ATTACHMENT 4

Scientists struggle to save seagrass from coastal pollution

By MICHAEL CASEY and ANDREW SELSKY

DURHAM, N.H. (AP) — Peering over the side of his skiff anchored in the middle of New Hampshire's Great Bay, Fred Short liked what he saw.

Just below the surface, the 69-year-old marine ecologist noticed beds of bright green seagrass swaying in the waist-deep water. It was the latest sign that these plants with ribbon-like strands, which had declined up to 80% since the 1990s, were starting to bounce back with improved water quality. Seven rivers carry pollution from 52 communities in New Hampshire and Maine into the 1,020-square-mile (2,650-square-kilometer) watershed for the bay.

"It actually looks better than it did last year at this time and better than has in many years," said Short, a noted seagrass expert who coordinates the monitoring of 135 sites around the world from his University of New Hampshire lab.

"You see here," he said, glancing into the water. "It's nearly 100% cover. You look to the bottom. You can't see the mud. You just see eelgrass. That is as dense as it gets. That's a really good sign."

Seagrass beds in New Hampshire and along shorelines around the world are important because they have been found to provide food and shelter for fish, shellfish and sea turtles. They also blunt the impacts of ocean acidification, reduce coastal erosion and keep the water clean by filtering out excessive nutrients.

Their comeback in the Great Bay gives hope for recovery elsewhere.

The more than 70 species of seagrasses are among the most poorly protected but widespread coastal habitats — more than 116,000 square miles (300,000 square kilometers) have been mapped, though there could be 10 times that. They are found along coastlines around the world except Antarctica's.

Seagrasses, which cover less than 0.2% of the world's oceans, store twice as much carbon in a given area as temperate and tropical forests, a study by the United Nations-affiliated Blue Carbon Initiative found. But seagrass meadows in many places are imperiled by coastal development, overfishing, runoff from farm waste, and the growing threat from climate change. They have declined roughly 7% annually since the 1990s, a peer-reviewed study found. That is on par with the declines of tropical rain forests and coral reefs.

Some seagrass declines have occurred with stunning speed. Central California's scenic Morro Bay has lost more than 90% of its eelgrass since 2007.

"It's certainly not a pretty picture and may not get any prettier because of the climate change issues we are all dealing with," said Virginia Institute of Marine Science's Robert Orth, a professor who has studied seagrass for decades. "These plants are very sensitive to environmental characteristics — water quality, temperature."

In parts of the United States and other developed countries, there is growing recognition of the importance of seagrass and its sensitivity to nitrogen-rich runoff from sewage treatment plants and other sources. Too much nitrogen can spike algae growth, which clouds the water and blocks the sunlight seagrass needs to grow.

"We think this is a problem that has to be solved," said Ken Moraff, water division director for U.S. Environmental Protection Agency's New England region. Communities around the Great Bay have spent about $200 million to upgrade wastewater treatment plants, resulting in some cutting nitrogen releases by up to 70%, according to EPA and officials in several Great Bay communities.

"We've seen other areas where reductions in nitrogen do result in the ecosystem starting to come back," Moraff said.

Studies have documented seagrass recovery in Boston, Tampa Bay and Long Island Sound.

Boston Harbor was once known as the dirtiest harbor in America because most waste went into the waters untreated.

Then the state invested $3.8 billion in a treatment facility on Deer Island that was completed in 2001 and allowed wastewater to be piped almost 10 miles (16 kilometers) out into Massachusetts Bay. The state has documented an 80% decline in nitrogen levels in the harbor.

Tay Evans, a seagrass specialist with the Massachusetts Division of Marine Fisheries, said there has been a corresponding 50% increase in eelgrass from 2006 to 2016. Now seagrass is growing in Governors Island Flats near Logan International Airport.

"It was astounding me," Evans said. "I dove there and saw what we would call a moonscape that was just mud. You come back and it's a lush meadow and then you're going to see all the animals — the winter flounder swimming through there, lobster walking around."

In Tampa Bay, seagrass beds are reaching levels not seen since the 1950s.

More than $2.5 billion was spent on upgrades to sewage treatment plants, measures to address stormwater runoff and curbs on nitrogen emissions from power plants. That resulted in two-thirds less nitrogen going into the bay compared to the 1970s, according to Ed Sherwood, executive director of the Tampa Bay Estuary Program.

Seagrass area nearly doubled to about 63 square miles (163 square kilometers). The water quality improvement along with a gill net ban has contributed to the recovery of several fish species including striped mullet, red drum and spotted sea trout.

But such stories can't mask the challenges.

Some recoveries such as those in parts of the Boston Harbor and the Great Bay are at risk from dredging. In other places, such as Chesapeake Bay, a decline in nitrogen has benefited many underwater plants but not eelgrass, which has declined since the 1990s.

Brooke Landry, a Maryland Department of Natural Resources biologist who monitors the bay's underwater vegetation, said that eelgrass, a coldwater species, may be more susceptible to heat events as seen in 2005 and 2010 — or to overly cloudy waters in the bay.

Scientists are also struggling to understand why eelgrass hasn't come back in California's Morro Bay.

"We have some theories," said Jennifer O'Leary, who studied the bay as a California Sea Grant researcher. She said the eelgrass decline has occurred in waters that are warmer, saltier, cloudier and less oxygenated than the bay's mouth, where eelgrass did well.

In New Hampshire, eelgrass has recovered about 20% in parts of the Great Bay, though it hasn't returned to several areas.

Some conservationists argue that bayside communities need to further reduce nitrogen releases through tens of millions of dollars in treatment plant improvements.

But several towns counter they have already made significant upgrades to their plants and that they should focus on cheaper options.

"You want to put your money where it's going to do the most good," said Portsmouth Deputy City Attorney Suzanne Woodland.

The EPA is considering allowing communities to hold off on treatment plant upgrades while they try to reduce nitrogen from stormwater runoff and septic tanks. Some communities upgraded sewage treatment voluntarily while others made upgrades to settle EPA enforcement actions.

Walking to his lab with his latest seagrass samples, University of New Hampshire's Short says that approach allows communities to avoid the painful steps necessary to ensure full recovery.

"It's easier to say no, no let the next guy pay for it," he said. "But now we are at the point where it's causing a huge issue. You don't have to believe the science. Go out there and look."

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Selsky reported from Salem, Oregon.

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Follow Casey on Twitter: @mcasey1, and Selsky on Twitter: @andrewselsky

ATTACHMENT 5

2-28-2020

Eelgrass Distribution in the Great Bay Estuary and Piscataqua River for 2019: Final Project Report submitted to the Piscataqua Region Estuaries Partnership

Seth Barker

Eelgrass Distribution in the Great Bay Estuary and Piscataqua River for 2019

Final Project Report submitted to the Piscataqua Region Estuaries Partnership

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February 28, 2020

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Abstract

Eelgrass distribution in Great Bay, Little Bay, and the Piscataqua River Estuary was mapped from aerial photography acquired on August 2, 2019. The total area of eelgrass beds with 10% or greater cover and a polygon area equal to or greater than 100 square meters was 625.9 hectares or 1677.7 acres. Eelgrass polygons were coded for Assessment Zone (http://www.granit.unh.edu/data/search?dset=greatbayestuaryassessmentzones_current) location and the results reported for each zone. The largest concentration of eelgrass was found in Great Bay with lesser amounts in the vicinity of Portsmouth Harbor. The total area of eelgrass beds has increased by 131 acres which is approximately an 8.5% increase from 2017 and very nearly equal to that mapped in 2013. This number includes some areas where both eelgrass and widgeon grass were present. As noted, in addition to eelgrass, widgeon grass was mapped in areas where field work confirmed its presence. There were 257.4 acres of widgeon grass (and eelgrass combined) identified and this was found primarily in Great Bay.

Introduction

The report that follows provides details of the mapping of eelgrass distribution in Great Bay, Little Bay, the Piscataqua River, Portsmouth Harbor and a small portion of the Atlantic Coast for the year 2019. In addition to eelgrass, widgeon grass and a mix of widgeon grass and eelgrass was mapped in areas where field visits confirmed the presence of widgeon grass. Aerial photography was obtained on August 2, 2019 and was followed by field work in September and early October to establish signatures for photointerpretation and to aid in the accurate mapping of eelgrass distribution. At the time of this report, this mapping is the latest regional documentation of eelgrass beds in the area. The project area is described and illustrated in the Appendix A.1.

Methods

Procedures followed the guidelines articulated in the project Quality Assurance Project Plan (QAPP), which can be found at: https://scholars.unh.edu/prep/431/ Mapping of the distribution of eelgrass was based on photointerpretation of aerial photography obtained on August 2, 2019, under a contract with Cornerstone Mapping, Inc, Bangor, Maine. Preliminary, georeferenced images were made available at the end of August 2019 and were used for field logistics. This initial draft photography did not have the locational accuracy of the final photomosaic and had not been color balanced but provided sufficient detail to locate features of interest, conduct initial mapping, and to select stations to be visited. Stations were selected in Great Bay, Little Bay, the Piscataqua River, Portsmouth Harbor and the Atlantic Coast and field visits by boat were made in the September/October time period. The boat and operator were provided by PREP for assistance with field verification. Location of observations was recorded as track files using high accuracy Trimble GeoXT GPS equipped with an external antenna. Since there can be a variety of photographic signatures and signatures change from year to year and with conditions at the time, field stations are important for the understanding of the nature of the signatures. The water-based field visits were made on September 5,11,12,18,19, 23, and October 2. In addition, several stations were visited on foot on October 2.

A total of 165 numbered stations and several unnumbered stations were visited (Figure 1). Subsurface observations were made with a Seaviewer drop camera equipped with a surface monitor at most of these stations. In a few cases, the bottom could be clearly viewed without the use of the drop camera. Video recordings were made at most but not all stations. Observations were made and videos recorded as the boat either drifted or motored at low speed over a station and one or more observations were recorded on a field sheet (Appendix A.2). Observations included the presence of eelgrass, whether eelgrass cover was judged to be equal to or greater than 10 % (Appendix A.3), the presence and type of macroalgae (where possible), and in some cases, substrate. The time of the observation was recorded and used in conjunction with the time of GPS observations which were recorded as points in GPS files. In most locations, a video recording was made which was time stamped. This allowed for location specific review at a later date in a GIS with the GPS file providing a guide to the approximate location. A total of 380 unedited video files of a minute or less were recorded and are provided as part of the ancillary data.

The final photomosaics were received from Cornerstone Mapping in December, 2019. These were added to a GIS along with field information and other data layers to aid in photointerpretation. Eelgrass beds were first outlined and screen digitized using the GIS software package, QGIS, and saved to an ESRI shape file. Final digitizing was generally done at a screen scale of 1:1000 or less. The projection used was New Hampshire State Plane, NAD83, and the units were feet (EPSG:102710; https://epsg.io/102710).

During the initial digitizing process, all eelgrass that was easily discerned was digitized in a polygon file. After beds were outlined to form polygons, areas with less than 10% eelgrass coverage as visible from the aerial photography were then deleted from the GIS file leaving the polygons of 10 percent cover or greater. Also, polygons of less than 100 square meters were also deleted. Database file attributes for 2019 are as follows: "id", a unique consecutive number; "Hectares", the area of the polygon in hectares; "Acres", the area of the polygon in acres; "Year", equal to 2019, the year of the aerial photography, "Label" for the assessment zone, and "type" to distinguish between polygons mapped as eelgrass, widgeon grass, or both. Additional details are provided in the project metadata file.

The QAPP describes a process by which the accuracy of the digitized polygon boundary is verified in the field. To meet this requirement a total of 12 points were recorded using the Trimble Geo XT on 9-12-2019 and an additional 12 points were recorded on 9-23-2019 (Figure 3). These points represent the location were eelgrass was first observed using a drop camera as the boat traversed from the navigation channel to shallow depths. The distance from this point to the polygon boundary was measured with the "measure tool" in QGIS and reported in Table 1.

During the digitizing process and when the final file was produced, the topology of the shapefile was checked using the QGIS topology routine. The topology rules enforced were no gaps, no duplicates, no overlap, no invalid geometry, or no multi-part geometry.

Results and Discussion

The distribution of eelgrass for 2019 is shown in Figure 2 along with higher resolution maps at 1:24000 scale (Appendix A.4, Figures 1-3)

The total area of eelgrass mapped in the entire project area was 1677.7 acres. This has been broken down by Assessment Zone and shown in Table 2. As in past years, Great Bay had by far the greatest amount of eelgrass, 1450.6 acres. Little Bay had 20.3 acres. The Portsmouth Harbor zone had 87.1 acres. The Little Harbor and Back Channel zone had 41.9 acres. The Gerrish Island area had 58.4 acres with additional area for these beds reported in both the Atlantic Coast, Piscataqua River, and other Assessment Zones.

Widgeon grass was found in abundance at several locations in Great Bay. The densest concentrations were found in a swath from Woodman Point to Pierce Point. Large beds were also found extending from Strongs Landing to Shackford Point. The only other location where it was observed was the head of Spinney Creek. Though it very likely is present at low density throughout the estuary it was not found in sufficient density to map at other locations where field visits were carried out. The lack of a clear signature also contributed to limitations in mapping. Widgeon grass was found growing alongside macroalgae in shallow and intertidal areas and was mixed with eelgrass in other shallow locations. It is assumed but not know that freshwater input is one of the factors that favored widgeon grass growth in these locations. Though widgeon grass has been found repeatedly in the vicinity of the mouth of the Winnicut River, this is the first year that it has been included in this series of mapping efforts.

It is felt that areas of dense eelgrass that contained macroalgae could be adequately differentiated from dense stands of only macroalgae or macroalgae and widgeon grass. In locations where eelgrass was not dense (10-30% for example), it was often difficult to differentiate eelgrass from other vegetation and required field verification. In many locations macroalgae was found growing in dense concentrations around the stems of eelgrass plants. In this situation, dense eelgrass was visible in the aerial photography but the macroalgae was often much less evident or not detected.

As in past years, oysters provided another signature that was clearly detected in some locations. If a large number of oysters was present on the surface of a mud bottom, the signature was distinctive. If found in the presence of eelgrass but not macroalgae, the eelgrass signature was clear and to a lesser extent oysters could be detected. However, if oysters were present along with macroalgae and eelgrass, the signature was confounded such that only the predominate feature could be discerned. The hard bottom and different types of macroalgae also produced signatures that were difficult to separate from that of eelgrass and therefore required field verification.

The work done to provide information on the accuracy of mapping at polygon boundaries was productive but the procedure used can be improved upon. Table 2 contains measurements in meters of the difference between the observed and mapped edge. The mean and standard deviation of these measurements was within the QAPP specification of 5 meters. A graphic showing the location of points in Great Bay is shown in Figure 3. Depending on wind and tide the velocity of the boat varied at time during this exercise. The GPS antenna was not a constant distance from the camera location, a point that was not accounted for in the analysis and any delay in recording the point resulted in additional error in the recorded point as the boat drifted. These things combined make this estimate conservative at best. It also must be noted that the line drawn for the polygon boundary smooths the boundary and does not take into account the very irregular boundary that would be observed on the ground. This makes it an estimate at best and though the results of work carried out on these two days is encouraging there should be a review of this specification in the QAPP and possible revision.

Figure 1. Field stations and GPS track logs.

