University of Massachusetts Amherst
ScholarWorks@UMass Amherst

Doctoral Dissertations | Dissertations and Theses

November 2016

Proactive Assessment of Climate Change and Contaminant Spill Impacts on Source Water Quality

Lillian C. Jeznach
University of Massachusetts, Amherst

Follow this and additional works at: https://scholarworks.umass.edu/dissertations_2

Part of the Civil Engineering Commons, and the Environmental Engineering Commons

Recommended Citation

Jeznach, Lillian C., "Proactive Assessment of Climate Change and Contaminant Spill Impacts on Source Water Quality" (2016). *Doctoral Dissertations*. 720.
https://scholarworks.umass.edu/dissertations_2/720

This Open Access Dissertation is brought to you for free and open access by the Dissertations and Theses at ScholarWorks@UMass Amherst. It has been accepted for inclusion in Doctoral Dissertations by an authorized administrator of ScholarWorks@UMass Amherst. For more information, please contact scholarworks@library.umass.edu.

PROACTIVE ASSESSMENT OF CLIMATE CHANGE AND CONTAMINANT SPILL IMPACTS ON SOURCE WATER QUALITY

A Dissertation Presented

by

Lillian C. Jeznach

Submitted to the Graduate School of the
University of Massachusetts Amherst in partial fulfillment
of the requirements for the degree of

DOCTOR OF PHILOSOPHY

September 2016

Department of Civil and Environmental Engineering

© Copyright by Lillian C. Jeznach 2016

All Rights Reserved

PROACTIVE ASSESSMENT OF CLIMATE CHANGE AND CONTAMINANT SPILL IMPACTS ON SOURCE WATER QUALITY

A Dissertation Presented

by

Lillian C. Jeznach

Approved as to style and content by:

____
John E. Tobiason, Chair

____
Mi-Hyun Park, Member

____
David P. Ahlfeld, Member

____
Christine E. Hatch, Member

____
Richard Palmer, Department Chair
Department of Civil and Environmental Engineering

DEDICATION

"To all of my family, for their support and encouragement"

ACKNOWLEDGEMENTS

I would like to express my sincere gratitude to my advisor John Tobiason for his support and academic guidance throughout my years in graduate school. Thank you for your dedication to my success and well-being as a graduate student. I will continue to look to you as an example during my future academic career. Thank you to Mi-Hyun Park for her research insights and thoughtful comments on my work as a member of the DCR team. I would like to extend further appreciation to the other members of my committee, David Ahlfeld and Christine Hatch, for all their helpful advice and direction on my research. I would also like to acknowledge the Massachusetts Department of Conservation and Recreation (MA DCR) Division of Water Supply and Protection, specifically Pat Austin, for the support throughout my time at UMass and the opportunity to work on such an interesting project for a drinking water reservoir so close to home.

To my EWRE graduate student peers, thank you for your support, collaborative attitudes, and friendship. It has truly been a wonderful experience working with you all. I also owe a huge thank you to the women of GWIS, who have been invaluable peer mentors and central to my personal and professional development throughout my academic studies.

Finally, I am incredibly grateful for the love and support from my family, especially my parents, who have always encouraged curiosity and creativity in life. Thank you for being such positive role models and inspiring me to pursue my career goals. A special thank you to my husband, Chris, who has been there for both the celebrations and also the pep talks when I needed them most. You are the best partner and teammate in life that one could ask for.

ABSTRACT

PROACTIVE ASSESSMENT OF CLIMATE CHANGE AND CONTAMINANT SPILL IMPACTS ON SOURCE WATER QUALITY

SEPTEMBER 2016

LILLIAN C. JEZNACH

B.S., WORCESTER POLYTECHNIC INSTITUTE

M.S., UNIVERSITY OF MASSACHUSETTS AMHERST

Ph.D., UNIVERSITY OF MASSACHUSETTS AMHERST

Directed by: Professor John E. Tobiason

Managing the water quality of surface drinking water sources has become an increasingly difficult task for water suppliers due to increased watershed urbanization and climate change. Changes in source water quality may affect public perceptions, treatment effectiveness, and ultimately costs to treat water to drinking standards. Although there are increased threats to current and future drinking water quality, current approaches to managing these threats are typically reactionary. Prior detailed modeling efforts of hypothetical events that may impair raw water quality allow for an understanding of constituent fate and transport, including potential maximum concentrations and travel times to the drinking water intake for constituents which may be of concern. The primary goal of this dissertation was to present proactive frameworks that utilize a hydrodynamic and water quality model to aid in developing scientifically-based management plans prior to an accidental or natural event occurring. The Wachusett Reservoir, a major drinking water supply for metropolitan Boston, Massachusetts, was used as a case study to illustrate proactive modeling efforts to quantify water quality impacts after both short and long-

term potential events. This work used a process-based modeling approach to simulate reservoir hydrodynamic and water quality responses to changes in various model inputs (streamflow, constituents, meteorology) and also evaluated current and future management decisions which may mitigate water quality impacts. The approach is demonstrated through a series of proactive modeling studies conducted to evaluate water quality at the drinking water intake in response to contaminant incidences, long-term increasing air temperatures, and extreme precipitation events. For the Wachusett Reservoir, proactive contaminant modeling highlighted the importance of a rapid response by managers to contain a contaminant spill and therefore minimizing the mass of contaminant that is able to enter the water column following an event. In scenarios that simulated long term increasing air temperature increases, model results suggested increases in epilimnion and hypolimnion water temperatures, decreased ice cover, and increased stratification duration by 2112. Extreme precipitation event simulations during the spring and summer resulted in organic matter concentrations that exceeded recorded maximums at the drinking water intake while nutrients for this particular reservoir remained low. The modeling results provide valuable insights into water quality responses to changes in water body inputs and can help inform short and long-term management strategies, prior to water quality degrading events.

TABLE OF CONTENTS

Page

LIST OF TABLES

Table **Page**

LIST OF FIGURES

Figure **Page**

INTRODUCTION

Managing the water quality of drinking water reservoirs has become an increasingly difficult task for many water suppliers in the face of increasing urbanization and climate change threats. In spite of increased threats to water quality impairment, current approaches to risk management and planning for drinking water supplies are typically reactionary, and not a process-based, system specific approach that is needed to monitor and maintain high quality drinking water (Baum et al., 2016). Risks to drinking water degradation include short-term low probability but high-risk events (e.g. contaminant spills, extreme weather events) and longer-term events (e.g. increasing air temperatures due to climate change). Changes in source water quality may affect public perceptions, treatment effectiveness, and ultimately costs to treat water to drinking standards. Managing the impacts of these water quality events is difficult, largely because predicting constituent fate and transport in reservoirs is complicated, often with multiple degradation pathways and interactions with other water quality variables in addition to reservoir hydrodynamics. Also, while these events are often high risk, the probability of them occurring in normal day to day operation can be small, highlighting the importance of proactive model applications to better understand and plan for events prior to an occurrence that may disrupt high quality water service.

Increasing urbanization and chemical storage/transport in water supply watersheds as well as the increase in highly publicized contamination incidents, such as the recent West Virginia chemical spill into the Elk River, are indications that water managers need to proactively plan for emergency contaminant scenarios (Bahadur and Samuels, 2014; Whelton et al., 2014). Hydrodynamic and water quality models have been used to understand the fate and transport of contaminants in drinking water sources and to evaluate management responses after an event has occurred (Chung and Gu, 2009; Chung and Gu, 1998; Gu and Chung, 2003; Martin et al., 2004). However, there are few published studies documenting proactive contaminant modeling efforts to guide emergency response planning prior to an event occurring. This may be due to the sensitive nature of particular contaminants of concern to a waterbody as well as the potential emergency response plans. A proactive modeling framework to simulate potential contaminant spills into a particular surface water supply, evaluate the impacts on water quality and the effectiveness of operational responses can provide significant scientific insight into emergency plan development.

Models can also be used to gain more insight into the ranges of water quality responses a water supply system is likely to experience on a local scale as a result of climate changes such as increasing air temperatures and more frequent occurrences of low probability events (Fang and Stefan, 1998; Fang and Stefan, 1999; Fang and Stefan, 2009; George et al., 2007; Komatsu et al., 2007; Lee et al., 2012; Sahoo and Schladow, 2008; Sahoo et al., 2011; Samal et al., 2012). Climate change impacts on water quality will vary by source, and utility managers and operators will need to understand how these changes may affect the fluctuations in water quantity and quality that will impact treatment and delivery. For example, natural inputs of water quality constituents from tributary sources play a large role in governing the daily water quality of a surface water reservoir, but these historical fluctuations in inputs may be altered by short and long-term changes in climate. Increasing air temperatures may lengthen the duration a reservoir is stratified and may decrease the number of days in which a reservoir is ice covered, which has water quality implications (Jankowski et al., 2006; Livingstone, 2003; Livingstone, 1993; Magnuson et al., 2000; Peeters et al., 2002). Climate projections can often be confusing to water managers due to the nature of imprecise models and local variability. However, models can be used to evaluate the sensitivity of reservoir water quality to probable ranges in climate changes as well as scientifically and quantitatively inform decisions that may mitigate potential impacts.

This dissertation work is motivated by the need for more proactive, as opposed to reactive, hydrodynamic and water quality modeling studies to guide water managers in making scientifically based management decisions to best protect the quality of drinking water supplied from surface water sources. The Wachusett Reservoir, a major drinking water supply to metropolitan Boston Massachusetts, is used as a case study example application of a proactive modeling framework and water quality modeling assessment. A series of proactive modeling studies were conducted to ultimately evaluate water quality at the drinking water intake in response to contaminant incidences, long-term increasing air temperatures, and extreme precipitation events. The modeling results provide valuable insights into water quality responses to changes in water body inputs and can help inform short and long-term management strategies, prior to water quality degrading events.

I.1 Drinking water contaminants

A major category of drinking water contaminants of concern to drinking water managers, particularly those in more urban watersheds, are contaminants that originate from anthropogenic sources. An example of a recent anthropogenic point source contaminant input to a drinking water supply was the West Virginia chemical spill into the Elk River in January 2014 from an industrial source located 1.61 km (1 mile) upstream from the Kanawha Valley Water Treatment Plant, the drinking water supply to Charleston, WV (Bahadur and Samuels, 2014; Whelton et al., 2014). An estimated 37,854 L (10,000 gal) of 4-methycyclohexane methanol (MCHM) and propylene glycol phenyl ether (PPH), organic solvents used in coal processing, spilled into the river, which shortly overwhelmed treatment capacity of the drinking water treatment plant and led to a water ban for an estimated 300,000 West Virginia residents (Bahadur and Samuels, 2014). The long-term health effects of this spill on the residents (15% of the state's population) is largely unknown, since little to no toxicological data and physiochemical properties are available for many of the solvent's ingredients (Whelton et al., 2014). Other anthropogenic contaminants to surface water include excessive nutrient inputs from wastewater or agricultural runoff, pesticides, pharmaceuticals, industrial compounds, and deicing agents (Chaudhury et al., 1998; Forman and Alexander, 1998; Sansalone and Buchberger, 1997; Sharpley et al., 1994; Wauchope, 1978).

Constituents of natural origin which may be of concern to drinking water managers include sediments, organic matter, and nutrients which may become abnormally high during heavy precipitation events (Caverly et al., 2013; Correll et al., 1999; Dhillon and Inamdar, 2013; Dhillon and Inamdar, 2013; Inamdar et al., 2006; Jung et al., 2014; Yoon and Raymond, 2012). During precipitation storm events, constituent loads to a surface water body generally increase due to tributary flow increases. However, concentrations of various constituents may increase or decrease depending on the physical characteristics of the watershed and the dominant processes governing their concentrations in the tributaries. Comparisons among different storms and watersheds have shown that season, precipitation amount, and antecedent moisture conditions effect solute concentrations and loads (Dhillon and Inamdar, 2014; Inamdar et al., 2006).

The Wachusett Reservoir, in central Massachusetts, is an oligo-mesotrophic surface water body that supplies metropolitan Boston (approximately 2.5 million people) with drinking water. The watershed is primarily forest (67%), followed by urban/developed land (17%), open water/wetland (10%), and agricultural land (8%) (Hagemann and Park, 2014). Nutrient and organic matter loads to the reservoir from the watershed tributaries are low and have decreased in the last decade due to implementation of storm water best management practices (BMPs) (Hagemann and Park, 2014). Organic matter and nutrient loads have been observed to increase during precipitation events (Hagemann and Park, 2014). However, anthropogenic contaminants, such as those carried by tanker truck or freight train, are also of concern due to the proximity of several major roadways and a railway to the reservoir, as seen in Figure I.1. As seen in the figure, there are two roadway bridge crossings and one railway bridge crossing over the reservoir. These locations are the highest concern for accidental contamination from an overturned vehicle or railcar.

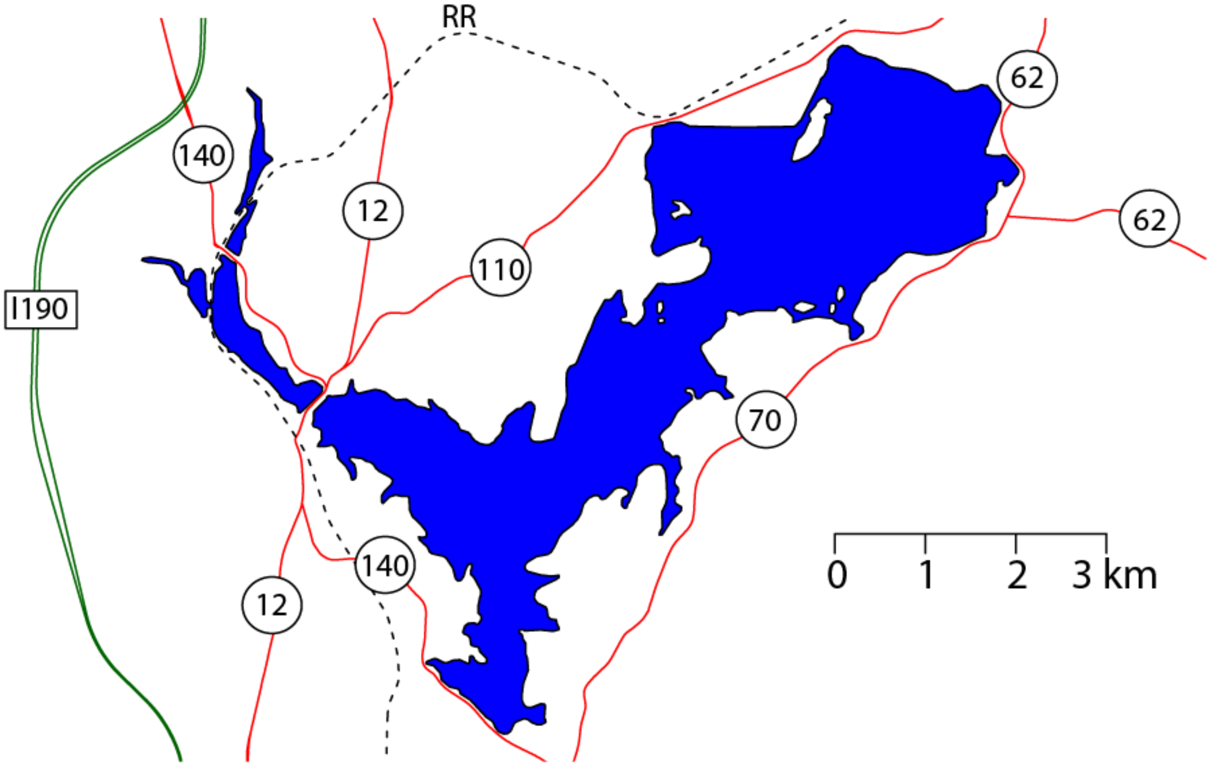

Figure I.1 Major roads and a railway surrounding Wachusett reservoir

I.2 Current and future climate trends and implications for water quality

Historical records of global mean surface air temperatures indicate a rise of 0.7ºC since the start of the 20th century (World Meteorological Organization (WMO), 2014). The average surface temperature in the U.S. has increased by 1.3 – 1.9 ºF (0.73 – 1.1 ºC) since 1895 with the majority of this increase occurring after 1970 (Melillo et al., 2014). By the end of the century, air temperatures in the U.S. are projected to increase by roughly 3 – 5 ºF in lower emission scenarios and possibly 5 – 10 ºF in higher emission scenarios (Melillo et al., 2014). Projections for the northeast region of the U.S. anticipate air temperature increases of about 3 – 6 ºF for reduced emission scenarios and about 4.5 – 10 ºF for increased emission scenarios by the 2080s (Horton et al., 2014). Changes in historic precipitation over the U.S. have been observed but these vary significantly with region. In general, there has been a national trend towards a greater amount of very heavy precipitation events (the heaviest 1% of all events) throughout the U.S. from 1958 – 2012 with the largest observed change occurring in the northeast region, which has experienced a 71% increase in the last 50 years (Horton et al., 2014). Recent examples of extreme heavy precipitation events in the northeastern U.S. include Hurricane Irene (summer 2011) and Hurricane Sandy (fall 2012).

Increasing air temperature affects physical, chemical, and biological processes in surface water bodies that impact water quality. Of all the meteorological drivers, air temperature has the most significant effect on water temperature variability (Henderson-Sellers, 1988). Long term changes in water temperature as a response to increasing air temperatures have been observed in lakes and reservoirs throughout the globe. Historical records of surface water bodies indicate that gradual changes in air temperatures due to climate change have increased thermal stratification intensity and the length of stratification (Arhonditsis et al., 2004; Coats et al., 2006; Jankowski et al., 2006; Livingstone, 2003; Livingstone, 1993; Peeters et al., 2002; Verburg and Hecky, 2009). Ice cover duration and thickness in temperate regions has also been observed to decline over the last century (Magnuson et al., 2000). Physical changes related to water temperature changes promotes hypoxic and anoxic conditions, particularly in the hypolimnion, and has been observed to extend these conditions in water bodies that already experience them (Jankowski et al., 2006; Porter et al., 1996). Changes in water temperatures also impact nutrient cycling within the water column, contributing to anoxic conditions in the hypolimnion and resulting in the release of nutrients (N and P). The impacts of long term air temperature increases on biological processes that effect water quality is complex because the dynamics depend on nutrient availability, density dependence, predation, and species specific dynamics which vary across ecosystems (Adrian et al., 2009)

There are few published studies analyzing water quality impacts from extreme precipitation events, despite the knowledge that large precipitation events are a major driver for the export of terrigenous organic carbon and organic-bound nutrients to receiving water bodies. Impacts of events depend on different precipitation volumes and intensities as well as watershed characteristics. In studies of forested watersheds during hurricane Irene in August of 2011, DOC and POC fluxes to a receiving water body were observed to contribute up to 40% and 50% of the annual masses of the respective constituents (Dhillon and Inamdar, 2013; Yoon and Raymond, 2012). Ratios of DOC to POC after extreme precipitation events in forested watersheds have been observed to be less than 1 while agricultural watersheds have been observed to be greater than 1. Nutrient concentrations and exports during extreme precipitation events are even less well understood. However, nutrient exports from catchments have been shown to be primarily associated with episodes of high discharge and sediment loads, similar to organic matter (Correll et al., 1999; Inamdar et al., 2006; Meyer and Likens, 1979).

I.3 Modeling approaches

Hydrodynamic and water quality models are commonly used to simulate water quality of a receiving water body in response to changes in watershed inputs and/or climate change. The two-dimensional (2-D), laterally averaged, model CE-QUAL-W2 is an example of a commonly used model that has been applied to over 200 water bodies around the world (Cole and Wells, 2015). Other examples of models that can be used in hydrodynamic and water quality modeling studies include one-dimensional (e.g., LCM, DYRESM, MINLAKE, CE-THERM-R1), two-dimensional (e.g., GVLHT), and three-dimensional (e.g., GLLVHT, EFDC, GEMSS) models.

In most cases, hydrodynamic and water quality models for drinking water sources are developed in response to accidental or natural events that resulted in impaired water quality. Modeling studies are typically carried out post-event to guide operational response, data collection, or remediation efforts (Clark et al., 1989; Grayman and Males, 2002; Gullick et al., 2003). For example, modeling studies were undertaken to develop operational strategies post-spills of conservative contaminant methyl isothiocyanate (MITC) in the Shasta Reservoir in California and the herbicide atrazine from agricultural runoff into the Saylorville Reservoir in Iowa (Chung and Gu, 2009; Chung and Gu, 1998; Gu and Chung, 2003). Many utilities around the world are now incorporating early warning systems into their source water protection efforts, which often include 1-D models that can be run with real-time data during a contamination event (Gullick et al., 2003). However, reservoir managers, particularly of non-riverine systems with vertical and/or lateral gradients, could also benefit from more comprehensive (2-D or 3-D) pre-event proactive hydrodynamic and water quality modeling to provide more accurate realizations of contaminant fate and transport for emergency plan development.

Hydrodynamic and water quality models can be used to proactively model future climate change impacts to water quality in surface water reservoirs and develop long term management strategies. A common modeling approach to simulating future climate change impacts on water quantity and quality is to employ the use of downscaled general circulation models (GCMs) to drive a watershed model, a systems model, and a hydrodynamic and water quality model. GCMs are a popular tool for generating climate scenarios to use in impact assessments evaluating long term changes in air temperature on water temperature and quality (Fang and Stefan, 1998; Fang and Stefan, 1999; Fang and Stefan, 2009; Komatsu et al., 2007; Sahoo et al., 2013; Sahoo et al., 2015; Samal et al., 2012). However, GCMs are also criticized for their inherent biases and their inability to capture the full range of future climate uncertainty (Brown and Wilby, 2012; Brown et al., 2012). Additionally, since the small temporal and spatial scales of extreme events cannot currently by accurately represented by a GCM, they are considered inappropriate for use in assessments simulating future impacts of short term extreme weather scenarios (Baker and Peter, 2008; Willems et al., 2012). In fact, to the author's knowledge, there are no published studies linking future extreme precipitation events with modeled watershed constituent loads and receiving water body quality using either GCMs or another future meteorological realization. There are, however, many studies citing qualitative observations and predictions of water quality impacts in raw drinking water sources due to extreme precipitation events. Reported impacts include increased organic matter and nutrient loads, algal blooms, microbial contamination, and well as color and turbidity issues, as summarized in Stanford et al (2014).

The primary contribution of this dissertation is to present proactive frameworks utilizing hydrodynamic and water quality models to aid in developing management plans prior to an accidental or natural event occurring. Each chapter highlights a different need for proactive management for the example drinking water supply reservoir, models a short or long-term potential event of concern, and quantifies key outcomes of the event which may affect current and future management decisions. The following 3 chapters present a forward thinking scientific and quantitative approach to managing drinking water quality in the face of increased watershed urbanization and an uncertain future climate.

CHAPTER 1

A FRAMEWORK FOR MODELING CONTAMINANT IMPACTS ON RESERVOIR WATER QUALITY

Authors

Jeznach, L.C., Jones, C., Matthews, T., Tobiason, J.E., and Ahlfeld, D.A., 2016. A framework for modeling reservoir contaminant impacts on water quality, Journal of Hydrology, Vol 537, pp. 322-333

1.1. Abstract

This study presents a framework for using hydrodynamic and water quality models to understand the fate and transport of potential contaminants in a reservoir and to develop appropriate emergency response and remedial actions. In the event of an emergency situation, prior detailed modeling efforts and scenario evaluations allow for an understanding of contaminant plume behavior, including maximum concentrations that could occur at the drinking water intake and contaminant travel time to the intake. A case study assessment of the Wachusett Reservoir, a major drinking water supply for metropolitan Boston, MA, provides an example of an application of the framework and how hydrodynamic and water quality models can be used to quantitatively and scientifically guide management in response to varieties of contaminant scenarios. The model CE-QUAL-W2 was used to investigate the water quality impacts of several hypothetical contaminant scenarios, including hypothetical fecal coliform input from a sewage overflow as well as an accidental railway spill of ammonium nitrate. Scenarios investigated the impacts of decay rates, season, and inter-reservoir transfers on contaminant arrival times and concentrations at the drinking water intake. The modeling study highlights the importance of a rapid operational response by managers to contain a contaminant spill in order to minimize the mass of contaminant that enters the water column, based on modeled reservoir hydrodynamics.

The development and use of hydrodynamic and water quality models for surface drinking water sources subject to the potential for contaminant entry can provide valuable guidance for making decisions about emergency response and remediation actions.

1.2. Introduction

Protecting surface drinking water sources from point and non-point sources of contamination is an important element to maintaining high drinking water quality and minimizing treatment costs. However, if contaminants enter a drinking water source, it is important to have an understanding of their fate and transport in the surface water body and how they may impact drinking water quality. Hydrodynamic modeling studies are typically carried out after a contamination incident to guide operational response, data collection, or remedial action, while few contaminant modeling studies have been published that help to understand contaminant transport in anticipation of incidents (Clark et al., 1989; Grayman and Males, 2002; Gullick et al., 2003). Hydrodynamic and water quality models are useful tools for determining contaminant plume behavior and impacts to water quality at a drinking water intake, making simulations especially useful for accessing multiple scenarios and proactively developing response plans for rapid and appropriate action in the event of an actual contaminant event in a drinking water reservoir (DiGiano and Grayman, 2014; Henderson-Sellers, 1991).

Water quality models can be used to simulate hydrodynamics, heat transfer, and water quality processes in water bodies in order to predict the fate and transport of water quality constituents or contaminants. Models simulate concentrations of nutrients, pathogens, microbes, and other constituents in the water body based on analytical or numerical solutions to equations describing physical, chemical, and biological processes of importance. Simulating conservative constituents is important for confirming that a model adequately describes the hydrodynamics of a system.

Reactive water quality constituents are affected by both fluid transport processes and reactions, which may result in degradation and lower concentrations at a location of interest such as a drinking water intake. Contaminants which may be of concern to reservoir operators include natural substances (natural organic matter (NOM)) and inorganic species, pathogens and other microorganisms, as well as the accidental release of anthropogenic substances (e.g. pesticides, pharmaceuticals, industrial compounds).

A contaminant may enter a water body from point and nonpoint sources. Point sources of contaminants are inputs occurring at a single location and may include inflows such as those from a tributary, discharges from a wastewater treatment plant, or an accidental contaminant spill. For example, in the year 2011, Hurricane Irene in the Northeastern US, a 200-yr event, caused discharge and dissolved organic matter concentrations in the Esopus Creek to increase 330 and 4-fold, respectively (Yoon and Raymond, 2012). The Esopus Creek drains 16,500 ha of the Catskill Mountains and eventually discharges into the Ashokan Reservoir, a primary drinking water source for New York City. The inflows from this tributary are closely monitored, since high stream flows from large events are associated with large point sources of nutrients, sediment, and pollutant transport to the reservoir. A recent anthropogenic point source contaminant input to a drinking water supply was the West Virginia chemical spill into the Elk River in January 2014 from an industrial source located 1.61 km (1 mile) upstream from the Kanawha Valley Water Treatment Plant, the drinking water supply to the Charleston, WV (Bahadur and Samuels, 2014). An estimated 37,854 L (10,000 gal) of 4-methycyclohexane methanol (MCHM) and propylene glycol phenyl ether (PPH), organic solvents used in coal processing, spilled into the river, which shortly overwhelmed treatment capacity of the drinking water treatment plant and led to a water ban for an estimated 300,000 West Virginia residents (Bahadur and Samuels, 2014).

Nonpoint sources of contaminants can include distributed runoff from a watershed, subsurface flow, atmospheric inputs, or roosting waterfowl. Diffuse pollution in agricultural watersheds from cropland and livestock are often significant sources of nutrients and pesticides to a surface water body resulting in impaired water quality (Sharpley et al., 1994; Wauchope, 1978). Field studies of roosting gulls on the Quabbin Reservoir, a drinking water supply reservoir for metropolitan Boston, MA, investigated this non-point source of fecal coliform to understand how gull roost location impacts coliform concentration at the drinking water intake (Garvey et al., 1998). Water from the Quabbin Reservoir has a filtration waiver from the Surface Water Treatment Rule (SWTR). Therefore, managers are required to conduct studies to understand the fate and transport of contaminants, like fecal coliform, and develop source water management plans to deal with nighttime roosting of gulls on the reservoir to maintain water quality. The field study, in conjunction with hydrodynamic and water quality modeling efforts, later described, helped reservoir management understand the importance and impact of their gull harassment program.

Hydrodynamic and water quality models are commonly used for simulating the response of a water body to changes in nutrient loads or contaminant inputs from both point and non-point sources. Selection of an appropriate water quality model depends of the geometry of the water body and the transport process of interest. In temperate climates the fate and transport of contaminants in drinking water reservoirs are influenced by seasonal thermal stratification. Impacts of thermal stratification on the transport of an inflow containing a contaminant of concern is of particular interest to reservoir operators because studies indicate that thermally stratified water bodies significantly decrease vertical mixing, causing contaminants to spread laterally, potentially leading to higher contaminant concentrations at a drinking water intake compared to well mixed conditions (Chung and Gu, 1998; Gu et al., 1996; Jeznach et al., 2014; Marti et al., 2011).

Hydrodynamic and water quality modeling can be used to evaluate various contaminant scenarios and impacts of management response decisions specific to a particular water body. Contaminant types and sources of potential contaminants in a reservoir will depend on hydrologic and watershed characteristics. Although a drinking water treatment plant can have a robust treatment system to deal with variations in water quality characteristics, source water protection is an important barrier to water quality impairment from contaminants.

