Impact of Rochester and Dover WWTF Nitrogen Load Reductions on Water Quality of the Upper Piscataqua River

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Abstract

This report provides an analysis of the nitrogen reductions implemented by the Dover and Rochester WWTFs and subsequently observed water quality response in the Upper Piscataqua River. The system loading and response data reflect that a substantial reduction in total nitrogen concentration occurred due in part to these nitrogen reductions. The effect of this reduction on "nutrient-related" water quality parameters (*e.g.*, chlorophyll-a, transparency, and dissolved oxygen) was evaluated. Comparisons of chlorophyll-a concentrations and light extinction coefficients before and after nutrient reduction indicated that there was no change in these parameters. This information indicates that nitrogen control has a negligible effect on the algal dynamics of the Upper Piscataqua River (consistent with previous evaluations of long term phytoplankton trends conducted by PREP). An evaluation of the effects of nitrogen reduction on DO in the Upper Piscataqua River was complicated by the lack of simultaneous monitoring data for the two monitoring stations during the period of elevated nitrogen concentrations. Data for the post-reduction period illustrate the effects of other confounding factors on DO variability through the monitoring period. This variability was compared with observations from the earlier period and assessed based on the occurrence and magnitude of minimum dissolved oxygen, diel DO range, and dissolved oxygen deficit at each monitoring station. Based on this evaluation, the pre-TN reduction metrics for minimum DO and diel range are essentially identical to the metrics observed after a 50% reduction in TN concentration. The metric for DO deficit showed mixed results for the two stations. The results for Station UPR4 suggest a potential effect, but this effect appears to be associated with tidal cycles. Consequently, the effect of nitrogen reduction on DO in this system appears to be negligible, as would be expected given the lack of change in algal dynamics.

Background

Since 2009, there has been considerable debate regarding the degree to which changing nitrogen concentrations/loads have impacted the ecology of the Great Bay estuary. EPA, in particular, has asserted that DO, chlorophyll-a, and transparency levels are negatively affected by TN concentration, and the adverse TN impact on water clarity, algal growth and DO has been significant. Linear regressions developed by DES indicated that major increases in minimum DO (~ 1 mg/l) would result from reducing TN levels from 0.5 mg/l to 0.25 mg/l, as implied by the regression analysis presented below (Figure 1)[1] from an early attempt to derive numeric nutrient criteria for the estuary.

[1] NHDES, 2009. Numeric Nutrient Criteria for the Great Bay Estuary. Available at http://des.nh.gov/organization/divisions/water/wmb/wqs/documents/20090610_estuary_criteria.pdf.

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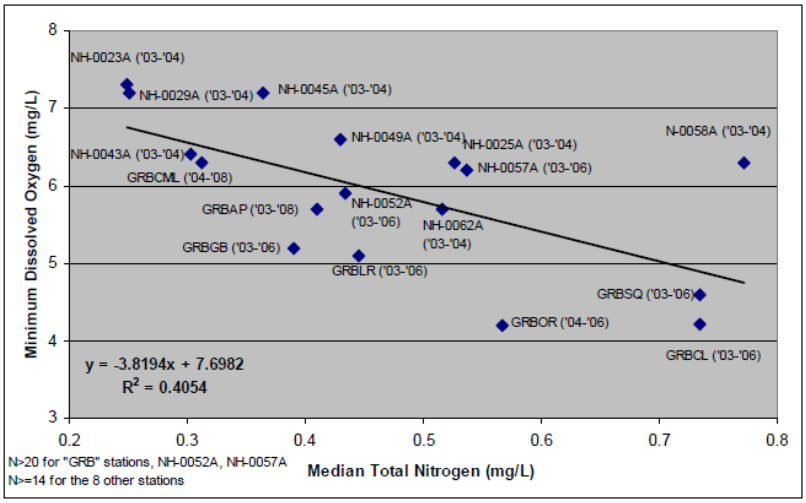

Figure 1: NHDES 2009 Nutrient Criteria Regression Analysis

In February 2014, a peer review panel of four nationally-recognized water quality experts determined that adverse ecological impacts from TN levels in Great Bay Estuary have not been demonstrated.[2] On June 30, 2014, the peer reviewers issued follow-up comments regarding additional water quality analyses that would improve the understanding of this system and provide confirmation regarding the efficacy of TN reduction. Given the scheduled voluntary nitrogen reductions planned by the Cities of Dover and Rochester, the peer reviewers indicated the following:

> **Dr. Reckhow**: This is a great opportunity for a before-after assessment to observe the effects of TN load reductions. You could use the model to evaluate the before and after conditions. Then you could design a monitoring program to reflect those results.
>
> **Dr. Bierman**: The issue is data to calibrate. You need to analyze different scenarios on the effect of increases/decreases of TN. You need to get multiple datasets along load response curve. Agrees with JCH that this would confirm whether TN levels are controlling phytoplankton growth/transparency in system or whether other factors (detention time/other sources) are controlling system responses.

Consistent with these recommendations, the Great Bay Municipal Coalition documented Rochester and Dover WWTF effluent improvements and conducted concurrent system monitoring (including the deployment of data sondes) to determine the effects, if any, of nitrogen reductions on Great Bay Estuary chlorophyll-a, DO, and light attenuation.

The assessment presented below is based on the general conceptual model for the effect of nutrients on estuarine water quality. The primary effect of increased nutrient loads is to

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

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stimulate plant growth (*i.e*., phytoplankton) as measured by chlorophyll-a concentration. As the level of algal biomass increases, water clarity (measured as a light extinction coefficient) decreases. In systems where macrophytic plant growth (*i.e.*, eelgrass) is important, increasing light extinction may adversely influence such growth. Increasing algal biomass also influences DO. Diel swings in DO concentration are directly influenced by algal biomass. As the algal biomass increases, the potential for large swings about the mean DO level increase due to photosynthesis and respiration. However, photosynthetic increases in DO concentration are mediated by available light. When algal growth becomes excessive, the settling of algae can create an increase in sediment oxygen demand, which would depress the overall average DO. This depression in the overall average DO would be exemplified by an increase in DO deficit. Superimposed upon these effects are the other factors which influence DO concentration and overall deficit (*e.g.*, biochemical oxygen demand, reaeration, tidal exchange, stratification).

Based on this conceptual model, we would expect to see a decrease in overall algal biomass with a reduction in the overall level of nutrients entering the estuary if nutrient load is the controlling factor on algal growth. However, if other factors (*e.g.*, retention time, light transmission) control algal growth, the reduction in algal biomass would be muted or non-existent. Assuming that decreased nutrient load has an overall effect on algal biomass, we would also expect an improvement in water clarity, a decrease in diel DO swing, and a decrease in the overall deficit. Prior analyses prepared by HDR-HydroQual and Dr. Steven Chapra indicated that TN was not controlling phytoplankton growth, because the level of phytoplankton growth in the system was far lower (over 10 times less) than what could be supported by the available inorganic nitrogen in the system.[3] This "real world" test would, through direct observation, confirm whether or not this assessment was accurate.

Wastewater Facility TN Reductions

The City of Rochester began upgrading its WWTF in the summer of 2013. The effects of the enhanced nitrogen removal and concurrent load reduction from the city's Waste Management landfill are clear from the quarterly effluent TN data (Fig. 2). From Jan. 2010-May 2013, the effluent TN averaged 36.5 mg/L and frequently ranged between 40-45 mg/L. Thereafter, additional facility upgrades were implemented to enhance TN removal resulting in nearly fourfold effluent concentration reductions. From July 2013-October 2015, the effluent averaged 10.4 mg/L TN. The most current summer performance (July-October 2015) indicates TN levels approaching 8 mg/L, as a seasonal average, are attainable.