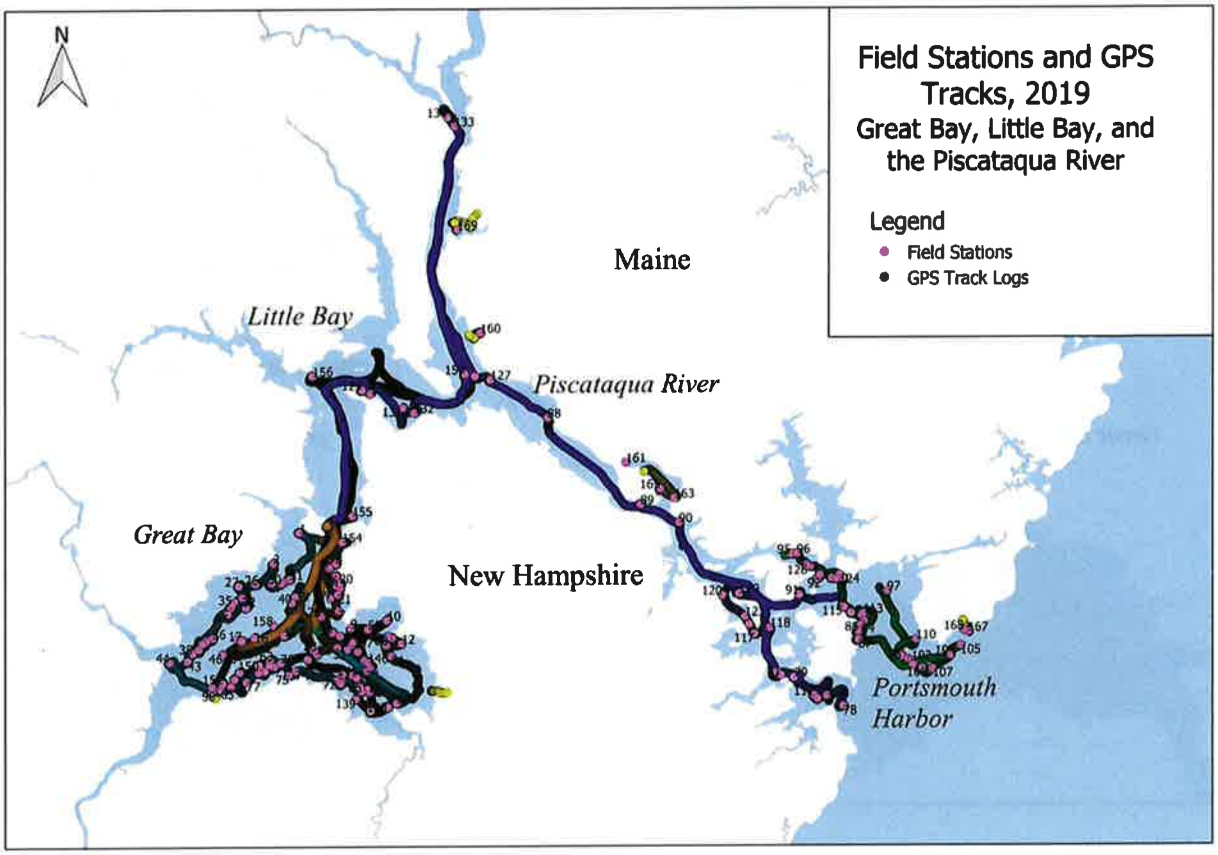


Figure 2. Distribution of eelgrass, 2019.

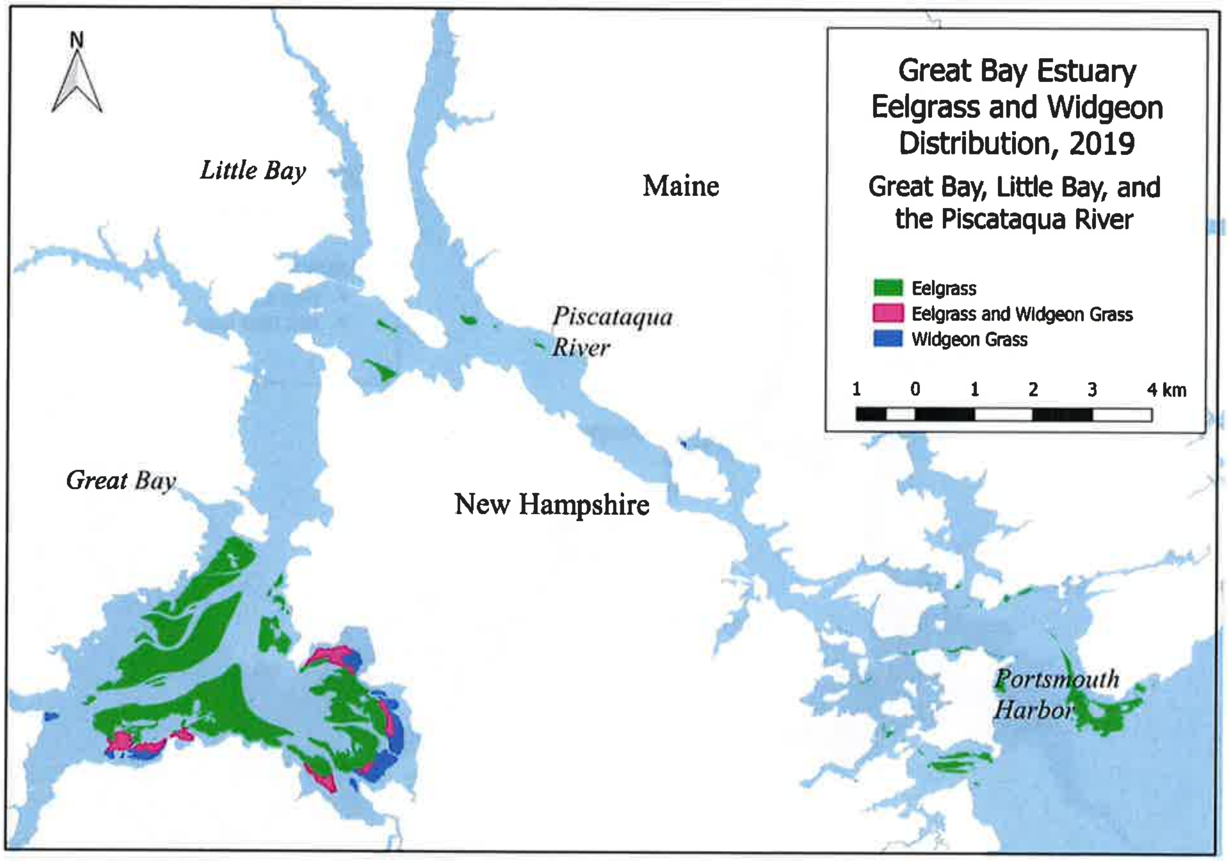


Figure 3. Screen shot showing location of edge check points

Table 1. Results of polygon edge check

9-12-2019, West Side of Channel					**9-23-2019, West Side of Channel**		
Point ID	**Distance(m)**	**Relative Position**			**Point ID**	**Distance(m)**	**Relative Position**
9	7.4	inside			114	8.7	inside
10	2.9	inside			115	3.7	inside
11	2.6	inside			116	5.4	inside
12	4.7	inside			117	0.1	outside
13	5.7	inside			118	4.2	inside
14	6.3	inside			119	0.1	outside
9-12-2019, East Side of Channel					**9-23-2019, East Side of Channel**		
Point ID	**Distance(m)**	**Relative Position**			**Point ID**	**Distance(m)**	**Relative Position**
2	0.4	inside			105	3	inside
4	1.1	inside			108	4.3	inside
5	1	inside			110	8.7	inside
6	1	inside			111	6.9	inside
7	3.9	inside			112	4	outside
8	4.6	inside			113	1.4	outside

Mean = 3.84 meters
SD = 2.545
95% Probability
3.84 ± 1.075 meters

Table 2. Area of polygons by Assessment Zone

Area in Acres – 2019					
Assessment Zone	Eelgrass (EG)	EG and WG	WG	**Total Eelgrass**	**Total**
Atlantic Coast	1.05			**1.05**	**1.05**
Gerrish Island Beds	58.43			**58.43**	**58.43**
Great Bay	1344.99	105.57	143.44	**1450.56**	**1594.01**
Little Bay	20.34			**20.34**	**20.34**
Little Harbor/Back Channel	41.89			**41.89**	**41.89**
Lower Piscataqua River North	8.57			**8.57**	**8.57**
Lower Piscataqua River South	3.55			**3.55**	**3.55**
Odiorne Point Beds	1.27			**1.27**	**1.27**
Portsmouth Harbor	87.08			**87.08**	**87.08**
Sagamore Creek	1.51			**1.51**	**1.51**
Spinney Creek			1.49		**1.49**
Upper Piscataqua River	2.18			**2.18**	**2.18**
Winnicut River		1.29	2.57	**1.29**	**3.87**
Total	**1570.87**	**106.87**	**147.50**	**1677.74**	**1825.24**

EG = Eelgrass
WG = Widgeon Grass

Appendix

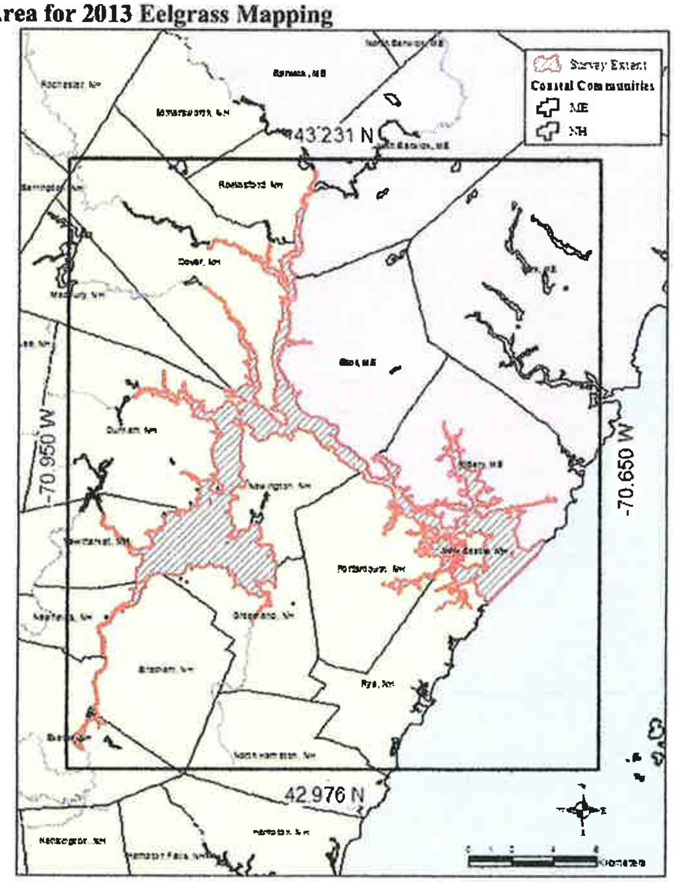
A.1 Description of study area.

The description from the 2019 QAPP is as follows:

A5 – Problem Definition/Background

Submerged aquatic vegetation (SAV), including seagrasses such as eelgrass (*Zostera marina*) and widgeon grass (*Ruppia maritima*) are essential to estuarine ecology because they filter nutrients and suspended particles from water, stabilizes sediments, provide food for wintering waterfowl, and provide habitat for juvenile fish and shellfish, as well as being the basis of an important estuarine food web. Healthy SAV both depends on and contributes to good water quality. Therefore, PREP tracks the presence of SAV in the Great Bay Estuary as an indicator of estuarine health. Note that seaweeds also provide some of these functions, but they are not considered SAVs as they are not vascular, rooted plants. The objective of this project is to map SAV habitat in the Great Bay Estuary during the summer growing period. The Great Bay Estuary is 21 square miles of tidal waters located in southeastern New Hampshire. The area for SAV mapping encompasses downstream portions of all tidal rivers and to the mouth of Portsmouth Harbor. The mouth of Portsmouth Harbor is defined by lines extending from Odiorne Point in Rye, NH to White Island to Horn Island to Sewards Point on Gerrish Island in Kittery, ME. The total area to be mapped is approximately 21 square miles. The study area in which SAV will be mapped for this project is shown in Figure 2. This is the same as the 2013 project area.

Figure 2: Study Area for 2013 Eelgrass Mapping



ATTACHMENT 6

Smast
University of Massachusetts Dartmouth
The School for Marine Science and Technology

Coastal Systems Program
SMAST

January 27, 2020

Dear Mr. Peschel:

As requested, we have reviewed the document[1] detailing the hydrodynamic and nitrogen model of the Great Bay Estuary relative to its appropriateness to support management decisions related to nitrogen concentrations as affected by nitrogen loading and system hydrodynamics. Our review also considered whether the model was appropriate for predicting nitrogen in-situ concentrations under different nitrogen loading scenarios. As part of this review, it was also necessary to examine if there is sufficient evidence to claim that nitrogen is a primary cause of water quality impairment and eelgrass loss in the Great Bay Estuary. It appears that the hydrodynamic/nitrogen model is sufficiently robust, calibrated and verified to make useful predictions of nitrogen concentrations and gradients in the Great Bay Estuary under different loading scenarios. However, it does not appear that the cause of ecological impairments in this estuary resulted from or recovery is being prevented by nitrogen enrichment, which is fundamental to conducting effective management. We recommend that the primary cause(s) of eelgrass loss be clearly determined prior to implementing any management actions.

Great Bay Estuary System Total Nitrogen Model[1] Review: HDR has been contracted to develop a hydrodynamic/nitrogen model of the Great Bay Estuary. They are using HDR's ECOMSED hydrodynamic model, which uses a three-dimensional time-dependent estuarine circulation model[2]. The model domain includes, Great Bay, Little Bay, upper and lower Piscataqua River and Cocheco River and includes an appropriate offshore boundary area. The model uses weather conditions (wind and incident solar radiation), river inflows, tide, temperature and salinity (at open boundaries) to predict water surface elevation, water velocity (3-D), temperature, salinity and turbulence throughout the estuary. The model has been used for similar studies around the world (see HDR report). As per good practice, the model output is compared to field observations to assess performance.

Hydrodynamic Model calibration used stage data from 2010, 2011 and 2017, of which the 2017 parameterization of the boundary salinity and temperature was the "best" of the 3 years, as new data was available. Temperature calibration (comparing predicted and observed) was very good for each of the 7 monitoring stations sampled throughout each of the 3 years. Salinity is highly variable due to the inter-annual differences and seasonal differences in freshwater input. Nonetheless, the model was well calibrated for salinity at most stations (less so for Squamscott River in 2010 and Lamprey River in 2011). The model is very well calibrated for temperature and salinity at both Great Bay stations in each year. These results are due to the stronger horizontal gradients in salinity in Squamscott and

[1] HDR Memorandum to Dean Peschel by C. Mancilla, T.W. Gallagher and N. Joshua. *Development of Great Bay Estuary System Total Nitrogen Model*, December 2, 2019.

[2] Blumberg and Mellor (1987) with Mellor and Yamada (1982) level 2 ½ turbulent closure scheme. Wetting/drying (flooding/draining tidal flats) was simulated (Flather and Heaps 1975) and incorporated into ECOMSED.

Lamprey basins compared to the more stable salinities in Great Bay. The salinity calibration is sufficient to give confidence in the model and it appears adequate to examine the effect of nitrogen loading on concentrations throughout the estuary.

Once the hydrodynamic model was calibrated and verified, a nitrogen model was added. This follows standard practice and was the approach used by the Massachusetts Estuaries Project (MEP) to allow for prediction of nitrogen concentrations and distribution (spatial gradient) in tidal estuaries under different nitrogen loading conditions. However, unlike the MEP where the sources and sinks of nitrogen were available and sediment recycling directly measured, the GBES nitrogen model was not as well supported by site-specific data. Therefore, the decision by HDR to build the nitrogen model based upon conservative transport was appropriate. It is important to note that the Great Bay Estuary does not appear to have the same level of water column-sediment exchange as the smaller MEP estuaries where the sediments are highly organic, resuspension is very low and nitrogen regeneration and denitrification play a significant role in nitrogen cycling. Great Bay has larger sandy and intertidal areas and sediment resuspension that supports the HDR approach. On a practical note, estuarine modeling is a sequential process where nitrogen models are developed and tested and may be refined as new datasets become available. Developing the conservative nitrogen model will allow testing of nitrogen loading and water column response in the Great Bay System. Moreover, since the model actually calibrated, the approach reduced the need for a model which includes all sources and sinks at this time.

It appears that the non-point and point sources loads are relatively well constrained. Point source loads are directly measured and account for about one third of the total loading, which makes up for some of the uncertainty in the non-point source data. The comparisons of the nitrogen loading from the 7 rivers measured versus computed show good agreement, although a root mean square (rms) error or other estimate of the fit to the 1:1 line would be helpful. Based upon visual inspection, the fit is sufficient to support the nitrogen model.