2-D models have been used in many applications to simulate constituent fate and transport and to evaluate management responses. Garvey et al. (1998) used CE-QUAL-W2 in conjunction with field studies, previously described, to simulate the fate and transport of fecal coliforms from gull roosting in the Quabbin Reservoir, an oligotrophic drinking water source in Massachusetts. The model results suggested that the gull roost location, wind speed, and wind direction impact the magnitude and variability of the outlet coliform concentration. Chung and Gu (1998) used the two-dimensional generalized longitudinal-vertical hydrodynamics and transport model (GLVHT) to simulate the transport and mixing of a spill of conservative contaminant methyl isothiocyanate (MITC) in Shasta Reservoir in California. GLVHT was developed from the laterally averaged reservoir model (LARM), which was later used to develop CE-QUAL-W2. The CE-QUAL-W2 model was modified to include a toxics sub model to better simulate the fate and transport processes of toxic contaminants, including sorption, desorption, photolysis, hydrolysis, oxidation, biotransformation, volatilization, diffusive exchanges between the sediments and water column, and sediment transport and deposition in the reservoir (Gu and Chung, 2003). The same modified model was used to simulate the herbicide atrazine in the Saylorville Reservoir in Iowa from agricultural runoff (Chung and Gu, 2009). The results from the study were useful in developing reservoir operation strategies to minimize contaminant concentration in the intake water (Chung and Gu, 2009).

Hydrodynamic and water quality models used in conjunction with a Spill Management Information System (SMIS) can help to effectively manage the risks of a potential spill into a water body (Martin et al., 2004). The GIS based system incorporates CE-QUAL W2 V3.1 as its surface water contaminant transport model and Computer-Aided Management of Emergency Operations (CAMEO) to model atmospheric dispersion. The SMIS application was designed to evaluate the short-term impacts of a chemical spill and to facilitate the development of a comprehensive response plan. This application was tested on the Cheatham Reach, a part of the Cumberland River, where the model simulated a 50,000 L spill of benzene that occurred over 1 hour. The combination of model results from CE-QUAL and CAMEO, and information from GIS layers, provides real-time planning and analysis capabilities for first-responders, facility operators, and emergency response organizations (Martin et al., 2004).

Knowledge of potential contaminant types and sources in combination with knowledge of the hydrodynamics and transport specific to surface water reservoirs can lead to more informed management decisions in the event of an actual contamination scenario. The use of models to simulate scenarios and evaluate management responses can be a critical exercise for developing appropriate and timely responses to emergency situations, guiding measurements of a contaminant plume as it travels, and reducing short and long term impacts to drinking water supplies. This paper presents a framework for using models to understand the hydrodynamics and transport of potential contaminants in a surface drinking water source and an example case study applying the framework to assess contaminant impacts on the Wachusett Reservoir, in central Massachusetts.

1.3. Framework

Figure 1.1 lays out a framework for assessing contaminant impacts on reservoir water quality and developing management response plans. Although the framework is presented in a linear order, the framework should be reevaluated frequently as new information becomes available or hypothetical contaminant scenarios no longer are relevant, for example. Also, it may be beneficial in some cases to conduct a simplified first screen of impacts using a generic conservative constituent tracer to evaluate the outcomes to a certain scenario, and then complete a more extensive assessment later or if more detailed information on specific contaminants would benefit the study. Prior to a contaminant incident, this proactive modeling framework can guide the placement of in-reservoir early warning monitoring systems and improve emergency response plans. Immediately following an incident, proactive modeling can guide responders in tracking a plume as it moves downstream and provide scientifically based estimations of contaminant travel time and concentrations at an intake location.

Step	Components
1. Determine Reservoir Characteristics	Bathymetry; inflows/outflows; operations
2. Model Development	Model selection; calibration and validation
3. Scenario Planning	Locations of concern; contaminant characteristics; reservoir conditions
4. Scenario Evaluation	Contaminant arrival time, concentration, and variability at intake
5. Risk Management	Management guidance; impacts of operational decisions

Figure 1.1 Framework for assessing contaminant impacts on reservoir water quality

1.3.1. Step 1: Determine reservoir characteristics

Preliminary studies detailing the reservoir physical characteristics, hydrology, and reservoir operations allow for an understanding of the hydrodynamics and transport of contaminants, and are therefore a logical first step in assessing contaminant impacts on reservoir water quality. Bathymetry of a reservoir is a key factor dictating whether a 1-D, 2-D, or 3-D model is an appropriate selection for simulating hydrodynamics and water quality. An overview of the reservoir's watershed and hydrology should be undertaken to determine major natural inflows and outflows. Other controlled inflows (treated wastewater, inter-reservoir transvers), outflows (releases, drinking water withdrawals) should also be accounted for. Reservoir operations, including rules of transfers from other reservoirs, releases downstream, withdrawals for drinking water treatment, and water surface elevation bounds, can greatly influence the overall water balance and transport of contaminants and should be included in data collection. Density currents can be created by inflows such as tributaries or inter-reservoir transfers can add complexity to hydrodynamic and transport behavior and arise from temperature or solute differences between water depths (Ahlfeld et al., 2003). A portfolio of possible operational water quality management decisions is also important later in the analysis for assessing the impact of operational responses which may affect contaminant behavior at a drinking water intake.

In-situ measurements of velocities, temperature, and water quality constituents across depths, as well as longitudinally and laterally across the reservoir should be completed to aid in determining appropriate model types. Measurements are also necessary for calibration and validation during model development.

1.3.2. Step 2: Model development

The assessment of reservoir characteristics described above is important for selection of an appropriate model for capturing the hydrodynamics and transport of a reservoir. One dimensional (1-D) models are appropriate for completely mixed reservoirs, with negligible lateral or vertical gradients. 1-D models are unable to simulate lateral and vertical gradients in velocities, temperature, and constituents, and therefore do not accurately represent physical processes occurring in many reservoirs. Stratifying water bodies are best simulated using two-dimensional (2-D) or three-dimensional (3-D) models. 2-D models can simulate hydrodynamics and transport in a reservoir during periods of stratification and are therefore more appropriate for waterbodies where longitudinal and vertical variability dominate lateral variability. 3-D models can simulate longitudinal, lateral, and vertical gradients, but are more computationally demanding than 1-D or 2-D models. However, a pre-contamination assessment using a 2-D or 3-D model helps to gain insight into the flow fields and allows for a more detailed understanding of the spread of the contaminant plume.

After data for boundary conditions are collected for inflows, outflows, inter-reservoir transfers, releases downstream, and drinking water withdrawals, a simple water balance analysis should be completed to compare calculated reservoir volumes and water surface elevations to measurements. This step ensures an adequate representation of historical inflows, outflows, and reservoir operations and can also allow one to make simplifying assumptions regarding the importance of smaller inflows and outflows to the water balance, such as those from groundwater or small tributaries. After this, additional boundary conditions, such as inflow temperatures and nutrient concentrations, and initial conditions can be incorporated if necessary. Once the model is calibrated and validated to measured in-situ water quality profiles and outflow measurements, contaminant scenarios should be developed.

1.3.3. Step 3: Scenario planning

A comprehensive list of locations where point and non-point sources of contaminants can enter the reservoir should be determined with the help of reservoir and watershed management expertise. These locations may include point sources of contamination such as roadways, railways, wastewater treatment plants, and tributaries. Non-point sources of contaminant entry may include roosting waterfowl, watershed runoff, or groundwater. Contaminants of concern may include anthropogenic contaminants such as industrial chemicals, fuel, fertilizer, or oil. Large mass loads of nutrients or fecal coliforms may be natural contaminants of concern. Locations and associated contaminant types should be ranked in a logical order of importance and relevance including factors such as location vulnerability, type and mass of contaminant, and probability of spill event occurrence.

Scenarios should be developed from ranked contaminants and locations that are anticipated to cause the greatest threat to drinking water quality. Contaminant scenarios should be developed with the following contaminant characteristics in mind:

- Volume
- Concentration
- Duration
- Density
- Miscibility
- Environmental fate
- Volatility
- Toxicity
- Reactivity

Additionally, the time of year in which a contaminant enters the water may be an important factor to consider, depending on the water body of interest and its geographic location. Seasonal thermal stratification and complete mixed conditions can impact contaminant travel time and concentration (Ahlfeld et al., 2003; Chung and Gu, 1998; Jeznach et al., 2014). Scenarios that simulate the fate and transport of contaminants during all seasons and mixed/non-mixed conditions provide a more complete understanding of reservoir characteristics on contaminant fate, transport, and impact on water quality. Wind speed and direction can also be a factor to consider including in scenarios, since wind may be of significance to lower density spills travelling along the top of the water column.

A baseline scenario of no management or operational response to contain, dilute, or slow down the contaminant should be simulated. In addition to this, a portfolio of management or operational decisions, as determined in Step 1, should be discussed and included in scenario simulations that may decrease contaminant travel time to a drinking water intake or reduce peak concentrations. These may include altering inter-reservoir transfer volumes or releases downstream, using booms to contain the spill to an area, aerating the reservoir, or using a different drinking water intake location or elevation, as examples.

It should be noted that conservative contaminant studies of a generic constituent (such as a tracer) are beneficial for gaining an initial understanding of hydrodynamics and transport in a reservoir prior to investigating scenarios of specific contaminant types which may decay or degrade in the natural environment. For example, density differences between an incoming contaminant and the ambient water can impact advection and dispersion of a contaminant plume. Conservative constituent scenarios can be investigated initially and used as a screening tool to help identify the most vulnerable locations or contaminants of concern.

1.3.4. Step 4: Scenario evaluation

Scenarios should be evaluated based on several characteristics, with the two key characteristics being contaminant arrival time and peak concentration at the drinking water intake. Contaminant arrival time can be defined as the number of days after a contaminant incident occurs where the contaminant concentration at the drinking water intake is a certain percent (e.g. 10%) of the complete mixed concentration in the reservoir. Contaminant arrival time is not limited to this definition however, and it should be defined in a manner appropriate to a specific contaminant, factoring in particular concentrations of concern if necessary. Scenarios may also be evaluated by the variability in contaminant concentration at the drinking water intake. Contaminant variability is directly related to the completely mixed or stratified nature of the reservoir during the year, where little variability in concentration at the intake is reflective of a well mixed reservoir and greater variability in concentration may be indicative of thermally stratified conditions (Jeznach et al., 2014). Peak concentrations, arrival times, and variability in concentration at a drinking water intake also depend on season, density, decay rate, and wind speed (Jeznach et al., 2014).

The use of a hydrodynamic and water quality model to simulate various scenarios allows for the evaluation of the impacts of management and operational decisions on contaminant arrival time, peak concentration, and variability. Impacts of operational decisions may lead to little change in contaminant behavior at a drinking water intake, or they may suggest improved or perhaps worsened conditions by management responses. Model simulations are very useful for evaluating various contaminant scenarios and a variety of management responses, with no risk to degrading actual drinking water quality, and provide a quantitative basis for decision-making in response to a contaminant event.

1.3.5. Step 5: Risk management

A major objective of a contaminant fate and transport study should be to develop a living guideline document that reservoir management can use to respond to a contaminant input to a drinking water reservoir. Modeling studies are useful because they can help in planning appropriate and timely responses to minimize short and long-term impacts on water quality. A model can also simulate emergency responses to minimize exposure risk to humans and reduce natural resource damage. Knowledge of contaminant plume behavior and travel time under various scenarios can guide in-situ measurements after contaminant entry and direct remedial actions. The contaminant response guidelines should include, but are not limited to, the following suggestions.

- A list of contaminant entry locations for the reservoir and associated contaminant types, with most vulnerable locations determined by conservative contaminant modeling studies, proximity to the drinking water intake, contaminant type, probability of occurrence, etc.

- Estimates of ranges in potential contaminant mass inputs based on potential contaminant properties and estimated volumes
- Ranges of contaminant travel times to the intake, depending various hydrodynamic, seasonal conditions, and contaminant properties
- Contaminant concentrations at the drinking water intake that correspond to ranges of potential contaminant masses and properties, specific contaminant entry locations, and hydrodynamic and seasonal conditions
- An evaluation of all management operational decisions that could be carried out and the impacts, if any, on contaminant travel time and maximum concentrations at the drinking water intake

In the event of a contaminant incident, the proactive modeling study and documentation is available to guide the emergency response. In addition to this, ideally emergency hydrodynamic and water quality modeling capabilities could be employed immediately following the incident to refine possible contaminant outcomes described in the guidance document based on known information about the actual contaminant and current reservoir conditions. Results from emergency model runs could be communicated to reservoir managers and water utilities (if modeling is done externally) in the forms of relevant tables and graphs with estimations of travel times and concentrations. However, the focus of the framework described in the manuscript is proactive hydrodynamic and water quality modeling prior to an event occurring. Training, communications, and reporting to decision-makers and the public in response to a contaminant incident are a part of the overall emergency response framework internal to a water utility.

1.4. Case study: Wachusett Reservoir

The Wachusett Reservoir, located in central Massachusetts, is a dimictic waterbody characterized by stratification, density currents from incoming tributaries, and varying spatial patterns in temperature and constituents. The reservoir supplies drinking water to up to 51 communities in the greater Boston and central Massachusetts area. The Quabbin Reservoir, completed in the 1930's, is located 65 miles west of Boston and is the most upstream reservoir in the system with a capacity of 1.6 billion m^3 (412 billion gallons). Periodically throughout the year water is transferred from the Quabbin about 30 miles east through the Quabbin Aqueduct to the Wachusett Reservoir, completed in 1908 with a volume of approximately 0.25 billion m^3 (65 billion gallons). Water from Wachusett Reservoir is withdrawn at the Cosgrove Intake and treated at the John J. Carroll Water Treatment Plant, from which it flows east to Boston. Treatment includes disinfection (ozone and UV primary; chloramines in distribution system), fluoride addition, and pH and water chemistry adjustment to prevent corrosion in the distribution system. The Wachusett Reservoir source water has a waiver from the filtration requirements of the US EPA Surface Water Treatment Rule (SWTR). The MWRA manages the treatment and distribution of the water while the Massachusetts Department of Conservation and Recreation (DCR) has the responsibility of managing the watersheds. Understanding the fate and transport of potential contaminants, both from external and internal loadings to the reservoir, is an important part of the management of water quality and source water protection.

1.4.1. Reservoir characteristics

The Wachusett Reservoir is the second largest water body in Massachusetts with a maximum depth of 36.6 m, a length of 13.5 km, and a surface area of 16.8 km^2. Inflows to the reservoir include precipitation, runoff, flow from nine tributaries, and water transferred from the Quabbin reservoir. The combined Quabbin and Wachusett reservoir system has a safe yield of approximately 13.1 m3/s (300 MGD). The largest tributaries, Stillwater and Quinapoxet, enter the reservoir from the northwestern end and contribute approximately 30 to 40% of the total annual inflow (Figure 1.2). Water from the Quabbin Reservoir is typically transferred to Wachusett from June through November into the mouth of the Quinapoxet River to help maintain water surface elevation (WSE) while meeting water demands and contributes from 30 to 60% of the annual inflow, depending on the year. Secondary transfer objectives include introducing water with lower organic content and generating hydropower. The major withdrawal from Wachusett is the Cosgrove drinking water intake, located at the eastern end of the reservoir, which supplies drinking water to metropolitan Boston and nearby towns. Water also leaves the reservoir through evaporation, minor withdrawals to a nearby town, and to supply the Nashua River through both a controlled sleeve valve release and a spillway.

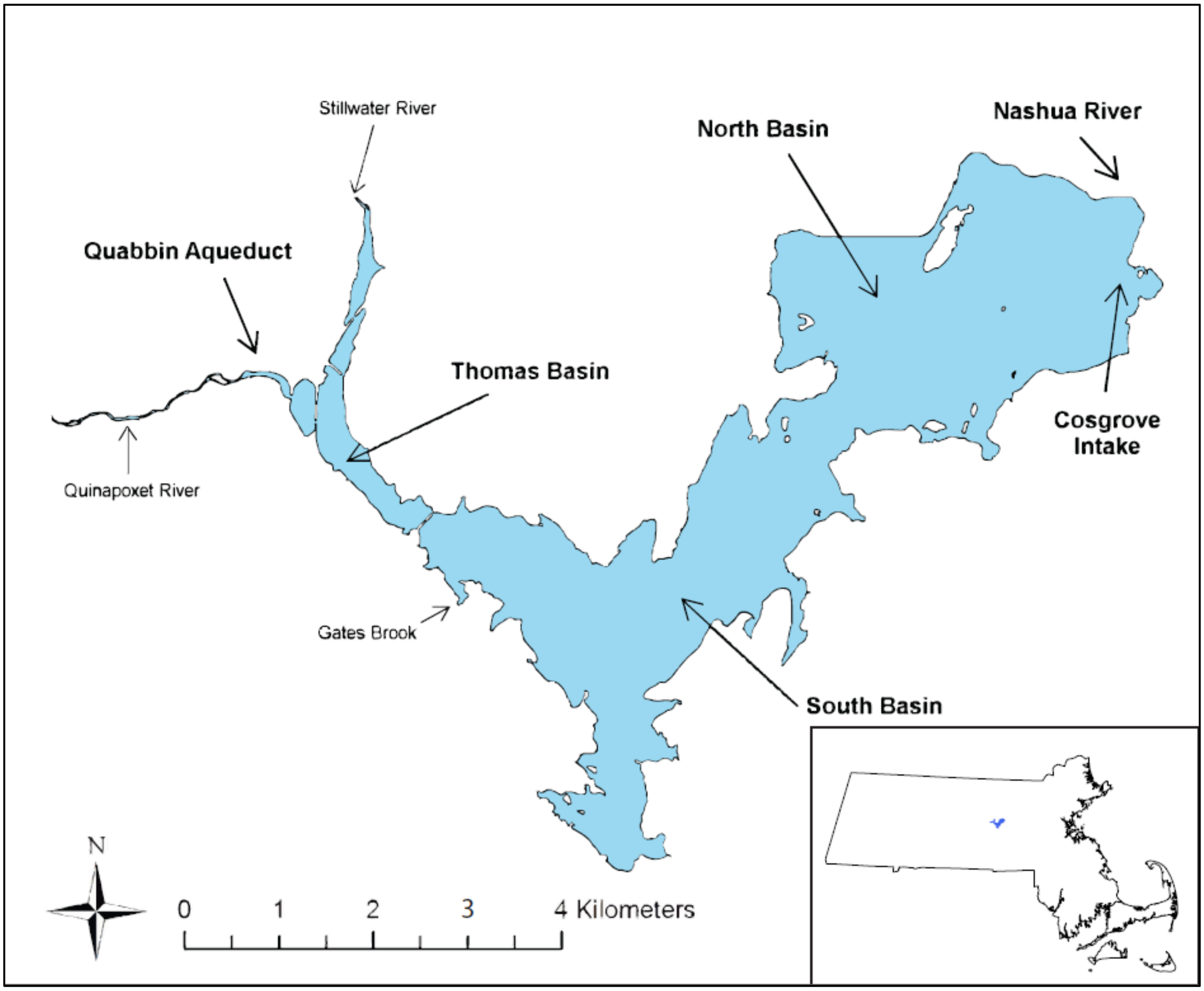

Figure 1.2 Wachusett Reservoir major features and locations of inflows/outflows

1.4.2. Potential contaminants

Water quality constituents may enter the Wachusett Reservoir via tributary inflows, direct runoff, groundwater inputs, shoreline litter, precipitation, animals, and accidental contaminant spills from roadways or railways. Microorganisms such as pathogenic bacteria, total and fecal coliform, *Giardia*, *Cryptosporidium*, and viruses are some examples of water quality constituents of concern. Fecal coliform are the primary indicator organism used to assess the presence of potential pathogenic bacteria and virus contamination associated with fecal matter in water supplies. The source of this type of contamination in Wachusett Reservoir could be from wildlife or a sewage overflow from a nearby pump station used to pump wastewater to a sanitary sewer and treatment system outside of the watershed. The concentrations of total and fecal coliform in raw wastewater are $10^7 – 10^9$ CFU/100 mL and $10^6 – 10^8$ CFU/100 mL, respectively (Metcalf and Eddy, 2003). The SWTR for unfiltered water supplies, such as Wachusett Reservoir, allow for no more than 10% of source water samples prior to disinfection (i.e. the Cosgrove Intake) over any six-month period to have more than 20 CFU/100 mL. Coliform loss in the reservoir can be caused by death in the natural environment as affected by general water quality, temperature and light, and by settling.

A contaminant may also enter the Wachusett reservoir via an accidental spill from a highway bridge or railroad line near the reservoir. A previous study discussed the fate and transport of a hypothetical spilled conservative contaminant into the Wachusett Reservoir, representing a worst-case concentration scenario and assuming no spill response intervention from reservoir operations staff (Jeznach et al., 2014). However, a contaminant may decay due to chemical degradation, volatilization, or microbial uptake, which may impact the arrival time and maximum concentration of the contaminant detected at the Cosgrove Intake. Mechanisms of degradation of a contaminant spill may include volatilization, microbial uptake, chemical reaction, and photochemical reactions.

1.4.3. Model development

The 2-D hydrodynamic and water quality model, CE-QUAL-W2 is an appropriate model for the Wachusett Reservoir because the length to width ratio is approximately 11 and because longitudinal and vertical gradients in the reservoir dominate lateral gradients (Ahlfeld et al., 2003). A 2-D model was also chosen for its relative simplicity and short model run time, compared to a 3-D model. CE-QUAL-W2 is a two-dimensional, longitudinal and vertical, hydrodynamic and water quality model developed at the U.S Army Corps of Engineers Waterways Experiment Station (USACE WES). The model has been used to simulate a combination of rivers, lakes, reservoirs, and estuaries (Cole and Wells, 2008). CE-QUAL-W2 directly couples hydrodynamics and water quality algorithms. To model water hydrodynamics and mass transport, CE-QUAL-W2 solves six laterally and layer averaged equations for water surface elevation (WSE), pressure, horizontal velocity, vertical velocity, constituent concentrations, and temperature/density using the finite difference method. The governing equations for the model, detailed by Cole and Wells (2008), are for horizontal-momentum, constituent transport, free WSE, hydrostatic pressure, continuity, and density. The original model was developed in 1975 by Edinger and Buchak and was known as LARM (Laterally Averaged Reservoir Model) (Cole and Wells, 2008).

The CE-QUAL-W2 modeling grid for the Wachusett Reservoir, pictured in Figure 1.3 (a) and (b), was modified from the original grid developed by Camp, Dresser, and McKee (CDM, 1995). The grid for this study consisted of 5 branches, and 64 laterally averaged segments each with up to 47 layers varying in thickness from 0.5 m to 1.5 m. The time step in the simulations was automatically determined to guarantee numerical stability, with a maximum time step of 1 hour. The ULTIMATE numerical solution scheme was used for all simulations. The Cosgrove drinking water intake is represented by segment 46 in the model grid and is modeled as a selective line sink at an elevation of 104.3 within layer 33.

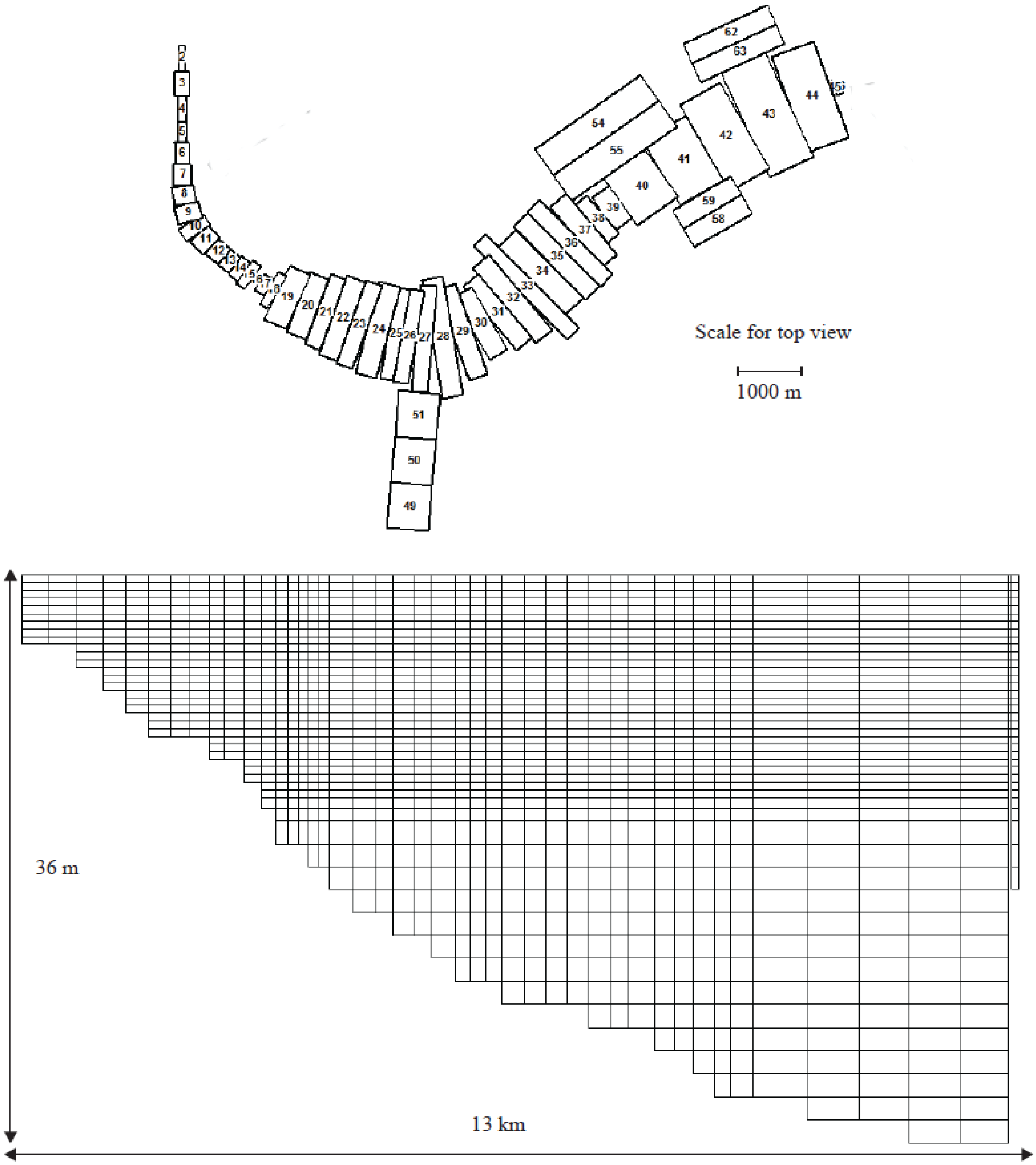

Figure 1.3 Wachusett Reservoir modeling grid plan view and side view

1.4.4. Input data and calibration

Modeled inflows to the Wachusett Reservoir included the Stillwater and Quinapoxet Rivers, seven minor tributaries, the Quabbin transfer, direct runoff, and precipitation. The Stillwater and

Quinapoxet Rivers are gaged for flow by the USGS and account for the drainage of approximately 73% of the watershed area. Flow from minor tributaries was estimated based on the daily watershed yield of the Stillwater River, as described in Tobiason et al (2002). The inflows to the Wachusett Reservoir from the Quabbin Reservoir were measured daily by the MWRA at the aqueduct outlet. Direct runoff was calculated based on the ratio of Stillwater daily discharge to Stillwater watershed area multiplied by the entire direct runoff area. Hourly precipitation data from the Worcester Regional Airport, approximately 10 miles southwest of the reservoir, were obtained from the National Oceanic and Atmospheric Administration (NOAA). Hourly meteorological data such as air temperature, dew point temperature, wind speed, wind direction, and cloud cover were also acquired from NOAA.

MWRA daily measured outflows from the reservoir included withdrawals from the Cosgrove Intake, discharge to the Wachusett Aqueduct, as well as releases and spillway discharges to the Nashua River at the Wachusett Dam. Direct withdrawals by the local town of Clinton were recorded daily by the MWRA. Evaporation loss was not directly measured but was calculated based on the algorithm used within CE-QUAL-W2 and work by Edinger et al. (1974). Inflow and outflow data for the calendar year were used to develop an annual water balance external to CE-QUAL-W2 to verify that daily calculated WSEs based on inflow and outflow volumes were within 0.15 m (0.5 ft.) of measured WSEs by the MWRA as discussed in Jeznach et al (2014).

Model input files were created using data for meteorology, bathymetry, water quality, initial flow and constituent conditions, outlet descriptions, and the adjusted inflow and outflow data. MWRA and DCR temperature and specific conductivity measurements from the Cosgrove Intake and the North Basin were compared with modeled values at segments 42 and 46, the approximate locations of the profile measurements collected at the North Basin and the Cosgrove Intake, respectively. The average absolute mean errors (AME) between modeled and measured temperature and specific conductivity at the drinking water intake were 3.2 ºC and 19.6 µS/cm, respectively. The average root mean square errors (RMSE) between modeled and measured temperature and specific conductivity profiles at the intake were 3.9 ºC and 22.2 µS/cm, respectively.

Figure 1.4 compares the temperature profile measurements (indicated by circles) in the North Basin with the model results (a solid line) in segment 42 on three different days in the year 2004. Temperature profiles were nearly uniform with depth during the early spring with a temperature of about 8 ºC; by late May the water surface temperature was beginning to increase due to increasing air temperatures. The timing and the extent of the thermocline development was accurately simulated during the summer months, with warmer temperatures on the top of the water column and cooler on the bottom. Average AME and RMSE between modeled and measured temperature profiles were 2.5 ˚C and 3.1 ˚C, respectively. When present, the depth and extent of the thermocline and the turnover were accurately simulated. Profiles during the summer and the fall have the highest RMSEs, most likely due to the difficulty of capturing the effect of the Quabbin transfer inflows because the water temperature is an estimation based on water temperature measured in the Quabbin Reservoir, 30 miles west, prior to travelling through the aqueduct to Wachusett.

