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June 18, 2013

Ms. Susan Murphy
U.S. Environmental Protection Agency – Region 1
5 Post Office Square
Suite 100 (OEP06-1)
Boston, MA 02109-3912

Re: NPDES Permit No.: MA0100897
Public Notice Number: MA-010-13
Taunton, MA
Comments on Draft Permit

Dear Ms. Murphy:

The City of Taunton ("Taunton or the "City") submits the comments herein on the proposed modifications of Taunton's NPDES Permit No. MA0100897 that were published for comment by EPA on March 20, 2013. The deadline for filing comments was extended at the request of the City, by EPA, to June 20, 2013. This new nitrogen limit for the Taunton permit is reflective of EPA's and the Massachusetts Department of Environmental Protection's (MassDEP) concern about nutrient loadings to the Taunton River and ultimately Mount Hope Bay. Taunton shares the concern of the federal and state governments about the health of Mount Hope Bay and acknowledges that it and other point sources discharge nitrogen from its wastewater treatment facilities (WWTF) into the Taunton River. Taunton also recognizes that there are significant non-point sources of nitrogen contributing to the Taunton River Watershed. We appreciate that upgrades to the Taunton WWTF, and others, may be necessary to ensure compliance with applicable standards.

The comments filed today by the City indicate that it is not possible to reliably identify the degree of nitrogen control required to ensure compliance with applicable standards using the methodology employed by EPA. Many changes in plant performance have been implemented in this and other basins since 2004/2005. Moreover, the conditions governing dissolved oxygen concentrations in Mount Hope Bay differ significantly from those in the Taunton River. This reality impacts the degree to which the City and other municipal wastewater treatment plants discharging into Taunton River must reduce their nitrogen loading. The question is whether the nitrogen limit included in the draft permit (a monthly average concentration of 3 mg/l) is supported by current data and analyses. The data used in the Fact Sheet for the Draft NPDES Permit is from 2004-2005. Since that time, water quality in Mount Hope Bay has improved markedly due to the CSO deep tunnel project in Fall River, the construction of cooling towers at the Brayton Point Station and improvements to some upstream wastewater treatment plants. The beneficial effect of these changes on the Taunton River and Mount Hope Bay is apparent in more recent data, but was not assessed by EPA in rendering this permit decision. Therefore, more recent data should be used for analysis of nitrogen loading for the WWTP point source discharges to the Taunton River.

The City has committed to begin promptly planning for an upgraded WWTF that will achieve appropriate total nitrogen concentrations in its discharge. A "Draft Environmental Impact Report and Final Comprehensive Wastewater Management Plan" was submitted to MassDEP in July 2009. Although discussions of nitrogen removal technologies were presented in the plan, it was never finalized as permit limitations for Total Nitrogen had not been developed by regulating authorities. Work to complete the plan will commence as soon as all comments regarding the draft NPDES permit are considered and the final permit is issued.

The MADEP has initiated a program to publish TMDLs for watersheds throughout Massachusetts. Rhode Island is also in the process of TMDL evaluation for Narragansett Bay. The MADEP has been underfunded and understaffed in its effort to complete the TMDLs. Because the State does not have enough money the EPA has imposed an economic hardship on the three largest WWTP discharges to the Taunton River by requiring the most restrictive "Limits of Technology" for the upgrades of their wastewater treatment plants. Two of these communities Taunton and Brockton have significant Environmental Justice areas that support he need for reconsidering this decision.

Based on the comments provided in Attachment 1, Taunton requests that EPA and MassDEP reconsider their decision to impose a Limit of Technology standard for total nitrogen in Taunton's NPDES Permit. Other conditions established by the draft permit are also questioned. Additional comments developed on behalf of Taunton, by Hall and Associates, are included in Attachment 2 to this correspondence.

Thank you for your careful consideration of these comments.

Respectfully,

Thomas C. Hoye, Jr.
Mayor

Attachments:

cc: John M. McCaul, Council President
Jason D. Buffington, City Solicitor
Fred Cornaglia, DPW
Joseph Federico, BETA Group, Inc.
John C. Hall, Hall & Associates

Attachment 1: Comments Submitted by the City of Taunton

Attachment 1

Comments Submitted by the City of Taunton

1. Inappropriate Interpretation of the Massachusetts Narrative Criteria

There remains significant uncertainty with respect to appropriate numeric nutrient criteria that should be used to establish discharge limits for treatment facilities in the Taunton River, Mount Hope Bay, and Narragansett Bay systems. The MassDEP and the Rhode Island Department of Environmental Management have not adopted numeric nutrient criteria for these surface water bodies and existing Surface Water Criteria in both states rely on narrative criteria, only. (See comments by Hall & Associates, provided in Attachment 2, also addressing this issue).

To include the proposed nitrogen limit in the draft NPDES permit, EPA has relied on interim, un-adopted numeric criteria serving as a translator of the narrative criteria established in State's Surface Water Quality Standards. The numeric criteria used were presented in an interim report (Massachusetts Estuaries Project – Site Specific Nitrogen Thresholds for Southeastern Massachusetts Embayments: Critical Indicators) prepared by the School for Marine Science and Technology at the University of Massachusetts Dartmouth. However these numeric thresholds, which were developed for three Cape Cod embayments in the Town of Falmouth, MA, were never subject to public comment and may not be applicable to the Taunton River, Mount Hope Bay and Narragansett Bay. Relying on data from dissimilar water bodies brings a high level of uncertainty with respect to the numeric criteria needed to protect the Taunton River, Mount Hope Bay and Narragansett Bay.

The report states: "it is not possible at this time to put quantitative nitrogen levels on each Water Quality Class. In fact, initial results of the Massachusetts Estuary Project (Chatham Embayment Report 2003) indicate that the total nitrogen level associated with a particular ecological response can vary by over 1.4 fold". The report goes on to conclude that "before final criteria are established, several habitat quality classification issues need to be resolved, including, but not limited to: variation in multiple indicators, embayments versus salt marsh habitat, upper versus lower embayment thresholds, and stable versus transitional habitat quality". Since such activities have not occurred, reliance on the Critical Indicators report to classify the Taunton River as nutrient impaired or to set ambient water quality targets is inappropriate and unsupported.

2. Proposed Nitrogen Limits are Unattainable

As stated above, Taunton does not believe EPA has a sound scientific basis to impose a limit of technology nitrogen limit. Even if EPA had sound reason to establish a limit of technology limit, the EPA has insufficient basis to establish that limit at 3 mg/l for several reasons. The first is that limits of technology need to be discussed in the context of a time period. What is achievable on an annual or seasonal average basis is different than what is achievable on a monthly average basis. EPA has inappropriately taken average seasonal limit of technology expectations and applied them as monthly limits. Section VI B. 5 of the Fact Sheet states: "The permit limit is 3.0 mg/l total nitrogen as a seasonal average, and a mass limit of 210 lbs/day….". Attachment D to the Fact Sheet (Page 8) also refers to the Total Nitrogen limit as seasonal and specifically states "The seasonal limit shall be applied on a rolling basis (e.g. the average reported for June shall include May and June of the reporting year as well as July through October of the preceding year)". However, the concentration and mass limits in the permit are identified as monthly averages not seasonal averages. Seasonal (May thru October/6-month rolling average) total nitrogen limit are the more appropriate permit basis.

EPA's Municipal Nutrient Removal Technologies Reference Document (2008, p. 2-80) references several factors that affect nitrogen removal efficiency. One factor that can influence how low the TN can be reduced is the dissolved organic nitrogen (DON) concentration. At this point, the DON concentration in Taunton's wastewater is not known and its impact on water quality is anticipated to be negligible. This will be explored in more depth as part of the Final Comprehensive Wastewater Management Plan. Effluent DON concentrations reported in various literature sources range from 0.4 mg/l to 2.2mg/l with an average concentration of approximately 1.3 mg/l. EPA's reference document also states that "The DON concentration is a critical variable for determining TN standards because the chemicals have limited availability for biological removal". Likewise, this parameter is not shown to have a stimulatory effect on plant growth in the River.

Absent this data, EPA cannot set the standard at the limit of technology with certainty or claim control of DON is necessary to protect the River. In the absence of DON data, EPA should consider a total inorganic nitrogen limit consisting of nitrite and nitrate nitrogen plus ammonia since these are the forms of nitrogen that are biologically available. This concept is further supported by an EPA publication entitled " An Urgent Call to Action Report in the State-EPA Nutrient Innovations Task Group" (August 2009) that discusses technology based limits for nitrogen in terms of nitrate and nitrite, only (see **Attachment 1.A**). We have included ammonia (ammonium) in the nitrogen standard because of its bio-availability.

Over the past few years, Connecticut communities have had to upgrade treatment facilities with state of the art technology to reduce nitrogen levels to the limits of technology in order to meet the requirements of the Long Island Sound total maximum daily load. The table below is a compilation of the 2010 data from ten of the recently upgraded plants in Connecticut.

Although these plants are producing low total nitrogen concentrations, individual monthly data (maximum month) from April through October indicates that the 3 mg/l limit cannot be achieved at all times. This also holds true for the average monthly concentration over the same April through October period. Setting a permit concentration at the limit of technology, requires a treatment facility to achieve discharge concentrations below that limit. By definition, this cannot be accomplished on a consistent basis and will result in persistent permit violations.

At a minimum, the EPA should consider defining total nitrogen as the sum of nitrite-N, nitrate-N and ammonia. Additionally, the permit limit for total nitrogen should be established as a rolling average seasonal limit over the May through October period.

CONNECTICUT WWTFs 2010 DATA MONTHLY AVERAGE TOTAL NITROGEN CONCENTRAION (mg/l)				
Town	**Process**	**Average** **12-month**	 **Apr. – Oct.**	**Max. Month Apr. – Oct.**
Branford	4-stage Bardenpho	3.4	3.1	4.7
Cheshire	Denite Filters	1.8	2.0	2.9
Jewett City	Phased Oxidation Ditch	2.3	2.1	3.0
Southington	Trickling Filter/Denite Filter	5.4	5.2	7.7
Suffield	MLE Oxidation Ditch	2.1	1.9	2.9
Stamford	4-Stage Bardenpho	3.5	2.8	3.2
New Canaan	MLE Oxidation Ditch	3.1	2.4	3.1
Milford Housatonic	4-Stage Bardenpho	4.7	4.4	5.1
Westport	4-Stage Bardenpho	2.6	2.1	2.6
Waterbury	4-Stage Bardenpho	4.1	3.7	5.4

* Reference **Attachment 1.B** for complete 2010 data.

3. Proposed Mass Limit Restricts the City's Ability to Expand Sewer Service

The proposed mass limit for total nitrogen effectively caps future plant flow rates to the current permitted flow of 8.4 mgd. Since the permit, as written, sets the total nitrogen concentration in the effluent at the limit of treatment technology, no further reduction in total nitrogen is possible and therefore no increase in flow is possible to prevent the mass limit from being exceeded. Given the lack of current data or analyses (see Attachment 2 for further information), it is nor reasonable or appropriate to impose the equivalent of a growth moratorium on the City.

In Section VI.A of the Fact Sheet, EPA acknowledges that in the Draft Environmental Impact Report (DEIR) for the Comprehensive Wastewater Management Plan, the City has identified 14 priority areas currently served by on-site wastewater disposal systems to which sewer system expansion has been proposed. Subsequent to the completion of the DEIR, the City has initiated planning to redevelop the Dever School property as an industrial park to enhance the City's economic base. Other future development opportunities are present in existing industrial zoned areas likely to contribute wastewater to the wastewater collection system. The proposed design flow rate to Taunton's wastewater treatment facility, in the DEIR, increases from 8.4 mgd to 10.2 mgd. This flow rate will be re-evaluated in the Final Environmental Impact Report.

Septic systems in general contribute a significant nitrogen load to the Taunton River watershed. By expanding the wastewater collection system to encompass the sewer needs areas, this will transfer treatment of wastewater to the WWTF and reduce the non-point nitrogen load to the River.

Establishing a mass total nitrogen limit in the discharge would effectively prohibit expansion of the wastewater collection and treatment system beyond its present design capacity. Anti-degradation provisions in the clean water act could restrict future expansion of the wastewater treatment facility. Therefore, the mass limit should be removed from the permit.

4. **Allowable Total Nitrogen Load**

Section VI.B.f.ii of the Fact Sheet develops an allowable total nitrogen load from the watershed, and more specifically point sources that would result in a concentration at or below the 0.45 mg/l threshold that was derived in other sections of the fact sheet. That validity of that threshold is questioned in other comments offered by the City but is used here for illustrative purposes.

The analysis performed by USEPA in the Fact Sheet relies on sampling performed by SMAST as part of the Mount Hope Bay Estuarine Monitoring Program, during the months of June, July and August of 2004 through 2006. Under that program, samples were collected on two occasions from 22 sampling stations each month for a total of 18 sampling events. In USEPA's analysis of allowable total nitrogen loading, data from 2006 was not used due to significant wet weather events that occurred in June. Although flows in the Taunton, Three Mile and Segreganset Rivers were elevated during that month, the 3-year average flow for June through August is more indicative of historic flows over the entire 6-month seasonal permitting period of May through October. The analysis should not be limited to selected low flow periods only.

Assuming EPA's approach is valid, we have recalculated the allowable total nitrogen load following the procedures established by USEPA and incorporating the 2006 monitoring data. The calculation is provided in **Attachment 1.C** for consideration and a brief summary of the results is provided as follows:

- The average total freshwater flow was 881 cfs
- Ocean flow was determined as 1,458 cfs based on an average salinity of 18.7 ppt.
- Based on a target TN concentration of 0.45 mg/l, the targeted nitrogen load was 5,672 pounds per day (ppd)
- The allowable load from watershed sources was determined as 3,472 ppd
- The required nitrogen load reduction was 756 ppd
- Based on a 20-percent reduction in nitrogen from non-point sources, the available nitrogen load from wastewater discharges was 2,187 ppd.
- Applying a uniform nitrogen concentration to wastewater discharges, the allowable total nitrogen concentration is 8.8 mg/l.

Based on the above, establishing a total nitrogen limit of 8.0 mg/l for all identified wastewater treatment facilities discharging to the Taunton River is warranted.

5. **Use of year round CBOD analyses**

The City finds the permit language pertaining to $CBOD_5$ analyses and nitrogen removal requirements to be contradictory and could put the City at risk for unwarranted violations.

The permit utilizes $CBOD_5$ as the measure of oxygen demand due to high nitrogenous oxygen demand in the effluent during the summer nitrifying season, as allowed under 40 CFR 103.102(a)(4). Page 9 of the Fact Sheet states: "The use of CBOD instead of BOD is not necessary in the colder season as the facility discontinues the nitrifying process, making the use of CBOD tests unnecessary. The City disagrees with this general premise. The fact that the facility is not fully nitrifying does not mean that such organisms are not present in the effluent in sufficient numbers to provide a misleading BOD reading. In addition, the City finds Footnote 12 on Page 6 contradictory as it requires the City to operate the treatment facility to reduce the discharge of total nitrogen during the months of November through April to the maximum extent possible even though there are no permit limitations for ammonia or total nitrogen during this period. If some degree of total nitrogen removal must be attempted in the colder season, the use of year round CBOD analyses would be necessary and appropriate to minimize the impacts from nitrogenous oxygen demand.

The statement in the Fact Sheet indicates that the nitrification process can be ceased from November through April. Therefore, Footnote 12 should be deleted in its entirety.

In the event that Footnote 12, takes precedent over the Fact Sheet in regard to the need to remove nitrogen from November through April, the City takes exception to the following statement:

> *"All available treatment equipment in place at the facility shall be operated unless equal or better performance can be achieved in a reduced operational mode"*

This sentence appears to give EPA and MassDEP the authority to dictate to the City means and methods of complying with its NPDES permit or to dictate more restrictive operation even when unnecessary to meet applicable standards. Neither EPA nor MassDEP have such authority. We do not want to be subject to a violation in an instance where a regulator demands a particular piece of equipment be activated even though it does not improve the quality of the discharge, particularly in a situation where there is no established numerical standard.

The City retains licensed and experienced wastewater operators who will make the determination as to what equipment must be operated to meet permit conditions. To illustrate this point, the provision allowing discontinued use of a supplemental carbon source from November through April may warrant that some equipment such as denitrification filters, be removed from service as they would provide little, if any, water quality benefit. Removing the filters from service would result in significant energy savings and reduce the carbon footprint of the WWTF during this period. The subject permit statement appears to give EPA and MassDEP the authority to challenge this prudent and viable decision and impose a permit violation where none is warranted.

The first sentence in Footnote 12 requiring the facility to be operated to reduce the discharge of total nitrogen to the maximum extent possible during this period is sufficient.

6. Inconsistent pH Limitations

Section VI.B.3 of the Fact Sheet states that: "MassDEP has stated that a permitted pH range of 6.0 to 8.5 SU is protective of State water quality standards, and this range has been included in the draft permit". This range is more restrictive than the range of 6.0 to 9.0 set forth in 40 CFR 133.102(c). However, the allowable range for pH in the Taunton WWTF discharge, as written in the permit, is 6.0 to 8.3 SU. There, does not appear to be any valid reason for the upper limit for pH being set at 8.3 SU instead of 8.5 SU.

7. **7Q10 River Flow**

Based on a review of the sections pertaining to the 7Q10 established in the Draft NPDES Permit (MA0100897) for the Taunton Wastewater Treatment Facility that was issued on March 20, 2013, the following comments were generated:

In the 2001 NPDES Permit Reissuance, the 7Q10 flow was defined as 30.4 cfs at Station No. 01108000, Taunton River near Bridgewater gauge and 41.85 cfs at the point of discharge. In the present draft NPDES permit, the 7Q10 flow has been revised downward by EPA to 22.9 cfs at the gauge and 31.6 cfs at the point of discharge using EPA's in-house DFLOW analysis of USGS stream flow data for, for the years 1931 through 2002.

It is difficult to understand why the 7Q10 in the Taunton River at the Bridgewater gauge would drop by nearly 25-percent from one used in a permit issued in 2001 and a calculation performed on data through 2002. A review of daily flow data at gauging station 01108000 for the years 2003 through 2012 shows that the lowest 7 day flow during this 10-year period was 47 cfs, which occurred twice; once in August 2005 and again in September 2007. Therefore, we request that the 7Q10 flow be re-evaluated through 2012, as inclusion of the recent flow data will likely alter the statistical analysis.

In fact, a printout from DFLOW provided by USEPA that was done after the 2007 draft permit was issued (using flow data from 1931 through 2008 rather than 2002) indicates that the 7Q10 is 23.7 CFS. This value is slightly higher than that used in the draft permit, although it is still much lower than the value used in the 2001 final permit. It does however provide justification that flow data through 2012 should be used in the evaluation.

The 7Q10 flow directly impacts the dilution factor at the discharge of the WWTF, which in turn impacts the allowable copper and chlorine residual concentrations established by the permit. EPA correctly reclassified the Taunton River at the point of discharge as a salt water body, immediately places more restrictive limits on total copper. Lowering the dilution factor places further restrictions on the discharge. These stringent standards, if enforced as they are, will require Taunton to treat its wastewater for copper. This does not appear to be justified, as Taunton's wastewater discharge has been in compliance with whole effluent toxicity testing.

8. **Schedule in ACO not Permit**

The Compliance Schedule included in the Draft permit is too restrictive and does not take into account the existing Administrative Order that the City of Taunton has with the EPA, Administrative Order Docket No. 08-042. The City of Taunton has applied for State Funding through the Clean Water State Revolving Fund and is listed on the Intended Use Plan for $15 million for three more projects. It is at the end of these projects that we believe the City will have completed elimination of all known cross connections between the sewer system and the storm drain system and removed sources of infiltration and inflow that are cost-effective. In addition to Sewer Separation and Infiltration/Inflow removal projects, the City is scheduled to complete its Comprehensive Wastewater Management Plan (CWMP) and Final Environmental Impact Report (EIR). As part of the CWMP and final EIR pilot testing will be required for determination of the most cost-effective and reliable means of achieving nitrogen reduction. Therefore, we are requesting that the compliance schedule be removed from the permit and negotiated through a separate Administrative Consent Order. The negotiated schedule must be more realistic in its duration and consider the long term economic needs of the City. The City believes that deferral of major Total Nitrogen reduction should occur until we know what improvements are necessary under current conditions. The City cannot afford to spend resources on multiple plant improvements as occurred in Upper Blackstone or to extend all of its resources on a "limit of technology facility" only to find that such a treatment requirement was not actually needed.

9. Economic Impact

The City has spent a significant amount of money related to wastewater utility improvements since the WWTF was upgraded in 2000. As a result of past projects and the existing CMOM Program, the average sewer rate for FY2014 is estimated to be $516. We are concerned that further large expenditures, as would be required to again upgrade the WWTF to meet limit of technology nitrogen limits, will bear a great financial burden on our users.

The City has several Environmental Justice (EJ) areas in various census tracts within its sewer district boundary (refer to **Attachment 1.D**). We are duly concerned that rising sewer rates will adversely affect these populations. The EJ population actually makes up about 35 percent of the total sewered population. The median household incomes in the various EJ areas range from $21,440 to $39,632.

As stated in EPA's *Interim Economic Guidance for Water Quality Standards*: "if the average annual cost per household (sewer rate) exceeds 2.0 percent of median household income, then the project may place an unreasonable financial burden on many households within the community" Based on the estimated sewer rate for FY2104 and applying EPA's screening criteria of 2 percent results in a median household income of $25,800 below which there would an unreasonable financial burden.

The table provided below identifies future wastewater related projects that need to be completed in Taunton. These projects include those required to complete the sewer separation and infiltration/inflow reduction program, to generally improve the collection system, and upgrade the WWTF for nitrogen removal. As a result of these projects, the annual sewer rate is expected to increase to more than $1,000. Based on an annual sewer rate of $1,000 all households with a median income of less than $50,000 would be adversely affected, which represents about 50% of the sewered households.

The City is requesting relief from the schedule so we can properly plan the required work and protect the economic viability of the City and the sewered population. The City is also requesting another analysis with more recent water quality data before upgrading the WWTP to achieve Technology Based Limits for nitrogen reduction. Pursuant to 40 CFR 131.01(g), we request EPA's determination on whether the current cost impact of EPA's "limit of technology" standard may be considered "substantial and widespread economic impact" , which would allow deferral of the high cost total nitrogen reduction measures or the approval of a variance by MassDEP.

Future Wastewater Related Design and Construction Projects		
Project	**Timeframe**	**Opinion of Project Cost**
Phase 10 SSES	By 2016	$5,500,000
Phase 11 SSES	By 2018	$5,500,000
Phase 12 SSES	By 2018	$5,500,000
New Main Lift Pump Station	By 2018	$11,500,000
CSO Mitigation Facility	------	$9,000,000
Wastewater Treatment Facility Improvements	------	$45,000,000
Total Project Costs		**$82,000,000**
Anticipated User Fee Increase Due to Debt Service [1]		**$495**

1. User rate increases by $6 per $1,000,000 of expenditure. Does not include increases in operations and maintenance costs associated with nitrogen removal. All costs to be redefined during the preparation of the Final CWMP and Environmental Impact Report.

10. Ambiguity in the Reporting of Unauthorized Discharges

The permit identifies the towns of Dighton and Raynham as co-permittees "for specific activities required in Sections I.B – Unauthorized Discharges and I.C – Operations and Maintenance of the Sewer System, which include conditions regarding the operation and maintenance of the collection system owned and operated by the Towns". Comments on the draft permit submitted on April 18, 2013 by the Upper Blackstone Water Pollution Abatement District (UBWPAD) specifically question the legal basis through which the EPA has authority to regulate Towns as co-permittees. The City of Taunton concurs with the comments issued by the UBWPAD (refer to **Attachment 1.E**) and they are included herein as Taunton's comments also.

EPA Region 1 does not possess legal authority to add or amend the existing NPDES rules *(Pennsylvania Mun. Authorities Ass'n v. Horinko, 292 F.Supp.2d 95 (D.D.C. 2003))*. EPA has never adopted the co-permittee requirements that the Region is seeking to impose. That such requirements may have been imposed on others is not relevant to their legality. Therefore, we request that the co-permittee provisions be stricken from this permit as arbitrary and capricious and otherwise not in accordance with law.

In addition, Section I.B of the permit states that "Discharges of wastewater from any other point source, including sanitary sewer overflows (SSOs), are not authorized by this permit and must be reported to EPA and MassDEP orally within 24-hours of the time the permittee becomes aware of the circumstances and a written submission shall also be provided within 5 days of the time the permittee becomes aware of the circumstances". The City of Taunton, who is designated as the permittee, in no way has control over the operation of wastewater collection systems in satellite communities and is not responsible for its functionality. Accordingly, the permittee (City of Taunton) will not be responsible for reporting SSOs that occur outside its municipal boundary and legal jurisdiction.

Taunton's inter-municipal agreements with contributing communities only regulate the quantity and character of the wastewater that enters the Taunton collection system to ensure that the integrity and performance of its wastewater infrastructure are protected. Taunton assumes no further responsibility.

11. **Wet Weather Limits**

    Taunton is requesting that consideration be given to providing a higher concentration limit during wet weather events. Maximizing wet weather flow treatment and simultaneously minimizing effluent nitrogen loads can be competing goals and provisions should be made in the permit to acknowledge different limits during wet weather events. Although the final plan to reduce the frequency and volume discharged from the West Water Street CSO, it is likely that more wastewater/stormwater will be directed to the WWTF during significant wet weather events. USEPA Region I has acknowledged this issue and issued "two tiered" permit limits to account for wet weather events in many locations including, New Haven, CT, Bangor ME, and Boston MA. New York City, in Region II, has similar accommodations for wet weather in their permits, as does Ohio, in Region V.

    40 CFR 122.44(d) and CWA Section 301(b)(1)(C) only require more restrictive limitations as “necessary to attain water quality standards…”. The permits various water quality-based limits are not necessary under high flow conditions as the wastewater facility has basically no meaningful impact on ambient water quality when such flows occur. Therefore, the discharge should not have to meet the more stringent limitations under these conditions – only technology-based requirements should apply (e.g., secondary treatment). The permit should be modified to specify that continued operation of all facilities is required under these conditions but the more restrictive water quality-based limits are suspended under these conditions.

12. **Comments from Hall and Associates**

    Attachment 2, prepared by Hall & Associates, provides further comments on the reasonableness of the proposed nitrogen and copper limitations. Based on those comments the City requests that both limitations by stricken from this permit. At a minimum, the present need for nitrogen limitations must be based on an assessment that fully accounts for effluent reduction requirements presently enacted or anticipated in this watershed and the watersheds affecting Mount Hope Bay. These include actions affecting CSO, organic loadings and nutrient loadings that all affect the dissolved oxygen regime. Moreover, a rational connection between nutrient levels, algal growth and dissolved oxygen conditions must be developed (at least for the Taunton River) to allow for the identification of actions that will ensure minimum dissolved oxygen compliance. Lastly, it is apparent that the dissolved oxygen water quality criterion for the estuary is out of date and inconsistent with those adopted for Narragansett Bay. It would seem most reasonable to ensure that the updated standards are adopted and to reassess the need for total nitrogen reduction given the best available science, using current standards.

recreational areas, and undeveloped tracts of land. Impervious lands include roofs, parking lots and streets. Stormwater collects fertilizers and other applied nutrients, as well as other pollutants on impervious surfaces, before it is discharged to receiving waters. As noted in the EPA SAB report *Urban Stormwater Management in the United States* (NRC 2008b):

> *Urban stormwater may actually have slightly lower pollutant concentrations than other nonpoint sources of pollution, especially for sediments and nutrients. The key difference is that urban watersheds produce a much larger annual volume of runoff waters, such that the mass of pollutants discharged is often greater following urbanization.*

Urban stormwater discharges via municipal separate storm sewer systems (MS4s) and combined storm sewer systems (CSSs) are regulated under the National Pollutant Discharge Elimination System (NPDES) permit program of the CWA. There are several thresholds for MS4 stormwater regulations. However, a significant number of communities and a substantial amount of urban growth occur outside of MS4s and are only subject to construction stormwater general permits.

Municipal Wastewater Treatment

Municipal wastewater treatment plants, also known as publicly owned treatment works (POTWs), usually discharge both phosphorus and nitrogen. Depending on the local ecological conditions and their relative contribution, POTW discharges can be a significant source of nutrients in some watersheds. People produce about 18 million tons of solid waste (feces) annually (based on Freitas Jr. 1999; MERCK 2007). U.S. municipal wastewater treatment facilities currently treat about 34 billion gallons of wastewater per day (USEPA 2008c).

For most of the country, municipal wastewater treatment generates two waste streams—biosolids and discharges of treated wastewater to surface water—which are regulated under the provisions of sections 301, 402, and 405 of the CWA, respectively. Municipal or sewage waste biosolids that are to be land applied must meet specific CWA and state regulatory standards to protect surface water and groundwater from contamination. Treatment for surface water discharges is regulated through NPDES permits, which must reflect both the technology-based requirements of secondary treatment (biological oxygen demand (BOD), total suspended solids (TSS), and pH) and applicable water quality standards. However, only a subset of POTW permits currently contain nitrogen and phosphorus limits. Of more than 16,500 municipal POTWs nationwide (USEPA 2008c), approximately 4 percent have numeric limits for nitrogen[2] and 9.9 percent for phosphorus (USEPA 2009e). Estimated costs for municipal nutrient removal can vary widely depending on level of treatment and process used, wastewater characteristics, plant capacity, existing treatment facilities, and other site-specific factors.

The estimated cost to upgrade all the POTWs in the United States to achieve the more stringent technology-based limits—3 mg/L for nitrate and nitrite and 0.1 mg/L for phosphate—would be about $44 billion to remove nitrogen, about $44.5 billion to remove phosphorus, and approximately $54 billion to include capabilities to simultaneously remove both nitrogen and phosphorus (based on USEPA 2008c). In addition, our growing population will result in

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[2] Although 43.5 percent of POTW permits have limits for ammonia, limiting ammonia generally does not reduce overall nitrogen loadings because nitrates and nitrites continue to be discharged.

Attachment 1B - 2010 Discharge Monitoring Data for Several Connecticut WWTFs

Plant	Date	Permit	FlowMGD	FTKN	NO2NO3	TN	TNMASS	TN Monthly Average
BRANFORD WPCF	01/06/2010	CT0100048	3.8	6.82	0.9	7.7	244	
BRANFORD WPCF	01/13/2010	CT0100048	3.4	3.01	1.17	4.2	119	
BRANFORD WPCF	01/20/2010	CT0100048	3.6	8.25	0.14	8.4	252	
BRANFORD WPCF	01/27/2010	CT0100048	[illegible]	4.26	1.83	[illegible]	204	6.6
BRANFORD WPCF	02/03/2010	CT0100048	3.5	2.61	1.49	4.1	120	
BRANFORD WPCF	02/10/2010	CT0100048	3.3	3.2	1.54	4.7	129	
BRANFORD WPCF	02/17/2010	CT0100048	3.3	3.23	2.04	5.3	146	
BRANFORD WPCF	02/24/2010	CT0100048	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]
BRANFORD WPCF	03/03/2010	CT0100048	5	2.34	0.9	3.2	133	
BRANFORD WPCF	03/10/2010	CT0100048	4	0.8	0.79	1.6	53	
BRANFORD WPCF	03/17/2010	CT0100048	5.9	0.95	0.97	1.9	94	
BRANFORD WPCF	03/24/2010	CT0100048	7	1.59	1.08	2.7	158	
BRANFORD WPCF	03/31/2010	CT0100048	11.2	8.4	[illegible]	10.3	962	[illegible]
BRANFORD WPCF	04/07/2010	CT0100048	5	0.51	0.74	1.3	54	
BRANFORD WPCF	04/14/2010	CT0100048	4	2.2	1.15	3.4	113	
BRANFORD WPCF	[illegible]	CT0100048	[illegible]	2.7	1.09	[illegible]	[illegible]	[illegible]
BRANFORD WPCF	05/07/2010	CT0100048	3.6	6.4	1.34	7.7	231	
BRANFORD WPCF	05/12/2010	CT0100048	3.4	1.08	1.77	2.9	82	
BRANFORD WPCF	05/19/2010	CT0100048	4.1	1.08	0.97	2.1	72	
BRANFORD WPCF	05/26/2010	CT0100048	[illegible]	3.87	0.31	4.2	[illegible]	4.2
BRANFORD WPCF	06/02/2010	CT0100048	3.3	2.76	1.31	4.1	113	
BRANFORD WPCF	06/09/2010	CT0100048	3.3	1.54	1.97	3.5	96	
BRANFORD WPCF	06/16/2010	CT0100048	3.2	2.6	0.74	3.3	88	
BRANFORD WPCF	06/23/2010	CT0100048	3.3	2.6	0.87	3.5	96	
BRANFORD WPCF	06/30/2010	CT0100048	3.1	2.2	0.64	[illegible]	67	[illegible]
BRANFORD WPCF	07/07/2010	CT0100048	3.1	1.81	2.08	3.9	101	
BRANFORD WPCF	07/14/2010	CT0100048	3.7	2.2	1.07	3.3	102	
BRANFORD WPCF	07/21/2010	CT0100048	3.2	2.3	0.67	3	80	
BRANFORD WPCF	07/28/2010	CT0100048	3.2	1.52	0.76	2.3	61	[illegible]
BRANFORD WPCF	08/04/2010	CT0100048	3	2.1	0.65	2.8	70	
BRANFORD WPCF	08/11/2010	CT0100048	3.1	0.54	0.79	1.3	34	
BRANFORD WPCF	08/18/2010	CT0100048	3	0.92	0.83	1.8	45	
BRANFORD WPCF	08/25/2010	CT0100048	3.2	[illegible]	0.29	1.5	40	1.9
BRANFORD WPCF	09/01/2010	CT0100048	3	0.92	0.96	1.9	48	
BRANFORD WPCF	09/08/2010	CT0100048	2.9	1.08	0.86	1.9	46	
BRANFORD WPCF	09/15/2010	CT0100048	2.8	1.17	1.44	2.6	61	
BRANFORD WPCF	09/22/2010	CT0100048	2.8	0.84	1.74	2.6	61	
BRANFORD WPCF	09/29/2010	CT0100048	2.8	0.99	1.65	2.6	61	2.3
BRANFORD WPCF	10/06/2010	CT0100048	3	0.93	1.87	2.8	70	
BRANFORD WPCF	10/13/2010	CT0100048	2.8	0.57	0.75	1.3	30	
BRANFORD WPCF	10/20/2010	CT0100048	2.8	0.5	0.92	1.4	33	
BRANFORD WPCF	10/27/2010	CT0100048	2.9	0.92	0.92	[illegible]	44	1.8
BRANFORD WPCF	11/03/2010	CT0100048	2.7	0.27	0.63	0.9	20	
BRANFORD WPCF	11/10/2010	CT0100048	3	0.49	0.61	1.1	28	
BRANFORD WPCF	11/17/2010	CT0100048	3.5	0.92	1.28	2.2	64	
BRANFORD WPCF	11/23/2010	CT0100048	2.8	[illegible]	1.34	2.2	[illegible]	1.6
BRANFORD WPCF	12/01/2010	CT0100048	3.3	0.57	0.57	1.1	30	
BRANFORD WPCF	12/08/2010	CT0100048	2.8	0.46	0.98	1.4	33	
BRANFORD WPCF	12/15/2010	CT0100048	3.2	0.25	0.72	1	27	
BRANFORD WPCF	12/21/2010	CT0100048	3.1	0.76	0.9	1.7	44	
BRANFORD WPCF	12/28/2010	CT0100048	[illegible]	1.34	1.21	2.6	65	1.6
					Average	3.1745	110.745098	3.4
					April - October	2.8	77.06896552	3.1
					Max	10.3	962	6.6
					Min	0.9	20	1.6
					Max (Apr-Oct)	7.7	231	4.7
					Min (Apr-Oct)	1.3	30	1.8

Plant	Date	Permit	FlowMGD	FTKN	NO2NO3	TN	TNMASS	TN Monthly Average
CHESHIRE WPCF	01/14/2010	CT0100081	2.8	1.2	1.3	2.5	58	
CHESHIRE WPCF	01/21/2010	CT0100081	2.7	0.9	0.4	1.3	29	
CHESHIRE WPCF	01/28/2010	CT0100081	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]
CHESHIRE WPCF	02/04/2010	CT0100081	3.2	1.3	0.57	1.9	51	
CHESHIRE WPCF	02/11/2010	CT0100081	2.8	1.2	0.29	1.5	35	
CHESHIRE WPCF	02/18/2010	CT0100081	2.5	1.4	0.35	1.8	38	
CHESHIRE WPCF	02/25/2010	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	45	[illegible]
CHESHIRE WPCF	03/04/2010	CT0100081	4.6	0.71	0.29	1	38	
CHESHIRE WPCF	03/11/2010	CT0100081	3.3	0.49	0.26	0.8	22	
CHESHIRE WPCF	03/18/2010	CT0100081	5.1	1.3	0.2	1.5	64	
CHESHIRE WPCF	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	129	[illegible]
CHESHIRE WPCF	04/01/2010	CT0100081	6.1	1	1.8	2.8	142	
CHESHIRE WPCF	04/08/2010	CT0100081	4.3	0.6	0.27	0.9	32	
CHESHIRE WPCF	04/15/2010	CT0100081	2.8	0.6	0.4	1	23	
CHESHIRE WPCF	04/22/2010	CT0100081	2.2	1.3	1.1	2.4	44	
CHESHIRE WPCF	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]
CHESHIRE WPCF	05/06/2010	CT0100081	1.8	0.82	0.11	0.9	14	
CHESHIRE WPCF	05/13/2010	CT0100081	1.8	1	0.1	1.1	17	
CHESHIRE WPCF	05/20/2010	CT0100081	1.8	1.6	2.7	4.3	65	
[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]
CHESHIRE WPCF	06/03/2010	CT0100081	1.5	1.3	0.18	1.5	19	
CHESHIRE WPCF	06/10/2010	CT0100081	2	1.2	0.28	1.5	25	
CHESHIRE WPCF	06/17/2010	CT0100081	2	0.8	0.18	1	17	
CHESHIRE WPCF	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]
CHESHIRE WPCF	07/01/2010	ct0100081	1.9	3.1	0.9	4	63	
CHESHIRE WPCF	07/08/2010	ct0100081	1.8	1.3	0.9	2.2	33	
CHESHIRE WPCF	07/15/2010	ct0100081	1.9	2.4	0.15	2.6	41	
CHESHIRE WPCF	07/22/2010	ct0100081	1.9	2.6	0.53	3.1	49	
[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]
CHESHIRE WPCF	08/05/2010	CT0100081	1.7	1.2	0.13	1.3	18	
CHESHIRE WPCF	08/12/2010	CT0100081	1.8	1.5	0.15	1.7	26	
CHESHIRE WPCF	08/19/2010	CT0100081	1.8	1.4	0.52	1.9	29	
[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]
CHESHIRE WPCF	09/02/2010	CT0100081	1.9	0.6	0.52	1.1	17	
CHESHIRE WPCF	09/09/2010	CT0100081	1.8	1.1	0.7	1.8	27	
CHESHIRE WPCF	09/16/2010	CT0100081	1.9	1.3	0.65	2	32	
CHESHIRE WPCF	09/23/2010	CT0100081	1.7	1.3	0.22	1.5	21	
[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]
CHESHIRE WPCF	10/07/2010	CT0100081	2	2	2.3	4.3	72	
CHESHIRE WPCF	10/14/2010	CT0100081	2	1.4	1.3	2.7	45	
CHESHIRE WPCF	10/21/2010	CT0100081	2.1	1.2	0.53	1.7	30	
[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]
CHESHIRE WPCF	11/04/2010	ct0100081	2	1.3	1.73	3	50	
CHESHIRE WPCF	11/12/2010	ct0100081	2.1	1.4	0.49	1.9	33	
CHESHIRE WPCF	11/18/2010	ct0100081	2.7	1.9	0.34	2.2	50	
[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]
CHESHIRE WPCF	12/02/2010	CT0100081	2.7	1	0.46	1.5	34	
CHESHIRE WPCF	12/09/2010	CT0100081	2.5	0.9	0.61	1.5	31	
CHESHIRE WPCF	12/16/2010	CT0100081	4.4	0.7	0.39	1.1	40	
CHESHIRE WPCF	12/23/2010	CT0100081	3.1	0.9	0.73	1.6	41	
[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]
					Average	1.8569	38.8627451	**1.8**
					April - October	1.9871	35.29032258	**2.0**
					Max	4.3	142	**2.9**
					Min	0.8	14	**1.4**
					Max (Apr-Oct)	4.3	142	**2.9**
					Min (Apr-Oct)	0.9	14	**1.5**

Plant	Date	Permit	FlowMGD	FTKN	NO2NO3	TN	TNMASS	TN Monthly Average
JEWETT CITY WPCF	01/18/2010	CT0100269	0.32	1.3	1.31	2.6	7	
JEWETT CITY WPCF	01/22/2010	CT0100269	0.32	0.81	0.49	1.3	3	
JEWETT CITY WPCF	01/29/2010	CT0100269	0.36	0.91	0.43	1.3	4	1.7
JEWETT CITY WPCF	02/05/2010	CT0100269	0.29	0.5	1.11	1.6	4	
JEWETT CITY WPCF	02/12/2010	CT0100269	0.3	1.8	0.68	2.5	6	
JEWETT CITY WPCF	02/19/2010	CT0100269	0.29	2.4	0.56	3	7	
JEWETT CITY WPCF	02/26/2010	CT0100269	0.62	2	0.23	2.2	11	2.3
JEWETT CITY WPCF	03/05/2010	CT0100269	0.39	2	0.42	2.4	8	
JEWETT CITY WPCF	03/12/2010	CT0100269	0.34	1.8	0.28	2.1	6	
JEWETT CITY WPCF	03/19/2010	CT0100269	0.47	1.3	0.18	1.5	6	
JEWETT CITY WPCF	03/26/2010	CT0100269	0.56	1.5	0.28	1.8	9	2.0
JEWETT CITY WPCF	04/09/2010	CT0100269	0.62	1.1	0.27	1.4	7	
JEWETT CITY WPCF	04/16/2010	CT0100269	0.47	0.85	0.36	1.2	5	
JEWETT CITY WPCF	04/23/2010	CT0100269	0.37	0.98	0.64	1.6	5	
JEWETT CITY WPCF	04/30/2010	CT0100269	0.32	0.67	2.31	3	8	1.8
JEWETT CITY WPCF	05/07/2010	CT0100269	0.31	0.7	0.86	1.6	4	
JEWETT CITY WPCF	05/14/2010	CT0100269	0.31	1.1	2.21	3.3	9	
JEWETT CITY WPCF	05/21/2010	CT0100269	0.3	1.7	0.74	2.4	6	
JEWETT CITY WPCF	05/28/2010	CT0100269	0.28	1.4	0.48	1.9	4	2.3
JEWETT CITY WPCF	06/04/2010	CT0100269	0.26	1.3	1.03	2.3	5	
JEWETT CITY WPCF	06/11/2010	CT0100269	0.28	1.4	2.43	3.8	9	
JEWETT CITY WPCF	06/18/2010	CT0100269	0.27	2.1	0.77	2.9	7	
JEWETT CITY WPCF	06/25/2010	CT0100269	0.27	2.4	0.57	3	7	3.0
JEWETT CITY WPCF	07/02/2010	CT0100269	0.28	3.4	0.63	4	9	
JEWETT CITY WPCF	07/09/2010	CT0100269	0.28	1.9	0.52	2.4	6	
JEWETT CITY WPCF	07/16/2010	CT0100269	0.29	0.65	0.49	1.1	3	
JEWETT CITY WPCF	07/23/2010	CT0100269	0.25	0.5	1.23	1.7	4	
JEWETT CITY WPCF	07/30/2010	CT0100269	0.26	1.2	0.54	1.7	4	2.2
JEWETT CITY WPCF	08/06/2010	CT0100269	0.23	1.2	0.61	1.8	3	
JEWETT CITY WPCF	08/13/2010	CT0100269	0.25	0.9	0.59	1.5	3	
JEWETT CITY WPCF	08/20/2010	CT0100269	0.23	1.1	0.58	1.7	3	
JEWETT CITY WPCF	08/27/2010	CT0100269	0.23	1	0.48	1.5	3	1.6
JEWETT CITY WPCF	09/03/2010	CT0100269	0.31	1	0.73	1.7	4	
JEWETT CITY WPCF	09/10/2010	CT0100269	0.25	0.85	0.57	1.4	3	
JEWETT CITY WPCF	09/17/2010	CT0100269	0.25	0.82	0.71	1.5	3	
JEWETT CITY WPCF	09/24/2010	CT0100269	0.25	0.8	1.1	1.9	4	1.6
JEWETT CITY WPCF	10/01/2010	CT0100269	0.31	0.96	0.81	1.8	5	
JEWETT CITY WPCF	10/08/2010	CT0100269	0.23	0.78	1.21	2	4	
JEWETT CITY WPCF	10/15/2010	CT0100269	0.24	0.68	1.01	1.7	3	
JEWETT CITY WPCF	10/22/2010	CT0100269	0.27	0.85	1.41	2.3	5	
JEWETT CITY WPCF	10/29/2010	CT0100269	0.27	0.83	0.66	1.5	3	1.9
JEWETT CITY WPCF	11/05/2010	CT0100269	0.25	1.2	1.71	2.9	6	
JEWETT CITY WPCF	11/12/2010	CT0100269	0.25	1.2	1.31	2.5	5	
JEWETT CITY WPCF	11/19/2010	CT0100269	0.23	1.5	0.67	2.2	4	
JEWETT CITY WPCF	11/23/2010	CT0100269	0.25	1.4	1.01	2.4	5	2.5
JEWETT CITY WPCF	12/03/2010	CT0100269	0.24	1.8	1.31	3.1	6	
JEWETT CITY WPCF	12/10/2010	CT0100269	0.22	1.7	2.61	4.3	8	
JEWETT CITY WPCF	12/17/2010	CT0100269	0.24	1.8	1.91	3.7	7	
JEWETT CITY WPCF	12/21/2010	CT0100269	0.28	1.9	3.41	5.3	12	
JEWETT CITY WPCF	12/28/2010	CT0100269	0.25	2.4	2.15	4.6	10	4.2
					Average	2.298	5.64	2.3
					April - October	2.0533	4.933333333	2.1
					Max	5.3	12	4.2
					Min	1.1	3	1.6
					Max (Apr-Oct)	4	9	3.0
					Min (Apr-Oct)	1.1	3	1.6

Plant	Date	Permit	FlowMGD	FTKN	NO2NO3	TN	TNMASS	TN Monthly Average
SOUTHINGTON WPCF	01/13/2010	CT0100536	4.7	6.4	0.8	7.2	282	
SOUTHINGTON WPCF	01/20/2010	CT0100536	4.6	4	3.7	7.7	295	
SOUTHINGTON WPCF	01/27/2010	CT0100536	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]
SOUTHINGTON WPCF	02/03/2010	CT0100536	5	3.9	1.6	5.5	229	
SOUTHINGTON WPCF	02/10/2010	CT0100536	4.8	5.6	3	8.6	344	
SOUTHINGTON WPCF	02/17/2010	CT0100536	4.6	3.1	1	4.1	157	
SOUTHINGTON WPCF	02/24/2010	CT0100536	[illegible]	[illegible]	[illegible]	[illegible]	424	6.2
SOUTHINGTON WPCF	03/03/2010	CT0100536	6.2	1.5	4.6	6.1	315	
SOUTHINGTON WPCF	03/10/2010	CT0100536	5.5	3.2	2.6	5.8	266	
SOUTHINGTON WPCF	03/17/2010	CT0100536	7.6	2.1	2.3	4.4	279	
SOUTHINGTON WPCF	03/24/2010	CT0100536	8.1	2.3	5.4	7.7	520	
SOUTHINGTON WPCF	03/30/2010	CT0100536	11.1	1.8	3.1	4.9	454	5.8
SOUTHINGTON WPCF	04/07/2010	CT0100536	7.5	4.2	1.7	5.9	369	
SOUTHINGTON WPCF	04/14/2010	CT0100536	5.8	1.1	3.1	4.2	203	
SOUTHINGTON WPCF	04/21/2010	CT0100536	5.2	1.7	3	4.7	204	
SOUTHINGTON WPCF	04/28/2010	CT0100536	4.7	1.5	8.8	10.3	404	6.3
SOUTHINGTON WPCF	05/05/2010	CT0100536	4.5	1.6	2.1	3.7	139	
SOUTHINGTON WPCF	05/12/2010	CT0100536	4.4	1.5	3.8	5.3	194	
SOUTHINGTON WPCF	05/19/2010	CT0100536	4.6	2.5	2.7	5.2	200	
SOUTHINGTON WPCF	05/26/2010	CT0100536	4.8	2.2	3.6	5.8	232	5.0
SOUTHINGTON WPCF	06/02/2010	CT0100536	4.2	2.2	1.8	4	140	
SOUTHINGTON WPCF	06/09/2010	CT0100536	3.6	2	3.2	5.2	156	
SOUTHINGTON WPCF	06/16/2010	CT0100536	3.8	2.5	8.2	10.7	339	
SOUTHINGTON WPCF	06/23/2010	CT0100536	3.8	1.7	6.2	7.9	250	
SOUTHINGTON WPCF	06/29/2010	CT0100536	4.4	2	8.5	10.5	386	7.7
SOUTHINGTON WPCF	07/07/2010	CT0100536	3.1	2.6	2.7	5.3	137	
SOUTHINGTON WPCF	07/14/2010	CT0100536	3.4	2	1.9	3.9	111	
SOUTHINGTON WPCF	07/21/2010	CT0100536	3	1.8	0.2	2	50	
SOUTHINGTON WPCF	07/29/2010	CT0100536	2.8	1.5	1.1	2.6	61	3.5
SOUTHINGTON WPCF	08/04/2010	CT0100536	2.6	2	1.2	3.2	69	
SOUTHINGTON WPCF	08/11/2010	CT0100536	3.1	2	3.7	5.7	147	
SOUTHINGTON WPCF	08/18/2010	CT0100536	2.7	1.9	14.9	16.8	378	
SOUTHINGTON WPCF	08/25/2010	CT0100536	3	1.8	1.4	3.2	80	7.2
SOUTHINGTON WPCF	09/08/2010	CT0100536	3	1.3	1.1	2.4	60	
SOUTHINGTON WPCF	09/15/2010	CT0100536	2.9	2.7	2.1	4.8	116	
SOUTHINGTON WPCF	09/22/2010	CT0100536	2.7	2.1	1.1	3.2	72	
SOUTHINGTON WPCF	09/29/2010	CT0100536	2.8	2	1.2	3.2	76	3.4
SOUTHINGTON WPCF	10/06/2010	CT0100536	3.3	2.1	1.5	3.6	99	
SOUTHINGTON WPCF	10/13/2010	CT0100536	2.8	4.7	0.9	5.6	131	
SOUTHINGTON WPCF	10/20/2010	CT0100536	2.1	1.1	1.7	2.8	49	
SOUTHINGTON WPCF	10/27/2010	CT0100536	2.5	0.8	1.2	2	42	3.5
SOUTHINGTON WPCF	11/03/2010	CT0100536	3.2	4.9	1.8	6.7	179	
SOUTHINGTON WPCF	11/10/2010	CT0100536	3.1	2.1	2.6	4.7	122	
SOUTHINGTON WPCF	11/17/2010	CT0100536	4.1	2.5	1.8	4.3	147	
SOUTHINGTON WPCF	11/24/2010	CT0100536	3.4	1.6	1.2	2.8	79	
SOUTHINGTON WPCF	11/29/2010	CT0100536	2.9	5.5	1.5	7	169	5.1
SOUTHINGTON WPCF	12/08/2010	CT0100536	3.3	4.3	4	8.3	228	
SOUTHINGTON WPCF	12/15/2010	CT0100536	5	2.5	3.1	5.6	234	
SOUTHINGTON WPCF	12/22/2010	CT0100536	4.6	1.5	1.6	3.1	119	
SOUTHINGTON WPCF	12/27/2010	CT0100536	4.3	2.8	0.8	3.6	129	[illegible]
					Average	5.422	196.4	5.4
					April - October	5.3	165.6896552	5.2
					Max	16.8	520	7.7
					Min	2	42	3.4
					Max (Apr-Oct)	16.8	404	7.7
					Min (Apr-Oct)	2	42	3.4

Plant	Date	Permit	FlowMGD	FTKN	NO2NO3	TN	TNMASS	TN Monthly Average
SUFFIELD WPCF	01/13/2010	CT0100552	1.3	1.5	0.15	1.7	18	
SUFFIELD WPCF	01/20/2010	CT0100552	1.5	1.2	0.06	1.3	16	
SUFFIELD WPCF	01/27/2010	CT0100552	[illegible]	4.5	0.03	4.5	23	[illegible]
SUFFIELD WPCF	02/03/2010	CT0100552	1.4	1.4	0	1.4	16	
SUFFIELD WPCF	02/10/2010	CT0100552	1.3	2.7	1.1	3.8	41	
SUFFIELD WPCF	02/17/2010	CT0100552	1.2	2.3	4.11	6.4	64	
SUFFIELD WPCF	02/24/2010	CT0100552	2.1	5.6	1.06	6.7	117	4.6
SUFFIELD WPCF	03/03/2010	CT0100552	1.9	5.3	1.03	6.3	100	
SUFFIELD WPCF	03/10/2010	CT0100552	1.5	1.9	2.74	4.6	58	
SUFFIELD WPCF	03/17/2010	CT0100552	1.7	0.3	0	0.3	4	
SUFFIELD WPCF	03/24/2010	CT0100552	2.2	1.7	0.18	1.9	35	
SUFFIELD WPCF	03/31/2010	CT0100552	4.0	2.1	0	2.1	72	3.0
SUFFIELD WPCF	04/07/2010	CT0100552	1.8	0.9	0	0.9	14	
SUFFIELD WPCF	04/14/2010	CT0100552	1.5	3.2	0.2	3.4	43	
SUFFIELD WPCF	04/21/2010	CT0100552	1.4	3.7	0.13	3.8	44	
SUFFIELD WPCF	04/28/2010	CT0100552	1.3	3.4	0	3.4	37	2.9
SUFFIELD WPCF	05/05/2010	CT0100552	1.2	1.5	0	1.5	15	
SUFFIELD WPCF	05/12/2010	CT0100552	1.2	0.8	0.22	1	10	
SUFFIELD WPCF	05/19/2010	CT0100552	1.3	0.7	0	0.7	8	
SUFFIELD WPCF	05/26/2010	CT0100552	1.4	0.1	0	0.1	1	0.8
SUFFIELD WPCF	06/02/2010	CT0100552	1.1	1.2	0	1.2	11	
SUFFIELD WPCF	06/09/2010	CT0100552	1.2	0.6	0.14	0.7	7	
SUFFIELD WPCF	06/16/2010	CT0100552	1.2	0.7	0.07	0.8	8	
SUFFIELD WPCF	06/23/2010	CT0100552	1.1	0.98	0.05	1	9	
SUFFIELD WPCF	06/30/2010	CT0100552	1	0.80	0.07	0.9	8	0.9
SUFFIELD WPCF	07/07/2010	CT0100552	1	1.8	0.05	1.9	16	
SUFFIELD WPCF	07/14/2010	CT0100552	1.1	0.9	0	0.9	8	
SUFFIELD WPCF	07/21/2010	CT0100552	1	0.9	0.06	1	8	
SUFFIELD WPCF	07/28/2010	CT0100552	0.9	1.3	0.1	1.4	11	1.3
SUFFIELD WPCF	08/04/2010	CT0100552	1	0.6	0.32	0.9	8	
SUFFIELD WPCF	08/11/2010	CT0100552	0.9	0.7	0.41	1.1	8	
SUFFIELD WPCF	08/18/2010	CT0100552	0.9	0.8	0.95	1.8	14	
SUFFIELD WPCF	08/25/2010	CT0100552	1	1.2	0.49	1.7	14	1.4
SUFFIELD WPCF	09/01/2010	CT0100552	1	0.8	0.45	1.3	11	
SUFFIELD WPCF	09/08/2010	CT0100552	1	0.7	3.62	4.3	36	
SUFFIELD WPCF	09/15/2010	CT0100552	0.9	1	0.44	1.4	11	
SUFFIELD WPCF	09/22/2010	CT0100552	0.9	1	3.7	4.7	35	
SUFFIELD WPCF	09/29/2010	CT0100552	1	0.6	2.23	2.8	23	2.9
SUFFIELD WPCF	10/06/2010	CT0100552	1.4	0.6	2.5	3.1	36	
SUFFIELD WPCF	10/13/2010	CT0100552	1.1	0.5	4.72	5.2	48	
SUFFIELD WPCF	10/20/2010	CT0100552	1.1	0.8	2.33	3.1	28	
SUFFIELD WPCF	10/27/2010	CT0100552	1.1	0.2	0	0.2	2	2.9
SUFFIELD WPCF	11/03/2010	CT0100552	1	0	0.79	0.8	7	
SUFFIELD WPCF	11/10/2010	CT0100552	1.3	0	0.93	0.9	10	
SUFFIELD WPCF	11/17/2010	CT0100552	2	0.3	0.35	0.7	12	
SUFFIELD WPCF	11/24/2010	CT0100552	1.3	0.5	0.57	[illegible]	[illegible]	0.8
SUFFIELD WPCF	12/01/2010	CT0100552	1.6	0.5	1.44	1.9	25	
SUFFIELD WPCF	12/08/2010	CT0100552	1.2	0	1.78	1.8	18	
SUFFIELD WPCF	12/15/2010	CT0100552	1.8	0	0.64	0.6	9	
SUFFIELD WPCF	12/22/2010	CT0100552	1.3	0.5	2.14	2.6	28	
SUFFIELD WPCF	12/29/2010	CT0100552	1.2	0.9	3.29	4.2	42	2.2
					Average	2.1275	24.62745098	2.1
					April - October	1.8733	17.73333333	1.9
					Max	6.7	117	4.6
					Min	0.1	1	0.8
					Max (Apr-Oct)	5.2	48	2.9
					Min (Apr-Oct)	0.1	1	0.8

Plant	Date	Permit	FlowMGD	FTKN	NO2NO3	TN	TNMASS	TN Monthly Average
WATERBURY WPCF	01/04/2010	CT0100625	25.7	4.6	0.6	5.2	1115	
WATERBURY WPCF	01/05/2010	CT0100625	25.4	5.2	1.5	6.7	1419	
WATERBURY WPCF	01/10/2010	CT0100625	23	3.4	1.7	5.1	978	
WATERBURY WPCF	01/11/2010	CT0100625	22.2	2.9	0.7	3.6	667	
WATERBURY WPCF	01/12/2010	CT0100625	21.9	4.4	0.7	5.1	932	
WATERBURY WPCF	01/18/2010	CT0100625	23.6	3.5	1	4.5	886	
WATERBURY WPCF	01/19/2010	CT0100625	21.5	2.6	0.8	3.4	610	
WATERBURY WPCF	01/20/2010	CT0100625	21.2	2.9	1	3.9	690	
WATERBURY WPCF	01/24/2010	CT0100625	21.3	1.7	1.5	3.2	568	
WATERBURY WPCF	01/25/2010	CT0100625	35.1	20.1	2	22.1	6469	
WATERBURY WPCF	01/26/2010	CT0100625	33.6	2	4.1	6.1	1709	
WATERBURY WPCF	01/31/2010	CT0100625	[illegible]	[illegible]	1.5	3.5	[illegible]	6.0
WATERBURY WPCF	02/01/2010	CT0100625	26	1.7	1.2	2.9	629	
WATERBURY WPCF	02/02/2010	CT0100625	25.4	2.4	1.9	4.3	911	
WATERBURY WPCF	02/07/2010	CT0100625	23.4	1.9	1.7	3.6	703	
WATERBURY WPCF	02/08/2010	CT0100625	22.3	1.4	1.6	3	558	
WATERBURY WPCF	02/09/2010	CT0100625	22.8	4.1	3.3	7.4	1407	
WATERBURY WPCF	02/15/2010	CT0100625	21.1	1.3	1.5	2.8	493	
WATERBURY WPCF	02/16/2010	CT0100625	20.7	1.2	1.8	3	518	
WATERBURY WPCF	02/17/2010	CT0100625	21.1	1.9	2.1	4	704	
WATERBURY WPCF	02/21/2010	CT0100625	21.5	1.6	1.6	3.2	574	
WATERBURY WPCF	02/22/2010	CT0100625	20.6	1.8	1.3	3.1	533	
WATERBURY WPCF	02/23/2010	CT0100625	22	2.6	1.7	4.3	789	
WATERBURY WPCF	02/28/2010	CT0100625	[illegible]	2.8	2	4.8	1497	3.9
WATERBURY WPCF	03/01/2010	CT0100625	35.3	1.8	1.2	3	883	
WATERBURY WPCF	03/02/2010	CT0100625	33.4	3.4	1.8	5.2	1449	
WATERBURY WPCF	03/07/2010	CT0100625	29.2	3.1	0.8	3.9	950	
WATERBURY WPCF	03/08/2010	CT0100625	28.3	3.4	0.8	4.2	991	
WATERBURY WPCF	03/09/2010	CT0100625	28.4	4.1	1.1	5.2	1232	
WATERBURY WPCF	03/14/2010	CT0100625	44.3	3.2	2.8	6	2217	
WATERBURY WPCF	03/15/2010	CT0100625	48	3.5	3.2	6.7	2682	
WATERBURY WPCF	03/16/2010	CT0100625	43.4	3.5	2.7	6.2	2244	
WATERBURY WPCF	03/21/2010	CT0100625	30.8	1.7	1.1	2.8	719	
WATERBURY WPCF	03/22/2010	CT0100625	31.6	1.6	1.6	3.2	843	
WATERBURY WPCF	03/23/2010	CT0100625	43.9	4.9	2.3	7.2	2636	
WATERBURY WPCF	03/28/2010	CT0100625	31.1	1.3	1.9	3.2	830	
WATERBURY WPCF	03/29/2010	CT0100625	44.5	3.5	2	5.5	2041	
WATERBURY WPCF	03/30/2010	CT0100625	[illegible]	[illegible]	3.6	[illegible]	[illegible]	5.0
WATERBURY WPCF	04/04/2010	CT0100625	41	1.4	2.4	3.8	1299	
WATERBURY WPCF	04/05/2010	CT0100625	40.6	1.5	1.7	3.2	1004	
WATERBURY WPCF	04/06/2010	CT0100625	35	0.9	2.5	3.4	992	
WATERBURY WPCF	04/11/2010	CT0100625	29.5	1.4	2	3.4	837	
WATERBURY WPCF	04/12/2010	CT0100625	29.8	0.8	1.8	2.6	642	
WATERBURY WPCF	04/13/2010	CT0100625	29.5	1	2.7	3.7	910	
WATERBURY WPCF	04/18/2010	CT0100625	26.4	1.2	2.6	3.8	837	
WATERBURY WPCF	04/19/2010	CT0100625	23.5	1	2.9	3.9	764	
WATERBURY WPCF	04/20/2010	CT0100625	23.7	1.1	3.2	4.3	850	
WATERBURY WPCF	04/25/2010	CT0100625	23.7	1.5	3.6	5.1	1008	
WATERBURY WPCF	04/26/2010	CT0100625	23.7	1.6	2.8	4.4	870	
WATERBURY WPCF	04/27/2010	CT0100625	22.5	1.3	3.3	4.6	863	3.9
WATERBURY WPCF	05/02/2010	CT0100625	20.6	7.6	3.3	10.9	1873	
WATERBURY WPCF	05/03/2010	CT0100625	22.4	1.4	2.5	3.9	729	
WATERBURY WPCF	05/04/2010	CT0100625	20.8	1.5	3.5	5	867	
WATERBURY WPCF	05/09/2010	CT0100625	19	1	1.9	2.9	460	
WATERBURY WPCF	05/10/2010	CT0100625	20.1	1.3	1.7	3	503	
WATERBURY WPCF	05/11/2010	CT0100625	20	1.5	1.9	3.4	567	
WATERBURY WPCF	05/16/2010	CT0100625	19.7	1	1.7	2.7	444	
WATERBURY WPCF	05/17/2010	CT0100625	20.1	0.9	1.6	2.5	419	
WATERBURY WPCF	05/18/2010	CT0100625	21.9	0.8	1.6	2.4	438	
WATERBURY WPCF	05/23/2010	CT0100625	19.4	0.5	2	2.5	405	
WATERBURY WPCF	05/24/2010	CT0100625	20.1	0.5	2.1	2.6	436	
WATERBURY WPCF	05/25/2010	CT0100625	20.1	0.5	2.8	3.3	553	
WATERBURY WPCF	05/31/2010	CT0100625	19.2	1.5	2.4	3.9	625	3.8
WATERBURY WPCF	06/01/2010	CT0100625	19.9	1.2	2.6	3.8	631	
WATERBURY WPCF	06/02/2010	CT0100625	19.9	1.2	3.2	4.4	730	
WATERBURY WPCF	06/06/2010	CT0100625	19.2	2.1	2.5	4.6	737	
WATERBURY WPCF	06/07/2010	CT0100625	19.7	1.7	2	3.7	608	
WATERBURY WPCF	06/08/2010	CT0100625	19.7	1.6	1.9	3.5	575	
WATERBURY WPCF	06/13/2010	CT0100625	20.5	1.7	1.9	3.6	616	
WATERBURY WPCF	06/14/2010	CT0100625	19.9	1.1	2.1	3.2	531	
WATERBURY WPCF	06/15/2010	CT0100625	20.1	1.4	2.3	3.7	620	
WATERBURY WPCF	06/20/2010	CT0100625	17.7	1.2	3	4.2	620	
WATERBURY WPCF	06/21/2010	CT0100625	18.5	1.3	2.1	3.4	525	
WATERBURY WPCF	06/22/2010	CT0100625	17.6	1.5	2.6	4.1	602	
WATERBURY WPCF	06/27/2010	CT0100625	18	1.2	3.2	4.4	661	
WATERBURY WPCF	06/28/2010	CT0100625	18.6	1.4	2.6	4	621	
WATERBURY WPCF	06/29/2010	CT0100625	19.1	1.6	3.3	4.9	781	4.0
WATERBURY WPCF	07/05/2010	CT0100625	17.1	5.9	1.5	7.4	1055	
WATERBURY WPCF	07/06/2010	CT0100625	18.4	2	2.5	4.5	691	
WATERBURY WPCF	07/07/2010	CT0100625	18.1	1.6	2.9	4.5	679	
WATERBURY WPCF	07/11/2010	CT0100625	17	1.4	3.1	4.5	638	

WATERBURY WPCF	07/12/2010	CT0100625	18.1	0.6		3.9	4.5	679	
WATERBURY WPCF	07/13/2010	CT0100625	20.8	1.4		4.7	6.1	1058	
WATERBURY WPCF	07/19/2010	CT0100625	18.1	1.2		4.8	6	906	
WATERBURY WPCF	07/20/2010	CT0100625	17.8	1.3		6.4	7.7	1143	
WATERBURY WPCF	07/21/2010	CT0100625	17.9	1.3		7.1	8.4	1254	
WATERBURY WPCF	07/25/2010	CT0100625	17.6	1.3		2.4	3.7	543	
WATERBURY WPCF	07/26/2010	CT0100625	17.7	1.1		2.7	3.8	561	
WATERBURY WPCF	[illegible]	[illegible]	[illegible]	[illegible]		[illegible]	[illegible]	[illegible]	[illegible]
WATERBURY WPCF	08/01/2010	CT0100625	16.5	1.3		1.3	2.6	358	
WATERBURY WPCF	08/02/2010	CT0100625	17.4	1.4		1.4	2.8	406	
WATERBURY WPCF	08/03/2010	CT0100625	16.5	1.6		1.7	3.3	454	
WATERBURY WPCF	08/08/2010	CT0100625	16.2	1.1		1.6	2.7	365	
WATERBURY WPCF	08/09/2010	CT0100625	18	1.2		2.4	3.6	540	
WATERBURY WPCF	08/10/2010	CT0100625	17.3	1.1		2.5	3.6	519	
WATERBURY WPCF	08/15/2010	CT0100625	16.3	1.3		1.2	2.5	340	
WATERBURY WPCF	08/16/2010	CT0100625	20.1	1.2		1.6	2.8	469	
WATERBURY WPCF	08/17/2010	CT0100625	18.9	0.8		2	2.8	441	
WATERBURY WPCF	08/22/2010	CT0100625	21	1.6		1.5	3.1	543	
WATERBURY WPCF	08/23/2010	CT0100625	22.5	1.1		1.2	2.3	432	
WATERBURY WPCF	08/24/2010	CT0100625	19.3	1.9		1.2	3.1	499	
WATERBURY WPCF	08/30/2010	CT0100625	19	1.4		1.8	3.2	507	
WATERBURY WPCF	08/31/2010	CT0100625	[illegible]	1.3		2.4	3.7	586	[illegible]
WATERBURY WPCF	09/06/2010	CT0100625	17.6	1.4		0.1	1.5	220	
WATERBURY WPCF	09/07/2010	CT0100625	18.7	0.7		1.1	1.8	281	
WATERBURY WPCF	09/08/2010	CT0100625	18.2	1.5		2.3	3.8	577	
WATERBURY WPCF	09/12/2010	CT0100625	17.5	1.5		1.3	2.8	409	
WATERBURY WPCF	09/13/2010	CT0100625	18.1	1.9		1	2.9	438	
WATERBURY WPCF	09/14/2010	CT0100625	17.2	2.5		1.2	3.7	531	
WATERBURY WPCF	09/19/2010	CT0100625	17.2	2.6		1.3	3.9	559	
WATERBURY WPCF	09/20/2010	CT0100625	16.5	1.5		1.4	2.9	399	
WATERBURY WPCF	09/21/2010	CT0100625	14.4	1.7		1.7	3.4	408	
WATERBURY WPCF	09/26/2010	CT0100625	17.7	1.2		2.5	3.7	546	
WATERBURY WPCF	09/27/2010	CT0100625	18.3	1.3		2.4	3.7	565	
WATERBURY WPCF	[illegible]	[illegible]	[illegible]	[illegible]		2.6	[illegible]	[illegible]	[illegible]
WATERBURY WPCF	10/03/2010	CT0100625	19.2	1.3		1.8	3.1	496	
WATERBURY WPCF	10/04/2010	CT0100625	19.7	1.3		1.6	2.9	476	
WATERBURY WPCF	10/05/2010	CT0100625	19.8	1.7		1.7	3.4	561	
WATERBURY WPCF	10/11/2010	CT0100625	18.4	0.6		1.6	2.2	338	
WATERBURY WPCF	10/12/2010	CT0100625	19.2	0.5		1.5	2	320	
WATERBURY WPCF	10/13/2010	CT0100625	19.1	0.6		1.9	2.5	398	
WATERBURY WPCF	10/17/2010	CT0100625	18.4	1.3		1.8	3.1	476	
WATERBURY WPCF	10/18/2010	CT0100625	19.2	1.1		1.4	2.5	400	
WATERBURY WPCF	10/19/2010	CT0100625	19	1.3		1.8	3.1	491	
WATERBURY WPCF	10/24/2010	CT0100625	17.7	1.3		1.7	3	443	
WATERBURY WPCF	10/25/2010	CT0100625	18.9	1.3		1.7	3	473	
WATERBURY WPCF	10/26/2010	CT0100625	19.1	1.2		2.6	3.8	605	2.9
WATERBURY WPCF	11/01/2010	CT0100625	18.5	1.7		1.6	3.3	509	
WATERBURY WPCF	11/02/2010	CT0100625	18.2	1.4		1.9	3.3	501	
WATERBURY WPCF	11/07/2010	CT0100625	19	1.6		1.4	3	475	
WATERBURY WPCF	11/08/2010	CT0100625	19.2	1.6		0.2	1.8	288	
WATERBURY WPCF	11/09/2010	CT0100625	19.5	1.7		1.7	3.4	553	
WATERBURY WPCF	11/14/2010	CT0100625	18.1	1.9		1.4	3.3	498	
WATERBURY WPCF	11/15/2010	CT0100625	18.9	2.1		1.3	3.4	536	
WATERBURY WPCF	11/16/2010	CT0100625	19.2	2.5		1.2	3.7	592	
WATERBURY WPCF	11/21/2010	CT0100625	19	2.1		1.3	3.4	539	
WATERBURY WPCF	11/22/2010	CT0100625	19.6	2.9		1.3	4.2	687	
WATERBURY WPCF	11/23/2010	CT0100625	19.7	1.8		1.8	3.6	591	
WATERBURY WPCF	11/28/2010	CT0100625	18.3	1.8		1.8	3.6	549	
WATERBURY WPCF	11/29/2010	CT0100625	19	1.9		1.9	3.8	602	
WATERBURY WPCF	11/30/2010	CT0100625	19.4	1.2		2.1	3.3	534	3.4
WATERBURY WPCF	12/05/2010	CT0100625	22.6	1.2		1.7	2.9	547	
WATERBURY WPCF	12/06/2010	CT0100625	21.4	1.2		1.6	2.8	500	
WATERBURY WPCF	12/08/2010	CT0100625	20.6	2.8		2.6	5.4	928	
WATERBURY WPCF	12/12/2010	CT0100625	30.1	3.3		1.8	5.1	1280	
WATERBURY WPCF	12/13/2010	CT0100625	34.4	2.5		2.2	4.7	1340	
WATERBURY WPCF	12/14/2010	CT0100625	30.4	2.4		3	5.4	1369	
WATERBURY WPCF	12/19/2010	CT0100625	24.2	2.5		1.8	4.3	868	
WATERBURY WPCF	12/20/2010	CT0100625	23.7	2.2		1.3	3.5	692	
WATERBURY WPCF	12/21/2010	CT0100625	22.3	2.7		2.2	4.9	911	
WATERBURY WPCF	12/26/2010	CT0100625	19.7	2.1		1.7	3.8	624	
WATERBURY WPCF	12/27/2010	CT0100625	20	2.2		2.1	4.3	717	
WATERBURY WPCF	12/28/2010	CT0100625	19.9	2.6		1.9	4.5	747	4.3
					Average		4.0405	802.9084967	4.1
					April - October		3.709	623.1797753	3.7
					Max		22.1	6469	6.0
					Min		1.5	220	2.9
					Max (Apr-Oct)		10.9	1873	5.4
					Min (Apr-Oct)		1.5	220	2.9

Plant	Date	Permit	FlowMGD	FTKN	NO2NO3	TN	TNMASS	TN Monthly Average
WESTPORT WPCF	01/12/2010	CT0100684	1.73	1.4	0.8	2.2	32	
WESTPORT WPCF	01/19/2010	CT0100684	1.67	1.9	1.2	3.1	43	
WESTPORT WPCF	01/26/2010	CT0100684	2.19	3.4	0.5	3.9	71	3.1
WESTPORT WPCF	02/02/2010	CT0100684	1.67	3.6	1.1	4.7	65	
WESTPORT WPCF	02/09/2010	CT0100684	1.54	4.1	1.1	5.2	67	
WESTPORT WPCF	02/16/2010	CT0100684	1.28	3.4	1.4	4.8	51	
WESTPORT WPCF	02/23/2010	CT0100684	1.46	2.8	1.2	4	49	4.7
WESTPORT WPCF	03/02/2010	CT0100684	3.23	1.7	1.3	3	81	
WESTPORT WPCF	03/09/2010	CT0100684	2.29	1.2	2.4	3.6	69	
WESTPORT WPCF	03/16/2010	CT0100684	4.44	1.2	5.7	6.9	256	
WESTPORT WPCF	03/23/2010	CT0100684	3.6	1	2.8	3.8	114	
WESTPORT WPCF	03/30/2010	CT0100684	5.39	1.2	2.9	4.1	184	4.3
WESTPORT WPCF	04/06/2010	CT0100684	3.08	1	2.2	3.2	82	
WESTPORT WPCF	04/13/2010	CT0100684	2.2	1.1	1.3	2.4	44	
WESTPORT WPCF	04/20/2010	CT0100684	1.83	1	0.9	1.9	29	
WESTPORT WPCF	04/27/2010	CT0100684	2.2	1	1	2	37	2.4
WESTPORT WPCF	05/04/2010	CT0100684	1.99	1.3	0.8	2.1	35	
WESTPORT WPCF	05/11/2010	CT0100684	1.47	1.2	1.4	2.6	32	
WESTPORT WPCF	05/25/2010	CT0100684	1.45	1.1	1.7	2.8	34	
WESTPORT WPCF	05/28/2010	CT0100684	1.57	1.4	1.5	2.9	38	2.6
WESTPORT WPCF	06/01/2010	CT0100684	1.41	1	0.9	1.9	22	
WESTPORT WPCF	06/08/2010	CT0100684	1.49	1.2	0.7	1.9	24	
WESTPORT WPCF	06/15/2010	CT0100684	1.33	1.9	0.4	2.3	26	
WESTPORT WPCF	06/20/2010	CT0100684	1.34	1.4	0.8	2.2	25	2.1
WESTPORT WPCF	07/06/2010	CT0100684	1.06	1.1	0.8	1.9	17	
WESTPORT WPCF	07/13/2010	CT0100684	1.19	1.8	0.7	2.5	25	
WESTPORT WPCF	07/20/2010	CT0100684	1.5	1.3	0.6	1.9	24	
WESTPORT WPCF	07/28/2010	CT0100684	1.23	1.1	0.7	1.8	18	2.0
WESTPORT WPCF	08/03/2010	CT0100684	1.21	1	0.9	1.9	19	
WESTPORT WPCF	08/10/2010	CT0100684	1.2	1.1	0.6	1.7	17	
WESTPORT WPCF	08/17/2010	CT0100684	1.25	1.3	0.3	1.6	17	
WESTPORT WPCF	08/24/2010	CT0100684	1.46	1.2	0.6	1.8	22	
WESTPORT WPCF	08/31/2010	CT0100684	1.24	1.1	0.5	1.6	17	1.7
WESTPORT WPCF	09/06/2010	CT0100684	1.03	1.1	0.7	1.8	15	
WESTPORT WPCF	09/14/2010	CT0100684	1.23	1.9	0.6	2.5	26	
WESTPORT WPCF	09/21/2010	CT0100684	1.14	1.5	0.7	2.2	21	
WESTPORT WPCF	09/28/2010	CT0100684	1.3	1.3	0.5	1.8	20	2.1
WESTPORT WPCF	10/05/2010	CT0100684	1.51	1.5	0.5	2	25	
WESTPORT WPCF	10/12/2010	CT0100684	1.25	1.1	0.4	1.5	16	
WESTPORT WPCF	10/19/2010	CT0100684	1.31	1.3	0.8	2.1	23	
WESTPORT WPCF	10/26/2010	CT0100684	1.31	1.5	1.1	2.6	28	2.1
WESTPORT WPCF	11/02/2010	CT0100684	1.173	1.8	0.5	2.3	23	
WESTPORT WPCF	11/09/2010	CT0100684	1.37	1.1	1	2.1	24	
WESTPORT WPCF	11/16/2010	CT0100684	1.32	1.5	0.5	2	22	
WESTPORT WPCF	11/23/2010	CT0100684	1.4	1.4	0.4	1.8	21	
WESTPORT WPCF	11/30/2010	CT0100684	1.35	1.3	0.7	2	23	2.0
WESTPORT WPCF	12/07/2010	CT0100684	1.46	1.4	0.8	2.2	27	
WESTPORT WPCF	12/14/2010	CT0100684	1.93	1.3	2.2	3.5	56	
WESTPORT WPCF	12/21/2010	CT0100604	1.56	1.3	1.1	2.4	31	
WESTPORT WPCF	12/28/2010	CT0100684	1.27	1.3	0.9	2.2	23	2.6
					Average	2.624	42.2	2.6
					April - October	2.1172	26.82758621	2.1
					Max	6.9	256	4.7
					Min	1.5	15	1.7
					Max (Apr-Oct)	3.2	82	2.6
					Min (Apr-Oct)	1.5	15	1.7

Plant	Date	Permit	FlowMGD	FTKN	NO2NO3	TN	TNMASS	TN Monthly Average
STAMFORD WPCF	01/04/2010	CT0101087	17.9	2.3	1.1	3.4	508	
STAMFORD WPCF	01/05/2010	CT0101087	17.4	2.9	1.6	4.5	653	
STAMFORD WPCF	01/06/2010	CT0101087	17	2.6	1.9	4.5	638	
STAMFORD WPCF	01/07/2010	CT0101087	16.7	2.6	1.7	4.3	599	
STAMFORD WPCF	01/11/2010	CT0101087	16	2.7	1.8	4.5	600	
STAMFORD WPCF	01/12/2010	CT0101087	15.8	3	1.6	4.6	606	
STAMFORD WPCF	01/13/2010	CT0101087	15.7	2.3	1.9	4.2	550	
STAMFORD WPCF	01/14/2010	CT0101087	15.5	2.6	1.6	4.2	543	
STAMFORD WPCF	01/18/2010	CT0101087	15.9	2.3	1	3.3	438	
STAMFORD WPCF	01/19/2010	CT0101087	15.3	2.8	0.8	3.6	459	
STAMFORD WPCF	01/20/2010	CT0101087	15.3	2	1.1	3.1	396	
STAMFORD WPCF	01/21/2010	CT0101087	15.1	2	1.5	3.5	441	
STAMFORD WPCF	01/24/2010	CT0101087	14.7	2.9	1.2	4.1	503	
STAMFORD WPCF	01/25/2010	CT0101087	17.5	9.5	1.4	10.9	1591	
STAMFORD WPCF	01/26/2010	CT0101087	17.5	3.2	0.9	4.1	598	
STAMFORD WPCF	01/27/2010	CT0101087	17	3.2	1.7	4.9	695	
STAMFORD WPCF	01/28/2010	CT0101087	16.8	2.8	1.7	4.6	645	
STAMFORD WPCF	01/31/2010	CT0101087	16.2	6.2	1.4	7.6	1027	4.7
STAMFORD WPCF	02/01/2010	CT0101087	16	8.9	0.9	9.8	1308	
STAMFORD WPCF	02/02/2010	CT0101087	15.7	4.5	0.8	5.3	694	
STAMFORD WPCF	02/03/2010	CT0101087	15.5	2.7	1.1	3.8	491	
STAMFORD WPCF	02/04/2010	CT0101087	15.4	2.7	1.3	4	514	
STAMFORD WPCF	02/07/2010	CT0101087	15.1	2.8	1.8	4.6	579	
STAMFORD WPCF	02/08/2010	CT0101087	15	2.9	1.6	4.5	563	
STAMFORD WPCF	02/09/2010	CT0101087	14.5	3.2	1.4	4.6	556	
STAMFORD WPCF	02/10/2010	CT0101087	14.5	3.1	1.3	4.4	532	
STAMFORD WPCF	02/15/2010	CT0101087	14.3	2.7	0.6	3.3	394	
STAMFORD WPCF	02/16/2010	CT0101087	14.4	2.9	0.7	3.6	432	
STAMFORD WPCF	02/17/2010	CT0101087	14.3	2.4	1.2	3.6	429	
STAMFORD WPCF	02/21/2010	CT0101087	14.8	2.2	0.9	3.1	383	
STAMFORD WPCF	02/22/2010	CT0101087	14.8	6.2	1.2	7.4	913	
STAMFORD WPCF	02/23/2010	CT0101087	15.9	6.6	1.6	8.2	1087	
STAMFORD WPCF	02/28/2010	CT0101087	27.8	2.4	1.5	3.9	904	4.9
STAMFORD WPCF	03/01/2010	CT0101087	26.5	2.5	1.2	3.7	818	
STAMFORD WPCF	03/02/2010	CT0101087	25.4	2	1.8	3.8	805	
STAMFORD WPCF	03/03/2010	CT0101087	24.6	1.9	1.8	3.7	759	
STAMFORD WPCF	03/04/2010	CT0101087	23.4	1.9	1.7	3.6	703	
STAMFORD WPCF	03/07/2010	CT0101087	20.8	2.2	0.7	2.9	503	
STAMFORD WPCF	03/08/2010	CT0101087	20.2	2	0.7	2.7	455	
STAMFORD WPCF	03/09/2010	CT0101087	19.6	2.2	1.2	3.4	556	
STAMFORD WPCF	03/10/2010	CT0101087	19	2.2	0.9	3.1	491	
STAMFORD WPCF	03/11/2010	CT0101087	18.5	2.4	1	3.4	525	
STAMFORD WPCF	03/15/2010	CT0101087	34.3	13.1	0.2	13.3	3805	
STAMFORD WPCF	03/16/2010	CT0101087	30.3	11.9	0.3	12.2	3083	
STAMFORD WPCF	03/17/2010	CT0101087	26.8	5.1	0.4	5.5	1229	
STAMFORD WPCF	03/18/2010	CT0101087	25.7	1.8	0.5	2.3	493	
STAMFORD WPCF	03/21/2010	CT0101087	21.7	1.9	0.5	2.4	434	
STAMFORD WPCF	03/24/2010	CT0101087	30.4	5.9	0.6	6.5	1648	
STAMFORD WPCF	03/25/2010	CT0101087	27.3	3.3	1.2	4.5	1025	
STAMFORD WPCF	03/28/2010	CT0101087	22.8	1.9	0.5	2.4	456	
STAMFORD WPCF	03/29/2010	CT0101087	33.7	14	0.3	14.3	4019	
STAMFORD WPCF	03/31/2010	CT0101087	46.6	7.8	1.5	9.3	3614	5.4
STAMFORD WPCF	04/04/2010	CT0101087	26.6	2.4	0.3	2.7	599	
STAMFORD WPCF	04/05/2010	CT0101087	25.3	2.2	0.3	2.5	528	
STAMFORD WPCF	04/06/2010	CT0101087	24.1	2	0.4	2.4	402	
STAMFORD WPCF	04/07/2010	CT0101087	22.9	2.2	0.4	2.6	497	
STAMFORD WPCF	04/08/2010	CT0101087	22	2.4	0.4	2.8	514	
STAMFORD WPCF	04/11/2010	CT0101087	19.6	2.2	0.6	2.8	458	
STAMFORD WPCF	04/12/2010	CT0101087	19.3	2.3	0.6	2.9	467	
STAMFORD WPCF	04/13/2010	CT0101087	18.6	2.6	0.9	3.5	543	
STAMFORD WPCF	04/14/2010	CT0101087	18.3	2.2	1.1	3.3	504	
STAMFORD WPCF	04/15/2010	CT0101087	17.9	2.3	1.3	3.6	537	
STAMFORD WPCF	04/18/2010	CT0101087	17.6	2.1	1.3	3.4	499	
STAMFORD WPCF	04/19/2010	CT0101087	17.3	2.1	0.8	2.9	418	
STAMFORD WPCF	04/20/2010	CT0101087	16.9	2.6	0.9	3.5	493	
STAMFORD WPCF	04/21/2010	CT0101087	16.7	2.4	0.8	3.2	446	
STAMFORD WPCF	04/22/2010	CT0101087	16.6	2.4	0.7	3.1	429	
STAMFORD WPCF	04/25/2010	CT0101087	17.2	2.3	1	3.3	473	
STAMFORD WPCF	04/26/2010	CT0101087	18.7	2.8	0.6	3.4	530	
STAMFORD WPCF	04/27/2010	CT0101087	18.8	3.4	0.8	4.2	659	
STAMFORD WPCF	04/28/2010	CT0101087	16.6	1.8	1.8	3.6	498	
STAMFORD WPCF	04/29/2010	CT0101087	16.8	1.9	0.9	2.8	392	[illegible]
STAMFORD WPCF	05/02/2010	CT0101087	16.5	1.8	0.7	2.5	344	
STAMFORD WPCF	05/03/2010	CT0101087	18.4	1.9	0.6	2.5	384	
STAMFORD WPCF	05/04/2010	CT0101087	17.2	1.9	0.7	2.6	373	
STAMFORD WPCF	05/05/2010	CT0101087	16.8	1.9	0.7	2.6	364	
STAMFORD WPCF	05/06/2010	CT0101087	16.4	1.9	0.8	2.7	369	
STAMFORD WPCF	05/09/2010	CT0101087	15.8	2	0.6	2.6	343	
STAMFORD WPCF	05/10/2010	CT0101087	15.9	2.3	0.5	2.8	371	
STAMFORD WPCF	05/11/2010	CT0101087	15.6	2.4	0.6	3	390	
STAMFORD WPCF	05/12/2010	CT0101087	16.1	2.4	0.7	3.1	416	

STAMFORD WPCF	05/13/2010	CT0101087	15.7	2.2	1.2	3.4	445	
STAMFORD WPCF	05/16/2010	CT0101087	15.2	2.1	0.8	2.9	388	
STAMFORD WPCF	05/17/2010	CT0101087	15.3	2.2	0.7	2.9	370	
STAMFORD WPCF	05/18/2010	CT0101087	16.3	2.4	1	3.4	462	
STAMFORD WPCF	05/19/2010	CT0101087	16.2	1.5	1.3	2.8	378	
STAMFORD WPCF	05/20/2010	CT0101087	15.8	1.4	1.1	2.5	329	
STAMFORD WPCF	05/23/2010	CT0101087	15.2	1.5	0.9	2.4	304	
STAMFORD WPCF	05/24/2010	CT0101087	15.6	1.7	0.7	2.4	312	
STAMFORD WPCF	05/25/2010	CT0101087	15.4	1.8	0.6	2.4	308	
STAMFORD WPCF	05/26/2010	CT0101087	15.5	1.6	0.7	2.3	297	
STAMFORD WPCF	05/27/2010	CT0101087	15.1	1.6	0.8	2.4	302	
STAMFORD WPCF	05/31/2010	CT0101087	14.7	1.4	0.9	2.3	282	27
STAMFORD WPCF	06/01/2010	CT0101087	15.1	1.6	0.8	2.4	302	
STAMFORD WPCF	06/02/2010	CT0101087	14.8	1.7	0.8	2.5	309	
STAMFORD WPCF	06/03/2010	CT0101087	14.7	1.4	1	2.4	294	
STAMFORD WPCF	05/06/2010	CT0101087	14.6	1.5	0.9	2.4	292	
STAMFORD WPCF	06/07/2010	CT0101087	14.3	1.5	0.7	2.2	262	
STAMFORD WPCF	06/08/2010	CT0101087	14.2	2.1	0.9	3	355	
STAMFORD WPCF	06/09/2010	CT0101087	14.8	1.9	1.3	3.2	395	
STAMFORD WPCF	06/13/2010	CT0101087	14.2	1.7	0.6	2.3	272	
STAMFORD WPCF	06/14/2010	CT0101087	14.5	1.7	0.7	2.4	290	
STAMFORD WPCF	06/15/2010	CT0101087	14.4	2.1	0.6	2.7	324	
STAMFORD WPCF	06/16/2010	CT0101087	14.4	1.6	0.8	2.4	288	
STAMFORD WPCF	06/17/2010	CT0101087	14.6	1.8	1	2.8	341	
STAMFORD WPCF	06/20/2010	CT0101087	14.2	2	1.5	3.5	415	
STAMFORD WPCF	06/21/2010	CT0101087	14.3	1.9	1.2	3.1	370	
STAMFORD WPCF	06/22/2010	CT0101087	14.5	2.1	1.1	3.2	387	
STAMFORD WPCF	06/23/2010	CT0101087	14.8	1.6	1.1	2.7	333	
STAMFORD WPCF	06/24/2010	CT0101087	14.2	1.6	1.2	2.8	332	
STAMFORD WPCF	06/27/2010	CT0101087	13.8	1.7	0.9	2.6	299	
STAMFORD WPCF	06/28/2010	CT0101087	14.5	1.6	0.9	2.5	302	
STAMFORD WPCF	06/29/2010	CT0101087	14.2	1.8	0.8	2.6	308	
STAMFORD WPCF	06/30/2010	CT0101087	13.7	1.7	1.4	3.3	343	27
STAMFORD WPCF	07/05/2010	CT0101087	13	1.6	1.7	3.3	358	
STAMFORD WPCF	07/06/2010	CT0101087	13.4	1.6	1.4	3	335	
STAMFORD WPCF	07/07/2010	CT0101087	13.6	1.7	1.8	3.5	397	
STAMFORD WPCF	07/08/2010	CT0101087	13.8	1.4	1.7	3.1	357	
STAMFORD WPCF	07/11/2010	CT0101087	13.3	1.5	1.5	3	333	
STAMFORD WPCF	07/12/2010	CT0101087	13.6	1.5	1.1	2.6	295	
STAMFORD WPCF	07/13/2010	CT0101087	15.4	1.8	1	2.8	360	
STAMFORD WPCF	07/14/2010	CT0101087	15.2	1.6	0.8	2.4	304	
STAMFORD WPCF	07/15/2010	CT0101087	15	1.5	0.7	2.2	275	
STAMFORD WPCF	07/18/2010	CT0101087	13.6	1.5	0.8	2.3	261	
STAMFORD WPCF	07/19/2010	CT0101087	15.2	1.6	0.7	2.3	292	
STAMFORD WPCF	07/20/2010	CT0101087	14.4	2.5	0.7	3.2	384	
STAMFORD WPCF	07/21/2010	CT0101087	14.9	2.1	1.1	3.2	398	
STAMFORD WPCF	07/22/2010	CT0101087	15	2.3	1.5	3.8	475	
STAMFORD WPCF	07/25/2010	CT0101087	14.7	2.8	1.3	4.1	503	
STAMFORD WPCF	07/26/2010	CT0101087	14.5	2.4	1.5	3.9	472	
STAMFORD WPCF	07/27/2010	CT0101087	14.3	2.4	1.7	4.1	489	
STAMFORD WPCF	07/28/2010	CT0101087	14.3	1.9	1.9	3.8	453	
STAMFORD WPCF	07/29/2010	CT0101087	14.3	1.9	2	3.9	465	32
STAMFORD WPCF	08/01/2010	CT0101087	13.4	2	0.7	2.7	302	
STAMFORD WPCF	08/02/2010	CT0101087	13.8	2.1	0.8	2.9	334	
STAMFORD WPCF	08/03/2010	CT0101087	13.6	2.7	0.7	3.4	386	
STAMFORD WPCF	08/04/2010	CT0101087	13.8	1.9	0.9	2.8	322	
STAMFORD WPCF	08/05/2010	CT0101087	13.7	1.9	0.9	2.8	320	
STAMFORD WPCF	08/08/2010	CT0101087	13.2	1.8	1	2.8	308	
STAMFORD WPCF	08/09/2010	CT0101087	13.7	1.8	1.1	2.9	331	
STAMFORD WPCF	08/10/2010	CT0101087	13.6	1.9	1	2.9	329	
STAMFORD WPCF	08/11/2010	CT0101087	13.6	1.6	0.9	2.5	284	
STAMFORD WPCF	08/12/2010	CT0101087	13.7	1.6	0.9	2.5	286	
STAMFORD WPCF	08/15/2010	CT0101087	13	1.7	1	2.7	293	
STAMFORD WPCF	08/16/2010	CT0101087	14	1.6	1.1	2.7	315	
STAMFORD WPCF	08/17/2010	CT0101087	13.5	1.8	0.8	2.6	293	
STAMFORD WPCF	08/18/2010	CT0101087	13.1	1.7	1.1	2.8	306	
STAMFORD WPCF	08/19/2010	CT0101087	13	1.8	1.1	2.9	314	
STAMFORD WPCF	08/22/2010	CT0101087	14.6	1.8	0.8	2.6	317	
STAMFORD WPCF	08/23/2010	CT0101087	15.3	1.7	0.6	2.3	293	
STAMFORD WPCF	08/24/2010	CT0101087	14.1	1.8	0.7	2.5	294	
STAMFORD WPCF	08/25/2010	CT0101087	14.1	1.5	0.7	2.2	259	
STAMFORD WPCF	08/28/2010	CT0101087	13.9	1.7	0.7	2.4	278	
STAMFORD WPCF	08/29/2010	CT0101087	13.4	1.8	0.9	2.7	302	
STAMFORD WPCF	08/30/2010	CT0101087	13.8	1.7	0.8	2.5	288	
STAMFORD WPCF	08/31/2010	CT0101087	13.8	1.7	0.8	2.5	288	27
STAMFORD WPCF	09/01/2010	CT0101087	13.8	1.5	0.8	2.3	265	
STAMFORD WPCF	09/02/2010	CT0101087	13.7	1.5	0.9	2.4	274	
STAMFORD WPCF	09/06/2010	CT0101087	13.1	1.7	0.9	2.6	284	
STAMFORD WPCF	09/07/2010	CT0101087	13.5	1.7	0.6	2.3	259	
STAMFORD WPCF	09/08/2010	CT0101087	13.4	1.9	0.7	2.6	291	
STAMFORD WPCF	09/09/2010	CT0101087	13.4	1.9	0.8	2.7	302	
STAMFORD WPCF	09/12/2010	CT0101087	13.2	2.4	1.1	3.5	385	

STAMFORD WPCF	09/13/2010	CT0101087	13.5	2.1	1	3.1	349	
STAMFORD WPCF	09/14/2010	CT0101087	13.1	2.3	0.9	3.2	350	
STAMFORD WPCF	09/15/2010	CT0101087	13	2.1	1.2	3.3	358	
STAMFORD WPCF	09/16/2010	CT0101087	13.8	1.9	0.9	2.8	322	
STAMFORD WPCF	09/19/2010	CT0101087	13	2.1	0.9	3	325	
STAMFORD WPCF	09/20/2010	CT0101087	13	2	0.7	2.7	293	
STAMFORD WPCF	09/21/2010	CT0101087	12.9	2.2	0.7	2.9	312	
STAMFORD WPCF	09/22/2010	CT0101087	13.3	1.6	0.9	2.5	277	
STAMFORD WPCF	09/23/2010	CT0101087	13.2	1.6	0.8	2.4	264	
STAMFORD WPCF	09/26/2010	CT0101087	12.8	1.7	0.7	2.4	256	
STAMFORD WPCF	09/27/2010	CT0101087	13.9	1.9	0.6	2.5	290	
STAMFORD WPCF	09/28/2010	CT0101087	13.8	3.2	0.6	3.8	437	
STAMFORD WPCF	09/29/2010	CT0101087	13.5	2.5	0.7	3.3	372	2.8
STAMFORD WPCF	10/03/2010	CT0101087	15.5	1.7	0.8	2.5	323	
STAMFORD WPCF	10/04/2010	CT0101087	15.5	1.8	0.7	2.5	323	
STAMFORD WPCF	10/05/2010	CT0101087	15.3	2.1	0.7	2.8	357	
STAMFORD WPCF	10/06/2010	CT0101087	15.3	2	0.6	2.6	332	
STAMFORD WPCF	10/07/2010	CT0101087	15	2.3	0.7	3	375	
STAMFORD WPCF	10/11/2010	CT0101087	14.7	1.8	0.6	2.4	294	
STAMFORD WPCF	10/12/2010	CT0101087	14.7	1.8	0.6	2.4	294	
STAMFORD WPCF	10/13/2010	CT0101087	14.2	2.2	0.7	2.9	343	
STAMFORD WPCF	10/14/2010	CT0101087	15	1.7	0.7	2.4	300	
STAMFORD WPCF	10/17/2010	CT0101087	15	1.6	0.5	2.1	263	
STAMFORD WPCF	10/18/2010	CT0101087	14.9	2	0.5	2.5	311	
STAMFORD WPCF	10/19/2010	CT0101087	14.8	1.7	0.7	2.4	296	
STAMFORD WPCF	10/20/2010	CT0101087	14.6	1.9	0.8	2.7	329	
STAMFORD WPCF	10/21/2010	CT0101087	14.8	1.8	0.8	2.6	321	
STAMFORD WPCF	10/24/2010	CT0101087	14.4	1.6	0.7	2.3	276	
STAMFORD WPCF	10/25/2010	CT0101087	14.4	1.7	0.7	2.4	288	
STAMFORD WPCF	10/26/2010	CT0101087	14.4	1.9	0.9	2.8	336	
STAMFORD WPCF	10/27/2010	CT0101087	14.8	1.6	0.9	2.5	309	
STAMFORD WPCF	10/28/2010	CT0101087	14.4	1.7	1	2.7	324	
STAMFORD WPCF	10/31/2010	CT0101087	13.8	1.9	0.8	2.7	311	2.6
STAMFORD WPCF	11/01/2010	CT0101087	13.8	2.1	0.8	2.9	334	
STAMFORD WPCF	11/02/2010	CT0101087	13.9	2.1	0.8	2.9	336	
STAMFORD WPCF	11/03/2010	CT0101087	13.8	1.7	0.7	2.4	276	
STAMFORD WPCF	11/04/2010	CT0101087	15.3	1.8	0.5	2.3	293	
STAMFORD WPCF	11/07/2010	CT0101087	14.4	2	0.7	2.7	324	
STAMFORD WPCF	11/08/2010	CT0101087	15.2	2.1	0.9	3	380	
STAMFORD WPCF	11/09/2010	CT0101087	15.5	2.3	0.6	2.9	375	
STAMFORD WPCF	11/11/2010	CT0101087	14.1	2.4	1.3	3.7	435	
STAMFORD WPCF	11/14/2010	CT0101087	13.9	2.2	0.7	2.9	336	
STAMFORD WPCF	11/15/2010	CT0101087	13.9	2.3	0.7	3	348	
STAMFORD WPCF	11/16/2010	CT0101087	14	3.5	0.8	4.3	502	
STAMFORD WPCF	11/17/2010	CT0101087	15.4	2.4	0.8	3.2	411	
STAMFORD WPCF	11/18/2010	CT0101087	14.4	2.1	0.9	3	360	
STAMFORD WPCF	11/21/2010	CT0101087	14.1	2.3	0.6	2.9	341	
STAMFORD WPCF	11/22/2010	CT0101087	14	2.3	0.7	3	350	
STAMFORD WPCF	11/23/2010	CT0101087	14.1	2.1	0.9	3	353	
STAMFORD WPCF	11/28/2010	CT0101087	13.9	2.1	1.1	3.2	371	
STAMFORD WPCF	11/29/2010	CT0101087	13.7	2.1	0.9	3	343	
STAMFORD WPCF	11/30/2010	CT0101087	13.5	3.5	1	4.5	507	[illegible]
STAMFORD WPCF	12/05/2010	CT0101087	14.7	2.2	1	3.2	392	
STAMFORD WPCF	12/06/2010	CT0101087	14.6	2.1	0.9	3	365	
STAMFORD WPCF	12/07/2010	CT0101087	14.1	2.3	1.3	3.6	423	
STAMFORD WPCF	12/08/2010	CT0101087	14.1	2.3	1.6	3.9	459	
STAMFORD WPCF	12/09/2010	CT0101087	14	2.2	1.9	4.1	479	
STAMFORD WPCF	12/13/2010	CT0101087	17.1	2.5	0.7	3.2	456	
STAMFORD WPCF	12/14/2010	CT0101087	16	3.3	1	4.3	574	
STAMFORD WPCF	12/15/2010	CT0101087	15.7	2.5	1.2	3.7	484	
STAMFORD WPCF	12/16/2010	CT0101087	15.4	2.6	1.3	3.9	501	
STAMFORD WPCF	12/19/2010	CT0101087	15	2.4	1.4	3.8	475	
STAMFORD WPCF	12/20/2010	CT0101087	14.9	2.6	1.4	4	497	
STAMFORD WPCF	12/21/2010	CT0101087	14.8	2.2	2.1	4.3	531	
STAMFORD WPCF	12/22/2010	CT0101087	14.7	2.4	2	4.4	539	
STAMFORD WPCF	12/26/2010	CT0101087	13.8	2.3	1.5	3.8	437	
STAMFORD WPCF	12/27/2010	CT0101087	14	2.7	2.2	4.9	572	
STAMFORD WPCF	12/28/2010	CT0101087	14	2.4	2.1	4.5	525	
STAMFORD WPCF	12/29/2010	CT0101087	13.9	2.9	1.6	4.5	522	3.9
					Average	3.4147	487.0862069	3.5
					April - October	2.8146	354.9166667	2.8
					Max	14.3	4019	5.4
					Min	2.1	256	2.6
					Max (Apr-Oct)	4.2	659	3.2
					Min (Apr-Oct)	2.1	256	2.6

Plant	Date	Permit	FlowMGD	FTKN	NO2NO3	TN	TNMASS	TN Monthly Average
NEW CANAAN WPCF	01/12/2010	CT0101273	1	2	3.4	5.4	45	
NEW CANAAN WPCF	01/19/2010	CT0101273	0.9	2.4	1.9	4.3	32	
NEW CANAAN WPCF	01/26/2010	CT0101273	1.4	2	2.7	4.7	55	4.8
NEW CANAAN WPCF	02/02/2010	CT0101273	1	1.2	2.2	3.4	28	
NEW CANAAN WPCF	02/09/2010	CT0101273	0.9	1.4	3.2	4.6	35	
NEW CANAAN WPCF	02/16/2010	CT0101273	0.7	1.9	1.3	3.2	19	
NEW CANAAN WPCF	02/23/2010	CT0101273	0.8	2	1.9	3.9	26	3.8
NEW CANAAN WPCF	03/02/2010	CT0101273	2.1	1.5	3.4	4.9	86	
NEW CANAAN WPCF	03/09/2010	CT0101273	1.4	1.8	2.1	3.9	46	
NEW CANAAN WPCF	03/16/2010	CT0101273	2.7	1.5	2.6	4.1	92	
NEW CANAAN WPCF	03/23/2010	CT0101273	1.3	1.3	2.3	3.6	39	
NEW CANAAN WPCF	03/30/2010	CT0101273	[illegible]	[illegible]	[illegible]	[illegible]	160	[illegible]
NEW CANAAN WPCF	04/06/2010	CT0101273	1.6	1	2.3	3.3	44	
NEW CANAAN WPCF	04/13/2010	CT0101273	1.2	1.3	2.9	4.2	42	
NEW CANAAN WPCF	04/20/2010	CT0101273	1	1.9	0.9	2.8	23	
NEW CANAAN WPCF	04/27/2010	CT0101273	1	1.5	0.4	1.9	16	3.1
NEW CANAAN WPCF	05/04/2010	CT0101273	1.1	1.2	0.5	1.7	16	
NEW CANAAN WPCF	05/11/2010	CT0101273	0.9	1.9	0	1.9	14	
NEW CANAAN WPCF	05/18/2010	CT0101273	0.9	1.7	0.4	2.1	16	
NEW CANAAN WPCF	05/25/2010	CT0101273	0.9	1.8	0.7	2.5	19	2.1
NEW CANAAN WPCF	06/01/2010	CT0101273	0.7	1.2	0.5	1.7	10	
NEW CANAAN WPCF	06/08/2010	CT0101273	0.8	2	0.5	2.5	17	
NEW CANAAN WPCF	06/15/2010	CT0101273	0.8	2.2	0.9	3.1	21	
NEW CANAAN WPCF	06/22/2010	CT0101273	0.8	2.2	0.7	2.9	19	
NEW CANAAN WPCF	06/29/2010	CT0101273	0.7	1.9	0.6	2.5	14	2.5
NEW CANAAN WPCF	07/06/2010	CT0101273	0.6	1.4	0.8	2.2	11	
NEW CANAAN WPCF	07/13/2010	CT0101273	0.7	1.2	0.7	1.9	11	
NEW CANAAN WPCF	07/20/2010	CT0101273	0.8	1.5	0.5	2	13	
NEW CANAAN WPCF	07/27/2010	CT0101273	0.8	1.4	0.8	2.2	15	2.1
NEW CANAAN WPCF	08/03/2010	ct0101273	0.7	1.4	0.6	2	12	
NEW CANAAN WPCF	08/10/2010	ct0101273	0.7	1.4	0.4	1.8	11	
NEW CANAAN WPCF	08/17/2010	ct0101273	0.7	1.1	0.3	1.4	8	
NEW CANAAN WPCF	08/24/2010	ct0101273	0.8	1.5	1	2.5	17	
NEW CANAAN WPCF	08/31/2010	ct0101273	0.7	1.4	1.4	2.8	16	2.1
NEW CANAAN WPCF	09/07/2010	CT0101273	0.6	1.7	1.2	2.9	15	
NEW CANAAN WPCF	09/14/2010	CT0101273	0.6	1.9	1.2	3.1	16	
NEW CANAAN WPCF	09/21/2010	CT0101273	0.7	2.2	0.9	3.1	18	
NEW CANAAN WPCF	09/28/2010	CT0101273	0.8	1.8	0.5	2.3	15	2.9
NEW CANAAN WPCF	10/05/2010	CT0101273	1	1.5	1.7	3.2	27	
NEW CANAAN WPCF	10/12/2010	CT0101273	0.8	1.2	0.6	1.8	12	
NEW CANAAN WPCF	10/19/2010	CT0101273	0.9	1.4	1	2.4	18	
NEW CANAAN WPCF	10/26/2010	CT0101273	0.8	2.1	0.5	2.6	17	2.5
NEW CANAAN WPCF	11/02/2010	CT0101273	0.74	1.8	1.2	3	19	
NEW CANAAN WPCF	11/09/2010	CT0101273	0.9	1.5	1.6	3.1	23	
NEW CANAAN WPCF	11/16/2010	CT0101273	0.8	1.9	1.1	3	20	
NEW CANAAN WPCF	11/23/2010	CT0101273	0.9	2.2	1.1	3.3	25	
NEW CANAAN WPCF	11/30/2010	CT0101273	0.8	1.7	1.3	3	20	3.1
NEW CANAAN WPCF	12/07/2010	CT0101273	0.9	2.2	1.9	4.1	31	
NEW CANAAN WPCF	12/14/2010	CT0101273	1.5	2.5	2.6	5.1	64	
NEW CANAAN WPCF	12/21/2010	CT0101273	1	2.2	1.6	3.8	32	
NEW CANAAN WPCF	12/28/2010	CT0101273	0.9	2.1	[illegible]	3.9	[illegible]	4.2
					Average	3.0843	28.56062745	3.1
					April - October	2.44	17.43333333	2.4
					Max	5.4	160	4.8
					Min	1.4	8	2.1
					Max (Apr-Oct)	4.2	44	3.1
					Min (Apr-Oct)	1.4	8	2.1

Plant	Date	Permit	FlowMGD	FTKN	NO2NO3	TN	TNMASS	TN Monthly Average
MILFORD HOUSATONIC WPCF	01/12/2010	CT0101656	6.4	3.2	3.4	6.6	352	
MILFORD HOUSATONIC WPCF	01/19/2010	CT0101656	6.4	1.8	2.3	4.1	219	
MILFORD HOUSATONIC WPCF	01/26/2010	CT0101656	5.1	1.5	1.8	3.3	140	4.7
MILFORD HOUSATONIC WPCF	02/02/2010	CT0101656	6	1.4	2.2	3.6	180	
MILFORD HOUSATONIC WPCF	02/09/2010	CT0101656	5.5	2.1	2.8	4.9	225	
MILFORD HOUSATONIC WPCF	02/16/2010	CT0101656	5.3	2.5	2	4.5	199	
MILFORD HOUSATONIC WPCF	02/23/2010	CT0101656	5.5	1.5	1.7	3.2	147	4.1
MILFORD HOUSATONIC WPCF	03/02/2010	CT0101656	11	5.1	2.4	7.5	688	
MILFORD HOUSATONIC WPCF	03/09/2010	CT0101656	7.9	4.9	1.5	6.4	422	
MILFORD HOUSATONIC WPCF	03/16/2010	CT0101656	16.1	3.3	3.6	6.9	927	
MILFORD HOUSATONIC WPCF	03/23/2010	CT0101656	7.5	3.9	1	4.9	307	
MILFORD HOUSATONIC WPCF	03/30/2010	CT0101656	15.3	2	2.2	4.2	536	6.0
MILFORD HOUSATONIC WPCF	04/06/2010	CT0101656	10.61	2.6	2.3	4.9	434	
MILFORD HOUSATONIC WPCF	04/13/2010	CT0101656	7.74	1.7	2	3.7	239	
MILFORD HOUSATONIC WPCF	04/20/2010	CT0101656	6.61	1.5	2.4	3.9	215	
MILFORD HOUSATONIC WPCF	04/27/2010	CT0101656	7.6	1.4	2.2	3.6	228	4.0
MILFORD HOUSATONIC WPCF	05/04/2010	CT0101656	6.7	1.6	3	4.6	257	
MILFORD HOUSATONIC WPCF	05/11/2010	CT0101656	5.7	1.4	3.7	5.1	242	
MILFORD HOUSATONIC WPCF	05/18/2010	CT0101656	4.5	1.4	4.1	5.5	206	
MILFORD HOUSATONIC WPCF	05/25/2010	CT0101656	6.1	1.5	3.6	5.1	259	5.1
MILFORD HOUSATONIC WPCF	06/01/2010	CT0101656	5.6	1.4	3.6	5	234	
MILFORD HOUSATONIC WPCF	06/08/2010	CT0101656	5	1.7	3	4.7	196	
MILFORD HOUSATONIC WPCF	06/15/2010	CT0101656	5.5	2.4	3.5	5.9	271	
MILFORD HOUSATONIC WPCF	06/22/2010	CT0101656	4.9	2	1.6	3.6	147	
MILFORD HOUSATONIC WPCF	06/29/2010	CT0101656	5.1	2.9	1.4	4.3	183	4.7
MILFORD HOUSATONIC WPCF	07/06/2010	CT0101656	5.1	1.6	3.2	4.8	204	
MILFORD HOUSATONIC WPCF	07/13/2010	CT0101656	4.6	1.9	3.3	5.2	200	
MILFORD HOUSATONIC WPCF	07/20/2010	CT0101656	5.2	1.6	3.2	4.8	208	
MILFORD HOUSATONIC WPCF	07/27/2010	CT0101656	4.9	1.5	3.1	4.6	188	4.9
MILFORD HOUSATONIC WPCF	08/03/2010	CT0101656	5.1	1.3	2.8	4.1	174	
MILFORD HOUSATONIC WPCF	08/10/2010	CT0101656	4.6	1.7	3.2	4.9	188	
MILFORD HOUSATONIC WPCF	08/17/2010	CT0101656	4.6	1.9	1.9	3.8	146	
MILFORD HOUSATONIC WPCF	08/24/2010	CT0101656	5.2	1.5	2.4	3.9	169	
MILFORD HOUSATONIC WPCF	08/30/2010	CT0101656	4.6	1.4	2.5	3.9	150	4.1
MILFORD HOUSATONIC WPCF	09/07/2010	CT0101656	4.5	2.1	2.3	4.4	165	
MILFORD HOUSATONIC WPCF	09/14/2010	CT0101656	4.2	2.3	1.4	3.7	130	
MILFORD HOUSATONIC WPCF	09/21/2010	CT0101656	4.2	2.5	1.6	4.1	144	
MILFORD HOUSATONIC WPCF	09/28/2010	CT0101656	4.5	2.1	2.2	4.3	161	4.1
MILFORD HOUSATONIC WPCF	10/05/2010	ct0101656	4.9	2.2	0.5	2.7	110	
MILFORD HOUSATONIC WPCF	10/12/2010	ct0101656	3.6	1.3	1.8	3.1	93	
MILFORD HOUSATONIC WPCF	10/19/2010	ct0101656	4.5	2.3	1.9	4.2	158	
MILFORD HOUSATONIC WPCF	10/26/2010	ct0101656	4.6	1.8	3.6	5.4	207	3.9
MILFORD HOUSATONIC WPCF	11/02/2010	CT0101656	4.5	4.9	2.7	7.6	285	
MILFORD HOUSATONIC WPCF	11/09/2010	CT0101656	4.9	5.3	1.4	6.7	274	
MILFORD HOUSATONIC WPCF	11/16/2010	CT0101656	4.7	7.3	1.2	8.5	333	
MILFORD HOUSATONIC WPCF	11/23/2010	CT0101656	4.9	2.6	1.5	4.1	168	
MILFORD HOUSATONIC WPCF	11/30/2010	CT0101656	4.6	2	3.2	5.2	200	6.4
MILFORD HOUSATONIC WPCF	12/07/2010	CT0101656	5.1	2.3	1.8	4.1	174	
MILFORD HOUSATONIC WPCF	12/14/2010	CT0101656	7	2.2	1.8	4	234	
MILFORD HOUSATONIC WPCF	12/21/2010	CT0101656	5.6	2.2	1.8	4	187	
MILFORD HOUSATONIC WPCF	12/28/2010	CT0101656	6	5.2	1.4	6.6	330	4.7
					Average	4.7586	243.7043137	4.7
					April - October	4.3933	196.8666667	4.4
					Max	8.5	927	6.4
					Min	2.7	93	3.9
					Max (Apr-Oct)	5.9	434	5.1
					Min (Apr-Oct)	2.7	93	3.9

Attachment 1.C Calculation of Allowable Total Nitrogen Load/Concentration
Using June-August 2004-2006 Data

I. DATA

Taunton River Flow at Bridgewater Gauge (CFS)	Estimated Taunton River Flow at Mouth (CFS)	Three Mile River Flow at North Dighton Gauge (CFS)	Three Mile River Flow at Mouth (CFS)	Segreganset River Flow at Dighton Gauge (CFS)	Segreganset River Flow at Mouth (CFS)	Assonet River based on Segreganset (CFS)	QuequechanRiver based on Segreganset (CFS)	Total Fresh Water Flow (CFS)
417.3	655.5	129.6	131.1	14.9	20.9	30.8	42.9	881.3

II. Calculations

Salinity	18.7	ppt	(from 2007 SMAST report)
Ocean Flow	1458.4	CFS	
Target N Conc.	0.45	mg/l	
Target N Load	5672.4	lb/day	
N Conc. At Sea Boundary.	0.28	mg/l	
Ocean N Load	2200.0	lb/day	
Allowable Load from Watershed Sources	3472.3	lb/day	
Actual Load from Watershed Sources	4,228	lb/day	(EPA)
Required Load Reduction	755.7	lb/day	
Required Percent Reduction	17.9	percent	
Non Point Source Load	1428.0	lb/day	(EPA)
Assumed reduction from non-point sources	20	percent	
Available load for Wastewater Discharges	2329.9	lb/day	
Uniform N Concentration	8.8	mg/l	

Note: 🟨 Calculated Value

Attachment 1.D - Census Income and Population Data

ID / EJ Class	ID	Total Population	Sewered Population[3]	Total Household Units	Est. Sewered Housing Units	Median Household Income	Total Area	Sewered Area[4]	Area Ratio	Percent Area Sewered
6139.02		4,751	4,225	1,993	1,772	$50,658	2,049	1,822	0.89	89
Language	-1	1,328	1,328	591	591	$45,110	81	81	1.00	100
6139.01		2,801	2,801	1,363	1,363	$34,838	314	314	1.00	100
6141.01		6,833	5,927	2,861	2,482	$55,280	2,812	2,439	0.87	87
6131		6,327	2,837	2,773	1,243	$58,272	6,805	3,051	0.45	45
6137		4,369	4,369	2,005	2,005	$53,433	883	883	1.00	100
Income	-2	1,916	1,916	982	982	$39,632	20	20	1.00	100
6138		4,519	4,519	2,464	2,464	$32,906	429	429	1.00	100
Income	-1	662	662	321	321	$21,440	10	10	1.00	100
Minority Income	-2	1,750	1,750	882	882	$23,730	11	11	1.00	100
Minority Income	-3	946	946	513	513	$38,542	7	7	1.00	100
Minority Income	-4	1,161	1,161	748	748	$26,321	12	12	1.00	100
6136		3,771	3,771	1,732	1,732	$37,024	485	485	1.00	100
Income	-2	1,712	1,712	748	748	$29,956	32	32	1.00	100
6134		3,812	1,173	1,436	442	$68,015	3,931	1,210	0.31	31
Minority	-2	1,658	1,658	640	640	$50,851	44	44	1.00	100
6133		7,201	906	2,738	345	$81,422	5,533	696	0.13	13
6140		4,472	4,472	1,956	1,956	$40,231	487	487	1.00	100
Minority Income	-1	2,073	2,073	877	877	$21,833	23	23	1.00	100
6141.02		7,018	2,250	2,575	826	$79,897	7,255	2,326	0.32	32
Totals		55,874	37,251	23,896	16,630	--	30,983	14,143	--	--

🟨 Environmental Justice Area contained within Tract

Median Household Income[1]	53401
Sewered Household Income[2]	48230

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[1] The median household income for the City of Taunton is from the 2007-2011 American Community Survey 5-Year Estimates

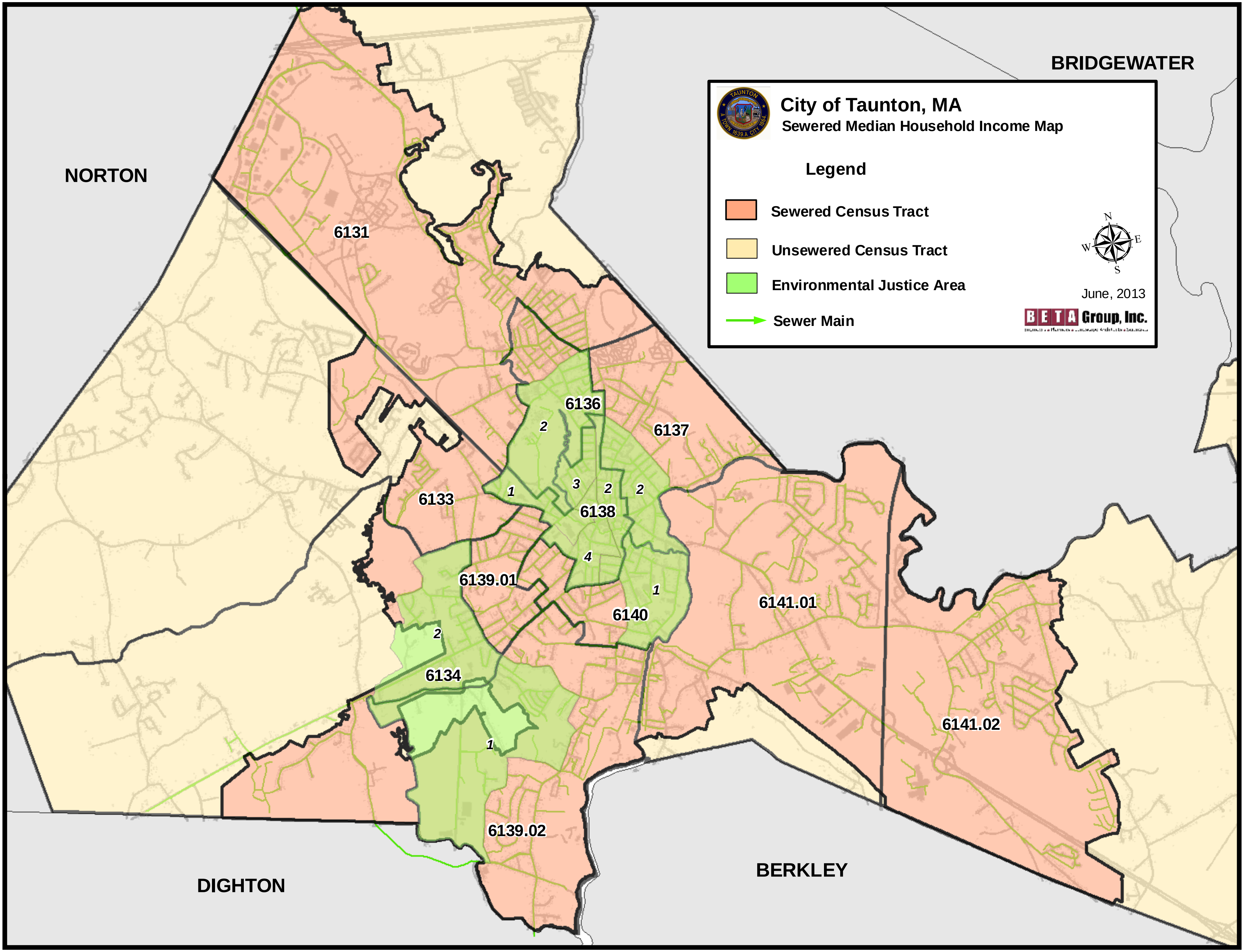
[2] The sewered household income is the weighted average based on the median household income and estimated sewered housing units for the U.S. Census Tracts

[3] Sewered population is the product of the percent sewered area and the total population

[4] Sewered area is the area of the Tract which is within the city of Taunton's sewered sub areas. Sub areas were generated by BETA Group, Inc.

[5] The Total Environmental Justice Population is 13206

Source: U.S. Census Bureau | American FactFinder



City of Taunton, MA

Sewered Median Household Income Map

Legend

🟧 Sewered Census Tract

🟨 Unsewered Census Tract

🟩 Environmental Justice Area

→ Sewer Main

June, 2013

BETA Group, Inc.

UBWPAD

UPPER BLACKSTONE
WATER POLLUTION ABATEMENT DISTRICT

Engineer Director / Treasurer Karla H. Sangrey, P.E.

April 18, 2013

Via E-Mail and Hand Delivery

Ken Moraff, Acting Director - moraff.ken@epa.gov
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Re: Comments of the Upper Blackstone Water Pollution Abatement District on the "Co-Permittee" Provisions of the Draft NPDES Permit No. MA0100897 Issued to The City of Taunton

Dear Messrs. Moraff and Ferris:

The Upper Blackstone Water Pollution Abatement District (the "District") hereby comments on the co-permittee provisions of the draft National Pollution Discharge Elimination System ("NPDES") Permit No. MA0100897 issued on March 20, 2013 to The City of Taunton, for discharges from the Taunton Wastewater Treatment Plant ("Taunton"). The draft permit names the Towns of Raynham and Dighton (the "Towns") as co-permittees "for specific activities required in Sections I.B - Unauthorized Discharges and I.C - Operation and Maintenance of the Sewer System, which include conditions regarding the operation and maintenance of the collection systems owned and operated by the Towns."

The District was a party to, and challenged similar co-permittee provisions in its NPDES permit, in the matter of *Upper Blackstone Water Pollution Abatement District*, NPDES Appeal Nos. 08-11 to 08-18 & 09-04, 14 E.A.D. (*Order denying review in part and remanding in part*, EAB, May 28, 2010 ("*Upper Blackstone* EAB Remand Order") in which the U.S. EPA Environmental Appeals Board ("EAB") remanded to Region 1 permit provisions that sought to regulate sewer lines owned, operated and maintained by separate municipalities as "co-permittees." In the *Upper Blackstone* EAB Remand Order, the EAB found that "[t]he Region has not sufficiently articulated in the record of this proceeding a rule-of-decision, or interpretation, identifying the statutory and regulatory basis for expanding the scope of NPDES authority beyond the treatment plant owner and operator to separately owned and operated collections systems that discharge to the treatment plant." Remand Order, at 18.

In the draft permit issued to Taunton, the Region again fails to identify a legal basis for its position that it has authority to regulate the Towns as co-permittees. While the draft Taunton permit fact sheet and document entitled *Analysis Supporting EPA Region I NPDES Permitting Approach for Publicly Owned Treatment Works that include Municipal Satellite Sewage Collection Systems* ("Region I's Analysis") seeks to respond to questions raised by the EAB in the Remand Order concerning EPA's legal authority to regulate separately owned municipal collection systems, the Region simply sets forth a series of old and new arguments to justify the regulatory position it previously staked out: that satellite systems can be included in the POTW permit. At footnote 10 of Region I's Analysis, the Region acknowledges that its "position differs from that taken by the Region in the *Upper Blackstone* litigation. There, the Region stated that the treatment plant was the discharging entity for regulatory purposes." Now, according to the Region, it "has clarified this view upon further consideration of the statute, EPA's own regulations and case law and determined that a municipal satellite collection system in a POTW is a discharging entity for regulatory purposes."

The Region makes this change with no basis to justify it. In the *Upper Blackstone* matter, and before the EAB, the satellite collection systems were not "discharging," but the Region could nonetheless regulate them. In the face of EAB's rejection of this argument, and in light of the Region's "clarified view," the Region now says satellite collection systems are "dischargers."

The Region's explanation for its change in position is insufficient and contrary to law. "[A]n agency changing its course must supply a reasoned analysis." Motor Vehicle Manufacturers Association v. State Farm Mutual Automobile Insurance Co., 463 U.S. 29, 57 (1983). In Region I's Analysis, it says only that it has "clarified [its] view." The Region, however, must "explain the evidence which is available" supporting that change and "must offer a 'rationale connection between the facts found and the choice made."' Id. 52. The Region does not, and cannot, identify new evidence or facts. The discharge point, at Outfall 001, has not changed. The owners or operators of the POTW and satellite collection systems have not changed.

In sum, the fact sheet and the Region I's Analysis fail to demonstrate that EPA has legal authority under the Clean Water Act ("CWA") or any NPDES regulation or sound factual basis to include the Towns as "co-permittees" to a NPDES permit. For the reasons set forth in this letter, EPA should strike the co-permittee provisions from the draft Taunton permit.

In Section III, Legal Authority, of its Analysis, EPA seeks to justify the imposition of co-permittee requirements upon the Towns based upon the definition of "publicly owned treatment works" or "POTW." Citing to the broad definition of "POTW" which includes the term "sewage collection systems," EPA contends that a POTW includes not only the treatment works, owned and operated by Taunton, but also the miles of sewers, pipes, equipment, and other systems owned, operated and maintained by the Towns Based on the definition of POTW at 40 CFR 122.2, EPA concludes,

> . . . a satellite collection system owned by one municipality that transports municipal sewage to another portion of the POTW owned by another municipality can be classified as part of a single POTW system discharging to waters of the U.S.

Analysis, p. 10.

> Under this approach, the POTW in its entirety will be subject to NPDES regulation as a point source discharger under the Act.

Attachment I, p. I

Missing from EPA's Analysis is any acknowledgement of or reference to the operative terms of the CWA that trigger NPDES permitting: "discharge of any pollutant by any person" from a point source. CWA § 301(a). It is the act of discharging a pollutant from a point source that gives rise to NPDES permitting. The ownership of a collection system, as part of a greater POTW, does not require a NPDES permit under the CWA. The Towns' collection systems have no point source. The Towns do not own, operate or control any point source. Instead, the Towns send waste water to a separately owned treatment plant for treatment and discharge at a point source.

Taunton, not any Town, is a person who discharges from a point source. Consequently, the reach of EPA's authority to regulate "dischargers" is limited to Taunton.

The CWA at Section 301(a) provides that "except in compliance [with a NPDES Permit] the discharge of any pollutant by any person shall be unlawful." The term "discharge of a pollutant" means "any addition of any pollutant to navigable waters from any point source." CWA § 502(12). The CWA authorizes EPA to "issue a permit for the discharge of any pollutant." CWA § 402(a)(1). Thus, under the CWA it is only those persons who discharge a pollutant from any point source to navigable waters who are subject to NPDES permitting requirements. CWA § 502(14) (defining point source as "any discernable, confined and discreet conveyance . . . from which pollutants are . . . discharged").

EPA incorrectly states that the "NPDES regulations . . . identify the "POTW" as the entity subject to regulation," citing to 40 C.F.R. § 122.21(a). Analysis, p. 8. The "entity" subject to regulation is the "person who discharges or proposes to discharge." 40 C.F.R. § 122.21(a)(1). Such persons are required make application for a permit and "[a]pplicants for new or exiting POTWs must submit information required" by 40 C.F.R. § 122.21(j), using Form 2A. 40 C.F.R. § 122.21(a)(2)(B).

EPA says "[w]hen a municipal satellite collection system conveys wastewater to the POTW treatment plant, the scope of NPDES authority extends to both the owner/operators of the treatment facility and the municipal satellite collection system, because the POTW is discharging pollutants. Analysis, p. 8. According to the permit, at Part I. A.1., "the permittee [*i.e.* Taunton] is authorized to discharge treated industrial and sanitary wastewater from outfall serial number 001 to the Taunton River," and at B, "[t]his permit authorizes discharge only from the outfall listed in Part I. A.1." The Towns do not own or operate outfall 001.

The Towns are not persons who discharge from a point source. The Towns do not "discharge a pollutant" as the term is defined under CWA. No doubt, the Towns "discharge" – as that term commonly used – wastewater via conveyance systems to a point source. The CWA, however, is specific: persons who discharge pollutants from a point source need a NPDES permit to do so. The Towns have no "direct discharge." See 40 CFR 122.2 (defining "direct discharge" to mean "discharge of a pollutant").

At footnote 12 of the Analysis, EPA states that some municipal satellite collection systems have erroneously "argued that the addition of pollutants to waters of the United States from pipes, sewer or other conveyances that go to a treatment plant are not a "discharge of a pollutant" under 40 CFR § 122.2." See 40 CFR 122.2 (persons who "discharge[] through pipes, sewers, or other conveyances owned by a . . . municipality which do not lead to a treatment works" are persons who "discharge of a pollutant" under 40 CFR 122.2. (emphasis supplied)). In support of this position, EPA says that there is "[o]nly one category of such discharges . . . excluded: indirect discharges" and that "the satellite system discharges at issue here are not indirect discharges." While it is true that the definition of "discharge of a pollutant" at 40 CFR 122.2 excludes pollutants from "indirect discharges," that does not mean that only "indirect dischargers" fall outside the scope of "discharge of a pollutant" or that an interpretation of the definition of "discharge of a pollutant" which excludes wastewater from separately owned collection systems to a treatment plant is not reasonable in light of the definition of other terms, described above, that require permitting from point sources. The use of the term "treatment works" as it appears in the regulatory definition of "discharge of a pollutant" does not preclude this interpretation.

EPA seeks to conflate the term "discharge" used in "discharge of a pollutant" with the "transfer of flow" or "conveyance" from a municipal conveyance system to the POTW treatment plant or works that has a point source "from which pollutants are discharged." The word "discharge" is a defined term: "when used without qualification [it] means the "discharge of a pollutant." 40 CFR 122.2. There is no "discharge" from a municipal conveyance system. And in this case, there is but one discharge point from a POTW. See draft permit Part I. A. 1. and B. It is that point source "from which pollutants are discharged" that triggers NPDES permitting, and only those persons who own or operate that point source are subject to such permitting. That point source is not owned by the Towns. In short, the jurisdictional reach under the CWA does not include persons, such as the Towns that own, operate and maintain sewer lines, that provide a conveyance for waste waters for treatment and discharge by another person from its point source.

The Region's rationale for seeking to impose co-permittee requirements upon the Towns is not consistent with the references to "municipality" in the definition of POTW found at 40 C.F.R. § 403.3(q), and the definition's statement that "[t]he term also means the municipality . . . which has jurisdiction over the Indirect Discharges to and the discharges from such a treatment works." The final sentence of the regulatory definition of POTW in the pretreatment regulations at 40 C.F.R. § 403.3(q), refers to municipalities that have "jurisdiction over . . . the discharges from such a treatment works." The term "municipality" as defined in CWA § 502(4) "means a city, town, borough, county, parish, district, association, or other public body created by or pursuant to State law and having jurisdiction over disposal of sewage, industrial wastes, or other wastes…." (emphasis supplied). The Towns have jurisdiction over only their collection systems. They have no jurisdiction over the treatment plant or point source of discharge. Thus, the Region's view that a satellite collection system is part of a POTW is inconsistent with the final sentence of the regulatory definition of POTW in the pretreatment regulations. **That that sentence provides that "POTW" may "also" mean a municipality has no bearing on this limitation.**

The absence of EPA authority to make the Towns co-permittees is borne out by the permitting process and EPA's regulations at 40 CFR § 122.21, Subpart B, Permit Application Requirements. 40 CFR § 122.21(a), entitled "Duty to Apply," provides that "[a]ny person who discharges or proposes to discharge pollutants . . . must submit a complete application . . . in accordance with this section [122.21] and part 124 of this chapter." 40 CFR § 122.21(a)(i). (emphasis supplied). Consistent with the CWA, EPA regulations require persons "who discharge pollutants" have a NPDES Permit. See CWA § 301(a)("except in compliance with this section and [other sections] of this title, the discharge of any pollutant by any person shall be unlawful"), and CWA § 402(a)(authorizing EPA to issue a permit "for the discharge of any pollutant"). Throughout, the permit application regulations at 40 CFR § 122.21 contemplate that it is the "person" who discharges pollutants who must obtain a NPDES Permit. No where in 40 CFR § 122.21 is there any reference to "co-permittee" or any suggestion that separately owned and operated conveyance systems are subject to NPDES permitting. Consistent with CWA, it is the person who discharges a pollutant from a point source who is subject to NPDES permitting requirements.

While 40 CFR § 122.21(a)(1) requires an application only from those persons who discharge from a point source, the regulations anticipate circumstances when a facility may be owned or operated by separate entities. The permit application regulations provide that "[w]hen a facility or activity is owned by one person but is operated by another person, it is the operator's duty to obtain a permit." 40 CFR § 122.21(b). Thus, it is operator of the **"point source" that must have the permit. "Owner or operator" means "the owner or operator of any "facility or activity" subject to regulation under the NPDES program." 40 CFR § 122.2. "Facility or activity" means "any NPDES "point source" or any other facility or activity (including land or appurtenances thereto) that is subject to regulation under the NPDES program." 40 CFR § 122.2. (emphasis supplied).**

Nothing in 40 CFR § 122.21 requires or suggests that "satellite collection systems" need to make application for a NPDES permit. While the regulations contemplate that "[m]ore than one application form may be required from a facility," multiple applications are only required where there may be multiple point sources, not multiple owned parts of a POTW. See, 40 CFR § 122.21(a)(2)(i) ("More than one application form may be required from a facility depending on the number and types of discharges or outfalls found there."). Again, the regulations require persons who discharge from point sources to have the NPDES permit.

Nowhere in Application Form 2A is there any reference to a "co-permittee" or suggestion that a person may make application, with a treatment works applicant, as co-permittee. See http://www.epa.gov/npdes/pubs/final2a.pdf. At page 1 of 21 of Form 2A, applicants "must complete questions A.8. through A.8. A treatment works that discharges effluent to surface waters of the United States must also answer questions A.9. through A.12." Part A.1 through A.8. of Form 2A asks for information about the facility and applicant, and asks "is the applicant the owner or operator (or both) of the treatment works?" (A.1., A.2.). Form 2A asks for collection system information; specifically, "information on municipalities and areas served by the facility . . . type of collection system (combined vs. separate) and its ownership (municipal, private, etc.)." (A.4.). Form 2A asks for information about the "collection system(s) used by the treatment plant." (A.7.). If the NPDES regulations contemplated permitting of collection systems, one would expect to see in each of these parts of the NPDES Application Form 2A some reference to the owners or operators of collection systems as "co-permittees." There is none. Form 2A also requires information on discharges. At Part A.8.a., Form 2A asks "Does the treatment works discharge effluent to waters of the U.S.? ___ Yes ___ No." Form 2A obviously contemplates "discharges" from a "treatment works," not a POTW. Finally, at Part A. 1.8.a.(i)-(v), Form 2A seeks information on the "types of discharge points the treatment works uses." No "collection system" or "satellite collection system" is listed here. This should be no surprise; collection systems and satellite collection systems do not have "discharge points" under the NPDES regulations.

In its Analysis, EPA would "waive" the Towns' permit applications and all requirements of 40 CFR § 122.21. In its effort to justify including the Towns as co-permittees, EPA both misapplies and takes 40 CFR § 122.21(j) entirely out of context. First, waivers can only be granted to those persons who have submitted applications. Nothing in the fact sheet suggests that the Towns applied for any NPDES permit. § 122.21(j) provides that:

> Permit applicants <u>must submit</u> all information available at the time of permit application. . . . The Director may waive any requirement <u>of this paragraph</u> if he or she has access to substantially identical information. (emphasis supplied).

40 CFR § 122.21(j) does not support the EPA's proposed waiver of any application by the Towns; it allows only for the waiver of certain information in a permit application submitted by the applicant.

Second, EPA can not unilaterally waive requirements of an application without a request to do so; the person must seek a waiver and that waiver must be approved by EPA. 40 CFR § 122.21(e) requires a compete application before EPA may issue a permit "([EPA] shall not issue a permit before receiving a complete application for a permit"), and a "waiver application" must be made, and approved, or not acted upon by EPA. 40 CFR § 122.21(e)(2) provides:

> A permit application shall not be considered complete if a permitting authority has waived application requirements under paragraphs (j) or (q) of this section and EPA has disapproved the waiver application. If a waiver request has been submitted to EPA more than 210 days prior to permit expiration and EPA has not disapproved the waiver application 181 days prior to permit expiration, the permit application lacking the information subject to the waiver application shall be considered complete.

Nothing in the fact sheet suggests that the Towns have made application for a waiver from the application requirements. 40 CFR § 122.21(j) says only that the "Director may waive any requirement of this paragraph if he or she has access to substantially identical information." This provision, in context, is obviously designed to allow waiver of some of the detailed and often duplicate information required under Section 122.21 and in EPA's permit application forms. As noted above, Form 2A consists of 21 pages and requires detailed information about the "treatment works." See Form 2A at <u>http://www.epa.gov/npdes/pubs/final2a.pdf</u>. Nothing in Section 122.21(j) suggests EPA may waive the requirement at 40 CFR § 122.21(a)(1) mandating an application from those persons who discharge from a point source. Likewise, nothing in Section 122.21(j) suggests EPA may waive the requirement for application signatures and certifications and authorizations required by 40 CFR § 122.22, none of which the Towns have provided. EPA seeks to ignore its own regulations and to issue a permit the Towns who have not applied for an NPDES permit.

EPA would further seek to cause the Towns to "consult and coordinate with the regional POTW treatment plant operators to ensure that any information provided to EPA about their respective entities is accurate and complete." Exhibit C to Analysis. EPA would then use its authority, under CWA § 308, to compel information from the Towns, should EPA deem information provided by the permit applicant incomplete. CWA § 308, however, applies to "the owner or operator of any point source." CWA § 308(a) (A). Information may be obtained only from such owner or operator of the "point source," the "effluent source" or "the owner or operator of such source." CWA § 308(a)(B)(i) and (ii). Again, because the Towns do not own or operate any point source, CWA § 308 would not apply to them. Under EPA's Analysis, it would read out of the regulations the entire Section 122.21. EPA's cobbled approach and legal analysis toward finding authority where there is none is not supported by its own regulations.

Nothing in the EPA's permit writers' manual evidences any authority to permit satellite collection systems as part of a greater POTW. Indeed, EPA's permit writers' manual make no reference to permitting of satellite collection systems or to the owner of such systems being subject to a NPDES permit as a co-permittee. See EPA NPDES Permit Writers' Manual, September 2010 http://www.epa.gov/npdes/pubs/pwm_2010.pdf. Instead, the Permit Writers' Manual supports the analysis provided above. It says: "Under the national program, NPDES permits are issued only to direct dischargers." Permit Writers' Manual Section 1.3.4. (emphasis supplied). As noted above, a "direct discharge" means the "discharge of a pollutant" and "discharge of a pollutant" means "any addition of any pollutant to navigable waters from any point source." CWA § 502(12). 40 CFR 122.2.

Section 4.1 of Permit Writers' Manual addresses "Who Applies for a NPDES Permit?" No mention is made in this section to satellite collection systems or to the owners of such systems. Instead, the Permit Writers' Manual states:

> The NPDES regulations at Title 40 of the *Code of Federal Regulations* (CFR) 122.21(a) require that any person except persons covered by general permits under § 122.28, who discharges pollutants or proposes to discharge pollutants to waters of the United States must apply for a permit. Further, § 122.21(c) prohibits the permitting authority from issuing an individual permit until and unless a prospective discharger provided a complete application. This regulation is broadly inclusive and ties back to the Clean Water Act (CWA) section 301(a) provision that except as in compliance with the act, "...the discharge of any pollutant by any person shall be unlawful." In most instances, the permit applicant will be the owner (e.g., corporate officer) of the facility. However the regulations at § 122.21(b) require that when a facility or activity is owned by one person but is operated by another person, it is the operator's duty to obtain a permit. The regulations also require the application to be signed and certified by a high-ranking official of the business or activity. The signatory and certification requirements are at § 122.22. Permits (and applications) are required for most discharges or proposed discharges to waters of the United States; however, NPDES permits are not required for some activities as specified under the *Exclusions* provision in § 122.3.

Section 4.3. of the Permit Writers' Manual addresses what forms must be submitted and at Exhibit 4-3 describes "the types of dischargers required to submit NPDES application forms, identifies the forms that must be submitted, and references the corresponding NPDES regulatory citation." Again, in Section 4.3 there is no mention of satellite collection systems or need for the owners of such systems to have a NPDES permit.

EPA's position that the collection system is part of the POTW does not advance its argument that "satellite collection systems" should be deemed "co-permittees" in NPDES permits. If the collection system is part of the POTW, it should matter not who owns what part or portions as it is the "person" who owns or operates that portion of the POTW that "discharges a pollutant" from a point source who is required to have a permit for that discharge. EPA acknowledges that the Towns do not own or operate the entire POTW. While EPA seeks "to refashion permits issued to regionally integrated POTWs to include all owners/operators of the treatment works (*i.e.*, the regional centralized POTW treatment plant and the municipal satellite collection systems)," permit conditions "pertain only to the portions of the POTW collection system that the satellites own." Analysis, p. 7. See Permit I.1.C. Because the Towns do not own or operate the point source – Outfall 001 – they are not a person who may be subject to a NPDES permit.

While the Analysis addresses generic problems associated with municipal sanitary sewer collection systems, including SSO's and I/I, nothing in the fact sheet or Analysis indicates that SSO's or I/I are not being appropriately addressed by the Towns or is a problem that requires or calls for the Towns to be identified as a co-permittee in this permit, or that co-permittee status may advance any I/I or SSO problem. Exhibit B of the Analysis, entitled "Analysis of extraneous flow trends and SSO reporting for representative systems," has nothing to do with Taunton or the Towns. EPA improperly seeks to use information not material to Taunton or the Towns to justify imposition of co-permittee requirements.

Nor does the fact sheet or Analysis explain why operation and maintenance of the Towns' sewer systems are not being adequately regulated by under State regulations at 310 CMR 12.00. 312 CMR 12.02 defines "Sewer Systems" to mean "pipelines or conduits, pumping stations, force mains, and all other structures, devices, appurtenances, and facilities used for collecting and conveying wastes to a site or works for treatment or disposal." The purpose of 314 CMR 12.00 is to insure "proper operation and maintenance of . . . sewer systems within the Commonwealth," and sets forth numerous requirements for the proper operation and maintenance of such systems. See 314 CMR 12.03(4), (10), and (11); 12.04(4); 12.05(5), (6) and (12); and 12.07(7).

