EFFECT OF WATER QUALITY MODEL UNCERTAINTY ON THE PASSAIC TOTAL MAXIMUM DAILY LOAD AND WATER QUALITY TRADING PROGRAM FOR TOTAL PHOSPHORUS

by

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A dissertation submitted to the Graduate School – New Brunswick

Rutgers, The State University of New Jersey

In partial fulfillment of the requirements

For the degree of

Doctor of Philosophy

Graduate Program in Environmental Sciences

Written under the direction of

Professor Christopher C. Obropta

And approved by

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New Brunswick, New Jersey

October, 2009

ABSTRACT OF THE DISSERTATION

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Professor Christopher C. Obropta

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 is the first study to examine the effects of water quality model uncertainty on a nutrient trading program. The method introduced in this study involved application of simple statistical tools to assess the credibility of the uncertainty analysis when compared to observed data. The method's efficiency and practicality directly address a main obstacle that has hindered a wider practice of uncertainty analyses of water quality models.

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 wastewater treatment plants (WWTPs). The TMDL margin of safety was found to be sufficient with respect to attaining dissolved oxygen (DO) surface water quality standards at Dundee Lake, and achieving a 70% reduction in

diverted TP load from the Wanaque South intake to the Wanaque Reservoir. Although the TMDL scenario showed greater than 10% probability of exceeding the target for chlorophyll-*a* (chl-*a*) at Dundee Lake, the efficacy of TMDL measures was clearly demonstrated when compared directly to actual conditions in the critical drought period of Water Year 2002. The uncertainty analysis found no evidence to suggest that the outcome of trades between WWTPs, as compared with command and control regulation, will significantly increase uncertainty in the attainment of DO surface water quality standards, site-specific chl-*a* criteria, and reduction targets for diverted TP load at affected potential hot spots in the watershed. Each simulated trading scenario demonstrated parity with or improvement from the baseline at the TMDL critical locations and low risk of hot spots elsewhere.

Finally, research on risk communication techniques was synthesized to help the New Jersey Department of Environmental Protection in its future public participation efforts on the Passaic water quality trading program. A strategy based on the principles of 'outrage management' was outlined for conducting a public meeting.

Acknowledgements

I would like to thank my advisor, Professor Obropta, for his boundless kindness and generosity, leadership, vision, and dedication to his students and staff. I am very fortunate to have received his guidance and support throughout my time in the Department of Environmental Sciences. His commitment to improving the environment and achieving real change sets an example I aspire to follow. I am grateful to Professor Goldfarb, Professor Strom, and Professor Uchrin for taking me under their wings since I first began here. It has been an honor to learn from and work with them. I appreciate the extra efforts Dr. Ferrara made to guide my research as the outside committee member.

In addition to my committee, I received invaluable help from other experts. I am grateful to Dr. Rob Miskewitz for his help in constructing the research methodology, and to Professor Green for further guidance in statistics questions. Professor Poe, Professor Boisvert and Yukako Sado of Cornell University taught me a great deal about environmental economics. Dr. Marcelo Cerucci and Tom Amidon of Omni Environmental Corporation provided a large amount of insight and assistance with the inner workings of the WASP model.

The Rutgers Cooperative Extension Water Resources Program staff has been a joy to work with and I will miss them very much. In particular, I am grateful to Lisa Evrard for helping me to improve my writing, and to Mehran Niazi for his deep friendship and his teaching me the value of the k.a. coefficient. I have kept my head above water and shared many good times with Sean Walsh, Greg Rusciano, Cheryl Burdick, Sandy Goodrow, Katie Giacalone, Eileen Althouse, Mike Mak, Steve Yergeau, Amy Boyajian, Jillian Thompson, Elaine Rossi, and Karan Bhandari of blessed memory.

I am indebted to my family for the tremendous support they have shown me on my journey back to school, and back to the place where I grew up. I could not have done this without the love and encouragement of my study buddy and soul mate April. My parents, Nelson and Paula, set a high standard for academic achievement and intellectual curiosity that continually inspires me. My sister Olivia has always been there for me with her warm spirit. Uncle Barry helped me to approach my dissertation with a calm perspective that saw me through hard times. And as my son to be kicked harder and harder in the womb, I typed faster and faster - thanks for pushing me when I needed it most.

I am blessed to have received the spiritual and emotional nurturing of the Gershman Y Congregation, Temple Brith Achim, Congregation Or Shalom, and the West Philadelphia Reevaluation Counseling community during my time of study.

This research would not have been possible without the Passaic WASP7 modeling software resources shared by Omni Environmental Corporation, and the ARRAMIS software shared by Sandia National Laboratories. The research described in this paper has been funded by USEPA under the Science to Achieve Results (STAR) Graduate Fellowship Program. EPA has not officially endorsed this publication, and the views expressed herein may not reflect the views of the EPA.

Finally, a portion of this research was previously published in the 81st WEFTEC Conference Proceedings. In the small number of cases where the results differ, the results reported in this dissertation supersede those reported in that article.

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Chapter 1: Introduction

1.1 The need for uncertainty analysis in water quality trading

Section 303(d) of the Clean Water Act (CWA) requires states to identify impaired water bodies that cannot meet ambient water quality standards. Regulators are then required to determine the total maximum daily load (TMDL) of pollutants. The TMDL calculates the maximum pollutant load that a water body can assimilate and still meet water quality standards, and then allocates allowable loads to point and nonpoint pollutant sources. A TMDL is thus akin to a "pollution budget." Loads from natural background sources and a margin of safety are accounted for in setting load allocations for point and nonpoint sources (Chen et al., 1999).

In terms of TMDL implementation, water quality trading (WQT) offers a management alternative to regulations that specify effluent levels or particular abatement technologies for each source of emissions (i.e., a command and control approach). WQT is a watershed-based and market-based approach that allows flexibility in individual emissions or abatement levels while meeting ambient water quality standards (Faeth, 2000). WQT is based on the premise that sources in a watershed can face very different costs to control the same pollutant. A trading program allocates a certain number of pollution allowances to sources in the watershed. The sources can either discharge under their allocation and sell their excess allowance or discharge over their allocation and purchase allowances. With appropriate restrictions on trade, the net effect will be to achieve targeted ambient levels of water quality throughout the watershed, ideally at a lower cost than requiring each individual source of emissions to implement pollution controls that comply with the TMDL reductions. Trading can occur among point sources and nonpoint sources. Depending on the market structure of the program, sources can trade directly or indirectly with each other (USEPA, 2004).

The U.S. Environmental Protection Agency (EPA) supports WQT and issued policy guidance in 2003 on trading (USEPA, 2003a). EPA support for WQT is driven by the expectations that trading can reduce the cost of TMDL implementation, and engage unregulated nonpoint sources such as farmers in reducing pollutant discharges to waterbodies (Grumbles, 2006; USEPA 2003a, 2004, 2006a). Despite the high hopes for WQT, it continues to progress at a slower pace than expected. Widespread efforts to develop local trading rules and guidelines have not raised the small number of actual trades taking place (King, 2005). A recent review found only four ongoing trading programs that have “experienced a large number of trades” (Morgan and Wolverton, 2005). The 2005 EPA Environmental Economics Research Strategy (EERS) reported that “existing attempts at TMDL trades have been difficult to establish and have not always been successful. TMDLs provide situations that are less clearly defined than the successful air pollution trades, both in terms of monitoring and with respect to pollutants” (USEPA, 2005, p. 2-6).

The EERS reference to “less clearly defined” TMDL situations acting as a hindrance to WQT success speaks directly to the problem of uncertainty. 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, across the science, policy, and economic dimensions. In terms of environmental science concerns, uncertainty about water quality models raises doubts that trading ratios (used to account for differential pollutant attenuation among sources) are adequate, or that trades will protect water quality and avoid creating hot spots, i.e., localized areas of degraded water quality (Nelson and Keeler, 2005). In terms of the social sciences, concerns about uncertainty are an institutional hindrance to WQT that deter stakeholder interest and involvement. A failure to characterize water quality model uncertainty impedes risk assessments and economic evaluations (Powers, 2006).

Uncertainty analyses "avoid the mistaken impression that assessments are precise and well understood" (Reckhow, 1994, p.1). It is a critical component in realistically estimating the benefits of environmental regulation (Krupnick et al., 2006). Environmental managers and the public need to know the expected uncertainty in assessed responses, rather than single point estimates, in order to better evaluate alternatives and guide future data collection efforts (Reckhow, 1994). The National Research Council (NRC) (2001) recommended that "uncertainty must be explicitly acknowledged both in the models selected to develop TMDLs and in the results generated by those models". This study contends that as with TMDLs, analysis of model uncertainty impacts on water quality trading programs is also necessary to provide better decision support to policy makers and increased transparency to affected stakeholders. Specifically, this research has examined a water quality trading program in development for the Non-Tidal Passaic River Basin (NTPRB) as a case study to analyze the effects of model uncertainty on the trading of total phosphorus (TP) between point sources.

1.2 Background of study area: Development of Passaic TMDL and WQT Program

The non-tidal portion of the Passaic River watershed encompasses 803 square miles, with 669 square miles of the watershed in New Jersey and the remainder in New

York (Figure 1-1). Approximately ¼ of New Jersey's population (i.e., two million people) lives in this watershed. Three of New Jersey's twenty watershed management areas (WMAs), WMAs 3, 4, and 6, are in the NTPRB. WMA 3 includes the Pompton, Pequannock, Wanaque, and Ramapo Rivers; WMA 4 includes the Lower Passaic and Saddle Rivers; and WMA 6 includes the Upper and Middle Passaic, Whippany, and Rockaway Rivers. In addition, 23 reservoirs, which provide potable water to the residents of New Jersey, are located within the NTPRB. The Wanaque Reservoir is the largest potable water source in the watershed, and it receives surface water from natural tributaries and from diversions of the Ramapo, Pompton, and confluence of the Pompton and Passaic Rivers in order to supply water to the North Jersey District Water Supply Commission (NJDWSC) and the Passaic Valley Water Commission (PVWC). Overall, about 50% of New Jersey's population receives drinking water from the NTPRB (NJDWSC, 2002a-c).

A TMDL (NJDEP, 2008a) for total phosphorus has been adopted and approved for the NTPRB. Surface water samples were collected at over 70 sampling stations within the watershed, including 24 wastewater treatment plants (WWTPs). For the TMDL study, a hydrodynamic model, nonpoint source load model, and a water quality model were developed for the NTPRB (Omni Environmental, 2007a). These models, coupled with the Najarian Associates (2005) model of the Wanaque Reservoir, were used to identify the Wanaque Reservoir and Dundee Lake as the critical locations where phosphorus is causing excessive primary productivity. As part of the TMDL, the New Jersey Department of Environmental Protection (NJDEP) proposed watershed criteria in accordance with N.J.A.C. 7:9B-1.5(g)3 in these locations as the best means to ensure protection of the designated uses. The watershed criteria were proposed in terms of a seasonal average concentration (June 15-September 1) of the response indicator, chlorophyll-*a* (chl-*a*). The proposed criteria were tailored to the unique characteristics of each critical location and were proposed as a seasonal average of 10 μg/L chl-*a* in the Wanaque Reservoir and a seasonal average of 20 μg/L chl-*a* in Dundee Lake (NJDEP, 2008a). These criteria each contain an implicit 10% margin of safety based on the underlying model predictions.

Phosphorus loading in the watershed is currently dominated by point sources, namely WWTPs. The wasteload allocations needed to meet the watershed criteria at Wanaque Reservoir and Dundee Lake were based on a long term average (LTA) effluent concentration of 0.4 mg/L of TP for all WWTPs. The 2007 discharger monitoring report (DMR) data illustrates that only 2 of the 22 main WWTPs discharged effluent with an LTA below 0.4 mg/L of TP (NJDEP, 2008b). There is an expected variance in the degree to which WWTPs discharging greater than LTA 0.4 mg/L TP can upgrade to comply with the new requirement, thus rendering the NTPRB as favorable for the implementation of a WQT program to achieve the TMDL. A water quality trading program, funded by USEPA Targeted Watershed Grant Agreement No. WS97284104-0, has been developed to increase the cost effectiveness of TMDL implementation (Passaic Trading Project, 2005). Twenty-two WWTPs, ranging in capacity from 0.1 to 16 million gallons per day (MGD), are expected to be the main trading participants (Table 1-1).

Extensive water quality modeling of the NTPRB has been completed by Omni Environmental Corporation (2007a), and their water quality model is hereafter referred to as the "TMDL model." (The TMDL model was linked to the Wanaque Reservoir model via the prediction of TP load diverted at the Wanaque South intake. The uncertainty of predicted TP load diverted at the Wanaque South intake was analyzed, which functions as a critical input to the Wanaque Reservoir model. However, the scope of this uncertainty analysis included only the TMDL model and not the Wanaque Reservoir model). The TMDL model is the basis for both establishing the TMDL allocations and implementing the TMDL via water quality trading. TMDL allocations for TP 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 the likelihood that TMDL allocations and trades of TP 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 (2003a and 2004) and critics of water quality trading (e.g., Steinzor, 2003). Model uncertainty analysis would yield an explicit approximation of the probability that the TMDL and phosphorus trades will have a positive impact on the NTPRB.

1.3 Literature review of uncertainty analysis

1.3.1 Typology of uncertainty

Water quality models, ranging from simple to complex in structure, represent the waterbody and/or watershed through mechanistic, empirical, or stochastic processes, and are applied to simulate waterbody responses to various pollutant loading scenarios. Water quality model predictions are critical in helping decision makers to establish

TMDL allocations to point and nonpoint sources (NRC, 2001). However, water quality models are imperfect representations of natural systems, and are subject to uncertainties. Several typologies of water quality model uncertainty have been discussed in the literature (Beck, 1987; Hession and Storm, 2000; Shirmohammadi et al., 2006). This study utilized the framework outlined in Yen et al. (1986) and Melching (1995) whereby four types of uncertainty are distinguished: a) uncertainty due to natural randomness (i.e., aleatory uncertainty), b) measurement uncertainty, c) uncertainty in model parameters and input values, and d) uncertainty of the model structure itself. Analysis of natural variability in contrast with parameter uncertainty is the subject of Walker (2003). The study of measurement uncertainty is often neglected although Shirmohammadi et al. (2006) highlight its importance. The study of model structure uncertainty is the most difficult and requires either a computationally demanding method such as Generalized Likelihood Uncertainty Estimation (Beven and Freer, 2001), or the comparison of predictions from several different models (Beck, 1987), neither of which is feasible in this case. While each area of uncertainty is important, and all are interrelated to some degree, this study focused on model parameter and input uncertainty and its effect on model output. As detailed in chapters 2 and 3, the study methodology was designed to evaluate if parameter and input uncertainty analysis provide an adequate approximation for overall model uncertainty.

1.3.2 Applications of uncertainty analysis

Model uncertainty analysis is widely acknowledged as essential for conducting reliable environmental decision making (Reckhow, 1994; NRC, 2001; Wu et al., 2006). A modeling framework that considers uncertainty can be applied to evaluate and rank feasible alternatives based on their risks of exceeding the target water quality criteria (Wu et al., 2006). Previous water quality model uncertainty analyses have generally focused on model predictions of observed data (e.g., Carroll and Warwick, 2001; Abrishamchi et al., 2005; Muleta and Nicklow, 2005) and alternate scenarios such as best management practices (Wu et al., 2006; Arabi et al., 2007), critical low flow (Melching and Yoon, 1996) and reduced pollutant loading (Borsuk et al., 2002; Zhang and Yu, 2004). Interestingly, 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 water quality 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.

Past uncertainty analyses of alternate scenarios are particularly relevant to the research objectives, which explore the uncertainty of TMDL and trading scenarios. A key question rarely asked is “how credible is the uncertainty estimate of alternate 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 are often predicted for alternate scenarios without first comparing the uncertainty analysis to available observed data (e.g., Melching and Yoon, 1996; 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 introduced in this study 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. Furthermore, the study produced not only a credible uncertainty analysis, but an efficient analysis whose method could easily be replicated by regulators charged with administering a water quality trading program and assessing its various risks.

1.3.3 Methods of uncertainty analysis

> Uncertainty analysis is the computation of the total uncertainty induced in the output by quantified uncertainty in the inputs and models, and the attributes of relative importance of the input uncertainties in terms of their contributions. Failure to engage in systematic sensitivity and uncertainty analysis leaves both analysts and users unable to judge the adequacy of the analysis, and the conclusions reached – Morgan and Henrion (1990, p.39).

An uncertainty analysis provides a probabilistic range of model output, rather than a single-value fixed model output (see Figure 1-2). There are two general methods in water quality model uncertainty analysis: first-order error analysis (FOEA), and Monte Carlo simulation (MCS) (Chapra, 1997). MCS is akin to a "brute force" approach, and is a much more robust method than FOEA (Summers et al., 1993). MCS involves sampling from the probability distribution of each uncertain parameter in order to obtain a probability distribution of model output. In contrast with the classical branch of statistics, MCS is derived from the Bayesian branch, and is based on propagation of a priori probability distributions; model parameters themselves are random variables sampled from an a priori probability distribution (Omlin and Reichert, 1999). MCS can analyze non-linear systems and non-normal input distributions, but the subjective choice of parameter distributions and lengthy computations are a criticism of this method (Zhang and Yu, 2004).

In contrast, FOEA is computationally efficient and provides a clear approach to uncertainty analysis by decomposing the variance of each output into the sum of contributions from each input. FOEA is typically conducted by calculating the first two terms of the Taylor series expansion of the model output function, where the expansion point is the mean value of the parameter set. Thus the mean and variance of the model output can be approximated. It is easy to update the risk estimation with FOEA when new information becomes available (Yen et al., 1986; Morgan and Henrion, 1990; Melching and Yoon, 1996; Zhang and Yu, 2004). However FOEA has several limitations. It assumes the system being studied can be approximated by a first order linearization. This may be a poor assumption for highly non-linear systems. In addition, FOEA assumes no parameter covariance, and FOEA sensitivity coefficient values are highly dependent on the magnitude of perturbation (Maskey and Guinot, 2003). FOEA is also not suitable when the parameter coefficients of variation exceed 20% (Tyagi and Haan, 2001), or when the parameter distributions have skewed tails (Summers et al., 1993). Although some studies have demonstrated that FOEA can produce results that are satisfactorily similar to MCS (Bobba et al., 1996; Melching and Yoon, 1996), Sohrabi et al. (2003) found that MCS can yield mean output values that are very different from a FOEA approximation.

The computational efficiency of MCS can be vastly improved by using stratified sampling methods (Cullen and Small, 2004; Krupnick et al., 2006). An example is Latin Hypercube Sampling (LHS), in which input probability distributions are broken into n non-overlapping ranges of equal probability, where n equals the number of times the model will be run. A single value is then selected n times from each range of each uncertain parameter without replacement and randomly combined with n samples from each of the other uncertain parameters to form n random parameter sets; every possible combination of parameter values is equally likely unless restricted pairing is imposed (McKay et al., 1979, Helton and Davis, 2000; Shirmohammadi et al., 2006) (Figs. 1-3 and 1-4). The output distribution and statistics can be obtained from the sample of n output values. Iman and Helton (1985) advised that successful LHS can be achieved with $n \geq (4/3) \cdot$(number of uncertain parameters). Thus, LHS involves fewer simulations than both MCS and FOEA; FOEA requires that $n \geq (2) \cdot$(number of uncertain parameters). LHS is more practical than MCS and more robust than FOEA (Cullen and Frey, 1999; Morgan and Henrion, 1990). Helton and Davis (2002) found LHS to produce more stable results than MCS.

LHS was originally developed in the 1970s by staff at Los Alamos Scientific Laboratory to analyze the uncertainty and reliability of nuclear reactors (Helton and Davis, 2003). LHS has since become a popular method of uncertainty analysis and is utilized in diverse applications spanning water quality modeling in Belgium (Melching and Bauwens, 2001) to radioactive waste disposal at Yucca Mountain (USDOE, 1998). Iman and Conover (1982) pioneered the method of “restricted pairing”, which accounts for parameter covariance and limits the parameter sample space accordingly in LHS and other random sampling methods. Thus, the parameter sample space and resulting uncertainty analysis are not overestimated. DiToro and van Straten (1979) demonstrated that failure to account for parameter covariance can lead to large overestimations of model uncertainty.

Method efficiency is a critical need in uncertainty analysis of complex mechanistic models (Chapra, 2003) such as the TMDL model. Conventional 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 LHS with restricted pairing (Iman and Conover, 1982) as the main method of uncertainty analysis. Chapter 2 provides an overview of the study methodology.

1.4 Study objectives

The primary objective of this study was to identify how model uncertainty affects model outputs and decision risks related to the Passaic TMDL (NJDEP, 2008a) and potential trades of TP between WWTPs. A secondary objective was to demonstrate that uncertainty analysis of water quality models is an essential step for the development of future water quality trading programs.

The following hypotheses were tested by model uncertainty analysis:

1. The Passaic TMDL will result in attainment of dissolved oxygen surface water quality standards and site-specific chlorophyll-*a* criteria at Dundee Lake, with less than 10% expected exceedance and 10% exceedance probability, respectively, at critical drought conditions;

2. The Passaic 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 intake, with less than 10% exceedance probability;

3. The outcome of trades between WWTPs, as compared with command and control regulation, will significantly increase uncertainty in the attainment of dissolved oxygen surface water quality standards, site-specific chlorophyll-*a* criteria, and reduction targets for diverted total phosphorus load at affected potential hot spots in the watershed.

The 10% exceedance probability and 10% expected exceedance values were chosen as thresholds in order to correspond with the 10% value suggested by EPA water quality guidance documents as a tolerable exceedance frequency (USEPA, 1997).

1.5 Research contributions to the field

There are three main contributions this research has made. First, it has provided a water quality model uncertainty analysis of a nutrient trading program, of which no examples to date can be found in the literature. Second, it has introduced a simple and efficient method to assess the credibility of an uncertainty analysis. The method's efficiency and practicality directly address a main obstacle that has hindered a wider practice of uncertainty analyses of water quality models (Chapra, 2003; Stow et al., 2007). Finally, it should be noted that trades in the study area are primarily anticipated to occur between point sources, e.g. WWTPs. Since there is almost no agriculture in the watershed, trades with nonpoint sources are not expected. The fact that only point-point source trade scenarios were studied is useful. Consider that trades with nonpoint sources are generally likely to contain more uncertainties than trades with point sources. By conducting a thorough uncertainty analysis of a point-point source nutrient trading program, a lower bound on the range of uncertainty regarding nutrient programs in general has been obtained, which could benefit nutrient trading programs nationwide.

Table 1-1: Twenty-two main wastewater treatment plants located in the Non-Tidal Passaic River Basin (Data source: NJDEP, 2008b)

Wastewater Treatment Plant	**2005-2007 avg. flow (MGD)**	**Permitted flow (MGD)**	**2007 avg. total phosphorus effluent concentration (mg/L)**
Berkeley Heights STP	1.58	3.1	0.48
Bernards Twp STP	1.71	2.5	2.80
Caldwell STP	3.92	4.5	1.62
Cedar Grove STP	1.49	2	1.58
Chatham Twp/ Chatham Glen STP	0.12	0.15	3.54
Florham Park SA	0.91	1.4	1.24
Hanover SA	2.07	4.6	0.81
Livingston Twp STP	2.21	4.6	3.04
Long Hill Township STP	1.05	0.9	2.68
Molitor Water Pollution (Madison-Chatham) STP	2.49	3.5	3.70
Morris Township - Butterworth STP	2.01	3.3	1.37
Morristown STP	2.58	6.3	0.37
Parsippany - Troy Hills RSA	12.57	16	3.55
Pompton Lakes STP	0.90	1.2	0.37
Rockaway Valley SA	10.51	12	1.52
Two Bridges SA	5.80	10	0.95
Verona STP	2.37	3	2.85
Wanaque Valley RSA	1.06	1.25	0.12
Warren Stage I-II STP	0.38	0.47	2.02
Warren Stage V STP	0.17	0.38	3.13
Warren Township SA Stage IV STP	0.30	0.8	2.19
Wayne Twp STP	8.22	13.5	1.98

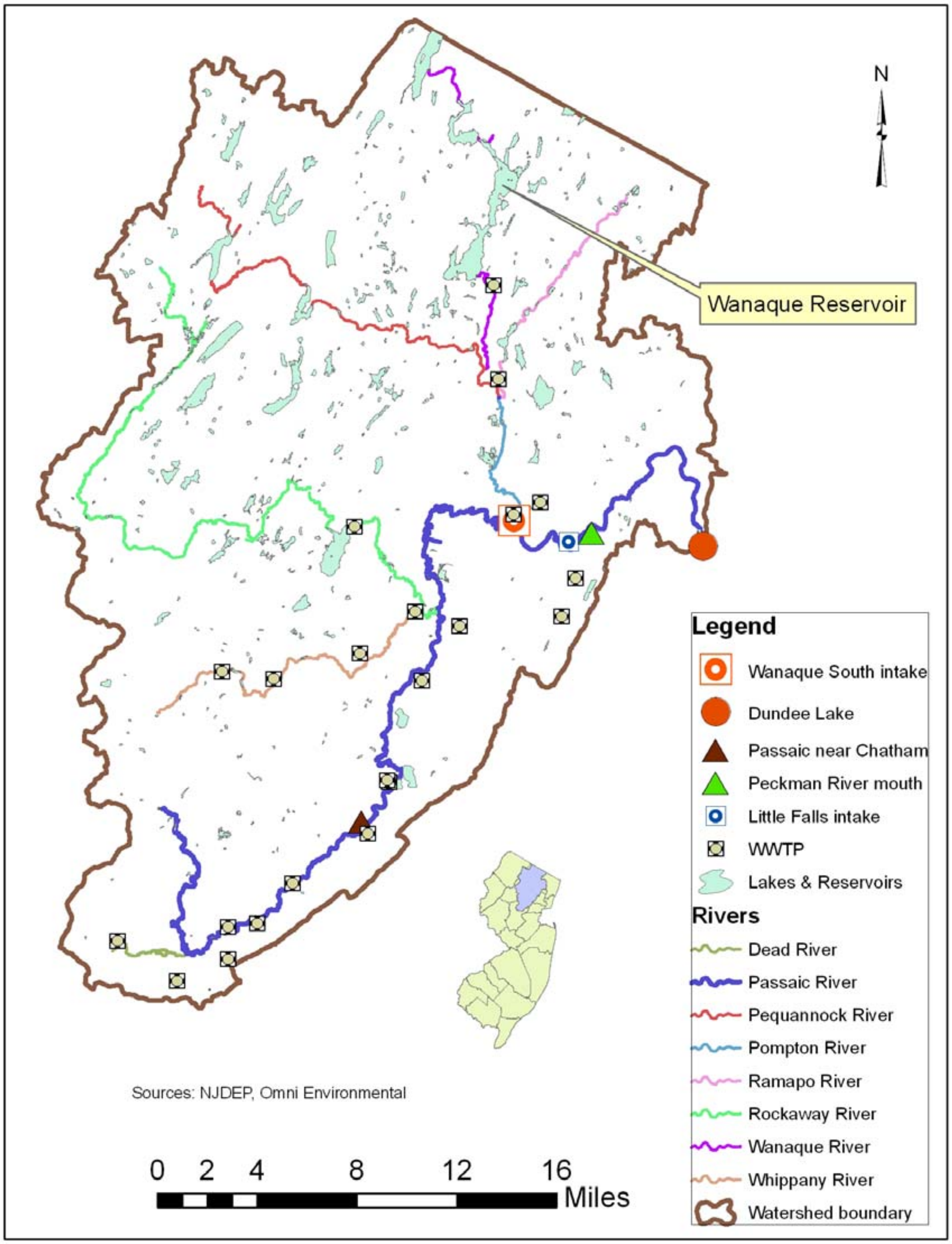

Figure 1-1: Non-Tidal Passaic River Basin, New Jersey

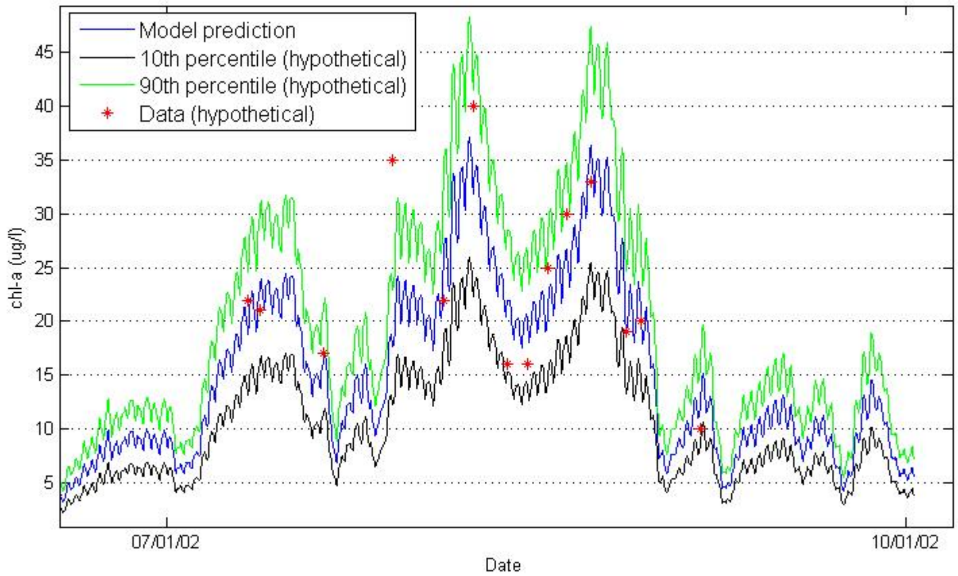

Figure 1-2. The fixed value model prediction is usually only shown, but an estimate of model uncertainty is more realistic. The predicted margin of uncertainty should capture most of the observed data.

Figure 1-3a shows a non-stratified probability distribution, suitable for Monte Carlo simulation. Figure 1-3b shows a stratified probability distribution, suitable for Latin Hypercube Sampling. (Adapted from Wyss and Jorgensen, 1998).

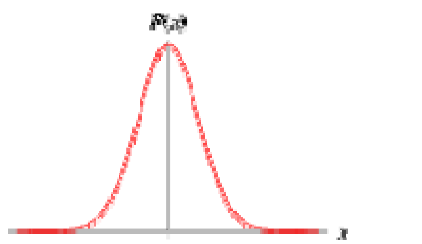

Figure 1-3a

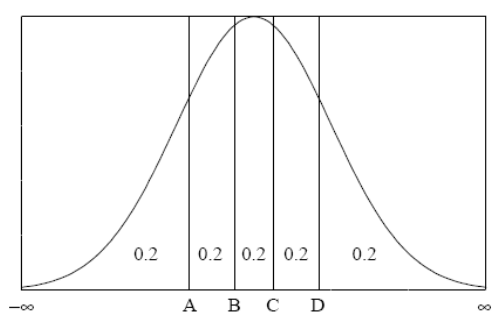

Figure 1-3b

Chapter 2: Methodology overview

This chapter provides an overview of the study methodology. Further details of the procedure for each of the three stages of the analysis are provided in Chapters 3 through 5, respectively.

2.1 Description of the TMDL model

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). The WASP modeling schematic is shown in Figures 2-1 and 2-2. The fundamental WASP modeling equations, expressed in Wool et al. (2003) and Ambrose et al. (2006), govern the dynamic relationships between the photosynthesis, respiration, growth and death of free floating and attached algae, light and nutrients that limit growth (the latter according to Monod kinetics), temperature, and DO in the water column and/or benthos. DO is also consumed by sediment oxygen demand (SOD), carbonaceous and nitrogenous biochemical oxygen demand (CBOD and NBOD), and replenished by reaeration from the atmosphere as a function of water depth and velocity. Nutrients are partitioned into dissolved and particulate fractions, and organic and inorganic species. Detrital processes include settling and mineralization of organic matter. A copy of the TMDL model was obtained from Omni Environmental Corporation in October 2006. The TMDL model network is shown in Figure 2-3. The TMDL model simulation timeframe was October 1, 1999 through November 30, 2003.

2.2 Sources of uncertainty in the TMDL model

The TMDL model is complex, sophisticated and state of the art (Rutgers EcoComplex, 2006). Nevertheless, like all models it contains numerous sources of uncertainty. In fact, a high number of model parameters such as in the TMDL model serves to increase rather than decrease model uncertainty (Doherty and Johnston, 2003). The TMDL water quality model is also linked to separate hydrodynamic and nonpoint source load models. Notable sources of uncertainty in the three models are described in Omni Environmental (2007a) and are categorized and listed below.

In the hydrodynamic model, sources of uncertainty include: the mixing algorithm used for diversion simulation at the Wanaque South (WS) intake; variability of discharger flow; hydraulic geometry; nonpoint source flows; spatial aggregation due to the placement of tributary network nodes; and the size of the time step, i.e., temporal aggregation.

In the nonpoint source load model, sources of uncertainty include: calculation of runoff load; calculation of baseflow load; and subbasin delineation and aggregation.

In the water quality model, sources of uncertainty include: lumping of macrophytes and benthic algae; lumping of phytoplankton species; the absence of zooplankton; the absence of an explicit link between organic matter deposition and SOD; the absence of an explicit link between low DO and nutrient recycling; spatial aggregation due to tributary network segmentation; size of the time step, i.e., temporal aggregation; one-dimensional system representation; waterfall load of DO at Little Falls and Great Falls; global kinetic parameter values; local kinetic parameter values;

phosphorus boundary conditions at the Passaic River and Ramapo River headwaters; and phosphorus boundary conditions at WWTP discharge points.

In addition, there are measurement uncertainties in the observed data of water quality parameters (e.g., TP, chl-*a*, DO), stream temperature, and solar radiation.

2.3 Scope of uncertainty analysis

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 was feasible to analyze using the methods described below. It was 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 items were outside the scope of the uncertainty analysis, although with the exception of the last item (i.e., LAWATERS model), a statistical comparison of observations to uncertainty estimates evaluated the decision to exclude these items. The hydrodynamic model uncertainty was not considered due to practicality, and because its predictions closely matched observations throughout the watershed. In addition, the hydrodynamic model parameters were generally based on a large quantity of data, thus mitigating uncertainty. The nonpoint source load model uncertainty was not considered due to practicality and because the Passaic system is dominated by point sources, even in a wet year such as Water Year (WY) 2003 (Omni Environmental, 2007a). Model structure uncertainty was not considered due to practicality. Measurement uncertainty was not considered because it was assumed to be smaller than water quality model uncertainty. In addition, the proprietary LAWATERS model (Najarian Associates, 2005) of the Wanaque Reservoir was outside the scope of the uncertainty analysis.

2.4 Application of Latin Hypercube Sampling

As noted in Chapter 1, method practicality is a key concern in uncertainty analysis of complex models (Chapra, 2003) such as the TMDL model. This study applied LHS with restricted pairing (Iman and Conover, 1982) as the main method of uncertainty analysis. The LHS samples 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).

Table 2-1 lists the parameters included in the first stage of the uncertainty analysis, which compared the uncertainty estimates to observed data. Selection was based on Omni Environmental (2007a), a WASP uncertainty analysis conducted by Lindenschmidt (2006), and a preliminary local sensitivity analysis of the TMDL model. Twenty-six kinetic parameters (sixteen global and ten local parameters) and nine boundary conditions for phosphorus were selected.

In the second and third stages of the uncertainty analysis, which examined the TMDL and water quality trading scenarios, 11 of the 26 kinetic parameters in Table 2-1 were instead modeled as fixed values based on global sensitivity analysis findings from the first stage. Phosphorus boundary conditions of 14 other WWTPs were added to the uncertainty analyses of the TMDL and trading scenarios to reflect the specific uncertainties of WWTP phosphorus effluent levels in the alternate scenarios. A summary of the parameter types included in the uncertainty analysis is shown in Table 2-2.

Separate probability distributions were estimated for each of the parameters listed in Table 2-1. 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 SOD the distribution properties were based on available data reported in Omni Environmental (2007a), and for other kinetic variables the distribution properties were based when possible on available literature (DiToro and van Straten, 1979; Scavia et al., 1981; Bowie et al., 1985; Brown and Barnwell, 1987; Chapra, 1997; Manache and Melching, 2004; Lindenschmidt, 2006). Through interpretation of the fundamental WASP model equations (Wool et al., 2003; Ambrose et al., 2006), certain kinetic parameters were assumed to have covariance with a correlation coefficient of either (+) or (–) 0.5; DiToro and van Straten (1979) suggested the sign of the correlation is more important than the magnitude. 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 (2007a). Full details are provided in Chapters 3 through 5.

The equation suggested by Iman and Helton (1985) to determine an adequate number of model runs in an LHS procedure is

$$n \geq (4/3) \cdot x \qquad \text{(Eq. 2.1)}$$

where

n = number of model runs, and

x = number of uncertain variables.

With x = 35, the TMDL model was run 50 times in each of the three stages of the uncertainty analysis.

2.5 Three stage approach

As alluded to above, 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.

The purpose of Stage 1 was to examine the credibility of the uncertainty analysis before extending it to alternate scenarios such as the TMDL and potential trades. This step is frequently omitted in uncertainty analyses of alternate scenarios of complex water quality models (e.g., Melching and Yoon, 1996; Zhang and Yu, 2004). Stage 1 also served to evaluate the assumption that the scope of the uncertainty analysis was adequate and the sources of uncertainty considered are the primary sources of uncertainty.

To make the uncertainty analysis more manageable, the simulation timeframe was shortened by excluding WY2000 (10/1/00 – 9/30/01). This left WY2001, WY2002, and WY2003 within the uncertainty analysis timeframe. Of the four water years used to calibrate and validate the TMDL model, WY2002 represents an extreme drought period; chl-*a* concentrations and diverted phosphorus loads at the WS intake were highest in WY2002. WY2001 contains a normal drought period, while WY2003 represents a wet period. Therefore the uncertainty analysis captured a wide range of hydrologic conditions in the watershed (Table 2-3).

In Stage 1, the performance of the uncertainty analysis was tested against observed data (compiled in Omni Environmental, 2007a) from WY2001 through WY2003 for the parameters and locations listed in Table 2-4. Since the Passaic TMDL bases seasonal average chl-*a* criteria on the period of June 15-September 1, only data from this time span was used when comparing chl-*a* and DO predictions. However, since TP is diverted at the WS intake almost year-round, TP data from the entire year was used to evaluate predictions. The statistical procedure for evaluating the uncertainty estimates against observed data is provided in Chapter 3; it essentially consisted of comparing the proportion of observations that fell inside the predicted 80% confidence intervals against the expected proportion of success.

In Table 2-4, locations 1, 3, 4 and 8 are considered potential hot spots in the watershed due to the negative impact of high phosphorus levels on DO, and locations 2, 5 and 6 are listed because of the increased cost of drinking water treatment associated with high phosphorus levels, as described in Omni Environmental (2007a) and Obropta et al. (2008). Location 7 is included in Stage 1 because of high measurements of chl-*a* there. Location 9 is included in Stage 1 because of its proximity to Dundee Lake.

A global sensitivity analysis performed at the end of Stage 1 identified key input variables that affected model output uncertainty. Input variables that were not sensitive were removed from the uncertainty analysis for Stages 2 and 3.

Stage 2 focused on the uncertainty of the TMDL scenario at future critical drought conditions in which all WWTPs discharge a LTA of 0.4 mg/L TP effluent at permitted flows, and nonpoint source loads are reduced by 60%. Besides the sensitive 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. Stage 2 outputs were produced to test Hypotheses 1 and 2 in the study objectives (i.e., calculate the probability of attaining chl-*a*, DO, and diverted TP load targets at key locations).

Stage 3 focused on the uncertainty of five trade scenarios and three baseline scenarios. Three baseline scenarios were necessary in order to reflect each of the three general diversion conditions in the watershed, as described in Obropta et al. (2008). The same variables that were modeled probabilistically in Stage 2 were repeated in Stage 3. In Stage 3, WWTP boundary conditions for phosphorus were adjusted so that buyers discharged LTA TP concentrations greater than 0.4 mg/L and sellers discharged less than 0.4 mg/L. Stage 3 outputs were produced to test Hypothesis 3 in the study objectives (i.e., compare the uncertainties of trading and no-trading approaches in attaining chl-*a*, DO, and diverted TP load targets at key locations).

Table 2-1: WASP Parameters included in Stage 1 of the uncertainty analysis

<table>
<tr><th colspan="12">Global kinetic parameters [unit] (type of probability distribution) [a]</th></tr>
<tr><td colspan="3">Phytoplankton maximum growth rate @ 20°C [/d] (Normal)</td><td colspan="3">Phytoplankton carbon: chlorophyll ratio (Triangular)</td><td colspan="3">Phytoplankton endogenous respiration rate @ 20°C [/d] (Triangular)</td><td colspan="3">Phytoplankton death rate, non-zooplankton predation [/d] (Triangular)</td></tr>
<tr><td colspan="3">Phytoplankton optimal light saturation [langleys/d] (Triangular)</td><td colspan="3">Phytoplankton half-saturation constant for nitrogen [mgN/L] (Normal)</td><td colspan="3">Phytoplankton half-saturation constant for phosphorus [mgP/L] (Triangular)</td><td colspan="3">Benthic algae maximum growth rate @ 20°C [$gD/m^2/d$] (Triangular)</td></tr>
<tr><td colspan="3">Benthic algae respiration rate @ 20°C [/d] (Lognormal)</td><td colspan="3">Benthic algae death rate @ 20°C [/d] (Beta)</td><td colspan="3">Benthic algae ammonia preference [mgN/L] (Beta)</td><td colspan="3">Benthic algae light constant for growth [langleys/d] (Beta)</td></tr>
<tr><td colspan="3">Benthic algae nitrogen half-saturation constant for growth [mgN/L] (Normal)</td><td colspan="3">Benthic algae phosphorus half-saturation constant for growth [mgP/L] (Beta)</td><td colspan="3">Mineralization rate of dissolved organic phosphorus @ 20°C [/d] (Normal)</td><td colspan="3">Nitrification rate @ 20°C [/d] (Normal)</td></tr>
<tr><th colspan="12">Local kinetic parameters [unit] (type of probability distribution) [a]</th></tr>
<tr><td colspan="3">Sediment oxygen demand [$g/m^2/d$] (Normal)</td><td colspan="6">Settling velocity of particulate phosphorus [cm/s] (Normal) [b]</td><td colspan="3">Fraction of bottom segment covered with benthic algae (Normal) [c]</td></tr>
<tr><td colspan="3">Dissolved fraction of orthophosphate at B12-N38 (Normal)</td><td colspan="3">Dissolved fraction of orthophosphate at B17-N5 (Normal)</td><td colspan="3">Dissolved fraction of orthophosphate at B16-N19 (Normal)</td><td colspan="3">Dissolved fraction of orthophosphate at B17-N29 (Normal)</td></tr>
<tr><th colspan="12">Headwater boundary conditions (type of probability distribution)</th></tr>
<tr><td colspan="6">Passaic River headwater scaling factor for phosphorus (Normal)</td><td colspan="6">Ramapo River headwater scaling factor for phosphorus (Normal)</td></tr>
<tr><th colspan="12">WWTP effluent [mg/L] (type of probability distribution)</th></tr>
<tr><td colspan="3">Two Bridges: orthophosphate (1 cluster Normal, 2 clusters Lognormal) [d]</td><td colspan="3">Two Bridges: organic phosphorus (Lognormal) [d]</td><td colspan="3">Verona: orthophosphate (Lognormal) [d]</td><td colspan="3">Berkeley Heights: organic phosphorus (Normal) [d]</td></tr>
<tr><td colspan="4">Berkeley Heights: orthophosphate (Normal) [d]</td><td colspan="4">Rockaway Valley: orthophosphate (1 cluster Normal, 2 clusters Lognormal) [d]</td><td colspan="4">Rockaway Valley: organic phosphorus (1 cluster Normal, 2 clusters Lognormal) [d]</td></tr>
</table>

[a] Available references for specific parameter probability distributions provided in Chapter 3, Tables 3-2 through 3-4.

[b] Variable was modeled separately at multiple locations, yielding a total of three variables, as detailed in Chapter 3.

[c] Variable was modeled separately at multiple locations, yielding a total of two variables, as detailed in Chapter 3.
[d] According to analysis of effluent data, as detailed in Chapter 3.

Table 2-2: Summary of WASP parameters included in uncertainty analysis

Category	Number of parameters, Stage 1	Number of parameters, Stages 2 and 3
Global kinetic parameters	16	8
Local kinetic parameters	10	7
Headwater boundary conditions	2	2
WWTP effluent	7	18
Total	**35**	**35**

Table 2-3: Hydrologic and water quality characteristics of Stage 1 uncertainty analysis timeframe

Water Year	Annual discharge at USGS Gage 01381900[a] (Passaic River at Pine Brook) [m^3]	TP load diverted from WS intake to Wanaque Reservoir[b] [kg]	Seasonal average chl-*a* at Dundee Lake[b] [μg/L]
2001	492,114,211	9,767	41.9
2002	229,213,565	42,975	73.6
2003	802,280,506	7,581	8.0

[a] U.S. Geological Survey (USGS) Gage 01381900 selected because of its minimal impact from surface water diversions which would otherwise skew the annual discharge figures
[b] Based on fixed-value simulation of the TMDL model (Omni Environmental, 2007a)

Table 2-4: Stage 1 Uncertainty analysis output (locations shown in Figure 2-3)

Location / Branch-node / segment number	Key water quality parameter(s)	Significance	Period of available data (data source)
1. Dundee Lake (watershed outlet) / B17-N29 / seg. 327	chl-*a*, DO	TMDL critical location (NJDEP, 2008a) 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 in NJ) / B5-N23 / seg. 54	TP	TMDL critical location (NJDEP, 2008a) due to potential effect of diverted water with high TP to stimulate algal blooms in reservoir	2003 (Omni)
3. Peckman River mouth / B16-N19 / seg. 298	DO	Area of concern (Omni Environmental, 2007a) due to low DO and high diurnal DO swing	2003 (Omni)
4. Passaic River near Chatham / B11-N33 / seg. 181	DO	Area of concern (Omni Environmental, 2007a) due to low DO	2002 (NJDEP)
5. Little Falls intake / B15-N10 / seg. 275	TP	Area of concern (Omni Environmental, 2007a) due to effect of TP on drinking water treatment	Data unreliable
6. Passaic River at confluence with Pompton River / B12-N38 / seg. 256	TP	High TP poses risk during extreme drought if Upper Passaic River flow is diverted to Wanaque Reservoir	TP: 2000-2003 (USGS, Omni)
7. B17-N4 / seg. 302	chl-*a*	High chl-*a*	chl-*a*: 2001-2003 (Omni, PVSC[a])
8. Station PA10 (~ 5 km upstream of Dundee Lake) / B17-N20 / seg. 318	DO	High chl-*a*, high diurnal DO swing	DO: 2003 (Omni)
9. B17-N25 (~ 3 km upstream of Dundee Lake) / seg. 323	chl-*a*	High chl-*a*	chl-*a*: 2001-2003 (PVSC[a])

[a] Passaic Valley Sewerage Commissioners

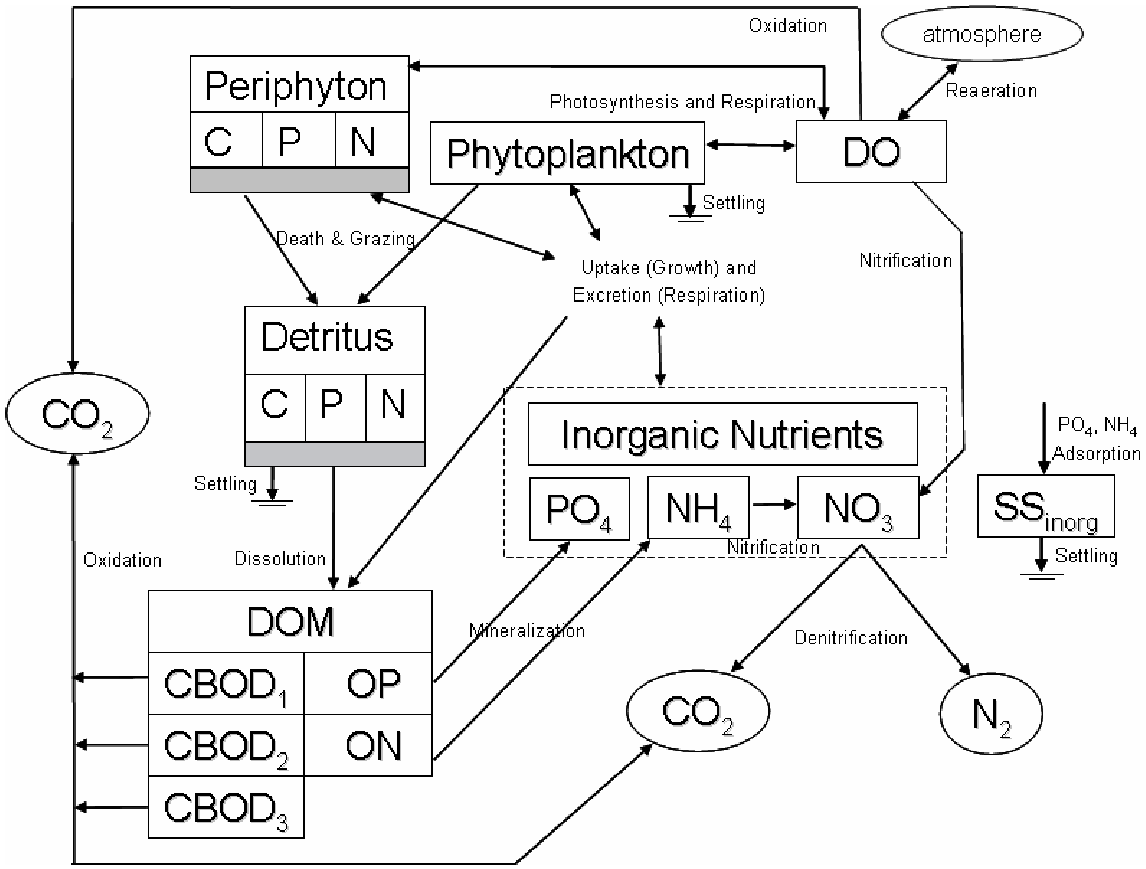

Figure 2-1: WASP7 Eutrophication Model Schematic (from Ambrose et al., 2006)

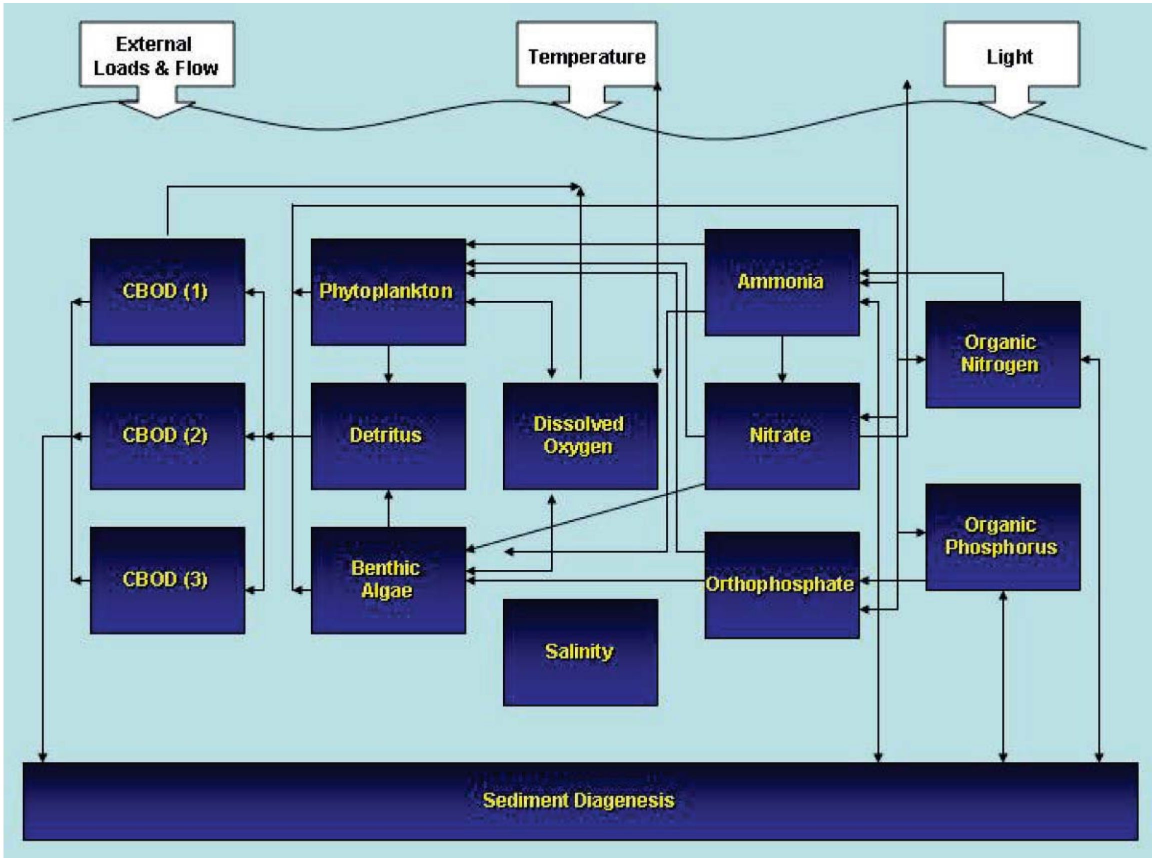

Figure 2-2: Alternate representation of WASP model schematic (from USEPA, 2006b)

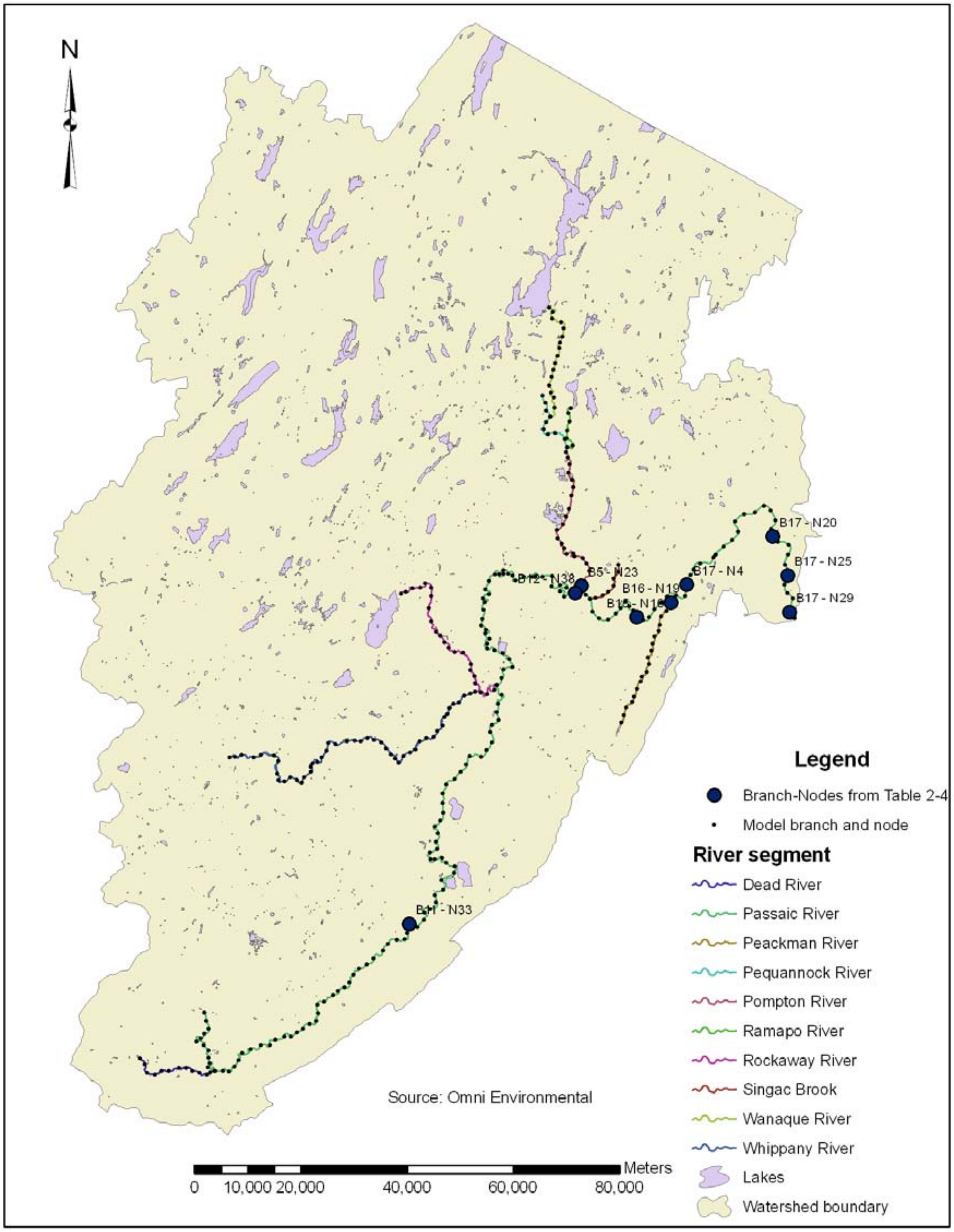

Figure 2-3: Omni Environmental (2007a) model segmentation network. Branch-nodes listed in Table 2-4 are highlighted**.**

Chapter 3: Uncertainty analysis of actual conditions (Stage 1)

3.1 Introduction

In Stage 1, the credibility of the uncertainty estimates were assessed through comparison to observed data. This chapter explains in detail the methodology of applying LHS to generate multiple input variable sets and obtain a probabilistic model output, which was then compared with observed TP, chl-*a* and DO data at key locations.

3.2 Methodology

3.2.1 Application of LHS to select input variables

As described in Chapter 2, the scope of the uncertainty analysis was restricted to four types of variables: select i) global and ii) local kinetic parameters, and select iii) headwater and iv) point source boundary conditions.

3.2.2 Selection of variables

The TMDL model contains 52 kinetic parameters that were calibrated and 288 boundary condition variables that were either estimated or entered directly from available data (Omni Environmental, 2007a). The tally of kinetic parameters increases considerably if local parameters are counted separately with each spatial variation. To make the uncertainty analysis feasible, only a fraction of these input variables were included in the analysis. The selection was guided by interpretations of Omni Environmental (2007a) and a WASP uncertainty analysis conducted by Lindenschmidt (2006), and a preliminary screening of the TMDL model via a local sensitivity analysis. The local sensitivity analysis was based on incrementally adjusting one variable at a time and comparing the degree of change in the model output of interest (i.e., chl-*a*, DO, or TP depending on the location). Through local sensitivity analysis, the variables in Table 3-1 were found to be of negligible sensitivity and were excluded from the uncertainty analysis. Ultimately, 26 kinetic parameters (15 global and 11 local parameters) and 9 boundary conditions for phosphorus were selected.

3.2.3 Global kinetic parameters

Table 3-2 lists the 16 global kinetic parameters that were modeled as uncertain in Stage 1. Separate probability distributions were estimated for each of the parameters and based, if possible, on available literature. Using literature values to estimate probability distributions is an established approach in water quality model uncertainty analysis when local data of the uncertain parameter is unavailable (e.g., Yoon and Melching, 1996; Zhang and Yu, 2004; and Wu et al., 2006). The calibrated values of the TMDL model served as mean values for symmetric distributions and mode or geometric mean values for skewed distributions.

The probability density functions (PDFs) for each variable in Table 3-2 are shown in Figures 3-1 through 3-16.

In the case of phytoplankton parameters, each instance of a triangular distribution was based on Manache and Melching (2004). Although Melching and Yoon (1996) used normal distributions to model these parameters, their study applied the FOEA method, whereas Manache and Melching (2004) applied LHS. Since this analysis used LHS, the Manache and Melching (2004) study was given preference in informing the distribution types of phytoplankton parameters.

DiToro and van Straten (1979) and Scavia et al. (1981) also provide information on setting probability distributions for several of the phytoplankton parameters; however, the coefficients of variation (COVs) they recommended were considered too high to apply in this analysis, and the smaller margins of uncertainty for these parameters reported in other studies (e.g., Bowie et al., 1985; and Brown and Barnwell, 1987) were used instead.

Note that in the TMDL model, periphyton and macrophytes were simulated together as a single state variable, termed “benthic algae” (Omni Environmental, 2007a). Macrophytes are abundant in the watershed, especially in the Passaic River near the Pompton/Passaic confluence, and their position in the model of being “lumped” with periphyton influenced the selection of probability distributions for benthic algae parameters. Beta distributions for WASP benthic algae parameters were mostly assigned to characterize skewed distributions. Beta distributions are often used as a rough model in the absence of sufficient data (Wyss and Jorgensen, 1998).

3.2.4 Local kinetic parameters

In contrast with global parameters, local parameters are spatially variable. For example, a given local parameter might have a different value at each of the 327 segments in the TMDL model. Such a parameter would ideally be handled in an uncertainty analysis as 327 separate variables; accordingly, in an LHS analysis the parameter value at each segment would be sampled separately. However, this approach is computationally prohibitive because it drastically increases the total numbers of variables and minimum required model runs in the LHS analysis.

Instead a multi-pronged approach was adopted for this study to make the problem manageable. First, it was established that analyzing output at only 9 of the 327 segments in the TMDL model is relevant to the study objectives. These 9 locations and the reasons for their selection are listed in Table 2-4. Second, for each of the local parameters of interest, a spatial sensitivity analysis was conducted in two parts.

In the first part of the spatial sensitivity analysis, a target location (segment $_{t}$) was selected (e.g., Peckman River mouth), and a local parameter (e.g., SOD) was varied by the same amount one upstream segment at a time (segment $_{t-1,\ t-2 \ldots t-n}$). This determined how far upstream the segment values should be varied in the uncertainty analysis for each local parameter at each target location, essentially treating that reach of segments (segment $_{t-n}$ through segment $_{t}$) as a homogenous unit with respect to a given local parameter. In order to preserve the assumption of localized spatial homogeneity, a constraint was placed that segment values would not be changed at or upstream of a tributary or point source inflow, or a diversion outflow (i.e., segment $_{t-n}$ > segment $_{\text{inflow or outflow}}$). For example, DO at the Peckman River mouth (segment 298) was sensitive to changes in the variable "fraction of bottom segment covered by benthic algae", or X_F, as far as 6 segments upstream at segment 292. Because a WWTP discharged into segment 291, segment 292 was the farthest segment upstream to have X_F modeled probabilistically in the uncertainty analysis.

The second part of the spatial sensitivity analysis examined whether changes in local parameter values meant to affect target location segment $_{t}$ also affected model output at other target locations (segments $_{u,\ v,\ w \ldots z}$) further downstream. Returning to the previous example, did changes in X_F not only affect DO at segment 298, but also DO at other target locations downstream such as segment 318? If there were effects at other target locations further downstream, then the local kinetic parameter Y_t at segment $_{t-n}$ through segment $_{t}$ was modeled as a separate variable in the uncertainty analysis, so that the uncertainty of Y_t would be kept separate from the effect of the uncertainty of Y_u on segment $_u$.

On the other hand, if there were no effects at other target locations further downstream, then the approach of Melching and Bauwens (2001) was adopted. Suppose that for a given local parameter Y, Y_t only affects segment $_t$, Y_u only affects segment $_u$ …Y_z only affects segment $_z$. Then Y_t, Y_u…. Y_z can each be considered as a standardized variable. Thus a local kinetic parameter Y can maintain a different mean and variance at each location, and the parameter Y can be counted in the uncertainty analysis as one overall variable instead of as one variable for each target location. Then for each LHS sample of parameter Y, the local parameter value for Y_t, Y_u…. Y_z is varied by the same standardized amount. In this way, spatial variations of a local kinetic parameter are "lumped" in the error propagation and the number of overall variables is kept at a manageable amount. The assumed covariance between Y_t, Y_u…. Y_z is allowable because Y_t only affects segment $_t$, Y_u only affects segment $_u$ …, etc.

Through the execution of this process, the local kinetic parameters in Table 3-3 were modeled as lumped, and local kinetic parameters in Table 3-4 were modeled as separate. Except for SOD, the probability distributions that were estimated for each of the parameters were based to the extent possible on available literature; a sparse set of local data was used to aid characterization of the SOD distributions (Table 3-5). As with the global kinetic parameters, the calibrated values served as mean values for the symmetric probability distributions of the local kinetic parameters. Tables 3-3 and 3-4 list the segment numbers, calibrated values, and distribution characteristics for each of the local kinetic parameters. For the lumped variables in Table 3-3, the PDFs for the corresponding surrogate variables are shown in Figures 3-17 and 3-18. The approximate PDFs of the segment-specific variables in Table 3-3 are shown for illustrative purposes in Figures 3-19 through 3-27; the actual values of these variables were not generated via LHS but rather calculated via the process described in the following paragraph. The PDF for each variable in Table 3-4 is shown in Figures 3-28 through 3-32.

Parameters in Table 3-3 had surrogate normal distributions sampled in the uncertainty analysis. Two surrogate normal distributions were sampled - one each for the variables 'SOD' and 'fraction of bottom segment covered with benthic algae' (figs. 3-17 and 3-18). Thus, the surrogate variables 'SOD_s' and 'Fraction of bottom segment covered with benthic $algae_s$' were the actual variables used in the LHS sample set execution, rather than the variables in Table 3-3. The values sampled from the surrogate distributions were then translated to a local value using the segment-specific mean and standard deviation. The formula to translate the local value from the surrogate value equates the z-scores of the surrogate and local variables:

$$x = \mu + (\sigma \bullet xs / \sigma s) \qquad \text{(Eq. 3.1)}$$

Where

the surrogate variable is normally distributed with mean value = 0,

the local variable is normally distributed,

x = local variable value ≥ 0,

μ = mean of local variable,

σ = standard deviation of local variable,

xs = surrogate variable value,

and

σs = surrogate variable standard deviation.

For example, one of the 50 samples of the SOD surrogate distribution had a value xs of 0.768. The mean of the surrogate distribution was 0 and σs was 0.335, therefore xs is 2.29 standard deviations above the mean. For SOD at segments 295-298, where μ is 2 and σ is 1.3, the translated value x is 4.98, which is 2.29 standard deviations above the local mean value. If xs had been negative, (i.e., below the surrogate variable mean), then x would be less than 2. If xs was negative enough that x is also calculated to be negative, then x was changed to zero.

Note that the settling rate variable was assumed to have a normal distribution in the uncertainty analysis. At first, this may appear to counter the conventional interpretation of settling velocity as having a lognormal distribution due to its direct dependence on particle size, which also tends to be lognormally distributed (Uchrin, 1980). However, in the TMDL model the settling rate was used not just to represent physical settling of organic and inorganic particulate phosphorus, but also to describe the adsorption of orthophosphate to the sediment bed, and the extra phosphorus uptake by macrophytes in certain areas characterized as wetland meadows (Omni Environmental, 2007a). The latter two phenomena are not explicitly simulated in WASP and were therefore lumped into the settling rate variable. For the uncertainty analysis, it was assumed that the effect of this lumping in the settling rate term would lead to higher values than if physical settling were only considered, thereby countering the skewness of a lognormal distribution. Thus a normal distribution with the calibrated value as the mean value was assumed for the settling rate variables. In addition, Manache and Melching (2004) modeled settling rate with a normal distribution.

3.2.5 Parameter covariance

Parameter covariance refers to a correlation between certain input parameters. It quantifies the belief “that a particular value for one variable implies something about the possible values for one or more other variables” (Helton and Davis, 2003). A positive covariance indicates that a high value in one input parameter implies a high value in a related input parameter. Conversely, a negative covariance indicates that a high value in one input parameter implies a low value in a related input parameter.

One of the weaknesses of the FOEA method is it assumes no parameter covariance. However, given the large amount of kinetic parameters in the TMDL model, it is reasonable to suspect some degree of parameter covariance, rather than to assume that each parameter has no relationship at all with any others. Indeed some kinetic parameters in the TMDL model are likely to have covariance, e.g., the growth and respiration rates of phytoplankton. Therefore, an uncertainty analysis is enhanced by considering the possibility of parameter covariance.

Iman and Conover (1982) pioneered the method of “restricted pairing” which accounts for parameter covariance and limits the parameter sample space accordingly in LHS and other random sampling methods. Essentially, the method induces a rank correlation structure between input variables to reflect desired correlations among the parameters (Helton and Davis, 2003). Thus, the parameter sample space and resulting uncertainty analysis output are not overestimated. Although LHS has been applied in other water quality model uncertainty analyses (e.g., Melching and Bauwens, 2001), restricted pairing has generally not been utilized, with the exception of Kanso et al.

(2006). DiToro and van Straten (1979) demonstrated that failure to account for parameter covariance can lead to large overestimations of model uncertainty.

DiToro and van Straten (1979) analyzed the uncertainty of a eutrophication model whose fundamental equations were the foundation of the WASP model. In their analysis, they found that the mathematical relationship between input parameters in predicting a state variable is a guide to setting terms of parameter covariance. If a state variable **A** is positively correlated with both variables **x** and **y**, i.e., **A** increases as **x** and **y** increase, then if **x** is overestimated **y** needs to be underestimated in order to avoid overprediction of **A**. Thus **x** and **y** have a negative covariance. A similar logic was followed in interpreting the fundamental WASP model equations (Wool et al., 2003) to assume a covariance among certain kinetic parameters. DiToro and van Straten (1979) stated that the sign of the covariance is more important than the magnitude. In this study, a correlation coefficient of either (+) or (–) 0.5 was applied to parameters with assumed covariance.

Correlation coefficients between specific parameters were entered into the LHS software and are listed in Table 3-6 and illustrated in Figure 3-33.

3.2.6 Headwater boundary conditions for phosphorus

In this study, as in Omni Environmental (2007a), the term ‘headwaters’ refers to the boundary at the uppermost river segment included in the model domain, and not necessarily the actual headwaters themselves. For example, the Ramapo River ‘headwaters’ is modeled at the outlet of Pompton Lakes; this is the upper boundary in the TMDL model where its river segmentation begins. This is much further downstream than the actual Ramapo River headwaters, located in New York. Figure 3-34 illustrates the locations of the four model headwater boundaries discussed in the following paragraph.

Phosphorus boundary conditions at the Ramapo River and Passaic River headwaters were included in Stage 1, in order to affect the uncertainty of phosphorus output at the WS intake (segment 54) and the Passaic River just upstream of the Pompton/Passaic confluence (segment 256), respectively. The Singac Brook and the Peckman River headwater boundary conditions for phosphorus were excluded from the uncertainty analysis based on a sensitivity analysis screening.

The Passaic River headwater boundary condition sits at the outlet of the Great Swamp. Using available data, the TMDL model estimated a typical profile of orthophosphate and organic phosphorus throughout each water year (Omni Environmental, 2007a). Those boundary conditions, spanning 10/1/00 – 11/30/03, are shown in Figures 3-35 and 3-36. The profile is a repeating one-year long series of six clusters, with each cluster spanning two months. A constant value was selected in the TMDL model for each cluster based on observations made during the corresponding two-month period of the particular cluster. (In executing the TMDL model, observed values were left unchanged and thus deviate from the corresponding cluster value). The following sequence was designed to calculate the uncertainty of the profile.

The COV was calculated for each cluster based on available data published in Omni Environmental (20007a), both for orthophosphate and for organic phosphorus (Tables 3-7 and 3-8). The average COV was then calculated for orthophosphate and organic phosphorus, yielding 0.39 for each.

For the uncertainty analysis, orthophosphate and organic phosphorus at the headwaters were assumed to have complete covariance in order to treat them as one overall variable. This choice was justified by their similar COVs. The COV for the single Passaic River headwater boundary condition variable was estimated at 0.30; this is less than the 0.39 values calculated from the data for both orthophosphate and organic phosphorus. A lower number was used because it was believed that had there been additional observations, a lower COV would have been measured. A normal distribution was then generated with 1.0 as the mean value with a standard deviation of 0.30. This distribution functioned as a "scaling distribution", i.e., a distribution of scaling factors (Fig. 3-37). In each model run of the uncertainty analysis, a sample was drawn from the scaling distribution. Since the mean of the distribution was 1, the calibrated value (i.e., the TMDL model assumed value) was the mean value. Sampled values α greater than 1 scaled the profile higher to α·calibrated value, and sampled values β less than 1 scaled the profile lower to β·calibrated value. The profile for orthophosphate and organic phosphorus were scaled either up or down by the same amount together, in line with their assumed total covariance. Actual measured values were left unchanged in the uncertainty analysis.

A similar process was followed for the Ramapo River headwater boundary condition for phosphorus. However, in contrast with the Passaic River headwater boundary condition, the Ramapo River headwater boundary conditions for organic phosphorus and orthophosphate were modeled by Omni Environmental (2007a) as constant throughout the entire water year, except for observed values (Figs. 3-38 and 3-39). According to the available data published in Omni Environmental (2007a), orthophosphate and organic phosphorus had comparable COVs at 0.49 and 0.38 (Table 3-9), respectively, and were thus assumed to have complete covariance in order to treat them in the uncertainty analysis as one overall variable. The average of the two COVs weighted by TMDL model value, 0.42, was chosen as the COV for the single Ramapo River headwater boundary condition variable. As with the Passaic headwater boundary condition, a normal distribution was then generated with 1.0 as the mean value with a standard deviation of 0.42 (Fig. 3-40). This distribution functioned as a scaling distribution and was used in the uncertainty analysis in the same way as the Passaic headwater scaling distribution.

3.2.7 WWTP boundary conditions for phosphorus

WWTPs in the watershed collected a varying amount of effluent data, ranging from monthly to daily samples. The effluent data from each WWTP was the basis for defining WWTP boundary conditions in the TMDL model. For the days that lacked effluent data, linear interpolation was used in the TMDL model to assign effluent boundary condition values. However, actual conditions may not have necessarily followed a linear pattern between data points that were days or even weeks apart. An alternate approach to linear interpolation was taken in the uncertainty analysis to assign effluent values for days without data.

First, the scope was defined by only selecting WWTPs that were large, had sizeable data gaps, and were the dominant source of phosphorus loading to a key location. Specifically these constraints were defined so as to only select WWTPs that met all of the following criteria: i) greater than 3 MGD in permitted flow capacity; ii) 10 or less phosphorus effluent measurements per month in any month from October 2000 through November 2003; and iii) functioned as the single largest plant directly upstream of any key location in Table 2-4. The first constraint ruled out 10 WWTPs. Of the remaining 12 WWTPs, the two largest - Parsippany-Troy Hills RSA and Wayne Township STP – were ruled out since each had 11 or more measurements in every month; in fact, the former plant had daily data. The last constraint left 4 WWTPs remaining in Stage 1 of the uncertainty analysis, as listed in Table 3-10 and shown in Figure 3-41.

In the WASP model, separate boundary conditions are input for orthophosphate and organic phosphorus. However, 3 of the 4 plants, i.e., Rockaway RSA, Berkeley Heights STP, and Verona STP, only measured TP data. The fourth plant – Two Bridges SA – measured orthophosphate and organic phosphorus for most but not all of the analysis timeframe. This presented a dual problem to model both the uncertainty of TP, and the ratio of orthophosphate to organic phosphorus for days without the constituent data. (In the case of Verona STP, the TP concentration was assumed to be 98% orthophosphate based on Omni Environmental effluent sampling. Therefore Verona STP organic phosphorus was considered negligible and the uncertainty analysis was only applied to the assumed orthophosphate values; assumed organic phosphorus values were left unchanged).

In terms of modeling the uncertainty of TP, available data from each of the WWTPs in Table 3-10 were plotted and analyzed to identify distinct clusters (Figs. 3-42 through 3-47). (Assumed orthophosphate values in the TMDL model, rather than TP, were plotted for Verona STP). Following removal of outliers, a separate probability distribution was then estimated for each cluster based on the data (Figs. 3-48 through 3-

59. Note that in each figure, histogram bin widths were set to best describe each dataset, and were not set to a common width). An LHS process was then applied. Each distribution was stratified into *x* equally probable intervals, where *x* is the number of days in the cluster without data. For each cluster, values were sampled from its stratified distribution in order to "fill in" values for the days without data. This was done 50 times in order to generate 50 different cluster sets, since the model needed to be run 50 times in Stage 1. Each of the 50 cluster sets was then randomly combined with other cluster sets of the same WWTP, thus yielding 50 different sample sets of TP for each WWTP. Since Two Bridges SA had separate data on orthophosphate and organic phosphorus from 5/1/01 – 11/30/03, the above process was applied to generate 50 different sample sets of orthophosphate and organic phosphorus, rather than 50 sample sets of TP, for that timeframe. It is important to note that actual observed values were not changed.

An example of the Cluster 1 TP effluent from Berkeley Heights STP is shown in Figure 3-60. The observed TP values are plotted along with 1 of the 50 sample sets generated from LHS. The observed and LHS values are listed in Table A-1.

The validity of this method is supported by a simple experiment. Only one WWTP in the watershed, Parsippany-Troy Hills RSA, had daily measurements of TP. The TMDL model assumed a 33:1 ratio of orthophosphate to organic phosphorus in the Parsippany-Troy Hills RSA effluent. Since the model essentially assumed organic phosphorus concentrations to be negligible, the daily modeled values of orthophosphate were extracted (Fig. 3-61). The first cluster (daily values from October 1, 1999 to April 22, 2001) was identified and analyzed. Half of the cluster dataset (the input value from every other day) was deleted. The remaining half of the dataset, termed the "abridged dataset", was subjected to the steps described above. The objective was to compare the sampled values to actual values that had been deleted. If the sampled values compared well to actual values, the method was valid.

The abridged dataset was characterized with a lognormal distribution (Fig. 3-62). The distribution was stratified into 285 intervals, corresponding to the 285 days without data, and then sampled 50 times. At each day of the 285 x 50 dataset, the 10^{th} and 90^{th} percentiles were calculated from the 50 samples for each day, giving an 80% confidence interval for each day (Fig. 3-63).

The actual values of the "skipped" days were then compared to the predicted 80% confidence intervals. The proportion of actual values that fell inside the predicted 80% confidence intervals, i.e., the proportion of success, was calculated. A 95% confidence interval about the proportion 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 interval was credible; otherwise the predicted 80% confidence interval was not credible. Table 3-11 shows a positive result in that 0.80 was inside the 95% confidence interval about the proportion of success. Therefore the method was validated.

Once the TP sample sets were generated for the appropriate WWTP clusters, a second step was needed to translate them into orthophosphate and organic phosphorus sample sets. The TMDL model assumed a plant-specific ratio of orthophosphate to organic phosphorus, termed here as "PSRP", for each WWTP based on effluent measurements reported in Omni Environmental (2007a). For the 3 WWTPs analyzed in Stage 1, those ratios are shown in Table 3-12. (Verona STP is not included because only its orthophosphate effluent was treated as an uncertain variable in Stage 1; organic phosphorus was assumed to be negligible and treated as a fixed variable). Each WWTP was assigned a normal distribution to characterize uncertainty in PSRP. The TMDL model ratios listed in Table 3-12 acted as mean values for each distribution; the standard deviations were derived from the COVs assumed in Stage 1. Note that the assumed COV values in Stage 1 were lower than the measured values; this was done to roughly reconcile the disparity between fixed model PSRPs and the measured mean PSRP values by reducing the variability around the fixed model values. Also, for Berkeley Heights STP and Rockaway Valley RSA, the PSRP was assumed to be constant over all effluent cluster periods.

Each distribution of PSRP was stratified into x intervals, where x equals the total number of days in the analysis timeframe without specific data on orthophosphate and organic phosphorus. The PSRP distributions were then sampled 50 times, and paired with the TP sample sets to produce 50 orthophosphate and organic phosphorus sample sets. Continuing from the previous example, in Table A-1 the measured value for TP on 10/3/2000 was 4.30 mg/L. To translate that into 50 different pairs of orthophosphate and organic phosphorus values, 50 sample sets of PSRPs were generated. In PSRP sample set 1, the value for 10/3/2000 was 2.21. Hence in sample set 1, the 10/3/2000 values for orthophosphate and organic phosphorus are 2.96 and 1.34, respectively. This process was repeated so that each day in each sample set had an orthophosphate and organic phosphorus pair adding up to the sampled TP value.

3.2.8 Execution of uncertainty analysis

The preceding methodology steps generated 50 sample sets of i) global and local kinetic parameters, and ii) headwater and WWTP boundary conditions for organic phosphorus and orthophosphate spanning 10/1/00 – 11/30/03. Table A-2 lists all the sample sets, except the WWTP boundary conditions, which are too lengthy to include. (The probability distribution types of the WWTP boundary condition are listed in Table 2-1).

3.2.9 Comparing the uncertainty analysis output to observed data

Fifty model simulations were run - one for each sample set. The resulting probabilistic model output was compared to available water quality data at the key locations in Table 2-4, in order to assess the credibility of Stage 1.

For a given parameter and location, the predicted 80% confidence interval at each timestep resulting from all the Stage 1 model runs was compared to observed data. The number 80% was chosen because in Stage 2, 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% margin outside the 80% confidence interval at the extreme of concern; the high extreme pertains to TP and chl-*a*. In contrast, an expected exceedance (i.e., the mean value of a distribution of exceedance frequencies (Borsuk et al., 2002)) of greater or less than 10%, a measure used to evaluate daily average and minimum DO predictions in Stage 2, does not have a direct relation to the 10% margin outside the 80% confidence interval. However, because the 80% confidence interval of predicted DO is a general indicator of the prospect of exceedance[1], it is still

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[1] A DO measurement lower than the water quality standard is more accurately described as a 'violation', rather than an 'exceedance'. However, in keeping with the standard terminology in the water quality

valuable to assess in Stage 1 the credibility of the 80% confidence interval before calculating expected exceedances in Stage 2. Figures 3-64 and 3-65 illustrate these concepts.

The 80% predicted confidence interval was then compared to observed data. Intuitively, if the majority of measurements fall inside the predicted 80% confidence interval, it indicates a better uncertainty estimate than if none of the measurements were to fall inside the predicted 80% confidence interval; the latter would indicate the confidence interval is too small. Conversely, if all the measurements were to fall inside the predicted 80% confidence interval, it indicates the confidence interval is too large. A simple statistical approach, the calculation of the confidence interval about a binomial proportion, was used to formally evaluate the performance of the uncertainty analysis. For each uncertainty analysis output, the number of measured values inside the predicted 80% confidence interval (i.e., successes) was compared to the total number of measured values, yielding a proportion of success. 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 interval was credible; otherwise the predicted 80% confidence interval was not credible. Since the sample sizes were small, the Agresti-Coull confidence interval formula was used (Agresti and Coull, 1998):

$$CI_{AC} = \tilde{p} \pm \kappa (\tilde{p}\,\tilde{q})^{1/2}\,\tilde{n}^{-1/2} \qquad \text{(Eq. 3.2)}$$

where

CI_{AC} = Agresti-Coull confidence interval,

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modeling uncertainty analysis literature (e.g., Melching and Yoon, 1996; Borsuk et al., 2002), the term ‘exceedance’ is used here also.

$\tilde{n}$ = 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.

3.2.9.1 Applying the Agresti-Coull confidence interval: Defining a successful Bernoulli trial

In applying the Agresti-Coull confidence interval, each comparison of the measured value to the corresponding 80% confidence interval is assumed to be an independent Bernoulli trial. In Stage 1, it was not necessary to require that an 80% confidence interval had to envelop an observation at the exact time of the observation in order to consider it a success. Instead, some flexibility was given in that a predicted 80% confidence interval for DO that enveloped the observation within ± 4.8 hours was sufficient to qualify as a successful trial. Similarly, a predicted 80% confidence interval for chl-*a* or TP that enveloped the observation within ± 12 hours was sufficient to qualify as a successful trial. However, an exception was made for TP at B12-N38, i.e., the Passaic River at the confluence with the Pompton River, in that a predicted 80% confidence interval that enveloped the observation within ± 48 hours was sufficient to qualify as a successful trial; justification for this exception is provided in section 3.3.1.

In order to carry this out, the model outputs had to be processed on a moving time scale – sub-daily for DO, daily for chl-*a*, daily for TP at the Wanaque South intake, and semi-weekly for TP at B12-N38. The predictions of DO concentrations were processed on a moving sub-daily time scale of ± 4.8 hours (equivalent to ± 2 model time steps), TP at the WS intake and chl-*a* concentration predictions were processed on a moving daily time scale of ± 12 hours (equivalent to ± 5 model time steps), and TP concentration predictions at B12-N38 were processed on a moving semi-weekly time scale of ± 48 hours (equivalent to ± 20 model time steps). To generate the 80% confidence intervals at the moving time scales, at each timestep the 10th percentile of the minimum values within ± x time steps was calculated along with the 90th percentile of the maximum values within ± x time steps, where x is 2 for DO, 5 for chl-*a*, 5 for TP at the WS intake, and 20 for TP at B12-N38. The processed model output was then compared to observed data.

3.2.9.2 Applying the Agresti-Coull confidence interval: Compiling observed datasets

3.2.9.2.1 TP

Year-round TP observations at the WS intake and B12-N38 were compiled. In order to maintain the condition of independent Bernoulli trials, at the WS intake only observations at least 27 hours apart were considered (Table A-3) thus yielding an interval between observations greater than the ± 12 hours of processed TP model output, and at B12-N38 only observations at least 100 hours apart were considered (Table A-4) thus yielding an interval between observations greater than the ± 48 hours of processed TP model output. The data sources were Omni Environmental and USGS. Data from Passaic Valley Water Commission (PVWC) and NJDWSC were not used because of suspect measurement reliability – several measurements from these sources recorded orthophosphate as higher than total phosphorus.

3.2.9.2.2 Chl-a

Chl-*a* observations from June 15 to September 1 in 2001 through 2003 at B17-N25 and B17-N4 were compiled. This matches the period in the seasonal average chl-*a* criteria. Dundee Lake data was not used because only 3 summer measurements were available. In order to maintain the condition of independent Bernoulli trials, only observations at least 27 hours apart were considered, thus yielding an interval between observations greater than the ± 12 hours of processed chl-*a* model output (Tables A-5 and A-6). The data sources were Omni Environmental and Passaic Valley Sewerage Commissioners (PVSC), where each PVSC observation is taken as the average of 3 transect measurements.

3.2.9.2.3 DO

A dataset of available grab samples and random diurnal meter data from June 15 to September 1 in 2002 and 2003 was prepared for the locations: Dundee Lake, Peckman River mouth, Passaic River near Chatham, and station PA10 (Tables A-7 through A-10). This matches the period in the seasonal average chl-*a* criteria (June 15 to September 1). Since the minimum DO standard at the Passaic River near Chatham was violated earlier than June 15, its dataset began at June 4 rather than June 15. In order to maintain the condition of independent Bernoulli trials, only observations at least 12 hours apart were considered, thus yielding an interval between observations greater than the ± 4.8 of processed DO model output. Data sources included Omni Environmental, NJDEP and PVWC.

