Effect of Water Quality Model Uncertainty on the Passaic TMDL and Water Quality Trading Program for Total Phosphorus

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ABSTRACT

Water quality modeling is a major source of scientific uncertainty in the Total Maximum Daily Load (TMDL) process. The effects of these uncertainties extend to water quality trading programs designed to implement TMDLs. This study identified how water quality model uncertainty affects outcomes related to the Non-Tidal Passaic River Basin TMDL for total phosphorus (TP) and potential trades of TP between point sources. Although the TMDL scenario showed greater than 10% probability of exceeding TMDL targets for chlorophyll-*a* (chl-*a*) at Dundee Lake and TP load diverted to the Wanaque Reservoir, the efficacy of TMDL measures was clearly demonstrated when compared directly to actual conditions in Water Year (WY) 2002. Trading scenario simulations suggest that trading ratios have been well designed. Each scenario demonstrated parity with or improvement from the baseline at the TMDL critical locations of Dundee Lake and the Wanaque South intake, and low risk of hot spots elsewhere.

KEYWORDS: water quality trading, uncertainty analysis, total maximum daily load, water quality modeling, latin hypercube sampling, WASP.

INTRODUCTION

Water Quality Trading and Uncertainty

Water quality trading (WQT) represents a watershed-based and market-based approach to meeting and exceeding water quality goals (Faeth, 2000). U.S. Environmental Protection Agency (USEPA) support for water quality trading is driven by the expectations that trading can reduce the cost of Total Maximum Daily Load (TMDL) implementation and engage unregulated nonpoint sources, such as farmers, in reducing pollutant discharges to waterbodies (USEPA 2003, 2004). Despite the high hopes for WQT, it continues to progress at a slower pace than expected (King, 2005).

A recent USEPA review (2005) suggested that uncertainty in the TMDL process has prevented the onset of significant trading activity. Besides monitoring, water quality modeling is a major source of scientific uncertainty in the TMDL process (NRC, 2001); the effects of these uncertainties extend to water quality trading programs designed to implement TMDLs.

Background of Study Area: Development of Passaic TMDL and WQT Program

Approximately 25% of New Jersey's population (i.e., two million people) lives in the non-tidal portion of the Passaic River Basin. The New Jersey portion of the watershed covers 669 square miles (Figure 1). To address over 200 stream miles impaired due to excessive total phosphorus (TP) in the Non-Tidal Passaic River Basin, the New Jersey Department of Environmental Protection (NJDEP) adopted the *Total Maximum Daily Load for the Non-Tidal Passaic River Basin Addressing Phosphorus Impairments* which set phosphorus load allocations for point and nonpoint sources in the watershed (NJDEP, 2008). Phosphorus loading in the basin is dominated by point sources. A water quality trading program is in development to increase the cost effectiveness of TMDL implementation (Obropta et al., 2008). Twenty-two WWTPs are expected to be the main trading participants.

Extensive water quality modeling of the Non-Tidal Passaic River Basin has been completed by Omni Environmental Corporation (2007), and their water quality model is hereafter referred to as the "TMDL model." The TMDL model is the basis for both *establishing* the TMDL allocations and *implementing* the TMDL via water quality trading. TMDL allocations for total phosphorus were derived from predictions of the TMDL model. In addition, the model was applied to predict water quality outcomes of various trading scenarios proposed to implement the TMDL. Given the importance of the TMDL model in informing the TMDL allocations and trading program development, analysis of the model uncertainty is vital to examining if the TMDL allocations and trades of total phosphorus will achieve water quality improvements. Uncertainty analysis is especially needed to verify that trades are not likely to create "hot spots," or localized areas of degraded water quality, a concern of both the USEPA (2003 and 2004) and critics of water quality trading (e.g., Steinzor, 2003). Model uncertainty analysis would yield an explicit approximation of the likelihood that the TMDL and phosphorus trades will have a positive impact on the Non-Tidal Passaic River Basin.

Model uncertainty analysis is widely acknowledged as essential for conducting reliable environmental decision making (Reckhow, 1994; NRC, 2001; Shirmohammadi et al., 2006). Previously, Ng and Eheart (2005) analyzed the uncertainty of a hypothetical biochemical oxygen demand (BOD) water quality trading program. However, in contrast with Ng and Eheart (2005), this is the first study that addresses the effect of model uncertainty on nutrient trading, where the dynamics of nutrients, algae and dissolved oxygen (DO) are more complex and uncertain than that of BOD and DO.

Study Objective

The objective of the study was to identify how model uncertainty affects model outputs and decision risks related to the Passaic TMDL (NJDEP, 2008) and potential trades of

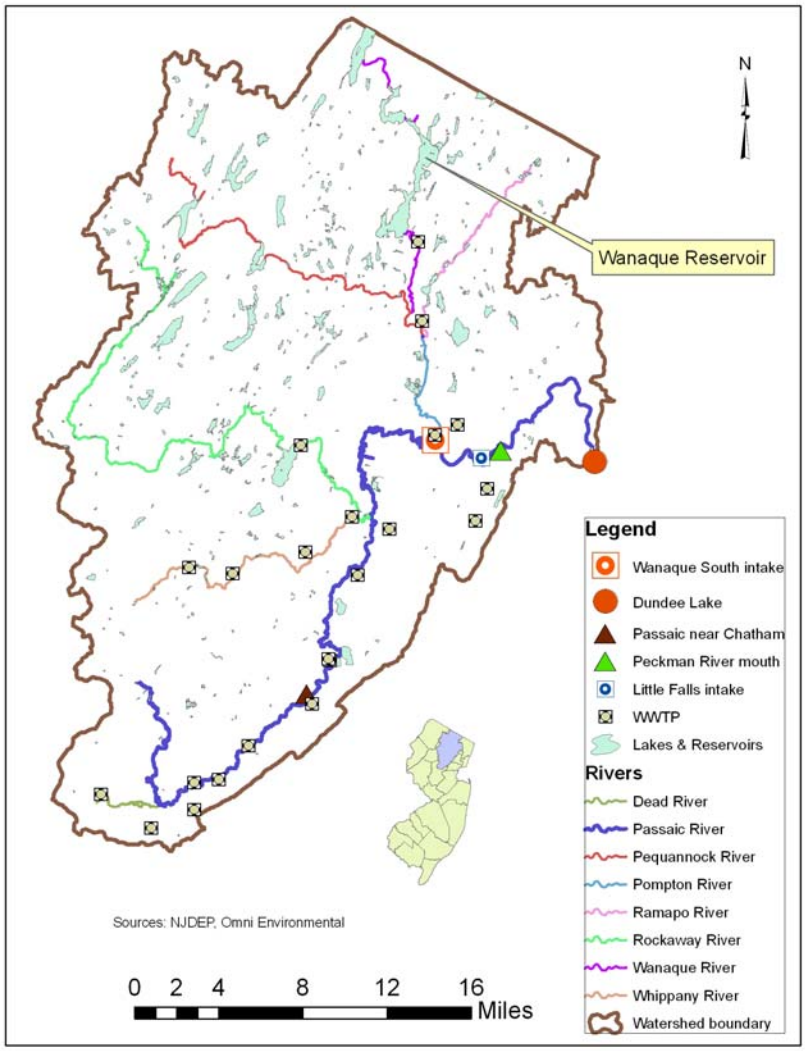

Figure 1. Non-Tidal Passaic River Basin, New Jersey.