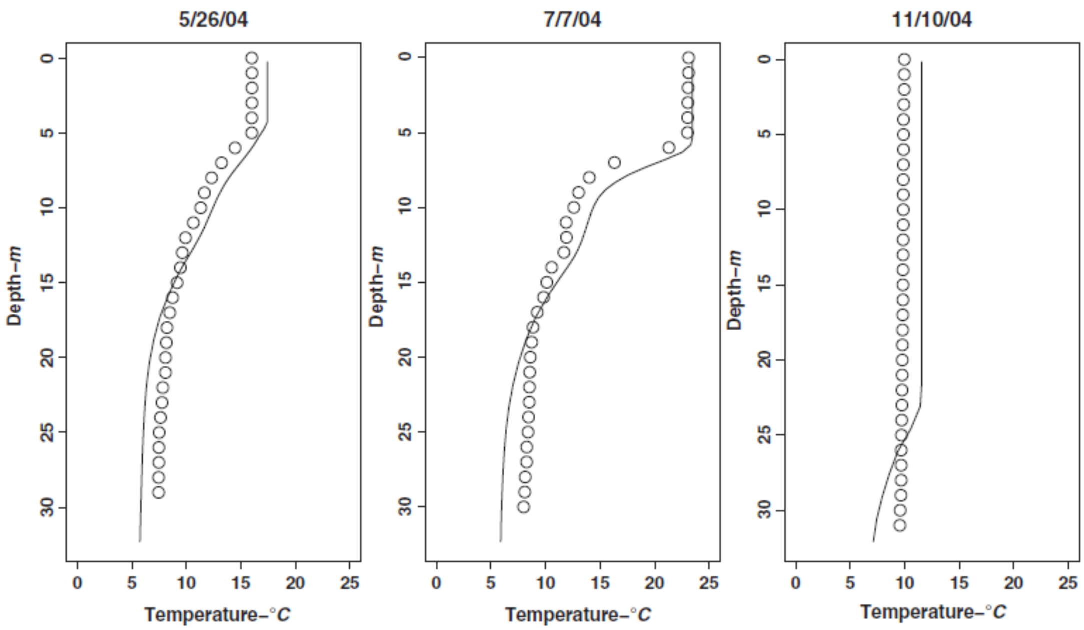

Figure 1.4 Temperature profiles in North Basin (Segment 42) for 5/26/04, 7/7/04, and 11/10/04

Specific conductivity measured in the North Basin of the reservoir and was used to calibrate the transport of non-reactive constituents in lieu of tracer studies since for this reservoir it is the only parameter that is significantly variable throughout the seasons and is measured daily or weekly in the inputs and outputs as well as in-situ. Specific conductivity data for the tributaries was collected weekly by the DCR and precipitation specific conductivity was collected from two nearby National Atmospheric Deposition Program (NADP) stations. Specific conductivity measurements were converted to total dissolved solids (TDS in mg/L) for model simulations based on a site specific relationship (0.6 times specific conductivity in μS/cm), since CE-QUAL-W2 does not model specific conductivity as a constituent (Tobiason et al., 2002).

Figure 1.5 compares specific conductivity profile measurements (circles) in the North Basin with model results (solid line) for segment 42 for three different days in 2004. Average AME and RMSE between modeled and measured specific conductivity profiles was 13 μS/cm and 15 μS/cm, respectively, and the model accurately simulated the specific conductivity in the epilimnion and hypolimnion throughout most of the year. The signature of the cooler and lower conductivity Quabbin transfer water was also evident in the profiles and is an important verification of the transport of conservative constituents by the model. Water from the Quabbin has lower specific conductivity (approximately 50 µS/cm) compared to Wachusett water (typically around 110 µS/cm) and is usually cooler than the Wachusett surface water, depending on the time of the year. The interflow is approximately 10 m thick and predominantly occurs 5 to 15 m below the surface of Wachusett (DCR, 2011). The Quabbin interflow is detected in Wachusett specific conductivity profiles by a region of lower specific conductivity located at approximately the same depth as the Wachusett thermocline. When the water reaches the Cosgrove Intake, the combination of Quabbin and Wachusett water is withdrawn and sent to the Carroll Water Treatment Plant to be disinfected and distributed. Further calibration and validation data from the year 2004 and other developed model years (2003, 2005-2009) is documented in Matthews (2007), Stauber (2009), Devonis (2011), and Clark (2013).

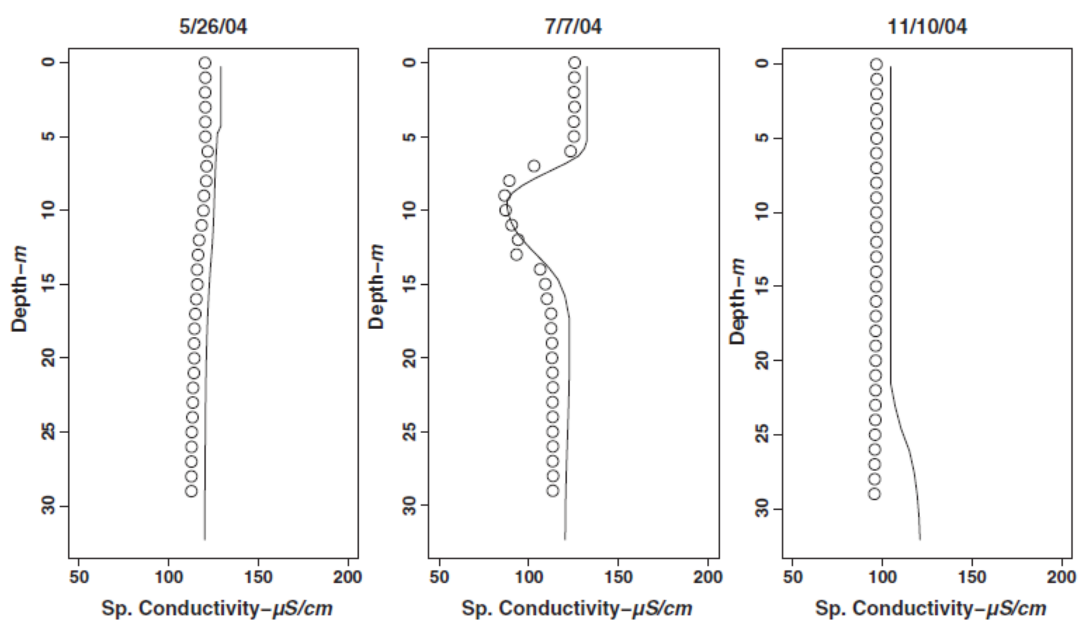

Figure 1.5 Specific conductivity profiles in North Basin (Segment 42)

1.4.5. Hypothetical contaminant spill scenarios

Modeled conservative contaminant spills into the reservoir, previously reported, provided valuable information to reservoir operators on the effects of contaminant density, season, the Quabbin transfer water, and wind on arrival time and concentration of a contaminant at the Cosgrove Intake (Jeznach et al., 2014). Model simulations highlighted the need for a fast management response to contain a contaminant spill if possible, as results indicated contaminant arrival at the drinking water in significantly less time than the mean hydraulic residence time of approximately 200 days. Simulating the movement of non-reactive or conservative constituents in the reservoir was important to understanding the impacts of a worst-case scenario, yielding the highest contaminant concentrations at the drinking water intake. However, in addition to being transported with water, many water quality constituents react, including decay, or degradation, yielding concentrations that differ from conservative constituents. Therefore, an understanding and assessment of the physical, biological and chemical processes in the Wachusett Reservoir that affect concentrations of potential contaminants can lead to improved management, response, and operational practices for maintaining water quality. Two constituents of interest to reservoir management are a large fecal coliform input from a sewage overflow into a tributary as well as an ammonium nitrate accidental spill from a railway (Matthews, 2007; Stauber, 2009).

For the sewage overflow simulations, an input of 180,000 gallons of raw wastewater with a fecal coliform concentration of 10^8 CFU/100 mL (a typical concentration in raw wastewater) occurred over 12 hours on Julian Day 170 (6/18/04), when the reservoir was not yet fully stratified. Previous research has shown that unstratified conditions in the spring and fall result in earlier conservative contaminant arrival times at the Cosgrove Intake compared to conservative contaminant spills occurring in the summer (Jeznach et al., 2014). Temperature dependent general decay, settling, and light induced decay, all pseudo first order decay mechanisms, were used to simulate the loss of coliform during transport through the reservoir. For these simulations, the Arrhenius temperature rate multiplier (θ) was 1.04, the first order decay rate at 20˚C for the fecal coliforms (k_{20}) was 0.75 day^{-1}, the settling rate (ω) was 0.29 m/day, and the proportionality constant relating irradiance-induced decay rate coefficient to irradiance (α) was 0.014 cm^2/cal. Values were chosen based on rates used in a study by Tobiason et al (1998) of the Quabbin Reservoir. A simulation was also completed with no decay in order to understand the transport of the coliforms throughout the reservoir.

A hypothetical spill of ammonium nitrate, commonly used in fertilizer, was simulated to occur on Julian day 230 (8/17/04) and included a mass input of 200,000 lbs of ammonium nitrate from segment 9 in CE-QUAL (Stauber, 2009). The stratified conditions in the summer months can result in contaminant concentrations up to 2 times greater than in the spring and fall for conservative contaminants (Jeznach et al., 2014). The spilled ammonium nitrate was assumed to be partially removed and contained within three hours, but over this time period, a tenth of the load was assumed to dissolve into the water. Ammonium and nitrate were modeled separately, due to their different first order uptake rates by plankton. Ammonium and nitrate were modeled as generic constituents in CE-QUAL-W2 with first order decay (uptake) rates of 0.13 day^{-1} and 0.023 day^{-1}, respectively.

The effects of different seasonal conditions as well as the impacts of Quabbin transfer operation on contaminant concentration at the Cosgrove Intake were investigated. A possible operational response to a contaminant spill in the Wachusett Reservoir could be to transfer, or increase the rate of transfer of water from the Quabbin Reservoir, since water from this reservoir plays a large role in the hydrodynamics and water quality in the Wachusett. Results from previous studies indicated that there is minimal impact in altering the Quabbin transfer during the spring and fall when the reservoir is essentially completely mixed, however, turning the transfer off in the summer when it is normally operating may reduce the variability of the contaminant concentration at the intake (Jeznach et al, 2014). Results also suggest that altering the transfer flow has a minimal impact on arrival time of a contaminant at the intake after a summer spill. A scenario investigated in this study included the Quabbin transfer operating normally in the summer (approximately 300 MGD), turned off for 30 days, and with increased flow of 400 MGD (17.51 m^3/s) for 30 days in response to the hypothetical ammonium nitrate contaminant spill.

1.4.6. Spill assessment

The calibrated and verified CE-QUAL-W2 model was used to simulate the fate and transport of hypothetical spills of fecal coliform and ammonium nitrate into the Wachusett Reservoir. The concentrations of these decaying constituents at the Cosgrove Intake were compared to those of a conservative water quality constituent.

1.4.6.1. Fecal coliform contamination

In the reservoir, fecal coliforms undergo general temperature dependent first order decay, light induced decay, and settling. Figure 1.6 shows simulated fecal coliform concentrations at the Cosgrove Intake as a result of a 180,000 gallon sewage spill with a 10^8 CFU/100 mL concentration of fecal coliforms on Julian Day 170 (6/18/04) (well mixed conditions) at segment 20. The figure illustrates the predicted concentrations if decay occurred from exposure to light, settling, or general temperature dependent decay. The results for no decay are not shown due to the difference in magnitude of concentration, which reached as high as 360 CFU/100 mL. The variable peaks and troughs in concentration at the intake were observed in the modeled coliform with no decay (not shown on figure) as well as for concentration with only light induced decay.

The peaks and troughs are consistent with results described in Jeznach et al. (2014) and are associated with contaminants becoming isolated in a vertical layer during the summer when the reservoir is stratified, leading to less vertical mixing and the withdrawal of alternating layers of high and low concentration at the intake. Peak concentrations for individual loss mechanisms of light, settling, and general decay were 2.65, 0.29, 0.007 CFU/100 mL, respectively. General, temperature dependent, first order decay was the predominant decay mechanism resulting in a 99.99% decrease in fecal coliform concentration at the intake when compared to no decay. Loss due to only light or only settling induced decay resulted in 99% and 99.9% reductions, respectively, in concentrations at the Cosgrove Intake compared to no decay.

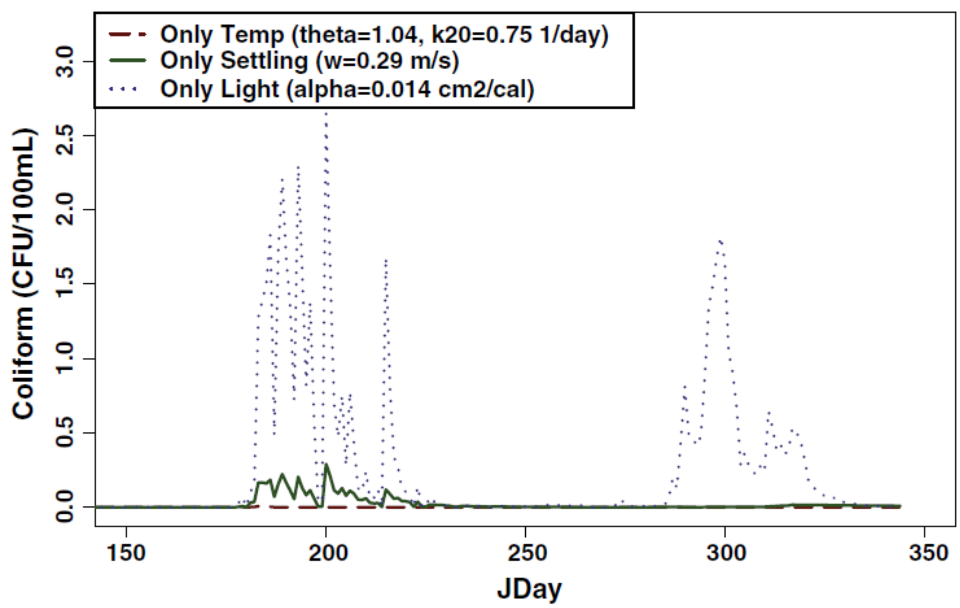

Figure 1.6 Fecal Coliforms at the Cosgrove as a result of wastewater spill at segment 20, JDay 170 (6/18/04)

Based on model simulations incorporating all decay mechanisms, decay coefficient values from literature, and for a sewage spill at segment 20, fecal coliform concentrations would not exceed the SWTR criteria of 20 CFU/100 mL at the Cosgrove Intake. Additional simulations with the same decay coefficients suggest that an inflow load of $7.0\text{x}10^9$ CFU/100 mL, 70 times larger than the simulated baseline concentration, discharged into the Wachusett for 12 hours would be necessary in order for the Cosgrove concentration to approach the SWTR criteria.

1.4.6.2. Ammonium nitrate contamination

Ammonium nitrate input from a spill would be subject to loss (decay) due to uptake by phytoplankton if sufficient phosphorous were present in the reservoir. Ammonium and nitrate were modeled individually with separate decay rates, representing different uptake rates by phytoplankton. The effects of decay on the decrease in constituent concentration were compared to the case of no ammonium nitrate decay. Ammonium and nitrate concentrations at the Cosgrove Intake for the decay scenarios were significantly lower than for the no decay condition. For a 20,000 lb dissolved spill of ammonium nitrate on Julian Day 230 (8/17/04) during stratified conditions, peak concentrations at the Cosgrove Intake were 0.075 mg/L nitrate and 0.002 mg/L ammonium when decay was included, while for no decay, peak concentrations were 0.125 mg/L nitrate (note that the Maximum Contaminant Level for nitrate is 10 mg/L) and 0.025 mg/L ammonium.

Figure 1.7 shows the relative concentrations of ammonium and nitrate at the Cosgrove Intake (simulated concentrations with decay divided by concentrations for a conservative contaminant) for a spill occurring on Julian Day 230 (8/17/04) from segment 9.

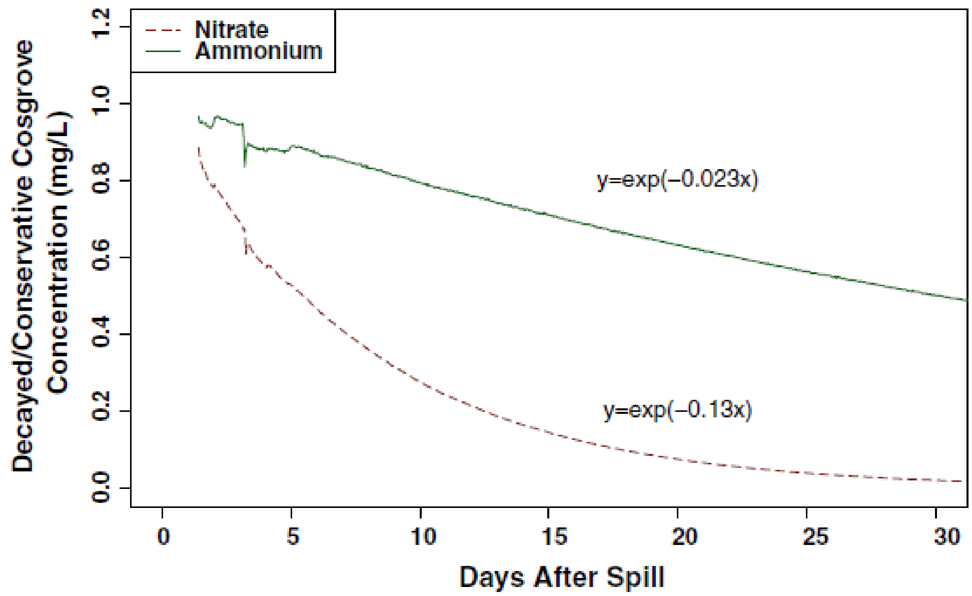

Figure 1.7 Comparing decay rates for spill of ammonium nitrate at segment 9 JDay 230 (8/17/04)

The figure illustrates a trend of exponential decay in the reservoir for ammonium and nitrate and the modeled first order rates of decay reflect the first order reaction uptake rates of 0.13 day^{-1} and 0.023 day^{-1} for ammonium and nitrate, respectively. Using a first order exponential decay expression (Equation 1.1) and the conditions simulated in this study, results for conservative constituent concentrations could be used to determine concentrations of reactive constituents at the Cosgrove Intake for the conditions, without performing a simulation using the generic reactive term in the model,

$$\frac{Concentration\ With\ Decay}{Conservative\ Concentration} = \exp(-k * t) \qquad \text{Equation 1.1}$$

Where k is the first order decay rate (day^{-1}) and t is the number of days after the spill.

The exponential expression could also be used to determine the decay rate required to decrease the maximum relative concentration at the Cosgrove Intake by 99%. For example, if the peak concentration of ammonium nitrate was approximately 0.125 mg/L and arrived in 23 days, a decay rate of 0.20 day^{-1} would be needed to decrease the peak concentration by 99% or to a concentration of 0.001 mg/L.

1.4.6.3. Seasonal impacts

Seasonal and stratification impacts that were evident for conservative contaminant simulations, as discussed by Jeznach et al. (2014), were also evident for decaying contaminant simulations from this study. In the spring and the fall, the reservoir was essentially completely mixed and uniform, as indicated by temperature and specific conductivity profiles. This condition generally leads to earlier arrival times and lower concentrations of contaminants at the Cosgrove drinking water intake, consistent with results from Jeznach et al. (2014).

The effects of the date during the year of a wastewater overflow on fecal coliform were evaluated. Figure 1.8 illustrates the fecal coliform concentrations (including general temperature, light, and settling decay) at the Cosgrove Intake for spills originating from Gates Brook and occurring on four different days during the spring and the fall of 2004.

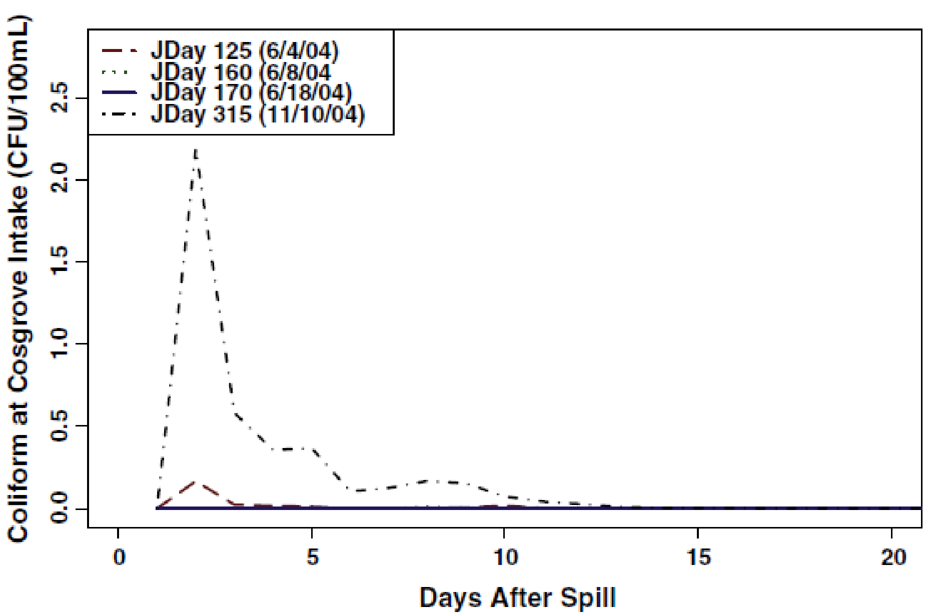

Figure 1.8 Effects of variations of date of wastewater spill into segment 20 at Cosgrove Intake

On Julian Day 125 (5/4/04) the reservoir was becoming stratified and the Quabbin transfer was not yet initiated. On Julian Days 160 and 170 (6/8/04 and 6/18/04), the reservoir was stratified and the Quabbin transfer was operating to maintain the WSE. The spill simulation on Julian Day 315 (11/10/04) was when the reservoir was completely mixed and there was no Quabbin transfer water entering the reservoir. Based on the results in Fig. 8, coliforms from a spill occurring in June were essentially non detectable at the Cosgrove Intake. However, fecal coliforms dispersed throughout the reservoir faster when the reservoir was completely mixed in early spring and fall, producing detectable fecal coliform concentrations at the intake, especially for the November spill. The cooler water temperatures and low sunlight in the late fall inhibits decay, resulting in greater fecal coliform concentrations at the intake (about 2 CFU/100 mL) compared to spills that occurred in the summer. An early spring spill may also be diluted by spring runoff and elevated tributary discharges.

1.4.6.4. Quabbin transfer impacts

In the event of a conservative contaminant spill into the reservoir, Quabbin transfer may not have a significant impact on the arrival time of a contaminant but it may help to decrease peak contaminant concentrations at the drinking water intake (Jeznach et al., 2014). The current study indicates that the impact of the Quabbin transfer on decaying contaminants is consistent with these results. Figure 1.9 shows ammonium and nitrate concentrations at the Cosgrove Intake for a summer spill on Julian Day 230 (8/17/04) for three different Quabbin transfer operation scenarios for 30 days after the spill occurred.

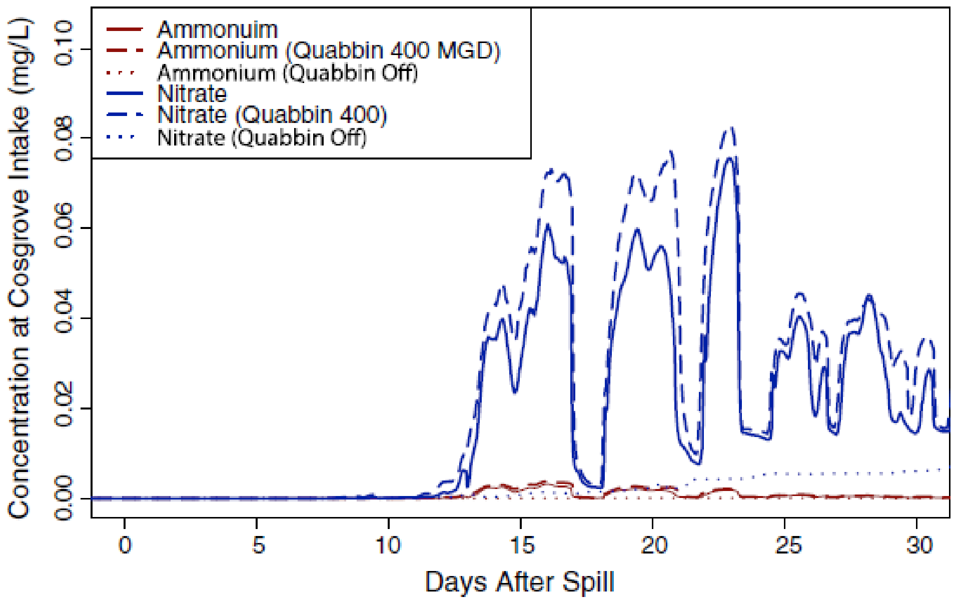

Figure 1.9 Ammonium nitrate concentration at the Cosgrove after increasing and turning off the Quabbin transfer (spill occurring on JDay 230)

Decreasing the Quabbin transfer to zero decreased the peak arrival concentration by 90%. Increasing the Quabbin transfer from the normal flow of about 300 MGD to 400 MGD in the summer had a negligible effect on the contaminant arrival time but increased peak concentrations by about 0.01 mg/L at the intake. Specific conductivity and tracer profiles have shown that the Quabbin transfer signature was evident in the water column, just lower in magnitude, when the transfer operation was disrupted and turned off for 30 days. Results suggest that even if the Quabbin transfer was altered for a month in response to a contaminant spill, there would be little impact on arrival time because the volume of water introduced via the Quabbin relative to the volume of water already circulating in the reservoir is small. Therefore, based on studies to date, the best response in regards to the Quabbin transfer in the event of a contaminant spill (decaying or conservative) is to turn off the transfer to potentially decrease peak concentrations at the Cosgrove Intake.

1.4.6.5. Management response/risk management

For an 180,000 gallon wastewater overflow into the reservoir with a 10^8 CFU/100 mL concentration of fecal coliforms, rapid loss by temperature, light, and settling prevented coliform concentrations at the drinking water intake from exceeding the SWTR of 20 fecal coliforms per 100 mL. Results from a modeled 200,000 lb spill of ammonium nitrate into the reservoir indicated that concentrations at the intake were reduced by up to 60% when loss due to plankton uptake was included in the simulations. Peak concentrations at the Cosgrove Intake were 0.075 mg/L nitrate and 0.002 mg/L ammonium.

An accidental release of a decaying contaminant into the reservoir would be of concern to management based on the modeled short arrival times at the intake (between 2 – 15 days for all seasons) and the varying rates of loss for different contaminants. Simulations suggest that contaminant spills into the Thomas Basin occurring in the spring and fall, when the reservoir is completely mixed, result in arrival of contaminants at the drinking water intake earlier than spills occurring in the summer months. However, concentrations of contaminants at the intake are greater and more variable during the summer months. A possible operational response to a contaminant spill could be to turn the Quabbin transfer on or off, depending on its state at the time of a spill. Simulations suggest that turning the Quabbin transfer water off in the event of a summer spill can reduce the variability in contaminant concentrations at the intake.

Information from this study, in conjunction with results presented in Jeznach et al (2014), has been used to develop guidelines for reservoir management to follow in the event a contaminant scenario arises. A confidential reservoir management document outlines a process in which to systematically and scientifically evaluate the severity of a situation and respond to a scenario based on contaminant mass, volume, density, miscibility, environmental fate (if known), and season of occurrence. In the event of a contaminant incident, there is capability to run emergency hydrodynamic and water quality modeling simulations to refine guidance document estimations of contaminant travel time and concentrations, based on known approximate contaminant characteristics and current conditions, which can then be communicated to the water utility theoretically within several hours.

1.5. Conclusions

In this article, a framework is presented for using hydrodynamic and water quality models to assess contaminant impacts on reservoir water quality. The framework follows a set of steps based on personal experience modeling contaminant impacts on reservoir drinking water quality and working with reservoir management to develop appropriate risk management responses and guidelines.

A case study assessment of the Wachusett Reservoir in central Massachusetts provides an example of an application of the framework and how it can be used to quantitatively and scientifically guide management in response to varieties of contaminant scenarios. In this study, after model development and calibration, two potential contaminant scenarios are investigated. A hypothetical fecal coliform input from a sewage overflow and an accidental ammonium nitrate spill from a tanker truck were analyzed, including the impacts of decay rates, season, and inter-reservoir transfers on contaminant concentrations at the drinking water intake. The modeling study highlights the importance of a rapid operational response by managers to contain a contaminant spill in order to minimize the mass of contaminant that enters the water column, based modeled reservoir hydrodynamics.

The framework and case study presented in the article present a useful approach to assessing contaminant impacts and management responses on surface drinking water sources prior to an event occurrence. While protecting source water quality is the most important safeguard against drinking water degradation, in the event of an emergency situation, prior detailed modeling efforts and scenario evaluations allow for an understanding of contaminant plume fate and transport, including potential maximum concentrations that could occur at the drinking water intake and contaminant travel time to the intake. It is recommended that hydrodynamic and water quality models be developed for all surface water bodies where the potential for contaminant entry exists in an effort to guide management response and remediation.

1.6. Acknowledgements

This project was supported by the Massachusetts Department of Conservation and Recreation (DCR) and the Massachusetts Water Resources Authority (MWRA). The findings are the opinions of the authors and do not represent the official findings of the DCR or MWRA.

CHAPTER 2

FUTURE CLIMATE EFFECTS ON THERMAL STRATIFICATION IN THE WACHUSETT RESERVOIR

Authors

Jeznach, L.C., Tobiason, J.E., 2015. Future Climate Effects on Thermal Stratification in the Wachusett Reservoir, JOURNAL - American Water Works Association , Vol 107, No. 4, pp. E197-E209.