Comparisons of the predicted nitrogen concentrations and observed nitrogen concentrations on a daily basis shows good agreement at each of the 5 monitoring stations. While there are some periods of disagreement (Great Bay 2017), the main Great Bay station and other 3 stations generally agreed well with the observations in the time-varying conservative transport nitrogen model. Overall, this analysis lends confidence that the model is adequately calibrated and validated for predicting water column nitrogen concentrations under different nitrogen loading scenarios.

If nitrogen is a primary factor controlling eelgrass coverage/recovery (see next section), then nitrogen loading ~200 kg/ha/yr results in a growing season TN concentration of 0.36 mg/L. This is a relatively low TN concentration and was found by the MEP to generally support high quality eelgrass habitat in shallow basins. Under this loading condition one would not reasonably expect that resulting TN concentrations would be significantly impacting eelgrass resources. In Great Bay eelgrass has had high coverages at historically higher TN concentrations (>0.4 or even 0.5 mg/L). This represents evidence that a 200 kg N/ha/yr loading or even greater loadings should be protective of eelgrass in this system (if nitrogen is even the principle factor causing or contributing to eelgrass impairment). It is important to note that our previous analysis indicated that the Eelgrass Coverage-NLM relationship (Latimer and Rego 2010) should not be used to define an acceptable nitrogen loading threshold for a TMDL. However, if that approximate approach to threshold analysis were to be used, a value of 200 kg N/ha/yr is accommodated as there is no justification for selecting a lower value, e.g. 100 kg N/ha/yr. The eelgrass coverage and nitrogen concentration data from Great Bay are consistent with the higher estimate, as protective of eelgrass resources, although it is likely, based upon historical data, that even a higher loading rate may still be protective of eelgrass habitat.

Linkage Between Nitrogen and Eelgrass Decline is Not Supported by Observations: Although the hydrodynamic/nitrogen model has value for predicting changes in nitrogen concentrations and resolving gradients throughout the Great Bay Estuary, the role of nitrogen in resource impairments within this system has not been sufficiently documented by available data. Therefore, it is likely that managing the water and habitat quality within this estuary based upon nitrogen probably won't have the positive ecological effects that are sought. Reviewing the variety of documents indicates the following:

(a) N concentrations are relatively low within this estuary compared to other New England estuaries and chlorophyll-a concentrations are also low (typically <5 ug/L) compared to basins impaired by nitrogen enrichment. This does not indicate a nitrogen impaired system.

(b) Dissolved inorganic nitrogen concentrations have historically been on the level of 0.1 mg N/L or ~7 uM, above the level that is generally thought to create non-limiting nitrogen availability for phytoplankton (e.g. phytoplankton production has sufficient N so N is not the limiting factor). This availability of N suggests that other factors are controlling phytoplankton biomass in this system. The issue of nitrogen controlling phytoplankton biomass and therefore water column transparency is not supported by the system response to nitrogen reductions in wastewater discharges from Dover and Rochester WWTFs. Even with the large decrease in nitrogen loading, there was little observed change in phytoplankton biomass, again calling into question if nitrogen is an important factor in water quality and eelgrass decline in this system.

(c) Eelgrass has historically been prevalent at higher nitrogen concentrations than in the present period of decline. Valiela and Cole (2002) noted that TN loadings were calculated to be about 250 kg/ha-yr in the mid-1990s when there were extensive eelgrass beds within the Great Bay system.

(d) Eelgrass in this system has been lost from wasting disease and other factors have been indicated as to controlling coverages (light attenuation from non-phytoplankton, e.g. CDOM, turbidity from resuspension, unstable or unsuitable sediments, etc). As noted in the 2014 Peer Review[3], "*Eelgrass growth, abundance and distribution are also controlled by temperature, nutrient availability (primarily nitrogen and phosphorus), tidal range, water motion, wave action, water residence time, bathymetry, substrate type, substrate quality, severe storms, disease, plant reproduction and anthropogenic disturbances* [...] (Kenworthy, 13). As of this writing it does not appear that alternative causes of the recent eelgrass decline have been examined except for documented losses due to wasting disease in the previous decade. Furthermore, eelgrass has historically declined and rapidly recolonized over short time scales (1-3 years). At present, the question is why has there not been the same full recolonization as previously observed, even though there is large coverage of eelgrass in Great Bay.

(e) Two other pathways for nitrogen to effect eelgrass coverages is through large accumulations of drift macroalgae and stimulation of epiphytic growth on eelgrass leaves. Macroalgae has been examined relative to eelgrass coverage/decline but does not appear to explain the decline and cannot explain the decline/recolonization cycles in previous years. As stated in the Peer Review, "*The data and arguments provided in the DES 2009 Report to support the weight of evidence for a relationship between nitrogen concentration, macroalgal abundance and eelgrass loss are neither compelling nor scientifically defensible.* [Subsequent data from 2008, 2009, and 2010 indicate]

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[3] Bierman, V., Diaz, R., Kenworthy, W. and Reckhow, K. February 13, 2014. Joint Report of Peer Review Panel for Numeric Nutrient Criteria for the Great Bay Estuary. New Hampshire Department of Environmental Services. June, 2009.

macroalgae were not limiting eelgrass growth" (Kenworthy, 27). Similarly, although epiphytes have been observed on Great Bay eelgrass the levels have not been sufficient to explain the declines in eelgrass coverages. This was also pointed out in the Peer Review[2], "If *epiphytes are not contributing significantly to light attenuation, and chlorophyll-a is only a minor contribution to light attenuation, nitrogen cannot be directly implicated as the major cause of light attenuation and eelgrass declines in the Great Bay estuary*" (Kenworthy, 12).

Based upon: 1) the lack of clear linkages between nitrogen concentrations and phytoplankton biomass, 2) the fact that phytoplankton appear to play a minor role in light attenuation and 3) the lack of observed effects on eelgrass of epiphytes and macroalgae, it is not proper to implement nitrogen management actions to restore eelgrass in Great Bay at this time. Restoration of eelgrass coverages demands a clear understanding of the cause of the decline so that the costs of actions can be justified and the desired response can be predicted with a reasonable degree of certainty. Determining the cause(s) of the eelgrass decline is fundamental to design of any actions for promoting eelgrass coverage. This is standard practice in estuarine restoration. The lack of a clear linkage was also stated by the Peer Reviewers, "*There is no basis for a scientifically defensible linkage between nitrogen impairment and eelgrass impairment presented in the report*" (Kenworthy, 19).

We appreciate the opportunity to review and comment on the modeling and approaches for nitrogen threshold development for eelgrass restoration/protection in Great Bay. However, at this time we strongly recommend that the cause(s) of the recent decline in eelgrass coverage be quantitatively determined and that further nitrogen reductions not be implemented until a reasonable understanding of the factors controlling eelgrass dynamics in this system is developed. Fortunately, if nitrogen was involved in the eelgrass loss in Great Bay, it appears that the current nitrogen loading level (post reductions in Dover and Rochester WWTF) should be adequately protective.

Sincerely,

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ATTACHMENT 7

TUFTS UNIVERSITY
School of Engineering

Professor and Louis Berger Chair in Computing and Engineering

Mr. Dean Peschel
Great Bay Municipal Coalition
c/o City of Portsmouth
680 Peverly Hill Road
Portsmouth, NH 03801

March 22, 2019

Re: Analysis of Technical Justification for Proposed Watershed TN Load Limitations for Great Bay Estuary

Dear Mr. Peschel:

In March 2019, I was contacted by the Great Bay Municipal Coalition (GBMC) to provide technical input on a "new scientific approach" being proposed by USEPA and NHDES to prescribe nitrogen load reductions for the Great Bay Estuary and its watershed. Based on the information provided, I understand that the state and federal agencies are proposing to utilize a 100 kg/ha-yr TN loading cap as necessary for the entire Great Bay watershed to protect eelgrass growth in the system. This nitrogen target was developed primarily from an eelgrass loss-TN loading nomograph created by Latimer and Rego in 2010.[1] This "load cap" is being proposed to form the basis of new nitrogen reduction requirements for wastewater facilities, stormwater contributions, and other non-point sources (such as septic systems). Because I had previously provided analyses of the prior state and federal regulatory efforts (see Chapra 2013[2]) and contributed to the 2014 Great Bay independent peer review, you have requested my opinion on the validity of the new approach being suggested by the regulatory authorities.

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[1] Latimer, J.S. and Rego, S.A. 2010. Empirical relationship between eelgrass extent and predicted watershed-derived nitrogen loading for shallow New England estuaries. *Estuarine, Coastal and Shelf Science*, 90:231-240.

[2] Chapra, S.C. 2013. Assessment of whether the department of environmental service's approach to nutrient criteria derivation for the great bay estuary used reliable, scientifically defensible methods to derive numeric nutrient criteria. Declaration before the Environmental Appeals Board of the United States Environmental Protection Agency.

Materials Reviewed and Questions Presented

In addition to Latimer and Rego, 2010, I was provided the following documents:

- March 8, 2019 DES PPT Slides – "Adaptive Management Permitting for Great Bay" (see slides 4-10)
  - Valiela and Cole (2002)[3] – source for % Seagrass cover lost vs. nitrogen loading figure (slide 6)
- 2007 Technical Advisory Committee (including Dr. Latimer as a participant) meeting notes which considered this simplified TN-loading eelgrass loss approach
- A list of technical questions submitted to Dr. Latimer by the Coalition regarding application of Latimer and Rego (2010) nitrogen targets to the Great Bay system
- Dr. Latimer's responses to technical questions and a Word document organizing Dr. Latimer's responses with the corresponding inquiries
- A Great Bay Municipal Coalition letter to EPA/DES dated November 19, 2018 Re: Inapplicability of Latimer and Rego, 2010 to Great Bay
- 2014 Great Bay Peer Review report

You have suggested that I prepare my analysis of Latimer and Rego's approach (as well as the related technical studies) considering the following questions:

1. Is the Latimer and Rego, 2010 approach consistent with accepted scientific methods for assessing TN impacts on estuarine systems?
2. Is the Latimer and Rego, 2010 approach applicable to Great Bay Estuary and does the approach provide reasonable confirmation that TN has impaired eelgrass growth in Great Bay or is preventing its recovery?
3. Is the Latimer and Rego, 2010 method contrary to the 2014 Peer Review and EPA's 2010 Stressor Response peer review?

Analysis of the Latimer and Rego, 2010 Approach

The approach employed by Latimer and Rego (2010) is a generalized and greatly simplified approach (e.g., a screening tool) based upon limited data, hypothetical eelgrass loss/coverage assumptions, and a limited set of ecological/estuarine conditions (primarily small embayments, subject to significant groundwater loading influences and minimal riverine inputs). The results of the nomograph, on its face, suggest an extreme variation of eelgrass "responses" for similar TN system loadings. If this paper was based on "real," not assumed, eelgrass losses and TN loading was the true cause of reported eelgrass "losses" (due to excessive plant growth precluding eelgrass growth as assumed in the paper) this extreme variation in results would not be expected.

As noted in Dr. Latimer's responses to the questions posed, this was a theoretical analysis with no apparent applicability to managing the Great Bay system. The analysis, being generalized and assumption-based, made no effort to scientifically confirm the report conclusions or to claim that it should be universally applied to other systems with significantly different physical, hydrodynamic and/or biochemical conditions governing the occurrence or loss of eelgrass

[3] Valiela, I. and Cole, M.L. 2002. Comparative Evidence that Salt Marshes and Mangroves May Protect Seagrass Meadows from Land-derived Nitrogen Loads. Ecosystems (2002) 5:92-102.

populations in complex ecosystems such as the Great Bay Estuary. Thus, this paper cannot be used to reasonably or reliably forecast eelgrass responses to TN loading for the Great Bay system without explicit confirmation that (1) the predicted eelgrass losses exist and (2) the excessive phytoplankton or macrophyte growth is, in fact, preventing eelgrass recovery in this system.

With respect to other analyses presented such as Valiela and Cole, 2002, those authors also focused on small, protected embayments that had confirmed, extreme macroalgae growth, due to nutrient enrichment. The extreme macroalgae growth prevented eelgrass recovery due to smothering of the eelgrass shoots. These conditions have no apparent relevance to the Great Bay system where such smothering has not been documented as the cause of the existing eelgrass condition.

Responses to Specific Questions Posed

1. Is the Latimer and Rego, 2010 approach consistent with accepted scientific methods for assessing TN impacts on estuarine systems?

   No. This simplified analysis does not address the numerous physical, chemical, or biological factors that need to be considered to produce a scientifically defensible conclusion that nitrogen is impairing a specific estuarine system. There is no EPA-approved or "generally accepted by the scientific community" method for TN loading/eelgrass response that is applicable to estuarine systems, as there can be for lakes assuming sufficient observed response data (not unverified data points) are available to relate nutrient loading to a form of excessive plant growth that may be detrimental to the system.

2. Is the Latimer and Rego, 2010 approach applicable to Great Bay Estuary and does the approach provide reasonable confirmation that TN has impaired eelgrass growth in Great Bay or is preventing its recovery?

   No. For the reasons expressed by Dr. Latimer himself, this approach has no apparent applicability to the Great Bay system. In fact, the data for the Great Bay system confirm it is inapplicable as TN loadings have greatly exceeded the upper TN loading Latimer and Rego indicate will eradicate all eelgrass growth (100 kg/ha-yr) while robust eelgrass growth was maintained in the 1990s through 2005. These data for the Great Bay system are a direct, unambiguous empirical indicator of the "safe" systemwide TN loading at this time, particularly as excessive macrophyte or phytoplankton growth did not occur with those loadings. The more recent data for Great Bay suggest an eelgrass loss of about 30% from historical levels, not the 100% loss expected if the Latimer model was applicable. That would place Great Bay among the least impacted systems assessed by Latimer. Moreover, the factors that would suggest a linkage to TN are not reflected in present measurements. In comparison with the earlier period, phytoplankton levels are essentially unchanged, and epiphytes are not reported to be excessive. Macrophytes are present, but apparently are not preventing eelgrass regrowth each year.

3. Is the Latimer and Rego, 2010 method contrary to the 2014 Peer Review and EPA's 2010 Stressor Response peer review?

   Yes to both aspects of this question. The 2014 Peer Review determined that the available system data did not confirm that TN was the cause of eelgrass decline or periodic low dissolved oxygen readings. The Latimer and Rego, 2010 analysis is not "new" nor is it "data" for this system nor is it reflective of the conditions controlling nutrient dynamics in the Great Bay Estuary. Thus, it cannot be used to demonstrate that the prior peer review conclusions are, in any way, in error.

   EPA's 2010 Stressor-Response methodology specifically requires consideration of the relevant factors (sometimes called "confounding factors") affecting an ecological response of concern when developing system wide nutrient criteria. This analysis fails to consider any of those relevant physical, chemical, or biological factors.

I hope that you find my observations helpful in determining the best path forward for protecting eelgrass resources in the Great Bay system. At this point, I do not see any scientifically defensible basis presented for asserting that additional TN reductions are currently required to protect or restore eelgrass resources. As noted by the 2014 Peer Review, it would be best to focus on the other factors known to affect that form of plant growth to better understand eelgrass dynamics for this system.

Sincerely,

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ATTACHMENT 8

SMAST
University of Massachusetts Dartmouth
The School for Marine Science and Technology

Coastal Systems Program
SMAST

January 20, 2020

Dear Mr. Peschel:

As requested, we have reviewed the publication, *Empirical relationship between eelgrass extent and predicted watershed-derived nitrogen loading for shallow New England estuaries* (Latimer and Rego, 2010), and several earlier related publications (e.g. Valiela et al. 1997, Valiela and Cole 2002) relative to its applicability to setting nitrogen management thresholds for the Great Bay Estuary, NH. The study, which will be referred to as the Eelgrass-NLM approach, has merits by bringing forward the cautionary note that external N loading to estuaries can result in eelgrass loss and therefore source reductions are needed in some areas for eelgrass protection. However, the Eelgrass-NLM study is more of a "quick look" survey across estuaries to see what relationships might exist between N-loading and eelgrass loss, rather than a quantitative estuary specific analysis to support watershed management actions, a conclusion that appears to be supported by the lead author as well.

