

# Quantitative Models Describing Past and Current Nutrient Fluxes and Associated Ecosystem Level Responses in the Narragansett Bay Ecosystem



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## **ABSTRACT**

Multiple drivers, including nutrient loading and climate change, affect the Narragansett Bay ecosystem in Rhode Island/Massachusetts, USA. Managers are interested in understanding the timing and magnitude of these effects, and ecosystem responses to restoration actions. To provide scientific support for nutrient management in Narragansett Bay, we developed a set of models based on past and current data for hydrodynamics, water quality, and ecology, calibrated to current conditions. We used linked mechanistic hydrodynamic and water quality models (EFDC/WASP), developed empirical approaches to quantitatively relate nutrient and dissolved oxygen concentrations to seagrass habitat (both bio-optical and seagrass predictive models), benthic macroinvertebrates, and fish; and applied a general model (the ECOPATH/ECOSIM model) for the foodweb. Our integrated set of models was generally able to reproduce observed data for the estuary and characterize scientifically valid patterns. Current modeling challenges include limited data availability for water quality and selected inputs, variability associated with invertebrate and fish populations, and issues of scaling for model linkage. We assessed the feasibility of using modeling for decision-making in the context of integrated nutrient management, with a goal of transferability to other Northeastern estuaries. While the models seem useful to answer questions related to Narragansett Bay, some aspects of this approach (i.e., the hydrodynamics and seagrass models) are more transferrable than others. Future directions include the testing of scenarios (e.g., land use change, habitat restoration, marine spatial planning), development of modeling approaches for zooplankton and shellfish, and the incorporation of economics.

## 1. INTRODUCTION

Eutrophication and associated ecological effects are increasing concerns in coastal environments. Estuarine eutrophication is often, but not always, linked to nutrient enrichment (Bricker et al. 2007), and in coastal marine environments, except in the tidal fresh portions of estuaries, nitrogen is most often the limiting nutrient and a major contributor to eutrophication (Oviatt and Gold 2005). Large increases in nitrogen flux delivered to ground or surface waters through direct runoff, human and animal wastes, and atmospheric deposition can favor rapidly-growing producers such as algae, leading to undesirable changes in aquatic resources, including reduced water clarity, seagrass decline, high organic input to sediments, hypoxia, nonlethal adverse effects on fish and benthos, fish kills, loss of biodiversity, and increases in nuisance species (Cloern 2001, deJonge et al. 2002, Nielsen et al. 2004, Bricker et al. 2007). Declines in fluxes of phosphorous through major New England rivers from 1965 thru 2000 have been documented by USGS (2003), but phosphorous loading remains an important stressor in surface fresh water resources. Eutrophication in surface, estuarine, and marine waters may reduce dissolved oxygen below the levels needed to meet Clean Water Act designated uses (e.g., the maintenance of waters that are swimmable, fishable, and drinkable). It may also affect human well-being through increased exposure to harmful algal toxins, drinking water nitrates and the reduction in economic opportunities throughout the region. An excellent summary of contemporary nutrient loading to and through Eastern U.S. watersheds is provided in USGS (2013), summarizing rates of nitrogen and phosphorous delivery to the coast, and estimates of the percentages of sources contributing to these river fluxes. These summaries do not include the additional anthropogenic nutrient loads from Wastewater Treatment Facilities discharging directly into estuaries, where tides result in bi-directional flows.

Management efforts include reduction of the stressor, as well as restoration (Elliott et al. 2007). The management of excess nutrients remains a complex challenge, and in the case of nitrogen should be considered in a multi-media (air, land, and water), and multi scale interventions approaches (Compton et al. 2011). Also, eutrophication problems in watersheds are embedded in changing aspects of land use and development, hence opportunities to improve water quality, may relate to the restoration of natural flows of water, restoration and protection of wetlands and riparian zones, in addition to further reductions in point and non-point sources of nutrient loading. In the achievement of cost effective, socially acceptable, and environmentally sound solutions, watersheds and estuaries can benefit from coordinated and intentional use of Federal, State/Tribal, and municipal authorities and capacities, and a transdisciplinary research and management approach for holistic problem-solving.

Kemp and Boyton (2012) note that synthesis studies, such as biophysical simulation modeling for specific systems, are important for addressing complex ecological problems in estuaries. The need to couple hydrodynamic and ecological models to address research and management questions in estuaries is clear because dynamic feedbacks between biotic and physical processes are important within estuarine systems (Ganju et al. 2015). Ecological models provide advances for environmental decision-making, by allowing managers to consider the effects of alternative solutions (Schmolke et al. 2010, Rose and Allen 2013). In particular, end-to-end modeling has the potential to combine models for physical systems through higher trophic levels (Rose et al. 2010). In the US, integrated estuarine modeling has a long history of use in Chesapeake Bay (e.g., Cerco and Noel 2013). Also within the

Northeastern region, there have been significant modeling efforts in Long Island Sound, using their System Wide Eutrophication Model (O'Donnell et al. 2014). A continuing challenge relates to the collection of sufficient observational data to characterize environmental variability and change, both for calibration of key model parameters, and for understanding key processes affecting fate, transport and effects of nutrients. Also, as model complexity and the number of model parameters increase, there is a risk that related increases in model skill assessments may result from curve fitting, rather than from additional model skill needed to characterize systems dynamics, such as inter-annual variability and expected longer term change in the magnitude and frequency of hypoxia. Ganju et al. (2015) provide a recent review of estuarine modeling approaches and issues.

Here, we assess the feasibility of using modeling to support decision-making in integrated nutrient management, primarily focusing on management of nitrogen, and on achieving more cost-effective, sustainable solutions for Narragansett Bay (NB). NB has experienced significant nutrient loading in the past; current management efforts are directed at reductions and restoration to reduce nutrient concentrations and ameliorate associated ecological effects. Modeling has been used to some extent in NB (Table 1). These modeling approaches form the basis of future work; however, we suggest that there is a need for a more integrated approach. The purpose of this effort was to apply and/or develop an integrated set of models to evaluate the effects of physical changes (e.g., drivers of change, disturbances, and enhancements), specifically related to nutrients, and how they affect ecosystem structure and function. Additionally, this task evaluates the efficacy of restoration actions. For example, the capacity and potential for biological restoration (e.g., shellfish) to ameliorate nutrient loads. Major objectives of this study are: 1) calibration of a hydrodynamic-water quality model for the bay; 2) development of stressor-response models for nutrients, 3) assessment of future scenarios, and 4) consideration of transferability to other estuaries and future modeling directions, with a goal of transferability to other Northeast estuaries. This report summarizes the progress made to date.

Table. 1. Previous modeling work in Narragansett Bay, RI/MA.

References	Model Description
<i>Hydrodynamics</i>	
Applied Science Associates, 2005	A 3-D model (BFHYDRO) with fine segments calibrated to surface elevation and currents data. They concluded that additional refinement was needed to resolve salinity and temperature.
Zhao et al. 2006	A Finite-Volume Community Ocean Model (FVCOM) model applied to Mt. Hope Bay and Narragansett Bay
Sankaranarayanan and Ward 2006	A 3D semi-implicit model, using orthogonal coordinates used to predict surface elevations, currents, salinities and temperatures.
Abdelrhman 2007	Simple tidal prism model applied at the scale of the bay, used to calculate the time for transport of constituents in embayments
Kremer et al. 2010	An approach to modeling estuarine hydrodynamics and water quality for the bay using Regional Ocean Modeling System (ROMS), and also using these predictions in a simplified way for a 30-element model of NB.

References	Model Description
<i>Water Quality</i>	
Kremer and Nixon 1978	A coarse spatial NB model using 8 geographic areas, with seasonal loading and daily computation of chemical and biological components
Lemerise 1994	Estuarine pelagic ecosystem model
Brawley et al. 2003, Brush and Brawley 2009 Applied Science Associates, 2005	Modeled phytoplankton primary production using a “light •biomass” model for computing phytoplankton productivity BFWASP water quality model was calibrated using model predicted currents from the calibrated BFHYDRO hydrodynamics model and water quality and nutrient data
Brush 2014,	EcoGEM modeling approach applied to 30 spatial elements in NB, used with Officer Box modeling approach for exchange among boxes
Vaudrey and Kremer 2015	EcoGEM modeling approach, applied to 30 spatial elements in NB used with hydrodynamics presented in Kremer et al. 2010 to generate a gross exchange matrix
Abderhman 2015b	EFDC water quality module application to NB, linked with EFDC hydrodynamics
<i>Ecology</i>	
Byron et al. 2011	ECOPATH, a steady-state mass balance model, used in NB to consider the role of oysters in the coastal foodweb
Meng et al. 2002	Developed an index of biotic integrity for fishes in Narragansett Bay, but in general, more impacted sites had higher scores, possibly indicating that the more impacted sites in the upper bay were also better juvenile habitat.
Fiksel et al. 2013	Triple Value Simulation model, a system dynamics approach with empirical relationships among stressors, responses, and links to social and economic factors

## 2. DESCRIPTION OF NARRAGANSETT BAY ECOSYSTEM

The NB watershed drains 4,836 square kilometers, and is located in the U.S. states of Rhode Island (RI, 40%) and Massachusetts (MA, 60%). The watershed encompasses approximately 100 cities and towns, with more than two million residents (Fig. 1). Providence, the capital of Rhode Island, is the largest city in the watershed, with a city population of approximately 177,000. The region was settled by Europeans in 1600s, and the primary use of the land in this area was agriculture, until industrialization, including textile and jewelry industries, and associated urbanization began in the 1900s. The textile industry peaked in the mid-1900s, as the cotton industry moved to the southern United States. The number of people working in the jewelry industry reached its peak during the 1970s (Nixon 1995). This watershed is more urban than other watersheds in the Northeastern U.S. that discharge into the North Atlantic, and NB supports a higher nitrogen loading than most of these estuaries (Latimer and Charpentier 2010). Currently, the watershed is about 37.7% urban land use. The largest tributaries are the Blackstone and Taunton Rivers (Fig. 1); wastewater treatment facilities (WWTF) discharging into the Seekonk and Providence River estuaries are point sources for most of the sewage effluents entering the bay (Nixon et al. 2008). River water inflow has a seasonal variability, with the highest flow in the spring and the minimum flow in early fall.

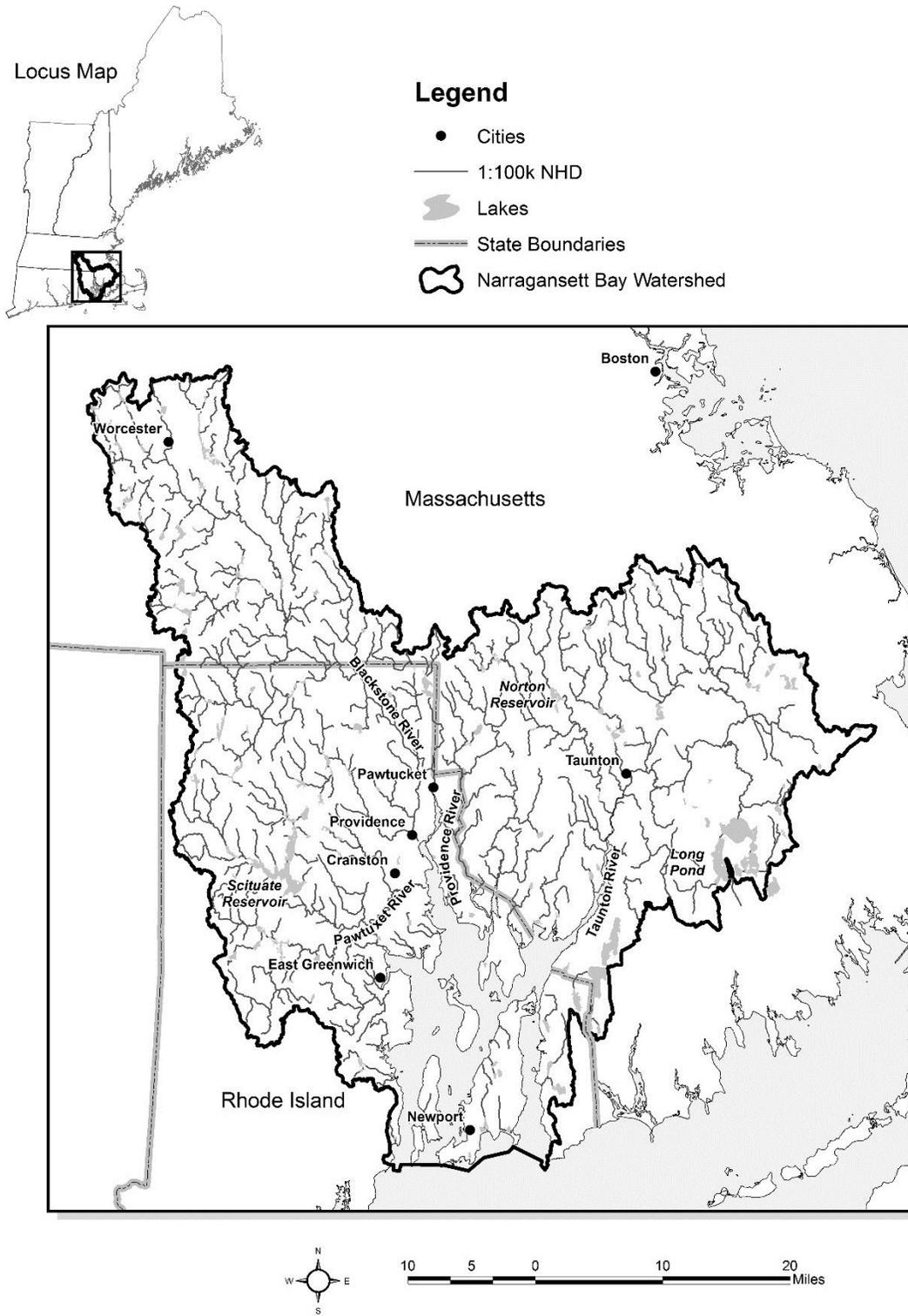


Figure 1. Location of Narragansett watershed and estuary, including cities, rivers and streams (from National Hydrography Database, NHD), lakes, and state boundaries.

The NB estuary covers 380 square kilometers, with 6% in Massachusetts. The estuary has three major islands (Conanicut, Prudence, and Aquidneck), that confine the flow into three outlets, the Sakonnet River, the East Passage, and the West Passage (Fig. 1). NB is relatively shallow, with mean depth of 8.0 m (Pilson 1985) and a maximum depth of 46 m in the east passage (Zhao et al. 2006). NB has semi-diurnal tides in the range of 0.9-1.2 m, and the residence time in Narragansett Bay ranges from 10 to 40 days with the average time being 26 days (Pilson 1985). Salinities range from about 24 ppt at the north end of the estuary to 32 ppt at the south end. Salt marshes cover a relatively small portion of the bay. The estuary provides valuable ecosystem services to the region through tourism, recreation, and commercial shellfishing. In NB, oysters (*Crassostrea virginica*) were harvested in the greatest quantities of all shellfish until 1940, when quahog (*Mercenaria mercenaria*) harvests exceeded that of the oyster. Although quahog harvests increased from 1940, current harvests are less than a quarter of 1980s harvests. A more detailed history of the region and its resources can be found in Desbonnet and Lee (1990), Desbonnet and Costa-Pierce (2008), Pastore (2011), and Pesch et al. (2012).

