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Modeling Natural, Organic Matter in an Unfiltered Surface Water Supply

Daniel R. Buttrick
University of Massachusetts - Amherst

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MODELING NATURAL ORGANIC MATTER IN AN UNFILTERED SURFACE WATER SUPPLY

A Master's Project Presented By

Daniel R. Buttrick

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Daniel R. Buttrick

Environmental Engineering Program
Department of Civil and Environmental Engineering
University of Massachusetts

Approved as to style and content by:

Dr. John E. Tobiason
Committee Chairperson

Dr. David P. Ahlfeld
Committee Member

Dr. John E. Tobiason
Graduate Program Director
Environmental Engineering Program

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Much of the work presented herein is based on studies by other DCR and Metropolitan District Commission (MDC) funded researchers. I have met many of these individuals, worked alongside a few of them, and read the words of the rest. Their findings are scattered throughout this document just as their names fill the reference pages. This particular research would not have been possible without such a foundation.

I also owe great thanks to Dr. John Tobiason, my academic and research advisor, for his valuable support, guidance, and patience. His suggestions shaped the work in this document more than those of anyone else. I also wish to thank Dr. David Ahlfeld, my other committee member, for his comments.

Everyone involved ensured that Wachusett Reservoir remained in my mind as a living system, from which the abstract numerical construction on my screen was very distantly removed. During these two years, I have narrowed my eyes against the sunlight scattered by rippling waves as I watched for the Old Church emerge from behind Davenport point. Darleen Bryan and I have stood together on the DCR boat's bow while Dave Worden and Dave Getman coaxed it through South Bay ice two days before Christmas. I have stayed out all night in the rain, soaked to the core, hunching over sparking car batteries while persuading turbid water through cranky filters. I have broken ice on Gates Brook with my grandfather's hatchet. I have seen the Great Blue Heron flying low above Thomas Basin, seeking the solitude interrupted by our motor.

Thank you, family, friends, fellow graduate students, and officemates. I appreciate all your support.

Thank you all for this tremendous learning experience.

Abstract

Wachusett Reservoir, located in central Massachusetts, supplies water to the Boston, Massachusetts metropolitan area. The reservoir receives water from a watershed of 117 square miles, as well as water transferred from the Quabbin Reservoir to the west. Quabbin Reservoir water generally has lower levels of most water quality constituents than water received from the Wachusett Reservoir watershed. The Massachusetts Department of Conservation and Recreation (DCR) manages the watershed and monitors tributary and in-reservoir water quality, while the Massachusetts Water Resources Authority (MWRA) is responsible for treatment and distribution.

CE QUAL W2 is a two dimensional, laterally averaged water quantity and quality modeling program. Version 2 of this program, used in this research, provides the ability to model 21 water quality constituents in addition to water surface elevation and temperature. Constituents modeled in this study include: total organic carbon (TOC) consisting of labile dissolved organic matter (LDOM) refractory dissolved organic matter (RDOM) algae, and detritus; nutrients including nitrate/nitrite, ammonium, orthophosphate; and the absorbance of 254 nm ultraviolet light (UV254). This study implemented CE QUAL W2 to study the sources, fate, and transport of these constituents in Wachusett Reservoir.

The water quality model was calibrated using data from 2001 and 2002. All required input and initial condition data, including inflow and outflow quantities, temperatures, constituent levels, and ambient meteorology were available from DCR, MWRA, the United States Geological Survey (USGS) and the National Oceanic and Atmospheric Administration (NOAA). Values for model parameters were determined to ensure best fit between model predictions and field data for Cosgrove Aqueduct, the main withdrawal of Wachusett Reservoir. The water quality model was then validated with data from 2000 before simulations were run.

NOM levels were relatively constant throughout the calibration period, although seasonal variation is significant. Lower NOM levels generally occur when the majority of withdrawn water originated in Quabbin Reservoir, and higher levels occur when water originating in the Wachusett Tributaries dominates. Measured TOC levels varied from

1.8 to 3.3 mg/L at Cosgrove during 2001 - 2002. These TOC trends were captured by defining 95% of inflow TOC as dissolved organic carbon (DOC), defining the remaining 5% detritus, and then defining 20% of inflow DOC as LDOM, and the remaining 80% as RDOM. First order LDOM and RDOM decay rates of 0.008 and 0.0008 day^{-1}, as well as a first order RDOM to LDOM decay rate of 0.0008 day^{-1} were most appropriate. Maximum algal growth and respiration rates of 1.9 and 0.1 day^{-1} were used. UV254 varied from 0.03 to 0.08 cm^{-1} during the calibration period. These trends were captured by a first order temperature dependent decay rate of 0.0008 day^{-1} and a value of 2.6E-5 cm^2/cal for a constant (α) relating the impact of sunlight irradiance on UV254 decay.

The parameter values determined through calibration were successfully used to validate the model using data from 2000, despite the lack of constituent data for Quabbin Transfer for that year (constituent levels were assumed to be the average of 2001 – 2002 levels). TOC levels at Cosgrove Aqueduct ranged from 1.7 to 3.4 mg/L for that year, while UV254 levels ranged between 0.04 and 0.08 cm^{-1}.

Two simulations showed that transferring water from Quabbin to Wachusett at 8.7 m^3/s during periods of high tributary runoff may reduce TOC levels at Cosgrove by up to 0.2 mg/L and UV254 levels by up to 0.008 cm^{-1}. A third simulation demonstrated that a large runoff event occurring in late summer/ early fall may lead to large increases in TOC and UV254 levels at Cosgrove, and may result in an unusual algal bloom. A fourth simulation was run to evaluate the impact of bypassing Wachusett Reservoir with Quabbin Transfer; the model predicts that TOC and UV254 levels at Cosgrove will increase, but the increased mean hydraulic residence time within Wachusett Reservoir results in more decay of those constituents. The resulting mixture of Quabbin and Wachusett water in similar proportions to those that actually occurred contains lower NOM levels than would exist if the bypass did not occur.

These results show that Wachusett Reservoir constituent levels at the Cosgrove withdrawal are strongly source driven, although in-reservoir processes are also important. The calibration and validation results indicate that CE QUAL W2 can be effectively used to predict NOM levels at Cosgrove. The simulation results suggest that controlled and uncontrolled events may impact water quality at Cosgrove.

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1. INTRODUCTION

The subject of this study is the Wachusett Reservoir, located in central Massachusetts. This reservoir is an unfiltered water supply for 2.2 million consumers in Boston, Massachusetts and surrounding communities. The Massachusetts Department of Conservation and Recreation (DCR) is responsible for managing and protecting the reservoir and its watershed, and the Massachusetts Water Resources Authority (MWRA) operates the water supply system. DCR rigorously manages the watersheds of Wachusett Reservoir and the other major reservoir in the system, the Quabbin Reservoir. These relatively pristine watersheds, coupled with long hydraulic residence times, provide high quality water to consumers. Studying the complex processes that occur within the reservoirs can lead to improved management and operational practices for the protection of water quality.

1.1 Objectives and Scope of Work

Of particular interest to this study of Wachusett Reservoir are the origin, fate, and transport of natural organic matter (NOM). Treatment of Wachusett water to prevent adverse affects on human health consists only of disinfection; coagulation and filtration are not performed. The lack of these treatment processes was not a concern until recently, as the low turbidity of the water would not interfere with disinfection and taste and odor was not usually a problem. However, the reaction of chlorine with NOM and the formation of disinfection by-products (DBPs) have become of particular interest in environmental engineering since the 1970s. The operators of Wachusett have few tools for control of DBPs.

The objective of this research was to investigate the origin, fate, and transport of water quality parameters (constituents) that are components of natural organic matter (NOM) within the Wachusett Reservoir. The scope of work included data gathering, modeling, and limited field work. The investigation was based on the previous organic matter characterization and modeling studies performed on Quabbin Reservoir (Garvey 2000; Roberts 2003), as well as hydrodynamic modeling studies performed on Wachusett Reservoir (Joaquin 2001). CE QUAL W2, a two dimensional, laterally averaged hydrodynamic and water quality modeling program was used to assemble and calibrate a model for the reservoir for the 2001 and 2002 calendar years. The resulting calibration was then validated for 2000. Constituents modeled include algae and detritus (algae and detritus together may be referred to as particulate organic carbon, POC; while detritus may be referred to as labile particulate organic matter, LPOM); labile and refractory dissolved organic matter (LDOM and RDOM) as components of dissolved organic carbon (DOC); nutrients that impact the generation of algae including orthophosphate, ammonium, and nitrate/nitrite; and absorbance of ultraviolet light at a wavelength of 254 nm (UV254) as a surrogate for DOC. Total organic carbon (TOC) was modeled indirectly as the sum of POC and DOC, or the sum of algae, detritus, LDOM and RDOM. The resulting model was then used to run simulations to determine the impact of different variables on the quality of water at the Cosgrove Intake, the main water supply withdrawal from the reservoir.

1.2 DCR/MWRA system

The Wachusett and Quabbin Reservoirs are the main supply components of a system that supplies drinking water to 2.2 million residents of more than 40 communities in eastern Massachusetts. Quabbin Reservoir, the newer and more pristine of the two reservoirs, holds 412 billion gallons of runoff from a watershed of 187 square miles located in Western Massachusetts. Water from Quabbin is selectively discharged to the Wachusett Reservoir via the 24.6 mile Quabbin Aqueduct. This aqueduct may also be utilized to transfer water from the Ware River to Quabbin through an intake partway between the two reservoirs. Ware River transfer can only occur when the transfer of Quabbin water to Wachusett is not occurring.

Wachusett Reservoir is a smaller reservoir, located in Central Massachusetts north of Worcester. From Wachusett, water is withdrawn and currently receives free chlorine primary disinfection, pH and alkalinity adjustment, and chloramination only. Ozonation for primary disinfection began during the summer of 2005. The system has a total safe yield of about 300 million gallons per day (MGD).

The system has been administered by two Massachusetts state agencies since 1984. The DCR, formerly the Metropolitan District Commission (MDC) is the watershed and water body steward responsible for protecting water quality through land management, water and watershed monitoring, and emergency response. The MWRA is responsible for operation of the reservoir system, including Quabbin Transfer, treatment, and the transmission system

1.3 Wachusett Reservoir

Wachusett Reservoir has a capacity of 65 billion gallons and, collecting runoff from a watershed of 117 square miles in Central Massachusetts, and has a safe yield of about 100 MGD. Wachusett is 8.4 miles long with a surface area of 6.3 square miles, a 37 mile shoreline, and a maximum depth of 36.6 meters. Through transfer of Quabbin water, MWRA operates Wachusett to maintain a water surface elevation of between 391.5 feet (119.3 m) above Boston base and 390 feet (118.9 m), an elevation below which exposed ledges provide unwanted bird roosting. The reservoir spills to the Nashua River via adjacent upper and lower spillway sections, the lower with a crest elevation of 392 ft (119.5 m) and the upper with a crest elevation of 395 ft (120.4 m). Stop logs are sometimes used to raise the lower spill elevation.

Figure 1.1 shows Wachusett Reservoir and its major inflows and outflows. The reservoir occupies a valley where the Stillwater and Quinapoxet Rivers meet to form the Nashua River. The former headwaters of Nashua River are now beneath the Thomas Basin which, with the Stillwater Basin (where Waushacum Brook enters), constitute the north westernmost portion of the reservoir. Stillwater River flows from the north and joins the east-flowing Quinapoxet River here. The Oakdale Power Station (Shaft 1) where the Quabbin transfers enter Wachusett is located on the Quinapoxet River just upstream of Thomas Basin. The reservoir extends southeast to Davenport Point on the north shore opposite South Bay. From Davenport Point, the main channel lies northeast. The next segment, between the point and a constriction known informally as the narrows, is referred to as South Basin. Beyond the narrows the reservoir is widest at North Basin which is defined by, clockwise from the narrows, the concave ‘V’ of North Dike, the masonry Wachusett Dam and spillway, the Cosgrove withdrawal structure, and South Dike.

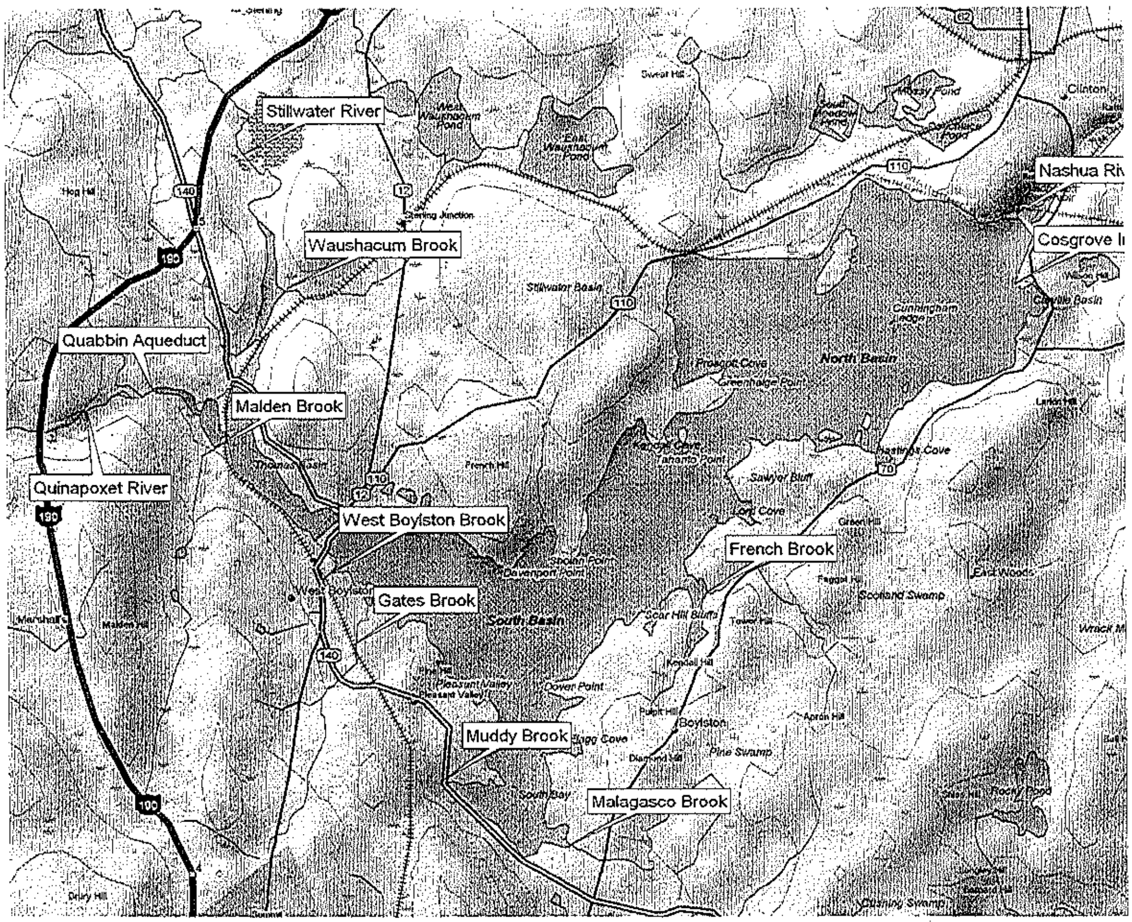

Figure 1.1 Major Wachusett Reservoir inflows and outflows (Delorme TopoUSA 3.0 2001, shown in Ahlfeld et al. 2003).

1.3.1 Tributaries

The major tributaries of the reservoir are the Stillwater and Quinapoxet Rivers. The watersheds of these rivers are the largest within the Wachusett Reservoir basin, representing about 73% of the basin area (Tobiason *et al.* 2002). Minor tributaries include Waushacum Brook, Malden Brook, West Boylston Brook, Gates Brook, Muddy Brook, Malagasco Brook, and French Brook. The watershed also includes several intermittent streams, including Potash Brook, Hastings Cove Brook, Oakdale Brook, Meadow Brook, and a few unnamed streams. Water received from tributaries generally accounts for 30 to 55 percent of the annual water budget of the reservoir, while Quabbin transfers generally account for 40 to 65 percent and precipitation directly on the surface of the reservoir accounts for about 5 percent.

1.3.2 Withdrawals

The main withdrawal from the reservoir is the intake for the Cosgrove Aqueduct, through which water is discharged to metropolitan Boston. This discharge accounts for 90 to 95% of the annual water budget of the reservoir. Cosgrove was built as a replacement for the Wachusett Aqueduct, which is maintained to backup the newer aqueduct. Normal operation includes a 2 MGD discharge through the Wachusett aqueduct, though it underwent a period of 220 MGD testing in October of 2002, in preparation for the shutdown of Cosgrove for a maintenance period in 2003.

Other withdrawals from Wachusett Reservoir include withdrawals by the towns of Worcester, Clinton, and Leominster, seepage through the North Dike, evaporation, and spillway and controlled discharges to the Nashua River.

1.3.3 General Water Quality

Profiles measured in Wachusett Reservoir during 2001 and 2002 indicate high water quality. During this period, the reservoir was slightly acidic with an average pH of 6.4 and a range of 5.0 to 7.3. Average alkalinity was 4.9 mg/L. Reservoir conductivity generally ranges between 80 and 100 μS/cm. Wachusett is considered borderline oligo-mesotrophic based on total phosphorus data ranging from 0.001 to 0.038 mg/L during 1994 (CDM, 1995) and ranging from 0.005 to 0.037 mg/L during the period of this study.

Wachusett Reservoir is dimictic, demonstrating typical summer stratification though with a somewhat unusual thermocline that is influenced by transfer from Quabbin Reservoir. The reservoir shows some seasonal pH dependence and pH variability with depth, as well as strong seasonal dissolved oxygen dependence and variability with depth. Hypolimnion dissolved oxygen concentrations do not generally fall below 4 mg/L and have never been observed to become anaerobic (Worden 2004).

1.3.4 Quabbin Transfer Interflow

Quabbin Transfer has a significant impact on Wachusett water quality as it is characterized by lower specific conductivity (~40 μS/cm), and lower concentrations of NOM and nutrients. It is withdrawn from Quabbin at a depth of between 9 and 13 meters where water temperatures are generally between 13 and 14 °C in summer (Worden 2003).

Upon discharge to Wachusett Reservoir (during stratified periods), Quabbin water warms slightly, but not to the temperature of the epilimnion of Wachusett. Warmer than the hypolimnetic waters, but colder than those of the epilimnion, the Quabbin water travels through the metalimnion as an interflowing density current.

This metalimnetic current is easily identifiable in measured profiles as a depth interval within the thermocline with little spatial change in temperature. It is also identified as a depth interval of low conductivity and low nutrient water compared to intervals deeper and more shallow within the water column. Three to 5 weeks and 5.5 to 8 billion gallons of transfer from Quabbin are required for the interflow to be identifiable at Cosgrove Intake (Worden, 2003). An extensive study of this phenomenon indicated that interflow travel time is highly dependent on the thermocline gradient of Wachusett Reservoir (Joaquin 2001; Ahlfeld *et al.* 2003b)

1.4 Data Availability

Data utilized in the course of this research were obtained from several state and federal agencies as discussed in this section.

1.4.1 Water Quantity Data

The United States Geological Survey (USGS) and DCR are the primary sources of water quantity data. USGS maintains stream gages on both the Stillwater and Quinapoxet Rivers. Fifteen minute instantaneous depth data are available at both locations. The Stillwater River has been gaged since April 22, 1994, and Quinapoxet since November 20, 1996. Staff gages and rating curves are maintained by USGS at Malden, West Boylston, Gates, Muddy, and Malagasco Brooks. DCR records instantaneous depths from these staff gages weekly. Daily discharge estimates for the smaller tributaries including those with staff gages are generated using measurements from the Stillwater gage.

MWRA records daily measured Quabbin Transfer discharges at the outlet of the aqueduct. MWRA also records daily Cosgrove and Wachusett Aqueduct discharges and Nashua River discharges. The quantity of water withdrawn by Clinton, Leominster, and Worcester are obtained and recorded by DCR.

DCR measures the water surface elevation (WSE) of Wachusett Reservoir daily. These measurements are used to determine reservoir storage through a rating curve.

Groundwater infiltration to the reservoir is not measured. The method of water budget calibration used by the University of Massachusetts (UMass) implicitly incorporates groundwater inflows by increasing stream discharges, as discussed by Tobiason *et al.* (2002).

Precipitation data is measured at a series of cooperative weather stations in the Wachusett Reservoir Watershed, as well as at a National Oceanic and Atmospheric Administration (NOAA) station at Worcester Airport, at Stillwater River stream gage, and at the MWRA station the Cosgrove withdrawal (Clinton, MA). Data for 2000 and 2001 was taken from the cooperative and NOAA stations for this research, while data for 2002 was taken from the MWRA and NOAA stations.

1.4.2 Water Quality Data

DCR measures conductivity and temperature and analyzes fecal and total coliform at Hastings, French, Malagasco, Muddy, Malden, Waushacum, Gate, West Boylston, Cook, Jordan Farm, and Rocky Brooks and the Stillwater and Quinapoxet Rivers on a weekly basis. Constituent concentrations were also measured in many of these tributaries on a monthly basis during 2001. Constituents analyzed include nitrate-nitrogen, nitrite-nitrogen, ammonia, silica, total phosphorus, UV-254, total suspended solids (TSS), and TOC. Monthly analyses of these parameters were continued in 2002 for the Stillwater and Quinapoxet Rivers and reduced to biannually for the other tributaries. It is notable that nutrient data does not exist for Waushacum Brook, which is the third largest tributary of the reservoir.

Some of the above constituents are measured in the precipitation at two National Atmospheric Deposition Program (NADP) stations in Massachusetts, one located on the Prescott Peninsula of the Quabbin Reservoir and one at Lexington. Numerous nutrients are measured at these locations, including nitrate and ammonium; however, phosphorus data does not exist. Precipitation concentrations of phosphorus and TOC are based on work by Garvey *et al.* (2002).

In-reservoir data is recorded by and obtained from DCR. This agency records temperature, dissolved oxygen, pH, and conductivity profiles at three locations on a monthly basis, starting just before stratification and continuing until the end of the year. Profiles may exist more or less frequently depending on weather conditions or specific needs. Additionally, DCR collects and analyzes for nitrate-nitrogen, ammonia-nitrogen, total Kjeldahl nitrogen (TKN), silica, alkalinity, and total phosphorus at the epilimnion, metalimnion, and hypolimnion of the three profiling stations on a quarterly basis.

The phytoplankton ecology of Wachusett Reservoir is monitored by DCR when the population is active. Sampling within the reservoir is conducted monthly or more frequently when necessary, and weekly at Cosgrove Intake.

Also at Cosgrove Intake, data is collected by MWRA for 42 water quality parameters at variable temporal frequencies. These parameters include nitrate, nitrite, orthophosphate, specific conductance (conductivity), TOC, and UV-254.

In September and December 2004 and May of 2005, UMass collected and analyzed samples from the epilimnion, metalimnion, and hypolimnion of the three DCR profiling locations for TOC, DOC, and UV-254 to supplement DCR/MWRA data and partially characterize reservoir NOM.

1.4.3 Meteorological Data

Additional Meteorological data are available at the NOAA station at Worcester Airport and at a weather station operated by MWRA, situated at the Cosgrove outlet building. Data available at the NOAA station include temperature, dew point, relative humidity, wind speed, wind direction, cloud cover, and visibility. Data recorded at the MWRA station include barometric pressure, humidity, air temperature, wind speed, wind direction, and rainfall. The MWRA data are not typically used during analysis as the proximity of the weather station to the building affects local wind currents, rendering the wind speed and wind direction data unreliable. It is likely that this station will soon be moved to a better location.

2. Literature Review

The literature review presented in this section presents background information for this study. Topics discussed include the importance of natural organic matter (NOM) in drinking water treatment (Section 2.1); computational reservoir modeling (Section 2.2); a discussion of applications of CE QUAL W2, the modeling software selected for this research (Section 2.3); the nature, origin and decay of NOM in lakes (Section 2.4); and parameter values necessary for implementing a model of NOM (Sections 2.5 and 2.6).

2.1 NOM in Drinking Water

NOM in drinking water is of particular concern. It imparts color, increases coagulant demand, disinfectant demand, and may react with disinfectant to form potentially harmful disinfectant by-products (DBPs). Color is generally not of concern in Wachusett Reservoir water, and coagulation is not currently implemented. However, high DBP levels are occasionally measured in the distribution system; total trihalomethanes (TTHMs) sometimes exceed 100 µg/L (MWRA concern level is 80 µg/L) and the sum of five regulated haloacetic acids (HAA5) sometimes exceed the MWRA concern level of 60 µg/L. Trihalomethanes (THMs) and chloroform are the most prevalent DBP forms measured in the MWRA distribution system (Sung *et al.* 2000). DBPs may be carcinogenic, may cause adverse reproductive and developmental effects, and may be mutagenic and hepatotoxic (AWWA 1999). Hundreds of DBP compounds have been identified. TTHMs and HAA5 are currently regulated by the U.S. Environmental Protection Agency (EPA 1998). Modeling NOM sources, fate, and transport in Wachusett Reservoir may be useful for predicting reservoir response to controlled and uncontrolled events, and possible impacts on DBPs

2.2 Reservoir Modeling

Models are often used to explain complicated hydrologic and ecologic systems, such as reservoirs. A model serves to represent the response of a physical system to external stimuli (Chapra 1997). Numerous methods are available to represent aquatic systems in a computational environment, each with unique benefits and drawbacks.

Modeling of the DCR/MWRA system was first undertaken to investigate levels of coliform bacteria at the primary withdrawal of Wachusett Reservoir. Camp, Dresser, and McKee (1995) evaluated 10 hydraulic and water quality modeling programs that were then available to determine which was best suited for modeling the reservoir.

Five of the evaluated programs represented aquatic systems in 1 dimension (1-D). 1-D models often assume lateral and vertical homogeneity. This assumption is adequate for modeling a long, narrow, and shallow water body where processes along the length of the water body dominate (i.e. a river). This type of model can be solved analytically for steady-state conditions, and often consists of a series of complete-mix segments that are solved numerically for transient conditions (Chapra 1997). In a recent study, Westphal *et al.* (2004) utilized a 1-D longitudinal model for Wachusett Reservoir during unstratified conditions. The model divided the reservoir into 5 segments and was used to model TOC and UV-254 as a decision support system (DSS) tool for reservoir operations.

Vertical 1-D models may also assume lateral and axial homogeneity for modeling two or more layers. A vertically stratified impoundment might be modeled with two layers, the upper layer representing the epilimnion and the lower layer representing the hypolimnion. Oppenheimer *et al.* (1994) developed a one segment, two layer model for simulating TOC in the 11 billion gallon Lake Youngs, a distribution reservoir in the water supply system for the City of Seattle, Washington. The model successfully predicted the spring diatom bloom that occurs annually in that reservoir. Simulations were conducted to predict the impact of changes in operational hydraulics on water quality.

2-dimensional (2-D) models generally combine a system of layers and segments to form a grid of complete-mix boxes representing impoundment or riverine geometry. This arrangement is appropriate when axial and vertical variability dominate lateral variability. 2-D models are generally more computationally intensive than 1-D models, but are capable of modeling complex systems more accurately. Westphal *et al.* (2004) added three layers to a five segment 1-D model to model TOC and UV-254 in Wachusett Reservoir during periods of stratification. Three layers were necessary to capture effects of Quabbin Interflow (see Section 1.3.4). The model was simplified to limit advection to the metalimnetic layer, with no direct interaction between the adjacent segments in the upper and lower layers.

Some 2-D models assume vertical homogeneity and have segments associated laterally and longitudinally. The WASP5 modeling program, distributed and supported by the EPA (EPA 1993) may be used in this manner. Tufford and McKellar (1999) used this program to model the shallow, eutrophic Lake Marion in South Carolina. The reservoir remains unstratified in some years, and measured longitudinal and latitudinal gradients in water velocity and quality suggested that a vertical homogeneity assumption was appropriate.

3-dimensional (3-D) models are often used to model impoundments with large gradients in three dimensions. 3-D models are generally computationally expensive, requiring a third set of equations to be solved for each element. CDM (1995) evaluated one 3-D modeling package for Wachusett Reservoir. More recently, the Thomas Basin of Wachusett Reservoir was modeled with a 3-D computational fluid dynamics (CFD) program (Kennedy 2003; Pease 2004). Thomas Basin is an area characterized by complex hydraulics that are not accurately captured by a 2-D model. Kennedy (2003) used Fluent 6.0 along with the mesh-generation program Gambit 2.0 to model Thomas Basin. The irregular mesh selected only allowed for unstratified conditions to be modeled. In a separate study, a hexahedral mesh was generated for modeling thermally stratified cases (Pease 2004). The results led to the understanding of a number of unusual phenomena in Thomas Basin, including backflow into the Quinapoxet Basin during periods of high Quabbin transfer, and exposing the pattern of gyres that form within the basin. Results from these analysis were compared to field data. Large computational requirements make these 3-D CFD models impractical for modeling reactive constituents.

2.3 CE QUAL W2

CE QUAL W2 is a 2-D, laterally averaged hydrodynamic and water quality model under continuous development and support by the United States Army Corps of Engineers (Cole and Buchak 1995). Through the addition of branches, CE QUAL W2 can be used in a quasi 3-D manner to model side channels. Version 2 of CE QUAL W2 is capable of modeling water temperature and 21 separate model quality constituents. Table 2.1 presents these compartments, along with their constituent numbers.

Table 2.1 Possible Constituents Modeled by CE QUAL W2 Version 2 with UV254 replacing coliform bacteria (UMass modification).

Constituent	No.
Conservative Tracer	1
Inorganic Suspended Solids	2
UV254	3
Total Dissolved Solids	4
Labile DOM	5
Refractory DOM	6
Algae	7
Detritus	8
Phosphate	9
Ammonium	10
Nitrate-Nitrite	11
Dissolved Oxygen	12
Sediment	13
Total Inorganic Carbon	14
Alkalinity	15
pH	16
Carbon Dioxide	17
Bicarbonate	18
Carbonate	19
Iron	20
CBOD	21

CE QUAL W2 has been used in a number of studies regarding reservoir water quality and operation. Bartholow *et al.* (2001) implemented a CE QUAL W2 model to predict the impact of a temperature control device (TCD) on Shasta Lake in northern California which was installed to improve endangered salmonid habitat downstream. Results showed that the TCD impacted reservoir water quality less than expected.

A similar application of the program was used to model the impact of changes in water quality, including reduction in organic matter and phosphorus loadings, on the suitability of areas in Brownlee Reservoir, Idaho as sturgeon habitat (Sullivan *et al.* 2003).

Bowen and Hieronymus (2003) implemented a hydrodynamic and water quality CE QUAL W2 model to support a total maximum daily nutrient load (TMDL) analysis for the Neuse Estuary in North Carolina. The program was appropriate for the application as that estuary has shown significant longitudinal and vertical gradients in water quality.

Cannonsville Reservoir of the water supply system for New York City was modeled using CE QUAL W2. Gelda *et al.* (1998) reported successful calibration and verification of a hydrodynamic model using the program which was applied to simulate the spill of a generic conservative contaminant.

CDM (1995) evaluated four 2-D models for Wachusett Reservoir, selecting CE QUAL W2 (See Section 3.1). The modeling grid consisted of 5 branches, 26 layers of 1.5 meter depth each, and 62 segments. The shallow depth of each layer allowed for vertical temperature and specific conductivity profiles to be predicted and compared to existing data. The model was calibrated with data from the 1987, 1990, and 1992 calendar years, validated with data from 1994, and used to predict temperature, specific conductivity, fecal coliform, dissolved oxygen, phosphorus, and chlorophyll.

The Wachusett Reservoir model was further developed at the University of Massachusetts at Amherst (UMass) Department of Civil and Environmental Engineering. The modeling grid was refined by reducing layer thickness in the upper 15 m of the model from 1.5 m to 0.5 meters to better predict epilimnion temperatures, and by adding two segments at the downstream end of the main branch to improve withdrawal geometry. The model was updated for calendar years 1998 and 1999 and was used to analyze the impact of Quabbin transfer and study the Quabbin interflow density current (Joaquin 2001; Tobiason *et al.* 2002; Ahlfeld *et al.* 2003b). Simulations found that, given adequate transfer quantity and stratification, up to 95% of the water being withdrawn from Wachusett Reservoir at the Cosgrove Intake originated at Quabbin with a travel time of only a few weeks, despite the 6 to 7 month average hydraulic retention time of the reservoir.

For water years 1987 to 1999, water budget calibration was accomplished using multiplicative factors for every hydrologic input except precipitation. The factors were adjusted by trial and error to minimize the difference between measured and predicted water surface elevation (WSE). The model was updated again for the 2000 and 2001 calendar years using a water balance external of the modeling program to determine the input calibration factor values (Ahlfeld *et al.* 2003a). The result of the external analysis was then applied to input files for CE QUAL W2 and confirmed via comparing modeled temperature and specific conductivity profiles to measured profiles. This method was then used to recalibrate water balances for 1994 through 1999.

2-D water quality modeling using CE QUAL W2 has also been conducted for Quabbin Reservoir using similar methods. A model was established to study hydraulics and coliform (Tobiason *et al.* 1996; Tobiason *et al.* 1998). The CE QUAL W2 model was then implemented to study NOM fate and transport and was modified to include a UV-254 absorbance subroutine in place of coliform. This work was based on extensive data collection and analysis by Garvey (2001) and is reported in Roberts (2003).

2.4 Lacustrine Organic Matter

Natural organic matter (NOM) within an aquatic system can be classified by several methods. Of interest in this research is the origin of the organic material, (either allochthonous or autochthonous) the bulk physical characteristics of the material (dissolved or particulate), and the environmental persistence of the material (labile or refractory). The organic matter pool of freshwater can comprise 80% humic material, often with 30 to 40% of this fraction composed of aromatic carbon (Malcolm 1990). Humic material, responsible for the recalcitrant component of organic carbon in lakes, generally consists of humic and fulvic acids. Humic acids have high molecular weight, high aromaticity, are colloidal in structure, and contain carboxyl, hydroxyl, phenol, and methoxyl groups (Steinberg and Muenster 1985, Aiken *et al.* 1992). Fulvic acids are more hydrophilic and generally include carboxyl, hydroxyl, and carbonyl groups surrounding an aromatic ring (Wetzel 2001). Fulvic acids are generally more labile than humic acids. It is often difficult to discern from the literature if a reference to humic materials refers specifically to humic acid, or instead to compounds that impart a dark color to the water, including but not limited to humic and fulvic acids.

Non-humic lacustrine organic matter consists of proteins, fats, carbohydrates, peptides, amino acids, waxes, resins, and other low molecular weight organic compounds. Non-humic material is more biologically labile and more readily synthesized; although it cycles rapidly and is present at low concentrations, it represents important energy fluxes and carbon pathways (Wetzel 2001). Aquatic fulvic acids are generally 20-80 percent of dissolved organic carbon (DOC), while hydrophilic acids constitute 5-20 percent of DOC (Aiken and Cotsaris 1995).

Common measures of the quantity of aquatic organic matter are DOC and total organic carbon (TOC) content (mass per volume). For a single lake, DOC generally varies little with depth and season, although fractionation may vary. Additionally, high variation in DOC content is often found within a geographical region (Steinberg and Muenster 1985). DOC within tributaries can be more variable; in a study of several tributaries flowing into Wachusett Reservoir, Bryan (2005) noted variations in DOC and TOC that were dependent on discharge, temperature, and season.

It is also common to measure the absorbance of light by a water sample as a surrogate of organic matter content. Measuring the absorbance of a spectrum of visible and ultraviolet light is often conducted during ecological studies. In the drinking water industry, measurement of the absorbance of ultraviolet light at a wavelength of 254 nm (UV254) is most common. Aromatic organic compounds, and other organic compounds with conjugated double bonds, most effectively absorb UV light (Edzwald *et al.* 1985). UV254 absorbance can therefore be used as a surrogate indicator of humic material. UV254 is often strongly correlated with DOC (Bryan 2005, Garvey 2000) or nonpurgeable total organic carbon (NPTOC; Edzwald *et al.* 1985) and correlated to disinfection by-product formation potential (DBPFP) (Bryan 2005; Edzwald *et al.* 1985; Garvey 2000; Sung 2003; Weishar *et al.* 2003).

In a study of the Grasse River and Glenmore Reservoir, Edzwald *et al.* (1985) proposed that calculating the ratio of UV to TOC (later known as specific UV absorbance, SUVA) could be used to judge the applicability for correlation equations established for one water source to another. In another study, the SUVA of 13 organic matter isolates was shown to correlate strongly with ^{13}C nuclear magnetic resonance percent aromaticity (Weishaar *et al.* 2003). However, SUVA is sometimes poorly correlated with specific DBPFP (Bryan 2005; Garvey 2000; Weishar *et al.* 2003), especially with waters from diverse sources. Bryan (2005) observed that highest DBPFP was observed at moderate SUVA (~3.5 L/mg-m) suggesting that a mixture of hydrophilic (low SUVA) and hydrophobic (high SUVA) may be most effective at producing DBPs. The only mechanistic reservoir model for UV254 prediction found in the literature is based implicitly on constant SUVA (Westphal *et al.* 2004).

In summary, aquatic organic matter contains humic and non humic material. Humic material contains humic and fulvic acids which are relatively recalcitrant in the environment. Non-humics decay rapidly. A combination of relatively simple measurement techniques are often used to determine bulk organic matter quantities and characteristics, although they must be used with discretion.

2.4.1 Allochthonous Sources

Allochthonous sources of organic matter include tributary and direct runoff (fluvial) inputs, groundwater inputs, shoreline litter, and precipitation. Precipitation is the only non-terrestrial source of NOM, with the organic matter bound in pollen, dust, bacteria, spores, and natural and anthropomorphic volatile organic carbons (Jordan and Likens 1975). Terrestrial sources receive much of their organic matter from the soil in a watershed (Aiken and Cotsaris 1995).

Thurman (1985) discussed in detail the quantity and character of allochthonous organic carbon. In summary, the quantity of DOC in a river varies with climate, river size, and type of vegetation. Locations of cooler climate, and runoff originating in cooler seasons, generally have lower DOC levels since productivity is low. DOC of rivers in cool temperature regions is typically 2 – 8 mg/L with a mean of 3 mg/L. For small rivers, DOC generally dominates POC, and in lakes, DOC typically comprises 90% of TOC (Thurman 1985).

Generally, organic matter in small, canopied streams is assumed to be of terrestrial origin, while autochthonous production may be a significant organic matter source in streams and rivers with decreased velocity and cover (Wetzel 1983). Often, allochthonous DOC inputs to reservoirs can exceed autochthonous production by several times (Wetzel 2001). Fluctuation in riverine TOC levels resulting from flow variation is often noted (Thurman 1985). This flushing effect was observed by McDowell and Fisher (1976) in Roaring

Brook at Mt. Toby State Forest, Sunderland, Massachusetts. DOC concentrations ranged between 1 and 2.5 mg/L during baseflow conditions and increased to 3 to 5 mg/L during high flow conditions. A study of western (Arizona) river systems noted shifts from predominantly autochthonous DOC sources to allochthonous sources during runoff events (Westerhoff and Anning 2000).

Quantities of allochthonous organic matter entering a system are relatively simple to determine. Doing so requires the identification of input sources, determining or estimating the quantity of inputs, and determining the influent organic matter levels. Rivers are often the largest and easiest to quantify allochthonous sources of organic matter. Contributions of organic matter from direct runoff can be estimated with precipitation data, estimated runoff coefficients, and estimated organic matter content. Atmospheric deposition can be estimated with local precipitation quantity and organic matter content data, although Jordan and Likens (1975) suggest that precipitation DOC concentrations are comparable at diverse locations. Gains or losses of NOM via groundwater can be estimated by examining local hydraulic gradients and groundwater organic matter content.

Jordan and Likens (1975) constructed a carbon budget for Mirror Lake in New Hampshire. The authors quantified all significant organic carbon fluxes into the lake excluding groundwater. During the study period, fluvial sources accounted for 60% of organic carbon inputs to the lake, shoreline litter accounted for ~25%, and direct precipitation for ~15%. The impact of precipitation on the organic carbon budget was more significant than the authors expected.

In a study of Quabbin Reservoir, Garvey (2001) estimated that the mass of stored TOC was an order of magnitude larger than the net monthly flux into the reservoir. These results indicate that changes in tributary inputs would have a delayed impact on reservoir water quality. The study also reported a larger organic carbon loading to the reservoir than was discharged, indicating a net loss in the system. Using data from Garvey (2001) and DCR, Roberts (2003) showed that the large surface area of Quabbin Reservoir relative to its watershed makes it particularly susceptible to precipitation effects.

It is also possible to predict inputs of allochthonous sources of organic matter. Boyer *et al.* (1996) implemented a hydrologic catchment model to simulate streamflow response to hydrologic events. This hydrologic model was then coupled with a simple model to predict changes of organic matter within a terrestrial 'reservoir,' in which DOC accumulates during periods of low flow, and is then flushed during large flow events. The model predicted DOC levels in the stream with reasonable accuracy.

Allochthonous DOC originating from terrestrial and marsh plants is generally dominated by humic and fulvic acids. This organic matter is very low in nitrogen content (~2%) compared to autochthonous organic matter (~8.3%) (Wetzel 2001). CE QUAL W2 utilizes one parameter defining the nitrogen content of algae and DOM. Cole and Buchak (1995) recommend that 8% be used for this value.

Aiken and Cotsaris (1995) discuss the influence of soils on the characteristics of organic matter in surface water. The authors note that riverine DOC content is often not dependent on soil DOC content, but on soil structure (i.e. clays have been shown to retain DOC) and composition (Ca^{2+} and Mg^{2+} have been shown to decrease the solubility of high molecular weight organic matter, essentially removing it from the water column).

2.4.2 Autochthonous Sources

The growth of algae generally accounts for most autochthonous organic matter within an aquatic system, although macrophytes and predators also contribute NOM. The relative contribution of autochthonous and allochthonous sources depends largely on the hydraulic characteristics of the water body. Systems with short retention times tend to have less phytoplankton production and biomass, and thus are generally dominated by allochthonous sources (Likens 1983). Additionally, larger water bodies tend to contain larger proportions of autochthonous organic matter than smaller water bodies, as the size of the littoral zone of a large lake is smaller relative to the overall lake size than that of a smaller lake (Thurman 1985).

Jordan and Likens (1975) determined that, despite the oligotrophic nature of Mirror Lake, autochthonous production is responsible for 83% of its organic matter. 90% of this production is generated by algae and the remaining 10% is generated by epilithiphyton, macrophytes, and bacteria. Garvey (2000) reached a similar conclusion for oligotrophic

Quabbin Reservoir. It was found that autochthonous production of organic matter was of the same order of magnitude as allochthonous inputs. Quabbin Reservoir tributary TOC was less than 2 to greater than 10 mg/L, and UV254 ranged from 0.06 to 0.28 cm^{-1}. The resulting SUVAs were generally between 2.9 to 3.9 L/mg-m, indicating an influent mix of humic and fulvic acids. In-reservoir, TOC generally ranged from 2 to 3 mg/L, UV254 from 0.02 to 0.03 cm^{-1}, and SUVA from between 1 and 1.5 L/mg-m. The lower in-reservoir SUVA values indicate that the nature of the NOM in the reservoir has changed significantly compared to the tributary inputs.

Pathways of autochthonous NOM generation include production of algal biomass, soluble extracellular products production, and generation of detritus through death. Extracellular products can include glycolic acid, carbohydrates, polysaccharides, amino acids, peptides, organic phosphates, VOCs, enzymes, and others (Wetzel 1983). These products are non-humic and generally biologically labile. Several studies have discussed the autochthonous generation of humic material through secondary pathways. Wetzel (2001) states that fulvic acid is a common autochthonous humic material. McKnight *et al.* (1994) examined two Antarctic ponds with watersheds containing no higher plants. Both ponds contained significant algal populations, and the organic matter in both ponds contained between 16 and 21% aromatic or olefinic carbon atoms. Such products contain relatively large amounts of nitrogen (McKnight *et al.* 1994 and Croué *et al.* 1996).

Steinberg and Muenster (1985) suggest that humics can be produced when organic matter molecules are brought within binding distance. Potential opportunities occur within decaying organisms, within organic matter particles or colloids, within the digestive systems of filterfeeders, or from aggregating surface active molecules. Tranvik and Kokalj (1998) propose that interactions between algal DOC and dissolved humic material (DHM) may increase the recalcitrance of the latter. A solution of carbon labeled algal extract was dissolved in artificial lakewater and irradiated with UV light in the presence and absence of dissolved humic material. It was found that the photoproducts of algal DOC were less bioavailable when irradiated with the humic material.

2.5 Dissolved Organic Matter Decay

Pathways of organic matter degradation in natural systems include microbial consumption and solar photolysis. Biologically labile organic matter, which includes the relatively low molecular weight, non-humic substances that originate from many biological processes and are cycled rapidly within a system, is (by definition) more easily degraded by microbial processes. Humic and fulvic acids may be more subject to photolysis; humic substances have long been thought to be microbially stable (Wetzel 1975, cited by Kouassi and Zika 1992) although it is possible that certain easily degradable compounds (glucose, lactate) may stimulate microbial degradation of recalcitrant compounds (Steinberg and Muenster 1985).

2.5.1 Biological Decay

Table 2.2 presents literature values for biological organic matter decay rates and suggested ratios between decay rates and other parameters as found in the literature by Roberts (2003) and Garvey (2000). The acronyms included are those implemented by CE QUAL W2 and are described in detail in Table 3.4. CE QUAL W2 models biologically labile DOM and recalcitrant (refractory) DOM as separate parameters with separate decay rates.

It is notable that decay rates span four orders of magnitude depending on the nature of the material. The labile DOM decay rates noted are generally between 0.1 and 2.3 day^{-1}, and refractory decay rates are generally two to three orders of magnitude lower. A notable exception is the modeling study of Quabbin Reservoir (Roberts 2003). The calibrated labile DOM decay rate of 0.003 day^{-1} is more consistent with refractory decay rates from the other studies, whereas the refractory decay rate, 0.0003 day^{-1}, is even lower. In a large system with long detention times, much organic matter would be mineralized. Especially recalcitrant DOM would bias the decay rate toward low values.

Numerous studies have examined microbial decay rates of organic matter. In studying the decomposition of algal detritus in Lake Loosdrecht, The Netherlands, Otten *et al.* (1992) subjected algal detritus to 3 weeks of dark, aerobic decay. DOC remaining after this period was deemed refractory for the purposes of the study. Assuming first order decay, and that 95% degradation of labile DOM occurred, the corresponding decay rate is

0.14 day^{-1}. Similarly, Wetzel and Manny (1971) discussed the properties of organic carbon in leaf litter leachate, defining a labile component with a half life of 2 days (first order decay rate k = 0.346 day^{-1}) and a refractory component with an 80 day half life (k = 0.0087 day^{-1}) (Wetzel and Manny, 1971).

Table 2.2 Literature Values for Microbial Degradation of Organic Carbon (condensed from Roberts, 2003)

Parameter	Value or Range	Unit	Source
LDOMDK	0.1	day^{-1}	Garvey (2000)
RDOMDK - LDOMDK	0.0008 - 2.3	day^{-1}	Roberts (2003) (literature)
LDOMDK	0.11 - 0.64	day^{-1}	Cole and Buchak (1995)
LDOMDK	0.003	day^{-1}	Roberts (2003)
RDOMDK	0.0003	day^{-1}	Roberts (2003)
DOC Consumption Coefficient of Bacteria	0.0178	day^{-1}	Cole *et al.* (2002)
LDOMDK:RDOMDK	~100	-	Cole and Buchak (1995)
LRDK:RDOMDK	~1	-	Cole and Buchak (1995)
Aromatic Compounds:Total DOM	0.8	-	Wetzel *et al.* (1995)

Bryan (2005) conducted a leaf leachate biodegradation experiment. Leaves from red maple (*Acer rubrum*), white oak (*Quercus Alba*), and white pine (*Pinus Strobus*) were collected and leached for eight days. The resulting solution was then biodegraded in a dark incubator at 22°C for 81 days. The solution was analyzed periodically for UV254 and TOC. DBPFP was measured at the end of the experiment. Table 2.3 summarizes some of the results and presents first order decay rates for litter from each tree species for the first 5 days, the next 76 days, and the overall period.

Table 2.3 Results of Leaf Litter Leachate Biodegradation Study (Bryan 2005)

	Maple		Oak		Pine	
Period	% TOC_i lost	k, day^{-1}	% TOC_i lost	k,day^{-1}	% TOC_i lost	k, day^{-1}
Day 1-5	25	0.0575	34	0.0831	50	0.1386
Day 6-81	26	0.0040	31	0.0049	1	0.0001
Overall	51	0.0088	65	0.0130	51	0.0088

Results indicate rapid decay during the first five days, ($k = 0.06$ to $0.14\ day^{-1}$) followed by slower decay during the next 76 days. Total decay for days 6 through 81 was similar to decay for days 1-5 for the maple and oak derived TOC, although 50% of TOC from the pine leachate was lost in the first 5 days, while only 2% of the total initial TOC was lost during the next 76 days. Results indicate that a single first order decay rate is not adequate to describe biodegradation of all the leaf-released compounds. It is notable that the decay rates resulting from the study are similar to those found in the literature.

2.5.2 Photolysis

Light degradation of organic matter is a significant environmental process. Allard *et al.* (1994) noted that weekly UV and TOC sampling of the Kalix River in Northern Sweden suggested a larger humic fraction in the winter than in summer, a relationship attributed to photolysis. The impact of irradiance on DOM quantity and composition has been the subject of much study in recent years. Moran and Zepp (1997) suggest that photolysis produces biologically available compounds including low molecular weight organic compounds (carbonyl compounds, molecular weight <200), carbon gases including CO and CO_2, bleached organic matter, and nitrogen- and phosphorus-rich compounds including NH^{4+} and PO_4^{3-}. Bertilsson and Tranvik (2000) irradiated water from 38 Swedish lakes with an artificial UV light source. With 8 hours of mild dose, production of dissolved inorganic carbon (DIC) and carboxylic acid corresponded to approximately 4.7% of DOC, indicating that photolysis is a significant process in DOC degradation.

Irradiance wavelength distribution is an important factor in photolysis. Moran and Zepp (1997) plotted literature data for irradiance quantum yield for production of CO and H_2O_2 (quantum yield is the fraction of absorbed light that results in a photoreaction) versus irradiance wavelength for diverse water sources (ocean, lake, river, and wetland). Quantum yield generally decreased with increasing wavelength, ranging from 3E-4 to 1E-5 (unitless) between 300 and 450 nm wavelengths for CO and from 2E-3 to 3E-5 between 300 to 410 nm for H_2O_2. The authors also conclude that the maximum rate of near surface photoproduct formation occurs with 330 nm wavelength light. However, with increasing depth, the impact of longer wavelength light becomes increasingly important. Koussi and Zika (1992) state that irradiance of a certain wavelength tends to reduce the absorbance property of a sample at that wavelength (i.e. UV254 absorbance is most effectively decayed with UV254 light).

Several studies have examined the rate of photolysis of organic matter. Koussi and Zika (1992) studied marine humic substances that were isolated from samples of Gulf of Mexico water. The authors assembled a global model from their findings, proposing that the change in UV absorbance of humic material is a photodependant, first order process, at rates ranging between 0.001 and 0.017 $hour^{-1}$, dependent on latitude, cloud cover, season, and depth.

In a similar study, Reche *et al.* (1999) examined photolysis of water from 36 lakes in Wisconsin, Michigan, New York, and Connecticut, characterized by widely ranging color, DOC levels, total phosphorus, trophic state, and hardness. The authors were primarily interested in change in sample absorbance properties. Water was exposed to sunlight in borosilicate bottles. The authors determined first order, cumulative sunlight dose-dependent photobleaching rate coefficients to vary from 7E-4 to 43.9E-4 $(E\ m^{-2})^{-1}$. Examination of the variability between lakes indicated that variations in acid neutralizing capacity (ANC) correlated best with varying rate coefficient, indicating that differences in ionic conditions may influence humic and fulvic acid configuration. It is also important to note that the borosilicate glass used proved to be opaque to UV-B irradiance (irradiance below 320 nm). In a study of humic water from Lake Savojarvi in Finland, Backlund (1992) irradiated samples with UV light only (70 cal/cm^2-day), and with a combination of UV light and H_2O_2. After 60 minutes, 25% reduction in UV254 absorbance was observed with UV irradiation only, while after 30 minutes, 96% reduction was observed with UV irradiation and H_2O_2. Examination of the molecular size distribution showed that large DOM molecules were degraded to smaller fragments during irradiation.

The current photolysis framework included in CE QUAL W2 (presented in Section 3.1) was developed by Wolfram (1996) as a modification to the coliform bacteria subroutine of the program. The original subroutine predicted coliform loss as a first order process, with rate of decay dependent on temperature. This method proved inadequate, and the model was modified to include a light-induced decay rate that varied with depth within the water column. The same light induced decay subroutine was implemented to predict decay of UV254 absorbance with little modification (Roberts, 2003). The temperature dependent UV254 decay rate at 20°C was set equal to the calibrated refractory DOM decay rate (due to the recalcitrant nature of the materials for which UV254 is a surrogate) and a value for α, the photolysis coefficient relating light induced decay to irradiance, was determined through calibration. These values were 0.0003 day^{-1} and 2.6E-6 cm^2/cal, respectively.

The impact of photoproducts on microbial growth is the subject of several studies. While some photoproducts of DOM photolysis are known, and although photolysis products often lead to increased bacterial growth, the mechanisms are not understood. Lindell *et al.* (1995) exposed lakewater with 12 mg/L DOC to simulated sunlight for 0 to 100 hours. Bacteria were then added and the mixture incubated. The lakewater exposed to sunlight increased bacterial numbers by 65% and bacterial volume by 360% with increasing UV radiation. Wetzel (1995) exposed DOM from aquatic plants to natural and artificial UV radiation. Few changes were noted in the DOM pool before and after photolysis; however, the photolyzed substances were more easily metabolized by the bacteria.

A study using filter sterilized lake water demonstrated that cycling periods of sunlight and microbial activity resulted in more complete degradation of DOM and humic substances; photolysis and bacterial decay resulted in three times the decay observed in samples with microbial degradation only (Miller and Moran, 1997). A similar study with water from Kolbudzkie and Straszyn Lakes in Poland (Grzybowski 2002) showed increases in bacterial respiration of 17 to 54% after exposure of filter-sterilized samples to daylong sunlight. There was poor correlation between change in light absorption characteristics and the increases in respiration. It is notable that the release of orthophosphorus from fulvic acid-phosphorus associations has been observed (Steinberg and Muenster 1985)

Photolysis of DOM in the environment may accelerate the decay of other organic compounds. Chin *et al.* (2004) measured photolysis of bisphenol A (BPA), an organic compound used in several industrial applications. BPA is known to be biologically labile, with a half life of days to weeks. Photolysis of BPA alone is slower than photolysis of BPA in the presence of DOM, although the authors could not correlate the photoreaction to a structural component of DOM. The result indicates that photolysis of BPA could be as significant as biodegradation in the natural environment.

Some studies have lead to contradictory results. Tranvik and Kokalj (1998) noted that photolysis of algal products is slower in the presence of dissolved humic material, possibly due to the interaction between photoproducts. In a separate study, additional UV exposure after an initial 6 hour irradiance period could lead to reduced bacterial utilization of the photoproducts (Wetzel 2000).

In summary, organic matter is degraded by both microbial action and photolysis. Microbial decay rates span up to five orders of magnitude depending on the source and nature of the organic matter. Photolysis is also an important process in reducing the humic fraction of DOM and making recalcitrant organic matter more biologically available, although it also seems to play a role in making biologically labile material more refractory. It is likely that photoproducts can enhance degradation of other compounds that are not as photosensitive.

2.5.3 Detritus Decay and Settling

Detritus (also known as particulate organic matter (POM)), represents nonliving particulate organic carbon (POC) within the water column (as defined in CE QUAL W2). Natural detritus removal mechanisms include decay and sedimentation. Particulate organic carbon is often operationally defined as organic carbon that does not pass a 0.45 μm pore size filter. This rejected material includes colloids (decomposition in days or weeks) to leaf litter (decomposition in weeks to months) to woody debris (decomposition in years) (Wetzel 1983). Resulting from the variability in size and composition, variability in detritus decay rate and settling velocity is similar to the variability of DOM decay rates.

Table 2.4 presents literature values for detritus decay rates as reported by Roberts (2003). Values range from 0 to 0.4 day^{-1} with no consistent cluster. Most notable is the three fraction decay rate scheme implemented by Berner (1980) and Westrich and Berner (1984). Three decay rates allow the user to partition similarly decaying material into separate compartments. CDM (1995) recommended that 0.007 day^{-1} be implemented for a detritus decay rate for Wachusett Reservoir, a value used by Roberts (2003) for Quabbin as well. POC levels are low within Wachusett Reservoir (Worden 2004); the system should be insensitive to this parameter. Averaged TOC and DOC data from Cosgrove withdrawal for 2000 suggest that POC levels are insignificant (averaged TOC and DOC for this period was 2.51 and 2.49 mg/L, respectively).

Table 2.4 Literature Values for Maximum Detritus Decay Rate (Roberts 2003)

Parameter	Value or Range	Unit	Source
Three fraction Detritus Decay	0, 0.0018, 0.035	day^{-1}	Berner (1980), Westrich and Berner (1984)
Detritus Decay Rate	0.001 - 0.01	$day^{-1}\,°C^{-1}$	Canale (1976)
Detritus Decay Rate	0.001	day^{-1}	Chen and Orlob ((1975)
Detritus Decay @ 10°C	0.1 - 0.4		Jorgensen (1976)
Detritus Decay Rate	0.4	day^{-1}	O'Connor *et al.* (1973)
Detritus Decay @ 20°C	0.001 - 0.2	day^{-1}	Schnoor (1996)
Temperature coefficient, Q	1.072	-	Jorgensen (1976)

Literature values for detrital settling velocities are similarly variable. Table 2.5 presents literature values reported in the literature (Roberts 2003). Rates range between 0.001 and 0.66 m/day. Most of the values are near the upper end of this range. Also notable are the results of a sediment trap study of Cannonsville Reservoir, New York. POC sedimentation rates of 1.14, 0.63, and 0.32 m/day were reported in the West Branch Delaware River at the reservoir inlet, the transition zone between riverine and lacustrine environments, and the lacustrine zone, respectively (Auer and Forrer 1998). Pastres and Ciavatta (2005) report a first order detrital settling loss rate of 0.384 day^{-1}.

Table 2.5 Literature Values for Settling Rate (Roberts 2003)

Parameter	Value or Range	Unit	Source
Maximum Detritus Settling Rate	0.66	m/day	Baines and Pace (1994)
Detritus Settling Rate	0.35	m/day	CDM (1995)
Detritus Settling Rate	0.2	m/day	Chapra (1997)
Detritus Settling Rate	0.25	m/day	Chapra and Reckhow (1983)
Detritus Settling Rate	0.002 and 0.0019	m/day	Jorgensen (1976)
Detritus Settling Rate	0.001 - 0.3	m/day	Schnoor (1996)

2.6 Algal Modeling

Numerous studies have modeled algae with variable success. Different approaches are frequently implemented. CDM (1995), Garvey (2000) and Roberts (2003) conducted extensive literature reviews for algal parameter values to be implemented in studies of the MWRA/DCR reservoir system. Wachusett and Quabbin Reservoirs are somewhat similar in trophic state and phytoplankton ecology; both are dominated by diatoms. CDM (1995) investigated parameter values for diverse phytoplankton species and determined average values for implementation by weighting the relative abundance of genera. Analysis for Wachusett Reservoir by Worden (2003) suggests that phytoplankton dynamics are fairly stable over long periods (1995 through 2002 was studied) although there are shorter term cyclic trends that are of some importance. For example, in years when spring diatom densities are relatively low (below ~1000 ASU/mL) summer densities of the chrysophyte *Syurna,* a critical taste and odor organism, become larger (above 20 ASU/mL).

2.6.1 Algal Growth Rate

Table 2.6 presents literature values for first order maximum algal growth rates as found by Roberts (2003). The values range over two orders of magnitude, from 0.09 through 3.9 day^{-1}. Literature values found by CDM (1995) ranged from 0.5 to 3.5 day^{-1}. Values most frequently reported are towards the center of this range, between 1 and 2.5 day^{-1}.

Table 2.6 Literature Values for Maximum Algal Growth Rate (Roberts 2003)

Value or Range	Unit	Source
0.5 - 3.5	day^{-1}	CDM (1995) (literature)
2.3 and 2.53	day^{-1}	Jorgensen, in Orlob (1983)
1.6 - 2.1	day^{-1}	Canale *et al.* (1976)
2.0	day^{-1}	Chapra and Reckhow (1983)
1.5 - 2.5	day^{-1}	Thomann and Mueller (1987)
0.09 - 0.78	day^{-1}	Sterner *et al.* (1995)
1.0 - 2.0	day^{-1}	Schnoor (1996)
1.8 - 3.9	day^{-1}	O'Connor *et al.* (1973)
0.84 - 2.0	day^{-1}	Garvey (2000)

Algal parameter values in eutrophic Cannonsville Reservoir, New York were determined through a series of light bottle/dark bottle experiments at varied nutrient levels and light intensities as presented in Auer and Forrer (1998). While the dominant phytoplankton species differ between Cannonsville and Wachusett, diatoms are the dominant species in both ecologies. The maximum algal growth rate determined under phosphorus limiting conditions as seen in Cannonsville in the early summer was 1.67 +/- 0.6 day^{-1}. A second series of experiments were conducted under nitrogen limiting conditions (as tends to occur in late summer in Cannonsville) resulting in a maximum algal growth rate of 1.08 +/- 0.6 day^{-1}. The average measured ambient specific growth rates were between 66 and 78% of these values, respectively.

In a study of the lagoon of Venice, Italy, Pastres and Ciavatta (2005) describe the sensitivity of a 3-D water quality model to varied parameter values. The model was calibrated with a maximum algal growth rate of 0.12 hr^{-1} (2.9 day^{-1}). A separate set of input parameter values, including a nominal growth rate of 0.0965 hr^{-1} (2.3 day^{-1}) was implemented in a Monte Carlo random sampling scheme for sensitivity analysis.

Garvey (2000) did not implement CE QUAL W2, but instead used the algal modeling framework to estimate algal fluxes. Using the literature values presented in Table 2.6 (see Garvey 2000), predicted algal productivity ranged from 22 to 199 mg C/m^2-day (at maximum growth rate = 0.84 day^{-1}) and 53 to 472 mg C/m^2-day (at maximum growth rate = 2.0 day^{-1}), similar to ranges reported for other oligotrophic systems, and well within ranges given for many systems (Wetzel 1983). However, in a CE QUAL W2 model of Quabbin Reservoir, Roberts (2003) predicted algae in terms of concentration at a withdrawal and at sampling profile stations. It was necessary to increase the maximum growth rate to 3.5 day^{-1} (still within the range of values reported in the literature) to achieve predicted algal densities similar to measured concentrations.

2.6.2 Algal Half Saturation Coefficient

CE QUAL W2 implements Monod kinetics to model the relationship between nutrient (or substrate) concentration and the algal growth rate. The algal growth rate approaches the maximum growth rate as the nutrient concentration becomes large. A nutrient half saturation coefficient defines the shape of the curve by specifying the nutrient concentration at which the growth rate is half the maximum growth rate. CE QUAL W2 treats nitrogen and phosphorus as potentially limiting nutrients, each with a half saturation coefficient and each capable of limiting growth. Table 2.7 presents half saturation coefficients for nitrogen and phosphorus from the literature as reported by Roberts (2003).

Table 2.7 Literature Values for Nutrient Half Saturation Coefficients (Roberts 2003)

Value or Range	Unit	Nutrient	Source
0.001 - 0.008	mg/L	Phosphorus	CDM (1995)
0.002	mg/L	Phosphorus	Chapra (1997)
0.006 - 0.025	mg/L	Phosphorus	Schnoor (1996)
0.001 to 0.005	mg/L	Phosphorus	Thomann and Mueller (1987)
up to 0.15	mg/L	Phosphorus	Lehman *et al.* (1975)
0.062	mg/L	Nitrogen	CDM (1995)
0.015	mg/L	Nitrogen	Chapra (1997)
0.001 - 0.020	mg/L	Ammonia	Schnoor (1996)
0.01 to 0.02	mg/L	Nitrogen	Thomann and Mueller (1987)
up to 0.4	mg/L	Nitrogen	Chen and Orlob (1975)

Values for both nitrogen and phosphorus are similar. Phosphorus values range from 0.001 to 0.15 mg/L, although most values are less than 0.01 mg/L. Nitrogen half saturation coefficients range from 0.001 (as ammonia) to 0.4 mg/L. Cannonsville Reservoir light bottle/dark bottle experiments yielded a phosphorus half saturation coefficient of 0.0005 mg/L (Auer and Forrer 1998). Calibration for the Venice Lagoon yielded half saturation coefficients of 0.01 and 0.05 mg/L for phosphorus and nitrogen, respectively.

Limiting of algal growth by light is determined with a user defined saturation light intensity at the maximum algal growth rate.Cole and Buchak (1995) report saturation light intensities for algal growth from 10 to 86 W/m^2. CDM (1995) selected 50 W/m^2 from the literature, a value that was subsequently used by Garvey (2000) and Roberts (2003) for Quabbin Reservoir. Auer and Forrer (1998) report values ranging from 16 to 90 $\mu E/m^2$-s (approximately 5.2 to 29 W/m^2). Chapra (1997) reports optimal light levels for algal growth to be approximately 100 to 400 Langley/day (48.5 to 194 W/m^2). Laws and Chalup (1990) used a value of 25.9 W/m^2.

2.6.3 Algal Respiration Rate

Table 2.8 presents values for first order maximum algal respiration rates reported in a literature review by Roberts (2003).

Table 2.8 Literature Values for Maximum Algal Respiration Rate (Roberts 2003)

Value or Range	Unit	Source
30-40	% of AGROW	Wetzel (1983)
0.05 - 0.3	day^{-1}	CDM (1995)
0.286 - 0.6	day^{-1}	Biswas (1981)
0.05 - 0.25	day^{-1}	Thomann and Mueller (1987)
0.025	day^{-1}	Chapra and Reckhow (1983)
0.088 and 0.13	day^{-1}	Jorgensen, in Orlob (1983)
0.015	day^{-1}	Orlob (1983)
variable	algal density dependent	Jorgensen (1976)
0.06	day^{-1}	Gargas *et al.* (1976)
0.03	day^{-1}	Canale *et al.* (1976)

The maximum respiration rates range between 0.015 and 0.6 day^{-1}, a broader range than found for the maximum growth rate. The most commonly reported values are less than 0.3 day^{-1} and often less than 0.1 day^{-1}. CDM (1995) selected a value of 0.1 day^{-1} from a literature search. Garvey (2000) also implemented 0.1 day^{-1} for predicting algal production with reasonable results. Roberts (2003) began calibration with 0.1 day^{-1} but 0.2 day^{-1} resulted in improved predictions. A recent study on the lagoon of Venice, Italy (Pastres and Ciavatta 2005) determined 0.12 day^{-1} to be appropriate.

The series of light bottle/dark bottle experiments conducted in Cannonsville Reservoir (Auer and Forrer 1998) under variable nutrient and light conditions yielded maximum algal respiration rates of 0.3 ± 0.2 day^{-1}. Measured values ranged from 0.08 to 1.1 day^{-1}, consistent with the variability reported in literature values. It is possible that respiration varies with algal activity, suggesting that a two step model, such as that developed by

Biswas (1981), or a model implementing a variable respiration rate, as in Jorgensen (1976), might be appropriate.

2.6.4 Algal Excretion Rate

Algal excretion is often not included in models, although algae are known to excrete organic carbon, nitrogen, and phosphorus (Jorgensen, 1979). CE QUAL W2 does include this process; the excretion of labile DOM is included as a first order release rate. Table 2.9 presents maximum algal excretion rate values from literature as reported by Roberts (2003).

Table 2.9 Literature Values for Maximum Algal Excretion Rate (Roberts 2003)

Value or Range	Unit	Source
0.02	day^{-1}	CDM (1995)
0.012	day^{-1}	Garvey (2000)
<20% of productivity	-	Wetzel (1983)

Cole and Buchak (1995) report a range of excretion rates from 0.014 day^{-1} to 0.044 day^{-1} for various algal species as determined by Nalewajko (1966). It is notable that only two of the six genera studied are known to exist in Wachusett Reservoir, although both are Chlorophytes and neither is dominant. Based on estimates of productivity in Quabbin Reservoir, Garvey (2000) assumed the excretion rate to be 20% of productivity after Wetzel (1983) (0.012 day^{-1} was used). Resulting estimated extracellular product release was between 3 and 60 mg C/m^2-day, on the high end of what was expected. Roberts (2003) implemented a value of 0.012 day^{-1} Considering that algae comprises no more than 20% of TOC in Wachusett Reservoir (see Section 3.3) model results are expected to be relatively insensitive to this parameter.

2.6.5 Algal Mortality and Settling

Some models distinguish between algal mortality and settling, while others do not. It can be difficult to distinguish between causes for algal loss, which include predatory loss, settling, and parasitic or pathogenic mortality. Table 2.10 presents algal mortality rates from the literature as reported by Roberts (2003). The mortality rates range between 0.01 and 0.9 day^{-1} , although the majority of the values are between 0.01 and 0.1. Cole and

Buchak (1995) suggest that the maximum mortality rate represents both predatory and nonpredatory losses, and should be less than 10% of the maximum growth rate. Garvey (2000) suggests that an appropriate mortality rate for Quabbin is about 1% of the maximum growth rate.

Table 2.10 Literature Values for Maximum Algal Mortality Rate (Roberts 2003)

Value or Range	Unit	Source
0.8	day^{-1}	Jassby and Goldman (1974)
0.09	day^{-1}	Jorgensen (1976)
0.05 to 0.25	day^{-1}	Schnoor (1996)
0.07, 0.26 and 0.87	day^{-1}	Sterner *et al.* (1995)
0.03	day^{-1}	CDM (1995)
0.01 - 0.03	day^{-1}	Cole and Buchak (1995)
<10% of AGROW	-	Cole and Buchak (1995)
~1% of AGROW	-	Garvey (2000)

In a study of the lagoon surrounding Venice, Italy, the suggested algal mortality rate is 0.12 day^{-1} (Pastres and Ciavatta 2005). Roberts (2003) used AMORT = 0.03 day^{-1} the same mortality rate as determined by CDM (1995) from literature review and as used by Garvey (2003).

Algal settling velocities are dependent on many factors including cell properties, fluid properties, and fluid turbulence (Orlob 1983, in Roberts 2003). Table 2.11 presents literature values reported by Roberts (2003) for algal settling rates.

Table 2.11 Literature Values for Maximum Algal Settling Rate (Roberts 2003)

Value or Range	Unit	Source
0.29	day^{-1}	CDM (1995)
0 - 1	day^{-1}	Canale (1976)
0 - 2 (0.2 typical)	day^{-1}	Schnoor (1996)
0.2	day^{-1}	Chapra (1997)
0.21 to 0.22	day^{-1}	Baines and Pace (1994)

The predicted settling flux in CE QUAL W2 is independent of fluid velocity, turbulence and direction, and depends on settling velocity. Other models have included settling rates that depend on fluid turbulence and algal physiological state. Reported settling velocities range between zero and 2 m/day (as reviewed by Schnoor 1996). For Quabbin Reservoir, Roberts (2003) implemented 0.29 m/day as proposed by CDM through literature review. Pastres and Ciavatta (2005) modeled settling via first order decay, implementing a rate of 0.0384 day^{-1} for the Venice Lagoon.

In a comprehensive study of settling in Cannonsville Reservoir, New York, Effler and Brooks (1998) deployed a series of sediment traps along the length of the reservoir. The relatively long, narrow reservoir receives most of its inflow from the West Branch Delaware River in the northeast. From there, the water flows generally southwest towards the dam. The consistent flow allows the reservoir to be divided into riverine, lacustrine, and transition zones. Sediment traps were placed at four stations in the lacustrine zone and at one station each in the riverine and transition zone. Total suspended solids (TSS), POC, particulate phosphorus (PP) and total chlorophyll settling velocities were calculated at each location. Of these measurements, total chlorophyll is probably the most representative of algal settling, although this measurement would likely include living and nonliving material. Total chlorophyll settling velocities ranged from 0.29 m/day in the riverine zone to 0.17 m/day in the lacustrine zone.

2.7 Nutrient Modeling

CE QUAL W2 includes a number of processes that affect nutrients, including nitrification, denitrification, release of nitrogen and phosphorus from the sediments, and the adsorption of phosphorus to inorganic suspended solids. Most of these processes are ignored in this research. Inorganic suspended solids are not currently included (nor do they impact Wachusett Reservoir water quality). Denitrification and sediment release occur significantly only under anaerobic conditions, which have never been observed in Wachusett Reservoir (Worden 1994). Nitrification is of interest, however.

Table 2.12 presents literature values from Roberts (2003) for nitrification rates (ammonia decay rate). CDM (1995) selected 0.01 day^{-1} from the literature, but the value was changed to 0.03 day^{-1} following calibration. This value was implemented by Roberts (2003) as well.

Table 2.12 Literature Values for Ammonia Decay Rate (Roberts 2003)

Value or Range	Unit	Source
0.01 - 0.03	day^{-1}	CDM (1995)
0.03 - 0.2	day^{-1}	Orlob (1983)
0.008	$day^{-1}\,°C^{-1}$	Canale (1976)
0.1	day^{-1}	Chapra (1997)

Another important consideration is the quantity of phosphorus available to biological growth. Zhang *et al.* (2004) reports that the sediments of phosphorus limited Florida Bay contain 45% phosphorus bound to minerals, 24% refractory organic phosphorus, 19% reductant soluble inorganic phosphorus, and 8% readily exchangeable phosphorus. Auer *et al.* (1998) report that 25% of total phosphorus is bioavailable in Cannonsville Reservoir.

3. Model Selection and Development

Upon evaluation of the strengths and limitations of each impoundment modeling method described in Section 2.3, a 2-D arrangement was selected. In Wachusett Reservoir, longitudinal and vertical hydrodynamic and temperature gradients dominate lateral gradients. Thus, a laterally averaged model is most appropriate (Tobiason *et al.* 1996). CE QUAL W2 was selected since it provides the ability to model the water quality constituents of interest in this study (TOC components, nutrients, and UV254). Section 3 describes the attributes, data requirements, and parameter requirements for CE QUAL W2 as implemented in this study. Topics discussed include the modeled grid of Wachusett Reservoir, modeled constituent interactions, and data requirements.

3.1 Model Description

Modeling water quality constituents requires the successful implementation of a hydrodynamic and thermodynamic model. This section presents and describes the attributes of the CE QUAL W2 model used in this research that represent the physical properties of the reservoir.

3.1.1 Hydrodynamic Representation of the Reservoir

The Wachusett Reservoir modeling grid, (shown in Figure 3.1) established by CDM (1995) and revised by Joaquin (2001), consists of 63 segments in 5 branches, with 47 layers at maximum depth. Branch 1 represents the main body of the reservoir and includes segments 2 though 46. Branch 2, consisting of segments 49 though 51, represent the South Bay. The remaining branches represent physical features that would have been poorly represented by widening segments in Branch 1.

Other notable features of the modeling grid are the constriction at Segment 15 and the outflow geometry of Segments 45 and 46. Segment 15 represents the narrowing of the reservoir at the Route 12 bridge, separating the Thomas Basin (Segments 2-14) from the main basin. Before this constriction was added, the model predicted less warming of Quabbin transfer water than DCR measurements indicate, causing the Quabbin transfer density current to appear as an underflow rather than an interflow as data suggests (CDM 1995).

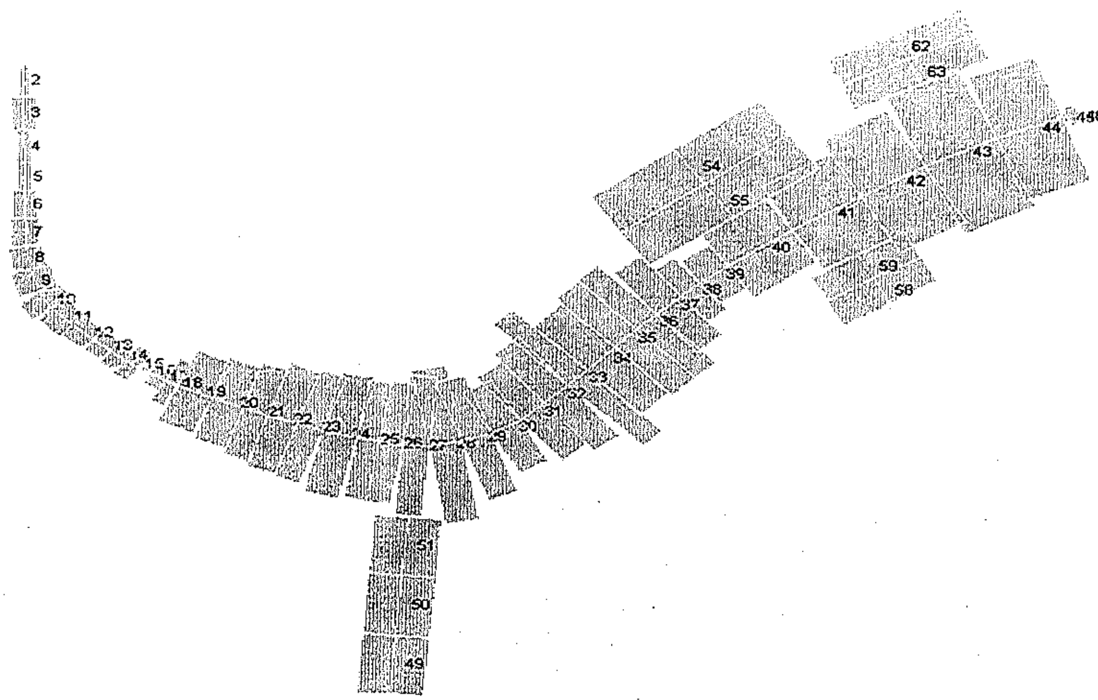

Figure 3.1 Plan view of Wachusett Reservoir modeling grid

Segments 45 and 46 were included to better represent the geometry created by segments of cofferdam that were never removed after construction of the Cosgrove intake structure. Segment 45 simulates the gap in the cofferdam that water must pass through to reach the intake, and Segment 46 represents the space that is enclosed by the intake structure and the remnants of the cofferdam. The presence of this feature causes warmer water from the upper layers of the reservoir to be withdrawn, despite the deep elevation of the withdrawal.

Figure 3.2 shows a profile view of the segments 42 through 46 as represented in the modeling grid. Layers are numbered from top to bottom. The topmost and bottommost layers (1 and 47, respectively) are boundary layers. The top 31 layers are 0.5 m in thickness. Layers 32 and 33 are 0.75 m in thickness, and layers 34 through 47 are 1.5 m in thickness.

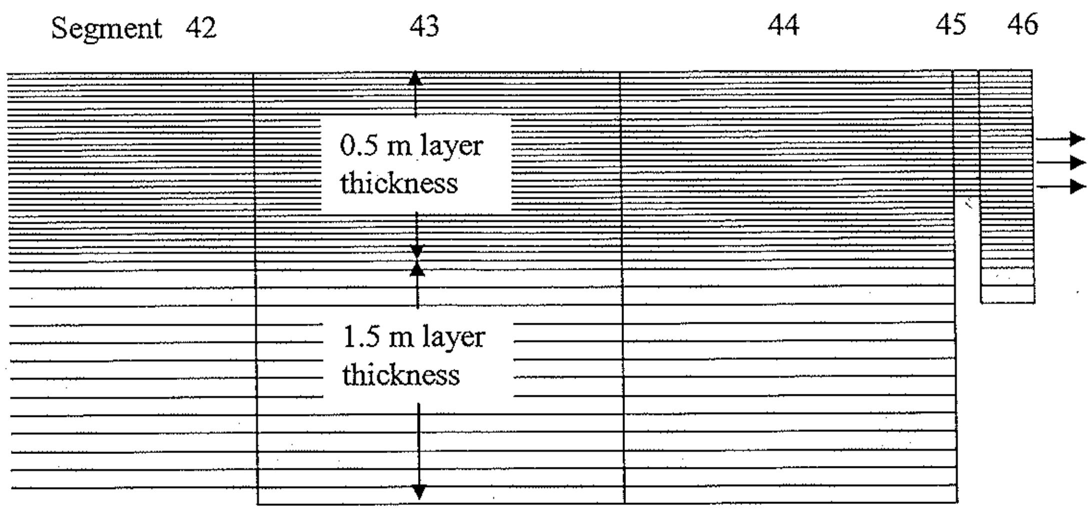

Figure 3.2 Profile View of (left to right) Segments 42 through 46.

3.1.2 Tributary Representation

The model grid described in Section 3.1.1 forms the reservoir into which the Wachusett watershed tributaries flow. The Wachusett model includes tributaries contributing ~1% or more of the annual water budget. Nine tributaries and the Quabbin Aqueduct, also modeled as a tributary, meet this criterion. Table 3.1 lists these tributaries and the model segments they enter.

Table 3.1 Tributaries modeled by CE QUAL W2

Tributary Name	Segment of Entry
Stillwater River	1
Waushacum Brook	3
Quinapoxet River	8
Quabbin Aqueduct	9
Malden Brook	10
West Boylston Brook	17
Gates Brook	20
French Brook	33
Malagasco Brook	49
Muddy Brook	50
Direct Runoff	branch 1

All of the tributaries except Stillwater and direct runoff enter the reservoir at a depth where the density of the tributary water corresponds to the density of the reservoir water. A branch inflow such as Stillwater is modeled as an upstream flow boundary condition, distributed over the depth of the water column. The tributaries that are too small to model individually are defined as direct runoff. Their estimated combined discharges are apportioned to each segment in branch 1 (based on segment surface area) as a non-point source. CE QUAL W2 defined non-point sources as distributed tributaries.

3.1.3 Withdrawal Representation

CE QUAL W2 provides several methods for modeling reservoir outlets, not all of which are used in this research. The Wachusett Reservoir model utilizes a selective withdrawal algorithm for representing Cosgrove aqueduct, and ordinary segment withdrawal to model other water losses except evaporation. The selective withdrawal algorithm calculates the layers from which water is withdrawn based on total outflow, structure type and elevation, and computed upstream density gradients (Cole and Buchak 1995).

The Cosgrove withdrawal structure includes two outlets structures, one at 343 ft. (104.5 m) elevation, and the other at 363 ft. (110.6 m) (CDM 1995). The lower outlet is typically used and is included in the model. This structure is modeled as two selective withdrawal line sinks at elevation 104.3 m (within layer 33). Layer 39 is set as the lowest layer of influence. Details may be found in the control file, presented in Appendix A.

Other withdrawals, comprising Nashua River discharges, North Dike seepage, Wachusett Aqueduct discharges, and withdrawals by towns, are modeled as the removal of water from a particular layer and segment. The locations of these usually minor withdrawals are presented in Table 3.2. Since North Dike seepage and town withdrawals occur within the same layer and segment, they are summed and represented as one withdrawal.

Table 3.2 Minor Withdrawal locations in the model.

Withdrawal	Layer	Segment
North Dike Seepage	11	44
Town Withdrawals	11	44
Nashua River	5	44
Wachusett Aqueduct	36	44

3.1.4 Data Requirements

The 11 inflows and 4 outflows included in the Wachusett Reservoir model, along with meteorological requirements, make this model relatively data intensive. Fortunately, CE QUAL W2 allows for data provided at varying temporal frequencies. The program recognizes the time interval between supplied data and interpolates as needed.

For every reservoir inflow, discharge and temperature files must be specified, along with a constituent file if water quality parameters are active. Flow data was provided to CE QUAL W2 on a daily average basis for all tributaries, direct runoff (m^3/s) and precipitation (m/s). A portion of an inflow file is presented as a sample in Appendix B. Inflow temperature data as measured by DCR is included on a weekly basis, except for precipitation where it is assumed to be the air dew point temperature at Worcester Airport, and is provided to the model on an hourly basis. A sample inflow temperature file is presented in Appendix C. Inflow constituent files include monthly or quarterly data as available. Cole and Buchak (1995) recommend that constituent data should be provided monthly or more frequently. Jordan and Likens (1975) offer that, although stream DOC data in their study varied from 1 to 7 mg/L (and up to four times these values during heavy rains), runoff ranged four orders of magnitude, thus accurate runoff measurements are more important than accurate DOC data when determining fluvial organic matter inputs.