The following key points should not be taken as criticisms of the scientists, given the state of the science in while they were conducting their work a decade or more ago. Rather, this summary addresses issues related to the use of this approach at an estuarine specific level as a scientifically defensible method for implementing watershed N-management actions, (detail in sections below):

1) The land-use loading model (NLM) has problems with nitrogen attenuation in groundwater, and has not been sufficiently calibrated (eg. sometimes it calibrates, sometimes it does not).

2) The Eelgrass-NLM approach does not account for tidal flushing/circulation which significantly modify effects of a N load relative to eelgrass habitat quality and the level of estuarine response (eg. eutrophication).

3) The Eelgrass-NLM approach does not account for the positive/negative effects on nitrogen levels from sediment processes (denitrification) and recycling nor does it address the varied forms of nitrogen (from groundwater, river inflows, sediment releases), which differ in ability to cause adverse ecological impacts.

4) The Eelgrass-NLM approach does not account for other factors (CDOM, turbidity) that are not directly related to nitrogen loading, but effect eelgrass habitat quality.

5) The Eelgrass-NLM approach shouldn't be presumed to be generally applicable. There is evidence, even in Latimer and Rego (2010), indicating that eelgrass coverage is not always lost at high nitrogen levels or is robust at low nitrogen levels.

6) Other watershed-estuarine approaches are available that produce quantitative and site specific management targets that are also more scientifically defensible. Such methods are based upon site-specific data and system parameters, that can be calibrated and verified for the estuary being managed.

While there are multiple issues of concern if one is considering using the Eelgrass-NLM approach as a management tool, points 1-4 listed above are of the most concern and need to be addressed for the Great Bay Estuary threshold analysis. Therefore, for clarification, an expansion on each point is provided as follows:

1) A critical element of the Eelgrass-NLM approach is the estimation of nitrogen loading to a given estuary and how that watershed load relates to the presence or absence of eelgrass, however, the land-use loading values are based upon the NLM model (Valiela et al 1997), which has numerous problems with its attenuation terms (particularly during aquifer transport) and lack of true calibration and states, "*First, loading rates*[1] *calculated using the model should not be interpreted and used as hard, well-defined values of thresholds, but rather as fuzzy guidelines derived from much data and many best guesses as to the effects of the various factors*." We agree, and in the intervening years research has shown problems with key parts of the NLM. As such, the groundwater driven N-loads utilized in the Eelgrass-NLM approach may not be representative of the load actually reaching the estuary and therefore weakens the relationship being drawn between N-load and eelgrass presence/absence (the critical concern for estuarine habitat management/restoration.

The NLM approach is aimed at producing a research model which tracks nitrogen from all sources and uptake within the watershed, and attempts to predict the nitrogen discharges to the estuary. The approach is similar in construct to other land-use loading models including the Massachusetts Estuaries Project (MEP) watershed module. The major difference between the MEP land-use model and most others used in watersheds with sandy outwash aquifers is in regards to the attenuation of nitrogen during transport through the aerobic aquifer soils (in Waquoit Bay Watershed, 35% removal). Uptake of nitrogen is commonly observed in surface water systems where biological cycling of nitrogen results in a portion of the inorganic nitrogen being lost due to either direct or coupled denitrification. However, a multitude of researchers studying nitrogen transformations in aerobic sandy outwash aquifers have concluded that nitrogen attenuation is generally negligible in these situations. Watershed nitrogen loading models developed by the USGS, CCC, Buzzards Bay Project and the MEP are based upon these results. Other studies have found validation of the various factors employed in the Nitrogen Loading Model (NLM) is not always clear from available information, although some factors are well developed and nearly identical to other watershed models in general use. However, it has not always been possible to rectify differences in watershed areas, nitrogen loads, and freshwater discharge volumes from the various reports and papers. More importantly, validation of the NLM model by its developers was based upon groundwater well point measurements, which sampled only a small portion of the full cross-section of the groundwater discharge boundary and only inorganic nitrogen forms. As the NLM is based upon inorganic nitrogen forms, it important to note that this contrasts with larger estuarine systems like Great Bay, which receive treated effluent discharges from WWTFs and large surface freshwater inflows which include organic forms as well as inorganic forms in the TN pool. In the NLM groundwater sampling there was no fractionation of the groundwater nitrogen pool or any salinity data presented in historic work describing the development of the NLM approach, it is not possible to evaluate whether the groundwater sampling for calibration taken at the "high tide mark at the seepage face" is representative of the groundwater flow. Limitations in this approach to

[1] Valiela et al. 1997, p. 374 referring to nitrogen loading rates derived by the land use model that enter the estuary.

measurement of groundwater nitrogen discharges were found in the MEP assessment of the Waquoit Bay Estuary where the NLM was developed (Valiela et al 1997).[2]

The MEP Nitrogen Loading Assessment found a " very large discrepancy in the Sage Lot Pond sub-system which receives little anthropogenic loading (modeled versus measured from Valiela et al., 2000, Table 2, 147 versus 846 kg N yr^{-1}, respectively). In addition, the "measured" loads to Hamblin Pond, Jehu Pond, and Quashnet River using the watershed areas presented in Valiela et al., 2000 yield agreements to modeled loading of 54%, 73% and 118% respectively (see Table 2 in Valiela et al., 2000)." Further, "based on a general review of the Waquoit Bay Nitrogen Loading Model (NLM) results published to date, there appeared to be *significant bias* in the model at higher nitrogen mass loadings. However, this research model was a unique attempt to capture all of the sources of transformations of nitrogen during passage through each major element of the soil system (biotic surface layer, vadose zone and aquifer) for each of the land-use types. It clearly fulfilled a critical role as a *research model* in indicating areas to direct additional future studies (e.g. aquifer attenuation, validation approaches). It should be noted that the model stops at the freshwater/salt water interface, and does not include the estuary itself (just the watershed)." Subsequent to the NLM, the MEP Linked Watershed-Embayment Modeling approach addressed attenuation in its N-loading module by empirical measure in a more integrated manner.

A key problem with the NLM for watershed loading determinations is that it is not robust, is only sometimes calibrated and then to inorganic nitrogen concentrations (which generally represent a small fraction of the total nitrogen pool), and does not account for circulation or dispersion of nitrogen within the receiving waters. Since the NLM was developed, new information on the lack of nitrogen attenuation in sandy outwash aquifers, simple tools for determining attenuation during passage through ponds and streams, have been incorporated into management assessments and threshold development. The lack of specificity, problems with attenuation and other loading issues likely explains the wide range of eelgrass coverage per watershed nitrogen load (Latimer and Rego 2010, graph 2). The uncertainty in the actual loading, lack of verification of the NLM further reduces the utility of the Eelgrass-NLM approach, and reduces its validity for setting defensible N thresholds for restoration of eelgrass coverage.

2) The Eelgrass-NLM approach uses static watershed N load for comparison to eelgrass habitat quality (declining, improving, stable). This has a major conceptual flaw, it is not the nitrogen loading rate from the watershed but the concentration of nitrogen in estuarine waters that controls eelgrass habitat quality[3]. Nitrogen loading effects are moderated by tidal flushing (exchange with low nitrogen boundary waters), which is further complicated by the location which nitrogen enters the estuary (headwaters, mid, near tidal inlet). The same mass of nitrogen entering at the headwaters has a much greater impact per kg, than if it entered nearer the tidal inlet due to the amount of time (residence time of water) a mass of N has to be influenced by internal biological processes as well as physical processes. From a management point of view, these factors are only

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[2] Howes B.L., S. Kelley, E. Eichner, R. Samimy, J. S. Ramsey, D. Schlezinger, P. Detjens (2011). Massachusetts Estuaries Project Linked Watershed-Embayment Approach to Determine Critical Nitrogen Loading Thresholds for the Waquoit Bay and Eel Pond Embayment System, Towns of Falmouth and Mashpee, MA, Massachusetts Department of Environmental Protection. Boston, MA.

[3] This is true of estuaries that (1) have production controlled by N (eg. N is the nutrient causing eutrophication) and (2) has watershed load dominated by inputs of inorganic nitrogen (eg. not refractory N compounds).

lightly addressed by the Eelgrass-NLM approach and without any specificity relative to the Great Bay Estuary. Hydrodynamics was not in the approach itself, but used some generic factors during data interpretation. Said approach attempts to addresses complex estuarine hydrodynamics using both a generic flushing factor inclusive for the whole system and a dilution factor (i.e. potential for flushing and dilution to affect N-load once it enters the estuarine system), however, this is inadequate for resolving N-concentration spatial gradients needed for clarifying how N-loading effects eelgrass distribution and density.

In general, to render scientifically defensible nutrient management decisions, water quality (nutrients) studies of tidally influenced estuaries must include a thorough evaluation of the hydrodynamics of the estuarine system. Estuarine hydrodynamics control a variety of processes including tidal flushing, pollutant dispersion, tidal currents, sedimentation, erosion, and water levels. Numerical models provide a quantitative and cost-effective method for evaluating tidal hydrodynamics since they require limited data collection and may be utilized to numerically assess a range of management alternatives to improve water quality and necessarily habitat quality (seagrasses, benthic micro/macro fauna). Once the hydrodynamics of an estuary system are understood, computations regarding the related coastal processes become relatively straightforward extensions to the hydrodynamic modeling. For example, the spread of pollutants (e.g. nutrient concentration gradients) can be analyzed from tidal current information developed by the numerical models and related to seagrass/benthic infauna distribution and density.

Regarding the determination of in situ N-concentrations given both external and internal N-loading to an estuary, several key points must be given consideration as follows. Since the magnitude of freshwater inflow into a given estuary can be smaller or larger in comparison to the tidal exchange through each inlet, the primary mechanism controlling estuarine water quality within a given estuarine system is tidal exchange. A rising tide offshore creates a slope in water surface from the ocean into the system of concern. Consequently, water flows into (floods) the system. Similarly, each estuary drains into offshore waters on an ebbing tide. This exchange of water between the estuarine system and the ocean is defined as tidal flushing. Numerical modeling tools must be invoked to evaluate the complexities of estuarine circulation/exchange and the effects on N-concentrations to then quantitatively assess tidal flushing in a system and how that relates to water residence times and changing N-concentration.

Flushing rate, or residence time (system vs. local), is defined as the average time required for a parcel of water to migrate out of an estuary from points within the system and has a critical effect on how N-loads translate to concentration gradients along an estuary and changes over time. **System residence times** are considered as the average time required for a water parcel to migrate from a point within the each embayment to the entrance of the system. In addition to system residence times, a second residence, the **local residence time**, is defined as the average time required for a water parcel to migrate from a location within a sub-embayment to a point outside the sub-embayment. Using Great Bay as an example, the **system residence time** is the average time required for water to migrate from Great Bay through Little Bay, into the Piscataqua River and into the Gulf of Maine, where the **local residence time** is the average time required for water to migrate from Great Bay to just Little Bay (not all the way to the Gulf of Maine).

Residence times are provided as a first order evaluation of how loading translates to estuarine water quality (N-concentrations). Lower residence times generally correspond to higher water quality (lower N-concentrations); however, residence times may be misleading depending upon pollutant/nutrient loading rates and the overall quality of the receiving waters. As a qualitative guide, **system residence times** are applicable for systems where the water quality within the entire estuary is degraded (high N-concentrations) and higher quality waters (e.g. low N-concentration water from the Gulf of Maine) provide the only means of reducing the high nutrient concentrations within the estuary. The rate of pollutant/nutrient loading and the quality of water outside the estuary both must be evaluated in conjunction with residence times to obtain a clear picture of water quality. Efficient tidal flushing (low residence time) is not an indication of high water quality if pollutants and nutrients are loaded into the estuary faster than the tidal circulation can flush the system. Neither are low residence times an indicator of high water quality if the water flushed into the estuary is of poor quality. Advanced understanding of water quality (N-concentrations and the forms of N present) must be obtained using numerical models to reconcile external and internal N-loads, pollutant/nutrient dispersion (circulation) and residence times to ultimately clarify N-concentration gradients and how those gradients contribute to eelgrass presence or absence. A key problem related to determining nitrogen concentrations in the Great Bay Estuary from watershed and other nitrogen inputs is accounting for the boundary condition nitrogen. Higher boundary conditions result in higher nitrogen levels in an estuary than if the boundary condition was lower. In the Eelgrass-NLM approach the boundary condition is insufficiently accounted for or not at all. Setting a nitrogen threshold concentration (from which the load is derived) to restore eelgrass in an Estuary like Great Bay requires inclusion of boundary conditions. Numerical hydrodynamic and water quality modeling provides a quantitative tool to include boundary conditions and to evaluate the complex mechanisms governing estuarine nutrient concentrations and how nitrogen load reducing actions taken for estuarine management translate to improvements in water quality (e.g. lower nutrient concentrations, greater water clarity, increased dissolved oxygen concentrations, lower chlorophyll concentrations) and increased eelgrass presence and density where nitrogen is the key determinant. Information available for the Great Bay system, indicates that system residence time is low, in comparison to the small embayments cited by Latimer and Valiela. This key factor confirms that application of the simplified assessment methods are not relevant to the Great Bay system.

One additional factor effecting the nitrogen threshold for eelgrass in an estuary is the tide range, the height of water over the sediment at high vs low tide. Empirical studies have found that systems with larger tide ranges are able to sustain eelgrass coverage compared to basins with a smaller tide range at the same nitrogen level. The underlying reason relates to light penentration, which is enhanced at low tide in a large tidal range system. Basically the eelgrass can withstand higher turbidity if a portion of the tide range (low tide) allows sufficient light for growth even if at high tide the light is much lower. This helps to explain some of the wide variation in eelgrass response to watercolumn nitrogen levels and loading rates, as this is not accounted for in survey studies.

3) Recycling of nitrogen within the water column and estuarine sediments generally contributes (positively, e.g. release; or negatively, net uptake, e.g. denitrification) significantly (internal nutrient loading) to water column nitrogen balance. In some estuaries, sediment release during summer accounts for up to 50% of the nitrogen that supports plant (microalgae {phytoplankton}, macroalgae) growth during summer. Recycling of N is not part of the Eelgrass-NLM approach.

Background: In addition to the nitrogen transport from land to estuarine receiving water, the amount of direct atmospheric deposition on each embayment surface must be determined as well as the amount of nitrogen recycling within the embayment, specifically net nitrogen regeneration from sediments (also considered legacy nutrients). Sediment nitrogen recycling results primarily from the settling and decay of microalgae (phytoplankton) and macroalgae (and eelgrass when present). During decay, organic nitrogen is transformed to inorganic forms, which may be released to the overlying waters or lost to denitrification within the sediments. Permanent burial of nitrogen in the sediments is generally small relative to the amount cycled. Sediment nitrogen regeneration can be a seasonally important source of nitrogen to embayment waters or in some cases a sink for nitrogen reaching the bottom of an estuary. Failure to include the nitrogen balance of estuarine sediments and the watershed attenuation generally leads to errors in predicting water quality (water column N concentrations), particularly in the determination of how summertime nitrogen load to embayment waters translate to phytoplankton production, changes in water clarity and the associated eelgrass loss documented in many estuaries.

In general the fraction of the phytoplankton population which enters the surficial sediments of a shallow embayment: (1) increases with decreased hydrodynamic flushing, (2) increases in low velocity settings, (3) increases within small enclosed basins.

Once organic particles become incorporated into surface sediments, they are decomposed by the natural animal and microbial community. This process can take place both under oxic (oxygenated) or anoxic (no oxygen present) conditions. It is through the decay of the organic matter with its nitrogen content that bioavailable nitrogen is returned to the embayment water column for another round of uptake by phytoplankton. This recycled nitrogen adds directly to the eutrophication of the estuarine waters in the same fashion as watershed inputs. In some systems that have been investigated by the MEP, recycled nitrogen can account for about one-third to one-half of the inorganic nitrogen supply to phytoplankton blooms during the warmer summer months. It is during these warmer months that estuarine waters are most sensitive to inorganic nitrogen loadings. Failure to account for this recycled nitrogen generally results in significant errors in determination of threshold nitrogen loadings for management and habitat restoration. In addition, since the sites of recycling can be different from the sites of nitrogen entry from the watershed, both recycling and watershed data are needed to determine the best approaches for nitrogen mitigation. is important to be able to account for the net nitrogen flux from the sediments within each part of each sub-system. This requires that an estimate of the particulate input and nitrate uptake be obtained for comparison to the rate of nitrogen release. Only sediments with a net release of nitrogen contribute a true additional nitrogen load to the overlying waters, while those with a net input to the sediments serve as an "in embayment" attenuation mechanism for nitrogen.