Nutrient loading is a major stressor affecting the NB ecosystem, with nitrogen (N) being the limiting nutrient, as in most estuarine and marine systems (Kremer and Nixon 1978, Oviatt et al. 1995). Ross (2014) considered the role of phosphorous becoming limiting, and concluded that under current and future scenarios, limitation by nitrogen will continue to be most significant. Vadeboncoeur et al. (2010) found that “total N loading to the bay increased 250% from 1850 to 2000, and 80% from 1900 to 2000.” Current concentrations of total nitrogen (TN) range from 0.2-1.0 mg/L in NB. NB has a distinct north-south gradient in nutrients (Oviatt 2008), reflecting the higher urbanization around the upper portions of the Bay. Point sources of nitrogen from suburban and urban environments are the dominant source of nutrient loading to the estuary (Deacutis 2008, and references therein), and hypoxia is more severe after higher river flows (Codiga et al. 2009) and depends on stratification (Bergondo et al. 2005). There are 32 sewage treatment facilities that discharge waste water into the Narragansett Bay basin – 22 of these discharge to rivers and streams, one discharges into Mt. Hope Bay, and the other 9 discharge into the Seekonk-Providence River estuary or Narragansett Bay itself (Nixon et al. 1995). An assessment of the mass balance of N suggests that the three main plants around Providence (Fields Point, Bucklin and East Providence) account for about 80% of the sewage that is discharged directly into Narragansett Bay (Nixon et al. 1995). Rivers and the WWTF that discharge into them are a major pathway for N, with 70% of the N entering the bay in the form of DIN (partitioned between  $\text{NH}_4$  and oxidized forms), with DON accounting for about 20% (Nixon et al. 1995). The transport of dissolved N in groundwater has been historically difficult to monitor and predict (Nowicki and Gold 2008). While N inputs from groundwater may be an important source of N in some of the smaller coves and embayments in the bay (Carey et al. 2005), research suggests that the volume entering the bay is not large, probably less than 10% of the freshwater input (Nixon et al. 2008). Atmospheric deposition of N contributes a relatively small fraction of the total anthropogenic N load to the surface of the bay, roughly 10-15% (Hamburg et al. 2008).

Deacutis (2008) provides an excellent review of the ecological impacts of excess nutrients in the system. Hypoxia occurred in 2001 and 2002 (Deacutis et al. 2006) and a mussel (*Mytilus edulis*) die-off event occurred during summer 2001 (Altieri and Witman 2006). Codiga et al. (2009) note the following effects of hypoxia in Narragansett Bay: “highest direct mortality risk is to sensitive and moderately sensitive

sessile species in the northern West Passage and western Greenwich Bay, with some risk to the Upper Bay; direct risk to mobile species may be ameliorated by weak spatial synchronicity; and indirect impacts, including reduced growth rates and shifts in predator–prey balances, are very likely throughout the sampled area due to observed suboxic and hypoxic conditions.” It is very clear that seagrass was abundant in appropriate shallow soft-bottom habitats throughout the Bay in the late 1800s based on anecdotal information and earliest surveys, growing well at certain spots even up through the Providence and Seekonk Rivers. The first dramatic decline in seagrass was due to Wasting Disease in the 1930s – 40s and the Hurricane of 1938, which together wiped out many beds. A fair amount of seagrass remained in the Bay until the early 1970s, after which many beds were lost, mostly likely due to nutrient pollution (Latimer and Rego 2010). Many seagrass beds are currently returning to NB (Chintala et al., 2015). Excess nutrients may possibly also support the growth of harmful algae blooms in NB (Borkman et al. 2012).

Additional stressors affect NB, including pathogens, contaminants, and climate change. Pathogens from WWTF are an issue for shellfish, water supply, and recreation (Desbonnet and Lee 1990), this is being addressed through the use of Total Maximum Daily Load analysis (e.g., RIDEM 2009, Massachusetts DEP 2010). Organic and metal contaminants have been detected in NB, with higher concentrations in the upper bay near industrial and residential areas (Desbonnet and Lee 1990, Calabretta and Oviatt 2008, Taylor et al. 2012). These contaminants have limited impacts on aquatic life, but can limit restoration efforts due to the possibility of resuspension from sediments (Kutcher 2009). Climate change affects the bay, not only through increased temperature and altered runoff, but also altered patterns for wind, clouds, and circulation. NB has experienced a 0.94°C increase in mean air temperature between 1905 and 2006, sea surface temperatures have warmed 2.2°C since the 1960s; there is a 3mm/year increase in precipitation since 1900; a 20 percent decrease in average wind speed since 1950; and an increase of about 60 days a year in cloud-covered days (CRMC 2010). Studies on NB have shown several climate associated patterns: a change in the timing and size of spring diatom blooms (Oviatt 2004, Smith et al. 2010); reduction in biochemical exchange with sediments (Fulweiler and Nixon 2009, 2011, Hyland et al. 2005); an increase of jellyfish, due to overwintering, and associated increased predation on fish larvae (Sullivan et al. 2008); a decrease in summer zooplankton (Oviatt 2004); and a change in the timing and composition of the fish community in NB (Jeffries and Terceiro 1985, Collie et al. 2008). There is also a threat of acidification in coastal waters, including NB (e.g., Wallace et al., 2014). Acidification can occur when excess carbon dioxide either enters or is produced in coastal waters and lowers pH, which can affect the ability of shellfish and other calcifiers to form shells. The effects of acidification and eutrophication may be synergistic (e.g., Gobler et al. 2014).

The states of RI and MA, along with EPA Region 1 are committed to identifying and solving current priority water quality problems related to excess nutrients in this highly urbanized watershed. In 2003, a large fish kill in Greenwich Bay, a sub-embayment of NB, spurred the RI legislature to take action. The State of RI has stated a goal to reduce landside loading of N to NB by 50%, requiring a combination of control measures (air, land, water). RI and MA are working with EPA Region 1 to help achieve this ambitious near term goal. One response involves engineering retrofits (tertiary treatment) for point source generators. All of the major sewage treatment plants discharging to the estuary and associated

tributaries have been upgraded, or are in the process of upgrading, to tertiary treatment, which will reduce average effluent N concentrations from 16-20 mg/L to  $\leq 5$ mg/L. A tunnel to divert combined sewer overflows (CSOs) was completed in 2008 to capture stormwater and reroute it through wastewater treatment facilities (WWTFs). NB is not currently using a Total Maximum Daily Load approach; instead the RI Department of Environmental Management (RIDEM) is using adaptive management, and expects to develop numeric criteria for Narragansett Bay by 2020. The MA Bays Program provides some guidance for target total nitrogen concentration: 0.5 mg/L in Buzzards Bay to minimize hypoxia risk, and 0.4 mg/L to protect eelgrass, but allows for some consideration of additional site-specific factors. Guidance can vary depending on site specific considerations. Managers are interested in understanding the timing and magnitude of the effects of nutrients and other stressors, as well as ecosystem responses to restoration actions. In particular, there is interest in understanding the effects of management actions other than point source reductions (e.g., the potential for restoring shellfish to ameliorate nutrient loads), since future point source reductions are likely to be costly. Modeling approaches can be useful to understand the system response to alternative actions, which can guide management decisions.

### 3. MODEL DEVELOPMENT

The approach used here was the application of a set of linked models, summarized below. An overview of the flow of information among models is presented in Fig. 2.

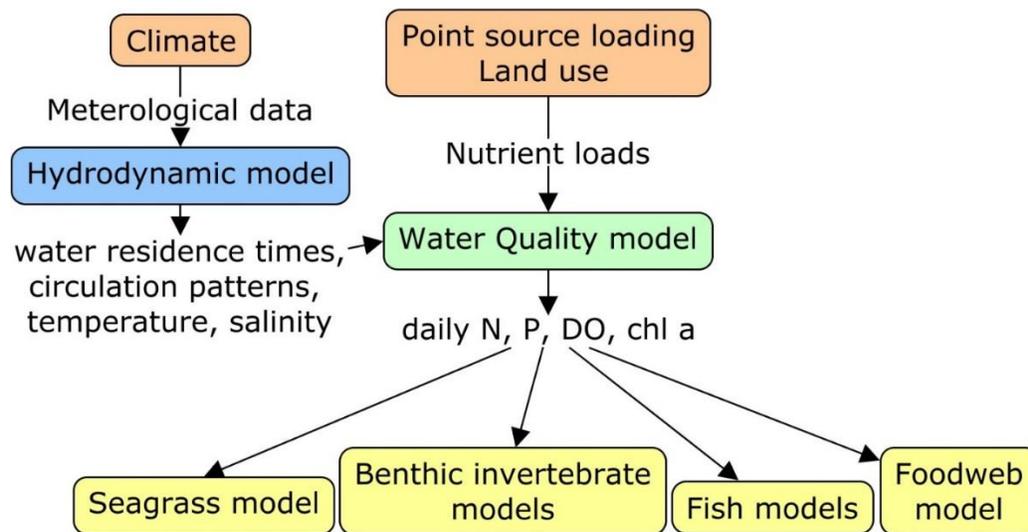


Figure 2. Schematic of information flow across models in the project.

- *Hydrodynamics (Section 3.1)* – A high-resolution hydrodynamic simulation of NB was modelled using the Environmental Fluid Dynamics Code (EFDC) model, which successfully replicated field observations of temperature and salinity in the bay. EFDC output served as input for the water quality model. While calibration and validation efforts in this approach are extensive, similar success can be expected when applying the model to other estuarine systems.
- *Water quality (Section 3.2)* – Distributions of water quality variables, including chlorophyll, dissolved oxygen, and N were modelled with moderate success using the mechanistic Water Quality Analysis Simulation Program (WASP) model, linked to EFDC. Problems encountered included limited availability of nutrient and solar radiation data, inadequate WASP documentation, and technical difficulties linking WASP and EFDC. Output from WASP scenarios can be linked to the ecology models for seagrass, benthic invertebrates, fish, and the foodweb.
- *Seagrass (Section 3.3)* – A fitted-parameter bio-optical model was developed to predict light availability vs depth in NB, based on water clarity parameters available from WASP. The understanding of light availability provides limitations on where seagrass communities may flourish in the bay. Additionally, regression models were developed based on biophysical parameters and locations of current seagrass communities. These models can be used to estimate how the seagrass coverage area may respond to reduction of TN loads. Output from the WASP model can potentially be linked to these seagrass models.
- *Benthic invertebrates (Section 3.4)* – Quantile regression models were used to examine relationships between eutrophication measures and benthic invertebrate community structure. The current distribution of invertebrate communities were found to respond to gradients within the bay. Additional analysis found significant dependence of community structure and function on extent of hypoxia. When available, WASP model simulations will allow examination of possible future benthic condition scenarios.
- *Fish (Section 3.5)* – Quantile regressions models were constructed to examine responses of species and community metrics to water quality measures. Significant relationships were detected with DO, chlorophyll, depth, eelgrass, and temperature, but inherent variability made it difficult to find strong, consistent patterns among species. Similar modelling elsewhere will require region-specific analyses, but the approach is transferable to other estuarine systems.
- *Foodweb dynamics (Section 3.6)* – The ECOPATH/ECOSIM (EwE) model was used to examine potential responses of food web trophic groups to changing N availability in NB. Clear responses were found; biomass of all groups diminished with decreasing N. EwE modelling should be readily transferable to other estuaries.

Details of these models and their applications are below. All of the research conducted for this report was covered by approved EPA Quality Assurance Plans and followed generally accepted good modeling practices (e.g., Jakeman et al. 2006, Laniak et al. 2013). None of the analyses conducted for this report required or involved primary data collection, either from environmental media or from human subjects. The modeling results discussed in this report were all generated using established software programs and freely available, open-source models – Microsoft Excel, R, EFDC, WASP, ECOPATH/ECOSIM.

### **3.1 Hydrodynamics**

The inherent hydrodynamic complexities in coastal and estuarine systems arise from salinity and temperature stratification, estuarine circulation, meteorological (e.g., wind) impacts, and astronomical (e.g., tide) forcing. These complexities require a 3-D model to provide adequate predictions of forces on and movements of water masses (with their associated constituents) in both space and time. The EPA-recommended Environmental Fluid Dynamics Code (EFDC, Hamrick 1996) was chosen for this project. EFDC is a 3-dimensional hydrodynamic, water quality, and ecological modeling system, which has been used for more than 80 modeling studies of rivers, lakes, estuaries, coastal regions and wetlands in the United States and abroad (USEPA: <http://www2.epa.gov/exposure-assessment-models/efdc-read-me-epa-version-10>.) Practical and realistic modeling of hydrodynamics is an important step towards proper modeling of ecological phenomena at the scale of whole estuaries because the hydrodynamics control the movement and dispersion of constituents important to ecology. The practicality of a hydrodynamic model lies in the reasonable spatial and temporal resolutions that dictate the ease and timely production of flow, and transport and dispersion of constituents. Results of the physical model are often checked against observations that are unaffected by biology in temperate systems (e.g., temperature, salinity). The generated hydrodynamics can be used by other models to study ecological and water quality dynamics within the Bay.

Abdelrhman (2015a) presents the methodology used to apply, calibrate, and validate a three-dimensional hydrodynamic model EFDC to the Narragansett Bay to generate sample predictions of hydrodynamics and transport for the year 2009; the final grid included 754 horizontal segments and 8 vertical layers, a total of 6,032 segments (Fig. 3). The typical segment size was 642 m (wide) and 1,218 m (long). The full duration of the majority of simulations was 365 days with a timestep of 120 s (two minutes). The major meteorological forces include precipitation, evaporation, solar radiation, wind speed and direction, air temperature, and atmospheric pressure. Inflow was represented from nine sub-watersheds, eight of which draining surface and ground water through gauged rivers; the ninth sub-watershed (498.03 km<sup>2</sup>) lies along the Bay's shore line, drains directly into the Bay, and is considered to be "riparian." Inflow water was also documented from all major wastewater treatment facilities (Fig. 4). The seaward open boundary of the Bay is forced by the semidiurnal tide, temperature, and salinity in the RI Sound. Predictions for temperature and salinity were compared to measured data at fixed buoy locations in NB (Codiga et al. 2009, Fig. 4), and to NOAA data for observations at two sites (PR and FR1, Fig. 4).

The EFDC model simulation was in good agreement with available field observations for water surface elevation, temperature, salinity, and velocity (Abdelrhman 2015a). Model predictions for salinity mimic observations throughout the year with an average difference < 0.5 ppt for surface waters and < 1.0 ppt for bottom waters. The scatter plots for the overall surface and bottom salinity show linear agreements with  $R^2 = 0.8$  and  $0.7$ , respectively. Water temperature is the most important parameter that influences most of the biological activities in the Bay. Thus, predicted and observed overall temperature behaviors have to be close. Overall predicted surface and bottom water temperatures are very close to observations throughout the year with an average difference < 0.5°C for surface waters and < 1.0°C for

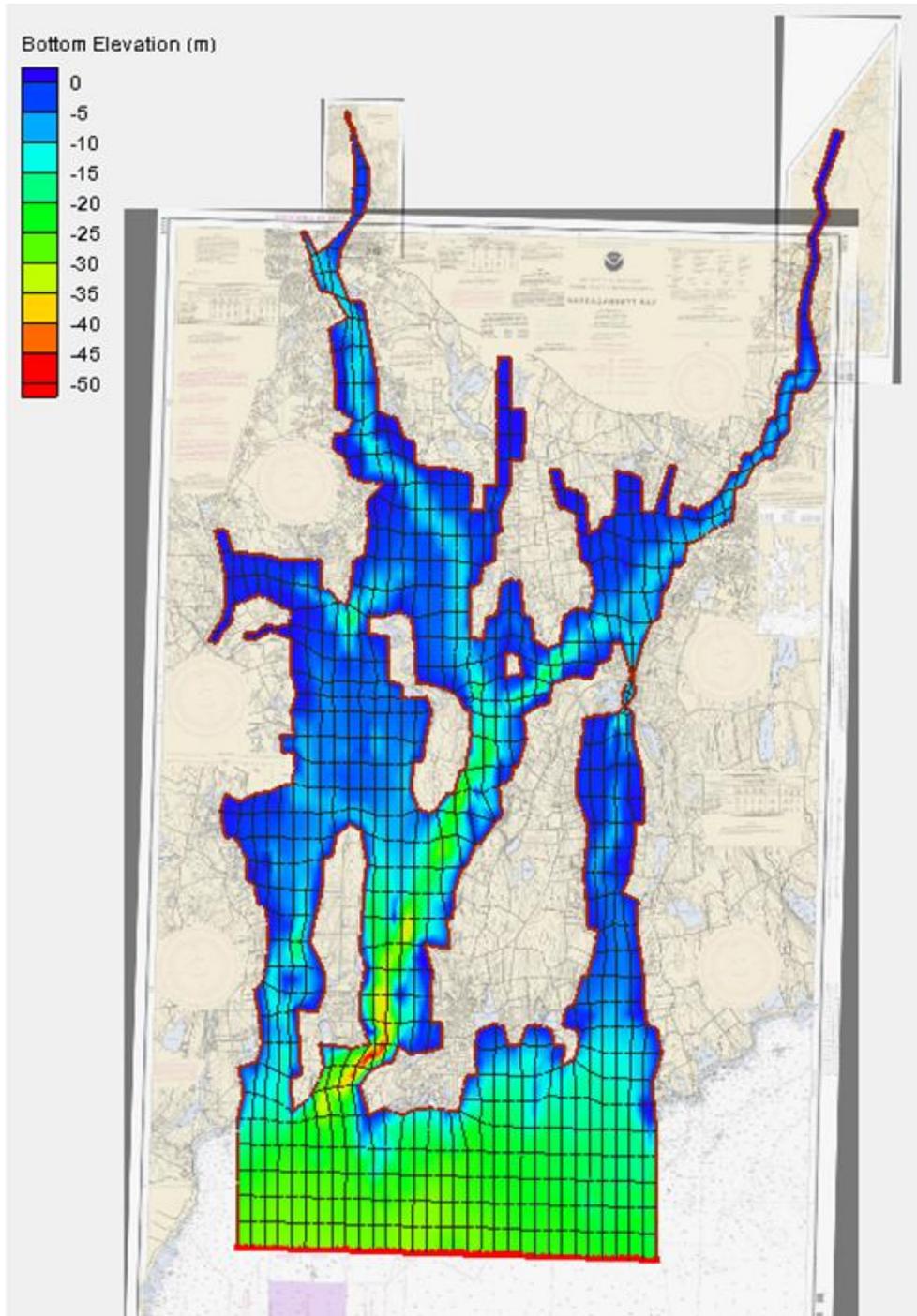


Figure 3. Environmental Fluid Dynamics Code (EFDC) numerical grid and bottom elevation (m) for Narragansett Bay. Segment size is variable -- Typical segment size: 640 m east-west, 1220 m north-south.