total phosphorus between wastewater treatment plants (WWTPs). The following hypotheses were tested by model uncertainty analysis:

1. The TMDL for total phosphorus will result in attainment of DO standards and site-specific chl-*a* criteria at Dundee Lake, with less than 10% expected exceedance and 10% exceedance probability, respectively, at critical drought conditions;
2. The TMDL will result in attainment of a 70% reduction, at critical drought conditions, of total phosphorus load diverted to the Wanaque Reservoir from the Wanaque South pump intake, with less than 10% exceedance probability;
3. Trades between WWTPs will achieve compliance with TMDL water quality targets at each of five potential hot spots in the watershed, with less than 10% expected exceedance or exceedance probability, at critical drought conditions.

METHODOLOGY

Scope of Uncertainty Analysis

The TMDL model is a calibrated and validated application of the Water Quality Analysis Simulation Program (WASP) version 7.0, released by the USEPA Office of Research and Development in Washington DC. WASP simulates nutrient kinetics and algal growth in a river network. WASP 7.0 is a dynamic compartment-modeling program for aquatic systems, including the water and the underlying benthos (Di Toro et al., 1983; Wool et al., 2003; Ambrose et al., 2006). A copy of the TMDL model was obtained from Omni Environmental Corporation.

This study focused on four sources of uncertainty in the TMDL model: i) global kinetic parameter values, ii) local kinetic parameter values, iii) phosphorus boundary conditions at the Passaic River and Ramapo River headwaters, and iv) phosphorus boundary conditions from selected WWTPs. The effect of these sources of uncertainty is feasible to analyze using the methods described below. It is assumed that these are the primary sources of uncertainty in the model. A statistical comparison of observations to uncertainty estimates evaluated that assumption. Also note that the vast majority of water quality model uncertainty analyses only focus on parameter and input uncertainty (e.g., Melching and Bauwens, 2001; Zhang and Yu, 2004; Lindenschmidt, 2006).

The following areas were outside the scope of the uncertainty analysis: i) hydrodynamic model uncertainty, ii) nonpoint source load model uncertainty, iii) model structure uncertainty, iv) measurement uncertainty, and v) the LAWATERS model of the Wanaque Reservoir (Najarian, 2005).

Application of Latin Hypercube Sampling

Method practicality is a key concern in uncertainty analysis of complex models (Chapra, 2003) such as the TMDL model. Conventional Monte Carlo simulation (MCS) involving many hundreds or thousands of simulations is not practical in this study because multiple trading scenarios need to be analyzed, and it is not feasible to repeat the entire process of conventional MCS for each scenario. Therefore, this study applied Latin Hypercube Sampling (LHS) with restricted pairing (Iman and Conover, 1982) as the main method of uncertainty analysis.

Table 1 lists the parameters included in the uncertainty analysis. Selection was based on Omni Environmental (2007), Lindenschmidt (2006), and a sensitivity analysis of the TMDL model.

Fifteen kinetic parameters (eight global and seven local) and nine boundary conditions for phosphorus were selected for inclusion in the uncertainty analysis, yielding a total of twenty four variables. The equation to determine the number of model runs needed in an LHS uncertainty analysis (Iman and Helton, 1985) is provided below.

$n \geq (4/3) * x$ (1)

Where

n = number of model runs, and

x = number of uncertain variables.

With x = 24, the TMDL model was run 50 times.

Table 1. Parameters included in uncertainty analysis.

Global kinetic parameters [unit] (type of probability distribution) [a]			
Phytoplankton max growth rate @ 20°C [/d] (Normal)	Phytoplankton carbon: chlorophyll ratio (Triangular)	Phytoplankton respiration rate @ 20°C [/d] (Triangular)	Phytoplankton death rate [/d] (Triangular)
Phytoplankton optimal light saturation [langleys/d] (Triangular)	Benthic algae max growth rate [$gD/m^2/d$] (Triangular)	Benthic algae respiration rate [/d] (Lognormal)	Benthic algae death rate [/d] (Beta)
Local kinetic parameters [unit] (type of probability distribution) [a]			
Sediment oxygen demand [$g/m^2/d$] (Normal)	Dissolved fraction of orthophosphate (Normal)	Settling rate of particulate phosphorus [/d] (Normal) [b]	Fraction of bottom segment covered with benthic algae (Normal) [b]
Headwater boundary conditions (type of probability distribution)			
Passaic River headwater scaling factor for phosphorus (Normal)		Ramapo River headwater scaling factor for phosphorus (Normal)	
WWTP effluent [mg/l] (type of probability distribution)			
Two Bridges: orthophosphate (1 cluster Normal, 2 clusters Lognormal) [c]	Two Bridges: organic phosphorus (Lognormal)	Verona: orthophosphate (Lognormal)	Berkeley Heights: organic phosphorus (Normal)
Berkeley Heights: orthophosphate (Normal)	Rockaway Valley: orthophosphate (1 cluster Normal, 2 clusters Lognormal) [c]	Rockaway Valley: organic phosphorus (1 cluster Normal, 2 clusters Lognormal) [c]	

[a] Probability distribution characteristics of global and local kinetic parameter cited in DiToro and van Straten (1979), Scavia et al. (1981), Bowie et al. (1985), Brown and Barnwell (1987), Chapra (1997), Manache and Melching (2004), and Lindenschmidt (2006).

[b] Variable was modeled separately at multiple locations, yielding a total of seven variables for this category.

[c] According to analysis of effluent data.

Separate probability distributions were estimated for each of the parameters listed in Table 1. For kinetic parameters, the TMDL model calibrated values served as mean values for symmetric distributions and mode or geometric mean values for skewed

distributions. For sediment oxygen demand (SOD), the distribution properties were based on available data in Omni Environmental (2007), and for other kinetic variables the distribution properties were based on the literature cited below Table 1. Through interpretation of the fundamental WASP model equations (Wool et al., 2003), certain kinetic parameters were assumed to have covariance, with a correlation coefficient of either (+) or (–) 0.5. To economize on model runs, certain local kinetic parameters were each considered as a standardized variable, as described by Melching and Bauwens (2001). In this way, local kinetic parameters were "lumped" in the error propagation and the number of LHS samples was kept at a manageable amount. Probability distributions for phosphorus boundary conditions were based on available data found in Omni Environmental (2007).