In its Determination on Remand issued to the District on July 7, 2010, the Region indicated it would "coordinate broadly within EPA in developing a response" to the *Upper Blackstone* EAB Remand Order. Nothing in Region 1's Analysis indicates this was done. Because EPA's authority to permit satellite collection systems impacts not only the Region, but is of national significance, and because the issues raised by the EAB concerning EPA's legal authority to regulate co-permittees were limited to those raised by the District, the Region's effort to permit satellite collection systems as co-permittees or otherwise through separate permits should be presented to the public for review and comment on a national level.

In June 2010, EPA did seek through "listening sessions" information from the public concerning permitting of satellite collection systems. See 75 Fed. Reg. 30395 (June 1, 2010) ("EPA is considering whether to propose modifying the [NPDES] regulations as they apply to municipal sanitary sewer collection systems"). In contemplating a potential regulatory change, EPA asked specifically for input on the question: *Should EPA propose to require permit coverage for municipal satellite collections systems?* Because EPA was "considering clarification of the framework for regulating municipal satellite collection systems under the NPDES program," and doing so via a regulatory change, the Region should not include at this time, and based on unsupported legal authority outlined above, the Towns as co-permittees in this permit. Until such time as EPA addresses this issue on a national level and gives the public the opportunity review and comment on the legal Analysis set forth by the Region, it should not include co-permittee provisions in this permit.

EPA's attempt to change the legal requirements applicable to satellite systems is a legislative rule that EPA is issuing without formal notice and comment rulemaking in violation of the Administrative Procedure Act ("APA"). In trying to distinguish between legislative rules and policy statements, courts have found that "if a document expresses a change in substantive law or policy the agency intends to make binding, or administers with binding effect, the agency may not rely upon the statutory exemption for policy statements, but must observe the APA's legislative rulemaking procedures." *Gen. Elec. Co. v. E.P.A.*, 290 F.3d 377,383-84 (D.C. Cir. 2002. *See also Appalachian Power Co. v. EPA*, 208 F.3d 1015 (D.C. Cir. 2000) (finding that an EPA guidance document that imposed new monitoring requirements relating to the operation of permit programs under the Clean Air Act was a legislative rule because it was treated as binding), *Nat'l Mining Ass'n v. Jackson*, 816 F. Supp. 2d 37, 42-49 (D.D.C. 2011) (finding a violation of the Administrative Procedure Act where EPA sought to impose a new process for obtaining section 404 permits without notice and comment rulemaking), *New Hope Power Co. v. U.S. Army Corps of Eng'rs*, 746 F. Supp. 2d. 1272, 1283-84 (S.D. Fla. 2010) (striking Corps guidance purporting to amend the prior converted croplands exclusion because it amounted to new legislative rules that created a binding norm and the Corps failed to comply with the APA).

In the case of the draft Taunton permit, there is no question that EPA intends its new position regarding satellite system to have binding effect. Moreover, it is telling that in 2001, EPA began a rulemaking that purported to give the agency direct authority over satellite systems, in the context of a propose rule pertaining to sanitary sewer systems. *See* National Pollutant Discharge Elimination System (NPDES) Permit Requirements for Municipal Sanitary Sewer Collection Systems, Municipal Satellite Collection Systems, and Sanitary Sewer

Overflows (proposal signed Jan. 4, 2001) (formerly available at http://cfpub.epa.gov/npdes/regresult.cfm?program_id=4&view=all&type=3, but now withdrawn from EPA's website). EPA later withdrew that proposed rule.

For these reasons, the co-permittee provisions of the draft Taunton permit should be stricken.

Very truly yours;
UPPER BLACKSTONE WATER
POLLUTION ABATEMENT DISTRICT

Karla H. Sangrey, P.E.
Engineer Director / Treasurer

C: The City of Taunton, Department of Public Works
Town of Raynham Sewer Department
Town of Dighton Sewer Department

Attachment 2: Comments Submitted by Hall & Associates on Behalf of the City of Taunton

City of Taunton
Comments on the Taunton Wastewater Treatment Plant Draft NPDES Permit
No. MA0100897

Prepared by
Hall & Associates
Washington, D.C.

Comments on the Taunton Wastewater Treatment Plant Draft NPDES Permit (MA0100897)

The draft effluent limitation for total nitrogen ("TN") is based on EPA‟s determination of a "protective" threshold nitrogen concentration for the Taunton River Estuary to preclude an impairment. The basis for this determination is presented in the Fact Sheet. (*See* Fact Sheet, at 12 – 34). Over these 23 pages, EPA presents an alleged impairment threshold of 0.45 mg/L TN, estimates the TN loads from point and non-point sources entering the receiving waters, and concludes that the Taunton Wastewater Treatment Facility ("WWTF") must meet the limits of technology (3 mg/L TN) to mitigate exceedances of the dissolved oxygen ("DO") water quality standard in the Taunton River Estuary and Mount Hope Bay.

The basis for the TN threshold determination is limited to a consideration of water quality monitoring data collected over a three year period (2004 – 2006) from a single location in Mount Hope Bay. EPA determined this threshold by identifying a location, outside the Taunton River Estuary, where water quality standards for DO are not violated in order to identify a nitrogen concentration consistent with unimpaired conditions. EPA asserts that this approach is consistent with EPA guidance regarding the use of reference conditions for the purposes of developing nutrient water quality criteria. (Fact Sheet, at 29). Based on an examination of the available data, EPA determined that Station MHB16 was an appropriate sentinel site because DO standards were met at this site. This site had a growing-season average total nitrogen concentration of 0.45 mg/L for the 2004-2005 period. Therefore, EPA selected 0.45 mg/L TN as the threshold protective of the dissolved oxygen water quality standard of 5.0 mg/L and claimed that the Taunton River Estuary must meet this same TN concentration at Station MHB19 to achieve compliance with the DO water quality standard.

Comments on the Legal/Regulatory Issues

The following provides comments on the legal/regulatory issues arising from the Region‟s proposed permit and fact sheet.

1. Organic enrichment is not a nutrient impairment designation, therefore, there is no demonstration that a nutrient requirement under 40 C.F.R. § 122.44(d) is triggered for the Taunton River.

In the Fact Sheet, the Region concludes that an organic enrichment impairment designation is equivalent to designating that waters as nutrient impaired. (Fact Sheet, at 19). Based on this assumption, the Region concludes that nutrients and chlorophyll a levels are excessive and that stringent TN reduction is needed to address low DO occurring in the estuary pursuant to 40 C.F.R. § 122.44(d).[1] However, the Region"s assessment addresses the wrong impairment in the draft permit; the Taunton River is impaired for *organic enrichment* which is *not* equivalent to a nutrient impairment. Because EPA has regulated an impairment that was not determined to exist by the agency that is given statutory authority to render such decisions (*i.e.,* MassDEP), EPA"s proposed permit limitations for TN should be withdrawn as it is inconsistent with the adopted, EPA-approved impairment listing.

a) EPA's action violates Clean Water Act ("CWA") procedures and requirements.

The Massachusetts 2010 § 303(d) list ("MA § 303(d) list" or "MA § 303(d) report") has the Taunton River, Segment MA62-02 listed as impaired due to pathogens.[2] The segments downstream of MA62-02 from the mouth of the River at the Braga Bridge in Fall River, are listed as impaired for pathogens and organic enrichment/low dissolved oxygen.[3] Further downstream, in Mount Hope Bay, a "nutrient" impairment is designated. An "organic enrichment" impairment designation is *not* equivalent to a "nutrient" impairment designation as evidence by MassDEP having two separate impairment designations for the pollutant causes. If MassDEP believes waters are "nutrient" impaired then such waters are designated as such. (*See, e.g.,* designations for certain sections of Mount Hope Bay). Thus, the state does not presently identify the Taunton Estuary as impaired by nutrients regardless of any potential "indicators"

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[1] *See* discussion on nutrients and chlorophyll a levels in DEP/SMAST Massachusetts Estuaries Project report, *Site-Specific Nitrogen Thresholds for Southeastern Massachusetts Embayments: Critical Indicators – Interim Report* (Howes *et. al.*, 2003) ("Critical Indicators Interim Report").

[2] Fact Sheet, at 4-5.

[3] *Id.*

discussed in the Critical Indicators Interim Report. It is clear, EPA has unilaterally amended the state‟s published, EPA-approved impairment designation via this permit action. EPA had the opportunity to follow specific statutory procedures (discussed below) to amend the Massachusetts impairment listing; however, no such action was ever undertaken by EPA. EPA never notified MassDEP that the impairment designation was in error as required by Section 303(d)(2). Thus, EPA‟s action violates the requirements of the Act regarding designation and determination of impairments and their causes.

b) EPA’s action is inconsistent with adopted state procedures for narrative criteria implementation.

As the MA § 303(d) report makes evident, “organic enrichment” is linked to low dissolved oxygen impairment instead of a nutrient impairment. (*See* MA § 303(d) report, at 15-16, Table listing Water Body System cause codes with the accompanying Assessment Database cause code and “organic enrichment/low DO” is paired with “[d]issolved oxygen saturation; oxygen, dissolved; and organic enrichment (sewage) biological indicators” while “nutrients” is paired with “nitrogen (total); phosphorus (total) and nutrient/eutrophication biological indicators”). There are no indications in the state‟s section 303(d) procedures that the low nutrient or chlorophyll a levels identified in the Critical Indicators Interim Report control whether or how organic enrichment designations are interpreted or nutrient impairment designations are rendered. According to Massachusetts impairment listing procedures, state waters are only identified as nutrient impaired where excessive algal growth *causes* DO related violations. *These procedures constitute the Department’s methodology for interpreting it narrative criteria with respect to nutrients. In determining that Taunton was nutrient impaired, EPA abandoned those procedures and created a new approach to identifying nutrient impairments, presuming that nitrogen levels were excessive. Specifically, EPA’s new approach assumed that elevated nutrients directly impair dissolved oxygen levels, which has no basis in state or federal law or the state’s published approach to evaluating nutrient impacts via its narrative standard. Thus, EPA’s action effectively amends existing state law, which is patently illegal.*[4]

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[4] *See, e.g., Iowa League of Cities v. EPA*, __ F.3d __, No. 11-3412, 2013 U.S. App. LEXIS 5933 (8th Cir. Mar. 25, 2013).

c) EPA failed to adhere to applicable statutory and regulatory requirements.

EPA‟s action compounds a series of legal and regulatory errors. EPA never adhered to its statutory responsibility of notifying Massachusetts and/or the public of its decision to reject the "organic enrichment" impairment determination made by the state and instead list the Taunton River as nutrient impaired. *See* 40 C.F.R. § 303(d)(2). Similarly, contrary to statutory procedures, EPA never notified Massachusetts or the public of its decision that Massachusetts‟ impairment identification procedures, as they pertain to nutrients, were insufficient or deficient in any matter. *Id.* Likewise, EPA never informed MassDEP that their application of state narrative criteria was misplaced and should instead allow for a *presumption*, rather than an actual demonstration, that nutrients are causing excessive algal growth or low DO based on the Critical Indicators Interim Report. This theory was specifically challenged by the New England Interstate Water Pollution Control Commission as technically flawed. (*See* Attachment A- the Commonwealth of Massachusetts is part of the New England Interstate Water Pollution Control Commission).

Under the CWA, EPA must review and either approve or disapprove a state‟s § 303(d) list. 33 U.S.C. § 1313(d)(2); 40 C.F.R. § 130.7(d)(2). If EPA disapproves the list, then it must, amongst other things, identify the deficiency and propose a proper revision. *Id.* EPA is only authorized to modify a state listing after it expressly disapproves of a state determination. *Id.* Therefore, in this case, if EPA believed that the Taunton River was impaired for nutrients it should have rejected the MA § 303(d) list. It is improper for EPA, after approving the MA § 303(d) list to later, in a draft NPDES permit, attempt to change an impairment listing by creating a water quality criterion for nutrients when the waters are impaired for organic enrichment/low dissolved oxygen. Likewise, if EPA disagreed with the MassDEP approach to narrative criteria implementation with respect to nutrients, EPA should have raised that objection pursuant to procedures under CWA Section 303(c). The Critical Indicators Interim report, cited by EPA as a basis to indicate the water quality that would constitute nutrient impairment, is not even referenced in the MassDEP 303(d) procedures for rendering nutrient impairment determinations.

Section 122.44(d) plainly indicates that state regulatory interpretation regarding narrative criteria compliance need to be respected (unless obviously incorrect). *See Kentucky Waterways Alliance v. Johnson*, 540 F.3d 493, 469 n.1 (6th Cir. 2008) ("In interpreting a state‟s water quality standard, ambiguities must be resolved by „consulting with the state and relying on authorized state interpretations."); *Marathon Oil Co. v. EPA*, 830 F.2d 1346, 1351-1352 (5th Cir. 1987) (EPA is merely an "interested observer" as to how a state interprets its WQS provisions); *American Paper Inst. v. EPA*, 996 F.2d 346, 351 (D.C. Cir. 1993) ("Of course, that does not mean that the language of a narrative criterion does not cabin the *permit writer's* authority at all; rather, *it is an acknowledgement that the writer will have to engage in some kind of interpretation to determine what chemical-specific numeric criteria--and thus what effluent limitations--are most consistent with the state's intent as evinced in its generic standard.*") (emphasis added)). EPA‟s entire permitting approach discards those technical and regulatory findings.

Adherence to the state‟s current procedures for confirming whether a nutrient impairment exists or that excessive algal growth is the cause of low DO readings is required by federal law. EPA has violated federal law and misapplied 40 C.F.R. § 122.44(d) by creating (or assuming) a nutrient impairment exists where one has not been determined to exist by the agency statutorily responsible for such determinations. *See, e.g., Ass'n of Pac. Fisheries v. EPA*, 615 F.2d 794, 811-812 (9th Cir. 1980) (As these records confirmed that EPA ignored the relevant information and "proceed[ed] upon assumptions that were entirely fictional or utterly without scientific support" EPA‟s action is not legally defensible). EPA has also violated federal law by substituting assumptions, unadopted numeric nutrient and chlorophyll a thresholds as the basis for presuming a nutrient impairment exists in Massachusetts waters to trigger permit requirements under § 122.44(d). (*See infra* note 9). As the NPDES regulations provide no such authority to EPA, this permit action must be withdrawn pending a demonstration that (1) algal growth levels are excessive and (2) such excessive plant growth is the cause of low DO conditions in the Taunton Estuary.

2. EPA provides no rational or substantive demonstration of a DO-related, nutrient impairment occurring in the Taunton River.

As noted above, state and federal law require a demonstration that the nutrient is in fact causing the impairment to demonstrated that more restrictive water quality based limitations are necessary. (*See e.g.,* CWA § 301(b)(1)(C) and 40 C.F.R. § 122.44(d) where both use the word "necessary" in authorizing the imposition of water quality-based limitations). The federal Administrative Procedure Act also requires technical conclusions to be based on substantial evidence.[5] EPA‟s Fact Sheet (at 26), simply concludes that excessive nutrients are the cause of DO impairments in the Taunton River. The entire analysis is nothing more than a series of unsupported assumptions that nowhere demonstrates that (1) the nutrients are causing excessive plant growth in the Taunton River or (2) that periodic low DO occurring in the Taunton Estuary is significantly related to algal growth and not some other factor unrelated to algal growth (*e.g.,* organic loadings from wastewater or CSO discharges known to exist in the system, periodic system stratification, natural deposition of organic materials from the watershed, or low DO entering the estuary from Mount Hope Bay). Without consideration of these conditions, it is simply impossible to determine whether or how nutrients could possibly be responsible for any low DO conditions.

a) Missing technical assessments preclude a determination that EPA's approach is rational and scientifically based.

Missing technical assessments needed to render a defensible permit evaluation include: (a) how TN affects algal growth in this part of the system; (b) how algal growth affects DO; (c) the form of nitrogen controlling plant growth; (d) where the algae found in the estuary are growing (upstream in fresh waters, in the Bay or in the tidal river); (e) the degree to which non-algal factors control DO in the system; (f) whether low DO is caused by SOD, diurnal DO variation or stratification; (g) how system hydrodynamics affect the occurrence of low DO; and (h) whether natural factors are responsible for the DO condition. Without such evaluations of these factors which are well documented as affecting DO of any tidal river, EPA‟s contention that nutrients

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[5] 5 U.S.C. § 706(2)(E); *see Citizens to Preserve Overton Park, Inc. v. Volpe*, 401 U.S. 402, 414 (1971) ("the agency action is to be set aside if the action was not supported by „substantial evidence.‟").

are the cause and, therefore, the solution to the DO condition is all presumption, pure speculation, and guesswork. In short, as there is no substantial evidence supporting this scientific conclusion and therefore is no objective way to know that it is scientifically correct, EPA"s proposed TN limitation is arbitrary and capricious.[6]

b) EPA's claim that an impairment exists without demonstrating causation violates federal and state law.

EPA"s approach (presuming a pollutant is causing a specific adverse ecological effect or causing a narrative criteria violation) is precisely what the CWA does not allow. *See* 40 C.F.R. § 131.11 (criteria determinations must be based on scientifically defensible information); 40 C.F.R. § 122.44(d) (demonstrating that limitations are necessary must be based on all available scientific information); *see also Natural Res. Def. Council v. EPA*, 16 F.3d 1395, 1398 (4th Cir. Va. 1993) ("The court agrees with EPA that its duty, under the CWA and the accompanying regulations, is to ensure that the underlying criteria which are used as the basis of a particular state"s water quality standard, are scientifically defensible . . ."); *Chem. Mfrs. Ass'n v. EPA,* 28 F.3d 1259, 1265 (D.C. Cir. 1994) (stating, when challenged, EPA must provide a "full analytical defense of its model" and show "there is a rational relationship between the model and the known behavior of the …pollutant to which it is applied."); *Columbia Falls Aluminum .v EPA,* 139 F. 3d 914, 923 (D.C. Cir 1998) (EPA "retains the duty to examine key assumptions as part of its affirmative burden of promulgating a non-arbitrary, non-capricious rule."). Likewise, EPA may not rely on a flawed or inaccurate study to render decisions under the Act. *Texas Oil & Gas Ass'n v. EPA*, 161 F. 3d 923, 935 (5th Cir. 1998). In this case as basic information is missing to determine that EPA"s approach is in fact necessary, the decision is *per se* flawed and unsupported.

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[6] As noted before, a central presumption of EPA"s effluent limit determination is that station MHB16 defines the level of nutrients (and presumably algal growth) that would be protective of the Taunton Estuary. *See supra,* at 1. It should be obvious to all that these open waters in a bay, highly influenced by the ocean, bear no objective resemblance to the physical setting occurring at Taunton River station (MHB19) where EPA chose to apply the Mount Hope Bay nutrient concentration. At a minimum, EPA would need to demonstrate that the conditions influencing TN dynamics and the DO regime at MBH16 are similar to the Taunton River site to support its position. No such demonstration is made because the physical conditions are radically different and there is no rational basis to believe that TN effects at MHB16 are similar in any way to TN effects at MHB19. Had EPA even conducted a cursory analysis it would have been obvious that (1) the algal growth in the Taunton River is *less than that occurring at MHB16* and (2) stratification, not algal growth, is the primary factor influencing DO levels in MHB16.

EPA decisions may not be based on "sheer guess work". *Leather Indus. of Am. v. EPA*, 40 F.3d 392, 408 (D.C. Cir. 1994) (citing *Am. Petroleum Inst.,* 665 F. 2d 1176, 1186-87 (D.C. Cir. 1981)). EPA may not regulate based on "probabilistic evidence" or "correlations" without proving causation. *Tex Tin Corp. v. EPA*, 992 F. 2d 353, 356 (D.C. Cir. 1993). Likewise, EPA may not claim that nitrogen is the cause of impairment in the Taunton River because it has caused impairment in other waters. The CWA and applicable state law require a site-specific demonstration of an impairment and its cause. (*See, e.g.*, § 303(d), 40 C.F.R. § 130; 314 CMR 4.05(5)(c)). Consequently, evidence that a TN level in a remote section of Mount Hope Bay is apparently not associated with DO violations at that location does not provide any credible evidence that the same TN level is necessary for the Taunton River, a physically distinct area. Without an assessment of the major factors known to affect DO in tidal estuaries and a demonstration of the degree to which TN is causing excessive algal growth and causing DO violation in the Taunton estuary, EPA"s approach is pure guesswork and therefore, arbitrary and capricious. *Leather Industries of Am., 40 F.3d 392.* Consequently, EPA lacks a credible, objective scientific basis for imposing the stringent TN limitations proposed in the draft NPDES permit.

3. EPA's approach is inconsistent with accepted scientific methods for assessing nutrient and DO impacts in flowing waters.

The Fact Sheet indicates that EPA chose an area of Mount Hope Bay that was meeting DO criteria as a "reference station" and simply presumed that whatever TN level that existed at that station would be the necessary TN level to be achieved in the Taunton River. (Fact Sheet, at 30). This was a form of truncated "stressor-response" evaluation the likes of which have been previously expressly rejected by EPA"s Science Advisory Board and EPA"s own published guidance on nutrient criteria derivation. The claim that the method is appropriate is thoroughly unsupported, not scientifically defensible, objectively irrational and without any known basis in accepted scientific methods for choosing necessary and appropriate nutrient controls for estuarine waters.[7] As such, this method for setting the nitrogen limit in the permit is arbitrary and capricious.

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[7] Based on the Supreme Court"s decision in *Daubert v. Merrell Dow Pharms*., no agency may base an analysis on scientific information that fails to meet minimum standards of reliability. 509 U.S. 579, 590 n.9 (1993). Daubert incorporates the administrative law principle that an agency cannot disregard the advice of its own experts or take

a) EPA ignored its own relevant guidance and procedures identifying the necessary analyses to establish a defensible nutrient criteria.

EPA has numerous documents showing how to relate nutrients to algae to DO in flowing waters. *See* EPA, *Nutrient Criteria Technical Guidance Manual: Estuarine and Coastal Marine Waters*, (Oct. 2001) ("Estuaries Guidance Document"); EPA, *Nutrient Criteria Technical Guidance Manual: Rivers and Streams* (July 2000).[8] Each of these documents requires EPA to account for the particular physical conditions influencing nutrient dynamics in the estuary to reasonably determine how the DO regime is impacted. These approaches all require detailed scientific data assessments and modeling. Likewise, EPA"s 2010 document entitled "Using Stressor-Response Relationships to Derive Numeric Nutrient Criteria" ("Stressor Response Guidance") stresses that a proper assessment must account for the factors that could influence the endpoint of concern (*e.g.,* DO) to ensure that nutrient criteria are necessary and properly established. For estuarine settings, that means that the evaluation must account for the physical setting, water column transparency, hydrology, hydrodynamics (in particular stratification), factors affecting algal growth rate, temperature, and detention time. EPA"s Fact Sheet did not present a single data plot or analysis to show any relationship exists between DO, chlorophyll a and TN for either the Taunton Estuary or Mount Hope Bay. Thus, there is nothing that shows the presumed conceptual model (TN caused excessive algal growth and low DO) is applicable to this estuary. There is no evidence in the record showing that achieving a 0.45 mg/L TN level is required in the Taunton River is necessary or sufficient to achieve DO standards. No information showing that TN reduction is required to correct a 0.5 mg/L DO deficit occurring in frequently in the Taunton River. Finally, there is nothing in the record to show that other options, such as adding DO to Taunton and Brockton effluent would be insufficient to offset low DO in the River if the impairment in fact still exists.

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action inconsistent with the facts demonstrated in the record. *Id.* at 593. Thus, for scientific evidence to be considered reliable for agency decision making, it must be based on an analysis that is accepted in the scientific community.

[8] *See also infra* note 31.

b) EPA’s simplified method is not accepted in the scientific community.

It is not accepted within the scientific community that stressor-response analyses used to identify numeric criteria, can be based on mere assumption. EPA has been harshly admonished by its own Science Advisory Board in drawing broad-based, unsupported and unverified conclusions with respect to nutrient control in similar circumstances:

> In order to be scientifically defensible, empirical methods must take into consideration the influence of other variables.

EPA, *SAB Stressor Response Review*, at 24 (Apr. 27, 2010).

> The statistical methods in the Guidance require careful consideration of confounding variables before being used as predictive tools…. *Without such information, nutrient criteria developed using bivariate methods may be highly inaccurate*.

Id. EPA"s latest approach is fundamentally flawed because EPA seeks to compare areas with radically different ecological settings- enclosed tidal rivers and well flushed open bay waters, without any analysis of the relevant factors influencing nitrogen impacts and other related factors influencing DO at these different locations.[9] There is no treatise or EPA guidance manual that indicates such an assessment is scientifically defensible or in any way accepted in the scientific community. In fact, in April 2010, EPA"s SAB has expressly stated the opposite- that only similar ecological settings should be evaluated when developing nutrient criteria and conducting stressor/response analyses based on empirical evidence.

> For criteria that meet EPA"s stated goal of “protecting against environmental degradation by nutrients,” the underlying causal models must be correct. *Habitat condition is a crucial consideration in this regard (e.g., light [for example, canopy cover], hydrology, grazer abundance, velocity, sediment type) that is not adequately addressed in the Guidance*. Thus, a major uncertainty inherent in the Guidance is accounting for factors that influence biological responses to nutrient inputs. *Addressing this uncertainty requires adequately accounting for these factors in different types of water bodies*.

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[9] This is the same error Dr. Steven Chapra informed EPA was fundamentally flawed when reviewing the EPA supported approach to generate nutrient criteria for Great Bay. (Attachment B- Dr. Chapra Declaration). His expert affidavit is applicable here because the same error is made in this instance and is even more egregious as EPA did not even attempt to show that the TN level caused excessive algal growth or that such algal growth was the likely cause of low DO conditions when proposing the Taunton permit.

Id. at 36, 37.

> *Numeric nutrient criteria developed and implemented without consideration of site specific conditions can lead to management actions that may have negative social and economic and unintended environmental consequences without additional environmental protection*.

Id. at 37. The analytical approach used by EPA to derive the required nutrient criteria and permit limits is also directly at odds with EPA‟s own 2010 Stressor Response Guidance[10] on proper derivation of nutrient criteria:

> "…, in the first step of the analysis, *classification*, the analyst attempts to control for the possible effects of other environmental variables by identifying classes of waterbodies that have similar characteristics and are expected to have similar stressor-response relationships."

Id. at 32.

> "… prior to estimating the stressor-response relationships, classes of waterbodies identified that are as similar as possible, except with regard to nutrient concentrations."

Id. at 56.

> "Beyond the possible effects of confounding variables, one should also consider whether assumptions inherent in the chosen statistical model are supported by the data."

Id. at 67. EPA completed none of these necessary evaluations for producing a defensible nutrient objective for the Taunton River Estuary, assuming that the system even exhibits a nutrient-induced DO impairment.

As noted earlier, EPA itself has put out different guidance manuals for rivers, lakes (bays) and estuaries because of the need to consider the effects of such different settings on nutrient impacts and criteria assessment.[11] None of these documents indicate it is acceptable to plot data from these different settings on the same chart to predict the impact of nitrogen or any other nutrient.

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[10] EPA, *Using Stressor-Response Relationships to Derive Numeric Nutrient Criteria* (Nov. 2010).

[11] EPA, *Technical Guidance Manual for Developing Total Maximum Daily Loads Book 2: Rivers and Streams; Part 1: Biochemical Oxygen Demand/ Dissolved Oxygen and Nutrients/Eutrophication*, at 4-27 (Mar. 1997).

Because EPA has used procedures that are not demonstrated to be scientifically defensible in any published treatise, are directly at odds with the Science Advisory Board admonitions and are contrary to EPA‟s own published guidance on how to properly evaluate a claimed nutrient-related DO impairment in an estuarine water, EPA‟s proposed approach is not scientifically defensible and cannot be ascribed to agency expertise. Consequently, these unproven and arbitrary procedures may not be used as a basis to establish water quality-based limitations under § 122.44(d).

4. EPA failed to account for existing treatment affecting Taunton River DO.

When determining the need for and level of nutrient control, EPA based all of its analysis on data and conditions occurring 8-9 years ago and did not account for any changed conditions occurring since then. (Fact Sheet, at 19 - 26). The Taunton River and tributaries to Mount Hope Bay have had extensive reduction of organic discharge due to CSO corrective measures and nutrient reduction since 2004. Effluent CBOD and nutrient levels have decreased dramatically from all discharges in the past 8 years. EPA‟s failure to account for these federally mandated actions impacting the need for TN reductions in the Taunton River, is a facial violation of applicable NPDES rules and the requirements of the Act.

It is axiomatic that an agency‟s permitting decisions should be based upon the latest available scientific information regarding the receiving water conditions and related regulatory efforts to address water quality. *See* 40 C.F.R. § 122.44(d)(1)(ii) (states in determining the need for permit limitations "the authority *shall* use procedures that account for *existing* controls on point and non-point sources…") (emphasis added); *see also Nw. Ecosystem Alliance v. Rey*, 380 F. Supp. 2d 1175, 1195-1996 (W.D. Wash. 2005) (finding an agency may not "simply rest on the previous EIS or [supplemental] EIS if there is new information that may alter the environmental analysis" and ultimately finding the agencies improperly relied upon outdated data in determining the supplemental EIS). Nowhere in EPA‟s analysis has the agency accounted for the extensive changes in facility operations that have reduced nutrients and CSO discharges impacting this estuary as well as Mount Hope Bay. Thus, EPA‟s proposed permit asserting a need for stringent TN limitations at the Taunton facility is plainly in violation of federal law because it is not based on the latest available scientific information or even remotely current water quality information for either Mount Hope Bay or the Taunton River.[12]

a) **Major improvements in water quality have occurred since 2004/5 that must be accounted for in setting permit limitations.**

Under the structure of the Act and its implementing regulations, it is plainly inappropriate to exclude consideration of current information that provides insight on whether or not historical water quality has significantly improved and the proper derivation of a narrative translator. *See, e.g.,* CWA Section 304(a) (requiring EPA to use the latest scientific information); 40 C.F.R. Part 130 (requiring impaired waters list be updated every 2 years in order to be based on current information for the estuary).[13]

In this case, EPA relied upon data from 2004/5 to conclude that major nutrient reductions were required to address DO concerns in both the Taunton River and, indirectly Mount Hope Bay. (Fact Sheet, at 29-30). Since 2004/5 there has been dramatic reductions in organic and nutrient loadings to these waters, therefore, the readings from 2004/5 cannot possibly reflect current

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[12] As the preamble to § 122.44(d) states, when developing a defensible water quality based limitation the "permitting authority should use all available scientific information on the effect of a pollutant on human health and aquatic life." 54 Fed. Reg. 23,868, 23,876 (June 2, 1989). EPA Region 1 has admitted that NPDES permits must be based on "all available scientific information." *See* EPA Response to Newmarket EAB NPDES Appeal 12-05, at 47. If the information used is not based on current conditions and fails to reflect known improvements in water quality occurring in the past 8 years, the analysis is neither "reliable" nor "scientific".

[13] The 11th Circuit Court of Appeals stated:

> The CWA requires that states identify all waterbodies within their boundaries that do not meet or are not expected to meet water quality standards. *See* 33 U.S.C. § 1313(d)(1)(A); 40 C.F.R. §§ 130.2(j), 130.7(b)(1). EPA regulations require states to ‚assemble and *evaluate all* existing and readily available water quality-related data and information to develop [their impaired waters lists].‟ 40 C.F.R. § 130.7(b)(5) (emphasis added).
>
> While § 130.7(b)(6)(iii) implies that Florida has a right to decide not to *use* certain data, it does not obviate the requirement in § 130.7(b)(5) that Florida *evaluate* all existing and readily available data. By taking the hard-line approach of not considering any data older than 7.5 years--even when there is no more current data for a particular waterbody--Florida has not fulfilled § 130.7(b)(5)'s evaluation requirement. Moreover, states are required by the CWA to identify *all* waterbodies that fail to meet water quality standards, 33 U.S.C. § 1313(d)(1)(A); states cannot shirk this responsibility simply by claiming a lack of current data. The district court misinterpreted the CWA's statutory and regulatory scheme when it held to the contrary, and we must therefore remand this issue for an analysis under the correct legal standard.

Sierra Club v. Leavitt, 488 F.3d 904, 913 (11th Cir. 2007).

conditions.[14] The reports entitled *Spatial and Temporal Patterns in Nutrient Standing Stock and Mass-Balance in Response to Load Reductions in a Temperate Estuary* (Attachment C)[15] and *Draft Nutrient Conditions in Narragansett Bay & Numeric Nutrient Criteria Development Strategies for Rhode Island Estuarine Waters* (Attachment D)[16], discuss the extent of nutrient reduction measures implemented by both Rhode Island and Massachusetts. From October 2003 to June 2008, at least eight Rhode Island wastewater treatment facilities, including the bay‟s second largest, upgraded to tertiary sewage treatment to remove excess nitrogen.[17] The largest, Field‟s Point WWTF, plans to complete its tertiary treatment system by December 2013 which will further reduce the bay‟s nitrogen levels.[18] In fact, it is expected that once the Field‟s Point WWTF upgrades are complete, the bay will meet the nitrogen target goal set by Rhode Island General Law § 46-12-3(25).[19]

Between the years 2000 and 2010, both the Taunton River and Narragansett Bay experienced significant reductions in TN loads. In the Taunton River, the average annual load of TN dropped from 1.64 x 10^6 kg to 5.28 x 10^5 kg from the periods 2003-2004 to 2008-2010. Adjusting for the difference in average annual flow, this represents a TN concentration reduction of 48%.[20] These reductions have greatly decreased total nitrogen levels in Mount Hope Bay and such levels are now well below the level EPA has indicated would be protective for Mount Hope Bay – 0.45 mg/L. *Infra* at 37-40.

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[14] After the 2003 fish kill in the Providence River, the Rhode Island legislature directed facilities to achieve a 50% reduction in nitrogen discharges. Tom Uva of the Narragansett Bay Commission indicated that the present TN discharges from Rhode Island have decreased by 48% and that ambient TN levels are the lowest measured to date. (Personal communication with John C. Hall on June 11, 2013).

[15] Jason Seth Krumholz, *Spatial and Temporal Patterns in Nutrient Standing Stock and Mass-Balance in Response to Load Reductions in a Temperate Estuary,* (2012).

[16] Christopher Deacutis and Donald Pryer, *Draft Nutrient Conditions in Narragansett Bay & Numeric Nutrient Criteria Development Strategies for Rhode Island Estuarine Waters* (June 2011).

[17] *Id.* at 2, 28.

[18] Krumholz, *supra* note 15, at 286.

[19] *Id.* at 97.

[20] *Id.* at 167.

A comparison of nutrient and organic loadings for the Taunton River demonstrates that major reductions in both parameters have occurred since 2004/5. The City of Brockton is in the process of undertaking additional modifications that will reduce its nitrogen loading even further. Overall point source nitrogen loadings to the estuary have decreased by approximately 25% since 2005 (excluding the CSO related TN reductions).

WWTF	Design Flow (MGD)	Receiving Stream	EPA Calculation Average 2004-05 Summer TN Discharge (lb/day)	May to October BETA Calculation Avg. 2004-05 Summer Discharge (lb/day)		May to October BETA Calculation Avg. 2011-12 Summer Discharge (lb/day)	
				BOD	TN	BOD	TN
Taunton[2]	8.4	Taunton River Estuary	610	474	681	116	502
Somerset[1]	4.2	Taunton River Estuary	349.5	244	412	160	398
MCI Bridgewater	0.55	Taunton River	37	202	No Data	341	24
Brockton[2]	18	Salisbury River	1303	358	1,434	117	618
Bridgewater	1.44	Town River	137.5	43	164	43	208
Mansfield	3.14	Three Mile River	375.5	24	431	19	383
Middleboro[2]	2.16	Nemasket River	207.5	11	282	11	397
		Total Load:	3,020	1,355	3,404	807	2,530

Notes:

1: Nitrogen data provided was monthly maximum day value.
2: CBOD measured during summer reporting period.
3: Values calculated with reported monthly averages unless otherwise noted.

The algal levels have also dropped in Mount Hope Bay by approximately 25%. Moreover, the Cities of Taunton and Fall River (at the mouth of the estuary) have implemented extensive wet weather controls that have reduced organic loadings to the river since 2004. *See* chart below detailing the degree of CSO reduction occurring. (Personal communication between Joe Federico, Beta Inc. and Nancy Beaton, CDM Smith).

Description	**Pre-CSO Program**	**Current**	**Reduction**
Estimated Annual CSO Volume	**1293 MG/year**	**278 MG/year (Overall)** **<65 MG/year (South/Central)**	**78% (Overall)** **>94% (South/Central)**

EPA‟s analyses, frozen in time failed to account for how these changes would alter the DO conditions in the Taunton River, 8 years later. Finally, the Brayton Point generating facility (at the mouth of the estuary) has implemented two new cooling towers that will lower temperatures in the Bay and Taunton River. (*See* Attachment E- Brayton Point Station Fact Sheet). The lower temperature will have a direct impact on promoting higher DO by (1) increasing DO saturation and (2) reducing the organic deoxygenation rates of the system. EPA‟s failure to account for the impact of these changes in treatment affecting algal growth and the DO regime is contrary to the requirements of 40 C.F.R. § 122.44(d).[21]

The effect of these measures since 2004/5 on DO in the Taunton River would be profound, assuming EPA‟s position regarding the factors controlling low DO is correct. The Bay delivers the vast majority of the water entering the Taunton River every day. EPA itself estimates that the salt water contribution is triple the fresh water component. (Fact Sheet, at 31). Improved DO would now be associated with these tidal flows as well as reduced algal levels. Likewise, millions of gallons of untreated wastewater have been reduced since 2004 via CSO control. This would reduce the organic enrichment of the estuary and reduce the low DO load associated with those combined sewer overflows. Given the scope of pollution reduction efforts occurring since 2004/5, it is inappropriate for EPA to claim that nutrient controls are necessary based on data reflecting 2004/5 conditions. It is certainly possible, if not likely, that the minor DO violations found to occur in the Taunton River based on 2004/5 conditions, no longer exist. In any event, the failure to account for these changes influencing the need for and extent of TN reduction is contrary to applicable rules and norms of administrative agency decision making.

In summary, to support it‟s claim that Taunton‟s nutrient discharge is the cause of narrative or DO criteria violation, EPA must utilize current data since numerous changes promoting improved DO have occurred since 2005. Therefore, EPA must update its analyses to reflect the known water quality improvements occurring since 2005 and determine, based on current data,

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[21] EPA was responsible, in part for mandating that nutrient reduction occur broadly in the Narragansett Basin and CSO reduction in Massachusetts. Those and other changes have produced major improvements in water quality such that the 2004/5 conditions referenced by EPA are no longer relevant.

whether or not the Taunton River Estuary is actually still impaired for DO and, if so, what factors are controlling that impairment.

5. EPA failed to provide a cause and effect demonstration as required by state and federal law.

As noted earlier, the Fact Sheet is bereft of analyses confirming that nutrients are the actual cause of low DO measured in the Taunton River in 2004/5. This is a fatal deficiency of EPA"s proposed permit action. Rather, EPA has employed a simplified form of "reference waters" assessment to select the "protective" TN concentration that must be achieved in the Taunton River. (Fact Sheet, at 30). As noted earlier, EPA"s selection of a TN end point for Mount Hope Bay was not based on a demonstrated impairment threshold needed to produce a minimum DO of 5.0 mg/L in the Taunton River. Moreover, the selection of the TN level failed to identify the relevant algal growth response which is necessary to produce the specific level of DO improvement to meet applicable numeric standards (assuming that the algal component is significant in controlling DO in the Taunton River) as required by state law.[22] Choosing a TN level without confirming that it is (1) necessary to produce the protective algal level and (2) that it can ensure DO compliance violates the requirement that the approach is sufficient to ensure standards compliance. (*See* 40 C.F.R. § 122.44(d)(1)(vi)(A) (requiring a narrative standard-based effluent limitation to "fully protect the designated use")). This plainly fails to meet regulatory prerequisites.

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[22] When EPA recently proposed estuarine nutrient criteria for Florida, EPA proposed chlorophyll a levels that were deemed sufficient to protect beneficial uses.

> EPA is proposing this [reference] approach to derive numeric chlorophyll *a* criteria for Florida"s coastal waters because the scientific data and information available were insufficient to establish accurate quantifiable relationships between TN and TP concentrations and harmful, adverse effects due to the limited TN and TP data available. Therefore, EPA is proposing to rely upon the reference condition approach to identify numeric chlorophyll-a criteria concentrations that protect the designated uses, and avoid any adverse change in natural populations of aquatic flora or fauna in Florida"s coastal waters.

EPA, *Water Quality Standards for the State of Florida's Estuaries, Coastal Waters, and South Florida Inland Flowing Waters* (2012), at 87.

a) The Clean Water Act requires a causal demonstration.

The CWA is a "science-based" statute that requires the establishment of criteria "accurately reflecting the latest scientific information" regarding "…the *effects* of pollutants on biological community diversity, productivity and stability…" 33 U.S.C. § 1314(a)(1); *accord*, 40 C.F.R. § 131.3(c) (criteria developed by EPA are based on "the *effect* of a constituent on a particular aquatic species"). No criteria (including a narrative criteria interpretation) can be approved unless it is "based on a sound scientific rationale". *Id.* § 131.11 (a).[23] Impairment listings only occur where it is demonstrated that the applicable criteria are exceeded. *See* 33 U.S.C. §1313(d).[24] Given the language of the Act and the implementing regulations, it is not surprising that courts have determined "that neither the language of the Act nor the intent of Congress appears to contemplate liability without causation." *See Nat'l Metal Finishers Ass'n*, 719 F.2d. at 640; *Ark. Poul. Fed. v. EPA*, 852 F. 2d 324, 328 (8th Cir. 1988) (stating the discharge must at least be "a cause" of the violation).

b) The state narrative criteria required cause and effect and excessive plant growth demonstrations.

The state narrative criteria require a "cause and effect" demonstration that nutrients actually caused excessive plant growth and such growth caused the low DO condition to claim a narrative violation exists. The Critical Indicators Interim Report specifies that nutrients "shall not exceed site-specific limits *necessary* to control accelerated or cultural eutrophication." (Critical Indicators Interim Report, at 9) (emphasis added).[25] However, nowhere does EPA present an analysis showing the Taunton River is subject to "cultural eutrophication" or that the specific

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[23] The Agency"s guidance on nutrient criteria development broadly discusses the need to address how causal (nutrients) and response (algal growth) is documented for particular water bodies.

[24] It is a general principle of the CWA, or any environmental statute for that matter, that pollutants be regulated if, and only if, they are causing harm or impairment. In generating numeric water quality criteria, EPA must abide by the same principle. *See* 33 U.S.C. §§ 1313(c)(2)(A), 1314 (a); 40 C.F.R. § 131.3(b); *Leather Indus. of Am.*, 40 F.3d at 401 ("EPA"s mandate to establish standards „adequate to protect public health and the environment from any reasonably anticipated adverse effects of each pollutant," does not give the EPA blanket one-way ratchet authority to tighten standards.").

[25] *See also* 314 CMR 4.05(5)(c) (Nutrients –"unless naturally occurring, all surface waters shall be free from nutrients in concentrations that would cause or contribute to impairment of existing or designated uses …").

values chosen from station MHB16 are "necessary" to ensure control of such unacceptable conditions in the Taunton River. As no such analysis is presented in the fact sheet, it is apparent that EPA has not properly interpreted or applied state law. Moreover, the Fact Sheet should have contained some demonstration that a specific reduction in algal level is needed to produce a specific improvement in DO in the Taunton River as state law is expressly intended to control excessive eutrophication (*i.e.,* excessive algal growth). No such analysis presented in this fact sheet. However, state rules do not regulate or prohibit "elevated nutrient levels" the applicable rules only prohibit such nutrient levels to the degree that they are the cause of "cultural eutrophication".[26] These are the required demonstrations under state law and EPA"s analysis failed to provide them to support the proposed limitations.

c) Federal rules and guidance require a demonstration of causation.

A "cause and effect" (*e.g.,* cause or contribute)[27] demonstration is necessary under 40 C.F.R. § 122.44(d) to regulate nutrients (*i.e.*, setting limits based on specific information confirming such effects actually occurred rather than generalizations regarding nutrient effects).[28] On its face, § 122.44(d) itself indicates that more restrictive limits only apply if the discharge "causes" a water quality criteria excursion.[29] The *Upper Blackstone* decisions repeatedly refer to the fact that

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[26] This "reference station" approach was also used by EPA to develop numeric nutrient criteria for streams in Florida based on a narrative standard and was struck down by the Court (*Fla Wildlife Fed'n, Inc., et. al. v. Jackson*, Case 4:08-cv-00324-RH-WSC, Doc. 351; N.D. Fla., Feb. 18, 2012) as insufficient to show that the criteria were necessary to maintain designated uses.

[27] The Region"s claim that § 122.44(d) requires that *no* discharge cause or contribute to a violation is a facial misreading of the provision.

[28] EPA"s latest position seems to be that it may impose nutrient requirements without such a demonstration. This, however, is a major reinterpretation of 40 C.F.R. § 122.44(d), without rulemaking and contrary to the structure of the Act. It is therefore illegal and may not be applied in this instance. *U.S. Telecom. Ass'n v. Fed. Commc'ns Comm'n*, 400 F.3d 29, 35 (D.C. Cir. 2005) („a substantive change in the regulation," requires notice and comment) (*quoting Shalala v. Guernsey Mem'l Hosp.*, 514 U.S. 87, 100 (1995)).

[29] The "or contributes" language means it is contributing to the "cause" of the violation. The structure of the rule and "relevant" preamble discussion confirms this approach. Under §122.44(d)(1)(ii), the permit writer first determines if "a discharge… causes or contributes to an instream excursion". In the case of a narrative standard one looks to see if the characteristics that are intended to be prevented are evidenced in the waters (*i.e.,* cultural eutrophication causing some type of system imbalance). If it is determined that an excursion is occurring (or likely to occur) then, and only then, under § 122.44(d)(1)(iii) "the permitting authority must establish effluent limits using one or more of the following methods…" The structure of the rule is clear, the methods for picking an protective instream level are only used to set the effluent limits, *not* to decide that the waters are in violation of the narrative standard. The 1989 preamble discussion confirmed this sequence:

nutrients were demonstrated to be "causing" extensive "cultural eutrophication" as the basis for imposing more restrictive limitations.

> Both the MERL model and the field measurements demonstrated that as nitrogen loadings increase, dissolved oxygen decreases and chlorophyll *a* increases, with both becoming less stable and subject to greater swings at higher levels of nitrogen. The EPA concluded that the basic causal relationship demonstrated in the MERL experiments "corresponds to what is actually occurring in the Providence/Seekonk River system."

Upper Blackstone v. EPA, 690 F.3d 9, 25-26 (1st Cir. 2012).[30]

The Rhode Island narrative criteria at issue in *Upper Blackstone* were also based on preventing "cultural eutrophication" as evidenced by nutrients causing excessive algal growth, low DO and related effects. In that case, the court first looked to see if the effects of "cultural eutrophication" existed and were documented to be caused by nutrients: "An influx of nitrogen and phosphorus from sewage treatment plants is causing serious problems for the River's waters and those downstream. The Blackstone, Seekonk, and Providence Rivers, and Narragansett Bay, *all suffer from severe cultural eutrophication*." *Id.* at 11 (emphasis added). The court observed "[h]ere, the EPA states, and the record reflects, that the MERL *model demonstrated the relationship between nitrogen loading, dissolved oxygen, and chlorophyll a production* for a range of loading scenarios *in a water environment similar to the Bay's.*" *Id.* at 27 (emphasis added). Further, the court noted:

> *Subsequently,* in order to address the severe and ongoing phosphorus-driven cultural eutrophication in the Blackstone River, the EPA incorporated a more stringent phosphorus limit into the 2008 permit. In formulating this limit, the EPA

> Subparagraph (i) should assist the permitting authority in determining whether it is necessary, under Federal regulations, to establish limits for a pollutant. *Note, however, this is different from calculating water quality-based effluent limits*. …Proposed subparagraph (iv) addresses the situation in which…the permitting authority does not have a numeric criteria to use *in deriving a water quality-based limit.*

54 Fed. Reg. 1,303, 1,304 (Jan. 12, 1989) (emphasis added).

[30] *Upper Blackstone*, 690 F.3d at 14 ("State water quality standards generally supplement these effluent limitations, so that where one or more point source dischargers, otherwise compliant with federal conditions, are nonetheless *causing a violation of state water quality standards*, they may be further regulated to alleviate the water quality violation. [30 U.S.C.] § 1311(b)(1)(C) …") (emphasis added).

> considered the national and regional guidance criteria and recommended values it had recently published.

Id. at 31 (emphasis added).

The April 2010 SAB Report on EPA"s stressor –response evaluations underscored the need for science-based "cause and effect" demonstrations when regulating nutrients: "Without a mechanistic understanding and a *clear causative link* between nutrient levels and impairment, there is no assurance that managing for particular nutrient levels will lead to the desired outcome." *Id.* at 4 (emphasis added). For criteria that meet EPA"s stated goal of "protecting against environmental degradation by nutrients," the underlying *causal models* must be correct." *Id.* at 37 (emphasis added). As noted earlier, EPA"s 2010 Stressor Response guidance issued in response to the SAB concerns recognized the need to establish the "cause and effect" relationship when regulating nutrients. No such analyses were presented in this permit action.

Because the proposed limits are not based on any demonstrated "cause and effect" relationship for the Taunton Estuary regarding "cultural eutrophication" and its current impact on the DO regime, the analysis is facially deficient and therefore, arbitrary and capricious and otherwise not in accordance with law. As discussed later in these comments, had EPA attempted to show a causal relationship between increasing nutrients, increasing algal levels and low DO for the Taunton River data, such an assessment would have shown those relationships do not exist in this estuary.

6. Natural conditions are not regulated as impairments and EPA lacks information confirming that DO conditions are anything but natural in the Taunton River.

The Fact Sheet confirms that natural conditions are not considered to be in violation of either numeric or narrative criteria (Fact Sheet, at 17). It is widely understood that low DO conditions may exist naturally in estuarine waters. Such low DO conditions due to natural factors have been confirmed in the Great Bay estuary (*see* Attachment F- Pennock, 2004 Lamprey River Dissolved Oxygen Study) due to periodic stratification of such waters. The studies of the Squamscott River (another Great Bay tidal river) also determined that low DO was not caused by elevated algal growth. (*See* Attachment G- letter from University of New Hampshire Professors to Mayors of Great Bay communities and Attachment H- Hydroqual assessment). It is apparent that the Taunton River may be performing similarly to these other tidal rivers in the nearby estuary that have undergone detailed scientific assessment. There is no information in the record showing that the periodic low DO is not natural, given the stratification that occurs in this system which causes low DO to occur.

The existing analysis of DO and chlorophyll a and its relationship to TN concentrations confirms that the minor, in frequent low DO is not apparently algal driven (*i.e.,* this is not a situation where diurnal DO changes are causing the occurrence of low DO). The low DO is produced by stratification and the condition is influenced by (1) the low DO entering from the Bay and (2) the deoxygenation of stratified waters due to sediment oxygen demand in the tidal river.

Given the dramatic CSO reductions that have taken place over the past 10 years, SOD would have been reduced. There is no reason to know whether or not the remaining DO condition (to the degree that it exists) is anything other than natural. Therefore, there is no basis at this time to assert that the discharge is presently causing or contributing to either a violation of the DO criteria for the Taunton River or any narrative criteria related to nutrients. As in the Great Bay tidal rivers, the stratification condition is a natural occurrence that, under certain conditions, will inevitably produce lower DO conditions. However, until EPA can demonstrate that the existing DO still fails to meet applicable criteria and that the remaining DO condition is a result of man-induced factors related to excessive algal growth, it is not reasonable to presume that nutrient regulation is necessary.

General Technical Comments on TN Limits

7. The TN endpoint used to derive the TN effluent limit is not scientifically defensible.

The "sentinel station" approach is not a rational or scientifically defensible basis for establishing a water quality standard because:

- It is contrary to EPA"s own guidance[31], and,
- It presumes, without any demonstration, that the factors influencing DO conditions at station MHB16 are the same factors that influence DO in the Taunton River Estuary.

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[31] *See* Estuaries Guidance Document; EPA, *Technical Guidance Manual for Performing Wasteload Allocations: Book III – Estuaries* (Part 1) (1990) ("WLA Guidance Document").

EPA likens the selection of a sentinel station as being consistent with the use of reference conditions to establish water quality criteria for nutrients. The "reference station" approach was used by the EPA to develop numeric nutrient criteria for streams in Florida and was struck down by the Court (*See Florida Wildlife Federation, Inc., et. al. v. Jackson*, Case 4:08-cv-00324-RH-WSC, Doc. 351) as insufficient to show that the criteria were necessary to maintain designated uses. As in Florida, the "reference" approach is also insufficient for use in Massachusetts. In this case, EPA cannot make a scientifically justified claim that the TN endpoint is necessary to meet a minimum DO concentration of 5.0 mg/L because EPA has not demonstrated that a TN concentration of 0.45 mg/L is a threshold, above which the DO criterion will be violated at station MHB16.

EPA‟s guidance documents on the development of numeric nutrient criteria and the development of wasteload allocations for dissolved oxygen in estuaries confirm that the primary effect of nutrients is to stimulate algal growth, which may influence DO in the estuary. However, many other factors influence DO levels and EPA presents no assessment to determine to what extent TN is causing the observed affects. Consequently, establishing a wasteload allocation for TN to address DO impairments in the estuary is arbitrary and capricious. Moreover, EPA has not demonstrated that DO at the Bay station (MHB16) responds in the same way as DO in the Taunton River Estuary (MHB19) or that the physical/chemical/hydrodynamic conditions at station MHB16 make it an appropriate reference site for the Taunton River Estuary. Consequently, the draft TN effluent limit based on this TN endpoint is arbitrary and capricious. EPA has not made any demonstration that the observed DO concentration is caused by the observed TN concentration. Without such a cause-and-effect demonstration, there is no reasonable assurance that controlling for TN will have any influence on minimum DO.

In developing the proposed TN endpoint, EPA noted that Massachusetts has not adopted numeric criterion for TN. (Fact Sheet, at 17). Rather, MassDEP uses a number of indicators to interpret its narrative nutrient standard. EPA asserts that MassDEP developed the Critical Indicators Interim Report for this purpose. However, the Critical Indicators Interim Report notes that the recommended ranges of appropriate TN thresholds must be further refined based on the specific physical, chemical, and biological characteristics of the system being evaluated. (*See* Critical

Indicators Interim Report, at 20). No such consideration was made for the Taunton River Estuary. Instead, EPA identified a threshold TN concentration for a site in *Mount Hope Bay* furthest from the Taunton River Estuary and assumed that this threshold concentration was appropriate in the Taunton River Estuary without any demonstration that the two locations behave in the same manner. In fact, the physical, chemical, and biological characteristics of the two areas are dramatically different. Station MHB16 is one of the deepest stations in the bay and is closest to the Ocean and Narragansett Bay while the Estuary consists of a very narrow channel of variable depth. These and other critical characteristics that dramatically affect how TN could possibly contribute to low DO via excessive algal growth were not considered in EPA‟s highly simplistic analysis. Thus, EPA‟s approach is not consistent with the methods described in the Critical Indicators Interim Report or with EPA‟s own guidance.

8. EPA completely ignores the conceptual model of significant factors that affect DO.

As described above, EPA identified a sentinel station (MHB16) and merely assumed, without any analysis, that the average TN concentration at the station should equal the allowable TN endpoint. This approach does not demonstrate that the conceptual model identified in the Fact Sheet is applicable to the Taunton River. (*See* Fact Sheet, at 14). This conceptual model is based on a well-recognized progression of symptoms that begins with the excessive growth of phytoplankton and macroalgae. As discussed in the Fact Sheet, the "primary" symptoms of nutrient over enrichment include an increase in the rate of organic matter supply (*e.g.,* phytoplankton), changes in algal dominance, and the loss of water clarity. These primary symptoms are followed by one or more secondary symptoms such as the loss of submerged aquatic vegetation, nuisance/toxic algal blooms, and low dissolved oxygen. While such conditions *may* occur, the presented analysis in the Fact Sheet nowhere demonstrates that they are occurring in the Taunton River.

a. Algal growth is *not* demonstrated to be excessive.

The primary effect of nutrient over enrichment is excessive algal growth. If algal growth is not excessive the secondary symptoms, particularly low DO, do not occur due to nutrient enrichment. Consequently, EPA must show that nutrients are stimulating algal growth (measured as chlorophyll-a), the levels of chlorophyll-a in the water column are excessive, and that the excessive levels of algae are, in fact, causing the observed low DO. In making this demonstration, EPA needs to identify a level of chlorophyll-a that is excessive and it must also include an evaluation showing that the nutrient reduction target selected will reduce algal growth to non-excessive levels that will raise DO levels to comply with the MassDEP water quality standards. The analysis presented in the Fact Sheet establishing the TN endpoint did not address *any* of these considerations. Rather, EPA identified a sentinel station that meets the DO standard and presumed that the annual average TN concentration at this station was the reason such compliance occurred. However, the average chlorophyll-a level found at this station (*i.e.,* the factor EPA presumes controls the occurrence of low DO) is 10.3 – 14.1 µg/L. (*See* Fact Sheet at 23, Table 5). This average algal level is *higher* than that present in the Taunton River at MHB19, which ranges from 5.5 – 10.5 µg/L. *Id.* Therefore, based on the DO response to algal growth at MHB16, it is apparent that excessive algal growth is (1) *not* occurring in the Taunton River Estuary and (2) some other factor *must* be causing the DO to drop below 5.0 mg/L in that area.[32]

b. The conceptual model does not support the sentinel station approach.

This "sentinel station" approach is not scientifically defensible for numerous reasons. First and foremost, the sentinel station approach presumes that the observed DO is caused by the observed TN. However, the proposed limits on TN have not been demonstrated to be necessary to attain the dissolved oxygen water quality standard. Many non-nutrient factors influence dissolved oxygen in the receiving waters, including natural and man-made conditions. EPA did not provide any assessment to evaluate the cause of low DO or to assess what fraction of the DO deficit is attributed to TN versus those other factors. Consequently, the proposed effluent limit is merely a guess. The "sentinel station" approach is demonstrably incorrect based on a consideration of the conceptual model, as illustrated in EPA"s Estuaries Guidance Document. TN has no direct impact on DO. Figure 2-4 (below) from the Estuaries Guidance Document illustrates the role of nutrients in phytoplankton growth:

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[32] This is the same conclusion reached by technical studies evaluating similar tidal rives in the Great Bay estuary. *See* Attachment G.

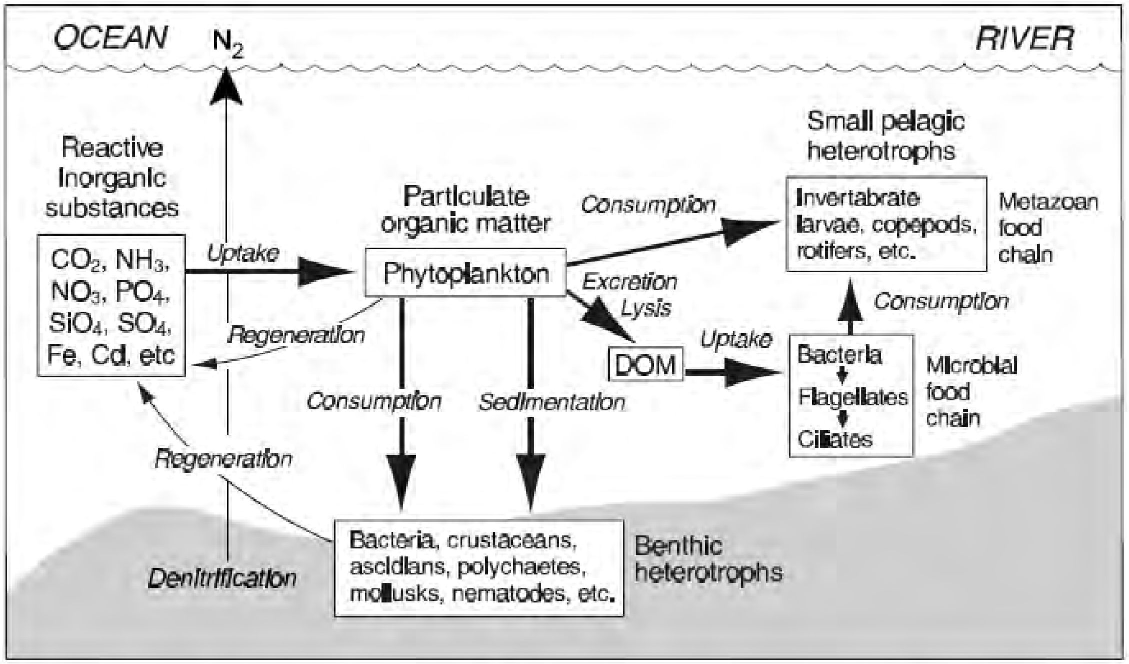

Figure 2-4. Schematic illustrating the central role of phytoplankton as agents of biogeochemical change in shallow coastal ecosystems. Phytoplankton assimilate reactive inorganic substances and incorporate these into particulate (POM) and dissolved organic matter (DOM) which support the production of pelagic and benthic heterotrophs. Arrows indicate some of the material fluxes between these different compartments. Denitrification has been added to the figure. Source: Cloern 1996.

Figure 2-9 (below) from the Estuaries Guidance Document illustrates the relationship between nutrients, phytoplankton and deep-water DO:

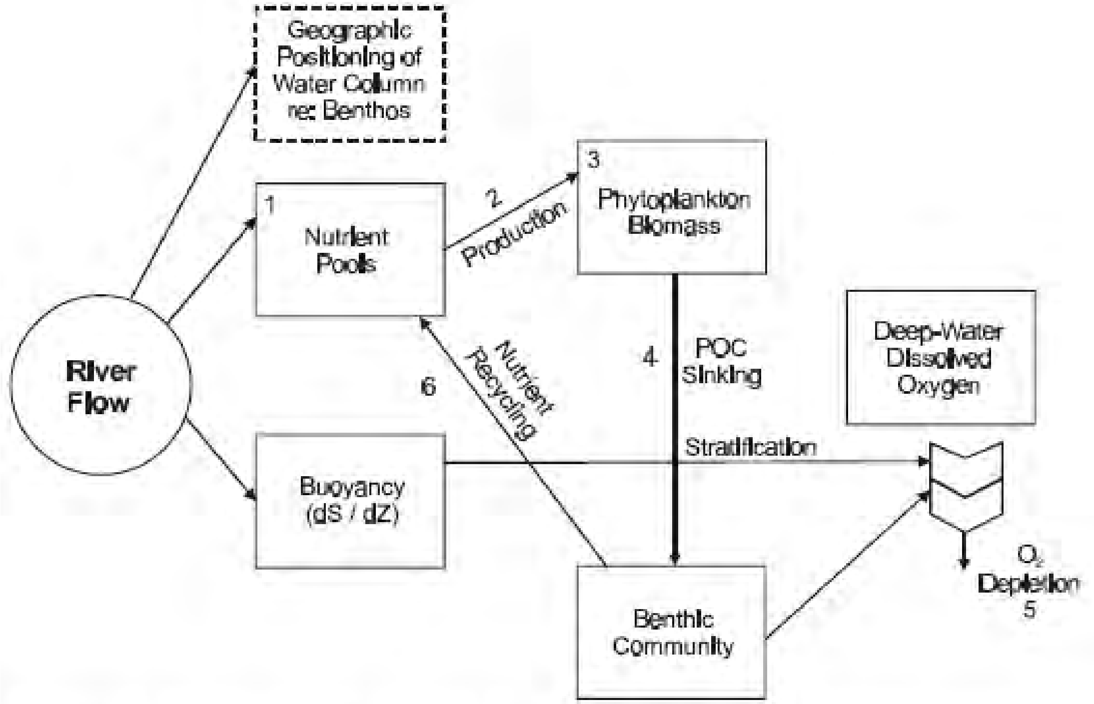

Figure 2-9. Simple schematic diagram showing the influences of river flow on ecosystem stocks and processes examined in this study. The mechanistic relationships between river flow and the stocks and processes shown in the diagram are explained in the text. Source: Boynton and Kemp 2000.

These figures *only* address the manner in which nutrients *may* influence phytoplankton growth and, subsequently, DO. It is obvious that this possible relationship does not provide "proof" that algal growth caused the existence of periodic low DO in the Taunton River Estuary. DO is also influenced by reaeration, organic matter (BOD), photosynthesis, and non-algal sediment oxygen demand as discussed in EPA"s WLA Guidance Document. Figure 2-6 and Figure 2-7 (below) from the WLA Guidance Document illustrates these interactions.

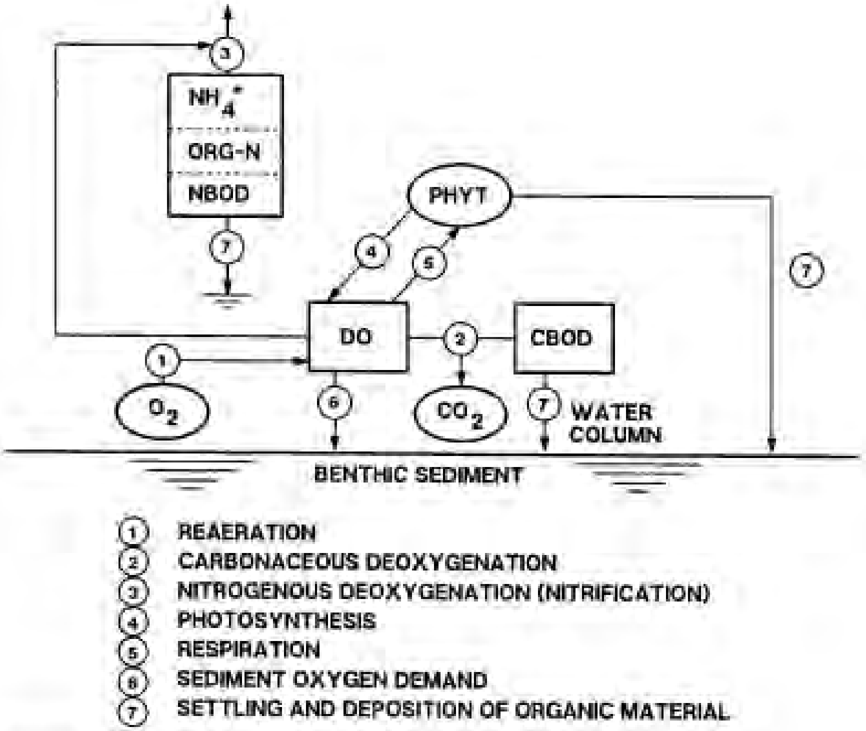

Figure 2-6. Basic variables and processes for dissolved oxygen.

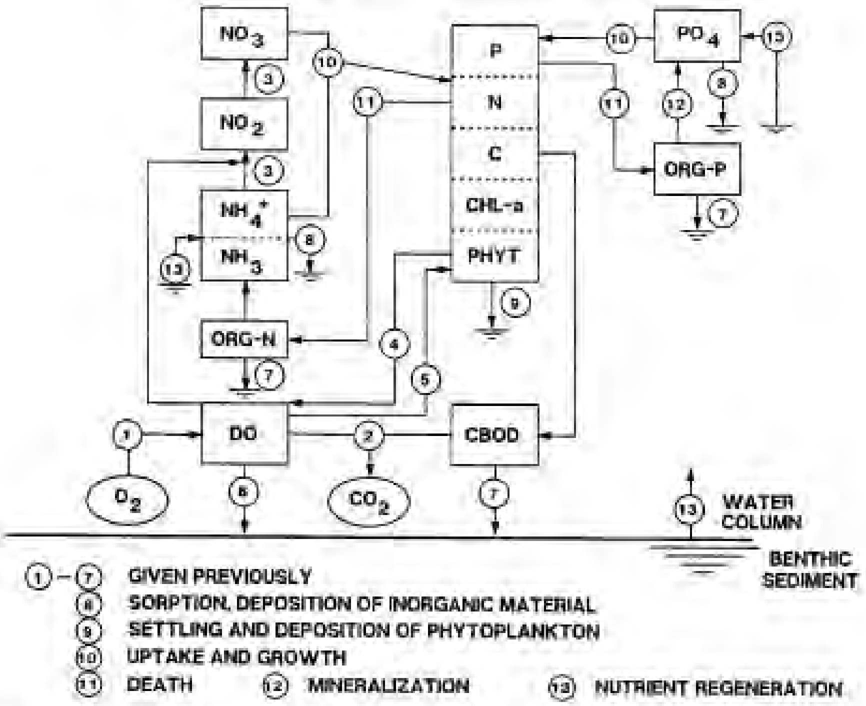

Figure 2-7. Standard variables for eutrophication and DO.

Together, these figures illustrate the complex relationship between nutrients, numerous other factors, and DO that must be address to competently determine what is causing a particular DO condition to occur. TN does not directly affect DO. Rather, any influence of TN is mediated through the growth of algae. Algae influences DO through photosynthesis (in the upper, photic zone), respiration, and decay (typically after settling). The influence of sediment oxygen demand on DO may be exacerbated by stratification which limits mixing between the upper and lower layers of water. System DO is also influenced by the decay of organic substances entering the system and the DO entering the system. However, the Fact Sheet presents no evaluation to determine the degree to which each of these factors influence DO in the Taunton River Estuary or Mount Hope Bay. Consequently, it is not possible to determine whether TN reduction is necessary or appropriate to address DO conditions in the Estuary.

c. EPA ignored the influence of stratification.

All of EPA"s guidance and SAB-issued commentary, as well as MassDEP guidance, states that the physical conditions of the receiving water must be evaluated to determine whether or how nutrients may cause adverse impacts. Stratification is particularly important with regard to the development of minimum DO conditions in the Estuary and Bay. When fresh and saline waters interact, they may become stratified with the denser, cold bottom saline water isolated from the less saline and warmer surface water. This situation is demonstrated to occur in the Bay and to be the primary factor triggering low DO conditions where the waters are deeper and less subject to turbulent mixing. Under stratified conditions, oxygen exchange with the surface waters is reduced and the effect of sediment oxygen demand (affected by algal and non-algal particulates) is pronounced, particularly when stratified conditions are prolonged. Thus, (1) the depth of the water, (2) the duration of the stratification event, and (3) the degree of the SOD all act to control the resultant DO condition in the stratified segment. Figure 1 (below) illustrates the pattern of temporal DO at the MHB-"Data Sonde" station operated by the Narragansett Bay Water Quality Monitoring Network (near MHB13) in relation to the tidal cycle.[33] Based upon the figure, periods of low DO in the bottom waters and maximum difference in surface-to-bottom-water DO

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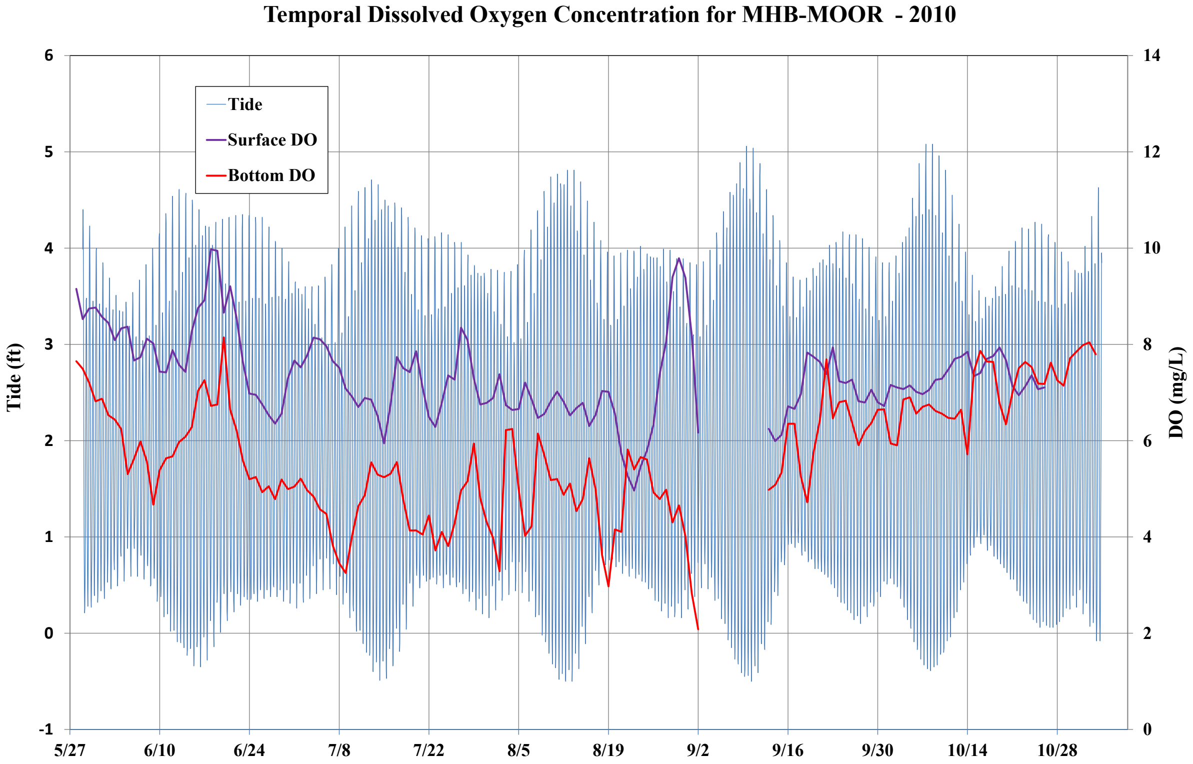
[33] Tidal stage data were obtained from NOAA for the Wickford gauging station. (Station I.D.: 8454538).

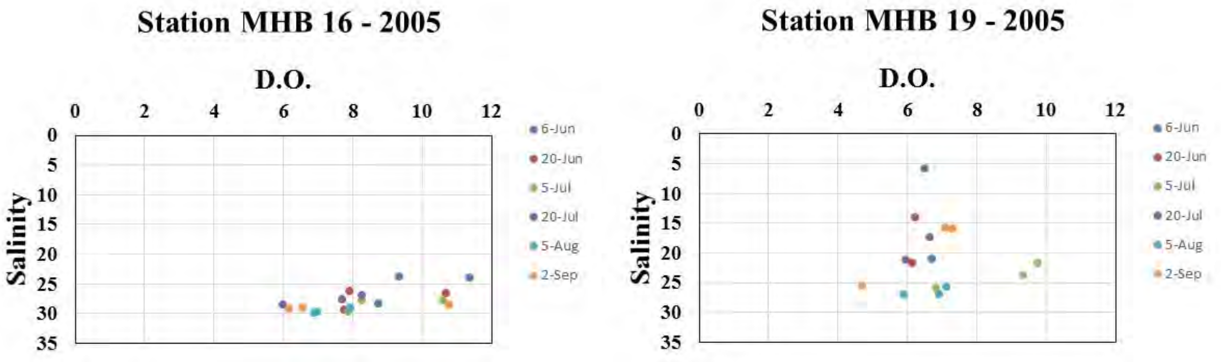
appear to coincide with neap tides, when tidal displacement in the Bay is at a minimum and stratification is prolonged.

Figure 1 – Tidal Stage versus Dissolved Oxygen in Mt. Hope Bay



Further upstream in the Estuary, stratification is far less intense and primarily caused by the tides. During the flood tide, marine waters rush in to the estuary with denser saline waters flowing below the less-dense fresh water. When the tide ebbs, these marine waters flow back into the bay. One consequence of this movement is that stratified conditions do not persist in the estuary because mixing and tidal exchange is much greater than at station MBH16 (the "sentinel station"). Consequently, the DO differences between the surface and bottom waters are far less than in the Bay and minimum DO concentrations tend to be associated with saline bay water that moves upstream during the flood tide. This means that DO in Mount Hope Bay has a *primary* control on the DO condition present in the Taunton estuary, *not* algal growth occurring in the Taunton River. Figure 2 (below) illustrates the differences in DO and salinity for the sentinel station in Mount Hope Bay (MHB16) and the upper Taunton River Estuary (MHB19) showing the physical condition are *not* comparable based on the 2005 database.

Figure 2 – Salinity and D.O. variability in Mt. Hope Bay and the Upper Taunton River Estuary



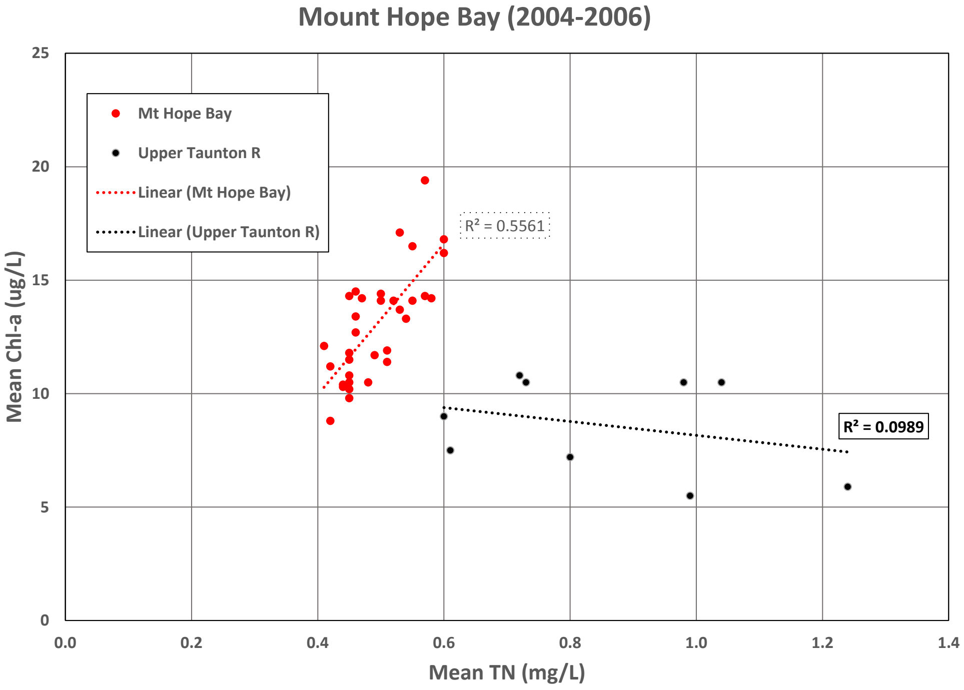
As discussed above, the conditions that create minimum DO conditions in the Bay are not the same as the conditions causing low DO in the Taunton River Estuary. Far less stratification occurs in the Taunton River for a shorter period and far less frequently. Consequently, the Taunton River station (MHB19) has a maximum DO variation of 0-3 mg/L (top to bottom). MHB16 has a variation of 1-5 mg/L. Therefore, unlike the Bay, the low DO condition and stratification in the Taunton River is very infrequent and far less intense. Consequently, the use of the Bay sentinel station to project the effect of TN on DO in the Taunton River estuary is arbitrary and capricious as the physical conditions controlling DO are markedly different at these two sites.

d. The response to TN differs in the Taunton River Estuary as compared to Mount Hope Bay.

EPA took the sentinel TN concentration at station MHB16 to prepare a mass balance analysis for the Taunton River Estuary at station MHB19. In doing so, EPA presumed, without any demonstration, that the conditions responsible for the DO readings in Mount Hope Bay are the same as in the Taunton River Estuary. Using the data presented in the Fact Sheet on Table 5 (Fact Sheet, at 23) it is apparent that Bay stations and Estuary stations do not respond in a similar manner. (*See* below Figure 3 and Figure 4). Figure 3 illustrates the apparent response of mean chlorophyll a to mean TN in the Mount Hope Bay stations in comparison with the response in the upper Taunton River stations (stations MHB18, MHB19, and MHB21). The apparent response in the Taunton River is flat over a wide range of TN concentrations while the response in Mount Hope Bay suggests a significant influence of inorganic nitrogen on plant growth.

Based on this comparison, it should be apparent that these systems behave very differently and the response at the sentinel station cannot be superimposed to predict how TN concentrations affect waters in the Taunton River estuary or the acceptable level of TN for the Taunton River.

Figure 3 – Mean Chlorophyll-a Concentration versus Mean TN in Mt. Hope Bay and Upper Taunton River (Stations 18, 19, 21)

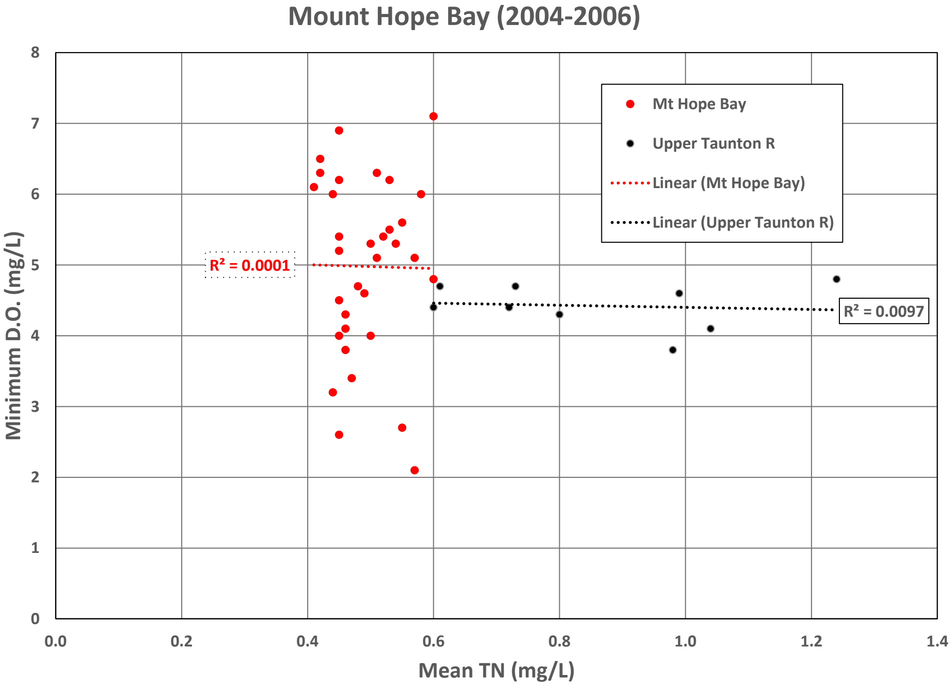


As these analyses indicate that EPA‟s conceptual model does not apply in the Taunton River, application of that model to derive more restrictive TN limitations is inappropriate. (*See* EPA Stressor Response Guidance, at 37).

e. Unique conditions which exist in Mount Hope Bay are not relevant to Taunton River Estuary.

EPA is regulating TN in the Taunton NPDES Permit under the belief that such control will "cure" low DO conditions in the Taunton River Estuary. This presumption is plainly incorrect based on the available monitoring data. Figure 4 (below) illustrates the apparent response of minimum DO to mean TN in the Mount Hope Bay stations in comparison with the response in the upper Taunton River stations. Again, the apparent response in the Taunton River is flat over a wide range of TN concentrations while the response in Mount Hope Bay suggests no relationship between TN concentration and minimum DO. In Mount Hope Bay, minimum DO levels range from 2 – 7 mg/L for essentially identical TN levels, ranging from 0.4 – 0.6 mg/L, with an $R^2 = 0.0001$. This exceedingly low R^2 indicates that minimum DO varies randomly with regard to TN concentration (*i.e.,* the two parameters are unrelated). The Taunton River Estuary shows a much smaller range in minimum DO levels (3.8 – 4.8 mg/L) over a far *larger* TN range of 0.6 – 1.2 mg/L, with an $R^2 = 0.0097$. This exceedingly low R^2 means there is no apparent relationship between TN and minimum DO (*i.e.,* TN explains less than 1% of the variation in minimum DO in the Taunton River Estuary). EPA‟s failure to analyze such available data was itself, arbitrary and capricious.

Figure 4 – Minimum DO Concentration versus Mean TN in Mt. Hope Bay and Upper Taunton River (Stations 18, 19, 21)



This complete lack of any meaningful relationship between TN and minimum DO in the Mount Hope Bay stations confirms that other factors, unrelated to TN, are strongly influencing minimum DO and nitrogen control is not likely to achieve compliance with the DO standard. The data assessment also confirms it is improper to presume that the Taunton River Estuary would respond to TN inputs in the same manner that Mount Hope Bay does, as one data set (Mount Hope Bay) indicates vertical response while the Taunton River has a horizontal response.

EPA, itself, has noted that nutrient criteria should not be developed if the impairment is insensitive to changes in nutrient concentration.

> Endpoints that were found to be insensitive to changes in nutrient concentrations in a particular estuarine system were not considered further in deriving numeric nutrient criteria for a system.

77 Fed. Reg. 74,924, 74,950 (Dec. 18, 2012).

Site-specific data for Mount Hope Bay and for the Upper Taunton River Estuary show that the minimum DO concentration does not show a response to increasing TN concentration. Since the purpose of this TN endpoint is to significantly mitigate exceedances of the minimum DO criterion in the Taunton River Estuary, consistent with EPA"s approach to numeric nutrient criteria development in Florida, the proposed endpoint for TN should be deleted from the permit. Consequently, the proposed effluent limit, which is based on restoring a use that is insensitive to increasing TN concentration, is arbitrary and capricious.

Other Technical Comments on TN Limit Derivation

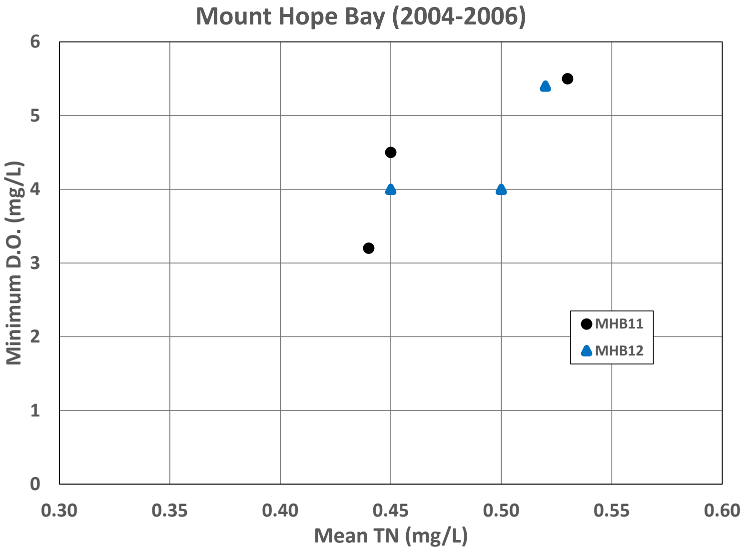
9. The TN endpoint was miscalculated.

Assuming, arguendo, that the sentinel station method is appropriate for establishing a TN threshold, EPA miscalculated the appropriate TN endpoint. The purpose of the calculation was to establish a TN concentration to ensure compliance with the applicable DO water quality standard. The selected TN endpoint, 0.45 mg/L, corresponds with a minimum DO concentration of approximately 6.0 mg/L, but the actual criterion target is 5.0 mg/L. (*See* Fact Sheet, at 23, Table 5). The data for MHB16 in 2006 show a minimum DO of 5.3 mg/L with a mean TN of 0.50 mg/L. Using these data, the TN endpoint necessary to achieve the DO criterion of 5.0 mg/L is a TN concentration *greater than* 0.50 mg/L, assuming that the Taunton River Estuary responded to TN in the same manner as observed in Mount Hope Bay. If a sentinel approach is defensible, it requires adjustment to reflect the TN load required to meet applicable standards (5 mg/L DO), not a 6.0 mg/L DO criteria.

10. The proposed TN endpoint is insufficient to achieve the DO criterion.

Water quality data presented in Table 5 of the Fact Sheet (at 23) show that several Mount Hope Bay stations do not achieve the DO criterion while in compliance with the proposed "protective" TN endpoint. These stations, MHB 11 and MHB 12, are illustrated in Figure 5 (below). Station MHB11 achieved the TN endpoint in 2004 and 2005, but was significantly below the minimum DO water quality standard in both of those years. Conversely, in 2006 this station exceeded the TN endpoint by a significant margin but was in full compliance with the minimum DO criterion. Similarly, station MBH12 was below the TN endpoint in 2004, but was also well below the DO criterion. In the subsequent years, this station exceeded the TN endpoint but alternatively failed (2005) and then exceeded (2006) the DO criterion.

Figure 5 – Minimum D.O. Concentration versus Mean TN (Stations 11, 12)



These data indicate that the selected TN endpoint is not needed to be protective of the applicable water quality standard. Moreover, the trend exhibited by the data indicates that the minimum DO *improves* with increasing TN concentration, contrary to EPA‟s conceptual model. This discrepancy with the conceptual model is a clear indication that other factors control the DO response. It is arbitrary and capricious for EPA to ignore this data confirming the simplified sentinel approach is not effective in controlling low DO conditions and chose a single "sentinel" location that fits EPA‟s regulatory theory.

11. TN is the wrong parameter to regulate for DO control in short detention systems such as the Taunton River Estuary.

EPA selected TN as the parameter to regulate without any demonstration that TN control is the appropriate form of nitrogen to achieve compliance with the DO water quality standard. As discussed above, the conceptual model for eutrophication in estuaries and coastal waters utilizes loads of dissolved inorganic forms of nitrogen as the basis for limiting algal growth and subsequently improving benthic DO levels. Notwithstanding the fact that EPA ignored its own guidance (*e.g.,* the Estuaries Guidance Document and the WLA Guidance Document) regarding selection of the nitrogen form to regulate, a consideration of the system hydrodynamics confirms that TN regulation is not appropriate. Assuming the Taunton River Estuary actually exhibited excessive algal growth, the form of nitrogen to control is DIN, not TN because of the systems short detention time. If the permit limit was based on DIN, it would completely alter the degree of treatment that would be required to reduce algal growth, since the background concentration of DIN in the ocean is negligible.

By regulating TN, EPA assumes that particulate and dissolved organic forms of nitrogen are available for stimulating algal growth in the Taunton River Estuary. The conversion of these organic forms to the form used by algae, DIN, requires that the residence time in the Taunton River Estuary and Mount Hope Bay is sufficient to allow this conversion. Based on the information presented in the Fact Sheet, Mount Hope Bay covers an area of 13.6 square miles, with a volume of 53.3 billion gallons at mean low water and a tidal range averaging approximately 4.5 feet. (*See* Fact Sheet, at 13). Assuming a tidal cycle of 12.3 hours, the total volume in the Bay is exchanged in 2.1 days. The exchange time in the Taunton River Estuary, itself, is projected to be less than one day based on the mean tidal exchange. This amount of time is insufficient to convert a significant amount of particulate and organic forms of nitrogen to DIN and EPA has provided no evaluation suggesting that such conversion occurs in the estuary or Bay to a significant extent. (*See* EPA, *Rates, Constants, and Kinetics Formulations in Surface Water Quality Modeling* (1985)).

If the regulated form of nitrogen is changed to the form controlling algal growth (*i.e.,* DIN), the necessary load reduction to meet DO standards would be significantly relaxed because the ocean boundary concentration of DIN is close to zero and the tidal exchange from the ocean provides significant dilution to the system.

12. EPA's analysis is based on outdated information.

EPA relied on water quality data collected by The School for Marine Science and Technology (SMAST) at the University of Massachusetts – Dartmouth to develop the TN endpoint of 0.45 mg/L. These data were collected from 2004 – 2006, but EPA only used the data from 2004 – 2005 for station MHB16 to calculate its protective threshold concentration. (*See* Fact Sheet, at 30). At the same time, SMAST collected data from 21 other stations that were summarized in Table 5 of the Fact Sheet (at 23). One of those stations, MHB-MOOR, centrally located in Mount Hope Bay, reported an average TN concentration of 0.48 mg/L over the same period.

The TN endpoint for this draft NPDES permit is based on data that are seven to eight years old and fail to reflect current conditions regarding TN and chlorophyll a levels in this system. Since 2004/5, many facilities that discharge to Narragansett Bay have implemented nutrient control and reduced the overall concentration of nitrogen and organic loadings to the Bay. Additional extensive reductions in nutrient load are associated with CSO controls being implemented by the City of Taunton and Fall River.[34] Ongoing monitoring data at Station MHB-MOOR, contained in a report by the Narragansett Bay Estuary Program[35], demonstrate that annual average nutrient concentrations ranged from 0.3 – 0.4 mg/L from 2006 – 2009 (illustrated in the following figure on page 35 of the report). The May – October average concentration (approximately, Julian date 120 – 304) are even lower, particularly in 2009. The 2009 TN concentration at the MHB-MOOR station was only 0.22 mg/L for the period from May – October. Thus, TN concentrations are within the range EPA has asserted reflect "excellent" water quality for Bay systems. (Fact Sheet, at 18). Under EPA"s own characterization, TN levels should be considered "excellent." (Fact Sheet, at 28 - citing a 0.3 – 0.39 TN level as "excellent").

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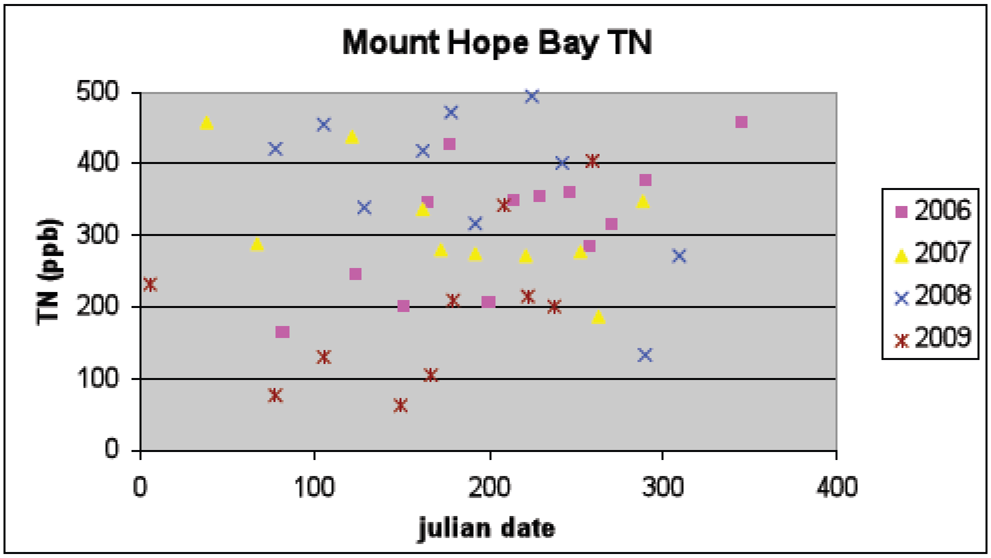
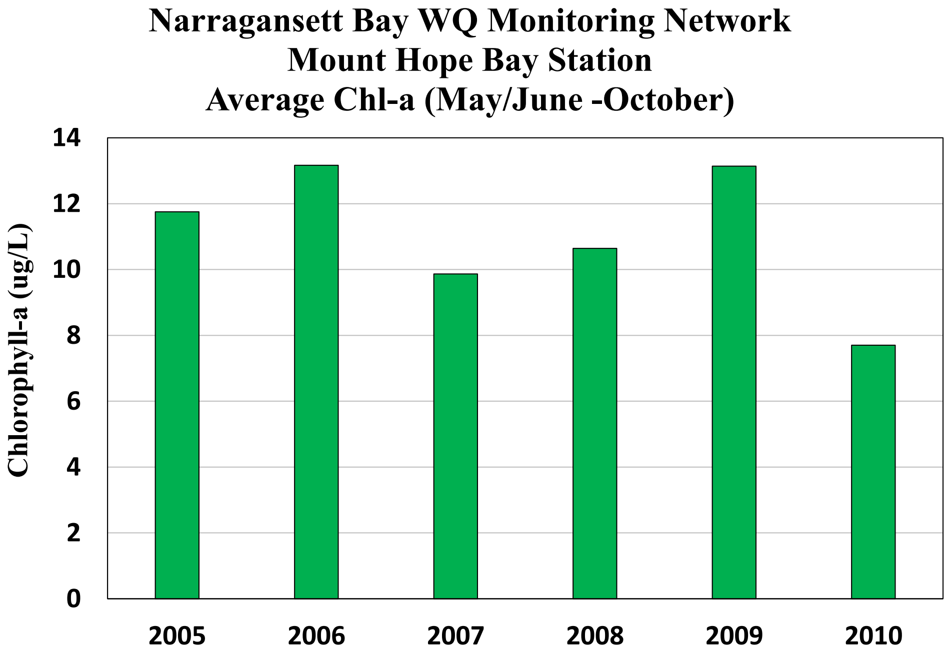
[34] *See* Attachment I– Excerpts from: City of Taunton Infiltration/Inflow Summary Report Jan 1, 2012- Dec. 31, 2012.

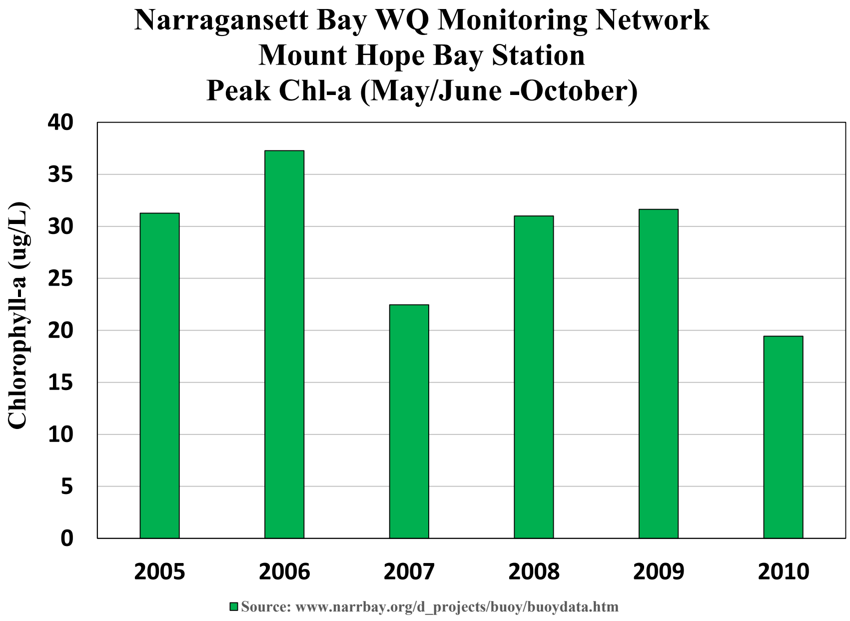
[35] Deacutis and Pryor, *supra* note 16.



Algal levels in Mount Hope Bay have dropped significantly since 2004/5, as illustrated in the charts below based on daily data collected by the Narragansett Bay Water Quality Monitoring Network near MHB-13 over the period from 2005 - 2010.



🟩 Source: www.narrbay.org/d_projects/buoy/buoydata.htm


Narragansett Bay WQ Monitoring Network
Mount Hope Bay Station
Peak Chl-a (May/June -October)

🟩 Source: www.narrbay.org/d_projects/buoy/buoydata.htm

Peak and average algal levels are at all-time lows. Assuming the algal levels are controlling system SOD and causing low system DO, these changes would produce far better DO conditions in the Bay, which greatly influences DO in the Taunton River.

As noted earlier, the TN levels in the Taunton River have also dropped dramatically over this period of time. *Supra*, at 15. Significant TN reductions have been achieved by facilities tributary to the river. These data indicate at least a 25% reduction in direct point source TN loadings. BOD discharge, which affects DO, has also improved. CSO reductions have also reduced TN and organic loads. These changes in nitrogen loading have produced about a 50% reduction in the Taunton system TN concentrations based upon a recently published PhD thesis. (Krumholtz, *supra* note 15).[36] Based on this information, the Taunton River likely meets EPA‟s suggested TN objective of 0.45 mg/L at MHB19, since the average TN concentration at this location was 0.70 mg/L TN. A 50% reduction in TN concentration would place TN concentration levels well below the 0.45 mg/L target EPA has chosen. Therefore, the need for further reduction at Taunton is not evident based upon current data.

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[36] The concentration of TN in the Taunton River has decreased from 1.74 mg/L in 2003-2004 to 0.91 mg/L in 2008-2010. Krumholtz, *supra* note 15, at 167, Table 3-2.

These data demonstrate that significant improvements in TN and algal concentration have occurred since the earlier SMAST study, with present annual average TN concentration of approximately 0.3 mg/L and average chlorophyll a less than 8 µg/L in the Bay. The conditions in the Bay will improve DO levels in the Taunton River Estuary because so much of the flow in the estuary originates from the Bay. At a minimum, the more-relevant new data must be used to assess current conditions in the Taunton River Estuary and the need for TN reductions at the Taunton WWTF.

Copper Limits not Necessary/Miscalculated

The draft NPDES permit includes revised water quality-based effluent limits for copper of 0.008 mg/L (monthly average) and 0.015 mg/L (daily maximum). The rationale for these effluent limits is presented in the Fact Sheet (at 36).

> The current permit for this facility contains an effluent limit for total recoverable copper based on the freshwater criteria for class B waters. The correct criterion for SB wasters is set forth below in terms of dissolved metals (form used for water quality standard) and total recoverable metals (used for permit limits). See 314 CMR 4.05(5)(e).
>
> Permit limits are calculated based on the [sic] meeting the criteria in the receiving water under 7Q10 conditions after accounting for the background concentration in the receiving water.

The final limits were determined based on compliance with the SB criteria using a mass balance equation:

$$Limit = \frac{Criteria\ x\ (7Q10 + Taunton\ WWTP\ Design\ Flow) - Background\ x\ 7Q10}{Taunton\ WWTP\ Design\ Flow}$$

This approach is premised on the assumption that the copper present in the effluent is in a toxic dissolved form such that an exceedance of the effluent limitation could adversely affect aquatic life. (*See* EPA Streamline Water-Effect Ratio Procedure for Discharges of Copper (Mar. 2001)). However, research confirms that copper from municipal effluents is chelated with dissolved organic carbon present in the treated wastewater such that is it not present in a toxic form. Consequently, there is no basis to claim an ecological concern with the discharge. This is further confirmed through consideration of whole effluent toxicity testing performed by the facility. The facility conducts whole effluent toxicity testing using organisms that are very sensitive to copper (*i.e., Ceriodaphnia dubia*). The results of this testing confirms that the copper in the effluent is not present in a toxic form given that no acute effects are found at concentrations that would produce such effects if copper were in a toxic forms. Consequently, the existing copper discharge cannot cause an impairment of designated uses and the proposed limits are not necessary. Moreover, even if the copper was present in a toxic form, the limits were calculated using the wrong mixing flow.

1. Copper is not in a toxic form in the Taunton River Estuary.

Performance data provided in Table 1 of the Fact Sheet (at 48-51) shows that the effluent is not toxic to *C. dubia*. These data, along with the corresponding copper concentration present in the test water, are summarized in the table below.

Date	Acute WET	Chronic WET	Copper (Average) (mg/L)	Copper (Max) (mg/L)
08/31/2010	100	100	0.0058	0.007
11/30/2010	100	100	0.0102	0.012
02/28/2011	100	100	0.012	0.014
05/31/2011	100	100	0.006	0.008
08/31/2011	100	100	0.009	0.011
11/30/2011	100	100	0.009	0.012
02/29/2012	100	100	0.01	0.012
05/31/2012	100	100	0.0063	0.0063

In every case, the whole effluent toxicity test indicated no toxicity in 100% effluent, with copper concentrations ranging from 0.006 – 0.014 mg/L. These results confirm that the copper present in the effluent is in a non-toxic state and should not be regulated as if it was toxic. Given these results, it is arbitrary and capricious for EPA to propose effluent limits assuming that the discharge has the reasonable potential to cause toxicity. The proposed limits for copper should be withdrawn.

2. Effluent limits were calculated improperly.

As described above, the water quality-based effluent limits in the current permit were calculated under the assumption that the facility discharged to Class B (fresh) waters. If this was the case, it would be appropriate to calculate the WQBEL using the 7Q10 flow as the dilution flow since this is the only flow into which the effluent mixes. However, EPA notes in the Fact Sheet, that the effluent actually discharges into saline (SB) waters. (Fact Sheet, at 16). Saline water is tidal and the dilution flow includes a tidal component of the flow that also provides dilution. This tidal flow was estimated to be 1,192 cfs (Fact Sheet, at 31). If copper limits are required for this discharge, the calculated limits must include the tidal dilution flow as well as the 7Q10 flow, and the WQBEL must also factor in the water effect ratio associated with the effluent.

A revised average monthly limit was calculated to account for this additional dilution flow, assuming that the dissolved copper concentration present in the ocean is negligible.

$$Limit = \frac{Criteria\ x\ (7Q10 + Ocean\ Flow +\ Taunton\ WWTP\ Design\ Flow) - Background\ x\ 7Q10}{Taunton\ WWTP\ Design\ Flow}$$

$$Limit = \frac{\frac{3.7\mu g}{L} x\ (31.6\ cfs + 1{,}192\ cfs +\ 13\ cfs) - \frac{2\mu g}{L} x\ 31.6\ cfs}{13\ cfs}$$

$$Limit = 347\ \mu g/L$$

Given this limit is far greater than existing effluent quality no reasonable potential exists to exceed the saline copper criteria and this limitation should be deleted from the permit.

List of Attachments

Attachment	
A	Letter from Ronald Poltak, Executive Director of the New England Interstate Water Pollution Control Commission to Lisa Jackson, EPA Administrator re: Nutrient Pollution (Jan. 3, 2011)
B	Declaration of Steven C. Chapra, Ph.D., F.ASCE (Feb. 27, 2013)
C	Jason Seth Krumholz, *Spatial and Temporal Patterns in Nutrient Standing Stock and Mass-Balance in Response to Load Reductions in a Temperate Estuary* (2012)
D	Christopher Deacutis and Donald Pryor, *Draft Nutrient Conditions in Narragansett Bay & Numeric Nutrient Criteria Development Strategies for Rhode Island Estuarine Waters* (June 2011)
E	Brayton Point Station, Somerset, MA, *Final National pollutant Discharge Elimination System Permit Fact Sheet* (Oct. 2003)
F	Jonathan Pennock, Ph.D., *2004 Lamprey River Dissolved Oxygen Study* (Mar. 31, 2005)
G	Letter from Richard Langan, Ph.D. & Stephen Jones, Ph.D. University of New Hampshire to the Mayors of Portsmouth, Dover, and Rochester, N.H. (Feb. 19, 2013)
H	Hydroqual, *Review of New Hampshire DES Total Nitrogen Criteria development for the Great Bay Estuary* (Jan. 10, 2011)
I	Excerpts from *City of Taunton Infiltration/Inflow Summary Report* J*an 1, 2012-Dec. 31, 2012*

Attachment A

NEIWPCC
New England Interstate Water Pollution Control Commission

Fostering Collaboration
on Water Issues

Training Environmental
Professionals

Coordinating
Water Research

Educating
the Public

January 3, 2011

Administrator Lisa Jackson
USEPA Headquarters
Ariel Rios Building
1200 Pennsylvania Avenue, N.W.
Mail Code: 1101A
Washington, DC 20460

Dear Administrator Jackson,

The Northeast states recognize that nutrient pollution is a significant environmental problem that impacts many waterbodies in our region and nationwide. Efforts such as the Long Island Sound and Lake Champlain TMDLs and the Massachusetts Estuaries Project provide concrete examples of our commitment to reducing nutrient inputs to our waters. We appreciate EPA's continued focus on this issue and fully support EPA Region 1's attention to how nutrient issues in the Northeast are distinct from those in other parts of the country. Furthermore, all of our states have put significant effort and resources into the process of developing numeric nutrient criteria. While we have no intention of abandoning our efforts to develop and establish these criteria, we have significant concerns with the direction EPA is now taking regarding the independent applicability of numeric nutrient criteria. The New England Interstate Water Pollution Control Commission recently represented its member states at an Office of Water briefing hosted by EPA Region 1. There, we had the opportunity to share some of our concerns with your staff, and have highlighted them for you below.

A number of Northeast states have advanced numeric nutrient criteria development to the point of initiating the rulemaking process within their state to establish these criteria as part of their Water Quality Standards. The technical approach favored by many states bases criteria on strong scientific evidence using stressor-response relationships, where nitrogen and phosphorus are the stressors and environmental indicators are the response (e.g. chlorophyll-a, Secchi disk, indices of biological health). Because the relationship between nutrients and environmental responses is based on many site-specific factors and varies from waterbody to waterbody, these responses consolidate the many site-specific factors that must be considered for efficient application of criteria, and therefore are the most appropriate indicators of a waterbody's impairment status.

Thus, both Maine and Vermont are proposing criteria for freshwater that are based on a decision framework that takes into account both causal variables (nitrogen and phosphorus) and environmental responses relevant to each waterbody. While EPA has argued that single number criteria approaches should be used, no such uniformity of condition exists in the natural world. Because nutrients are not toxic contaminants with threshold responses, conditions demonstrated by acceptable biological responses that are reflective of a range of nutrient conditions are the most appropriate way to

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apply criteria. While ambient concentrations may be helpful in screening potential impairments, under a decision framework approach, a waterbody would be considered impaired only if one or more measured environmental response criteria did not meet limits, regardless of whether or not the established phosphorus or nitrogen criteria were exceeded. In the case that all measured environmental response criteria are met, the waterbody would not be considered impaired, even if nitrogen or phosphorus concentrations were above the state's numeric criteria.

Based on the final criteria established by EPA for the state of Florida, and feedback provided to the states of Maine and Vermont by EPA Region 1, EPA is not supportive of response-based approaches. EPA has taken the position that states can incorporate response variables but must include numeric nutrient criteria for both nitrogen and phosphorus and that each criterion must be independently applicable to determine a waterbody's impairment status. By taking this position, a waterbody could be determined to be in violation of water quality standards even when a biological impairment does not exist. In addition, by requiring both nitrogen and phosphorus criteria to be incorporated into state water quality standards and applied independently, technological controls could be required to remove both nutrients even though most systems are controlled by the most limiting nutrient (i.e., typically phosphorus in freshwater and nitrogen in marine waters). This added burden could result in significant increases in sludge production and treatment and energy costs, despite not being necessary to control eutrophication in most cases. We recognize that there are some POTWs that discharge to both freshwater and marine systems, but this is the exception and not the rule.

EPA Region 1 has recently suggested a framework that allows for a waterbody exceeding a numeric criterion but meeting acceptable levels for environmental response variables to be listed as "indeterminate" for its attainment status. We appreciate the Region's continued dedication to finding a solution that is workable for both parties, but we still have the same fundamental objection that a waterbody that is meeting environmental response criteria should be listed as attaining standards even if it exceeds a numeric nutrient criterion. We understand that EPA has concerns about implementing response-based criteria, but we feel that this is a question that is dealt with in permitting, not standards development. Further, the Northeast states have solid experience in crafting defensible and robust permits with effluent limits derived from these same response-based criteria. We are committed to working with both of our EPA regions to continue implementing these valid and defensible limits using already endorsed EPA methodologies.

In summary, the Northeast states believe that EPA has failed to produce sufficient scientific evidence or a viable legal or policy basis for the imposition of independent applicability of numeric nutrient criteria. In addition, the Northeast states do not agree that numeric criteria for both nitrogen and phosphorus are necessary for all waterbodies. Numeric criteria should only be required for the limiting nutrient in a system unless dual limitation is demonstrated.

The Northeast states have amply demonstrated that using environmental response variables to develop nutrient criteria is a scientifically valid approach that is highly protective of water quality. Many years of data collection and analysis have gone into development of these criteria. Furthermore, in their review of EPA's *Technical Guidance on Empirical Approaches for Numeric Nutrient Criteria Development,* EPA's Scientific Advisory Board (SAB) recognized that a stressor-response approach is a legitimate, scientifically-based method for developing numeric nutrient criteria when it is applied appropriately, such as part of a tiered weight-of-evidence approach. The approaches being proposed by the Northeast states fall in line with this recommendation by the SAB, especially with respect to the potential range of acceptable nutrient concentrations, and their site-specificity, that a weight-of-evidence approach supports.

The Northeast states are very appreciative of the assistance provided by EPA Region 1 throughout the nutrient criteria development process and have every intention of continuing the scientific work that will build the foundation of their numeric nutrient criteria. We also plan to continue to address nutrient impairments through NPDES permitting, TMDLs, and adaptive watershed management, while criteria are being developed and put in place. However, the Northeast states are concerned about EPA's approach, and many states are taking the position that they will not proceed any further with adoption of numeric nutrient criteria until EPA has provided sufficient explanation of the legal requirement and scientific basis for the requirement for independent applicability of criteria. Once those concerns can be addressed, we will renew our commitment to the process of establishing these important criteria in earnest.

Thank you for your consideration of the concerns we have described. We are eager to continue working with you on this important environmental issue and look forward to your response.

Sincerely,

Ronald Poltak
Executive Director

Cc: Curt Spalding, Regional Administrator, EPA Region 1
Judith Enck, Regional Administrator, EPA Region 2
NEIWPCC Executive Committee

Attachment B

BEFORE THE ENVIRONMENTAL APPEALS BOARD
UNITED STATES ENVIRONMENTAL PROTECTION AGENCY
WASHINGTON, D.C.

In re:)
Town of Newmarket)
NPDES APPEAL No. 12-05)
NPDES Permit No. NH0100196)

Declaration of Steven C. Chapra, Ph.D., F.ASCE[1]

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

Executive Summary

This document provides an expert review of the New Hampshire Department of Environmental Services (DES) approach to nutrient criteria development for the Great Bay Estuary. The methodologies under review are those presented in the document entitled "Numeric Nutrient Criteria for the Great Bay Estuary" (2009). My analysis is specifically directed at addressing whether the Division's use (and EPA's acceptance) of the "stressor-response" methodology in that document to derive the recommended nutrient criteria for total nitrogen employed scientifically defensible methods and whether those methods, as applied, are consistent with generally accepted scientific norms applicable to the use of such statistical methods. Upon review, it is my opinion that the DES criteria document did not use scientifically defensible methods and it failed to apply stressor-response methods in a manner accepted by the scientific community. *The methods applied are, in fact, grossly incorrect, internally inconsistent and have produced results that bear no reasonable relationship to reality.* Consequently, the analysis was fundamentally flawed and the proposed TN criterion of 0.3 mg/l is not demonstrated to be either necessary or appropriate to protect aquatic resources in the Estuary.

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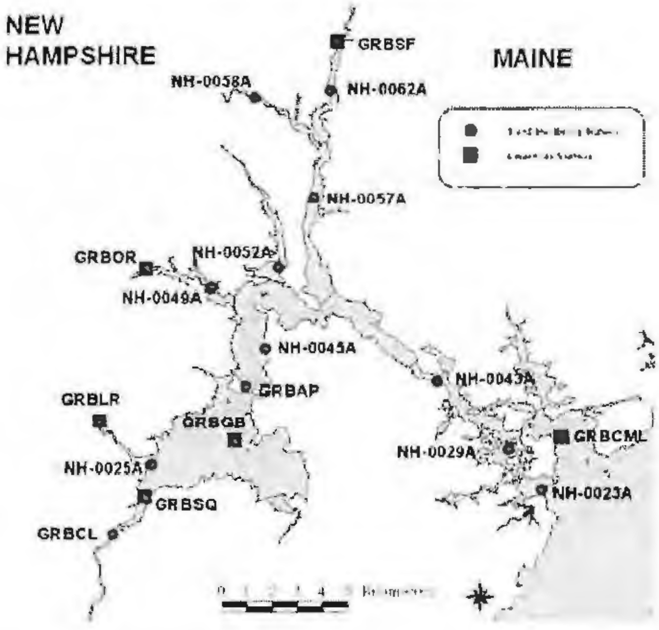
[1] Professor and Berger Chair in Computing and Engineering; Civil and Environmental Engineering Department; Tufts University; Medford, MA 02155

Assessment of whether the 2009 Numeric Nutrient Criteria document employed scientifically defensible methods in criteria derivation

The DES numeric criteria document (hereafter, the "Criteria Document") was completed in June 2009[2] and relied extensively on simple linear regression analyses (1) to show nitrogen was causing certain adverse system responses and (2) to select the level of nitrogen that would control and eliminate those adverse responses. The adverse responses of concern were (1) low dissolved oxygen (D.O.) occurring in the tidal rivers and (2) poor water column transparency caused by excessive algal (phytoplankton) growth. The document also included limited references to excessive macroalgae growth for Great Bay proper, but this concern did not control the derivation of the recommended TN criteria for either the tidal rivers or the bay systems.

Figure 2 from the Criteria Document, presented below, indicates the scope of the monitoring program used to supply the data in the regression analyses. The various locations are physically very heterogeneous and include near ocean bays, tidal straights, inland bays, and tidal rivers.

Figure 2: Trend Monitoring Stations for Water Quality in the Great Bay Estuary



Data from these various locations throughout the estuary, representing dramatically different physical habitats and hydrodynamic conditions, were averaged for use in subsequent regression analyses. Charts were prepared claiming to demonstrate how key nutrient concentrations and response variables (e.g., chlorophyll a, transparency) changed

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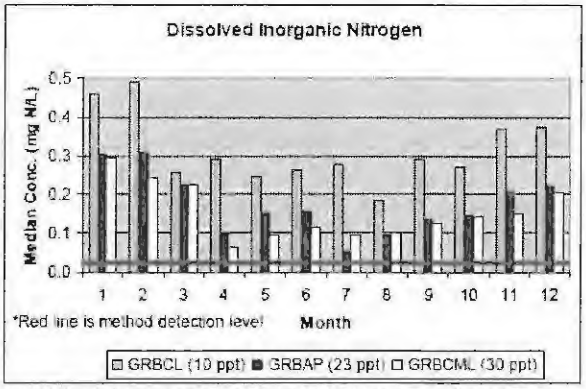
[2] Numeric Nutrient Criteria for the Great Bay Estuary. New Hampshire Department of Environmental Services. June 2009.

through the system as a function of each other. Figure 8 from the Criteria Document illustrates monthly changes in inorganic nitrogen levels for a tidal river (Station GRBCL; Squamscott River), an inland bay (Station BRBAP; Great Bay-Adams Point), and the mouth of the estuary (Station BRBCML). The figure shows that inorganic nitrogen concentrations are significantly higher in the tidal river and decrease towards the mouth of the estuary. This decrease generally aligns with the average salinity at each station.

Figure 8: Seasonal Pattern for Dissolved Inorganic Nitrogen at Trend Stations with Different Salinities



All available data for these stations in 2000 - 2008 were included in this graph, which amounts to:
GRBAP: (Jan-Mar) 2000, 2001, 2006, 2007, 2008; (Apr-Dec) 2000 through 2008
GRBCL: (Jan-Mar) 2000, 2001; (Apr-Dec) 2000, 2001, 2002, 2003, 2004, 2005, 2006, 2007, 2008
GRBCML: (Jan-Mar) 2001; (Apr-Dec) 2000, 2001, 2002, 2003, 2004, 2005, 2006, 2007, 2008

Figure 13 from the Criteria Document illustrates the long term algal levels at various sites within the estuary, while Figure 16 illustrates monthly changes in median chlorophyll-a in a tidal river (Squamscott), Great Bay, and at the mouth. The long term average algal levels are higher in certain tidal rivers (e.g., Squamscott) but lower as one proceeds into waters with greater flushing characteristics (Great Bay and the Piscataqua River). It should be noted that the algal levels occurring throughout the system are, on average, generally quite low. Even in the higher detention time areas of Great Bay, the average concentration is only about 3 μg/l while in areas of very high tidal exchange (Piscataqua River) the average concentration ranges from 1-2 μg/l. This low level of primary productivity indicates that this system is not conducive to producing significant algal growth as a result of current nutrient inputs. [3]

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[3] For example, a 100 μgN/L level of dissolved inorganic nitrogen in Great Bay has the potential to grow about 30 μg/L chlorophyll-a. This is an absolute upper limit as is borne out by the fact that the median algal growth in Great Bay is one tenth of this potential. This indicates that other factors (i.e., water column transparency, detention time, nutrient recycle, etc.) are controlling the amount of plant growth that occurs.

Figure 15: 90th Percentile Concentrations of Chlorophyll-a in Regions of the Great Bay Estuary Calculated from Samples Collected in All Seasons in 2000-2008

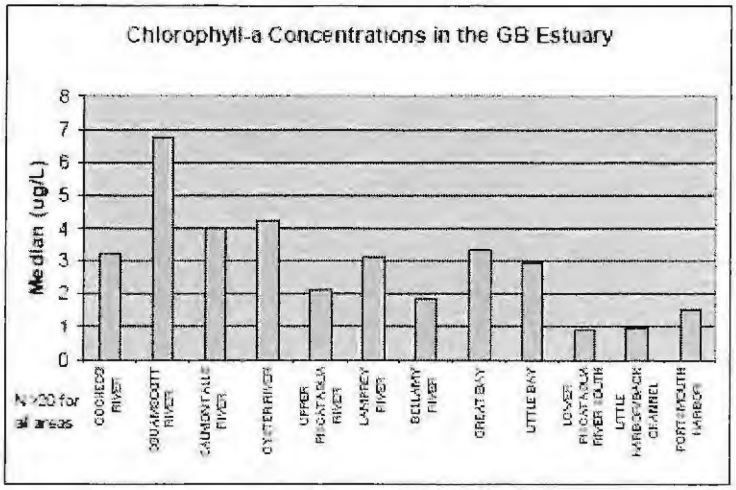
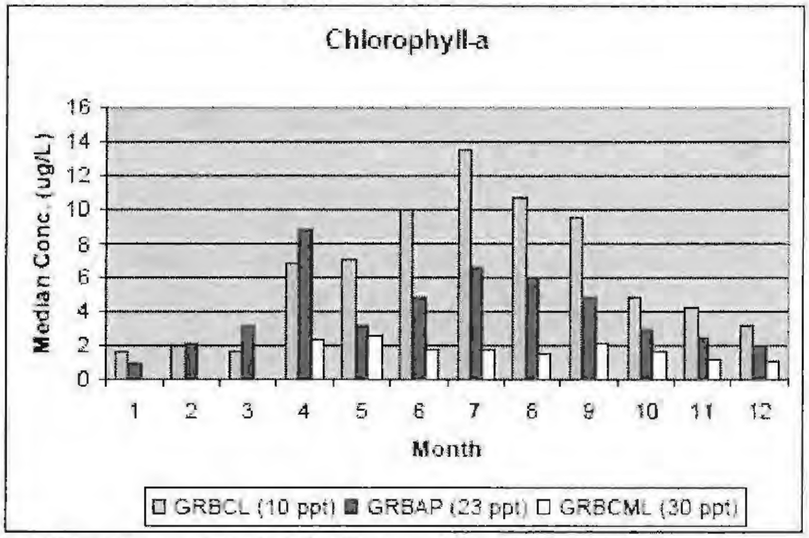


Figure 16: Seasonal Patterns of Chlorophyll-a at Trend Monitoring Stations with Different Salinities



All available data for these stations in 2000 - 2008 were included in this graph, which amounts to
GRBAP: (Jan-Mar) 2000, 2001, 2006, 2007, 2008; (Apr-Dec) 2000 through 2008
GRBCL: (Jan-Mar) 2000, 2001; (Apr-Dec) 2000, 2001, 2002, 2003, 2004, 2005, 2006, 2007, 2008
GRBCML: (Jan-Mar) None; (Apr-Dec) 2001, 2002, 2003, 2004, 2005, 2006, 2007, 2008

The DES considered this information and concluded that the observed algal chlorophyll-a was in response to the spatial pattern of nitrogen. DES then prepared a regression analyse relating the 90th percentile chlorophyll-a concentration to total nitrogen (Figure 17 from the Criteria Document). It then claimed that this regression proves that primary productivity (as indicated by phytoplankton blooms) is associated with the concentration of nitrogen. [4]

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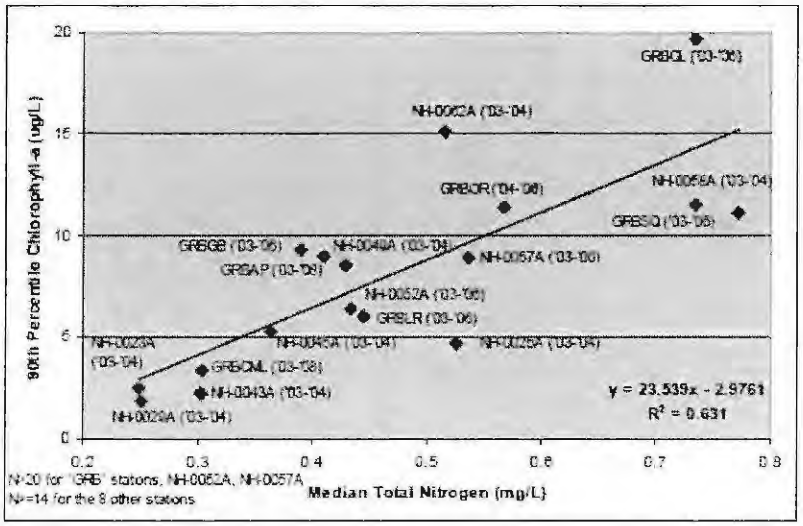
[4] This conclusion was directly at odds with the 2013 State of the Estuaries report that confirmed algal levels in the system have not materially changed over a 30 year period despite wide fluctuations in available inorganic nitrogen. This would only occur if TN was NOT the factor presently limiting algal growth in this system.

Figure 17: Relationship between Nitrogen and Chlorophyll-a Concentrations at Trend Stations



N=20 for "GRB" stations, NH-0052A, NH-0057A
N=14 for the 8 other stations

This regression does not provide any of the "proof" claimed by DES, and as discussed below, has gross methodological flaws. For a regression analysis to be scientifically defensible, confounding factors that influence the response variable (chlorophyll-a) must be controlled so that the stressor variable (total nitrogen) is the only factor (or at least the primary factor) influencing the response. DES did not considered any confounding factors when it prepared this simple regression. Consequently, all that can be determined from this analysis is that chlorophyll-a levels and total nitrogen levels co-vary. Such omission of confounding factors leads to what are formally called in the statistics literature "spurious correlations."[5]

If the data are re-plotted and classified according to biotype it is readily apparent that the observed light attenuation response reflects the hydrologic conditions of the monitoring station. The apparent relationship between light attenuation and TN is an artifact caused by the concurrent decrease in TN concentration caused by dilution with the tides. Virtually all of the regression evaluations presented in the Criteria Document plot data from highly different systems (riverine, bay, ocean) without accounting for the many factors that make these systems respond differently. Such evaluations are not scientifically defensible, are not accepted within the scientific community and yield unreliable results.

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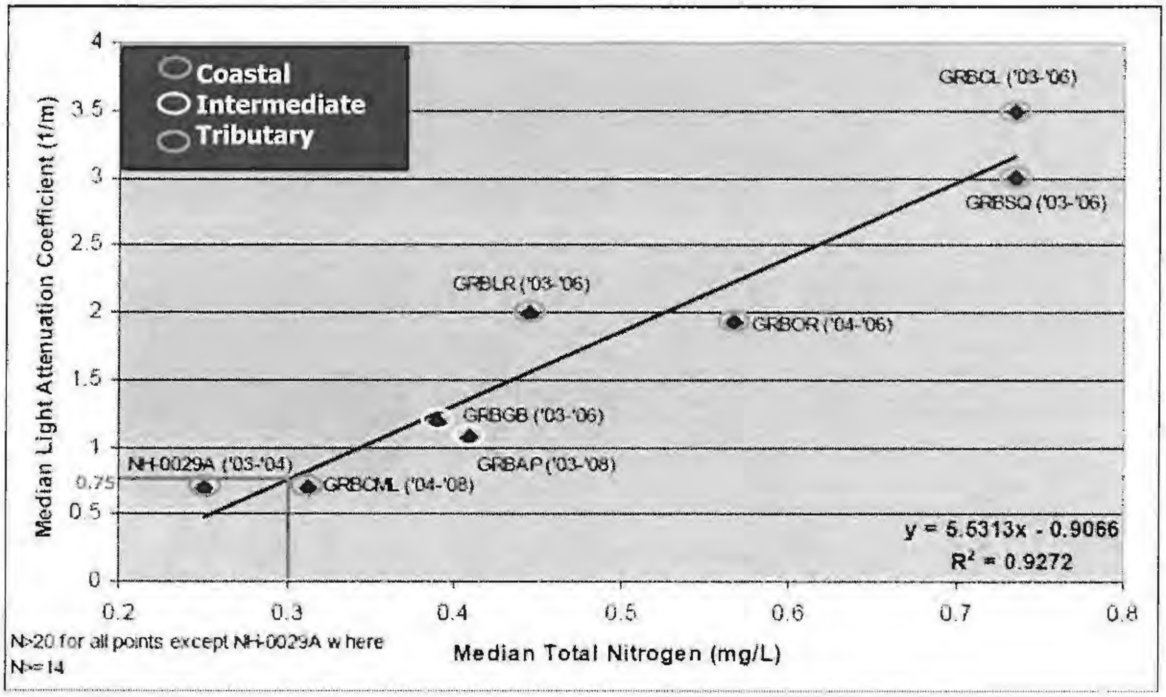
[5] Pearl, J. 2000. Causality: Models, Reasoning and Inference, Cambridge University Press.



N>20 for all points except NH-0029A where N>=14

Dissolved Oxygen Impact Analyses

The Criteria Document presented several simple regressions relating dissolved oxygen levels to chlorophyll-a concentration (Figure 26) and total nitrogen (Figure 29). In Figure 26, the minimum and maximum reported dissolved oxygen concentrations are plotted against the 90th percentile concentration of chlorophyll-a in the various Assessment Zones of the estuary. The Criteria Document claims that these regressions clearly show both a decrease in the minimum D.O. and an increase in the maximum D.O. with increasing chlorophyll-a.[6] This regression evaluation is unreliable for several reasons. First, as with other graphs, it combines results from hydrologically distinct areas, which has no basis in proper ecological data assessment. Many factors influence D.O. and it is certain that these factors are not uniform among all of the assessment zones and seasonal data (e.g., temperature, salinity, time of sampling). Secondly, the supposed influence of algal level on minimum D.O. yields a very flat response, confirming that nutrients cannot be the primary factor influencing the response. Consequently, nutrient control cannot materially improve water quality with regard to attainment of the D.O. criterion. Finally, Figure 26 implies that the diurnal range in D.O. varies from 7 – 12 mg/L for chlorophyll-a ranging from 2 – 17 µg/L. Modeling estimates using well calibrated models predict a diurnal D.O. range of only 1 – 3 mg/L for such a narrow range of algal growth. Consequently, some other unconsidered factors must contribute significantly to the observed results, not TN.

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[6] It is not apparent that this graph is even plotting the D.O. condition occurring when the 90th percentile chlorophyll-a concentrations occurs. If this is not the case, the entire relationship is a statistical fabrication based on unrelated information.

Figure 26: Relationship between Dissolved Oxygen and Chlorophyll-a in Assessment Zones

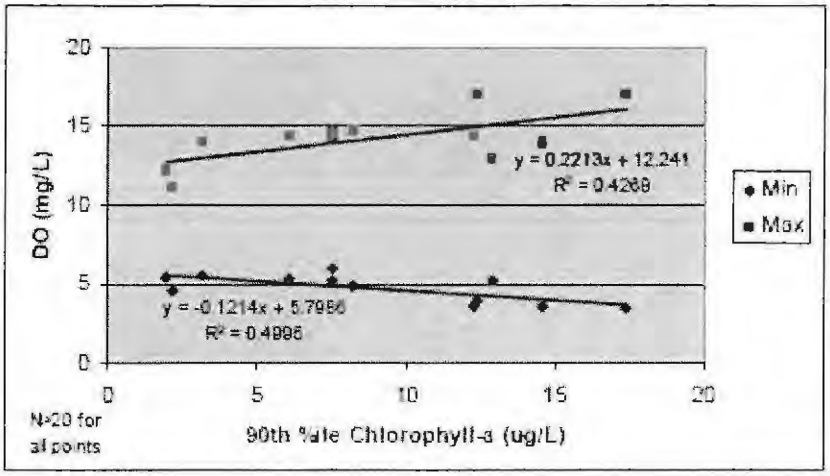


Figure 29 presents minimum dissolved oxygen at the Trend Stations in relation to median total nitrogen. This type of analysis has no basis in the literature or any published method of acceptable DO impact assessment. TN does not have a direct effect on dissolved oxygen and attempting to relate these two parameters is not accepted within the scientific community. Rather, DES must first show the relationship between TN and chlorophyll-a and then show the relationship between chlorophyll-a and D.O. If this is done by comparing Figure 17 and Figure 26, it shows a very minor influence of TN on minimum D.O. However, the regression in Figure 29 suggests a very significant influence of total nitrogen on minimum D.O. This discrepancy is a clear indication that these regression analysis are producing diametrically opposed results.

Figure 29: Relationship between Dissolved Oxygen and Nitrogen at Trend Stations

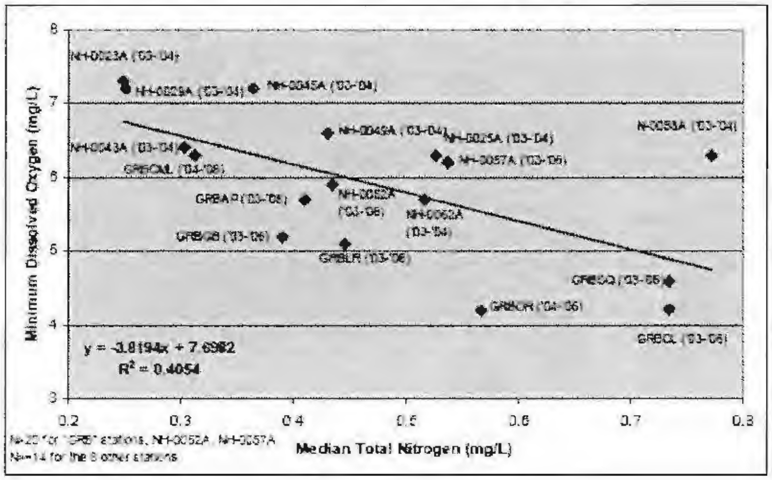
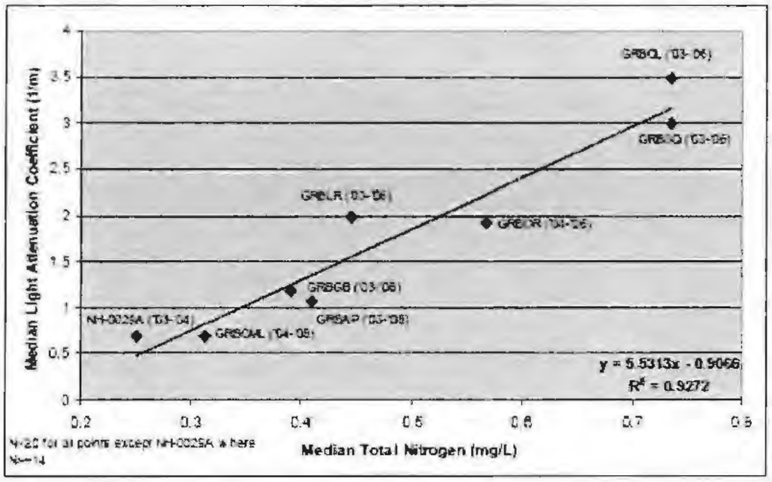


Figure 39 from the Criteria Document presents a regression of the measured light attenuation coefficient versus median total nitrogen at the Trend Stations. Based on this regression analysis, and targeting light penetration depth to support eelgrass populations, DES established a TN criterion of 0.3 mg/L. As with the other regressions, light attenuation is influenced by many other factors (e.g., color, turbidity) that were not considered when the data for all the Trend Stations were pooled to develop the regression. As a result, the analysis is not scientifically defensible. However, other data are available to confirm that this regression is only an artifact of the analysis. The data presented in Figure 13 show that median algal levels vary from about 1 – 7 µg/L through the system. These concentrations cannot physically cause the change in transparency suggested in Figure 39. Moreover, an independent study on the factors influencing transparency determined that chlorophyll-a is only a minor factor. (Morrison et al. 2008) Therefore, TN cannot cause the change in transparency presented in Figure 39.

Figure 39: Relationship between Light Attenuation Coefficient and Total Nitrogen at Trend Stations



The fundamental errors common to all of these analyses are:

1. The analyses combine data sets from greatly different physical settings; this is a simply not acceptable.
2. The predicted impacts from algal growth on transparency and DO are physically impossible, but that reality was not recognized by the document author.
3. None of the co-varying or confounding factors that must be considered to allow such regression analyses to produce reliable results were conducted.
4. The results are directly at odds with published State of the Estuary reports and tributary assessments confirming that TN has not caused material changes in algal growth nor is it controlling minimum DO, verifying these analyses have no connection to reality in this system.

The Criteria Document discusses the work of Morrison et al., 2008 (at 61) which confirmed that algal growth was a minor component affecting system transparency – as would be expected given the low algal growth in the system. That analysis confirmed that color from the tidal rivers was the main factor limiting light throughout the system. Color is NOT a factor influenced by the total nitrogen inputs to the system but is a natural condition occurring in certain watersheds throughout the country. The steady improvement in transparency through this system is most readily explained by dilution of color inputs from the tidal rivers – not any TN influence on excessive algal growth.

Likewise, with respect to system D.O., the Criteria Document (at 51) indicates that low D.O. in the Lamprey River is documented to be caused by the system hydrodynamics. However, this factor is nowhere assessed in any of the D.O.-related evaluations. Thus, it is clear that the report's conclusions based on these graphs are not scientifically defensible and fail to conform to even basic principles of environmental data analysis (i.e., to draw inferences from ecological responses to pollutants (such as nutrients), causal relationships and confounding factors must be identified and controlled in the assessment). This is a strict requirement to ensure that the analysis does not become confounded by factors unrelated to the variable of concern.[7]

Where complex and second order effects are involved, which may be controlled by a host of factors unrelated to nutrients (such as transparency and dissolved oxygen), the analysis must account for the other factors to demonstrate that the parameter of concern (in this case nutrients) is the parameter controlling the system response. No treatise accepts the position that it is proper to plot TN or chlorophyll a versus an instream D.O. concentration or measurement of transparency to demonstrate a scientifically defensible causal relationship. D.O., in particular, is easily affected by a dozen chemical, physical and biological factors that interact to cause a particular response.[8] Algal growth may affect dissolved oxygen via two routes: (1) diurnal changes due to plant photosynthesis and respiration and (2) creation of additional oxygen demand through cell death (e.g., sediment oxygen demand or "SOD"). However, neither of these factors are assessed. At a minimum, measurements of SOD could have confirmed whether algal growth is having any significant effect on this component. Likewise, transparency is controlled by four main factors: water, color, non-algal turbidity, and algal growth. There is no direct relationship between TN and transparency. Any regression showing such a relationship must first demonstrate the connection between transparency and chlorophyll-a, but no such relationship was provided in the Criteria Document.

Unless this is confirmed and quantified, the other factors known to be changing between the locations due to system hydrodynamics and differing external inputs could completely explain these graphs.[9] Such a sub-system response analysis would have provided the necessary level of confirmation that reducing TN levels will have a

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[7] It is a basic principle of environmental assessment and water quality criteria development that tests and evaluations are run under stable (steady state) conditions to ensure that the effect of the parameter of concern, and not some other changing variable, is occurring. The graph present a vision of "single parameter ecology" which is a uniformly rejected theory of data and ecological impact assessment.

[8] Thomann, R.V., Mueller, J. A. 1987. Principles of Surface Water Quality Modeling and Control. Harper-Collins; Chapra, S.C. 1997. Surface Water Quality Modeling, McGraw-Hill.

[9] HydroQual (2012) demonstrated that algal levels in the Squamscott River were heavily influenced by the discharge of algae from the Exeter lagoon system. The average impact on algal levels was approximately 6 ug/l. Since these algae do not grow in the system, it was totally inappropriate to plot data from the Squamscott River along with other tidal river algal levels and attribute those changes to TN inputs. As shown in Figure 16 (average monthly chlorophyll a levels for three system locations) the average algal in the Squamscott River (at Chapman's landing) ranges from 10- 14 ug/l June to September. Approximately 50% of this algal growth appears to be an artifact of the Exeter discharge. Eliminating this artifact would have resulted in a graph demonstrating little difference in algal growth between this tidal river and Adams Point in Great Bay. This would likely have had an even greater impact on Figure 17 given the importance of the Squamscott River data to the regression line.

demonstrable benefit to improving D.O. and transparency. At this point, the only thing that this analysis demonstrates is that as one moves from the tidal rivers to the ocean, minimum D.O. levels increase and transparency improves. That is a thoroughly unremarkable finding that would apply to almost any estuarine system since transparency is typically better and D.O. concentrations less variable in the ocean but poorer (often naturally) in the tidal rivers due to marsh and other watershed/system hydrodynamic influences.

In summary the analysis presented in the document entitled "Numeric Nutrient Criteria for the Great Bay Estuary" (2009) are (1) not based on methods generally accepted by the scientific community, (2) are contrary to the methods published in dozens of treatises on this topic (3) utilize obviously incorrect and physically impossible relationships attributed to algal growth and nitrogen influences and (4) are so thoroughly confounded and unexplained as to render them worthless for the purposes of numeric nutrient criteria development.

Acceptable Scientific Methods Governing Use and Application of Stressor-Response Methodologies

The following provides additional information regarding the degree of analysis necessary to allow this type of "stressor-response" assessment to be considered scientifically defensible and useful in nutrient criteria development.

The proper use of statistical methods to develop scientifically defensible nutrient criteria has been a highly controversial subject. In 2008, EPA began to apply regression analyses in an effort to set nutrient endpoints for use in TMDLs in lieu of site-specific modeling evaluations. At that time, I participated in an effort to get these methods reviewed by EPA's Science Advisory Board.

In August 2009, EPA released a draft Guidance document on use of the "stressor – response" approach to derive numeric nutrient criteria that recommended simply plotting the nutrient level versus various ecological endpoints (e.g., macroinvertebrate indices) under the assumption that the nutrients present in the water column were the cause of the change in the response variable (e.g., invertebrate index).[10] The fundamental scientific error impacting the validity and scientific reliability of this approach was that it presumed, rather than demonstrated "cause and effect." It is widely understood in the scientific community that response variables such as invertebrate indices and chlorophyll a level are impacted by a broad range of factors that may co-vary with nutrient levels. Moreover, as nutrients themselves are not toxics, one would, in general, need to first demonstrate that the nutrient level caused some change in plant growth that then caused a change in habitat and other water quality factors. This fact is reflected in an example "mechanisms" diagram contained in EPA's final stressor-response guidance, below.

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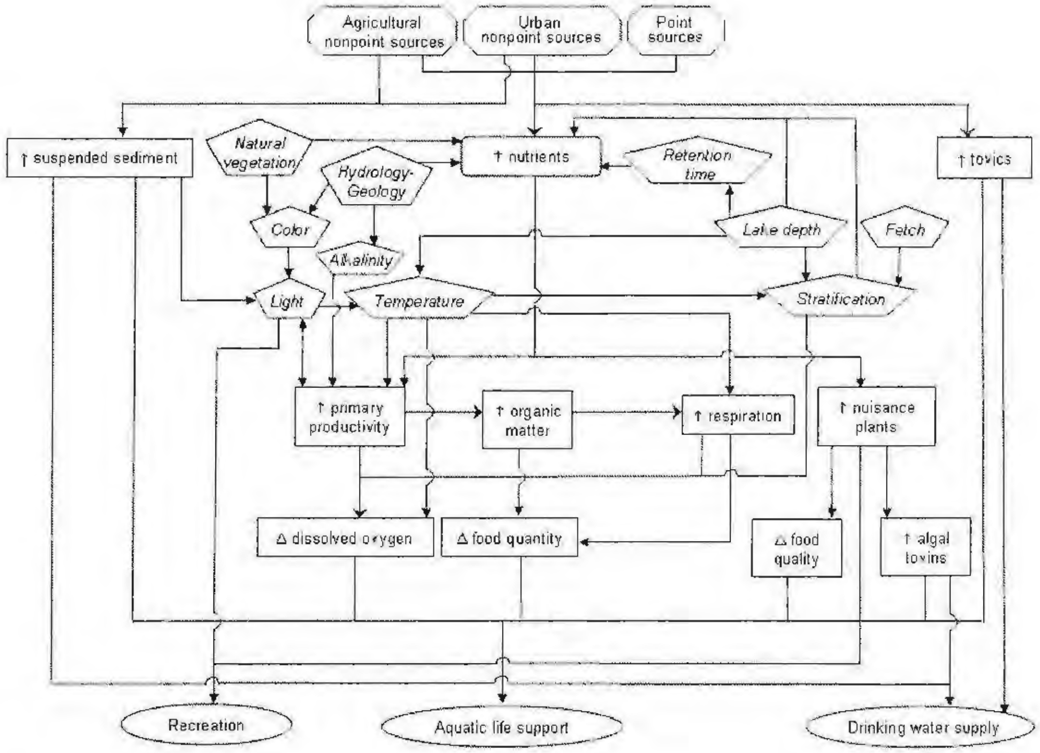
[10] Empirical Approaches for Nutrient Criteria Derivation (Science Advisory Board Review Draft) USEPA August 17, 2009.



EPA 2010 Stressor-Response Guidance at 10

Due to the numerous technical concerns voiced over developing nutrient criteria using these simplified methods, EPA used its Science Advisory Board (SAB) to conduct an independent peer review in September 2009 (three months after the 2009 Numeric Nutrient Criteria document was finalized by New Hampshire DES). Expert's from across the country were brought together to hear testimony and review the validity of EPA's approach. The SAB review clearly determined that the use of these methods for nutrient criteria development were not "scientifically defensible" unless major revisions and restrictions were incorporated to ensure that the statistical relationships reasonably reflected what was actually occurring in the receiving water.[11] In any event, the SAB determined that EPA's recommended approach to employing various simplified regression approaches to predict complex ecological response to nutrients were not scientifically defensible for a series of reasons including:

- The methods do not demonstrate "cause and effect";
- The methods failed to consider confounding and co-varying factors such as habitat and physical/chemical differences independently affecting the response variables;
- The methods failed to address first-order impacts (plant growth) that must precede any more complex impacts; and
- The statistical methods, by themselves, do not verify that the changes in condition

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[11] SAB Ecological Processes and Effects Committee, April 27, 2010 Final – Review of Empirical Approaches for Nutrient Criteria Derivation.

> are biologically significant.

