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WATER QUALITY MODEL UNCERTAINTY ANALYSIS OF A POINT-POINT SOURCE PHOSPHORUS TRADING PROGRAM[1]

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ABSTRACT: Water quality modeling is a major source of scientific uncertainty in the Total Maximum Daily Load (TMDL) process. The effects of these uncertainties extend to water quality trading programs designed to implement TMDLs. This study examines the effects of water quality model uncertainty on a nutrient trading program. The study builds on previous work to design a phosphorus trading program for the Nontidal Passaic River Basin in New Jersey that would implement the watershed TMDL for total phosphorus (TP). The study identified how water quality model uncertainty affects outcomes of potential trades of TP between wastewater treatment plants. The uncertainty analysis found no evidence to suggest that the outcome of trades between wastewater treatment plants, 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 TP load at potential hot spots in the watershed. Each simulated trading scenario demonstrated parity with or improvement from the command and control approach at the TMDL critical locations, and low risk of hot spots elsewhere.

(KEY TERMS: water quality trading; modeling; uncertainty analysis; TMDL; WASP; Latin Hypercube Sampling.)

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INTRODUCTION

The Need for Uncertainty Analysis in Water Quality Trading

Water quality trading (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 functions as an alternative to a command and control regulatory approach that prohibits trading and aims to specify effluent levels or particular abatement technologies for each pollutant source. 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 each source in the watershed and allows each source to discharge this amount, discharge less and sell the excess allowances, or discharge more and purchase additional allowances. With appropriate restrictions on trade, the net effect will be to achieve water quality goals, theoretically at a lower cost than through command and control

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regulation. Trading can in principle occur among point sources, among nonpoint sources, or between point and nonpoint sources (USEPA, 2004).

United States Environmental Protection Agency (USEPA) support for WQT is driven by the expectations that trading can reduce the cost of Total Maximum Daily Load (TMDL) implementation and engage unregulated nonpoint sources, such as farmers, in reducing pollutant discharges to water bodies (USEPA, 2003, 2004). 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 2005 review found only four ongoing trading programs that have "experienced a large number of trades" (Morgan and Wolverton, 2005). Information accessed recently from the USEPA WQT website reflected a modest increase (USEPA, State and Individual Trading Programs. Accessed April 19, 2011, http://www.epa.gov/owow/watershed/trading/tradingmap.html) in that actual trades had occurred in 17 states. However, the majority of trading programs continue to be small; only five to eight programs could be considered to have experienced a large number of trades as of April 2011.

A USEPA review (USEPA, 2005) suggested that inherent uncertainties in the TMDL process have prevented the onset of significant trading activity. Water quality modeling is a major source of scientific uncertainty in the TMDL process (NRC, 2001); the effects of these uncertainties extend to WQT programs designed to implement TMDLs, across the science, economic, and policy dimensions. In terms of environmental science, uncertainty about water quality models raises doubts that trades will protect water quality and avoid creating hot spots, that is, localized areas of degraded water quality (Nelson and Keeler, 2005). In terms of economics, a failure to characterize water quality model uncertainty impedes risk assessments and economic evaluations (Powers, 2006). In terms of policy, uncertainty can reduce the willingness to accept WQT by citizens and nongovernmental organizations.

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 have an estimate of uncertainty in assessed responses, rather than single point estimates, to better evaluate alternatives and guide future data collection efforts (Reckhow, 1994). The 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 WQT programs is also necessary to provide better decision support to policy makers and increased transparency to affected stakeholders. Specifically, this research has built on previous work by Obropta *et al.* (2008, 2010) which designed a phosphorus trading program for the Nontidal Passaic River Basin (NTPRB) in New Jersey. That trading program, which has been recommended to the New Jersey Department of Environmental Protection (NJDEP) but not yet implemented, functions as a case study in this article to analyze the effects of model uncertainty on the trading of total phosphorus (TP) between point sources. Ng and Eheart (2005) analyzed the uncertainty of a hypothetical biochemical oxygen demand (BOD) WQT program. However, in contrast with Ng and Eheart (2005), this study 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 (Kardos and Obropta, 2008). This is especially relevant given that the bulk of WQT programs in the United States (U.S.) center around nutrient trading. (Nutrients in the context of WQT refer to nitrogen or phosphorus, and not carbon or silicon.) Other trading programs have utilized water quality models (e.g., Long Island Sound, Connecticut; Minnesota River Basin, Minnesota; Neuse River Basin, North Carolina), but have not conducted uncertainty analyses of those models. Although many trading programs have introduced conservative trading ratios as a way to account for uncertainty (e.g., Dillon Reservoir, Colorado; Great Miami River Watershed, Ohio), those are typically arbitrary safety factors and have not been followed with any formal water quality model uncertainty analyses to verify their adequacy; this trend echoes the problem of arbitrary margins of safety that are often built into TMDLs (NRC, 2001; Shirmohammadi *et al.*, 2006).

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

The reader is referred to Obropta *et al.* (2008, 2010) for detailed explanation of the trading program design. A summary of the watershed, TMDL, and trading program is provided here as background information. This summary is admittedly long, but necessary to clarify the complexities of the study area and give proper context to the uncertainty analysis.

Study Area. Approximately 25% of New Jersey's population (i.e., 2 million people) lives in the nontidal portion of the Passaic River Basin. The New Jersey

and New York portions of the nontidal watershed cover 1,733 and 347 square kilometers, respectively (Figure 1). The presence of surface water diversions fundamentally alters the hydrology of the watershed. The Wanaque Reservoir system supplies drinking water to 4 million people and is the state's largest reservoir system (NJDWSC, 2002). Surface water is pumped to the Wanaque Reservoir from intakes located downstream at a rate according to consumer demand, water availability, and regulatory restriction. The Wanaque South intake, located on the Pompton River just upstream of its confluence with the Passaic River, is the primary diversion intake (Figure 2). Diversions to the Wanaque Reservoir transform basic relationships of upstream and downstream between certain locations in the watershed. For example, when the Wanaque Reservoir does not require diverted inflow, that is, the No Diversion condition, the Passaic River is not a natural tributary or source of water to the reservoir; the river continues unimpeded to Dundee Dam (the outlet of the nontidal basin), and then beyond to Newark Bay (the outlet of the tidal basin). But when the Wanaque Reservoir does require high volumes of diverted inflow as occurred in a 2002 drought, both the Pompton and the Upper Passaic River waters can be diverted to the reservoir, and the latter river effectively becomes "upstream" of the reservoir (Najarian Associates,

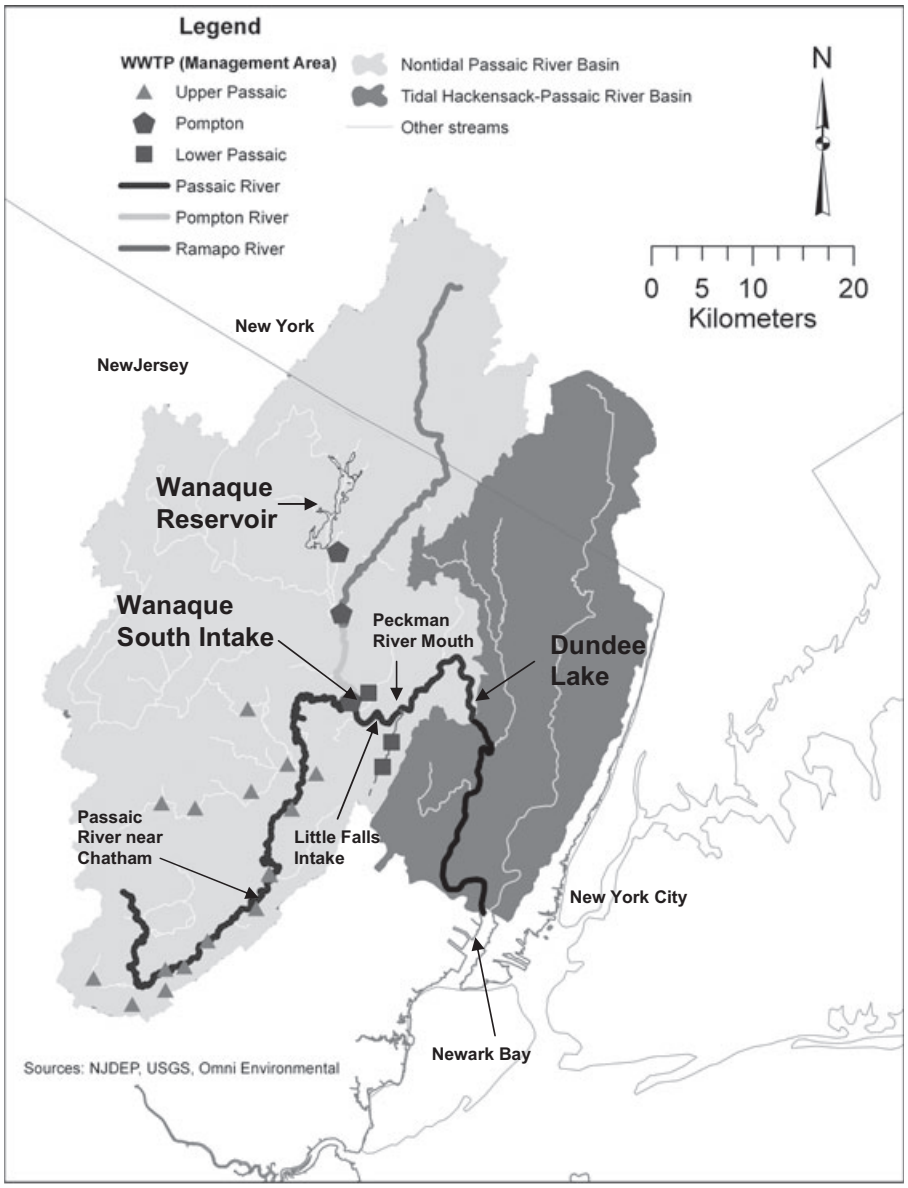


FIGURE 1. Nontidal Passaic River Basin (approx. 2,080 km^2) Lies to the West of the Tidal Basin. Wastewater treatment plants (WWTPs) are displayed according to management area, a concept further illustrated in Figure 3.