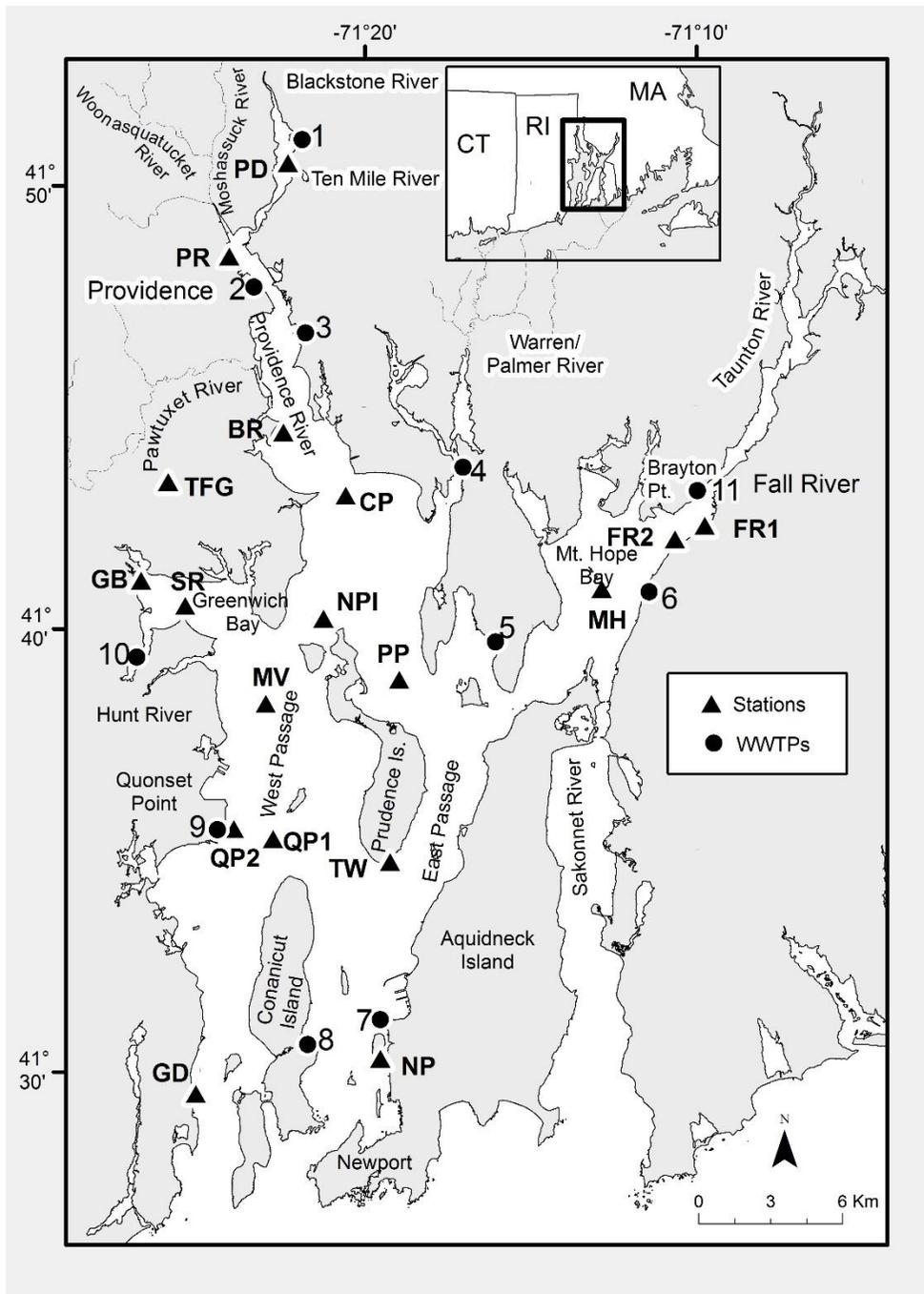


Figure 4. Waste water treatment facilities with direct discharge into NB (1 Bucklin Point, 2 Fields Point, 3 E. Providence, 4 Warren, 5 Bristol, 6 Fall River, 7 Newport, 8 Jamestown, 9 Quonset Point, 10 E. Greenwich, 11 Somerset) and fixed stations (BR Bullock Reach, CP Conimicut Point, FR Fall River, GB Greenwich Bay, GS URI/GSO, MH Mount Hope Bay, MV Mount View, NP Newport, NPI North Prudence Island, PD Phillipsdale, PP Poppasquash Point, PR Providence River, QP Quonset Point, Sally Rock SR, TFG T.F. Green Airport, TW T-Wharf, described in Codiga et al. 2009).

bottom waters. The scatter plots for the overall surface and bottom temperatures show good linear agreement with an  $R^2 = 0.99$ . Also, the model was able to reasonably capture the overall behavior of temperature stratification during the warming period from May to August, 2009. The scatter plot reflected a linear agreement ( $R^2 = 0.86$ ) between predicted and observed values of water surface elevation ( $R^2=0.86$ ) and subtidal variations ( $R^2=0.98$ ) at site PR. The model showed that flushing time depends on the horizontal and vertical location within the water body and that it can reach 60 days in parts of the water column within some regions in NB (e.g., Greenwich Bay).

EFDC model results were examined using skill assessment. Skill assessment for coupled biological/physical models of marine systems was presented by Stow et al. (2009). The assessment analysis compares a model prediction, which has an unknown “prediction error” from the true value (or truth), to observation, which has unknown “observation error” from the truth. The difference between the observation and the prediction is known and is identified as the “misfit” or “residual”. The skill analysis examines the closeness of predictions to “truth” by quantifying the misfits which should be small and noisy. It is worth mentioning that a slight phase error (e.g., in time) between a prediction and an observation can result in a big misfit. For EFDC salinity, six of the 10 skill metrics averaged across top and bottom measurements for 14 stations indicated >90% skill, and for temperature 10 of the 10 metrics indicated >90% skill. For elevation and subtidal elevation, 7 of the 9 metrics compared to NOAA observations at two sites each (PR, used for calibration and FR1, used for validation) indicated >90% skill. Predicted salinities showed values lower than observed at some locations, especially near the bottom. Similar difficulties in matching salinity predictions to observations were reported by Applied Science Associates (ASA, 2005) after applying their 3-D model (BFHYDRO) with fine segments (size 100–400 m) and eleven sigma layers. Their predictions lacked stratification in the water column. Although the reported uncertainty range in the USGS stream flows is 6%–19%, (Harmel et al. 2006), which may have an impact on salinity predictions, all stream flows were kept at the values reported by USGS in this work. Further investigation may be necessary to address this issue. The impact of defining an accurate concentration time series at the open boundary (e.g., for salinity) should not be underestimated. Such concentrations would reflect the values in the ebbing or flooding water across the boundary. However, based on the location of the open boundary, some of the flooding flow would have concentrations related to the preceding ebbing flow and actual entry of the assigned boundary values would be delayed.

One important advantage of this system is its ability to provide adequate predictions on a coarse numerical mesh (e.g., 640 m x 1200 m for NB), which facilitated running its water quality and ecology routines simultaneously with the hydrodynamics. Predictions for a full year simulation took only 13 hours on a DELL- Precision - T5600 computer, which is very convenient for similar applications. In addition to hydrodynamics, EFDC provides the possibility to model water quality, and some ecological end points, which are being explored as a possible comparison to the WASP modeling (Abdelrhman 2015b). Among other modeling systems, the Finite Volume Coastal and Ocean Model (FVCOM) and the Regional Ocean Modeling System (ROMS) cover the hydrodynamics like EFDC. Although the FVCOM has some water quality capabilities, its ecological abilities were not tested. The ROMS does not include any water quality or ecological capability; other models are needed to provide these capabilities. The

communication process between ROMS and such models is challenging and involves considerable data transfer, time, effort, resources, and operators. Additionally, the ROMS model run time is much greater than EFDC. After building the necessary capacity, the EFDC system can be utilized by a single modeler on a single desktop computer to provide adequate results within a reasonable execution time. For NB, the EFDC application supports water quality modeling, which is significant for nutrient management. Future directions for the SFDC modeling include modeling additional years with different hydrology, in order to explore the robustness of the model, and using future climate scenarios as input, in order to understand how future climate will alter hydrodynamics.

### **3.2 Water quality**

Water quality is being simulated by EPA's WASP model (version 7.52) to relate nutrient loading to frequency and duration of hypoxia. WASP is a widely-used process-based simulation model freely available from the US EPA (<http://www.epa.gov/athens/wwqtsc/html/wasp.html>). WASP can model the dynamics of individual nutrient species, organic matter, dissolved oxygen, alkalinity, inorganic solids, periphyton, and multiple phytoplankton groups (Fig. 5). WASP has previously been used to model estuaries, for example Tampa Bay in Florida (Wang et al. 1999), and Neuse River estuary in North Carolina (Wool et al. 2003). It can be used in a linked mode with EFDC, where EFDC provides hydrodynamics, temperature, and salinity, and WASP calculates water quality using the EFDC output and biochemical kinetics. This linked approach was used for NB: this WASP application with EFDC used the same spatial scale as EFDC (Fig. 3), with a maximum timestep of a day (WASP adjusts the timestep as necessary). The application presented here is for the year 2009, details of the application are presented in Dettmann and Charlestra (2015).

Input data were taken from several sources. Modeled solar radiation data for Newport, Rhode Island were retrieved from the National Solar Radiation Database (Wilcox 2012). It is noted that no direct solar radiation measurements are available on NB after 2008. Tidal data were extracted from the NOAA database historical archive (<http://tidesandcurrents.noaa.gov/>). Boundary conditions were set using stream and WWTF inflows communicated by the EFDC model (see section 3.1), with the open ocean boundary nutrient concentrations based on Krumholz (2012). Concentrations for river nutrients were derived from the Narragansett Bay Commission database (<http://snapshot.narrabay.com/app/WaterQualityInitiatives/NutrientMonitoring>). Data for nutrient inputs from two WWTFs were provided by the NBC; data for nutrient loading from the remaining WWTFs was downloaded from the US EPA's Integrated Compliance Information System (ICIS) database, which contains data submitted to the Agency by permitted dischargers, based on monitoring requirements specified in their National Pollution Discharge Elimination System (NPDES) discharge permits. Since only total phosphorus (TP) was measured, dissolved inorganic phosphorous concentrations were assumed to be 80% of TP, and that the remaining 20% of TP is dissolved organic phosphorus (David Pincumbe, USEPA Region 1, pers. comm. 2014). Biochemical oxygen demand was calculated based on effluent measurements from the Bucklin Point, East Providence, and Field's Point WWTFs in 1989 (Doering et al. 1990, Dettmann et al. 1992). Atmospheric deposition for nutrient and carbon state variables was calculated using data from the literature (Nixon et al. 1995, Bowen and Valiela 2001, Jurado et al. 2008). Kinetic constants were taken from published studies and WASP

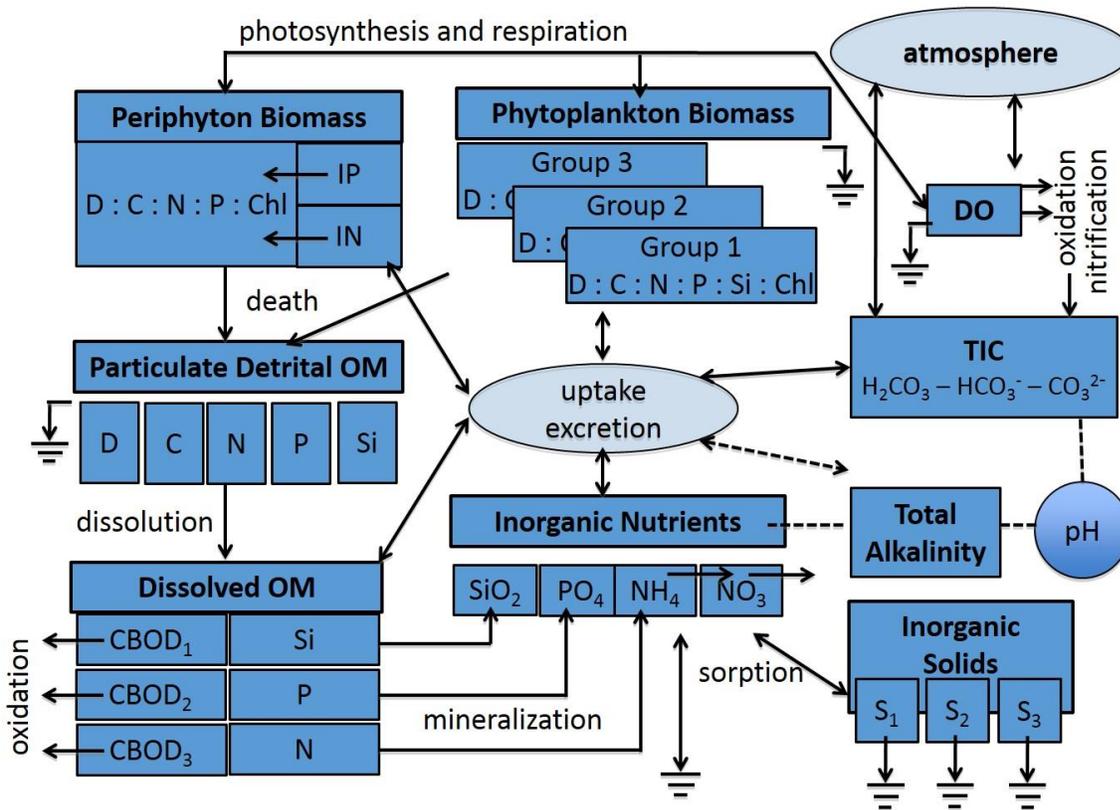


Figure 5. Water Quality Analysis Simulation Program (WASP) model state variables and processes. WASP includes nutrients, dissolved and particulate organic matter (OM), dissolved oxygen (DO), total inorganic carbon, alkalinity, periphyton, and three groups of phytoplankton.

documentation (Kremer and Nixon 1978, Wang et al. 1999, Ambrose and Wool 2009). Data on benthic nutrient flux rates, including temperature dependence, were obtained from a variety of sources: Lindsey Fields (pers. comm.), Fulweiler et al. (2010), Kremer and Nixon (1978), with SOD based on Fulweiler et al. (2010). At present, one phytoplankton group is being simulated.