In response to these criticisms, EPA significantly revised the draft stressor-response document and republished the methods in November 2010.[12] That document largely reflected the technical recommendations of the Science Advisory Board. Most importantly, EPA's final document specified that the methods would only be considered sufficient if data are available on "causal variables, response variables and confounding factors" (EPA Guidance @ 4). Absent such information, a "scientifically defensible' relationship generally cannot be developed. Ensuring that data are properly "classified" is a key factor for ensuring the evaluated relationship reflects nutrient impacts and is not unduly impacted by other changing ecological (confounding or co-varying) conditions (EPA Guidance @ 55, 56) Consequently, EPA notes that "many confounding factors must be considered when estimating the effects of nitrogen/phosphorus on a measure of aquatic life in streams (e.g., macroinvertebrate index)." (EPA Guidance @ 11) This concept applies also to endpoints such as D.O. and transparency that are not directly influenced by nutrients. Consequently, EPA includes extensive discussion on the importance of properly conducting the "confounding factors" analysis and further indicates that when parameters co-vary (such as nutrients, color, turbidity, solids, algal levels) it is critical to determine which parameter is actually controlling the response variable. (EPA Guidance @ 26-29).

The following quotes from EPA's guidance document further illustrate the methodology that must be used and factors that must be considered to ensure a "stressor-response" assessment is scientifically defensible:

Recommendations from 2010 USEPA Stressor-Response Guidance

> **Need to ensure Data Evaluation is Only Conducted for Similar Ecological Settings**
>
> [I]n the first step of the analysis, classification, the analyst attempts to control for the possible effects of other environmental variables by identifying classes of waterbodies that have similar characteristics and are expected to have similar stressor-response relationships. Classifications for a stressor-response analysis are typically based on statistical analysis; however, existing classes can be used as a starting point. The most widely used existing classification for analyses of nutrient data are the fourteen national nutrient ecoregions.
>
> (EPA Stressor-Response Guidance at 32)
>
> Classifying data is a key step in analyses of stressor-response relationships because the expected responses of aquatic ecosystems to increased N and P can vary substantially across different sites.
>
> (EPA Stressor-Response Guidance at 55)

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[12] Using Stressor response Relationships to Derive Numeric Nutrient Criteria, USEPA November 2010.

> The first step for classifying data is to identify variables to include in the analysis that will help improve the accuracy and precision of estimated stressor-response relationships.
>
> * * * * *
>
> [E]xploratory data analysis can indicate other variables that should be included in the classification analysis. In particular, other variables that are strongly correlated with the stressor variable or with the response variable should be evaluated for inclusion in classification analysis.
>
> (EPA Stressor-Response Guidance at 56 – 57)
>
> **The Impact of Confounding and Co-varying Factors Must be Assessed**
>
> [M]any confounding variables must be considered when estimating the effects of nitrogen/phosphorus pollution on a measure of aquatic life in streams (e.g., a macroinvertebrate index).
>
> (EPA Stressor-Response Guidance[13] at 11)
>
> [W]hen the effects of a possible confounder are not controlled, the relationship estimated between the nutrient variable and the response variable may partially reflect the unmodeled effect of the confounding variable.
>
> (EPA Stressor-Response Guidance at 65)
>
> The possible influences of confounding factors are the main determinants of whether a statistical relationship estimated between two variables is a sufficiently accurate representation of the true underlying relationship between the two variables. ...
>
> Before finalizing candidate criteria based on stressor-response relationships, one should systematically evaluate the scientific defensibility of the estimated relationships and the criteria derived from those relationships. More specifically, one should consider whether estimated relationships accurately represent known relationships between stressors and responses and whether estimated relationships are precise enough to inform decisions.
>
> (EPA Stressor-Response Guidance at 65)
>
> Beyond the possible effects of confounding variables, one should also consider whether assumptions inherent in the chosen statistical model are supported by the data.
>
> (EPA Stressor-Response Guidance at 67)

The 2009 Numeric Nutrient Criteria document clearly did not meet any of these pre-requisites for applying simple linear regression analysis in the development of numeric

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[13] EPA. November 2010. Using Stressor-response Relationships to Derive Numeric Nutrient Criteria. EPA-820-S-10-001.

nutrient criteria. The findings presented in the Criteria Document are based on procedures that the SAB rejected, which is not surprising given the timing of its development (pre SAB).

A cursory review of the 2009 Numeric Nutrient Criteria Document confirms that is did not rely on accepted, scientifically defensible methods. The evaluation errors were extensive and included virtually every major factor that EPA has identified in its final Stressor-Response guidance document, including:

- Combining data from different biotypes that affect D.O. and transparency;
- Failing to consider co-varying pollutants and parameters;
- Failing to evaluate key confounding factors;
- Presuming that the pollutant was the cause of the changing system response parameter when the available data confirmed it was not; and,
- Failing to assess the accuracy and reliability of the suggested relationships based on data and studies from specific areas within the Great Bay system.

Is the Department's use of simplified regression methods scientifically defensible and consistent with accepted scientific methods?

The short answer is clearly - no. The key to the proper/defensible use of the stressor-response methods lies in addressing the factors that could otherwise explain the relationship being assessed. Since both DO and transparency are affected by numerous ecological, chemical and biological factors, any valid defensible assessment must reasonably account for these factors, prior to reaching any conclusion that nutrients are the primary cause of changing transparency and D.O. in this system. Both the SAB and EPA itself have identified the prerequisites that must be met to utilize these methods to produce reliable and scientifically defensible results. The Department has plainly failed to address the confounding factors and similar system prerequisites and has simply ignored other admonitions contained in the SAB report and the applicable federal guidance regarding proper use of this method.

Moreover, as an expert in the field of environmental impacts and effects analysis, I am aware of no treatise that would support the position that an acceptable analysis may plot data from multiple habitat types with major hydrologic difference on the same graph in assessing complex ecological phenomena. Consequently, the estuary-wide nutrient criteria generated by using the approach described in the Department's technical report is not scientifically reliable, not scientifically defensible, not a method generally accepted within the scientific community and has produced a result that is, consequently, demonstrably incorrect.

I swear that the forgoing statements are true to the best of my knowledge.

Steven C. Chapra, Ph.D., F.ASCE

STATE OF MASSACHUSETTS

COUNTY OF MIDDLESEX

Signed and sworn to before me on this 27TH day of February, 2013 by Steven C. Chapra.

Notary Public

My Commission Expires: August 10, 2018

(Notary Seal)

NANDI P. BYNOE
Notary Public
Commonwealth of Massachusetts
My Commission Expires
August 10, 2018

Notarized this Day, 27 FEB 2013

Attachment C

SPATIAL AND TEMPORAL PATTERNS IN NUTRIENT STANDING STOCK AND MASS-BALANCE IN RESPONSE TO LOAD REDUCTIONS IN A TEMPERATE ESTUARY

BY

JASON SETH KRUMHOLZ

A DISSERTATION SUBMITTED IN PARTIAL FULFILLMENT OF THE
REQUIREMENTS FOR THE DEGREE OF
DOCTOR OF PHILOSOPHY
IN
OCEANOGRAPHY

UNIVERSITY OF RHODE ISLAND
2012

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DOCTOR OF PHILOSOPHY DISSERTATION

OF

JASON KRUMHOLZ

APPROVED:

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Major Professor_________Candace Oviatt____________

________Scott Nixon________________

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_______Nasser Zawia___________
DEAN OF THE GRADUATE SCHOOL

UNIVERSITY OF RHODE ISLAND
2012

ABSTRACT

The addition of excess organic matter into a system, commonly referred to as eutrophication (Nixon, 1995), is a widespread problem in estuaries throughout much of the world. To combat this trend, many management agencies are imposing regulations limiting the amount of nutrients (nitrogen and phosphorus) which can be discharged into coastal waters through wastewater treatment and agriculture. In 2005, the Rhode Island Department of Environmental Management (RIDEM) enacted legislation mandating that wastewater treatment facilities (WWTF) discharging their effluent into Narragansett Bay and its tributaries reduce the concentration of nitrogen in their effluent. This legislation will reduce wastewater nitrogen loading to the bay by 50% by 2014 with the ultimate goal of improving water quality, reducing hypoxia, and restoring lost ecosystem services (e.g. seagrass) to the bay. Early stages of this reduction took place between 2005-2009, reducing loadings at 11 WWTF's which discharge into the bay from 16-20mg/l total nitrogen to either 8 or 5mg/l.

Response of other estuaries to similar reductions in loading has been varied and complex, with relatively few ecosystems showing straightforward linear reductions in concentration, productivity, and chlorophyll with reduced load. The overall goal of this study is to quantify the impact of these initial loading reductions on the standing stock (Chapter 1), seasonal cycling (Chapter 2), and mass-balance (Chapter 3) of nitrogen and phosphorus in Narragansett Bay.

To accomplish this goal, we first reviewed data from a five-year study of surface nutrient concentration at 13 stations throughout Narragansett Bay (Chapter 1). Because Narragansett Bay is aligned along a north-south gradient of decreasing

urbanization and most sources of nutrients to the bay are located in or around the city of Providence, at the head of the estuary, we can establish down-bay relationships of nutrient constituents to see how their concentrations change spatially throughout the bay, and compare these relationships to past studies. We can also use established volume relationships to estimate the total standing stock of nutrients in the bay at any given time, and compare how this changes over the course of a year during the present survey and during past surveys. In response to a 30% reduction in the total annual load of dissolved inorganic nitrogen from all sources, which corresponds to a 17% reduction in total nitrogen, we saw measurable reductions in downbay concentrations and standing stocks approximately on par with these reductions. Phosphorus concentrations in the bay have declined dramatically (30-50%) in part due to recent loading reductions, but also in part due to management action in the 1980's and 1990's to remove phosphates from detergents and industrial surfactants. We also see changes in the way nitrate, nitrite, and ammonium are used on a downbay gradient, which we hypothesize are related to the loading reductions.

In order to fully understand the impact of load reductions on the ecosystem, we must also consider how the nutrients in the system have changed over the long-term, both in terms of annual cycling, and in terms of response to changing climate in the bay. This analysis constitutes the second chapter of the dissertation. Over the last 50-100 years, Narragansett Bay has grown measurably warmer, and weather patterns have changed, bringing increased cloud cover, more storms, and more precipitation. All of these changes impact the way nutrients enter the bay, and the way phytoplankton use the nutrients. We examined the impact of these potential changes using a long-term weekly dataset of nutrient concentrations collected by the MERL lab at the University of Rhode Island Graduate School of Oceanography since 1978. We use both conventional statistics and a state-space model formulated in the computing language R (SSPIR). Our results show virtually no long-term trend or change in timing of seasonal cycling of nutrients or chlorophyll. However, we do see changes in the seasonal patterns of concentration of both nutrients and chlorophyll at the GSO station, with measurable changes in cumulative distribution function for phosphate, silicate, ammonium, and chlorophyll. We also observe statistically significant reductions over the course of the time series for nitrate, nitrite, ammonium, and phosphate, though it is difficult to ascribe causality to these changes. Model results were largely inconclusive, but show a marginally significant intervention effect attributable to the loading reduction in the ammonium signal at the GSO dock, with no significant long-term trend observed for any analyte.

Finally, we conduct a mass-balance nutrient budget assessment for nitrogen and phosphorus in Narragansett Bay (Chapter 3). Mass-balance is a common way of tracing the sources, sinks, and reservoirs of nutrients in a system, and seeing how these components might change with time. Nutrient budgets for Narragansett Bay have been compiled approximately every decade, but recent and future loadings compel a reanalysis to determine how the system is responding to initial stage reductions. We see a reduction in WWTF loading to the bay of just over 100 million moles of nitrogen and 4 million moles of phosphorus, which constitutes about 20 and 16 percent of the net annual load of nitrogen and phosphorus from all sources. However, much of this reduction is realized in tributary rivers, and variable riverine abatement rates in those rivers mean that some of the net reduction is not felt by the bay proper. Furthermore, evidence from literature suggests that changes in bay sediment net denitrification rate may be offsetting some or all of the loading reductions.

ACKNOWLEDGMENTS

It is the academic tradition for acknowledgements to be listed at the end of a peer reviewed manuscript, yet the irrefutable and iron clad university of Rhode Island dissertation format template lists acknowledgements first. I would like to think that the reason for this is because without the support of so many people, no dissertation, least of all this one, could possibly be completed. Dissertation writing is a marathon, not a sprint, and no marathoner can be successful without a great support team. So, before we dive headlong into the results of several (many?) years of hard labor, I, like many before me, would like to take a moment to thank all of those who made this milestone possible for me.

Virtually every acknowledgement section (including the acknowledgement section of my advisor) begins with some sort of remark about the thesis advisor's unending patience, and here too I will not disappoint. My mentor, Dr. Candace Oviatt, has shown near infinite patience for the tortuous path down which we have traveled together in pursuit of this degree. Make no mistake, this has not been a flat and straight pavement marathon. This has been an up and down, through mud and rocks endurance event. Yet through all the highs and lows, your steady hand and composed demeanor have kept me on my feet and moving (generally) forwards with a smile on my face. I cannot thank you enough.

To the rest of my dissertation committee, Drs. Scott Nixon, Jeremy Collie, Art Gold, and Graham Forrester I also extend my sincerest gratitude. You have opened your doors to me, shared your research and life experiences with me, from DGS and IGERT to the BVI, to words of encouragement in the hallways of GSO and the corridors of the YMCA, you have always been there for me, often with a wisecrack at the ready. I consider myself a better and more thorough researcher and scientist thanks to each of your influences.

The GSO community as a whole has been a remarkable place to spend the last several (many?) years. Virtually every door is open, and any faculty, staff or student is willing to help in any way possible. For this assistance, in whatever form it takes, from technical help to logistical help, to friendship and emotional support I am extremely grateful. To list each and every person who has contributed to the work on the following pages would take virtually as long as the manuscript itself. I am, however, particularly indebted to my group of GSO peers with whom I have walked this road, and with whom I have learned, that science, like floor hockey, is truly a team sport. Particular thanks among this group for guidance and technical assistance with this dissertation are due to Wally Fulweiler, Matt Horn, and Rich Bell.

Team MERL has been a fixture in my graduate tenure, and I could not be prouder of my time in MERL, nor think of a better group of individuals with which to work. I have benefited greatly from my association with dozens of MERL alumni from the very first pioneers in the 70's to those of us who still man the tiller and keep the ship afloat to this day. Particular thanks among this group are due to Chris Calabretta, Brooke Longval, Heather Stoffel, Edwin Requintina, Conor McManus, Jeff Mercer, Leslie Smith, Matt Schult, Chris Melrose and Kim Hyde. Also to MERL Interns and technicians who assisted with the data collection and analysis for my dissertation: Ashley Bertrand, Danielle Dionne, and Rossie Ennis.

I would also like to thank all the faculty, staff and students of the Coastal Institute IGERT program for your support and guidance. Being able to see the entire project through from the beginnings as a member of the 'guinea pig' class of Co-05 to offering graybacked veteran support for later cohorts was a tremendous experience. The skills and perspective I have acquired through this program have shaped my path through grad school and continue to shape my career goals. I am particularly indebted to the tireless work of Pete, Judith, Q, Deb, Jim, Candace, and Art, as well as the Co-05 and '06 cohorts with which I shared my 'active duty' rotation. Thank you for showing me what it means to be truly interdisciplinary.

There are many collaborators whose willingness to exchange data and ideas have greatly improved the quality of the manuscripts herein. I would like to thank Angelo Liberti and the scientific staff at RIDEM and NBC for sharing data and ideas, brainstorming, and helping shape the outputs of this work. I would also like to thank collaborators Claus Dethlefsen and Jamie Vaudrey.

Last, but certainly not least, I want to thank all of my friends and family. The unflagging support of my wife Emily and the exuberant smile of my wonderful son Charlie have been a constant ray of light. This victory is as much yours as it is mine, and I promise, as soon as this thing is done, to do better with the housework. Thanks to my dad, Alan, for his sense of humor and for helping me to focus on the important things, to my mom Robin, for always believing in me and encouraging me to follow my dreams, and to my brother, Steven, for never questioning my motives, but always questioning my methods, and being my unfailing allies for 32 (28) years. I love you all so much.

DEDICATION

This work is dedicated to my son Charlie. I hope that in some small way, we can contribute to the furthering of the science associated with sustainable use and management of marine ecosystems on behalf of yours and future generations. It is my sincerest hope that we will be able to devise sound management practices for the sustainable use of marine resources, such that the wonderful mysteries of the ocean will continue to yield a sufficient spawning stock of research questions to support sustainable harvest of dissertation topics for generations to come.

I fancy myself to be a decent writer of this sort of thing, but I think your friend Dr. Seuss says it best:

"… now that you're here, the word of the Lorax seems perfectly clear. UNLESS someone like you cares a whole awful lot, nothing is going to get better. It's not."

-The Old Once-Ler

PREFACE

As described in the URI Graduate School guidelines for thesis preparation, this thesis is organized in a manuscript format. The body of the text is divided into three sections, corresponding to the format of journal articles. The first manuscript is submitted to Estuaries and Coasts, with co-author Candace Oviatt. The second manuscript will be formatted for Northeastern Naturalist, and will be submitted with co-authors Candace Oviatt, Rich Bell, and Claus Dethlefsen. The third manuscript will be submitted to Estuarine Coastal and Shelf Science, and will be co-authored by Candace Oviatt, Jaimie Vaudrey, Scott Nixon, and Rosmin Ennis. There are three appendices, divided into A) supplemental methods, B) Plant and River discharge calculations, C) Matlab and R scripts for code used within the chapters. The appendices provide additional background that was excluded from the chapters for brevity's sake.

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CHAPTER 1

CHANGES IN NUTRIENT STANDING STOCK IN A TEMPERATE ESTUARY WITH DECREASED NITROGEN AND PHOSPHORUS LOADING

ABSTRACT

We review the initial impact of decreased summer nitrogen and phosphorus loading between 2004 and 2007 into Narragansett Bay, RI. Biological nitrogen removal at 11 of 29 sewage treatment facilities which discharge their effluent either directly into Narragansett Bay or into its tributaries has reduced effluent nitrogen concentration at those plants by half or more during summer months. This results in a 30% decrease in the inorganic load and a 17% decrease in the total annual nitrogen load to the system. The reduction in load is visible in a reduction of the standing stock of dissolved inorganic nitrogen, but no statistically significant change in total nitrogen in the bay over time was detected. We do see significant differences in downbay patterns of dissolved and total nitrogen when compared by analysis of covariance (ANCOVA), as well as several interaction effects, which may be an indication that utilization patterns are changing. In contrast, dissolved inorganic phosphorus shows a consistent reduction throughout the bay, likely caused by a combination of legislative efforts in the 1990's and removal of phosphorus at several treatment plants which discharge into tributary rivers. Taken together, our data indicate that the early response of the ecosystem to reduction is within the bounds of what might be expected, particularly given high inter-annual variability in nutrient concentrations.

Keywords: Nitrogen, Phosphorus, Nutrients, Management, Hypoxia, Estuary

INTRODUCTION

In the 21st century anthropogenic pressure on coastal ecosystems continues to grow. Despite accounting for only 17% of the land in the continental United States, coastal counties account for over 153 million people (53%), a number which has increased by more than 30 million since 1980, an increase of roughly 28% (Crossett et al. 2004). Many of our nation's largest cities, particularly on the East coast, are positioned on or near estuaries, which brings great benefit in terms of commerce, industry, recreation, and tourism, but also great responsibility, as estuarine ecosystems are both highly productive, and highly sensitive to change. A recent review of literature on our nation's estuaries found 64 out of the 99 estuaries assessed exhibited moderate to high levels of anthropogenic enrichment, with 65% of systems for which data were available predicted to worsen by 2020 (compared to 19% predicted to improve) (Bricker et al. 2007). The same assessment found the Mid-Atlantic Region (Cape Cod to Virginia) to be the most impacted region in the country, with 20 out of 22 estuaries considered moderately or highly eutrophic, and eight systems declining since 1999, while only one (Gardiners Bay) improved (Bricker et al. 2007).

Fortunately, as awareness about anthropogenic impact on coastal water bodies grows, an increasing number of management organizations are beginning to consider measures to limit nutrient input to estuaries, in the hopes of addressing the many impacts of increased eutrophication, such as hypoxia, reduced water quality, loss of SAV (submerged aquatic vegetation), beach and fishing closures, etc. (Carstensen et al. 2006, Deacutis 2008, Duarte et al. 2009, Dam et al. 2010). The implementation of tertiary, or 'advanced' wastewater treatment techniques at wastewater treatment facilities (WWTF's) (defined herein as processes, whether biological, chemical, physical or any combination of those, which remove nutrients from wastewater effluent prior to discharge), often referred to as biological nutrient (or nitrogen) removal (BNR), is one such method which is being implemented widely, as increases in technology and utilization drive the cost of this treatment down and its efficacy up (Lishman et al. 2000, Jeong et al. 2006). The efficacy of this management option to generate system wide improvements in water quality is a topic of great interest to scientists and managers alike.

Decreased nutrients have had dramatically different patterns in different ecosystems. While in some cases management strategies to reduce nutrient loading have resulted in rapid declines in nutrient standing stocks, in many cases the ecosystem responds either slowly, or less dramatically than anticipated (e.g. Artioli et al. 2008; Carstensen et al. 2006; Boynton et al. 2008; Nixon 2009), which is attributed to a wide range of causal factors, including sediment release, shifting baselines, and non-linear response types (e.g. Duarte et al. 2009, Taylor et al. 2011). The lack of predictable response is particularly evident with respect to the use of BNR in WWTF's to mitigate hypoxia in estuarine waters. While certain key physical parameters (e.g. residence time, stratification, temperature, etc.) are causally linked to hypoxia (e.g. Codiga et al. 2009, Rabalais et al. 2009, Bianchi et al. 2010), the direct link between changes in nutrient supply and reduced hypoxia is weak, ecosystem specific, and often nonlinear (Artioli et al. 2008, Kemp 2009).

Decreased nitrogen and phosphorus loading to the bay may cause a wide range of ecological impacts, ranging from straightforward to more complex. At the most basic level, reduction of loadings may cause a subsequent drop in the standing stock and total annual budget of nitrogen and phosphorus in the bay, or it is possible that other terms of the nutrient budget (e.g. sediment and water column recycling) may change to preserve the overall standing stock and annual budget (Carstensen et al. 2006, Fulweiler et al. 2007, Duarte et al. 2009). While re-mineralization of sediment nutrients has been implicated as a possible mechanism for delayed response in some heavily impacted ecosystems (Carstensen et al. 2006, Clarke et al. 2006), in other ecosystems (Boynton et al. 2008), including a mesocosm study in Narragansett Bay (Oviatt et al. 1984) the sediments have a short memory, and the ecosystem responds rapidly to changes in nutrient loading. Reduction of nutrients may result in a decrease in primary productivity in some or all regions (Carstensen et al. 2006, Boynton et al. 2008), a change in nutrient ratios which may impact the frequency with which a given nutrient (N,P, or potentially even Si) is limiting and/or cause a shift in the phytoplankton species assemblage (de Vries et al. 1998, Turner et al. 1998, Tomasky et al. 1999, Artioli et al. 2008). Nutrient reduction may lead to a decrease in the extent or severity of hypoxia in the bay by reducing primary productivity, and therefore export of organic matter to the benthos, or alternatively, the supply of nutrients and organic matter may not be limited, and/or variability in hypoxia may be driven primarily by physical forcings (Robinson and Napier 2002, Codiga et al. 2009, Duarte et al. 2009, Kemp 2009). The combination of these many variables makes it difficult to predict how future oligotrophication of the bay will impact its ecology (e.g. Nixon 2009, Nixon et al. 2009)

With increased awareness of the potential impacts of low oxygen conditions in the Providence River Estuary, Upper Bay, and Greenwich Bay, Rhode Island Department of Environmental Management (RIDEM) has required that several of the major sewage treatment plants which serve Narragansett Bay be upgraded to tertiary sewage treatment, with most other large plants planning upgrades in the next few years (RIDEM 2005). The overall goal of RI General Law § 46-12-3(25), the driving force behind these changes, is to reduce nitrogen loading to the bay from WWTF's by 50%, a task which, based on percentage reductions achieved at the plants which have already upgraded, will be achieved once the largest plant discharging into the bay, located at Fields Point (Fig. 1-1) completes upgrades, presently scheduled to be sometime in late 2013 or 2014.

Plants that have upgraded use bacterially mediated coupled nitrification/denitrification to convert ammonium to nitrate and nitrite aerobically, then anaerobically to di-nitrogen gas, which is out-gassed to the atmosphere (Lishman et al. 2000, Jeong et al. 2006). This process has reduced rates of ammonium discharge at some plants by nearly an order of magnitude, and DIN concentrations by more than half during summer months (Liberti, unpublished data), since the rate of bacterially mediated denitrification is temperature dependent (e.g. Dawson and Murphy 1972, Lishman et al. 2000, Pell et al. 2008). The implementation of a combined sewer overflow reservoir in 2008 has further reduced nutrient input during high flow periods by delaying storm water runoff, and running it through treatment plants before discharge into the bay. The combination of these factors has reduced annual sewage based total nitrogen loading by 27% (Table 1-1) which constitutes a reduction of approximately 15% of the annual TN load to the ecosystem(Nixon et al. 2008).

Phosphorus loading reductions have also markedly decreased over the past decades, but this reduction is due in large part to legislative changes during the 80's and 90's, in particular RI general law § 49-26-3, passed in 1995, which dramatically limited the use of phosphate in detergents (Litke 1999). Several of the WWTF's that discharge into tributary rivers rather than directly into the bay have undertaken phosphate removal efforts to reduce loading from their effluent. In many cases, these efforts have been highly successful, removing upwards of 80-90% of the phosphate from effluent (see Appendix B). However, the impact of this reduction on the overall phosphorus budget of the bay (see chapter 3) is not large, in part because WWTF's contribute a smaller percentage of the overall phosphorus budget of the bay than for nitrogen, and in part because the plants with the largest phosphorus reductions are not the largest in terms of volume or total phosphorus flux.

By reviewing the impact of this management action on the standing stocks of dissolved inorganic nitrogen and phosphorus (DIN and DIP) as well as total nitrogen and phosphorus (TN and TP) in the bay, we can gain a better understanding with respect to how the ecology responds, on the short term, to changes in nutrient loading and compare our results with those observed in other ecosystems In this paper, we examine the short-term impact of a large (≈30% of annual sewage based N loading, (Table 1-1)) reduction in nutrient loading on nutrient standing stocks in Narragansett Bay, RI resulting from the implementation of advanced wastewater treatment at several facilities discharging either directly into the bay, or into tributary rivers.

STUDY SITE

Narragansett Bay, including Mount Hope and Greenwich Bays, but not the Sakonnet River (which is connected to the bay proper by only a very small channel and has very limited exchange) is a relatively shallow (average depth 8.6 meters) temperate estuary of approximately 328 km^2 (Pilson 1985a). Freshwater input is relatively low, approximately $100m^3s^{-1}$, and circulation is predominantly tidally driven, with ocean water typically moving in the east passage, and out the west passage (Kincaid et al. 2008). As a result of the combination of these factors, and the generally shallow depth of the bay (with the exception of the lower parts of the East Passage), Narragansett Bay is typically only weakly stratified throughout most of its mid to upper reaches, and salinity remains high (>20 psu) throughout virtually the entire estuary, and increases on a generally north-south gradient to roughly 32psu at the bay mouth (Pilson 1985a, Kincaid et al. 2008).

In Narragansett Bay a significant amount of historical baseline data exists on nutrient dynamics in the bay, through field studies (Nixon et al. 1995, Nixon et al. 2008), and through experimental treatments in the MERL mesocosms (e.g. Oviatt 1980, Nowicki and Oviatt 1990, Oviatt et al. 1995, Oviatt et al. 2002). Past research indicates that the bay is a nitrogen limited ecosystem, with a strong North-South gradient of nitrogen and phosphorus concentration caused by WWTF and river inputs, which are the two largest sources of these nutrients and which are concentrated in the Providence River and Upper Bay (e.g. Nixon et al. 1995, Oviatt et al. 2002). Previous nutrient budgets suggest that the bay was a net autotrophic ecosystem, and that the majority of the nutrients exported into Rhode Island Sound from the bay are in inorganic form, rather than as organic material (Nixon et al. 1995). Compared to other temperate estuaries, Narragansett Bay has a relatively densely populated watershed, and about 63% of the total nutrient flux into the bay comes directly or indirectly (via rivers) from WWTF's (Nixon et al. 2008), as compared to an average for 74 temperate estuaries of about 36% (Latimer and Charpentier 2010).

Primary production in the bay has been estimated at about 320 gC/m^2 on a baywide average (Oviatt et al. 2002) and the community is phytoplankton dominated, traditionally experiencing a strong winter/spring diatom bloom, and several subsequent blooms throughout the summer which are lesser in intensity, duration and areal extent (Nixon et al. 1995, Oviatt et al. 2002, Smith et al. 2010). The frequency and intensity of this winter/spring bloom has declined over the past several years, and has not occurred at all in some years (e.g. Oviatt et al. 2002, Oviatt 2004, Smith et al. 2010), although in the last few years the ecosystem has experienced large winter diatom blooms correlated with colder winter water temperatures. Furthermore, average chlorophyll levels have also been generally trending downward, with a 70% drop reported for a mid-bay site since the early 1970's (Fulweiler and Nixon 2009), though again, with the return of the Winter/Spring diatom bloom, this trend may also be reversing in Narragansett Bay and other similar Northeast U.S. estuaries (e.g. Dam et al. 2010). Given some evidence of changes in NAO it is reasonable to suspect that New England may see more years with strong winter-spring blooms than without in the near future (e.g. Knight et al. 2005, Keenlyside et al. 2008).

In addition to the above mentioned loading reductions, significant changes in the climate and phenology of the bay over the last several decades have been documented (Oviatt 2004, Melrose et al. 2009, Smith et al. 2010). Over the last century, we have seen an annual precipitation increase of over 30 cm/y (nearly a 30% increase) (Pilson 2008, Melrose et al. 2009), and the frequency of severe precipitation events has increased nearly 90% (Madsen and Figdor 2007). Over the last half century, average water temperature has increased by 1.2^{o}C and the average number of cloudy days per year has increased by 61 (Melrose et al. 2009). These shifts in climate have impacted the way that nutrients cycle through the bay and are taken up by biota, sequestered in sediments, recycled, and flushed from the bay (Pilson 2008, Fulweiler and Nixon 2009, Nixon et al. 2009). In addition, the intermittency of the Winter/Spring bloom in many recent years, may contribute to variability in nutrient standing stocks during this time period (e.g. Li and Smayda 1998, Oviatt et al. 2002, Oviatt 2004, Fulweiler et al. 2007).

We aim to compare downbay concentration gradients and total standing stocks of nitrogen and phosphorus in the bay since the implementation of advanced wastewater treatment with past studies of the bay to determine if and how the WWTF upgrades have impacted the distribution and standing stocks of nutrients in the bay. We will also investigate how chlorophyll has responded to changes in nutrient stocks. This exercise will help us to understand which areas of the bay (if any) are most susceptible to changes from present and future reductions in nutrient load.

METHODS

Surface nutrient samples were collected from 2006-2010 (inclusive) at thirteen stations throughout the bay (Fig. 1-1) representing a broad geographical coverage including four stations each in the East and West Passages, three stations in the

Providence River Estuary, a station at the mouth of the bay south of Jamestown, and a station in Mt. Hope Bay. Samples were collected monthly on cruises using the RIDEM R.V. John Chafee, supplemented with additional biweekly summer (May-September inclusive) sampling using the Marine Ecosystem Reserarch Laboratory (MERL) 20' Wellcraft. Surface samples were collected (by bucket), and stored in 1L opaque polycarbonate bottles on ice until returned to the MERL facility for processing. Since the cruise track did not go into Greenwich Bay, a small dataset of nutrients collected at the Greenwich Bay DEM fixed monitoring network site was used (Figure 1-1). Apart from the sporadic nature of the collection dates at this site, these samples were processed identically to regular cruise samples and run on the same instrument.

Immediately upon returning to the lab, a 40 ml aliquot from each station was filtered (by 0.45 micron nucleopore filter using a syringe) for dissolved inorganic nutrients (NO_2, NO_3, PO_4, NH_3, and SiO_4), and a 40 ml whole water aliquot was collected for total nutrients (TN and TP). Samples were frozen at $-4^{o}C$ prior to analysis. Total nutrient samples were extracted using the Alkaline Persulfate method (Valderrama 1981, Patton and Kryskalla 2003). Traditional colorimetric analysis techniques were used for each analyte modified slightly to achieve maximum accuracy and precision on each instrument (Table 1-2).

From 2006-2008 samples were analyzed on a Technicon autoanalyzer. Beginning in 2009, samples were analyzed on a newly purchased Astoria SFA autoanalyzer. A thorough intercalibration between the two instruments was conducted prior to the switch-over, with samples from 1/09-6/09 as well as additional intercalibration test samples, run on both instruments. All analytes with the exception of nitrate and total nitrogen were directly comparable between instruments with no correction. Nitrate (and total nitrogen, which is run on the nitrate channel) required the implementation of an empirically derived correction factor, after which results were directly comparable.

In all cases, yearly averages were first computed by calculating monthly averages from each station, to avoid biasing toward the more heavily sampled summer period. In order to fill data gaps caused by missed sampling cruises or lost/damaged samples, gaps at a given station of less than 2 samplings were linear interpolated. Infrequent gaps of more than 2 samplings were filled by averaging the values for all samples collected in the month in question during other years in the survey (See Appendix D for more details). This was done to avoid bias in yearly averages caused by the presence or absence of sampling in a given month (particularly December and January, where sampling was often infeasible due to weather, and concentrations are typically highest).

Data were natural log normalized (to meet the linearity assumptions of tests used) and spatial patterns in nutrient concentration on a downbay gradient were compared between the present study and past studies at similar sampling locations (Fig. 1-1) (Oviatt 1980, Oviatt et al. 2002). This analysis was performed with analysis of covariance (ANCOVA) in MATLAB, where distance downbay from Field's Point (the furthest north sampling station) is the covariate, and the slope and intercept of the linearized downbay gradient in concentration were compared both within the present study and between this study and past studies. Analysis of covariance essentially functions as a combination of a regression and an analysis of variance (ANOVA), by removing the variance associated with the covariate (distance downbay) and then conducting an ANOVA. This can greatly increase the power of the ANOVA by removing the variability attributed to the covariate (in this case, an order of magnitude or more).

Standing stocks were calculated by multiplying surface nutrient concentration by volumes for each section of the bay derived from the General Ecosystem Model (GEM) box model (see Kremer et al. 2010). In cases where a model box did not have an associated station, or had more than one station the numerical average of stations in surrounding boxes, or the numerical average of all stations in the box was used, respectively. The GEM model does have separate surface and bottom boxes for each element, but we elected to use surface nutrient values only because only a very limited number of bottom samples were collected as part of this study, and no relationship could be established between surface and bottom values. Data from past studies (e.g. Kremer and Nixon 1978, Oviatt 1980) indicate that surface and bottom values are frequently very similar (since the water column is often well mixed), and in times when they vary, these datasets do not provide a consistent relationship between surface and bottom to justify developing an algorithm to calculate bottom values. A recent study by Hefner (2009) using data from two mid-bay stations confirmed that surface and bottom nutrient levels are highly correlated, and residuals were not easily explained.

Water samples from the DEM Buoy station in Greenwich Bay were used for both GEM boxes in Greenwich Bay, however, since sampling frequency at this station (particularly in winter months) was highly sporadic, it was not feasible to calculate averages for each year of the study, rather, a single average was computed for the period of 2006-2009 by averaging all monthly samples collected in a given month (n= 2-12) during the sampling period, and using these monthly averages to calculate annual and summer (June-Sept.) averages during the sampling period. Due to its comparatively low volume, these two boxes contribute less than 2% of the total baywide standing stock, so the lack of precision in this region is unlikely to significantly influence results.

Annual and summer standing stocks were compared to each other and to prior standing stocks estimated by applying the methods above to data from the 1979-1980 survey (Oviatt 1980) for inorganic nutrients and the 1998 survey (Oviatt et al. 2002) for total nutrients. Statistical comparisons were two tailed T-tests using SigmaPlot. Prior to analysis, the Shapiro-Wilk test was used to confirm normality. Because only one year of data was available for past studies, equal variance was assumed for these tests, while unequal variance was assumed when testing summer vs. annual standing stock.

Spatial maps of major nutrient constituents were calculated by Inverse Distance Weighting (IDW) interpolation of the combined shuttle cruise and buoy datasets. The interpolation does not consider circulation dynamics or local geography (e.g. changes in bathymetry) within the bay when determining values intermediate to the sampling stations. However barrier vectors were manually drawn at the latitudes of Aquidneck and Prudence Islands to prevent the software from interpolating across these landmasses. The resulting interpolation was masked with the RI state outline (which includes 32 islands) from RIGIS.org. This analysis was carried out in ArcGIS 9.2 according to methods described by Peterson and colleagues (2010).

RESULTS

The changes in downbay concentration are perhaps easiest viewed by comparing absolute concentration before the data have been normalized to meet the assumptions of the statistical tests used. Compared to previous studies (e.g. Oviatt 1980)(Fig. 1-2), the bay shows a reduction in annual dissolved inorganic nitrogen of 15-20% which is a significant reduction at upper bay stations, and a reduction of 35-50% in ortho-phosphate, which is significant throughout the bay (Fig. 1-2). While TP had a similar pattern to DIP, TN shows no significant reduction, though one station (station 11 in the Providence River Estuary) does appear to be consistently lower than past studies. The decrease in DIN was most noticeable in the mid to upper bay region, although the furthest North station (immediately adjacent to the outfall from the Fields Point WWTF) did not show a measurable reduction (upgrades for this plant are scheduled for 2013). However, interannual variability in nutrient concentration was also greatest in this mid-upper bay region (Fig. 1-2). DIP followed a similar downbay pattern to DIN, but with less interannual variability.

Analysis of Covariance reveals more details regarding the overall nutrient dynamics on the downbay gradient (Fig. 1-3). While DIN does not show statistically significant changes in estimated slope or intercept parameters, the ammonium intercept, which is a measure of the level in the upper bay, drops significantly, while nitrate+nitrate has an increased slope but no change in intercept (Table 1-3). Both TN and TP show reduction in intercept, while total phosphorus also changes slope (Table

1-3). As expected, ANCOVA identifies a very strong correlation among all nutrient parameters with distance downbay (Table 1-4). Once the variability associated with the covariate is removed the ANOVA portion of the test reveals significant changes between studies for all parameters tested both as annual averages and during the summer with the exception of TN during the summer. There is also a significant interaction effect (change in slope over time) for nitrate+nitrite and TP, with the interaction effect for DIN as a whole approaching statistical significance. Silicate shows a significant change between studies, both in terms of slope and intercept, but this is driven almost entirely by changes in the station 12 (Fields Point) data. None of the other stations show significant changes.

The pattern in baywide standing stocks shows many of the same patterns seen in the downbay gradients. More specifically, a drop was present for all parameters except silicate, though this relationship was only statistically significant for phosphorus (Fig. 1-4,Table 1-5), though the decrease in DIN on an annual average basis approaches significance (T-test df=4, T=2.17, P=0.09). However, while the reduction in TN is not statistically significant, the average value for the study period is approximately 17% less than the average value calculated for the 1998 survey (Oviatt et al. 2002), which is similar in magnitude to the observed 17% loading reduction, so it is possible that we simply lack the statistical precision to detect this change in light of inter-annual variability. Of note, however, is that the improved reduction efficiency anticipated during the summer (to the impact of temperature on the coupled nitrification-denitrification process) is not evident at all in the standing stock of TN. Similarly, while DIN exhibited a 62% decrease in the summer compared to the annual average, the rest of the constituents did not exhibit this pattern (Fig. 1-4, Table 1-4), and the reduction, when compared to past studies, is not significantly different (36% during summer vs. 34% on an annual basis.

Spatial patterns in nutrient dynamics showed expected trends when extrapolated across the entire bay (Fig. 1-5). Virtually all constituents mapped behaved similarly, decreasing exponentially with north south distance away from the Providence River Estuary and the major point sources of nutrients (WWTF's) therein. In general, concentration in the east passage was slightly lower than concentration in the west passage at equivalent latitude. Mount Hope Bay seemed to be a source of both Nitrogen and Phosphorus to the bay proper, with slightly higher concentrations inside than outside for all constituents, while Greenwich Bay appears to be a source only for elevated concentrations of DIN, with concentrations of DIP, TN, and TP roughly equivalent to, or even lower than surrounding waters (Fig. 1-5).

The ratio of N:P is commonly used as an indicator of potential nutrient limitation in marine ecosystems (Doering et al. 1995, de Vries et al. 1998, Tomasky et al. 1999, Guildford 2000). While not conclusive evidence of one type of limitation or another, DIN:DIP are frequently compared to the ratio of N:P in Redfield organic matter (16:1 N:P). A ratio below 16:1 is typically interpreted as an indication of nitrogen limitation, while ratios above 16:1 are considered indicative of phosphorus limitation (Oviatt et al. 2002, Artioli et al. 2008, Boynton et al. 2008, Nixon et al. 2008). For total nutrients (TN:TP) the inflection point between N and P limitation is typically higher and more variable. This has been attributed to the fact that organic and particulate nutrients are not as readily available for biological uptake and have variable, but usually greater than 16:1 N:P ratios in nitrogen limited systems (e.g. Guildford 2000). When averaged over the year, the bay showed evidence of nitrogen limitation throughout (Fig.1-6), with ratio approaching, but never reaching 16:1 in Greenwich Bay and the Upper Bay, and below 4:1 throughout much of the mid and lower bay. Similarly, N:Si ratio is well below 1:1 through most of the bay, approaching 1:1 in the Providence River where both species are abundant. Comparing DIN:DIP to TN:TP ratios, demonstrates the large amount of nitrogen which is locked up in organic and particulate material, particularly in the Providence River Estuary and Greenwich Bay, but also in the Ohio Ledge region. Despite DIN:DIP ratios around 10:1, these areas showed TN:TP ratios well above 16:1 and in some places, above 20:1. 20:1 is the threshold indicated by a meta-analysis by Guildford and colleagues (2000) as the bottom cut-off for potential N/P co-limitation (Fig.1-6).

DISCUSSION

Nutrient reductions observed

While significant reduction in DIN compared to levels in the late 70's was evident, there is no evidence of a system-wide reduction in TN since 1998 which would be associated with WWTF upgrades. Unfortunately, no TN data from the 70's is available with which to compare, as this survey pre-dates the widespread adoption of the alkaline persulfate technique for colorimetric determination of TN (Valderrama 1981). Stoichiometric and regression based calculations by Oviatt (2008) suggest that a reduction in load of 20% would be minimally detectable under present conditions, and our result corroborates that conclusion. Concentrations of all nutrient constituents remained high in the upper bay year round, and both 2006 and 2009 demonstrated high spatial and temporal extents of hypoxia in the bay (Codiga et al. 2009, Deacutis pers. comm.), indicating that at present, load reductions do not appear to be having a large enough impact on nutrient dynamics to measurably reduce the severity, aerial extent or duration of upper bay hypoxia.

Furthermore, it was difficult to discern if the reductions presently observed are even the direct result of activities at the WWTF's. Since BNR is most effective at warm temperatures (e.g. Dawson and Murphy 1972, Lishman et al. 2000, Pell et al. 2008), one would expect to see a much larger reduction during the summer months, and less so over the remainder of the annual cycle. In contrast, the data from this study (Fig. 1-4,Fig. 1-5) show a relatively consistent reduction over the summer and the entire year, when compared to past studies. The lack of a stronger reduction in the summer is particularly puzzling given that research in similar polyhaline ecosystems typically point to stronger nitrogen limitation during the summer months, with evidence of light or other factors becoming important in colder months (Hecky and Kilham 1988, Cloern 1999, Tomasky et al. 1999). A possible explanation of this is the observed decrease in net denitrification rates observed in the bay over the last several years (see Chapter 3, Fulweiler et al. 2007, Fulweiler and Nixon 2011).

One strong indication that at least some of the observed trends in nutrient patterns can be attributed to loading reduction comes from the percentage of DIN in the bay which was ammonium. Past studies have shown the majority (≈60%) of the DIN in the Providence River Estuary and upper bay to be ammonium, with a decreasing percentage moving down bay (Kremer and Nixon 1975, Oviatt 1980) (Fig. 1-2). This pattern of decreasing proportion moving downbay is consistent with high point source loading of sewage in the upper bay, since secondary treated sewage has very high ammonium concentration, but ammonium is preferentially selected by many plankton species. However, the present study shows lower (≈40%) ammonium concentration in the upper bay, and no decrease moving down bay (Fig. 1-2e), which would be expected if tertiary treatment was converting much of the ammonium to nitrate and nitrite (whose concentrations have actually increased in the effluent streams of many plants which have upgraded). Furthermore, standing stocks of silicate remain unchanged, which reduces the likelihood that the observed reductions are caused by increased drawdown by diatom blooms.

Relationship with primary productivity

The nutrient observations can be compared to recent primary productivity measurements in the bay which have not decreased since the 2005 implementation of advanced wastewater treatment (Smith 2011). The present reduction constitutes about a 17% reduction in the total annual loading of nitrogen to the ecosystem (slightly higher as a fraction of summer N load) when the sewage load is considered alongside riverine, direct deposition, and runoff values (Table 1-1). Mesocosm experiments conducted at the MERL facility in the 1980's indicate log-linear response of primary productivity to nutrient loading, and indicated an 18% reduction in primary productivity in response to a halving of nutrient concentration at loading levels similar to those presently observed in the Providence River Estuary (Oviatt 1986). Another possible explanation for the lack of observed response is that many ecosystems, even those dominated by sewage inputs, may take several years to respond to load reductions (e.g. Carstensen et al. 2006, Duarte et al. 2009). However, similar mesocosm experiments in the Narragansett Bay ecosystem show rapid response of sediment and water column to loading reduction (Oviatt et al. 1984).

The loading reduction does not directly result in a reduction of chlorophyll-a in the bay. Although others have reported a long-term decline in average chlorophyll in Narragansett Bay (e.g. Li and Smayda 1998, Fulweiler et al. 2007, Nixon et al. 2009) weekly data from the GSO dock station and data from a fixed buoy operated by RIDEM located at Bullocks Reach (in the southern reaches of the Providence River Estuary) both exhibit no change in annual average between the first and second half of the 00's (Two tailed equal variance T-Test: df=5 T=-0.4 P=0.70, df=5 T=-2.05 P=0.10 for BR and GSO respectively) or maximum chlorophyll (Two tailed equal variance T-Test: df=5 T=-0.94 P=0.38, df=5 T=-0.96 P=0.37 for BR and GSO respectively) which would be associated with the WWTF reductions; occurring primarily in 2005 and 2006 (Fig. 1-7). Furthermore, there is little long term change in GSO dock data collected by Pilson and colleagues in the late 70's and early 80's (see chapter 2, Pilson 1985b). If anything, chlorophyll has increased during the latter part of the 00's, though this is unlikely to have been caused by the WWTF reduction; more likely the return of large winter-spring blooms in these years.

Sources and sinks of nutrients

Applying a statistical technique to spatially average concentrations showed the location of primary sources and sinks of nutrients in the bay and an exponential decrease with distance down bay. Concentrations in the East Passage were slightly lower than the West Passage, as the circulation patterns of the bay tend to bring oceanic water in the East passage, and advect fresher water from up the bay out the West passage (Kincaid et al. 2008, Rogers 2008). Mt. Hope Bay and the Taunton River were a source of nutrients to the bay proper, while Greenwich Bay may pulse nutrients into the ecosystem after storm events, but on an annual average, has concentrations similar to surrounding bay water for most constituents. On the whole, the circulation dynamics of the bay appeared to be exporting nutrients to Rhode Island Sound, although these nutrients appeared to be primarily in organic form, rather than inorganic (Fig. 1-5). However, caution should be taken in over-interpreting the results of this portion of the analysis, since the model does not take into consideration circulation, depth, wind, or other parameters, and simply extrapolates nutrient concentration based on distance between sampling points.

Discussion in the literature has regarded the role of sediment nutrient flux in Narragansett Bay, and how the contribution of the sediments to the overall nutrient budget of the bay may have changed over the past several decades (Fulweiler et al. 2007, Nixon et al. 2009, Fulweiler et al. 2010). Changes in sediment nutrient flux, particularly the observed reductions in net sediment denitrification, could potentially mask any observable changes resulting from decreased loading. When scaled up to a whole bay average, the results of Fulweiler and colleagues (2007) indicated that the sediments may now be contributing roughly 100 million moles of nitrogen during the summer period, compared with past studies which showed denitrification throughout the annual cycle (Seitzinger et al. 1984, Nowicki 1994b). This change is on the same order of magnitude as the presently observed reductions in sewage loading (90 million moles,Table 1-1) and could explain the lack of a reduction in nutrient concentration and standing stock during the summer (Fig. 1-4).

Nutrient ratios

Similar caution should be used in interpreting N:P ratio data. The data indicated that the ecosystem as a whole, on an annual average, remained strongly nitrogen limited (based on DIN:DIP data, Fig.1-6) as observed in past literature (e.g. Oviatt et al. 1995) While TN:TP ratios are typically not used as a metric for nutrient limitation, the difference between DIN:DIP and TN:TP indicated, in that for the most part, phosphorus behaved conservatively in the bay, with DIP:TP ratios remaining fairly constant down bay while DIN:DIP ratio decreased on a downbay gradient (presumably as N, the limiting nutrient, is consumed). In contrast, DIN:TN ratio (Fig.1-6b) is not at all consistent, with large amounts of particulate and organic N observed in the Upper Bay, Greenwich Bay, and the Ohio Ledge area, changing the N:P ratio in these areas, a possible indication of higher nutrient utilization in these areas (Fig.1-6). The pooling of organic material also may be related to the short residence time of water in the Providence River Estuary (Pilson 1985a) limiting the amount of biological activity which can take place in that region, and/or advecting large amounts of phytoplankton into the upper bay and Ohio Ledge. N:Si ratios follow a consistent north/south gradient, as silicate concentrations appear to fall of linearly rather than exponentially moving downbay (Fig. 1-2). While in the upper bay, N:Si ratio approaches 1:1, in most cases, both species are abundant in this region.

Analyzing the N:P ratios in the form of average of annual averages over a four year period smoothes the data a great deal, and tends to flatten out many of the finer scale details. DIN:DIP ratio, like everything else presented here, was highly variable, and while the smoothed data suggest that the bay is strongly nitrogen limited, there were several individual instances where DIN:DIP ratio exceeded 16:1, particularly in the winter (see Appendix D). This seasonal pattern is consistent with literature from other similar ecosystems (e.g. Fisher et al. 1999, Tomasky et al. 1999, Saito 2008), and suggests that, particularly in this time period (which includes the winter bloom period) both phosphorus and nitrogen may be of concern to management.

Comparison with other ecosystems

Direct comparison of the impact of nutrient reductions between ecosystems can be difficult, as many complex biological, chemical, and physical variables play a large role in how an ecosystem responds to a stimulus. It is, nevertheless, worth the exercise of placing the results observed here in the context of other ecosystems, with the caveat that this is intended merely as a reference, and not as an indication of relative success or failure of the management effort. To this end, we briefly compiled results and compared loading reduction, concentration reduction, and biological response (generally either chlorophyll or primary productivity) from several similar (predominantly temperate estuarine) ecosystems which have undergone nutrient loading reductions (Table 1-6).

In general, the results of this study fall well within the range of observed patterns in other ecosystems. For most ecosystems, response was less than the loading reduction, and Narragansett Bay is no exception to this pattern. Most ecosystems do show some biological response (while not quantified in a method comparable with the other studies presented, Carstensen and colleagues show a correlation between TN and chlorophyll, and therefore, a consequent reduction (Carstensen et al. 2004, Carstensen et al. 2006)) though the range of observed responses is very large. Some general trends which emerge from this comparison are that highly eutrophic ecosystems require greater reduction to elicit response. Greening and Janecki (2006) broke down their analysis to different sections of Tampa Bay, and show less response in highly eutrophic sections of the bay, despite large loading reductions, with greater response in less impacted regions. Residence time may also be a concern, particularly for poorly mixed ecosystems. In general, polyhaline N limited ecosystems did not show significant time lags unless groundwater was a major contributor of loading, though sediment P release may be a larger concern.

CONCLUSION

In general, the results of this study suggest that nutrient concentrations and standing stocks are responding predictably to the instituted loading reductions, and that changes in observed concentrations and standing stocks represent a reduction proportional to the percentage reduction in loading to within the confidence intervals imposed by inter-annual variability in all sampled terms. This reduction is detectable at a statistically significant level for DIN, for which the reduction constitutes approximately 30% of the annual ecosystem budget, but for TN, for which the reduction constitutes only about 17% of the total annual ecosystem budget, some evidence of reduction can be seen in some tests, but not in others. Both total and inorganic phosphorus show statistically significant reductions of 35-50%, though these reductions are likely due just as much from legislative action removing phosphates from detergents and surfactants as to the limited phosphorus removal activities going on at the WWTF's.

While the nutrient standing stocks in the bay have responded to the implemented reduction, no observable reduction in annual average chlorophyll (Fig. 1-7) or primary productivity (Smith 2011) were observed. Past experiments in this ecosystem (Oviatt 1986) have indicated that nutrient levels in the upper bay are sufficiently high that concentrations would have to be reduced by half or more to elicit a response that might be detectable against the inter-annual variability (Oviatt 2008). While present reductions do not approach this level, once all plants discharging into the ecosystem have upgraded to tertiary treatment, we estimate that the annual nitrogen budget will be reduced by approximately 50% (Table 1-1), which would justify a reanalysis of nutrient dynamics and primary productivity of the ecosystem at that time.

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Table 1-1 Estimated major sources of Nitrogen (10^6 Moles N as TN) to Narragansett Bay, and potential future change resulting from impending management strategies. 2010 change values are from this study (Chapter 3).

Nitrogen Source	**2003**[a]	**2010 change**	**2014 potential change**[b]	**Notes**
Direct Sewage	170	143 (16% reduction)	up to 60% decrease	2014 value based on RIDEM estimates of loading: 3mg/l for major plants for 2014, 8mg/l for smaller plants.[b]
Indirect (into rivers) Sewage	193	120 (37% reduction)	up to 50-60% decrease	Assumes above plus MA compliance with proposed reductions. Does not account for riverine abatement.
Other riverine inputs & surface drainage	145	129 (11% Reduction	?	may improve slightly due to reduction in ISDS usage, fertilizer restriction, and improved land-use practices. Changes may take years-decades to manifest.
Direct Atmospheric Deposition	30	30	?	unlikely to change significantly, but may decrease slightly due to air quality regulations.
Urban Runoff	37	62(67% increase)	up to 20-30% decrease	Increased precipitation and land-use changes. Potential future decrease from improvements in CSO abatement and land usage regulations.
TOTAL (10^6 Moles/yr)	**575**	**484**[c]	**approx. 270-320**	

[a] Data from Nixon et al. 2008 [b] Estimates from Liberti, 2009 pers. comm. [c] assuming no change in un-estimated parameters.

Table 1-2 Autoanalytic methodologies and empirically determined detection limits for each nutrient analyte

Analyte	Technicon Method (used 2006-2008)	Technicon MDL	Astoria Method (used 2009-present)	Astoria MDL
Nitrite	Greiss Reaction (NH_4Cl buffered Napthyethelene/Sulfanilimide (NED/SAN)) (Strickland and Parsons 1968, Technicon 1972a, Fox 1979)	0.02 µM	Greiss reaction (Imidazole Buffered NED/SAN) (Strickland and Parsons 1968, Fox 1979, Astoria-Pacific 2005)	0.02 µM
Nitrate	Greiss reaction (NED/SAN w/packed cadmium reduction) (Strickland and Parsons 1968, Technicon 1972a)	0.2 µM	Greiss reaction (NED/SAN w/ open tubular cadmium reduction) (Strickland and Parsons 1968, Astoria-Pacific 2005, Scott et al. 2005)	0.1 µM
Phosphate	Heteropoly Blue (molybdic+ascorbic) (Technicon 1971, Hager et al. 1972, EPA 1983c)	0.12 µM	Heteropoly Blue (molybdic + ascorbic acid) (EPA 1983c, Scott et al. 2005)	0.06 µM
Ammonia	Berthelot Indophenol blue (crystalline phenol+hypochlorite) (Solorzano 1969, Technicon 1973, EPA 1983a)	0.1 µM	Modified Berthelot (liquid phenol, hypochlorite, tartarate) (Solorzano 1969, Scott et al. 2005, Schmidt and Clement 2009)	0.05 µM
Silica	Silico-heteropoly blue (ascorbic, oxalic, molybdic) (Brewer and RIley 1966, Technicon 1972b)	0.06 µM	Silico-heteropoly blue (molybdic, tartaric, stannous chloride) (Sakamoto et al. 1990, Scott et al. 2005)	0.08 µM
Total Nitrogen	Alkaline Persulfate Oxidation + Greiss reaction (as above) (Technicon 1972a, Solorzano and Sharp 1980, Valderrama 1981)	1.1 µM	Alkaline Persulfate Oxidation + Greiss reaction (as above) (Solorzano and Sharp 1980, Valderrama 1981, Astoria-Pacific 2005)	0.5 µM
Total Phosphorus	Alkaline Persulfate Oxidation + Heteropoly Blue (as above) (Technicon 1971, Solorzano and Sharp 1980, Valderrama 1981)	0.12 µM	Alkaline Persulfate Oxidation + Heteropoly Blue (as above) (Solorzano and Sharp 1980, Valderrama 1981, Scott et al. 2005)	0.06 µM

Table 1-3 Parameter estimation by analysis of covariance (ANCOVA) comparing various nutrient parameters from the present study (2006-2010 average) with past studies (Oviatt 1980, Oviatt et al. 2002) over annual and summer (June-Sept.)periods with the covariate of distance south of Fields Point. Parameters are natural log transformed to meet assumptions of ANCOVA, so estimates are ln (concentration) in μM with variation between studies expressed as ±. A p value below 0.05 (boldface) indicates a parameter which has changed significantly between studies

	Annual				**Summer**			
Analyte	Estimate	St. Err.	T	P	Estimate	St. Err.	T	P
DIN								
Slope	-0.063+/-0.010	0.0056	1.70	0.10	-0.078+/-0.004	0.010	0.42	0.68
Intercept	3.20+/-0.22	0.13	1.64	0.12	2.55+/-0.44	0.23	1.9	0.07
DIP								
Slope	-0.031+/-0.002	0.0033	0.60	0.56	-0.045+/-0.003	0.003	0.98	0.33
Intercept	**.897+/-0.24**	**.078**	**3.08**	**0.006**	**1.33+/-0.54**	**0.070**	**7.61**	**>0.001**
Ammonium								
Slope	-0.073+/-0.000	0.0050	0.02	0.98	-0.074+/-0.009	0.009	0.93	0.36
Intercept	**2.62+/-0.41**	**0.12**	**3.43**	**0.003**	**1.89+/-0.68**	**0.22**	**3.07**	**0.006**
Nitrate+Nitrite								
Slope	**-0.052+/-0.005**	**0.0051**	**2.93**	**0.008**	-0.081+/-0.0012	0.011	0.11	0.91
Intercept	2.56+/-0.09	0.12	0.73	0.48	1.67+/-0.22	0.26	0.86	0.40
Silicate								
Slope	**-0.035+/-.006**	**0.002**	**3.76**	**0.001**	-0.028+/-0.06	0.003	1.96	0.065
Intercept	**3.37+/-0.18**	**0.03**	**4.69**	**>0.001**	3.25+/-0.09	0.07	1.29	0.21
TN								
Slope	-0.041+/-.004	0.025	1.73	0.10	-0.041+/-0.003	0.0027	1.07	0.29
Intercept	**3.88+/-0.22**	**0.059**	**3.69**	**0.001**	3.82+/-0.11	0.063	1.7	0.10
TP								
Slope	**-0.043+/-0.007**	**0.0021**	**3.5**	**0.002**	**-0.049+/-0.007**	**0.0022**	**3.04**	**0.005**
Intercept	**1.19+/-0.36**	**0.050**	**7.17**	**>0.001**	**1.53+/-1.30**	**0.052**	**29.5**	**>0.001**

Table 1-4 Statistical results of ANCOVA test comparing present (2006-2010 average) downbay gradient to past (Oviatt 1980, Oviatt et al. 2002) studies over the annual cycle and during the summer (June-Sept.) with covariate distance downbay from Fields Point. All values were ln transformed prior to analysis to meet criterion for normality (by Shapiro-Wikes test). Parameters with P <0.05 are considered statistically significant and are presented in bold

Annual				**Summer**			
Analyte	df	F	P	Analyte	df	F	P
DIN							
Survey	**1**	**31.9**	**>0.001**	**Survey**	**1**	**8.14**	**0.001**
Distance	**1**	**123**	**>0.001**	**Distance**	**1**	**65.2**	**>0.001**
Survey*Distance	1	2.88	0.10	Survey*Distance	1	0.17	0.68
DIP							
Survey	**1**	**43.6**	**>0.001**	**Survey**	**1**	**166**	**>0.001**
Distance	**1**	**88.8**	**>0.001**	**Distance**	**1**	**239**	**>0.001**
Survey*Distance	1	0.36	0.56	Survey*Distance	1	0.97	0.33
Ammonium							
Survey	**1**	**39.6**	**>0.001**	**Survey**	**1**	**17.6**	**>0.001**
Distance	**1**	**222**	**>0.001**	**Distance**	**1**	**65.7**	**>0.001**
Survey*Distance	1	>0.1	0.98	Survey*Distance	1	0.87	0.36
Nitrate+Nitrite							
Survey	**1**	**10.2**	**0.004**	**Survey**	**1**	**3.07**	**0.09**
Distance	**1**	**96.1**	**>0.001**	**Distance**	**1**	**54.5**	**>0.001**
Survey*Distance	**1**	**8.56**	**0.008**	Survey*Distance	1	0.01	0.91
Silicate							
Survey	**1**	**8.64**	**0.008**	Survey	1	0.40	0.53
Distance	**1**	**437**	**>0.001**	**Distance**	**1**	**86.3**	**>0.001**
Survey*Distance	**1**	**13.57**	**0.002**	Survey*Distance	1	3.78	0.065
TN							
Survey	**1**	**19.1**	**>0.001**	Survey	1	2.37	0.13
Distance	**1**	**271**	**>0.001**	**Distance**	**1**	**235**	**>0.001**
Survey*Distance	1	2.99	0.09	Survey*Distance	1	1.14	0.29
TP							
Survey	**1**	**68.3**	**>0.001**	**Survey**	**1**	**36.6**	**>0.001**
Distance	**1**	**409**	**>0.001**	**Distance**	**1**	**491**	**>0.001**
Survey*Distance	**1**	**12.2**	**0.002**	**Survey*Distance**	**1**	**9.84**	**0.005**

Table 1-5 Statistical Results of standing stock analysis comparing total average standing stock of nutrients from present study (2006-2010 average) to past studies (Oviatt 1980, Oviatt et al. 2002). Results were tested using two tailed T-test with pooled variance. Negative T values indicate that the parameter decreased between studies, positive T values indicate an increase. Parameters with T>Tcrit (2.77 for past vs. present studies, 2.30 for annual vs. summer) are considered statistically significant and presented in bold

Past vs. Present Annual			
Analyte	Df	T	P
DIN	4	-2.17	0.09
DIP	**4**	**-5.57**	**0.005**
Silicate	4	0.48	0.66
TN	4	-1.01	0.37
TP	**4**	**-2.84**	**0.04**
Past vs. Present Summer			
Analyte	Df	T	P
DIN	4	-1.43	0.23
DIP	**4**	**-3.49**	**0.03**
Silicate	4	-0.15	0.89
TN	4	-0.23	0.83
TP	4	-1.12	0.33
2006-2010 Annual vs. Summer			
Analyte	Df	T	P
DIN	**8**	**5.39**	**0.001**
DIP	8	-0.42	0.69
Silicate	8	0.17	0.87
TN	8	-0.08	0.93
TP	**8**	**-2.42**	**0.05**

Table 1-6 Response of selected similar estuarine systems to reduction in nutrient loadings. For each ecosystem, response parameter, loading reduction, observed reduction of concentration, and biological response (generally either chlorophyll concentration or primary productivity) are presented (where available) along with the reference citation. NR: not reported NS: No Significant reduction observed

Ecosystem	**Parameter**	**% load reduction**	**% conc. reduction**	**% biological response**	**reference**
Narragansett Bay	DIN	30	34	NS	This study, Smith (2010), Oviatt et al. (2002)
	TN	17	17 (*NS)	NS	"
	TP	NR	28	NS	"
Lajalati Bay	TN	90	30-40	30-40	Clarke et al. (2006)
Pawtuxent R. Estuary	TN	10	NS	NS	Boynton et al. (2008)
Danish Straits	TN	50	Up to 44	NR	Carstensen et al. (2006)
	TP	80	22-57	NR	"
Gulf of Riga	TN	50	NR	NS	Duarte et al. (2009)
Odense Fjord	TN	33	NR	22	"
Helgoland	TN	50	NR	20	"
Marsdiep	TN	43	NR	30	"
Boston Harbor	TN	80-90	35	29,50**	Taylor et al. (2011)
	TP	80-90	32	29,50**	"
Tampa Bay	TN	60+	NR	20-60	Greening and Janecki (2006)
* result not statistically significant			** Chlorophyll-a, Primary Productivity		

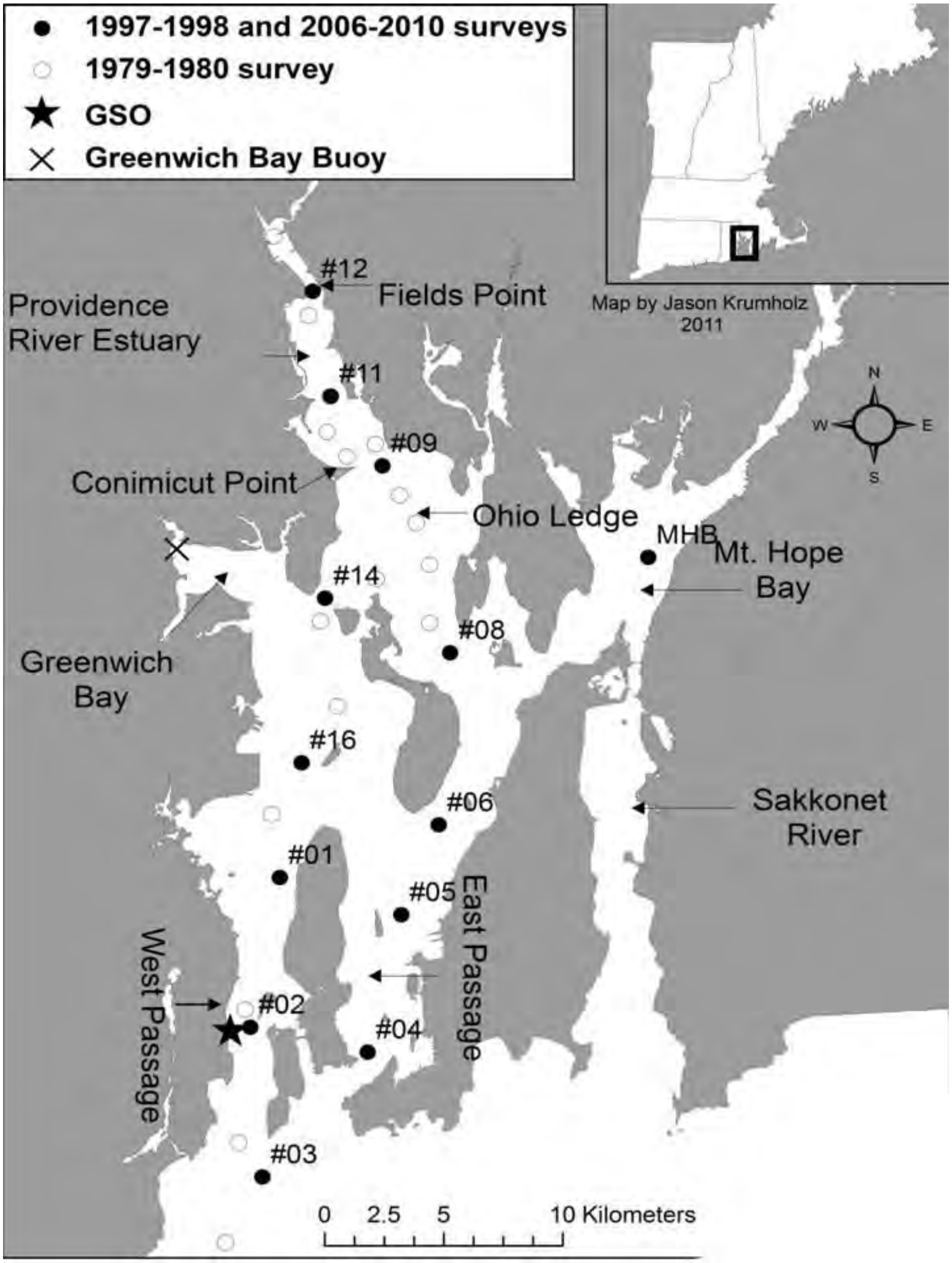

Fig. 1-1 Map of Narragansett Bay, Rhode Island. Solid dots indicate surface water sampling locations for this study, which were the same stations used by Oviatt et al. (2002). Hollow dots indicate sampling locations from Oviatt et al. (1980) used for comparison. The Graduate School of Oceanography (GSO) is marked with a star. Bay landmarks referred to in the manuscript are identified for reference

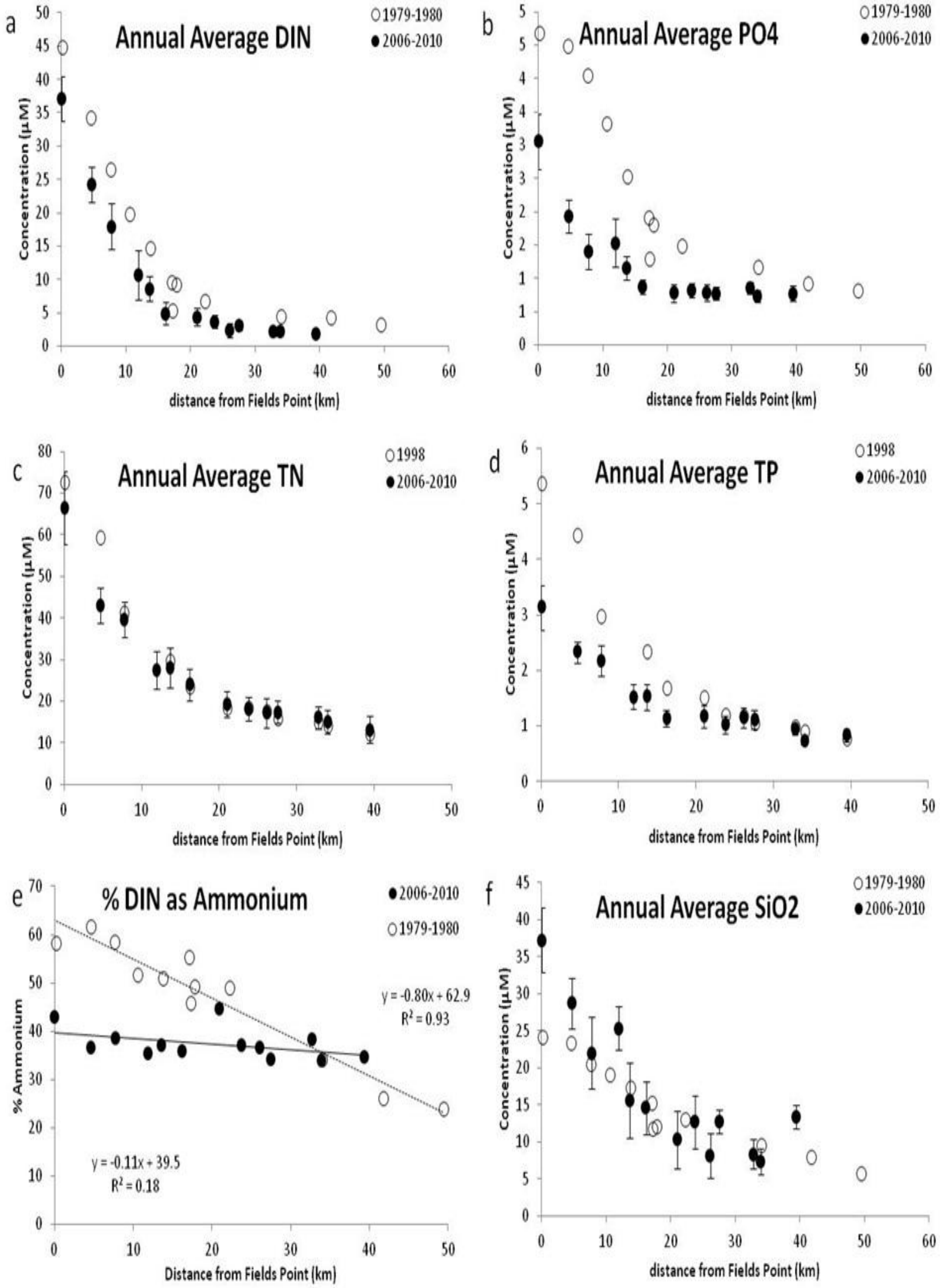

Fig. 1-2 Annual nutrient averages on a downbay gradient from Fields Point. Data from this survey (2006-2010) were compared with previous surveys (Oviatt et. al 2002, Oviatt 1980). Error bars are 1σ of annual averages

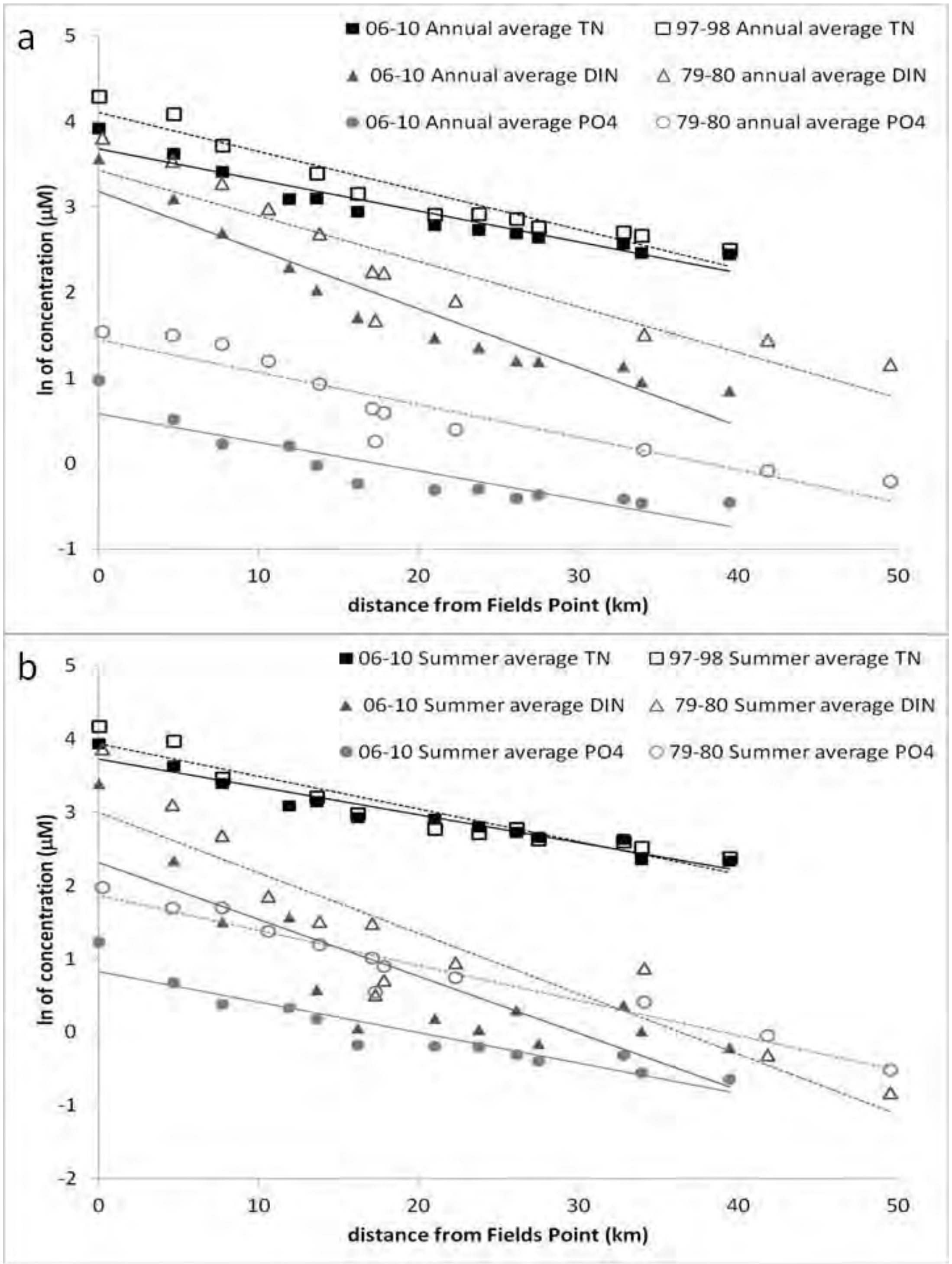

Fig. 1-3 Natural log of annual (a) and summer (June-September) (b) average total (TN) and dissolved (DIN) nitrogen and ortho-phosphate (PO_4) concentration on a downbay gradient during the present study (2006-2010) compared with past studies (Oviatt et al. 2002; Oviatt et al. 1980). Each relationship was compared by ANCOVA (Table 3 and 4)

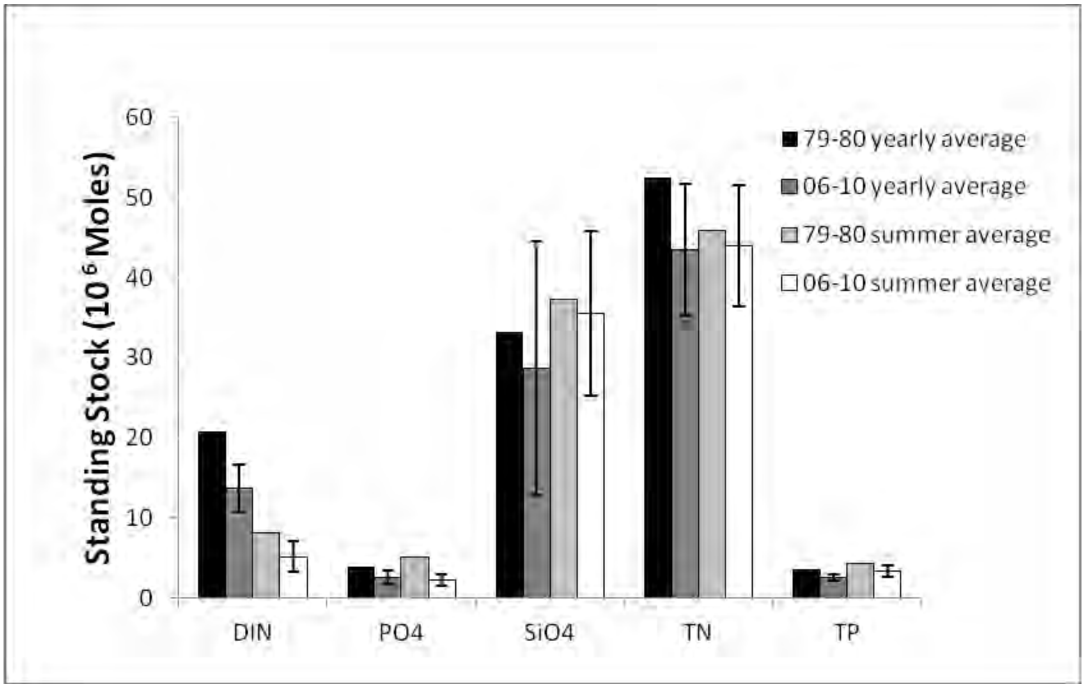

Fig. 1-4 Annual and summer standing stock of nutrients in Narragansett Bay. The 06-10 data were based on annual and summer (June-September) averages of monthly survey averages from this study. Historical TN &TP data from 1997-1998 survey (Oviatt et. al 2002), historical DIN, Si0$_4$ *and PO*$_4$ *data from 1979-1980 survey (Oviatt 1980). Statistical results for this analysis can be found in Table 5*

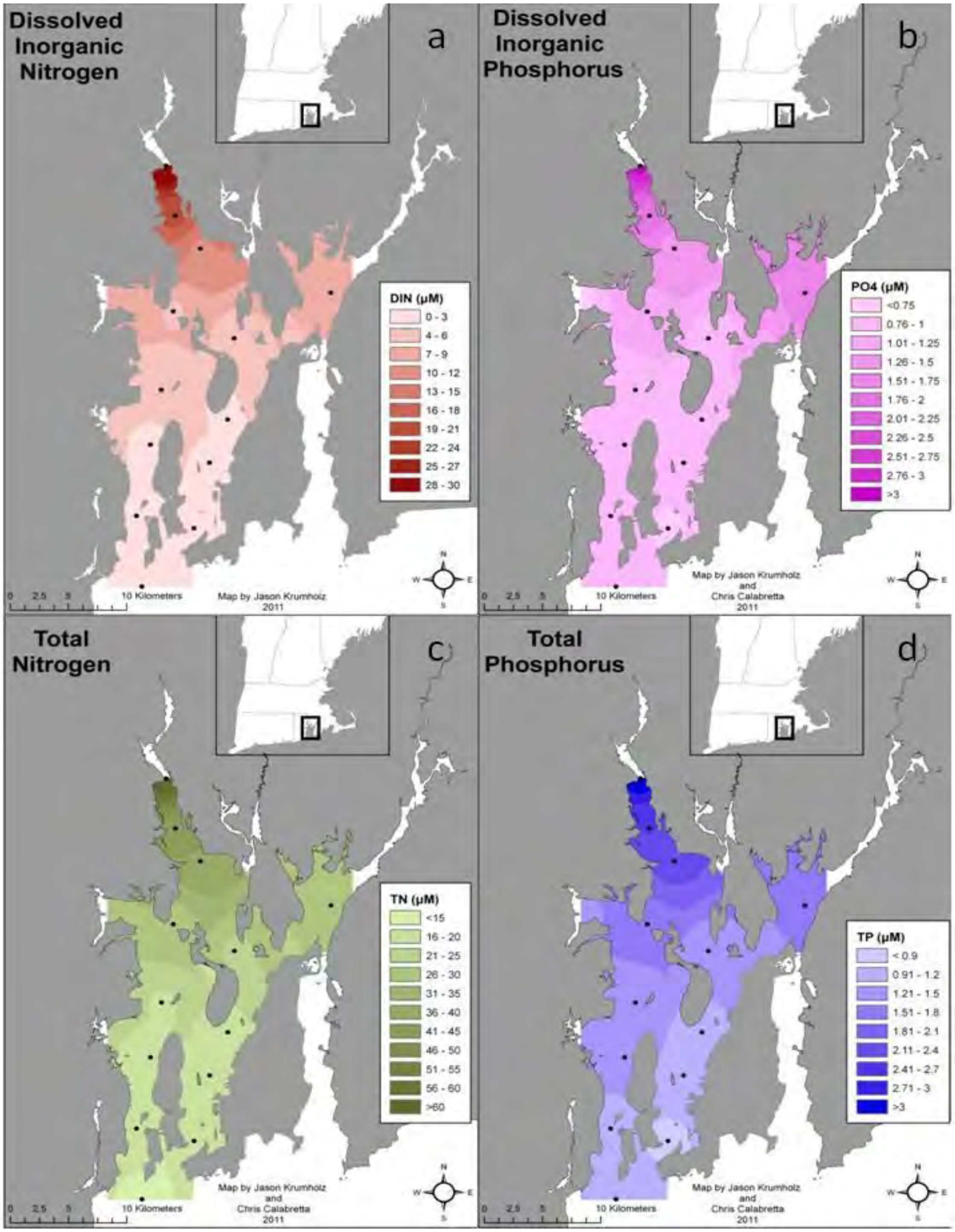

Fig. 1-5 Spatial maps of annual average surface nutrient concentration in Narragansett Bay for dissolved inorganic nitrogen (a) and Phosphorus (b) as well as total nitrogen (c) and phosphorus (d). Spatial interpolation was accomplished by inverse distance weighting of 2006-2010 annual averages of monthly average cruise and buoy data

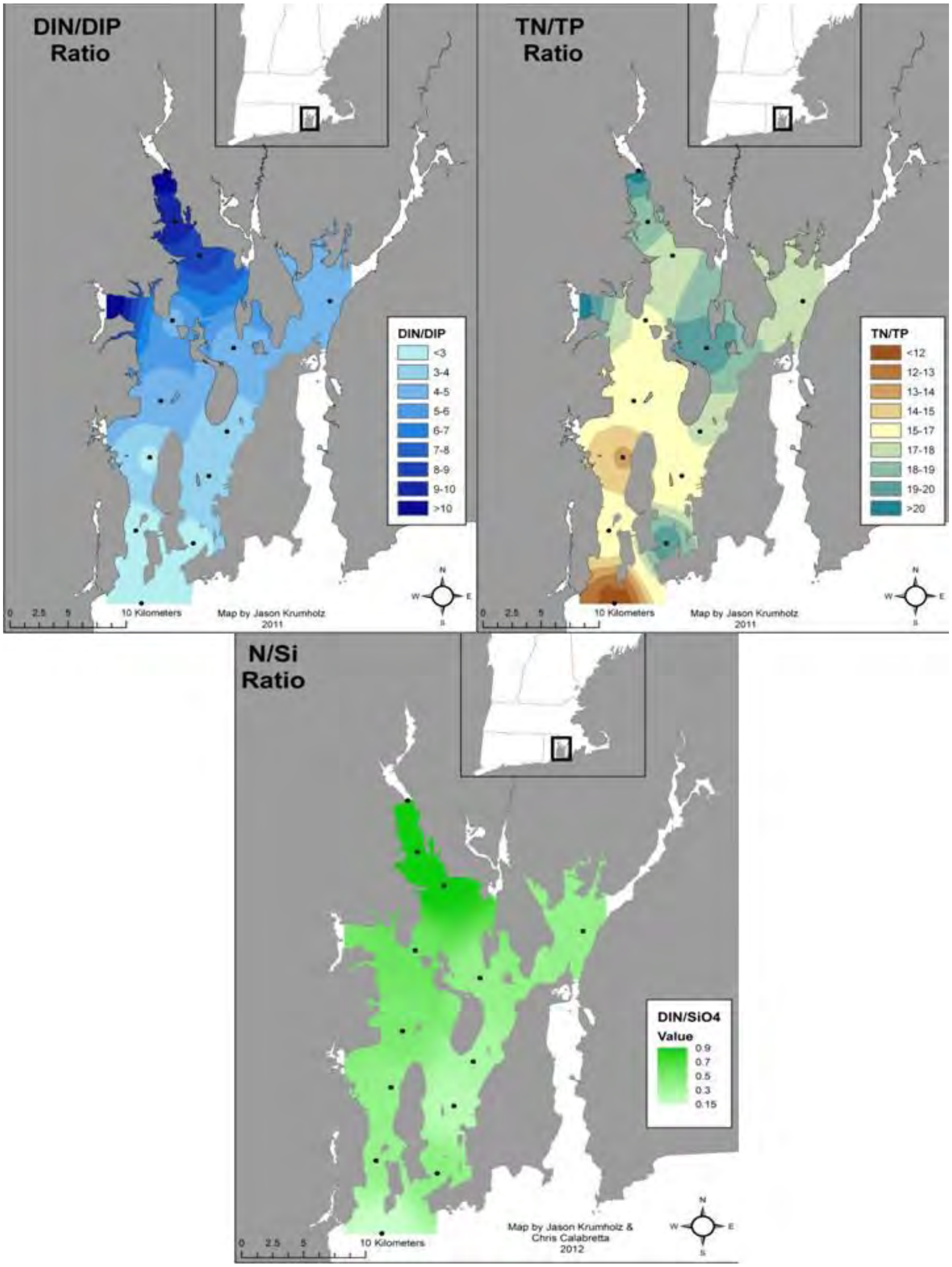

Fig.1-6 Spatial interpolation of dissolved inorganic (a) and total (b) nitrogen to phosphorus ratio and DIN:SiO4 ratio (c) in Narragansett Bay. Values are 2006-2010 annual averages of monthly average cruise and buoy data. Spatial interpolation completed in ARC 9.2 using IDW technique

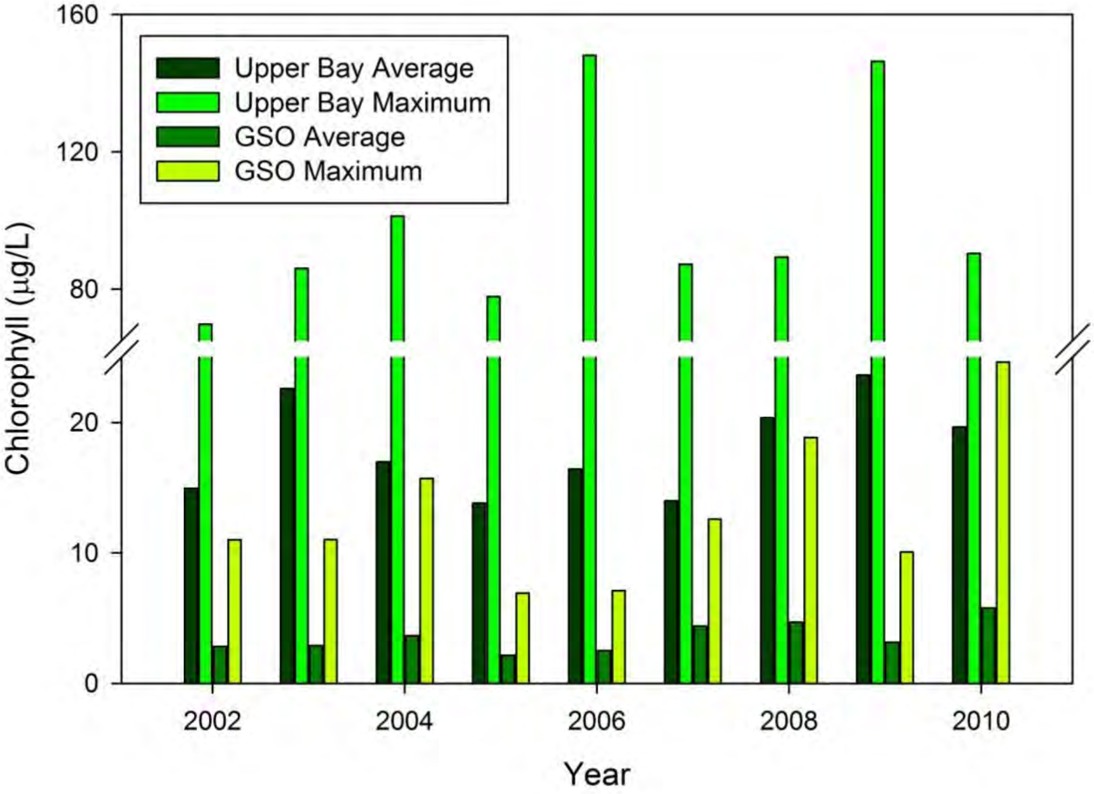

Fig. 1-7 Annual average and maximum chlorophyll-a levels at the GSO Dock station (measured weekly) (Fig. 1) and Bullock's Reach Buoy (Upper Bay) (Data from Heather Stoffel, www.narrbay.org). Upper bay data are seasonal (May-Oct) average and seasonal maximum of daily averages calculated from 15 minute in situ fluorescence data.

CHAPTER 2

AN ANALYSIS OF ANNUAL NUTRIENT CYCLING IN NARRAGANSETT BAY, RI: 1978-2010

ABSTRACT

Annual patterns in nutrient cycling are important to furthering our understanding of how the biology, physics, and chemistry of estuarine ecosystems interact. We use a 40+ year long dataset of weekly water quality and nutrient parameters in Narragansett Bay to analyze long-term and seasonal nutrient trends which may be associated with climate change as well as to investigate changes attributable to recent reductions in nutrient inputs to the bay from implementation of advanced wastewater treatment at several facilities which discharge into the bay.

Comparing the beginning of this dataset to the five years of data available after nutrient plant upgrades (2006-2010, there are statistically significant decreases in concentrations of nitrate, nitrite, ammonium, and phosphate, no change in chlorophyll, and a statistically significant increase in silicate. We also observed changes in the cumulative distribution function of phosphate, ammonium, silicate and chlorophyll. While seasonal cycling was much stronger in the lower bay than the upper bay, no long-term changes in timing of the seasonal cycle in either region of the bay were evident.

INTRODUCTION

In many estuaries, nutrient mitigation strategies are being considered to slow or reverse the progression of anthropogenic eutrophication caused by large sewage, industrial, or agricultural loads (Carstensen et al. 2006, Clarke et al. 2006, Boynton et al. 2008, Vaudrey pers. comm., Nixon et al. 2008). However, the implications of these reductions are not uniform. While in some cases response is relatively linear and predictable- perhaps with a time lag (Carstensen et al. 2006, Artioli et al. 2008, Kemp 2009), in many cases, response is non-linear and for systems with a long history of eutrophication, rapid reductions may not produce the desired result (Duarte et al. 2009, Kemp 2009, Nixon 2009, Taylor et al. 2011).

Increased awareness of the adverse impacts of excessive nutrient loading, combined with falling cost of advanced wastewater treatment upgrades has led Rhode Island Department of Environmental Management (RIDEM) to require that several of the major sewage treatment plants which serve Narragansett Bay be upgraded to tertiary sewage treatment procedures. Between 2002 and 2006, eight plants which discharge into the bay or its tributaries upgraded to tertiary treatment, with three more upgrading between 2007 and 2010 and most other large plants planning upgrades in the next few years (RIDEM 2005). These advanced wastewater treatment procedures include bacterial nutrient removal, which has reduced DIN concentrations in the effluent of these plants by more than half during summer months (the rate of bacterially mediated denitrification is temperature dependent) (e.g. Dawson and Murphy 1972, Lishman et al. 2000, Pell et al. 2008). The implementation of a combined sewer overflow reservoir in 2008 has further reduced nutrient input during high flow periods by delaying storm water runoff, and allowing it to be run through treatment plants before discharge into the bay. The combination of these factors has the potential to reduce total annual nitrogen loading by approximately 30% (Table 2-1).

Often it can be difficult to tease apart the impact of an intervention on an estuarine system in light of various other long-term anthropogenic and natural (e.g. decadal oscillations) variability. Given that most management interventions are not designed as scientific experiments, replication and other forms of scientific controls are often not practical, and in many cases, sufficient long-term baseline monitoring data are not available. Although recent advances in technology have brought automated in situ nutrient analysis within reach (if not quite firmly in hand), the monitoring of nutrients in coastal waters is still, for the most part, accomplished with colorimetric nutrient analysis techniques which have changed little over the past few decades. Modern technology, however, provides continually advancing capability to assimilate, analyze, and communicate data, and as interest in tracking the impacts of remediation activities grows, so too does the body of readily available datasets and tools designed for this purpose.

One such package is SSPIR (Dethlefsen and Lundbye-Christensen 2006), a State-space Model (SSM) package written for the computing language R (R Development Core Team 2005). SSM's are commonly used in the pollution literature for time series data with both an annual and long-term trend (Fanshawe et al. 2008, Lundbye-Christensen et al. 2009, Dadvand et al. 2011) and the SSPIR package allows the differentiation of seasonal cycle, long-term trend and one-time intervention (such as caused by a legislative change or facility upgrade). This package is therefore ideal for this type of study, because it allows us to parse the various changes observed in the seasonal cycle and/or long-term trend separately, rather than perceiving the seasonal cycle as variability in the long-term trend.

In this study, we analyze changes in the annual cycling of nutrients in a temperate estuary (Narragansett Bay, RI) resulting from loading reductions to this system, but also in light of changes in climate and phenology of the region (e.g. Nixon et al. 2009, Fulweiler et al. 2010). The key questions we aim to answer are whether loading reductions at wastewater treatment plants in the upper bay have impacted the seasonal patterns of nutrient concentration in the upper bay, and whether these changes persist further down the estuary and/or impact the seasonal distribution of chlorophyll-a (a frequently used proxy for primary productivity) in the mid to lower bay region. We will also investigate whether long-term changes in the abundance or cycling of nutrients exist in the lower bay, presumably related to changes in climate and phenology.

STUDY SITE

Narragansett Bay is a 328-km^2 shallow phytoplankton based temperate ecosystem with a mean depth of about 8.6 m and a mean water residence time of 26 days (Pilson 1985a, Nixon et al. 1995). Freshwater input is only about 100 m^{3}s^{-1} (Pilson 1985a), resulting in a generally well-mixed system with relatively high salinity ranging from about 20psu in the surface waters at the head of the estuary to about 32psu at the mouth (Oviatt et al. 2002).

The watershed is home to approximately 2 million people, most of whom are concentrated in the northernmost urbanized portions of the watershed. As a result, the bay tends to have a generally north-south gradient in salinity, nutrient and other pollution loading, which in turn creates a similar gradient in eutrophication and primary productivity (Oviatt et al. 2002, Nixon et al. 2008, Oviatt 2008). In contrast to other similar estuaries, approximately 60-65% of the annual nitrogen load to Narragansett Bay comes from sewage (Nixon et al. 1995, Nixon et al. 2008) which is much higher than the average of 36% found by Latimer and Charpentier (Latimer and Charpentier 2010) for 74 New England Estuaries (including Narragansett Bay).

METHODS

Since the fall of 1976, water samples have been collected weekly from the end of the dock at the Graduate School of Oceanography, Narragansett, RI (Figure 2-1). Surface water samples were collected at approximately 9AM each Wednesday morning, irrespective of tide, although if significant precipitation or scheduling conflicts were anticipated, the sample was occasionally collected slightly early or late. Sampling commenced in August, 1976 and has continued virtually without interruption (two short periods, one in 1977 and 1983 had no samples for a few months) through the present. For the purposes of this analysis, only complete years (1978-2010, excluding 1983) were used, constituting a total of 1715 discrete samples over this 33 year period; slightly over 51 samples per year on average.

The sampling location has changed very little over the time period sampled. During 1977 and 1978, samples were collected by Niskin bottle from 2m depth at the

GSO pier. During the operational phase of the Marine Ecosystem Research Lab (MERL), water was collected from the indoor header tank supplying water to the mesocosm facility. The supply intake for this tank was located in approximately 2-3m of water under the dock (Pilson 1985b, Oviatt 2004). When this facility ceased full-time operation in June 1997, sampling returned to the pier, although samples are now collected by bucket from the surface, rather than by Niskin bottle.

Each sample was measured for temperature immediately by thermometer, then a one liter subsample was collected in an opaque polycarbonate bottle and returned to the MERL facility for analysis. Samples were analyzed immediately (after a 30 minute rest in a dark room) for fluorescence and a 10ml aliquot buffered with two drops of supersaturated magnesium carbonate buffer was filtered onto 25mm Whatman GFF filters for chlorophyll extraction (Yentsch and Menzel 1963) as modified by Lorenzen (1966). Prior to July 1984, all chlorophyll analysis was conducted with a Turner Model III fluorometer; from July of 1984 until August 2002, a Turner Designs Model 10 Series Field Fluorometer (Oviatt and Hindle 1994) was used. In August 2002 this instrument was replaced by a Turner Design Model 700, and in May, 2007 by a Turner model 10-AU. In each case, an intercalibration of the two instruments was performed. Specifics of the MERL application of this procedure can be found in the MERL methods manual (Oviatt and Hindle 1994). With a few small exceptions, most notably a switch from freezing chlorophyll filters for later extraction to immediate extraction in November, 2008 (MERL, unpublished) these methods have changed little over time. To correct the chlorophyll dataset, a correction factor was empirically derived using a set of side-by-side samples over the course of a year, and put in place to account for possible differences resulting from the switch from freezing to immediate extraction (as per Graff and Rynearson 2011). Because the majority of the dataset used freeze-and-extract methodology, the most recent two years of data were corrected to resemble earlier data, rather than correcting 30+ years of data, even though it is likely that immediate extraction results are more accurate.

Separate 40 ml aliquots were withdrawn for salinity and dissolved inorganic nutrients. Salinity samples were sealed with parafilm and stored at room temperature awaiting analysis on a Guildline model 8400B Autosal salinometer. This model instrument has been continuously employed since the commencement of the dataset, although it was replaced partway through with a nearly identical model.

Nutrient samples were filtered through a 0.45µM nucleopore filter and stored frozen until analysis. For the majority of the sampling period, nutrients were analyzed on a Technicon model 2 Autoanalyzer (Technicon Industrial Systems, Tarrytown, NY). In 2009, nutrient analysis in the MERL facility switched to an Astoria SFA analyzer (Astoria-Pacific, Clackamas, OR). A thorough intercalibration between these two instruments was conducted to ensure continuity of data (See Appendix A). Colorimetric techniques used by the two instruments were similar, although some changes do exist (See Appendix A for a thorough review or *Table 1-**2*** for summary). Prior to 1982, Nitrite was not run separately, so only a ‘nitrate+nitrite’ measurement was available; however, this does not impact the determination of dissolved inorganic nitrogen because any nitrite present in the sample (generally a small amount of the total DIN (see Chapter 1)) would have been detected in the ‘nitrate+nitrite’ channel. Methodology for the preparation and storage of nutrient samples has not changed with instrument switchover, with the exception that starting in 2009, salinity for upper bay samples was recorded for matrix matching purposes (matrix matching was not used for the Technicon, rather a salinity correction factor was applied when necessary, see appendix A) and is outlined along with standard operating protocols for the Technicon analyzer in the MERL manual (Oviatt and Hindle 1994). Seawater operating procedures for the Astoria analyzer can be found in Scott et al. (2005), and SOP for the MERL Astoria Analyzer can be found in Appendix A.

In order to correct for any potential bias caused by missed or lost samples, linear interpolation was used to fill any gaps in the dataset. Of the 10620 discrete values in the dataset, 639, or roughly 6% were interpolated. Most gaps occur in the early portions of the dataset and only 5 are more than 2 weeks in duration between samples.

Annual averages, minima, and maxima for each analyte were calculated and compared via regression analysis to examine long-term trends. Nutrient and chlorophyll data were also compared to climate variables such as precipitation at T.F. Green airport in Providence (NOAA 2008), and NAO to identify any long-term trends. These lower bay data were compared with similar data collected at several upper bay stations during similar time periods, using two separate discrete datasets. During 1979-1980 surface nutrient samples were collected from 17 stations around the bay (but not including the lower east passage) approximately biweekly as part of a separate study (Oviatt 1980). Similarly, from 2006-2010 surface water samples were collected from 13 stations throughout the bay for a separate project. However, the sampling methodology used in both of these data sets is virtually identical to the protocol used for the GSO dock sample (for more detail, see Chapter 1) with the exception that no chlorophyll samples were taken, and all samples were run on the MERL autoanalyzers, so the data are comparable. However, because the sampling locations in these two studies do not line up exactly, and because nutrient concentrations in the Providence River Estuary are strongly spatially variable, it was necessary to average the values over a larger area in order to make these datasets comparable. By averaging over the entire Providence River Estuary (defined as from Fields Point south to Conimicut Point, encompassing 4 stations for the 79-80 survey, and 3 stations for the 2006-2010 survey) it is possible to directly compare these two datasets. Averaging in this way also eliminates any small scale spatial variability which could impact the results.

Data were compared across time and space with two-sided two-tailed Kolmogorov-Smirnov (KS) tests to determine if the distribution or magnitude of the nutrient data have changed over time. This test is commonly used to test the assumption of normalcy in a dataset by comparing a given dataset to a normal dataset with the same mean and standard deviation (often referred to as a one sided KS test). However, it can also be used to compare two observed distributions, and calculate the likelihood that those observations are drawn from the same larger dataset or are independent (two sided KS test).

The KS test is useful to determine whether the data are drawn from the same or statistically different distributions, but it does not distinguish between temporal shifts and magnitude shifts. To attempt to isolate any temporal changes, the data were normalized to cumulative percentage of observed nutrients over the course of a year, such that on 12/31, 100% of each analyte has been realized. This allows us to view the percentage of the nutrients which can be found in each given season. This procedure creates a visualization whereby a constant concentration across the annual cycle would cause a straight line with slope approximately 2% per week, and a strong seasonal cycle would produce a sigmoid response. The normalized data were then tested again with the KS test to determine if any temporal shifts were statistically significant.

The time series analysis package SSPIR (Dethlefsen and Lundbye-Christensen 2006), written for R (2005) was used to parse the observed effects into long-term, seasonal, and intervention driven changes. SSPIR is a state-space model (SSM) which is similarly treated in R to a generalized linear model (GLM), with the exception that the SSM allows the parsing of time series terms (e.g. harmonic and unstructured seasonal patterns, interventions, etc…). The model is then fitted to the data using extended Kalman filtering (Dethlefsen and Lundbye-Christensen 2006).

Because of the high amount of interannual variability, and the strong serial autocorrelation in the data (correlation coefficient of timestep t with timestep t-1 was about 0.8), fitting a state-space model like SSPIR to the data is a good choice to try to increase the resolution. Because SSPIR cannot predict variance (and therefore provide a confidence interval around a prediction) it was necessary to calculate variance with another function. For this, we chose StructTS (Ripley 2002) and removed the annual cycle using ‘sumseason’ to average the past 52 (weekly) data points to white noise, which reduced the trend to a random walk, and produced appropriately uncorrelated residuals.

By separating the long-term trend from the intervention in this periodic dataset, we can isolate whether reductions at Upper Bay treatment facilities have a measurable impact on average nutrient concentrations, seasonal nutrient cycles, or chlorophyll levels at this lower bay station or whether this area of the bay is relatively insulated from upstream changes.   To do this, we calculated the magnitude and confidence intervals for an intervention term on various nutrient analytes taking place in January 2006 (When the Bucklin point plant came online, although several other smaller plants upgraded within a few months of this time), and for comparison sake, a phosphate intervention term taking place in January 1995, immediately after legislation passed to reduce phosphate loadings from detergent (Litke 1999). This comparison will allow us to test the sensitivity of the model to the intervention term, because unlike DIN reductions, phosphorus reductions were gradual, beginning well before the passage of legislation, and continuing to gradually fall throughout the 90's and 00's. R-Code and specific application notes pertaining to the model can be found in Appendix C.

RESULTS

Virtually all nutrient components exhibited a seasonal cycle, with concentration highest in the late winter/early spring, falling off sharply with the winter-spring diatom bloom (or less sharply in years where this bloom is weak or absent), remaining lower through the summer, then rising again in the fall as primary productivity tapers off (Oviatt et al. 2002) (Figure 2-2). The absolute magnitude and timing of the yearly maximum was variable, and appeared to show little trend over time, with the possible exception of a lack of extremely high values during the last 5 years or so (Figure 2-2). Throughout the year, the ratio of N:P was typically well under the 16:1 Redfield ratio, an indication of nitrogen limitation, although at times during the summer, both nitrogen and phosphorus became quite low (Figure 2-2, 2-3).

A first pass comparison can be made by observing side-by-side, the annual cycle at the beginning of the dataset and the annual cycle from the most modern years, to detect whether a change in absolute magnitude or seasonal timing can be observed in either the GSO dock or Providence River Estuary datasets (Figure 2-3, 2-4 respectively). For the upper bay dataset, discreet sampling was done on a monthly (bi-weekly in the summer) basis at several stations in the Providence River Estuary both in 1979-1980 (Oviatt 1980) and from 2006-2010 (this study). To account for differences in sampling locations (due to proximity to nearby WWTF's, nutrient concentrations in the PRE were highly spatially variable) all stations within the area north of Conimicut Point and South of Fields Point (3 for the 2006-2010 survey and 4 for the 1979-1980 survey) were averaged.

Seasonal magnitudes and patterns of nitrogen constituents (nitrate, nitrite, ammonium) have not changed dramatically over time, though some small changes in seasonal pattern (most notably a sharper drop off of nitrate+nitrite in the modern data due possibly to the return of larger winter spring blooms) and magnitude (e.g. less ammonium in the fall in the modern data) may be observed (Figure 2-3). There was a dramatic reduction in the concentration of phosphate throughout the annual cycle. In contrast, silicate shows a small, but statistically significant increase in concentration, particularly during the summer months.

It is apparent from the data that there was a large amount of inter annual variability, both in range and in pattern. This was particularly true in the winter-spring period and in late summer, which is expected, because of variability associated with bloom dynamics during these periods, documented in past literature on the bay (e.g. Pilson 1985b, Oviatt et al. 2002, Smith et al. 2010), and observed in high variability in the chlorophyll data during those times of year (Figure 2-3). When taken as yearly averages, there was no long-term trend observed in the chlorophyll data, however, again possibly due to weakening of spring blooms (e.g Nixon et al. 2009) there is a slight downward trend in annual maximum chlorophyll over time (Figure 2-5). Precipitation was a small but significant contributor to DIN and SiO_4 concentration in the bay, with a slight positive relationship between the average DIN concentration in a given month at the GSO dock, and the total precipitation fallen during that month (R^2=0.02, df=359, F=6.6, P=0.01), and similarly, over an annual cycle, for silicate (R^2=0.22, df=30, F=8.2, P=0.007) (Figure 2-6a,d). Because the sample was not collected when rain is falling, we chose not to attempt correlation on a shorter time scale than monthly for fear of biasing the result due to the sampling method. No relationship was found between PO_4 and precipitation or between NAO (December, January, February index) and nutrients at the GSO station, though NAO exhibits a slight negative correlation with chlorophyll (R^2=0.14, df=32, F=5.17, P=0.03). There were small but significant negative relationships between chlorophyll and nutrients (R^2= 0.13, df=383 , F=58 , $P<0.001$ for DIN and R^2= 0.11, df= 383, F=47 , $P<0.001$ for PO_4), though the relationship with DIN has both steeper (relative to Redfield) slope and higher R^2 (Figure 2-6).

By using the KS test to compare data from the beginning of the dataset to data after the onset of loading reductions (2006-2010) we can determine whether the upgrades, or other changes to the system, have altered the distribution of nutrients, either in timing (likely associated with climate change), or in magnitude (likely attributed to load reductions). One output visualization of the Kolmogorov-Smirnov test is to compare the cumulative frequency distribution (with frequency on the Y axis and concentration on the X) of the two datasets. This analysis for the GSO dock data, indicated that the nutrient analytes have responded differently over time. While nitrate and nitrite showed virtually identical curves to data from 30 years ago, ammonium showed a small but statistically significant drop across the entire range of observed values (Figure 2-7). DIN showed similar maximum magnitude, indicating that peak DIN concentrations have not changed over time, and a small but not statistically significant increase in the frequency of moderate values (between 2-4 μM), with a not statistically significant corresponding decrease in the frequency (but not magnitude) of high (>8μM) values.

In contrast, phosphate showed a continuous reduction across all dates, with the largest reduction (>50%) present in the peak values. For example, 90% of observed phosphate values in the modern dataset are below 1.3 μM, while only about 50% of the historical values are below this threshold (Figure 2-7).

Silicate shows a statistically significant increase at the GSO site, and nearly statistically significant in the upper bay, with the increase appearing to result from more very high values in the recent data, rather than fewer low values (Figure 2-7, 2-8).

Chlorophyll also shows a statistically significant decline though this reduction appears to come exclusively from a drop in peak values (Figure 2-7, 2-5). While the KS test does not discriminate as to whether a statistically significant change is due to a drop in peak values, or a change in distribution, the associated K statistic shows the maximum difference observed between the two datasets, which in the case of chlorophyll, is located at the very peak of the distribution (Figure 2-7). Furthermore, regression analysis shows no change in annual average, but a measurable downward trend in annual maximum (Figure 2-5). Unfortunately, bloom dynamics in the lower bay are difficult to discern from this dataset because an observed chlorophyll peak may be due to favorable local conditions, or due to advection of a bloom from the upper bay, and the weekly sampling frequency is insufficient to reliably capture shorter events. Nevertheless, this portion of the bay has experienced a significant reduction in the frequency and magnitude of high chlorophyll values over the last few decades.

The upper bay data (Figure 2-4, 2-8) have similar absolute patterns to the lower bay data, though the concentrations are (expectedly) higher, and the seasonal variability is somewhat lower. For this dataset we also have total nutrients (from Oviatt et al. 2002), which show a significant decrease in very high TN events, and a nearly significant reduction in TP, which appears to be relatively constant across concentration (Figure 2-8)

We observed interesting patterns in the magnitude shifts in nutrients associated with the last few decades in Narragansett Bay, but in order to investigate whether changes in seasonality are observed, it was necessary to isolate and remove these magnitude changes, in order to look strictly at the seasonal patterns (Figure 2-9). To do this, we normalized the maximum concentration observed in any given year to 1, and examined the cumulative fraction of the total nutrient load observed during the course of the year. A normalized cumulative percent contribution curve that is close to linear indicates consistent concentration throughout the year. Areas with steep slopes have disproportionately high concentrations, and vice versa. Some variables (e.g. nitrate) exhibit a much stronger seasonal cycle than others (e.g. chlorophyll, phosphate). In general, however, few changes between the datasets are observed. The spring bloom may be occurring slightly earlier (evidenced by an earlier drawdown of nitrate and DIN), and there may be a slightly stronger seasonal cycle in phosphate and ammonia, but none of these observations were statistically significant. On a seasonal basis, a much weaker cycling in the upper bay occurred than we observed in the lower bay, particularly in nitrate+nitrite, which was relatively constant in the upper bay, but showed a strong seasonal cycle in the lower bay (Figure 2-10, 2-9 respectively). In contrast, silicate shows very weak seasonal cycling in both parts of the bay, possibly because it does not flux into or out of the sediments as much as nitrogen.

Similarly to the lower bay, there were only very slight differences in seasonal pattern which can be observed between the datasets, none of which were statistically significant (Figure 2-10). Increased variability in the upper bay dataset may be related to variations in discharge associated with precipitation, but also may be an artifact of the way the data were handled. While the lower bay are weekly data points from a single source, the upper bay data are monthly averages of several stations located across a strong spatial gradient. In many cases, the concentration at Conimicut point (the southernmost extent of stations categorized as ‘Providence River Estuary’) are half or less the value observed at Field’s Point (the northernmost extent and location of the outfall for the largest plant), a reduction due in part to dilution and in part, presumably to utilization.

None of the analytes showed a statistically significant (confidence interval not overlapping zero) intervention effect relating to a phosphorus reduction pinpointed in January 1995 at α=0.10, and most estimated intervention terms (with the exception of phosphate) were very small, indicating minimal impact. Intervention terms for the DIN reduction associated with the WWTF upgrades were much larger, but so too are the associated confidence intervals. No intervention parameters were significant at α=0.05, and only ammonium was significant at α=0.10 (-0.54±0.46 μM). The intervention term for chlorophyll was positive (though not statistically significant), indicating that, if anything, chlorophyll in the lower bay has increased since the reductions came online.

Another benefit of the model is that it can be used to compare the relative magnitudes of the various signals within the dataset (Figure 2-11). The model pulls out a seasonal signal of approximately 10 μM. With annual cycling removed by compiling a one year moving average, we can also display a long-term trend in the data (Figure 2-11a). While the time series shows some prolonged periods of relatively high DIN concentration in the 1990’s, and an extended period of low average values from 2003-2008, 2009 and 2010, the last two years of the model are quite high, which casts doubt on any long-term trend. While the model does show some interesting patterns, the remaining residual after long-term trend and seasonal cycle have been removed is still quite large (Figure 2-11c); larger than the magnitude of the seasonal cycle and the long-term trend combined. This term also appears to show an annual pattern, a possible indication that not all of the annual signal is captured appropriately by the model.

DISCUSSION

When directly comparing the early and late parts of the dataset, there are some clear changes despite the large amount of interannual variability. Virtually every analyte (with the exception of chlorophyll) showed a statistically significant change from the early to the later part of the dataset (with NO_2, NO_3, DIN, NH_4, and PO_4 decreasing, and SiO_4 increasing). The aspect most directly associated with the upgraded WWTF processing is the observed decrease in ammonium during the summer. These reductions significantly (up to 90% in some cases, see chapter 3) reduce ammonium loading from several of the plants discharging into the bay and its tributaries. This hypothesis is strengthened by the fact that the changes in ammonium were larger in the upper portions of the bay, nearer to the WWTF's (Figure 2-3, 2-4). The trend was weaker when considering DIN as a whole, as nitrate and nitrite have decreased only slightly in the lower bay, and not at all in the upper bay.

In contrast, the observed large reduction in phosphorus was likely less related to WWTF upgrades (though a few plants have implemented phosphorus reduction procedures), but rather due to changes in legislation removing phosphates from detergents which occurred throughout the 80's and 90's and continue into the present (e.g. Litke 1999). We suspect this because phosphate showed a gradual decline throughout the dataset (Figure 2-7, 2-8, 2-12), rather than a punctuated drop in the highest values as observed with the nitrogen species (Figure 2-7, 2-8).

Unfortunately, the model cannot confirm the impact of legislation on phosphate concentrations result, as there was no significant response of phosphate to intervention either in the mid 1990's or 2005. The phosphate response should be more gradual because phasing out began before the passage of legislation, continued to reduce through the 90's and 00's, and may also have been delayed due to sediment remineralization (Pomeroy et al. 1965, Litke 1999, Carstensen et al. 2006).

We also observed an increase in dissolved silicate between the beginning and end of the survey period. It is possible that the observed increase in silicate in both the upper and lower bay (Figure 2-3, 2-4) is related to increased precipitation, as silicate concentration shows a positive correlation with total precipitation on an annual basis (R^2=0.23, df=29,F=8.28 P=0.007). It is also possible that this pattern was related to decreased diatom based primary productivity, and therefore decreased demand. The trend holds for both the upper and lower bay datasets (Figure 2-3, 2-4), reducing the likelihood that it is anecdotal or site related (e.g. increased sedimentation at GSO dock site). However, there is a great deal of interannual variability in silicate concentration, and while several other studies have shown a decrease in chlorophyll over time in Narragansett Bay (e.g. Fulweiler et al. 2007, Nixon et al. 2009), this dataset does not show any long-term reduction of average chlorophyll concentration in the bay (Figure 2-3, 2-5), though there may be some evidence of decrease in the intensity of blooms (Figure 2-5, 2-7). While a shift in the biological community of primary producers might explain the observed trend, it is beyond the scope of this study to speculate on causality.

In the upper bay a steep reduction in maximum ammonium values occurred, likely caused by the removal of ammonium from the WWTF's. The biological nitrogen removal process used at these plants is typically coupled nitrification-denitrification, whereby the DIN in secondary treated wastewater is super-oxygenated and bacteria oxidize ammonium into nitrite then nitrate, after which the wastewater is allowed to become anoxic, and other bacteria convert it into nitrogen gas (N_2). If the aerobic process is run near to completion but the anaerobic portion is not, a dramatic reduction in ammonium discharge occurs (up to 90% for some plants during summer months), with little change, or even an increase in nitrate and nitrite discharge. This transformation explained the upper bay data (Figure 2-8), but the trend weakened in the lower bay as the relative contribution of ammonium to DIN decreased. This is somewhat puzzling, since typically ammonium is more readily bioavailable than nitrate or nitrite, however it is possible that decreased loading of ammonium paired with stable or even slightly increasing nitrate and nitrite loads may have increased the relative percentage of nitrate and nitrite taken up, simply because there was insufficient available ammonium.

The lack of a strong seasonal cycle in nitrate in the upper bay is a potential indication that nutrients were not limiting production in the upper bay. While concentrations in the lower bay were drawn down to near zero during the summer months when productivity was high, concentrations in the upper bay remained relatively constant throughout, as a steady supply of nutrients from the plants exceeded that which can be utilized by the plankton during its short residence time of about 3 days in the Providence River Estuary (Pilson 1985a). This is confirmed by results from the GEM box model, which showed, in general, that light limited production in the Providence River Estuary during most of the year (Kremer et al. 2010, Vaudrey pers. comm.). Literature from other systems also provides evidence of light limitation under similar nutrient loads, especially in the winter time (e.g. Cloern 1999, Sin et al. 1999, Saito 2008). Furthermore, mesocosm experiments (Oviatt et al. 1986, Oviatt et al. 1995) showed decreased ‘return on investment’ with nitrogen loading at or near concentrations observed in the Providence River Estuary.

Although definite differences in seasonal cycling between the upper and lower bay occur, when we compare the seasonal patterns at the same site over time, there is little evidence of any changes. After standardizing to remove changes in absolute magnitude of nutrient concentration, we see no change in annual cycle over the dataset. This is an indication that the many other climate related factors which might be influencing nutrient dynamics in the bay by altering phenology have not, at least as of yet, impacted the seasonal cycling of nutrients.

The model results were relatively inconclusive in terms of discerning whether an instantaneous ‘intervention’ occurred in concentration associated with the plant reductions, rather than a gradual decrease or simply interannual variability. While the model predicts a decrease in all nitrogen species associated with the intervention, the residuals produced by the StructTS function which are used to calculate the confidence interval for the model were cripplingly large. As such, the only analyte with a 90% confidence interval not overlapping zero was ammonium (intervention term 0.54μM±0.46). This is the analyte from which we would expect the greatest response, since the majority of the plant reductions is in the form of ammonium. While the state-space modeling approach may be an interesting and appropriate technique to parse trends and responses in this dataset, additional work is necessary before the model will provide further insight.

One possible issue is that because the residuals were estimated with StrucTS, which does not include an intervention term, variability associated with the reduction would be interpreted by StrucTS as 'noise', increasing the residuals from the model, and therefore, the variance in SSPIR. It is also possible that the 'solution' provided by StrucTS was a local maximum rather than the global maximum likelihood, artificially inflating our estimation of variance as well.

Another issue with the model is the high amount of variability in the data not captured by either the long-term trend, the seasonal cycle, or the intervention term. The model residuals appear to still have an annual signal in them as well, though perhaps the period of this signal is not exactly 52 weeks from year to year, which may explain why the model does not attribute this variability to the annual cycle term. We attempted to fit the annual cycle term using the 'polytrig' function in SSPIR, which would allow the periodic (seasonal) cycle to vary from year to year both in amplitude and in period, but could not get this function to work, and so settled for the simpler 'sumseason' command which uses a fixed amplitude and 52 week period. It is quite possible that the uncaptured variability in the seasonal cycle has to do with the timing of the winter-spring bloom. We suspect this because there is a downward spike in the residuals virtually every year in the February-March time frame, and the spike tends to be smaller in years with no winter-spring bloom (e.g. 1998, 2005, 2006) (Figure 2-11), which we anticipate is indicative of nutrient drawdown correlated with the bloom. The ability to capture and incorporate some of this variability would greatly improve the utility, and probably the predictive capacity of the model.

CONCLUSION

Some marked changes have occurred in the way nutrients cycle in the bay over the last several decades. There is a strong decrease in phosphorus in both the upper and lower bay (Figure 2-3, 2-4, 2-7, 2-8), due to legislative changes removing phosphates in detergents, surfactants, and other industrial and household products. WWTF load modifications have resulted in significant reductions of ammonium and to a lesser degree DIN in both the upper and lower bay (Figure 2-3, 2-4, 2-7, 2-8). While the lower bay appears to have a stronger seasonal cycle, particularly for nitrate+nitrite, than the upper bay, neither location exhibits statistically significant shifts in timing or seasonal pattern (only magnitude) (Figure 2-9, 2-10). Furthermore, the WWTF reductions appear to have had no impact on chlorophyll concentrations in either the upper or the lower bay (Figure 2-3, 2-5). However, a statistically significant reduction in annual maximum chlorophyll value had occurred in the lower bay over the course of the entire dataset (Figure 2-5, 2-7).

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Table 2-1: Estimated major sources of Nitrogen (10^6 Moles N as TN) to Narragansett Bay, and potential future change resulting from impending management strategies.

Nitrogen Source	**2003**[a]	**2010 change**	**2014 potential change** [b]	**Notes**
Direct Sewage	170	143 (16% reduction)	up to 60% decrease	2014 value based on RIDEM estimates of loading: 3mg/l for major plants for 2014, 8mg/l for smaller plants. [b]
Indirect (into rivers) Sewage	193	120 (37% reduction)	up to 50-60% decrease	Assumes above plus MA compliance with proposed reductions. Does not account for riverine abatement.
Other riverine inputs & surface drainage	145	129 (11% Reduction	?	may improve slightly due to reduction in ISDS usage, fertilizer restriction, and improved land-use practices. Changes may take years-decades to manifest.
Direct Atmospheric Deposition	30	30	?	unlikely to change significantly, but may decrease slightly due to air quality regulations.
Urban Runoff	37	62(67% increase)	up to 20-30% decrease	Increased precipitation and land-use changes. Potential future decrease from improvements in CSO abatement and land usage regulations.
TOTAL (10^6 Moles/yr)	**575**	**484**[c]	**approx. 270-320**	

[a] Data from Nixon et al. 2008 [b] Estimates from Liberti, 2009 pers. comm. [c] assuming no change in un-estimated parameters.

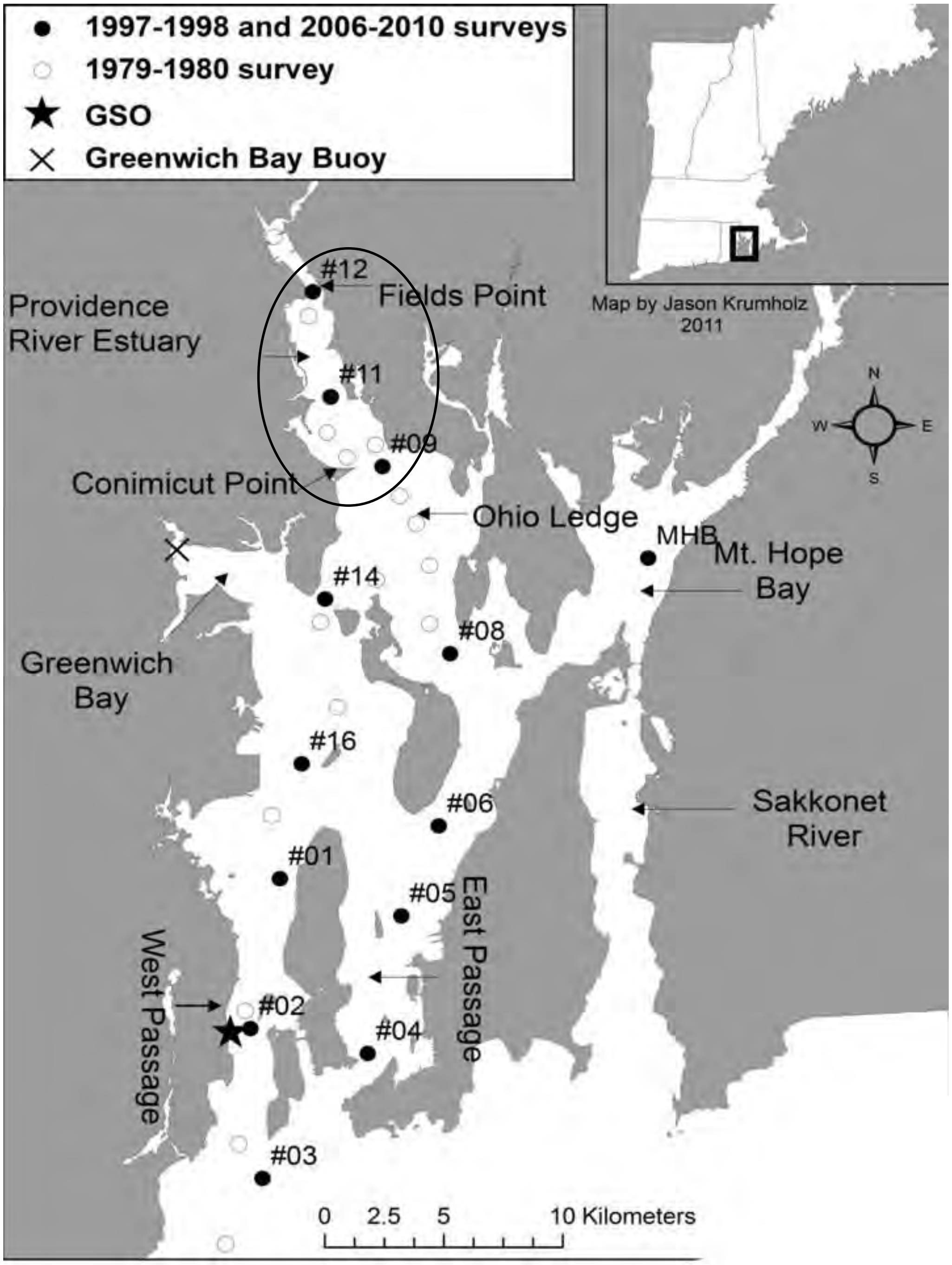

Figure 2-1: Map of Narragansett Bay and landmarks referred to in this manuscript. Sampling stations from the Providence River Estuary averaged in this manuscript to generate ‘upper bay’ values are enclosed in the circle.

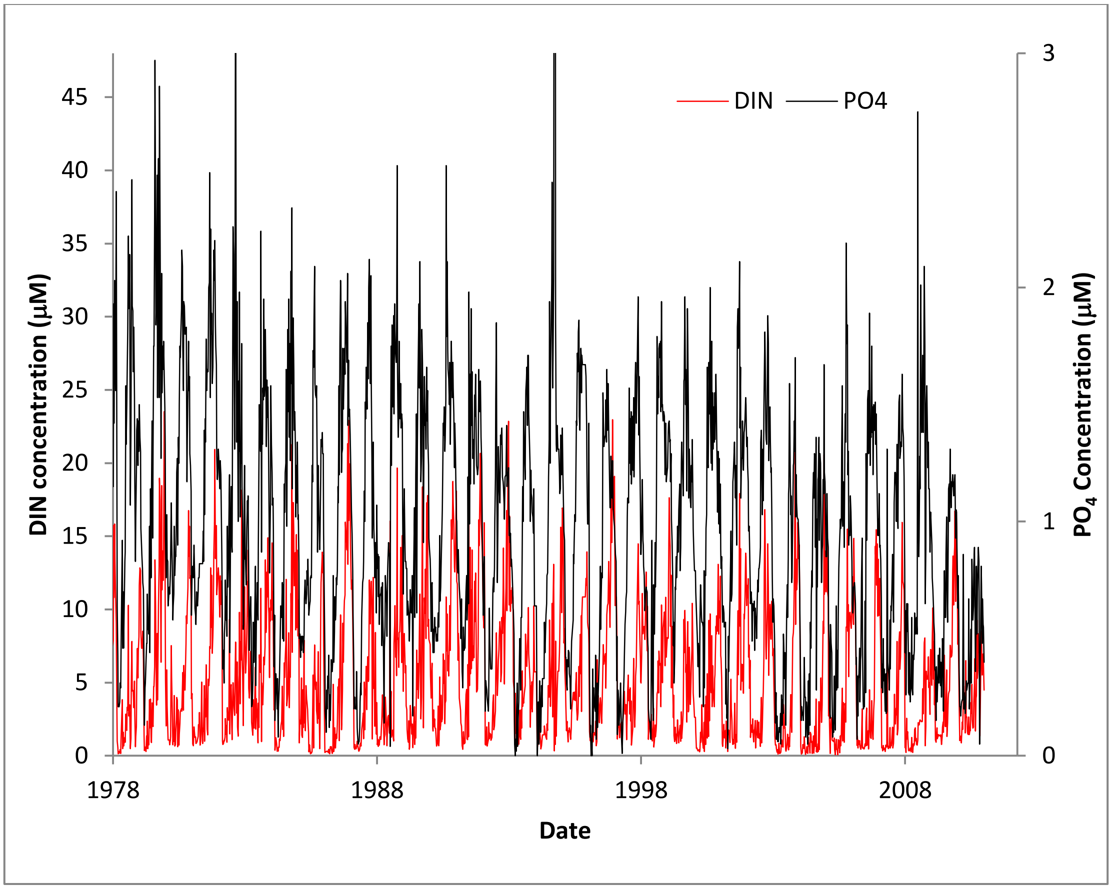

Figure 2-2: Weekly dissolved Inorganic nitrogen and phosphorus concentrations over the 35 year dataset at GSO Pier. Nitrogen (left axis) and phosphorus (right axis) axes are scaled at 16:1.

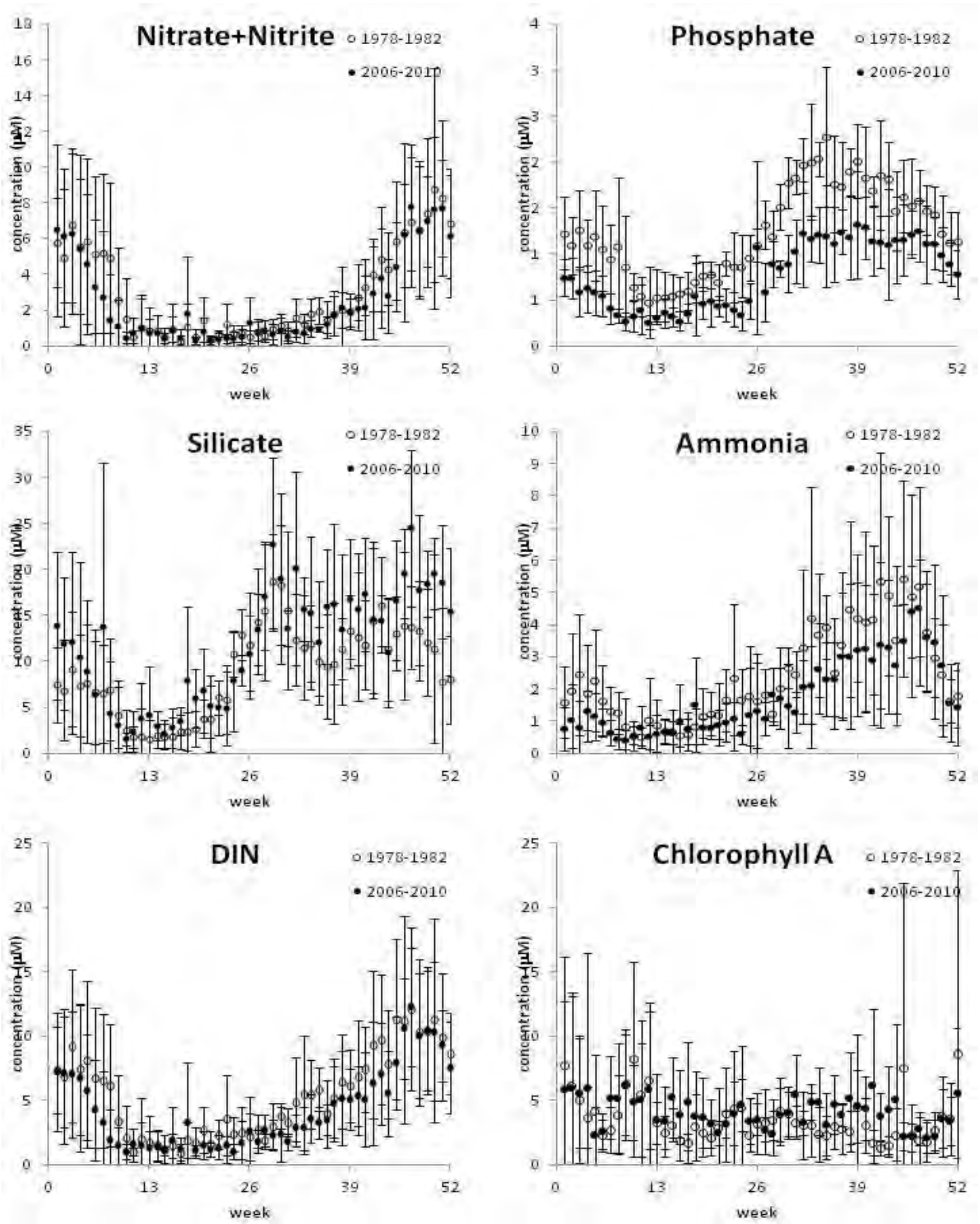

Figure 2-3: Seasonal cycle of nutrient analytes at GSO dock station. Data are annual averages by week for the periods 1978-1982 (inclusive) and 2006-2010 (inclusive). Error bars are the standard deviation of annual values for the given week within the 5 year survey period.

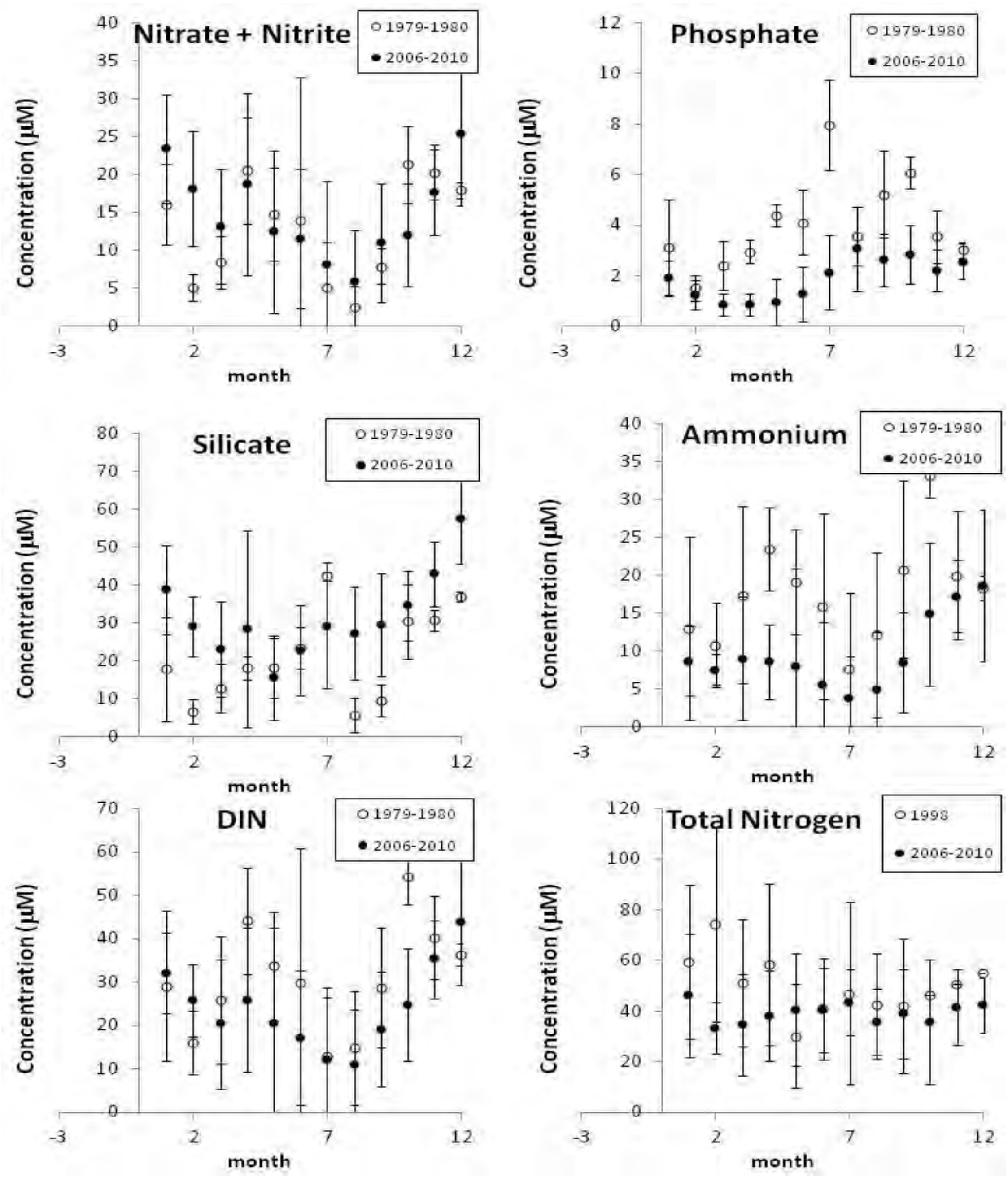

Figure 2-4: Seasonal cycle of nutrient analytes in the Providence River Estuary. Data are averages of all observed values at 3 (2006-2010) or 4 (1979-1980) stations between Conimicut Point and Fields Point during the given month (N= 3-12) for the 1979-1980 survey (Oviatt et al. 1980) and 2006-2010 (inclusive). Error bars are the standard deviation of all values for the given month within the survey period.

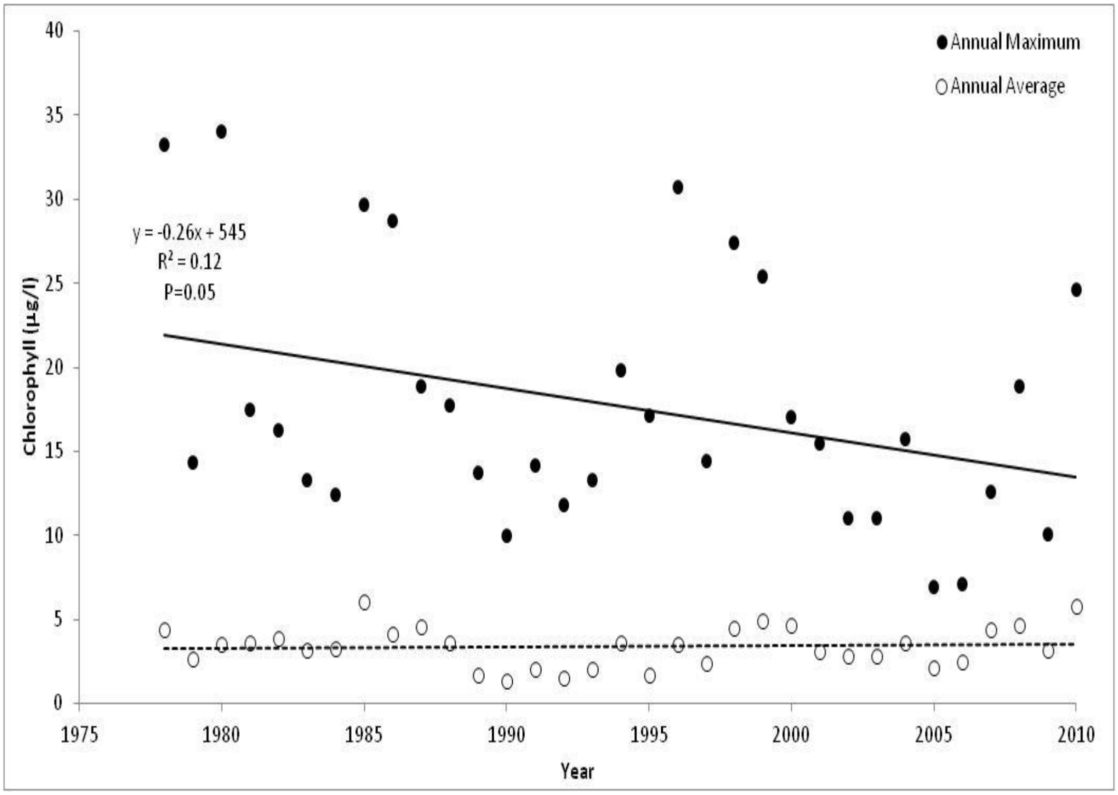

Figure 2-5: Annual average (solid bars) and maximum (hollow bars) chlorophyll at the GSO station over the course of the time series. Annual average chlorophyll shows no long-term trend, while annual maximum shows a slight downward trend of about 0.25 μg/l/y (R^2=0.13, df=33,F-4.14, P=0.05).

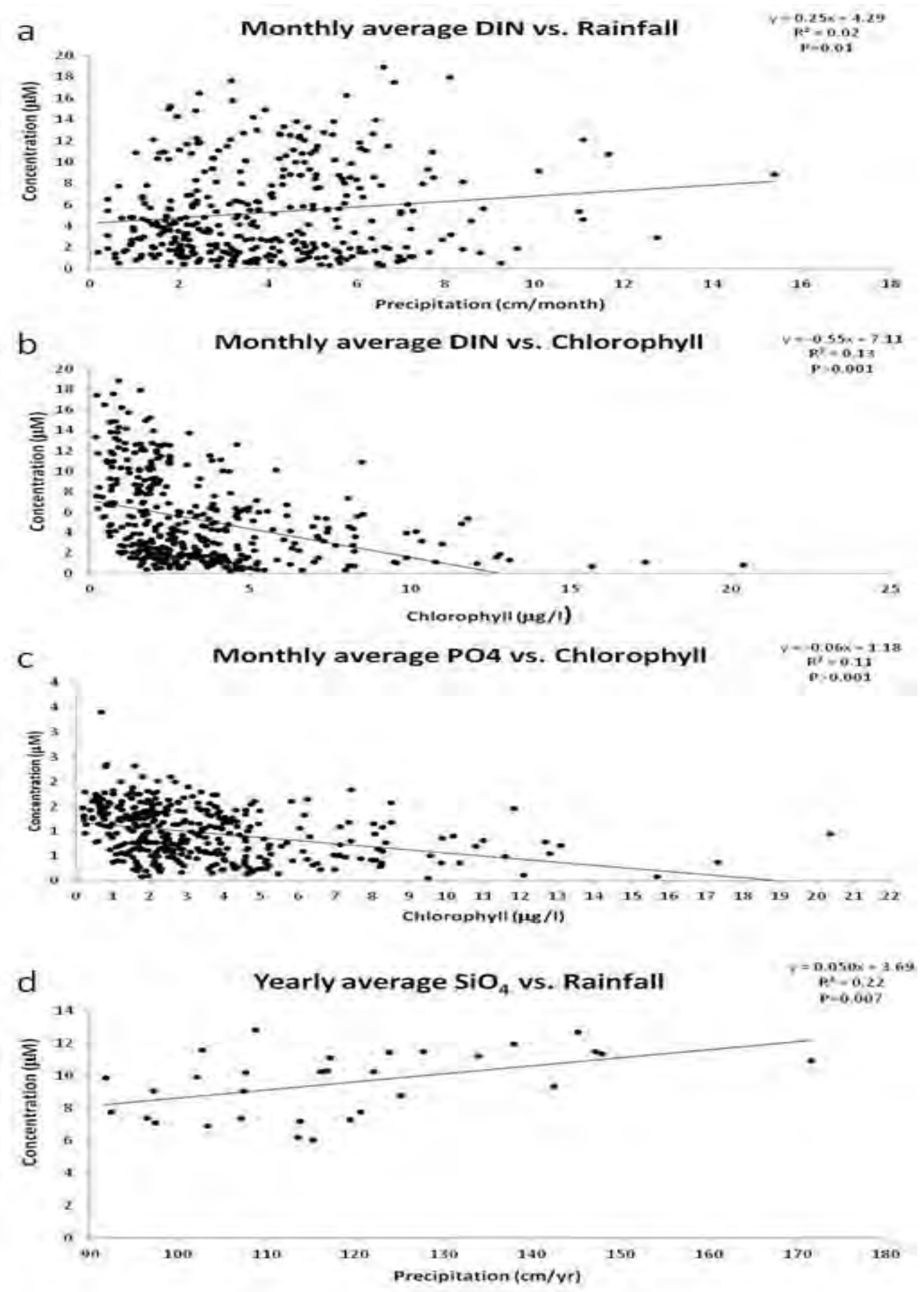

Figure 2-6: Relationships between monthly average DIN and precipitation (a) and chlorophyll (b), and between monthly average PO4 and chlorophyll (c), and yearly average SiO_4 and chlorophyll (d) at the GSO dock station from 1978-2010. Concentration data are the average of all samples taken in that month, and precipitation data are the total monthly precipitation (in rainfall equivalent) at TF Green airport in Providence (NOAA 2011).