[3] See, for example, the February 27, 2013 Declaration of Steven C. Chapra in the Town of Newmarket NPDES Appeal No. 12-05

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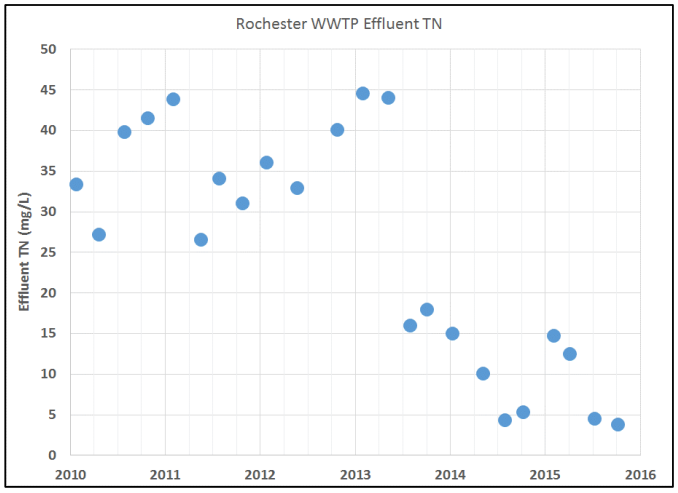

Figure 2: Rochester WWTF Effluent TN

In the summer of 2015, the City of Dover also upgraded its WWTF to enhance TN removal. The improved nitrogen removal is evident in the facility's post-August 2015 weekly effluent TN concentration data (Fig. 3).

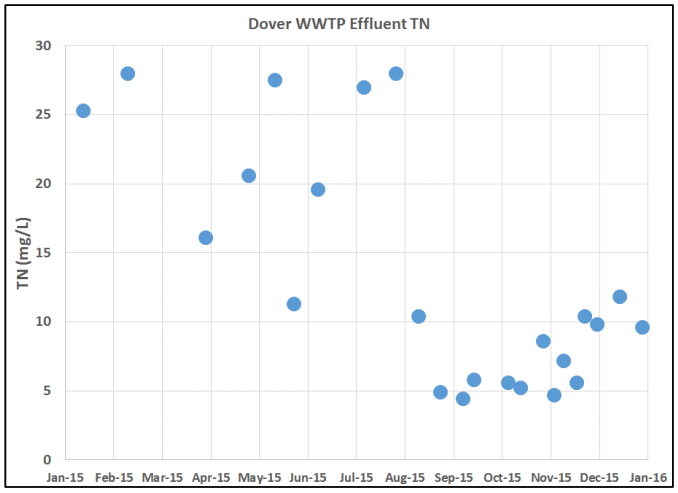

Figure 3: Dover WWTF Effluent TN

From January to July 2015, the effluent TN averaged 22.6 mg/L. After facility upgrades were fully operational, the September-October effluent TN averaged under 6 mg/L, a nearly fourfold reduction. The resulting TN reductions are also evident from the facilities' effluent loading data (Fig. 4).

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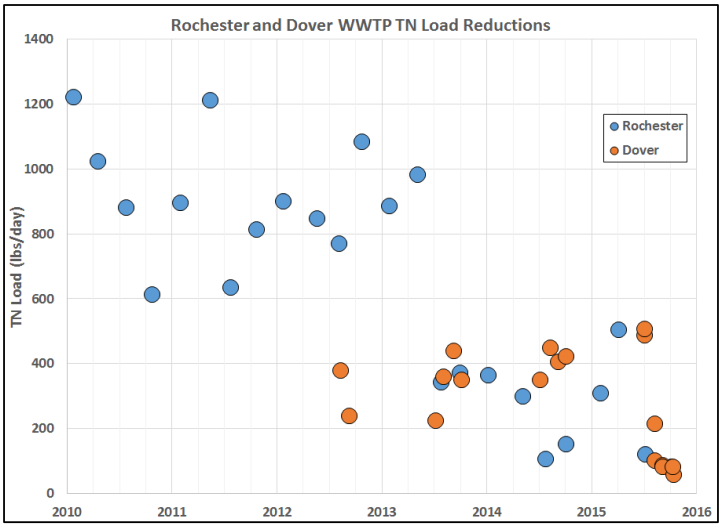

Figure 4: Rochester and Dover WWTF TN Loads

When only summer data (July-Oct.) are considered, the combined Rochester and Dover WWTF TN loads have substantially declined from an average of 1,230 lbs/day in 2012 to about 300 lbs/day in 2015. A greater average reduction would be expected in 2016 as the 2015 data include both pre-upgrade and post-upgrade Dover data. The post-upgrade Dover data were averaged to project 2016 conditions with an expected combined TN load of 200 lbs/day, an 84% reduction compared to 2012 data (Fig. 5).

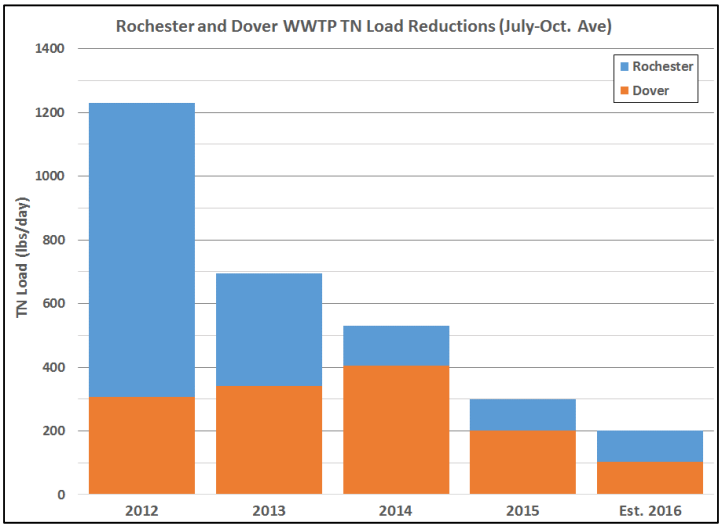

Figure 5: Rochester and Dover WWTF TN Load Reductions

These reductions have led to a significant reduction in the instream concentration of TN in the Piscataqua River.

Effect of TN Reductions on Upper Piscataqua River Water Quality

In order to evaluate the efficacy of TN reduction for the system, water quality data were evaluated for TN and several key parameters that could be adversely influenced by existing nutrient levels. There are several monitoring stations on the Piscataqua River where grab sample data have been collected over the period of time encompassing the WWTP load reductions. Long term monitoring data are available for Station NH-0057A, with grab samples collected once per month at low and high tide. More intensive data collection was initiated in 2015 at Stations UPR4 and UPR6A, with duplicate grab samples collected in the afternoon. In addition, data sondes have been deployed at Stations UPR4, UPR6, and UPR6A to monitor diel dissolved oxygen changes and other water quality characteristics. (See Figure 6 for station locations). All of these stations are located on the Upper Piscataqua River. Station UPR4 is approximately 0.5 km upstream of NH-0057A, which is located adjacent to the Dover WWTP outfall. Station UPR6 is located approximately 0.5 km downstream of NH-0057A. Station UPR6A is located 0.09 km (300 feet) west of UPR6.