Modeled nutrient output was compared primarily to Krumholz (2012); For example, Fig. 6 shows simulated and observed water concentration of TN in transects along the East and West passages of NB. Agreement between model results and observations is excellent for the West Passage transect; agreement is good between Conimicut Point and Station 5 in the East Passage, but concentrations are over-predicted at Stations 11 and 12 in the upper portion of the Providence River. Modeled dissolved oxygen (DO) and chlorophyll a (Chl *a*) data were compared to the fixed station buoy data. The fixed-site monitoring system (installed on buoys and docks) around NB provides high frequency (every 15 minutes) measurements for some parameters, such as salinity, temperature, DO and Chl *a* (Codiga et al. 2009). At Conimicut Point, simulated DO values at the bottom capture the general observed seasonal trend, but differ in details (Fig. 7a). The seasonal observed minimum occurs at approximately August 19; the minimum simulated values occur approximately a week later and are about 1-2 mg/L higher than the observed minimum. Simulated values are also approximately 2 mg/L higher than observed values for much of the period of declining values between May and August, and somewhat low during the period

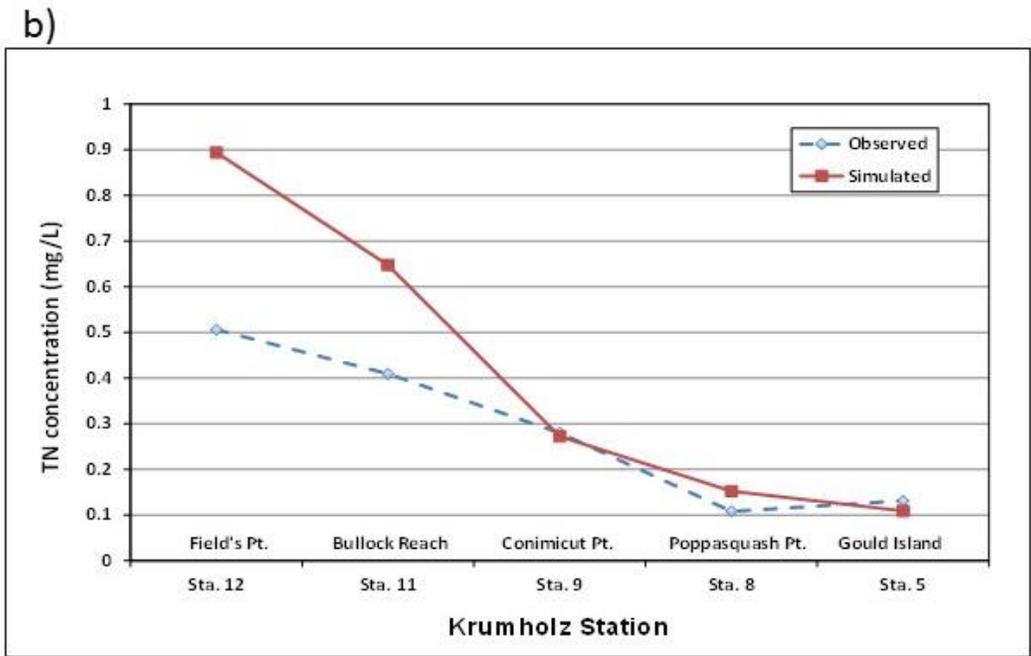
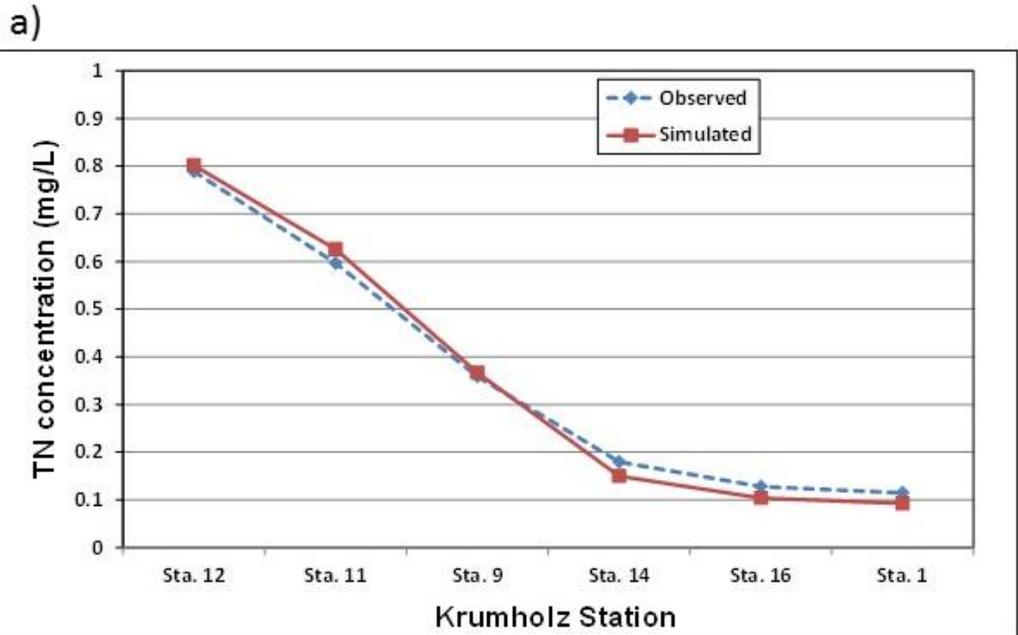


Figure 6. Simulated and observed surface-water concentrations of total nitrogen (at noon) along a longitudinal transect in a) West Passage – Fields Point to Krumholz (2012) station 1 near northern Jamestown, 7 July 2009; and b) East Passage – Fields Point to Krumholz (2012) station 5 near Gould Island), 16 June 2009.

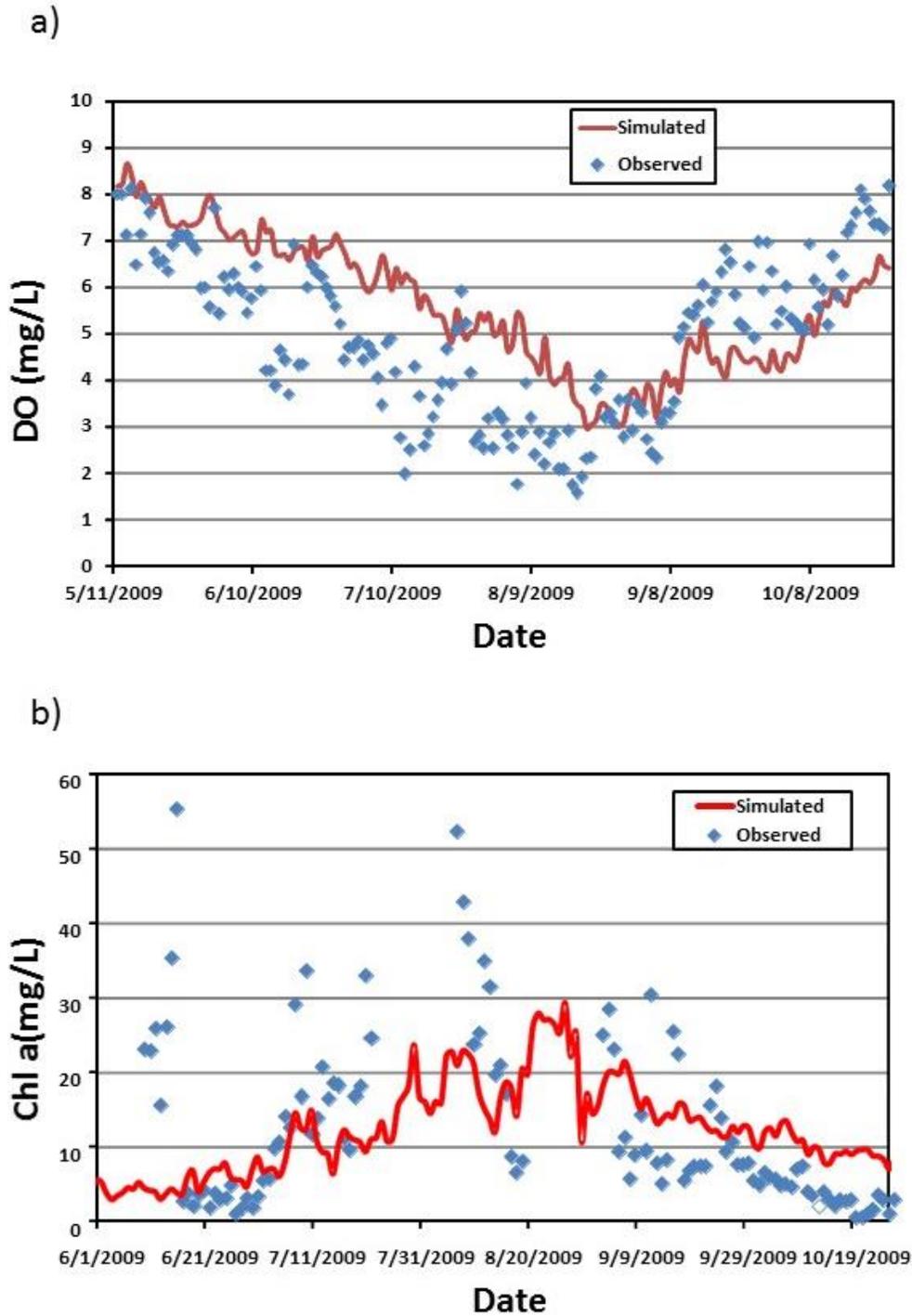


Figure 7. WASP water quality model simulated and observed concentrations for the Conimicut Point Buoy: a) noon near-bottom dissolved oxygen for May – October 2009. b) noon near-surface (layer 3) chlorophyll a for June – October 2009. Layers were chosen to approximate sensor depths.

of increasing DO concentrations following the minimum. O'Donnell et al. (2014) found a similar issue of differences between DO predictions and observations from their models of Long Island Sound. Simulated Chl *a* values are in general accord with most observations throughout the simulated period, but do not represent the occasional observed short-lived spikes (Fig. 7b). The Chl *a* data suggest that the period from 6/21-8/21 may be a single event rather than a grouping of short term fluctuations, but missing data within this range creates uncertainty. The model does better at simulating seasonal patterns of DO and chlorophyll *a* than short-term dynamics – this is likely caused, at least in part, by the low frequency of boundary data, and the use of surrogate solar radiation data. Additional comparisons will be based on number of days of hypoxia, i.e., how often during the simulation year the daily average DO concentrations fall below certain criteria (e.g., < 2.8 mg/L or 4.8 mg/L).

While simulation results presented here capture the overall seasonal trends of TN, Chl *a* and DO, further calibration is required to improve agreement with observations. However, it is noted that in-estuary nutrient measures are limited; this should be a consideration for further data collection. As part of the ongoing calibration, Dettmann and Charlestra (2015) are focusing on three areas. First, they are exploring the sensitivity of model results to various mixing regimes for inputs from tributaries and municipal WWTFs. They initially assumed that these inputs are completely mixed throughout the total depth of the water column, but recently acquired dye study data indicate that WWTF effluents are often concentrated near the surface. Second, they will use recently-acquired data on daily water-column metabolism to check the model simulation of rates and dynamics of phytoplankton production and respiration. This will serve as a further constraint on the kinetic parameters for the phytoplankton model. Third, they are exploring options for improving the representation of actual local time series of solar radiation for Narragansett Bay. Thus far, modeled data that represents the statistics of local solar radiation was used, but not its actual time-dependence in the year being simulated. They continue to explore other possible reasons for the limited ability of the model to reproduce high-frequency variability. There are several future directions for WASP modeling in NB. First, the WASP version 7 release includes a sediment diagenesis model, which predicted sediment oxygen demand (SOD) and nutrient fluxes from the underlying sediments, so an additional future direction is to include this module for NB. Second, additional algae groups can be added, to better simulate the different seasonal blooms. Third, we will continue to explore linkages with ecological models, in terms of converting and transferring output in useful formats. Finally, varying nutrient scenarios can be run with WFDC-WASP, as described in section 4.2.

### **3.3 Seagrass**

#### **3.3.1 Bio-optical model**

While prediction of suitable seagrass habitat must consider parameters beyond water clarity (see 3.3.2), water clarity is one of the most influential predictors of seagrass distribution. Bio-optical models have been developed for estimating the maximum depth of seagrasses for a number of coastal areas (e.g., Biber et al. 2008, Gallegos 1994, 2001, Gallegos and Kenworthy 1996, and Kenworthy et al. 2014). Thursby et al. (2015) have developed calibration data for the use of an existing bio-optical model (see Kenworthy et al. 2014) for Narragansett Bay. With the existence of Narragansett Bay specific calibration

data, information on the concentrations of colored dissolved organic matter (CDOM), total suspended solids (TSS) and Chl *a* are used to model estimates of changes in light quantity and quality with depth which can be used, among other things, to predict the maximum depth likely for seagrass. In the case of NB, the seagrass is *Zostera marina*, and the light limit is usually around 20% of surface light (Duarte 1991).

The measurement of light extinction with depth can be a complicated process, and requires specialized equipment—the attenuation coefficients for light relative to water quality parameters (i.e., CDOM, TSS and Chl *a*) are wavelength dependent. With a bio-optical model and appropriate calibration data, a complicated process is reduced to just requiring data on CDOM, TSS and Chl *a*. Therefore, a potentially complex monitoring process for water clarity is greatly simplified, as the above parameters are often standard measurements for coastal monitoring programs, or can be provided from water quality models or remote sensing. In the case of Narragansett Bay, the most significant influence on water clarity having the potential to be managed is Chl *a*. This is mediated through the relationship between nutrients (usually N) and Chl *a* (phytoplankton).

The management implications for the bio-optical model are straightforward. The bio-optical model uses the wavelength-dependent absorption and scattering coefficients for the three optically active constituents (CDOM, TSS and phytoplankton) to calculate the diffusive attenuation coefficient ( $K_{dPAR}$ )—integrated over the photosynthetically active wavelengths (400 to 700 nm). These values can be easily converted into any depth of interest. For example, what is the distribution of depths within Narragansett Bay where 20% ( $Z_{20}$ ) of surface light can reasonably be expected to reach the bottom? Fig. 8a shows the distribution of the calculated extinction coefficients using remotely sensed estimates for CDOM, TSS and Chl *a*. These data were derived from a December 2009 scene using the Hyperspectral Imager for the Coastal Ocean (HICO) on the International Space Station. Over 24,000 individual data points were generated from the covered area. These data were then converted to estimates of the depth for each point at which the available light was 20% of the surface light. Because light availability is not the only predictor of seagrass presence, one can then easily overlay these data with other information, such as sediment type, water current, etc., as described below in 3.3.2. As an example, Fig. 8b has removed all of the bathymetric depths greater than 3 m. A very useful future image would be one where the data from Fig. 8 are culled based on sediment and wave energy parameters. This process provides a useful method to predict expected changes in seagrass potential area as water clarity might be improved through nutrient reduction scenarios.

Calibration information for bio-optical models are available from regions throughout the globe. Many of these datasets are for the open ocean, and others are for coastal areas within Arctic and European waters. A future direction for this work will be to run sensitivity analyses to test the effect of using various other calibration data. If a “universal” calibration could be derived, the usefulness of remote sensing to estimate water clarity would greatly increase.