All constituents for a specific inflow source are included in one file; one column is provided for each constituent present. CE QUAL W2 must be instructed, via the control file, to read a particular constituent in a particular inflow. Constituents are ordered by ascending constituent number, presented in Table 2.1. If data for some but not all constituents do not exist for a sampling time, the missing value may not be entered as blank or zero, as CE QUAL W2 will assume that the value is zero, and interpolate to and from zero for the interval surrounding that data point. A value must be generated by manual interpolation or averaging of existing data. A sample constituent inflow file is presented in Appendix D. CE QUAL W2 allows constituent files for different inflows to have inconsistent data frequencies

Outlet discharges must also be specified. Each branch withdrawal (i.e. Cosgrove) has a separate outflow file, while segment-layer withdrawals (i.e. Wachusett Aqueduct) are allotted a column in the withdrawal file. Temperature and constituent concentrations need not be specified for outflows; they are calculated by CE QUAL W2. However, it is possible to specify downstream constituent concentration and temperature boundary conditions. A sample withdrawal file is presented in Appendix E.

Meteorological data necessary for this model include air temperature, dew point temperature, wind speed, wind direction, cloud cover, and solar radiation. All of these data but solar radiation are available from the NOAA station at Worcester Airport, within 10 miles of the reservoir. Solar radiation is calculated by the preprocessing program W2MET (JEEAI, 1998).

Temperature and constituent initial conditions for the start of simulation must be established from available data. If in-reservoir gradients are small, spatially uniform initial conditions may be set in the control file. When gradients are present, vertical and longitudinal initial conditions can be manually set in the vertical profile (VPR) and longitudinal profile (LPR) initial condition files.

3.2 Hydrodynamic Modeling

3.2.1 Volume and Water Surface Elevation

The first step to prepare data for use with the Wachusett Reservoir model is to construct a water volume balance for the reservoir. This process allows for calibration and the pinpointing of instances when data might be inaccurate or missing. The balance is constructed by adding the total inflow volume and subtracting total outflow for a particular day (using daily-averaged data) to the total storage volume of the previous day, thus determining the new storage volume. Storage is then converted into a predicted WSE which is then compared to daily WSE as measured by DCR. The quadratic correlation of volume and WSE, based on a historic table of reservoir bathymetry that was assembled prior to inundation, is:

$$V_{m,t} = (0.040)z^2_{m,t} - (7.934)z_{m,t} + 398.3 \tag{3.1}$$

where $V_{m,t}$ is the "measured" storage volume ($x10^6$ m^3) of the reservoir at time t and $z_{m,t}$ is the measured water surface elevation (m) of the reservoir at time t. A more thorough discussion of this equation is presented in Ahlfeld *et al.* (2003a). Figure 3.3 presents a plot of the volume – WSE relationship for Wachusett Reservoir (Equation 3.1); the relationship is almost linear. The upper limit of the graph, 119.5 m, is the lower spillway elevation. The upper spillway elevation is 120.4 m. Spill elevation may be raised with stop logs. The lower elevation shown is well below typical lower water levels; between 1994 and 2002, WSE was below 118 m on only two occasions.

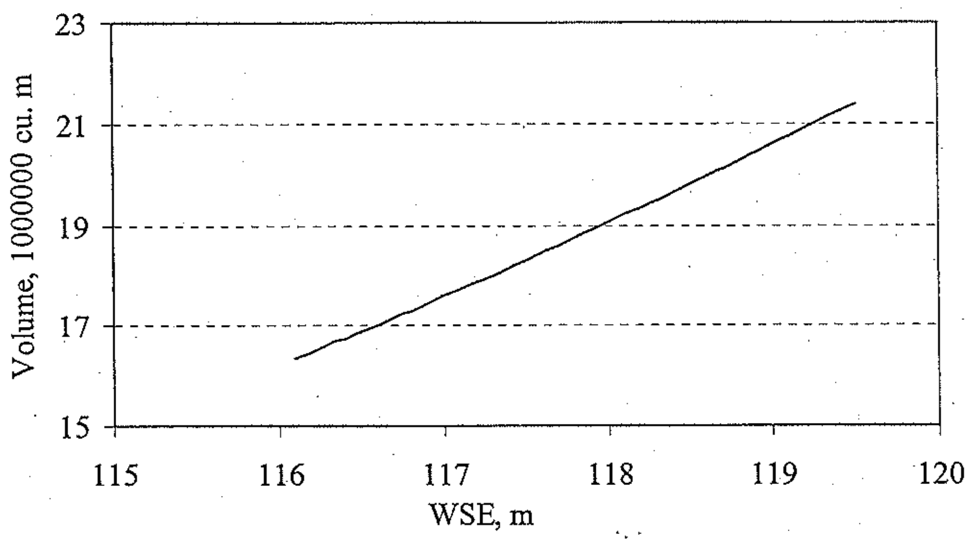

Figure 3.3 Relationship between volume and WSE for Wachusett Reservoir.

3.2.2 Preparation of Inflow Data

Discharge data is available for only a few of the tributaries. It is therefore necessary to estimate daily discharges for the others. To accomplish this task, the daily average discharge for a minor tributary on a certain day is assumed to be equal to the discharge of Stillwater River for that day, multiplied by the ratio of the area of the tributary watershed under consideration to the area of the Stillwater watershed. Stillwater discharges were chosen instead of Quinapoxet discharges, as the latter can be influenced by discharges from upstream reservoirs that are operated by the City of Worcester. A discussion of tributary discharges is presented in Tobiason *et al.* (2002).

Once discharges are estimated, discrepancies between the measured and modeled WSEs are minimized by multiplying each tributary inflow and the Quabbin Transfer by a calibration factor, determined separately for each calendar year. The calibration factors are determined by the SOLVER algorithm package within Microsoft Excel. This algorithm is used to minimize the sum of square errors between predicted volume and volume determined from measured WSE and Equation 3.1. A more detailed description of this process is found in Ahlfeld *et al.* (2003a).

Once the inflow data has been adjusted, a runoff coefficient for each tributary watershed may be determined. A dimensionless runoff coefficient for a tributary, C, during a specific calendar year is defined as:

$$C = (V/A)/d \quad (3.2)$$

where

V = total annual volume of discharge from a tributary (m^3/yr)

A = area of the watershed of that tributary (m^2)

d = total annual rainfall depth (m)

Chapter 4 presents a detailed analysis of the hydrology for each year studied in this research.

3.2.3 Preparation of Outflow Data

All outflow data, except evaporation, is measured or estimated by DCR and MWRA. The larger outflow discharges, such as the Cosgrove Aqueduct, are measured with reasonable accuracy via venturi meter. Small discharges, such as North Dike seepage, are likely to be inaccurate, but are of little consequence, and inaccuracies are accounted for in the inflow calibration factors. Therefore, outflow discharges are not adjusted.

3.2.3.1 Estimating Evaporation

Evaporation is a significant loss of water from the reservoir and is included in the calculations of CE QUAL W2. It must therefore be accounted for in the water budget calibration made external of the modeling software, so it is necessary to estimate it manually. Test runs of a new version of the CE QUAL W2 programming code, edited by UMass to model UV254 for the Quabbin Reservoir model (Roberts 2003), showed significant WSE discrepancies when run with inflow files for the CE QUAL W2 model of the 2001 calendar year, presented in Ahlfeld *et al.* (2003a). Investigation showed that the new code version was estimating more evaporation than the version typically used for Wachusett. The difference between the two codes could not be determined, so evaporation for Wachusett was reestimated based on work by Edinger *et al.* (1974) in the same manner as for Quabbin (Garvey 2000; Roberts 2003). Figure 3.4 presents monthly total evaporation predicted by the previously used method and by the newly implemented method.

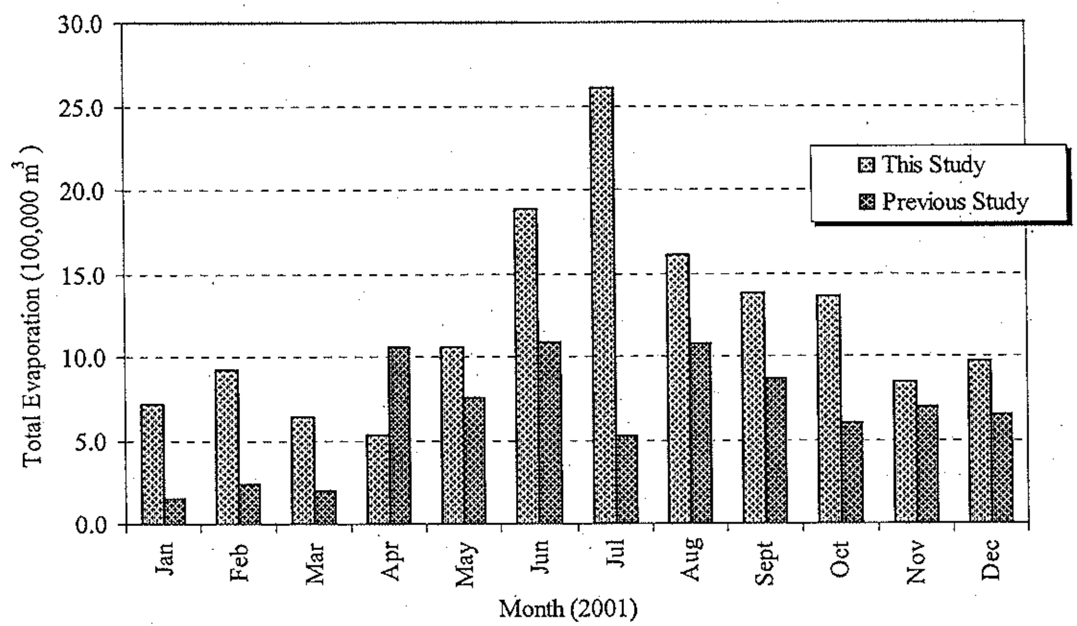

Figure 3.4 Comparison between monthly total evaporation for Wachusett Reservoir in 2001 by the evaporation estimate methods used in this study and in a previous study.

In the Edinger *et al.* (1974) method, the rate of evaporative water loss, Q_e (m/s), from a body of water may be summarized as

$$Q_e=\beta\,(T_s-T_d)f(W)/\,\rho\Delta_e \quad (3.3)$$

where

$$\beta=0.35+0.015(T_s+T_d)/2+0.0012[(T_s+T_d)/2]^2 \quad (3.4)$$

$$f(W)=9.2+0.46W^2 \quad (3.5)$$

and

T_s= water surface temperature (°C)

T_d= dew point temperature (°C)

ρ= density of water (1000 kg/m^3)

Δe= latent heat of evaporation (J/g @ 20°C)

W= wind speed at 7 m above water surface (m/s)

Refer to Garvey (2002) for details regarding the derivation.

The new method estimates more evaporation for every month in 2001 but April, peaking in the summer months with up to four times the evaporation as estimated in the colder seasons. On the other hand, the previously used method predicts fairly constant evaporation, except at the beginning of the year. Ahlfeld *et al.* (2003a) mention that evaporation was underestimated during the summer months, a statement which is also suggested by this analysis. If the current CE QUAL W2 code is to be used for Wachusett data that have been reported previously, it will be important to revise the water budget and inflow calibration factors to include the revised evaporation method.

3.2.3.2 Ice Cover

CE QUAL W2 includes an algorithm to calculate ice cover for the reservoir. This option was not implemented to avoid modeling and calibration complications. However, it is notable that not including ice cover may result in the overprediction of winter evaporation, solar heat transfer, and light-induced decay of constituents; each of these processes may inhibited by ice in the reservoir system.

3.2.4 Physical Model Coefficients

A summary of values for physical coefficients used in this study, along with values from CDM (1995), Joaquin (2001), and Roberts (2003), is presented in Table 3.3. The values for AX (the dispersion of momentum in the X direction), DX (dispersion of heat and constituents in X direction) and CHEZY (impact of bottom friction) are all model default values. The value for BETA (fraction of light reflected by the water surface) was determined by calibration (Joaquin 2001). Some parameters presented in Joaquin (2001) are not present in the version of CE QUAL W2 implemented in this study (wind function constant and quadratic terms, and a solar radiation multiplier) while Joaquin (2001) and CDM (1995) do not present values for some parameters in Table 3.3. This discrepancy indicates that a slightly different code version may have been used in those studies, possibly resulting in the evaporation discrepancy discussed in Section 3.2.3.1.

The combination of light extinction coefficients used by the program are difficult to separate from one another without data intended specifically for that purpose. Cole and Buchak (1995) present the following equation for estimating the net light extinction coefficient from secchi disk depth:

$$\gamma_{net}=1.11z_s^{-0.73} \qquad (3.6)$$

where z_s = secchi disk depth

Secchi disk depth measurements represent total light attenuation within the water column, including dissolved compounds and organic and inorganic particles.

Table 3.3 Physical Model Parameters

CE QUAL W2 Parameter	Description	CDM (1995)	Joaquin (2001)	Roberts (2003)	This Study
AX	Longitudinal Eddy Viscosity (m^2/s)	-	-	1.0	1.0
DX	Longitudinal Eddy Diffusivity (m^2/s)	-	-	1.0	1.0
CHEZY	Chezy Coefficient ($m^{0.5}/s$)	-	-	70	70
WSC	Wind Sheltering Coefficient	0.65, 0.85	0.65	0.70	0.65
BETA	Fraction of Solar Radiation lost at water surface	0.28	0.45	0.23	0.45
EXH2O	Light Extinction Coefficient for Water (m^{-1})	0.295	0.45	0.24	0.29
EXINOR	Light Extinction Coefficient for Inorganic Solids ($m^3/m\text{-}g$)	-	-	0.001	0
EXORG	Light Extinction Coefficient for Organic Solids ($m^3/m\text{-}g$)	-	-	0.001	0
CBHE	Coefficient of Bottom Heat Exchange (m^2/sec)	7.0E-07	7.0E-07	7.0E-07	7.0E-07
TSED	Sediment Temperature (°C)	9-11	10	12	10

In this case, γ_{net} can be included in CE QUAL W2 for EXH2O as a total light extinction coefficient for water. Alternatively, CE QUAL W2 includes an algorithm that can determine a variable net light extinction coefficient for the water column, where:

$$\gamma_{net} = EXH2O + EXINOR*\Phi_{iss} + EXORG*\Phi_{oss} \qquad (3.7)$$

and

Φ_{iss} = total concentration of inorganic suspended solids, mg/L

Φ_{oss} = total concentration of organic suspended solids, mg/L

where the suspended solids concentrations are predicted by the model. In this case, EXH2O is taken as background attenuation (i.e. with no suspended solids) to which attenuation due to suspended solids is added. In Wachusett Reservoir, suspended solids levels are generally low (turbidity is typically 0.1 NTU), and secchi disk data is abundant. Thus, γ_{net} was calculated from secchi disk data, and the contribution of suspended solids ignored by setting EXINOR and EXORG to zero.

Calculation of EXH2O was performed using 2003 secchi disk transparency data, shown in Figure 3.1. Data from 2003 was used as 2001 and 2002 data was not readily available.

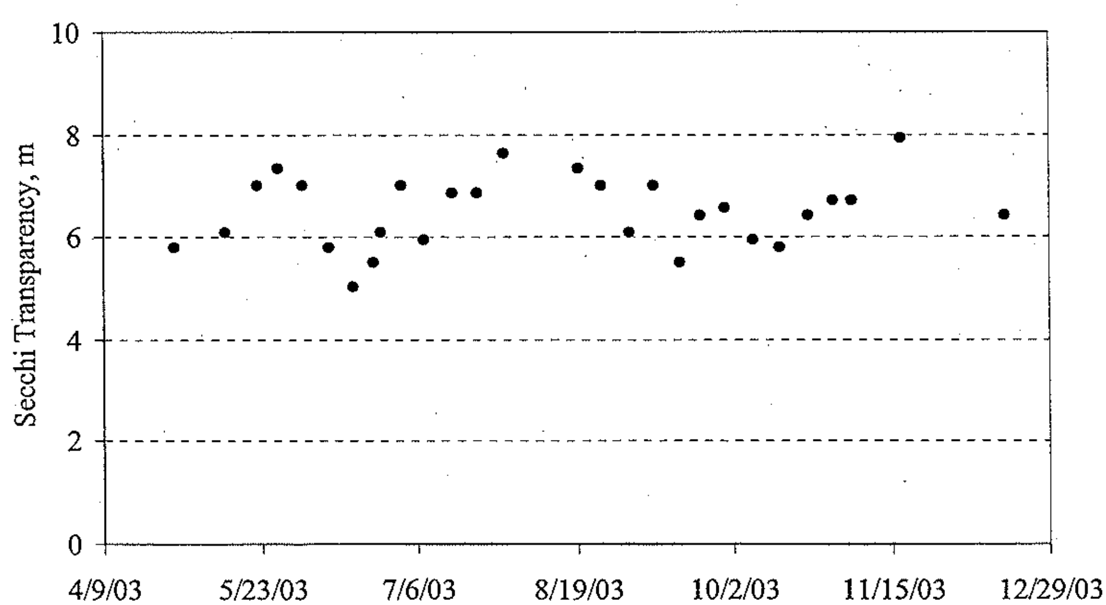

Figure 3.5 2003 secchi disk depth data for Wachusett Reservoir

During this period, the average secchi disk depth was 6.5 m with a standard deviation of 0.70 m, resulting in EXH2O equal to 0.28 m^{-1} (from equation 3.6). Back-calculating secchi disk depth from the previously used EXH2O value of 0.45 m^{-1} gives a z_s equal to 3.5 m, which does not reflect actual conditions. The value implemented by Roberts (2003), 0.24 m^{-1}, corresponds to a secchi transparency of 8.1 m. Quabbin Reservoir is less turbid and has low organic matter levels compared to Wachusett, resulting in greater transparency.

3.2.5 Temperature and Conductivity Calibration Methods

Upon completion of the water budget calibration, CE QUAL W2 input files are prepared and the model is run. Assuming adequate agreement between measured and modeled WSE, the calibration is confirmed by comparing measured in-reservoir profiles of temperature and conductivity to those predicted by the model. While WSE comparison confirms the bulk water balance, temperature and conductivity agreement confirms other significant hydrodynamic attributes. Temperature profile agreement indicates that supplied meteorological data is adequate and that heat exchange and advection coefficients are appropriate. Important to note are whether predicted heating, cooling, and stratification attributes agree with reservoir data. Conductivity profile comparison indicates that calibrated inflow proportions are appropriate (i.e. no accumulation or loss of conductivity over time) and that Quabbin interflow is predicted at the correct depth and magnitude. Temperature and conductivity profile disagreement has been noted in previous studies and resulted in grid modifications (See CDM 1995; Joaquin 2001).

The most frequent in-reservoir measurements of temperature and conductivity are recorded at stations within Thomas Basin (TB), South Basin (Station 3412), and North Basin (Station 3417). The Thomas Basin station is located towards the east shore in the north-south center of the basin, corresponding to Segment 10 in the model. The South Basin station location is near the Scar Hill Bluffs at the deepest point in the water column, where the Old Church by the Route 12 bridge is almost hidden by Davenport point, in Segment 33 in the model. The North Basin station is towards the center of the basin, where a boat depthfinder reads 28 to 30 meters (depending on WSE), corresponding to Segment 42 (refer to Figure 1.1).

CE QUAL W2 does not model conductivity as a constituent, but it does model total dissolved solids (TDS), a closely related parameter. The majority of TDS in a water are inorganic ions that contribute conductivity. Although the relationship between TDS and conductivity is dependent on the ions present, for modeling purposes it can be generalized that

$$TDS = 0.6 * Conductivity \quad (3.8)$$

where TDS is in mg/L and conductivity is in microsiemens per cm (μs/cm). This ratio was used in the CDM model (1995) and was recently confirmed for Wachusett using data from Malagasco Brook (Tobiason *et al.* 2000). TDS is modeled as a conservative constituent. Measurements of conductivity for inflows are converted to TDS for use in inflow constituent concentration files. The TDS profiles determined by CE QUAL W2 are converted into conductivity and compared to profiles measured by DCR.

3.3 Water Quality Modeling

Upon completion of the hydrodynamic calibration, water quality constituents other than TDS can be modeled. This section presents the algorithms used by CE QUAL W2 to model the constituents that were of interest in this study. These constituents include LDOM, RDOM, detritus, algae, ammonium-nitrogen, nitrate-nitrogen, orthophosphate, and UV254.

CE QUAL W2 calculates incremental changes in conditions (i.e. density, velocity, concentration) during timesteps of variable duration (timestep length is determined by an internal algorithm). In this section, the rate of change of constituent concentrations due to various processes is presented in differential equation form. These equations are discretized in the model for calculation of the change that occurs during each timestep. The constituents are presented in order of their constituent numbers (see Table 2.1).

3.3.1 UV254 Absorbance

Modeling UV254 with CE QUAL W2 is accomplished using modifications to the coliform bacteria modeling subroutine. The original subroutine modeled coliform with no autochthonous sources and only first-order, temperature dependent, decay. Tobiason *et al.* (1998) then modified the subroutine to include a settling term and a light-induced decay term to more accurately capture coliform dynamics. With little modification, Roberts (2003) was able to implement the improved subroutine to model UV254 instead of coliform, using both the temperature dependent and light induced decay terms and ignoring the settling term. The program computes UV254 as a concentration in g/m^3 while it is actually an absorbance of light. As there is no interaction of UV254 with other model constituents, no change in the internal units is necessary. In this subroutine, the time rate of change of UV254 absorbance is:

$$\frac{\partial UV254}{\partial t} = -K_{UV} UV254 \tag{3.9}$$

where

UV254 = the absorbance of UV light at 254 nm wavelength, cm^{-1}

and K_{UV} is the total first-order decay coefficient (day^{-1}). K_{UV} is calculated as

$$K_{UV} = K_{UV,temp} + K_{UV,light} \tag{3.10}$$

where $K_{UV,temp}$ is decay dependent on water temperature, sometimes called 'dark decay,' and $K_{UV,light}$ represents photolysis caused by sunlight.

The impact of temperature on UV254 decay is modeled using the following simplified form of the Arrhenius equation:

$$K_{UV,temp} = K_{UV,20}\theta^{T-20} \tag{3.11}$$

where

$K_{UV,20}$ = decay of UV254 absorbance at 20 °C, day^{-1}

θ = an empirical constant, based on reaction activation energy, temperature, and ideal gas constant.

The typical range for θ is 1.02 to 1.08 for temperatures and reactions in natural systems (Chapra 1997).

Light-induced decay is calculated using Beer's Law and a proportionality constant:

$$K_{UV,light} = \alpha I_0 e^{-\gamma z} \tag{3.12}$$

where

α = effect of irradiance on decay of UV254, cm^2/cal

I_0 = irradiance just below water surface, cal/cm^2-day (I_0 = I – BETA)

γ = irradiance extinction coefficient, m^{-1} (EXH2O used)

z = depth below water surface

3.3.2 Labile Dissolved Organic Matter (LDOM)

Labile DOM is input from sources and lost to outlets, gained from algal excretion and mortality, and has two decay pathways: one which produces ammonium, phosphorus and inorganic carbon while consuming dissolved oxygen, and one that produces RDOM. An additional pathway of light induced decay from refractory DOM to LDOM has also been included. Figure 3.6 provides a schematic of these processes. The time rate of change of labile DOM is:

$$\frac{\partial \Phi_{ldom}}{\partial t} = \left(K_{ae}\Phi_a + (1-P_{am})K_{am}\Phi_a\right) - \gamma_{om} A3\left(K_{ldom} + K_{lrdk}\right)\Phi_{ldom} + K_{om,light}\Phi_{rdom} \quad (3.13)$$

where

Φ_{ldom} = labile DOM concentration, g/m^3

Φ_{rdom} = refractory DOM concentration, g/m^3

Φ_a = algal concentration, g/m^3

K_{ae} = algal excretion rate, sec^{-1}

K_{am} = algal mortality rate, sec^{-1}

K_{ldom} = labile DOM decay rate, sec^{-1}

K_{lrdk} = labile to refractory DOM decay rate, sec^{-1}

P_{am} = partition coefficient for algal mortality, sec^{-1}

γ_{om} = temperature rate multiplier for organic matter decay

$A3$ = specifier for aerobic or anaerobic decay processes and

$$K_{om,light} = \alpha_{om} I_0 e^{-\gamma Z} \tag{3.14}$$

where α_{om} (cm^2/cal) is a constant relating irradiance to the decay of RDOM, and with other variables discussed in Section 3.3.1.

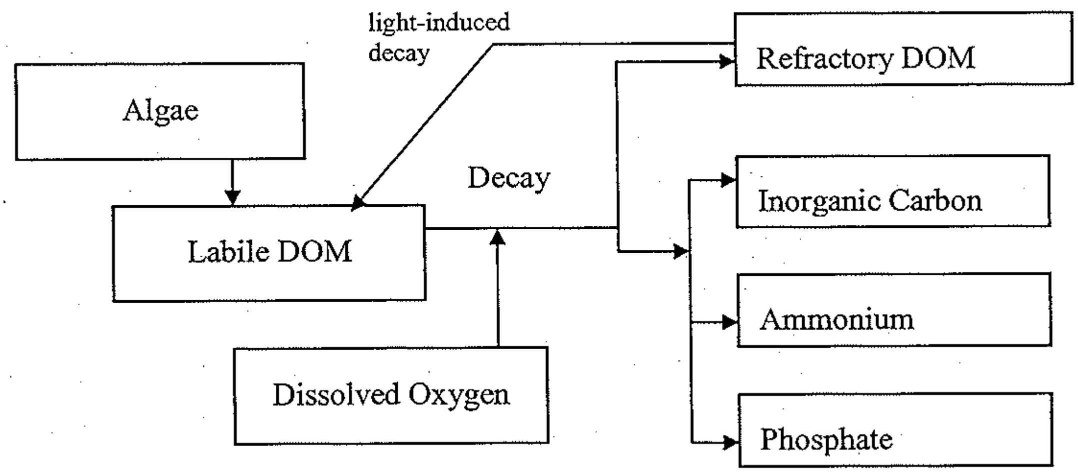

Figure 3.6 Schematic of internal decay and generation processes affecting labile DOM

3.3.3 Refractory Dissolved Organic Matter (RDOM)

Refractory dissolved organic matter is input from sources and lost to outlets. It is formed by the decay of labile DOM, and is decayed to inorganic carbon, ammonium, and phosphorus while consuming dissolved oxygen. A pathway of light induced decay from RDOM to LDOM has also been included. Figure 3.7 presents a schematic describing these processes. The time rate of change of refractory DOM is:

$$\frac{\partial \Phi_{rdom}}{\partial t} = \gamma_{om} A3\left(K_{lrdk}\Phi_{ldom} - K_{rdom}\Phi_{rdom}\right) - K_{om,light}\Phi_{rdom} \tag{3.15}$$

where

Φ_{rdom} = refractory DOM concentration, g/m^3

Φ_{ldom} = labile DOM concentration, g/m^3

K_{rdom} = refractory DOM decay rate, sec^{-1}

K_{lrdk} = labile to refractory DOM decay rate, sec^{-1}

γ_{om} = temperature rate multiplier for organic matter decay

$A3$ = specifier for aerobic or anaerobic decay processes

and

$$K_{om,light} = \alpha_{om} I_0 e^{-\gamma z} \qquad (3.16)$$

where α_{om} (cm^2/cal) is a constant relating irradiance to the decay of RDOM, and with other variables as discussed in Section 3.3.2.

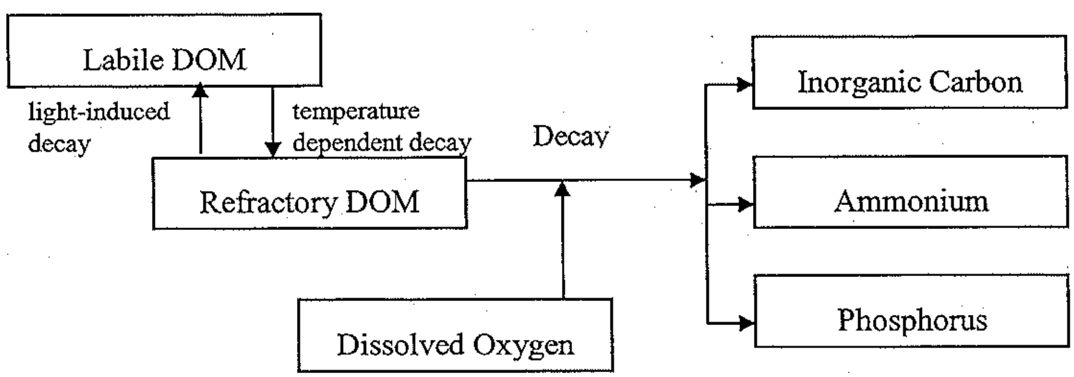

Figure 3.7 Schematic of internal decay and generation processes affecting RDOM

3.3.4 Algae

CE QUAL W2 version 2, used in this study, models algae as a single compartment to represent all phytoplankton present in an ecosystem. Determining model parameters therefore causes the user to create a modeled algal species that represents either the average characteristics of algae in the water body, or that represents one species. Version 3 of the software allows the user to model up to six algal compartments.

The processes affecting phytoplankton in CE QUAL W2 are limited to algal growth by photosynthesis, algal respiration, algal excretion, and algal mortality. The rate of change of phytoplankton is:

$$\frac{\partial \Phi_a}{\partial t} = \left(K_{ag} - K_{ar} - K_{ae} - K_{am}\right)\Phi_a - \frac{\omega_a}{\Delta z}\Phi_a \qquad (3.17)$$

where

Φ_a = algal concentration, g/m^3

K_{ag} = algal growth rate, sec^{-1}

K_{ar} = algal respiration rate, sec^{-1}

K_{ae} = algal excretion rate, sec^{-1}

K_{am} = algal mortality rate, sec^{-1}

ω_a = algal settling rate m/sec

Δz = layer thickness, m

Each rate coefficient is determined from a maximum rate determined by calibration and included in the control file, and rate multipliers determined by environmental factors. The algal growth rate is determined by the equation:

$$K_{ag} = \gamma_{ar}\gamma_{af}\min(\lambda_I, \lambda_P, \lambda_N)K_{ag\max} \qquad (3.18)$$

where

γ_{ar} = temperature rate multiplier for temperatures lower than optimum for algal growth

γ_{af} = temperature rate multiplier for temperatures higher than optimum for algal growth

λ_I = light-limited growth factor

λ_P = phosphorus-limited growth factor

λ_N = nitrogen-limited growth factor

$K_{ag\max}$ = maximum algal growth rate, AGROW in control file, sec^{-1}

Determination of the rate multipliers and limiting growth factors are somewhat complex processing procedures. Garvey (2000) discusses the rate multipliers, which are generated using the four ALGT and four AGK Thornton and Lessem (1978) rate multipliers for specific temperatures. Both the rate multipliers and limiting growth factor are discussed in detail in Cole and Buchak (1995).

The light limited growth factor is again used for determining algal excretion that generates LDOM, determined by the equation:

$$K_{ae} = (1 - \lambda_I) K_{ae\max} \quad (3.19)$$

where

$K_{ae\,max}$ = maximum algal excretion rate, AEXCR in the control file, sec^{-1}

Algal respiration, which produces inorganic carbon, nitrate/nitrite, ammonium, and phosphate, is described by the equation

$$K_{ar} = \gamma_{ar} K_{ar\max} \quad (3.20)$$

where

K_{armax} = maximum algal respiration rate, ARESP in control file, sec^{-1}

and γ_{ar} was previously defined.

Algal mortality produces particulate matter, modeled as detritus, and labile DOM. Algal mortality is approximated by the equation:

$$K_{am} = \gamma_{af} K_{am\max} \quad (3.21)$$

where

K_{armax} = maximum algal mortality rate, AMORT in control file, sec^{-1}

and γ_{af} was previously defined.

A schematic of the internal relationships between algae and other compartments are shown in Figure 3.8. Generally, an increased algal growth rate leads to increased phytoplankton production, while increased algal respiration, excretion, and mortality leads to decreased phytoplankton production, although through different pathways and yielding different products. Algal biomass is also lost by settling. The model calculates the settling flux exiting an upper layer and entering a lower layer based on algal concentration and a settling velocity, ALGS, specified by the user.

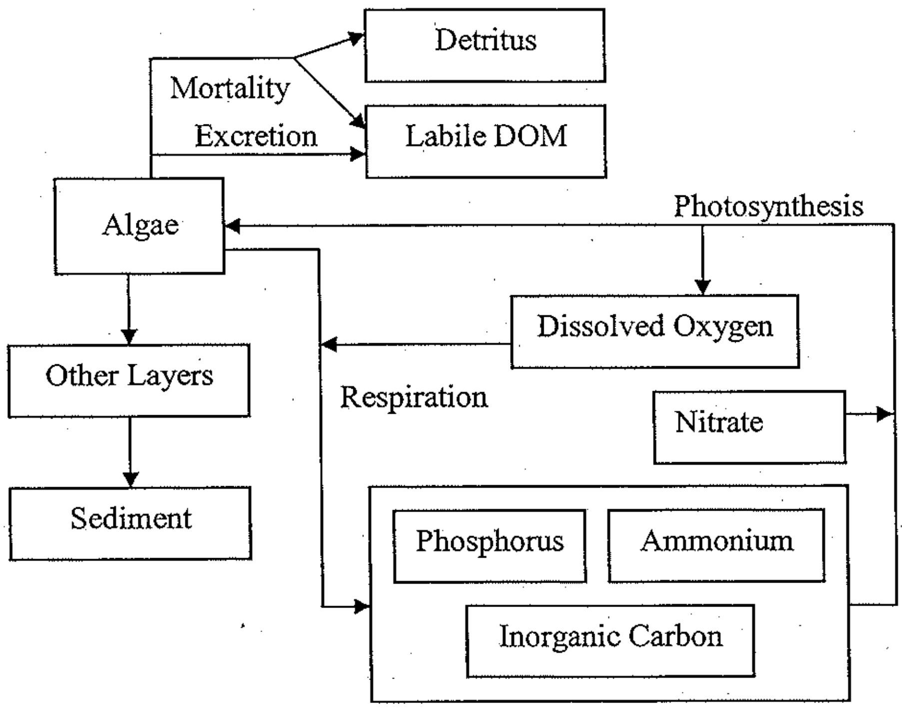

Figure 3.8 A schematic representation of the internal processes affecting algae.

3.3.5 Detritus

Detritus represents non-algal particulate organic matter in CE QUAL W2. Algal mortality is typically the largest source of this constituent in quiescent water bodies. Detritus is gained from sources, lost to outlets, generated by dying algae, decayed to inorganic carbon, ammonium, and phosphorus, and settled to lower layers and to the sediment. Accumulated detritus may be decayed in the sediment. A schematic showing these relationships may be seen in Figure 3.9. The rate equation for detritus is:

$$\frac{\partial \Phi_{dt}}{\partial t} = P_{am} K_{am} \Phi_a - K_{dt} \gamma_{om} \Phi_{dt} - \frac{\omega_{dt}}{\Delta z} \Phi_{dt} \qquad (3.21)$$

where

Φ_{dt} = detritus concentration, g/m^3

P_{am} = partition coefficient for algal mortality, sec^{-1}

K_{am} = algal mortality rate, sec^{-1}

Φ_a = algal concentration, g/m^3

K_{dt} = detritus decay rate, sec^{-1}

γ_{om} = temperature rate multiplier for organic matter decay

ω_{dt} = detrital settling rate m/sec

Δz = layer thickness, m

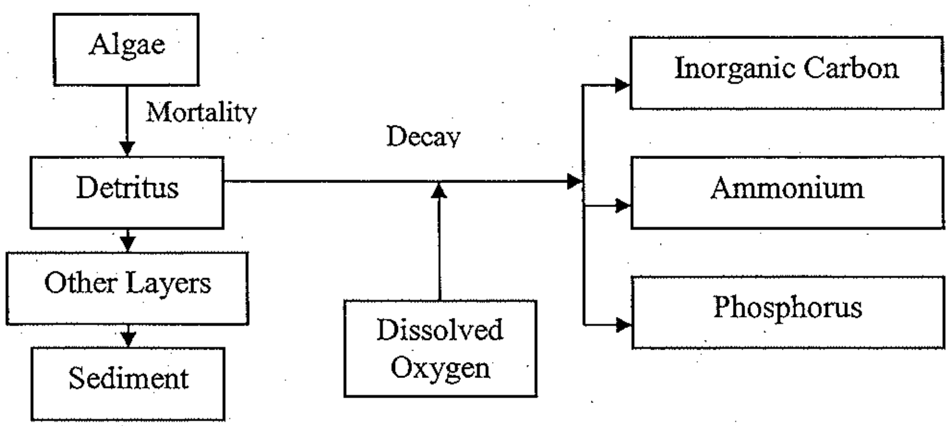

Figure 3.9 Schematic of internal decay and generation processes affecting detritus

3.3.6 Phosphorus

Algal growth in Wachusett Reservoir is phosphorus limited, as is the case for most temperate lakes and reservoirs (Worden 2003). Modeling this nutrient well is therefore important as variations in phosphorus will cause variations in algae when light and temperature growth conditions are favorable. Care needs to be taken when using phosphorus data, as measurements for total phosphorus do not reflect bioavailable phosphorus concentrations. If only total phosphorus data is available, sensitivity analysis must be performed to determine a scaling factor for phosphorus data. Ideally, measurements of soluble reactive phosphorus or orthophosphate should be used in inputs.

Phosphorus is gained from sources, from the decay of labile and refractory DOM, and detritus, through release from sediment, and from algal release through respiration. It is lost to outlets, to algal growth and through adsorption to the surface of suspended solids that settle.

The adsorption process only occurs under oxic conditions, following the low-concentration, and therefore linear, region of a Langmuir isotherm. The CE QUAL W2 factor O2LIM is the minimum dissolved oxygen concentration at which adsorption is allowed. Once adsorbed, phosphorus settles with the suspended solids. As different suspended solids have different settling rates as defined by the user, modeling phosphorus settling as such essentially allows phosphorus to settle at different rates depending on the dominant suspended solid. Inorganic suspended solids were not modeled in this study as they occur at very low levels in Wachusett Reservoir so adsorption of phosphorus is not important here. Phosphorus release occurs if the dissolved oxygen concentration falls below O2LIM. Release would not occur in Wachusett Reservoir, as anoxic conditions are not observed, as mentioned in Section 1.3.3. A schematic showing phosphorus dynamics is shown in Figure 3.10. Phosphorus dynamics are described by the equation:

$$\frac{\partial \Phi_p}{\partial t} = (K_{ar} - K_{ag})\delta_P \Phi_a + K_{ldom}\delta_P \gamma_{om}\Phi_{ldom} + K_{dt}\partial_P \gamma_{om}\Phi_{dt} + K_{rdom}\delta_p \gamma_{om}\Phi_{rdom} + \frac{S_{od}\gamma_{om}A_s}{V_{box}} - \frac{P_P(\omega_{ss}\Phi_{ss} + \omega_{dt}\Phi_{dt} + \omega_{FE}\Phi_{FE})}{\Delta z}\Phi_P \quad (3.22)$$

where

Φ_p = phosphorus concentration, g/m^3

Φ_{SS} = inorganic suspended solids concentration, g/m^3

Φ_{FE} = particulate iron concentration, g/m^3

δ_P = stoichiometric coefficient for phosphorus

P_P = adsorption coefficient for phosphorus, sec^{-1}

S_{od} = sediment release rate, g/m^2-sec

A_s = sediment area, m^2

V_{box} = volume of a segment-layer box, m^3

ω_{SS} = inorganic suspended solid settling rate, m/sec

ω_{FE} = particulate iron settling rate m/sec

Δz = layer thickness, m

and other variables and parameters as previously defined.

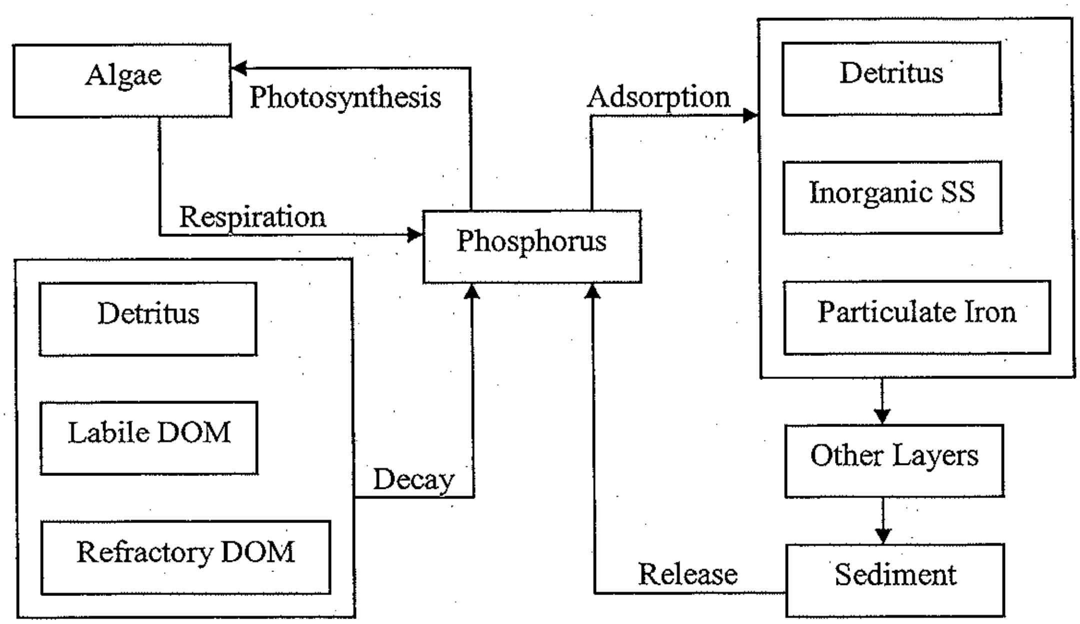

Figure 3.10 Schematic of internal phosphorus dynamics

3.3.7 Ammonium

Ammonium-nitrogen is received from sources, released by algal respiration, and by the decay of detritus, labile DOM, refractory DOM, and sediment (aerobic conditions), and released from sediment during anaerobic conditions in a zero-order process (which does not occur in Wachusett Reservoir) or as set by the model parameter O2LIM. Ammonium does not adsorb to sediment as does phosphorus, but instead constitutes a portion of the organic sediment. Additionally, ammonium is lost through algal growth, nitrification under oxic conditions (i.e. concentration of dissolved oxygen > O2LIM). These relationships are shown in Figure 3.11 and the equation:

$$\frac{\partial \Phi_{NH4}}{\partial t} = K_{ar}\delta_N \Phi_a - K_{ag}\delta_N \Phi_a \frac{\Phi_{NH4}}{\Phi_{NH4} + \Phi_{NO3}} + K_{ldom}\delta_N \gamma_{om} \Phi_{ldom}$$
$$+ K_{rdom}\delta_N \gamma_{om} \Phi_{rdom} + K_{dt}\delta_N \gamma_{om} \Phi_{dt} + K_S \delta_N \gamma_{om} \Phi_S + \frac{S_{od}\gamma_{om} A_S}{V_{box}} \quad (3.23)$$
$$- K_{NH4}\gamma_{NH4}\Phi_{NH4}$$

where

Φ_{NH4} = ammonium concentration, g/m^3

Φ_{NO3} = nitrate-nitrite concentration, g/m^3

Φ_S = mass of sediment, g

δ_N = stoichiometric coefficient for nitrogen

γ_{NH4} = temperature rate multiplier for nitrification

K_{NH4} = nitrification rate, sec^{-1}

K_S = sediment decay rate, sec^{-1}

and all other variables as previously defined

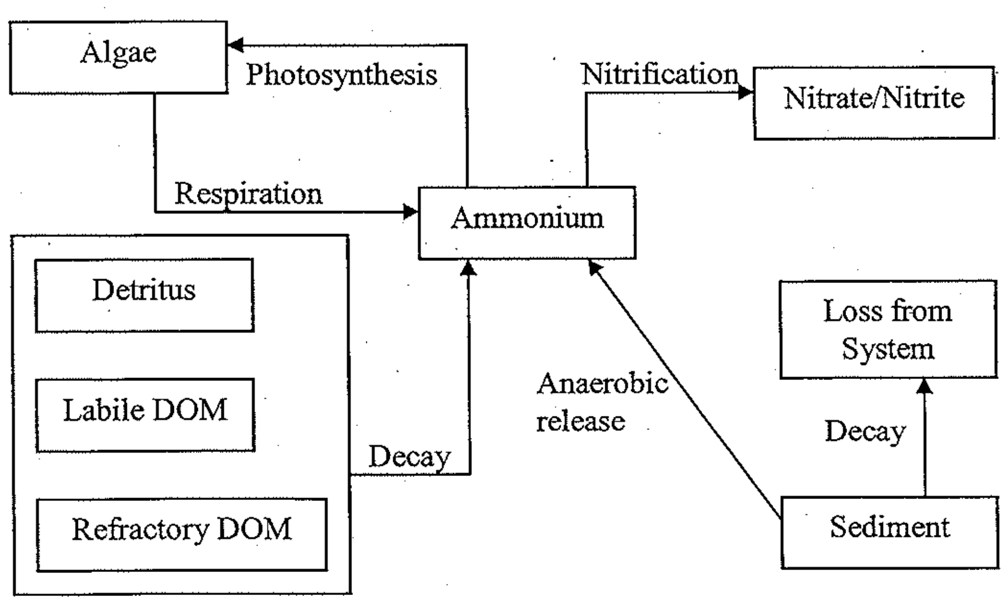

Figure 3.11 Schematic of internal ammonium dynamics

3.3.8 Nitrate-Nitrite

Nitrate and nitrite are modeled as one constituent in CE QUAL W2, as nitrite is a short-lived product of nitrification (only occurring when dissolved oxygen is greater than O2LIM) that always decays to nitrate and is typically low in concentration. It is likely that denitrification does not occur in Wachusett Reservoir as dissolved oxygen levels are too high. Nitrate is used to produce algae during photosynthesis. Figure 3.12 shows the interactions between nitrate-nitrite and other parameters, as does the equation:

$$\frac{\partial \Phi_{NO3}}{\partial t} = K_{NH4}\gamma_{NH4}\Phi_{NH4} - K_{NO3}\gamma_{NO3}\Phi_{NO3} - K_{ag}\delta_N\Phi_a\left(1 - \frac{\Phi_{NH4}}{\Phi_{NH4} - \Phi_{NO3}}\right) \quad (3.24)$$

where all variables have been previously defined.

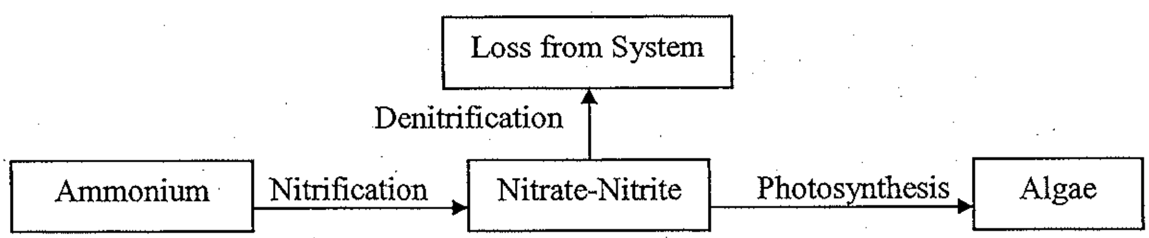

Figure 3.12 Schematic of internal nitrate-nitrite dynamics

3.3.9 Water Quality Parameter Values

Calibration exercises conducted during CE QUAL W2 modeling studies on Quabbin and Wachusett Reservoirs have yielded generally good results. However, there is some variation in parameter values chosen, as the studies have had different goals. Calibrating this model required the selection of a set of initial values for each of the parameters mentioned in Sections 3.3.1 through 3.3.8. Table 3.4 presents parameters values that have resulted from previous studies. Initial values for this study were taken from Roberts (2003) as the goals and means of the two studies are quite similar.

Table 3.4 Parameter values obtained from previous Quabbin and Wachusett modeling studies

CE QUAL W2 Parameter	Description	Units	Source		
			CDM	Garvey	Roberts
ALPHA	Impact of irradiance on UV254	cm^2/cal	-	-	2.6E-6
THETA	Arrhenius constant for UV254 decay		-	-	1.03
COLDK	UV254 dark decay at 20 °C	day^{-1}	-	-	0.0003
OMT1	Lower limit of OM decay	°C	0	-	0
OMT2	Lower limit of maximum-rate OM decay	°C	15	-	15
OMK1	Decay rate multiplier at OMT1		0.1	0.1	0.1
OMK2	Decay rate multiplier at OMT2		0.98	0.98	0.98
LDOMDK	Labile DOM decay rate	day^{-1}	0.3	-	0.003
LRDK	Labile to refractory DOM decay rate	day^{-1}	0.003	-	0.0003
RDOMDK	Refractory DOM decay rate	day^{-1}	0.003	-	0.0003
LPOMDK	Detrital decay rate	day^{-1}	0.007	-	0.007
LPOMS	Detrital settling rate	m/day	0.35	-	0.35
AG	Maximum algal growth rate (AGROW)	day^{-1}	0.9	0.84	3.5
AE	Maximum algal excretion rate (AEXCR)	day^{-1}	0.02	0.012	0.012
AM	Maximum algal mortality rate (AMORT)	day^{-1}	0.03	0.03	0.03
AR	Maximum algal respiration rate (ARESP)	day^{-1}	0.1	0.1	0.2
ALGS	Algal settling rate	m/day	0.29	-	0.29
ASAT	Saturation light intensity at AGROW	W/m^2	50	50	50
APOM	Detritus:algal biomass for algal mortality		0.8	-	0.8
ALGT1	Lower limit of algal growth	°C	0	0	0
ALGT2	Lower limit of max algal growth	°C	17	17	17
ALGT3	Upper limit of max algal growth	°C	22	22	22
ALGT4	Upper limit of algal growth	°C	28	28	28
AGK1	Fraction of AGROW at ALGT1		0.1	0.1	0.1
AGK2	Fraction of AGROW at ALGT2		0.98	0.98	0.98
AGK3	Fraction of AGROW at ALGT3		0.98	0.98	0.98
AGK4	Fraction of AGROW at ALGT4		0.1	0.1	0.1
AHSP	Algal 1/2 saturation constant for PO4	g/m^3	0.001	0.016	0.016
PARTP	Phosphorus adsorption coefficient		1.2	-	1.2
PO4REL	Sediment release-rate of PO4 (fraction of SOD)		0.005	-	0.05
AHSN	Algal 1/2 saturation constant for NH4	g/m^3	0.062	0.062	0.062

Table 3.4 Parameter values obtained from previous Quabbin and Wachusett modeling studies (continued)

CE QUAL W2 Parameter	Description	Units	CDM	Garvey	Roberts
NH4DK	Ammonia decay rate	day^{-1}	0.03	-	0.03
NH4T1	Lower limit of ammonium decay	°C	0	-	0
NH4T2	Lower limit of maximum-rate ammonium decay	°C	15	-	15
NH4T3	Fraction of nitrification rate at NH4T1		0.1	-	0.1
NH4T4	Fraction of nitrification rate at NH4T2		0.98	-	0.98
NO3DK	Nitrate decay rate	day^{-1}	0.1	-	0.1
NO3T1	Lower limit of nitrate decay	°C	0	-	0
NO3T2	Lower limit of maximum-rate nitrate decay	°C	15	-	15
NO3K1	Fraction of denitrification rate at NH4T1		0.1	-	0.1
NO3K2	Fraction of denitrification rate at NH4T2		0.98	-	0.98
CO2REL	Sediment CO2 release rate (fraction of SOD)		0.1	-	0.1*
O2NH4	Oxygen stoichiometric equiv. for NH4 decay		3.43	-	4.57*
O2ORG	Oxygen stoichiometric equiv. for OM		1.4	-	1.4*
O2RESP	Oxygen stoichiometric equiv. for dark respiration		1.2	-	1.4*
O2ALG	Oxygen stoichiometric equiv. for algal growth		1.1	-	1.4*
BIOP	Ratio of Phosphorus to OM		0.004	-	0.011*
BION	Ratio of Nitrogen to OM		0.067	-	0.08*
BIOC	Ratio of Carbon to OM		0.5	-	0.45*
O2LIM	Maximum DO concentration for anaerobic processes	g/m^3	0.2	-	0*

*Values recommended by Cole and Buchak (1995) for all studies

3.3.9.1 UV254 Absorbance Parameters

Roberts (2003) initially selected values for ALPHA, THETA, and COLDK of 0.014 cm^2/cal, 1.03-1.07, and 0.014 day^{-1}, respectively. These values were determined by Tobiason *et al.* (1998) as optimum for modeling coliform bacteria in Quabbin, and were selected because no literature values existed for UV254 decay parameters. Subsequent analysis determined that a COLDK value of 0.0003 day^{-1} was appropriate, the same value as used for the first order refractory DOM decay rate. Since UV254 is a surrogate for measuring humic material, the decay rates should be similar. Subsequent calibration then determined a value of 2.6×10^{-6} cm^2/cal for ALPHA. These values are reasonable starting values for the Wachusett calibration, as both reservoirs have watersheds that are somewhat similar, and greater than 50% of water in Wachusett originates at Quabbin annually. However, Quabbin has a minimum mean residence time of 3.7 years (during periods of transfer to Wachusett (Garvey 2000)) while the mean residence time of Wachusett is 0.6 years. It is therefore likely that organic matter in Wachusett is more allochthonous, and decay rates may be affected.

3.3.9.2 Organic Matter Parameters

Three organic matter decay rates are included in CE QUAL W2: LDOMDK describes decay from labile DOM to nutrients and inorganic carbon, LRDK describes decay from labile DOM to refractory DOM, and RDOMDK describes decay from refractory DOM to nutrients and inorganic carbon. These rates can be estimated from biological oxygen demand (BOD) rates if they are known, and Cole and Buchak (1995) suggest that LDOMDK should be two orders of magnitude larger than RDOMDK. Roberts (2003) selected decay rates equating LDOMDK to LRDK, RDOMDK to COLDK, and setting LDOMDK one order of magnitude larger than RDOMDK. Unfortunately, there is little data available on the character of in-reservoir organic carbon, so this study will follow these guidelines.

It is possible that the inclusion of a light-induced pathway for decaying refractory DOM to labile DOM might better reflect in-reservoir decay conditions. However, as that pathway would induce a continuous feedback loop between labile and refractory DOM, more data and laboratory experiments would be necessary to estimate and calibrate the quantity of organic matter that decays in that manner.

Additionally, OMTI and OMT2 are parameters that set the minimum and maximum temperatures used in defining the curve that adjusts organic matter decay for temperature. OMK1 and OMK2 are the fractions of the organic matter decay rates that occur at these temperatures. The values selected by CDM (1995) are similar to default values recommended by Cole and Buchak (1995).

3.3.9.3 Algae Parameters

The parameters AG, AR, AE, AM, and AS were initially based on the study by Roberts (2003) on Quabbin, although the algal characteristics of the reservoirs are quite different. Quabbin is so nutrient poor that AG had to be set at a high value to cause CE QUAL W2 to predict algae production on the same order of magnitude of measured production in the reservoir. Wachusett water is somewhat more nutritive and most likely a smaller maximum growth rate is appropriate. AR was also adjusted by Roberts (2003) although the other parameters used were selected by Garvey (2000) and validated by Roberts.

ALGT1, ALGT2, ALGT3, and ALGT4 are temperatures selected by the user that set the curve that adjusts the algal growth rate. AGK1, AGK2, AGK3, and AGK4 determine the fraction of maximum algal growth that occurs at those temperatures. In this study, these parameters were initially set to values determined by CDM (1995). With ALGT2 and ALGT3 set at 17 and 22 °C, algae that favor warm water conditions will be predicted better than other species.

ASAT, AHSP and ASHN impact the response of algae to light, nitrogen, and phosphorus conditions. The initial values used in this study (ASAT and AHSN are based on CDM (1995) and AHSP based on Garvey (2000)) were the same as used by Roberts (2003).

3.3.9.4 Detritus Parameters

All nonliving organic particles are modeled as detritus by CE QUAL W2, so the decay and settling rates selected represent average rates for POM in the reservoir. The initial value for LPOMDK was set to the value determined by CDM (1995), as was initial value for the detritus settling rate, LPOMS. The parameter APOM defines the fraction of algal biomass after mortality that is lost to detritus. Cole and Buchak (1995) refer to a study by Otsuki and Hayna (1972) that determined this value to be 0.8. This value was used in all previous Wachusett and Quabbin modeling studies. The remaining biomass is lost to labile DOM.

3.3.9.5 Nutrients

The absence of anoxic conditions in Wachusett Reservoir make consideration of the parameters PO4REL, NH4REL, NO3DK, NO3T1, NO3T2, NO3K1, and NO3K2 unnecessary. Ignoring inorganic suspended solids and iron makes consideration of the adsorption coefficient PARTP irrelevant.

Nitrification does occur under aerobic conditions, however, so it is necessary to consider NH4DK, the decay rate for ammonium to nitrate. The initial value selected for this parameter originated with the CDM (1995) study and was validated by Roberts (2003). NH4T1 and NH4T2 are also necessary to consider. They are the lower and upper temperature limits describing the curve used to determine the fraction of NH4DK that occurs at certain temperatures. NH4K1 and NH4K2 are the fractions of the maximum decay rate at these points.

3.3.9.6 Stoichiometry

Table 3.4 includes values for several stoichiometric parameters included in CE QUAL W2. Cole and Buchak (1995) state that the default values of these parameters should be maintained unless data exists that suggest otherwise. Default values are based on the empirically derived Redfield ratios (Redfield 1934) of organic matter composition, and other sources. CDM (1995) used different values for Wachusett in their study. The reason for this difference is unknown, although CDM modeled algae as chlorophyll A (Chl.A) instead of as carbon, so it is possible that the values used are based on organic matter as Chl.A. The stoichiometric parameters hold for all organic matter, but CDM did not consider DOM or detritus, so the discrepancy would not have impacted their results.

3.3.10 Water Quality Initial Concentrations

Initial concentrations must be defined for each constituent in each layer and segment for the beginning of the model run period. Initial conditions may be uniform throughout the reservoir, or vary vertically or longitudinally. In Quabbin Reservoir, Roberts (2003) set uniform initial conditions for all constituents except UV254, which exhibited strong longitudinal gradients.

In this study, uniform initial conditions were used. Model runs began in winter or early spring before the reservoir becomes vertically stratified. The reservoir does exhibit some longitudinal concentration gradients, but they are relatively small and vary rapidly, especially during periods of transfer. The initial concentration was set within the range of withdrawal concentrations as measured at Cosgrove and corresponding to the beginning of the model run.

3.3.11 Water Quality Constituent Data

It is recommended by Cole and Buchak (1995) that water quality constituent data for inputs be provided at a monthly frequency. CE QUAL W2 then interpolates between input data points as necessary. Unfortunately, during the period of this study, monthly water quality data are only available for the tributary inputs during 2000 and 2001. In 2002, data for the minor tributaries are only available on a biannual basis. Fortunately, Stillwater and Quinapoxet data are available monthly. As they are more significant sources of water than the minor tributaries, having periodic data from Stillwater and Quinapoxet is more important. Tributary inflow data does not exist for certain constituents, so it is necessary to estimate or adjust data to meet the requirements of the model. No algae, POC, RDOM or LDOM data exists for any of the tributaries. TOC data does exist, however, so POC was assumed to be 5% of tributary TOC. The remaining 95% was assumed to be DOM, of which 20% was assumed to be labile and 80% and assumed to be refractory. This assumption is based on work by Roberts (2003) Garvey (2000), Hodgkins (1999), and Jordan and Likens (1975). Algal inputs from tributaries are assumed to be minor and were therefore ignored.

Tributary orthophosphate (soluble reactive phosphorus) data did not exist for the study period. However, total phosphorus data does exist for these sources. A comparison of outlet orthophosphate and total phosphorus indicated that approximately 50% of total phosphorus was orthophosphate. This assumption was utilized to adjust inflow phosphorus data, and then confirmed during model calibration (see section 4.4.2.1). Figure 3.13 provides a time-series plot of total phosphorus and orthophosphate as measured at Cosgrove. Note that many values are actually the detection limit of 0.0025 mg/L.

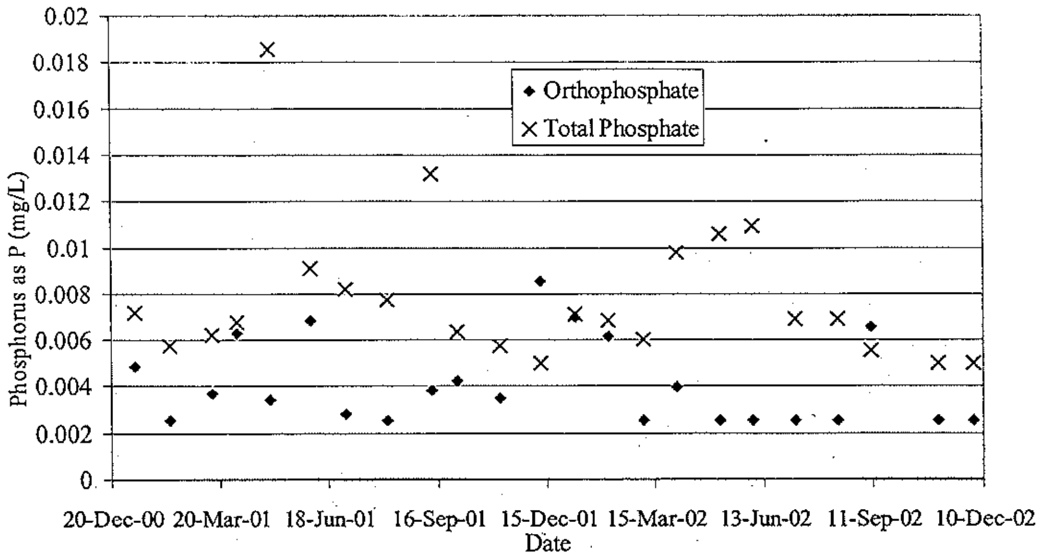

Figure 3.13 Time series plot of total phosphorus and orthophosphate.

MWRA does not measure constituent concentrations for Quabbin Aqueduct. However, all constituents of interest are measured at the Chicopee Valley Aqueduct (CVA) withdrawal from Quabbin. This outlet provides water to the Towns of Chicopee, South Hadley, and Wilbraham in western Massachusetts. Constituent concentrations in Quabbin tend to be stable, and Garvey (2000) reports that Cosgrove and CVA constituent levels are similar. Water discharged to Quabbin Aqueduct was assumed to contain water of the same composition as that discharged to the CVA. Similarly, no constituent data exists for Waushacum Brook, although the brook is the third largest tributary of the reservoir. Data from Stillwater River was applied to Waushacum Brook for the purposes of this research. This assumption should be checked with data.

There is also little data regarding direct runoff. For the purpose of this study, nutrient and NOM data from Purgee Brook at Quabbin Reservoir was used for direct runoff constituent levels. Purgee Brook drains a forested basin thought to be representative of the area of direct runoff to Wachusett Reservoir, as much of the direct runoff area is DCR owned and maintained.

Precipitation constituent data are also very scarce. The NADP stations at Lexington, MA and Quabbin Reservoir measure precipitation ammonia and nitrogen, but not phosphorus, UV254, or organic carbon constituents. Estimates for these parameters were based on those used by Roberts (2003) and Garvey (2000).

3.3.12 Water Quality Calibration Method

The primary means of calibrating the model is to compare model predictions of constituent concentration at the Cosgrove withdrawal to measured concentrations at that location. This method was implemented since understanding outlet water quality is the ultimate goal of all research conducted on the DCR/MWRA water system. Additionally, very little in-reservoir NOM data (including profiles) exist.

Starting with the initial parameter values, inflow concentration data, and in-reservoir conditions, adjustments were made to optimize the fit of modeled constituent concentrations to measured concentrations. Parameters were changed individually so the exact impact of one change would be known.

3.4 Model Execution

Input files were generated based on the requirements dictated by the programming code. Approximately 70 input files are required, along with one executable file containing the program, and a preprocessor to check the input files (if used). Changes to model geometry or number of inputs or withdrawals requires adjusting an include file (W2.INC) that must be compiled with the programming code. The control file (W2_CON.NPT) can be generated in any text editing program, as was done in most of this study, or generated using the software W2 Studio (JEEAI 1998). This software can also be used for postprocessing, although Microsoft Excel was used in this study.

4. Calibration Results and Discussion

Hydrodynamic calibrations were prepared for the calendar years 2000, 2001, and 2002 in this study. The years were calibrated separately, as each was characterized by unique hydrologic conditions that lead to distinct yet plausible results. Additionally, 2000 data used for modeling was implemented separately from 2001 and 2002 data, which was modeled sequentially for constituents in CE QUAL W2, so it was advantageous to prepare hydraulic data separately for the individual years.

A constituent calibration was conducted for the combined period of 2001 and 2002. These results were then validated with the 2000 hydrodynamic and constituent calibration. This arrangement proved best because less inflow constituent data exists for 2002 and because a hydrodynamic inconsistency arose in the 2002 calibration..

4.1 Hydrodynamic Modeling - 2000

4.1.1 Reservoir Inflows

The year 2000 was characterized as a typical year for precipitation received in the vicinity of Wachusett Reservoir. The Clinton meteorological station received 44.8 in (114 cm) of precipitation that year, as presented in Table 4.1 along with other precipitation statistics.

Table 4.1 2000 Precipitation Statistics

Total Precipitation, in. (cm)	44.8 (114)
Average Daily Precipitation, in/day (cm/day)	0.12 (0.31)
Number of Days with Precipitation	217
Average on Days with Precipitation, in/day (cm/day)	0.21 (0.52)

Precipitation fell during more than half of the days that year, with the largest precipitation events in the late spring. Figure 4.1 shows daily precipitation quantity for 2000. Two large storm events in April, where daily precipitation reached 1.7 and 2.1 in (4.3 and 5.3 cm), along with one day in June where 1.9 in (4.8 cm) fell, significantly raise the quantity of precipitation to fall in those months relative to other months.

The impact of these storm events on tributary discharge can be seen in the adjusted Stillwater and Quinapoxet hydrographs, shown in Figure 4.2 along with Quabbin transfers.

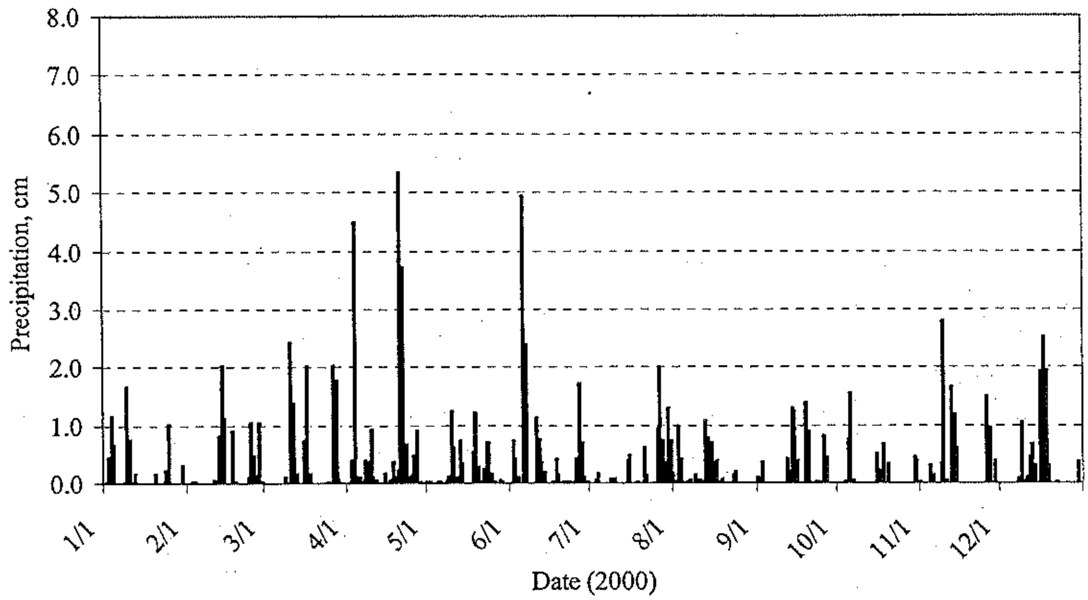

Figure 4.1 Daily precipitation for Wachusett Reservoir during 2000

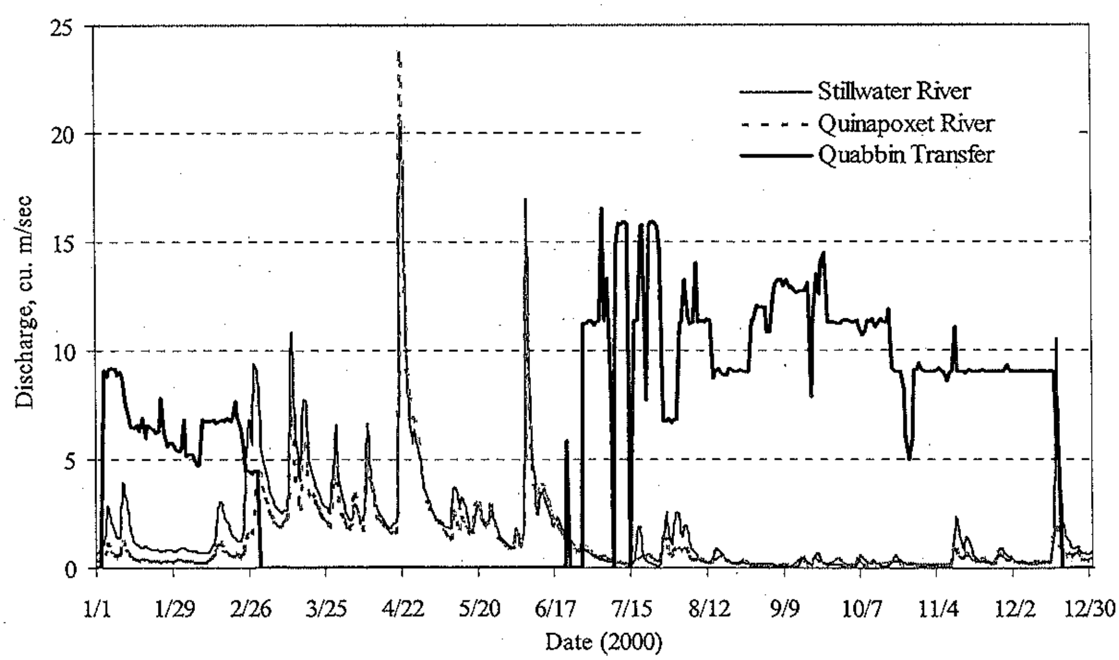

Figure 4.2 Stillwater and Quinapoxet River hydrographs with Quabbin Transfer (2000) composed of edited data (not USGS direct data).

It is notable that the discharges of the two tributaries are quite similar during the large storm events but deviate somewhat during small events and base flow conditions. It is probable that operation of the reservoirs that discharge into Quinapoxet have a significant impact on flow, impacting the ratio of Stillwater to Quinapoxet discharges. It is important to note that Figure 4.2 presents the data as adjusted by the water balance discussed in Section 3.2.1 instead of data as measured at the USGS gages.

Figure 4.3 presents the WSE for Wachusett Reservoir in 2000 along with the Quabbin transfer discharge rate. The horizontal lines are the limits of the DCR/MWRA reservoir operating range. The lower limit exists to prevent excessive exposure of shallow areas in the North Basin to reduce waterfowl roosting, while the upper limit is 0.15 m below the lower spill elevation of the reservoir.

Tributary discharges and precipitation quantity in the late winter and spring are large compared to those in the summer and fall. This trend is typical for Wachusett and was observed during 1998 and 1999 as well (Tobiason *et al.* 2002). During wet periods such as this, the yield of the Wachusett watershed is adequate to meet demand and maintain WSE. Thus, Quabbin transfer generally does not occur in spring. During periods where tributary discharges approach base flow conditions and precipitation is less frequent, Quabbin transfers are initiated to maintain WSE. In 2000, the majority of transfer occurred between the middle of June and the middle of December. Transfer discharge during this period was generally between 8.8 and 15.8 m^3/s. Some transfer also occurred in winter, between the beginning of January and the end of February.

Figure 4.4 presents the relative quantity of water received from each reservoir source for 2000. Wachusett received 51.5% of its water from Quabbin Transfers, 15.0% and 11.2% from Stillwater and Quinapoxet, respectively, 10.0% from direct runoff, 5% from direct precipitation, and the remaining 7.2% from the minor tributaries.

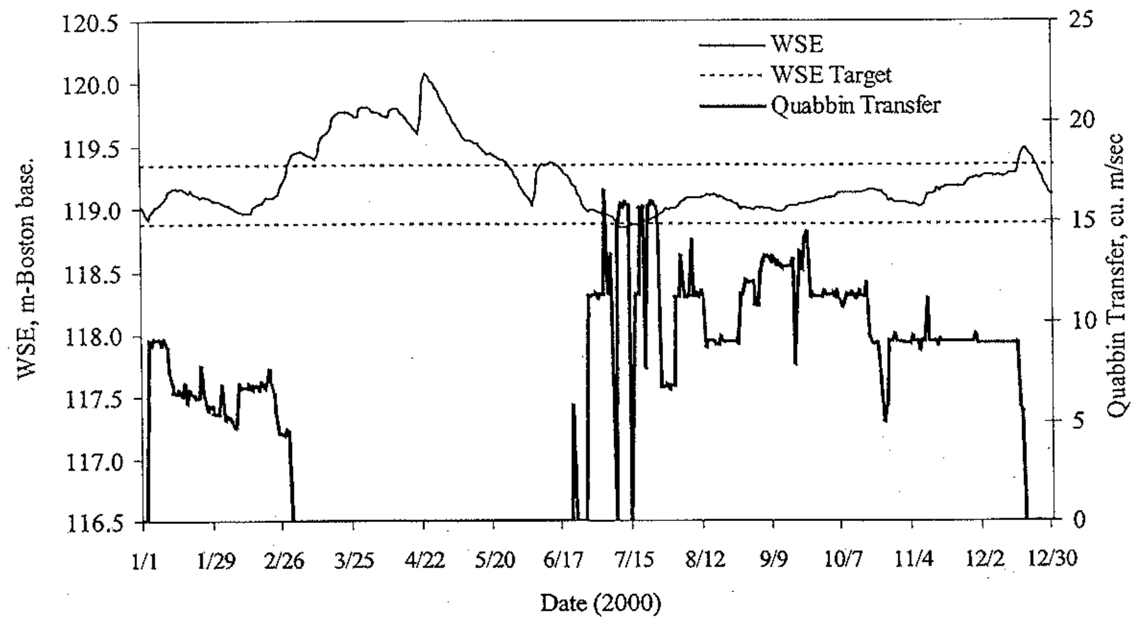

Figure 4.3 Quabbin transfer and water surface elevation for 2000

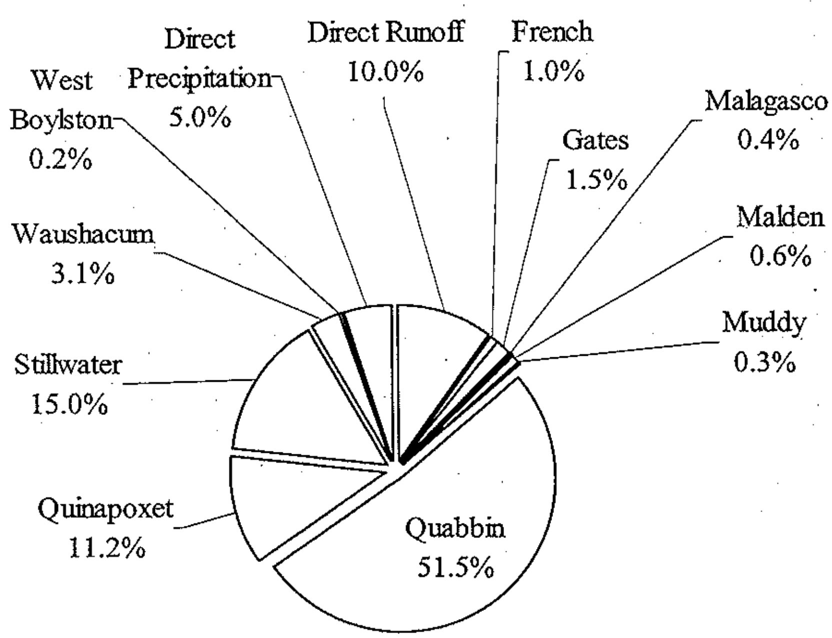

Figure 4.4 Relative contribution of 2000 inflows to Wachusett Reservoir

4.1.2 Reservoir Losses

The significant precipitation that occurred in June resulted in a large quantity of spill to the Nashua River, which reached a maximum of 25 m^3/sec on April 24. Figure 4.5 shows this discharge along with discharges through Cosgrove intake, WSE, and WSE operating range. Reservoir losses were dominated by this short period of Nashua River spilling and Cosgrove demands.

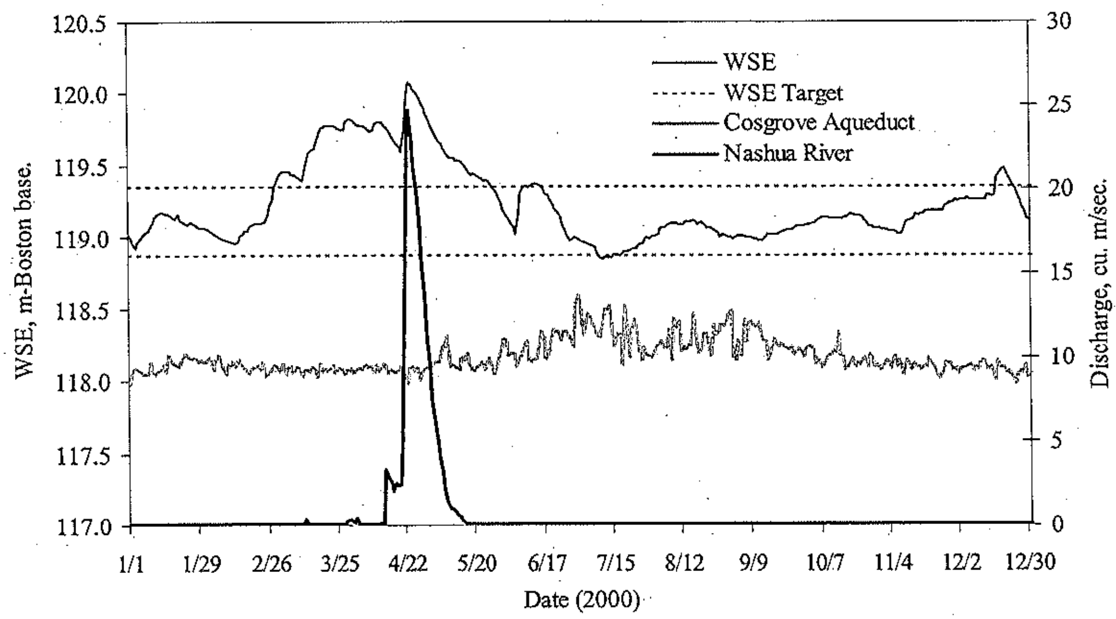

Figure 4.5 Major water losses and water surface elevation for 2000

Evaporative water loss from Wachusett Reservoir was estimated as described in Section 3.2.3.1, using meteorological data from the NOAA station at Worcester Airport. Estimates of water surface temperatures were generated by interpolating between surface measurements included in measured DCR in-reservoir profiles measurements available and by estimation based on 2001 and 2002 Cosgrove withdrawal temperatures during unstratified periods. Figure 4.6 presents daily estimated evaporation for 2000. Evaporation ranged between 0 and 1.4 m^3/s and averaged 0.41 m^3/s. More evaporation occurred during the second half of the year, when water surface temperatures were warmer.

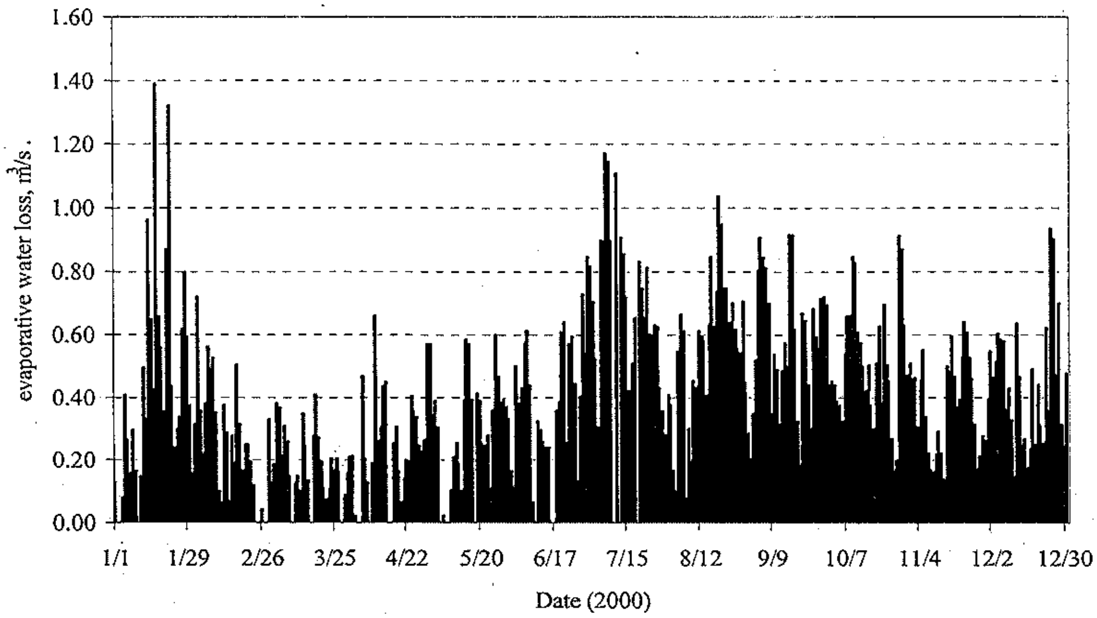

Figure 4.6 Estimated daily average evaporation rates for 2000

Figure 4.7 presents the relative quantity of water losses considered in this research for 2000. Cosgrove withdrew 88% of the annual water budget of the reservoir during that period, while 6.6% was discharged to the Nashua River (including required minimum flow) and 3.6% was lost to evaporation. The remaining 1.8% either was withdrawn by towns, was discharged to Wachusett Aqueduct, or seeped through North Dike.

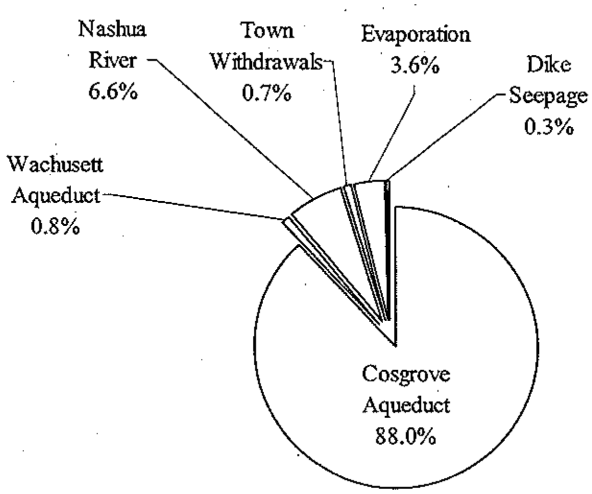

Figure 4.7 Relative quantity of water lost to the outflows of Wachusett Reservoir.

4.1.3 Calibration Results

Upon assembling all necessary inflow and outflow data, SOLVER was used to optimize calibration factors for the reservoir inflows as discussed in Section 3.2.2. Table 4.2 presents the inflow calibration factors determined by this analysis along with a range of inflow factors for 1994 through 1999 as reported in Ahlfeld *et al.* (2003a).

Table 4.2 Summary of 2000 and historic calibration factors

Inflow	Annual Average (1994-1999)	Range (1994-1999)	2000
Quabbin	1.04	1.0-1.1	1.19
Stillwater	0.98	0.70-1.28	1.27
Quinapoxet	1.14	1.04 -1.30	0.82
Waushacum	1.31	1.11-1.65	1.30
Direct Runoff	1.30	1.11-1.62	1.30
Malden	1.20	1.00-1.35	1.30
W. Boylston	1.26	1.11-1.35	1.30
Gates	1.36	1.11-2.00	1.30
Muddy	1.20	1.00-1.35	1.30
Malagasco	1.20	1.00-1.35	1.30
French	1.20	1.00-1.35	1.30

Most of the 2000 calibration factors are within the range established by the 1994 to 1999 values. The Quabbin transfer calibration factor calls for increasing discharge measurements by 19%, almost double the maximum change during the period of record. DCR estimates that Quabbin transfer measurements are accurate within 1 to 2%, suggesting that the calibration factor should range from 0.98 to 1.02. A value of 1.19 is therefore unexpected. It is possible that transfer data is missing, but there are no intervals of exceptionally poor fit. .Additionally, there is no systematic deviation, so the value was accepted. The calibration factor for Quinapoxet, 0.82, was the only other value to deviate from its 1994-1999 range. However, this value is within the overall range of historic factors (in 1995 a value of 0.70 was used for Stillwater) and is not far from the USGS determination that 95% of the values recorded at this gage are within 10% of the actual value (USGS 2003). It should be noted that data from the Stillwater gage is rated 'fair' or 'poor,' a result of beaver activity that influences river depth.