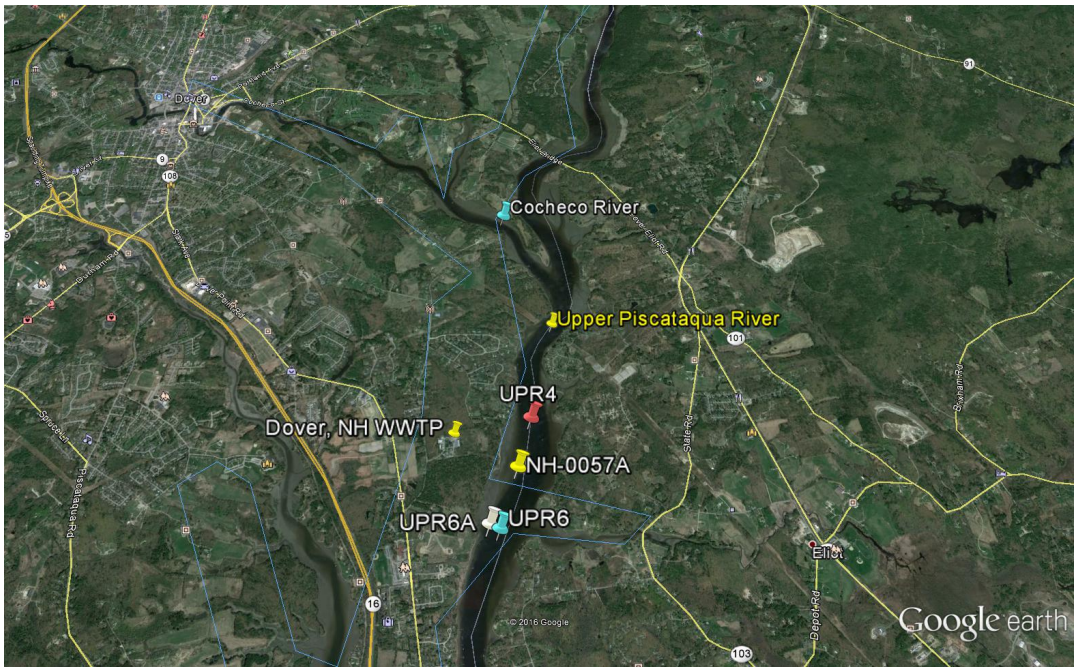

Figure 6: Monitoring Station Map

Average August-September TN data monitored at Station NH-0057A, UPR4, and UPR6A reflect the substantial nutrient reductions recorded at the Dover and Rochester WWTFs. When all of the available TN data are plotted for the August-September timeframe, it is apparent that the seasonal TN concentration has decreased from approximately 0.5 mg/L (2010 – 2012; prior to TN reductions at Rochester) to about 0.25 mg/L (2015; post-TN reduction at Rochester and implementation of TN reduction at Dover). (See, Figure 7). Although not shown in Figure 7, the results for 2013 and 2014 at Station NH-0057A show TN concentrations averaging 0.35 mg/L for this seasonal

period. From the figure below, it is noted that post-upgrade TN readings at UPR4 and UPR6A are, for all practical purposed, identical. NH-0057A is located between these two stations. Therefore, plotting the data for these stations on the same chart, for comparison purposes, is considered reasonable (i.e., differences in readings are not a function of the location of the sampling).

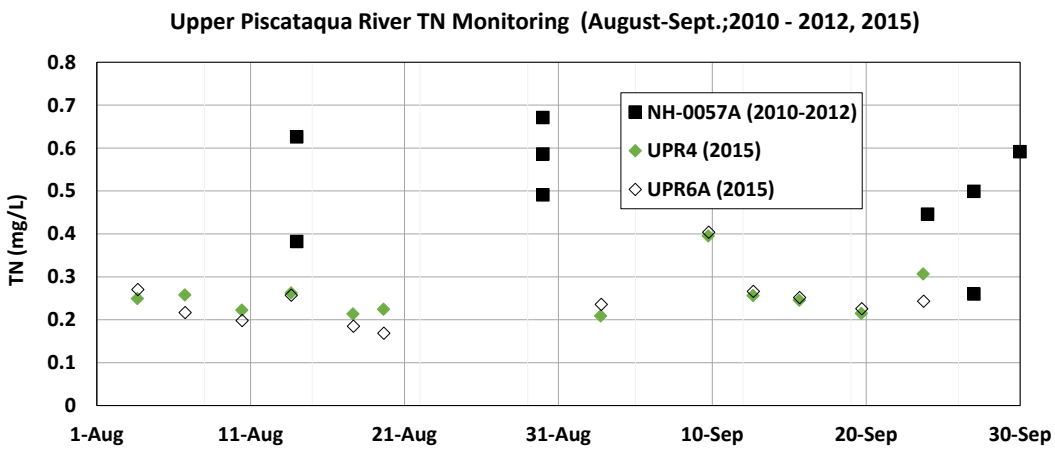

Figure 7: Ambient TN Concentration

Nitrogen Impacts on "Related" Water Quality Parameters

As noted earlier, the Rochester and Dover WWTF nitrogen reductions were implemented to test the hypothesized nitrogen-induced eutrophication and light attenuation impairments. Given such significant reductions in (primarily inorganic) nitrogen during the algal growing season, some improvement in chlorophyll-a would be expected if the hypothesized causal relationship between nutrients and chlorophyll-a accurately reflected this system's dynamics.

- Chlorophyll-a

All of the available chlorophyll-a monitoring data since 2010 are illustrated in Figure 8. The data collected at NH-0057A include monthly grab samples at high and low tide (*e.g.*, two samples on one day) over the entire monitoring period (April – December) from 2010 through 2014. The entire period is presented to illustrate variability over each year. The data collected at UPR4 and UPR6A are semi-weekly grab samples, collected in the afternoon, in August and September 2015. Duplicate samples were collected during each sampling event and the duplicate results were averaged and presented in Figure 8.

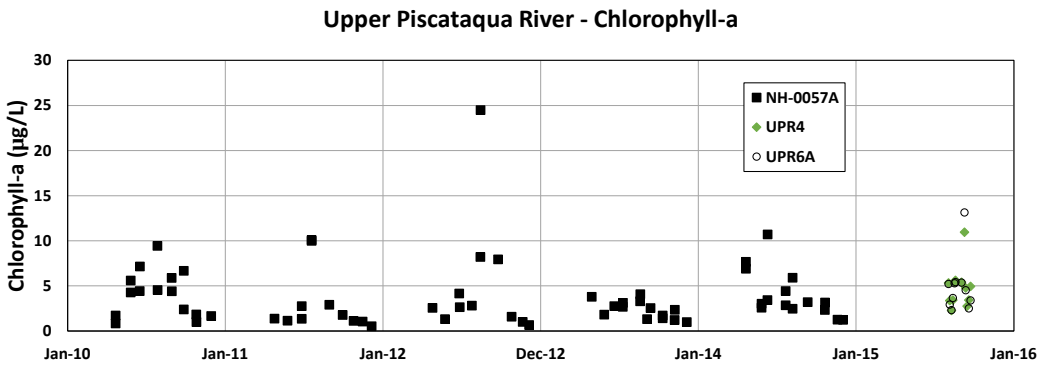

Figure 8: Ambient Chlorophyll-a Concentration

Figure 8 illustrates several aspects concerning chlorophyll-a levels in the Upper Piscataqua River. Over the period from 2010 through 2014, chlorophyll-a levels have fluctuated without showing any particular long term trend. Concentrations tend to be low at the beginning and end of the monitoring season, with higher concentrations occurring between June and September, with occasional peak concentrations which may represent blooms. Following TN load reductions at Rochester in 2013, chlorophyll-a concentrations were lower than in previous years. However, in 2014 chlorophyll-a concentrations increased slightly although the ambient TN concentration was nearly identical to the previous year. Aside from one very elevated concentration observed in August 2012, the profiles look nearly identical for each year. [4]

A closer review of the August-September 2015 sampling results is presented in Figure 9. This figure shows that the samples collected at the two stations are essentially identical, as might be expected for two stations that are in relatively close proximity on the Piscataqua River.