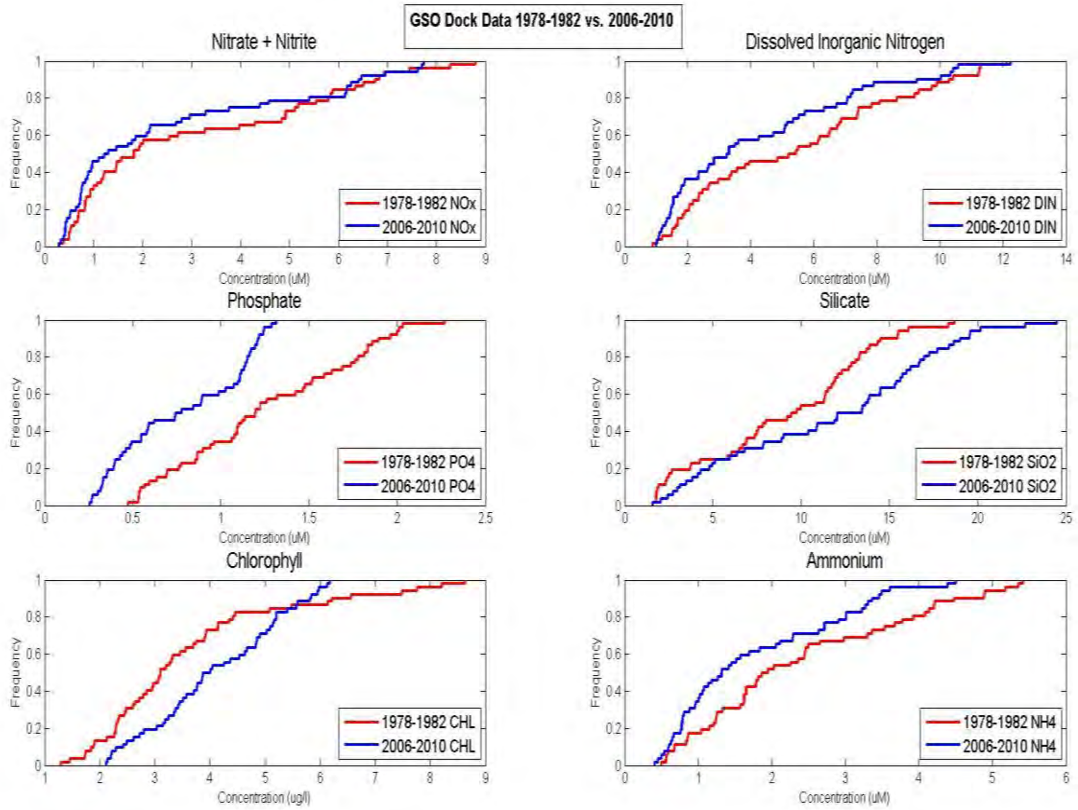

Figure 2-7: Comparison of Cumulative Distribution function of various nutrient analytes at GSO Dock station between 1978-1982 (inclusive) and 2006-2010 (inclusive). 2 sided 2 tailed Kolmogorov-Smirnov testing showed significant differences for Phosphate ($p<.001$, $K=0.42$), Silicate ($p=0.05$, $K=0.36$), Ammonium ($p=0.02$, $K=0.26$), and Chlorophyll ($p=0.005$, $P=0.32$).

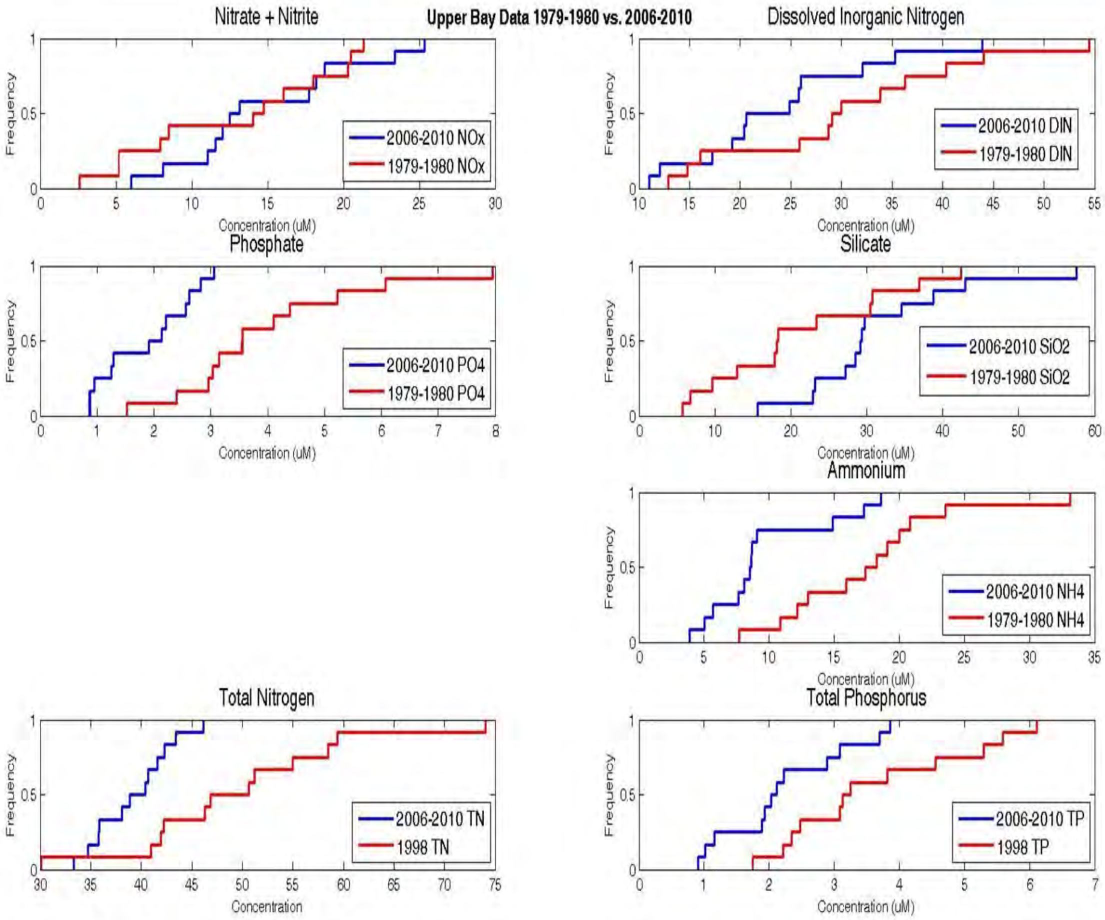

Figure 2-8: Comparison of Cumulative Distribution function of various nutrient analytes at Upper Bay stations in 1979-1980 and 2006-2010 (inclusive). 2 sided 2 tailed Kolmogorov-Smirnov testing showed significant differences for Phosphate ($p<.001$,$K=0.75$), Ammonium ($p=0.004$, $K=0.66$), and Total Nitrogen ($p=0.004$, $K=0.66$), and nearly significant difference for Total Phosphorus ($p=0.06$, $K=0.5$) and Silicate ($p=0.06$, $K=0.5$).

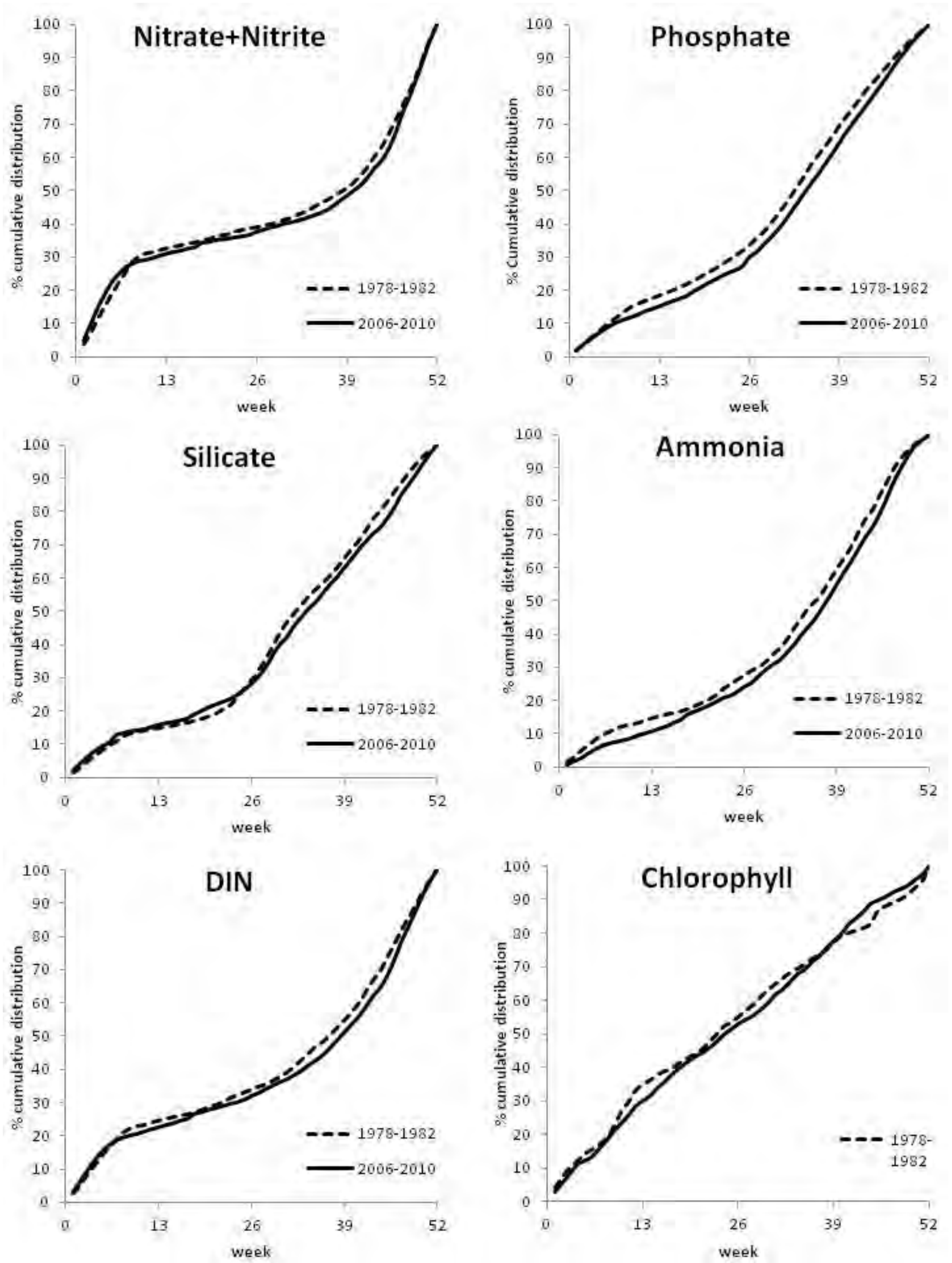

Figure 2-9: Normalized (to % of total observed) seasonal nutrient patterns at GSO Dock Station during the periods 1978-1982 (inclusive) and 2006-2010 (inclusive). Data are annual averages of values in a given week. Y-axis labels are cumulative percent contribution for that analyte at that time of year.

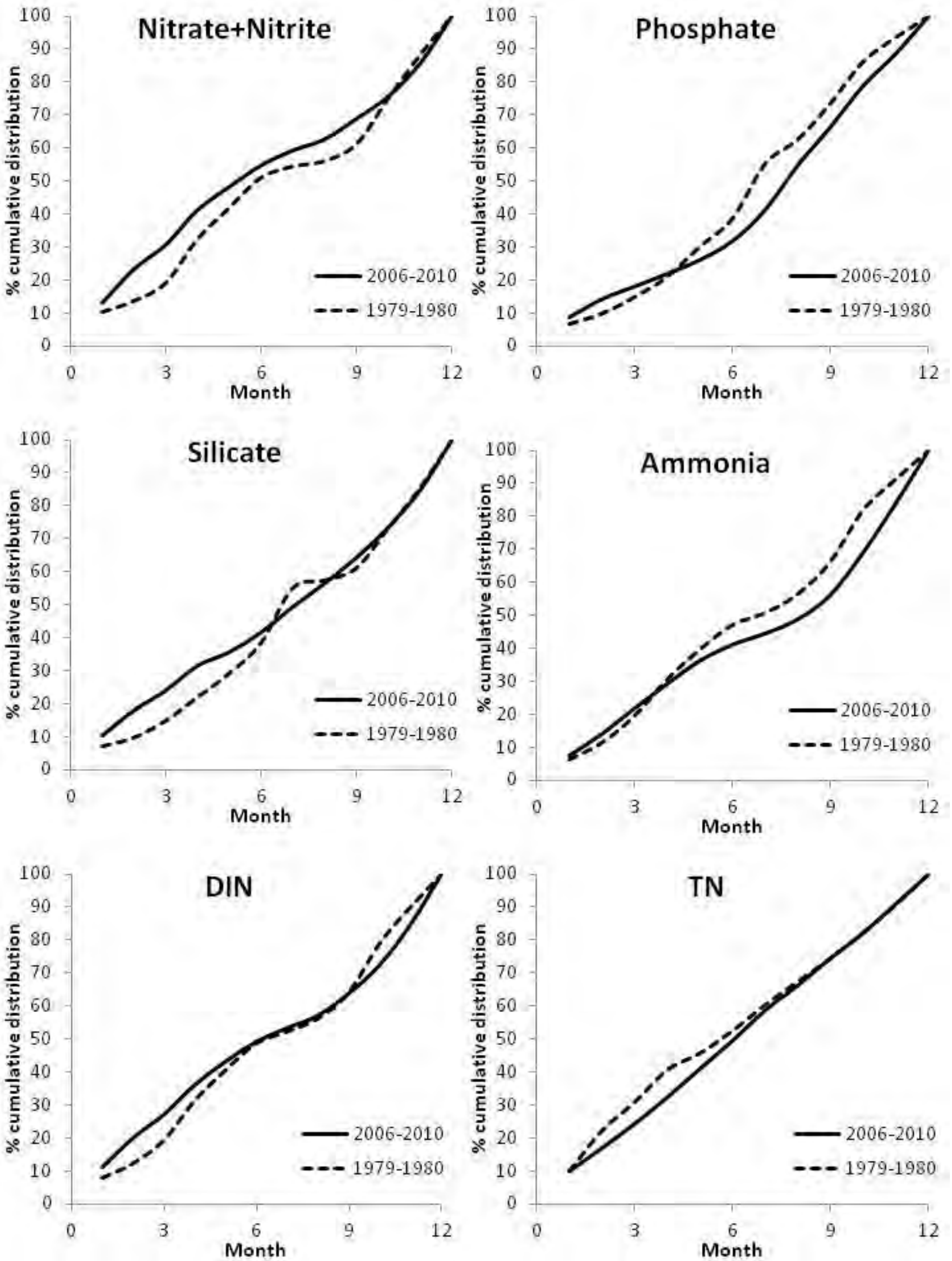

Figure 2-10: Normalized (to % of total observed) seasonal nutrient patterns for the average of 4 (1979-1980) or 3 (2006-2010) stations in the Providence River Estuary (Between Conimicut Point and Fields Point) during 1979-1980 and 2006-2010 (inclusive). Data are average of monthly averages for each year surveyed. Y-axis labels are cumulative percent contribution for that analyte at that time of year.

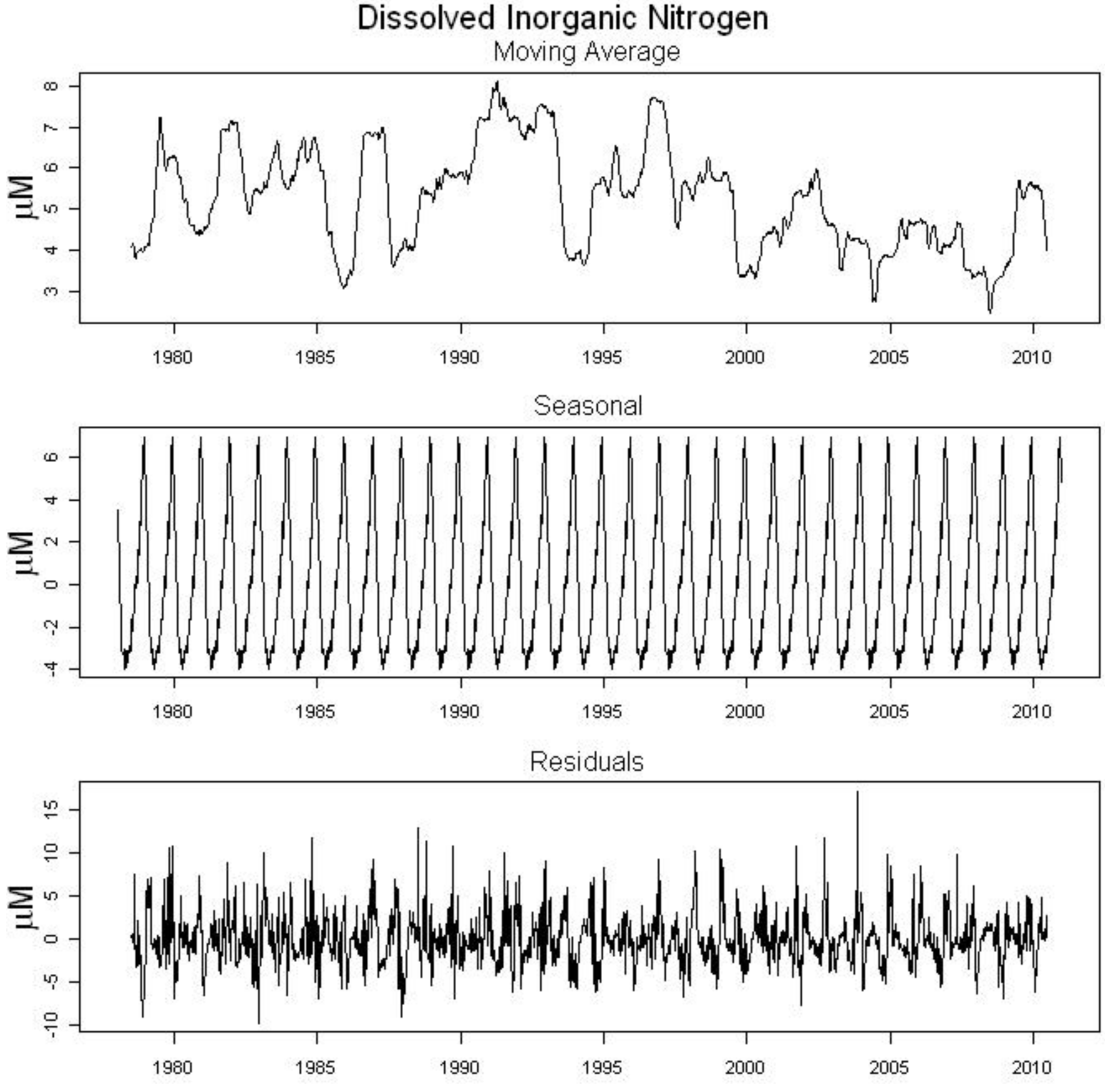

Figure 2-11: SSPIR model results for dissolved inorganic nitrogen showing 52 week moving average (top), seasonal cycle (middle) and residual signal (bottom) of the modeled trend.

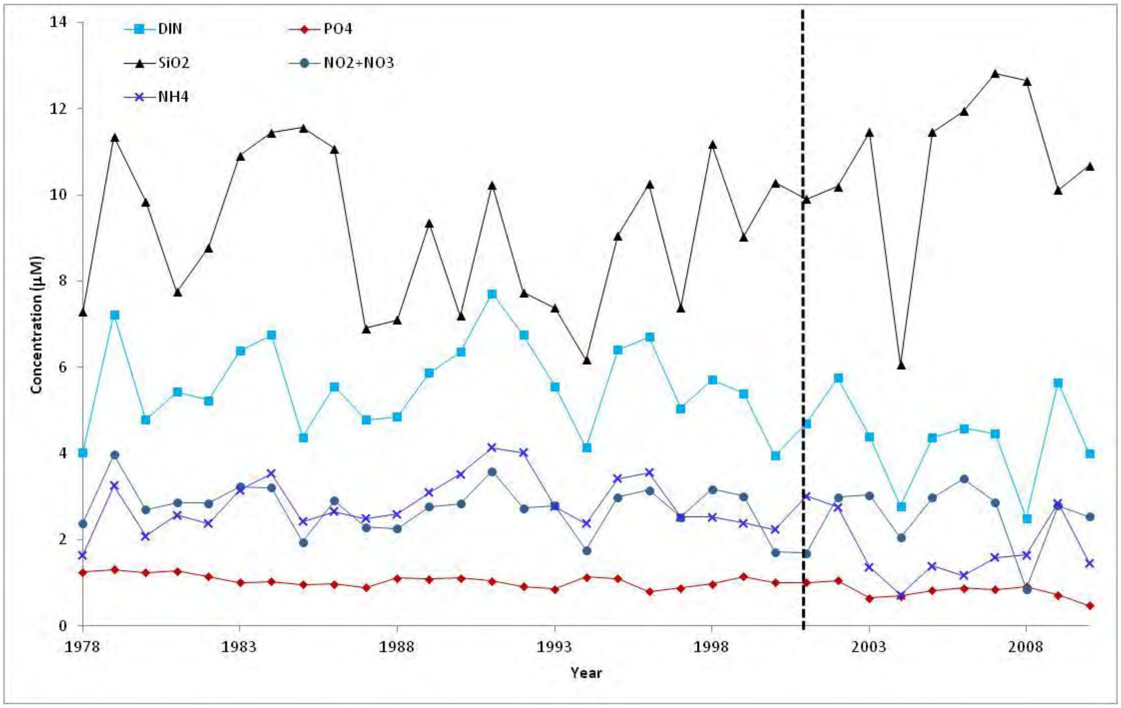

Figure 2-12: Annual average of nutrient analytes at GSO dock station 1978-present. Dashed line shows beginning of implementation of advanced wastewater treatment. Only phosphate shows significant reduction (regression $R^2=0.44$, df=27, F=20.7, P>0.001) prior to the implementation of wastewater treatment. All analytes except silicate show significant reduction between pre and post treatment year

CHAPTER 3

AN ASSESSMENT OF THE IMPACT OF NUTRIENT LOADING REDUCTIONS ON THE ANNUAL MASS-BALANCE OF NITROGEN AND PHOSPHORUS IN NARRAGANSETT BAY

ABSTRACT

Narragansett Bay is a relatively well mixed, high salinity estuarine ecosystem with low fresh water inflow. Much of the shoreline is developed, and most of the sources of nutrient load to the bay are located in the head of the estuary. Recently, several wastewater treatment facilities which discharge into the bay or its tributaries have upgraded to advanced wastewater treatment, with upgrades at the remaining plants following within 2-4 years. We review the mass-balance of nitrogen and phosphorus in the bay, examining the contribution of inorganic and total nitrogen and phosphorus to the bay from atmospheric deposition, river loading, wastewater treatment plants, groundwater and urban run-off, and loss terms from fisheries, denitrification, sediment burial, and export. For the first time in a mass-balance of this system, we attempt to calculate flux across the bay/sound interface rather than estimating it by difference.

Our results show a total load to the system of 488 million moles total nitrogen (TN) and 25.8 million moles total phosphorus (TP) per year. This works out to about 1.48 moles and 0.078 moles of TN and TP per square meter per year respectively, a value which falls near the center of the range of similar urban estuaries (e.g. Bricker et al. 2007, Boynton et al. 2008), though the overall N:P of inflows is nearly 19:1, while

most similar systems are below 16:1 (Boynton et al. 2008). The reduction in total system loading from sewage of roughly 100 million moles TN and slightly more than 4 million moles TP, constitutes reductions of roughly 28% and 22% of sewage based nitrogen and phosphorus respectively, which translates to roughly 17% of the total load of both nitrogen and phosphorus to Narragansett Bay from all sources. Most of these reductions reach the bay, though some of the upgrades to plants in the Blackstone River are mitigated before that river meets the estuary proper. Sewage, whether directly or indirectly discharged into the bay, accounts for just over half of the TN and TP discharged to the system, a reduction when compared to past studies. Our estimates of offshore flux indicate that approximately 65% of the TN load, but slightly higher than 100% of the annual TP load are fluxed offshore from the bay. The former estimate is in line with past estimates, but the latter, if correct, may indicate that the system is not at steady state with regard to P.

INTRODUCTION

The compilation of elemental mass-balances for estuarine systems is a topic which has been of interest to science for over a century since James Johnstone compiled a nitrogen budget for the North Sea (Johnstone 1908). Concern over eutrophication in Narragansett Bay, similarly, has roots stretching back over a century to the pioneering work of George Field and colleagues at the RI College of Agriculture experimental station (Field 1898), and perhaps even further to the work of Justus von Leibig in the mid 19th century. Furthermore, most modern nutrient budgets address many of the same components addressed by Johnstone (1908) in his initial attempts (though often our estimates are somewhat better constrained). Yet this tool continues to be of great interest to scientists and managers alike, with the Thompson ISI web of knowledge (apps.webofknowledge.com) reporting 384 marine or freshwater nutrient mass-balances published in the last three years (2009-2011) alone.

The question of why nutrient mass-balances (nutrient budgets herein) have garnered attention through the years and yet, have remained fundamentally unchanged in their execution, has to do primarily with the fact that a mass-balance is rooted in simple arithmetic and basic physical properties. A body of water must, over the long-term, balance what comes in and what goes out, and the physical vectors for these fluxes have changed little over the last century. Nutrients enter the estuary through flow from tributary rivers, and in the case of nitrogen, through direct deposition from the atmosphere. The standing stock within the estuary exchanges nutrients with the sediments through burial and resuspension/remineralization, and with the open ocean through tidal flushing and circulation. Biota can assimilate nutrients, changing their form (from inorganic to organic) and, to a limited degree, can export nutrients from the system via advection, migration, or anthropogenic capture. Recently, we have also begun to consider the ability of biota to export and import nitrogen into the system via nitrogen fixation and/or denitrification, though the magnitude of this flux can be very variable and is often not well constrained (e.g. Lipschultz and Owens 1996, Larsson 2001, Fulweiler and Nixon 2011).

While the principles of nutrient mass-balance have changed little over the last century, the level of technology with which the problem can be approached has dramatically increased over the last few decades. While traditionally most nutrient budgets have assumed a closed system, and calculated at least one major term of the budget by difference, increasing availability of computer driven circulation models such as the Regional Ocean Model System (ROMS) has made estimation of flux at the ocean estuary interface (typically the most difficult of the terms to estimate) more feasible. Availability of GIS based tools has also greatly improved the accuracy of estimating fluxes from the watershed such as urban runoff, atmospheric deposition, and land-use changes.

Contemporary with these improvements in technology, management attitudes with respect to nutrient loadings in marine systems have begun to shift as well. Up until a few decades ago, estuaries around the world were on a general trend of eutrophication, predominantly at the hands of anthropogenic processes such as fertilizer use, wastewater disposal, and increases in impervious surface (Clarke et al. 2006, Bricker et al. 2007, King et al. 2008). Recently, however, a sharp increase in the number of management actions to curtail, and in some cases reverse, this trend of eutrophication has renewed interest in conducting mass-balances in these systems to determine the impact of these management actions on the various exchanges of nutrients within the estuary and the response of the system to load reductions (e.g. Carstensen et al. 2006, Artioli et al. 2008, Boynton et al. 2008, Eyre et al. 2011).

This manuscript aims to update the nitrogen and phosphorus budget for Narragansett Bay, a temperate New England estuary. Past budgets for this system have been conducted approximately once per decade (e.g. Nixon et al. 1995, Nixon et al. 2008). The most recent budget was published in 2008, using a combination of data collected during the 2003-2004 field season and ‘carry over’ data from the 1995 budget, most of which were collected in the 1980’s. Recently, Rhode Island Department of Environmental Management (RIDEM) has required that several of the major wastewater treatment facilities (WWTF) which serve Narragansett Bay be upgraded to tertiary sewage treatment, with most other large plants planning upgrades in the next few years (RIDEM 2005). The overall goal of RI General Law § 46-12-3(25), the driving force behind these changes, is to reduce nitrogen loading to the bay from WWTF’s by 50%, a task that, based on percentage reductions achieved at the plants which have already upgraded, will be achieved once the largest plant discharging into the bay, located at Fields Point (Figure 3-1) completes upgrades, presently scheduled to be sometime in late 2013 or 2014. RIDEM is also imposing phosphorus loading limits on plants which discharge into tributary rivers of the bay.

While nitrogen reduction is typically accomplished by bacterially mediated coupled nitrification/denitrification (Lishman et al. 2000, Jeong et al. 2006), phosphorus reduction is typically accomplished by chemical scavenging, though more advanced biological techniques may be on the horizon (Strohm 2006). These processes result in very different limits being imposed for the different nutrients. Because the bacterial nitrogen removal process is often temperature dependent, both in nature and in WWTF's (Nowicki 1994a, Lishman et al. 2000) most plants discharging into the bay are required to reduce total nitrogen load in effluent to either 8 or 5 mg/l (0.6 or 0.4 millimolar) during the active (May-October) season, and to the maximum extent possible during the colder winter months (Liberti, pers. Comm). During the active season, this is typically a reduction of 60-70% from the concentration before upgrade (see appendix B). In contrast, phosphorus is chemically scavenged from the wastewater, which is a process that is not temperature dependent and capable of much higher removal rates. Many plants which are upgrading to remove phosphorus have or soon will have limits of 0.1 mg/l (3.2 mM), a reduction of 90% or more. This changes the molar ratio of N:P in effluent at these plants from about 7:1 to somewhere between 22-35:1.

In light of these changes, a re-assessment of the nitrogen and phosphorus budget of the bay is justified. The main question we aim to answer through this exercise is whether other parameters of the budget have also changed in response to reductions from the WWTF's. As such, we have made efforts to update estimates of as many parameters of the budget as possible while adhering to the general framework laid by the most recent (Nixon et al. 2008) system budget, so as to isolate sources of change to the system from changes to our estimates of system parameters resulting from improved estimation techniques (which also occur).

We reassessed loading of nitrogen and phosphorus from rivers, wastewater treatment plants, groundwater and urban run-off. We also reassessed the role of the sediments as a source/sink of nitrogen, export resulting from secondary production (fish), and attempt to close the budget by using the EcoGEM model (Kremer et al. 2010) to predict the flux of nitrogen and phosphorus across the bay/sound interface. However, in many cases, sufficient data for a new parameter estimate were not available. In these cases, parameters were carried over from the most recent budget, rather than risking estimation based on incomplete or insufficient data.

STUDY SYSTEM

For the purposes of this paper, we will adhere to the convention used in past budgets, of defining the bay as all of the portions of the bay proper, the East and West Passages, Mount Hope and Greenwich Bays (as well as many smaller bays and harbors), and the Providence River Estuary. All of these sections of the bay exchange freely with each other and with Rhode Island Sound on the southern boundary (Figure 3-1). Similarly, we choose to exclude the Sakonnet river, as have past budgets, because its exchange with the bay proper is limited to a very small breachway and it receives little direct input of fresh water or sewage (Nixon et al. 1995).

When considered in this way, the bay has an area of 328 km^2 with an average depth of about 8.6 meters and a watershed to surface area ratio of roughly 11:1 (Chinman and Nixon 1976, Pilson 1985a). Circulation in the bay is predominantly tidally driven, with the mean flow direction in the East Passage and out the West

Passage (Kincaid et al. 2008, Rogers 2008). Freshwater input is small, presently averaging about 103.8 m^3/s (Spaulding and Swanson 2008), which is virtually identical to the value used by Nixon and colleagues in past budgets of 105 m^3/s which was calculated by Pilson in the 80's (Pilson 1985a). A more detailed description of the ecology of the bay can be found in Kremer and Nixon (1978) or Desbonnet and Costa Pierce (2008).

When compared to other similar temperate estuaries, Narragansett Bay is generally considered moderately eutrophic (e.g. Bricker et al. 2007), with a nitrogen and phosphorus load per square kilometer which ranks 11th and 10th highest out of 35 estuaries surveyed by Boynton and colleagues (2008) and 8th in nitrogen load/km^2 among 33 systems surveyed by Latimer and Charpentier (2010). Prior to upgrades, Narragansett Bay received approximately 65% of its nitrogen load from sewage discharged either directly into the bay or into its tributaries (Nixon et al. 2008), which is nearly double the average of 36% found by Latimer and Charpentier (2010). This loading makes it an excellent candidate for assessing the impact of load reductions from WWTF's on components of the system budget.

METHODS/DATA SOURCES

STANDING STOCKS AND WATER COLUMN CONCENTRATIONS

Water column nutrients in the bay were measured from monthly surface water collection at 13 stations throughout the bay (Figure 3-1) from 2006-2010 collected as part of the CHRP/NuShuttle and MERL sampling cruise and augmented with data from QA/QC samples collected sporadically at the Greenwich Bay fixed buoy station. To buffer against interannual variability, which can be significant, typically the average of 2006-2010 annual averages is presented, with confidence interval given as the standard deviation of annual averages. In virtually all cases, the natural variability exceeds any sources of measurement error by at least two orders of magnitude (see appendix A), so it was deemed unnecessary to propagate sources of error. Standing stocks were calculated using volume estimates from the GEM box model (Kremer et al. 2010). For greater detail on the methodologies associated with the collection and analysis of these data, and the compilation of standing stock values, please refer to Chapter 1 and Appendices A and D.

RIVERS

Data for river concentrations of phosphate (PO_4), nitrite (NO_2), nitrate (NO_3), ammonium (NH_4), and total nitrogen (TN) were provided by the Narragansett Bay Commission (NBC). Data were collected approximately biweekly during the time period of 2006-2010, with a total of 107 samples collected during these five years (slightly more than 21 per year on average). Samples were collected from 15 stations on rivers discharging into the bay. However, for this study, only the stations closest to the mouth of the Blackstone, Pawtuxet, Taunton, Woonasquatucket, Moshassuck, and Ten Mile rivers were used (Figure 3-2). Combined, these five rivers account for nearly 80% of the flow entering the bay (Ries et al. 1990, Nixon et al. 1995). These data were analyzed by NBC personnel, using standard colorimetric autoanalysis techniques (NBC 2008). An intercalibration between the instrument used for these samples, and the one used for the water column samples was conducted in 2005 to ensure inter-comparability of data (NBC 2008).

To estimate flux requires flow and concentration. Daily average flow data for the rivers in question are available for download on the USGS website: http://waterdata.usgs.gov/ma/nwis/current/?type=flow. Flow was corrected for ungauged area below the monitoring stations using ratios calculated by Pilson (1985a) and Boucher (1991) as per Nixon and colleagues (1995, 2008). To arrive at daily flux by combining daily flow measurements with periodic concentration values, there are a number of techniques used in literature. In this case, we chose Beale's unbiased estimator (Beale 1962) for several reasons: flow and concentration are weakly correlated, flow data are positively skewed, and the sample size in any given year is relatively small (<50). Comparisons of results using different estimation techniques to estimate flux in this way show Beale's to be well suited to these types of data, and in most cases, show little difference between techniques (Tin 1965, Fulweiler 2003). Furthermore, Beale's estimator was used by Nixon and colleagues in past budgets, so given no indication that a different technique would produce superior results, Beale's is the logical choice. Briefly, Beale's estimator works by comparing the flow on measured days to the mean flow, and correcting the estimated flux for any bias imposed by the less regular concentration sampling régime. A more thorough review of the application of Beale's estimator can be found in Dolan et al. (1981) or Fulweiler(2003).

There is a significant amount of interannual variability in river load, based in large amount due to variability in precipitation. To arrive at a better estimate of the average loading to the bay from this source, we calculated the average loading from each river from a three year period, 2008-2010, and compared this loading to the most recent published values from Nixon et al. (2003-2004 for all rivers but the Taunton, for which Nixon and colleagues used a dataset from the 80's). We were able to make direct comparisons for dissolved and total nitrogen, and dissolved inorganic phosphorus; unfortunately, we did not have data for total phosphorus in this dataset. To arrive at an estimate of this parameter for the budget, we calculated the average ratio of total phosphorus to inorganic phosphorus from the several surveys presented in Nixon et al. (2008, table 5.9) for each river. Finding relatively consistent relationships (RSD<30% in all cases) we used this value to extrapolate total phosphorus from inorganic. However, Nixon and colleagues were unable to get data from the Taunton River at that time.

TREATMENT PLANTS

There are 29 WWTF's that discharge their effluent into Narragansett Bay. Of those, 10 discharge their effluent directly into the bay and 19 discharge into tributary rivers which subsequently drain into the bay. A total of 21 of the plants including four of the five largest plants, discharge either directly into the Providence River Estuary or into its tributaries, with four discharging into the Taunton River and Mt. Hope Bay and one into Greenwich Bay. The remainder discharge directly into the mid or lower bay. For the purposes of calculating total nutrient load to the bay, plants discharging into rivers are considered as part of the flux from those respective rivers (to avoid double counting and allow for river abatement). RIDEM was able to provide data for discharges from 17 of these plants including 8 out of the 10 plants which discharge directly into the bay. In most cases, these data were collected weekly, though larger plants were sampled more often, and smaller plants as infrequently as every other month. In all cases samples were 24-hour average composites of samples collected every 30 minutes. (NBC 2008 Liberti pers. comm., see appendix B for more details) Beale's estimator (Beale 1962) was again used to calculate flux from flow and concentration data provided.

Where data were not available, we adjusted numbers from past budgets to account for changes in population served by those plants. In some cases, past budgets had estimated data using an average value of N and P load per person multiplied by the number of people served by the plant. In these cases, we used the same technique, but we found that the average load per person per day (even among plants which have not upgraded) has changed since Nixon and colleagues estimated it, so we revised the estimated load from 0.9 and 0.035 moles of N and P per person per day (Nixon et al. 2008) to 0.8 and 0.045 moles of N and P, respectively, per person per day (Appendix B).

Because most plants which upgraded did so in the 2005-2006 time frame, we compare annual averages from the years 2008-2010 to annual averages from 2000-2003 (from Nixon et al. 2008) to ascertain the impact of advanced treatment on loadings. A few plants (North Attleboro in 2008 and Worcester in 2009) upgraded after the others, and for those plants, we calculate the 'post upgrade' averages using only the available data after the upgrade was completed. For plants where 2000-2003 data were available, we recalculated fluxes to ensure that our methods were comparable to Nixon and colleagues, and found excellent agreement, typically to within rounding error.

ATMOSPHERIC DEPOSITION

Atmospheric deposition onto the watershed of Narragansett Bay is accounted for in river loading, therefore only the direct wet and dry deposition of nitrogen and phosphorus onto the surface bay are of concern. These loadings were estimated by Nixon and colleagues from data collected at the Graduate School of Oceanography (for P) in the late 1970's and on Prudence Island (for N) in the 1980's (Nixon et al. 1995) and have in the past generally been found to be a small (<5%) portion of the overall budget. While no new direct measurements of deposition were made for this study, we did compare the results from these studies to more recent estimates of deposition rates from the New England area (Howarth et al. 2007, Howarth 2008) and found the results to be similar. While environmental regulations have improved the emissions of NO_x from automobile and industrial exhaust, the number of car miles driven on New England roads has increased 70% since 1970 (Howarth 2008), resulting, it seems, in an overall deposition figure which has likely changed little since it was last measured. Furthermore, while direct deposition is a major factor in some systems, contributing 4-35% of the load incident on 40 major coastal watershed surveyed by Alexander et al. (2001), it is a relatively minor player in Narragansett Bay, despite a relatively high flux per unit area (Howarth 2008). For these reasons, lacking more recent direct measurements, and with no evidence suggesting that loading from this vector has changed significantly in the intervening time-span, we chose to carry over estimates of direct deposition from past budgets.

URBAN RUN-OFF

A previous study of nutrient loading from various land-use types during 12 storms over the course of 1979-1980 (Carter 1982) has provided the basis of estimates of urban run-off for the last several mass-balances conducted. While this study has the distinct benefit of being conducted in the Narragansett Bay watershed, the amount of data available and the number of land-use types surveyed was very limited, with the flux per acre coefficients for many land-use types determined by only a few data points. An estimate of flux from urban run-off was calculated by multiplying the coefficients determined by Carter (1982) by the long-term average precipitation at the time of 1.19 m/y, and by the approximate number of acres of each land-use type in cities and towns which discharge their stormwater directly into the bay (Nixon et al. 1995). This estimate of the flux from urban run-off has been used, essentially without revision, for the last 30 years.

We made several adjustments to this value. First, we used the identical method to Nixon et al. (1995), adjusting only for changes in land-use and precipitation. Land-use was adjusted by comparing present and historical GIS land-use coverage in the towns surveyed using ArcGIS 9.2, and precipitation was adjusted to the 10 year average between 2000-2010. Next, we considered all land-use types occurring within the areas which discharge directly to the bay. Though the majority of land-use types in terms of acreage are covered by the four categories used in Carter's survey (Residential, Commercial, Industrial, and Highway), remaining land-use types are ignored in that study, and subsequently in acreage estimates used by Nixon et al. (Carter 1982, 1995). Rather than ignoring these other land use types (e.g. mixed use, transitional, institutional, and open space) we assigned each to the land-use category from Carter's work which most closely approximated it. With the exception of open space, we were able to arrive at a reasonable analogue from Carter's work (sometimes averaging her coefficients for areas zoned as mixed use). For areas zoned as open space, we used the nationwide average coefficient from an NRC report (NRC 2008) on urban storm water. Finally, we considered the variability inherent in this prediction by comparing the results derived from using the coefficients determined by Carter (1982) to results derived if the NRC coefficients (NRC 2008) were used for all land-use types. While the NRC coefficients gain several additional coverage types, and benefit from a large number of samples within each coverage, these samples are nationwide averages, and the amount of nutrient in urban storm water run-off is very system specific. Thus, while it is impossible to tell which set of coefficients is more 'correct', this analysis at least gives us an idea of the variability inherent in our ability to estimate this term of the budget.

PRIMARY PRODUCTION

While primary production does not, in itself, change the amount of nutrients coming into or out of the bay, it is an important vector for moving nutrients between the various pools and sinks (e.g. transforming inorganic nutrients to organic, moving nutrients from the water column to the sediment, etc…) and also is highly relevant to the discussion of loading reductions from a management perspective. A very robust survey of primary productivity in Narragansett Bay over an annual cycle was conducted by Oviatt and colleagues in 1998 (Oviatt et al. 2002). Data collected by Smith (2011) in 2006-2008 do not show conclusive evidence of changes in primary productivity since the 1998 survey at 5 stations in the Providence River estuary and the West Passage. Since the latter survey occurs after the majority of the WWTF upgrades (including the largest, at the Bucklin Point facility in East Providence) we assume that primary productivity in the bay has not changed significantly since 1998, and therefore because the 1998 study has greater spatial coverage, we use the regressions established therein.

DENITRIFICATION

It is fortunate that Narragansett Bay has been the site for several studies on the net flux of nitrogen into and out of estuarine sediments. The estimates used by Nixon et al. (Nixon et al. 1995, Nixon et al. 2008) are built upon a series of studies conducted in the bay (Seitzinger et al. 1984, Nowicki and Oviatt 1990, Nowicki 1994a) and at the MERL mesocosm facility at the Graduate School of Oceanography which established in situ denitrification rates, and extrapolated those values using regressions between temperature and denitrification rate established by mesocosm study.

More recently, Fulweiler and colleagues have measured denitrification at the same mid-bay station as well as several other stations throughout the bay, and observed dramatic differences in sediment nitrogen and phosphorus flux (e.g. Fulweiler et al. 2007, Fulweiler et al. 2010, Fulweiler and Nixon 2011). In 2005 and 2006, they noted a large reduction in denitrification rate, with the sediments serving as a net source (nitrogen fixation) rather than a sink (denitrification) of nitrogen during parts of the year (Fulweiler et al. 2007). Furthermore, Fulweiler and colleagues noted a lack of the previously established pattern of spatial and temporal relationships in denitrification rate in the bay (Fulweiler et al. 2010, Fulweiler and Nixon 2011). We revise estimates of net sediment nitrogen flux calculated in past budgets by re-estimating baywide flux using data from these manuscripts.

SEDIMENTS

As with many such systems (Carstensen et al. 2006, Clarke et al. 2006, Boynton et al. 2008), the sediments of Narragansett Bay are a key storage term in the nutrient budget, since most of the sediment that enters the bay likely remains within the system (Nixon et al. 1995). Mesocosm experiments in the MERL facility have shown the sediments of Narragansett Bay to have generally short ‘memory’ and rapidly achieve equilibrium with overlying water via remineralization within an annual cycle (Kelly and Nixon 1984, Oviatt et al. 1984, Kelly et al. 1985).

However, long-term burial in the sediments is a form of export from the system which must be considered. Nixon and colleagues estimated the amount of N and P buried in this way by multiplying sedimentation rate determined from radiometric dating of ^{210}Pb and ^{137}Cs as well as other organic pollutants and metals in sediment cores (Corbin 1989) with measurements of N and P in sediments below the zone of biological activity (e.g. Nixon et al. 1986, Nixon et al. 1995). While it would be ideal to have revised estimates of this parameter, sedimentation rates in the bay do not appear to have changed dramatically (Hartmann et al. 2005). Furthermore, given the amount of time it takes for sediments in the bay (and the nutrients they contain) to be buried (Nixon et al. 1986, Corbin 1989), it seems unlikely that the concentration of nutrients in the sediment being buried would have changed significantly as a result of loading reductions which occurred only a few years ago. Therefore, carrying over estimates of sediment nutrient burial seems to be a reasonable assumption.

As a point of reference, we can also estimate the ‘standing stock’ of nutrients stored in the sediments which is theoretically bioavailable. Using published estimates of N and P concentration in bay sediments from mesocosm work by Nowicki and Oviatt (1990), and assuming that sediments are bioavailable down to a depth of 10 cm and that concentrations measured by Nowicki and Oviatt remain constant throughout this bioturbated layer, we can arrive at a cursory estimate of the amount of nitrogen and phosphorus are in short term storage in the bioavailable sediments at any given time, using concentrations from Nowicki and Oviatt’s 8X enriched experiment for the Providence River Estuary, and the control sediments for the rest of the bay (Nowicki and Oviatt 1990).

FISHERIES LANDINGS

Export of nutrients from the bay from fish and fisheries landings is extremely difficult to quantify. Most of the commercially captured finfish species in the bay are migratory, spending only part of the year within the bay. Thus, it is a grossly inappropriate assumption to calculate finfish (or even lobster) landings in the bay, determine the amount of nitrogen in that biomass, and assume it is an export of bay sourced nitrogen. In past budgets, Nixon and colleagues have been limited to estimating hard clam landings (the only major sessile species harvested in the bay) as a source of export.

Recently Longval (2009) calculated biomass spectra for the Narragansett Bay fish community. As part of this study, she compared the biomass spectra across a seasonal cycle. In this analysis, a clear peak in biomass exists which matches size with several very common small age 0 fish (typically species such as scup, butterfish, and clupeids) which recruit in the spring and grow over the course of the summer. By comparing the biomass in this peak when the fish first recruit to the net (1cm mesh) in the spring to the biomass in the same peak in the fall, one can achieve a rudimentary, albeit highly conservative estimate of fish biomass which can directly be attributed to Narragansett Bay, virtually all of which is exported from the bay, either as fisheries landings, or in the stomachs of other fish which either move offshore, or are captured. We therefore supplement a revised hard clam harvest estimate with the estimate of fish biomass export achieved in this way.

BOUNDARY FLUXES

We attempt herein to model, rather than calculating by difference, the flow of nutrients across the bay/sound interface, and thus, to ‘close’ the total system budget. Fully closed nutrient budgets are becoming more common as more advanced computer simulations improve our ability to model water flow in and out of a system. In this case, we use the GEM model (Kremer et al. 2010) to handle nutrient movement, into and out of the 15 model boxes (Figure 3-3) which are parameterized for flow into and out of Narragansett Bay by the ROMS circulation model (e.g. Kincaid et al. 2008, Rogers 2008). Circulation data exist only for 2006, so we use nutrient data for this year to estimate flux.

We have a robust dataset of water column concentration from monthly sampling, however because this dataset was collected from the back of a moving boat, it was not possible to sample bottom water on a regular basis. We used two past datasets of surface and bottom nutrients, one from 1972-72 and one from 1979-1980 (Kremer and Nixon 1975, Oviatt 1980) to develop relationships between surface and bottom concentrations in the lower east and west passages and from this, were able to estimate bottom concentrations from our surface data (Figure 3-4). While the relationship between surface and bottom is generally complex and variable on any given day, particularly in the southern portions of the bay, where concentrations tend to be very low, there does appear to be a clear seasonal pattern which we were able to discern by combining these two datasets (Figure 3-4).

The GEM model shows that, over the course of one day, virtually no water exchanges between the sound and anywhere past the south end of Prudence Island, so it was only necessary to extend these relationships to the lower east and lower west passages, for which these two surveys have a reasonable density of data.

Having established a parameterization for the surface and bottom boxes of the GEM model for the bay/sound boundary and the lower east and west passages, we first calculated the amount of nitrogen and phosphorus which flux into the bay on an annual basis by setting the concentration in all bay boxes to zero, and initializing the model with appropriate conditions for the sound. We ran the model for one day, ‘captured’ the amount of nutrients in each box in the bay, reset the bay concentrations to zero, and advanced the model one day. We repeated the process for the entire year of 2006 (the ROMS model is parameterized with 2006 weather and forcing data). The sum of these gives us an estimate of the flux in from the sound to the bay. We then reversed the process, parameterizing bay boxes with modeled nitrogen and phosphorus concentrations, and setting the boundary condition to zero. By monitoring the net change in nutrients over the course of each day, we can calculate the amount of nutrients the model is exporting from the system on that day. This can again be summed for the year and subtracted from total import to provide an estimate of net total flux into or out of the bay.

In order to assess the variability associated with the assumptions we are forced to make with respect to this calculation (most notably the extrapolation of bottom concentration from surface) we parameterize the model using several different estimation techniques and ran it to get a range of estimates. We also ran the model with the 2006-2010 average concentrations in addition to the 2006 data to see how much interannual variability changes this estimate, with the caveat that when using 2006-2010 data rather than only the 2006 data the weather forcing no longer lines up with nutrient concentrations which accurately correspond to those conditions.

RESULTS

INPUTS

DIRECT DEPOSITION

Nixon et al. estimated 30 +/- 6 million moles of nitrogen per year deposited directly on the surface of the bay. Using Howarth's (2008) regional estimate of 1200 KgN/km^2/y and a bay area as above of 328 km^2 yields a very similar estimate of 28 million moles. This vector is therefore still responsible for roughly 5% of the annual total nitrogen budget of the bay (Table 3-1). Assuming ratios of DIN:TN are similar to those observed in 1995, approximately 80% of this is in dissolved inorganic form. Phosphorus flux measurements exist only from a 1977 dataset by Graham (Graham

1977 in Nixon et al. 1995), who measured 390 μmol/m^2/y incident on the lower West Passage. Assuming this rate is consistent across the bay yields an estimate of 0.13 million moles of phosphorus deposited in this way. While this is perhaps not the best assumption, this rate is roughly comparable to literature values (e.g. Davis and Ogden 1994, Jassby et al. 1994) which also do not show a great deal of spatial or temporal variability within the same system (Jassby et al. 1994) and the total flux of phosphorus by this method constitutes less than 1% of the phosphorus budget (Table 3-1), so the budget is highly insensitive to changes in this parameter.

RIVERS

Rivers are the single largest contributor of both nitrogen and phosphorus to the bay when sewage discharged into the rivers is considered as part of the river flow. However, due in part to improvements in plant efficiency on the rivers, and presumably in part to changes in the watershed not well measured in this study (e.g. vegetated buffer strips, reduced fertilizer use on lawns and agriculture, less phosphates in detergents, etc…), the nutrient load coming down most of the rivers has declined dramatically since the last assessment (Table 3-2). The Taunton was not measured directly in the most recent budget, so our comparison here is with data from the late 80's (Boucher 1991, Nixon et al. 1995), but shows a reduction of more than 50% in nitrogen and nearly 90% in phosphorus. A large portion of this difference is due to the fact that the previous estimates relied on a large correction factor to scale flows at the Bridgewater gauge station up for 250 square miles of watershed below this station. While we use this technique for the other rivers, we do not feel that it is appropriate for the Taunton because of vast concentration differences between the Taunton at the mouth and the Taunton at Bridgewater. Instead, we add these 250 square miles to the 'ungauged flow' term initially proposed by Reis et al. (1990) and employed in both recent budgets (Nixon et al. 1995, Nixon et al. 2008). If we were to apply the same correction factor used in past studies, we would get 82.1 million moles TN and 1.23 million moles TP, a 30 and 77% reduction respectively.

Most other rivers show modest reductions in TN load, which are typically associated with, and less than or equal to reductions that took place at plants discharging into those rivers, though the Ten Mile River shows a slight increase (Table 3-2). Similarly, phosphate reductions in the Pawtuxent and Ten Mile can be attributed to permit limits for phosphorus discharge on those rivers, while the Blackstone, which has no such limits at this time, shows an increase in P loading. The smaller rivers which do not have any plants on them (Moshassuck and Woonasquatucket), also show significant P loading reductions. Though the source of these reductions is not clear, the magnitude of flux from those rivers is very small, and thus, the change in the budget from these vectors is small in light of other changes.

WASTEWATER TREATMENT FACILITIES

Of the 29 facilities which discharge into the bay and its tributaries, 11 plants have upgraded to advanced wastewater treatment for nitrogen since the last assessment (Bucklin Point, East Providence, East Greenwich, Woonsocket, Smithfield, Cranston, Warwick, West Warwick, and Burrillville in Rhode Island, and Worcester, and North Attleboro in Massachusetts). Three of those plants (Bucklin Point, East Providence and East Greenwich) discharge directly into the bay, while the rest discharge into the tributary rivers. In that same time period, five plants (Woonsocket, Smithfield, Cranston, Warwick, and West Warwick) discharging into tributary rivers, have undergone upgrades to remove phosphorus from effluent. This removal has resulted in a reduction in the total sewage load to the bay of approximately 100 million moles of TN and 4.2 million moles of TP (Table 3-3, Figure 3-4). About three fourths of the nitrogen reduced, and all of the phosphorus reduction comes via the tributary rivers, with only about 27 million moles of TN per year in reductions at plants that discharge directly into the bay, and a slight increase in TP load at those same plants (Table 3-3). A thorough review of the plant-by-plant loading, permit levels, and upgrade status for each plant can be found in appendix B.

From a budget standpoint (Table 3-1) it is important to distinguish between the two sources (because reduction from plants discharging into tributaries is realized in the river value, not in the sewage value for the overall budget). For the purpose of presenting and discussing results, we felt it would be easier to discuss all plant discharges together. In examining the plant discharges, we wanted to determine whether the reductions we see in total load can be conclusively attributed to advanced wastewater treatment practices at the upgraded plants. As such, we looked at the change over time in total, active season, and inactive season discharge from plants which have, and have not upgraded (Figure 3-4, Figure 3-5, Figure 3-6). While there is a fair amount of interannual variability, we universally see a clear and statistically significant improvement among plants upgraded for nitrogen removal, both individually and as a group, as these plants mirror the un-upgraded plants for the first few years of the dataset before diverging as the upgraded plants come online in 2005-2009 (Figure 3-4). As more plants continue to come online, and several plants with early stages of reduction continue upgrades to meet permit limits of 5 or even 3mg/l, we can expect the total contribution from sewage to drop even further.

Contrary to expectations, there is minimal difference between 'active' (Figure 3-5B) and 'inactive' (Figure 3-5C) season loading reductions among plants which upgraded for nitrogen reduction. While these plants are only bound to their permit limits during the warmer months, they are required to operate advanced wastewater treatment to 'maximum extent' during the rest of the year, which appears to, at least on average, approach the efficiency achieved during warmer months(Figure 3-5D). With phosphorus, on the other hand, because removal is done by chemical scavenging, and is not mandated in the colder months, a clear difference can be seen in the amount of reduction achieved during active (Figure 3-6B) vs. inactive season (Figure 3-6C) despite a fair amount of noise in this signal (Figure 3-6D). While fewer plants have phosphorus limits, those limits typically specify reductions of 80-90% vs. untreated water, so the net effect is similar percentage wise. With several other plants preparing to remove phosphorus down to 0.1 or 0.2 mg/L (Liberti, pers. Comm.), the reduction in the coming years could be even more significant.

The data presented here include data from the 2010 year. In late March of 2010, Rhode Island received a massive rainstorm, which dumped more than 8" of rain on parts of the state (NOAA 2011). As a result of this storm (considered a 100-year storm), virtually all plants violated their permits for a short period of time. Due to severe flooding on the Pawtuxent river, three plants; Cranston, Warwick, and West Warwick were forced to close for several days, and discharged a large volume of minimally treated sewage into the bay until they became operational again, after which they still required up to a few months to get tertiary treatment systems back online and fully operational. Reductions at these plants were much greater during the 2008 and 2009 seasons. However, because high flow events do happen from time to time, we decided not to remove 2010 from consideration in our analysis, but rather to simply note its impact on plant discharges.

URBAN RUN-OFF

Nixon and colleagues (1995) partitioned the un-gauged flow determined by Ries et al. (1990), by calculating the portion of un-gauged acreage which falls within these qualifications (6-9m^3/s), and separating it from the roughly 25 m^3/s of unmeasured flows determined by Ries (1990). They then assigned coefficients from Carter (1982) for each of four land-use types (industrial, residential, commercial, and highway) to the acreage from each of the municipalities above. Doing so yielded a contribution of 37 million moles TN and 4 million moles TP from this vector. This component of the budget was used 'as is' in the more recent budget by Nixon and colleagues (Nixon et al. 2008).

Adhering to all of the conventions and assumptions laid out by Nixon et al., and adjusting only for changes in precipitation and changes in land-use yields 28% and 25% increases in total nitrogen and phosphorus loadings, respectively. However, since 2008, the Narragansett Bay Commission (NBC) has been collecting stormwater in the first phase of a stormwater reduction project, which directs 14 combined sewer overflows (CSO) into a large underground tunnel during storm events, for later treatment at the Fields Point treatment plant, rather than discharging it via the combined sewer overflows (CSO) directly into the bay. At present, the Fields Point has not upgraded to advanced treatment, so this process has little impact on the total nitrogen and phosphorus discharged into the bay (though secondary treatment does remove some nutrients). However, because this water is now diverted through the plant, it was counted in our direct sewage discharge, and so, should be removed from the urban run-off estimate. Based on preliminary data from NBC (Comeau, pers. comm), the tunnel treats about 4 million cubic meters of water per year, which amounts to <1% of the stormwater load to the bay by volume, but the concentration is quite high. While the monitoring program within the tunnel is preliminary, we estimated that it diverts 2.5 million moles of TN and about 250,000 moles TP per year from the CSO's into the Field's Point plant. Subtracting this amount gives a net increase in urban stormwater of 22% or 8.1 million moles of TN and an increase of 19% or 0.75 million moles of TP (Table 3-4C).

Further modification of the urban run-off figure comes from a re-analysis of the approximately 140,000 acres falling within cities and towns which discharge their run-off directly into the bay. Carter (1982) considers in her analysis only acreage which fits into the land-use categorizations she sampled. This leaves a large amount of acreage unassigned. Much of this land is open space, which has very low per acre coefficients (NRC 2008) (Table 3-4A), but some of it falls into categories such as transitional area, mixed use, transportation (railroad tracks, bus terminals, port facilities) and institutional usage (e.g. schools, courthouses, etc…) (Table 3-4B). We assigned these acreages to the coefficient most closely resembling their usage, and added coefficients from the NRC stormwater report (NRC 2008) where necessary. This results in a large increase in the loading of both nitrogen and phosphorus, caused in large part from the additional acres surveyed in this way, but also from the improvements in the accuracy of the GIS techniques used to conduct the assessment, and the improved classification provided by having additional coefficients. The change calculated from this step is approximately 38 million additional moles of TN and 2 million additional moles of TP. However, since these changes represent improved accuracy, and not a change in the actual loading to the bay, they should be considered separately from the above discussed changes, which do represent an increased loading to the bay. We therefore present the urban run-off figure as a range, with calculation from Nixon et al. modified for land-use change, precipitation, and CSO abatement as the low end, and our modified calculation as the high end (Table 3-1).

GROUNDWATER

Estimates of nutrient contribution from groundwater have not been included in past budgets. However, groundwater can be a locally important phenomenon worthy of some, if cursory, consideration. Particularly in older neighborhoods with high densities of septic tanks, of which some may be old and leaky (modern septic systems contribute much less nitrogen to groundwater), groundwater nitrate concentrations may be an order of magnitude elevated from surface water flows (Valiela and Costa 1988, Nowicki and Gold 2008).

This is locally true in Greenwich Bay, where recent efforts at sewering large portions of the population are underway, but for a long time, residents living very near to the water were reliant on septic systems and ISDS for disposal of wastewater. Urish and Gomez (Urish and Gomez 2004) estimate the groundwater flux of nitrogen into this embayment using three different sets of attenuation coefficients, and arrive at an estimate of 47-57 metric tons of N, or 3.5-4.1 million moles. This value is roughly consistent with a budget for Greenwich Bay of 10-16 million moles N, of which slightly more than ½ is sourced from the bay proper (DiMilla et al. 2011). While in the grand scheme of the budget, this constitutes less than 1% of the total flow of N into the bay, it should be noted that this value is 4-5 times larger than the flow from the East Greenwich WWTF (which has recently upgraded). Groundwater is not thought to be a significant contributor of phosphorus in most situations, due to the high capacity of soil to absorb phosphorus.

OUTPUTS

DENITRIFICATION

Sediment denitrification (the microbial conversion of DIN to N_2O and N_2 gas) is particularly difficult to quantify because it does not follow easily predictable patterns. While some systems at some times show clear relationships between denitrification rate and temperature and/or organic material loading (Jorgensen 1989, Nowicki and Oviatt 1990, Seitzinger and Giblin 1996, Cabrita and Brotas 2000, Lishman et al. 2000), recent studies have repeatedly shown no clear correlation with either in Narragansett Bay (Fulweiler et al. 2007, Fulweiler and Nixon 2009, Fulweiler et al. 2010, Fulweiler and Nixon 2011) and even the first direct measurements of denitrification in Narragansett Bay showed no impact of increased organic matter loading or temperature (Seitzinger et al. 1984).

Earlier budget estimates attributed 85-170 million moles of N loss to denitrification (Nixon et al. 1995). There is strong evidence suggesting that this number has gone down significantly, though by how much is uncertain. Data collected by Fulweiler and colleagues for 2005, when extrapolated over the entire bay, suggest an average net denitrification rate of about 40μmol/m^2/h (Fulweiler et al. 2007, Fulweiler and Nixon 2011). This number scales up to just under 80 million moles per year if extrapolated across the soft bottomed area of the bay and through the entire year. In the summer of 2006, however, Fulweiler and colleagues observed strong net nitrogen fixation, and postulated that if the rates observed in that summer were paired with denitrification at the rate observed in 2005 during the remaining 9 months of the year, the net result would be fixation of 40 million moles over the course of the year. The summer of 2006 was a year with no winter/spring diatom bloom in the bay and relatively low average chlorophyll in the mid-bay where these samples were collected (see Chapter 1). The working hypothesis of the authors was that reduction in flux of organic material to the benthos as a result of decreased chlorophyll and the lack of a large winter/spring bloom, coupled with warming water is facilitating these changes (e.g. Nixon 2009, Nixon et al. 2009, Fulweiler et al. 2010). However, we have recently had several years with strong winter/spring blooms, and average chlorophyll in our lower bay dataset (see chapter 2) shows no trend with time (due in part to high annual values in 2008, 2009, and 2010). So it is possible that the 2006 values observed by Fulweiler et al. are ‘worst case’ numbers. However, it is also possible that they are indicative of the future, since 2006 is also the first summer during which many of the upgraded plants discharged reduced effluent loads into the bay. Fulweiler and colleagues have continued this sampling program, but the data are not yet available for publication. It will be interesting to see how this term of the budget refines itself as more data become available. For the time being, we see no alternative other than to use the 2005 and 2006 measurements as a range, which yields an estimate of -20±60 million moles TN per year contributed by net sediment processes.

BURIAL

The loss of nutrients through burial in the sediments is an important term of the budget, but one which is difficult to quantify. The benthos of Narragansett Bay is very active, and much of the organic material which falls to the bottom is recycled and returned to the system. Mesocosm studies at the MERL facility have shown that nutrients in the sediment are rapidly returning to the overlying water even from heavily enriched sediments (Oviatt et al. 1984, Nowicki and Oviatt 1990). More recent measurements of sediment nutrient flux by Fulweiler et al. confirm the trend of rapid release of nutrients from the sediment, particularly the release of phosphate in low oxygen conditions, which are becoming more common in the Upper Bay regions (Melrose et al. 2007, Codiga et al. 2009, Smith 2011) where phosphate concentrations are also the highest (Fulweiler et al. 2010).

Quantification of burial requires an estimate of sedimentation rate, coupled with measurements of nitrogen and phosphorus concentration in the zone of sediments below bioturbation. Nixon et al. (1995, 1986) make this calculation based on two studies of deposition rates at different areas in the bay (Santschi et al. 1984, Corbin 1989). Because these studies measure carbon, not nitrogen, established C/N ratios in accumulating sediments (Nixon and Pilson 1984, Frithsen et al. 1985) were used to estimate N burial. This indirect method was chosen because of a paucity of direct nitrogen measurements in sequestered sediment. By this technique, Nixon and colleagues estimated a burial rate of 45-100 million moles per year for nitrogen. Data for phosphorus burial in the bay are similarly limited by lack of available sedimentation rate data (see Nixon and Pilson 1984, Nixon et al. 1995), but do align with reported values for other similar systems (Lukkari et al. 2009, Hartzell et al. 2010, Eyre et al. 2011), and thus with similar caution as originally urged by Nixon and colleagues (1995), we can adopt their estimate of 5-8 million moles per year for phosphorus burial. While Nixon et al. (1995) lament the lack of resolution in estimating these parameters, there is not sufficient supplementary data readily available to justify a reanalysis. Furthermore, given the relatively long amount of time it takes sediment to settle below the zone of bioturbation, it is unlikely that burial rates have changed in response to WWTF upgrades which are only a few years old. Over time, it is possible that decreased loading could reduce nutrient flux to the benthos, and therefore decrease burial rates, but for the time being, we can carry over the estimates from the past budget with some confidence that they are reasonably accurate.

FISHERIES

A remaining export of biomass comes from the fishery. Nixon et al. (1995) estimate nitrogen removal from the quahog fishery by calculating the meat weight of landings, and using a percentage (2.7%) of biomass N determined from literature, and 16:1 N:P ratio to estimate removal by this vector (Nixon et al. 1995). Though hard clam landings have been very variable, landings over the last few years for which we were able to get data average to about 1.85 million kilograms per year, up very slightly from the earlier estimate of 1.75, resulting in a net removal of 3.5 million moles of N and 0.22 million moles of P by this vector.

We also estimate removal from growth of fish biomass which can be directly ascribed to growth in the bay. To do this, we account only for the growth in biomass of age 0 fish which come into the bay in the spring, and grow over the course of the summer. Using biomass data summarized from the DEM monthly fish trawl in the bay by Longval (2009), we estimate that biomass growth from this vector is approximately $0.95gC/m^2/y$ wet biomass. Since this is a baywide average of sampling stations roughly evenly distributed throughout the bay, we can scale it up over the area of the bay to get approximately 311 metric tons of fish biomass per year supported in this way. Furthermore, because we are using biomass rather than abundance, this estimate accounts for loss due to mortality and assimilation efficiency. To convert this to nitrogen, we used a biomass:carbon ratio of 3:1 and Redfield C:N:P, which works out to 2.3% N by weight. This value is similar to the value found by Nixon et al. (1995) for Quahog and is also roughly comparable to values found in a similar study on fish nutrient export in coastal Louisiana (Deegan 1993). By this calculation, 7.5 million moles of N and 0.45 million moles of P are exported by this vector. While this is a conservative estimate, at the very least, it makes some attempt at quantifying the role of secondary production on nutrients. Combined with the hard clam data, this sums to 11 million moles of TN and 0.67 million moles of TP.

EXPORT

Flux across the bay/sound interface has historically been extremely difficult to quantify. Past budgets have, at least in some part, calculated this term by difference, assuming that the system is, on average, at steady state. Nixon et al. (1995) estimated inflow of nitrogen and phosphorus from the sound to be 115 and 27 million moles of nitrogen and phosphorus, respectively, by calculating inflow volume using a salt balance model (Pilson 1985b) and concentration using the bottom water concentration in the lower East Passage from a yearlong survey in the early 70's (Kremer and Nixon 1975). Outflow of organic nutrients was calculated by estimating export of carbon from primary production (whose creation and burial are easier to quantify) and using the Redfield ratio to estimate N and P loss at 90-185 million moles of N and 7-14 million moles of P per year, plus an additional 72 and 2.4 million moles N and P, respectively, from riverine DOM. The budget is then 'balanced' by difference, assuming inorganic export of the remainder of the inputs of nitrogen and phosphorus to the bay, or 240-470 million moles N and 41-51 million moles TP (Nixon et. al 1995, Table 21).

We attempt herein to use a modeling approach to more accurately quantify these fluxes. The GEM Box model (Kremer et al. 2010) was designed as an eco-physical model to simulate property exchange in Narragansett Bay in order to look at the drivers of hypoxia in the bay. However, it uses the highly accurate ROMS model for property exchange and flow between a series of model boxes which correspond well to the stations sampled in this study (Figure 3-3). By parameterizing the GEM model with the river and plant loadings above, and parameterizing the bay/sound boundary using data from station 3 in Chapter 1, we can generate estimates of exchange between elements, and therefore, produce an estimate of flux into and out of the bay.

The modeled nitrogen concentration in the 15 bay boxes (Figure 3-3) does an excellent job of paralleling measured concentrations from our baywide survey. To test this, we calculated Relative Operating Characteristic (ROC) scores comparing measured and modeled nitrogen and phosphorus at each station over the entire year at 19 evenly divided thresholds between the minimum and maximum value observed in each GEM box. The summed ROC scores for the entire model (all boxes, across the entire year) are .92 for nitrogen and .96 for phosphorus, where 1.0 is a perfect match, and a score above 0.5 is indicative of a skilled model (Figure 3-7).

Doing so yields inflow estimates of 251 million moles DIN and 75 million moles DIP per year, both significantly higher than estimates of 115 and 27 million moles N and P respectively put forth by Nixon et al. (1995) in previous budgets. However, similar to past budgets, the model predicts net fluxes out of the bay for both N and P, calculating net export of 102 and 283 million moles inorganic and total nitrogen, and 29.8 and 32 million moles inorganic and total phosphorus respectively. This calculation indicates that significantly less nitrogen and more phosphorus are fluxed out of the bay in inorganic form than estimated by Nixon et al. (1995) but upholds the conclusions of that study that the vast majority of both N and P incident on the bay are exported to the sound in one form or another, and that most of the P export is inorganic. Using these estimates to close the budget, we are very close to balancing the nitrogen budget for the bay, with inputs and outputs overlapping to within the significant margin of error necessary with this type of calculation. However, our estimate of net phosphorus export makes the bay slightly net negative for phosphorus, as total load to the system as quantified above, is only about 25 million moles.

STORAGE TERMS

A thorough review of the standing stock of nutrients stored in the water column of the bay can be found in Chapter 1. Relative to the magnitude of other terms in the budget, the water column standing stocks are small; 15 and 45 million moles of DIN and TN respectively, and 2.5 and 3 million moles of DIP and TP respectively. With standing stock and input, we can make a cursory estimate at residence time, arriving at a residence time of 15.5, 33, 67, and 42 days for DIN, TN, DIP, and TP respectively. Compared to a residence time of about 30 days for water in the bay (Pilson 1985a), DIN appears to be rapidly assimilated, while phosphorus (both inorganic and organic) may be being retained in the bay for longer than the average residence time of water, possibly either through recycling, or sediment flux and resuspension.

The storage terms in the sediment were much larger than in the water column. Our ability to estimate this from existing data is limited, as we have only very limited data on nutrient concentration in the surficial sediments from mesocosm experiments. However, when we scale these concentrations up to account for the top 10 centimeters across the bay, we estimate approximately 1770 million moles TN are and 377 million moles TP are stored in this reservoir. There is a tremendous amount of uncertainty associated with these terms, as we do not know if concentration is constant throughout the bioavailable sediments, and we have only 2 data points, both of which come from mesocosm and not in situ data. However, the resolution here is sufficient to arrive at the conclusion that the short term storage in sediments far exceeds the annual inflow for both N and P and thus, the reservoir is more than large enough to potentially buffer short term changes in supply, or even mask a management intervention for a short period of time.

DISCUSSION

INPUTS

DEPOSITION

Though only nutrients which fall directly onto the bay's surface are quantified here as a budget term, and this term is a relatively small contributor to the overall budget of the bay, less than 10% of the nitrogen budget and less than 1% of the phosphorus budget, atmospheric deposition onto the watershed is an important part of the budget. Its predominant manifestation is in the rivers term, and we can estimate its magnitude by subtracting the plant discharges from the total river flow numbers. Of the 221 million moles ascribed to river flow, a maximum of 118 million moles can be ascribed to plant discharges, and this assumes no loss term for utilization or burial ascribed to the stream flow. Similarly for phosphorus, of the 9.35 million moles which enters the bay through the rivers, only half could possibly come from the plants, and phosphorus in freshwater systems is typically taken up very quickly. The remainder in both cases is caused by processes in the watershed, either deposition on the watershed and subsequent run-off or other anthropogenic processes (e.g. fertilizer, septic systems, run-off from roads, etc…)

In reality, the fraction of this river flow due to watershed processes is likely even higher. We can make a first pass approximation of this in rivers which experienced upgrades, such as the Blackstone. Between measurements made by Nixon and colleagues in 2003-2004 and those made by NBC in 2006-2010, approximately 50 million moles/year of nitrogen were removed from effluent discharged from this river (Table 3-3), yet we see only a 15 million mole reduction (approximately) in flux to the bay (Table 3-2). The difference is somewhat mitigated by the observed 10% increase in flow, to which we can, estimating by percentages, attribute an additional 7.5 million moles of loading. Even still, our results indicate that at most half of impact of the reductions implemented is felt by the bay proper. This is an indication that the 50% riverine abatement estimate used by the DEM in assessing the impact of reductions may be close to accurate for the Blackstone (RIDEM 2005). In comparison, the Pawtuxent River, the only other river which had substantial upgrades to its plants, shows reductions in nitrogen and phosphorus which almost exactly match the plant reductions of about 20 million moles per year nitrogen and 2 million moles per year phosphorus, a potential indication of very little abatement. This is not surprising, since the travel time on the Blackstone from Worcester to Narragansett Bay allows much more time for biological, physical and chemical processes than the short run down the Pawtuxent from Cranston and Warwick to the bay.

RIVERS

As mentioned earlier, the rivers are the primary vector of nutrients into the bay, despite relatively low total freshwater input compared to other similar systems

(Bricker et al. 2007). Most of our river calculations seem in line with what we expect to see given the plant reductions, tempered slightly by increases in precipitation which have, on average, increased flow by around 10%. There are two discrepancies in river estimates between our work and past budgets which warrant attention. The first is the order of magnitude reduction in phosphorus load in the smaller rivers. While we expect reduction in phosphorus load in the Pawtuxent and Ten Mile loadings (and soon the Blackstone) due to plant upgrades on those waterways, no such upgrade occurs on the Moshassuck or Woonasquatucket rivers, and flow between the surveys seems relatively consistent. It is difficult to establish a firm causal mechanism here, as we are not aware of any management action to reduce loadings in these stream reaches. However, the contributions of these rivers to the overall budget are very small, so the resolution of our data may be limited. Despite order of magnitude phosphorus reductions in both of these rivers, this change accounts for only 1-2% of the phosphorus budget.

The other, and far more significant difference is the Taunton River. The discrepancy in measurement comes in part from the fact that Nixon et al. (1995, 2008) scaled up the flow of the Taunton to account for the large un-gauged area between the measurement station, at State Farm in Bridgewater MA, and the mouth of the river. By land area, slightly more than half of the watershed is un-gauged because the river has tidal influence for about 10 miles from its mouth. This results in increasing the flow from the Bridgewater gauge by about 40%, as calculated by (Boucher 1991). We elected not to scale this flow up primarily because the Taunton River at Bridgewater, where it was sampled both for flow and for concentration, during low flow periods is more than half sewage effluent by volume. Even during high flow periods, the effluent from the Brockton WWTF, at a relatively constant 17-20 million gallons per day, is close to 10% of the total flow of the river. Therefore, we feel it may not be accurate to apply concentration data taken at the Bridgewater gauge, and assume that it will hold constant as the volume essentially doubles with 300 square miles of un-gauged area below this station. This is much less of a concern for other rivers, where the volume of effluent is small compared to the volume of water, and the ratio of gauged to un-gauged area is small (for most of the other rivers, the ratio of gauged to total area is <1.2).

When we calculate the Taunton River using Boucher's (1991) coefficient, we get 82 million moles TN and about 1.22 million moles TP. This TN estimate is still a 30% reduction over Nixon et al. and the phosphorus reduction is still about 77% of the earlier estimate. These numbers are probably a more accurate representation of the change which has gone on over time in that system. We expect the large phosphorus reduction, since Nixon et al.'s values are from data collected in the 1980's, before large scale reductions in phosphorus load became mainstream (Litke 1999). However, for the purpose of attempting to quantify as accurately as possible the total flows into and out of the system, we believe that adding the un-gauged portion of the Taunton River to our 'unmeasured drainage' term, and representing it with the average load per acre across the entire system provides a more accurate picture of the actual contribution from the Taunton, though we admit there is a fair amount of uncertainty either way on this matter.

WASTEWATER TREATMENT PLANTS

The flows from the treatment plants are perhaps the easiest to quantify, and the most data-rich portion of the assessment. There were very few ‘surprises’ in this analysis either. Most plants with upgrade permits in force met or exceeded their targets virtually without fail. Many plants did almost as well during the winter as they did during the summer. This was a bit of a surprise, because we expected a temperature-dependent relationship here (e.g. Lishman et al. 2000, Jeong et al. 2006), but we hypothesize that since the tanks are generally underground, and receiving water also from underground and/or partially indoor facilities, the water in these tanks may be fairly well insulated, and remain warmer than expected despite cold air temperatures, which would improve efficiency.

We elected to use 2010 data from Warwick, and West Warwick, even though those plants were physically flooded for a prolonged period of time, and not fully back online for several months after the large flood in late March. The 2010 average numbers from these two plants are 50% higher for total nitrogen, and nearly double for total phosphorus compared to 2008 and 2009. Over the long-term, we expect most years to be more like 2008 and 2009, and hopefully these plants will implement procedures which will assist them in recovering quickly from flood events when they do occur, minimizing excess flux. However, if the past 50 years have been any indication, the climate of Narragansett Bay is shifting towards increased precipitation and increased storms (Madsen and Figdor 2007, Pilson 2008, Smith et al. 2010), so removal of these data as a ‘fluke’ seems shortsighted.

URBAN RUN OFF

The impact of industrialization and build up in nearshore communities is felt in the nutrient mass-balance through the urban run-off term. Precipitation that falls on these areas, with high amounts of impervious surface, is collected in sewers and discharged directly into the bay, rather than moving gradually through the water table, where much of the nutrient load may be alleviated. This is true of nearshore communities throughout the Providence area, as well as Fall River, Newport, East Greenwich and North Kingstown.

This term of the budget also is more complex than the analysis suggests. Much of the trick with urban run-off involves effectively partitioning the run off so that it is not double counted as part of either a WWTF discharge or in the ‘unmeasured flow’ term associated with the rivers. Nixon et al. (1995) thoroughly review the assumptions that go into the parceling of space so as to avoid, or at least minimize double counting here, with the only major change that has occurred on this front being the institution of the CSO catchment tunnel, which actually diverts a significant portion of what was formerly part of this term of the budget into the ‘direct plant discharge’ term. Even so, the combination of land-use change and increased precipitation causes this term to rise.

The reassessment of acreage not originally assessed by Carter (1982), and therefore by Nixon et al. (1995, 2008) is difficult, because the decision to use nationwide coefficients specific to a land-use type rather than a more generic coefficient that is more specific to the watershed is a difficult trade off, and can be argued either way. For the most part, there is reasonable agreement in coefficients between the two studies, but there are some exceptions, notably the phosphorus coefficient associated with highways, which in the case of the earlier study, is based only 3 data points (Table 3-4B). The other issue arises from the consideration of land use types not measured by Carter. We felt it necessary to provide some value for these previously unconsidered acres, and when we did a sensitivity analysis by varying how we assign coefficients (Dionne et al. 2009), we found the overall estimate to be relatively insensitive to how we handled this issue. However again here, a large part of the difference between our assessment and the past assessment is due to a change in methodology, so the increase in actual load from this source since it was last assessed in the 80's is probably about 20%.

One interesting thing to point out is that road miles are a large driver of load from this vector, since the runoff coefficients from this land use type are so high. One thing that neither our assessment nor Nixon et al.'s work takes into consideration is the recent trend towards the creation of vegetated buffer strips and retention wetlands. In virtually all systems studied with low to moderate loading rates (Narragansett Bay would be considered moderate, as compared to dense agriculture or concentrated feed lot operations, which would be considered high), this technique reduced N and P load by more than 90%, in some cases as high as 99% (Haycock 1993, Lee et al. 1998, Greenway et al. 2001). This is particularly true during the growing season, but even in winter, buffer strips with trees are >95% efficient for nitrogen removal, while grass is 84% efficient for nitrogen removal and up to 50% efficient in phosphorus removal, even during simulated heavy rain events, presumably due in large part to subsoil microbes as well as above ground biomass (Haycock 1993, Lee et al. 1998). These advances in urban planning could explain some of the reductions in the smaller rivers, and could mitigate future land-use changes as the upper bay SAMP continues to prioritize vegetated buffers (SAMP 2005).

Retention wetlands also have great potential for mitigation. Greenway and colleagues (2001) show that >25 % of TP and 80-85% of TN can be removed by constructed wetlands, while Lin et al. (2002) show similar patterns with even higher rates of P removal. Small ‘wetlands’ are an ancillary result of digging borrow pits to create overpasses, but if these wetlands are managed (with appropriate drainage and above-ground biomass removal) they can sequester as much as 1.5 tons of carbon per hectare per year (McCarty and Ritchie 2002). A meta analysis of nitrogen uptake rates in retention wetlands (Crumpton et al. 2008) shows wide range of nitrate consumption rates, from 200-1200kgN/ha/y with a mean of 400 though some of this is likely due to denitrification.

While these numbers sound enticing, it seems logical to ask whether these retention wetlands can offset the additional nutrients incident on a system from a construction project. To accomplish this, we analyzed a recent construction project in North Kingstown RI, where several small retention wetlands which, were periodically mowed, were created adjacent to new overpasses associated with an expansion of route 403 coming from the Quonset Point port facility (Figure 3-8). We used Google Earth to measure the amount of newly constructed highway (4 additional lanes in some places and expansion from 2 lanes to 4 lanes in others), and to map and quantify the areas of wetlands created (Figure 3-8). If we apply our highway urban run-off coefficient, our annual rainfall of about 127cm/yr puts about 500-700 moles N and 40-

50 moles TP on an acre of highway, while Crumpton et al. (2008) measured that an acre of wetland could remove (average) 11,500 moles TN. The N:P ratio in live marsh grass is close to 16:1(Dame 1991), but literature suggests that a significant amount of the N removal from marshes is the result of denitrification, not above ground biomass growth (Valiela and Teal 1977, Dame 1991, Mitsch and Gosselink 1993). In the case of natural salt marshes, probably most of the vegetative uptake is recycled (since little of this organic matter is exported), but in constructed wetlands, the plant material is mowed and composted. Lacking estimates of the amount of biomass removed by mowing, if we estimate denitrification to be responsible for 50% of the N loss, and an N:P ratio of 16, 350 moles TP would be removed in this way. Our Google earth calculation suggests that the Route 403 expansion in North Kingstown adds 27,000 m^2 of created wetland or about 6.7 acres across several small ponds, and about 1.3 million square feet of roads assuming 12 foot lanes, which is about 30 acres. This gives us a total increased load from construction of 18,000 moles N and 1350 moles P, while the wetland could remove 77,000 moles N and 2400 moles P; in this case several times more than the road adds. A similar study on retention ponds in Saskatchewan showed that a pond of roughly 9,000 square meters (2.2 acres) removed approximately 18,000 moles of nitrate per year (Wang et al. 2008), which is similar in magnitude to the estimate presented here.

In fact, the role of wetlands in general as a nutrient sink may be a mitigation pathway deserving more attention. Heffner and Nixon are presently calculating rates of nitrogen removal from salt marshes exposed to varying levels of anthropogenic nitrogen loading, but these data are not ready for publication, and the total acreage of salt marsh in the bay proper is small, slightly less than 1500 hectares (Wigand 2008) perhaps leading to why this term has not been included in past budgets of Narragansett Bay, despite inclusion in budgets of many other systems (Woodwell et al. 1977, Woodwell et al. 1979, Boynton et al. 2008, Eyre et al. 2011). If we were to make a first-pass approximation at the amount of nitrogen which could be removed in this way, we might use the low end figure from Crumpton et al. (2008) of 200kgN/ha/y, which would give us a removal term of about 20 million moles per year, a small but significant contributor to the budget. Thus, the restoration of natural wetlands may be a management strategy worth considering moving forward.

Another factor which has received a great deal of attention is the new CSO collection tunnel. At present, all this tunnel does from a nutrient budgeting perspective is move about 2.5 million moles per year of TN and about 0.25 million moles of TP from the urban run-off term to the direct sewage term because the Fields Point plant has not instituted advanced treatment yet. In practice, the impact on the ecosystem may be more pronounced, because it will delay these nutrients (and the freshwater in which they are suspended) from entering the bay during a time of already high freshwater flux, and therefore may slightly reduce the extent or severity of hypoxia which typically follows large rainfall events.

In theory, once the Fields Point plant upgrades, this will result in a net removal of about 2 million moles of nitrogen per year. Further upgrades to the CSO system will capture 15 more overflows within the next 3 years, and create a second tunnel with feeds the Bucklin Point plant, intercepting another 17 CSO's by 2021. All told, the system could capture and treat as much as half of the stormwater nitrogen incident on the Providence metro area, resulting in a reduction of 10 million moles TN or more. It should be pointed out that the primary goal of the retention project is not nutrient removal, but rather reducing beach and fishing closures, by reducing the discharge of untreated wastewater and its bacterial load, so analyzing this reduction on a ‘cost per mole’ basis does not capture the full benefit of the tunnel.

GROUNDWATER

Compared to many other systems, particularly those with large agricultural inputs, the groundwater contribution to Narragansett Bay is very small (Boynton et al. 2008, Kincaid et al. 2008, Nowicki and Gold 2008). Based on salinity budget measurements, Pilson estimated this avenue to account for less than 10% of the total freshwater to the bay (Pilson 1985a). This falls roughly in line with estimates made by Kincaid and colleagues using the ROMS model (Kincaid et al. 2008). Furthermore, some of this may be captured in the ‘unmeasured flows’ estimated by Ries (1990), which are included in above and past mass-balance calculations. By subtracting the sewage estimate from the river load and adding the urban run-off number to the remainder, we can roughly quantify the amount of nutrients which freshwater flows bring into the bay at about 150 million moles TN, 10% of which is about 15 million moles. Our only quantifiable source of groundwater comes from Greenwich Bay, which we estimate at 4 million moles TN. Some of the remaining groundwater is likely counted by the ‘unmeasured flows’ term (which we used to scale up observed flows to match predicted flows). In general, while we may be underestimating this term somewhat, the magnitude of the discrepancy is not a major concern within the scope of the budget writ large.

OUTPUTS

SEDIMENT FLUX

Narragansett Bay is extremely fortunate to have a rich history of denitrification studies measuring flux of nitrogen to and from the sediment (Seitzinger et al. 1984, Nowicki and Oviatt 1990, Nowicki 1994b, Fulweiler et al. 2007). This biological process has traditionally been viewed as an important removal mechanism by which 13-26% of the annual input is removed from the bay (by conversion to N_2 gas) (Nowicki 1994b, Nixon et al. 1995, Fulweiler et al. 2007). However, recent changes to the bay, brought presumably by changes in climate and phenology (e.g. Nixon 2009, Nixon et al. 2009, Fulweiler et al. 2010) and possibly in part by decreased loading have altered the net denitrification rates in the bay.

This term of the budget also has perhaps the most uncertainty associated among any of the terms we can directly measure. Measurements of denitrification in the bay, both past and present appear to be patchy, variable, and not well correlated to other physical processes in the bay (e.g. organic material loading or temperature) (Seitzinger et al. 1984, Nowicki 1994b, Fulweiler et al. 2007, Fulweiler et al. 2010, Fulweiler et al. 2011). Even as the amount of data on this topic has increased rapidly over the last few years, it has served mostly to help us realize how much more we need to do in order to truly understand the benthic-pelagic coupling in this ecosystem.

With that being said, there are certainly enough data available to make a reasonable estimate at the contribution of this term. However, it also seems likely that the denitrification rate in the bay is not constant, and is likely to vary greatly from season to season, based on the amount of organic matter fluxed to the benthos in any given year (Nowicki and Oviatt 1990, Fulweiler and Nixon 2009, 2011) and the availability of oxygen during summer months, which is also variable (Melrose et al. 2007, Codiga et al. 2009). The two years sampled by Fulweiler et al. (2007) and presented in this study, 2005 and 2006 both represent years with no winter-spring bloom, and 2006 was one of the most severe years on record for hypoxia in terms of spatial extent, severity, and duration (Codiga et al. 2009, Smith 2011). In contrast, very large blooms occurred in 2009 and 2010, with blooms smaller but still present in 2008. This may cause the estimate of denitrification to more closely resemble earlier measurements by Seitzinger, Nowicki, and colleagues (Seitzinger et al. 1984, Nowicki 1994b). However, the opposite can also be argued; that because 2006 was the first year after loading reductions, we ought to expect that conditions in this year would be the norm moving forward. We therefore provide a large range (-20±60 million moles) for the estimate of denitrification, but our range does not overlap with the estimate of Nixon and colleagues of 85-170 million moles denitrification (Table 3-1). If net sediment N flux truly varies from the maximum of the range calculated in this study (40 million moles net nitrogen fixation) to the minimum of the range calculated by Nixon and colleagues (170 million moles net denitrification), it would be the single largest term in the nutrient budget. Even our estimate of the interannual variability in this term (-20± 60 million moles), which may well be too small, makes this the third largest term in the budget, and something we should keep our eye on closely as we move into the future.

It should also be noted that the change in estimates of denitrification between Nixon's estimate; 130±45 million moles and the present estimate; 20±60 million moles, is almost exactly the same magnitude as the reduction in loading of nitrogen to the bay associated with upgrades to wastewater treatment facilities (Table 3-1). It is difficult to establish a positive causal link between these two factors, and it is entirely possible that this similarity is a coincidence, but it is also possible that changes in the flow of nitrogen into and out of the sediments may ‘counteract’ a significant portion of continued reduction efforts, whether causally or driven by another (e.g. climate) factor.

Especially given the present and pending phosphorus reductions, this could have interesting implications for the N:P ratios in the bay. While presently, the bay remains nitrogen limited on average, the ratio of N:P approaches or exceeds 16 particularly during winter months in the upper portions of the bay, though both species are typically abundant during this time of year (see chapter 1). If total N loads to the system remain constant (e.g. reductions to the load are balanced by changes in sediment flux) while P load continues to drop, this may tip the scales even further towards phosphorus limitation.

Another interesting corollary of this research is that while we expected to find a proportionately larger impact on standing stocks during the summer as a result of WWTF upgrades, we found similar magnitudes of decrease when comparing summer and annual totals (see Chapter 1). If indeed the benthos is contributing a significant amount of nitrogen to the water column by fixation (or even denitrifying less) during the summer, this could explain why the decreased loads during the summer are not evident in the standing stocks.

The lack of phosphorus balance in the bay is a key point of discussion, and the sediments are the lynchpin which might enable this phenomenon. The sinks of phosphorus we document in this study are actually about 25% less than found by Nixon et al. (1995). However, our total sources of phosphorus over the same time period have dropped by more than half. If indeed sinks of phosphorus exceed sources at this time by the 15 million moles estimated (Table 3-1), these losses are likely coming from the large reservoir of phosphorus stored in the sediments.

Flux from the sediments of phosphorus is traditionally considered to be net zero, whereby flux from water column to sediment is balanced by burial and remineralization. Mesocosm studies using Narragansett Bay sediments confirm this trend, showing sediment and water column reaching relative equilibrium within 6 months of a disturbance (Oviatt et al. 1984, Nowicki and Oviatt 1990, Nowicki 1994b). However, it is worth considering that the year of data used for the bay/sound modeling flux is the year during which most of the plants which upgraded completed their upgrades. Thus, any short term imbalance resulting from this reduction in supply would be reflected in our results. Furthermore, 2006 was a particularly severe year for hypoxia in the bay, and hypoxic conditions are well known to flux phosphorus from sediment to water column (Nowicki and Oviatt 1990, Fulweiler et al. 2010), which could cause additional short term flux out of the bay from the sediment storage term.

In addition to their nitrogen measurements, Fulweiler and colleagues also measured net sediment phosphate flux at 3 stations in the Providence River estuary, Greenwich Bay, and the Upper Bay (Fulweiler et al. 2010). This relationship does show weak temperature dependence, with the strongest fluxes out of the sediment at warm (>20^{o}C) water temperatures, and flux dropping to essentially zero in cold (<10^{o}C) water. Fitting the regression applied by Fulweiler et al. to average annual water temperature data for the bay, we can calculate that the sediments of the Providence River estuary might supply some 4.2 million moles of inorganic phosphorus to the water column over the course of 2006, with net flux from Greenwich Bay of only a few thousand moles, and the Upper Bay station close to zero (Fulweiler et al. 2010). The authors do not measure flux from the lower bay, but their measured flux from the Providence River Estuary alone would account for more than half of the 'missing' phosphorus in our budget (Table 3-1). This may be a key area for future study, because if the ecosystem truly is 'balancing the budget' by exporting several million moles of phosphorus from sediment storage per year, there could be further changes in productivity and N:P ratio once the system reaches equilibrium, especially given additional future loading reductions.

FISHERIES

Our estimates to quantify fisheries removal herein are preliminary and certainly conservative. There are a number of literature attempts to quantify the impact of fish and fish biomass on nutrient dynamics (Vanni et al. 1997, Vanni 2002, Sereda et al. 2008), mostly for freshwater systems. However Deegan (1993) attempted to quantify the role of fish biomass export on an estuarine nutrient budget in Louisiana, arriving at an estimate of about 3.1 grams N/m^2 and about 0.9 grams P/m^2, which constituted about 5-10% of the total nutrient budget of that system. Our estimate of fish export is between 2-3% of the total inputs to the system. If we were to assume the same rate of export as Deegan found, Narragansett Bay would export about 70 million moles of TN and 9.5 million moles of TP per year as fish biomass (Deegan 1993). This export would represent about 22 grams carbon/m^2 according to Deegan's calculations, which is less than 10% of the total primary productivity in the bay of 323 gC/m^2/y as estimated by Oviatt and colleagues (Oviatt et al. 2002).

The key difficulty in estimating fish export from this system is that much of the fish biomass is not year round resident. Virtually all of the biomass of fish leaves the bay in the winter, migrating offshore and/or south. Species like bluefish and striped bass, which constitute the majority of the recreational catch, are highly mobile, and even more site associated demersal fish like tautog, black seabass, and scup tend to move into deeper water during the winter. The other major fishery in the bay which may be easily quantifiable is for lobster, which is responsible for about 1,700 metric tons of landings (RIDFW 2008), but these animals also are mobile, and a portion of their diet is thought to come from lobster pots, which are typically baited with skate, herring, or other fish whose source is unknown (Saila et al. 2002).

BURIAL

Especially as the inputs to the system continue to change over the next years and decades, it may be wise to systematically reevaluate whether the rate of nutrient removal from the system by burial is changing. Based on recent estimates of sedimentation rate in the bay which range from 0.5-2cm/y (Hartmann et al. 2005), it would take a minimum of 5-10 years for sediment to settle out of the top 10-20 cm, which is typically the zone considered to be most biologically active (Calabretta and Oviatt 2008, Shumchenia and King 2008). Since the majority of upgrades did not occur until 2005, it may still be too early to detect any change which is occurring in all but the most sensitive locations in the bay.

FLUSHING

Estimates of exchange across the bay/sound interface are limited not by the modeling capacity, but by the relative paucity of data used to inform the process. While we are quite confident in the ability of the GEM model to provide reasonable estimates of water exchange between the bay and the sound and circulation within the bay (Figure 3-8), our nutrient data are on a much coarser scale than the model truly needs.

We have only one station representing RI sound, and two stations representing the lower east and west passages respectively. Each station was sampled 12 times during the year 2006, and the samples are surface only. From this, we must create a matrix of daily surface and bottom nutrient concentration estimates to parameterize the flux into the model, which, by the nature of their being estimates, do not really ‘line up’ with any particular weather events associated with the circulation parameterization of the model (e.g. if the modeled weather data dictate a wind shift from the North to the South on a given day, this will intensify the flow up the East Passage of the bay, bringing in more nutrients from the Sound, but if we did not sample that day, we may be using inappropriately interpolated concentrations to parameterize those fluxes.

Furthermore, we are estimating bottom concentrations from averaged relationships derived between surface and bottom concentration from two surveys several decades ago, and these relationships are very variable (Table 3-5). We must assume that the relative relationships between surface and bottom concentrations have not changed with time, which is probably a reasonable assumption, but one on which we do not have enough data to conclusively comment either way. However, even after this assumption is considered, there are still problems associated with this technique. While the actual relationship between surface and bottom concentration is likely to be correlated to weather, wind, river flow, tide and other factors which are considered by the model, the concentration relationships we are using would be blind to these variations. This could bias the model one way or the other.

The model appeared to be relatively insensitive to changes in the estimation technique used to extrapolate bottom water concentrations from surface, with the net flux across the boundary changing by a maximum of 4% for nitrogen and 2% for phosphorus across the three estimation techniques we attempted. Interannual variability was a larger concern, causing a change of about 10% between the 2006 data and the 2006-2010 average concentration values. Even this is likely an underestimate of interannual variability, since this controls only the sound concentration, and weather and circulation as well as load from rivers and plants is still driven by the model, which, in this case remains parameterized with 2006 data. As discussed earlier, 2006 was a year with high precipitation (137 cm as opposed to a 10 year average of about 119), and high spatial hypoxia extent (Codiga et al. 2009). Therefore the forcings associated with this year may overestimate flux from the bay to the sound relative to a more ‘average’ year.

Because of the way the model runs, we were also forced to either treat TN as a conservative tracer, and not allow it’s uptake at all by biology, or treat it identically to DIN, and allow it to be immediately taken up by the biology in the model. We also do not have appropriate particulate nitrogen (PN) data to parameterize the modeled river flows, and therefore had to use a ratio of TDN:TN derived from previous work (Dionne et al. 2009). Though PN is a relatively small contributor, these assumptions are the cause of the larger uncertainty in the TN flux.

Our fluxes for DIN calculated by the GEM model seem entirely reasonable. Exchange across the bay/sound boundary is much larger than estimated by Nixon et al. (1995) (Figure 3-9), likely because the model considers nutrients flowing in and out with the tidal cycles, while Nixon et al. measure only net transport in and out. However net flux out of the system is slightly less than calculated by Nixon, totaling just over 100 million moles. This could be an artifact of the totally different methodology, or it could be a reflection of reductions in loading. We do see changes in the way DIN constituents behave on a downbay gradient after the reduction (see Chapters 1,2), particularly ammonium, so it would not be unreasonable to attribute some or all of this reduction to actual decreases in the concentration of water leaving the bay (and/or increases in the concentration entering from increased regional atmospheric deposition).

On the other hand, phosphorus fluxes across the bay/sound interface of about 30 million moles, 90% of which is in inorganic form, are dramatically different than past estimates of 50-70 million moles export across this boundary (Figure 3-9), though Nixon's (1995) results also suggest that the vast majority of the export (about 80% in that study) is inorganic. The GEM model ascribes a much larger portion of the total phosphorus budget of the bay to import from offshore, and consequently, predicts much higher export, though net export is actually lower than calculated by Nixon et al.

(1995) (Figure 3-9). The model estimates that the flux of phosphorus out of the system is approximately 25% greater than the combined fluxes of phosphorus into the system from all sources. While we have few other quantifiable fluxes out of the system, and thus, expect a great deal of the phosphorus incident on the system to export to the sound one way or another (organically or inorganically), we would ideally have more data to try to determine whether these measurements are accurate, because it is important to understand whether we have captured a short term imbalance in the budget, whether there is continued consistent loss from the sediment storage reservoir into the water column, or whether we are missing another source of phosphorus to the bay, especially given recent management efforts to control phosphorus loading to the bay.

Here again, a conceivable mechanism for the imbalance might be sediment regeneration. If for many years, phosphorus inputs have greatly exceeded readily quantifiable outputs (as postulated in past budgets), it seems logical that a large storage term of phosphorus would exist in the sediments of the bay, which could conceivably take a while to flux out in response to reduced loadings. This has been shown true in many other estuarine systems (e.g. Carstensen et al. 2006, Artioli et al. 2008, Boynton et al. 2008, Lukkari et al. 2009), but most of those systems have lower salinity than observed in Narragansett Bay, and past mesocosm experiments in this system (e.g. Oviatt et al. 1984, Kelly et al. 1985, Nowicki and Oviatt 1990) have shown rapid response of the system to changes in loading, such that it would seem unlikely that phosphorus deposited in the the 1980's and earlier when loading was much higher would still be remineralizing and contributing to flux out of the system at this point. It is, however, possible that our measurements captured a short term event, and in reality, fluxes from the bay into the sound are somewhat lower.

COMPARISONS WITH OTHER SYSTEMS

Narragansett Bay appears to respond similarly to nutrient loading reduction as other similar systems for which budgets have been compiled (e.g. Artioli et al. 2008, Boynton et al. 2008, Duarte et al. 2009). It is difficult to tell at this early and intermediate stage in the reduction process what the ultimate impact on the system will be. Many systems with smaller reductions in load have shown no or minimal biological response (Carstensen et al. 2006, Artioli et al. 2008, Duarte et al. 2009) to the reduction. At this point, Narragansett Bay shows no measurable decline in chlorophyll (see chapter 1, 2) or primary productivity (Smith 2011) as a result of the load reductions. In contrast, systems with dramatic loading reductions almost always show biological response (Greening and Janicki 2006, Taylor et al. 2011), so it is possible that as loading reductions approach the 50% threshold predicted by RIDEM (RIDEM 2005) we will begin to see reduction of chlorophyll and primary productivity (Oviatt 2008).

Narragansett Bay falls in the middle of many similar systems in terms of nitrogen and phosphorus loadings (Figure 3-10) in terms of load per acre. While our study shows the system to be nitrogen limited on the large scale, and this result is consistent with past studies of Narragansett Bay (e.g. Nixon et al. 1995), the present N:P ratio of loading to the bay is 19:1. Proposed reductions will bring the system more in line with a 16:1 input ratio of N:P. Continued management efforts to reduce phosphorus from several additional plants stand to remove 2-3 million additional moles of phosphorus from the bay, at which point, the phosphorus load to the bay will be only about 110% of prehistoric levels, while nitrogen load will still be in the vicinity of 4 times prehistoric estimates (Nixon 1997).

CONCLUSION

In compiling the budget of clearly defined inflows to the bay, marked reductions in the contribution of sewage to the total nitrogen and phosphorus budget of the bay occurred (Figure 3-4, 3-5). Of the 11 plants in the bay which have upgraded their systems, virtually all plants are meeting or falling below permitted concentrations throughout the year, with only a few very short violations (see appendix B). Many plants are exceeding expected reduction levels during the winter months. The combination of these factors has resulted in a reduction in the sewage load to the bay of just over 100 million moles, or about 27% of the total 2003 sewage nitrogen load to the bay as estimated by Nixon et al. (2008) (Table 3-3). Given the excellent performance relative to targets of plants which have upgraded to date, there is little reason to believe that planned upgrades to other plants scheduled for 2012, 2013, and 2014 will not combine to reach the targeted 50% nitrogen load reduction set out by RIDEM (RIDEM 2005).

Several of the upgraded plants are located along rivers, which seem to have highly variable abatement rates. While virtually all of the reductions calculated for plants discharging into the Pawtuxent River are realized in reduction in flux from that river, only about 30% of the nitrogen reductions calculated for the Blackstone River (about 50 million moles per year) are realized in reduction in annual flux for this river (about 15 million moles per year reduction) (Table 3-2, Table 3-3). While some of the decreased effluent discharge is mitigated by increased flow, driven by increased precipitation, it is clear that if the overall goal is reduction of load delivered to the bay proper, management effort should be focused on plants discharging either directly into the bay, or into tributaries which drain rapidly into the bay. This observation, however, does not take into consideration the improvement in ecosystem function which might be realized within these rivers by reducing load discharged into them.

Fluxes of nutrients from the sediment to the water column appear to have changed dramatically over time. Recent estimates are highly variable, but show significantly lower rates of denitrification and phosphorus flux in all observed cases than past estimates (Table 3-1, Fulweiler et al. 2007, Fulweiler et al. 2010, Fulweiler and Nixon 2011). The magnitude of the denitrification decrease approximately parallels the observed decrease in nutrient load to the bay proper from advanced wastewater treatment. Whether a result of changes in climate and/or phenology, or a direct result of loading reductions, the sedments, which were formerly a sink through denitrification for approximately 20% of the nitrogen incident on the system, now appear to be close to neutral in terms of net nitrogen flux. This change has the ability to mask or mitigate a great deal of the impact of present and future loading reductions if the sediments continue to be net neutral over an annual cycle.

The fluxes of nutrients across the bay/sound interface remain difficult to quantitatively estimate, but the flux of nitrogen from the bay into the sound may have decreased in response to loading reductions (Table 3-1).

Biological parameters (primary productivity, fish export, etc…) do not appear to have changed at this time (see chapters 1 &2, Oviatt 2008, Longval 2009). This is not surprising, given the small magnitude of loading reduction, logarithmic relationship between nutrient load and productivity, and response of the system (e.g. less denitrification, less river abatement, less flux across the bay/sound interface) to load reductions. It seems that, for the present time at least, there are still ample nutrients to support the sustained level of primary productivity observed before reductions. However, future reductions may be large enough to have an impact.

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Table 3-1: Nutrient budget for Narragansett Bay with sources for each flux. Units are in millions of moles nitrogen and phosphorus. Positive numbers indicate a source of nutrients to the bay, negative numbers represent sinks. Confidence intervals on river and plant loading are standard deviation of annual averages since upgrade (see appendix B for calculation).

Source	DIN	TN	DIP	TP	Notes
Direct Deposition	24± 5	30 ±6	-	0.13	Nixon et al. 1995
Rivers	173±43	249±62	4.7±1.2	10.54±2.6	Calculated, TP estimated by ratio
Direct Sewage Discharge	100±12	143±17	4.18±0.5	9.4±1.1	Calculated, DIP estimated by ratio
Urban Run-off	29±9	62±17	2.8±0.5	5.8±1	Reassessed based on Nixon et al. 1995
Groundwater	4	4	-	-	Urish and Gomez 2004
TOTAL INPUTS	**330±46**	**488±67**	**11.7±1.4**	**25.8±3**	
Denitrification	-20 ±60	-20 ± 60			Fulweiler et al. 2007, 2010, 2011
Burial	-	-70±26	-	-6.5±1.5	Nixon et al. 1995
Fisheries export	-	-11	-	0.65	Calculated from Longval 2009
Net Export to Sound	-102±12	-283±60	-29.8±3.3	-32±3.5	Calculated using GEM
TOTAL OUTPUTS	**-122±62**	**-384±94**	**-29.8±3.3**	**-39.1±4.0**	
STORAGE TERMS					
Standing Stock	15±3	45±8	2.5±.9	3±.4	Chapter 1
Sediments		1770±590		377±112	Calculated from Nowicki and Oviatt (1990)

Table 3-2: Comparison of river flow and nutrient flux from rivers between this survey and the 2003-2004 survey presented by Nixon et al. (2008). Units are millions of m^3/day for flow and millions of moles per year for flux.

	2003-2004		2008-2010	
	N	P	N	P
Blackstone River				
Mean Daily Flow	2.57		2.76	
Dissolved Inorganic	68.88	1.69	59.34	2.18
Total	98.63	3.87	84.73	5.36[a]
Pawtuxet River				
Mean Daily Flow	1.00		1.11	
Dissolved Inorganic	44.61	1.96	25.67	0.77
Total	59.29	3.61	36.78	1.63[a]
Woonasquatucket River				
Mean Daily Flow	0.28		0.29	
Dissolved Inorganic	6.62	0.16	4.10	0.03
Total	8.59	0.32	5.72	0.10[a]
Moshassuck River				
Mean Daily Flow	0.19		0.12	
Dissolved Inorganic	3.50	0.07	2.04	0.01
Total	4.77	0.13	2.68	0.02[a]
Ten Mile River				
Mean Daily Flow	0.35		0.33	
Dissolved Inorganic	9.86	0.24	11.84	0.08
Total	14.07	0.81	14.39	0.27[a]
Taunton River				
Mean Daily Flow	2.58[c]		1.59	
Dissolved Inorganic	86[c]	3.3[c]	23.53	0.35
Total	117[c]	5.3[c]	37.68	0.56[b]
Unmeasured Flow				
Mean Daily Flow	1.48[d]		2.90[e]	
Dissolved Inorganic	48.3	1.6	46.8	1.27
Total	66.5	3.1	67.3	2.85
GRAND TOTAL				
Mean Daily Flow	8.43		9.10	
Dissolved Inorganic	267.8	9.05	173.3	4.70
Total	368.9	17.13	249.3	10.54

[a] Calculated from the average ratio of inorganic to total phosphorus (Nixon et al. 2008)
[b] Calculated from the average of the average ratios of inorganic to total phosphorus (Nixon et al. 2008)
[c] data from (Boucher 1991) as presented in (Nixon et al. 1995)
[d] based on calculation of area of gauged to ungauged river area by (Ries et al. 1990) as modified by (Nixon et al. 1995)
[e] based on Ries et al. (1990) plus flow from 304 mi^2 of un-gauged flow in Taunton basin.

Table 3-3: Average wastewater treatment facility discharge for the time period from 2007-2010 at wastewater treatment facilities discharging into the bay or its tributaries. All values with the exception of flow, which is in millions of gallons per day, are in millions of moles per year.

PLANT	Flow	NH_4^+	NO_2	NO_3	DIN	TN	TP
Discharges to: **Narragansett Bay**							
Field's Point	44.45	37.40	3.23	5.84	46.47	63.50	3.19
Bucklin Point	21.37	1.27	0.23	13.70	15.20	18.90	3.14
Newport	9.20	*Nutrients*	*not*	*monitored*		10.50	0.59
East Providence	7.11	3.28	0.13	2.93	6.34	7.53	0.52
Bristol	3.57	1.94	0.17	1.93	4.04	6.27	0.18
Warren	1.92	1.35	0.02	0.22	1.59	1.86	0.05
East Greenwich	1.07	0.86	0.01	0.46	1.33	0.87	0.42
Quonset Point	0.47		0.04	0.46		0.73	0.10
Jamestown		0.05	0.00	0.10	0.15	0.16	0.02
Fall River*		22.90			24.95	33.20	1.15
Total		**69.04**	**3.83**	**25.64**	**100.06**	**143.52**	**9.37**
Blackstone River							
Worcester[1]	31.09	3.01			14.02	16.60	1.07
Woonsocket	7.48	0.97	0.06	3.24	4.27	4.99	0.56
Smithfield	2.01	0.18	0.07	1.04	1.29	1.46	0.02
Grafton*		2.00			3.34	3.28	0.14
Millbury*		1.96			2.42	2.44	0.24
Northbridge*		1.48			1.91	3.06	0.17
Burrillville	0.85	0.99	0.07	0.23	1.29	1.40	0.02
Hopedale*		0.13					0.02
Leicester*		0.03					0.00
Douglas*		0.10			0.14	0.20	0.02
Upton*		0.07			0.08	0.12	0.00
Total		**10.92**	**0.20**	**4.51**	**28.75**	**33.55**	**2.27**
Ten Mile River							
Attleboro	4.07	0.45				7.67	0.02
North Attleboro	4.28	0.41				2.98	0.03
Total		**0.86**				**10.65**	**0.06**

Table 3-3 continued.

PLANT	Flow	NH_4^+	NO_2	NO_3	DIN	TN	TP
Pawtuxent River							
Cranston	11.33	3.96	0.12	5.98	10.06	12.50	0.43
West Warwick	6.00	1.01	0.36	5.37	6.74	8.03	0.45
Warwick	5.00	1.43	0.06	2.39	3.88	4.75	0.21
Total		**6.40**	**0.54**	**13.74**	**20.68**	**25.28**	**1.09**
Taunton River							
Brockton*		15.72			27.56	36.51	0.83
Taunton*		2.04				4.18	0.29
Somerset*		2.68			3.44	8.28	0.17
Total		**20.43**			**30.99**	**48.97**	**1.28**
GRAND TOTAL						**262.0**	**14.1**

[1] Flow value is the average of flows from 2009-2010 instead of 2007-2010 as there was no flow data available for 2007 and 2008.

* Parameter values were calculated by scaling previous values, 2000-2003 (Nixon, 2008), by the population change from 2000-2010.

Table 3-4: Changes in urban run-off attributable to different sources of variability. A: Land use coefficients from Carter 1982 (used by Nixon et al.) and from NRCDS 2008 (used by this study) in moles per acre per centimeter of rain. B: Total acreage (in thousands of acres) of each land use type which discharges to Narragansett Bay as calculated in the two studies. C: Changes in urban runoff attributable to different vectors. All changes are relative to urban run-off figures presented in Nixon et al. (1995) and based upon the central assumptions presented therein.

Table 3-4A	Res.	Com.	Ind.	Hwy	Inst.	open
Nitrogen						
Carter 1982	3.23	3.53	1.33	5.5	-	-
NRCDS 2008	4.58	4.96	5.34	6.49	4.20	1.53
Phosphorus						
Carter 1982	0.16	0.028	0.21	6.1	-	-
NRCDS 2008	0.26	0.39	0.26	0.26	0.39	0.026

Table 3-4B	Res.	Com.	Ind.	Hwy	Inst.	open	other	TOTAL
Nixon 1995	33.2	6.88	29.75	3.31*	-	-	-	73.14
Present	64.65	6.97	7.80	3.54	4.61	37.4	15.1	140.1

*Our estimate of 1990 loadings corrects an mathematical error in Nixon et al. (1995) which incorrectly publishes this value as 8.49

Table 3-4 C: Changes to estimates of Urban Run-off into Narragansett Bay		
Constituent	**% Change TN**	**% Change TP**
Increased precipitation 10 year avg. 2000-2010 vs. Nixon et al. 1995	9	9
Land-use Change Primarily from increased # of lane-miles of roads & highways offset by loss of industrial acreage	19	14
CSO retention tunnel Based on phase one, complete 11/2008	-6	-6
TOTAL ATTRIBUTABLE TO CHANGES IN LOADING	**22%**	**17%**
Changes in Assessment Method: Use of GIS to categorize previously unconsidered sewered acreage, change to NRCDS coefficients.	102	52
TOTAL	**124% increase**	**71% increase**

Table 3-5: Conversion factors used to estimate bottom nutrient concentration in Rhode Island Sound from surface concentration. Conversion factors were established by comparing known surface and bottom concentrations from a 1972-1973 survey (Kremer and Nixon 1974) and a 1979-1980 survey (Oviatt 1980) and are the mean of all bottom/surface ratios for the given month at all stations located at the mouth of the bay in each study.

Month	**NH4**	**PO4**	**DIN**	**NO2+NO3**
Jan	0.49	0.89	0.95	0.98
Feb	0.58	1.08	1.01	1.06
Mar	0.78	1.22	0.62	1.34
Apr	1.14	1.10	0.62	0.48
May	1.48	1.02	0.74	0.76
Jun	1.79	0.92	2.36	2.36
Jul	2.82	0.85	3.09	2.86
Aug	2.38	1.07	2.35	1.84
Sept	0.92	0.99	1.01	0.98
Oct	1.00	0.91	0.86	0.73
Nov	0.38	0.68	0.42	0.43
Dec	0.34	0.82	0.61	0.69

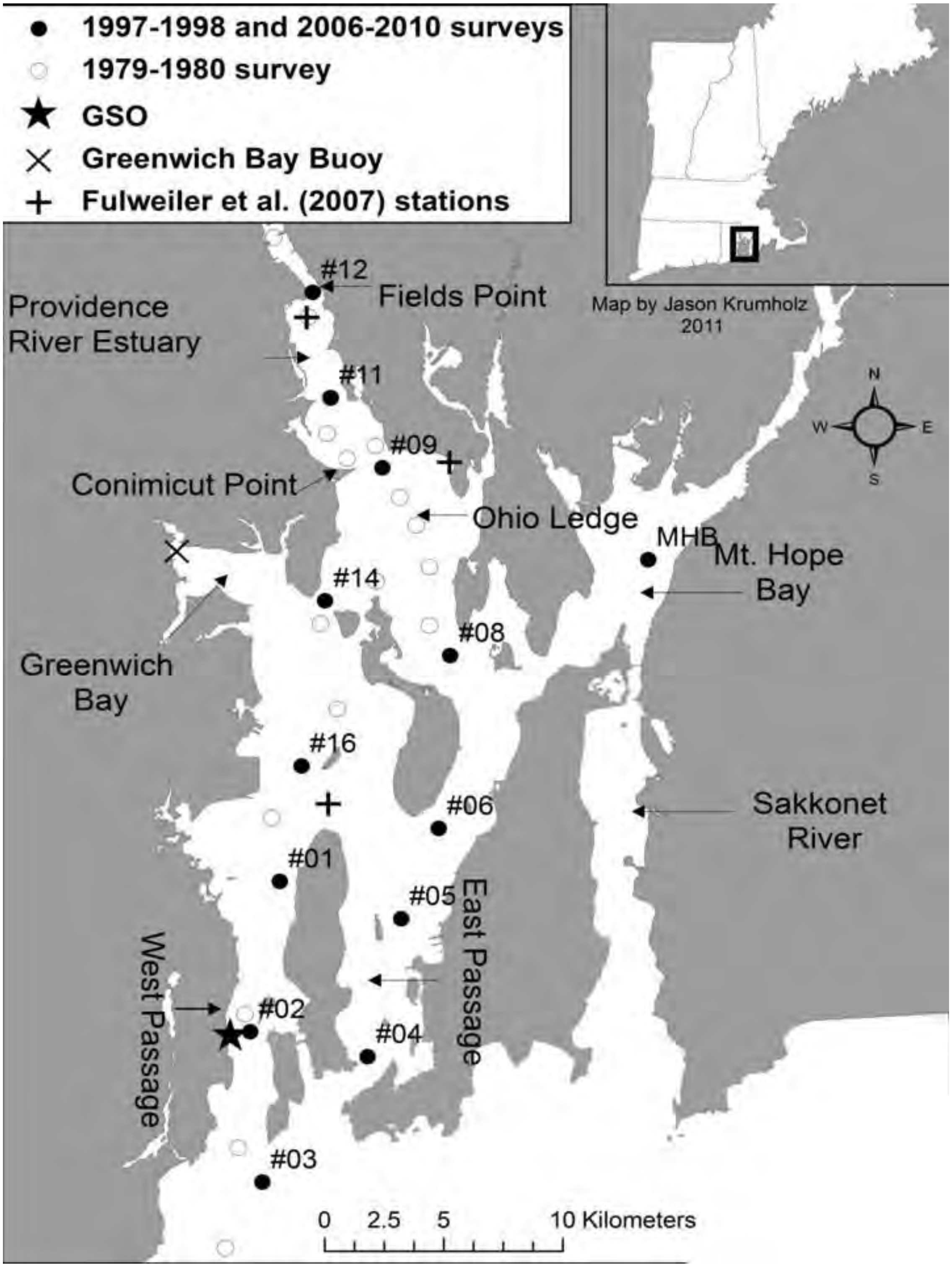

Figure 3-1: Map of Narragansett Bay showing the sampling stations and landmarks used by various studies cited within this manuscript.

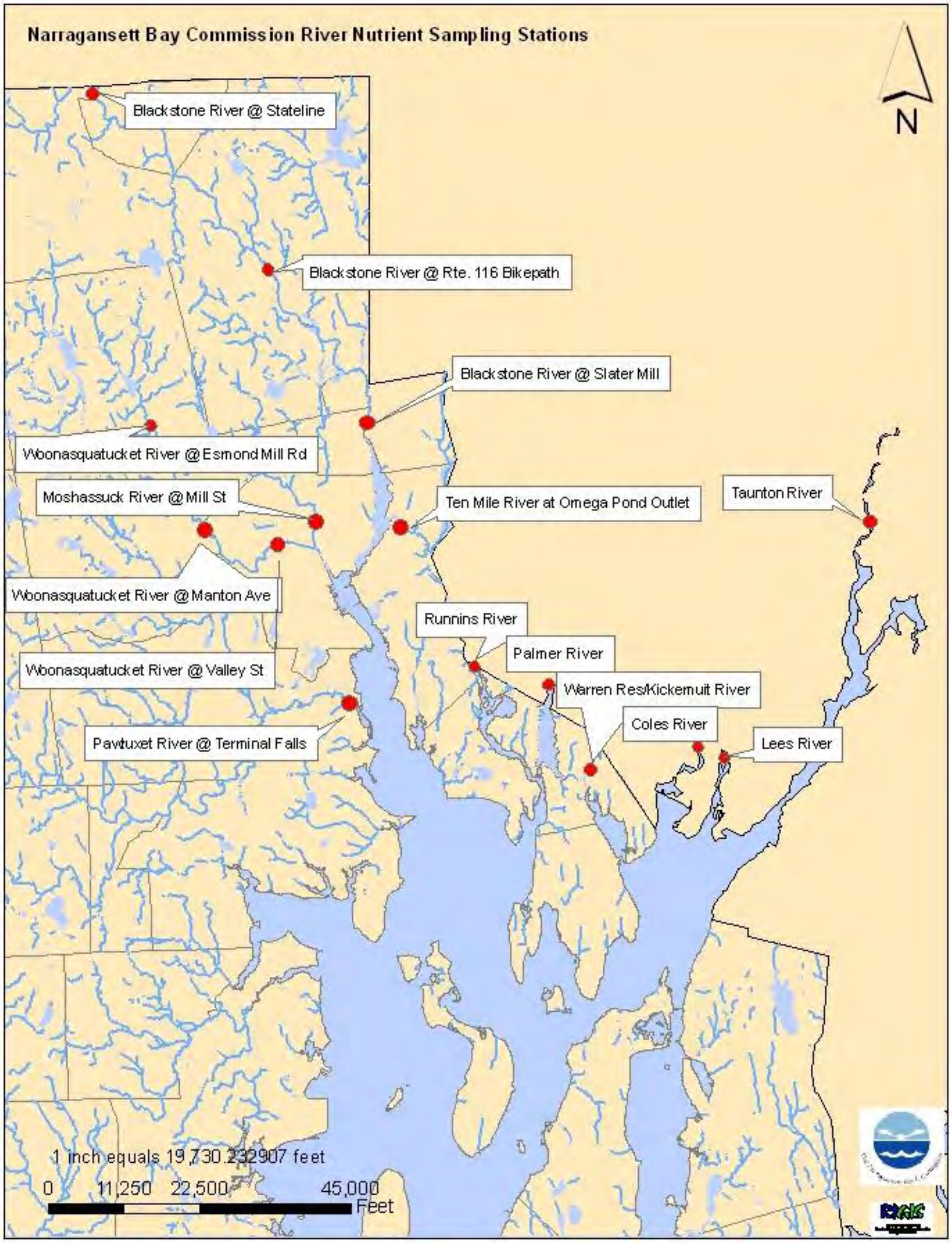

Figure 3-2: Map of Upper Narragansett Bay showing river sampling stations used by the Narragansett Bay Commission for nutrient sampling.

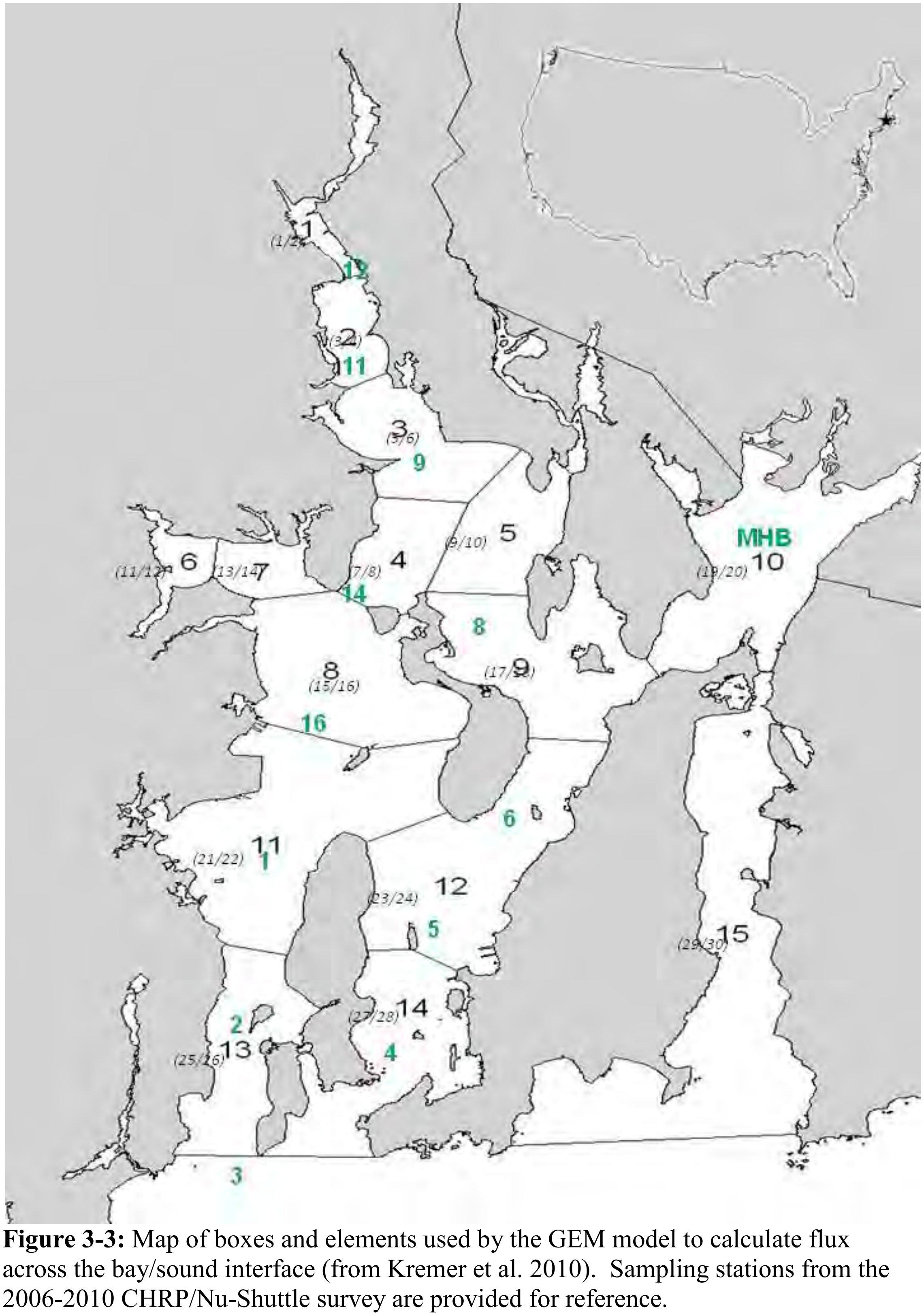

Figure 3-3: Map of boxes and elements used by the GEM model to calculate flux across the bay/sound interface (from Kremer et al. 2010). Sampling stations from the 2006-2010 CHRP/Nu-Shuttle survey are provided for reference.

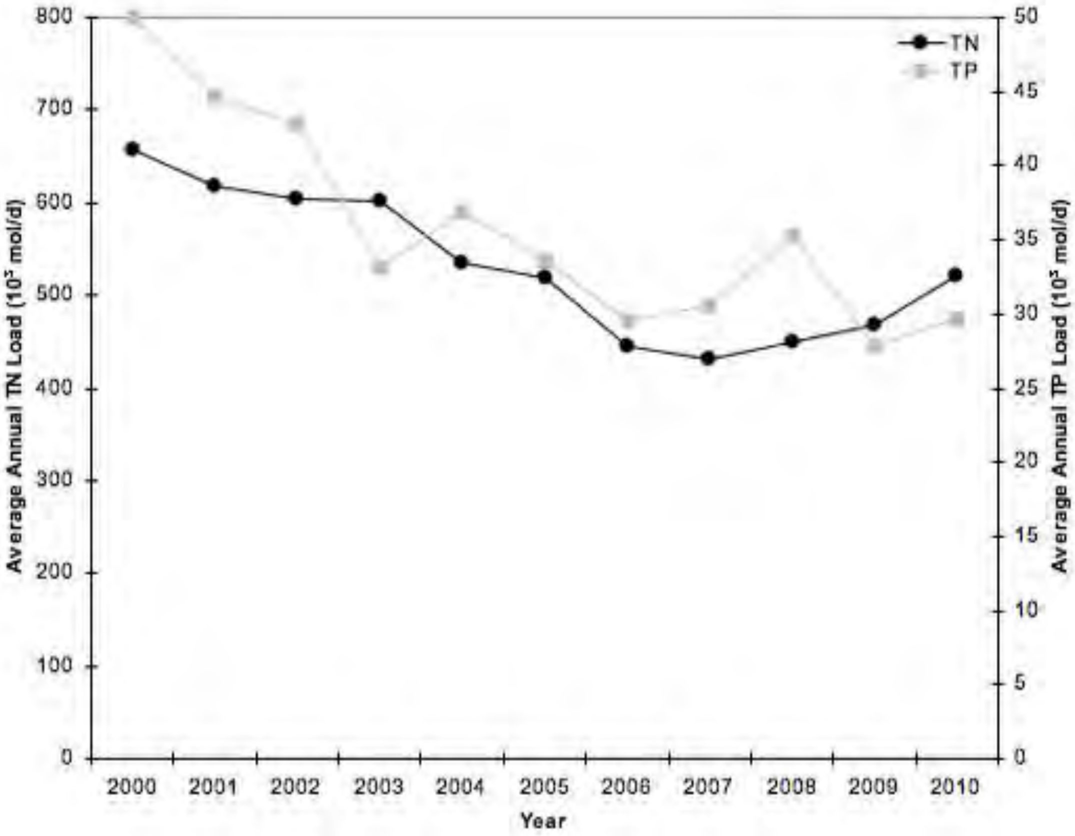

Figure 3-4: Estimated average daily total nitrogen (black, left axis) and phosphorus (grey, right axis) load to Narragansett Bay from sewage for the years 2000-2010. This load includes estimates from all plants discharging into the bay and tributary rivers. Units are thousands of moles per day.

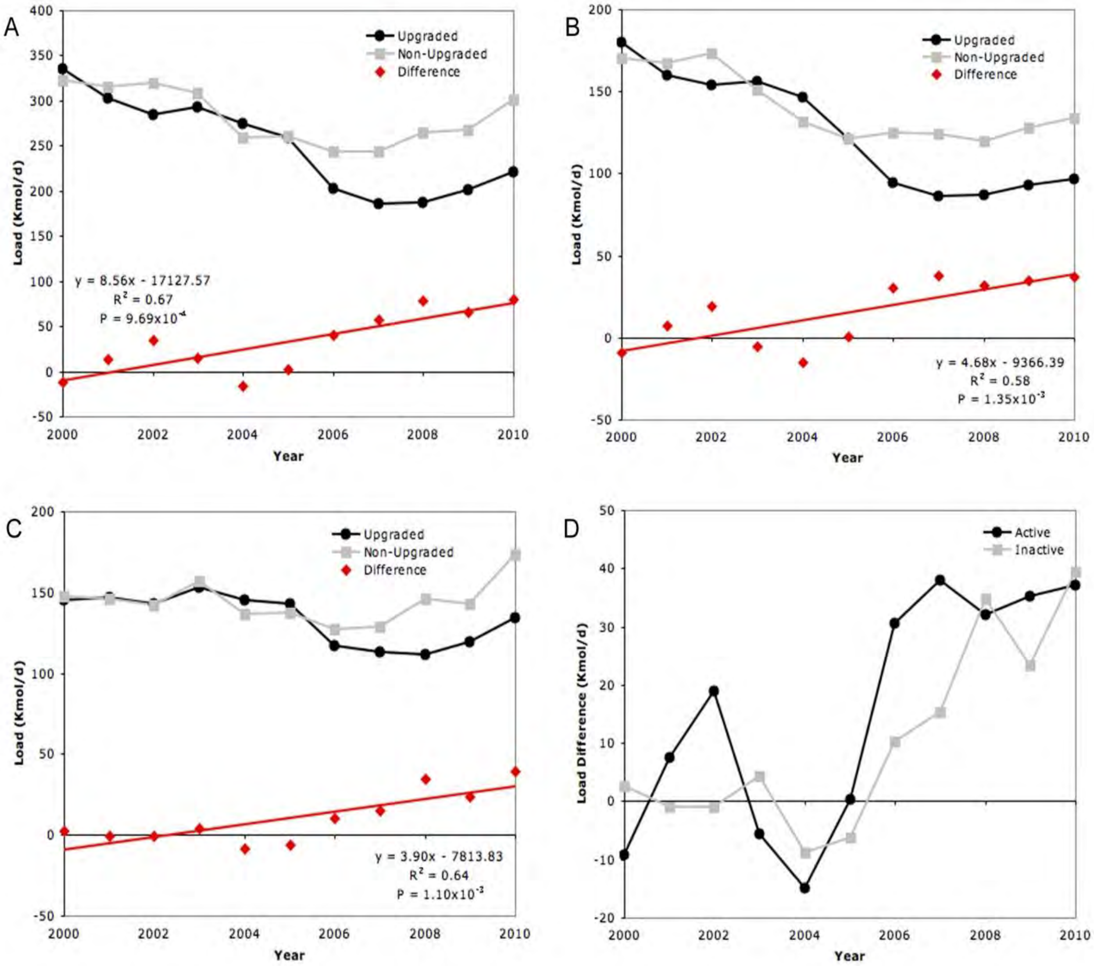

Figure 3-5: Total nitrogen (TN) load at 17 WWTF's for which data were available in thousands of moles per day. A) Annual TN load from facilities which underwent upgrades (black) and those which did not (grey) with the difference between the two (red). B) Active season (May-Oct.) TN load discharged from upgraded (black) and un-upgraded (grey) facilities with the difference in red. C) Inactive (Nov.-April) season difference (red) between upgraded (black) and un-upgraded (grey) plants. D) Improvement during active (May-Oct., black) relative to inactive (Nov.-Apr., grey) season difference among upgraded plants.

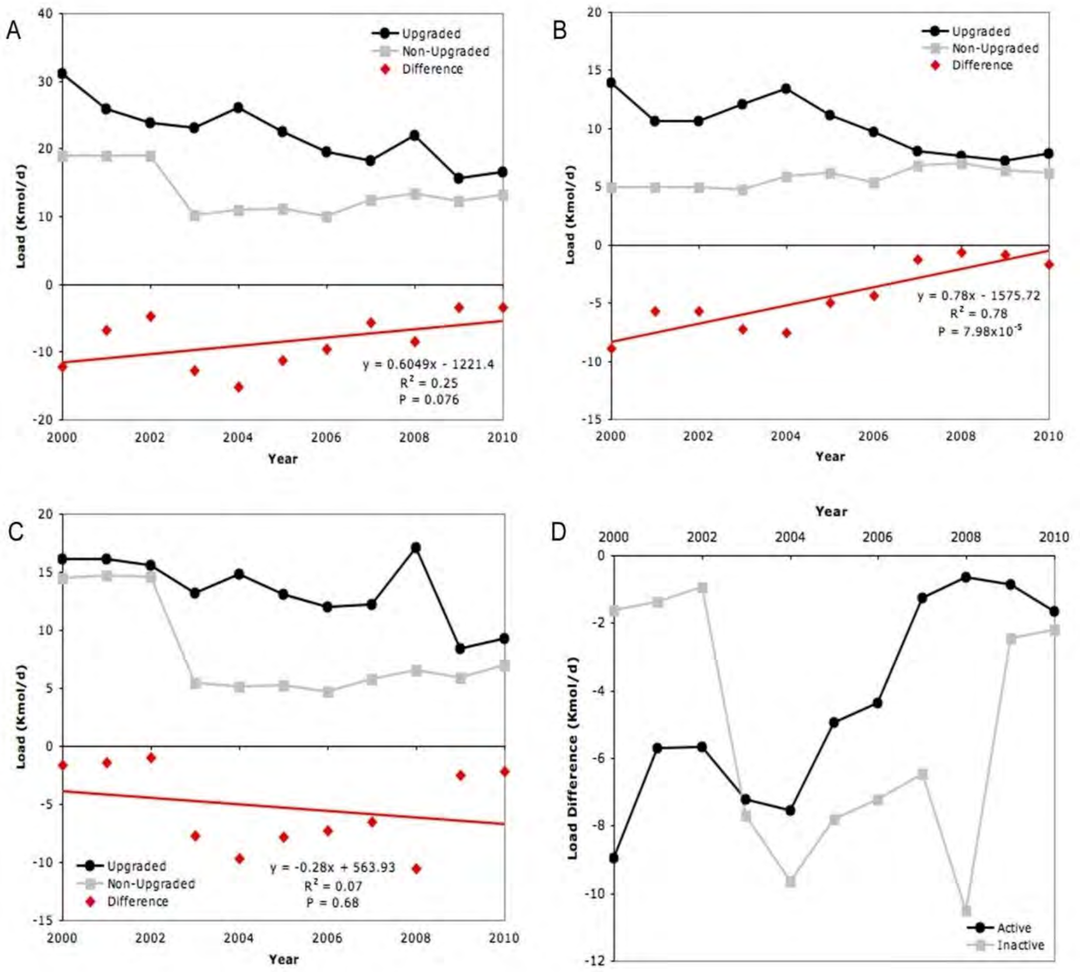

Figure 3-6: Total phosphorus (TP) load at 17 WWTF's for which data were available, in thousands of moles per day. A) Annual TP load from facilities which underwent upgrades (black) and those which did not (grey) with the difference between the two (red). B) Active season (May-Oct.) TP load discharged from upgraded (black) and un-upgraded (grey) facilities with the difference in red. C) Inactive (Nov.-April) season difference (red) between upgraded (black) and un-upgraded (grey) plants. D) Active (May-Oct., black) vs. inactive (Nov.-Apr., grey) season difference among upgraded plants.

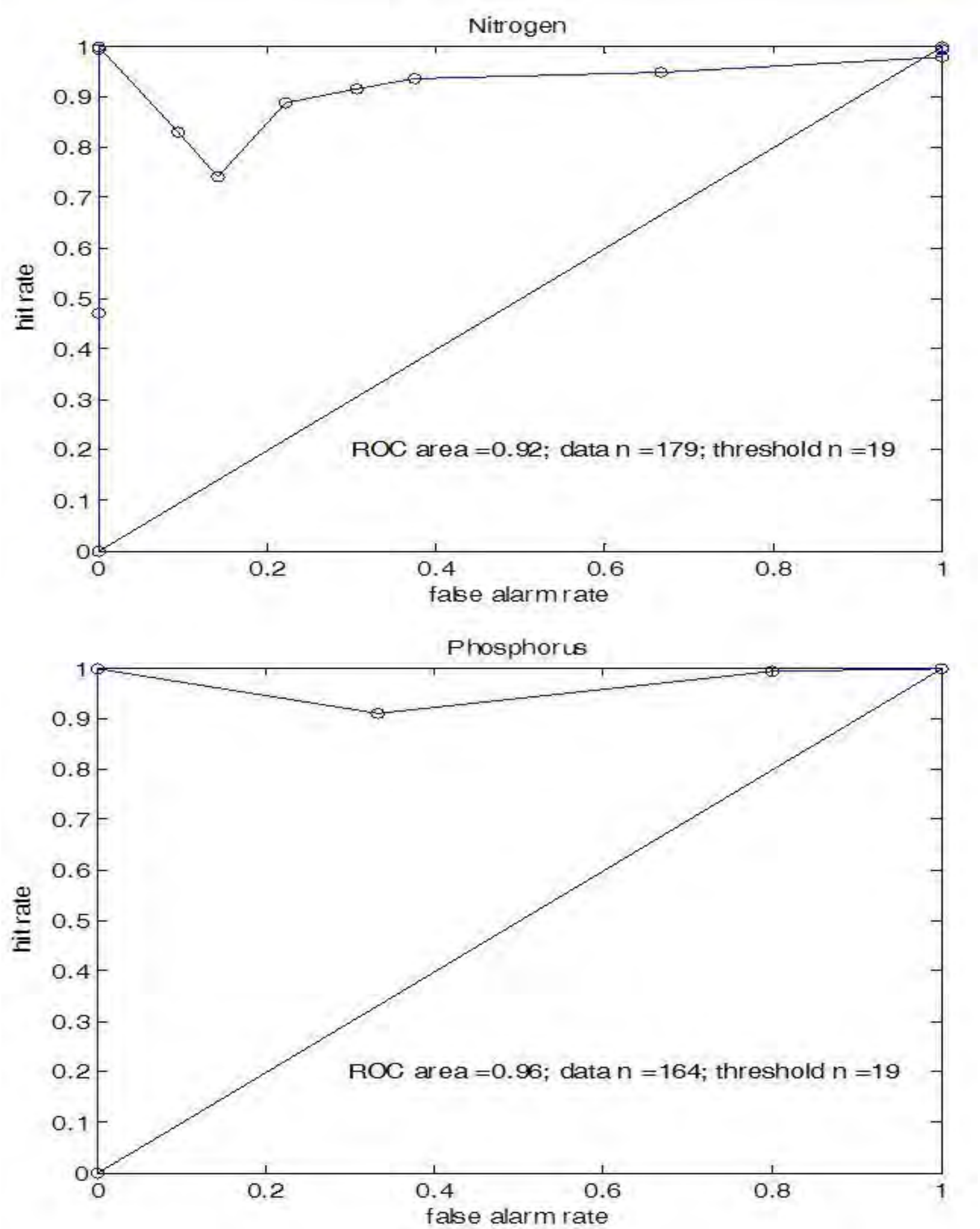

Figure 3-7: Relative Operating Characteristic (ROC) scores for GEM box modeled nitrogen and phosphorus concentration relative to observed concentration (Chapter 1). Scores presented are cumulative for all boxes, across the entire year (15 boxes, 12 months) and represent the model's ability to correctly match the observed data relative to 19 threshold concentrations. The area under the ROC curve is an indication of model skill, ranging from 0-1 where 1 is perfect and >0.5 (black line) is considered skilled.

