γ space is defined as

$$\omega = \frac{1}{\psi_\gamma} [w - \psi'_x u - \psi'_y v]$$

(17)

ω can be determined by solving the conservation equation of water mass in the transformed γ space, (3). The vertical velocity w in (x, y, z) space is then calculated from (17). Vertical velocity and eddy viscosity/diffusivity are specified in the layer interface while other flow quantities (S, T, C, κ , ε) are located in the center of each layer.

Boundary conditions:

At the free surface, the boundary conditions are as follows:

$$\omega(x', y', 0, t') = 0; \quad [\tau_{wx}, \tau_{wy}] = \rho_a C_d [u_w, v_w] [u_w^2 + v_w^2]^{1/2} \quad (18)$$

$$\frac{\rho_0 N_v}{\psi_\gamma} \left(\frac{\partial T}{\partial \gamma}, \frac{\partial S}{\partial \gamma}, \frac{\partial C}{\partial \gamma} \right) = (T, 0, 0); \quad \frac{\partial(\kappa, \varepsilon)}{\partial \gamma} = 0 \quad (19)$$

where (τ_{wx}, τ_{wy}) = wind stress components in the x- and y- directions, respectively; (u_w, v_w) = wind velocity components in the x- and y- direction, respectively; ρ_{air} = air density; C_d = air-water drag coefficient; and T = net ocean surface heat flux. Eq. (19) is used when no wind stress is applied. Otherwise, the surface turbulent kinetic energy κ_s and dissipation ε_s are estimated by the following relations (Rodi 1980; ASCE Task Committee 1988)

$$\kappa_s = \frac{u_*^2}{\sqrt{C_\mu}}; \quad \varepsilon_s = \frac{|u_*|^3}{\kappa \Delta z} \quad (20)$$

where u_* = surface velocity shear; Δz = distance to the surface at the first grid point below surface, as suggested by Raithby et al.(1988).

At all solid boundaries, no normal flux of momentum, salt, energy, and dissolved constituent is allowed. The bottom stress is determined from a quadratic law similar to that at the surface and is given by

$$[\tau_{bx}, \tau_{by}] = \rho_0 C_f [u_b, v_b] [u_b^2 + v_b^2]^{1/2} \quad (21)$$

where (τ_{bx}, τ_{by}) = bottom stress components in the x- and y-directions, respectively; u_b, v_b = bottom velocity components in the x- and y-directions, respectively; ρ_0 = mean water density; and C_f = empirical coefficient for bottom friction. if sufficient resolution near the bottom boundary is provided, C_d was estimated so that velocities match the logarithmic law of the wall

$$C_d = \left[\frac{1}{\kappa} \ln(\Delta z_b)/z_0 \right]^{-2}$$

(22)

where the karman constant $\kappa = 0.42$; Δz_b = distance from the bottom in the grid point nearest the bottom; and z_0 = parameter dependent on the local bottom roughness and set to 1 cm (Blumberg and Mellor 1987).

The bottom turbulent kinetic energy κ_b and dissipation ε_b are determined from relations in the form of (20) with the bottom velocity shear u_* and the distance $\Delta z = \Delta z_0$ at the grid point nearest the bottom.

At inflow boundaries, κ and ε can be prescribed from fully developed channel flow data (Demuren and Rodi 1983) as

$$k_d = 0.004u_d^2; \varepsilon_d = C_\mu^{3/4} \frac{k_d^{3/2}}{0.09b} \quad (23)$$

where u_d = inflow velocity; and b = inflow inlet width.

At open boundaries, the surface elevation or mass flux is required as are fluxes of salt, energy, dissolved constituent in to the system. On outflow simplified conservation equations, using only the advection terms, are used to transport salt, heat, turbulent kinetic energy and dissipation

$$\frac{\partial(S, T, C, k, \varepsilon)}{\partial t} + V_n \frac{\partial(S, T, C, k, \varepsilon)}{\partial x_n} = 0 \quad (24)$$

where V_n = velocity normal to the open boundary.

Appendix B. Selected Web Pages Related to MHB

1. NARRAGANSETT BAY PORTS DATA: (CONDUCTIVITY, SALINITY, SPECIFIC GRAVITY, WATER TEMPERATURE)

http://ports-infohub.nos.noaa.gov/ports/hubs/narragansett_bay/index.shtml

2. NARRAGANSETT BAY PORTS: FALL RIVER COMPOSITE

- a) Fall River water levels – observation and prediction
- b) Fall River winds
- c) Fall River currents – along and cross channel profiles
- d) Fall River currents – observed along and cross channel time series
- e) Fall River currents – predicted along and cross channel time series

http://www.co-ops.nos.noaa.gov/nbports/Composite_fr.html

3. NARRAGANSETT BAY / MOUNT HOPE WATERSHED

The Mount Hope Bay Shores and Narragansett Bay Watershed is located in southwestern Massachusetts and a small portion of eastern Rhode Island. The watershed drains into Mount Hope Bay, which is located at the mouth of the Taunton River, and five smaller rivers. Proceeding in a westerly direction from Mt. Hope Bay, the five rivers include the Lees, Cole, Kickamuit, Palmer and Runnins River, which generally flow in a southern direction through Rhode Island and empty into to Narragansett Bay.

<http://www.state.ma.us/envir/mwi/mthopenarragansett.htm>

4. REMOTE SENSING OF NARRAGANSETT BAY

This is an investigation of the dynamics of Narragansett Bay using remote sensing technology in a manner that has not been possible with more traditional methods. Multispectral satellite imagery offers a truly synoptic view of various water properties over the entire estuary and can be used to understand the spatial and temporal interaction of both physical and biological characteristics of the bay. This project is unique in its direct application of high technology to immediate and local environmental concerns.