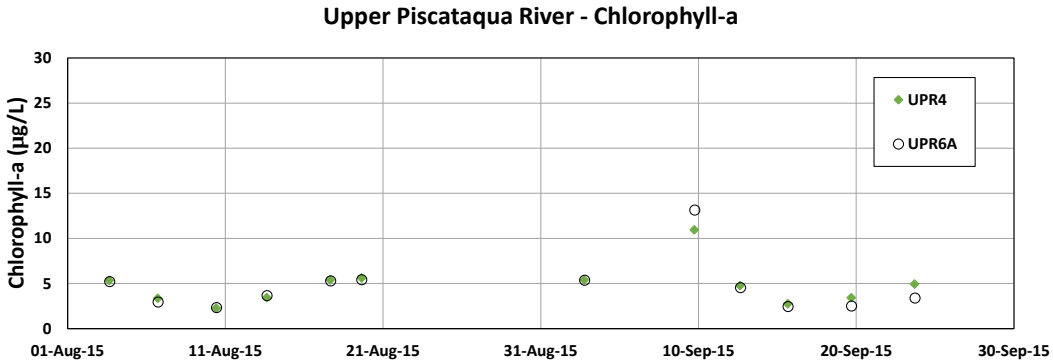

Figure 9: Ambient Chlorophyll-a Concentrations in 2015

[4] The same pattern of algal growth showing no significant change in chlorophyll a level in response to changing TN concentrations has been documented by PREP at the Adams Point sampling location between Great Bay and Little Bay. (See, 2013 State of the Estuaries Report at 17)

The August-September chlorophyll-a concentrations for 2010-2012 (Station NH-0057A) were compared with the chlorophyll-a concentrations observed in 2015 (UPR4, UPR6A), as illustrated in Figure 10, using analysis of variance. This analysis shows that there are no significant differences among the data sets (P = 0.34). This analysis was repeated, removing the apparent outliers (the peak values observed at each station. These peaks may represent short-term blooms. They were removed to see if the resulting decrease in variance affects the statistical comparison of the two data sets). The analysis again indicated no significant difference among the stations (P = 0.37).

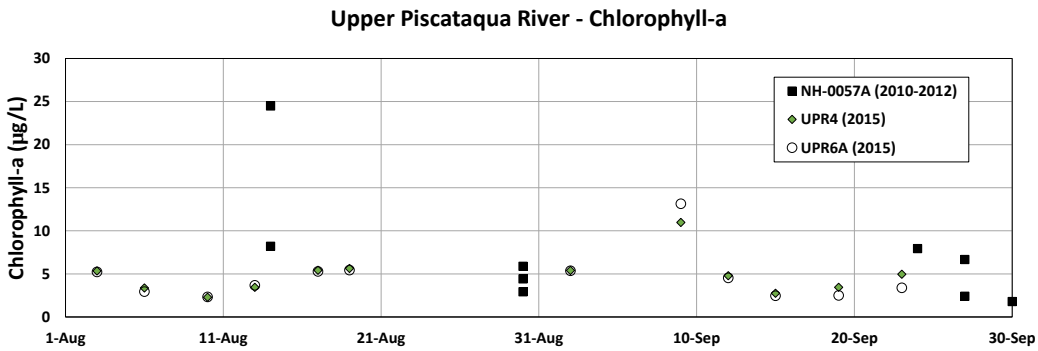

Figure 10: Ambient Chlorophyll-a Concentrations for August-September before and after TN Load Reduction

Based on the available data, the reduction in ambient TN concentration did not result in any material decrease in the ambient concentration of chlorophyll-a in the Upper Piscataqua River, as had been predicted by earlier technical evaluations.

- Light Extinction

Light attenuation has been measured routinely at stations throughout the Great Bay Estuary, including the Upper Piscataqua River. The methods used to collect light attenuation data, reported as a light extinction coefficient (K_d, 1/meter) were described in a field sampling and monitoring report (December 31, 2015) prepared by Jones et al. (University of New Hampshire, Jackson Estuarine Laboratory).

> Water column light attenuation readings were taken at each station during the Slack Tide Studies. Measurements of underwater photosynthetically active radiation (PAR) were made at up to eight depths, station water depth permitting, using a Licor LI-193 model Spherical Underwater Quantum Sensor. To normalize for fluctuations of solar irradiance during a light cast, a Licor LI-190 was used to record in-air PAR for each underwater measurement. Data were recorded with a Licor LI-1400 Datalogger.
>
> Results for these data were given as diffuse attenuation coefficients (Kd). Values for Kd were calculated as the slope of the line of measurement depth versus the natural log of surface irradiance-corrected underwater PAR measurements. Any cast with data resulting in regression $R^2 < 0.95$ are

considered bad and are not reported. Duplicate light casts were occasionally performed to assess reproducibility.

The methods used in surveys prior to 2015 are believed to be similar, since these data were also collected by the Jackson Estuarine Laboratory.

All of the available extinction coefficient data since 2010 are illustrated in Figure 11. The data collected at NH-0057A include monthly monitoring results with one or two samples collected on a given day over the entire monitoring period (April – December) from 2010 through 2014. If duplicate measurements were made, these results were averaged. As with the chlorophyll-a data, the entire period is presented to illustrate variability over the months. The data collected at UPR4 and UPR6A are semi-weekly monitoring results, collected during slack tide, in August and September 2015.

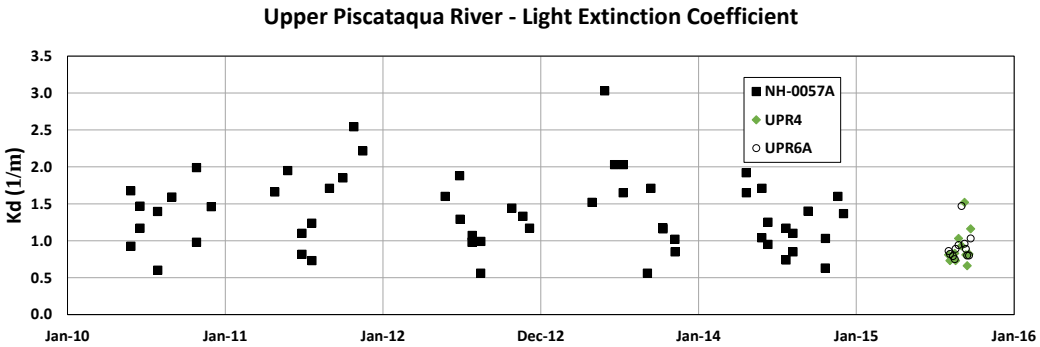

Figure 11: Ambient Light Extinction Coefficients

Figure 11 illustrates several aspects concerning light extinction levels in the Upper Piscataqua River. Over the period from 2010 through 2014, light extinction levels tend to be greater at the beginning and end of the monitoring period than in the middle (July-August). Following TN load reductions at Rochester in 2013, light extinction coefficients appear to be much lower than in previous years. However, this is an artifact attributed to the seasonality of the 2015 survey.

The August-September light extinction levels for 2010-2014 (Station NH-0057A) were compared with the K_d levels observed in 2015 (UPR4, UPR6A), as illustrated in Figure 12. The figure shows that the extinction coefficients prior to TN reduction and after initiation of TN reduction exhibited a similar range as the 2015 measurements.