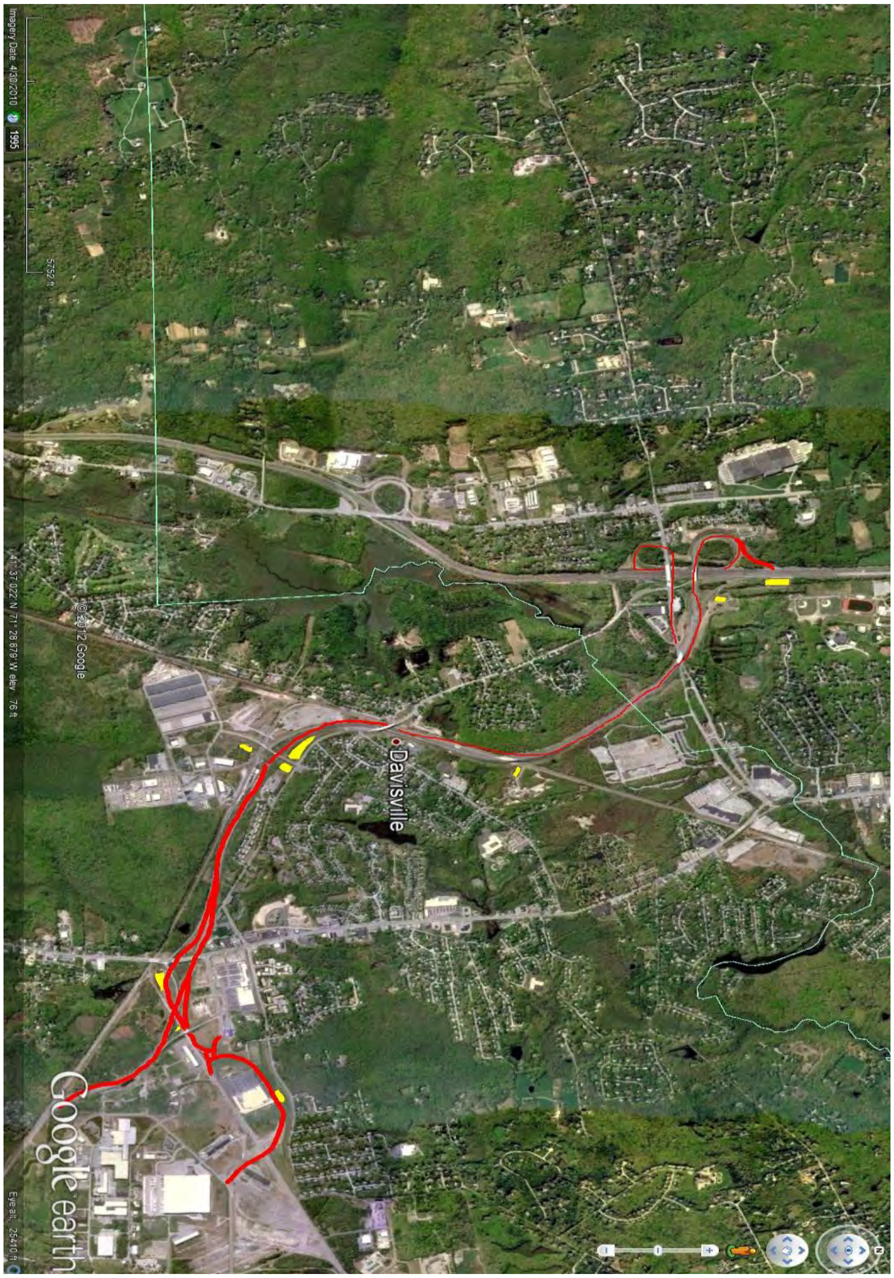

Figure 3-8: Map of areas of North Kingstown, Rhode Island impacted by recent construction of an extension for route 403. Newly created treatment wetlands are shown in yellow, while newly created roads are shown in red, with a thick red line indicating the addition of 4 new lanes of road, and a thin red line indicating expansion from 2 to 4 lanes.

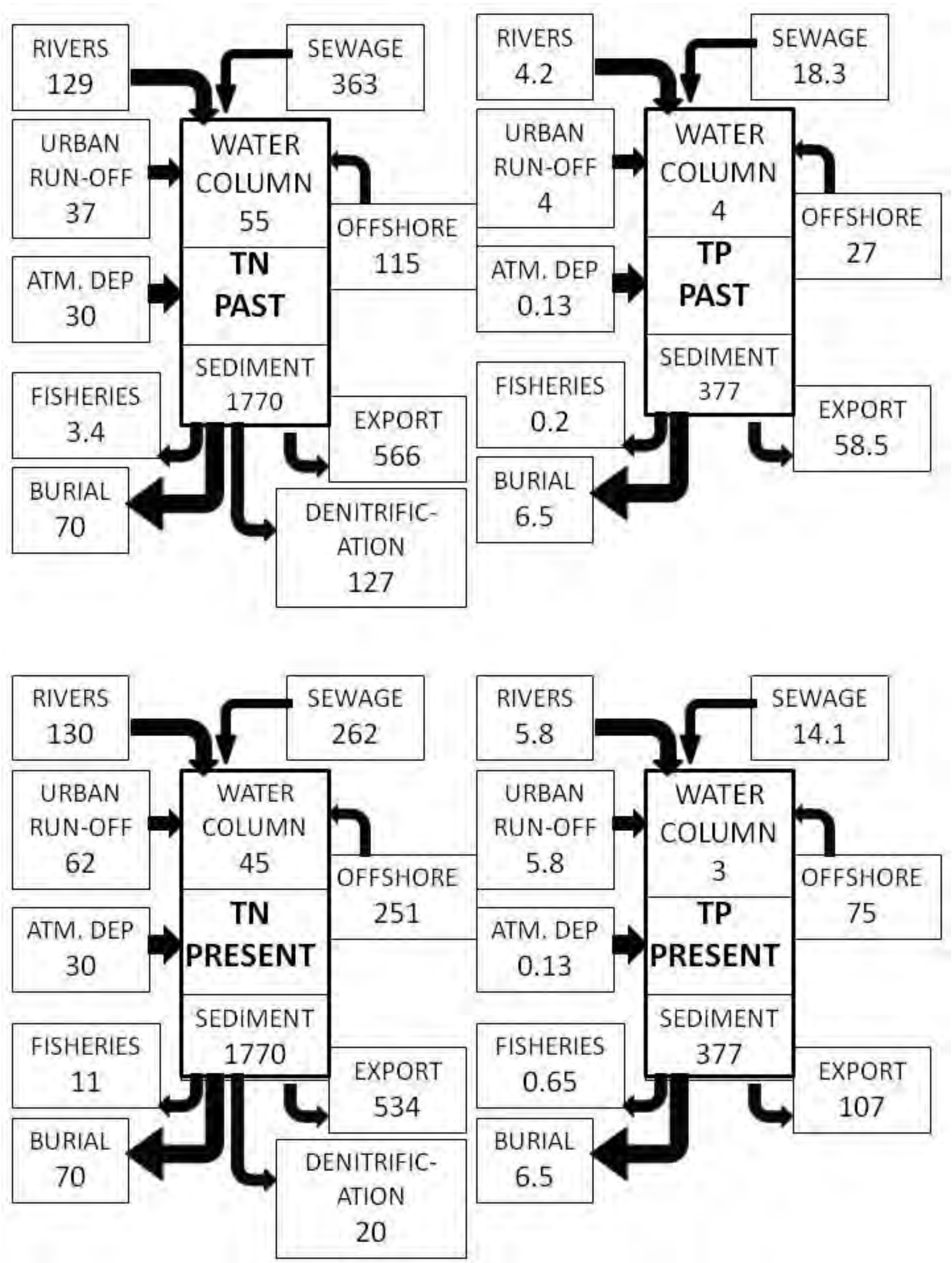

Figure 3-9: Box diagram of sources and sinks of nutrients to Narragansett Bay past and present. Past data are most recent available estimates from previous budgets by Nixon and colleagues (1995, 2008). Present data are 2006-2010 average, except export which is for 2006. Sewage value includes direct and indirect discharge, and river loading here is estimated as total river loading – sewage discharge into rivers. Export is presented as gross export. All units are millions of moles per year.

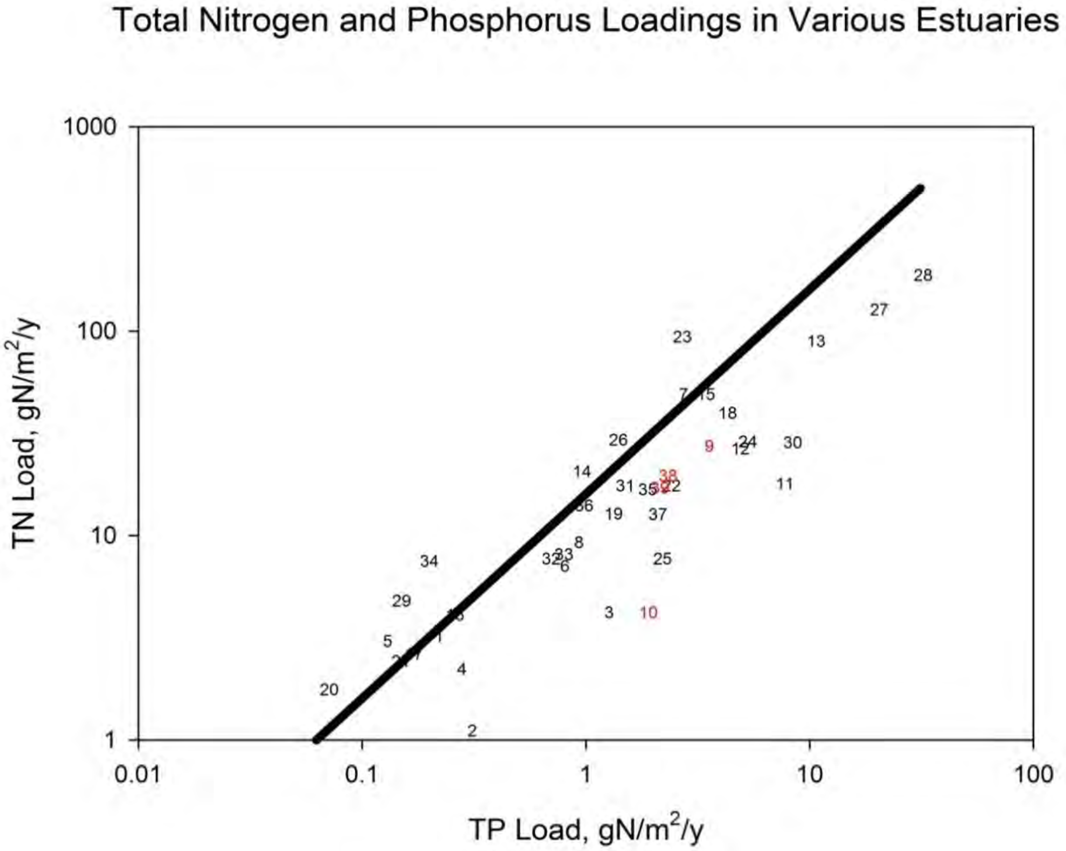

Figure 3-10: Total nitrogen and phosphorus loads to various ecosystems. Figure adapted from Boynton et al. 2008. Narragansett Bay points are shown in red, with point 9 representing the 1995 Nixon et al. budget, point 10 indicating estimates of prehistoric load to Narragansett Bay by Nixon et al. 1997, point 38 representing this survey, and point 39 representing the projected loadings for Narragansett Bay for 2014 once additional WWTF upgrades are complete. The line represents a 16:1 N:P loading ratio.

APPENDIX A

Supplemental Methods

This Appendix contains 3 sections. The first details the autoanalytic methodologies used on the two instruments presented in the study, their differences from each other and from the literature on which they were based. The second details the intercalibration procedure for the two instruments. The third is a Standard Operating Protocol and troubleshooting guide for the Astoria Analyzer, provided for reference purposes.

SECTION 1: AUTOANALYTIC METHODOLOGIES

Nitrate/Nitrite

Both the Astoria and Technicon autoanalyzers use a very similar chemical reaction to measure nitrate and nitrite. In both instruments, nitrite is detected by the formation of an azo dye during the Greiss reaction- the diazotization of Sulfanilimide (SAN) and subsequent coupling with N-1 napthyelthylenediamine (NED)(Fox 1979). This reaction takes place in a buffered acidic medium. The absorbance of the resulting dye is read at 540nm on both instruments. Nitrate is measured by reducing nitrate to nitrite using cadmium coated with copper (Wood et al. 1967)

This methodology was developed throughout the 1960's and is reviewed by (Strickland and Parsons 1968). The respective manufacturers detail their specific variations on this methodology used by each instrument (Technicon 1972a, Astoria-Pacific 2005), the recommended techniques for each instrument are followed exactly except that the Imidazole buffer called for in the Astoria Pacific methodology is replaced with the Ammonium Chloride/Ammonium Hydroxide buffer used in the

Technicon methodology. A comparison between the two buffers showed no increased precision with the Imidazole buffer, and since it is expensive and difficult to prepare, we retained the original method. Thus, the only differences between the two methods are as follows:

1) The Astoria technique uses a slightly lower concentration of the SAN reagent
2) Both the NED and SAN reagents are filtered at 0.45 mM before use in the Astoria, while the Technicon prodecure only calls for the filtration of NED
3) The Technicon methodology calls for a single mixed NED/SAN reagent (50/50) while the Astoria method calls for the reagents to be separated, but injects them sequentially in a 1:1 ratio.
4) The Astoria methodology calls for a small amount of surfactant (Brij-35 or TX-10) to be added to the SAN and the buffer, while the Technicon does not use surfactants.

These methodologies differ significantly from the standard EPA methodology for colorimetric determination of nitrate/nitrite in that they lack EDTA in the buffer, and use much lower ratio of reagent/sample (EPA 1983b). However, the use of EDTA was shown to be problematic, and the lower reagent concentrations reduce the blank value, and thus, are commonly used for the determination of low level nitrate/nitrite (Strickland and Parsons 1968, Grasshoff et al. 1983).

Phosphate

The phosphate methodology used by both instruments is very similar, and is essentially unchanged from the recommended Technicon Industrial Method (Technicon 1971). This methodology is based on the formation of phosphomolybdic acid (by mixing phosphate ions with molybdic acid in an acidic medium). The phosphomolybdic acid is subsequently reduced. This reaction produces heteropoly blue, which can be read at 660nm or 880nm. The reduction is typically accomplished using ascorbic acid, however alternate methodologies call for hydrazine sulfate or stannous chloride. The method was initially described by(Murphy and Riley 1962), and modified for use on autoanalyzers by (Hager et al. 1972). This method is very similar (stochiometrically identical) to the established EPA procedure for autoanalytic phosphate measurement (EPA 365.1), with the only minor difference being the diameter of the diluent line, which is slightly different between the EPA method, the Technicon method, and the Astoria method (EPA 1983c)

Astoria Pacific has methodologies for both ascorbic acid reduction (A205) and hydrazine (A204) (Scott et al. 2005) but in order to maintain maximum continuity in the transition between instruments, it was deemed best to continue using ascorbic acid reduction, since the only major downside of this methodology is that the reagent is relatively unstable, and must be prepared daily. Although both instruments use essentially identical reagent chemistries, the following minor differences exist:

1) The Astoria regent has a small amount of surfactant (SLS) added, while the Technicon reagents do not use surfactant.

2) The Technicon procedure calls for 4.9N Sulfuric Acid, while the Astoria procedure calls for 5.0N acid.

3) The Astoria reagent is filtered at 0.45mM before use.

4) The Astoria uses an 880nM filter while the Technicon uses an 820nM filter

Ammonia

There are a wide range of commercially available techniques for the measurement of Ammonia. Both the Astoria and the Technicon use methods based on the Berthelot reaction. In this reaction, hypochlorite (bleach), alkaline phenol, and ammonia are combined and heated in a heat bath at 65°C to produce indophenol blue. The intensity of this colorimetric reaction is intensified by the addition of sodium nitroferricyanide (also referred to as nitroprusside).

Both the Astoria and Technicon methods are based on the technique detailed by (Solorzano 1969). MERL uses a Solorzano modified version (order of reagents flipped) of the original Technicon method (Technicon 1973) on the Technicon analyzer. MERL procedure uses two reagents; a combined phenol/nitroferricyanide reagent, and a sodium citrate/sodium hydroxide/sodium hypochlorite complexing reagent. The air line for this cartridge is scrubbed through a 10% sulfuric acid solution to remove airborne ammonia contamination (a major problem). On the Astoria analyzer, MERL uses a modification of Astoria method A026 (Scott et al. 2005). The Astoria method is similar to the Technicon method stoichiometrically, except that it calls for a third reagent. In this case, a weaker nitroferricyanide/phenol reagent, a separate sodium hydroxide/sodium hypochlorite reagent, and a complexing reagent of sodium citrate, potassium sodium tartarate, and sodium hydroxide are used. The addition of tartarate to the complexing reagent is intended to remove any crystallization of calcium and/or magnesium which can occur during the reaction process, and which interferes with the reading as the sample passes through the flowcell. While the Technicon does not appear to suffer from this problem even without the tartarate (Oviatt and Hindle 1994), the Astoria was experiencing irregular baselines and random spikes attributed to the precipitation of calcium by this reaction. To combat this, the amount of hydroxide used in the reagents was reduced by half from the published values, in order to lower the pH of the reaction and inhibit crystallization. This modification is based on work done by Dr. Christopher Schmidt at Texas A&M (Schmidt and Clement 2009). To combat airborne interference, this cartridge is injected with ultrapure (99.95%) N_2 gas, rather than air.

The differences between the MERL Technicon and Astoria methods can be summarized as follows:

1) The Astoria method uses a potassium sodium tartarate addition to the complexing reagent to prevent crystallization. The Technicon does not experience this problem

2) The Astoria method separates the hypochlorite from the complexing reagent

3) The Astoria method uses a weaker mixture of phenol/nitroferricyanide

4) The Astoria method uses dinitrogen gas rather than scrubbed air to segment flow

5) The Astoria method uses a small amount of surfactant (TX-10 or Brij-35) added to the complexing reagent. The Technicon does not require surfactant.

6) The Astoria measures at 640nM, the Technicon measures at 630nM

Silicate

The Technicon and Astoria use different methods for the analysis of silicate in seawater. The MERL method for the Technicon is based on Technicon method 186-72W (Technicon 1972b). This involves the reaction of silica with an acidic molybdate solution to produce silico-molybdic acid, which are reduced (similarly to colorimetric ortho-phosphate methods) to produce a heteropoly blue complex. This method was first tuned for autoanalysis by Brewer and Riley (Brewer and RIley 1966). The Technicon method calls for the addition of oxalic acid prior to the reaction with molybdate to eliminate interference from ortho-phosphate (since the colorimetry for phosphate is very similar), and uses ascorbic acid as the reductant.

The Astoria method uses Astoria method A026 (Scott et al. 2005) wherein a similar ammonium molybdate solution to form silico-molybdic acid. Subsequently, tartaric acid is used to destroy any phospho-molybdic acid compounds which have formed (essentially different ways of dealing with the same phosphate interference problem). Stannous chloride is then used as the reducing agent. This method is discussed in detail by Sakamoto et al. (Sakamoto et al. 1990) Both instruments read the resulting silicoheteropoly blue at 820 nM.

The Ascorbic/Oxalic/Molybdic technique used by the Technicon is far more popular among general use (Gilbert and Loder 1977, Gordon et al. 1993), however, this technique does not appear to be compatible with the surfactant (SLS) required for the Astoria to run smoothly. After several attempts to modify this technique to achieve consistent results, it was abandoned in favor of the above discussed method. In summary, methods differences between the Astoria and the Technicon are as follows:

1) The Astoria uses tartaric acid rather than oxalic to eliminate phosphate interference
2) The Astoria uses stannous chloride rather than ascorbic acid as the reductant
3) The Astoria uses surfactant (SLS) in the molybdic acid reagent. The Technicon uses no surfactant
4) All Astoria reagents are filtered at 0.45mM. Technicon reagents are not filtered.

Total Nitrogen (TN)/Total Phosphorus (TP)

The analysis of total nitrogen and total phosphorus is accomplished by the use of a persulfate oxidation reaction conducted on whole (unfiltered) seawater. 22.5ml of seawater is digested by boiling for 30 minutes with 2.5 ml of potassium persulfate/boric acid/sodium hydroxide oxidizing reagent. This breaks down organic nutrients, converting them to dissolved inorganic form, at which point they are run on the autoanalyzer in an identical fashion to Nitrate and Ortho-Phosphate. This method was initially described by Valderrama (Valderrama 1981), and is used frequently for seawater (Grasshoff et al. 1983).

The measurement of TP in seawater using this technique is fairly robust, however the measurement of TN by this technique has been the subject of some debate. Prior to the use of the alkaline persulfate digestion, the primary technique in use was the Kjeldahl digestion, which is rapid and robust, but has several key drawbacks, most notabily, the toxicity of the reagents, and the fact that the resultant value (often referred to as TKN, or total Kjeldahl nitrogen) is a measure of ammonia plus organic nitrogen, and does not include nitrate and nitrite, two major inorganic constituents which are captured by the alkaline persulfate methodology. The major drawback of the alkaline persulfate technique is that it is dependent on a high and consistent conversion rate of ammonia and organics into nitrate and nitrite. This conversion efficiency is highly sensitive to the temperature and time of the extraction process, and incomplete extraction, if not appropriately corrected for, can bias results. Furthermore, because the estuarine TN values are significantly higher than typical estuarine nitrate values (TN values in upper Narragansett Bay routinely exceed 60mM and can reach 100mM, while nitrate values rarely exceed about 20mM), issues with nonlinearity of standards and cadmium reduction efficiency can emerge, as well as the potential for depletion of the cadmium column during the run day, causing efficiency loss (Scott et al., 2005: Scott, pers. comm.). USGS recently compared the two techniques, and found that while TP and TKP reliably produce consistent values, TN (minus nitrate and nitrite) and TKN do not always agree, particularly at high nutrient levels. The cause for this discrepancy is uncertain, but the reports suggests that this is likely due to nitrate interference in the TKN methodology, but potentially due to extraction efficiency problems with the alkaline persulfate technique (Patton and Kryskala, 2003).

SECTION 2: INTERCALIBRATION RESULTS

Nitrite/Nitrate

Intercalibration of nitrite was relatively straightforward. The relationship is approximately 1:1, and the R^2 is around 0.99 (Figure A-1). It should be noted that the tightness of the fit sometimes breaks down somewhat at low (< 0.3mM) concentrations, with the Astoria showing detectable levels of nitrite, while the Technicon values are near the detection limit (Figure A-2). This may be a factor of increased low range sensitivity in the Astoria technique, which is more precise, and uses a higher SAN concentration. In all cases (with and without high point), the relationship is not significantly different from 1:1 by ANCOVA.

Intercalibration of nitrate on the other hand, was extremely problematic. On any given day of intercalibration, the relationship between the two machines is typically fairly strong ($R^2<0.9$), but the slope is inconsistent, and not close to 1:1. During some run days, the slope even appears to change mid-run (Figure A-3). These mid-run changes do not appear to be precipitated by any change in methodology, and are likely due to a rapid change in Cadmium reduction efficiency, perhaps caused by a blockage in the Technicon column. The shift is not likely to have been precipitated by a change in the efficiency of the Astoria unit, since during the run day, that instrument performs regular tests of its cadmium efficiency, all of which were within specification.

The Technicon always produces higher values, with slope varying from approximately 1.3:1 up to 1.8:1, and averaging about 1.6:1. To test whether one instrument or the other was the source of the problem, identical samples were run on both instruments as well as a Teledyne model 2003 Nitrous Oxide sensor, which uses a vanadium/sulfate reduction (as per (Braman and Hendrix 1989) which eliminates the potentially troublesome cadmium reduction step. This instrument is much more precise and accurate than either the Technicon or the Astoria (although it is very time consuming and cannot be used in segmented flow autoanalysis). Results from this inter-comparison suggest that the newer Astoria analyzer was producing reasonably accurate results, while the Technicon appeared to be severely overestimating, especially at higher concentrations (Figure A-4).

Given the relative reliability of nitrite results, it was deduced that the likely culprit for this variability is the Cadmium reduction process. Approximately four years ago, the Technicon was switched from Cadmium columns intended for use on that machine to columns designed for a Lachat brand analyzer, with a much lower inner diameter. This was done without any sort of intercalibration or testing. It is hypothesized that this lower inner diameter results in incomplete reduction of standards at higher concentrations, producing an artificially shallow standard curve, and causing over-estimation of actual nitrate levels in samples with high concentrations. This is further complicated by the fact that the analyst applied a 'correction' to all nitrate data based on a one point 'check' of cadmium reduction efficiency. Given that the loss of efficiency appears to be dependent on concentration, this may have caused an underestimate of samples with low concentrations.

In order to test this hypothesis, old Technicon Cadmium columns were repacked according to the procedure detailed in the MERL manual (Oviatt and Hindle 1994). When the Cadmium efficiency 'correction' was removed, a relationship of 1.05:1 was observed, with an R^2 of <0.99 (Figure A-5). This relationship is not significantly different from 1:1 by ANCOVA. This provides strong evidence that the combination of incomplete reduction from the smaller diameter coil and an incorrectly applied 'correction' are the source of the disagreement between instruments.

However, in order to use the data which was run on the Technicon (which is essential for the compilation of nutrient budgets, and the comparison of present nutrient standing stocks with those of the previous decade), it was necessary to derive an empirical correction factor which relates concentration on the Technicon (using Lachat Cadmium columns) to appropriate values. In order to do this, it was necessary to go back to the raw data sheets, and re-calculate the Cadmium efficiency 'correction' for each run day, and then remove this correction from the data, after which Astoria and Technicon values were compared across the pooled intercalibration samples (approximately 200), and a consistent correction factor was determined (Figure A-6). A linear regression for the slope of the correction factor has intercept not significantly different from zero ($P=0.50$) and a highly statistically significant slope ($P<0.0001$). Analysis of covariance shows corrected data have a relationship not significantly different from 1:1 against the Astoria data.

Phosphate

The intercalibration of ortho-phosphate between the two instruments proceeded very smoothly. The relationship between the two instruments is consistent, very close to 1:1, and displays good correlation across the entire range of samples measured (Figure A-7). This relationship is not different from 1:1 by ANCOVA. This seems logical given that the two chemistries are virtually identical, and this technique is used almost unilaterally, with little variation, for colorimetric analysis of Ortho-phosphate in seawater; a surefire indication of its reliability.

Ammonia

The intercalibration of ammonia between the Technicon and the Astoria has met with somewhat mixed results. Once the Astoria technique was modified to remove any interference from precipitates, the relationship is approximately 1:1, especially at higher levels and the correlation is reasonable (R^2 approximately 0.98) (Figure A-8). However there is a bit of variability and noise in the data. On different run days, the relationship can be slightly greater or less than 1:1, and the R^2 can be as low as 0.97 (Figure A-8). At present, the only explanation for this variability is the inherent noise in this analytical technique. Ammonia baselines are noisy and tend to drift on both instruments, and attempts to correct for this are not always completely successful. Furthermore, even with the nitroferricyanide, the absolute amplitude of the signal (intensity of the color reaction) is low on both instruments (the absorbance peak of the high ammonia standard is less than 10% as intense as the nitrate high standard).

Another possible explanation for the variation is that colorimetric ammonia determination is slightly salinity dependent. Because the Astoria uses a ‘matrix matching’ analysis technique (where the rinse water is approximately the same salinity as the sample), it would be susceptible to variation if the salinity of the sample varies significantly from the salinity of the rinse water. Similarly, the Technicon might experience variability if the salinity of the standards were different from the salinity of the sample (it can be corrected with an equation. To correct for this, we have begun testing samples for salinity, and will alter the Astoria matrix as necessary to account for low salinity samples.

In order to determine whether the difference between the instruments is variable (and therefore uncorrectable in an intercalibration) or whether one machine consistently reads higher or lower than the other, additional intercalibration samples were run on three additional separate days. Once salinity corrected, data above 3X MDL (deemed by the EPA to be the functional reporting limit) show a very strong relationship not statistically different from 1:1 (Figure A-9). While the Astoria appears to be able to resolve samples significantly below this concentration, replication on the Technicon at very low concentrations becomes problematic and the correlation between instruments is poor.

Silicate

Despite significant differences in the methodology, silicate intercalibration proceeded smoothly. The relationship is very close to 1:1 (it improves further with the reduction of the two outliers) and the correlation is good (R^2>.99). This relationship is not statistically different from 1:1 (Figure A-10). Two outliers are present in the dataset, which were sequential samples when run, but since no concrete explanation can be arrived at for why these samples deviate from the expected pattern, they are not excluded from the analysis.

Total nitrogen (TN)/Total phosphorus (TP)

Given the fact that, from an autoanalytic standpoint, the measurement of TN/TP is identical to the measurement of nitrate and phosphate, one would expect to get similar results for the intercalibration of TN and TP to the results achieved for nitrate (highly problematic) and phosphate (extremely reliable). For the most part, this is the case, although the measurement of TN/TP proves to introduce significantly more variability in the data, lowering R^2 values for both TN (Figure A-11) and TP (Figure A-12). The significantly greater than 1:1 relationship on the nitrate channel persists, as expected, into TN analysis. What is rather unexpected is the degree of variability in TN observed in this intercalibration. While each individual run day produces a relatively strong correlation between the Astoria and Technicon results (individual R^2 values range from approximately 0.91-0.98), the slope of the relationship is highly variable (ranging from almost exactly 1:1, to as high as 2:1), resulting in a very weak relationship when the data is pooled, which is not only poorly correlated (R^2=0.57), but also shows signs of a potential baseline or blanking problem (intercept 5.1mM) (Figure A-11).

In attempting to account for the increased variability caused by the TN/TP procedure over the inorganic analogues, and other inconsistencies observed in preliminary data analysis (e.g. some samples with Ortho-P values higher than TP) a thorough review of MERL TN/TP procedures compared to recommended literature procedures (Grasshoff et al. 1983, Oviatt and Hindle 1994) was conducted. The following inconsistencies were identified:

1) Protocols call for vials to be dried at 200^{o}C after cleaning. Present MERL procedures utilize a 60^{o}C oven for this purpose

2) Literature protocols call for the use of fructose 1-6-diphosphate (TP) and glycene (TN) standard curves rather than traditional sodium nitrate and potassium phosphate standards used for DIN analysis. Using an organic standard corrects for extraction efficiency losses during the extraction process (typically nonlinear). MERL uses inorganic standards with a one point extraction efficiency check, and does not apply a correction.

3) Literature recommends pre-diluting any samples expected to have TN above 50mM as extraction efficiency falls off at this point. MERL does not pre-dilute samples anticipated to be above this threshold (e.g. Fields Point station).

4) Literature also recommends multiple recrystalizations of Potassium Persulfate, and that persulfate be stored in a vacuum jar with sulfuric acid and potassium permanganate to scavenge organics

These deviations from protocol are likely to cause two potential problems. Failure to properly clean vials before extraction could cause blanks to be too high, and indeed, upon inspection MERL TN blanks range from about 2-10mM TN as compared to literature values of 1-2 mM, and MERL TP blanks range from about 0.3-1.0 as compared to literature values of 0.3-0.5mM (Grasshoff et al. 1983, Oviatt and Hindle 1994, Patton and Kryskalla 2003). Given that the low end of observed MERL blank values is in line with literature values, and only 1-2 blanks were run for each run day, sometimes with significant variability between the blanks, it can only be assumed that different vials possess different amounts of contamination, and as such contamination variability could be passed along to the sample, which would more than explain the approximately 5% loss of correlation between total nutrient and dissolved nutrient intercalibrations. While this problem cannot be corrected for in the existing dataset, it can be rectified moving forward, to improve the precision of our measurements. Further experimentation on this matter revealed that with 3 recrystalizations and proper storage of persulfate, MERL blanks can be brought into the 2mM range

The use of improper standards is perhaps a more serious problem. A preliminary analysis comparing inorganic to organic standards was conducted to assess the severity of the potential loss. As suggested in the literature, TN samples above approximately 50mM TN showed decreased extraction efficiency. No such problem was observed for TP extraction efficiency, which remains reliable and linear up to approximately 50mM (much higher than the highest observed field values). TN standards of 12, 24, 36, and 48mM closely paralleled equivalent DIN standards, but by 200mM, extraction efficiency loss was about 30% (Figure A-13). This means that high TN values in the existing dataset will be under-represented, and any loss or change in extraction efficiency from day to day would not be corrected for in the data.

Conclusion

With some minor modifications to procedure, we were able to successfully intercalibrate all analytes between the two instruments. Nitrite, phosphate, ammonium, silicate and total phosphorus can be directly compared between instruments without the need for a correction factor. These channels show strong regression relationships with high R^2 and statistically significant slopes, with intercepts not significantly different from zero. All also showed no significant difference in slope between the established relationship and a 1:1 line (Figure A-14).

Nitrate and TN data required significant additional attention, however once an erroneously applied cadmium correction coefficient was removed from the data, and dilutions were appropriately treated, the data show a reliable and correctable pattern of underestimation by the Technicon in both TN and nitrate (which is to be expected since they run on the same channel). Once a correction factor is applied to the Technicon data they show reasonable comparability with the associated Astoria data, and have slope and intercept not significantly different from 1:1.

After intercalibration, all analytes showed EPA Method Detection(Ripp 1996) limits similar to literature values(Grasshoff et al. 1983) (Table A-1).

SECTION 3: STANDARD OPERATING PROCEDURE (SOP) FOR MERL NUTRIENT SAMPLE ANALYSIS USING ASTORIA 5 CHANNEL SFA

Procedure compiled 3/2012 by:

Jason Krumholz

Rosmin Ennis

M. Conor McManus

Preface

This appendix is designed to serve as an operational guide for daily use, maintenance, and routine troubleshooting of the MERL Astoria-Pacific 5 channel Segmented Flow Nutrient Analyzer. While many parts of this document are specific to the MERL lab set-up and designed to aid in transitioning the use of the instrument between operators and technicians, many portions may be of use to others using this, or a similar colorimetric nutrient analyzer. See earlier sections of this Appendix for more specifics about the colorimetric techniques used on this instrument.

I. SAMPLE COLLECTION AND STORAGE

Prior to Collecting Samples

1. Build Nutrient Filters
    a. Rinse all parts of the filter with DI water.
    b. Place the circular disk onto the large piece and press an O ring into the groove around the circular disk.

    c. Use tweezers to place a polycarbonate 0.45 micron filter (part # K04CP04700) onto the circular disk.
    d. Press the round piece with a "tail" on top of the filter and O ring and screw on the last piece tightly.
    e. Be sure to build at least 13 nutrient filters.
2. Labeling Bottles: There are 13 stations from which samples are collected. Two samples are collected from each station: one filtered sample for Dissolved Inorganic Nutrient (DIN) analysis and one whole water (unfiltered) sample for Total Nutrient (TNTP) analysis.
    a. Gather 26 clean nutrient bottles (translucent HDPE with Polypropylene screw caps, Fisher ID 02-895A). Inspect all bottles for damage; bottles should be full of DI water. A bottle that is less than full has a high probability of having a leak.
    b. Use one color of tape for DIN and another color for TNTP (makes them easier to separate later and prevents mistakes). Put a ring of tape around each bottle about halfway up ensuring that the tape ring goes at least 1.5 times around the bottle so it won't come off when the bottle gets wet.
    c. Label each bottle with permanent marker with the following information: the cruise date (mm/dd/yyyy), the sample type (DIN or TNTP) and the station number
    d. The station numbers we use are 1, 2, 3, 4, 5, 6, 8, 9, 11, 12, 14, 16, and MHB (Mount Hope Bay). These station numbers are chosen to line up

with a previous study, but we don't use all of the sampling sites from the previous study, so some numbers are missing.

3. Sample bottles are 1L opaque HDPE narrow mouth bottles (Fisher part No.:312004-0032). Samples bottles are stored full of DI water in a cooler in the hallway. During collection, samples are stored on ice before being returned to the lab for filtration

Filtering Nutrient Samples

1. Take the first station's brown sample bottle and invert it 5 times.
2. Place the tube attachment of the syringe inside the brown bottle. Do not remove it until you switch bottles.
3. Connect the syringe to the tube and draw about 20 ml of water into the syringe and rinse it. Repeat this two more times.
4. Empty the DI water from the first station's corresponding clear sample bottles. Place a nutrient filter on top of your clear DIN sample bottle for that station. Draw a full pull of water into the syringe and filter 1/3 of the contents of the syringe into the clear DIN sample bottle. Shake the water in the bottle and pour it out. Repeat this 2 more times with the remainder of the water in the syringe.
5. Draw a full pull of water into the syringe and filter into the clear DIN sample bottle to fill it until about where the top of the tape is being sure to leave room for the water to expand as it freezes.
6. **TNTP samples are NOT filtered**. They are rinsed 3 times with water directly from the brown sample bottle and filled with water directly from the brown

sample bottle. Once again, they are filled to the top of the tape leaving enough room to allow for expansion during freezing.

7. Repeat steps 1 – 6 for the 12 other stations, but change your nutrient filter between each station.
8. If the nutrient filter is severely leaky, first try tightening the cap. If that fails, get a new filter. If you run out of filters, you can rinse and rebuild one of the ones that leaked, making sure to rinse it thoroughly with DI water **then with sample** before proceeding.
9. The analyst for the ASTORIA nutrient analyzer needs to know the salinity of the samples being run (i.e. if it is below about 20 ppt). After you have finished filtering, take a small amount of water (~0.5 ml) from the brown station 12 bottle (the furthest north station) with a pipette and place it on the refractometer to measure salinity. If the salinity is below 20 ppt, take a small piece of tape, write the salinity on it, and place it over the top of the DIN and TNTP vials from that station. Continue measuring salinity at downbay stations until one of them is >20ppt. The downbay station order is: 12, 11, and 9, then 8, 14, and MHB, then 6 and 8. If station 9 is below 20 ppt, measure 8, 14**, and** MHB, if one of those is below 20 ppt, measure 6 **and** 8. Typically, either all of the stations will be OK, or only station 12 will be below 20 ppt. If you believe more than station 12 and 11 to be below 20 ppt, find someone to double check and make sure you're using the refractometer right before proceeding.

Storing Nutrient Samples

1. All samples are placed into the nutrient freezer. Check all caps for tightness before placing in freezer.
2. Put DIN samples on the DIN shelf, TNTP samples on the TNTP shelf, and buoy samples, if applicable, on the door.
3. Log samples (quantity and date) put in the freezer on the door so that if a station was not sampled on a given cruise day someone doesn't spend 30 minutes going through the freezer looking for the missing sample.

Cleaning Filters

1. Take apart the nutrient filter apparatus.
2. Throw away the polycarbonate filter.
3. Rinse all plastic pieces and O ring 3 times.
4. Place all the plastic pieces into a 10% hydrochloric acid bath.
5. Place the O ring into a beaker with DI water as it will disintegrate in the acid bath.
6. Take all plastic pieces out of the acid bath after at least 24 hours. Rinse 3 times with DI water and set out to dry in a clean place.
7. ONCE DRY, PUT AWAY. DO NOT LEAVE INDEFINITELY ON THE COUNTER!!!

II. SAMPLE PREPARATION

Preparing DIN Samples: DIN samples do not require any special treatment prior to analysis. DIN samples remain frozen until the day of analysis.

Preparing TNTP Samples:

1. Recrystallizing Potassium Persulfate: Be sure to make recrystallized potassium persulfate before the day you need to do the TNTP extraction. The glassware, thermometer, and funnel need to be washed in an acid bath, rinsed with ultrapure DI water, and dried in a drying oven prior to TNTP extraction.
    a. Dissolve 48 g of potassium persulfate in 300 ml ultrapure DI water in a 1500 ml Erlenmeyer flask. You can double this recipe if desired.
    b. Heat to 65°C, hand stirring and swirling until all potassium persulfate has dissolved. While solution is heating, create an ice bath large enough to fit the flask.
    c. Continue to heat with swirling and bring temperature up to 75°C.
    d. Remove and place immediately in the ice bath. Cool solution to <10C. Crystals should form.
    e. Using a 3” Buckner funnel with a #42 Whatman Qualitative filter cut down to size, first rinse the filter through the funnel with ultrapure DI water then pour potassium persulfate crystals and remaining liquid through the funnel with vacuum (5 psi) to draw off the water. Note: When doubling recipe the solid will almost completely fill the funnel.
    f. Scoop the remaining crystals out of the beaker, rinsing with a small amount of ultrapure DI water if necessary (adding water reduces the return).
    g. Dry and fluff crystals in funnel for 5min.

h. Transfer to clean dish and put in desiccator or 60°C oven until dry (approx. 24 hrs/overnight in oven).

i. Store crystals in a dessicator jar to prevent accumulation of moisture. Ideally, add a small dish of 36N sulfuric acid and a small dish of potassium permanganate to the dessicator jar to scavenge any impurities out of the air.

NOTE: To maximize purity, potassium persulfate should be recrystallized a minimum of 2 times, preferably 3 times. After each recrystallization, estimate the percent return and reduce the amount of water added when starting the process proportionately, otherwise it may be difficult to get all the crystals out of solution with an ice bath. A saltwater ice bath can ameliorate this issue somewhat. Ideally, there should be just enough water in the flask so that the last of the crystals dissolve right at 75°C.

2. Cleaning and Drying TNTP Vials

   a. Create a water bath and begin heating to 80°C while the vials are prepared as it takes a while to get to the correct temperature.

   b. Set up TNTP vials into racks and get the beaker of TNTP vial caps. For each run you will need enough vials for your samples, extraction standards (2N & 2P), and blanks (2). It is always a good idea to do extra vials than you will need to allow for breakage and extras.

   c. Make up a solution of potassium persulfate in a volumetric flask to be used for cleaning.

      Recipe: 25 g potassium persulfate (does not have to be recrystallized)

15 g boric acid

175 ml 1M NaOH

Fill to 500 ml with ultrapure DI water

d. Pour some of the potassium persulfate solution into a clean beaker from which to pipette.
e. Pipette 3.5 ml of the potassium persulfate solution into each TNTP vial and screw on the cap. Keep any extra potassium persulfate solution in a bottle for future cleanings.
f. Place TNTP vials in the water bath when it reaches 80°C.
g. Bring water bath to a boil (100°C). This is a critical time for the TNTP vials so make sure the vials are put in when the water bath is 80°C and then bring it up to a boil. It is during this time period that crucial chemical reactions occur so it is best not to mess it up.
h. Start a timer for 15 minutes when the water comes to a boil.
i. At the end of 15 minutes, remove the vials from the water bath and let them cool to room temperature.
j. Empty the contents into a hazardous waste receptacle and rinse vials with DI water.
k. Turn the TNTP vials upside down in the rack and place them in the drying oven.
l. Place the caps in a 10% hydrochloric acid bath for about 8 hours. After acid washing, check the integrity of the caps and discard any that are showing excessive signs of wear.

3. Extracting TNTP Samples
    a. Remove TNTP samples out of the nutrient freezer and thaw them. This can be accomplished by placing the sample bottles in a **warm** water bath or by running them under warm water. Be sure to check the tightness of the sample bottle caps, ensure there are no cracks in the sample bottle, and not to submerge the bottles to prevent contamination of the sample. Make sure they are completely thawed before proceeding. After thawing, rinse the sample bottles with DI water and dry them before pouring sample out. Even a single drop of tap water can severely contaminate a sample.
    b. Create a water bath and begin heating to 80°C while the samples are prepared as it takes a while to get to the correct temperature.
    c. Make up a solution of potassium persulfate in a volumetric flask to be used for extraction.

       Recipe: 12.5 g **recrystallized** potassium persulfate
       7.5 g boric acid
       87.5 ml 1M NaOH
       Fill to 250 ml with ultrapure DI water

NOTE: This recipe is sufficient for nearly 100 samples. For smaller batches, it can be reduced proportionately.

Useful variation: 10 g **recrystallized** potassium persulfate
6 g boric acid
70 ml 1M NaOH

Fill to 200 ml with ultrapure DI water

d. Mix on a heated stir place on medium heat until potassium persulfate has dissolved completely.
e. Remove clean TNTP vials from the oven and gather **completely** dry acid washed caps. Rack TNTP vials and be sure to write down which vials correspond to which samples.
f. Take the thawed sample bottle and gently agitate to mix the sample. Unscrew the cap and wipe the neck of the bottle with a kimwipe to remove any remaining DI water. This is to ensure the sample is not contaminated.
g. Fill TNTP vials with 22.5 ml of sample (up to the etched line).
h. Pour recrystallized potassium persulfate solution into a clean beaker and pipette 2.5 ml of the recrystallized potassium persulfate solution into each vial and screw on the cap.
i. FOR BLANKS: fill 2 additional vials to the line with artificial seawater (or ultrapure DI water for freshwater analysis), add 2.5 ml of the recrystallized potassium persulfate solution, and screw on the cap.
j. FOR EXTRACTION STANDARDS: reserve 2 vials each for phosphorus and nitrogen extraction standards.
    i. <u>Phosphorus</u>: Add 200 µl of 1000 µM fructose 1, 6-diphosphate stock to a 100 ml volumetric flask and fill to the line with artificial seawater (or ultrapure DI water for freshwater analysis). Mix the solution then add 22.5 ml (to the etched line)

    to the corresponding P standard TNTP vials. Pipette 2.5 ml of the recrystallized potassium persulfate into the vials and screw on the caps. This makes a 2mM extraction standard check

    ii. Nitrogen: Add 2 ml of 1000 µM glycine stock to a 100 ml volumetric flask and fill to the line with artificial seawater (or ultrapure DI water for freshwater analysis). Mix the solution then add 22.5 ml (to the etched line) to the corresponding N standard TNTP vials. Pipette 2.5 ml of the recrystallized potassium persulfate into the vials and screw on the caps. This makes a 20mM extraction standard check

  k. Keep any extra recrystallized potassium persulfate solution in a bottle to use for TNTP vial cleanings in the future.
  l. Place TNTP vials in the water bath when it reaches 80°C
  m. Bring the water bath to a boil (100°C). This is a critical time for the samples so make sure they are put in when the water bath is 80°C and then bring it up to a boil.
  n. Start the timer for 30 minutes when the water comes to a boil.
  o. After 30 minutes have passed, turn off the heat for the water bath and let the TNTP vials cool gradually to room temperature.
  p. Remove from the water bath and tighten caps. Samples are stable at room temperature for at least 30 days after extraction.

4. Vial Care between Extractions
  a. Discard extra sample in waste container.

   b. Triple rinse caps and vials with DI water.
   c. Vials should be acid washed after every usage.

III. SAMPLE ANALYSIS

Prior to Run Day:

1. Make sure the DI water pump is functioning properly. It should read about 18.
2. Check all chemicals used to make nutrient reagents to ensure they have not gone bad. If any have gone bad, remake them. The most common chemicals to go bad are:
   a. Stock molybdic acid- commonly precipitates along walls of bottle, check bottle carefully the day before, generally cannot be re-heated to get back into solution.
   b. Ammonium molybdate- commonly precipitates along walls of bottle, check bottle carefully the day before, generally cannot be re-heated to get back into solution.
   c. SLS: If crystals have precipitated, place on heat and stir until they go back into solution.
3. The chemicals used to remake nutrient chemicals can be found in Table A-2:
4. Nutrient chemicals are made as can be found in Table A-3:

On the Run Day:

1. Starting the Machine
   a. Dump, rinse, and refill water reservoir with ultrapure DI water and place lines in bottle.
   b. Latch all the platens down on the machine.

- c. Lock the auxiliary pump in the back and turn it on.
- d. Open the nitrogen gas.
- e. Turn on the surge protector.
- f. Run machine for 7 minutes.

2. Rinsing the Machine

NOTE: Rinse line goes through all of these steps, but not the coolant reservoir line, which always stays in water.

- a. Run the machine on water for 7 minutes and check for a regular bubble pattern before proceeding to the next step.
- b. Run the machine on 10% hydrochloric acid for 5 minutes.
- c. Run the machine on ultrapure DI water for 5 minutes.
- d. Run the machine on Chemwash for 5 minutes.
  - i. While the machine is running on Chemwash, turn on the computer, open FasPac II, and create a new run.
  - ii. Click the hand icon to connect the computer to the machine. A green light indicates they are connected.
  - iii. Fill SR 20 with Chemwash and under “System”, click “Clean System”. When done cleaning (sampler returns to original position), clean again.
- e. Once cleaned, place machine in start up/shut down mode until ready for reagents (usually exceeds the 7 min needed to be online).

3. Conditioning the Cd Column

NOTE: Remember to only put the Cd column line on when the reagents/standards/samples go on. No water can go through the Cd column.

   a. Cd column is online when the colored lines are hooked up together (Green-Green and Red-Red).
   b. To clean, first hook up green end to waste tube.
      i. Inject 10 ml ultrapure DI water into red end of Cd column.
      ii. Inject 10 ml 2% $CuSO_4$ over 30 seconds. **If you push through too slow the column will clog, but if you push too fast the column won't clean/react with chemicals inside the column.**
      iii. **QUICKLY/AS FAST AS RESAONABLY POSSIBLE** put buffer through the column. Buffer should be injected both forward and backward. This requires you to switch the waste end to the red end.

4. Make Reagents (Table A-4)
   a. Rinse all reagent bottles with DI water.
   b. Reagents should be made while machine is being rinsed and the Cd column is being cleaned.
   c. Recipe quantities are for an 8 hour run day. Typically if you plan to run longer, multiply the NED, SAN, Ammonium complexing reagent and silicate molybdate and Tartaric reagents by 1.5.
   d. NOTE: the Stannous chloride and phenol reagents tend to be marginally stable. On a good day, you can get 12 hours out of them,

but you need to watch them vigilantly for decay in amplitude of your check standard after about 6-8 hours.

NOTE: Astoria Pacific calls for the use of an imidazole buffer for this analyte to preserve Cd column life. We found this buffer to produce undesirable results in saltwater use, and have defaulted back to the Ammonium Chloride buffer used in the Technicon method. However, to improve column life, we always flush and store the column filled with the imidazole buffer after each run (see Table A-4 for recipe).

NOTE: While the Ammonium Chloride buffer works well in most cases, for extremely high values, such as porewater samples, or samples with pH significantly different from 8, it isn't strong enough and can severely damage the column. In these cases we have had good luck with a buffer composed of 85g ammonium chloride (NH_4Cl), and .1g EDTA mixed to a total volume of 900 ml then adjusted with Sodium Hydroxide (NaOH) to a pH of 8.5.

5. Put all Reagents Online
    a. Before putting reagents online, turn on the heat baths.
    b. For silicate, the stannous chloride reagent goes on after (5 min delay) the molybdate and tartaric acid reagents.
    c. Once the reagents have been online for a few minutes, put the Cd column online.
    d. At this time, switch the rinse from ultrapure to ASW.
    e. If you have not done so already, initialize FasPac and connect to the instrument. Display all signals and Zero all signals so you can see your

    baseline. Expect a baseline jump when the machine goes to reagents/ASW. Sometimes there is also bubble introduction into the flow cells from this process. This is the first culprit if you do not have stable baselines. Once all baselines are stable, proceed to step 7 (step 6 is done concurrently to step 5)

    f. The Ammonium channel tends to produce a lot of crystalline precipitate which partially obscures the flowcell and impairs baseline detection when it first goes onto ASW carrier. This USUALLY resolves in about 15-20 minutes, sometimes it takes as long as 30 minutes. It is not really well known why this takes so long to stabilize. It has been empirically shown that vigilant watching, cursing, yelling, and threats extend this time exponentially, while soft music, encouragement, and simply walking away to check your e-mail tend to shorten it.

6. Make Standards (Table A-5)
    a. Rinse standard bottles with DI water
    b. Standards should be made while the machine is being put on the reagents. Standards should be made as follows (values in ml of 1000mM stock added to each 100ml **plastic** volumetric)
    c. Standard bottles are then filled with artificial seawater (ultrapure DI water for freshwater samples) and inverted several times to mix.
7. Put Standards Online

The instrument uses specific ‘identifiers’ to recognize specific types of samples, for a complete list of the available identifiers, see the FasPac manual. The identifiers commonly used are described here, it is important to note that the format is case sensitive.

SYNC = Synch standard. Used to line up the timing on different channels and account for differences in transit time. Typically a high standard with all analytes being run in it.

W= baseline check. A water (ASW or ultrapure DI) sample for which you want the instrument to reset the baseline.

w= A blank for which you do NOT want the instrument to reset the baseline to zero, used often when you’re going from a high standard to a low standard and want to eliminate the possibility of carryover. NOTE: the difference in case between w and W has a huge difference in how the machine interprets.

CO= Carryover check. A water (ASW or DI) sample placed immediately following a high standard. This preprogrammed identifier calculates the percentage of the amplitude of the previous peak which ‘carries over’ into the next peak. If automatic carryover correction is enabled, it will use this value to correct subsequent high samples followed by low samples

NOX%= preprogrammed identifier for cadmium efficiency check. This is a high nitrite sample (red 4) placed immediately after a high nitrate (black 4) sample. The instrument calculates the percentage return on the cad column and (if enabled) can perform a range of actions if this value is outside of an acceptable parameterization (e.g. 95%)

C1, C2, …CX = preprogrammed identifiers for calibrants. In our case, C1 is a zero standard (ASW of the appropriate matrix or Ultrapure DI), C2-C6 are the black (mixed) standards in order, and C7-C10 are the red (nitrite) standards in order. The instrument reuses C1 as the zero standard for both curves. TNTP uses a single mixed curve. The values of the calibrants can of course be changed in the System menu. See the FasPac Manual for more details here.

The racking order for the standards with # of reps in parentheses () can be found in Table A-6.

8. Check Calibrants
    a. The software options for monitoring check calibrants are severely buggy, and my recommendation is to turn them off and manually monitor your check calibrants. Should you choose to enable calibrant checks, be aware that the instrument will occasionally restart a run with no warning or explanation. Without extreme vigilance, this will cause the instrument to draw the first sample tube dry and introduce air into the lines, which will cause FAR more problems for you than monitoring your own check calibrants.
    b. For DIN runs, typically the CC1 (check cal 1) identifier is used for the mixed high standard (black 4) and the CC2 identifier is used for a cad check (red 4). CC1 is racked in slot 1:1 and CC2 in slot 1:2 with the initialization marker (right click to set) set on 1:1. I allow the instrument to set the check cal frequency (20) and wash frequency (20 in the system menu, but uncheck 'monitor check calibrants'. This

means that you must manually inspect the run to make sure your check calibrants are within bounds.

   c. For TN runs, I use the same identifiers for check calibrants, but rack them in the Standards rack (typically in the open SR17 and SR18 spots). This allows the analyst to easily line up the sample ID's from the extraction sheet with the sample ID's in the sample table, minimizing the chance for confusion and a sample to get mis-racked. If you do this, you must reset the initialization block marker (right click) onto SR17 (if not already done) and make sure you set the 'first CC row' to SR17 in the system menu or the instrument will malfunction.
   d. Because the instrument takes up about 2-3 ml per sample, you can get about 4 checks from a 16 ml vial (the bottom 2ml are unusable- the needle doesn't go that far down) before it needs to be refilled. Monitor this closely, as if these vials run dry, you will inject air into the instrument, which puts unnecessary wear on the cd column and can ruin your bubble pattern and your day very easily.

9. Preparing and Racking Samples
   a. If DIN samples are being run, begin thawing samples under warm water. Be sure all caps are tight before thawing, and that the water does not come up to the caps. A single drop of tap water can severely contaminate a sample. Once thawed, rinse in DI water and dry thoroughly to remove any tapwater from the sample.

    b. If TNTP samples are being run, make sure extracted sample tubes are in the correct order according to the sample sheet.
    c. Once DIN samples are thawed, rinse in DI water, dry bottles completely, and order them.
    d. Place an appropriate amount of tubes in the plastic racks.
    e. Gently mix samples (DIN or TNTP) and begin pouring into tubes in order going down each column working from left to right.
    f. The machine batch downloads data from the sample table (in FasPac) to the instrument every 4 samples. This means that you must have the sample table filled in at least 4 samples ahead of where the sampler is sampling at all times (or the instrument will malfunction). The transit time for the longest line (ammonium) is about 7 minutes. The default sample time is 35 sec. with a 55 sec. wash, so this means that you must be racked at least 15 replicates (5 samples in triplicate) ahead of what you see on the screen for results, or you will crash the software.

10. During the Run
    a. Make sure you are either manually inserting, or using FasPAC to control autowashes (capital W's if doing it manually) to monitor baseline and check standards to monitor colorimetric response and Cd column efficiency.
    b. If Cd column efficiency falls well below 95%, you can pause the run (use the PAUSE command), make sure you put it into the sample table at least 4 samples ahead of where the machine is presently sampling),

reactivate the column and continue, or abort the run at the analyst's discretion

c. DILUTIONS AND RERUNS: If you have offscale samples that need to be diluted and re-run, or other problems (e.g. bubbles) cause you to lose a sample, you can add it to the end of the sample run. If you are planning on doing this, make sure you either get the re-runs entered into the end of sample table before the machine gets close to the end of the run (see 9F above) or put a string of 5-6 waters at the end of the sample table, which will allow time for all of the samples to get through the flowpath, and for the analyst to figure out which samples require dilution and get them into the sample table. Make sure you put them in the sample table FIRST, then dilute the sample and put it in the rack. The FasPAC sample table has a column for 'total dilution' which, if you use it to enter your dilution factor, will automatically calculate the correct concentration. We have found the various colorimetries to be relatively linear up to about 100 mM, thus, while samples still need to be rerun if they are more than about 120% of the high standard, the concentration of the original sample can be used to estimate the dilution factor (e.g. if your curve goes from 0-8mM, and the original sample runs through at 40mM, a 10X dilution is ideal.) Dilutions can be done to a total volume of 10ml (to simplify math) and the instrument can still get 3 replicates reliably. We have not had much luck with dilutions past about 20X. In these cases, the recommended procedure would be to refreeze the sample and rerun it with a higher standard curve.

11. Shutting Down the Machine
    a. Take the Cd column offline
    b. Take the stannous chloride offline and put line in start up/shut down
    c. Turn off the heat baths
    d. Flush the Cd column with Imidazole buffer and store it closed (attach inflow line to outflow line) and filled with Imidazole buffer.
    e. Move the rest of the reagents to start up/shut down **EXCEPT** for the tartaric acid and molybdate reagents.
    f. Take the tartaric acid and molybdate reagents off after 5 minutes.
    g. Let the machine run on start up/shut down for about 7 minutes.
    h. Run machine on 10% HCl for 7 minutes.
    i. Run machine on ultrapure DI water for 7 minutes.
    j. Run machine on Chemwash for 7 minutes.
    k. Run machine on water for 7 minutes.
    l. Run machine dry.

NOTE: If running again in the near future, Steps H-L are unnecessary. Run the machine on start-up/shut down solution for about 15 minutes, followed by water for 7 minutes and shut it down.

    m. Detach platens on main and accessory pump
    n. Close nitrogen pillow
    o. Turn off main power switch

p. Place catch cup under sampler incase water backflows

q. Leave all reagent lines in the water beaker (if running soon) or a clean dry covered container (if pumped dry)

r. OPTIONAL: detatch all pump tubes from the right side stretcher to take the tension off the tubes. This can extend their life, especially if you're not planning on running again soon.

12. Run Day Troubleshooting

NOTE: Use this section like a dichotomous key. Find the problem you are having, and drill down. I've organized by most likely to least likely issues for each situation.

a. UNSTABLE BASELINE
    i. Check for bubbles in the flow cells
        1. Clear bubbles from flowcells
    ii. Check for good bubble pattern, capsule shaped bubbles at even intervals. Approximately even ratio of bubble/sample
        1. Make sure all reagents in the problematic sample are delivering (remove straw from solution, introduce a bubble and follow it through the system)
        2. Make sure there's not a leak or a fitting that's allowing air into the system (evident from jerky bubble motion)
        3. Try turning up the accessory pump a little to deliver more flow.
        4. Consider replacing the offending pump tube

            5. Call technical support

        iii. Is there junk (crystals) in the ammonia flow cell

            1. Wait 30 minutes and try again
            2. Wait 15 more minutes and try again
            3. Test pH coming out of heat bath, should be about 9-10
                a. Remake complexing reagent and adjust pH to 10

        iv. Are all of the filters in the flowcells in good condition?

            1. If not, replace them. Refer to brown maintenance manual or call technical support for assistance.

        v. Walk away for 15 minutes

            1. Sometimes the machine just takes a while to figure itself out in the morning. If this fails, proceed to vi.

        vi. Call technical support

    b. NO/INSUFFICIENT SYNC PEAK

        i. Are all of the lights OK (unlikely but easy to fix)?

            1. Go to 'system> show light %' and compare light percentages to recent runs to make sure the lights are still good
                a. If it's too high or low, you can loosen the set screw and adjust the position of the light to get it within nominal range
                b. If you still have no/insufficient light, consider replacing the fiber optic

- ii. Were all reagents made correctly?
  1. Is phosphorus reagent (if PO4 is the problem) a nice straw yellow?
     - a. Remake (once), if that fails, proceed to b.iii
  2. Did you reactivate the Cd column this AM (if NO2 is good but NO3 is bad)
     - a. If no, do that now, if yes, go to b.iv
- iii. Is it a flow path problem?
  1. Are all reagents on the offending channel drawing appropriately (see a.ii.1 above)
     - a. If not, check for a clog in the straw or one of the fittings
  2. Is sample being delivered efficiently
     - a. Look for backflow in offending lines, introduce a bubble by removing the sample needle from the washpot and follow it through the system
     - b. The flowpath of sample is NH4>SIO4>NO2>NO3>PO4. If the interruption is in line with this (e.g. you have NH4, SIO4, and NO2 but no NO3 or PO4) this is the likely problem, inspect the flowpath for leaks and clogs, clean all metal fittings, replace if

necessary. If not (e.g. you have all but NH4 or SiO4) this is not the problem.

iv. Is one of the reagents bad?

    1. Check for precipitate in reagent bottle. As above, the most likely offenders for this are (in order)

        a. Either of the molybdate reagents (silicate or Phosphate)
        b. The Citric acid (phosphate)
        c. The complexing reagent (ammonium)
        d. Not likely a reagent problem (NO2, NO3)

v. Is the Cd column bad (NO2, NO3)

    1. Check the pH of the sample coming out of the column. This can be problematic for anoxic, very high concentration, or poorly buffered (freshwater) samples. It should be around 2. If not (usually too low), adjust the buffer so the pH is around 2 or slightly above.
    2. Reactivate the column

        a. First do the daily reactivation again (water, copper, buffer). If that fails:
        b. Do the more aggressive reconditioning in the brown troubleshooting manual. If that fails

            c. Consider replacing the column, especially if it's over 200 hours old. Remember to activate and 'burn in' a new column before use

    vi. Are all the stocks/standards good?

        1. They're good for a year, and don't tend to go 'all the way' bad. If you're off by 10% or so, consider remaking your standards, or stocks, if they're old
        2. If you're not getting any peak at all, this is unlikely to be the problem. Attempt all other troubleshooting methods (e.g. flowpath or reagent issues) before proceeding to 3 below.
        3. If you're not getting any peak at all, and the stocks are appropriate age, consider attempting a benchtop titration to see if you get any color (use straight 1000uM stock, you're looking for blue for PO4 and SiO4, and pink for nitrogen species). If not, remake the stock.

c. FLOWPATH/BUBBLE PATTERN ISSUES

    i. Consider a.ii and b.ii above

    ii. Can you trace the problem to a specific line?

        1. Check all reagents to that line to make sure they're delivering
        2. Make sure the Nitrogen pillow is open and full

            3. Make sure the air line pump tube (for lines not on the N2 pillow) are not obstructed and are in good condition
        iii. Are all/multiple lines malfunctioning
            1. In this case it's probably a sample line issue, see B.iii.2 above
            2. Check to make sure the needle is properly positioned in the washpot and not drawing up too many bubbles
                a. Adjust the needle, or if there's too much air in the washpot, try turning up the accessory pump a little
                b. Make sure none of the lines going into or out of the accessory pump are kinked or trapped under anything, even a small restriction can be deadly here.
            3. Walk away for 15 minutes and see if the problem persists
                a. Seriously, sometimes the machine just takes a while to sort itself out.
                b. Call tech support.
    d. CADMIUM COLUMN ISSUES
        i. Did you remember to activate it this morning?
            1. If not, activate it and start over
        ii. Is it clogging, tearing up bubbles excessively, or back flowing?

        1. Flush extensively with imidazole buffer in BOTH directions
        2. Try cleaning out the edges of the column with the paperclip probe tool (a piece of 0.020 wire rubber banded to a ½ paperclip)
        3. Check/replace the PE tubing coming in and out of the column, the fittings which link that tubing to the column (0.90 PE with 0.33 silicone sheathed inside) and the pins that connect it to the PE tubing on the system. Clean and replace if necessary
        4. Perform a more extensive cleaning procedure from the brown troubleshooting manual
        5. If it's old, consider replacing it. If not, call tech support

    iii. Is the efficiency dropping off rapidly?
        1. Try the harsh reactivation step in the brown binder
        2. Test the pH of the sample coming out of the column to make sure it's 2ish.
        3. If you are running porewater, brackish samples, potentially anoxic samples, or potentially very high concentration seawater samples, switch buffers to the ammonium chloride/EDTA buffer and see if that helps
        4. Consider replacing the column if old, otherwise call tech support

Machine Maintenance

1. 50 hour preventative maintenance. (NOTE: This needs to be performed EVERY 50 hours, sometimes a little earlier, sometimes a little later. Usually you will notice pump tubes starting to go bad. If you can replace 1 and get a run day in, go for it, if more than 1 is bad, you should probably scrap the run day and do the maintenance, because it's likely that others will go bad during the run day and ruin your data
    a. Run warm 20% contrad (heat to 65C in water bath) through all lines except sample rinse line and ammonium waterbath line to clean the glass coils and flowcells.
    b. Run water through the system for 30 minutes to flush the contrad
    c. Clean the platens by removing them and cleaning them with ethanol then with lubricant (tri Flow silicone lubricant, ordered from Astoria)
    d. Clean the rollers by undoing one side of the tubes for each roller and holding a Kimwipe with ethanol over them as they move. Repeat with lubricant.
    e. Change all pump tubes.
        i. Be sure to trim pump tubes to appropriate length to avoid (minimize) the massive tangle of tubes. Trim with the GREEN or YELLOW cutters or a razor blade.
        ii. The pump tubes can be dipped in ethanol to ease putting them back on, and can also be stretched a little with the probe tool or WHITE pliers.

f. Change all Poly Flow (bluish tubing)
    i. You have to use Astoria brand poly flow. You cannot substitute generic .034 PTFE tubing (I tried, I know it's much cheaper, trust me)
    ii. It must be trimmed with a razor or guillotine, NO cutters
    iii. You can ease replacement by priming the tip with the probe tool
    iv. If the tube kinks, you need to trim it off at the kink and try again. For this reason, it's often a wise idea to cut the tube a bit longer than you think you need!

g. Clean autosampler
    i. Clean any salt stains and wipe down the sampler with water or ethanol.
    ii. Use Tri-Flow to oil the sample arm gears and the crossbeam

h. Clean instrument
    i. Inspect under cartridges for leaks
    ii. Inspect flowcells, 'coffins' and sliders. Oil sliders with tri-flow making sure not to get any oil on the flowcell!
    iii. Wipe off all surfaces with water and/or ethanol to clean any spills
    iv. Inspect all glass/glass junctions, fittings etc. for cracking, wear, or damage

i. Rotate platens to ensure they wear evenly

j.

2. TWICE PER YEAR
    a. Clean all reagent, rinse, and ASW bottles by filling partly with 10% bleach. Let sit for 30 minutes then dump and rinse with DI water. Fill bottle partly with 10% hydrochloric acid. Let sit for 30 minutes then dump and rinse with DI water.
    b. Change all PE (grey) tubing. (note, this can be done at discretion when it appears worn, stretched out, or overly stained, between once and twice per year)
        i. Make sure you use a razor or the YELLOW cutters to cut PE. You can use Astoria brand or generic 0.34 Polyethelene tubing
        ii. You can ease the replacement by dipping in ethanol, but try not to use the tool, this will only increase the frequency with which the tubes have to be changed
        iii. It doesn't matter if PE kinks (unlike PolyFlow)
    c. Carefully inspect all junctions and fittings, replace worn junctions, inspect and replace any worn, stained, or skuzzy reagent straws,
    d. Carefully remove and flush out flowcells with warm contrad then water to remove any accumulated sediment.
    e. Inspect (replace if worn/skuzzy) the coiled sample line. Typically this has a lifespan of about one year. Be sure to mark it's in service date
    f. Inspect all platens for excessive wear. Replace as necessary. Platens have a lifespan of 500-1000 hours depending on usage

g. Inspect all stocks, reagents, surfactants, and dry chemicals and replace any that have expired. Stocks and wet chemicals are good for 1 year, dry chemicals are good for 5 years. Surfactants vary. **BRIJ-35 (Astoria proprietary surfactant used for ammonia) actually does go bad, and has to be replaced if expired.** TX-10/100 seems to be more reliable.

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Table A-0-1. Autoanalytic methodologies and empirically determined EPA detection limits for each nutrient analyte.

Analyte	**Technicon Method (used 2006-2008)**	**Technicon MDL**	**Astoria Method (used 2009-present)**	**Astoria MDL**
Nitrite	Greiss Reaction (NH_4Cl buffered Napthyethelene/Sulfanilimide (NED/SAN)) (Strickland and Parsons 1968, Technicon 1972a, Fox 1979)	0.02 mM	Greiss reaction (Imidazole Buffered NED/SAN) (Strickland and Parsons 1968, Fox 1979, Astoria-Pacific 2005)	0.02 mM
Nitrate	Greiss reaction (NED/SAN w/packed cadmium reduction) (Strickland and Parsons 1968, Technicon 1972a)	0.2 mM	Greiss reaction (NED/SAN w/ open tubular cadmium reduction) (Strickland and Parsons 1968, Astoria-Pacific 2005, Scott et al. 2005)	0.1 mM
Phosphate	Heteropoly Blue (molybdic+ascorbic) (Technicon 1971, Hager et al. 1972, EPA 1983c)	0.12 mM	Heteropoly Blue (molybdic + ascorbic acid) (EPA 1983c, Scott et al. 2005)	0.06 mM
Ammonia	Berthelot Indophenol blue (crystalline phenol+hypochlorite) (Solorzano 1969, Technicon 1973, EPA 1983a)	0.1 mM	Modified Berthelot (liquid phenol, hypochlorite, tartarate) (Solorzano 1969, Scott et al. 2005, Schmidt and Clement 2009)	0.05 mM
Silica	Silico-heteropoly blue (ascorbic, oxalic, molybdic) (Brewer and RIley 1966, Technicon 1972b)	0.06 mM	Silico-heteropoly blue (molybdic, tartaric, stannous chloride) (Sakamoto et al. 1990, Scott et al. 2005)	0.08 mM
Total Nitrogen	Alkaline Persulfate Oxidation + Greiss reaction (as above) (Technicon 1972a, Solorzano and Sharp 1980, Valderrama 1981)	1.1 mM	Alkaline Persulfate Oxidation + Greiss reaction (as above) (Solorzano and Sharp 1980, Valderrama 1981, Astoria-Pacific 2005)	0.5 mM
Total Phosphorus	Alkaline Persulfate Oxidation + Heteropoly Blue (as above) (Technicon 1971, Solorzano and Sharp 1980, Valderrama 1981)	0.12 mM	Alkaline Persulfate Oxidation + Heteropoly Blue (as above) (Solorzano and Sharp 1980, Valderrama 1981, Scott et al. 2005)	0.06 mM

Table A-2. Chemicals used to make nutrient chemical listed by name with its manufacturer and number.

Chemical	**Company**	**Number**
Ammonium Molybdate	Fisher	A674
5.0 Sulfuric Acid	Ricca	8325
Ascorbic Acid	Fisher	A61
Potassium Antimony Tartrate	Aldrich	244791
Sodium Dodecyl Sulfate	Fisher	BP166
Ammonium Chloride	Fisher	A661
N-1-Napthylethylenediamine	Sigma-Aldrich	222488
Sulfanilamide	Sigma	S9251
Ammonium Hydroxide	Fisher	A669
Sodium Hydroxide	Fisher	S318
Sodium Hypochlorite Solution	Fisher	SS290
Sodium Citrate	Fisher	S279
Potassium Sodium Tartrate	Fisher	S387
Sodium Nitroferricyanide	Fisher	S350
Phenol Liquid	Fisher	A931I
36N Sulfuric Acid	Fisher	
Tartaric Acid	Fisher	A314
Chloroform	MP	194002
Hydrochloric Acid	Fisher	A144C
Stannous Chloride	Fisher	T142
Sodium Chloride	Fisher	S271
Magnesium Chloride	Fisher	M63

Table A-3. Procedure for making nutrient chemicals.

NO_2 + NO_3 REAGENTS	
Ammonium Chloride	1. 30 g Ammonium Chloride/L ultrapure DI water 2. Mix with stir bar 3. Store on shelf
Napthylethylene (NED)	1. 1.0 g N-1-Napthylethylenediamine/L ultrapure DI water 2. Filter at 0.045 µm 3. Store in small chemical fridge
Sulfanilamide (SAN)	1. 10 g Sulfanilamide/L 10% HCl 2. Filter at 0.045 µm 3. Store in small chemical fridge
Ammonium Hydroxide	Straight from bottle in chemical fridge
NH_4 REAGENTS	
0.125N Sodium Hydroxide	1. 5.0 g Sodium Hydroxide/L ultrapure DI water 2. Mix with stir bar 3. Store on shelf
Sodium Hypochlorite Solution	Straight from bottle in small chemical fridge. Use fisher (or similar) brand hypochlorite. **Do not** use household bleach.
Ammonia Complexing Reagent	1. 56 g Sodium Citrate + 0.75 g hydroxide + 9.6 g Potassium Sodium Tartrate/500 mL ultrapure DI water 2. Filter at 0.045 µm 3. Store in small chemical fridge
Sodium Nitroferricyanide	1. 0.5 g Sodium Nitroferricyanide/L ultrapure DI water 2. Mix with stir bar 3. Store in small chemical fridge
Phenol liquid	Straight from bottle in enclosed section of chemical shelf
SiO_4 REAGENTS	
Stock Molybdic Acid	1. 10.8 g Ammonium Molybdate + 2.8 mL 36 N Sulfuric Acid/L ultrapure DI water. **Add Ammonium Molybdate and 700-800 mL ultrapure then add acid and remaining ultrapure DI water.** 2. Filter at 0.045 µm 3. Store in chemical fridge
Tartaric Acid	1. 200 g Tartaric Acid/L ultrapure 2. Add 2 drops of chloroform 3. Store in small chemical fridge
10% Hydrochloric Acid	1. 100 mL HCl/900 mL ultrapure DI water. **Fill with ultrapure then add acid.**

	2. Store on shelf
Stannous Chloride	1. 50 g Stannous Chloride + 50 mL HCl/250 mL ultrapure DI water. **Add some ultrapure DI water to Stannous Chloride then add acid and remaining ultrapure.** 2. Store in freezer
PO_4 REAGENTS	
Ammonium Molybdate	1. 40 g Ammonium Molybdate/L ultrapure DI water 2. Mix with stir bar 3. Filter at 0.045 µm 4. Store in chemical fridge
4.9N Sulfuric Acid	1. 20 mL ultrapure DI water filled to 1L with 5.0N Sulfuric Acid 2. Store on shelf
Ascorbic Acid	1. 54 g Ascorbic Acid/L ultrapure DI water 2. Store in small chemical fridge
Potassium Antimony Tartrate	1. 0.68 g Potassium Antimony Tartrate/500 mL ultrapure DI water 2. Mix with stir bar 3. Store on shelf
SLS	1. 15 g Sodium Dodecyl Sulfate/85 mL ultrapure. **Be sure to wear a mask.** 2. Mix with stir bar 3. Store on shelf
OTHER	
Artificial Seawater (28 psu)	1. 51 g Sodium Chloride + 16 g Magnesium Sulfate/2L ultrapure DI water 2. Mix with stir bar 3. Store on shelf 4. This makes 28PSU artificial seawater. For other salinities adjust accordingly
Start-up/Shut-down	Add the following surfactants to 250 ml ultrapure DI water: 1. Nitrate,Nitrite, TN: 3.5 ml TX-10 2. Phosphate (and TP) and Silicate: 10ml SLS 3. Ammonium: 1ml Brij-35 (30 drops)
ChemWash	1. 40 g Sodium Hydroxide/ 1L ultrapure DI water 2. Stir with stir bar 3. Add 4 ml Triton X100
Imidazole Buffer	1. 34 g Imidazole + 30 ml Stock Ammonium Chloride – Copper Sulfate/2L ultrapure DI water 2. Fill with about 1.5 ml ultrapure DI water 3. Add about 67 ml 10% Hydrochloric Acid 4. Fill to top with remaining ultrapure DI water

Table A-4. Procedure for making nutrient reagents.

$\mathbf{NH_4}$	
Hypochlorite	
60 ml	0.125 Sodium Hydroxide
1.2 ml	Sodium Hypochlorite Solution
Citrate/Tartrate/Hydroxide	
100 ml	Complexing Reagent
20 drops	Brij
Nitroferricyanide/Phenol	
60 ml	Sodium Nitroferricyanide
1.2 ml	Phenol liquid

$\mathbf{SiO_4}$	
Molybdate	
100 ml	Molybdic Acid Reagent
6.5 ml	SLS
Tartaric Acid	
60 ml	Tartaric Acid
Stannous Chloride	
60 ml	10% HCl
1.2 ml	Stannous Chloride

$\mathbf{NO_2+NO_3}$	
Ammonium Chloride Buffer*	
50 ml	Ammonium Chloride
100 ml	Ultrapure DI water
0.25 ml	Ammonium Hydroxide
1.33 ml	TX-10
NED	
60 ml	Napthyethylene (NED)
SAN	
80 ml	Sulfanilamide (SAN)
1.6 ml	TX-10

$\mathbf{PO_4}$	
ADD IN ORDER	
10 ml	Ammonium molybdate
33 ml	4.9 Sulfuric Acid
6.65 ml	Ascorbic Acid
6.65 ml	Potassium Antimony Tartrate
10 ml	Ultrapure DI water
FILTER @ 0.45 µM	
5.5 ml	SLS

Table A-5. Guide for making nutrient standards. All values are in ml.

DIN	**1**	**2**	**3**	**4**	**4.5**	**5**	**SYNC**
PO_4	0.2	0.4	0.6	0.8	1.0		0.8
SiO_4	0.5	1.0	1.5	2.0	4.0		2.0
NO_2+NO_3	0.3	0.6	0.9	1.2	1.8		0.0
NH_4	0.2	0.4	0.6	0.8	1.0		0.8
NO_2	0.1	0.2	0.3	0.4	0.6	1.2	1.2

TNTP	**1**	**2**	**3**	**4**	**4.5**	**5**	**SYNC**
NO_2+NO_3	1.2	2.4	3.6	4.8	7.2		
PO_4	0.2	0.4	0.6	0.8	1.0		0.8
NO_2						4.8	4.8

Table A-6. Racking order for nutrient standards with the number of reps in parentheses ().

Rack Position	DIN	TNTP
SR1	SYNC (1)	SYNC (1)
SR2	CO (1)	CO (1)
SR3	W (1)	W (1)
SR4	w (1)	w (1)
SR5	B4 (2)	B4 (2)
SR6	NOX% (2)	NOX% (2)
SR7	W (1)	W (1)
SR8	w (1)	w (1)
SR9	C1 (2)	C1 (2)
SR10	C2 (2)	C2 (2)
SR11	C3 (2)	C3 (2)
SR12	C4 (2)	C4 (2)
SR13	C5 (2)	C5 (2)
SR14	C6(2)	C6(2)
SR15	C7(2)	C7(2)
SR16	C8 (2)	W
SR17	C9(2)	CC1
SR18	C10(2)	CC2
SR19	W	

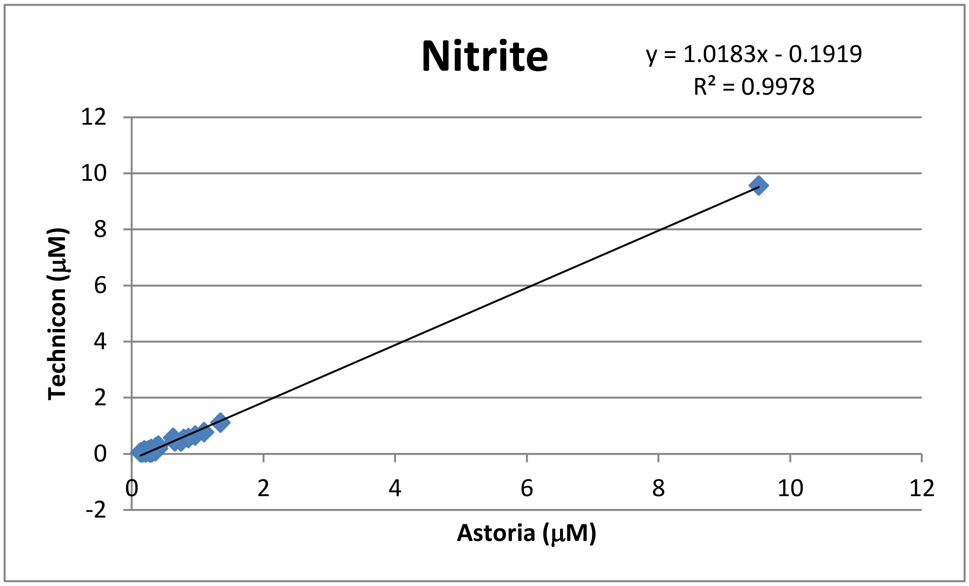


Figure A-1 Comparison of Nitrite values between Astoria and Technicon Autoanalyzers. Data run 11/30/2009

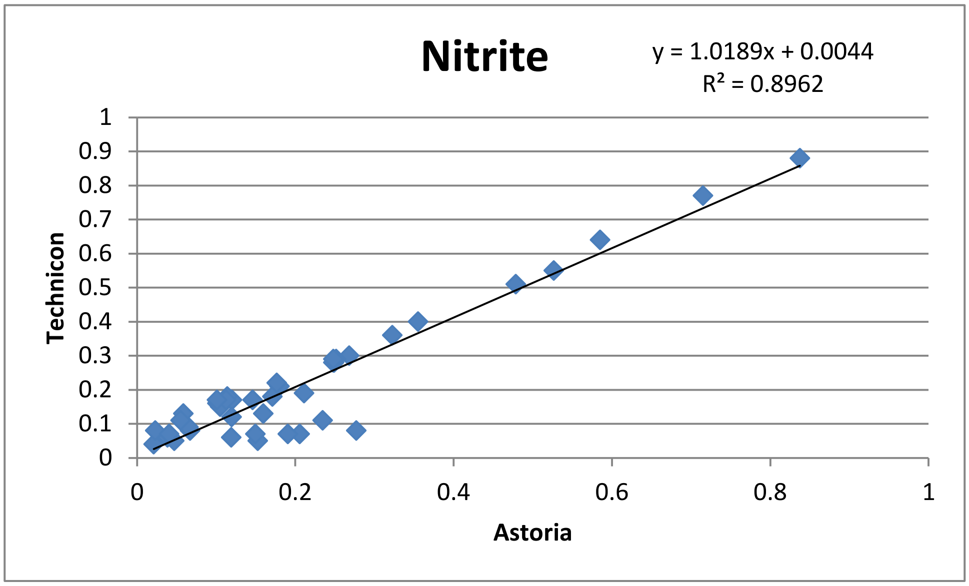


Figure A-2 Low range Nitrite comparison between Astoria and Technicon Autoanalyzer. Samples run 11/4/2009. While the overall relationship remains solid, the Astoria appears to be able to detect lower levels than the Technicon.

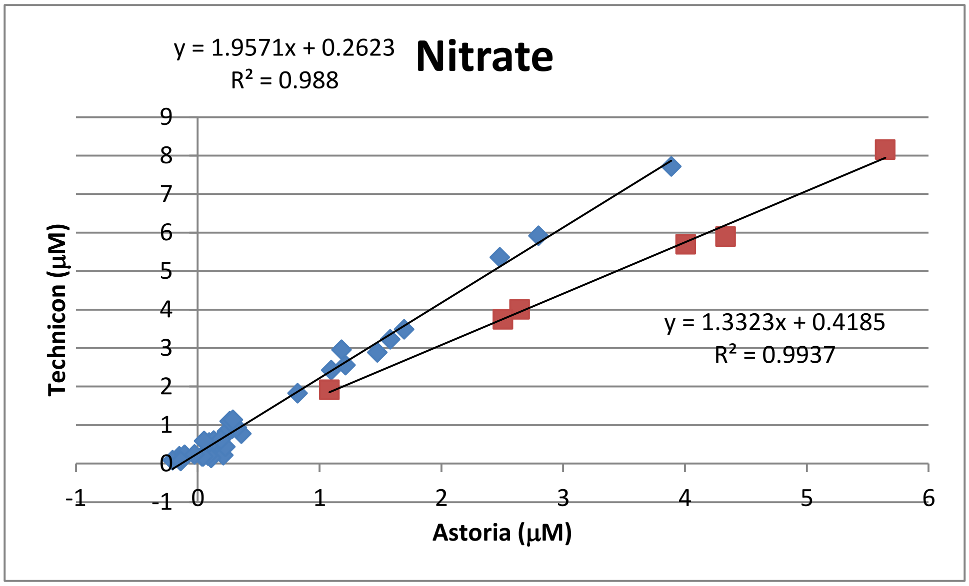


Figure A-3 Intercomparison of Nitrate data between Technicon and Astoria Autoanalyzers on 11/4/2009. Squares represent the first six samples run during this day, and diamonds represent remaining samples. High correlation on both 'sets' indicates a possible rapid shift in Cd reduction efficiency on the Technicon.

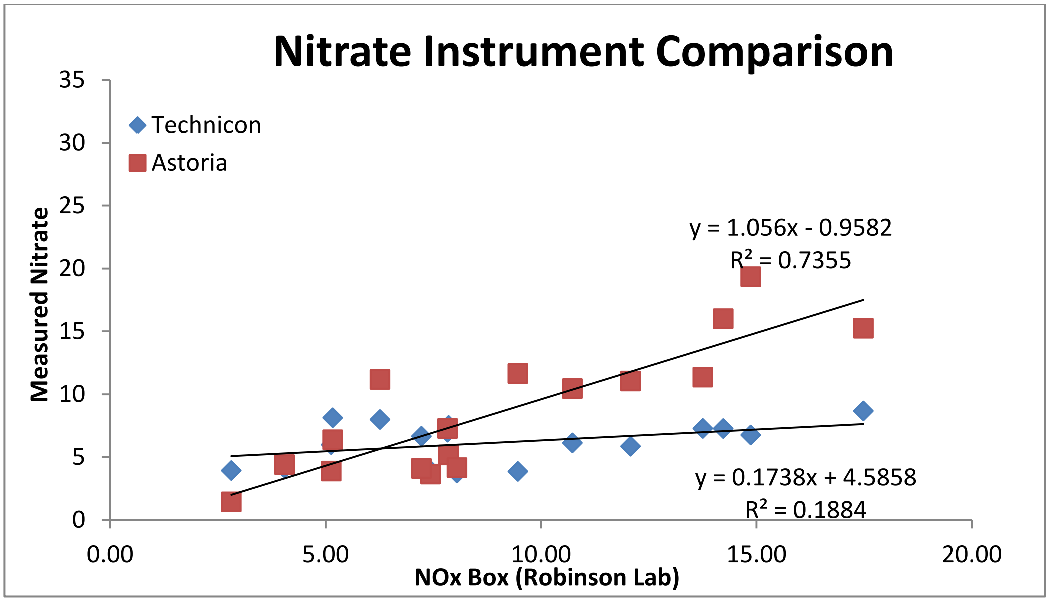
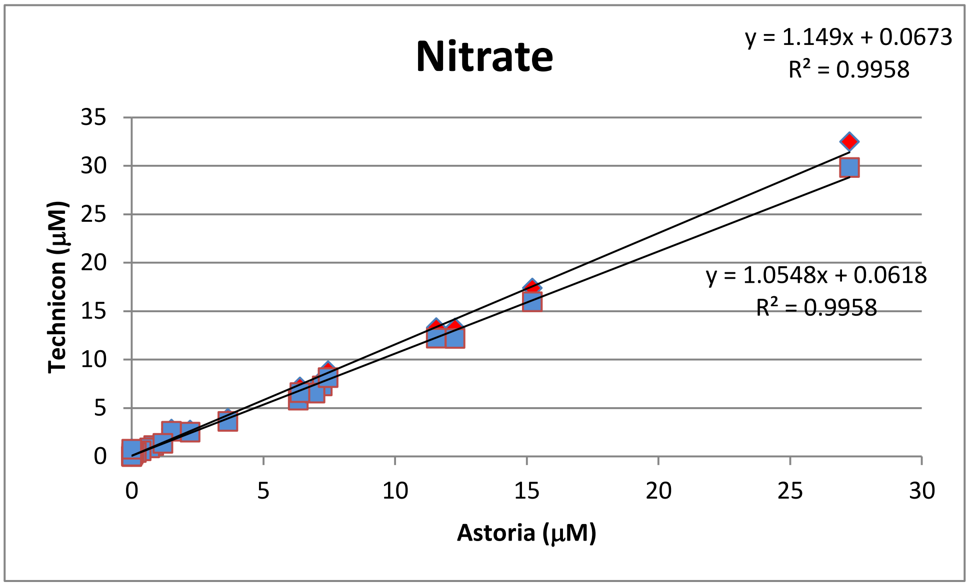


Figure A-4 Comparison of MERL measured nitrate+nitrite for both instruments to measurements on Teledyne instruments nitrous oxide sensor.



FigureA-5: Intercalibration data from 11/30/2009 showing Technicon values against Astoria values after the Technicon was retrofitted with a refurbished old style Cadmium column. Diamonds show data with a one point 'correction' for Technicon Cd efficiency. Squares show data without the correction.

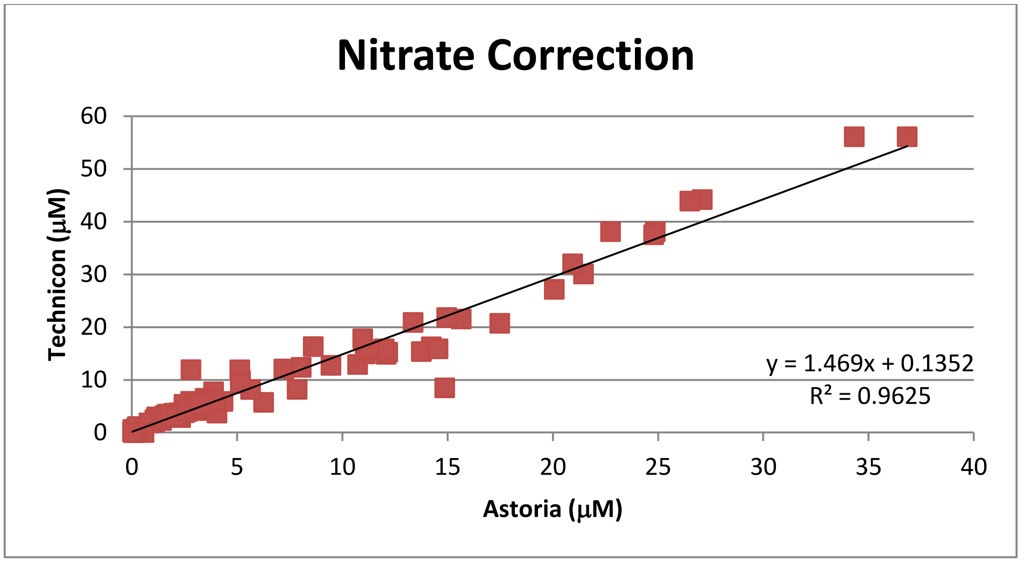


Figure A-6. Final pooled corrected nitrate data for all intercalibration samples run showing relationship between Astoria (X) and Technicon (Y) results once the erroneous cad efficiency correction was removed.

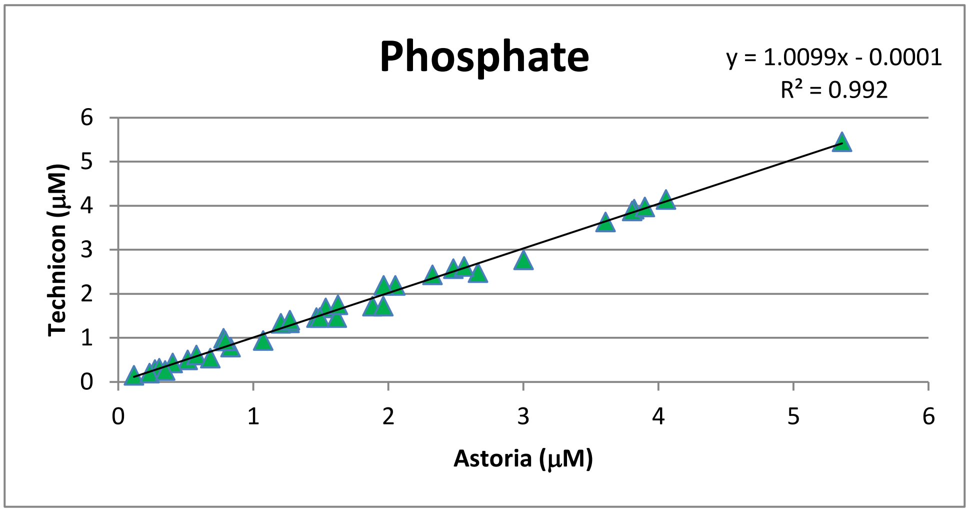


Figure A-7: Sample intercalibration curve for Ortho-Phosphate from intercalibration data run 10/28/2009.

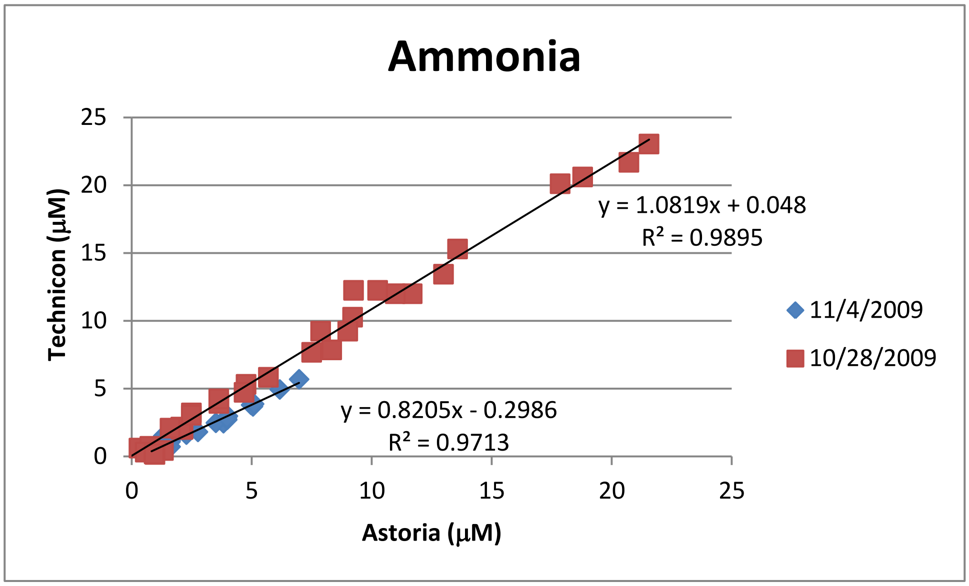


Figure A-8. Intercalibration of ammonia between Astoria and Technicon autoanalyzers. Data from 11/4/09 in blue diamonds, data from 10/28/09 in red squares.

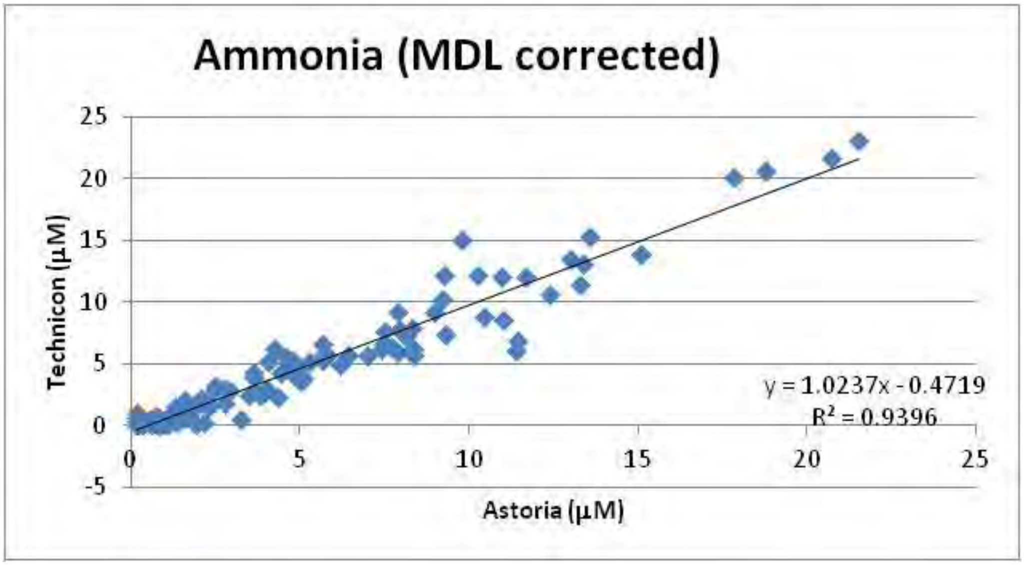


Figure A-9. Pooled and salinity corrected intercalibration data for Astoria vs. Technicon ammonium channels. These data were corrected such that values below the EPA reporting limit of 0.3mM are not considered in the analysis

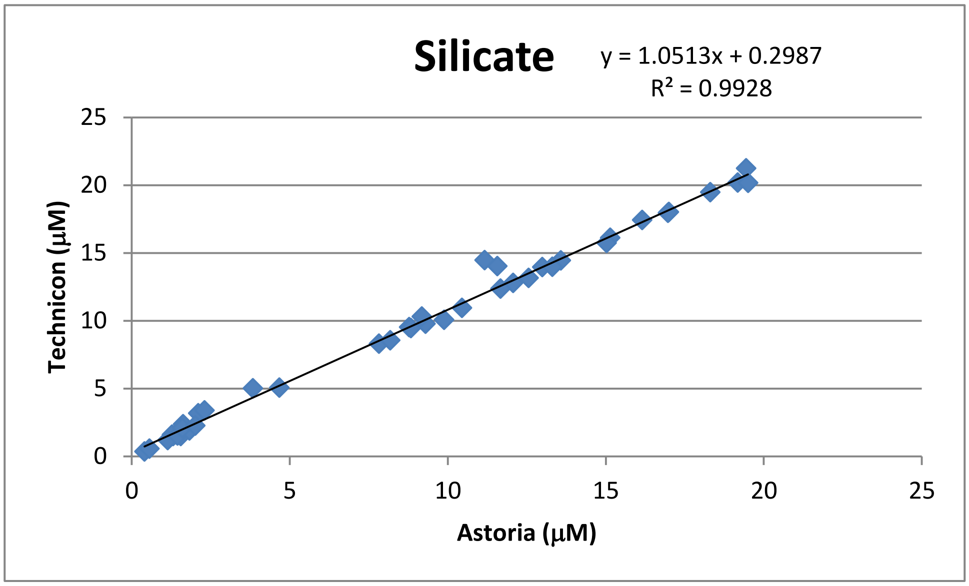


Figure A-10. Intercalibration results between Technicon and Astoria autoanalyzers for silicate. Samples run on 11/4/2009.

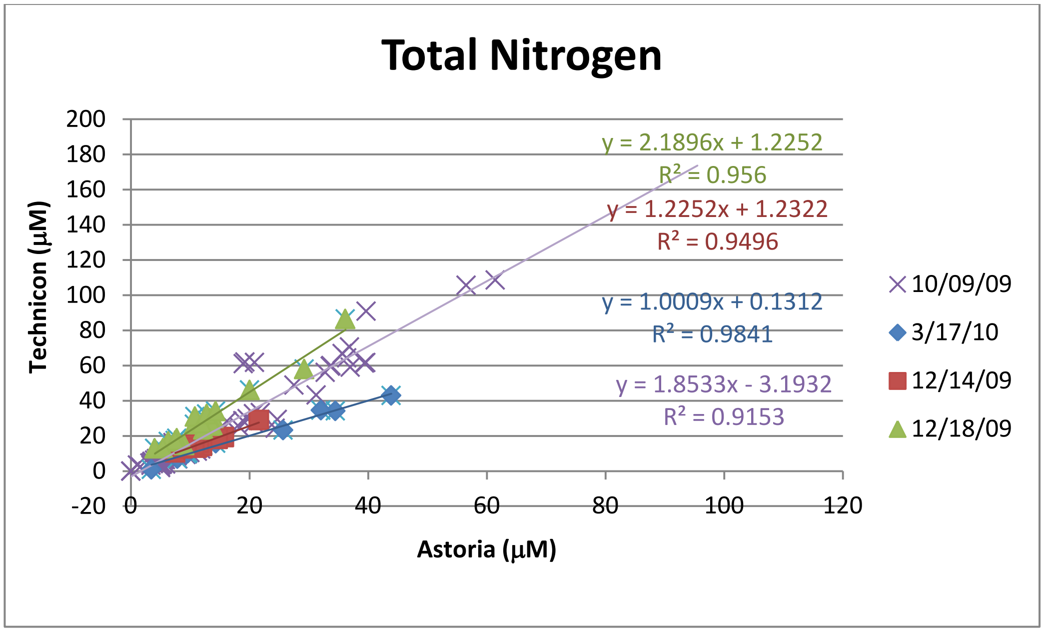


Figure A-11. Intercalibration of Total Nitrogen (TN) between Astoria and Technicon autoanalyzers with samples broken down by date run. R^2 of pooled sample is 0.57 with equation Y=1.27X+5.11

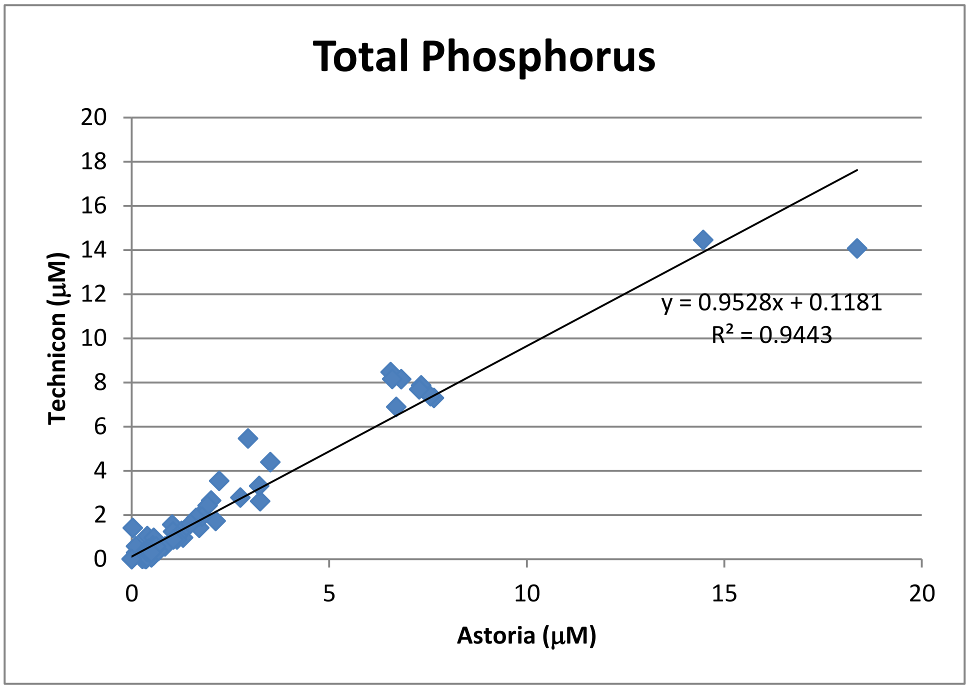


Figure A-12. Intercalibration of Total Phosphorus (TP) between Astoria and Technicon autoanalyzers. Samples run 10/09/2009.

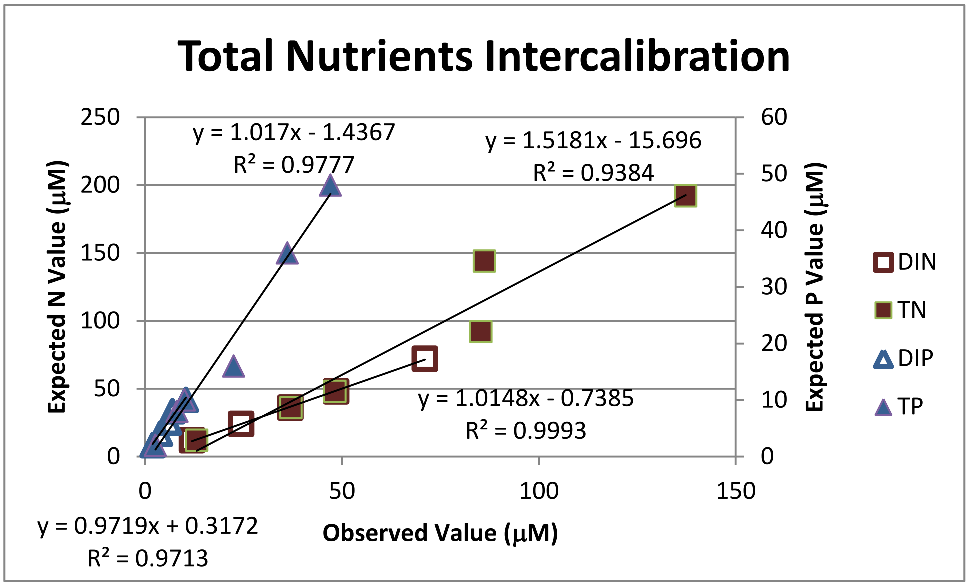


Figure A-13. Total nutrients vs. dissolved nutrients standards tests. TN/DIN is on the left Y-axis, while TP/DIP is on the right Y-axis. Dissolved nutrients are hollow markers, total nutrients are filled. Data run 12/14/2009

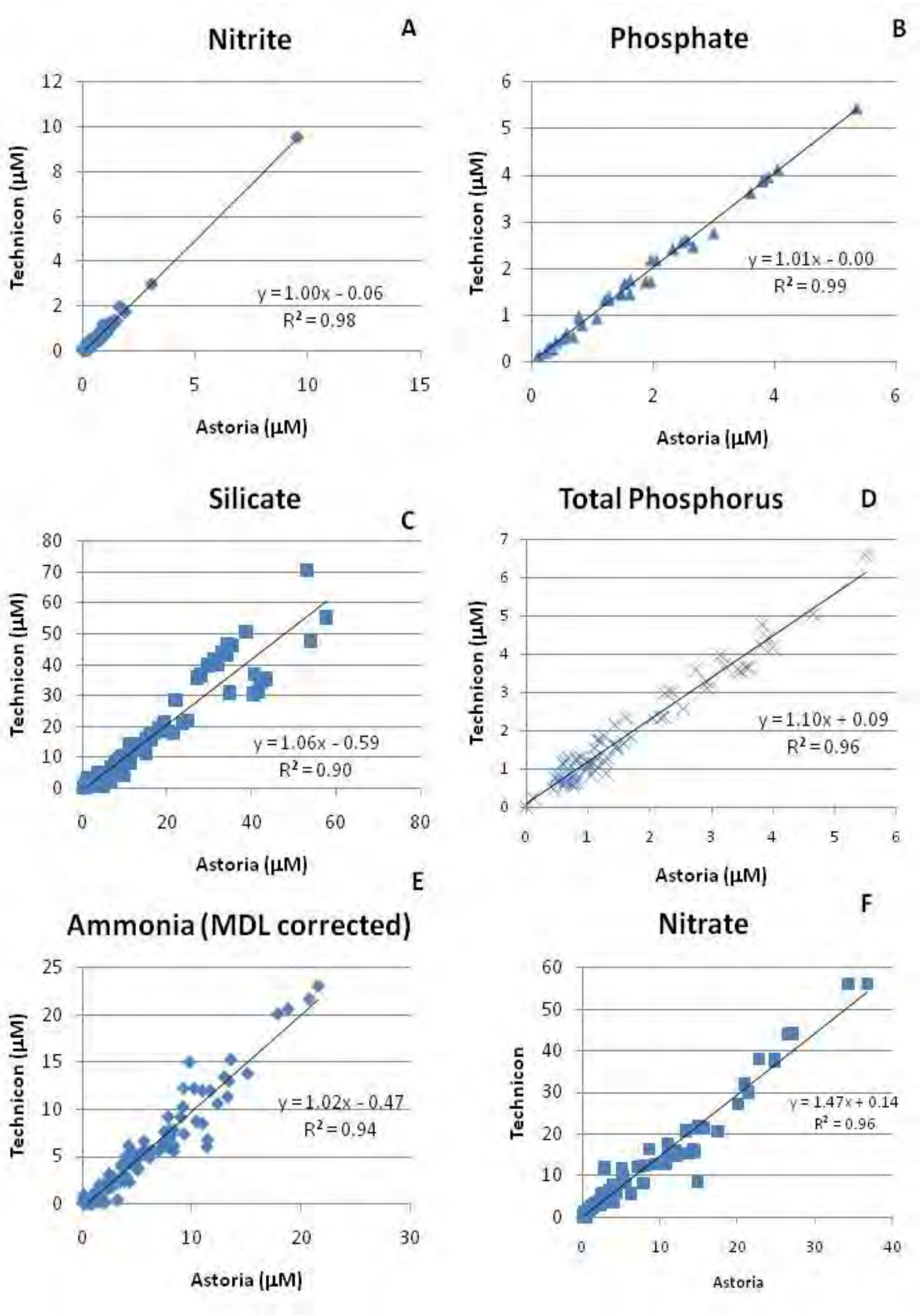


Figure A-14 Summary of pooled intercalibration data for all analytes measured. All concentrations are in mM with Astoria values on the X axis and Technicon values on the Y. Nitrate data include both nitrate and TN data run on the same channel.

APPENDIX B

NUTRIENT INPUT FROM WASTEWATER TREATMENT FACILITIES IN THE NARRAGANSETT BAY WATERSHED, 2000 – 2010

Preface

This appendix is based in large part upon the results of an independent study project by Rosmin Ennis undertaken under the supervision of Jason Krumholz and Candace Oviatt in the spring of 2011.

Executive Summary

Wastewater treatment facilities (WWTF) have been the primary source of nitrogen and phosphorus into Narragansett Bay for many years. Upgrades to 10 facilities in the Narragansett Bay watershed have been completed in the first stage of a project with the overall goal of reducing nitrogen and phosphorus loading to the Bay from WWTF by 50%. As expected, after upgrade, the majority of those facilities showed a reduction in nitrogen and/or phosphorus when compared to their load prior to upgrade and to those facilities that have not yet upgraded. With this in mind, there are a few additional main points of our study that should be highlighted.

The Bucklin Point facility in East Providence, RI reduced total nitrogen in effluent by about 50%. This reduction has been relatively consistent year-round since upgrade completion.

The Worcester and Woonsocket plants have shown large reductions in total nitrogen since implementation of advanced treatment, but are significantly upstream from Narragansett Bay proper, so it is difficult to tell at this stage what impact the reductions may have on the riverine abatement rate in the Blackstone River and therefore the overall impact on the downstream system; especially for the Worcester plant, which first upgraded in 2009.

The North Attleboro, MA facility has shown a large reduction in total phosphorus since its upgrade completion in 2008; however, the full impact of the upgrade is uncertain due to how recently it was completed. The Attleboro, MA facility showed an equally large reduction in total phosphorus in 2007-2010 when compared to 2000-2003. All facilities on the Pawtuxet River (Cranston, Warwick, and West Warwick) showed a large reduction in total phosphorus since their upgrade completions. However, a similar reduction in their total nitrogen loads was not observed most likely due to their difficulties with flooding in 2010. When this year of data is removed, all facilities' total nitrogen reductions improved.

Overall it appears that the upgraded facilities are indeed reducing their total nitrogen and total phosphorus loads to Narragansett Bay. However, the majority of these facilities are on rivers that discharge into Narragansett Bay not the Bay itself, which makes the full effect of the upgrades on the total load to the Bay difficult to determine.

Introduction

History of Nutrient Introduction into Narragansett Bay

Human interactions with the Narragansett Bay have had noticeable impacts on the ecosystem. Since the dawn of the Industrial Revolution in the mid 1800s, humans have been dredging the bottom of the Bay, inadvertently or purposefully introducing exotic species, and polluting the waters through the discharge of numerous chemicals and excess nutrients in the form of human and animal waste and agricultural fertilizers (Nixon et al., 2005; Nixon et al., 2008; Hamburg et al., 2008).

Prior to the Industrial Revolution, nutrient concentrations in Narragansett Bay were relatively low (Nixon et al., 2008). This kind of environment allowed vast eelgrass meadows to thrive, as eelgrass meadows are very sensitive to nutrient inputs (Nixon et al., 2008). However, a community shift occurred after the rapid industrialization, nitrogen pollution, and population growth associated with the Industrial Revolution (Nixon et al., 2008; Hamburg et al., 2008; Kelly 2008). The majority of these meadows quickly disappeared indicating an increase in nutrient concentrations in Narragansett Bay (Nixon et al., 2008).

The explosive population growth of the 19th century increased the demand for protein rich food imported from nearby areas, which in turn increased the amount on nitrogen in human waste (Nixon 1995; Hamburg et al., 2008). When coupled with the almost 55,000 people connected to established sewer systems in 1889, the amount of nitrogen being discharged into the Narragansett Bay and its major tributaries steadily increased and has continued to do so with population growth (Nixon et al., 2005; Nixon et al., 2008; Hamburg et al., 2008; King et al., 2008). Traditional agricultural practices also changed during the 19th century from the use of no synthetic fertilizers to their use on almost every farm (Hamburg et al., 2008). However, although synthetic fertilizers and other non-point sources of pollution are important when discussing the history of nitrogen introduction in Narragansett Bay, the single largest contributor of nitrogen to the Bay is sewage, which until very recently contributed about 65% of the Bay's total load of nitrogen (Nixon et al., 2008). This increased loading of nitrogen into Narragansett Bay quickly exhibited unwanted effects on the ecosystem.

Excess Nutrient Input Leads to Eutrophication

Phosphorus and primary production limiting nitrogen are essential nutrients in the maintenance of a healthy estuarine system (Latimer and Charpentier 2010; RI DEM 2005; Oviatt 2008; Bowen and Valiela 2001; Caraco and Cole 1999). However, the amount of reactive nitrogen in aquatic systems has increased every year until recently due to anthropogenic practices and is causing eutrophication, an increase in the input of organic matter to an ecosystem (Nixon et al., 2008; Latimer and Charpentier 2010; King et al., 2008; Caraco and Cole 1999; Howarth and Marin 2006).

Eutrophication is detrimental to aquatic ecosystems because it promotes increased algal growth, which prevents sunlight from penetrating the water column to sustain benthic plants (Bowen and Valiela 2001; RI DEM 2005). Decomposing algae strip the water of its dissolved oxygen, creating hypoxic or anoxic conditions leading to fish kills and possible changes in food web structures (Latimer and Charpentier 2010).

The occurrence of eutrophication in Narragansett Bay due to anthropogenic nutrient input has been increasing over the last century. Previous studies have determined that nitrogen input to coastal waters is greatest in areas of agricultural and urban activity (Howarth and Marino 2006). Observed trends in carbon and nitrogen concentrations also provide strong evidence that eutrophication is occurring in the upper Narragansett Bay due to anthropogenic causes (King et al., 2008). Additionally, studies of ^{15}N in the Bay have suggested eutrophication and decreased dissolved oxygen concentrations as a result of sewage discharge (King et al., 2008).

Advances in Wastewater Treatment and Reduction of Nitrogen

The establishment of sewer systems and sewage treatment facilities in the Narragansett Bay watershed in the late 1880s brought waste from a large number of people to one central location for discharge into the water (Nixon et al., 2005). Previously, waste had been left in the soil on land as fertilizer (Nixon, et al., 2005; Hamburg et al., 2008). However, the newly established wastewater treatment facilities received raw sewage and did little other than undertake rudimentary treatment methods aimed at protecting public health and safety (Latimer and Charpentier 2010). The introduction of secondary treatment in the 1970's, and subsequently tertiary treatment in the 2000's has provided better options for treatment of wastewater prior to its discharge into the Bay (Nixon et al., 2008). Primary treatment, or more simply disinfection, of wastewater was the first advance in wastewater treatment followed by secondary treatment, more advanced filtration and removal of suspended solids (Hamburg et al., 2008). By the late 20th century, all public sewage treatment facilities were equipped for secondary treatment of wastewater. However, wastewater treatment facilities are currently the largest source of nitrogen to Narragansett Bay and further reduction in nitrogen is needed (RI DEM 2005).

The motivation to further reduce nitrogen was accelerated by the occurrence of intense algal blooms and fish kills associated with eutrophication in 2003 (Oviatt 2008). Rhode Island General Law now requires the Department of Environmental Management (DEM) to not only reduce nitrogen loadings from wastewater treatment facilities by 50% by 2014 and provide reports of their reduction status, but also to implement a plan of action designed to manage excess nutrients and their effects on Rhode Island water to prevent eutrophic conditions (RI DEM 2005; Section 46-12-2; Section 46-12-3). Additionally, the Federal Clean Water Act requires each state to create a schedule for water quality restoration in impaired waters (RI DEM 2005).

Further reduction of nitrogen has been accomplished by the development of tertiary treatment methods (Hamburg et al., 2008). The addition of anaerobic denitrification by bacterial growth as the last step in wastewater treatment converts nitrate to inert nitrogen gas, which is released from the facility into the atmosphere (RI DEM 2005; Nixon et al., 2008). The reduction of nitrogen in discharged effluent is anticipated to reduce the amount of primary productivity thereby restoring habitable dissolved oxygen concentrations to the benthic community and sediments (Nixon et al., 2008). In recent years, there has been a decrease in the amount of nitrogen discharged into Narragansett Bay due to the establishment of tertiary treatment at several facilities and stricter environmental regulations (King et al., 2008). However, some of the larger wastewater treatment facilities still remove only a small amount of the total nitrogen they collect in untreated sewage (Hamburg et al., 2008).

It is difficult to determine how the reduction of nitrogen in wastewater effluent will translate to Narragansett Bay as a whole because the Bay has been changing dramatically over the years (Nixon et al., 2008). Long-term upward trends in temperature of almost 1˚C have put stress on the ecosystem (Pilson 2008; Hamburg et al., 2008). Increases in precipitation and river flow into the Bay have also increased over the last century (Pilson 2008). Freshwater input from the Bay's major tributaries largely influence residence time of water and dissolved substances in the Bay (Pilson 1985; 2008). Nutrient cycling and retention in the coastal environment must be assessed prior to determining the allowable amount of nutrients discharged into the water (Doering et al., 1990). Topography, geology, and oxygen concentration in the water, among other factors, must also be taken into account because they influence the retention of nitrogen in a system (Caraco and Cole 1999).

Objectives

The primary objective of this study is to determine the load of nitrogen in the form of nitrite (NO_2), nitrate (NO_3), ammonium (NH_4^+), and total nitrogen (TN) in the discharged effluents of wastewater treatment facilities (WWTF) in the Narragansett

Bay watershed. The load of phosphorus in the form of total phosphorus (TP) was also determined for the same WWTF. The load of nitrogen and phosphorus forms were also determined for the six major rivers that discharge into the Narragansett Bay.

Many methods exist to calculate annual loads based on measurements of flow and concentration. Although most ratio estimators are virtually equal when using a large sample size, in this study, Beale's unbiased ratio estimator (Beale 1962) was deemed the most suitable for several reasons. Beale's unbiased ratio estimator is ideally used in situations in which there are limited concentration data, but daily flow data are available (Dolan et al., 1981). Beale's unbiased ratio estimator also places different emphasis on concentration values based on their deviation from the mean, therefore, creating an almost unbiased estimate in cases where the distribution of values is not normal (Dolan et al., 1981; Tin 1965). An unbiased estimate is useful to data sets with samples from different times of the year, as there may be great variation throughout the year. It was also determined through comparison to other methods, means over a time period or log-linear regressions, by Dolan et al. (1981), that Beale's unbiased ratio estimator is superior in removing bias while still retaining high precision and accuracy (Dolan et al., 1981). Finally, Beale's unbiased ratio estimator has been used before in similar kinds of studies (Nixon et al., 1995; Nixon et al., 2008; Fulweiler 2003).

These load values will then be examined to determine the effectiveness of nitrogen reduction in WWTF upgraded to tertiary treatment methods and how this reduction translates to changes in concentrations of these nutrients in Narragansett Bay and its major tributaries. It is expected that WWTF upgraded to tertiary treatment methods will discharge lower loads of nitrogen into Narragansett Bay.

Methods

Data Contribution

Total nitrogen (TN), ammonium (NH_4^+), nitrite (NO_2), nitrate (NO_3), and total phosphorus (TP) concentrations in effluent discharged from wastewater treatment facilities (WWTF) in the Narragansett Bay watershed and nutrient loading of rivers emptying into Narragansett Bay were examined in this study. Facility flow data associated with each parameter measurement were also considered. All WWTF data was in the form of MS Excel files. Angelo Liberti and Deb Merrill of the Rhode Island Department of Environmental Management (RI DEM) contributed all Rhode Island WWTF data as well as all data for the Attleboro, North Attleboro, and Worcester facilities. All remaining facilities were estimated from previous measurements. All data concerning the nutrient loading of rivers emptying into Narragansett Bay was processed and contributed by Steve Granger of the University of Rhode Island's Graduate School of Oceanography.

Data Processing

The data contributed by the RI DEM contain many different parameter and flow measurement intervals (ie. daily, weekly, monthly, etc.). For consistency, the monthly average of each parameter and flow from each facility was used for analysis. In some cases, the monthly average is the average of several measurements taken over the course of each month. All flow values were the monthly average of continuous flow measurements (Table B-1).

All relevant flow and parameter data were isolated from the RI DEM data and separated into its own MS Excel file by facility. From there, all flow data was converted from millions of gallons per day (Mgal/d) as it was in the RI DEM data to liters per day (L/d) and then to cubic meters per day (m^3/d). All parameter concentration data was converted from milligrams per liter (mg/L) as it was in the RI DEM data to moles per liter (mol/L). A flux value in moles per day (mol/d) for each month was determined from flow (L/d) and parameter concentration (mol/L). All monthly flux values were moles of nitrogen per day for all nitrogen related parameters and moles of phosphorus per day for all total phosphorus (TP) measurements.

Once flux values had been calculated from parameter concentration and flow (L/d) for all years of available data, an annual load in kilomoles per year (Kmol/y) was determined by using a Beale’s unbiased ratio estimator macro in MS Excel (modified from Ganger, pers. comm.). The same process was repeated for both the active treatment season, defined by the RI DEM as May to October, and the inactive treatment season, defined as November to April. Each seasonal load (Kmol/season) was calculated by using only each season's months of data with the Beale's macro then converting to kilomoles per day (Kmol/d) then multiplying by the number of days in each season to arrive at a seasonal load in kilomoles per season. This process was repeated with available data for all WWTF.

Estimating Missing Data

The data contributed by the RI DEM did not contain data for every year from 2000-2010 for all WWTF. It also did not include all facilities being examined in this study as was previously described. This problem was solved in one of two ways: scaling available load data by population change or by using a multiplication factor with population. The cities and towns served by each facility were provided by the RI DEM website. The annual total populations of the cities and towns served by each facility from 2000-2010 were found on the U.S. Census Bureau website. The actual population of the total served by each facility in 2000 was provided by the RI DEM website. The percent of the total population for each city or town served in 2000 was calculated from these values. This percentage was used for the remaining years in the decade to calculate the actual population served by each facility for each year from 2000-2010. The population change from one year to the next from 2000-2010 was then calculated from the annual actual population served by each facility. This technique assumes that growth occurs proportionally in sewered and unsewered areas, which, for the most part, is likely to be a robust assumption. Furthermore, population change rates were generally low (ranging from -3.8% -2.9%), so the sensitivity of the overall loading estimate to this parameter is low.

For facilities included in the data provided by the RI DEM, individual years of missing load data from 2000-2010 were estimated by scaling the previous year of available load data by the change in population served by the facility. For the Massachusetts facilities that were not included in the RI DEM data, individual years of load data were not estimated. Instead, a 2007-2010 annual load average was estimated by scaling the 2000-2003 annual load average calculated by Nixon (2008) by the change in population served by each facility from 2000-2010.

The RI DEM data did not include total nitrogen or total phosphorus data for all facilities. For those facilities that had no data for total nitrogen or total phosphorus, annual and seasonal loads for total nitrogen or total phosphorus were calculated by using a multiplication factor of 0.8 moles of nitrogen per person per day or 0.045 moles of phosphorus per person per day. Similar multiplication factors (0.9 mol N/person/day and 0.035 mol P/person/day) were previously calculated by Nixon, et al. (2008) using earlier data. The multiplication factors used in this study were calculated in the same way using available data from this study. The appropriate multiplication factor was multiplied by the actual population served by the facility with missing data to get a daily load. The daily load was then multiplied by the number of days in the year, 365, or in each season to arrive at an annual load in moles per year or a seasonal load in moles per season.

Data Analysis

All load values were formatted into tables and graphs were created using MS Excel. “Pre” and “post” values were calculated from these tables to illustrate the effect upgrade completion has had on the load of upgraded and non-upgraded facilities. “Pre” values are defined as the average of load values from 2000-2004, except at the Burrillville (2000-2001) and Woonsocket (2000-2002) facilities, which upgraded in 2002 and 2003, respectively. The Burrillville and Woonsocket facilities use different years to avoid averaging over the year of upgrade completion. “Post” values are defined as the average of load values from 2007-2010, except at the Worcester and North Attleboro facilities. The Worcester facility upgraded in 2009, so the only “post” value is the 2010 load. “Post” values for the North Attleboro facility were the average of 2009 and 2010 data to avoid averaging over the year of upgrade completion. The percent difference between the pre and post loads were also calculated. T-tests were used to determine significance between the pre and post both annual and seasonal load values and any other load difference.

Results

The results presented below are the most interesting and relevant results to this study. Results are first presented as the total load to Narragansett Bay and subsequently divided by the body of water into which each facility discharges. Dotted lines in figures indicate that the load value was estimated from population data and recent load values. Several facilities show load reductions immediately prior to upgrade completion, which can most likely be attributed to the facilities' ability to begin reducing before the upgrade was officially reported complete (Liberti, pers. comm.; Travers, pers. comm.). A complete record of the status and trends of all plants for which data are available can be found in the appendix.

Total Sewage Discharge to Narragansett Bay

The average annual total sewage nitrogen from 2007-2010 discharged from each facility was added together to achieve an average grand total amount of nitrogen discharged into Narragansett Bay annually during that time period. The same was repeated for the average annual total sewage phosphorus discharged from each facility from 2007-2010. The average grand total amount of sewage nitrogen discharged into Narragansett Bay per year from 2007-2010 was 262.0 million moles and the average grand total amount of sewage phosphorus discharged per year was 14.1 million moles (Table B-2). This nitrogen load is 101.5 million moles, or 38.5%, less than the grand total nitrogen load calculated for 2003 and the phosphorus load is 4.2 million moles, 27.7% less (Nixon et al. 2008).

The average annual and active season total nitrogen concentrations from 2000-2004 and 2007-2010 were calculated for all facilities that had total nitrogen concentration data available. The Worcester, Woonsocket, Burrillville, and North Attleboro used 2010, 2000-2002, 2000-2001, and 2009-2010 averages, respectively, to avoid averaging over upgrades. These values were compared to existing and future nitrogen limits. The Bucklin Point and North Attleboro facilities are the only two that were in compliance with their nitrogen limits throughout the year and specifically during the active season after their upgrades were completed. The East Greenwich and Cranston facilities were in compliance with their limits only during the active season after their upgrades were completed. Due to flooding in 2010, all facilities on the Pawtuxet River (Cranston, West Warwick, Warwick) were examined more closely. Only average annual total nitrogen concentrations from 2007-2009 were calculated for all three facilities as the flood occurred in March, which is not included in the active season. The average annual total nitrogen concentrations from 2007-2009 for the Cranston, West Warwick, and Warwick facilities were 11.2 mg/L, 12.3 mg/L, and 8.3 mg/L, respectively. Many facilities have nitrogen limits set to go into effect in several years and it can be seen that these facilities have already begun total nitrogen concentration reductions to meet those limits by their deadlines (Table B-3).

The average annual and active season total phosphorus concentrations from 2000-2004 and 2007-2010 were calculated for all facilities with available total phosphorus concentration data. The Worcester, Woonsocket, Burrillville, and North Attleboro used 2010, 2000-2002, 2000-2001, and 2009-2010 averages, respectively, to avoid averaging over upgrades. These values were compared to existing and future phosphorus limits. The Smithfield and Cranston facilities are the only two that were in compliance with their phosphorus limits throughout the course of the year and, more specifically, during the active season after their upgrades were completed. The Warwick facility was in compliance with its phosphorus limit during the year and the Woonsocket facility was in compliance with its phosphorus limit during the active season. Due to flooding in 2010, all facilities on the Pawtuxet River (Cranston, West Warwick, Warwick) were examined more closely. Only average annual total phosphorus concentrations were calculated as the flood occurred in March. These facilities have average annual total phosphorus concentrations of 0.89 mg/L, 1.4 mg/L, and 0.62 mg/L, respectively, from 2007-2009 (Table B-4).

The annual total nitrogen load of upgraded facilities was on average 7% higher than that of non-upgraded facilities from 2000-2004. However, the annual total nitrogen load of upgraded facilities was significantly less, by about 70%, than that of non-upgraded facilities from 2007-2010 (df = 7, T = -3.31, P = 9.68×10^{-4}). The average total nitrogen load difference between upgraded and non-upgraded facilities during the active season and the inactive season was 1.54×10^{4} moles per day and 1.03×10^{4} moles per day, respectively. The average total nitrogen load difference during the active season was not significantly different than the average total nitrogen load difference during the inactive season (df = 20, T = 0.26, P = 0.523; Fig. B-1).

The average total phosphorus load difference between upgraded and non-upgraded facilities during the active season and the inactive season was 4.45×10^{3} moles per day, and 5.28×10^{3} moles per day, respectively. The average total phosphorus load difference during the active season was not significantly different than the average total phosphorus load difference during the inactive season (df = 20, T = 0.18, P = 0.558; Fig. B-2).

Direct Discharge to Narragansett Bay

After its upgrades were completed in 12/2005, the Bucklin Point facility significantly reduced its average annual total nitrogen load (df = 7, T = 5.79, P = 3.11×10^{-6}; Fig. B-3). Bucklin Point also significantly reduced its annual ammonium load (df = 7, T = 11.71, P = 2.67×10^{-7}) while its annual nitrate load significantly increased after upgrades were completed (df = 7, T= -7.49, P = 5.60×10^{-6}; Fig. B-4). The Bucklin Point facility also significantly reduced its active season nitrite load (df = 7, T = 2.46, P = 0.005; Fig. B-5). Load reductions during the active and inactive season showed a similar pattern to annual load reductions for all parameters.

Although the East Greenwich facility did not significantly reduce its average annual total nitrogen load after upgrades were completed, it did significantly reduce its average active season total nitrogen load by about 40% more than the annual reduction (df = 7, T = 4.34, P = 1.96×10^{-4}; Fig. B-6). The East Greenwich facility also significantly reduced its annual nitrite load after upgrades were completed (df = 7, T = 1.55, P = 0.039; Fig. B-7). Both active and inactive season nitrite load reductions followed a similar pattern to the annual load reduction. The East Greenwich facility significantly reduced its average active season nitrate load (df = 7, T = 2.74, P = 0.003), but it significantly increased during the inactive season (df = 7, T = -2.46, P = 0.005; Fig. B-8). Additionally, it should be noted that the East Greenwich facility experimented with nitrogen removal during June and July of 2005, which may account for early reductions observed before upgrade construction was completed (Travers, pers. comm.).

Discharge to the Blackstone River

Upon upgrade completion, the Burrillville facility significantly reduced its average active season total sewage nitrogen (df = 4, T = 2.17, P = 0.037) and total sewage phosphorus loads (df = 4, T = 2.03, P = 0.045; Fig. B-9). However, during the inactive season, the Burrillville facility significantly increased its average ammonium load (df = 4, T = -2.28, P = 0.032; Fig. B-10).

The Woonsocket facility significantly reduced its average annual nitrite load after upgrades were completed in 9/2001 (df = 5, T = 5.95, P = 2.85×10^{-4}; Fig. B-11). Both the active and inactive season load reductions were similar to the annual reduction. The Woonsocket facility also significantly reduced its inactive season ammonium load with similar reductions during the active season and the year overall (df = 5, T = 2.09, P = 0.025; Fig. B-12).

After upgrades were completed in 6/2006, the Smithfield facility significantly reduced its annual total nitrogen load (df = 7, T = 3.05, P = 0.002; Fig. B-13). Both the active season and inactive season total nitrogen load reductions followed a similar pattern to annual reductions. The Smithfield facility also significantly reduced its annual ammonium load (df = 7, T = 7.57, P = 5.20×10^{-6}), but its annual nitrate load significantly increased after upgrades were completed (df = 7, T = -6.25, P = 1.87×10^{-6}; Fig. B-14). A similar reduction pattern in ammonium and nitrate was seen seasonally. A significant reduction in average annual total phosphorus discharged from the Smithfield facility occurred after upgrades were completed (df = 7, T = 10.03, P = 7.72×10^{-7}; Fig. B-15).

Significant changes in the average annual or seasonal loads for any parameter from the Worcester facility could not be determined as this facility recently upgraded to advanced wastewater treatment in 2009. However, a large reduction in ammonium and total phosphorus occurred after the upgrade was completed (Fig. B-16). It should be noted that the Worcester facility participated in nutrient removal training and assistance during 2007 and 2008, which is most likely the cause of reductions seen prior to the upgrade being reported complete (Travers, pers. comm.).

Discharge to the Pawtuxet River

Due to excessive flooding in 2010, all facilities that discharge to the Pawtuxet River were evaluated for two sets of years after upgrades were completed: 2007-2010 and 2007-2009. The purpose is to illustrate the effect the flood had on post upgrade load values.

After upgrades were completed in 11/2004, the Warwick facility significantly reduced its average annual total nitrogen load (df = 7, T = 3.09, P = 0.001; Fig. B-17). Seasonal total nitrogen load reductions followed a similar pattern to annual reductions. The Warwick facility significantly reduced its average annual ammonium load after upgrades were completed and both seasons showed comparable reductions (df = 7, T = 2.83, P = 0.002; Fig. B-18). Average annual nitrite loads were significantly reduced (df = 7, T = 2.35, P = 0.006) while average annual nitrate loads significantly increased after upgrades were completed (df = 7, T = -1.86, P = 0.019; Fig. B-19). The Warwick facility also significantly reduced its average annual total phosphorus load upon upgrade completion (df = 7, T = 3.32, P = 0.001; Fig. B-20). When flooding is accounted for, the average total nitrogen, ammonium, nitrite, and total phosphorus loads after upgrades were completed both annually and seasonally were on average about 10% lower than when 2010 load values were included. The average nitrate load after upgrades were completed both annually and seasonally was about 10% higher than when 2010 load values were included. Additionally, it should be noted that the Warwick facility underwent several nitrogen removal trial periods from 2001-2003, which may account for reductions observed prior to upgrade construction completion (Travers, pers. comm.).

Although it completed upgrades in 1/2006, the Cranston facility did not significantly reduce its average annual total nitrogen load (df = 7, T = 1.16, P = 0.101). However, its average active season total nitrogen load was significantly reduced (df = 7, T = 2.33, P = 0.007; Fig. B-21). Additionally, the Cranston facility significantly reduced it average total phosphorus load year round (df = 7, T = 2.69, P = 0.003; Fig. B-22). When flooding is accounted for, the Cranston facility still did not significantly reduce its average annual total nitrogen load (df = 7, T = 1.16, P = 0.143).

The West Warwick facility significantly reduced its average annual ammonium load (df = 7, T = 2.56, P = 0.004) while its average annual nitrate load significantly increased after upgrades were completed in 7/2005 (df = 7, T = -4.39, P = 1.81×10^{-4}; Fig. B-23). Seasonal ammonium and nitrate loads had comparable reductions to the annual load reductions. When flooding is taken into account, all parameter load reductions were relatively unchanged with the exception of active season total phosphorus, which was significantly reduced (df = 7, T = 1.28, P = 0.023).

Discharge to the Ten Mile River

The upgrades completed at the North Attleboro facility in 2008 have not yet shown any significant change for any parameter either annually or seasonally, though mean values for total nitrogen and ammonium in upgraded years show an 8% increase and 25% reduction, respectively, over mean values pre-upgrade. However, the average annual total phosphorus discharged from the facility has dramatically decreased by an average of 75% annually and during the active season since upgrade completion (Fig. B-24).

River Loading

The grand total dissolved inorganic and total nitrogen load from all rivers combined was each on average about 25% less in 2008-2010 than the load from 2003-2004. The Pawtuxet, Woonsquatucket, Moshassuck, and Taunton Rivers reduced both their dissolved inorganic and total nitrogen by an average of 30%, 36%, 43%, and 35%, each, respectively, in 2008-2010 when compared to 2003-2004. The grand total dissolved inorganic phosphorus and total phosphorus load from all rivers combined was on average 45% and 83% less, respectively, in 2008-2010 than the load in 2003-

2004. In 2008-2010, the Blackstone River increased its dissolved inorganic and total phosphorus loads by about 50% and 40%, respectively. The Pawtuxet River reduced its dissolved inorganic and total phosphorus loads by over 50% each in 2008-2010. The Woonasquatucket, Moshassuck, and Taunton Rivers reduced their dissolved inorganic and total phosphorus loads by about 80% each. The Ten Mile River reduced its dissolved inorganic and total phosphorus load by about 70% each (Table B-5).

Discussion

Advanced wastewater treatment for the removal of nitrogen is a two part process that includes aerobically converting ammonium to nitrite then to nitrate, or nitrification, then anaerobically converting nitrate to nitrogen gas, or denitrification ("Nitrogen Removal from Wastewater"; RI DEM, 2005). A common trend observed among upgraded facilities that utilize this process to remove nitrogen was a dramatic decrease in their ammonium loads with a large increase in their nitrate loads. This most notably occurred at the Bucklin Point, Smithfield, Warwick, and West Warwick facilities, all of which had significant reductions in ammonium loads with significant increases in nitrate loads. Additionally, the Warwick facility significantly reduced its nitrite load while its nitrate load significantly increased. This occurrence is most likely caused by the nitrification-denitrification process described above ("Nitrogen Removal from Wastewater"; RI DEM, 2005). However, the nitrate loads of three out of the four previously mentioned facilities had quite substantial increases, some by several orders of magnitude. It could be speculated that the increase in nitrate observed at these facilities is due to an insufficient holding time of wastewater that does not allow for effective denitrification of nitrate. When this phenomenon is evaluated by examining DIN (NH_4^+ + NO_2 + NO_3) discharge from facilities where it was most common, it was found that DIN discharge significantly decreased is almost all cases, meaning the ammonium reduction was greater than the nitrate increase. Despite this observation, the reduction of ammonium and nitrite and increase in nitrate is indicative that the process of advanced wastewater treatment is functioning properly (“Nitrogen Removal from Wastewater”; RI DEM, 2005).

The total nitrogen load per year from all facilities combined in 2007-2010 was almost 40% lower than the total nitrogen load per year from all facilities combined calculated for 2000-2003 by Nixon et al. (2008). This reduction is likely attributed to the completion of upgrades as completed facilities accounted for almost 90% of the total load reduction in 2007-2010. The Worcester facility alone accounts for about half of the total load reduction. However, it is difficult to tell if this large reduction is due to the upgrade or annual variation as this facility was completed very recently in 2009. However, the Bucklin Point facility showed a consistent year round total nitrogen reduction of about half, which accounts for almost 20% of the grand total load reduction. Additionally, this facility is in now compliance with Rhode Island General Law stating that wastewater treatment facilities must reduce their nitrogen load by 50% (Section 46-12-2).

It should be noted that the Warwick facility has also consistently shown a significant reduction of its total nitrogen load, but due to the large flood in 2010, which overwhelmed all facilities on the Pawtuxet River (Warwick, Cranston, and

West Warwick), it is not in compliance with Rhode Island General Law. However, prior to flooding in 2010, the Warwick facility was in compliance with Rhode Island General Law with an annual nitrogen reduction of about 50% (Section 46-12-2). The Cranston and West Warwick facilities follow a similar reduction pattern but neither are in compliance either annually or seasonally, with or without the flood.

The total phosphorus load per year from all facilities combined in 2007-2010 was about 30% less than the total phosphorus load per year from all facilities combined calculated for 2000-2003 by Nixon et al. (2008). This reduction is largely due to the efforts of upgraded facilities to remove phosphorus from their effluent as they accounted for over 90% of the grand total phosphorus load reduction. The most successful of these facilities were the Smithfield, Cranston, Warwick, and Worcester facilities, which had consistent reductions of about 90%, 70%, 60%, and 50%, respectively, year round. Of those facilities, the Smithfield, Cranston, and Warwick facilities have phosphors permits issued. The reductions of the Worcester, Smithfield, and Cranston facilities are especially noteworthy as they are the largest and third largest facilities on the Blackstone River and largest on Pawtuxet River.

Several facilities on the rivers that drain to Narragansett Bay showed significant decreases in their average annual total nitrogen and total phosphorus loads after their upgrades were completed. Phosphorus is essential to river ecosystems as it is the limiting nutrient for primary productivity (Kelly 2001); therefore, facilities located on rivers in the Narragansett Bay watershed also focused on removing phosphorus from their effluent (RI DEM 2005). As mentioned earlier, the Worcester, Woonsocket, and Smithfield facilities on the Blackstone River had large decreases in their total phosphorus loads. The Cranston, Warwick, and West Warwick facilities located on the Pawtuxet River also had large reductions in their total phosphorus loads. On the Ten Mile River, the North Attleboro and Attleboro facilities both largely reduced their total phosphors loads. Although they were not as great, most of these river facilities also had reductions in their total nitrogen loads. However, it is very difficult to tell the impact that these reductions will have on the overall Narragansett Bay ecosystem as only about 50% of river phosphorus loads reach Narragansett Bay proper (Nixon et al., 1995). Attenuation of sewage phosphorus in the Blackstone River removes about 25% of the total phosphorus load discharged (Nixon et al., 2008). Additionally, phosphorus reaching Narragansett Bay from the Pawtuxet and Ten Mile Rivers may not be purely from sewage as it has been observed that there are additional sources of phosphorus, such as storm water runoff, in these rivers (Nixon et al., 2008; RI DEM 2005). Discharged sewage nitrogen also has the ability to be released to the atmosphere through denitrification or stored in river sediments, which makes it difficult to determine the source of nitrogen entering Narragansett Bay (Nixon et al., 2008). Therefore, upgrades completed on rivers may have an immediate impact on the river in which they discharged but the impact they have on the Narragansett Bay system may be less apparent as of yet.

The wastewater treatment facilities examined in this study commonly enforce limits for nitrogen and/or phosphorus concentrations in effluent prior to discharge during the summer months of May to October, or the active season as it is referred to in this study (RI DEM 2005). Concentration limits are enforced during this time period because it is thought that greatest reductions will occur during this time reducing primary productivity so that benthic dissolved oxygen concentrations will rise to prevent anoxia (Nixon et al., 2008). It was anticipated that upgraded facilities would dramatically reduce their loads during the active season because the process of advanced wastewater treatment is temperature dependent (“Nitrogen Removal from Wastewater”). Warmer temperatures increase the efficiency of the nitrification-denitrification process meaning increased nitrogen reduction (“Nitrogen Removal from Wastewater”). However, no significant difference was observed during the active and inactive seasons. Since there was no significant difference between load reductions during the two seasons, there may be other factors that influence the efficiency of nitrogen reduction. It could be speculated that there is no significant difference in seasonal loads because the underground cement wastewater holding tanks are well insulated and seasonal changes in the surrounding environment have little effect on the temperature of the wastewater. Whatever the reason may be, inactive season load reductions from upgraded facilities have been more efficient than originally expected.

Although facility upgrades accounted for the majority of the large nitrogen and phosphorus reductions to Narragansett Bay, it is difficult to tell the full effect these reductions will have on the Narragansett Bay ecosystem. Management strategies aim to reduce nutrient concentrations as much as possible to return Narragansett Bay to its condition before human nutrient introduction (Nixon et al., 2008). However, Narragansett Bay is a very dynamic ecosystem that has been affected by a multitude of environmental changes and natural fluctuations since the human introduction of nutrients, such as temperature changes, freshwater input, and chlorophyll concentrations (Nixon et al., 2008; Pilson 2008; Hamburg et al., 2008; Duarte et al., 2009). These changes have shifted the original state of Narragansett Bay to something different that may not be attainable even with reductions in nutrient inputs (Duarte et al., 2009; Oviatt et al., 1984). A complete reversal may not occur once wastewater treatment facilities reduce their nutrient input or it may occur to a lesser degree after several years (Duarte et al., 2009). Therefore, the original state of Narragansett Bay should not be the ultimate goal of reducing nitrogen and phosphorus loads from wastewater treatment facilities, yet maintaining the Narragansett Bay ecosystem in a state that provides worthwhile ecosystem services (Duarte et al., 2009). Despite the frustration that Narragansett Bay may not revert to its original state, it has been argued that Narragansett Bay has been stable for almost 100 years and completely removing all nutrients could in fact be detrimental (Nixon et al., 2008). However, reasonable nitrogen and phosphorus reductions in wastewater treatment facility effluent are important, as they will prevent any further degradation to the Narragansett Bay ecosystem.