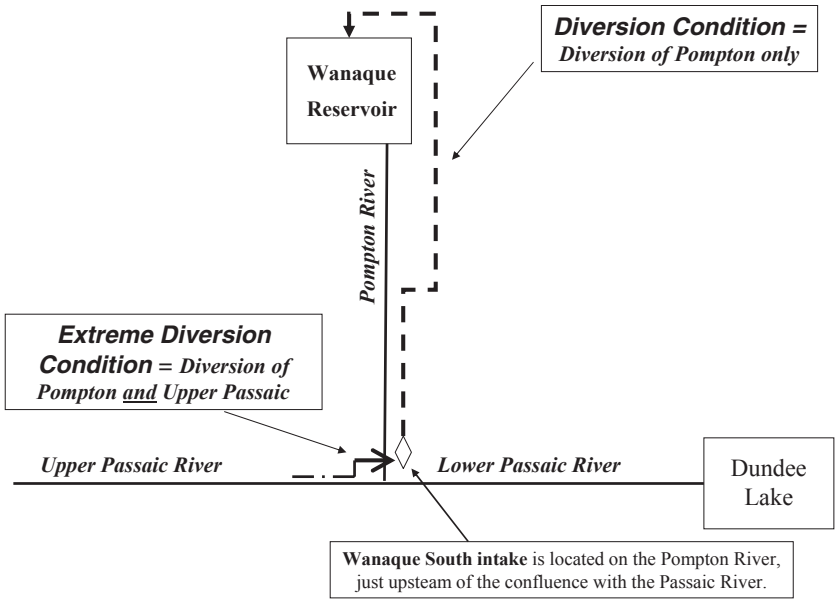


FIGURE 2. Schematic of Diversion Conditions. The Wanaque South intake diverts water from the Pompton River and/or Upper Passaic River to the Wanaque Reservoir. During the No Diversion condition, the Wanaque South intake is not operated.

2005); this scenario is termed the Extreme Diversion condition. Alternatively, an intermediate condition exists whereby the Pompton River but not the Upper Passaic River waters are diverted; this is termed the Diversion condition.

The activation of the Wanaque South intake is highly variable both within a single year and between years. Thus in terms of the three diversion conditions outlined above, a shift from one scenario to another can occur multiple times in a year. The watershed hydrology fluctuates with the extent of surface water diversions, resulting in dynamic relationships of upstream and downstream. The range of possibilities for TP loadings affecting the Wanaque South intake and Dundee Lake (i.e., the impoundment behind Dundee Dam) is shown in Table 1, and was accounted for in the trading program design. Note that Dundee Lake always receives TP loadings from the Upper Passaic and Pompton Rivers, whereas the particular diversion condition determines if and from where the Wanaque South intake receives TP loadings.

TABLE 1. Effect of Diversion Condition on Total Phosphorus (TP) Loading to Wanaque South Intake and Dundee Lake.

	Wanaque South Intake Is Affected by TP Load From:		Dundee Lake Is Affected by TP Load From:	
Diversion Condition	**Upper Passaic River**	**Pompton River**	**Upper Passaic River**	**Pompton River**
No Diversion	No	No	Yes	Yes
Diversion	No	Yes	Yes	Yes
Extreme Diversion	Yes	Yes	Yes	Yes

Total Maximum Daily Load. High loadings of TP, primarily from wastewater treatment plants (WWTPs), have periodically contributed to algal blooms at two critical locations – the Wanaque Reservoir and the watershed outlet identified as Dundee Lake. Besides producing taste and odor problems for drinking water, algal blooms also harm the aquatic environment through increased consumption of DO, due to the respiration of living algae and decomposition of dead algae. Intense fluctuations of diurnal DO are also observed during algal blooms. To address this impairment from TP, NJDEP adopted the *Total Maximum Daily Load for the Nontidal Passaic River Basin Addressing Phosphorus Impairments* which set phosphorus load allocations for point and nonpoint sources in the watershed (NJDEP, 2008a). (TP loads that enter the watershed from New York are covered by a separate TMDL (NJDEP, 2008b).) As TP loading in the NTPRB is dominated by WWTPs, the majority of reductions are allocated to those sources. The Passaic TMDL proposed site-specific watershed criteria in terms of a seasonal average (June 15-September 1) concentration of the response indicator, chlorophyll α (chl-a), a measure of the algal population. To meet the chl-a criteria, the TMDL requires each WWTP to discharge a maximum TP effluent concentration of 0.4 mg/l, calculated as a long-term (i.e., annual) average.

The TMDL primarily aims to alleviate phosphorus loads that the Wanaque Reservoir receives via the Wanaque South diversion intake, located in the midst of the basin, and the loads that Dundee Lake receives from all upstream sources. As phosphorus has been identified as the limiting nutrient to growth of algae, a reduction in phosphorus loading is expected to result in attainment of seasonal average chl-a criteria, DO daily average and instantaneous minimum criteria, and reduced diel DO fluctuations which are not subject to a criterion but are recognized as an indicator of algal activity. The TMDL identifies the Wanaque Reservoir and Dundee Lake as critical locations; these are also the watershed outlets. The Wanaque South intake is a surrogate critical location for the Wanaque Reservoir, since the intake is the source of most of the phosphorus loading that reaches the reservoir (Najarian Associates, 2005).

Water Quality Trading Program. A WQT program, focused on point sources, was developed through a USPEA Targeted Watershed Grant to increase the cost effectiveness of TMDL implementation (Obropta *et al.*, 2008, 2010). Twenty-two WWTPs, all located in New Jersey, are expected to be the primary eligible trading participants (Table 2). (NJDEP required that the trading program only include New Jersey dischargers.) At the time of writing, trading had not yet begun in the basin.

The trading program was designed primarily to address NJDEP concerns regarding water quality. In this article, "trading program design" refers to the rules and structure (e.g., allocations, eligible trades) that would govern trading in the NTPRB; it does not refer to a forecast of the least cost allocation of phosphorus permits. NJDEP expressed that the trading program design should (1) account for the surface water diversions in the watershed; (2) recognize that achieving the TMDL goals is driven primarily by discharger *concentration* of TP rather than *load* (NJDEP, 2008a), thus trades should be based on attaining the net effect of 0.4 mg/l TP effluent from each discharger; and (3) function so that water quality outcomes from trading, in terms of diverted phosphorus load at the Wanaque South intake and seasonal average chl-a at Dundee Lake, must be equal to or better than they would have been under a command and control approach that prohibited trading (NJDEP Division of Watershed Management, April 30, 2007, personal communication).

TABLE 2. Twenty-Two Main Wastewater Treatment Plants in the Nontidal Passaic River Basin.

Wastewater Treatment Plant	Management Area*	Average Flow (million gallons per day, or mgd)**	Permitted Flow (mgd)	Average Total Phosphorus Effluent Concentration (mg/l)***
Berkeley Heights Sewage Treatment Plant (STP)	Upper Passaic	1.58	3.1	0.48
Bernards Twp STP	Upper Passaic	1.71	2.5	2.80
Caldwell STP	Upper Passaic	3.92	4.5	1.62
Cedar Grove STP	Lower Passaic	1.49	2	1.58
Chatham Twp/Chatham Glen STP	Upper Passaic	0.12	0.15	3.54
Florham Park Sewerage Authority (SA)	Upper Passaic	0.91	1.4	1.24
Hanover SA	Upper Passaic	2.07	4.6	0.81
Livingston Twp STP	Upper Passaic	2.21	4.6	3.04
Long Hill Township STP	Upper Passaic	1.05	0.9	2.68
Molitor Water Pollution (Madison-Chatham) STP	Upper Passaic	2.49	3.5	3.70
Morris Township-Butterworth STP	Upper Passaic	2.01	3.3	1.37
Morristown STP	Upper Passaic	2.58	6.3	0.37
Parsippany-Troy Hills Regional Sewerage Authority (RSA)	Upper Passaic	12.57	16	3.55
Pompton Lakes STP	Pompton	0.90	1.2	0.37
Rockaway Valley SA	Upper Passaic	10.51	12	1.52
Two Bridges SA	Pompton	5.80	10	0.95
Verona STP	Lower Passaic	2.37	3	2.85
Wanaque Valley RSA	Pompton	1.06	1.25	0.12
Warren Stage I-II STP	Upper Passaic	0.38	0.47	2.02
Warren Stage V STP	Upper Passaic	0.17	0.38	3.13
Warren Township SA Stage IV STP	Upper Passaic	0.30	0.8	2.19
Wayne Twp STP	Lower Passaic	8.22	13.5	1.98