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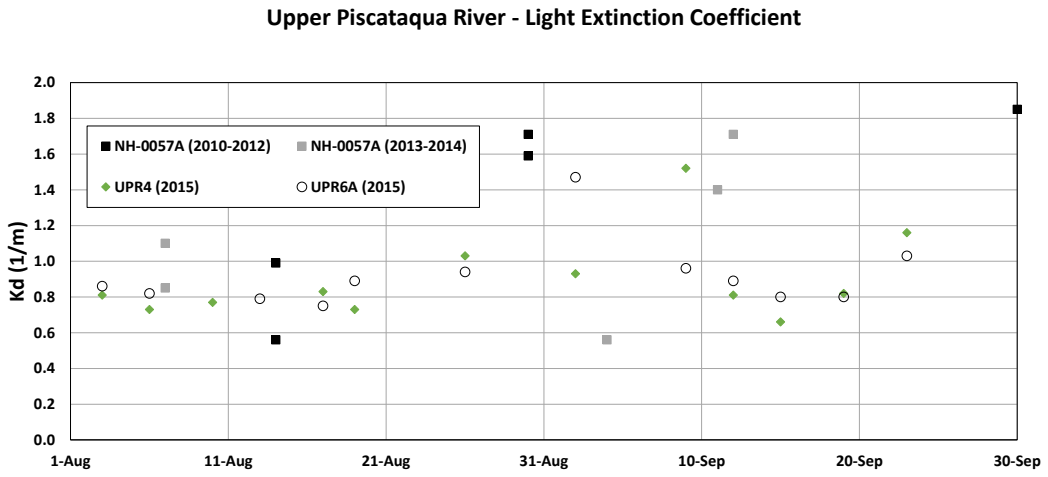

Figure 12: August-September Light Extinction Coefficients before and after TN Reduction

This observation may be explained by basic factors controlling light extinction in the Great Bay Estuary. Morrison et al. (2008)[5] evaluated the factors controlling light extinction in the estuary and determined that non-algal particulate (NAP) turbidity and colored dissolved organic matter (CDOM) accounted for 56% of the observed attenuation, while phytoplankton (measured as chlorophyll-a) accounted for only 12%. (Figure 13). CDOM and NAP are associated with runoff from the watershed. [6]

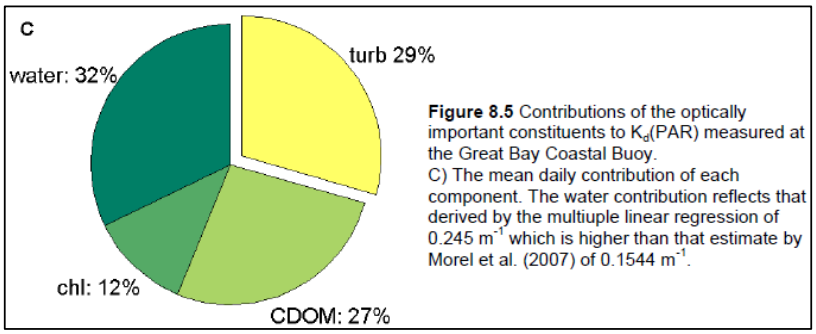

Figure 13: Relative Contributions of Light Attenuating Substances

CDOM levels are naturally high in all of the tributaries for this system, as exemplified by this flume discharge located on the Salmon Falls River, a tributary to the Upper

[5] Morrison, J.R., T.K. Gregory, S. Pe'eri, W. McDowell & P. Trowbridge, 2008. Using Moored Arrays and Hyperspectral Aerial Imagery to Develop Nutrient Criteria for New Hampshire's Estuaries.

[6] The Piscataqua River has, in general, poorer light transmission readings (i.e., higher K_d) than occurs in Great Bay, even though algal growth is lower, in general, in the Piscataqua River. Prior analyses by HDR-Hydroqual, utilizing the relationships developed by Morrison indicated that only a very small fraction (~5%) of measured light transmission occurring in the Upper Piscataqua River was due to algal particulates.

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Piscataqua River (Fig. 14). A recent peer-reviewed study also concluded that CDOM in Gulf of Maine watersheds is correlated to increased tributary discharge.[7]

Figure 14: CDOM in Salmon Falls River

When the K_d measurements are plotted along with the USGS-metered flow for the Cocheco River (Station 01072800; near Rochester, NH), this relationship becomes more apparent (Figure 15). For periods when the river flows are low and steady (*e.g.*, July 2010-2012; August-September, 2015) the extinction coefficients are equivalent before and after the TN load reductions. The increases in K_d measurements appear to be associated with increases in river flow, which convey higher tributary loads of CDOM and NAP into the system. Moreover, since the chlorophyll-a concentrations are not materially different between pre-TN load reduction (2010 – 2012) and post-TN load reduction (2015), the observed reduction in TN concentration has no material influence on light extinction.

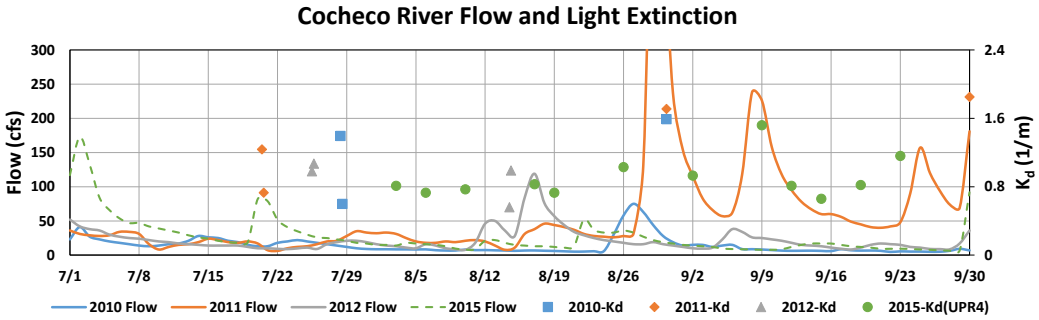

Figure 15: Relation between Light Extinction and Cocheco River Flow

[7] Balch, W., T. Huntington, G. Aiken, D. Drapeau, B. Bowler, L. Lubelczyk, and K. Butler (2016), Toward a quantitative and empirical dissolved organic carbon budget for the Gulf of Maine, a semienclosed shelf sea, Global Biogeochem. Cycles, 30, 268–292, doi:10.1002/2015GB005332.

- Dissolved Oxygen

Data sondes were deployed in the Upper Piscataqua River in 2012 and 2015, before and after the major TN reductions occurred. The sondes were installed approximately one meter off the bottom. These sondes continuously recorded measurements of temperature, salinity, DO, percent DO saturation, pH, chlorophyll-a[8], and meter depth, in 15-minute intervals. The results of the 2012 survey have been QA/QC'd. The results for the 2015 survey have not been similarly finalized. Consequently, the assessments presented below should be considered preliminary until the 2015 results are finalized.

It should be noted that the relationship between TN reduction and system DO is far more indirect than phytoplankton growth or transparency, given the increased number of non-nutrient factors that impact DO measurements in this system on a day to day basis. These non-nutrient factors include (1) tributary inputs (DO, organic loadings) (2) tidal variation (3) stratification (4) temperature, (5) salinity (6) DO inputs from Gulf of Maine and (7) organic loadings from the WWTPs. Given the lack on algal response previously evaluated, it would seem apparent that no material change in DO could be attributed to the load reductions achieved. Nonetheless, assessment of the DO measurements may provide further insight on the dynamics of this system.

- Survey Results

<u>DO Concentration</u>

The 2012 survey included a sonde installation at UPR4 from July 26 to August 8. The sonde was then redeployed at UPR6 from August 8 to August 21. The 2015 survey included installations of individual sondes at UPR4 and UPR6A from August 3 to August 26. The sondes were retrieved on August 26 for data download and maintenance. They were then reinstalled from September 2 to September 24. The raw data for DO concentration are illustrated in Figure 16 for both of these surveys.