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Table B-1. All WWTF included in this study are listed below by the body of water into which they discharge. Facilities that have upgraded to advanced wastewater treatment for the removal of nitrogen are noted below with the year in which they upgraded. All parameters included in the RI DEM dataset are listed below. The frequency of measurements for each parameter is listed in its respective column followed by the years of data included in the RI DEM dataset. “Active” refers to the active season, May to October, and “inactive” refers to the inactive season, November to April. **W = weekly, 3W = 3x/week, 2W = 2x/week, M = monthly, 2M = 2x/month.**

Discharges to: **Narragansett Bay**	**Upgraded**	**TN**	**NH_4^+**	**NO_2 & NO_3**	**TP**
Field’s Point	Sched. 12/2013	W 2002-05 3W 2005-10	2W 2002-10	W 2002-05 3W 2005-10	W 2002-10
Bucklin Point	2006	W 2002-05 3W 2005-10	2W 2002-10	W 2002-05 3W 2005-10	W 2002-10
Newport			*No*	*data*	
East Providence	Sched. 9/2012	W 2002-10	M 2002-07 W 2007-10	W 2002-10	W 2002-04
Bristol		2M 2000-10	2M 2001-04	2M 2000-10	2M 2000-04
Warren	Sched. 12/2015	W 2003-04	W 2003-04	W 2003-04	
East Greenwich	3/2006	W active 2M inactive, 2000-10		W active 2M inactive, 2000-10	
Quonset Point		2000-04	2000-04	2000-04	2000-04
Jamestown		2001-03	2000	2000-03	2000
Fall River, MA			*No*	*data*	
Blackstone River	**Upgraded**	**TN**	**NH_4^+**	**NO_2 & NO_3**	**TP**
Worcester	2009	3W active 2W inactive, 2009-10	3W 2000-02, 2009-10		3W Apr-Oct 2000-02, 2009-10, M Nov-Mar 2000-08, 2W Nov-Mar 2008-10
Woonsocket	2002	3x/week 2000-10, M Nov-Mar 2008-10	3W Jun-Oct W Nov-May 2000-10	3W 2001-10 M Nov-Mar 2008-10	3W 2000-10
Smithfield	6/2006	3W active M inactive	3W Jun-Oct W Nov-May	3W 2000-07 M inactive	3W Jun-Sep W Oct-May

		2007-10	2000-10	2007-10	2001-10
Grafton	*No data*				
Millbury	*No data*				
Northbridge	*No data*				
Burrillville	2001	W 2000-10 3W active 2006-10	W 2000-10 3W active 2006-10	W 2000-10 3W active 2006-10	W 2000-10 3W active 2006-10
Hopedale	*No data*				
Leicester	*No data*				
Douglas	*No data*				
Upton	*No data*				
Ten Mile River	**Upgraded**	**TN**	**NH_4^+**	**NO_2 & NO_3**	**TP**
Attleboro		3W active W inactive 2008-10	3W active 2W inactive 2000-10		3x/week 2000-10, 2W Nov-Mar 2009-2010
North Attleboro	2008	3W active W inactive 2007-10	2W 2000-10		2x/week 2000-10, 3W Apr-Oct 2008-2010
Pawtuxet River	**Upgraded**	**TN**	**NH_4^+**	**NO_2 & NO_3**	**TP**
Cranston	1/2006	W Jun-Sep 2M Oct-May 2000-10	W 2000-10	W Jun-Sep 2M Oct-May 2000-10	W 2000-10
West Warwick	7/2005	W Jun-Oct 2M Nov-May 2000-10	W 2000-10	W Jun-Oct 2M Nov-May 2000-10	W 2000-10
Warwick	11/2004	W Jun-Oct 2M Nov-May 2000-10	W 2000-10	W Jun-Oct 2M Nov-May 2000-10	W 2000-10
Taunton River	**Upgraded**	**TN**	**NH_4^+**	**NO_2 & NO_3**	**TP**
Brockton	*No data*				
Taunton	*No data*				
Somerset	*No data*				

Table B-2. The average value from 2007-2010 for each parameter discharged per year from each facility in the Narragansett Bay watershed is displayed below. All values with the exception of flow are in millions of moles per year. Flow values are in thousands of cubic meters per day. "NO_2 + NO_3" is the sum of nitrite (NO_2) and nitrate (NO_3). "DIN" is the sum of ammonium (NH_4^+), nitrite (NO_2), and nitrate (NO_3). Nutrients were not monitored at the Newport facility. * indicates that parameter values were calculated by scaling previous values, 2000-2003 (Nixon, 2008), by the population change from 2000-2010.

Discharges to:	**Flow**	**NH_4^+**	**NO_2**	**NO_3**	**NO_2+NO_3**	**DIN**	**TN**	**TP**
Narragansett Bay								
Field's Point	168.3	37.40	3.23	5.84	9.07	46.47	63.50	3.19
Bucklin Point	80.9	1.27	0.23	13.70	13.93	15.20	18.90	3.14
Newport	34.8	*Nutrients*	*not*	*monitored*			10.50	0.59
East Providence	26.9	3.28	0.13	2.93	3.06	6.34	7.53	0.52
Bristol	13.5	1.94	0.17	1.93	2.10	4.04	6.27	0.18
Warren	7.3	1.35	0.02	0.22	0.24	1.59	1.86	0.05
East Greenwich	4.1	0.86	0.01	0.46	0.47	1.33	0.87	0.42
Quonset Point	1.8		0.04	0.46	0.50		0.73	0.10
Jamestown		0.05	0.00	0.10	0.10	0.15	0.16	0.02
Fall River*		22.90			2.05	24.95	33.20	1.15
Total		**69.04**	**3.83**	**25.64**	**31.51**	**100.06**	**143.52**	**9.37**
Blackstone River								
Worcester[1]	117.7	3.01			11.01	14.02	16.60	1.07
Woonsocket	28.3	0.97	0.06	3.24	3.30	4.27	4.99	0.56
Smithfield	7.6	0.18	0.07	1.04	1.11	1.29	1.46	0.02
Grafton*		2.00			1.34	3.34	3.28	0.14
Millbury*		1.96			0.46	2.42	2.44	0.24
Northbridge*		1.48			0.43	1.91	3.06	0.17
Burrillville	3.2	0.99	0.07	0.23	0.30	1.29	1.40	0.02
Hopedale*		0.13						0.02
Leicester*		0.03						0.00
Douglas*		0.10			0.05	0.14	0.20	0.02
Upton*		0.07			0.01	0.08	0.12	0.00
Total		**10.92**	**0.20**	**4.51**	**18.00**	**28.75**	**33.55**	**2.27**
Ten Mile River								
Attleboro	15.4	0.45					7.67	0.02
North Attleboro	16.2	0.41					2.98	0.03
Total		**0.86**					**10.65**	**0.06**
Pawtuxet River								
Cranston	42.9	3.96	0.12	5.98	6.10	10.06	12.50	0.43
West Warwick	22.7	1.01	0.36	5.37	5.73	6.74	8.03	0.45
Warwick	18.9	1.43	0.06	2.39	2.45	3.88	4.75	0.21
Total		**6.40**	**0.54**	**13.74**	**14.28**	**20.68**	**25.28**	**1.09**
Taunton River								
Brockton*		15.72			11.84	27.56	36.51	0.83
Taunton*		2.04					4.18	0.29
Somerset*		2.68			0.76	3.44	8.28	0.17
Total		**20.43**			**12.60**	**30.99**	**48.97**	**1.28**
GRAND TOTAL							**262.0**	**14.1**

[1] Flow value is the average of flows from 2009-2010 instead of 2007-2010 as there was no flow data available for 2007 and 2008.

Table B-3. Average annual and active season total nitrogen concentrations during 2000-2004 and 2007-2010 for all facilities with nitrogen concentrations available. Nitrogen limits, when applicable, are listed below the average concentrations for each time period. All values are in mg/L. Gray shading indicates compliance with the limit, while yellow shading indicates non-compliance with limits currently in effect.

Discharges to:	ANNUAL		ACTIVE	
Narragansett Bay	**2000-2004**	**2007-2010**	**2000-2004**	**2007-2010**
Field's Point	14.8	13.6	14.8	13.5
	5.0	*mg/L*[a]	*5.0*	*mg/L*[a]
Bucklin Point	15.4	7.8	15.9	7.6
	8.0	*mg/L*[b]	*8.0*	*mg/L*[b]
East Providence	15.3	11.3	15.0	11.8
	5.9	*mg/L*[c]	*5.9*	*mg/L*[c]
Bristol	24.6	25.0	27.6	27.0
Warren	12.7	*no data*	14.6	*no data*
			5.0	*mg/L*[d]
East Greenwich	10.5	8.1	9.8	3.8
	5.0	*mg/L*	*5.0*	*mg/L*
Quonset Point	16.3	*no data*	16.7	*no data*
Jamestown	7.7	*no data*	7.7	*no data*
Blackstone River				
Worcester	*no data*	6.0	*no data*	6.3
	5.0	*mg/L*[e]	*5.0*	*mg/L*[e]
Woonsocket	17.0	6.7	16.9	5.9
	5.0	*mg/L*[f]	*5.0*	*mg/L*[f]
Smithfield	19.3	7.9	19.3	7.9
	max	*extent*	*max*	*extent*
Burrillville	16.8	15.8	14.4	10.4
	max	*extent*	*max*	*extent*
Ten Mile River				
Attleboro	*no data*	21.8	*no data*	23.5
	8.0	*mg/L*[e]	*8.0*	*mg/L*[e]
North Attleboro	*no data*	6.8	*no data*	6.5
	8.0	*mg/L*[g]	*8.0*	*mg/L*[g]
Pawtuxet River				
Cranston	16.2	11.0	15.0	8.0
	8.0	*mg/L*	*8.0*	*mg/L*
West Warwick	15.3	13.5	15.4	10.6
	8.0	*mg/L*	*8.0*	*mg/L*
Warwick	20.5	9.6	19.0	9.4
	8.0	*mg/L*	*8.0*	*mg/L*

[a] Planned to be completed 12/6/13.

[b] Nitrogen limit of 5.0 mg/L planned to be completed 3/1/14.

[c] Planned to be completed 9/1/12.

[d] Nitrogen limits of 5.0 mg/L (May-Oct) and 14.3 mg/L (Nov-Apr) planned to be completed 12/1/15.

[e] Was planned to be completed by the end of 2011.

[f] Nitrogen limit of 3.0 mg/L planned to be completed 3/31/14.

[g] Planned to be completed by the close of 2012.

Table B-4. Average annual and active season total phosphorus concentrations during 2000-2004 and 2007-2010 for all facilities with phosphorus concentrations available. Phosphorus limits, when applicable, are listed below the average concentrations for each time period. All values are in mg/L. Gray shading indicates compliance with the limit, while yellow shading indicates non-compliance with limits currently in effect.

Discharges to:	**ANNUAL**		**ACTIVE**	
Narragansett Bay	**2000-2004**	**2007-2010**	**2000-2004**	**2007-2010**
Field's Point	1.2	1.4	0.9	1.6
Bucklin Point	2.2	2.4	1.7	2.3
East Providence	2.6	*no data*	2.9	*no data*
Bristol	1.3	*no data*	1.3	*no data*
Warren	*no data*	*no data*	*no data*	*no data*
East Greenwich	13.5	*no data*	*no data*	*no data*
Quonset Point	*no data*	*no data*	*no data*	*no data*
Jamestown	4.7	*no data*	*no data*	*no data*
Blackstone River				
Worcester	1.4	0.6	1.5	0.9
Woonsocket	3.6	1.6	3.8	0.5
		1.0 mg/L[a]		*1.0 mg/L*[a]
Smithfield	3.2	0.2	3.2	0.2
		0.2 mg/L		*0.2 mg/L*[b]
Burrillville	0.8	0.6	0.9	0.7
Ten Mile River				
Attleboro	0.4	0.1	0.4	0.1
North Attleboro	0.7	0.1	0.7	0.1
Pawtuxet River				
Cranston	3.5	<0.1	3.5	0.7
		1.0 mg/L[c]		*1.0 mg/L*[c]
West Warwick	2.8	1.7	3.1	1.3
		1.0 mg/L[d]		*1.0 mg/L*[d]
Warwick	2.9	1.0	3.4	1.1
		1.0 mg/L[e]		*1.0 mg/L*[e]

[a] Phosphorus limit of 0.1 mg/L planned to be completed 3/31/14.
[b] Phosphorus limit planned for April – October as of 12/20/12.
[c] Phosphorus limit of 0.1 mg/L planned to be completed 3/31/13.
[d] Phosphorus limit of 0.1 mg/L planned to be completed 4/1/14.
[e] Phosphorus limit of 0.1 mg/L planned to be completed 9/30/13.

Table B-5. Flow, nitrogen, and phosphorus discharged from rivers that drain to Narragansett Bay in 2003-2004 (Nixon, et al., 2008) and from 2008-2010. All flow values are in millions of cubic meters per day and nitrogen and phosphorus values are in millions of moles per year.

	2003-2004		2008-2010	
	N	P	N	P
Blackstone River				
Mean Daily Flow	2.57		3.14	
Dissolved Inorganic	68.88	1.69	67.32	2.48
Total	98.63	3.87	96.13	5.36[a]
Pawtuxet River				
Mean Daily Flow	1.00		1.28	
Dissolved Inorganic	44.61	1.96	29.73	0.89
Total	59.29	3.61	42.60	1.63[a]
Woonasquatucket River				
Mean Daily Flow	0.28		0.29	
Dissolved Inorganic	6.62	0.16	4.10	0.03
Total	8.59	0.32	5.72	0.07[a]
Moshassuck River				
Mean Daily Flow	0.19		0.12	
Dissolved Inorganic	3.50	0.07	2.04	0.01
Total	4.77	0.13	2.68	0.02[a]
Ten Mile River				
Mean Daily Flow	0.35		0.33	
Dissolved Inorganic	9.86	0.24	11.84	0.08
Total	14.07	0.81	14.39	0.27[a]
Taunton River				
Mean Daily Flow	2.58[c]		3.46	
Dissolved Inorganic	86.00[c]	3.30[c]	51.25	0.76
Total	117.00[c]	5.30[c]	82.09	1.22[b]
Unmeasured Flow				
Mean Daily Flow	1.48[d]		1.48[e]	
Dissolved Inorganic	48.30	1.60	27.70	0.75
Total	66.50	3.10	39.80	1.65
GRAND TOTAL				
Mean Daily Flow				
Dissolved Inorganic	267.80	9.05	193.98	5.00
Total	368.90	17.13	283.41	2.87

[a] Calculated from average ratio of inorganic to total phosphorus (Nixon, et al., 2008).

[b] Calculated from the average of the average ratios of inorganic to total phosphorus (Nixon, et al., 2008).

[c] Data from (Boucher, 1991) as presented in (Nixon, et al., 1995).

[d] Based on calculation of area of gauged to ungauged river area by (Ries, et al., 1990) as modified by (Nixon, et al., 1995).

[e] Based on Ries, et al., (1990) plus flow from 304 mi^2 of un-gauged flow in the Taunton basin.

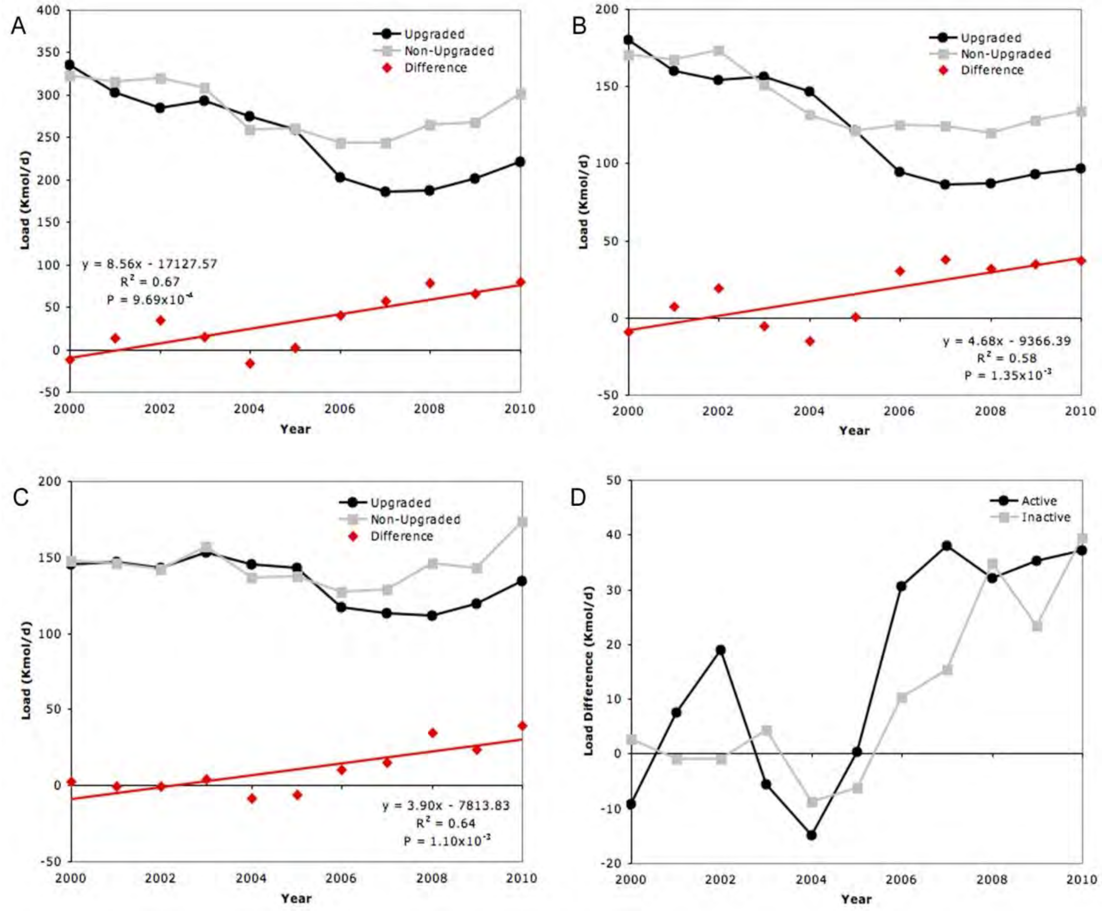

Figure B-1. Total nitrogen load from 2000-2010. (A) Annual total nitrogen load discharged from upgraded facilities (black) and non-upgraded facilities (gray) with the difference between the two (red). (B) Active season total nitrogen load discharged from upgraded facilities (black) and non-upgraded facilities (gray) with the difference between the two (red). (C) Inactive season total nitrogen load discharged from upgraded facilities (black) and non-upgraded facilities (gray) with the difference between the two (red). (D) The difference in total nitrogen between upgraded and non-upgraded facilities during the active season (black) and the difference in total nitrogen between upgraded and non-upgraded facilities during the inactive season (gray).

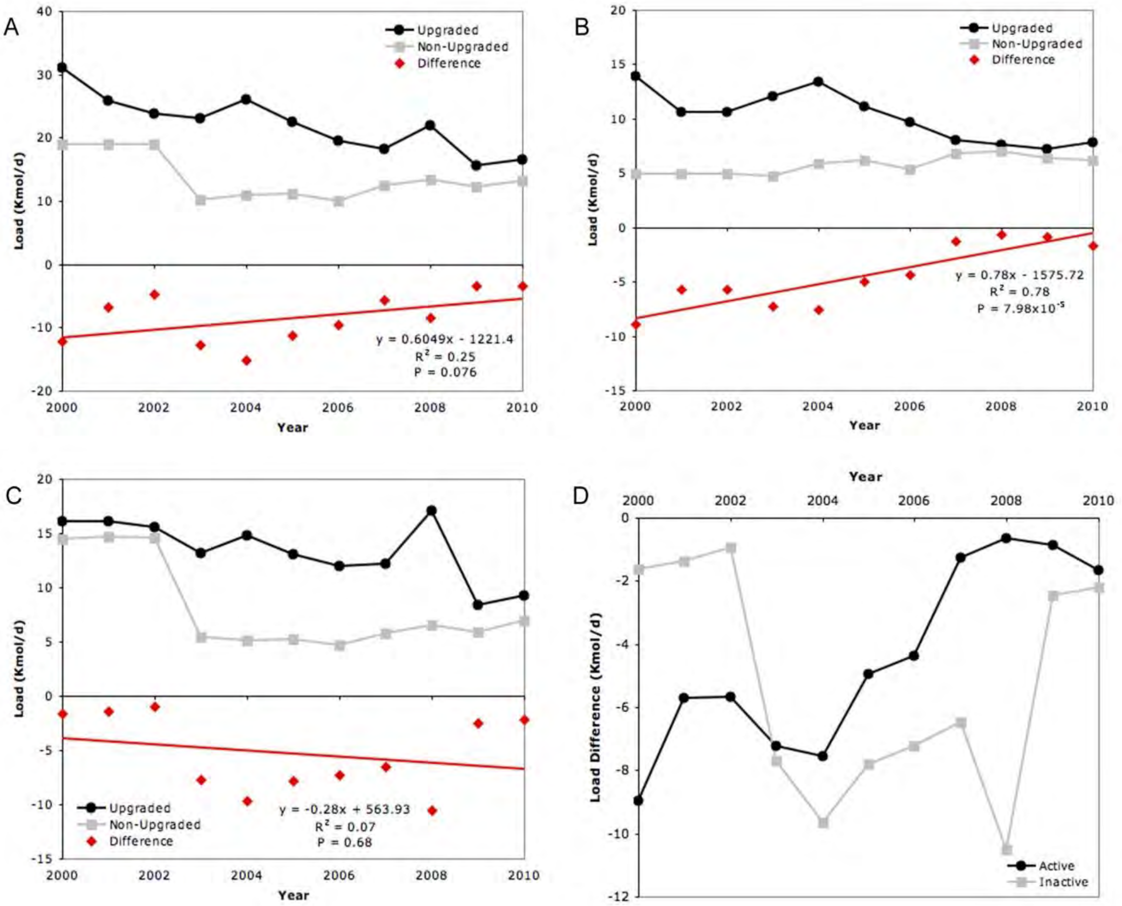

Figure B-2. Total phosphorus load from 2000-2010. (A) Annual total phosphorus load discharged from upgraded facilities (black) and non-upgraded facilities (gray) with the difference between the two (red). (B) Active season total phosphorus load discharged from upgraded facilities (black) and non-upgraded facilities (gray) with the difference between the two (red). (C) Inactive season total phosphorus load discharged from upgraded facilities (black) and non-upgraded facilities (gray) with the difference between the two (red). (D) The difference in total phosphorus between upgraded and non-upgraded facilities during the active season (black) and the difference in total phosphorus between upgraded and non-upgraded facilities during the inactive season (gray).

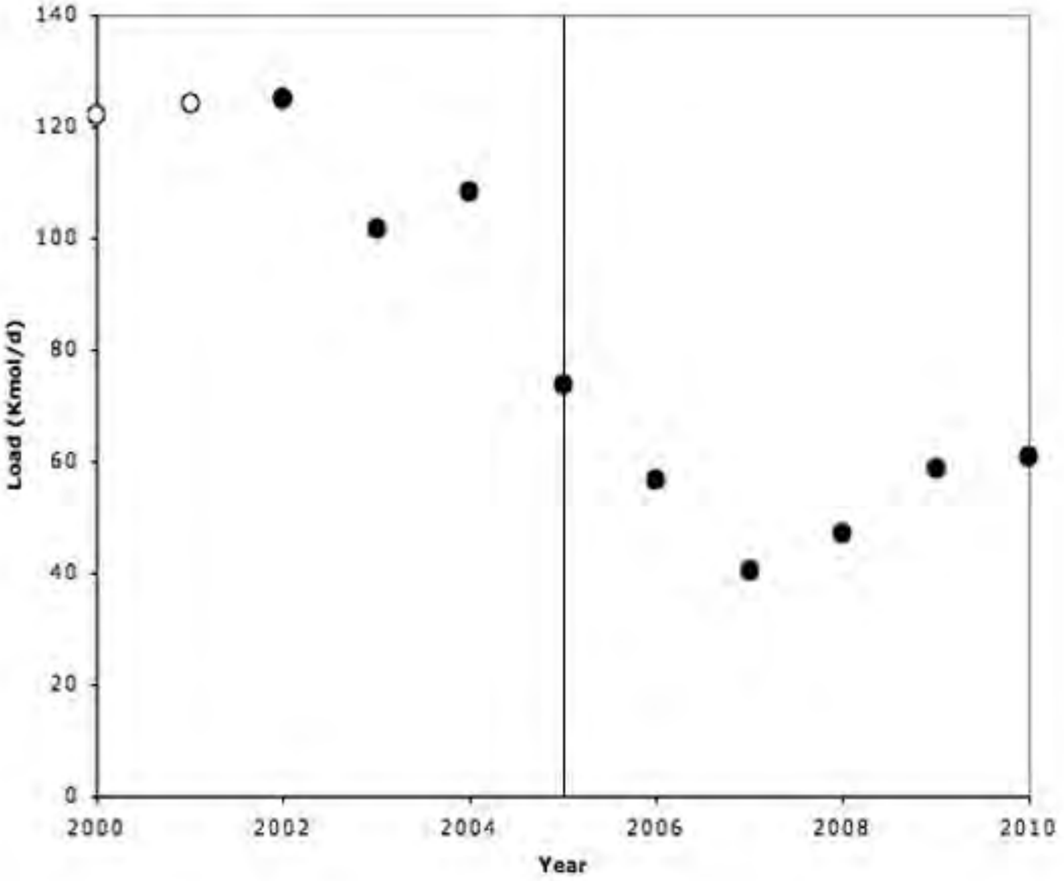

Figure B-3. Average annual total sewage nitrogen discharged from the Bucklin Point facility from 2000-2010. The vertical line represents upgrade completion in 12/2005. Open circles represent data that was estimated using population data and available load data. Closed circles represent actual data.

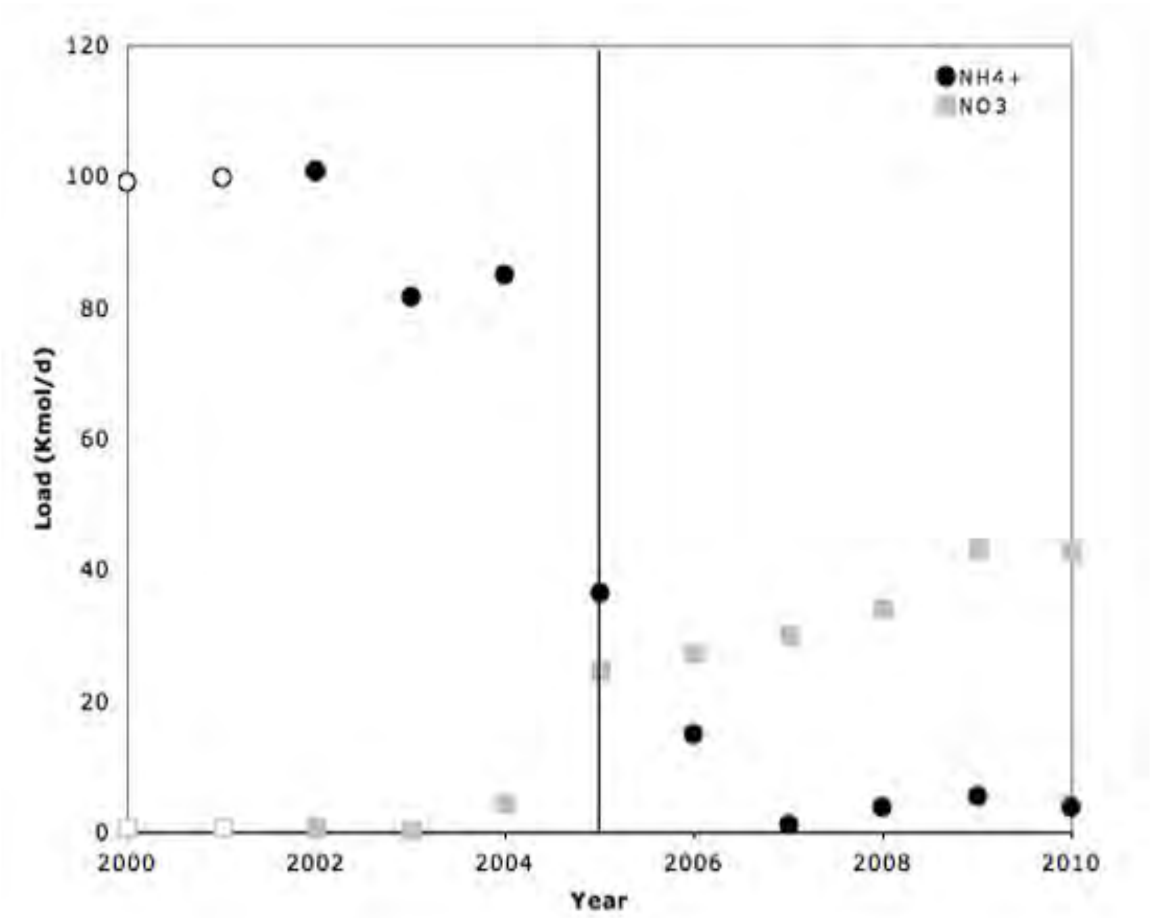

Figure B-4. Average annual sewage ammonium (NH_4^+) and nitrate (NO_3) discharged from the Bucklin Point facility from 2000-2010. The vertical line represents upgrade completion in 12/2005. Open points represent data that was estimated using population data and available load data. Closed points represent actual data.

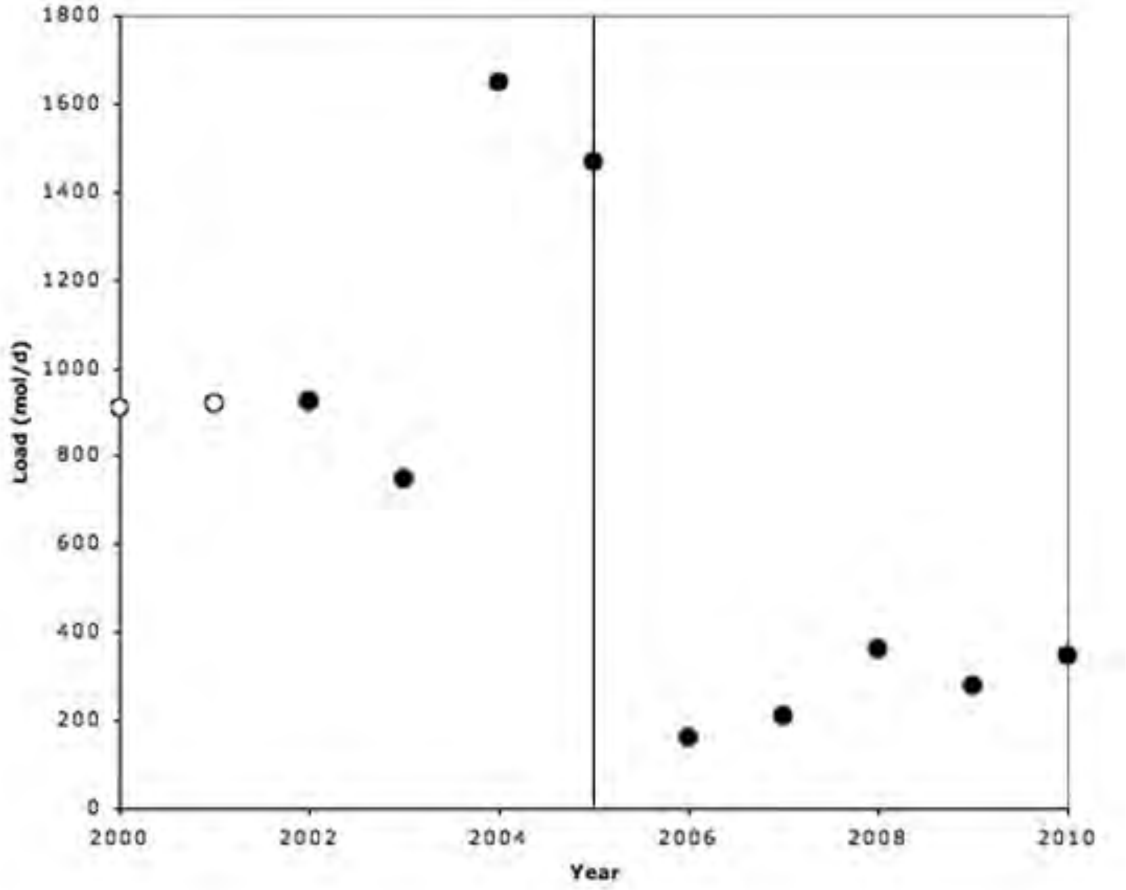

Figure B-5. Average active season sewage nitrite discharged from the Bucklin Point facility from 2000-2010. The vertical line represents upgrade completion in 12/2005. Open circles represent data that was estimated from population data and available load data. Closed circles represent actual data.

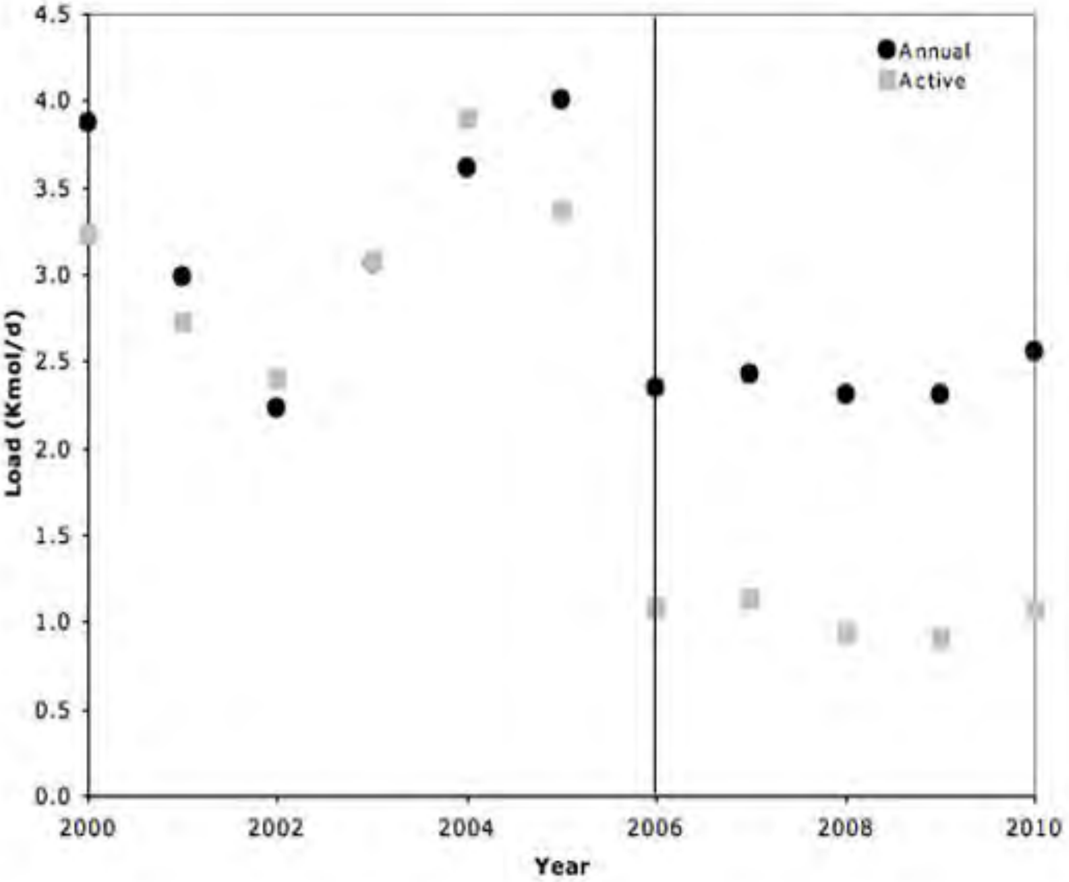

Figure B-6. Average annual and active season total sewage nitrogen load discharged from the East Greenwich facility from 2000-2010. The vertical line represents upgrade completion in 3/2006.

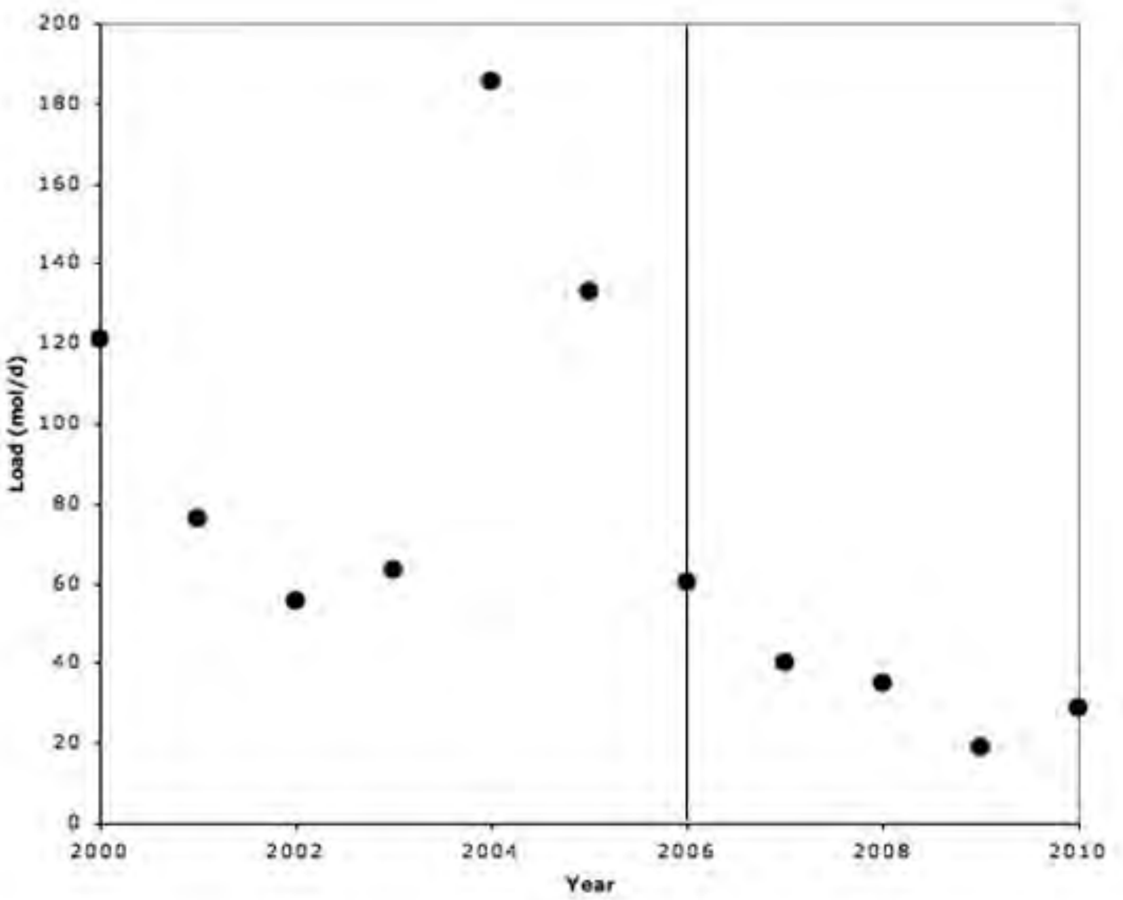

Figure B-7. Average annual sewage nitrite discharged from the East Greenwich facility from 2000-2010. The vertical line represents upgrade completion in 3/2006.

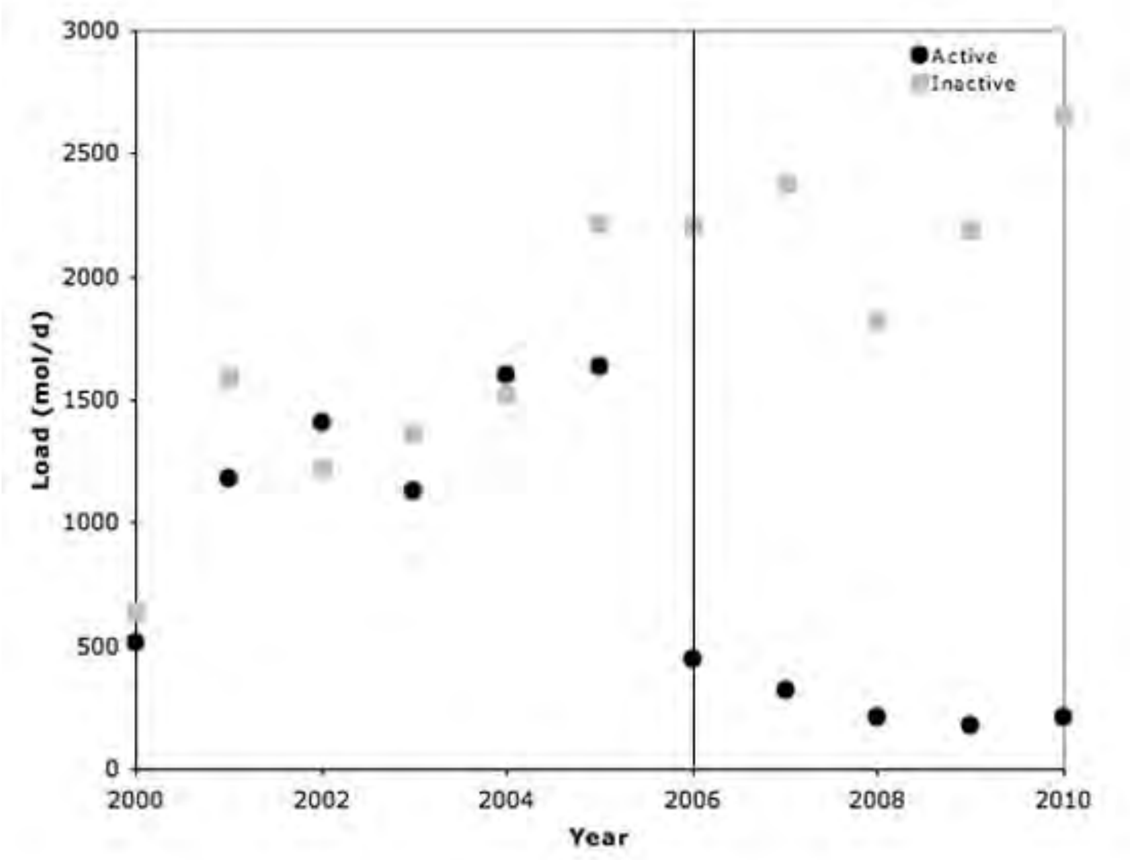

Figure B-8. Average active and inactive season sewage nitrate discharged from the East Greenwich facility from 2000-2010. The vertical line represents upgrade completion in 3/2006.

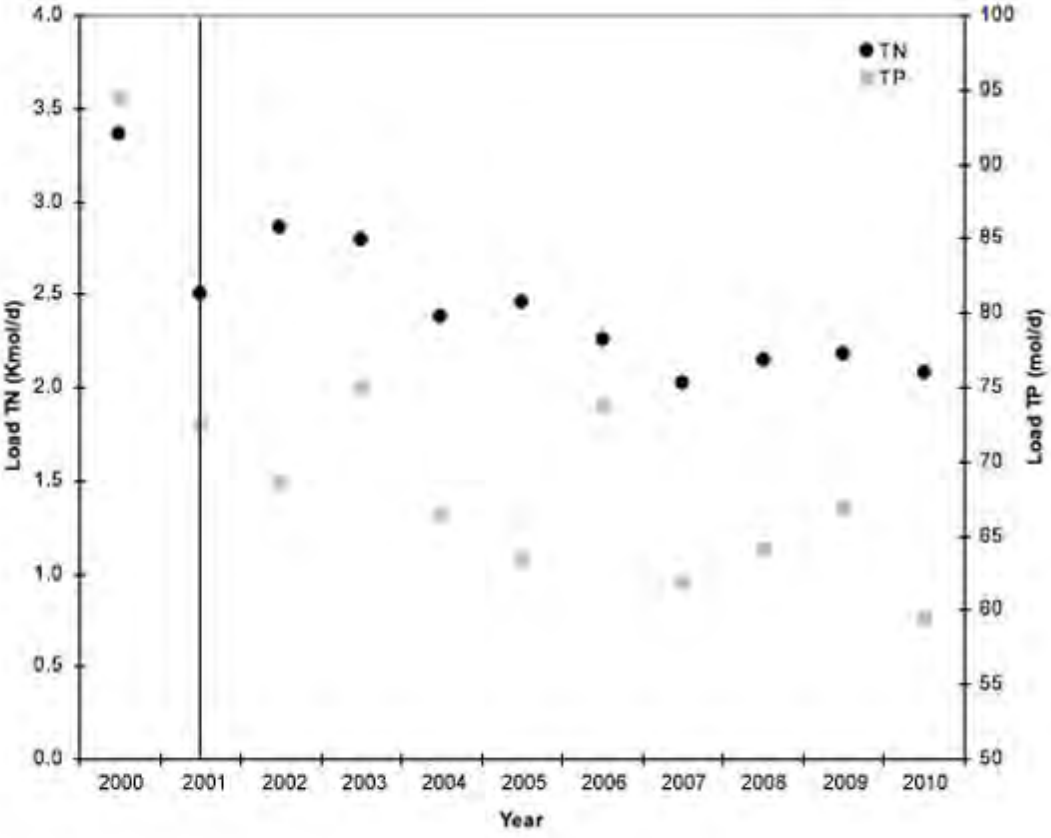

Figure B-9. Average active season total sewage nitrogen (TN) and phosphorus (TP) discharged from the Burrillville facility from 2000-2010. The vertical line represents upgrade completion during 2001.

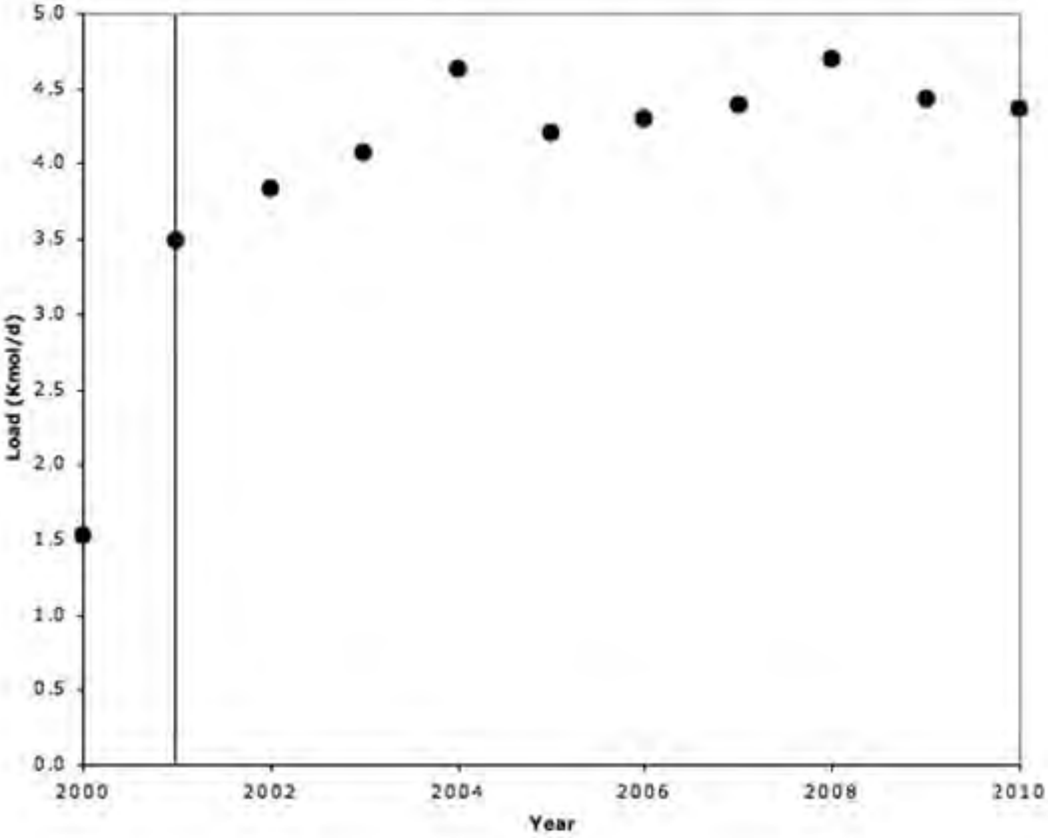

Figure B-10. Average inactive season sewage ammonium discharged from the Burrillville facility from 2000-2010. The vertical line represents upgrade completion during 2001.

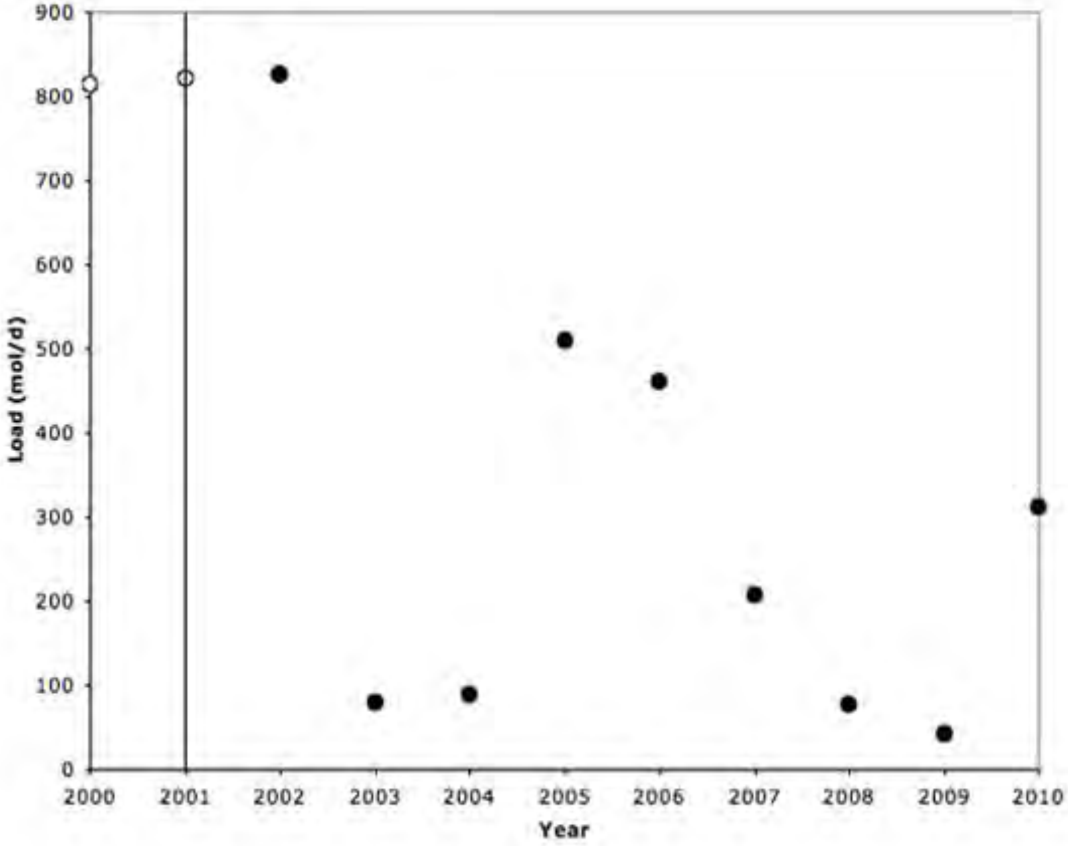

Figure B-11. Average annual sewage nitrite discharged by the Woonsocket facility from 2000-2010.The vertical line represents upgrade completion in 9/2001. Open circles represent data that was estimated with population data and available load data. Closed circles represent actual data.

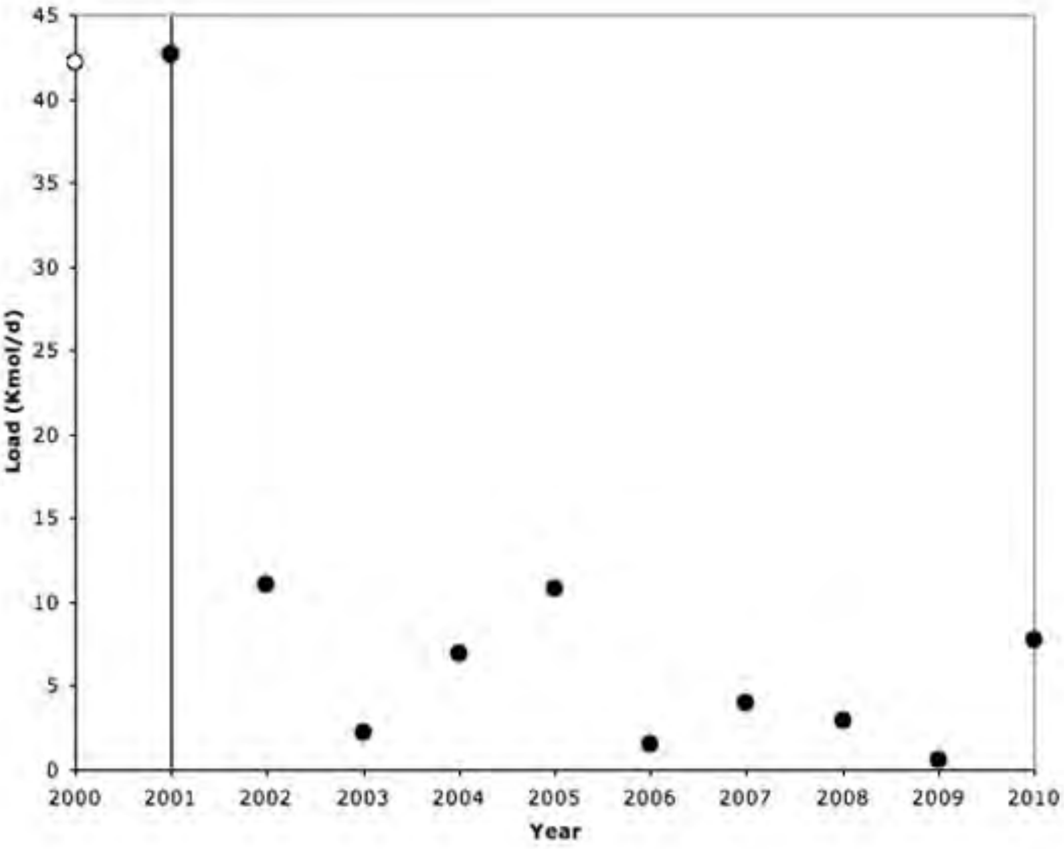

Figure B-12. Average inactive season sewage ammonium discharged from the Woonsocket facility from 2000-2010. The vertical line represents upgrade completion in 9/2001. Open circles represent data that was estimated from population data and available load data. Closed circles represent actual data.

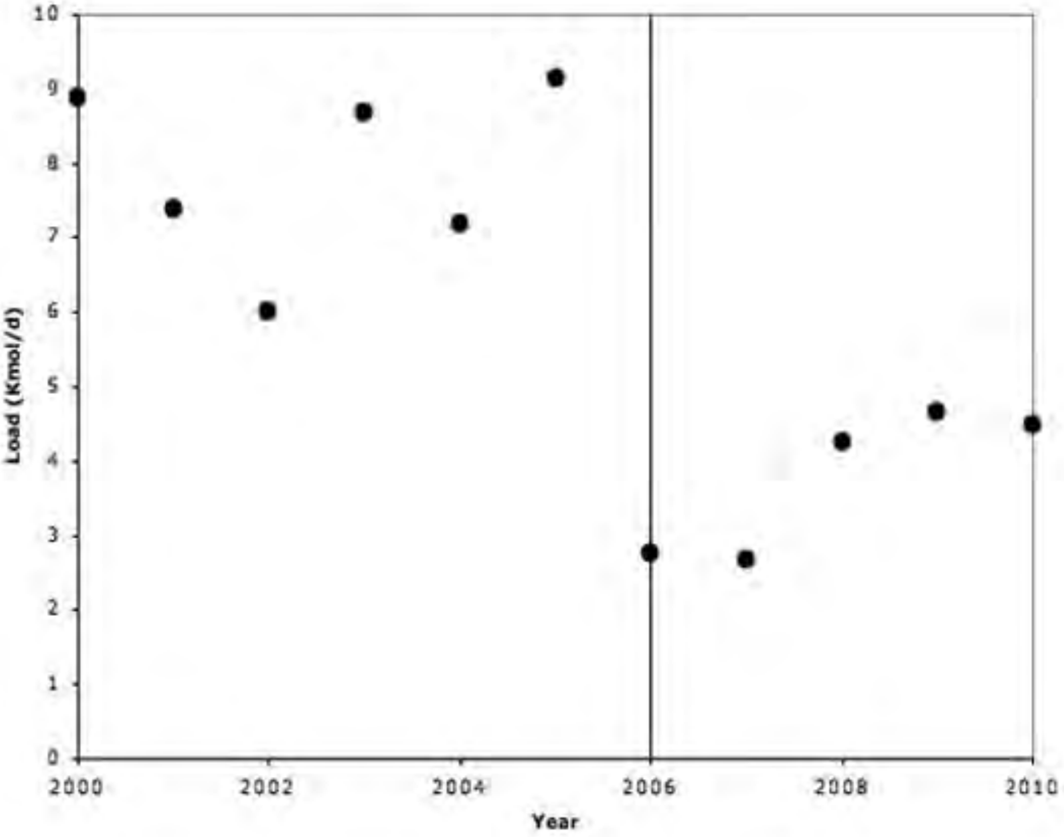

Figure B-13. Average annual total nitrogen discharged from the Smithfield facility from 2000-2010. The vertical line represents upgrade completion in 6/2006.

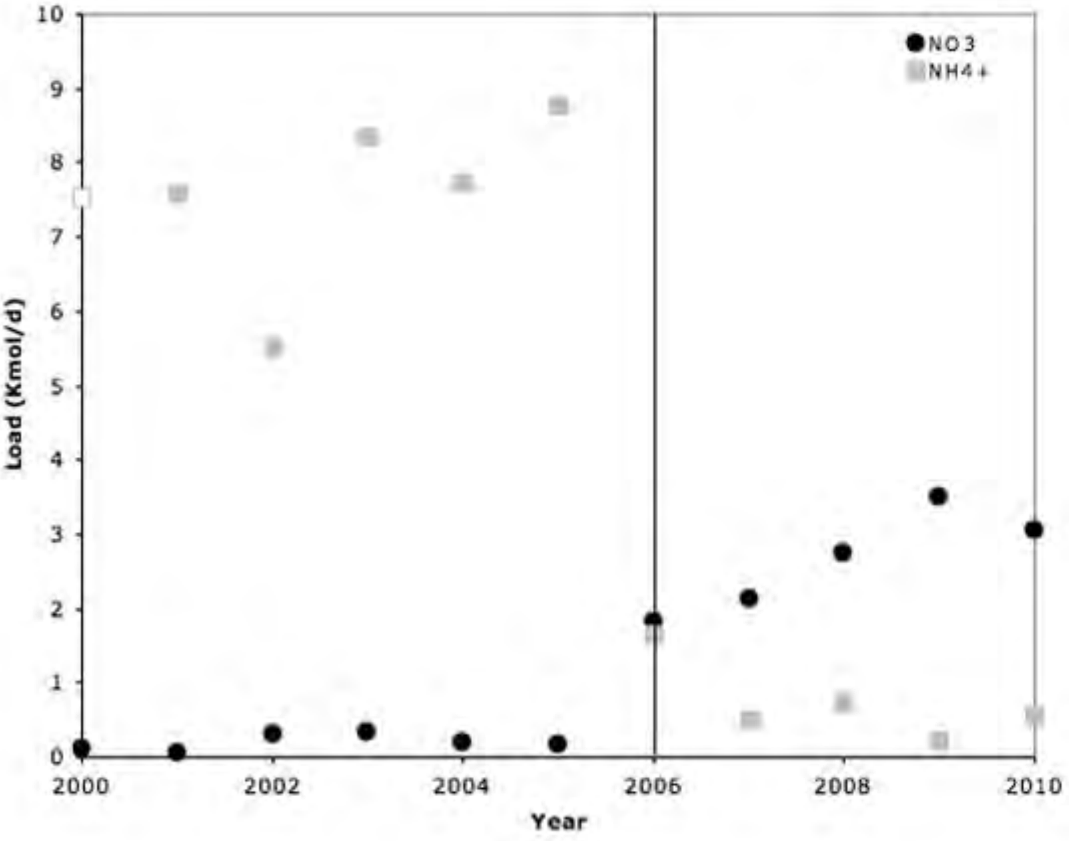

Figure B-14. Average annual ammonium (NH4+) and nitrate (NO3) loads discharged from the Smithfield facility from 2000-2010. The vertical line represents upgrade completion in 6/2006. Open points represent data that was estimated with population data and available load data. Closed points represent actual data.

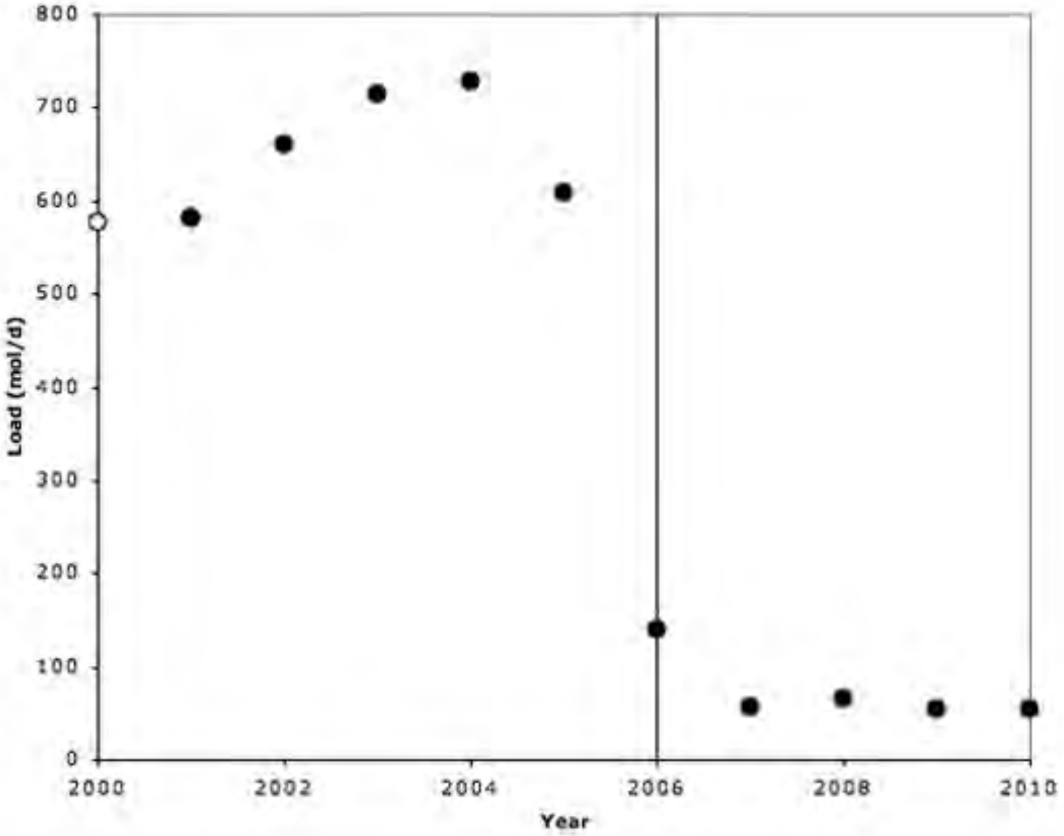

Figure B-15. Average annual total phosphorus discharged from the Smithfield facility from 2000-2010. The vertical line represents upgrade completion in 6/2006. Open circles represent data that was estimated from population data and available load data. Closed circles represent actual data.

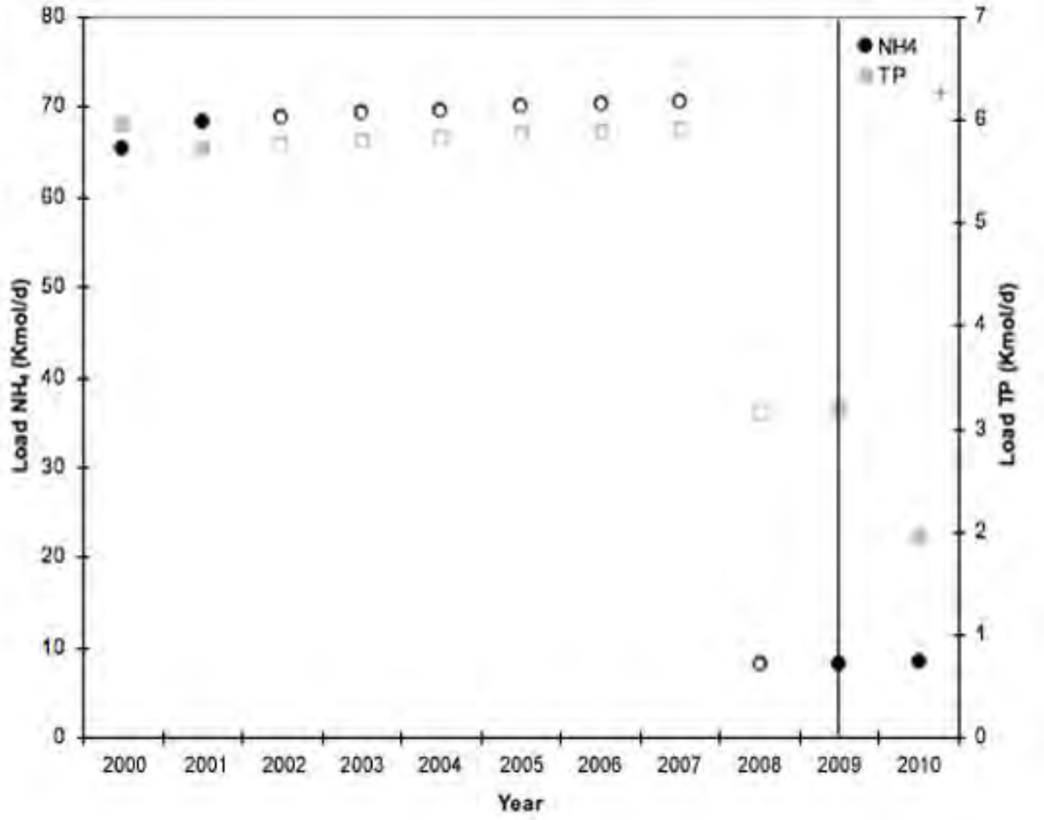

Figure B-16. Average annual sewage ammonium (NH4+) and total sewage phosphorus discharged from the Worcester facility from 2000-2010. The vertical line represents upgrade completion in 2009. Open points represent data that was estimated with population data and available load data. Closed circles represent actual data.

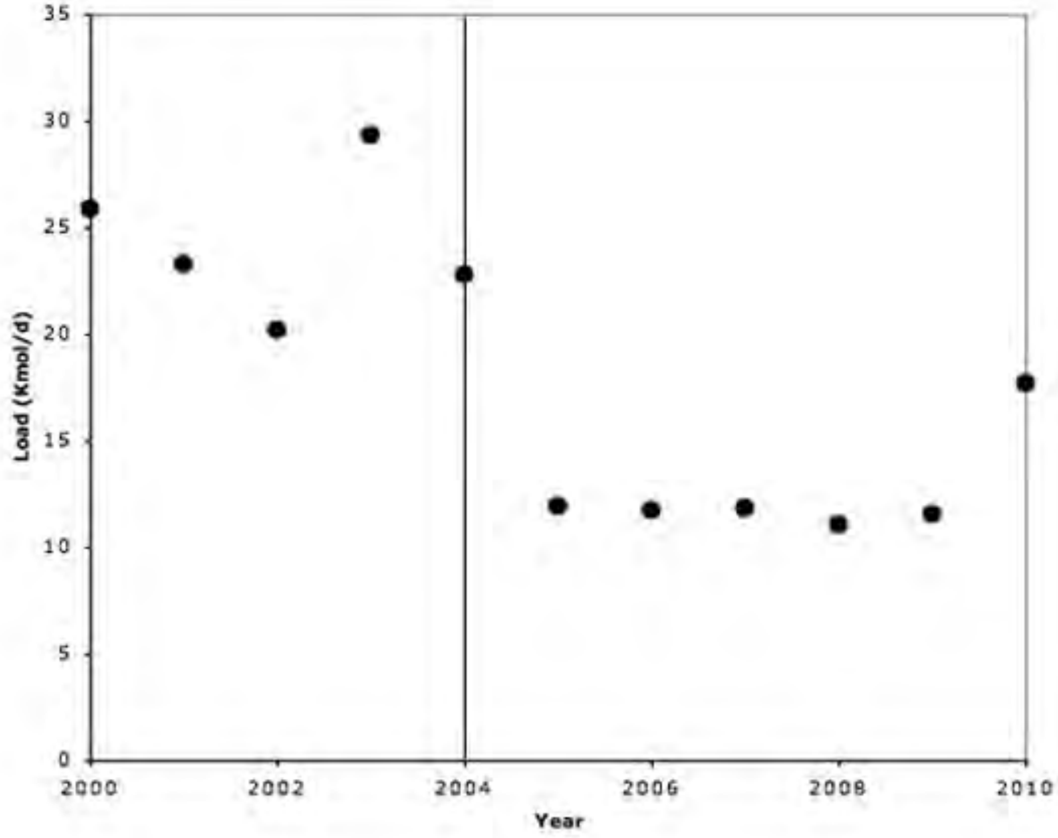

Figure B-17. Average annual total sewage nitrogen discharged from the Warwick facility from 2000-2010. The vertical line represents upgrade completion in 11/2004.

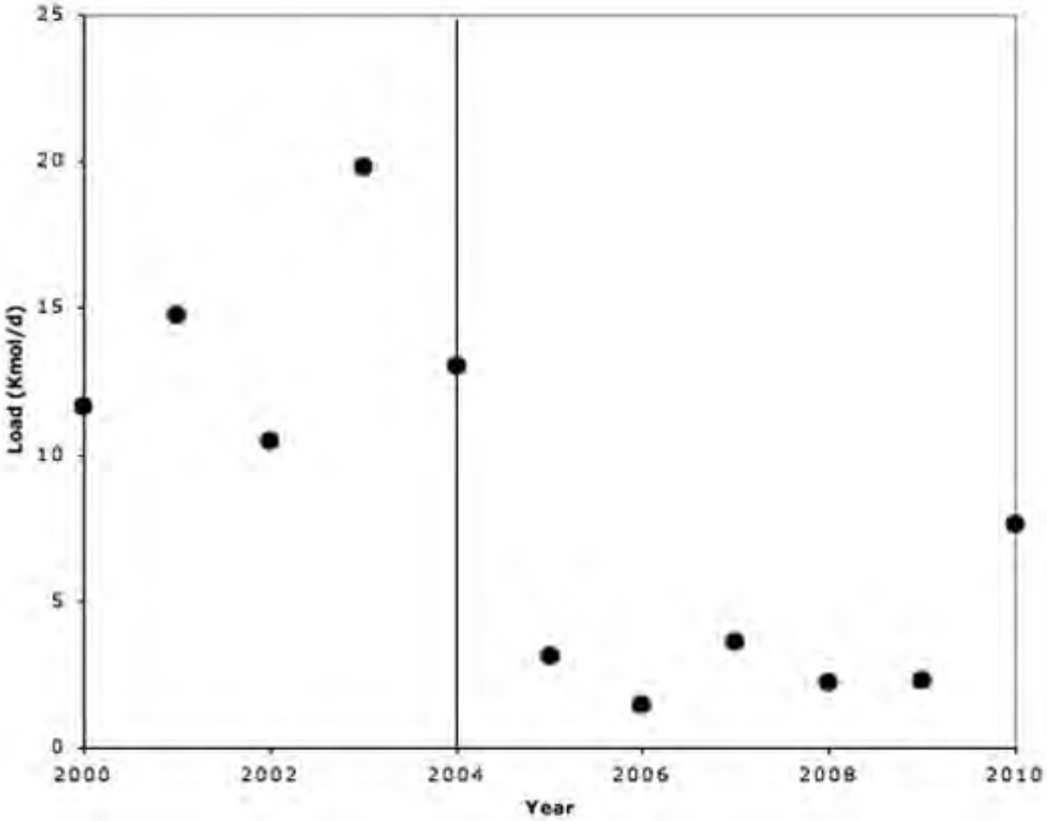

Figure B-18. Average annual sewage ammonium discharged from the Warwick facility from 2000-2010. The vertical line represents upgrade completion in 11/2004.

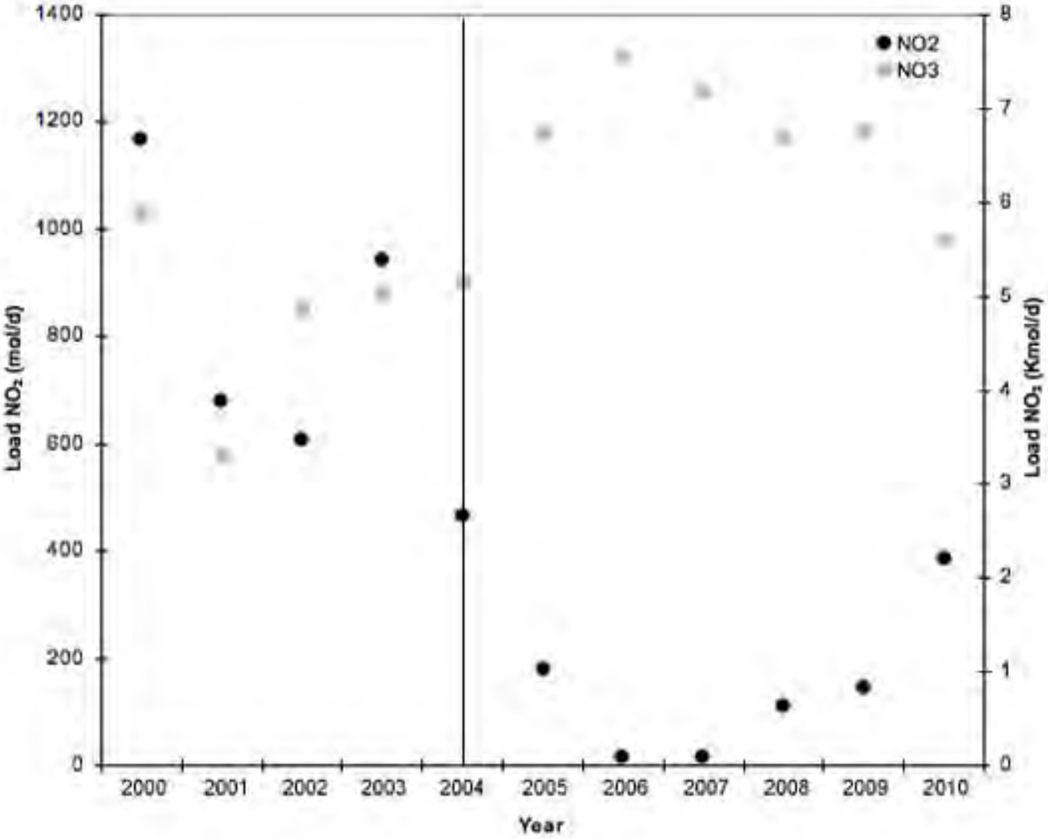

Figure B-19. Average annual nitrite (NO2) load and nitrate (NO3) load discharged from the Warwick facility from 2000-2010. The vertical line represents upgrade completion in 11/2004.

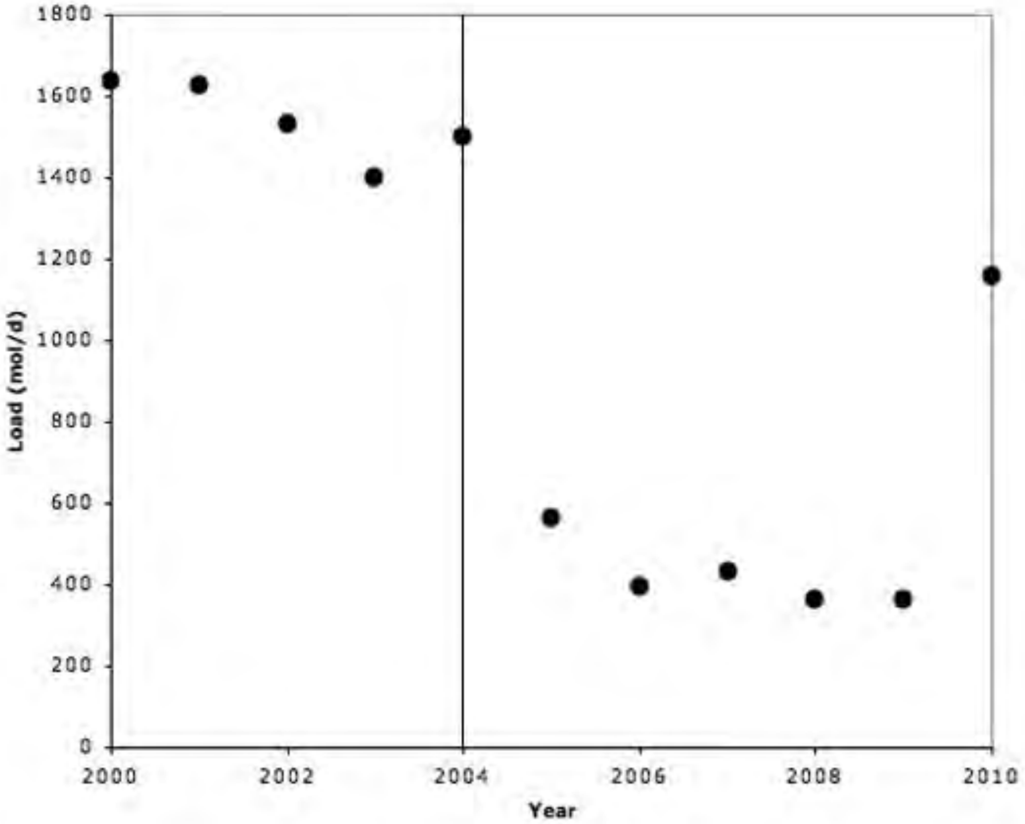

Figure B-20. Average annual total sewage phosphorus load discharged from the Warwick facility from 2000-2010. The vertical line represents upgrade completion in 11/2004.

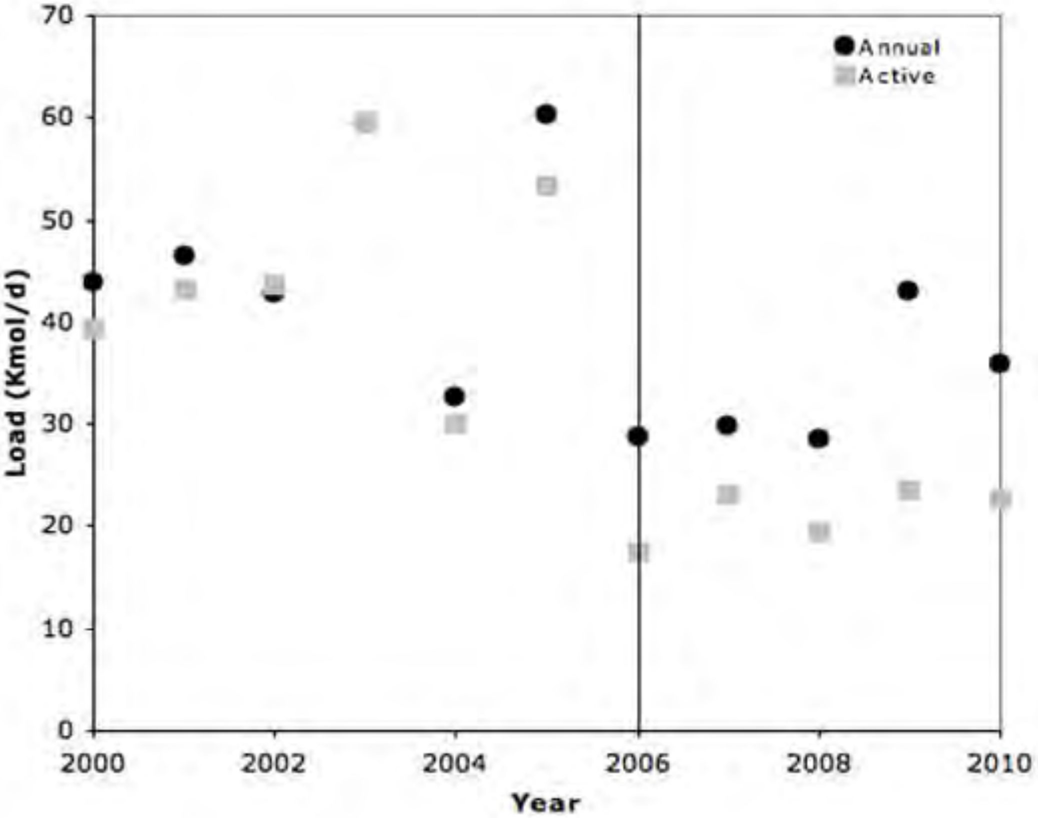

Figure B-21. Average annual and active season total nitrogen discharged from the Cranston facility from 2000-2010. The vertical line represents upgrade completion in 1/2006.

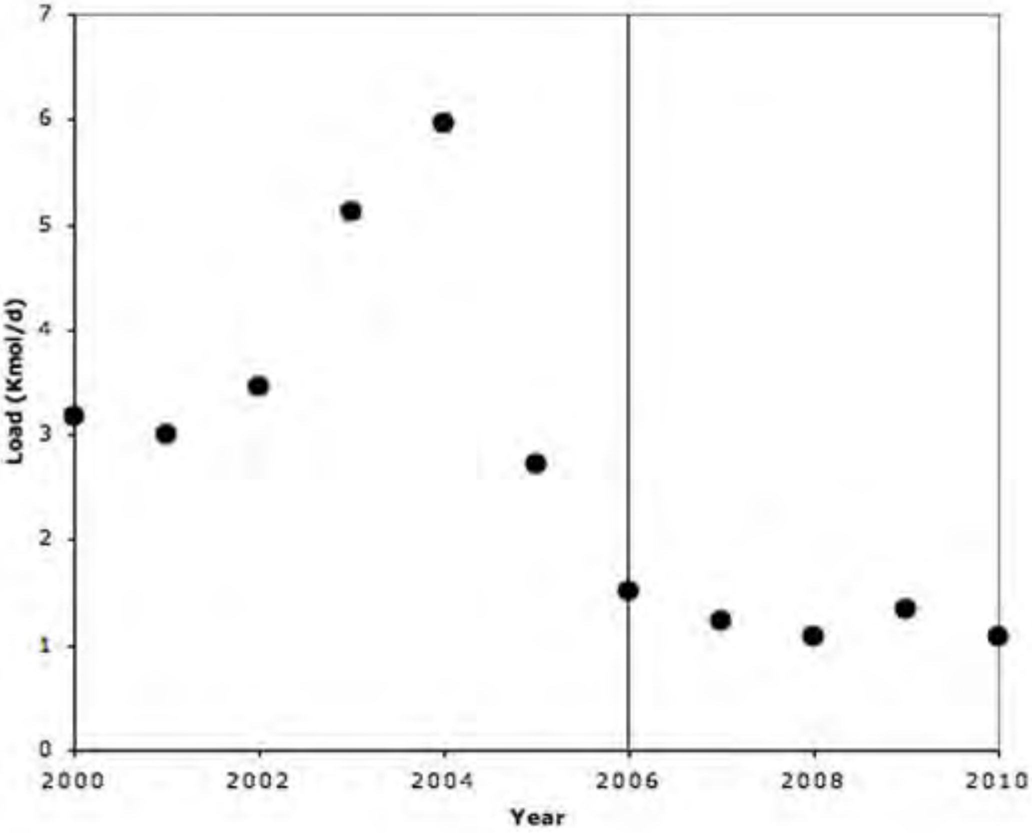

Figure B-22. Average annual total phosphorus load discharged from the Cranston facility from 2000-2010. The vertical line represents upgrade completion in 1/2006.

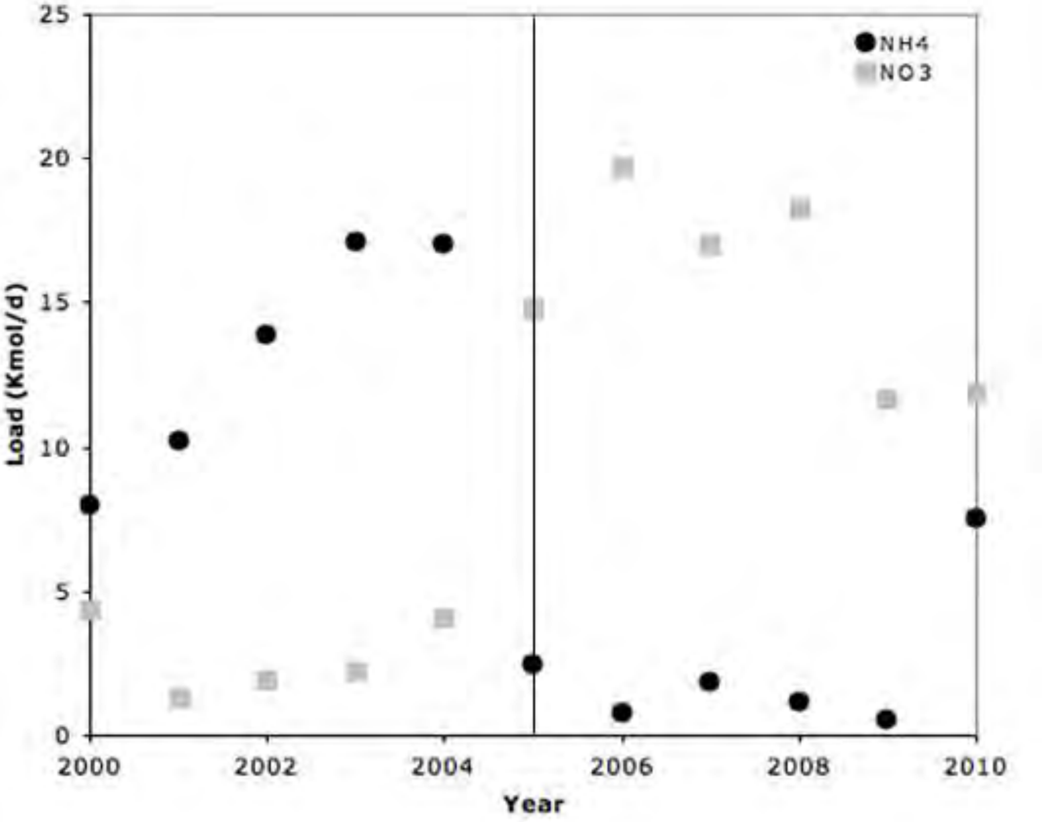

Figure B-23. Average annual ammonia (NH4) and nitrate (NO3) loads discharged from the West Warwick facility from 2000-2010. The vertical line represents upgrade completion in 7/2005.

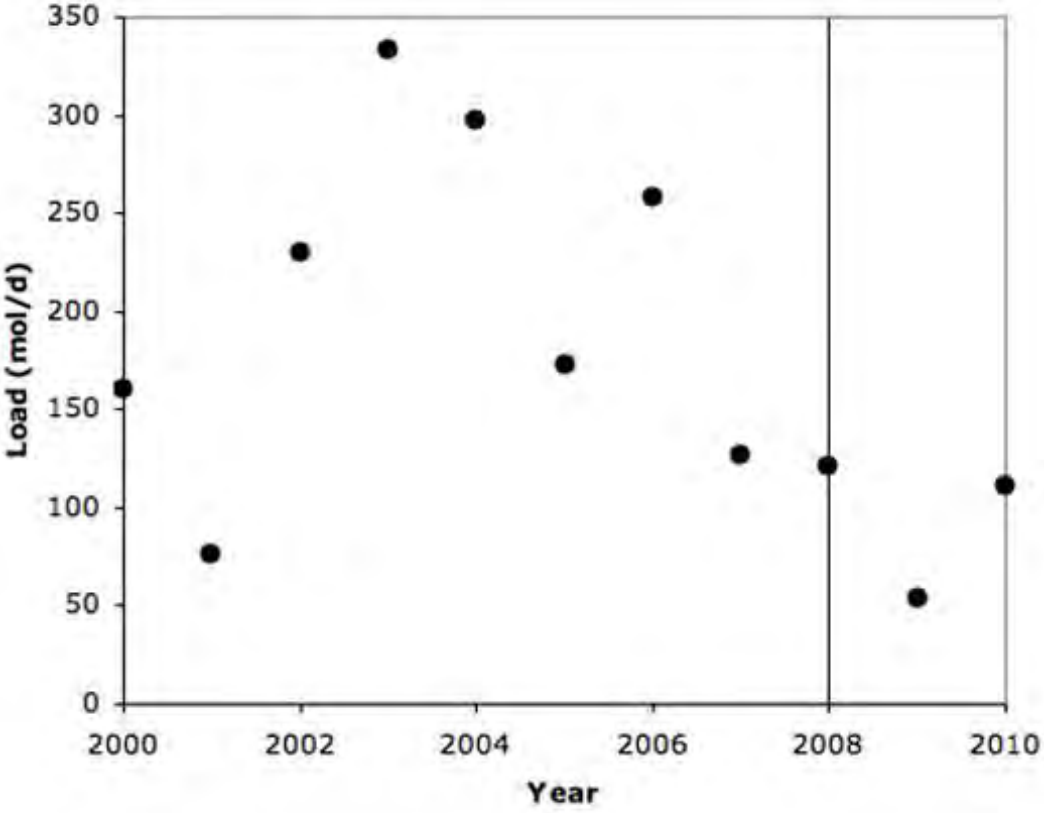

Figure B-24. Average annual total sewage phosphorus discharged from the North Attleboro facility from 2000-2010. The vertical line represents upgrade completion in 2008.

SUPPLEMENTAL FIGURES

Table B-6. Annual total nitrogen load discharged from each facility from 2000-2010. All values are in millions of moles N per year.

Discharges to:	**2000**	**2001**	**2002**	**2003**	**2004**	**2005**	**2006**	**2007**	**2008**	**2009**	**2010**
Narragansett Bay											
Field's Point	76.84[a]	77.42[a]	77.94	71.78	59.99	57.99	54.05	53.18	61.55	63.87	75.55
Bucklin Point	44.51[a]	45.26[a]	45.58	37.08	39.52	26.96	20.61	14.82	17.26	21.46	22.17
Newport[b]	11.21	11.17	11.14	11.07	10.96	10.70	10.91	10.46	10.39	10.35	10.64
East Providence	9.24[a]	9.31[a]	9.37	11.74	7.93	10.81	8.11	7.60	6.94	7.18	8.38
Bristol	8.97	5.94	6.81	6.44	5.49	5.47[c]	5.46	7.61	7.40	6.04	4.03
Warren	3.10[d]	3.11[d]	3.12[d]	3.10	1.93	1.91[c]	1.89[c]	1.88[c]	1.87[c]	1.85[c]	1.83[c]
East Greenwich	1.41	1.09	0.82	1.12	1.32	1.46	0.86	0.88	0.84	0.84	0.93
Quonset Point	0.98	1.00	0.78	0.94	0.75	0.74[c]	0.73[c]	0.73[c]	0.73[c]	0.73[c]	0.73[c]
Jamestown	0.11[e]	0.11	0.09	0.16	0.16[d]	0.16[d]	0.16[d]	0.16[d]	0.16[d]	0.16[d]	0.15[d]
Fall River*											
Blackstone River											
Worcester	15.74[f]	15.85[f]	15.98[f]	16.11[f]	16.22[f]	16.33[f]	16.41[f]	16.45[f]	16.88[f]	16.93	16.55
Woonsocket	21.93	10.90	5.76	3.82	3.88	7.94	7.20	6.58	4.68	3.56	5.15
Smithfield	3.24	2.69	2.19	3.17	2.62	3.34	1.01	0.97	1.56	1.69	1.64
Grafton*											
Millbury*											
Northbridge*											
Burrillville	1.27	1.33	1.41	1.36	1.37	1.35	1.36	1.33	1.61	1.32	1.35
Hopedale*											
Leicester*											
Douglas*											
Upton*											
Ten Mile River											
Attleboro	7.19[f]	7.26[f]	7.35[f]	7.39[f]	7.38[f]	7.38[f]	7.39[f]	7.41[f]	7.41[f]	7.45	8.42
North Attleboro	2.72[g]	2.76[g]	2.79[g]	2.80[g]	2.81[g]	2.81[g]	2.80[g]	2.80[g]	2.81	2.42	3.53
Pawtuxet River											
Cranston	16.04	16.95	15.59	21.65	11.88	21.97	10.50	10.90	10.42	15.69	13.06
West Warwick	5.82	5.28	6.27	9.28	12.41	7.81	8.76	8.90	7.96	5.38	9.86
Warwick	9.44	8.48	7.37	10.70	8.33	4.37	4.28	4.32	4.03	4.20	6.47
Taunton River											
Brockton*											
Taunton*											
Somerset*											

* Did not have annual data. 2007-2010 values were estimated by scaling 2000-2003 values (Nixon, et al., 2008) by population change.

[a] Estimated with population data from the U.S. Census Bureau and load data from 2002.

[b] Calculated assuming 0.8 moles nitrogen per person per day by 365 days per year.

[c] Estimated with population data from the U.S. Census Bureau and load data from 2004.

[d] Estimated with population data from the U.S. Census Bureau and load data from 2003.

[e] Estimated with population data from the U.S. Census Bureau and load data from 2001.

[f] Estimated with population data from the U.S. Census Bureau and load data from 2009.

[g] Estimated with population data from the U.S. Census Bureau and load data from 2008.

Table B-7. Annual total phosphorus load discharged from each facility from 2000-2010. All values are in millions of moles P per year.

Discharges to:	**2000**	**2001**	**2002**	**2003**	**2004**	**2005**	**2006**	**2007**	**2008**	**2009**	**2010**
Narragansett Bay											
Field's Point	4.94[a]	4.97[a]	5.01	2.04	2.41	2.57	2.09	3.07	3.41	2.96	3.30
Bucklin Point	3.24[a]	3.27[a]	3.29	2.23	2.56	3.01	3.04	2.68	4.69	2.38	2.79
Newport[b]	0.63	0.63	0.63	0.62	0.62	0.60	0.61	0.59	0.58	0.58	0.60
East Providence	0.88[a]	0.89[a]	0.89	0.62	0.53	0.53[c]	0.52[c]	0.52[c]	0.52[c]	0.52[c]	0.51[c]
Bristol	0.17[d]	0.17	0.14	0.22	0.18	0.18[c]	0.18[c]	0.18[c]	0.18[c]	0.18[c]	0.19[c]
Warren[e]	0.05	0.05	0.05	0.05	0.05	0.05	0.05	0.05	0.05	0.05	0.05
East Greenwich	0.41	0.41[f]	0.42[f]	0.42[f]	0.43[f]	0.43[f]	0.42[f]	0.42[f]	0.42[f]	0.42[f]	0.42[f]
Quonset Point[b]	0.10	0.10	0.10	0.10	0.10	0.10	0.10	0.10	0.10	0.10	0.10
Jamestown[g]	0.03	0.03	0.03	0.03	0.03	0.03	0.02	0.02	0.02	0.02	0.02
Fall River*											
Blackstone River											
Worcester	2.18	2.09	2.10[d]	2.12[d]	2.13[d]	2.14[d]	2.15[d]	2.16[d]	1.16[h]	1.16	0.71
Woonsocket	2.93	1.19	0.11	0.16	0.24	0.65	0.24	0.21	0.73	0.71	0.59
Smithfield	0.21[d]	0.21	0.24	0.26	0.27	0.22	0.05	0.02	0.02	0.02	0.02
Grafton*											
Millbury*											
Northbridge*											
Burrillville	0.03	0.03	0.02	0.03	0.02	0.02	0.02	0.02	0.02	0.03	0.02
Hopedale*											
Leicester*											
Douglas*											
Upton*											
Ten Mile River											
Attleboro	0.10	0.08	0.09	0.03	0.03	0.01	0.06	0.01	0.01	0.02	0.03
North Attleboro	0.06	0.03	0.08	0.12	0.11	0.06	0.09	0.05	0.04	0.02	0.04
Pawtuxet River											
Cranston	1.16	1.09	1.26	1.87	2.18	1.00	0.56	0.45	0.39	0.49	0.39
West Warwick	0.53	0.50	0.60	0.67	1.02	0.47	0.39	0.46	0.37	0.35	0.61
Warwick	0.60	0.59	0.56	0.51	0.55	0.21	0.14	0.16	0.13	0.13	0.42
Taunton River											
Brockton*											
Taunton*											
Somerset*											

* Did not have annual data. 2007-2010 values were estimated by scaling 2000-2003 values (Nixon, et al., 2008) by population change.

[a] Estimated with population data from the U.S. Census Bureau and load data from 2002.

[b] Calculated assuming 0.045 moles phosphorus per person per day by 365 days per year.

[c] Estimated with population data from the U.S. Census Bureau and load data from 2004.

[d] Estimated with population data from the U.S. Census Bureau and load data from 2001.

[e] Estimated with population data from the U.S. Census Bureau and load data from 1996.

[f] Estimated with population data from the U.S. Census Bureau and load data from 2000.

[g] Estimated with population data from the U.S. Census Bureau and load data from 1994.

[h] Estimated with population data from the U.S. Census Bureau and load data from 2009.

Table B-8. Active season total nitrogen load discharged from each facility from 2000-2010. All values are in millions of moles N per year.

Discharges to:	**2000**	**2001**	**2002**	**2003**	**2004**	**2005**	**2006**	**2007**	**2008**	**2009**	**2010**
Narragansett Bay											
Field's Point	42.75[a]	43.07[a]	43.36	35.95	29.02	26.16	27.29	26.73	26.05	29.58	31.57
Bucklin Point	23.37[a]	23.59[a]	23.76	18.12	20.28	10.51	8.54	6.46	7.41	9.72	7.42
Newport[b]	5.65	5.63	5.62	5.58	5.53	5.39	5.50	5.28	5.24	5.22	5.37
East Providence	4.47[a]	4.51[a]	4.54	4.82	4.81[c]	4.18[d]	4.15	3.84	2.48	2.99	4.22
Bristol	4.02	2.34	4.30	3.20	3.20[c]	3.18[d]	3.16	4.05	4.38	3.47	2.27
Warren	1.72[c]	1.73[c]	1.73[c]	1.72	1.72[c]	1.70[c]	1.69[c]	1.68[c]	1.67[c]	1.65[c]	1.63[c]
East Greenwich	0.59	0.50	0.44	0.57	0.72	0.62	0.20	0.21	0.17	0.17	0.20
Quonset Point	0.52	0.51	0.38	0.48	0.48[c]	0.48[c]	0.47[c]	0.47[c]	0.47[c]	0.47[c]	0.47[c]
Jamestown	0.07[e]	0.07	0.05	0.11	0.11[c]	0.11[c]	0.11[c]	0.11[c]	0.11[c]	0.11[c]	0.10[c]
Fall River*											
Blackstone River											
Worcester	7.63[f]	7.69[f]	7.75[f]	7.81[f]	7.86[f]	7.91[f]	7.95[f]	7.97[f]	8.18[f]	8.21	7.57
Woonsocket	12.47	4.79	2.89	2.13	2.13	3.69	3.40	2.46	2.29	1.40	1.37
Smithfield	1.81	1.58	1.29	1.85	1.54	1.96	0.59	0.46	0.67	0.67	0.74
Grafton*											
Millbury*											
Northbridge*											
Burrillville	0.62	0.46	0.53	0.51	0.44	0.45	0.42	0.37	0.39	0.40	0.38
Hopedale*											
Leicester*											
Douglas*											
Upton*											
Ten Mile River											
Attleboro	3.05[g]	3.08[g]	3.12[g]	3.14[g]	3.13[g]	3.13[g]	3.13[g]	3.14[g]	3.14[g]	3.16[g]	3.16
North Attleboro[b]	5.13	5.21	5.26	5.27	5.29	5.30	5.29	5.29	5.30	5.31	5.44
Pawtuxet River											
Cranston	7.23	7.95	8.03	10.97	5.52	9.82	3.22	4.25	3.58	4.34	4.17
West Warwick	2.49	2.54	2.55	4.68	6.36	2.06	2.67	2.06	2.05	1.81	4.47
Warwick	4.30	3.90	3.67	5.15	3.34	1.91	2.09	1.91	1.79	1.80	3.44
Taunton River											
Brockton*											
Taunton*											
Somerset*											

* Did not have annual data. 2007-2010 values were estimated by scaling 2000-2003 values (Nixon, et al., 2008) by population change.

[a] Estimated with population data from the U.S. Census Bureau and load data from 2002.

[b] Calculated assuming 0.8 moles nitrogen per person per day by 184 days per summer season, 181 days per winter season, and 182 days per leap year winter season (2000, 2004, 2008).

[c] Estimated with population data from the U.S. Census Bureau and load data from 2003.

[d] Estimated with population data from the U.S. Census Bureau and load data from 2006.

[e] Estimated with population data from the U.S. Census Bureau and load data from 2001.

[f] Estimated with population data from the U.S. Census Bureau and load data from 2009.

[g] Estimated with population data from the U.S. Census Bureau and load data from 2010.

Table B-9. Active season total phosphorus load discharged from each facility from 2000-2010. All values are in millions of moles P per year.

Discharges to:	**2000**	**2001**	**2002**	**2003**	**2004**	**2005**	**2006**	**2007**	**2008**	**2009**	**2010**
Narragansett Bay											
Field's Point	0.75[a]	0.76[a]	0.76	0.95	1.35	1.48	1.14	1.74	1.81	1.58	1.50
Bucklin Point	0.80[a]	0.81[a]	0.81	1.11	1.23	1.56	1.50	1.09	1.23	1.14	1.14
Newport[b]	0.32	0.32	0.32	0.31	0.31	0.30	0.31	0.30	0.29	0.29	0.30
East Providence	0.49[a]	0.49[a]	0.49	0.24	0.24[c]	0.24[c]	0.24[c]	0.24[c]	0.24[c]	0.24[c]	0.23[c]
Bristol	0.08[d]	0.08	0.07	0.10	0.10[c]	0.10[c]	0.10[c]	0.10[c]	0.10[c]	0.10[c]	0.10[c]
Warren[e]	0.04	0.04	0.04	0.04	0.04	0.04	0.04	0.04	0.04	0.04	0.04
East Greenwich[f]	0.39	0.40	0.40	0.40	0.41	0.41	0.40	0.40	0.40	0.40	0.40
Quonset Point[b]	0.05	0.05	0.05	0.05	0.05	0.05	0.05	0.05	0.05	0.05	0.05
Jamestown[g]	0.02	0.02	0.02	0.02	0.02	0.02	0.02	0.02	0.02	0.02	0.01
Fall River*											
Blackstone River											
Worcester	1.10	0.95	0.96[d]	0.97[d]	0.97[d]	0.98[d]	0.98[d]	0.99[d]	0.69[h]	0.70	0.46
Woonsocket	1.66	0.59	0.05	0.09	0.12	0.37	0.15	0.10	0.12	0.04	0.06
Smithfield	0.10[d]	0.10	0.10	0.13	0.13	0.11	0.01	0.01	0.01	0.01	0.01
Grafton*											
Millbury*											
Northbridge*											
Burrillville	0.02	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01
Hopedale*											
Leicester*											
Douglas*											
Upton*											
Ten Mile River											
Attleboro	0.06	0.05	0.05	0.02	0.02	0.01	0.04	0.01	0.01	0.01	0.01
North Attleboro	0.03	0.02	0.05	0.07	0.06	0.04	0.05	0.03	0.03	0.01	0.00
Pawtuxet River											
Cranston	0.44	0.49	0.82	0.94	1.04	0.44	0.26	0.18	0.16	0.19	0.13
West Warwick	0.23	0.26	0.27	0.37	0.58	0.08	0.09	0.07	0.08	0.09	0.34
Warwick	0.30	0.27	0.40	0.30	0.35	0.06	0.06	0.06	0.06	0.07	0.30
Taunton River											
Brockton*											
Taunton*											
Somerset*											

* Did not have annual data. 2007-2010 values were estimated by scaling 2000-2003 values (Nixon, et al., 2008) by population change.

[a] Estimated with population data from the U.S. Census Bureau and load data from 2002.

[b] Calculated assuming 0.045 moles phosphorus per person per day by 184 days per summer season, 181 days per winter season, and 182 days per leap year winter season (2000, 2004, 2008).

[c] Estimated with population data from the U.S. Census Bureau and load data from 2003.

[d] Estimated with population data from the U.S. Census Bureau and load data from 2001.

[e] Estimated with population data from the U.S. Census Bureau and load data from 1996.

[f] Estimated with population data from the U.S. Census Bureau and load data from 1999.

[g] Estimated with population data from the U.S. Census Bureau and load data from 1994.

[h] Estimated with population data from the U.S. Census Bureau and load data from 2009.

Table B-10. Inactive season total nitrogen load discharged from each facility from 2000-2010. All values are in millions of moles N per year.

Discharges to:	**2000**	**2001**	**2002**	**2003**	**2004**	**2005**	**2006**	**2007**	**2008**	**2009**	**2010**
Narragansett Bay											
Field's Point	32.86[a]	33.37[a]	33.15	35.71	31.10	31.54	28.49	29.33	35.00	34.18	43.91
Bucklin Point	21.07[a]	21.15[a]	21.30	18.70	19.11	15.28	10.70	8.66	10.14	11.73	14.65
Newport[b]	5.59	5.54	5.52	5.49	5.47	5.31	5.41	5.19	5.18	5.13	5.28
East Providence	4.69[a]	4.69[a]	4.72	6.90	4.47	4.42[c]	3.93	3.74	4.45	4.17	4.18
Bristol	4.91	3.57	2.50	3.23	3.10	3.07[c]	3.05[c]	3.05[c]	3.06[c]	3.02[c]	3.11[c]
Warren	1.39[d]	1.39[d]	1.39[d]	1.39	1.08	1.06[c]	1.05[c]	1.05[c]	1.05[c]	1.03[c]	1.02[c]
East Greenwich	0.58[e]	0.59	0.37	0.55	0.61	0.84	0.65	0.67	0.66	0.67	0.72
Quonset Point	0.45	0.48	0.40	0.46	0.42	0.41[c]	0.41[c]	0.41[c]	0.41[c]	0.41[c]	0.41
Jamestown[b]	0.25	0.25	0.25	0.25	0.25	0.25	0.24	0.24	0.24	0.24	0.24
Fall River*											
Blackstone River											
Worcester	8.45[f]	8.51[f]	8.58[f]	8.65[f]	8.70[f]	8.76[f]	8.80[f]	8.82[f]	9.05[f]	8.45[f]	8.95
Woonsocket	3.68[g]	3.69[g]	3.72[g]	3.77[g]	3.78[g]	3.75	3.79	4.08	2.41	2.15	3.73
Smithfield	0.87[h]	0.87[h]	0.88[h]	0.89[h]	0.90[h]	0.90[h]	0.89[h]	0.89[h]	0.89	1.02	0.90
Grafton*											
Millbury*											
Northbridge*											
Burrillville	0.66	0.87	0.88	0.85	0.93	0.89	0.94	0.94	1.21	0.91	0.96
Hopedale*											
Leicester*											
Douglas*											
Upton*											
Ten Mile River											
Attleboro	3.82[i]	3.85[i]	3.90[i]	3.93[i]	3.92[i]	3.92[i]	3.93[i]	3.94[i]	3.94[i]	3.96	5.20
North Attleboro	1.48[h]	1.50[h]	1.52[h]	1.52[h]	1.53[h]	1.53[h]	1.52[h]	1.52[h]	1.53	1.40	2.02
Pawtuxet River											
Cranston	8.95[e]	8.99	7.55	10.73	6.38	12.16	7.25	6.64	6.63	11.31	8.59
West Warwick	2.69[e]	2.70	3.70	4.59	6.08	5.69	6.03	6.73	5.87	3.55	5.40
Warwick	4.58[e]	4.58	3.70	5.54	5.00	2.45	2.18	2.40	2.25	2.40	3.04
Taunton River											
Brockton*											
Taunton*											
Somerset*											

* Did not have annual data. 2007-2010 values were estimated by scaling 2000-2003 values (Nixon, et al., 2008) by population change.

[a] Estimated with population data from the U.S. Census Bureau and load data from 2002.

[b] Calculated assuming 0.8 moles nitrogen per person per day by 184 days per summer season, 181 days per winter season, and 182 days per leap year winter season (2000, 2004, 2008).

[c] Estimated with population data from the U.S. Census Bureau and load data from 2004.

[d] Estimated with population data from the U.S. Census Bureau and load data from 2003.

[e] Estimated with population data from the U.S. Census Bureau and load data from 2001.

[f] Estimated with population data from the U.S. Census Bureau and load data from 2010.

[g] Estimated with population data from the U.S. Census Bureau and load data from 2005.

[h] Estimated with population data from the U.S. Census Bureau and load data from 2008.

[i] Estimated with population data from the U.S. Census Bureau and load data from 2009.

Table B-11. Inactive season total phosphorus load discharged from each facility from 2000-2010. All values are in millions of moles P per year.

Discharges to:	**2000**	**2001**	**2002**	**2003**	**2004**	**2005**	**2006**	**2007**	**2008**	**2009**	**2010**
Narragansett Bay											
Field's Point	4.41[a]	4.48[a]	4.45	1.08	1.07	1.12	0.94	1.32	1.62	1.38	1.79
Bucklin Point	2.48[a]	2.49[a]	2.51	1.11	1.34	1.44	1.53	1.59	3.46	1.24	1.65
Newport[b]	0.31	0.31	0.31	0.31	0.31	0.30	0.30	0.29	0.29	0.29	0.30
East Providence	0.37[a]	0.38[a]	0.38	0.38	0.27	0.27	0.27	0.27	0.27	0.27	0.26
Bristol	0.09[c]	0.09	0.08	0.12	0.11	0.11[d]	0.11[d]	0.11[d]	0.11[d]	0.11[d]	0.11[d]
Warren[e]	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01
East Greenwich	0.21	0.21[f]	0.21[f]	0.21[f]	0.22[f]	0.22[f]	0.21[f]	0.21[f]	0.21[f]	0.21[f]	0.21[f]
Quonset Point[b]	0.05	0.05	0.05	0.05	0.05	0.05	0.05	0.05	0.05	0.05	0.05
Jamestown[g]	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01
Fall River*											
Blackstone River											
Worcester	1.09	1.12	1.13[c]	1.14[c]	1.15[c]	1.15[c]	1.16[c]	1.16[c]	1.19[c]	0.26[h]	0.26
Woonsocket	0.70[i]	0.70[i]	0.71[i]	0.72[i]	0.72[i]	0.71[i]	0.71[i]	0.71[i]	0.71	0.65	0.52
Smithfield	0.14[a]	0.14[a]	0.14	0.13	0.14	0.11	0.04	0.01	0.02	0.01	0.01
Grafton*											
Millbury*											
Northbridge*											
Burrillville	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01
Hopedale*											
Leicester*											
Douglas*											
Upton*											
Ten Mile River											
Attleboro	0.03[j]	0.03[j]	0.03[j]	0.03[j]	0.03[j]	0.03[j]	0.03[j]	0.03[j]	0.03[j]	0.03[j]	0.03
North Attleboro	0.02[i]	0.02[i]	0.02[i]	0.02[i]	0.02[i]	0.02[i]	0.02[i]	0.02[i]	0.02	0.01	0.04
Pawtuxet River											
Cranston	0.60[c]	0.60	0.44	0.92	1.14	0.56	0.29	0.27	0.24	0.30	0.26
West Warwick	0.30	0.24	0.33	0.31	0.44	0.38	0.29	0.38	0.29	0.26	0.27
Warwick	0.32[c]	0.32	0.16	0.22	0.20	0.14	0.08	0.09	0.07	0.07	0.13
Taunton River											
Brockton*											
Taunton*											
Somerset*											

* Did not have annual data. 2007-2010 values were estimated by scaling 2000-2003 values (Nixon, et al., 2008) by population change.

[a] Estimated with population data from the U.S. Census Bureau and load data from 2002.

[b] Calculated assuming 0.045 moles phosphorus per person per day by 184 days per summer season, 181 days per winter season, and 182 days per leap year winter season (2000, 2004, 2008).

[c] Estimated with population data from the U.S. Census Bureau and load data from 2001.

[d] Estimated with population data from the U.S. Census Bureau and load data from 2004.

[e] Estimated with population data from the U.S. Census Bureau and load data from 1996.

[f] Estimated with population data from the U.S. Census Bureau and load data from 2000.

[g] Estimated with population data from the U.S. Census Bureau and load data from 1994.

[h] Estimated with population data from the U.S. Census Bureau and load data from 2009.

[i] Estimated with population data from the U.S. Census Bureau and load data from 2008.

[j] Estimated with population data from the U.S. Census Bureau and load data from 2010.

Table B-12. Nitrogen and phosphorus loads from rivers that drain to Narragansett Bay from 2006-2010. All nitrogen and phosphorus values are in millions of moles per year.

Blackstone River	**NO_3+NO_2**	**NO_2**	**NO_3**	**NH_4**	**PO_4**	**SiO_2**	**TN**	**DIN**	**DON**
2007	45.07	1.03	44.05	14.00	1.94	40.39	68.34	58.74	9.77
2008	75.25	1.52	73.72	20.60	2.45	66.28	146.16	90.98	55.19
2009	56.83	1.07	55.76	14.65	1.64	80.85	87.23	71.48	15.75
2010	34.29	0.67	33.67	6.59	3.34	56.81	55.00	39.51	14.93
Pawtuxet River									
2005	30.67	0.61	30.06	11.57	1.78	71.09	50.80	42.24	
2006	21.77	0.45	21.33	5.11	0.82	52.49	42.86	26.88	
2007	23.67	0.37	23.30	5.49	1.11	39.41	36.40	29.16	
2008	30.26	0.48	29.78	3.69	0.85	50.40	55.15	33.96	
2009	27.16	0.59	26.57	7.71	1.15	69.56	45.12	34.87	
2010	14.04	0.24	13.80	6.32	0.69	42.50	27.54	20.36	
Woonasquatucket River									
2006	2.67	0.05	2.63	0.71	0.15	3.01	6.68	3.38	3.26
2007	2.85	0.05	2.79	0.20	0.01	3.69	3.79	3.04	0.81
2008	3.78	0.06	3.72	0.60	0.04	3.59	6.19	4.38	1.78
2009	3.81	0.04	3.76	0.16	0.02	6.97	5.44	3.97	1.47
2010	3.66	0.09	3.57	0.29	0.04	6.22	5.52	3.96	1.57
Moshassuck River									
2006	1.06	0.03	1.03	0.39	0.01	1.44	2.77	1.45	1.35
2007	1.55	0.03	1.52	0.27	0.00	3.31	2.26	1.82	0.44
2008	1.85	0.03	1.82	0.36	0.01	3.63	2.88	2.21	0.67
2009	1.74	0.03	1.71	0.24	0.01	4.50	2.52	1.99	0.53
2010	1.59	0.02	1.57	0.32	0.01	3.43	2.63	1.91	0.72
Ten Mile River									
2006	9.63	0.19	9.43	1.21	0.11	5.92	15.31	10.88	4.52
2007	6.60	0.17	6.43	0.95	0.11	4.58	11.97	7.55	2.26
2008	12.47	0.12	12.35	0.62	0.11	7.91	16.60	13.09	3.52
2009	11.30	0.12	11.18	0.49	0.08	8.23	14.03	11.78	2.12
2010	9.86	0.22	9.65	0.78	0.06	6.93	12.52	10.65	2.49
Taunton River									
2006	23.97	0.69	23.27	9.82	0.99	23.86	97.04	33.94	62.09
2007	34.83	0.73	34.09	8.51	0.74	28.07	63.07	43.34	19.73
2008	44.55	0.61	41.14	9.34	0.75	31.64	78.59	53.88	24.71
2009	56.55	0.93	55.62	5.67	0.95	64.13	111.07	66.09	43.72
2010	29.07	0.68	28.48	4.40	0.58	22.33	56.60	33.76	23.27

Table B-13. Average and standard deviation of flow, nitrogen, and phosphorus for wastewater treatment facilities in the Narragansett Bay watershed from 2006-2010. All flow values are in cubic meters per day and all nitrogen and phosphorus values are in moles per year.

Discharges to:	**Flow**		**DIN**		**TN**		**DIP** [d]		**TP**	
Narragansett Bay	**Mean**	**St Dev**	**Mean**	**St Dev**	**Mean**	**St Dev**	**Mean**	**St Dev**	**Mean**	**St Dev**
Field's Point	$1.7x10^{5}$	$4.2x10^{4}$	$4.6x10^{7}$	$7.6x10^{6}$	$6.2x10^{7}$	$9.1x10^{6}$	$1.5x10^{6}$	$9.1x10^{4}$	$3.0x10^{6}$	$5.2x10^{5}$
Bucklin Point	$8.5x10^{4}$	$2.7x10^{4}$	$1.5x10^{7}$	$3.3x10^{6}$	$1.9x10^{7}$	$3.1x10^{6}$	$2.4x10^{6}$	$2.7x10^{5}$	$3.1x10^{6}$	$9.1x10^{5}$
Newport	$3.5x10^{4}$	$1.1x10^{4}$			$1.1x10^{7}$	$2.3x10^{5}$	$4.0x10^{5}$	$2.8x10^{2}$	$5.9x10^{5}$	$1.3x10^{4}$
East Providence	$2.7x10^{4}$	$8.1x10^{3}$	$6.7x10^{6}$	$1.8x10^{6}$	$7.6x10^{6}$	$6.0x10^{5}$	$4.5x10^{5}$	$9.1x10^{1}$	$5.2x10^{5}$	$6.8x10^{3}$
Bristol	$1.4x10^{4}$	$5.1x10^{3}$	$3.9x10^{6}$	$5.2x10^{5}$	$6.1x10^{6}$	$1.5x10^{6}$	$1.2x10^{5}$	$2.1x10^{1}$	$1.8x10^{5}$	$2.0x10^{3}$
Warren	$7.1x10^{3}$	$2.8x10^{3}$	$1.6x10^{6}$	$1.8x10^{4}$	$1.9x10^{6}$	$2.5x10^{4}$	$3.6x10^{4}$	$1.2x10^{1}$	$5.2x10^{4}$	$8.0x10^{2}$
East Greenwich [a]	$4.1x10^{3}$	$8.5x10^{2}$	$1.3x10^{6}$	$7.2x10^{4}$	$8.7x10^{5}$	$4.3x10^{4}$	$2.9x10^{5}$	$5.8x10^{-1}$	$4.2x10^{5}$	$4.9x10^{2}$
Quonset Point	$1.8x10^{3}$	$4.7x10^{2}$			$7.3x10^{5}$	$2.2x10^{3}$	$6.8x10^{4}$	$9.3x10^{-1}$	$1.0x10^{5}$	$3.0x10^{2}$
Jamestown			$1.5x10^{5}$	$9.5x10^{2}$	$1.6x10^{5}$	$1.4x10^{3}$	$1.7x10^{4}$	$1.9x10^{0}$	$2.5x10^{4}$	$2.2x10^{2}$
Fall River*										
Blackstone River										
Worcester [b]	$1.1x10^{5}$	$4.2x10^{4}$			$1.7x10^{7}$		$4.8x10^{5}$		$7.1x10^{5}$	
Woonsocket	$2.9x10^{4}$	$8.9x10^{3}$	$4.5x10^{6}$	$1.2x10^{6}$	$5.4x10^{6}$	$1.5x10^{6}$	$3.4x10^{5}$	$1.3x10^{5}$	$5.0x10^{5}$	$2.5x10^{5}$
Smithfield [a]	$7.6x10^{3}$	$1.9x10^{3}$	$1.3x10^{6}$	$2.4x10^{5}$	$1.5x10^{6}$	$3.3x10^{5}$	$1.4x10^{4}$	$1.6x10^{2}$	$2.1x10^{4}$	$1.8x10^{3}$
Grafton*										
Millbury*										
Northbridge*										
Burrillville	$3.2x10^{3}$	$8.4x10^{2}$	$1.3x10^{6}$	$1.2x10^{5}$	$1.4x10^{6}$	$1.2x10^{5}$	$1.6x10^{4}$	$1.3x10^{2}$	$2.4x10^{4}$	$1.8x10^{3}$
Hopedale*										
Leicester*										
Douglas*										
Upton*										
Ten Mile River										
Attleboro	$1.5x10^{4}$	$4.3x10^{3}$			$7.6x10^{6}$	$4.5x10^{5}$	$1.9x10^{4}$	$1.6x10^{4}$	$2.8x10^{4}$	$2.1x10^{4}$
North Attleboro [c]	$1.6x10^{4}$	$4.4x10^{3}$			$3.0x10^{6}$	$7.8x10^{5}$	$2.1x10^{4}$	$7.0x10^{3}$	$3.0x10^{4}$	$1.5x10^{4}$
Pawtuxet River										
Cranston [a]	$4.3x10^{4}$	$1.0x10^{4}$	$1.0x10^{7}$	$2.4x10^{6}$	$1.3x10^{7}$	$2.4x10^{6}$	$2.9x10^{5}$	$5.0x10^{3}$	$4.3x10^{5}$	$4.6x10^{4}$
West Warwick	$2.3x10^{4}$	$6.3x10^{3}$	$6.9x10^{6}$	$1.8x10^{6}$	$8.2x10^{6}$	$1.7x10^{6}$	$3.0x10^{5}$	$2.4x10^{4}$	$4.4x10^{5}$	$1.0x10^{5}$
Warwick	$1.9x10^{4}$	$2.0x10^{3}$	$3.8x10^{6}$	$9.4x10^{5}$	$4.7x10^{6}$	$1.0x10^{6}$	$1.3x10^{5}$	$8.0x10^{4}$	$2.0x10^{5}$	$1.3x10^{5}$
Taunton River										
Brockton*										
Taunton*										
Somerset*										

* indicates facilities that do not have annual data.

[a] Average and standard deviation values are for 2007-2010 to avoid averaging over upgrade completion.

[b] Average nitrogen and phosphorus load values are 2010 load values as this is the only year of data available after upgrades were completed.

[c] Average and standard deviation values are for 2009-2010 to avoid averaging over upgrade completion.

[d] Average DIP load values for the Field's Point, Bucklin Point, and East Providence facilities were calculated using the ratio between DIP and TP values from earlier measurements (Nixon, et al., 1995). The ratio between DIP and TP for the remaining facilities was calculated by taking the average of the DIP to TP ratios of the Field's Point, Bucklin Point, and East Providence facilities. Average DIP load values for the remaining facilities were calculated using this average ratio.

Table B-14. Average and standard deviation of flow, nitrogen, and phosphorus for rivers that drain to Narragansett Bay from 2006-2010. All flow values are in millions of cubic meters per day and all nitrogen and phosphorus values are in millions of moles per year.

	Flow		DIN		TN		DIP		TP	
	Mean	**St Dev**	**Mean**	**St Dev**	**Mean**	**St Dev**	**Mean**	**St Dev**	**Mean**	**St Dev**
Blackstone River [a]	2.574	2.874	66.821	18.452	89.185	40.222	2.341	0.744	5.758	1.641
Pawtuxet River	1.071	1.366	29.047	6.333	41.413	10.272	0.923	0.199	1.949	0.943
Woonasquatucket River	0.225	0.266	3.744	0.603	5.526	1.094	0.052	0.058	0.172	0.069
Moshassuck River	0.115	0.163	1.877	0.310	2.612	0.241	0.006	0.001	0.014	0.002
Ten Mile River	0.320	0.333	10.780	2.226	14.087	1.922	0.095	0.024	0.028	0.000
Taunton River	1.502	1.516	45.340	13.191	81.273	22.824	0.804	0.168	1.290	

[a] Average and standard deviation values are for 2007-2010.

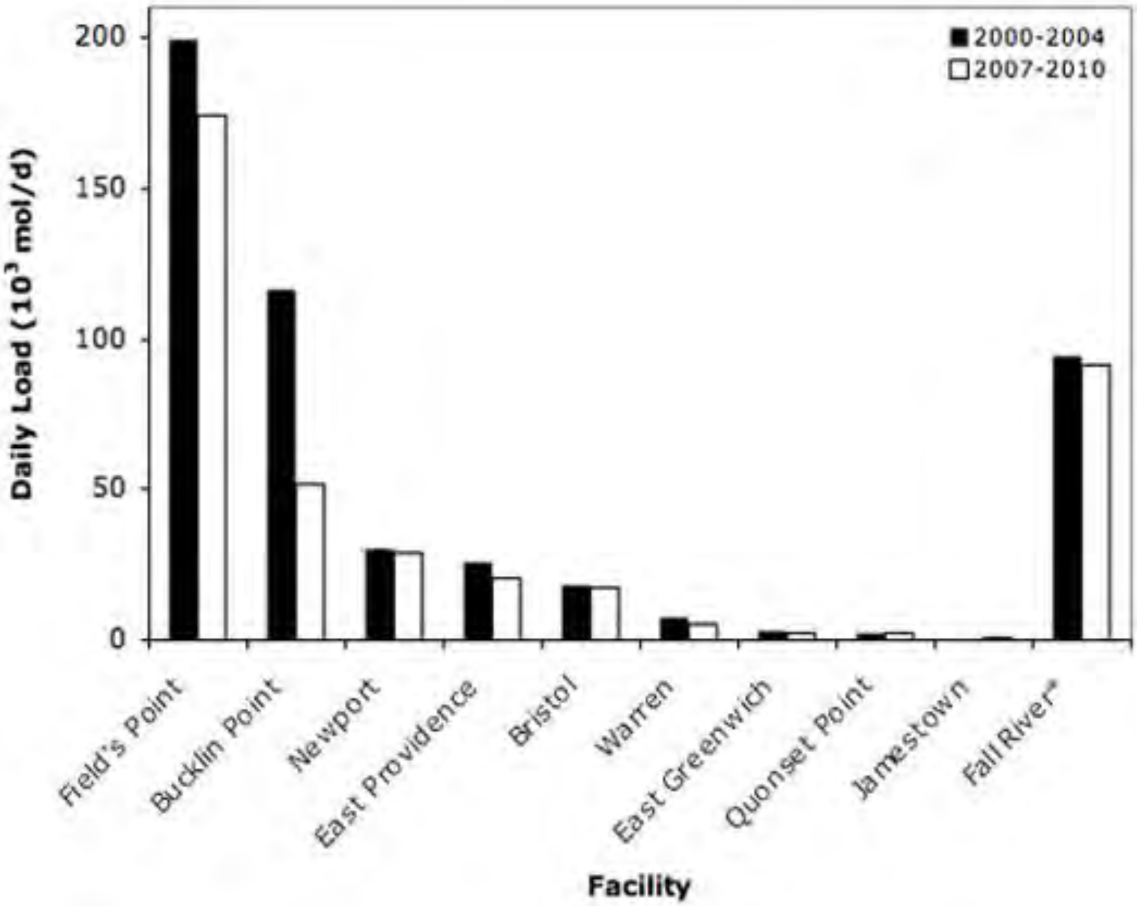

Figure B-25. Annual daily total nitrogen load from facilities that directly discharge to Narragansett Bay.

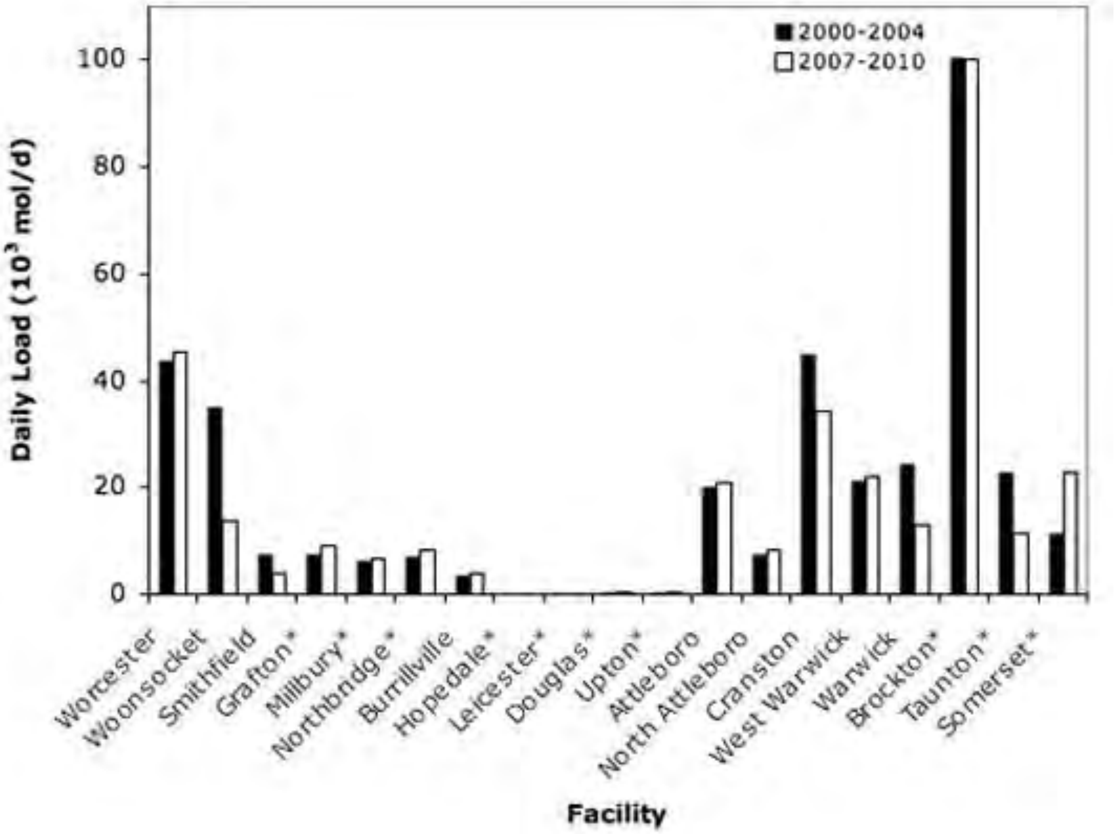

Figure B-26. Annual daily total nitrogen load discharged from facilities on rivers that drain to Narragansett Bay. * indicates that facilities were estimated with previous values (Nixon, et al., 2008) and population data.

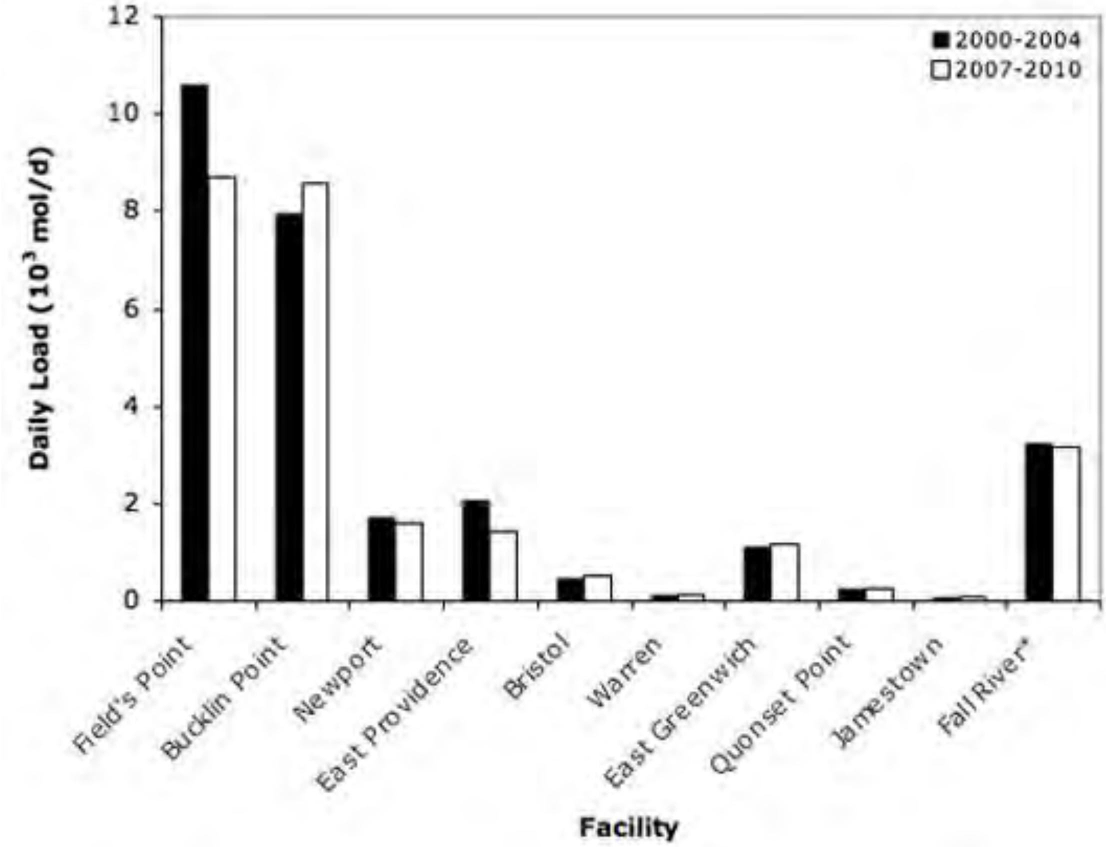

Figure B-27. Annual daily total phosphorus load from facilities that directly discharge to Narragansett Bay.

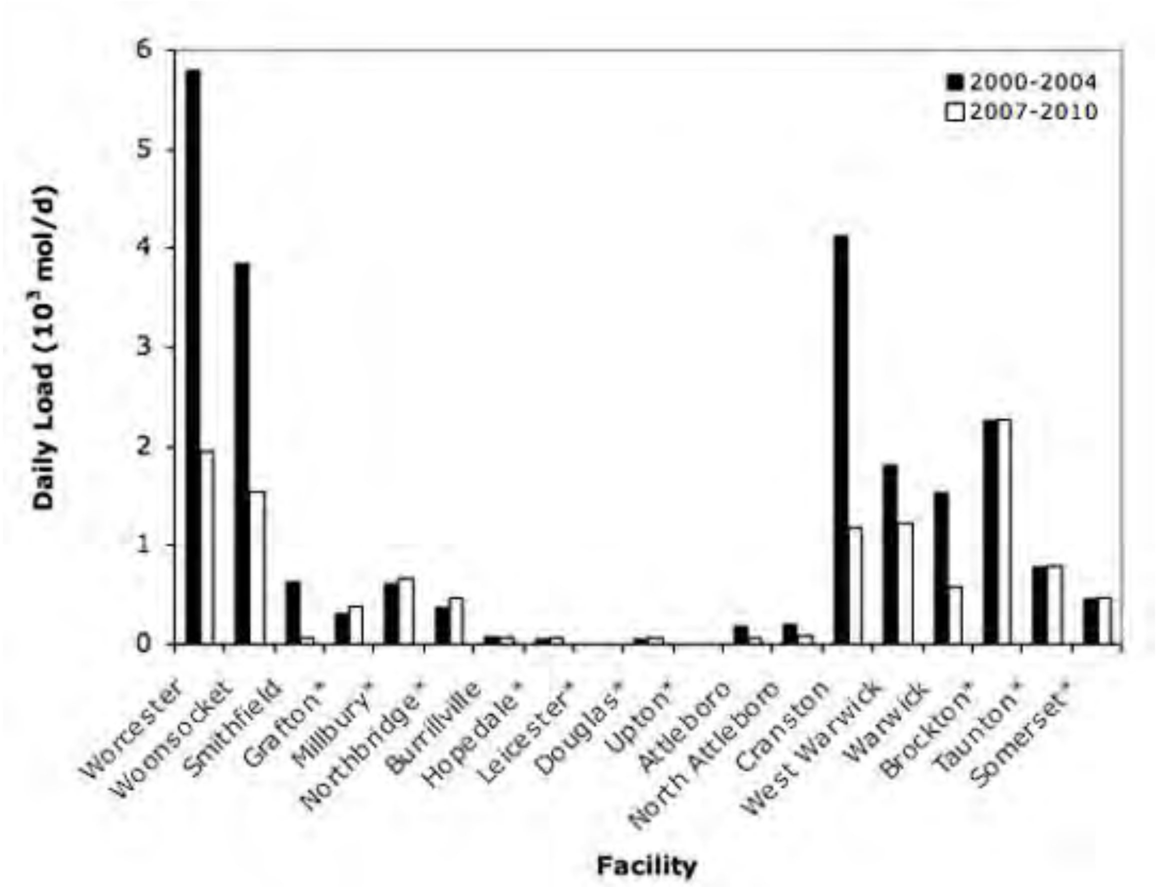

Figure B-28. Annual daily total phosphorus load discharged from facilities on rivers that drain to Narragansett Bay. * indicates that facilities were estimated with previous values (Nixon, et al., 2008) and population data.

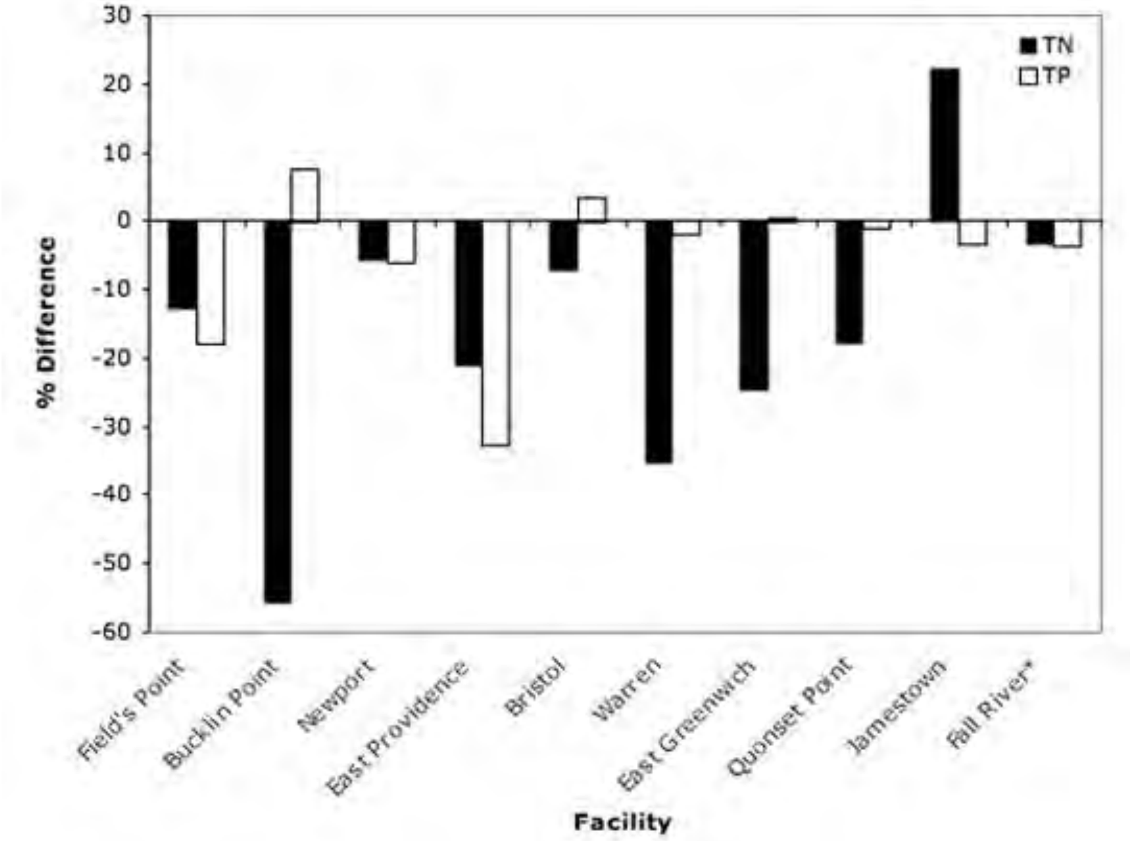

Figure B-29. Annual percent difference in total nitrogen and total phosphorus loads from facilities that directly discharge to Narragansett Bay in 2007-2010 relative to 2000-2004.

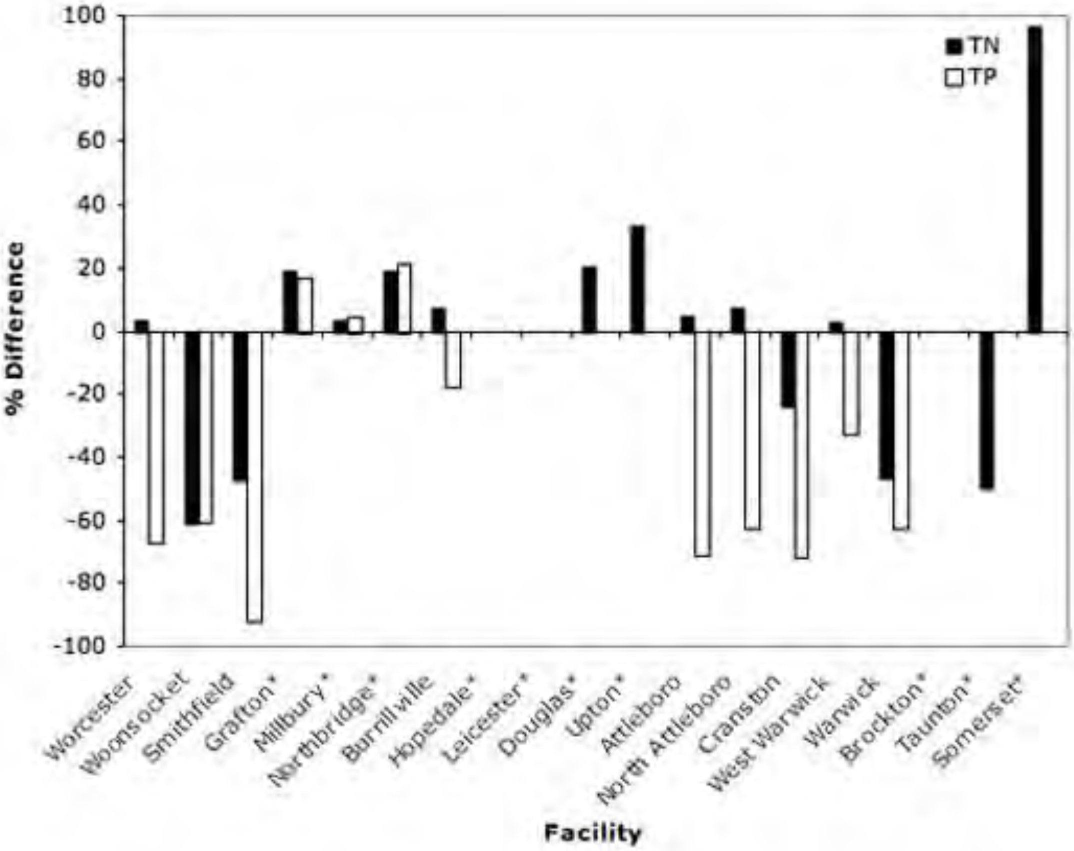

Figure B-30. Annual percent difference in total nitrogen and total phosphorus loads from facilities that discharge to rivers that drain to Narragansett Bay in 2007-2010 relative to 2000-2004.

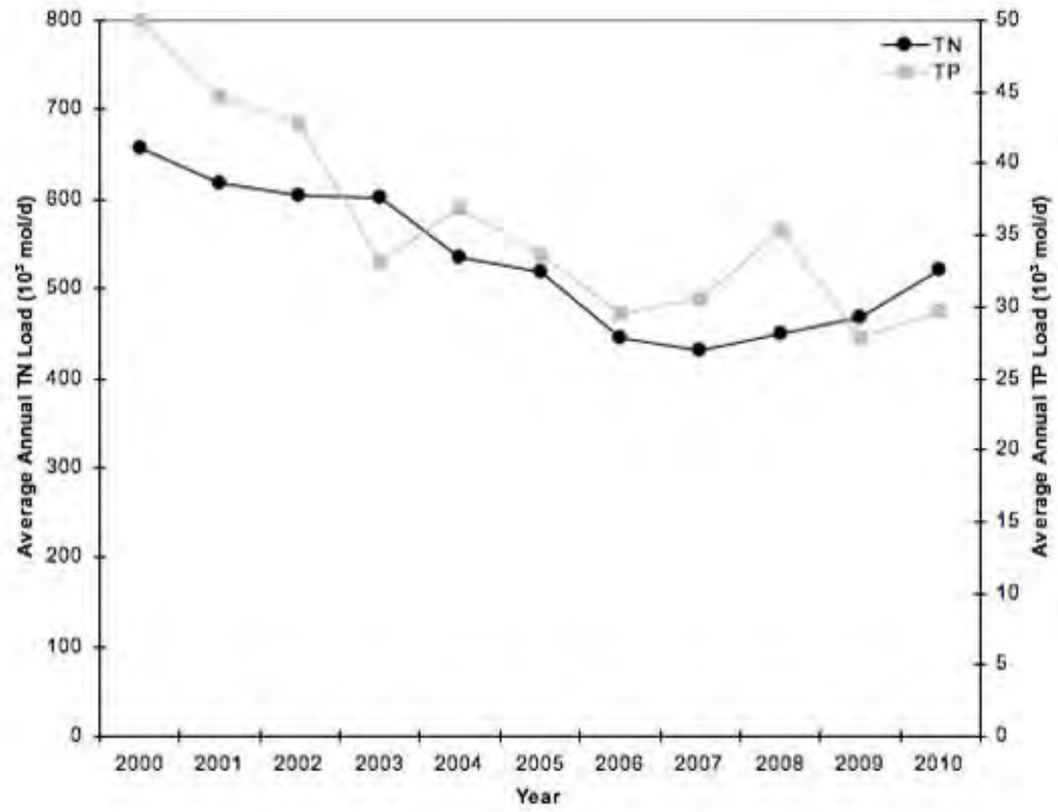

Figure B-31. Average annual total sewage nitrogen and total sewage phosphorus load discharged from all facilities combined over the 2000-2010 time period.

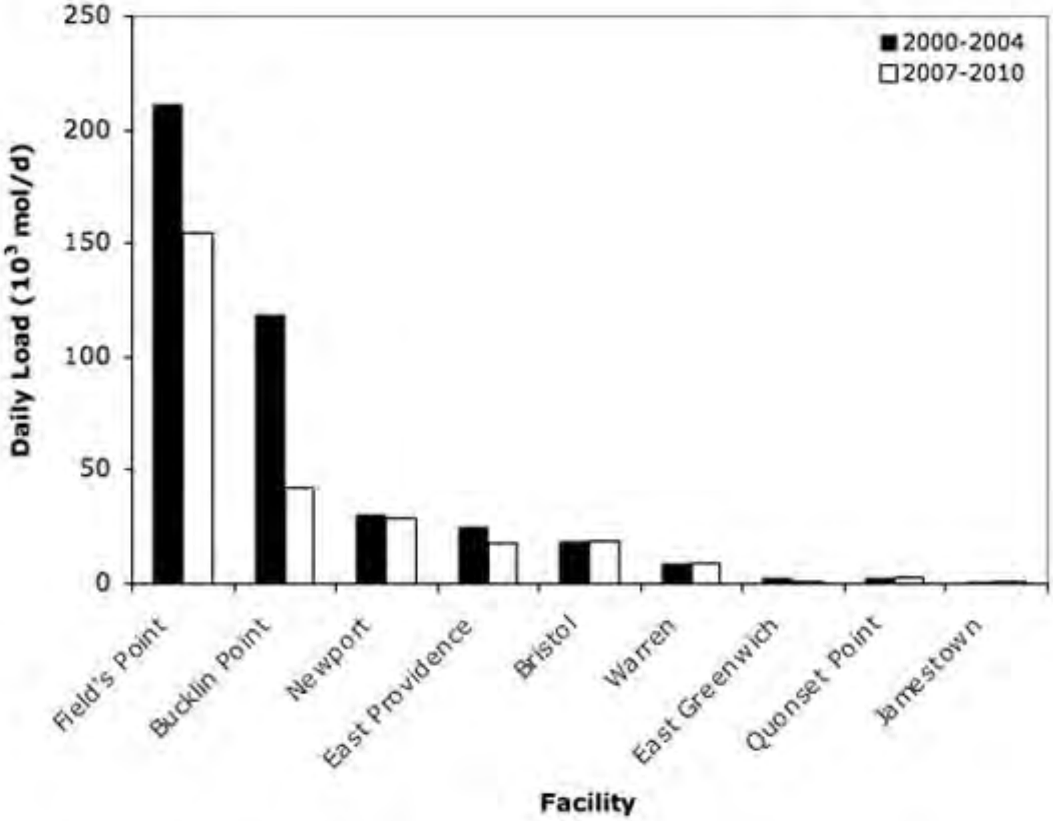

Figure B-32. Active season daily total nitrogen load from facilities that directly discharge to Narragansett Bay. The Fall River facility was not included as there was no seasonal data available.