The LHS samples with restricted pairing were generated using the ARRAMIS TM Risk and Reliability software package, version 0.5 Beta, developed by Sandia National Laboratories in Albuquerque, New Mexico (Wyss and Jorgensen, 1998).

Probing the Credibility of Uncertainty Estimates

Past uncertainty analyses of *alternate* scenarios are particularly relevant to the study objective, which focuses on the uncertainty of TMDL and trading scenarios. A key question not always asked is "how *credible* is the uncertainty estimate of future scenarios?" Or in this case, "why should one believe the estimate of uncertainty for a TMDL condition or trading scenario…how well does the uncertainty analysis compare to actual data in the first place?" In the literature, uncertainties for complex mechanistic models are often predicted for future scenarios without first comparing the uncertainty analysis to available observed data (e.g., Zhang and Yu, 2004). Two branches of methods that address this concern are not feasible or applicable here. Bayesian parameter identification methods, as applied in Gallagher and Doherty (2007), explicitly relate parameter uncertainty to observed data, however these methods are too computationally demanding for the TMDL model, which requires 2 hours on a 1.6 GHz PC to simulate one year. Borsuk et al. (2002) demonstrated in a landmark paper a method to account separately for residual variability and parameter uncertainty; however, it implicitly assumes that residual patterns will be unchanged for alternate scenarios, an assumption that cannot reasonably be applied to trading scenarios. The method applied here involved application of simple statistical tools to assess the robustness of the uncertainty analysis when compared to observed data. In this manner, the credibility of the uncertainty estimate for an alternate scenario was better established.

The uncertainty analysis was executed in three stages. In Stage 1, the credibility of the uncertainty estimates were assessed through comparison to observed data. In Stages 2 and 3, the uncertainty analysis was extended to the TMDL and trading scenarios, respectively.

In Stage 1, the performance of the uncertainty analysis was tested against observed data (compiled in Omni Environmental (2007)) from 2001 to 2003 for the parameter and location listed in Table 2. Since the Passaic TMDL bases seasonal average chl-*a* criteria

on the period June 15-August 31, only data from this time span was used when comparing chl-*a* and DO predictions. However, since TP is diverted at the Wanaque South intake almost year-round, TP data from the entire year was used to evaluate predictions.

Table 2. Stage 1 Uncertainty analysis output.

Location	Key water quality parameter(s)	Significance	Period of available data (data source)
1. Dundee Lake (watershed outlet)	chl-*a*, DO	TMDL critical location (NJDEP, 2008) due to high chl-*a* and high diurnal DO swing.	DO: 2003 (Omni) chl-*a*: insufficient
2. Wanaque South intake (point of major surface water diversion to Wanaque Reservoir, the largest source of drinking water for New Jersey)	TP	TMDL critical location (NJDEP, 2008) due to potential effect of diverted water with high TP to stimulate algal blooms in reservoir.	2003 (Omni)
3. Peckman River mouth	DO	Area of concern (Omni Environmental, 2007) due to low DO and high diurnal DO swing.	2003 (Omni)
4. Passaic River near Chatham	DO	Area of concern (Omni Environmental, 2007) due to low DO	2002 (NJDEP)
5. Little Falls intake	TP	Area of concern (Omni Environmental, 2007) due to effect of TP on drinking water treatment.	Data unreliable
6. PA10 (~ 5 km upstream of Dundee Lake)	DO	High chl-*a*, high diurnal DO swing.	DO: 2003 (Omni)
7. 17N25 (~ 3 km upstream of Dundee Lake)	chl-*a*, DO	High chl-*a*, high diurnal DO swing.	DO, chl-*a*: 2001-2003 (Passaic Valley Sewerage Commission)

In Table 2, locations 1-5 are considered potential hot spots in the watershed. Locations 6-7 are included in Stage 1 because of their proximity to Dundee Lake.

The 80% confidence intervals at each timestep from all 50 model runs were compared to observed data. The number 80% is chosen because in Stages 2 and 3, the objective for chl-*a* and TP predictions was to compare water quality targets against the 10% exceedance probability. The 10% exceedance probability corresponds to the 10% outside

the 80% confidence interval at the extreme of concern; the high extreme pertains to phosphorus and chl-*a*. In contrast, the 10% expected exceedance, a measure used to evaluate daily average and minimum DO predictions in Stages 2 and 3, does not have a direct relation to the 10% outside the 80% confidence interval. However, because 80% confidence intervals of DO predictions are a general indicator of the overall likelihood of exceedance, it is still valuable to assess in Stage 1 the credibility of these 80% confidence intervals before calculating expected exceedances in Stages 2 and 3.

A simple statistical approach, the calculation of the confidence interval about a binomial proportion, was used to evaluate the performance of the uncertainty analysis. For each uncertainty analysis output, the number of measured values inside the predicted 80% confidence intervals (i.e., successes) was compared to total measured values. A 95% confidence interval about the proportion of successes was then calculated. If 0.80 (the expected proportion of success) fell inside the 95% confidence interval about the proportion, the result was positive, indicating the predicted 80% confidence intervals were credible; otherwise the predicted 80% confidence intervals were not credible. Since the sample sizes were small, the Agresti-Coull confidence interval formula was used (Agresti and Coull, 1998):

$$CI_{AC} = \tilde{p} \pm \kappa (\tilde{p}\,\tilde{q})^{1/2}\,\tilde{n}^{-1/2} \qquad (2)$$

Where

CI_{AC} = Agresti-Coull confidence interval,
$\tilde{n}$ = number of independent Bernoulli trials + 4,
$\tilde{p}$ = (number of successes + 2) / $\tilde{n}$,
$\tilde{q}$ = 1- $\tilde{p}$, and
$\kappa = z_{\alpha/2}$ = 1.96 at 95% confidence.

In Stage 1, dissolved oxygen concentrations were processed on a moving sub-daily time scale, and total phosphorus and chl-*a* concentrations were processed on a moving time scale, before comparing model output to observed data. This was done because prediction accuracy of those parameters at those scales is an adequate measure of success, and accuracy at a finer temporal resolution is not necessary for the purposes of this study.

Stage 2 focused on the uncertainty of the TMDL scenario at future critical drought conditions in which all WWTPs discharge a long term average of 0.4 mg/l total phosphorus effluent at permitted flows. Besides the variables that were modeled probabilistically in Stage 1, additional WWTP boundary conditions for phosphorus were modeled as uncertain in Stage 2 to reflect the specific uncertainties of the TMDL scenario. In Stage 2, all WWTP boundary conditions for phosphorus were assumed to have a lognormal distribution with an arithmetic mean of 0.4 mg/l and a coefficient of variation equal to 0.6, as assumed in NJDEP (2008). As in Stage 1, separate probability distributions were generated for each uncertain variable. With x = 35, the model was run 50 times to satisfy Equation 1.