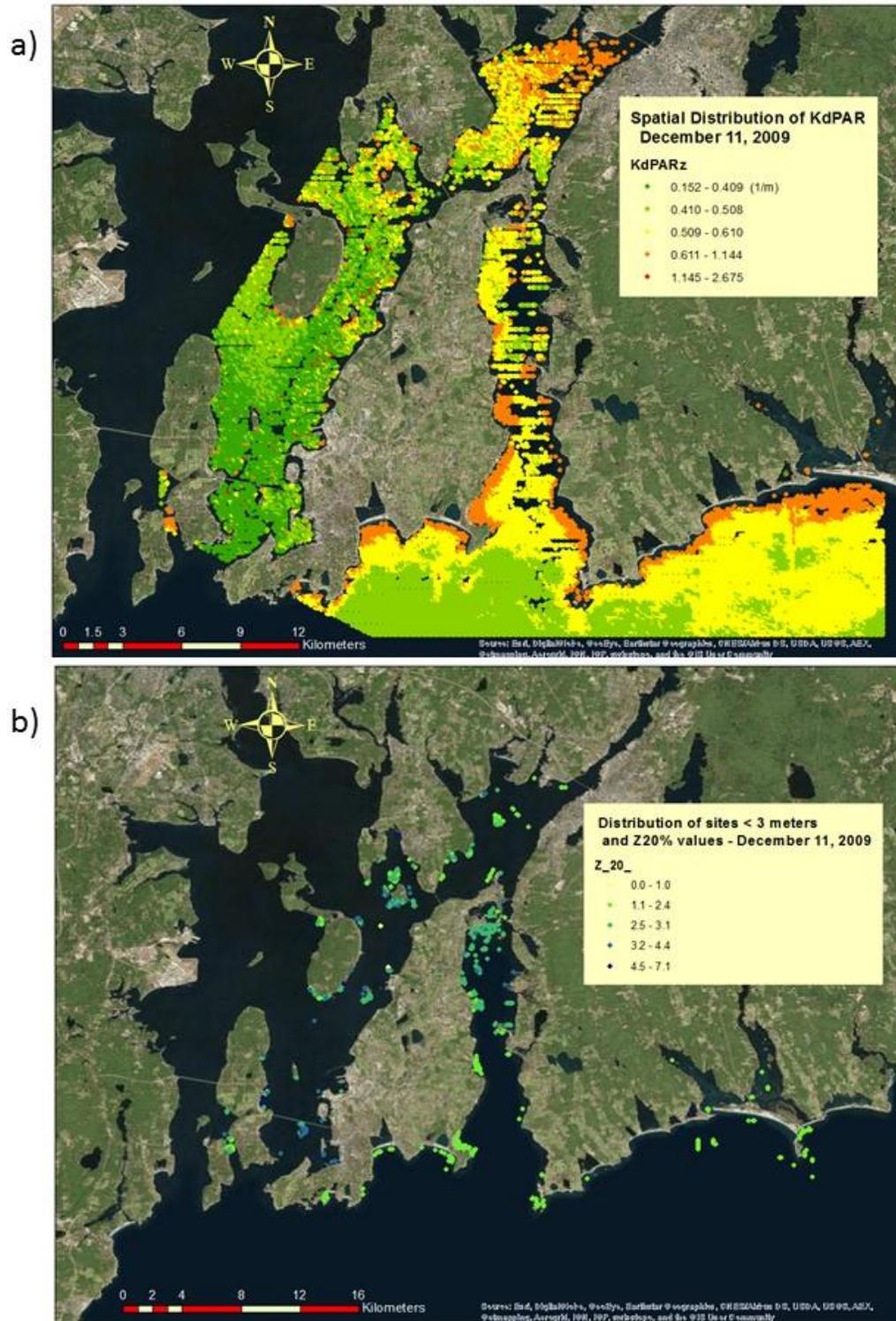


Figure 8. Bio-optical model output. a) Spatial distribution of diffuse attenuation coefficients ( $K_dPAR$ ) along eastern passages of Narragansett Bay from a 11 December 2009 Hyperspectral Imager for the Coastal Ocean (HICO) image, b) Spatial distribution of depth of 20% of surface light calculated from data above, culled to remove any depths > 3 m.

### 3.3.2 Seagrass model

Restoration of seagrass habitats in estuaries requires an understanding of the roles of multiple interacting stressors including nutrients, climate change and habitat modification. Detenbeck and Rego (2015) explored the application of generalized linear mixed models (glmm) and generalized additive mixed models to describe the simple and interactive effects of environmental factors on the distribution of *Z. marina* in NB. They used a random shoreline effect to account for random colonization or extinction effects. They developed a spatial framework describing the coordinates of spatial autocorrelation in estuarine systems, with the main axis parallel to the shoreline and a secondary axis perpendicular to the shoreline, and used an approach to incorporate a term for residual autocorrelation in glmm first introduced by Crase et al. (2012). They dealt with covariance of environmental factors by excluding correlated variables where necessary, and using multiple strategies to describe the interaction of the light environment and wave energy with depth.

Detenbeck and Rego (2015) predicted seagrass distribution at the scale of 10-m grid cells, as presence/absence or average presence/absence associated with shoreline locations spaced at 10-meter intervals, and minimum or maximum depth of distributions at those locations. Prediction of seagrass absolute or average presence/absence at shoreline locations was very robust, with area under the Receiver Operating Characteristic curves of 0.95 – 0.98 following 10-fold cross-validation of models. For the model predicting seagrass presence/absence at the grid cell scale, the most influential predictor of fixed effects was Secchi depth, followed by (in order): shoreline isolation, sediment percent total organic carbon, sediment type, and salinity. For the model predicting seagrass presence at shoreline locations, the most influential predictor was sediment type, followed by sediment percent total organic carbon (at low Secchi depth), then salinity (as an indicator of downstream gradients in TN). For all shorelines combined, this model predicts that a 40% reduction in TN loads (and concentration) would support a 12% increase in the area colonized by seagrass in the 0 to 5 m depth zone to about 63% in the short term and slightly more over future decades as sediment organic carbon recovers.

### **3.4 Benthic Invertebrates**

Benthic macroinvertebrates are commonly used as biomonitors to detect pollution impacts in estuaries, because they are relatively sedentary, respond predictably to pollution (Pearson and Rosenberg 1978), and act as integrators of stress over months to years (Weisberg et al. 1997, Paul et al. 2001). A healthy macroinvertebrate community is diverse, with large, deep-burrowing organisms. As organic enrichment increases, the bigger, longer lived organisms start to disappear, the community may increase in abundance (more numbers) but will contain smaller, opportunistic organisms. This corresponds to the upper oxygenated level of the sediment becoming shallower. Mass mortality of the benthos ('benthos kills') is likely when bottom DO is  $\sim 1$  mg/L (Diaz and Rosenberg 1995), and a bacterial mat may be present. Contemporary work has shown that north-south gradient of nutrients in NB is reflected in the benthic invertebrate community (Calabretta and Oviatt 2008), where impacted areas were characterized by lower diversity and early successional species. There may also have been shifts in the benthic community due to changing plankton communities within the Bay, including the modification of the

winter-spring bloom (Borkman and Smayda 2009, Smith et al. 2010). Here, we describe two efforts directed at understanding the current patterns in NB.

### 3.4.1 Response of benthic macroinvertebrates to eutrophication related stressors

Benthic macroinvertebrate data, bottom DO, and sediment total organic carbon (TOC) were assembled from two major sources. Data from 1990-2006 were obtained from EPA's National Coastal Assessment ([www.epa.gov/emap](http://www.epa.gov/emap)). In addition, benthic surveys from 2004-2006 conducted in Newport, Greenwich Bay and Providence River using similar procedures (Pelletier, unpublished) were also compiled. Bottom DO represents a single point measure taken ~ 1 m off the bottom during the day. These measurements likely underestimate hypoxic events. TOC is a more stable measure of organic enrichment and environmental stress (Hyland et al. 2005, Pelletier et al. 2011). The benthic invertebrate data were summarized by total number of species. Quantile regression models (90% quantile) were developed to determine the relationship between eutrophication measures (DO, organic carbon) and invertebrate condition. All species were classified into one of five pollution tolerance groups (Gillett et al. 2015), defined originally by Borja et al. (2000). Species range from Group I (pollution sensitive) to Group V (pollution tolerant). Selected species from these groups were examined to determine if their distribution in NB relative to oxygen and carbon levels corresponded to their expected response.

Benthic diversity decreased with increasing sediment organic carbon ( $p=0.0017$ ) and increased with increasing bottom DO ( $p=0.0033$ ; Fig. 9). Group I and II species are considered to be pollution sensitive. *Nucula proxima* is a Group I bivalve which was most abundant at low TOC levels. Its highest abundance occurred when sediment TOC was between 1.5% and 2%, although high abundances were apparent up to ~3% TOC (Fig. 10). *Nucula proxima* abundance was higher when bottom dissolved oxygen was high, with the highest abundance observed at 8 mg/L. *Ilyanassa trivittata*, a Group II gastropod (mud snail) skewed towards higher abundances at lower % TOC. This species appears to be less sensitive to DO - abundance was at its highest levels when DO was between 3 and 7 mg/L. *Nephtys incisa*, a Group II predatory polychaete had its highest abundance when TOC was <3%, and DO was >3 mg/L. Overall, the individual species responded as expected to organic enrichment and dissolved oxygen stress. Group III species are tolerant organisms that may be enhanced by organic enrichment (Borja et al. 2000). *Ampelisca abdita* is a classic, Group III mat forming amphipod. Its highest abundances were seen between 1.5% and 3.5% TOC (Fig. 11). This is consistent with Diaz et al. (2008) who estimated that ideal conditions for *Ampelisca* mat building was between 2% and 3% TOC. *Ampelisca abdita* also showed a relationship with DO; highest abundance were seen between 2.5 mg/L and 5 mg/L. DO. The Group III spionid polychaete *Polydora cornuda* showed no apparent relationship between abundance and organic carbon; abundance was high when TOC ranged from 1% and 8% (Fig. 11). There was a suggestion of DO tolerance, but its range appeared to be quite broad (2-5.5 mg/L). Group IV and V are opportunistic species that are often also classified as pollution tolerant. *Streblospio benedicti*, a spionid polychaete, is a classic example of this group (Fig. 11). This cosmopolitan species is found in high abundance at all TOC and DO levels, but had its highest abundance at ~6.5% TOC and ~2 mg/L DO. This is the expected response of this species to organic enrichment and dissolved oxygen stress.

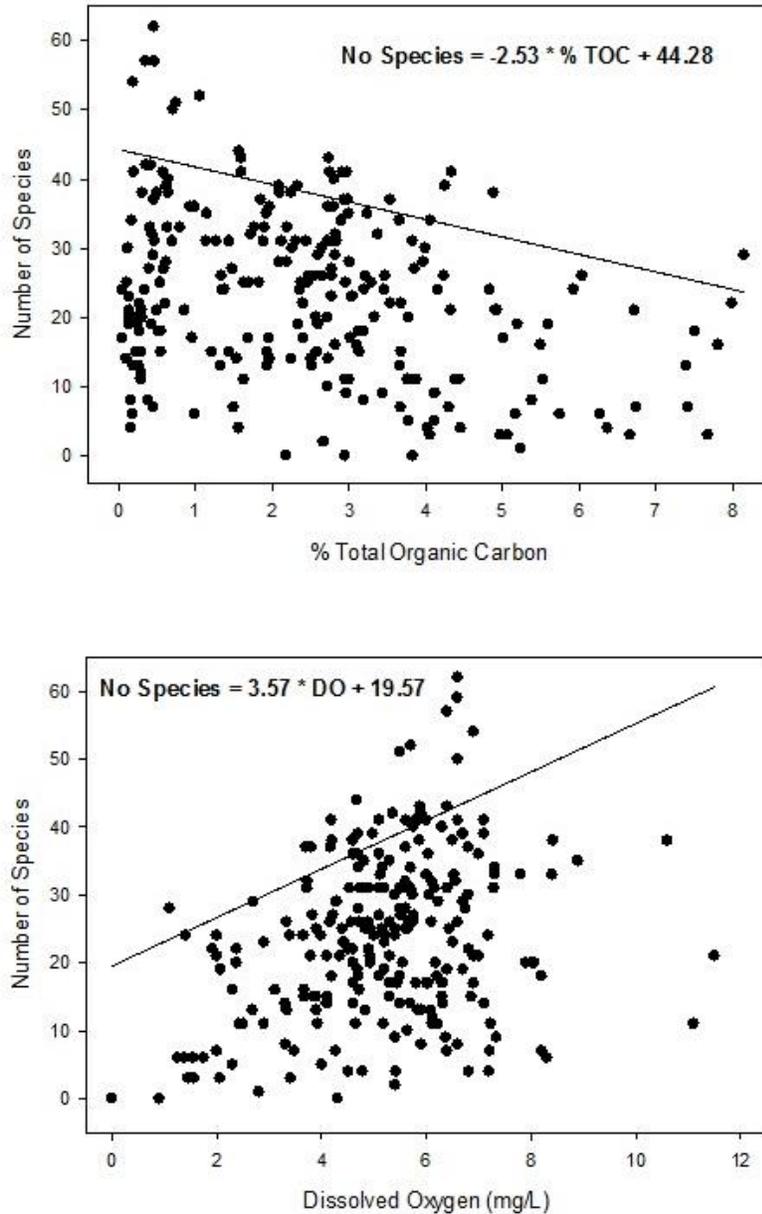


Figure 9. The relationship of number of benthic macroinvertebrates to sediment organic carbon and bottom dissolved oxygen (lines show quantile regression,  $\tau=0.9$ ).

Overall, the benthic invertebrates appear to be responding to oxygen and sediment organic gradients. This can be seen at the community level (diversity), and in the responses of individual species. However, there is variability in the responses, as other factors and habitat also influence invertebrates (e.g., Frithsen 1989). We expect that data from benthic surveys conducted between 2012 and 2014 (Pelletier, unpublished) will be available in the near future. These data were collected near the RIDEM buoys (Codiga et al. 2009), which should allow the patterns described above to be further examined with better DO data that can correspond to Rhode Island's current dissolved oxygen criteria standard.

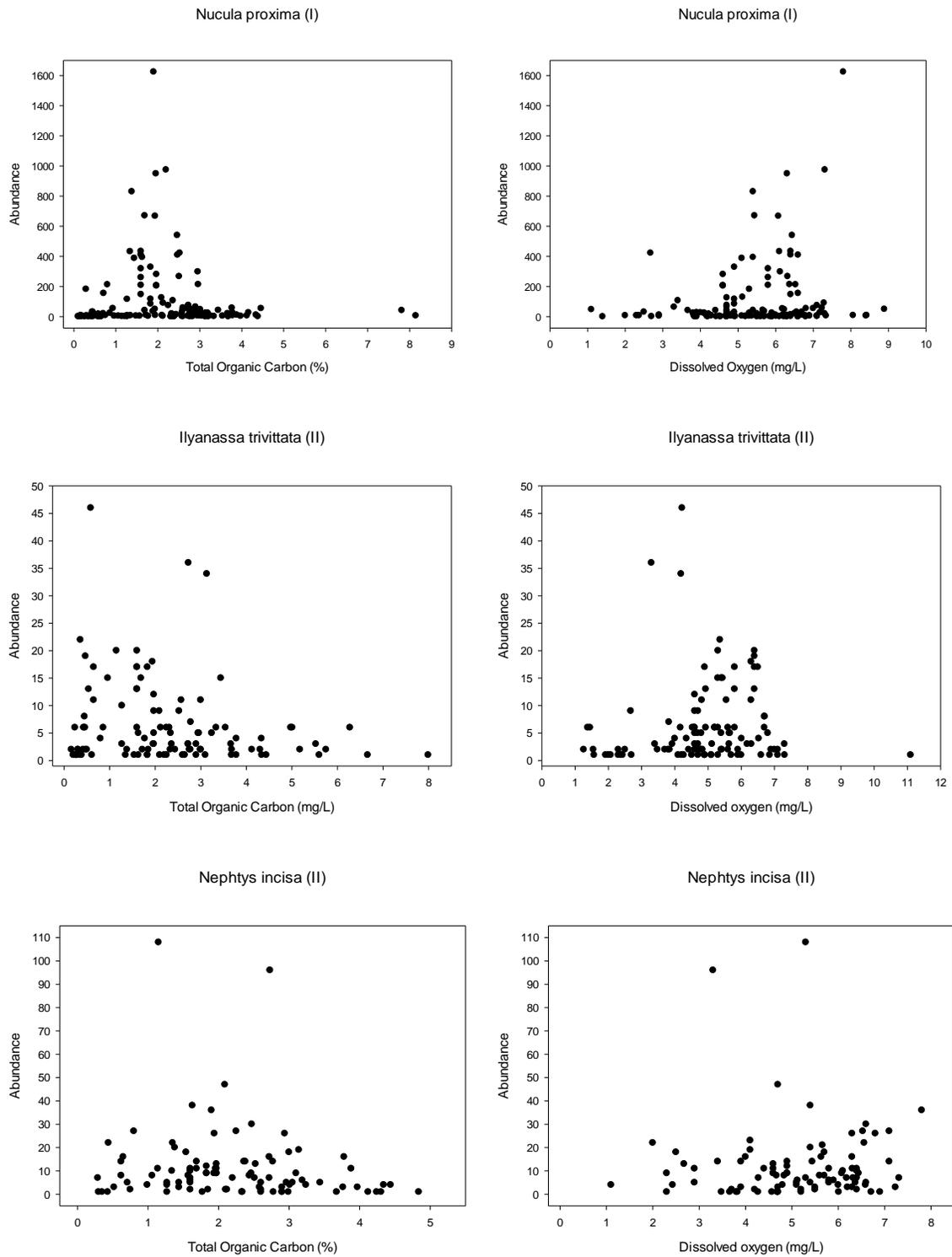


Figure 10. Response of Group I and II benthic invertebrate species (pollution sensitive) to varying sediment organic carbon and bottom dissolved oxygen levels.