Simply put, without accounting for sediment N uptake/release, it is not possible to determine the water column N concentrations with accuracy and even when quantitative flushing and mixing is determined for and estuary. At present there is not a good determination of flushing/circulation of Great Bay relative to dilution/dispersion/flushing of nitrogen added to the water column nor is there a determination of sediment uptake/release throughout the estuary. Therefore, it is not possible to determine the N threshold load with accuracy which would be supportive of eelgrass and a healthy habitat.

Nitrogen uptake/release from the sediments of Great Bay is not taken into account if the Eelgrass-NLM is employed, greatly increasing the uncertainty in any threshold based upon knowing the nitrogen load. As mentioned above, using the extremely simplistic Eelgrass-NLM approach (while adequate as a macro-level screening tool) does not support development of a robust N Threshold needed for development of a Great Bay TMDL and cost effective N management / habitat restoration. While it may be used as a general guideline for prioritizing which estuaries are most in need of detailed assessment, there is little confidence that costly implementation measures should be based upon its output and there is a good possibility that over or under management may occur. However, as sediment processes can result in both net removal or release in summer, it is not even possible to determine with confidence that the sediment N load in Great Bay is over or underestimated at this time.

4) Other factors can cause eelgrass decline and this was not assessed in the NLM. We agree with both Valiela and Latimer that in many settings, excess nitrogen is a major cause of eelgrass decline, working the general sequence of nitrogen load/concentration increase, increased phytoplankton, decreased light penetration/increased epiphytes (lowering eelgrass growth) and loss of eelgrass health and eventually coverage. However, other factors can play important roles, such as increased CDOM (noted in Chesapeake Bay restorations) and turbidity due to re-suspension or surface water inputs of particulates. Both of these latter factors play the same role as increased phytoplankton in decreasing light penetration, lowering light for eelgrass growth. Before any restoration threshold or action plan can be developed based on nitrogen, these factors and any others, such as unstable sediments due to a change in circulation, dredging, shellfishing, and other direct disturbances need to be evaluated. Given the question of using the Eelgrass-NLM approach, it is important to note that Latimer and Rego (2010) state that 5 of the 62 basins in their study are "anomalous" in that they have low loading rates and no to small eelgrass coverages and appear to fall outside of the eelgrass – water quality paradigm. Although some speculation of why this occurs in these systems is presented, they remain anomalous and with the Great Bay data (see #5 below), raise questions about the validity of merely using a 100 kg/ha/yr loading rate, without higher level analysis.

In the Massachusetts Estuaries Project's analysis of 70 s.e. Massachusetts estuaries, eelgrass loss was deemed a key indicator of nitrogen enrichment only if: (a) there was evidence that eelgrass historically existed in the basin, (b) other factors (dredging, sediment stability, moorings/disturbance, etc) were first ruled out, and (c) the basin was nitrogen limited (phytoplankton production was stimulated by nitrogen additions). These factors were not included in the Eelgrass-NLM approach as it was a survey study to examine if there were any general relationship between eelgrass coverage (not eelgrass loss) and present nitrogen loading.

In the case of the Great Bay Estuary, only turbidity/CDOM and nitrogen appear to be possible determinants to eelgrass loss. The role of turbidity, mainly from re-suspension has been a concern. While we did not have access to a lot of turbidity data, it is interesting that the Mothers Day Storm was observed to result in significant re-suspension and high turbidity for an extended period. This was not a nitrogen-induced effect. It is not clear how often this occurs or how wide spread the occurrence, however, it does indicate that re-suspension of sediments is of concern for light penetration in this estuary and therefore it is currently not clear how much of the eelgrass loss in Great Bay is related to nitrogen loading versus turbidity from resuspension (or possibly CDOM etc). This information indicates that a higher order approach should be invoked to be certain that nitrogen is the key to eutrophication in this system and that lowering N levels will restore historic eelgrass coverage within the Great Bay Estuary.

5) The Eelgrass-NLM approach has not been verified to be generally applicable. We reviewed Valiela and Cole 2002 as Great Bay is listed within the tables of that publication, but examination of the document reveals no recommendations or information on eelgrass loss that is relevant to Great Bay. However, the TN loading to Great Bay was noted as 252 kg/ha-yr (Table 1 at 94) citing Short and Mathieson (1992), but does not contain an independent loading analysis or level of eelgrass present in the system. None-the-less, it is significant that the presented eelgrass mapping data for the system (1990-1996) confirms robust eelgrass growth throughout Great Bay but at an apparently higher TN loading rate well above the threshold of 100 kg/ha-yr suggested in Latimer and Rego (2010). Similarly, in Figure 1, there appears to be no significant difference in the % coverages in the 50-100 kg/ha-yr range than in the 100-150 kg/ha-yr range. The large amount of variation in the overall data set and the very low numbers in the 150 – 250 kg/ha-yr range greatly increase the risk of error in using a 100 kg/ha-yr threshold based upon this date.

6) Since there are major limitations to using the NLM nitrogen loads coupled to a generalized eelgrass distribution to set nitrogen limits for management, others have used more estuarine specific quantitative assessment and modeling approaches. Among many, we herein give the example of the Massachusetts Estuaries Project (MEP) approach to setting nitrogen thresholds for eelgrass restoration. The MEP Linked Watershed-Embayment Management Model Approach was established because many of the previously developed tools (like the Eelgrass-NLM Approach) for predicting loads and concentrations tend to be generic in nature, and overlook some of the specific characteristics of a given water body as well as details of estuarine dynamics that drive habitat function to varying degrees. The MEP approach focuses on linking water quality model predictions, based upon watershed nitrogen loading (inclusive of integrated measure of attenuation across the entirety of the watershed) and embayment recycling and system hydrodynamics, to actual measured values for specific nutrient species within estuarine waters. The linked watershed-embayment approach is built using embayment specific measurements, thereby enabling calibration of the prediction process for specific conditions in each of the coastal embayments of southeastern Massachusetts. To date, MassDEP and USEPA have been developing TMDLs for 70 estuaries in Massachusetts based upon the MEP assessment and modeling approach.

Conclusions: Based upon review of the Eelgrass-NLM approach (Latimer and Rego 2010, Valiela et al. 1997, Valiela and Cole 2002) it is clear that there are substantial weaknesses for its application for management of the Great Bay Estuary and we must conclude that it is not sufficiently robust for determining eelgrass restoration targets in this estuary.

- The NLM suffers from large uncertainty in several of its attenuation factors, some of which are now known to be incorrect, and the fact that it has not been sufficiently calibrated and verified for any estuary and Great Bay in particular.
- It is the concentration of nitrogen in estuarine waters that controls eelgrass habitat quality, not the loading[4]. Nitrogen loading effects are moderated by tidal flushing (exchange with low nitrogen boundary waters), which is further complicated by the location which nitrogen enters the estuary (headwaters, mid, near tidal inlet). Nitrogen loading effects are also modified by the tidal range and basin volume. In addition, the forms of nitrogen present have different impacts on eutrophication depending upon tidal flushing. From a management point of view, these factors are only incompletely and inadequately addressed by the Eelgrass-NLM approach. Said approach attempts to addresses complex estuarine hydrodynamics using both a flushing factor and a dilution factor (i.e. potential for flushing and dilution to affect N-load once it enters the estuarine system), however, this is inadequate for resolving N-concentration gradients needed for clarify how N-loading effects eelgrass distribution and density.
- Equally important, the Eelgrass-NLM approach as presently applied does not account for a major nitrogen source/sink in the sediment during the critical summer period which typically has a large impact on water column N levels and the level of nitrogen enrichment. Not accounting for this process and its variation throughout the Great Bay Estuary creates substantial uncertainty and can result in either an under or overestimate of the amount of nitrogen source reduction that may be required to restore eelgrass coverage (if N is the primary cause of decline).
- While nitrogen enrichment does cause eelgrass decline in many estuarine settings, it is not totally clear that turbidity resulting in decreased light penetration with associated eelgrass loss is not a primary or even the primary factor in the Great Bay system. This nitrogen versus resuspension driven turbidity has been a point of discussion for several years, while CDOM can also be a major factor. CDOM has been documented to play a major role in limiting light penetration for this system (Morrison, et al 2008) In our review of the data, we could not determine the magnitude of the role of sediment resuspension, but caution that if this is a major cause of eelgrass decline, nitrogen source reduction will not have the anticipated positive effect on restoration after the funds are expended. The role of other factors was discussed by Latimer and Rego (2010) where 5 of the 62 basins had low N loading and no to low eelgrass coverages and were deemed outside of the eelgrass – nitrogen loading paradigm.
- The significant variability in the overall relationship between eelgrass coverage and nitrogen loading relationship greatly increase the risk of error in using a 100 kg/ha-yr threshold based upon this data. Due to the large amount of variation, the there appears to be no significant

[4] This is true of estuaries that (1) have production controlled by N (eg. N is the nutrient causing eutrophication) and (2) has watershed load dominated by inputs of inorganic nitrogen (eg. not refractory N compounds).

difference in the % coverages in the 50-100 kg/ha-yr range than in the 100-150 kg/ha-yr range. Use of this approach is further complicated by the very low numbers in the 150 – 250 kg/ha-yr range, which would include the historical loading estimate for Great Bay (252 kg/ha/yr) when there was significant eelgrass coverage.

Taken together, it is not possible to recommend the Eelgrass-NLM approach as a scientifically defensible method for setting a nitrogen threshold or target or to use as the basis for watershed nitrogen load reductions. There are simply too many data gaps, uncertainty in the NLM loadings and a wide variation in the eelgrass coverage at similar watershed nitrogen loadings (graph 2, Latimer and Rego 2010). Further the developer of the NLM noted the issues in the 1997 paper, where he directly stated that the "loading rates calculated using the model should not be interpreted and used as hard, well-defined values of thresholds, but rather as fuzzy guidelines" [5]. Similarly, Dr. Latimer has indicated (personal communication) that his 2010 paper was intended to seek new information on the general relationship between N loading and eelgrass coverage, which has spawned new research, but is not robust enough for developing and implementing nitrogen thresholds. Moreover, he concurred that it would be inappropriate to apply this method to derive nutrient reduction requirements for the Great Bay system given its unique hydrodynamic and physical characteristics that earlier assessments did not address. Since other approaches are now available to increase the certainty of threshold analysis and which cover the data gaps mentioned above, employing some of these seems reasonable to produce a robust, quantitative, defensible nitrogen threshold concentration and load for the Great Bay Estuary.

Sincerely,

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[5] Valiela et al. 1997, p. 374 referring to nitrogen loading rates derived by the land use model that enter the estuary.

ATTACHMENT 9

SMAST
University of Massachusetts Dartmouth
The School for Marine Science and Technology

Oct 7, 2019

Dear Mr Peschel:

As requested, I have reviewed the documentation provided regarding various total nitrogen targets/thresholds for the restoration/protection of eelgrass and benthic resources in various estuaries in New England that have EPA-approved Total Maximum Daily Loads (TMDL) under the Clean Water Act. SMAST was involved in the majority of those projects and is cited heavily in their TN threshold analyses. As a result, I am quite familiar with the level of water quality modeling and water quality monitoring that were used to develop these protective nutrient targets for the eelgrass endpoint. The characteristics of those studies and data collection efforts supporting the development of the TN endpoints and TMDL reductions are stated below.

Background on SMAST Endpoint and TMDL Development

Virtually all of the studies were conducted to address the impacts on TN on saltwater "ponds" and embayments. The nitrogen sources to these systems were generally "non-point" (i.e., groundwater, septic, and surface runoff) with point sources limited to a only a few estuaries. These systems were not associated with major river systems, significant upland watersheds or substantial WWTPs. For these systems, the groundwater, groundwater fed streams and sediment components would have been primarily contributing dissolved inorganic nitrogen forms (i.e., nitrate from septic tanks or ammonium from sediment organic matter decay). TN loading to the estuaries was determined from individual parcel data coupled to nitrogen source strengths from local studies (watershed module) and directly measured stream nitrogen discharges and directly measured sediment nitrogen release.

The system data used to identify the protective TN targets and calibrate the embayment specific water quality models were collected during the growing season (typically May – September), as the critical period for management. The growing season was selected as the focus as it supports the poorest water and habitat quality of the year. Embayment specific hydrodynamic models were used to ensure that the impact of the external loading sources were properly considered, with respect to system transport, dilution (dispersion), flushing and boundary exchanges and freshwater volumetric inputs as they affect the distribution of TN concentrations throughout the system of interest. The forms of nitrogen present were typically composed of bioavailable or readily degradable plus more refractory forms (refractory is defined as not biologically available within its residence time in a basin). As the external loading sources to the watershed (e.g., septic tanks) did not vary seasonally for the most part, the TMDLs have only recommended annual average load reductions. If the sources had a major seasonal component (like riverine or WWTP inputs), this temporal variation was integrated into the water quality models to assess growing season impacts. This was relatively straightforward as most embayments had water exchanges on the order of weeks to a month.

The eelgrass TN thresholds developed by SMAST were fundamentally based on intra and intersystem comparisons of eelgrass and measured water quality (including TN, water clarity, salinity, depth) with tidally averaged TN from the validated numerical water quality modeling. However, the modeling of TN was a refinement and is not always critical to developing a threshold.

It is the comparison of TN levels across a variety of eelgrass sites (areas with healthy eelgrass and stable beds, areas with thinning beds, areas where eelgrass beds have been declining or have recently disappeared) that underpins threshold analysis for the eelgrass endpoint. This comparative approach is used for to develop a variety of threshold in aquatic systems and is generally accepted as the best available approach because it is based upon actual measurements of the constituent of interest (nitrogen) and the"health" of the selected endpoint (in this case eelgrass, but also benthic animals). The approach is both robust and verifiable and can be augmented by the use of indexes or models,

Regarding to the questions that you posed, please see my answers below:

Appropriate nitrogen concentrations for the protection of eelgrass resources;

For eelgrass, the protective growing season TN concentration identified by SMAST typically ranged 0.32-0.45 mg/l, as you have properly identified in the summary attachment. (Enclosure). For the Great Bay system, selecting a growing season average in the range of 0.32-0.35 mg/l should be protective of that resource based on our experiences with the nearby Massachusetts estuarine waters. But analysis of the Great Bay available data on water quality and eelgrass is needed to be sure. Also, the range in TN across the SMAST threshold analyses stems from differences between estuaries, particularly in terms of depth, tidal range, amount of inorganic material versus organic material in watercolumn and data on site specific temporal trends in eelgrass coverage. For example, eelgrass in shallow water can tolerate higher TN and turbidity levels than in adjacent sites in deeper water, etc.

Timing and Forms of Nitrogen to Regulate

I understand that the Great Bay system is relatively well flushed (relatively short residence time) and that the form of nitrogen likely includes components that are not biologically active during their short time in the estuary, for example bulk DOC/DON can be 100's of years old and typically makes up the majority of TN entering through the offshore boundary on incoming tides. Given that the point and non-point system loads to Great Bay Estuary can vary significantly seasonally the peak seasonal loads need to be used in the modeling and the model verified with growing season measured TN levels throughout the basins. The growing season loading needs to account for regeneration of nitrogen from the sediments, as this can be a significant input during summer. I suggest that the modeling also include bioactive nitrogen (DIN+PON) as it has been found to be more accurate in large basins, as it does not include bulk DON which is generally refractory. Having both the TN and bioactive N models should allow better targeting of N load reductions and over-management.

This system would have a higher particulate N loading than the systems evaluated by SMAST, given the large watershed that feeds into Great Bay and the Piscataqua River. Some particles would be expected to settle within the system. Therefore, it is recommended that the impact of sediment release of bioavailable forms of nitrogen be assessed. In some systems, this is significant, others less so. This would provide insight on the need to address the control of particulate forms of nitrogen from the watershed in runoff that could settle and create adverse impacts during the growing season. The model calibration for TN and bioactive N should also yield insight into the importance of summer sediment N release to the overall N load to the watercolumn.