2.1. Abstract

A two-dimensional hydrodynamic and water quality model, CE-QUAL-W2, was used to simulate the effects of increasing future air temperatures on water temperature, stratification timing and duration, as well as changes in winter ice cover in the Wachusett Reservoir, a major drinking water supply for metropolitan Boston. Historic model years 2003-2012 provided a framework for future synthetic climate scenarios to evaluate the sensitivity of the reservoir thermal processes over 100 years. Average epilimnion and hypolimnion temperatures increased by about 12% and 7%, respectively, after 100 years of increasing air temperatures. Stratification duration increased by 1-2 weeks, beginning earlier and ending later than the historical stratification period. Additionally, the average number of days with ice cover decreased 18-57% by the end of all scenarios. Results from this study provide insight into the sensitivity of Wachusett Reservoir water temperatures and the potential impacts of increasing air temperatures due to climate change.

2.2. Introduction

The impact of climate change on water resources is a topic of concern and interest to water managers around the world, as many water bodies are already exhibiting responses to an estimated rise in global surface temperatures of approximately 0.7ºC since the start of the 20th century (World Meteorological Organization (WMO), 2014). Climate change concerns broadly include increasing short term extreme events, and long term changes in rainfall patterns, drought, and surface and air temperatures. The implications of these meteorological changes on surface water bodies includes changes in the thermodynamic balance across the air and water interface, the timing and volume of stream water inflows, and the amount of wind energy inputs the system experiences. Meteorology is the driving force for lake internal heating, cooling, mixing, and circulation, which are processes that then have an effect on many other ambient water quality parameters.

A body of water exchanges heat mainly at the air-water interface by short wave radiation, long wave radiation, conduction, and evaporation (Edinger et al., 1968). The physical processes governing water temperature include vertical mixing, attenuation of penetrating radiation, and stratification. The temperature response of a waterbody is not immediate, and the speed of the response is slower with increasing depth into the water column. The top of the water column (epilimnion) exhibits a faster response, corresponding with short term meteorological changes over a 24 hour period, and this is usually not apparent further down in the water column. Water temperatures of the epilimnion and metalimnion reflect seasonal changes in air temperature. The lower portion of the water column (hypolimnion) is shielded from much of the seasonal meteorological variability and shows a slower response to increasing air temperatures, with the strongest response exhibited during turnover in temperate lakes (Livingstone, 1993).

The sensitivity of a water body to climate changes can be modeled with a physically based process model that simulates the seasonal responses to meteorological forcing (Komatsu et al., 2007; Lee et al., 2012; Lee et al., 2012; Sahoo and Schladow, 2008; Sahoo et al., 2011; Sahoo et al., 2013; Samal et al., 2012). This type of model has become an important tool for exploring a water body's response to single or multiple climate stressors, which may change independently or dependently at different rates. Of all the meteorological drivers, air temperature has the most significant effect on water temperature variability (Henderson-Sellers, 1988). An understanding of the sensitivity of thermal aspects of a water body to changes in air temperature is the key to understanding how climate change may impact water quality, since water temperature affects processes such as seasonal dissolved oxygen distribution, nutrient cycling, rates of reactions, and algal dynamics.

Many modeling studies have used future climate projections from General Circulation Models (GCM) based on International Panel on Climate Change (IPCC) emission scenarios (typically A1B, A2, and B1) to provide the meteorological forcing for future model scenarios (Fang and Stefan, 1998; Fang and Stefan, 1999; Fang and Stefan, 2009; George et al., 2007; Komatsu et al., 2007; Lee et al., 2012; Sahoo and Schladow, 2008; Sahoo et al., 2011; Samal et al., 2012). Future impact studies will move toward using the IPCC representative concentration pathways to represent a broad range of future greenhouse gas concentrations and climate outcomes to provide forcing to GCMs. Downscaled GCM projections can be used as meteorological inputs to a watershed model to generate future streamflows which are then used as inputs, in conjunction with the meteorological data, to a hydrodynamic process-based model to simulate a water body of interest. One dimensional (1-D) and two dimensional (2-D) hydrodynamic models are commonly used. For example, Samal et al. (2012) used GCM values of mean daily air temperature, wind, and solar radiation projections for 2081 – 2100 to produce change factors that were applied to a 39 year record for a watershed model and 1-D lake model of Cannonsville Reservoir in New York. A 2-D reservoir water quality model was developed by Komatsu et al

(2007) using inputs from the A2 GCM scenario and a runoff model to evaluate the sensitivity of the Shimajigawa reservoir to long term effects of climate change on water quality and the aquatic ecosystem. Simulations suggested that increasing air temperatures due to climate change will increase epilimnion and hypolimnion water temperatures, and lead to earlier thermal stratification periods, deeper thermoclines, and later turnovers (Fang and Stefan, 1999; Hondzo and Stefan, 1993; Komatsu et al., 2007; Samal et al., 2012). Greater differences between epilimnion and hypolimnion water temperatures results in more stable stratification and the gradual decrease of deep mixing events in monomictic and facultatively dimictic lakes, so much so that in some cases water bodies are unable to overcome density differences and no longer become completely mixed during cooler months (Peeters et al., 2002; Sahoo and Schladow, 2008; Sahoo et al., 2011). Warmer water temperatures also delay ice formation, decrease the number of days with ice cover, and reduce maximum ice thickness in the winter months (Fang and Stefan, 1998; Fang and Stefan, 2009).

Climate change impacts on water temperatures consequentially have an effect on many other water quality processes within a water body, many of which have been modeled and observed in previous studies. Warmer water can lead to an increase in the depth of the anoxic layer in the hypolimnion during the open water season but may increase the minimum under-ice dissolved oxygen (DO) values by up to 8 mg/L in lakes over the contiguous US (Fang and Stefan, 2009; Komatsu et al., 2007). Changes in the timing and duration of stratification will likely impact nutrient cycling and anaerobic conditions in the hypolimnion. Increased transport of nutrients, particularly soluble reactive phosphorous and ammonium nitrogen, from the sediments to the epilimnion during turnover, could lead to greater biological activity (algae) (Komatsu et al., 2007; Sahoo et al., 2013). Higher water temperatures along with changes in the timing, delivery, and amount of nutrients in the water column may lead to increased frequency and earlier occurrence of algal blooms in the future (Komatsu et al., 2007; Peeters et al., 2002). Harmful blooms of toxin producing cyanobacteria are a major concern for many drinking water resources in the future, since these microorganisms can take advantage of earlier onset, later turnover, and more intense vertical stratification as well as changes in anthropogenic nutrient loading, internal cycling, and increased atmospheric CO_2 supplies (Paerl and Paul, 2012). Climate change impacts on the thermal stratification and water quality of water bodies are important to evaluate on a case by case basis, since all water bodies will respond differently due to different hydrology, geography, and operational rules.

2.2.1. MWRA/DCR system

The Quabbin and Wachusett Reservoir System is the drinking water source for 51 communities in the Boston metropolitan area and Central Massachusetts (approximately one-third of the Massachusetts population). The Massachusetts Water Resources Authority (MWRA) is responsible for the delivery and distribution of the water to the communities while the Massachusetts Department of Conservation and Recreation (DCR), Division of Water Supply and Protection, manages the watersheds surrounding the reservoirs. The Wachusett Reservoir was filled in 1908 and has a volume of approximately 202,678 acre-ft (0.25 billion m^3 or 65 billion gallons). By the 1930's Boston's increasing demand for water spurred the development of the Quabbin Reservoir, 30 miles west of Wachusett and 65 miles west of Boston. The reservoir was completed in the 1930's and has a capacity of approximately 1,297,141 acre-ft (1.6 billion m^3 or 412 billion gallons), resulting in a combined reservoir system safe yield of approximately 13.1 m^3/s (300 MGD). Periodically throughout the year, water is transferred from the Quabbin Reservoir about 30 miles east through the Quabbin Aqueduct to the Wachusett Reservoir, where it is then withdrawn at the Cosgrove drinking water intake, treated at the John J. Carroll Water Treatment Plant, and sent to Boston. Water withdrawn at the intake is treated by disinfection, fluoride addition, and pH and water chemistry adjustment to prevent corrosion in the distribution system.

The Wachusett Reservoir, the focus of this study, is the second largest water body in Massachusetts with a maximum depth of 36.6 m, a length of 13.5 km, and a surface area of 16.8 km^2 (Table 2.1). Approximately 30 to 60% of the annual inflow to Wachusett is from the Quabbin Reservoir, allowing for the water surface elevation (WSE) to be highly managed and typically maintained throughout the year between the elevations of 118.9 m (390 ft) and 119.3 m (391.5 ft). When the reservoir is iced over in the winter months the lower bound is 118.3 m (388 ft). Secondary objectives for transferring water from Quabbin to Wachusett include introducing water with lower natural organic matter content and generating hydropower.

Table 2.1 Wachusett Reservoir characteristics

Characteristic	Value
Length	8.4 (13.5 km)
Max width	1 mi (1.6 km)
Surface area	6.3 sq mi (16.3 km^2)
Max depth	120 ft (37 m)
Volume	65 x 10^9 gal (0.25 km^3)
Mean water age	~ 200 days
Watershed area	108 mi^2 (280 km^2)

Other inflows to the Wachusett Reservoir include precipitation, runoff, and flow from nine tributaries, as indicated on Figure 2.1. The largest tributaries, the Stillwater and Quinapoxet, enter the reservoir from the northwest and contribute approximately 30 to 40% of the total annual inflow. The Cosgrove Intake, located at the eastern end of the reservoir, is the major withdrawal from Wachusett but water also leaves through evaporation, direct withdrawals to the nearby town of Clinton, as well as releases and spills to the Nashua River.

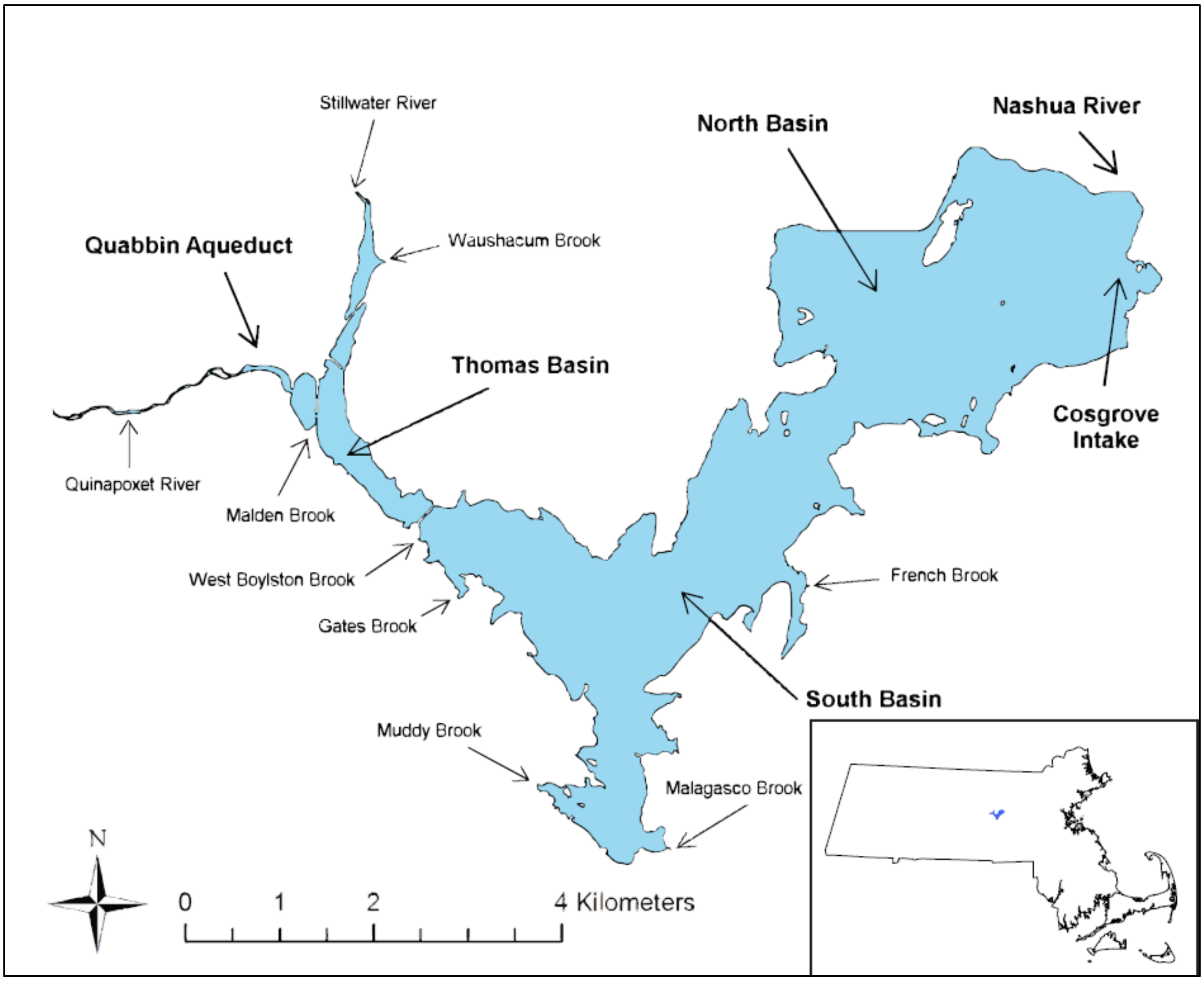

Figure 2.1 Wachusett Reservoir major inflows and outflows

The impacts of climate change on the Quabbin and Wachusett system have been investigated previously, but from the perspective of water quantity (e.g. safe yield) and not water quality (Kirshen and Fennessey, 1995; Pica, 2012; Yates and Miller, 2010). A study done by Kirshen and Fennessey (1995) showed serious decreases in reservoir safe yield during some future GCM climate scenarios but increases in safe-yield during other scenarios, when increases in temperature were offset by increases in precipitation and streamflow. The contradictory results of the GCM's indicated the uncertainty of predicting climate change impacts on the system. Pica (2012) used an ensemble of 112 GCM projections to evaluate climate change impacts on performance metrics such as reliability and safe yield. In general, the study showed an increase in safe yield with the future scenarios as air temperature and precipitation are generally predicted to increase for the region (Pica, 2012). However, the variability in the GCM projections sometimes produced opposite results, indicating decreased safe yield in the future. A study by Yates and Miller (2010) indicated that the Wachusett and Quabbin system was able to maintain safe yield and moderate increases in safe yield in most future climate scenarios, due to the large over-year storage of the water in the system. Climate change impacts on water quality parameters in the reservoir have not been previously investigated, and given the sometimes contradictory results of previous studies, investigating and planning for multiple future sets of conditions is warranted.

For this study, the sensitivity of reservoir water temperature, thermal stratification behavior, and ice cover were analyzed by simulating a range of possible future air temperature scenarios 100 years into the future. Comparisons of the future scenarios to a baseline scenario (no change from historic air temperatures for the years 2003-2012) were used to evaluate trends in reservoir thermal processes, such as stratification onset, duration and turnover during the open water season, and the onset, growth, and melting of ice during the winter months. An understanding of the sensitivity of reservoir thermal processes is an important first step to determining other water quality characteristics impacted by water temperature and future changes in reservoir process drivers.

2.3. Model development

2.3.1. Model description

For this study, the two-dimensional longitudinal and vertical, hydrodynamic and water quality model, CE-QUAL-W2 (Version 3.7), was used to simulate historical data and future climate change scenarios for the Wachusett Reservoir (Cole and Wells, 2008). The model is best used for relatively long and narrow water bodies that have longitudinal and vertical water quality gradients because lateral variations in velocities, temperatures, and constituents are assumed to be negligible. The model has been used to simulate a combination of rivers, lakes, reservoirs, and estuaries around the world (Cole and Wells, 2008). The model was a logical choice for this study because of its extensive calibration, verification, and application in many previous studies of the Wachusett Reservoir (Buttrick, 2005; Camp et al., 1995; Devonis, 2011; Jeznach et al., 2014; Joaquin, 2001; Matthews, 2007; Sojkowski, 2011; Stauber, 2009).

The modeling grid for Wachusett Reservoir, shown in Figure 2.2, was modified from the grid developed by Camp, Dresser, and McKee (CDM, 1995). The bathymetry of the reservoir was represented by 5 branches, made up of 64 laterally averaged segments with up to 47 layers varying in thickness from 0.5 m to 1.5 m. Segment 46 represented the Cosgrove Intake, where water was withdrawn using a selective withdrawal algorithm. The intake was modeled as a selective line sink at an elevation of 104.3 m, within layer 33 and layer 35 was the bottom layer from which selective withdrawal could not occur. The actual intake has multiple intakes, however the shallower intake is typically used and was included in the model.

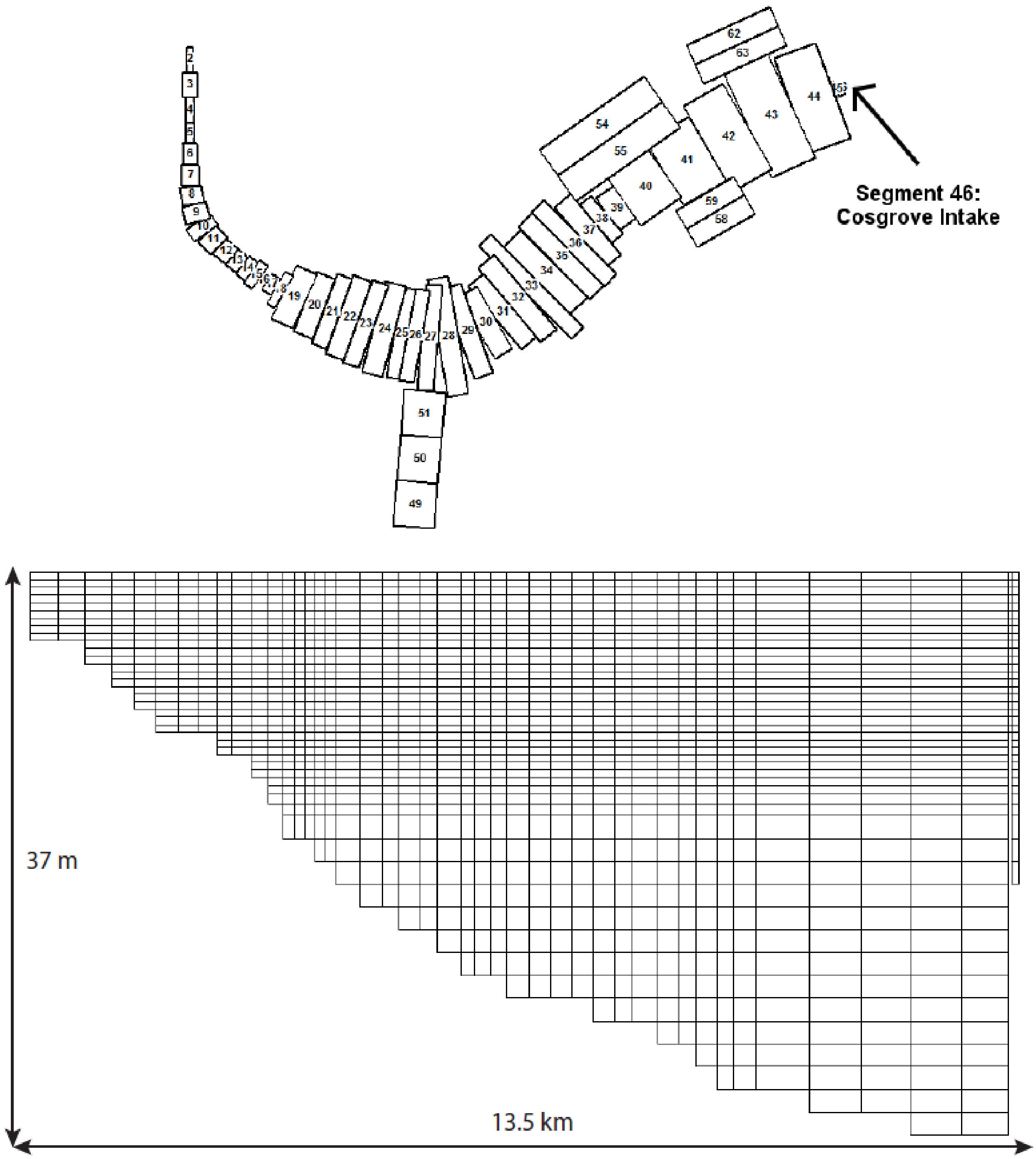

Figure 2.2 The plan view and side view of the reservoir modeling grid

2.3.2. Input data

Historical data (2003-2012) available for this study were used to drive all model simulations and included measured time series of daily reservoir inflows and associated temperatures and specific conductivity, meteorological data (hourly air temperature, dew point, solar radiation, wind speed, and wind direction) and daily outflows from withdrawals, spills, and releases. The Stillwater and the Quinapoxet Rivers, the two largest tributaries, are gaged for flow by the United States Geological Survey (USGS) and account for the drainage of approximately 73% of the watershed area. Flow from the ungaged minor tributaries was estimated based on the daily watershed yield of the Stillwater River, as described in Tobiason et al (2002). Inflows from the Quabbin Reservoir were measured by the MWRA at the Quabbin Aqueduct. Direct runoff was estimated based on the ratio of Stillwater daily discharge to Stillwater watershed area multiplied by the total direct runoff area. Hourly meteorological data, including precipitation, for the Worcester Regional Airport, approximately 10 miles southwest of the reservoir, were obtained from the National Oceanic and Atmospheric Administration (NOAA). Daily outflows from the reservoir were recorded by the MWRA.

2.3.3. Historical simulation

Model input files for the years 2003-2012 were created using data for meteorology, bathymetry, inflow and outflow, water quality, initial flow and constituent conditions, and outlet descriptions. Uniform temperature and constituent initial conditions were applied to the reservoir on January 1, 2003 (Julian day 1 of the model simulation) because the reservoir was assumed to be completely mixed on this day of the year. Measured WSE and in-situ profiles for temperature and specific conductivity were compared to modeled WSE and profiles from the years 2003-2012. Modeled WSE was within 0.15 m (0.5 ft) of the MWRA measured WSE.

Figure 2.3 compares the temperature profile measurements (indicated by circles) in the North Basin with the model results (a solid line) in segment 42 on three different days in the year 2008, as an example profile comparison. Temperature profiles were nearly uniform in the winter months through the early spring and became stratified into the three layers (epilimnion, thermocline, and hypolimnion) around May as the surface water temperature began to increase in response to increasing air temperature. Cooler weather in the fall decreases the epilimnion temperature, reducing the density difference between the epilimnion and the hypolimnion until the water column becomes unstable. In the processes called turnover, cooler water from the epilimnion plunges down in the water column mixing with the hypolimnion water and wind helps to mix the entire reservoir. The timing and the extent of the thermocline development and turnover was accurately simulated and temperatures were, on average, within 1.4˚C of the measured data throughout the 10 years.

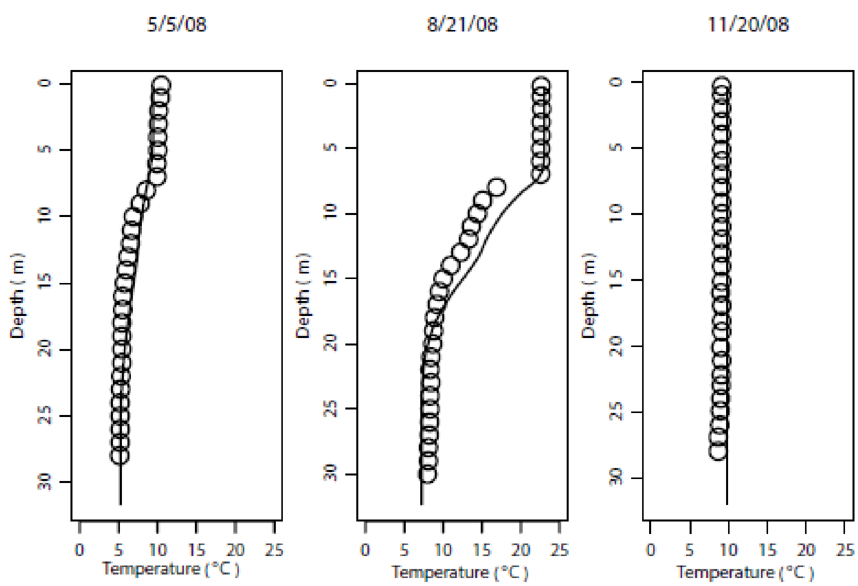

Figure 2.3 North Basin (segment 42) water temperature profiles for three days in 2008

Specific conductivity profile measurements were used to calibrate and validate the transport of the non-reactive constituents in the model. Since specific conductivity is not simulated by the CE-QUAL-W2 model, measurements were converted to total dissolved solids (TDS) concentrations, which are included as a model water quality parameter, using a site specific relationship for a Wachusett Reservoir tributary (Tobiason et al., 2002). This relationship, as expressed in Equation 2.1, can be used if it is assumed that TDS in water consists mainly of inorganic ions that conduct electricity and that there is a constant relative ionic composition in the reservoir throughout the simulations.

$$TDS\left(\frac{mg}{L}\right) = 0.6 \times Specific\ Conductivity\ (\frac{\mu S}{cm}) \qquad \text{Equation 2.1}$$

Figure 2.4 compares specific conductivity profile measurements (circles) in the North Basin with model results (solid line) for segment 42 for three different days in 2008 The model accurately simulated the specific conductivity in the epilimnion and hypolimnion throughout most of the years and model results were on average within 15 µS/cm of the measured values. Water that was transferred from the Quabbin Reservoir was typically lower in specific conductivity and cooler than the Wachusett Reservoir surface temperature. The transfer exists as an interflow density plume approximately 10 m thick and about 5 to 15 m below the water surface of the Wachusett Reservoir. The signature of the interflow water was evident in the specific conductivity profiles when the Quabbin transfer was operating and can be seen in Figure 2.4.

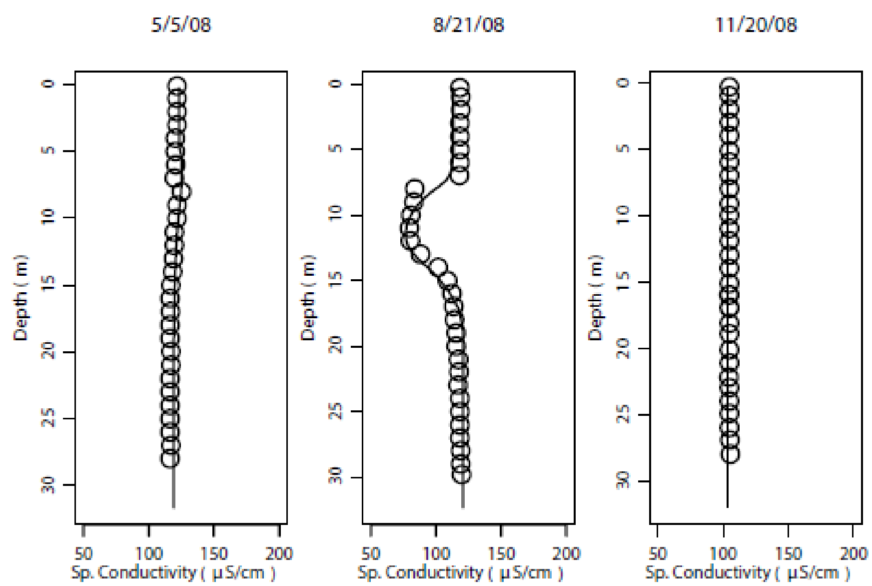

Figure 2.4 North Basin (segment 42) specific conductivity profiles for three days in 2008

2.3.4. Future climate scenarios

Downscaled annual average temperature and precipitation from 112 GCM projections for three different IPCC emission scenarios (A1B, A2, and B1) over the Quabbin and Wachusett watersheds are discussed in Pica (2012). The GCMs generally suggest increases in annual average temperature and precipitation for this region in the future, although the variability of these projections with increasing projected future time also increases. In general, the average annual temperature is projected to increase by about 1.5˚C by 2030, 2.5˚C by 2050, and nearly 3˚C by 2070 (Pica, 2012). The average annual precipitation is expected to increase 5% by 2030, 7% by 2050, and almost 8.5% by 2070 (Pica, 2012). The increase in temperature and precipitation would affect the amount and timing of streamflow in the region. Increased precipitation would likely result in increased streamflows, but higher temperatures may impact the timing of these flows and the amount of flow lost to evapotranspiration. GCMs are commonly used as a starting point to drive climate assessments of water bodies. Downscaled meteorology output from the GCMs are input to a hydrology model to generate streamflows, then meteorology and streamflows are used as inputs to a hydrodynamic and water quality model. A criticism of GCM models is that they are inherently biased and the variability in the results may not truly capture the range future climate uncertainty (Brown and Wilby, 2012; Brown et al., 2012). GCM projections are often perceived as forecasts by decision makers and stakeholders, making management decisions unclear when results present sometimes contradictory information. An alternative approach is based on a sensitivity analysis of a system of interest and uses GCMs to inform, rather than drive the analysis. System sensitivity can be explored using scenarios referred to as synthetic scenarios where a particular climatic element, such as temperature, is changed by realistic amounts, often in line with climate model projections for the area (Anandhi et al., 2011; Carter et al., 1994). With this approach, climate conditions that may stress system thresholds and vulnerabilities can be identified and the synthetic conditions can be compared to the range of conditions suggested by GCM projections as a way to prioritize management decisions.