[8] The sonde measurements of chlorophyll-a include wide swings in discrete measurements that cannot be real. Consequently, these results were not considered reliable and were not evaluated for this report.

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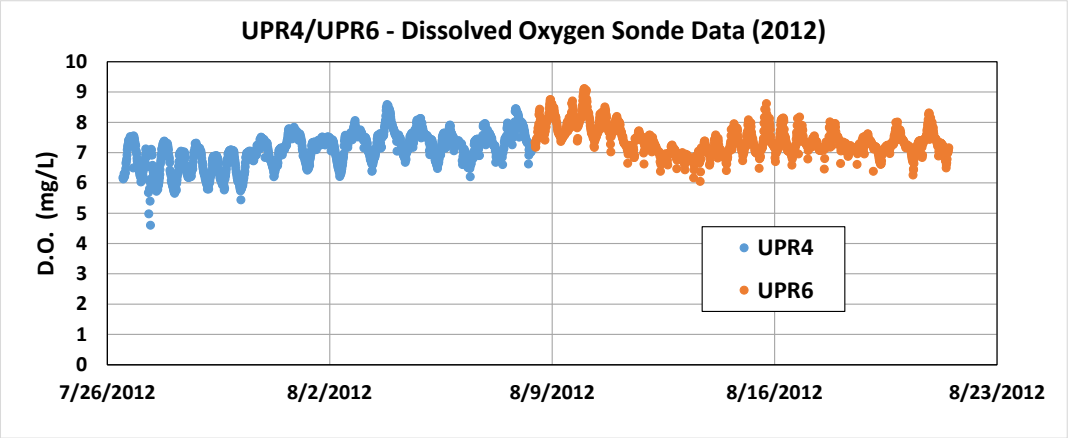
(a)



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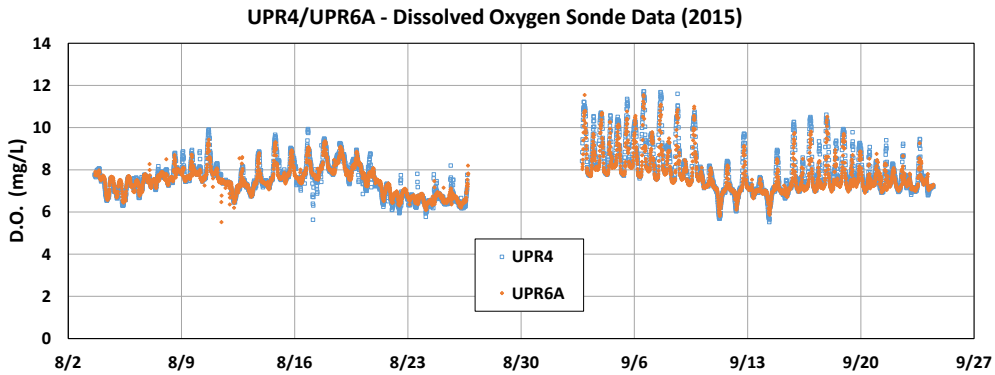

Figure 16: Sonde Measurement of DO in (a) 2012 and (b) 2015

The temporal data presented in Figure 16 clearly illustrates the diel pattern of DO augmentation followed by DO depression due to photosynthesis/respiration and tidal oscillations which bring in generally higher DO concentrations from the Gulf of Maine. In addition, a longer-term cyclical pattern of increasing and decreasing DO concentration is evident and is likely due to the spring-neap tidal cycle. These patterns are clearly visible in 2012 due to the sequential deployment of the sonde at the two stations. However, the results for 2015 are obscured because the data for the two stations overlap. In order to make the diurnal variability more apparent, the data were replotted as discrete day observations in Figure 17.

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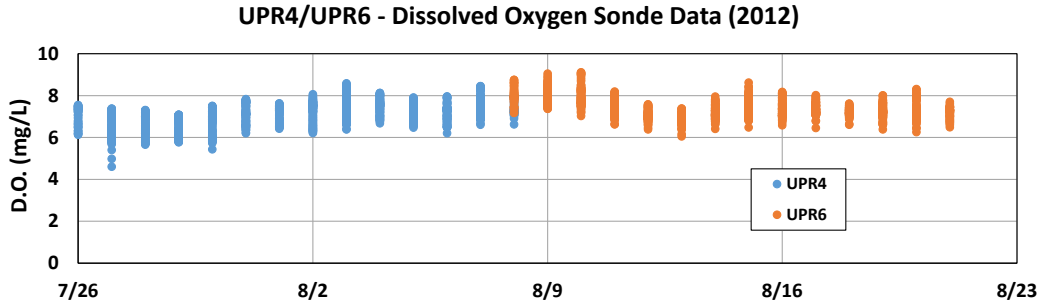
(a)



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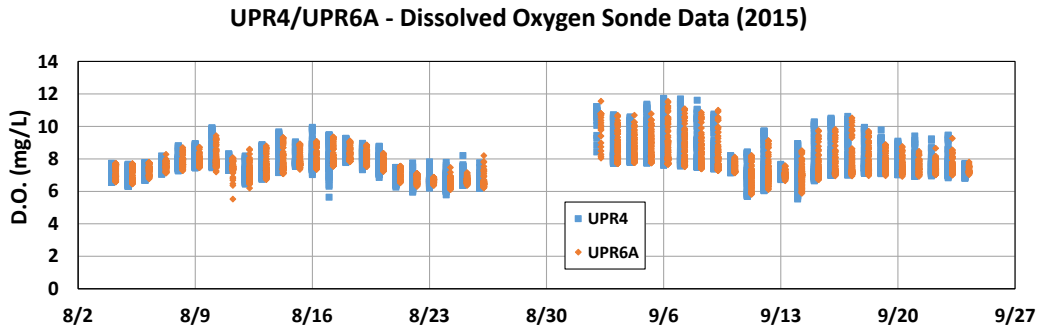

Figure 17: Ambient DO Diurnal Range for (a) 2012 and (b) 2015

The 2012 data show that the DO concentrations are generally well above the water quality criteria and the median diurnal range was 1.6 mg/L. The overall average DO for the measurement period was 7.09 mg/L and 7.39 mg/L for UPR4 and UPR6, respectively, with a minimum DO of 5.7 mg/L (UPR4) and 6.3 mg/L (UPR6). These differences do not appear to be due to differences in temperature or salinity (Table 1). In fact, the higher DO levels observed at UPR6 occurred in response to conditions favoring a lower DO saturation concentration in the river. The temporal variation in average DO concentration appears to be related to the tidal cycle and variability in the DO response.

The 2015 data show that the DO concentrations are well above the water quality criteria. During the August installation, the median diurnal range was 1.6 mg/L and the minimum DO was 5.9 mg/L. During the subsequent installation, the median diurnal range increased to 2.8 mg/L while the minimum DO decreased to 5.5 mg/L. The overall average DO for the first installation period was 7.5 mg/L for both UPR4 and UPR6A. The overall average DO for the second installation period was 8.0 mg/L for UPR4 and 7.8 mg/L for UPR6A. The observations at station UPR4 mimic the observations at UPR6A, as might be expected for estuarine stations situated 1.0 kilometer apart.

The 2015 DO response for the August and September installations is obviously different. These differences do not appear to be due to differences in temperature or salinity, which did not change significantly based on the measurements obtained for these parameters (Table 1).