Stage 3 focused on the uncertainty of three trade scenarios and one baseline scenario at critical drought conditions. These scenarios are listed in Table 3 and described further in Obropta et al. (2008). The enumeration of the scenarios is maintained to be consistent with Obropta et al. (2008). The same variables that were modeled probabilistically in Stage 2 were repeated in Stage 3. In Stage 3, all WWTP boundary conditions for phosphorus were assumed to have a lognormal distribution with an arithmetic mean of their presumed long term average total phosphorus effluent concentration and coefficient of variation equal to 0.6, as assumed in NJDEP (2008). Buyers discharged long term average TP concentrations greater than 0.4 mg/l and sellers discharged less than 0.4 mg/l. As in Stages 1 and 2, separate probability distributions were generated for each uncertain variable. With x = 35, the model was run 50 times to satisfy Equation 1.

Water quality trading allocations for WWTPs in the basin are expected to be based on a recent history of average plant flows. Therefore, the baseline scenario for Stage 3 consisted of all WWTPs discharging at a long term average of 0.4 mg/l TP at flow levels representative of 2004-2006. These flows are less than the permitted flows applied in the Stage 2 TMDL scenario simulation. No trading was simulated in the baseline scenario.

Table 3. Description of trading scenarios for water quality simulation.

Scenario	General description
Baseline	All WWTPs discharging at 0.4 mg/l long term average total phosphorus effluent
1	Simple trade with seller upstream; buyer and seller in same subwatershed; Seller is upstream of Passaic River near Chatham.
8	Complex trades with buyers and sellers in different subwatersheds; buyers concentrated upstream; several buyers upstream of Passaic River near Chatham; sellers upstream of Wanaque South intake.
9	Complex trades with buyers and sellers in different subwatersheds; buyers spread upstream and downstream; buyers upstream of Passaic River near Chatham and Peckman River mouth; sellers upstream of Wanaque South intake.

In Stages 2 and 3, model output was processed as follows. For chl-*a*, a distribution of seasonal averages from June 15 through August 31, 2002 at Dundee Lake was calculated. The hydrology of 2002 represents a critical drought condition, and June 15-August 31 corresponds to the NJDEP (2008) site-specific seasonal average criteria. For DO, distributions of exceedance frequencies of the daily average and minimum DO standards over the period of June 15 through August 31, 2002 were calculated at Dundee Lake, the Peckman River mouth, and the Passaic River near Chatham. As described in Borsuk et al. (2002), the expected exceedance is the mean value of the exceedance frequencies, and the confidence of compliance is the area of the distribution with exceedance frequencies ≤ 10%. The distribution of daily diurnal DO swings over the period of June 15 through August 31, 2002 was also calculated at Dundee Lake, the Peckman River mouth, and the Passaic River near Chatham. For TP, the distribution of load diverted at the Wanaque South intake over the entire Water Year (WY) 2002 (10/1/01–9/30/02) was calculated, as well as the distribution of average concentration at the Little Falls intake over WY2002.

RESULTS

Stage 1: Assessing the credibility of the uncertainty analysis

As indicated by the 95% Agresti-Coull confidence intervals about a proportion listed in Table 4, the DO simulations compared well to measurements at Dundee Lake (Figure 2), Peckman River mouth, and location PA10, but did not compare well at the Passaic River near Chatham. The discrepancy at the Passaic River near Chatham is likely due to either measurement error or positive bias in the SOD mean value. DO at this location is sensitive to SOD. DO was generally overpredicted (Figure 3) suggesting that SOD was underpredicted. Thus, estimates of DO uncertainty are credible at all locations in Table 4 except the Passaic River near Chatham.

Table 4. 80% confidence intervals of DO simulations compared to observed data.

Location	Measurements	Successes	$\tilde{p}$	95% Agresti-Coull Confidence Interval about proportion
Dundee Lake	18	13	0.69	(0.49, 0.88)
Peckman River mouth	15	11	0.68	(0.48, 0.89)
Passaic River near Chatham	13	6	0.47	(0.23, 0.71)
PA10	20	15	0.71	(0.53, 0.89)

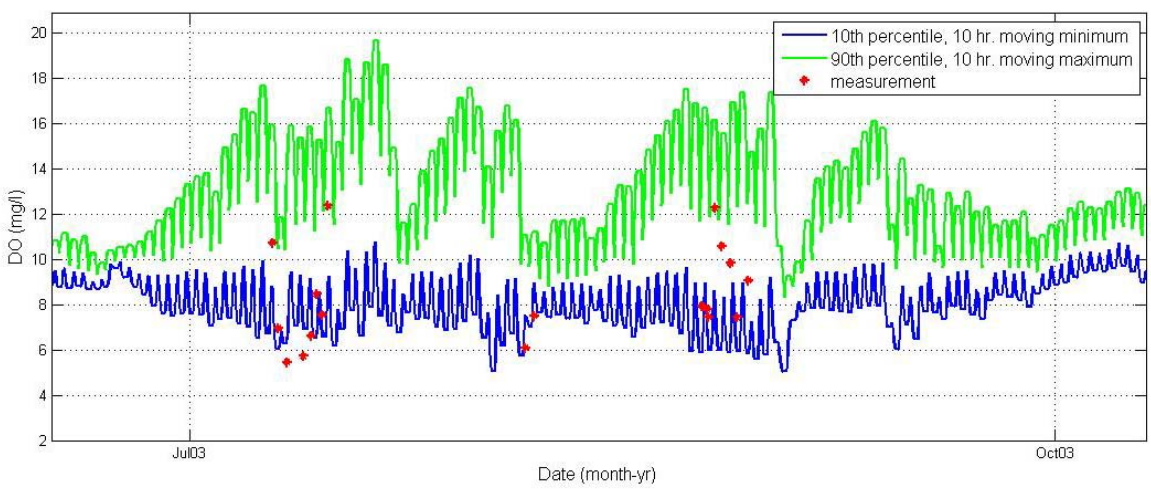

Figure 2. 80% Confidence intervals of DO at Dundee Lake and 2003 observed data.

Chl-*a* data at Dundee Lake was insufficient, so measurements at a nearby upstream location, 17N25, were used instead for comparison to simulations (Figure 4). The results in Table 5 suggest a positive outcome, although the 95% confidence interval is inflated by the small sample size. Therefore, estimates of chl-*a* uncertainty are credible at 17N25

and assumed to be credible at Dundee Lake; however, the credibility is based on a small sample size of observations.