Table 4.3 presents unitless historic runoff coefficients for each tributary watershed for 1994 through 1999 as reported in Ahlfeld *et al.* (2003a) and for 2000.

Table 4.3 Summary of 2000 and historic runoff coefficients

Inflow	Annual Average (1994-1999)	Range (1994-1999)	2000
Stillwater	0.48	0.24-0.75	0.59
Quinapoxet	0.74	0.54-0.94	--
Waushacum	0.61	0.37-0.75	0.61
Direct Runoff	0.61	0.37-0.75	0.61
Malden	0.57	0.37-0.75	0.61
W. Boylston	0.59	0.37-0.75	0.61
Gates	0.63	0.37-0.75	0.61
Muddy	0.57	0.37-0.75	0.61
Malagasco	0.57	0.37-0.75	0.61
French	0.57	0.37-0.75	0.61

All runoff coefficients are similar to the average runoff coefficient as determined for 1994-1999 and within the range of 1994-1999 values. It is important to note that a runoff coefficient is not calculated for Quinapoxet River. Since that tributary loses water to the City of Worcester, withdrawals and spilling from Quinapoxet Reservoir impact river discharge and should be considered in runoff coefficient calculation as discussed in Ahlfeld *et al.* (2003a).

Figure 4.8 shows a comparison between Wachusett Reservoir measured WSE for 2000 and the WSE as predicted using the water quantity data and calibration factors determined by this research. Figure 4.9 presents the deviation between the modeled and measured WSEs. As shown, the model reasonably simulates the measurements with no systematic deviation. Statistics describing the deviation of the model from the measurements are presented in Table 4.4.

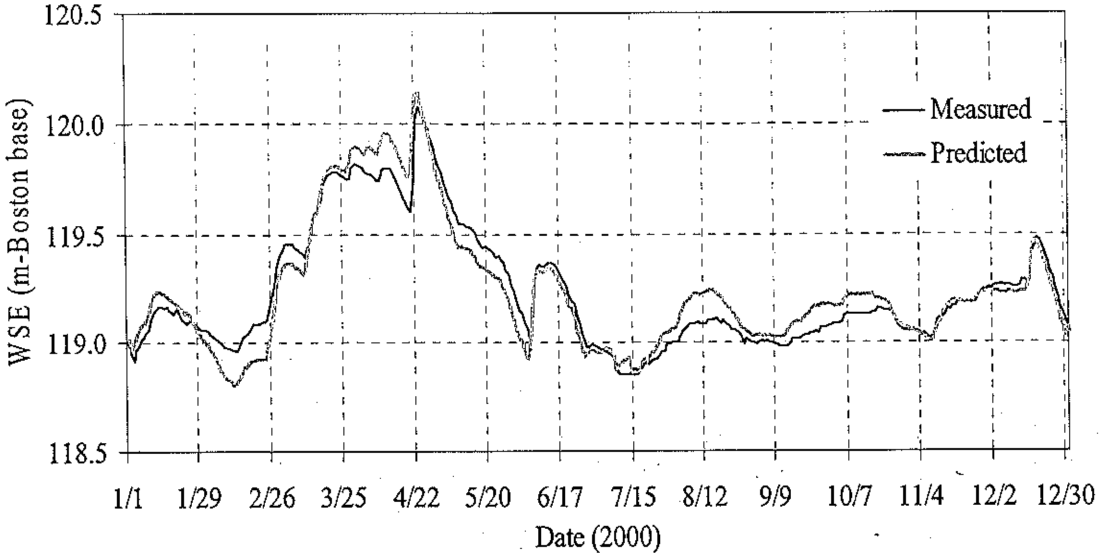

Figure 4.8 Wachusett Reservoir water budget calibration for 2000

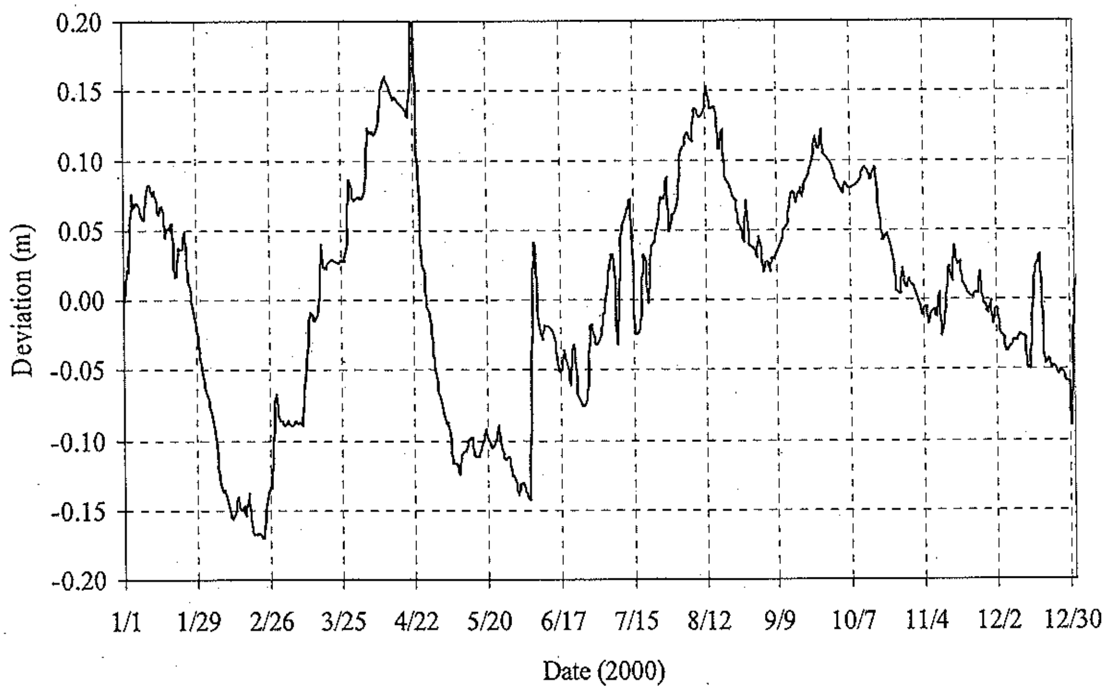

Figure 4.9 Deviation between 2000 Wachusett Reservoir modeled and measured WSE

Table 4.4 Selected 2000 statistics describing water budget calibration results

Statistic	Value
RMS error, m	0.082
Average Absolute Deviation, m	0.067
Maximum Positive Deviation, m	0.279
Maximum Negative Deviation, m	-0.171
No. Days Greater then +/- 0.15 m Deviation	18

These results demonstrate that the water budget calibration is reasonable. The goal of this calibration method is to maintain a deviation of less than 0.15 m between the predicted and measured WSEs. This criteria was not met on 18 days, or 4.9% of the year. Considering that the criteria was met during 95% of the year, and that the average absolute deviation is less than 0.1 meters relative to the 36.6 m maximum depth, the results will be considered reasonable and sufficient for this research. The relatively frequent exceedance of the deviation criteria most likely results from the wet nature of the year, where inflow quantities based on less accurate stream gage estimates and estimation account for nearly half of the water budget.

4.1.3.1 Temperature Profile Comparison

Figure 4.10 through Figure 4.24 show comparisons between measured temperature profiles in Wachusett Reservoir and those modeled by CE QUAL W2 for 2000. In most cases, the model predicted observed trends well. Modeled epilimnion and hypolimnion temperatures were generally within one degree C of data from measured profiles. In one instance, on May 10th, the modeled profile indicates 10 °C warmer temperatures at the surface. This discrepancy is resolved by the next profile date (June 15) and is indicative of the model predicting the onset of stratification too early. Modeled and measured profiles on August 31 and October 26 agree well and show that stratification ended between those dates.

The largest systematic discrepancies occur in the metalimnetic portions of the profiles. Here, modeled profiles somewhat underpredict temperature by 1 to 2 °C on most days and up to 5 °C on July 19th by predicting the thermocline to be 1 to 2 m too shallow. However, the overall shape of modeled thermocline described the measured thermocline well, with the exception of underpredicting temperature over the interval between ~9 and

13 m depth from July 17 to July 31. Between these depths, the presence of the Quabbin interflow causes the measured profile to show a smaller temperature gradient than those just above and below. The model does not capture this feature well.

It is also important to note that the model does not predict profiles measured in Thomas Basin well. Thomas Basin is characterized by very complex hydrodynamics as discussed in Kennedy (2003) and Pease (2004). The necessary assumption of lateral homogeneity made for CE QUAL W2 modeling does not hold in the portion of Thomas Basin where DCR records profiles. However, evidence presented in Section 4.3.3 suggests that the 2D model could adequately predict profiles measured at the Route 12 bridge constriction where the Thomas Basin joins the bulk of the reservoir.

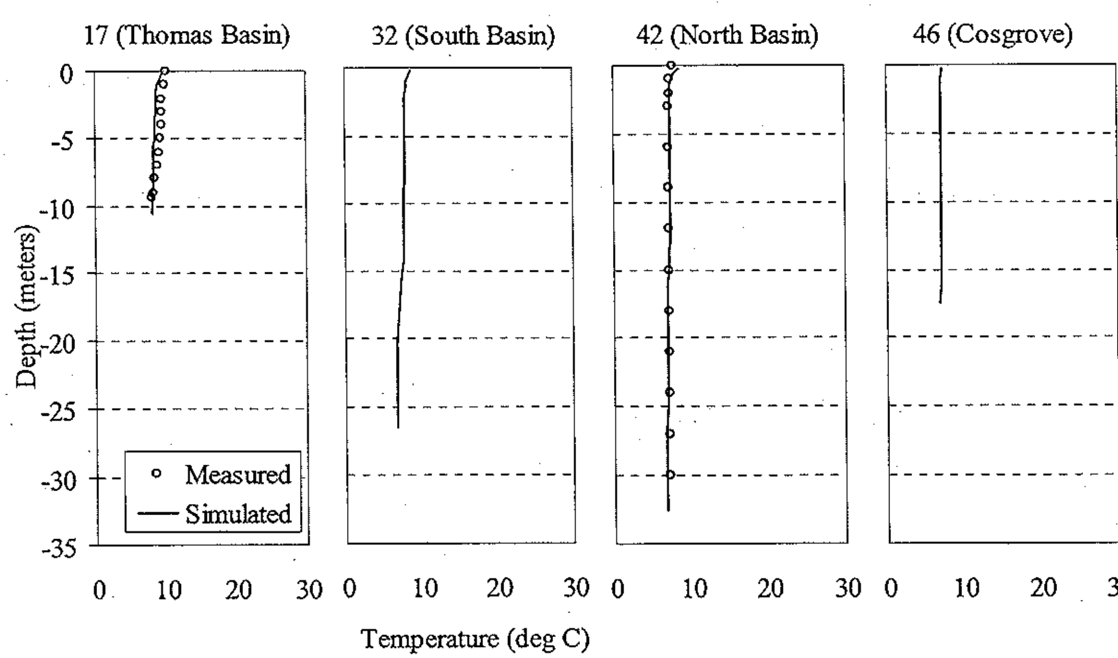

Figure 4.10 April 20, 2000 temperature profiles (°C)

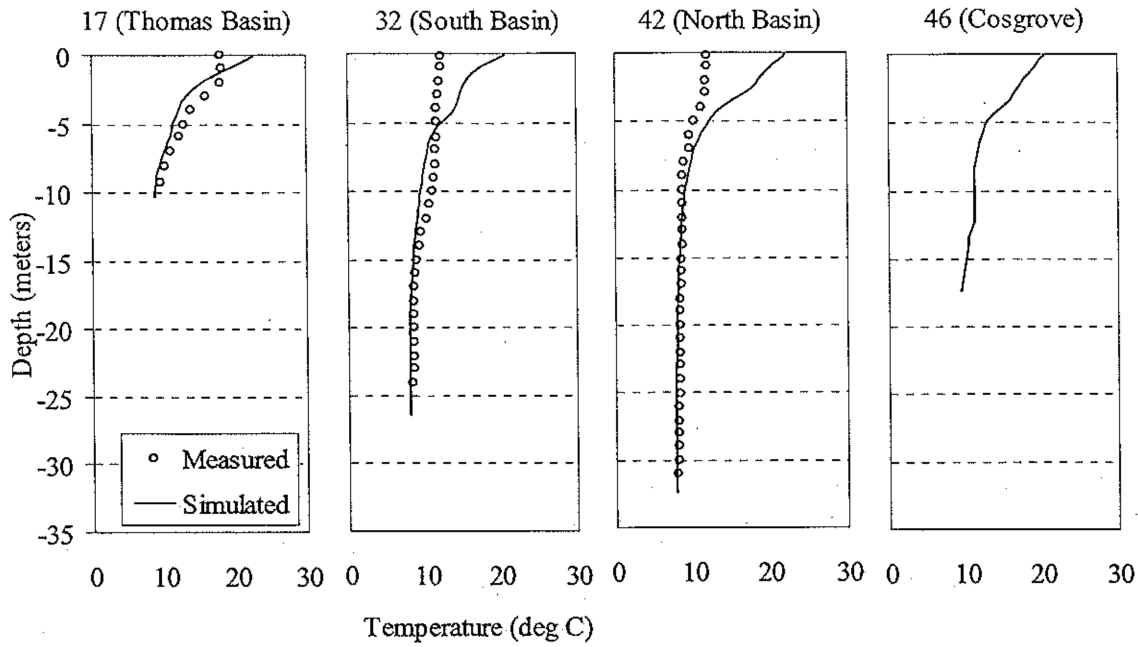

Figure 4.11 May 10, 2000 temperature profiles (°C)

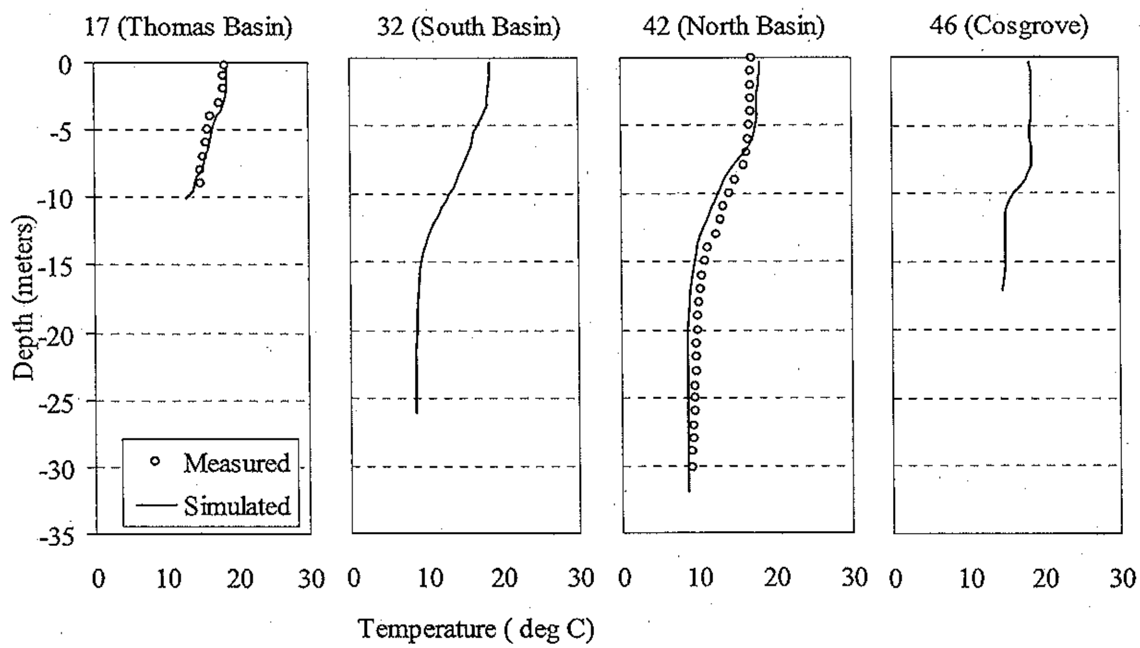

Figure 4.12 June 15, 2000 temperature profiles (°C)

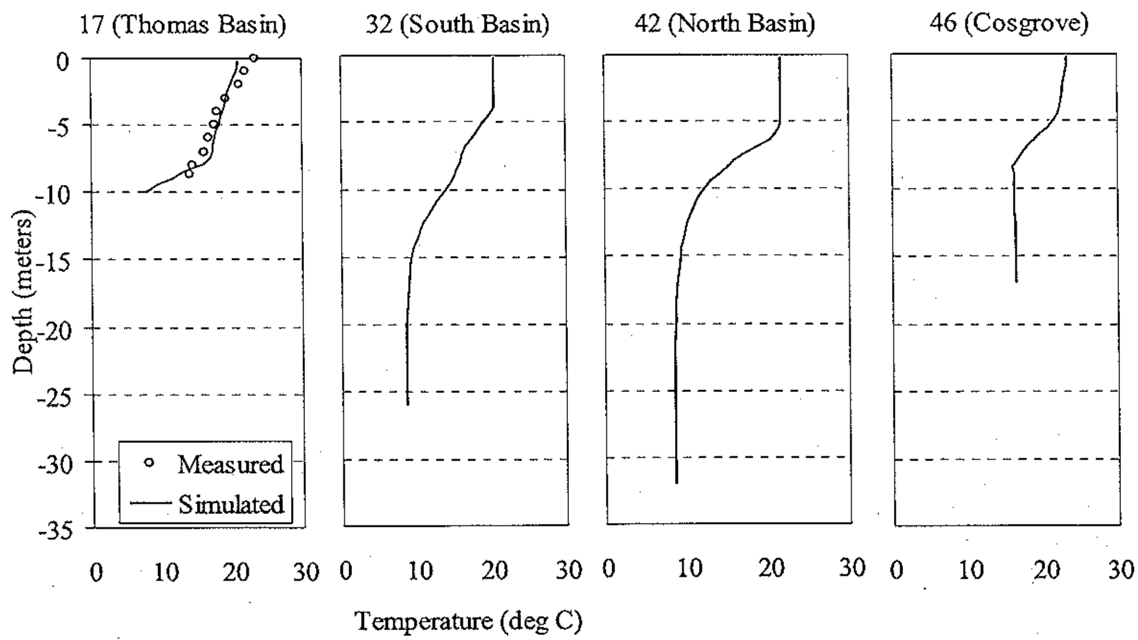

Figure 4.13 June 22, 2000 temperature profiles (°C)

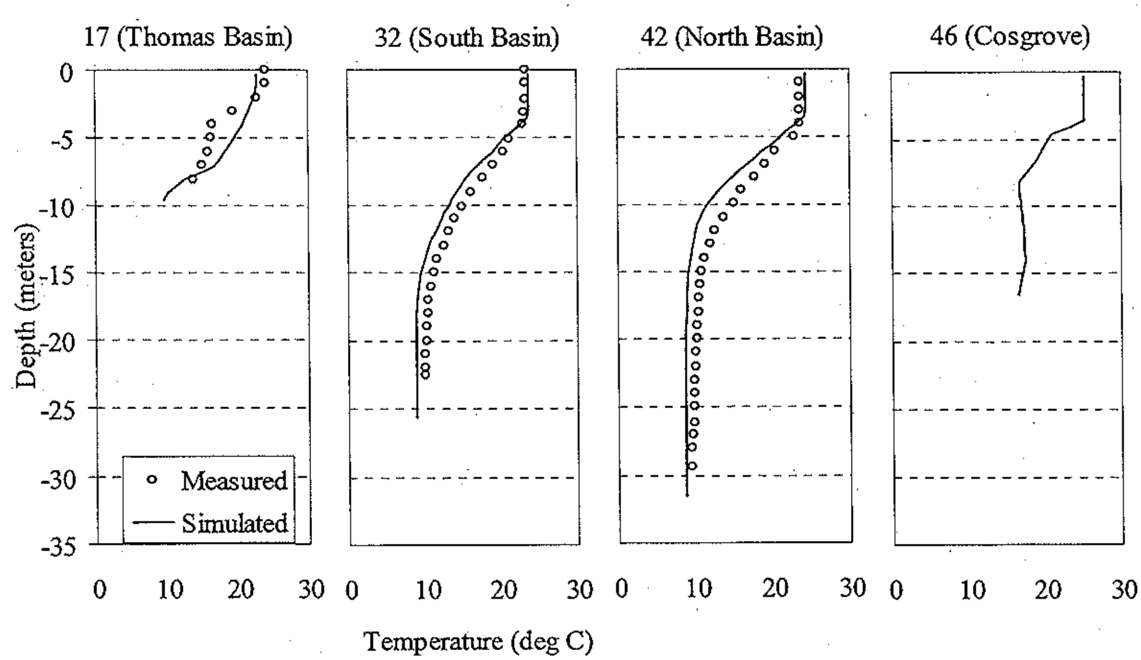

Figure 4.14 June 30, 2000 temperature profiles (°C)

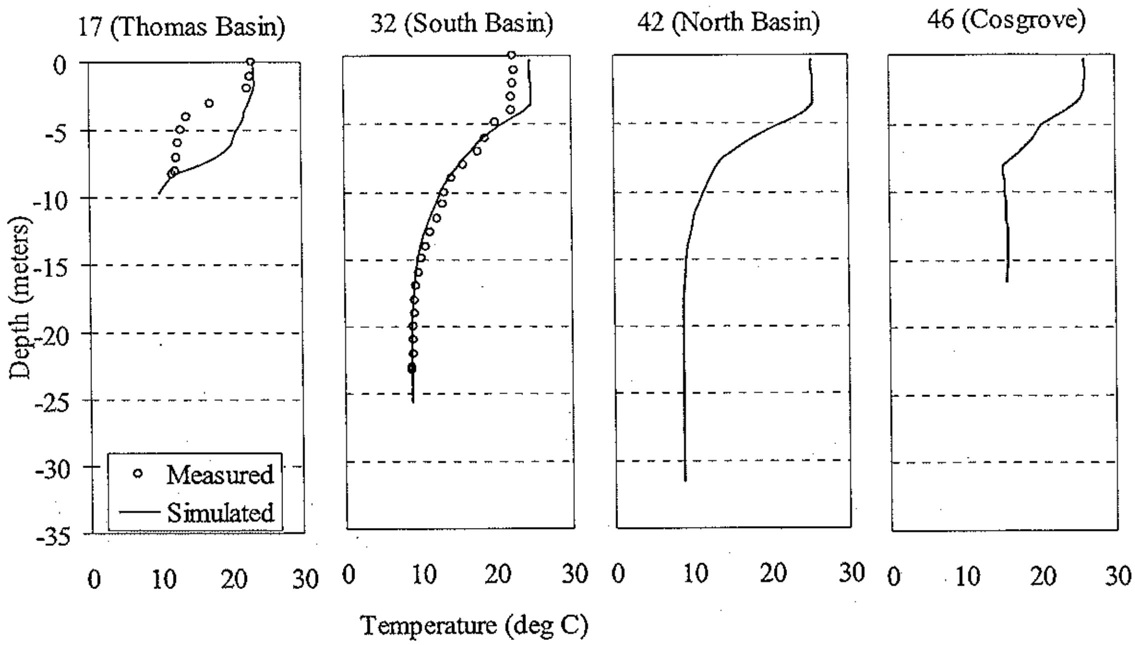

Figure 4.15 July 5, 2000 temperature profiles (°C)

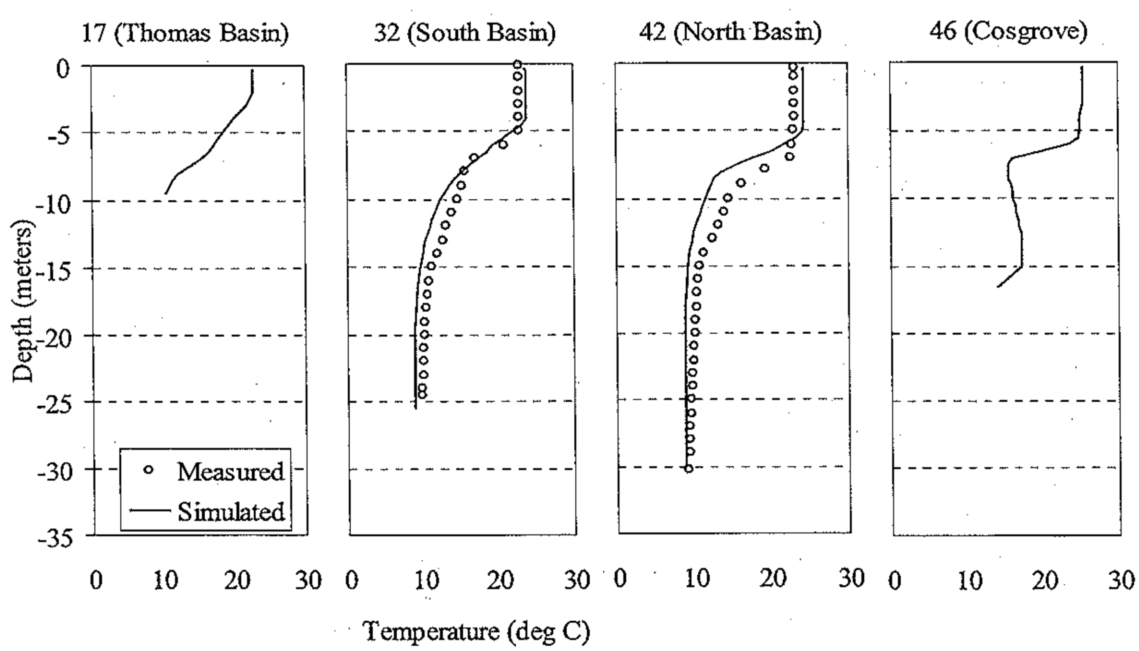

Figure 4.16 July 10, 2000 temperature profiles (°C)

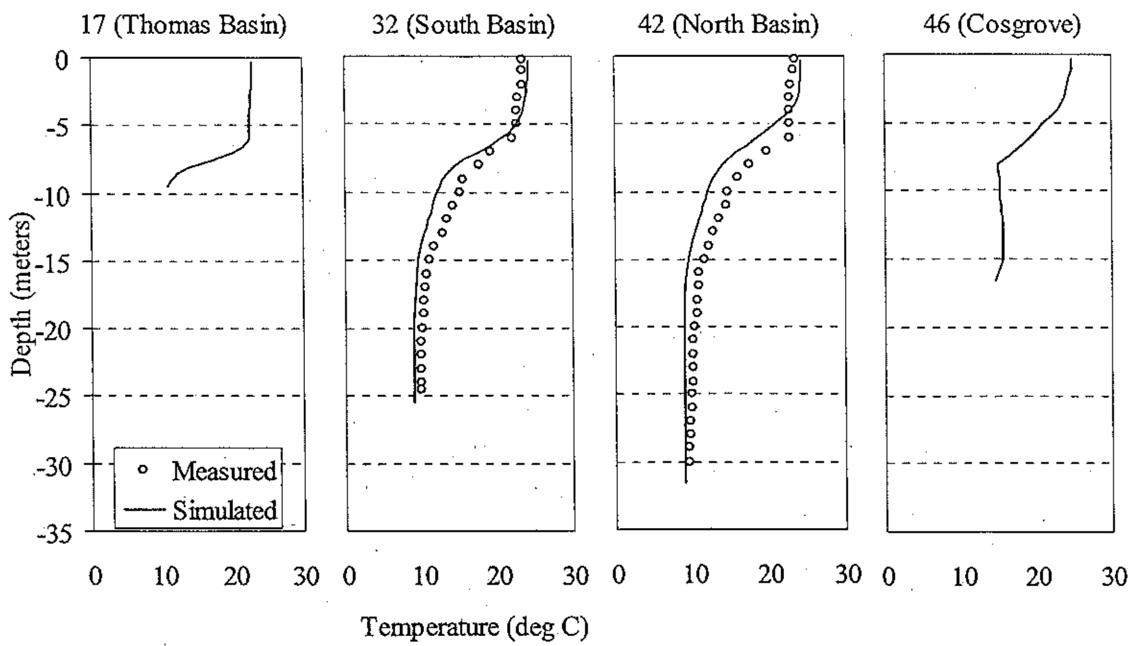

Figure 4.17 July 12, 2000 temperature profiles (°C)

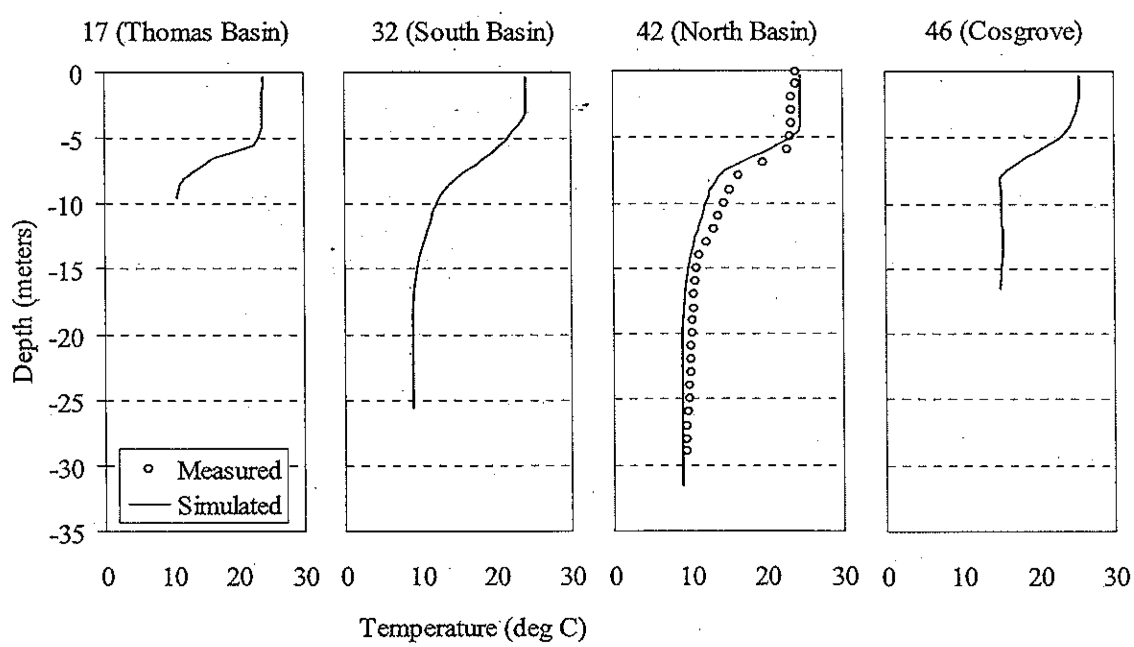

Figure 4.18 July 14, 2000 temperature profiles (°C)

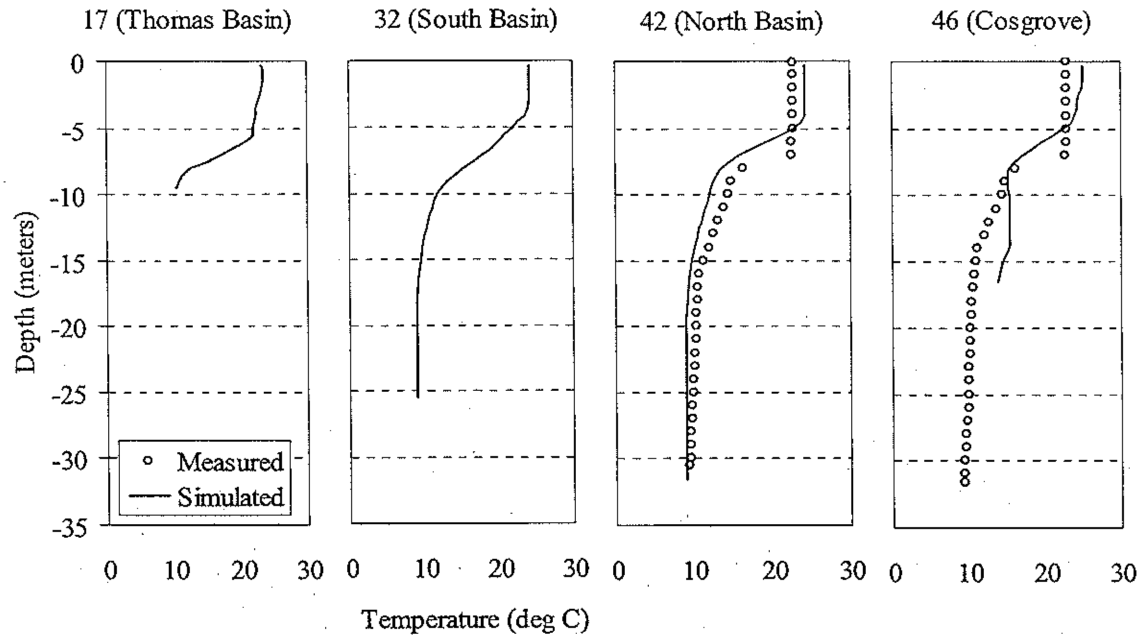

Figure 4.19 July 19, 2000 temperature profiles (°C)

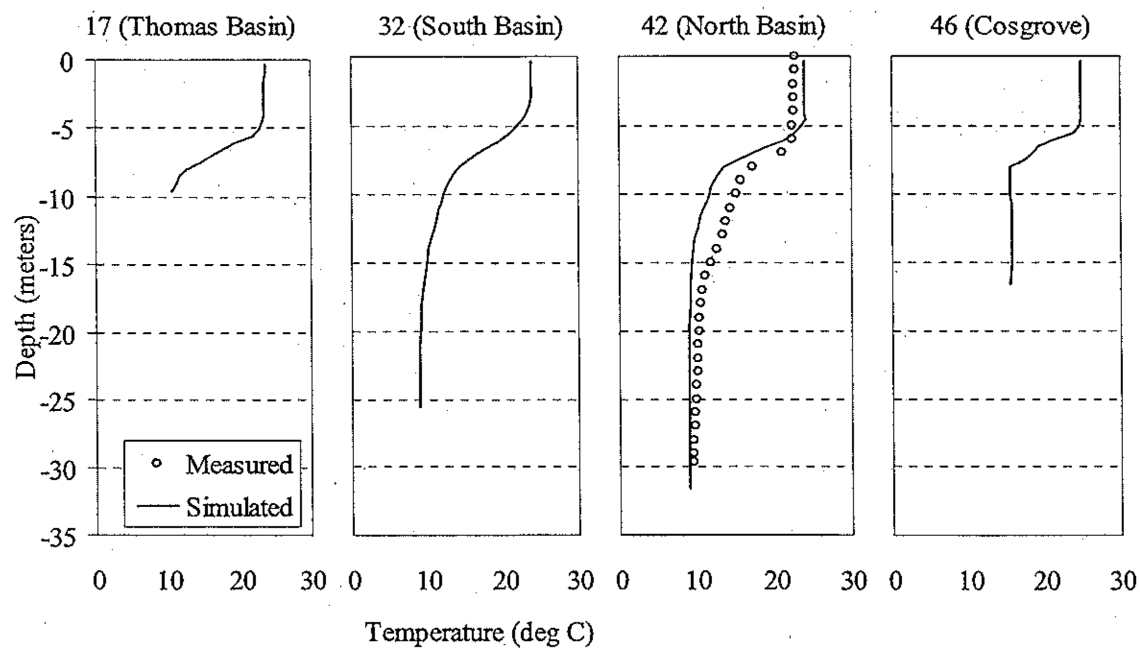

Figure 4.20 July 21, 2000 temperature profiles (°C)

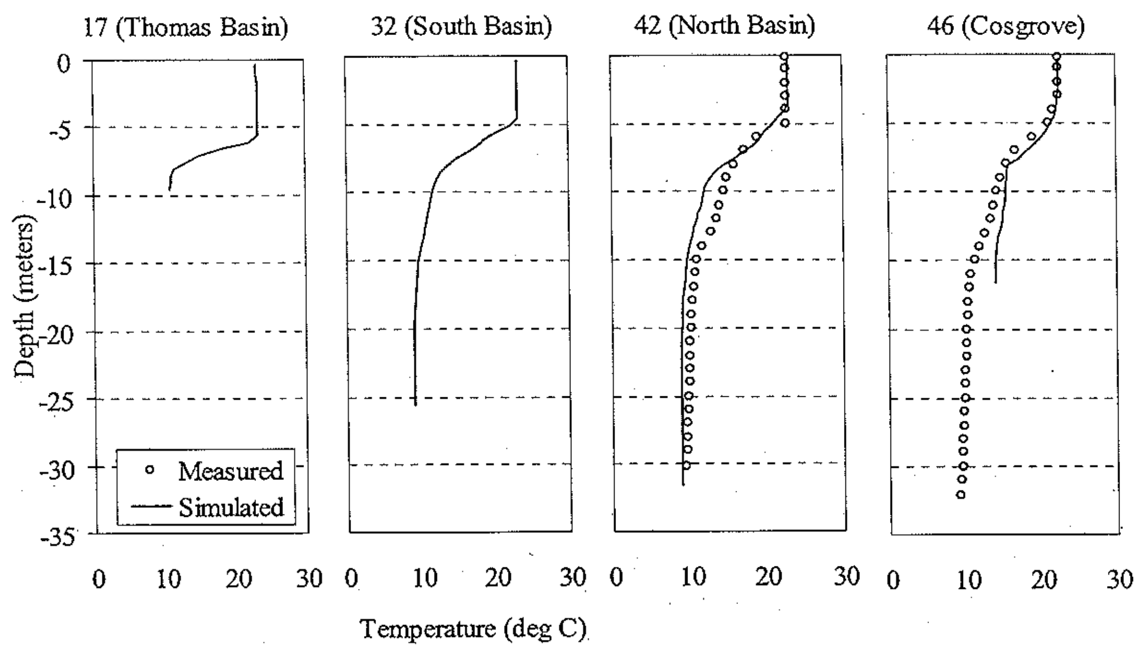

Figure 4.21 July 26, 2000 temperature profiles (°C)

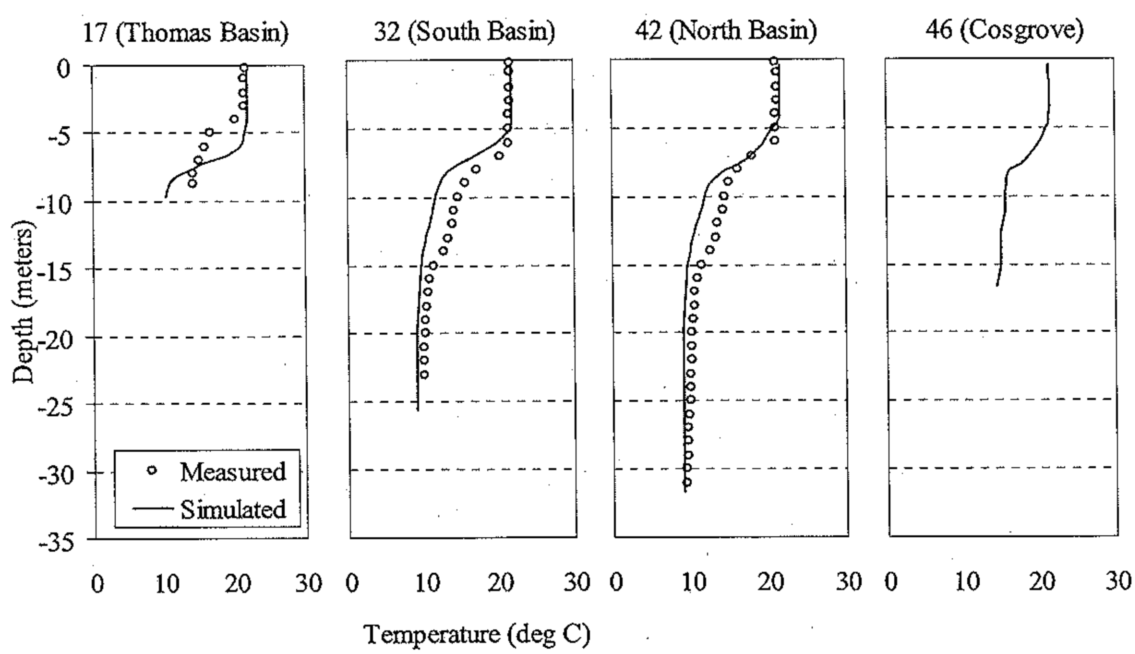

Figure 4.22 August 1, 2000 temperature profiles (°C)

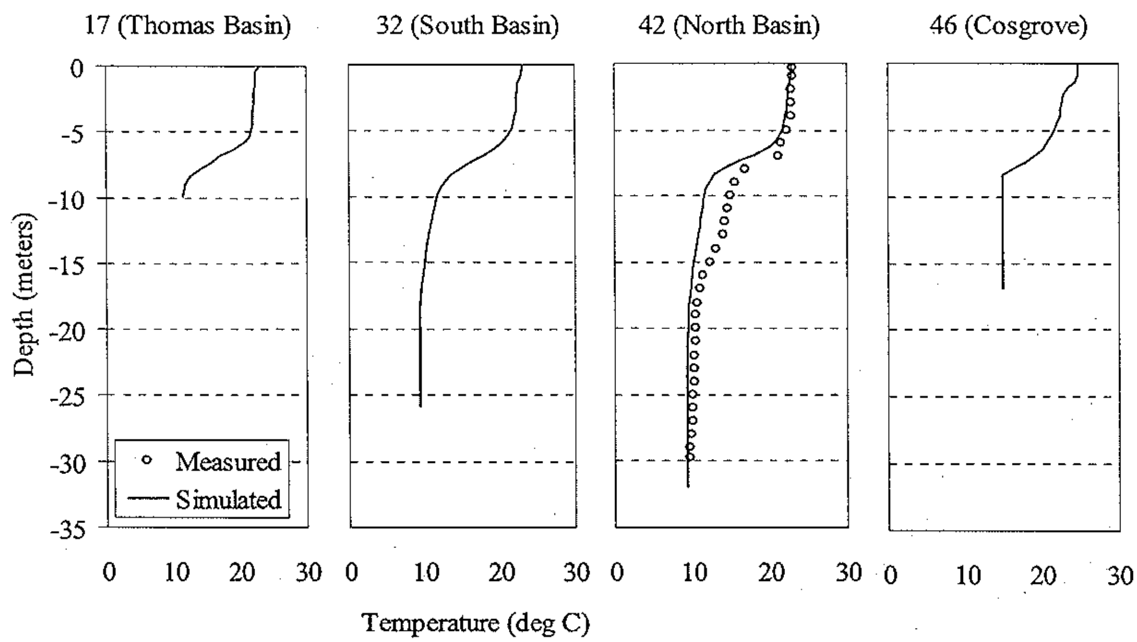

Figure 4.23 August 31, 2000 temperature profiles (°C)

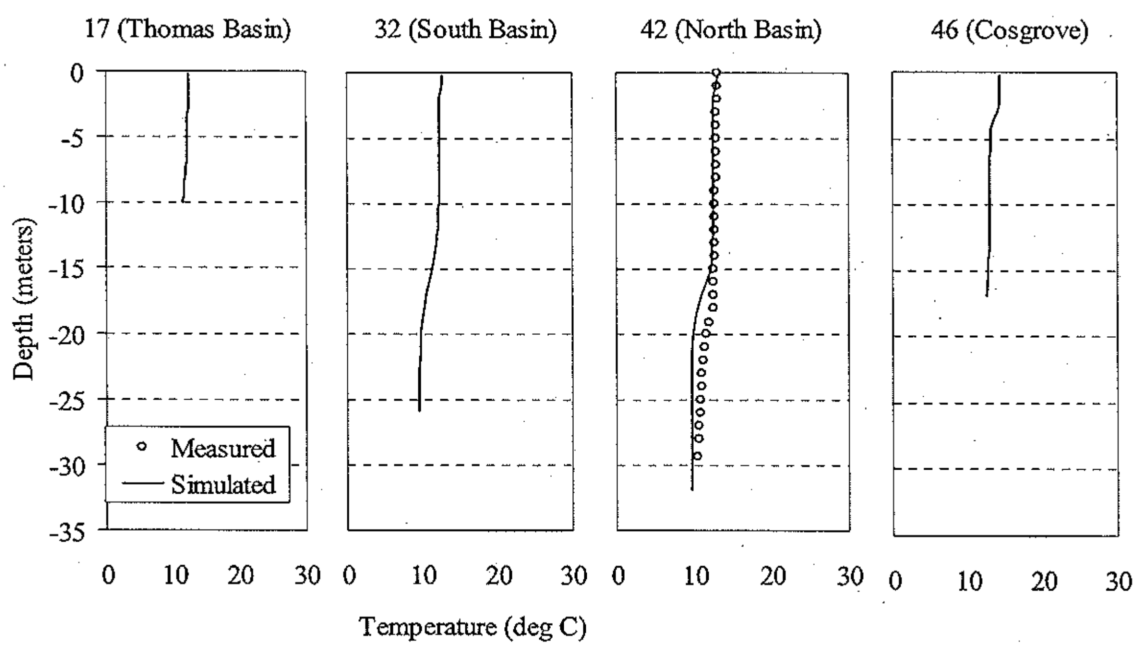

Figure 4.24 October 26, 2000 temperature profiles (°C)

4.1.3.2 Conductivity Profile Comparison

Figure 4.25 through Figure 4.39 presents comparisons between conductivity profiles measured in 2000 in Wachusett Reservoir and corresponding profiles predicted by CE QUAL W2. For North Basin and South Basin profiles, the modeled profiles are generally similar to the epilimnion and hypolimnion portions of the measured profiles. Modeled epilimnion conductivity values are generally within 5 μS/cm of measured values, with the largest deviation of 6 μS/cm occurring on August 31st. Deviation between modeled and measured hypolimnion conductivity values is also relatively small, within 7 μS/cm throughout the year. Generally, predicted epilimnion conductivity is higher than the modeled conductivity, while hypolimnion conductivity is generally lower.

It is notable that Quabbin Reservoir water is characterized by low conductivity, generally between 35 and 40 μS/cm. Wachusett Reservoir water is generally higher in conductivity. Conductivity of Wachusett Reservoir tributaries is generally between 100 and 500 μS/cm which, when diluted with Quabbin Transfer water, results in reservoir conductivity of 70 to 110 μS/cm.

In the metalimnion, presence of the Quabbin interflow, characterized by an interval of low conductivity, can first be seen in the measured profiles on July 5 in South Basin, followed by its appearance in the North Basin profile on July 19. These observations are, for the most part, reflected in the model predictions. Occasionally, CE QUAL W2 predicts a more shallow decrease in conductivity with a lower minimum (see Figure 4.37 Figure 4.38) than is observed in the measurements. For example, on August 31 at North Basin the minimum predicted conductivity of 59.7 μS/cm occurs at 8.5 meters depth, while a minimum conductivity of 69.3 μS/cm was recorded at 10 meters depth in the measured profile on that day. The largest deviations between measured and modeled metalimnion are 3 m and 11 μS/cm, although predictions are generally within 8 μS/cm and 2 m of the measurements.

As expected, modeled profiles for Thomas Basin do not describe profiles measured in that basin well. Modeled and measured profiles deviate by as much as 5 m and 35 μS/cm. This discrepancy occurs because the necessary assumption of lateral homogeneity required for modeling with CE QUAL W2 is incorrect for this portion of the reservoir. However, evidence suggests that the model is able to predict profiles measured at the outlet of the basin successfully, as presented in Section 4.3.3.

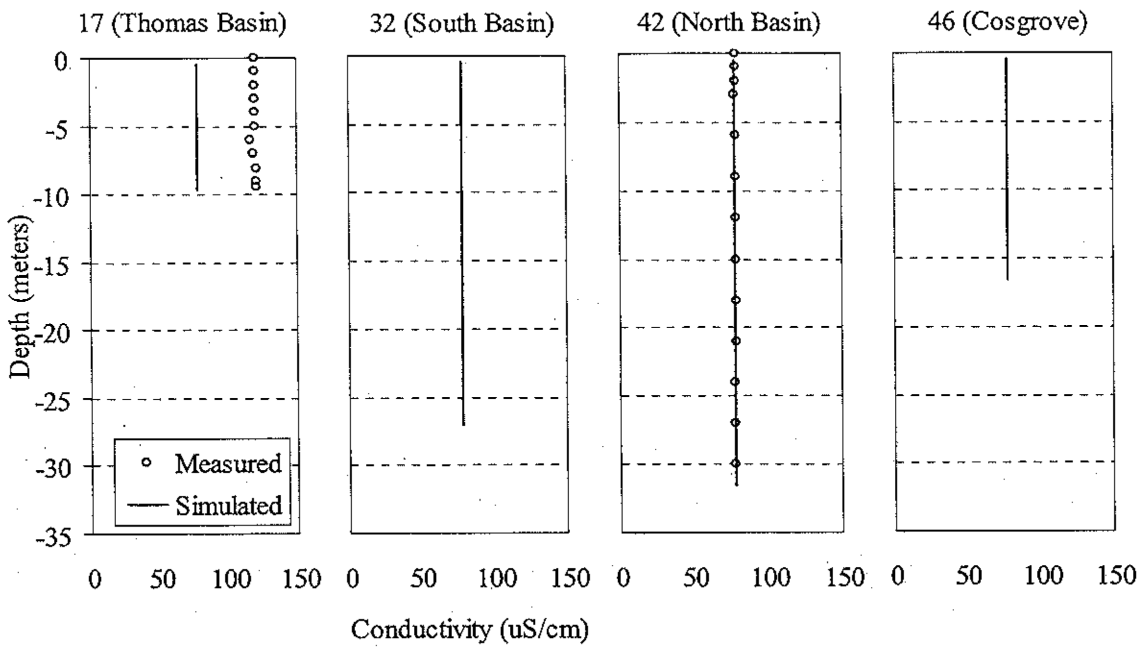

Figure 4.25 April 20, 2000 conductivity profiles (μS/cm)

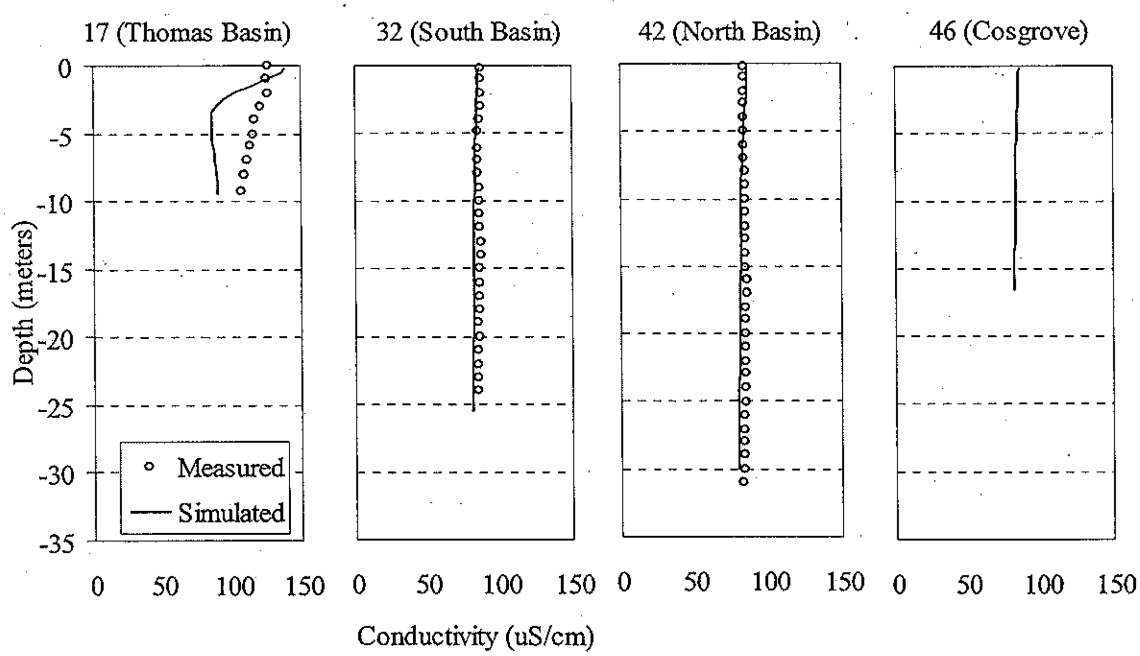

Figure 4.26 May 10, 2000 conductivity profiles (μS/cm)

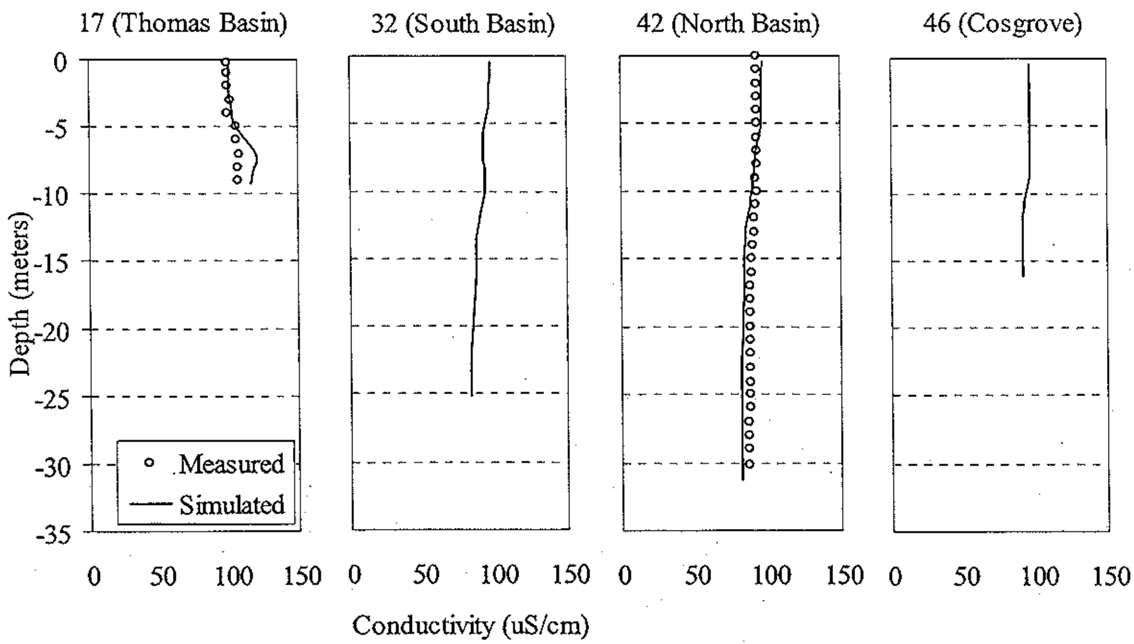

Figure 4.27 June 15, 2000 conductivity profiles (µS/cm)

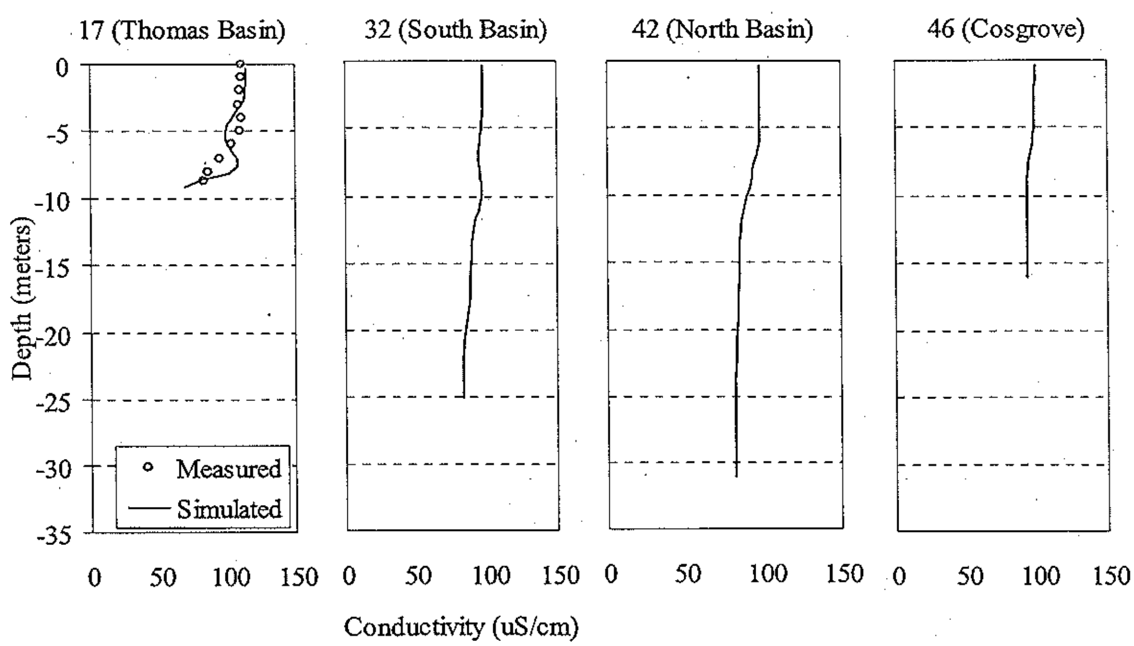

Figure 4.28 June 22, 2000 conductivity profiles (µS/cm)

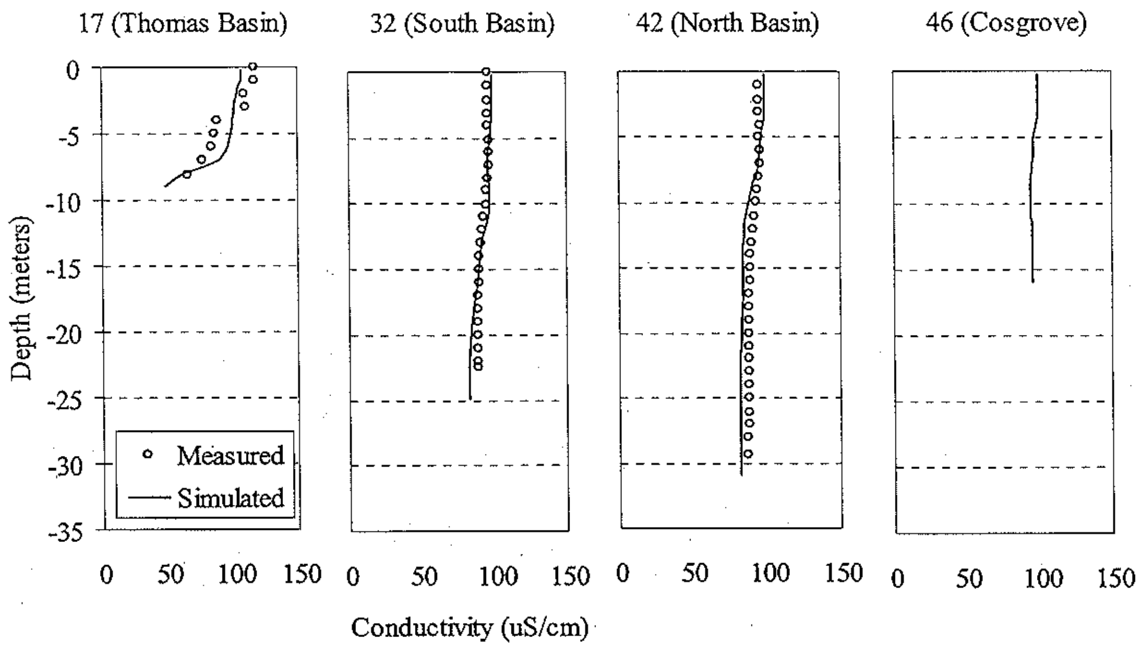

Figure 4.29 June 30, 2000 conductivity profiles (µS/cm)

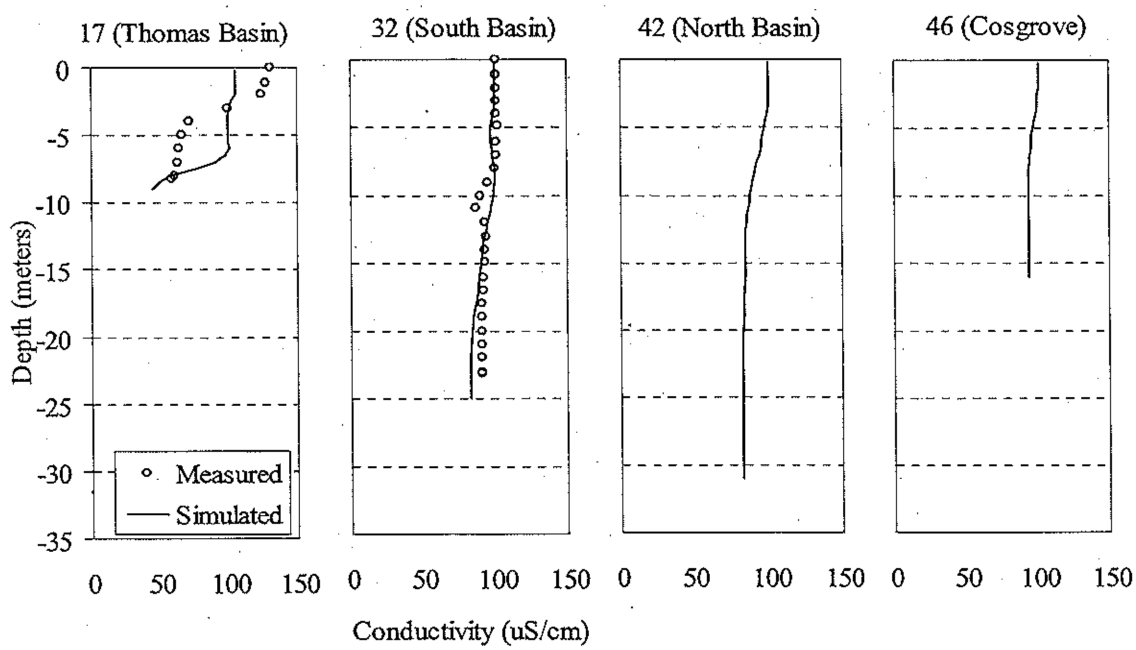

Figure 4.30 July 5, 2000 conductivity profiles (µS/cm)

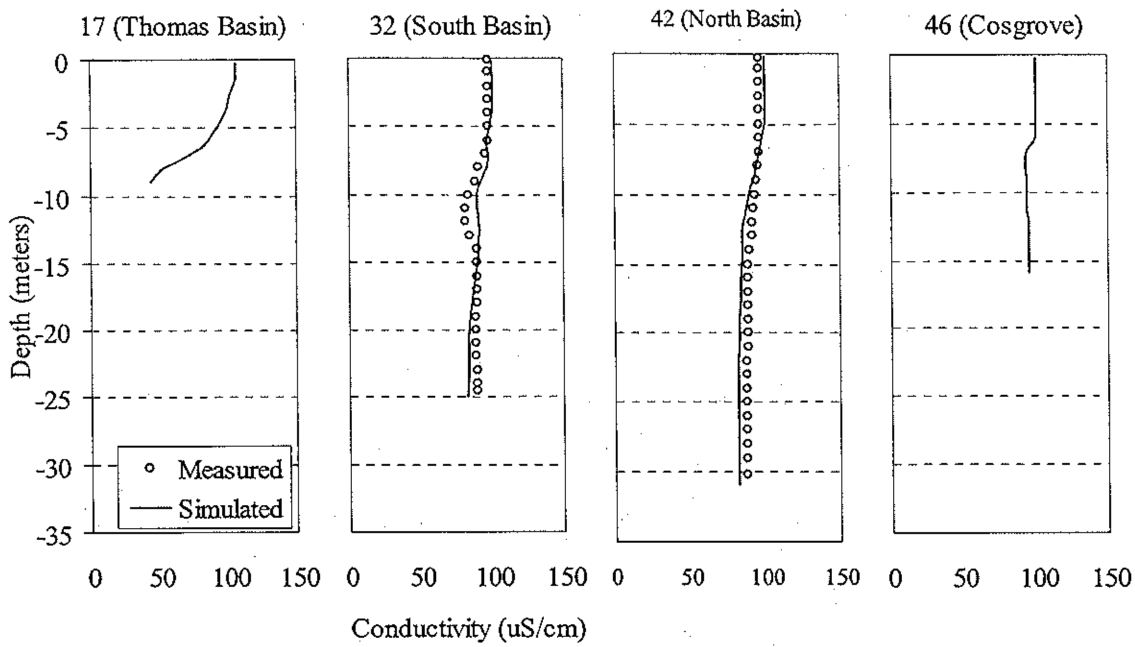

Figure 4.31 July 10, 2000 conductivity profiles (µS/cm)

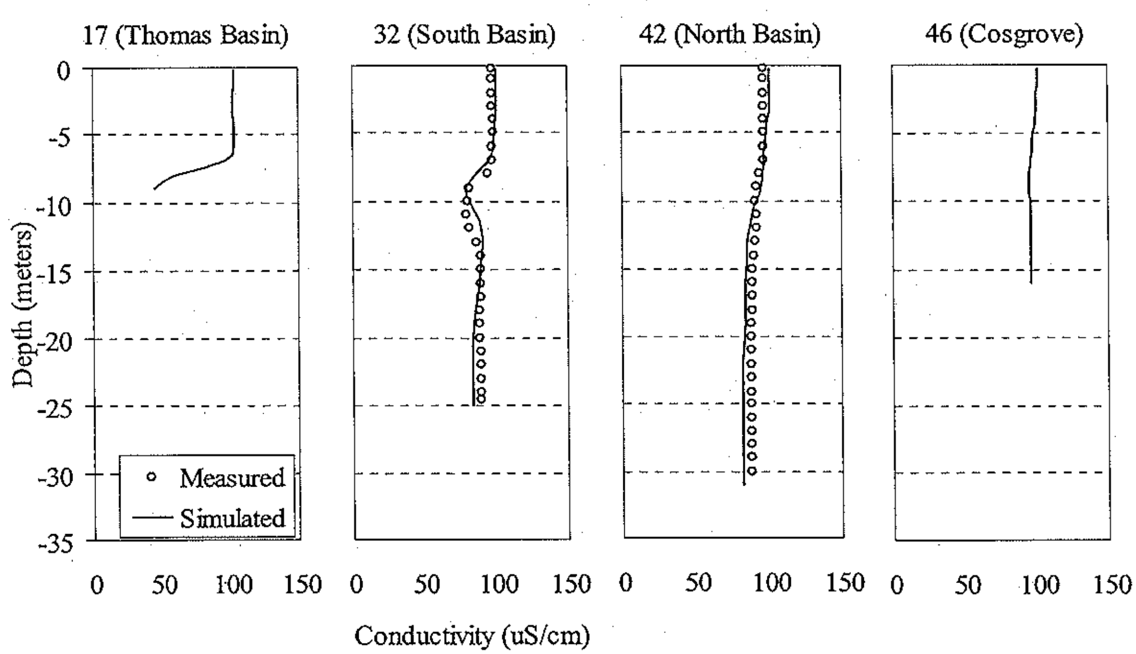

Figure 4.32 July 12, 2000 conductivity profiles (µS/cm)

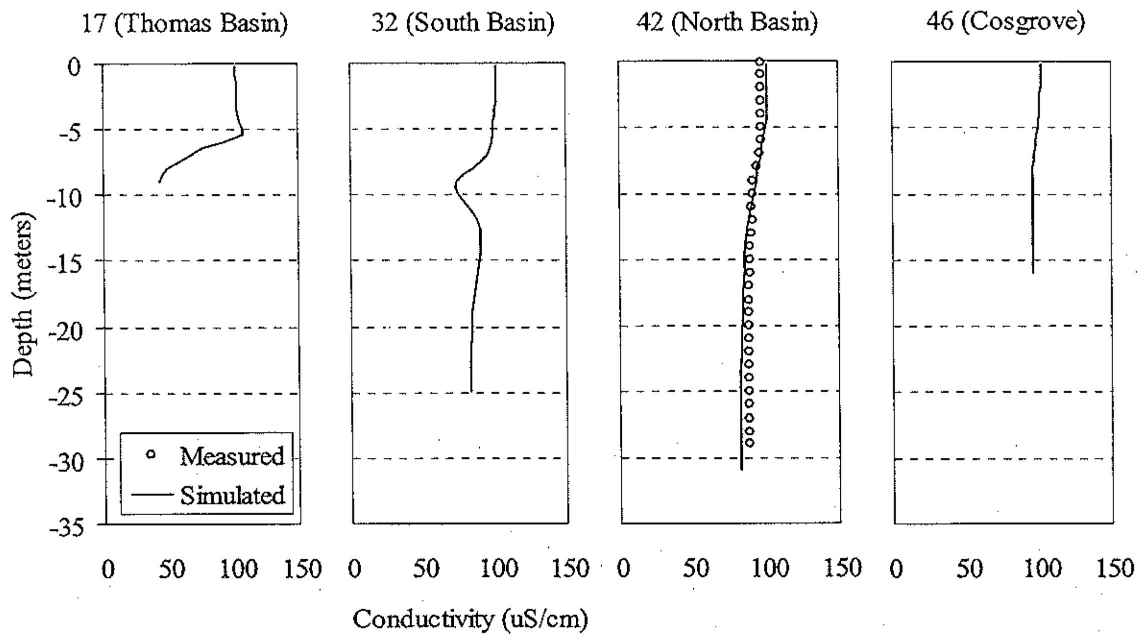

Figure 4.33 July 14, 2000 conductivity profiles (μS/cm)

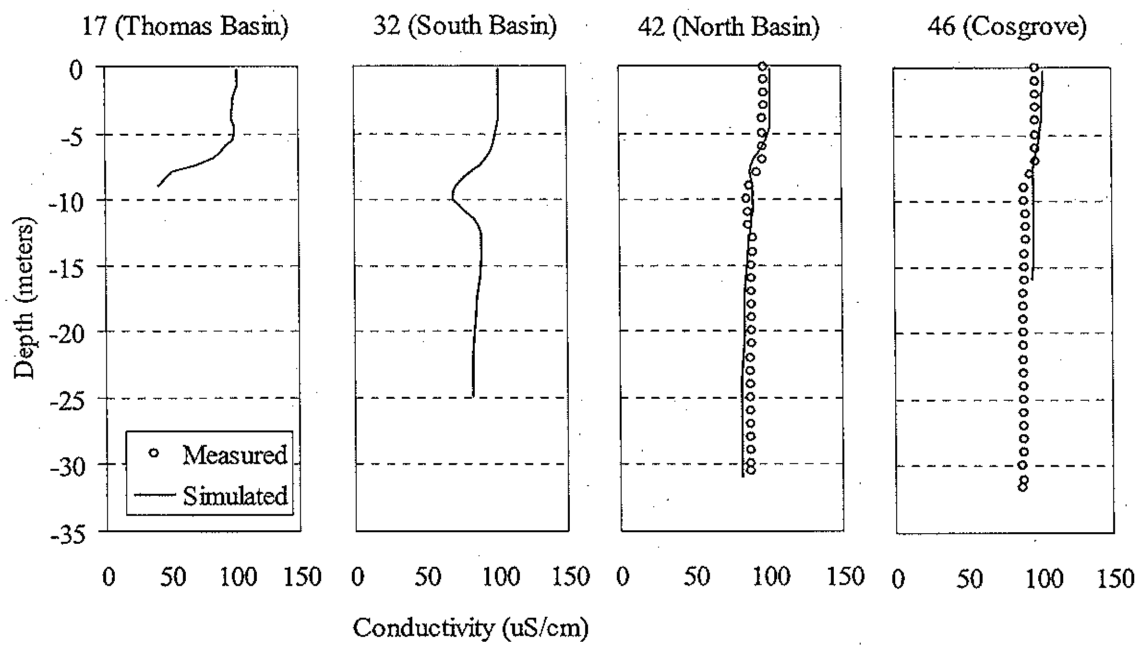

Figure 4.34 July 19, 2000 conductivity profiles (μS/cm)

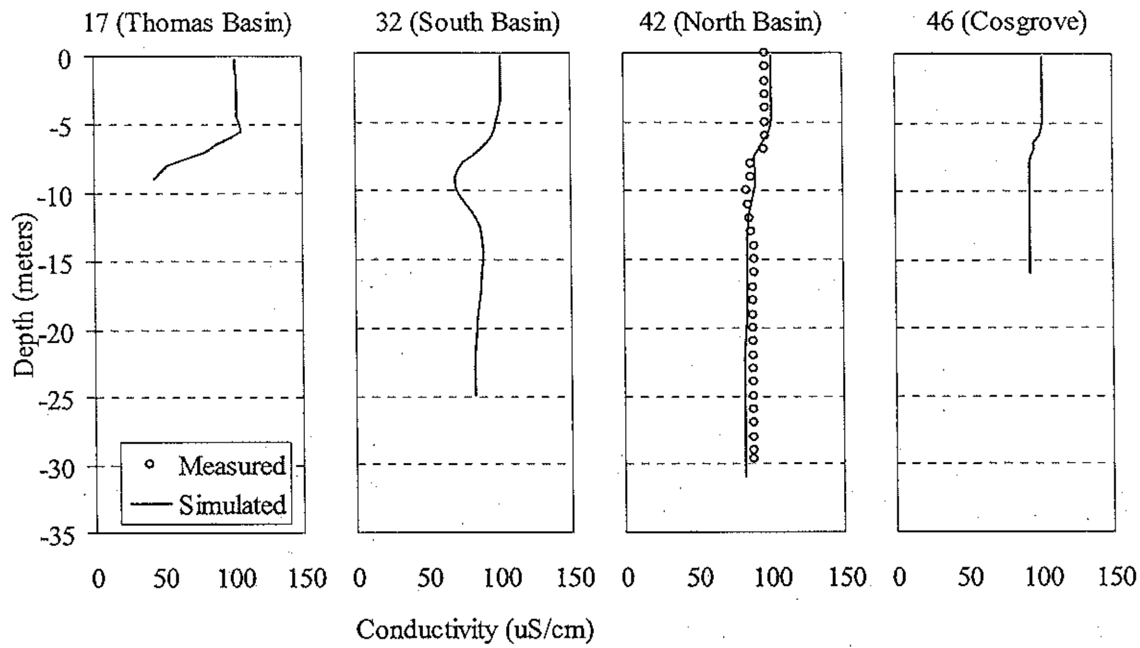

Figure 4.35 July 21, 2000 conductivity profiles (μS/cm)

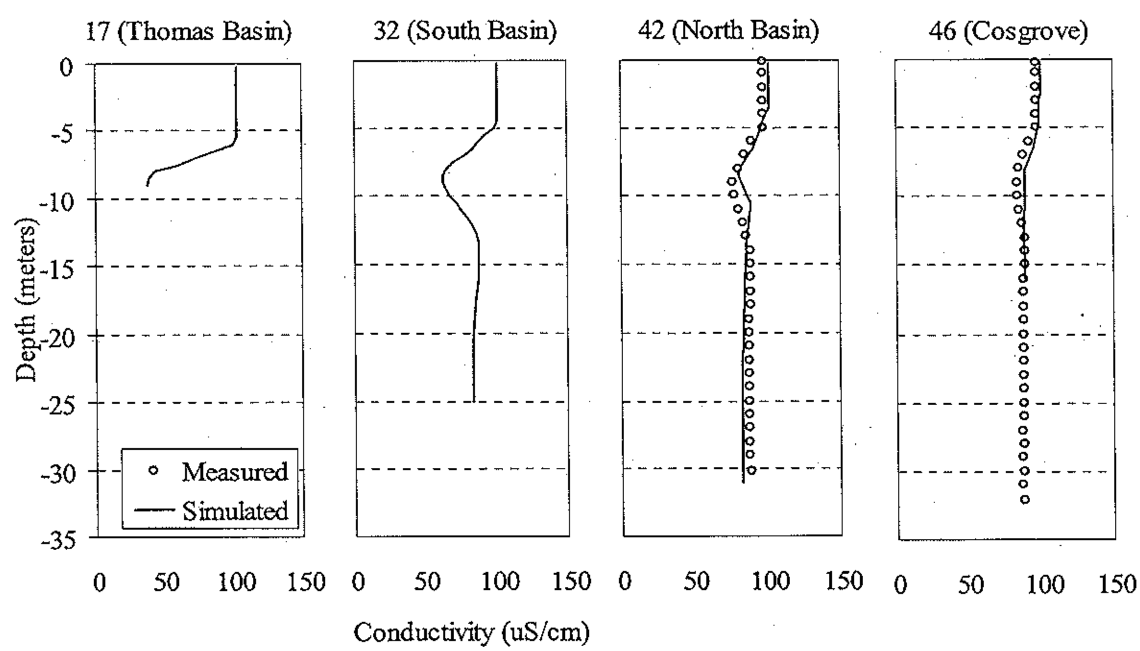

Figure 4.36 July 26, 2000 conductivity profiles (μS/cm)

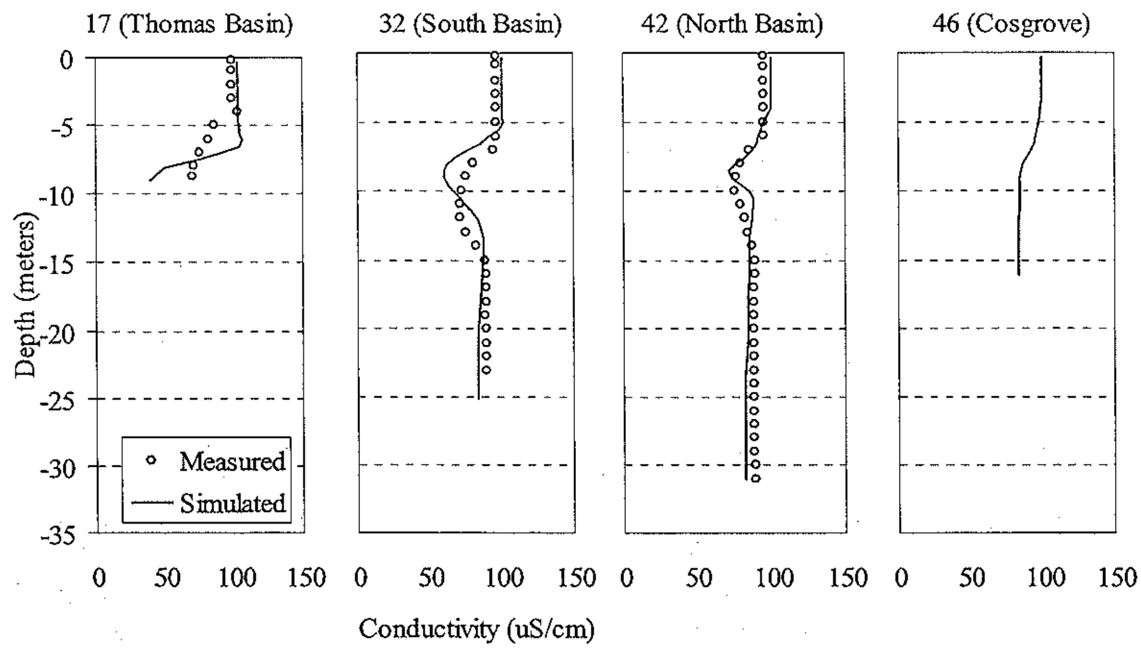

Figure 4.37 August 1, 2000 conductivity profiles (µS/cm)

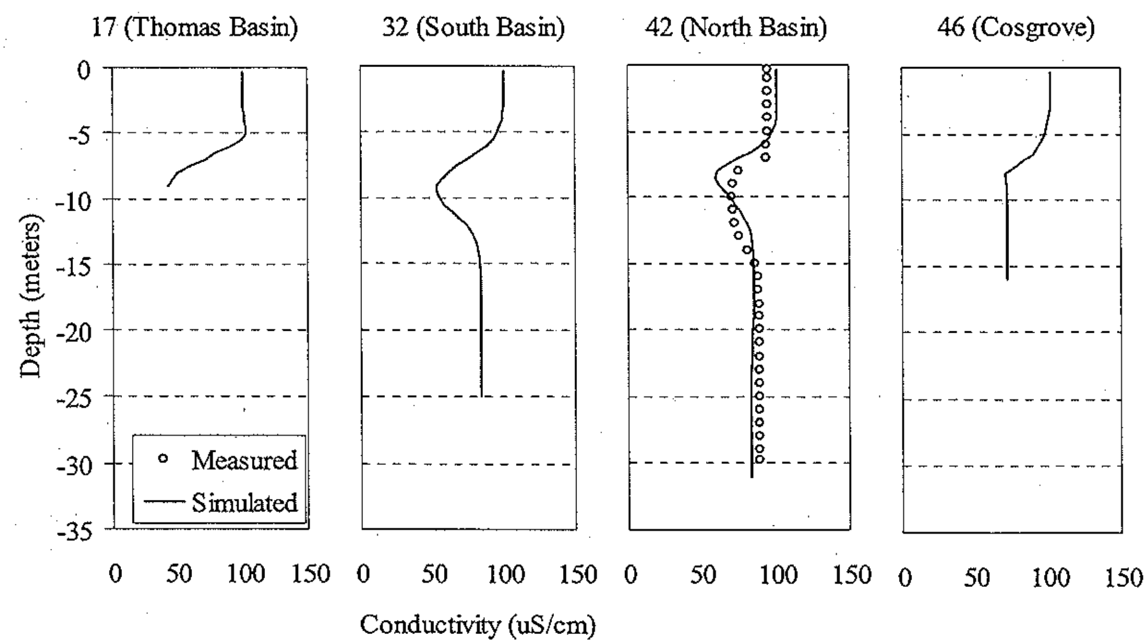

Figure 4.38 August 31, 2000 conductivity profiles (µS/cm)

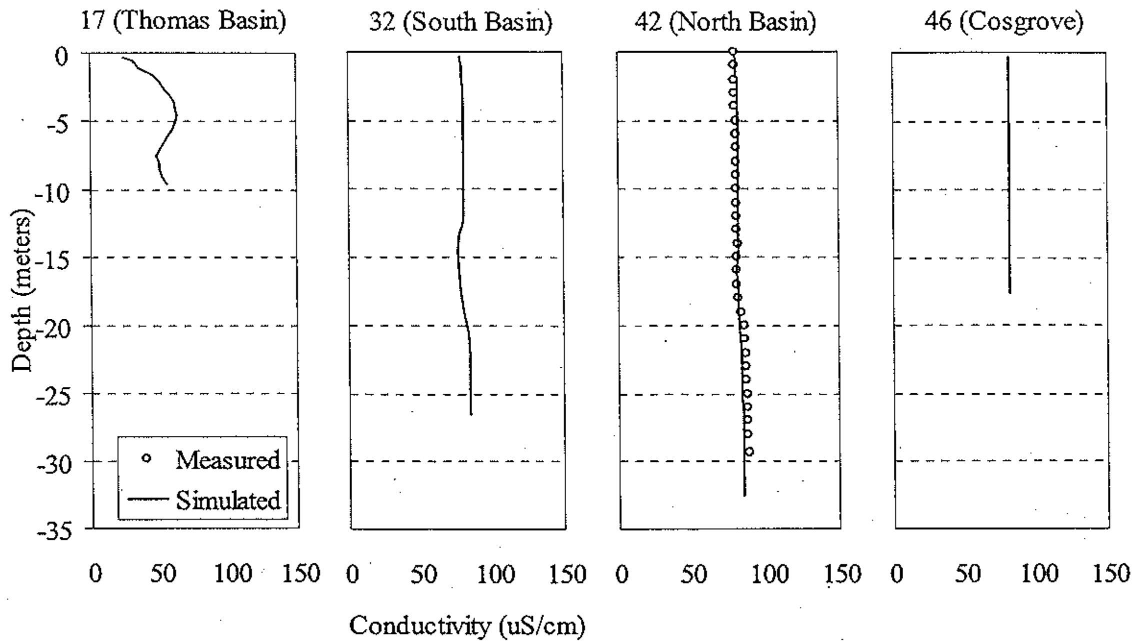

Figure 4.39 October 26, 2000 conductivity profiles (µS/cm)

4.2 Hydrodynamic Modeling – 2001

4.2.1 Reservoir Inflows

The hydrology of Wachusett Reservoir in 2001 followed typical patterns, where in a wet spring reservoir inflows are dominated by precipitation and tributary discharges, and in a dry summer inflows are dominated by Quabbin transfer. However, 2001 differed from 2000 in that it was an extremely dry year for New England. Less than 30 inches (76 cm) of rain fell at the Clinton meteorological stations near Wachusett Reservoir that year, as shown in Table 4.5.