Diurnal DO meter data alone was too sparse to make meaningful comparisons to the uncertainty analysis output. Only 3 to 9 days of adequate meter data were available at the 5 locations listed in the previous paragraph (Table 3-13). Although the Agresti-Coull confidence interval is designed for small samples, its limits in this application become apparent as the sample size shrinks to 5; the confidence intervals become too wide for meaningful interpretation as the sample size decreases to such small levels. If more days of diurnal meter data had been available, a direct comparison of predicted and observed daily average and daily minimum DO values would have been possible. However this was not the case, and therefore grab samples were utilized along with random diurnal meter data to compare predicted and observed DO at a sub-daily time scale.

3.3 Results

3.3.1 TP

Omni Environmental measurements at the WS intake were only made during times without any diversion or when only the Pompton River was diverted; no measurements were made during a diversion of both the Passaic and Pompton Rivers, i.e., an “extreme diversion” as termed in Obropta et al. (2008). The positive results in Table 3-14 suggest that estimates of TP uncertainty at the WS intake are credible during times without any diversion or when only the Pompton River is diverted (Fig. 3-66).

During an extreme diversion, a sizeable fraction of the diverted load is contributed by the Passaic River. For example in the drought year of WY2002, when an extreme diversion occurred on 216 days, 41% of the total load diverted at the Wanaque South intake was estimated to have come from the Passaic River (Omni Environmental, 2007b). Because a reliable dataset was not available at the WS intake during times of extreme diversion, the credibility of the uncertainty analysis during an extreme diversion was evaluated indirectly through separately considering its performance first in the Pompton River, and then in the Passaic River near the WS intake.

The estimated uncertainty of predicted TP from the Pompton River that reaches the WS intake has already been shown to be credible (Table 3-14 and Fig. 3-66). With regard to the Passaic River, TP measurements were made at B12-N38 just upstream of its confluence with the Pompton River. This sampling location represents the portion of the Passaic that enters the WS intake during an extreme diversion. The results in Table 3-14 suggest that when model output is processed at a moving daily time scale, estimates of TP uncertainty at B12-N38 are not credible and that the 80% confidence interval is too small (Fig. 3-67a). However when model output is processed at a longer moving time scale of 4 days rather than 1 day, estimates of TP uncertainty at B12-N38 are credible (Table 3-14 and Fig. 3-67b).

The discrepancy in the predicted uncertainty of TP at B12-N38 is likely due to model structural uncertainty in that macrophytes are not explicitly modeled. Macrophytes are abundant in that section of the Passaic River, and their intake and release of phosphorus were described in the model through lumping with benthic algae parameters and the local settling velocity of particulate phosphorus. This appears to have resulted in a narrowly flawed simulation of TP at the daily time scale, which is to be expected since the phosphorus dynamics of macrophytes and periphyton do not function in reality at the same rates. Similarly, physical settling velocity of particulate phosphorus and phosphorus uptake by macrophytes also may not function at the same rates when evaluated at a daily time scale. However, Figure 3-67b clearly demonstrates that at the semi- weekly time scale, the predicted uncertainty of TP at B12-N38 is credible.

The reduced COV in the Passaic River headwater boundary condition for phosphorus that was described in section 3.2.6 is unlikely to be the cause of the discrepancy in the predicted uncertainty of TP at B12-N38. A global sensitivity analysis found that the farthest downstream location where the Passaic River headwater scaling factor significantly affected TP was B11-N33, located 46 km upstream of B12-N38.

*3.3.2 Chl-*a

Chl-*a* data at Dundee Lake was insufficient, so measurements at a nearby upstream location, B17-N25, were used instead for comparison to simulations (Fig. 3-68). The results in Table 3-15 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 B17-N25 and assumed to be credible at Dundee Lake; however, the credibility is based on a small sample size of observations at a location 3 km upstream.

The 90th percentile prediction at B17-N25 appears very high in the summers of 2001 and 2002, reaching peak values of about 250 μg/L in each summer. A comparison to a location upstream of the Great Falls (B17-N4) demonstrates that the 90th percentile prediction there underpredicted the 2001 summer peak measurement of 163 μg/L (Fig. 3-69). That measurement, on August 2, 2001, coincides with the height of the predicted bloom. Unfortunately a same-day measurement was not made at B17-N25, otherwise it would be possible to directly compare the 90th percentile 250 μg/L bloom prediction at B17-N25. Therefore, although the 90th percentile prediction at B17-N25 seems high, it is a reasonable estimate based on a comparison of the predicted and observed bloom at B17-N4 in 2001.

3.3.3 DO

As indicated in Table 3-16, the DO simulations compared well to measurements at Dundee Lake (Fig. 3-70), the Peckman River mouth (Fig. 3-71), and station PA10 (Fig. 3-72), but did not compare well at the Passaic River near Chatham (Fig. 3-73).

The discrepancy at the Passaic River near Chatham is likely due to positive bias in the SOD mean value. DO at this location is sensitive to SOD, as demonstrated in the next section. DO was generally overpredicted (Fig. 3-73) suggesting that SOD was underpredicted.

Estimates of DO uncertainty are therefore credible at all locations in Table 3-16 except the Passaic River near Chatham.

3.3.4 Global sensitivity analysis

Following the execution of the Stage 1 uncertainty analysis and comparison to previous observations, a global sensitivity analysis (GSA) was conducted to identify kinetic parameters that significantly affected model output for TP, chl-*a* and DO at key locations. Non-sensitive parameters were removed from Stages 2 and 3 of the uncertainty analysis to streamline those phases of the research. Boundary condition variables were not included in the GSA.

A GSA is global in that through the LHS process, all the variables of interest were varied simultaneously rather than one at a time. This allows the sensitivity analysis to account for possible parameter covariance; a GSA is thus more realistic than a local sensitivity analysis (Muleta and Nicklow, 2005). Using all 50 sets of input kinetic parameters and model outcomes, separate GSAs were performed for the: i) average DO at Dundee Lake, Peckman River mouth, Passaic River near Chatham and station PA10 over the span of May through September 2001; ii) average diurnal DO swing at Dundee Lake, Peckman River mouth, Passaic River near Chatham and station PA10 over the span of May through September 2001; iii) average TP at the WS intake, B12-N38, and Little Falls intake over the spans of October 2000 through April 2001 and then May through September 2001; and iv) average chl-*a* at Dundee Lake over the span of May through

September 2001. May through September 2001 corresponds to the algal growing season of that year.

In order to eliminate any bias effects of pre-defined parameter covariance on the GSA results, the Stage 1 uncertainty analysis was redone without any restricted pairing; no correlations were defined between any of the kinetic parameters. Any possible correlations between input parameters and model output were thus able to emerge without prior judgments.

For each of the above locations and water quality parameters, the GSA procedure was to univariately correlate each set of rank-transformed model outcomes with a rank-transformed input variable set. Rank-transformation eliminates the effect of monotonic nonlinear relations (Manache and Melching, 2004). Input variables that had significant correlations with the model outcome (i.e., $p < 0.05$) were identified. Seven dummy input variables were used to screen out significant yet weakly correlated input variables; in a few GSAs some of the dummy variables were found to be significant, showing weak correlations of $r < |0.5|$ in each occurrence. Therefore only significant input variables with correlation coefficients $\geq |0.5|$ were screened as meaningful results.

Significant input parameters with correlation coefficients $\geq |0.5|$ are shown in Tables 3-17 through 3-21 for the appropriate location and water quality parameter. Corresponding subplots are shown in Figures 3-74 through 3-88. Of the 26 kinetic parameters, 8 were found to have correlation coefficients $\geq |0.5|$. Average DO and diurnal DO swing were exclusively affected by benthic processes concerning either benthic algae variables or SOD. TP and chl-*a* were each highly correlated with only one kinetic parameter, settling velocity of particulate phosphorus and phytoplankton maximum growth rate, respectively. Furthermore, TP at the WS intake was not highly correlated with any kinetic parameters. A subsequent GSA that accounted for both the kinetic parameters and headwater boundary conditions found that TP at the WS intake was strongly correlated with the scaling factor of the Ramapo River headwater boundary condition for phosphorus, with correlation coefficients of 0.99 and 0.95 in October 2000-April 2001 and May 2001-September 2001, respectively.

Although only 8 of the 26 kinetic parameters had correlation coefficients ≥ |0.5|, a conservative decision was made to not discard all the remaining 18 kinetic parameters from Stages 2 and 3. Seven of the eighteen parameters were retained based on findings from the preliminary local SA undertaken at the beginning of Stage 1. The 15 kinetic parameters carried forward into Stages 2 and 3 are shown in Table 3-23, and the 11 removed are shown in Table 3-24.

Interestingly, phytoplankton variables were not significantly correlated with DO outcomes in the model. This suggests that exceedances of chl-*a* criteria do not necessarily imply exceedances of DO water quality standards at the Dundee Lake site. This is because DO at Dundee Lake is much more correlated with benthic algae variables in the model. The lack of a correlation between phytoplankton processes and DO at Dundee Lake suggests that if a seasonal average chl-*a* criteria exceedance occurs at Dundee Lake, then in order to fully describe water quality at the site, care should be taken to also report the exceedance frequency of the daily average and minimum DO standards, as well as the distribution of diurnal DO swings over the June 15 to September 1 period. At the same time, it can be argued that the chl-*a* criteria is fully justifiable based on i) its relation to protection of designated uses, and ii) its response indicator status of excessive phosphorus loading. Excess phosphorus also stimulates growth of the periphyton and macrophytes that the model predicts to have such a strong effect on DO at Dundee Lake.

3.3.5 Discussion

Although the NRC (2001) highlighted the importance of water quality model uncertainty analysis in TMDL development, the rigorous practice of uncertainty analysis has been rare (Stow et al., 2007). This research has sought to address this problem through developing a simple and computationally efficient method for water quality model uncertainty analysis. By using a stratified sampling approach, LHS, estimates of model uncertainty were realized with much fewer model runs than a Monte Carlo approach. Furthermore, the credibility of the uncertainty estimates were assessed through a basic statistical tool, the Agresti-Coull confidence interval about a binomial proportion, that works well with small sample sizes of observations. At least 9 or 10 observations are necessary to assess the uncertainty analysis credibility; with less observations the Agresti-Coull confidence intervals become too large for meaningful interpretation of the results. With at least 15 observations, the method can detect if the predicted 80% confidence interval of a water quality parameter is either too large or too small. With less than 15 observations, the method can only detect if the predicted 80% confidence interval of a water quality parameter is too small.

An important qualification is that this research benefited from having a calibrated model already in place. Without a calibrated model, estimates of parameter probability distributions would have been more difficult, and more model runs would have been required. However, the simplicity of assessing the uncertainty analysis credibility would have remained the same, and the research contribution of that portion of the methodology – the most powerful piece of the methodology - would have remained intact.

Through application of the above methodology, it was established that estimates of DO uncertainty are credible at Dundee Lake, the Peckman River mouth, and station PA10, but not at the Passaic River near Chatham. Estimates of chl-*a* uncertainty are credible at Dundee Lake, although this conclusion was reached based on a small sample size of observations at a location 3 km upstream. Estimates of TP uncertainty at the Wanaque South intake are credible during times without any diversion or when only the Pompton River is diverted; when extreme diversions occur, estimates of TP uncertainty are credible provided that model output of TP is processed at a 4-day moving time scale. These findings on credibility were crucial to establish before proceeding into uncertainty analysis of the TMDL and trading scenarios; a reliable investigation of the research objectives would not be possible without having first probed the uncertainty analysis credibility.

In addition, the findings on credibility generally support the assumption that the selected kinetic parameters and boundary conditions are the primary sources of model uncertainty. Of the potential uncertainty sources not considered, only model structural uncertainty appears to have had a slight impact at one location – the absence of explicit modeling of macrophytes at B12-N38. Although not perfect, the scope of the uncertainty analysis was demonstrated to be sound and reasonable.

The GSA yielded important insights into which kinetic parameters have the greatest effect on predicted TP, chl-*a* and DO at key locations. As an aid to adaptive management, the GSA findings could be used as a tool to guide future monitoring efforts and gain more information on sensitive kinetic parameters. That information could then feed back into the probability distribution inputs and produce an updated estimate of model uncertainty. Specifically, Tables 3-17 through 3-21 list the variables that most significantly affect TP, chl-*a* or DO at each location. Of all the kinetic parameters shown, measuring the phytoplankton maximum growth rate at Dundee Lake should be the top priority, followed by SOD at the Passaic River near Chatham. These parameters would be more straightforward to measure than the other parameters listed. A better understanding of those parameters would enhance the forecast of chl-*a* uncertainty at Dundee Lake and DO uncertainty at the Passaic River near Chatham. Given the current information, both of those locations are predicted to have high exceedance frequencies in the TMDL scenario, as will be shown in Chapter 4. More information on the phytoplankton maximum growth rate at Dundee Lake and SOD at the Passaic River near Chatham could serve to either verify or refute the related findings in Chapter 4.

Finally, to resolve the two instances of water quality data gaps that hindered Stage 1, at least 15 surface water samples each should be collected of chl-*a* at Dundee Lake during June 15 to Sep 1, and TP at the Wanaque South intake during periods of extreme diversion, in order to verify the findings reported here.

Table 3-1: Variables excluded from uncertainty analysis through screening by preliminary sensitivity analysis

Parameter	Units
Detritus dissolution rate	d^{-1}
Dissolved fraction of organic phosphorus	-
Dissolved organic nitrogen mineralization rate	d^{-1}
Fraction of dead phytoplankton recycled to organic nitrogen	-
Fraction of dead phytoplankton recycled to organic phosphorus	-
Light extinction coefficient	d^{-1}
Peckman River boundary conditions for phosphorus	mg/L
Singac Brook boundary conditions for phosphorus	mg/L

Table 3-2: Probability distributions of WASP global kinetic parameters

WASP parameter	Unit	Calibrated value	Distribution characteristics						Notes[a]
			Type	Mean (unless otherwise noted)	Standard deviation	Coefficient of variation (COV)	Min.	Max.	
Nitrification rate @ 20°C	d^{-1}	0.25	Normal	0.25	0.025	0.10	-	-	1, 2
Phytoplankton maximum growth rate @ 20°C	d^{-1}	1.25	Normal	1.25	0.1875	0.15	-	-	1, 2
Phytoplankton carbon to chlorophyll ratio	-	20	Triangular	20	-	-	12	28	3, 4
Phytoplankton endogenous respiration rate @ 20°C	d^{-1}	0.15	Triangular	0.15	-	-	0.10	0.20	3, 5
Phytoplankton death rate, non-zooplankton predation	d^{-1}	0.1	Triangular	0.10	-	-	0.01	0.19	3, 6
Phytoplankton optimal light saturation	Langleys/d	320	Triangular	320	-	-	290	350	3, 4
Phytopl. nitrogen half-saturation constant for growth	mgN/L	0.025	Normal	0.025	0.005	0.20	-	-	1, 7

Phytoplankton phosphorus half-saturation constant for growth	mgP/L	0.0025	Triangular	0.0025 = apex	-	-	0.0005[b]	0.01[b]	3, 8
Benthic algae maximum growth rate @ 20°C	$gD/m^2/d$	60	Triangular	60	-	-	35	85	9
Benthic algae respiration rate @ 20°C	d^{-1}	0.01	Lognormal	0.01 = geometric mean	0.0072	0.61	-	-	9
Benthic algae death rate @ 20°C	d^{-1}	0.005	Beta (with p = 1.3, q = 10.0)	0.005 = mode	0.0086	0.70	0.001	0.10	9
Benthic algae ammonia preference	mgN/L	0.1	Beta (with p = 1.5, q = 4.0)	0.1 = geometric mean	0.0076	0.63	0.0025	0.50	9
Benthic algae light constant for growth	Langleys/d	350	Beta (with p = 4.0, q = 0.5)	350 = mode	13.42	0.04	250	350	9
Benthic algae nitrogen half-saturation constant for growth	mgN/L	0.025	Normal (bounded)	0.025	0.005	0.20	0.015	0.035	9
Benthic algae phosphorus half-saturation constant for growth	mgP/L	0.0025	Beta (with p = 0.5, q = 10.0)	0.0025 = mode	0.0013	0.37	0.0025	0.025	9

Mineralization rate of dissolved organic phosphorus @ 20°C	d^{-1}	0.2	Normal (bounded)	0.2	0.01	0.05	0	0.22	7, 10

[a]1 = COV based on the typical range for the coefficient of variation for such parameters reported by Brown and Barnwell (1987); 2 = Normal distribution assumed based on Melching and Yoon (1996) and Manache and Melching (2004); 3 = Triangular distribution assumed based on Manache and Melching (2004); 4 = parameter range based on Wool et al. (2003); 5 = parameter range based on Wool et al. (2003), Manache and Melching (2004), and Lindenschmidt (2006); 6 = parameter range based on Bowie et al. (1985), Wool et al. (2003), Manache and Melching (2004), and Lindenschmidt (2006); 7 = Normal distribution assumed based on Manache and Melching (2004); 8 = parameter range based on Bowie et al. (1985) and Brown and Barnwell (1987); 9 = based on engineering judgment with consideration of parameter value range reported in Ambrose et al. (2006), uncertainty of periphyton and macrophyte lumping in TMDL model, and trials of different values; 10 = parameter range based on Brown and Barnwell (1987), Wool et al. (2003) and Lindenschmidt (2006).

[b] Based on Stage 2 findings reported in Chapter 4.3.5, it is believed that the minimum and maximum values are more plausibly 0.0012 and 0.0038 in this system. Parameter measurements should be made on Dundee Lake phytoplankton samples to confirm this belief.

Table 3-3: Probability distributions of WASP local kinetic parameters modeled as lumped variables

WASP parameter	Unit	Target location segment number	Segments varied	Calibrated value	Distribution characteristics						Notes[a]
					Type	Mean	Standard deviation	COV	Min.	Max.	
SOD	$g/m^2/d$	181	179-181	8.0	Normal (bound-ed)	8.0	3.0	0.375	0	20	1, 2
		256	251-256	8.0		8.0	3.0	0.375	0	20	1, 3
		54	54	3.5		3.5	1.3125	0.375	0	10	1, 2
		298	295-298	2.0		2.0	1.3	0.65	0	10	1, 3
Fraction of bottom segment covered with benthic algae $_1$	-	181	179-181	0.25	Normal	0.25	0.0375	0.15	-	-	4, 5
		298	292-294	0.20		0.20	0.03	0.15	-	-	4, 5
			295-298	0.35		0.35	0.0525	0.15	-	-	4, 5
		256	255-256	0.15		0.15	0.0225	0.15	-	-	4, 5
		318	316	0.20		0.20	0.03	0.15	-	-	4, 5

			317-318	0.30		0.30	0.045	0.15	-	-	4, 5
		323	319-323	0.15		0.15	0.0225	0.15	-	-	4, 5
			324-325	0.15		0.15	0.0225	0.15	-	-	4, 5
		327	326	0.30		0.30	0.045	0.15	-	-	4, 5
			327	0.70		0.70	0.105	0.15	-	-	4, 5

[a]1 = Normal distribution assumed based on Melching and Yoon (1996); 2 = COV set equal to COV in Segment 256; 3 = Standard deviation set equal to the difference between measured and calibrated SOD value; 4 = Normal distribution chosen to enable application of Melching and Bauwens (2001) approach for local parameters; 5 = COV assumed based on engineering judgment and trials with higher COV values.

Table 3-4: Probability distributions of WASP local kinetic parameters modeled as separate variables

WASP parameter	Unit	Target location segment number	Segments varied	Calibrated value	Distribution characteristics						Notes[a]
					Type	Mean	Standard deviation	COV	Min.	Max.	
Settling rate of particulate phosphorus $_1$	cm/s	275	267, 272-274	0.6	Normal (bound-ed)	0.6	0.18	0.30	0	1	1,2,3,4
Settling rate of particulate phosphorus $_2$		318	299-300, 304-306, 312, 315	0.1		0.1	0.03	0.30	0	1	
		256	219-256								
Settling rate of particulate phosphorus $_3$		298	284-298	0.01		0.01	0.003	0.30	0	1	
Dissolved fraction of orthophosphate $_1$	-	256	247-256	0.60	Normal (bound-ed)	0.60	0.15	0.25	0	1	5
Dissolved fraction of orthophosphate $_2$	-	298	292-298	0.60		0.60	0.15	0.25	0	1	
Dissolved fraction of orthophosphate $_3$	-	303	299-303	0.60		0.60	0.15	0.25	0	1	
Dissolved fraction of orthophosphate $_4$	-	327	316-327	0.60		0.60	0.15	0.25	0	1	

Fraction of bottom segment covered with benthic algae $_{2}$	-	303	299-303	0.15	Normal (bound-ed)	0.15	0.06	0.40	0	1	6,7

[a]1 = Segments varied for settling rate of particulate phosphorus correspond exactly to the segments varied in the TMDL model; 2 = Normal distribution assumed based on Manache and Melching (2004) and consideration of macrophyte uptake of phosphorus included in this variable; 3 = COV based on Brown and Barnwell (1987) and Manache and Melching (2004); 4 = parameter range based on Brown and Barnwell (1987) and Chapra (1997); 5 = Normal distribution and COV based on calibrated value and parameter range in Wool et al. (2003); 6 = Normal distribution assumed to correspond with Fraction of bottom segment covered with benthic algae $_{1}$; 7 = Higher COV based on trials with lower values and engineering judgment.

Table 3-5: Summary of SOD observations at Stage 1 target locations (adapted from Omni Environmental, 2007a)

Location	Number of observations	Observed SOD [g/m^2/d]	Calibrated value
Segment 256	1	5.0	8.0
Segment 298	1	3.3	2.0

Table 3-6: Variable correlations entered into ARRAMIS

Variable 1	Variable 2	Correlation coefficient
Benthic algae ammonia preference	Benthic algae maximum growth rate @ 20°C	0.5
Benthic algae ammonia preference	Benthic algae respiration rate @ 20°C	0.5
Benthic algae death rate @ 20°C	Benthic algae ammonia preference	0.5
Benthic algae death rate @ 20°C	Benthic algae maximum growth rate @ 20°C	0.5
Benthic algae death rate @ 20°C	Fraction of bottom segment covered with benthic algae $_1$	0.5
Benthic algae death rate @ 20°C	Fraction of bottom segment covered with benthic algae $_2$	0.5
Benthic algae death rate @ 20°C	Phytoplankton death rate, non-zooplankton predation	-0.5
Benthic algae maximum growth rate @ 20°C	Benthic algae light constant for growth	0.5
Benthic algae maximum growth rate @ 20°C	Benthic algae nitrogen half-saturation constant for growth	-0.5
Benthic algae maximum growth rate @ 20°C	Benthic algae phosphorus half-saturation constant for growth	0.5
Benthic algae maximum growth rate @ 20°C	Phytoplankton maximum growth rate @ 20°C	-0.5
Benthic algae light constant for growth	Phytoplankton optimal light saturation	-0.5
Benthic algae respiration rate @ 20°C	Phytoplankton endogenous respiration rate @ 20°C	-0.5
Benthic algae nitrogen half-saturation constant for growth	Benthic algae phosphorus half-saturation constant for growth	-0.5
Benthic algae phosphorus half-saturation constant for growth	Fraction of bottom segment covered with benthic algae $_1$	0.5
Benthic algae phosphorus half-saturation constant for growth	Fraction of bottom segment covered with benthic algae $_2$	0.5
Benthic algae phosphorus half-saturation constant for growth	Phytoplankton phosphorus half-saturation constant for growth	-0.5
Mineralization rate of dissolved	Settling rate of particulate	0.5

organic phosphorus @ 20°C	phosphorus $_1$	
Mineralization rate of dissolved organic phosphorus @ 20°C	Settling rate of particulate phosphorus $_2$	0.5
Mineralization rate of dissolved organic phosphorus @ 20°C	Settling rate of particulate phosphorus $_3$	0.5
Nitrification rate @ 20°C	SOD	-0.5
Phytoplankton carbon:chlorophyll ratio	Phytoplankton maximum growth rate @ 20°C	0.5
Phytoplankton carbon:chlorophyll ratio	Phytoplankton endogenous respiration rate @ 20°C	0.5
Phytoplankton maximum growth rate @ 20°C	Phytoplankton death rate, non-zooplankton predation	0.5
Phytoplankton maximum growth rate @ 20°C	Phytoplankton optimal light saturation	0.5
Phytoplankton maximum growth rate @ 20°C	Phytoplankton endogenous respiration rate @ 20°C	0.5
Phytoplankton maximum growth rate @ 20°C	Phytoplankton nitrogen half-saturation constant for growth	-0.5
Phytoplankton maximum growth rate @ 20°C	Phytoplankton phosphorus half-saturation constant for growth	0.5
Phytoplankton phosphorus half-saturation constant for growth	Phytoplankton nitrogen half-saturation constant for growth	-0.5

Table 3-7: Summary of data underlying the Passaic River headwater boundary condition for organic phosphorus in the TMDL model (adapted from Omni Environmental, 2007a)

Cluster	Period	Number of observations	Observed COV	TMDL model value
1	Oct-Nov	7	0.55	0.045
2	Dec-Jan	3	0.26	0.037
3	Feb-Mar	3	0.38	0.023
4	Apr-May	4	0.54	0.052
5	Jun-Jul	8	0.26	0.082
6	Aug-Sep	6	0.35	0.096
Average COV: 0.39				

Table 3-8: Summary of data underlying the Passaic River headwater boundary condition for orthophosphate in the TMDL model (adapted from Omni Environmental, 2007a)

Cluster	Period	Number of observations	Observed COV	TMDL model value
1	Oct-Nov	7	0.65	0.033
2	Dec-Jan	3	0.38	0.027
3	Feb-Mar	3	0.40	0.017
4	Apr-May	4	0.06	0.038
5	Jun-Jul	8	0.44	0.059
6	Aug-Sep	6	0.43	0.070
Average COV: 0.39				

Table 3-9: Summary of data underlying the Ramapo River headwater boundary conditions for organic phosphorus and orthophosphate in the TMDL model (adapted from Omni Environmental, 2007a)

	Number of observations	COV	TMDL model value	Average COV, weighted by TMDL model value
Organic phosphorus	26	0.38	0.06	0.42
Orhophosphate	26	0.49	0.03	

Table 3-10: WWTPs included in Stage 1 of the uncertainty analysis

WWTP	Permitted flow (MGD)	Total number of phosphorus effluent measurements*	Number of months with ≤ 10 phosphorus effluent measurements*	Nearest affected target location (Branch-node)	Separate measurements of organic phosphorus and orthophosphate
Berkeley Heights STP	3.1	172	38	Passaic River near Chatham (B11-N33)	No
Rockaway Valley RSA	12	470	3	Passaic River at confl. with Pompton River (B12-N38)	No
Two Bridges SA	10	300	38	WS intake (B5-N23)	No: Oct. 2000 – Apr. 2001 Yes: May 2001 – Nov. 2003
Verona STP	3.0	74	38	Peckman River mouth (B16-N19)	No

*Between October 2000-November 2003, i.e., the Stage 1 timeframe

Table 3-11: Performance of predicted 80% confidence interval for Cluster 1 orthophosphate effluent from Parsippany-Troy Hills RSA

Measurements	Successes	$\hat{p}$	95% Confidence Interval about $\hat{p}$
285	215	0.75	(0.7044, 0.8044)

Table 3-12: Ratio of orthophosphate to organic phosphorus in effluent from select WWTPs modeled as uncertain in Stage 1

WWTP	Mean, measured	Standard deviation, measured	COV, measured	Number of measurements	Ratio used in TMDL model	COV, Stage 1
Berkeley Heights STP	5.07	2.39	0.47	6	4.06	0.25
Rockaway Valley RSA	15.06	14.13	0.94	6	5.07	0.40
Two Bridges SA	13.05	10.80	0.83	188	50.53	0.50

Table 3-13: Number of days (n) in June through August with complete DO diurnal meter data (adapted from Omni Environmental, 2007a)

Location	n
Dundee Lake	5
Peckman River mouth	3
Passaic River near Chatham	9
Station PA10	5

Table 3-14: Predicted 80% confidence interval of TP compared to observed data

Location	Measurements	Successes	$\tilde{p}$	95% Agresti-Coull confidence interval about proportion
WS intake	20	15	0.71	(0.53, 0.89)
B12-N38	32 [a]	15 [a]	0.47 [a]	(0.31, 0.64) [a]
	31 [b]	22 [b]	0.69 [b]	(0.54, 0.84) [b]

[a] Model output processed on moving ± 12 hour time scale
[b] Model output processed on moving ± 48 hour time scale

Table 3-15: Predicted 80% confidence interval of chl-*a* compared to observed data

Location	Measurements	Successes	$\tilde{p}$	95% Agresti-Coull confidence interval about proportion
B17-N25	9	5	0.54	(0.27, 0.81)

Table 3-16: Predicted 80% confidence interval of DO compared to observed data

Location	Measurements	Successes	$\tilde{p}$	95% Agresti-Coull confidence interval about proportion
Dundee Lake	17	12	0.71	(0.49, 0.92)
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)

Table 3-17: Significant kinetic parameters with r ≥ |0.5| that affect average TP from October 2000 through April 2001, as determined by GSA

Location	Kinetic parameter	Correlation coefficient
WS intake	None	NA
B12-N38	Settling rate of particulate phosphorus $_2$	-0.91
Little Falls intake	Settling rate of particulate phosphorus $_2$	-0.74

Table 3-18: Significant kinetic parameters with r ≥ |0.5| that affect average TP from May through September 2001, as determined by GSA

Location	Kinetic parameter	Correlation coefficient
WS intake	None	NA
B12-N38	Settling rate of particulate phosphorus $_2$	-0.88
Little Falls intake	Settling rate of particulate phosphorus $_2$	-0.78

Table 3-19: Significant kinetic parameters with r ≥ |0.5| that affect average chl-*a* from May through September 2001, as determined by GSA

Location	Kinetic parameter	Correlation coefficient
Dundee Lake	Phytoplankton maximum growth rate @ 20°C	0.88

Table 3-20: Significant kinetic parameters with r ≥ |0.5| that affect average DO from May through September 2001, as determined by GSA

Location	Kinetic parameter	Correlation coefficient
Dundee Lake	Benthic algae maximum growth rate @ 20°C	0.66
	Benthic algae respiration rate @ 20°C	-0.65
Peckman River mouth	Benthic algae death rate @ 20°C	0.58
	Benthic algae maximum growth rate	0.56
	SOD	-0.51
Passaic River near Chatham	SOD	-0.87
PA10	Benthic algae maximum growth rate @ 20°C	0.57
	Benthic algae respiration rate @ 20°C	-0.73

Table 3-21: Significant kinetic parameters with r ≥ |0.5| that affect average diurnal DO swing from May through September 2001, as determined by GSA

Location	Kinetic parameter	Correlation coefficient
Dundee Lake	Benthic algae maximum growth rate @ 20°C	0.77
	Fraction of bottom segment covered with benthic algae$_1$	0.67
Peckman River mouth	Benthic algae maximum growth rate @ 20°C	0.80
	Fraction of bottom segment covered with benthic algae$_1$	0.67
Passaic River near Chatham	Benthic algae maximum growth rate @ 20°C	0.85
	Fraction of bottom segment covered with benthic algae$_1$	0.60
PA10	Benthic algae maximum growth rate @ 20°C	0.82
	Fraction of bottom segment covered with benthic algae$_1$	0.53

Table 3-22: Matrix of input variables for subplots in Figures 3-74 through 3-88

Benthic algae ammonia preference	Benthic algae death rate @ 20°C	Benthic algae maximum growth rate @ 20°C
Benthic algae light constant for growth	Benthic algae respiration rate @ 20°C	Benthic algae nitrogen half-saturation constant for growth
Benthic algae phosphorus half-saturation constant for growth	Dummy variable $_1$	Dissolved fraction of orthophosphate $_4$
Dissolved fraction of orthophosphate $_1$	Dissolved fraction of orthophosphate $_3$	Dissolved fraction of orthophosphate $_2$
Mineralization rate of dissolved organic phosphorus @ 20°C	Fraction of bottom segment covered with benthic algae $_1$	Fraction of bottom segment covered with benthic algae $_2$
Dummy variable $_2$	Nitrification rate @ 20°C	Phytoplankton carbon:chlorophyll ratio
Phytoplankton death rate, non-zooplankton predation	Phytoplankton maximum growth rate @ 20°C	Phytoplankton optimal light saturation
Phytoplankton endogenous respiration rate @ 20°C	Phytoplankton nitrogen half-saturation constant for growth	Phytoplankton phosphorus half-saturation constant for growth
Dummy variable $_3$	Dummy variable $_4$	Settling rate of particulate phosphorus $_1$
Settling rate of particulate phosphorus $_2$	Settling rate of particulate phosphorus $_3$	SOD
Dummy variable $_5$	Dummy variable $_6$	Dummy variable $_7$

Table 3-23: WASP kinetic parameters retained for use in Stages 2 and 3 of the uncertainty analysis

Global kinetic parameters [unit]			
Phytoplankton maximum growth rate @ 20°C [/d]	Phytoplankton carbon: chlorophyll ratio []	Phytoplankton endogenous respiration rate @ 20°C [/d]	Phytoplankton death rate, non-zooplankton predation [/d]
Phytoplankton optimal light saturation [langleys/d]	Benthic algae maximum growth rate @ 20°C [gD/m^2/d]	Benthic algae respiration rate @ 20°C [/d]	Benthic algae death rate @ 20°C [/d]
Local kinetic parameters [unit]			
SOD [g/m^2/d]	Settling velocity of particulate phosphorus$_{1,2,3}$ [cm/s]		Fraction of bottom segment covered with benthic algae$_{1,2}$ []
Dissolved fraction of orthophosphate$_1$ []			

Table 3-24: WASP kinetic parameters treated as fixed variables in Stages 2 and 3 of the uncertainty analysis

Global kinetic parameters [unit]			
Mineralization rate of dissolved organic phosphorus @ 20°C [/d]	Phytoplankton half-saturation constant for nitrogen [mgN/L]	Phytoplankton half-saturation constant for phosphorus [mgP/L]	Nitrification rate @ 20°C [/d]
Benthic algae ammonia preference [mgN/L]	Benthic algae light constant for growth [langleys/d]	Benthic algae nitrogen half-saturation constant for growth [mgN/L]	Benthic algae phosphorus half-saturation constant for growth [mgP/L]
Local kinetic parameters [unit]			
Dissolved fraction of orthophosphate$_{2,3,4}$ []			

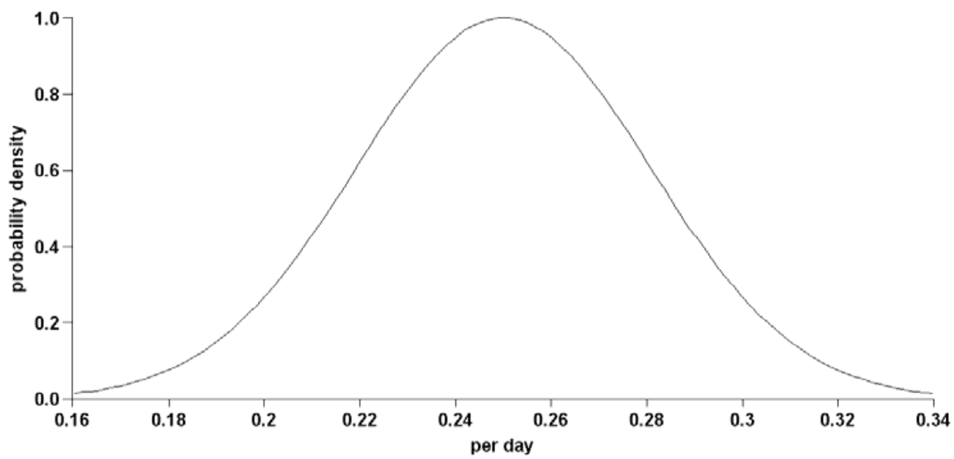


Figure 3-1: PDF of nitrification rate @ 20°C

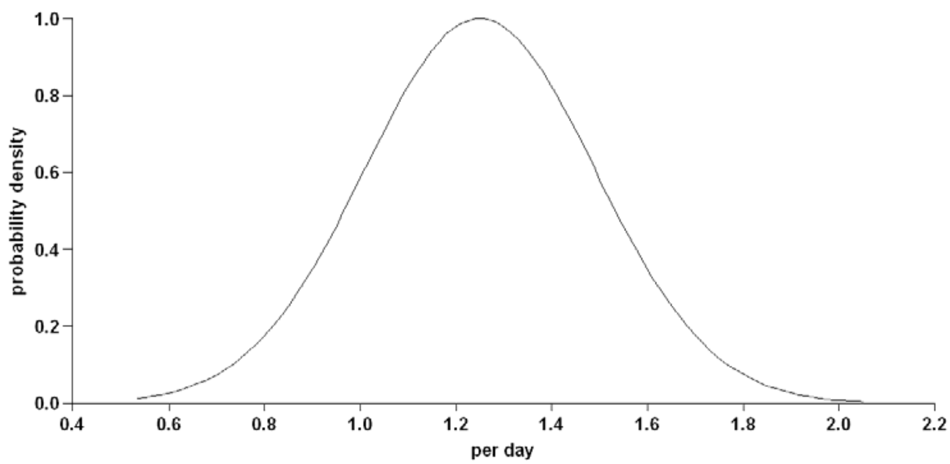


Figure 3-2: PDF of phytoplankton maximum growth rate @ 20°C

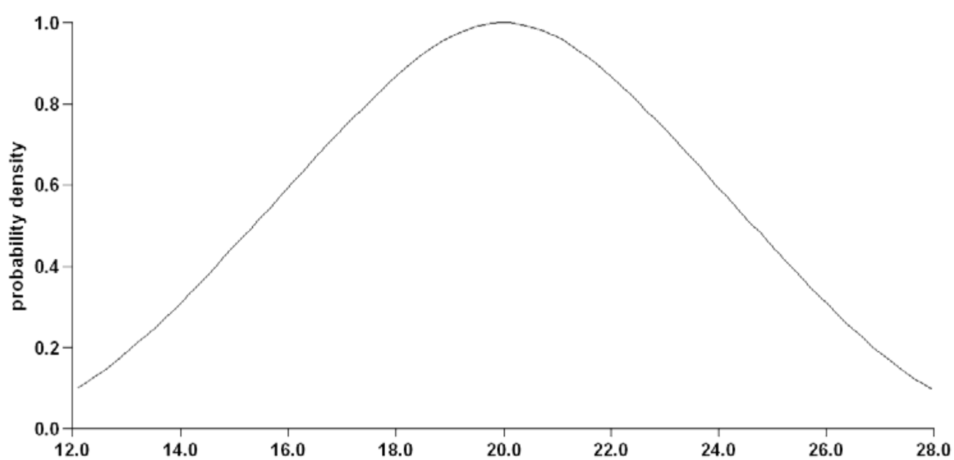


Figure 3-3: PDF of phytoplankton carbon:chlorophyll ratio

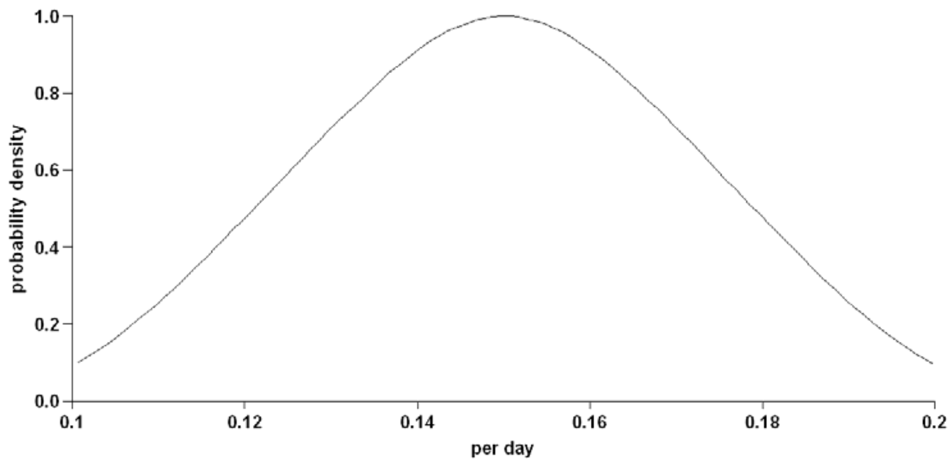


Figure 3-4: PDF of phytoplankton endogenous respiration rate @ 20°C

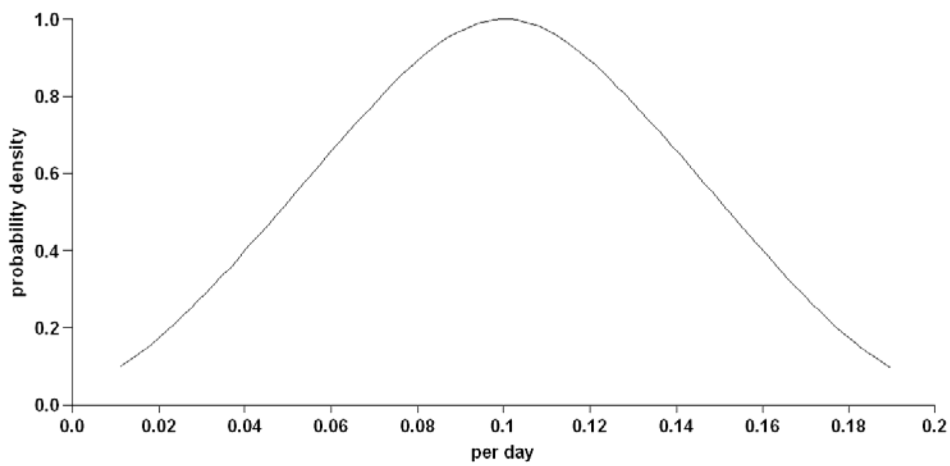


Figure 3-5: PDF of phytoplankton death rate, non-zooplankton predation

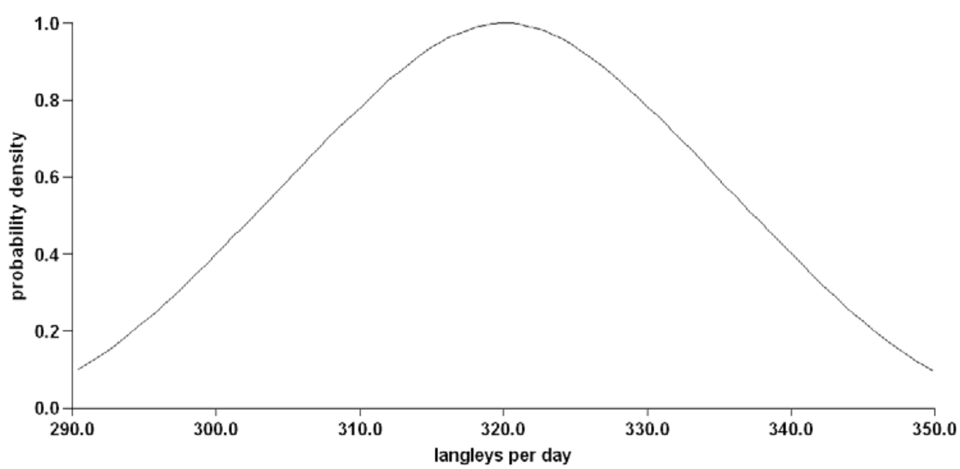


Figure 3-6: PDF of phytoplankton optimal light saturation

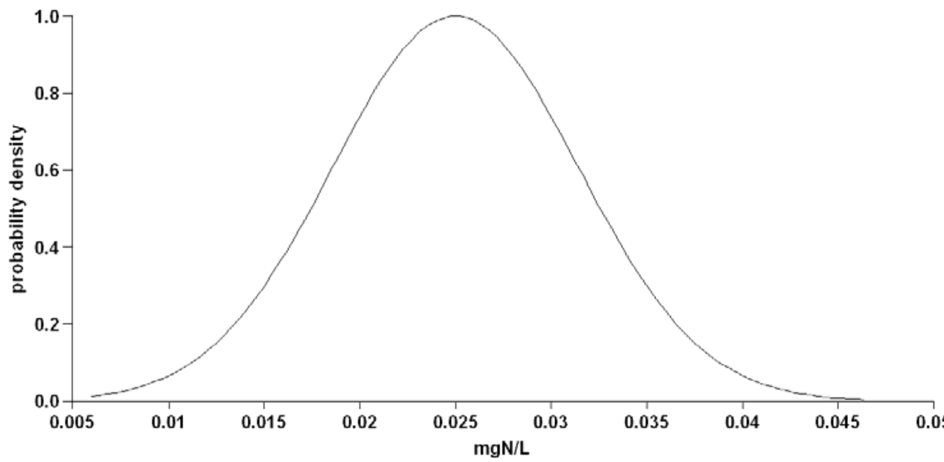


Figure 3-7: PDF of phytoplankton nitrogen half-saturation constant for growth

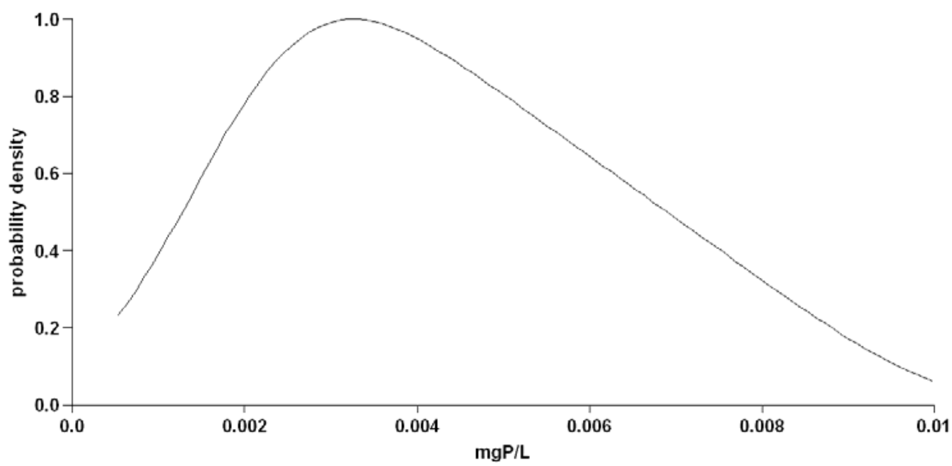


Figure 3-8: PDF of phytoplankton phosphorus half-saturation constant for growth

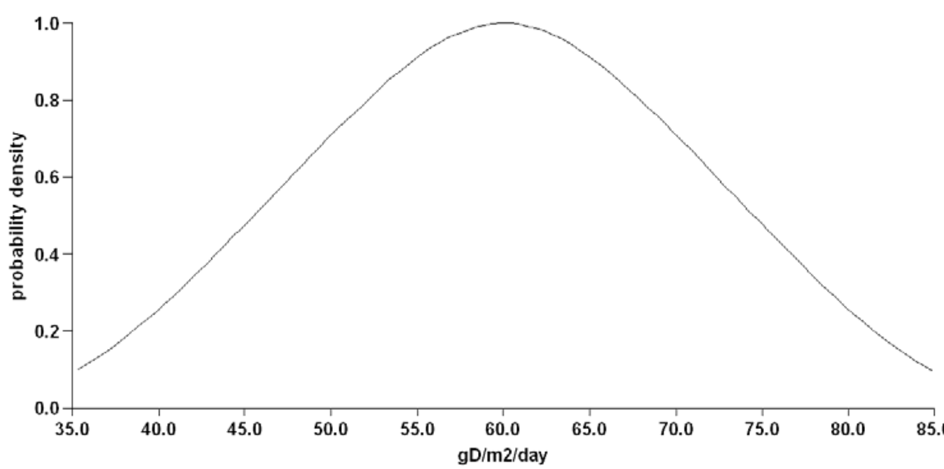


Figure 3-9: PDF of benthic algae maximum growth rate @ 20°C

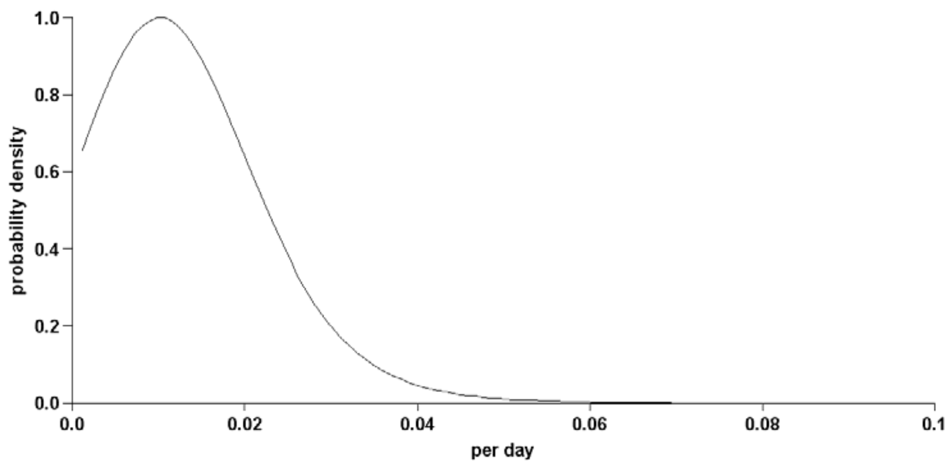


Figure 3-10: PDF of benthic algae respiration rate @ 20°C

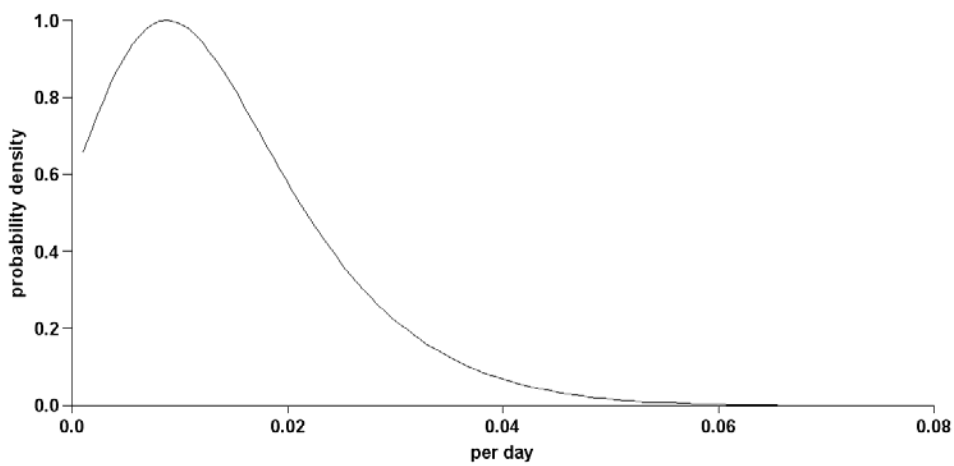


Figure 3-11: PDF of benthic algae death rate @ 20°C

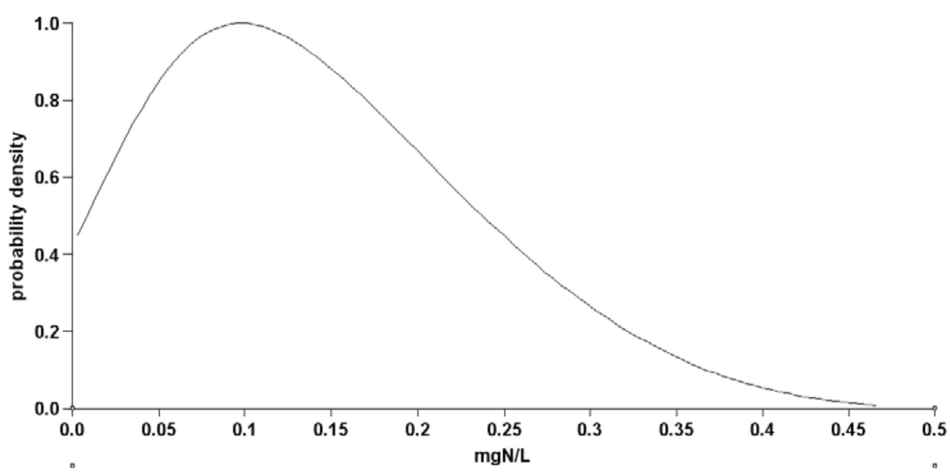


Figure 3-12: PDF of benthic algae ammonia preference

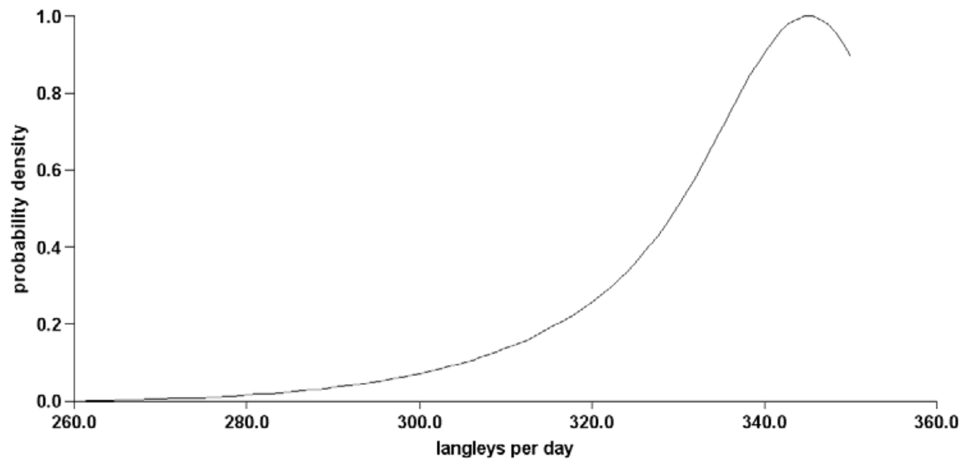


Figure 3-13: PDF of benthic algae light constant for growth

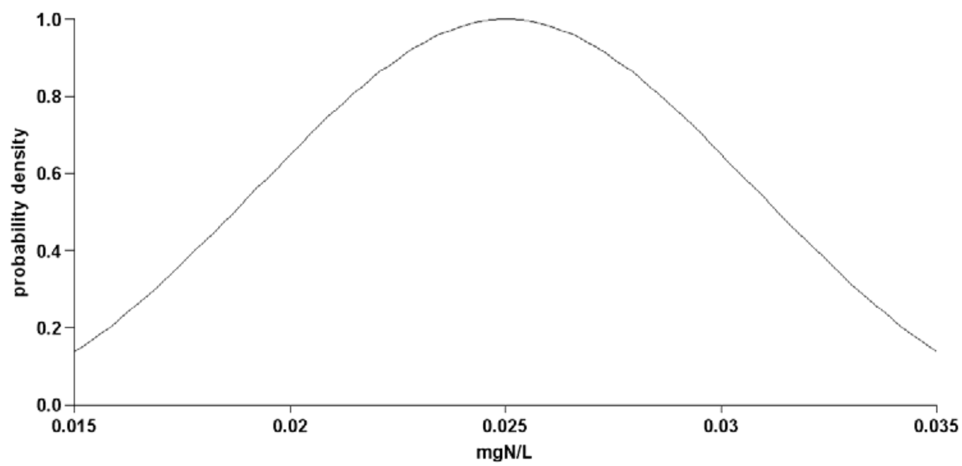


Figure 3-14: PDF of benthic algae nitrogen half-saturation constant for growth

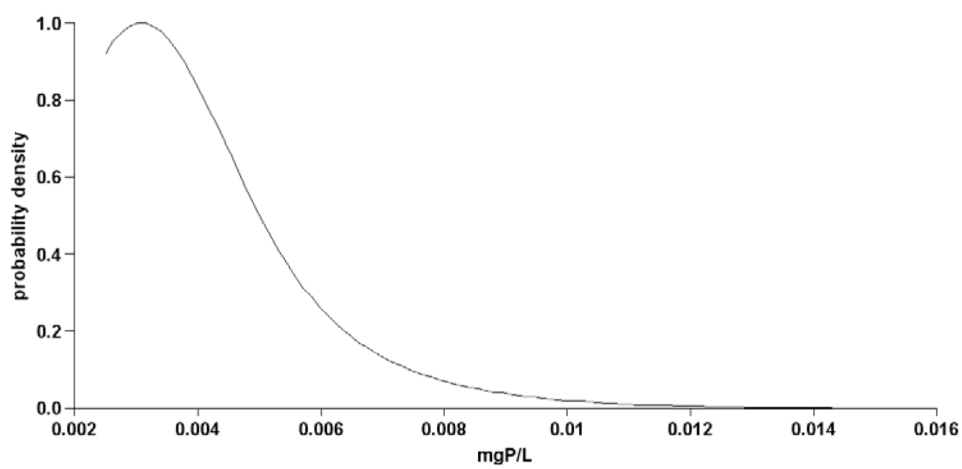


Figure 3-15: PDF of benthic algae phosphorus half-saturation constant for growth

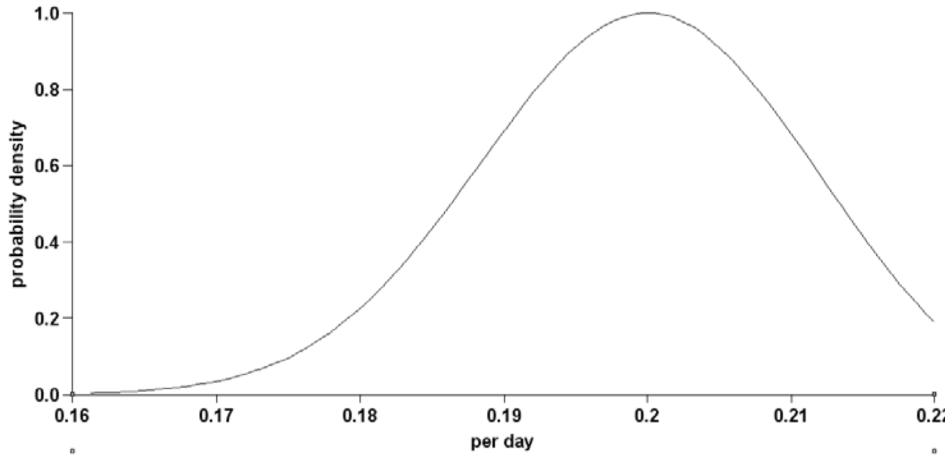


Figure 3-16: PDF of mineralization rate of dissolved organic phosphorus @ 20°C

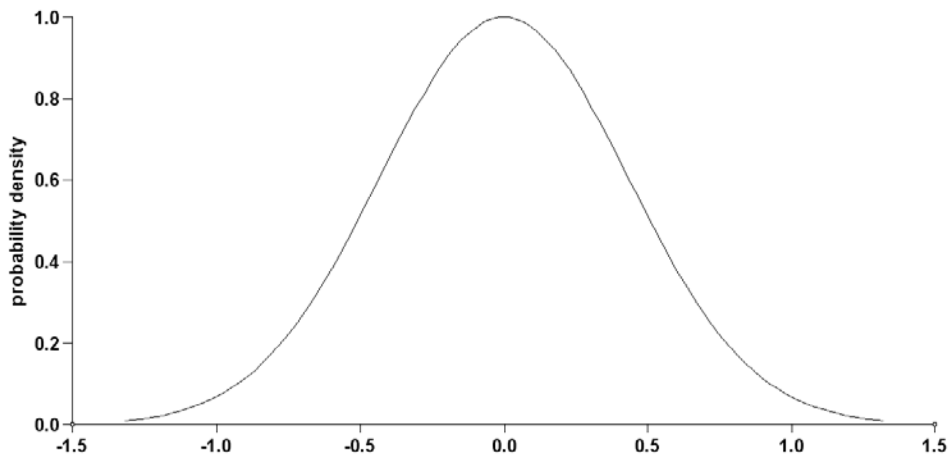


Figure 3-17: PDF of surrogate variable for SOD

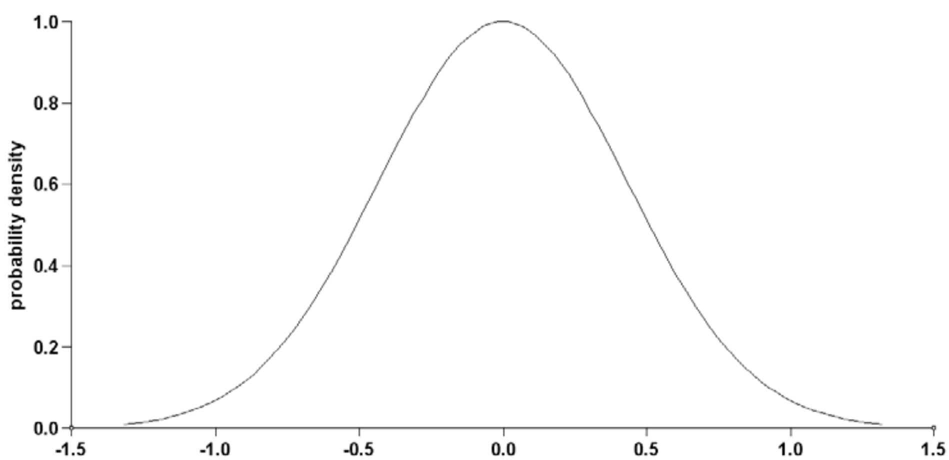


Figure 3-18: PDF of surrogate variable for Fraction of bottom segment covered with benthic algae $_1$

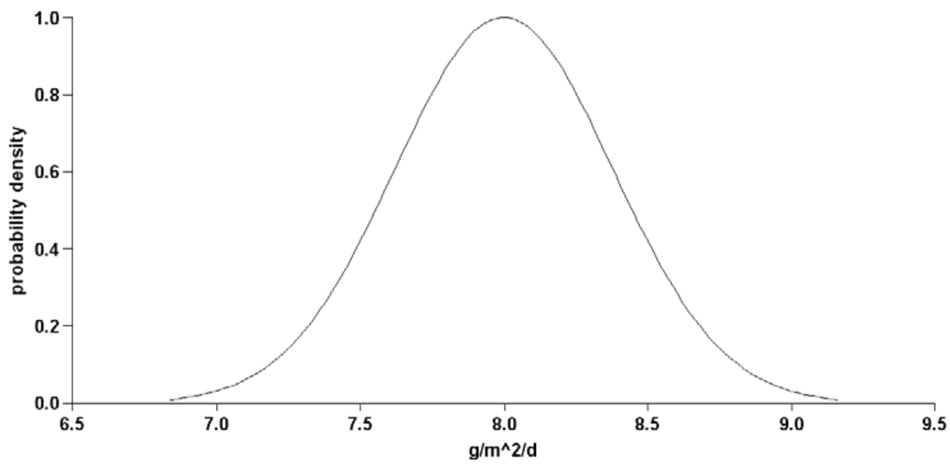


Figure 3-19: PDF of SOD at segments 179-181 and 251-256

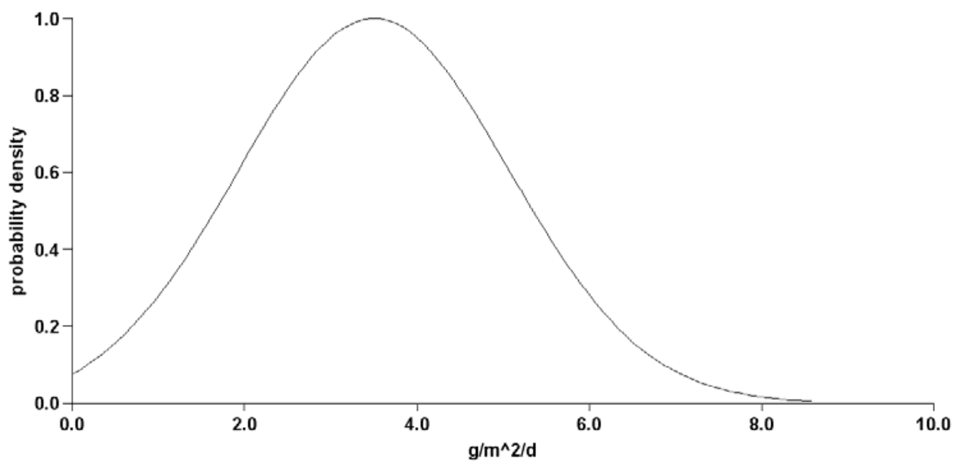


Figure 3-20: PDF of SOD at segment 54

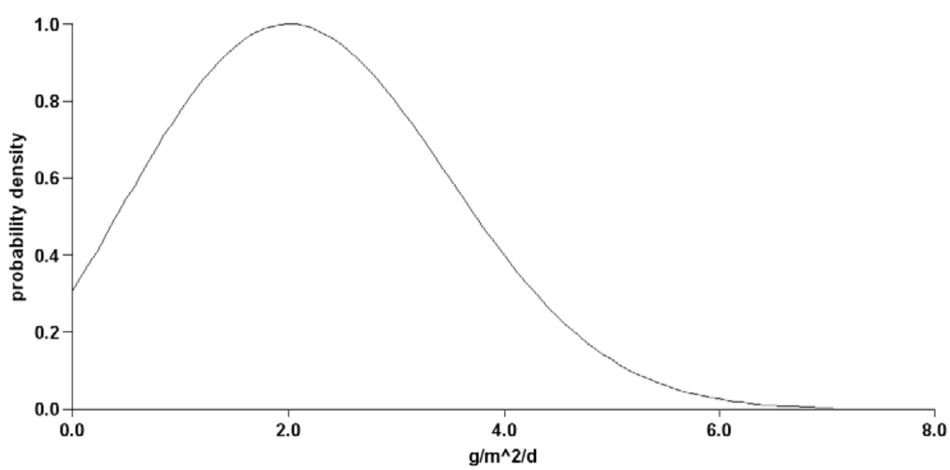


Figure 3-21: PDF of SOD at segments 295-298

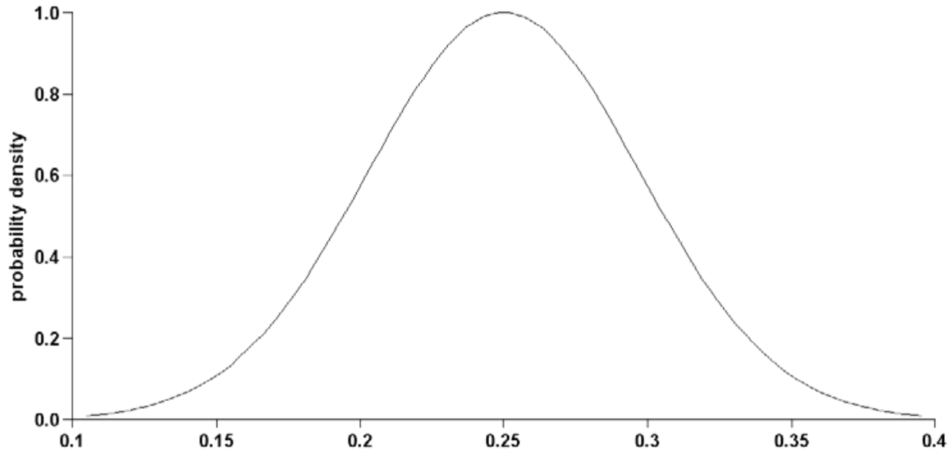


Figure 3-22: PDF of fraction of bottom segment covered with benthic algae at segments 179-181

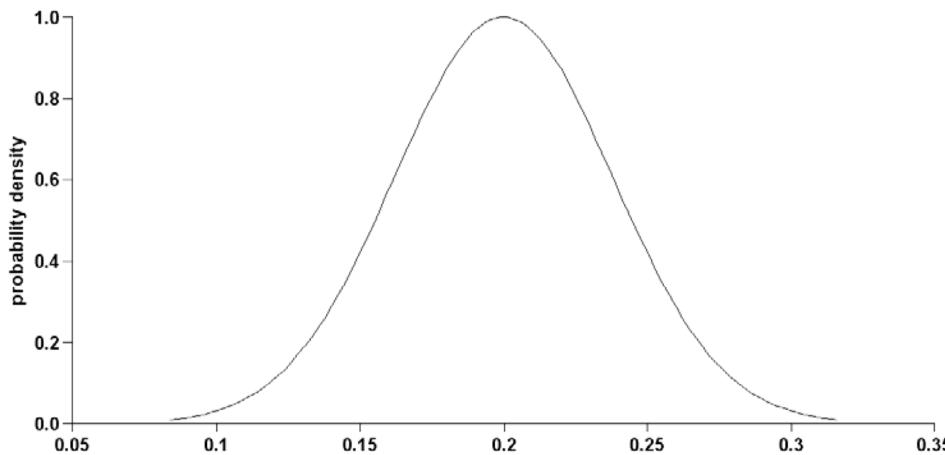


Figure 3-23: PDF of fraction of bottom segment covered with benthic algae at segments 292-294 and 316

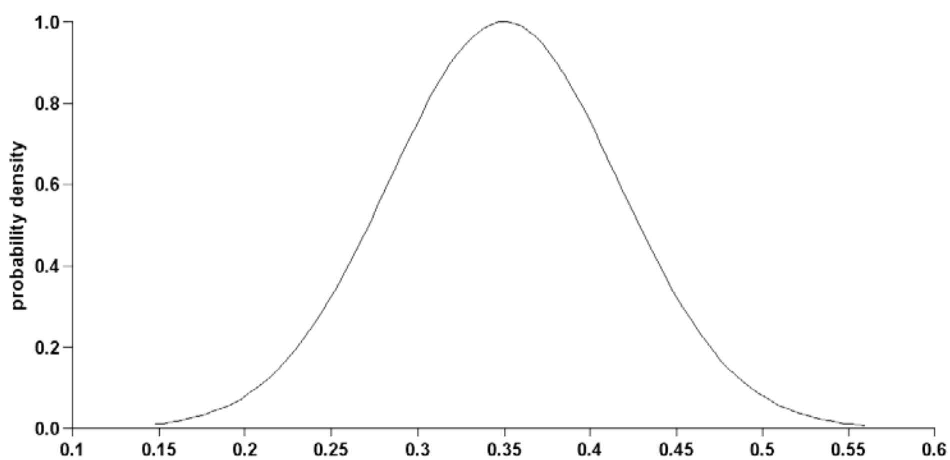


Figure 3-24: PDF of fraction of bottom segment covered with benthic algae at segments 295-298

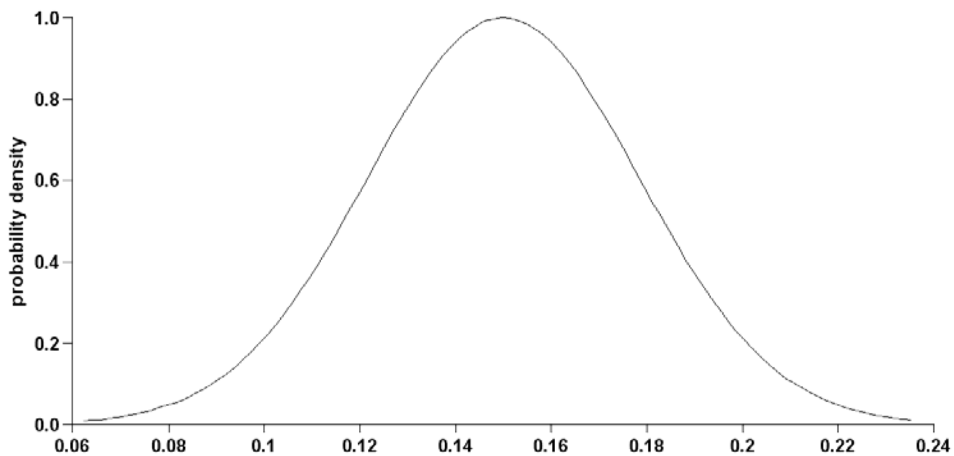


Figure 3-25: PDF of fraction of bottom segment covered with benthic algae at segments 255-256 and 319-325

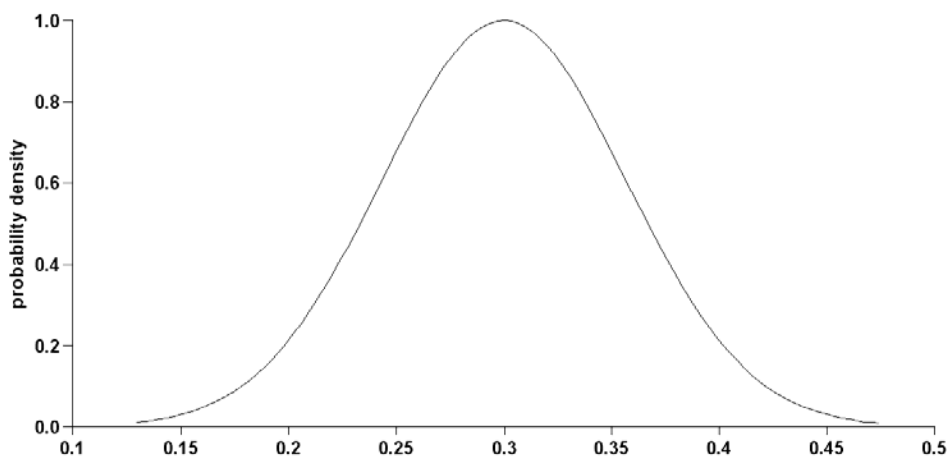


Figure 3-26: PDF of fraction of bottom segment covered with benthic algae at segments 317-318 and 326

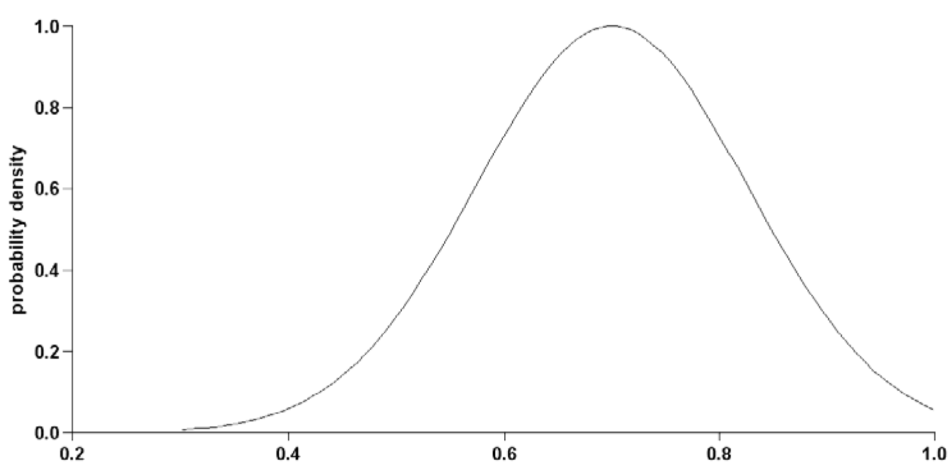


Figure 3-27: PDF of fraction of bottom segment covered with benthic algae at segment 327

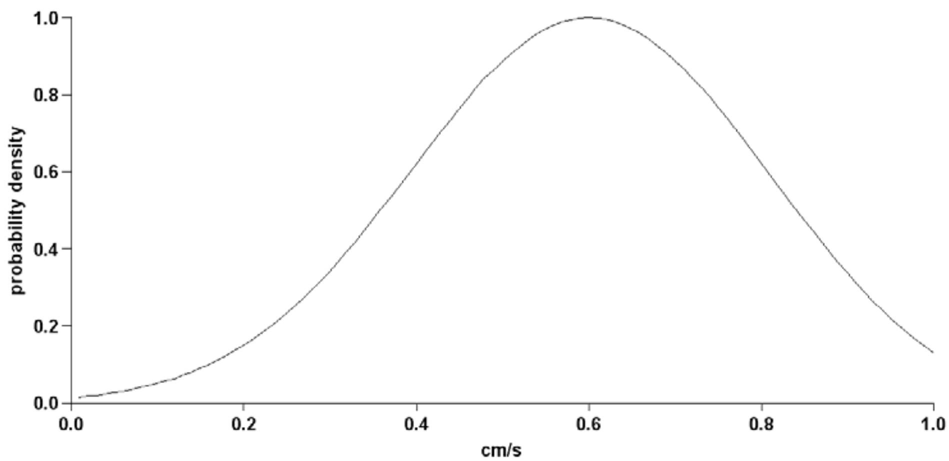


Figure 3-28: PDF of settling rate of particulate phosphorus $_1$

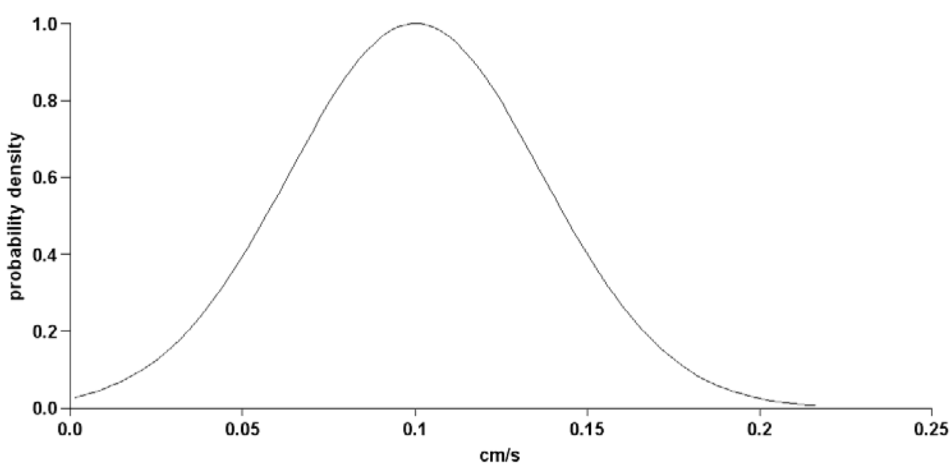


Figure 3-29: PDF of settling rate of particulate phosphorus $_2$

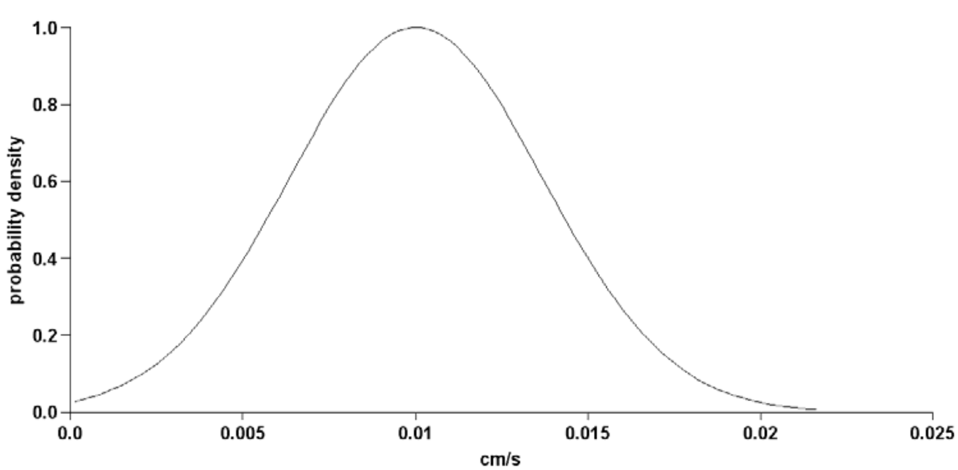


Figure 3-30: PDF of settling rate of particulate phosphorus $_3$

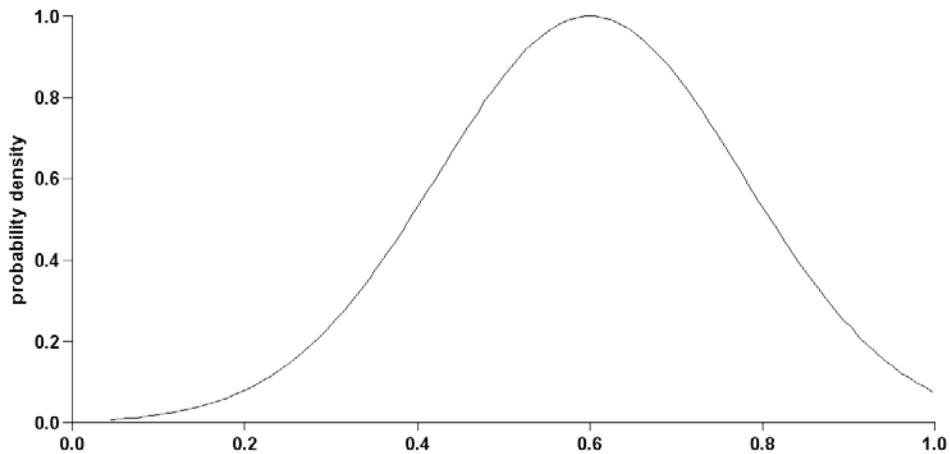


Figure 3-31: PDF of dissolved fraction of orthophosphate $_{1, 2, 3, 4}$

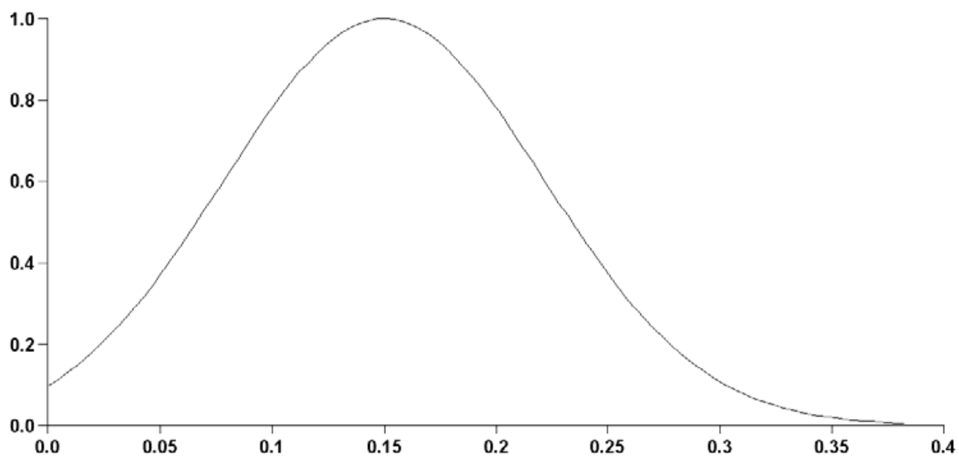


Figure 3-32: PDF of fraction of bottom segment covered with benthic algae at segments 299-303

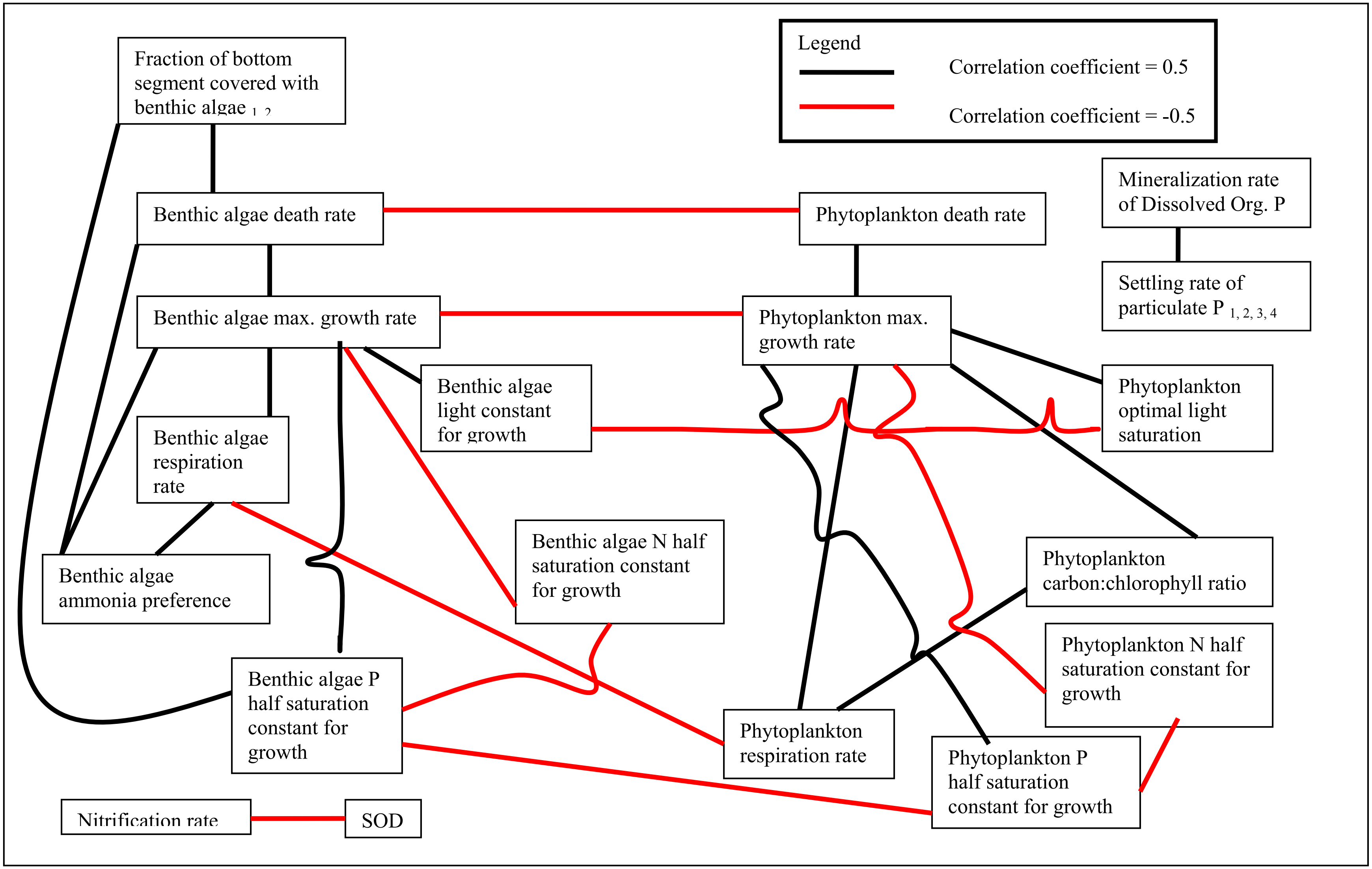


Figure 3-33: Illustration of parameter covariances listed in Table 3-6

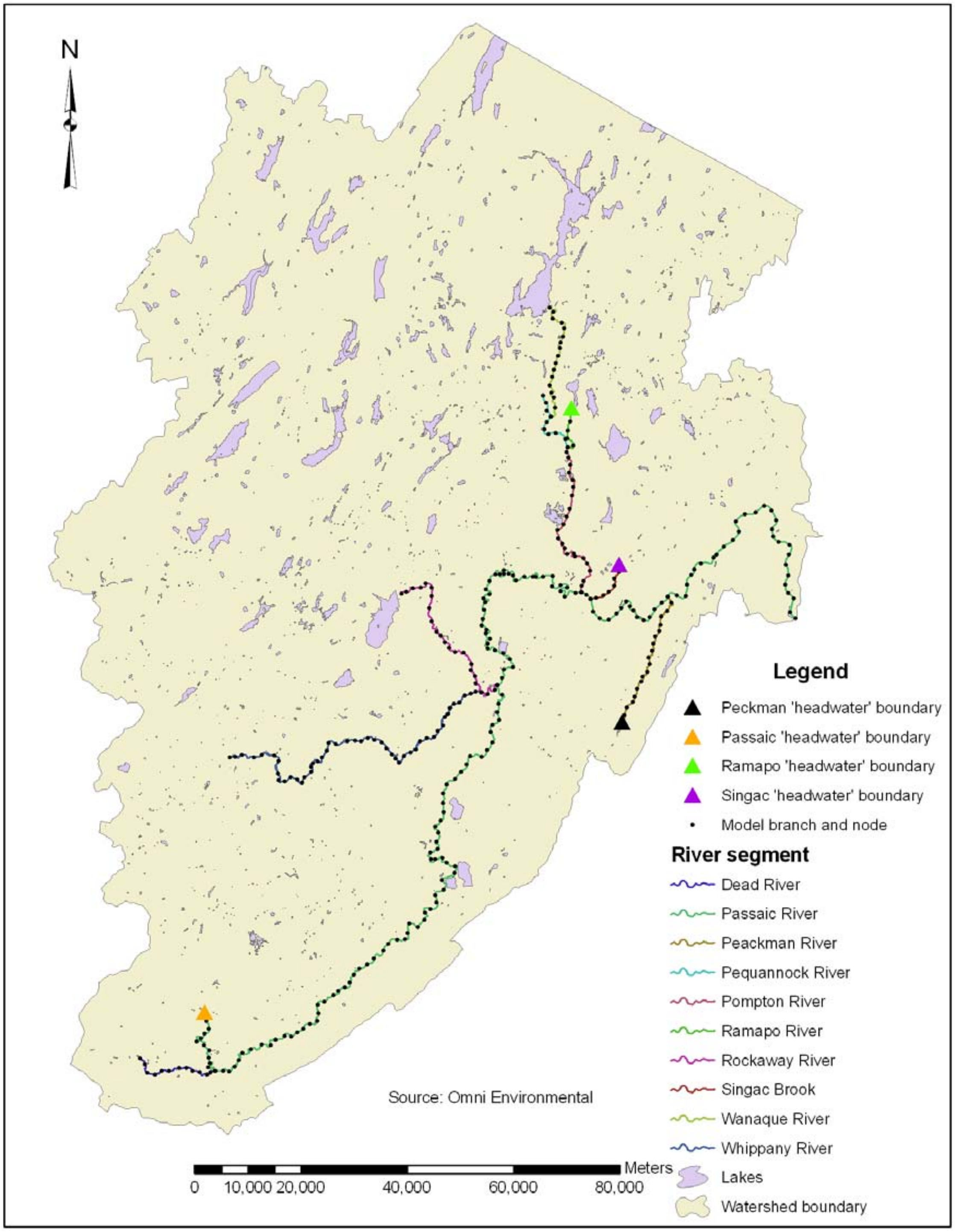


Figure 3-34: Locations of Peckman River, Passaic River, Ramapo River, and Singac Brook 'headwaters' in TMDL model domain