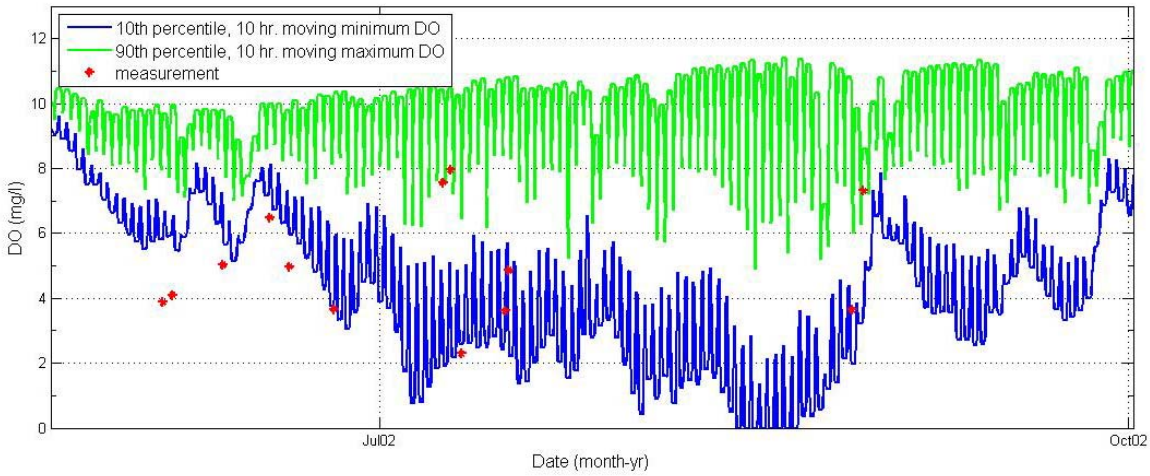

Figure 3. 80% Confidence intervals of DO at Passaic River near Chatham and 2002 observed data.

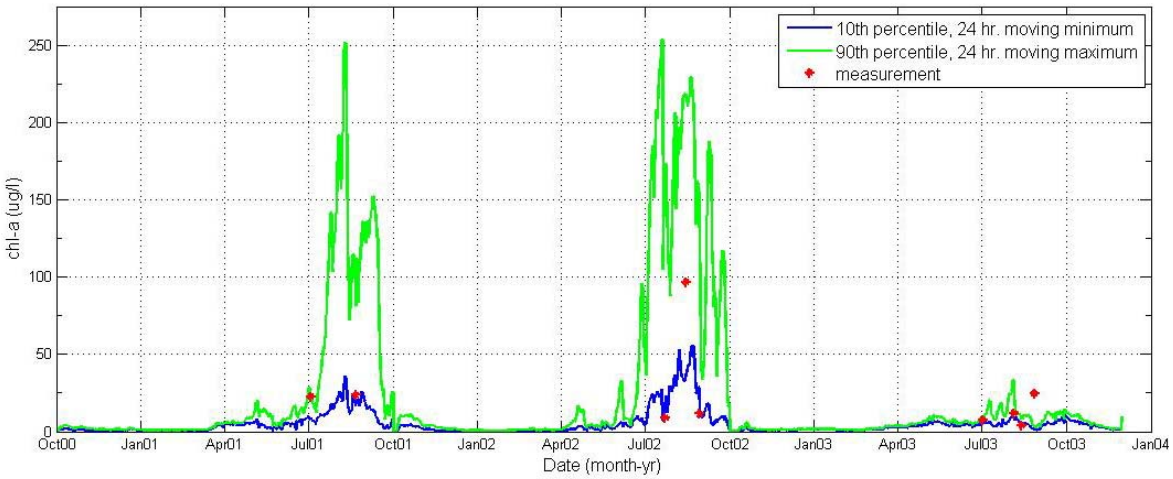

Figure 4. 80% Confidence intervals of chl-*a* at location 17N25 and 2001-2003 observed data.

Table 5. 80% confidence intervals of chl-*a* simulations compared to observed data.

Location	Measurements	Successes	$\tilde{p}$	95% Agresti-Coull Confidence Interval about proportion
17N25	9	5	0.54	(0.27, 0.81)

The positive results in Table 6 suggest that simulations of TP concentrations at the Wanaque South intake are credible (Figure 5). Since the diverted flow is known with certainty, uncertainty estimates of the TP load diverted to the Wanaque Reservoir are

considered credible. Although the observed dataset does not contain any samples during diversions to the Wanaque Reservoir, there were samples taken during less intense diversions to a Passaic Valley Water Commission treatment facility.

Table 6. 80% confidence intervals of TP simulations compared to observed data.

Location	Measurements	Successes	$\tilde{p}$	95% Agresti-Coull Confidence Interval about proportion
Wanaque South intake	20	15	0.71	(0.53, 0.89)

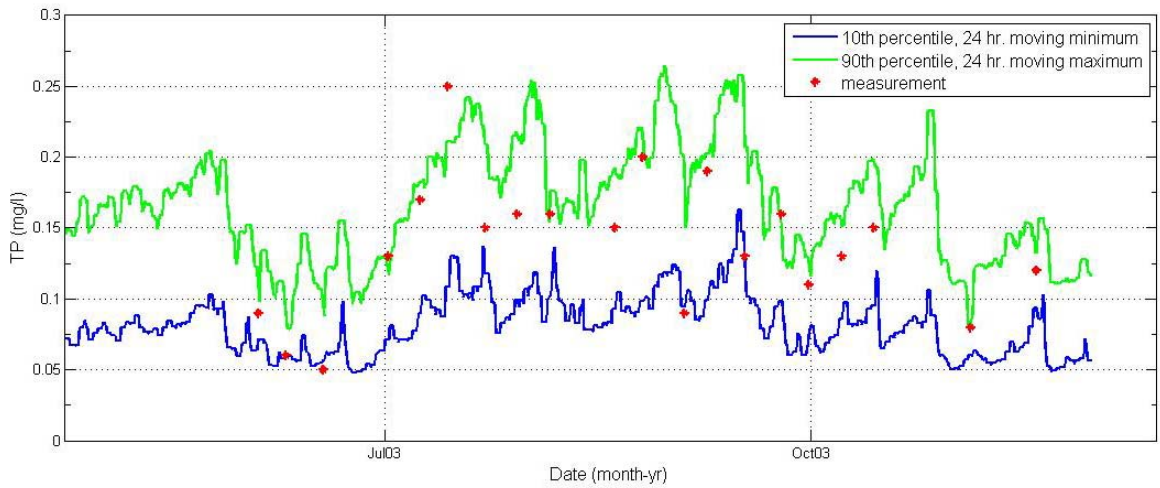

Figure 5. 80% Confidence intervals of TP at Wanaque South intake and 2003 observed data.