Table 4.5 2001 precipitation statistics (Clinton station)

Total Precipitation, in (cm)	28.2 (72)
Average Daily Precipitation, in/day (cm/day)	0.08 (0.20)
Number of Days with Precipitation	86
Average on Days with Precipitation (cm/day)	0.33 (2.12)

Average monthly precipitation in 2001 was 2.35 in (6.0 cm), as compared to 3.74 in. (9.5 cm) in 2000; Figure 4.40 shows monthly precipitation in 2001 (a plot of daily precipitation may be seen in Ahlfeld *et al.* 2003a). In each of the months of April, May, June, August, October, and November, (totaling 6) less than 2 in (5.1 cm) of precipitation fell, while in 2000 there was only one month (October) that saw that little rain. Significant is that no precipitation was observed for two sequential months, in May and June. The influence of precipitation on tributary discharges is shown in Figure 4.41.

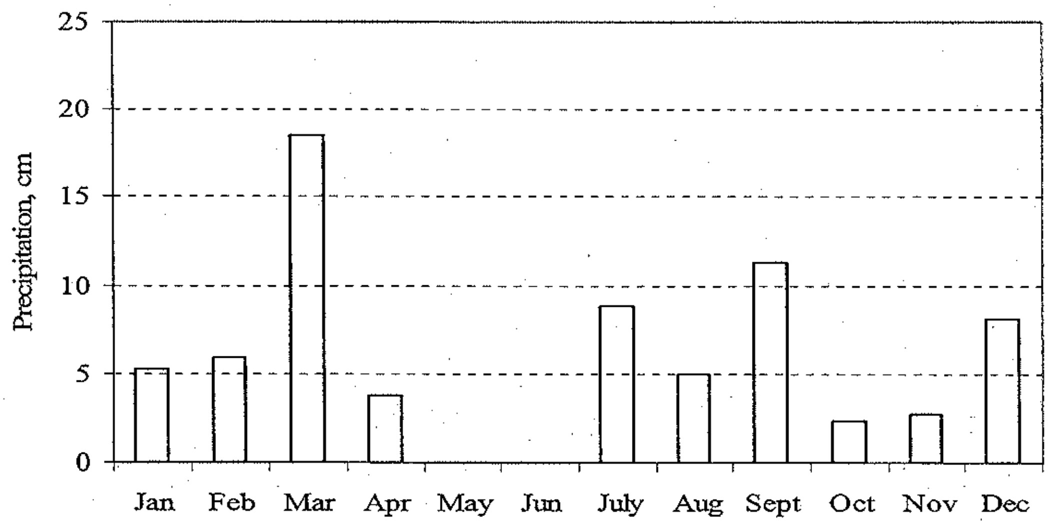

Figure 4.40 Total monthly precipitation for 2001

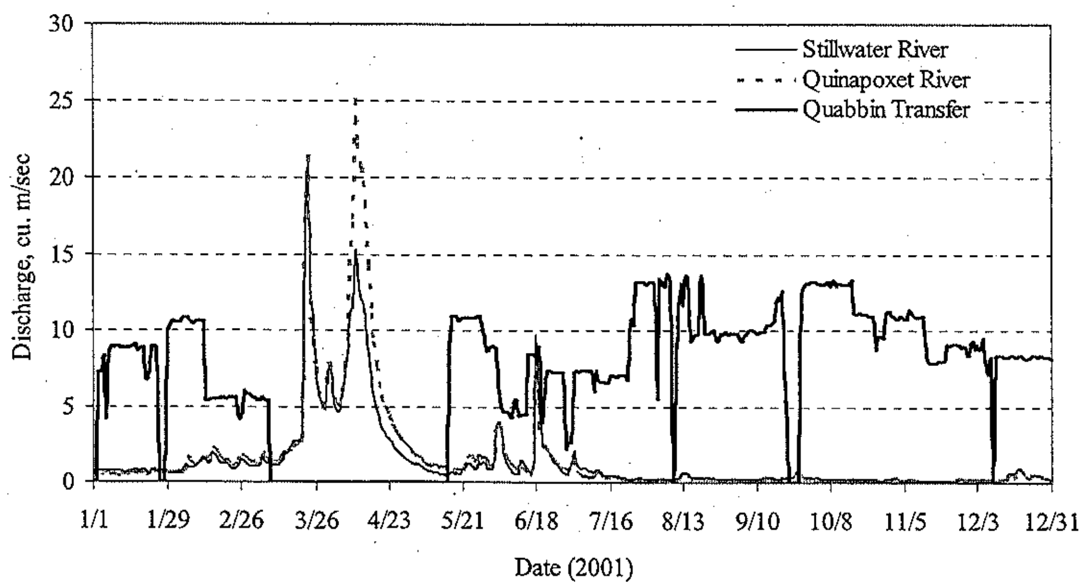

Figure 4.41 2001 hydrograph for Stillwater and Quinapoxet Rivers and Quabbin Transfer

The 7.3 in (18.5 cm) of precipitation that fell in March resulted in discharges for Stillwater and Quinapoxet Rivers that approached 25.5 m^3/s as can be seen in Figure 4.41. Early March through mid-May was the only time of year in which Quabbin transfer did not occur (Figure 4.41). During other periods, transfer generally ranged between 5 and 14 m^3/s, with an average daily transfer of 9.0 m^3/s for days when transfer was occurring, or 7.0 m^3/s for the year. Although precipitation returned in July, Stillwater and Quinapoxet discharges generally did not become larger than 0.28 to 0.42 m^3/s base flow levels, possibly because precipitation was refilling impoundments and replenishing soil moisture.

Figure 4.42 presents the relative contributions of inflows to Wachusett in 2001. Quabbin transfer accounted for 7.2% more of the annual water budget than in 2000 (58.7 vs. 51.5%). It is also notable that Quinapoxet contributed more water than Stillwater in 2001, while the reverse was true in 2000. Proportions of flow from direct runoff and the minor tributaries are necessarily the same as in 2000, as they are estimated based on the ratio of watershed size to that of Stillwater. The impact of inflows on water surface elevation is shown in Figure 4.43, while the impact of outflows is shown in Figure 4.44

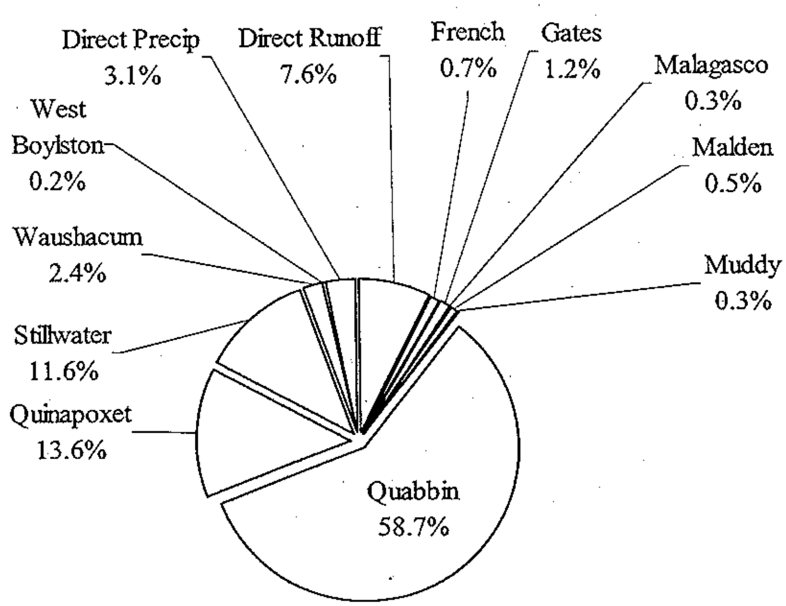

Figure 4.42 Relative contribution of 2001 inflows to Wachusett Reservoir

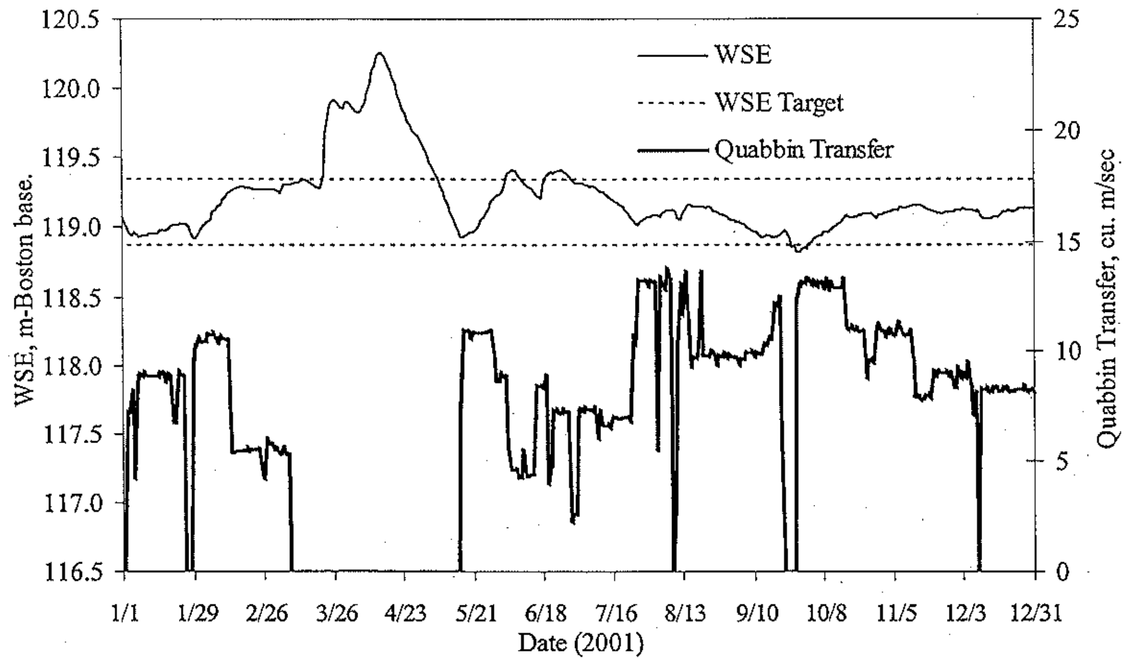

Figure 4.43 Impact of Quabbin Transfer on water surface elevation for 2001

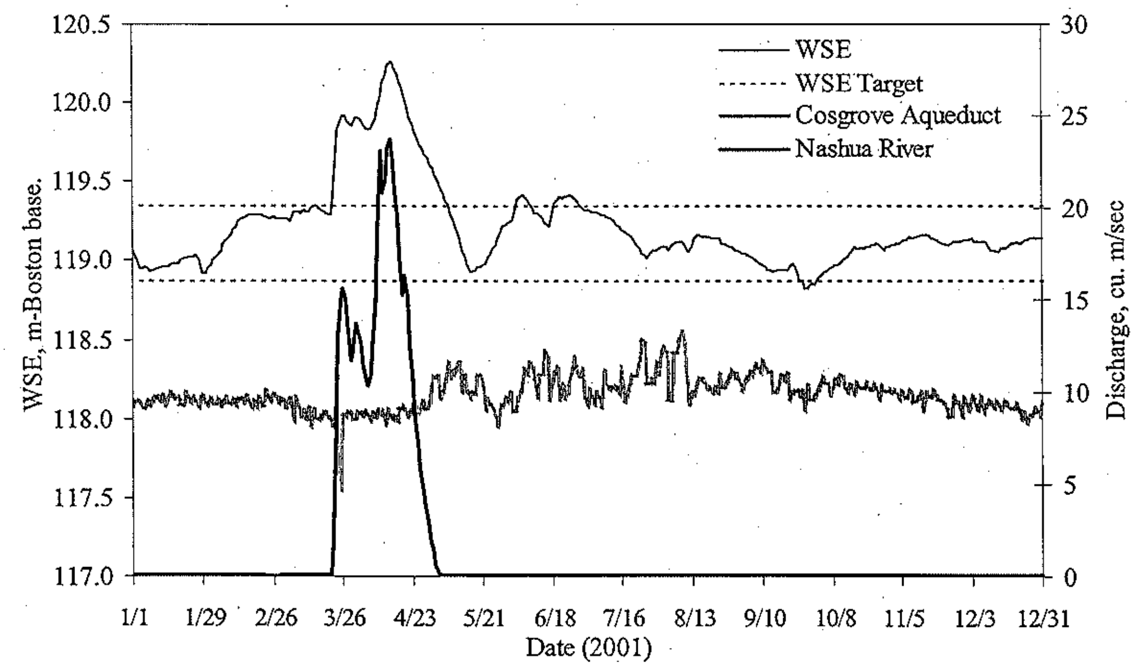

Figure 4.44 Major water losses and water surface elevation for 2001

The rapid increase in WSE that resulted from large quantities of precipitation and tributary discharge in March and April reverses to a rapid decrease following spilling to the Nashua River. The WSE decrease ends just before the WSE reaches the lower target of 118.9 m by the start of Quabbin transfer at a rate of approximately 11 m^3/s. Since demand for the following two weeks (May 17 through May 31) averages 9.6 m^3/s, the 1.4 m^3/s surplus causes WSE to rise and approach the upper limit. Quabbin Transfer is then varied to maintain WSE for the remainder of the year. It is interesting to note the brief cessation of transfer from September 22 through September 26, during the largest precipitation event of the year where 6.9 cm fell on September 25. Operators were most likely expecting large tributary discharges to meet demand, but these never occurred (Stillwater discharge increased from 0.28 to 0.76 m^3/s while Quinapoxet increased from between 0.14 and 0.28 to 0.68 m^3/s) and it became necessary to commence 13.1 m^3/s of transfer from Quabbin.

4.2.2 Reservoir Losses

As in 2000, the principal discharges from Wachusett Reservoir in 2001 consisted of demand discharge to the Cosgrove intake and spilling plus base flow to the Nashua River. Despite the relatively small quantity of precipitation in 2001, spilling to Nashua accounted for a greater percentage of losses than in 2000 (13% as compared to 6.6%). Evaporation for the two years was quite consistent in terms of percentage (3.6% in 2000 and 3.8% in 2001) as were the other minor losses (dike seepage, Wachusett Aqueduct, withdrawals by towns). Figure 4.45 presents the relative influence of each loss on the reservoir water budget in 2001.

Figure 3.4 presents monthly total evaporation for 2001, computed using two methods. Figure 4.46 presents the daily average evaporation rate as used in this study. When comparing this figure to Figure 4.6 presenting the average daily evaporation rate for 2000, it is apparent that evaporative trends for the two years are similar with higher rates in the warmer months. In 2001, the period of largest evaporative losses is centered on the early summer, in late June and early July, while in 2000 evaporation rates were more consistent starting during this time and continuing into November and December.

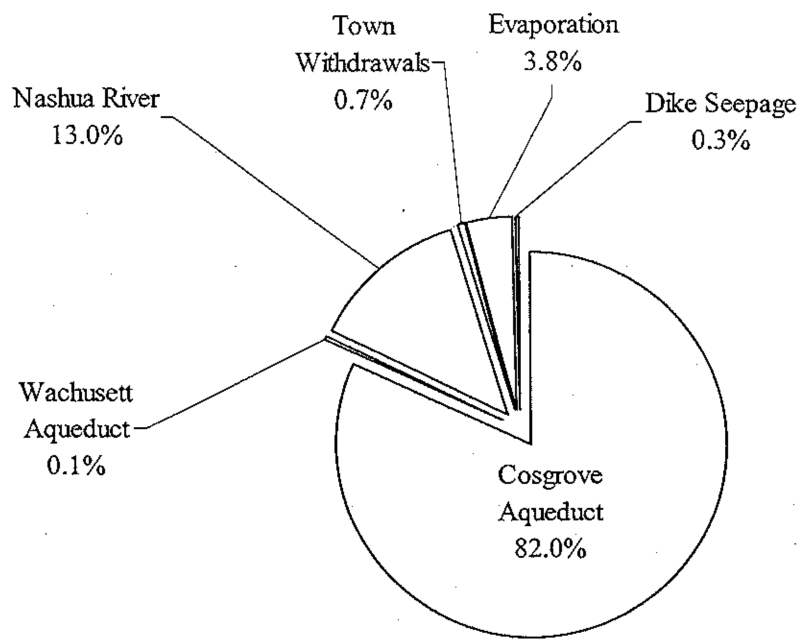

Figure 4.45 Relative quantity of water lost to the major sinks of Wachusett Reservoir in 2001

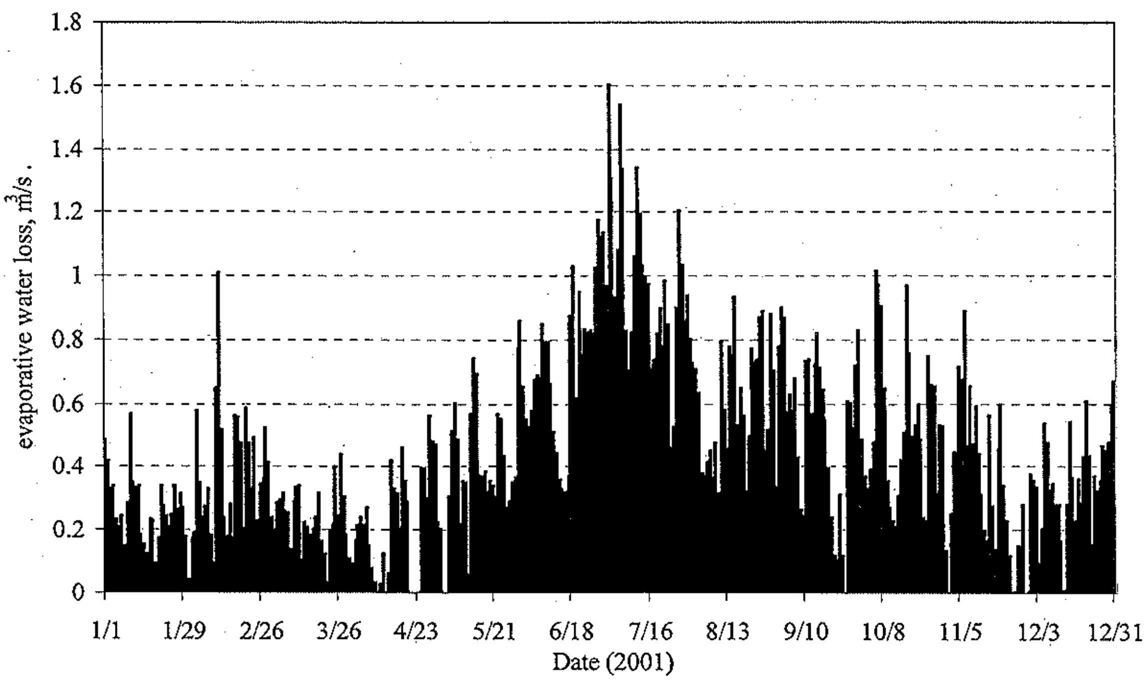

Figure 4.46 Average daily evaporation rates as estimated for 2001

4.2.3 Calibration Results

Implementing SOLVER to balance the required inflow and outflow data presented in Section 4.2.1 resulted in inflow calibration factors presented in Table 4.6.

Table 4.6 Summary of 2001 and historic calibration factors

Inflow	Range (1994-2000)	2001
Quabbin	1.0-1.19	0.97
Stillwater	0.70-1.28	1.16
Quinapoxet	0.82-1.30	1.13
Waushacum	1.11-1.65	1.16
Direct Runoff	1.11-1.62	1.16
Malden	1.00-1.35	1.16
W. Boylston	1.11-1.35	1.16
Gates	1.11-2.00	1.16
Muddy	1.00-1.35	1.16
Malagasco	1.00-1.35	1.16
French	1.00-1.35	1.16

All but one of the calibration factors determined for 2001 are within the range of 1994 through 2000 values. The outstanding value of 0.97 for Quabbin transfer is close to the range of +/- 2% as expected by DCR, so it should be considered reasonable. The Quinapoxet calibration factor results in less of an adjustment of discharges for that tributary than for Stillwater. Data from the Quinapoxet gage is designated as 'good' quality by USGS while the Stillwater gage is rated 'fair' or 'poor,' which is consistent with the results of this calibration. The relative inaccuracy of Stillwater data results from periodic beaver activity near the gage that causes unpredictable impounding.

Table 4.7 provides runoff coefficients determined by this analysis. All the coefficients with the exception of that for Quinapoxet are at the extreme upper end of the range for 1994 through 2000. The small response of the Stillwater and Quinapoxet Rivers following the September storm event as discussed in Section 4.2.1 suggests that runoff coefficients vary throughout the year. It is important to note that the Quinapoxet runoff coefficient for 2001 was calculated using a method presented in Ahlfeld *et al.* (2003a).

Table 4.7 Summary of 2001 and historic unitless runoff coefficients

Inflow	Range (1994-2000)	2001
Stillwater	0.24-0.75	0.75
Quinapoxet	0.54-0.94	0.78
Waushacum	0.37-0.75	0.75
Direct Runoff	0.37-0.75	0.75
Malden	0.37-0.75	0.75
W. Boylston	0.37-0.75	0.75
Gates	0.37-0.75	0.75
Muddy	0.37-0.75	0.75
Malagasco	0.37-0.75	0.75
French	0.37-0.75	0.75

Table 4.8 presents statistics describing the water budget calibration results. The 2001 calibration is characterized by smaller root mean square (RMS) error than for 2000, as well as a smaller average absolute deviation and does not at any time exceed the +/- 0.15 m maximum deviation criteria. The higher quality of this calibration is likely a result of the higher proportion of Quabbin transfer in the 2001 water budget. Thus, a larger proportion of the annual inflow is measured with an accurate gage (venturi meter) as opposed to a stream gage where high variability exists, or through estimation.

Table 4.8 Selected 2001 statistics describing water budget calibration results

Statistic	Value
RMS error, m	0.063
Average Absolute Deviation, m	0.051
Maximum Positive Deviation, m	0.131
Maximum Negative Deviation, m	-0.150
No. Days Greater then +/- 0.15 m Deviation	0

Figure 4.47 presents the measured daily and predicted Wachusett WSE results. Figure 4.48 presents a time series plot of deviation of the predicted from the measured values. There is no noticeable trend in deviation throughout the year.

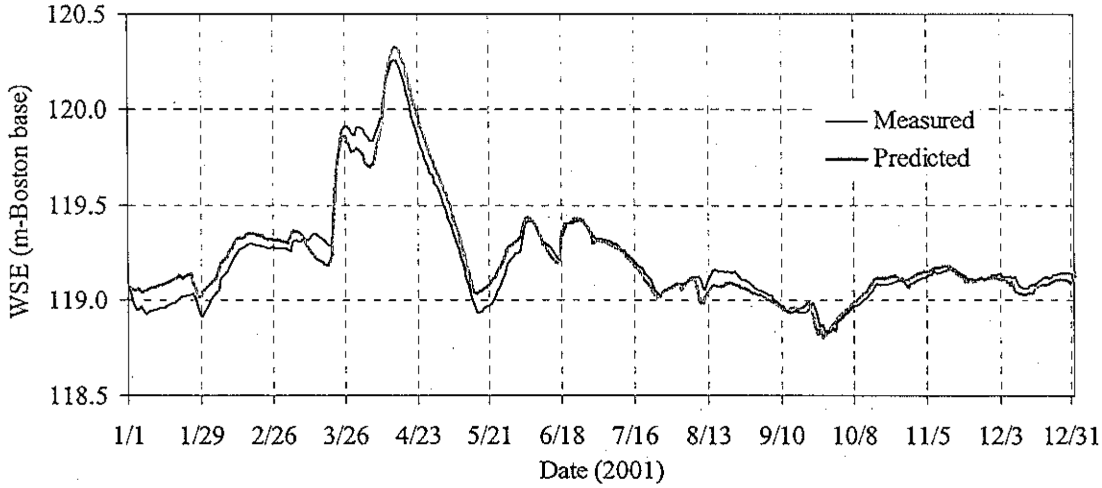

Figure 4.47 Wachusett Reservoir water budget calibration for 2001

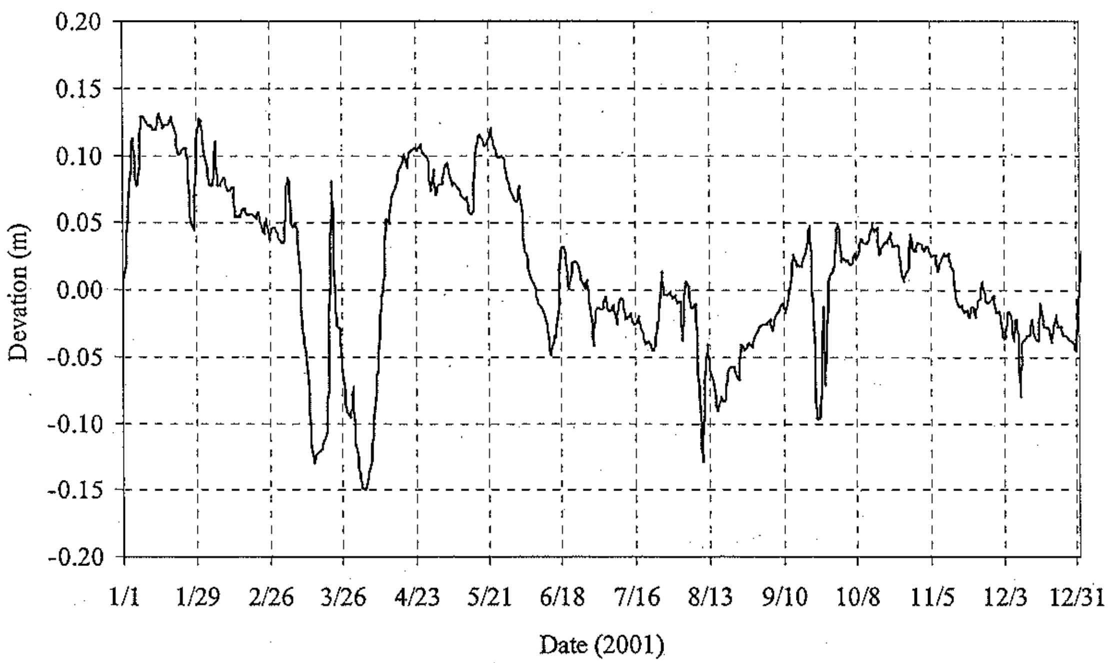

Figure 4.48 Deviation between 2001 Wachusett Reservoir modeled and measured WSE

4.2.3.1 Temperature Profile Comparison

In general, temperature profiles for Wachusett Reservoir predicted by CE QUAL W2 matched measured profiles closely for North and South Basins (See Figure 4.49 through

Figure 4.58). At those locations, the epilimnion portion of the profiles are almost exactly matched on every day. The largest deviation, 1.2 °C occurs on April 26, the first profile date. The reservoir had already started to stratify, thus, the uniform temperature initial condition applied was not completely accurate. However, the deviation is small as the vertical temperature gradient was very small. The hypolimnetic portions of the profiles predict measurements almost as well. The largest deviation here was 1.5 °C, occurring on August 22, although most of the hypolimnetic portions of the modeled profiles are within 1 °C of the measurements.

The thermocline at North and South Basin in the early summer is characterized by a very large temperature gradient in the measured profiles. On May 15, for example, a 7 °C temperature difference over a 2 meter interval was measured at South Basin. The thermocline became less pronounced on May 29, and by June the data shows a constant rate of change in temperature between the water surface and 13 m depth. The model predicted correct thermocline depth on May 15 and May 29, although a less pronounced gradient was predicted. However, the model predicted the thermocline to extend all the way to the surface on June 14 and 26, matching the data very closely. The July 14^{th} profiles are somewhat more typical, with relatively constant epilimnion and hypolimnion temperatures and a stepped thermocline. The model captures the correct shape of the thermocline, though predicting the epilimnion to be too deep by 2 m and with a temperature difference of up to 7 °C. The model is in relative agreement with the data in that overturn occurs between the end of October and early November.

As expected, the model does not accurately predict Thomas Basin temperatures due to the necessary but locally invalid assumption of lateral homogeneity required by CE QUAL W2. The predicted profiles are characterized by a much deeper thermocline than is observed in the reservoir.

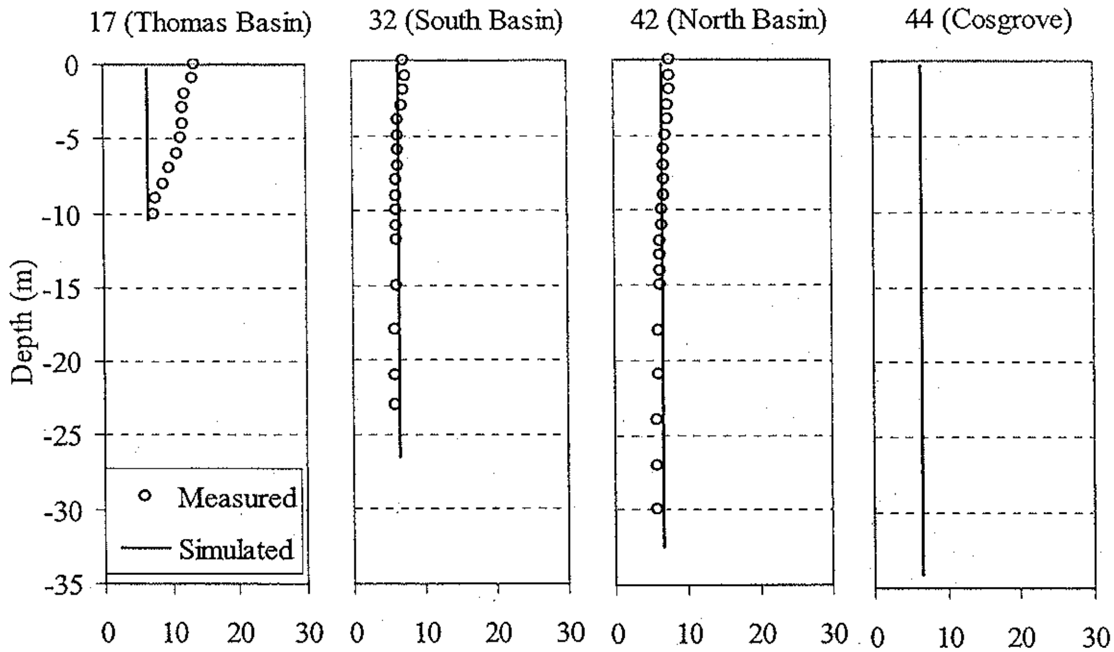

Figure 4.49 April 26, 2001 temperature profiles (°C)

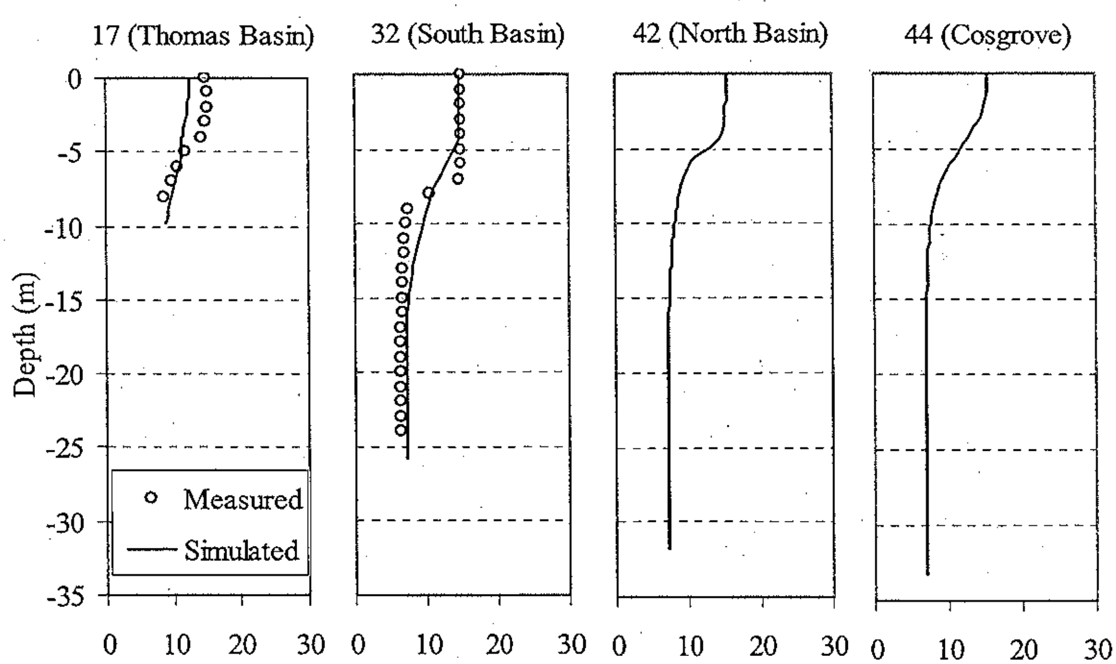

Figure 4.50 May 15, 2001 temperature profiles (°C)

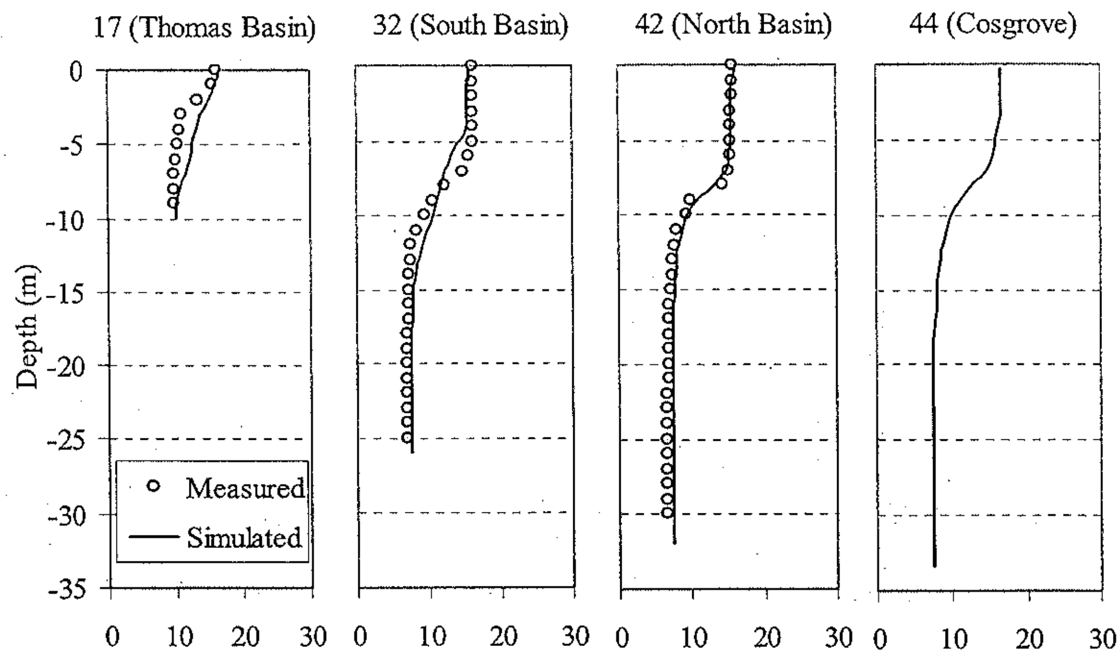

Figure 4.51 May 29, 2001 temperature profiles (°C)

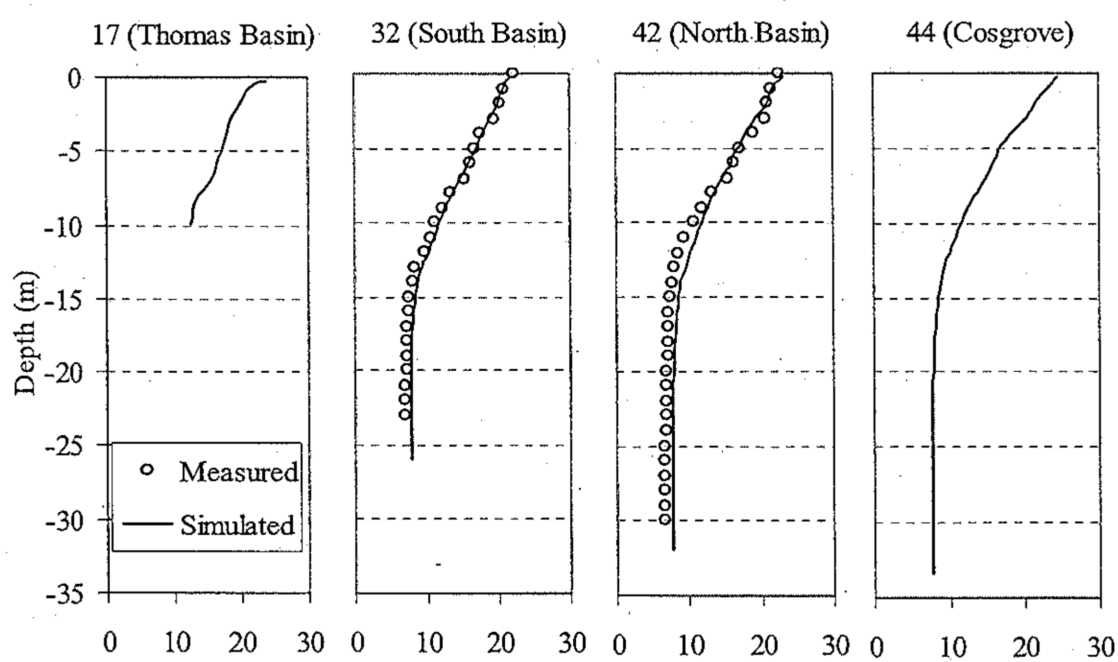

Figure 4.52 June 14, 2001 temperature profiles (°C)

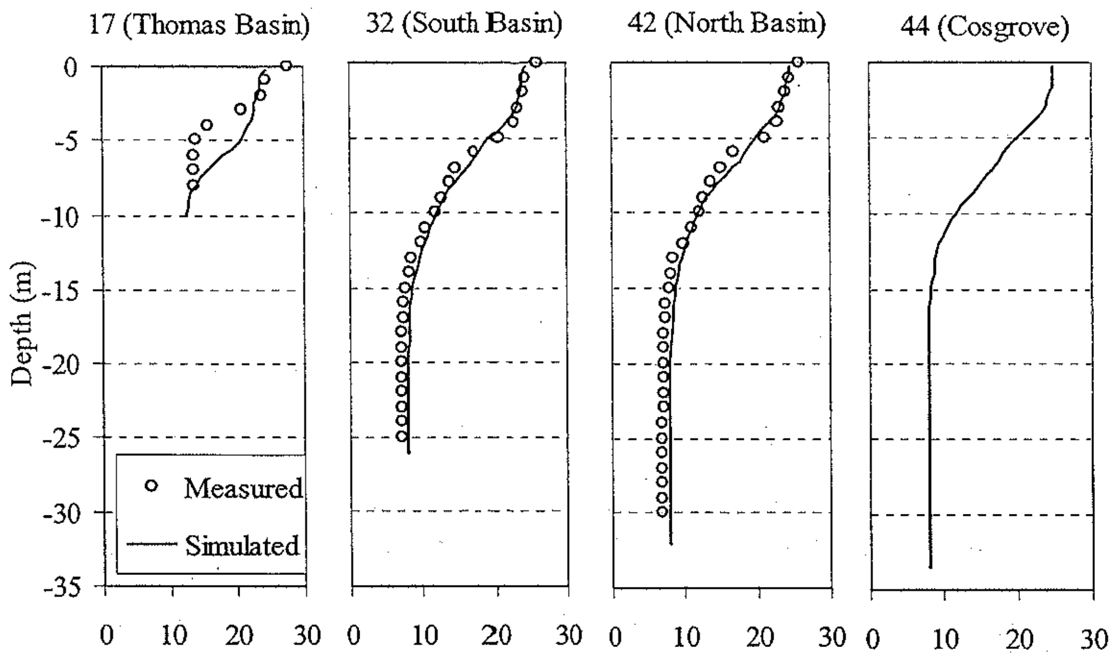

Figure 4.53 June 26, 2001 temperature profiles (°C)

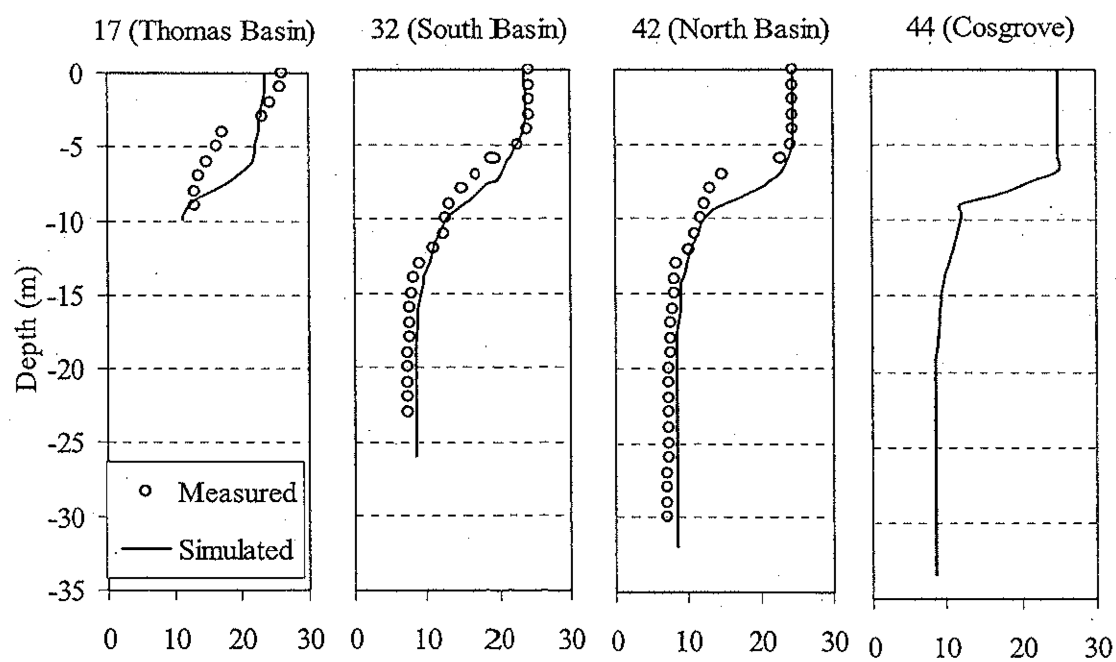

Figure 4.54 July 24, 2001 temperature profiles (°C)

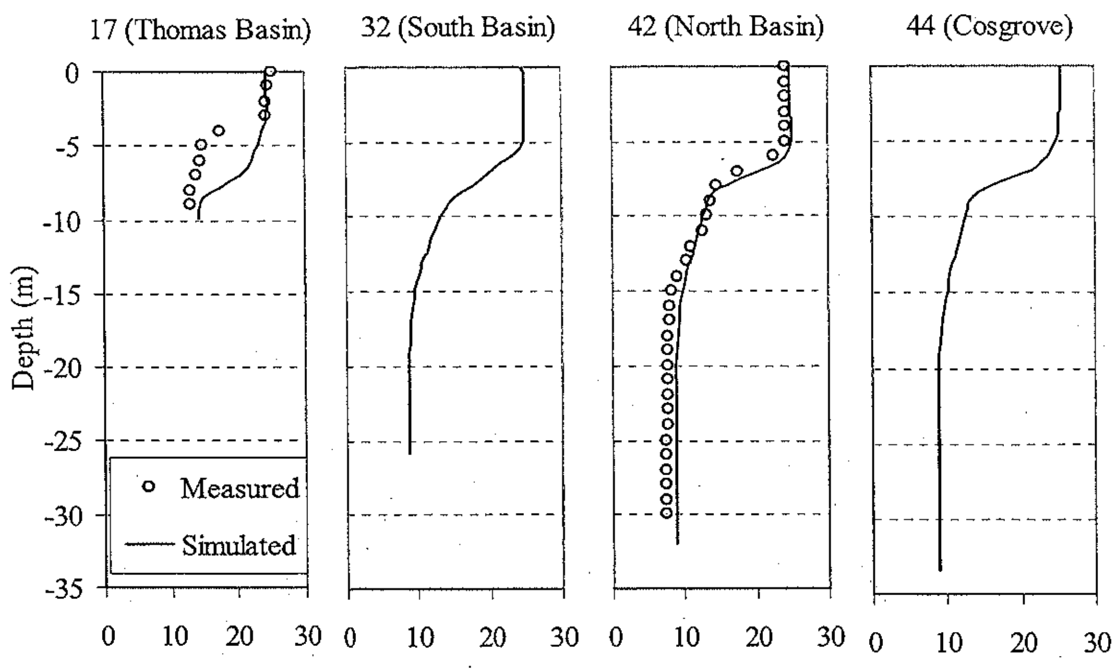

Figure 4.55 August 22, 2001 temperature profiles (°C)

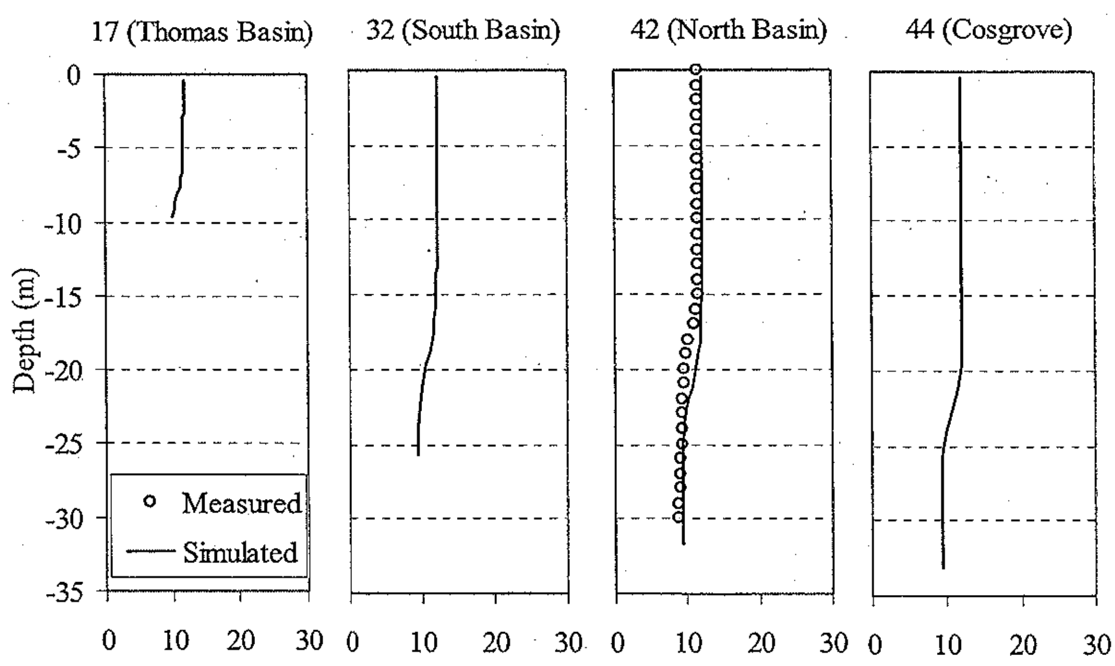

Figure 4.56 October 31, 2001 temperature profiles (°C)

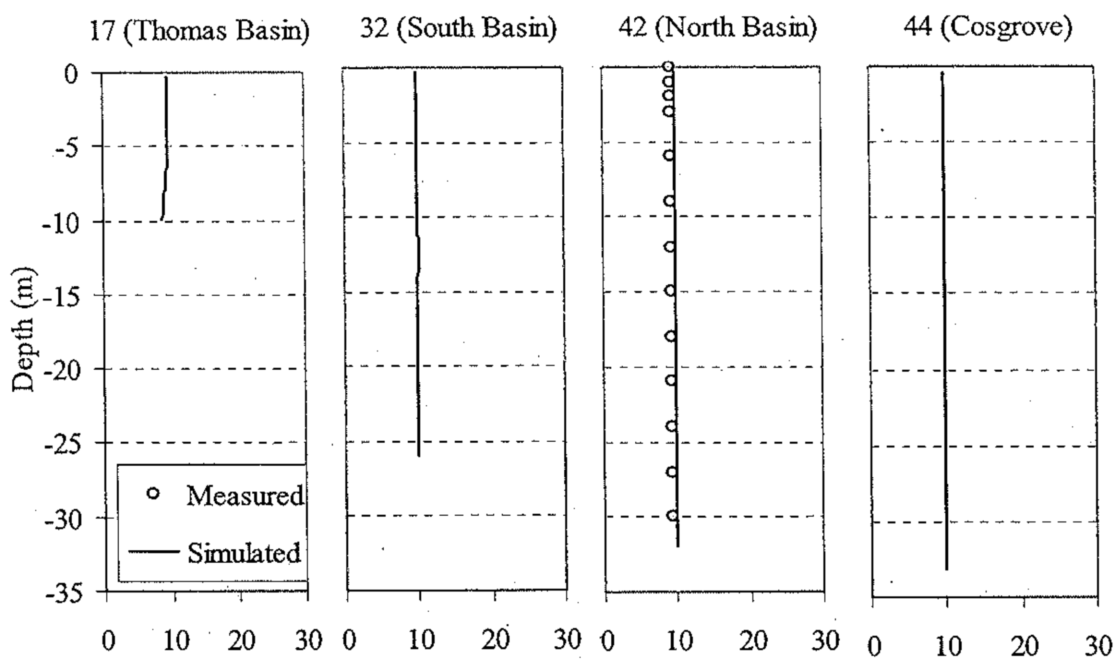

Figure 4.57 November 14, 2001 temperature profiles (°C)

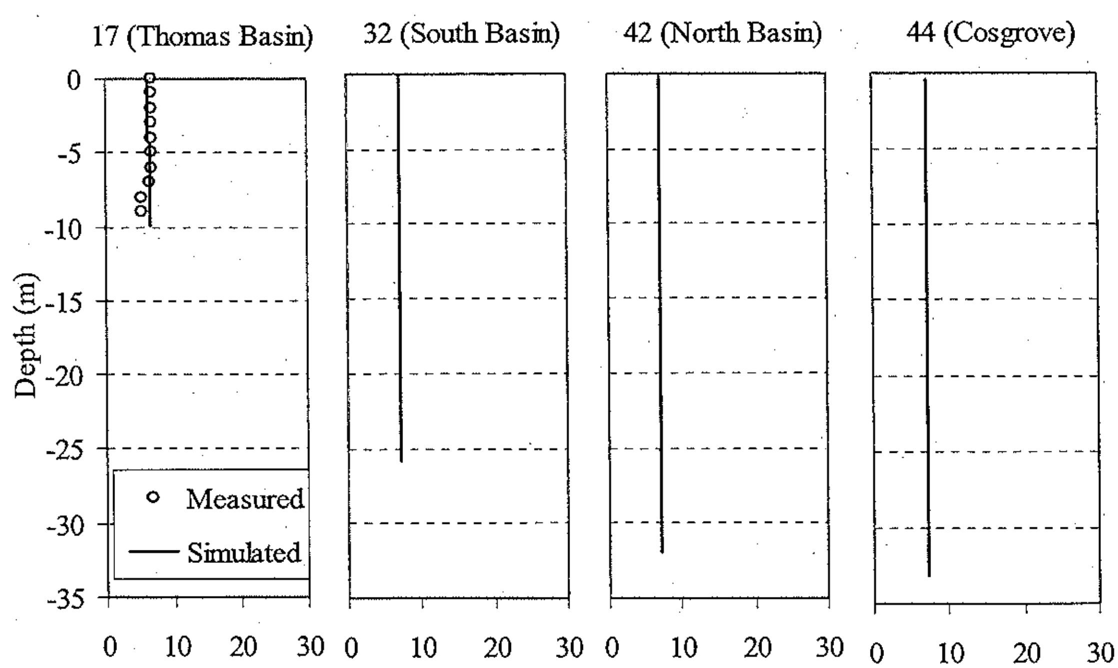

Figure 4.58 December 18, 2001 temperature profiles (°C)

4.2.3.2 Conductivity Profile Comparison

The 2001 predicted conductivity profiles (Figure 4.59 though Figure 4.68) match measured profiles well, especially in the metalimnion and hypolimnion at North and South Basins. The epilimnion portion of the measured profiles deviate slightly, but by not more than 11 μS/cm, occurring near the surface on June 16. In the hypolimnion, the largest deviations are on October 31 and November 14 at North Basin, with the model within 5 μS/cm of the data. The largest actual deviation occurs on April 26, resulting from the assumption of longitudinal uniformity for selecting an initial concentration for the reservoir.

In the metalimnion, the characteristic minimum of profile conductivity due to Quabbin transfer is apparent beginning on May 29 at South Basin and continuing through August 22. On June 14, June 26, and July 24 at South Basin and June 14^{th}, June 26, and August 22 at North Basin this characteristic minimum and surrounding gradients in the model occur at the same depths as in the measurements, deviating by no more than 9 μS/cm, but usually only deviating by 2 to 4 μS/cm. The model most inaccurately predicted the data on July 24 at North Basin, where a deviation of up to 27 μS/cm and 2 m occurred in the hypolimnion

As expected, the model did not accurately predict 2001 Thomas Basin conductivity profiles due to the incorrect assumption of lateral homogeneity.

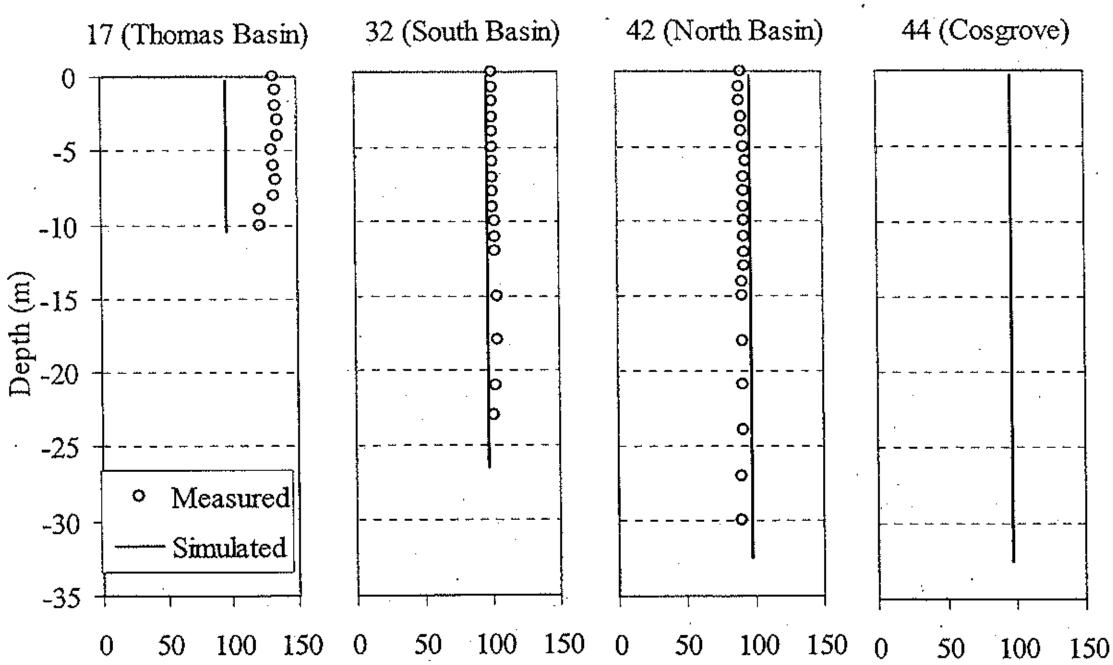

Figure 4.59 April 26, 2001 conductivity profiles (µS/cm)

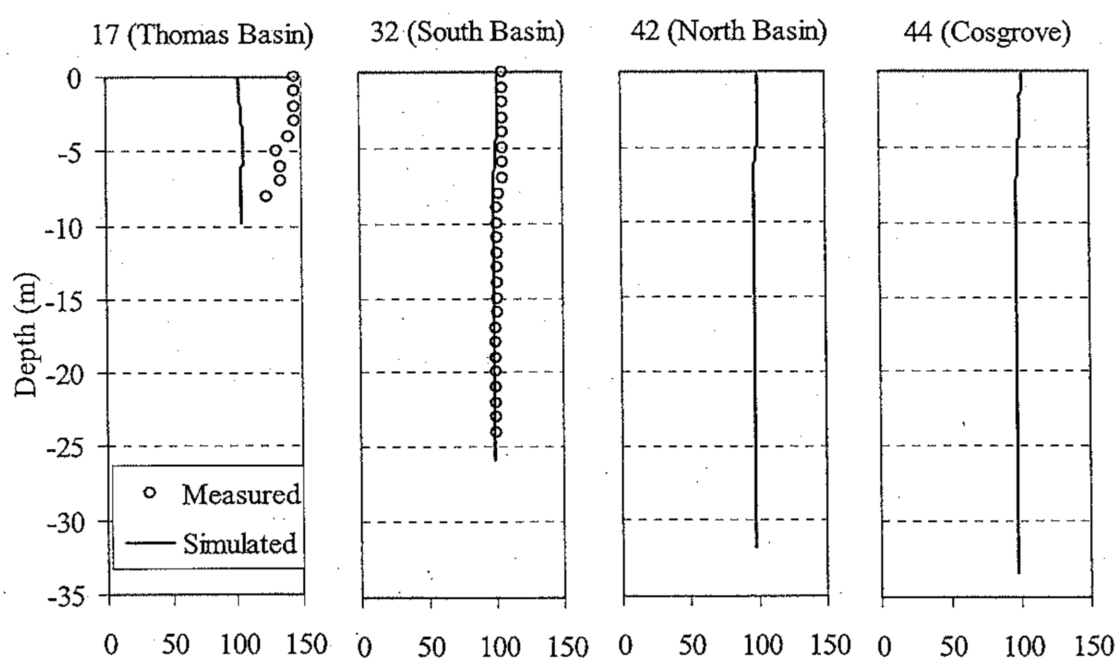

Figure 4.60 May 15, 2001 conductivity profiles (µS/cm)

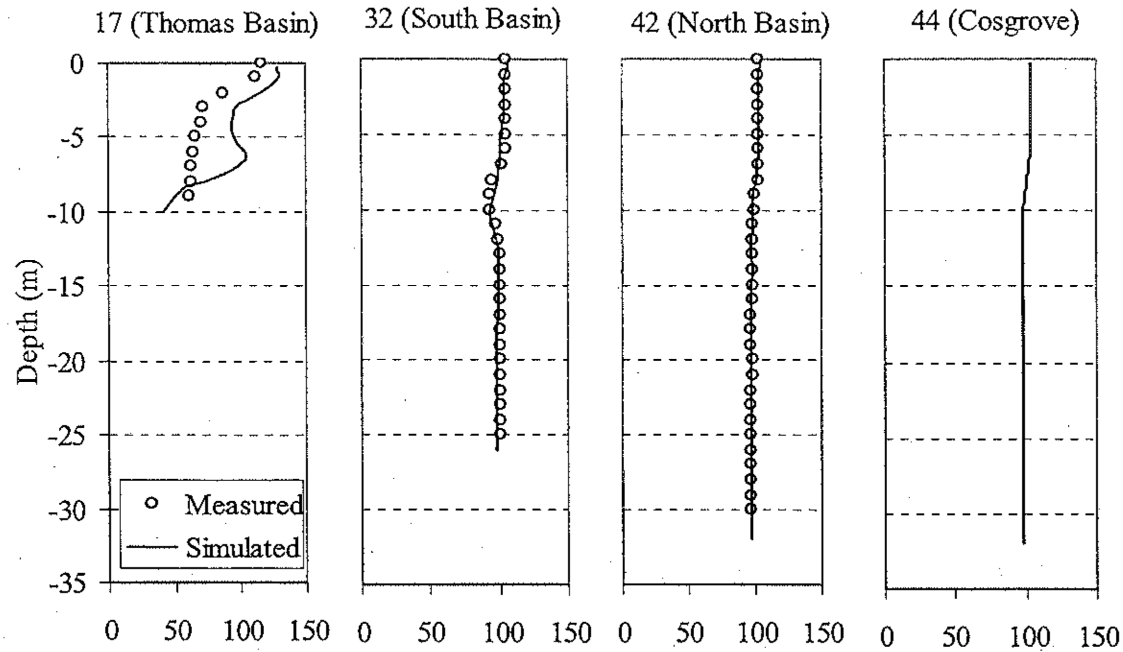

Figure 4.61 May 29, 2001 conductivity profiles (μS/cm)

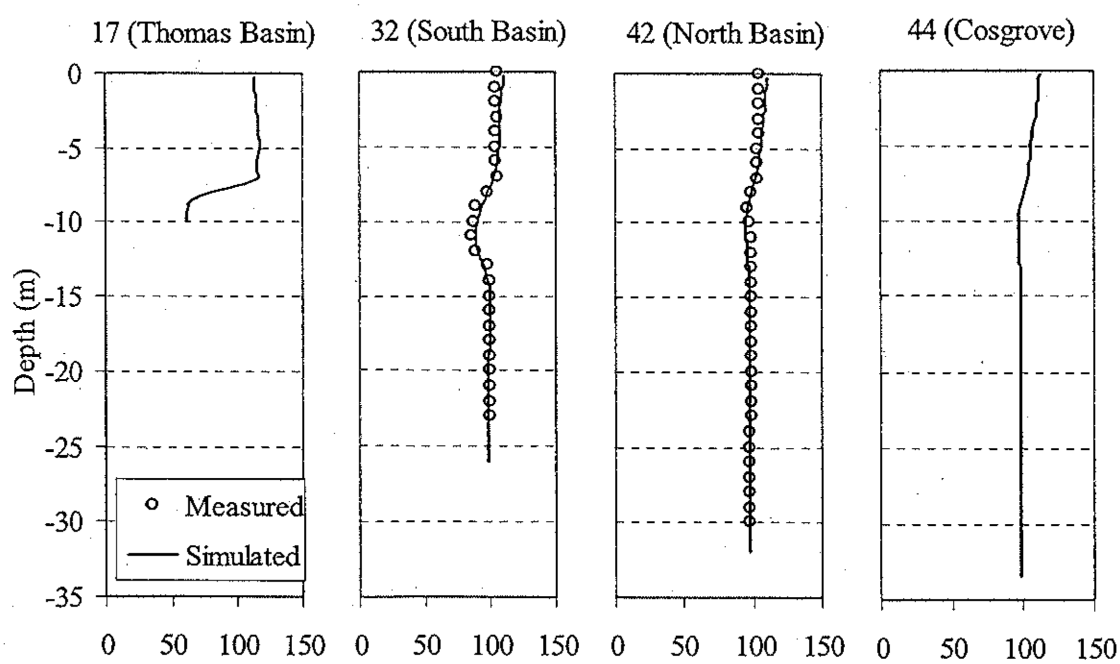

Figure 4.62 June 14, 2001 conductivity profiles (μS/cm)

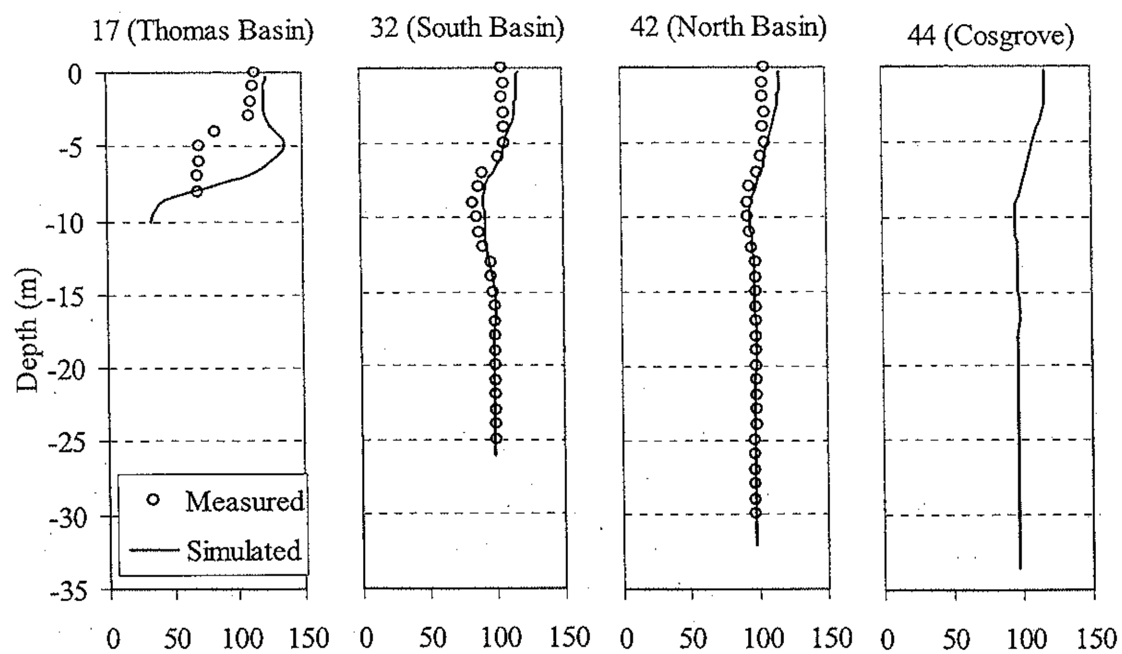

Figure 4.63June 26, 2001 conductivity profiles (µS/cm)

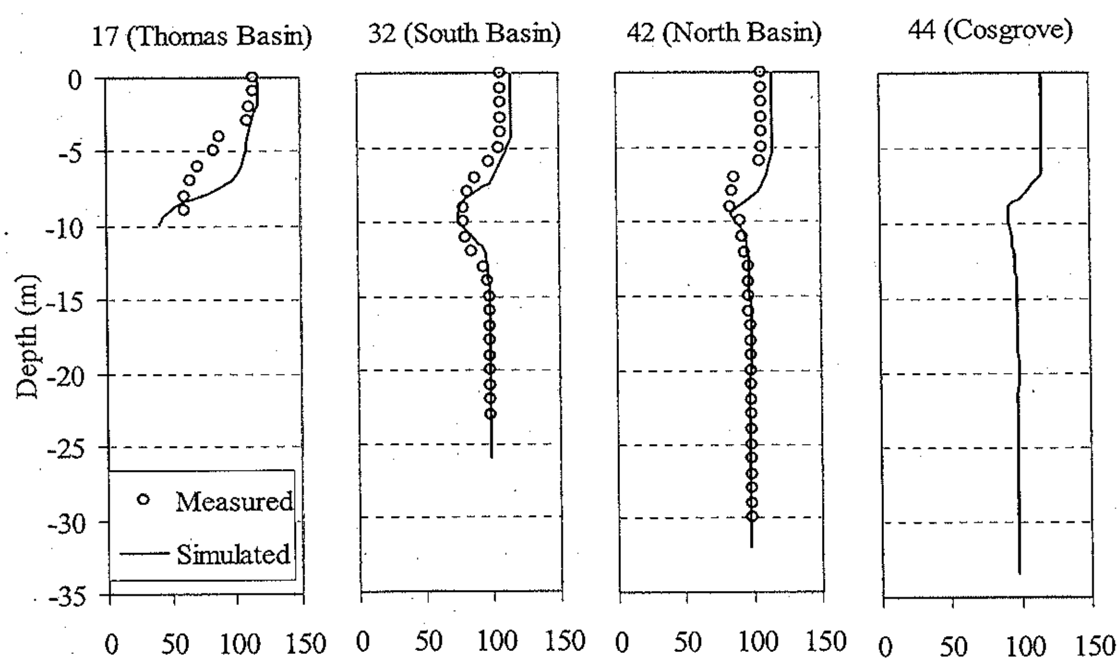

Figure 4.64 July 24, 2001 conductivity profiles (µS/cm)

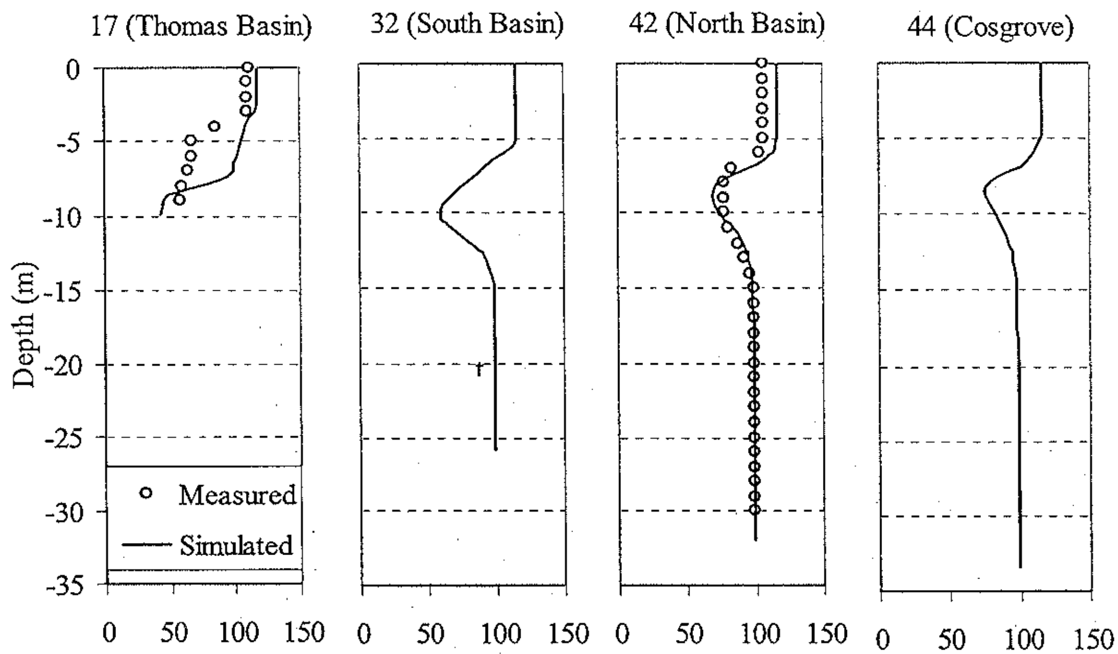

Figure 4.65 August 22, 2001 conductivity profiles (µS/cm)

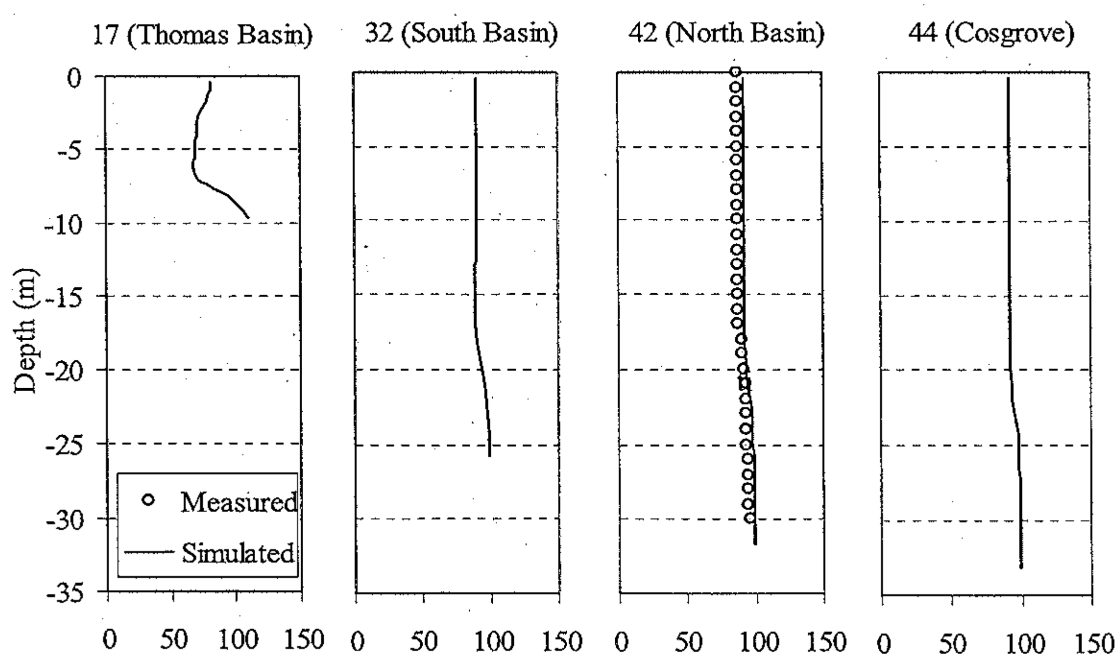

Figure 4.66 October 31, 2001 conductivity profiles (µS/cm)

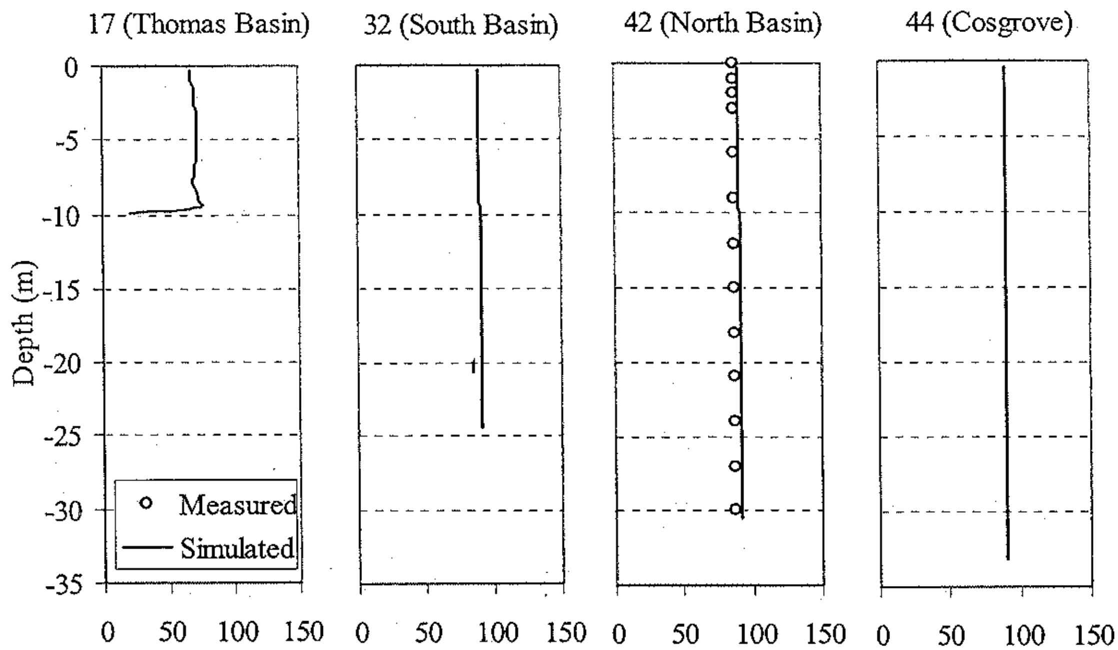

Figure 4.67 November 14, 2001 conductivity profiles (µS/cm)

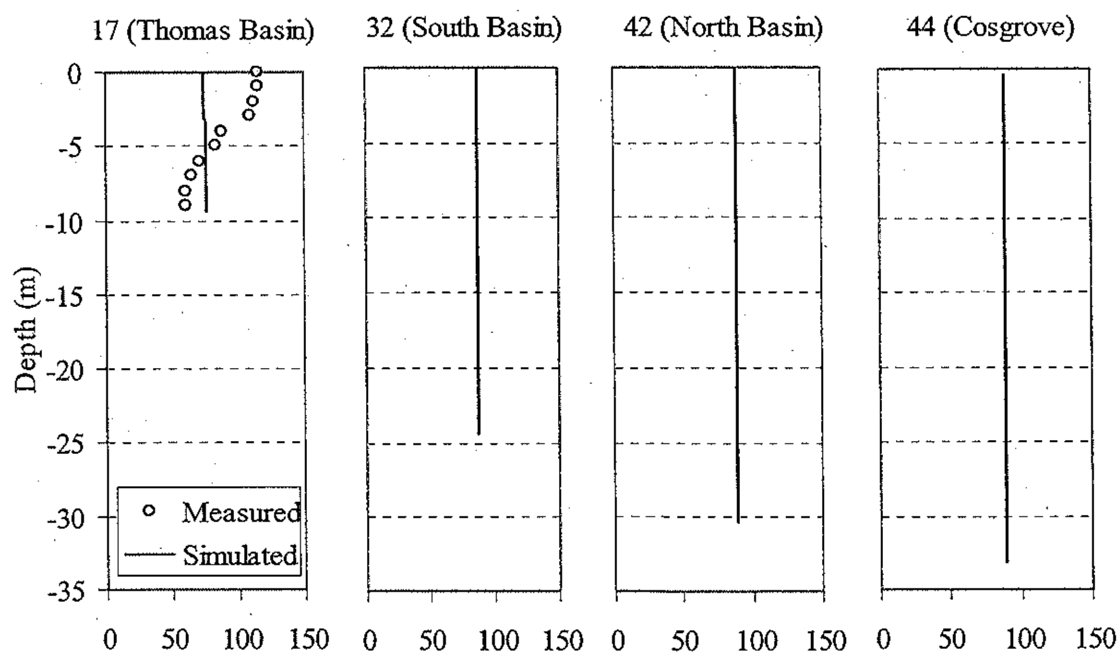

Figure 4.68 December 18, 2001 conductivity profiles (µS/cm)

4.3 Hydrodynamic Modeling – 2002

4.3.1 Reservoir Inflows

Precipitation returned to typical levels in 2002 following dry 2001. As measured at the co-operative weather stations surrounding Wachusett, precipitation was relatively consistent throughout the year. Figure 4.69 presents a precipitation hydrograph synthesized from daily averaged data for 2002, and Figure 4.70 presents total monthly precipitation for that year.

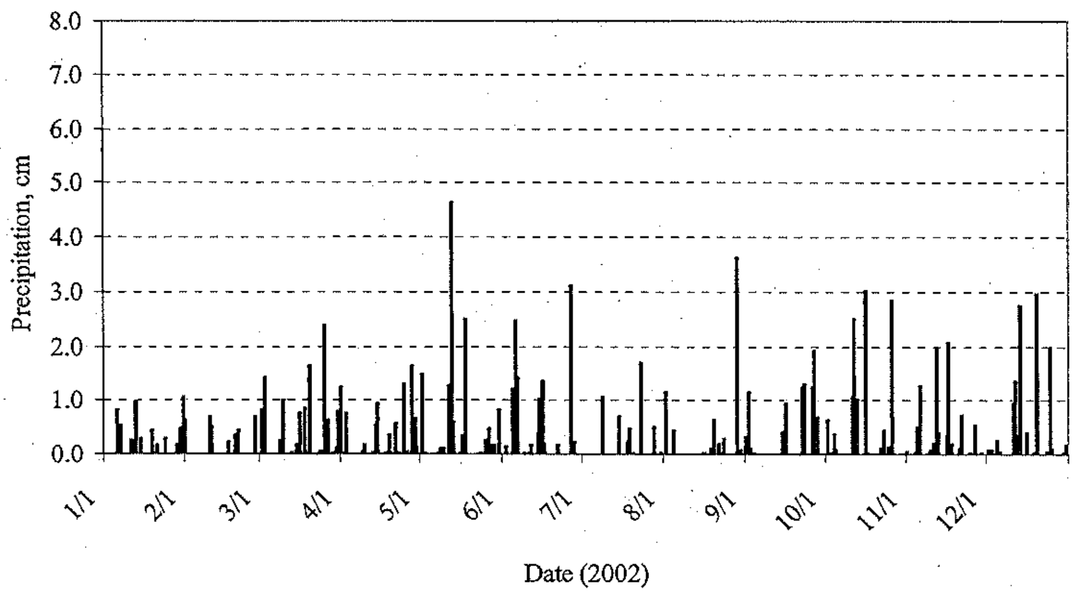

Figure 4.69 Precipitation Hydrograph for Wachusett Reservoir in 2002

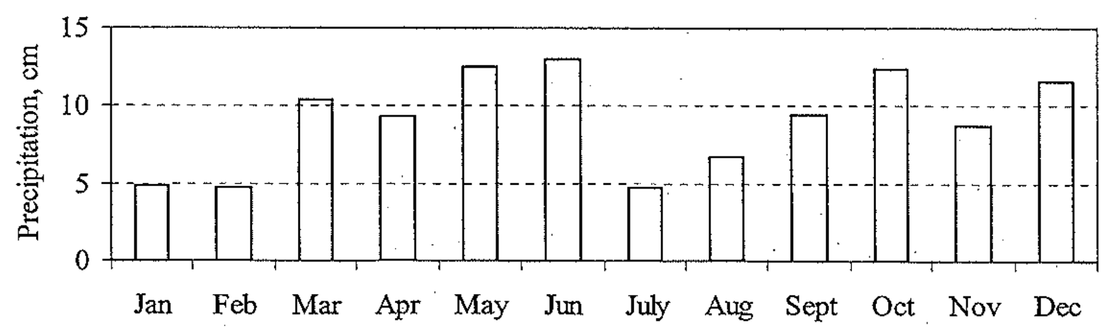

Figure 4.70 Total monthly precipitation accumulation for 2002

June received the most precipitation at 13.0 cm, though in March, May, October, and December between 10 and 13 cm fell. In three months, January, February, and July, less than 5.5 cm fell. Table 4.9 presents selected statistics for 2002 precipitation.

Table 4.9 Precipitation Statistics for 2002

Total Precipitation (cm)	109
Average Daily Precipitation (cm/day)	0.30
Number of Days with Precipitation	214
Average on Days with Precipitation (cm/day)	0.51

These statistics are very similar to those from 2000, (Table 4.1), wherein an average of 0.53 cm fell on each of 217 days with precipitation, totaling 114 cm. Figure 4.71 presents quarterly total precipitation for 2000 through 2002 where the quarters correspond roughly to the seasons.

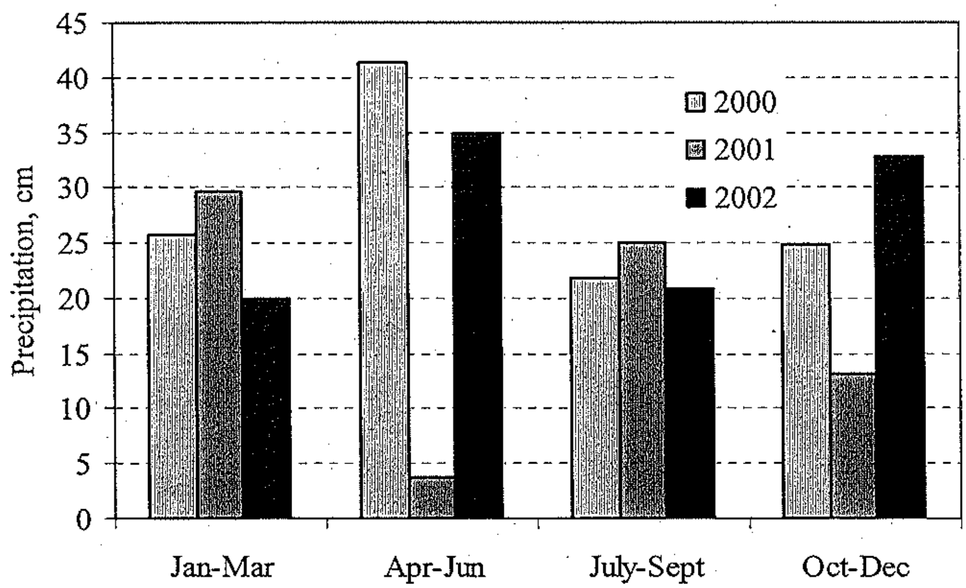

Figure 4.71 Total quarterly precipitation for 2000-2002

It is notable that the relative uniformity of precipitation during 2002 along with the slightly greater quantity of precipitation late in the year had a significant impact on tributary discharges in 2002. Figure 4.72 presents a hydrograph for the Stillwater and Quinapoxet Rivers in 2002.