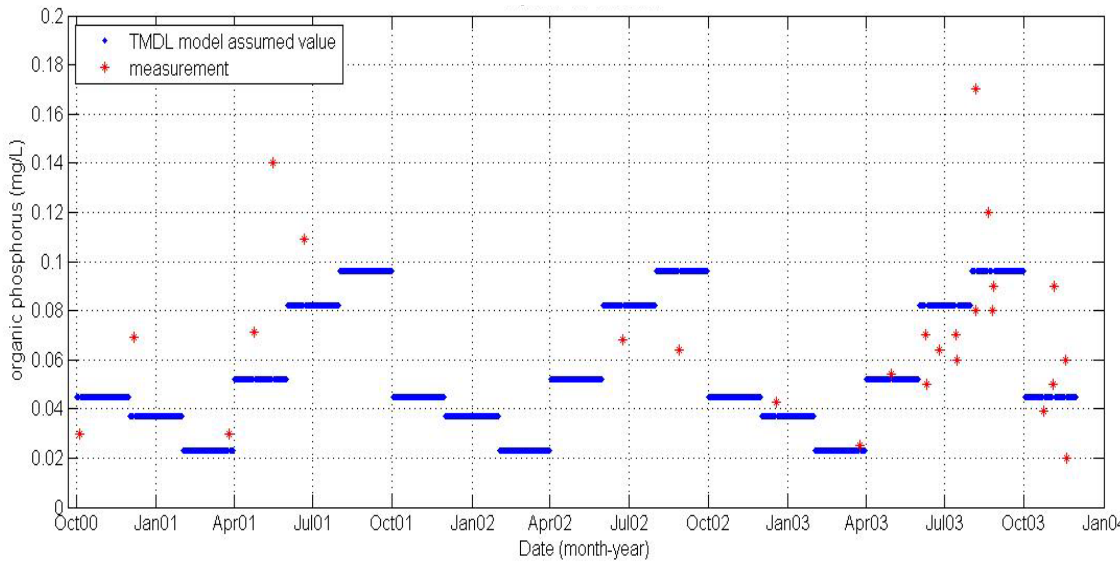


Figure 3-35: Simulated time series of organic phosphorus at Passaic River headwater boundary condition in the TMDL model (adapted from data published in Omni Environmental, 2007a)

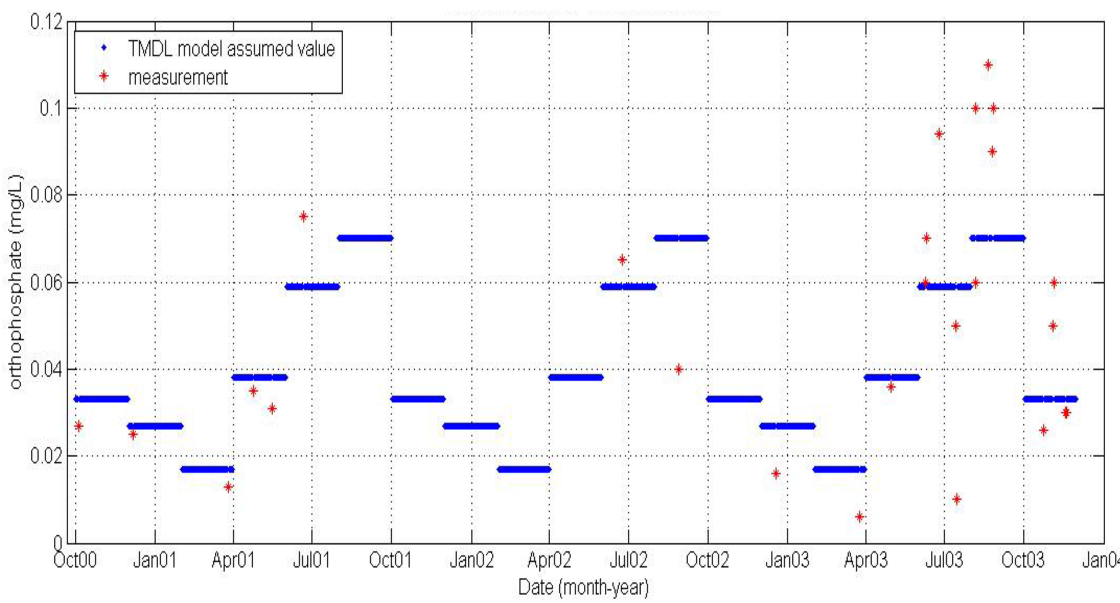


Figure 3-36: Simulated time series of orthophosphate at Passaic River headwater boundary condition in the TMDL model (adapted from data published in Omni Environmental, 2007a)

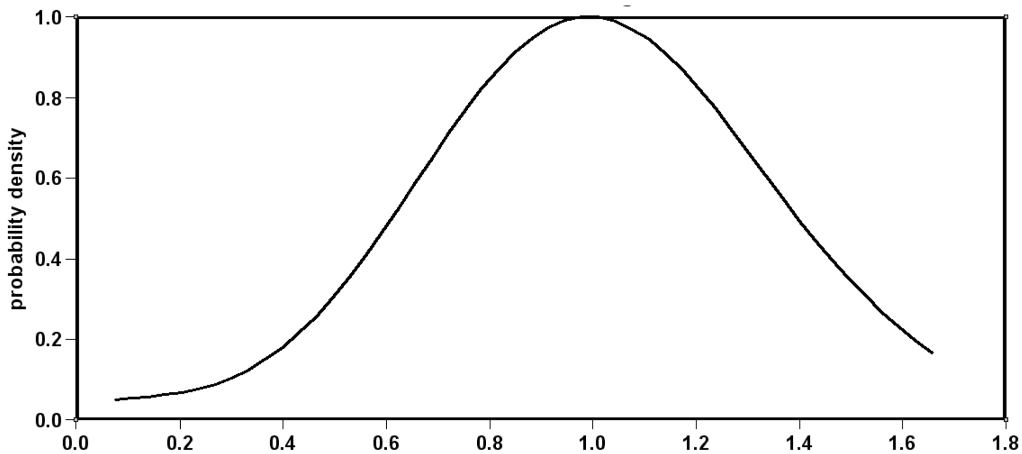


Figure 3-37: PDF of scaling factor applied to Passaic River headwater boundary condition for phosphorus

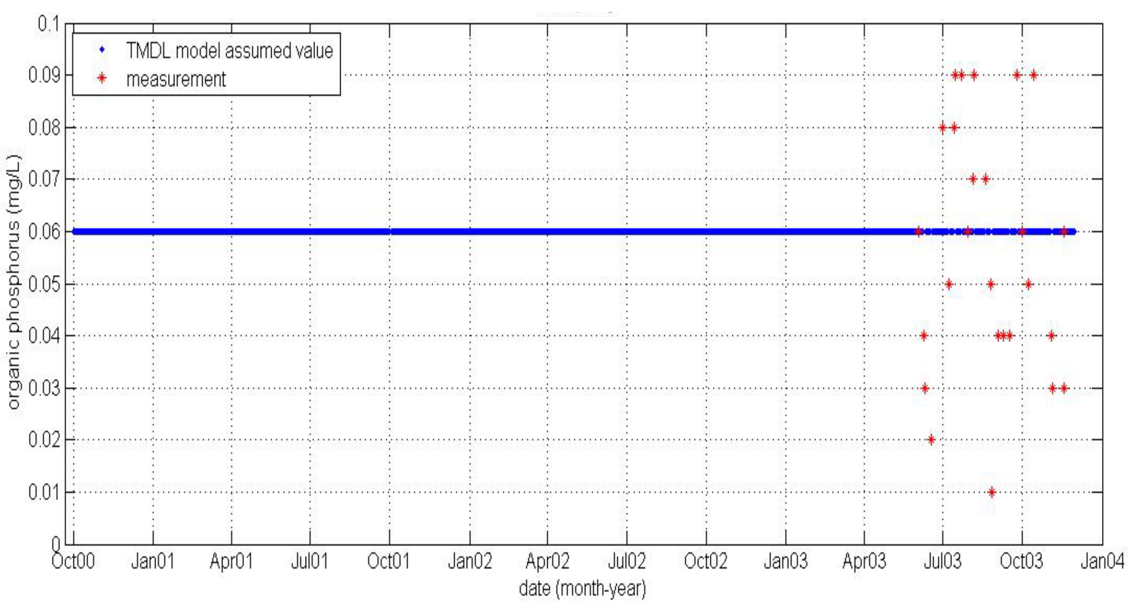


Figure 3-38: Simulated time series of organic phosphorus at Ramapo River headwater boundary condition in the TMDL model (adapted from data published in Omni Environmental, 2007a)

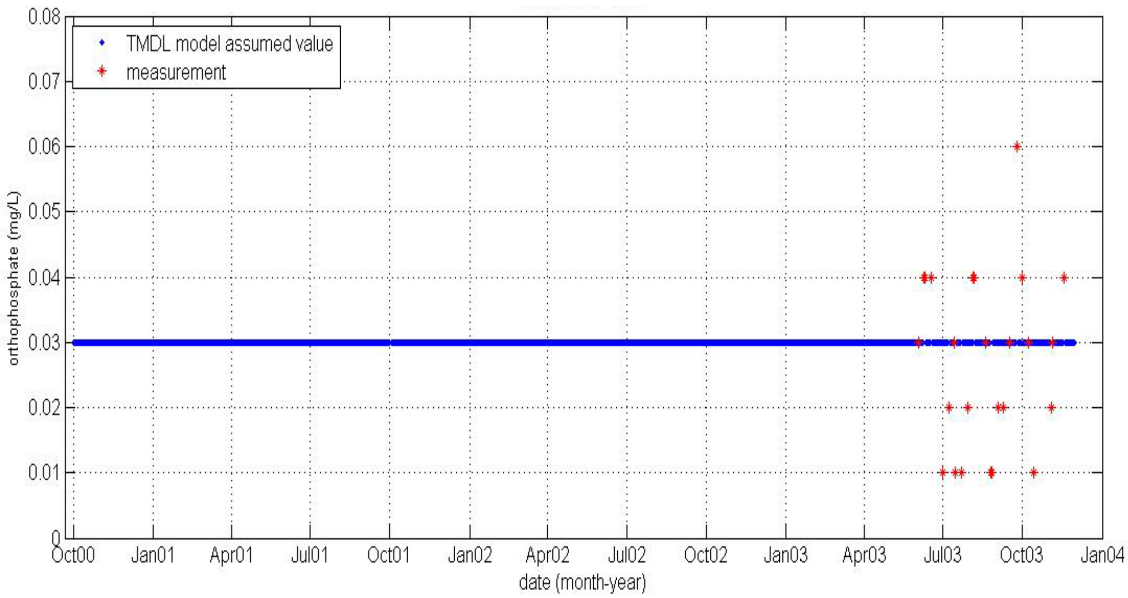


Figure 3-39: Simulated time series of orthophosphate at Ramapo River headwater boundary condition in the TMDL model (adapted from data published in Omni Environmental, 2007a)

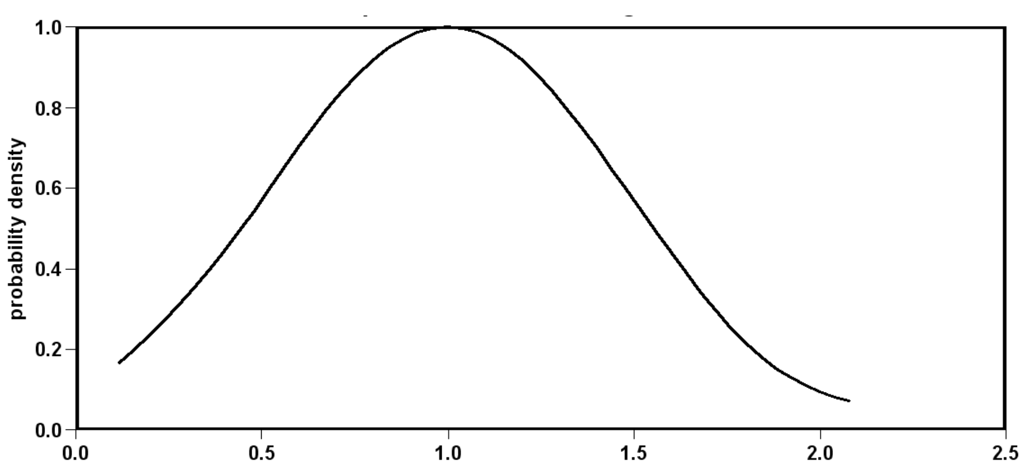


Figure 3-40: PDF of scaling factor applied to Ramapo River headwater boundary condition for phosphorus

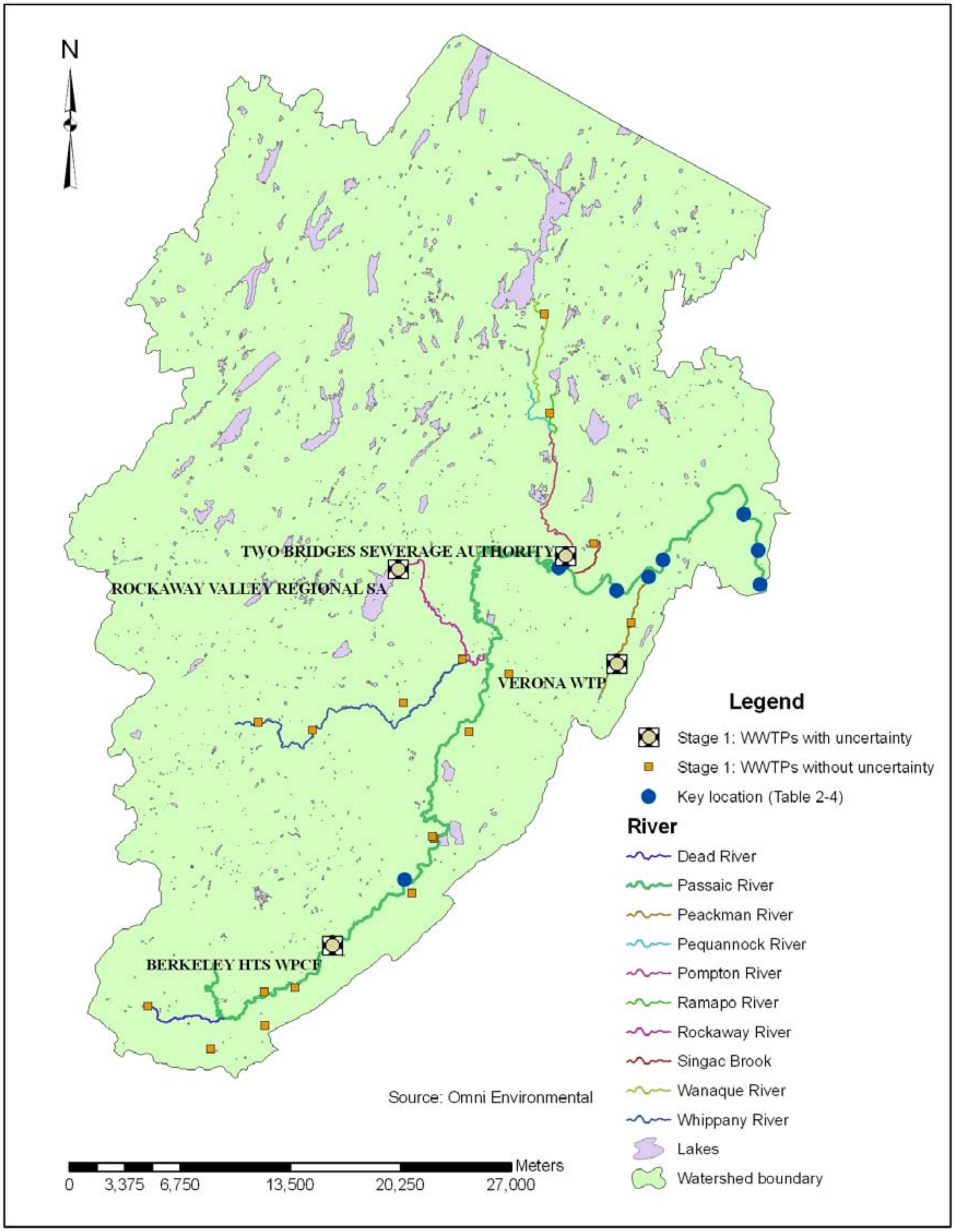


Figure 3-41: WWTPs included in Stage 1 of the uncertainty analysis

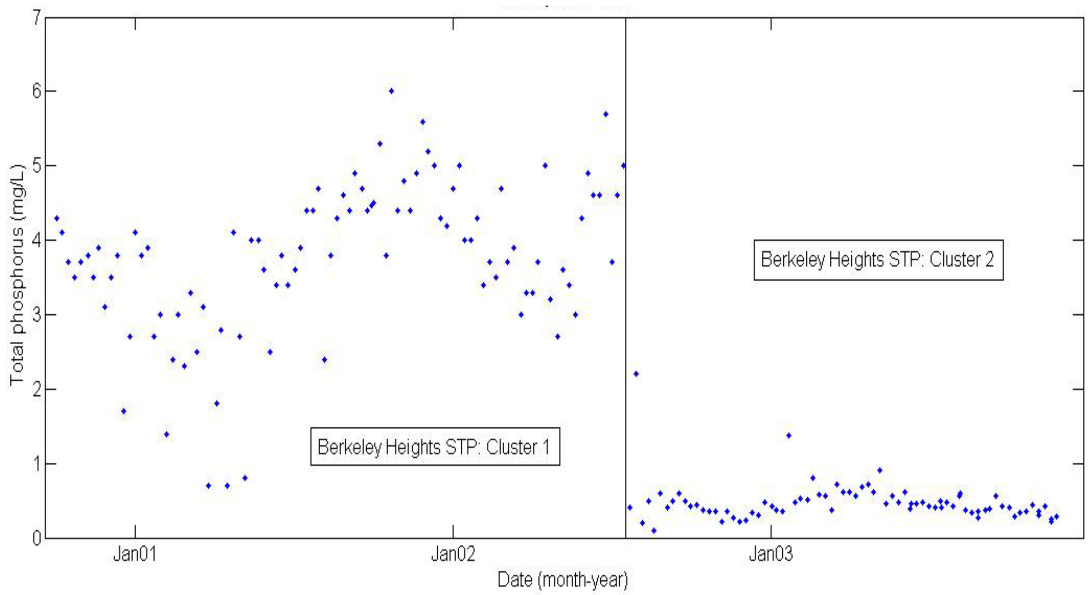


Figure 3-42: TP effluent data of Berkeley Heights STP from 10/1/00 – 11/30/03
(adapted from data published in Omni Environmental, 2007a)

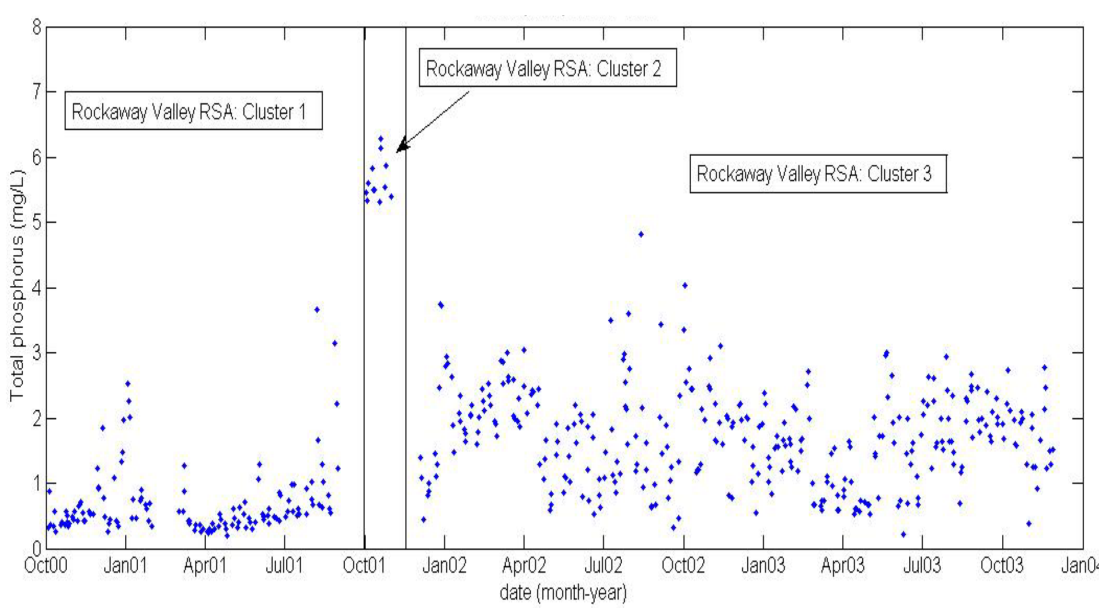


Figure 3-43: TP effluent data of Rockaway Valley RSA from 10/1/00 – 11/30/03
(adapted from data published in Omni Environmental, 2007a)

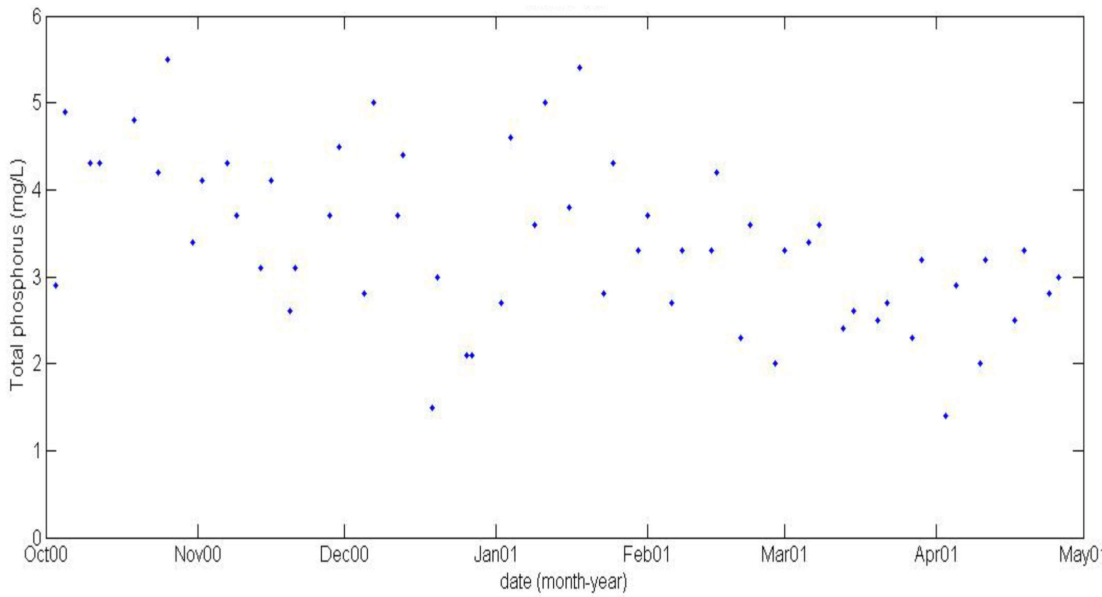


Figure 3-44: TP effluent data of Two Bridges SA from 10/1/00 – 4/30/01 (adapted from data published in Omni Environmental, 2007a)

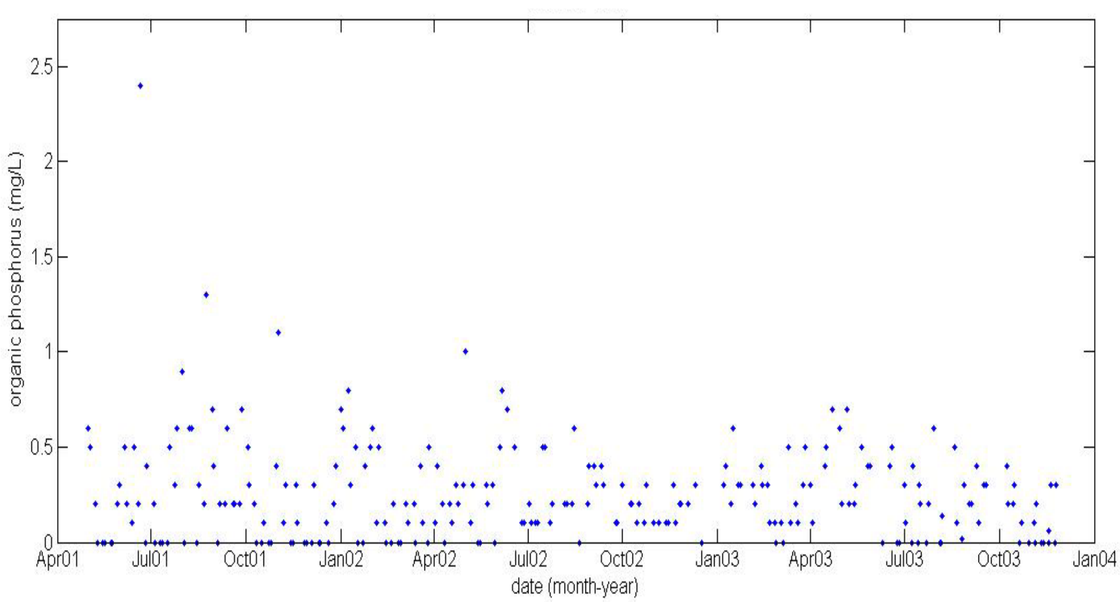


Figure 3-45: Organic phosphorus effluent data of Two Bridges SA from 5/1/01 – 11/30/03 (adapted from data published in Omni Environmental, 2007a)

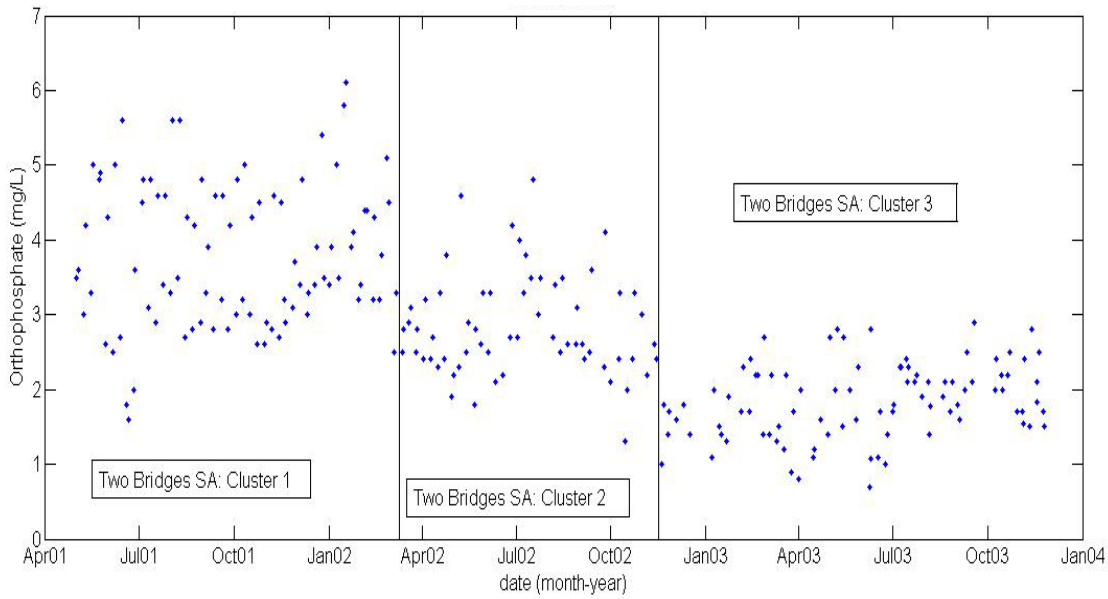


Figure 3-46: Orthophosphate effluent data of Two Bridges SA from 5/1/01 – 11/30/03 (adapted from data published in Omni Environmental, 2007a)

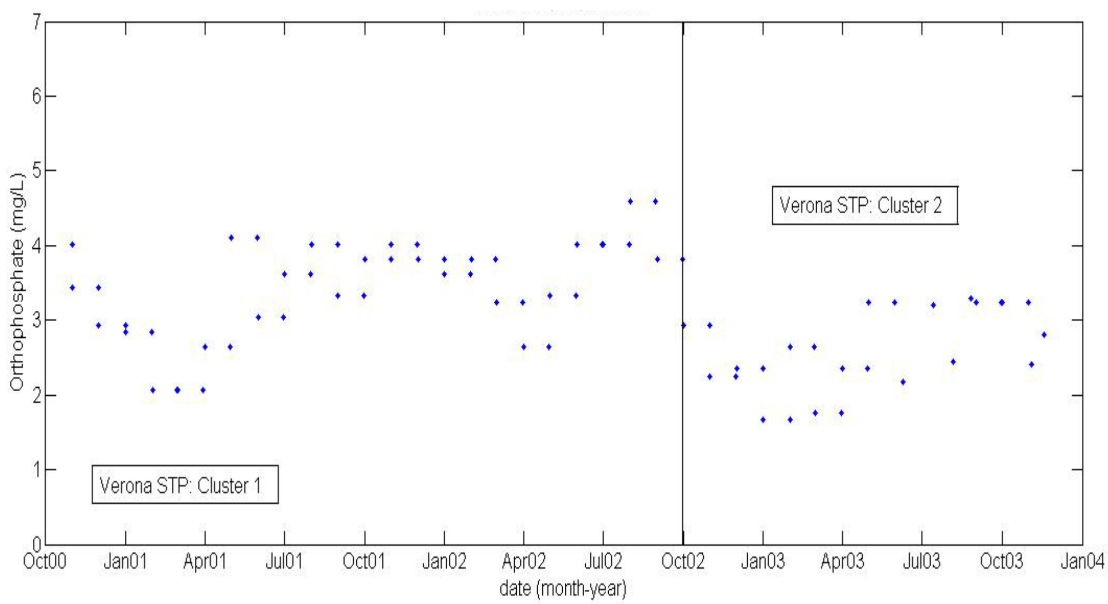


Figure 3-47: TMDL model-simulated orthophosphate effluent of Verona STP from 10/1/00 – 11/30/03 (adapted from data published in Omni Environmental, 2007a)

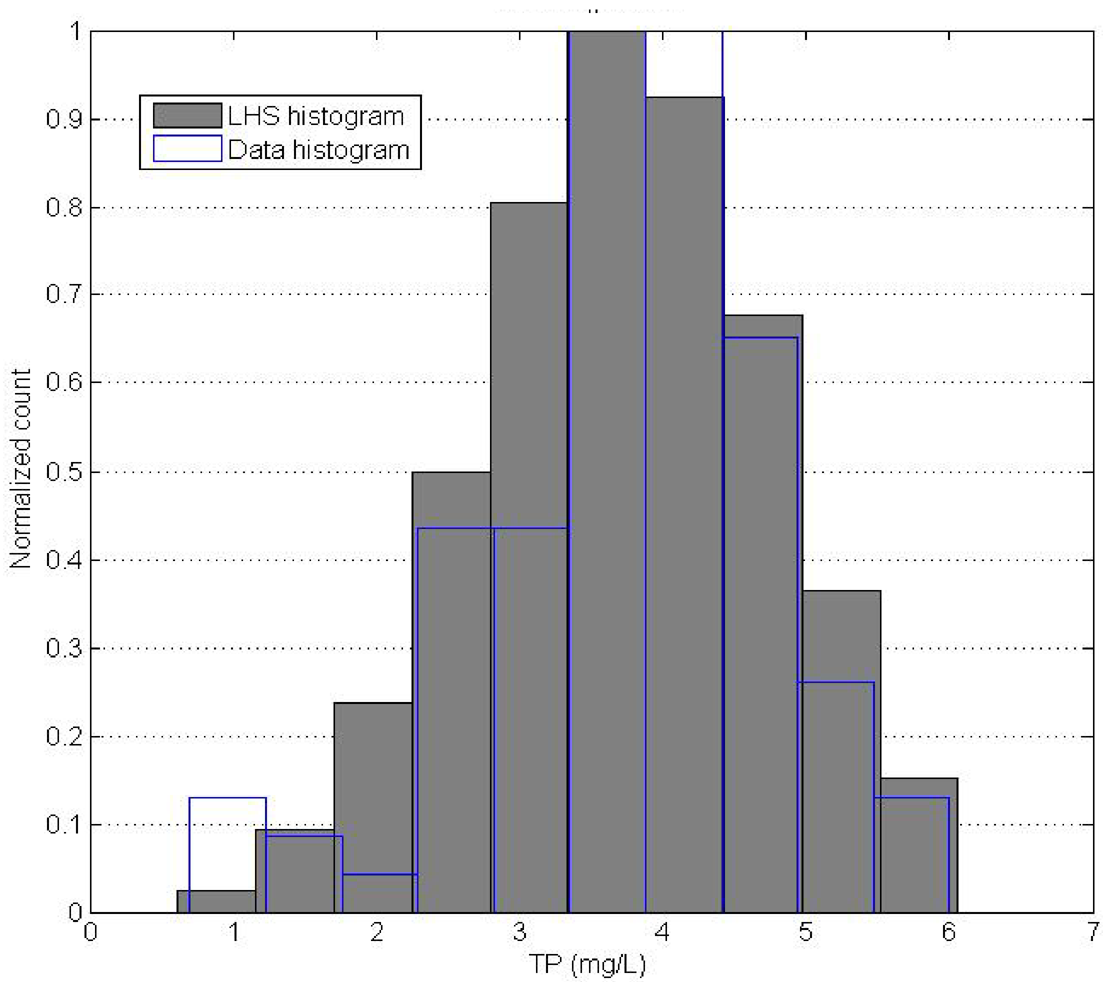


Figure 3-48: Comparison of histograms for Cluster 1 TP effluent from Berkeley Heights STP

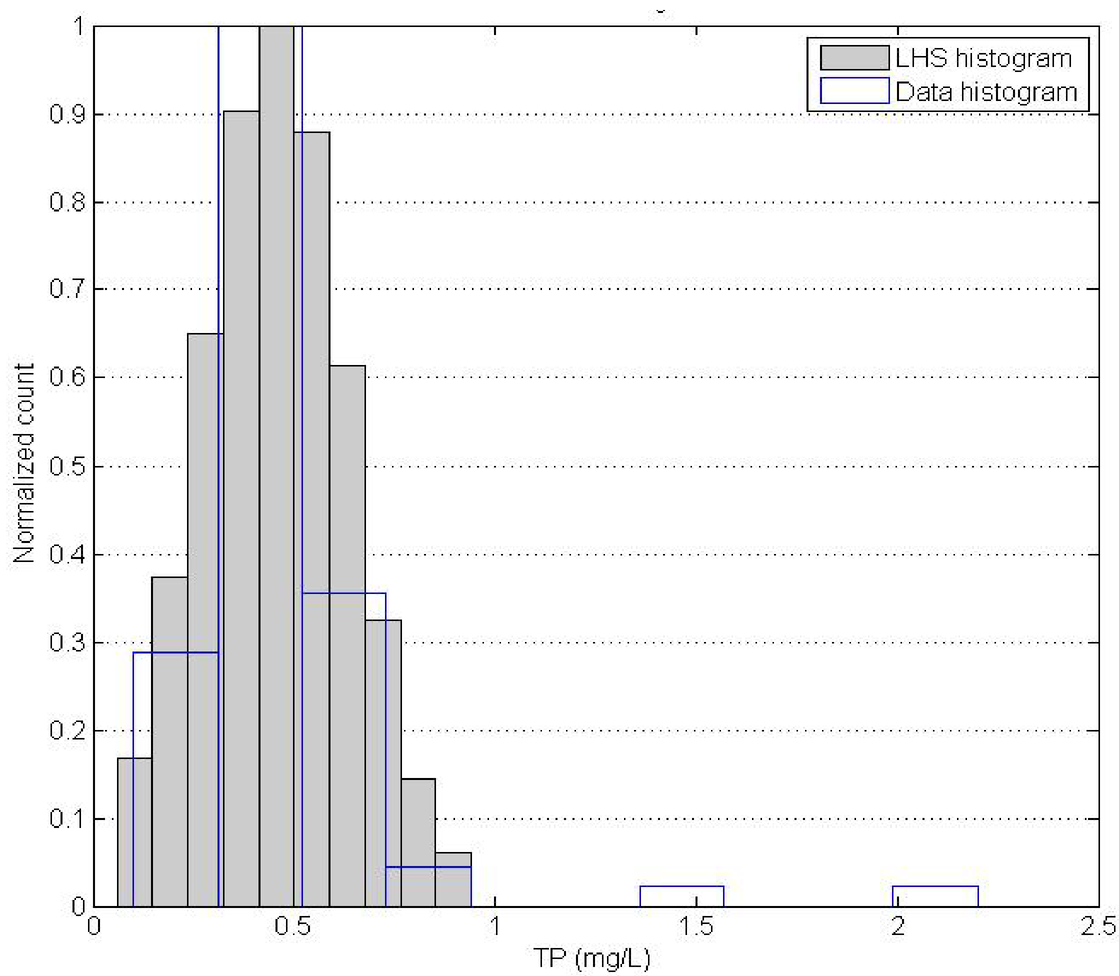


Figure 3-49: Comparison of histograms for Cluster 2 TP effluent from Berkeley Heights STP

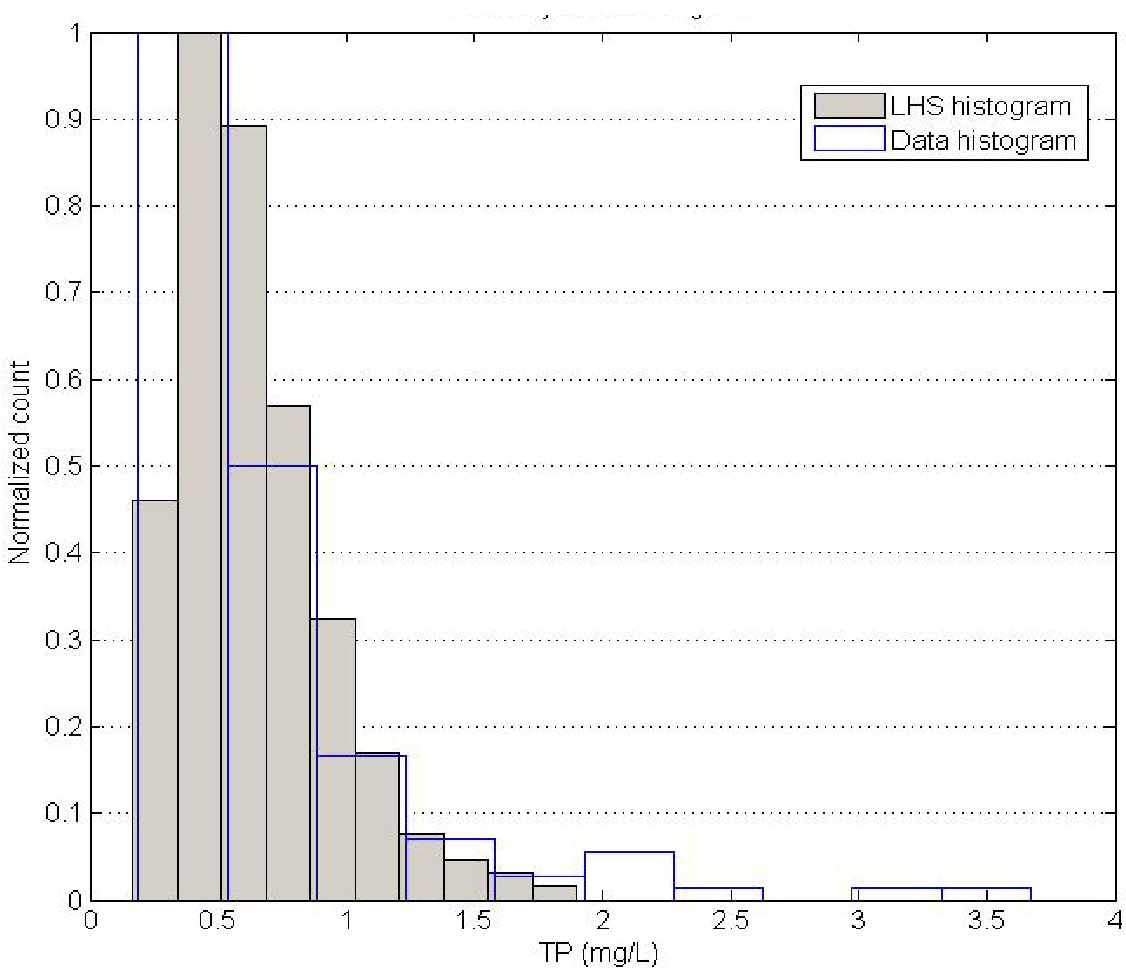


Figure 3-50: Comparison of histograms for Cluster 1 TP effluent from Rockaway Valley RSA

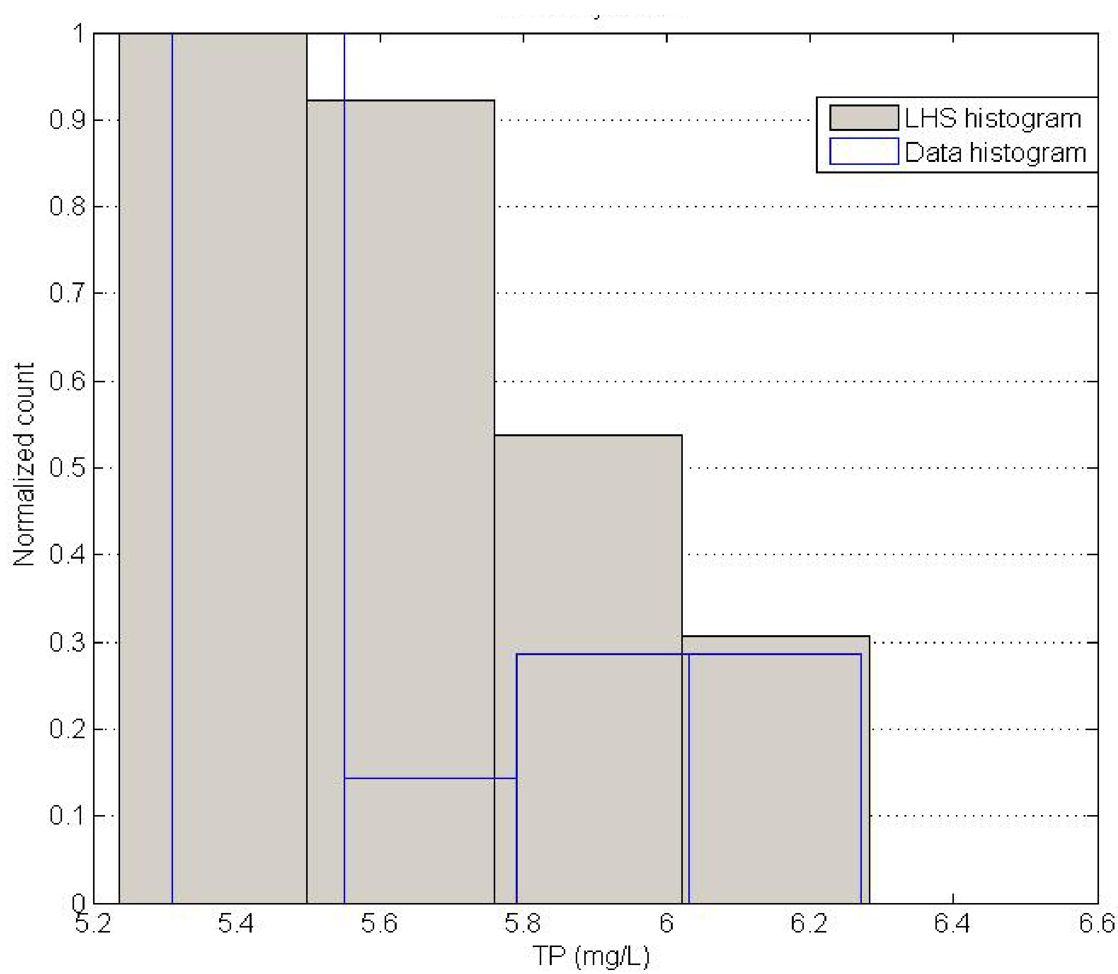


Figure 3-51: Comparison of histograms for Cluster 2 TP effluent from Rockaway Valley RSA

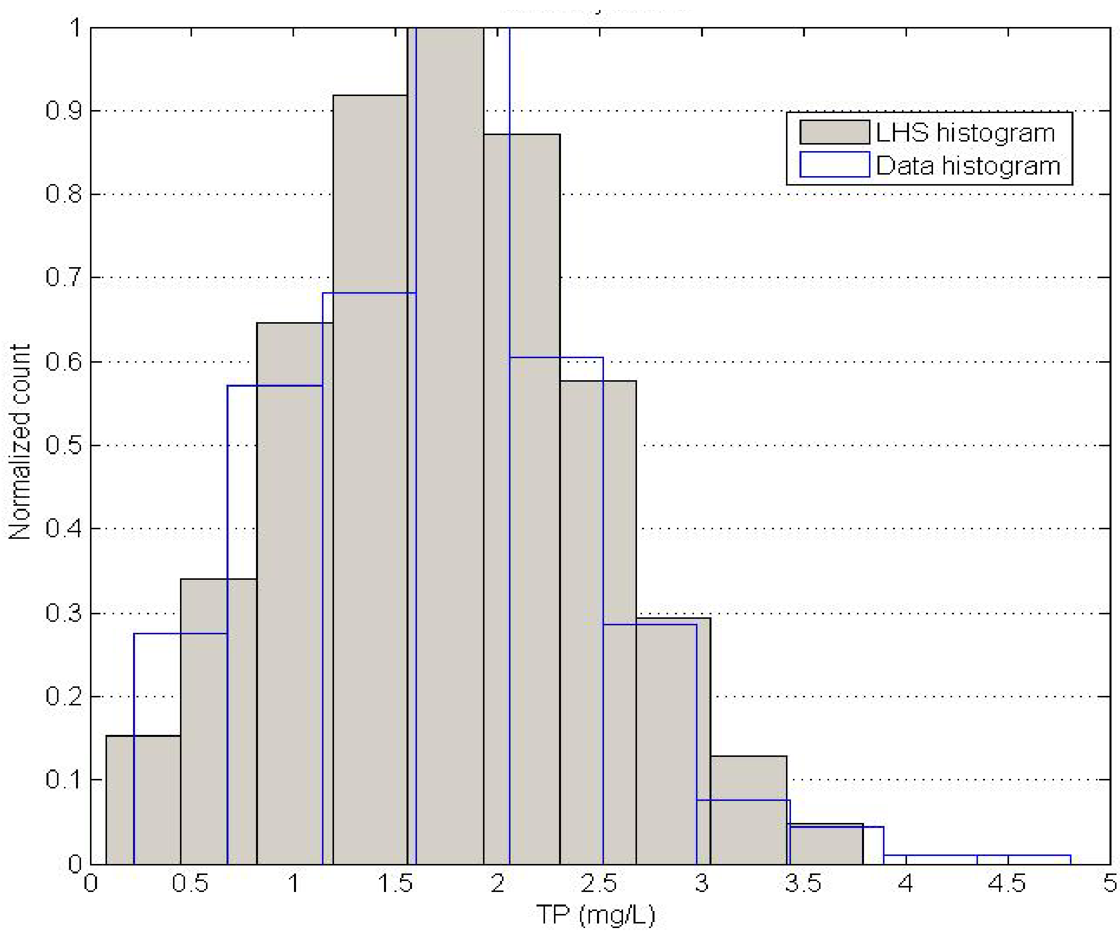


Figure 3-52: Comparison of histograms for Cluster 3 TP effluent from Rockaway Valley RSA

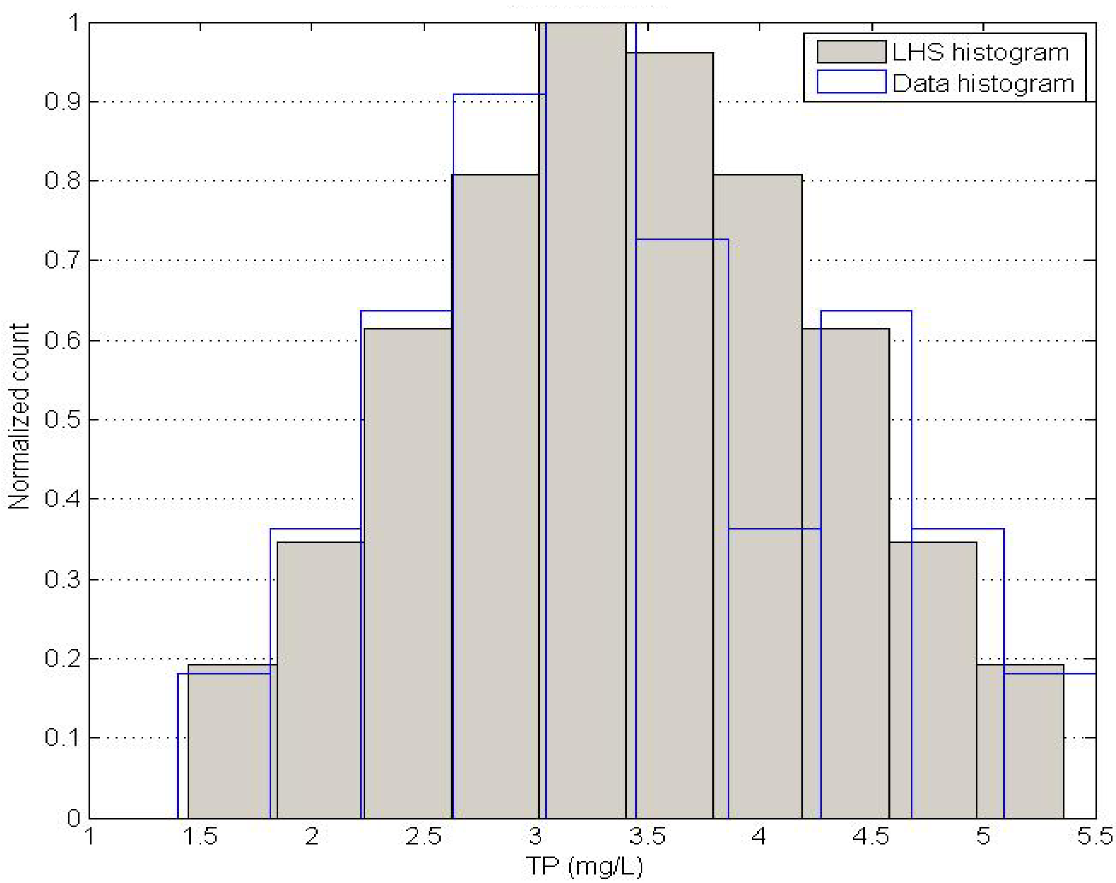


Figure 3-53: Comparison of histograms for Cluster 1 TP effluent from Two Bridges SA

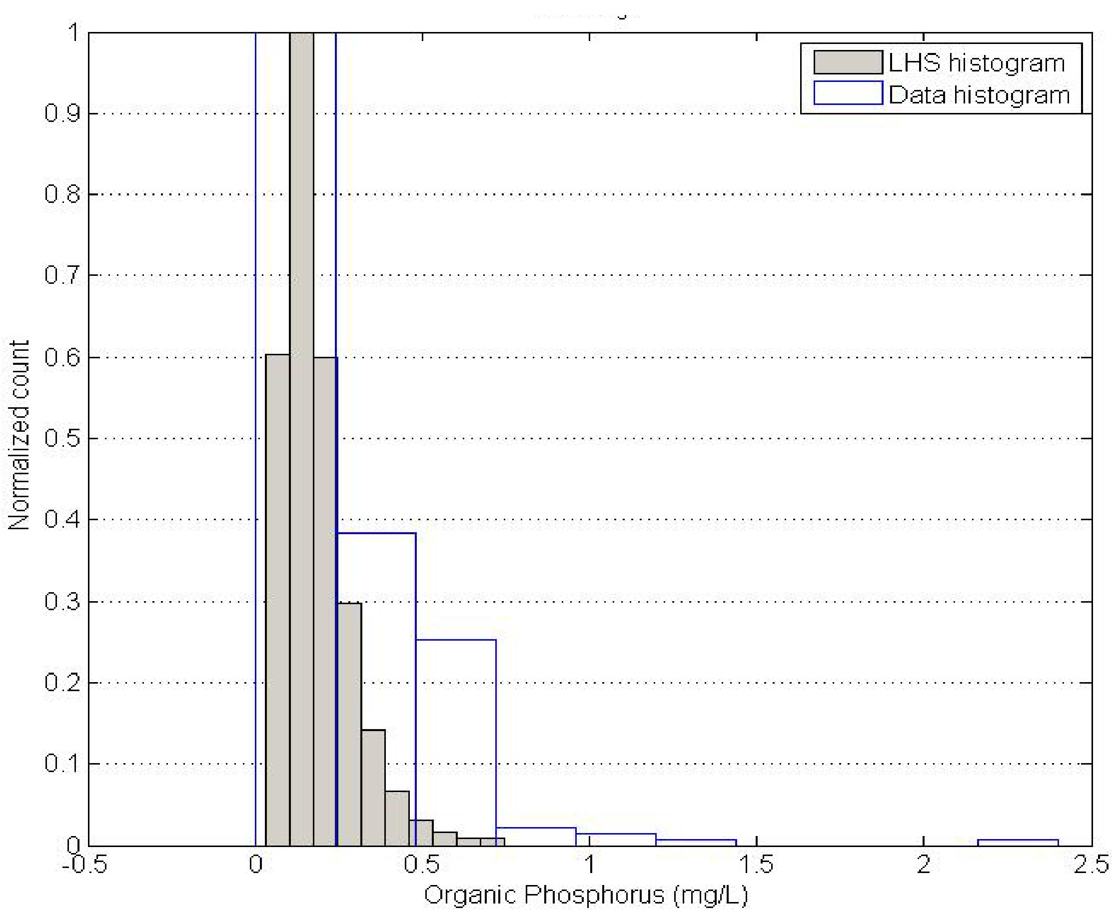


Figure 3-54: Comparison of histograms for Cluster 1 Organic phosphorus effluent from Two Bridges SA

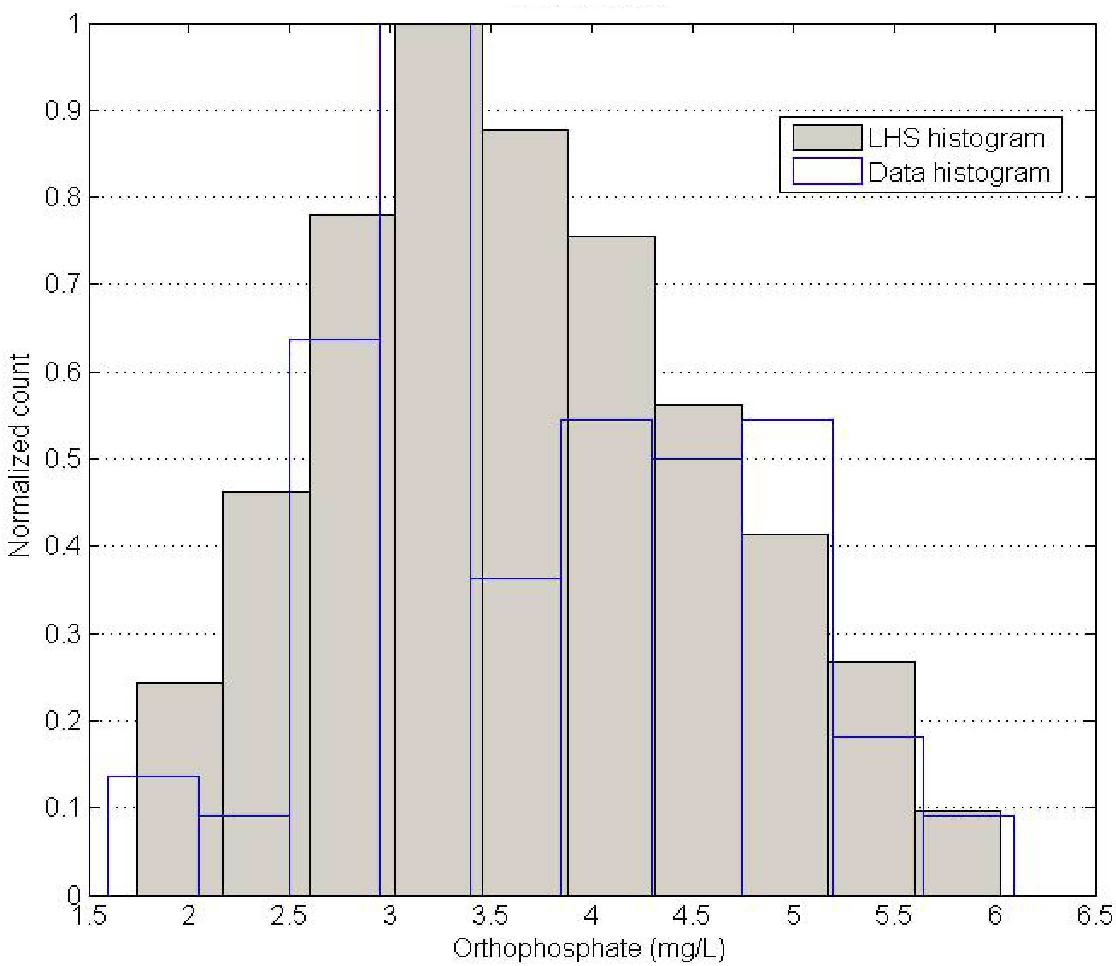


Figure 3-55: Comparison of histograms for Cluster 1 Orthophosphate effluent from Two Bridges SA

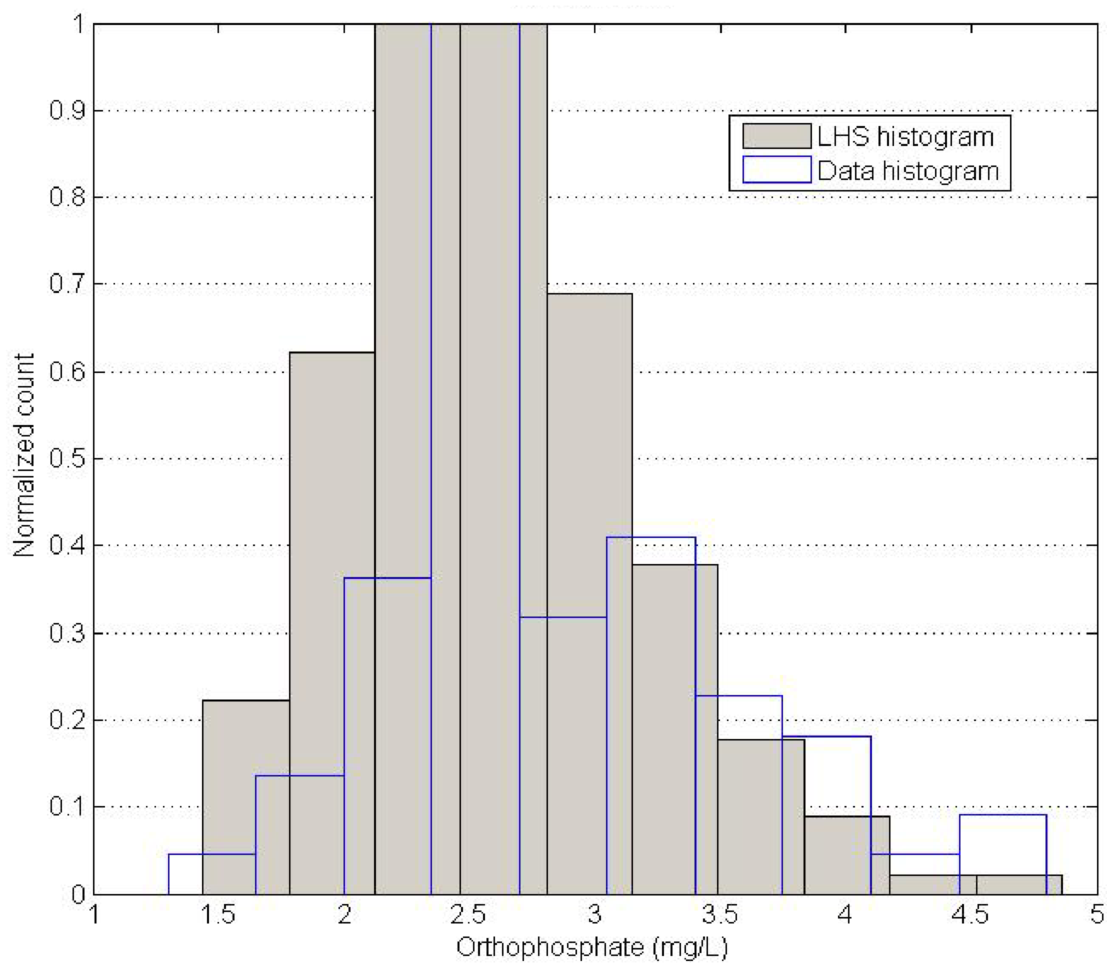


Figure 3-56: Comparison of histograms for Cluster 2 Orthophosphate effluent from Two Bridges SA

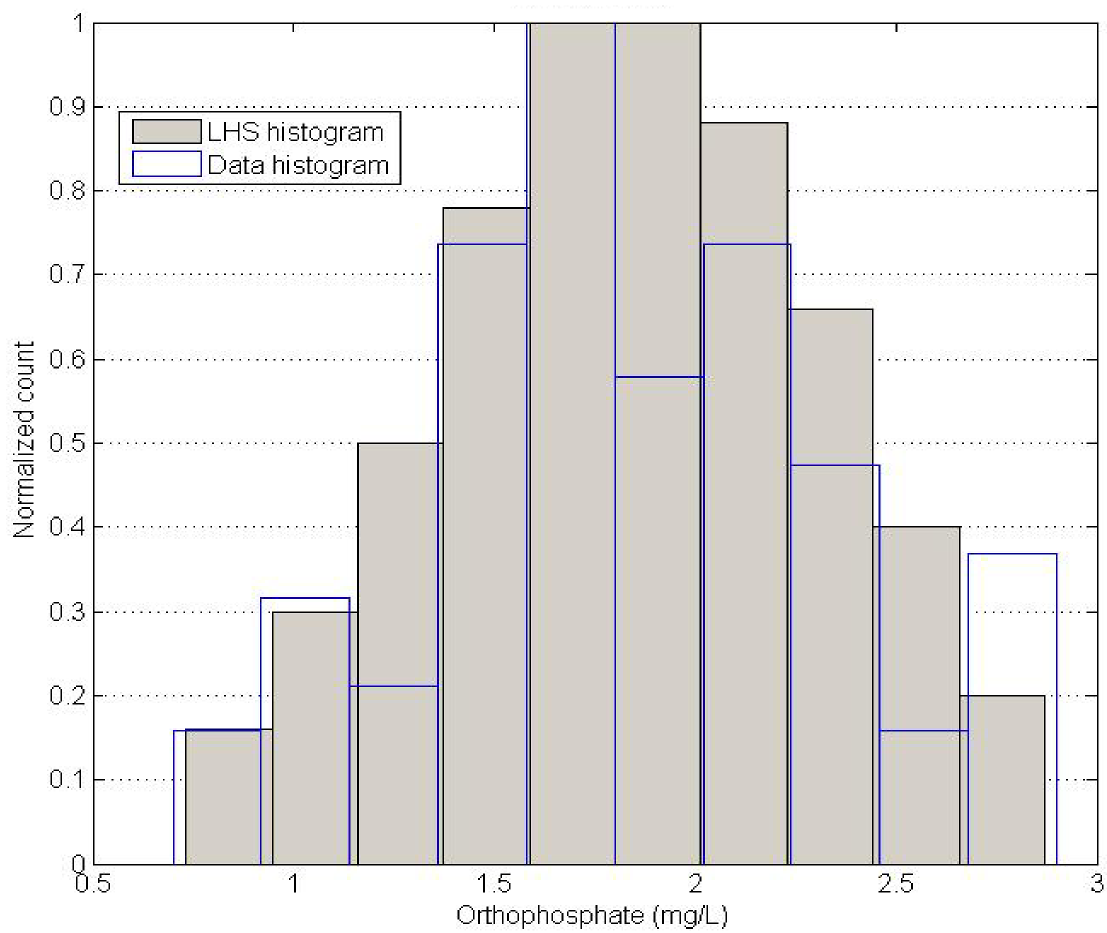


Figure 3-57: Comparison of histograms for Cluster 3 Orthophosphate effluent from Two Bridges SA

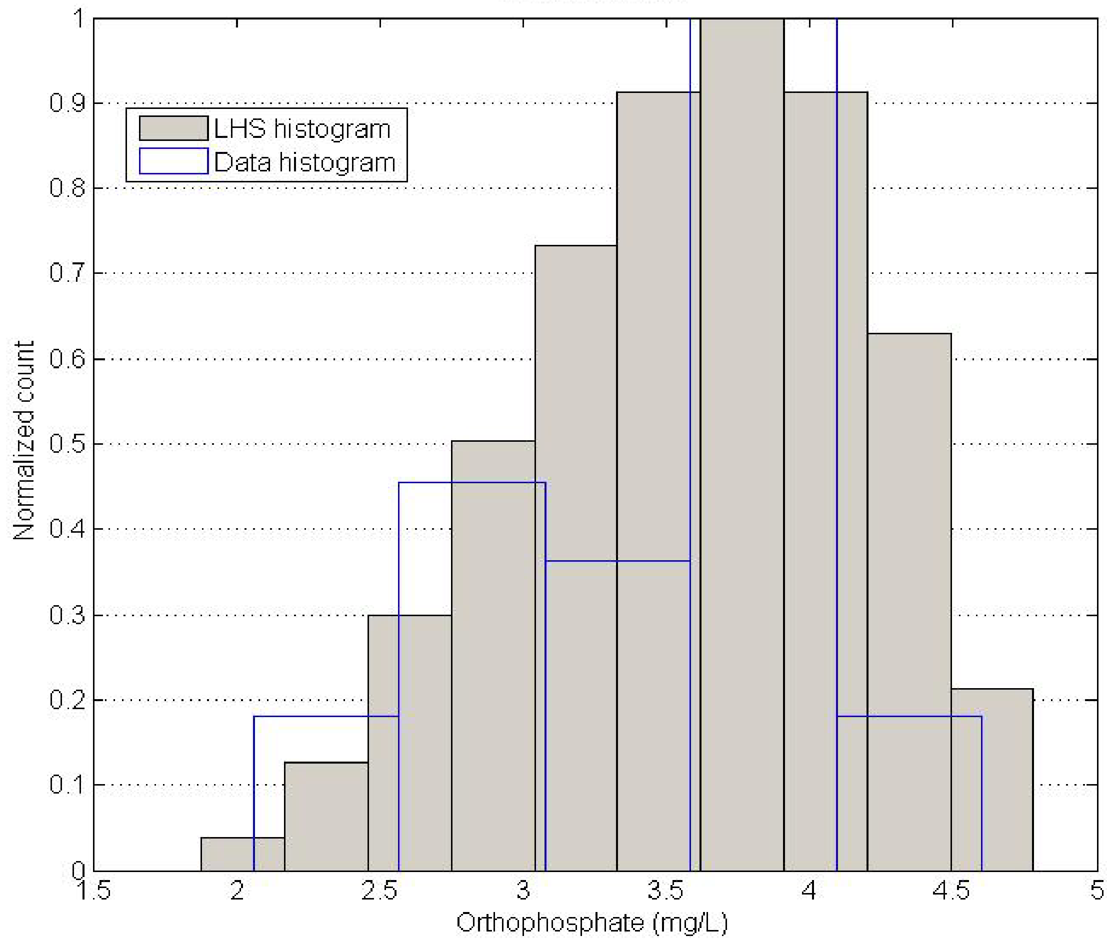


Figure 3-58: Comparison of histograms for Cluster 1 Orthophosphate effluent from Verona STP

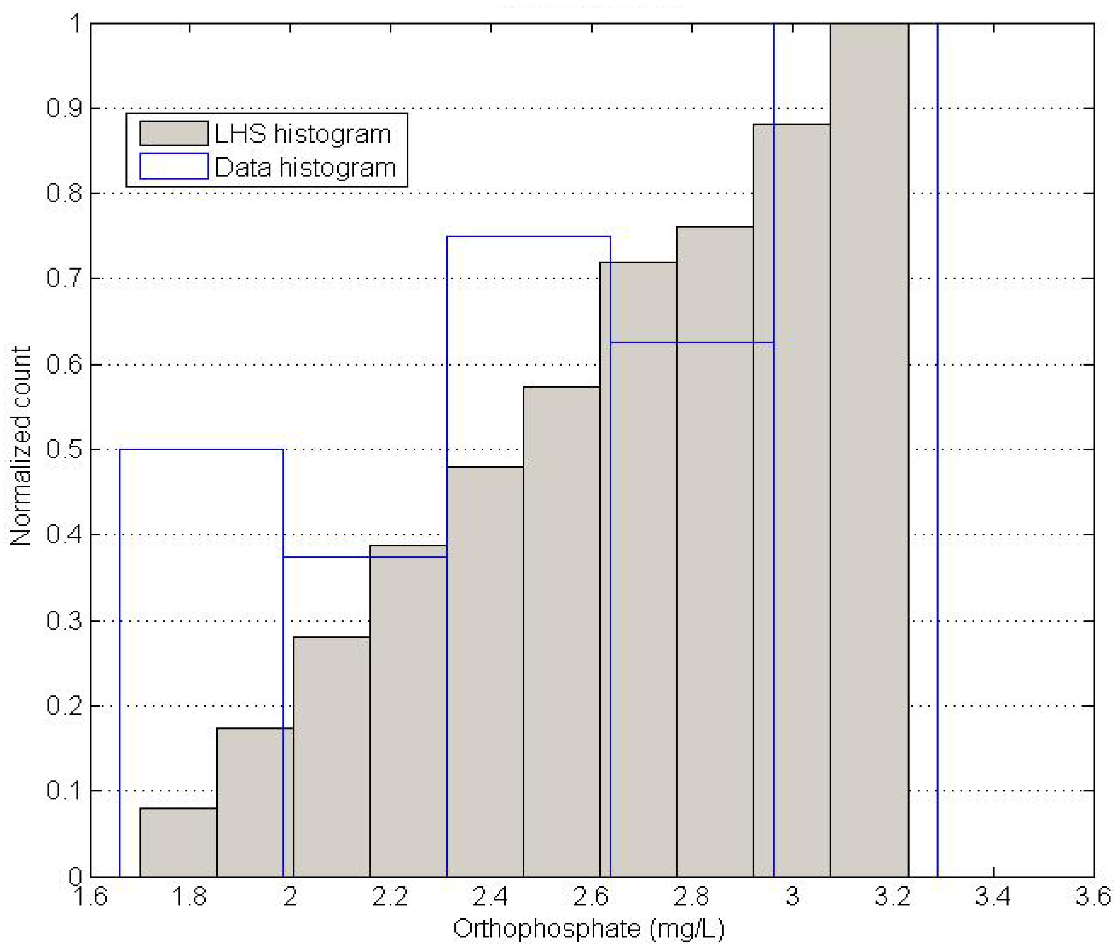


Figure 3-59: Comparison of histograms for Cluster 2 Orthophosphate effluent from Verona STP

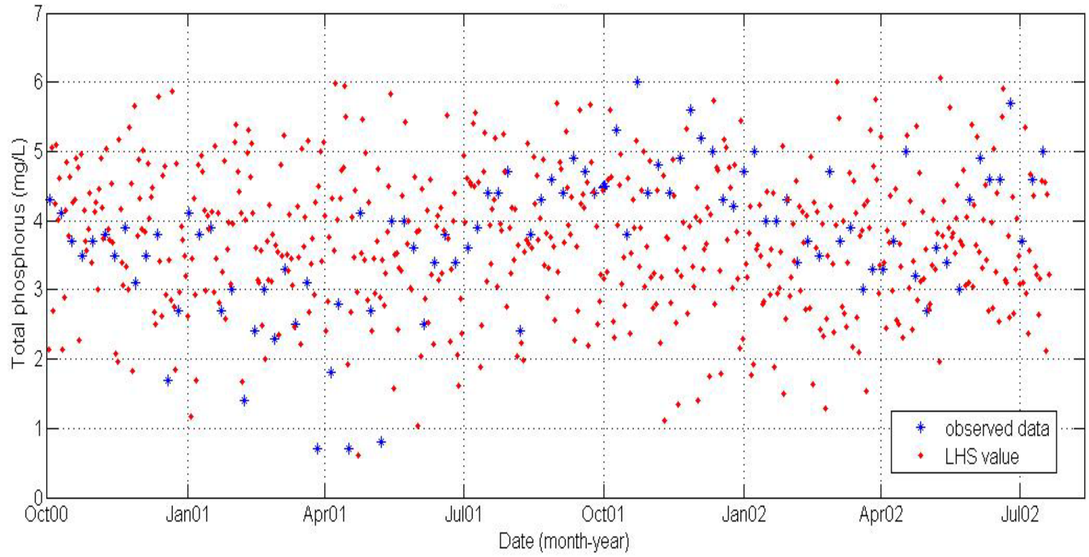


Figure 3-60: Observed and LHS-generated values for Cluster 1 TP effluent from Berkeley Heights STP

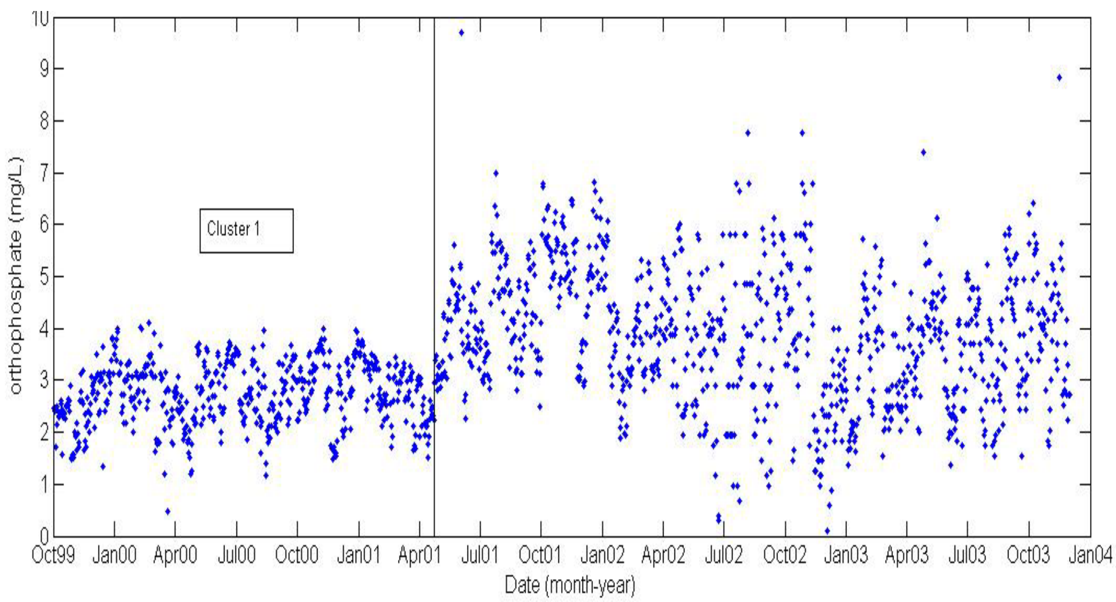


Figure 3-61: TMDL model-simulated orthophosphate effluent of Parsippany-Troy Hills RSA from 10/1/99 – 11/30/03 (adapted from data published in Omni Environmental, 2007a)

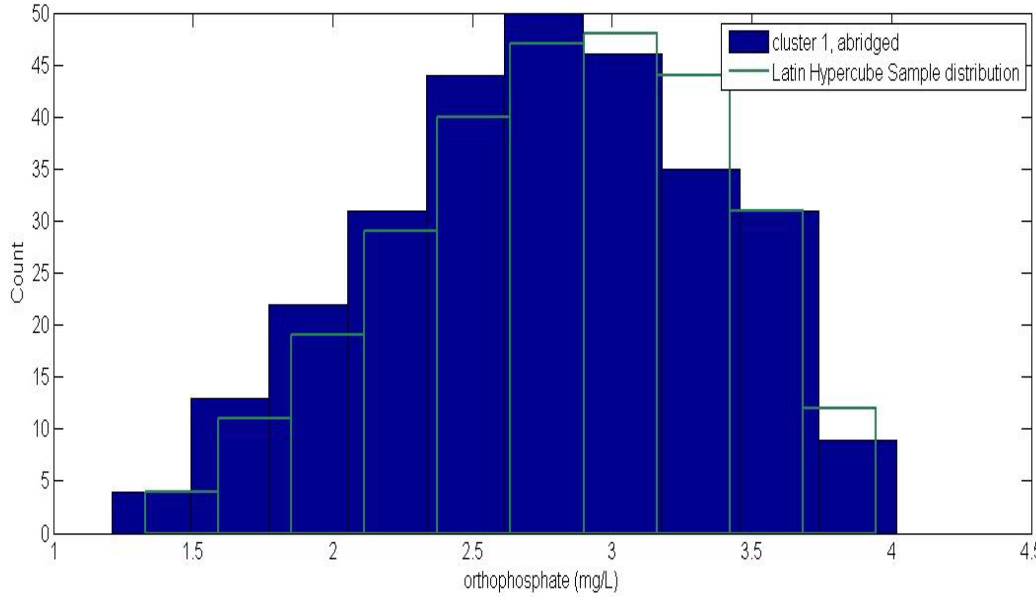


Figure 3-62: Comparison of histograms for Cluster 1 Orthophosphate effluent from Parsippany-Troy Hills RSA

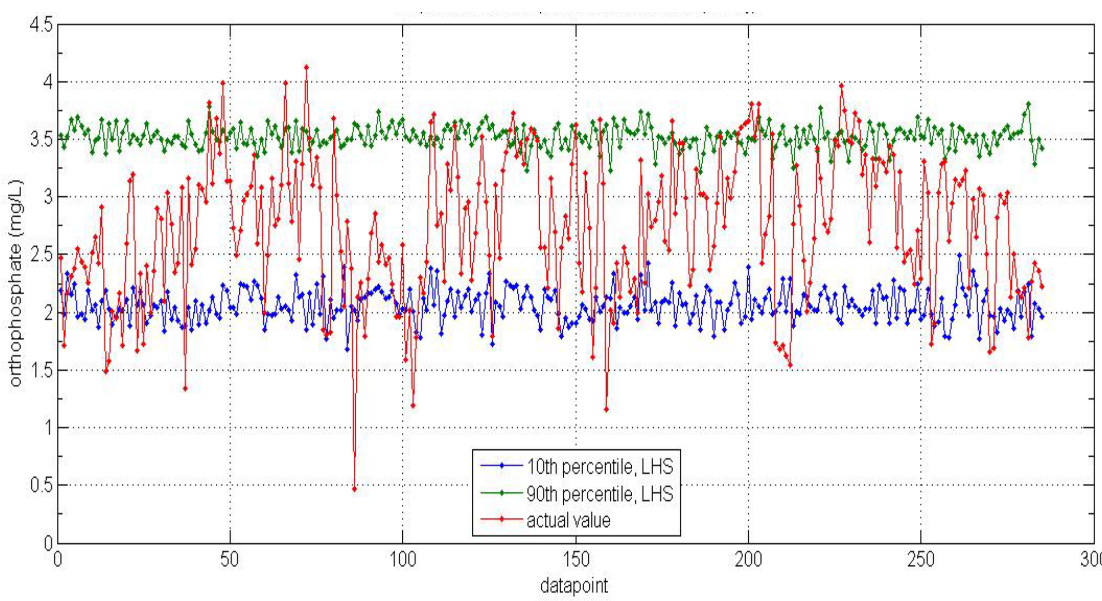


Figure 3-63: Comparison of actual values and predicted 80% confidence interval for Cluster 1 Orthophosphate effluent from Parsippany-Troy Hills RSA

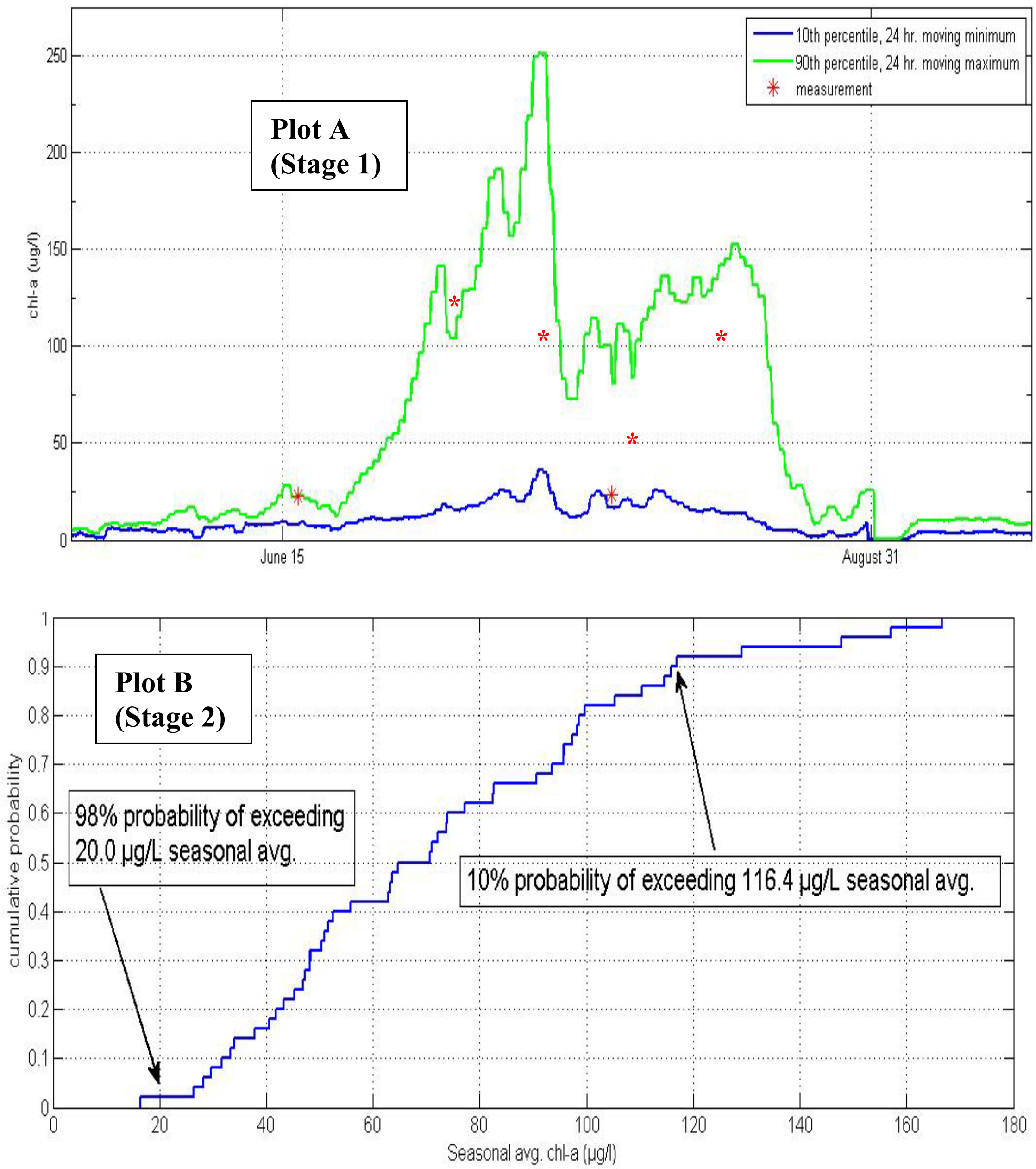


Figure 3-64: Example illustration of Stage 1 and Stage 2 chl-*a* plots and the relationships between the 90th percentiles

Plot A: Calculate the 80% confidence interval (at each timestep) for chl-*a* between June 15-September 1.

Plot B: Calculate the seasonal average (June 15-September 1) chl-*a* for each model run, then plot the cumulative distribution of all seasonal averages. The 90th percentile, i.e., 10% exceedance probability, of seasonal average chl-*a* in Plot B corresponds to the 90th percentile in Plot A. The credibility of the 90th percentile value in Plot B is based directly on the credibility of the 80% confidence interval in Plot A, which in turn is based on the proportion of measurements that fall inside the 80% confidence interval.

The process is the same for TP, except that Plots A and B are calculated for the entire water year, where Plot A is an 80% confidence interval of TP concentration at each timestep and Plot B is a cumulative distribution of diverted load over the water year. (The uncertainty of diverted flow is ignored in Stage 2, so that uncertainty of diverted load is strictly a function of uncertainty in TP concentration).