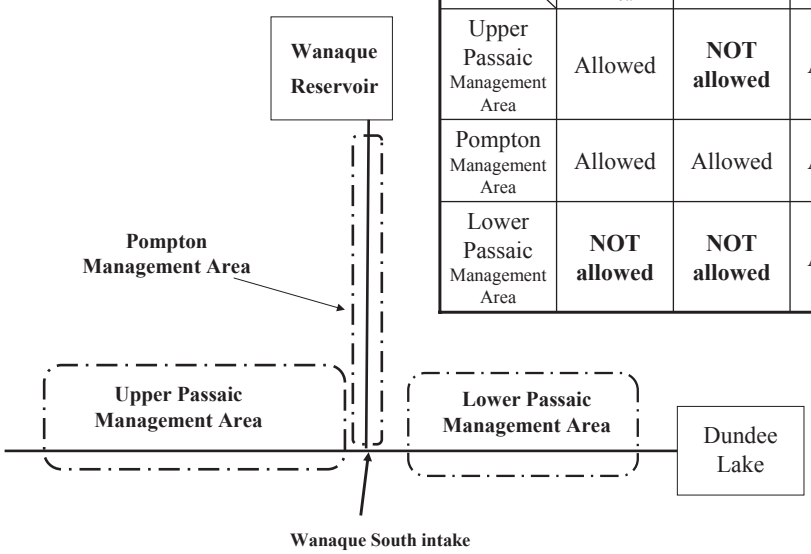
Note: *Data source*: NJDEP, NJPDES DMR data by NJPDES Permit Number. Accessed July 2008, http://datamine2.state.nj.us/DEP_OPRA/OpraMain/get_long_report?
*The watershed is divided into three management areas, as shown in Figure 3.
**Based on 2005-2007 data.
***Based on 2007 data.

In response, the trading program was designed with the following features. The watershed was divided into three management areas. Bidirectional trades (i.e., the seller can be upstream or downstream of the buyer) are allowed within management areas, but trades between management areas are restricted according to the framework shown in Figure 3, so as to protect water quality at both critical locations under all diversion conditions.

Trades are subject to a trading ratio. The trading ratio is based on the relative delivery of TP load from two point sources to a shared critical location (i.e., Wanaque South intake or Dundee Lake) downstream. A trading ratio is necessary to equalize the load exchanged relative to the shared critical location. A trading ratio of 0.9 means that 1,000 kg abated by the seller, has the same effect at the shared critical location as 900 kg abated by the buyer. Trading ratios vary depending on the buyer and seller in question. To account for multiple diversion conditions and both critical locations, each trading ratio is chosen based on the worst-case scenario, that is, the most vulnerable diversion condition and critical location, for a given buyer-seller pair.

To realize the concentration-driven goal of the TMDL, while acknowledging that transactions must occur in units of mass, trading allocations were mass loads equal to the product of a TP effluent concentration of 0.4 mg/l and a recent history of *actual* discharger volumetric flow rate rather than *permitted* volumetric flow rate. Although the permitted flow rate was used to calculate TMDL allocations, it was not used to determine trading allocations, otherwise sellers would receive allowances for more kilograms than they had actually removed from their effluent, thus posing a risk to the water body. Therefore unused capacity cannot be sold, resulting in trading allocations being less than TMDL allocations. Trades conducted in this manner are expected to achieve a net effect of 0.4 mg/l TP effluent from each discharger.

The trading program design was validated via a simulated "stress test." Thirteen trading scenarios, detailed in Obropta *et al.* (2010), were devised that imposed a series of critical conditions (e.g., heavy trading activity across tributaries and between management areas, buyers clustered in portions of the watershed). The buyer-seller pairs in each of the 13 scenarios were considered feasible, given forecasts of each WWTP's ability to upgrade TP removal. Each trading scenario was simulated at its most vulnerable of the three diversion conditions. (The simulation model is detailed in the Methodology section.) For purposes of comparison, a baseline (i.e., no-trade) scenario was simulated for each of the three diversion conditions. In each trading scenario, simulations of water quality outcomes at the critical locations were equal to or better than a no-trade approach (Table 3). Thus, the trading program design passed the stress test, and was deemed capable of protecting water quality under the most adverse conditions.



Seller \ Buyer	Upper Passaic Management Area	Pompton Management Area	Lower Passaic Management Area
Upper Passaic Management Area	Allowed	**NOT allowed**	Allowed
Pompton Management Area	Allowed	Allowed	Allowed
Lower Passaic Management Area	**NOT allowed**	**NOT allowed**	Allowed

FIGURE 3. Trading Framework. The basin is divided into three management areas. Trading between management areas is restricted according to the matrix at upper right, to protect the Wanaque Reservoir under all diversion conditions.

TABLE 3. Evaluation of Trading Scenario Performance at TMDL Critical Locations.

Trading Scenario	Diversion Condition	Ratio of Trading Scenario to No-Trade Scenario for: TP Load Diverted at Wanaque South Intake	Seasonal Average Chlorophyll *a* at Dundee Lake
1	Extreme Diversion	0.98	0.97
2	No Diversion	NA	1.00
3	Diversion	0.98	0.96
4	Diversion	0.98	0.95
5	Extreme Diversion	0.75	0.97
6	Extreme Diversion	NA*	0.95
7	Extreme Diversion	0.72	0.92
8	Extreme Diversion	0.77	0.98
9	Extreme Diversion	0.75	0.98
10	No Diversion	NA	0.94
11	Extreme Diversion	0.70	0.96
12	Extreme Diversion	0.81	0.97
13	Extreme Diversion	0.94	0.97

Note: Results <1.0 indicate the trading scenario was superior to the no-trade approach.
*Trading parties are all in Lower Passaic Management Area, with no effect on Wanaque South intake.

Less conservative – and more economically favorable – approaches toward trading ratios were developed; however, none of them passed the stress test (Obropta *et al.*, 2010). A team of economists did model the trading program with an approach based on minimizing total abatement costs under a regime of structured bilateral trades (Boisvert *et al.*, 2008); they estimated that even with the most conservative trading ratios, net savings of 16% were possible when compared to the costs of command and control regulation. These findings agree with Feng *et al.* (2009) who assessed the validity of trading ratios derived with a water quality model and the benefits of a least cost allocation approach.

Role of Water Quality Modeling in TMDL and Trading Scenarios

The TMDL allocations were based on the coordinated execution of four models: (a) an instream flow model that simulated tributary hydrodynamics; (b) a nonpoint source load model which routed stormwater runoff loads into the tributary network; (c) an instream water quality model of the tributary network, which simulated nutrient-algae-DO processes, and also output a time series of diverted phosphorus loads from the Wanaque South intake; and (d) a deterministic water quality model of the Wanaque Reservoir (NJDEP, 2008a). (These models are discussed further in the Methodology section.)

As explained in the next section, this study focused exclusively on the instream water quality model (model *c* in the above paragraph). Instream water quality modeling of the basin was completed by Omni Environmental Corporation (2007), and their water quality model is hereafter referred to as the "Passaic Water Quality Model," or PWQM. The PWQM was instrumental in identifying appropriate TMDL allocations, and was the primary tool used to predict water quality outcomes of various trading scenarios. Given the importance of the PWQM in informing the trading program design, analysis of the model uncertainty is vital to examining the probability that trades of TP will not degrade water quality. Uncertainty analysis is especially needed to verify that trades are not likely to create hot spots, a concern of both the USEPA (2003, 2004) and critics of WQT (e.g., Steinzor, 2003).

Study Objective

The primary objective of the study was to identify how model uncertainty affects model outputs and decision risks related to 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 WQT programs. Hypothesis testing was conducted to investigate if 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, the chl-a criterion, and reduction targets for diverted TP load, at potential hot spots in the watershed.

METHODOLOGY

Materials

The PWQM was calibrated and validated by Omni Environmental Corporation (2007), and provided to the authors for research purposes. It is an application of the Water Quality Analysis Simulation Program (WASP) version 7.0, released by the USEPA Office of Research and Development. 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 (Figure 4) (Di Toro *et al.*, 1983; Wool *et al.*, 2003; Ambrose *et al.*, 2006). To simulate diurnal

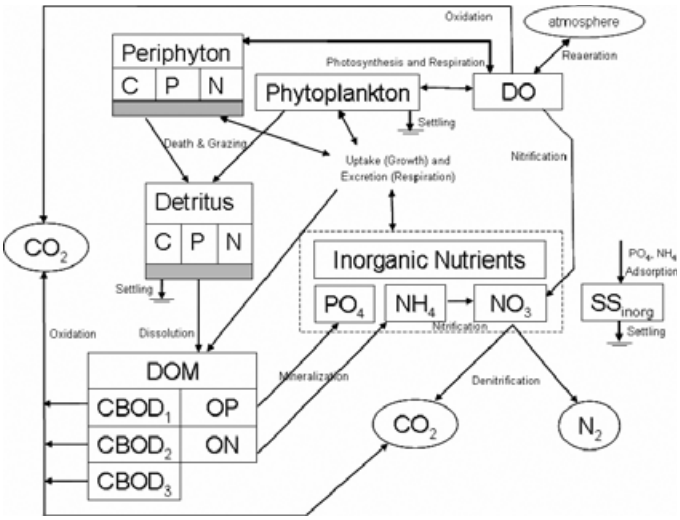


FIGURE 4. WASP7 Eutrophication Model Schematic (from Ambrose *et al.*, 2006). Phytoplankton and periphyton state variables are related to the phosphorus, nitrogen, and dissolved oxygen (DO) cycles. Light, temperature, and sediment oxygen demand are additional variables not depicted in this figure.