Stage 2: TMDL scenario (critical drought conditions and permitted flows from WWTPs).

The distribution of predicted seasonal averages of chl-*a* at Dundee Lake indicates a mean value below the 20 µg/l criteria, but a 28% probability of exceeding the criteria (Figure 6). Note that the distribution is skewed, and would be expected to approach normality with a larger sample size under the Central Limit Theorem.

The distribution of predicted TP load diverted to the Wanaque Reservoir from the Wanaque South intake indicates a mean value below the 70% reduction target, but an 18% probability of exceeding the target (Figure 7).

Tables 7 and 8 indicate less than 10% expected exceedance of daily average and minimum DO at Dundee Lake and the Peckman River mouth, but greater than 10% expected exceedance of daily average and minimum DO at the Passaic River near Chatham. Results from Stage 1 suggest that expected exceedances at the latter location are likely even higher than what was calculated in Stage 2.

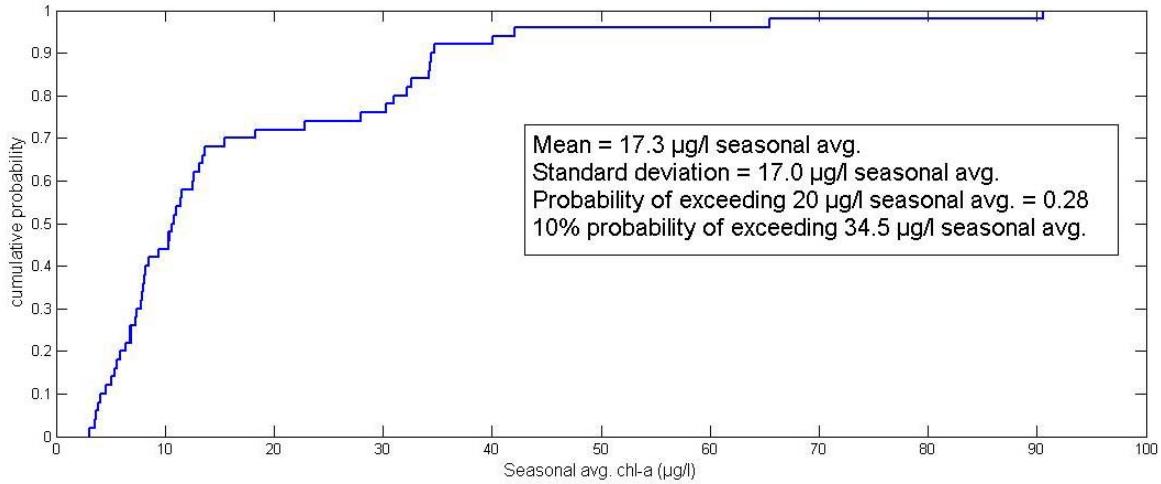

Figure 6. TMDL scenario: cumulative distribution function (CDF) of chl-*a* seasonal average at Dundee Lake, June 15-August 31, 2002.

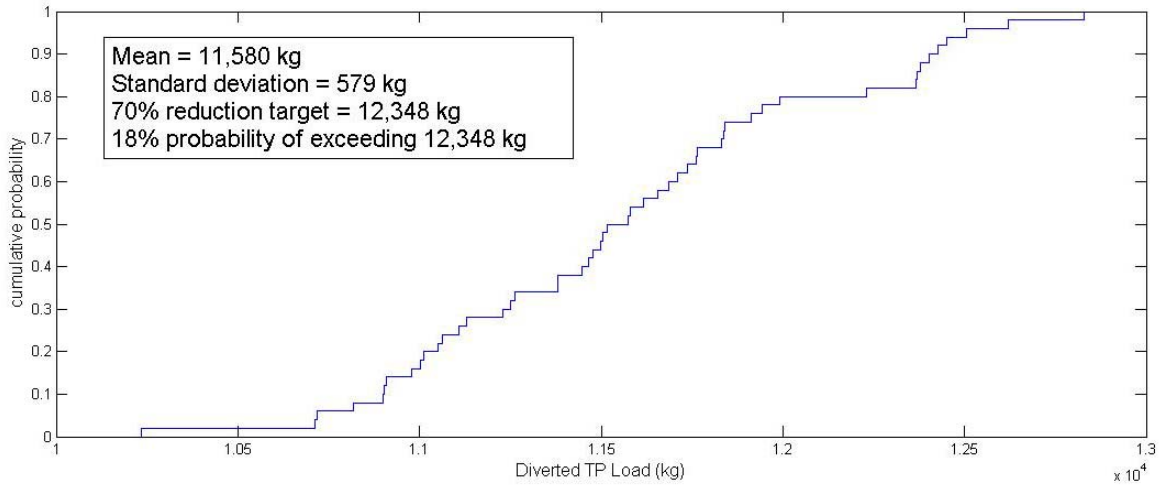

Figure 7. TMDL scenario: CDF of TP load diverted from Wanaque South intake to Wanaque Reservoir in WY2002.

Table 7. TMDL scenario: distribution of daily average DO exceedance frequencies.

Location	Expected Exceedance	Confidence of Compliance
Dundee Lake	0	1.0
Peckman River mouth	0	1.0
Passaic River near Chatham	0.20	0.50

Table 8. TMDL scenario: distribution of daily minimum DO exceedance frequencies.

Location	Expected Exceedance	Confidence of Compliance
Dundee Lake	0	1.0
Peckman River mouth	0.06	0.74
Passaic River near Chatham	0.43	0.20

TMDL *efficacy* scenario: the effect of TMDL measures on actual WY2002 conditions

The TMDL scenario assumed permitted flows are discharged from WWTPs. The uncertainty analysis of the TMDL scenario found instances of water quality target exceedance probabilities and expected exceedances greater than ten percent. However, implementation of TMDL measures through reduction of point and nonpoint source loads would still bring a substantial improvement to water quality over actual conditions. To demonstrate this, TMDL measures (i.e., long term average 0.4 mg/l TP effluent from WWTPs and 60% reduced nonpoint source loads) were applied to actual WY2002 conditions where WWTPs mostly discharged less than permitted flows at TP concentrations greater than the TMDL goal of 0.4 mg/l.

Diverted TP load would have shown a significant improvement (p=0 according to Kolmogorov-Smirnov (K-S) test) over actual conditions if TMDL measures had been in place in WY2002 (Figure 8, Table 9).

Seasonal average chl-*a* at Dundee Lake would have shown a significant improvement (p=0 according to K-S test) over actual conditions if TMDL measures had been in place in WY2002 (Figure 9, Table 10).

The impact of TMDL measures on DO is clearest when analyzing the reduced daily diurnal DO swing at each location in Table 11. A high diurnal DO swing is an indicator of excessive productivity and poor water quality (NJDEP, 2003). For example, a significant decrease in diurnal DO swing (p=0 according to K-S test) at the Peckman River mouth would have occurred if TMDL measures had been in place in WY2002.