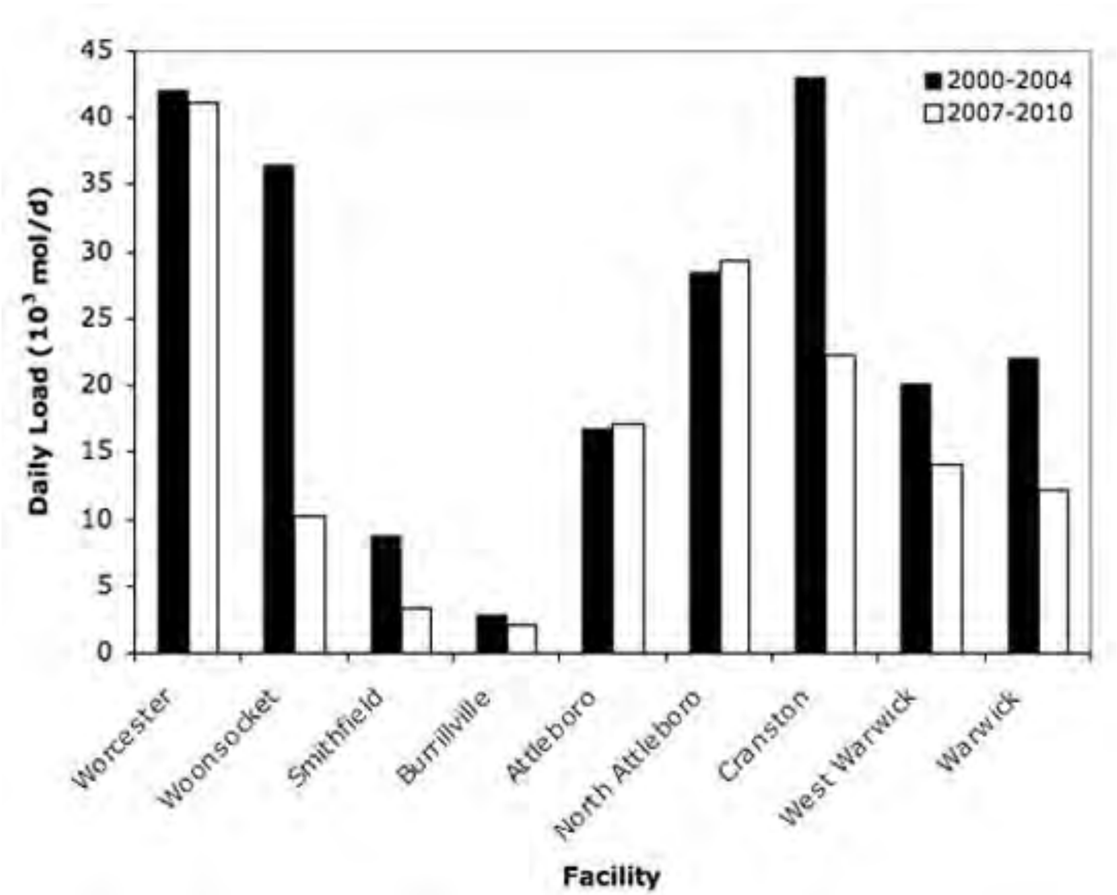

Figure B-33. Active season daily total nitrogen load discharged from facilities on rivers that drain to Narragansett Bay. The Grafton, Millbury, Hopedale, Leicester, Douglas, Upton, Brockton, Taunton, and Somerset facilities were not included as there was no seasonal data available.

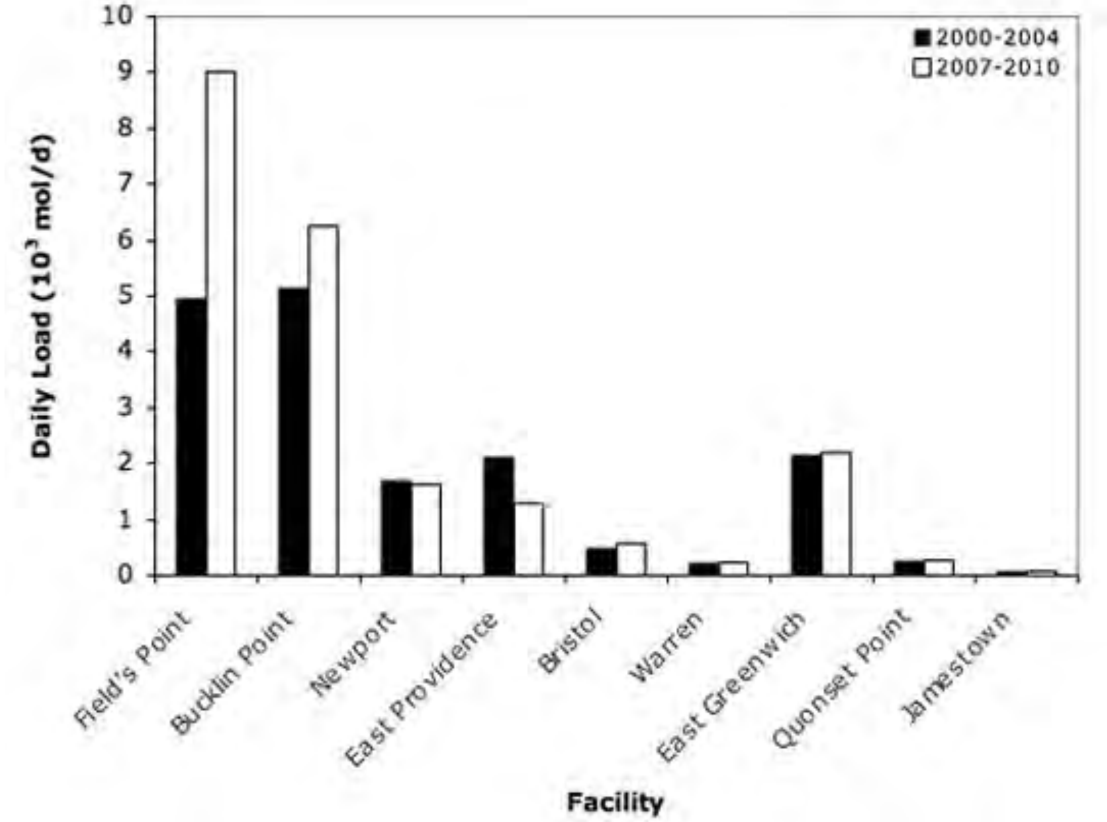

Figure B-34. Active season daily total phosphorus load from facilities that directly discharge to Narragansett Bay. The Fall River facility was not included as there was no seasonal data available.

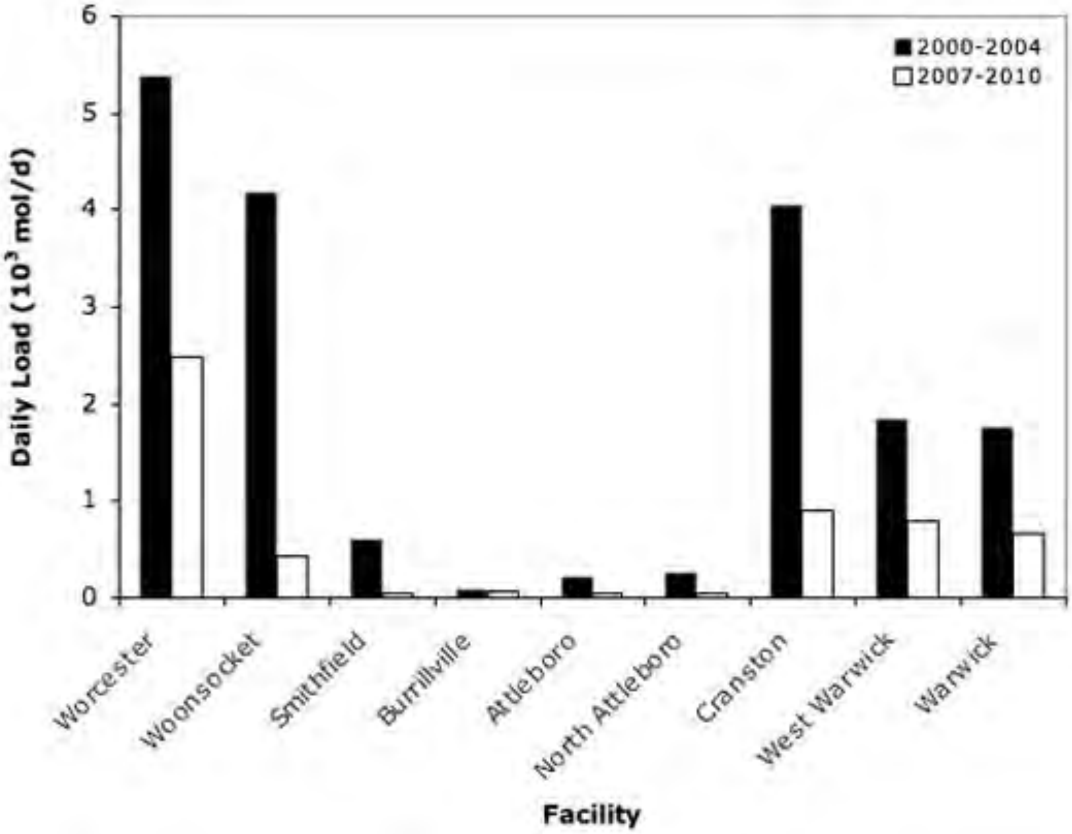

Figure B-35. Active season daily total phosphorus load from facilities discharged from facilities on rivers that drain to Narragansett Bay. The Grafton, Millbury, Hopedale, Leicester, Douglas, Upton, Brockton, Taunton, and Somerset facilities were not included as there was no seasonal data available.

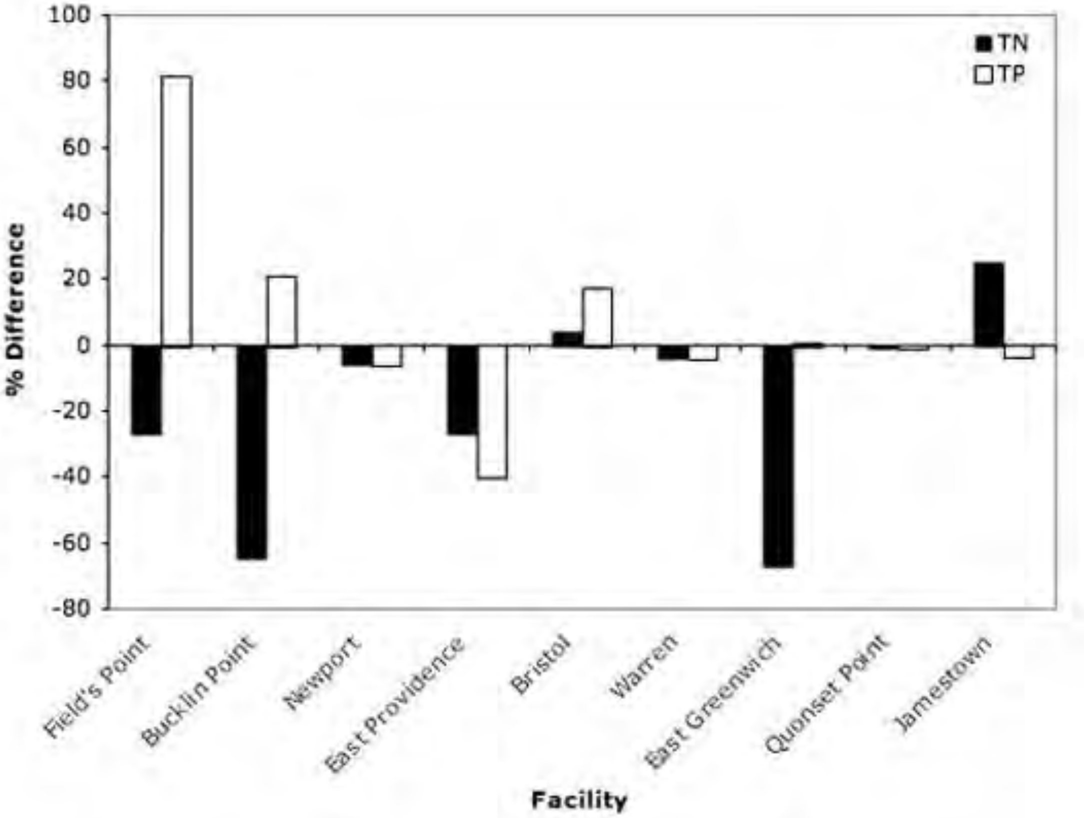

Figure B-36. Active season percent difference in total nitrogen and total phosphorus loads from facilities that directly discharge to Narragansett Bay in 2007-2010 relative to 2000-2004. The Fall River facility was not included as there was no seasonal data available.

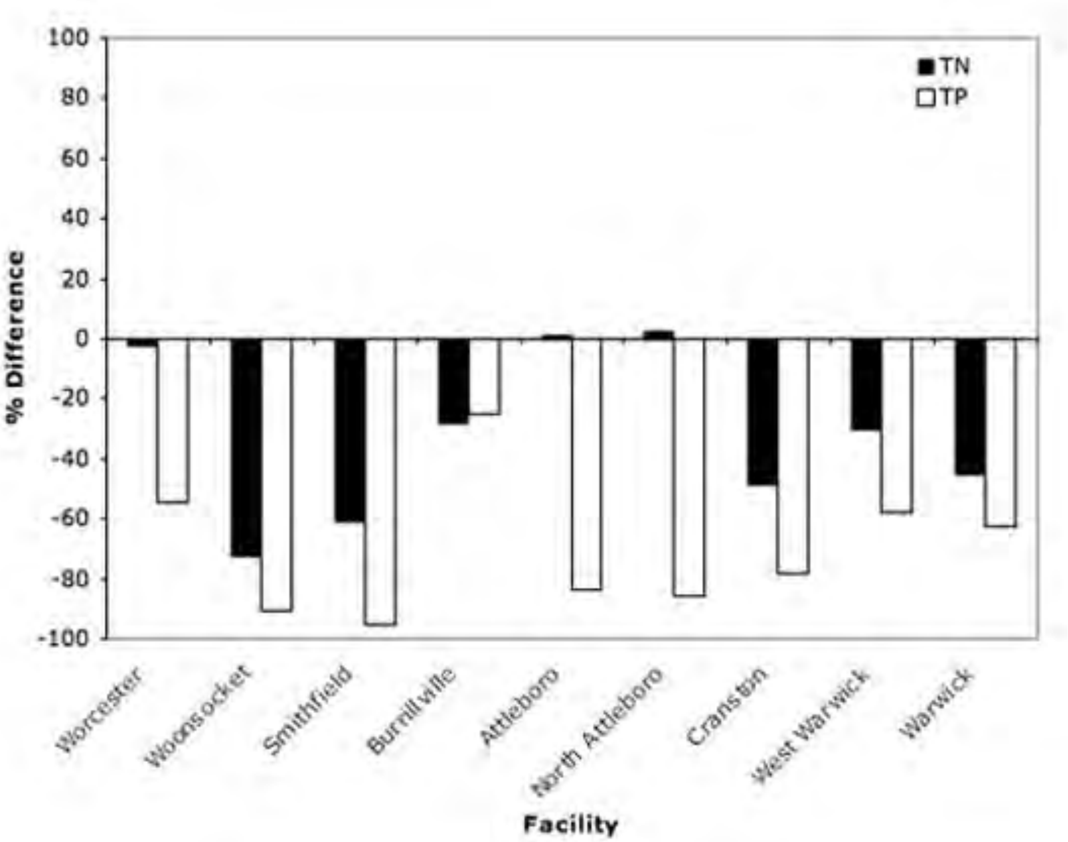

Figure B-37. Active season percent difference in total nitrogen and total phosphorus loads discharged from facilities on rivers that drain to Narragansett Bay in 2007-2010 relative to 2000-2004. The Grafton, Millbury, Hopedale, Leicester, Douglas, Upton, Brockton, Taunton, and Somerset facilities were not included as there was no seasonal data available.

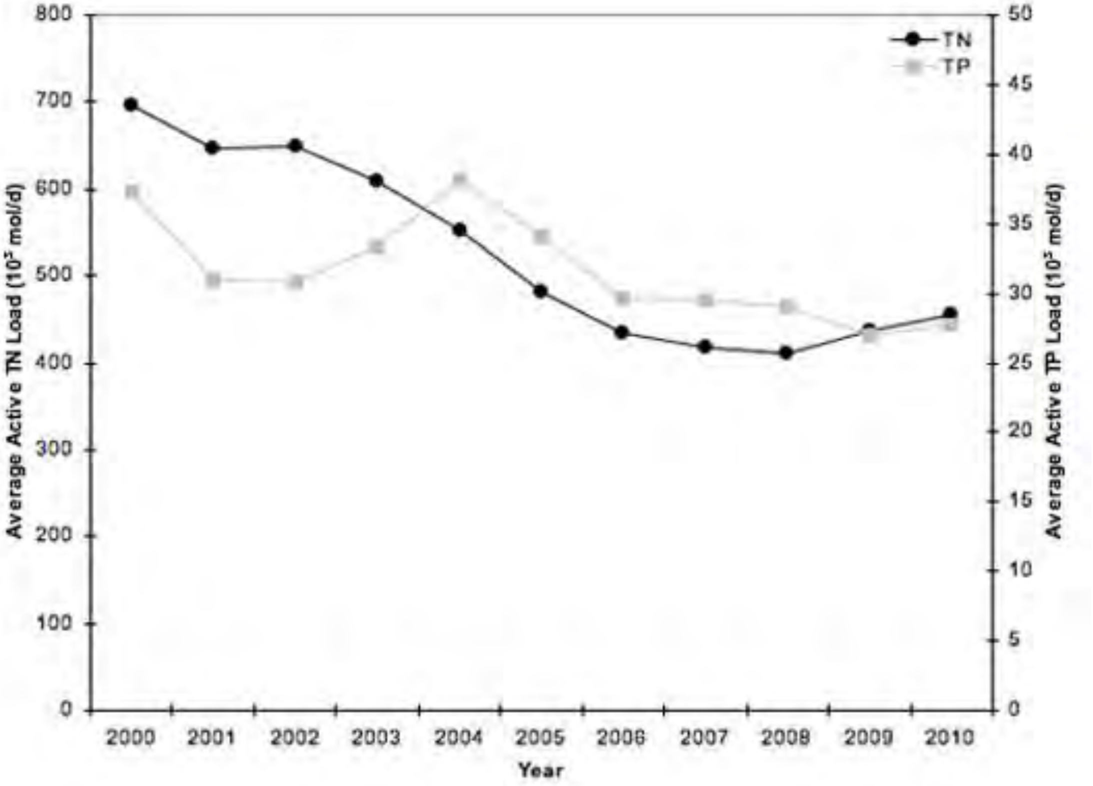

Figure B-38. Average active season total sewage nitrogen and total sewage phosphorus load discharged from all facilities with load data available combined over the 2000-2010 time period.

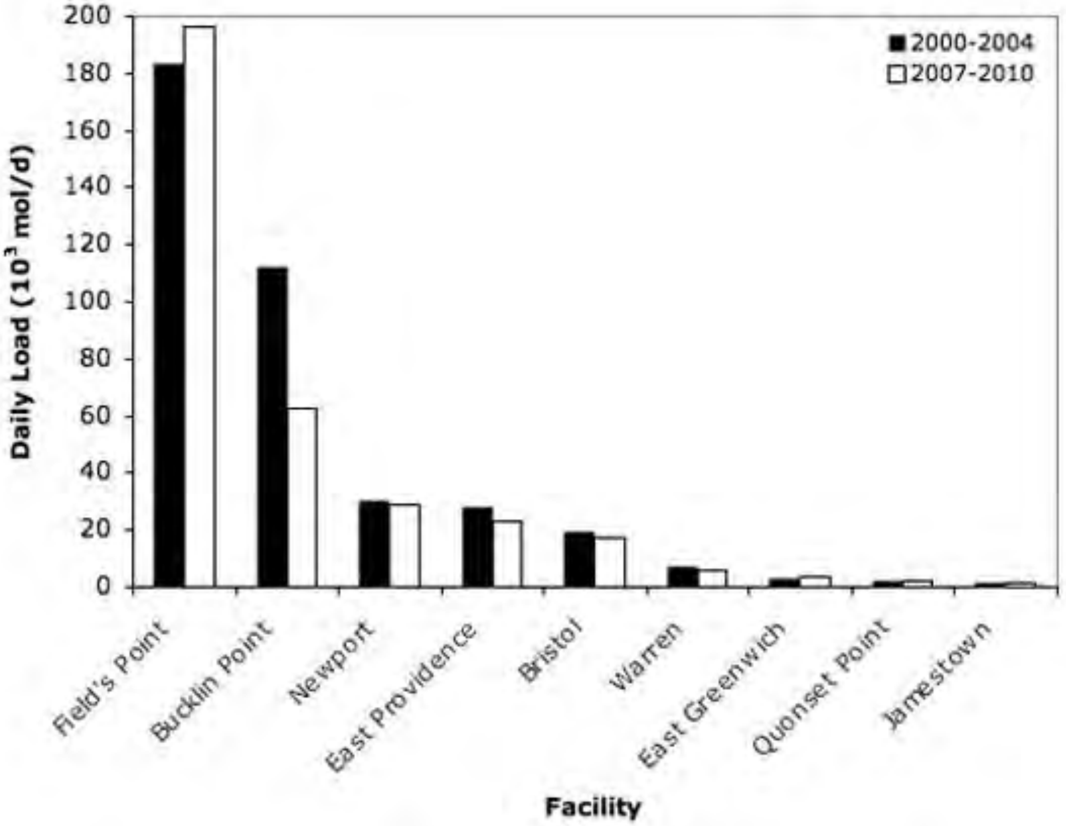

Figure B-39. Inactive season daily total nitrogen load from facilities that directly discharge to Narragansett Bay. The Fall River facility was not included as there was no seasonal data available.

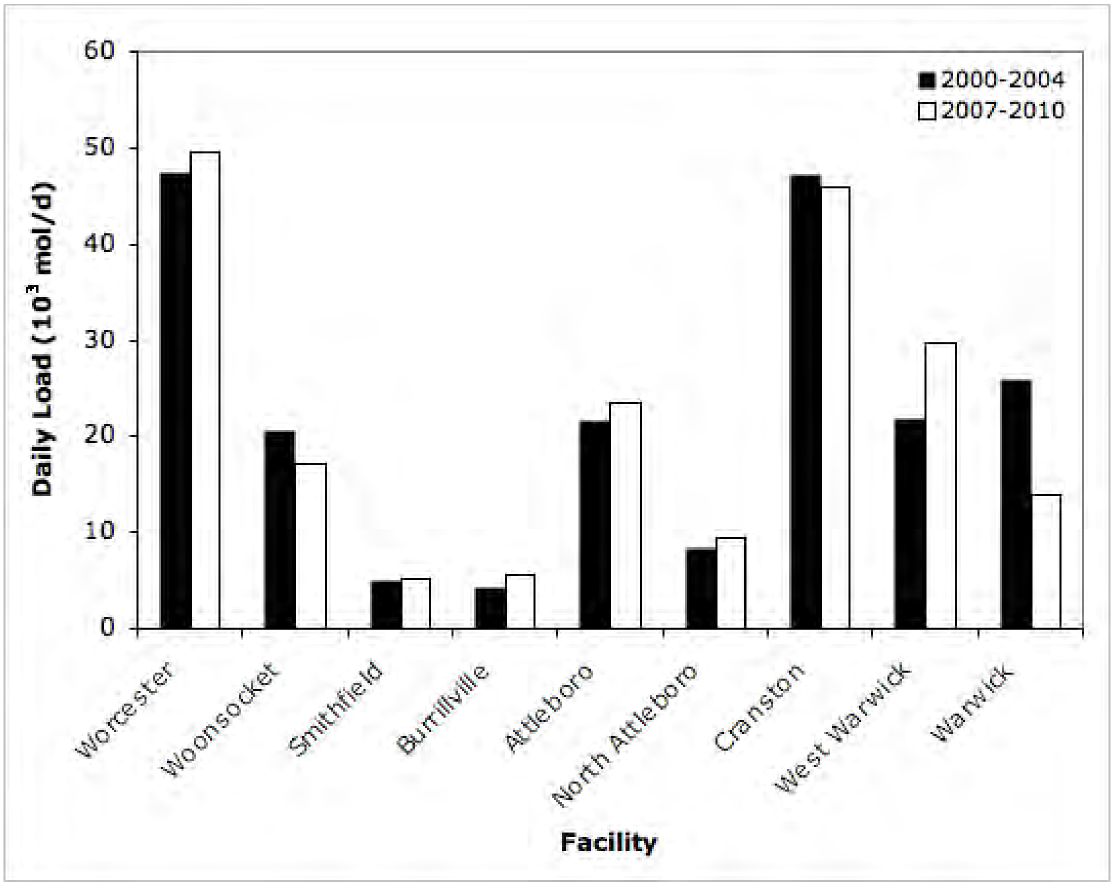

Figure B-40. Inactive season daily total nitrogen load discharged from facilities on rivers that drain to Narragansett Bay. The Grafton, Millbury, Hopedale, Leicester, Douglas, Upton, Brockton, Taunton, and Somerset facilities were not included as there was no seasonal data available.

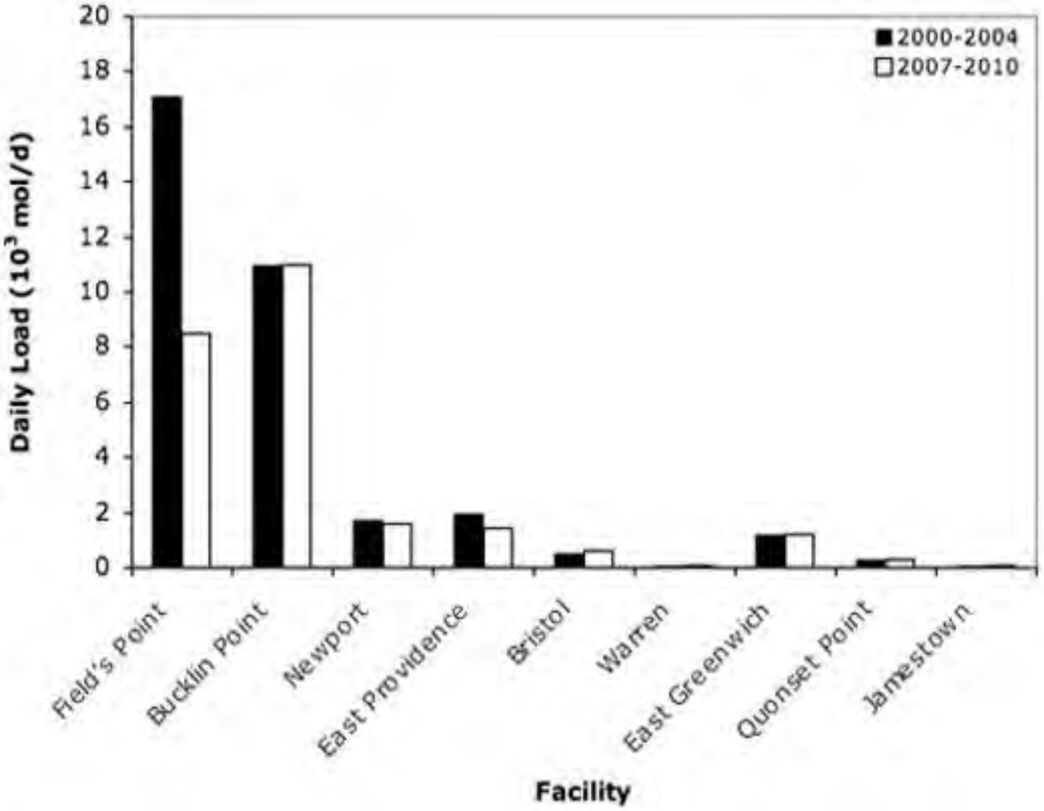

Figure B-41. Inactive season daily total phosphorus load from facilities that directly discharge to Narragansett Bay. The Fall River facility was not included as there was no seasonal data available.

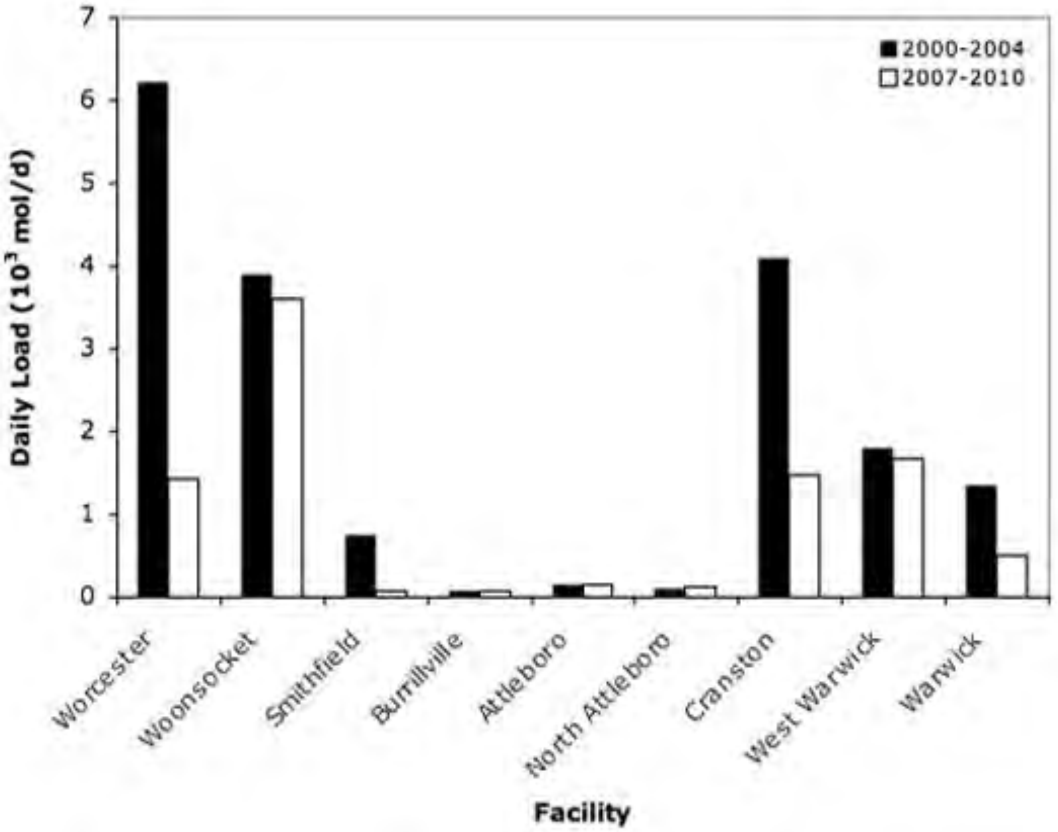

Figure B-42. Inactive season daily total phosphorus load discharged from facilities on rivers that drain to Narragansett Bay. The Grafton, Millbury, Hopedale, Leicester, Douglas, Upton, Brockton, Taunton, and Somerset facilities were not included as there was no seasonal data available.

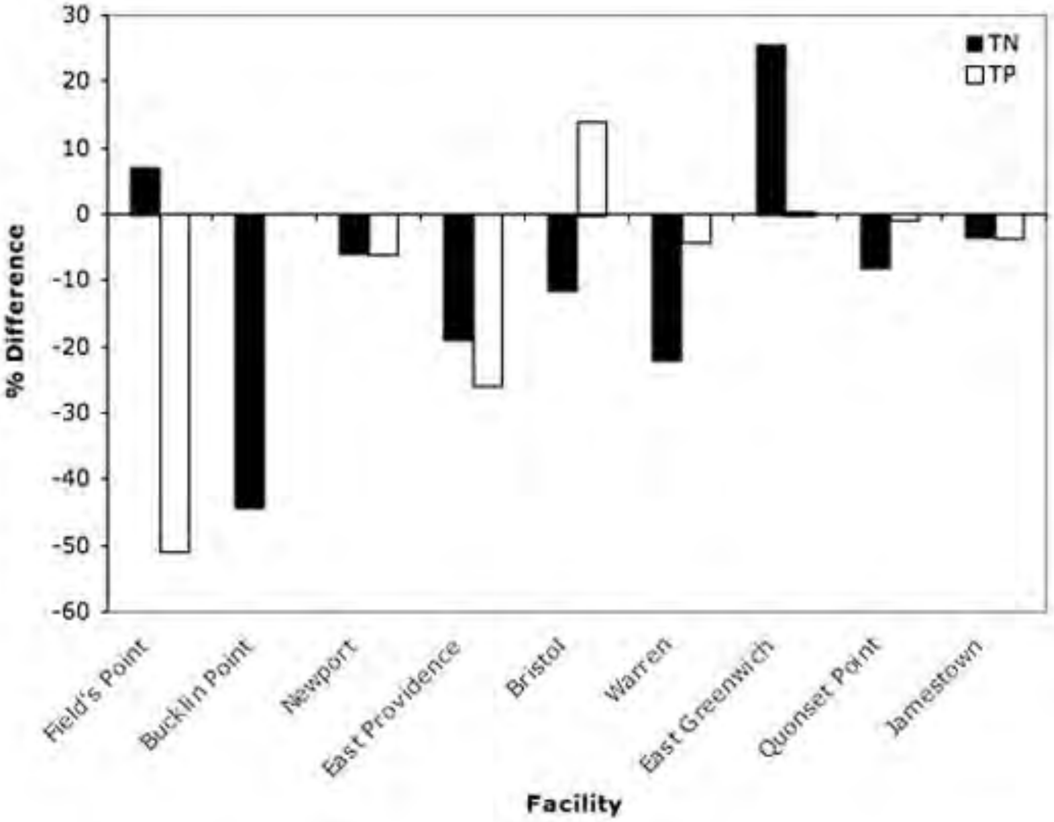

Figure B-43. Inactive season percent difference in total nitrogen and total phosphorus loads from facilities that directly discharge to Narragansett Bay in 2007-2010 relative to 2000-2004. The Fall River facility was not included as there was no seasonal data available.

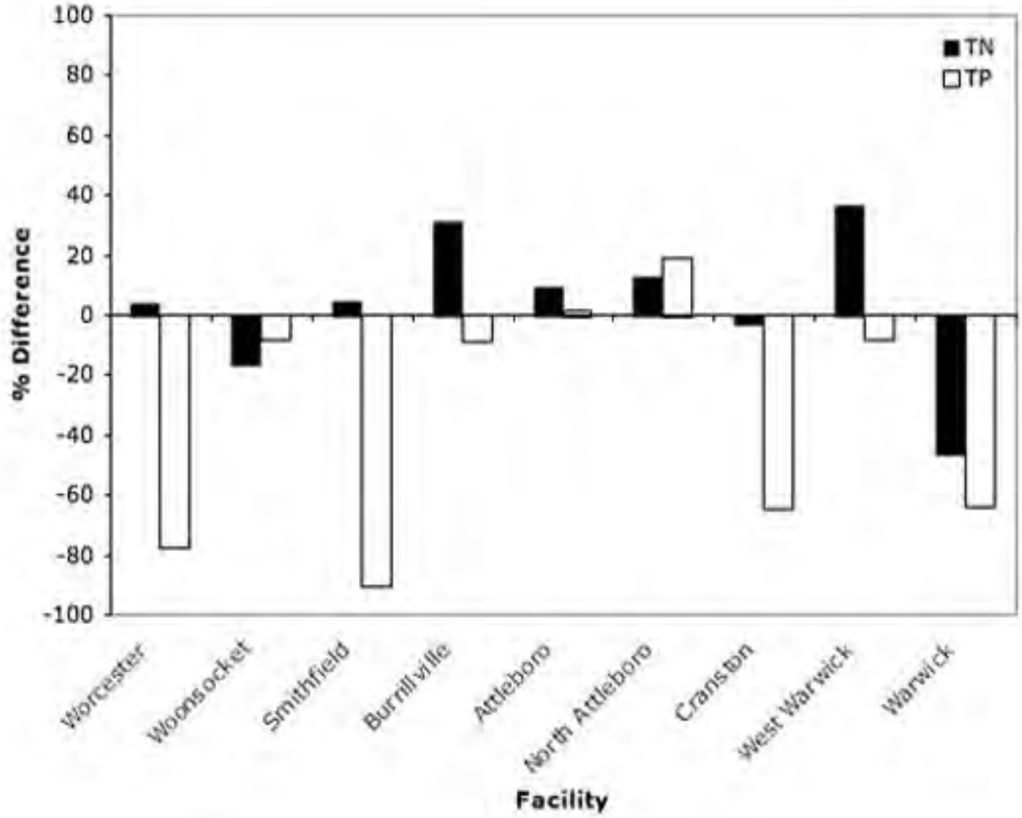

Figure B-44. Inactive season percent difference in total nitrogen and total phosphorus loads discharged from facilities on rivers that drain to Narragansett Bay in 2007-2010 relative to 2000-2004. The Grafton, Millbury, Hopedale, Leicester, Douglas, Upton, Brockton, Taunton, and Somerset facilities were not included as there was no seasonal data available.

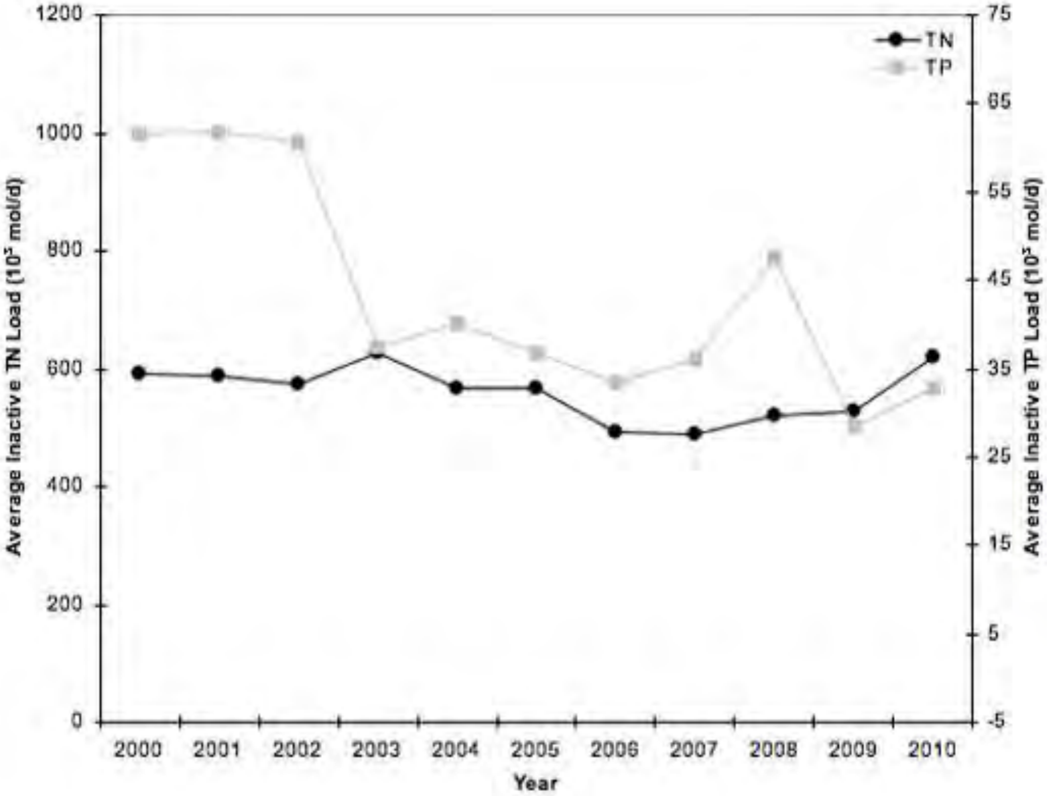

Figure B-45. Average inactive season total sewage nitrogen and total sewage phosphorus load discharged from all facilities with load data available combined over the 2000-2010 time period.

APPENDIX C

CODE FOR MATLAB AND R

```
                          ANALYSIS OF COVARIANCE IN MATLAB
This code was written with the assistance of Matt Horn
%First input your data
%Next rename them using the colheaders or textdata to identify what is
%what.
Year=data(:,1);
Distance=data(:,2);
DIN=data(:,3);
PO4=data(:,4);
NH3=data(:,5);
NOx=data(:,6);
SiO4=data(:,7);
%Clear extraneous data and keep your "data" which is equal to "raw"
clear colheaders textdata

%Take the log transform of 4 variables
ln_DIN=log(DIN);
ln_PO4=log(PO4);
ln_NH3=log(NH3);
ln_NOx=log(NOx);
ln_SiO4=log(SiO4);
%Use indeces to find the point identifier for given years - note... this is
%NOT the value... it's the location of those values in the matrix.
index1980=find(Year==1980);
index2006=find(Year==2006);
index2007=find(Year==2007);
index2008=find(Year==2008);
index2009=find(Year==2009);
index2010=find(Year==2010);

%%Make a matrix that is your year labels.
%yearlabel=char('1980','2006','2007','2008','2009','2010');




%Make an average that includes 2006-2010
temp_mean_ln_DIN=[ln_DIN(index2006) ln_DIN(index2007) ln_DIN(index2008)
ln_DIN(index2009) ln_DIN(index2010)];
temp_mean_ln_PO4=[ln_PO4(index2006) ln_PO4(index2007) ln_PO4(index2008)
ln_PO4(index2009) ln_PO4(index2010)];
temp_mean_ln_NH3=[ln_NH3(index2006) ln_NH3(index2007) ln_NH3(index2008)
ln_NH3(index2009) ln_NH3(index2010)];
temp_mean_ln_NOx=[ln_NOx(index2006) ln_NOx(index2007) ln_NOx(index2008)
ln_NOx(index2009) ln_NOx(index2010)];
temp_mean_ln_SiO4=[ln_SiO4(index2006) ln_SiO4(index2007) ln_SiO4(index2008)
ln_SiO4(index2009) ln_SiO4(index2010)];

mean_06_10_ln_DIN=mean(temp_mean_ln_DIN,2);
mean_06_10_ln_PO4=mean(temp_mean_ln_PO4,2);
```

```
mean_06_10_ln_NH3=mean(temp_mean_ln_NH3,2);
mean_06_10_ln_NOx=mean(temp_mean_ln_NOx,2);
mean_06_10_ln_SiO4=mean(temp_mean_ln_SiO4,2);
clear temp_mean_ln_DIN temp_mean_ln_PO4 temp_mean_ln_NH3 temp_mean_ln_NOX
temp_mean_ln_SiO4

%Plot up the raw data based upon year.
figure(1);clf;hold on;
subplot(5,1,1);
    plot(Distance(index1980),ln_DIN(index1980),'.k');hold on;
    plot(Distance(index2006),ln_DIN(index2006),'xr');hold on;
    plot(Distance(index2007),ln_DIN(index2007),'ob');hold on;
    plot(Distance(index2008),ln_DIN(index2008),'+g');hold on;
    plot(Distance(index2009),ln_DIN(index2009),'*m');hold on;
    plot(Distance(index2010),ln_DIN(index2010),'<k');hold on;
    plot(Distance(index2006),mean_06_10_ln_DIN,'cd');hold on;
    title('DIN');
        xlabel('distance (km)');ylabel('DIN')
        legend('1980','2006','2007','2008','2009','2010','06-10 mean')

subplot(5,1,2);
    plot(Distance(index1980),ln_PO4(index1980),'.k');hold on;
    plot(Distance(index2006),ln_PO4(index2006),'xr');hold on;
    plot(Distance(index2007),ln_PO4(index2007),'ob');hold on;
    plot(Distance(index2008),ln_PO4(index2008),'+g');hold on;
    plot(Distance(index2009),ln_PO4(index2009),'*m');hold on;
    plot(Distance(index2010),ln_PO4(index2010),'<k');hold on;
    plot(Distance(index2006),mean_06_10_ln_PO4,'cd');hold on;
     title('PO4');
        xlabel('distance (km)');ylabel('PO4')

subplot(5,1,3);
    plot(Distance(index1980),ln_NH3(index1980),'.k');hold on;
    plot(Distance(index2006),ln_NH3(index2006),'xr');hold on;
    plot(Distance(index2007),ln_NH3(index2007),'ob');hold on;
    plot(Distance(index2008),ln_NH3(index2008),'+g');hold on;
    plot(Distance(index2009),ln_NH3(index2009),'*m');hold on;
     plot(Distance(index2010),ln_NH3(index2010),'<k');hold on;
    plot(Distance(index2006),mean_06_10_ln_NH3,'cd');hold on;
     title('NH3');
        xlabel('distance (km)');ylabel('NH3')

subplot(5,1,4);
    plot(Distance(index1980),ln_NOx(index1980),'.k');hold on;
    plot(Distance(index2006),ln_NOx(index2006),'xr');hold on;
    plot(Distance(index2007),ln_NOx(index2007),'ob');hold on;
    plot(Distance(index2008),ln_NOx(index2008),'+g');hold on;
    plot(Distance(index2009),ln_NOx(index2009),'*m');hold on;
     plot(Distance(index2010),ln_NOx(index2010),'<k');hold on;
    plot(Distance(index2006),mean_06_10_ln_NOx,'cd');hold on;
     title('NOx');
        xlabel('distance (km)');ylabel('NOx')
subplot(5,1,5);
    plot(Distance(index1980),ln_SiO4(index1980),'.k');hold on;
    plot(Distance(index2006),ln_SiO4(index2006),'xr');hold on;
    plot(Distance(index2007),ln_SiO4(index2007),'ob');hold on;
```

```
    plot(Distance(index2008),ln_SiO4(index2008),'+g');hold on;
    plot(Distance(index2009),ln_SiO4(index2009),'*m');hold on;
     plot(Distance(index2010),ln_SiO4(index2010),'<k');hold on;
    plot(Distance(index2006),mean_06_10_ln_SiO4,'cd');hold on;
     title('SiO4');
        xlabel('distance (km)');ylabel('SiO4')

%ANCOVA-tron
%This version tests means
xval=[Distance(index1980); Distance(index2006)];%Distance
yval=[ln_SiO4(index1980); mean_06_10_ln_SiO4];%experimental variable
gval=[Year(index1980); Year(index2006)];% year
%this version tests years independently
%xval=[Distance(index1980);Distance(index2006);Distance(index2007);Distance(i
ndex2008);Distance(index2009);Distance(index2010);Distance(index2006)];
%DISTANCE
%yval=[ln_NOx(index1980);ln_NOx(index2006);ln_NOx(index2007);ln_NOx(index2008
);ln_NOx(index2009);ln_NOx(index2010);mean_06_10_ln_NOx];      %LN_DIN
%gval=[Year(index1980);Year(index2006);Year(index2007);Year(index2008);Year(i
ndex2009);Year(index2010);1;1;1;1;1;1;1;1;1;1;1;1;1];          %YEAR

[h,atab,ctab,stats] =
aoctool(xval,yval,gval,0.05,'Distance','ln_SiO4','Year');

multcompare(stats,0.05,'on','','intercept');% multiple comparison of
intercepts
%multcompare(stats,0.05,'on','','slope');% multiple comparison of slopes
```

Shapiro-Wilk Test in Matlab

This code was obtained through the Matlab File Exchange at:
http://www.mathworks.com/matlabcentral/fileexchange/13964

```
function [H, pValue, W] = swtest(x, alpha, tail)
%SWTEST Shapiro-Wilk parametric hypothesis test of composite normality.
%   [H, pValue, SWstatistic] = SWTEST(X, ALPHA, TAIL) performs
%   the Shapiro-Wilk test to determine if the null hypothesis of
%   composite normality is a reasonable assumption regarding the
%   population distribution of a random sample X. The desired significance
%   level, ALPHA, is an optional scalar input (default = 0.05).
%   TAIL indicates the type of test (default = 1).
%
%   The Shapiro-Wilk hypotheses are:
%   Null Hypothesis:        X is normal with unspecified mean and variance.
%      For TAIL =  0 (2-sided test), alternative: X is not normal.
%      For TAIL =  1 (1-sided test), alternative: X is upper the normal.
%      For TAIL = -1 (1-sided test), alternative: X is lower the normal.
%
%   This is an omnibus test, and is generally considered relatively
%   powerful against a variety of alternatives.
%   Shapiro-Wilk test is better than the Shapiro-Francia test for
%   Platykurtic sample. Conversely, Shapiro-Francia test is better than the
%   Shapiro-Wilk test for Leptokurtic samples.
%
%   When the series 'X' is Leptokurtic, SWTEST performs the Shapiro-Francia
%   test, else (series 'X' is Platykurtic) SWTEST performs the
%   Shapiro-Wilk test.
%
%    [H, pValue, SWstatistic] = SWTEST(X, ALPHA, TAIL)
%
% Inputs:
%   X - a vector of deviates from an unknown distribution. The observation
%     number must exceed 3 and less than 5000.
%
% Optional inputs:
%   ALPHA - The significance level for the test (default = 0.05).
%
%   TAIL  - The type of the test (default = 1).
%
% Outputs:
%  SWstatistic - The test statistic (non normalized).
%
%   pValue - is the p-value, or the probability of observing the given
%     result by chance given that the null hypothesis is true. Small values
%     of pValue cast doubt on the validity of the null hypothesis.
%
%     H = 0 => Do not reject the null hypothesis at significance level ALPHA.
%     H = 1 => Reject the null hypothesis at significance level ALPHA.
%

%
% References: Royston P. "Algorithm AS R94", Applied Statistics (1995) Vol.
44, No. 4.
%   AS R94 -- calculates Shapiro-Wilk normality test and P-value
%   for sample sizes 3 <= n <= 5000. Handles censored or uncensored data.
%   Corrects AS 181, which was found to be inaccurate for n > 50.
%
```

```
%
% Ensure the sample data is a VECTOR.
%

if numel(x) == length(x)
    x  =  x(:);                % Ensure a column vector.
else
    error(' Input sample ''X'' must be a vector.');
end

%
% Remove missing observations indicated by NaN's and check sample size.
%

x  =  x(~isnan(x));

if length(x) < 3
   error(' Sample vector ''X'' must have at least 3 valid observations.');
end

if length(x) > 5000
    warning('Shapiro-Wilk test might be inaccurate due to large sample size (
> 5000).');
end

%
% Ensure the significance level, ALPHA, is a
% scalar, and set default if necessary.
%

if (nargin >= 2) && ~isempty(alpha)
   if numel(alpha) > 1
      error(' Significance level ''Alpha'' must be a scalar.');
   end
   if (alpha <= 0 || alpha >= 1)
      error(' Significance level ''Alpha'' must be between 0 and 1.');
   end
else
   alpha  =  0.05;
end

%
% Ensure the type-of-test indicator, TAIL, is a scalar integer from
% the allowable set [-1 , 0 , 1], and set default if necessary.
%

if (nargin >= 3) && ~isempty(tail)
   if numel(tail) > 1
      error('Type-of-test indicator ''Tail'' must be a scalar.');
   end
   if (tail ~= -1) && (tail ~= 0) && (tail ~= 1)
      error('Type-of-test indicator ''Tail'' must be -1, 0, or 1.');
   end
else
```

```
   tail  =  1;
end

% First, calculate the a's for weights as a function of the m's
% See Royston (1995) for details in the approximation.

x       =   sort(x); % Sort the vector X in ascending order.
n       =   length(x);
mtilde  =   norminv(((1:n)' - 3/8) / (n + 0.25));
weights =   zeros(n,1); % Preallocate the weights.

if kurtosis(x) > 3

    % The Shapiro-Francia test is better for leptokurtic samples.

    weights =   1/sqrt(mtilde'*mtilde) * mtilde;

    %
    % The Shapiro-Francia statistic W is calculated to avoid excessive
rounding
    % errors for W close to 1 (a potential problem in very large samples).
    %

    W   =   (weights' * x) ^2 / ((x - mean(x))' * (x - mean(x)));

    nu      =   log(n);
    u1      =   log(nu) - nu;
    u2      =   log(nu) + 2/nu;
    mu      =   -1.2725 + (1.0521 * u1);
    sigma   =   1.0308 - (0.26758 * u2);

    newSFstatistic   =   log(1 - W);

    %
    % Compute the normalized Shapiro-Francia statistic and its p-value.
    %

    NormalSFstatistic =   (newSFstatistic - mu) / sigma;

    % the next p-value is for the tail = 1 test.
    pValue   =   1 - normcdf(NormalSFstatistic, 0, 1);

else

    % The Shapiro-Wilk test is better for platykurtic samples.

    c    =   1/sqrt(mtilde'*mtilde) * mtilde;
    u    =   1/sqrt(n);

    PolyCoef_1   =   [-2.706056 , 4.434685 , -2.071190 , -0.147981 , 0.221157
, c(n)];
    PolyCoef_2   =   [-3.582633 , 5.682633 , -1.752461 , -0.293762 , 0.042981
, c(n-1)];
```

```
    PolyCoef_3    =   [-0.0006714 , 0.0250540 , -0.39978 , 0.54400];
    PolyCoef_4    =   [-0.0020322 , 0.0627670 , -0.77857 , 1.38220];
    PolyCoef_5    =   [0.00389150 , -0.083751 , -0.31082 , -1.5861];
    PolyCoef_6    =   [0.00303020 , -0.082676 , -0.48030];

    PolyCoef_7    =   [0.459 , -2.273];

    weights(n)    =   polyval(PolyCoef_1 , u);
    weights(1)    =   -weights(n);

    % Special attention when n=3 (this is a special case).
    if n == 3
        weights(1)   =   0.707106781;
        weights(n)   =   -weights(1);
    end

    if n >= 6
        weights(n-1) =   polyval(PolyCoef_2 , u);
        weights(2)   =   -weights(n-1);

        count   =   3;
        phi     =   (mtilde'*mtilde - 2 * mtilde(n)^2 - 2 * mtilde(n-1)^2) /
...
                (1 - 2 * weights(n)^2 - 2 * weights(n-1)^2);
    else
        count   =   2;
        phi     =   (mtilde'*mtilde - 2 * mtilde(n)^2) / ...
                (1 - 2 * weights(n)^2);
    end

    %
    % The vector 'WEIGHTS' obtained next corresponds to the same coefficients
    % listed by Shapiro-Wilk in their original test for small samples.
    %

    weights(count : n-count+1)  =  mtilde(count : n-count+1) / sqrt(phi);

    %
    % The Shapiro-Wilk statistic W is calculated to avoid excessive rounding
    % errors for W close to 1 (a potential problem in very large samples).
    %

    W   =   (weights' * x) ^2 / ((x - mean(x))' * (x - mean(x)));

    %
    % Calculate the significance level for W (exact for n=3).
    %

    newn    =   log(n);

    if (n > 3) && (n <= 11)

        mu      =   polyval(PolyCoef_3 , n);
```

```
        sigma    =   exp(polyval(PolyCoef_4 , n));
        gam      =   polyval(PolyCoef_7 , n);

        newSWstatistic  =   -log(gam-log(1-W));

    elseif n >= 12

        mu       =   polyval(PolyCoef_5 , newn);
        sigma    =   exp(polyval(PolyCoef_6 , newn));

        newSWstatistic  =   log(1 - W);

    elseif n == 3
        mu       =   0;
        sigma    =   1;
        newSWstatistic  =   0;
    end

    %
    % Compute the normalized Shapiro-Wilk statistic and its p-value.
    %

    NormalSWstatistic       =   (newSWstatistic - mu) / sigma;

    % The next p-value is for the tail = 1 test.
    pValue       =   1 - normcdf(NormalSWstatistic, 0, 1);

    % Special attention when n=3 (this is a special case).
    if n == 3
        pValue  =   1.909859 * (asin(sqrt(W)) - 1.047198);
        NormalSWstatistic =   norminv(pValue, 0, 1);
    end

end

% The p-value just found is for the tail = 1 test.
if tail == 0
    pValue = 2 * min(pValue, 1-pValue);
elseif tail == -1
    pValue = 1 - pValue;
end

%
% To maintain consistency with existing Statistics Toolbox hypothesis
% tests, returning 'H = 0' implies that we 'Do not reject the null
% hypothesis at the significance level of alpha' and 'H = 1' implies
% that we 'Reject the null hypothesis at significance level of alpha.'
%

H  = (alpha >= pValue);
```

2 sided 2 tailed Kolmogorov Smirnov test in Matlab

```
%Script for Importing and running data for 2 sided Kolmogorov Smirnov
%distribution test%
%Jason Krumholz September, 2011
%1 Import the data to a matrix of 12 rows by X columns called 'data'
%Name the variables
NOx0610=data(:,2);
DIN0610=data(:,3);
PO40610=data(:,4);
SiO20610=data(:,5);
NH30610=data(:,6);
TN0610=data(:,7);
TP0610=data(:,8);
NOx7980=data(:,9);
DIN7980=data(:,10);
PO47980=data(:,11);
SiO27980=data(:,12);
NH37980=data(:,13);
TN1998=data(:,14);
TP1998=data(:,15);
[hNOx,pNOX,kNOx] = kstest2(NOx0610,NOx7980)
[hDIN,pDIN,kDIN] = kstest2(DIN0610,DIN7980)
[hPO4,pPO4,kPO4] = kstest2(PO40610,PO47980)
[hSiO2,pSiO2,kSiO2] = kstest2(SiO20610,SiO27980)
[hNH3,pNH3,kNH3] = kstest2(NH30610,NH37980)
[hTN,pTN,kTN] = kstest2(TN0610,TN1998)
[hTP,pTP,kTP] = kstest2(TP0610,TP1998)
%Plot cumulative distribution frequencies
figure
subplot(4,2,1)
A0610 = cdfplot(NOx0610);
hold on
A7980 = cdfplot(NOx7980);
set(A0610,'LineWidth',2,'Color','r');
set(A7980,'LineWidth',2);
legend([A0610 A7980],'2006-2010 NOx','1979-1980 NOx','Location','SE');
subplot(4,2,2)
B0610 = cdfplot(DIN0610);
hold on
B7980 = cdfplot(DIN7980);
set(B0610,'LineWidth',2,'Color','r');
set(B7980,'LineWidth',2);
legend([B0610 B7980],'2006-2010 DIN','1979-1980 DIN','Location','SE');
subplot(4,2,3)
C0610 = cdfplot(PO40610);
hold on
C7980 = cdfplot(PO47980);
set(C0610,'LineWidth',2,'Color','r');
set(C7980,'LineWidth',2);
legend([C0610 C7980],'2006-2010 PO4','1979-1980 PO4','Location','SE');
subplot(4,2,4)
D0610 = cdfplot(SiO20610);
hold on
D7980 = cdfplot(SiO27980);
set(D0610,'LineWidth',2,'Color','r');
set(D7980,'LineWidth',2);
legend([D0610 D7980],'2006-2010 SiO2','1979-1980 SiO2','Location','SE');
```

```
subplot(4,2,7)
E0610 = cdfplot(TN0610);
hold on
E7980 = cdfplot(TN1998);
set(E0610,'LineWidth',2,'Color','r');
set(E7980,'LineWidth',2);
legend([E0610 E7980],'2006-2010 TN','1998 TN','Location','SE');
subplot(4,2,6)
G0610 = cdfplot(NH30610);
hold on
G7980 = cdfplot(NH37980);
set(G0610,'LineWidth',2,'Color','r');
set(G7980,'LineWidth',2);
legend([G0610 G7980],'2006-2010 NH4','1979-1980 NH4','Location','SE');
subplot(4,2,8)
F0610 = cdfplot(TP0610);
hold on
F7980 = cdfplot(TP1998);
set(F0610,'LineWidth',2,'Color','r');
set(F7980,'LineWidth',2);
legend([F0610 F7980],'2006-2010 TP','1998 TP','Location','SE');
```

SSPIR code in R

This code was written with the assistance of Claus Dethlefsen and Rich Bell.

```
#       7/15/11

#               Krumholz nutrient data

#        we shall try with SSPIR

library(sspir)

t98 <-read.table( "T98interpolated.csv", header=T, sep=',',stringsAsFactors=F)
head(t98)

# pick out essential info
fav <- c("CHLa","NO2.NO3","PO4","NH4","DIN","Nint","Pint")
t98.small <- t98[,fav]
# (I found an NA in "SiO2" so I left this one out)

tt <- 1:nrow(t98.small)
t98.small$tt <- 1:nrow(t98.small)
t98.small$s1 <- sin(t98.small$tt*2*pi/52)
t98.small$c1 <- cos(t98.small$tt*2*pi/52)
t98.small$s2 <- sin(t98.small$tt*2*2*pi/52)
t98.small$c2 <- cos(t98.small$tt*2*2*pi/52)
t98.small$s3 <- sin(t98.small$tt*3*2*pi/52)
t98.small$c3 <- cos(t98.small$tt*3*2*pi/52)
t98.small$s4 <- sin(t98.small$tt*4*2*pi/52)
t98.small$c4 <- cos(t98.small$tt*4*2*pi/52)
t98.small$Nint <- as.factor(t98.small$Nint)
t98.small$Pint <- as.factor(t98.small$Pint)

t98.ts <- ts(t98[,fav], frequency = 52, start = c(1978, 1))
plot(t98.ts[,fav])

require(graphics)
t98.decomp <- decompose(t98.ts[,fav],type="additive")
plot(t98.decomp$trend) # moving average

library(rms)
n.group <- 1
d <- datadist(t98.small)
options(datadist="d")

describe(t98.small)
```

```
# inspect a histogram of the CHLa
hist(t98.small$CHLa)
hist(log(t98.small$CHLa))
# use log instead of raw measurements.
par(mfcol=c(5,2))
for (i in 1:5) hist(t98.small[,i])
for (i in 1:5) hist(log(t98.small[,i]))
par(mfrow=c(1,1))

## ordinary least squares models
# just trend and interventions
# trend is a restricted cubic spline with 7 knots
m1 <- ols(log(CHLa)~rcs(tt,7)+Nint+Pint,data=t98.small)
m1 <- ols(log(DIN)~rcs(tt,7)+Nint+Pint,data=t98.small)
m1 <- ols(log(PO4)~rcs(tt,7)+Nint+Pint,data=t98.small)

anova(m1)
summary(m1)
# Nint: 0.35 (0.05;0.66) ie (exp(0.35)-1)*100%=42% increase, p=2%
# Pint: -0.26 (-0.57;0.05) ie 23% decrease, p=11%

# adjust for one sine-cosine
m2 <- ols(log(CHLa)~rcs(tt,7)+c1+s1+Nint+Pint,data=t98.small)
m2 <- ols(log(NH4)~rcs(tt,7)+c1+s1+Nint+Pint,data=t98.small)
m2 <- ols(log(DIN)~rcs(tt,7)+c1+s1+Nint+Pint,data=t98.small)
m2 <- ols(log(PO4)~rcs(tt,7)+c1+s1+Nint+Pint,data=t98.small)

anova(m2)
summary(m2)
# Nint: 0.28 (-0.02;0.59) ie 32% increase, p=7%
# Pint: -0.32 (-0.62;-0.01) ie 27% decrease, p=4%

# adjust for four sine-cosines
m3 <- ols(log(CHLa)~rcs(tt,7)+c1+s1+c2+s2+c3+s3+c4+s4+Nint+Pint,data=t98.small)
m3 <- ols(log(DIN)~rcs(tt,7)+c1+s1+c2+s2+c3+s3+c4+s4+Nint+Pint,data=t98.small)
anova(m3)
summary(m3)
acf(resid(m3))
# Nint: 0.24 (-0.06;0.53) ie 27% increase, p=12%
# Pint: -0.35 (-0.66;-0.05) ie 30% decrease, p=2%
# note that the autocorrelation function of the residuals looks
# "terrible". There is a strong serial correlation. That's why the
# simple models do not work and we turn to time series models, such as
# state space models.

# A "simple" state space model is the Basic Structural Model, built in
```

```
# to R. It includes level, slope, "sum-to-season" and residuals. It is
# very efficient at maximizing the likelihood and estimating the
# variance parameters.
t98.i <- StructTS(log(t98.ts[,1]),type="BSM")
plot(cbind(fitted(t98.i),resids=resid(t98.i)))
print(t98.i$coef)
acf(resid(t98.i))
#level    slope season    eps
#0.1867393 0 5.350514e-06 0.2672746
phihat <- c(0.1867393, 0, 5.350514e-06, 0.2672746)
# note that the slope variance parameter is estimated to 0, meaning
# that the slope is not time-varying. Thus the trend reduces to a
# local level model.

# the bad thing about StructTS is that it cannot handle covariates.
# That's why we turn to sspir and formulate the same model as BSM in
# StructTS but add the two covariates Nint and Pint.

# The bad thing about sspir is that it does not estimate the variance
# parameters. You need to use some kind of numerical maximization
# algorithm and it might take forever. We thus just take the estimated
# parameters from StructTS and plug in. This is not quite legal since
# the parameters are estimated without taking the covariates into
# account. We ignore that for now.... If you were to do it right, you
# would take this as initial values and then find the
# phi-configuration that maximizes kfs(yourmodel)$loglik


#######################

#         Chl a is dependent variable


## the big model with timevarying season and trend.
## variance parameters are taken from the BSM model from StructTS

sm1 <- ssm( log(t98.ts[,1]) ~ -1+tvar(polytime(tt,1)) + tvar(sumseason(tt,52)) +
   t98.ts[,6] + t98.ts[,7],fit=FALSE)
phi(sm1)[c(4,1,2,3)] <- phihat
sm1.fit <- kfs(sm1)
Nint <- sm1.fit$m[1,54] # since it is static, all m's are the same
Pint <- sm1.fit$m[1,55]
# Nint: 0.23, ie 26% increase
# Pint: 0.51, ie 67% increase
sdNint <- sqrt(diag(sm1.fit$C[[1]])[54])
sdPint <- sqrt(diag(sm1.fit$C[[1]])[55])
```

```
# thus a 95% confidence interval can be obtained:
(exp(c(Nint - 1.96*sdNint, Nint + 1.96*sdNint))-1)*100
#very wide ...

# local level model for the trend
sm2 <- ssm( log(t98.ts[,1]) ~ tvar(1) + tvar(sumseason(tt,52)) +
   t98.ts[,6] + t98.ts[,7],fit=FALSE)
phi(sm2)[c(4,1,3)] <- phihat
sm2.fit <- kfs(sm2)
Nint <- sm2.fit$m[1,53]
Pint <- sm2.fit$m[1,54]
# Nint: 0.07, ie 7% increase
# Pint: -0.54, ie 42% decrease
sdNint <- sqrt(diag(sm2.fit$C[[1]])[53])
sdPint <- sqrt(diag(sm2.fit$C[[1]])[54])
# thus a 95% confidence interval can be obtained:
(exp(c(Nint - 1.96*sdNint, Nint + 1.96*sdNint))-1)*100
(1-exp(c(Pint - 1.96*sdPint, Pint + 1.96*sdPint)))*100


########################################

#        Nitrogen as dependent variable

bad.egg<-which(log(t98.ts[,4])==min(log(t98.ts[,4])))
t98.ts[bad.egg,4]<-0.05
tt<-1:1716

## the big model with timevarying season and trend. for N and P
## variance parameters are taken from the BSM model from StructTS

sm1 <- ssm( log(t98.ts[,5]) ~ -1+tvar(polytime(tt,1)) + tvar(sumseason(tt,52)) +
   t98.ts[,6] + t98.ts[,7],fit=FALSE)  #        DIN

sm1 <- ssm( log(t98.ts[,4]) ~ -1+tvar(polytime(tt,1)) + tvar(sumseason(tt,52)) +
   t98.ts[,6] + t98.ts[,7],fit=FALSE)  #        NH4

phi(sm1)[c(4,1,2,3)] <- phihat
sm1.fit <- kfs(sm1)
Nint <- sm1.fit$m[1,54] # since it is static, all m's are the same
Pint <- sm1.fit$m[1,55]
# Nint: 0.50,
# Pint: -0.07
sdNint <- sqrt(diag(sm1.fit$C[[1]])[54])
sdPint <- sqrt(diag(sm1.fit$C[[1]])[55])
# thus a 95% confidence interval can be obtained:
```

```
(exp(c(Nint - 1.645*sdNint, Nint + 1.645*sdNint))-1)*100
#very wide ...

(exp(Nint)-1)*100      #        %=42% increase, p=2%

(exp(-0.54)-1)*100
##################################

#        Nitrogen as dependent variable, w/o 2010 data

x.2009<-t98.ts[1:1664,]
par(family='serif',mfrow=c(3,3),mar=c(2,2,2,2))
for(i in 1:length(x.2009[1,]))
plot(x.2009[,i],typ='l',main=colnames(x.2009)[i])
}
tt<-1:1664
## the big model with timevarying season and trend. for N and P
## variance parameters are taken from the BSM model from StructTS

sm1 <- ssm( log(x.2009[,5]) ~ -1+tvar(polytime(tt,1)) + tvar(sumseason(tt,52)) +
   x.2009[,6] + x.2009[,7],fit=FALSE)
phi(sm1)[c(4,1,2,3)] <- phihat
sm1.fit <- kfs(sm1)
Nint <- sm1.fit$m[1,54] # since it is static, all m's are the same
Pint <- sm1.fit$m[1,55]
# Nint: 0.54
# Pint: -0.70
sdNint <- sqrt(diag(sm1.fit$C[[1]])[54])
sdPint <- sqrt(diag(sm1.fit$C[[1]])[55])
# thus a 95% confidence interval can be obtained:
(exp(c(Nint - 1.96*sdNint, Nint + 1.96*sdNint))-1)*100
#very wide ...



##### NH4


#        Nitrogen as dependent variable, w/o 2010 data

x.2009<-t98.ts[1:1664,]
par(family='serif',mfrow=c(3,3),mar=c(2,2,2,2))
for(i in 1:length(x.2009[1,]))
plot(x.2009[,i],typ='l',main=colnames(x.2009)[i])
}
tt<-1:1664
```

```
## the big model with timevarying season and trend. for N and P
## variance parameters are taken from the BSM model from StructTS

sm1 <- ssm( log(x.2009[,4]) ~ -1+tvar(polytime(tt,1)) + tvar(sumseason(tt,52)) +
   x.2009[,6] + x.2009[,7],fit=FALSE)
phi(sm1)[c(4,1,2,3)] <- phihat
sm1.fit <- kfs(sm1)
Nint <- sm1.fit$m[1,54] # since it is static, all m's are the same
Pint <- sm1.fit$m[1,55]
# Nint: -1.66
# Pint: --1.020
sdNint <- sqrt(diag(sm1.fit$C[[1]])[54])
sdPint <- sqrt(diag(sm1.fit$C[[1]])[55])
# thus a 95% confidence interval can be obtained:
(exp(c(Nint - 1.96*sdNint, Nint + 1.96*sdNint))-1)*100
#very wide ...


plot((log(x.2009[,4])),typ='l')

bad.egg<-which(log(x.2009[,4])==min(log(x.2009[,4])))

x.2009[bad.egg,4]<-0.05

t98.ts[bad.egg,]


plot(lowess(log(x.2009[,4])),typ='l')
```

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Attachment D

DRAFT

Nutrient Conditions in Narragansett Bay & Numeric Nutrient Criteria Development Strategies for Rhode Island Estuarine Waters

Provided to the R.I. Dept. of Environmental Management
Office of Water Resources

NARRAGANSETT BAY
ESTUARY PROGRAM

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Numeric Nutrient Criteria Development Strategies for R.I. Estuarine Waters

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Background

This document is provided to the RIDEM OWR to lay out a conceptual approach to developing numerical nutrient criteria for marine waters of the state. As the State agency responsible for administering water quality standards and criteria, the Rhode Island Department of Environmental Management (RIDEM) previously committed to developing refined nutrient criteria to strengthen protection of Rhode Island's surface waters. The negative effects of excessive loadings of nutrients to both fresh and estuarine waters constitute a well recognized water quality concern in the State. Over the last decade, a range of actions have been taken to better control and mitigate nutrient pollution. Refining water quality standards through the adoption of numeric nutrient criteria is intended to strengthen the basis for future mitigation and management of nutrient water pollution including allocation of acceptable pollutant loadings as determined in new total maximum daily load (TMDL) studies.

Point Sources

In its current water quality regulations, RIDEM has specified a numeric limit for total phosphorus (TP) in freshwater lakes and ponds, and rivers at the point they enter lakes and ponds, but otherwise relied on narrative criteria to support management decision-making. It is important to note that while the narrative criteria are inherently more general, RIDEM has successfully relied on them to advance management of nutrient pollutant loadings from major point sources in Rhode Island. Specific effluent limits for Total Nitrogen or Total Phosphorus, or both, have been incorporated into 12 of 19 permits for major public wastewater facilities regulated under the Rhode Island Pollutant Discharge Elimination System (RIPDES) Program. Rhode Island is in the midst of implementing a strategy to mitigate the adverse effects of eutrophication in the upper Narragansett Bay by reducing the nitrogen pollutant loadings from eleven (11) Rhode Island wastewater treatment facilities (WWTFs) by 50% (as compared to 1995-1996 levels). Permits for these WWTFs include effluent limits for Total Nitrogen (TN). Major investments to upgrade these wastewater treatment facilities have been completed or are planned for advanced (tertiary) treatment through 2014 (see Table 1). Eight WWTFs have completed upgrades while four others are in varying stages of planning, design or construction. The upgrade of the Narragansett Bay Commission (NBC) WWTF at Bucklin Point was completed in late 2005, and the NBC Field's Point plant, the State's largest, is currently targeted for completion by the end of 2013. Rhode Island is seeking similar reductions in nitrogen loadings from WWTFs in Massachusetts that discharge into rivers tributary to the upper Bay region.

Rhode Island Waste Water Treatment Facility Upgrades Timeline

WWTF	Permitted Max Flow (MGD)	Average Annual Flows (avg. MGD)	Initiate Secondary Treament	Pre-treatment Program Approved	Dechlorination (UV)	Tertiary Complete Construction		Permit or Modification	Advanced Treatment Status (permit limit in mg/L)
						Due	Finished		
BRISTOL	3.8	2.8	02/06/90	02/29/84	1996			7/28/99	Nitrogen Not Evaluated
BURRILLVILLE	1.5	0.8	11/1/81		pre 1993	NA		9/7/00	Ammonia (8.9), Nitrogen (Max extent)
CRANSTON	19	10.6	03/20/87	09/13/84	2001	11/1/05 (TN & P)	1/26/06	6/1/00	Ammonia (2), Nitrogen (8), Phosphorus (1)
EAST GREENWICH	1.24	1.4	03/15/89	10/03/84	(2003)	3/06	3/06	9/28/01	Nitrogen (5)
EAST PROVIDENCE	10.4	7.9	08/01/77	09/07/83	1998			6/28/05	Nitrogen (8)
JAMESTOWN	0.75	0.4	11/17/81		NA			9/25/00	Not Anticipated
NARRAGANSETT	1.4	0.6	09/26/80		1996			3/29/00	Ammonia (32)
NARRAGANSETT BAY COMM-BUCKLIN (former BVDC)	46	26.2	03/30/86	09/07/83	1999 (2005)	9/06	12/05	12/31/90	Ammonia, Nitrogen (design target of 8)
								6/28/05	Nitrogen Modified to 5
NARRAGANSETT BAY COMM-FIELDS PT	65	49.7	06/15/88	09/14/84	1997			9/30/92	Ammonia, Nitrogen
								6/28/05	Nitrogen Modified to 5
NEW SHOREHAM	0.3	0.135	08/01/78		1995		5/27/05		Ammonia (11.2)
NEWPORT	10.7	8.4	02/28/91	03/30/85	2002				Nitrogen Not Evaluated
RIEDC/QUONSET	2.4	1		09/24/86	NA				Nitrogen Not Evaluated
SMITHFIELD	3.5	1.4	08/27/79	09/10/96	1996	7/21/06	6/6/08		Ammonia (2.7), Nitrogen (max extent) and Phosphorus (0.2)
SOUTH KINGSTOWN	5	2.4	1976	09/13/84	1997				Not Required
WARREN	2	1.8	11/1/90	09/10/98	1999				Ammonia, Nitrogen
WARWICK	5	3.4	12/11/90	02/09/84	2000	11/1/04	11/1/04		Ammonia (2), Phosphorus (1), Nitrogen (8)
WEST WARWICK	7.9	5.2	06/15/93	09/07/83	2000 (2005)	7/05	7/05		Ammonia (2), Phosphorus (1), Nitrogen (8)
WESTERLY	3.3	2.8	6/1/79	03/30/85	2002	10/03	10/03		Ammonia (5.5), Nitrogen (design target of 12)
WOONSOCKET	16	8.4	3/1/87	3/10/86	2000	9/01	9/01		Ammonia (2) Nitrogen (10), Phos (1) Nitrogen Mod (5)

*Source : RIDEM Office of water Resources June 2008.

Nonpoint and Stormwater Sources

Mitigating nutrient pollution from non-point sources presents a challenge. Given Rhode Island's densely developed landscape, the management focus has been on on-site wastewater systems and stormwater discharges. In 2008 RIDEM revised the state regulations governing on-site wastewater treatment systems to require advanced treatment for de-nitrification in areas deemed environmentally sensitive; e.g. watersheds of certain coastal ponds. Additional rules to compel the phase out of cesspools within proximity to certain waters became effective in August 2010. Rhode Island has adopted a new stormwater design manual, effective in early 2011, that requires the use of low impact development practices and will compel that stormwater discharges receive treatment to reduce pollutant loadings.

Need for Numeric Nutrient Criteria

EPA has established that excessive amounts of certain nutrients (nitrogen and phosphorus) in U.S. surface waters, including estuarine and salt waters, are a form of pollution leading to significant adverse ecological impacts violating the CWA. While encouraging states to continue the progress in decreasing nutrient loads, the EPA is requesting that states also move towards development of *numerical* nutrient criteria for estuarine and marine waters as loading or concentration limits due to the limitations of narrative criteria in a legal context such as permit concentration limits and TMDLs. Rhode Island, as with the other states, is developing nutrient limits for fresh waters , and beginning efforts to develop numerical nutrient criteria for marine waters. The major process and potential directions this work could focus on are discussed below. A practical summary of this issue is described in a report by Battelle for USEPA and the state of Maine , and we borrow heavily from that document, which outlines various options and approaches to this task (Battelle 2008).

"…The major problem with a numeric limit is that there are large variations in the natural physical, chemical, and biological characteristics of water resources (and adjacent lands) that influence how a particular waterbody responds to changes in nutrient loads. In order to take these variations into account, nutrient criteria must be established on appropriate spatial scales and not merely dictated on a national scale…Temporal scales may also be considered as nutrient dynamics can change seasonally…A technical guidance manual for developing nutrient criteria in estuarine and coastal marine waters was published in 2001 (EPA 2001)…(and)… highlights the importance of Nitrogen as the limiting nutrient in most coastal marine waters.[1] "

Rhode Island, like most states, has focused on the development of numeric nutrient criteria in freshwater systems (lakes/reservoirs and rivers/streams), while saltwater criteria are narrative in nature. This is not unexpected since development of salt water / estuarine numeric nutrient criteria is expected to be much more complex in nature due to the strong local hydrodynamic factors that influence responses in these tidally-influenced waterbodies. The present RI nutrient criteria for saltwater states that nutrients shall not be "in such concentration that would impair any usages specifically assigned to said Class, or cause undesirable or nuisance aquatic species associated with cultural eutrophication" (RI DEM WQ Regs 2009). The RI freshwater phosphorous limits are more specific and require a numeric limit for specific

[1] Battelle . 2008. Conceptual Plan for Nutrient Criteria Development in Maine Coastal Waters. Report to USEPA Region I & Oceans & Coastal Protection Divisions & State of Maine .February 22, 2008. Work Assignment No. 4-53 Project No. G921353

quantitative limits: “Average Total Phosphorus shall not exceed 0.025 mg/l in any lake, pond, kettle hole or reservoir, and average Total P in tributaries at the point where they enter such bodies of water shall not cause exceedance of this phosphorus criteria, except as naturally occurs, unless the Director determines, on a site-specific basis, that a different value for phosphorus is necessary to prevent cultural eutrophication.” (RI DEM 2009).

A number of states have developed or are proposing estuarine numeric nutrient criteria for N, P and/or response parameters (CT, DE, HI, MA; MD, NH, NY, and VA). Appendix A provides more detailed available information on many of these criteria The Maryland, Delaware and Virginia criteria were developed voluntarily as part of the Chesapeake Bay criteria effort (EPA 2003) and the Connecticut and New York criteria are only for dissolved oxygen in Long Island Sound.[1] Some states are in the process of developing site-specific estuarine / salt water numeric nutrient criteria (e.g., Great Bay, NH ; salt ponds and lagoons/harbors of Cape Cod, MEPS program,MA). Due to the complex nature of this issue, the timeframe for nutrient criteria development for most coastal states is considered a complex multi-year process (see Figure 1 from Battelle 2008).

1st Steps

Our proposed first steps outline the critical priority tasks required to begin the process of developing a scientifically justifiable numeric nutrient criteria for marine and estuarine waters. It is critical to begin with a valid database of all available values for the key measurements recommended for nutrient criteria development from RI marine waters. The choice of these indicator variables should be discussed with a technical advisory group made of local marine researchers and applied environmental managers experienced in this issue, but will likely include chl *a* , TN, DIN species, TP , clarity as secchi depth of K_d . In addition, other estuaries have uses eelgrass extent, depth of eelgrass deepwater edge, and more recently, nuisance macroalgae extent (NH) Some reasonable range of the nutrient concentration (TN) would be examined for significant correlation with these quantitative measures of negative nutrient impacts such as low D.O. concentration, change in extent of SAV, etc. That statistical relationship would be used to project the reasonable criteria

Another method to develop criteria would be to take a large dataset of such indicators and nutrient concentrations involves use of percentiles (e.g., 75th “good” level percentile based on a large number of waterbodies, with 25th percentile as a potential criteria. The statistical data population characteristics for these variables is key to the evaluation of any criteria chosen in order to ensure that the chosen value(s) is reasonably protective , but not exceeding the natural variability of an unimpacted system. Decisions will need to be made on adequate spatial and temporal resolution linked to the state’s assessment resolution (e.g., Bay-wide?, subembayment level? Below subembayment level?). The scale should match both the temporal variability of the measured indicator and the assessment level at which the criteria will be applied. Temporal decisions will need to include applicable seasonal & frequency requirements for minimal applicable data sets and adequate data to characterize the assessment scale used by the state (usually the subembayment levels e.g., Wickford Harbor, subareas of Greenwich Bay etc.). Note: several scientists have indicated (personal communications) that thist is *not* a particularly good method because it assumes there are adequate “pristine” or “healthy” areas within your dataset to quantify the characteristics of “good” areas. In addition, it does not address critical factors, like flushing, that often drive the system response to nutrient levels

In order to deal with physical characteristics that often drive local responses to nutrients, marine areas should be categorized by important physical factors that are known to affect response to excess nutrients. The Battelle report (2008) summarizes some of these and provides references to NOAA and USEPA methods of categorization. The minimal break out would be to group areas by level of FW (salinity)/ river influence and residence time/ flushing rate. Depth and width to area ratios have also been applied in this concept as a proxy for flushing rate. USEPA AED has a simplified model they have applied to many coves and harbors in Narragansett Bay that may be useful in this regard (Abdelrhman 2005). The application of a subembayment width to narrowest mouth ratio categorization could be completed by an intern with some knowledge of GIS tools.

The figure provided below highlights the steps needed to develop a numeric nutrient criteria, while the following text discusses the different approaches suggested by US EPA for numeric nutrient criteria development (both from Battelle 2008).

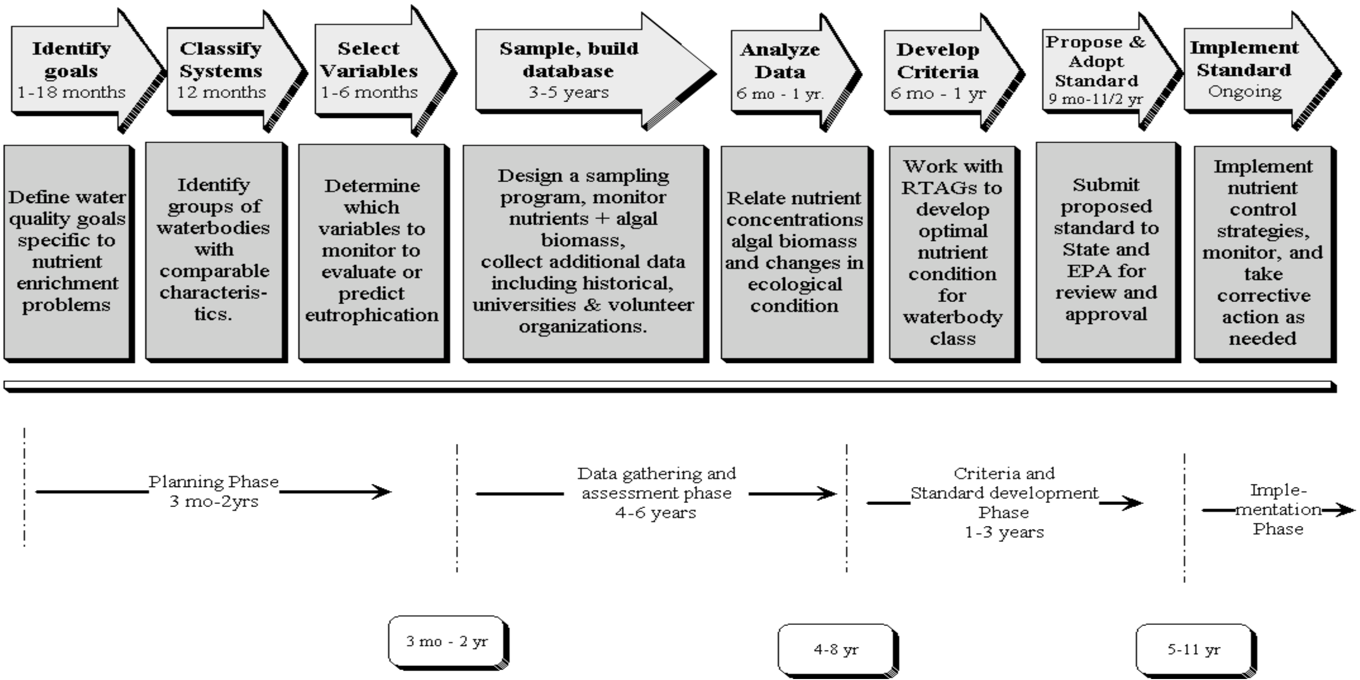


Fig. 1 Multi-year planning process for development of numerical nutrient criteria (Battelle, 2008[1]).
[1] Battelle . 2008. Conceptual Plan for Nutrient Criteria Development in Maine Coastal Waters. Report to USEPA Region I & Oceans & Coastal Protection Divisions & State of Maine .February 22, 2008. Work Assignment No. 4-53 Project No. G921353

Conceptual Model

A graphic conceptual model is a critical recommendation by the state of NH at the recent USEPA nutrient criteria workshop (6-2-11, Boston, MA) to help explain to both the public and the legislatures the basis for the nutrient criteria. We have presented several figs from the USEPA and other sources as recommended examples. The RIDEM should review these and decide which presents the clearest example.

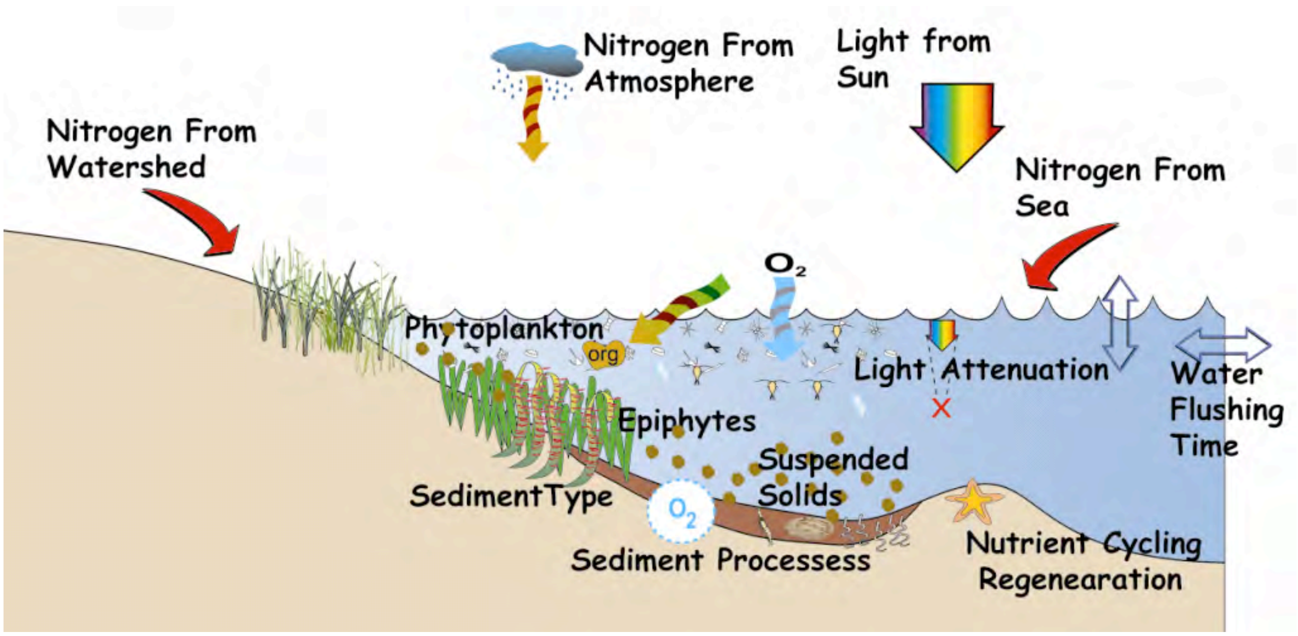


Fig.1. Marine System Response to nutrients (Nitrogen). Source : US EPA , J. Latimer

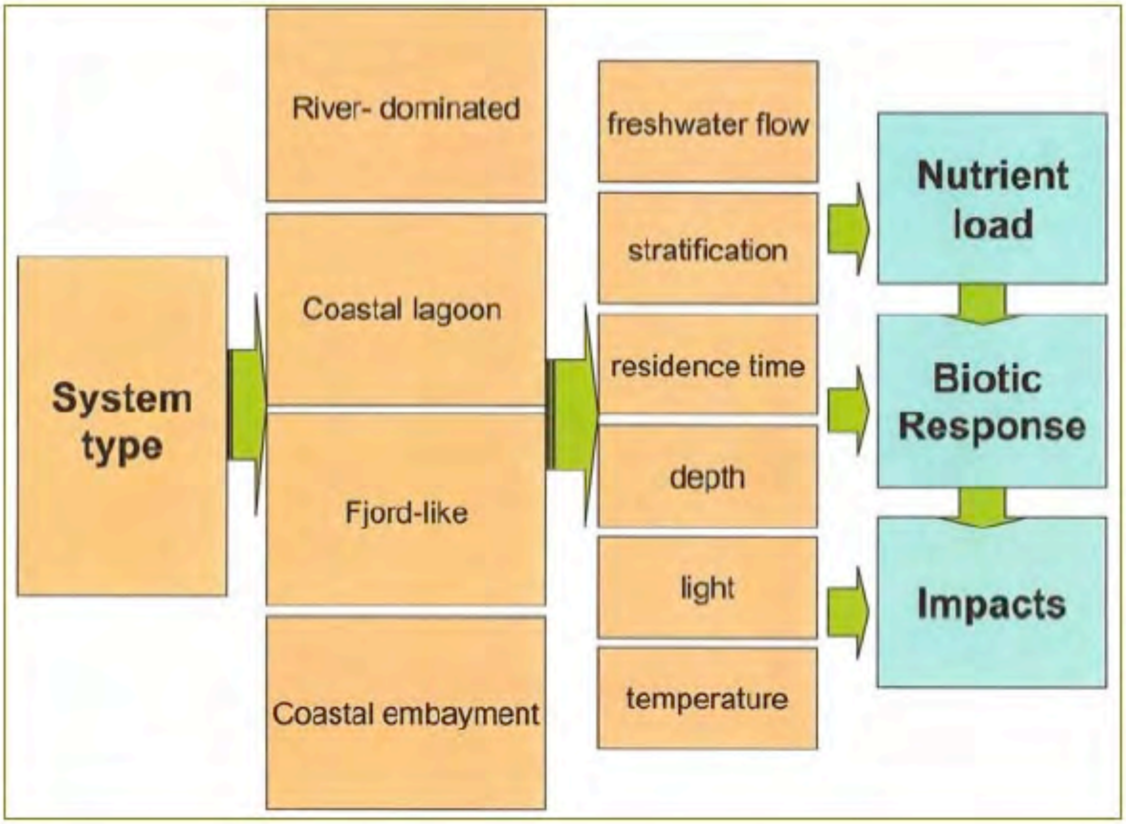


Fig.2. Marine Waters Categories. Source: USEPA Nutrient Criteria Report 2010

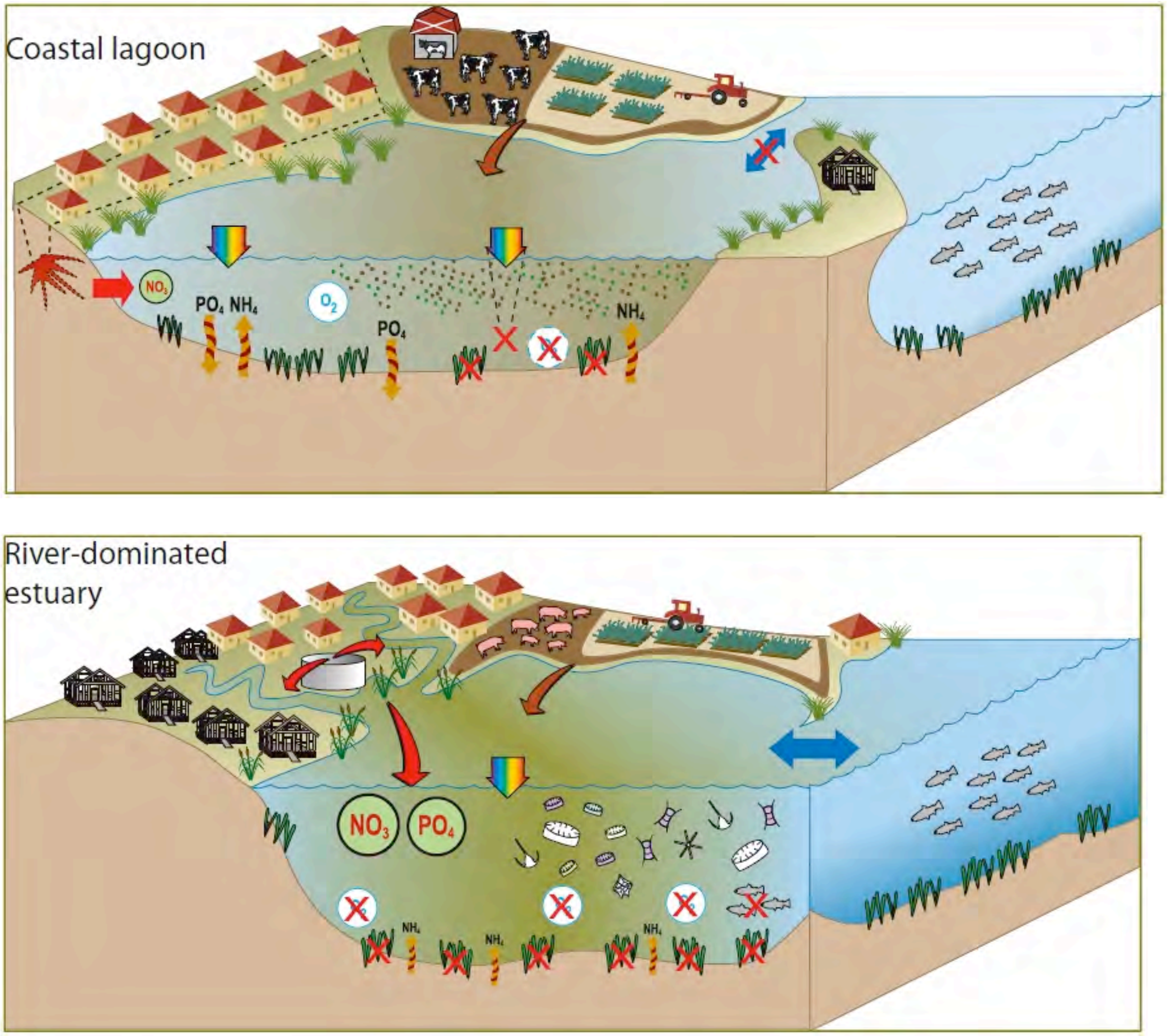
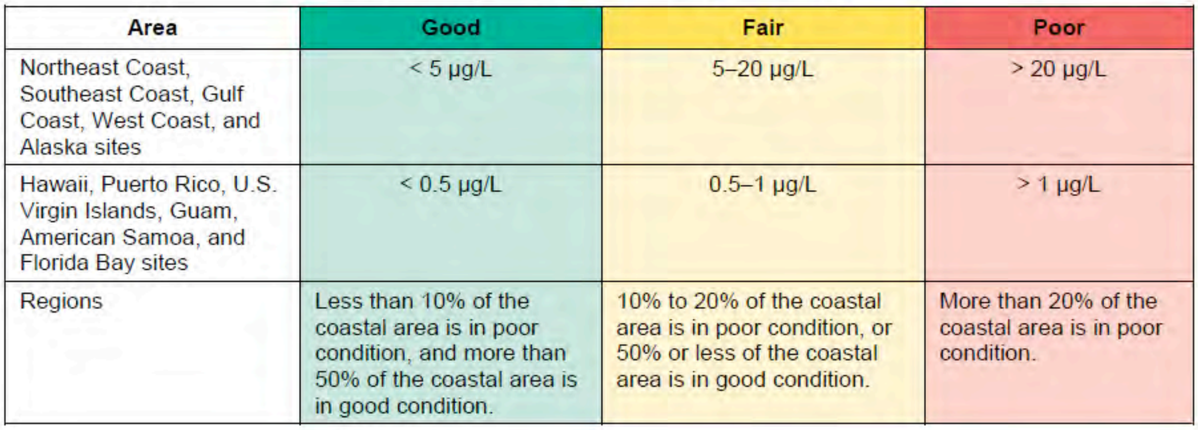


Fig.3. Marine System Response to nutrients (Nitrogen & Phosphorous).

Table 1-5. Cutpoints for Assessing Chlorophyll *a*



Area	Good	Fair	Poor
Northeast Coast, Southeast Coast, Gulf Coast, West Coast, and Alaska sites	< 5 µg/L	5–20 µg/L	> 20 µg/L
Hawaii, Puerto Rico, U.S. Virgin Islands, Guam, American Samoa, and Florida Bay sites	< 0.5 µg/L	0.5–1 µg/L	> 1 µg/L
Regions	Less than 10% of the coastal area is in poor condition, and more than 50% of the coastal area is in good condition.	10% to 20% of the coastal area is in poor condition, or 50% or less of the coastal area is in good condition.	More than 20% of the coastal area is in poor condition.

Source: of above table & fig : USEPA Nutrient Criteria Report 2010

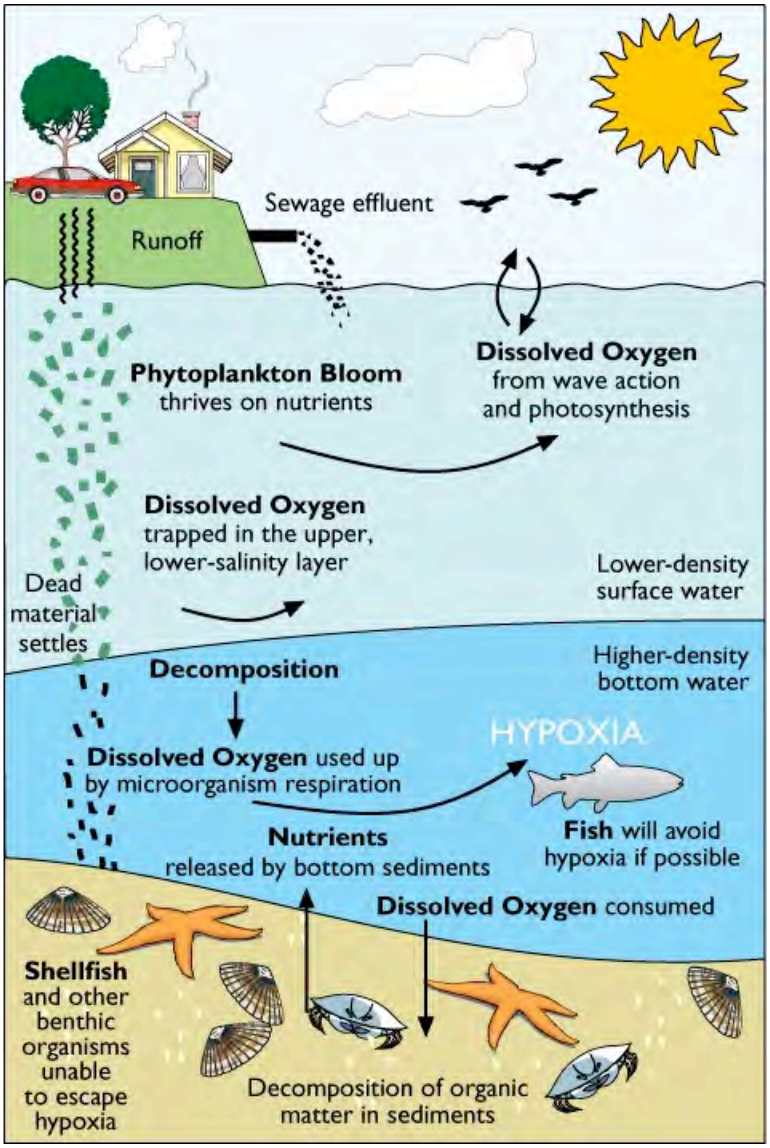


Fig.4. Eutrophication Impacts to Marine Waters. Source USEPA NCA 2010

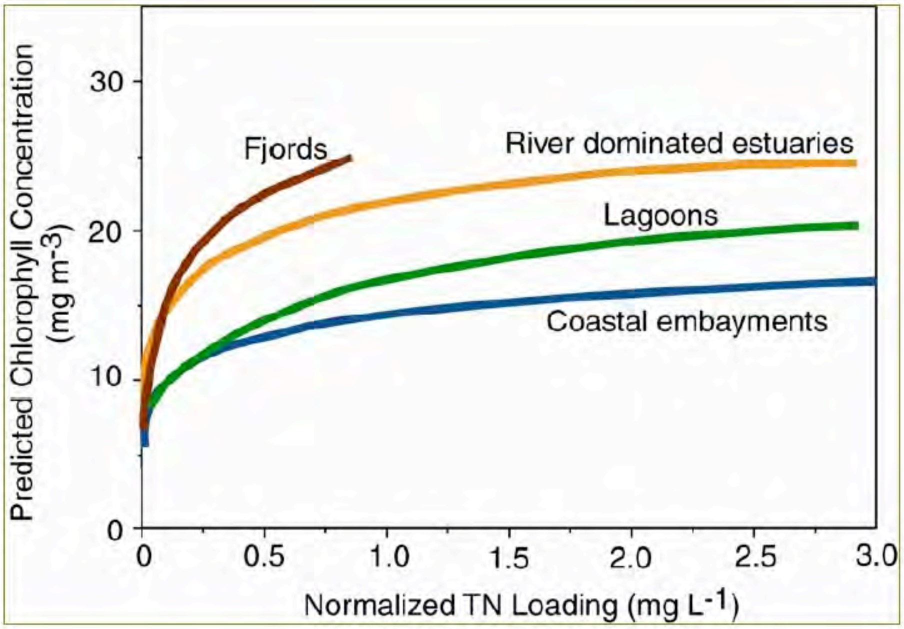
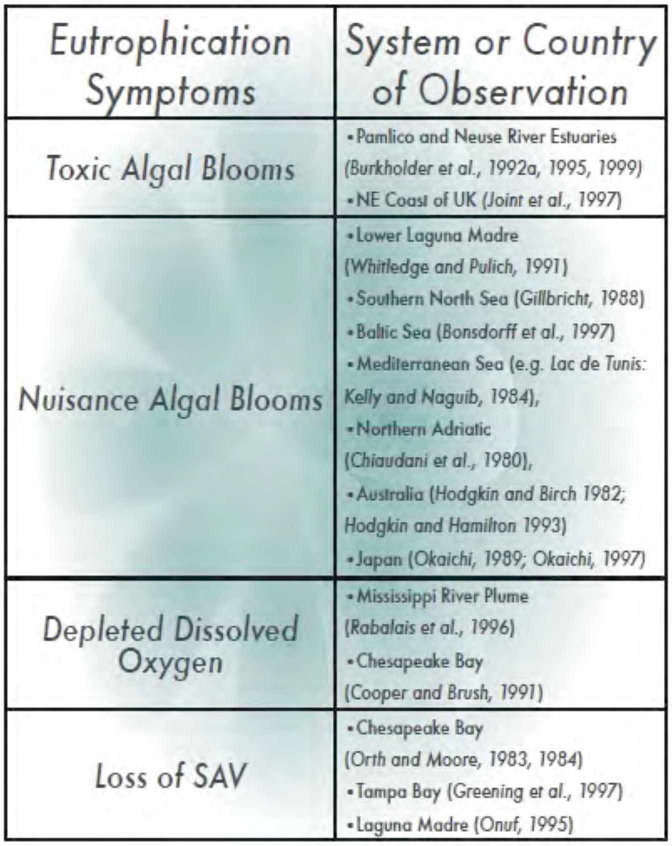


Fig. 5. Example relationship between TN load and response indicator for Chl *a*

NOTE: assumes adequate data available + the chl a values shown are too low for the concentration - should be in the 15-20 mg/L range by 0.5 - 1.0 for coastal embayments for NE



Eutrophication Symptoms	System or Country of Observation
Toxic Algal Blooms	•Pamlico and Neuse River Estuaries (Burkholder et al., 1992a, 1995, 1999) •NE Coast of UK (Joint et al., 1997)
Nuisance Algal Blooms	•Lower Laguna Madre (Whitledge and Pulich, 1991) •Southern North Sea (Gillbricht, 1988) •Baltic Sea (Bonsdorff et al., 1997) •Mediterranean Sea (e.g. Lac de Tunis: Kelly and Naguib, 1984), •Northern Adriatic (Chiaudani et al., 1980), •Australia (Hodgkin and Birch 1982; Hodgkin and Hamilton 1993) •Japan (Okaichi, 1989; Okaichi, 1997)
Depleted Dissolved Oxygen	•Mississippi River Plume (Rabalais et al., 1996) •Chesapeake Bay (Cooper and Brush, 1991)
Loss of SAV	•Chesapeake Bay (Orth and Moore, 1983, 1984) •Tampa Bay (Greening et al., 1997) •Laguna Madre (Onuf, 1995)

Source: NEAA Update Workshop Report 2002

Below- two graphics from NEEA Report 2007

Figure 2. A conceptualization of the relationship between overall eutrophic conditions, associated eutrophic symptoms, and influencing factors (nitrogen loads and susceptibility).

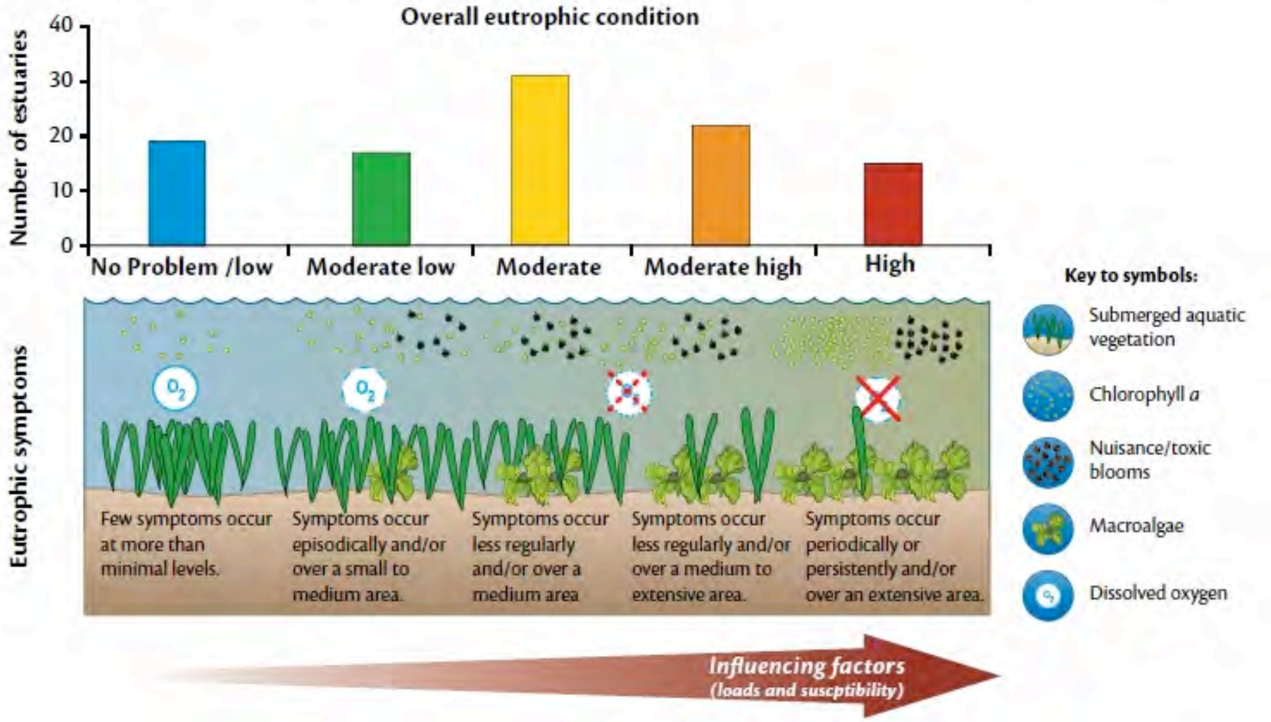
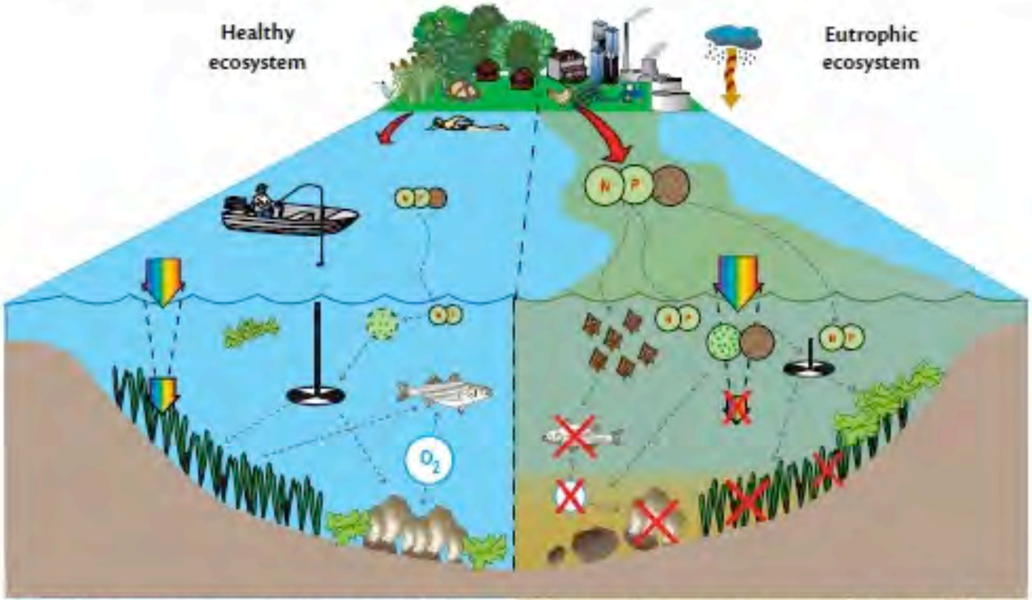


Figure 1.1. Conceptual diagram comparing a healthy system with no or low eutrophic condition to an unhealthy system exhibiting eutrophic symptoms.



In healthy ecosystems, nutrient inputs, specifically nitrogen and phosphorus , occur at a rate that stimulates a level of macroalgal and phytoplankton (chlorophyll *a*) growth in balance with grazer biota. A low level of chlorophyll *a* in the water column helps keep water clarity high , allowing light to penetrate deep enough to reach submerged aquatic vegetation . Low levels of phytoplankton and macroalgae result in dissolved oxygen levels most suitable for healthy fish and shellfish so that humans can enjoy the benefits that a coastal environment provides.

In a eutrophic ecosystem, increased sediment and nutrient loads from farming , urban development , water treatment plants , and industry , in combination with atmospheric nitrogen , help trigger both macroalgae and phytoplankton (chlorophyll *a*) blooms, exceeding the capacity of grazer control. These blooms can result in decreased water clarity , decreased light penetration , decreased dissolved oxygen , loss of submerged aquatic vegetation , nuisance/toxic algal blooms and the contamination or die off of fish and shellfish .

Review of Specific Approaches Recommended for Criteria Development

There are several approaches that can be taken to develop nutrient criteria based on US EPA Recommendations. The relative value and attributes of each are summarized below based on the Battelle report (2008).

1. **Reference Condition Approach –** This approach relies on the use of nutrient concentration data collected in "reference" areas that are determined to be relatively pristine / minimally impacted. Nutrient concentration thresholds are selected from the distribution of the collected nutrient data for these sites (e.g. 50th percentile for concentrations of N in reference site(s).

Advantages:

• High confidence that waters attaining the nutrient criteria are good quality with all uses protected (*assuming* good temporal nutrient concentration record and thorough WQ assessments in such waters).

- Relatively simple means to calculate threshold (statistical descriptive statistics of the ref. nutrient population) .
- Straightforward implementation

Disadvantages:

- Potential lack of adequate number of good reference "unimpacted" or "minimally impacted" sites where data can be collected or lack of historical reference quality data. For RI, possible reference sites: outer shore of Newport areas, but application problem with that area : less applicable to upper Bay conditions due open exchange with RI Sound vs upper Bay etc. This issue needs further discussion with local experts on adequacy of reference sites for RI.
- Assumes adequate nutrient records for an adequate number of reference sitesto decently characterize the population statistics as well as data for most RI waters, and *thorough and accurate* WQ assessment of such reference sites as fully meeting *all* variables potentially impacted by excess nutrients.
- In addition, analytical methods need to be discussed with experts. Battelle recommended that Maine go to measurement of TN rather than TKN to minimize potential impact of laboratory methods when comparing to TN data.
- Subjective selection of threshold value. Some "reference" waters ("unperturbed and high quality") may be above the nutrient threshold in certain periods. This could lead to erroneous assessments indicating potential violation of threshold criteria clearly needed for other waters. [Unlikely for RI. This problem should be more common when dealing with an area of high flushing rate with the open ocean and the major source of nutrients is incoming ocean water such as Maine areas]. The opposite risk is also true and has a higher risk of occurring in RI waters : chosen "reference" criteria value might actually be a "low bar" due to lack of adequate "unimpacted" reference sites and acceptance of nutrient values as ref values based on subpar sites due to inadequate "unimpacted" waterbody data available in the nutrient WQ database.
- Will require separate data sets for different salinity regimes since salinity is a key factor in response to nutrients in other estuaries.
- **Key drawback** : this method *does not account well for other factors that can affect response to nutrients* (e.g., local hydrodynamics/flushing rates etc.) unless one uses separate ref sites for each waterbody-type category , areas that flush their load into an

adjacent water might be assessed as meeting, while a nearby lesser-flushed area is impacted by the "uptide" area and is violating due to low flushing. It would require a "downstream impacts" clause at the least to ensure source areas are not assessed as meeting (due rapid flushing) and not in need of a TMDL.

- Will require decisions about representativeness of samples both spatially and temporally (area coverage and seasonal windows). State should get input from experts panel/advisers.

2. **Data Distribution Approach –** This approach utilizes *all* nutrient data collected from waters of *all* designated classes and conditions. As with the reference condition approach, thresholds are selected from the distribution of the data (usually a lower percentile because some large fraction of the data is assumed to be from waters with altered or impaired quality). Battelle's report indicated that the target concentrations are usually selected so there is reasonable expectation that most waters *will be able to attain criteria* (e.g., near the median). This method seems significantly biased towards the WQ of the lower end *of the sampled population*, even if this population includes areas of poor WQ conditions. Selection of threshold(s) is also supposed to include examination of expected attainable conditions based on implementation of best attainable treatment and best management practices for all discharging facilities. This approach sets a goal of bringing all waters to some reasonably attainable nutrient concentration target that should put most waters into compliance. However, it seems to require TMDLs based on BAT and BMPs and seems more like a UAA approach. The burden of implementation is on the sources (point and nonpoint) to meet technology standards, and there is a high likelihood of a false sense of meeting criteria as was experienced with the Chesapeake Bay Program goals in the 1990's, along with the likelihood this would require a *waterbody-specific approach* in the end. A number of experts have indicated this method to have serious drawbacks because of the above issues (personal communication from several prominent experts).

Advantages:

- Uses all available data (but ***expect significant additional data will be needed*** to get an adequate population estimate).
- Multiple thresholds may be selected representing different conditions based on classification (SA, SB, etc.)
- Relatively simple means to calculate threshold. Most waters could attain criteria and maintain designated uses.
- Simple to implement.

Disadvantages:

- **Requires very large dataset** that includes the range of conditions good to poor that are expected to occur under all conditions, natural and human-influenced.
- Will require separate data sets for different salinity regimes since salinity is a key factor in response to nutrients in other estuaries.
- Selection of nutrient concentration threshold value based solely on available database may not be ecologically defensible. Significant threat that "low bar" criteria values are not protective due to lack of adequate high quality reference sites and acceptance of subpar sites based solely on low percentile values of database ("best" of the bad WQ areas).

- Does not account well for other factors that can affect response to nutrients (e.g., local hydrodynamics/flushing rates etc.).

3. **Predictive Model Approach –** This approach selects criteria thresholds based on use of predictive models (e.g. regressions) that correlate adequate datasets of nutrient concentrations with other environmental effects such as low D.O. or chlorophyll *a,*loss of eelgrass, etc.

Advantages:

- Can attempt to account for other factors that can influence nutrient function in the environment.
- Reasonable expectation of a statistical correlation with other key WQ criteria like D.O. or phytoplankton biomass/ loss of water clarity.
- Multiple thresholds may be selected representing different conditions based on the State's current classification system (SA, SB)
- Commonly used for other criteria development.
- Might be feasible to base on simple BZI (biomass, photic depth, and incident irradiance), which are considered an acceptable, general method for predicting daily net phytoplankton production (NPP) in well-mixed, nutrient replete estuaries. Corrections for shallow areas of the Bay area available (Brush & Nixon 2009, Brush & Brawley 2008). This type of model links nutrient concentration to productivity, although loading actually seems better here.

Disadvantages:

- Requires **development of one or more models** that correlate nutrient levels to various environmental effects. Models do need to be calibrated for both physical aspects like flushing rate of a specific area (e.g., Greenwich Bay, Wickford Harbor etc.) and salinity regime, as well as biological responses to both nutrient concentrations and physical changes like seasonal temperature changes .
- Problem of potential limited availability of data for model construction (nutrients, other independent variables, and dependent response variables like D.O. , chl *a*, or SAV extent etc.) across a range of WQ conditions good to poor that are expected to occur.
- *Not* simple to implement: dependent on complexity of model used. *May* require development of complex ecological-physical model of estuarine waters if simple model does not predict response (phytoplankton and macro algae productivity) well, and requires staff technically capable of running such models and choosing correct process factors like metabolic rates, primary productivity rates etc. based on published or recommended values from local experts.
- Difficult to control amount of error (variance) in the model(s), with a high likelihood of decent general trend response modeling (i.e., model indicates decreasing load/concentration to X will lead to general overall decrease in phytoplankton biomass/ chl a level Y) but likely incapable of predicting accurate response of system to specific nutrient levels due local hydrodynamics / flushing etc. at a subembayment resolution (e.g., Greenwich Bay) .

4. **Effects-based Approach –** This approach establishes **nutrient criteria as "screening" values** : they are not enforced until some other impaired "response" is demonstrated. Appropriate

response criteria need to be established (e.g. oxygen, chlorophyll, cell counts, marine life response pattern, etc.). The screening thresholds for nutrient concentrations are developed by one of the above approaches (e.g., adequate nutrient population stats with “unimpaired” indicator response such as healthy eelgrass habitat as being done for MA by B. Howes UMA / SMAST).

Advantages:

- High confidence that designated uses *are* attained (direct measurement of designated use). Attainment is based on *response criteria* (actual detection of positive/negative effects in the ecosystem such as condition of eelgrass, frequency of low oxygen events, etc.).
- Takes into account other variables that affect nutrient function.
- Multiple thresholds may be selected representing different conditions based on classification (SA, SB)
- Opportunity for site-specific criteria.

Disadvantages:

- Lack of adequate data on correlation of nutrients to suitable response criteria. Preferably use indicator already existing in statute or rule or well-established relationship between indicator (e.g., eelgrass health) and nutrient concentration or loading rate.
- Need to develop scientifically-defensible relationship of nutrients to chosen response criteria. (e.g., need to choose “detrimental” chl a levels, including temporal sampling / seasonal window minimal requirements)
- Not simple application : Adequate measurements of several response criteria are required to assess water quality condition and designated uses that could be affected by nutrients.
- Two data types are required to make an assessment (nutrient *and* response criteria … but measurements *could* be captured *together* with adequate field monitoring).
- Increased monitoring requirements and cost.
- Implementation is complex. Results not always clear since nutrients could be low when response criteria are measured as violating (e.g., summer periods when biology absorbs all available nutrients and nutrient concentrations are low in the water column but low DO or poor eelgrass health indications, or conversely, the measured nutrients are high in winter but there is no clear violation of response criteria like low DO due to seasonality of biological responses and lag times between loading and clear negative responses. Different salinity regimes may respond differently and require different criteria thresholds since salinity is a key factor in response to nutrients in other estuaries.
- There is a clear need for minimum temporal and spatial sampling requirements for both nutrients and the response indicators.

Issues for RI Criteria Development :
Data Sources and Sampling Locations

There have been a number of nutrient surveys completed over the years by URI researchers in Narragansett Bay proper, but many of the subembayments (e.g., Wickford Harbor, Allens Harbor, Bristol Harbor) do not have any or minimal data. In addition, only one published survey included measurement of Total Nitrogen (TN) (see section x p.) Two stations: one just north of Fox Island / south of Quonset Point and one at the URI Bay Campus dock provide a long term weekly sample database of dissolved nutrient concentrations as well as response variables such as chlorophyll a, but not TN. Sources of data for response variables of nutrient-sensitive habitat (SAV) are limited to 2 aerial overflights (1996 and 2006, Bradley et al 2007) and a newly scheduled flight (2011). The NBNERR has been developing a video rapid assessment procedures method that could provide one measure of response from this indicator on a more consistent basis. There is some verbal documentation of historical beds (Dougherty 1986) which could provide nonquantitative historical context to the shallow water habitats of Narragansett Bay waters that were once high quality / moderately low nutrient areas. New habitat models available from NOAA can estimate area of potential eelgrass habitat based on depth contours and wind exposure and may be able to provide maps for targeting areas for restoration potential;.

A database would need to be developed that can encompass all pertinent data, with some criteria for minimum acceptable frequency (i.e., a single measurement may not be useful unless other data is available). There is a critical data gap which requires a significant increase in monitoring data on TN and TP concentrations for marine waters of RI. Data should also be broken out on a seasonal basis rather than an annual mean in order to separate out seasonal responses (e.g., June-Sept.) **Federal assistance for such an effort should be pursued**. Both Nitrogen and Phosphorous should be included, as well as Silica where feasible in order to examine ratio issues which many researchers believe have significant influence on local responses. Nutrient form (TN, NH4, NO3 etc) should be broken out where data is available. **All marine waters need to be categorized by mean or modal depth and salinity range. This could be done through GIS analysis of the bathymetric and salinity data coverage. Some measure of flushing rate should be included for specific subembayment area level , perhaps using information developed by the USEPA AED lab in Narragansett for many areas of Narragansett Bay.** .

In most coastal states dealing with criteria development, the marine systems are being categorized based upon salinity regimes. The importance of freshwater inputs need to be taken into account for RI, but the small amount of oligohaline (lowest salinity regime) waters due to damming of most river mouths probably will minimize the salinity issue into perhaps 2 salinity regimes (mesohaline and polyhaline waters). Data will need to be organized to examine these waters separately for statistical population characterization.

A potential classification scheme to consider for RI might use stressor-response relationships to group waterbodies by how they respond to nitrogen loading as the stressor (Dettmann and Kurtz ,2006). They use two separate responses – extent of eelgrass habitat and phytoplankton biomass response (as chlorophyll concentration) in 10 estuaries and compare chl a to TN concentration. They concluded that there is a consistent phytoplankton response related to ambient TN concentrations, but that other factors (e.g., water clarity from TSS) may reduce the response, leading to lower production for same level of concentration. Areas with strong river influences had similar relationships, but the response was more complex due to a wide range in TSS levels.

These issues should be vetted with the local researchers who are highly knowledgeable in this topic. Issues such as depth averaging etc will also need to be discussed since some data is surface only and some is surf + bottom averages. Summary statistics and graphical presentations of the surface, summer data from all stations should be developed, including overall mean, minimum and maximum values, seasonal means, standard deviation, and percentiles (10th, 25th, 75th, and 90th) for each parameter of interest (TN, TP, chlorophyll a, DO, and the dissolved inorganic nitrogen concentrations). Box and whisker plots can summarize these results graphically (see below). Frequency plots should be produced to describe the overall data distribution, and GIS maps produced to depict the spatial distribution of these parameters.

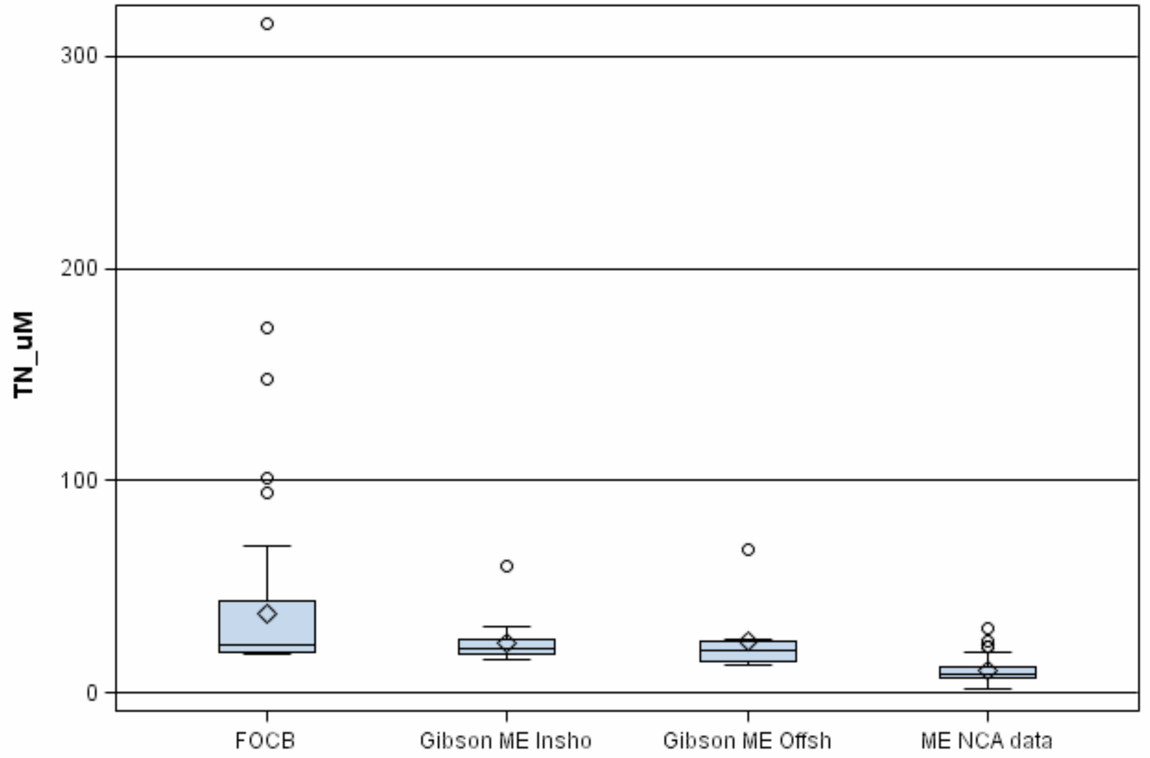


Figure 3. Example Box and whisker plots of summer, surface TN concentrations (µM) for the state of Maine. The various symbols represent values as follows: the box = 25th to 75th quartile range, the line in the box = median, the diamond = mean, open circles are outliers, and the whiskers extend to the furthest value below and above the quartiles that is within 1.5 times the interquartile range (IQR). Figure from [1].

Needed steps for Nutrient Criteria Development for RI

- **Identify and acquire all available raw nutrient related data from all sources** – federal, state or local monitoring efforts, scientific research efforts, etc. This should include data that could be used to classify waterbodies (salinity, etc). Annual and seasonal averages (available from the lit.) are not useful for gathering the full statistical population characterization. This step will should include meeting with local experts who have the data and developing an agreement on how the data will be treated in order to protect their intellectual rights for publication of their unpublished data. TN data should be especially pursued as being the most useful for nutrient concentrations.
- **Because of data gaps for TN, a critical parameter for numerical criteria development, a TN monitoring program for all RI marine waters should be initiated ASAP, with a design based on consideration of likely gradients within each waterbody measured (vs randomizd sampling).** (Other data should also be sampled simultaneously through this effort,including secchi, light irradiance, DIN and DIP , TP , as well as extended YSI D.O. and chl*a* deployments in summer and digital photographs or other measurements of the extent of eelgrass and/or macroalgae.
- **The methodology for TN should be using the persulfate method vs the Kjeldahl method, including at the RIDEM NBNERR (see App B for comments on this need in the section on available TN data)**
- **From a baywide monitoring perspective, a key missing piece is a clear understanding of surface vs bottom concentrations.** No long-term study has looked at bottom water concentrations on a regular basis since the late 70s, and even then, samples were only monthly and for only a year at a time.
- **Another key issue for comparing nutrient lab results between data sources is the lack of a regular series of intercalibrations between labs.** Each lab operates with its own sets of standards, which are handmade, and rarely checked against anything with a truly known concentration, and there is no standardized methodology between labs for collecting, processing and running samples with respect to preservation, holding time, handling requirements, etc. Many university labs , even within the same institution, use different methodologies and chemical reactions depending on their instruments or the type of samples run (e.g.,some labs use phenol/nitroferricyanide for ammonia, others use a similar indophenol blue reaction, but with an EDTA buffer (official EPA method)). **At a minimum, any labs receiving state or federal funds linked to the state's mgt needs should be required to complete an annual intercalibration to ensure that inter-lab variations don't impact results. (personal communication from J Krumholtz, URI 2011)**
- **There is a critical need to develop a comprehensive central database for this data with established data management procedures.** Because of the work already completed on this issue at URI GSO, it is likely this **could be a collaboration between URI and RIDEM**
- **Examine the possibility of federal (EPA AED) help using a stressor-response model since the authors are involved in nutrient criteria development and are in-state (e.g. Dettmann)**
- **Present pilot results to develop criteria** using the various above criteria approaches, including pros and cons, **to an "expert panel" advisor group and get feedback.**
- **Pursue federal funding mechanisms for these nutrient criteria development activities – from field work to data mining to public outreach.**

Workplan and Timelines

RIDEM and NBEP expect a first task toward pursuing any selected approach will be an assessment of data availability. While a large amount of nutrient data has been generated for Narragansett Bay, most data are for DIN, a less useful dataset vs Total N. Based on the extent to which sufficient data is available at the appropriate temporal and spatial scales, a sampling program will need to be designed to address data gaps and ensure collection of key data over the next several years. This period of time is important given the expected changes to nutrient loadings due to RI WWTF upgrades that will not be completed until 2013-2014. Based on the phased approach to infrastructure improvements agreed upon in RI, it is expected that an assessment of water quality conditions that result from the upgrades will be made prior to deciding on further reductions in nutrient loadings from the RI WWTFs.

RIDEM further expects that estuarine waters will have to be categorized for criteria development. The criteria developed for Narragansett Bay may not be applicable to certain restricted subembayments or the coastal ponds along the State's southern shore. Decisions on how to segregate coastal waters for criteria development will need to be made during selection of the most appropriate approach.
We recommend that the RI DEM continue to work with partners, including the NBEP and URI Coastal Institute and the US EPA AED to explore the best methodology for development of a numerical nutrient criteria for nitrogen for saltwaters/ estuarine waters of the state. The exact timeline is unclear for RI, because significant data gaps, especially for total nitrogen (TN) need to be filled before one can examine relationships between TN load and system responses. The planning and data gathering phases for concentration-based criteria can probably be collapsed together and completed in 3-5 years if resources are available to complete the needed tasks. Effects-based criteria should be considered for RI based on expert comments and recommendations. One possibility is to base nutrient criteria concentrations on eelgrass survivability linked to water clarity / chl a concentrations driven by those nutrient levels, as is being pursued in NH and MA. CT has also examined this concept and has produced a very good white paper on the issue of eelgrass threshold light needs etc. (Vaudrey 2008). Eelgrass areas in RI have been well-studied by RI experts, and significant literature has been developed on minimum light requirements (linked to chl a). Eelgrass habitat use could be considered the highest quality (SA) criteria, with some % of that threshold for areas expected no capable of ever sustaining that use due to historical urban development etc (e.g., Providence River). Such a percentile of eelgrass threshold should be matched to areas that are not experiencing oxygen levels below the state criteria.

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Appendix A Examination of presently available data for Narragansett Bay in regards to numeric nitrogen criteria development

This appendix discusses available nutrient data for various Rhode Island waters in detail. Sampling station maps for the various sources are provided at the end of this appendix.

Available TN data

The only published bay-wide data set for TN that we are aware of was reported by Oviatt (2008, in Science for Ecosystem-Based Management: Narragansett Bay in the 21st Century) based on a 1997-98 survey: She has more recent TN data for 2006-09 which she has not yet released (unpublished). The TN gradient for these data are not significantly different from the ’97-’98 data (Krumholtz, personal communication and NOAA CHRP presentations 2011), ranging from 70 uM/l at Fields Point to 40 uM/l at Bullocks Reach buoy, dropping to the 30 uM/l range at the mouth of Greenwich Bay, dropping to 20-15 uM/l further downbay..

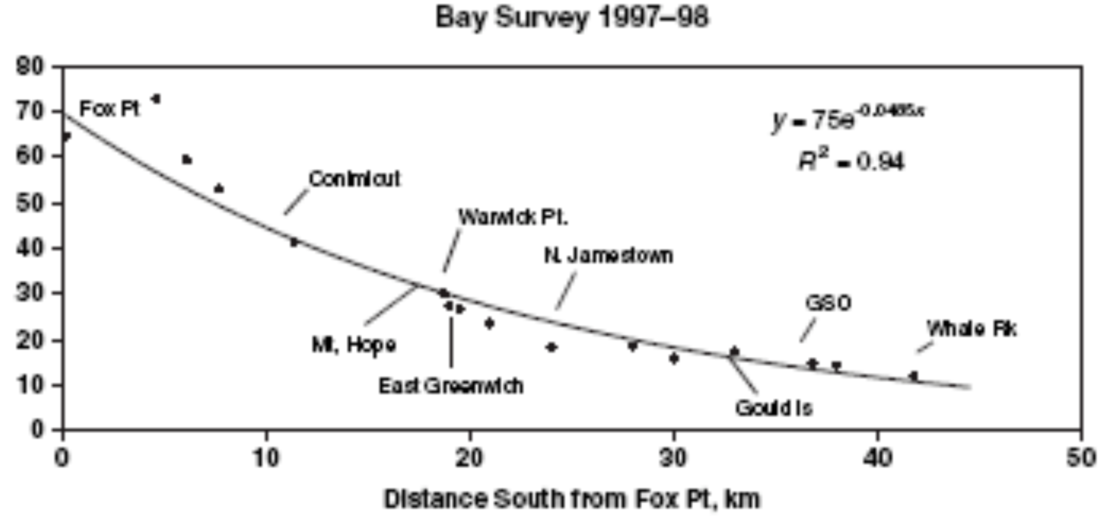


Fig. 18.2 Annual mean TN concentrations (Total Nitrogen, μM) plotted on the north–south axis of Narragansett Bay and regressed as a function of distance from Fox Pt at the head of the Providence River to Whale Rock at the mouth of the West Passage (F = 436, P = <0.001)

Although Oviatt (2008) presented a N to S profile, it seems better to present data as a S to N profile as above. Note that the regression Oviatt (2008) determined reaches approximately 70 micromolar (x 14 ugN/l= 980 ug/l = 0.98 mg/l) at Fox Point, close to averages in recent years at the India Point station (approximately same locations). Oviatt’s lower bay averages of 10-20 micromolar (140-280 ug/l=0.14-0.28mg/l) at Gould Island and GSO are similar to the more recent NuShuttle/MERL station 1 average which are also approximately the same locations.

NBC has been collecting total dissolved N data (TDN, but unfortunately not the particulate N needed to determine TN) above Conimicut Point since about 2006. Jason Krumholz of URI GSO is updating the bay-wide survey database for TN as part of his dissertation (expected completion Fall 2011). These data should be gathered, correlated with nutrient-related parameters such as chl-a, DO, and water clarity, and analyzed for usefulness is determining nutrient criteria. NBC, GSO and NBNERR should be encouraged to measure TN and all of its constituents.

WWTFs have been the source of the majority of TN delivered to Narragansett Bay. The RI Governor's Commission recommended and the RI General Assembly approved legislation calling for 50% reduction (from a 1995-96 baseline) in N loading from WWTFs by the end of 2008 (with provision to adjust that date consistent with permit modifications) (see RI DEM, 2005, Plan for Managing Nutrient Loadings to Rhode Island Waters, www.dem.ri.gov/pubs/nutrient.pdf).

Discharge reports from WWTFs show marked reductions in recent years (although 2010 will likely be a setback due to flood damage to plants). DEM estimates that operating reductions in WWTF effluent loads total about 35% of the 1995-96 baseline. All RI WWTFs with discharges to nutrient impaired areas have agreed upon schedules for upgrades to meet these requirements. The next major reduction (NBC's Field's Point plant) is under construction but not scheduled for completion until 2014. Unfortunately, the next largest treatment plant affecting Narragansett Bay (the UBWPAD plant near Worcester MA) is still resisting the permit issued for it. However, recent data indicates they have been achieving levels of TN well below 10 mg/L

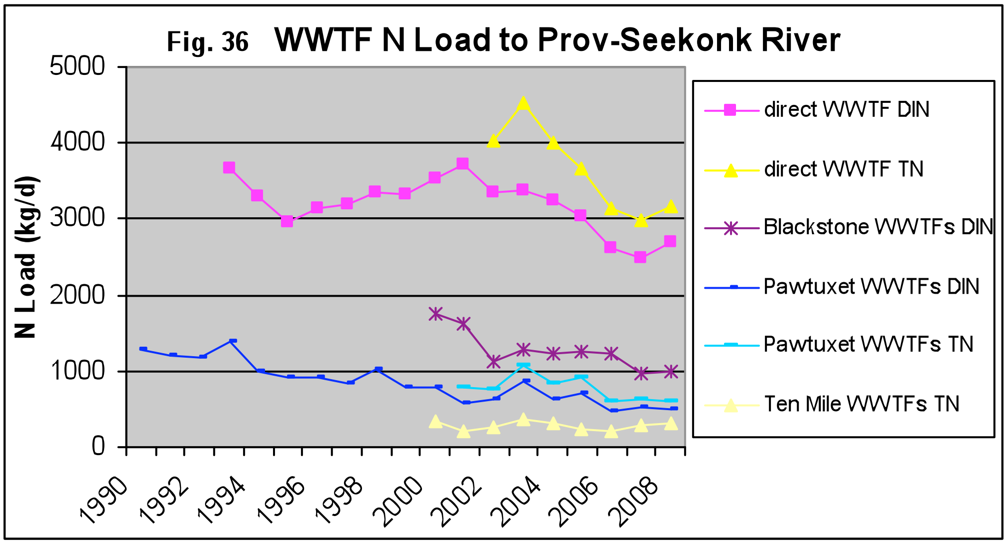


Fig. 36 WWTF N Load to Prov-Seekonk River

Dissolved Inorganic Nitrogen (DIN) and Issues of Seasonal Patterns and Variability

Leanna Heffner produced a report for the NB NERR(2009), "Nutrients in Mid-Narragansett Bay: A Spatial Comparison of Recent and Historical Data", which focused on DIN components (only DIN is measured at the NERR site, GSO and station 2). DIN has a strong seasonal variation and misses the assimilated N in phytoplankton as well as other organic forms, causing major loss of any correlation between N concentration and responses such as chl *a* levels (see Fig below). TN should be a complete measure of N concentrations and show markedly less seasonal variation. Almost all nutrient criteria proposed by other states focus on TN. As noted above, the only published bay-wide data set for TN that we have been able to locate is by Oviatt based on a 1997-98 survey:

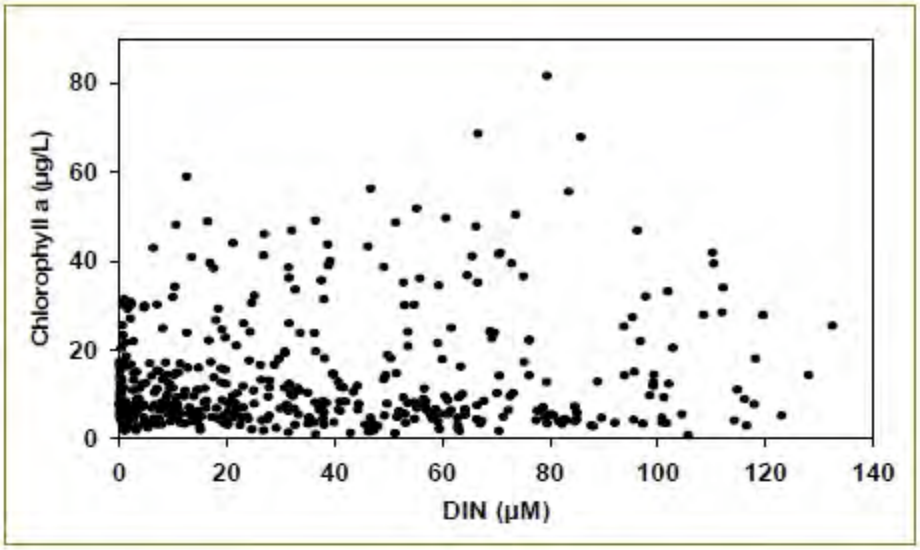


Figure 4.9. Scatter diagrams for the mid and lower Delaware Bay, for data from 1980 to 2003, illustrate that simple, synoptic correlations between nutrients and biomass often do not display characteristics of a dose-response relationship. From J. Sharp, unpublished.

Fig from Madden et al. (2010)

The DIN (ammonia + nitrate + nitrite) component of TN has been more studied. Heffner (2009) reported that the longest data series in the bay (which she called "Station 2" but which is called "Fox Island" here to avoid confusion with the NuShuttle stations) showed no change in ammonia or nitrate (nitrite concentrations are much smaller contributions to DIN) since the early 1970s.

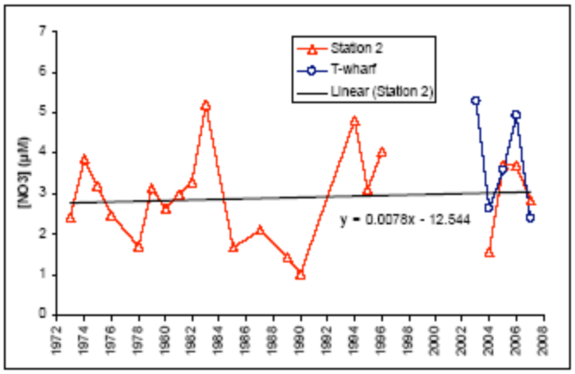


Figure 12. Mean annual nitrate concentrations from Station 2 historical data (1973-1996) and recent Station 2 and T-wharf data (2003/4-2007). A linear regression line for the Station 2 values shows no changes over time (p = 0.7385).

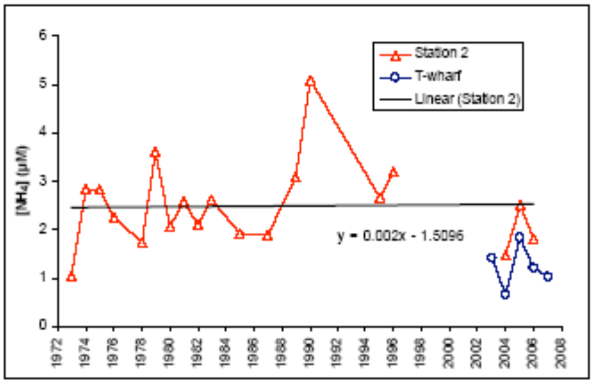


Figure 9. Mean annual ammonium concentrations from Station 2 historical data (1973-1996) and recent Station 2 and T-wharf data (2003/4-2007). A linear regression line for the Station 2 values shows no changes over time (p = 0.6818).

DIN shows strong seasonality as well as a strong N-S gradient in the bay.