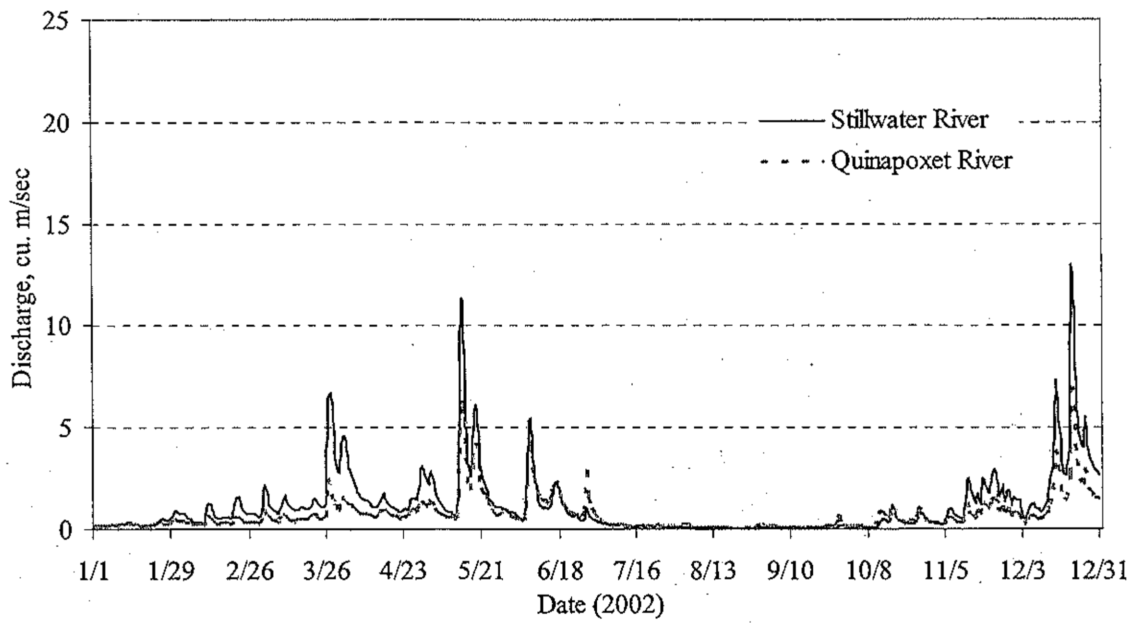

Figure 4.72 2002 hydrograph for Stillwater and Quinapoxet Rivers, composed of edited data

It is notable that there are two periods in 2002 where tributary discharges became larger than ~10 m^3/s, starting in February and lasting through June, and then again starting in October, compared to only one period each in 2000 and 2001 as seen in Figure 4.2 and Figure 4.41. It appears that these periods of large tributary discharge are dependent on long precipitation trends rather than monthly trends. This becomes apparent when the four periods of large discharge observed during this period of study are compared to the quarterly precipitation totals. In these cases, only when quarterly precipitation surpassed ~28 cm do the tributaries tend to respond significantly to precipitation events. The September 2001 storm event where 6.9 cm fell is an example. The three months preceding this event were not dry; 8.9 cm of precipitation fell in July, ~5 cm fell in August, and ~4.3 cm had fallen in September up to that point. However, total precipitation for the quarter was below a threshold, and there was little response by the tributaries.

By contrast, there were unusually significant storm events in late 2002 as can be seen in Figure 4.72, but total precipitation for the preceding quarter was 33 cm and the tributaries responded with large discharges.

2002 was also atypical in terms of Quabbin transfers. Despite the presence of two periods of relatively high tributary discharge, annual total discharge for Quinapoxet and Stillwater Rivers was below normal as can be seen in Figure 4.73.

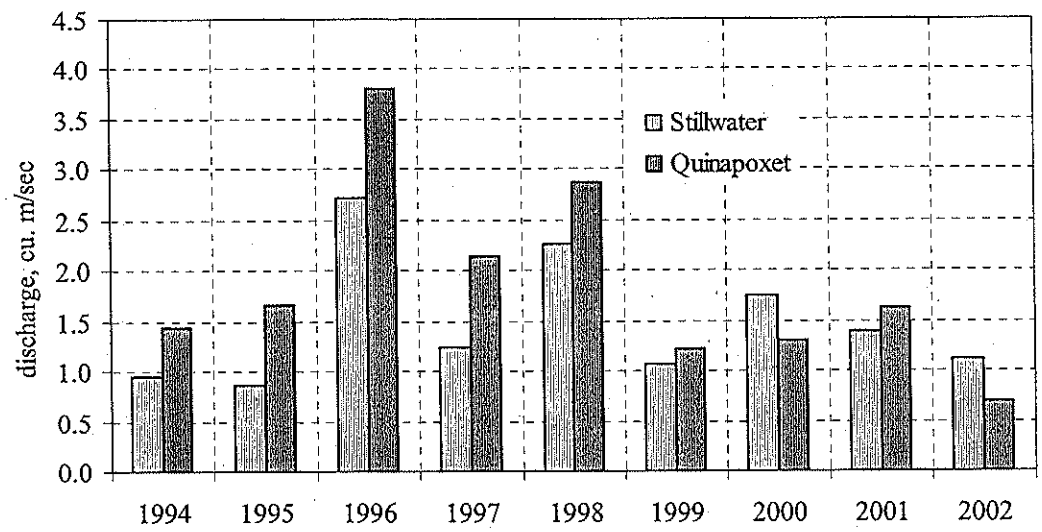

Figure 4.73 Average annual discharge (daily basis) for Quinapoxet and Stillwater Rivers

As a result of these relatively small discharges, the Quabbin Transfer quantity differed from past years. Figure 4.74 shows a time series plot of Quabbin transfer, which was not included in Figure 4.72 for readability, while Table 4.10 provides statistics describing Quabbin transfer and metropolitan Boston demand for the three years.

It is significant to note the operational reliance on Quabbin Transfer in 2002. Although a normal quantity of precipitation was received in 2002, a slightly greater quantity of transfer occurred than in 2001, and a much greater quantity occurred than in 2000. Even in the late spring, when tributary inflows are normally large, transfer had to occur to maintain WSE. As a result, Quabbin water accounts for a large proportion of water received by Wachusett in 2002 (68.3%), as shown in Figure 4.75. Differences in demand for the three years do not account for differences in transfer; in fact, demand in 2002 was slightly smaller than during the 2000 and 2001.

Table 4.10 Quabbin Transfer Statistics

Statistic	2000	2001	2002
Average Daily Transfer, m^3/s	6.0	7.0	7.2
Average on Days with Transfer, m^3/s	9.5	9.0	10.2
Number Transfer Days	231	285	258
Volume Transferred, 10^6 m^3	190	222	227
Demand (Cosgrove plus Wachusett), 10^6 m^3	319	311	307

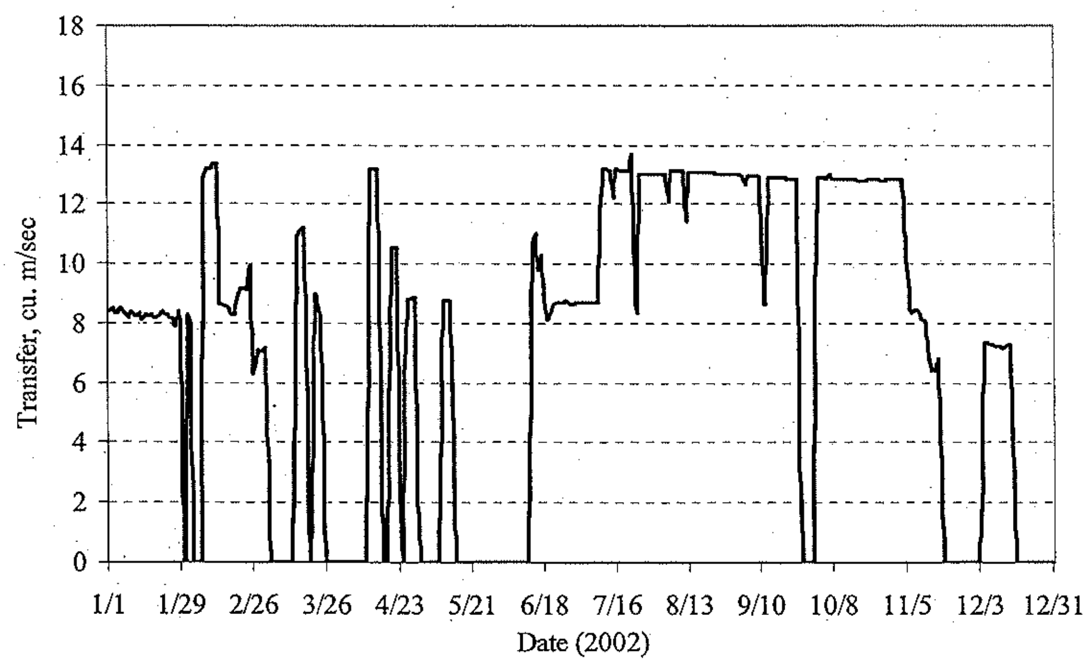

Figure 4.74 2002 Quabbin Transfer to Wachusett Reservoir

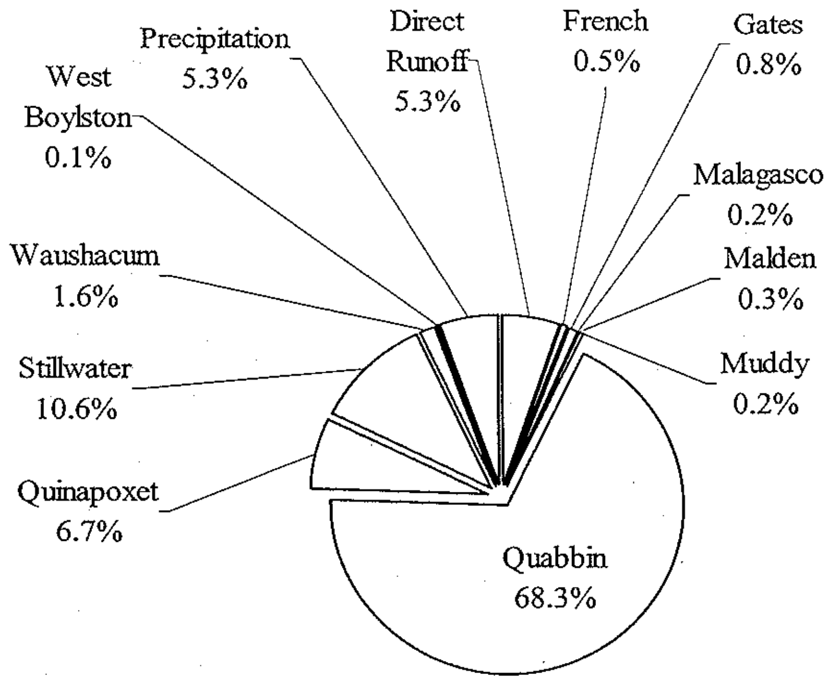

Figure 4.75 Comparison of water received from modeled Wachusett Reservoir sources in 2002

In 2002, water originating at Quabbin comprised 10% more of the total inflow to Wachusett than in 2001. As Quabbin water has lower levels of most constituents, a trend towards lower constituent concentrations in Wachusett is expected. Of the other significant sources of water to Wachusett in 2002, 6.7% came from Quinapoxet, 10.6% from Stillwater, 5.3% each from direct runoff and direct precipitation, and 3.8% comes from the seven minor tributaries.

4.3.2 Reservoir Losses

Wachusett reservoir outflows in 2002 were characterized by a period of simultaneous discharge to the Cosgrove and Wachusett aqueducts, a rather unusual event. Discharge to Wachusett Aqueduct occurred for approximately 18 days in October to test it for reinstatement to service during repairs to Cosgrove. The combination of this ~9.6 m^3/s discharge plus ~8.7 m^3/s average demand during this period overwhelmed the combined Quabbin and Wachusett system supply, causing a dramatic decline in WSE even though Quabbin was transferring at ~13 m^3/s.

Additionally, as the quantity of water discharged from tributaries into Wachusett during 2002 was smaller than in previous years, there was no period of spilling to the Nashua River in 2002 as there was in 2000 and 2001. As a result, discharges to this river in 2002 accounted for only 1% of water losses, versus 6.6% and 13% in the earlier years, respectively. The relative quantity of water lost to each sink is presented in Figure 4.76.

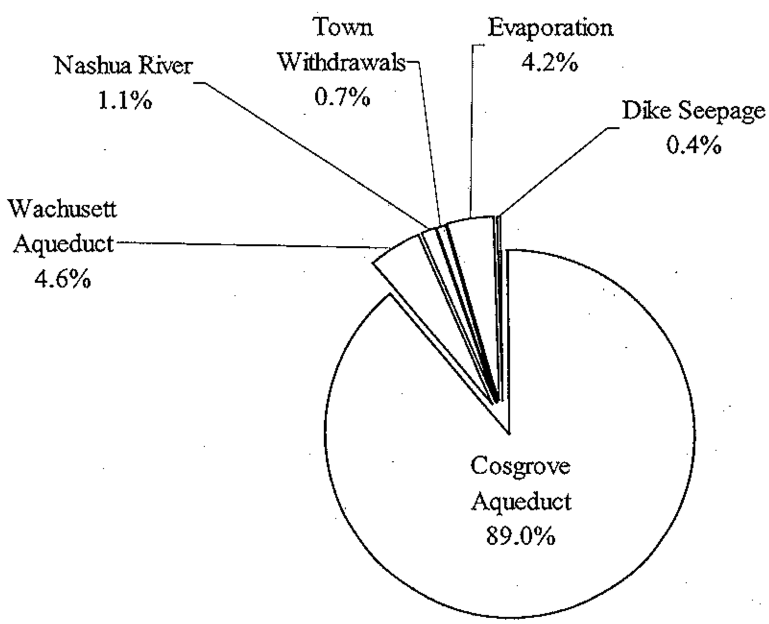

Figure 4.76 Relative quantity of water exiting Wachusett Reservoir through each considered outflow

4.3.3 Calibration Results

Table 4.11 presents the 1994 through 2001 range of calibration factors for each modeled tributary and Quabbin Transfer, as well as calibration factors for 2002. It is important to note that the calibration factors shown were only partially determined by the SOLVER algorithm package included with Microsoft EXCEL. A hydrodynamic inconsistency that arises in the CE QUAL W2 model in late 2002 introduced uncertainty into the calibration. The inconsistency was first noted through comparing measured and predicted conductivity profiles; conductivity accumulated throughout the year in the model. The SOLVER-determined calibration factor for Quabbin was 0.98. This factor was then set manually to unity with transfer measurements, and SOLVER was implemented to determine calibration factors for the tributary inflows, which are subject to much more uncertainty. This was done to reduce the accumulation of conductivity in Wachusett Reservoir by adding additional Quabbin water. The resulting calibration did not completely solve the problem. More details regarding the hydrodynamic inconsistency are presented in Section 4.3.3.2.

Table 4.11 Summary of 2002 and historic calibration factors

Inflow	Range (1994-2001)	2002
Quabbin	0.97-1.19	1.00
Stillwater	0.70-1.28	1.16
Quinapoxet	0.82-1.30	1.05
Waushacum	1.11-1.65	0.98
Direct Runoff	1.11-1.62	0.98
Malden	1.00-1.35	0.98
W. Boylston	1.11-1.35	0.98
Gates	1.11-2.00	0.98
Muddy	1.00-1.35	0.98
Malagasco	1.00-1.35	0.98
French	1.00-1.35	0.98

It is also notable that a series of precipitation events at the end of 2002 resulted in Stillwater and Quinapoxet River discharges that were inconsistent with responses to other precipitation events during the year (i.e. that portion of the year could be characterized by a significantly different runoff coefficient than the rest of the year). The reason for this inconsistency was not apparent. It was therefore necessary to estimate tributary runoff by a water balance that utilized measured water surface elevation, all measured and estimated non-riverine inflows and withdrawals, and estimated evaporation to back-calculate total tributary inflow. Total tributary inflow was then divided between tributaries and direct runoff based on relative watershed size, except for Stillwater and Quinapoxet discharges, which were apportioned based on historic runoff.

The determined calibration factors are generally below the range of values determined for 1994 through 2001. Each minor tributary calibration factor as well as the direct runoff calibration factor was determined to be 0.98, while the minimum calibration factor for the preceding seven year period was 1.00 for Malden, Muddy, Malagasco, and French Brooks, and 1.11 for Waushacum, West Boylston, and Gates Brooks and Direct Runoff. However, 0.98 is close to unity with the uncelebrated predicted discharges, and is therefore acceptable. The determined calibration factors for Stillwater and Quinapoxet Rivers, 1.16 and 1.05, respectively, are within the range of historic values for those tributaries. Further, the Stillwater River calibration factor suggests greater uncertainty than the Quinapoxet calibration factor, which agrees with the USGS ratings of the streamgages from which the data are obtained. These calibration factors are therefore considered reasonable. Table 4.12 presents the range of unitless runoff coefficients for each tributary for 1994 through 2001, along with the runoff coefficient determined by this analysis for 2002. The Quinapoxet runoff coefficient was not calculated; doing so requires additional data from the City of Worcester which was not made available.

Table 4.12 Summary of 2002 and historic runoff coefficients

Inflow	Range (1994-2001)	2002
Stillwater	0.24-0.75	0.39
Quinapoxet	0.54-0.94	--
Waushacum	0.37-0.75	0.30
Direct Runoff	0.37-0.75	0.30
Malden	0.37-0.75	0.30
W. Boylston	0.37-0.75	0.30
Gates	0.37-0.75	0.30
Muddy	0.37-0.75	0.30
Malagasco	0.37-0.75	0.30
French	0.37-0.75	0.30

The calculated 2002 runoff coefficient for each tributary except Stillwater is below the range of values for the 1994 through 2001 period. The Stillwater River runoff coefficient is near the low end of the range for that period. This is consistent with other observations; although a typical quantity of precipitation fell during 2002, annual average Stillwater and Quinapoxet runoff (presented in Figure 4.73) were lower than typical during the year. These results are therefore considered acceptable.

Table 4.13 presents statistics resulting from the 2002 water balance calibration. Root mean squared (RMS) error is 0.005 m larger than the value resulting from the 2001 calibration, although it is well below 0.15 m (1 ft). On three days the difference between measured and predicted WSE differed by more than 0.15 m. The calibration is therefore considered acceptable.

Table 4.13 2002 Water Balance Calibration Statistics

Statistic	Value
RMS error, m	0.068
Average Absolute Deviation, m	0.060
Maximum Positive Deviation, m	0.150
Maximum Negative Deviation, m	-0.1650
No. Days Greater then +/- 0.15 m Deviation	3

Figure 4.77 presents measured Wachusett WSE and WSE predicted with the 2002 water balance calibration. Figure 4.78 presents deviation between measured and predicted WSE. It is significant that a systematic deviation is present, tending towards negative deviation (predicted WSE less than measured WSE) from February through July and positive deviation during July through December.

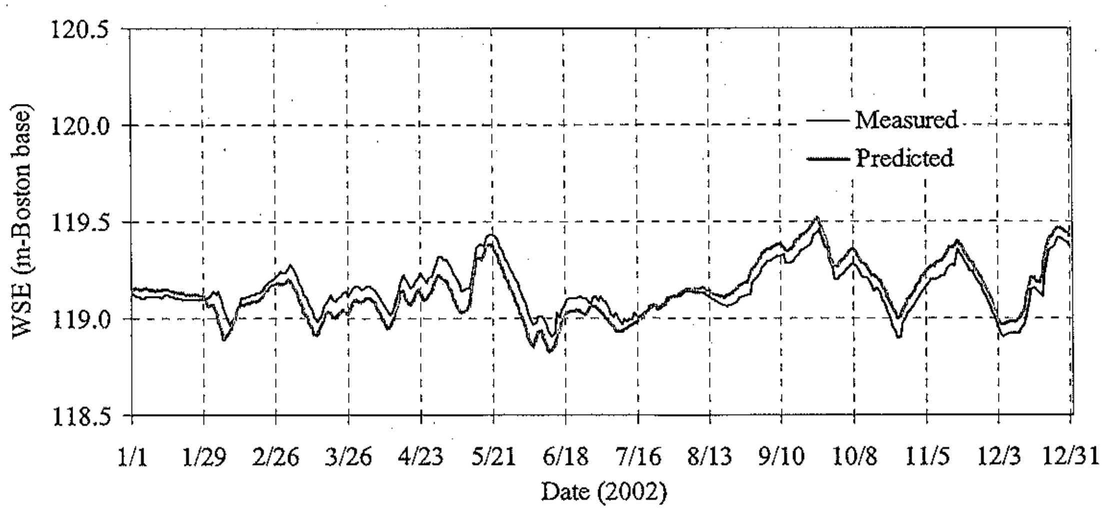

Figure 4.77 Measured Wachusett WSE and WSE predicted Microsoft Excel

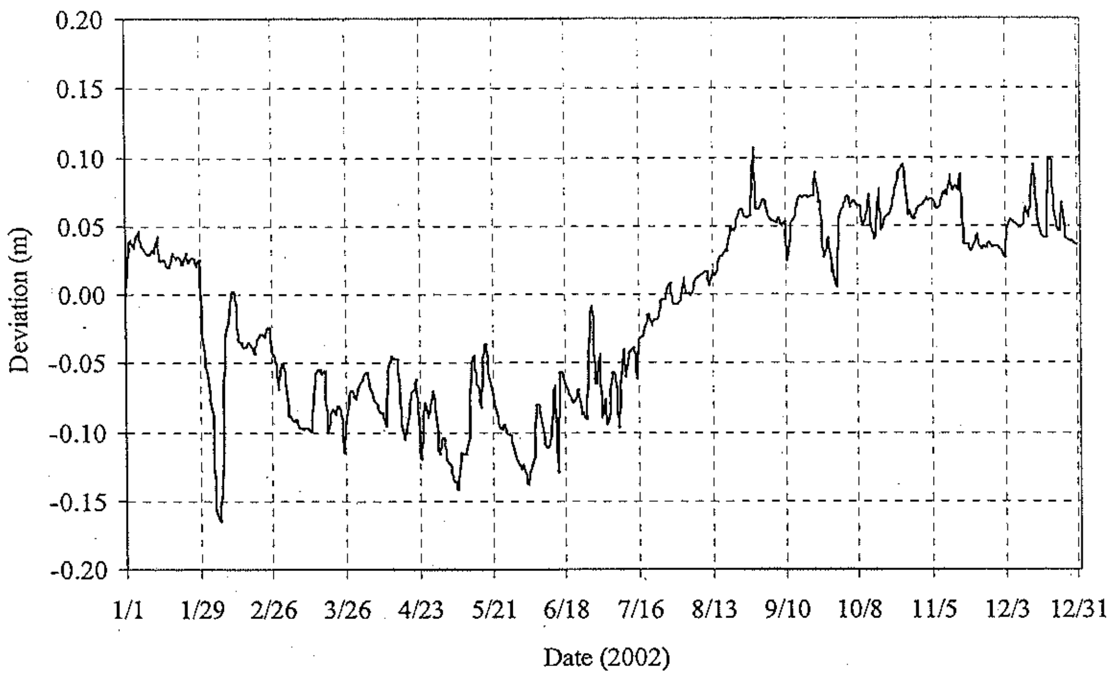

Figure 4.78 Deviation between Wachusett WSE as measured and as predicted

This deviation likely arises from holding the Quabbin transfer calibration factor to 1.00; increasing the transfer calibration factor lead to more inflow in the second half of the year (when the majority of transfer occurred), while decreasing tributary calibration factors decreased spring inflow. However, the magnitude of deviation is small and the calibration is therefore considered acceptable.

4.3.3.1 Temperature Profile Comparison

Figure 4.79 through Figure 4.89 show temperature profiles measured by DCR staff in Thomas Basin, South Basin, North Basin, and the vicinity of the Cosgrove intake, along with temperature profiles predicted by CE QUAL W2 at those locations. CE QUAL W2 generally captures the temperature profiles at South Basin, North Basin and Cosgrove Intake to within 3 °C. The model has a slight tendency to overpredict epilimnion temperatures and underpredict hypolimnion temperatures, although the depth of the thermocline is accurately predicted when present.

Thomas Basin temperature profiles are generally not well predicted. However, on November 26, 2002, DCR and UMass staff recorded three profiles within Thomas Basin to establish boundary conditions for a 3-D CFD model of the basin (Figure 4.88). The profiles were measured in the typical Thomas Basin measurement location (in the longitudinal center of the basin close to the east shore, at the railroad bridge under which Quinapoxet River water and Quabbin Transfer enters Thomas Basin and at the Route 12 bridge where water exits the basin and enters the main reservoir.

CE QUAL W2 was implemented to predict profiles in Segment 11 of the model (corresponding to the typical measurement location) and Segments 14 and 15; immediately upstream of and underneath the Route 12 bridge as represented in the model. Although there is significant deviation between the profile predicted for Segment 11 and the profile measured in that part of the basin, the profile measured at the Route 12 bridge is captured by the model (both segments) to within 1 °C.

Although the assumption of lateral homogeneity necessary to implement CE QUAL W2 has shown to be inapplicable to Thomas Basin, the water column beneath the Route 12 bridge is narrow and therefore more likely to be laterally uniform. Profiles measured at this location, as well as water sampled here, are likely more characteristic of the water quality in Thomas Basin and of the water entering the reservoir than water at the typical sampling location.

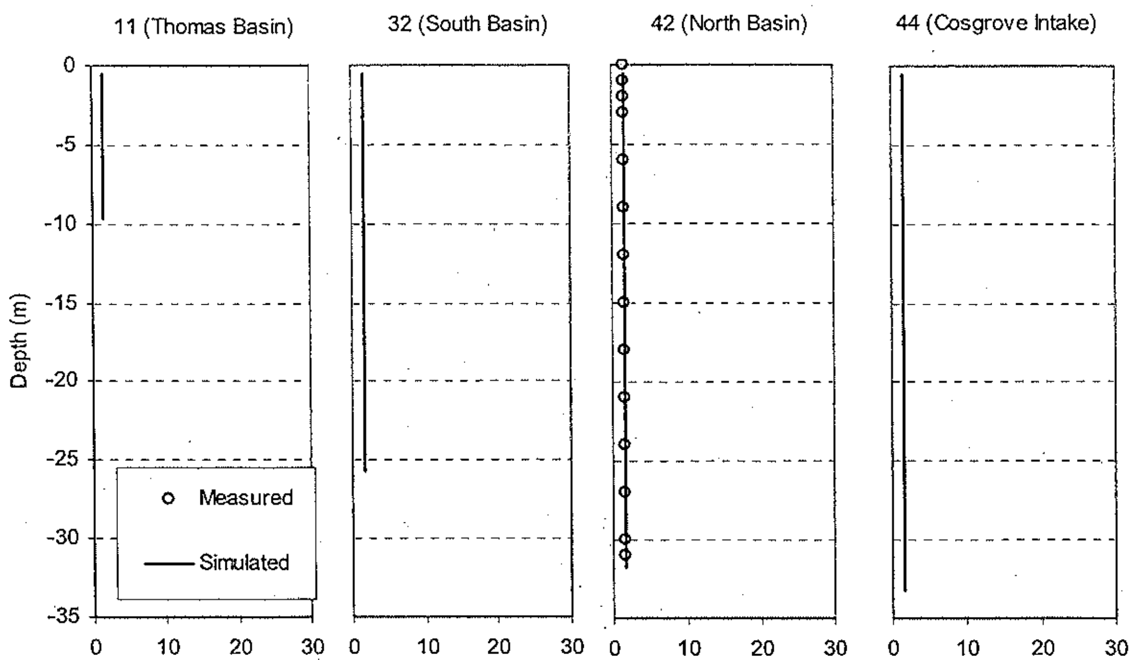

Figure 4.79 February 20, 2002 temperature profiles (°C)

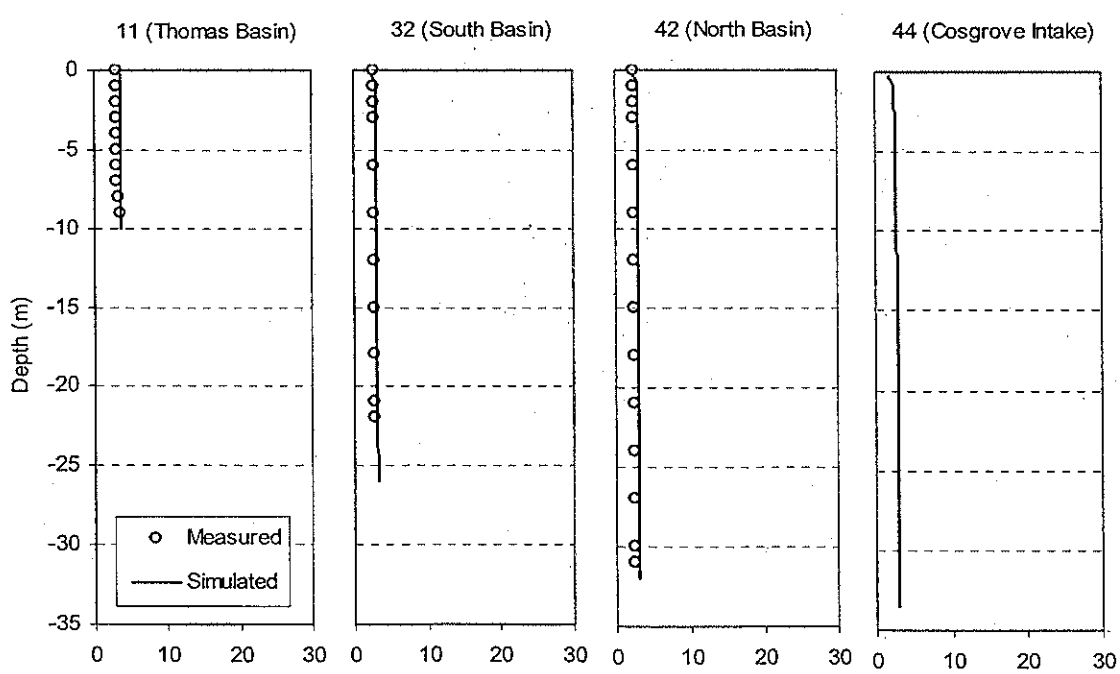

Figure 4.80 March 5, 2002 temperature profiles (°C)

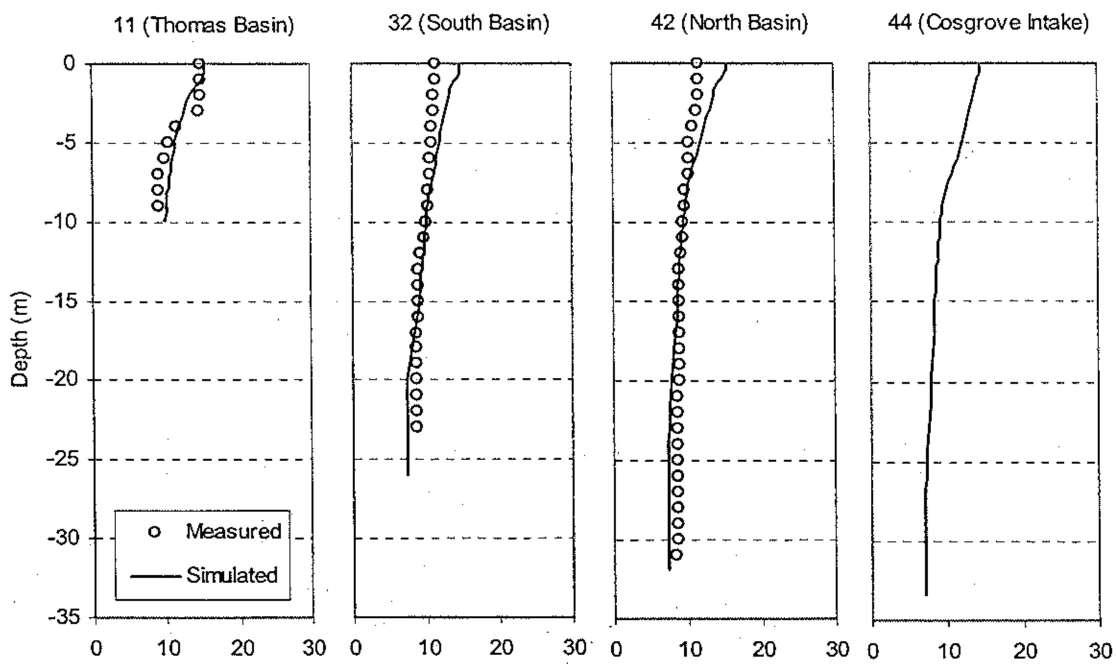

Figure 4.81 May 9, 2002 temperature profiles (°C)

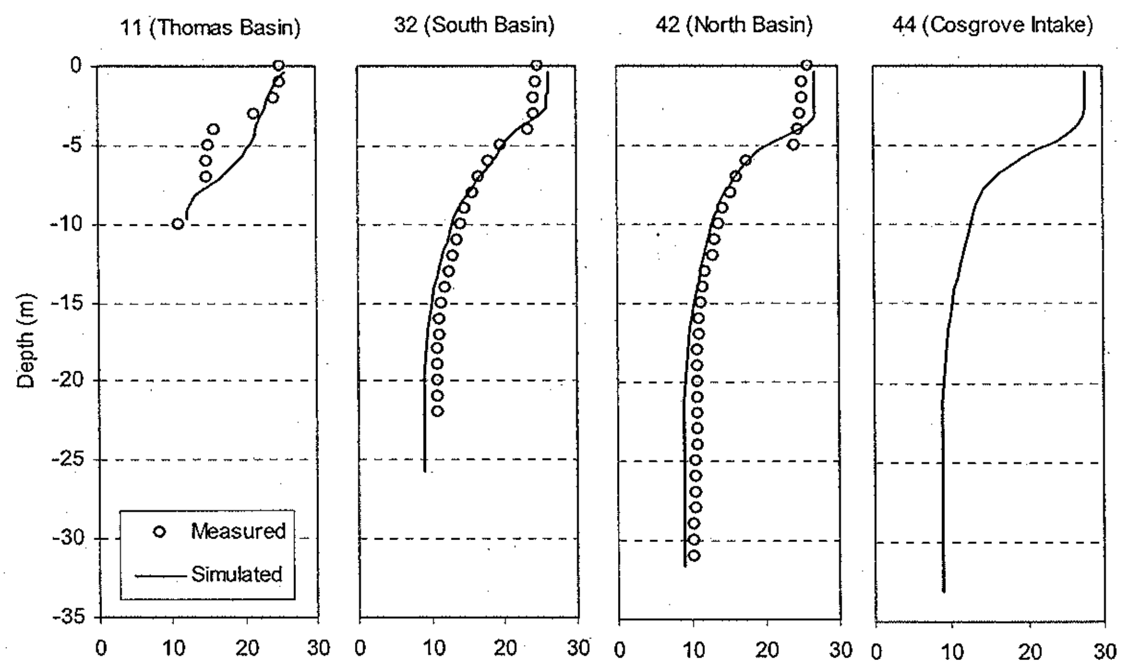

Figure 4.82 July 2, 2002 temperature profiles (°C)

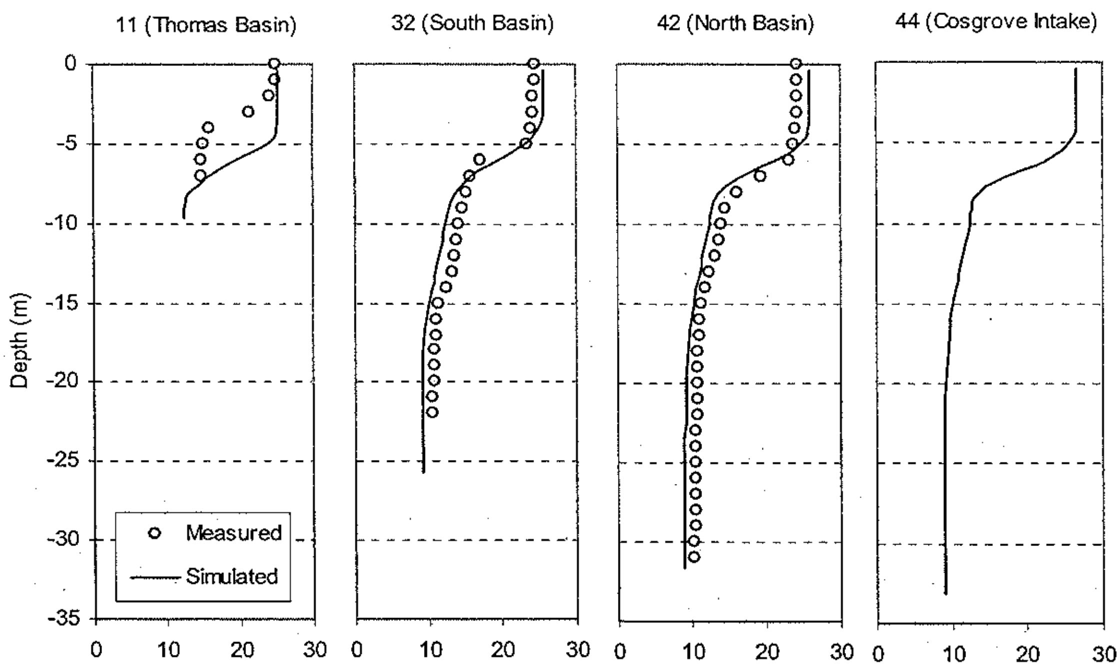

Figure 4.83 July 22, 2002 temperature profiles (°C)

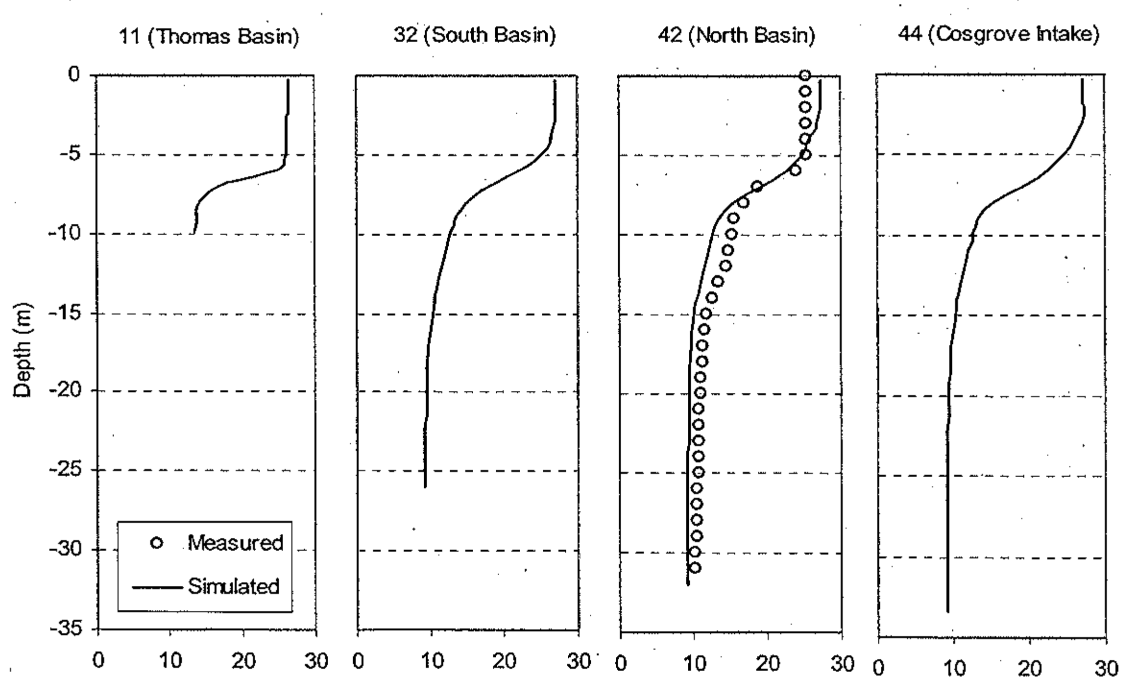

Figure 4.84 August 22, 2002 temperature profiles (°C)

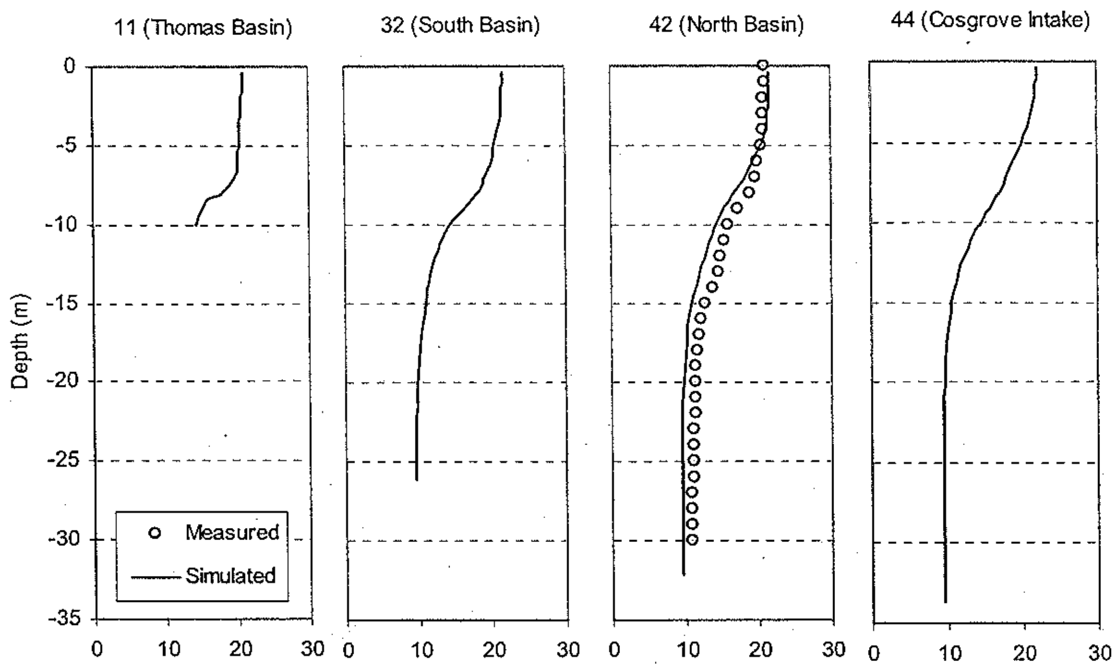

Figure 4.85 September 18, 2002 temperature profiles (°C)

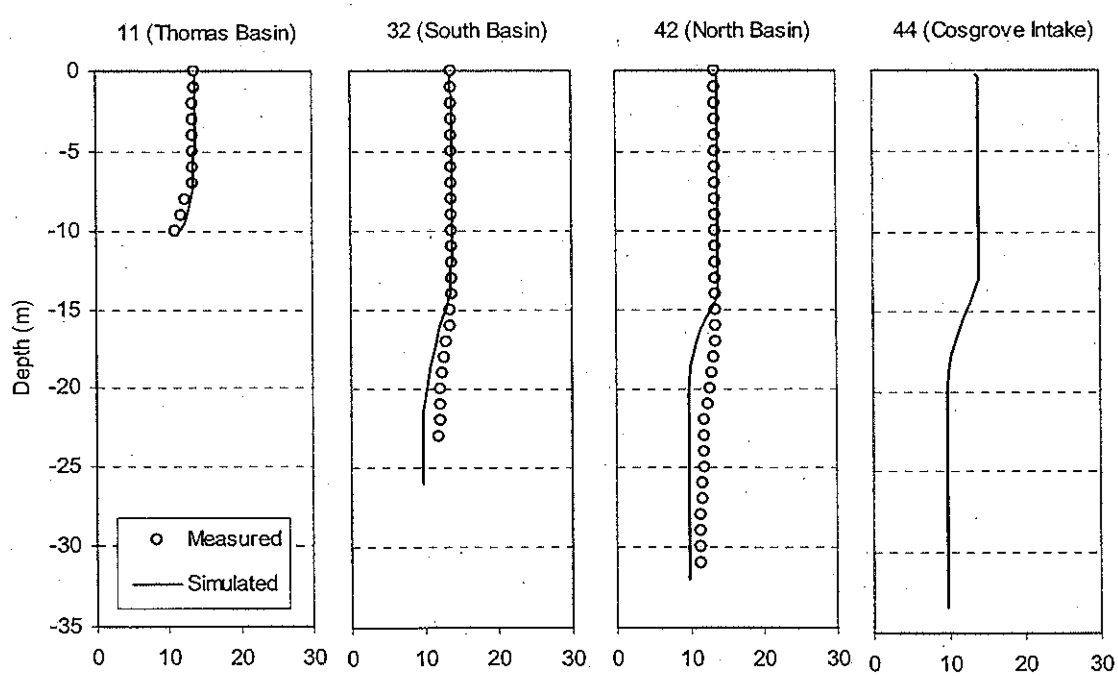

Figure 4.86 October 22, 2002 temperature profiles (°C)

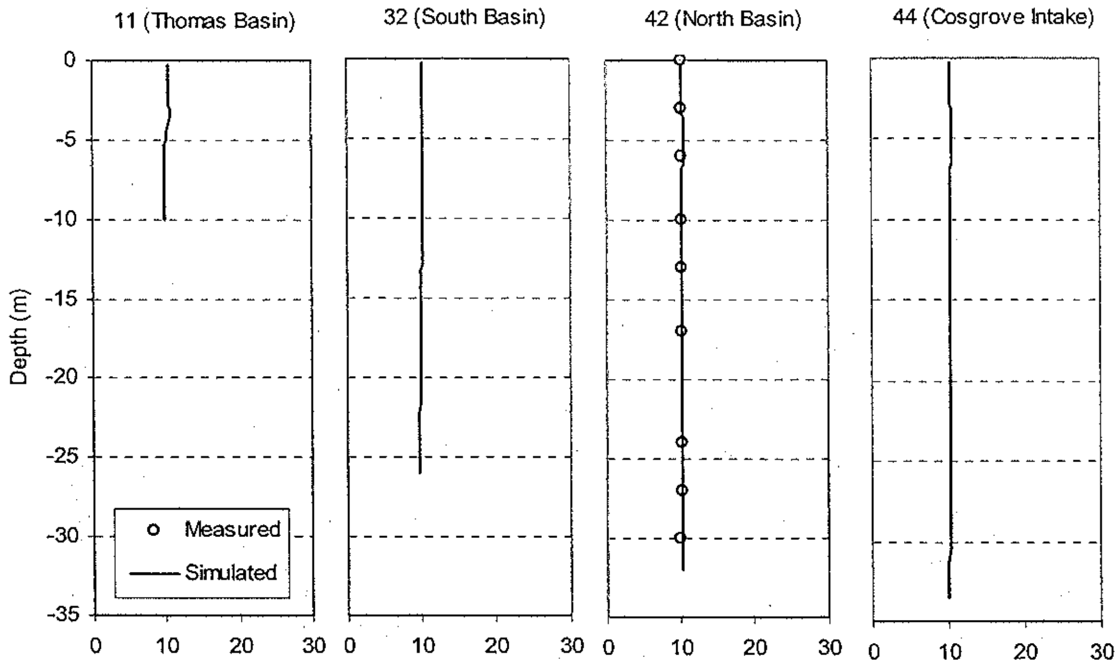

Figure 4.87 November 13, 2002 temperature profiles (°C)

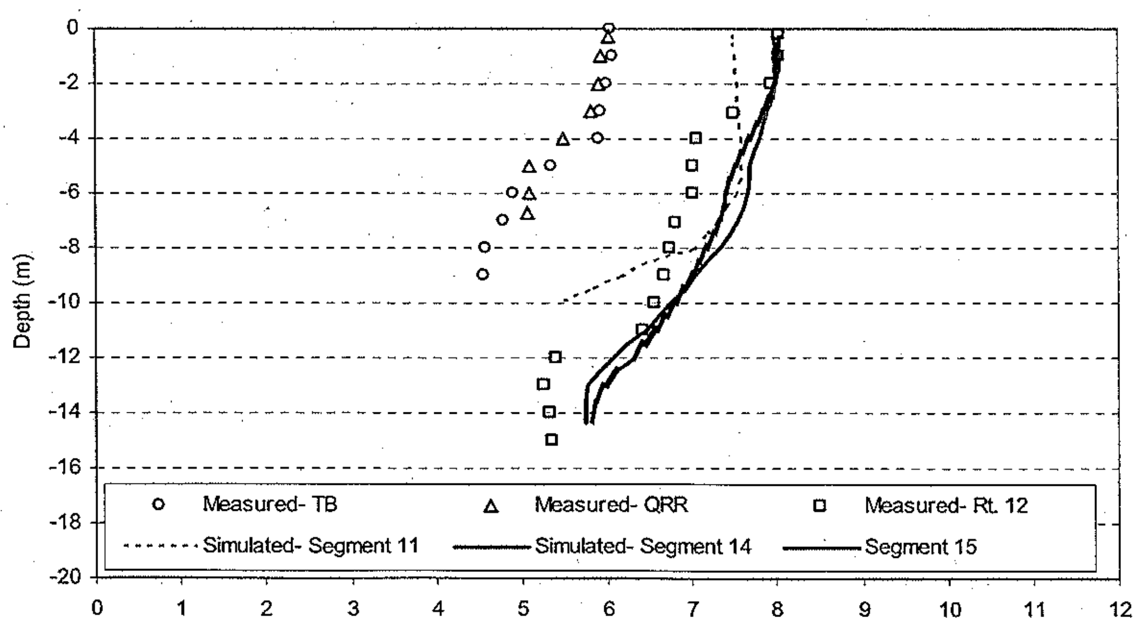

Figure 4.88 November 26, 2002 temperature profiles (°C)

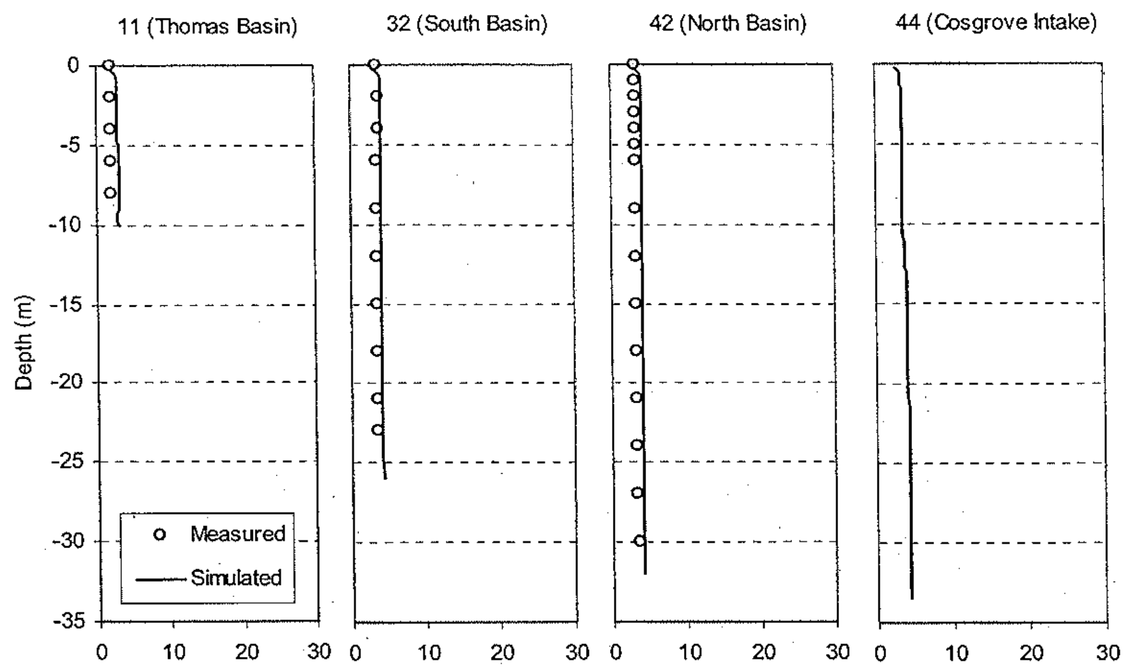

Figure 4.89 December 17, 2002 temperature profiles (°C)

4.3.3.2 Conductivity Profile Comparison

Figure 4.90 through Figure 4.100 present measured conductivity profiles in Wachusett Reservoir for 2002 at Thomas Basin, South Basin, North Basin, and in the vicinity of the Cosgrove intake, along with conductivity profiles predicted by CE QUAL W2 at those locations. The predicted profiles generally capture the magnitude and characteristics of the measured profiles for South Basin and North Basin, especially towards the beginning of the year. The minimum of conductivity resulting from hypolimnetic Quabbin Reservoir water deviates by no more than 2 m vertically and 10 μS/cm. Throughout the year, measured conductivity levels are slightly lower than those predicted by CE QUAL W2, except for the interval of minimum conductivity resulting from Quabbin Transfer in the reservoir metalimnion, which was characterized by generally lower levels than measured. Investigation indicated these deviations likely arose from inaccurately predicted vertical mixing.

It is important to note that although CE QUAL W2 predictions of Thomas Basin conductivity profiles were generally poor, the model did effectively capture the profiles measured under the Route 12 bridge on November 26 2002 (Figure 4.99), as the water column under the bridge is narrow and water beneath is more likely to approach lateral homogeneity than locations within the basin.

Vertical mixing in CE QUAL W2 is driven by wind, entered to the model through the meteorological input file. The model is highly sensitive to wind (Cole and Buchak, 1995). However, Version 2 of CE QUAL W2 as implemented allows for only a uniform wind sheltering coefficient. Analysis of wind patterns indicated increased frequency of wind from the northeast during the summer of 2002 (as compared to 2001). It is possible that the model as currently calibrated does not adequately account for mixing caused by wind from this direction. As the Wachusett Dam and North Dike are located in the northeast, winds blowing from this direction blow over the dam, which offers less shelter to the reservoir than the low hills and trees in other directions. It is possible that this inaccuracy is causing underprediction of mixing between the metalimnion and epilimnion, resulting in the predicted withdrawal of more low conductivity Quabbin water than occurred, and resulting in an accumulation of high conductivity water within the reservoir.

Figure 4.101 presents a time series plot of measured conductivity in water entering the Cosgrove Aqueduct, along with conductivity predicted with a wind sheltering coefficient (WSC) of 0.65 (as calibrated and presented in Joaquin, 2001 and Ahlfeld *et al.* 2003a), and as predicted with a wind sheltering coefficient of 1.0. The two predictions are consistent until July 24, when conductivity predicted with the smaller WSC begins to underpredict measured conductivity, and conductivity predicted with the larger WSC begins to overpredict measured conductivity. As a result of this analysis, CE QUAL W2 was reprogrammed to accept a WSC that varies depending on wind direction. However, results were inconclusive and the modification abandoned. As the model typically produces reliable conductivity predictions (see Sections 4.1 through 4.2, and Ahlfeld *et al.* 2003a) no adjustments were made to the WSC as implemented. It is notable that the underprediction of conductivity levels in the Cosgrove withdrawal during this period is similar to underprediction of UV254 (see UV254 calibration, Section 4.4.1) and DOC (see DOC calibration, Section 4.4.4) during the same period.

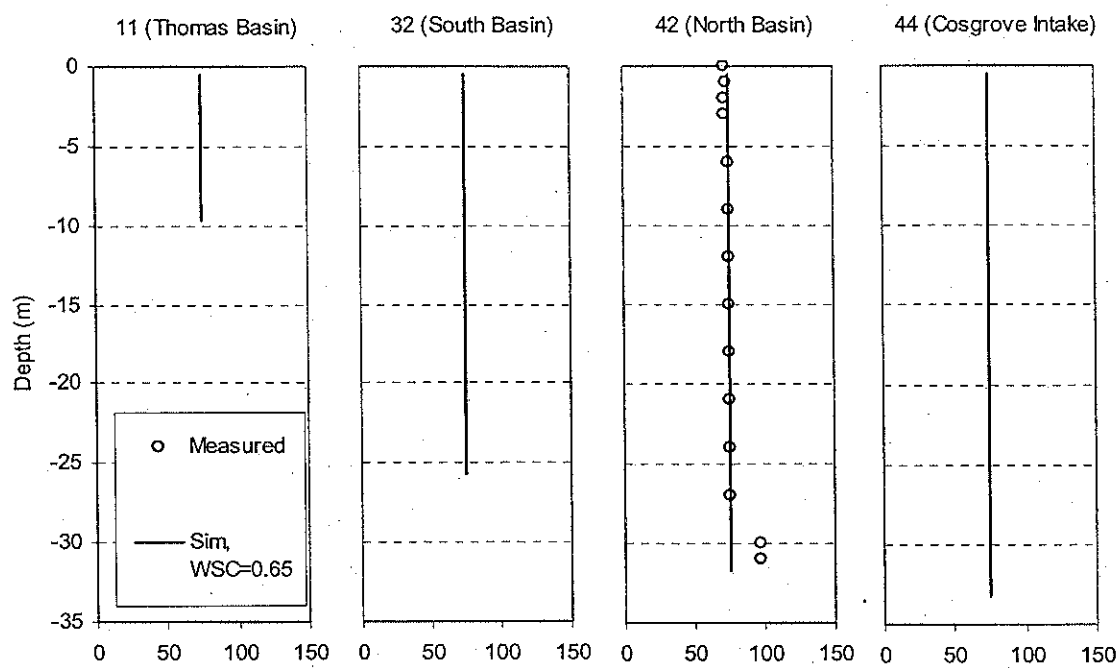

Figure 4.90 February 20, 2002 conductivity profiles (µS/cm)

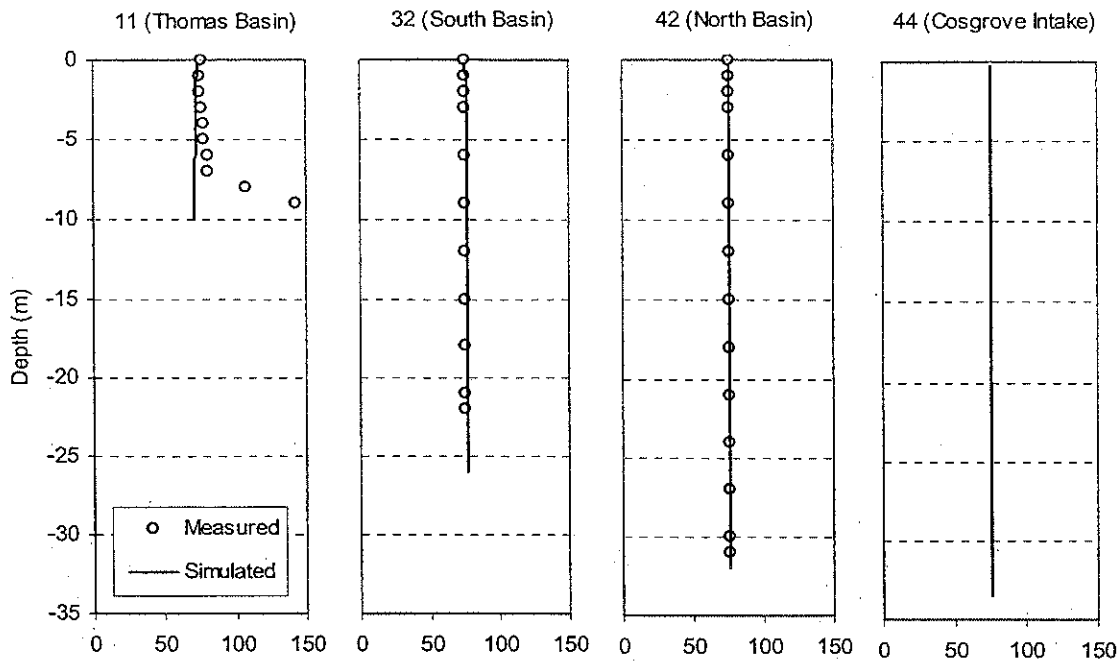

Figure 4.91 March 5, 2002 conductivity profiles (μS/cm)

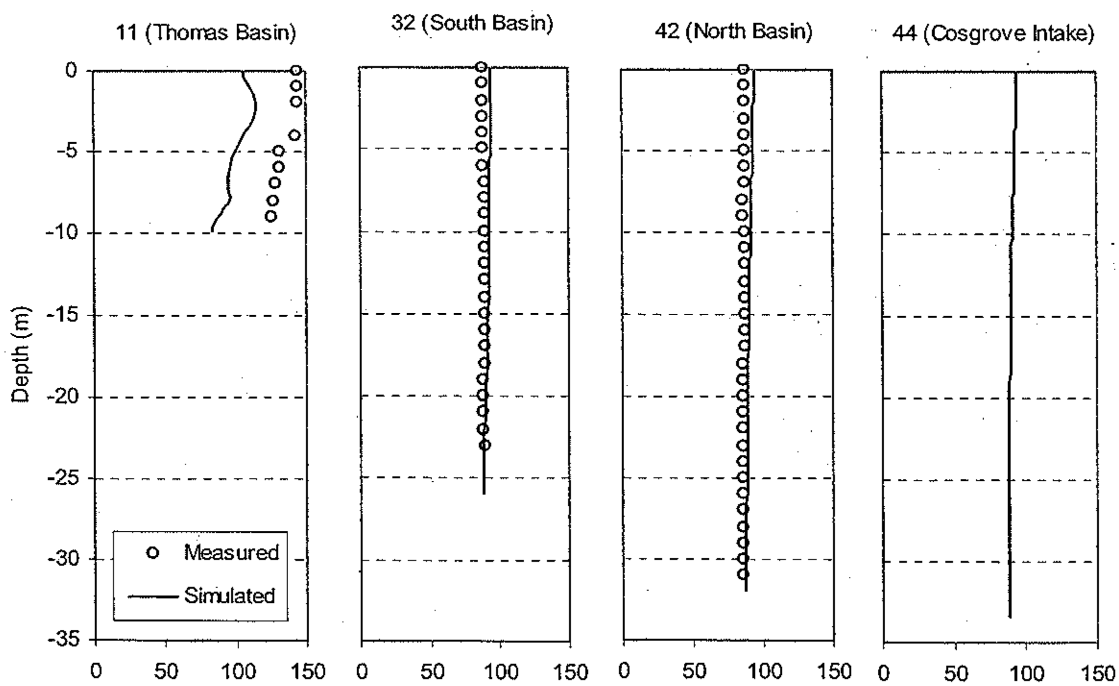

Figure 4.92 May 9, 2002 conductivity profiles (μS/cm)

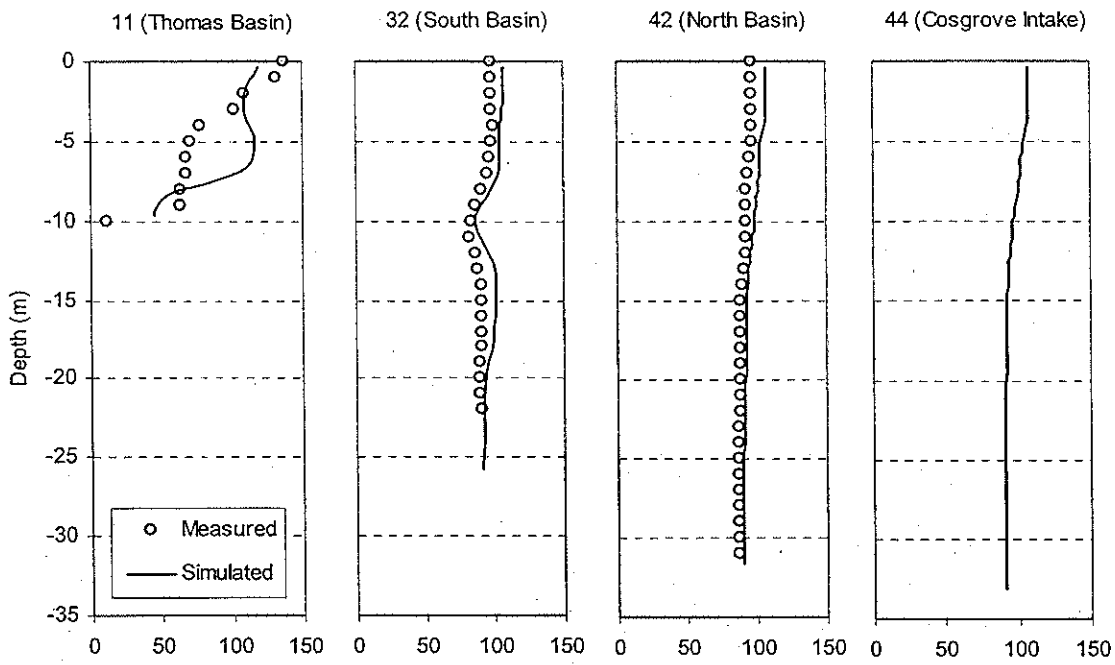

Figure 4.93 July 2, 2002 conductivity profiles (μS/cm)

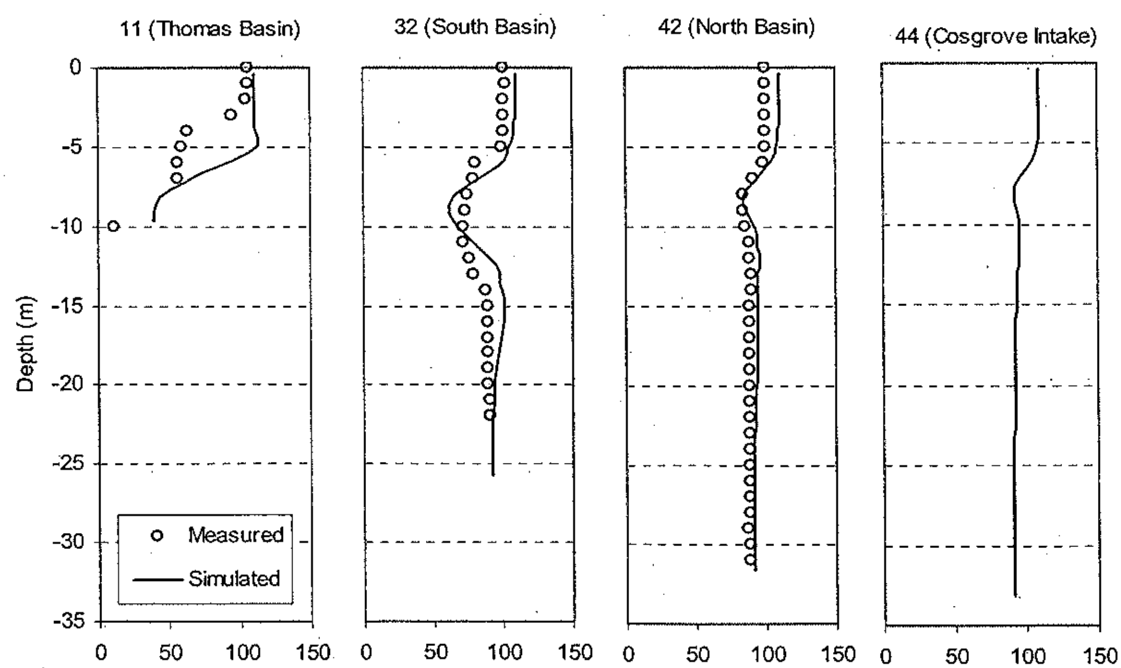

Figure 4.94 July 22, 2002 conductivity profiles (μS/cm)

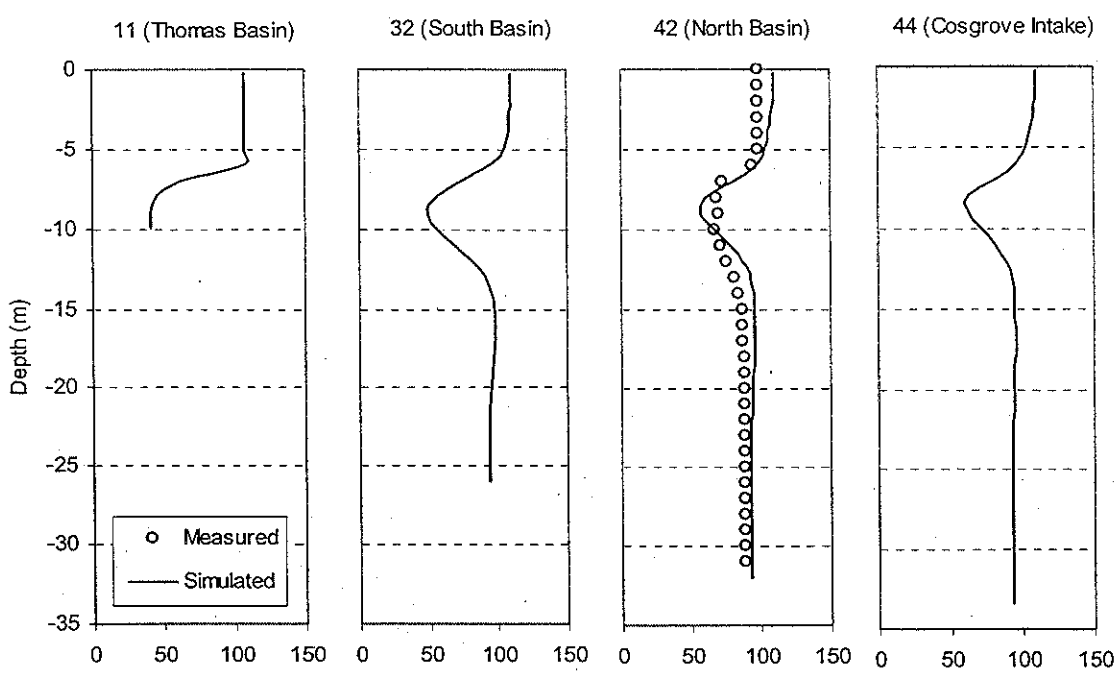

Figure 4.95 August 22, 2002 conductivity profiles (µS/cm)

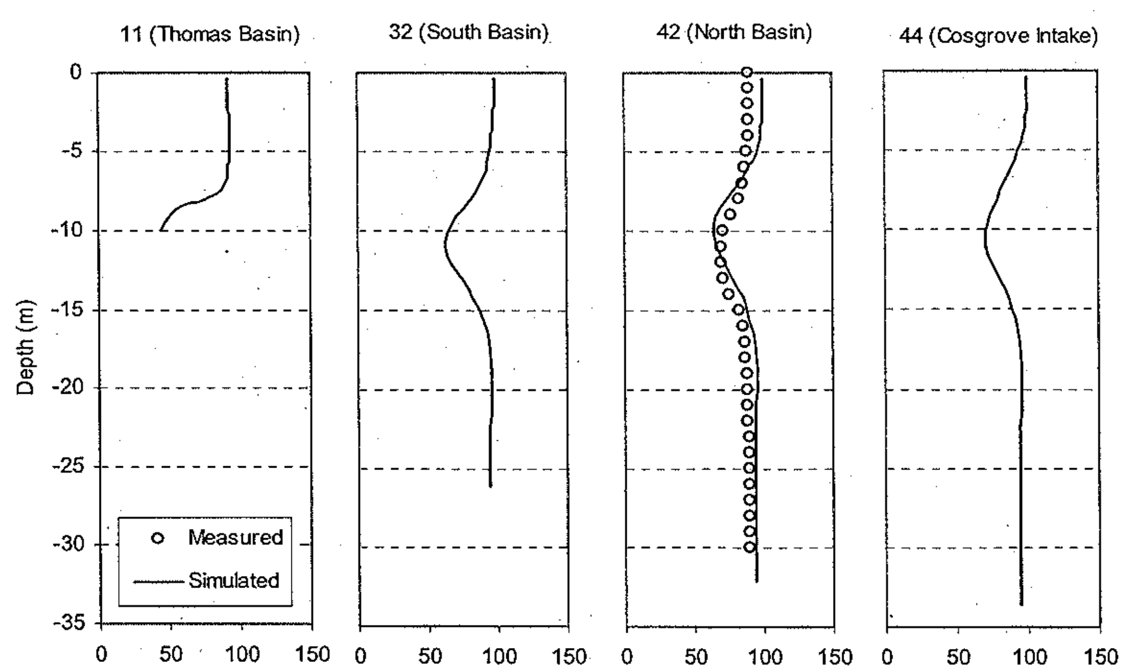

Figure 4.96 September 18, 2002 conductivity profiles (µS/cm)

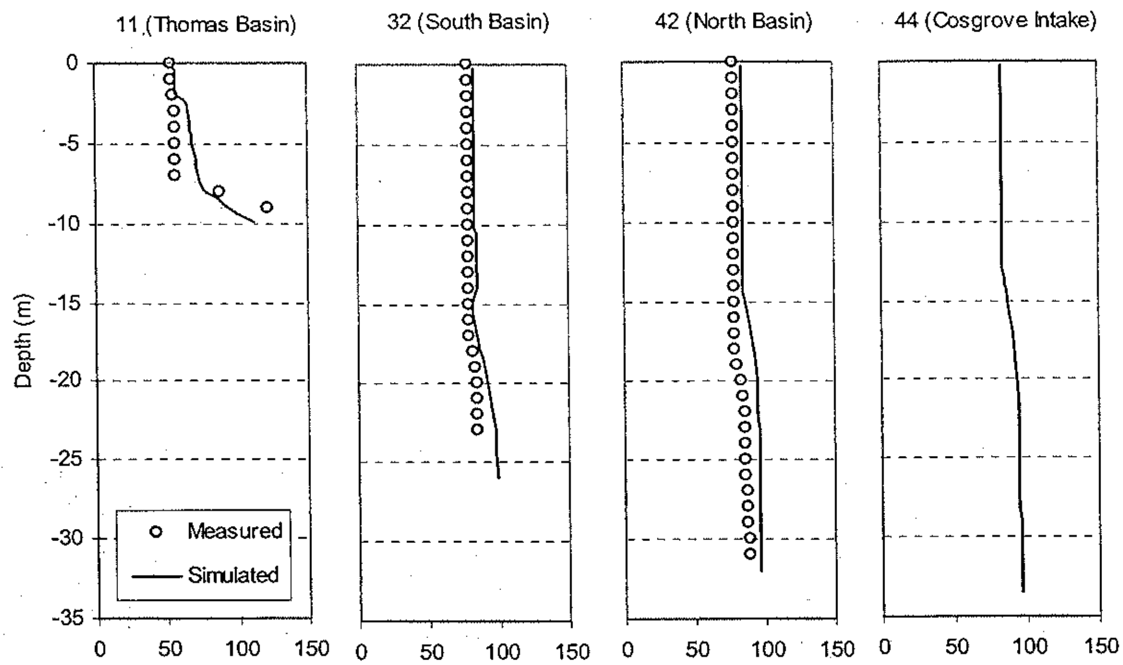

Figure 4.97 October 22, 2002 conductivity profiles (µS/cm)

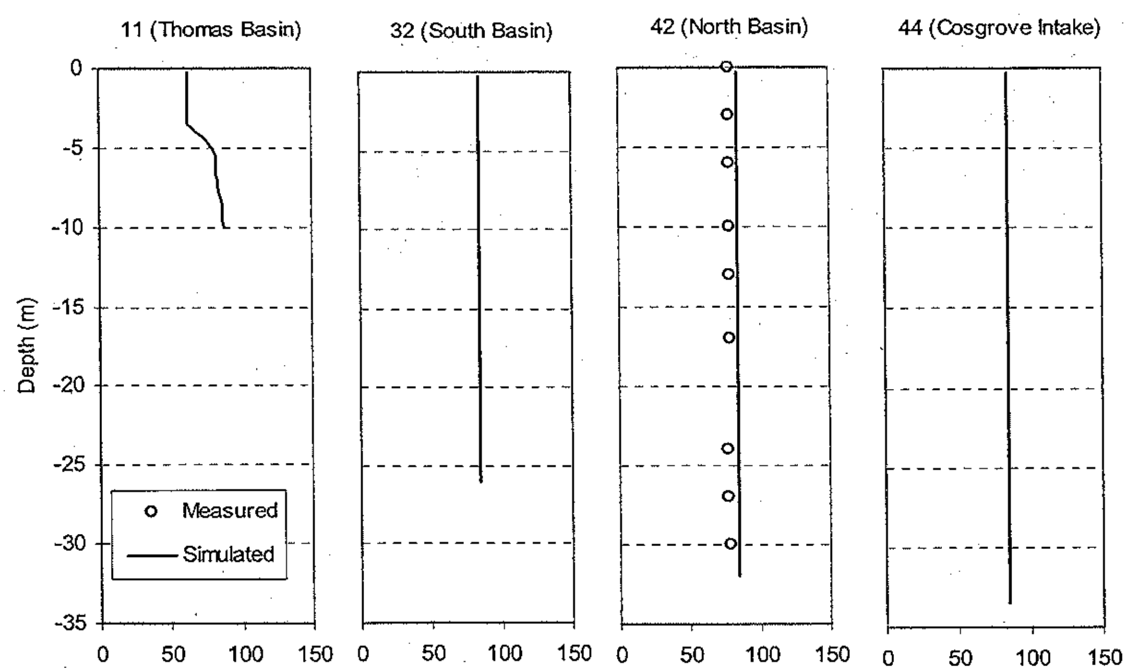

Figure 4.98 November 13, 2002 conductivity profiles (µS/cm)

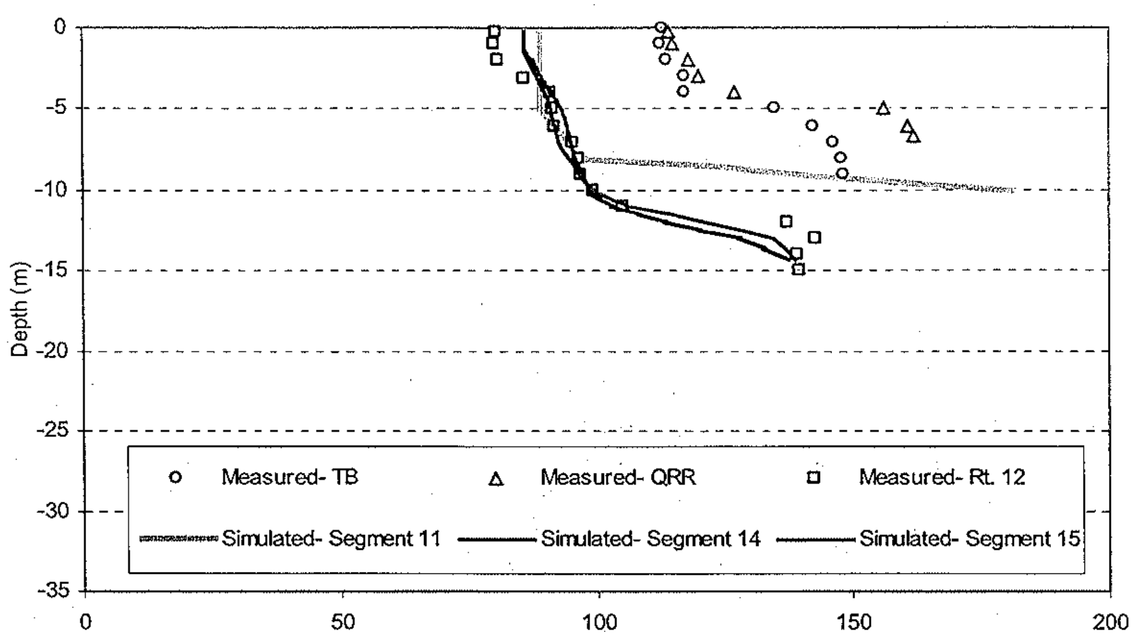

Figure 4.99 November 26, 2002 conductivity profiles (μS/cm)

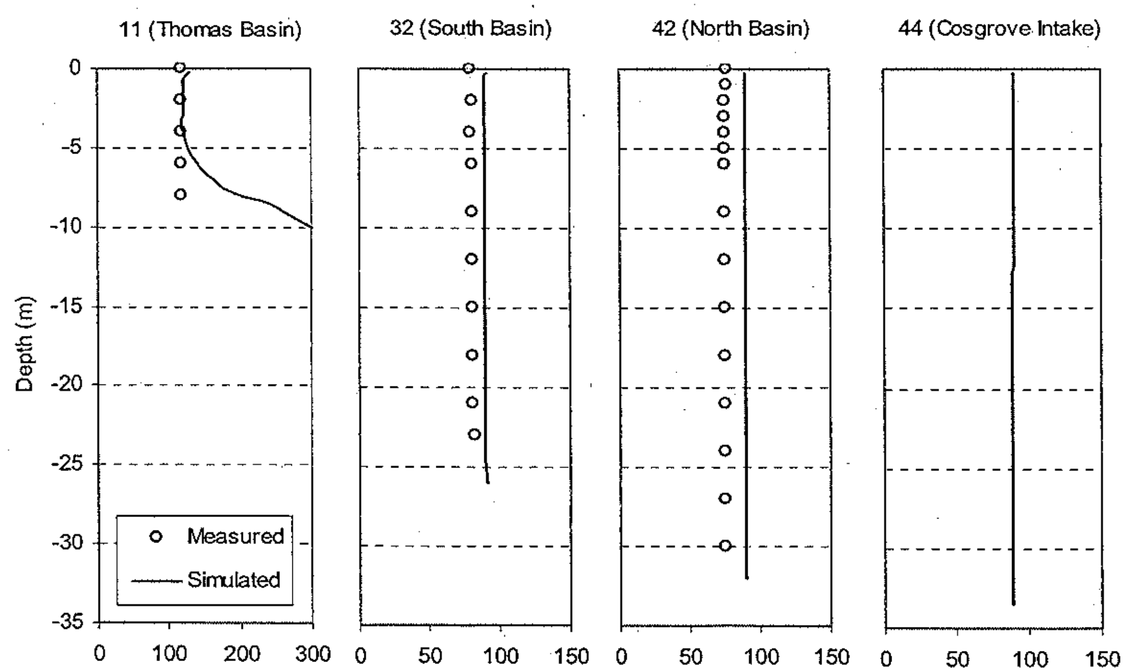

Figure 4.100 December 18, 2002 conductivity profiles (μS/cm)

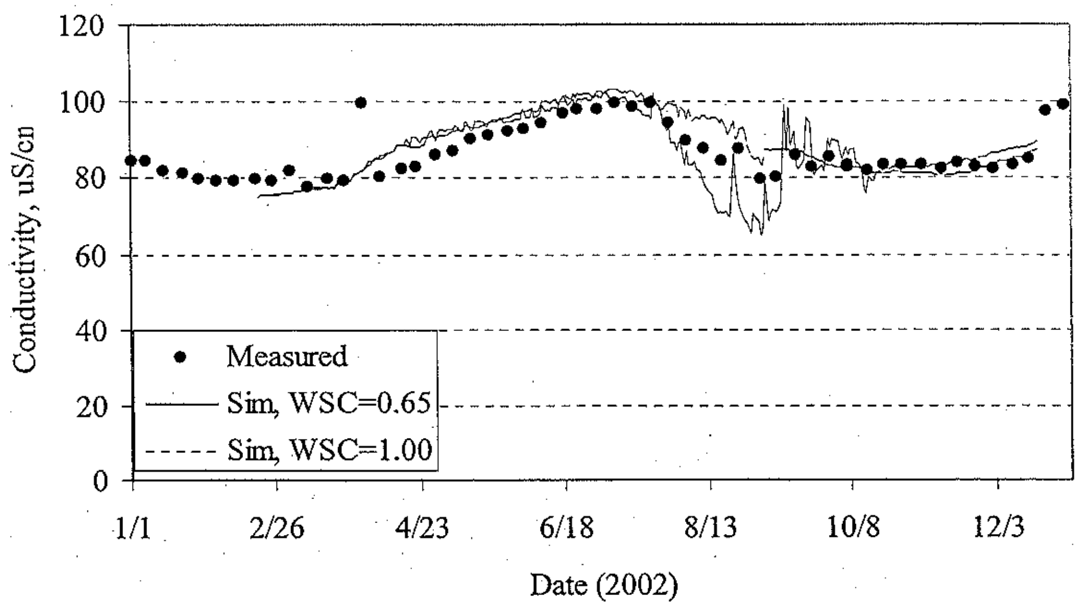

Figure 4.101 Measured conductivity at Cosgrove, with conductivity predicted using two wind sheltering coefficients.

4.4 Constituent Calibration

After successful calibration of the hydrodynamic model for Wachusett Reservoir (presented in Sections 4.1 through 4.3, as well as in CDM (1995); Joaquin (2001); and Ahlfeld *et al.* (2003a)), non-conservative water quality constituents were included. Constituents were modeled either as a series of interdependent compartments that relate to NOM (LDOM, RDOM, algae, detritus, and nutrients) or as an independent parameter impacted by allochthonous sources and decay only (UV254), as discussed in Section 3.3.

4.4.1 Organic Carbon Component Results

As discussed in detail in Section 3.3, CE QUAL W2 distinguishes between labile and refractory DOM. Autochthonous LDOM is generated by algal secretion and mortality and autochthonous RDOM is generated by decay from LDOM only, so these parameters were considered first during calibration.

Organic carbon data are relatively scarce for Wachusett Reservoir and its tributaries, so it was necessary to make several assumptions. Table 4.14 presents a summary of MWRA organic carbon data available for Wachusett Reservoir that was used in this research.

Table 4.14 MWRA Organic Carbon data available for Wachusett Reservoir.