Similar trends of improvement are seen at the Little Falls intake if TMDL measures had been implemented in WY2002 (Table 12).

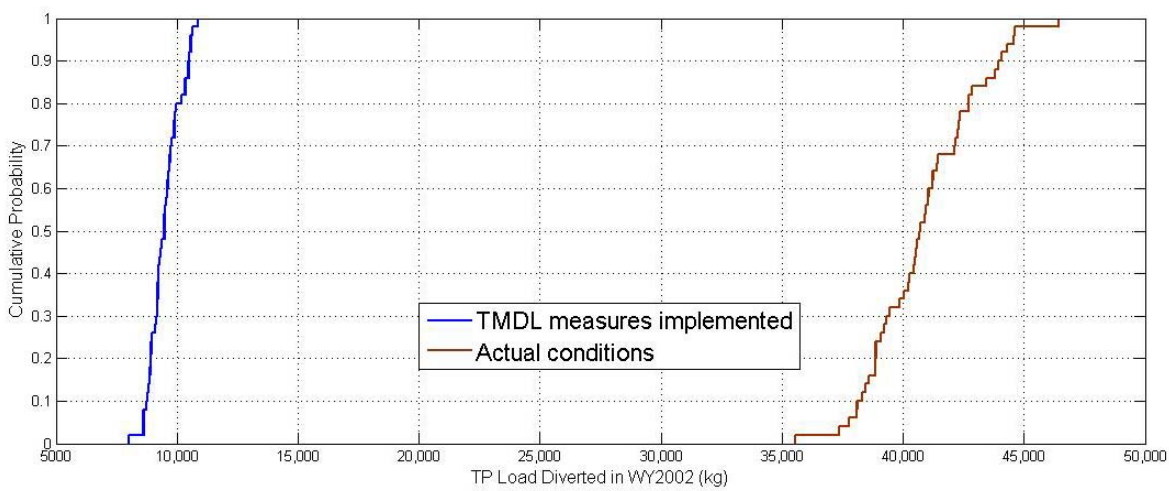

Figure 8. TMDL efficacy scenario: comparison of CDFs of TP load diverted from Wanaque South intake to Wanaque Reservoir in WY2002.

Table 9. TMDL efficacy scenario: total phosphorus load diverted from Wanaque South intake to Wanaque Reservoir.

Scenario	Mean (kg)	Standard deviation (kg)	10% probability of exceeding: (kg)
Actual	40,914	2207	44,010
TMDL implemented	9,504	646	10,490

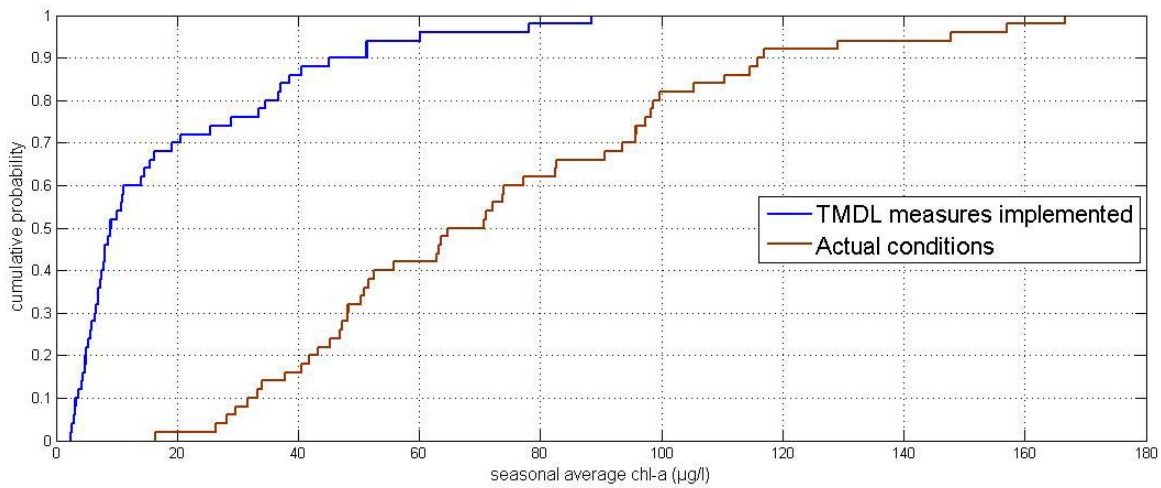

Figure 9. TMDL efficacy scenario: comparison of CDFs of seasonal average chl-*a* at Dundee Lake, June 15-August 31, 2002.

Table 10. TMDL efficacy scenario: seasonal average chl-*a* at Dundee Lake.

Scenario	Mean (µg/l)	Standard deviation (µg/l)	10% probability of exceeding: (µg/l)	Probability of exceeding 20 µg/l seasonal avg.:
Actual	73.0	35.3	116.4	0.98
TMDL implemented	18.9	20.1	48.2	0.29

Table 11. TMDL efficacy scenario: distribution of daily diurnal DO swing.

Location	Scenario	Mean (mg/l)	Standard deviation (mg/l)	10% probability of exceeding: (mg/l)
Dundee Lake	Actual	7.45	2.78	10.7
	TMDL implemented	2.27	1.34	4.1
Peckman River mouth	Actual	7.63	2.99	11.4
	TMDL implemented	5.43	2.15	8.3
Passaic River near Chatham	Actual	4.43	1.80	6.7
	TMDL	3.54	1.41	5.4

	implemented			

Table 12. TMDL efficacy scenario: distribution of annual average TP concentration at Little Falls intake.

Scenario	Mean (mg/l)	Standard deviation (mg/l)	10% probability of exceeding: (mg/l)
Actual	0.54	0.05	0.62
TMDL implemented	0.09	0.01	0.11

Stage 3: Trading scenarios

The seasonal average chl-*a* distribution of each trading scenario compared to the baseline was not significantly different (p>0.95 according to K-S test) (Figure 10). Each scenario including the baseline showed a 28% probability of exceeding the 20 µg/l target.

The diverted TP load distribution of trading scenarios 8 and 9 were each significantly different compared to the baseline (p=0 according to K-S test), while scenario 1 was not significantly different from the baseline (p=1 according to K-S test) (Figure 11). Scenarios 8 and 9 have lower loads because the WWTPs upstream of the Wanaque South intake are simulated as sellers. Each scenario showed a less than 10% probability of exceeding the 12,382 kg target.