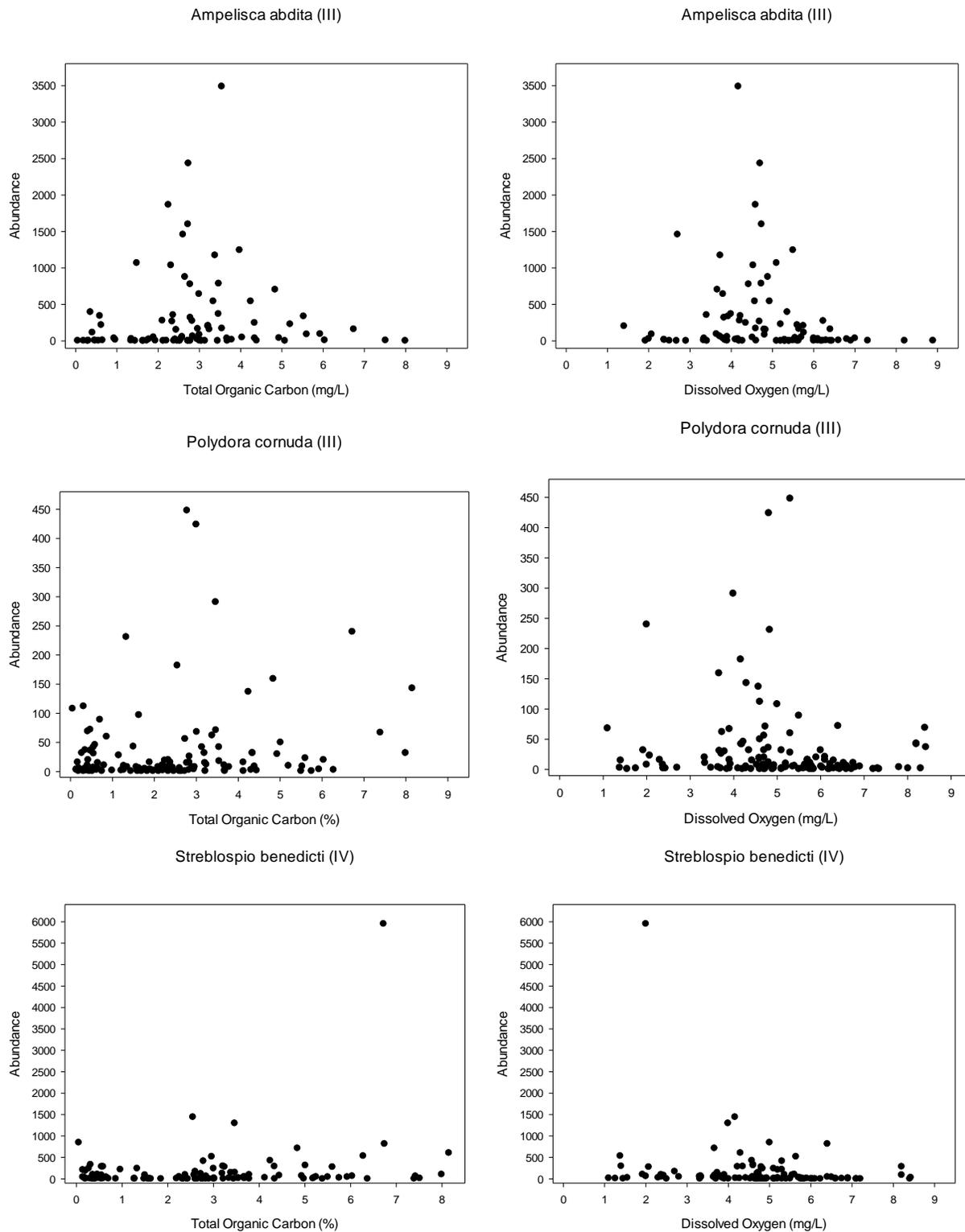


Figure 11. Response of Group III (intermediate sensitivity) benthic invertebrate species and *Streblospio benedicti*, a Group IV species to varying sediment organic carbon and bottom dissolved oxygen levels.

### 3.4.2 Invertebrate structure and function

To better understand how eutrophication-driven over-enrichment of the sediments and seasonal hypoxia of the bottom water have affected ecosystem functions and services of the benthic community, a benthic data set spanning 20 years and 152 stations in NB, along with ancillary data, was used to compare measures of community structure and function from stations in seasonally hypoxic areas with stations in normoxic areas (Hale 2015). Benthic data were taken from the EPA's EMAP Virginian Biogeographic Province program (sampled from 1990-1993); National Coastal Assessment (NCA 2015, sampled from 2000-2006); and National Coastal Condition Assessment (NCCA 2015, sampled in 2010). These data were supplemented with benthic macrofauna (> 2.0 mm) biomass data from a waterfowl feeding study at 30 stations throughout NB (Loring et al. 2013); sediment oxygen uptake, sediment-water nutrient, and N gas flux data at a mid-Bay station and a Providence River station (Fulweiler 2007); and Redox Potential Discontinuity (RPD) sediment depth and community successional stage data from a sediment profile camera study at 56 stations throughout the Bay (Valente et al. 1992). Hypoxic areas were those with frequent bottom water DO levels < 2.9 mg/L, occasional episodes of 0 mg/L, and summer (July and August) mean DO < 4.8 mg/L (Prell et al. 2004, 2006; Saarman et al. 2008). Stations with sediments contaminated by metal or organic pollutants were removed from the analysis. Data groups were compared with t-tests.

Differences between the hypoxic and normoxic areas are summarized in Table 2. Mean species richness in the seasonally hypoxic areas was lower than in normoxic areas (23 vs. 29;  $p=0.05$ ). Many rare species found in normoxic areas were not present in hypoxic areas—the number of species present in only one sample was 4 in hypoxic areas vs. 86 in normoxic areas. Mean abundances were not significantly different; however, mean abundance for Ostracoda, a hypoxia-sensitive group of crustaceans, was higher in normoxic areas than in hypoxic areas, and Spionidae, a hypoxia-tolerant group, showed the opposite pattern. Mean biomass and secondary production were both lower in hypoxic areas. Benthic communities in the hypoxic areas were at a significantly earlier successional stage. Lastly, the mean depth of the sediment Redox Potential Discontinuity, a measure of sediment bioturbation by macrofauna, in hypoxic areas was one-fifth that of normoxic areas. The hypothesis that these benthic changes are related to sediment characteristics is supported by analysis of Fulweiler (2007) data, which shows that mean oxygen uptake and ammonium flux out of the sediments was significantly higher in the hypoxic areas, as compared to normoxic areas (Table 2). This degradation of benthic community structure and function has implications for sustainable provision of ecosystem services that are desired by the human population surrounding and using the Bay. For future scenarios, this threshold finding can be applied by calculating mean July-August DO levels from WASP output under current and future scenarios, and plotting changes in the distributions of the community types.

### **3.5 Fish**

Estuarine fish are highly valued resources that are affected by several factors, including eutrophication, climate, landscape alteration, pollution, and fishing pressure, the effects of which are difficult to disentangle (Breitburg et al. 2009a and b). Several studies have shown that estuarine fish communities in the northeast and mid-Atlantic regions of the U.S. have been changing through time, for NB (Oviatt et

Table 2. Narragansett Bay benthic measures and sediment-water fluxes in periodically hypoxic areas -- those with frequent bottom water dissolved oxygen levels < 2.9 mg/l, occasional episodes of 0 mg/l, and a summer (July and August) mean of <4.8 mg/L -- versus normoxic areas (Hale 2015).

<b>Parameter (Data from NCA 2015, NCCA 2015, except where noted)</b>	<b>Hypoxic</b>	<b>(N)</b>	<b>Normoxic</b>	<b>(N)</b>	<b>p</b>
Mean # species	23	(23)	29	(92)	0.05
Mean abundances	653	(23)	536	(92)	0.52
Mean Ostracoda (hypoxia-sensitive)	16	(45)	114	(106)	<0.001
Mean Spionidae (hypoxia-tolerant)	5672	(45)	917	(102)	0.04
Multidimensional scaling of abundances	-		-		<0.001
Rare species (based on n=152):					
# One occurrences	4		86		-
# Two occurrences	7		44		-
Mean one occurrences	0.2	(23)	0.8	(92)	<0.001
Mean two occurrences	0.4	(23)	0.8	(92)	0.02
Mean biomass, g C dry wt/m <sup>2</sup>	10.1	(2)	52.0	(7)	0.10
Mean secondary production, ug C/ m <sup>2</sup> /day	28.3	(2)	104.2	(7)	0.15
Mean depth of apparent Redox Potential Discontinuity (RPD), cm (Valente et al. 1992)	0.7	(24)	3.3	(24)	<0.001
Mean successional stage (Valente et al. 1992)	1.5	(37)	2.3	(34)	0.03
<b>2000s sediment-water fluxes (Fulweiler 2007)</b>					
Mean oxygen uptake, mg/m <sup>2</sup> /hr	43.0	(33)	31.2	(12)	0.01
Mean ammonium flux out, uMol/m <sup>2</sup> /hr	122.9	(35)	46.6	(12)	0.05
Mean phosphate flux out, uMol/m <sup>2</sup> /hr	2.8	(39)	2.1	(12)	0.77
Mean nitrogen gas flux out, uMol/m <sup>2</sup> /hr	78.9	(33)	71.6	(12)	0.89

al. 2003), the Saco River estuary in southern Maine (Furey and Sulikowski 2011), Long Island Sound (Howell and Auster 2012), and Chesapeake Bay (Sobocinski et al. 2013, Buchheister et al. 2013). Temperature is a significant factor affecting these communities, with warmer temperatures favoring warm-adapted species (Howell and Auster 2012, Wood et al. 2009), influencing diversity (Sobocinski et al. 2013) and favoring pelagic over demersal species (Collie et al. 2008). Buchheister et al. (2013) found that the demersal fish community in Chesapeake Bay was primarily structured by the latitudinal salinity gradient, while water temperature influenced seasonal dynamics, and low DO concentrations suppressed catch rate, and diversity. Breitburg et al. (2009a) found that hypoxia may not reduce system-

wide fisheries landings, but it is a serious challenge to management. Previous work in NB quantified trends in fish assemblages over time to climatic influences in the bay, including an increase in taxonomic diversity, a shift from vertebrates to invertebrates, and a shift from demersal to pelagic species (Jeffries and Terceiro 1985, Collie et al. 2008). In an early study of NB fish, depth, sediment organic content, temperature, and wind speed were significant in explaining patterns of fish species distribution (Oviatt and Nixon 1973).

Here, we considered the effects of depth, habitat, nutrients, and temperature on fish. We used finfish data collected in a trawl survey conducted by RIDEM for the years 1980 – 2012 (Olszewski 2011, Fig. 12). The data are a combination of seasonal surveys (14 random stratified stations selected from a predefined grid) and monthly surveys (13 fixed stations, began in 1990). At each station an otter trawl with ¼ inch liner and 4.5 inch mesh is deployed. For this analysis, we used the bottom strata (which is more likely to experience hypoxia) for the seasonal data (which had a broader spatial range). We combined the data for “Little Skate”, and “Clearnose Skate” with “Skates”, since they were only reported separately in some years. We used a  $\log_{10}(n+1)$  transform on all abundance data, dropped any species occurring in less than 10% of the samples (to exclude vagrant species), and coded species as “Pelagic” or “Demersal” based on Collie et al. (2008) and Bigelow and Schroeder (1953). In our final dataset of samples with corresponding environmental data ( $n=130$ ), species richness in a sample ranged from 1-20, with a mean of 9; overall abundance ranged from 1-21906 individuals, with a mean of 2026; and the percent of pelagic species ranged from 0-100, with a mean of 47%.

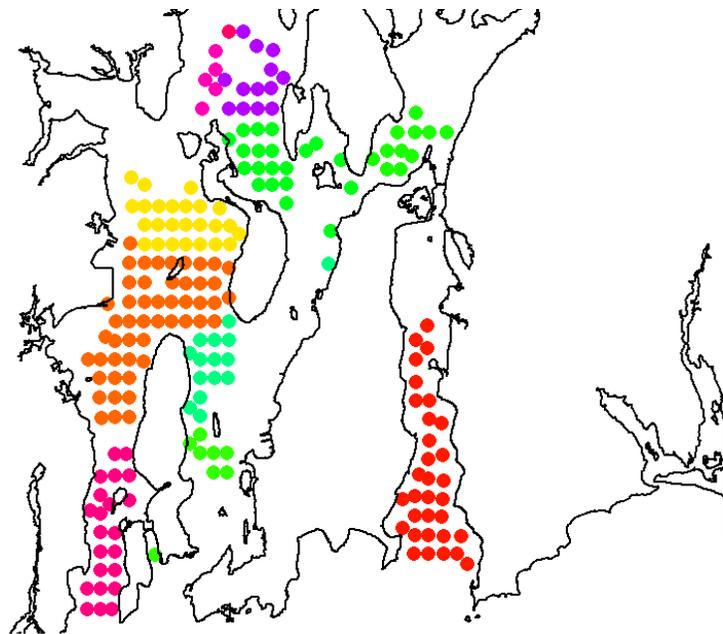


Figure 12. Map of Narragansett Bay with RIDEM fish trawl survey sampling locations color-coded by the bay segmentation used in Kremer et al. (2010).

Depth data (m) for sampling locations was obtained from a data layer compiled by the National Geophysical Data Center from NOAA hydrographic surveys. For habitat, we considered sediment class type (taken from NarrBay.org –because this source does not have data near the shoreline, proximity analysis was used to obtain data for five sites that did not overlay the substrate data layer), and linear distance to nearest eelgrass bed (km, calculated from the Rhode Island Geographic information System <http://www.edc.uri.edu/rigis/>). We used Nu Shuttle data (see narrbay.org for metadata and partial unprocessed data) for measures of Chl *a* (ug/L, calculated from bottle-calibrated fluorometer), DO (mg/L), temperature (°C), and salinity (ppt). The Nu Shuttle surveys were conducted monthly from 1998 to 2010 and consisted of a baywide boat track along which a multi-sensor instrument was towed. The instrument path was vertically sinusoidal and thus covered the full depth range. Due to high frequency data logging during each survey, the resulting data set for the 13 year period exceeds 4.2 million records. We created a spatial coverage of the bay segmentation used in Kremer et al. (2010), assigned each Nu Shuttle record and fish sampling site to the appropriate segment and depth layer using GIS tools, and then extracted the Nu Shuttle data from the most closely matching date to the fish sampling date (all dates were within a month, except for the samples in Sept. 2002, Sept. 2003, and April 2004).

Nu Shuttle data were collapsed into two depth layers (above and below 6.09 m) based on the RIDEM fish sampling protocol. These environmental variables were not significantly correlated with one another, except that salinity was negatively related to Chl *a* and seagrass, and positively related to depth (and so it was excluded from further analysis), and seagrass was positively related to depth. Measured DO ranged from 2.8-11.6 mg/L with a mean of 7.7 mg/L, measured Chl *a* ranged from 0.77-19.97, with a mean of 4.4. A regression for DO as a function of Chl *a* and temp was used to fill in 9 missing values.