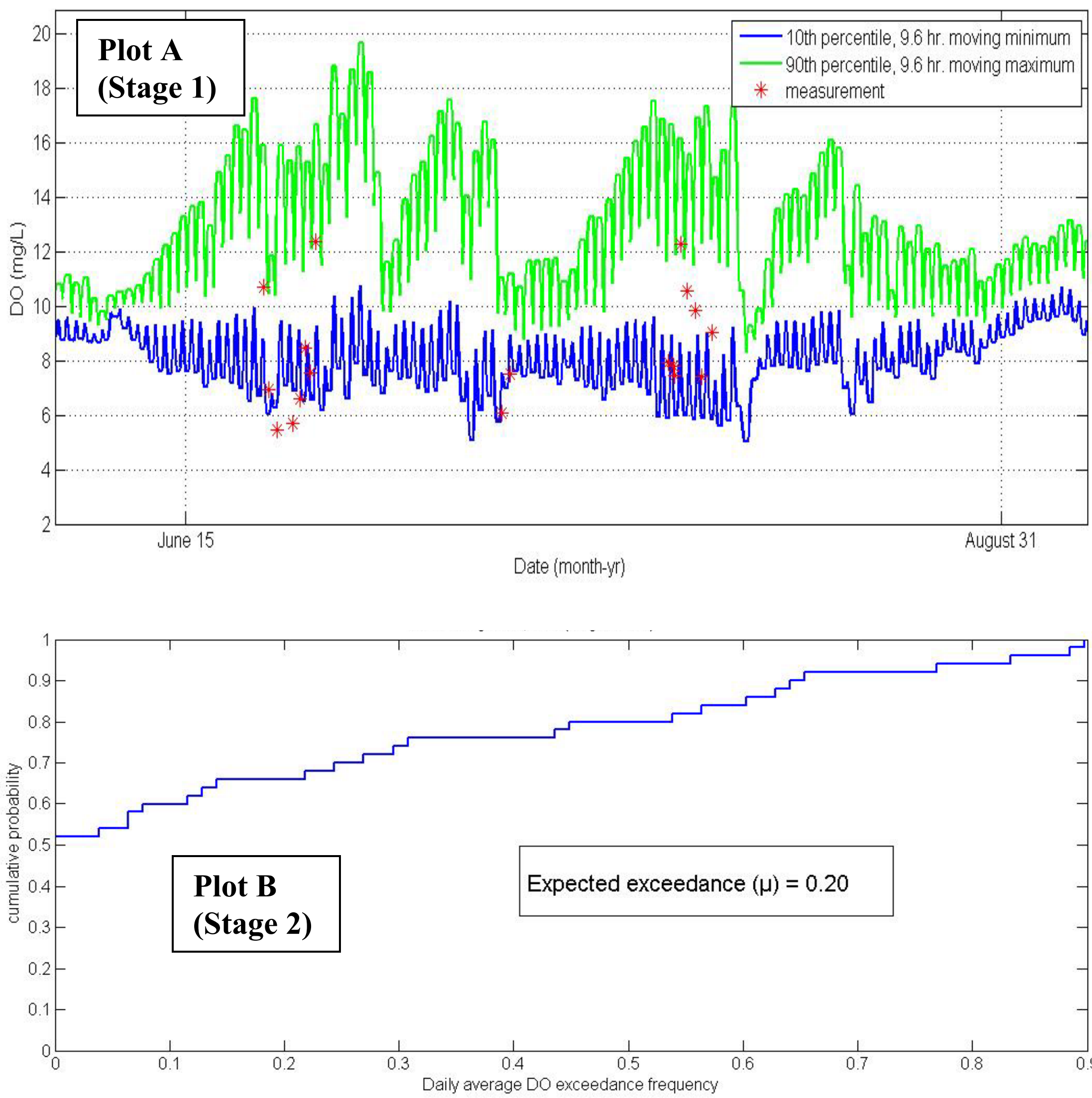


Figure 3-65: Example illustration of Stage 1 and Stage 2 DO plots

Plot A: Calculate the 80% confidence interval (at each timestep) for DO between June 15-September 1.

Plot B: For the period of June 15-September 1, plot the cumulative distribution of daily average DO exceedance frequencies, where each model run has an exceedance frequency equal to the number of days the daily average standard was exceeded divided by the total number of days from June 15- September 1. The expected exceedance is the mean of the distribution. The credibility of the expected exceedance and confidence of compliance in Plot B is based indirectly on the credibility of the 80% confidence interval in Plot A, which in turn is based on the proportion of measurements that fall inside the 80% confidence interval.

The process is the same for plotting the minimum DO standard exceedance frequencies, where each model run has an exceedance frequency = (number of days the minimum standard was exceeded) / (number of days from June 15- September 1).

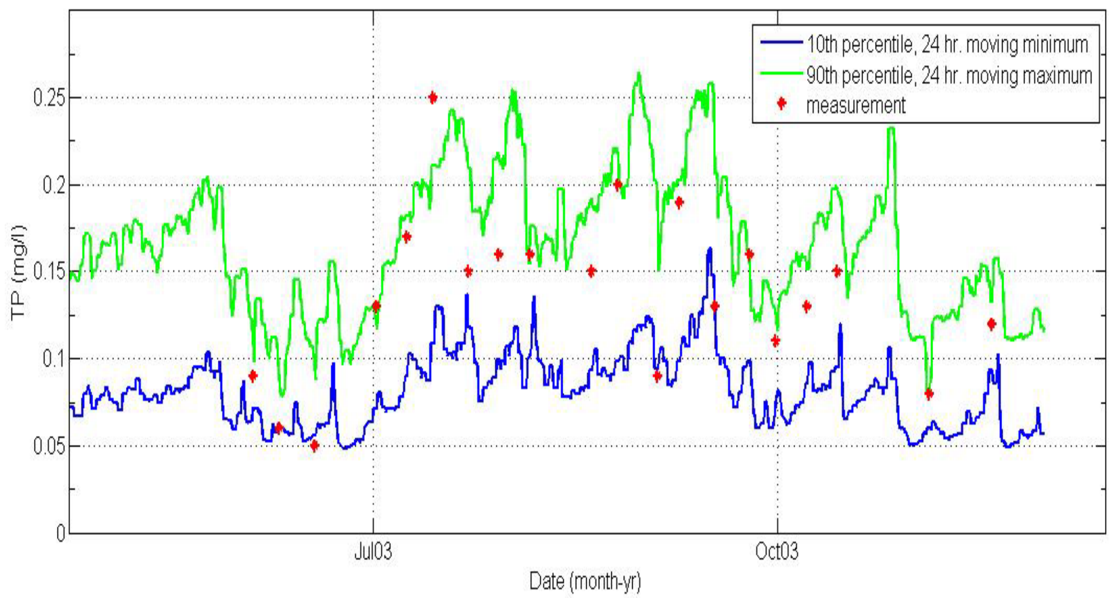


Figure 3-66: Predicted 80% confidence interval of TP at the Wanaque South intake and observed data

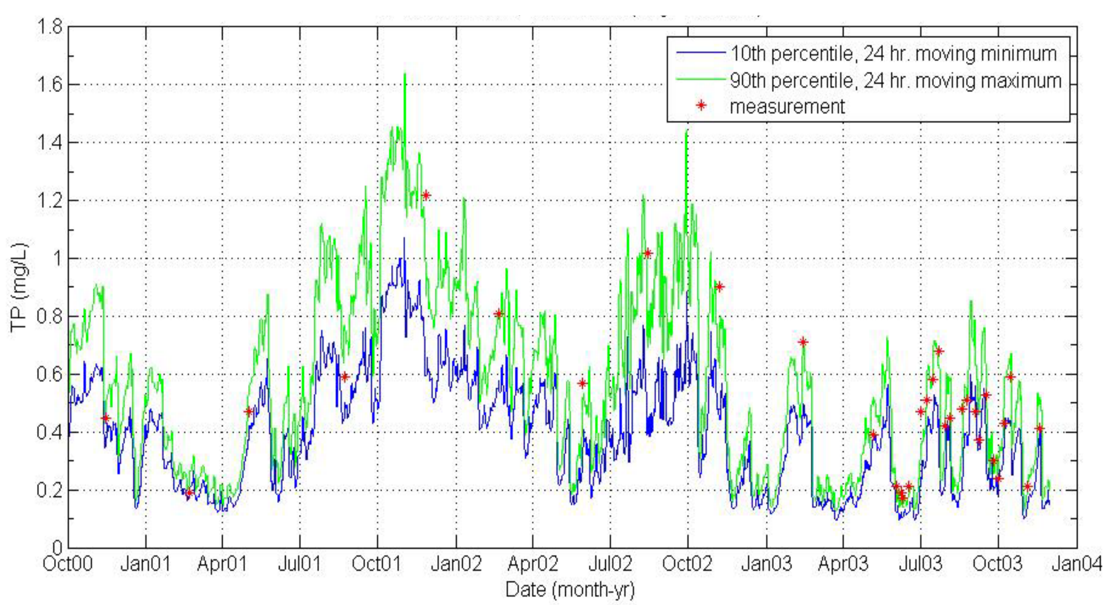


Figure 3-67a: Predicted 80% confidence interval of TP at B12-N38 and observed data; model output processed at 24 hour moving time scale.

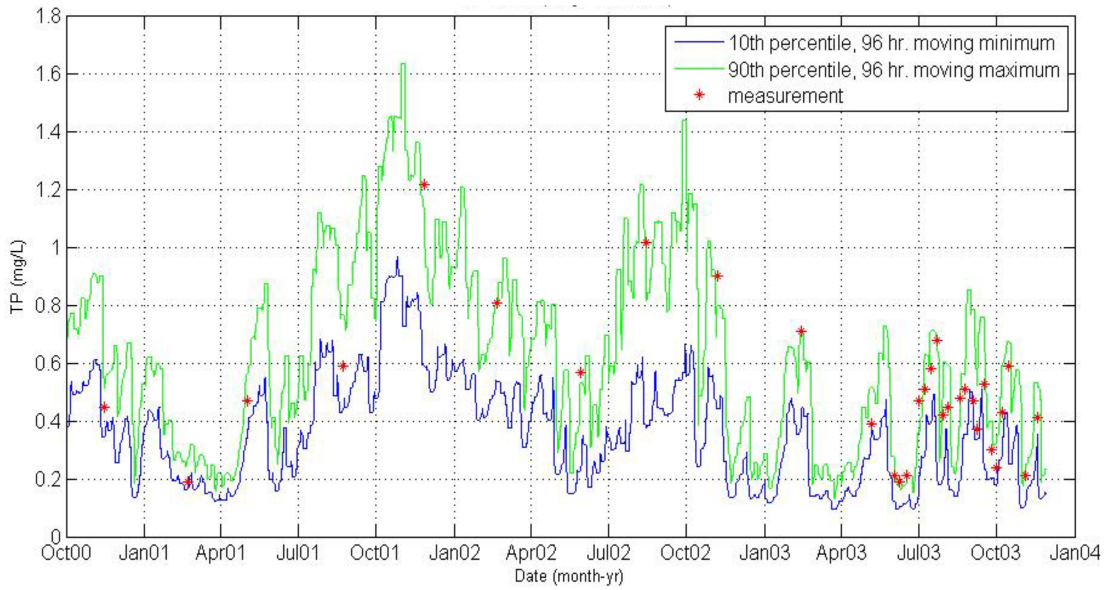


Figure 3-67b: Predicted 80% confidence interval of TP at B12-N38 and observed data; model output processed at 96 hour moving time scale.

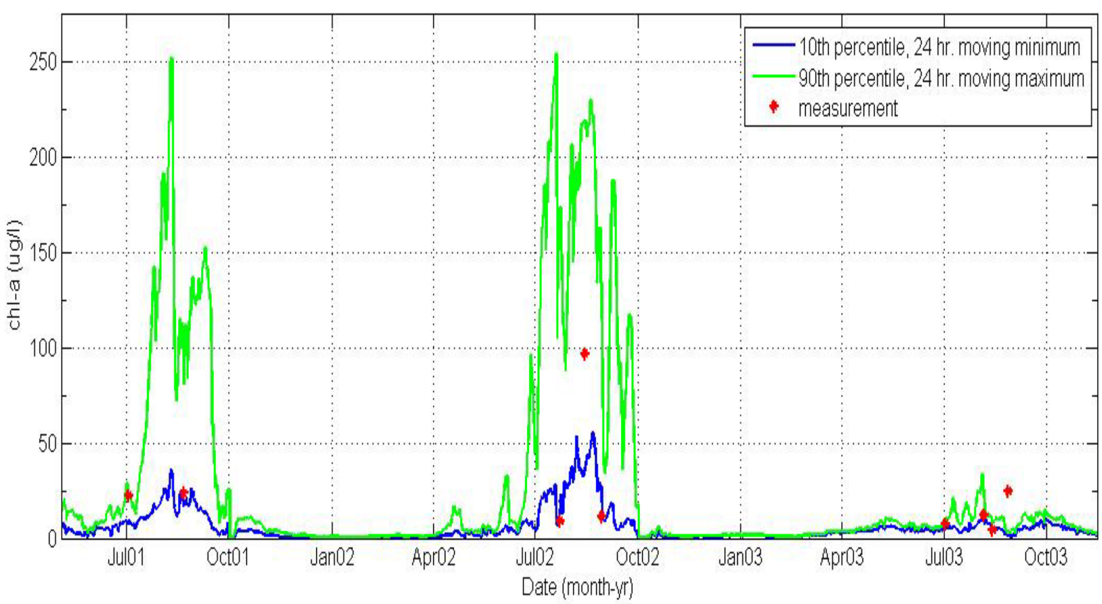


Figure 3-68: Predicted 80% confidence interval of chl-*a* at B17-N25 and observed data

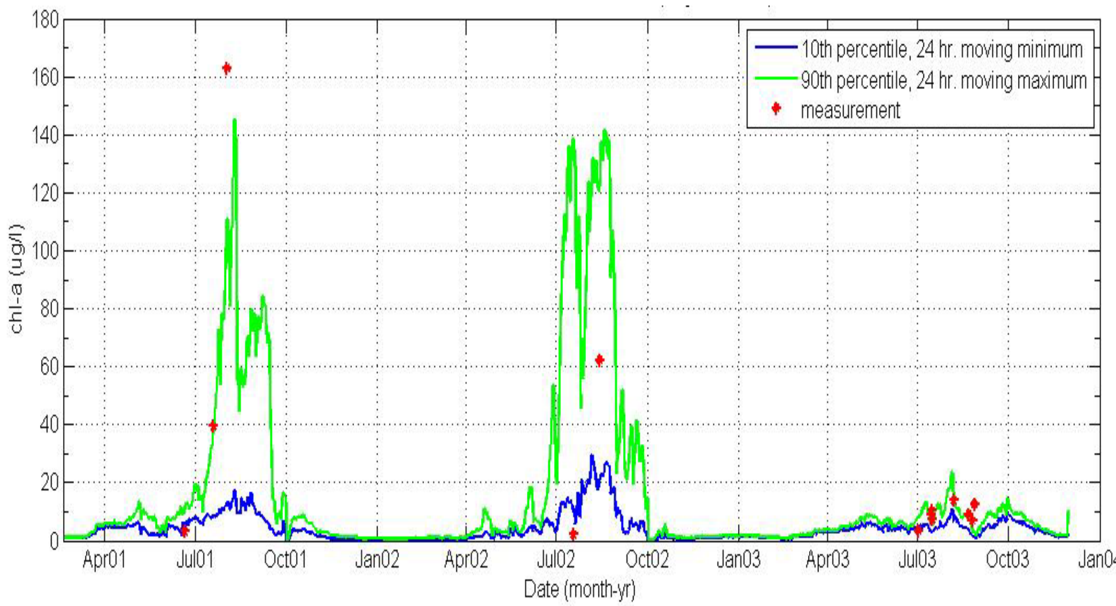


Figure 3-69: Predicted 80% confidence interval of chl-*a* at B17-N4 and observed data

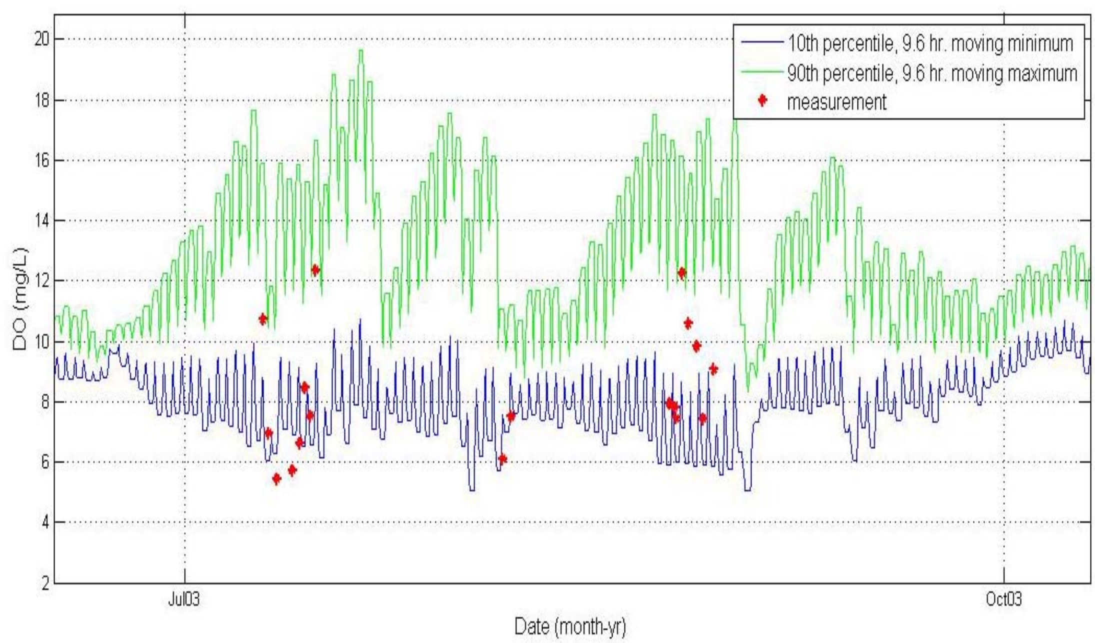


Figure 3-70: Predicted 80% confidence interval of DO at Dundee Lake and observed data

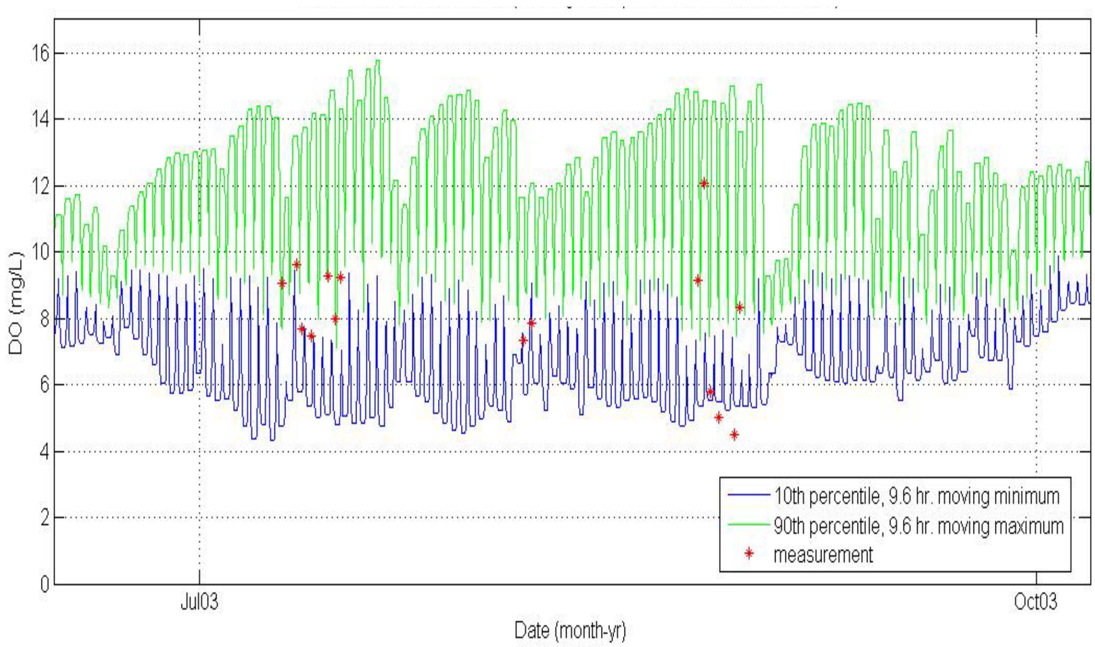


Figure 3-71: Predicted 80% confidence interval of DO at Peckman River mouth and observed data

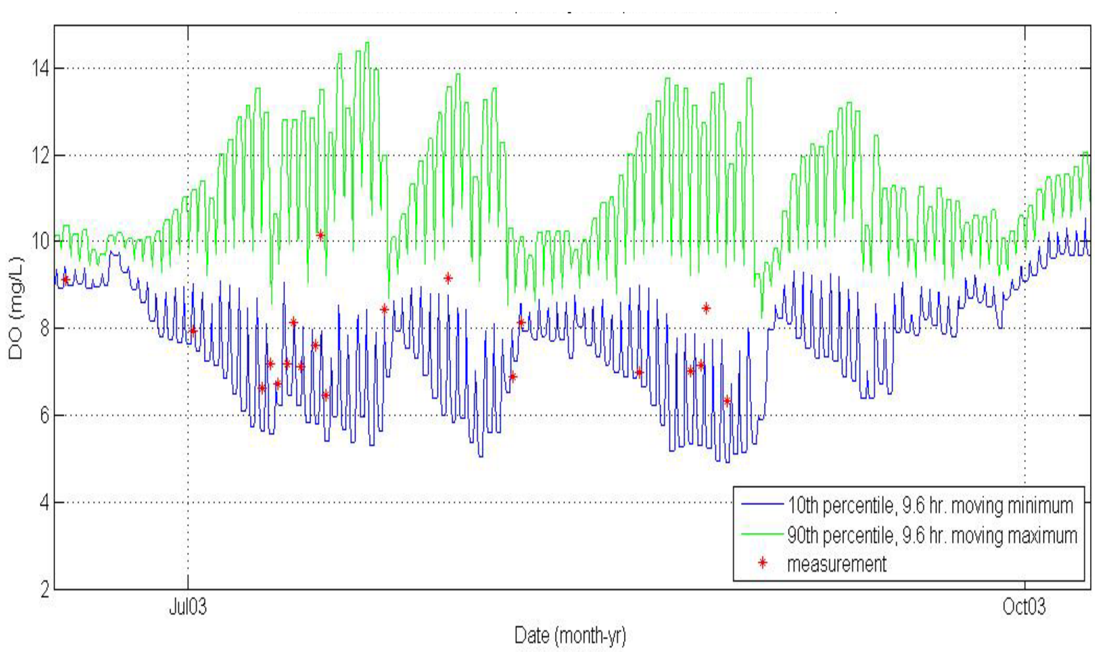


Figure 3-72: Predicted 80% confidence interval of DO at station PA10 and observed data

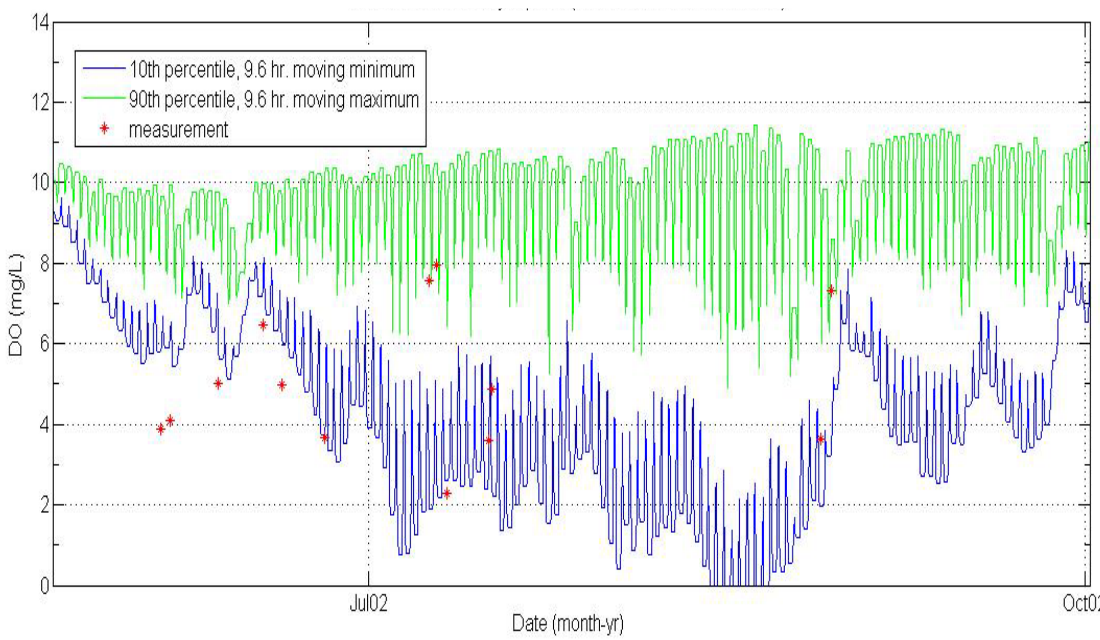
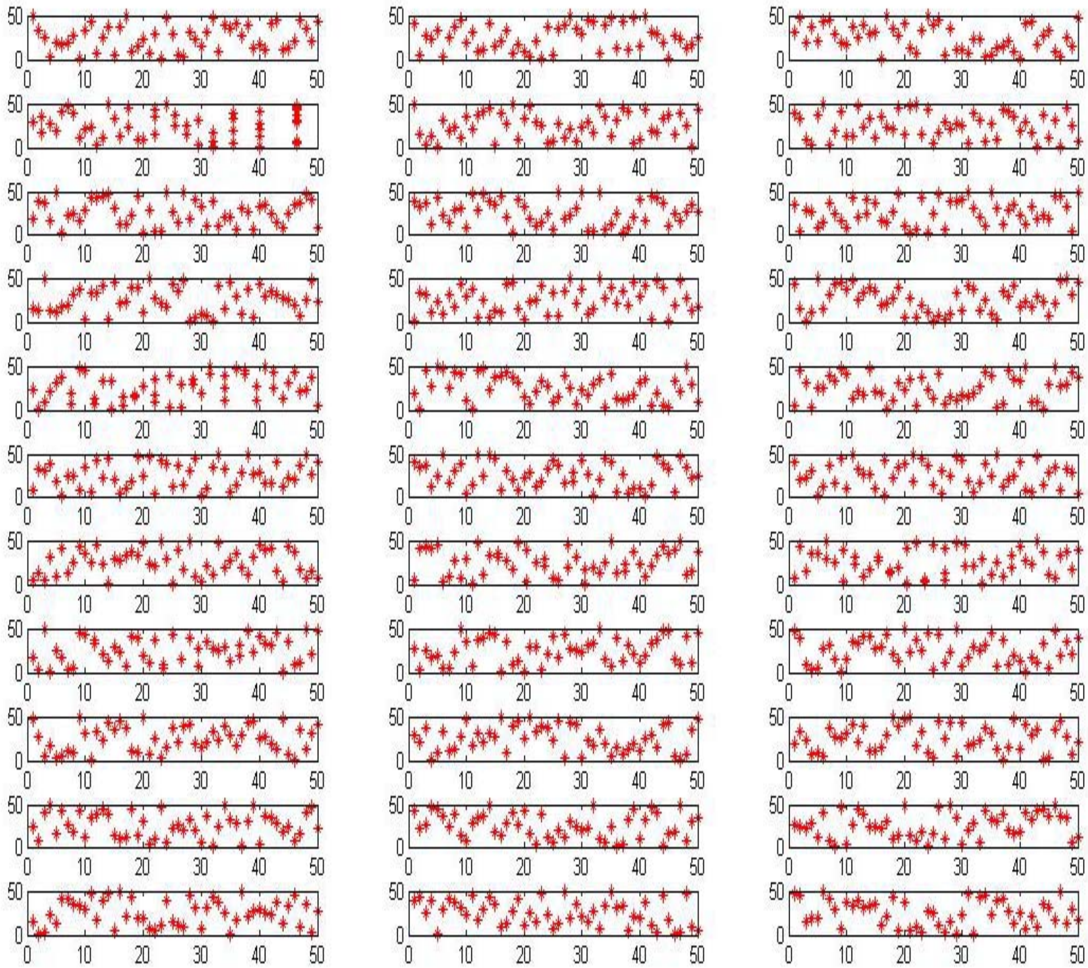


Figure 3-73: Predicted 80% confidence interval of DO at Passaic River near Chatham and observed data



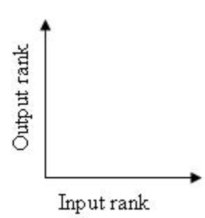


Figure 3-74: Subplots of GSA results of rank-transformed average TP at Wanaque South intake from October 2000 through April 2001. See Table 3-22 to identify kinetic parameters shown here (subplots arranged in same pattern as corresponding table entries). Significant parameters with r ≥ |0.5| are circled.

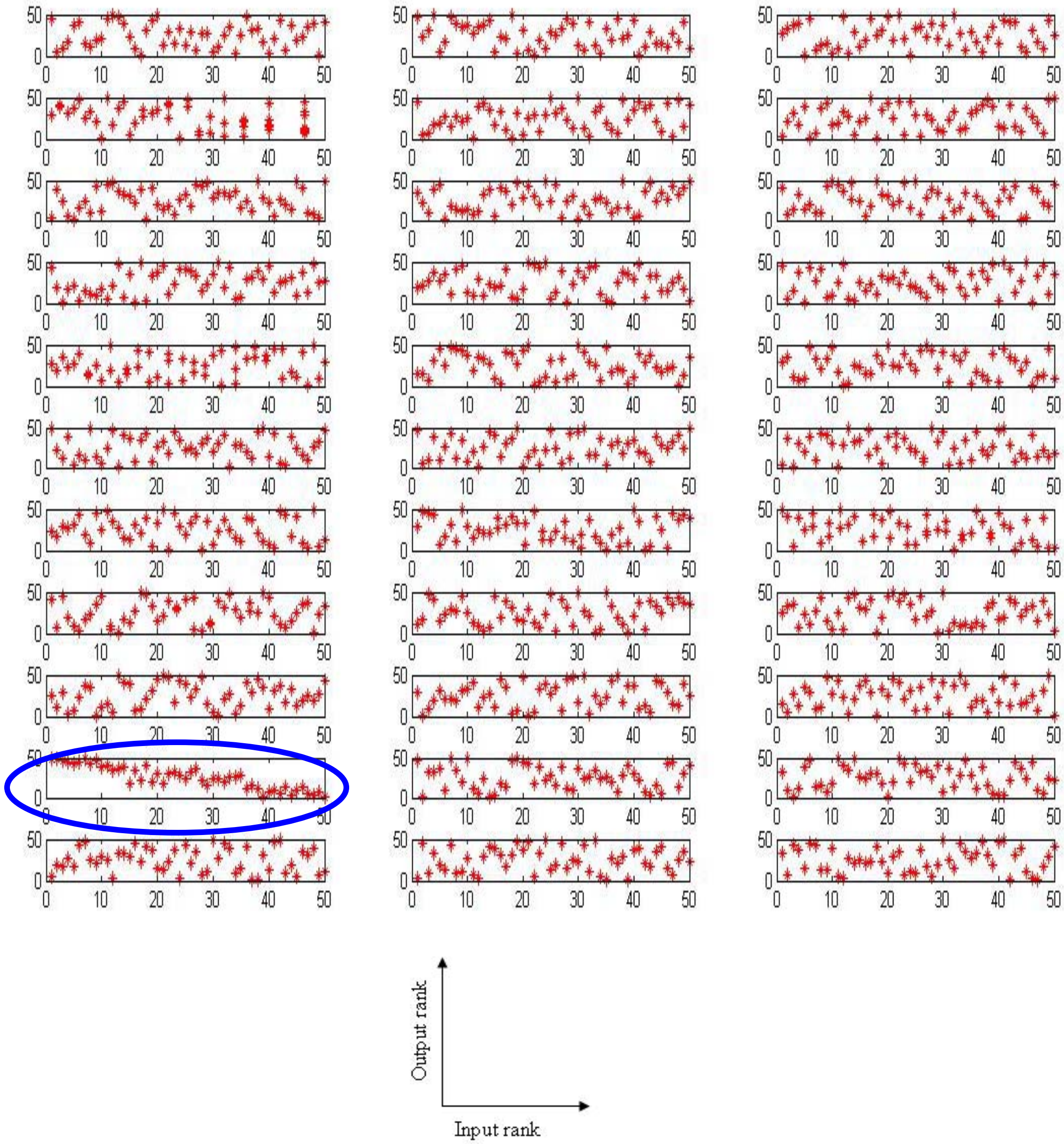


Figure 3-75: Subplots of GSA results of rank-transformed average TP at B12-N38 from October 2000 through April 2001. See Table 3-22 to identify kinetic parameters shown here (subplots arranged in same pattern as corresponding table entries). Significant parameters with r ≥ |0.5| are circled.

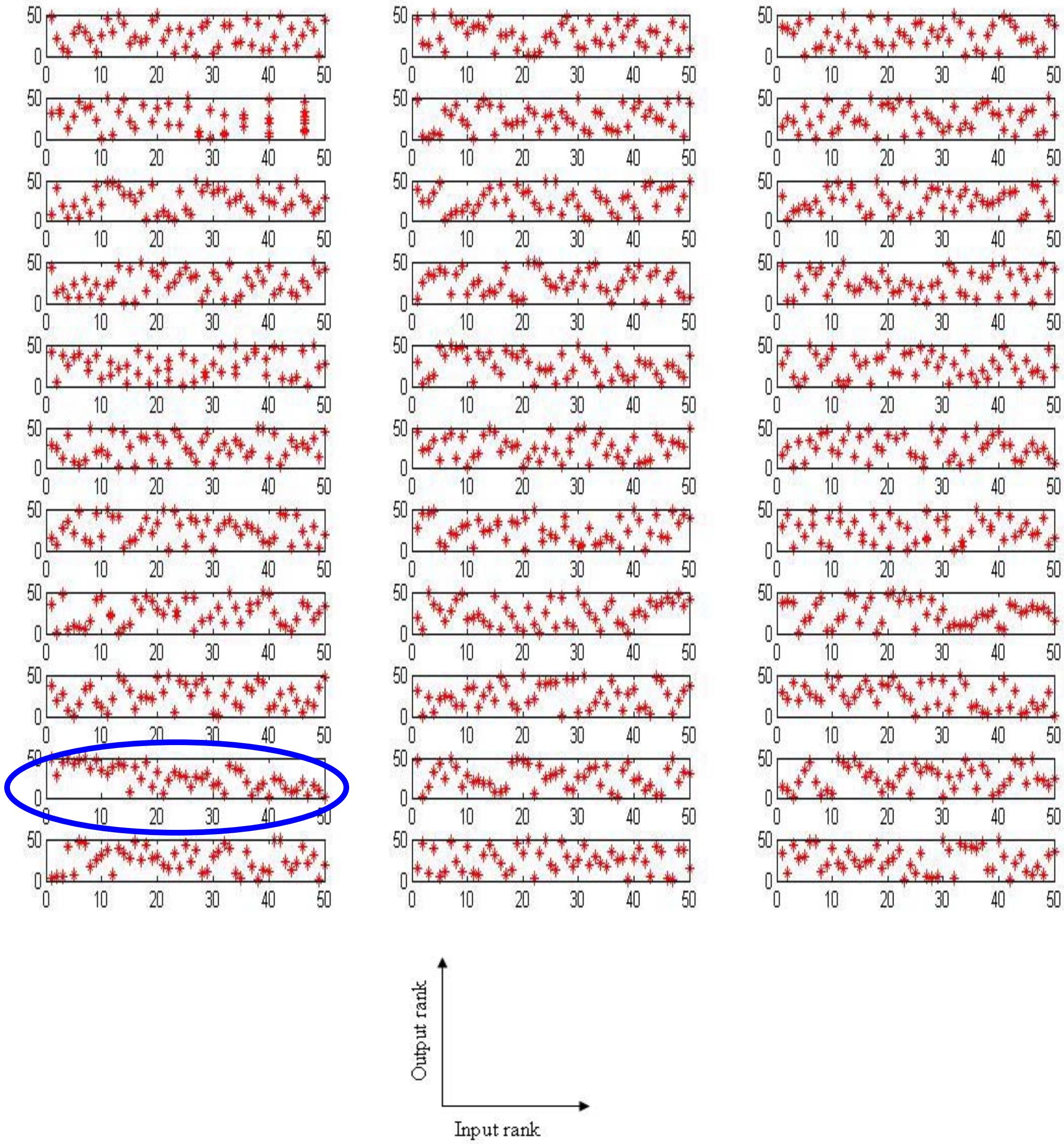


Figure 3-76: Subplots of GSA results of rank-transformed average TP at Little Falls intake from October 2000 through April 2001. See Table 3-22 to identify kinetic parameters shown here (subplots arranged in same pattern as corresponding table entries). Significant parameters with r ≥ |0.5| are circled.

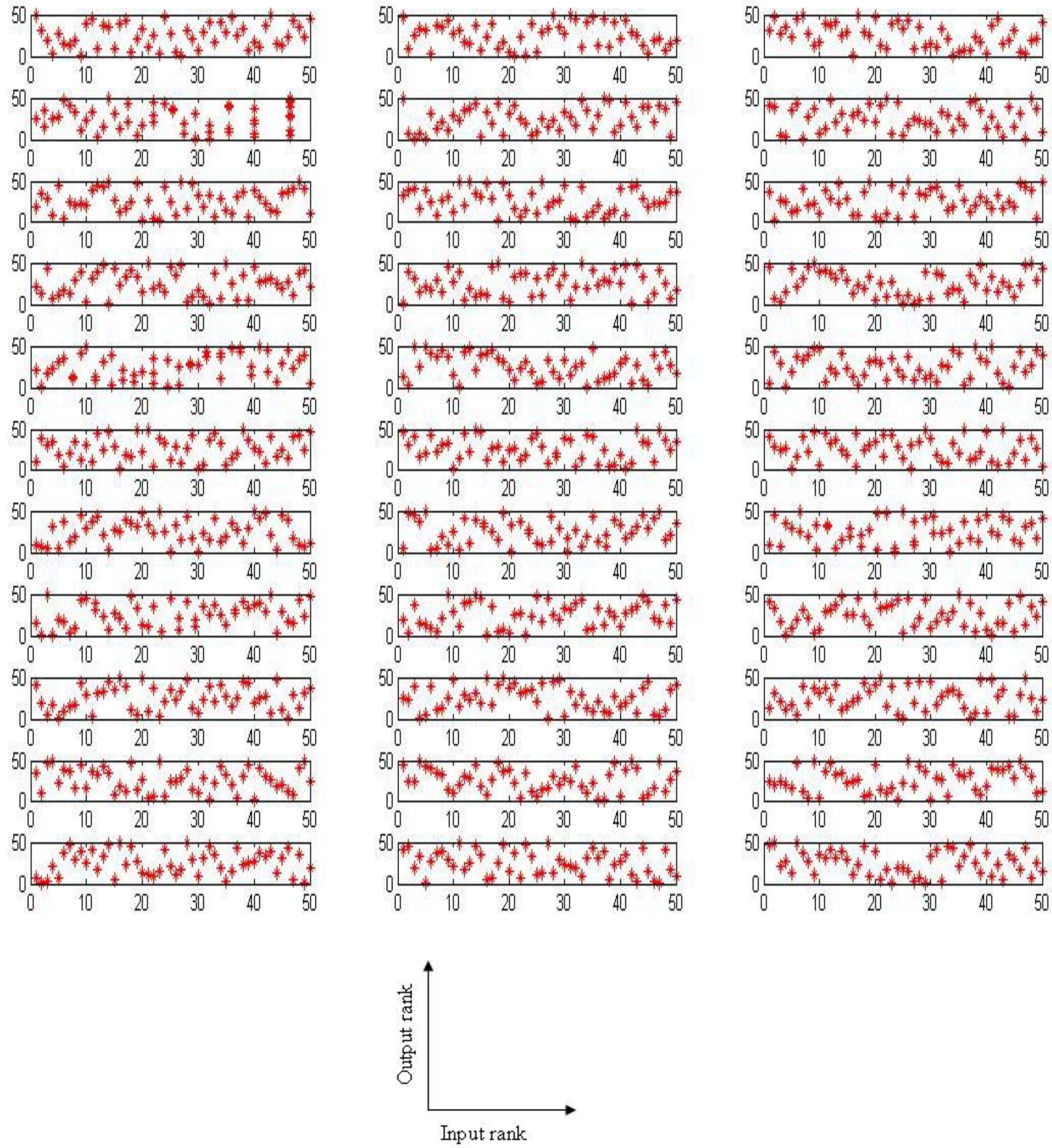


Figure 3-77: Subplots of GSA results of rank-transformed average TP at Wanaque South intake from May through September 2001. See Table 3-22 to identify kinetic parameters shown here (subplots arranged in same pattern as corresponding table entries). Significant parameters with r ≥ |0.5| are circled.

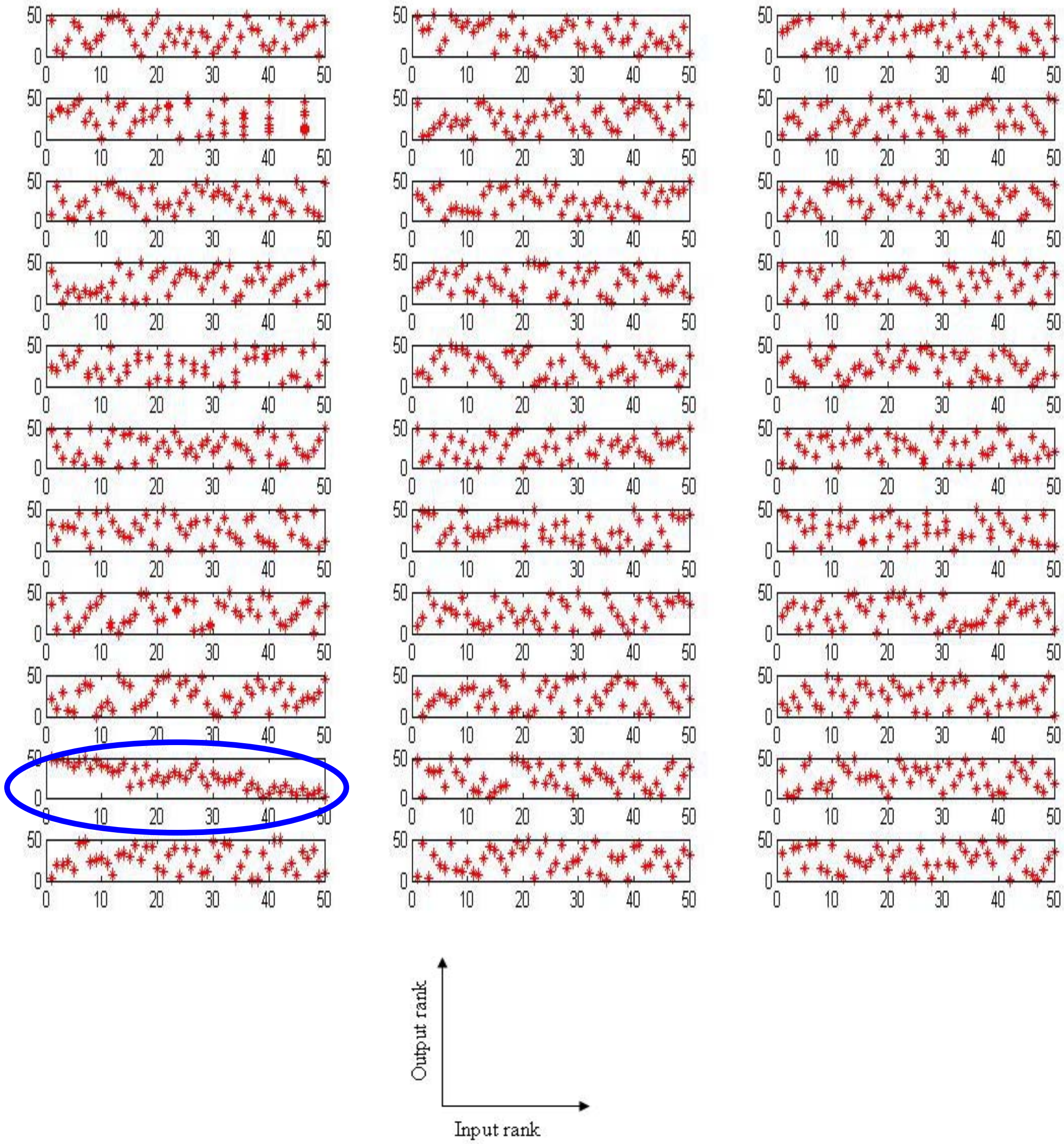


Figure 3-78: Subplots of GSA results of rank-transformed average TP at B12-N38 from May through September 2001. See Table 3-22 to identify kinetic parameters shown here (subplots arranged in same pattern as corresponding table entries). Significant parameters with r ≥ |0.5| are circled.

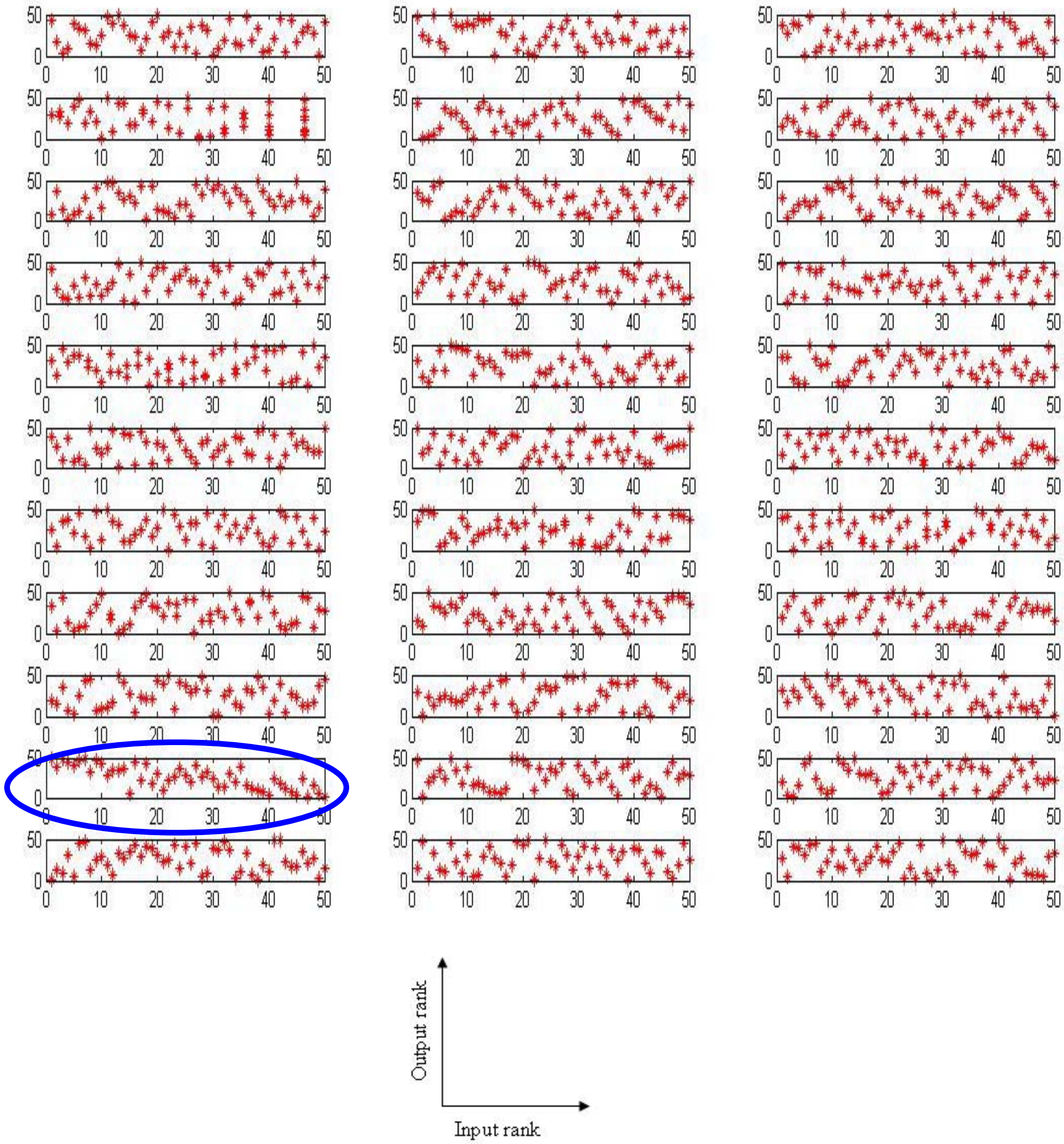
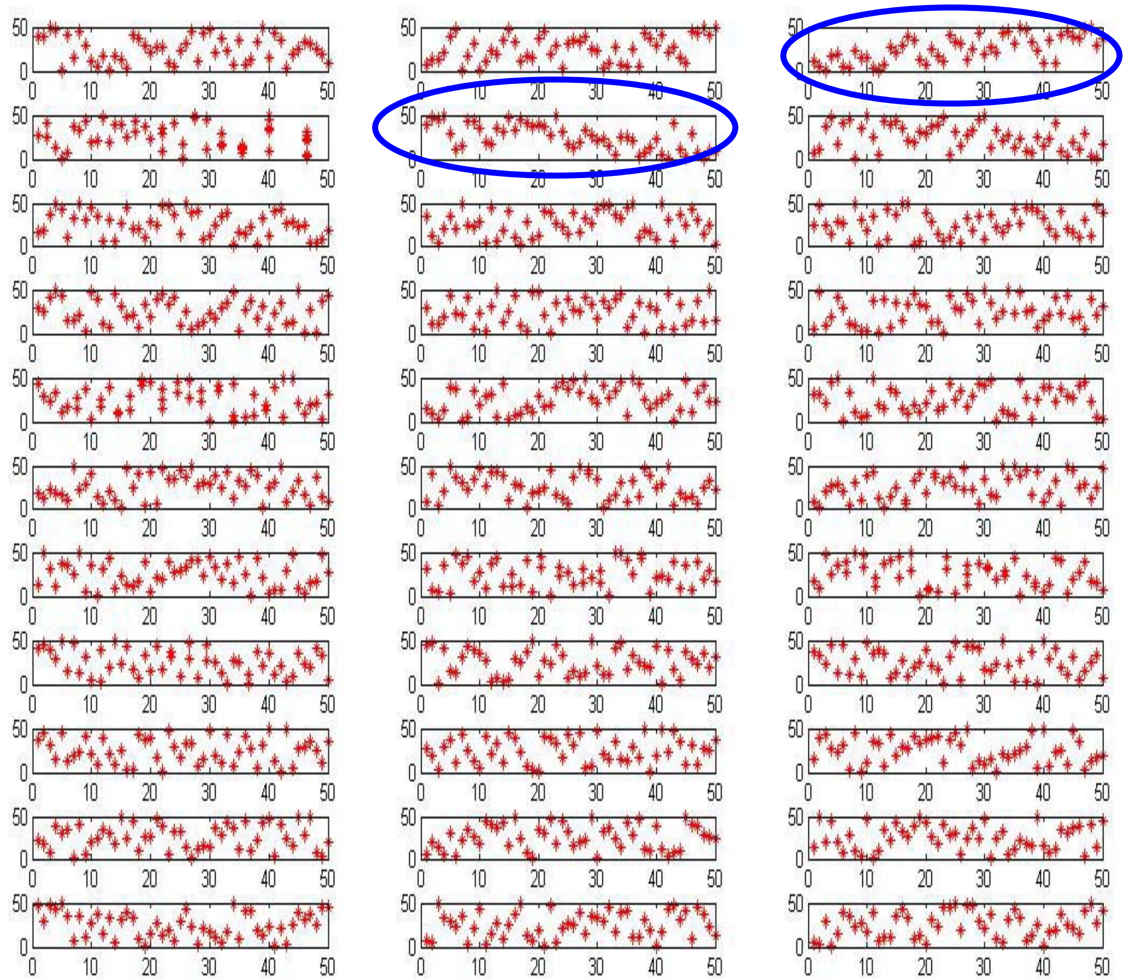


Figure 3-79: Subplots of GSA results of rank-transformed average TP at Little Falls intake from May through September 2001. See Table 3-22 to identify kinetic parameters shown here (subplots arranged in same pattern as corresponding table entries). Significant parameters with r ≥ |0.5| are circled.

Figure 3-80: Subplots of GSA results of rank-transformed average chl-*a* at Dundee Lake from May through September 2001. See Table 3-22 to identify kinetic parameters shown here (subplots arranged in same pattern as corresponding table entries). Significant parameters with r ≥ |0.5| are circled.



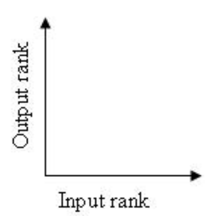


Figure 3-81: Subplots of GSA results of rank-transformed average DO at Dundee Lake from May through September 2001. See Table 3-22 to identify kinetic parameters shown here (subplots arranged in same pattern as corresponding table entries). Significant parameters with r ≥ |0.5| are circled.

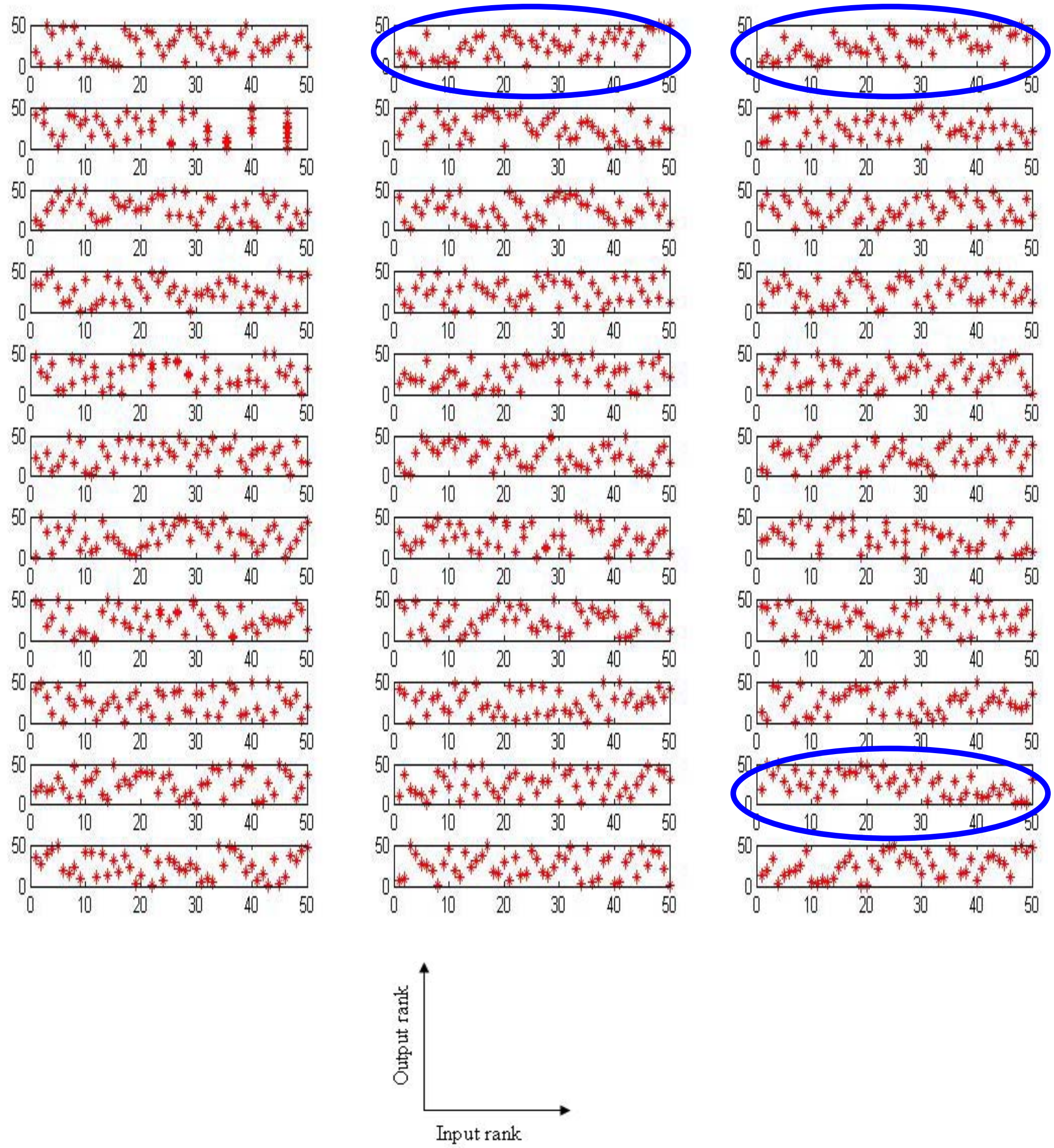


Figure 3-82: Subplots of GSA results of rank-transformed average DO at Peckman River mouth from May through September 2001. See Table 3-22 to identify kinetic parameters shown here (subplots arranged in same pattern as corresponding table entries). Significant parameters with r ≥ |0.5| are circled.

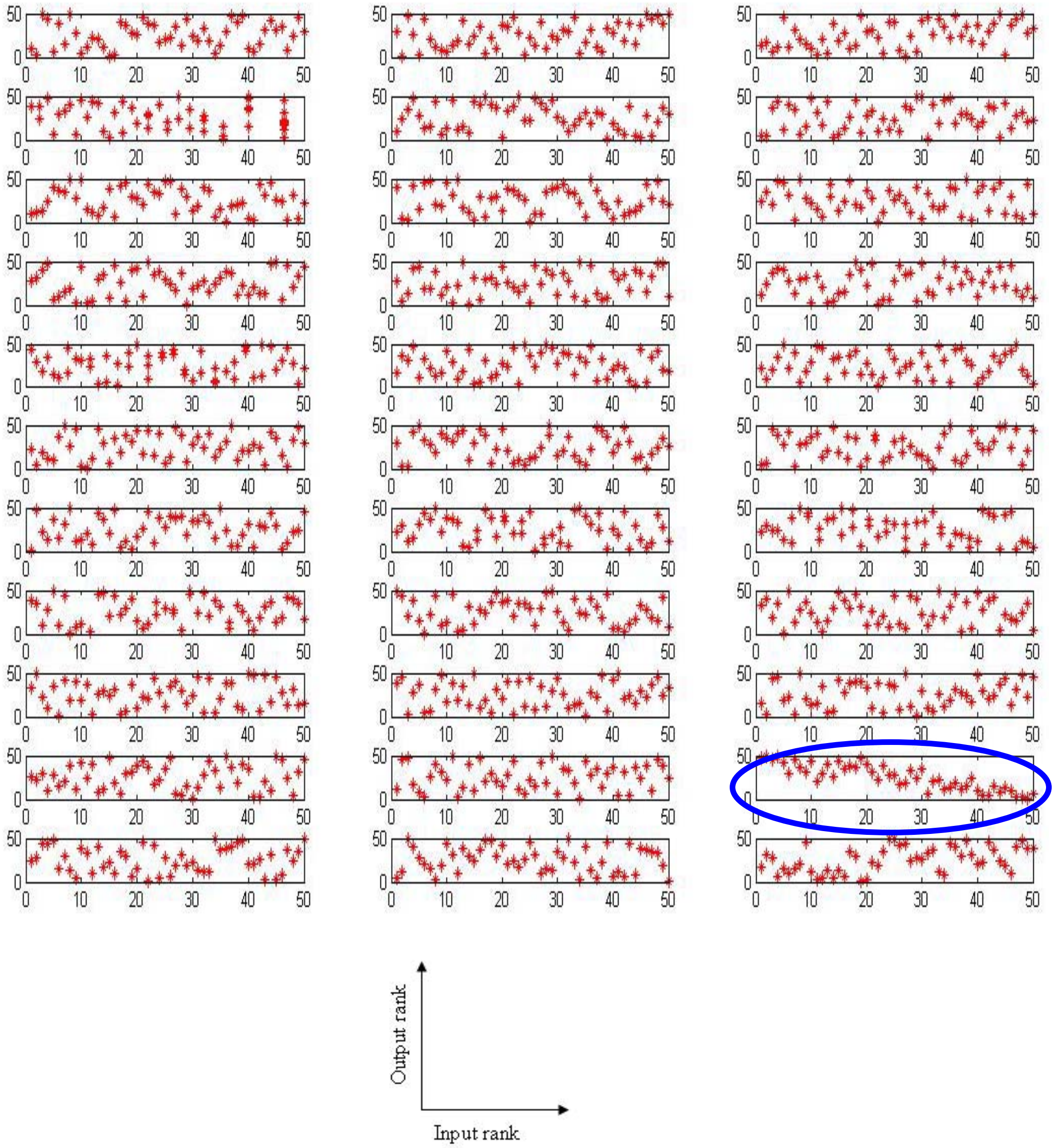


Figure 3-83: Subplots of GSA results of rank-transformed average DO at Passaic River near Chatham from May through September 2001. See Table 3-22 to identify kinetic parameters shown here (subplots arranged in same pattern as corresponding table entries). Significant parameters with r ≥ |0.5| are circled.

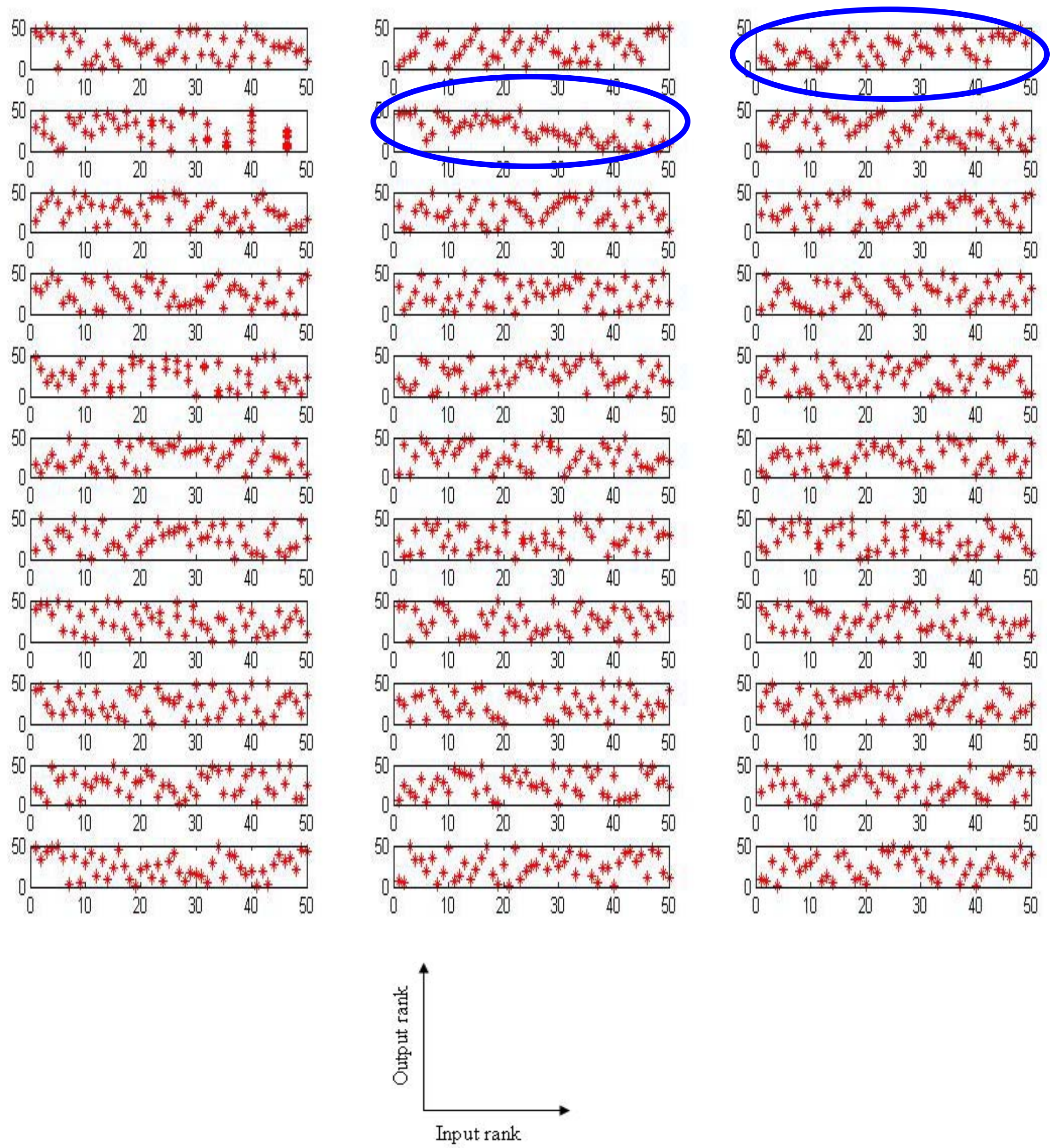


Figure 3-84: Subplots of GSA results of rank-transformed average DO at station PA10 from May through September 2001. See Table 3-22 to identify kinetic parameters shown here (subplots arranged in same pattern as corresponding table entries). Significant parameters with r ≥ |0.5| are circled.

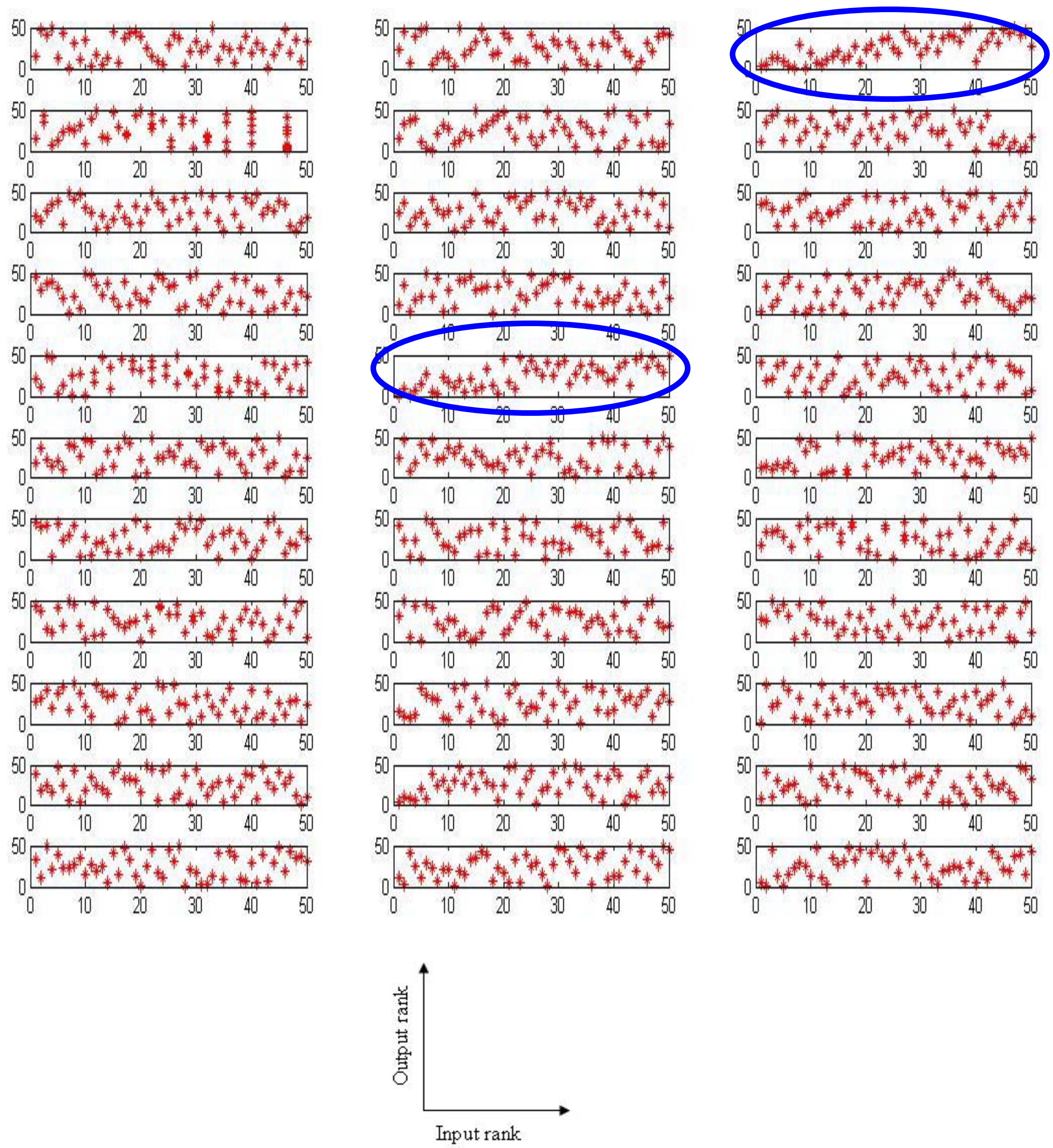


Figure 3-85: Subplots of GSA results of rank-transformed average diurnal DO swing at Dundee Lake from May through September 2001. See Table 3-22 to identify kinetic parameters shown here (subplots arranged in same pattern as corresponding table entries). Significant parameters with r ≥ |0.5| are circled.

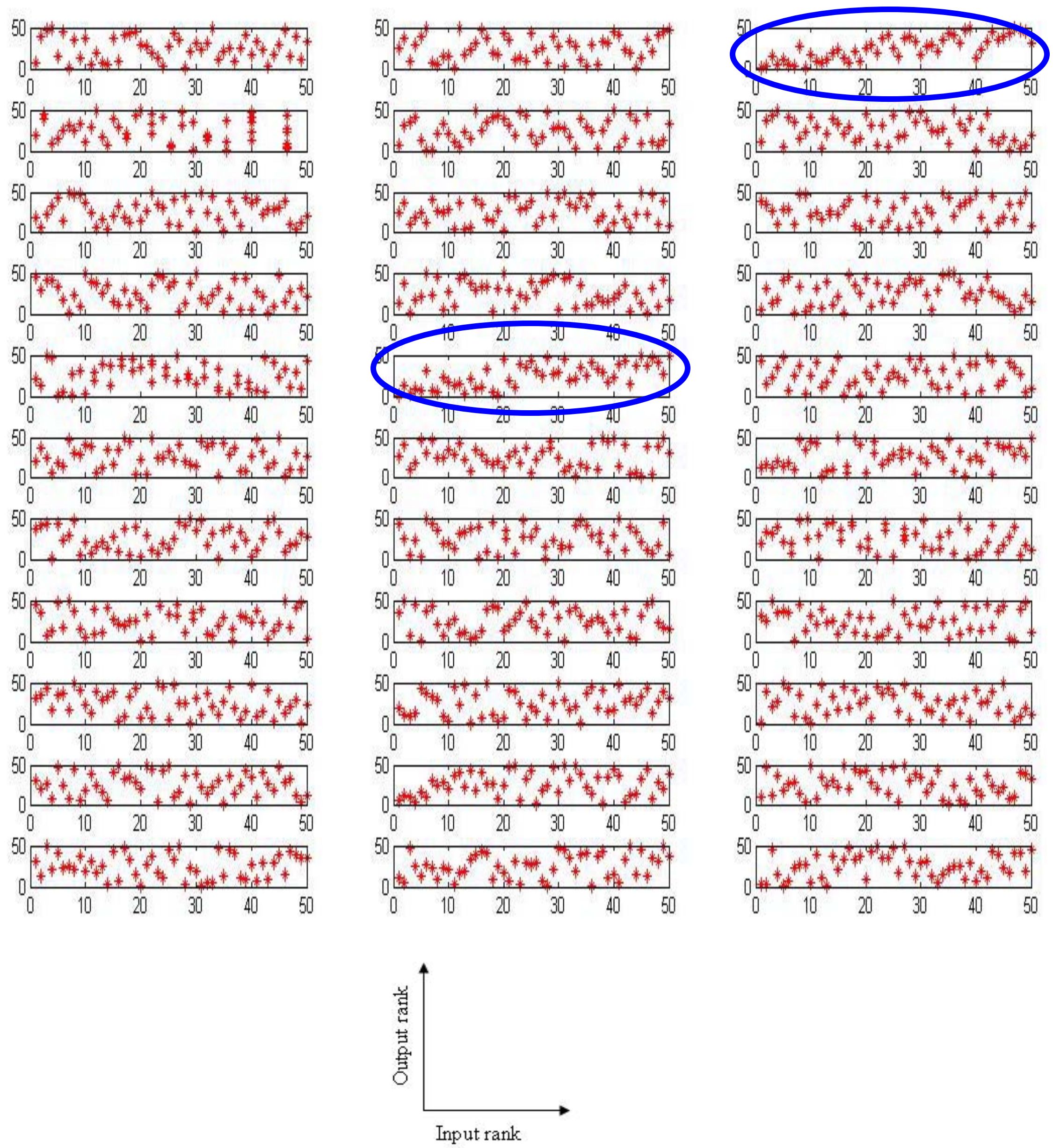


Figure 3-86: Subplots of GSA results of rank-transformed average diurnal DO swing at Peckman River mouth from May through September 2001. See Table 3-22 to identify kinetic parameters shown here (subplots arranged in same pattern as corresponding table entries). Significant parameters with r ≥ |0.5| are circled.

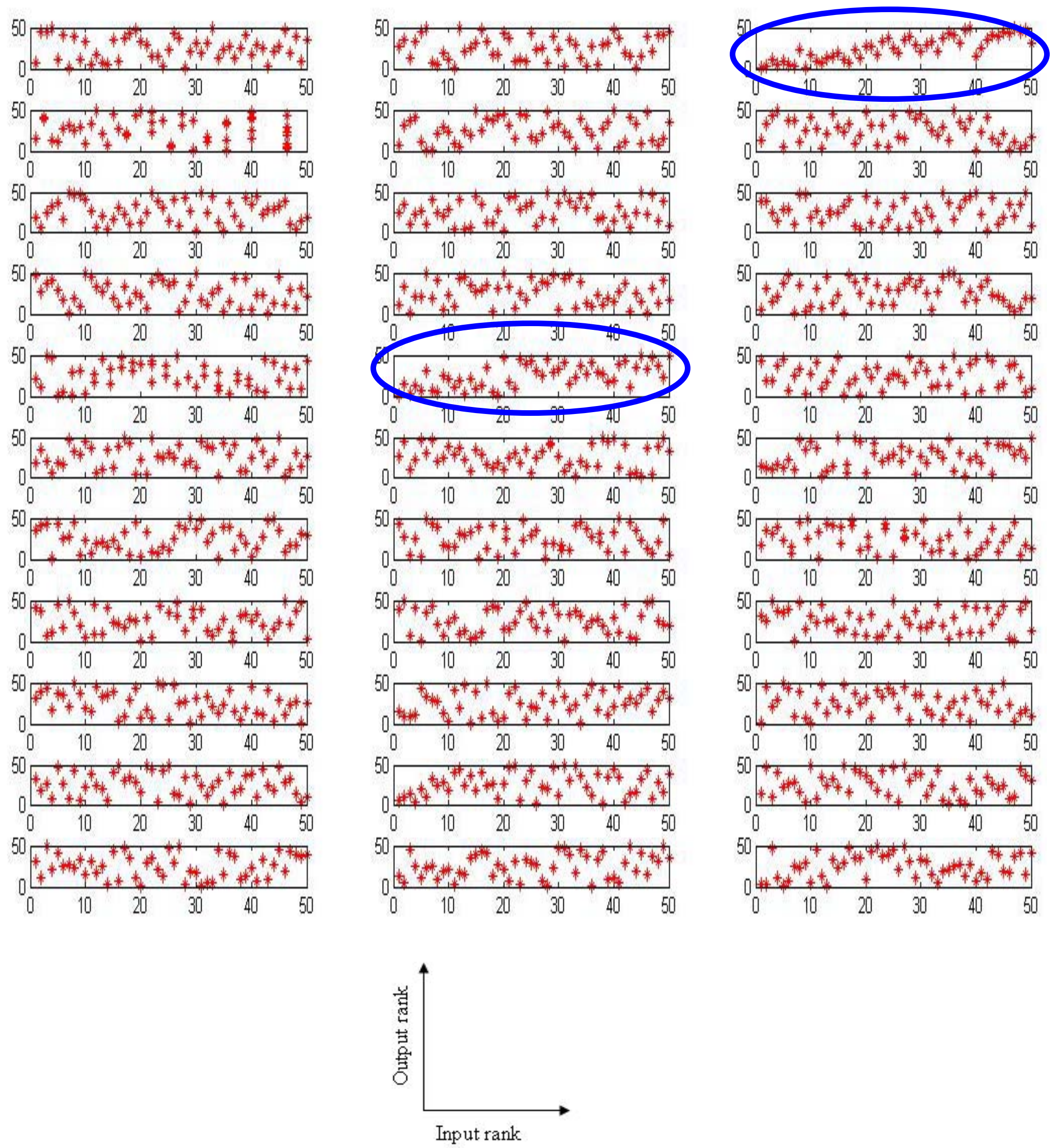


Figure 3-87: Subplots of GSA results of rank-transformed average diurnal DO swing at Passaic River near Chatham from May through September 2001. See Table 3-22 to identify kinetic parameters shown here (subplots arranged in same pattern as corresponding table entries). Significant parameters with $r \geq |0.5|$ are circled.

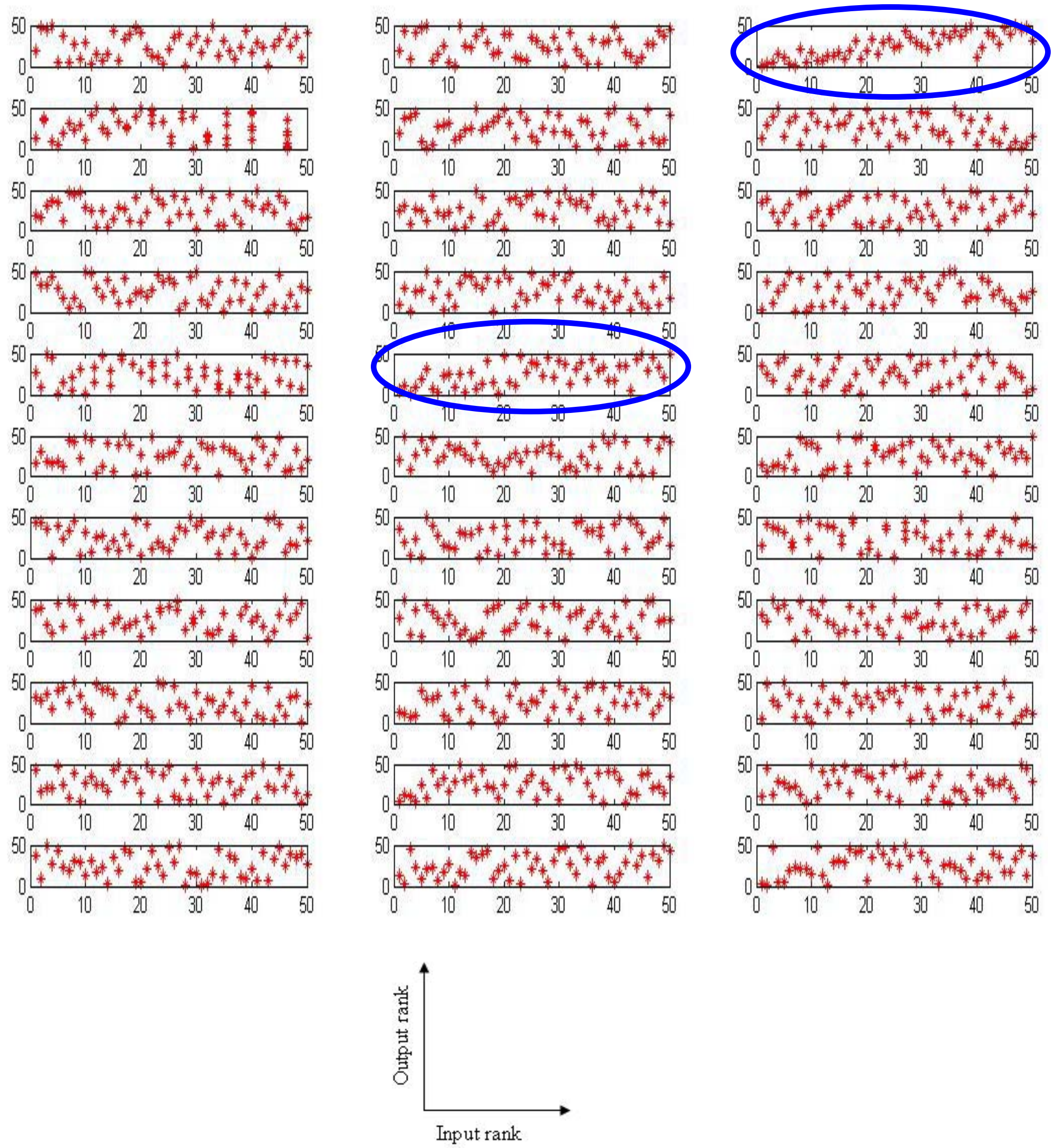


Figure 3-88: Subplots of GSA results of rank-transformed average diurnal DO swing at station PA10 from May through September 2001. See Table 3-22 to identify kinetic parameters shown here (subplots arranged in same pattern as corresponding table entries). Significant parameters with r ≥ |0.5| are circled.

Chapter 4: Uncertainty analysis of the TMDL scenario (Stage 2)

4.1 Introduction

Stage 2 focused on the uncertainty of the TMDL scenario at future critical drought conditions in which all WWTPs discharge a LTA of 0.4 mg/L TP effluent at permitted flows, and nonpoint sources achieve 60% reduction of TP loading (NJDEP, 2008a). Specifically, the objective of Stage 2 was to test the first two research hypotheses:

1. The Passaic TMDL will result in attainment of dissolved oxygen surface water quality standards and site-specific chlorophyll-*a* criteria at Dundee Lake, with less than 10% expected exceedance and 10% exceedance probability, respectively, at critical drought conditions;
2. The Passaic 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 intake, with less than 10% exceedance probability.

4.2 Methodology

4.2.1 Timeframe

In the TMDL scenario, hydrologic conditions of WY2000-WY2003 were simulated with the modifications of all WWTPs discharging at permitted flows, and a scenario of increased pumping at the WS intake to represent future growth in demand. WY2002 represents the year with critical drought conditions. The highest pumping demand was simulated to occur in that water year (Table 4-1). Since WY2002 is the critical year, with the fixed model predicting the highest load of TP diverted from the WS intake to the Wanaque Reservoir and the highest seasonal average concentration of chl-*a* in Dundee Lake (Tables 4-2 and 4-3), the uncertainty analysis focused only on WY2002.

4.2.2 Application of LHS to select input variables

As was done in Stage 1, the scope of Stage 2 was restricted to four types of variables: select i) global and ii) local kinetic parameters, and select iii) headwater and iv) point source boundary conditions.

4.2.2.1 Selection of variables

Fifteen kinetic parameters (8 global and 7 local parameters) and twenty boundary conditions for phosphorus were selected. The kinetic parameters were selected based on the GSA in Stage 1. Several more WWTP boundary condition variables were added in Stage 2. Whereas Stage 1 only covered 4 WWTPs (based on criteria described in Chapter 3), Stage 2 covered the 18 WWTPs from Table 1-1 with ≥ 1.0 MGD of permitted or average flow. The increase of WWTPs included in Stage 2 reflects the additional uncertainty related to WWTPs in the TMDL scenario. In Stage 1 there was existing data to characterize each WWTP, however such data is lacking for a future scenario where most WWTPs are simulated to discharge much lower TP effluent levels than at present conditions.

4.2.2.2 Global and local kinetic parameters

The global and local kinetic parameters included in Stage 2 are listed in Table 4-4. The same PDFs applied in Stage 1 were applied in Stage 2 also. The approach of Melching and Bauwens (2001) was once again used for the variables "SOD" and "Fraction of bottom segment covered with benthic algae".

4.2.2.3 Parameter covariance

For the kinetic parameters carried forward from Stage 1, the same covariance relationships from Stage 1 were assumed here. Correlation coefficients between specific parameters were entered into the LHS software (i.e., ARRAMIS), as listed in Table 4-5 and illustrated in Figure 4-1.

4.2.2.4 Headwater boundary conditions for phosphorus

The uncertainty of phosphorus boundary conditions at the Passaic and Ramapo River headwaters were modeled the same way as in Stage 1, using the same scaling distributions. Note that in the TMDL scenario, orthophosphate and organic phosphorus concentrations in the Passaic River headwaters are predicted to show no change from current conditions. However, in the Ramapo River headwaters those same constituents are predicted to show much lower concentrations due to assumed implementation of the Pompton Lakes TMDL (NJDEP, 2008a). This study assumed that the same proportion of uncertainty would continue to apply at the Ramapo River headwater boundary conditions for phosphorus in Stage 2 as in Stage 1. The scaling distributions applied in Stage 2 were the same as in Stage 1, and were applied to the assumed orthophosphate and organic phosphorus profiles for the Passaic River and Ramapo River headwaters.

4.2.2.5 WWTP boundary conditions for phosphorus

In the TMDL scenario, all WWTPs are simulated to discharge a TP effluent of 0.4 mg/L. This number is merely a long-term average and is expected to contain daily variation, a source of uncertainty. This phenomenon was reflected in Stage 2 of the uncertainty analysis through the following steps. The TMDL document assumes that upon TMDL implementation, the TP effluent from each WWTP will follow a lognormal distribution with an arithmetic mean of 0.4 mg/L and COV of 0.6 (NJDEP, 2008a). Stage 2 applied the NJDEP assumption; TP effluent from each WWTP was assumed to follow a lognormal distribution with an arithmetic mean of 0.4 mg/L and COV of 0.6

(Fig. 4-2). To increase efficiency of the uncertainty analysis, only the 18 WWTPs with ≥ 1.0 MGD of permitted or average flow were included in Stage 2. (The other 4 WWTPs (e.g., Chatham Glen STP, Warren Stage I-II STP, Warren Twp. SA Stage IV STP, and Warren Stage V STP) were modeled with fixed boundary conditions). For a single model run, the TP effluent from each of the 18 WWTPs was considered as a separate variable, and assigned 1 of 18 different sets of LHS-generated probability distributions, where each set contained 365 random values that had the same arithmetic mean and distribution shape as Figure 4-2.

4.2.2.5.1 Regression analysis of orthophosphate and TP concentrations in measurements of WWTP effluent in the NTPRB

The next step was to translate the randomly sampled TP daily value into orthophosphate and organic phosphorus components. In the absence of data on the future characteristics of WWTP effluent under the TMDL scenario, the approach of Stage 1 – developed for past rather than future conditions – was not taken. Instead in Stage 2, a data aggregation approach was used. In Stage 2, effluent samples taken in 2003 from each of the 22 WWTPs were aggregated in an attempt to discern any clear trends in the relationship of orthophosphate to total phosphorus. Following are the details of that analysis.

Omni Environmental collected 6 samples each of orthophosphate and TP between June through November 2003 at 22 WWTPs in the NTPRB. One outlier was removed from the analysis. Therefore 131 paired samples of orthophosphate and TP were included in the analysis. Based on previous surveys of the WWTPs, the facilities were categorized as using either a) chemical treatment, b) enhanced biological phosphorus removal (EBPR) or c) Activated Sludge (AS) processes to remove phosphorus (Table A-11).