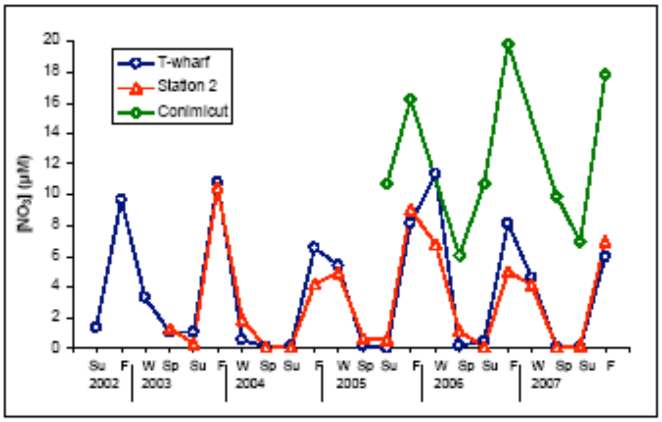
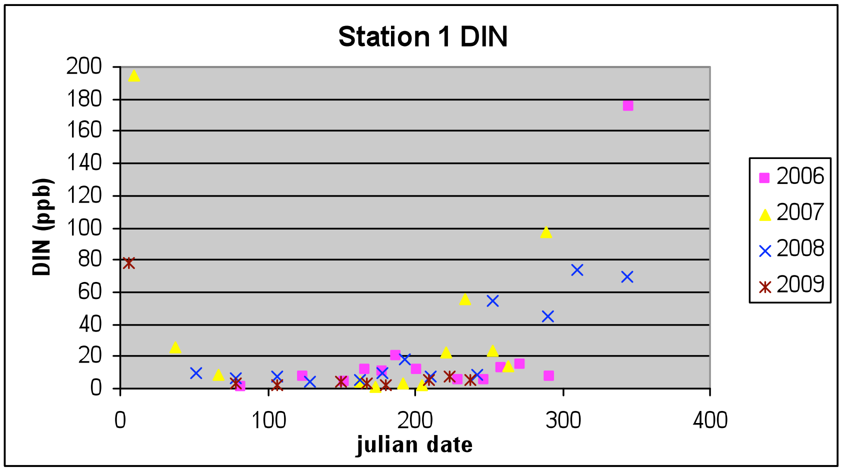


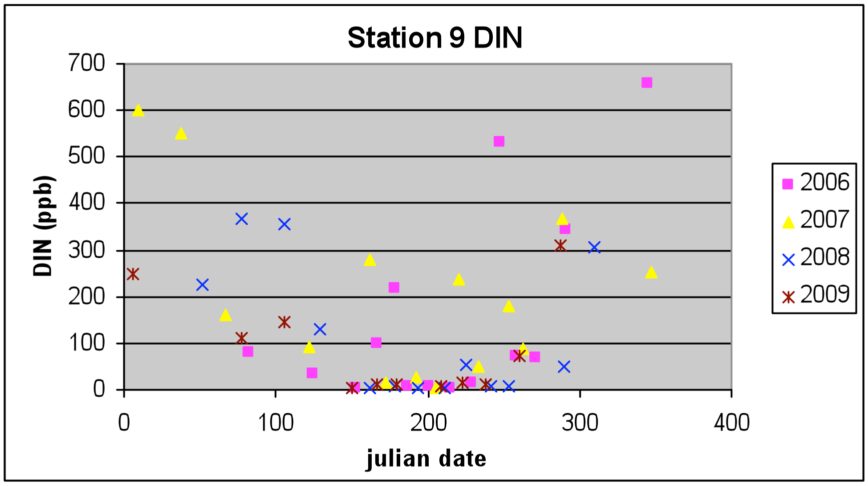
Figure 11. Mean seasonal nitrate concentrations for surface and bottom waters at T-wharf and Station 2 (W = winter, Sp = spring, Su = summer, F = fall). Coefficients of variation generally ranged from 0-210% at T-wharf, 2-200% at Station 2 and 25-95% at Conimicut Point.

Seasonal variations render DIN concentrations less useful as a water quality indicator. Nevertheless EPA's National Coastal Condition Assessment has used DIN for that purpose. For Northeast estuaries, they defined "good" levels of DIN as <0.1 mg/l, "fair as 0.1-0.5 mg/l, and "poor" as >0.5 mg/l. The 2008 assessment reported that approximately 40% of the area of the bay had "fair" or "poor" DIN levels based on 56 samples from stratified random sites during 2000 and 2001 generally during summer months. In the southern portion of the bay, DIN concentrations are drawn down almost to limits of detection during much of the year.



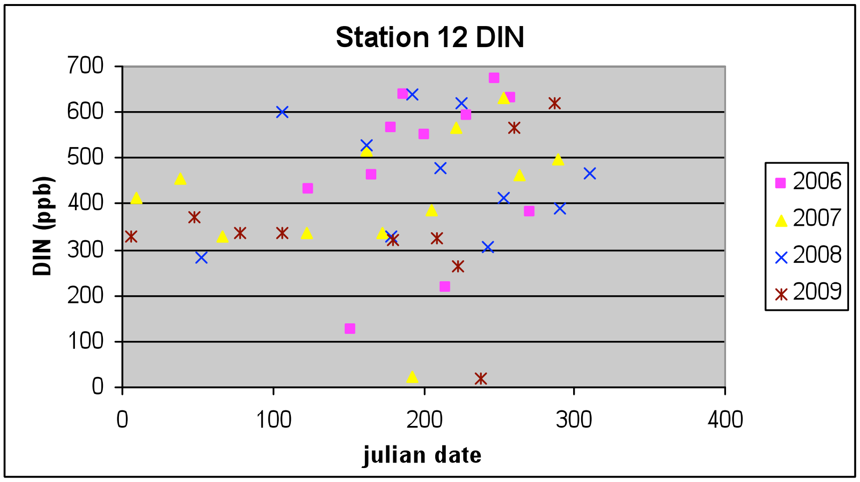
(NuShuttle/MERL station 1 is just south of the "Fox Island" station.)

Further north in the bay, the seasonal pattern is different with strong drawdown for a shorter period of time and higher concentrations at other times.

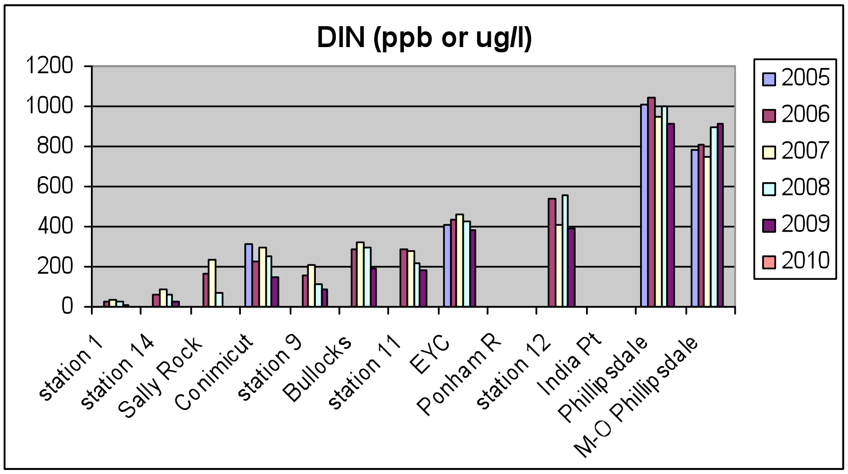


(NuShuttle/MERL station 9 is at Conimicut Point)

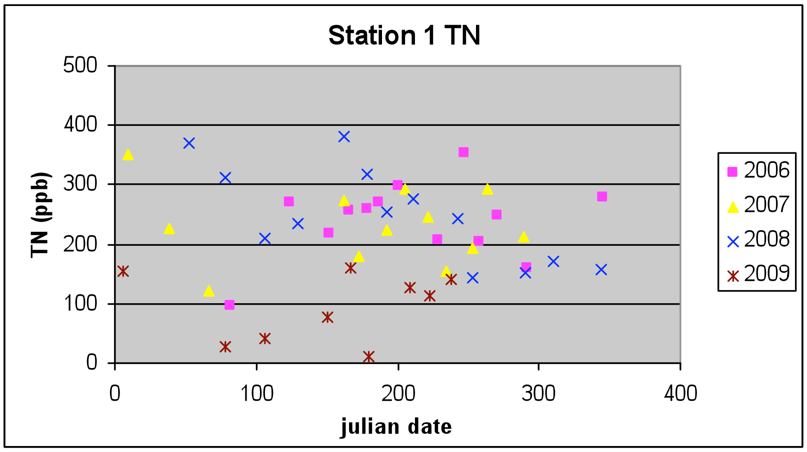
Seasonal patterns are effectively obliterated further north as nutrient loads appear to overwhelm assimilative capacity (station 12, below, is at Fields Point).



Annual averages of DIN can be significantly affected by timing of sampling, most noticeably in the southern part of the bay where the seasonal variation is greatest. Winter sampling is more difficult and, as a result, often less frequent. Even at Phillipsdale in the Seekonk (the most northern site sampled), May-October average values differ from the annual average.

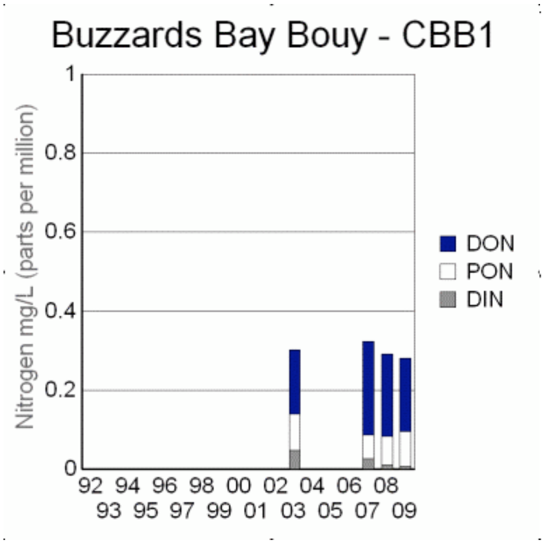


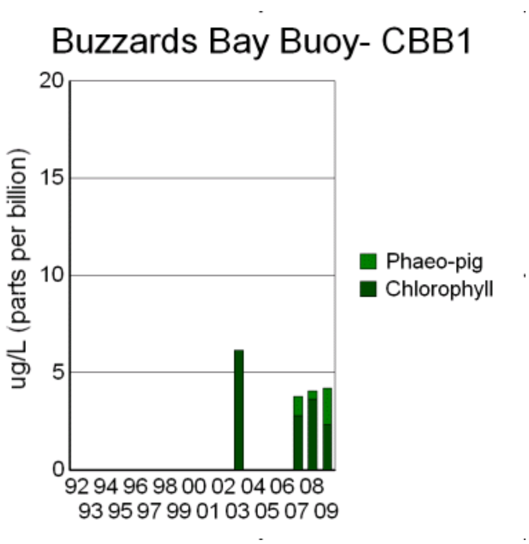
Seasonal variation of TN is much less distinct than DIN variations even in the lower bay.

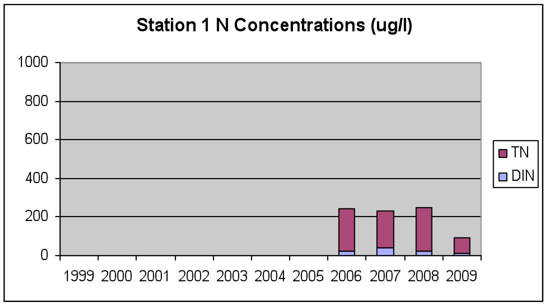


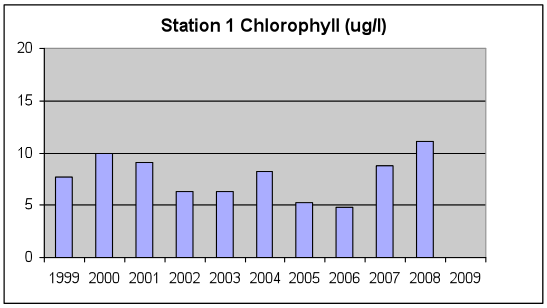
Total Nitrogen (TN) Composition and Behavior in Other Estuaries

Buzzards Bay estuaries have a much more comprehensive set of data on N concentrations and comparisons with Narragansett Bay locations might be useful. Measurements at a central bay buoy in Buzzards Bay are quite similar to those from NuShuttle/MERL station 1 in Narragansett Bay.









Particulate organic matter (PON) constituted roughly one-third of TN. DiMilla (2006), Oczkowski (2008) and Trowbridge (2009) have all suggested that phytoplankton might be only 1% of TN but both Howarth and Boynton, in reviews of Trowbridge's report, raised concerns about those calculations (see Appendix C). The ratio of PON**:** chl (g/g) at the central buoy in Buzzards Bay is approximately 20 which is consistent with the literature. Buzzards Bay data from many sites show that PON correlates closely with chl levels (on an annual average basis) and the PON **:** chl ratio ranges from 20 to 50. Although non-living detritus constitutes some portion of PON, its concentration is indicative of phytoplankton uptake (as indicated by chl) but chlorophyll calculations often underestimate its magnitude..

Phytoplankton growth models such as those compared by Smith and Yamanaka (2007) also show that chl/N may start very low and take two weeks or longer to stabilize at their maximum value (3 g N/g chl or about 0.25 mole N/gm chl).

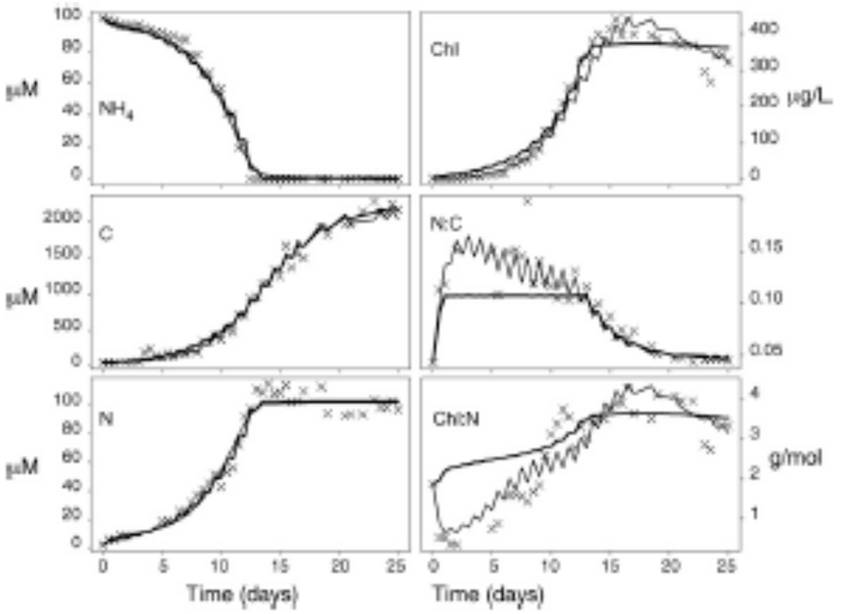
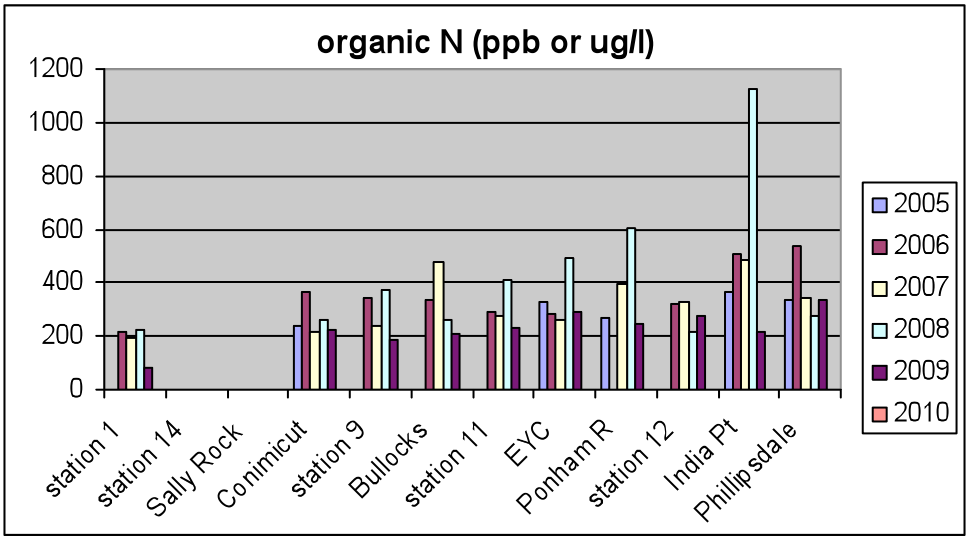


Fig. 1 – Best-fit simulations (thick lines, C model; thin lines, P model) and observations (X's) for NH$_4$, phytoplankton C (C), phytoplankton N (N) and chlorophyll (Chl), molar N:C ratio and Chl:N ratio.

From the viewpoint of tracking N, measuring only chl*a* is a poor indicator of actual uptake, particularly after nutrient concentrations are rapidly drawn down. In addition, shallow-water macroalgae, which has substantial capability to rapidly surge-uptake N is not accounted for by water chlorophyll sampling. This may explain the delay of chl concentrations after seasonal drops in N concentration observed at the GSO dock.

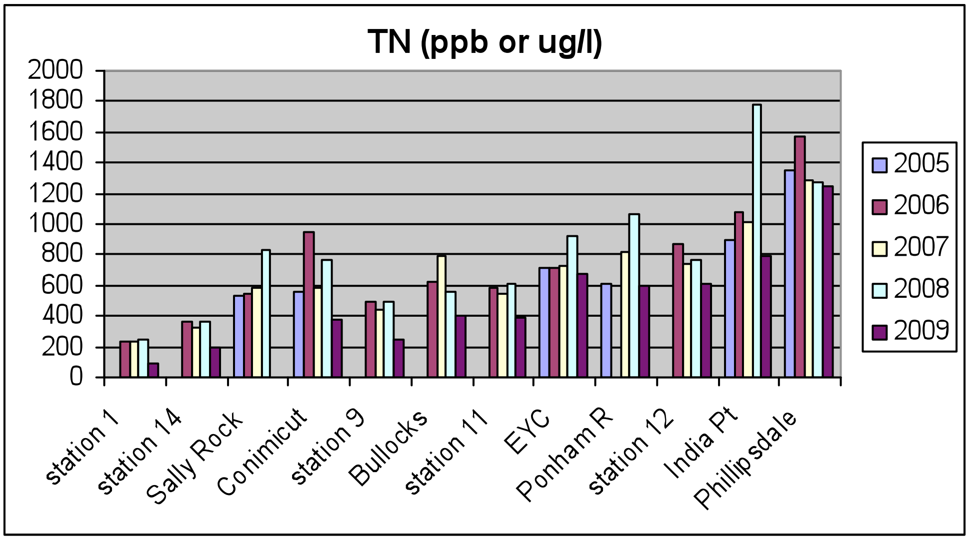
PN, minus DIN and POM, leaves dissolved organic N (DON). Organic N shows a significantly weaker N-S gradient than TN in Narragansett Bay – generally between 200 and 300 ug/l regardless of location (though occasional large anomalies deserve investigation).



Examination of Potential TN Criteria and Sources & Concentration Ranges by Waterbody

Based on a review of criteria proposed in surrounding states (particularly the Massachusetts Estuaries Program – see appendix B), if RI were to develop estuarine nutrient criteria, it is likely that Total Nitrogen would be the most useful nutrient measure, and target total nitrogen (TN) concentrations would probably be in the vicinity of 0.35-0.40 mg/l (approximately 25-30 micromolar). Target concentrations might be less, perhaps ~0.20 mg/l, in areas where eelgrass restoration is needed.

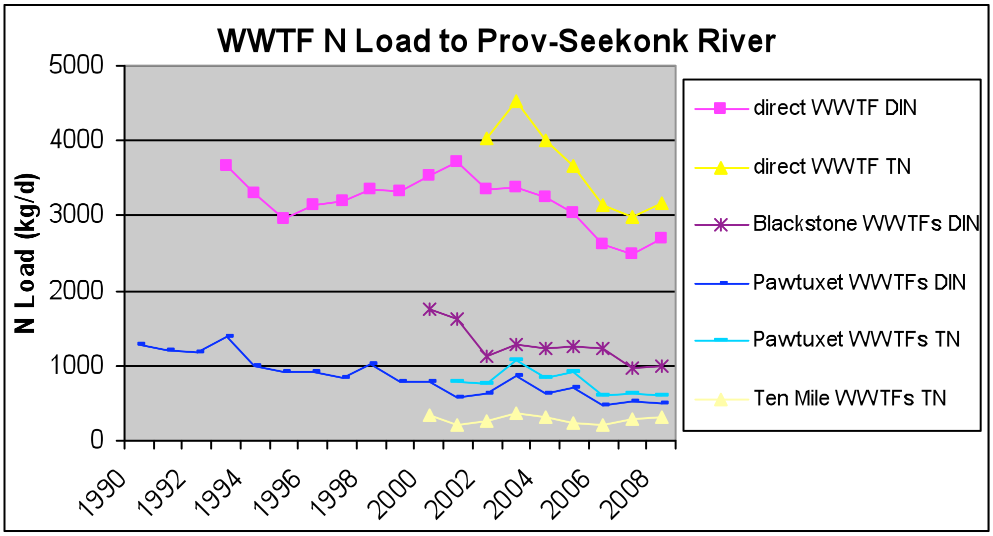
The below fig provides the more recent TN concentrations by Bay area from three sources (NuShuttle/MERL, NBC and URI Watershed Watch-see maps at end of this appendix p.40-43).



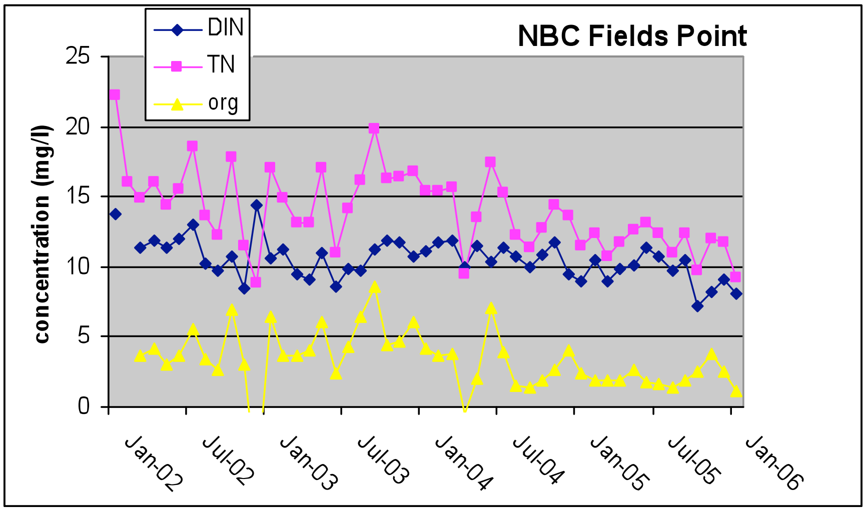
If TN concentration targets were set at 0.35 – 0.40 mg/l and the Oviatt TN nutrient gradient was accepted as representative, areas requiring nutrient reduction would be much the same as the areas presently identified as impaired by low DO levels. If eelgrass restoration required lower targets of ~0.2 mg/l, areas north of Jamestown would require nutrient reduction. Regulatory objectives would appear to be quite similar to objectives set now by DO standards.

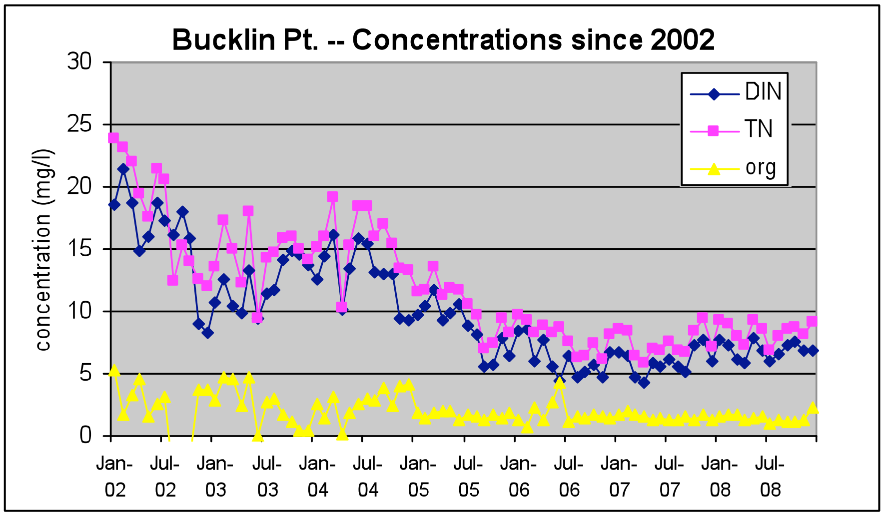
TN Sources

The largest source of N to the bay is from WWTFs. Based on discharge monitoring reports submitted to DEM, N load has decreased substantially since 2003. 85 to 90% of TN from WWTFs, on an annual average basis, has been DIN.



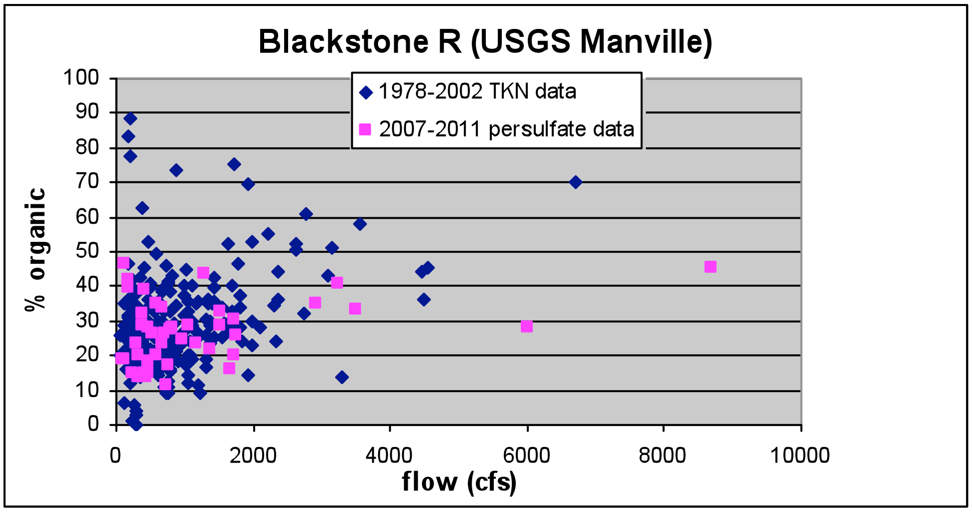
Effluent concentrations from the two largest WWTFs discharging to the bay are shown below:





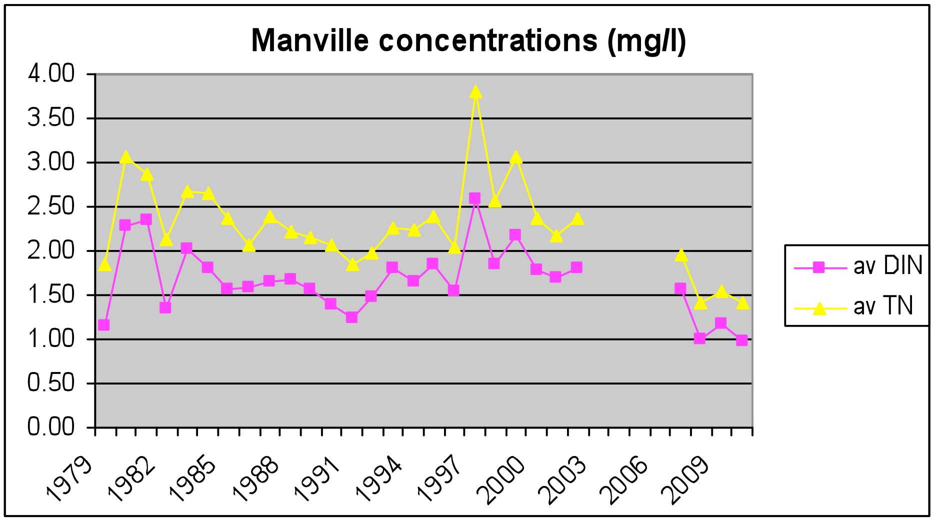
<u>River inputs</u>, based on data from USGS gauges, have a somewhat larger fraction of DON.

For the <u>Blackstone</u> (second largest tributary at an average 1,150 cubic feet per second):



(Note differences in measurement techniques – more about that later.)

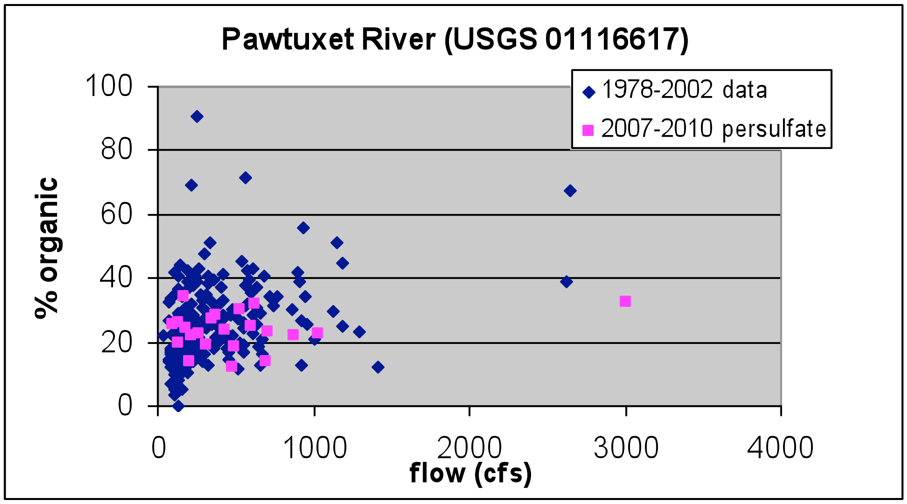
Concentrations remain high but have decreased substantially over the past 15 years:



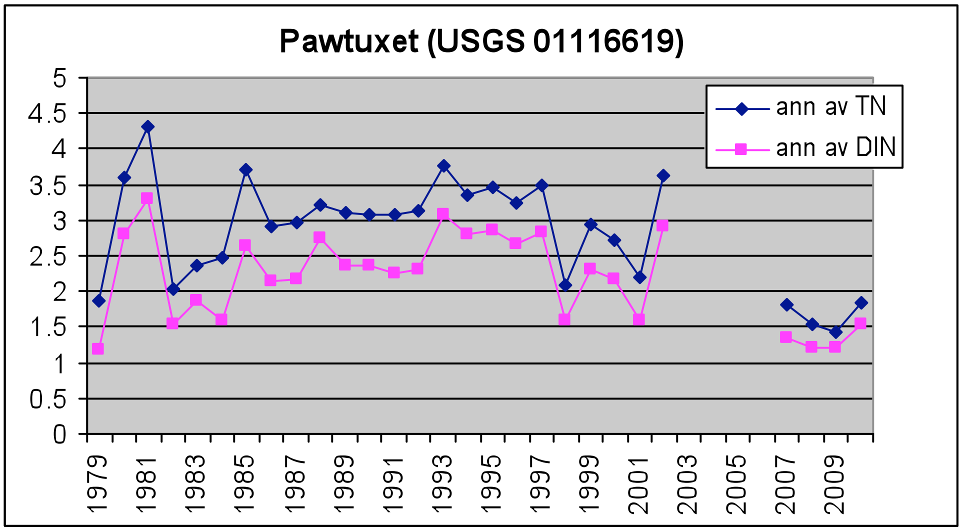
Most of the decrease appears to be due to large reductions in load from the Woonsocket WWTF (whose load dropped sharply around 2000) plus improvements in the UBWPAD facility in the last few years. (Note that USGS water quality monitoring on RI’s portion of the Blackstone River was suspended after 2002 and not restarted until 2007.)

EPA’s suggested riverine N criterion for Ecoregion XIV in which the Narragansett Bay watershed is located is 0.71 ppm (mg/l). Despite the substantial decreases in average annual concentrations, the Blackstone River remains a factor of two above the recommended threshold. The Blackstone River from the MA/RI state line to the Seekonk River is included on RI’s 2010 list of impaired waters for low DO and high TP with a 2018 TMDL planned – despite downstream impacts, it is not listed for TN.

For the Pawtuxet (long-term average flow of 575 cubic feet per second (Ries, 1989)), the organic fraction of TN appears to be somewhat lower than for the Blackstone.

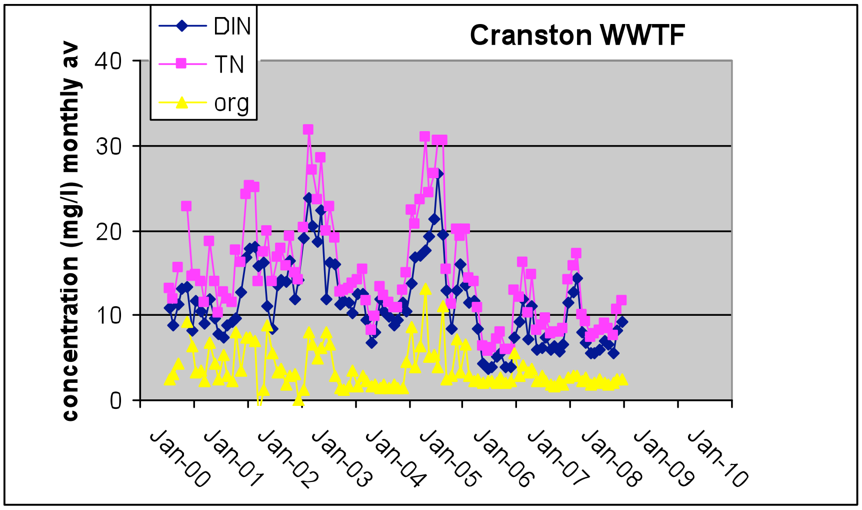


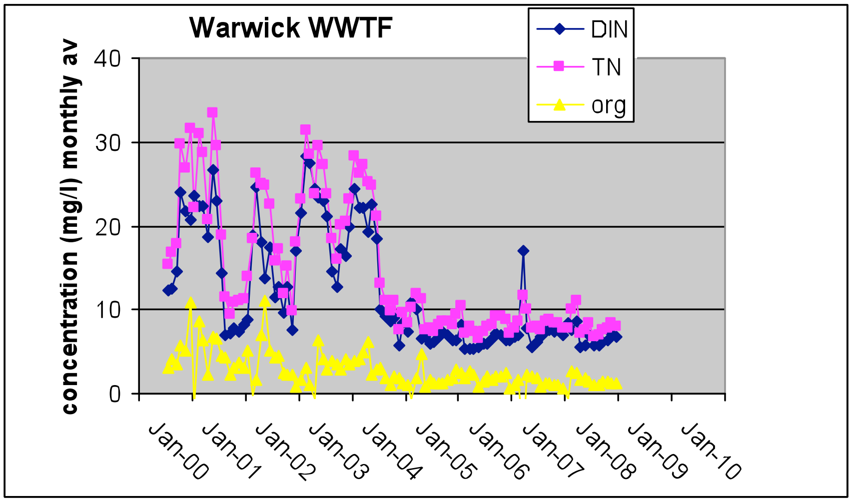
Average annual concentrations of N in the Pawtuxet have also decreased in recent years, presumably due to improvements to the three WWTFs discharging to the river.



Surprisingly, the average annual TN concentration for 2010 was not dramatically higher than 2008 or 2009 despite major flooding which severely damaged all three WWTFs along the river. The mainstem of the Pawtuxet River is listed as impaired by TP in RI's 2010 list of impaired waters. Similar to the Blackstone, the Pawtuxet is not listed as impaired by TN despite downstream impacts and concentrations well above EPA's suggested criteria.

Below are plots of the discharges from the two larger WWTFs along the river. The Cranston plant discharges about 10 MGD while the Warwick plant discharges about 5 MGD.





The Woonasquatucket River (average flow of 182 cubic feet per second (Ries, 1989)) has been sampled by NBC since 2007. Average annual TDN concentrations ranged from 790- 1050 ppb. The organic fraction rises during the growing season but is never more than half of the TN concentration. The DIN is predominately nitrate. The lower reaches of the Woonasquatucket are listed as impaired by low DO but not explicitly for nutrients. The measured averages would exceed EPA's suggested ecoregion TN criterion of 710 ppb for rivers and streams as well as any likely estuarine nutrient criterion (though none of the river is classified as marine).

The Moshassuck River (average flow of 55 cubic feet per second (Ries, 1989)) has also been sampled by NBC since 2007. Much of the river, including its lower reaches, is listed as impaired based on benthic macroinvertebrate bioassessments but not specifically for nutrients. Average annual TDN concentrations ranged from 940-1070 ppb. TN constituents vary in similar ways as for the Woonasquatucket.

The Ten Mile River (average flow of 145 cubic feet per second (Ries, 1989)) has also been included in NBC's sampling since 2007. Average annual TDN concentrations range from 1600-1950 ppb, predominately in the form of nitrate with organic content rarely more than 20%. These concentrations are well above suggested riverine and estuarine nutrient criteria. Two WWTFs contribute to this load. Attleboro's WWTF typically discharges about 5 MGD with relatively high TN concentrations.

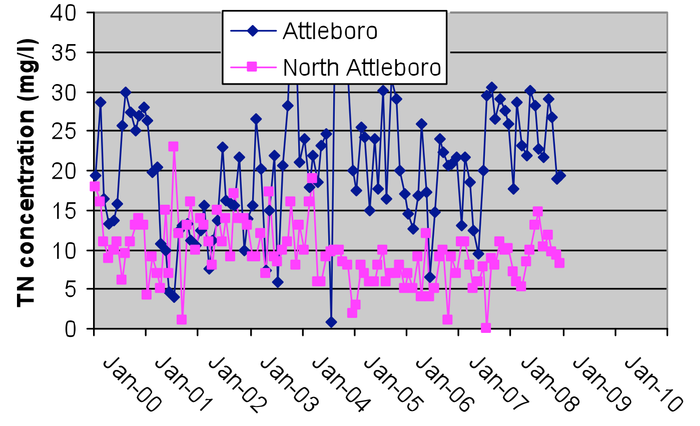


Figure 35 Ten Mile River WWTFs

TDN/orthophosphate ratios suggest that the sampling site at the outlet of Omega Pond, which is at the mouth of the Ten Mile River, is always P limited. Omega Pond is listed as impaired by low DO and high TP in RI's 2011 impaired waters list. Orthophosphate concentrations averaged approximately 20 ppb compared to EPA's suggested ecoregion criterion of 31 ppb for TP.

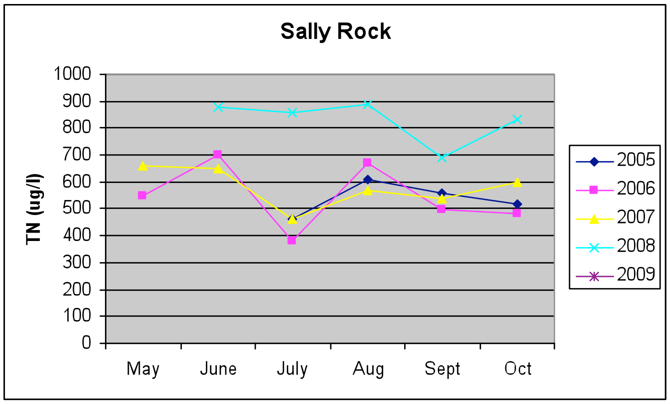
The Palmer River has an average flow of 120 cubic feet per second. Its lower reach from the MA-RI border is listed as impaired by TN and DO in Rhode Island's 2010 303(d) list. TDN concentrations, as measured by NBC, have averaged 490-680 ppb on an annual basis for 2007-2009. Note that these concentrations are below EPA's suggested riverine ecoregion TN criterion (710 ppb) but above likely levels of 350-400 ppb if estuarine nutrient criteria were to be adopted (the RI section of the Palmer is classified as marine). Organic content of TN ranges from less than 50% in winter to 100% in the late growing season. Ammonia is a significant fraction of DIN only when levels are drawn down.

In summary, upper bay tributaries have TN concentrations which, although reduced in recent years, remain above EPA's suggested N criteria of 0.71 ppm (mg/L) for rivers and streams in this region. The tributaries are listed as impaired by TP but not by TN despite downstream impacts. **Organic N appears to be generally less than 30% of the TN load from rivers to the upper bay and less than 20% of the TN load from WWTFs discharging directly to the Bay.** Particularly as treatment plants improve nutrient removal, the organic N in their effluent may become more recalcitrant (i.e., not fully bioavailable). Bioavailability of organic N has been a research topic for many years (see Seitzinger and Sanders (1997)). In recent years, with tightening permit limits, the topic is getting renewed attention by WERF which is developing standard procedures for measurement.

<u>Subembayments</u> of Narragansett Bay may have different characteristics than main bay.

<u>Subembayments</u> of Narragansett Bay may have different characteristics than main bay and require different criteria depending on flushing rates and response of ecosystem indicators like eelgrass and D.O.

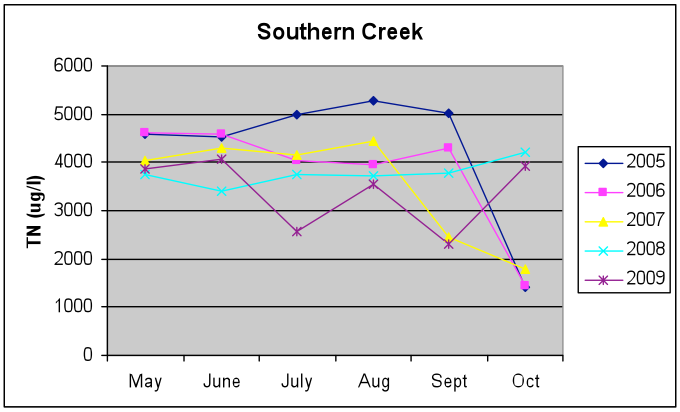
<u>Greenwich Bay</u>
Sally Rock, in the central area of Greenwich Bay, has been monitored by URI Watershed Watch since 2005 but, unfortunately, monitoring of all central bay sites by Watershed Watch was dropped after 2008. Sally Rock showed average (May through October) TN concentrations of about 600 ug/l or ppb (except in 2008 which was anomalously high).



Other stations at Middle Ground and The Brothers showed similar concentrations. Monitoring continues at 3 marinas around Greenwich Bay (Ponaug Marina, Little Rhody Boat Club and Warwick Cove Marina) – showing higher concentration than the central areas with typical TN monthly averages ranging from about 600 ug/l at Warwick Cove to 1000 ug/l at Ponaug.

Watershed Watch also continues to monitor nutrient concentrations in 4 tributaries to Greenwich Bay – the Maskerchugg River (feeding into Greenwich Cove), Hardig Brook (discharging to Apponaug Cove), and Tuscatucket Brook and Southern Creek (both flowing into Brushneck Cove). **All of these tributaries have TN concentrations well above EPA's suggested TN criterion of 0.71 mg/l for rivers and streams in this ecoregion.**

Lowest levels are reported for the Maskerchugg (typical monthly averages of 1200 ug/l).
The highest levels, remarkably high at 4-5 mg/L, are reported for Southern Creek:

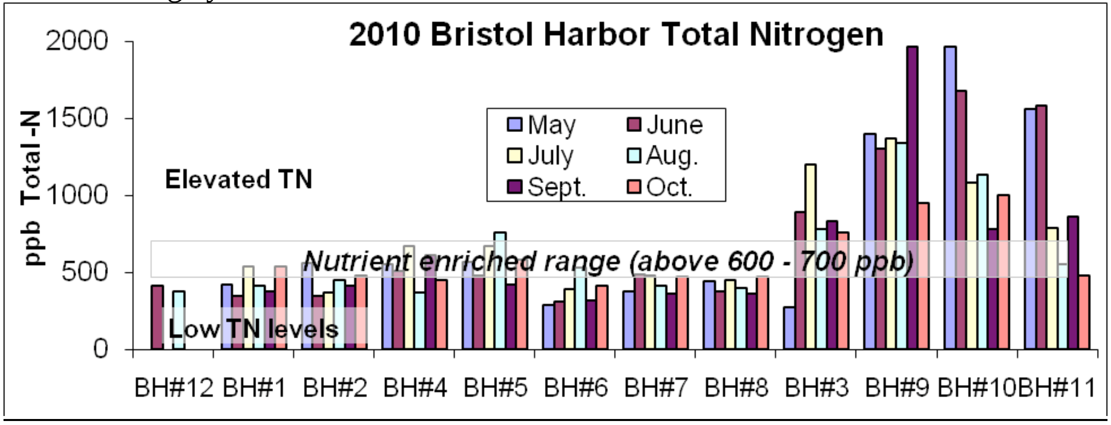


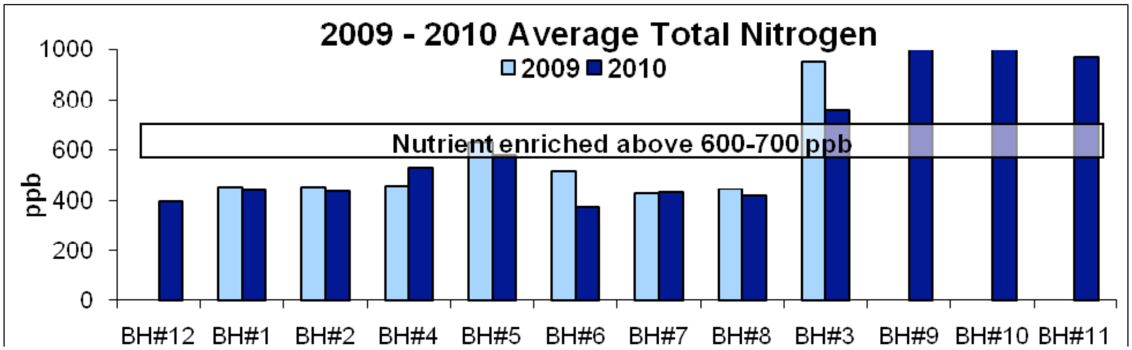
None of these tributaries are listed as nutrient impaired on Rhode Island's 2010 303(d) list. All are fairly small. The largest tributary, the Maskerchugg, carries an annual average of only about 12 cubic feet per second (Ries, 1989).

Organic N is usually less than 20% of TN in the tributaries and rarely more than a few percent in Southern Creek. The overwhelmingly DIN inputs are largely in the form of nitrate but ammonia forms a large part of the DIN measured in the central bay and periphery. This is in contrast with other subembayments described below. The tributary data appear to offer a valuable tool to prioritize work in the watershed. Greenwich Bay is included on RI's latest list of waters impaired by low DO and high TN. A TMDL is scheduled for 2016 if needed after WWTF upgrades and SAMP implementation.

Bristol Harbor

Bristol Harbor is listed in RI's latest list of impaired waters as meeting criteria for all designated uses. URI Watershed Watch data from samples collected by Save Bristol Harbor show annual average TN levels of 400-500 ug/l in the outer portion of Bristol Harbor (BH12 Herreshoff, BH1 Elks Club Dock, BH2 Bristol Harbor Inn, and BH8 Brito Dock – see Appendix B for map). Sites in the inner portion of Bristol Harbor have higher TN concentrations (a generalizable pattern in subembayments). Silver Creek (BH3, BH10 and BH11) shows concentrations of up to 2000 ug/l and may be a significant source of N load to the harbor. However, at the mouth of Silver Creek, organic N ranged from 65-70% of TN in May and June of 2009 to 30% in August in contrast to the Greenwich Bay tributaries where DIN was the predominant form of TN year around. But, similar to Greenwich Bay, DIN forms in Bristol Harbor were also largely ammonia.



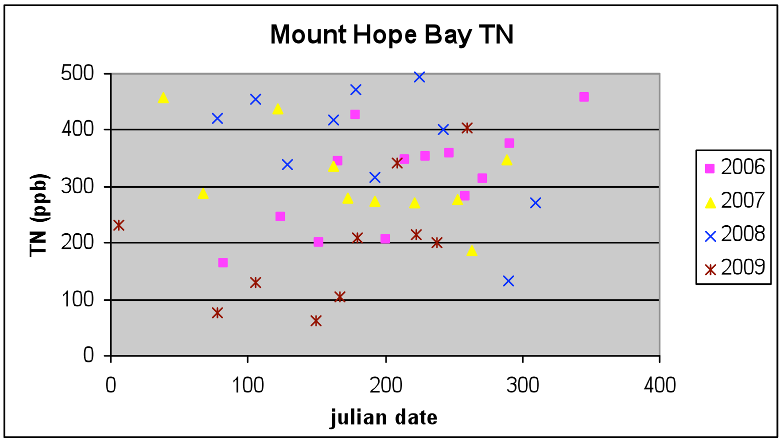


NuShuttle/MERL station 8 is in the east passage of the bay near the mouth of Bristol Harbor (and just north of the mouth of Mt. Hope Bay). Average annual TN concentrations there are typically close to

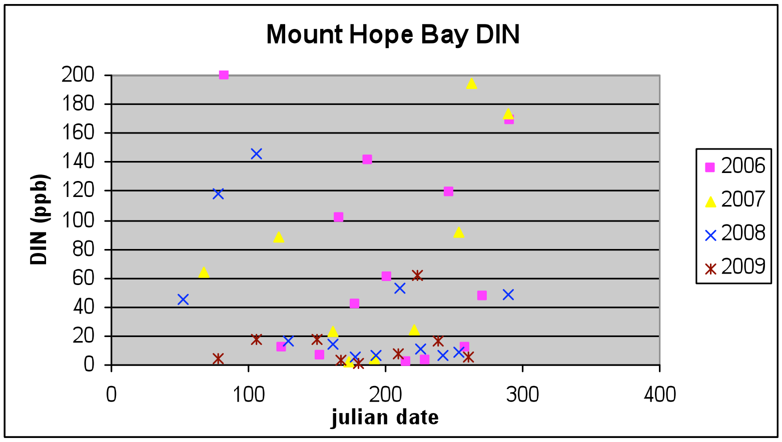
300 ug/l or ppb. Similar to the BH#12 site at Herreshoff, DIN concentrations are close to detection limits throughout the May-October growing season.

Mount Hope Bay

NuShuttle/MERL data show annual average TN concentrations of 300-400 ug/l or ppb, slightly higher than at NuShuttle/MERL station 8 outside the mouth of Mount Hope Bay.



DIN was often drawn down during the growing season but not as regularly as at open mid-Narragansett Bay sites.



Mount Hope Bay is included on RI's 2010 list of impaired waters for low DO and high TN. A TMDL is planned for 2014 "pending EPA/MA action". Massachusetts' portion of Mount Hope Bay is also included on that state's 2010 list of impaired waters for high TN, low DO (in segment between Braga Bridge and the mouth of the Cole River) and high chl a.

The Taunton River, which is the major freshwater source to Mount Hope Bay, had average annual TDN concentrations of 1100-1500 ppb in 2007-2009 according to NBC data. Concentrations at the mouth of the river were approximately 30% organic N (approximately 300 ppb) regardless of season. The TN concentrations were well above EPA's suggested TN criterion for rivers in Ecoregion XIV of 710 ppb. Orthophosphate concentrations averaged 28-77 ppb compared to the suggested ecoregion criterion of 31 ppb for TP. The Taunton River is included on Massachusetts' 2010 list of impaired waters for organic enrichment/low DO. The Taunton River drains an area of 562 square miles and carries a long-term average annual flow of 1,050 cubic feet per second according to Ries (1989).

The Cole River, a much smaller tributary to Mount Hope Bay, draining 13.4 square miles and carrying an average annual flow of 28.7 cubic feet per second (Ries, 1989), showed average annual TDN concentrations of 560-750 ppb in 2007-2009, very close or possibly exceeding EPA's suggested TN criterion for rivers in Ecoregion XIV of 710 ppb. 50-90% of the TN at the Cole River mouth was organic N with a distinct seasonal pattern highest during the growing season. The Cole River from route 6 to its mouth on Mount Hope Bay is included on Massachusetts' 2010 list of impaired waters for TN, DO and chl a with a TMDL required. Orthophosphate concentrations sampled by NBC at Milford Road in Swansea ranged from 6-21 ppb, well below the suggested ecoregion criterion for TP of 31 ppb.

The Kickemuit River, an even smaller tributary to Mount Hope Bay, draining 8.6 square miles, showed average annual TDN concentrations of 630-840 ppb in 2007-2009 (NBC data), above the suggested ecoregion TN criterion of 710 ppb in most years sampled. Similar to the Cole River, organic N contributions showed a distinct seasonal pattern from as low as 40% in winter to 100% in summer. The mainstem of the Kickemuit River is included on RI's list of impaired waters for P. Orthophosphate concentrations sampled by NBC at the lower end of the Warren reservoir ranged from 6-10 ppb, well below the suggested ecoregion criterion for TP of 31 ppb.

Salt Ponds/Coastal Ponds

RI DEM (2006) suggested that a TN target of 0.31 mg/l may be appropriate for both Green Hill and Ninigret Ponds. Green Hill Pond had an average TN concentration of 0.612 mg/l based on URI Watershed Watch/Salt Ponds Coalition data from 2000-2006 (6-8 values per year). For 2007, the average of 5 stations, 6 months data from each, sampled weekly, was 0.67 mg/l; for 2008, 0.53 mg/l; and for 2009, 0.58 mg/l. RIDEM conducted continuous DO monitoring and, on that basis, decided to list Green Hill Pond as impaired by low DO with a TMDL to be developed.

Ninigret Pond had an average TN concentration of 0.45 mg/l for 2000-2006. Average values from 6-8 stations (see Appendix B for map), 6 months per year (weekly samples), were 0.53 mg/l for 2007, 0.53 mg/l for 2008 and 0.47 mg/l for 2009. Ninigret Pond is listed as fully supporting use designations except habitat which was not assessed.

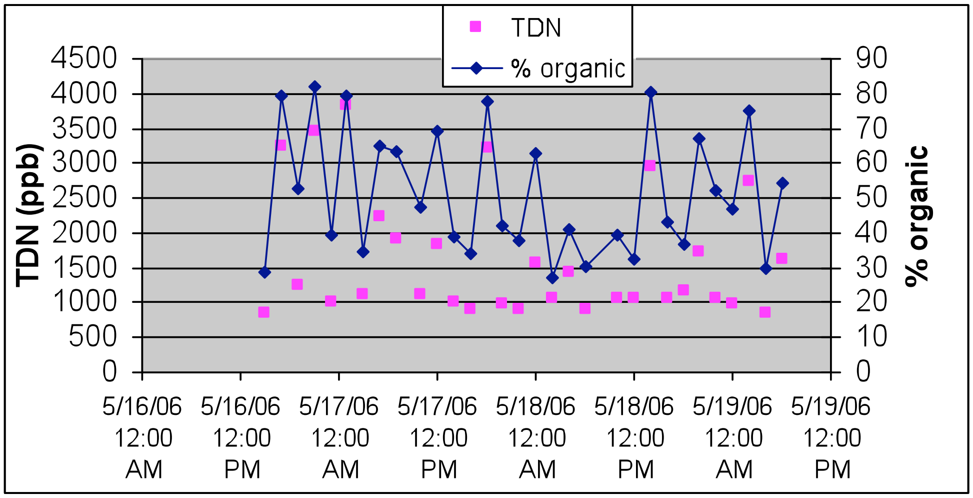
DEM (2006) used the Buzzards Bay Eutrophication Index (EI) methodology to arrive at its TN target. The overall eutrophication index is an average of indices for 5 parameters (DO, secchi depth, chl, DIN and TON – secchi has since been removed since the ponds are often too shallow to allow a clarity depth to be determined). DIN points vary linearly between 0 points for concentrations of 0.14 mg/l and greater and 100 points for 0.014 mg/l and less. (Note that this scale is more stringent than that used by the National Coastal Condition Assessment which rated concentrations between 0.1 and 0.5 mg/l as "fair".) TON (TN – DIN) points vary linearly between 0 points for concentrations of 0.6 mg/l and greater and 100 points for 0.28 mg/l and less. Both Green Hill and Ninigret Ponds have been designated as Special Resource Protection Waters and, in parallel with the Buzzards Bay approach, should have an EI goal of 65 or better. Assuming DIN concentrations are very low during the growing season and that the TON (which would be equal to TN if DIN is negligible) element of the EI should individually support the goal of 65, acceptable TN concentrations would be up to 0.39 mg/l. RIDEM’s suggested target of 0.31 mg/l was based on review of data for Green Hill and Ninigret Ponds.

	303(d) status	**av. Mo. TN(ppb) (sites)**			**Watershed N loading* (kg/ha/yr)**	**Salt Ponds Coalition 2009/2008**
		2009	**2008**	**2007**		
Pt. Judith Pond	no listed nutrient impairment	464 (7)	464 (9)	604 (5)	66-85	fair+ fair-
Potter Pond	fully supporting	425 (1)	400 (1)	NA	88-163	good fair+
Cards Pond	not assessed	no monitoring low salinity			132-245	
Trustom Pond	no listed nutrient impairment	no monitoring low salinity			76-138	
Green Hill Pond	low DO	584 (5)	532 (5)	671 (5)	60-87	fair- poor
Ninigret Pond	habitat not assessed	470 (8)	526 (8)	528 (6)	39-63	fair+ fair-
Quonochontaug Pond	fully supporting	387 (10)	429 (13)	467 (12)	29-44	fair+ good
Winnapaug Pond	fully supporting	528 (4)	617 (4)	488 (4)	69-125	fair- poor
Little Maschaug Pond	fully supporting habitat	NA	1087 (1)	1052 (1)	40-66	NA fair-

*** Nixon and Buckley (2007) – threshold for eelgrass = 30 kg/ha/yr (27 lb/acre/yr)**

Sampling and Analysis Issues

N concentrations vary with time at a wide range of scales but, in part because continuous monitoring instruments are not available, temporal variations can distort data. Seasonal variation has been discussed above. Irregular sampling intervals (as in all these monitoring programs to some degree) can bias annual averages. Many WWTF permits and associated monitoring are different in the May-October growing season. Watershed Watch also operates just during those months (although their partner in the coastal ponds, the Salt Ponds Coalition, uses only June-October data). Over short time scales, we have little information about the variation of N concentrations. In May of 2006, NBC found large variations among surface samples collected at Phillipsdale on the Seekonk every two hours over three days. Neither tidal nor diurnal patterns are evident.



Spatial variations are apparent not only as a gradient in the bay but also in subembayments. TN concentrations are almost always higher around the periphery of water bodies like Greenwich Bay and Bristol Harbor and a gradient is superimposed from the inner to the outer portions. Conditions of the Coastal Ponds are being evaluated based on averages of generally peripheral stations. In contrast, the Massachusetts Estuaries Project has defined its TN thresholds at “sentinel sites” in inner reaches of subestuaries. Analysis of N loading to Great Bay, NH, calculated steady state concentrations of TN (watershed N load divided by total water flushing time). In Narragansett Bay, the most significant limitation is simply the lack of TN monitoring in much of the bay. Termination of NuShuttle coverage last year leaves no bay-wide TN monitoring.

Sample analysis methodologies also complicate the TN picture in Narragansett Bay. NBC measures TN as the sum of Total Kjeldahl Nitrogen (TKN) plus nitrate and nitrite. Costa et al. (1992) described concerns of oceanographers that this method was not sufficiently sensitive to measure low ambient concentrations. A persulfate digestion was preferred. USGS has converted from TKN measurements to the persulfate method after careful comparison of the two techniques. (Presumably the differences shown above in river concentrations are, therefore, real changes not artifacts.) MERL/NuShuttle and URI Watershed Watch also use the persulfate method.

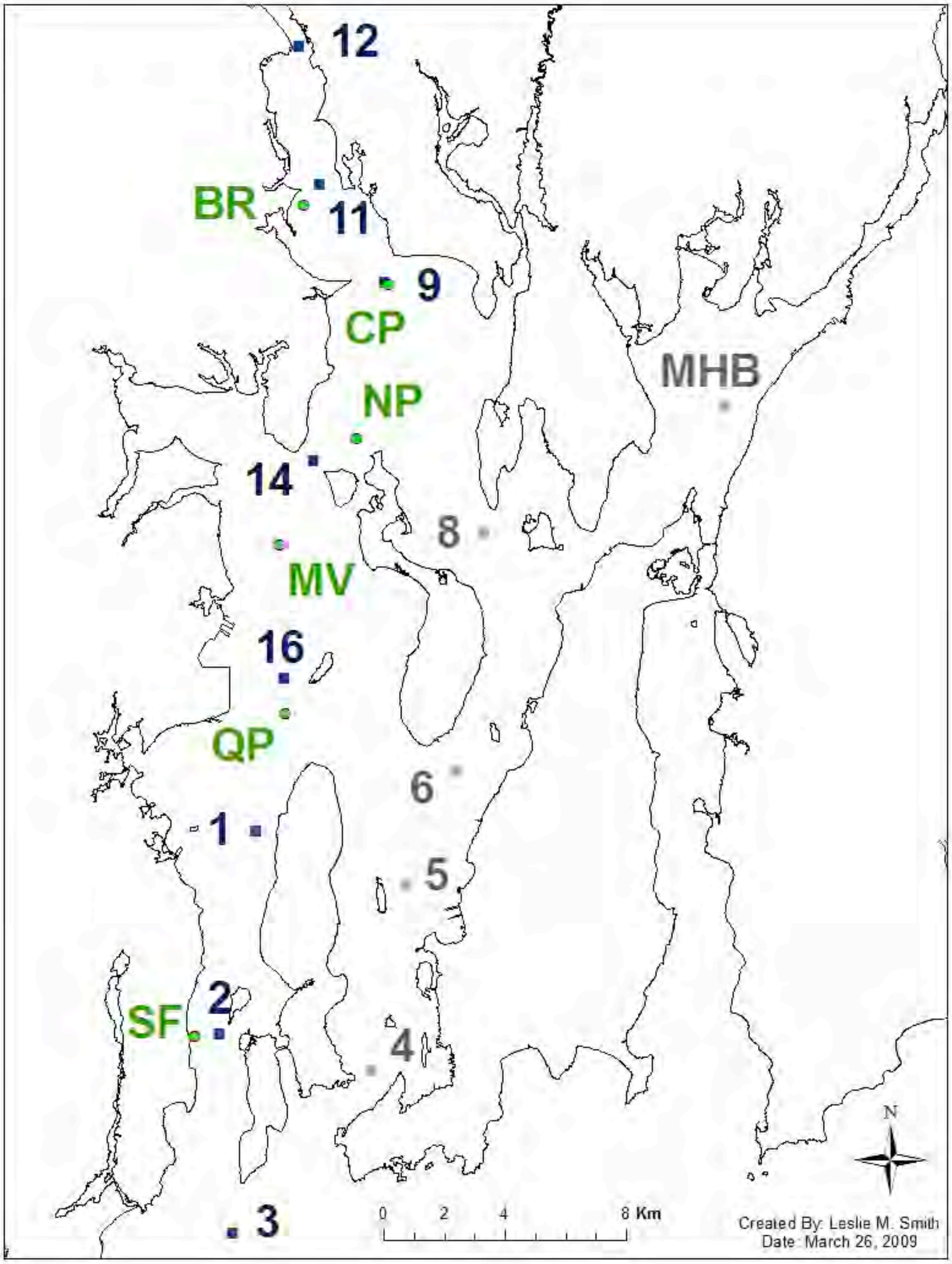
J. Krumholtz has provided the following discussion (2011) concerning the need to shift all TN measurements to the persulfate method vs the TKN method used by several groups in RI:

"The center of the Total Kjeldahl Nitrogen (TKN) vs. Alkaline persulfate issue is that TKN requires long digestion times, hazardous chemicals (concentrated sulfuric acid and mercuric chloride, although the latter has sometimes been replaced with copper), and does not capture all N endmembers. TKN converts organic nitrogen into ammonia, which is then read using typical colorimetric methods. It gives a fair estimation of TN when combined with a separate NO2+NO3 reading, but comparison of TKN+NO2+NO3 to Alkaline Persulfate TN are not consistent and often not close to 1:1 (see Bronk et al 2000, Patton et al 2003, Sharp et al 2002, Solorzano and Sharp 1980.). In contrast, TN using alkaline persulfate captures >95% of the nitrogen by digestion, is simple, and uses only mildly caustic reagents (approximately 1N sodium hydroxide with potassium persulfate). It also allows determination of TN with a single assay, because it converts all nitrogen end products to Nitrate and Nitrite, which can then be reduced to nitrite using a standard cadmium copper reactor, and measured by Greiss reaction. Furthermore, digested TN samples are stable on a benchtop at room temperature for extended periods, making laboratory analysis easier."

"Another serious problem for interstudy data usage is that there is not a regular series of intercalibrations between labs. Each lab operates with its own sets of standards, which are handmade, and rarely checked against anything with a truly known concentration, and there is no standardized methodology for collecting, processing and running samples (with respect to preservation, holding time, handling requirements, etc.) plus, many of us use different methodologies and chemical reactions depending on our instruments or the type of samples we run (for example, some labs use phenol/nitroferricyanide for ammonia, and others use a similar indophenol blue reaction, but with an EDTA buffer: the EPA method). At a minimum, we should be doing regular (yearly) intercalibrations to ensure that these variations don't impact results."

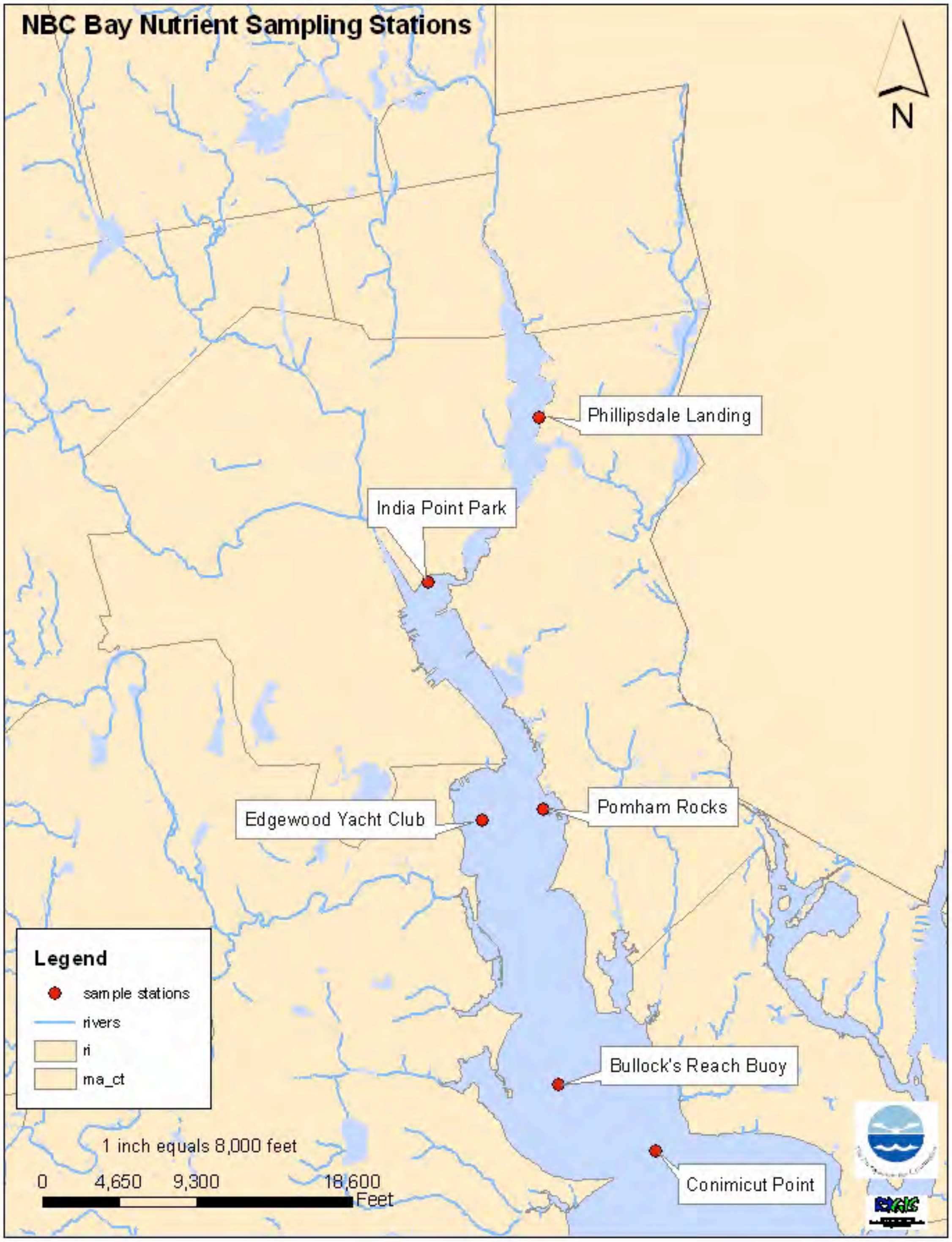
Appendix A – Maps of Narragansett Bay Nutrient Monitoring

1. NuShuttle/MERL



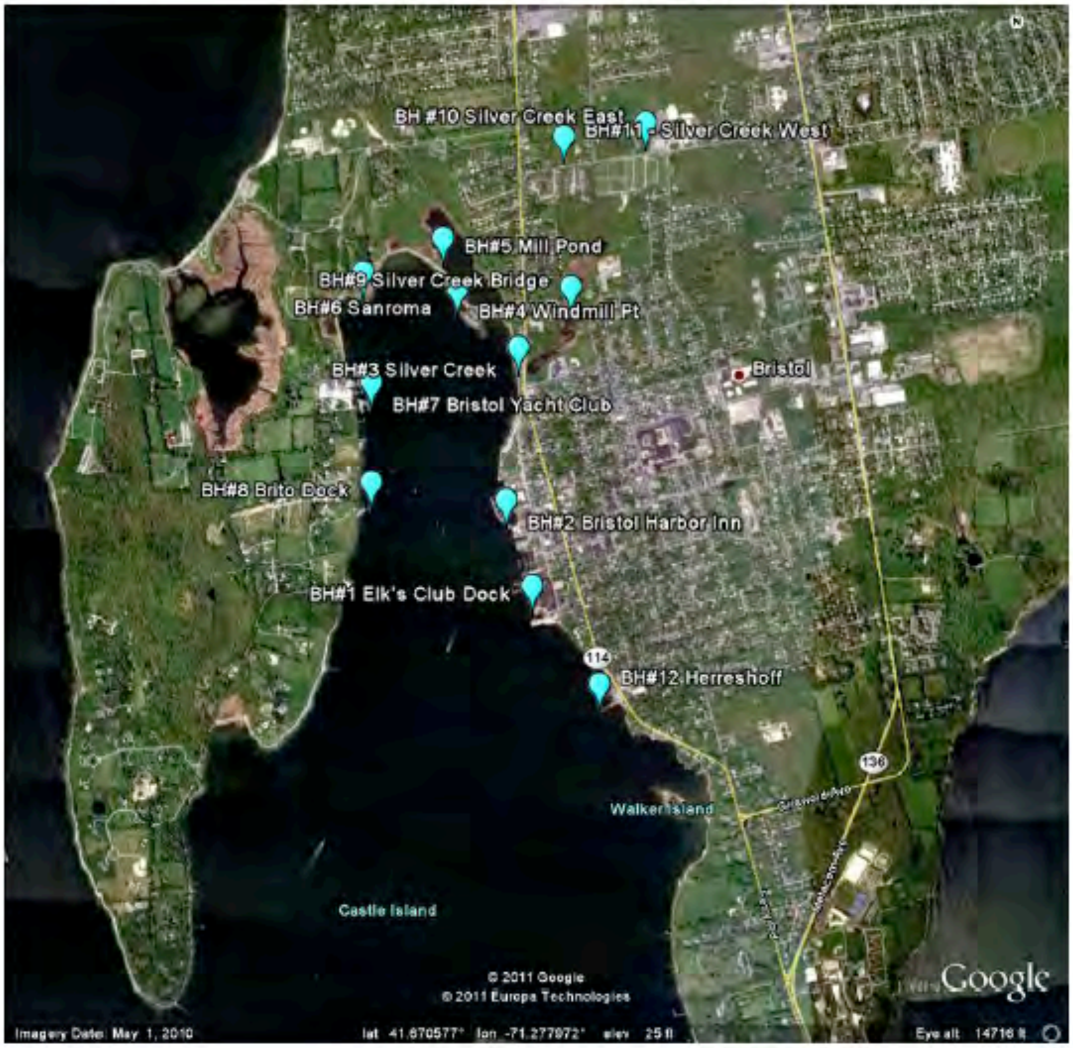
NuShuttle sampling locations (blue squares on map, designated “Station nn” in text)

2. Narragansett Bay Commission (NBC)



3. Save Bristol Harbor/URI Watershed Watch

Figure 1. 2010 Bristol Harbor Monitoring Sites



4. Salt Ponds Coalition/URI Watershed Watch

Aquatic Health of Green Hill Pond

June through October 2009

The aquatic health of each southern Rhode Island coastal salt pond was determined by applying the Buzzards Bay Water Quality Index (WQI). The WQI concept uses % dissolved oxygen saturation (DO), chlorophyll-a (an estimate of algae abundance), water transparency (Secchi depth, open-water stations only), and nitrogen abundances (the nutrient that fertilizes algae growth) to construct aquatic health measures. The average of the lowest 20% of DO measurements during the summer months are used in the WQI, and HIGHER DO saturations correspond to better water quality. Chlorophyll-a, measured in micrograms per liter (ug/L), is the opposite--LOWER chlorophyll-a concentrations (less algae) correspond to better water quality. Nitrogen abundance is measured in micrograms per liter (ug/L) two ways: as dissolved inorganic nitrogen (DIN), which equals nitrate plus ammonia, and total organic nitrogen (TON), which measures the organic matter in the water column. For both DIN and TON, LOWER concentrations correspond to better water quality. Finally, for water transparency measured using Secchi depth, HIGHER depth numbers indicate better water quality. Each of these parameters was assigned a WQI score based on its measured value at each sampling station; the results are presented on the above map and in the table below.

Water Quality Index	Score
Good	> 65
Fair +	50 – 65
Fair -	35 – 50
Poor	< 35

Green Hill Pond	Station	DO	Secchi	Chl	DIN	TON	Overall
Overall score:	In Pond (center)	41.0	---	83.9	35.8	48.0	44.8
38.5	Indigo Point	24.9	---	19.9	18.9	7.1	17.7
Fair -	Sea Lea	44.3	---	86.8	65.0	64.1	65.1
	Teal Road	30.7	---	43.1	0.0	31.2	26.2

Date: April 2010
Data Sources: Edward Callender
Elise Torello

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Appendix B – Other State Criteria

Recent History of Nutrient Criteria Development

Eutrophication impairs the majority of estuaries around the US and there has been little or no progress in improving conditions over the past decade (Bricker et al., 2007. Effects of Nutrient Enrichment in the Nation's Estuaries: A Decade of Change).

Despite EPA having a nutrient reduction strategy in place for a decade, a 2008 survey ("State Adoption of Numeric Nutrient Standards, 1998-2008, www.epa.gov/waterscience/criteria/nutrient/strategy/status.htm) found that only 3 of the 24 states that have estuaries have adopted numeric nutrient standards for one or more parameters (TN, TP, chlorophyll and clarity) for all of their estuaries, seven for part of their estuaries and 14 states had not adopted numeric nutrient standards for their estuaries.

As a result, EPA is being pressed to act on estuarine nutrient criteria and standards by a number of forces including:

1. A July 2008 petition by conservation groups from nine states along the Mississippi River and two national groups (NRDC and Sierra Club) calling for EPA to set and enforce nutrient standards to limit nutrient pollution in the river because it contributes to the "dead zone" in the Gulf of Mexico. (Actually EPA is being urged to set nutrient standards in the federal waters of the Gulf, then require states to establish consistent regulations for their waters. Three reports from the National Research Council (2008, 2009 and 2010) have also urged action.
2. Executive Order 13508 issued on May 12, 2009, called for the Federal Leadership Committee (chaired by EPA) to develop and implement a new strategy for protection and restoration of the Chesapeake Bay region, responding to several consent decrees and many evaluations of insufficient progress. The strategy was issued, as called for, on May 12, 2010. A draft TMDL ("pollution diet") – the largest ever developed by EPA (actually 294 TMDLs, one for each of N, P and suspended solids for each of 98 impaired Bay segments) – was issued in September 2010 and those limits, by jurisdiction and major river basin, are being incorporated into Watershed Implementation Plans by the states involved. The final TMDL is to be established by December 31, 2010.
3. A report by EPA's IG of August 26, 2009, "EPA Needs to Accelerate Adoption of Numeric Nutrient Water Quality Standards" (www.epa.gov/oig/reports/water.htm).
4. An August 2009 consent agreement signed by EPA with the Florida Wildlife Federation, Sierra Club and others on nutrient standards for FL, agreeing to set standards for fresh water by January 15, 2010 (which was done – see www.epa.gov/waterscience/standards/rules/florida) and for estuaries by January 2011. Both are to be finalized within 9 months of proposal. It is a first-time use of EPA authority to impose standards on a state. The freshwater documents are huge and drafts were criticized as a Chinese menu rather than absolute standards. Final regulations were issued in early December. The state has proposed estuarine nutrient criteria but these have been based, in almost if not all cases, on the contention that current conditions are sufficiently protective of designated uses. A panel of EPA's Science Advisory Board is set to review the methodologies of the state and EPA in mid-December.

5. A suit filed in August 2010 by CLF and the Coalition for Buzzards Bay against EPA alleging that the agency (and others) have not met their responsibilities for reducing nutrient pollution around Cape Cod and Buzzards Bay in MA.

EPA's Office of Water responded to the agency IG on November 24, 2009, committing to a corrective action plan including an update of the state efforts every two years and, by May 2010, a methodology to determine on a state-by-state basis whether numerical nutrient criteria are required and, if so, what priority they need to have (states that have active nutrient reduction efforts might not be pushed hard on standards). As far as I know, the promised May action has not yet been completed.

In April of 2010 EPA's Science Advisory Board provided a "Review of Empirical for Nutrient Criteria Derivation" recommending that load-response models be recognized as alternative and complementary tools to empirical numerical nutrient concentration criteria techniques, pointed out associated uncertainties and urged use of multiple methodologies, and drew attention to the need for downstream protective values. The Office of Water is revising its guidance on nutrient criteria and the SAB has been asked to continue its review role, particularly focusing on the Florida estuarine criteria.

Massachusetts

Mass. Estuaries Project (MEP)(www.oceanscience.net/estuaries)
TMDLs at www.mass.gov/dep/water/resources/tmdls.htm

Massachusetts has not initiated an effort to develop nutrient criteria statewide. The major effort in that direction has been the Massachusetts Estuaries Project centered at UMass/Dartmouth and covering Cape Cod, Buzzards Bay, Mt Hope Bay and the Islands. Roughly half of the 89 planned embayments have been completed and TMDLs have been issued by MA DEP for most of those areas. A lawsuit was filed in August 2010 by CLF and the Coalition for Buzzards Bay alleging that EPA and others have failed to meet responsibilities to reduce N loadings. (Under a series of settlement agreements, MWRA has upgraded treatment, built an ocean outfall and is pursuing CSO abatement, resulting in substantially improved water quality in Boston Harbor and Massachusetts Bay and relieving pressure to set nutrient limits there.)

The MEP effort involves intensive data collection and modeling supported by a combination of federal, state and local funding. Restoration of historical eelgrass, if records indicate it was present, as well as benthic infauna condition are important considerations in setting nutrient thresholds (reference conditions). A water quality model developed by the Corps of Engineers (RMA-4) is used to determine tidally-averaged TN thresholds at representative "sentinel site" (or compliance site) in each embayment.

Town	Embayments	TN threshold (tidal average at sentinel sites)
--- Cape Cod		
Orleans/Eastham	Rock Harbor system (no historical eelgrass)	0.50 mg/l (1.00 mg/l in low salinity Namskaket and Little Namskaket Marsh/Creeks)
Orleans/Harwich/ Brewster/Chatham	Pleasant Bay system (bioactive N only 25-50% of TN; >1/2 total N load generally)	0.16-0.20 mg/l bioactive N (DIN+PON) – MEP 0.12-0.25 mg/l (except 0.41 mg/l benthic N flux for upper Muddy Creek) – TMDL
Chatham	Stage Harbor system (benthic N flux 36% of total N load)	0.38 mg/l – MEP, TMDL
Barnstable/Yarmouth	Lewis Bay	0.38-0.50 mg/l (except 1.0 mg/l

		for low salinity Halls Creek)
Barnstable	Centerville River system	0.37 mg/l
Barnstable	Three Bays (Cotuit Bay = net sink; North Bay = big source of N due to benthic flux)	0.38 mg/l – MEP, TMDL
Barnstable	Rushy Marsh (benthic N flux = sink for 1/3 total N load)	0.50 mg/l
Mashpee/Barnstable	Popponesset Bay (benthic = small net sink)	0.38 mg/l
Mashpee	Waquoit Bay system (benthic N flux = 54% of total N load for Hamblin/Jedu Pond system)	0.38 mg/l (except 0.45 mg/l for Jehu Pond and 0.50 mg/l for Quashnet system)
Falmouth	Bournes/Green/Great/ Perch Ponds (benthic N flux > ½ total N load except sink for Green Pond)	0.40-0.45 mg/l
Falmouth	Little Pond	0.45 mg/l
Falmouth	Oyster Pond (2-4 ppt salinity)	0.63 mg/l
--- Buzzards Bay		
Falmouth	West Falmouth Harbor (benthic = sink for ~1/3 total N load)	0.35 mg/l
Bourne	Phinneys Harbor/Back Bay	0.35 mg/l
Dartmouth	Slocums and Little River	0.37 mg/l
--- Islands		
Martha's Vinyard		
Edgartown	Edgartown/Great Pond	0.50 mg/l
Nantucket		
Nantucket	Nantucket Harbor	0.36 mg/l
Nantucket	Sesechacha Pond (little tidal exchange)	1.00 mg/l

MA regulations, in addition to the TMDLs and wastewater management plans, require and constrain nutrient reductions. For example, Title 5 requires upgrades of septic systems at sale; groundwater discharges >10,000 gpd require permits; and ocean outfalls off Cape Cod are prohibited by the Coastal Sanctuaries Act.

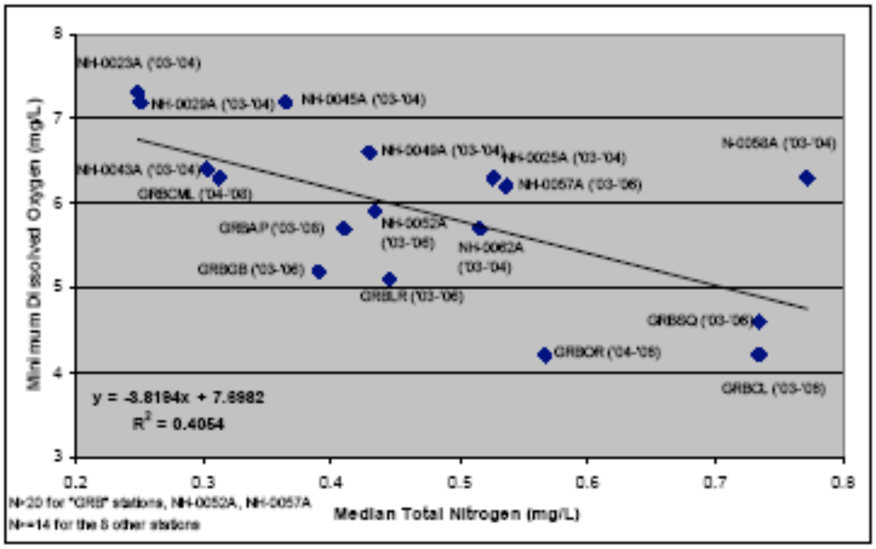
The CLF/Coalition for Buzzards Bay suit argues that (1) EPA should have classified septic systems as point sources and (2) EPA, the Cape Cod Commission and Barnstable County have failed to update and implement wastewater management plans. Septic systems are generally the largest controllable source of N loading to Cape Cod estuaries (roughly 85%). Consultants for Barnstable County have examined alternatives for wastewater treatment and estimate costs to be between $4-8 billion (Barnstable County Wastewater Cost Task Force (2010) Comparison of Costs for Wastewater Management Systems Applicable to Cape Cod. www.ccwpc.org). Centralized systems are more cost-effective but the largest portion of costs are related to collection systems. Estimated costs range from low of $230/lb N removed/yr for a 3 MGD centralized treatment system to $830/lb N removed/yr for an N-removing ISDS. A mix of ISDSs, cluster systems and sewers/centralized treatment is likely to be needed. The Cape Cod Water Protection Collaborative's web site includes much of the planning and analysis reports (www.ccwpc.org).

New Hampshire

New Hampshire has developed nutrient criteria for the Great Bay Estuary which includes most, if not all, its estuarine waters (see Trowbridge (2009) Numeric Nutrient Criteria for the Great Bay Estuary, www.prep.unh.edu/resources/nutrient/20090601_nutrient_criteria.pdf).

NH DES used an empirical effects-based (or weight of evidence) approach. Water quality measurements from different sections of the estuary were used to develop linear regressions between nitrogen concentrations and chlorophyll-a, dissolved oxygen and water clarity. Low dissolved oxygen and loss of eelgrass habitat were considered the most important impacts to aquatic life from nutrient enrichment in the Great Bay Estuary. Specifically, in order to maintain instantaneous dissolved oxygen concentrations greater than 5 mg/l and average daily concentrations greater than 75% saturation, the annual median TN concentration should be less than or equal to 0.45 mg/l and the 90th percentile chlorophyll-a concentration should be less than or equal to 10 ug/l.

Figure 29: Relationship between Dissolved Oxygen and Nitrogen at Trend Stations



For the protection of eelgrass habitat, the annual median TN concentration should be less than 0.25-0.30 mg/l and the annual mean light attenuation coefficient should be less than or equal to 0.5-0.75/m depending on the eelgrass restoration depth.

On average, N associated with organic matter (both dissolved and particulate) accounted for 59-62% of TN. However N in phytoplankton (calculated as 6% of the biomass indicated by chlorophyll a concentration assuming that chl-a was 5% of the biomass) was only 1% of TN. DIN was 36-41% of the TN.

This analysis has been interpreted in the press as requiring that N loading be reduced by roughly half. Estimated costs particularly for the WWTF upgrades raise public concern (the possibility of ocean outfalls has been raised). However, 65% of current N load is estimated to come from non-point sources. The state DEP is criticized for not having a more targeted approach to reduce those non-point sources.

Municipalities around Great Bay petitioned NH DES in May of 2101 to delay its process, use a formal rule-making procedure and carry out an independent peer review (see www.cityofportsmouth.com/publicworks/wwmp/Volume1-WMP/AppendixC/AppendixC.pdf). The peer review was completed in June of 2010 (see www.des.nh.gov/organization/divisions/water/wmb/coastal/documents/20100629-peer-review.pdf).

Maine

The Maine legislature passed a resolution in 2007 calling for ME DEP to develop a conceptual plan for establishing nutrient criteria for its coastal waters. In response, ME DEP submitted a response in June of 2008 (www.maine.gov/dep/blwq/report/2008/nutrient_criteria_report.pdf -- this report includes a 2008 report by Battelle, Conceptual Plan for Nutrient Criteria Development in Maine Coastal Waters). The approach recommended by Battelle was a hybrid of the Reference Condition/Data Distribution approaches as used for Yaquina Estuary, OR and Pensacola Bay, FLA pilot studies (see below) . They recommend using the median or percentile approach as potential criteria level for TN, DIN, chlorophyll, and DO. There are drawbacks to this approach as noted above and in the Yaquina and Pensacola work (see below). Maine is probably not a good example state for RI because much of the Maine salt waters are influenced mainly by offshore nutrient loads, with highest levels offshore and decreased levels until you reach the mouths of developed large river systems.

Maine's preferred plan is to follow an empirical effects-based (or weight of evidence) approach. At present, however, data to support such an approach are lacking. Since the DEP lacks a comprehensive database on nutrient effects for marine waters, the department recommended that it proceed to implement nutrient criteria using a data-distribution approach. DEP has not found a reliable reference condition or reference waters. The department intends to select threshold values that are achievable and plans to consider costs, technology, etc. (Courtemanch presentation at NEIWPCC October 2008). Maine intends to complete drafting estuarine nutrient criteria by 2012.

Based on Dettman and Kurtz (2006), the state expects that a threshold of 0.5-0.7 mg/l TN will be protective of Maine's coastal waters (due to the high concentration from offshore bottom waters). Concentrations in most coastal areas of the state are below those thresholds although Portland, Down East and some other areas are expected to be affected..

DEP's response report includes estimates of costs for upgrades to WWTFs but does not address techniques or costs to reduce N loading from non-point sources.

Dettman and Kurtz (2006) Responses of Seagrass and Phytoplankton in Estuaries of the United States to Nutrients: Implications for Classification. US EPA document AED-06-102

Connecticut

Connecticut is reportedly not working on nutrient criteria presently. A TMDL was issued in 2000 to achieve water quality standards for DO for Long Island Sound based on extensive modeling (and rough cost estimates). It called for 58.5% reduction in N loading to the sound. Although loads have been reduced (through a largely successful innovative trading program) and DO conditions improved, there are concerns that the planned reductions may not actually improve DO conditions to the extent required. Modeling work continues and further upstream sources are being evaluated as possible TMDL revisions are considered. In February of 2010, EPA Region 1 took the rare step of blocking a state-issued draft WWTF discharge permit for Hartford, VT's Quechee WWTF citing the impact of additional N on Long Island Sound.

Long Island Sound:

After 15 years of monitoring and related modeling and synthesis, a Total Maximum Daily Load (TMDL) for nitrogen loading to the Sound was approved by the EPA and the states of New York and Connecticut. This TMDL was established in order to meet DO water quality criteria in LIS. A multiyear effort has been phased in by these States to meet the TMDL. Cumulative point and nonpoint nitrogen load of all in-basin sources are to be reduced by 58.5% (10% reduction of total non-point load of N + 63.5% reduction of point source discharges) over a 15 year period (5-year incremental targets). The TMDL estimates even after these reductions have occurred, the state WQS for D.O. in the Sound would not be achieved. The TMDL therefore also requires reductions in nitrogen from out-of-basin sources in Phase Four, and the implementation of non-treatment alternative technologies in Phase Five.

Delaware:

Indian River, Rehoboth Bay, and Little Assawoman Bay tidal portions of the stream basins require controls needed to attain submerged aquatic vegetation growth season (approximately March 1 to October 31). Thresholds: Average levels for dissolved inorganic nitrogen of (no more than) 0.14 mg/L as N, for dissolved inorganic phosphorus of (no more than) 0.01 mg/L as P, and for total suspended solids of (no more than) 20 mg/L shall be instituted. DE has also adopted dissolved oxygen and Secchi disk criteria linked to nutrients for its tidal ChesapeIndian River, Rehoboth Bay, and Little Assawoman Bayake Bay waters.

Chesapeake Bay:

In Chesapeake Bay, criteria have been developed for DO, water clarity, and chlorophyll *a* (EPA 2003). DO criteria have been assigned to five different regions of the bay defined by uses and depth and water clarity criteria have been assigned to four different salinity regimes. For chlorophyll *a*, a narrative standard was established for the entire bay. The large number of regional criteria is due to large amount of research and monitoring data that is available for this estuary.

Pilot Attempts – **Florida & Oregon 2007 - 2008** (Brown *et al*. 2007, Hagy *et al*. 2008).

In **Yaquina Estuary, Oregon**, existing data were used to examine spatial and temporal trends and a "weight of evidence" approach was used to develop criteria to protect eelgrass habitat (considered highly sensitive to nutrient addition). Criteria were derived for the 'dry season' (May-October). The estuary was divided into 2 zones for criteria development. Zone 1(lower estuary) is highly influenced by offshore coastal water and nutrient loading from the ocean. Zone 2 (upper estuary) is influenced by river NPS and point source nutrient inputs. Overall, water quality conditions in the estuary are presently good and support existing seagrass habitat. They followed EPA guidance (EPA 2001), and proposed criteria use median values from the existing dataset for DIN, phosphate, chlorophyll a, and water clarity (Brown et al. 2007). Oregon has an existing water quality standard for DO of 6.5 mg/L. Although this was closer to the 25th percentile it was recommended to keep this standard for Yaquina Estuary because the only apparent DO problem was an intermittent incursion of hypoxic waters that enters the estuary from offshore coastal waters.

A weight of evidence approach was also used in **Pensacola Bay, FLA** (Hagy *et al*. 2008). The use of historical data to develop a reference condition was evaluated, but for this bay the historical condition was actually more impacted by toxic (point) discharges than the current state. Nutrient loading to the system has decreased somewhat since 1980 although significant agricultural sources still exist. Present water quality was considered protective of the desirable uses, although some areas are experiencing loss of eelgrass. Hypoxic conditions appear to be the result of natural processes (high salinity water and local hydrodynamics) and a propensity toward low DO in the system and loss of seagrass in the bay were considered related to pre-1980 degraded water quality. It is unclear from some comments by Hagy whether in fact there are still nutrient impacts taking place on some eelgrass beds so this seems a potential weak argument. The goal was to keep water quality at its current levels and not to have it degrade. Criteria were proposed for Pensacola Bay based on the relative freshwater and seawater influences along the salinity gradients with separate criteria for oligohaline (<5 PSU), mesohaline (5-18 PSU), and polyhaline (>18 PSU). Use of summer median levels were proposed as criteria for chlorophyll *a*, Secchi depth, DIN, phosphate, TN (<35 µM), and TP (Hagy *et al*. 2008). Since DIN and TN concentrations track opposite to chl a and secchi in most cases, it seems odd to use the summer DIN and TN levels. Non-biologically active periods (winter) seem more appropriate for nutrient concentrations unless loadings are being used to estimate concentration if biological uptake were not occurring.

More Recent Florida Criteria Development

The consent agreement with Florida Wildlife Federation, Sierra Club and others calls for EPA to proposed estuarine nutrient criteria for FL by mid-January of 2011 (recently revised to mid-November of 2011). Although EPA rarely, if ever, imposes water quality standards on states, the court judged that it was necessary in this case. Since the judgment, FL DEP has put a large effort into developing criteria that would alleviate the need for EPA action. The state has drafted reports for ~30 estuarine systems (see www.dep.state.fl.us/water/wqssp/nutrients/estuarine.htm).

EPA has asked its SAB to review a technical support document that describes methods and approaches for developing numeric nutrient criteria for Florida's estuarine and coastal waters, downstream protection values to protect those waters, and criteria for flowing waters in the south Florida region. This document was made available in mid-November (see yosemite.epa.gov/sab/sabproduct.nsf/02ad90b136fc21ef85256eba00436459/c439b7c63eb914f8525773b004e53calOpenDocument) and the SAB panel will meet December 13-14, 2010. Florida DEP provided its "Proposed Methodology for the Assessment of Numeric Nutrient Criteria for South Florida Estuaries and Coastal Waters" in September 2010 on the web site listed above.

EPA's proposed numeric nutrient criteria for inland surface freshwaters, issued in January of 2010 per the consent agreement (and which could be promulgated by October of 2010), addresses the need for downstream protection by setting allowable loadings each estuary can receive. The proposal derives allowable loadings based on USGS' SPARROW model estimates, calling for half of the anthropogenic loading (difference between current and "natural" loading) to be reduced. The load can be allowed to be higher if TN is assimilated prior to "delivery" and vice versa. This proposal was modified when the final regulations were issued on November 14, 2010 (see water.epa.gov/lawsregs/florida_index.cfm).

Tampa Bay's current nutrient loading, under which it is making significant water quality progress according to Tampa Bay's National Estuary Program, is acceptable under the currently accepted TMDL. SAV coverage has been steadily improving. Average TN concentration is 0.56 mg/l.

However, EPA's downstream protective value approach would have required 50% reduction in anthropogenic loads.

Sarasota Bay has had >50% reduction in TN loads since 1988 and TN concentration has dropped proportionally. However changes in chl-a and water clarity have been less dramatic (and in some instances such as Roberts and Blackburn Bays opposite to expectations). SAV coverage has increased both in acreage and density since 1988 but less than the load reduction. Delays and nonlinear response to nutrient reduction have been reported elsewhere. EPA's methodology would require TN to be less than 0.54 mg/l (assuming 90% delivery from upstream) – a 50-70% reduction in some areas of the bay. Questions are being raised whether the biologically relevant TN load is being measured. Stormwater is estimated to be 62% organic N, WWTF effluent 38% organic, and atmospheric deposition is argued to be mostly organic.

For Pensacola Bay, Hagy et al. in a 2008 report ("An Approach for Developing Numeric Nutrient Criteria for a Gulf Coast Estuary", EPA report 600R-08/004) concluded that "water quality criteria for nutrients and nutrient-related water quality measures could be based reasonably on currently observed conditions because evidence that more stringent criteria are scientifically defensible, necessary, or even achievable, is lacking." Low DO events are reported to be associated with natural salinity stratification and natural organic material delivered by river systems, not nutrient enrichment. Turbidity is a temporary problem associated with storm events. Chlorophyll concentrations remain at levels that do not interfere with SAV photosynthesis. Ammonia toxicity problems associated with SAV loss, hypoxia and fish kills in the 1950s and 1960s have been resolved. Phytoplankton blooms, epiphyte growth and macroalgal problems have not been reported (see FL DEP presentation by Frydenborg of Aug., 2010, www.dep.state.fl.us/water/wqssp/nutrients/docs/estuarine/tallahassee/pensacola_bay_082410.pdf).

FL DEP has analyzed other FL estuaries and is proposing estuarine nutrient criteria as outlined in the following table (in order from western panhandle down to Keys then up the east coast):

		TN (mg/l)		
Segment	Existing Long Term Geometric Mean	Maximum Allowed Long Term Geo. Mean	Annual Geo. Mean for Network of Stations (<2/5yr exceed)	Annual Geo. Mean for Single Site (<2/5 yr ex)
Perdito Bay		later		
Pensacola Bay System				
East Bay	0.34	0.37	0.47	0.53
Escambia Bay	0.55	0.60	0.75	0.84
Pensacola Bay	0.37	0.41	0.51	0.55
Santa Rosa Sd	0.35	0.38	0.60	0.62
Choctawhatchee Bay System				
Central	0.37	0.41	0.52	0.52
Middle	0.34	0.38	0.45	0.46
East	0.40	0.44	0.45	0.45
Bay-wide	0.37	0.41	0.50	0.51
St. Andrew Bay System				
Central	0.42	0.46	0.63	0.65
East		N/A		

Grand Lagoon	0.41	0.45	0.55	0.57
Mouth	0.32	0.35	0.51	0.53
North Bay		N/A		
West Bay	0.42	0.46	0.56	0.58
Bay-wide	0.42	0.46	0.58	0.63
St. Joe Bay	0.225	0.25	0.28	0.31
Apalachicola Bay System				
Applachicola	0.69	0.76	1.00	1.03
East Bay	0.68	0.74	0.85	0.89
St. George Sd	0.45	0.50	0.57	0.59
St. Vincent Sd	0.64	0.70	0.75	0.75
Bay-wide	0.68	0.75	0.95	1.00
Alligator Harbor	0.27	0.30	0.41	0.46
Ochlockonee Bay		not enough data yet		
Apalachee Bay System				
St. Marks	0.36	not enough data yet		
Aucilla	1.1	not enough data yet		
Econfina	0.89	not enough data yet		
Steinhatchee	0.92	not enough data yet		
Suwannee Estuary				
Nearshore	0.72 (0.74-1.20)	0.79	0.97	1.08
Offshore	0.42	0.46	0.56	0.60
Waccasassa Estuary				
Nearshore	0.63 (0.57)	0.69	0.77	0.84
Offshore	0.48	0.53	0.61	0.66
Withlachoochee Estuary				
Nearshore	0.43 (0.60)	0.47	0.54	0.55
Offshore	0.33 (0.33)	0.36	0.41	0.41
Springs Coast				
St. Joseph/Clearwater				
Tampa	report discusses objections to EPA recommendations and argues that the existing TMDL should remain as the governing standard			
Sarasota (based on current chl + 1 std dev as protective of SAV and regressing for TN)				
Roberts Bay	0.54			
Little Sarasota	0.60			
Blackburn Bay	0.43			
Sarasota Bay	0.28-1.34 (based on ambient water color for period 1998-2009)			
Palma Sola Bay	0.93			
Charlotte Harbor		no recommendations yet		
Rookery Bay				
10,000 Islands				
Florida Bay System (based on maintaining existing conditions; no TN plots provided)				
Central	0.72	0.80	1.02	1.05
Southern	0.49	0.54	0.66	0.69
Western	0.30	0.33	0.39	0.42
East Central	0.52	0.57	0.68	0.70

Northern	0.55	0.60	0.71	0.72
Coastal Lakes	0.94	1.03	1.31	1.31
Florida Keys (based on maintaining existing conditions: no TN plots provided)				
Marquesas	0.14	0.16	0.20	0.21
Back Country	0.20	0.22	0.25	0.27
Bayside	0.20	0.22	0.26	0.28
Dry Tortugas	0.13	0.14	0.18	0.19
Oceanside	0.15	0.16	0.19	0.20

(note: this table is incomplete)

Summary of Other State approaches

State	N	P	Chl a	Clarity / other
CT	No criteria work planned planned Using complex DO ecomodel for TMDL N loading- 58.5% reduction of N load over 15 yrs	Y	Part of complex DO ecomodel for TMDL N loading	Secchi depth as part of complex DO ecomodel for TMDL N loading
DE	DIN ≤ 0.14 mg/L as N	DIP ≤ 0.01 mg/L as P	TSS 20 mg/L	DO + secchi of Ches Bay
ME	Expect to use median or %tile approach vs ref approach – expect range of 0.5-0.7 mg/l TN (2012)	TP draft	draft	
MD	N	N	SAV surrogate	SAV Ches.Bay restoration goal for clarity
MA	TN conc site-specific based on site specific seasonal surveys- ranges from 0.37-0.63 mg/L Most common is .37-.45 mg/L	NA	NA	eelgrass used as indicator of acceptable TN concentration
NH	site-specific TN conc (Great Bay) annual median TN ≤0.45 mg/l for D.O. TN≤0.25-0.30 mg/l for eelgrass habitat		90th percentile ≤ 10 ug/l	eelgrass habitat protection target annual mean light atten. Coeff. ≤ 0.5-0.75/m [dep. On eelgrass restoration depth]
LIS	See Ct above		See CT	See CT

Note Table 1 – (still in development) State approaches to Nitrogen Criteria

Addendum - **Mississippi River/Gulf of Mexico – Nutrient Controls**

The following is an excerpt from the 2010 NRC report, "Improving Water Quality in the Mississippi River and Northern Gulf of Mexico: Strategies and Priorities":

NUMERIC WATER QUALITY CRITERIA FOR THE NORTHERN GULF OF MEXICO

The 2008 NRC report recommended that "the EPA should develop water quality criteria for nutrients in the Mississippi River and the northern Gulf of Mexico" (NRC, 2008). That report explained that even if all of the ten states along the Mississippi River developed and fully implemented state-level water quality criteria, their cumulative efforts would not necessarily begin to reduce the areal extent of the NGOM hypoxic zone. Further, the report explained that"[a]n adequate approach to remediating northern Gulf of Mexico hypoxia would entail establishing numeric nutrient criteria for the mouth of the Mississippi and Gulf of Mexico waters that permit no more nutrient flow into the Gulf than could be accommodated by natural processes without significant oxygen depletion" (NRC, 2008, p. 126).

Establishing numeric criteria for nutrients in the federal territorial waters of the northern Gulf of Mexico would establish an overall goal of MRB nutrient water pollution management.[1] This action would have only one immediate legal consequence under the Clean Water Act: Louisiana, Mississippi, and Texas would have to determine whether their state Gulf waters meet the new criteria. From there, assuming that the state waters of the Gulf of Mexico did NOT meet the new criteria, each state would have to list its coastal waters as "impaired" in the next state Section 303(d) list and begin the Section 303 Total Maximum Daily Load prioritization process, which is designed to identify sources of pollutants across a watershed and create a plan for reducing pollutant loadings.[2] Section 303(d) of the Clean Water act requires each state to identify "impaired waters" of the state. This impairment determination is based on a comparison of observed conditions to state-promulgated water quality standards. Existence of numeric criteria makes this process of determining whether a waterbody is impaired more straightforward and transparent. Subsequent to this listing, each state also is required to prioritize its impaired waters and develop a Total Maximum Daily Load determination, which includes a determination of the maximum allowable loading of the problematic pollutant or pollutants. The resulting TMDL plan must be allocated to point sources and nonpoint sources, and include a margin of safety. If states do not meet these requirements, EPA is required to do so. Hence, establishing specific numeric criteria for Gulf coastal waters will set into motion requirements to develop plans for pollutant load reductions to meet those standards (see NRC, 2008 for details regarding implications of interstate water quality standards and TMDLs).

If the new numeric nutrient criteria apply to the federal waters of the Gulf of Mexico, the process for the EPA is less clearly specified in the Clean Water Act in Section 303, which by its language, applies only to states. Nevertheless, the Act itself clearly applies to the federal zones of the ocean, and the EPA has frequently issued National Pollutant Discharge Elimination System (NPDES, as specified within Clean Water Act Section 402; see NRC, 2008) permits for discharges of pollutants into federal ocean waters. Any apparent ambiguities regarding the application of the TMDL process to the NGOM are superseded by the authority clearly vested with EPA elsewhere in the Clean Water Act to protect water quality in the federal zones of the ocean. Along with comprehensive authority given to EPA over oceans and interstate pollution, the Clean Water Act allows for the crafting of a flexible and long-term implementation plan for achieving MRB water quality improvements throughout the Mississippi River basin, with goal to eventually reduce minimize NGOM hypoxia (See NRC, 2009 for discussion of CWA Sections 102, 104, and other relevant authorities.).

Importantly, and as already noted, numeric nutrient criteria for the northern Gulf of Mexico would represent a goal for the entire Mississippi River basin. Establishing numeric criteria for the northern Gulf would act as a driver and allow EPA and the Mississippi River states to begin working upstream in this large, complex watershed, as numeric criteria would provide an end point that could serve as the basis for setting standards in upstream states of the basin. Moreover, implementing numeric nutrient criteria in the federal waters of the northern Gulf of Mexico could provide the EPA with leverage when encouraging or mandating establishment of state numeric standards. Establishing NGOM numeric nutrient criteria also would complement the MRBI in moving toward a more systematic, adaptive, and coordinated basin-wide approach to managing nutrients and water quality.

To reaffirm and reemphasize a recommendation from the 2008 NRC report, the EPA should establish numeric criteria for nutrients for the waters of the northern Gulf of Mexico.

[1] An alternative to establishing criteria for nutrients as a water quality goal would be to establish a dissolved oxygen goal. Criteria for dissolved oxygen have been established and used as a goal for reducing hypoxia in the Chesapeake Bay, but the primary strategy to achieve that goal has been reduction in nutrient loads. In the Mississippi River basin and northern Gulf, nutrient loads from nonpoint sources are the prevailing driver of Gulf of Mexico hypoxia. Nutrient criteria thus represent a more direct means of addressing nutrient loadings across the basin and into the Gulf.

[2] Section 303 of the Clean Water Act addresses water quality standards and the Total Maximum Daily Load (TMDL) process. For more detail on Section 303 and its programs, see NRC, 2008, esp. pp. 78-85.

NRC. 2008. Mississippi River Water Quality and the Clean Water Act: Progress, Challenges and Opportunities. National Academies Press, Washington, DC

NRC, 2009. Nutrient Control Actions for Improving Water Quality in the Mississippi River Basin and Northern Gulf of Mexico. National Academies Press, Washington,

Appendix C – Particulate Organic Matter

Trowbridge (2009) reported (page 17) that "N associated with organic matter (both dissolved and particulate) accounted for 59-62% of TN. However N in phytoplankton was only 1% of the total." That seems surprising. (Note that Oczkowski at al. (2008) cites DeMilla (2006) as asserting that phytoplankton may account for less than 1% of total suspended solids less than 150um filtered from surface water in a fall collection – probably based on a calculation similar to Trowbridge's.)

Earlier in his report (page 5), Trowbridge wrote that "N in phytoplankton was calculated from the chlorophyll-a concentration in the sample and assuming that chlorophyll-a, carbon and nitrogen comprised 5%, 50%, and 6% of biomass by dry weight, respectively. The percentage for N was calculated from the ratio of particulate carbon to particulate nitrogen in 127 samples from the estuary. This calculated percentage is consistent with estimates from EPA modeling guidance." Thus N in phytoplankton would be 6/5 x chl-a. Even relatively high chl-a concentrations (10-20 ug/l) would comprise only 12-24 ug/l N -- small amounts by comparison to recommended thresholds of 0.25-0.45 mg/l N (250-450 ug/l N).

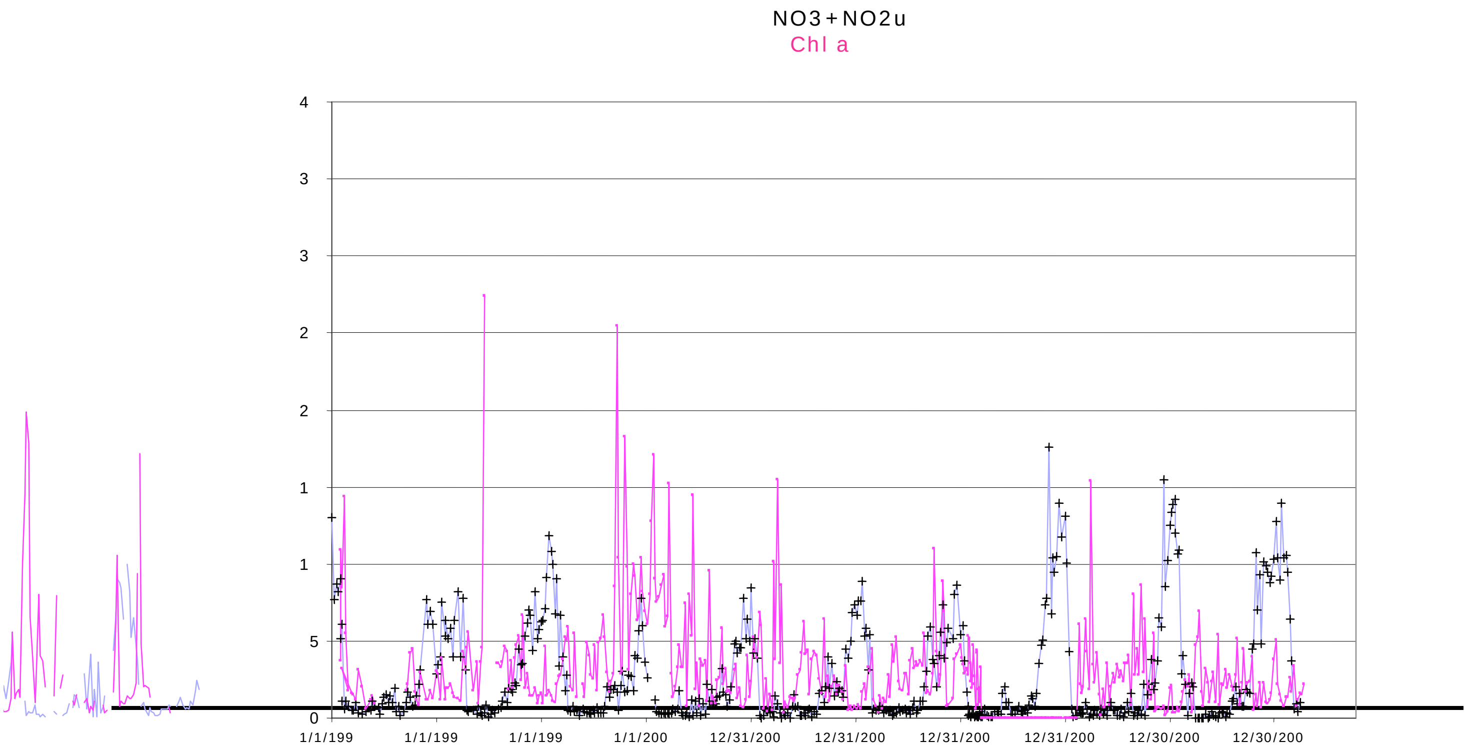
While Phil describes how he arrived at the 6%N in phytoplankton biomass, I can't find an explanation of the 5% estimate of chl-a in phytoplankton biomass. Valiela's 1995 book on Marine Ecological Processes (page 23) states that "for phytoplankton the ratio of biomass to chlorophyll averages about 62 and varies between 22 and 154." Unfortunately he doesn't give references for that statement. However Phil's 5% equates to a ratio of 20 which is below the range he reported. If Phil used the 62 average Valiela reported then the phytoplankton N would be estimated at a bit more than 3 times what he seems to have estimated -- perhaps 36-72 ug/l -- still smaller than I would have guessed but closer. If the upper end of Valiela's range was used (~150) then phytoplankton N would be estimated to be 7.5 times Phil's estimate -- so 90-180 ug/l. Even those concentrations, reflecting pretty strong blooms which the thresholds are set to avoid, would be less than half of the TN.

Bob Howarth's review of Trowbridge's report stated: "The report assumes that phytoplankton biomass is composed of 50% carbon by weight and 6% nitrogen (page 6). This gives a molar C:N ratio of 9.7 which is fairly high. I think using a lower value for carbon might be more reasonable, perhaps 42-45%. I would also suggest a higher value for nitrogen, perhaps 7.5%. This would give a molar C/N ratio that is consistent with the Redfield ratio (approximately 6.8 for C:N). Using total particulate matter concentrations of nitrogen to infer the nitrogen content in living phytoplankton (as the report does) is problematic, as much of the particulate matter is non-living deitritus, probably derived from terrestrial sources and seagrasses as well as from phytoplankton. " Giordano et al. (2011) used a similar argument to estimate PN by assuming a carbon/chl-a ratio of 60 gC/g chl-a. based on data from Cloern et al. (1995) for light sufficient, nutrient limited cultures and values from nearby Chesapeake Bay and Redfield stoichiometry (6.8/60=0.11 chl-a/N). They noted that "Our method of estimating PN likely underestimates PN contributed from deitritus, so our value reflects a conservative estimate."

Walt Boynton also commented on this point in his review of Trowbridge's report, stating: "Clarify the 5%, 50% and 6% sentence. What biomass is being referred to here? Is this water column POC? I'm not at all sure doing this (despite EPA guidance) is worthwhile. These ratios really vary widely in my experience. Whatever is decided, this is a weak approach and not much should be inferred from these results."

Buzzards Bay data show that PON correlates with chl levels, at least on an annual average basis. Chl/PON (g/g) ranged roughly between 0.02 and 0.05. Smith and Yamanaka (2007) reported use of maximum ratio of Chl to N parameter set at 0.30 in two models. Chl:N rose from low levels (<0.10) to the max limit in about 14 days as the plankton acclimated. Liu et al. (2007) reported a cluster of samples with chl:PN ratios of 0.001-0.01 at high ammonia levels and a more diffuse cluster between 0.01 and 0.1 at low ammonia levels. They followed analysis of chl:POC done by Cifuentes et al (1988) for the Delaware. They noted that the Chl:PN ratio reflected nitrogen uptake associated with the production of autochthonous POM. Samples associated with lower ammonia levels (less preferred by phytoplankton) all had del-15 N values distinctive from those of the highly polluted samples with very high concentration of ammonium." Hasegawa et al. (2000) reported Chl a:PON ranging from 0.05 to 0.17 in their experiments. Caperon et al. (1976) reported Chl a:PN ranging from 0.023 to 0.081 in their samples.

From the viewpoint of tracking N, chl can be a poor indicator of actual uptake, particularly after nutrient concentrations are rapidly drawn down. This may explain the delay of chl concentrations after seasonal drops in N concentration observed at the GSO dock.



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Caperon et al. (1976) Particulate organic carbon, nitrogen and chlorophyll as measures of phytoplankton and deitritus standing crops in Kaneohe Bay, Oahu, Hawaiian Islands. Pacific Science 30(4):317-327

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Cloern et al. (1995)

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Smith and Yamanaka (2007) Quantitative comparison of photoacclimation models for marine phytoplankton. Ecological Modelling 201: 547-552

Trowbridge, Phillip (2009) Numeric Nutrient Criteria for the Great Bay Estuary. New Hampshire Department of Environmental Services

Valiela, Ivan (1995) Marine Ecological Processes (page 2)

Attachment E

Public Involvement

On July 22, 2002, EPA and the Massachusetts DEP jointly issued a new proposed National Pollutant Discharge Elimination System (NPDES) Permit to Brayton Point Station and opened a public comment period on the permit. The agencies held information meetings on August 5 and 6, 2002, in Somerset, Massachusetts and Bristol, Rhode Island, respectively, to explain the draft permit and answer questions. The agencies held public hearings in Somerset and Bristol on August 26 and 27, 2002, respectively, to accept comments on the draft permit. The comment period, originally scheduled to close on September 4, 2002, was extended to October 4, 2002.

During this 2 1/2 month comment period, EPA received more than 150 comments from elected officials, federal, state and local government agencies, private organizations, individual citizens and the permittee. Careful consideration was given to these comments in development of the final permit.

EPA's response to these comments, published in a document of the same name, specifies which provisions of the draft permit have been changed in the final permit and the reasons for the change, and summarizes and responds to all significant comments on the draft permit submitted during the public comment period. This document can be reviewed at:

www.epa.gov/ne/braytonpoint

For More Information

Call EPA toll free at 888-372-7341 and ask for the following extensions:

Damien Houlihan Engineering Project Manager	81586
Phil Colarusso Biology	81506
Mark Stein Legal	81077
Angela Bonarrigo Community Relations	81034

or call

MA Department of Environmental Protection
David Johnston, Deputy Regional Director
(508) 946-2708

What Impact will this Permit have on Electricity Rates for New England Consumers?

Even after its upgrades, Brayton Point Station's three coal and one oil / gas units will continue to be capable of producing more than 1500 megawatts of electricity at full capacity, while remaining a low cost producer of electricity for New England's energy market.

Using conservative (i.e., worst case) assumptions, the average household, using 500 KWh per month, would see long-term monthly increases of $0.06 to $0.18 in electricity rates as a result of the construction of a closed-cycle cooling system. The short-term impacts of unit outages during the construction period could result in a short-term rate effect of approximately $0.70 per month, but only for nine months.

For More Detailed Information

The final requirements for Brayton Point Station's thermal discharges and cooling water withdrawal are stated in the Final NPDES permit issued to the plant. The permit, along with EPA's response to comments, is available for review at the following locations:

Somerset Public Library
1464 County Street
Somerset, MA

Rogers Free Library
525 Hope Street
Bristol, RI

U.S. EPA Records Center
1 Congress Street
Boston, MA

information is also available for review on the world wide web at:

www.epa.gov/ne/braytonpoint

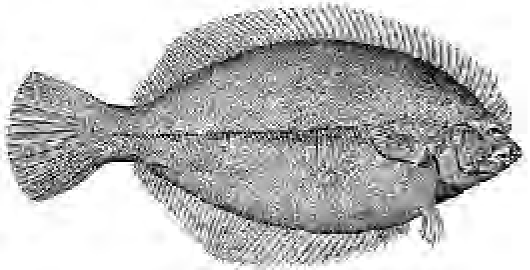
All documents may be downloaded and printed. (Adobe Acrobat Reader is required)

Brayton Point Station Somerset, MA

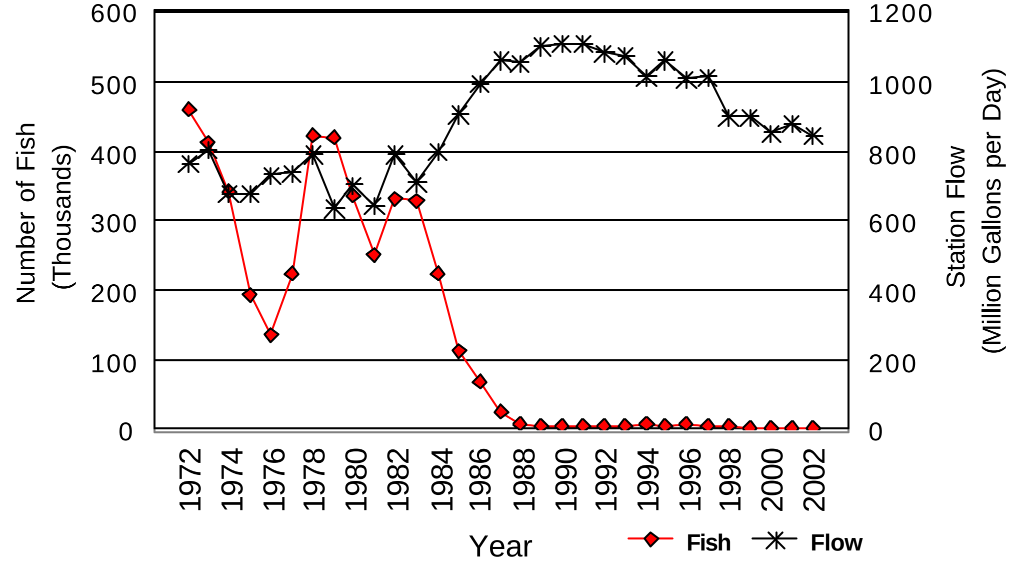
Final National Pollutant Discharge Elimination System (NPDES) Permit October 2003

EPA has developed a final permit for the Brayton Point Station power plant together with the MA Department of Environmental Protection (DEP) and in close coordination with the RI Department of Environmental Management (DEM) to meet requirements of the Clean Water Act. This permit seeks to substantially reduce the facility's impact on Mount Hope Bay. Compliance with this permit will be an essential complement to broader public and private efforts to restore and maintain the health of Mount Hope Bay and the greater Narragansett Bay ecosystem. These other efforts include fishing management, projects to improve sewage treatment, abatement of pollution from combined sewer overflows, and scientific research.

Brayton Point Station is the largest industrial source affecting Mount Hope Bay. Based on the scientific analyses to date, EPA, MA DEP and others have concluded that stronger controls are needed on the power plant's withdrawal of water from the bay and discharge of heated water back to the bay in order to satisfy Clean Water Act standards. These limits will help to protect the bay and give the fishery a chance to recover. The technology exists for Brayton Point Station to both meet the performance standards required by this permit <u>and</u> continue to produce reliable, inexpensive electricity for New England.



Mount Hope Bay
Winter Flounder Abundance and Flow
versus Year



Fish populations declined by more than 87% after 1984 when Brayton Point Station began a 45% increase in cooling water withdrawal from the bay. (It should be noted that the facility's thermal discharge increased by a similar percentage at that time). Despite decreased fishing, many species have shown no signs of recovery. The above graph shows the decline of winter flounder relative to the increase in cooling water use. Similarly dramatic declines can be demonstrated for other fish species as well.

Average annual losses of fish eggs and larvae due to existing cooling water withdrawals at Brayton Point Station include:

- 251 million winter flounder
- 11.8 billion bay anchovy
- 375 million windowpane flounder
- 3.5 billion tautog

Brayton Point Station's Impact on Mount Hope Bay

Located in Mount Hope Bay at the confluence of the Taunton and Lee Rivers, the Brayton Point Station power plant produces about 6% of the electricity consumed in New England. In producing this electricity, however, Brayton Point Station destroys trillions of marine organisms each year and significantly alters the temperature of the bay.

Each day, the station withdraws nearly one billion gallons of water from the bay and circulates it through the facility to condense the steam used to produce electricity. The water is then discharged back to the bay at elevated temperatures of up to 95º Fahrenheit. This "once through" cooling system has contributed to the collapse of the Mount Hope Bay fishery in the following ways:

- **Destroying trillions of organisms.** Water taken from the bay by the facility contains trillions of organisms, including billions of fish eggs and larvae. These organisms are pulled through (or "entrained") in the facility and killed by severe physical and chemical impacts and extreme water temperatures. For example, 251 million winter flounder larvae, 3.5 billion tautog eggs and 375 million windowpane flounder eggs are harmed in an average year.

Cooling water withdrawals also create a water velocity at the intake pipes which traps (or "impinges") many juvenile and mature fish against the intake screens. For example, in 1999, more than 75,000 Atlantic Menhaden were killed during a month long impingement event.

Altogether, trillions of organisms are lost to entrainment and impingement each year, including species of commercial and recreational importance, and forage fish and other organisms integral to the food web.

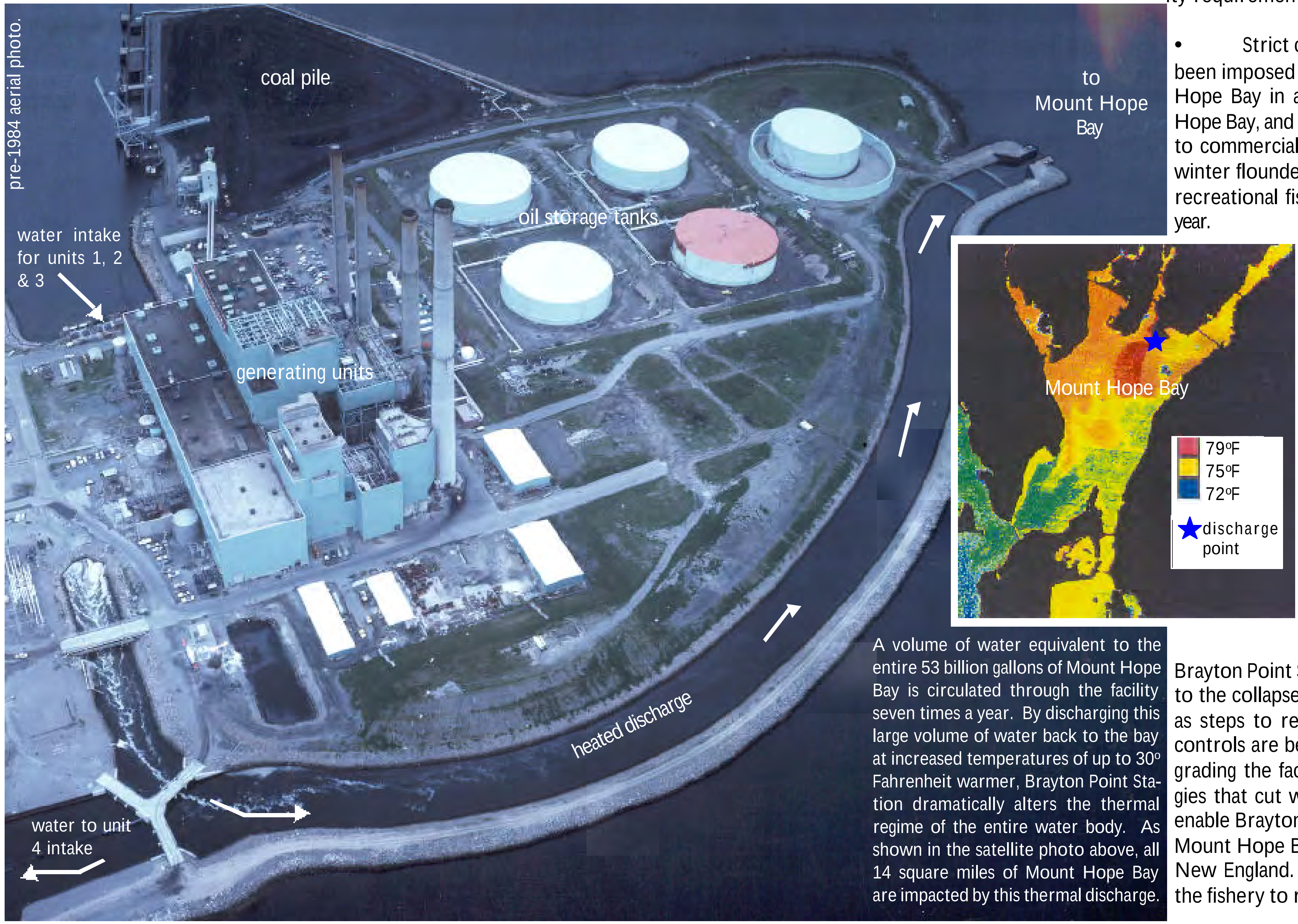
- **Dramatically altering the water temperature in the bay.** As a result of Brayton Point Station discharges of heated water, the temperature in the bay is about 1.5º Fahrenheit greater than other similar water bodies locally. This is a significant temperature difference in a fragile ecosystem. Altering the natural temperature of the bay has degraded the habitat, making areas inhospitable to native fish species, disrupting normal fish migration, and undermining the balanced, indigenous community of fish that should exist in Mount Hope Bay.

What Does EPA's Permit Require?

Consistent with the Clean Water Act, EPA is requiring thermal discharge limits that protect the marine life that should thrive in Mount Hope Bay. In addition, EPA is setting cooling water intake flow limits so that Brayton Point Station's cooling system reflects the best technology available to minimize the facility's adverse environmental impacts. The permit specifically requires Brayton Point Station to:

- Reduce total annual heat discharge to the bay by 96%, from 42 trillion British Thermal Units (BTUs) a year to 1.7 trillion BTUs a year, and
- Reduce water withdrawal from the bay by approximately 94%, from nearly 1 billion gallons a day to 56 million gallons a day. This flow requirement is consistent with well-established closed-cycle cooling technology using wet, mechanical draft cooling towers for generating units 1 through 4.

Compliance with these permit limits will eliminate annual fishery losses by an estimated 94% and improve habitat quality, thereby helping to give the bay an opportunity to recover.



A volume of water equivalent to the entire 53 billion gallons of Mount Hope Bay is circulated through the facility seven times a year. By discharging this large volume of water back to the bay at increased temperatures of up to 30º Fahrenheit warmer, Brayton Point Station dramatically alters the thermal regime of the entire water body. As shown in the satellite photo above, all 14 square miles of Mount Hope Bay are impacted by this thermal discharge.

Protecting Mount Hope Bay

While many federal, state and local efforts have been underway to protect Mount Hope Bay and the larger Narragansett Bay estuary, Brayton Point Station has continued to operate with nearly the same "once-through" cooling technology that was installed almost 40 years ago. Requiring the power plant to meet limits consistent with modern cooling system equipment complements these other efforts, which include:

- Sewage treatment improvements in Fall River, including a $115 million combined sewer overflow abatement program, being implemented to meet state and federal water quality requirements.
- Strict commercial and recreational fishing limits have been imposed in Massachusetts and Rhode Island for Mount Hope Bay in an effort to help restore fish stocks. Mount Hope Bay, and most areas of upper Narragansett Bay, is closed to commercial trawlers. In addition, recreational fishing for winter flounder is closed for 10 months of the year. A small recreational fishing effort is allowed for two months of the year.
- At the regional level, the National Marine Fisheries Service has spent $160 million in the last 10 years buying back fishing vessels and licenses from fishermen in the northeast to reduce fishing pressure on groundfish, including winter flounder. Moreover, additional stringent federal fishing restrictions are expected to be put in place next year.
- Enhancing knowledge about the Narragansett Bay estuary and implementing activities to protect and restore the estuary and its resources through the Narragansett Bay Estuary Program, which has spent approximately $15 million in federal and state matching funds on this effort since 1984.

Brayton Point Station's cooling water system has contributed to the collapse of the fishery and inhibited its recovery, even as steps to reduce fishing pressure and improve pollution controls are being taken to facilitate the bay's recovery. Upgrading the facility's cooling system with modern technologies that cut water withdrawals and thermal discharges will enable Brayton Point Station to reduce its harmful effects on Mount Hope Bay while continuing to generate electricity for New England. These improvements are expected to allow the fishery to recover and restrictions on fishing to be eased.

Attachment F

2004 Lamprey River Dissolved Oxygen Study

A Final Report to

The New Hampshire Estuaries Project

Submitted by

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Durham, NH, 03824

March 31, 2005

This report was funded by a grant from the New Hampshire Estuaries Project, as authorized by the U.S. Environmental Protection Agency pursuant to Section 320 of the Clean Water Act.

NewHampshire
EstuariesProject

Table of Contents

Introduction

As part of the National Estuarine Research Reserve System, the Great Bay System-Wide Monitoring Program (SWMP) produces *in situ* water quality data for four sites in and around Great Bay. In recent years, DataSondes deployed in the upper Lamprey River have documented dissolved oxygen concentrations that do not meet federal standards during a significant portion of the summer and fall period. These low oxygen concentrations, if they persist may have a negative effect on benthic and pelagic organisms in the river and will necessitate management action to improve water quality.

Project Goals and Objectives

UNH completed this project under contract to the NH Estuaries Project (Project ID #04-M-2; CE-991711-06 and CE-991711-08). The project goals and objectives per the contract were to carry out surveys of the Lamprey River during the summer and fall to:

(1) confirm the accuracy of the DataSonde data;

(2) assess whether the DataSonde data are generally representative of the upper reaches of the river; and

(3) gain insight into the potential causes of low oxygen in the bottom waters of the river.

The final work product was agreed to be a summary analysis of survey data and Excel data files containing survey data, relevant DataSonde records along with appropriate meta-data for these data.

Methods

DataSonde deployments followed the procedures generally prescribed by the National Estuarine Research Reserve Central Data Management Office (CDMO) and detailed in Small et al. (2003).

Briefly, YSI 6600 DataSondes are programmed to obtain measurements of specific conductivity, salinity, dissolved oxygen, percent saturation, pH, temperature, water level, and turbidity every half-hour. The instruments are deployed continuously during ice-free seasons, except for brief periods when they are removed for cleaning, maintenance and recalibration. Pre and post-deployment calibrations are performed using the diagnostics menu of the YSI Ecowatch program and QA/QC procedures developed by NERR Research Coordinators and YSI engineers. VWR conductivity and pH standards are used for calibration. YSI formazin is used to calibrate turbidity probes.

DataSondes are deployed approximately one meter from the bottom and recovered for data download every 2-4 weeks depending upon the time of year. Files are first examined and graphed using Ecowatch software. Missing and/or anomalous data are noted. Files are then transferred to a Macintosh computer and opened in Excel software and edited. Missing data due to routine YSI maintenance and probe failure or communication errors are inserted into the spreadsheet. Edited files are merged to contain one full month of data. Files are verified by means of CDMO Excel macros. The CDMO cdmomac3.xls macro allows the user to automatically format column widths to the correct number decimal places based on the YSI sensor specifications. It also allows the user to QA/QC each data logger generated file for missing data points, fill all cells that do not contain data with periods, and find all data points that fall outside the range of what the DataSonde is designed to measure (outliers). The CDMO import.xls macro will allow PC users with 30-minute data to automatically create a monthly Excel file from a two-week deployment and insert periods for missing data. Edited files are merged to contain one full month of data. In addition, in November 1999 a graphing capability was added to this macro allowing users to produce single parameter and missing point graphs on a monthly basis. All files are graphed in Excel and examined in order that anomalous data points can be identified and removed.

Surveys were carried out by small boat on four days in the summer and fall of 2004; 16 July, 29 July, 12 August and 26 October. Originally the surveys were designed to be in response to low dissolved oxygen events observed using near real-time telemetry; however, telemetry for this site could not be established during 2004. As a result, the surveys dates were chosen based on past experience of the time and tidal stages for which low dissolved oxygen, if present, would be expected.

During each survey, sampling was conducted at ~15 stations in the upper basin, ~7 in the tidal river between the basin and Great Bay, and between 2 and 3 times at the DataSonde location (Figure 1). At each station, vertical profiles of specific conductivity, salinity, dissolved oxygen, percent saturation, pH, and temperature were taken using a YSI 6600 DataSonde in approximately 0.5 meter vertical increments. The profiling DataSonde was calibrated on the day of the survey following the CDMO methods outlined above. In addition, location coordinates were obtained using a Magellan Sport Trac hand-held GPS.

Comparisons between DataSonde and survey data were made by using the DataSonde data point (collected every 30 minutes) taken closest to the survey profile and by using the survey profile sample depth closest to the depth of the DataSonde. Maximum differences between the data used in the comparisons was thus, 15 minutes in time and 0.5 meters in depth.

Results and Discussion

The lack of near real-time telemetry during the study period resulted in a shift in study design from the proposed goal of one detailed survey (~22 stations) and four additional surveys (~10 stations each) to four detailed surveys (~22 stations each). This resulted in >88 station profiles as compared to the >62 proposed.

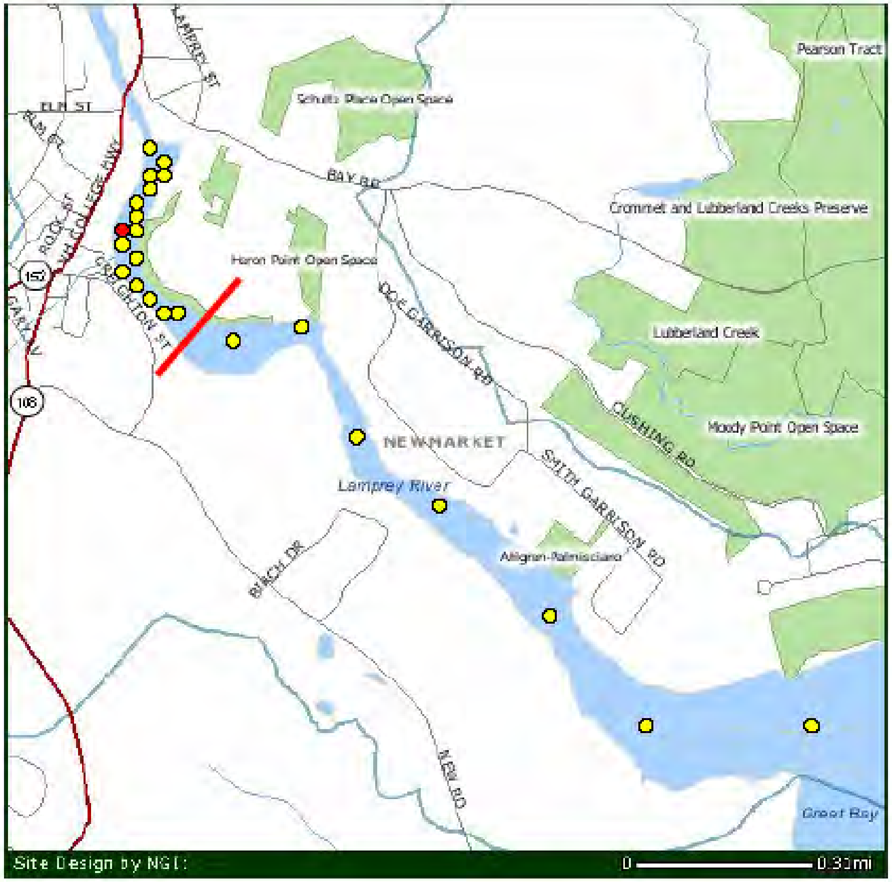
Comparisons of dissolved oxygen saturation and salinity data from the DataSondes with vertical profiles from the four surveys (Figures 2-5) were used to assess the accuracy and reliability of the DataSonde data. For the 16 July, 12 August and 26 October surveys (Figures 2, 3 & 5) the survey profile data was consistent with the DataSonde data. On 29 July, the survey profile data showed higher oxygen levels than the corresponding DataSonde data. The data also displayed a consistent trend of decreasing oxygen concentration/percent saturation with salinity (and depth; see data files).

Figure 1

Lamprey River DO Study

Approximate Station Locations

- ● **Sonde Station**
- ○ **Cast Stations**
- / Arbitrary Division between upper and lower basins



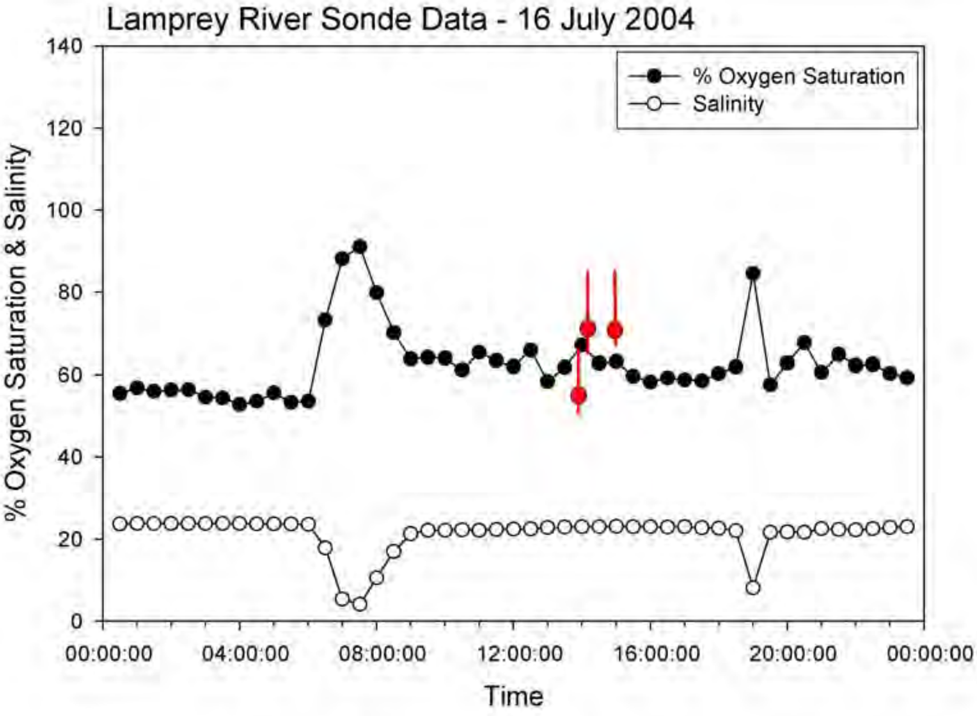


Figure 2 –Oxygen Saturation (solid dots) and Salinity (open dots) data from the Lamprey River DataSonde on 16 July. Red dots (DataSonde depth) and line (range) for vertical casts taken adjacent to the DataSonde during spatial survey on the same date.

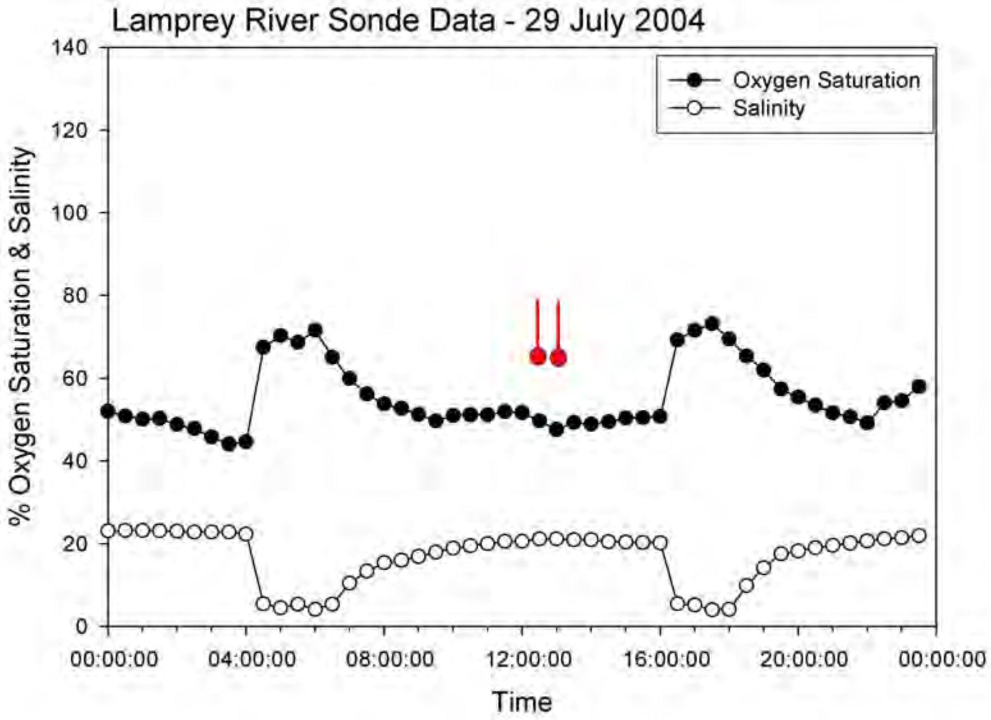


Figure 3 –Oxygen Saturation (solid dots) and Salinity (open dots) data from the Lamprey River DataSonde on 29 July. Red dots (DataSonde depth) and line (range) for vertical casts taken adjacent to the DataSonde during spatial survey on the same date.

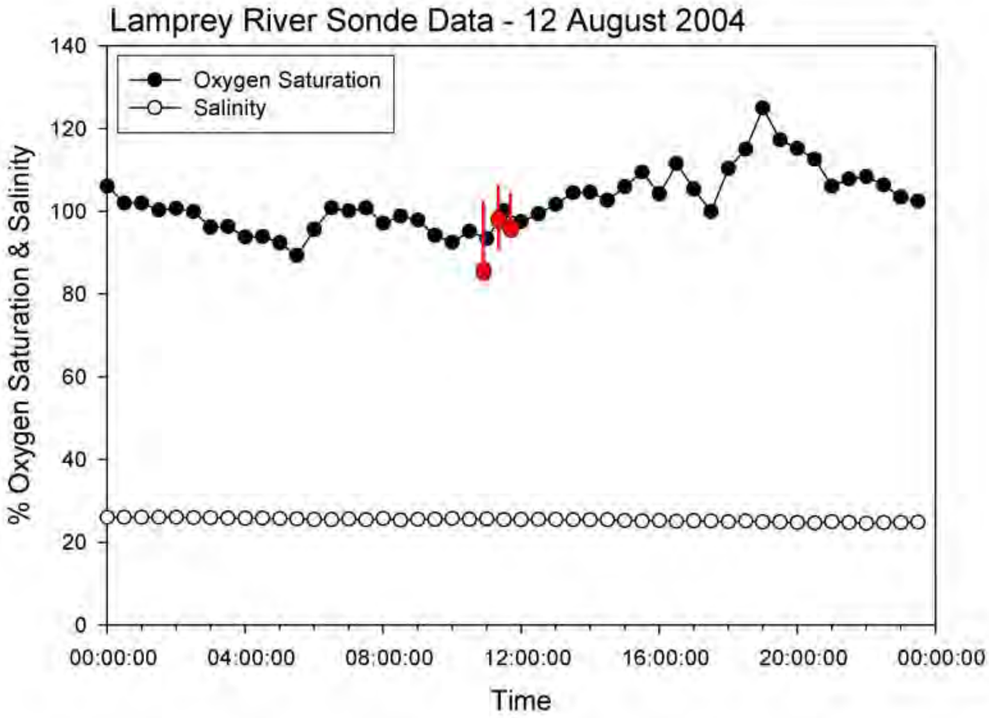


Figure 4 –Oxygen Saturation (solid dots) and Salinity (open dots) data from the Lamprey River DataSonde on 12 August. Red dots (DataSonde depth) and line (range) for vertical casts taken adjacent to the DataSonde during spatial survey on the same date.