For analysis, we considered response variables of species richness, total abundance, % pelagic species, and abundance of pelagics, demersals, invertebrates, and individual species, using the R package 'quantreg' (Koenker 2015). Quantile regression estimates the change in a specified quantile of the response variable, particularly an upper boundary of a scatter plot – this approach is useful for data with a wedged-shaped scatter plot, which results from other stressors co-occurring with the modeled stressor than can cause negative effects on the response ([http://www.epa.gov/caddis/da\\_basic\\_3.html](http://www.epa.gov/caddis/da_basic_3.html)). For the full dataset, 16 of the 30 response variables showed significant positive responses to temperature and negative responses to DO, which we interpreted as a seasonal pattern. Therefore, we reran the analysis for just the fall season. For fall, there was no significant relationship between DO and temperature, however, DO was significantly correlated with Chl *a* ( $r=0.52$ ,  $p<0.0001$ ). Results from the fall analysis are presented in Table 3. Total abundance, pelagics, invertebrates, lobster, and bay anchovy showed negative responses to Chl *a*, while demersals, menhaden, black sea bass, scup, striped sea robin, summer flounder, and windowpane flounder showed positive relation with Chl *a*, and a negative relationship with DO. Three additional species (bluefish, Atlantic moonfish, and longfin squid) showed a positive relationship with DO.

To some extent, we would expect a positive relationship with Chl *a*, representing a food source, so the negative relationship observed with some response measures may indicate nutrient effects. There are multiple reasons why a stronger response to DO was not observed: for the fall data, only 12% of samples have  $DO<5$  mg/L, the level at which DO concentration may be expected to have a significant effect

(US EPA 2000) and compensatory mechanisms may be taking place (Breitburg et al. 2009a). Possibly more detailed analysis of the monthly data would yield a clearer pattern. A limitation of this analysis is that the effect of fishing pressure was not accounted for, due to lack of data. Estuarine fish community changes may be modeled in a detailed, mechanistic way with a representation of age classes (Gamble and Link 2009); however, data availability needs are significant for this type of approach, which are not currently available for NB. Adamack et al. (2012) used an individual-based model for bay anchovy that was linked with watershed and water quality models in the Patuxent River and Chesapeake Bay in Maryland, to investigate effects of hypoxia. Future work in NB could focus on species-level models for key species such as winter flounder. Modeling of estuarine fish is challenged by the need to appropriately represent behavioral movement, which Rose et al. (2010) identified as an area for future research. Future efforts for fish modeling in NB include examination of data in other seasons, and relationships with finer-scale modeled data.

Table 3. Significant quantile regression coefficients for fall fish trawl data ( $\tau=0.9$ , \* $p<0.01$ , \*\* $p<0.0001$ ).

Measure (N=66)	DO	Chl $\alpha$	Depth	Eelgrass	Temp
Species richness (# species)				-0.4090	
Total abundance (# individ.)		-0.0634*			
Pelagics (# individ.)		-0.0732			
Demersal (# individ.)	-0.2514	0.1241			-0.1404
Pelagics (%)					
Invertebrates (# individ.)		-0.1278*	0.0861**		
Alewife ( <i>Alosa pseudoharengus</i> )				-0.1448*	
American lobster ( <i>Homarus americanus</i> )		-0.0926*		-0.1444**	
Atlantic cod ( <i>Gadus morhua</i> )	Not sufficient data				
Atlantic herring ( <i>Clupea harengus</i> )					
Atlantic moonfish ( <i>Selene setapinnis</i> )	0.2191		-0.0816	-0.2640**	
Atlantic silverside ( <i>Menidia menidia</i> )	Not sufficient data				
Atlantic menhaden ( <i>Brevoortia tyrannus</i> )	-1.2596**	0.0281	-0.0776	-0.1577*	
Bay anchovy ( <i>Anchoa mitchilli</i> )	0.2531*	-0.1106		0.0571	
Black Sea Bass ( <i>Centropristis striata</i> )	-0.6440**	0.1832*			-0.1876*
Bluefish ( <i>Pomatomus saltatrix</i> )	0.1987		0.1096**	-0.1266**	0.3127**
Butterfish ( <i>Peprilus triacanthus</i> )					
Horseshoe Crab ( <i>Limulus polyphemus</i> )				0.1478	
Longfin squid ( <i>Loligo pealei</i> )	0.1477*		0.0823**		
Northern kingfish ( <i>Menticirrhus saxatilis</i> )					
Red hake ( <i>Urophycis chuss</i> )					
Scup ( <i>Stenotomus chrysops</i> )	-0.2309*	0.1307**	-0.0426**	-0.0803**	-0.1446**
Skate (Little Skate, <i>Leucoraja erinacea</i> and Clearnose Skate, <i>Raja eglanteria</i> )			0.0861		
Spotted Hake ( <i>Urophycis regia</i> )			0.0537**		
Striped Sea Robin ( <i>Prionotus evolans</i> )	-0.2560*	0.0252*		0.0736*	

Summer flounder ( <i>Paralichthys dentatus</i> )	-0.0629*		
Tautog ( <i>Tautoga onitis</i> )	-0.2626		
Weakfish ( <i>Cynoscion regalis</i> )	-0.0629*		
Windowpane flounder ( <i>Scopthalmus aquosus</i> )	-0.3732*	0.1375*	
Winter flounder ( <i>Pseudopleuronectes americanus</i> )	-0.3296*	0.1335	-0.1720**

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### 3.6 Foodweb dynamics

In a comprehensive view of NB, is it important to consider the level of the foodweb. For example, as the amount of N entering the bay and the subsequent growth of phytoplankton is reduced through management activities, there is a potential for the available food for fish and shellfish to be reduced as well. Alternatively, shellfish can be considered as a possible avenue for reducing the N in the bay. The relationship between nitrogen in the water and the abundance of fish and shellfish needs to be delineated to assist in managing these resources. Ecopath with Ecosim (EwE) is one type of model that is based on the trophic structure of the functional groups in a system. It has been used several times in the NB system, originally by Monaco (1995) and later by Byron (2010). The model that was developed for this purpose was built from Byron’s model investigating the carrying capacity of the bay for oyster aquaculture. Ecopath is a model used in fisheries and ecosystem applications where trophic interactions and fisheries management strategies are of interest. It utilizes a static, mass-balanced approach with trophic interactions with biomasses that are reported on an annual average basis. Input parameters for Ecopath are divided into trophic groups, and, given information on the status of the current system (i.e., the biomass (B), production (P), and consumption (Q) for each trophic group), other criteria can be calculated and used to formulate a model of flow through the system. Examples of models using Ecopath include Christensen (1995), Christensen and Pauly (1993), and Monaco and Ulanowicz (1997). It has been used in nearby estuaries of Long Island Sound (Nuttall et al. 2011) and Chesapeake Bay (Cerco et al. 2010). Ecosim is a time-dynamic simulation version of Ecopath, which allows the system to be driven by a time series of N.

Here the Ecopath application of Byron (2010) was updated, diagnostics were run according to Link (2010), and Ecosim (version 6.4.3) was used to run simulations of reduced N. Trophic groups first defined in Monaco and Ulanowicz (1997, Table 4) and used in Byron (2010) were modified by: removing cultured oysters; adding groups for softshell clams, quahogs, mussels and oysters, and their associated input parameters (B, P/B, Q/B, and ecotrophic efficiency, EE); and adjusting inputs for phytoplankton, benthic deposit feeder, and benthic suspension feeder biomasses (Table 4). The shellfish groups were added to allow the ability to consider the effects of shellfish on phytoplankton, as a restoration option. Benthic suspension feeder and benthic deposit feeder biomass were from the U.S. EPA’s Environmental Monitoring and Assessment Program (EMAP) data from NB from 1990 through 1994 taken using a stratified random design applied to the bay (<http://www.epa.gov/emap2/html/data/>). Several parameters that Byron adjusted for balancing were reset to the original values before balancing to test

Table 4. Trophic groups used in EwE modeling, Biomass (B, g/m<sup>2</sup>), Ecotrophic Efficiency (EE), Production to Biomass ratio (P/B), Consumption to Biomass ratio (Q/B), and diet composition changes (%) used in Byron (2010) based on Kraeuter and Castagna (2001).

Code	Trophic Group	Principal species B	EE <sup>e</sup>	P/B <sup>e</sup>	Q/B <sup>f</sup>	% Diet composition changes			
						IC	Qua	MO	SSC
BA	Benthic algae		0.9	80					
BDF	Benthic Deposit Feeders	worms	5.05 <sup>c</sup>	0.9		0.2			
BIC	Benthic Invertebrate Carnivores	lobsters and crabs	7.82 <sup>d</sup>						
BSF	Benthic Suspension Feeders	Shellfish, worms	16.81 <sup>c</sup>	0.82	2				
CF	Carnivorous fish								
Det	Detritus						0.1	0.1	0.1
IC	Invertebrate Carnivores	squid and ctenophores							
Meso	Mesozooplankton				25				
Micro	Microzooplankton		0.95	140					
MO	Mussels and Oysters <sup>b</sup>	<i>Mytilus edulis</i> , <i>Crassostrea virginica</i>	0.23 <sup>b</sup>	0.95	0.15	2.18			
Para	Parabenthos	<i>Crangon</i> , <i>Palaemonetes</i> shrimp							
PB	Pelagic bacteria								
PF	Planktivorous fish					0.15			
POC	Bacterial Sediment Particulate Organic Carbon				1190		0.1	0.1	0.1
PP	Phytoplankton		1854.8 <sup>a</sup>				0.8	0.8	0.8
Qua	Quahogs <sup>b</sup>	<i>Mercenaria mercenaria</i>	0.71 <sup>b</sup>	0.95	1.02	5.1			
SSC	Soft shell clams <sup>b</sup>	<i>Mya arenaria</i>	0.17 <sup>b</sup>	0.95	0.45	2.18			

<sup>a</sup> derived from the average concentration of surface and bottom chlorophyll a in NB for 2010 and converted to grams of carbon per meter squared using the average depth of NB and the relationship of 42 g carbon/g chlorophyll a. Source: <http://www.gso.uri.edu/phytoplankton/#Data>

<sup>b</sup> Atlantic Coast Landings; <http://accsp.org/>

<sup>c</sup> From U.S. EPA's Environmental Monitoring and Assessment Program data from NB from 1990 through 1994 in a stratified random design applied to the bay <http://www.epa.gov/emap2/html/data/>

<sup>d</sup> original Byron 2010 data

<sup>e</sup> Christensen et al. 2009

<sup>f</sup> Nuttall et al. 2011

whether they met conditions in the shellfish model. Ecotrophic efficiencies (EE) are not needed if enough input parameter values are added, but can be added for more rigor. Values from Christensen et al. (2005) were utilized for available trophic groups.

The validity of the data used for the model was assessed by using pre-balancing diagnostics recommended by Link (2010) and sensitivity analysis. Link summarizes these diagnostics into five areas of comparisons: 1) biomass across taxa and across the trophic level, 2) biomass ratios, especially for predator/prey relationships, 3) vital rates (Q, P, respiration), 4) ratios of vital rates compared across taxa, and 5) total production and consumption. These diagnostics were also used by Byron et al. (2011). Examples of these diagnostics include a biomass comparison across trophic levels that showed a decrease in biomass as the trophic level increased from phytoplankton to carnivorous fish (Fig. 13a), and a comparison of vital rates across taxa (Fig. 13b). Critical in the vital rate comparison is that consumption must be greater than production for each taxa, which is shown in Fig. 13b. Overall, this analysis showed that the criteria of Link (2010) were satisfied. Once the static portion of EwE was balanced, simulations were applied with Ecosim for a 30-year time period with a yearly timestep. We first ran a control simulation with no forcing functions, examined the simulation, and adjusted the default setting for defined predator/prey interactions--the switching power parameter, controlling how narrowly a predator's diet is defined. Increasing the value of the switching power parameter allows a predator to consume alternative prey when their preferred prey reaches low concentrations, keeping the prey from being grazed too heavily and adding a realistic diet substitution. Next, N forcing functions of +6%, -11%, -27%, and -50% were applied to the model (Vadeboncoeur et al. 2010), resulting in increased biomass of all trophic groups with increased N and decreased biomass of all trophic groups with decreased N (Fig. 13c). We also examined how an increase in quahogs affected the phytoplankton group (as a surrogate for bay water quality). Even with initial quahog biomass multiplied by 10, the response on phytoplankton was <1%.

The EwE model shows promise for simplifying systems-level trophic interactions. Within the software there are many variables that can be altered for specific systems. However that also means that there are many variables that need to be accounted for within the system, some of which may not be available. Availability of data can be a limiting factor. When using N as a forcing function in the shellfish model, the shellfish responded as might be predicted: lowering N lowers the available food and, hence, lowers the total biomass of the quahogs, softshell clams, and mussels and oysters. The magnitude of the change in N was reflected in the biomass also. To be used in a predictive capacity, though, more of the default variables need to be replaced with system-specific information. There are two components that would be very helpful in strengthening the NB model: first, a time series of historical data for biomass and N, and second, better information on the differences in diet and predator/prey interactions on different life stages for the trophic groups, which could be incorporated into EwE.

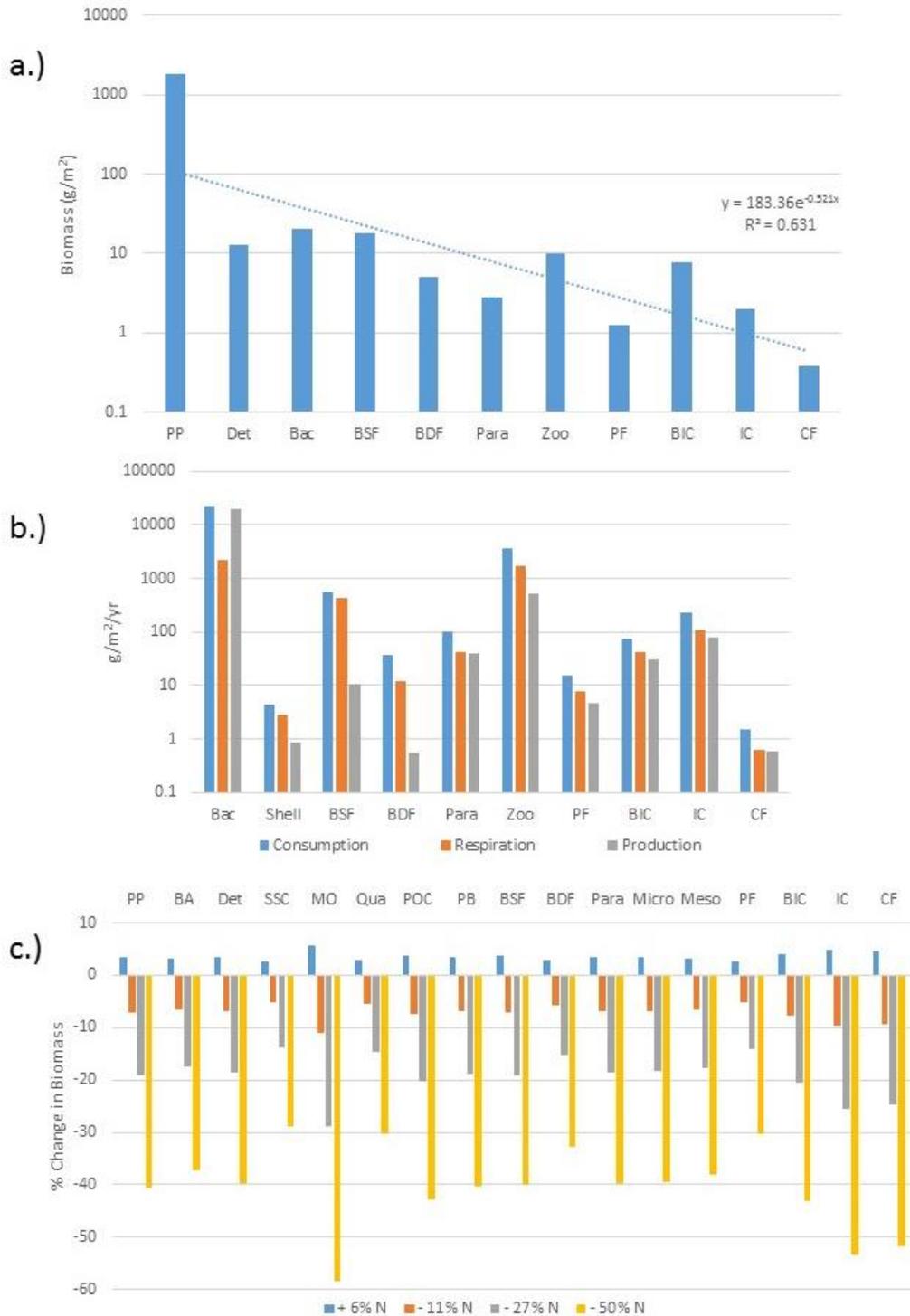


Figure 13. Results of the ECOPATH/ECOSIM model: a) Plot of biomass of functional groups, ordered by increasing trophic level; b) Vital rates in g/m<sup>2</sup>; and c) Simulated changes in annual biomass in % over 30 years by trophic group in response to changes in nitrogen in NB. Functional group codes are defined in Table 4 (additionally, Zoo (zooplankton) = Meso + Micro, Bac (bacteria) = PB + POC, and Shell (shellfish) = Qua + MO + SSC).