Calibrated and verified CE-QUAL-W2 models for 10 years, based on historical data for Wachusett Reservoir, were used as the framework for future synthetic climate scenario development. Systematic increases in air temperature were used to evaluate the sensitivity of the reservoir thermal processes over a period of 100 years into the future (2013-2112). A 100 year-long future time series of inputs was created from the 10 years of historic calibrated and validated data (2003-2012), as it was assumed that 10 years was long enough to provide a representative description of the base, or current, conditions. These historic years were repeated 10 times in the order in which they occurred to maintain a realistic WSE within the highly managed elevation range. Rearranging the order of the 10 historic years, without additionally altering the inflows or outflows, produced unrealistically high or low WSEs if two similar years (e.g. two higher inflow years) occurred adjacent to one another. To estimate the potential effect of changing air temperatures on the sensitivity of the thermal processes in the reservoir, the same historical meteorological data was used but with the increased air temperature scenarios. A "base scenario" with no change from historic air temperature over 100 years, was used to compare to four increasing air temperature scenarios. The increasing air temperature scenarios were developed by imposing an increasing linear trend on the historic air temperatures of 1, 2, 3, and 4 ˚C over the 100 year long time series. The range of temperatures is within the range projected by GCMs for the region (Pica, 2012). For these simulations, other historical meteorological parameters and precipitation remained unchanged from historic values. Increases in precipitation in the region, as projected by many GCM models, would likely reduce the need to transfer water from the Quabbin Reservoir for quantity purposes and possibly increase the releases to the Nashua River to maintain WSE. However, increasing precipitation could also increase nutrient loading from tributaries and direct runoff into the reservoir, possibly requiring more water from the Quabbin Reservoir to be transferred to improve water quality, a subject of a future study.

The timing and the amount of inflow from the tributaries was unaltered for this study. However, the tributary temperatures were adjusted in response to the increasing air temperature trends imposed on the meteorology data, since tributary temperatures are reflective of short-term (weekly) changes in air temperature. A nonlinear regression model developed by Mohseni et al (1998) was used to simulate weekly stream temperatures using weekly air temperatures. Equation 2.2 was used to estimate stream temperature (T_s),

$$T_s = \mu + \frac{\alpha - \mu}{1 + e^{\gamma(\beta - T_\alpha)}} \qquad \text{Equation 2.2}$$

Where parameters α and μ specify the estimated maximum and minimum weekly stream temperatures, respectively, γ is a measure of the slope of the function, and β is the air temperature at the point of inflection (Mohseni et al., 1998). The stream temperature model simulated historic weekly stream temperatures for the nine tributaries in the Wachusett watershed well, with Nash-Sutcliffe (NSC) values and root mean square error (RMSE) values in the range of 0.64 – 0.86 and 1.95 - 4.04, respectively. Direct runoff temperature for historic simulations was calculated as an average of the tributary temperatures, and was calculated using the same method for future simulations.

It was assumed that the temperature of the Quabbin transfer water remained unchanged from the historical temperatures. The Quabbin Reservoir is over 6 times larger than Wachusett and the water temperature response to air temperature increases would likely by lagged compared to Wachusett, based on the volumes. Since there was no trend evident in available historical Quabbin water temperature data, it was unclear how to speculate to what extent the Quabbin water temperatures may change in response to air temperature increases without modeling the Quabbin reservoir processes in addition to Wachusett. Since the withdrawal to the Quabbin Aqueduct is deep in the water column, it was assumed that temperatures at this location would not respond to increases in air temperatures as rapidly as the surface water temperatures. Initial Wachusett Reservoir model simulations showed average water temperature increases of less than 2 °C with a 4 °C air temperature increase. A sensitivity test was also conducted with the Wachusett CE-QUAL-W2 model where Quabbin transfer water temperatures were linearly increased up to 2 ˚C over 100 years with a 4 degree air temperature increase and the results showed no significant difference in Wachusett epilimnion, hypolimnion, or Cosgrove Intake water temperature trends compared to historical water temperature simulation throughout the decades. This indicates that Wachusett water temperatures are driven more by air temperatures than the temperature of the Quabbin water, even though 30 to 60% of the annual inflow to Wachusett is from the Quabbin Reservoir. This result is consistent with literature stating that meteorological forcing is the main driver of water temperature in single lake systems (Edinger et al., 1968; Henderson-Sellers, 1988). However, increasing air temperature impacts on the Quabbin water temperatures may warrant further investigation in future studies. Initial temperature and specific conductivity the first day (January 1, 2013 and Julian day 1) of the future simulations (2013-2112) were uniform, as the reservoir was assumed to be completely mixed.

2.4. Results and discussion

2.4.1. Epilimnion and hypolimnion temperatures

Increasing air temperature in the future will have an effect on epilimnion and hypolimnion water temperatures in the Wachusett reservoir. For this study, epilimnion and hypolimnion temperatures were defined as the average water temperature in the upper and lower 5 m of the water column, respectively. Figures 2.5 and 2.6 illustrate the sensitivity of the Wachusett Reservoir epilimnion and hypolimnion water temperatures when increasing air temperature scenarios are imposed. The figures show the future scenario water temperature difference from the base scenario (100 years into the future with no air temperature trend) at three 10 year long time slices along the x-axis: the first (2013-2022), middle (2053-2062), and last (2103-2112) decades for each of the four increasing air temperature scenarios, indicated by the colors. Each box shows the median, 1st quartile, 3rd quartile, and range (minimum and maximum) daily water temperature differences from the corresponding day in the base scenario during a given decade. A positive difference in water temperature from the base scenario along the y-axis indicates future water temperatures that are higher than the base scenario and a negative difference indicates future water temperatures that are less than the base scenario.

In general, median water temperatures, represented by the line through the box plots in Figure 2.5 and Figure 2.6, for both the epilimnion and hypolimnion increased with increasing air temperature. The results illustrate how changes in air temperature affect water temperature at different depths in the water column. A greater water temperature increase occurred in the epilimnion, as expected since the epilimnion is at the air/water interface and directly exposed to the changing meteorological forcing (Livingstone, 2003; Livingstone, 1993). Median daily epilimnion water temperature increased in the range of 0.4 to 1.5 ˚C for the 1˚ to 4˚C air temperature change, respectively. Hypolimnion median water temperature increased less, in the range of 0.1 to 0.4 ˚C for the same future air temperature range. Average future epilimnion and hypolimnion water temperatures increased by as much as 12% and 7%, respectively.

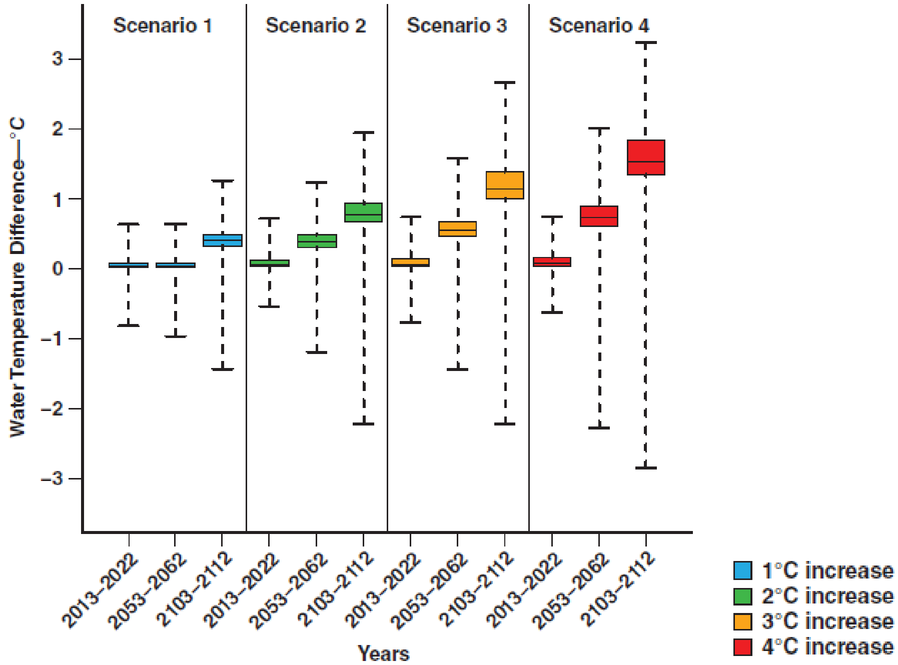

Figure 2.5 Future scenario epilimnion water temperature difference from the base scenario

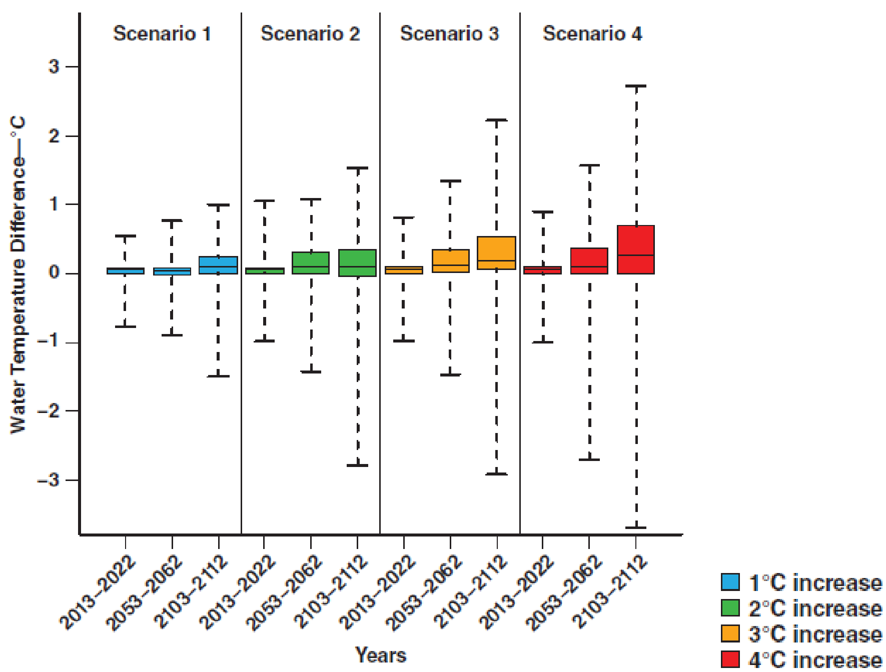

Figure 2.6 Future scenario hypolimnion water temperature difference from the base scenario

These average increases in epilimnion and hypolimnion water temperatures are within the range of increases simulated by Samal et al (2012) to occur by 2100 in the Cannonsville Reservoir (about 14.5 times larger volume than Wachusett with an approximate volume of 293,689 acre-ft and residence time of 237 days) when the future meteorology was based on A1B and A2 IPCC emission scenarios. The Cannonsville epilimnion and hypolimnion water temperatures were simulated to increase on average between 10-12% and 6-14%, respectively, during the thermal stratification period. Future changes in water temperatures predicted by Komatsu et al (2007) for the Shimajigawa reservoir in Japan, (12 times smaller in volume than Wachusett with an approximate volume of 16,700 acre-ft) are larger than those predicted in this study. GCM A2 scenario predictions over the region of the Shimajigawa reservoir suggest air temperature increases of 2.1 ˚C in the summer and 3.5-4.0 ˚C in the winter by the years 2091-2100, could result in an increase in the average epilimnion and hypolimnion water temperatures by 3.6 and 2.8 ˚C, respectively.

The minimum and the maximum water temperature differences from the base scenario became lower and higher respectively when air temperature was increased, indicating that throughout the 10 year time periods shown in Figures 2.5 and 2.6 there were days when water temperature was greater than the base scenario and also less than the base scenario. Negative water temperature differences occurred during the winter months, when the higher air temperatures decreased or eliminated ice formation and melt, as discussed in the following section. Ice and snow cover during the winter shields reservoir water from meteorological conditions and provides insulation, which maintains slightly warmer water temperatures during the ice covered months (Doran et al., 1996; Livingstone, 1993). When there was less or no ice cover in the future simulations, water was exposed to the cold winter air temperatures and became cooler than what it had been historically. Decreases in hypolimnion temperatures in future scenarios have also been observed in simulations of deep (24 m) Minnesota lakes by Hondzo and Stefan (1993), although winter ice cover was not included in those simulations and cooler temperatures were attributed to an earlier onset of stratification, which shielded the hypolimnetic water from surface heating.

Figure 2.7 show the average monthly epilimnion and hypolimnion water temperatures for the last 10 years of the 100 year long future scenarios with 4˚ C increasing air temperatures. The monthly water temperatures were averaged over each decade to identify and illustrate the general monthly changes and trends as a response to increasing air temperatures scenarios. Average epilimnion water temperatures increased throughout the year by about 2 ˚C, except during the months of February and March, where water temperatures were essentially the same as base scenario water temperatures. The maximum water epilimnion temperature occurred during July and August, when air temperatures were the highest. Average hypolimnion water temperatures increased by about 1 ˚C in November, December, and January, and were about 0.5 ˚C higher during the summer months. Hypolimnion water temperature in February was, on average, cooler than the base scenario by about 0.2 ˚C and average March hypolimnion water temperature was approximately the same as it was in the base scenario. As mentioned previously, the colder water temperatures during February and March were the result of decreased ice cover in the winter with higher air temperatures, therefore the open water was exposed to cold winter air temperatures. Maximum hypolimnion water temperatures occurred in November, at approximately the same time as the fall turnover. Average monthly water temperatures for other scenarios showed similar trends but changes of lesser magnitude.

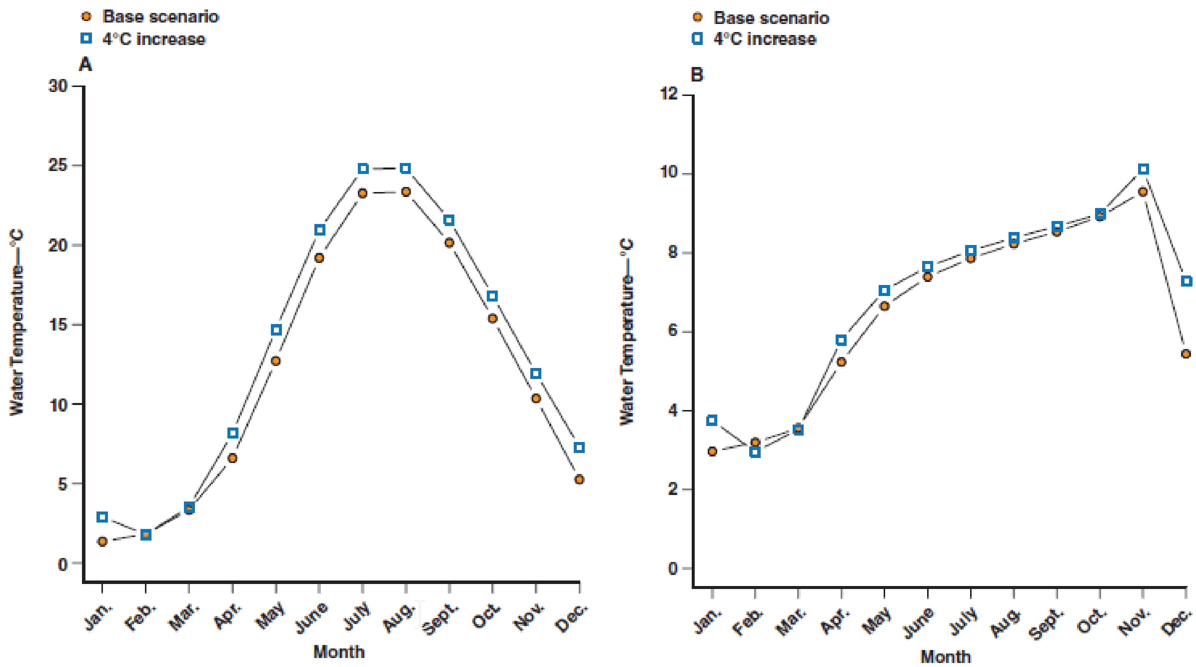

Figure 2.7 Average monthly epilimnion (a) and hypolimnion (b) water temperatures for the last 10 years (2103-2112) of the 4 ˚C increasing air temperature scenario

Changes in epilimnion and hypolimnion water temperatures affected the length of the summer stratification period. The difference between epilimnion and hypolimnion water temperatures of a water body is an indicator of the strength of the stratification. For this study, the reservoir was considered stratified when the temperature difference between the epilimnion and the hypolimnion was greater than 1 ˚C, a criteria defined in Hondzo and Stefan (1996) and used in subsequent studies (Fang and Stefan, 1999; Fang and Stefan, 2009). Table 2.2 shows the average and the range of the number of days that the reservoir was stratified in the base scenario and for all the increasing air temperature scenarios. The average length of stratification throughout the 10 historical years of calibrated and validated data was about 207 days in length with a range of approximately 183 – 235 days. With increasing air temperatures, the average number of days that the reservoir was stratified increased to 212 days (a 5 day increase) by the middle decade and to 220 days (a 13 day increase) by the last decade of the scenarios. During the first 10 years of each of the scenarios there was essentially no change in the length of stratification. Simulated climate change impacts on the length of stratification in the Cannonsville Reservoir by Samal et al (2012) indicated similar increases in the number of stratified days (7 days longer for A1B and 12 for A2 scenarios) by the year 2100.

Table 2.2 Changes in stratification duration (in days) with increasing air temperatures

Scenario	**2013-2022**		**2053-2062**		**2103-2112**	
	Average	**Range**	**Average**	**Range**	**Average**	**Range**
Base	207	183 - 235	207	182 – 234	207	183 - 235
1 ˚C	207	183 - 235	207	183 – 235	208	184-237
2 ˚C	207	183 - 235	208	184 – 236	213	185-241
3 ˚C	207	183 - 235	210	185 – 238	215	187-244
4 ˚C	207	183 - 235	212	186 - 245	220	188-254

In addition to the changes in the length of the stratification periods, the timing of stratification and turnover was altered by increasing air temperatures. By the last decade of each of the increasing temperature scenarios, stratification occurred on average between 0 and 5 days earlier and a maximum of 10 days earlier with a 4 ˚C increase. Fall turnover in the reservoir occurred between 1 and 8 days later on average than the average turnover date in the base scenario. Turnover occurred as much as 18 days later in the scenario with the 4 ˚C air temperature increase. Changes in the timing and the duration of stratification in the reservoir could have an impact on nutrient cycling and could result in changes in the timing and duration of algal blooms. However, Wachusett Reservoir and Quabbin Reservoir are characterized as oligotrophic water bodies, and therefore impacts to nutrient cycling and algal blooms would likely be less than those observed and modeled in previous studies (Komatsu et al., 2007; Lee et al., 2012; Lee et al., 2012; Peeters et al., 2002; Sahoo and Schladow, 2008; Sahoo et al., 2011; Sahoo et al., 2013; Samal et al., 2012).

2.4.2. Ice cover

In addition to increasing water temperatures, changes in meteorological conditions affect ice cover formation, growth, and melting on the reservoir. CE-QUAL-W2 can simulate the onset, growth, and breakup of ice cover. When the surface water temperature becomes lower than the freezing point, the negative temperature is converted to an equivalent ice thickness and equivalent heat is added to the heat source and sink term for the water. Once there is a net gain of heat to the surface and the surface temperature becomes greater than the freezing temperature, the ice begins to melt. Ice cover, growth, and breakup depend on locations of temperatures of inflows and outflows, evaporative wind variations over the ice surface, as well as turbulence and water movement beneath the ice. Ice growth or melt at the ice-water interface in CE-QUAL-W2 can be described by the following Equation 2.3 (Cole and Wells, 2008).

$$\Delta\theta_{iw}^{n} = \frac{1}{\rho_i L_f}[K_i \frac{T_f - T_s^n}{\theta^{n-1}} - h_{wi}\left(T_w^n - T_f\right)] \qquad \text{Equation 2.3}$$

Where θ_{iw} is ice growth/melt at the ice-water interface (m), ρ_i is density of ice (kg/m^3), L_f is latent heat of fusion (J/kg), K_i is thermal conductivity of ice (W/m˚C), T_f is freezing point temperature (˚C), θ is the ice thickness (m), T_s is ice surface temperature, (˚C), h_{wi} is the coefficient of water-to-ice heat exchange through the meal layer (W/m^2˚C), and T_w is water temperature below the ice (˚C). For the historical years 2003-2012, CE-QUAL-W2 simulated the duration of ice cover in the North Basin of the reservoir within 8 days, on average, of the observed duration.

Figure 2.8 illustrates how the average number of days with ice cover during each decade changes with increasing air temperature scenarios. The number of days with ice cover were averaged over each decade in the scenarios to identify and illustrate the general changes and trends as a response to the changing meteorological driver. The sensitivity of individual years to changes in ice cover varied by year, depending on the particular meteorological and hydrologic conditions of that particular year. The average number of days that the reservoir was ice covered throughout the base scenario was about 49 days per year. By the last decade of each of the scenarios, the average number of days with ice cover in the North Basin each year was approximately 41, 32, 31, and 22 days for the 1, 2, 3, and 4 ˚C increases in future air temperatures, respectively.

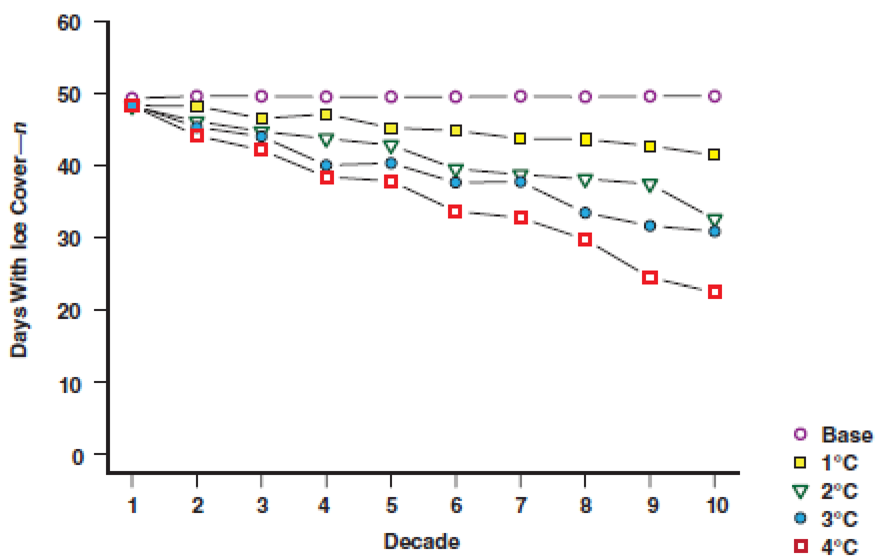

Figure 2.8 Average number of days with winter ice cover in the North Basin for increasing air temperature scenarios

Average annual ice cover thickness also decreased with increasing air temperature scenarios, as seen in Figure 2.9. The average annual ice thickness in the North Basin during each decade for the base scenario was approximately 0.15 m. By the last decade of each of the scenarios, the average ice thickness in a winter decreased to 0.13, 0.12, 0.10, 0.09 m for the 1, 2, 3, and 4 ˚C increases in future air temperatures, respectively. The average maximum ice thickness during the base scenario was approximately 0.44 m throughout the future 100 years and decreased by as much as 0.14 m by the last decade of the 4 ˚C air temperature scenario. Ice thickness over the contiguous United States under climate change scenarios was simulated to decrease by an average of 0.21 m in a study done by Fang and Stefan (1998). Results from this study are consistent with results of Fang and Stefan (1998; 1999), which indicated delayed ice formation, earlier melt, and reduction in ice thickness under scenarios investigating a possible doubling of atmospheric CO2.

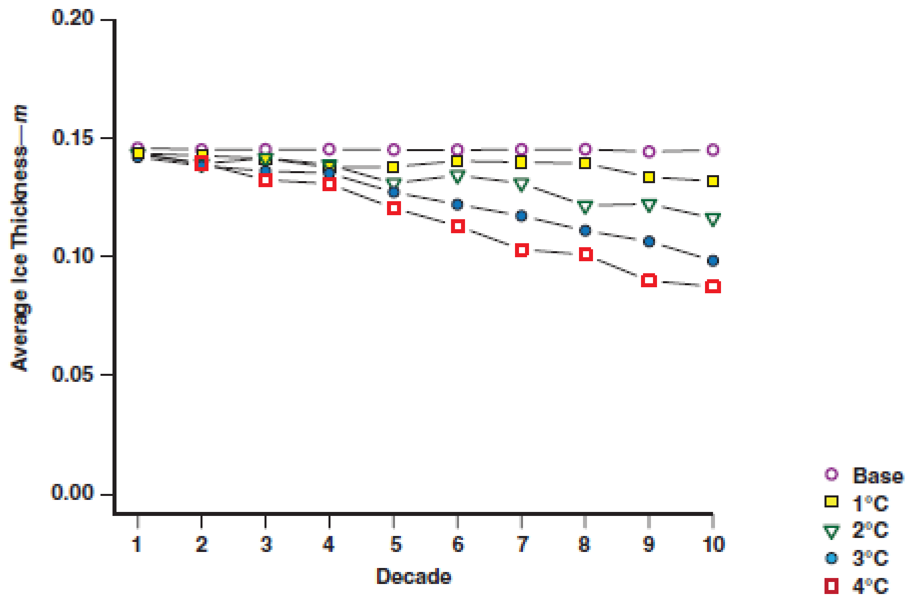

Figure 2.9 Average ice thickness in the North Basin for increasing air temperature scenarios

2.4.3. Cosgrove intake temperatures

The effect of increasing air temperatures on water temperature at the Cosgrove Intake is seen in Figure 2.10. The plot shows the first, middle, and last decade within each of the four increasing air temperature scenarios, as indicated by the colors, and the Cosgrove Intake water temperature difference from the base scenario. With increasing air temperature the median water temperature at the Cosgrove Intake was greater than the base scenario. With a 4˚C air temperature increase over 100 years, the median water temperature was approximately 1 ˚C higher by the last decade.

With increasing air temperatures, there was also increasing variability in water temperature differences from the base scenario. If air temperatures increase by 4˚C over the next 100 years, simulations suggest that water temperatures could be up to 4˚C warmer (during the summer months) and 6˚C cooler (during the winter months) than historical water temperatures in a given year. Similar to results discussed previously, cooler water temperatures occurred during the winter months and were a result of decreased ice growth, later onset of ice cover, and earlier ice melt.

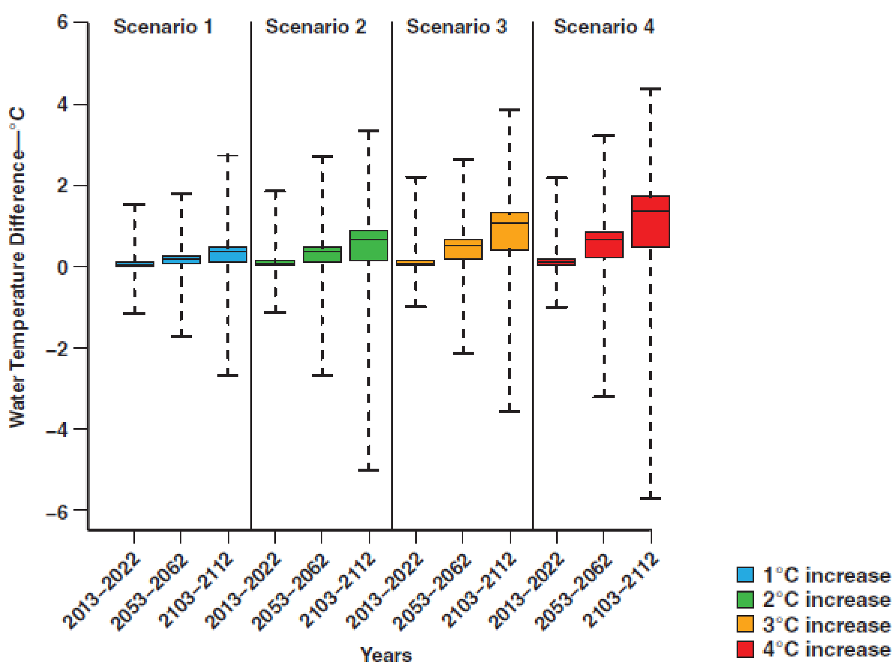

Figure 2.10 Future scenario Cosgrove Intake water temperature differences from the base scenario

2.5. Conclusions

Future changes in climate will have varying degrees of impact on the thermal stratification processes of water bodies across the globe. The impacts of climate change on the stratification processes are important to understand, since water temperature affects many other water quality processes, therefore making the topic especially important to drinking water reservoirs, where source water quality is of importance. Changes in thermal stratification as a response to changes in meteorological conditions will influence, for example, nutrient cycling, dissolved oxygen concentrations throughout the water column, and the timing and duration of algal blooms.

The purpose of this study was to investigate the sensitivity of water temperature to changes in one meteorological parameter, temperature, which is a main driver of many processes occurring in the reservoir. This analysis assumed no other changes from historic meteorological data and that the historic water balance was maintained. Although the method used in this study limits the degree of variability in the future to the variability of 10 historic years, the method still allows for an understanding of the trends of changes the reservoir might undergo. Table 2.3 summarizes the effects of increasing air temperatures on water temperatures in the Wachusett Reservoir during the middle and last decades of the 100 year long future scenarios.

Table 2.3 Summary of metrics for the middle and last decade of the four scenarios

10 Year Average	2053-2062	2103-2112
Epilimnion Temp Increase (˚C)	0.2 – 0.7	0.4 – 1.4
Hypolimnion Temp Increase (˚C)	0.1 – 0.2	0.1 – 0.4
Cosgrove Temp Increase (˚C)	0.1 – 0.5	0.3 -1.1
Stratification Duration (days)	1 – 6	1 - 13
Fewer Days of Ice Cover	4 - 11	9 - 28
Decrease in Max Ice Thickness (m)	0.1 – 0.3	0.2 - 0.6

In general, the average of epilimnion and hypolimnion water temperature increased with increasing air temperatures by about 0.4-1.4˚C and 0.1-0.4 ˚C, respectively by the end of all scenarios. These temperatures correspond to average increases in epilimnion and hypolimnion water temperatures by up to 12% and 7%, respectively, compared to base scenario temperatures. Water temperatures in the epilimnion and hypolimnion in the future scenarios were generally warmer throughout most of the year, and slightly cooler during the winter months, when there were changes in ice cover growth, onset and duration. The changes in water temperatures resulted in greater differences between epilimnion and hypolimnion temperatures and therefore stronger stratification during the summer months. On average, the length of stratification increased by about 1-2 weeks, beginning earlier and ending later, towards the end of all scenarios. In the base scenario, the reservoir was stratified on average about 57% of the year. Under increasing air temperatures, simulations suggest that the reservoir could remain stratified up to 8% longer in the future. The average number of days with ice covering the reservoir during the winter months decreased by about 8-22% by the middle decade of each scenario and by about 18-57% by the last decade. When the reservoir was covered with ice, the thickness was less, by about 9-40% during the last decade for all scenarios. Additionally, Cosgrove water temperatures increased by an average of 0.3-1.1˚C by the last decade of each of the scenarios.