Inflow	Type	Average (mg/L)	Standard Deviation (σ)	Range (mg/L)	Period (MM/YY), number of samples
Stillwater	TOC	4.62	1.5	1.97-8.67	01/00 - 12/02, 40
Quinapoxet	TOC	5.29	1.64	2.35-10.20	01/00 - 12/02, 39
CVA*	TOC	1.95	0.24	1.04-3.03	01/00 - 12/02, 227
Malden	TOC	4.08	2.7	1.45-15.30	01/00 - 11/02, 28
Gates	TOC	3.23	1.22	1.37-6.14	01/00 - 11/02, 30
French	TOC	8.12	3.58	3.77-14.50	01/00 - 11/02, 19
Malagasco	TOC	12.17	7.45	1.14-31.50	01/00 - 11/02, 28
West Boylston	TOC	2.61	1.42	1.03-5.93	01/00 - 11/02, 28
Muddy	TOC	4.17	1.84	2.02-10.20	01/00 - 11/02, 26
Outflow					
Cosgrove	TOC	2.39	0.37	1.56-4.64	01/00 - 12/02, 487
Cosgrove	DOC	2.48	0.43	1.20 - 6.11	01/00 - 12/00, 185

*No Quabbin Aqueduct constituent concentrations were measured so Chicopee Valley Aqueduct data is substituted.

It is notable that the highest concentrations of organic carbon are present in the smallest tributaries entering the reservoir. For example, TOC concentrations in Malagasco, French, and Malden Brooks all exceeded 10 mg/L. However, each of these tributaries is responsible for less than 1% of the reservoir water budget. Conversely, the largest sources of water are generally characterized by relatively low concentrations of organic carbon. Quabbin Aqueduct TOC concentrations were generally near 2 mg/L. Stillwater and Quinapoxet TOC concentrations, though generally not the lowest of the tributaries, are typically below 10 mg/L and average 4.6 and 5.3 mg/L, respectively. MWRA data also exists for water entering Cosgrove Aqueduct. Water at this location generally contains less than 3 mg/L of TOC and DOC, though occasionally higher levels are observed. The data also indicates that nearly all of organic carbon at Cosgrove is dissolved.

Table 4.15 presents organic carbon data, recorded by UMass, for the three largest inflows to Wachusett Reservoir on four dates in 2000 and 2001. This data is consistent with the MWRA data presented in Table 4.14 in terms of total concentrations. Additionally, DOC measurements allow the calculation of DOC to TOC ratios, as necessary for CE QUAL W2 input data. For Stillwater and Quinapoxet Rivers, the ratio of DOC to TOC ranged from 0.91 to 1 (omitting values greater than 1, which result from experimental error), indicating that most of the organic carbon entering from these sources is dissolved. DOC to TOC ratios for Quabbin Aqueduct were similar, ranging from 0.94 to ~1.

Table 4.15 UMass Organic Carbon data for major inflows of Wachusett Reservoir (Takiar 2001).

	Stillwater River			Quinapoxet River			Quabbin Aqueduct		
Date	TOC, mg/L	DOC, mg/L	DOC/ TOC	TOC, mg/L	DOC, mg/L	DOC/ TOC	TOC, mg/L	DOC, mg/L	DOC/ TOC
Jun-00	5.24	5.20	0.99	6.24	6.25	1.00	2.16	2.21	1.02
Sep-00	5.05	4.94	0.98	5.65	5.78	1.02	2.11	1.98	0.94
Jan-01	3.42	3.28	0.96	4.67	4.34	0.93	1.98	1.87	0.94
Apr-01	3.93	3.59	0.91	4.44	4.08	0.92	2.12	2.08	0.98

Table 4.16 presents additional tributary NOM data from a study of many of the Wachusett tributaries (Bryan 2005). The data shown are the average values from as many as six sampling dates at a location along the tributary, except for Stillwater River, which was sampled at four locations. Several of these tributaries are characterized by very low flow and are accounted for in the Wachusett Reservoir model as direct runoff, rather than as a tributary. The TOC and DOC values shown are within the ranges presented in Table 4.14 and Table 4.15. The largest average TOC concentration is for Malagasco Brook in both the MWRA data sets and the data from Bryan (2005) at about 10 mg/L. In both data sets, the Quinapoxet River contains a higher concentration of TOC than Stillwater (5.6 mg/L v. 4.1 – 4.6 mg/L). Most significantly, ratios of DOC to TOC are between 0.92 and 1, which is within the ranges of data from MWRA and Takiar (2001).

Table 4.16 TOC and DOC sampling data for Wachusett Reservoir tributaries, based on thrice-yearly data from 2001 - 2005 (Bryan 2005)

Tributary	TOC, mg/L		DOC, mg/L			
	Value	σ	Value	σ	n	DOC/TOC
Stillwater	4.12	1.75	3.90	1.59	19	0.95
Waushacum	5.34	1.42	4.89	1.39	6	0.92
Quinapoxet	5.34	2.31	5.06	2.28	6	0.95
Malden	3.03	2.45	2.88	2.51	6	0.95
Gates	3.76	2.79	3.44	2.76	6	0.92
French	7.62	2.28	7.33	2.04	6	0.96
Malagasco	10.95	6.91	10.97	7.43	6	1.00
Muddy	3.68	2.60	3.46	2.72	5	0.94
Justice Brook	4.79	1.08	4.93	1.47	5	1.03
Houghton Brook	5.06	3.10	4.94	3.16	4	0.98
Scalon Brook	5.18	3.75	5.15	3.93	4	0.99
Ball Brook	5.03	3.96	5.04	3.87	4	1.00
Wachusett Brook	4.39	2.63	4.32	2.63	3	0.99
Rocky Brook	4.58	3.47	5.24	4.38	3	1.15
Bailey Brook	2.19	0.23	2.03	0.25	2	0.93
Keyes Brook	3.89	n/a	4.15	n/a	1	1.07

In September and December 2004, epilimnetic, metalimnetic, and hypolimnetic samples were collected at three Wachusett Reservoir sampling locations corresponding to the Route 12 bridge at the southeastern boundary of Thomas Basin, and the South and North Basin sampling stations. These samples were analyzed for TOC, DOC, and UV254 as snapshots of Wachusett Reservoir conditions on those dates. Table 4.17 presents a summary of the organic carbon portion of this data, locationally averaged.

Table 4.17 In-reservoir organic carbon data (mg/L) for Wachusett Reservoir

Date	Depth	Thomas Basin			South Basin			North Basin		
		TOC	DOC	DOC/TOC	TOC	DOC	DOC/TOC	TOC	DOC	DOC/TOC
9/30/04	E	2.13	2.00	0.94	2.02	2.04	1.01	2.11	2.14	1.02
9/30/04	M	2.51	2.36	0.94	2.02	2.06	1.02	1.76	1.79	1.01
9/30/04	H	2.82	2.64	0.94	1.89	1.86	0.98	1.96	2.08	1.06
12/23/04	E	4.41	4.63	1.05	2.34	2.27	0.97	1.96	1.94	0.99
12/23/04	M	4.67	4.55	0.97	2.23	2.19	0.98	1.96	1.86	0.95
12/23/04	H	4.59	4.45	0.97	2.33	2.22	0.95	1.93	2.01	1.04

Note that the Thomas Basin organic carbon concentrations for December 23, 2004 are between 1.6 and 2 times higher than in September of that year. This difference is most likely the result of the diluting effect of low NOM Quabbin Transfer, which was occurring on and prior to the September sampling but not prior to the December sampling. On both dates there was a slight but noticeable difference in organic carbon concentrations between the three sampling locations, with Thomas Basin concentrations being the highest and North Basin concentrations the lowest. This difference indicates a net loss in organic carbon across the reservoir on these dates. Conversely, the ratio of DOC to TOC generally increased between the Thomas Basin and North Basin sampling locations. This is likely due to settling of allochthonous POC along the length of the reservoir. Algal growth and mortality may reverse this trend during the summer months, but no data exists to support this conjecture.

4.4.1.1 Organic Carbon Characterization Estimates for Model Input

As presented in Section 3.3, CE QUAL W2 models organic carbon as DOM and POM. Since the majority of tributary NOM data is in the form of TOC, and since DOC data is scarce, it is necessary to estimate the fraction of NOM as DOM and POM. All data from MWRA, Takiar (2001) and this study show that the DOC:TOC ratio is between 0.90 and 1, and generally between 0.94 and 1. Further, ratios of DOC to TOC are generally higher near Cosgrove and are generally lower near Thomas Basin. Thus, it was assumed that DOM was 95% of TOC for all inflow data, and that the remaining 5% was POM. These ratios were the same used for modeling Quabbin Reservoir (Roberts 2003).

It was further necessary to divide these fractions for use with CE QUAL W2. As no data characterizing POM was available, it was assumed that 100% of allochthonous POM is in the form of nonliving POM, or Labile POM as modeled by CE QUAL W2. This implies that there are no allochthonous algae sources. Most likely, autochthonous algal growth would dominate allochthonous algae, and the quantity of allochthonous POM is so small that this assumption is likely reasonable.

More care must be taken when determining the ratio of labile to refractory DOM. Very little DOC characterization data exists for Wachusett Reservoir. Hodgkins (1999) measured biodegradable dissolved organic carbon (BDOC) at various points in the

MWRA supply and distribution system, including measurements of Cosgrove Aqueduct DOC and BDOC. A summary of those results is presented in Table 4.18

Table 4.18 Summary of DOC and BDOC data at Cosgrove (Hodgkins 1999).

Date	Cosgrove Aqueduct		
	DOC	BDOC	BDOC/DOC
10/23/97	2.11	0.27	0.13
5/26/98	2.80	0.58	0.21

The data indicates that a relatively small percentage of DOC is biodegradable. Roberts (2003) performed a sensitivity analysis of results of a CE QUAL W2 model of Quabbin Reservoir to tributary ratios of refractory and labile DOM. That analysis determined that assuming 20% of DOM to be labile and the remaining 80% to be refractory was most appropriate. This assumption was implemented for all inputs in this study for the majority of the scenarios presented in Section 4.4. However, organic matter in Quabbin water has decayed for a long period and may be characterized by a larger refractory DOM to labile DOM ratio. An analysis of the impact of varied Quabbin Transfer RDOM to LDOM ratio is presented in Section 4.4.3. Figure 4.102 presents a flowchart for the divisions of NOM input to CE QUAL W2 in this study.

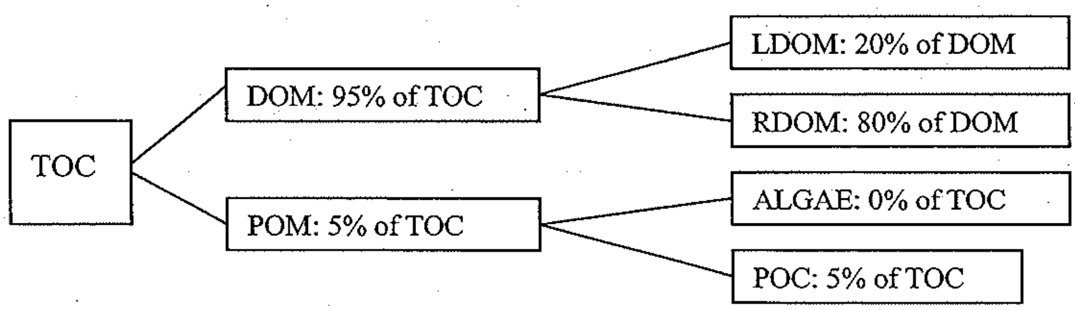

Figure 4.102 Characterization of NOM for tributary inputs to CE QUAL W2 Wachusett model

Additionally, Garvey (2000) estimated precipitation DOC to be 1.3 mg/L on average. This concentration is within the range suggested by Jordan and Likens (1975) and was used for the Wachusett model. LDOM was assumed to constitute 20% of DOC from this source while RDOM was assumed to constitute the remaining 80%.

Figure 4.103 presents a time series plot of the TOC concentration of water entering Cosgrove Aqueduct for the 2001 and 2002 study period used for constituent calibration. Notable features of this trend include a net loss of ~0.5 mg/L TOC (approximately 20%) from the beginning of 2001 to the end of 2002. Between these dates, there are two notable peaks; the first occurs in late May 2001 at ~3.3 mg/L, and the other in late July 2002 at ~2.5 mg/L.

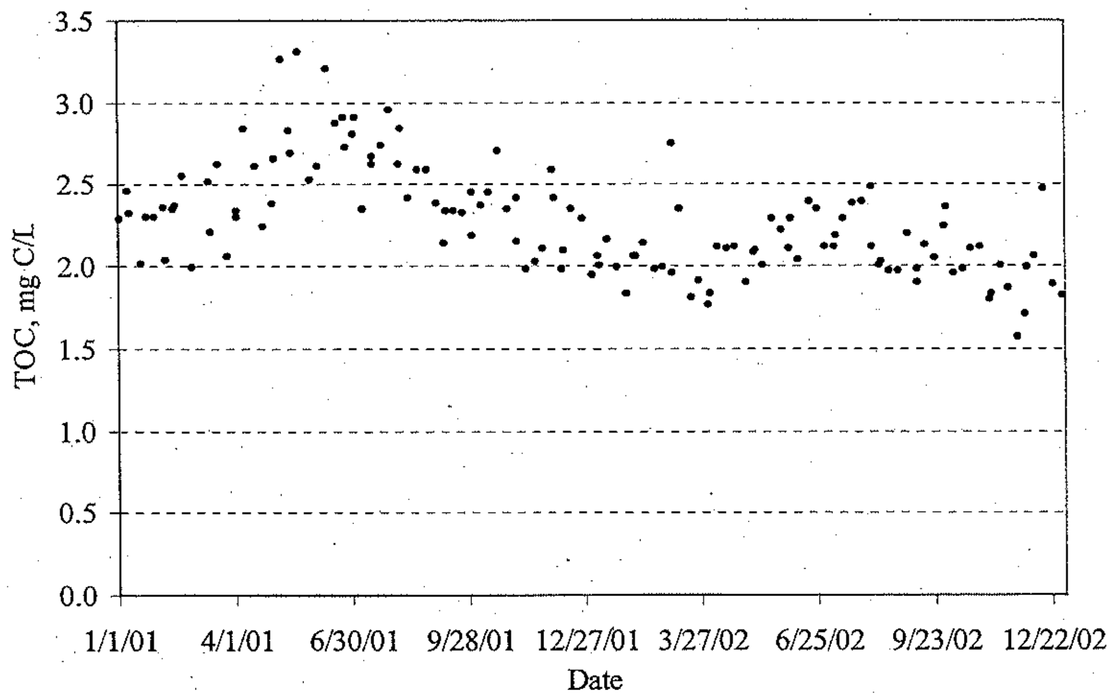

Figure 4.103 Measured TOC concentrations at the Cosgrove Intake of Wachusett Reservoir

4.4.1.2 DOC Calibration Results

Initial DOM concentration was first estimated from measured TOC at Cosgrove corresponding to the start of the model run. DOC was estimated from this value based on the relationship presented in Figure 4.102. The resulting DOM concentration was determined to be 2.18 mg/L. It was then estimated that 90% of the DOM in Wachusett is refractory, on average. This value was selected because LDOM decays at a faster rate than RDOM; although it is assumed that 20% of the DOM entering Wachusett is labile, a lower percentage of the DOM in the reservoir could be characterized as such.

Figure 4.104 shows model results for three cases of DOM decay and a series of TOC data at Cosgrove, meant for use as a reference. In the first of these cases, designated 'No Decay,' the labile DOM decay rate (LDOMDK), refractory DOM decay rate (RDOMDK), and labile to refractory decay rate (LRDK) were all set to zero, forming a case where advection is the only pathway of DOM loss. In this case, there is almost no net change in RDOM, and LDOM accumulates to about 200% of the initial value.

Algae was included as a predicted constituent in these runs; algal processes contribute LDOM, so this impact was included. The algal parameters were not yet calibrated however, adding uncertainty to this analysis.

It is notable that DOM component trends with no decay are similar to measured TOC trends with no DOM decay or POC components included. These trends include peak levels in summer, followed by a decline in fall and winter. Although LDOM is predicted to accumulate, after August 2001 there is little net change in this constituent (i.e. the output concentration has approached the conservative input concentration). Predicted RDOM concentrations (no decay) remain relatively constant as well. Roberts (2003) showed that modeling OM components with no decay resulted in a steady accumulation of organic matter within Quabbin Reservoir, a result that differs significantly from these findings for Wachusett. It is therefore apparent that advection is a dominant process in organic matter dynamics in Wachusett, and that influent organic matter strongly influences outlet concentrations.

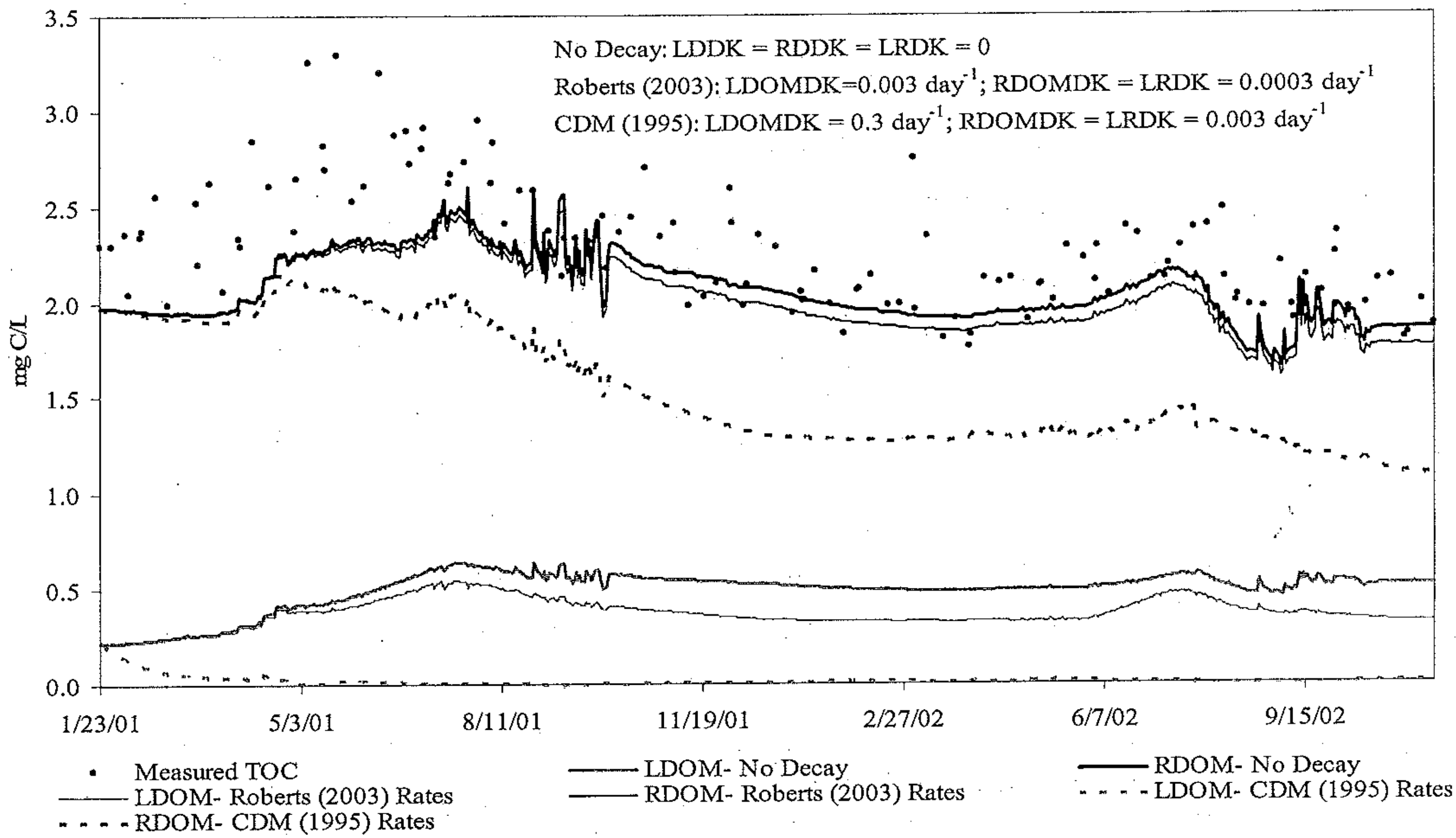

Figure 4.104 Measured TOC and modeled DOC at Cosgrove, using no decay, decay rates from Roberts (2003) and rates from CDM (1995)

In the second case, DOM decay rates are set to those used by Roberts (2003) in her study of Quabbin Reservoir. The model predicts RDOM concentrations to decline slightly during the year with RDOMDK =0.0003 day^{-1}, but RDOM on December 31 is only about 1% lower than the initial value of 1.97 mg/L. This decline does not mirror the decline observed in the data. Because LDOMDK is an order of magnitude larger (0.003 day^{-1}), the net gain in LDOM during the year is only 46% with this decay rate. It is important to note that the net LDOM decay rate is actually 0.0033 day^{-1} due to LRDK = 0.0003 day^{-1} in this case, and that net RDOM decay is somewhat less than 0.0003 day^{-1} due to receiving some of the labile DOM decay product. Considering that decay rates approximate average conditions, the shorter Wachusett detention time reduces the need to account for the extremely slow decay of very refractory components that is an important part of NOM dynamics in Quabbin.

The final case presented in this figure show results of decay rates chosen from literature by CDM (1995). With LDOMDK= 0.3, and LRDK and RDOMDK = 0.003, these values conform to the suggestion by Cole and Buchak (1995) that the labile decay rate be two orders of magnitude larger than that for refractory DOM. However, as a result, the model predicts labile and refractory DOM concentrations declining to a degree not supported by data.

The modeled DOM concentrations resulting from the varied decay rate scenarios demonstrate the ability of the model to predict the general trends of DOM at Cosgrove with reasonable accuracy. Each model prediction shows peaks in concentration in the late summer and early fall that approximately correspond to peaks in TOC during those periods. These results suggest that further investigation of DOM decay rates is appropriate.

Three more decay rate scenarios for DOM were investigated, as shown in Figure 4.105. In one case, decay rates between the very low values required for Quabbin (Roberts 2003) and the larger values identified in the literature (see Section 2.5.1) were selected. Changing LDOMDK to 0.03 day^{-1} and retaining the LRDK and RDOMDK rates as used for Quabbin (0.0003 day^{-1}) maintains conformity with the Cole and Buchak (1995) criteria of the labile decay rate being about two orders of magnitude larger than the refractory decay rate. These decay rates predicted a decrease in RDOM from 1.97 mg/L to 1.75 mg/L, and a decrease in LDOM from 0.22 to 0.06 mg/L. The decline in RDOM throughout the year seemed somewhat low, while the decline in LDOM seemed somewhat high.

The LDOM decay rate was then halved, and the RDOM and LR decay rates doubled in an attempt to improve fit. The selected values of LDOMDK = 0.015 day^{-1}, and LRDK and RDOMDK = 0.0006 day^{-1} resulted in the series designated 'case 2' in Figure 4.105. These rates resulted in a RDOM decline from 1.97 mg/L to 1.67 mg/L, a decline 36% larger than with RDOMDK= 0.0003 day^{-1}. Additionally, this scenario resulted in a predicted LDOM concentration of 0.12 mg/L at the end of the year.

A third set of DOM decay rates was then implemented, shown as 'case 3' in Figure 4.105. These rates were determined by further reducing LDOMDK and increasing RDOMDK and LRDK (still set at equal values) such that the labile and refractory decay rates only differed by one order of magnitude, considered by Cole and Buchak (1995) to be the smallest acceptable difference between these rates. Values of 0.008 day^{-1} for LDOMDK and 0.0008 day^{-1} for RDOMDK and LRDK, about 2.7 times the rates determined by Roberts (2003) for Quabbin were selected. These values result in an end of year concentration of 0.19 mg/L of LDOM and 1.62 mg/L of RDOM.

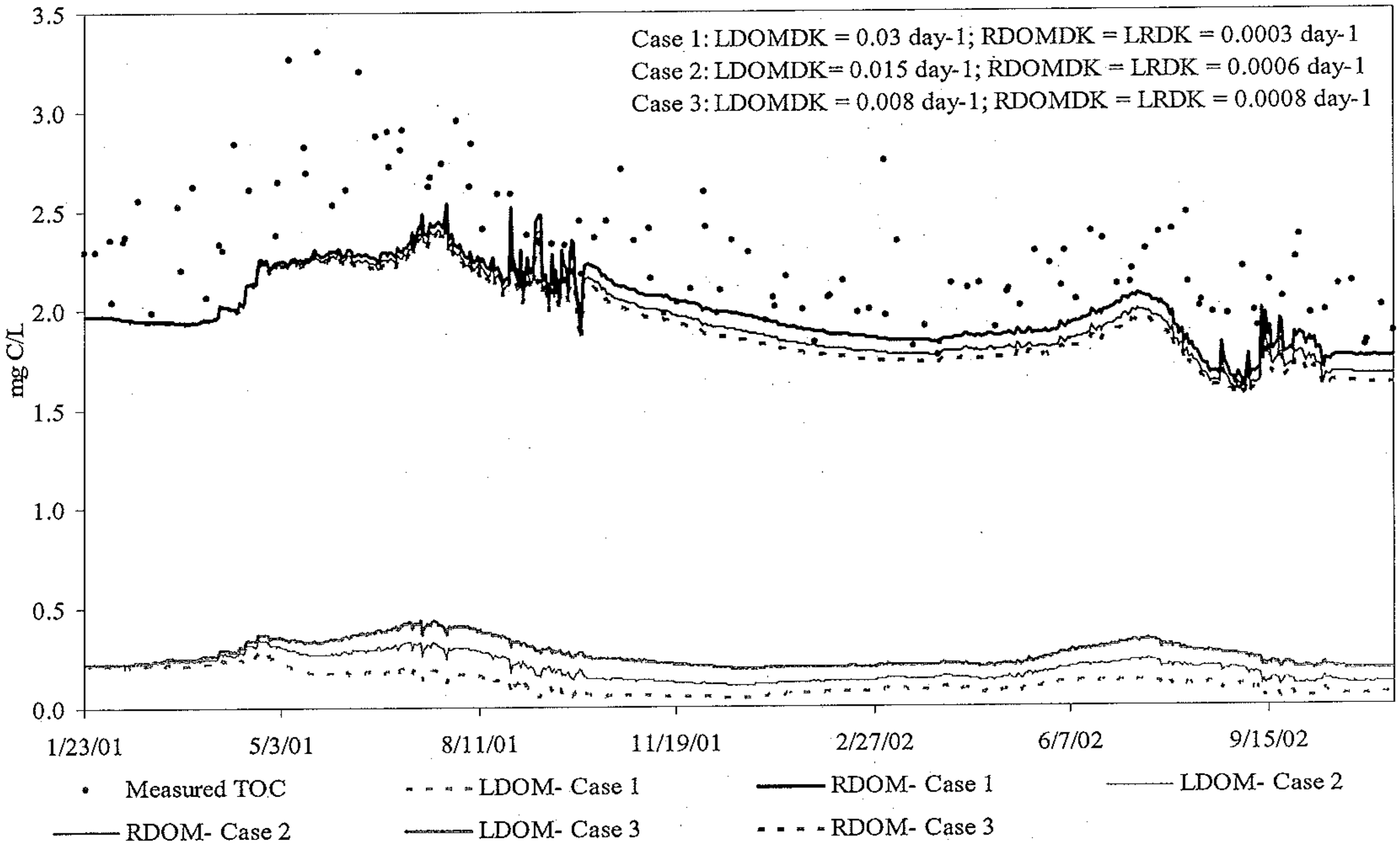

Figure 4.105 Measured TOC and modeled DOM at Cosgrove for three scenarios of varied DOM parameter decay rates.

The results shown in Figure 4.105 are similar to each other and difficult to evaluate considering the scarcity of DOM data for the system. A net loss of RDOM occurs in all three cases, although the loss is slightly larger for Case 3, which consists of the largest refractory DOM decay rate (a 19% loss in Case 3 occurred as compared to 11% and 15% losses for Cases 1 and 2, respectively). In Cases 1 and 2, with higher rates of labile DOM decay than Case 3 (LDOMDK=0.03 day^{-1} in Case 1 and 0.015 day^{-1} in Case 2, as compared to 0.008 day^{-1} in Case 3), the model predicts a net loss of LDOM throughout the two year interval (losses of 72% and 47%, respectively), while a loss of only 20% occurs in Case 3, which is proportional to the approximately 20% net reduction of TOC as measured at Cosgrove during that period.

It is notable that light induced decay of biologically recalcitrant organic matter has been shown to produce organic matter that is more bioavailable (see Section 2.5.2). This is akin to adding a light induced decay pathway from RDOM to LDOM in CE QUAL W2. This pathway was added and is identical to the light induced decay pathway of UV254. Details are presented in Section 3.3.3. This modification required the introduction of OMALP, a parameter that relates the impact of irradiance to the decay of RDOM (α_{om} in Equation 3.16). As organic matter data for Wachusett is scarce, assuming that LRDK = 0 was necessary to prevent feedback between the two parameters. However, LRDK = 0 eliminates autochthonous formation of RDOM in the model, which may not be representative of the natural system. RDOMDK was also set equal to zero for the initial calibration. Figure 4.106 presents a time series plot of RDOM and LDOM as modeled with RDOMDK = LRDK = 0.0008 day^{-1} and LDOMDK = 0.008 day^{-1} (as presented in Figure 4.105), and for two cases as modeled with OMALP = 1.3E-5 cm^2/cal. (LDOMDK = 0.012 day^{-1} and LDOMDK = 0.016 day^{-1}). It was necessary to increase LDOMDK to account for the gain in LDOM from RDOM and to account for the loss of the LRDK pathway. The temperature dependent decay and light induced decay pathways produce similar results. DOM in both scenarios follow the trends of the data, and the light induced decay scenario predicts the same net change in RDOM throughout the year (-0.35 mg/L), although slightly lower concentrations are predicted in the summer and slightly higher concentrations are predicted in winter.

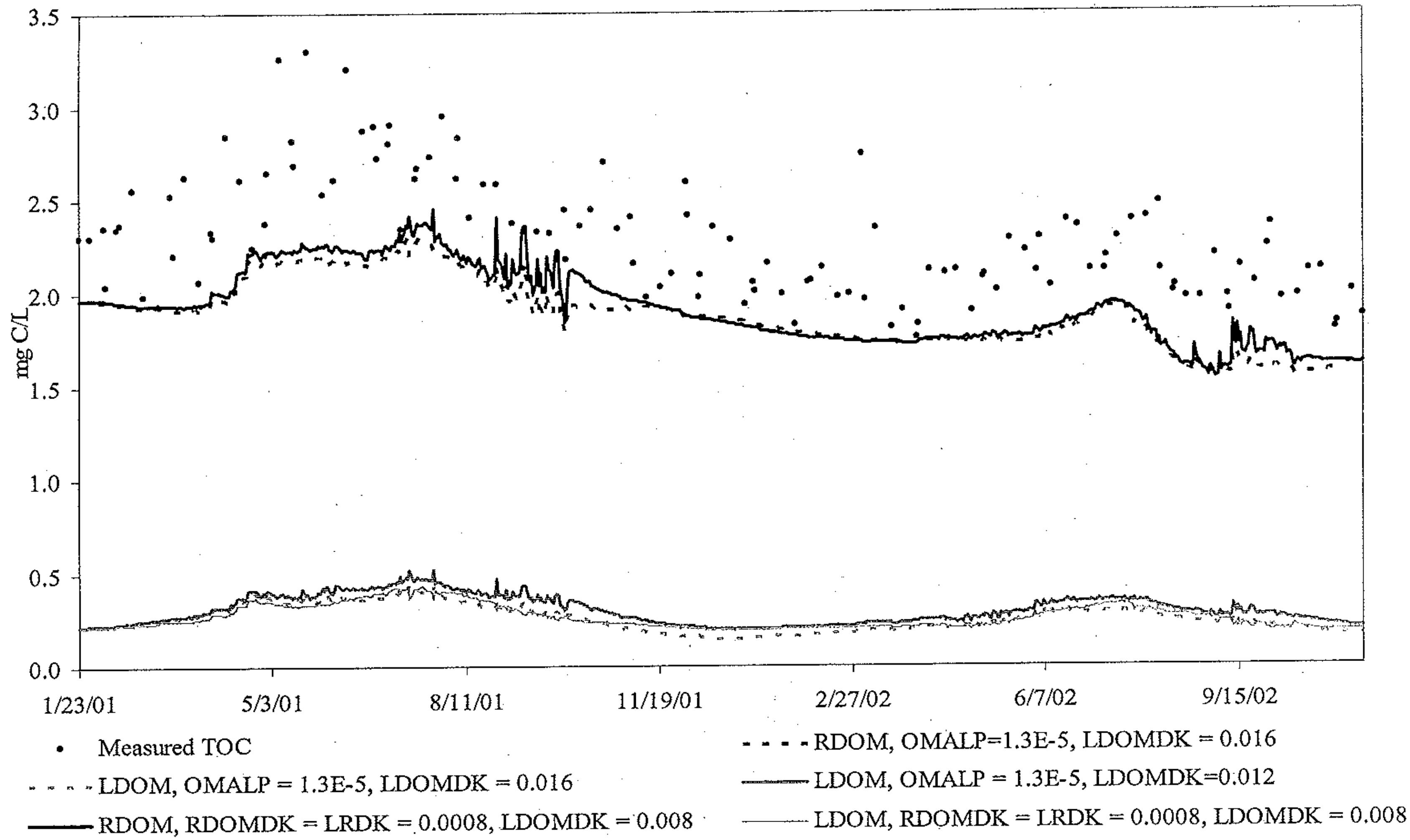

Figure 4.106 Measured TOC with modeled DOM for one value of OMALP decay and two values of LDOMDK.

Figure 4.107 presents the impact of variation in the Quabbin transfer refractory DOM to labile DOM ratio to levels of those constituents at Cosgrove (RDOM to LDOM = 80:20 was used for results presented in Figure 4.104 through Figure 4.106). Results of modeling these conditions with CE QUAL W2 are as expected; as the RDOM fraction of Quabbin Transfer DOM increases, predicted RDOM levels at Cosgrove increase as predicted LDOM levels decrease. Increasing the ratio from 80:20 to 95:5 increased predicted RDOM levels at the end of 2002 from 1.62 to 1.83 mg/L, and decreased end of the year LDOM levels from 0.19 to 0.10 mg/L. These effect of varying the ratio is noticeable but small.

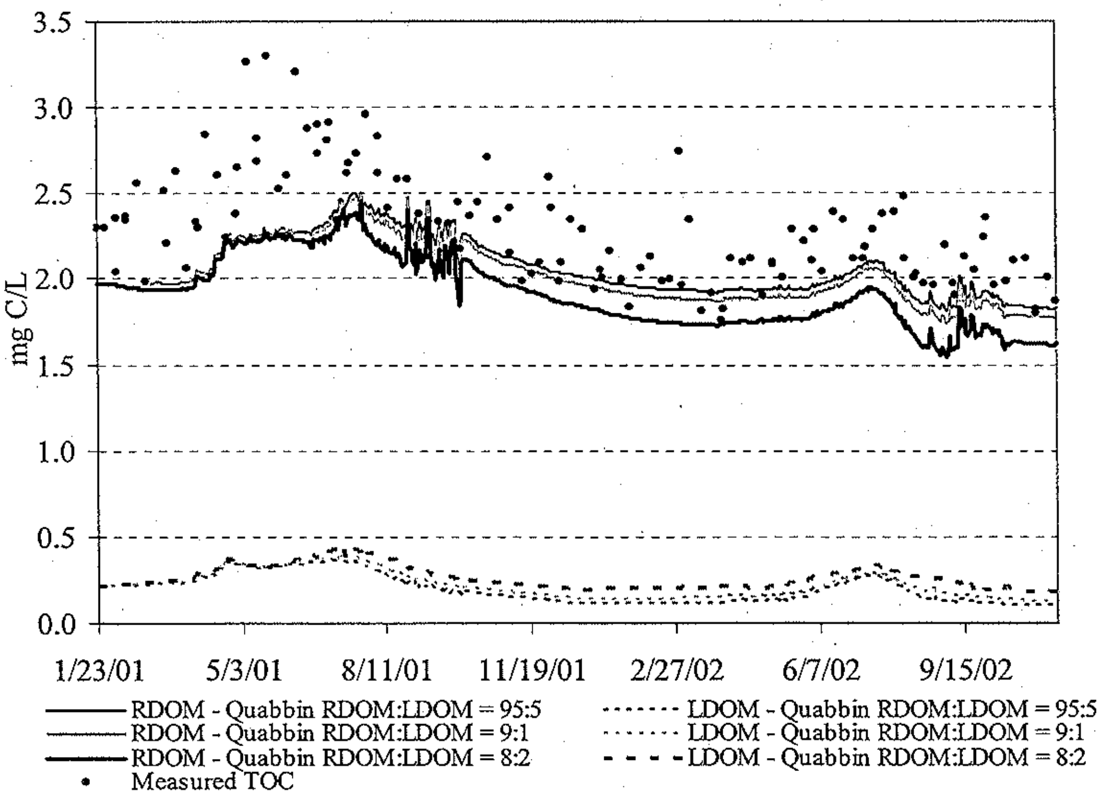

Figure 4.107 Impact of varied Quabbin Aqueduct RDOM to LDOM ratio on levels of those constituents as predicted as Cosgrove

4.4.2 POC and Nutrients

Phytoplankton dynamics in Wachusett Reservoir are typical of temperate, oligotrophic, softwater lakes (Worden 2003). Diatoms and chrysophytes (yellow-green algae) dominate, although lower quantities of cyanophytes (blue green algae), chlorophytes (green algae), and dinoflagellates are common. Phytoplankton samples are collected at the Cosgrove Intake weekly. In-reservoir samples were collected monthly in 1998 and 1999, but not during the study period.

Phytoplankton data were collected and enumerated by DCR (Worden 2003) and the results reported in areal standard units (ASUs; 1 ASU = 400 microns^2). Concentrations at Cosgrove ranged from 2.5 ASUs/mL to 760 ASU/mL at Cosgrove during 2001 and 2002. Algal enumeration data were converted to mg Carbon/L as described in Garvey (2000) and as presented in Roberts (2003).

In a typical year, phytoplankton activity is low in winter when temperatures are cold and sunlight is weak. In the spring, a rapid increase (bloom) in diatom levels occurs which depletes nutrients and reduces density to a minimum. A secondary bloom may then occur in late summer (Worden 2003). Figure 4.108 presents phytoplankton enumeration data converted to mg C/mL for April 10, 2001 through the end of 2002. Included are diatoms, algae (all phytoplankton not including diatoms), and total phytoplankton as the sum of algae and diatoms.

A bloom of phytoplankton occurred starting in April of 2001. This bloom was dominated by diatoms, which reached densities of 526 ASU/mL (~0.43 mg C/L) on May 22. There was a corresponding algae bloom of 166 ASU/mL (~0.10 mg C/L) two weeks later. After these blooms, a decline in both algal and diatom density occurred. Diatom density decreased to 2-5 ASU/mL (approximately 0.0014 to 0.0042 mg C/L) in late July and remained at low levels until the beginning of 2002. From July through the end of the year, algae dominated diatoms, with densities ranging from 80 to 200 ASU/mL (approximately 0.05 to 0.12 mg C/L). Dynamics followed a similar trend in 2002, although the spring diatom bloom was larger at 761 ASU/mL (0.55 mg C/L) on June 13, and late summer algal densities were lower, most likely because nutrients were depleted by the earlier diatom bloom (Worden 2003).

It is important to note that MWRA applies copper sulfate to the area around the Cosgrove Intake when phytoplankton densities become large or when densities of nuisance genera including *Anabaena* (a Cyanophyte) and *Synura* (a Chrysophyte) become large (MDC

2003). This chemical is an algicide, and the resulting algal death may affect modeling accuracy. Copper sulfate was applied near Cosgrove intake once each in June and September 2001, once in June and 2002 and twice in August 2002.

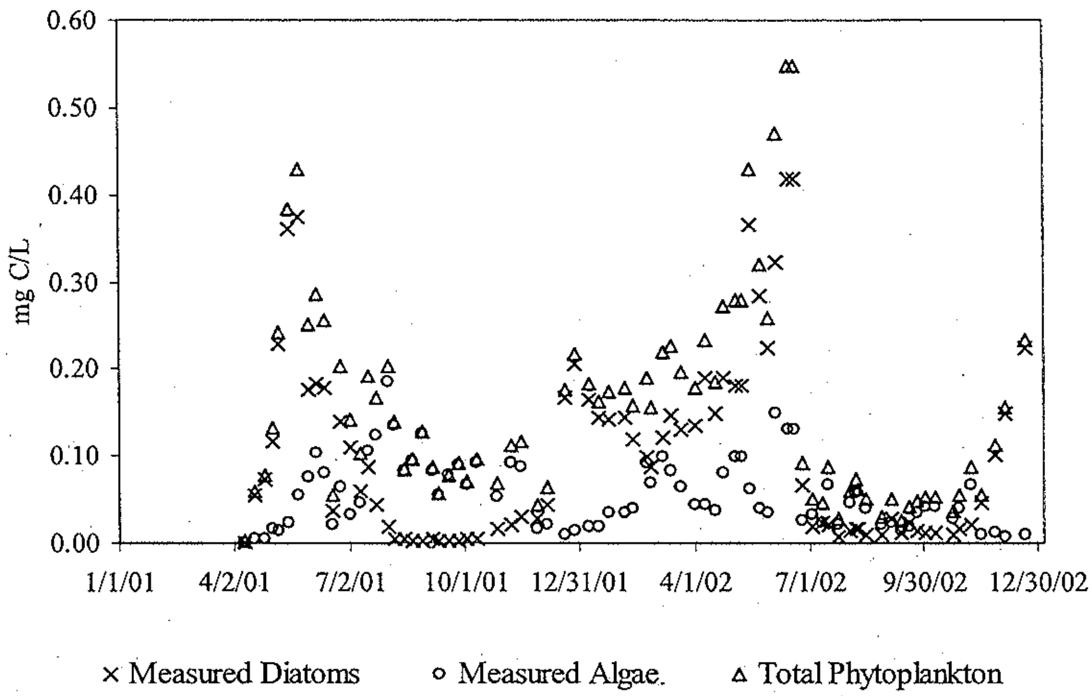

Figure 4.108 Phytoplankton enumeration data for Wachusett Reservoir at Cosgrove from DCR (converted to mg C/L).

As discussed in Section 4.4.1.1, algae were assumed to be exclusively autochthonous; therefore, this constituent was not included in the inflow concentration files of the model. An initial concentration of 0.05 mg C/L was used for the model; larger values decayed rapidly, causing the model to predict increased nutrient concentrations, thereby increasing the magnitude of the spring diatom blooms.

Table 4.19 and Table 4.20 present a summary of nitrate-nitrogen, ammonia-nitrogen, and total phosphorus data for Wachusett Reservoir tributaries and in-reservoir, respectively. In-reservoir nutrient data is more abundant for the study period, as its collection is included in the DCR sampling program.

Table 4.19 Inflow Nutrient Data for Wachusett Reservoir, 2000 - 2002 (from DCR and MWRA)

Water	Nitrate		Ammonia		Total Phosphorus		Number of Samples
	Average	Range	Average	Range	Average	Range	
Stillwater River	0.238	0.085 - 0.641	0.016	<0.005 - 0.048	0.051	0.013 - 0.468	39
Quinapoxet River	0.401	0.016 - 0.873	0.022	<.005 - 0.098	0.057	0.010 - 0.212	39
Quabbin - CVA	0.014	<0.005 - 0.028	0.009	<0.005-0.035	0.006	<0.005 - 0.013	22
Malden Brook	0.612	0.172 - 1.53	0.013	<0.005 - 0.058	0.052	0.014 - 0.272	28
Gates Brook	1.775	0.948 - 2.51	0.010	<0.005 - 0.037	0.044	0.014 - 0.105	30
French Brook	0.111	<0.005 - 0.297	0.039	<0.005 - 0.131	0.043	0.010 - 0.12	19
Malagasco Brook	0.673	0.204 - 1.04	0.020	<0.005 - 0.063	0.034	0.012 - 0.079	28
West Boylston Brook	2.866	1.57 - 4.19	0.021	<0.005 - 0.090	0.025	0.007 - 0.066	26
Muddy Brook	0.164	0.057 - 0.391	0.030	<0.005 - 0.073	0.044	0.008 - 0.273	27
Cosgrove Aqueduct	0.064	0.013 - 0.131	0.012	<0.005 - 0.035	0.008	0.005 - 0.019	36

Table 4.20 In-Reservoir Nutrient Data for Wachusett, 1998 - 2002 (condensed from MDC 2003)

Sampling Station	Nitrate (mg N/L) 1998 - 2002	Ammonia (mg N/L) 1998 - 2002	Total Phosphorus (mg P/L) 1998 - 2002
Thomas Basin (E)	<0.005 - 0.201	<0.005 - 0.018	<0.005 - 0.023
Thomas Basin (M)	<0.005 - 0.205	<0.005 - 0.018	<0.005 - 0.022
Thomas Basin (H)	<0.005 - 0.236	<0.005 - 0.021	<0.005 - 0.022
Basin South (E)	<0.005 - 0.172	<0.005 - 0.014	<0.005 - 0.017
Basin South (M)	0.011 - 0.184	<0.005 - 0.026	<0.005 - 0.022
Basin South (H)	0.049 - 0.224	<0.005 - 0.044	<0.005 - 0.037
Basin North(E)	<0.005 - 0.124	<0.005 - 0.012	<0.005 - 0.013
Basin North (M)	<0.005 - 0.138	<0.005 - 0.036	<0.005 - 0.017
Basin North (H)	0.049 - 0.190	<0.005 - 0.041	<0.005 - 0.014

Nutrient levels in the reservoir inflows follow the same trends as organic matter, presented in Section 4.4.1.1. Water entering from Quabbin is nutrient poor compared to most of the other tributaries: the average Quabbin nitrate concentration is at least one order of magnitude lower than all of the other inflows except French Brook, the average Quabbin ammonia concentration is the lowest of all inflows, and the average Quabbin total phosphorus concentration is only 24% that of the next lowest inflow concentration (West Boylston Brook).

Water from Stillwater and Quinapoxet Rivers have nitrate concentrations that are significantly higher than that of Quabbin, (0.238 and 0.401 mg N/L, respectively, as compared to 0.014 mg/L) but are within the range of average values from the other tributaries (0.111 mg N/L in French Brook to 2.886 in West Boylston Brook). Average ammonia concentrations from Stillwater and Quinapoxet are also within the range of values from the other tributaries (0.010 to 0.039 mg N/L). However, the average total phosphorus concentration for Stillwater River is the third highest of the inflows, and Quinapoxet River water had the highest average total phosphorus concentrations during the period of data. As Stillwater and Quinapoxet Rivers are the largest tributary inflows, the impact of this phosphorus on algal dynamics is likely significant. Some caution is necessary, however, because an unknown quantity of this phosphorus is probably not biologically available. Table 4.21 presents the extent of orthophosphate data from Wachusett Reservoir inflows. This data is limited, but suggests that just more than half of inflow total phosphorus is biologically available (orthophosphate) (see Figure 3.13 for a time series plot of measured total phosphorus and orthophosphate at Cosgrove). Considering that 64% of the orthophosphate data is at or below the detection limit, tributary orthophosphate input to CE QUAL W2 was assumed to be 50% of the total phosphorus measured.

Table 4.21 Orthophosphate Data for Wachusett Inflows, 2000-2002 (MWRA and DCR)

Inflow	Orthophosphate	
	Average	Range
Quabbin - CVA	0.0039	<0.0025 - 0.0084
French Brook	0.0120	n/a
Cosgrove Aqueduct	0.0044	<0.0025 - 0.0091

The in-reservoir data shown in Table 4.20 are arranged to show ranges of values at locations and depths throughout the reservoir. Most of the minimum levels shown for all three nutrients are below the detection limit of 0.005 mg/L. Since most inflow concentrations of nitrate and phosphorus were greater than the detection limit, the low in-reservoir values result from a combination of dilution with Quabbin water and loss through reservoir processes. The maximum concentrations at these locations show strong longitudinal and weak vertical gradients. The largest maximum nitrate concentrations were observed in Thomas Basin, while the lowest maximum nitrate combinations were observed at North Basin. Nitrate concentrations also generally increase with depth at all three locations. Longitudinal ammonia gradients generally increased with distance from Thomas Basin and with depth. Maximum total phosphorus concentrations decreased with distance from Thomas Basin and increased with depth. Figure 4.109 shows a time series plot of nutrient concentration as measured at the Cosgrove Aqueduct for 2001 and 2002.

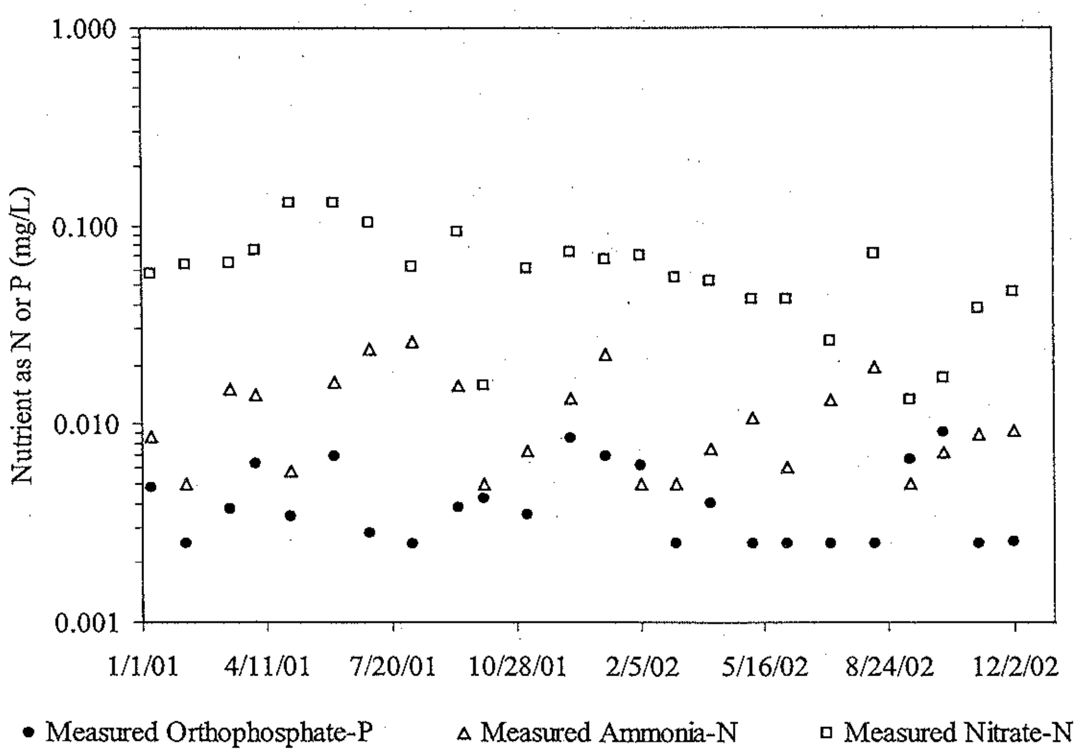

Figure 4.109 Nutrient Concentration at Cosgrove, 2001–2002 (MWRA data)

4.4.2.1 Phytoplankton and Nutrient Calibration Results

As discussed in Section 3.3.4, maximum algal growth (AGROW), respiration (ARESP), settling (ALGS), mortality (AMORT) and excretion (AEXCR) impact the rate of change of predicted phytoplankton concentrations in the reservoir. These rates were examined while maintaining LDOMDK = 0.008, LRDK = 0.0008 and RDOMDK = 0.0008. Generally, increasing AGROW or decreasing ARESP, ALGS, AMORT, or AEXCR will yield larger algal concentrations and shift blooms forward in time.

Sensitivity of phytoplankton concentrations to AGROW was first examined. Figure 4.110 presents a time series plot of algal concentrations at Cosgrove at three different maximum algal growth rates (AGROW). The rates shown include 3.5 day^{-1} as selected by Roberts (2003), 1.9 day^{-1}, an intermediate value, and 1.0 day^{-1}, similar to the values selected by CDM (1995) for Wachusett and Garvey (2001) for Quabbin (see Table 3.4). Note that all other algal parameters were set to the values chosen for Quabbin (Roberts 2003) except for the maximum algal respiration rate, ARESP, which was set at 0.1 day^{-1} as determined by CDM (1995). The phytoplankton data shown includes both diatoms and algae.

All three values of AGROW predict more algae exiting Cosgrove in 2001 than in 2002, whereas measurements show higher concentrations in 2002. AGROW = 1.0 day^{-1} predicted very little algae at Cosgrove, with concentrations reaching approximately 0.02 mg C/L at a maximum in 2001 (compared to 0.43 mg C/L as estimated from enumeration data) and predicting almost no algae in 2002.

Setting AGROW = 1.9 day^{-1} yields similar model predictions for both years. At this rate, maximum phytoplankton concentrations of 0.51 and 0.46 mg C/L are predicted in 2001 and 2002, respectively, as compared to the measured values of 0.43 and 0.54 mg C/L, respectively. The model predicts these blooms to occur at approximately the same time as they occurred in the reservoir; the model predicts the maximum value of the 2001 bloom 6 days late, May 28, while the 2002 bloom is predicted 15 days early, on May 29. After the peak algal concentration, the model prediction of phytoplankton growth is poor. Modeled algal concentrations decline more rapidly than occurred in the reservoir, and fall and winter concentrations are significantly underpredicted.

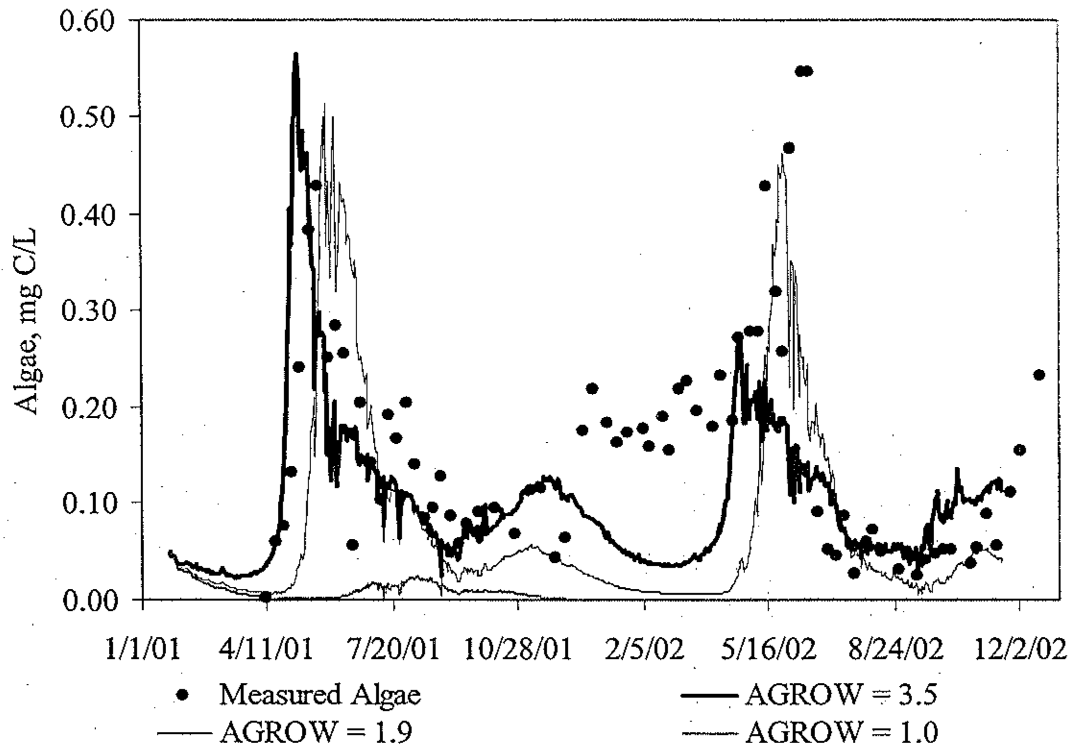

Figure 4.110 Modeled and measured total algae at Cosgrove, 2001 and 2002 (AGROW in day^{-1}).

The largest maximum growth rate, AGROW = 3.5 day^{-1}, predicts the 2001 bloom 16 days early and overpredicts concentrations by approximately 0.1 mg/L, and significantly underpredicts the 2002 bloom. This growth rate predicts larger fall and winter concentrations than AGROW = 1.0 day^{-1} and AGROW = 1.9 day^{-1}, but the model predicts decreasing levels between November 2001 and February 2002, while concentrations in the reservoir increased during that period. It was decided to focus on modeling the late spring/early summer phytoplankton blooms for calibration, as CE QUAL W2 seems more able to predict the species that bloom at this time.

Figure 4.111 presents the sensitivity of modeled maximum algal concentration in 2001 to AGROW. Each point corresponds to the date and concentration of the largest algae concentration predicted at a value of AGROW. Maximum predicted concentrations generally advance in time with increasing AGROW. Maximum concentrations ranged between 0.50 and 0.59 mg C/L. Maximum predicted concentrations are independent of

AGROW between AGROW = 1.4 day^{-1} and AGROW = 3.5 day^{-1}. There is high sensitivity to AGROW between values of 1.4 and 1.9 day^{-1} and variable sensitivity between AGROW = 1.9 and 2.4 day^{-1}.

Consumption and release of nutrients by algae are also important to consider. Figure 4.112 presents a time series plot of orthophosphate at Cosgrove as measured and as predicted by CE QUAL W2 for three values of AGROW. A uniform initial concentration of 5E-3 mg P/L was selected for the reservoir based on the January 9, 2001 measurement of 0.0048 mg/L. During 2001 and 2002, orthophosphate concentrations at Cosgrove ranged from 0.0025 (detection limit) to 0.009 mg P/L. Phosphorus levels were low in July 2001 and May through August 2002, and peaked in June 2001, early 2002, and September 2002.

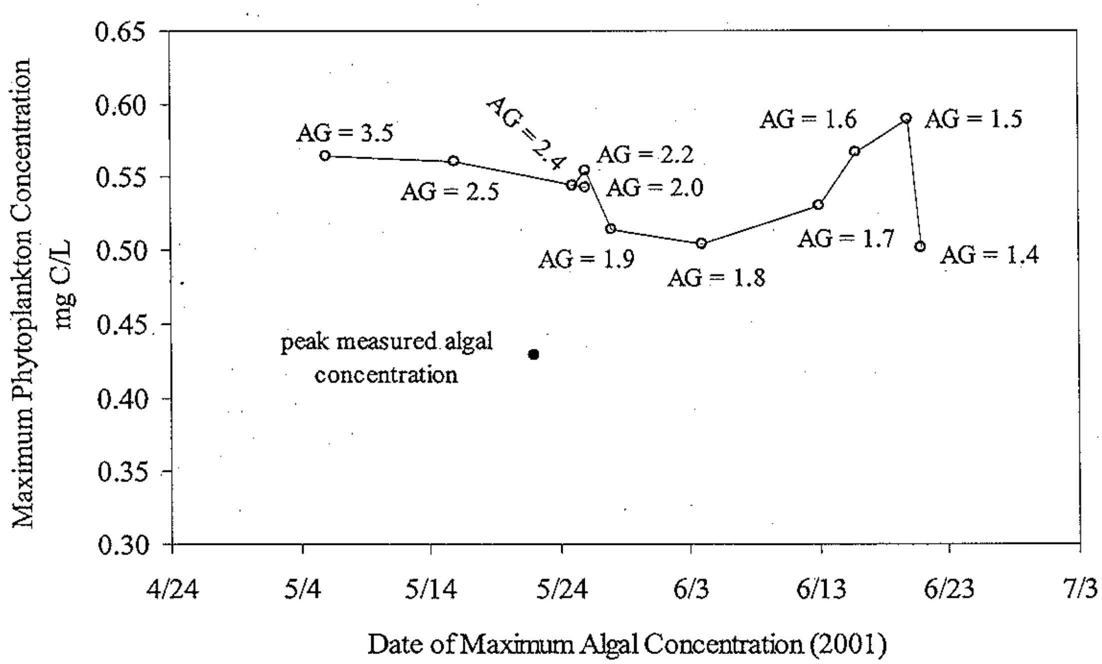

Figure 4.111 Sensitivity of maximum 2001 phytoplankton concentration to the maximum algal growth rate AGROW (shown as AG, in day^{-1}).

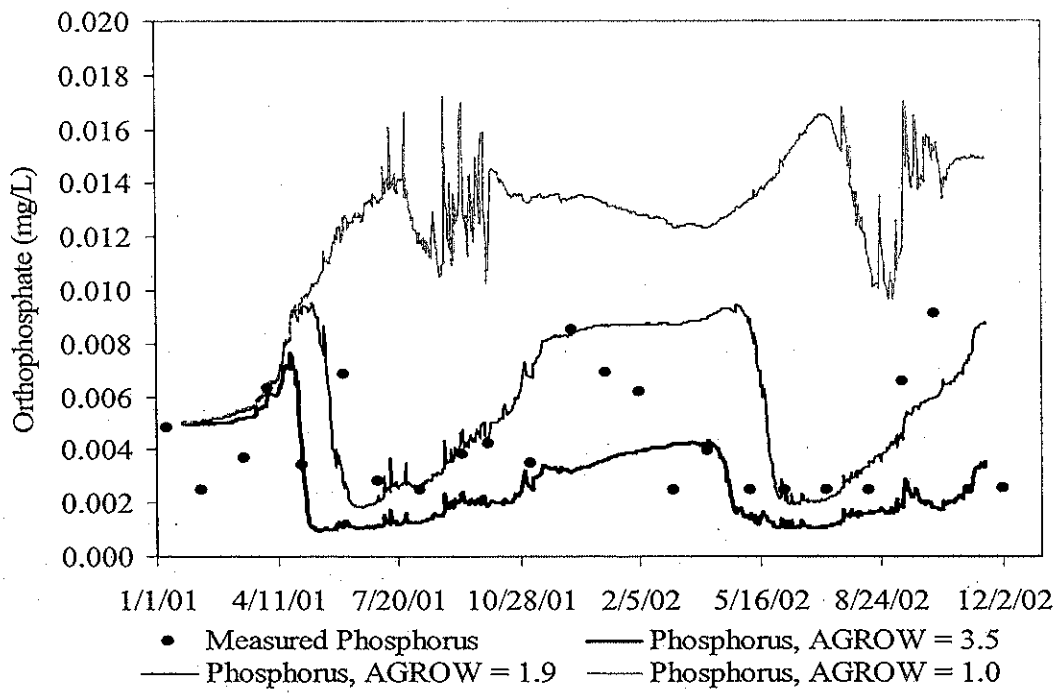

Figure 4.112 Modeled and measured orthophosphate at Cosgrove for 2001 and 2002 (AGROW in day^{-1}).

Orthophosphate exhibits a high sensitivity to AGROW since the nutrient is assimilated by phytoplankton growth. Since low phytoplankton concentrations are modeled at Cosgrove with AGROW = 1 day^{-1}, predicted phosphorus concentrations increase threefold from 5.0E-3 to 1.5 E-2 mg/L and are significantly larger than measured values. With AGROW = 3.5 day^{-1} the predicted phosphorus concentration at the end of 2002 is low (0.0035 mg/L) but close to 0.0025 mg/L, the measured value (and the detection limit). However, this maximum growth rate does not capture the wintertime increase in measured phosphorus that occurred between August 2001 and March 2002. Setting AGROW = 1.9 day^{-1} does capture these trends in terms of magnitude and rate of increase. Inaccuracy between November and May is a result of the model underpredicting winter phytoplankton levels.

The impacts of varying algal growth rates on ammonia concentrations can be seen in Figure 4.113. For this research, ammonia is generated by decay of organic matter and respiration by algae and lost through nitrification and photosynthesis. Therefore, algal growth consumes ammonia, but the presence of algae also generates it.

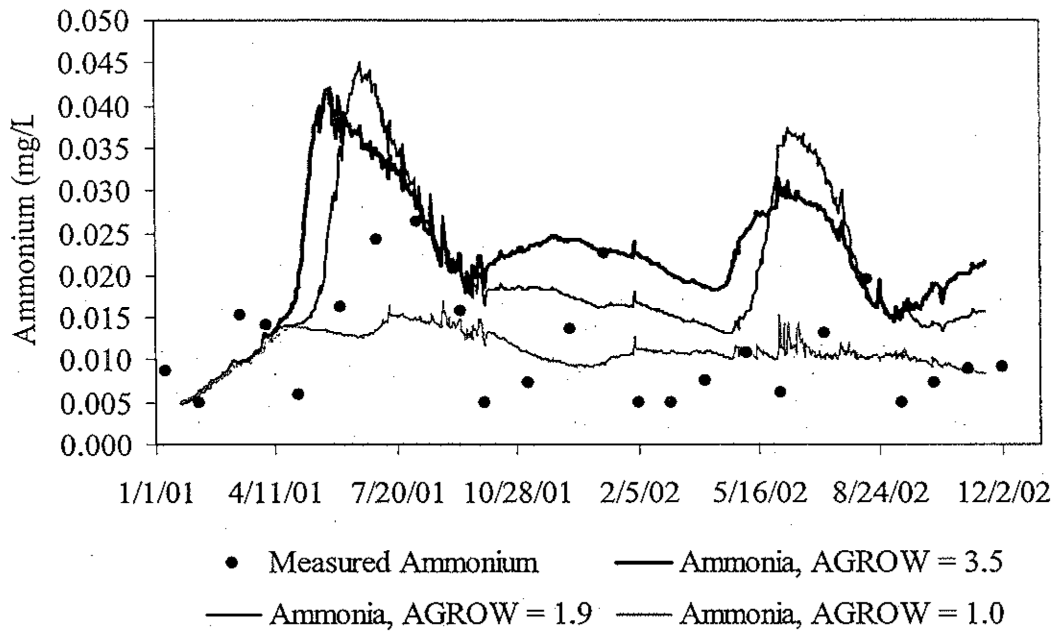

Figure 4.113 Modeled and measured ammonia at Cosgrove, 2001 and 2002 (AGROW in day^{-1}).

Ammonia concentrations during 2001 and 2002 ranged from the detection limit of 0.005 mg N/L to 0.026 mg N/L. Ammonia trends appear to cycle more frequently than phosphorus trends; low concentrations occurred in February, May, and October 2001 and February, June, and September 2002. Peak ammonia concentrations occurred in July and August 2001, and January and August 2002, occurring just after peak phytoplankton densities in each case.

At AGROW = 1.0 day^{-1}, ammonia concentrations increase from the uniform initial concentration of 0.005 mg/L (established from withdrawal data) to approximately 0.014 mg/L in April 2001, with little variation for the rest of the year. At AGROW = 1.9 day^{-1}, significantly more ammonia is released in early summer corresponding to the large phytoplankton blooms. Modeled concentrations become approximately triple the concentrations predicted at the lower maximum growth rate at these peaks. Additionally, winter ammonia concentrations are elevated by 50 to 90%. This algal growth rate overpredicts ammonia concentrations at Cosgrove At AGROW = 3.5 day^{-1}, predicted peak ammonia concentrations are lower and occur sooner than at AGROW = 1.9 day^{-1}.

Nitrate concentrations varied by an order of magnitude during the study period, from 0.013 in September 2002 to 0.13 in May and June 2001. Nitrate generally declined throughout the study period and did not exhibit the short term, seasonal trends apparent in phosphorus and ammonia. Nitrate is formed from ammonia by nitrification, and because it is lost through photosynthesis of algae (which are generally low in number in this oligotrophic system) and denitrification under anaerobic conditions (which do not occur in Wachusett), it has only secondary dependence on seasonal conditions. Figure 4.114 presents nitrate as measured at Cosgrove and nitrate/nitrite as modeled by CE QUAL W2 (assumed to be nitrate only).

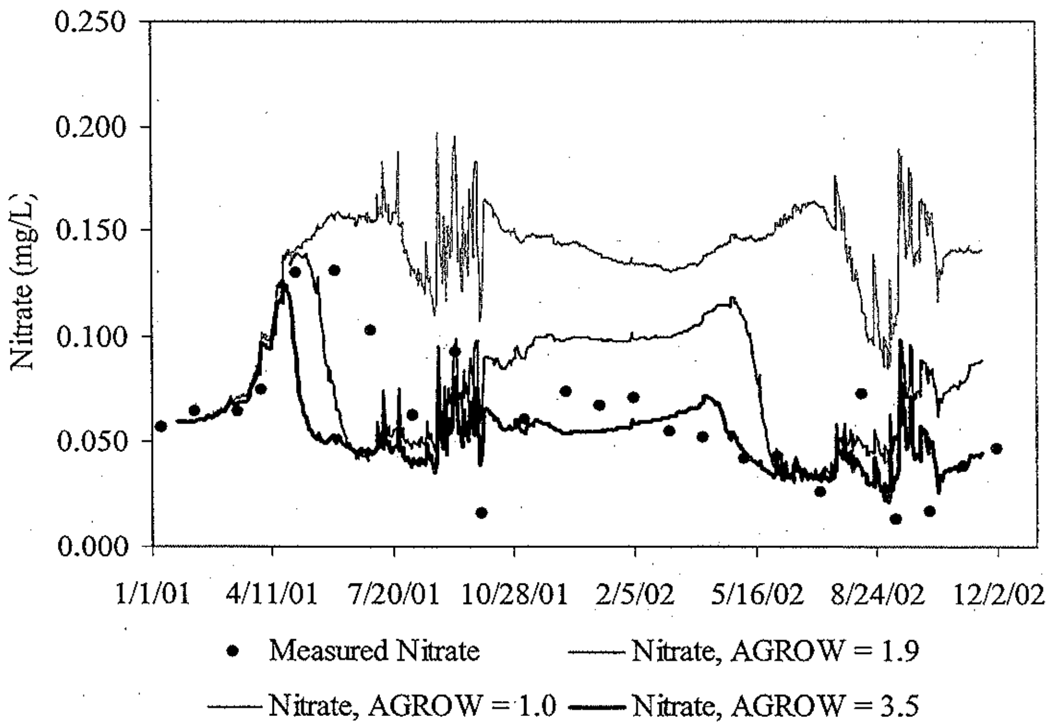

Figure 4.114 Modeled and measured nitrate at Cosgrove, 2001 and 2002 (AGROW in day^{-1}).

The low maximum growth rate, AGROW = 1.0 day^{-1}, predicts significantly greater nitrate concentrations than were measured. This curve also peaks in July 2002, although nitrate levels in the reservoir were low at that time. AGROW = 1.9 day^{-1} and AGROW = 3.5 day^{-1} both predict nitrate trends reasonably well. The former predicts generally larger nitrate concentrations than the latter during the winter, and the prediction of the latter is more consistent with the data.

The impact of maximum algal growth rate on nutrient concentrations depends on the nutrient. For orthophosphate, the intermediate value, AGROW = 1.9 day^{-1}, is most representative of the data, while for ammonia AGROW = 1.0 day^{-1} is most appropriate, and for nitrate AGROW = 3.5 day^{-1} is most appropriate. However, as nutrient concentrations in Wachusett Reservoir are low, and both the model and physical system susceptible to minute variations, the maximum algal growth rate equal to 1.9 day^{-1} appropriately models all nutrients.

The impacts of ARESP, AEXCR, AMORT, and ALGS on algal growth were then examined. The impact of adjustments to initial nutrient concentrations were examined but are not presented because sensitivity was similar to that observed while examining AGROW. The model proved to be insensitive to AEXCR; the model was run three times, once with a value of 0.012 day^{-1}, selected by Garvey (2000) and Roberts (2003), and then once each with that value halved and doubled. These adjustments yielded no change in algal or nutrient concentration, and changes in LDOM were very minor.

Sensitivity to ARESP is presented in Figure 4.115. In conducting this sensitivity analysis, AGROW = 1.9 day^{-1} and all other parameters are those used in the AGROW analysis. ARESP = 0.2 day^{-1} as implemented by Roberts (2003) caused algae to be significantly underpredicted during the 2002 bloom. Setting ARESP = 0.1 day^{-1}, the value implemented by CDM (1995) for Wachusett and Garvey (2000) for Quabbin, caused the model to better predict the 2002 bloom in terms of magnitude and timing. Implementing ARESP = 0.15 day^{-1} resulted in the best prediction of the peak 2002 algal concentrations, predicting them within 0.03 mg C/L, but the increase in algae to this peak, and the decrease from this peak were predicted unacceptably late in the year. Additionally, the model prediction of the 2001 bloom was also delayed. Setting ARESP = 0.05 day^{-1} yielded a result similar to that predicted by ARESP = 0.2 day^{-1}, although shifted forward in time, predicting larger peak algal concentrations, and predicting concentrations in October and November 2001 that were within the range of values estimated from enumeration data. ARESP = 0.1 day^{-1} resulted in the best agreement between model predictions and the data.

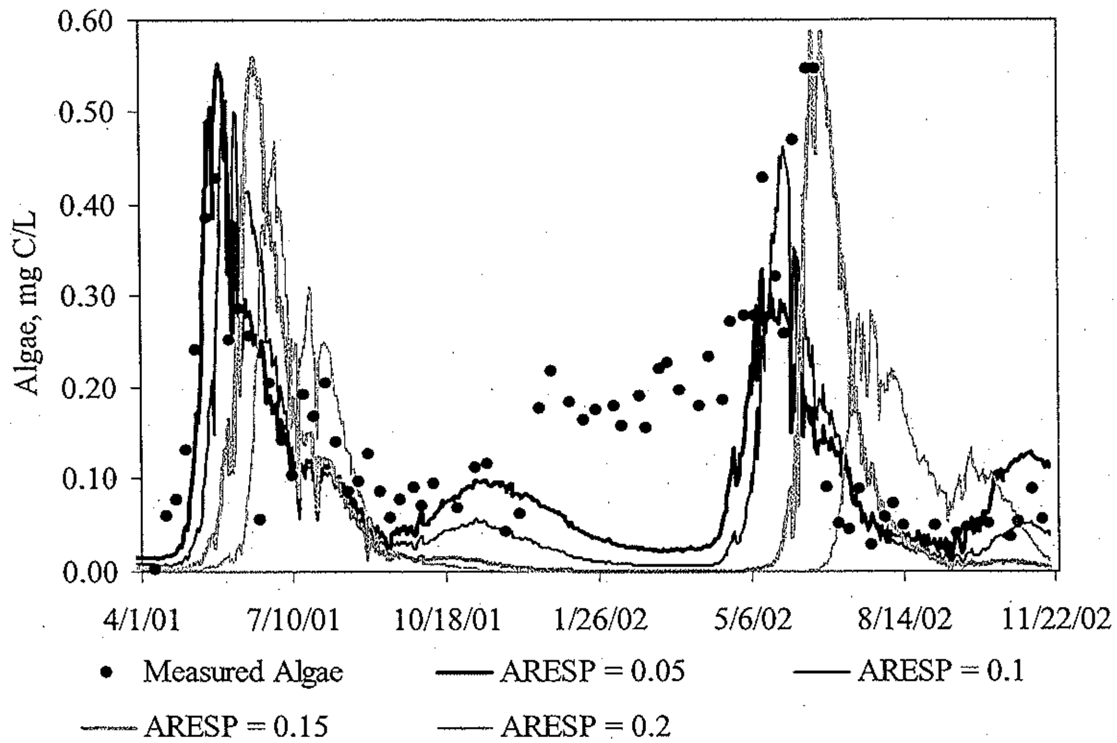

Figure 4.115 Modeled and measured total algae at Cosgrove, 2001 and 2002 (ARESP in day^{-1}).

Sensitivity of modeled algae at Cosgrove to AMORT is shown in Figure 4.116. The value of 0.03 day^{-1} for this parameter was implemented in studies by CDM (1995), Garvey (2000) and Roberts (2003). Cole and Buchak (1995) recommend a mortality rate of less than 10% of the maximum growth rate, corresponding to values of 0.08 to 0.35 day^{-1} based on these studies. However, they also report values ranging from 0.0096 to 0.031 day^{-1}. Garvey (2000) suggests that, for Quabbin, mortality is small relative to growth and that AMORT equaling ~1% AGROW is appropriate. Running CE QUAL W2 with AMORT = 0.06 day^{-1} predicted lower and later 2001 algae bloom concentrations, and predicted generally less algae in 2002 (based on the area under the algal curve), although peak concentration was similar to that predicted by the lower maximum mortality rates. Setting AMORT = 0.01 day^{-1} predicted generally larger summer 2001 and winter 2001 – 2002 algae levels, and shifted the 2002 bloom forward in time slightly. AMORT = 0.03 day^{-1} was selected as it is most appropriate in terms of maximum algal concentrations and the timing of the resulting predicted blooms.

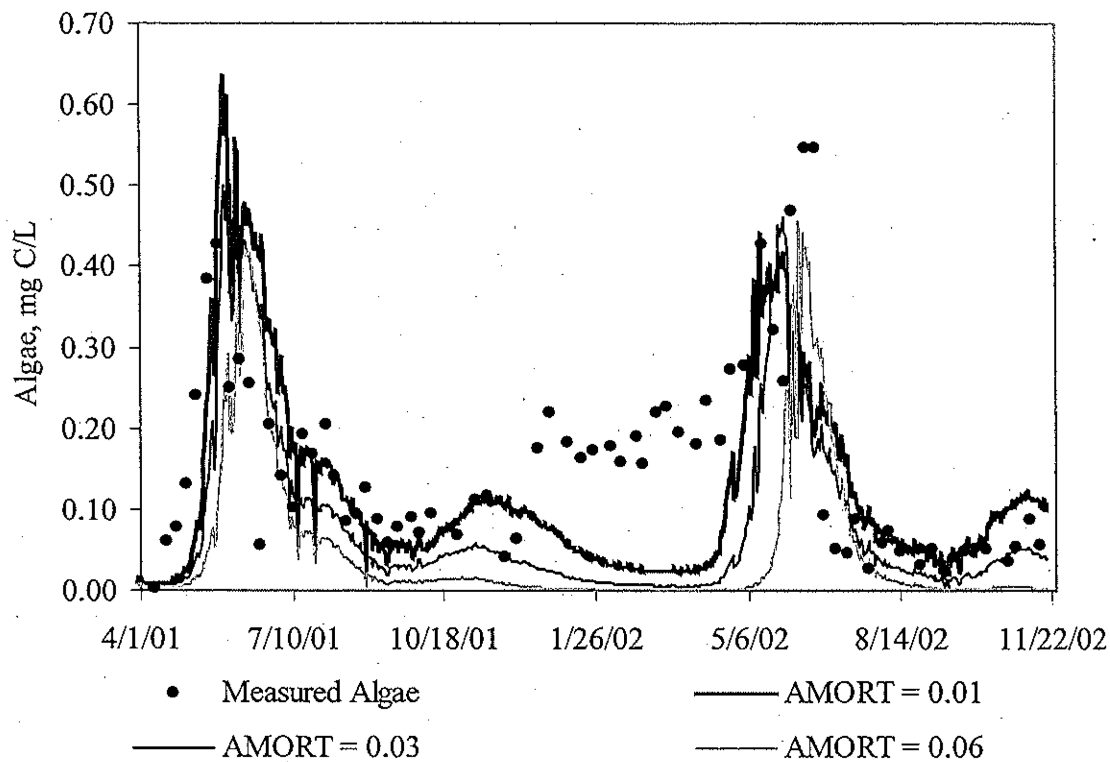

Figure 4.116 Modeled and measured total algae at Cosgrove, 2001 and 2002 (AMORT in day^{-1}).

Sensitivity of modeled algae at Cosgrove to algal setting rate is shown in Figure 4.117. Results are somewhat mixed. Decreasing the settling velocity generally increased algal levels as predicted at Cosgrove, especially following the predicted summer algal blooms.

The lowest settling velocities shown, ALGS = 0 m/day and ALGS = 0.1 m/day, predict larger peak algal concentrations in 2001 than found for the larger velocities. However, the following decline in algae levels parallels the decline of the data, indicating that perhaps very low settling velocities are appropriate. However, in 2002, these low settling velocities predict slower declines in algae concentrations than the data suggests, and the larger settling velocities are more effective. This analysis is therefore inconclusive, and the settling velocity of 0.29 m/day, implemented by Roberts (2003) and CDM (1995) was retained.

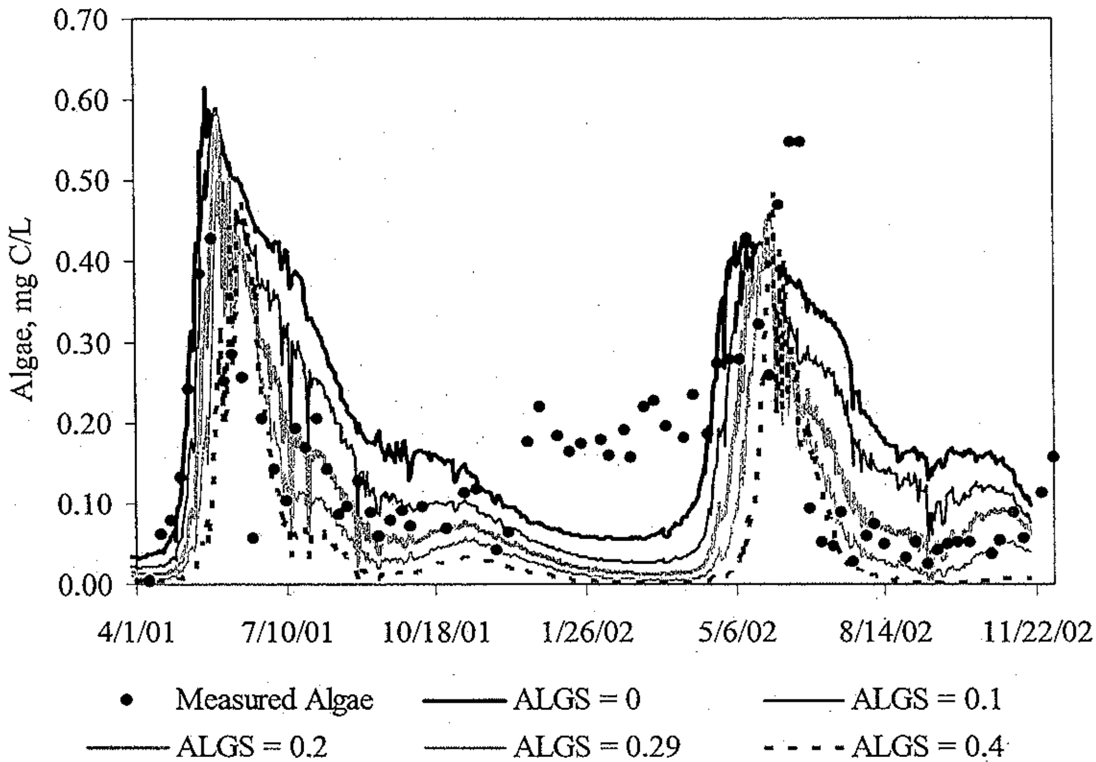

Figure 4.117 Modeled and measured total algae at Cosgrove, 2001 and 2002 (ALGS in m/day).

Nitrification is the only nutrient decay process included in CE QUAL W2 that applies to Wachusett Reservoir since aerobic conditions are always observed in the reservoir. Nitrification is modeled with first-order decay set by the rate NH4DK. CDM (1995) and Roberts (2003) implemented NH4DK = 0.03 day^{-1}. Model results show no sensitivity of phytoplankton to NH4DK; halving the value, doubling the value, and setting the value to zero yielded the same result. Thus, the model predicts that nitrogen is in excess in Wachusett Reservoir and that phosphorus is the limiting nutrient. This is consistent with findings from limnological research (Worden 2003, MDC 2003). Predicted ammonia and nitrate are sensitive to NH4DK, as shown in Figure 4.118 and Figure 4.119, respectively. Setting NH4DK to zero causes ammonia to accumulate to eight times the measured concentration in December 2002. Halving NH4DK to 0.015 and 0.06 day^{-1} yielded ammonia concentrations of 2 to 3 times measured values throughout the study period. Doubling NH4DK to 0.06 day^{-1} resulted in improved prediction of ammonia concentrations, with no deviation between measured and modeled ammonia in December 2002, and predicted ammonia lying within the scatter of data for most periods except those following the spring algal blooms.

Increasing NH4DK to 0.06 day^{-1} decreased the agreement between the modeled nitrate results and nitrate data at Cosgrove, although not significantly. For nitrate, the intermediate values of 0.015 and 0.03 day^{-1} for NH4DK produce a better fit. Inaccuracies in modeling nitrate likely arise in part from inaccuracies in modeling algae; thus, NH4DK = 0.03 day^{-1} was retained for this study.