<http://www.planetary.brown.edu/~mustard/apurva/>

5. NARRAGANSETT BAY ESTUARINE PROGRAM

- a) to prevent further degradation and incrementally improve water quality in developing coastal areas with deteriorating water quality;
- b) to protect diminishing high quality resource areas throughout the Bay watershed;
- c) to more effectively manage commercially, recreationally, and ecologically important estuarine-dependent living resources;
- d) to rehabilitate degraded waters in the Bay watershed and restore water quality-dependent uses of Narragansett Bay;
- e) to establish necessary interstate and interagency agreements and mechanisms to coordinate and oversee implementation of the

Narragansett Bay Comprehensive Conservation and Management
Plan.

<http://www.nbep.org/index.html>

6. SATELLITE IMAGES OF MOUNT HOPE BAY/MASSACHUSETTS
COAST

<http://coast.mit.edu>

7. NARRAGANSETT BAY DRAINAGE BASIN

<http://ma.water.usgs.gov/basins/narrag.htm>

8. FACTS ON 5 POWER STATIONS EMISSIONS:

Salem Harbor Power Plant

Mount Tom Power Plant

Brayton Point Power Plant

Canal Station Power Plant

Mystic Power Plant

<http://www.pirg.org/masspirg/enviro/cleanair/plants.htm>

9. ISSUES FACING NARRAGANSETT BAY

a) Narragansett Bay Habitat Restoration

b) Recent Restoration Activities

c) Narragansett Bay Restoration Projects:

- d) Eelgrass
- e) Salt Marshes
- f) Fish Runs
- g) Water Quality
- h) Narragansett Bay Method
- i) National Restoration Efforts

<http://www.savebay.org/bayissues/>

10. NARRAGANSETT BAY PROJECT

This project is part of a larger comprehensive study of Narragansett Bay being managed jointly by NMFS and RIDEM. The comprehensive study includes enhanced assessments of Narragansett Bay fish and fisheries, a survey of sediment pollution in Narragansett Bay, a study of Bay currents, and a network of fixed-site samplers recording a continuous record of hydrographic data.

<http://www.na.nmfs.gov/nargbayproj.html>

11. UNIVERSITY OF RHODE ISLAND SEDIMENT CHEMISTRY STUDY

View of the data table description page

http://www.state.ma.us/czm/University_of_Rhode_island.htm

12. MASSACHUSETTS RESOURCE IDENTIFICATION PROJECT

(MRIP)

MRIP is a component of the EPA Region I. Resource Protection Project that specifically focuses on the identification and protection of Massachusetts' most important natural resources. It was designed to help agencies and organizations jointly target places for attention and make the most of their limited time and money. MRIP is a joint project of MassGIS and the U.S. Environmental Protection Agency (EPA).

<http://www.state.ma.us/mgis/mrip.htm>

13. MASSACHUSETTS SHORELINE CHANGE ANALYSIS INDEX

Applied Geographics, Inc. (AGI) performed a vector-based historic shoreline change analysis using Arc/Info vector coverages, AML, and C. Linear historic shoreline data as early as 1844 and as recent as 1982 were provided and an analysis was undertaken to define and execute a procedure for deriving the historic rate of shoreline change using a vector-based methodology. Programs written in C were modified to handle the complexities of the Massachusetts historic shoreline data.

http://www.appgeo.com/atlas/project_source/czmcc/methods/p350.htm

14. HISTORIC USGS MAPS OF NEW ENGLAND

<http://docs.unh.edu/nhtopos/nhtopos.htm>

15. STATE OF RHODE ISLAND DEPARTMENT OF ENVIRONMENTAL
MANAGEMENT

This site will give you information on Parks, Coastal Resources, Fish and Wildlife, and Forest Environment. You will also find links to the Geo-Data Viewer, an exciting new tool for the delivery of interactive maps over the internet.

<http://www.state.ri.us/dem/maps/index.htm>

16. EMPACT PROJECT FOR NARRAGANSETT BAY / MOUNT HOPE
BAY, PROVIDENCE, RHODE ISLAND / FALL RIVER,
MASSACHUSETTS

Objective: This project will provide the public with timely information about water quality in the Narragansett Bay and its tributaries and the effects of natural and man-made influences on the Bay. Approach: The project will monitor water quality parameters and develop a web site to deliver data to the public in addition to using existing communications and outreach programs to disseminate the data through public displays, teacher training, and programs with community groups.

http://es.epa.gov/ncer_abstracts/grants/99/envmon/pavignano.html

17. THE LIVING BAY - TOPOGRAPHY AND GEOLOGY

Narragansett Bay: An individual's perspective

http://www.providenceri.com/NarragansettBay/the_living_bay.html

18. SAMPLING THE BAY OVER THE LONG TERM

This article emphasizes the importance of long-term fish trawl recording of marine life in Narragansett Bay.

http://www.gso.uri.edu/maritimes/Text_Only/00Winter/text/taylor.htm

19. COASTAL 2000 - COMPREHENSIVE MARINE MONITORING
PROGRAM SAMPLING SCHEMATIC

This summer, as part of the U.S. Environmental Protection Agency's (EPA) Coastal 2000 Monitoring effort, Massachusetts began one of the most comprehensive assessments of its coastal waters and sediments ever undertaken. The following schematic shows the Coastal 2000 sampling sites. <http://www.state.ma.us/czm/coastal2k.htm>