Overall, it appears that a comparative analysis of key water quality metrics and eelgrass health/stability will support a site-specific TN threshold for Great Bay. Also, nitrogen modeling needs to provide distribution of TN and bioactive N throughout the system and allow validation using actual data to ensure proper N load reductions are developed.

Presently, I am not aware of any other papers or studies addressing the level of nitrogen that would be protective of eelgrass resources in the New England area. I hope you find this information helpful in completing your analyses of the Great Bay System.

Sincerely,

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Evaluation of TN Endpoint for the Protection of Eelgrass

Prepared by
Great Bay Municipal Coalition

In November 2018, EPA Region I identified a paper published by Dr. James Latimer (Latimer and Rego, 2010)[1] as appropriate for setting nitrogen load restriction in Great Bay estuary for the protection of eelgrass. In subsequent meetings, NHDES requested that the Great Bay Municipal Coalition identify an alternative approach, based on literature and other relevant scientific information, that could be considered protective of eelgrass resources and used to set nutrient limitations while site-specific studies are being conducted in the Estuary. This memorandum provides a Summary Table of various TN endpoints identified as being protective of eelgrass resources in nearby New England estuarine systems. The table primarily reflects a subset of TN endpoints from approved TMDLs developed to protect eelgrass habitat, prepared by MassDEP as part of the Massachusetts Estuaries Project (MEP). The MEP program relied on verified exposure data and resulting system response (i.e., the values are based on conditions documented to be protective, not theoretical model loading analyses). The subset was limited to about 20 approved endpoints (instead of pulling all of the TMDL endpoints) because the TN targets all clustered within a small range and our purpose was to select an interim value supported by a preponderance of accepted values.

Each of the MassDEP TMDL endpoints was developed for a relatively small embayment, using the "sentinel" station approach to develop the target endpoint. The target TN endpoint was selected from a station near the mouth of the embayment system with higher quality waters that supported eelgrass habitat. Each of the embayments was primarily under the influence of TN loading from groundwater sources associated with septic systems and land usage. As such, the TN load was primarily in the form of dissolved inorganic nitrogen (DIN). Each of the reported endpoints in the Summary Table is a growing season average concentration. Therefore, if an interim endpoint value is selected from the Summary Table for application to the Great Bay Estuary, it should also be applied as a growing season average. For added conservativism, the criteria would be applied as total nitrogen.

As part of this literature review the Coalition also examined the *Long Island Sound Comprehensive Conservation and Management Plan 2015* and the *Long Island Sound Nitrogen Reduction Strategy* (December 2015) which, among other things, establish goals for restoring eelgrass and limiting hypoxia in Long Island Sound. This Plan was developed and approved by multiple parties, including EPA Region 1, to protect eelgrass resources. An overview of the nitrogen reduction strategy was presented by EPA via public webinar on November 8, 2017. One conclusion of the strategy was to differentiate between coastal embayments with small watersheds influenced primarily by groundwater loadings and those which received loadings from larger riverine systems (such as that present in Great Bay). The USEPA Fact Sheet with the Nitrogen Reduction Strategy specifically noted that the empirical relationships between nitrogen

[1] Latimer, J.S., and Rego, S.A.. 2010. Empirical relationship between eelgrass extent and predicted watershed-derived nitrogen loading for shallow New England estuaries. Estuarine, Coastal and Shelf Science 90 (2010) 231 – 240.

loads and eelgrass health, such as that developed by Latimer and Rego (2010), may not be valid for larger riverine systems and, consequently, was not employed as the basis for developing nutrient loading targets. (*Nitrogen Reduction Strategy, Fact Sheet #2* at 1). As noted in the LIS documents, the direct loading approach suggested by Latimer and Rego (2010) does not addresses (1) actual site-specific system responses, (2) relevant forms of nitrogen, (3) systems where the major loading are from riverine sources or (4) the unique hydrodynamics of an estuary impacting plant growth responses to nitrogen inputs. Consequently, as with the LIS Strategy, the use of this approach is not scientifically defensible for assessing TN impacts in the Great Bay Estuary system.

EPA (through Tetra Tech) also prepared a literature review memo summarizing its technical approach for establishing nitrogen thresholds in Long Island Sound[2]. The literature review memo is organized by watershed groupings including separate evaluations for smaller embayments and those affected by large riverine systems. For each of these groupings, EPA is developing nitrogen thresholds to translate the narrative water quality standard into a numeric target concentration (as done in the MEP TMDLs summarized in the table) and identifying where nitrogen watershed loading results in exceedances of the identified threshold. Based on the literature review of median growing season TN concentration necessary to protect eelgrass, page F-3 of the Report stated the following:

> *For embayments, Tetra Tech selected a median value of 0.40 mg/L TN to protect the seagrasses in embayments. This value is the rounded value of the median TN protective of seagrasses (0.39 mg/L; range: 0.30 to 0.49 mg/L). Values above the literature review maximum TN concentration of 0.49 mg/L were not considered protective of eelgrass (see Table F-1).*

Once a TN endpoint was identified, the load necessary to meet the endpoint was calculated considering the system hydrodynamics. (See, *Establishing Nitrogen Endpoints for Three Long Island Sound Watershed Groupings. Subtasks F and G. Summary of Empirical Modeling and Nitrogen Endpoints*. April 13, 2018) From the LIS studies and peer review (discussed below), it is clear EPA Region 1 is not using the Latimer and Rego (2010) loading approach to establish reduction requirements for eelgrass protection in Long Island Sound, even in the smaller embayments. Rather, first a TN concentration necessary to protect eelgrass resources is identified. Then, the load necessary to ensure that the TN endpoint is not exceeded is determined. This is the same approach used in the MEP TMDLs that are summarized in the Endpoint Summary Table and is consistent with the approach the Coalition has undertaken here.

Finally, an independent peer review of the proposed LIS approach was completed in January 29, 2019 by EPA Region 1. The independent peer review Technical Review Team, funded by EPA, included Dr. Victor J. Bierman. Dr. Bierman was also on the peer review team that evaluated the 2009 Draft Nutrient Criteria for Great Bay. In that analysis, Dr. Bierman stated the following:

[2] Literature Review Memo. March 27, 2018. Long Island Sound (LIS): Application of Technical Approach for Establishing Nitrogen Thresholds and Allowable Loads for Three LIS Watershed Groupings: Embayments, Large Riverine Systems and Western LIS Point Source Discharges to Open Waters.

> [E]elgrass and aquatic life are the assessment endpoints. If appropriate analyses are conducted with all of the relevant site-specific data, then TN concentration targets can be developed that will protect the assessment endpoints. In turn, an appropriate site-specific, load-response model can then be used to determine TN loads from the watershed that can meet the in-water TN concentration targets. This is the approach currently being used with the linked watershed-embayment model in the 89 MEP embayments (Howes et al., 2006).

This is the approach that the Great Bay Municipal Coalition is currently pursuing. Therefore, consistent with EPA's own findings and approaches in LIS, it is appropriate to employ the literature review approach presented in this memorandum, to identify a range of growing season average TN endpoints (0.35-0.45 mg/l) for use as an interim target, pending completion of the site-specific studied for the Great Bay system. The interim TN target can be used to evaluate interim TN load limitations using the hydrodynamic model as we are currently doing.

TN Endpoint Summary Table

No.	Receiving Water/Source	Author	Protected Use	TN Source	Avg. Period	TN Endpoint	Page Citation
1	Wild Harbor Estuarine System TMDL for TN	MassDEP November 2017	Eelgrass Cover	Ground water (septic)	Summer Seasonal Avg.	0.35 mg/L	iii
2	Parkers River Embayment System TMDL for TN	MassDEP May 2017	Eelgrass Habitat	Ground water (septic)	Summer Seasonal Avg.	0.42 mg/L	iii
3	Fiddlers Cove and Rands Harbor Embayment Systems TMDL for TN	MassDEP November 2017	Benthic Community Structure	Ground water (septic)	Summer Seasonal Avg.	0.50 mg/L	iv
4	Quissett Harbor Embayment System TMDL for TN	MassDEP November 2017	Eelgrass Habitat	Ground water (septic)	Summer Seasonal Avg.	0.34 mg/L	iv
5	Bass River Estuarine System TMDL for TN	MassDEP May 2017	Eelgrass, Benthic Habitat	Ground water (septic)	Summer Seasonal Avg.	0.42 mg/L	V
6	Tampa Bay	Barriers and Bridges in abating Coastal Eutrophication March 2019	Eelgrass	Point Sources	Annual	~0.32 mg/L	See Figure 2 at 9
7	Lagoon Pond TMDL for TN	MassDEP July 2015	Eelgrass Habitat	Ground water (septic)	Summer Seasonal Avg.	0.35 mg/L	V
8	Nantucket Harbor TMDL for TN	MassDEP January 28, 2009	Eelgrass Habitat	Ground water (septic)	Summer Seasonal Avg.	0.35 – 0.36 mg/L (no macroalgae present)	iii

TN Endpoint Summary Table (continued)

No.	Receiving Water/Source	Author	Protected Use	TN Source	Avg. Period	TN Endpoint	Page Citation
9	Green Pond TMDL for TN	MassDEP April, 2006	Eelgrass Habitat	Ground water (septic)	Summer Seasonal Avg.	0.42 mg/L	13
10	Great Pond TMDL for TN	MassDEP April, 2006	Eelgrass Habitat	Ground water (septic)	Summer Seasonal Avg.	0.40 mg/L	13
11	Bournes Pond TMDL for TN	MassDEP April, 2006	Eelgrass Habitat	Ground water (septic)	Summer Seasonal Avg.	0.45 mg/L (shallow)	13
12	Tisbury Great Pond Black Point Pond Estuarine System TMDL for TN	MassDEP December 2017	Eelgrass Habitat	Ground water (septic)	Summer Seasonal Avg.	0.46 mg/L (limited habitat; bathymetry)	iv
13	Three Bays System TMDL for TN	MassDEP September 2007	Eelgrass Habitat	Ground water (septic)	Summer Seasonal Avg.	0.38 – 0.50 mg/L	iii
14	Swan Pond River Estuarine System TMDL for TN	MassDEP May 2017	Eelgrass Habitat	Ground water (septic)	Summer Seasonal Avg.	0.40 mg/L	v
15	West Falmouth Harbor Embayment System TMDL for TN	MassDEP November 2007	Eelgrass Habitat	Ground water (WWTP, septic)	Summer Seasonal Avg.	0.35 mg/L	iii
16	Pleasant Bay System TMDL for TN	MassDEP May 2007	Eelgrass Habitat	Ground water (septic)	Summer Seasonal Avg.	0.16 – 0.20 mg/L bioactive N conc. (DIN + DON) 0.52 mg/L TN	iii
17	Waquoit Bay System TMDL for TN – Jehu Pond/Great River	MassDEP January 2006	Eelgrass Habitat	Ground water (septic)	Summer Seasonal Avg.	0.446 mg/L	12

TN Endpoint Summary Table (continued)

No.	Receiving Water/Source	Author	Protected Use	TN Source	Avg. Period	TN Endpoint	Page Citation
18	Waquoit Bay System TMDL for TN – Hamblin Pond/Little River	MassDEP January 2006	Eelgrass Habitat	Ground water (septic)	Summer Seasonal Avg.	0.38 mg/L	12
19	Waquoit Bay System TMDL for TN – Quashnet River	MassDEP January 2006	Benthic Habitat	Ground water (septic)	Summer Seasonal Avg.	0.50 mg/L	12
20	MEP Linked Watershed-Embayment Approach – Waquoit Bay	MassDEP May 2012	Eelgrass Habitat	Ground water (septic)	Summer Seasonal Avg.	0.327 mg/L	197

ATTACHMENT 10

Summary Report

Technical Review of Select Memorandums Supporting the Development of Nitrogen Endpoints for Three Long Island Sound Watershed Groupings: 23 Embayments, 3 Large Riverine Systems, and Western Long Island Sound Open Water

Prepared for:

U. S. Environmental Protection Agency Region 1
U.S. EPA Contract Number 68HE0118A0001
Order Number 68HE0118F0006

Prepared by:

HydroAnalysis, Inc.

HydroAnalysis

On behalf of:

PARS Environmental, Inc.

PARS ENVIRONMENTAL
a Montrose Environmental Group company

Comprehensive Environmental, Inc.

January 29, 2019

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1. Introduction

Long Island Sound (LIS or "Sound") suffers from periods of low dissolved oxygen (DO) that have led to adverse ecological effects. Concentrations of DO greater than 5 mg/L are considered protective of aquatic life in Long Island. During the summer, DO concentrations in the bottom waters of the Sound often fall below 3 mg/L, an occurrence referred to as hypoxia. Excess loading of nitrogen is the primary cause of hypoxia in the Sound. In addition to the adverse effects to aquatic life, excess nitrogen can also produce algal blooms, decrease water clarity, and limit the growth of submerged aquatic vegetation (Long Island Sound Study, 2018).

In 2016, the U.S. Environmental Protection Agency (USEPA) Region 1 contracted with Tetra Tech to provide technical support with the development of nitrogen endpoints for Long Island Sound, and the calculation of nitrogen load allocations for the LIS watershed. The development of nitrogen endpoints the Sound focused on three categories of waterbodies: 1) 23 embayments; 2) three large riverine systems (Connecticut, Housatonic, and Thames Rivers); and 3) Open water in Western LIS. The project, entitled *Application of Technical Approach for Establishing Nitrogen Thresholds and Allowable Loads for Three LIS Watershed Groupings: Embayments, Large Riverine Systems and Western LIS Point Source Discharges to Open Water,* was completed in March 2018. In order to ensure that the work was conducted using scientifically-sound methodologies consistent with professional and relevant scientific practices, USEPA commissioned an independent technical review of the following technical memorandums (hereinafter, "technical memorandums" or "memorandums") from the project:

1. Summary of Hydrodynamic Analysis (Subtask E Memorandum) (USEPA, 2018a).
2. Summary of Empirical Modeling & Nitrogen Endpoints (Subtask F/G Memorandum) (USEPA, 2018b).

The Hydrodynamic Analysis subtask (Subtask E; USEPA, 2018a) used output from the System Wide Eutrophication Model (SWEM) and other sources to accomplish two key objectives: 1) Define the areas of influence for the Connecticut, Housatonic, and Thames Rivers (i.e., "regions within which water from the rivers exerts a predominant effect on water quality condition"), and calculate their estimated nitrogen loading contributions to select LIS embayments and throughout all of Long Island Sound; and 2) Calculate the relative mixing between open water in LIS and individual embayments.

The results of the Hydrodynamic Analysis subtask (Subtask E; USEPA, 2018a) were used to support the Empirical Modeling & Nitrogen Endpoints subtask (Subtasks F/G; USEPA, 2018b). The objective of the Empirical Modeling & Nitrogen Endpoints subtask was to develop nitrogen endpoints for each of the selected embayments that are protective of seagrass and that prevent adverse effects related to macroalgae and DO. The results from both analyses (Subtask E and Subtask F/G) are going to be used to support the calculation of nitrogen load allocations for the LIS watershed, and to estimate source specific load reductions to meet the nitrogen endpoints (Subtask H).

The goal of the *Empirical Modeling & Nitrogen Endpoints* subtask (Subtasks F/G; USEPA, 2018b) was to develop nitrogen endpoints for the watersheds selected for the study (see Figure F-1 in USEPA 2018b). The candidate endpoints for total nitrogen were developed using the following three empirical approaches (also referred to as "lines of evidence" in the memorandums) (USEPA, 2018b):

1. Scientific Literature Analysis
    a. Identify literature-based nitrogen endpoints (loads and concentrations) from similar estuaries associated with the protection of key assessment/response variables for LIS (e.g., seagrass, aquatic life).
2. Stressor-Response Analysis
    a. Develop nitrogen endpoints using existing water quality data from LIS to establish empirical statistical models of the relationship between chlorophyll *a* and total nitrogen.
    b. Develop chlorophyll *a* endpoints using empirical statistical models of the relationship between key assessment/response variables (seagrass and aquatic life), light availability (Secchi depth or light attention), and DO, as a function of chlorophyll *a*.
3. Distribution-Based Approach
    a. Develop nitrogen endpoint concentrations using the 25th percentile of total nitrogen concentration distributions for LIS embayments and open water stations.