In addition, Two Bridges SA had about 80 samples each in 2003 of orthophosphate and TP collected in-house. However, these measurements were not included in the analysis because it skewed the linear regression equation too heavily towards the measurements of one facility.

Figure 4-3 plots the WWTP aggregated paired samples of orthophosphate and TP. A clear linear relationship exists, and is approximated by the equation:

$$y = 0.977 \cdot x - 0.07 \qquad \text{(Eq. 4.1)}$$

where

y = orthophosphate (mg/L),

and x = TP (mg/L).

Equation 4.1 has an r^2 value of 0.99.

The next step was to verify the applicability of Equation 4.1 to each of the phosphorus removal processes. Figures 4-4 through 4-6 plot the WWTP data while separately identifying each phosphorus removal process. Again, a clear linear relationship exists in each case, with a slightly different linear regression equation approximating each dataset. The slope values are very similar, ranging from 0.95 to 0.99. The equation for the AS removal data has a positive y-intercept (Fig. 4-5), which is impossible in reality since orthophosphate cannot be above 0 if TP equals 0. The chemical treatment dataset (Fig. 4-6) yields a regression equation with the lowest y-intercept at -0.087.

Given the overall similarities between the three different regression equations for each phosphorus removal process, it is reasonable to use Equation 4.1 in modeling of WWTP effluent for TMDL and trade scenarios. Furthermore, the low y-intercept value in Equation 4.1 compares well with that of Figure 4-6, which is important in predicting orthophosphate at low TP concentrations where chemical treatment is likely. Finally, the residual plot in Figure 4-7 demonstrates no bias in the error of Equation 4.1.

The phenomena of a strong linear relationship and high coefficient value for the orthophosphate term (i.e., 0.977) in Equation 4.1 are due to all the WWTPs having very low effluent Total Suspended Solids (TSS) concentrations. During the months that Omni Environmental collected the WWTP effluent samples, all the WWTPs had very low monthly average TSS concentrations, with most reporting below 5 mg/L TSS; the highest value was 14 mg/L (NJDEP, 2008b) (Fig. 4-8). The low concentration of effluent particulate matter is the reason for a low fraction of organic phosphorus in the effluent. Equation 4.1 should therefore not be generally applied, but only in this particular situation where all the WWTPs had low effluent TSS.

To summarize, WWTP effluent orthophosphate was modeled in Stage 2 according to Equation 4.1. Organic phosphorus was modeled as the difference between TP and orthophosphate. Because organic phosphorus and orthophosphate could each be expressed directly from a single random variable x in Equation 4.1, they were counted as one overall variable in Stage 2 of the uncertainty analysis.

4.2.3 Execution of uncertainty analysis

As with Stage 1, Stage 2 also had 35 random input variables. Accordingly, 50 LHS sample sets were generated of i) global and local kinetic parameters, and ii) headwater and WWTP boundary conditions for organic phosphorus and orthophosphate spanning 10/1/01 – 9/30/02.

4.2.4 Output processing

4.2.4.1 TP

In the TMDL model, the daily diverted load from the WS intake to the Wanaque Reservoir is assumed to equal the daily average TP concentration at the intake multiplied by the daily diverted flow. A time series of daily diverted flows in the WY2002 TMDL scenario was provided by NJDEP (Omni Environmental, 2007a); the diverted flow series was regarded in this study as a fixed quantity. Therefore, uncertainty in the diverted load was strictly a function of uncertainty in the TP concentration at the WS intake.

For each of the 50 model realizations, a 4-day moving average TP concentration at the intake was calculated for each day, and then multiplied by the given pump flow rate for that day to obtain a series of daily diverted loads. After summing the daily load values, 50 values for annual diverted load were obtained with one value for each model run. (In each annual load total the first 5 days were not included to remove any transient effects from initial conditions of the simulation; the last 2 days were not included due to the use of a 4-day moving average for TP concentration). These values were plotted as a cumulative distribution. The mean and standard deviation of the distribution were calculated. The 90th percentile value, which corresponds to the 10% exceedance probability, was directly compared to the TMDL 70% reduction target.

Note that with respect to the reduction target, the TMDL expressed the diverted load target as a ten-year average of 4088 kg/yr, spanning 1993-2002 (NJDEP, 2008a), rather than a single-year figure. For this study, the 70% reduction target for WY2002 was taken to be 70% reduced from the predicted diverted load under the actual conditions in WY2002. After removing days 1-5 and 364-365 from the total amount in order to match the timeframe of Stage 2 output processing, the reduction target amounts to 70% less than 39,997 kg, or 11,999 kg.

The cumulative distribution of WY2002 average TP concentration at the Little Falls intake was also plotted.

4.2.4.2 Chl-a

For each of the 50 model realizations, the seasonal average (June 15 to September 1) of chl-*a* at Dundee Lake was calculated. The 50 values for seasonal average were compiled and plotted in a cumulative distribution. The mean and standard deviation of the distribution were calculated. The 90th percentile value, which corresponds to the 10% exceedance probability, was directly compared to the TMDL target of 20 μg/L.

4.2.4.3 DO

Three different types of output were generated for DO: i) cumulative distribution of daily average DO exceedance frequencies over the period of June 15 to September 1, ii) cumulative distribution of minimum DO exceedance frequencies over the period of June 15 to September 1, and iii) cumulative distribution of diurnal DO swing over the period of June 15 to September 1. These outputs were generated at Dundee Lake, the Peckman River mouth, the Passaic River near Chatham and station PA10. Note that the June 15-September 1 period was chosen because it corresponds to the span of the seasonal average chl-*a* criteria and captures the time of peak productivity.

The distribution of daily average DO exceedance frequencies was calculated as follows. Dundee Lake, the Peckman River mouth, the Passaic River near Chatham and station PA10 are all classified as FW2 waters. The applicable New Jersey Surface Water Quality Standard for daily average DO is 5 mg/L (NJDEP, 2004). For each of the 50 model realizations, the exceedance frequency of the daily average DO standard over the period of June 15-September 1 was calculated by dividing the number of days with average DO less than 5 mg/L by the total number of days. The cumulative distribution of the 50 calculated exceedance frequencies was then plotted. As conceived by Borsuk et al. (2002), the mean of the distribution is termed the Expected Exceedance (EE), and the percentage below the 10% EPA guideline for exceedance frequency is termed the Confidence of Compliance (CC). The CC can also be thought of as "the probability that the true exceedance frequency is below the 10% EPA guideline" (Borsuk et al., 2002). The bounds of the middle 90% of the values are considered the 90% confidence interval.

The distribution of minimum DO exceedance frequencies was calculated as follows. For FW2 waters, the applicable New Jersey Surface Water Quality Standard for minimum DO is 4 mg/L (NJDEP, 2004). The same procedure was followed as in the preceding paragraph, except that for each of the 50 model realizations, the exceedance frequency of the minimum DO standard was calculated by dividing the number of days with DO less than 4 mg/L by the total number of days. The remaining steps in the preceding paragraph were repeated.

The distribution of diurnal DO swing was calculated as follows. The diurnal DO swing was calculated for each day in each model run over the period of June 15-September 1. The cumulative distribution of all the calculated diurnal swings was then plotted.

4.2.5 Stage 2B

An intermediate research step termed Stage 2B was carried out in order to investigate if 50 model runs were sufficient. In Stage 2B the same process was followed as in Stage 2, except to generate 100 LHS sample sets instead of 50. The same steps for output processing of TP, chl-*a* and DO were done on the set of 100 model realizations.

4.3 Results

4.3.1 Comparison of Stage 2 and Stage 2B results

The first step was to compare results from Stages 2 and 2B. A two-sample Kolmogorov-Smirnov (K-S) test was done to compare the various distributions generated for each of the TP, chl-*a* and DO outputs. The two-sample K-S test calculates the probability that two distributions are different under the null hypothesis that both samples come from the same distribution. Figures 4-9 through 4-22 show cumulative distribution function (CDF) comparisons of Stages 2 and 2B results.

Results of the K-S tests (Table 4-6) show that for average and minimum DO exceedance frequency, seasonal average chl-*a*, TP diverted load, and annual average TP concentration at Little Falls intake, there is no evidence to suggest significant differences in output between Stages 2 and 2B. Comparison of the diurnal DO swing K-S test results show much smaller p-values and significant differences at $\alpha = 0.05$ (though not significant at $\alpha = 0.01$) at two locations, Dundee Lake and station PA10. However at these two locations, the absolute value of percent differences in the mean, standard deviation, and 90th percentiles of the two distributions range from 1.1 to 5.5% (Table 4-7); in addition the distribution means are not significantly different (p= 0.31 at Dundee Lake and p = 0.10 at station PA10 according to two-sample t-tests). The K-S test results of the other distributions, and the negligible percent differences in the diurnal DO swing distributions are consistent with the claim that 50 model runs was a valid choice in the uncertainty analysis. This finding agrees with guidance from Iman and Helton (1985) on the number of model runs needed in an LHS analysis.

Results from Stages 2 and 2B are reported in Tables 4-8 through 4-14. Since Stage 2B results are propagated from a greater number of model runs, those are considered the definitive results for the TMDL scenario uncertainty analysis.

4.3.2 TP

Although Stages 2 and 2B distributions show no significant difference ($p = 0.94$), this is the one case where Stages 2 and 2B distributions show an important deviation at a key value. The deviation between the two distributions at the 90th percentiles is high enough that Stage 2B has a lower target exceedance probability by 44%; in fact with Stage 2B it dips just below the threshold 10% exceedance probability to 9%, whereas Stage 2 shows a 16% exceedance probability of the target load (Table 4-8 and Fig. 4-9).

To reconcile the difference between the two distributions at the 90th percentile, a normal distribution was fitted to each of the diverted load CDFs of Stages 2 and 2B (Fig. 4-10). The normal approximations were justified by i) Lillie test high p-values (0.44 and 0.50 for Stages 2 and 2B, respectively), and b) their identical mean and standard deviation values to the corresponding distributions without the normal approximation. After applying a normal approximation to each distribution, the two values for exceedance probability of the target load showed much closer agreement: 9.2% for Stage 2 and 9.8% for Stage 2B (Table 4-9). Again, the latter result is considered the definitive answer for the TMDL scenario.

The second hypothesis of the study is **supported**. The Passaic TMDL will result in attainment of a 70% reduction, at critical drought conditions, of TP load diverted to the Wanaque Reservoir from the WS intake, with less than 10% exceedance probability. The TMDL margin of safety, an implicitly derived number (NJDEP, 2008a), has been justified with respect to diverted TP load from the WS intake to the Wanaque Reservoir. Through rigorous uncertainty analysis, the TMDL margin of safety has been demonstrated to be sufficient with regards to reducing diverted TP load from the WS intake to the Wanaque Reservoir.

With respect to average TP concentration at the Little Falls intake, the TMDL scenario indicates a low margin of uncertainty, with a coefficient of variation of 7.7%. The 90th percentile concentration is 0.14 mg/L (Table 4-10 and Fig. 4-11). Reliable data at this location was not available for comparison in Stage 1, therefore credibility of the Stage 2B results cannot be ascertained.

4.3.3 DO

Tables 4-11 and 4-12 indicate less than 10% expected exceedance of daily average and minimum DO at Dundee Lake, the Peckman River mouth, and station PA10, but greater than 10% expected exceedance of daily average and minimum DO standards at the Passaic River near Chatham (Figs. 4-12 through 4-17). Results from Stage 1 suggest that expected exceedances at the latter location are likely even higher than what was calculated in Stage 2B. Stage 1 results also suggest that uncertainty estimates of DO at Dundee Lake, Peckman River mouth and station PA10 are credible in the TMDL scenario.

The critical location in the TMDL is Dundee Lake. Given that it has a zero expected exceedance and 100% confidence of compliance, the DO portion of the first research hypothesis is **supported**. The Passaic TMDL will result in attainment of DO surface water quality standards at Dundee Lake, with less than 10% expected exceedance at critical drought conditions.

In terms of diurnal DO swing, prior to the development of the Passaic TMDL, NJDEP had authorized the use of a document entitled the *Technical Manual for Phosphorus Evaluations* (NJDEP, 2003). The purpose of the document was to determine when point source discharge of phosphorus resulted in excessive growth of algae and adverse impacts to DO. Algae produce oxygen in daylight and respire at night. High diurnal DO swings indicate excessive levels of algae. The document used 6.0 mg/L as the maximum value for an acceptable diurnal DO swing. Using that criteria as a guideline, in the TMDL scenario two locations – Peckman River mouth and station PA10 – show exceedance probabilities above 10%; and two locations –Dundee Lake and Passaic River near Chatham - show exceedance probabilities below 10% (Table 4-13 and Figs. 4-18 through 4-21) . Dundee Lake is the TMDL critical location and the results there support the adequacy of the TMDL measures for diurnal DO swing at Dundee Lake. The Peckman River mouth shows a 10% exceedance probability at 7.88 mg/L, much higher than the 6.0 mg/L guideline.

4.3.4 Chl-a

The distribution of predicted seasonal averages of chl-*a* at Dundee Lake indicates a mean value below the 20 µg/L criteria, but the wide margin of uncertainty results in a 30% probability of exceeding the criteria (Table 4-14 and Fig. 4-22). Stage 1 results, based on analysis at a nearby location with a small sample size, suggest that uncertainty estimates of chl-*a* at Dundee Lake are credible in Stage 2.

The chl-*a* portion of the first research hypothesis is **rejected**. The Passaic TMDL will **not** result in attainment of site-specific chlorophyll-*a* criteria at Dundee Lake, with less than 10% exceedance probability at critical drought conditions.

Despite the high exceedance probability for chl-*a* at Dundee Lake, the expected exceedance there for DO is zero. DO is a more important water quality parameter than chl-*a*. Although the TMDL margin of safety is not sufficient to ensure at least a 90% probability of meeting the seasonal average chl-*a* standard, in terms of DO at Dundee Lake, the margin of safety is sufficient. This reinforces the point in Chapter 3 that exceedances of chl-*a* criteria do not necessarily imply exceedances of DO water quality standards at the Dundee Lake site. This phenomenon traces back to the model's much higher sensitivity of DO at Dundee Lake to benthic algae variables rather than phytoplankton variables.

In contrast with the diverted TP load portion of the TMDL, for chl-*a* at Dundee Lake the TMDL margin of safety has not been justified. Through rigorous uncertainty analysis, the margin of safety in the TMDL has been demonstrated to be insufficient with regards to achieving seasonal average chl-*a* criteria at Dundee Lake. However, perhaps more importantly, the TMDL margin of safety has been demonstrated to be sufficient in terms of meeting DO water quality standards at Dundee Lake.

4.3.5 TMDL efficacy scenario: The effect of TMDL measures on actual WY2002 conditions

The uncertainty analysis of the TMDL scenario found instances of water quality target exceedance probabilities and expected exceedances greater than ten percent in terms of chl-*a* at Dundee Lake, and DO measures at the Peckman River mouth, the Passaic River near Chatham, and station PA10. This raises a question - is the TMDL still worth implementing if it cannot reliably achieve all water quality targets? Or can it be shown that TMDL implementation would be effective in significantly improving water quality over actual conditions? To investigate this, TMDL measures (i.e., LTA 0.4 mg/L TP effluent from WWTPs and 60% reduced nonpoint source loads) were simulated to occur during actual WY2002 conditions when WWTPs mostly discharged less than permitted flows. (The TMDL scenario assumed that WWTPs discharged permitted flows). Also, the actual pumping scenario at the WS intake was simulated rather than the future pumping scenario.

Results were then compared to the WY2002 actual conditions without TMDL implementation, i.e. Stage 1 output, from the matching timeframe. The left-tailed two-sample K-S test was applied to compare output distributions of the two scenarios. The left-tailed two-sample K-S test determines the probability that one continuous distribution is smaller than another under the null hypothesis that both samples come from the same distribution. K-S test results are shown in Table 4-15.

To execute this, the 50 LHS sample sets in Stage 2 were re-run at WY2002 actual hydrodynamic conditions. The results for each water quality parameter are described below.

4.3.5.1 TP

Diverted TP load would have shown a significant improvement (p=0) over actual conditions if TMDL measures had been in place in WY2002 (Table 4-16 and Fig. 4-23).

Similar trends of improvement (p=0) are seen at the Little Falls intake if TMDL measures had been implemented in WY2002 (Table 4-17 and Fig. 4-24).

4.3.5.2 Chl-a

Seasonal average chl-*a* at Dundee Lake would have shown a significant improvement (p=0) over actual conditions if TMDL measures had been in place in WY2002 (Table 4-18 and Fig. 4-25).

4.3.5.3 DO

The impact of TMDL measures on DO is clearest when analyzing the reduced diurnal DO swing distribution at each location in Table 4-19 and Figures 4-26 through 4-29. Reductions of 70%, 29%, 20%, and 56% in the mean diurnal DO swing at Dundee Lake, Peckman River mouth, Passaic River near Chatham, and station PA10, respectively, would have occurred if TMDL measures had been in place in WY2002.

It is notable that in the TMDL efficacy scenario, the Passaic River near Chatham showed a significant increase (p=0.01) in expected exceedance of daily average DO from 22% to 35% (Table 4-20 and Fig. 4-30), while at the same time the mean diurnal DO swing was reduced. Concurrently, the site would have seen only marginal improvement in expected exceedance of the minimum DO standard. 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. Reduced phosphorus loading will actually have the effect of reducing daily average DO via reduced primary productivity.

This aligns with the Omni Environmental (2007a) finding that in the Upper Passaic River, productivity generally increases average DO.

Other results for distributions of daily average and minimum DO exceedance frequency show no significant differences (Tables 4-21 and 4-22 and Figs. 4-31 through 4-35).

4.3.6 Confirmation of kinetic parameters selection in Stage 2

As mentioned in Section 4.2.2.1, the selection of kinetic parameters for Stage 2 was based on GSA results in Stage 1. Following the execution of Stage 2, a local sensitivity analysis was done to verify that the discarded parameters found to be not sensitive at Stage 1 conditions (Table 3-24) were also not sensitive at Stage 2 conditions.

A local sensitivity analysis was done on high and low percentile model runs from Stage 2. All kinetic parameters in Table 3-24 showed negligible sensitivity except for "Phytoplankton half-saturation constant for phosphorus". For that particular parameter, high percentile parameter values greatly affected high percentile chl-*a* runs. For example, when the 90th percentile parameter value was applied to the Stage 2 90th percentile model run for seasonal average chl-*a*, the seasonal average diminished sharply from 34.8 to 16.9 μg/L. However this is a dubious result because the fixed-model seasonal average of 17.4 μg/L would then become greater than the 90th percentile outcome in a wide-ranging distribution. The cause of this doubtful result is that the assumed parameter range specified in Table 3-2 is probably too high. The parameter probability distribution specified in Table 3-2 has a COV of 47%. A 47% COV is considered high for this variable type according to Brown and Barnwell (1987); a normal COV for this variable type is reported as 10 to 20%. If a COV of 20% is assumed instead, then the parameter shows no sensitivity in Stage 2 and the Stage 2 results reported earlier are unaffected. The corresponding probability distribution for the variable would have a minimum and maximum of 0.0012 and 0.0038, respectively, rather than 0.0005 and 0.01 (Fig. 4-36).

Comparing these findings, it is believed that the narrower distribution in Figure 4-36 is more reasonable than the wider distribution because i) the narrower distribution has a COV that is considered normal rather than high (according to criteria in Brown and Barnwell, 1987), and ii) the narrower distribution yields a Stage 2 output distribution of seasonal average chl-*a* that is more plausible, with the fixed-model output falling at approximately the 70th percentile rather than the 90th to 95th percentile in a wide-ranging output distribution. (Stage 1 results would not be affected by the narrower parameter distribution since it was not a significantly sensitive parameter at Stage 1 conditions).

Measurements of the half-saturation constant for phosphorus should be made on phytoplankton samples collected from Dundee Lake to confirm this belief. If the COV approaches the high range of 50% then Stage 2 should be redone, since the exceedance probability of predicted seasonal average chl-*a* would most likely decrease to less than 10% and the first research hypothesis would not be rejected.

4.4 Discussion

Hydrologic model uncertainty was left outside the scope of the uncertainty analysis. However, consider the hydrologic effects of the TMDL requirement to reduce urban and agricultural nonpoint source runoff loads by 60%. (The nonpoint source load reduction was simulated in the TMDL model by reducing event mean concentrations (EMCs) specific to urban and agricultural land uses, and then recalculating flow-weighted runoff EMCs for each subbasin (Omni Environmental, 2007a)). That is an ambitious target that would require the implementation of structural best management practices (BMPs) throughout the watershed. The construction of retention basins would play a prominent role in any large scale attempt to reduce nonpoint source loads by 60%. A vast array of retention basins would have the hydrologic effect of decreased runoff and increased infiltration and baseflow.

The TMDL scenario (and the Stage 2) hydrologic model does not account for this – but if it did, what would be the likely effects? Increased baseflow would reduce phosphorus concentrations in the river network, and thus reduce algal growth and associated impacts to DO. Whether or not the phosphorus loading would be reduced is a delicate question. On the one hand lower runoff volumes would imply lower phosphorus loadings. On the other hand, the simulated reduction in EMCs can itself be interpreted as a direct result of BMP implementation, therefore the reduction in phosphorus loading has already been accounted for in the TMDL model, and to predict a further reduction would be double counting. Thus a conservative prediction would be that including the hydrologic effects of a 60% nonpoint source load reduction would lead to no change in predicted phosphorus loading, nor to the diverted phosphorus load to the Wanaque Reservoir.

The magnitude of these effects can not be determined in this study and should be the subject of future research. What is described above is only qualitative. Ultimately, it is an issue of model structure uncertainty in that a feedback link is absent between the nonpoint source load model and the hydrologic model at TMDL conditions.

In light of the above, Stage 2 uncertainty estimates for chl-*a* and DO can be viewed as conservative estimates. However, the predicted margin of exceedance regarding the chl-*a* criteria – a 30% exceedance probability - is high enough that even if it were conservative, the chl-*a* portion of the first research hypothesis would still be **rejected**. The Passaic TMDL would still **not** result in attainment of site-specific chlorophyll-*a* criteria at Dundee Lake, with less than 10% exceedance probability at critical drought conditions.

Despite the high exceedance probability for chl-*a* at Dundee Lake, the expected exceedance there is zero for daily average and minimum DO standards. The DO portion of the first research hypothesis is **supported**. The Passaic TMDL **will** result in attainment of DO surface water quality standards at Dundee Lake, with less than 10% expected exceedance at critical drought conditions. Although the TMDL margin of safety is not sufficient to ensure at least a 90% probability of meeting the seasonal average chl-*a* criteria, in terms of DO at Dundee Lake, the margin of safety is sufficient. Exceedances of chl-*a* criteria do not necessarily imply exceedances of DO water quality standards at the Dundee Lake site.

The second research hypothesis was **supported**. The Passaic TMDL **will** result in attainment of a 70% reduction, at critical drought conditions, of TP load diverted to the Wanaque Reservoir from the WS intake, with less than 10% exceedance probability. The TMDL margin of safety has been justified with respect to diverted TP load from the WS intake to the Wanaque Reservoir.

With regard to the other locations examined in Stage 2, the Passaic River near Chatham shows expected exceedances above 10% for the daily average and minimum

DO standards. In light of the Stage 1 findings on credibility of uncertainty estimates at this location, those outcomes are credible despite the hydrologic effects of the TMDL scenario discussed above. The Peckman River mouth also shows a 90^{th} percentile diurnal DO swing of 7.88 mg/L, well above the 6.0 mg/L criteria once used in NJDEP (2003). Station PA10 shows a 90^{th} percentile diurnal DO swing of 6.14 mg/L; this might decline to below the 6.0 mg/L threshold value if hydrologic effects of the TMDL scenario are accounted for. With respect to average TP concentration at the Little Falls intake, the TMDL scenario indicates a low margin of uncertainty, with a COV of 7.7%. The 90^{th} percentile concentration is 0.14 mg/L.

Even though the TMDL scenario shows expected exceedances or exceedance probabilities above the 10% target at certain locations, the efficacy of the TMDL measures was clearly demonstrated when compared directly to actual conditions in WY2002. Significant improvements were reflected in reduced chl-*a*, TP diverted load, and diurnal DO swings at all relevant locations.

Table 4-1: Diverted flow from the WS intake to the Wanaque Reservoir in the TMDL scenario, WY2000-2003 (adapted from Omni Environmental, 2007a)

Water Year	Diverted flow from WS intake to Wanaque Reservoir (cubic meters)
2000	29,798,476
2001	53,619,689
2002	111,565,278
2003	24,476,979

Table 4-2: Fixed-value model prediction of diverted TP load from the WS intake to the Wanaque Reservoir in the TMDL scenario, WY2000-2003 (adapted from Omni Environmental, 2007a)

Water Year	Diverted TP load from WS intake to Wanaque Reservoir [1] (kg)
2000	2,135
2001	4,861
2002	11,743
2003	1,805

1. The first five days of each water year were not included in order to remove any transient effects from the initial conditions in the model simulations.

Table 4-3: Fixed-value model prediction of seasonal average of chl-*a* at Dundee Lake in the TMDL scenario, WY2000-2003 (adapted from Omni Environmental, 2007a)

Water Year	Seasonal average (μg/L)
2000	5.6
2001	12.9
2002	17.4
2003	3.6

Table 4-4: WASP variables modeled as uncertain in Stages 2 and 3 of the analysis

Global kinetic parameters [unit]			
Phytoplankton maximum growth rate @ 20°C [/d]	Phytoplankton carbon: chlorophyll ratio []	Phytoplankton endogenous respiration rate @ 20°C [/d]	Phytoplankton death rate, non-zooplankton predation [/d]
Phytoplankton optimal light saturation [langleys/d]	Benthic algae maximum growth rate @ 20°C [gD/m^2/d]	Benthic algae respiration rate @ 20°C [/d]	Benthic algae death rate @ 20°C [/d]
Local kinetic parameters [unit]			
SOD [g/m^2/d]	Settling velocity of particulate phosphorus$_{1,2,3}$ [cm/s]		Fraction of bottom segment covered with benthic algae$_{1,2}$ []
Dissolved fraction of orthophosphate$_1$ []			
Boundary condition variables [a] [unit]			
Passaic River headwater scaling factor []	Ramapo River headwater scaling factor []	Berkeley Heights STP [mg/L]	Bernards Twp STP [mg/L]
Caldwell STP [mg/L]	Cedar Grove STP [mg/L]	Florham Park SA [mg/L]	Hanover SA [mg/L]
Livingston Twp STP [mg/L]	Long Hill Township STP [mg/L]	Molitor Water Pollution (Madison-Chatham) STP [mg/L]	Morris Township - Butterworth STP [mg/L]
Morristown STP [mg/L]	Parsippany - Troy Hills RSA [mg/L]	Pompton Lakes STP [mg/L]	Rockaway Valley SA [mg/L]
Two Bridges SA [mg/L]	Verona STP [mg/L]	Wanaque Valley RSA [mg/L]	Wayne Twp STP [mg/L]

[a] WWTP variables pertain to TP effluent

Table 4-5: Variable correlations entered into ARRAMIS

Variable 1	Variable 2	Correlation coefficient
Benthic algae death rate @ 20°C	Benthic algae maximum growth rate @ 20°C	0.5
Benthic algae death rate @ 20°C	Fraction of bottom segment covered with benthic algae$_1$	0.5
Benthic algae death rate @ 20°C	Fraction of bottom segment covered with benthic algae$_2$	0.5
Benthic algae death rate @ 20°C	Phytoplankton death rate, non-zooplankton predation	-0.5
Benthic algae maximum growth rate @ 20°C	Phytoplankton maximum growth rate @ 20°C	-0.5
Benthic algae respiration rate @ 20°C	Phytoplankton endogenous respiration rate @ 20°C	-0.5
Phytoplankton carbon:chlorophyll ratio	Phytoplankton maximum growth rate @ 20°C	0.5
Phytoplankton carbon:chlorophyll ratio	Phytoplankton endogenous respiration rate @ 20°C	0.5
Phytoplankton maximum growth rate @ 20°C	Phytoplankton death rate, non-zooplankton predation	0.5
Phytoplankton maximum growth rate @ 20°C	Phytoplankton optimal light saturation	0.5
Phytoplankton maximum growth rate @ 20°C	Phytoplankton endogenous respiration rate @ 20°C	0.5

Table 4-6: Results of two-sample K-S tests of Stage 2 and Stage 2B output distributions

Location	Output distribution	p-value
Dundee Lake	DO daily average exceedance frequency	1.0
Peckman River mouth		1.0
Passaic River near Chatham		0.98
PA10		1.0
Dundee Lake	Minimum DO exceedance frequency	1.0
Peckman River mouth		0.94
Passaic River near Chatham		0.50
PA10		1.0
Dundee Lake	Seasonal average of chl-*a*	0.41
Wanaque South intake	TP load diverted to Wanaque Reservoir, WY2002	0.94
Little Falls intake	Average TP concentration, WY2002	0.79
Dundee Lake	Diurnal DO swing	**0.03**
Peckman River mouth		0.07
Passaic River near Chatham		0.10
PA10		**0.02**

Table 4-7: Percent difference in diurnal DO swing distribution key values, comparing Stage 2B and Stage 2

Location	Mean (%)	Standard deviation (%)	90^{th} percentile (%)
Dundee Lake	1.1	3.8	5.5
Peckman River mouth	0.0	-2.1	-0.9
Passaic River near Chatham	-0.6	-1.7	-1.2
PA10	-1.6	-2.8	-3.4

Table 4-8: TP load diverted from WS Intake to Wanaque Reservoir during WY2002 TMDL scenario

Scenario	Mean (kg)	Standard deviation (kg)	Probability of exceeding 11,999 kg (%)	90^{th} percentile (kg)
Stage 2B	11,276	558	9	11,981
Stage 2	11,259	557	16	12,060

Table 4-9: TP load diverted from WS Intake to Wanaque Reservoir during WY2002 TMDL scenario, using Normal approximation of model output

Scenario	Mean (kg)	Standard deviation (kg)	Probability of exceeding 11,999 kg (%)	90^{th} percentile (kg)
Stage 2B	11,276	558	9.8	11,994
Stage 2	11,259	557	9.2	11,971

Table 4-10: Average TP concentration at Little Falls intake during WY2002 TMDL scenario

Scenario	Mean (mg/L)	Standard deviation (mg/L)	90^{th} percentile (mg/L)
Stage 2B	0.13	0.01	0.14
Stage 2	0.14	0.01	0.15

Table 4-11: Distribution of daily average DO exceedance frequencies during June 15-September 1, 2002 for TMDL scenario

	Expected Exceedance (%)		Confidence of Compliance (%)		90% confidence interval	
Scenario / Location	Stage 2B	Stage 2	Stage 2B	Stage 2	Stage 2B	Stage 2
Dundee Lake	0	0	100	100	0-0	0-0
Peckman River mouth	0	0	100	100	0-0	0-0
Passaic River near Chatham	20	20	56	54	0-71	0-65
PA10	0	0	100	100	0-0	0-0

Table 4-12: Distribution of minimum DO exceedance frequencies during June 15-September 1, 2002 for TMDL scenario

	Expected Exceedance (%)		Confidence of Compliance (%)		90% confidence interval	
Scenario / Location	Stage 2B	Stage 2	Stage 2B	Stage 2	Stage 2B	Stage 2
Dundee Lake	0	0	100	100	0-0	0-0
Peckman River mouth	6	6	81	76	0-35	0-29
Passaic River near Chatham	41	43	24	22	0-82	0-79
PA10	1	0	99	100	0-1	0-0

Table 4-13: Distribution of diurnal DO swing during June 15- September 1, 2002 for TMDL scenario

	Mean (mg/L)		Standard deviation (mg/L)		90^{th} percentile (mg/L)	
Scenario / Location	Stage 2B	Stage 2	Stage 2B	Stage 2	Stage 2B	Stage 2
Dundee Lake	2.78	2.75	1.84	1.77	5.49	5.19
Peckman River mouth	5.40	5.40	1.90	1.94	7.88	7.95
Passaic River near Chatham	3.51	3.53	1.19	1.21	5.06	5.12
PA10	3.71	3.77	1.78	1.83	6.14	6.35

Table 4-14: Seasonal average of chl-*a* at Dundee Lake during WY2002 TMDL scenario

Scenario	Mean (µg/L)	Standard deviation (µg/L)	Probability of exceeding 20 µg/L (%)	90^{th} percentile (µg/L)
Stage 2B	16.6	13.0	30	33.9
Stage 2	17.3	17.0	27	34.4

Table 4-15: Results of left-tailed two-sample K-S tests of TMDL efficacy and Stage 1 output distributions

Location	Output distribution	p-value
Dundee Lake	DO daily average exceedance frequency	1.0
Peckman River mouth		1.0
Passaic River near Chatham		**0.01** [1]
PA10		1.0
Dundee Lake	Minimum DO exceedance frequency	0.98
Peckman River mouth		0.18
Passaic River near Chatham		0.59
PA10		1.0
Dundee Lake	Seasonal average of chl-*a*	**0.00**
Wanaque South intake	TP load diverted to Wanaque Reservoir, WY2002	**0.00**
Little Falls intake	Average TP concentration, WY2002	**0.00**
Dundee Lake	Diurnal DO swing	**0.00**
Peckman River mouth		**0.00**
Passaic River near Chatham		**0.00**
PA10		**0.00**

1. TMDL efficacy scenario distribution was significantly larger than actual condition; p-value shown is from right-tailed two-sample K-S test

Table 4-16: TP load diverted from WS Intake to Wanaque Reservoir during WY2002 for actual condition and TMDL efficacy scenarios

Scenario	Mean (kg)	Standard deviation (kg)	90^{th} percentile (kg)
Actual condition	39,508	2079	42,414
TMDL efficacy	9,190	628	10,160

Table 4-17: WY2002 average TP concentration at Little Falls intake for actual condition and TMDL efficacy scenarios

Scenario	Mean (mg/L)	Standard deviation (mg/L)	90^{th} percentile (mg/L)
Actual condition	0.54	0.05	0.61
TMDL efficacy	0.09	0.01	0.11

Table 4-18: WY2002 seasonal average of chl-*a* at Dundee Lake for actual condition and TMDL efficacy scenarios

Scenario	Mean (µg/L)	Standard deviation (µg/L)	Probability of exceeding 20 µg/L (%)	90^{th} percentile (µg/L)
Actual condition	73.0	35.3	97	115.8
TMDL efficacy	18.9	20.1	29	45.1

Table 4-19: Distribution of diurnal DO swing during June 15- September 1, 2002 for actual condition and TMDL efficacy scenarios

Location	Scenario	Mean (mg/L)	Standard deviation (mg/L)	90^{th} percentile (mg/L)
Dundee Lake	Actual conditions	7.45	2.78	10.71
	TMDL efficacy	2.27	1.34	4.06
Peckman River mouth	Actual conditions	7.63	2.99	11.43
	TMDL efficacy	5.43	2.15	8.31
Passaic River near Chatham	Actual conditions	4.43	1.80	6.75
	TMDL efficacy	3.54	1.41	5.40
PA10	Actual conditions	7.15	2.77	10.34
	TMDL efficacy	3.15	1.61	5.44

Table 4-20: Distribution of daily average DO exceedance frequencies during June 15- September 1, 2002 for actual condition and TMDL efficacy scenarios

Location	Scenario	Expected Exceedance (%)	Confidence of Compliance (%)	90% confidence interval
Dundee Lake	Actual conditions	0	100	0-0
	TMDL efficacy	0	100	0-0
Peckman River mouth	Actual conditions	0	100	0-0
	TMDL efficacy	0	100	0-0
Passaic River near Chatham	Actual conditions	22	52	0-74
	TMDL efficacy	35	29	1-79
PA10	Actual conditions	0	100	0-0
	TMDL efficacy	0	100	0-0

Table 4-21: Distribution of minimum DO exceedance frequencies during June 15- September 1, 2002 for actual condition and TMDL efficacy scenarios

Location	Scenario	Expected Exceedance (%)	Confidence of Compliance (%)	90% confidence interval
Dundee Lake	Actual conditions	0	100	0-0
	TMDL efficacy	0	100	0-0
Peckman River mouth	Actual conditions	10	75	0-57
	TMDL efficacy	4	82	0-20
Passaic River near Chatham	Actual conditions	53	9	0-82
	TMDL efficacy	51	18	0-82
PA10	Actual conditions	0	100	0-0
	TMDL efficacy	1	98	0-8

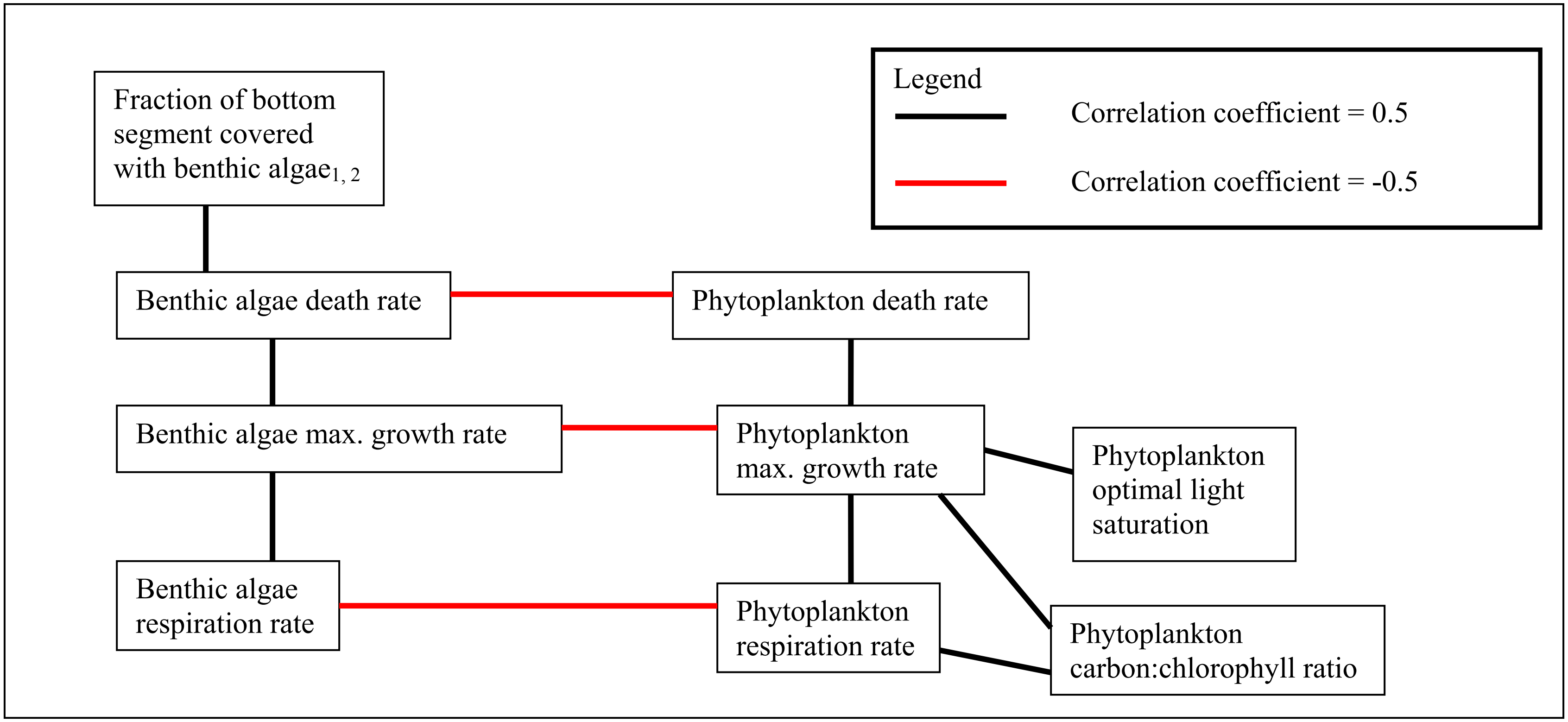


Figure 4-1: **Illustration of parameter covariance relationships assumed in Stage 2 of the uncertainty analysis**

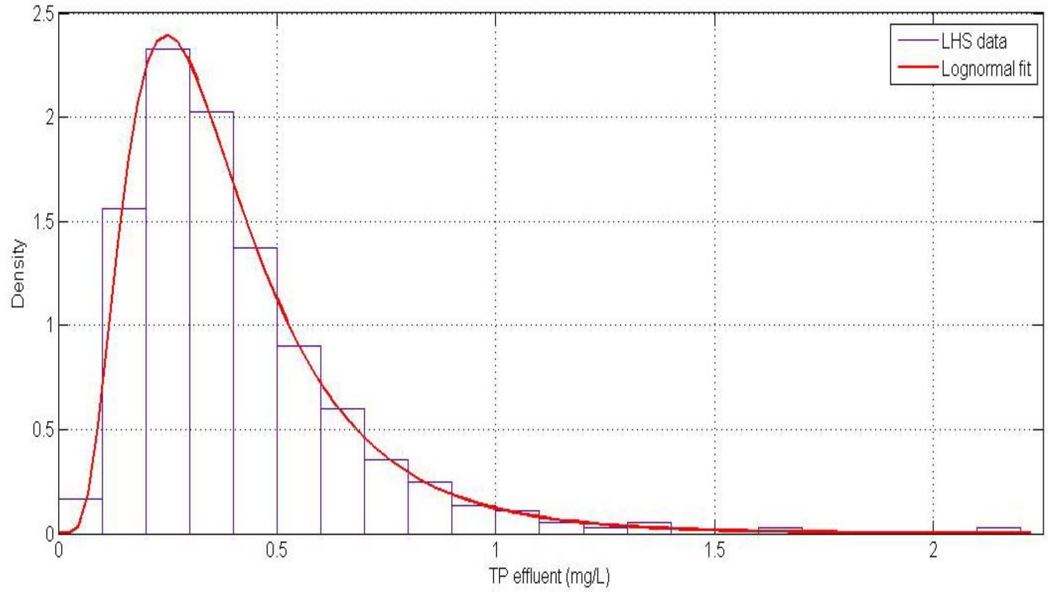


Figure 4-2: Probability distribution of TP effluent from a WWTP with ≥ 1.0 MGD permitted or average flow, as applied in Stage 2

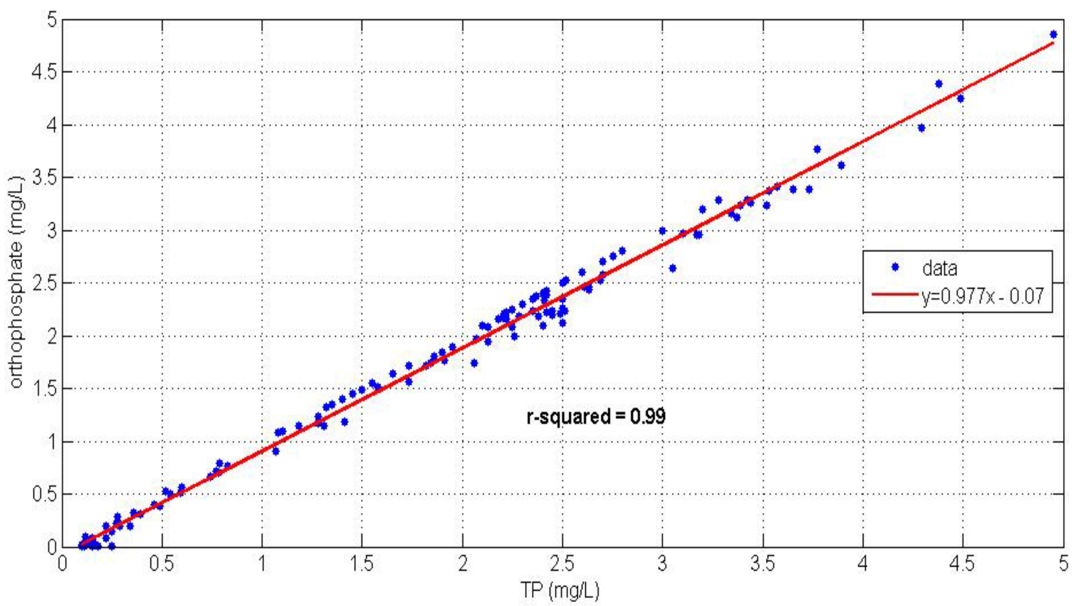


Figure 4-3: WWTP effluent data sampled by Omni Environmental (2007a), fitted with linear regression equation 4.1

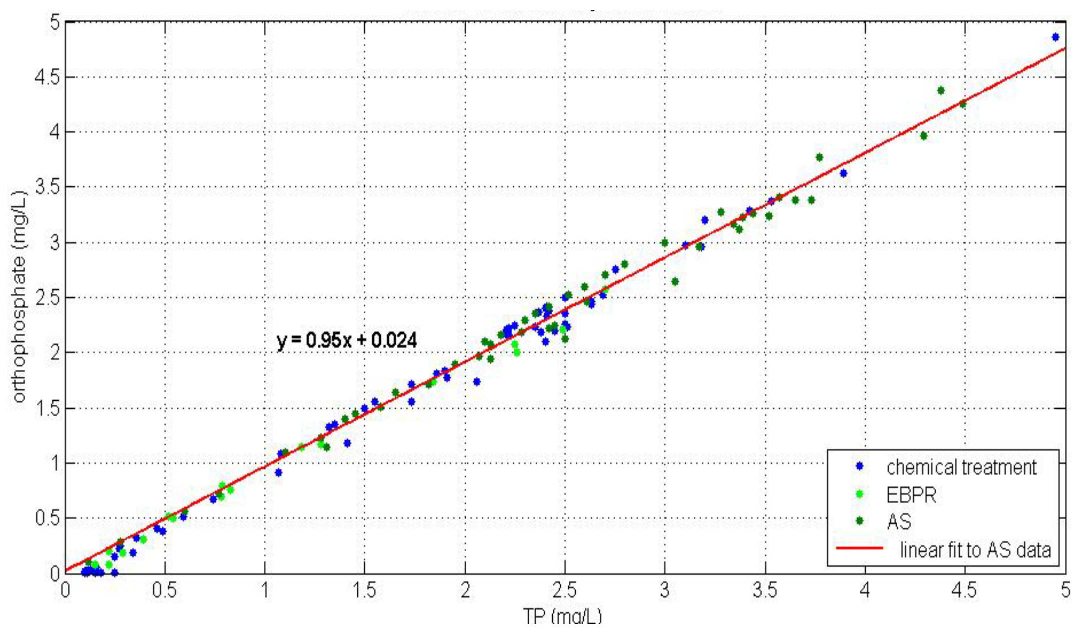


Figure 4-4: WWTP effluent data sampled by Omni Environmental (2007a), fitted with linear regression equation for Activated Sludge data only

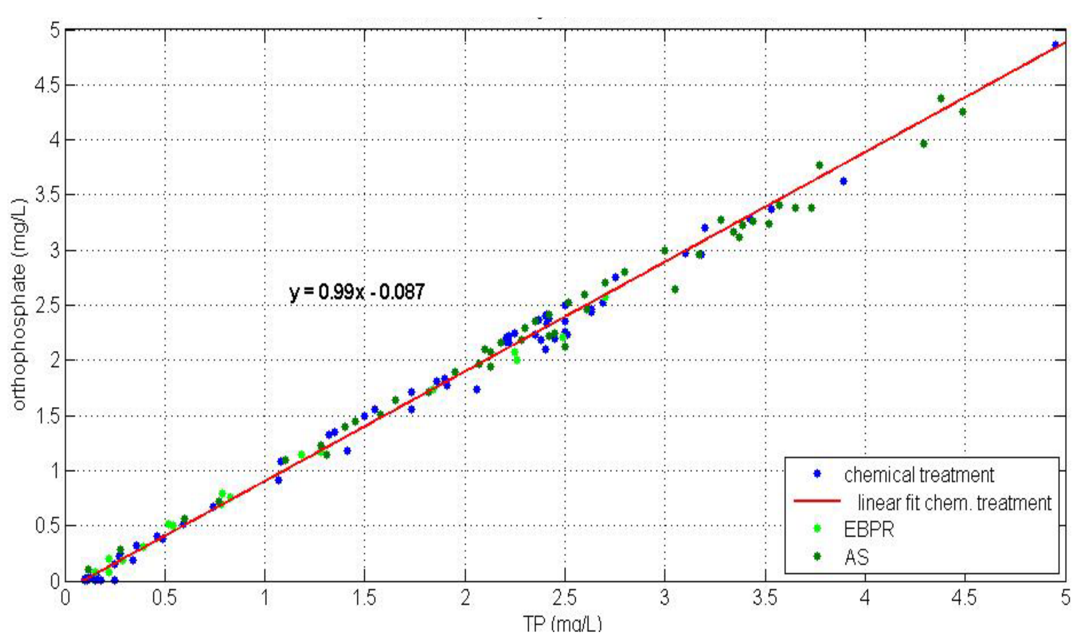


Figure 4-5: WWTP effluent data sampled by Omni Environmental (2007a), fitted with linear regression equation for chemical treatment data only

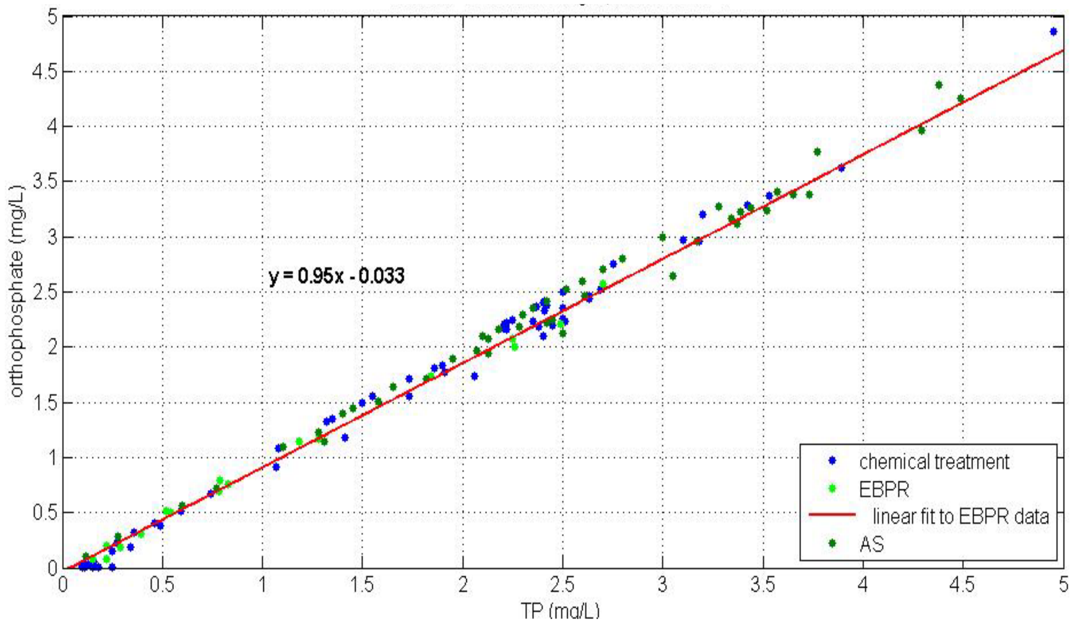


Figure 4-6: WWTP effluent data sampled by Omni Environmental (2007a), fitted with linear regression equation for EBPR data only

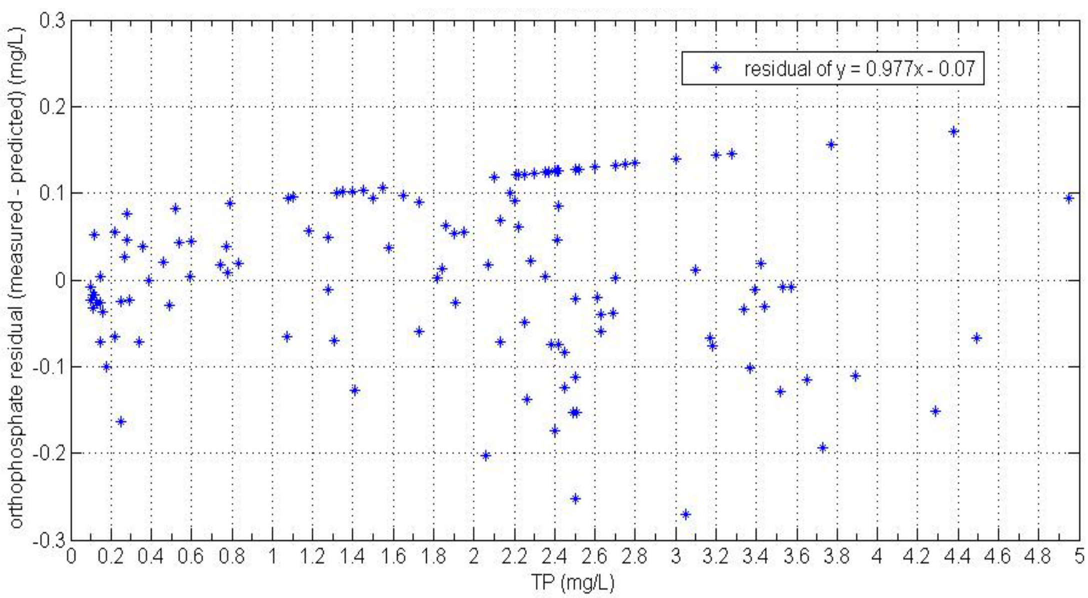


Figure 4-7: Residual plot of Equation 4.1

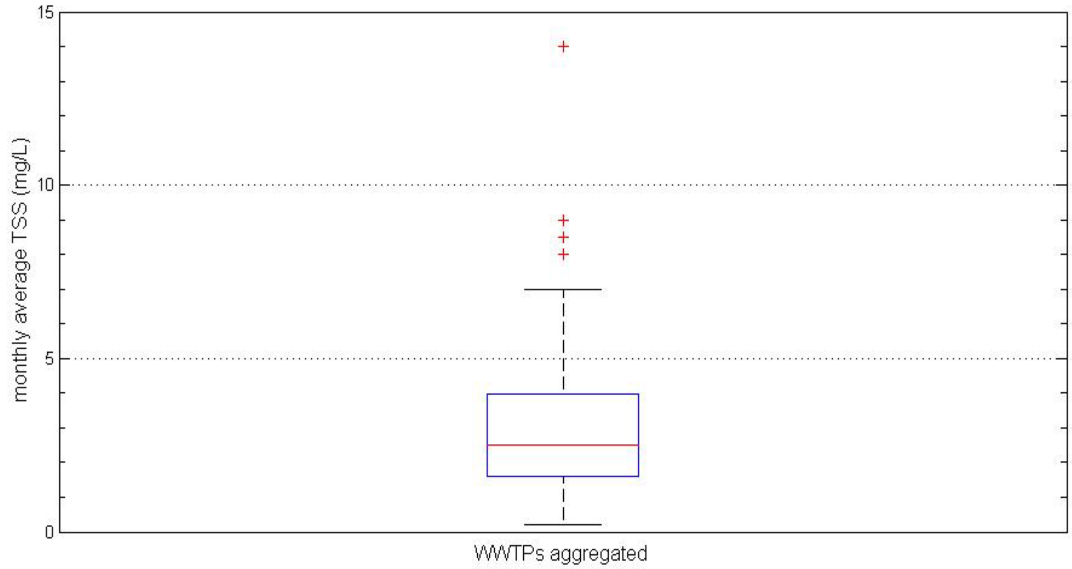


Figure 4-8: Boxplot of aggregated monthly average TSS effluent data in June-August and November 2003 from the 22 WWTPs analyzed in the study

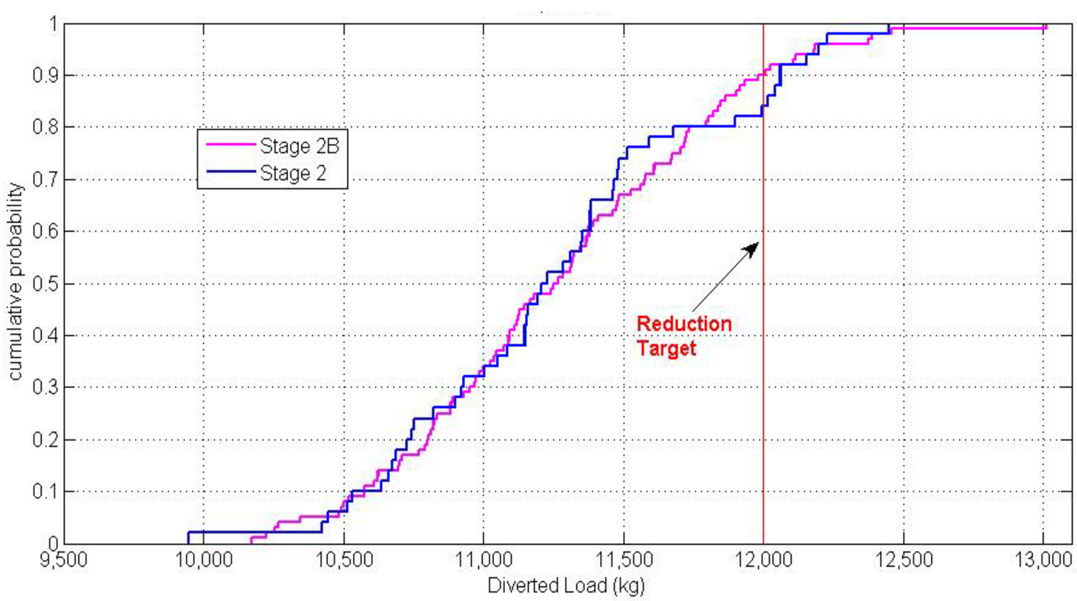


Figure 4-9: Stages 2 and 2B CDFs of WY2002 diverted TP load from WS Intake to Wanaque Reservoir in TMDL scenario

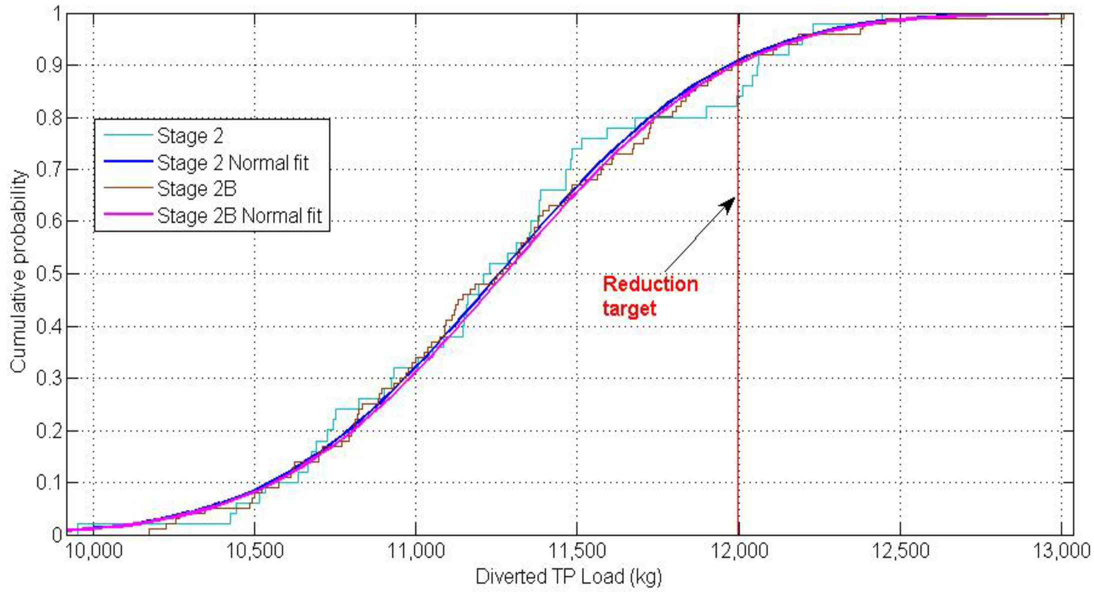


Figure 4-10: Stages 2 and 2B CDFs of diverted TP load from WS Intake to Wanaque Reservoir during WY2002 TMDL scenario, using Normal approximations of model outputs

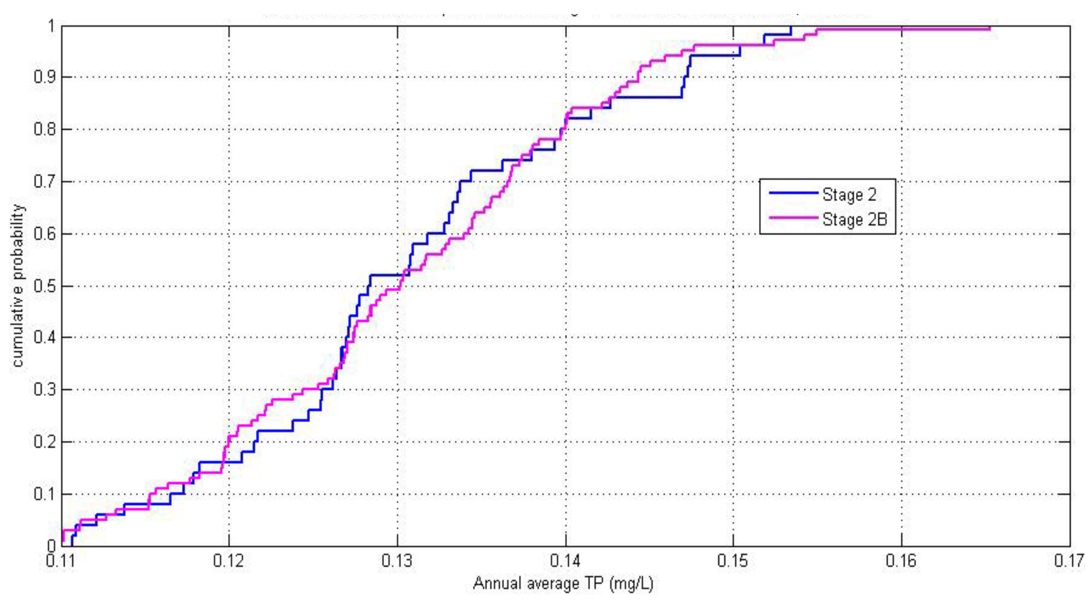


Figure 4-11: Stages 2 and 2B CDFs of average TP concentration at Little Falls intake during WY2002 TMDL scenario

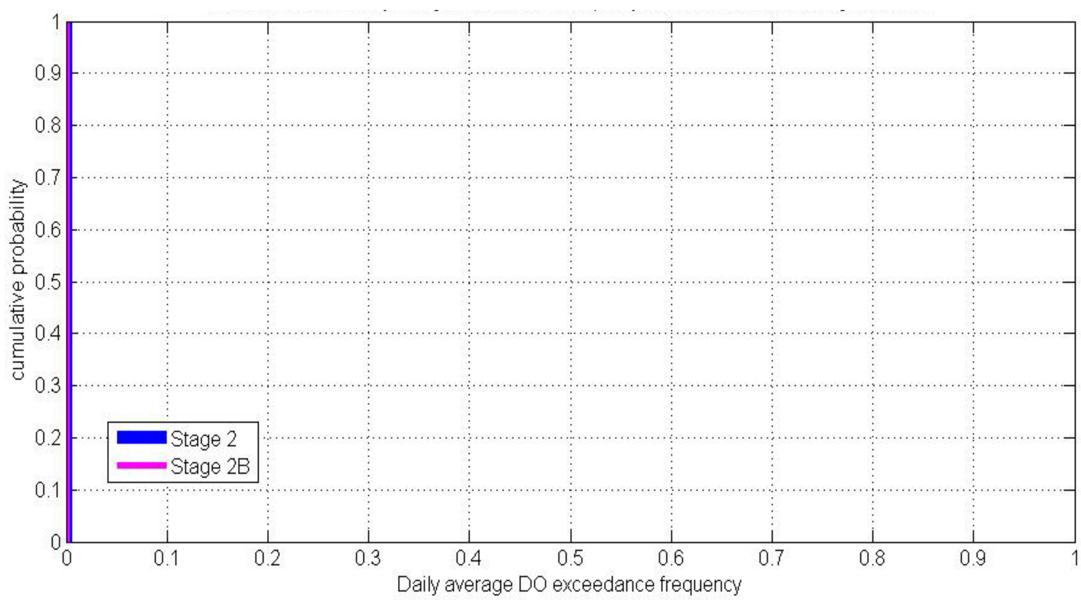


Figure 4-12: Stages 2 and 2B CDFs of June 15-Sep. 1, 2002 daily average DO exceedance frequency at Dundee Lake, Peckman River mouth and station PA10 in TMDL scenario

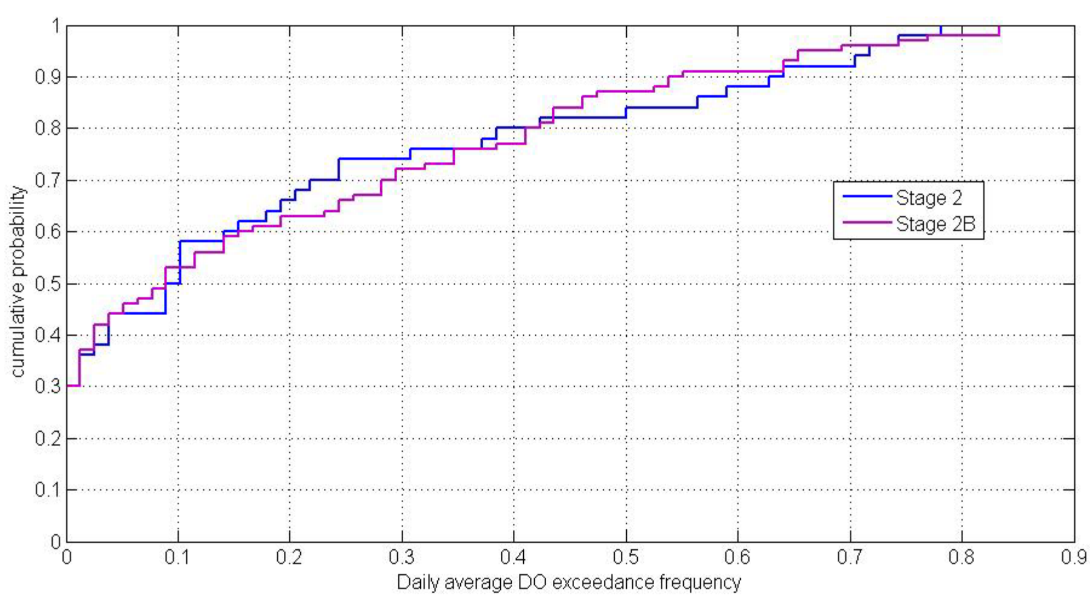


Figure 4-13: Stages 2 and 2B CDFs of June 15- Sep. 1, 2002 daily average DO exceedance frequency at Passaic River near Chatham in TMDL scenario

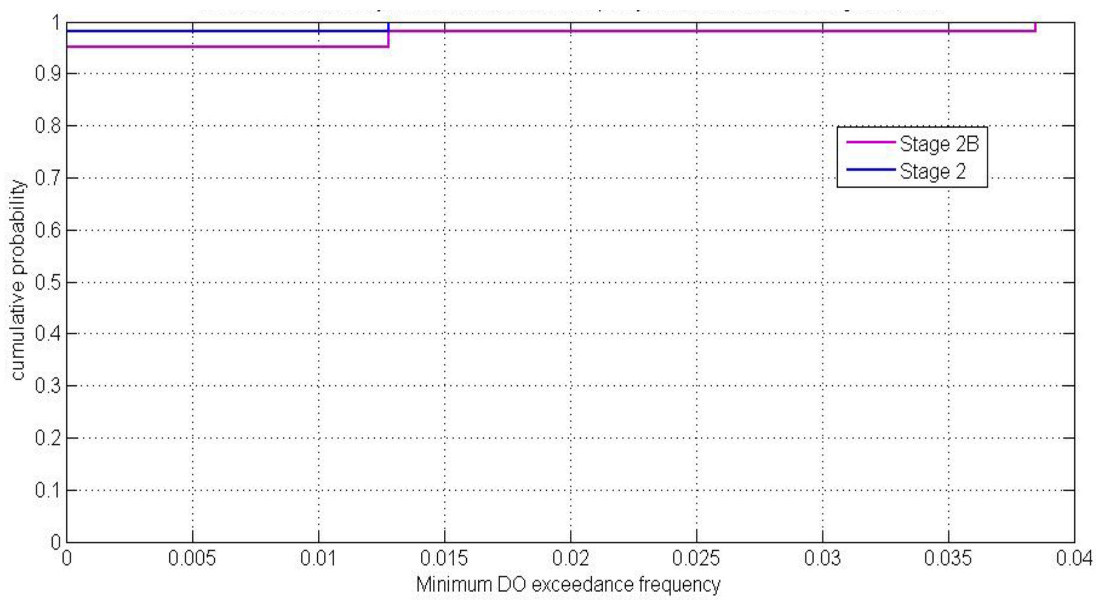


Figure 4-14: Stages 2 and 2B CDFs of June 15- Sep. 1, 2002 minimum DO exceedance frequency at Dundee Lake in TMDL scenario

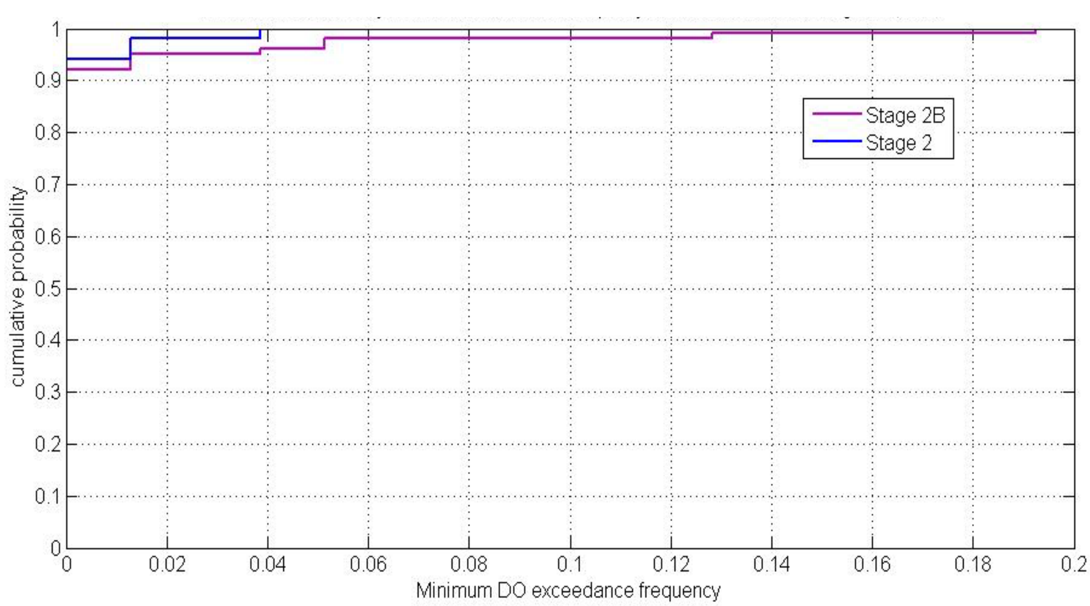


Figure 4-15: Stages 2 and 2B CDFs of June 15- Sep. 1, 2002 minimum DO exceedance frequency at station PA10 in TMDL scenario

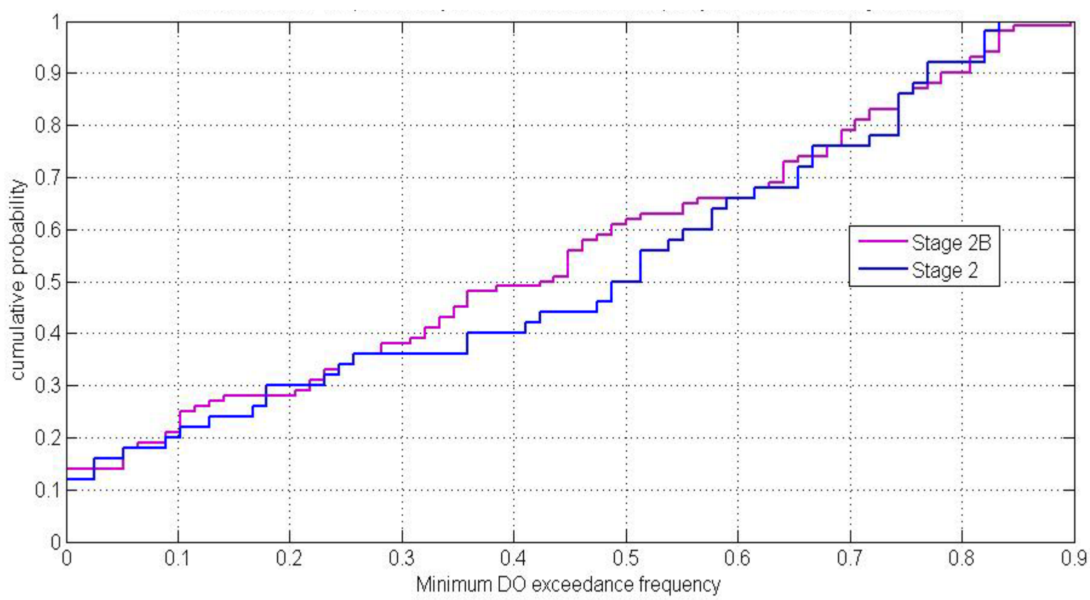


Figure 4-16: Stages 2 and 2B CDFs of June 15- Sep. 1, 2002 minimum DO exceedance frequency at Passaic River near Chatham in TMDL scenario

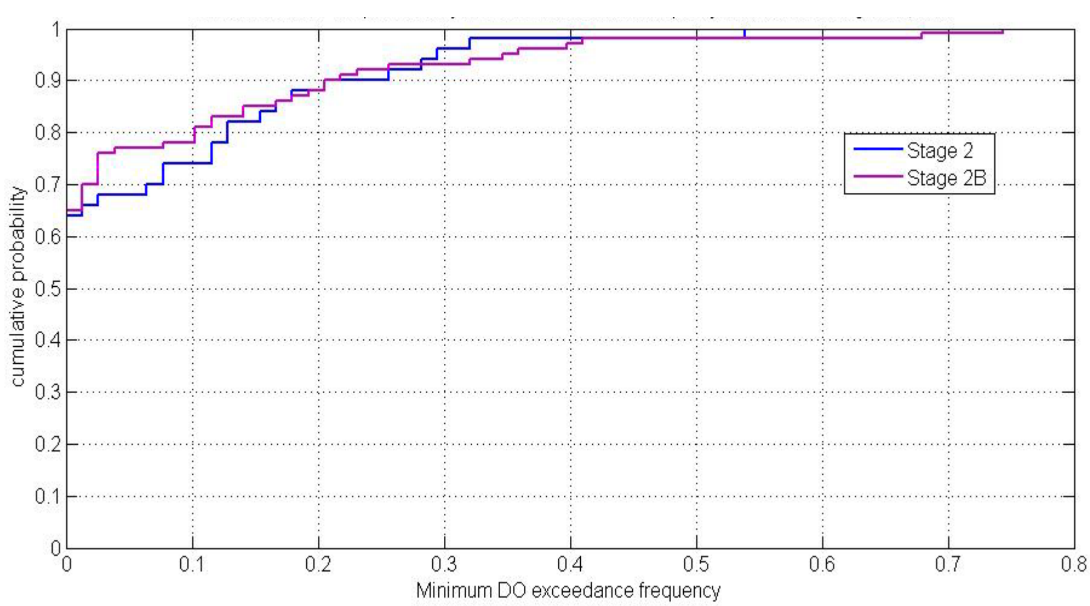


Figure 4-17: Stages 2 and 2B CDFs of June 15- Sep. 1, 2002 minimum DO exceedance frequency at Peckman River mouth in TMDL scenario

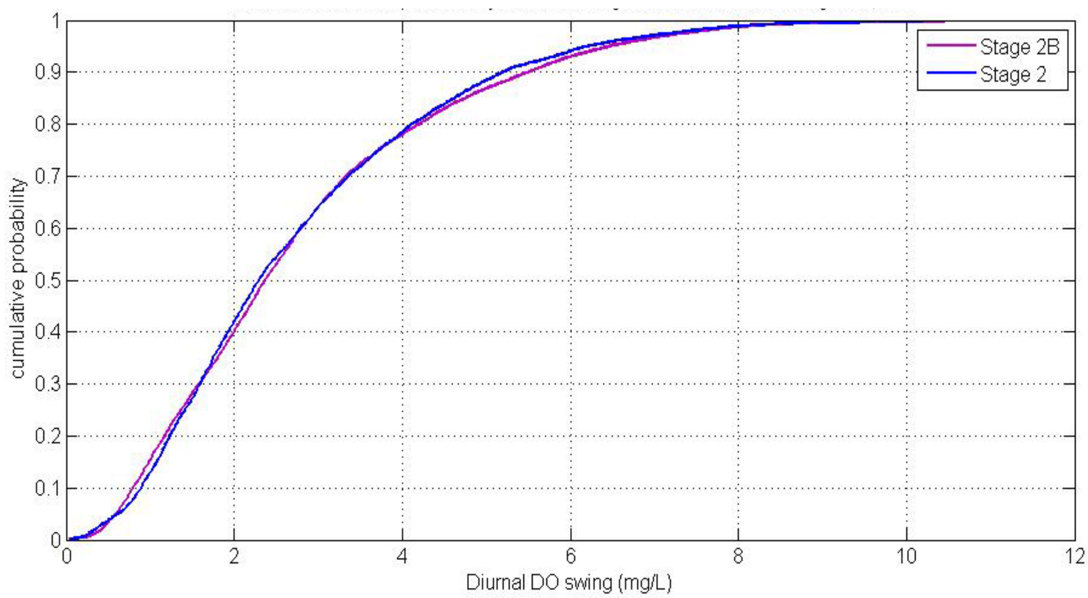


Figure 4-18: Stages 2 and 2B CDFs of diurnal DO swing from June 15- Sep. 1, 2002 at Dundee Lake in TMDL scenario

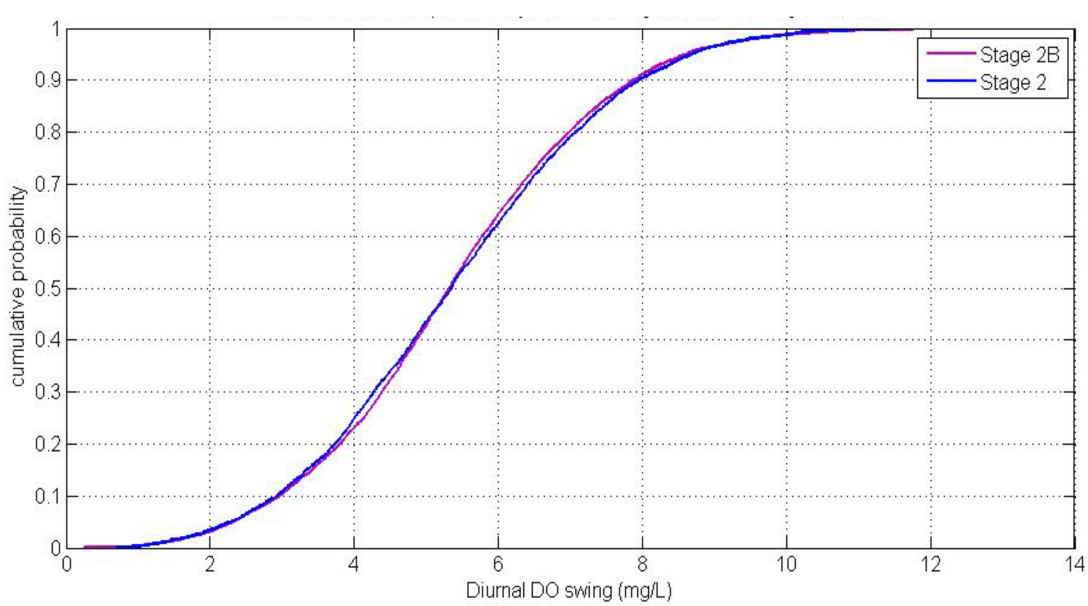


Figure 4-19: Stages 2 and 2B CDFs of diurnal DO swing from June 15- Sep. 1, 2002 at Peckman River mouth in TMDL scenario

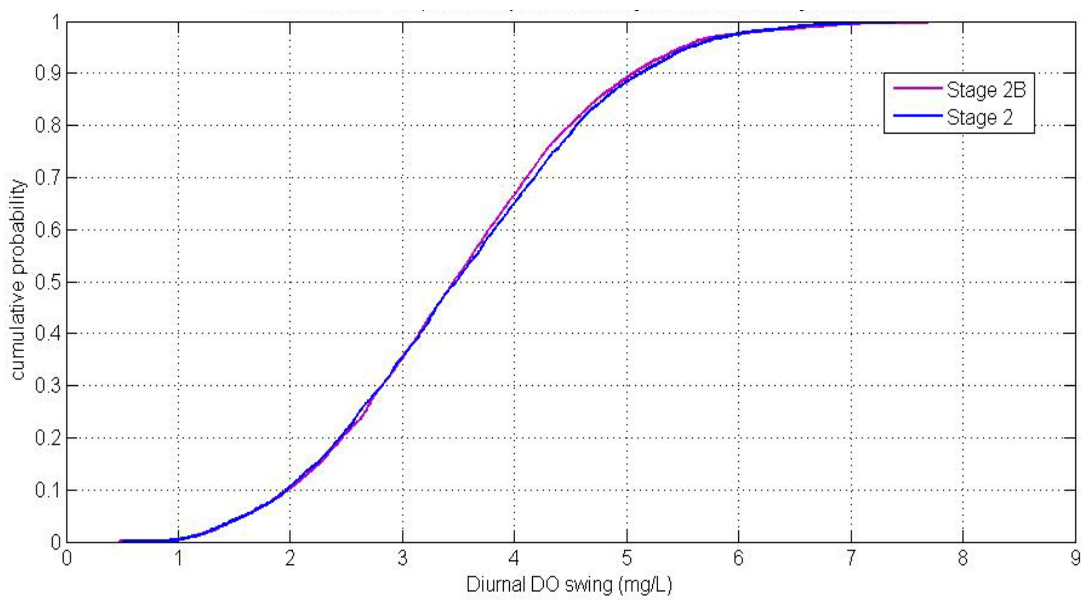


Figure 4-20: Stages 2 and 2B CDFs of diurnal DO swing from June 15- Sep. 1, 2002 at Passaic River near Chatham in TMDL scenario

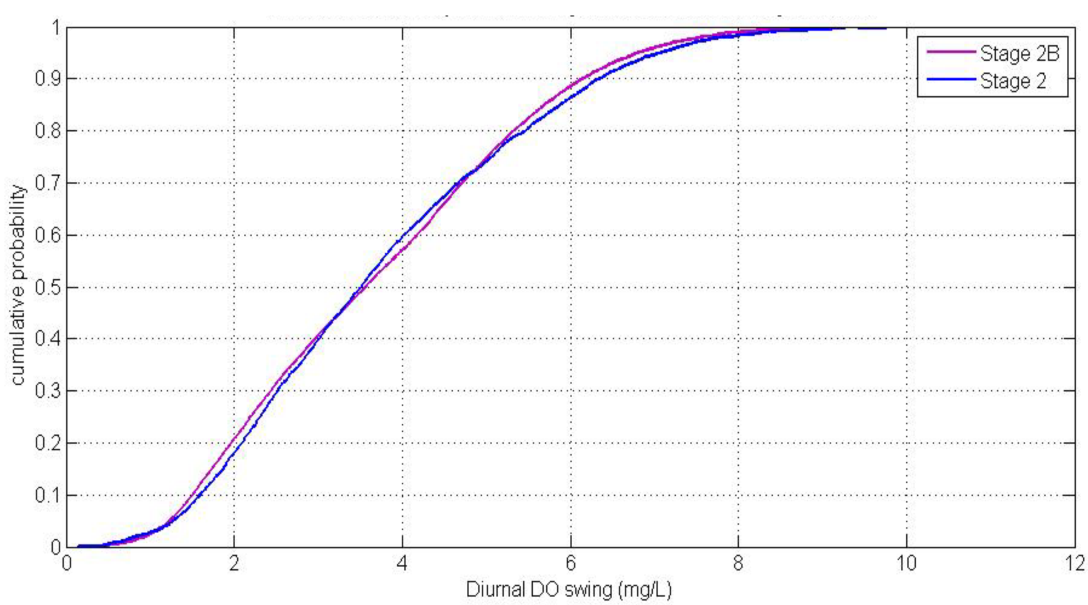


Figure 4-21: Stages 2 and 2B CDFs of diurnal DO swing from June 15- Sep. 1, 2002 at station PA10 in TMDL scenario

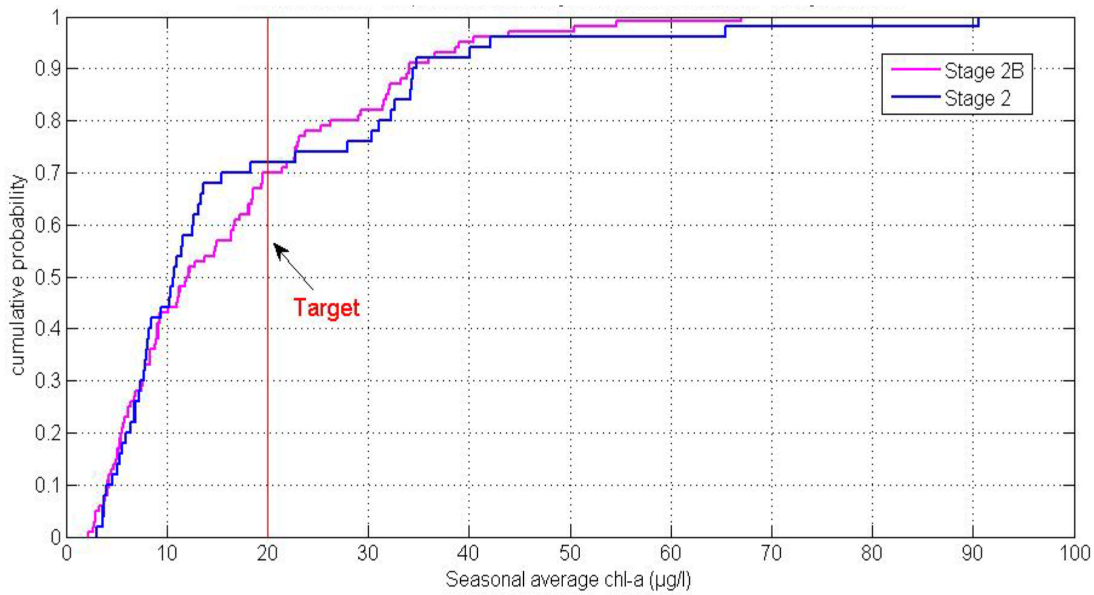


Figure 4-22: Stages 2 and 2B CDFs of WY2002 seasonal average chl-*a* concentration at Dundee Lake in TMDL scenario