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Table 1: Average Survey Temperature and Salinity

Survey	Temperature			Salinity		
	UPR4	UPR6	UPR6A	UPR4	UPR6	UPR6A
7/26-8/8/2012	22.0	-	-	24.2	-	-
8/8-8/21/2012	-	22.1	-	-	27.3	-
8/3-8/26/2015	22.4	-	21.8	26.2	-	28.2
9/2-9/24/2015	21.4	-	20.9	28.4	-	28.0

The differences between the August and September 2015 sonde measurements merit comment with regard to the overall system response to nutrients. Over this time period, the ambient TN concentration remained relatively stable at about 0.25 mg/L (See, Figure 7) and the algal biomass averaged less than 5 µg/L, with the exception of a slight bloom (12 µg/L) on September 9, 2015. (See, Figure 10). Although conditions were nearly identical between August and September, the overall average DO concentration increased and the diel swing doubled. The maximum diel swing occurred coincidently with the slight bloom, but the overall water quality was better than when the algal biomass was reduced (*e.g.*, the daily average DO was 8.8 mg/L when the diel swing was at its maximum). Thus, it appears that the short term algal bloom added additional DO to the system, on net, in excess of the change in diurnal variation.

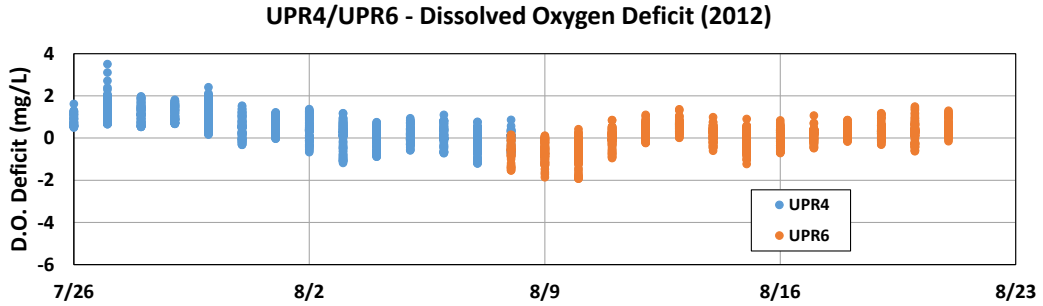
DO Deficit

The DO saturation results were converted into DO deficit to evaluate the overall differences from saturation for the monitoring locations and survey years (Figure 18). DO deficit is an indication of the net oxygen demand exerted by the multitude of factors that affect DO. Dissolved oxygen deficit is particularly useful since it integrates the effects of temperature and salinity on DO saturation. A “negative” deficit reading means that the overall DO condition in the water was above the anticipated saturation value, given the recorded salinity and temperature at that time. The 2012 UPR4 results show a diurnal average of 1.2 mg/L, at the start of the monitoring period (7/27 – 7/30). These deficits gradually reduced toward the end of the monitoring period, when deficits averaged 0.1 mg/L (8/3 – 8/6). The 2012 UPR6 results show a continuing reduction in the overall deficit (diurnal average -0.04 mg/L from 8/9 – 8/15) and then the deficit begins to rise toward the end of the monitoring period, averaging 0.2 mg/L (8/16 – 8/20).

The 2015 August survey results show a daily average deficit of -0.10 mg/L for UPR4 and a daily average of -0.04 mg/L for UPR6A. The corresponding daily average deficits for the 2015 September survey were -0.48 mg/L and -0.12 mg/L for UPR4 and UPR6A, respectively.

DOVER 001900

(a)



(b)

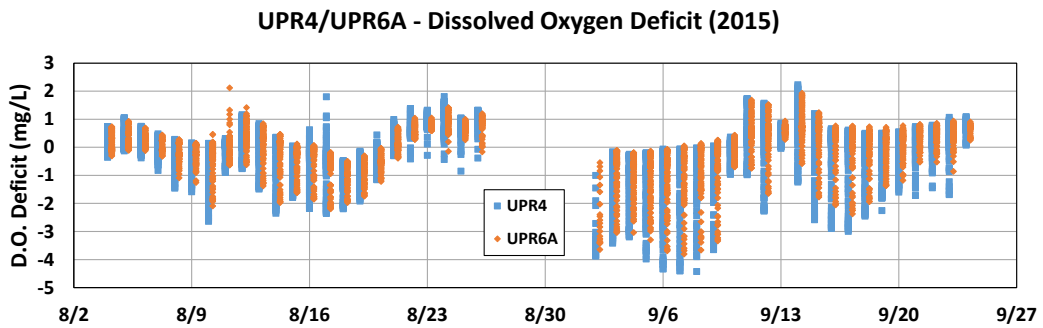


Figure 18: Calculated Ambient DO Deficit Concentration for (a) 2012 and (b) 2015

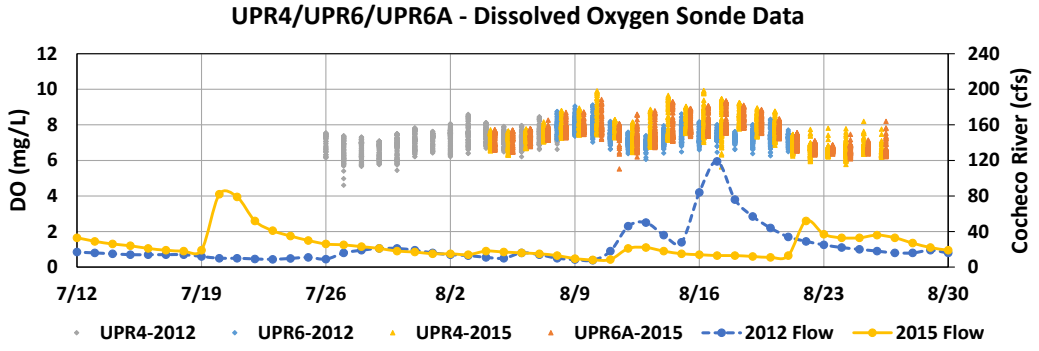
- Effect of TN Reduction on DO

Direct comparisons of the 2012 and 2015 sonde data, to evaluate the effect of TN reduction on this parameter, are complicated by the influence of multiple factors affecting DO. Differences in temperature and salinity are captured by looking at dissolved oxygen deficit. However, other factors influencing DO are not considered. As noted earlier, these include: external loads from point source dischargers, tidal exchange and tributary flows of DO deficit, carbonaceous biochemical oxygen demand, and nitrogenous oxygen demand; sediment oxygen demand; reaeration (affected by wind and tidal currents); and, algal photosynthesis/respiration (affected by ambient chlorophyll-a level, water clarity, cloud cover, and sun angle). Thus, any assessment of the data presented above does not provide a complete picture. However, the available DO data do provide further insight on the effects of TN reduction on this parameter.

As a first cut, the data from the two surveys were superimposed upon each other to provide a visual assessment of diurnal DO range. USGS flow data for the Cocheco River are also plotted to provide a potential indication of external loads (Figure 19).