Results from this study provide greater insight into the sensitivity of the Wachusett Reservoir water temperatures to potential projected increases in air temperature due to climate change. The use of models to simulate the physical response to climate change is valuable since a model can demonstrate that changes in water temperature are not one-for-one with air temperature changes and the interaction of ice cover plays an important role in seasonal variability. An understanding of how air temperature impacts water temperature for a specific water body is a first logical step in any larger climate study, since water temperatures influence many physical, chemical, and biological processes within the water body. Future work will evaluate impacts of changing multiple model inputs which may be subject to a variety of potential climate change impacts, such as additional changes to meteorology and hydrology. The combined effects of changes in inflow volumes, timing, quality, and meteorology is important to understanding and mitigating future water quality challenges, especially those related to extreme events.

2.6. Acknowledgements

This project was supported by the Massachusetts Department of Conservation and Recreation. The findings are the opinions of the authors and do not represent the official findings of the DCR or MWRA.

CHAPTER 3

PROACTIVE MODELING OF WATER QUALITY IMPACTS OF EXTREME HYDROLOGIC EVENTS. PART 2: DRINKING WATER RESERVOIR

Authors

Jeznach, L.C., Hagemann, M., Park, M., Tobiason, J.E.

3.1. Abstract

Extreme precipitation events are of concern to managers of drinking water sources because these occurrences can affect both water supply quantity and quality. However, little is known about how these low probability events impact organic matter and nutrient loads to surface water sources and how these loads may impact raw water quality. This study describes a method for evaluating the sensitivity of a water body of interest from watershed input simulations under extreme precipitation events. An example application of the method is illustrated using the Wachusett Reservoir, an oligo-mesotrophic surface water reservoir in central Massachusetts and a major drinking water supply to metropolitan Boston. Extreme precipitation event simulations during the spring and summer resulted in total organic carbon, UV-254 (a surrogate measurement for reactive organic matter), and total algae concentrations at the drinking water intake that exceeded recorded maximums. Nutrient concentrations after storm events were less likely to exceed recorded historical maximums. For this particular reservoir, increasing inter-reservoir transfers of water with lower organic matter content after a large precipitation event has been shown in practice and in model simulations to decrease organic matter levels at the drinking water intake, therefore decreasing treatment associated oxidant demand, energy for UV disinfection, and the potential for formation of disinfection byproducts.

3.2. Introduction

Future climate projections based on global climate models (GCMs) indicate that the average annual surface temperatures around the globe will increase while future changes in precipitation will vary with geographic region and seasons. The global mean surface temperature has risen approximately 0.7ºC since the start of the 20th century and is projected to exceed 1.5ºC by the end of the 21st century (IPCC, 2013; World Meteorological Organization (WMO), 2014). Warming throughout the century will increase the amount of water stored in the atmosphere, possibly leading to a more dynamic hydrologic cycle (IPCC, 2013). Global warming and changes in precipitation will not be uniform in time and space and will vary with region as well as wet and dry seasons (IPCC, 2013; Stanford et al., 2014).

There is growing evidence that historically low-probability "extreme" weather events such as floods, droughts, and heat waves are occurring more frequently and in different locations than they have occurred in the past (IPCC, 2013; NOAA, 2013). Extreme weather events are generally defined as those that have less than 1% to 5% probability of annually occurring in a specific region and are the result of any substantial change in weather type, severity, frequency, duration, or combination of events (Stanford et al., 2014). In the United States, for example, analyses of precipitation events during the 20th century demonstrated an increase in precipitation and an increase in precipitation intensity, especially during the last three decades of the 20th century in the eastern US (Groisman et al., 2004; Groisman et al., 2005). Increased precipitation and precipitation intensity in the eastern US have led to increased streamflows in the region, and GCM projections indicate a continuation of this trend (Groisman et al., 2004; Groisman et al., 2005).

Climate induced changes to watershed hydrology and water quality, such as changes to tributary volumes, timing of inflows, and constituent loads, have an effect on receiving water quality. If the receiving water is a drinking water supply reservoir, changes in raw water quality are of concern because lower quality raw water increases treatment costs, impairs finished water aesthetics, and possibly is a risk to public health. The ability of a waterbody to withstand the stress of these altered inflows and loads depends on how stressed the waterbody is in its current trophic state under current hydrologic conditions (Murdoch et al., 2000). Additionally, the hydrology of reservoirs is often anthropogenically controlled and therefore responses to climate change will be influenced by the specific features of the individual system. Gradual changes in meteorology and hydrology as well as the increased occurrences of low-probability, short-term events are equally important to understand with respect to water quality impacts. Continued climate stress may lead to exceeding system thresholds and can result in water quality degradation (Murdoch et al., 2000; Whitehead et al., 2009). Water bodies that are currently impaired will require less climate stress to exceed water quality thresholds, while less impaired water bodies will be able to tolerate higher stress and change while maintaining high water quality.

Increasing occurrences of low-probability extreme events can result in short-term water quality changes (e.g. spikes in nutrient loadings) and long-term water quality impacts (e.g. the compounding effect of greater annual nutrient loads) to drinking water sources. Water quality degradation as a result of extreme short-term changes in air temperature, precipitation in the form of rain or snow (as well as a lack of precipitation, i.e., droughts), tributary flow rates and timings, and runoff amounts all affect nutrient, organic matter, and sediment loads, in addition to receiving water organic matter composition, algae dynamics, and water age/flushing rates. The frequency and likelihood of event occurrence are also important to consider when evaluating impacts of extreme events on water quality, since combinations of several events can result in more gradual degradation in water quality over a period of months or years.

Extreme precipitation events are a major driver for the export of terrigenous organic carbon and organic-bound nutrients because erosion and sediment transport during large precipitation events are greater than during normal flow conditions. Higher streamflows can lead to greater mobility and dilution of constituents as well as greater sediment loads, altering the morphology of rivers and sediment transport to surface water bodies (Whitehead et al., 2009). Despite the knowledge of the importance of precipitation events controlling carbon and nutrient fluxes to water bodies, there have been few published studies that have analyzed fluxes from large or extreme precipitation events and most literature is focused on carbon fluxes.

Organic carbon has an important role in ecosystems since it is involved in the complexation and transport of toxic metals and organic contaminants. The costs to remove of organic matter during drinking water treatment scale with source water organic matter content. Greater organic matter concentrations can increase coagulant and oxidant demands (so higher doses needed) and increase the formation of regulated and unregulated disinfection byproducts (DBPs) during the disinfection process. It is widely accepted that dissolved organic carbon (DOC) and particulate organic carbon (POC) fluxes increase during precipitation events, however it is less clear how the ratio of DOC:POC changes with different precipitation volumes and intensities or watershed characteristics (Dhillon and Inamdar, 2013; Inamdar et al., 2006; Yoon and Raymond, 2012).

Studies of forested watersheds that contribute water to drinking water sources indicate that DOC and POC fluxes increase dramatically during extreme events. A recent notable example is

Hurricane Irene, a precipitation event with a 200 year return period, that impacted the east coast of the US in August 2011 (Dhillon and Inamdar, 2013; Yoon and Raymond, 2012). Yoon and Raymond (2012) used a series of high resolution measurements to determine the total amount of DOC and DON transported in Esopus Creek in New York during Hurricane Irene. The Esopus Creek drains 16,500 ha of the Catskill Mountains and eventually discharges into the Ashokan Reservoir, a primary drinking water source for New York City. During this event, flows increased 330 fold, and concentrations increased 4-fold, resulting in roughly 40% and 31% of the average annual DOC and DON mass inputs in only 5 days (Yoon and Raymond, 2012). Measurements of DOC and POC from a forested watershed draining the Maryland Piedmont during Hurricane Irene were 20% and over 50% of the annual 2011 DOC and POC fluxes, respectively (Dhillon and Inamdar, 2013). Measurements from this study indicate a large increase in POC fluxes relative to DOC fluxes at a certain event precipitation threshold of approximately 75 mm for the study watershed (Dhillon and Inamdar, 2013). Based on this study, extreme precipitation events may increase ratios of POC to DOC to greater than one and the threshold at which this change occurs will depend on properties of the watershed such as land-use, vegetation, and geology (Dhillon and Inamdar, 2013). In contrast, the DOC flux from an agricultural watershed on the Virginia Coastal Plain during two consecutive tropical storms was twice as much as the POC flux due to land use differences (Caverly et al., 2013). Additionally, events occurring in lower portions of a large watershed may have a greater and faster impact on raw drinking water quality at an intake, since there is less attenuation time for particles and organic matter in the system, as observed in a Phoenix, Arizona watershed (Barry et al., 2016).

Nutrient concentrations and exports during heavy precipitation events are even less well understood, since these vary widely with watersheds, seasonal conditions, land cover types, hydrology, geology, and other landscape characteristics. Concentrations of nitrogen species in the northeastern United States have been observed to be higher during spring events, which can be attributed to snowmelt and the flushing of nitrate (NO_3^-) accumulated in the soil during the dormant winter season (Correll et al., 1999; Inamdar et al., 2006). For a glaciated forested watershed in western New York, NO_3^- concentrations increased from pre-event conditions by 60% during spring snowmelt events from May 2003 through April 2004 but decreased as much as 92% during large precipitation events in the summer and fall (Inamdar et al., 2006). The maximum changes from pre-event concentrations for ammonium (NH_4^+) and DOC concentrations for the same watershed over 12 months of precipitation events were 240-3200% and 120-370%, respectively (Inamdar et al., 2006). Total phosphorous (TP) concentrations observed across four adjacent watersheds of differing land uses on the Atlantic Coastal Plain in Maryland were higher during summer storms than in the winter and spring (Correll et al., 1999). Nutrient concentrations in the forested watershed were the least impacted by increasing tributary discharges due to precipitation events compared to croplands and mixed-land use watersheds (Correll et al., 1999). Phosphorus exports from forested catchments have been shown to be primarily associated with episodes of high discharge and sediment loads (Meyer and Likens, 1979).

Hydrodynamic and water quality models are commonly used to simulate water quality of a receiving water body in response to changes in watershed inputs and/or climate change. The 2-D model CE-QUAL-W2 is an example of a commonly used model that has been applied to over 200 water bodies around the world and has been used to evaluate climate change impacts on water quality for many water bodies across the globe (Cole and Wells, 2015; Fang et al., 2007; Lee et al., 2012; Samal et al., 2013). Model simulations across many studies using various models indicate that increasing air temperatures due to climate change will result in increased epilimnion and hypolimnion water temperatures, earlier stratification periods, deeper thermoclines, later turnovers, and decreased ice cover in colder regions (Fang and Stefan, 1998; Fang and Stefan, 1999; Fang and Stefan, 2009; Hondzo and Stefan, 1993; Jeznach and Tobiason, 2015; Komatsu et al., 2007; Sahoo and Schladow, 2008; Sahoo et al., 2011; Samal et al., 2012). Changes in water temperature can affect other in-situ water quality parameters such as dissolved oxygen and processes such as nutrient cycling and algal dynamics (Fang and Stefan, 2009; Komatsu et al., 2007; Sahoo et al., 2013). Lake and reservoir models coupled with watershed models have simulated changes in nutrient and organic matter watershed loads (Debele et al., 2008; Narasimhan et al., 2010). Model simulations of future long-term increases in precipitation over a watershed indicate increased nutrient loads to a receiving water body (Chang et al., 2001).

This study and the Hagemann et al (2016) companion study, quantify the potential impacts of extreme precipitation events on tributary and receiving water quality, particularly those water bodies used as drinking water sources. A major objective of these studies was to develop a method to evaluate extreme precipitation event impacts on water quality in a watershed (as described in Hagemann et al (2016) and in a drinking water reservoir (as described in this study). The method was applied in both studies to the Wachusett Reservoir, a major drinking water supply reservoir to metropolitan Boston, as an example to ultimately quantify source drinking water quality sensitivity to potential extreme precipitation event nutrient loads from the surrounding watershed. To the authors' knowledge, this is the first published study that links hypothetical extreme short-term precipitation event watershed nutrient loads with simulated receiving water body quality and proactively models the impacts of such an event. The method and example application presented are beneficial to water managers since this type of study can aid in proactive management efforts to maintain or improve water quality in the face of climate change and can improve management responses to extreme events to mitigate potential increases in treatment costs.

3.3. Study area

The Wachusett Reservoir is the second largest water body in Massachusetts, USA, with maximum depth of 36.6 m, length of 13.5 km, surface area of 16.8 km^2, and an approximate volume of 0.25 billion m^3. The reservoir system includes the Quabbin Reservoir (approximately 1.6 billion m^3) located in western Massachusetts. Water transferred from Quabbin travels 48 km (30 miles) east through the Quabbin Aqueduct to the western end of the Wachusett Reservoir. Raw drinking water is withdrawn at the Cosgrove Intake, treated, and sent east to Boston via the Metrowest Water Supply Tunnel (Figure 3.1). Wachusett Reservoir is classified as oligo-mesotrophic, and water withdrawn at the intake is not filtered; treatment includes disinfection (ozone, ultraviolet light, and chloramines), fluoride addition, and pH and water chemistry adjustment to prevent corrosion in the distribution system. Together, the reservoirs have a safe yield of approximately 13.1 m^3/s, supplying 51 communities (approximately one-third of the Massachusetts population) in the Boston metropolitan and central Massachusetts area with drinking water. The Massachusetts Water Resources Authority (MWRA) assumes responsibility for the delivery and distribution of water to the communities while the Massachusetts Department of Conservation and Recreation (DCR) manages the surrounding watersheds.

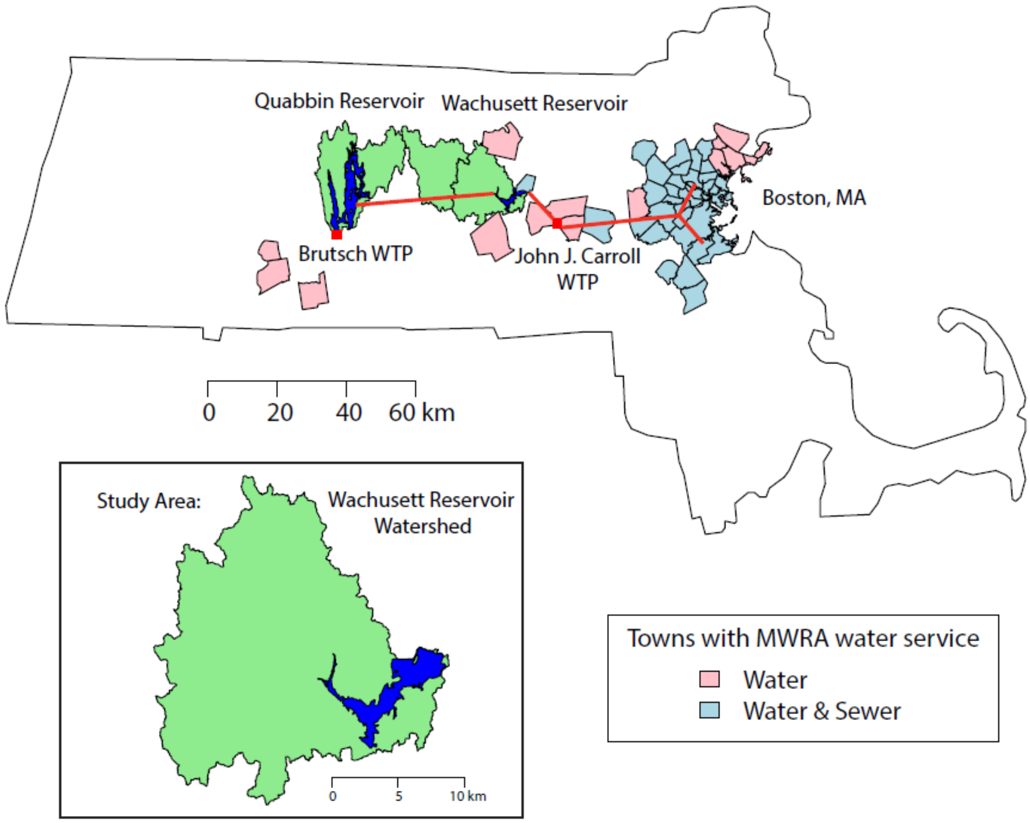

Figure 3.1 MWRA drinking water system

The Wachusett watershed (excluding the reservoir) area is approximately 286 km^2 (70,678 acres). Land use in the watershed is primarily forested (67.3%), followed by residential (10.8%), wetland (7.7%), agriculture (5.7%), open water (2.7%), commercial/industrial (2.3%), and other (3.4%) (MA DCR, 2013). The overall amount of impervious land in the Wachusett watershed is estimated to be 5.5% (MA DCR, 2013). Approximately 30 to 60% of the annual inflow to Wachusett is from the Quabbin Reservoir with the primary transfer objective of maintaining water surface elevation and secondary objectives of generating hydropower and introducing water with less natural organic matter content than Wachusett Reservoir water. Other major inflows to the Wachusett Reservoir include direct precipitation, direct runoff, and inflow from nine tributaries. The largest tributaries, the Stillwater and the Quinapoxet rivers, enter the reservoir from the northwest and contribute approximately 30 to 40% of the total annual inflow. The Cosgrove drinking water intake, located at the eastern most end of the reservoir, is the major withdrawal from Wachusett Reservoir but water also leaves through evaporation, minor withdrawals to nearby towns, as well as releases and spills to the Nashua River.

3.3.1. Data

Modeled inflows to the Wachusett Reservoir include the Stillwater and Quinapoxet Rivers, seven minor tributaries, the Quabbin Transfer, direct runoff, and precipitation. The Stillwater and Quinapoxet Rivers are gaged for flow by the USGS and account for the drainage of approximately 73% of the watershed area. Flow from minor tributaries was estimated based on the daily watershed yield of the Stillwater River, as described in Tobiason et al (2002). The inflows to the Wachusett Reservoir from the Quabbin Reservoir were measured daily by the MWRA at the aqueduct outlet. Direct runoff was calculated based on the ratio of Stillwater daily discharge to Stillwater watershed area multiplied by the entire direct runoff area. Hourly precipitation data from the Worcester Regional Airport, approximately 10 miles southwest of the reservoir, were obtained from the National Oceanic and Atmospheric Administration (NOAA). Hourly meteorological data such as air temperature, dew point temperature, wind speed, wind direction, and cloud cover were also acquired from NOAA. MWRA daily measured outflows from the reservoir included withdrawals from the Cosgrove drinking water intake, discharge to the Wachusett Aqueduct, as well as releases and spillway discharges to the Nashua River at the Wachusett Dam.

Water quality data used for the model boundary conditions include constituent concentrations for the nine tributaries, precipitation, direct runoff, and Quabbin transfer inflows. Measured constituents in addition to water temperature included in the simulations were specific conductivity, total organic carbon (TOC), total phosphorus (TP), nitrate (NO_3-N), ammonia (NH_4-N), and UV-254 (a surrogate measurement for DOC). Since POC and DOC were not directly measured in tributaries, it was assumed based on previous watershed work that 5% of TOC was POC and the remaining 95% was DOC, of which 20% was assumed to be labile dissolved organic matter (LDOM) and 80% was assumed to be refractory dissolved organic matter (RDOM) (Bryan, 2004; Buttrick, 2005; Hodgkins, 1999; Jordan and Likens, 1975; Roberts, 2003). Algal inputs from tributaries (LPOM) were assumed to be negligible therefore all POM in tributary inflows was considered from detritus and therefore refractory (RPOM). Tributary orthophosphate (PO_4^{-3}) data did not exist for the study period but a prior study comparison of outlet $PO_4^{-3}$and TP for the Quabbin Reservoir indicated that, on average, approximately 50% of TP was PO_4^{-3} (Garvey, 2000).

Water quality constituents in precipitation were measured at two National Atmospheric Deposition Program (NADP) stations in Massachusetts; one is located on the Prescott Peninsula of the Quabbin Reservoir and one in Lexington, Massachusetts (discontinued in September 2010). Constituents measured at these locations include specific conductivity, NO_3-N, and NH_4-N. PO_4^{-3} and TOC precipitation concentrations are not included in this data and concentrations for this study were based on the work by Roberts (2003) and Garvey (2000) for the Quabbin Reservoir.

Water quality constituent concentrations for the Quabbin transfer were based on measurements by the MWRA at the Chicopee Valley Aqueduct (CVA) withdrawal from Quabbin. The CVA withdrawal provides water to the western Massachusetts towns of Chicopee, South Hadley, and Wilbraham. For this study it was assumed that water quality at these two withdrawal locations were comparable, based on previous work on Quabbin Reservoir water quality Garvey (2000).

3.4. Methods

There are several approaches that can be utilized when modeling future climate change impacts on water quantity and quality, however a new approach was developed for this study due to the limitations of simulating future extreme precipitation events. The typical approach to simulating climate change impacts that generally employs the use of downscaled GCMs to drive watershed, systems, and hydrodynamic and water quality models was deemed inappropriate for this particular study for several reasons: 1) the poor sub-daily meteorological temporal resolution of downscaled projections, 2) the great uncertainty associated with regional precipitation projections, especially short term precipitation events (Baker and Peter, 2008; Willems et al., 2012). There are also limitations to methods commonly used to estimate or simulate watershed nutrient loads during average watershed conditions when they are applied to large precipitation events such as 1) the current inability to accurately model the relevant processes governing constituent concentrations during large events, 2) the lack of or limited temporal and spatial sampling of a variety of constituents during extreme rain events in most watersheds, 3) the inability to generalize measurements and observations from different studies across different watersheds, 4) and the inability in some cases to generalize measurements across one watershed.

Therefore, this study uses a new approach to understand how extreme precipitation events may impact surface drinking water quality. In this method, the sensitivity of water quality in the receiving water body was explored based on reasonable choices of precipitation amounts for the region of interest and statistically generated ranges in probable concentrations, reflecting the uncertainty of predicting concentrations at high tributary flows. Surface water quality was assessed using a process-based hydrodynamic and water quality model. The method is discussed in the following sections in the context of a case study.

3.4.1. Event Watershed Loads

An overview of the method used to generate watershed flows and constituent concentrations resulting from extreme precipitation events is described in this section, but a more detailed description of the methods can be found in the Hagemann et al (2016) Part 1 companion paper. This study employed a framework for generating watershed inputs (i.e. tributary flows and constituent concentrations) to ultimately evaluate water quality in a receiving drinking water source. The methodology for generating event watershed loads was a combination of imposed storm-rainfall depth hydrology scenarios and a probabilistic model for water quality constituent concentrations. This approach accounts for the uncertainties associated with generating constituent concentrations in tributaries during extreme precipitation events, since little is understood about the processes that effect concentrations during heavy rain events.

The approach was driven by the choice of a precipitation rate appropriate for the region and the particular extreme event investigation. For this case study, three precipitation depths of 101 mm (4 inches), 152 mm (6 inches) , and 203 mm (8 inches) over a 24 hour period were chosen, representing historical return intervals in the region of 5, 50, and 100 years, respectively. The rain was assumed to uniformly occur over the watershed and all excess precipitation was assumed to be converted to runoff within 7 days of the imposed rainfall scenario.

Inflows to the reservoir for a chosen historical base year were modified to account for the imposed event precipitation depths. For this study, the year 2011 was used as the base year on which to impose the event scenarios since the shape of the hydrographs reflected a typical year with moderate snowmelt in the early spring and occasional rainstorms throughout the year. During this year there was a mid-April rainstorm (65 mm, 2.6 in,) and a late August hurricane event (Hurricane Irene), which is the largest recent “extreme” precipitation event in the watershed. In central Massachusetts, Hurricane Irene produced approximately 112 mm (4.4 inches) of rain over 2 days, with a maximum precipitation rate of about 12.7 mm/h (0.5 in/hour) (NCDC Climate Data Online, Worcester Ma Regional Airport). Extreme event scenarios for this study were imposed on the year 2011 over the historic spring precipitation event on April 16th and over the August 28th hurricane, during which the reservoir was unstratified and stratified, respectively.

For all tributary inputs, the observed hydrograph was separated into baseflow and direct runoff. The extreme event scenario hydrograph was calculated by volumetrically scaling up the direct runoff portion of the observed hydrograph by the estimated excess precipitation. Baseflow and precipitation losses were assumed constant for an event magnitude, while direct runoff and rainfall excess were not (Hagemann et al 2016). Resulting streamflows for the two largest tributaries, Stillwater and Quinapoxet, are shown in Figure 3.2, with the original historic streamflow in black and the 101, 152, and 203 mm precipitation scenarios in green, orange, and red, respectively.

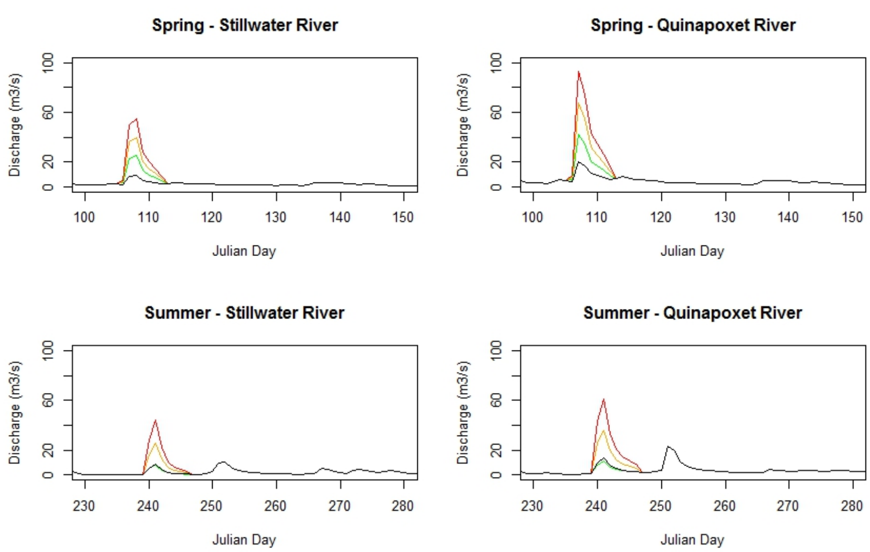

Figure 3.2 Original and simulated Stillwater and Quinapoxet River flows for 101, 152, and 203 mm (black, green, orange, red, respectively) spring and summer precipitation events

Constituent concentrations in the tributaries were generated probabilistically based on historic flow and concentration data from the case study watershed tributaries (Hagemann et al, 2016). A multivariate probability model was developed for all constituents in all tributaries, conditional on time and hydrological conditions. Predictions of concentrations at higher flows in these models have increased uncertainty due to the general lack of data for higher flow events. To account for this uncertainty, 100 quasi-Monte Carlo samples were drawn from a two-dimensional space, simplified from a multi-dimensional space using principle component analysis. These 100 samples for each precipitation event depth for each season (spring April event or summer August event) were used as boundary conditions to the process-based reservoir model.

3.4.2. Reservoir Modeling

The hydrodynamic and water quality processes of the Wachusett Reservoir were simulated using the two-dimensional process-based model CE-QUAL-W2 that directly couples hydrodynamic and water quality process algorithms. The CE-QUAL-W2 model of the Wachusett Reservoir was originally developed and calibrated by Camp, Dresser, and McKee (CDM) and FTN Associates using data from the years 1987, 1990, and 1992 (Camp et al., 1995). The Wachusett Reservoir is an appropriate application of the model since the relatively long and narrow bathymetry produces velocity, temperature and water quality gradients predominantly in two directions (longitudinal and vertical). The CE-QUAL-W2 model (various versions) has been used in many studies by researchers at the University of Massachusetts, Amherst since the early 1990’s, with work focusing on both the Quabbin and Wachusett Reservoirs and a range of water quality related topics (Ahlfeld et al., 2003; Buttrick, 2005; Devonis, 2011; Jeznach et al., 2014; Jeznach and Tobiason, 2015; Matthews, 2007; Sojkowski, 2011; Stauber, 2009). Version 4.0 was used for this study.

The Wachusett Reservoir model grid (Figure 3.3) is based on the original model grid developed by CDM (1995). The reservoir is divided into five branches and 64 laterally averaged segments, each with up to 47 layers. The top layers through layer 31 are 0.5 m thick, layers 32 and 33 are 0.75 m thick, and the bottom layers 37 through 47 are 1.5 m thick. Inflows from tributaries are matched up with the layers whose density most closely corresponds to inflow density. The major outflow from the reservoir, the Cosgrove drinking water intake, is represented by segment 46 and segment 45 represents the remnants of a coffer dam installed during the construction of the intake. Water is withdrawn at this location in the model using the selective withdrawal algorithm, which calculates the layers from which water is taken based on total outflow, structure, elevation, and computed upstream gradients (Cole and Wells, 2015). Other outflows include releases to the Nashua River via a sleeve valve and a spillway (both located at model segment 44). During simulated precipitation events, additional water was released downstream to the Nashua River with consideration for maximum release limits a week prior to and for several days after the event until the water surface elevation returned to historical levels.