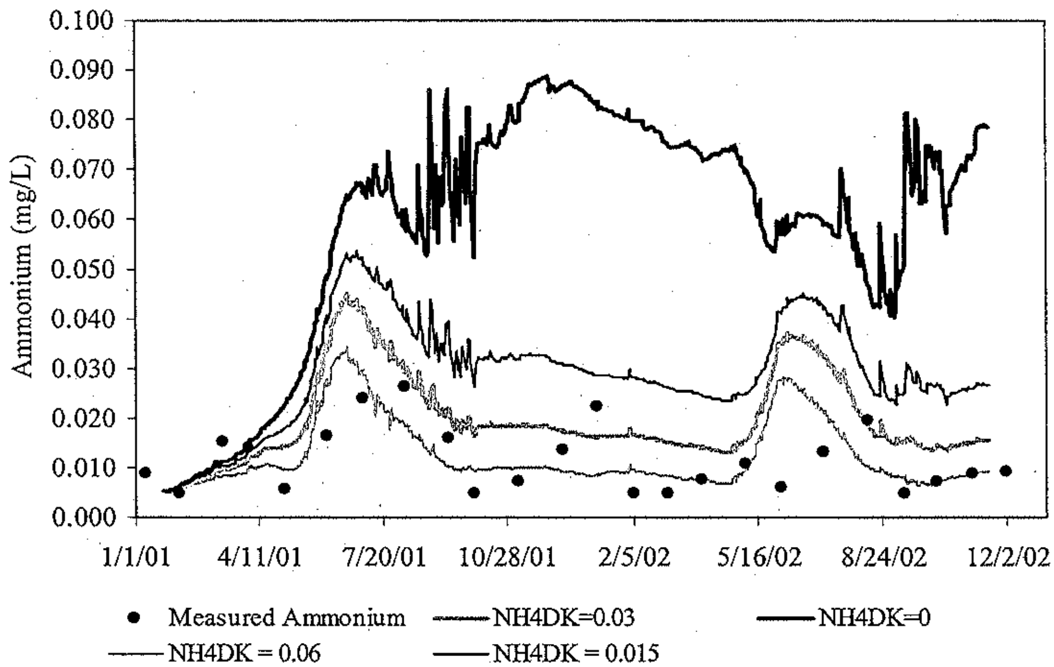

Figure 4.118 Modeled and measured ammonia at Cosgrove, 2001 - 2002 (NH4DK in day^{-1}).

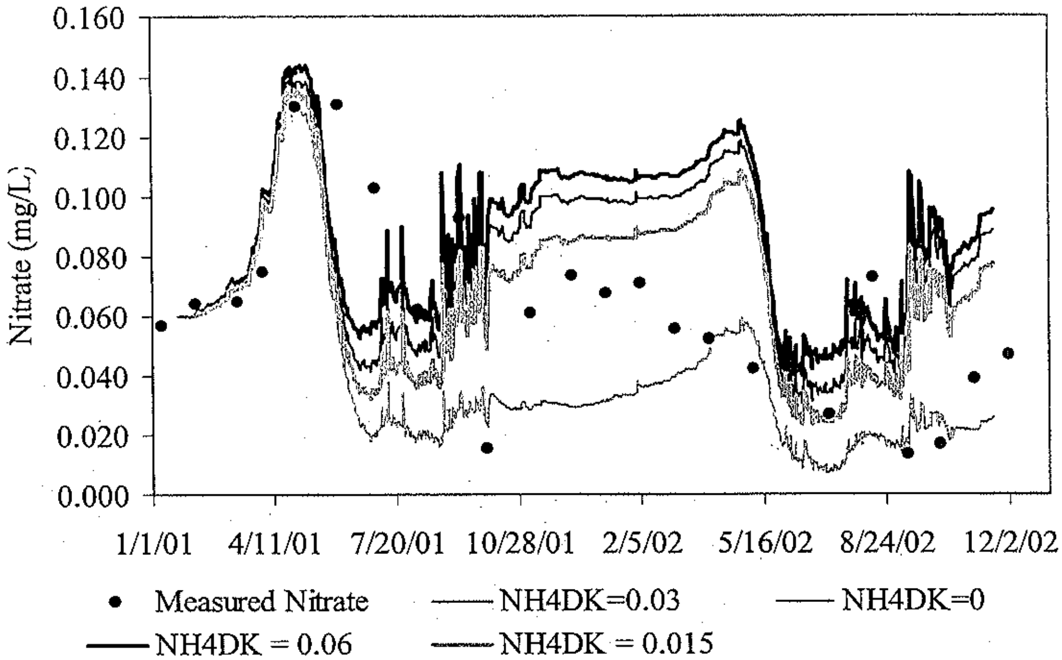

Figure 4.119 Modeled and measured nitrate at Cosgrove, 2001-2002 (NH4DK in day^{-1}).

4.4.2.2 Algal Modeling Limitations

It is important to note that there are significant limitations to modeling phytoplankton with CE QUAL W2 Version 2. The largest is that only one phytoplankton compartment exists. The user must therefore adjust the parameters presented in Section 3.3.4 to model a particular algal genera, or to model a generic species that mimics the behavior of several genera, or the phytoplankton ecosystem. More algal compartments would provide the ability to better represent algal ecology and thereby improve the nutrient calibration. Additionally, diatoms require silica to assemble a frustule, and silica can be the limiting nutrient to diatom growth. Worden (2003) notes that concentrations of silica in Wachusett are typically larger than the minimum required for diatom growth (0.5 mg/L from Wetzel 1983), but these dynamics are unknown and perhaps useful to study. CE QUAL W2 Version 2 cannot model silica, although Version 3 has that capability, as well as the capability to model additional algal compartments. Silica data for Wachusett Reservoir currently exists, as do data regarding predominance of phytoplankton taxa, so implementing Version 3 is feasible.

4.4.2.3 Particulate Organic Carbon Calibration Result

POC can be modeled as the sum of the algal and detrital constituents provided in CE QUAL W2. Detritus is the particulate product of algal decay. Detritus was included in all inflows, except precipitation, at 5% of measured TOC, and an initial value of 0.06 mg/L was assumed for the reservoir (approximately ½ of 5% of TOC, selected because water body processes would reduce in-reservoir concentrations). Sensitivity to the detritus decay and settling rate provided in CE QUAL W2 were not examined due to the scarcity of data. The parameter values LPOMDK = 0.007 day^{-1} and LPOMS = 0.35 m/day, as determined by CDM (1995) and Roberts (2003) were therefore implemented. Figure 4.120 presents modeled algae, detritus, and the sum of algae and detritus (POC), as modeled at Cosgrove, using the algae parameter values presented in Section 4.4.2.1.

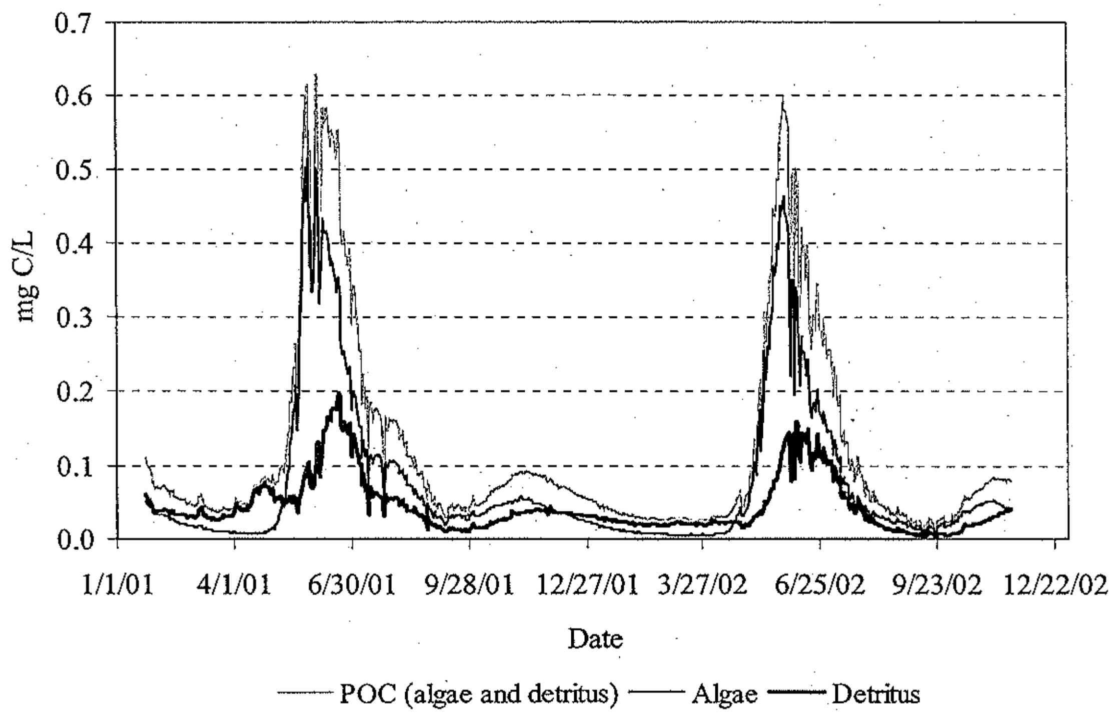

Figure 4.120 Modeled POC and POC components at Cosgrove, 2001 and 2002.

CE QUAL W2 predicts detritus at levels generally below 0.05 mg/L except following the peak spring/summer algae blooms. Since detritus is generated by algal mortality, peak detritus concentrations correspond to peak algae concentrations, though shifted slightly later in time. The largest predicted detritus concentration is 0.2 mg C/L, which corresponds to a POC concentration of 0.6 mg/L when added to algae.

4.4.3 Total Organic Carbon Calibration Results

Predicted DOM, algae, and detritus concentrations were added to predict TOC levels at Cosgrove. Scarcity of organic matter data for the Wachusett Reservoir system imparts uncertainty into this modeling analysis. As a result, three alternative calibrations are presented in this section: one in which refractory and labile DOM fractions are equal for tributary and Quabbin Aqueduct inputs, one in which larger refractory and labile DOM fractions are implemented for Quabbin Aqueduct than for the Wachusett Tributaries, and one in which refractory DOM decay is driven by photolysis.

4.4.3.1 TOC Calibration Alternative I – Consistent DOM fractionation

In this calibration alternative, it is assumed that the refractory and labile DOM fractions are equal for inputs from Wachusett Tributaries and Quabbin Aqueduct. This assumption is reasonable since autochthonous DOM generation in Quabbin is of the same order of magnitude as allochthonous generation, since BDOC measurements in Quabbin can be above 20% of total DOC (Garvey 2000), and since, in Quabbin, photolysis likely results in labile photoproducts as has been shown in numerous studies (see Section 2.5.2). Table 4.22 presents NOM parameter values for this alternative that differ from those implemented in previous studies (Table 3.4; Roberts 2003, Garvey 2000, CDM 1995).

Table 4.22 NOM Parameter Values Implemented in Alternative I

Parameter	Description	Units	Alternative I
LDOMDK	Labile DOM decay rate	day^{-1}	0.008
LRDK	Labile to refractory DOM decay rate	day^{-1}	0.0008
RDOMDK	Refractory DOM decay rate	day^{-1}	0.0008
OMALP	Impact of irradiance on RDOM	cm2/cal	0
$RDOM:LDOM_{trib}$	Tributary Refractory to Labile DOM ratio	-	8:2
$RDOM:LDOM_{QA}$	Quabbin Refractory to Labile DOM ratio	-	8:2
AG	Maximum algal growth rate (AGROW)	day^{-1}	1.9
AR	Maximum algal respiration rate (ARESP)	day^{-1}	0.1

It is also assumed in this alternative that there is no light induced decay of RDOM. This decay pathway is accounted for by the temperature induced first order decay rate RDOMDK. Figure 4.121 presents the time series results of predicted TOC compared to measured TOC at the Cosgrove Intake. The four TOC components, although calibrated separately, effectively model TOC concentrations when added.

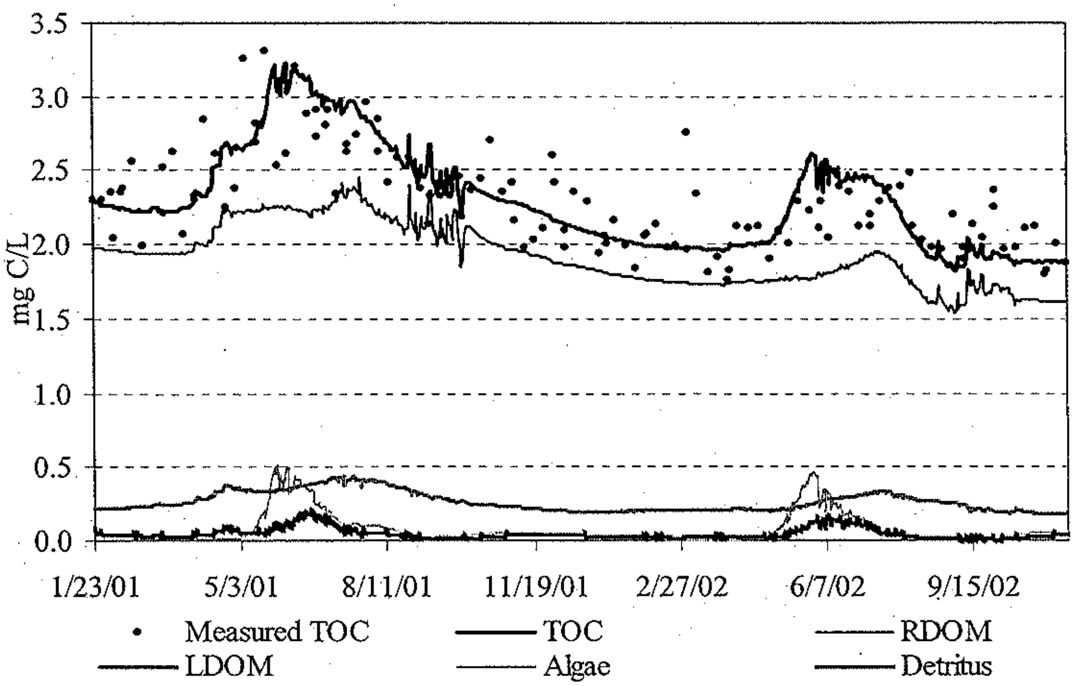

Figure 4.121 Measured and modeled TOC and TOC components at the Cosgrove withdrawal for Calibration Alternative I.

Predicted TOC is generally within the scatter of measured TOC for the study period, and the notable trends are captured. The largest deviations occur in the spring of 2001, the spring of 2002, and the late summer of 2002. In Spring 2001, the predicted diatom bloom occurs late, causing the highest TOC level to occur late, although the predicted TOC is within the range of measured values. In the spring of 2002, although measured algal concentrations as carbon are larger than in 2001, there is not as pronounced of a peak in TOC concentration. This discrepancy likely results from inaccuracy in estimating algal carbon concentration from enumeration data.

Some values of AGROW examined, including AGROW = 3.5 day^{-1} produced larger algal concentrations in 2001 than in 2002. However, algal data suggests AGROW = 1.9 day^{-1} is appropriate, and this value was retained. This issue may be revisited in a future study. The significant deviation occurred in July and August of 2002 involved TOC underprediction as a result of the wind mixing problem presented in Section 4.3.3.2.

A notable result of TOC calibration is the dynamic relationship between labile and refractory DOM throughout the study period. Although tributary inputs of DOM are assumed to be 20% labile and 80% refractory, at the end of the study period predicted labile DOM constitutes 10% of total DOM, while refractory constitutes the remainder. This ratio is the same as the assumption for the in-reservoir initial value at the beginning of the study period, confirming this assumption.

In summary, the calibration alternative summarized in Table 4.22 results in model predictions that fit the data at Cosgrove withdrawal, with parameter values and NOM component ratios that are consistent with results from Garvey (2000), Roberts (2003) and literature values presented in Sections 2.5 and 2.6.

4.4.3.2 TOC Calibration Alternative II – Inconsistent DOM fractionation

This alternative presents a TOC parameter calibration for increasing the RDOM fraction to 95% of DOM for the Quabbin Aqueduct input, and for decreasing the LDOM fraction to 5%. These fractions may be reasonable as the long detention time of Quabbin would result in the decay of all but the most refractory DOM. Light induced decay of refractory DOM was not implemented in this scenario; this process is accounted for by the 1^{st} order temperature dependent decay rate RDOMDK. The increased recalcitrance of organic matter from the Quabbin input required that the DOM decay rates be increased to maintain adequate fit of model predictions to data measured at Cosgrove. The resulting calibration is presented in Table 4.23. The results of this calibration alternative are presented in Figure 4.122.

TOC results for this alternative are essentially identical to those presented for Alternative I; the maximum predicted TOC concentration in 2001 is 3.22 mg/L, whereas 3.23 mg/L was predicted in Alternative I. At the end of 2002, TOC levels of 1.89 are predicted in both cases. Additionally, the average TOC concentration for the two year period was 2.30 mg/L as predicted with alternative I and 2.31 as predicted with alternative II.

Table 4.23 NOM Parameter Values Implemented in Alternative II

Parameter	Description	Units	Alternative II
LDOMDK	Labile DOM decay rate	day^{-1}	0.01
LRDK	Labile to refractory DOM decay rate	day^{-1}	0.001
RDOMDK	Refractory DOM decay rate	day^{-1}	0.001
OMALP	Impact of irradiance on RDOM	cm2/cal	0
$RDOM:LDOM_{trib}$	Tributary Refractory to Labile DOM ratio	-	8:2
$RDOM:LDOM_{QA}$	Quabbin Refractory to Labile DOM ratio	-	95:5
AG	Maximum algal growth rate (AGROW)	day^{-1}	1.9
AR	Maximum algal respiration rate (ARESP)	day^{-1}	0.1

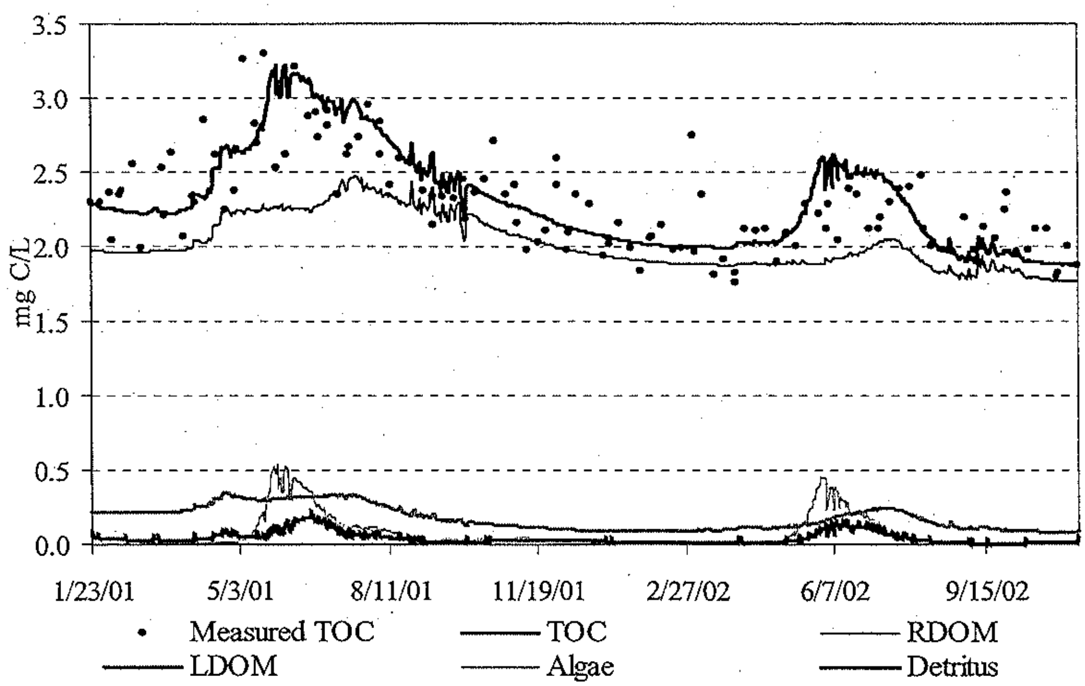

Figure 4.122 Measured and modeled TOC and TOC components at the Cosgrove withdrawal for Calibration Alternative II.

It is notable that the predicted LDOM and RDOM fractions are different than previously predicted; predicted RDOM comprises 95.5% of DOM for this alternative, as compared to 90% resulting from alternative I. Results from Hodgkins (1999) suggest that a larger fraction is appropriate (BDOC = 13 – 21 % of DOC in that study). Predicted algae and detritus levels are similar to levels as calibrated and as presented in alternative I. Slightly higher algal levels were predicted in 2001, most likely as a result of phosphorus release from increased organic matter decay.

In summary, the calibration alternative summarized in Table 4.23 results in model predictions that fit the data at Cosgrove withdrawal. The assumptions underlying this alternative are acceptable, and the results are consistent with expectations. It is notable, however, that the resulting refractory DOM fraction at Cosgrove is larger than in data reported by Hodgkins (1999).

4.4.3.3 TOC Calibration Alternative III – Photolysis of Refractory DOM

In a third calibration alternative, RDOMDK and LRDK were replaced with light-induced decay as presented in Section 3.3.3. These parameters were set to zero to simplify calibration and to prevent feedback Table 4.24 presents the model parameter values implemented in this case. A Quabbin transfer RDOM fraction of 80% (LDOM fraction of 20%) was implemented as in alternative I. Figure 4.123 presents a time-series result of calibration alternative III.

Table 4.24 Alternative III NOM Parameters, Including Light Induced Decay of RDOM

Parameter	Description	Units	Alternative III
LDOMDK	Labile DOM decay rate	day^{-1}	0.012
LRDK	Labile to refractory DOM decay rate	day^{-1}	0
RDOMDK	Refractory DOM decay rate	day^{-1}	0
OMALP	Impact of irradiance on RDOM	cm2/cal	1.30E-05
$RDOM:LDOM_{trib}$	Tributary Refractory to Labile DOM ratio	-	8:2
$RDOM:LDOM_{QA}$	Quabbin Refractory to Labile DOM ratio	-	8:2
AG	Maximum algal growth rate (AGROW)	day^{-1}	1.9
AR	Maximum algal respiration rate (ARESP)	day^{-1}	0.1

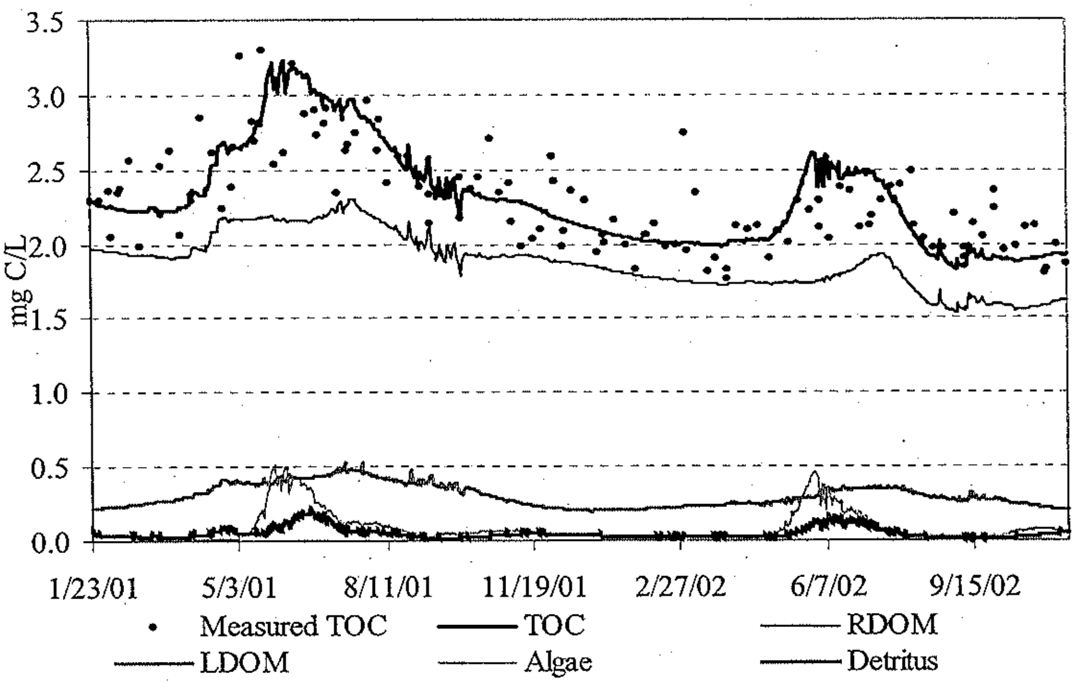

Figure 4.123 Alternative III TOC calibration with light induced decay of RDOM, showing measured and modeled TOC components.

At the end of 2002, predicted RDOM comprised 88.8% of predicted DOM. Maximum 2001 TOC was predicted to be 3.23 mg/L, essentially equal to results from Alternatives I and II. Average predicted TOC during the two year period is 2.30 mg/L; identical to alternative I and very similar to alternative II.

4.4.3.4 TOC Calibration Conclusions

Selecting from these three alternatives is difficult; results are similar, and each mimics environmental conditions. Presumably, each alternative sometimes represents natural conditions; DOM fractionation is variable, and light degradation impacts bioavailability of DOM. For the purposes of this study, Alternative I was selected for validation and for scenario simulation. Alternative II was not selected since Quabbin Transfer input DOM fractions of 20% LDOM and 80% RDOM is more consistent with BDOC results from Garvey (2000). Alternative III was not selected because calibration of the light induced decay parameter OMALP, the temperature dependent decay parameter RDOMDK, and the temperature dependent decay parameter LRDK would be too difficult, and little is known regarding the nature and decay pathways of organic matter within the system. The selected parameter values can be seen in Table 4.24.

4.4.4 UV254 Calibration

UV254 is modeled as an independent compartment that includes only light-induced and temperature dependent decay. Within CE QUAL W2, algal products, LDOM, and RDOM have no UV254 absorbance; therefore, the only source of UV254 is the reservoir tributaries, Quabbin Transfer, precipitation, and direct runoff.

Table 4.25 presents average UV254 levels for the Quabbin transfer and the minor tributaries as measured by MWRA and DCR.

Table 4.25 Inflow UV254 Data for Wachusett Reservoir, 2001 – 2002 (from DCR and MWRA)

Water	UV254 (cm^{-1})	
	Average	Range
Stillwater River	0.155	0.069 - 0.353
Quinapoxet River	0.193	0.096 - 0.311
Quabbin - CVA	0.020	0.015 - 0.030
Malden Brook	0.115	0.041 - 0.314
Gates Brook	0.084	0.025 - 0.134
French Brook	0.338	0.172 - 0.611
Malagasco Brook	0.561	0.033 - 1.727
West Boylston Brook	0.089	0.031 - 0.193
Muddy Brook	0.119	0.048 - 0.256
Cosgrove Aqueduct	0.047	0.030 - 0.078

Quabbin Transfer generally has the lowest UV254 of all the inputs; the maximum recorded value is lower than the lowest value recorded for the tributaries. Average UV absorbance levels for the Stillwater and Quinapoxet Rivers are greater than those of Malden, Gates, West Boylston, and Muddy Brooks, and are generally an order of magnitude larger than that of Quabbin Transfer. As in Roberts (2003), precipitation UV254 was assumed to be 0.020 cm^{-1}. Data from Purgee Brook, an inflow of Quabbin Reservoir, was used for Wachusett Reservoir direct runoff UV254, with an average value of 0.044 cm^{-1} for the year. It was assumed that the direct runoff area for the reservoir has similar characteristics to the watershed of Purgee Brook. The validity of this assumption and sensitivity of reservoir UV254 to direct runoff contribution warrants future study.

Table 4.26 summarizes DCR data for in-reservoir UV254 sampling (condensed from MDC 2003). It is notable that UV254 levels in Thomas basin vary more widely than in the South and North Basins. This variation occurs since Thomas Basin water quality is much more readily impacted by Quabbin Transfer than that of the other two locations. Lower limits of ranges in UV254 at South Basin and North Basin at all depths are similar, although maximum measured levels at North Basin are lower than those at South Basin. This gradient likely occurs due to decay of UV254 absorbing substances within the reservoir.

Table 4.26 In-Reservoir UV254 Data for Wachusett, 1998-2002 (from MDC 2003)

Sampling Station	UV254 (cm^{-1}) 1998 - 2002
Thomas Basin (E)	0.026 - 0.140
Thomas Basin (M)	0.026 - 0.147
Thomas Basin (H)	0.027 - 0.150
Basin South (E)	0.031 - 0.085
Basin South (M)	0.032 - 0.089
Basin South (H)	0.036 - 0.091
Basin North (E)	0.032 - 0.068
Basin North (M)	0.032 - 0.079
Basin North (H)	0.032 - 0.069

Figure 4.124 presents measured UV254 at Cosgrove for the 2001 – 2002 study period. Notable trends include increasing levels in spring that peak at 0.078 cm^{-1} in summer of 2001 and 0.060 cm^{-1} in summer of 2002, followed by decreasing levels in fall, and fairly consistent low levels in winter. There is an apparent net change in UV254 throughout the study period; withdrawal levels in January 2001 range near 0.045 cm^{-1} while December 2002 levels are approximately 0.030 cm^{-1}, a net reduction of one third.

A water volume weighted material balance for Wachusett for 2001 and 2002 using averaged data for the period indicates that the expected mixed UV254 of input water is

0.062 cm^{-1}. Averaged UV254 data at Cosgrove yields 0.047 cm^{-1} during the study period, indicating a net loss in the reservoir.

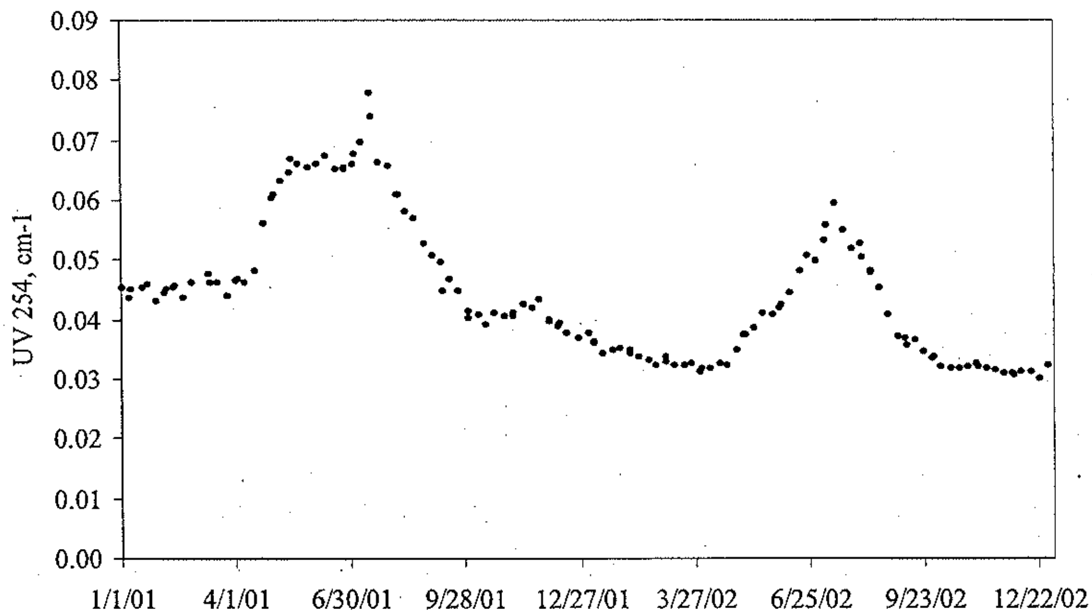

Figure 4.124 Measured UV254 at the Cosgrove withdrawal for 2001 and 2002 (MWRA data).

Setting the 1st order temperature dependent decay rate, COLDK, to zero and running CE QUAL W2 with varied values of the light-induced decay constant ALPHA yielded the results in Figure 4.125. A conservative case (i.e. no decay) is also presented in that figure. It is notable that after September 2001, the conservative case predicts UV254 to be consistently ~0.02 cm^{-1} greater than the data. This difference is reduced during early summer 2002 as measured UV254 levels increase. It is important to note that a significant decrease in predicted UV254 (all cases) occurs in July and August 2002. This is probably a result of the hydrodynamic inaccuracy presented in Section 4.3.3.2. As ALPHA increases UV254 levels approach measured levels. It is notable that ALPHA = 4.0E-5 cm^2/cal and larger predict a dip in measured UV254 in September 2001, indicating that reduction in UV levels during this period is strongly influenced by light. The impact of temperature dependent decay is presented in Figure 4.126, with THETA = 1.03 and ALPHA = 0 in all cases.

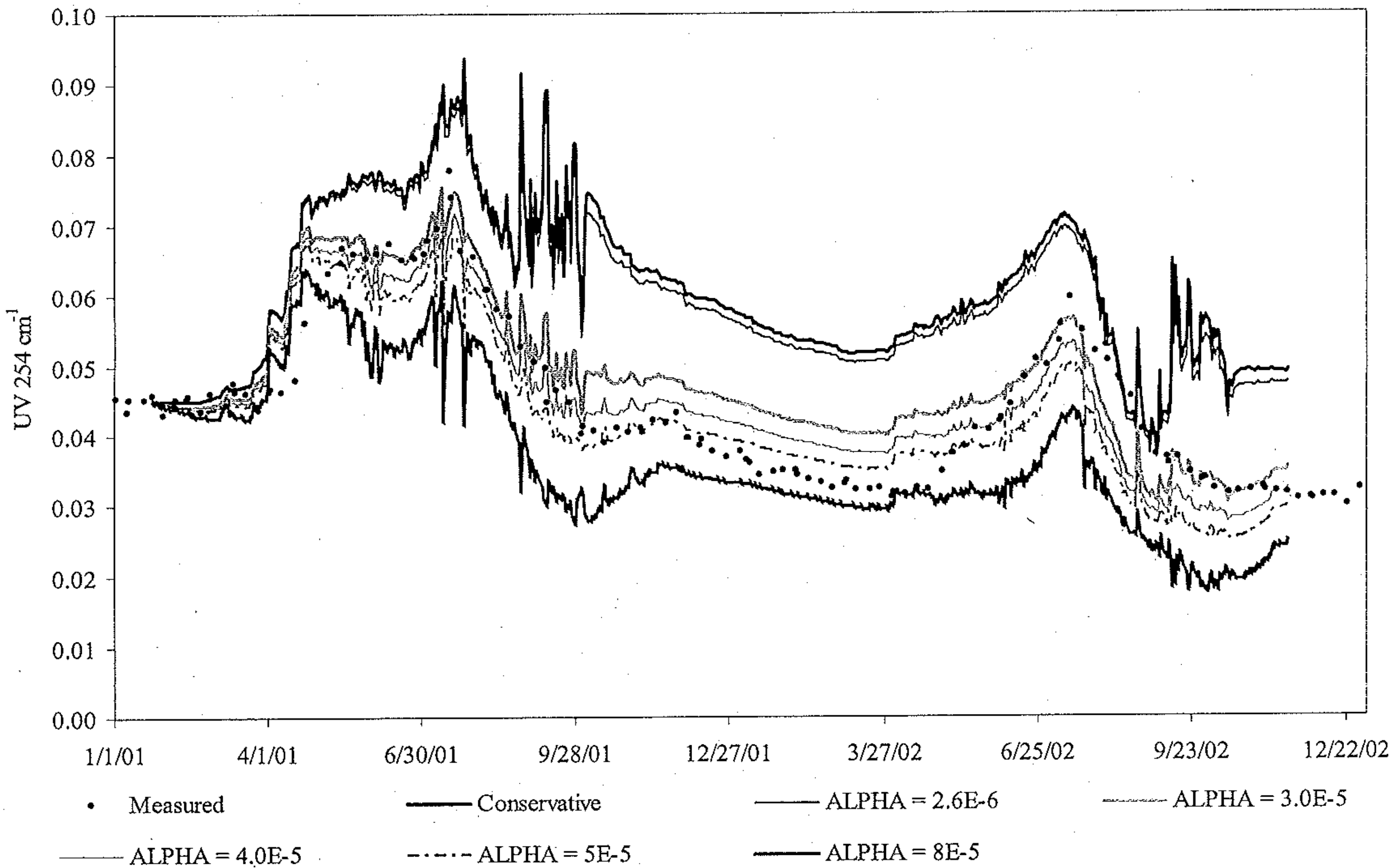

Figure 4.125 Modeled and measured UV254 at Cosgrove with COLDK = 0 and varied ALPHA values (in cm^2/cal).

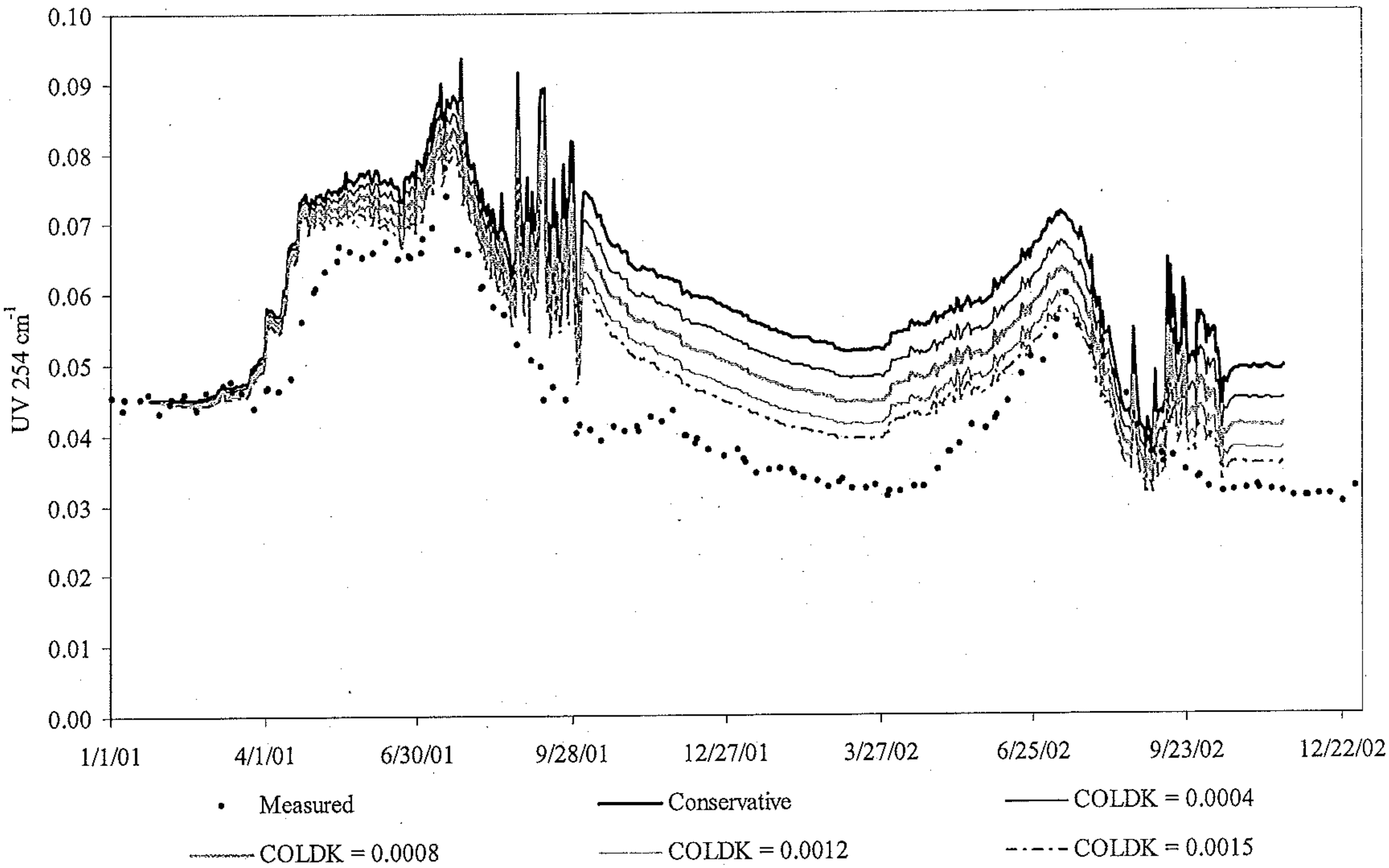

Figure 4.126 Modeled and measured UV254 at Cosgrove with ALPHA = 0, THETA = 1.03, and varied COLDK values (day^{-1}).

The values of COLDK presented are similar to the calibrated value of RDOMDK (0.0008 day^{-1}). UV254 absorbance reflects the presence of aromatics, which also indicates humic materials. Humic materials are environmentally refractory, therefore COLDK was based on RDOMDK. Results are similar to those presented in Figure 4.127, with COLDK = 0.015 resulting in similar trends to that of COLDK = 0 and ALPHA = 3.0E-5. Use of only temperature dependent decay does not predict the brief minimum in UV254 that occurred in September 2001, suggesting that light induced decay is an important process.

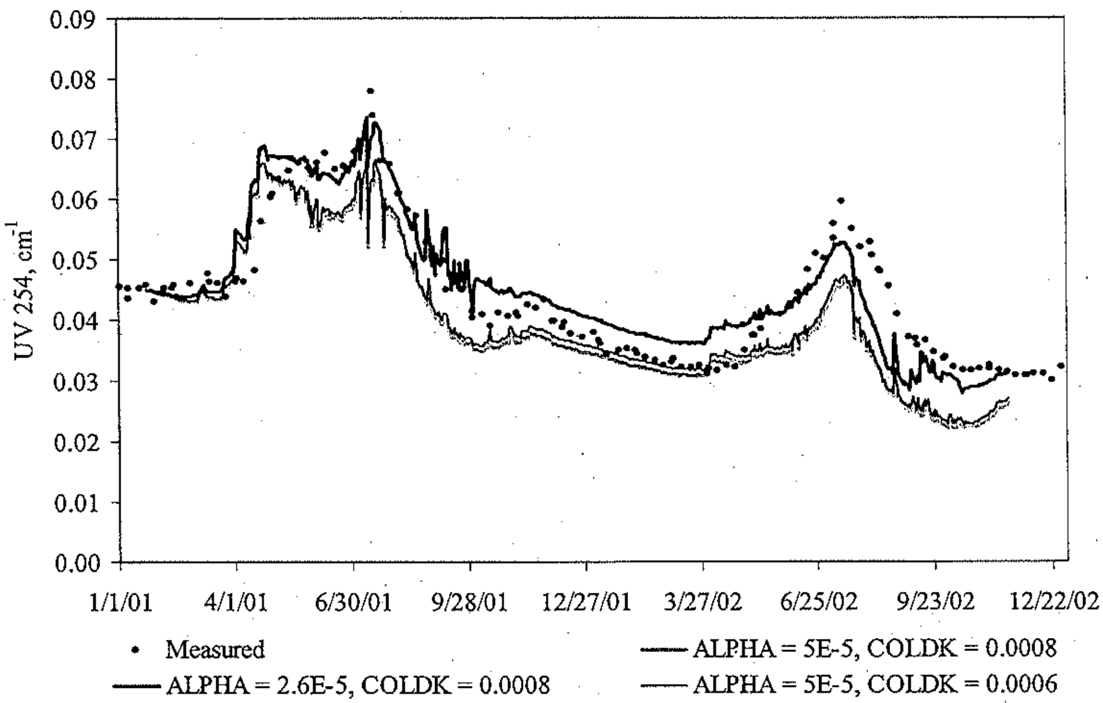

Figure 4.127 Modeled and measured UV254 at Cosgrove with THETA = 1.03, and varied COLDK values (day^{-1}) and ALPHA values (cm^2/cal).

Setting COLDK equal to RDOMDK or 0.0008 day^{-1} with ALPHA = 2.6E-5 cm^2/cal (one order of magnitude larger than that selected in Roberts, 2003) produces the best fit between modeled and measured UV254 for the period. UV254 is predicted to within 0.005 cm^{-1} throughout the study period, with the largest deviation resulting from inadequate wind mixing in July of 2002. However, increasing ALPHA to 5E-5 cm^2/cal and decreasing COLDK to 0.0006 day^{-1} more accurately predicts the September 2001 minimum (as shown, there is little difference between COLDK = 0.0008 day^{-1} and 0.0006 day^{-1} with ALPHA = 2.6E-5 cm^2/cal).

Generally, varying UV254 decay rates results in either overprediction during 2001 levels or underprediction of 2002 levels. Only two UV254 data points were available for each minor tributary during 2002, which may result in decreased modeling accuracy. It was decided to select ALPHA = 2.6E-5 and COLDK = 0.0008 as these values produce the best approximation of UV254 data.

4.4.5 SUVA Results

Modeled UV254 and DOC results can be used to predict specific UV absorbance of dissolved organic matter. SUVA is calculated by dividing UV254 absorbance/m by DOC in mg/L. Since UV254 is an indicator of aromatic rings in organic carbon molecules, this parameter indicates the composition of organic carbon in a system. High SUVA (4 or greater) indicates large quantities of aquatic humics and other compounds of high hydrophobicity. Low SUVA (2 and less) indicates the dominance of low molecular weight compounds. Figure 4.128 presents SUVA determined from measured UV254 and DOC data, and SUVA determined from modeled UV254 and DOC results. The modeled UV254 and DOC series implemented were those resulting from the TOC and UV254 calibrations presented in Section 4.4.3.1 and Section 4.4.4.

It is notable that the data shows a slight but significant long term decline in SUVA over the study period, from ~2.2 L/mg-m to 1.7 L/mg-m. SUVA predicted from modeled parameters decreases from 2.05 to 1.74 L/mg-m during the period. The model also captures short term trends; SUVA increases to ~2.5 L/mg-m during both summers; SUVA from the model is in the same range. Inaccuracy in modeled UV254 during the summer of 2002 results in an SUVA inaccuracy of 0.2 L/mg-m. It is interesting to note that a slight concave-down trend in SUVA data occurred between September of 2001 and April of 2002. A similar trend occurs in SUVA as predicted by the model. Reasonable agreement between SUVA from the model and measured data confirms successful DOC and UV254 calibration.

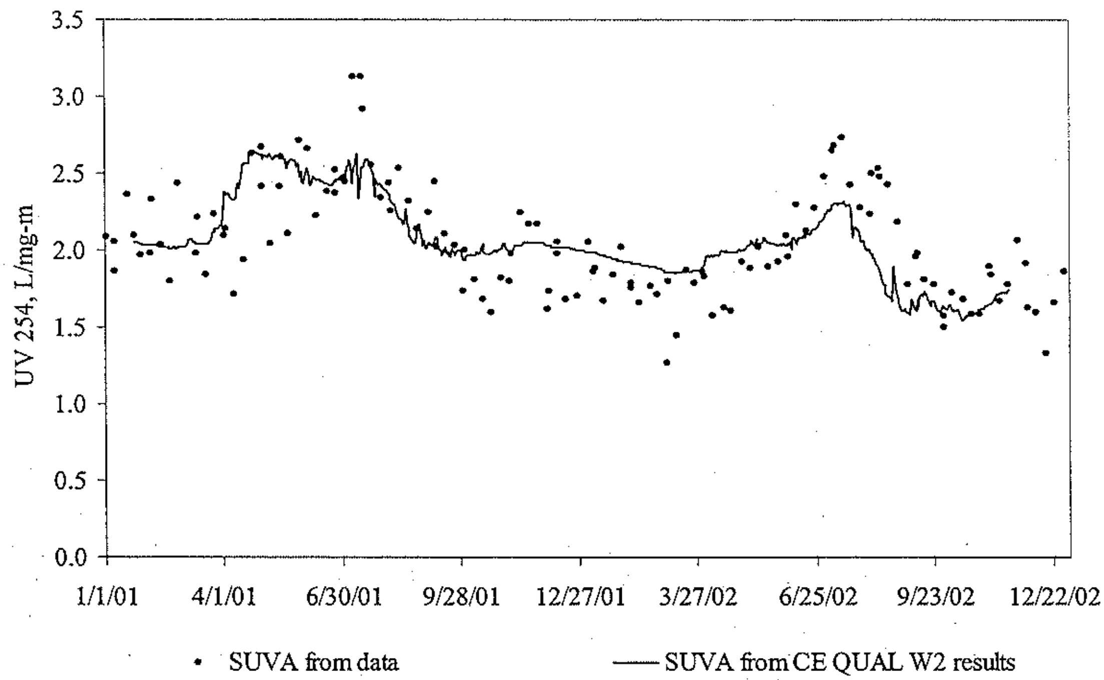

Figure 4.128 SUVA at Cosgrove as determined from data and model results for 2001-2002 (MWRA data).

5. Validation Results

Following calibration, CE QUAL W2 was used to predict constituent concentrations at Cosgrove by applying parameter values and constituent data in the 2000 hydrodynamic model (see Section 4.1). All assumptions and parameter values used during calibration were applied for validation, including the following:

- Inflow orthophosphate equals 50% of measured total phosphorus
- DOC equals 95% of inflow TOC, with POC constituting the remaining 5%
- RDOM:LDOM equals 4:1 of measured inflow DOC
- RDOM:LDOM equals 9:1 of measured Cosgrove DOC (initial condition)
- All TOC constituent parameters values implemented were used by Roberts (2003) except those varied in this study as presented in Table 4.22
- UV254 constituent parameter values implemented as presented in Section 4.4.4
- Constituent initial values are uniform in-reservoir and set equal to the concentration at Cosgrove at the beginning of the simulation
- Direct runoff constituent data is the same as for Purgee Brook at Quabbin during 1998 and 1999
- Precipitation constituent concentrations are assumed constant except for nitrate and ammonia, for which data exist for 2000

Two additional data preparation assumptions were necessary to successfully validate the model. Quabbin Reservoir outflow constituent data was available for Chicopee Valley Aqueduct water during 2001 and 2002; this data was used for constituents in Quabbin Transfer to Wachusett Reservoir. However, no CVA data was available between the end of data collected by Garvey (2000) in January 2000 and when MWRA constituent records at CVA begin in December 2000. Quabbin transfer concentrations were therefore assumed to be equal to the 2001 – 2002 mean CVA constituent concentrations, presented in Section 4.4. This assumption is reasonable since Quabbin constituent levels are relatively consistent.

The second departure from calibration methods required adjustment of inflow UV254 data. Notations appended to tributary data indicated that various absorbance cell path lengths were used to measure UV254. Large deviation between TOC/UV254 correlations indicated that some UV254 data were reported as absorbance/cm, while other data were recorded as absorbance/10 cm. An example may be seen in 2000 West Boylston Brook UV254 and TOC data, shown in Figure 5.1.

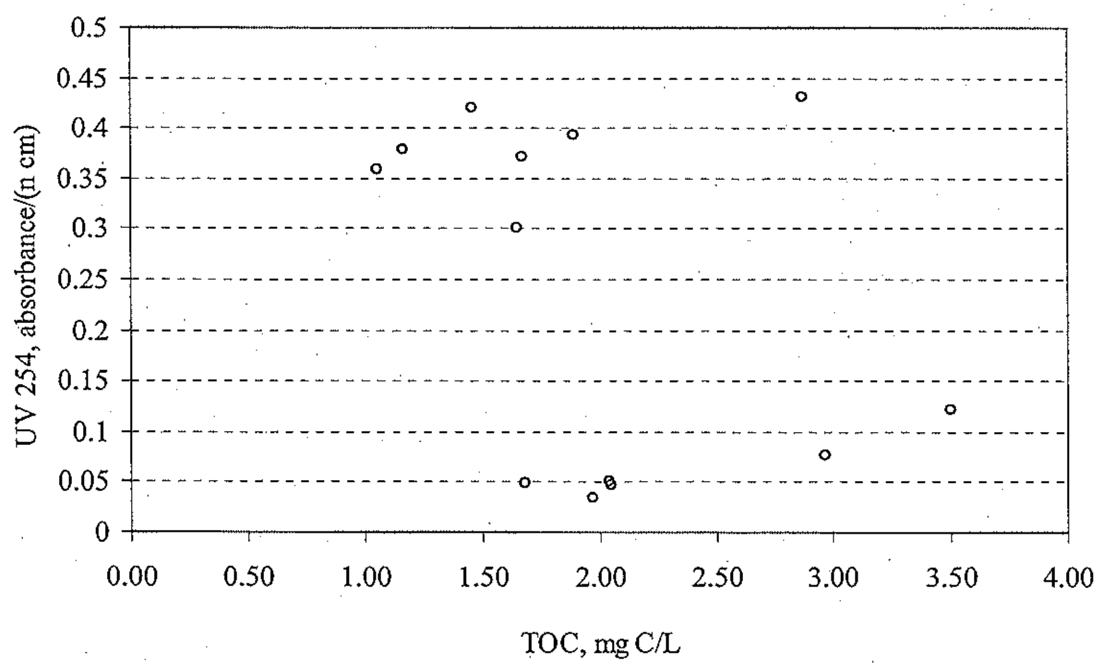

Figure 5.1 Measured UV254 vs. measured TOC at West Boylston Brook (DCR data)

The data are clustered in two areas, one between 1 and 3 mg/L TOC, and 0.3 and 0.45 cm^{-1} UV254, and the other between 1.7 and 3.5 mg/L TOC and 0.035 and 0.12 cm^{-1}. The domains of the clusters overlap significantly, while the majority of UV254 measurements of the former are approximately one order of magnitude larger than that of the latter. It was therefore apparent that the cluster of greater absorbance consists of data reported as absorbance/10 cm, and was therefore adjusted to absorbance/cm for consistency with the majority of the data. Errant data in other tributaries was adjusted similarly. It is important to note that these adjustments were made as consistently as possible, where shown to be applicable by the technique described above.

5.1 TOC Validation Results

Results of the TOC validation are shown in Figure 5.2. Algal productivity was higher in 2000 than in 2001 and 2002, with maximum concentrations of 1.1 mg of algae as carbon per liter estimated from enumeration data, as compared to 0.43 and 0.55 mg C/L as estimated in 2001 and 2002.

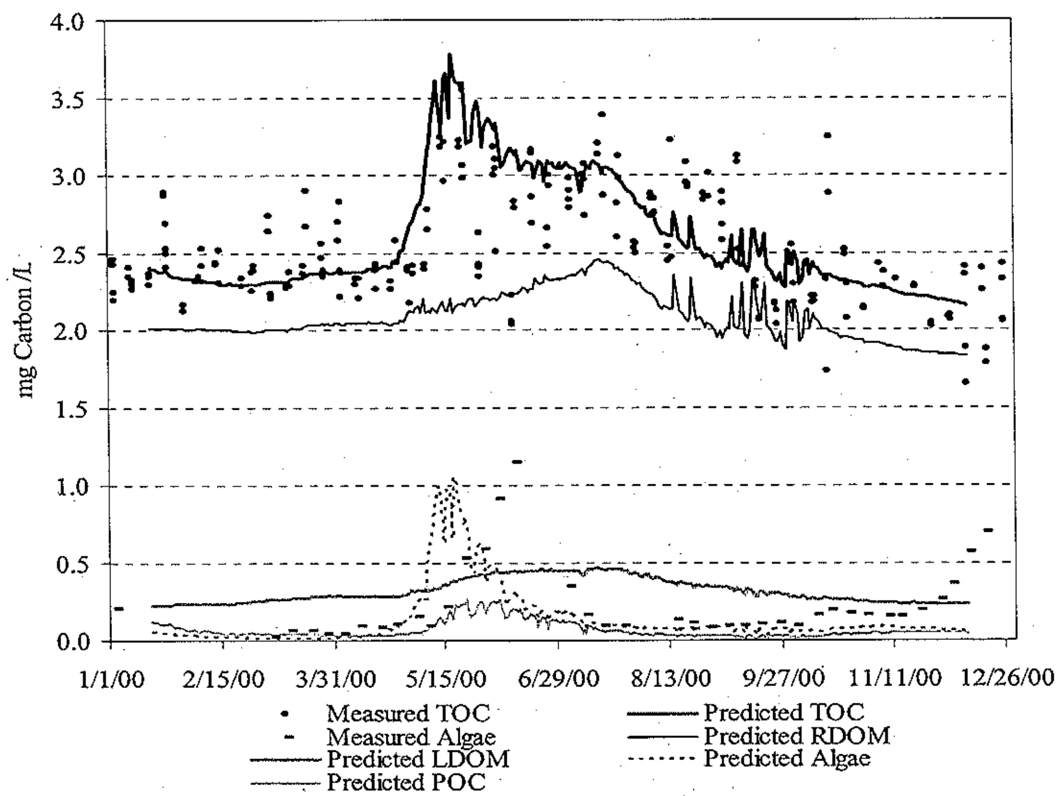

Figure 5.2 Measured TOC and Algae with modeled TOC components at Cosgrove withdrawal in 2002 (data from DCR and MWRA).

The majority of Cosgrove TOC measurements range from 2.0 to 3.4 mg/L, with the minimum occurring at the end of 2000, and the maximum in July. CE QUAL W2 predicts a peak in algae at 1.0 mg C/L on May 18. This concentration is within 10% of 1.1 mg/L, the value estimated from data, although it occurs approximately 1 month earlier than the peak measured value, which occurred on June 12. Additionally, the model does not predict the increase in phytoplankton that occurred in December of 2000, an understandable shortcoming as the model was calibrated to predict the spring and summer bloom. It is important to recall the algae modeling limitations presented in Section 4.4.2.2 when examining the algal levels predicted by this validation.

The predicted spring peak in algal concentration results in overprediction of TOC during May 2000. However, this is the only period when predicted TOC deviates from the range of values shown by the data. The model predicts a net decline in TOC from 2.40 to 2.16 mg/L throughout the year. This decline is driven primarily by a decline of RDOM (concentrations decrease from 2.0 to 1.8 mg/L during 2000); LDOM is generated by algal excretion and mortality, and thus remains constant (a net increase of 0.01 mg/L is predicted). As a result, the predicted ratio of refractory to labile DOM at Cosgrove decreases from 9:1 to 8:1 during the year.

5.2 Nutrient Validation Results

Figure 5.3 shows measured and predicted orthophosphate at Cosgrove in 2000. Measured orthophosphate ranges between the detection limit, 0.0025 mg P/L, and 0.0082 mg P/L on April 4. No significant seasonal trend occurred, although springtime levels were slightly higher than those recorded later in the year.

Predicted orthophosphate concentrations increase rapidly during February and March, with a maximum concentration of 0.015 mg P/L occurring in April. This value is just less than double the maximum measured value. Predicted phosphate then decreases rapidly, reaching a minimum concentration of 0.002 mg P/L in mid-May. This rapid decline is likely a result of the algal bloom that is predicted to occur at that time. After the predicted minimum, orthophosphate levels increase slowly, reaching a concentration of 0.0094 at the end of the year.

Predicted orthophosphate trends differ significantly from those that occur in the data, however, concentrations are so low that predictions are difficult to evaluate. Since predicted and measured orthophosphate are of the same order of magnitude and modeled orthophosphate does not exhibit a consistently increasing or decreasing trend, the results are considered reasonable.

Figure 5.4 shows measured ammonia and modeled ammonium at Cosgrove during 2000. Measured ammonia ranges between the detection limit of 0.005 mg/L on April 4 and

May 2, and 0.035 mg/L on July 11. Modeled ammonium increases steadily from the initial value of 0.008 to 0.03 on July 22, and then decreases slowly. Although predicted ammonium at the end of the year is approximately 2 times as large as the measured values, the low concentrations make accurate prediction difficult and this discrepancy is acceptable.

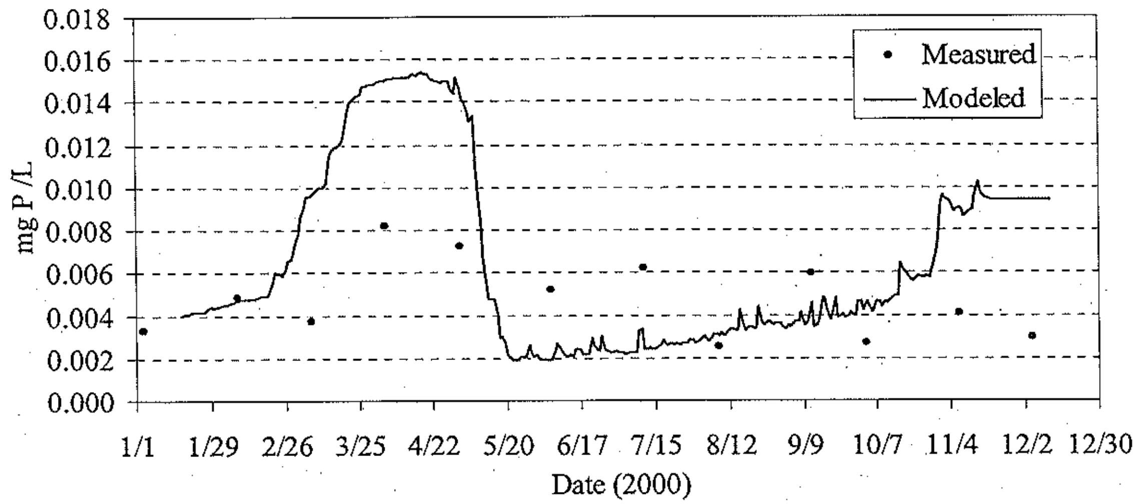

Figure 5.3 Measured and predicted orthophosphate at Cosgrove, 2000 (MWRA data)

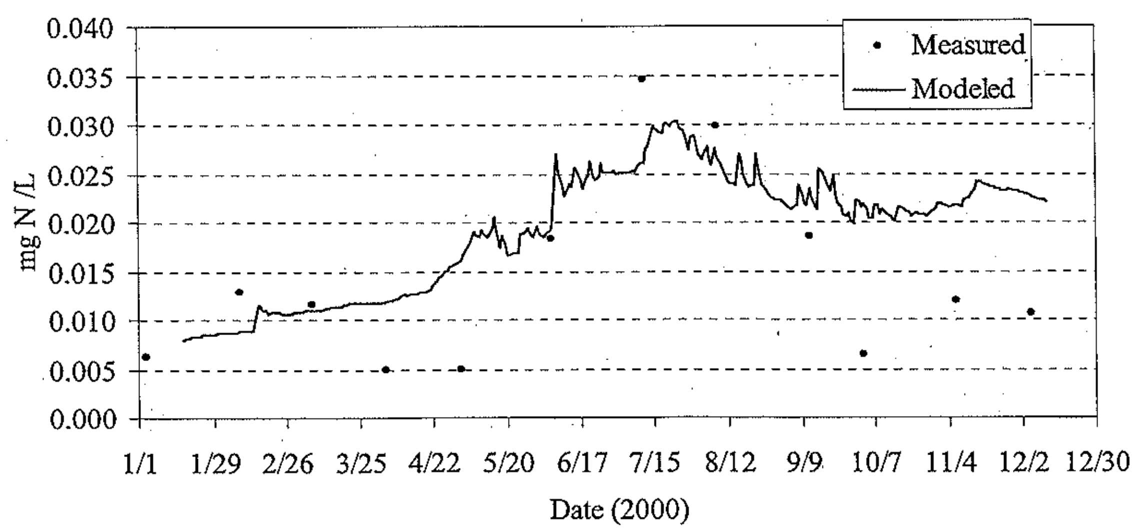

Figure 5.4 Measured ammonia and predicted ammonium at Cosgrove, 2000 (MWRA data)

Figure 5.5 shows measured and predicted nitrate at Cosgrove in 2000. Measured nitrate ranged between 0.027 mg N/L in October 2000 and 0.108 in June 2000. Modeled nitrate ranged between 0.0019 mg N/L in May 2000 and 0.99 mg/L in April. The sudden decline in predicted nitrate is a result of the predicted algal bloom that occurred in May.

Generally, nitrate is of the correct order of magnitude, although the predicted decreasing, then increasing trend occurring in the spring is the opposite of that occurring in the data. Due to the lack of Quabbin transfer constituent data and the fact that nitrate concentrations recover after this minimum indicate that the calibration is adequate.

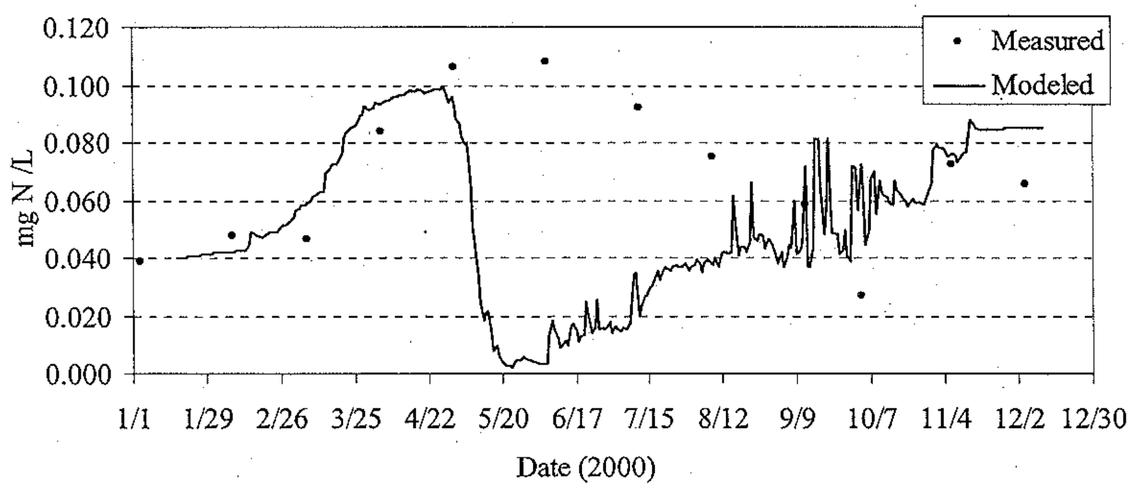

Figure 5.5 Measured and predicted nitrate at Cosgrove, 2000 (MWRA data)

5.3 UV254 Validation Results

Figure 5.6 presents predicted and measured UV254 at Cosgrove Aqueduct. Measured UV254 at Cosgrove generally ranged between 0.037 (measured in January) and 0.085 cm^{-1} (measured in July) during 2000 with several outliers at the beginning and end of the year. A net increase of approximately 0.008 cm^{-1} UV254 absorbance occurred between the beginning and the end of the year. Also shown in Figure 5.6 is predicted conservative UV. The conservative case predicts UV254 levels 0.017 cm^{-1} greater at the end of the year than with decay. It is notable that there is a large degree of variability in August and September, probably resulting from wind mixing.

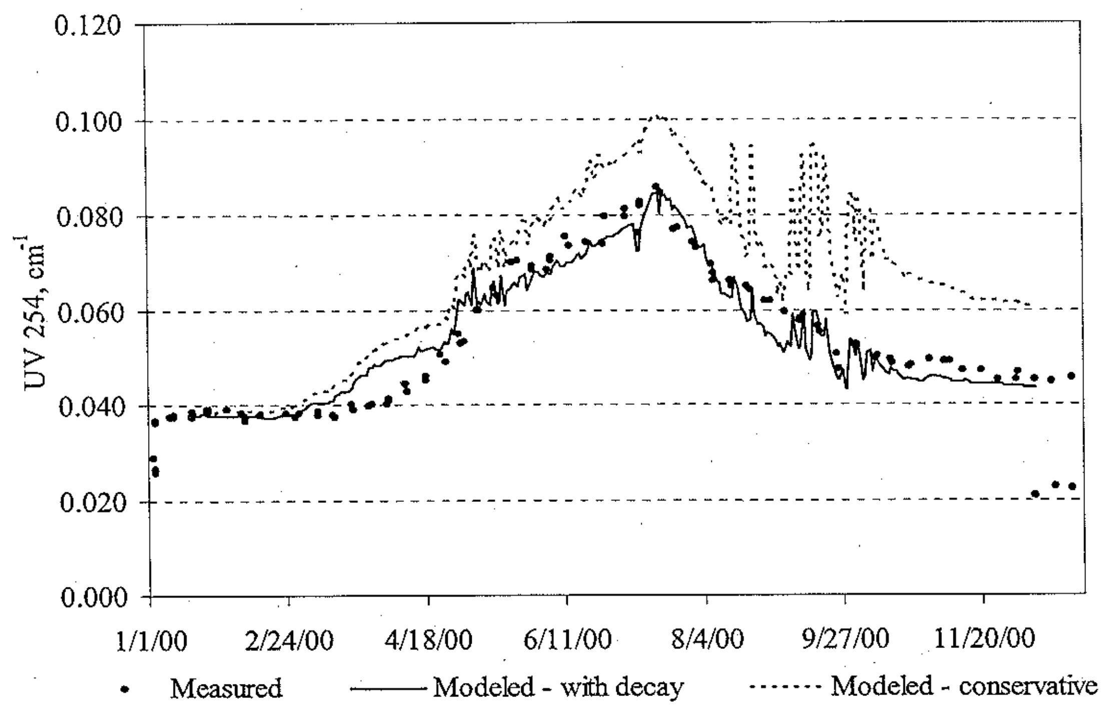

Figure 5.6 Measured and predicted UV254 at Cosgrove, 2000 (MWRA data)

UV254 modeled with decay generally follows the trend and magnitude of measured UV254 at Cosgrove. The largest deviations were 0.008 cm^{-1} on April 4 and 0.007 cm^{-1} on September 5, although these deviations were quickly corrected. There is a tendency of underprediction from the beginning of May through the end of the year, but peak predicted UV254 occurs on July 15, only two days prior to the maximum measured value. Note that UV254 levels measured on September 25 and 26 were lower than levels for the week before and the week after and that the model predicted a sudden decrease and increase in UV254 as well. However, it is also notable that the model predicted a similar decrease in late August and early September that did not actually occur. By December, predicted UV254 deviated from measured UV254 by 0.002 cm^{-1}. The additional UV254 data available for 2000 provides validation results that are better than the calibration results.

5.4 SUVA Validation Results

It is important to note that, unlike in 2001 and 2002, DOC data exist for Cosgrove Aqueduct for 2000. Examining SUVA during calibration required assuming that 95% of TOC was DOC, and calculating SUVA with calculated DOC values. For the 2000 validation, DOC measurements were used until they were discontinued on November 27. Figure 5.7 presents measured TOC and DOC at Cosgrove throughout 2000. It is notable that measured DOC values are within the range of TOC values throughout the study period except for eight data points beginning October 31. For the ensuing month, measured DOC was consistently greater than TOC, indicating sample contamination during measurement. The resulting DOC data is between ~1.5 and ~3.5 mg/L greater than TOC, resulting in declining SUVA values that are likely not representative of conditions in the reservoir. SUVA from measured data beyond October 31 was therefore ignored.

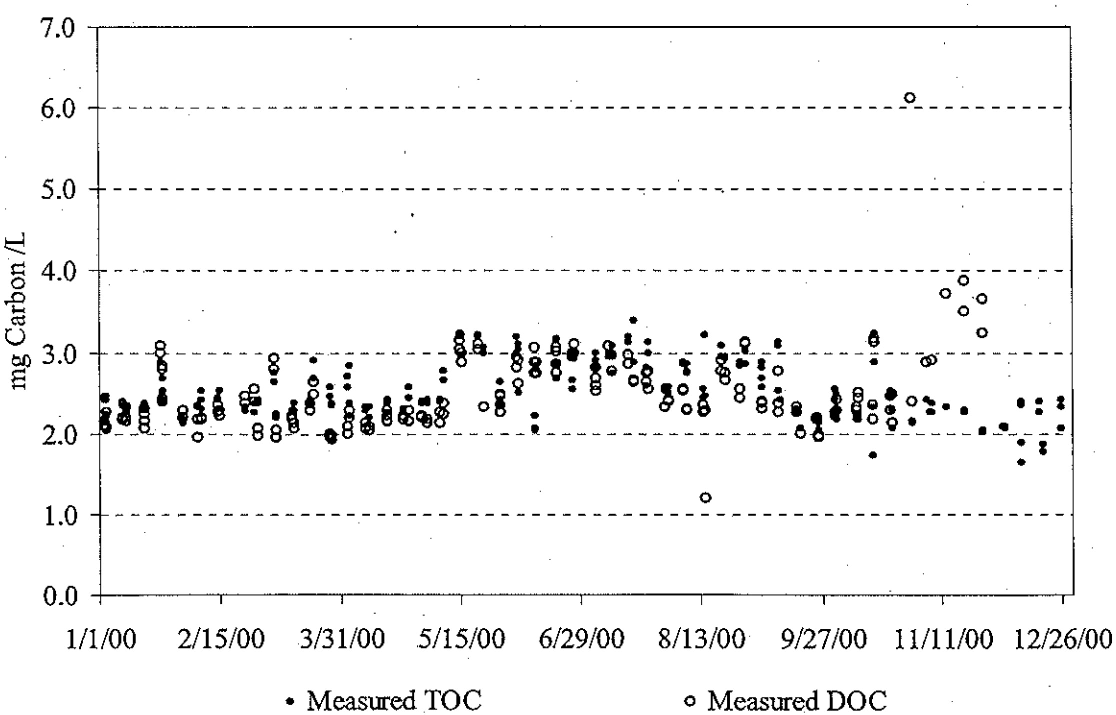

Figure 5.7 Measured TOC and DOC at Cosgrove during 2000 (MWRA data).

Figure 5.8 shows SUVA calculated from data and from model outputs. Calculated SUVA data increased from ~1.5 L/mg-m in January to ~3 .0 L/mg-m in mid-July. SUVA then declined to ~2.0 L/mg-m in late October. SUVA in November was neglected. SUVA from model results followed these trends closely and was within the range of data derived SUVA from January through June. SUVA levels from the July peak through November were underpredicted by 0.1 to 0.2 L/mg-m. In November and December, predicted SUVA is constant at approximately 2.1 L/mg-m. This cannot be corroborated by data derived SUVA, but the successful validation of the UV254 and TOC model predictions indicate that this result is appropriate and perhaps more representative of reservoir conditions than SUVA calculated from the apparently erroneous data.

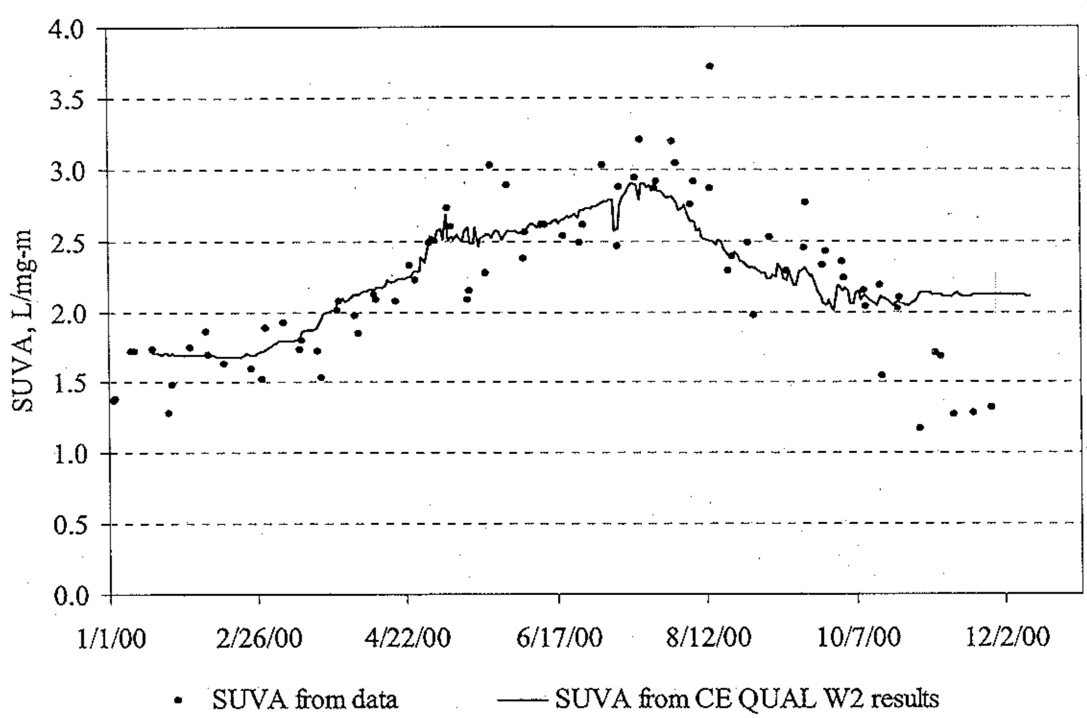

Figure 5.8 SUVA at Cosgrove as determined from data and model results for 2000 (MWRA data).

6. SIMULATIONS

Successful calibration and validation of the CE QUAL W2 water quality model for the Cosgrove withdrawal of Wachusett Reservoir indicate the possibility of simulating controlled and uncontrolled events. Several simulations, presented in this chapter, were run to predict the response of withdrawal water quality to increasing the quantity of Quabbin Transfer (Sections 6.1 and 6.2); to adding large quantity of watershed runoff in the late summer and early fall (Section 6.3); and to bypassing Wachusett Reservoir with Quabbin Transfer (Section 6.4).

6.1 Increased Quabbin Transfer during Dry Spring.

The 2002 calendar year was hydrologically unusual for Wachusett, in that tributary runoff was not adequate to maintain water surface elevation in the spring. In both 2000 and 2001, increased runoff caused the reservoir to spill, and no water was transferred from Quabbin. No spilling occurred in 2002, and periodic Quabbin Transfer was necessary.

CE QUAL W2 was used to evaluate the impact of additional Quabbin transfer during the Spring of 2002 on Wachusett Reservoir water quality. Since Quabbin Reservoir water is characterized by lower concentrations of various water quality constituents than water received from the tributaries, it is anticipated that increased transfer will decrease Wachusett constituent levels.

To test this conjecture, average daily transfer at 8.7 m^3/s (198 MGD) was included on days when no transfer was recorded. To maintain reservoir water surface elevation, the surplus water was discharged to the Nashua River. Figure 6.1 shows Quabbin Transfer as calibrated and as implemented for this simulation for 2001 and 2002. The additional transfer and discharge to Nashua River totals 54.5 million m^3 (14.4 billion gallons) for 79 days between March 4 and June 13 2002.

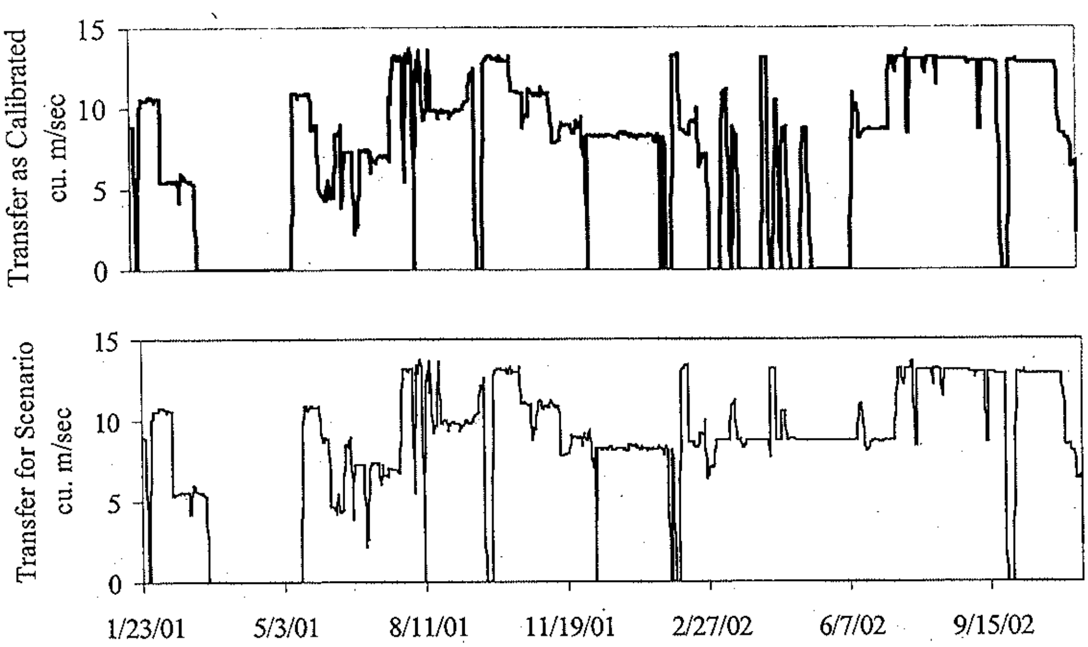

Figure 6.1 Quabbin Transfer resulting from water balance (top figure) and as implemented for the increased Spring 2002 transfer scenario (bottom figure).

Figure 6.2 presents TOC and RDOM at Cosgrove as predicted by this scenario as well as TOC and RDOM as predicted originally. Other TOC components are not shown as variation between those constituents as calibrated and resulting from the scenario are minimal. The impact of additional transfer begins on March 25 2002, three weeks after additional transfer begins. The additional transfer results in a reduction of TOC by as much as 0.2 mg/L during the spring algal bloom. The resulting reduction is not due to algae levels, however; it results from decreased RDOM due to dilution. These predicted peak TOC levels are not significantly different from those resulting from calibration. In general, CE QUAL W2 predicts that the impact of this additional transfer on TOC levels is noticeable but small.

Figure 6.3 presents UV254 as predicted by this scenario, as well as UV254 as predicted originally, at Cosgrove. As with TOC, the impact of increased transfer on UV254 becomes noticeable on March 25. The additional Quabbin reservoir water reduces peak UV254 levels by approximately 0.008 cm^{-1}, although it does not significantly impact

UV254 levels later in the year. This reduction during peak periods may be significant, especially if DBP levels in the distribution system are high.

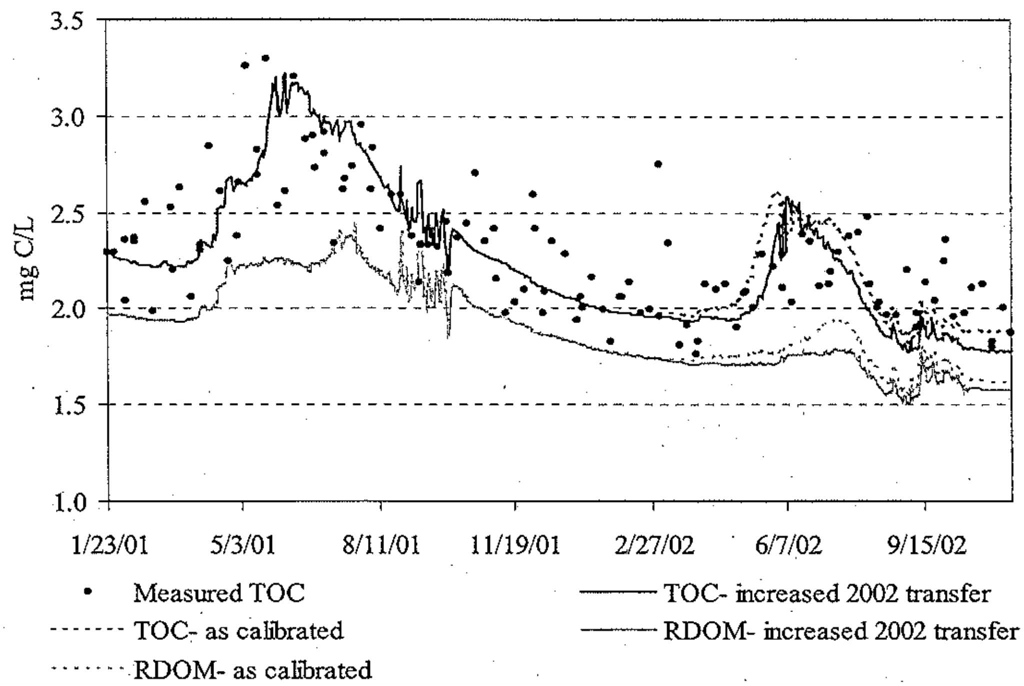

Figure 6.2 TOC and RDOM as predicted by CE QUAL W2 during calibration and by the increased 2002 Quabbin Transfer simulation.

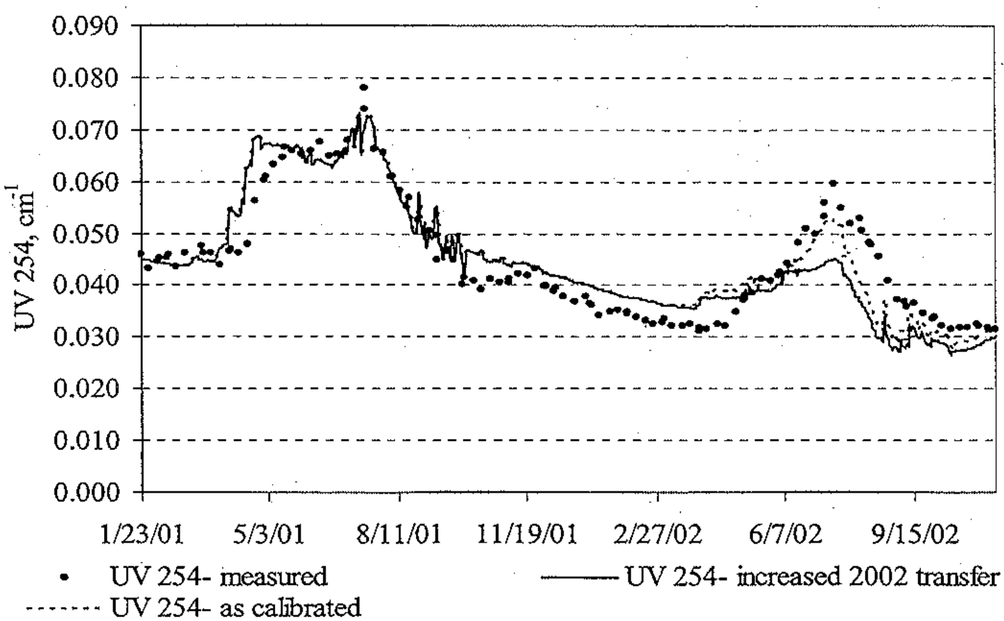

Figure 6.3 UV254 as predicted by CE QUAL W2 by calibration and by the increased 2002 Quabbin Transfer simulation.