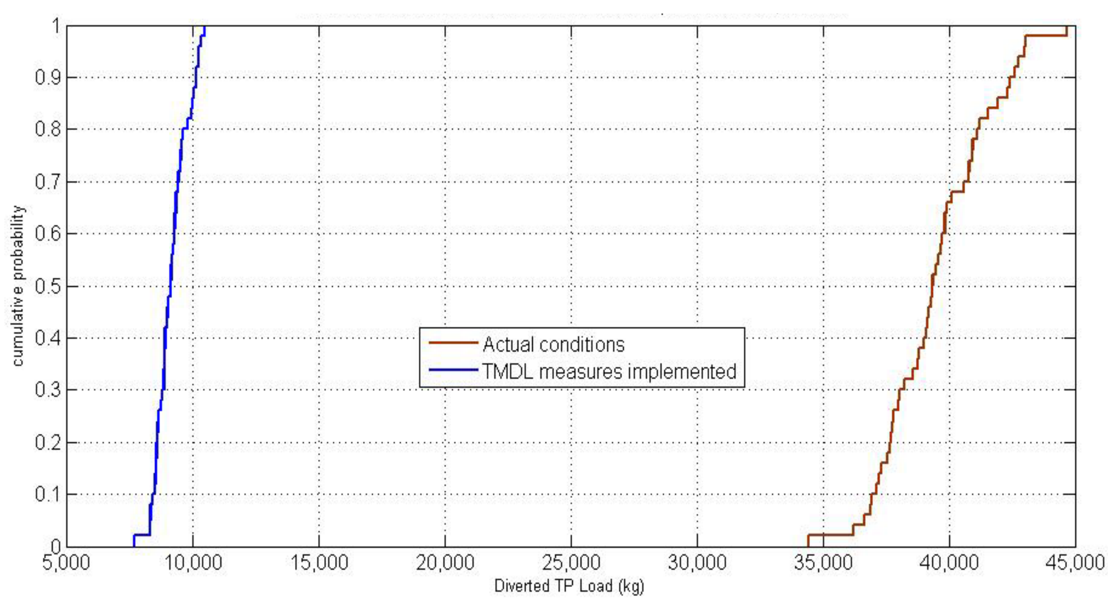


Figure 4-23: CDFs of WY2002 diverted TP load from WS Intake to Wanaque Reservoir in actual condition and TMDL efficacy scenarios

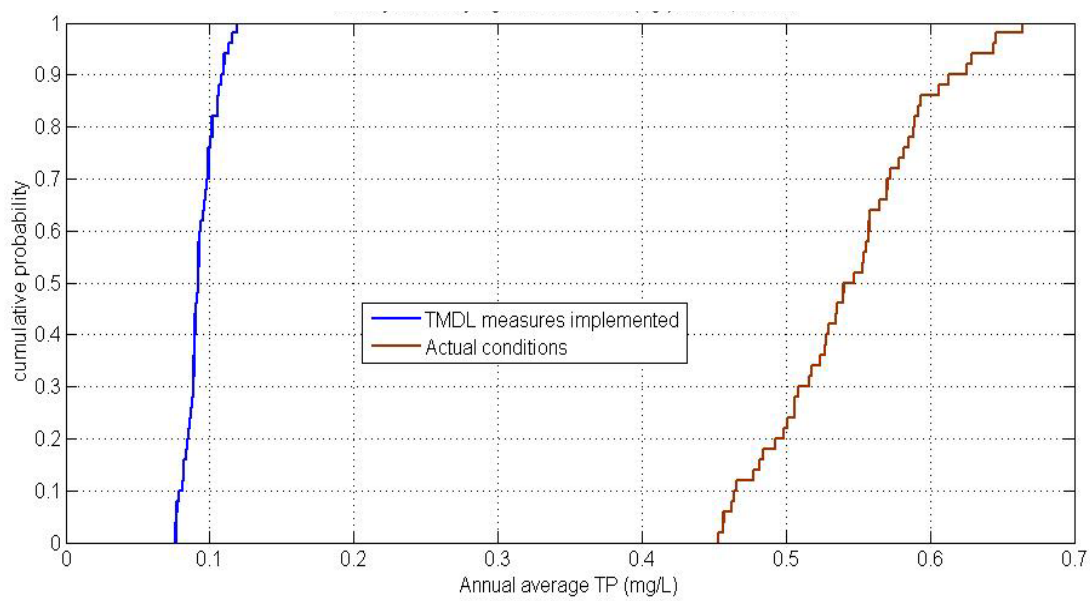


Figure 4-24: CDFs of WY2002 average TP concentration at Little Falls intake in actual condition and TMDL efficacy scenarios

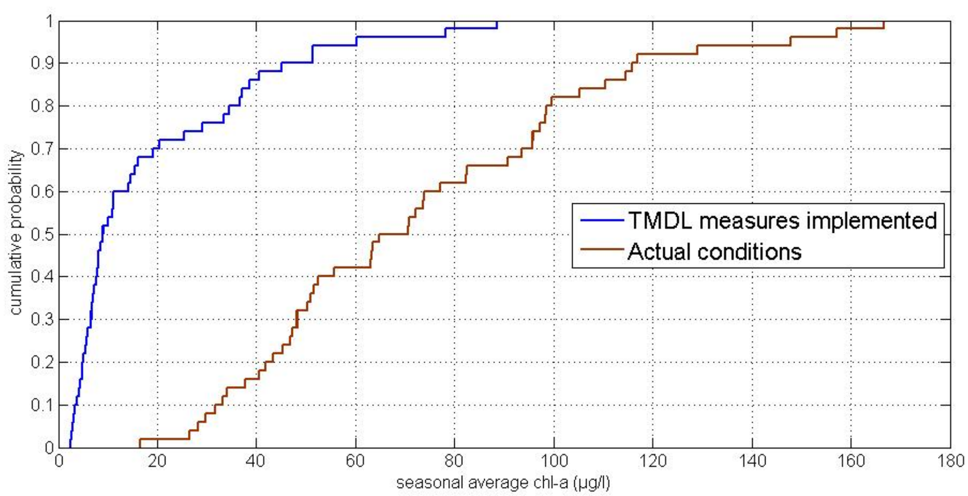


Figure 4-25: CDFs of WY2002 seasonal average of chl-*a* at Dundee Lake in actual condition and TMDL efficacy scenarios

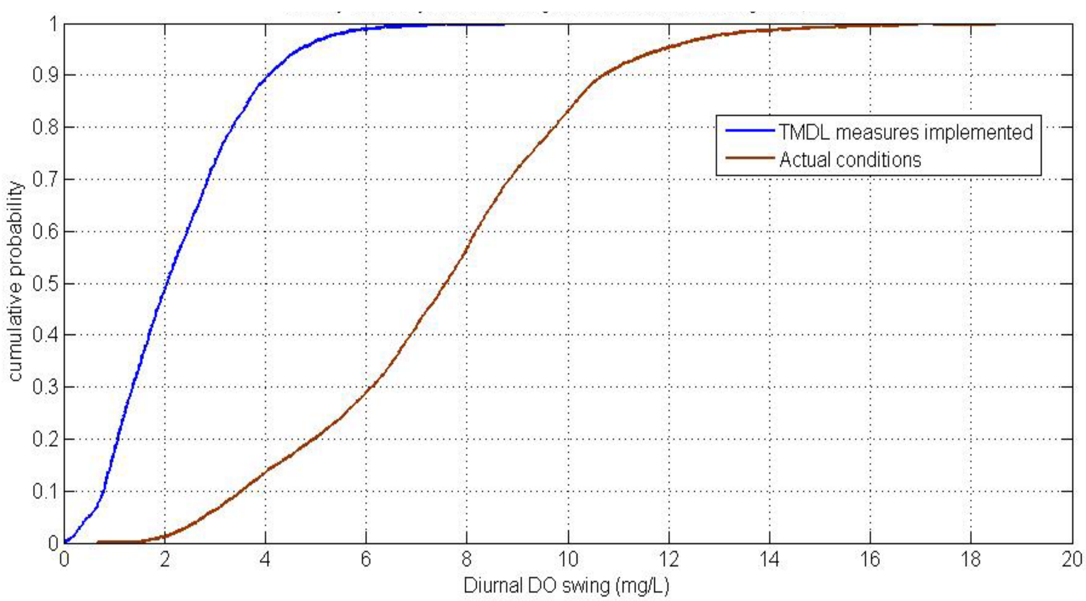


Figure 4-26: CDFs of diurnal DO swing from June 15- Sep. 1, 2002 at Dundee Lake in actual condition and TMDL efficacy scenarios

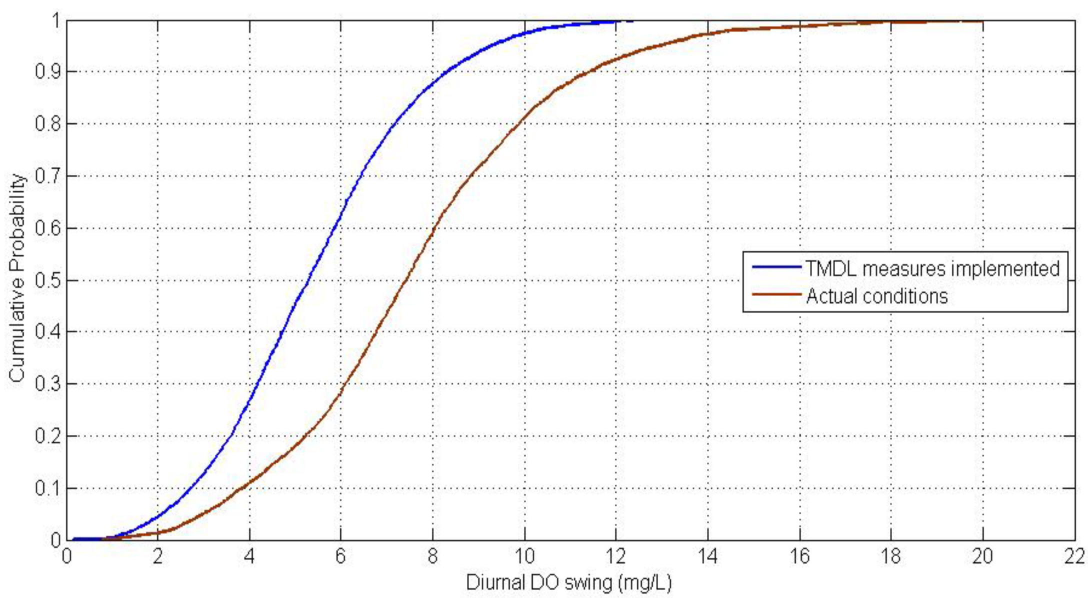


Figure 4-27: CDFs of diurnal DO swing from June 15- Sep. 1, 2002 at Peckman River mouth in actual condition and TMDL efficacy scenarios

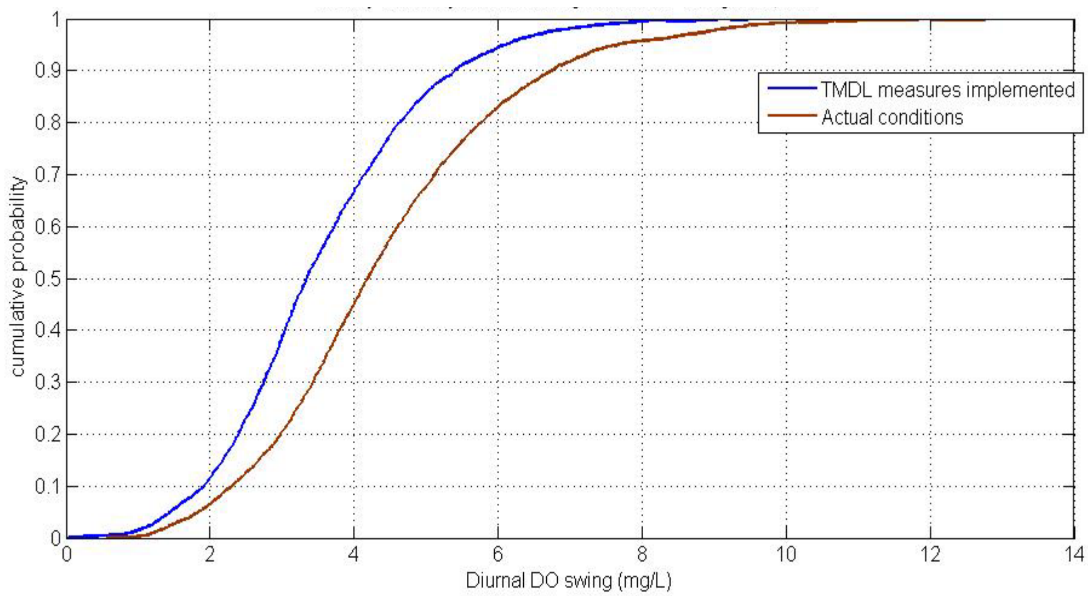


Figure 4-28: CDFs of diurnal DO swing from June 15- Sep. 1, 2002 at Passaic River near Chatham in actual condition and TMDL efficacy scenarios

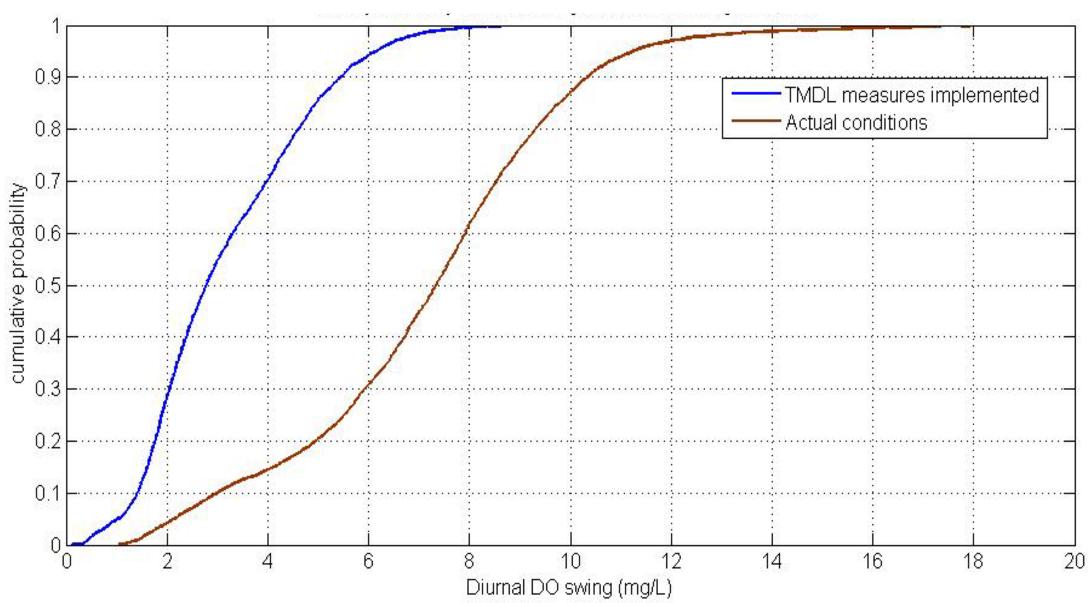


Figure 4-29: CDFs of diurnal DO swing from June 15- Sep. 1, 2002 at station PA10 in actual condition and TMDL efficacy scenarios

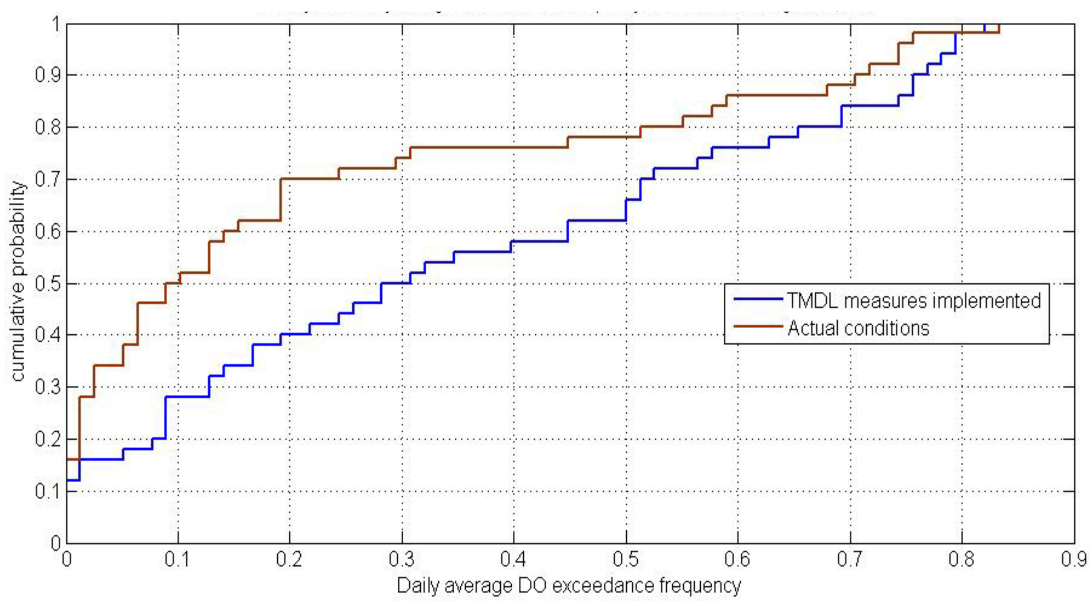


Figure 4-30: CDFs of daily average DO exceedance frequency from June 15- Sep. 1, 2002 at Passaic River near Chatham in actual condition and TMDL efficacy scenarios

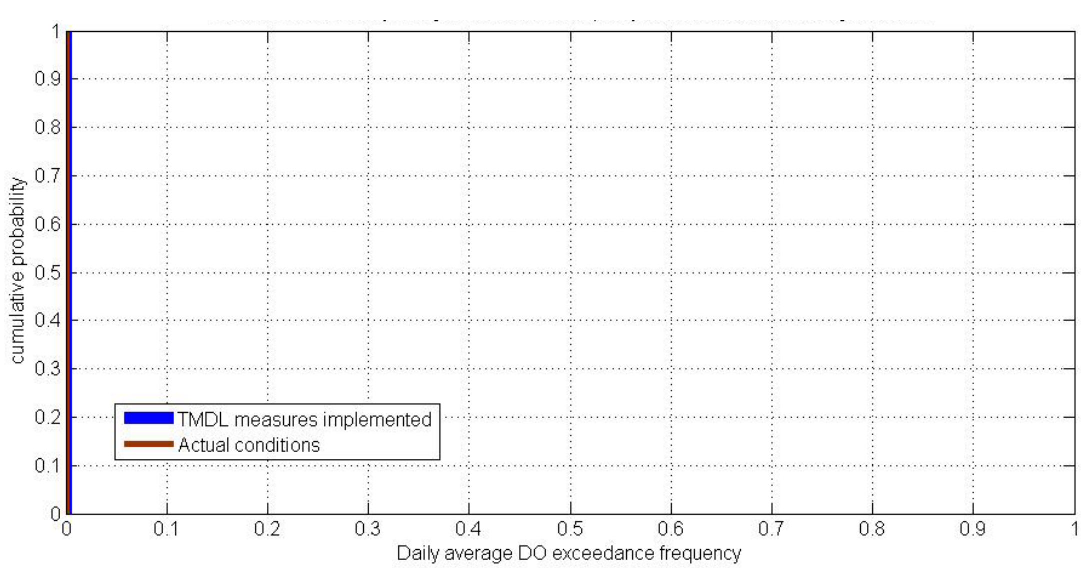


Figure 4-31: CDFs of daily average DO exceedance frequency from June 15- Sep. 1, 2002 at Dundee Lake, Peckman River mouth and station PA10 in actual condition and TMDL efficacy scenarios

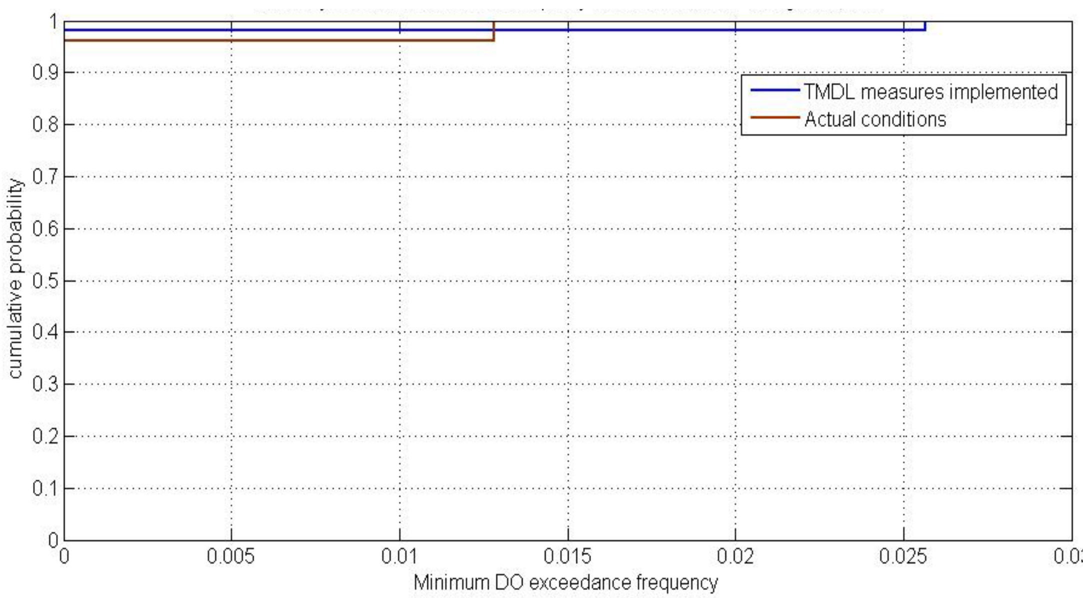


Figure 4-32: CDFs of minimum DO exceedance frequency from June 15- Sep. 1, 2002 at Dundee Lake in actual condition and TMDL efficacy scenarios

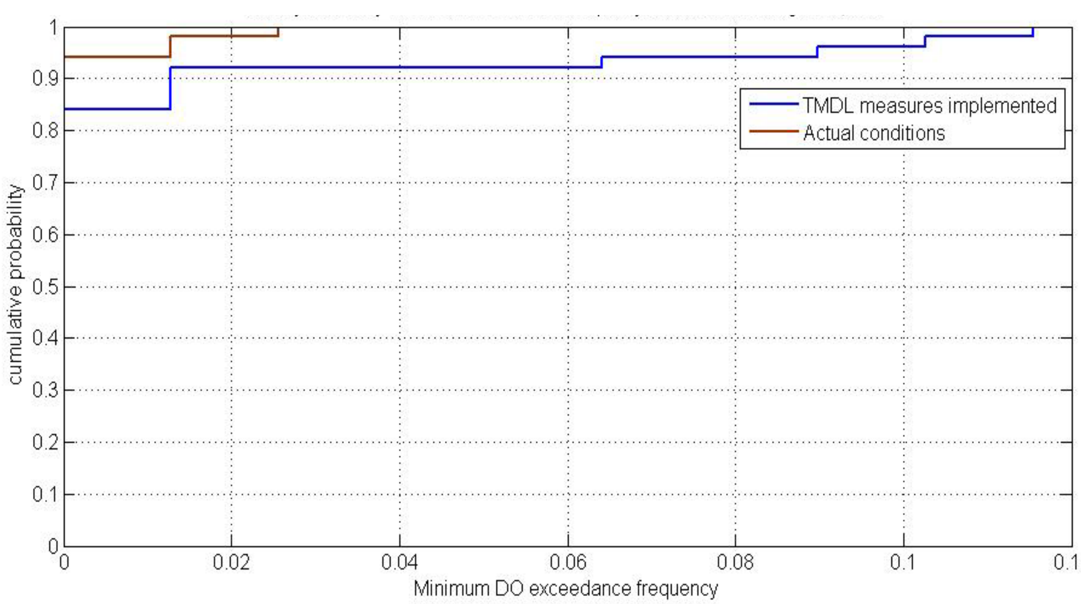


Figure 4-33: CDFs of minimum DO exceedance frequency from June 15- Sep. 1, 2002 at station PA10 in actual condition and TMDL efficacy scenarios

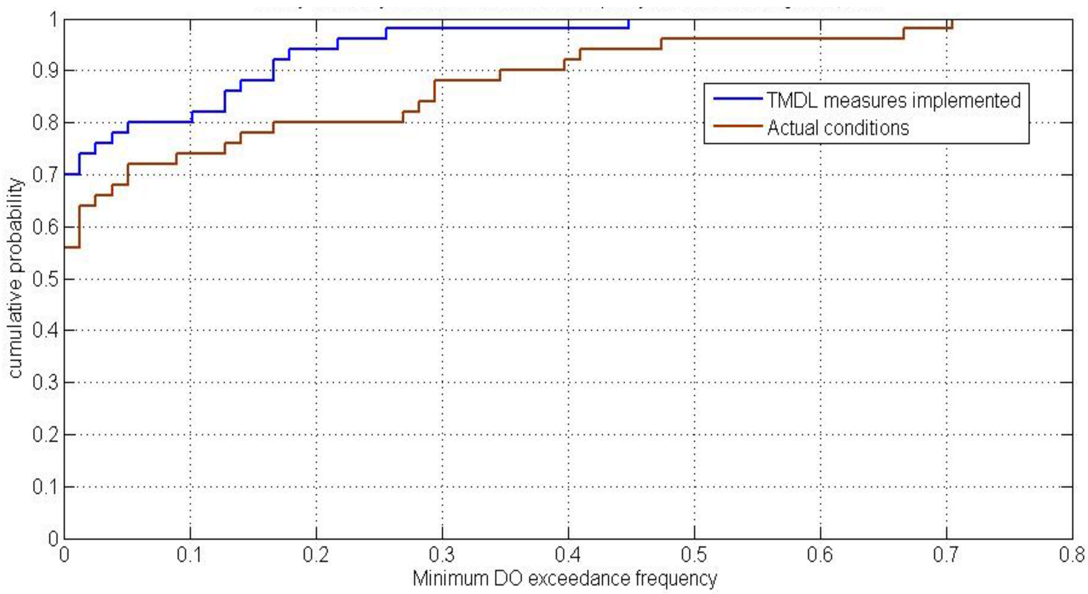


Figure 4-34: CDFs of minimum DO exceedance frequency from June 15- Sep. 1, 2002 at Peckman River mouth in actual condition and TMDL efficacy scenarios

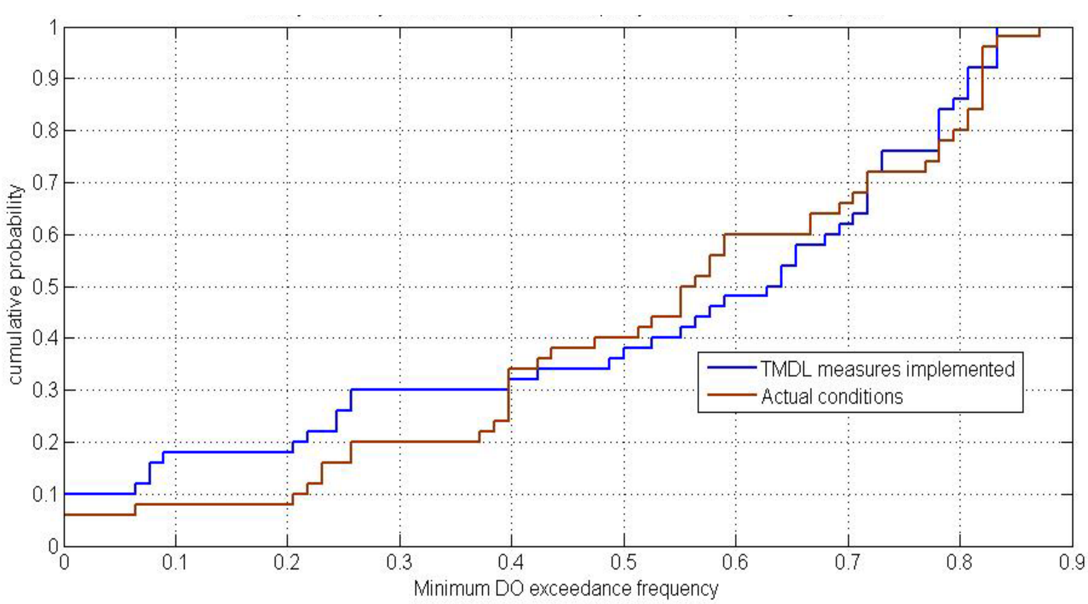


Figure 4-35: CDFs of minimum DO exceedance frequency from June 15- Sep. 1, 2002 at Passaic River near Chatham in actual condition and TMDL efficacy scenarios

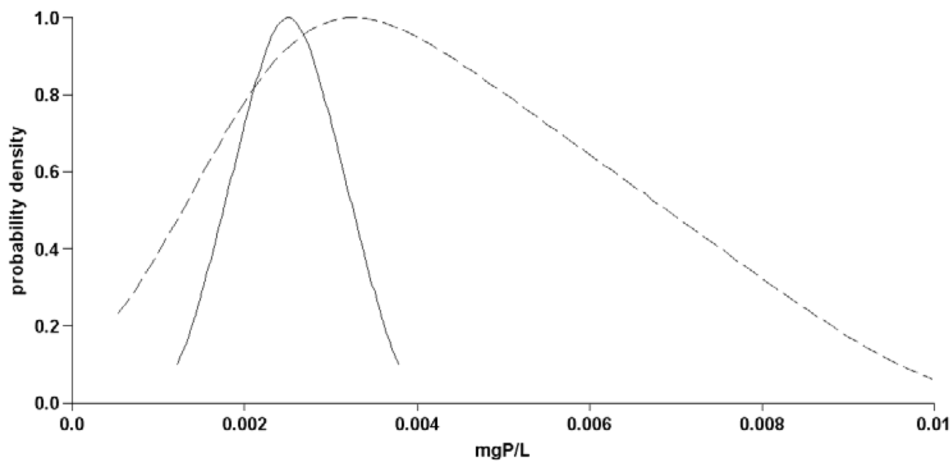


Figure 4-36: Comparison of PDFs for Phytoplankton phosphorus half-saturation constant for growth. The wider distribution was applied in Stage 1, but the narrower distribution is suggested as more reasonable based on Stage 2 findings.

Chapter 5: Uncertainty analysis of water quality trading scenarios (Stage 3)

5.1 Introduction

Stage 3 focused on the uncertainty of water quality trading scenarios at a variety of hydrologic conditions. In total, 5 trading scenarios and 3 corresponding baseline scenarios were simulated. The objective of Stage 3 was to test the third research hypothesis:

The outcome of trades between WWTPs, as compared with command and control regulation, will significantly increase uncertainty in the attainment of dissolved oxygen surface water quality standards, site-specific chlorophyll-*a* criteria, and reduction targets for diverted total phosphorus load at affected potential hot spots in the watershed.

5.2 Background information on the Passaic WQT program

The information in this section is drawn largely from Passaic Trading Project (2009). Refer to that source for more details on the development of the Passaic WQT program.

5.2.1 Hot spots

The main water quality concern with trading is that trades will lead to hot spots. Hot spots are localized areas with unacceptably high pollutant levels. Hot spots are particularly a concern downstream of a buyer. The main potential hot spots in the watershed are the two TMDL critical locations (NJDEP, 2008a) – the Wanaque Reservoir and Dundee Lake. Due to the diversion infrastructure, the WS intake is a surrogate hot spot for the Wanaque Reservoir. The areas of concern have lower susceptibility to excess phosphorus than the critical locations (Omni Environmental, 2007a). They include station PA10, the Peckman River mouth, the Passaic River near Chatham and the Little Falls intake (Fig. 5-1). All other locations in the watershed were shown in Omni Environmental (2007a) to have negligible DO impacts from reduced TP loading and are not considered potential hot spots.

The Passaic trading program was designed to specifically achieve compliance with TMDL water quality targets at the two critical locations. Compared to a command and control approach that prohibits trading, fixed-value model simulations of several trading scenarios all showed equal or better results at the 2 critical locations, and equal or only slightly lesser results at the areas of concern (Omni Environmental, 2007c). However, those trade scenario simulations did not account for model uncertainty. Stage 3 used uncertainty analysis to determine if trading significantly increases the uncertainty of key water quality outcomes at each of the above locations, relative to a command and control approach that prohibits trading.

5.2.2 Diversion conditions

Due to fluctuations in precipitation and demand for drinking water from the Wanaque Reservoir, three potential surface water diversion scenarios can occur with respect to the WS intake. These scenarios, termed “no diversion”, “diversion” and “extreme diversion”, are explained below.

In the “no diversion” scenario, the WS intake is not activated, thus the Wanaque Reservoir does not receive any phosphorus loads from the 22 main WWTPs in the watershed. All 22 WWTPs impact only Dundee Lake. (The “no diversion” scenario is estimated to have occurred on 63 percent of the days from October 1, 1999 through Nov 30, 2003 (Omni Environmental, 2007b)).

In the “diversion” scenario, the WS intake is operating under conditions where the Pompton River flow is enough to meet the pumping demand. The WS intake only diverts surface water to the Wanaque Reservoir from the Pompton River, and diverts nothing from the Upper Passaic River. (The “diversion” scenario is estimated to have occurred on 19 percent of the days from October 1, 1999 through Nov 30, 2003 (Omni Environmental, 2007b)). Therefore, Pompton River dischargers impact both the Wanaque Reservoir and Dundee Lake, while dischargers from the Upper and Lower Passaic River drainage areas only impact Dundee Lake.

In the “extreme diversion” scenario, the WS intake is operating under conditions where the Pompton River flow is not enough to meet the pumping demand. Flow from both the Pompton and Upper Passaic Rivers is diverted. (The “extreme diversion” scenario is estimated to have occurred on 19 percent of the days from October 1, 1999 through Nov 30, 2003 (Omni Environmental, 2007b)). Therefore, Pompton River dischargers impact only the Wanaque Reservoir, Upper Passaic River drainage area dischargers impact both the Wanaque Reservoir and Dundee Lake, and dischargers from the Lower Passaic River drainage area impact only Dundee Lake.

5.2.3 Passaic trading framework

Passaic phosphorus trading is bound by two constraints: to achieve compliance with TMDL water quality targets i) at two critical locations (Wanaque Reservoir and Dundee Lake), and ii) under all diversion conditions. The Passaic trading framework was designed to carry out trading under these constraints (Obropta et al., 2008). In total, three management areas (MAs) are delineated: the Pompton MA, the Upper Passaic MA and the Lower Passaic MA. All intra-MA trades are allowed. The acceptability of inter-MA trades depends on the ability to protect both critical locations under all diversion conditions. The end result is reflected in Table 5-1. Of the six possible inter-MA trades, three are allowed and three are not allowed. The Lower Passaic MA can buy from but not sell to the other MAs. Conversely, the Pompton MA can sell to but not buy from the other MAs. Consequently, the Upper Passaic MA can buy from the Pompton MA and sell to the Lower Passaic MA.

5.2.4 Trading ratios

Trading ratios function like an exchange rate to equalize the load traded between WWTPs. A trading ratio is necessary to equalize the load exchanged relative to the shared TMDL critical location. This is necessary to make sure trades have the same net impact as if both plants discharged at their allocated levels. The trading ratio is based on the relative attenuation of phosphorus discharged from two point sources toward a shared TMDL critical location. A trading ratio of 0.9 means that 1000 kg abated by the seller has the same effect at the TMDL critical location as 900 kg abated by the buyer, therefore the seller's 1000 kg is worth 900 kg to the buyer.

An analysis by Omni Environmental (2007d) determined attenuation rates of TP load discharged from each WWTP to both the WS intake and Dundee Lake, under each diversion condition. Trading ratios were then calculated for each pair of buyer and seller based on relative attenuation rates with respect to the critical locations; trading ratios were also calculated at each diversion condition, yielding a total of up to 6 trading ratios for each pair of buyer and seller. Lower trading ratios are more conservative. The minimum trading ratio, further reduced by 10% as a margin of safety, was chosen as the final trading ratio to be applied under all diversion conditions for the sake of simplicity, low transaction costs, and being conservative.

A trade between an eligible buyer-seller pair can occur under any of the three diversion conditions. The critical diversion condition is the diversion condition that yields the lowest (i.e., most conservative) trading ratio for a particular pair of buyer and seller. It varies for each pair of buyers and sellers. For most trades, the extreme diversion is the critical condition, but for other trades either the diversion or no diversion is the critical condition. Passaic trading ratios were designed to be conservative and function under the most critical diversion condition for each pair of buyers and sellers.

5.2.5 Trading formula

The trading program is based on attaining the net effect of LTA 0.4 mg/L TP from each discharger on an annual basis, while realizing that transactions must occur in units of mass rather than concentration for trading to be viable. To achieve these objectives, actual discharger flow rather than permitted flow is the basis for determining load allocations.

From a practical standpoint, allocations must be known prior to making trades. This facilitates the establishment of property rights, an essential precursor for a successful trading program (Boisvert et al., 2007). Therefore allocations in the Passaic trading program are based on a recent history of actual discharger flow, termed "anticipated actual discharger flow". The latter term is defined as the average of the three most recent years of actual discharger flow. The allocation for each WWTP is the product of 0.4 mg/L TP and the Anticipated Actual Discharger Flow.

The trading formula is:

Balance = Allocation – Load Discharged - Actual load sold + Equalized load purchased

(Eq. 5.1)

Where

Allocation = (0.4 mg/L LTA · Anticipated Actual Discharger Flow),

Load Discharged = (Load discharged),

Actual load sold = (Load below allocation that seller removed from wastewater and sold),

and

Equalized load purchased = (Actual load sold · Trading ratio $_{\text{seller to buyer}}$).

Note that in a given year, if a WWTP effluent flow is less than its anticipated actual discharger flow, then the WWTP could technically discharge a proportionate amount greater than 0.4 mg/L and still meet its allocation. This possibility is referred to as "trading at 'dry year' effluent flows" and is a critical condition that was analyzed. Accordingly, trading scenarios were simulated at "dry year" effluent flows.

5.3 Methodology

5.3.1 Trade scenario selection

In the study that reported on development and validation of the Passaic trading program (Passaic Trading Project, 2009), 13 trade scenarios were tested to confirm that the proposed management area framework and trading ratios would achieve compliance with TMDL water quality targets. Scenarios that would most stress the system and simulate critical conditions were developed to test the proposed trading program. Examples of "stressful" trade scenarios are those where trades occur i) between dischargers on different tributaries, ii) between dischargers in different management areas, iii) with the seller downstream, or iv) with trading ratios greater than 1. Additionally, each trade scenario was simulated at the critical diversion condition for that trade, and at “dry year” effluent flows.

From the set of 13 trade scenarios, 5 were chosen for the uncertainty analysis. Scenario 1 represents a simple and presumably safe type of trade – the buyer is downstream of the seller, and both are located on the same tributary. Each of the other four scenarios represents one of the diversion conditions, and one or more of the stressful factors cited above. As a whole, the five trading scenarios chosen for Stage 3 represent all three diversion conditions and all the stressful factors cited above. All 13 of the trading scenarios equaled or outperformed corresponding baseline scenarios; however Scenarios 2, 3, 8 and 9 came closest to exceeding the baseline when compared to other scenarios from the same diversion condition, therefore those 4 scenarios were chosen for Stage 3 analysis. Table 5-2 lists the 5 trade scenarios in detail, while Table 5-3 gives a general description of them. The enumeration of the scenarios is maintained to be consistent with Passaic Trading Project (2009). Figures 5-2 through 5-6 geographically illustrate each of the trade scenarios.

Three baseline simulations (Table 5-3) were developed for comparison to the appropriate trading scenario. In each baseline, no trading occurs and all plants discharge LTA 0.4 mg/L TP. Hydrologically, each of the baseline simulations reflects one of the three diversion conditions, and the dry year set of effluent flows, i.e., the minimum annual effluent flows from each WWTP in 2004-2006. Extreme diversion conditions were overlaid with WY2002 conditions, and the diversion and no diversion conditions were each overlaid with WY2001 conditions. All scenarios assume 60% reduction in nonpoint source loads from existing conditions.

Note that the TMDL scenario assumes permitted flows from WWTPs. In Stage 3, the trading scenarios (and corresponding baseline scenarios) assume actual, i.e., lower effluent flows from WWTPs. Therefore Stage 3 trade scenario results are not directly comparable to Stage 2.

In general terms, Scenario 1 was designed to test the safest form of trading, with the seller upstream on the same tributary; Scenario 2 was designed to test trades along the same tributary with the seller downstream; Scenario 3 was designed to test cross-tributary trading with a trading ratio above 1; Scenarios 8-9 were designed to test a complex variety of cross- tributary and inter-MA trades.

5.3.2 Execution of uncertainty analysis

Stage 3 utilized the same uncertain variables as in Stage 2, therefore 50 model runs were executed to generate the probabilistic output.

5.3.2.1 Kinetic variables and headwater boundary conditions for phosphorus

The same probability distributions and parameter sets that were used in Stage 2 were applied in Stage 3 for local and global kinetic parameters, and headwater boundary conditions for phosphorus.

5.3.2.2 WWTP boundary conditions for phosphorus

The baseline scenarios used the same boundary condition sets as in Stage 2. For the trade scenarios, the same process as Stage 2 was followed, except to replace 0.4 mg/L TP as the lognormal distribution arithmetic mean with 0.1 mg/L TP for sellers and the designated effluent concentration for the buyer. The COVs for TP effluent distributions were assumed to be 0.6, just as in Stage 2. Figure 5-7 depicts a probability distribution of TP effluent from a WWTP acting as a seller in any of the trade scenarios. Equation 4.1 described in Stage 2 was applied to derive orthophosphate and organic phosphorus boundary condition values from the total phosphorus values.

If a small WWTP with less than 1 MGD permitted flow was part of a trade scenario, its TP uncertainty was ignored, and Equation 4.1 was applied to its designated LTA to derive orthophosphate and organic phosphorus boundary condition values.

5.3.2.3 Output processing

The same procedure as in Stage 2 was followed for output processing of TP, chl-*a* and DO.

5.3.2.4 Method of comparing trade and baseline scenario outputs

The right-tailed two-sample Kolmogorov-Smirnov (K-S) test was applied to compare output distributions of trade and baseline scenarios. The K-S test can be applied to any type of continuous distribution. The right-tailed two-sample K-S test determines the probability that one continuous distribution is larger than another under the null hypothesis that both samples come from the same distribution. The usage of the right-tailed two-sample K-S test enabled the research question in Stage 3 to be answered – do trade scenarios significantly increase the output distribution above the baseline approach?

5.4 Results

5.4.1 Extreme diversion scenarios

Baseline 6 output was compared to Scenarios 1, 8 and 9. K-S test results are shown in Table 5-4.

5.4.1.1 TP

In Scenario 1, the distribution of diverted TP load at the WS intake showed no significant increase from Baseline 6. Scenarios 8 and 9 showed a significant improvement ($p = 0.00$) in diverted TP load, because the largest WWTP upstream of the WS intake –Two Bridges SA- acted as a seller (Table 5-5 and Fig. 5-8).

At the Little Falls intake, Scenarios 1 and 8 showed no significant increases in annual average TP concentration from Baseline 6. Scenario 9 showed a significant improvement ($p = 0.02$) because of the large number of upstream sellers coupled with buyers discharging at TP concentrations less than 1.1 mg/L (Table 5-6 and Fig. 5-9).

5.4.1.2 Chl-a

In all trade scenarios, the distribution of seasonal average chl-*a* showed no significant increases from Baseline 6 at Dundee Lake (Table 5-7 and Fig. 5-10).

5.4.1.3 DO

As in Stage 2, the DO output locations of concern were Dundee Lake, the Peckman River mouth, the Passaic River near Chatham and station PA10. Because of the design of Scenarios 1 and 8, impacts to the Peckman River mouth were not applicable in those scenarios.

In all trade scenarios, the distribution of daily average DO exceedance frequencies showed no significant increases from Baseline 6 at all locations (Table 5-8, Figs. 5-11 and 5-12).

In all trade scenarios, the distribution of minimum DO exceedance frequencies showed no significant increases from Baseline 6 at all locations (Table 5-9, Figs. 5-13 through 5-16).

At Dundee Lake and station PA10, the distributions of diurnal DO swing showed no significant increase from the baseline in all trade scenarios. However, there were three other cases where the distribution of diurnal DO swing showed a significant increase in uncertainty from the baseline: at the Passaic River near Chatham in Scenarios 8 and 9, and at the Peckman River mouth in Scenario 9. In each of these cases, a buyer was located directly upstream of the area of concern. The increased TP discharge from the buyer(s) above baseline levels contributed to the rise in diurnal DO swing at the area of concern. The magnitude of deviation was however fairly small. The percent differences from the baseline in mean value, standard deviation, and 90th percentile value for all three cases ranged from 7 to 11% (Tables 5-10 and 5-11, Figs. 5-17 through 5-20).

Note that significant increases in diurnal DO swing influenced daily average and minimum DO exceedance frequency distributions. At the Passaic River near Chatham, as the diurnal DO swing increased in Scenarios 8 and 9, the daily average DO expected exceedance decreased (Tables 5-8 and 5-10). Apparently, the increased productivity helps to raise naturally low DO; Omni Environmental (2007a) reported the same finding. In contrast, in Scenario 1 where a seller was located directly upstream of the Passaic River near Chatham, the diurnal DO swing decreased and the daily average DO expected exceedance increased.

At the Peckman River mouth in Scenario 9, the increased diurnal DO swing affected minimum DO results there in terms of an increase in expected exceedance, decline in confidence of compliance, and increase in the 90% confidence interval (Table 5-9).

5.4.2. Diversion scenario

Baseline 5 output was compared to Scenario 3. K-S test results are shown in Table 5-12.

5.4.2.1 TP

The distribution of diverted TP load at the Wanaque South intake in Scenario 3 showed no significant increase from Baseline 5 (Table 5-13 and Fig. 5-21).

At the Little Falls intake, Scenario 3 showed no significant increase in annual average TP concentration from Baseline 5 (Table 5-14 and Fig. 5-22).

5.4.2.2 Chl-a

The distribution of seasonal average chl-*a* at Dundee Lake showed no significant increase in Scenario 3 from Baseline 5 (Table 5-15 and Fig. 5-23).

5.4.2.3 DO

The DO output locations of concern were Dundee Lake and station PA10. Because of the design of Scenario 3, impacts to the Peckman River mouth and the Passaic River near Chatham were not applicable. Ignoring these 2 locations is justified because the diversion scenario is never a critical diversion condition for trades that could adversely affect either location, i.e., when a buyer is either upstream of the Passaic River near Chatham or the Peckman River mouth.

In Scenario 3, the distributions of daily average and minimum DO exceedance frequencies, as well as diurnal DO swing, showed no significant increases from Baseline 5 at Dundee Lake and station PA10 (Tables 5-16 through 5-18, and Figures 5-24 through 5-29).

5.4.3 No Diversion scenario

Baseline 4 output was compared to Scenario 2. K-S test results are shown in Table 5-19.

5.4.3.1 TP

At the Little Falls intake, Scenario 2 showed no significant increase in annual average TP concentration from Baseline 4 (Table 5-20 and Fig. 5-30).

5.4.3.2 Chl-a

The distribution of seasonal average chl-*a* at Dundee Lake showed no significant increase in Scenario 2 from Baseline 4 (Table 5-21 and Fig. 5-31).

5.4.3.3. DO

The DO output locations of concern were Dundee Lake, the Passaic River near Chatham and station PA10. Because of the design of Scenario 2, impacts to the Peckman River were not applicable. Ignoring this location is justified because the no diversion scenario is never a critical diversion condition for trades that could adversely affect this location, i.e., when a buyer is upstream of the Peckman River mouth.

The distribution of daily average and minimum DO exceedance frequencies showed no significant increases in Scenario 4 from Baseline 2 at each location (Tables 5-22 and 5-23, and Figures 5-32 through 5-36).

At Dundee Lake and station PA10, the distributions of diurnal DO swing showed no significant increase in Scenario 2 from Baseline 4. However, the diurnal DO swing showed a significant increase in uncertainty from the baseline at the Passaic River near Chatham. As in Scenarios 8 and 9, the cause was the location of a buyer directly upstream. The magnitude of deviation was however very small. The percent differences from the baseline in mean value, standard deviation, and 90th percentile value ranged from 4 to 6% (Tables 5-24 and 5-25, and Figures 5-37 through 5-39). Note that the higher diurnal DO swing increased the daily average DO confidence of compliance from the baseline value.

5.5 Discussion

The method and results of Stage 3 illustrate the importance of reporting the relative uncertainties of management and baseline approaches. This issue was raised by Arabi et al. (2007) in their study of BMP effectiveness uncertainty in an agricultural watershed. In this analysis, each trade scenario had either a DO standard expected exceedance above 10%, or a chl-*a* standard exceedance probability above 10%, or both. If the analysis had neglected to compare the relative uncertainty of trade scenario to baseline outcomes, the wrong conclusion about the performance of water quality trading would have been reached. Only when comparing the relative uncertainty of trading scenarios to the corresponding baseline approaches does it become apparent that there were no significant increases at any key location for the exceedance of DO standards, or the uncertainty of predicted chl-*a*, TP diverted load, and TP concentration at the Little Falls intake. In addition, a few cases saw significant improvements in achieving a water quality goal, e.g., Scenarios 8 and 9 with respect to diverted TP load, and Scenario 9 with respect to TP concentration at Little Falls intake. Overall, the right-tailed two-sample K-S test was shown to be a clear and straightforward way to compare if a management strategy increases outcome uncertainty relative to the baseline approach. The left-tailed two-sample K-S test should be used to determine if outcome uncertainty has significantly decreased due to a management strategy.

Four outcomes, all regarding diurnal DO swing, did show a significant increase from the baseline. The diurnal DO swing distribution at the Passaic River near Chatham and/or the Peckman River mouth increased significantly above the baseline in Scenarios 2, 8 and 9. However the magnitudes of the deviations were small and ranged from 4 to 11% in terms of mean value, standard deviation, and 90th percentile value. A degree of deviation is not surprising at these locations since the trading ratios were not specifically designed for them. Considering the modest deviations from the baseline in diurnal DO swing, it is not recommended to prohibit Scenarios 2, 8 or 9, especially since the latter two scenarios show significant improvements in diverted TP load from the WS intake to the Wanaque Reservoir.

The large proportion of trade scenario outcomes that showed no significant increase in the predicted uncertainty of DO, TP and chl-*a* can be attributed to the use of conservative trading ratios designed for worst-case conditions. Moreover, trading scenario outcomes at the critical locations of Dundee Lake and the WS intake never showed a significant increase above the baseline. The results fail to reject the assumption that phosphorus trading in the NTPRB can reliably achieve compliance with TMDL water quality goals under the most severe circumstances, such as cross-tributary or inter-MA trading, dry year effluent flows and critical diversion conditions. With respect to the final research hypothesis, the analysis found no evidence to suggest that the outcome of trades between WWTPs, as compared with command and control regulation, will significantly increase uncertainty in the attainment of DO surface water quality standards, site-specific chl-*a* criteria, and reduction targets for diverted TP load at affected potential hot spots in the watershed.

Finally, it is worth noting that the large majority of WQT programs in the United States involve nutrient trading. Examples include the Chesapeake Bay, Long Island Sound, Neuse River and Minnesota River trading programs. This study is the first to examine the effects of water quality model uncertainty on a nutrient trading program. Trading in the Passaic will primarily involve point sources, since there is almost no agriculture in the watershed. This is useful in that trading with nonpoint sources will contain more uncertainties than trading between point sources. By conducting a thorough uncertainty analysis of a point-point source nutrient trading program, a lower bound on the range of uncertainty regarding nutrient trading programs in general has been obtained, which could benefit nutrient trading programs nationwide.

Table 5-1: Passaic trading framework for management areas (adapted from Obropta et al., 2008)

Seller \ Buyer	Upper Passaic MA	Pompton MA	Lower Passaic MA
Upper Passaic MA	Yes	No	Yes
Pompton MA	Yes	Yes	Yes
Lower Passaic MA	No	No	Yes

Table 5-2: Trade scenarios analyzed in Stage 3

Scenario	Seller	Buyer	Seller flow (MGD)	Buyer flow (MGD)	Trading ratio	Seller LTA (mg/L)	Buyer LTA (mg/L)
1	Berkeley Hts	Florham Park	1.61	0.86	0.78	0.100	0.860
2	Caldwell	Bernards (Harrison Brook)	3.99	1.78	0.92	0.100	0.985
		Warren V		0.15			1.0
3	Parsippany-Troy Hills	Rockaway	12.48	10.06	1.03	0.100	0.930
	Caldwell		3.99		1.09		
8	Parsippany-Troy Hills	Bernards (Harrison Brook)	12.48	1.78	0.82	0.100	2.910
	Rockaway		10.06		0.65	0.100	
		Warren V		0.15	0.65		1.850
		Warren I-II		0.37	0.63		1.620
	Two Bridges	Morris-Butterworth	5.57	2.02	0.63	0.100	0.968
	Berkeley Heights	Warren IV	1.61	0.28	1.00	0.100	2.200
	Morristown	Florham Park	2.15	0.86	0.77	0.100	1.360
	Wanaque RSA		0.96		0.43	0.100	
	Pompton Lakes	Chatham Glen	0.89	0.12	0.46	0.100	1.530
9	Parsippany-Troy Hills	Cedar Grove	12.48	1.23	0.43	0.100	1.045
		Bernards (Harrison Brook)		1.78	0.90	0.100	1.000
		Florham Park		0.86	0.86	0.100	1.000
	Rockaway	Hanover	10.06	2.1	0.77	0.100	1.000
		Long Hill		1.03	0.70		1.000
		Molitor (Madison-Chatham)		2.48	0.67		0.927
	Two Bridges		5.57		0.60	0.100	

	Pompton Lakes		0.89		0.48	0.100	
	Morristown	Morris-Butterworth	2.15	2.02	1.00	0.100	0.927
	Wanaque RSA		0.96		0.56	0.100	
	Berkeley Heights	Warren IV	1.61	0.28	1.00	0.100	1.000
		Warren I-II		0.37	1.00		1.080
		Chatham Glen		0.12	1.00		1.000

Table 5-3: Description of baseline and trading scenarios analyzed in Stage 3

Scenario	General description	Diversion condition
Baseline 4	All WWTPs discharging at 0.4 mg/L LTA total phosphorus effluent	No Diversion
Baseline 5	All WWTPs discharging at 0.4 mg/L LTA total phosphorus effluent	Diversion
Baseline 6	All WWTPs discharging at 0.4 mg/L LTA total phosphorus effluent	Extreme Diversion
1	Simple trade with seller upstream; both located in Upper Passaic MA; Seller is upstream of Passaic River near Chatham.	Extreme Diversion
2	Seller is downstream of buyer; both located in Upper Passaic MA. Buyer is upstream of Passaic River near Chatham.	No diversion
3	Cross tributary trade with trading ratio > 1. Intra-Upper Passaic MA trades.	Diversion
8	Complex trades with buyers and sellers in different management areas; buyers concentrated upstream; several buyers upstream of Passaic River near Chatham; sellers upstream of WS intake.	Extreme Diversion
9	Complex trades with buyers and sellers in different management areas; buyers spread upstream and downstream; buyers upstream of Passaic River near Chatham and Peckman River mouth; sellers upstream of WS intake.	Extreme Diversion

Table 5-4: Results of right-tailed two-sample K-S tests for baseline and trade scenarios at extreme diversion condition

Location	Output distribution	SC1 p-value	SC8 p-value	SC9 p-value
Dundee Lake	DO daily average exceedance frequency, June 15-Sep. 1, 2002	1.0	1.0	1.0
Peckman River mouth		NA	NA	1.0
Passaic River near Chatham		0.47	1.0	1.0
PA10		1.0	1.0	1.0
Dundee Lake	Minimum DO exceedance frequency, June 15-Sep. 1, 2002	1.0	1.0	1.0
Peckman River mouth		NA	NA	0.35
Passaic River near Chatham		0.98	0.47	0.71
PA10		1.0	0.98	0.98
Dundee Lake	Seasonal average of chl-*a*	0.92	0.83	0.98
Wanaque South intake	TP load diverted to Wanaque Reservoir, WY2002	0.92	1.0	1.0
Little Falls intake	Average TP concentration, WY2002	0.92	0.83	0.98
Dundee Lake	Diurnal DO swing, June 15- Sep. 1, 2002	0.97	0.98	0.78
Peckman River mouth		NA	NA	**0.00**
Passaic River near Chatham		1.0	**0.00**	**0.00**
PA10		0.91	0.98	0.63

Table 5-5: TP load diverted from WS intake to Wanaque Reservoir during WY2002 for baseline and trade scenarios at extreme diversion condition

Scenario	Mean (kg)	Standard deviation (kg)	90^{th} percentile (kg)	Probability of exceeding 11,999 kg (%)
Baseline 6	9,576	624	10,540	0
Scenario 1	9,577	620	10,540	0
Scenario 8	8,100	606	9,026	0
Scenario 9	7,883	589	8,751	0

Table 5-6: Average TP concentration at Little Falls intake during WY2002 for baseline and trade scenarios at extreme diversion condition

Scenario	Mean (mg/L)	Standard deviation (mg/L)	90^{th} percentile (mg/L)
Baseline 6	0.10	0.01	0.12
Scenario 1	0.09	0.01	0.12
Scenario 8	0.10	0.01	0.12
Scenario 9	0.10	0.01	0.12

Table 5-7: 2002 seasonal average of chl-*a* at Dundee Lake for baseline and trade scenarios at extreme diversion condition

Scenario	Mean (μg/L)	Standard deviation (μg/L)	Probability of exceeding 20 μg/L (%)	90^{th} percentile (μg/L)
Baseline 6	18.74	19.45	28	44.77
Scenario 1	18.65	19.28	28	44.30
Scenario 8	18.58	19.32	28	45.74
Scenario 9	18.16	19.41	28	39.40

Table 5-8: Distribution of daily average DO exceedance frequencies during June 15-September 1, 2002 for baseline and trade scenarios at extreme diversion condition

	Expected Exceedance (%)				Confidence of Compliance (%)				90% confidence interval			
Scenario / Location	BL6	SC1	SC8	SC9	BL6	SC1	SC8	SC9	BL6	SC1	SC8	SC9
Dundee Lake	0	0	0	0	100	100	100	100	0-0	0-0	0-0	0-0
Peckman River mouth	0	NA	NA	0	100	NA	NA	100	0-0	NA	NA	0-0
Passaic River near Chatham	28	33	22	22	36	31	48	51	0-76	0-79	0-75	0-75
PA10	0	0	0	0	100	100	100	100	0-0	0-0	0-0	0-0

Table 5-9: Distribution of minimum DO exceedance frequencies during June 15-September 1, 2002 for baseline and trade scenarios at extreme diversion condition

	Expected Exceedance (%)				Confidence of Compliance (%)				90% confidence interval			
Scenario / Location	BL6	SC1	SC8	SC9	BL6	SC1	SC8	SC9	BL6	SC1	SC8	SC9
Dundee Lake	0	0	0	0	100	100	100	100	0-0	0-0	0-0	0-0
Peckman River mouth	7	NA	NA	11	78	NA	NA	66	0-28	NA	NA	0-43
Passaic River near Chatham	48	47	51	50	19	19	16	18	0-82	0-82	0-82	0-82
PA10	0	0	1	0	100	100	100	100	0-2	0-2	0-3	0-2

Table 5-10: Distribution of diurnal DO swing during June 15- September 1, 2002 for baseline and trade scenarios at extreme diversion condition

	Mean (mg/L)				Standard deviation (mg/L)				90^{th} percentile (mg/L)			
Scnrio. / Loctn.	BL6	SC1	SC8	SC9	BL6	SC1	SC8	SC9	BL6	SC1	SC8	SC9
Dundee Lake	2.32	2.32	2.29	2.33	1.41	1.41	1.38	1.44	4.20	4.22	4.17	4.28
Peckman River mouth	5.45	NA	NA	6.17	2.08	NA	NA	2.27	8.25	NA	NA	9.13
Passaic River near Chatham	3.62	3.29	4.00	3.91	1.33	1.27	1.47	1.42	5.35	4.97	5.96	5.79
PA10	3.23	3.24	3.18	3.23	1.66	1.66	1.62	1.66	5.56	5.59	5.50	5.62

Table 5-11: Percent difference from baseline at extreme diversion condition in diurnal DO swing distribution key values for trade scenarios with significantly increased distribution

	Mean (%)		Standard deviation (%)		90th percentile (%)	
Scenario / Location	SC8	SC9	SC8	SC9	SC8	SC9
Peckman River mouth	NA	11	NA	9	NA	11
Passaic River near Chatham	10	8	11	7	11	8

Table 5-12: Results of right-tailed two-sample K-S tests for baseline and trade scenarios at diversion condition

Location	Output distribution	SC3 p-value
Dundee Lake	DO daily average exceedance frequency, June 15- Sep. 1, 2001	1.0
Peckman River mouth		NA
Passaic River near Chatham		NA
PA10		1.0
Dundee Lake	Minimum DO exceedance frequency, June 15- Sep. 1, 2001	1.0
Peckman River mouth		NA
Passaic River near Chatham		NA
PA10		0.98
Dundee Lake	Seasonal average of chl-*a*	1.0
Wanaque South intake	TP load diverted to Wanaque Reservoir, WY2001	0.98
Little Falls intake	Average TP concentration, WY2001	1.0
Dundee Lake	Diurnal DO swing, June 15- Sep. 1, 2001	0.98
Peckman River mouth		NA
Passaic River near Chatham		NA
PA10		1.0

Table 5-13: TP load diverted from WS Intake to Wanaque Reservoir during WY2001 for baseline and trade scenarios at diversion condition

Scenario	Mean (kg)	Standard deviation (kg)	90th percentile (kg)	Probability of exceeding 4,088 kg (%)
Baseline 5	2,685	258	2,990	0
Scenario 3	2,677	258	2,978	0

Table 5-14: Average TP concentration at Little Falls intake during WY2001 for baseline and trade scenarios at diversion condition

Scenario	Mean (mg/L)	Standard deviation (mg/L)	90^{th} percentile (mg/L)
Baseline 5	0.08	0.01	0.09
Scenario 3	0.08	0.01	0.09

Table 5-15: 2001 seasonal average of chl-*a* at Dundee Lake for baseline and trade scenarios at diversion condition

Scenario	Mean (μg/L)	Standard deviation (μg/L)	Probability of exceeding 20 μg/L (%)	90^{th} percentile (μg/L)
Baseline 5	13.16	11.63	26	28.49
Scenario 3	12.66	11.23	26	28.34

Table 5-16: Distribution of daily average DO exceedance frequencies during June 15-September 1, 2001 for baseline and trade scenarios at diversion condition

	Expected Exceedance (%)		Confidence of Compliance (%)		90% confidence interval	
Scenario / Location	BL5	SC3	BL5	SC3	BL5	SC3
Dundee Lake	0	0	100	100	0-0	0-0
Peckman River mouth	NA	NA	NA	NA	NA	NA
Passaic River near Chatham	NA	NA	NA	NA	NA	NA
PA10	0	0	100	100	0-1	0-1

Table 5-17: Distribution of minimum DO exceedance frequencies during June 15-September 1, 2001 for baseline and trade scenarios at diversion condition

	Expected Exceedance (%)		Confidence of Compliance (%)		90% confidence interval	
Scenario / Location	BL5	SC3	BL5	SC3	BL5	SC3
Dundee Lake	0	0	100	100	0-2	0-2
Peckman River mouth	NA	NA	NA	NA	NA	NA
Passaic River near Chatham	NA	NA	NA	NA	NA	NA
PA10	2	1	96	96	0-9	0-9

Table 5-18: Distribution of diurnal DO swing during June 15- September 1, 2001 for baseline and trade scenarios at diversion condition

	Mean (mg/L)		Standard deviation (mg/L)		90^{th} percentile (mg/L)	
Scenario / Location	BL5	SC3	BL5	SC3	BL5	SC3
Dundee Lake	2.09	2.06	1.39	1.37	3.91	3.85
Peckman River mouth	NA	NA	NA	NA	NA	NA
Passaic River near Chatham	NA	NA	NA	NA	NA	NA
PA10	2.97	2.92	1.52	1.50	5.11	5.03

Table 5-19: Results of right-tailed two-sample K-S tests for baseline and trade scenarios at no-diversion condition

Location	Output distribution	SC2 p-value
Dundee Lake	DO daily average exceedance frequency, June 15-Sep.1, 2001	1.0
Peckman River mouth		NA
Passaic River near Chatham		1.0
PA10		1.0
Dundee Lake	Minimum DO exceedance frequency, June 15-Sep.1, 2001	1.0
Peckman River mouth		NA
Passaic River near Chatham		0.92
PA10		1.0
Dundee Lake	Seasonal average of chl-*a*	0.83
Wanaque South intake	TP load diverted to Wanaque Reservoir, WY2001	NA
Little Falls intake	Average TP concentration, WY2001	0.98
Dundee Lake	Diurnal DO swing, June 15-Sep. 1, 2001	0.99
Peckman River mouth		NA
Passaic River near Chatham		**0.00**
PA10		0.99

Table 5-20: Average TP concentration at Little Falls intake during WY2001 for baseline and trade scenarios at no-diversion condition

Scenario	Mean (mg/L)	Standard deviation (mg/L)	90^{th} percentile (mg/L)
Baseline 4	0.07	0.01	0.08
Scenario 2	0.07	0.01	0.08

Table 5-21: 2001 seasonal average of chl-*a* at Dundee Lake for baseline and trade scenarios at no-diversion condition

Scenario	Mean (μg/L)	Standard deviation (μg/L)	Probability of exceeding 20 μg/L (%)	90^{th} percentile (μg/L)
Baseline 4	12.71	9.03	22	22.68
Scenario 2	12.92	9.29	24	23.24

Table 5-22: Distribution of daily average DO exceedance frequencies during June 15-September 1, 2001 for baseline and trade scenarios at no-diversion condition

	Expected Exceedance (%)		Confidence of Compliance (%)		90% confidence interval	
Scenario / Location	BL4	SC2	BL4	SC2	BL4	SC2
Dundee Lake	0	0	100	100	0-0	0-0
Peckman River mouth	NA	NA	NA	NA	NA	NA
Passaic River near Chatham	11	10	68	74	0-44	0-42
PA10	0	0	100	100	0-0	0-0

Table 5-23: Distribution of minimum DO exceedance frequencies during June 15-September 1, 2001 for baseline and trade scenarios at no-diversion condition

	Expected Exceedance (%)		Confidence of Compliance (%)		90% confidence interval	
Scenario / Location	BL4	SC2	BL4	SC2	BL4	SC2
Dundee Lake	0	0	100	100	0-1	0-1
Peckman River mouth	NA	NA	NA	NA	NA	NA
Passaic River near Chatham	21	22	37	36	0-55	0-54
PA10	1	1	100	100	0-4	0-4

Table 5-24: Distribution of diurnal DO swing during June 15- September 1, 2001 for baseline and trade scenarios at no-diversion condition

	Mean (mg/L)		Standard deviation (mg/L)		90^{th} percentile (mg/L)	
Scenario / Location	BL4	SC2	BL4	SC2	BL4	SC2
Dundee Lake	2.19	2.16	1.38	1.37	4.04	4.00
Peckman River mouth	NA	NA	NA	NA	NA	NA
Passaic River near Chatham	2.98	3.09	0.95	1.01	4.21	4.45
PA10	2.94	2.90	1.35	1.35	4.84	4.83

Table 5-25: Percent difference from baseline in diurnal DO swing distribution key values for trade scenario with significantly increased distribution at no-diversion condition

	Mean (%)	Standard deviation (%)	90^{th} percentile (%)
Scenario / Location	SC2	SC2	SC2
Passaic River near Chatham	4	6	6

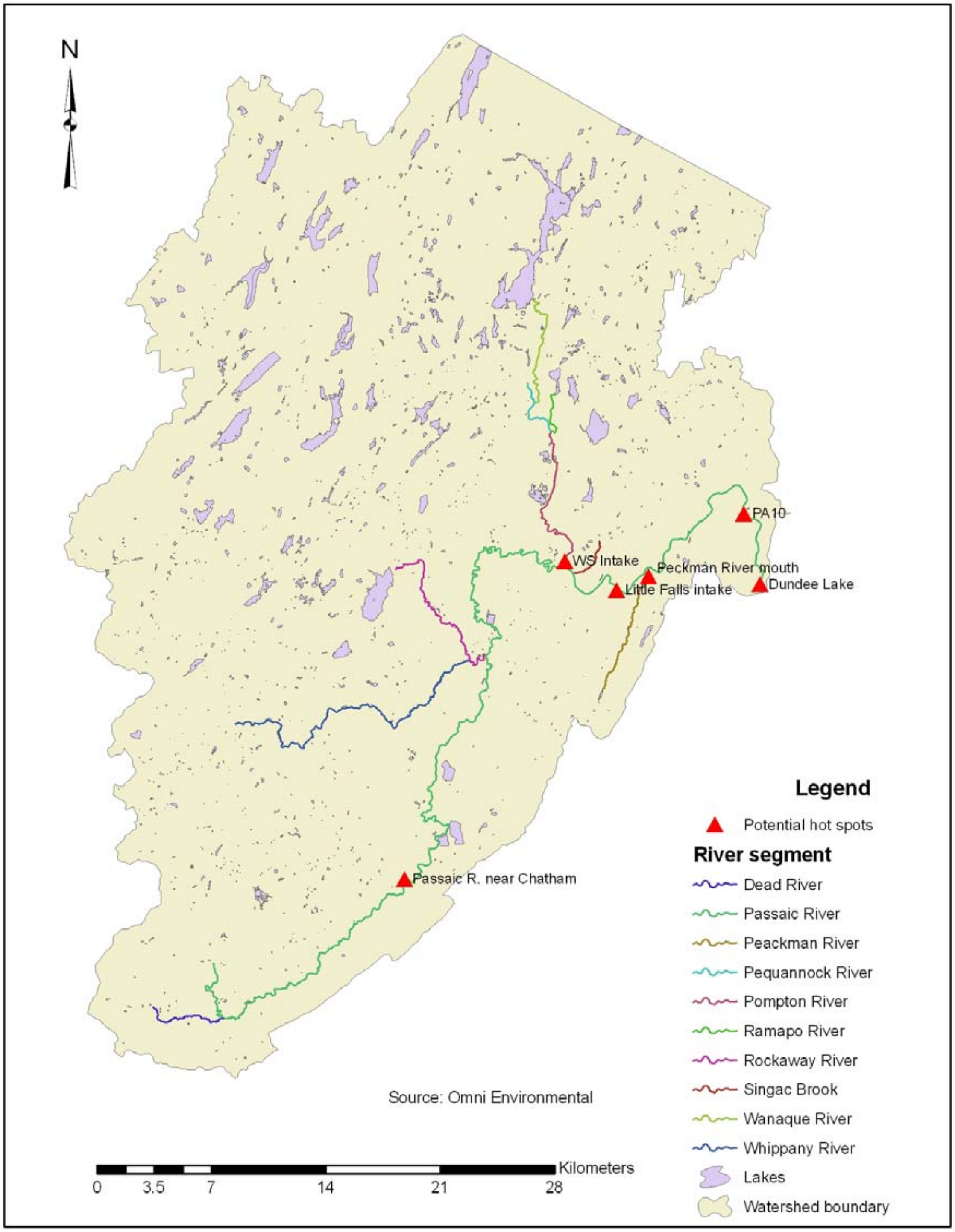


Figure 5-1: Potential hot spots analyzed in Stage 3

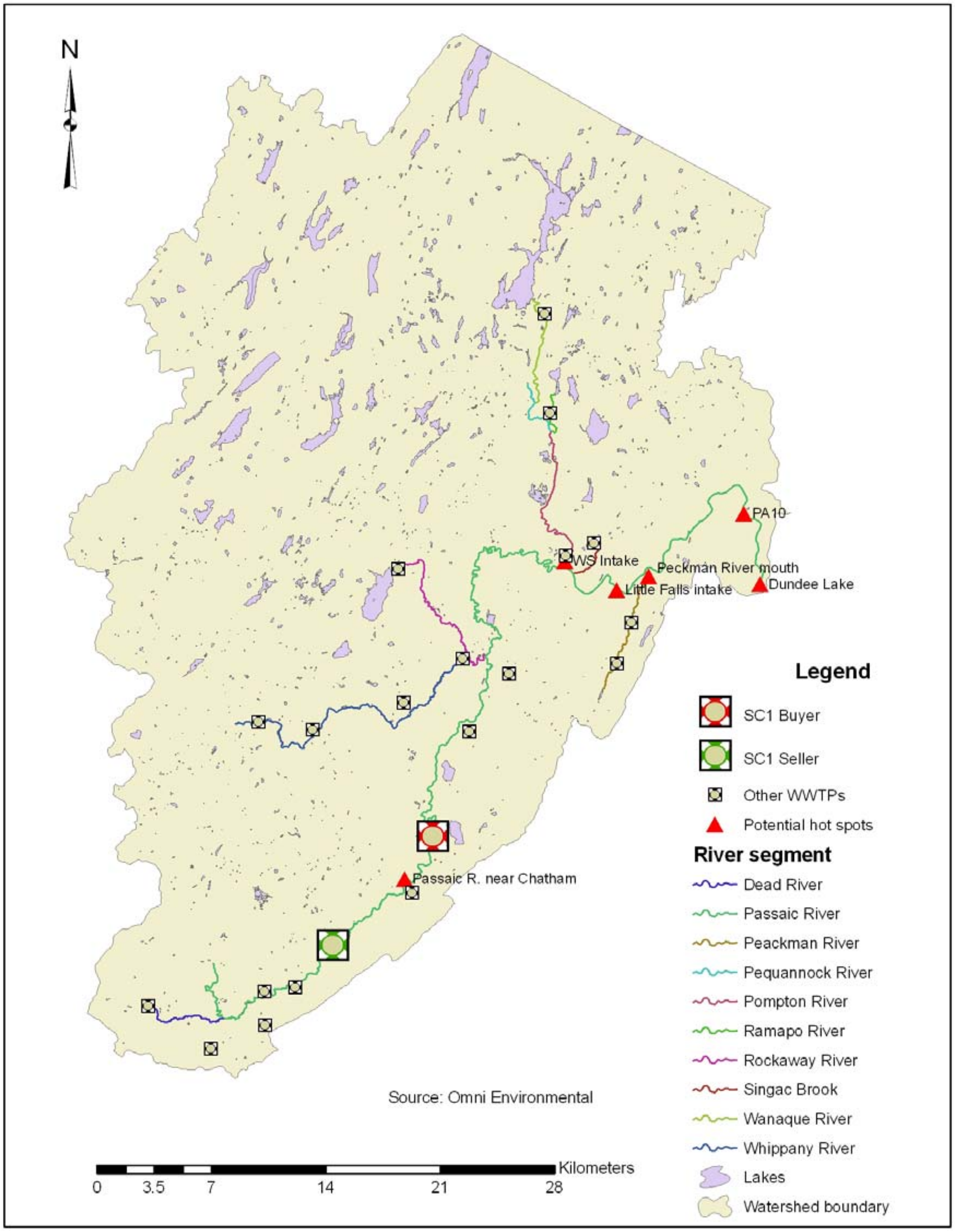


Figure 5-2: Trade Scenario 1

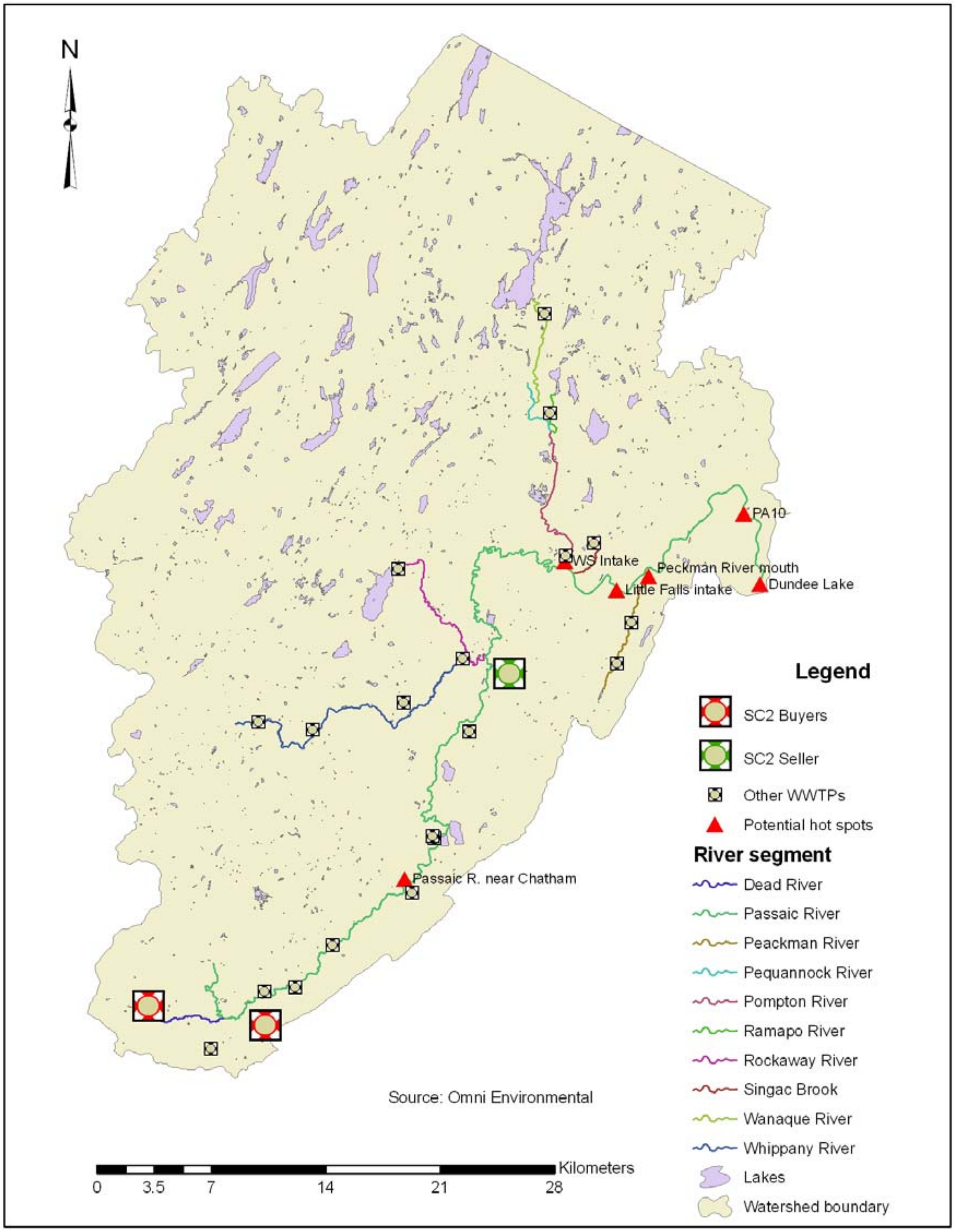


Figure 5-3: Trade Scenario 2

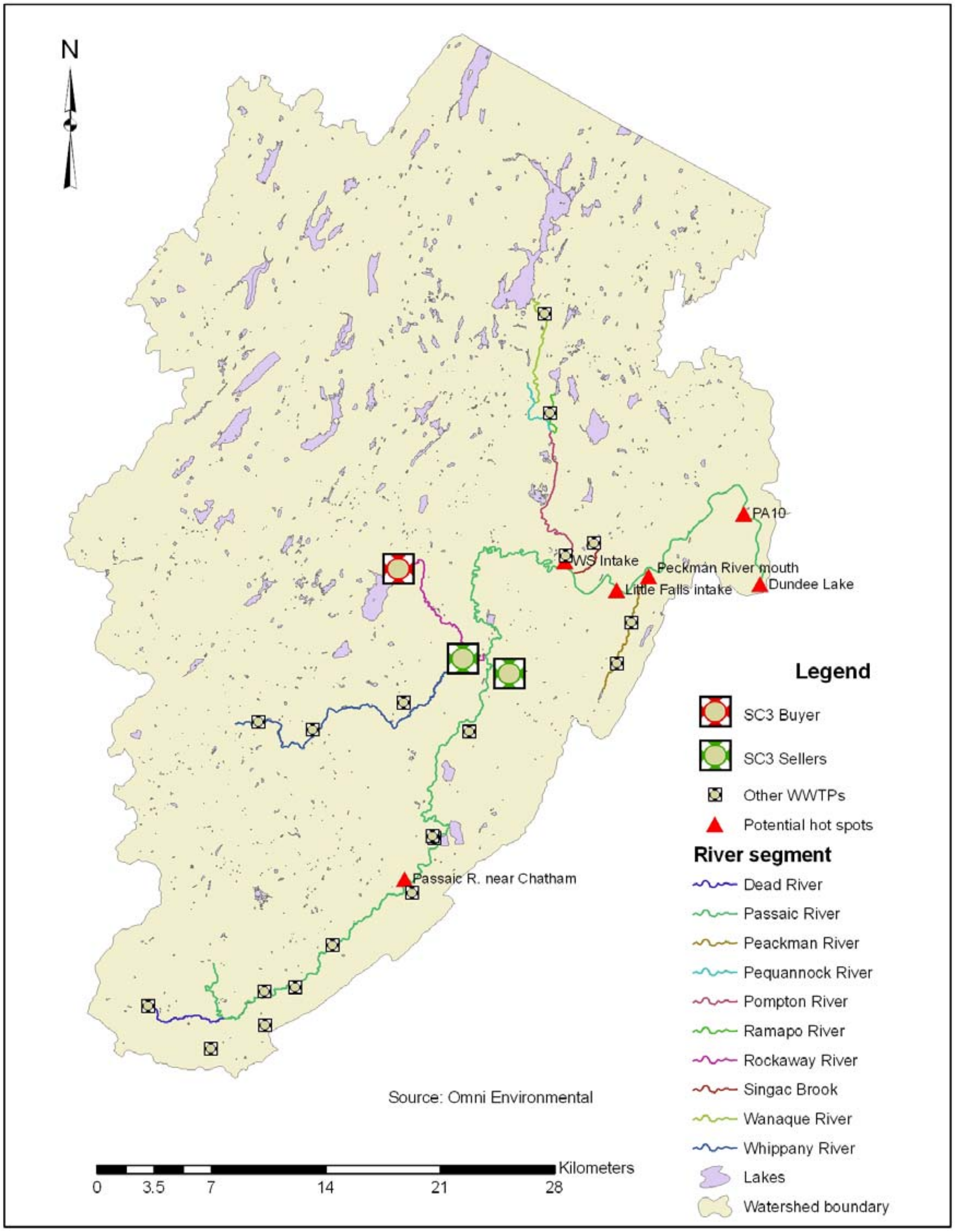


Figure 5-4: Trade Scenario 3

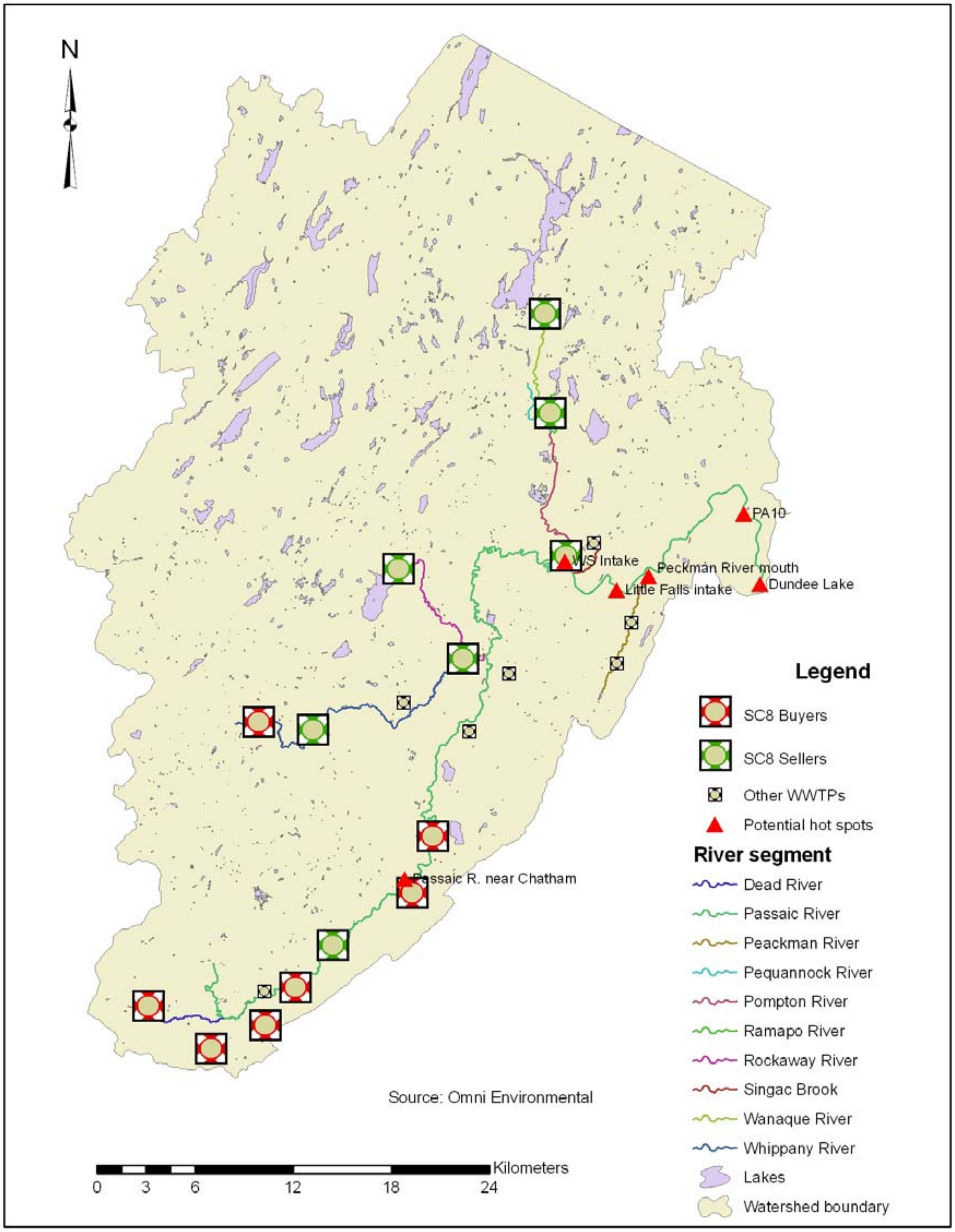


Figure 5-5: Trade Scenario 8

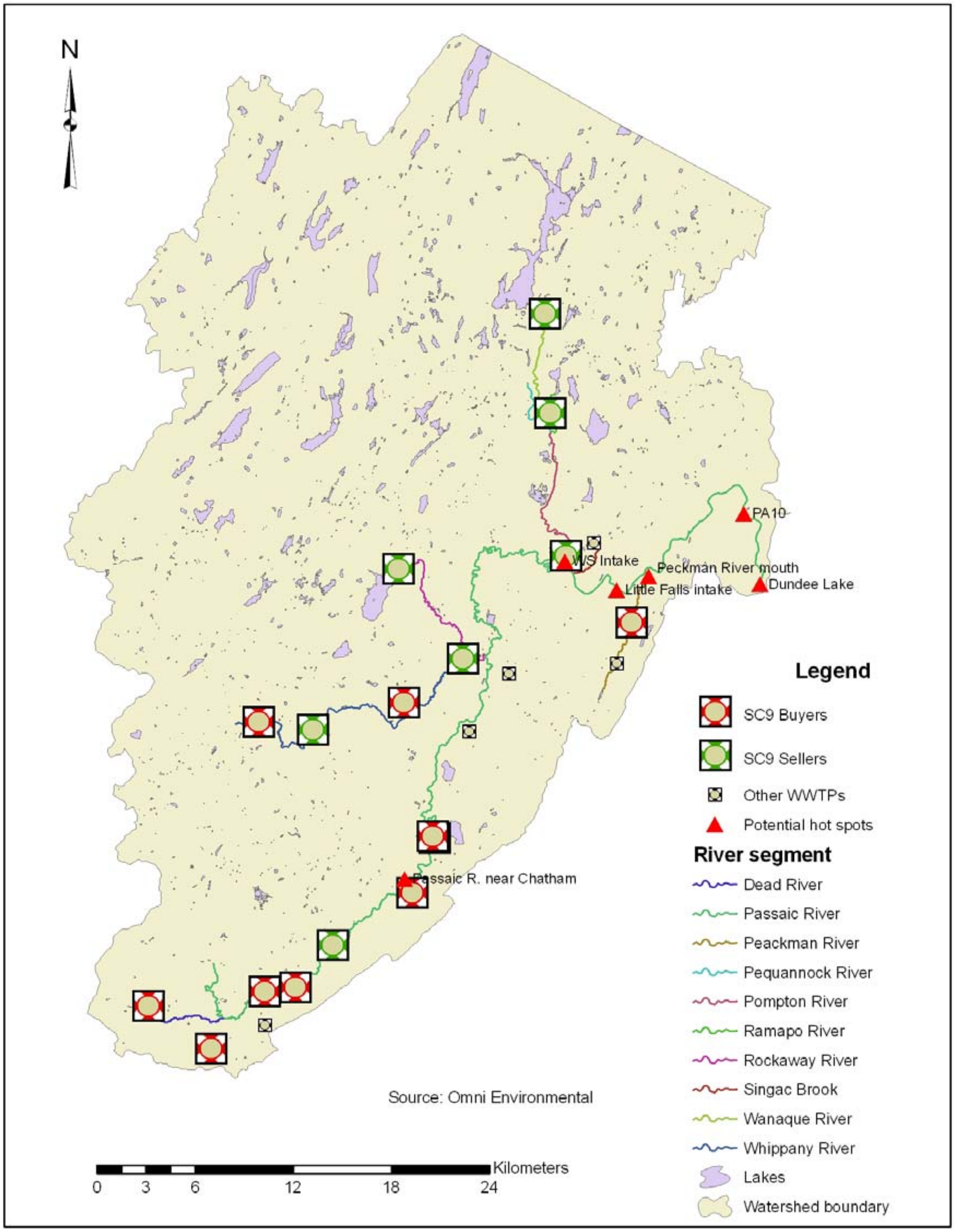


Figure 5-6: Trade Scenario 9

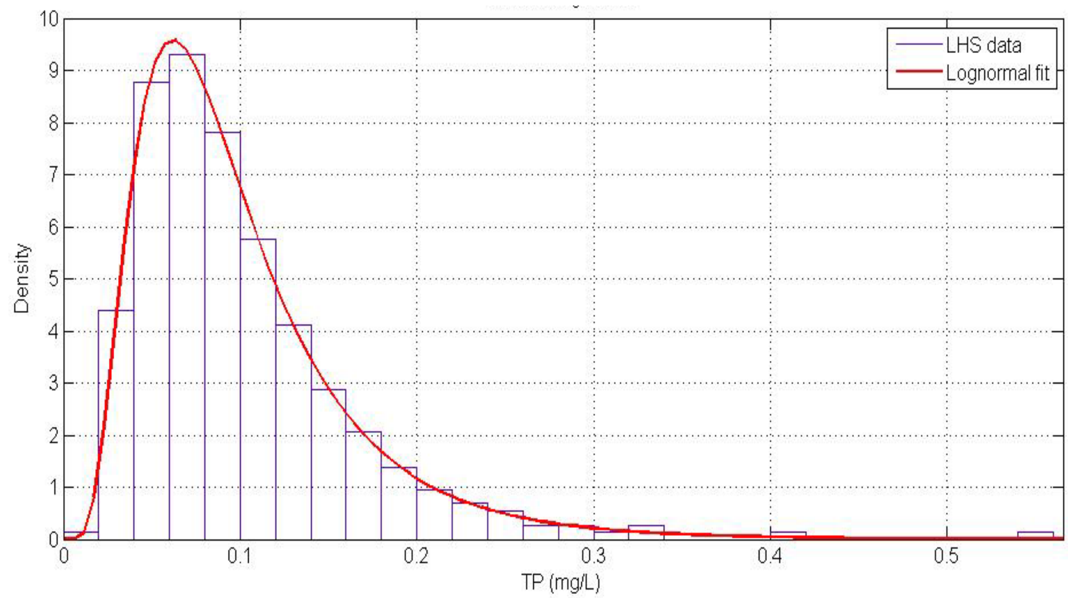


Figure 5-7: Probability distribution of TP effluent from a WWTP acting as a seller in Stage 3 trade scenarios