The following is a summary of the final total nitrogen endpoints selected for each of the above empirical approaches (USEPA, 2018b):

1. Scientific Literature Analysis: Median total nitrogen from literature-based values protective of seagrass
    a. Embayments: Range of 0.30–0.50 mg/L; median of 0.39 mg/L, rounded to 0.40 mg/L
    b. Open Water: Range of 0.30–0.60 mg/L; median of 0.41 mg/L, rounded to 0.40 mg/L
2. Stressor-Response Analysis: Mean total nitrogen associated with chlorophyll *a* endpoints
    a. Embayments: Range of 0.06 mg/L–2.52 mg/L
    b. Open Water: Not applicable
3. Distribution-Based Approach: 25th percentile of total nitrogen observed in LIS embayments and open water stations.
    a. Embayments: 0.27 mg/L
    b. Open Water: 0.24 mg/L

2. Technical Review Process and Review Team

2.1. Technical Review Process

HydroAnalysis, Inc. (under USEPA Contract No. 68HE0118A0001 with PARS Environmental and Comprehensive Environmental, Inc.) was commissioned by USEPA to coordinate and manage an independent technical review (hereinafter, "technical review") of two selected technical memorandums from the LIS nitrogen endpoints project (see Section 1). HydroAnalysis' responsibilities included identification and selection of technical reviewers (hereinafter, "technical reviewers", "reviewers", "Review Team", or "Technical Review Team"), coordination of the technical review, production of a summary report for the technical review, and development and delivery of a webinar to inform stakeholders of the outcomes of the review.

HydroAnalysis was given directive authority by USEPA for planning, coordinating, and managing all aspects of the technical review. The USEPA remained independent from the technical review, and did not play a role in the selection of technical reviewers or in the production of the summary report. The USEPA was given an opportunity to review the draft report prior to final publication, and ask for clarification on Review Team responses, if needed. Clarification was not needed.

HydroAnalysis assembled a group of four technical reviewers with expertise in the areas of estuarine water quality (e.g., eutrophication), estuarine ecology and biology (e.g., biological response indicators), and estuarine hydrodynamic and water quality modeling. The reviewer selection process included a screening for independence and conflict of interest. All four reviewers were asked a series of questions concerning potential conflict of interest, and signed forms certifying that they had no conflicts of interest related to the technical review. In addition to considerations of expertise, experience, and conflicts of interest, selection was also based on the reviewer's availability to complete the technical review during the timeframe allotted for the review.

The four technical reviewers were charged with performing an independent review of the two selected technical memorandums from the LIS nitrogen endpoints project, and given specific questions to respond to (see Section 2.3). Each technical reviewer submitted written responses to the review questions directly to HydroAnalysis. The technical reviewers did not communicate with one another during the review process. The reviewers also did not communicate with USEPA or with Tetra Tech during the review process or during the development of this summary report.

HydroAnalysis reviewed the Review Team responses, and coordinated closely with the reviewers to obtain clarification on responses as needed, and to obtain agreement for recommended edits to address major grammatical or spelling errors. None of the edits modified, interpreted, or enlarged upon the technical reviewer's responses. The reviewers were given an opportunity to review the draft report, and provide clarification or corrections, if needed. The responses of the Review Team as provided in this report (see Section 4) represent the individual opinions and assessments of each of the technical reviewers.

2.2. Technical Review Team

Brief descriptions of the experience and areas of expertise for each of the technical reviewers are provided below.

Victor J. Bierman, Jr., Ph.D., BCEEM

Dr. Victor Bierman is a Senior Scientist Emeritus at LimnoTech with 45 years of experience in the development and application of water quality models for eutrophication and the transport and fate of toxic chemicals, leading to his publication of over 100 technical papers and reports. He is a former USEPA National Expert in Environmental Exposure Assessment, and a former Associate Professor in the Department of Civil Engineering at the University of Notre Dame. He is also a Board Certified Environmental Engineering Member (by Eminence) of the American Academy of Environmental Engineers and Scientists. Dr. Bierman conducts research and development on projects for federal, state and regional government clients. He also provides scientific peer review, litigation support, and expert testimony on a variety of environmental issues for government agencies, and industrial, regulatory and private clients. Dr. Bierman is a leading expert in the assessment and solution of problems related to nutrients, DO, nuisance algal blooms, nitrogen fixation, exotic species, and ecosystem processes. He has conducted studies in watersheds, lakes, major rivers, estuaries, coastal marine systems, the Great Lakes, and at USEPA Superfund sites. Key accomplishments by Dr. Bierman related to the topic of this review include service as Panel Chair for a scientific peer review of the Massachusetts Estuary Project (MEP) linked watershed-embayment model for protection of eelgrass and aquatic life, service as a consultant to the USEPA Science Advisory Board for peer review of draft technical guidance on using stressor-response models to derive numeric nutrient criteria, and service on a scientific peer review panel for numeric nutrient criteria for protection of eelgrass in the Great Bay Estuary, New Hampshire.

Mark J. Brush, Ph.D.

Dr. Mark Brush is an Associate Professor of Marine Science at the Virginia Institute of Marine Science (VIMS) in Gloucester Point, VA, part of the College of William and Mary. Dr. Brush received his B.S. in Biological Sciences from Cornell University in 1995 and his Ph.D. in Biological Oceanography from the University of Rhode Island in 2002, and has been at VIMS since 2002 as a postdoctoral fellow, research scientist, and faculty member. His research program focuses on the ecology of coastal marine ecosystems such as estuaries and lagoons, through field- and lab-based ecological investigations, synthesis of water quality monitoring data, and interdisciplinary ecosystem simulation modeling. Recent projects have focused on modeling the response of coastal systems to nutrient enrichment and climate change, with a focus on water quality and ecosystem function, quantifying coastal

- Tables G-10 and G-12 have an extra footnote referencing a population model. Why was a different model used relative to the other tables (especially given all the data present in these two systems)?

Dr. Janicki's Response

The presentation of the TN endpoints and targets was adequate and should be understandable to most readers. The hierarchical modeling graphics also should be understandable to most readers.

Dr. Justic's Response

The TN endpoints and targets are clearly explained and the graphs are easily understandable.

3. Comment specifically on the methods used to recommend TN endpoints. Are the methods used to identify recommended TN endpoints and ranges scientifically valid and laid out in a clear way? Are the TN endpoint values reasonable for protection of the region? Are the assumptions clearly presented? What are the minimum data requirements for applying the methods to establish TN endpoints applicable to individual embayment whether for purposes of protecting Long Island Sound or the embayment itself? What considerations should be given to application of the methods to non-homogenous embayments to ensure that the TN endpoints are protective of all portions of the embayment?

Dr. Bierman's Response

The LRA method is scientifically valid and laid out in a clear way. It is always a good first step because it allows identification of TN concentrations and ranges corresponding to various assessment endpoints (e.g., eelgrass and aquatic life) in other similar waterbodies. It also allows identification of relevant response variables and confounding factors that should be considered in attempting to link TN concentrations to these assessment endpoints. Although the LRA method can provide a useful screening-level analysis, it should not be assumed that specific TN concentrations and ranges from other waterbodies can be directly translated to LIS because these concentrations are strongly site-specific.

The memorandum states on Pages F-2 and F-3 that a decision was made to focus primarily on TN values from the most proximate study areas (Massachusetts) and not to incorporate values from farther north (Great Bay, NH) or south (Chesapeake Bay) because those systems were considered substantially different. This approach assumed that the Massachusetts estuaries literature-based targets were appropriate for LIS, given the similarities in geography, climate, and species composition (e.g., Zostera marina) consistent with similar physical and chemical habitat requirements in both embayment as well as shallow and deeper open water habitats between the two regions. Consequently, many of my comments on the memorandum draw upon approaches, analyses, and findings from the Massachusetts Estuaries Program (MEP).

The SRM methods themselves are scientifically valid, but not laid out in a clear way in the memorandum. USEPA (2010) recommends summarizing and visualizing datasets before conducting SRM statistical analyses, but this was not done in the memorandum. In addition, the applications of the SRM methods to LIS contain conceptual flaws and questionable assumptions, and their results do not provide scientifically valid support for the TN endpoints.

The DbA is a broad, generic approach that can be useful at regional scales and is laid out in a clear way in the memorandum. Selection of TN concentration targets by using the 25th percentile of all TN samples in LIS embayments and open waters (Table F-10) is consistent with USEPA protocol; however, because the DbA in the memorandum did not explicitly use any site-specific data for eelgrass distributions, the primary response variables (chlorophyll *a*, K_d, DO) or eelgrass physical habitat requirements (sediment grain size and total organic carbon), there is no assurance that these 25th percentile TN targets will protect the LIS assessment endpoints (eelgrass, aquatic life).

The values from the LRA appear reasonable, but are not based on site-specific data from the LIS embayments. The values from the DbA appear reasonable, but they are based only on site-specific TN concentrations and not on any other parameters directly related to eelgrass or aquatic life. The values from the SRM are conceptually flawed and scientifically invalid (see my responses to Questions 10a – 10f for details and specific examples.

With regard to minimum data requirements, the memorandum states on Page F-1 that seagrasses (eelgrass) and other aquatic life were selected for developing nitrogen endpoints. It states that these assessment endpoints are principally reflected by water column chlorophyll *a* (through its effect on light for seagrass growth) and DO (through its effect on benthic fauna and fishes). These statements are accurate but do not reflect all of the site-specific parameters that should be considered for applying the methods to establish TN endpoints for purposes of protecting Long Island Sound or the embayments themselves. For example, as stated on Page 200 in Howes et al. (2006):

> "Determination of site-specific nitrogen thresholds for an embayment requires the integration of key habitat parameters (infauna and eelgrass), sediment characteristics data and nutrient related water quality information (particularly dissolved oxygen and chlorophyll a)."

Koch (2001) acknowledges that light and parameters that modify light (epiphytes, total suspended solids, chlorophyll *a*, nutrients) are the first factors to consider when determining habitat suitability for seagrass, but points out that these factors alone do not explain why seagrass does not occur in areas where light levels are adequate. He goes on to emphasize the importance of also considering physical-chemical factors such as current velocity, waves, tides, salinity, sediment grain size distribution (GSD), sediment total organic carbon (TOC), and sediment sulfide concentration.

In the memorandum, the TN endpoint values from the LRA are based on those developed for other, proximate systems and not on site-specific data from LIS. The values from the DbA are based only on site-specific TN concentrations and not on any of the other above parameters. The independent variables in the final SRMs include chlorophyll *a*, TN, pH, salinity, and temperature, but none of the other above parameters. It is not known whether any of these other parameters were considered in the SRMs because the memorandum lists only the independent variables in the final models, not all of those that were actually investigated.

To ensure that the TN endpoints are protective of all portions of the embayment when applying the methods to non-homogenous embayments, it would be appropriate to consider the sentinel station approach used in the MEP. As stated on Page 204 in Howes et al. (2006):

> *"The approach for determining nitrogen loading rates, which will maintain acceptable habitat quality throughout an embayment system, is to first identify a sentinel location within the embayment and second to determine the nitrogen concentration within the water column which will restore that location to the desired habitat quality (threshold nitrogen level). The sentinel location is selected such that the restoration of that one site will necessarily bring the other regions of the system to acceptable habitat quality levels."*

See my specific responses to Questions 8, 10 and 11, for related discussion on this topic, including on the manner in which the assumptions are presented in the memorandum.

Dr. Brush's Response

First, I strongly support the use of chlorophyll *a*, light attenuation, and DO as assessment endpoints; these are the exact endpoints used by the long-standing USEPA Chesapeake Bay Program (CBP) and were developed after extensive deliberation over many years of work. If USEPA wishes to further pursue benthic fauna, they could look into the CBP DO criteria which specifically addressed estuarine fauna by thoroughly evaluating the literature for faunal-DO relationships.

The use of a multiple lines of evidence approach to establish TN endpoints, with uncertainty ranges in the case of two methods, is in line with best practice and existing approaches, and in my view excellent. The three approaches are scientifically valid and clearly presented. The methods for each approach were also generally well explained, with some caveats provided in the relevant sections below. Some of these caveats relate to issues with textual clarity and terminology; these do not take away from the validity of the analyses and can be addressed with some relatively simple clarifications in the memo. Caveats in the Stressor-Response Modeling section raise more important methodological issues which I believe should be addressed prior to final acceptance of those TN endpoints. That said, I found the conclusions reached after each analysis to be well supported by the data and analyses.

ATTACHMENT 11

STATE OF MAINE
DEPARTMENT OF ENVIRONMENTAL PROTECTION

PAUL R. LEPAGE
GOVERNOR

PATRICIA W. AHO
COMMISSIONER

April 23, 2015

NAME
ADDRESS

RE: Effluent Nutrient Sampling for MEPDES Permits/Maine WDLs

Dear XXXX:

This letter is to inform you that the Maine Department of Environmental Protection (Department) is in the process of determining the potential need for water quality-based total nitrogen limits in Maine Pollutant Discharge Elimination System Permit/Waste Discharge Licenses (MEPDES/WDLs) for wastewater discharges directly to marine water, as well as for freshwater discharges located in reasonable proximity to marine Head of Tide (HoT). This letter has been sent to 32 select major marine dischargers and those above HoT for which no facility-specific effluent nitrogen or phosphorus data have been made available to the Department.

Via this letter, the Department is requesting that you voluntarily collect samples from your effluent discharge this summer. The Department has contracted with a Maine-certified, commercial laboratory to provide sample handling information, containers and pre-paid shipping labels, and to analyze the effluent samples at no cost to you (see Appendix A for sampling schedule and details).

Your participation will help ensure that the most accurate nutrient data possible are available to the Department when your facility MEPDES/WDL is renewed in the future. These data will enable the Department to determine if total nitrogen limits are necessary for your facility, and to establish them appropriately if they are required.

Background
The regulation of nutrients such as total phosphorus and total nitrogen in waste discharge permits under the Clean Water Act, has received increased national attention over the last several years. Nutrient enrichment can cause negative environmental impacts to surface waters, such as algal blooms, low dissolved oxygen concentrations, fish kills, and shifts in the biological community to more pollution tolerant species, all of which could cause non-attainment of water quality standards. To better manage nutrient enrichment, the EPA has required that states develop and adopt numeric criteria for phosphorus and nitrogen for all jurisdictional waters and requires states to report annually on progress toward this goal. The Department has been developing

nutrient criteria to incorporate into Maine's water quality standards for the last several years, and is initially focusing on nitrogen for marine waters.

Nitrogen is generally a limiting nutrient in marine waters, and phosphorus is generally a limiting nutrient in fresh waters. However, the Department regularly collects paired ambient nitrogen and phosphorus data. This sampling regime is based on recent EPA guidance[1] as well as the dynamic nature of marine waters, including estuaries, the differing nutrient requirements of marine algae, and the need to protect downstream waters (relevant for locations above HoT). A more complete understanding of nitrogen and phosphorus loading to marine receiving waters and freshwaters upstream of HoT will enable a comprehensive assessment of influences on water quality standards, and permit more informed decisions for nutrient reductions, if and where necessary.

Regulatory Authority
Department Regulation, Chapter 523 specifies that water quality-based limits are necessary when the Department has determined that a discharge has Reasonable Potential to cause or contribute to an excursion above any State water quality standard, including State narrative criteria.[2,3] In addition, Chapter 523 specifies that water quality based limits may be based upon criteria derived from a proposed State criterion, or an explicit State policy or regulation interpreting its narrative water quality criterion, supplemented with other relevant information. Supplemental information may include EPA's Water Quality Standards Handbook (October 1983), risk assessment data, exposure data, current EPA criteria documents, or using EPA's Water quality criteria, published under section 304(a) of the CWA supplemented where necessary by other relevant information.[4]

Recent correspondence with EPA indicates that all permits for discharges to fresh and marine waters must contain a Reasonable Potential analysis to determine if water quality based limits are needed for total nitrogen and/or phosphorus.