DOVER 001901

(a)



(b)

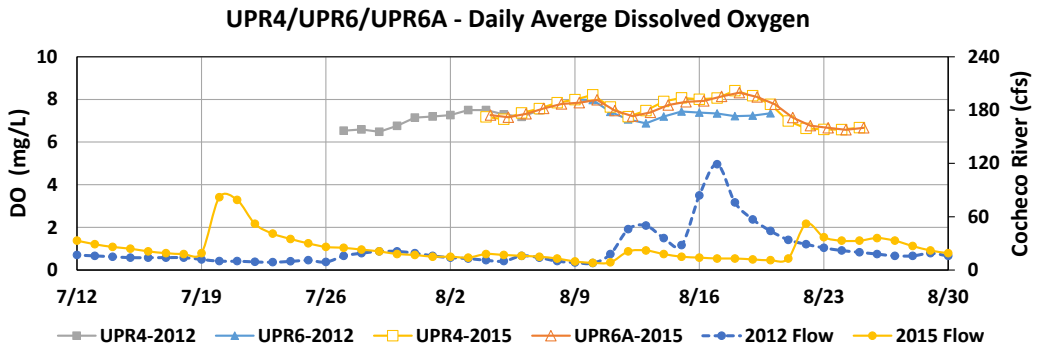
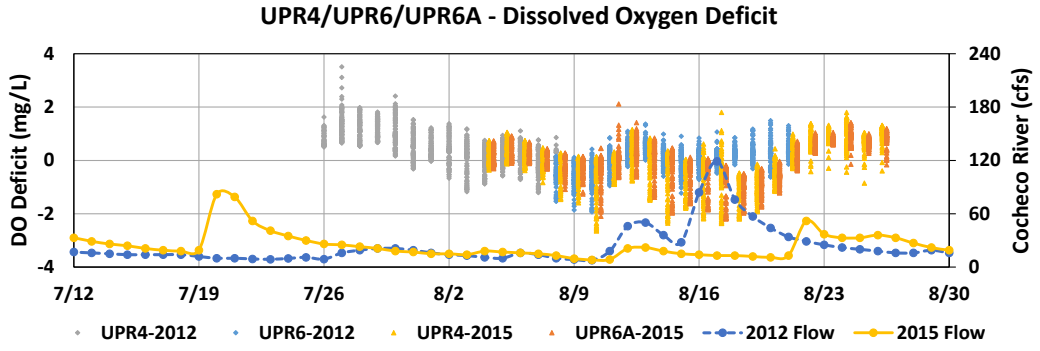


Figure 19: Combined 2012 and 2015 DO: (a) Daily Variability; (b) Daily Average

Figure 19 shows that the Cocheco River flow regime was nearly identical for the two monitoring periods. The pattern in DO concentration illustrated for the pre- and post-nutrient reduction periods suggest a nearly identical variability in diurnal range, daily average, and temporal variability. System DO concentration does appear to decrease in response to increased tributary flow (See, 2012 DO readings from August 12 to August 20), which may be expected since higher tributary flows would transport higher organic loads into the system and may also have a lower DO concentration than occurs in the receiving water. This observation would tend to refute the assertion that a decrease in ambient TN from 0.50 mg/L to 0.25 mg/L would increase the minimum DO by 1 mg/L.

Figure 20 presents a similar evaluation for dissolved oxygen deficit. The initial deficits observed at UPR4 in 2012 are higher than those experienced in 2015. However, as the temporal shift in deficit progresses, both the 2012 and 2015 deficits for UPR4 align. The 2012 measurements for UPR6, which commence on August 8, fall on top of the 2015 measurements and only deviate when the flow record for the Cocheco River indicates a significant increase in external load.

(a)



(b)

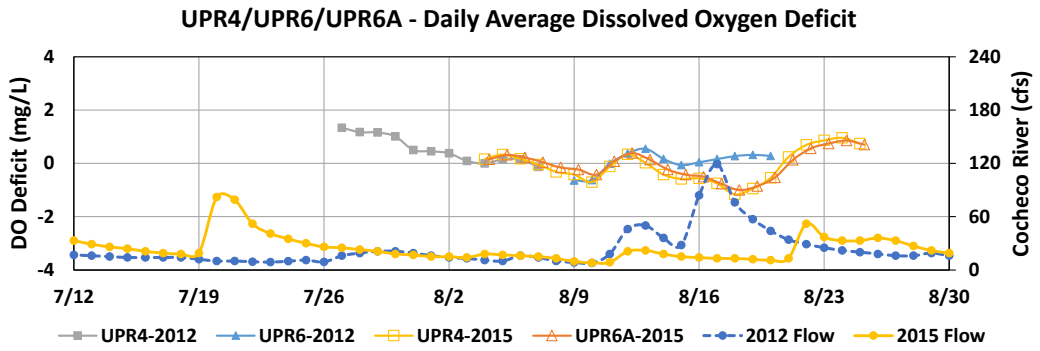


Figure 20: Combined 2012 and 2015 DO Deficit: (a) Daily Variability; (b) Daily Average

Summary data for average DO, minimum daily DO, diel DO swing, and average daily deficit are presented in Table 2. The data show similar results for minimum daily DO, diel DO swing, and average daily deficit, as would be expected given the lack of change in algal productivity mentioned above. For the most significant parameter with regard to aquatic life use attainment (minimum daily DO concentration), the results are nearly identical. In all cases, the minimum DO was well above the threshold for impairment. The overall range in diurnal DO is also essentially identical for the two periods. The August surveys show diurnal swings less than 2.0 mg/L, which indicates that eutrophic enrichment is not a significant concern for the Upper Piscataqua River. Finally, the average daily deficit is well below 1.0 mg/L, indicating that the system is not significantly stressed by oxygen demanding substances.

Direct comparison of the daily DO deficit in 2012 and 2015 suggests that the deficits were slightly higher in 2012. However, this appears to be attributed to tidal conditions and the short duration of the survey with regard to Station UPR4. The slightly elevated deficit observed at Station UPR6 may be attributed to additional loads associated with storm runoff as suggested by the flow data. These results suggest that the reduction in TN, from 0.5 mg/L to 0.25 mg/L did not affect the DO regime in the Upper Piscataqua River.

DOVER 001903

Table 2: Water Quality Pre- and Post-TN Reduction

Survey Year	2012		2015			
Survey Dates	7/26-8/8	8/8-8/21	8/3-8/26		9/2-9/24	
Station	UPR4	UPR6	UPR4	UPR6A	UPR4	UPR6A
Median TN (mg/L)	0.50*		0.24	0.21	0.25	0.25
Average DO (mg/L)	7.1	7.4	7.5	7.5	8.0	7.7
Minimum Diel DO (mg/L)	5.7	6.3	5.9	6.1	5.5	5.8
Diel Range (mg/L)						
Median	1.7	1.6	1.6	1.3	3.0	2.9
Average	1.8	1.6	1.8	1.5	3.0	2.6
DO Deficit (mg/L)						
Median	0.42	0.16	-0.08	0.06	-0.39	0.02
Average	0.52	0.07	-0.09	-0.04	-0.48	-0.12

* Median TN concentration for August-September, 2010-2012

Conclusion

In summary, the available "before and after" data indicate that despite the substantial TN load reductions from WWTFs (primarily a reduction in inorganic nitrogen) and associated ambient TN water quality improvements in the Upper Piscataqua River, no demonstrable improvements in algal biomass or transparency occurred. The lack of a marked response in chlorophyll-a concentrations indicates that factors other than TN concentration control algal biomass in this system. Without a change in algal biomass, the light extinction coefficient also does not respond to changes in TN concentration. The observed fluctuations in the light extinction coefficient are attributed to watershed runoff contributions of CDOM and NAP.

Ambient DO concentrations in the Upper Piscataqua River show high quality conditions prior to and after TN load and concentration reductions. Variations in ambient DO concentration are apparent in the "before and after" data. These variations appear to be associated with spring-neap tidal conditions, and the short duration of the 2012 survey preclude direct comparison of the concentration data. However, given the significant overlap of the data, adverse effects of elevated TN concentration on DO concentration are not apparent. Moreover, the other metrics used to evaluate changes in the DO regime (minimum daily DO, diel DO swing, and average daily DO deficit) do not show improvement with the reduction in TN to 0.25 mg/L. Consequently, ambient TN concentration and point source TN load do not appear to be significant influences on DO in the Upper Piscataqua River.

DOVER 001904