## 4. DISCUSSION AND FUTURE DIRECTIONS

### 4.1 Discussion of model application and transferability

Overall, we found that our set of models gave reasonable results. EFDC calibration for the year 2009 provided adequate results that compared favorably with field observations of temperature and salinity, as indicated by over eight skill parameters. EFDC is adequate for use in most water bodies and the methods developed herein can be adopted to facilitate the application of the models to new systems. Calibration of the WASP model has proven more challenging, in part due to a lack of appropriate data for observed nutrients and solar radiation in NB, as well as technical issues with the model, and lack of adequate WASP documentation. Also, issues arose with the model linkage between EFDC and WASP, requiring significant troubleshooting. The development of our mechanistic approach with EFDC-WASP has progressed in parallel with NOAA's Coastal Hypoxia Research Program modeling (Brush 2014, Vaudrey and Kremer 2015), which uses science-based empirical formulations for water quality calculations. Future work can include more extensive comparisons between the two approaches, with standardization of inputs, and comparison to the same output data, including characterization of past hypoxia from molybdenum (Boothman, personal communication, 2015). Seagrass models were relatively well-supported with data, and the combination of models provides a strong scientific framework for seagrass forecasts. Models for both benthic invertebrates and fish were able to detect broad patterns, but inherent variability made it difficult to find strong, consistent patterns.

A goal of this research was to consider the transferability of our coordinated modeling approach to other estuaries, particularly those within the National Estuary Program, for which eutrophication is a common problem. While the EFDC seems transferrable with some effort, we have found the EFDC-WASP approach to be challenging and a significant time investment. Although WASP is a mechanistic model, which can use default parameters for certain processes, it requires several site-specific inputs that need to be prepared before the models can be run. Consequently, additional water quality approaches could be considered as alternatives to achieve a more transferrable approach (e.g., Lehrter et al. in review, Abdelrhman 2015b). The bio-optical model and seagrass models will likely be transferrable; much of the data for application of the seagrass model to other U.S. estuaries is available through EPA's Estuary Data Mapper application ([www.epa.gov/edm](http://www.epa.gov/edm)). Empirical ecology models for invertebrates and fish, would ideally be based on data from the site of interest, but these should be relatively easy to develop. Rose and Allen (2013), in a review of marine ecosystem responses to climate change, noted that site-specific models were used for upper trophic levels in the 12 papers they reviewed; they note that there is a need for scaling between site-specific and regional models. Finally, the EwE model has been used widely for estuaries, and is therefore quite transferrable, although the lack of spatial detail and limited effects of stressors limit the applicability of this approach for more targeted use. The NB EwE application was based on a past similar approach, which simplified our efforts.

Despite the challenges of modeling difficulty and uncertainty, an integrated modeling approach such as ours provides opportunities for considering a broader understanding of estuarine systems, including an assessment of unintended consequences and feedbacks. For example, the dynamics of sediment diagenesis plays a significant role in determining nutrient concentrations. The mechanistic nature of the

hydrodynamic and water quality models, in particular, allow us the possibility of using these models for forecasting the results of future scenarios. Duarte et al. (2013) note that degradation and recovery may follow different pathways, and that recovery pathways may depend on ecosystem connectivity, and may vary among ecosystem components, which supports the need for a more mechanistic approach. Ideally, we would have used a more mechanistic approach for the full system, through the higher trophic levels, and “end-to-end” modeling approach (Rose et al. 2010), however, data availability in NB did not seem adequate to support this approach. The set of models can contribute to assessments of multiple stressors, and the relative importance of these, which is an important holistic view of the system. Multiple stakeholders across two states, and significant involvement with stakeholders is critical to the successful use of models in management and decision making (e.g., Fiksel et al. 2013). Additional considerations include the need for varying levels of model calibration needed to better inform policy deliberations, and the authorities and capacities of environmental management institutions needed to run models, and assess uncertainties. Additionally, in the view of climate change, “restoration” with a changing baseline requires new thinking.

#### ***4.2 Future application of scenario analysis***

Future management scenarios will include specific actions related to point and non-point sources of N, and estuarine restoration. Our efforts can build upon aspects of the work of Industrial Economics, Incorporated, IEC (2015), multiple interactions with stakeholders in EPA Region 1, and RIDEM to develop scenarios. IEC (2015) determined that future point source reductions at the larger WWTFs (e.g., WWTPs discharging >45,000 kg during summer months in 2014) to 3 mg/L could be considered. For nonpoint source scenarios in NB, IEC (2015) worked with stakeholders at RIDEM to develop scenarios to reduce the N content of residential fertilizer by 20%, and upgrade half of the Independent Sewage Disposal Systems (ISDS) in the watershed to systems with a removal efficiency of 30% from 2014 to 2025. It is also possible to consider the sewerage of these ISDSs. Additional nonpoint source scenarios are related to land use. Ross (2014) used the U.S. Department of Agriculture’s Soil and Water Assessment Tool (SWAT) to incorporate projected 2080 land use for NB taken from Tayyebi et al. (2013). This source projected that urban development decreased by 0.23% in the NB watershed, and in response, organic N and P increased only slightly by 3.22% and 3.20%, respectively, and nitrate loading decreased by 0.63%. This projection, or nearer-term projections, could be used from this model as input to the modeling system here. Scenarios for future land use are available for 2025 from the Rhode Island Statewide Planning Program (2006). Similar watershed models have been applied to subbasins within NB (e.g., Randhir and Tsvetkova 2011). Additional practices of interest in RI are related to use of green infrastructure, wetlands restoration, and improved stormwater management, including benefits of low-impact development (Horsley Witten Group, Inc. 2010). EPA’s Community Multiscale Air Quality model (CMAQ, Byun and Schere 2006) can provide projected changes in the air deposition of N, which is expected to decline in the future. CMAQ has been used in the calibration of USGS East Coast watershed models (USGS 2013), and it can be directly incorporated into water quality models like WASP. Finally, shellfish restoration can also be considered as a mechanism for removal of N in the estuary (Rose et al. 2014).

We could also consider effects of climate change, which is the major factor affecting most of the hydrodynamic forcing functions. Increased atmospheric temperature can cause changes in snow melt, freshwater flow, sea level, precipitation, evaporation, water salinity, and wind patterns. For a climate change scenario, sea level rise has to be estimated, and the mean sea level (MSL) has to be adjusted accordingly, which will impact the water depth (bathymetry) in the Bay. Assuming that sea level rise will continue at the same global rate of  $3.3 \pm 0.4$  mm per year (Nicholls and Cazenave 2010), after  $n$  years the increase in MSL will be  $\sim 3.3n$  mm. Tide elevations, at the seaward open boundary, have to be calculated from the major astronomical tidal constituents and be referenced to the new MSL. The time series of the typical yearly freshwater inflows from the eight rivers have to be adjusted (multiplied by a factor) to account for future wet/dry weather, or estimated for future applications of watershed models (Ross 2014). Changes in extreme values may be inferred from statistical analysis of present and expected future distributions. Salinity concentration at the open boundaries can be similar to current conditions. Water temperature increases throughout the year are expected at the seaward open boundary and also in river inflows, which can be predicted using recently-developed approaches (e.g., Wehrly et al. 2009). Evaporation rate will be calculated internally by the model. Climate models would be expected to provide: relative humidity, atmospheric pressure, rainfall, shortwave solar radiation, cloud cover, wind speed, and wind direction. EFDC can easily reproduce predictions for NB with future estimates of these forcing functions for future scenarios. Rose and Allen (2013) note the potential benefits of some standardization of global climate change scenarios used in marine ecosystem modeling.

#### ***4.3 Water quality model selection uncertainty based on observations***

The water quality modeling community has produced a large variety of modeling approaches for estuaries, including NB. Because most of these models contain complex biological models, typically as sets of differential equations, and also contain or are coupled to high resolution physical models, formal statistical relationships between observations and models are typically lacking. The challenges this produces for assessing model uncertainty and comparing alternative models combined with the problems caused by the tendency toward complex over-parameterized models are discussed in Ganju et al. (2015). Using the Nu Shuttle data (see section 3.5) in combination with medium complexity water quality models, we are applying Bayesian techniques (e.g., Monte Carlo Markov Chains, Metropolis-Hastings algorithms, and eventually Kalman filters) to calculate model likelihoods and to gain access to formal inference methods. These approaches are computationally expensive and may often require static representation of hydrodynamics, but would improve the transparency of model strengths and limitations for decision-making. As a simple example, it is tempting to incorporate zooplankton as a top-down control on important rates in a water quality model (see next section), but before adding this complexity, a formal approach to model inference would be needed to determine whether zooplankton rates can be estimated well enough from available data to overcome the statistical penalty (i.e., trade-off in accuracy vs. precision) of adding new parameters to the model. The goal in the inference step is to identify the most complex model alternative that the available data will support. Thus, in those cases where the best model, given the data, is still too coarse to support decision-making, the clear and logical solution may be that better monitoring and the continuation of adaptive management actions are needed.

#### **4.4 Future Directions: Role of zooplankton**

Based on a comprehensive study of food web structure and hypoxia, Capriulo et al. (2002) suggested that unconsumed copepod biomass was an important contributor to hypoxia in Long Island Sound. Such perspectives on the influence of grazers or higher trophic levels on hypoxia are rare in the water quality research community, but zooplankton have been shown to exert strong control over the onset, magnitude, and termination of phytoplankton blooms in temperate estuaries (George et al. 2015). This is important to water quality because phytoplankton blooms fuel the metabolic activity that consumes DO, produces carbon dioxide, and lowers pH in bottom waters. Since blooms begin when phytoplankton growth exceeds grazing, it has been suggested that milder winters would reduce or delay the onset of phytoplankton blooms by allowing winter persistence of larger grazer populations (Oviatt et al. 2002, Oviatt 2004, George et al. 2015). Based on modeling studies in coastal waters of the north Atlantic, Townsend et al. (1994) demonstrated that annual variability in the timing of phytoplankton blooms could be explained by the temperature effects of annual variations in solar radiation and cloudiness on grazers. Relatedly, Rice et al. (2015) described strong associations between increasing seawater temperature and characteristics of the copepod community over the past half century in Long Island Sound.

There have been numerous short-term or species-specific studies of zooplankton in NB but it is unclear whether the trends observed in nearby estuaries are also occurring in NB. In a review of mid-20th century field studies, Durbin and Durbin (1988) concluded that changes in sampling methods and net size prevented analysis of trends through time for small taxa (bivalve larvae, copepod nauplii, polychaetes, rotifers). For larger zooplankton forms in which trends could be analyzed, no trends were found for the period covered by their review (1950 to 1986). The 1998-2000 Nu Shuttle survey collected zooplankton data, but the data are considered unsuitable for analysis (Chris Melrose, NOAA, personal communication). The URI Phytoplankton Survey reported weekly zooplankton counts for Oct 2001 to Jan 2005. Within this 3.25 year period, the strong variability in both the timing and number of seasonal peaks illustrates the need for longer term data. These various data limitations frustrate efforts to identify environmental drivers of zooplankton communities in NB and the potentially strong effects they may have on nutrient cycling and production.

#### **4.5 Future Directions: Shellfish Modelling**

Additional modeling could be conducted to include effects of nutrient loading on shellfish, primarily quahogs (*Mercenaria mercenaria*). Grear et al. (in prep) have developed a population model for quahog response to pH changes related to ocean acidification, which was found to be related to eutrophication in NB (Wallace et al. 2014). The quahog model is constructed from size-abundance survey data from northeast estuaries using modeling methods developed in Grear et al. (2011). Dose-response relationships for carbon dioxide are used with this model to predict population-level effects of nutrient-enhanced coastal acidification. Such models can be used in conjunction with other stress-response relationships and can also form the basis for possible extension to source/sink analysis of bed closures and harvest sustainability. Weiss et al. (2002) have identified positive growth rates in quahogs and

softshell clams (*Mya arenaria*) associated with N enrichment, so it would be possible to incorporate these relationships into the population model.

#### **4.6 Future Directions: Economics**

Model outputs for shellfish and finfish could be used as input for economic valuation approaches. It is likely that there would be changes to both recreational and commercial shellfish harvests in NB, primarily through opening currently closed areas because of reductions in pathogens. There are data on commercial shellfish harvests and values that could be applied to projected changes in open shellfishing areas and harvests; however, data are not available on recreational shellfishing uses and harvests in the Bay, so without gathering primary data it will not be possible to value recreational shellfishing. Additional water quality modeling would be needed to assess changes in pathogens under future scenarios; it is possible to represent pathogens within the WASP model using a passive tracer approach. There are impacts on both commercial and recreational finfish; however, the majority of commercial fishing occurs outside NB (Desbonnet and Lee 1990). To estimate the producer surplus, or value, of commercial catch, cost estimates or typical profit percentages (available in Pacheco and Tyrrell 2003), as well as information on regulated catch limits, would be needed. This information could be used with our model predictions of changes in fish abundance, assuming that catch will change by the same percent as abundance, and is not already at regulated limits. To produce estimates of value for changes in recreational finfish, we could combine the estimates of change in abundance of recreational species from the models developed here with existing studies of the value of recreation catch, assuming that changes in abundance would be proportional to changes in catch rate.

Economic valuation approaches could also be used for beaches and water clarity. Valuation of beaches could be conducted by using prediction of bacteria populations to predict changes in beach closures. This information would be combined with the existing values of beach visits (e.g., Opaluch et al. 1999, Kline and Swallow 1998) and data on the number of visitors obtained from beach managers, if available. The ability to estimate beach values and recreational fishing and shellfishing values relies on availability of existing valuation and participation data. If such data are not available, it would be necessary to collect primary recreational data using survey methods. A longer-term goal would be to incorporate predictions of seaweed on the beach, especially sea lettuce (*Ulva* spp.) into beach valuation. Changes in property values due to changes in water clarity could be estimated using a recently conducted study of how water clarity affects property values around Narragansett Bay, applying the study results to model outputs for changes in clarity by location (Liu 2014).

More broadly, a full assessment of the impacts of policies aiming to address N pollution would include how the costs and benefits are distributed across the stakeholders in the watershed. Addressing problems associated with excess N pollution should be considered in a larger systems context. A systems approach would consider environmental governance authorities and capacities to implement solutions that could include: 1) reduction in sources of excess N and co-pollutants, 2) further investments in wastewater and water treatment, 3) innovative management practices involving recycling of wastewater and excess nutrients, 4) protection and enhancing watershed structures and functions in wetlands and riparian zones that help attenuate nutrient fluxes, and 5) restoring watershed connectivity

by removing obstructions to flowing water such as unneeded dams that can contribute to water quality and habitat degradation, and increase flooding risks. This approach would consider both the collection of needed revenue to pay for combinations of interventions, and the spatial distribution of benefits, including who pays, and who benefits. In RI, some of the burden of WWTF upgrades has been supported by State-wide bond issues, but these bonds need to be paid, and many of these costs are born by WWTF and water supply rate payers. Further adjustments to environmental finance policies may be needed to avoid unsustainable municipal, and household debt burdens, and achieve cost effective, socially acceptable and environmentally sound solutions.

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