6.2 Increased Quabbin Transfer during Wet and Dry Springs

During March and April 2001 several large precipitation events resulted in significant tributary discharges to Wachusett. Although Quabbin transfer was discontinued at the beginning of this period, Wachusett WSE increased and the reservoir spilled to Nashua River. As a result, TOC and UV254 at Cosgrove increased, coinciding with the spring algal blooms. The impact of transferring from Quabbin during this wet period was investigated in this scenario. Also included in this scenario is additional 2002 transfer (see Section 6.1).

Figure 6.4 shows Quabbin Transfer as implemented in the calibration model and in the scenario of increased Spring of 2001 and 2002 transfer. As for the increased 2002 transfer scenario, average daily transfer of 8.7 m^3/s (198 MGD) was added on days with no transfer. This additional transfer accounts for 51.1 million m^3 (13.5 billion gallons) of additional transfer during 68 days in 2001, in addition to 54.5 million m^3 (14.4 billion gallons) for 79 days during 2002. This water was released to Nashua River to prevent change in WSE. Resulting Nashua River discharges were as much as 32.4 m^3/s (740 MGD) and do not consider possible flooding (perhaps a significant omission).

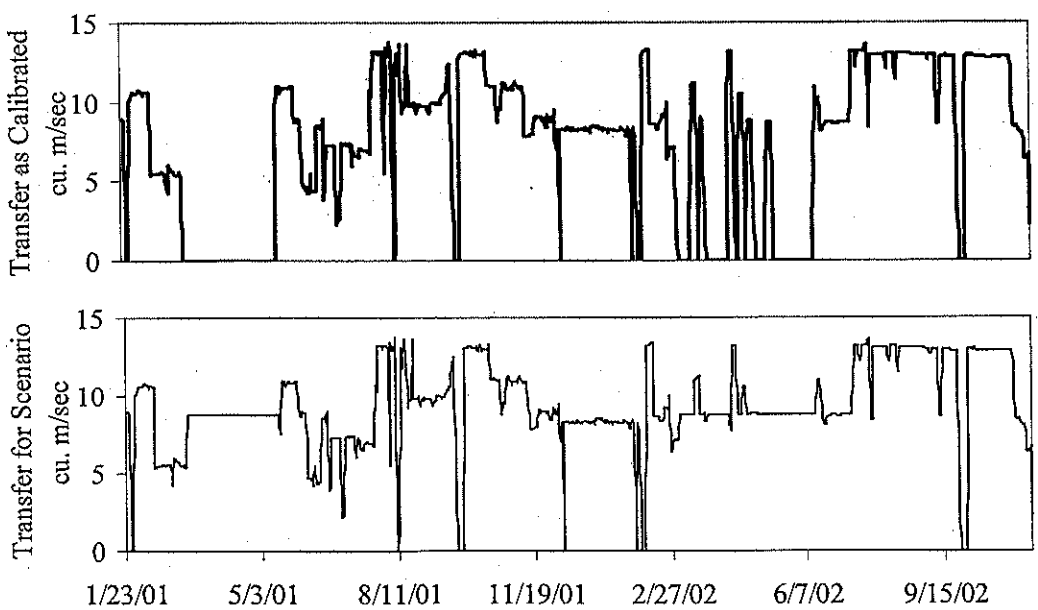

Figure 6.4 Quabbin Transfer resulting from water balance ('calibrated transfer') and as implemented for the increased Spring 2001 and 2002 transfer scenario.

Figure 6.5 presents TOC and RDOM as predicted by this scenario as well as TOC and RDOM resulting from the calibration presented in Section 4.4.3.1. Other TOC components are not shown as variation between those constituents as calibrated and resulting from the scenario are minimal. The impact of additional transfer is noticeable by April 7 2001, three weeks after additional transfer begins. The additional transfer results in a reduction of TOC of as much as 0.2 mg/L during the spring algal bloom. This reduction primarily results from decreased RDOM due to dilution. Predicted peak TOC levels are not significantly different from those resulting from calibration. TOC levels remain approximately 0.05 mg/L less than those predicted by calibration throughout the rest of 2001 and the beginning of 2002, although the difference increases in March 2002 resulting from the second instance of increased transfer. By the end of 2002, the predicted TOC concentration at Cosgrove is 1.83 mg/L with increased transfer during both years, and 1.84 mg/L with increased transfer during 2002 only. In general, CE QUAL W2 predicts that the impact of this additional transfer on TOC levels is noticeable but small; the impact is negligible within 1 to 1.5 years of the additional transfer period.

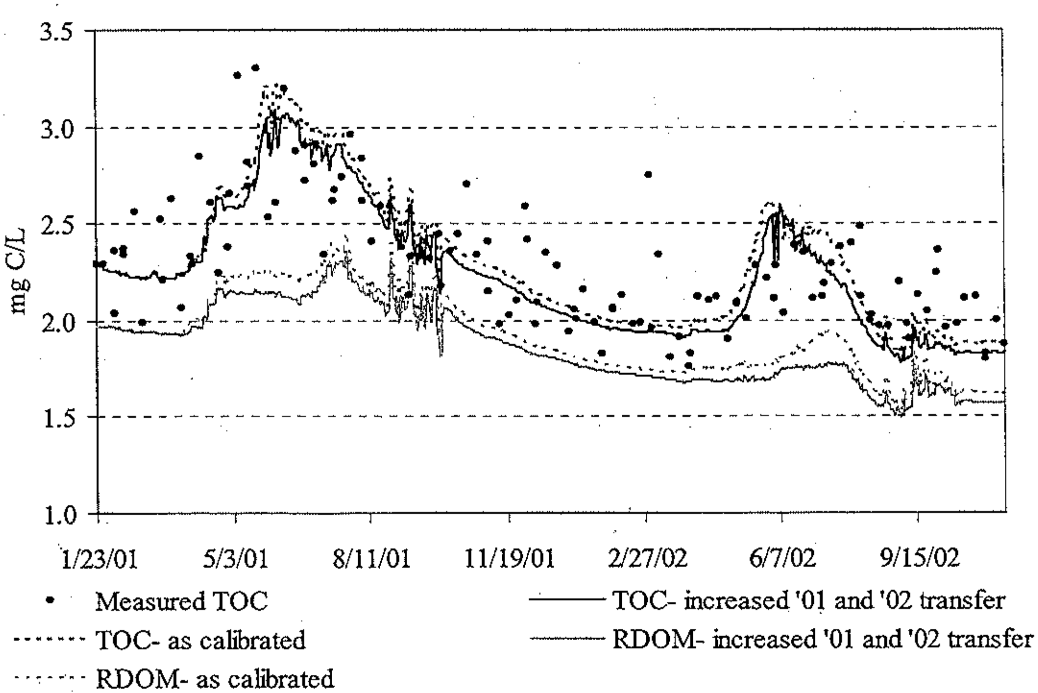

Figure 6.5 TOC and RDOM as predicted by CE QUAL W2 by calibration and by the increased 2001 and 2002 Quabbin Transfer simulation.

Figure 6.6 presents UV254 as predicted by the increased 2001 and 2002 scenario as well as UV254 as predicted by calibration. As with TOC, the impact of increased transfer on UV254 becomes noticeable on April 7 2001. The additional Quabbin reservoir water reduces UV254 levels by approximately 0.008 cm^{-1} between the two predicted peak levels that occur on April 26 and July 22 and does not significantly impact UV254 levels later in the year (UV254 predicted by calibration and by the increased transfer scenario differ by approximately 0.0015 cm^{-1}). UV levels in 2002 are also decreased by approximately 0.008 cm^{-1} during peak levels in spring; a similar decrease to that predicted with increased 2002 transfer only. The predicted impact of 68 days of 8.7 m^3/s transfer of water from Quabbin is noticeable but small.

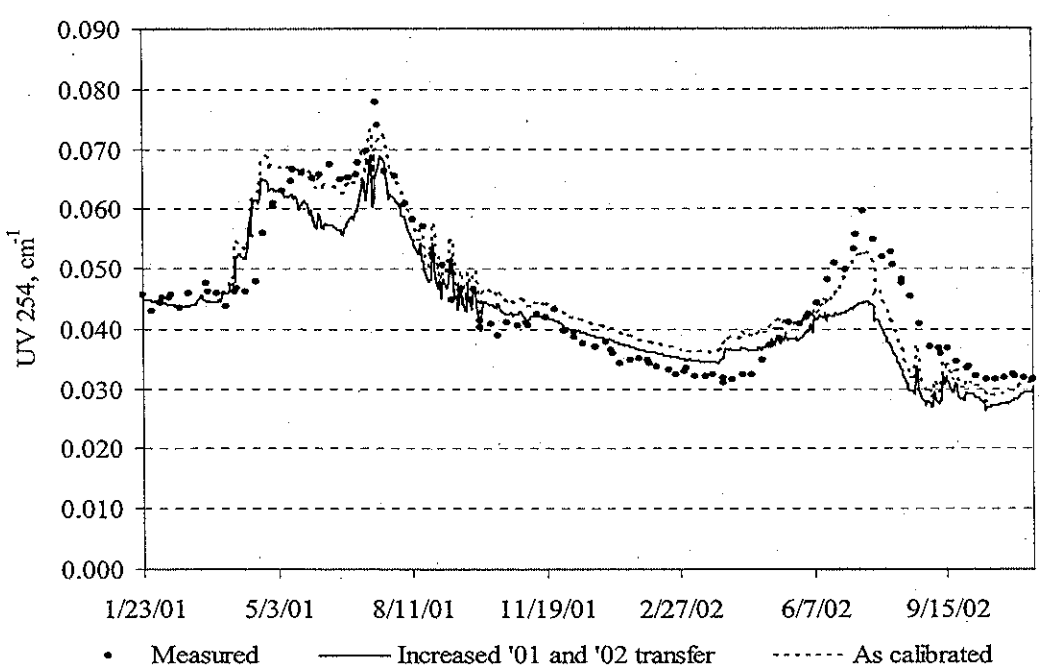

Figure 6.6 UV254 as predicted by CE QUAL W2 by calibration and by the increased 2001 and 2002 Quabbin Transfer simulation.

6.3 Additional 2001 Runoff Period

It is notable that, although 2001 was a dry year, significant tributary runoff in March and April input large quantities of water to Wachusett Reservoir. Water was not transferred from Quabbin during this period, and Wachusett spilled to Nashua River. As a result, water quality at Cosgrove was significantly affected. A relatively dry period followed in which water was constantly transferred from Quabbin, improving water quality at Cosgrove.

A simulation was run with CE QUAL W2 to predict the impact of a second series of wet weather in late summer 2001. It was anticipated that the occurrence of such an event might have prevented Quabbin transfers, thereby resulting in impacted water quality. The additional tributary runoff and precipitation was generated by setting daily inflows (except Quabbin Transfer) as input to CE QUAL W2 for August 1 through September 30 2001 equal to those for March 1 through April 30 of that year. Nashua River withdrawal was changed in a similar manner during the same, modified period. Quabbin Transfer was then adjusted manually to ensure approximate agreement of predicted and measured WSE after September 30 2001. Figure 6. presents inflows implemented in the model for calibration and for the additional storms scenario. Shown is the sum of all tributary inflows, as well as Quabbin Transfer, for both cases in 2001 only (although the model was run for 2001 through 2002 as in the other scenarios, no modifications were made after October 10, 2001 when Quabbin Transfer was restarted).

During the 61 day period of increased precipitation and runoff, 7.4 cm of additional precipitation was included, as well as average daily Quinapoxet and Stillwater River discharges of 6.7 and 5.22 m^3/sec, respectively, as compared to 0.19 and 0.29 m^3/sec as determined through calibration. The modified inflow resulted in a 45.9 million m^3 net increase of water to the 2001 budget. The majority of precipitation occurred on three dates; on March 5 (included on August 5) 4.5 cm fell, on March 21 (included on August 21) 5.3 cm fell, and on March 30 (August 30) 2.8 cm fell. There were also several days on which smaller precipitation events occurred. Peak runoff occurred on March 23 (August 23) and April 10 (September 10).

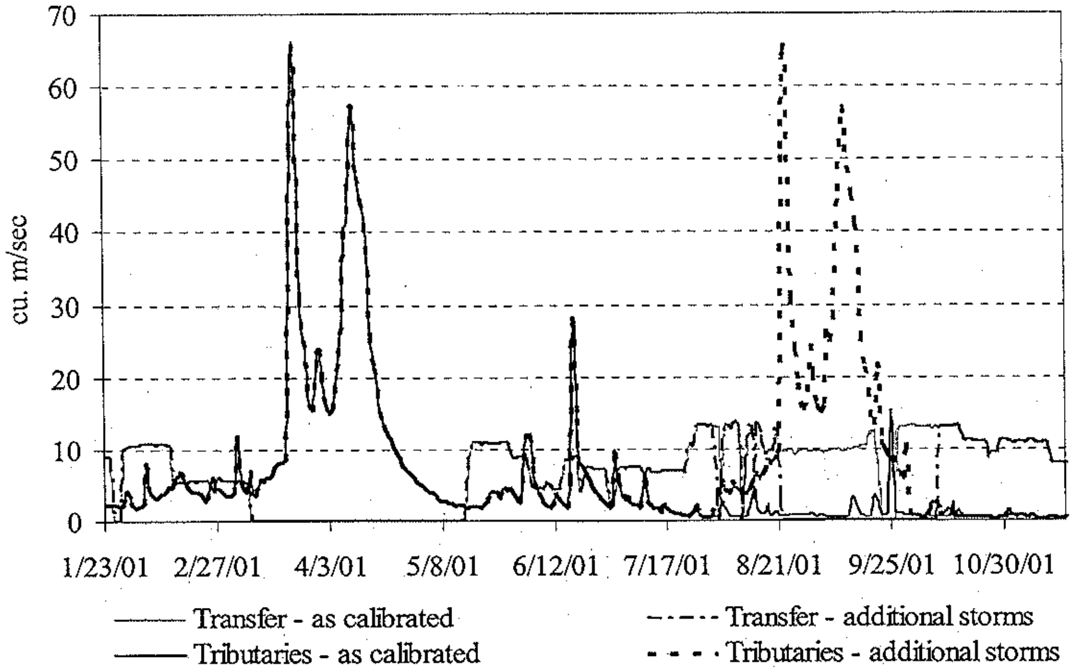

Figure 6.7 Tributary inflows and Quabbin Transfer as calibrated and as implemented in the additional 2001 runoff scenario.

Figure 6.8 presents measured WSE along with WSE as predicted by CE QUAL W2 with inflows as calibrated and with inflows as modified to include the additional 2001 storms. During the additional runoff period, predicted WSE rose to 120.2 m on September 16, approximately 1.3 m larger than the level measured at that time.

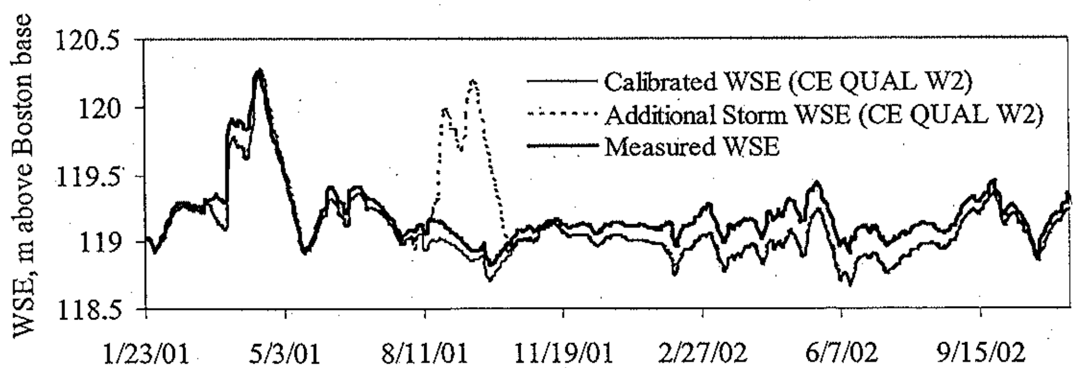

Figure 6.8 Measured WSE with WSE as predicted by CE QUAL W2 during calibration and during the additional 2001 runoff scenario.

The inflow concentration files were examined for the period of increased transfer. In general, constituent concentrations for each tributary were within the range of values measured for other portions of the two year period. It is possible that a large storm event could impact tributary constituent concentrations; however, additional, in-depth study beyond the scope of this research would be required to determine if the impact was generally towards increasing or decreasing concentration, and what the magnitude of the impact would be. Therefore, constituent concentrations as measured during the August through September 2001 period were implemented.

Figure 6.9 presents the impact of the additional 2001 runoff on TOC and TOC component levels in Wachusett Reservoir. The additional runoff results in a significant increase in TOC that begins on August 26 2001. This spike reaches a maximum of 4.5 mg/L on September 21. The spike results from increasing levels of every TOC component.

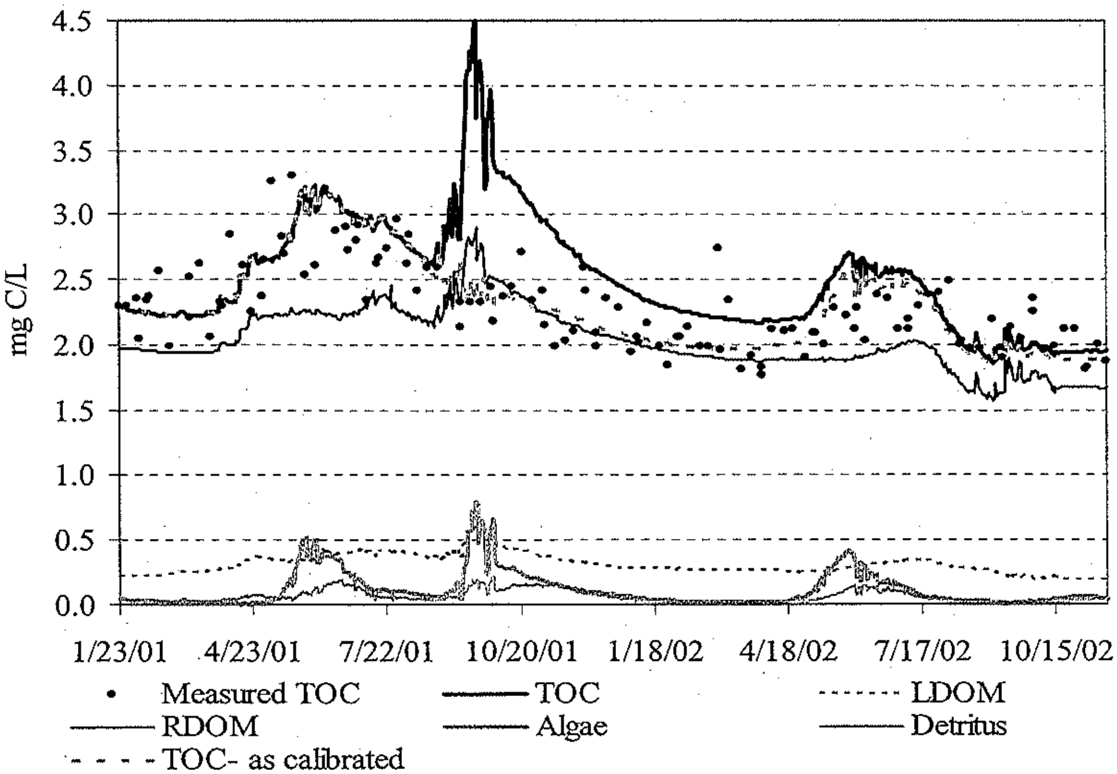

Figure 6.9 TOC as measured at Cosgrove, along with TOC as calibrated, and TOC and TOC components as predicted by the additional 2001 runoff scenario.

CE QUAL W2 predicts a large algal bloom occurring at this time, most likely as a result of increased phosphorus levels due to runoff. This bloom generates more LDOM and detritus than occurred in the calibration model. Decay of the additional LDOM results in additional RDOM. In addition to this autochthonous organic matter generation, allochthonous organic matter is likely also a factor in raising TOC levels. On October 7, after the peak of the bloom as predicted at Cosgrove, TOC levels decline and converge on TOC levels as predicted through calibration. By the beginning of the 2002 algae bloom (May 3) the residual impact of the additional runoff is approximately 0.15 mg TOC/L, and by the end of 2002, the impact is negligible.

Figure 6.10 presents the impact of additional 2001 runoff on UV254 as predicted at Cosgrove compared to measured UV254 and UV254 as calibrated with CE QUAL W2. The additional runoff and lack of Quabbin Transfer results in UV254 levels that are larger than any measured at Cosgrove during the study period. Levels rise to approximately 0.12 cm^{-1} on September 21, twice the suggested MWRA transfer trigger level of 0.06 cm^{-1} (Sung 2003).

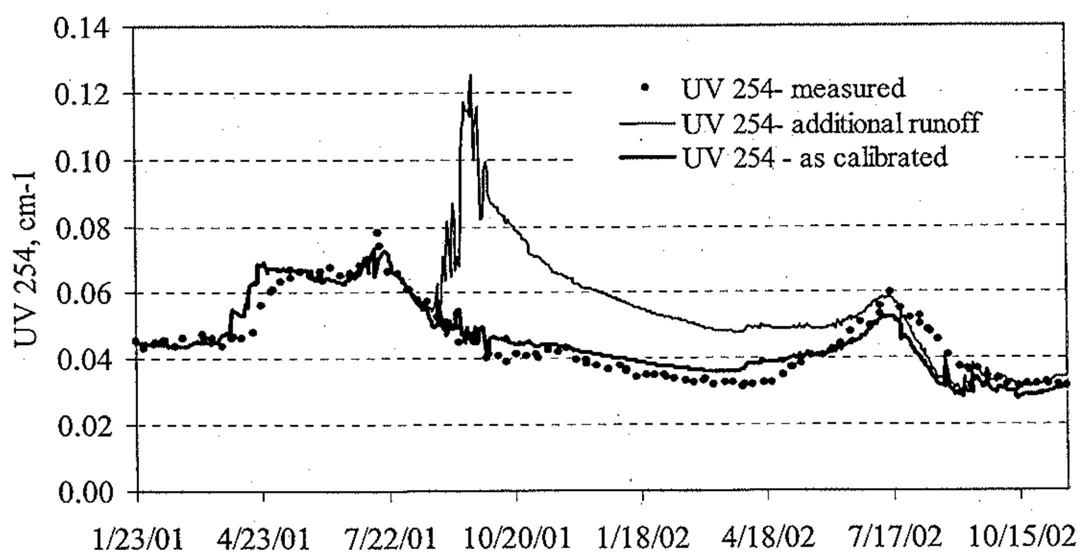

Figure 6.10 UV254 as measured at Cosgrove along with UV254 as calibrated and as predicted by the additional 2001 runoff scenario.

Most likely, the MWRA would not be able to implement transfer under these conditions as the additional water discharging to Nashua River would increase flooding. However, the resulting high UV254 levels may result in high chlorine demand and DBP formation (Bryan 2005; Edzwald *et al.* 1985; Garvey 2000; Sung 2003; Weishar *et al.* 2003).

It is difficult to evaluate if the magnitude of the algae bloom resulting from the increased runoff is appropriate as the model demonstrated during calibration that it could not adequately predict algal levels except during the spring diatom bloom. An improved phytoplankton calibration would improve the validity of the autochthonous organic matter prediction resulting from this scenario. However, both the TOC and UV254 predictions suggest that significant impairment in water quality would occur less than three weeks after the beginning of a series of storms.

6.4 Quabbin Bypass

A scenario was run in which water was withdrawn from both reservoirs to meet demand as though an aqueduct was constructed for water from Quabbin to bypass Wachusett. The Quabbin Bypass scenario was run with constituent, temperature, meteorological, and tributary input, and water system demand (assumed to be equal to measured Cosgrove discharges for the period) data from the 2000 through 2002 models. Quabbin discharge, Cosgrove discharge, and spilling to Nashua River were calculated with a water balance in Microsoft EXCEL.

Demand was met by defining a quantity of water to be withdrawn from Quabbin, and setting the difference between Quabbin withdrawal and demand (Quabbin plus Cosgrove) to be equal to Wachusett discharge to the Cosgrove aqueduct. To maintain Wachusett water surface elevation at reasonable levels, it was necessary to vary Quabbin discharge quarterly. Wachusett aqueduct and Cosgrove aqueduct withdrawals were set to 0.088 m^3/sec (2.0 MGD), town withdrawals were maintained at reported values, North Dike seepage was maintained at 0.039 m^3/sec (0.9 MGD) and evaporation was maintained as calibrated. When predicted WSE exceeded the spill elevation (119.5 m), the water was discharged to Nashua River as if it were spilled. Spill quantity was estimated using a water balance external to CE QUAL W2. A Nashua River base flow discharge of 0.088 m^3/sec was included in addition to estimated spilling. WSE was maintained above 117.5

m, as lower levels have not been observed, and the validity of model geometry below this level is not known. The resulting discharges and WSE as determined by these calculations are presented in Figure 6.11.

During 2000, large quantities of runoff made low Quabbin discharges in the spring and early summer possible. During the second quarter of 2000, demand was met with Wachusett reservoir water exclusively. During 2001 and 2002, larger Quabbin discharges were implemented; at the end of the forth quarter in 2001, although all demand was met with water from Quabbin, Wachusett WSE rose only slightly. The differing hydrology of the three years resulted in 46.0%, 34.2%, and 24.2% of demand being met by Wachusett discharges in 2000, 2001, and 2002, respectively.

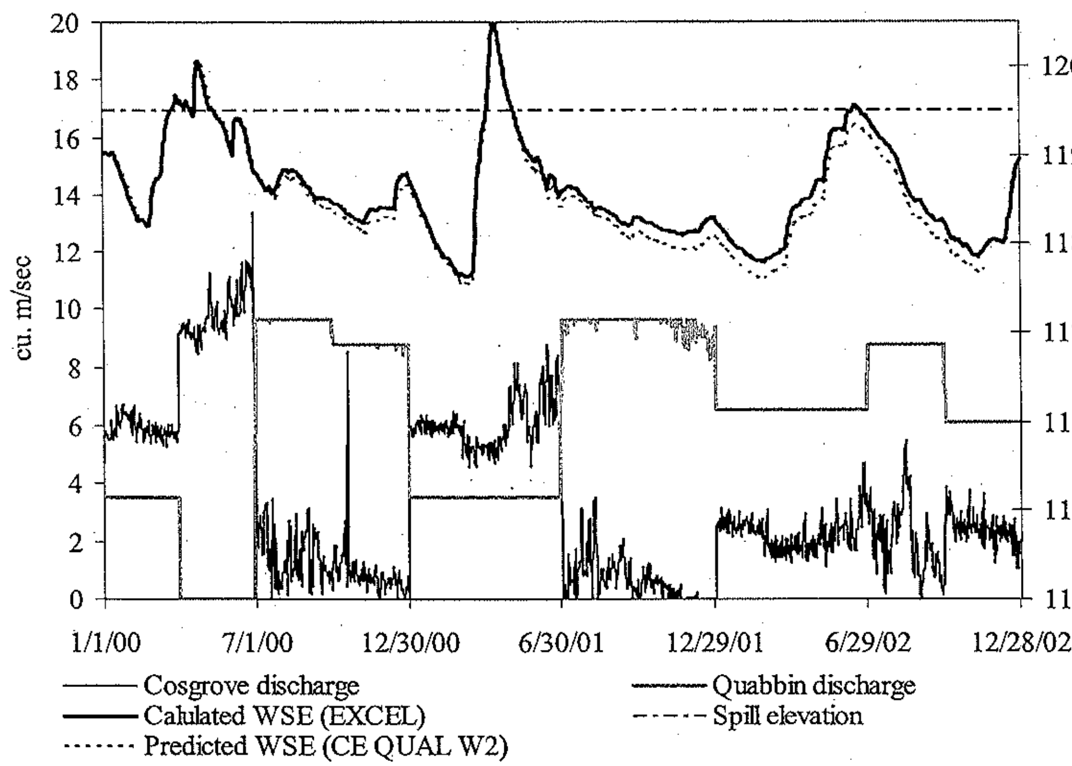

Figure 6.11 Wachusett Reservoir water surface elevation as calculated by Microsoft Excel and as predicted by CE QUAL W2, along with quantity of water discharged to Quabbin and Cosgrove aqueducts to meet demand in the Quabbin Bypass scenario.

It is notable that the total quantity of water discharged from Quabbin to meet consumer demand (with destinations of Wachusett Reservoir through transfer, and Boston in the bypass scenario) is slightly smaller in all three years of the Quabbin Bypass scenario than as implemented in the calibration model. This difference is likely due to smaller quantities of water being spilled in the bypass scenario than actually occurred. Total annual Quabbin discharge for both cases is shown in

Table 6.1

Table 6.1 Discharge from Quabbin to meet demand, m^3/yr

Scenario	2000	2001	2002
As Calibrated	1.90E+08	2.22E+08	2.27E+08
Quabbin Bypass	1.74E+08	2.06E+08	2.21E+08

Figure 6.12 presents TOC levels at Cosgrove that result from this scenario as predicted by CE QUAL W2 and TOC levels resulting from mixing that water with Quabbin Transfer water to meet demand, as well as Quabbin TOC levels, Wachusett TOC levels as predicted at Cosgrove through calibration, and the percentage of Wachusett water included in the final mixture. Note that Quabbin Transfer TOC for 2000 is assumed to be the average of TOC 2001 through 2002 data, and that the data is from the Chicopee Valley Aqueduct (no Quabbin Transfer data is available).

The resulting predicted average Cosgrove TOC level with no transfer is 2.8 mg/L, the average Quabbin Input TOC level was 1.8 mg/L, and the predicted TOC level of the mixture is 2.2 mg/L. It is notable that the mixed prediction represents average TOC levels for the three year period following the implementation of a Quabbin water bypass of Wachusett reservoir; the initial condition used is a measured value and thereby represents the mixture of waters from both reservoirs. Average measured TOC at Cosgrove during 2000 – 2002 was 2.4 mg/L This scenario suggests that bypassing Wachusett Reservoir with Quabbin Transfer could lead to reduced combined TOC levels (0.2 mg/L less, on average for 2000-2002).

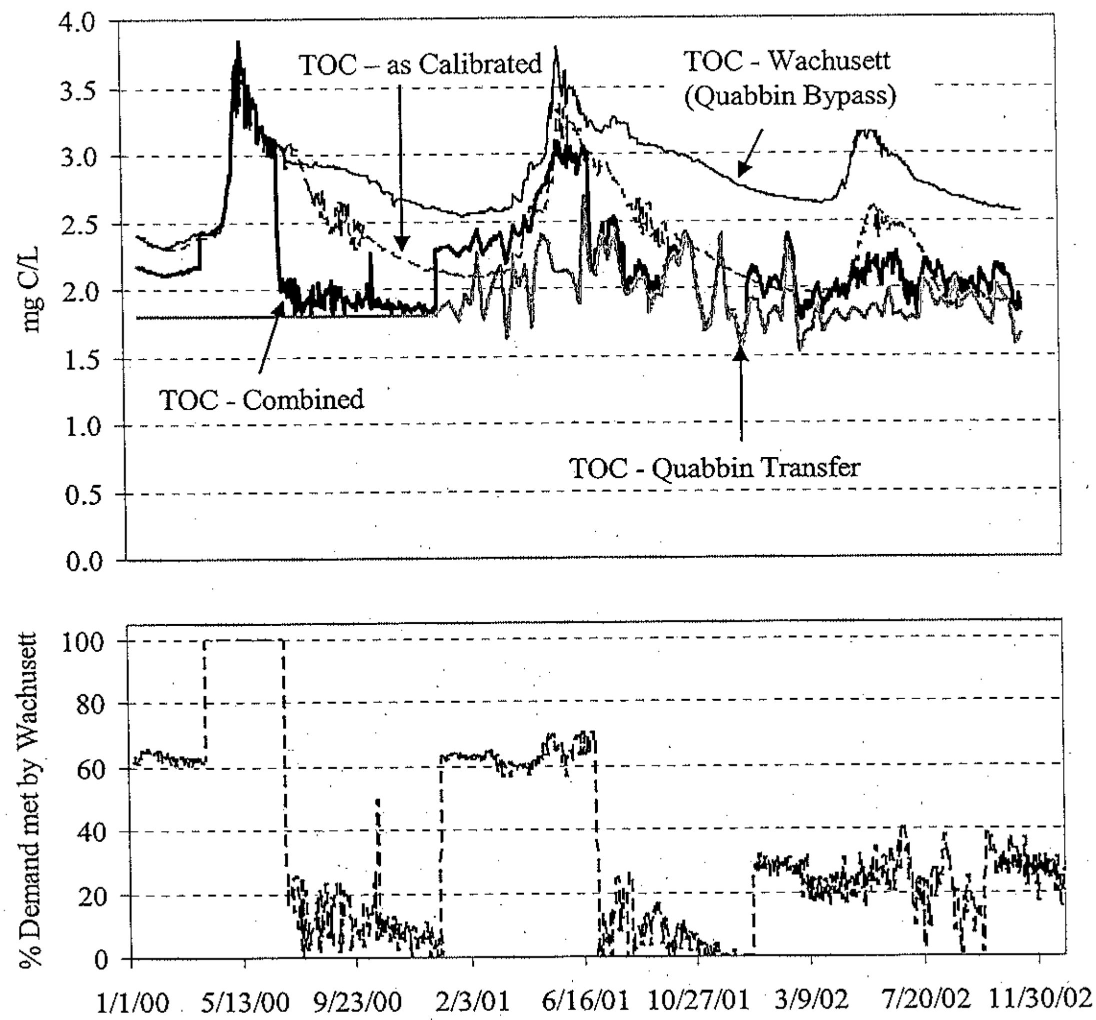

Figure 6.12 TOC predictions for Wachusett Reservoir with no Quabbin Transfer, Quabbin Transfer, the mixing of the two waters, and the percentage of water originating in Wachusett.

Figure 6.13 presents UV254 results for this scenario. Shown are predicted Wachusett UV254 levels, the Quabbin Transfer UV254 levels used (2001 and 2002 values measured at CVA, 2000 values the average of 2001 - 2002 values), and UV254 levels at Cosgrove as predicted with transfer. Also shown is the UV254 prediction resulting from mixing the Quabbin and Wachusett water to meet demand, and the percentage of that water originating in Wachusett.

Average CVA UV254 levels were 0.022 cm^{-1} during the three year period, while average predicted Wachusett levels at Cosgrove were 0.066 cm^{-1}. Mixing the two waters to meet demand resulted in average UV254 of 0.035 cm^{-1}, while measured UV254 levels at Cosgrove averaged 0.047 cm^{-1}. These results suggest that bypassing Wachusett Reservoir with Quabbin Transfer may result in a 25% combined reduction of UV254.

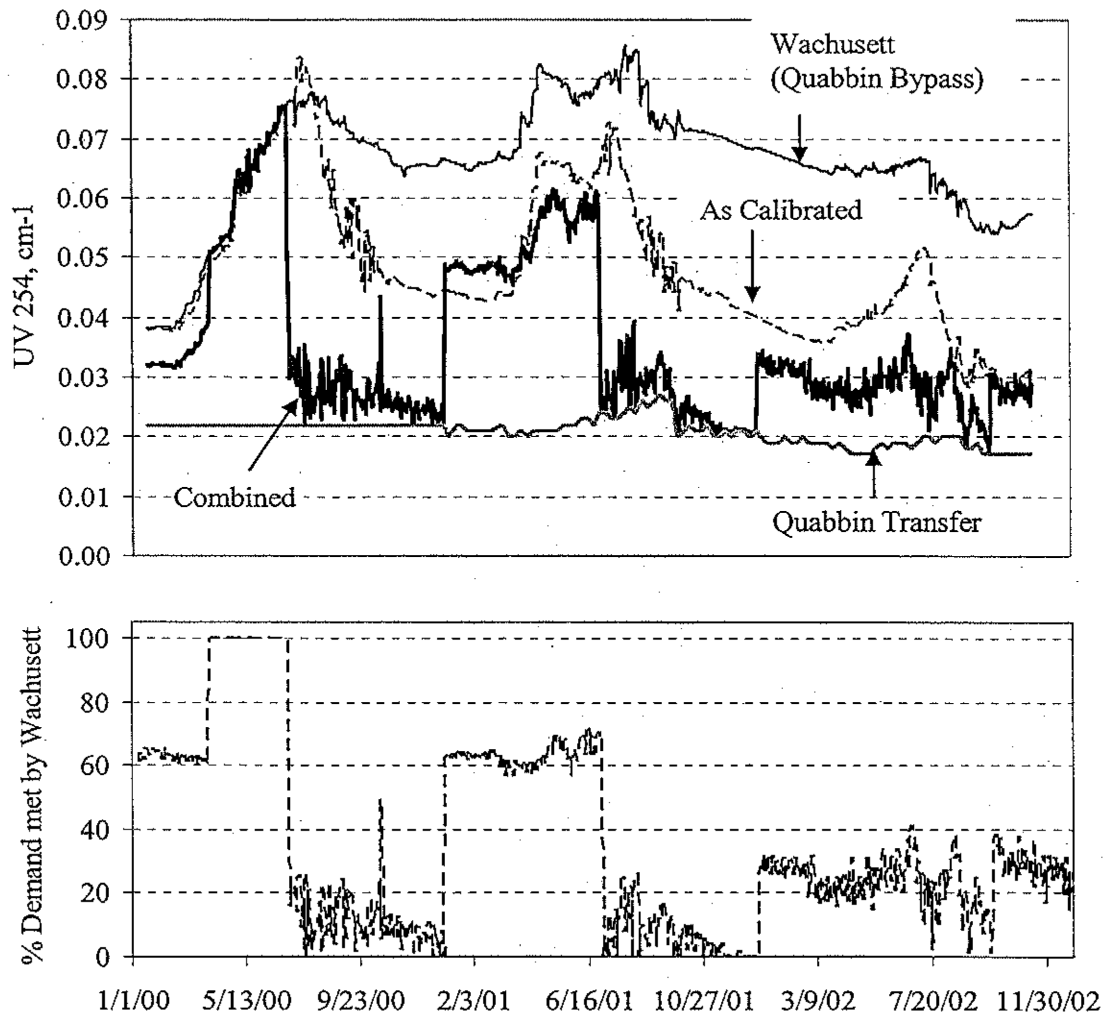

Figure 6.13 UV254 predictions for Wachusett Reservoir without Quabbin Transfer, Quabbin Transfer, the mixing of the two waters, and the percentage of water originating in Wachusett.

The reduced mixed TOC and UV254 levels result from increased residence time within Wachusett Reservoir. Residence times resulting from the bypass scenario were 1.3, 1.4, and 2.2 years in 2000, 2001, and 2002, respectively, as compared to the 0.61, 0.60, and 0.68 years calculated for actual conditions. Increased residence time results in longer exposure to sunlight and contact with microorganisms, allowing increased decay.

Decay was quantified by running the model for a conservative UV254 case. Light induced and temperature dependent decay rates were set to zero and the model run for cases with Quabbin transfer (as calibrated) and without transfer (this scenario). The difference between the conservative UV254 results at Cosgrove and the results with decay were divided by the conservative UV254 prediction for each case. A time series plot of the predicted decay percentage is shown in Figure 6.14.

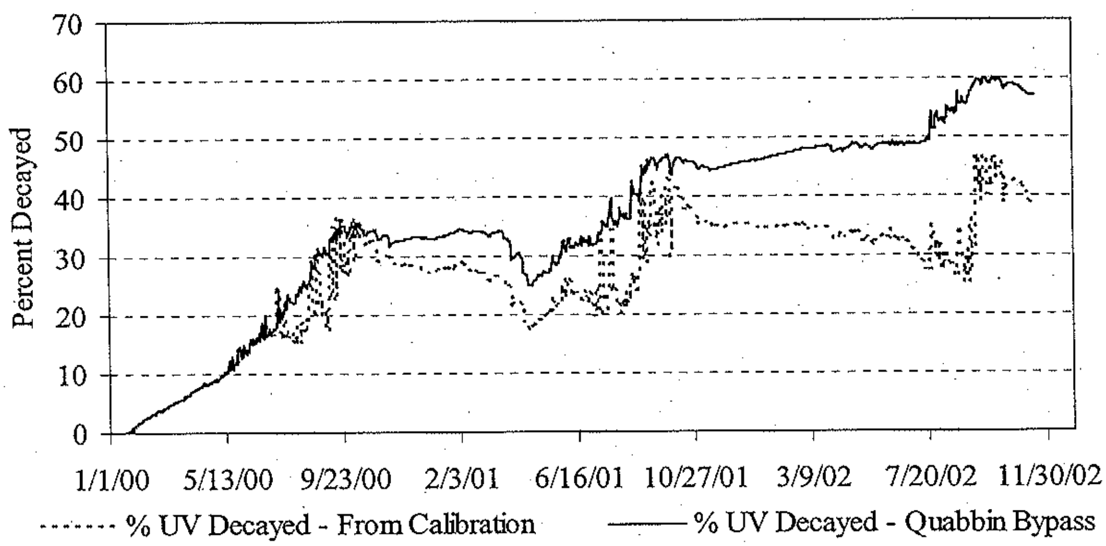

Figure 6.14 Percent of UV254 decayed in water withdrawn by Cosgrove Intake.

The UV254 initial value for the reservoir in both cases was set equal to the measured UV254 value at Cosgrove, so results for that year should be ignored. During 2001 and 2002, 28% and 34% of UV254 was decayed with Quabbin Transfer, respectively. With no transfer during those years, 37 and 51% of UV254 was decayed. The result of this increased decay is the improvement in UV254 levels predicted for Wachusett water combined with water from a Quabbin bypass. The decreased TOC levels, although not as dramatic, are a result of the same phenomenon.

7. SUMMARY, CONCLUSIONS, AND RECOMMENDATIONS

7.1 Summary

Wachusett Reservoir is an oligotrophic drinking water reservoir with TOC concentrations between 1.5 and 4.0 mg/L and UV254 levels between 0.030 and 0.085 cm^{-1}. Variation in NOM levels within these ranges is seasonal and dependent on the dominant inflow. In most years, more than half of the annual water budget of the reservoir is received from Quabbin Reservoir through inflow from the Quabbin Aqueduct. This transferred water is characterized by lower NOM levels than water received from the watershed of the Wachusett Reservoir. As a result, periods of high tributary runoff result in higher water quality constituent levels at Cosgrove Aqueduct, the main withdrawal. These periods typically occur in the spring and early summer. When tributary runoff is not adequate to meet demand by consumers and transfer from Quabbin Aqueduct is occurring, water received from the Wachusett Tributaries is diluted and water quality at Cosgrove Aqueduct improves.

CE QUAL W2 a two dimensional, laterally averaged water quantity and quality modeling program, was used to model NOM constituents in Wachusett Reservoir. The interaction between tributary inflows and Quabbin Transfer was of particular interest. Version 2 of this program, implemented in this research, can model 21 water quality constituents. Eight of these constituents were invoked in this study. Constituents modeled with the program include UV254, labile DOM, refractory DOM, algae, detritus, nitrate-nitrogen, ammonia-nitrogen, and soluble reactive phosphorus. From these constituents, TOC and SUVA were calculated.

The period of study was January 2000 through December 2002. The model was calibrated with data from 2001 and 2002, and validated with data from 2000. Meteorological, in-reservoir, and inflow and outflow quantity, temperature, and quality data was required. This data was available from NOAA, DCR, MWRA, USGS and NADP.

Water quantity and hydrodynamics were calibrated by comparing measured and predicted reservoir WSE as well as measured and predicted in-reservoir temperature and specific conductivity profiles for stations downstream of Thomas Basin. Model parameters impacting hydrodynamics were set to values resulting from Wachusett Reservoir studies by CDM (1995) and Joaquin (2001), Quabbin Reservoir studies by Garvey (2000) and Roberts (2003), and through calculation from DCR data.

Water quality constituents were calibrated by comparing measured and predicted concentrations leaving Cosgrove aqueduct. Parameter values impacting these constituents were based on results from CDM (1995) Garvey (2000) and Roberts (2003) and then adjusted to improve fit. The obtained values were compared to literature values. The model was then validated by applying all calibrated parameter values and data preparation methods to a separate data set. Results demonstrated the ability of the model to predict conditions without further adjustment.

The model was then used to predict the impact of varied controllable and uncontrollable hydrologic conditions on water quality at Cosgrove. Simulations included increasing Quabbin transfer during relatively wet periods, including several additional storms to simulate an unusually wet late summer and fall, and ceasing transfer of Quabbin Reservoir water to Wachusett Reservoir.

7.2 Conclusions

7.2.1 Data Availability

Water quantity data available for the study period was adequate to assemble a mass balance and calibrate hydrodynamics. However, data was sometimes difficult to collect, and sometimes inaccurate. In-reservoir temperature and conductivity profile data was adequate to validate hydrodynamics. CE QUAL W2 is able to successfully predict temperature and conductivity profiles under the Route 12 bridge. Profiles measured and samples taken upstream in Thomas Basin are likely not representative of temperature or constituent levels entering the reservoir. Water quality data was adequate to predict fate and transport of NOM based on tributary inflow and Cosgrove Withdrawal concentrations. Modeling accuracy was decreased for the 2002 calendar year since minor tributary constituent data was available on a biannual basis only. More periodic data collection would improve modeling accuracy. No in-reservoir TOC data was available for the study period, making comparison of gradients infeasible.

7.2.2 TOC Conclusions

TOC levels at Cosgrove Aqueduct varied seasonally within a range of approximately 1.5 to 3.0 mg/L during the study period. Measured concentrations at Cosgrove are generally lower than concentrations input by the tributaries. However, Cosgrove Aqueduct concentrations are generally higher than concentrations received from Quabbin Aqueduct, the largest inflow. A volume weighted material balance for the study period predicts larger average withdrawal TOC concentrations than were measured, indicating that loss of NOM is occurring.

TOC was modeled in CE QUAL W2 as the sum of labile DOM, refractory DOM, algae, and detritus. Using decay coefficients, settling rates, and stoichiometric parameter values from previous studies for Wachusett Reservoir and Quabbin Reservoir did not yield satisfactory results. Three alternative organic matter calibration results are appropriate based on existing data. The result selected for validation and scenario prediction includes the following first order, temperature dependent decay rates that differ from Roberts (2003):

- a labile DOM decay rate of 0.008 day^{-1},
- a refractory DOM decay rate of 0.0008 day^{-1},
- a labile DOM to refractory DOM decay rate of 0.0008 day^{-1}.

These labile, refractory, and labile to refractory decay rates are 2.7 times larger than the values determined by Roberts (2003) for Quabbin Reservoir. This difference most likely stems from the assumption that the refractory DOM decay rate represents decay for all refractory DOM in the system. In fact, refractory DOM represents a variety of heterogeneous compounds. In a system with long residence times, refractory DOM that decays more rapidly is consumed, and extremely refractory DOM remains; therefore, the resulting decay rate that must describe decay of both compounds is biased by the extremely refractory compounds

A second calibration alternative assumed a smaller labile DOM percentage for Quabbin Transfer DOM (5%, compared to 20% in other alternatives). Temperature dependent decay rates were increased to adequately predict DOM. This assumption is inconsistent with results from Garvey (2000) and was not selected for validation.

The temperature dependent nature of the decay rates are meant to simulate microbial degradation of these constituents. However, numerous studies show that photolysis of biologically recalcitrant DOM increases its bioavailability and is a major pathway in its degradation. Therefore, in a third calibration alternative, the temperature dependent refractory DOM decay pathway was replaced with a light induced decay pathway. To simplify calibration by preventing a feedback loop, the labile to refractory decay rate was set to zero. The resulting value for α_{om}, a parameter relating the effect of irradiance on refractory DOM decay, is 1.3 E-5 cal/cm^2. It was also necessary to increase the labile DOM decay rate to 0.0012 day^{-1}. This alternative is consistent with processes known to occur but was not selected due to difficulty of calibration and lack of data for calibration.

Phytoplankton were first modeled with parameter values determined by Roberts (2003) for Quabbin. The resulting model overpredicted and underpredicted 2001 and 2002 spring diatom blooms, respectively. Parameter adjustment resulted in values of 1.9 day^{-1} for the maximum algal growth rate and 0.1 day^{-1} for maximum algal respiration rate. All other algal parameter values were used as in Roberts (2003). The resulting calibration predicts spring diatom blooms to within 0.1 mg C/L, although other blooms, as well as algal levels throughout the year are not well predicted. These inaccuracies likely result from modeling limitations unique to Version 2.

Accurate algal prediction depends on accurate nutrient prediction. The majority of input and withdrawal phosphorus data is total phosphorus data; little orthophosphate data exists. Assuming that 50% of measured input total phosphorus was bioavailable was necessary for accurate algae prediction.

Detritus comprises a small portion of reservoir TOC; therefore, the model was relatively insensitive to detritus decay and settling rates. Values implemented by Roberts (2003) were selected.

The resulting DOM, algae, nutrient, and POC calibration accurately predicted TOC for 2001 through 2002. The only major deviation occurred in late summer 2002, when wind mixing was underpredicted and outflowing water is too heavily dominated by Quabbin Transfer water.

7.2.3 UV254 Conclusions

UV254 levels varied by approximately 0.05 cm^{-1} during the study period, from 0.03 to 0.08 cm^{-1}. Measured withdrawal UV254 levels were generally lower than tributary UV254 levels and higher than Quabbin Transfer UV254 levels. Running a conservative UV254 scenario in CE QUAL W2 predicted higher levels than were measured, indicating that decay of UV254 also occurred.

UV254 is a surrogate measure for NOM that indicates presence of humic materials. Humic materials are biologically recalcitrant, so the first order temperature dependent UV254 decay rate was based on the refractory DOM decay rate, 0.0008 day^{-1}. Subsequent calibration indicated that 2.6E-5 cal/cm^2 was appropriate for alpha, the parameter relating irradiance to decay of UV254. The selected temperature dependent decay rate is 0.0005 day^{-1} (2.7 times) larger than the value selected by Roberts (2003) while ALPHA is one order of magnitude larger than ALPHA as selected in that study. This again may be due to detention time differences between the reservoirs.

7.2.4 Constituent Validation Conclusions

Parameter values and assumptions resulting from calibration were implemented successfully to model constituents in 2000. TOC was predicted with reasonable accuracy, although peak TOC was predicted ~1 month early and 0.5 mg/L too high. Nutrient calibrations fluctuated significantly, but year end concentrations were reasonable. UV254 prediction was in close agreement with data, despite the lack of Quabbin Transfer constituent data.

7.2.5 Simulation Conclusions

7.2.5.1 Increased Quabbin Transfer during Dry Spring

Increasing Quabbin Transfer in Spring 2002 to 8.7 m^3/s resulted in small decreases in water quality constituent levels. An additional 14.4 billion gallons of water was transferred and consequently wasted to Nashua River. This additional discharge of relatively low UV254, low TOC water resulted in reducing peak spring TOC levels by

0.2 mg/L and UV254 levels by 0.008 cm^{-1} (compared to levels as predicted without additional transfer). TOC and UV254 levels are affected minimally once additional transfer ceases. Decrease in levels of water quality constituents at Cosgrove begins to occur after three weeks. The improvement in water quality may not be worth the water lost to Nashua River, unless

- Quabbin Reservoir is near full or spilling, or
- Distribution system DBP levels are approaching or exceeding regulatory levels (based on Sung 2003).

7.2.5.2 Increased Quabbin Transfer during Wet and Dry Springs

Quabbin Transfer was increased for two consecutive Springs (2001 and 2002). Additional transfer consisted of 13.5 billion gallons during 68 days in 2001 and 14.4 billion gallons for 79 days in 2002. The additional transfer of relatively low UV254, low TOC water resulted in TOC levels of consistently ~0.05 mg/L lower than as predicted with no additional transfer. During peak TOC in 2002, levels were briefly 0.2 mg/L lower than as predicted by calibration. The additional transfer decreased UV254 levels by 0.005 cm^{-1} during peak 2001 levels and by 0.008 cm^{-1} during peak 2002 levels. Decreases in constituent levels begin to occur three weeks after transfer begins. Constituent levels are minimally decreased when additional transfer ceases. As in the additional scenario of additional 2002 transfer only, the improvement in water quality may not be worth the water lost to Nashua River, unless

- Quabbin Reservoir is near full or spilling, or
- Distribution system DBP levels are approaching or exceeding regulatory levels (based on Sung 2003).

7.2.5.3 Additional 2001 Runoff Period

Modeling an unseasonably wet late summer/fall in 2001 as might occur for a hurricane event resulted in significantly increased levels of UV254 and TOC at Cosgrove Aqueduct. The combination of large tributary discharges containing high levels of TOC and UV254 and ceased transfer from Quabbin led to these water quality impacts. The magnitude of increase is not clearly known, as increased runoff would likely increase or decrease water quality constituent levels of the tributaries. Significant increases in water quality constituent levels at Cosgrove begin approximately four weeks after the beginning of the increased runoff and precipitation period.

7.2.5.4 Quabbin Bypass

Constructing an aqueduct for water from Quabbin Reservoir to bypass Wachusett Reservoir would result in a mean hydraulic residence time increase of ~1 year (based on 2000-2002 results) for Wachusett Reservoir. Water within Wachusett would no longer benefit from dilution with Quabbin water; thus, an increase in TOC and UV254 levels is predicted to occur at Cosgrove. However, the increased residence time within Wachusett results in increased degradation of UV254 and TOC in this scenario. When the resulting water withdrawn at Cosgrove is mixed with bypassed water from Quabbin Reservoir (in similar ratios to those that occurred) constituent levels are generally lower than measured levels. UV254 levels decrease more significantly than TOC levels (~25% compared to ~8%, respectively).

7.3 Recommendations

The following recommendations result from conclusions drawn in this study.

7.3.1 Recommendations for DCR/MWRA- Sampling

- Improve coordination of hydrodynamic data collection, specifically regarding Quabbin Transfer and Wachusett Aqueduct.
- Increase collection frequency of water quality constituent data collection for minor tributaries (perhaps to 4 times annually).
- Collect water quality constituent data for Waushacum Brook, as it is the third largest tributary of the reservoir.
- Collect Quabbin Transfer constituent data periodically (monthly or quarterly).
- Sample Thomas Basin, North Basin, and South Basin stations for TOC and DOC when sampling for UV254 and nutrients.
- Record temperature and conductivity profiles and collect nutrient samples under Rt. 12 bridge instead of at the currently used Thomas Basin station.

- Include DOC measurement when TOC is measured at Cosgrove.
- Include storm event sampling data in periodic tributary sampling data.
- Establish database for hydrodynamic and water quality data

7.3.2 Recommendations for DCR/MWRA – Quabbin Transfer

- Transferring water from Quabbin that results in spilling of Wachusett may result in small decreases in TOC and UV254 levels. This transfer is appropriate when Quabbin contains close to full capacity or when distribution system DBPs are nearing or exceeding concern levels.

7.3.3 Recommendations for DCR/MWRA – Capital Improvements

- Constructing an aqueduct to bypass Wachusett Reservoir with Quabbin Reservoir water merits additional study. Wachusett Reservoir water quality at Cosgrove would decrease, but the mixture of Quabbin Reservoir and Wachusett Reservoir water would contain decreased constituent levels. Cost of construction is likely high and would need to be compared to treatment plant construction costs.

7.3.4 Possibilities for Future Research

This section suggests future research that may be used to improve the understanding of NOM in Wachusett Reservoir. Suggestions are generally arranged from most to least strongly suggested. Several of these suggestions are similar to work conducted in Quabbin Reservoir by Garvey (2000), and several others may be currently underway.

- Implement CE QUAL W2 Version 3 for multispecies phytoplankton prediction, or another phytoplankton prediction method (i.e. artificial neural network).
- Implement other CE QUAL W2 constituents including inorganic carbon/pH.
- Implement overland flow model to predict discharge for ungaged tributaries and direct runoff.

- Collect comprehensive in-reservoir NOM data to establish concentration gradients with and without Quabbin Transfer occurring, and before and after significant storm events.
- Establish impact of storm events on inflow water quality
- Conduct laboratory experiments on Wachusett Reservoir water to:
  - Establish temperature dependent decay rates for DOM and UV254
  - Establish light induced decay rate for DOM and UV254
  - Establish the impact of light decay on organic matter bioavailability
  - Examine autochthonous generation of UV254
- Conduct in-reservoir light bottle/dark bottle experiments for determining algal growth and respiration rates
- Conduct sediment trap experiments to estimate algal and detritus settling rates.

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Appendix A – CE QUAL W2 Control File (W2_CON.NPT)

```
00-02.ascalibrated
Synthesis of 2000 through 2002 model years
TITLE C ..............................TITLE...................................
        Wachusett Reservoir (Jday 18-1051)
        All in- and outflows as calibrated
        Constituent parameters as calibrated
        ctr_tr1.npt = cin_br1.npt, cdt from Quabbin Purgee Brook
        OMLTDK.EXE code -
        UV254 and light decay of RDOM with revised constituent designations

TIME CON  TMSTRT   TMEND    YEAR
         18.0000 1051.00    2000

DLT CON      NDT  DLTMIN
               1 1.00000

DLT DATE    DLTD    DLTD    DLTD    DLTD    DLTD    DLTD    DLTD    DLTD    DLTD
         0.00000

DLT MAX   DLTMAX  DLTMAX  DLTMAX  DLTMAX  DLTMAX  DLTMAX  DLTMAX  DLTMAX  DLTMAX
         1440.00

DLT FRN     DLTF    DLTF    DLTF    DLTF    DLTF    DLTF    DLTF    DLTF    DLTF
         0.90000

BRANCH G      US      DS     UHS     DHS      NL
Br 1           2      46       0       0       2
Br 2          49      51       0      27       2
Br 3          54      55       0      40       2
Br 4          58      59       0      41       2
Br 5          62      63       0      43       2

LOCATION     LAT    LONG    EBOT
         42.3800 71.7400 84.7000

INIT CND     T2I    ICEI  WTYPEC
         2.00000 0.00000   FRESH

CALCULAT     VBC     EBC     MBC   PQINC     EVC     PRC
              ON      ON      ON      ON      ON      ON

INTERPOL   QINIC    TRIC    DTIC    HDIC  QOUTIC    WDIC   METIC
              ON      ON      ON      ON      ON      ON      ON

DEAD SEA   WINDC    QINC   QOUTC   HEATC
              ON      ON      ON      ON

ICE COVER   ICEC  SLICEC   SLHTC  ALBEDO   HWICE    BICE    GICE  ICEMIN   ICET2
             OFF  DETAIL      ET 0.25000 10.0000 0.60000 0.07000 0.05000 3.00000

TRANSPORT  SLTRC   THETA
        QUICKEST 0.00000

WSC NUMB    NWSC   WINDH
               1 10.0000

WSC DATE    WSCD    WSCD    WSCD    WSCD    WSCD    WSCD    WSCD    WSCD    WSCD
         0.00000
```

```
WSC COEF       WSC      WSC      WSC      WSC      WSC      WSC      WSC      WSC      WSC
           0.65000

HYD COEF        AX       DX    CHEZY     CBHE     TSED
           1.00000  1.00000  70.0000  7.0E-07  10.0000

SEL WITH       SWC      SWC      SWC      SWC      SWC      SWC      SWC      SWC      SWC
                ON      OFF      OFF      OFF      OFF

N STRUC       NSTR     NSTR     NSTR     NSTR     NSTR     NSTR     NSTR     NSTR     NSTR
                 2        0        0        0        0

K BOTTOM      KBSW     KBSW     KBSW     KBSW     KBSW     KBSW     KBSW     KBSW     KBSW
Br 1            39       39
Br 2
Br 3
Br 4
Br 5

SINK TYPE    SINKC    SINKC    SINKC    SINKC    SINKC    SINKC    SINKC    SINKC    SINKC
Br 1          LINE     LINE
Br 2
Br 3
Br 4
Br 5

E STRUC       ESTR     ESTR     ESTR     ESTR     ESTR     ESTR     ESTR     ESTR     ESTR
Br 1       104.300  104.300
Br 2
Br 3
Br 4
Br 5

W STRUC       WSTR     WSTR     WSTR     WSTR     WSTR     WSTR     WSTR     WSTR     WSTR
Br 1       24.0000  24.0000
Br 2
Br 3
Br 4
Br 5

N OUTLET      NOUT     NOUT     NOUT     NOUT     NOUT     NOUT     NOUT     NOUT     NOUT
                 0        0        0        0        0

O LAYER       KOUT     KOUT     KOUT     KOUT     KOUT     KOUT     KOUT     KOUT     KOUT
Br 1
Br 2
Br 3
Br 4
Br 5

N WDRWAL       NWD
                 3

W SEGMNT       IWD      IWD      IWD      IWD      IWD      IWD      IWD      IWD      IWD
                44       44       44

W LAYER        KWD      KWD      KWD      KWD      KWD      KWD      KWD      KWD      KWD
                11        5       36

N TRIBS        NTR
                 9

TRIB PLACE   PQTRC    PQTRC    PQTRC    PQTRC    PQTRC    PQTRC    PQTRC    PQTRC    PQTRC
```

```
            DENSITY DENSITY DENSITY DENSITY DENSITY DENSITY DENSITY DENSITY DENSITY

TRIB SEG        ITR     ITR     ITR     ITR     ITR     ITR     ITR     ITR     ITR
                  3       8       9      10      20      33      49      17      50

TRIB TOP       ETRT    ETRT    ETRT    ETRT    ETRT    ETRT    ETRT    ETRT    ETRT
                  0       0       0       0       0       0       0       0       0

TRIB BOT       ETRB    ETRB    ETRB    ETRB    ETRB    ETRB    ETRB    ETRB    ETRB
                  0       0       0       0       0       0       0       0       0

DST TRIB       DTRC    DTRC    DTRC    DTRC    DTRC    DTRC    DTRC    DTRC    DTRC
                 ON     OFF     OFF     OFF     OFF

SCR PRINT      SCRC    NSCR
                 ON       1

SCR DATE       SCRD    SCRD    SCRD    SCRD    SCRD    SCRD    SCRD    SCRD    SCRD
                  1

SCR FREQ       SCRF    SCRF    SCRF    SCRF    SCRF    SCRF    SCRF    SCRF    SCRF
            1.00000

SNAPSHOT       LJPC    UPRC    WPRC    TPRC  DLTPRC
                 IV     OFF     OFF      ON     OFF

SNP PRINT      SNPC    NSNP   NISNP
                OFF       1      45

SNP DATE       SNPD    SNPD    SNPD    SNPD    SNPD    SNPD    SNPD    SNPD    SNPD
            225.000

SNP FREQ       SNPF    SNPF    SNPF    SNPF    SNPF    SNPF    SNPF    SNPF    SNPF
            999.000

SNP SEG        ISNP    ISNP    ISNP    ISNP    ISNP    ISNP    ISNP    ISNP    ISNP
                  2       3       4       5       6       7       8       9      10
                 11      12      13      14      15      16      17      18      19
                 20      21      22      23      24      25      26      27      28
                 29      30      31      32      33      34      35      36      37
                 38      39      40      41      42      43      44      45      46

PRF PLOT       PRFC    NPRF   NIPRF
                 ON      21       6

PRF DATE       PRFD    PRFD    PRFD    PRFD    PRFD    PRFD    PRFD    PRFD    PRFD
            116.000 135.000 149.000 165.000 177.000 205.000 234.000 266.000 279.000
            298.000 299.000 300.000 301.000 302.000 303.000 304.000 305.000 306.000
            307.000 318.000 352.000

PRF FREQ       PRFF    PRFF    PRFF    PRFF    PRFF    PRFF    PRFF    PRFF    PRFF
            100.000 100.000 100.000 100.000 100.000 100.000 100.000 100.000 100.000
            100.000 100.000 100.000 100.000 100.000 100.000 100.000 100.000 100.000
            100.000 100.000 100.000

PRF SEG        IPRF    IPRF    IPRF    IPRF    IPRF    IPRF    IPRF    IPRF    IPRF
                 11      17      32      42      44      46

SPR PLOT       SPRC    NSPR   NISPR
                 ON      21       6

SPR DATE       SPRD    SPRD    SPRD    SPRD    SPRD    SPRD    SPRD    SPRD    SPRD
            116.000 135.000 149.000 165.000 177.000 205.000 234.000 266.000 279.000
```

```
         298.000 299.000 300.000 301.000 302.000 303.000 304.000 305.000 306.000
         307.000 318.000 352.000

SPR FREQ    SPRF    SPRF    SPRF    SPRF    SPRF    SPRF    SPRF    SPRF    SPRF
         100.000 100.000 100.000 100.000 100.000 100.000 100.000 100.000 100.000
         100.000 100.000 100.000 100.000 100.000 100.000 100.000 100.000 100.000
         100.000 100.000 100.000

SPR SEG     ISPR    ISPR    ISPR    ISPR    ISPR    ISPR    ISPR    ISPR    ISPR
              11      17      32      42      44      46

TSR PLOT    TSRC    NTSR
              ON       1

TSR DATE    TSRD    TSRD    TSRD    TSRD    TSRD    TSRD    TSRD    TSRD    TSRD
         1.00000

TSR FREQ    TSRF    TSRF    TSRF    TSRF    TSRF    TSRF    TSRF    TSRF    TSRF
         1.00000

VPL PLOT    VPLC    NVPL
             OFF       1

VPL DATE    VPLD    VPLD    VPLD    VPLD    VPLD    VPLD    VPLD    VPLD    VPLD
         1.00000

VPL FREQ    VPLF    VPLF    VPLF    VPLF    VPLF    VPLF    VPLF    VPLF    VPLF
         1.00000

CPL PLOT    CPLC    NCPL
             OFF       1

CPL DATE    CPLD    CPLD    CPLD    CPLD    CPLD    CPLD    CPLD    CPLD    CPLD
         1.00000

CPL FREQ    CPLF    CPLF    CPLF    CPLF    CPLF    CPLF    CPLF    CPLF    CPLF
         1.00000

RESTART     RSOC    NRSO    RSIC
             OFF       1     OFF

RSO DATE    RSOD    RSOD    RSOD    RSOD    RSOD    RSOD    RSOD    RSOD    RSOD
         364.000

RSO FREQ    RSOF    RSOF    RSOF    RSOF    RSOF    RSOF    RSOF    RSOF    RSOF
         100.000

CST COMP     CCC    LIMC     SDC     CUF
              ON     OFF     OFF       6

CST ACT      CAC     CAC     CAC     CAC     CAC     CAC     CAC     CAC     CAC
             OFF     OFF      ON     OFF      ON      ON      ON      ON      ON
              ON      ON      ON     OFF     OFF     OFF     OFF     OFF     OFF
             OFF     OFF     OFF

CST ICON     C2I     C2I     C2I     C2I     C2I     C2I     C2I     C2I     C2I
         0.00000 0.00000 0.03800 73.1500 0.22330 2.00900 0.05000 0.11750 0.00400
         0.00800 0.04000 10.0000 1.00000 0.00000 0.00000 0.00000 0.00000 0.00000
         0.00000 0.00000 0.00000

CST PRINT   CPRC    CPRC    CPRC    CPRC    CPRC    CPRC    CPRC    CPRC    CPRC
             OFF     OFF      ON     OFF      ON      ON      ON      ON      ON
              ON      ON      ON     OFF     OFF     OFF     OFF     OFF     OFF
```

```
              OFF      OFF      OFF

CIN CON     CINAC    CINAC    CINAC    CINAC    CINAC    CINAC    CINAC    CINAC    CINAC
              OFF      OFF       ON      OFF       ON       ON      OFF       ON       ON
               ON       ON       ON      OFF      OFF      OFF      OFF      OFF      OFF
              OFF      OFF      OFF

CTR CON     CTRAC    CTRAC    CTRAC    CTRAC    CTRAC    CTRAC    CTRAC    CTRAC    CTRAC
              OFF      OFF       ON      OFF       ON       ON      OFF       ON       ON
               ON       ON       ON      OFF      OFF      OFF      OFF      OFF      OFF
              OFF      OFF      OFF

CDT CON     CDTAC    CDTAC    CDTAC    CDTAC    CDTAC    CDTAC    CDTAC    CDTAC    CDTAC
              OFF      OFF       ON      OFF       ON       ON      OFF       ON       ON
               ON       ON       ON      OFF      OFF      OFF      OFF      OFF      OFF
              OFF      OFF      OFF

CPR CON     CPRAC    CPRAC    CPRAC    CPRAC    CPRAC    CPRAC    CPRAC    CPRAC    CPRAC
              OFF      OFF       ON      OFF       ON       ON      OFF      OFF       ON
               ON       ON       ON      OFF      OFF      OFF      OFF      OFF      OFF
              OFF      OFF      OFF

EX COEF     EXH2O     EXSS     EXOM     BETA
          0.29000  0.00100  0.00000  0.45000

COLIFORM   COLQ10    COLDK    ALPHA
          1.03000  0.00080   2.6E-5

S SOLIDS      SSS
          0.00000

ALGAE          AG       AM       AE       AR       AS     ASAT     APOM
          1.90000  0.03000  0.01200  0.10000  0.29000  50.0000  0.80000

ALG RATE      AT1      AT2      AT3      AT4      AK1      AK2      AK3      AK4
          0.00000  17.0000  22.0000  28.0000  0.10000  0.98000  0.98000  0.10000

DOM        LDOMDK     LRDK   RDOMDK    OMALP
          0.00800  0.00080  0.00080        0

POM        LPOMDK     POMS
          0.00700  0.35000

OM RATE      OMT1     OMT2     OMK1     OMK2
          0.00000  15.0000  0.10000  0.98000

SEDIMENT      SDK     FSOD
          0.01000  1.00000

S DEMAND      SOD      SOD      SOD      SOD      SOD      SOD      SOD      SOD      SOD
          0.19500  0.19500  0.19500  0.19500  0.19500  0.19500  0.19500  0.19500  0.19500
          0.19500  0.19500  0.19500  0.19500  0.19500  0.19500  0.19500  0.19500  0.19500
          0.19500  0.19500  0.19500  0.19500  0.19500  0.19500  0.19500  0.19500  0.19500
          0.19500  0.19500  0.19500  0.19500  0.19500  0.19500  0.19500  0.19500  0.19500
          0.19500  0.19500  0.19500  0.19500  0.19500  0.19500  0.19500  0.19500  0.19500
          0.19500  0.19500  0.19500  0.19500  0.19500  0.19500  0.19500  0.19500  0.19500
          0.19500  0.19500  0.19500  0.19500  0.19500  0.19500  0.19500  0.19500  0.00000
          0.00000

CBOD         KBOD     TBOD     RBOD
          0.00000  1.04700  1.85000

PHOSPHOR     PO4R    PARTP     AHSP
```

```
         0.00500 1.20000 0.01600

AMMONIUM    NH4R    NH4DK    AHSN
         0.01000 0.03000 0.06200

NH4 RATE   NH4T1   NH4T2   NH4K1   NH4K2
         0.00000 15.0000 0.10000 0.98000

NITRATE    NO3DK
         0.00000

NO3 RATE   NO3T1   NO3T2   NO3K1   NO3K2
         0.00000 15.0000 0.10000 0.98000

SED CO2     CO2R
         0.10000

IRON         FER     FES
         0.40000 0.00000

STOICHMT  O2NH4    O2OM    O2AR    O2AG    BIOP    BION    BIOC
         4.57000 1.40000 1.40000 1.40000 0.01100 0.08000 0.45000

O2 LIMIT  O2LIM
         0.00000

BTH FILE..............................BTHFN...................................
        coffer0102.npt

VPR FILE..............................VPRFN...................................
        vpr.npt

LPR FILE..............................LPRFN...................................
        lpr.npt - not used

RSI FILE..............................RSIFN...................................
        rso74.npt - not used

MET FILE..............................METFN...................................
        met.npt

QWD FILE..............................QWDFN...................................
        Qwd_new.npt

QIN FILE..............................QINFN...................................
Br 1    qin_br1.npt
Br 2    qin_br2.npt
Br 3    qin_br3.npt
Br 4    qin_br4.npt
Br 5    qin_br5.npt

TIN FILE..............................TINFN...................................
Br 1    tin_br1.npt
Br 2    tin_br2.npt
Br 3    tin_br3.npt
Br 4    tin_br4.npt
Br 5    tin_br5.npt

CIN FILE..............................CINFN...................................
Br 1    cin_br1.npt
Br 2    cin_br2.npt
Br 3    cin_br3.npt
Br 4    cin_br4.npt
```

```
Br 5    cin_br5.npt

QOT FILE..............................QOTFN..............................
Br 1    cosgrove_out.npt
Br 2    qot_br2.npt - not used
Br 3    qot_br3.npt - not used
Br 4    qot_br4.npt - not used
Br 5    qot_br5.npt - not used

QTR FILE..............................QTRFN..............................
Tr 1    qtr_tr1.npt
Tr 2    qtr_tr2.npt
Tr 3    qtr_tr3.npt
Tr 4    qtr_tr4.npt
Tr 5    qtr_tr5.npt
Tr 6    qtr_tr6.npt
Tr 7    qtr_tr7.npt
Tr 8    qtr_tr8.npt
Tr 9    qtr_tr9.npt

TTR FILE..............................TTRFN..............................
Tr 1    ttr_tr1.npt
Tr 2    ttr_tr2.npt
Tr 3    ttr_tr3.npt
Tr 4    ttr_tr4.npt
Tr 5    ttr_tr5.npt
Tr 6    ttr_tr6.npt
Tr 7    ttr_tr7.npt
Tr 8    ttr_tr8.npt
Tr 9    ttr_tr9.npt

CTR FILE..............................CTRFN..............................
Tr 1    ctr_tr1.npt
Tr 2    ctr_tr2.npt
Tr 3    ctr_tr3.npt
Tr 4    ctr_tr4.npt
Tr 5    ctr_tr5.npt
Tr 6    ctr_tr6.npt
Tr 7    ctr_tr7.npt
Tr 8    ctr_tr8.npt
Tr 9    ctr_tr9.npt

QDT FILE..............................QDTFN..............................
Br 1    qdt_br1.npt
Br 2    qdt_br2.npt - not used
Br 3    qdt_br3.npt - not used
Br 4    qdt_br4.npt - not used
Br 5    qdt_br5.npt - not used

TDT FILE..............................TDTFN..............................
Br 1    tdt_br1.npt
Br 2    tdt_br2.npt - not used
Br 3    tdt_br3.npt - not used
Br 4    tdt_br4.npt - not used
Br 5    tdt_br5.npt - not used

CDT FILE..............................CDTFN..............................
Br 1    cdt_br1.npt
Br 2    cdt_br2.npt - not used
Br 3    cdt_br3.npt - not used
Br 4    cdt_br4.npt - not used
Br 5    cdt_br5.npt - not used
```

PRE FILE..............................PREFN...............................

Br 1 qpr_br1.npt
Br 2 qpr_br2.npt
Br 3 qpr_br3.npt
Br 4 qpr_br4.npt
Br 5 qpr_br5.npt

TPR FILE..............................TPRFN...............................

Br 1 tpr_br1.npt
Br 2 tpr_br2.npt
Br 3 tpr_br3.npt
Br 4 tpr_br4.npt
Br 5 tpr_br5.npt

CPR FILE..............................CPRFN...............................

Br 1 cpr_br1.npt
Br 2 cpr_br2.npt
Br 3 cpr_br3.npt
Br 4 cpr_br4.npt
Br 5 cpr_br5.npt

EUH FILE..............................EUHFN...............................

Br 1 euh_br1.npt - not used
Br 2 euh_br2.npt - not used
Br 3 euh_br3.npt - not used
Br 4 euh_br4.npt - not used
Br 5 euh_br5.npt - not used

TUH FILE..............................TUHFN...............................

Br 1 tuh_br1.npt - not used
Br 2 tuh_br2.npt - not used
Br 3 tuh_br3.npt - not used
Br 4 tuh_br4.npt - not used
Br 5 tuh_br5.npt - not used

CUH FILE..............................CUHFN...............................

Br 1 cuh_br1.npt - not used
Br 2 cuh_br2.npt - not used
Br 3 cuh_br3.npt - not used
Br 4 cuh_br4.npt - not used
Br 5 cuh_br5.npt - not used

EDH FILE..............................EDHFN...............................

Br 1 edh_br1.npt - not used
Br 2 edh_br2.npt - not used
Br 3 edh_br3.npt - not used
Br 4 edh_br4.npt - not used
Br 5 edh_br5.npt - not used

TDH FILE..............................TDHFN...............................

Br 1 tdh_br1.npt - not used
Br 2 tdh_br2.npt - not used
Br 3 tdh_br3.npt - not used
Br 4 tdh_br4.npt - not used
Br 5 tdh_br5.npt - not used

CDH FILE..............................CDHFN...............................

Br 1 cdh_br1.npt - not used
Br 2 cdh_br2.npt - not used
Br 3 cdh_br3.npt - not used
Br 4 cdh_br4.npt - not used
Br 5 cdh_br5.npt - not used

```
SNP FILE............................SNPFN.................................
        00-02.noUVDK.snp

TSR FILE............................TSRFN.................................
        00-02.noUVDK.tsr

PRF FILE............................PRFFN.................................
        00-02.noUVDK.prf

VPL FILE............................VPLFN.................................
        00-02.noUVDK.vpl

CPL FILE............................CPLFN.................................
        00-02.noUVDK.cpl

SPR FILE............................SPRFN.................................
        00-02.noUVDK.spr
```

Appendix B – CE QUAL W2 Sample Inflow File (QIN_BR1.NPT)

```
Stillwater River
Inflow quantity 00-02 (00 & 01 evap edited)
    Jday cu. m/s
     0.5  0.6481
     1.5  0.6842
     2.5  0.9002
     3.5  1.1523
     4.5  2.9167
     5.5  2.7006
     6.5  1.9444
     7.5  1.5483
     8.5  1.3323
     9.5  1.4403
    10.5  3.8889
    11.5  3.2767
    12.5  2.3045
    13.5  1.8004
    14.5  1.1523
    15.5  1.0442
    16.5  0.9722
    17.5  0.9002
    18.5  1.0082
    19.5  0.9362
    20.5  0.8642
    21.5  0.8282
    22.5  0.8282
    23.5  0.7922
    24.5  0.7562
    25.5  0.7202
    26.5  0.8642
    27.5  0.8282
    28.5  0.7562
    29.5  0.7202
    30.5  0.8282
    31.5  0.9362
```

Appendix C – CE QUAL W2 Sample Tributary Temperature File (TTR_TR1.NPT)

```
Waushacum Brook
Weekly Temperature 2000-2002
JDAY    TIN (C)
     3.5     2.5
    10.5     2.1
    17.5     0.2
    25.5     0.1
    31.5     0.1
    38.5     0.1
    45.5     0.3
    52.5     0.4
    59.5     1.1
    66.5     3.4
    73.5     3.2
    80.5     4.7
    88.5    10.2
    94.5      10
   101.5     7.3
   108.5     9.7
   115.5     9.1
   122.5    11.2
   129.5    21.5
   136.5    14.2
   143.5    13.2
   150.5    14.5
   157.5    15.9
   164.5      16
   171.5    20.2
   178.5    24.9
   186.5    24.5
   192.5    23.4
   201.5    21.3
   206.5    21.6
   213.5    18.8
   220.5    23.3
```

Appendix D – CE QUAL W2 Sample Tributary Constituent File (CTR_TR3.NPT)

Quabbin Reservoir at CVA
Water Quality Constituents 2000 - 2002

JDAY	UV-254	L DOM	R DOM	POC	PO4	NH4	NO3	DO
1	0.02	0.3502	1.4008	0.184	0.004	0.009	0.01436	10
364	0.02	0.3502	1.4008	0.184	0.004	0.009	0.01436	10
366	0.022	0.3268	1.3072	0.172	0.0027	0.01	0.0146	10
372	0.02	0.3348	1.3392	0.176	0.0027	0.0101	0.0146	10
382	0.022	0.3456	1.3824	0.182	0.0027	0.00883	0.0153	10
387	0.022	0.332	1.328	0.175	0.0026	0.00755	0.016	10
393	0.021	0.3178	1.2712	0.167	0.0026	0.00628	0.0167	10
400	0.021	0.3538	1.4152	0.186	0.0025	0.005	0.0174	10
407	0.021	0.3392	1.3568	0.179	0.0025	0.00589	0.01835	10
415	0.021	0.4096	1.6384	0.216	0.0026	0.00678	0.0193	10
421	0.021	0.3204	1.2816	0.169	0.0026	0.00767	0.02025	10
435	0.022	0.377	1.508	0.198	0.0026	0.00856	0.0212	10
442	0.02	0.3924	1.5696	0.207	0.0033	0.00811	0.02207	10
449	0.02	0.2934	1.1736	0.154	0.0041	0.00767	0.02293	10
456	0.021	0.3978	1.5912	0.209	0.0048	0.00722	0.0238	10
463	0.02	0.3654	1.4616	0.192	0.0049	0.00678	0.0215	10
471	0.021	0.3924	1.5696	0.207	0.0049	0.00633	0.0192	10
477	0.021	0.3088	1.2352	0.163	0.0049	0.00589	0.0169	10
484	0.021	0.4104	1.6416	0.216	0.005	0.00544	0.0146	10
491	0.021	0.4338	1.7352	0.228	0.005	0.005	0.0123	10
498	0.021	0.4212	1.6848	0.222	0.0046	0.00569	0.01625	10
505	0.022	0.3834	1.5336	0.202	0.0041	0.00639	0.0202	10
513	0.022	0.3842	1.5368	0.202	0.0037	0.00708	0.02415	10
519	0.022	0.387	1.548	0.204	0.0033	0.00777	0.0281	10
526	0.023	0.379	1.516	0.199	0.0032	0.00746	0.02555	10
534	0.023	0.3392	1.3568	0.179	0.0031	0.00715	0.02301	10
540	0.022	0.485	1.94	0.255	0.003	0.00685	0.02046	10
547	0.024	0.3852	1.5408	0.203	0.0029	0.00654	0.01792	10
554	0.024	0.3798	1.5192	0.2	0.0028	0.00623	0.01537	10
561	0.023	0.4338	1.7352	0.228	0.0028	0.00592	0.01283	10
568	0.023	0.4014	1.6056	0.211	0.0027	0.00562	0.01028	10
575	0.024	0.441	1.764	0.232	0.0026	0.00531	0.00774	10

Appendix E – CE QUAL W2 Sample Withdrawal File (QWD_NEW.NPT)

Reservoir Withdrawals (Excluding Cosgrove)
Sum of (ONR, OWA, OT, OND)

JDAY	ND+T	Nashua	Wach A
0.5	0.1095	0.0701	0.0876
1.5	0.1139	0.0701	0.0876
2.5	0.1052	0.0701	0.0876
3.5	0.1139	0.0701	0.0876
4.5	0.1139	0.0701	0.0876
5.5	0.1139	0.0701	0.0876
6.5	0.1052	0.0701	0.0876
7.5	0.1139	0.0701	0.0876
8.5	0.1183	0.0701	0.0876
9.5	0.1095	0.0701	0.0876
10.5	0.1095	0.0701	0.0876
11.5	0.1139	0.0701	0.0876
12.5	0.1095	0.0701	0.0876
13.5	0.1052	0.0701	0.0876
14.5	0.1183	0.0701	0.0876
15.5	0.1095	0.0701	0.0876
16.5	0.1227	0.0701	0.0876
17.5	0.1183	0.0701	0.0876
18.5	0.1183	0.0701	0.0876
19.5	0.1008	0.0701	0.0876
20.5	0.1227	0.0701	0.0876
21.5	0.1183	0.0701	0.0876
22.5	0.1271	0.0701	0.0876
23.5	0.1139	0.0701	0.0876
24.5	0.1227	0.0701	0.0876
25.5	0.1183	0.0701	0.0876
26.5	0.1139	0.0701	0.0876
27.5	0.1139	0.0701	0.0876
28.5	0.1139	0.0701	0.0876
29.5	0.1183	0.0701	0.0876
30.5	0.1183	0.0701	0.0876
31.5	0.1183	0.0701	0.0876