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[1] USEPA. 2015. Preventing Eutrophication: Scientific Support for Dual Nutrient Criteria. EPA-820-S-15-001, USEPA, Office of Water, Office of Science and Technology, Washington, DC.
[2] *Waste Discharge License Conditions,* 06-096 CMR 523(5)(d)(1)(i) (effective date January 12, 2001)
[3] State narrative water quality criteria include descriptions of allowable impacts to marine habitat and water quality necessary to support designated uses such as recreation in and on the water. Standards for Classification of Estuarine and Marine Waters may be found at 38 MRSA Sec. 465-B.
[4] 06-096 CMR 523(5)(d)(1)(vi)(A)

Reasonable Potential (RP) Calculation
The RP calculation to determine if a total nitrogen limit is needed consists of the following:

$$C_{ff} = \frac{C_{eff}}{DF_{ff}} + C_a$$

C_a = ambient nutrient concentration
C_{eff} = effluent nutrient concentration
C_{ff} = far field, in-stream concentration
DF_{ff} = far field dilution factor

Based on this calculation, if the resulting concentration (C_{ffr}) is above the interim total nitrogen threshold for the receiving water (0.32 mg/L in proximity to eelgrass or 0.45 mg/L in the absence of eelgrass), the discharge is determined to have a Reasonable Potential to cause or contribute to an excursion above applicable water quality standards. These interim nitrogen thresholds are based on data from Maine, New Hampshire and Massachusetts, and are subject to change based on the Department's nutrient criteria development process. If an exceedance of a threshold value occurs based on the RP calculation, the Department will determine the potential need to establish water quality based limits and/or the appropriate monitoring requirements.

Request for Collection of Effluent Data
The Department is requesting that you collect monthly samples of your effluent this summer from June through October to enable an accurate characterization of effluent nitrogen and phosphorus concentrations. Each sample should be a 24-hour composite and collected on the Monday, Tuesday or Wednesday of the first week of each month to enable overnight shipping no later than Wednesday of each week. Each 24-hour composite sample should be collected and shipped within the following date ranges:

Event 1: June 1-3
Event 2: July 6-8
Event 3: August 3-5
Event 4: August 31-September 2
Event 5: October 5-7

Samples will be analyzed for Total Kjeldahl Nitrogen, nitrate + nitrite, orthophosphorus and total phosphorus. Effluent samples should be mailed to the Department-contracted and certified laboratory for this project, ALS Environmental, in Rochester, New York. Sample data and quality assurance information will be provided directly to the Department, and will be made available to the respective facilities after a complete data quality check has occurred. <u>Costs associated with effluent sample shipping and analyses will be covered entirely by the Department.</u>

Sample collection, handling, packaging and shipping details are provided in Appendix A. The Department will provide analytical laboratory SOPs, as requested.

Closing

The Department must include Reasonable Potential calculations for each discharge permit upon MEPDES/WDL renewal, and will be able to more accurately assess the Reasonable Potential of your discharge to cause or contribute to non-attainment of water quality standards with knowledge of your nutrient load. Your participation will help provide the most accurate nutrient data possible are available to the Department to ensure that permit limits can be established only when necessary. Your assistance in collecting these samples would be very much appreciated. If the Department is unable to obtain facility-specific effluent data through this process, we will rely on estimated total nitrogen levels based on existing data from other facilities for Reasonable Potential analysis.

Intent to Participate

Please contact Angela Brewer (angela.d.brewer@maine.gov; 207-592-2352) by *Friday, May 8*, to indicate intent to participate in effluent nutrient sampling.

Questions

Please direct any questions as follows:

Effluent Sampling: Angela Brewer (angela.d.brewer@maine.gov, 207-592-2352)
Regulatory: Brian Kavanah (brian.w.kavanah@maine.gov, 207-287-7700)

Sincerely,

Brian Kavanah, Director
Division of Water Quality Management
Bureau of Land and Water Quality

cc: Mick Kuhns, Don Witherill, Rob Mohlar, Angela Brewer, Gregg Wood, all DEP facility inspectors. – DEP
Tom Connolly – MeWEA
Kirsten Hebert - MRWA
Janice Jaeger – ALS Environmental, Inc.

Attachments: Appendix A – *Protocol for Effluent Nutrient Sampling*

Appendix A: Protocol for Effluent Nitrogen and Phosphorus Sample Collection

Please contact Angela Brewer (angela.d.brewer@maine.gov; 207-592-2352) by *Friday, May 8,* to indicate intent to participate in effluent nutrient sampling.

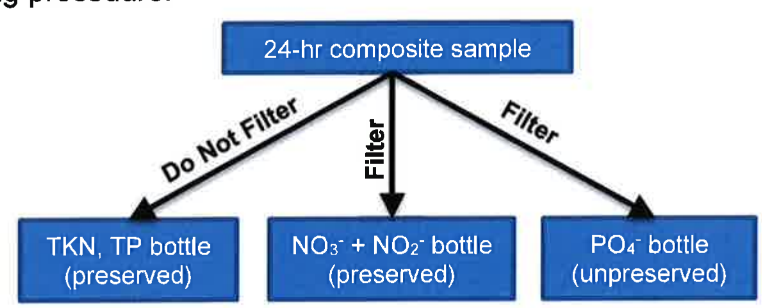
Collect post-treatment, pre-discharge composite samples of your effluent over a 24-hour period. Divide the composite sample into three sample bottles, and ship following the below schedule:

Event 1: June 1-3, 2015
Event 2: July 6-8, 2015
Event 3: August 3-5, 2015
Event 4: August 31-September 2, 2015
Event 5: October 5-7, 2015

During compositing, keep composite sample at 0-6 °C (without freezing) in a glass or polyethylene bottle or jug. Clean the bottle or jug prior to each use with dilute H_2SO_4. Follow cleaning with several rinses of distilled water. Commercially purchased, pre-cleaned sample containers are an acceptable alternative. Clean sampler hoses as needed.

The lab will provide three, 250 mL plastic bottles per sampling event. Two bottles will contain H_2SO_4 preservative (yellow sticker on top) and one will be unpreserved. All bottles will have pre-applied labels that will indicate the analyte and provide space to record the facility name, and date and time the composite sample was completed.

When the composite sample is complete, mix the bottle or jug contents and then pour a subsample into the bottle labeled "TKN, TP" until the bottle is ¾ full. The other two bottles require filtered sample. Syringes and filters will be provided by the Department. For each of the NO_3^- + NO_2^- bottle and the PO_4^- bottle, rinse the syringe barrel three times with the composite sample, then pull composite sample into barrel, screw on filter tip, and dispense sample into bottle. Repeat until each bottle is ¾ full. You may need to switch to a new filter if it becomes too difficult to dispense sample using the initial filter. Do not rinse the bottles prior to adding the composite sample. Cap each of three bottles tightly and refrigerate at 0-6 °C (without freezing) until shipping. See schematic for illustrated subsampling procedure:



A Chain of Custody (CoC) form will be provided by the lab. Fill out and mail a CoC with each cooler shipment. Pack lab-provided sample coolers with **wet ice** as tightly as possible and liberally tape cooler shut. Ship coolers overnight via FedEx using lab-supplied, pre-paid shipping labels. **Shipping should occur no later than Wednesday of the given week.**

ATTACHMENT 12

STATE OF MAINE
DEPARTMENT OF
ENVIRONMENTAL PROTECTION

PAUL R. LEPAGE
GOVERNOR

MELANIE LOYZIM
ACTING COMMISSIONER

December 1, 2018

Mr. Robert Clark
96 Clearwater Drive
Falmouth, ME. 04105

RE: Maine Pollutant Discharge Elimination System (MEPDES) Permit #ME0100218
Maine Waste Discharge License (WDL) #W002650-6D-I-R
Final Permit

Dear Mr. Clark:

Enclosed please find a copy of your **final** MEPDES permit and Maine WDL **renewal** which was approved by the Department of Environmental Protection. Please read this permit/license renewal and its attached conditions carefully. Compliance with this permit/license will protect water quality.

Any interested person aggrieved by a Department determination made pursuant to applicable regulations, may appeal the decision following the procedures described in the attached DEP FACT SHEET entitled "*Appealing a Commissioner's Licensing Decision.*"

If you have any questions regarding the matter, please feel free to call me at 287-7693. Your Department compliance inspector copied below is also a resource that can assist you with compliance. Please do not hesitate to contact them with any questions.

Thank you for your efforts to protect and improve the waters of the great state of Maine!

Sincerely,

Gregg Wood
Division of Water Quality Management
Bureau of Water Quality

Enc.

cc: Matt Hight, MDEP/SMRO
Sandy Mojica, USEPA
Ivy Frignoca, Casco Bay Keeper
Lori Mitchell, MDEP/CMRO
Marelyn Vega, USEPA

STATE OF MAINE
DEPARTMENT OF ENVIRONMENTAL PROTECTION
17 STATE HOUSE STATION AUGUSTA, MAINE 04333-0017

DEPARTMENT ORDER

IN THE MATTER OF

TOWN OF FALMOUTH PUBLICLY OWNED TREATMENT WORKS FALMOUTH, CUMBERLAND COUNTY, ME ME0100218 W002650-6D-I-R **APPROVAL**	)))))	MAINE POLLUTANT DISCHARGE ELIMINATION SYSTEM PERMIT AND WASTE DISCHARGE LICENSE **RENEWAL**

Pursuant to the provisions of the Federal Water Pollution Control Act, Title 33 USC, Section 1251, et. seq. and Maine Law 38 M.R.S., § 414-A et seq., and applicable regulations, the Department of Environmental Protection (Department hereinafter) has considered the application of the TOWN OF FALMOUTH (Town/permittee hereinafter), with its supportive data, agency review comments, and other related material on file and finds the following facts:

APPLICATION SUMMARY

The Town has submitted a timely and complete application to the Department for the renewal of combination Maine Pollutant Discharge Elimination System (MEPDES) permit #ME0100218/Maine Waste Discharge License (WDL) #W002650-6D-G-R (permit hereinafter) which was issued by the Department on February 21, 2013, for a five-year term. The 2/21/13 permit authorized the discharge of up to a monthly average flow of 1.56 million gallons per day (MGD) of secondary treated sanitary waste waters from a publicly owned treatment works facility to the Presumpscot River estuary, Class SC, in Falmouth, Maine.

PERMIT SUMMARY

This permitting action is carrying forward all the terms and conditions of the previous permitting actions <u>except</u> that it:

1. Removes a monthly average water quality based mass limitation and concentration reporting requirement for total copper as a recent statistical evaluation indicates none of the most current 60 months of test results exceeds or has a reasonable potential to exceed applicable ambient water quality criteria (AWQC).

2. Incorporates a special condition requiring the permittee to immediately report all discharges of untreated waste water to the Maine Department of Marine Resources (DMR). This information will assist the DMR in determining whether to close conditionally approved shellfish harvesting areas impacted by the discharges.

6. EFFLUENT LIMITATIONS AND MONITORING REQUIREMENTS (cont'd)

j. Nitrogen - The USEPA requested the Department evaluate the reasonable potential for the discharge of total nitrogen to cause or contribute to non-attainment of applicable water quality standards in marine waters, namely aquatic life use support. The permittee voluntarily participated in a Department-coordinated project to determine typical effluent nitrogen concentrations, and submitted monthly composite samples from May-October, 2008 (n = 6). The mean value of the permittee's six samples was 7.9 mg/L. Although a small sample size, this 2008 mean value compares well with internal total nitrogen data generated by the facility between 2011 and 2017 (n= ~200) that indicate a mean value differing by 0.1 mg/L. For this reasonable potential evaluation, the Department considers 7.9 mg/L to be representative of total nitrogen discharge levels from the Falmouth facility.

With the exception of ammonia, nitrogen is not acutely toxic; thus, the Department is considering a far-field dilution to be more appropriate when evaluating the more systemic types of influences associated with total nitrogen in the marine environment. Falmouth discharges to the estuarine portion of the Presumpscot River, which is a relatively confined, tidal flat-dominated embayment that empties into the inner Casco Bay. The tidally averaged flushing rate of the Presumpscot River estuary (head of tide to the Route 1 Bridge) is approximately 3,415 cfs (2,200 MGD). Based on the Department's hydraulic modeling of the Presumpscot River estuary, the far-field dilution factor for Falmouth's discharge has been determined to be approximately 1,410:1 (see calculation below).

Tidal Flushing Volume = 2,200 MGD
Discharge Flow Rate= 1.56 MGD

$$\frac{2{,}200 \text{ MGD}}{1.56 \text{ MGD}} = 1{,}410:1$$

Total nitrogen concentrations in effluent = 7.9 mg/L
Far-field dilution factor = 1,410:1

In-stream concentration after dilution: $\frac{7.9 \text{ mg/L}}{1{,}410} = 0.006$ mg/L

As of the date of this permitting action, the State of Maine has not promulgated numeric ambient water quality criteria for total nitrogen. According to several studies in USEPA's Region 1, numeric total nitrogen criteria have been established for relatively few estuaries, but the criteria that have been set typically fall between 0.35 mg/L and 0.50 mg/L to protect marine life using dissolved oxygen as the indicator. While the thresholds are site-specific, nitrogen thresholds set for the protection of eelgrass habitat range from 0.30 mg/L to 0.39 mg/L. Based on studies in USEPA's Region 1 and the

6. EFFLUENT LIMITATIONS AND MONITORING REQUIREMENTS (cont'd)

Department's best professional judgment of thresholds that are protective of Maine water quality standards, the Department is utilizing a threshold of 0.45 mg/L for the protection of aquatic life in marine waters using dissolved oxygen (DO) as the indicator, and 0.32 mg/L for the protection of aquatic life using eelgrass as the indicator. Due to the absence of mapped eelgrass within the estuary (see below paragraphs), the Department is using a threshold value of 0.45 mg/L to protect aquatic life using dissolved oxygen as the indicator.

Beyond the salt marsh channel to which the Falmouth effluent is discharged, the vast majority of the Presumpscot River estuary is intertidal and therefore the only suitable eelgrass habitat is along the low intertidal and shallow subtidal banks within the narrow channel (see low tide imagery in Fig. 1). The nearest suitable eelgrass habitat is approximately 0.6 km from the discharge location. Four known surveys have been completed within the Presumpscot River estuary that have documented presence/absence of eelgrass. The 1970's Timson (Maine Geological Survey) Coastal Marine Geological Environments information referenced in other marine discharge permits is not being utilized for this permit due to deficiencies in the aerial imagery and groundtruthing methods used for eelgrass delineation. The first and second eelgrass surveys considered in this permit occurred in 1993 and 2001 by the Maine Department of Marine Resources, and the third and fourth in 2013 and 2017 by the Maine Department of Environmental Protection. None of the four surveys documented eelgrass within the Presumpscot River estuary, and consistently identified eelgrass no closer than the southeastern side of Mackworth Island (4 km from the discharge location). June 2018 draft aerial imagery currently under review by a DEP contractor similary has not indicated eelgrass presence within the Presumpscot River estuary.

The Department and external partners have been collecting ambient total nitrogen data along Maine's coast. For the vicinity of the Falmouth discharge, the Department calculated a weighted mean background concentration of 0.34 mg/L (n = 35) based on surface water data collected at three sites (Figure 1, Table 1) within and just outside of the Presumpscot River estuary between May and October of a given year. The weighted mean value was calculated to account for differences in sample size between sites bracketing the estuary as well as considerably more water volume entering the estuary from Casco Bay as compared to the Presumpscot River between May and October. Further, and to avoid potential influence of the Falmouth discharge on the background calculation, total nitrogen data were only used from late ebb or slack low tides for Site #1 (PRV70), and from late flood or slack high tides for Site #2 (PRVRT1) and Site #3 (CBPR). Use of this data subset is intended to represent typical total nitrogen concentrations entering the estuary from the non-tidal River and Casco Bay, respectively (Figure 1, Table 1). Although additional total nitrogen data are available from sites in the vicinity of the East End of Portland, these data may be more directly influenced by the