DO fluctuations characteristic of algal activity, the PWQM was executed at a 144-min time step, or 10 time steps per day.

The PWQM reads in hydrodynamic conditions from a calibrated and validated application of DAFLOW, a model developed by the U.S. Geological Survey. DAFLOW is a one-dimensional diffusive wave flow model that uses a Lagrangian solution method to calculate streamflows at each time step and location (Jobson, 1989). The PWQM imports nonpoint source pollutant loads from a proprietary watershed loading model based on stormwater event mean concentrations (Omni Environmental Corporation, 2007).

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

Scope of Uncertainty Analysis

This study focused on four sources of uncertainty in the PWQM: global kinetic parameter values, local kinetic parameter values, phosphorus boundary conditions at the Passaic River and Ramapo River headwaters, and phosphorus boundary conditions from selected WWTPs. These sources can be broadly categorized as parameter and input uncertainty. It was assumed that these are the primary sources of uncertainty in the model. This assumption was validated by a statistical comparison that tested how well uncertainty estimates matched observations. Specifically, for a given location and state variable, the actual proportion of success (i.e., proportion of observations within the predicted 10th and 90th percentiles) fell within the 95% confidence interval about the expected proportion of success, as detailed in Kardos (2009).

The following areas were outside the scope of the uncertainty analysis, with the exclusion of all but the Wanaque Reservoir model justified by the aforementioned statistical analysis in Kardos (2009): hydrodynamic model uncertainty (including WWTP effluent flows), nonpoint source load model uncertainty, model structure uncertainty, measurement uncertainty, and the uncertainty associated with the proprietary LAWATERS model of the Wanaque Reservoir (Najarian Associates, 2005). LAWATERS uses diverted phosphorus load output from the PWQM, and phosphorus loads from tributaries naturally upstream of the reservoir, to simulate state variables in the Wanaque Reservoir. Thus, the uncertainty analysis included diverted phosphorus loads at the Wanaque South intake, but not the effects of phosphorus in the reservoir itself. The analysis did include the effects of phosphorus in the Passaic River and its tributaries, as simulated in the PWQM.

A probabilistic assessment of the willingness of each discharger to participate in trading would have been valuable to include in the uncertainty analysis. However, it was the authors' experience that municipal WWTPs were understandably reluctant to disclose information that could have compromised their negotiating positions in trading discussions. Therefore, quantitative assessments of willingness to trade cannot yet be made. A qualitative approach, implicit in the construct of the 13 trading scenarios mentioned above, is more appropriate given the available information.

Application of Latin Hypercube Sampling

Method practicality is a key concern in uncertainty analysis of complex models (Chapra, 2003; Stow *et al.*, 2007) such as the PWQM. Conventional Monte Carlo simulation (MCS) involving many hundreds or thousands of simulations is not practical in this study because multiple trading scenarios need to be analyzed, and it is not feasible to repeat the entire process of conventional MCS for each scenario. A single simulation requires 40 min on a 2.2-GHz PC to execute. Therefore, this study applied the more efficient approach of Latin Hypercube Sampling (LHS) with restricted pairing (Iman and Conover, 1982) as the main method of uncertainty analysis.

Latin Hypercube Sampling is an efficient method to sample without replacement from stratified probability distributions, whereas MCS samples with replacement from nonstratified distributions (McKay *et al.*, 1979; Helton and Davis, 2000; Shirmohammadi *et al.*, 2006). LHS parameter combinations can be constrained by restricted pairing, in which the probability of a particular combination being selected is its true probability rather than the product of the marginal probabilities. Iman and Helton (1985) advised that successful LHS can be achieved with n greater than or equal to the product of 4/3 and the number of uncertain parameters.

The parameters included in the uncertainty analysis are listed in Table 4. Selection was based on Omni Environmental Corporation (2007), Lindenschmidt (2006), and a sensitivity analysis of the PWQM. Details of the assumed parameter distributions are listed in Tables A1 to A4.

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

$$n \geq \left(\frac{4}{3}\right) * x, \tag{1}$$

where n is the number of model runs; and x, number of uncertain parameters. With $x = 35$, the model was run 50 times.

Separate probability distributions were estimated for each of the parameters listed in Table 4. The PWQM was calibrated and validated by Omni Environmental Corporation (2007), yielding a fixed set of parameter values that reasonably matched observations of multiple state variables throughout the watershed. Consequently, the PWQM calibrated values were used to develop probability distributions for the selected kinetic parameters. For kinetic parameters with symmetric distributions, the PWQM calibrated values served as mean values. Parameters with skewed distributions utilized the PWQM calibrated values as either the mode or geometric mean, whichever resulted in a more successful comparison of uncertainty estimates to observations, as determined by the statistical approach described in Kardos (2009). For sediment oxygen demand, the distribution properties were based on available data from Omni Environmental Corporation (2007), and for other kinetic parameters the distribution properties were based on available literature (Bowie *et al.*, 1985; Brown and Barnwell, 1987; Melching and Yoon, 1996; Chapra, 1997; Wool *et al.*, 2003; Manache and Melching, 2004; Ambrose *et al.*, 2006; Lindenschmidt, 2006). In the absence of available literature for a given parameter, trials with different distributions were performed until model output uncertainty estimates successfully bounded observations, as determined by the statistical approach described in Kardos (2009).

Through interpretation of the fundamental WASP model equations (Wool *et al.*, 2003; Ambrose *et al.*, 2006), certain kinetic parameters, shown in Figure 5, were conservatively assumed to have covariance with a correlation coefficient of either (+) or (−) 0.5; in the absence of actual site-specific kinetic parameter data, more explicit correlation coefficients could not be defined. Only parameter-pairs that had a potential counterbalancing effect consistent throughout all WASP state variable equations were assigned a correlation coefficient. For example, to prevent overprediction of

TABLE 4. Water Quality Analysis Simulation Program Parameters Modeled as Uncertain.

Global kinetic parameters [*unit*] (*type of probability distribution*)			
Phytoplankton maximum growth rate at 20°C [/day] (Normal)	Phytoplankton carbon: chlorophyll ratio [] (Triangular)	Phytoplankton endogenous respiration rate at 20°C [/day] (Triangular)	Phytoplankton death rate, nonzooplankton predation [/day] (Triangular)
Phytoplankton optimal light saturation [Langleys/day] (Triangular)	Benthic algae maximum growth rate at 20°C [gD/m^2/day] (Triangular)	Benthic algae respiration rate at 20°C [/day] (Lognormal)	Benthic algae death rate at 20°C [/day] (Beta)
Local kinetic parameters [*unit*] (*type of probability distribution*)			
Sediment oxygen demand [g/m^2/day] (Normal)	Dissolved fraction of orthophosphate [] (Normal)	Settling rate of particulate phosphorus [/day] (Normal)*	Fraction of bottom segment covered with benthic algae [] (Normal)*
Boundary condition parameters [*unit*] (*type of probability distribution*)			
Passaic River headwater scaling factor for phosphorus [] (Normal)	Ramapo River headwater scaling factor for phosphorus [] (Normal)	TP effluent from 18 WWTPs [mg/l] (Lognormal)	

Notes: TP, total phosphorus; WWTP, wastewater treatment plant.
*Parameter was modeled separately at multiple locations, yielding a total of seven parameters for this category.

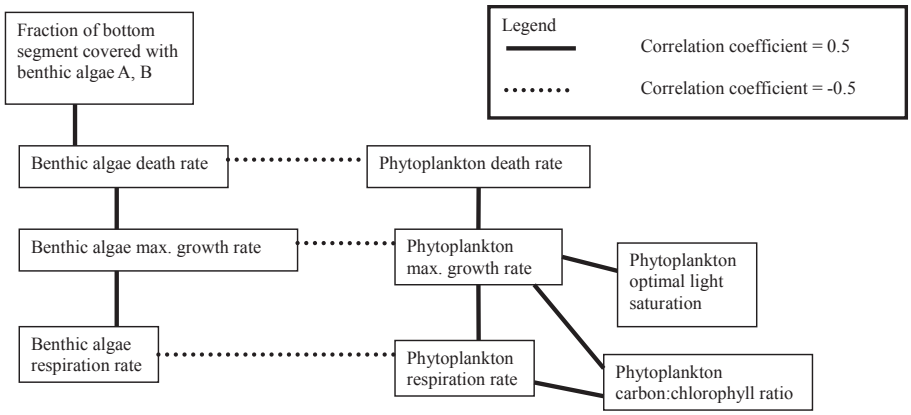


FIGURE 5. Illustration of Parameter Covariance Relationships Assumed in the Uncertainty Analysis. Parameters not shown (e.g., sediment oxygen demand, dissolved fraction of orthophosphate, settling rate of particulate phosphorus) were assumed to have no covariance with each other, nor with the parameters in this figure.