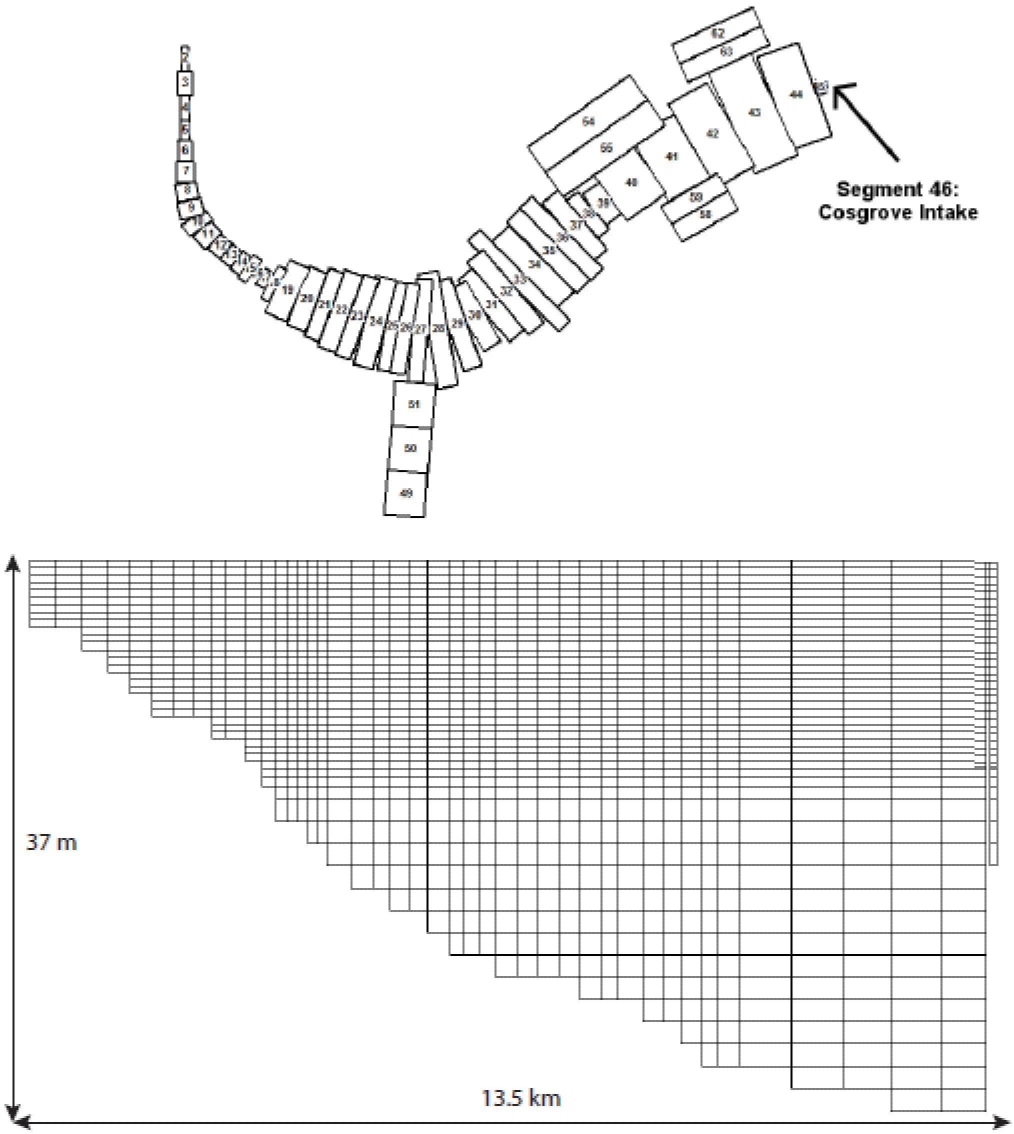

Figure 3.3 Wachusett Reservoir model grid

The hydrodynamic model algorithms simulate water surface elevations, velocities, and temperatures. The water quality constituents included in this application included total dissolved solids, dissolved organic matter (labile and refractory), particulate organic matter (labile and refractory), algae, phosphorus, ammonium, nitrate/nitrite, and dissolved oxygen. UV-254 (absorbance at a wavelength of 254 nm and a measure of the amount (and reactivity when normalized by DOC) of dissolved natural organic matter in source water) was modeled as a generic constituent with 1st order temperature dependent biochemical decay and an additional photodegradation decay rate that varies with depth in the water column, as described by Equation Equation 3.1,

$$\frac{d(UV254)}{dt} = -K_1\theta^{(T-20)}UV254 - \alpha I_0(1-\beta)e^{-\lambda z}UV254 \qquad \text{Equation 3.1}$$

where K_1 is the first order decay rate (sec^{-1}), θ is the temperature rate multiplier, T is the water temperature, α is the user defined photolysis coefficient, I_0 is the radiation at the water surface (W/m^2), β is the fraction of short wave solar radiation absorbed at the surface, λ is light extinction coefficient (m^{-1}), and z is the depth in the water column (m). The 1st order UV-254 decay rate (K_1) was set equal to the calibrated refractory organic matter decay rate (UV-254 is a surrogate for refractory organic matter) and a value for α, the new photolysis coefficient relating light induced decay to irradiance, was determined through calibration.

The CE-QUAL-W2 model was developed based on the years 2003-2012 using historical data for meteorology, bathymetry, inflow and outflow, water quality, initial flow and constituent conditions, and outlet descriptions. The ten years used to develop the model are a selection of the yearly hydraulic variability of the reservoir, representing a combination of wet and dry years. Water quality measurements taken at the intake and from in-situ profiles were used to calibrate and validate the model. Water quality profile measurements by the DCR include temperature, specific conductivity, pH, dissolved oxygen, and chlorophyll collected from the deepest portion of the reservoir (North Basin) approximately every other week throughout the year, except during ice cover. Additional profiles are measured less frequently at additional locations in the reservoir depending on weather conditions or specific needs. The phytoplankton ecology is monitored by the DCR in the reservoir at monthly intervals (or when necessary) and weekly at the intake. Additional water quality data are collected at the intake by the MWRA at variable temporal frequencies; temperature and specific conductivity are monitored continuously at 15 minute intervals while NH_4^+, NO_3^-, PO_4^{-3}, TOC, and UV-254 are monitored weekly.

Model simulations of temperature and specific conductivity are compared to measurements to verify adequate simulation of the reservoir's heat budget and the movement of non-reactive water quality constituents. The model was previously calibrated and validated for temperature and specific conductivity in a prior study (Jeznach and Tobiason, 2015). Additional water quality constituents were calibrated and validated to measurements taken at the Cosgrove drinking water intake and the root mean square error (RMSE) and absolute mean error (AME) of the final calibrated parameters are shown in Table 3.1. Algae were modeled as total algae with units of mg-C/L, where 1 mg/L is roughly equivalent to 1600 algal standard units (ASU) per mL, depending on the variety of algae in the particular sample. The calibrated model simulates the seasonal cycling and patterns of the ten hydraulically different years well, indicating that the chosen calibration parameters for the water quality variables are logical choices to generally capture the physical, chemical, and biological processes as a response to varying reservoir inputs and outputs.

Table 3.1 Measured and modeled water quality at the drinking water intake

Water quality parameter	**Measured Average (2003-2012)**	**Measured Range (2003-2012)**	**RMSE**	**AME**
Sp. Conductivity (µs/cm)	107	68 – 137	45	5
UV-254 (cm-1)	0.060	0.035 – 0.108	0.030	0.001
Algae (mg-C/L)	0.19	0.00 – 1.93	0.72	0.03
TOC (mg/L)	2.4	1.8 – 3.4	0.7	0.1
PO4 (mg/L)	0.006	0.003 – 0.020	0.014	0.001
NH4 (mg/L)	0.015	0.005 – 0.098	0.025	0.002
NO3 (mg/L)	0.098	0.011 – 0.200	0.086	0.006

3.5. Results and Discussion

The focus of the results presented in this study is that of the water quality impacts within the reservoir, particularly at the location of the drinking water intake. Extreme event flows as well as nutrient and organic matter load inputs from the watershed to the reservoir for these particular event scenarios are discussed in Hagemann et al (2016). Given the characterization of Wachusett Reservoir as oligo-mesotrophic, measured nutrient (nitrogen and phosphorous) concentrations as well as algae growth are low (Table 3.1). Drinking water treatment does not include coagulation and particle separation to remove natural organic matter (NOM), a measure of DPB precursors. UV-254 is frequently monitored at the drinking water intake because it is effective for estimating ozone doses and it can serve as a trigger to initiate Quabbin Reservoir transfer inflows to Wachusett Reservoir to reduce organic matter content. Therefore, in-reservoir extreme event scenario water quality in this study was evaluated with a focus on organic matter, since treatment costs and performance for this particular system are sensitive to fluctuations in organic matter.

Results are discussed in the context of the maximum spring and summer water constituent concentrations at the drinking water intake during the base year 2011 (Table 3.2), with each concentration representing a single measurement for a particular month. Rainfall events preceding constituent measurements had recurrence intervals of less than 2 years, with the exception of the maximum total algae measured on April 19, which occurred 2 days after a rain event on April 17 with a return interval of approximately 2 years. The spring in this particular watershed is characterized by snowmelt, spring precipitation in the form of rain, and high flows. The reservoir hydrodynamics during this time of year are similar to that of a complete mixed reactor, with unstratified conditions, low demand, and usually no (or very little) inter-reservoir transfer inflows from the Quabbin Reservoir. Although the theoretical mean hydraulic residence time (MHRT) of Wachusett Reservoir based on volume and throughput flow is approximately 200 days, previous modeling studies estimate that the travel time of constituents in the spring from the eastern end of the reservoir (where tributaries are located) can range from 2-7 days on average (Jeznach et al., 2014). In comparison, model simulations suggest that the summer travel times range from 5-15 days, on average, due to the thermally stratified nature of the reservoir during this time of year (Jeznach et al., 2014). The summer is also characterized by higher water demands offset by increased inflows from the Quabbin Reservoir primarily to maintain yield but also to dilute natural organic matter levels in Wachusett and generate hydropower. Nutrients are typically lower in the reservoir during the summer due to increased biological activity.

Table 3.2 Measured maximum concentrations from the calendar year 2011

Constituent	Spring Maximum	Summer Maximum	Year Maximum
UV-254 (cm-1)	0.073 (May 31)	0.077 (June 13)	0.093 (October 3)
Total Algae (mg-C/L)	0.4 (April 19)	0.5 (June 23)	0.5 (June 23)
TOC (mg/L)	2.3 (April 4)	2.7 (August 1)	2.89 (December 5)
PO4 (mg/L)	0.010 (May 2)	0.006 (July 11)	0.010 (May 2)
NH4 (mg/L)	0.013 (March 7)	0.024 (July 11)	0.024 (July 11)
NO3 (mg/L)	0.115 (May 2)	0.103 (June 6)	0.115 (May 2)

3.5.1. Spring precipitation event

The histograms in Figure 3.4 show the frequency distributions of maximum simulated concentrations at the drinking water intake for the three event magnitudes (101, 152, and 203 mm) resulting from the simulated tributary concentration input samples (100 samples per precipitation depth per season) following the spring event (Julian day 106, April 16 2011). In the spring scenarios, TOC and UV-254 concentrations exceeded 2011 measured springtime values. The maximum springtime TOC measurement was 2.3 mg/L at the intake on May 31 and the maximum value measured during the year 2011 was 2.9 mg/L on December 5. For the modeled scenarios, the median simulated peak TOC concentrations at the Cosgrove for a 101, 152, and

203 mm precipitation scenarios fell below 2.66, 2.76, and 2.83 mg-C/L, respectively. For a 101 mm precipitation event occurring over 24 hours, the range of simulated concentrations of TOC at the intake were between 2.53 and 2.90 mg-C/L, with the maximum concentration simulated for this event approximately reaching the 2011 measured maximum. Simulated TOC concentrations at the intake for an 203 mm event had a greater spread compared to the 101 mm event, as seen in Figure 3.4, with concentrations ranging from 2.54 to 3.37 mg-C/L. TOC concentrations increased within 3 days of the event, but the peak concentration did not occur until June 19, 56 days after the event occurrence. The maximum UV-254 springtime measurement was 0.073 cm^{-1} measured on May 31 2011 and the maximum measurement in 2011 was 0.093 cm^{-1} on October 3. For UV-254, the median simulated peak concentrations at the Cosgrove intake for a 101, 152, and 203 mm scenarios were 0.081, 0.091, and 0.099 cm^{-1}, respectively, which occurred 13 days after the event on April 29. For a 101 mm event occurring over 24 hours, the range of simulated concentrations at the intake was between 0.074 and 0.092 cm^{-1}, with the minimum concentration for this event size just above the maximum springtime 2011 measurement of 0.073 cm^{-1}. The spread of simulated concentrations for a 203 mm precipitation event in comparison is greater (0.083 – 0.128 cm^{-1}) indicating less certainty about UV-254 concentrations for this larger event.

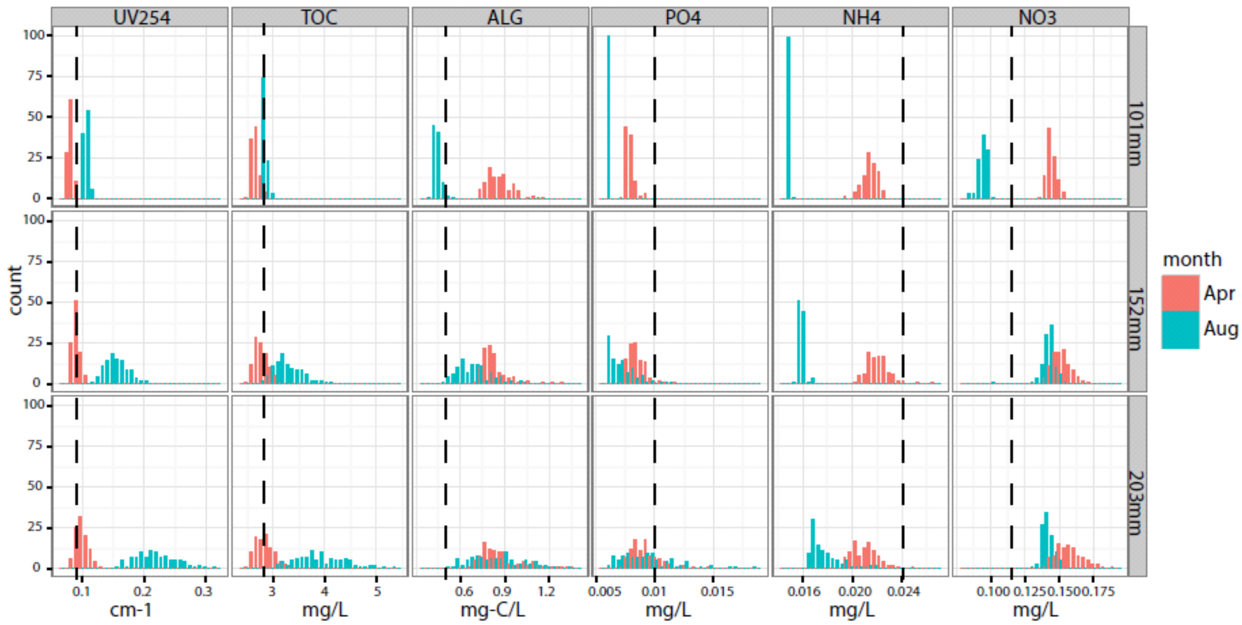

Figure 3.4 Maximum concentrations at the drinking water intake following the precipitation event scenarios (2011 year maximums noted by dotted lines)

A time series of TOC and UV-254 concentrations at the intake in the days leading up to the event and following the event is shown in Figure 3.5. In this figure, the solid line represents the median simulated concentration, the dark grey area is the first and third quartiles of the concentrations, and the bounds of the light grey area indicate the full range of simulated concentrations from a given precipitation depth. With increasing precipitation depth, the median concentration increases and the range of concentrations also increases, due to uncertainties associated with predicting concentrations during larger events. Impacts of a large event were simulated to last over a year, as indicated by the duration for which a range of concentrations at the intake is simulated (grey shaded area).

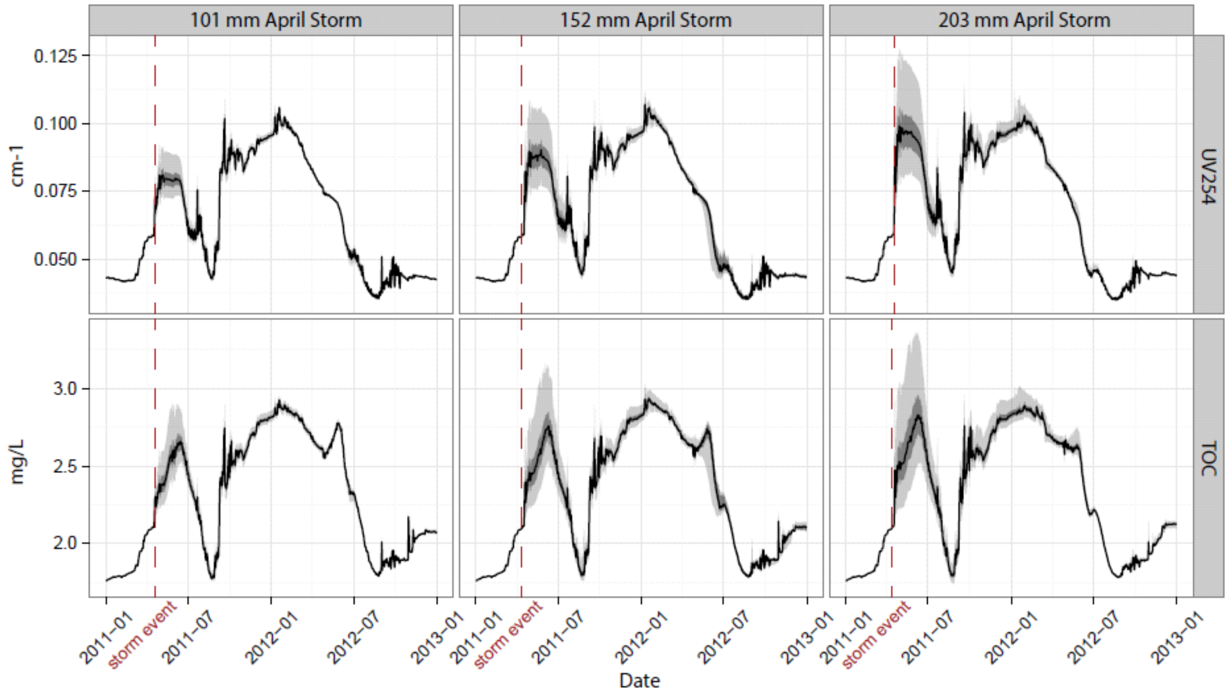

Figure 3.5 Time series of TOC and UV-254 concentrations at the drinking water intake in the days leading up to and following the spring precipitation event on April 16 (black, dark gray, and light gray regions represent the median, 25/75th percentile, and the range

The maximum algae measurement in the spring of 2011 was 0.4 mg-C/L (640 ASU/mL of total algae), on April 19 and the maximum recorded measurement for the year was 0.5 mg-C/L (800 ASU/mL) on June 23. Model simulations for spring extreme precipitation event scenarios indicated algae concentrations would likely exceed the maximum 2011 measurements. For a 101, 152, and 203 mm precipitation event, the median simulated algae peak concentrations were 0.86, 0.81, and 0.84 mg-C/L respectively. For a 203 mm precipitation event, the maximum simulated concentration was 1.37 mg-C/L, occurring 33 days after the event on May 19.

Contrary to the organic matter and algae results, it was less likely that extreme precipitation event scenarios resulted in concentrations of nutrients at the intake that equaled or exceeded the maximum 2011 measurements (Table 3.2). Nutrient concentrations in watershed tributaries are typically low and were simulated to remain low during imposed event scenarios (Hagemann et al, 2016). Nutrient loads from the watershed to the reservoir were simulated to increase with increasing precipitation depth due to the increased tributary flow volumes; however, reservoir model simulations suggest these increased loads and associated in situ processes that contribute to reservoir nutrient concentrations will result in concentrations remaining low. The maximum measurement of PO_4^{-3} during the year 2011 was 0.010 on May 2. For 101, 152, and 203 mm precipitation events, the simulated maximum PO_4^{-3} intake concentrations were 0.008, 0.008, and 0.009 mg/L, respectively. The maximum measurement of NH_4^+ during 2011 was 0.024 on July 11 whereas the maximum simulated event NH_4^+ maximum concentrations were 0.022, 0.022, and 0.021 mg/L, respectively. The maximum measurement of NO_3^- in the spring and in the year 2011 was 0.115 mg/L, measured on May 2. For 101, 152, and 203 mm events, the greatest NO_3^- maximum concentrations simulated to occur at the intake were simulated to be 0.14, 0.15, and 0.16 mg/L, respectively, which were slightly above the maximum measurement but not at levels of concern. Peak concentrations of PO_4^{-3}, NH_4^+, and NO_3^- occurred 3, 42, and 3 days after the start of the event.

3.5.2. Summer precipitation event

Maximum concentration histograms for constituents resulting from summer precipitation events on August 28 (Julian day 240) are shown in Figure 3.4. Similar to the spring event results, TOC and UV-254 concentrations at the intake became elevated to levels higher than the 2011 summertime maximums. The maximum TOC measurement in the summer of 2011 and on record was 2.7 mg/L on August 1 and the maximum measurement during the year was 2.89 mg/L on December 5. For 101, 152, and 203 mm summer precipitation events, the median of simulated TOC maximum concentrations were 2.8, 3.2, and 3.9 mg-C/L, respectively, with the 203 mm event concentrations exceeding the record measurement 12 days after the event.. Simulated TOC concentrations at the intake for a 203 mm event had a much greater spread compared to the 101 mm event, as seen in Figure 3.4, with concentrations ranging from 3.02 to 5.36 mg-C/L. TOC concentrations increased within 5 days of the event, but the peak concentration did not occur until September 19, 22 days after the event occurrence, as seen in Figure 3.6. The maximum UV-254 measurement in the summer was 0.077 cm^{-1} on June 13 and the maximum measurement during the year 2011 was 0.093 on October 3. In the 101, 152, and 203 mm events, 50% of the maximum UV-254 concentrations at the intake were below 0.103, 0.151 and 0.213 cm^{-1}, respectively, with all scenarios exceeding the summertime and year maximum measurement. The simulated peak concentration was on Sept 9, 12 days after the event. Similar to a spring event, increased UV254 and TOC concentrations after a large summer event were simulated over one year after the event occurrence.

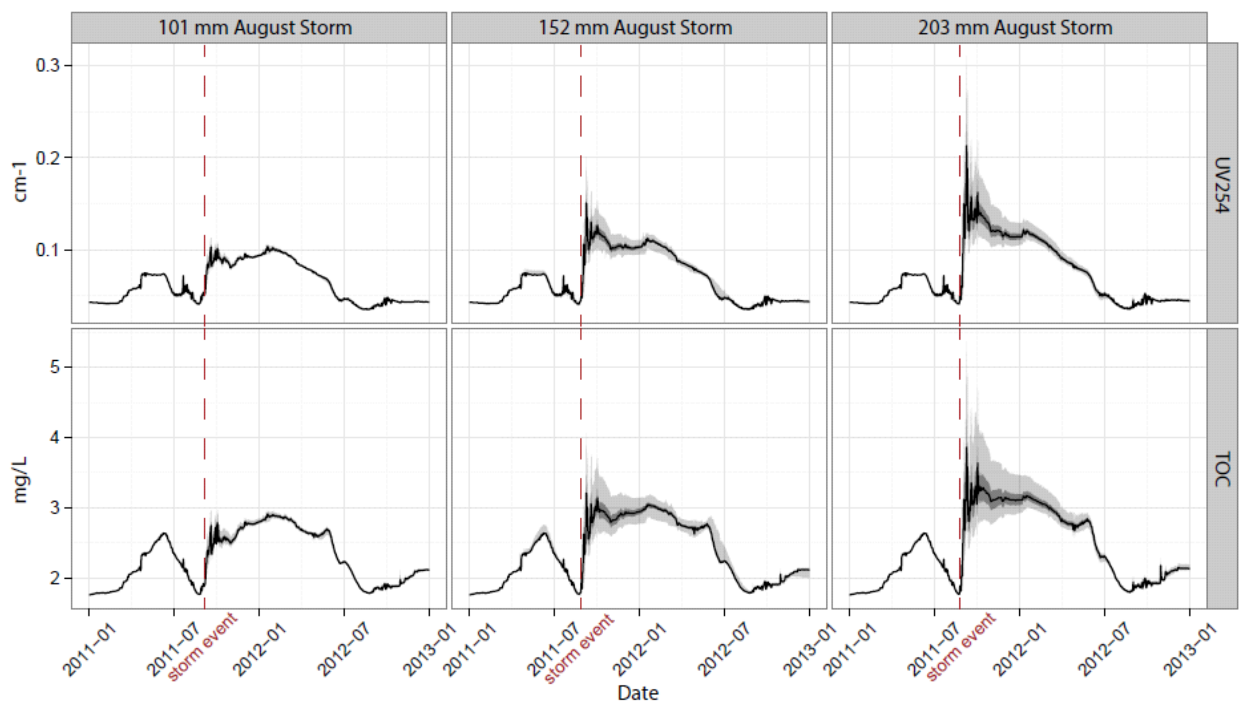

Figure 3.6 Time series of TOC and UV-254 concentrations at the drinking water intake in the days leading up to and following the summer precipitation event on August 28 (black, dark gray, and light gray regions represent the median, 25/75th percentile, and the ran

The maximum algae measurement during the summer and year of 2011 was 0.5 mg-C/L (800 ASU/mL total algae) on June 23. For 101, 152, and 203 mm events, the median simulated maximum total algae concentrations at the Cosgrove were 0.4, 0.7, and 0.8 mg-C/L, respectively, with the 152 and 203 mm events exceeding the maximum 2011 measurement. The peak algal concentrations during the summer event occurred on September 30, 33 days after the start of the event.

None of the event scenarios simulated during the summer resulted in PO_4^{-3} or NH_4^+ concentrations that equaled or exceeded the 2011 maximum measurements on record, as shown in Table 3.2. For a 101, 152, and 203 mm event in the summer, the simulated maximum PO_4^{-3} concentrations at the intake were 0.006, 0.007, and 0.009 mg/L, respectively, occurring 12 days after the event. The maximum NH_4^+ concentrations at the intake during the simulated events occurred on October 3, 36 days after the event start. The simulated NH_4^+ maximum concentrations were 0.015, 0.016, and 0.017 mg/L for a 101, 152, and 203 mm precipitation event, respectively. The maximum NO_3^- concentration at the intake during the simulated precipitation events occurred on September 6, 9 days after the event start. The median simulated NO_3^-concentration from precipitation events exceeded the 2011 summer and year maximum measurements. For 101, 152, and 203 mm events, the maximum simulated NO_3^- concentrations were 0.094, 0.142, 0.140 mg/L, respectively.

3.5.3. Impacts of management decisions

The calibrated hydrodynamic and water quality model can be used to evaluate the effectiveness of various management decisions that can be made if an extreme precipitation event were to occur. Management decisions that may improve water quality after a precipitation event will be case specific; however, the following is an example analysis for the Wachusett Reservoir illustrating the concept.

Elevated levels of organic matter in the reservoir, and particularly at the drinking water intake, are of concern to Wachusett Reservoir managers, while nutrients remain low throughout the year and are relatively stable. UV-254 is measured weekly and is a useful parameter for predicting oxidant demand, chlorine decay and DBP formation. Average UV-254 in Wachusett is about 0.06 cm-1 and average TOC is 2.4 mg/L. Elevated levels of UV-254 lead to increased chlorine and ozone demand resulting in the need for higher doses, which can increase costs and disinfection byproduct formation. Additionally, measurements of UV-254 in Wachusett can be important for timing the operation of the Quabbin Reservoir transfers to lower organic matter levels and optimize treatment results (Sung, 2003). In July 1998 and 2000, the MWRA observed dramatic decreases in chlorine demand and decay kinetics 2 weeks after Quabbin transfers were initiated (due to elevated organic matter levels), which further resulted in a decrease in chlorine dose and DBP formation (Sung, 2003).

Therefore, operational decisions should consider transferring water from the Quabbin Reservoir in an extreme precipitation event in order to lower resulting organic matter levels. However, decisions regarding inter-reservoir transfers must also weigh competing objectives, particularly the minimization of downstream flooding into the Nashua River, which is a large concern during large precipitation events. For example, the operational strategy employed during Hurricane Irene in August 2011 considered both of these objectives. Before this event, the Quabbin transfer was flowing (as typical of August conditions) but in anticipation of the forecasted hurricane, flow was stopped on August 25 and the water surface elevation was allowed to decrease to provide storage for storm inflows. Precipitation began on August 27 and predominantly on the

28. After the event, the Quabbin transfer inflow was initiated briefly from September 1 – 6 and stopped once again until September 20 when it resumed as normal operation. Water was also released downstream to the Nashua River before, during, and after the event to control the water surface elevation with care to minimize downstream flooding.

Water quality measurements were taken after Hurricane Irene at the intake on September 6, as shown in Table 3.3. These measurements can be compared to the September 6 model outputs from a scenario simulating a 101 mm precipitation event in August, since this is essentially recreating Hurricane Irene, but with inflows and concentrations generated with the method described previously. In the model scenario, the operation of the Quabbin transfer was unaltered from the historic operation during Hurricane Irene. The resulting simulated constituent concentrations from this event, in Table 3.3, are generally in line with the measured concentrations on September 6. UV-254 predictions are the most similar to the measurements while TOC, PO_4^{3-}, NH_4^+, and NO_3^- are slightly underpredicted and total algae is overpredicted. However, algae measurements several days after Sept 6 were recorded as 0.14 mg-C/L, which is very close to the simulated concentrations predicted by the model scenario.