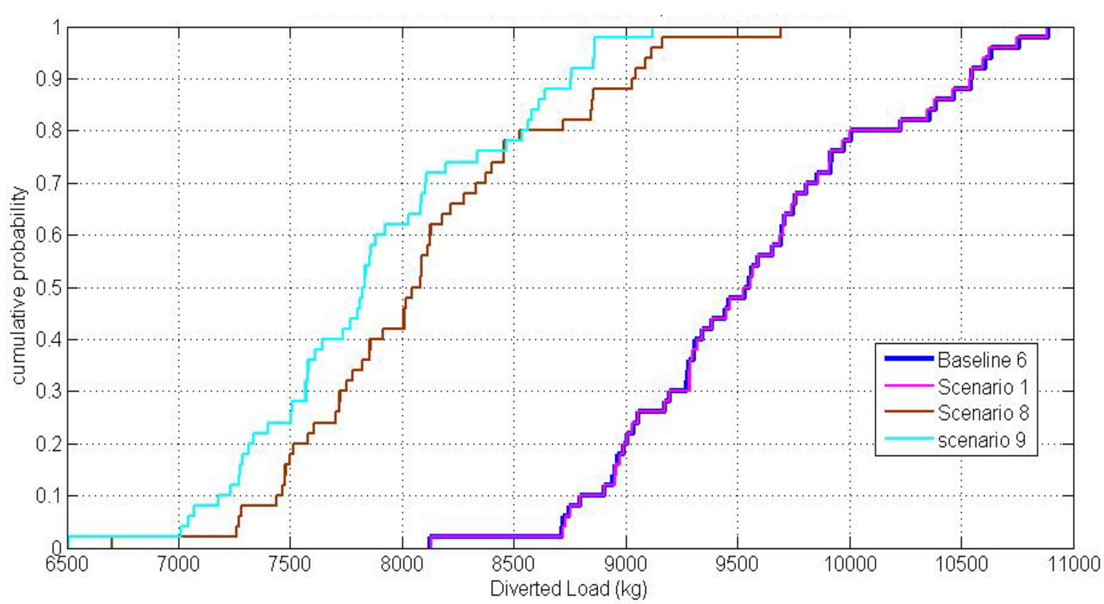


Figure 5-8: CDFs of WY2002 diverted TP load from WS intake to Wanaque Reservoir in baseline and trade scenarios at extreme diversion condition

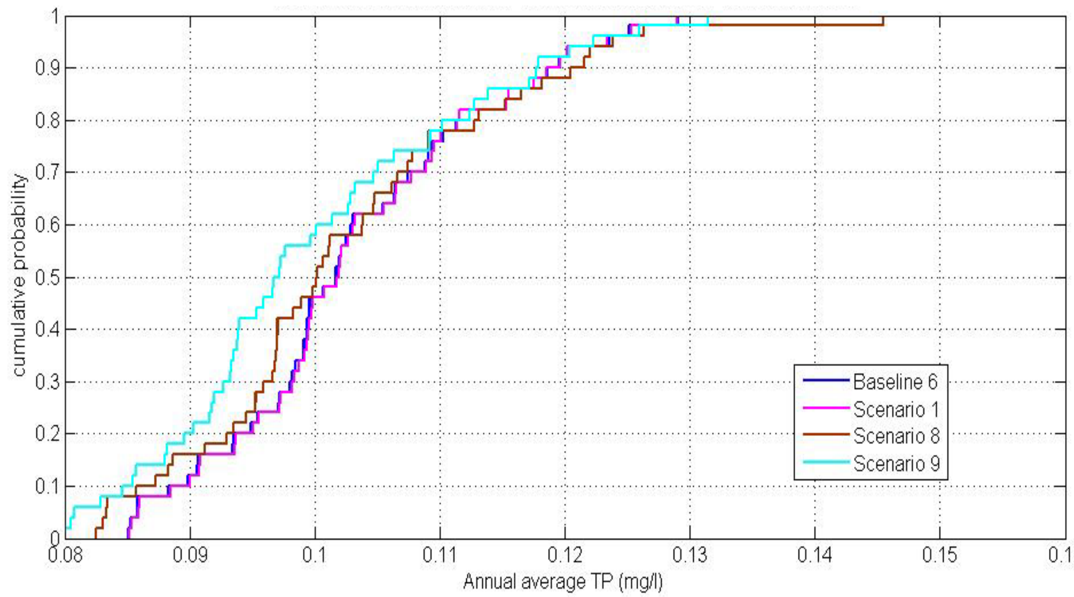


Figure 5-9: CDFs of WY2002 average TP concentration at Little Falls intake in baseline and trade scenarios at extreme diversion condition

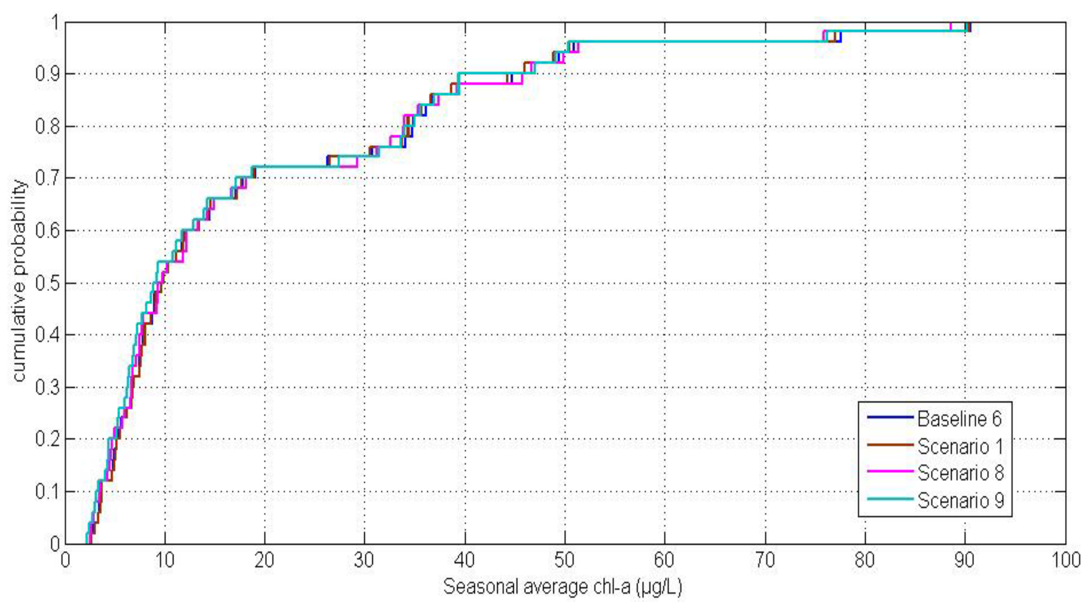


Figure 5-10: CDFs of WY2002 seasonal average chl-*a* concentration at Dundee Lake in baseline and trade scenarios at extreme diversion condition

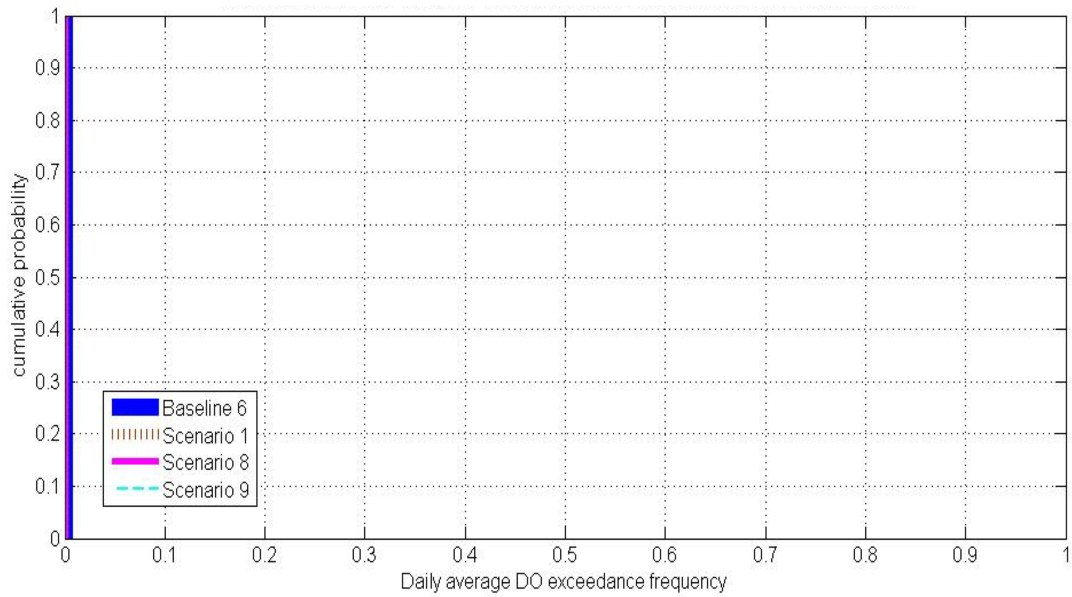


Figure 5-11: CDFs of June 15-Sep. 1, 2002 daily average DO exceedance frequency at Dundee Lake, Peckman River mouth and station PA10 in baseline and trade scenarios at extreme diversion condition

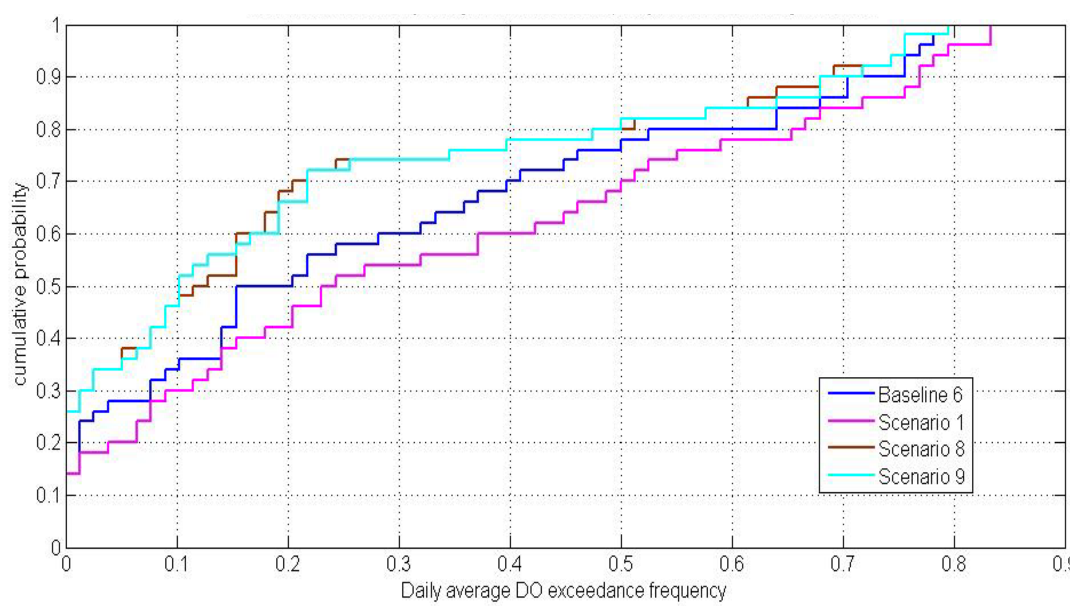


Figure 5-12: CDFs of June 15- Sep. 1, 2002 daily average DO exceedance frequency at Passaic River near Chatham in baseline and trade scenarios at extreme diversion condition

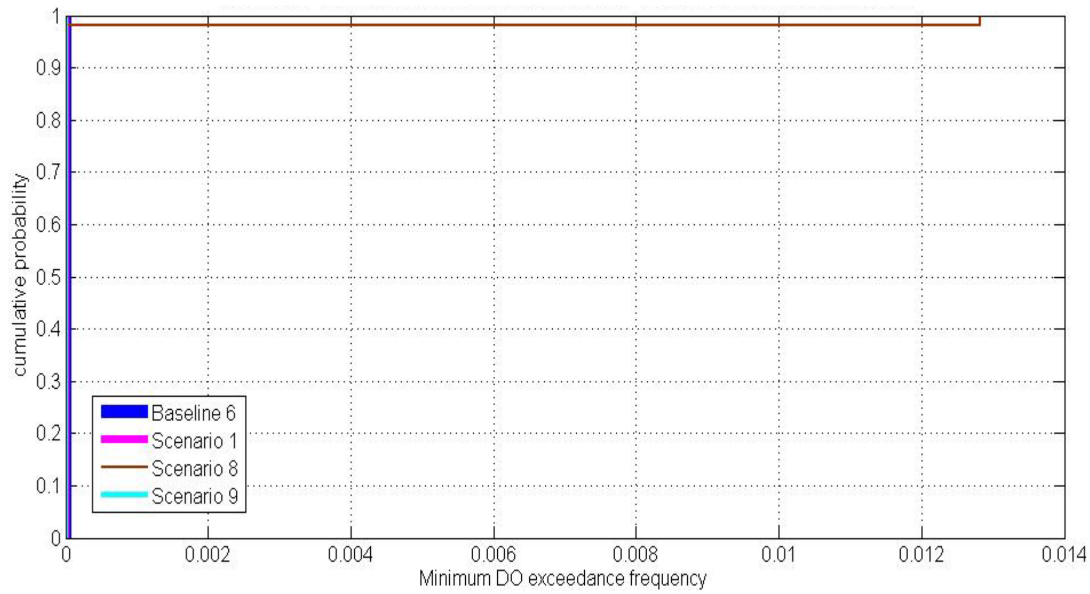


Figure 5-13: CDFs of June 15- Sep. 1, 2002 minimum DO exceedance frequency at Dundee Lake in baseline and trade scenarios at extreme diversion condition

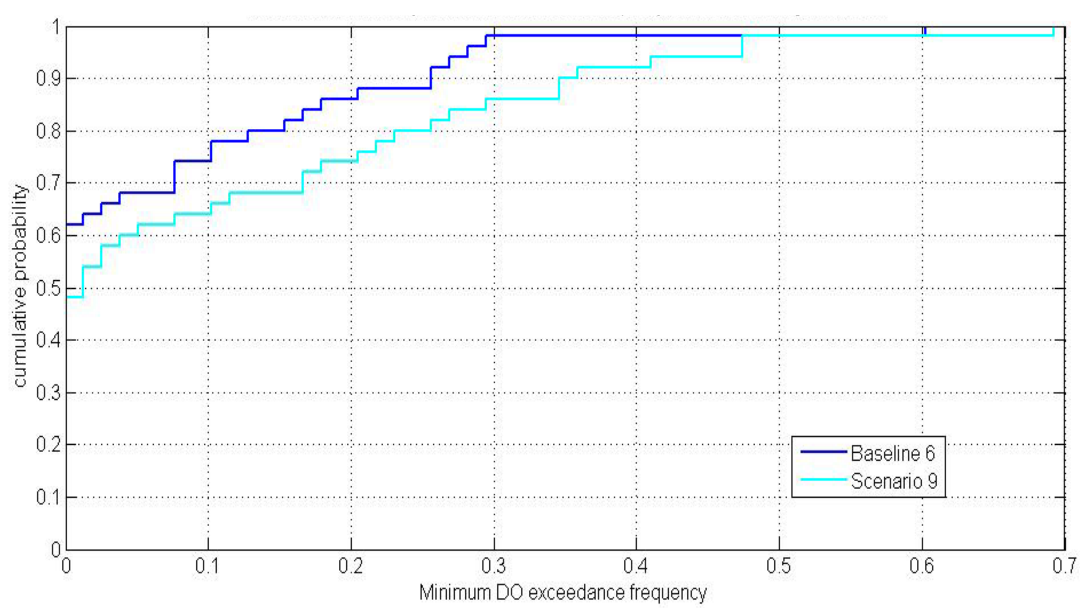


Figure 5-14: CDFs of June 15- Sep. 1, 2002 minimum DO exceedance frequency at Peckman River mouth in baseline and trade scenarios at extreme diversion condition

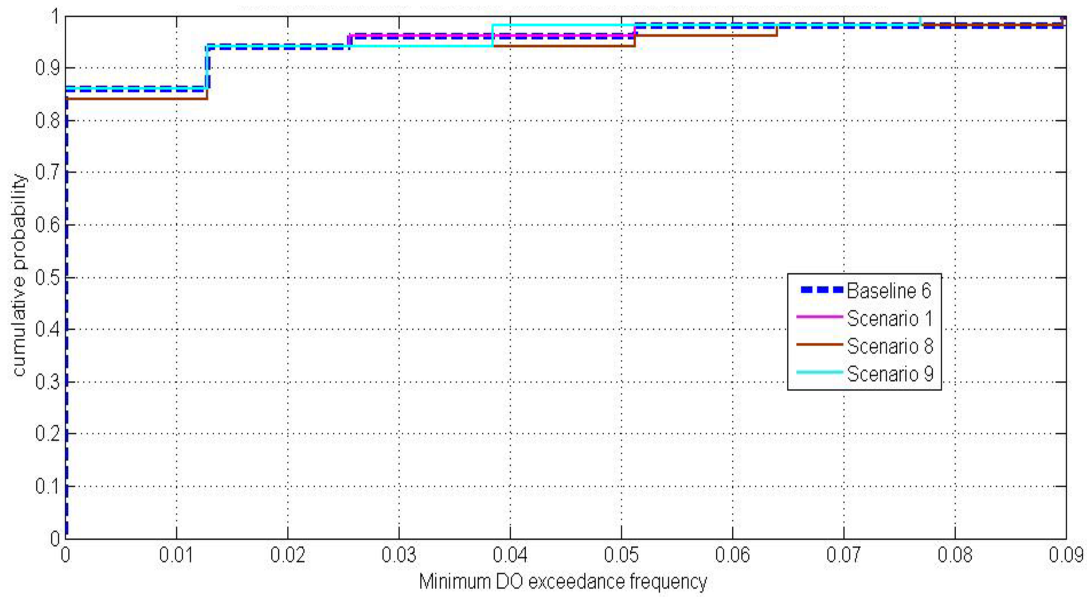


Figure 5-15: CDFs of June 15- Sep. 1, 2002 minimum DO exceedance frequency at station PA10 in baseline and trade scenarios at extreme diversion condition

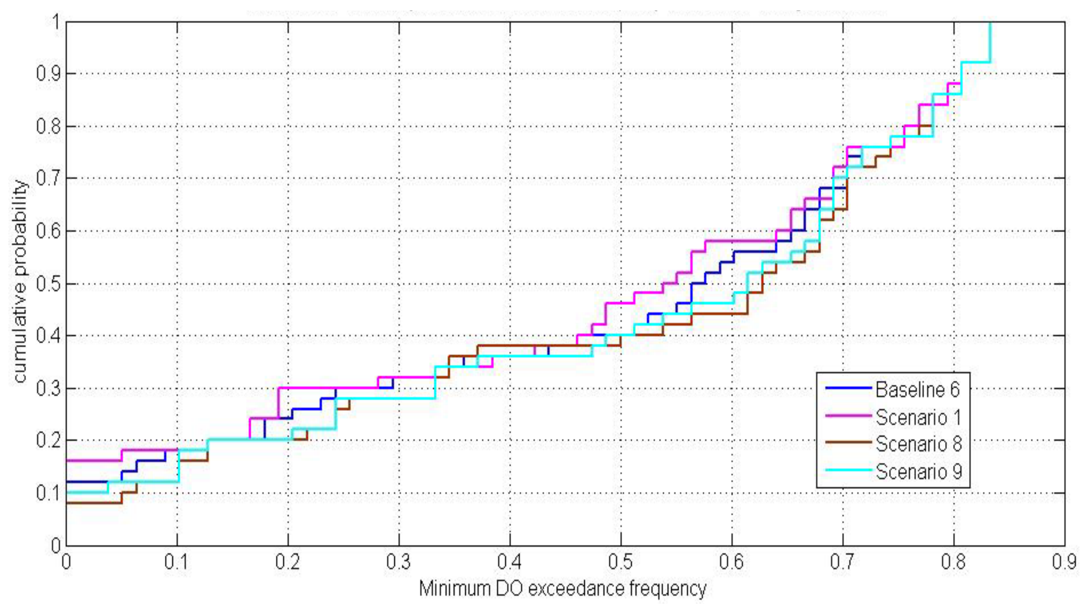


Figure 5-16: CDFs of June 15- Sep. 1, 2002 minimum DO exceedance frequency at Passaic River near Chatham in baseline and trade scenarios at extreme diversion condition

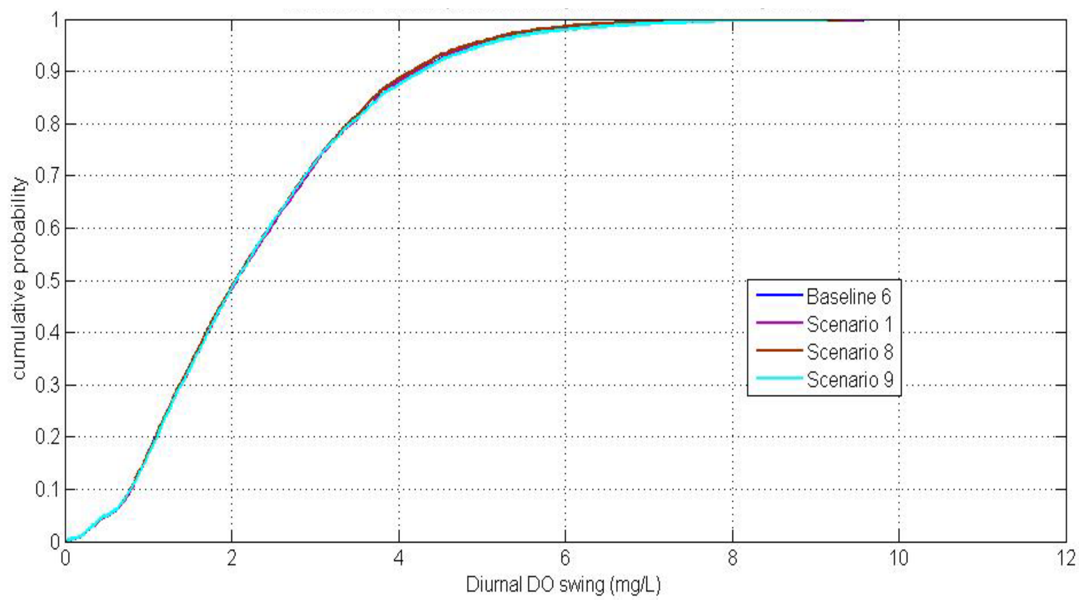


Figure 5-17: CDFs of diurnal DO swing from June 15- Sep. 1, 2002 at Dundee Lake in baseline and trade scenarios at extreme diversion condition

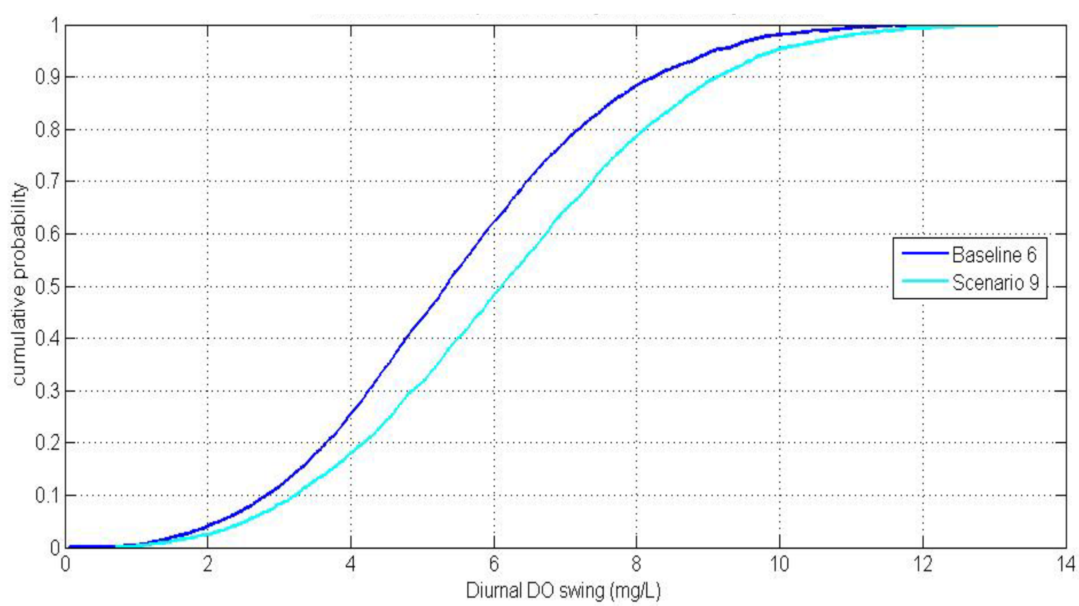


Figure 5-18: CDFs of diurnal DO swing from June 15- Sep. 1, 2002 at Peckman River mouth in baseline and trade scenarios at extreme diversion condition

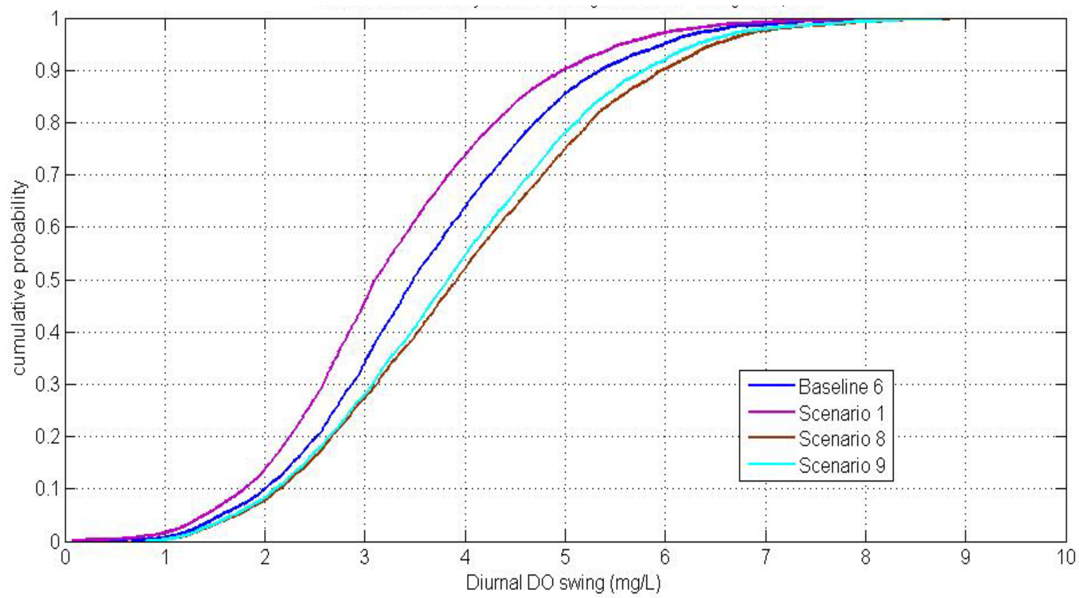


Figure 5-19: CDFs of diurnal DO swing from June 15- Sep. 1, 2002 at Passaic River near Chatham in baseline and trade scenarios at extreme diversion condition

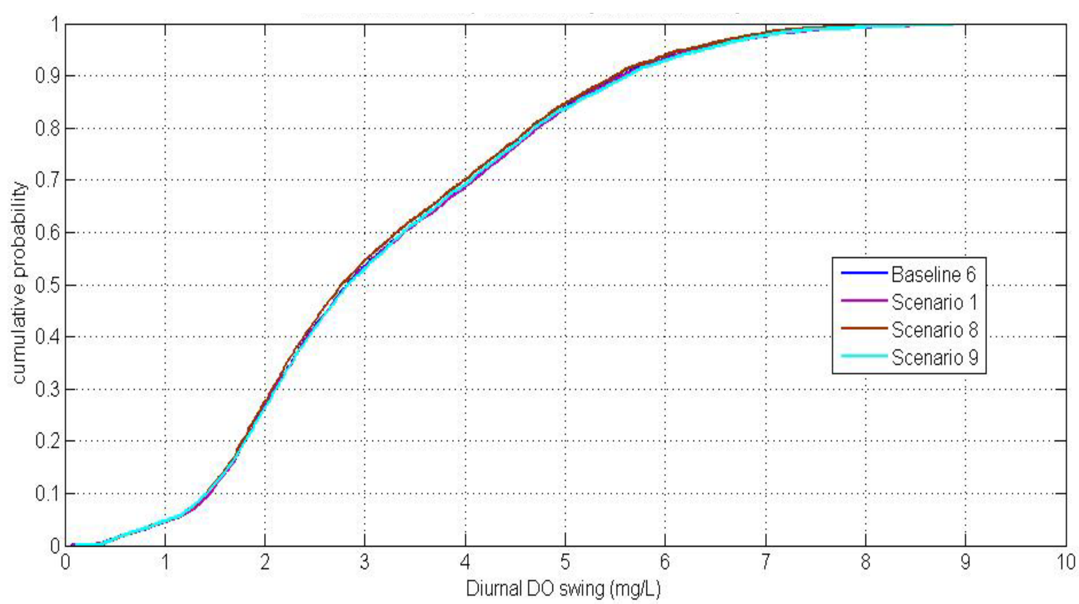


Figure 5-20: CDFs of diurnal DO swing from June 15- Sep. 1, 2002 at station PA10 in baseline and trade scenarios at extreme diversion condition

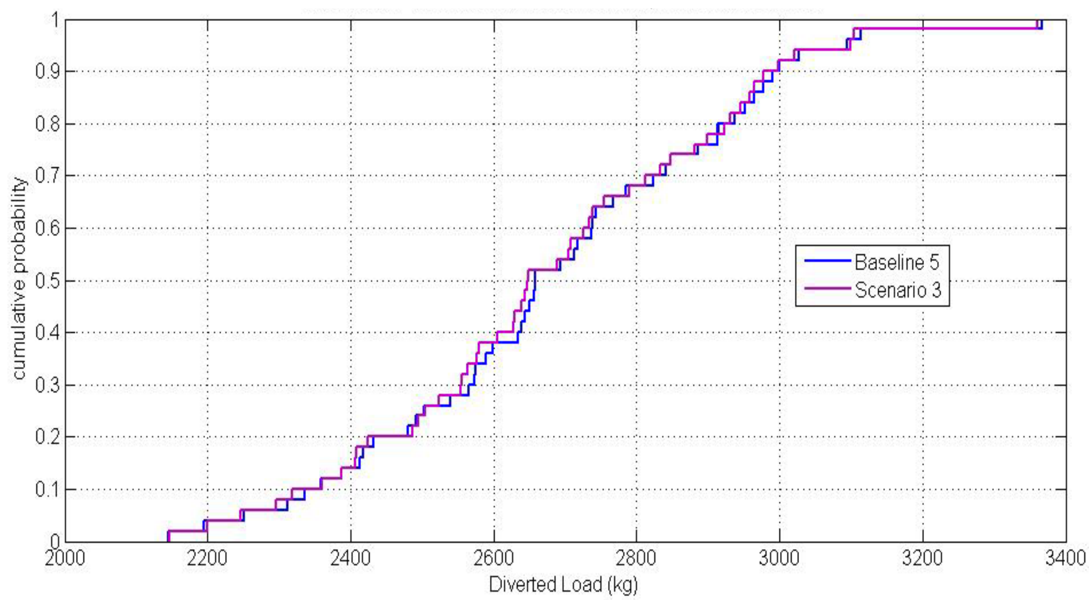


Figure 5-21: CDFs of WY2001 diverted TP load from WS intake to Wanaque Reservoir in baseline and trade scenarios at diversion condition

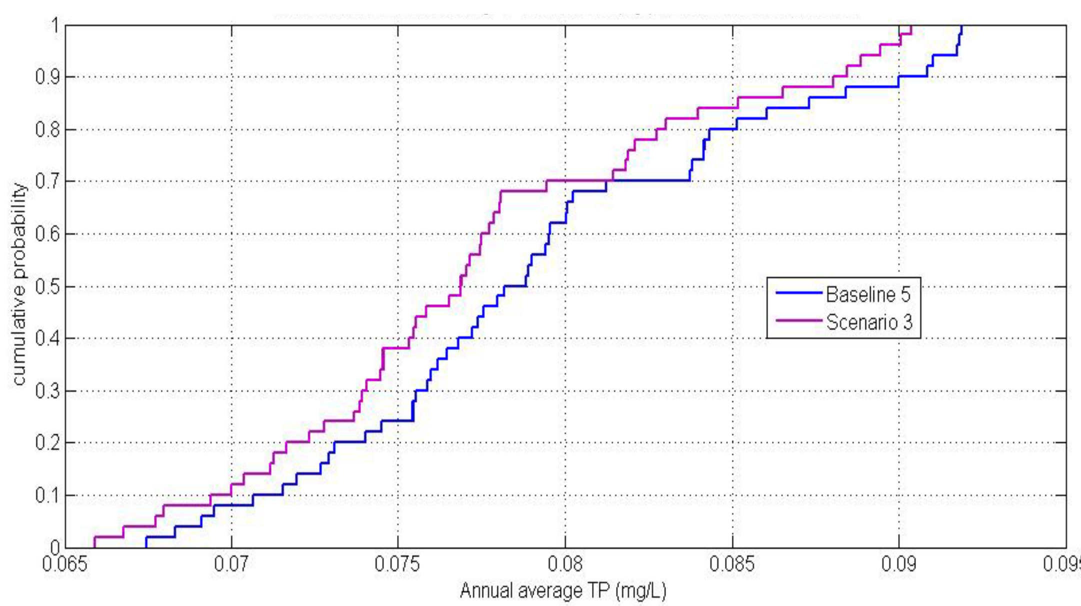


Figure 5-22: CDFs of WY2001 average TP concentration at Little Falls intake in baseline and trade scenarios at diversion condition

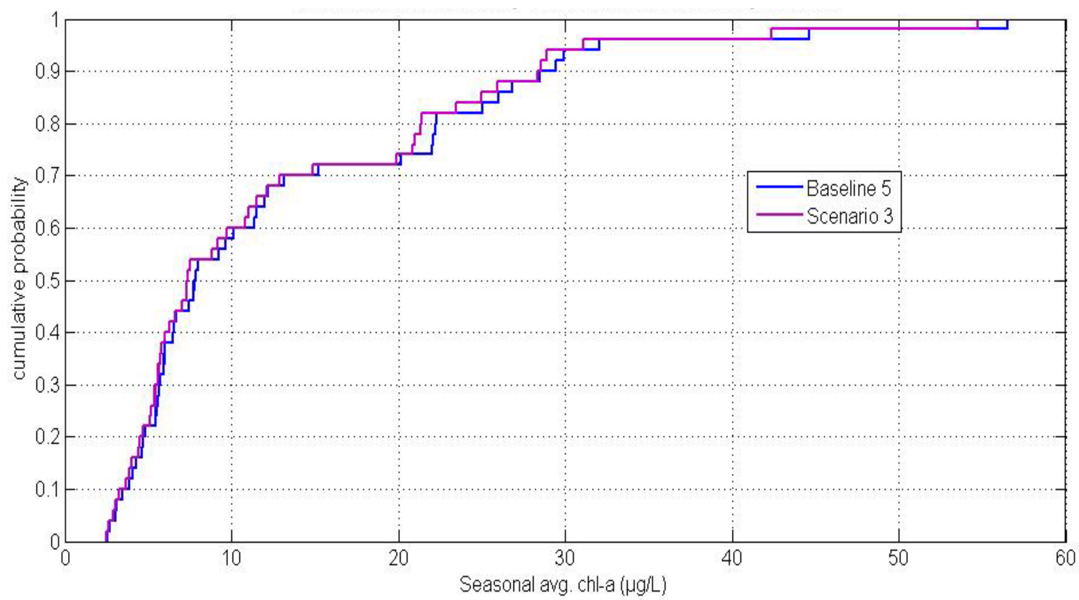


Figure 5-23: CDFs of WY2001 seasonal average chl-*a* concentration at Dundee Lake in baseline and trade scenarios at diversion condition

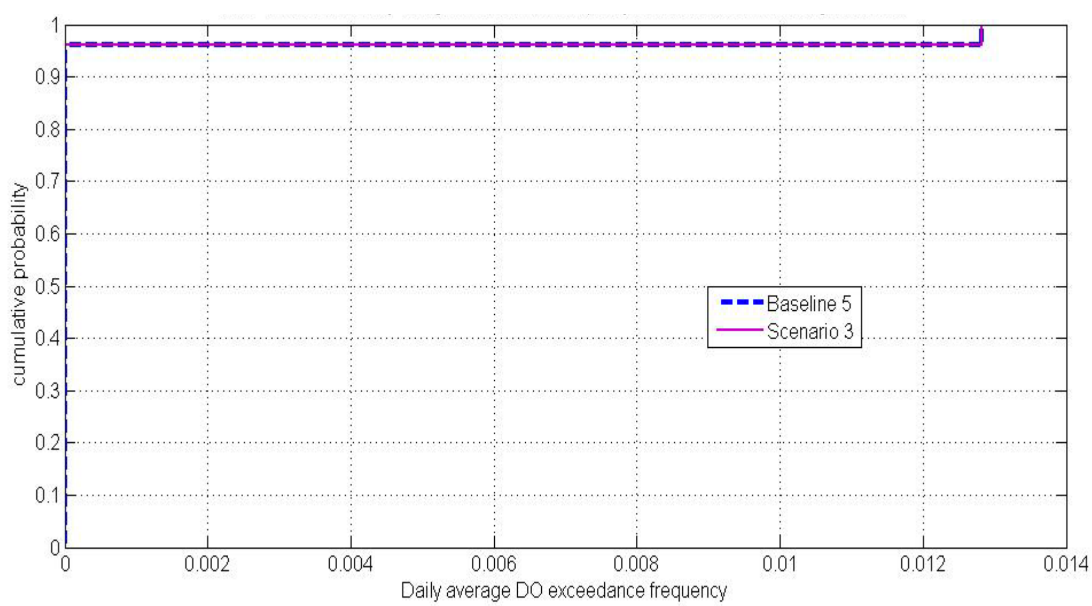


Figure 5-24: CDFs of June 15- Sep. 1, 2001 daily average DO exceedance frequency at Dundee Lake in baseline and trade scenarios at diversion condition

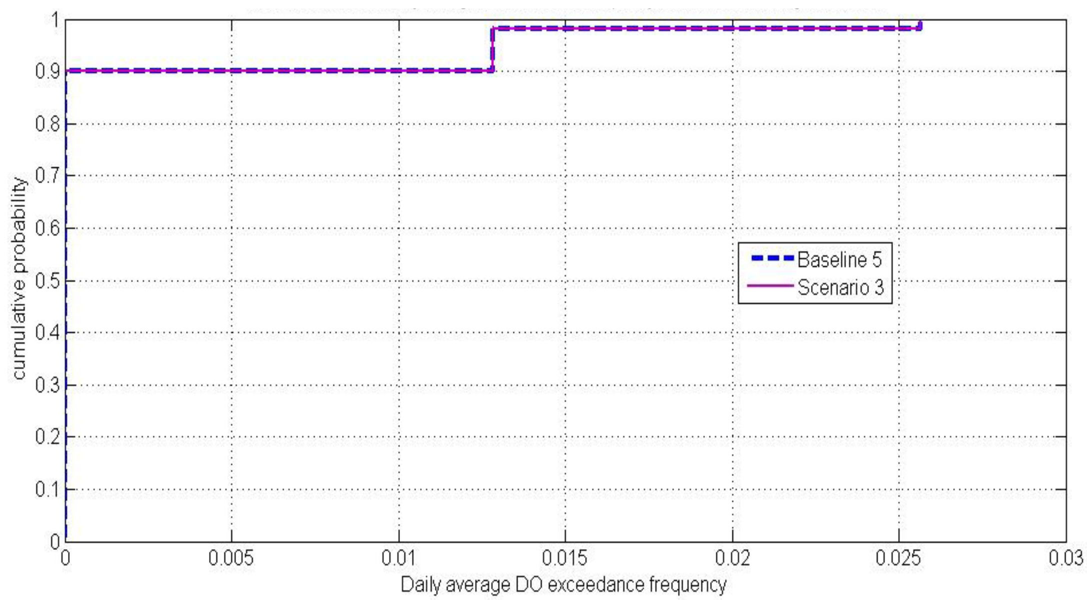


Figure 5-25: CDFs of June 15- Sep. 1, 2001 daily average DO exceedance frequency at station PA10 in baseline and trade scenarios at diversion condition

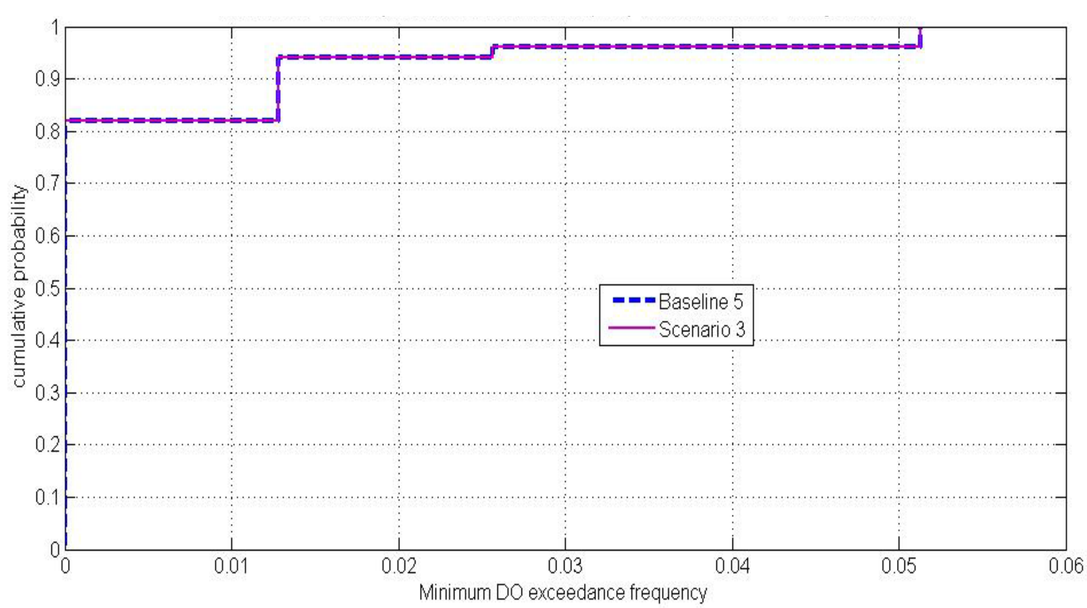


Figure 5-26: CDFs of June 15- Sep. 1, 2001 minimum DO exceedance frequency at Dundee Lake in baseline and trade scenarios at diversion condition

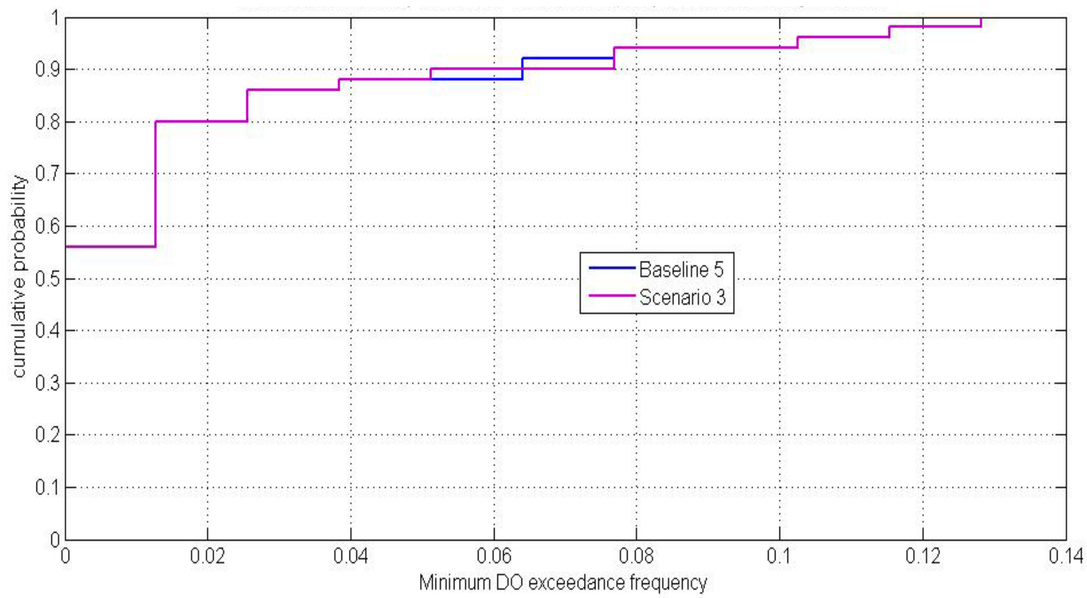


Figure 5-27: CDFs of June 15- Sep. 1, 2001 minimum DO exceedance frequency at station PA10 in baseline and trade scenarios at diversion condition

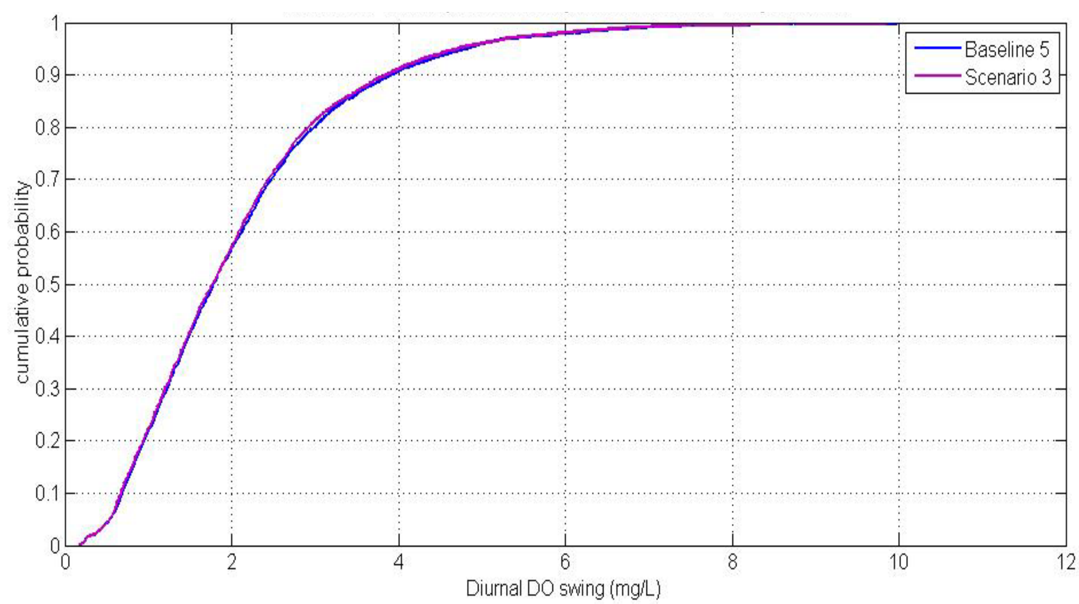


Figure 5-28: CDFs of diurnal DO swing from June 15- Sep. 1, 2001 at Dundee Lake in baseline and trade scenarios at diversion condition

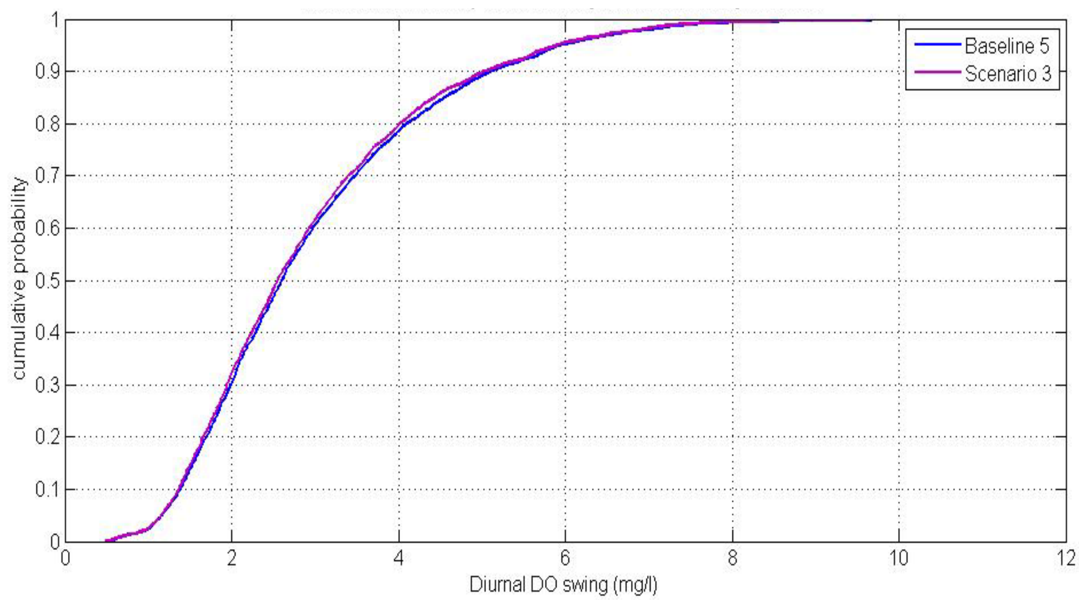


Figure 5-29: CDFs of diurnal DO swing from June 15- Sep. 1, 2001 at station PA10 in baseline and trade scenarios at diversion condition

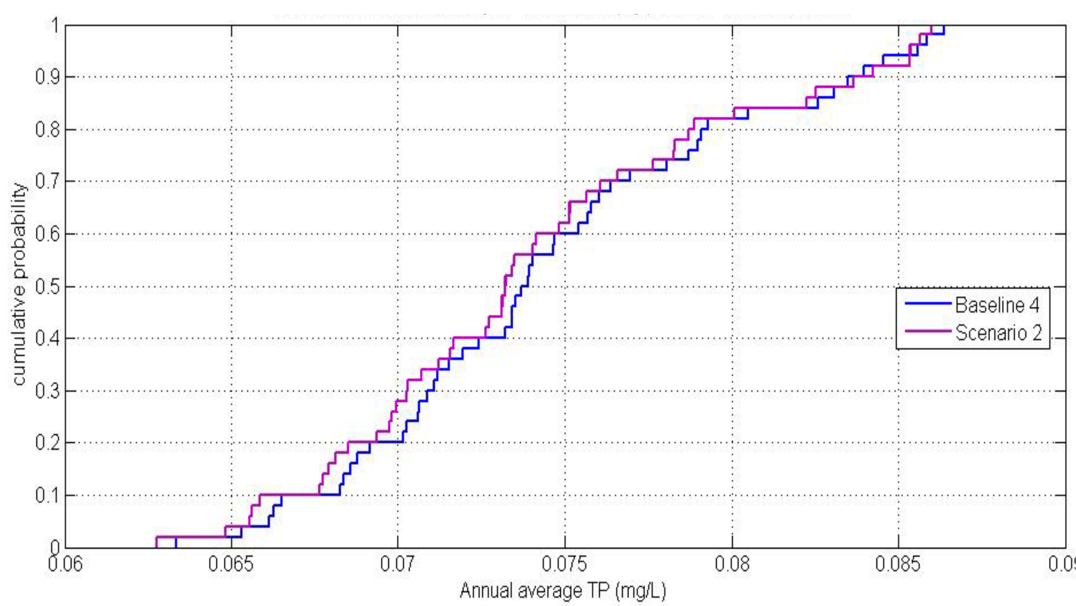


Figure 5-30: CDFs of WY2001 average TP concentration at Little Falls intake in baseline and trade scenarios at no-diversion condition

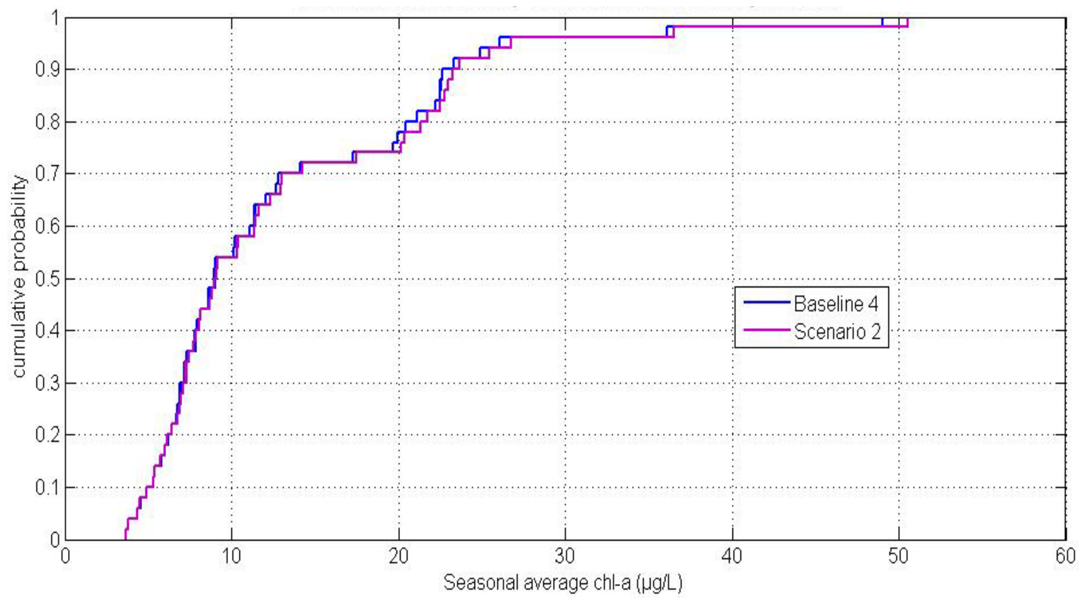


Figure 5-31: CDFs of WY2001 seasonal average chl-*a* concentration at Dundee Lake in baseline and trade scenarios at no-diversion condition

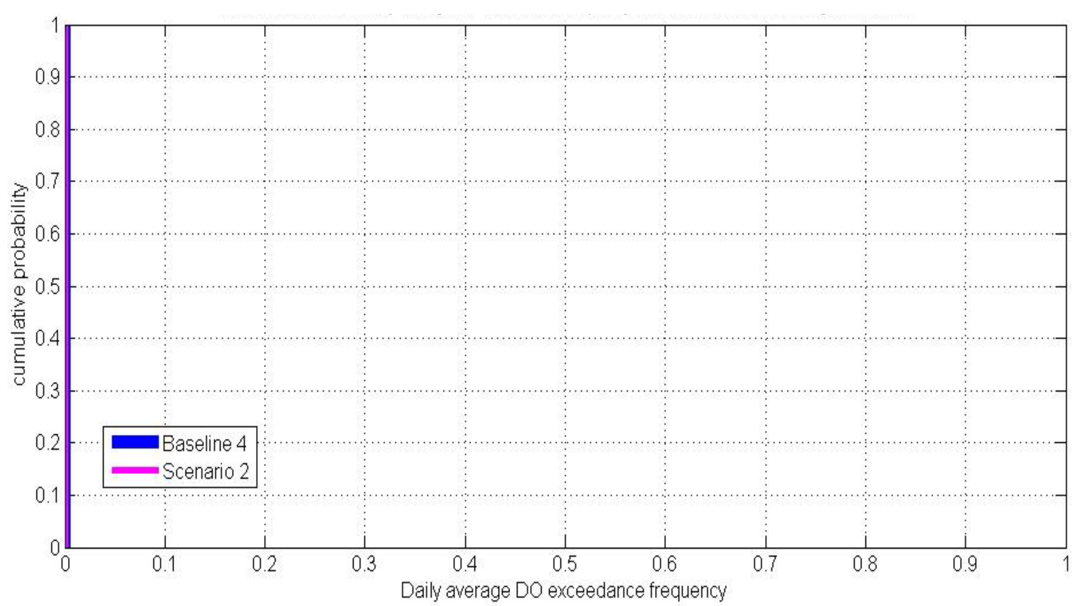


Figure 5-32: CDFs of June 15- Sep. 1, 2001 daily average DO exceedance frequency at Dundee Lake and station PA10 in baseline and trade scenarios at no-diversion condition

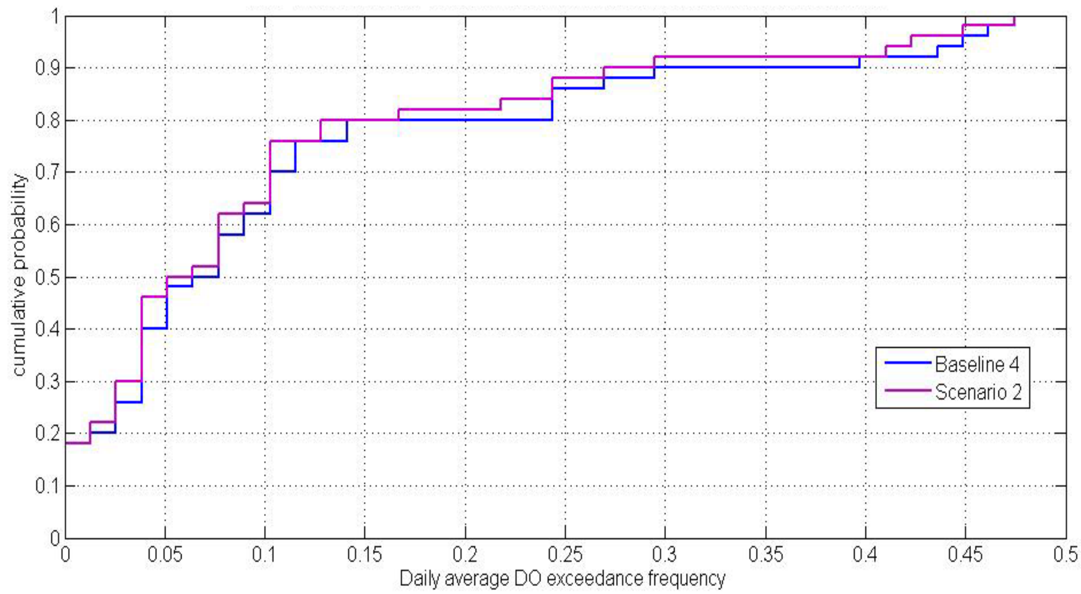


Figure 5-33: CDFs of June 15- Sep. 1, 2001 daily average DO exceedance frequency at Passaic River near Chatham in baseline and trade scenarios at no-diversion condition

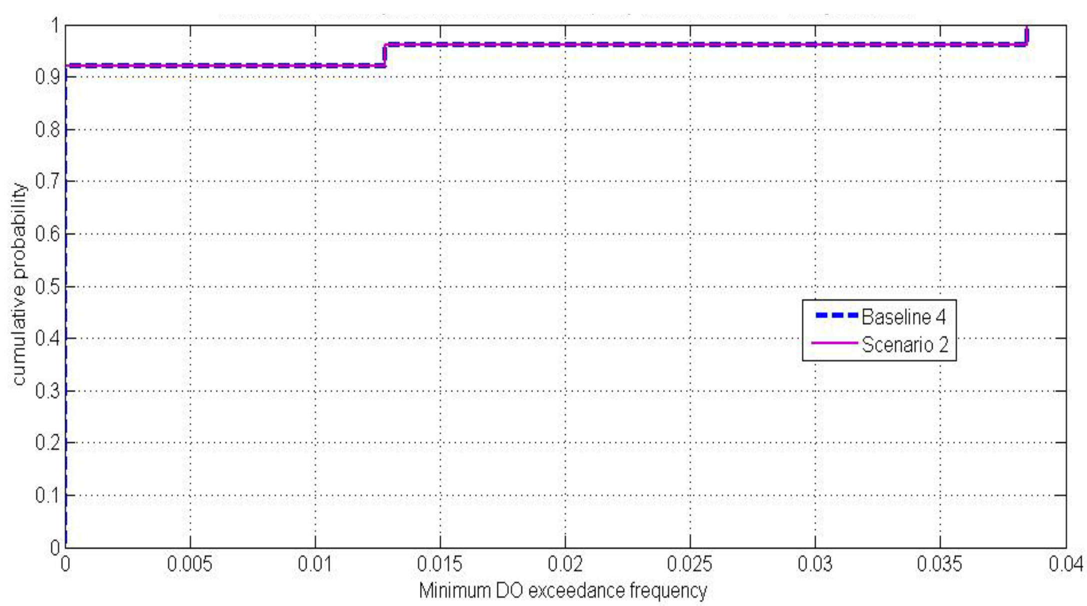


Figure 5-34: CDFs of June 15- Sep. 1, 2001 minimum DO exceedance frequency at Dundee Lake in baseline and trade scenarios at no-diversion condition

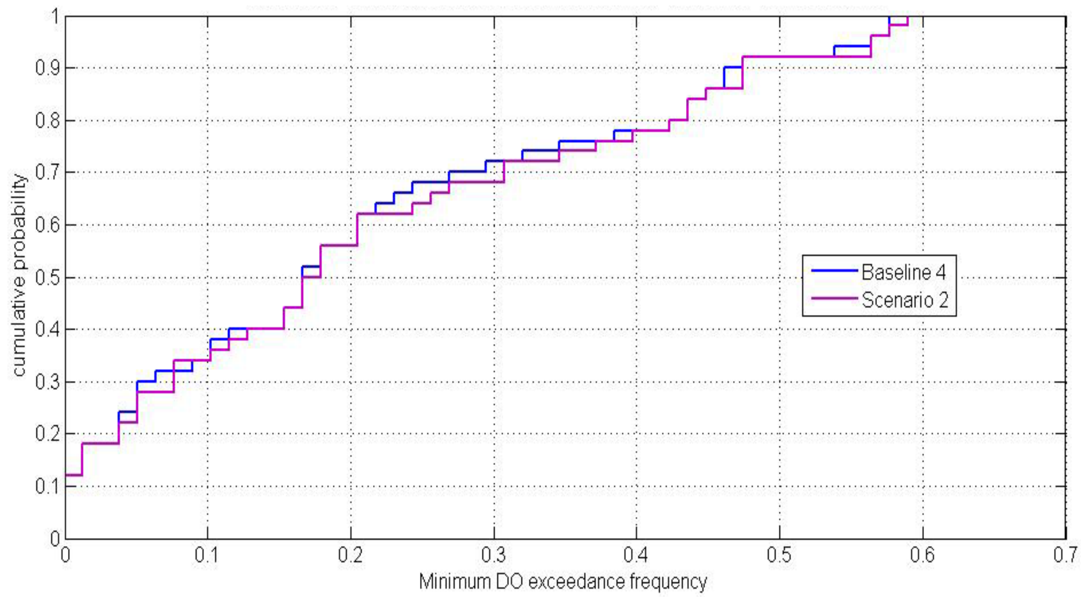


Figure 5-35: CDFs of June 15- Sep. 1, 2001 minimum DO exceedance frequency at Passaic River near Chatham in baseline and trade scenarios at no-diversion condition

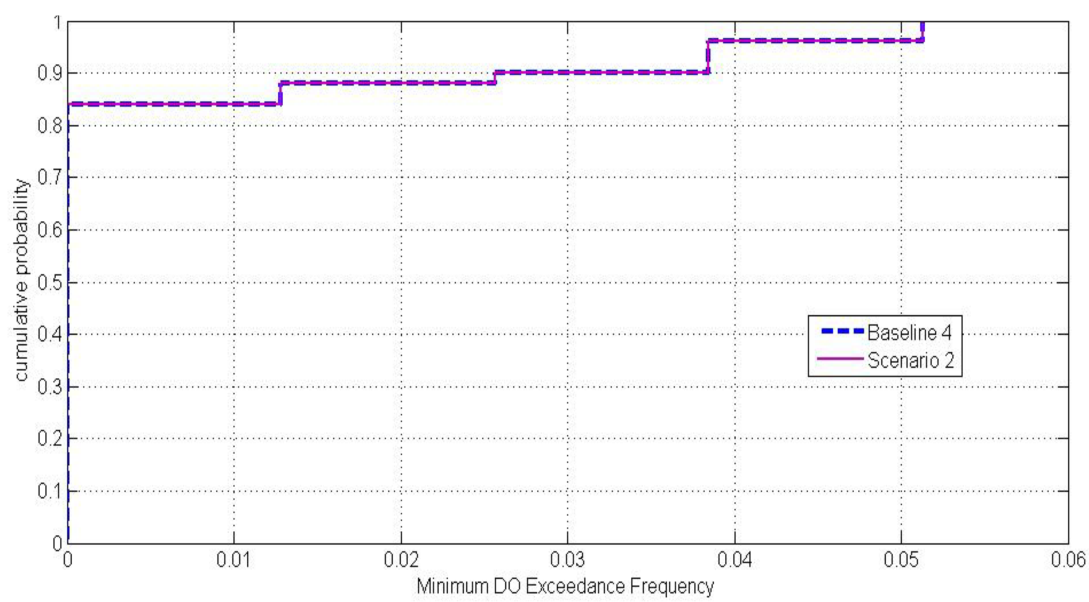


Figure 5-36: CDFs of June 15- Sep. 1, 2001 minimum DO exceedance frequency at station PA10 in baseline and trade scenarios at no-diversion condition

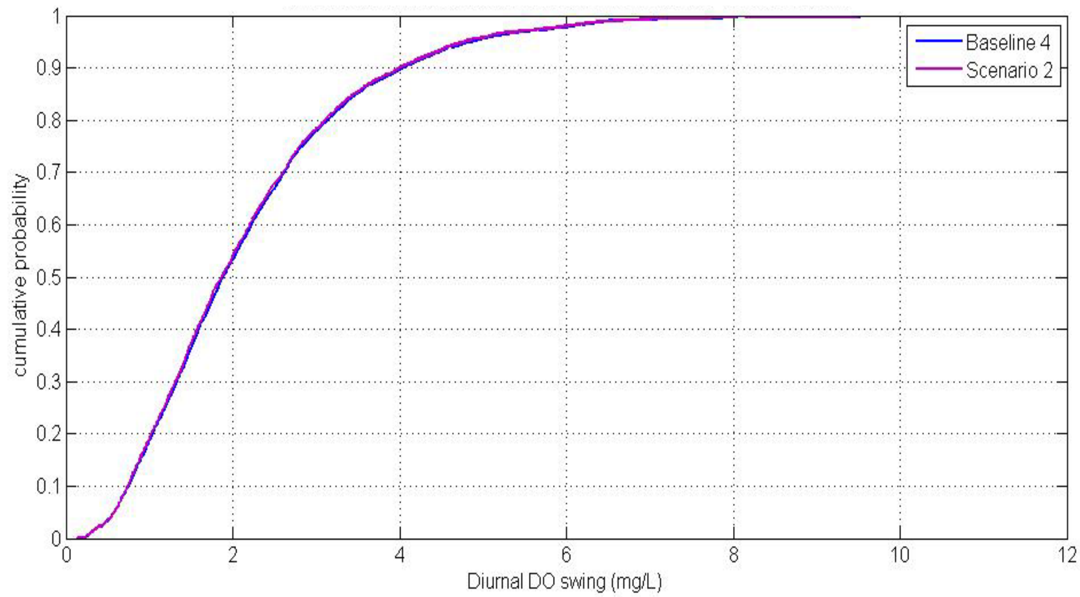


Figure 5-37: CDFs of diurnal DO swing from June 15- Sep. 1, 2001 at Dundee Lake in baseline and trade scenarios at no-diversion condition

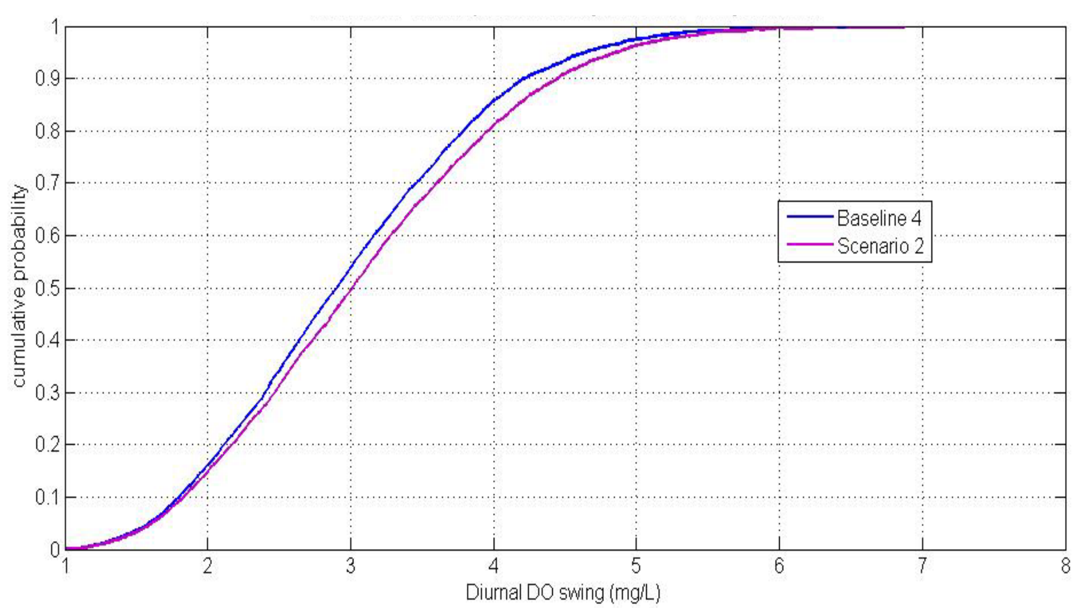


Figure 5-38: CDFs of diurnal DO swing from June 15- Sep. 1, 2001 at Passaic River near Chatham in baseline and trade scenarios at no-diversion condition

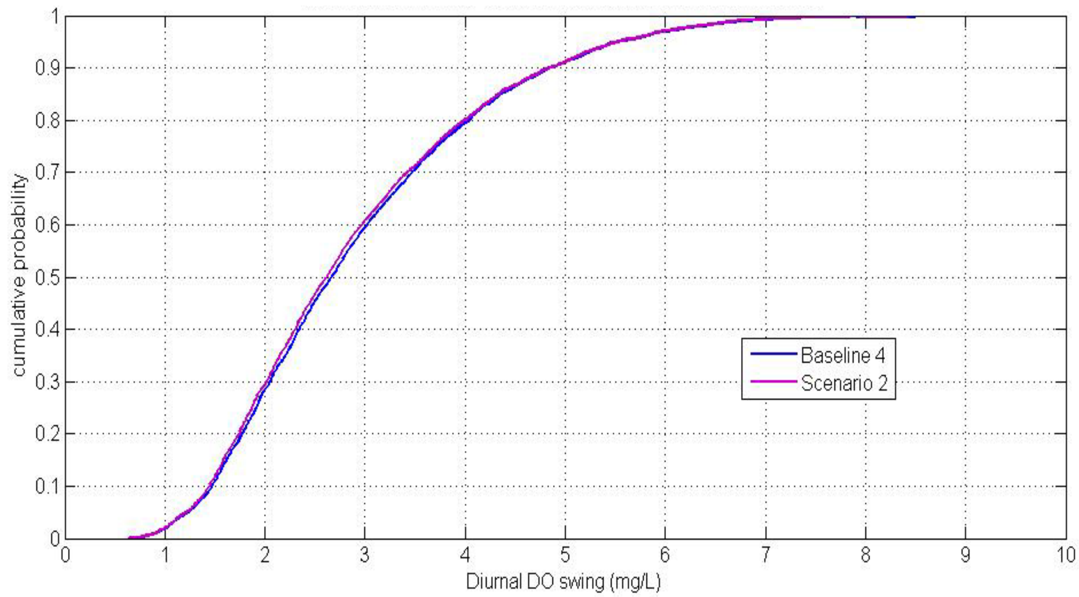


Figure 5-39: CDFs of diurnal DO swing from June 15- Sep. 1, 2001 at station PA10 in baseline and trade scenarios at no-diversion condition

Chapter 6: Communicating model uncertainty to the public

6.1 Introduction

The use of environmental models to inform policy and regulatory decisions in the U.S. has increased vastly in the past 25 years. Exponential growth in computer technology capability and data availability, combined with an increased understanding of environmental processes have allowed regulators and other stakeholders to further analyze the linkages of complex environmental problems and proposed policy options through the lens of environmental models. With respect to water pollution, the demand for water quality models has risen in response to the trend away from technology based standards toward water quality based effluent limits, and the need to develop numerous TMDLs (NRC, 2007).

Uncertainty analysis of environmental models plays a critical role in the dialogue between scientists and other stakeholders. As the influence of environmental models continues to grow and gain importance, the need for scientists to analyze and communicate about model uncertainty to other stakeholders becomes critical in maintaining or building trust in scientific findings.

6.2 Perspectives on models

The contemporary shift in environmental policy making from top-down decision making toward more stakeholder involvement in decision making has affected a change in the perception of what it takes to trust a model. “This not only involves the elements of model evaluation but also who will have a legitimate right to say whether they can trust the model and the decisions emanating from its application” (NRC, 2007). Communicating model uncertainty is very relevant to multi-stakeholder deliberations.

A related trend has been the change in perspectives on environmental models, especially within the modeling community. The perception of models has begun to transition from giving answers to offering insights. This reflects the acknowledgment that no matter how much more sophisticated modeling becomes, there can never be a perfect model that provides "the truth". Models, as abstractions of reality, fundamentally have uncertainty (NRC, 2007).

Two paradigms of models are present and in conflict amidst this period of transition. The newer paradigm regards the purpose of models as tools for deliberative problem solving among disparate stakeholders. In this paradigm, model uncertainty is an inevitable feature that needs to be explained. Model uncertainty does not necessarily undercut model reliability. Model evaluation centers on how well the model aided the problem solving process rather than its level of accuracy (Fisher et al., 2006). The fitness of a model's use is less dependent on eliminating uncertainty and more dependent on the transparency with which stakeholders reach decisions based on uncertain information (USEPA, 2003). In contrast, the older paradigm holds that the purpose of modeling is to prove that a regulation is supported by "sound science". According to this paradigm, model uncertainty is an undesirable feature that challenges the model's reliability and must be reduced. The model is primarily evaluated according to its level of accuracy. Modeling is strictly a scientific exercise in which public participation is inappropriate (Fisher et al., 2006).

6.3 Uncertainty in the science and policy arenas

There are differing views on the public reaction to model uncertainty. According to Fisher et al. (2006) the dominance of the older paradigm has built an unrealistic public expectation that a scientifically sound model contains no uncertainty, and any sign of uncertainty is a flaw that renders the model unreliable and unfit for use. In a larger sense, DeClercy (2005) points out that the public expects its leaders to convey certainty on issues that affect them. In contrast, Frewer (2004) argues that "elite groups in the science and policy community have underestimated the ability of non-experts to understand uncertainty" (p. 394). These groups have acted under the assumption that information on uncertainty would increase public distrust in science and scientific institutions, and cause panic and confusion on the issue. However, Frewer et al. (2002) argue that the converse appears to be true; when information on uncertainty is withheld, the public is skeptical and distrusts the motives of regulators and scientific advisors. Communicating about uncertainty can actually increase the credibility of the communicator, which in turn reduces perceived risk (Frewer, 2004).

When uncertainty is considered, the fundamentally different contexts of science and policy can cause different responses to it. Science and policy have different 'evidentiary standards'. Depending on the situation at hand and the perceived costs of being wrong, the policy maker may employ either stricter or looser evidentiary standards than the scientist. Consequently, a policy maker might misinterpret the degree of certainty reported by the scientist; the scientist might fail to report information that could be useful to the policy maker, or even withdraw from confronting uncertainty so as not to lose the confidence of the policy maker. Ultimately the response to uncertainty involves a debate about public values that scientists, who tend to bias toward considering themselves objective, are susceptible to being unaware of (Kinzig and Starrett, 2003).

There are several reasons why environmental regulatory agencies might seek to avoid analyzing or reporting model uncertainty. Once uncertainty is acknowledged, stakeholders could expect that models be updated regularly as new information is learned and that regulatory decisions be modified accordingly (NRC, 2007). Model uncertainty could paralyze policy actions if some stakeholders seek to delay a decision until more information is gathered. Some environmental officials have worried that if environmental model uncertainties are reported more often than uncertainties in other sciences, the public might get the wrong impression that environmental issues contain more uncertainty. Other officials express concern that uncertainty can be used as an excuse to avoid giving definite answers (Wardekker et al., 2008). Finally, the acknowledgment of model uncertainty could expose the agency to legal challenges, although in the U.S. the courts have generally sided with the agency on such challenges and explicitly recognized that models are simplifications of reality and do not require perfect accuracy to support a regulatory decision (McGarity and Wagner, 2003).

There are, however, compelling reasons that an agency should report model uncertainty. It promotes transparency and accountability through providing stakeholders with an assessment of the degree of confidence associated with model results, as well as information about which aspects of the model have the largest impact on its results (NRC, 2007). It is responsible policy and good scientific practice to report model uncertainty when decisions are made based on limited scientific knowledge (Pascual, 2005; Wardekker et al., 2008). Misrepresentation of model uncertainty can lead to hugely embarrassing and damaging outcomes if and when the truth is exposed (Janssen et al., 2005). Finally, if important decisions that affect the public are made based on the information provided by models, it obstructs the democratic process to exclude information concerning model uncertainty (NRC, 1996).

6.4 Effective communication of model uncertainty

Once a scientist or agency decides to report its findings on model uncertainty, the effective communication of those findings is critical. If not done properly, other stakeholders could get confused by the information or lose confidence in the overall analysis, thereby complicating the decision making process (NRC, 2007). Effective communication of model uncertainty is vital when dealing with legal challenges to the model (Pascual, 2005). It also serves the broader purpose of building public trust in science through countering the prevailing trend of insulating scientific discourse from the public (Patt and Dessai, 2005). Interestingly, early communications from the Intergovernmental Panel on Climate Change (IPCC) to decision makers explicitly neglected to mention probabilistic results because they assumed the audience could not understand that type of information. This damaged the IPCC's credibility and later efforts have completely turned around with full disclosure of uncertainty deemed essential (Patt and Dessai, 2005).

Effective communication of model uncertainty depends first on identifying the target audience. Knowledge of their concerns, needs, comfort with technical information, and the overall policy context will shape what information is essential to include and what can be left out (Krupnick et al., 2006). Key elements to communicate include the basic model concept, model assumptions and limitations, a history of the model development and evaluation process, quality of the data used, the sources of uncertainty, the probabilities of various outcomes, and the likelihood and impacts of reducing uncertainty (Frewer, 2004; NRC, 2007). Acknowledgement should be made of what is known and not known and the consequent policy implications. Comments on the rigor of the uncertainty analysis, and mention of any similar studies should be given. The information should be conveyed in a simple and concise manner that uses plain language, avoids abstractions and technical jargon, and offers clear graphs and tables (Sandman, 1987a; USEPA, 2003). Web-based tools are a promising way to promote widespread and interactive means of understanding model uncertainty (NRC, 2007).

The manner in which uncertainty is presented affects how it is perceived and can influence decisions. "An option framed in terms of its probability of success is seen as more attractive than the same option presented in terms of its complementary probability of failure" (Krupnick et al., 2006, p. 173). Fox (1984) recommends giving a narrative containing qualitative assessments with quantitative technical support. Graphs and tables can be helpful if designed thoughtfully. Krupnick et al. (2006) reports that box and whisker plots, CDF plots, and PDF plots perform well with decision makers, while area and volume plots should be avoided. Depending on the audience, Finkel (2002) suggests giving a point estimate such as the median or 90th percentile rather than the entire PDF.

There are different opinions on whether verbal or numeric terms are more effective for communicating uncertainty. In addition to providing graphs, the IPCC (2001) used a seven point scale of verbal terms such as 'likely' and 'very likely' to distinguish levels of certainty. Words are easier to remember than numbers; however, the disadvantage is that people subjectively define terms such as 'likely' or 'very likely' (Wardekker et al., 2008).

Comparisons can be an effective means of communicating uncertainty by helping to make the idea of probability less abstract (Patt and Dessai, 2005). Sandman (1987a) cautions that when making risk comparisons, the source of the risk comparison must be credible, the overall situation should not be heavily laden with emotion or hostility from stakeholders, the comparison must be conveyed with the intention of clarifying the issue rather than minimizing or dismissing it, and the comparison should acknowledge that factors other than relative risk are important.

The process through which communication takes place is vital to success. The most important, most obvious and yet easiest to ignore aspect of communication is that it is a two-way process. Early efforts at risk communication failed largely because scientists and agencies directed the public without being open to dialogue. Thus public values were not considered in expert assessments of risk and the public increasingly ignored their advice (Patt and Dessai, 2005). Attitudes have shifted somewhat. Several high profile reports (e.g., NRC, 1994, 1996, 2004, 2007; CRAM, 1997a,b) have been released which espouse the merits of stakeholder involvement throughout the problem solving process in achieving effective communication. However, if progress is to be judged by the state of adaptive management, a field that heavily advocates stakeholder involvement, major changes in social and institutional norms need to occur before agencies meaningfully collaborate or share power with diverse stakeholders from the beginning to end of a problem solving process (Allan and Curtis, 2005).

6.5 Lessons from risk communication: Outrage management

Risk assessment typically accounts for both the probability and severity in consequence of an outcome. For example, a highly severe highly probable event has more risk than an event of equal severity but less probability. The uncertainty analysis completed in this study is not a formal risk assessment, but its explicit consideration of the probability of outcomes and the magnitudes of difference between trade scenario and baseline outcomes has enough common ground with risk assessment that lessons from risk communication are relevant.

Practitioners of risk communication have come to realize that the public perception of risk is not just based on the technical data presented. Psychological factors can have a major influence on the perception of risk. For example, involuntary risks tend to be overestimated while voluntary risks are underestimated (Sandman, 1994). Sandman (2003) uses the term "outrage" to describe the nontechnical component of risk; it refers to the assortment of nontechnical factors such as voluntariness, trust, control, fairness, dread, and responsiveness that combine to affect the overall perception of risk. "Outrage is the principal determinant of perceived hazard" (Sandman, 2003). It is such a real and important variable that Sandman (1987b) asserts that risk is the sum of hazard and outrage. Therefore successful risk communication consists of two tasks – to explain the technical risk and take actions to reduce the level of outrage. Sufficiently addressing outrage is a precondition to successfully conveying technical information on the hazard.