DO, the phytoplankton and benthic algae maximum growth rates were assigned a negative correlation. A negative correlation for this parameter-pair would have similar counterbalancing effects on prediction of phosphorus and nitrogen state variables. The general approach of assigning positive or negative correlations based on the counterbalancing effects on state variables is consistent with Di Toro and van Straten (1979). A sensitivity analysis was performed to compare the effect of unrestricted pairing to the restricted pairing used in the uncertainty analysis. It found that restricted pairing with the assumed correlations reduced the margin of uncertainty by up to 5% across DO, TP, and chl-a outputs.

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. Lumped parameters implicitly had complete covariance; this was justified in that lumped parameters were verified to only affect their immediate downstream target location, and thus contained no interference across target locations.

Probability distributions for Ramapo River and Passaic River headwater boundary conditions were based on available data from Omni Environmental Corporation (2007).

The procedure for modeling uncertainty of WWTP phosphorus boundary conditions is described below. The TMDL assumes that upon implementation, the TP effluent from each WWTP will follow a lognormal distribution with an arithmetic mean of 0.4 mg/l and coefficient of variation (COV) of 0.6 (NJDEP, 2008a). This study applied the NJDEP assumption. To increase efficiency of the uncertainty analysis, only the 18 WWTPs with ≥1.0 million gallons per day (mgd) (54.6 cubic meters per sec) of permitted or average flow were modeled probabilistically. For each yearlong model run, to simulate the uncertainty of daily TP effluent concentrations from each of the 18 WWTPs, 18 different lognormal distributions were randomly generated, each with 365 values, COV of 0.6, and an arithmetic mean equal to the TP effluent concentration specified in the model scenario.

In modeling TP effluent concentrations, the relative fractions of orthophosphate and organic phosphorus must be considered, because higher fractions of orthophosphate result in greater growth of algae. To translate the daily TP effluent concentration value arrived at in the previous step into a percentage of orthophosphate and organic phosphorus constituents, a regression analysis was done based on (six) paired samples of orthophosphate and TP concentrations collected in 2003 at each of the main 22 WWTPs in the watershed (Omni Environmental Corporation, 2007). A clear linear relationship exists between the orthophosphate and TP concentration data (Figure 6), and is approximated by Equation (2):

$$y = 0.977x - 0.07, \tag{2}$$

where y is orthophosphate (mg/l); and x, TP (mg/l). Equation (2) has an r^2 value of 0.99.

The phenomena of a strong linear relationship and near-unity coefficient value for the orthophosphate term (i.e., 0.977) in Equation (2) are due to the discharge of very low total suspended solids (TSS) effluent concentrations from all the WWTPs. During the period that WWTP effluent samples were collected, all the WWTPs discharged very low monthly average

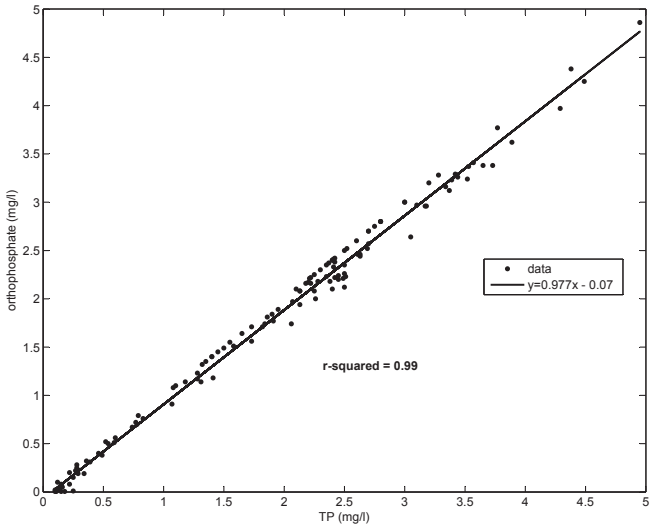


FIGURE 6. Wastewater Treatment Plant Effluent Data Sampled by Omni Environmental Corporation (2007), Fitted With Equation (2). A strong linear relationship exists between orthophosphate and total phosphorus (TP).

TSS concentrations, with most reporting below 5 mg/l TSS; the highest value was 14 mg/l (NJDEP, NJPDES DMR Data by NJPDES Permit Number. Accessed July, 2008, http://www.state.nj.us/dep//opra/online.html). Equation (2) should therefore not be generally applied, but only in this particular situation where all the WWTPs discharged very low effluent concentrations of TSS.

With respect to small WWTPs with <1 mgd (54.6 cubic meters per sec) effluent flow, TP effluent concentration was modeled deterministically without any daily variation, and Equation (2) was applied to derive orthophosphate and organic phosphorus boundary condition concentration values. Deterministic modeling of the small WWTPs is justified by their combined effluent flow comprising only 1% of the total effluent flow from all 22 dischargers.

Execution of Uncertainty Analysis and Output Processing

The LHS analysis used 35 random input parameters, as shown in Table 4. In accordance with Equation (1), 50 LHS sample sets were generated of global and local kinetic parameters, and headwater and WWTP boundary conditions for organic phosphorus and orthophosphate spanning Water Years (WY) 2001 and 2002, a period which captures all three diversion conditions. (A water year is defined as October 1-September 30.)

The two TMDL critical locations and three "areas of concern," as described by Omni Environmental Corporation (2007), were each considered a potential hot spot, or checkpoint, in the uncertainty analysis. Model output was processed at each checkpoint for the relevant water quality constituents (Table 5).

In the PWQM, the daily diverted load from the Wanaque South 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 WY2001 and WY2002 was provided in Omni Environmental Corporation (2007); 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 Wanaque South intake. After summing the daily load values, 50 values for annual diverted load were obtained with one value per model run and compiled as a cumulative output distribution.

Similarly, cumulative output distributions were assembled based on the 50 model results for seasonal average (June 15-September 1) of chl-a at Dundee Lake, and annual average TP concentration at the Little Falls intake, respectively.

Three different types of cumulative output distributions were calculated for DO, each spanning the

TABLE 5. Potential Hot Spots Examined.

Location	Key Water Quality Constituent(s)	Significance
Wanaque South intake	TP	Major surface water diversion intake to Wanaque Reservoir, the largest drinking water source in New Jersey; TMDL critical location (NJDEP, 2008a) due to potential effect of diverted water with high TP to stimulate algal blooms in reservoir
Dundee Lake	chl-a, DO	Watershed outlet; TMDL critical location (NJDEP, 2008a) due to high chl-a and high diurnal DO swing
Peckman River mouth	DO	Area of concern (Omni Environmental Corporation, 2007) due to low DO and high diurnal DO swing
Passaic River near Chatham	DO	Area of concern (Omni Environmental Corporation, 2007) due to low DO
Little Falls intake	TP	Area of concern (Omni Environmental Corporation, 2007) due to effect of TP on cost of drinking water treatment

Notes: TP, total phosphorus; chl-a, chlorophyll a; DO, dissolved oxygen; TMDL, Total Maximum Daily Load.

period of June 15-September 1: daily average DO violation frequencies, minimum DO violation frequencies, and diurnal DO swing. These outputs were generated at Dundee Lake, the Peckman River mouth, and the Passaic River near Chatham. Note that the June 15-September 1 period was chosen because it corresponds to the definition period of the seasonal average chl-a criterion and brackets the time of peak productivity.

The output distribution of daily average DO violation frequencies was calculated as follows. Dundee Lake, the Peckman River mouth, and the Passaic River near Chatham are each subject to a daily average DO criterion of 5 mg/l (NJDEP, 2004). For each of the 50 model runs, the violation 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 <5 mg/l by the total number of days. The cumulative output distribution of the 50 calculated violation frequencies was then compiled. Based on Borsuk *et al.* (2002), the mean of the distribution is termed the Expected Violation.

The output distribution of minimum DO violation frequencies was calculated as follows. The applicable 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 violation frequency of the minimum DO standard was calculated by dividing the number of days with DO <4 mg/l by the total number of days. The remaining steps in the preceding paragraph were repeated.

There is no criterion in New Jersey for diurnal DO swing; however, it is a useful indicator of algal activity. The cumulative output distribution of diurnal DO swing was calculated by including the diurnal DO swing for each day in each model run over the period of June 15-September 1.

Method of Comparing Trading and No-Trade Scenario Outputs

The right-tailed two-sample Kolmogorov-Smirnov (K-S) test (Gibbons and Chakraborti, 2003) was applied to compare output distributions of trading and no-trade scenarios. The latter are also termed "baseline" scenarios in this study since they represent the command and control approach. 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 the first population cumulative distribution is greater than the second population cumulative distribution, under the null hypothesis that both samples come from the same distribution.

For any scenario, trading or baseline, an output cumulative distribution is a profile of its uncertainty with respect to that output. For a given output (e.g., seasonal average chl-a at Dundee Lake), if the trading scenario cumulative distribution is significantly greater than the baseline scenario cumulative distribution, then the trading scenario has introduced a significant increase in uncertainty.

The usage of the right-tailed two-sample K-S test enabled the research question to be answered – is the output distribution of a trading scenario significantly greater than the output distribution of its baseline scenario, thus implying an increased uncertainty in achieving water quality targets?

