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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.

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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. S**ystem 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.

<u>Conclusions:</u> 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.