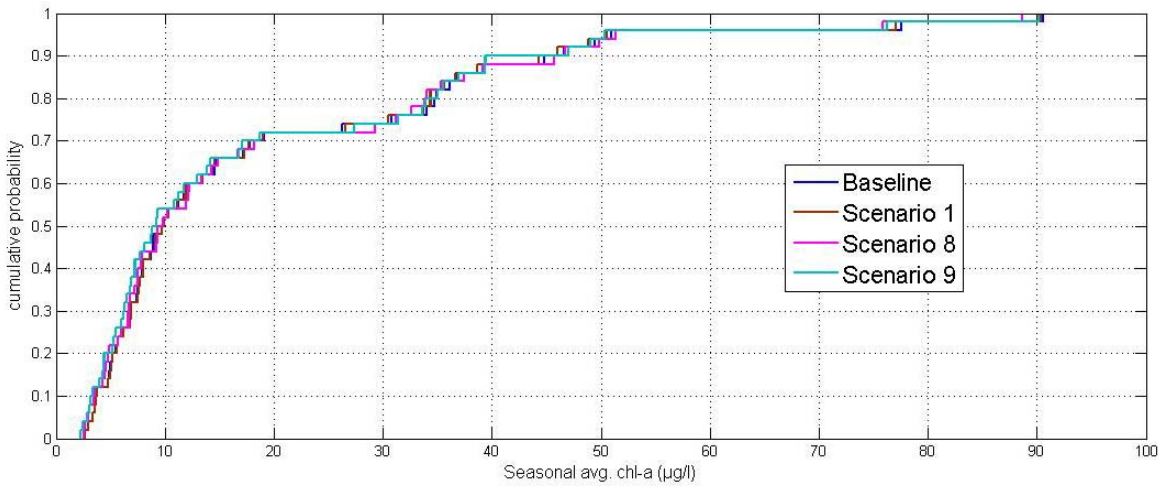

Figure 10. Trading scenarios: comparison of CDFs of seasonal average chl-*a* at Dundee Lake, June 15-August 31, 2002.

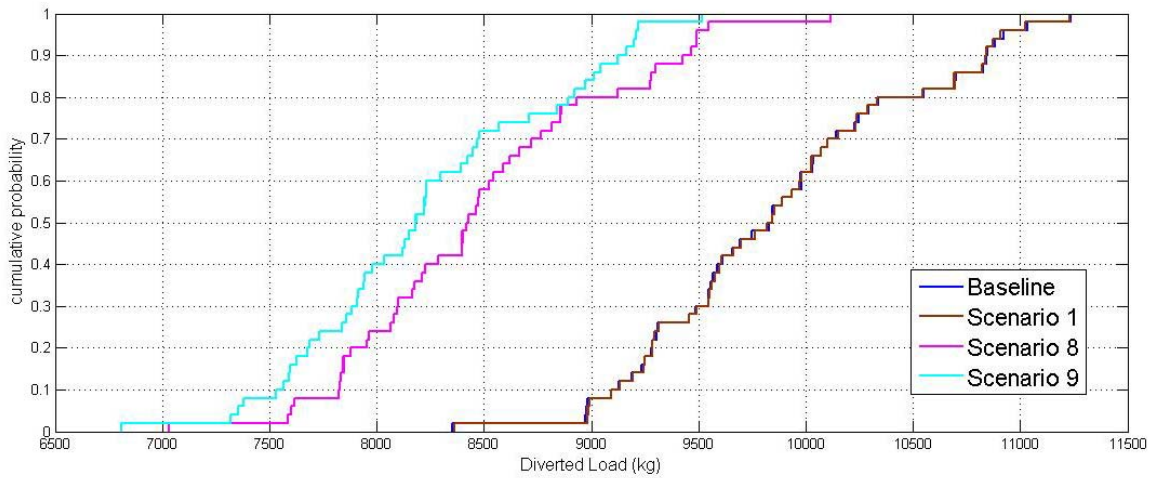

Figure 11. Trading scenarios: comparison of CDFs of TP diverted from Wanaque South intake to Wanaque Reservoir, WY2002.

The impact of all trading scenarios on DO at Dundee Lake showed no change from the baseline, and expected exceedances of daily average and minimum DO standards were zero in all cases. Scenarios where a buyer was upstream of either the Peckman River mouth or Passaic River near Chatham showed modest increases in expected exceedance of minimum DO at the location in question (increase from 0.07 to 0.11 at Peckman River mouth, 0.48 to 0.51 at Passaic River near Chatham). Remarkably, the Passaic River near Chatham showed a modest decrease in expected exceedance (from 0.28 to 0.22) of daily average DO when the buyer was upstream. This is because the added phosphorus discharged from the buyer stimulated algal growth which increased the diurnal DO swing and raised the daily average DO. Finally, results from Stage 1 suggest that expected exceedances at the Passaic River near Chatham are likely higher in each scenario than what was calculated in Stage 3.

The impact of all trading scenarios on TP at the Little Falls intake was equal to or slightly better than the baseline distribution of annual average concentrations (mean = 0.10 mg/l).

DISCUSSION and CONCLUSIONS

The credibility of the uncertainty analysis was assessed in Stage 1. Predictions of TP and chl-*a* were found to be credible at the Wanaque South intake and a location 3 km upstream of Dundee Lake, respectively. However, the credibility of the chl-*a* evaluation was based on a small sample size of observations. Predictions of DO were credible at all locations studied except the Passaic River near Chatham, where exceedances are likely underestimated throughout the analysis.

Although the TMDL scenario showed greater than 10% probability of exceeding TMDL targets for seasonal average chl-*a* at Dundee Lake and TP load diverted to the Wanaque Reservoir, the efficacy of the TMDL measures was clearly demonstrated when compared directly to actual conditions in WY2002. Significant improvements in chl-*a*, TP load diverted, and diurnal DO swing would have occurred in WY2002 if the TMDL measures

had been implemented. It is notable that in the TMDL efficacy scenario, the Passaic River near Chatham would have seen only marginal improvement in expected exceedance of the minimum DO standard, while at the same time seeing substantial improvement in diurnal DO swing. This suggests that the Passaic River near Chatham has naturally low DO and that management measures can only affect the magnitude of diurnal DO swing at this location.

Trading scenario simulations suggest that trading ratios have been well designed to account for differential fate and transport effects of phosphorus discharged throughout the watershed. Each scenario demonstrated parity with or improvement from the baseline at the TMDL critical locations of Dundee Lake and the Wanaque South intake. As expected, scenarios where the WWTPs upstream of the Wanaque South intake acted as sellers yielded significant decreases in TP load diverted; scenarios where WWTPs upstream of the Peckman River mouth or Passaic River near Chatham acted as buyers showed modest but acceptable increases in expected exceedance of minimum DO. Overall, trading scenario simulations indicated a low risk of hot spots compared to the baseline.

Future research will focus on analyzing the margin of uncertainty of additional trading scenarios under a wider range of trading conditions.

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