Trading Scenario Selection

Recall the trading scenario results in Table 3. Each trading scenario was simulated at its most vulnerable of the three diversion conditions. Corresponding baseline (i.e., no-trade) scenarios were simulated for each of the three diversion conditions. All 13 of the trading scenarios equaled or outperformed corresponding baseline scenarios; however, seasonal averages of chl-a at Dundee Lake in Scenarios 2, 3, 8, and 9 came closest to exceeding baseline values when compared to other scenarios from the same diversion condition (Table 3), therefore those four scenarios were chosen for this analysis. As a whole, the four scenarios represent all three diversion conditions and all the stress test factors described previously. In addition, Scenario 1 represents a simple and presumably safe type of trade, in which the buyer is located downstream of the seller on the same tributary. The five scenarios are listed in detail (Table 6) and with a more general description (Table 7). The enumeration of the scenarios is maintained to be consistent with Obropta *et al.* (2010). Extreme Diversion conditions were overlaid with WY2002 natural streamflow conditions, and the Diversion and No Diversion conditions were each overlaid with WY2001 natural streamflow conditions.

RESULTS

The effect of model uncertainty on trading was evaluated under each of the three diversion conditions – Diversion, No Diversion, and Extreme Diversion. To investigate the Diversion condition, Baseline 5 output was compared to Scenario 3. No significant increases in uncertainty due to trading were found in diverted TP load at the Wanaque

TABLE 6. Trading Scenarios.

Scenario	Seller	Buyer	Seller Flow (mgd)	Buyer Flow (mgd)	Trading Ratio	Seller TP Effluent, Annual Average (mg/l)	Buyer TP Effluent, Annual Average (mg/l)	TP Load Sold (kg)
1	Berkeley Hts	Florham Park	1.61	0.86	0.78	0.100	0.860	665
2	Caldwell	Bernards (Harrison Brook)	3.99	1.78	0.92	0.100	0.985	1,545
		Warren V		0.15			1.000	135
3	Parsippany-Troy Hills	Rockaway	12.48	10.06	1.03	0.100	0.930	5,223
	Caldwell		3.99		1.09			1,669
8	Parsippany-Troy Hills	Bernards (Harrison Brook)	12.48	1.78	0.82	0.100	2.910	5,223
					0.65			2,851
	Rockaway	Warren V	10.06	0.15	0.65	0.100	1.850	452
		Warren I-II		0.37	0.63		1.620	977
	Two Bridges	Morris-Butterworth	5.57	2.02	0.63	0.100	0.968	2,436
	Berkeley Heights	Warren IV	1.61	0.28	1.00	0.100	2.200	660
	Morristown	Florham Park	2.15	0.86	0.77	0.100	1.360	1,201
	Wanaque		0.96		0.43	0.100		440
	Pompton Lakes	Chatham Glen	0.89	0.12	0.46	0.100	1.530	406
9	Parsippany-Troy Hills	Cedar Grove	12.48	1.23	0.39	0.100	1.045	998
		Bernards (Harrison Brook)		1.78	0.82	0.100	1.000	1,458
		Florham Park		0.86	0.77	0.100	1.000	682
	Rockaway	Hanover	10.06	2.10	0.69	0.100	1.000	1,712
		Long Hill		1.03	0.63		1.000	850
					0.67			308
	Two Bridges	Molitor (Madison-Chatham)	5.57	2.48	0.60	0.100	0.927	1,451
	Pompton Lakes		0.89		0.48	0.100		29
	Morristown	Morris-Butterworth	2.15	2.02	1.00	0.100	0.927	1,201
	Wanaque		0.96		0.56	0.100		220
	Berkeley Heights	Warren IV	1.61	0.28	1.00	0.100	1.000	198
		Warren I-II		0.37	1.00		1.080	341
		Chatham Glen		0.12	1.00		1.000	99

TABLE 7. General Description of Baseline (i.e., No-Trade) and Trading Scenarios.

Scenario	General Description	Diversion Condition
Baseline 4	All WWTPs discharging at 0.4 mg/l TP effluent, annual average	No Diversion
Baseline 5	All WWTPs discharging at 0.4 mg/l TP effluent, annual average	Diversion
Baseline 6	All WWTPs discharging at 0.4 mg/l TP effluent, annual average	Extreme Diversion
1	Simple trade with seller upstream; both located in same MA; Seller is upstream of Passaic River near Chatham	Extreme Diversion
2	Seller is downstream of buyer with both located in same MA. Buyer is upstream of Passaic River near Chatham	No Diversion
3	Cross tributary trade with trading ratio >1. Sellers and buyer located in same MA	Diversion
8	Complex trades with buyers and sellers in different MAs; buyers clustered upstream; several buyers located upstream of Passaic River near Chatham; sellers located upstream of Wanaque South intake	Extreme Diversion
9	Complex trades with buyers and sellers in different MAs; buyers clustered in discrete areas; buyers located upstream of Passaic River near Chatham and Peckman River mouth; sellers located upstream of Wanaque South intake	Extreme Diversion

Notes: WWTP, wastewater treatment plant; TP, total phosphorus; MA, management area.

South intake, seasonal average chl-a at Dundee Lake, daily average DO violation frequency, minimum DO violation frequency, and diurnal DO swing at Dundee Lake, the Peckman River mouth and the Passaic River near Chatham, and annual average TP concentration at the Little Falls intake (Table 8).

Similar results were found for the No Diversion condition, by way of comparing Baseline 4 with Scenario 2 (Table 9), with one exception. As shown in Figure 7, the diurnal DO swing distribution at the Passaic River near Chatham realized a significant increase in uncertainty due to trading. The cause was

TABLE 8. Results of Right-Tailed Two-Sample Kolmogorov-Smirnov Tests for Baseline and Trade Scenarios at Diversion Condition.

Location	Output Distribution	Scenario 3 *p*-value*
Dundee Lake	DO daily average violation frequency, June 15-September 1, 2001	1.0
Peckman River mouth		NA**
Passaic River near Chatham		NA**
Dundee Lake	Minimum DO violation frequency, June 15-September 1, 2001	1.0
Peckman River mouth		NA**
Passaic River near Chatham		NA**
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-September 1, 2001	0.98
Peckman River mouth		NA**
Passaic River near Chatham		NA**

Notes: DO, dissolved oxygen; chl-a, chlorophyll *a*; TP, total phosphorus.
*At the 0.05 level of significance, $p < 0.05$ indicates trading scenario cumulative distribution is significantly greater than baseline scenario.
**Impacts to the Peckman River mouth and the Passaic River near Chatham were not applicable, because the Diversion condition is never critical for trades that could adversely affect either location.

TABLE 9. Results of Right-Tailed Two-Sample Kolmogorov-Smirnov Tests for Baseline and Trade Scenarios at No Diversion Condition.

Location	Output Distribution	Scenario 2 *p*-value*
Dundee Lake	DO daily average violation frequency, June 15-September 1, 2001	1.0
Peckman River mouth		NA**
Passaic River near Chatham		1.0
Dundee Lake	Minimum DO violation frequency, June 15-September 1, 2001	1.0
Peckman River mouth		NA**
Passaic River near Chatham		0.92
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-September 1, 2001	0.99
Peckman River mouth		NA**
Passaic River near Chatham		**0.00**

Notes: DO, dissolved oxygen; chl-a, chlorophyll *a*; TP, total phosphorus.
*At the 0.05 level of significance, $p < 0.05$ indicates trading scenario cumulative distribution is significantly greater than baseline scenario as indicated in boldface.
**Impacts to the Peckman River mouth were not applicable, because the No Diversion condition is never critical for trades that could adversely affect that location; the Wanaque South intake is not affected during the No Diversion condition.

the location of a buyer directly upstream. The magnitude of increase was however very small. The percent differences of diurnal DO swing from the baseline in mean value, standard deviation, and 90th percentile value were 4%, 6%, and 6%, respectively. Importantly, no significant differences were found between the baseline and trading scenario distributions of daily average DO and minimum DO violation frequencies, at the Passaic River near Chatham ($p = 1.0$ and 0.92, respectively).

The Extreme Diversion condition was analyzed by comparing Baseline 6 with Scenarios 1, 8, and 9. No significant increases in uncertainty were found in diverted TP load at the Wanaque South intake, seasonal average chl-a at Dundee Lake (Figure 8), daily average DO violation frequency and minimum DO violation frequency at Dundee Lake, the Peckman River mouth and the Passaic River near Chatham, and annual average TP concentration at the Little Falls intake (Table 10). In fact, under Scenarios 8 and 9, significant improvements ($p = 0.00$) were found in diverted TP load (Figure 9), because the largest WWTP upstream of the Wanaque South intake acted as a seller. Scenario 9 also yielded a significant improvement ($p = 0.02$) in annual average TP concentration at the Little Falls intake (Figure 10).

At Dundee Lake, no significant increase in uncertainty was found for the diurnal DO swing under the Extreme Diversion condition. However, there were three other cases where the distribution of diurnal

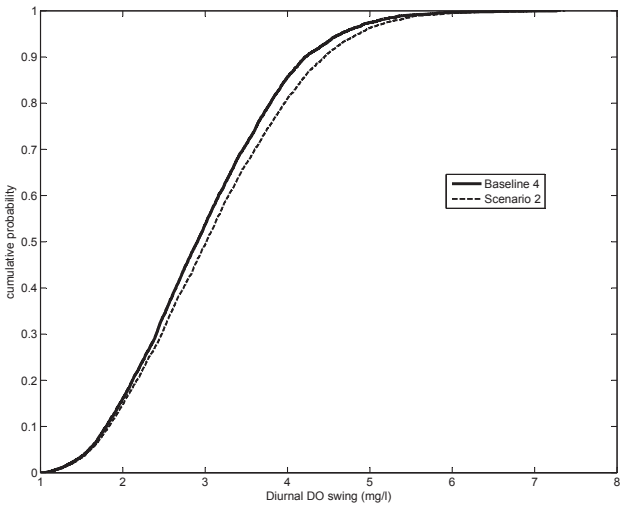


FIGURE 7. Cumulative Distributions of Diurnal Dissolved Oxygen (DO) Swing From June 15 to September 1, 2001 at Passaic River Near Chatham in Baseline and Trading Scenarios at No Diversion Condition. The trading scenario distribution is significantly greater than the baseline scenario distribution, although the magnitude of the increase is very small.