Water quality trading in the NTPRB is an issue that has provoked outrage among some local environmentalist NGOs. Judging from public comment #77 in the TMDL (NJDEP, 2008a) and an opinion expressed in a local media outlet (Tittel, 2008), their outrage stems from several factors. They claim that water quality trading will cause environmental harm, allow dischargers to get away with not upgrading pollution control technology, invite unrestrained and manipulative market forces to create economic and political inequities, and abandon the public to leave it unprotected. Outrage was also expressed at being excluded from the establishment of the trading system, the commoditization of a pollutant, and a distrust of government enforcement ability against violators. These complaints echo concerns in the literature about emissions trading (e.g., Solomon and Lee, 2000; Farrell and Lave, 2004; Berck and Helfand, 2005) pertaining to hot spots, threats to environmental justice, inequities, the immorality of commoditizing the environment, and market manipulation.

In a larger context, current political and economic conditions in the country might affect a wide-sweeping critique and reevaluation of water quality trading. Critics of water quality trading might argue that it is a Bush administration policy that seeks to deregulate environmental protection using a market-based approach. The severity of these charges would be amplified in light of the current financial crisis, which has increased public skepticism towards deregulation and unrestrained free markets. The lack of results in water quality trading activity nationwide, despite extensive efforts from EPA and USDA to promote it, renders WQT even more vulnerable to these potential charges. If these types of attacks are initiated, it would exacerbate the outrage of environmentalist NGOs opposed to WQT in the NTPRB.

Given these outrage factors, the communication of model uncertainty, i.e., the ‘hazard’ part of the risk equation, by itself will not be enough. Of course the communication of model uncertainty must be done effectively to prevent outrage from getting worse, but it cannot be the only component of a successful risk communication strategy.

Risk communication seeks a level of outrage that is commensurate with the level of hazard (Sandman, 2008a). In this case the hazard level is low, as demonstrated by Chapter 5 results that showed that under the most adverse conditions, there is no evidence to suggest that WQT in the NTPRB will lead to worse outcomes than a command and control approach that prohibits trading. Although the hazard level is low, the outrage level of some critics is very high. The risk communication strategy most appropriate to this situation is termed ‘outrage management’. Outrage management recognizes that when people are angry about an issue that is low in hazard, the problem is not that they do not understand the numbers, but rather are too angry or upset to calm down, trust the source that is conveying the technical information, and consider the data. In order to be listened to, the risk communicator must first do the listening and acknowledge why the stakeholder is entitled to be outraged. Only after the outrage is addressed can technical information be presented as trustworthy (Covello and Sandman, 2001). Essential outrage management methods include active listening, aiming for the middle ground position rather than the opposite extreme of the critic, acknowledging prior misbehavior and current problems, discussing achievements with humility, sharing control or at least being accountable, and subtly drawing out unvoiced concerns and motives (Sandman, 2008a).

It should be noted that the environmentalist NGOs who are opposed to WQT in the NTPRB may or may not represent a substantial fraction of local environmentalist and citizen attitudes to the issue. Furthermore, WQT in the NTPRB appears to have the support of other stakeholders in the watershed, such as the WWTPs and NJDWSC. However, successful risk communication requires engaging with critics, no matter how intractable they may seem. If they are ignored rather than engaged, their outrage will grow and they will probably find other means to disrupt the process, such as broadcasting their views through the media and/or organizing other activists and citizens to support their cause. They may not be won over or convinced through engagement, but at least other stakeholders including moderate critics and neutral parties might perceive that the agency is making a serious effort at reasoned dialogue with its fiercest opponents, and thus be less likely to join the opposition (Sandman, 2003b).

6.6 Conclusions: Recommended strategy for a public meeting on water quality trading in the Non-Tidal Passaic River Basin

Applying the principles of outrage management and communication of model uncertainty might help NJDEP in its future public participation efforts on the Passaic water quality trading program. This section outlines a strategy for conducting a public meeting with outraged stakeholders and other groups. A public meeting is probably not the only form of outreach that NJDEP will spearhead. A more general public information campaign will also likely take place. However this section focuses on the public meeting component because that is where real outrage management can occur – and successful risk communication will be determined by successful outrage management.

Some of these steps might seem surprising, counterintuitive, or even naïve, especially given the long history of stakeholder conflict in this watershed. However, the impedance that outrage poses to risk communication is supported by an extensive body of research (e.g., Johnson et al., 1992; Sandman et al., 1993; Sandman et al., 1998), and the techniques of outrage management have been sought by a wide variety of public, private and non-profit organizations working in sectors such as biotechnology, petrochemicals, defense, law enforcement, mining, and public health (Sandman, 2008b). This should be kept in mind while reviewing the recommended steps below.

Since the purpose of the public meeting is outrage management, the target audience is the local environmentalist NGOs who are most outraged about the trading program. Other invitees should include moderate environmentalist NGOs and community leaders from poor and minority areas in the downstream portion of the watershed. If outrage management is executed effectively, the participation of the latter two groups at the meeting increases the chances that they will either remain neutral or side with the agency. Agency leaders should attend so that the audience will grasp how seriously their concerns are taken. Technical experts should attend, including Rutgers University professors that helped to design the trading program, in order to credibly communicate technical information. Agency personnel and technical experts should meet in advance to review their objectives and rehearse the messages they aim to convey. Supporters such as the WWTPs and NJDWSC should not be invited because they will probably disagree with the degree of empathy outrage management requires, and instead pressure the agency to fight back, further entrenching the conflict (Sandman, 2008a).

Ultimately the goal of the meeting is to listen to the outrage, seek to address it, and gain enough trust from enough participants that technical information describing the low hazard level can be received as objectively as possible.

NJDEP should consider allowing an outside facilitator to help conduct the meeting. Although many facilitators seek to avoid allowing a discussion where outrage is vented (Sandman, 2008a), for the purposes of this meeting, the facilitator should encourage venting of outrage before proceeding to guide a substantive discussion of the issues.

Be resigned to a long meeting. Trying to shorten the meeting implies there is something to hide. By allowing the audience to determine when the meeting will end, the agency shows that it takes their concerns seriously, and makes an important gesture of relinquishing control (Reeves, 2007).

The meeting should begin by letting the critics vent their outrage. Agency personnel should listen, empathize, and acknowledge their reasons for outrage. The reasons for outrage, having been already voiced in the TMDL public comments and local media, will probably include:

- Water quality trading will cause environmental harm,
- Dischargers will get away with not upgrading pollution control technology,
- Unrestrained and manipulative market forces will create economic and political inequities,
- It is immoral to commoditize the environment and a pollutant,
- The public will be abandoned and left unprotected,
- The government cannot be trusted to enforce against violators,
- Their organizations were excluded from the establishment of the trading system.

To address those concerns the following are offered as talking points, all of which reflect the most current understanding of the water quality trading program as it will be implemented, unless otherwise noted:

- NJDEP will closely regulate the Passaic water quality trading program in a transparent manner. The proposed trading program will not function as a free and unconstrained deregulated market. Proposed trades will require approval from NJDEP. NJDEP has the right to veto proposed trades, and environmental justice will be one of the evaluation criteria. Trades will be written into draft discharger permits; those permits will be subject to public comment. Public participation will be sought at each significant decision point in the process.

- NJDEP is responsible for closely monitoring discharges of trading partners so that trading commitments are fulfilled. NJDEP will annually verify that the trading obligations included in a discharge permit have been met. NJDEP will ensure that extensive water quality monitoring throughout the watershed occurs to ensure that trading does not create hot spots. All the data described in this bullet item will be made publicly available to demonstrate the transparency of the program.
  - Furthermore, to demonstrate accountability and a sincere effort to involve stakeholders, NJDEP should strongly consider inviting those local environmentalist NGOs most critical of the trading program along with a neutral third party (e.g., TMDL advisory panel) to have oversight of the data collection tasks described above. While this recommendation might seem unusual or infeasible, note that in outrage management, sharing power is the ultimate path to gaining trust. Realizing that most institutions would decline to do that, the next best thing to sharing power is being accountable. Offering the role of oversight to the NGO builds trust without having to share power. Having a neutral third party also handle oversight ensures the NGO acts properly in its role (Sandman, 2002).

- In case of noncompliance by a trading partner, NJDEP may either bring enforcement proceedings or move to withdraw or modify a permit.
- An analysis by economists from Cornell University indicates the water quality trading program is expected to save up to 18% of the costs of a command and control approach (Boisvert et al., 2008). Those are costs that would otherwise be borne by the public. These savings are especially significant in a time of economic crisis.
- The water quality trading program was designed with numerous safety features, such as a management area approach with conservative trading ratios, to protect water quality under worst case conditions (Obropta et al., 2008).
- An extensive uncertainty analysis was done to investigate if water quality trading increases the chance of degradation compared to a command and control approach. Those results will be presented next.

Explaining those points in a calm and humble manner, as opposed to an agitated and pompous manner, should sufficiently address the stakeholder outrage that enough people attending the meeting would be willing to trust the information presented on model uncertainty (Sandman, 1994).

A sample fact sheet describing the model uncertainty analysis is provided. Note several features about the fact sheet which should be highlighted in the public discussion:

- Model uncertainty is normal and inevitable.
- The basic model concept and key assumptions are explained.
- The history of the model's development and evaluation with respect to this watershed are explained.

- The scope of the uncertainty analysis is described.
- The key sources of model uncertainty and their effects are described.
- An explanation is given of the methodology for comparing trade scenario to baseline scenario outcomes, along with simple instructions for how to interpret the graph.
- Clear maps are used to provide the context of potential hot spot locations, particularly with respect to environmental justice concerns.
- One simple table shows the results of the uncertainty analysis.
- A brief discussion sheds insight on the two outcomes where trading was not as good as the command and control approach.
- Future steps to reduce model uncertainty are mentioned.
- Nontechnical language is avoided as much as possible.

6.6.1 *Proposed fact sheet on model uncertainty and the Passaic water quality trading program*

Background

Computer models are commonly used in efforts to understand the environment. For example, climate change models have been critical to understanding the relationship between greenhouse gas emissions and global warming. Models are a useful tool because often times, the environment being studied is so big (e.g., a watershed, global climate, aquifer) that it cannot possibly be isolated and studied in a laboratory. A model is "a simplification of reality that is constructed to gain insights into [a system]" (NRC, 2007, p. 31). Models are particularly useful for trying out different what-if scenarios; models make predictions which help stakeholders to decide on the best management strategy for an environmental problem.

Models by definition are not reality – they are just a useful and simple way to describe reality. Because models are not perfect copies of reality, a model prediction inevitably contains some uncertainty. Fortunately, scientists are able to analyze the uncertainty of a model, and estimate the impact it has on model forecasts. In this case, an uncertainty analysis was done on the water quality model that was used to predict outcomes of various trading scenarios.

The Passaic Water Quality Model

Four models were linked to study the impact of phosphorus on the Non-Tidal Passaic River Basin (NTPRB). One of the models dealt exclusively with the Wanaque Reservoir. The other 3 models (e.g., flow model, nonpoint source load model, and water quality model) were used for the rest of the watershed. Of the four models, only the uncertainty of the water quality model was analyzed. This was because it contains significantly more uncertainty and influence on the overall results than the other three models. In the field of model uncertainty analysis, it is common practice to limit the scope of the analysis, since it is often too complex or inefficient to look at all the uncertainties. The uncertainty analysis of the Passaic model followed these common practices and focused on what was considered to be the main source of uncertainty.

The water quality model used for the NTPRB is an application of an EPA model called Water Quality Analysis Simulation Program version 7.0 (WASP 7). WASP has been around since the 1970s and has been widely used in both the U.S. and other countries. WASP is designed to represent the dynamic processes that link nutrients, algae and dissolved oxygen (DO) in rivers and lakes. A simple schematic of WASP is shown in Figure 6-1.

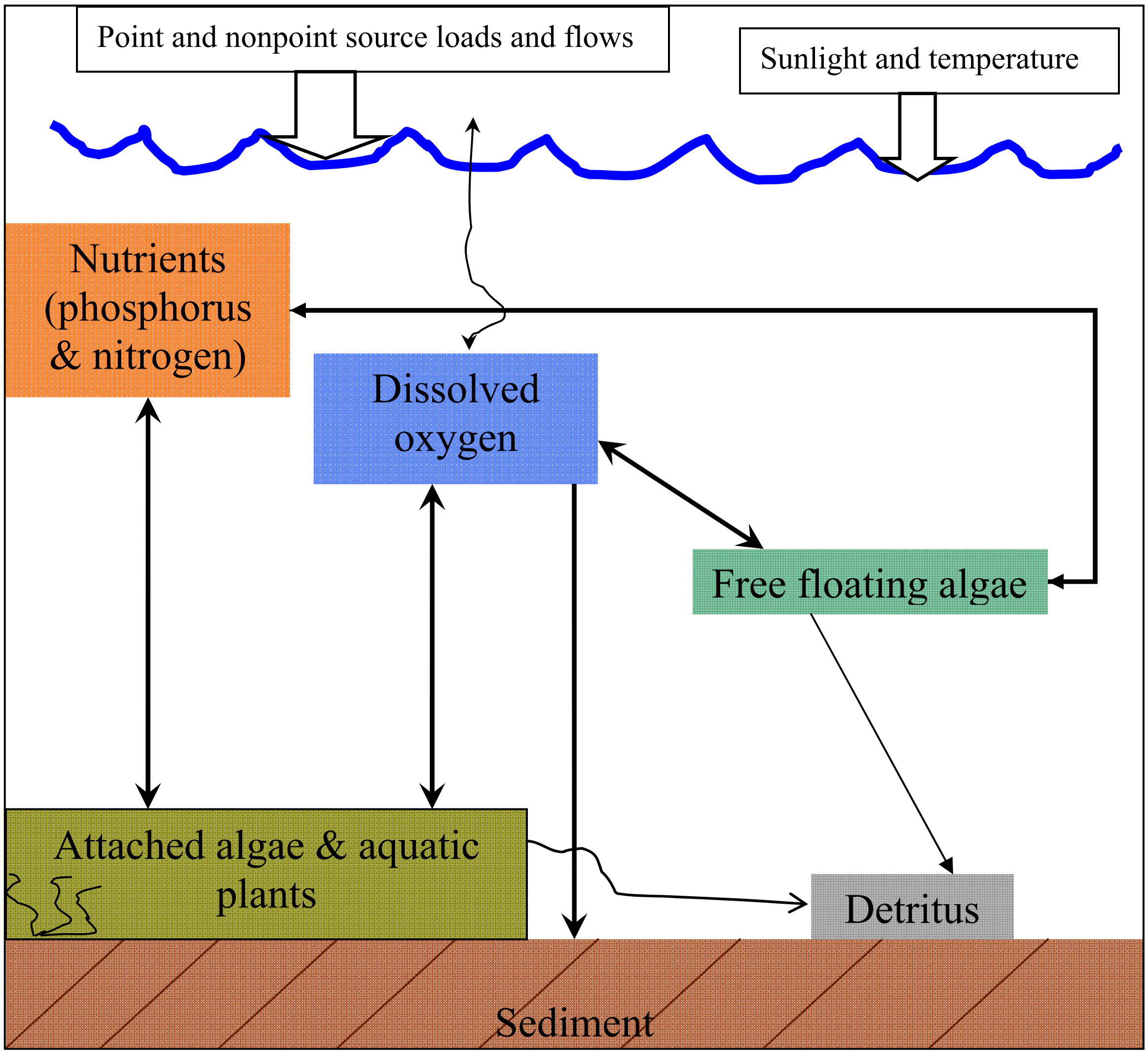


Figure 6-1: Simplified schematic of the WASP model

NJDEP contracted a private company (Omni Environmental Corp.) with modeling expertise to develop a WASP model of the NTPRB (Omni Environmental, 2007). An independent expert panel, the Rutgers EcoComplex TMDL advisory panel, reviewed the efforts of Omni Environmental and provided feedback and guidance as needed. The model results were evaluated against real watershed data collected from 1999-2003, and the model was judged to be state of the art. Model results were used to inform decisions in development of the Passaic TMDL for total phosphorus.

Uncertainty Analysis of the Passaic Water Quality Model

Like all models, the WASP model of the NTPRB contains uncertainties. A study determined that the uncertainty mainly comes from estimated rates of river processes (e.g., growth rate of algae) and the assumed pollutant content coming from wastewater treatment plants and upstream tributaries. The uncertainty analysis explicitly accounted for those sources of uncertainty because they greatly affect predictions of total phosphorus, algae, and DO levels. Other less important uncertainties which were not studied were a detailed portrayal of each algae and aquatic plant species, and all the ways that the water and sediment interact.

The uncertainty analysis was applied to model predictions of various trading scenarios. Specifically, the analysis examined if there was any evidence that water quality trading would create or exacerbate hot spots to any extent beyond the command and control approach. Model predictions of DO, total phosphorus, and free-floating algae from a range of worst-case trading scenarios were analyzed and compared to corresponding baseline scenarios in which trading did not occur. Standard statistical techniques were then used to determine if the trading scenario outcome was significantly worse than the baseline scenario outcome. (Worst-case trading scenarios were tested because if trading is ok under worst-case conditions, then it should be ok under less adverse conditions).

Here's an example. The graph below (Figure 6-2) compares the range of likely outcomes for two scenarios: "[Trading] Scenario 3" and "Baseline 5". What's shown is the amount of phosphorus load that would be diverted from the Wanaque South intake to the Wanaque Reservoir over the course of an entire year. The less phosphorus that is diverted, the better it is for the Reservoir. Each line in the graph represents the prediction for a particular scenario; the blue line is the baseline scenario, and the purple line is the trading scenario. Where the blue line is to the right of the purple line, the baseline scenario has a higher phosphorus load than the trading scenario. Notice that both lines look very alike. That means that both scenarios contain very similar magnitudes of uncertainty, and very similar probabilities of achieving the same outcomes. A simple statistical test can tell us if one outcome is significantly higher than the other. If the trade scenario is significantly higher, then we conclude that trading would be worse than a command and control approach. However in this case, the statistical test tells us the trade scenario is **not** significantly higher, so we conclude that there is insufficient evidence (or "no evidence" since the test results are very clear) to claim that trading would be worse than a command and control approach.

This method was applied to look at several types of worst-case trade scenarios and their outcomes relative to a command and control approach at several potential hot spots throughout the watershed. Figures 6-3 and 6-4 depict the potential hot spot locations, overlaid on 2000 census tract information regarding income levels and minority populations.

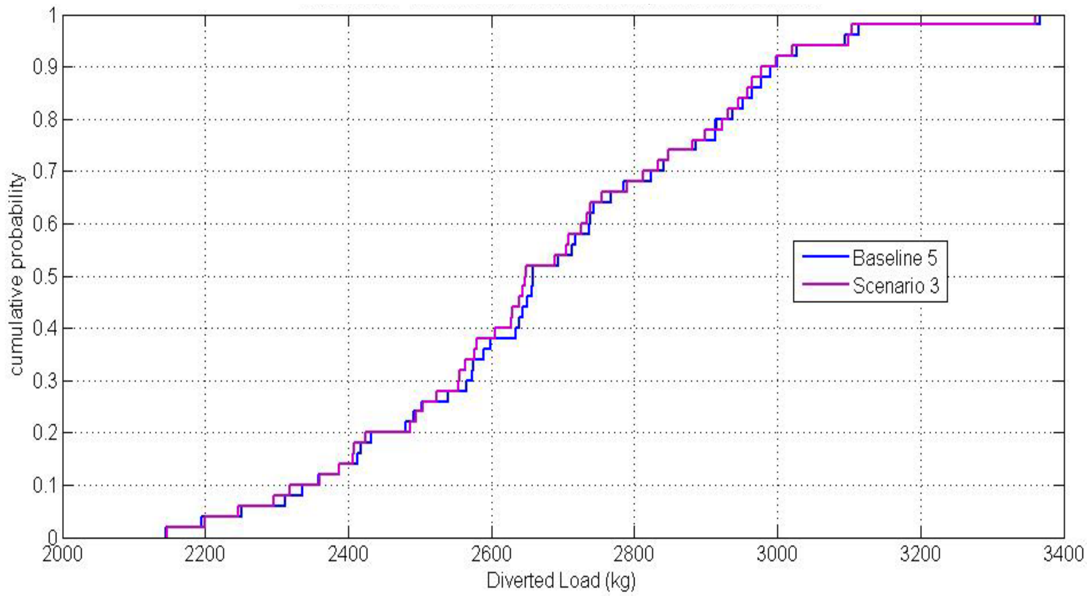


Figure 6-2: Comparison of uncertainty estimates for a trading and baseline approach. The graph shows diverted phosphorus load from the Wanaque South intake to the Wanaque Reservoir over the course of one year.

Table 6-1 shows the results of the uncertainty analysis on Passaic water quality trading. The results show that for almost all outcomes, there is no evidence to suggest that trading under the worst-case conditions will not be as good as the command and control approach. There were two worst-case outcomes that suggested a negative effect on diurnal DO swing at the Peckman River mouth and the Passaic River near Chatham; however in each of those particular cases trading also caused a positive effect at the Wanaque South intake, by reducing the phosphorus load diverted to the Wanaque Reservoir by 15-18%. The benefits of less phosphorus in the Wanaque Reservoir reach all residents in the watershed. It should also be noted that contrary to environmental justice concerns, of the two negatively affected locations, one is an affluent and predominantly white area – the Passaic River near Chatham. The precaution of additional monitoring of diurnal DO swings could be implemented at both the Peckman

River mouth and Passaic River near Chatham in the event that trades are implemented near those locations.

Future steps

The uncertainty analysis has provided a guide to future data collection efforts that will support an adaptive management approach. Learning more about specific variables in the model will allow the uncertainty analysis to be updated. Model uncertainty might be reduced, but keep in mind models can never be totally free of uncertainty.

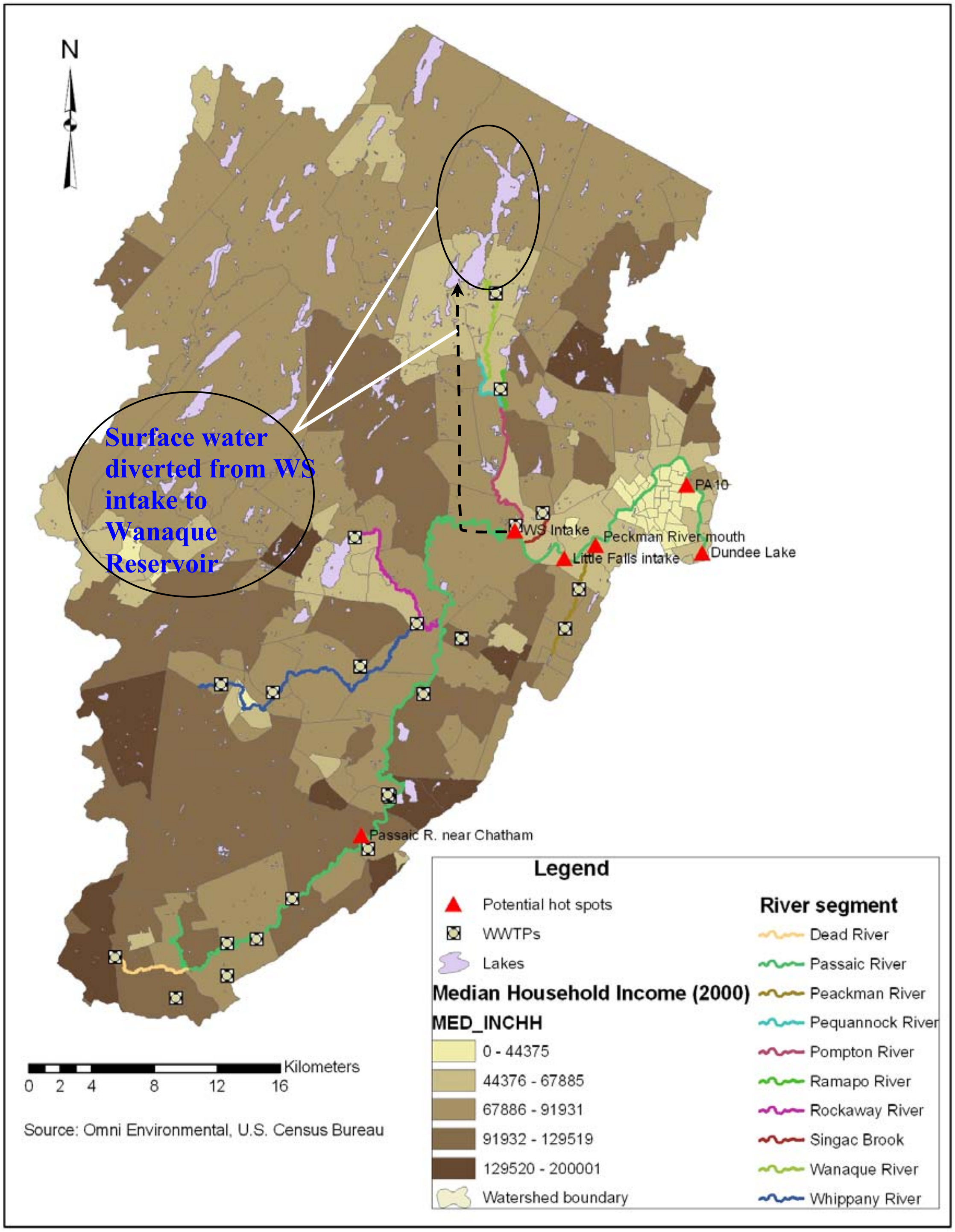


Figure 6-3: Potential hot spots in the Non-Tidal Passaic River Basin, overlaid with 2000 Census tract data on median household income

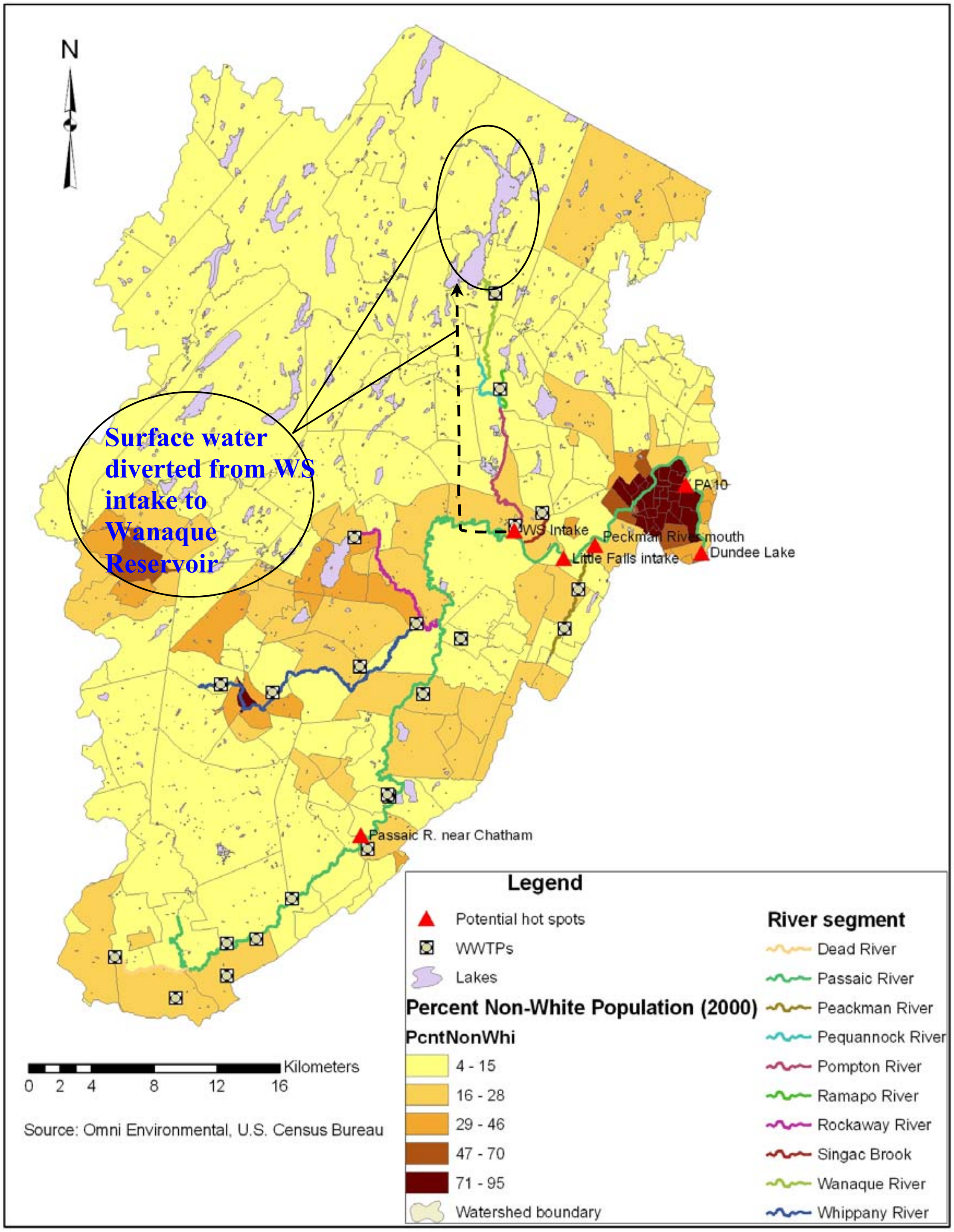


Figure 6-4: Potential hot spots in the Non-Tidal Passaic River Basin, overlaid with 2000 Census tract data on percentage minority population

Table 6-1: Results of uncertainty analysis on Passaic water quality trading

Location	Outcome	Is there evidence to claim that trading under worst-case conditions will not be as good as the command and control approach?	If there is evidence, what is the expected magnitude of degradation?
Wanaque South intake*	Annual diverted load of total phosphorus	No	-
Dundee Lake*	Seasonal average of chlorophyll-*a*	No	-
Dundee Lake*	Exceedance frequency of the daily average dissolved oxygen standard during the algae growing season	No	-
Peckman River mouth		No	-
Passaic River near Chatham		No	-
Station PA10		No	-
Dundee Lake*	Exceedance frequency of the minimum dissolved oxygen standard during the algae growing season	No	-
Peckman River mouth		No	-
Passaic River near Chatham		No	-
Station PA10		No	-
Dundee Lake*	Diurnal dissolved oxygen swing during the algae growing season	No	-
Peckman River mouth		Yes‡	11% (6.17 vs. 5.45 mg/L)
Passaic River near Chatham		Yes ^	10% (4.00 vs. 3.62 mg/L)
Station PA10		No	-
Little Falls intake	Annual average total phosphorus concentration	No	-

* Critical location as stated in TMDL (NJDEP, 2008a)
‡ Trading also caused an 18% improvement in phosphorus diverted to the Wanaque Reservoir
^ Trading also caused a 15% improvement in phosphorus diverted to the Wanaque Reservoir

Chapter 7: Conclusions

Water quality modeling is a major source of scientific uncertainty in the TMDL process; the effects of these uncertainties extend to water quality trading programs designed to implement TMDLs. A model uncertainty analysis strengthens decision support to policy makers and increases transparency to affected stakeholders. Models are tools, and their adeptness is increased by uncertainty analysis in many ways such as investigating whether the TMDL margin of safety is sufficient, or if pollutant trading will protect water quality and avoid creating hot spots. The primary objective of this research was to analyze the effects of model uncertainty on the total phosphorus TMDL for the Non-Tidal Passaic River Basin and an associated point-point source phosphorus trading program designed to implement the TMDL. A secondary objective was to demonstrate that uncertainty analysis of water quality models is an essential step for the development of future water quality trading programs.

7.1 Results

Through comparison of probabilistic model output with observed data, estimates of DO uncertainty are credible at Dundee Lake, the Peckman River mouth, and station PA10, but not at the Passaic River near Chatham. Estimates of chl-*a* uncertainty are credible at Dundee Lake, although this conclusion was reached based on a small sample size of observations at a location 3 km upstream. Estimates of TP uncertainty at the Wanaque South intake are credible during times without any diversion or when only the Pompton River is diverted; when extreme diversions occur, estimates of TP uncertainty are credible provided that model output of TP is processed at a 4-day moving time scale due to limitations caused by gaps in observed data. These findings on credibility were crucial to establish before proceeding into uncertainty analysis of the TMDL and trading scenarios.

With respect to the first research hypothesis, the Passaic TMDL will **not** result in attainment of site-specific chlorophyll-*a* criteria at Dundee Lake with less than 10% exceedance probability at critical drought conditions. However, despite the high exceedance probability for chl-*a* at Dundee Lake, the expected exceedance there is zero for daily average and minimum DO standards. The DO portion of the first research hypothesis is **supported**. The Passaic TMDL **will** result in attainment of dissolved oxygen surface water quality standards at Dundee Lake, with less than 10% expected exceedance at critical drought conditions. Although the TMDL margin of safety is not sufficient to ensure at least a 90% probability of meeting the seasonal average chl-*a* criteria, in terms of DO at Dundee Lake, the margin of safety is sufficient. Exceedances of chl-*a* criteria do not necessarily imply exceedances of DO water quality standards at the Dundee Lake site.

The second research hypothesis was **supported**. The Passaic 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. The TMDL margin of safety has been justified with respect to diverted total phosphorus load from the Wanaque South intake to the Wanaque Reservoir.

Even though the TMDL scenario shows expected exceedances or exceedance probabilities above the 10% target at certain locations, the efficacy of the TMDL measures was clearly demonstrated when compared directly to actual conditions in

WY2002. Significant improvements were reflected in reduced chl-*a*, TP diverted load, and diurnal DO swings at all relevant locations.

With respect to the final research hypothesis, the analysis **found no evidence** to suggest that the outcome of trades between WWTPs, as compared with command and control regulation, will significantly increase uncertainty in the attainment of dissolved oxygen surface water quality standards, site-specific chlorophyll-*a* criteria, and reduction targets for diverted total phosphorus load at affected potential hot spots in the watershed. The results fail to reject the assumption that phosphorus trading in the Non-Tidal Passaic River Basin can reliably achieve compliance with TMDL water quality goals under the most severe circumstances, such as cross-tributary or inter-management area trading, dry year effluent flows and critical diversion conditions.

Four outcomes, all regarding diurnal DO swing (for which there is no surface water quality standard), did show a significant increase in trading scenarios from the baseline. However the magnitudes of the deviations were small and ranged from 4 to 11% in terms of mean value, standard deviation, and 90th percentile value. Considering the modest deviations from the baseline in diurnal DO swing, it is not recommended to prohibit the three discrepant trading scenarios, especially since 2 of the 3 scenarios yielded significant improvements in diverted TP load from the Wanaque South intake to the Wanaque Reservoir.

Research on risk communication techniques was also synthesized to help NJDEP in its future public participation efforts on the Passaic water quality trading program. A strategy was outlined for conducting a public meeting with ‘outraged’ stakeholders and other groups. The strategy applies the principles of ‘outrage management’ and recognizes that sufficiently addressing outrage is a precondition to successfully conveying technical information on model uncertainty.

7.2 Contributions of the results

The results described above illustrate five main contributions of the study. The first three contributions directly address research needs cited in the literature. First, it provided a rare formal assessment of a TMDL margin of safety. Second, with respect to the water quality trading results, it demonstrated the importance of comparing relative uncertainties. Third, it served as the first study to examine the effects of water quality model uncertainty on a nutrient trading program. As an added benefit, by conducting a thorough uncertainty analysis of a point-point source nutrient trading program, a lower bound on the range of uncertainty regarding nutrient trading programs in general has been obtained, which could benefit nutrient trading programs nationwide. Fourth, it provided a useful caution that exceedances of chl-*a* criteria might not necessarily imply exceedances of DO water quality standards at the Dundee Lake site, and that in order to fully describe water quality at the site, care should be taken to also report the exceedance frequency of the daily average and minimum DO standards, as well as the distribution of diurnal DO swings over the June 15 to September 1 period. Fifth, it supports adaptive management through providing prioritized guidance on which model variables and water quality parameters to collect further data on.

7.3 Contributions of the methodology

The method introduced in this study 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. Furthermore, the study produced not only a credible uncertainty analysis, but an efficient analysis whose LHS-based method could easily be replicated by regulators charged with administering a water quality trading program and assessing its various risks. The method's smooth efficiency and practicality directly address a main obstacle that has hindered a wider practice of uncertainty analyses of water quality models.   An important qualification is that this research benefited from having a calibrated model already in place. Without a calibrated model, estimates of parameter probability distributions would have been more difficult, and more model runs would have been required. However, the simplicity of assessing the uncertainty analysis credibility would have remained the same, and the research contribution of that portion of the methodology – the most powerful piece of the methodology - would have remained intact.

Other methods such as Bayesian network models are more comprehensive in scope than what was applied here. However, without necessarily intending to do so, the methodology of this study appears to have offered one way to meet the recommendation by NRC (2007) to pursue "hybrid approaches" to uncertainty analysis that mix deterministic and stochastic methods and avoid probabilistic representation of every single uncertainty. In addition, in keeping with the life cycle approach to modeling recommended by NRC (2007), the parameter distributions that were estimated for this study can be easily updated once new information is acquired, thus enabling updating of model uncertainty estimates.

7.4 Future research

One avenue for future research could be coupling the nonpoint source load model with the hydrodynamic model. If structural BMPs such as retention basins are implemented on a large scale throughout the watershed in order to achieve the TMDL allocation of 60% phosphorus load reduction, the presumed effect of increased infiltration and baseflow and less runoff on surface water quality could be explicitly accounted for. A further step of including the uncertainties of BMP performance would expand the scope of the uncertainty analysis to include the nonpoint source load model and hydrodynamic model.

Another direction for future research would be to input the water quality model uncertainty analysis results of various trading scenarios into the economic model developed by Boisvert et al. (2008) and Sado (2006). The effect of this would be to obtain a probabilistic output for cost-effectiveness that could be achieved through water quality trading. This would expand the uncertainty analysis across disciplines and yield a powerful integrated assessment of the water quality trading program.

Yet another area for future research would be to compare the effect of different trading ratios. This would elucidate whether less conservative trading ratios pose a greater risk of creating hot spots than the current set of trading ratios.

Finally, the LAWATERS model (Najarin Associates, 2005) of the Wanaque Reservoir could be included in the uncertainty analysis. Results from this study on the uncertainty of diverted TP load from the WS intake to the Wanaque Reservoir could be directly linked to the LAWATERS model to obtain probabilistic output of state variables in the Wanaque Reservoir system.

APPENDIX

Table A-1: Observed and LHS-generated values for Cluster 1 TP effluent from Berkeley Heights STP (actual values in bold)

Date	TP (mg/L)	Date	TP (mg/L)	Date	TP (mg/L)	Date	TP (mg/L)
10/1/2000	2.87	11/22/2000	3.33	1/13/2001	2.98	**3/6/2001**	**3.30**
10/2/2000	2.14	11/23/2000	3.01	1/14/2001	4.07	3/7/2001	3.51
10/3/2000	**4.30**	11/24/2000	5.35	1/15/2001	3.88	3/8/2001	3.27
10/4/2000	5.06	11/25/2000	4.53	**1/16/2001**	**3.90**	3/9/2001	4.09
10/5/2000	2.7	11/26/2000	1.82	1/17/2001	4.13	3/10/2001	3.45
10/6/2000	4.24	11/27/2000	5.66	1/18/2001	2.97	3/11/2001	3.07
10/7/2000	5.09	**11/28/2000**	**3.10**	1/19/2001	5.08	3/12/2001	2.47
10/8/2000	4.01	11/29/2000	3.78	1/20/2001	3.46	**3/13/2001**	**2.50**
10/9/2000	4.61	11/30/2000	4.89	1/21/2001	4.11	3/14/2001	4.2
10/10/2000	**4.10**	12/1/2000	4.32	1/22/2001	2.82	3/15/2001	3.47
10/11/2000	2.14	12/2/2000	3.87	**1/23/2001**	**2.70**	3/16/2001	2.21
10/12/2000	2.89	12/3/2000	5.01	1/24/2001	3.26	3/17/2001	5.03
10/13/2000	4.15	12/4/2000	3.83	1/25/2001	4.87	3/18/2001	3.62
10/14/2000	4.84	**12/5/2000**	**3.50**	1/26/2001	2.58	3/19/2001	4.06
10/15/2000	3.79	12/6/2000	4.83	1/27/2001	3.59	**3/20/2001**	**3.10**
10/16/2000	4.64	12/7/2000	4.04	1/28/2001	3.98	3/21/2001	5.15
10/17/2000	**3.70**	12/8/2000	4.35	1/29/2001	3.08	3/22/2001	2.68
10/18/2000	4.26	12/9/2000	4.47	**1/30/2001**	**3.00**	3/23/2001	3.37
10/19/2000	4.3	12/10/2000	2.67	1/31/2001	3.95	3/24/2001	3.75
10/20/2000	4.9	12/11/2000	2.5	2/1/2001	5.13	3/25/2001	2.92
10/21/2000	4.77	**12/12/2000**	**3.80**	2/2/2001	5.38	3/26/2001	4.26
10/22/2000	2.27	12/13/2000	5.79	2/3/2001	4.7	**3/27/2001**	**0.70**
10/23/2000	4.96	12/14/2000	3.42	2/4/2001	4.42	3/28/2001	4.49
10/24/2000	**3.50**	12/15/2000	2.62	2/5/2001	4.1	3/29/2001	5
10/25/2000	4.14	12/16/2000	4.66	2/6/2001	1.68	3/30/2001	3.42
10/26/2000	3.71	12/17/2000	4.79	**2/7/2001**	**1.40**	3/31/2001	5.14
10/27/2000	3.56	12/18/2000	2.92	2/8/2001	4.02	4/1/2001	4.07
10/28/2000	3.87	**12/19/2000**	**1.70**	2/9/2001	4.98	4/2/2001	2.4
10/29/2000	4.4	12/20/2000	3.44	2/10/2001	5.31	4/3/2001	2.84
10/30/2000	3.4	12/21/2000	2.85	2/11/2001	4.63	4/4/2001	3.76
10/31/2000	**3.70**	12/22/2000	5.86	2/12/2001	5.11	**4/5/2001**	**1.80**
11/1/2000	4.12	12/23/2000	2.75	2/13/2001	4.11	4/6/2001	4.33
11/2/2000	4.27	12/24/2000	1.85	**2/14/2001**	**2.40**	4/7/2001	3.57
11/3/2000	3	12/25/2000	4.82	2/15/2001	3.6	4/8/2001	5.98
11/4/2000	4.46	**12/26/2000**	**2.70**	2/16/2001	3.14	4/9/2001	4.01
11/5/2000	4.91	12/27/2000	2.97	2/17/2001	3.11	**4/10/2001**	**2.80**
11/6/2000	4.19	12/28/2000	3.91	2/18/2001	3.57	4/11/2001	4.32
11/7/2000	3.74	12/29/2000	3.72	2/19/2001	2.48	4/12/2001	4.73
11/8/2000	**3.80**	12/30/2000	3.5	**2/20/2001**	**3.00**	4/13/2001	4.76
11/9/2000	5.03	12/31/2000	3.2	2/21/2001	2.01	4/14/2001	5.95
11/10/2000	3.87	1/1/2001	2.62	2/22/2001	3.7	4/15/2001	5.51

11/11/2000	3.72	**1/2/2001**	**4.10**	2/23/2001	3.14	4/16/2001	1.93
11/12/2000	4.53	1/3/2001	1.17	2/24/2001	4.23	**4/17/2001**	**0.70**
11/13/2000	3.68	1/4/2001	3.46	2/25/2001	3.12	4/18/2001	2.67
11/14/2000	**3.50**	1/5/2001	4.33	2/26/2001	3.8	4/19/2001	4.05
11/15/2000	2.08	1/6/2001	2.93	**2/27/2001**	**2.30**	4/20/2001	3.48
11/16/2000	1.97	1/7/2001	1.69	2/28/2001	3.65	4/21/2001	4.44
11/17/2000	5.18	1/8/2001	4.81	3/1/2001	3.54	4/22/2001	2.96
11/18/2000	4.16	**1/9/2001**	**3.80**	3/2/2001	2.35	4/23/2001	0.615
11/19/2000	3.39	1/10/2001	4.95	3/3/2001	4.8	**4/24/2001**	**4.10**
11/20/2000	3.06	1/11/2001	4.71	3/4/2001	3.49	4/25/2001	3.53
11/21/2000	**3.90**	1/12/2001	4.15	3/5/2001	5.23	4/26/2001	5.46
4/27/2001	3.43	6/18/2001	3.85	8/9/2001	1.99	9/30/2001	4.43
4/28/2001	4.97	**6/19/2001**	**3.80**	8/10/2001	3.55	**9/30/2001**	**4.47**
4/29/2001	2.85	6/20/2001	5.53	8/11/2001	3.72	10/1/2001	3.17
4/30/2001	4.26	6/21/2001	3.74	8/12/2001	5.12	10/1/2001	4.49
5/1/2001	**2.70**	6/22/2001	2.25	8/13/2001	3.66	**10/2/2001**	**4.50**
5/2/2001	2.41	6/23/2001	3.3	**8/14/2001**	**3.80**	10/3/2001	3.25
5/3/2001	3.46	6/24/2001	4	8/15/2001	3.61	10/4/2001	4.59
5/4/2001	2.95	6/25/2001	4.4	8/16/2001	4.92	10/5/2001	5.6
5/5/2001	3.7	**6/26/2001**	**3.40**	8/17/2001	4.24	10/6/2001	4.64
5/6/2001	4.74	6/27/2001	2.06	8/18/2001	4.65	10/7/2001	2.54
5/7/2001	2.9	6/28/2001	1.62	8/19/2001	3.73	10/8/2001	2.32
5/8/2001	**0.80**	6/29/2001	2.88	8/20/2001	4.03	**10/9/2001**	**5.30**
5/9/2001	3.44	6/30/2001	2.37	**8/21/2001**	**4.30**	10/10/2001	3.94
5/10/2001	2.78	7/1/2001	4.94	8/22/2001	4.89	10/11/2001	4.51
5/11/2001	3.24	7/2/2001	3.97	8/23/2001	3.36	10/12/2001	4.96
5/12/2001	3.83	7/3/2001	4.62	8/24/2001	3.82	10/13/2001	3.09
5/13/2001	4.49	**7/4/2001**	**3.60**	8/25/2001	2.89	10/14/2001	3.47
5/14/2001	5.83	7/5/2001	4.55	8/26/2001	3.31	10/15/2001	2.78
5/15/2001	**4.00**	7/6/2001	4.51	8/27/2001	3.77	**10/16/2001**	**3.80**
5/16/2001	1.58	7/7/2001	5.41	**8/28/2001**	**4.60**	10/17/2001	4.61
5/17/2001	3.51	7/8/2001	4.5	8/29/2001	3.63	10/18/2001	2.36
5/18/2001	3.52	7/9/2001	5.56	8/30/2001	2.57	10/19/2001	3.52
5/19/2001	2.43	**7/10/2001**	**3.90**	8/31/2001	5.7	10/20/2001	4.3
5/20/2001	3.32	7/11/2001	4.56	9/1/2001	3.25	10/21/2001	3.9
5/21/2001	3.28	7/12/2001	1.88	9/2/2001	4.84	10/22/2001	5.16
5/22/2001	4.36	7/13/2001	2.49	9/3/2001	4.75	**10/23/2001**	**6.00**
5/23/2001	**4.00**	7/14/2001	4.71	**9/4/2001**	**4.40**	10/24/2001	3.34
5/24/2001	5.43	7/15/2001	5.26	9/5/2001	3.69	10/25/2001	3.9
5/25/2001	3.67	7/16/2001	3.13	9/6/2001	5.28	10/26/2001	4.44
5/26/2001	3.99	**7/17/2001**	**4.40**	9/7/2001	4.48	10/27/2001	5
5/27/2001	3.03	7/18/2001	3.93	9/8/2001	3.93	10/28/2001	3.05
5/28/2001	3	7/19/2001	3.37	9/9/2001	3.8	10/29/2001	3.62
5/29/2001	**3.60**	7/20/2001	4.17	9/10/2001	4.35	**10/30/2001**	**4.40**
5/30/2001	4.63	7/21/2001	4.31	**9/11/2001**	**4.90**	10/31/2001	4.86
5/31/2001	3.83	7/22/2001	5.19	9/12/2001	2.28	11/1/2001	3.19
6/1/2001	1.04	7/23/2001	3.05	9/13/2001	3.63	11/2/2001	2.74

6/2/2001	3.85	**7/24/2001**	**4.40**	9/14/2001	2.69	11/3/2001	3.23
6/3/2001	2.04	7/25/2001	3.96	9/15/2001	5.59	11/4/2001	2.8
6/4/2001	4.45	7/26/2001	3.86	9/16/2001	4.25	11/5/2001	4.43
6/5/2001	**2.50**	7/27/2001	5.25	9/17/2001	4.38	**11/6/2001**	**4.80**
6/6/2001	2.87	7/28/2001	4.7	9/18/2001	4.19	11/7/2001	2.24
6/7/2001	4.15	7/29/2001	2.76	**9/19/2001**	**4.70**	11/8/2001	3.18
6/8/2001	2.53	**7/30/2001**	**4.70**	9/20/2001	4.56	11/9/2001	4.66
6/9/2001	4.6	7/31/2001	3.89	9/21/2001	2.19	11/10/2001	1.12
6/10/2001	3.21	8/1/2001	3.23	9/22/2001	5.68	11/11/2001	3.77
6/11/2001	2.22	8/2/2001	4.18	9/23/2001	4.85	11/12/2001	3.75
6/12/2001	**3.40**	8/3/2001	3.44	9/24/2001	4.41	**11/13/2001**	**4.40**
6/13/2001	3.91	8/4/2001	4.16	**9/25/2001**	**4.40**	11/14/2001	2.81
6/14/2001	4.08	8/5/2001	2.04	9/26/2001	3.92	11/15/2001	4.88
6/15/2001	3.15	8/6/2001	3.64	9/27/2001	2.51	11/16/2001	4.37
6/16/2001	4.18	**8/7/2001**	**2.40**	9/28/2001	2.83	11/17/2001	2.53
6/17/2001	3.24	8/8/2001	2.23	9/29/2001	3.21	11/18/2001	3.28
11/19/2001	1.35	1/10/2002	3.64	3/3/2002	6.01	4/24/2002	5.36
11/20/2001	**4.90**	1/11/2002	4.22	3/4/2002	5.48	4/25/2002	2.86
11/21/2001	3.2	1/12/2002	3.49	**3/5/2002**	**3.70**	4/26/2002	4.68
11/22/2001	3.98	1/13/2002	2.8	3/6/2002	2.75	4/27/2002	3.12
11/23/2001	3.61	1/14/2002	2.84	3/7/2002	3.99	4/28/2002	4.12
11/24/2001	5.32	**1/15/2002**	**4.00**	3/8/2002	2.66	4/29/2002	3.16
11/25/2001	2.66	1/16/2002	4.42	3/9/2002	2.46	4/30/2002	3.58
11/26/2001	3.91	1/17/2002	3.36	3/10/2002	3.4	**5/1/2002**	**2.70**
11/27/2001	**5.60**	1/18/2002	2.92	3/11/2002	4.69	5/2/2002	2.71
11/28/2001	3.34	1/19/2002	3.58	**3/12/2002**	**3.90**	5/3/2002	2.8
11/29/2001	3.04	1/20/2002	4.39	3/13/2002	2.18	5/4/2002	3.35
11/30/2001	4.06	1/21/2002	1.89	3/14/2002	4.58	5/5/2002	4.02
12/1/2001	4.48	**1/22/2002**	**4.00**	3/15/2002	4.1	5/6/2002	3.29
12/2/2001	1.41	1/23/2002	2.95	3/16/2002	2.61	**5/7/2002**	**3.60**
12/3/2001	4	1/24/2002	3.02	3/17/2002	2.1	5/8/2002	3.81
12/4/2001	**5.20**	1/25/2002	3.54	3/18/2002	3.78	5/9/2002	1.97
12/5/2001	3.3	1/26/2002	5.07	3/19/2002	3.86	5/10/2002	6.06
12/6/2001	3.58	1/27/2002	1.49	**3/20/2002**	**3.00**	5/11/2002	3.79
12/7/2001	3.67	1/28/2002	2.58	3/21/2002	4.23	5/12/2002	4.29
12/8/2001	4.18	**1/29/2002**	**4.30**	3/22/2002	1.54	5/13/2002	3.62
12/9/2001	1.75	1/30/2002	4.31	3/23/2002	4.28	**5/14/2002**	**3.40**
12/10/2001	4.09	1/31/2002	2.91	3/24/2002	3.89	5/15/2002	3.9
12/11/2001	**5.00**	2/1/2002	3.19	3/25/2002	5.3	5/16/2002	5.63
12/12/2001	5.73	2/2/2002	3.95	**3/26/2002**	**3.30**	5/17/2002	3.76
12/13/2001	2.79	2/3/2002	3.42	3/27/2002	4.79	5/18/2002	3.82
12/14/2001	4.77	2/4/2002	4.57	3/28/2002	5.75	5/19/2002	3.53
12/15/2001	4.72	**2/5/2002**	**3.40**	3/29/2002	4.59	5/20/2002	4.04
12/16/2001	3.28	2/6/2002	4.2	3/30/2002	2.95	5/21/2002	4.52
12/17/2001	1.79	2/7/2002	2.99	3/31/2002	3.36	**5/22/2002**	**3.00**
12/18/2001	**4.30**	2/8/2002	4.12	4/1/2002	5.21	5/23/2002	3.71
12/19/2001	5.17	2/9/2002	3.68	**4/2/2002**	**3.30**	5/24/2002	3.65

12/20/2001	5.06	2/10/2002	2.72	4/3/2002	3.39	5/25/2002	4.73
12/21/2001	3.73	2/11/2002	4.07	4/4/2002	2.45	5/26/2002	3.07
12/22/2001	4.77	**2/12/2002**	**3.70**	4/5/2002	3.85	5/27/2002	4.05
12/23/2001	4.47	2/13/2002	2.74	4/6/2002	3.69	5/28/2002	3.11
12/24/2001	3.02	2/14/2002	4.93	4/7/2002	4.37	**5/29/2002**	**4.30**
12/25/2001	**4.20**	2/15/2002	1.63	4/8/2002	2.99	5/30/2002	5.39
12/26/2001	4.81	2/16/2002	4.27	**4/9/2002**	**3.70**	5/31/2002	3.04
12/27/2001	3.84	2/17/2002	3.6	4/10/2002	4.45	6/1/2002	4.22
12/28/2001	3.17	2/18/2002	4.13	4/11/2002	2.56	6/2/2002	3.65
12/29/2001	2.16	**2/19/2002**	**3.50**	4/12/2002	2.51	6/3/2002	5.22
12/30/2001	5.45	2/20/2002	2.42	4/13/2002	3.67	6/4/2002	3.55
12/31/2001	2.3	2/21/2002	3.96	4/14/2002	3.33	**6/5/2002**	**4.90**
1/1/2002	**4.70**	2/22/2002	2.33	4/15/2002	2.31	6/6/2002	3.32
1/2/2002	3.38	2/23/2002	1.28	4/16/2002	3.95	6/7/2002	4.01
1/3/2002	3.18	2/24/2002	2.59	**4/17/2002**	**5.00**	6/8/2002	4.52
1/4/2002	3.81	2/25/2002	3.5	4/18/2002	5.24	6/9/2002	3.43
1/5/2002	4.21	**2/26/2002**	**4.70**	4/19/2002	3.02	6/10/2002	2.7
1/6/2002	1.77	2/27/2002	4.21	4/20/2002	4.29	**6/11/2002**	**4.60**
1/7/2002	1.92	2/28/2002	3.38	4/21/2002	2.43	6/12/2002	3.97
1/8/2002	**5.00**	3/1/2002	2.39	4/22/2002	3.41	6/13/2002	2.73
1/9/2002	3.59	3/2/2002	3.93	**4/23/2002**	**3.20**	6/14/2002	5.04
						6/15/2002	3.26
						6/16/2002	4.39
						6/17/2002	2.55
						6/18/2002	4.60
						6/19/2002	5.51
						6/20/2002	5.9
						6/21/2002	3.14
						6/22/2002	3.1
						6/23/2002	3.78
						6/24/2002	2.6
						6/25/2002	5.70
						6/26/2002	4.34
						6/27/2002	2.65
						6/28/2002	3.3
						6/29/2002	4.03
						6/30/2002	3.49
						7/1/2002	3.08
						7/2/2002	3.70
						7/3/2002	3.1
						7/4/2002	5.34
						7/5/2002	2.94
						7/6/2002	2.37
						7/7/2002	3.56
						7/8/2002	4.67
						7/9/2002	4.60
						7/10/2002	4.36
						7/11/2002	3.34

						7/12/2002	3.22
						7/13/2002	2.64
						7/14/2002	3.16
						7/15/2002	4.57
						7/16/2002	5.00
						7/17/2002	4.55
						7/18/2002	2.12
						7/19/2002	4.38
						7/20/2002	3.22

Table A-2: Stage 1 LHS sample sets for kinetic parameters and headwater boundary condition variables

Model run	Benthic algae ammonia preference	Benthic algae death rate @ 20°C	Benthic algae maximum growth rate @ 20°C	Benthic algae light constant for growth	Benthic algae respiration rate @ 20°C	Benthic algae nitrogen half-saturation constant for growth	Benthic algae phosphorus half-saturation constant for growth	Dissolved fraction of orthophosphate [4]	Dissolved fraction of orthophosphate [1]
1	0.2720	0.01801	52.9	339	0.02020	0.02586	0.00259	0.57	0.46
2	0.0320	0.00658	51.6	342	0.00555	0.02386	0.00275	0.81	0.41
3	0.0457	0.00294	45.1	337	0.00820	0.02895	0.00263	0.87	0.71
4	0.0223	0.00581	44.4	347	0.00384	0.03144	0.00262	0.65	0.44
5	0.0967	0.01500	41.7	322	0.00448	0.03223	0.00277	0.49	0.80
6	0.1290	0.00523	65.7	350	0.02341	0.02979	0.00378	0.51	0.41
7	0.1790	0.02364	58.4	340	0.00326	0.02899	0.00253	0.58	0.33
8	0.0742	0.00634	68.4	350	0.00519	0.02124	0.00354	0.53	0.57
9	0.0367	0.00933	55.6	340	0.00721	0.02406	0.00250	0.57	0.54
10	0.0683	0.00165	60.2	335	0.01306	0.02236	0.00341	0.75	0.50
11	0.1490	0.02122	50.7	349	0.00631	0.03292	0.00250	0.46	0.72
12	0.1950	0.01241	72.4	334	0.01263	0.01582	0.00741	0.41	0.69
13	0.2090	0.01324	67.9	345	0.00994	0.02058	0.00306	0.43	0.67
14	0.1590	0.01719	63	329	0.01220	0.02705	0.00265	0.63	0.56
15	0.0558	0.01168	55.5	302	0.00595	0.02172	0.00465	0.83	0.68
16	0.0624	0.00539	42.2	347	0.01081	0.03484	0.00282	0.47	0.81
17	0.1640	0.02031	78.9	350	0.01472	0.02146	0.00429	0.82	0.56
18	0.2250	0.04395	74.9	347	0.00691	0.02650	0.00496	0.45	0.47
19	0.2150	0.02922	73.5	343	0.01480	0.02292	0.00395	0.35	0.49
20	0.2510	0.01140	71.1	350	0.01136	0.02304	0.00254	0.66	0.60
21	0.0121	0.00110	51.2	349	0.00902	0.02754	0.00388	0.52	0.52

22	0.1170	0.02506	57.3	348	0.00758	0.02082	0.00285	0.77	0.63
23	0.1060	0.01051	46.2	347	0.01575	0.03028	0.00292	0.39	0.58
24	0.0904	0.00997	53.6	332	0.00618	0.02469	0.00293	0.74	0.70
25	0.2350	0.01395	61.4	311	0.03118	0.03096	0.00315	0.73	0.66
26	0.0834	0.01619	77.8	346	0.01889	0.01834	0.00473	0.70	0.87
27	0.0853	0.00421	63.9	350	0.00703	0.01975	0.00268	0.59	0.55
28	0.2460	0.00791	60.8	341	0.01349	0.02727	0.00252	0.54	0.61
29	0.1270	0.00311	53.8	317	0.00855	0.02674	0.00257	0.66	0.78
30	0.1220	0.01433	67.6	346	0.01632	0.02365	0.00416	0.37	0.74
31	0.1890	0.02889	81.4	350	0.01156	0.02519	0.00320	0.68	0.62
32	0.3860	0.01268	69.8	328	0.02519	0.02939	0.00252	0.62	0.76
33	0.0149	0.00230	38.9	315	0.01007	0.02628	0.00250	0.17	0.65
34	0.0645	0.00381	58.9	348	0.00934	0.02439	0.00261	0.48	0.52
35	0.1360	0.00347	54.5	338	0.01191	0.02199	0.00328	0.62	0.51
36	0.2780	0.02281	59.5	349	0.05506	0.03046	0.00251	0.96	0.60
37	0.1120	0.00881	70.3	350	0.00484	0.01897	0.00814	0.64	0.73
38	0.0773	0.00718	49.6	345	0.00810	0.02548	0.00331	0.72	0.67
39	0.1700	0.00676	65.6	349	0.02044	0.02862	0.00538	0.79	0.34
40	0.1480	0.00767	49.1	345	0.00497	0.02605	0.00281	0.69	0.26
41	0.0417	0.00843	48	343	0.00252	0.02792	0.00686	0.61	0.77
42	0.0515	0.00490	56.7	344	0.00787	0.01651	0.00272	0.52	0.86
43	0.1050	0.00939	75.7	349	0.01685	0.02542	0.00363	0.55	0.43
44	0.2020	0.01085	62	325	0.01415	0.01796	0.00560	0.33	0.48
45	0.1400	0.01821	66.7	350	0.01040	0.02825	0.00255	0.55	0.94
46	0.3000	0.00457	65	349	0.01748	0.02336	0.00310	0.59	0.63
47	0.1740	0.00234	63.2	331	0.01099	0.02003	0.00445	0.68	0.59
48	0.3300	0.01954	62.3	291	0.00652	0.02490	0.00351	0.61	0.54
49	0.0264	0.01595	57.5	323	0.00945	0.02270	0.00300	0.50	0.38
50	0.0962	0.03632	59.5	336	0.00870	0.01916	0.00604	0.71	0.64

Model run	Dissolved fraction of orthophosphate $_3$	Dissolved fraction of orthophosphate $_2$	Mineralization rate of dissolved organic phosphorus @ 20°C	[a]Fraction of bottom segment covered with benthic algae $_1$	Fraction of bottom segment covered with benthic algae $_2$	Passaic River headwater scaling factor	Nitrification rate @ 20°C	Phytoplankton carbon:chlorophyll ratio
1	0.47	0.95	0.207	-0.05	0.24	0.903	0.262	20.7
2	0.53	0.77	0.203	-0.15	0.13	1.43	0.259	26.3
3	0.80	0.66	0.207	-0.14	0.12	0.817	0.264	18
4	0.48	0.35	0.205	-0.17	0.15	1.01	0.292	16.3
5	0.51	0.58	0.201	0.22	0.12	0.796	0.241	27.1
6	0.37	0.48	0.201	-0.20	0.11	0.849	0.2	18.2
7	0.81	0.43	0.187	0.42	0.10	1.02	0.25	15.6
8	0.35	0.86	0.196	0.28	0.03	1.2	0.258	17.2
9	0.61	0.72	0.19	-0.22	0.14	0.723	0.229	15.4
10	0.72	0.64	0.186	-0.51	0.12	1.6	0.256	16.7
11	0.76	0.57	0.205	-0.02	0.23	1.41	0.253	18.9
12	0.46	0.78	0.191	-0.21	0.25	1.26	0.257	22
13	0.59	0.55	0.178	-0.35	0.18	1.29	0.27	20.1
14	0.26	0.71	0.2	0.10	0.10	1.09	0.245	19.8
15	0.54	0.80	0.198	0.26	0.21	0.59	0.233	12.2
16	0.40	0.62	0.199	0.14	0.17	1.34	0.223	17.5
17	0.60	0.51	0.194	0.67	0.17	1.47	0.237	22.8
18	0.66	0.83	0.198	0.18	0.29	0.751	0.243	21.7
19	0.76	0.81	0.219	0.11	0.16	1.22	0.267	21
20	0.56	0.56	0.211	-0.97	0.14	1.37	0.234	14.9
21	0.71	0.61	0.199	-0.30	0.20	0.994	0.232	20.6
22	0.61	0.47	0.192	-0.01	0.22	0.688	0.273	18.8

23	0.70	0.61	0.184	0.57	0.12	1.1	0.254	19.4
24	0.50	0.27	0.217	-0.39	0.21	1.24	0.265	25.6
25	0.55	0.53	0.2	-0.10	0.22	0.494	0.24	18.1
26	0.51	0.69	0.203	1.20	0.14	1.17	0.239	23
27	0.56	0.56	0.204	-0.53	0.08	0.355	0.209	23.5
28	0.44	0.74	0.193	-0.41	0.08	0.637	0.278	13.8
29	0.49	0.52	0.203	0.06	0.04	1.16	0.247	17.6
30	0.42	0.46	0.212	0.32	0.20	0.775	0.251	20.3
31	0.54	0.50	0.192	0.02	0.14	0.571	0.276	24
32	0.69	0.49	0.2	-0.31	0.05	1.63	0.248	19.3
33	0.64	0.59	0.197	-0.27	0.07	0.84	0.261	24.7
34	0.67	0.64	0.189	-0.11	0.11	1.12	0.287	22.5
35	0.66	0.68	0.209	0.19	0.06	0.729	0.303	24.2
36	0.58	0.66	0.195	0.01	0.17	0.924	0.269	22.3
37	0.69	0.60	0.211	0.38	0.16	0.877	0.298	16.2
38	0.92	0.73	0.208	0.34	0.09	0.977	0.226	21.1
39	0.62	0.55	0.206	0.21	0.19	1.04	0.272	19.1
40	0.29	0.53	0.195	0.08	0.16	1.19	0.197	24.9
41	0.64	0.65	0.186	0.13	0.26	1.07	0.28	20.5
42	0.43	0.41	0.209	-0.47	0.15	0.945	0.244	18.4
43	0.73	0.41	0.197	-0.05	0.15	0.442	0.282	21.2
44	0.65	0.33	0.206	0.47	0.19	1.13	0.251	14.7
45	0.77	0.70	0.202	-0.24	0.19	0.924	0.221	21.5
46	0.88	0.45	0.189	-0.07	0.09	0.655	0.22	23.3
47	0.75	0.76	0.194	-0.63	0.16	0.955	0.229	21.9
48	0.57	0.63	0.214	0.30	0.13	0.872	0.237	19.7
49	0.83	0.67	0.214	0.04	0.18	1.3	0.216	17
50	0.62	0.37	0.182	0.51	0.20	1.06	0.212	19.6

Model run	Phytoplankton death rate, non-zooplankton predation	Phytoplankton maximum growth rate @ 20°C	Phytoplankton optimal light saturation	Phytoplankton endogenous respiration rate @ 20°C	Phytoplankton nitrogen half-saturation constant for growth	Phytoplankton phosphorus half-saturation constant for growth	Ramapo River headwater scaling factor
1	0.10200	1.74	322	0.133	0.02578	0.00670	0.84
2	0.08527	1.63	324	0.18	0.02184	0.00458	1.36
3	0.15700	1.52	343	0.149	0.03162	0.00334	0.50
4	0.07797	0.853	312	0.131	0.02992	0.00216	1.27
5	0.04895	1.41	325	0.184	0.03055	0.00179	0.61
6	0.12200	1.09	316	0.118	0.02228	0.00144	1.06
7	0.09573	1.55	339	0.17	0.02042	0.00811	0.82
8	0.05127	0.544	293	0.152	0.02500	0.00270	0.65
9	0.06900	0.77	296	0.126	0.02365	0.00926	1.02
10	0.10900	0.945	330	0.122	0.02791	0.00284	1.41
11	0.11600	1.29	318	0.156	0.02835	0.00607	1.57
12	0.11500	1.6	347	0.157	0.02392	0.00316	0.96
13	0.13200	1.22	313	0.187	0.03010	0.00360	1.54
14	0.05536	0.683	326	0.141	0.03715	0.00118	0.67
15	0.10400	0.839	324	0.107	0.02639	0.00276	1.97
16	0.13900	1.67	328	0.165	0.01978	0.00351	1.26
17	0.12700	1.24	314	0.162	0.03205	0.00435	1.19
18	0.04458	0.937	305	0.17	0.02446	0.00298	1.34
19	0.06040	1.06	332	0.138	0.02687	0.00109	1.70
20	0.09799	1.08	316	0.146	0.02805	0.00372	0.44
21	0.17400	1.44	311	0.144	0.02143	0.00395	0.89
22	0.11300	1.35	307	0.143	0.01822	0.00343	0.99
23	0.07091	1.46	320	0.117	0.02241	0.00573	1.44
24	0.04008	1.33	314	0.174	0.01897	0.00508	1.08

25	0.08966	1.18	327	0.147	0.02411	0.00489	0.22
26	0.14700	1.57	337	0.141	0.01947	0.00725	0.89
27	0.09136	1.93	336	0.195	0.01334	0.00792	1.75
28	0.09867	1.16	334	0.129	0.02302	0.00406	0.72
29	0.07484	1.21	334	0.159	0.02457	0.00654	1.61
30	0.08320	0.978	299	0.159	0.02526	0.00500	1.23
31	0.06209	1.31	341	0.134	0.02053	0.00633	0.70
32	0.10700	1.12	318	0.123	0.01570	0.00451	1.16
33	0.14600	1.5	326	0.128	0.02286	0.00543	0.77
34	0.12900	1.42	310	0.179	0.02604	0.00694	0.32
35	0.11000	1.7	321	0.167	0.02529	0.00470	1.46
36	0.07540	1.49	302	0.14	0.01653	0.00839	1.02
37	0.14200	1.32	309	0.164	0.01772	0.00384	0.56
38	0.09322	1	303	0.155	0.02890	0.00170	0.80
39	0.12000	1.05	310	0.136	0.02676	0.00256	1.13
40	0.16700	1.89	317	0.168	0.02554	0.00562	0.93
41	0.10000	1.38	323	0.173	0.02176	0.00249	0.54
42	0.12300	1.14	319	0.136	0.03367	0.00424	1.10
43	0.08174	0.722	321	0.15	0.03418	0.00198	1.30
44	0.13500	1.25	320	0.112	0.02348	0.00597	0.74
45	0.06416	1.02	306	0.15	0.03100	0.00537	1.20
46	0.15100	1.27	305	0.161	0.02885	0.00242	1.14
47	0.08730	0.895	315	0.153	0.02746	0.00209	0.39
48	0.16000	1.38	329	0.176	0.02952	0.00308	0.97
49	0.02179	1.15	332	0.145	0.02114	0.00752	-0.01
50	0.03041	1.2	330	0.152	0.02733	0.00230	0.85

Model run	Settling rate of particulate phosphorus $_1$	Settling rate of particulate phosphorus $_2$	Settling rate of particulate phosphorus $_3$	[a]SOD
1	0.573	0.087	0.0129	-0.16
2	0.718	0.132	0.0082	-0.10
3	0.442	0.127	0.0096	-0.30
4	0.403	0.101	0.0106	-0.16
5	0.82	0.143	0.0065	0.41
6	0.461	0.148	0.0114	0.30
7	0.484	0.068	0.0106	-0.71
8	0.321	0.083	0.0123	-0.23
9	0.447	0.108	0.0079	0.04
10	0.505	0.091	0.0043	0.31
11	0.55	0.113	0.0143	0.68
12	0.521	0.075	0.0090	-0.05
13	0.298	0.063	0.0115	-0.21
14	0.415	0.091	0.0103	0.22
15	0.597	0.094	0.0098	-0.01
16	0.808	0.072	0.0066	-0.12
17	0.612	0.121	0.0076	0.28
18	0.705	0.119	0.0020	0.05
19	0.731	0.138	0.0150	-0.60
20	0.858	0.105	0.0091	0.02
21	0.694	0.061	0.0130	-0.28
22	0.8	0.013	0.0094	0.13
23	0.348	0.073	0.0094	0.50
24	0.603	0.116	0.0181	-0.39
25	0.564	0.077	0.0133	0.25
26	0.524	0.104	0.0141	0.10
27	0.56	0.116	0.0087	0.91
28	0.653	0.122	0.0120	-0.08

29	0.746	0.079	0.0108	-0.34
30	0.645	0.110	0.0116	-0.33
31	0.626	0.086	0.0085	-0.26
32	0.637	0.098	0.0072	0.11
33	0.472	0.094	0.0083	0.01
34	0.924	0.053	0.0073	-0.13
35	0.735	0.096	0.0127	-0.50
36	0.381	0.160	0.0056	-0.47
37	0.616	0.142	0.0100	0.17
38	0.965	0.102	0.0125	0.08
39	0.763	0.100	0.0159	0.35
40	0.54	0.089	0.0103	0.22
41	0.121	0.128	0.0068	-0.56
42	0.78	0.177	0.0111	-0.03
43	0.676	0.082	0.0061	-0.41
44	0.865	0.123	0.0051	-0.18
45	0.663	0.054	0.0111	-0.04
46	0.243	0.107	0.0101	0.14
47	0.682	0.066	0.0089	0.20
48	0.586	0.111	0.0118	0.47
49	0.497	0.135	0.0138	0.59
50	0.427	0.044	0.0080	0.39

[a] Indicates surrogate value; subsequently translated to local segment value according to Equation 3.1 using local mean and standard deviation values in Table 3-3.

Table A-3: Observed TP at Wanaque South intake (adapted from data published in Omni Environmental, 2007a)

Date and time	TP (mg/L)
6/3/2003 13:25	0.09
6/9/2003 12:24	0.06
6/17/2003 13:25	0.05
7/1/2003 14:10	0.13
7/8/2003 15:14	0.17
7/14/2003 13:30	0.25
7/22/2003 14:04	0.15
7/29/2003 13:54	0.16
8/5/2003 14:35	0.16
8/19/2003 14:10	0.15
8/25/2003 14:45	0.2
9/3/2003 15:11	0.09
9/8/2003 13:44	0.19
9/16/2003 14:10	0.13
9/24/2003 14:35	0.16
9/30/2003 11:35	0.11
10/7/2003 14:15	0.13
10/14/2003 13:54	0.15
11/4/2003 11:45	0.08
11/18/2003 12:45	0.12

Table A-4: Observed TP at B12-N38 (adapted from data published in Omni Environmental, 2007a)

Date and time	TP (mg/L)
11/14/2000 10:19	0.45
2/21/2001 12:30	0.19
5/2/2001 9:30	0.47
8/23/2001 13:19	0.59
11/26/2001 9:59	1.22
2/19/2002 12:59	0.81
5/29/2002 9:49	0.57
8/14/2002 9:40	1.02
11/7/2002 9:40	0.90
2/13/2003 9:59	0.71
5/6/2003 12:10	0.39
6/3/2003 13:15	0.21
6/9/2003 12:59	0.19
6/10/2003 16:40	0.17 [a]
6/17/2003 13:15	0.21
7/1/2003 14:29	0.47
7/8/2003 14:49	0.51
7/15/2003 11:55	0.58
7/22/2003 14:29	0.68
7/29/2003 14:15	0.42
8/5/2003 14:19	0.45
8/19/2003 14:25	0.48
8/25/2003 12:59	0.51
9/3/2003 15:30	0.47
9/8/2003 14:10	0.37
9/16/2003 14:29	0.53
9/24/2003 15:00	0.30
9/30/2003 11:45	0.24
10/7/2003 14:35	0.43
10/14/2003 14:25	0.59
11/4/2003 11:23	0.21
11/18/2003 13:22	0.41

[a] Measurement not used for comparison to 4-day moving model output

Table A-5: Observed chl-*a* at B17-N25 from June 15 to September 1 (adapted from data published in Omni Environmental, 2007a)

Date and time	chl-*a* (µg/L)
7/3/2001 12:04	22.53
8/21/2001 10:44	23.53
7/23/2002 11:49	8.90
8/14/2002 11:55	96.83
8/29/2002 12:34	11.41
7/2/2003 11:49	7.48
8/5/2003 11:29	12.07
8/13/2003 10:50	4.27
8/27/2003 10:40	24.57

Table A-6: Observed chl-*a* at B17-N4 from June 15 to September 1 (adapted from data published in Omni Environmental, 2007a)

Date and time	chl-*a* (µg/L)
6/20/2001 12:30	3.20
7/19/2001 11:45	39.50
8/2/2001 11:49	163.00
7/18/2002 11:45	2.14
8/13/2002 11:39	62.27
7/1/2003 11:19	3.56
7/14/2003 15:40	8.70
7/15/2003 11:49	10.32
7/15/2003 14:35	7.30
8/6/2003 11:29	13.90
8/21/2003 10:57	9.08
8/25/2003 15:00	6.70
8/27/2003 11:49	12.45

Table A-7: Observed DO from June 15 to September 1 at Dundee Lake (adapted from data published in Omni Environmental, 2007a)

Date and time	DO (mg/L)	Type of measurement
7/9/03 17:19	10.73	Diurnal meter
7/10/03 8:49	6.95	Diurnal meter
7/11/03 6:45	5.45	Diurnal meter
7/13/03 0:34	5.71	Diurnal meter
7/13/03 19:40	6.62	Diurnal meter
7/14/03 10:09	8.46	Diurnal meter
7/15/03 0:20	7.54	Diurnal meter
7/15/03 15:49	12.35	Diurnal meter
8/5/03 16:12	6.09	Grab sample
8/6/03 15:04	7.51	Grab sample
8/24/03 8:39	7.94	Grab sample
8/24/03 23:10	7.81	Diurnal meter
8/25/03 19:20	12.26	Diurnal meter
8/26/03 12:15	10.58	Diurnal meter
8/27/03 9:45	9.83	Diurnal meter
8/28/03 3:20	7.43	Diurnal meter
8/29/03 8:15	9.06	Grab sample

Table A-8: Observed DO from June 15 to September 1 at Peckman River mouth (adapted from data published in Omni Environmental, 2007a)

Date and time	DO (mg/L)	Type of measurement
7/10/03 2:39	9.07	Diurnal meter
7/11/03 17:35	9.62	Diurnal meter
7/12/03 6:49	7.70	Diurnal meter
7/13/03 7:40	7.48	Diurnal meter
7/15/03 4:44	9.29	Diurnal meter
7/15/03 21:49	7.99	Diurnal meter
7/16/03 12:45	9.22	Grab sample
8/5/03 15:30	7.32	Grab sample
8/6/03 13:27	7.84	Grab sample
8/24/03 18:55	9.13	Diurnal meter
8/25/03 13:10	12.07	Diurnal meter
8/26/03 4:10	5.77	Diurnal meter
8/27/2003 2:25	5.00	Diurnal meter
8/28/2003 19:50	4.50	Diurnal meter
8/29/03 8:55	8.35	Grab sample

Table A-9: Observed DO from June 4 to September 1 at Passaic River near Chatham (adapted from data published in Omni Environmental, 2007a)

Date and time	DO (mg/L)	Type of measurement
6/4/02 7:30	3.88	Diurnal meter
6/5/02 11:00	4.10	Diurnal meter
6/11/02 17:00	5.03	Diurnal meter
6/17/02 11:29	6.48	Diurnal meter
6/19/02 21:00	4.97	Diurnal meter
6/25/02 6:59	3.67	Diurnal meter
7/8/02 17:00	7.56	Diurnal meter
7/9/02 15:59	7.95	Diurnal meter
7/11/02 0:01	2.30	Diurnal meter
7/16/02 9:00	3.61	Diurnal meter
7/16/02 20:00	4.86	Diurnal meter
8/27/02 23:29	3.64	Diurnal meter
8/29/02 9:30	7.31	Diurnal meter

Table A-10: Observed DO from June 15 to September 1 at station PA10 (adapted from data published in Omni Environmental, 2007a)

Date and time	DO (mg/L)	Type of measurement
6/17/03 14:19	9.14	Grab sample
7/1/03 15:20	7.94	Grab sample
7/9/03 4:25	6.61	Diurnal meter
7/10/2003 2:15	7.19	Diurnal meter
7/10/2003 20:29	6.71	Diurnal meter
7/11/03 23:39	7.20	Diurnal meter
7/12/03 13:34	8.15	Diurnal meter
7/13/03 9:40	7.12	Diurnal meter
7/15/03 1:09	7.60	Diurnal meter
7/15/03 15:10	10.14	Grab sample
7/16/03 5:35	6.47	Diurnal meter
7/22/03 15:04	8.44	Grab sample
7/29/03 14:46	9.15	Grab sample
8/5/03 16:44	6.89	Grab sample
8/6/03 14:29	8.15	Grab sample
8/19/03 15:04	7.00	Grab sample
8/25/2003 4:50	7.01	Diurnal meter
8/26/03 10:05	7.16	Diurnal meter
8/26/03 22:10	8.46	Diurnal meter
8/29/03 7:35	6.33	Grab sample

Table A-11: WWTP effluent data used to develop Equation 4.1 (adapted from data published in Omni Environmental, 2007a)

WWTP	Phosphorus removal process	orthophosphate (mg/L)	TP (mg/L)	date
Berkeley Heights	chemical	0.40	0.46	6/10/03
		0.38	0.49	7/14/03
		0.51	0.59	8/6/03
		0.22	0.27	8/26/03
		0.32	0.36	11/4/03
		0.15	0.25	11/18/03
Bernards Twp.	Activated Sludge	0.28	0.28	6/10/03
		1.10	1.1	7/14/03
		1.14	1.31	8/6/03
		1.71	1.82	8/26/03
		0.56	0.6	11/4/03
		2.52	2.52	11/18/03
Caldwell	Activated Sludge	1.51	1.58	6/10/03
		3.00	3	7/14/03
		0.10	0.12	8/6/03
		3.41	3.57	8/26/03
		2.35	2.35	11/4/03
		2.70	2.7	11/18/03
Cedar Grove	chemical	1.56	1.73	6/10/03
		2.50	2.50	7/14/03
		1.18	1.41	8/6/03
		2.18	2.38	8/26/03
		2.16	2.22	11/4/03
		2.25	2.25	11/18/03
Chatham Glen	Activated Sludge	2.96	3.17	6/10/03
		3.16	3.34	7/14/03
		2.64	3.05	8/6/03
		3.38	3.73	8/26/03
		3.28	3.28	11/4/03
		3.24	3.52	11/18/03
Florham Park	Activated Sludge	0.72	0.77	6/10/03
		1.40	1.4	7/14/03
		1.94	2.13	8/6/03
		2.24	2.45	8/26/03
		1.64	1.65	11/4/03
		2.30	2.3	11/18/03
Hanover	chemical	2.33	2.41	6/10/03
		2.70	2.7	7/14/03
		2.23	2.51	8/6/03

		2.46	2.63	8/26/03
		2.37	2.37	11/4/03
		2.40	2.4	11/18/03
Livingston	Activated Sludge	1.23	1.28	6/10/03
		2.1	2.1	7/14/03
		2.12	2.5	8/6/03
		2.22	2.42	8/26/03
		2.16	2.18	11/4/03
		2.8	2.8	11/18/03
Long Hill	chemical	0.67	0.74	6/10/03
		3	3	7/14/03
		2.26	2.5	8/6/03
		2.52	2.69	8/26/03
		1.35	1.35	11/4/03
		2.21	2.21	11/18/03
Molitor	Activated Sludge	1.97	2.07	6/10/03
		3.38	3.65	8/6/03
		3.26	3.44	8/26/03
		2.42	2.42	11/4/03
		3.12	3.37	11/18/03
Morristown	EBPR	0.79	0.79	6/10/03
		0.5	0.54	7/14/03
		0.7	0.78	8/6/03
		0.31	0.39	8/26/03
		0.52	0.52	11/4/03
		0.2	0.22	11/18/03
Morristown-Butterworth	EBPR	0.08	0.15	6/10/03
		0.19	0.29	7/14/03
		2	2.26	8/6/03
		2.57	2.7	8/26/03
		0.76	0.83	11/4/03
		1.14	1.18	11/18/03
Parsippany Troy-Hills	Activated Sludge	2.80	2.8	6/10/03
		4.25	4.49	7/14/03
		3.97	4.29	8/6/03
		3.23	3.39	8/26/03
		3.77	3.77	11/4/03
		4.38	4.38	11/18/03
Pompton Lakes	chemical	0.005	0.15	6/10/03
		0.01	0.25	7/14/03
		0.005	0.11	8/6/03
		0.005	0.18	8/29/03
		0.05	0.16	11/4/03
		0.005	0.1	11/18/03

Rockaway Valley	EBPR	0.08	0.22	6/10/03
		2.08	2.25	7/14/03
		1.17	1.28	8/6/03
		2.21	2.49	8/26/03
		1.74	1.84	11/4/03
		2.08	2.13	11/18/03
Two Bridges	chemical	1.08	1.08	6/10/03
		2.40	2.4	7/14/03
		1.77	1.91	8/6/03
		1.71	1.73	8/26/03
		1.55	1.55	11/4/03
		1.84	1.9	11/18/03
Verona	chemical	2.17	2.2	6/10/03
		3.20	3.2	7/14/03
		2.44	2.63	8/6/03
		3.29	3.42	8/26/03
		2.41	2.41	11/4/03
		2.80	2.8	11/18/03
Wanaque	chemical	0.03	0.12	6/10/03
		0.19	0.34	7/14/03
		0.03	0.13	8/6/03
		0.05	0.15	8/26/03
		0.02	0.11	11/4/03
		0.02	0.1	11/18/03
Warren I	chemical	0.25	0.28	6/10/03
		2.35	2.5	7/14/03
		2.96	3.18	8/6/03
		2.23	2.35	8/26/03
		2.22	2.22	11/4/03
		1.4	1.4	11/18/03
Warren IV	chemical	0.91	1.07	6/10/03
		2.38	2.42	7/14/03
		1.74	2.06	8/6/03
		2.97	3.1	8/26/03
		1.32	1.32	11/4/03
		1.49	1.5	11/18/03
Warren V	chemical	2.2	2.45	6/10/03
		3.62	3.89	7/14/03
		3.37	3.53	8/6/03
		4.86	4.95	8/26/03
		2.75	2.75	11/4/03
		1.81	1.86	11/18/03
Wayne	Activated Sludge	1.45	1.45	6/10/03
		2.60	2.6	7/14/03
		2.08	2.13	8/6/03

		2.46	2.61	8/26/03
		1.89	1.95	11/4/03
		2.18	2.28	11/18/03

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ACKNOWLEDGEMENT OF PREVIOUS PUBLICATION

Kardos, J.S.; Obropta, C.C. (2008) Effect of Water Quality Model Uncertainty on the Passaic TMDL and Water Quality Trading Program for Total Phosphorus; *81st Annual Water Environment Federation Technical Exhibition and Conference (WEFTEC) Proceedings*.

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Publications

Obropta, C.C.; Niazi, M.; Kardos, J.S. (2008) Application of an Environmental Decision Support System to a Water Quality Trading Program Affected by Surface Water Diversions. *Journal of Environmental Management*, **42**, 946-956.

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