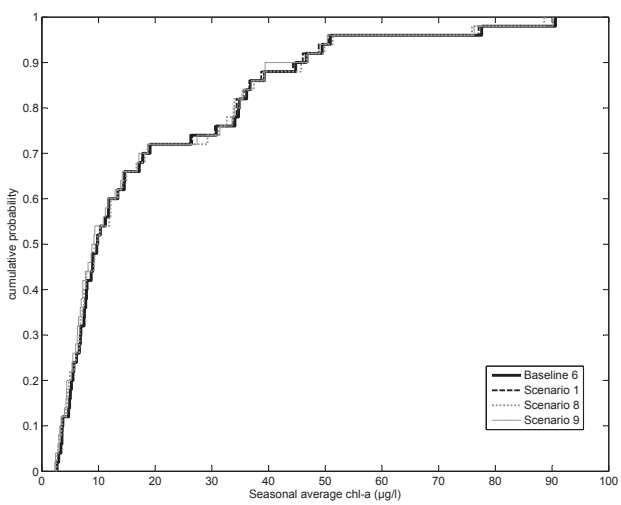


FIGURE 8. Cumulative Distributions of WY2002 Seasonal Average Chlorophyll *a* Concentration at Dundee Lake in Baseline and Trading Scenarios at Extreme Diversion Condition. For each trading scenario distribution, there is no significant increase from the baseline scenario distribution.

DO swing showed a significant increase in uncertainty from the baseline: at the Passaic River near Chatham in Scenarios 8 and 9 (Figure 11), and at the Peckman River mouth in Scenario 9 (Figure 12). In each of these cases, a buyer was located directly upstream of the area of concern, similar to Scenario 2. The increased concentration of TP in the discharge from the buyer(s) above baseline levels contributed to the rise in diurnal DO swing at the area of concern. The magnitude of increase 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%. Importantly, no significant differences were found between the baseline and trading scenario distributions of daily average DO and minimum DO violation frequencies, at both the Passaic River near Chatham and the Peckman River mouth (p ranged from 0.35 to 1.0; see Table 9).

DISCUSSION

The results illustrate the importance of reporting the relative uncertainties of trading and no-trading approaches. This issue was raised by Arabi *et al.* (2007) in their study of best management practice effectiveness uncertainty in an agricultural watershed. In our analysis, each trading scenario had either a DO criterion expected violation above 10%, or a chl-a criterion exceedance probability above 10%; these results, taken alone without attention to corresponding no-trade scenarios, might imply that trading cannot reliably achieve water quality targets. However, it is the relative uncertainties between trading and no-trading approaches that must be compared to assess the impact of trading. Then it becomes apparent that none of the trading scenarios introduced any significant increase in uncertainty in achieving water quality criteria, when compared to a no-trading approach, as shown by the two-sample K-S tests. In addition, a few cases saw significant improvements in achieving a water quality goal, for example, 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 determine if a management strategy increased 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 of the diurnal DO swing outcomes did show a significant increase from the no-trading baseline. The diurnal DO swing distribution at the Passaic River near Chatham increased significantly above the baseline in Scenarios 2, 8, and 9; a significant increase was also found at the Peckman River mouth in Scenario 9. However, the magnitudes of the increases were small, ranging from 4% to 11% in terms of mean

TABLE 10. Results of Right-Tailed Two-Sample Kolmogorov-Smirnov Tests for Baseline and Trade Scenarios at Extreme Diversion Condition.

Location	Output Distribution	*p*-Value* Scenario 1	Scenario 8	Scenario 9
Dundee Lake	DO daily average violation frequency, June 15-September 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
Dundee Lake	Minimum DO violation frequency, June 15-September 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
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-September 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**

Notes: DO, dissolved oxygen; chl-a, chlorophyll *a*; TP, total phosphorus.
*At the 0.05 level of significance, $p < 0.05$ indicates trading scenario cumulative distribution is significantly greater than baseline scenario as indicated in boldface.
**Neither Scenarios 1 or 8 contained buyers upstream of the Peckman River mouth, therefore only Scenario 9 was analyzed for trading scenario impacts at that location.

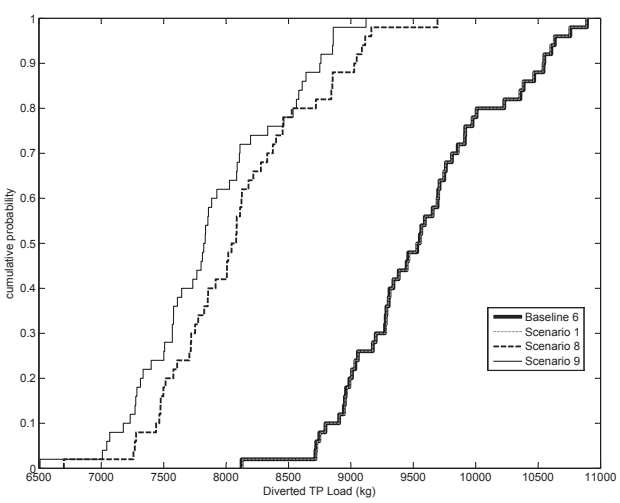


FIGURE 9. Cumulative Distributions of WY2002 Diverted Total Phosphorus (TP) Load From Wanaque South Intake to Wanaque Reservoir in Baseline and Trading Scenarios at Extreme Diversion Condition. The distributions of Scenarios 8 and 9 are significantly less than the baseline scenario distribution.

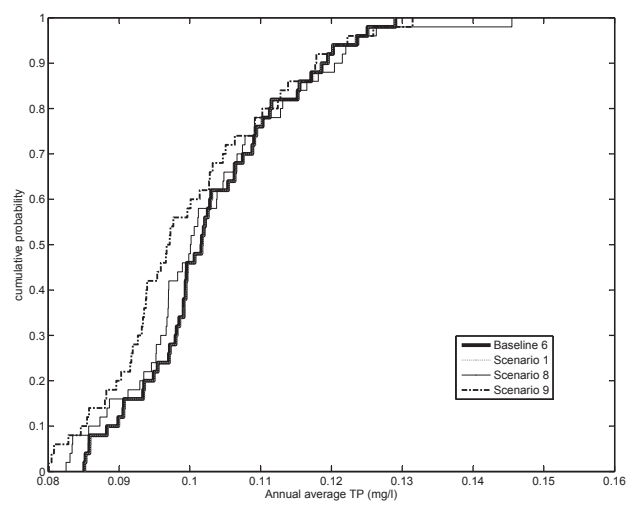


FIGURE 10. Cumulative Distributions of WY2002 Average Total Phosphorus (TP) Concentration at Little Falls Intake in Baseline and Trading Scenarios at Extreme Diversion Condition. The Scenario 9 distribution is significantly less than the baseline scenario distribution.

value, standard deviation, and 90th percentile value; a degree of increase is not surprising at these locations since the trading ratios were not specifically designed for them. Moreover, the violation frequencies of DO criteria were not significantly different between trading and baseline scenarios. Considering the modest increases 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 Wanaque South intake to the Wanaque Reservoir.

CONCLUSIONS

Overall, simulations of worst-case trading scenarios indicated a low risk of hot spots compared to the

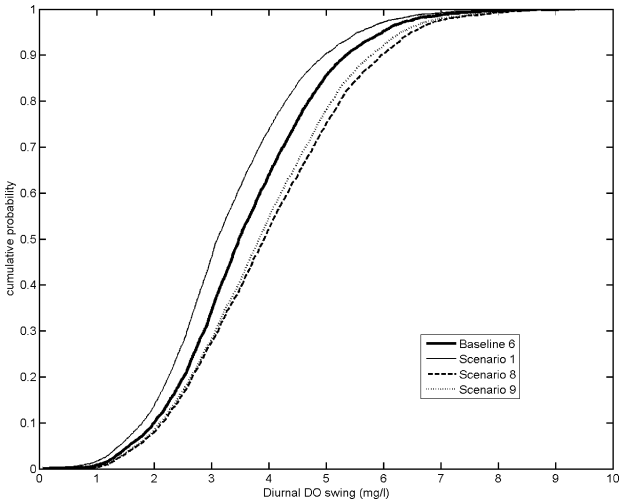


FIGURE 11. Cumulative Distributions of Diurnal Dissolved Oxygen (DO) Swing From June 15 to September 1, 2002 at Passaic River near Chatham in Baseline and Trading Scenarios at Extreme Diversion Condition. The distributions of Scenarios 8 and 9 are significantly greater than the baseline scenario distribution, although the magnitude of the increase is fairly small.

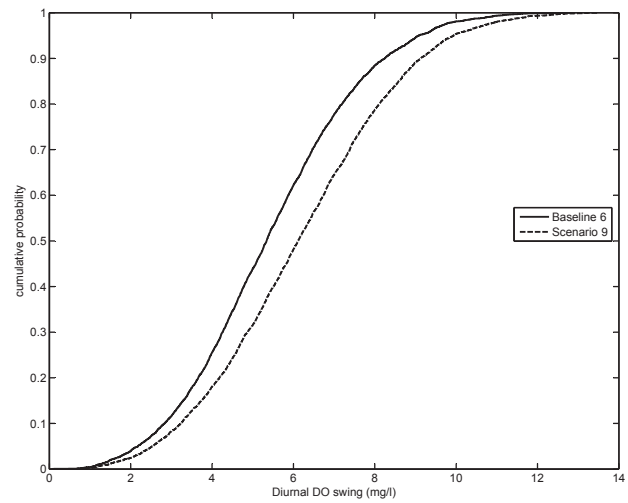


FIGURE 12. Cumulative Distributions of Diurnal Dissolved Oxygen (DO) Swing From June 15 to September 1, 2002 at Peckman River Mouth in Baseline and Trading Scenarios at Extreme Diversion Condition. The trading scenario distribution is significantly greater than the baseline scenario distribution, although the magnitude of the increase is fairly small.

no-trading baseline. The large proportion of trade scenario outcomes that showed no significant increase in the predicted uncertainty of DO, TP, and chl-a targets can be attributed to the use of conservative trading ratios designed for worst-case conditions (Obropta *et al.*, 2010). Moreover, trading scenario outcomes at the critical locations of Dundee Lake and the Wanaque South intake never showed a significant increase above the no-trading baseline in terms of distributions of violation frequencies of DO or distributions of seasonal average chl-a and diverted TP load. 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 potential hot spots in the watershed, even under the most severe circumstances such as inter-management area trading and critical diversion conditions.

It is worth noting that the large majority of WQT programs in the U.S. involve nutrient trading. Examples include the Chesapeake Bay, Long Island Sound, Neuse River Basin, and Minnesota River Basin trading programs. This study examined 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. In other watersheds, trading with nonpoint sources would contain more uncertainties than trading between point sources. By conducting a thorough uncertainty analysis of a point-point source nutrient trading program, this study has established a lower bound on the range of uncertainty regarding nutrient trading programs in general, which could benefit nutrient trading programs nationwide.

In addition, the study produced an efficient uncertainty analysis whose LHS based method could be replicated by regulators charged with administering a WQT program and assessing its various risks. The method's efficiency and practicality directly address a main obstacle that has hindered a wider practice of uncertainty analyses of water quality models.

Future research should focus on site-specific data collection of the WASP parameters modeled as uncertain, to better define probability distributions and parameter correlations. Further efforts could also include the LAWATERS deterministic model (Najarian Associates, 2005) of the Wanaque Reservoir in the uncertainty analysis; results from this study on the uncertainty of diverted TP load from the Wanaque South 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. Another avenue for research would be to compare the effect of different trading ratios. This would elucidate whether the less conservative trading ratios described as alternatives in Obropta *et al.* (2010) pose a greater risk of creating hot spots than the current set of trading ratios.

APPENDIX: DETAILS OF PROBABILITY DISTRIBUTIONS USED IN THE ANALYSIS

TABLE A1. Probability Distributions of Water Quality Analysis Simulation Program (WASP) Global Kinetic Parameters.

WASP Parameter	Unit	PWQM Calibrated Value	Distribution Characteristics: Type	Mean (unless otherwise noted)	SD	COV	Min.	Max.	Notes
Phytoplankton maximum growth rate at 20°C	day^{-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 at 20°C	day^{-1}	0.15	Triangular	0.15	–	–	0.10	0.20	3, 5
Phytoplankton death rate, nonzooplankton predation	day^{-1}	0.1	Triangular	0.10	–	–	0.01	0.19	3, 6
Phytoplankton optimal light saturation	Langleys/day	320	Triangular	320	–	–	290	350	3, 4
Benthic algae maximum growth rate at 20°C	$gD/m^2/day$	60	Triangular	60	–	–	35	85	7
Benthic algae respiration rate at 20°C	day^{-1}	0.01	Lognormal	0.01 = geometric mean	0.0072	0.61	–	–	7
Benthic algae death rate at 20°C	day^{-1}	0.005	Beta (with $p = 1.3$, $q = 10.0$)	0.005 = mode	0.0086	0.70	0.001	0.10	7

Notes: PWQM, Passaic Water Quality Model; COV, coefficient of variation.
1, COV based on the typical COV range 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, based on engineering judgment with consideration of parameter value range reported in Ambrose *et al.* (2006), structural uncertainty of periphyton and macrophyte lumping in the PWQM, and trials of different values.

TABLE A2. Probability Distributions of Water Quality Analysis Simulation Program (WASP) Local Kinetic Parameters Modeled as Lumped.

WASP Parameter	Unit	Target Location	PWQM Segments Varied	PWQM Calibrated Value	Distribution Characteristics: Type	Mean	SD	COV	Min.	Max.	Notes
SOD	$g/m^2/day$	Passaic River near Chatham	179-181	8.0	Normal (bounded)	8.0	3.0	0.375	0	20	1, 2
		Passaic River at confluence with Pompton River	251-256	8.0		8.0	3.0	0.375	0	20	1, 3
		Wanaque South intake	54	3.5		3.5	1.3125	0.375	0	10	1, 2
		Peckman River mouth	295-298	2.0		2.0	1.3	0.65	0	10	1, 3
Fraction of bottom segment covered with benthic algae A	–	Passaic River near Chatham	179-181	0.25	Normal	0.25	0.0375	0.15	–	–	4, 5
		Peckman River mouth	292-294	0.20		0.20	0.03	0.15	–	–	4, 5
			295-298	0.35		0.35	0.0525	0.15	–	–	4, 5
Fraction of bottom segment covered with benthic algae B	–	Passaic River at confluence with Pompton River	255-256	0.15	Normal	0.15	0.0225	0.15	–	–	4, 5
		Station PA10	316	0.20		0.20	0.03	0.15	–	–	4, 5
			317-318	0.30		0.30	0.045	0.15	–	–	4, 5
		B17-N25 (approx. 3km upstream of Dundee Lake)	319-323	0.15		0.15	0.0225	0.15	–	–	4, 5

TABLE A2. Continued.

WASP Parameter	Unit	Target Location	PWQM Segments Varied	PWQM Calibrated Value	Distribution Characteristics						Notes
					Type	Mean	SD	COV	Min.	Max.	
		Dundee Lake	324-325	0.15		0.15	0.0225	0.15	–	–	4, 5
			326	0.30		0.30	0.045	0.15	–	–	4, 5
			327	0.70		0.70	0.105	0.15	–	–	4, 5

Notes: PWQM, Passaic Water Quality Model; COV, coefficient of variation.
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 A3. Probability Distributions of Water Quality Analysis Simulation Program (WASP) Local Kinetic Parameters Modeled as Separate.

WASP Parameter	Unit	Target Location	PWQM Segments Varied	PWQM Calibrated Value	Distribution Characteristics						Notes
					Type	Mean	SD	COV	Min.	Max.	
Settling rate of particulate phosphorus A	cm/s	Little Falls intake	267, 272-274	0.60	Normal (bounded)	0.6	0.18	0.30	0	1	1-4
Settling rate of particulate phosphorus B		Station PA10	299-300, 304-306, 312, 315	0.10		0.1	0.03	0.30	0	1	
		Passaic River at confluence with Pompton River	219-256								
Settling rate of particulate phosphorus C		Peckman River mouth	284-298	0.01		0.01	0.003	0.30	0	1	
Dissolved fraction of ortho phosphate A	–	Passaic River at confluence with Pompton River	247-256	0.60	Normal (bounded)	0.60	0.15	0.25	0	1	5
Dissolved fraction of orthophosphate B	–	Peckman River mouth	292-298	0.60		0.60	0.15	0.25	0	1	
Dissolved fraction of orthophosphate C	–	B17-N4	299-303	0.60	Normal (bounded)	0.60	0.15	0.25	0	1	5
Dissolved fraction of orthophosphate D	–	Dundee Lake	316-327	0.60		0.60	0.15	0.25	0	1	
Fraction of bottom segment covered with benthic algae B	–	B17-N4	299-303	0.15	Normal (bounded)	0.15	0.06	0.40	0	1	6, 7

Notes: PWQM, Passaic Water Quality Model; COV, coefficient of variation.
1, Segments varied for settling rate of particulate phosphorus correspond exactly to the segments varied in the PWQM; 2, normal distribution assumed based on Manache and Melching (2004) and consideration of macrophyte uptake of phosphorus included in this parameter; 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 A; 7, higher COV based on trials with lower values and engineering judgment.

TABLE A4. Headwater Boundary Condition Distribution Characteristics.

WASP Parameter	Distribution Characteristics					Notes
	Type	Mean	SD	Min.	Max.	
Passaic River headwater scaling factor*	Normal (bounded)	1.0	0.30	0	2.5	Based on phosphorus data in Omni Environmental Corporation (2007)
Ramapo River headwater scaling factor*		1.0	0.42	0	2.5	

Note: WASP, Water Quality Analysis Simulation Program.
*Scaling factor applied to both orthophosphate and organic phosphorus boundary condition time series.

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