Table 3.3 Simulated and measured water quality constituents on September 6th 2011 during 101 mm extreme event in the summer with two different management options

Constituent	Quabbin flow turned ON		Quabbin flow OFF		Measured after Hurricane Irene
	Median	range	Median	range	
UV-254 (cm-1)	0.0517	0.050 – 0.053	0.066	0.064 – 0.0694	0.054
Algae (mg-C/L)	0.13	0.13 – 0.14	0.07	0.07 – 0.08	0.04
TOC (mg/L)	1.90	1.88 – 1.93	2.08	2.03 – 2.16	2.2
PO4 (mg/L)	0.0037	0.0036 – 0.0037	0.0050	0.0048 – 0.0055	0.007
NH4 (mg/L)	0.0060	0.0060 – 0.0061	0.0080	0.0079 – 0.0081	0.008
NO3 (mg/L)	0.0352	0.0342 – 0.0363	0.1365	0.1343 – 0.1396	0.071

Additional scenarios were simulated to evaluate the effect of not intermittently operating the Quabbin transfer before, during, and after the storm. In this scenario, inflows from the Quabbin were stopped August 25 and resumed September 20. When Quabbin water was not intermittently operated, concentrations of all water quality constituents on September 6 increased at the intake, with the exception of total algae, as seen in Table 3.3. Increases in concentrations ranged from about 100% to almost 400% greater than original simulated concentrations when Quabbin water was transferred following the event. These results are consistent with measured observations of lowered concentrations at the intake after initiating Quabbin transfer water to lower organic matter content in Wachusett.

3.6. Conclusions

Extreme precipitation events can have potentially large impacts on source water quality for drinking water supplies but relatively little is known about the nature and extent of these impacts for various water quality constituents. The work described in this manuscript coupled with a watershed input model simulation in Part 1 can be used to quantify the range of potential water quality impacts of these low probability events. The method is valuable for understanding more about the interactions between watersheds and receiving waterbodies to proactively develop scientifically based management decisions during extreme precipitation events, which can in turn improve treated water quality and ultimately reduce treatment costs.

Wachusett Reservoir, in central MA, was used to illustrate the method to predict raw water quality for extreme precipitation events occurring in the spring and summer. In the simulated spring event, TOC, UV-254, and algae concentrations exceeded the springtime records. In comparison, summer event TOC, UV-254, and algae concentrations were greater than those resulting from the spring event, exceeding 2011 maximum summertime measurements with the exception of the 101 mm event. Nutrient concentrations after the spring and summer events remained relatively low and did not exceed maximum measurements, with the exception of NO_3^- after the 152 and 203 mm events. When the observed water quality impacts of Hurricane Irene were compared to simulated concentrations at the intake resulting from a summer event of the same magnitude (101 mm or 4 inches), the results generally show similar concentrations at the intake, indicating this method is a reasonable way to predict water quality impacts.

Maintaining low organic matter concentrations is imperative for this drinking water source in order to decrease costs and minimize disinfection byproduct formation. The level of reactive organic matter (measured by UV-254) at the intake tends to increase every year between April and June, which is related to the biological activity within the watershed, resulting in higher NOM loadings. Algae growth within the reservoir during this time also has an impact on the nature and amount of NOM. A management strategy to lower NOM levels in Wachusett Reservoir is to dilute the organics by initiating transfers of water with lower NOM from the Quabbin Reservoir. This strategy was employed during Hurricane Irene in August 2011 and model simulations comparing intake concentrations to scenarios where water was not transferred via the Quabbin aqueduct illustrate the negative impacts of a management decision to not initiate a transfer indicated by increased constituent concentrations. Therefore, model simulations verify the effectiveness of the decisions made by management during this recent extreme event to decrease negative water quality impacts and simulations will likely be valuable information for understanding the impacts of future events.

3.7. Acknowledgements

This project was supported by the Massachusetts Department of Conservation and Recreation. The findings are the opinions of the authors and do not represent the official findings of the DCR or MWRA.

CONCLUSION

The primary goal of this dissertation was to present proactive frameworks utilizing a hydrodynamic and water quality model to aid in developing scientifically-based management plans prior to an accidental or natural event occurring. The Wachusett Reservoir, in central Massachusetts, was used as a case study to illustrate proactive modeling efforts to quantify water quality impacts after both short and long-term potential events of concern. This work used a process-based modeling approach to simulate reservoir hydrodynamic and water quality responses to changes in various model inputs (streamflow, constituents, meteorology) and also evaluated current and future management decisions which may improve water quality.

A contaminant spill modeling framework for drinking water reservoirs was developed to assess contaminant impacts and management responses on surface drinking water sources prior to an event occurrence (Chapter 1). Prior detailed modeling efforts and scenario evaluations improve the understanding of contaminant plume fate and transport, including potential maximum concentrations that could occur at the drinking water intake and contaminant travel time to the intake after an event. In the example study of the Wachusett Reservoir, modeled contaminant scenarios of a hypothetical fecal coliform input from a sewage overflow and an ammonium nitrate spill from a tanker truck highlighted the importance of a rapid management response to contain a contaminant spill in order to minimize the mass of contaminant that enters the water column. Modeling efforts have also guided the placement of additional in-reservoir monitoring devices, based on velocity profiles to detect changes in water quality and provide early warning to water quality concerns at the drinking water intake.

Hydrodynamic and water quality models are also valuable tools for proactively evaluating and managing water quality impacts of the short and long-term impacts of climate change. The first logical step in any larger climate study investigating water quality impacts of climate change is to simulate the impacts of increasing air temperatures on water temperatures, since water temperature influences many physical, chemical, and biological processes within a water body. Scenarios investigating water temperature sensitivity to increasing air temperatures in the Wachusett Reservoir watershed indicate that water temperatures changes are not one-for-one with air temperature increases and that seasonal variability in water temperatures is linked to winter ice cover (Chapter 2). A more challenging question is the potential effects of the combined changes in tributary inflow volumes, timing, and water quality as well as changes in meteorology due to climate change, especially those changes related to extreme events. Hydrodynamic and water quality modeling in conjunction with statistically generated watershed inputs simulating extreme precipitation events is one approach to try to quantify water quality impacts in a receiving water body (Chapter 3). Future impacts of these low-probability events to raw drinking water quality are highly uncertain, but the method presented in this work is one approach to quantifying and characterizing the potential impacts in an effort to proactively develop scientifically based management plans.

The work presented in this dissertation provides several examples of how hydrodynamic and water quality models can be effective tools for proactive water quality management. The work also reveals a number of future research needs that will become increasingly important to drinking water management due to increasing watershed urbanization and climate change impacts around the globe.

In general, there is a great need for better science guiding contaminant response efforts in drinking water sources. Recent events, such as the West Virginia chemical spill, have highlighted both the lack of information on the chemical properties, fate, and transport as well as deficiencies in risk management planning. Contaminant storage and transport is often in very close proximity to freshwater drinking sources, yet there is little information on the fate and transport of these contaminants in freshwater systems. For example, crude oil is frequently transported via roadways and railways near drinking water sources, yet of the few fate and transport studies of crude oil, almost all have been for oil spills in the marine environment. Fundamental fate and transport studies of contaminants in freshwaters stored near drinking water supplies would improve site-specific emergency and risk management planning for utilities.

In addition, future studies quantifying water quality impacts of climate change, particularly extreme events, would be improved with more frequent and spatially diverse measurements of water quality parameters in both watersheds and surface water bodies. Greater frequency of measurements across varying areas would better capture the short and long-term variability in concentrations and would improve current abilities to model the physical, chemical, and biological processes in watersheds and lakes. This is particularly important for low-probability extreme events, where data for heavy precipitation events (the top 1%) is limited but constituent loads (particularly organic matter) have been observed to have significant impacts on water quality.

APPENDIX A

MODEL CALIBRATION AND VALIDATION

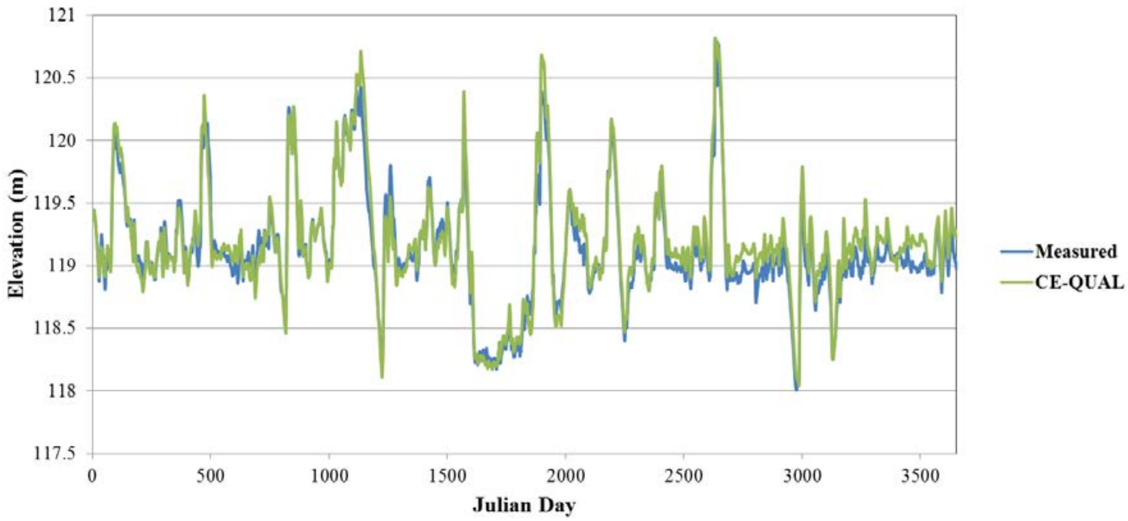

Figure A.1 Measured and modeled water surface elevation (2003-2012)

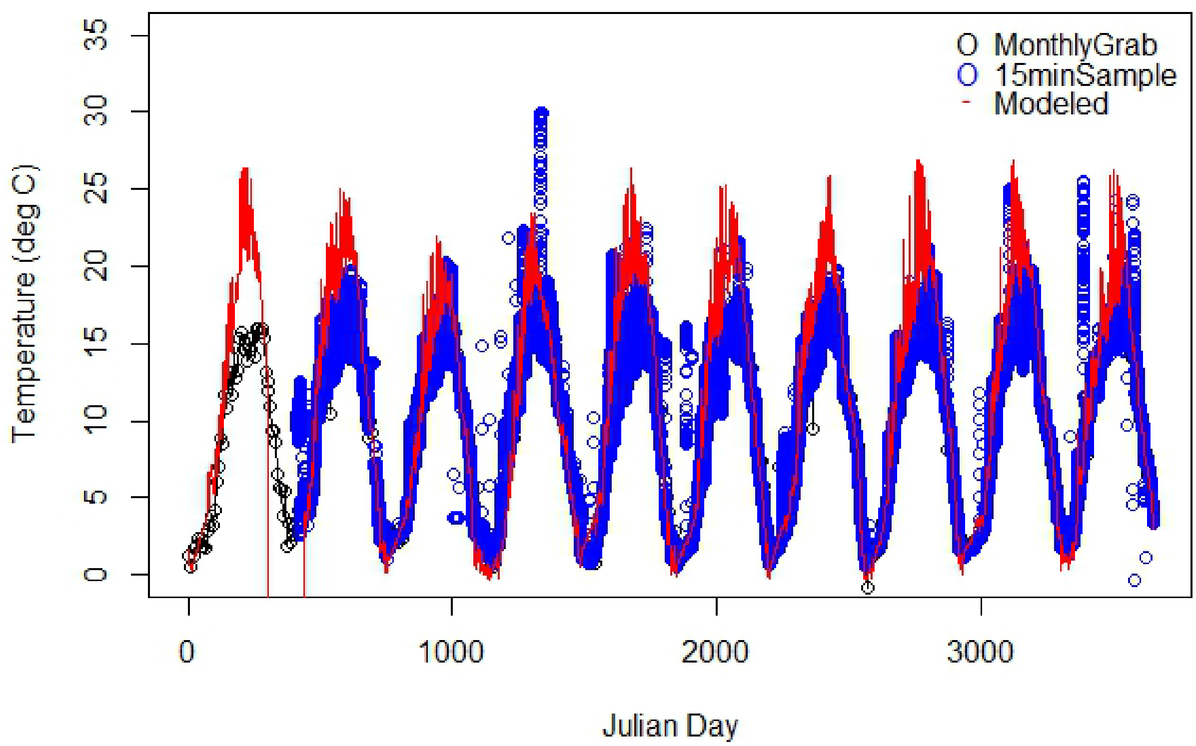

Figure A.2 Measured and modeled water temperature at the Cosgrove drinking water intake (2003-2012)

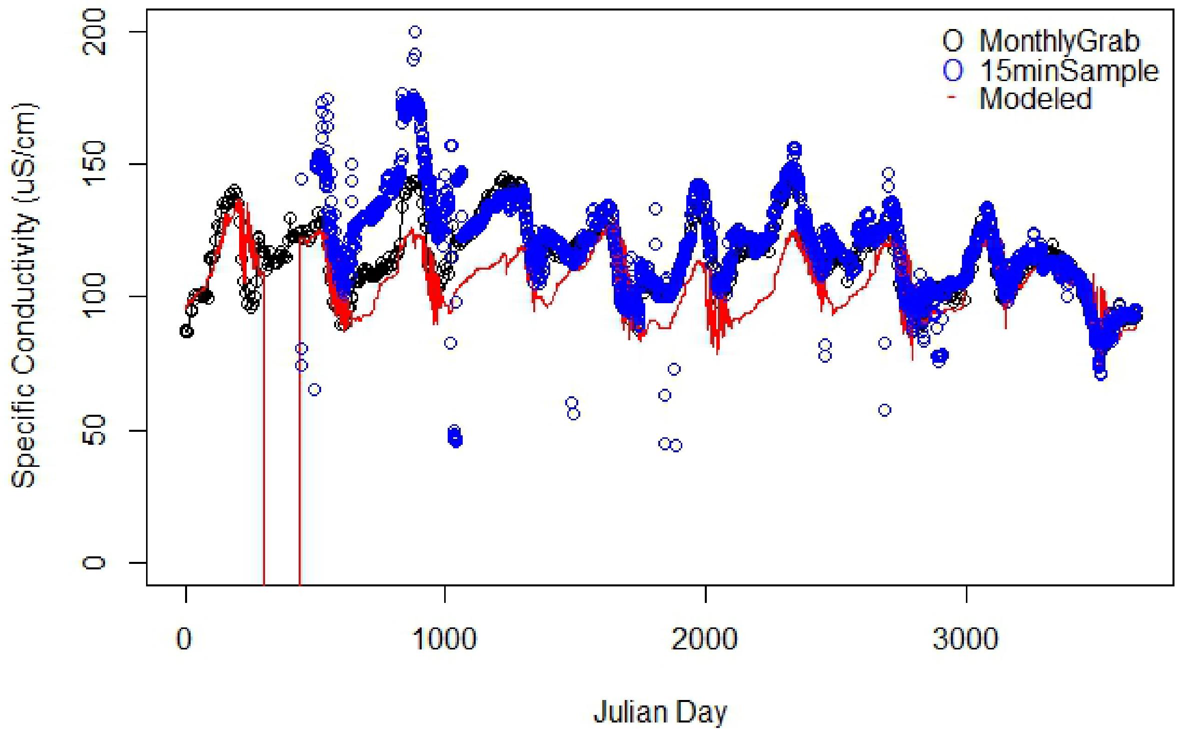

Figure A.3 Measured and modeled specific conductivity at the Cosgrove drinking water intake (2003-2012)

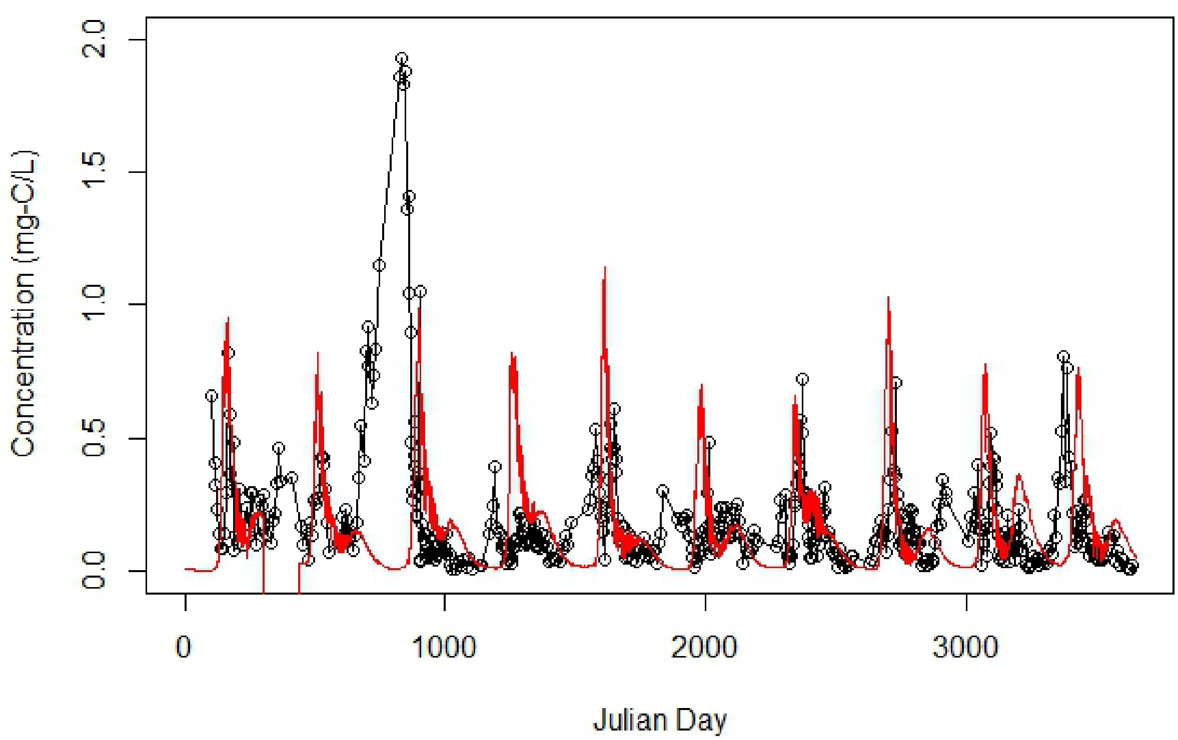

Figure A.4 Measured and modeled total algae at the Cosgrove drinking water intake (2003-2012)

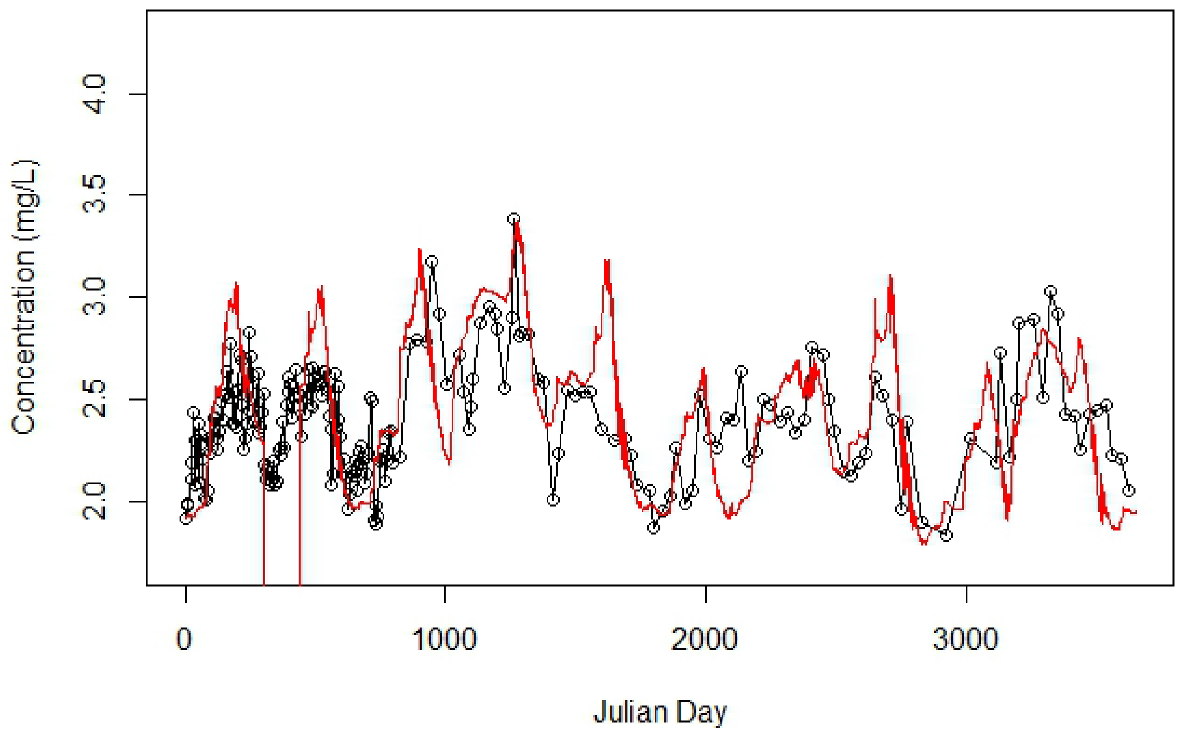

Figure A.5 Measured and modeled TOC at the Cosgrove drinking water intake (2003-2012)

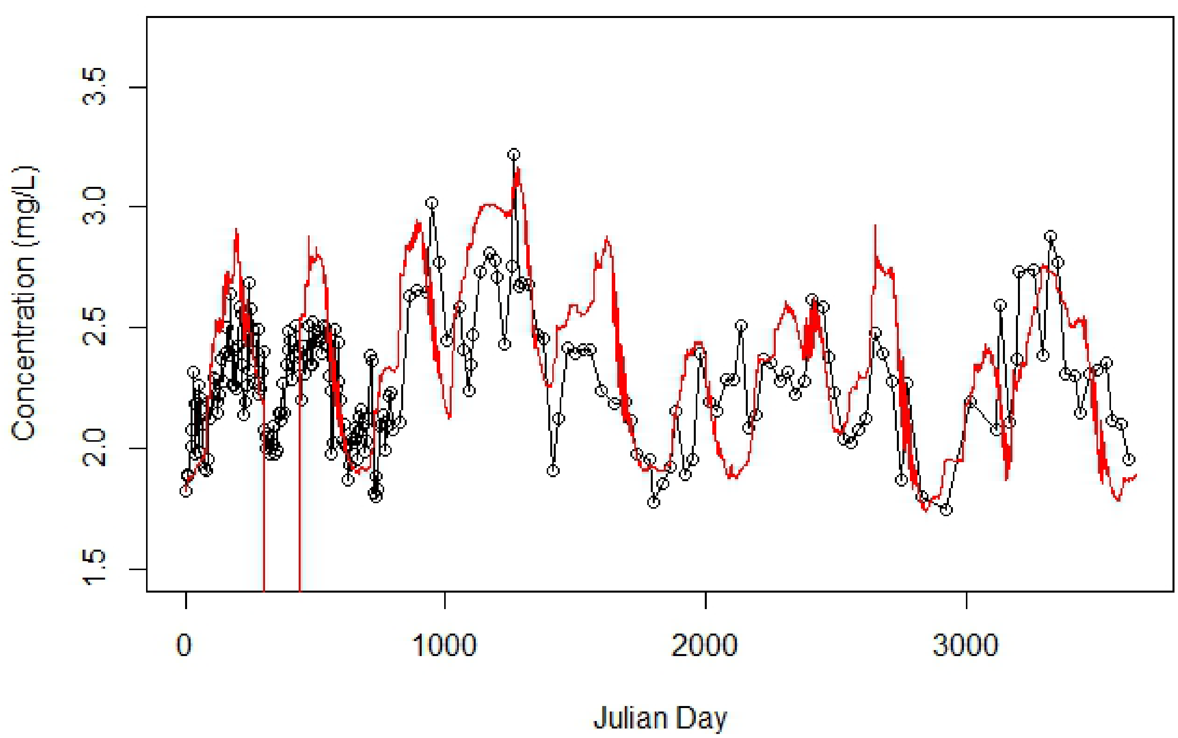

Figure A.6 Measured and modeled DOC at the Cosgrove drinking water intake (2003-2012)

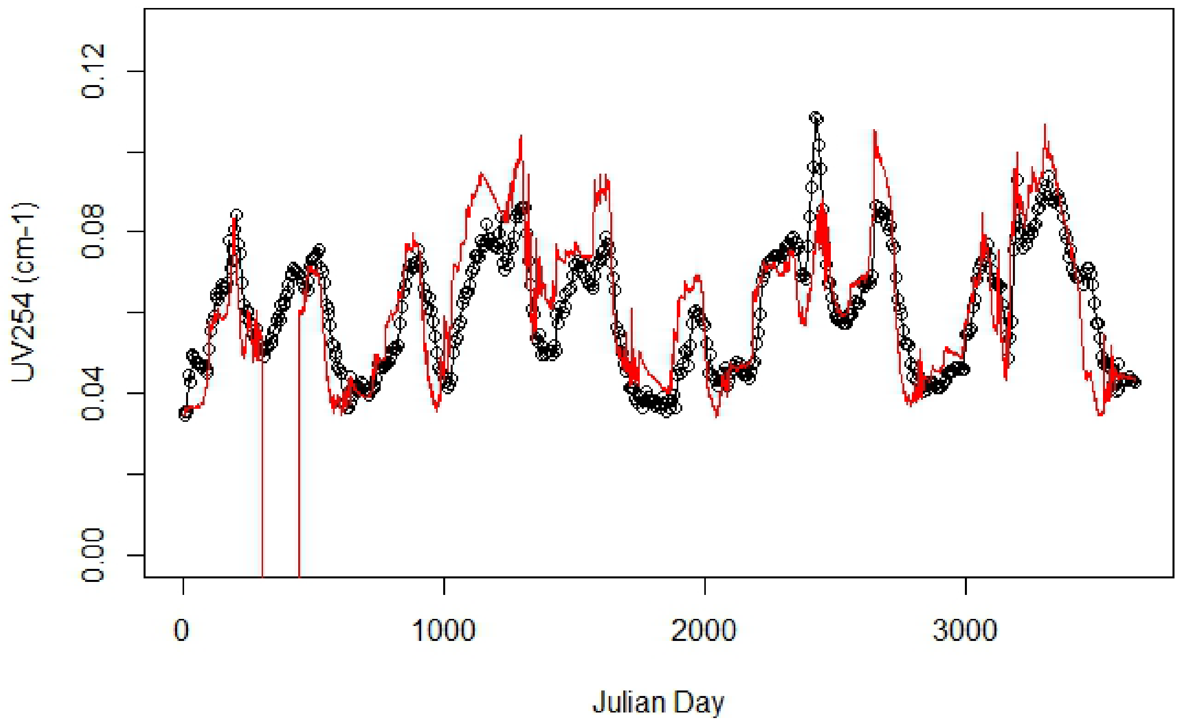

Figure A.7 Measured and modeled UV-254 at the Cosgrove drinking water intake (2003-2012)

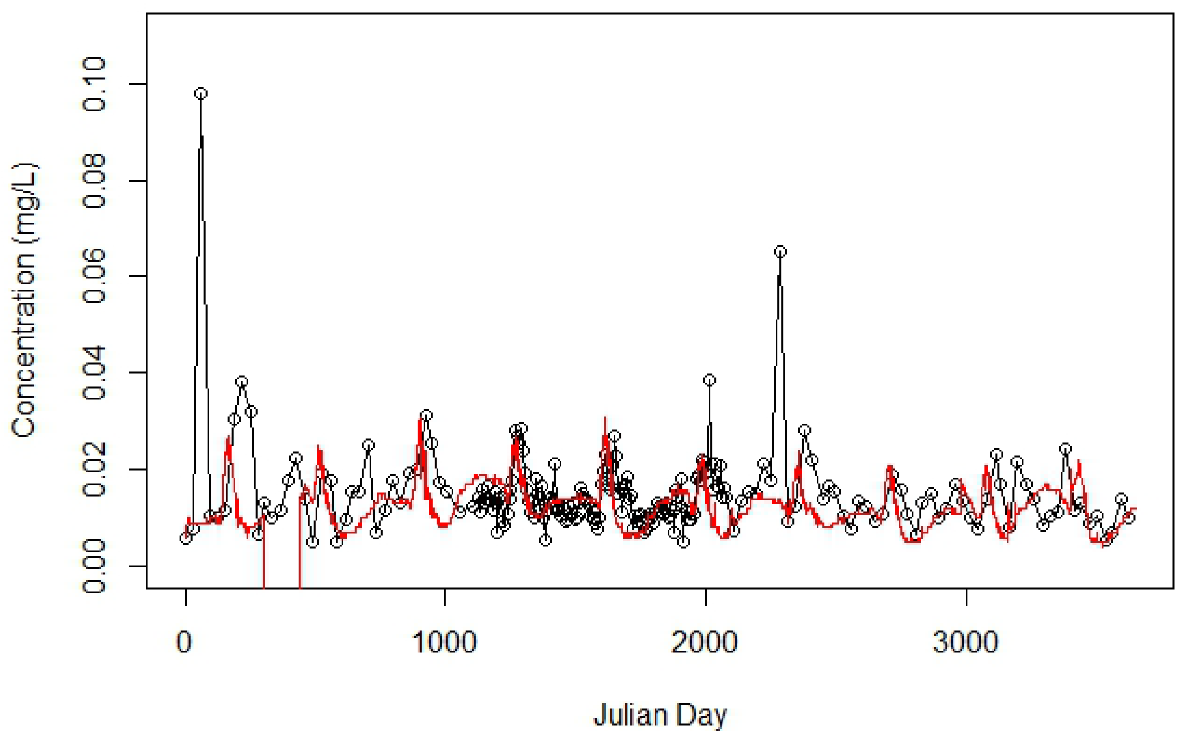

Figure A.8 Measured and modeled NH_3-N at the Cosgrove drinking water intake (2003-2012)

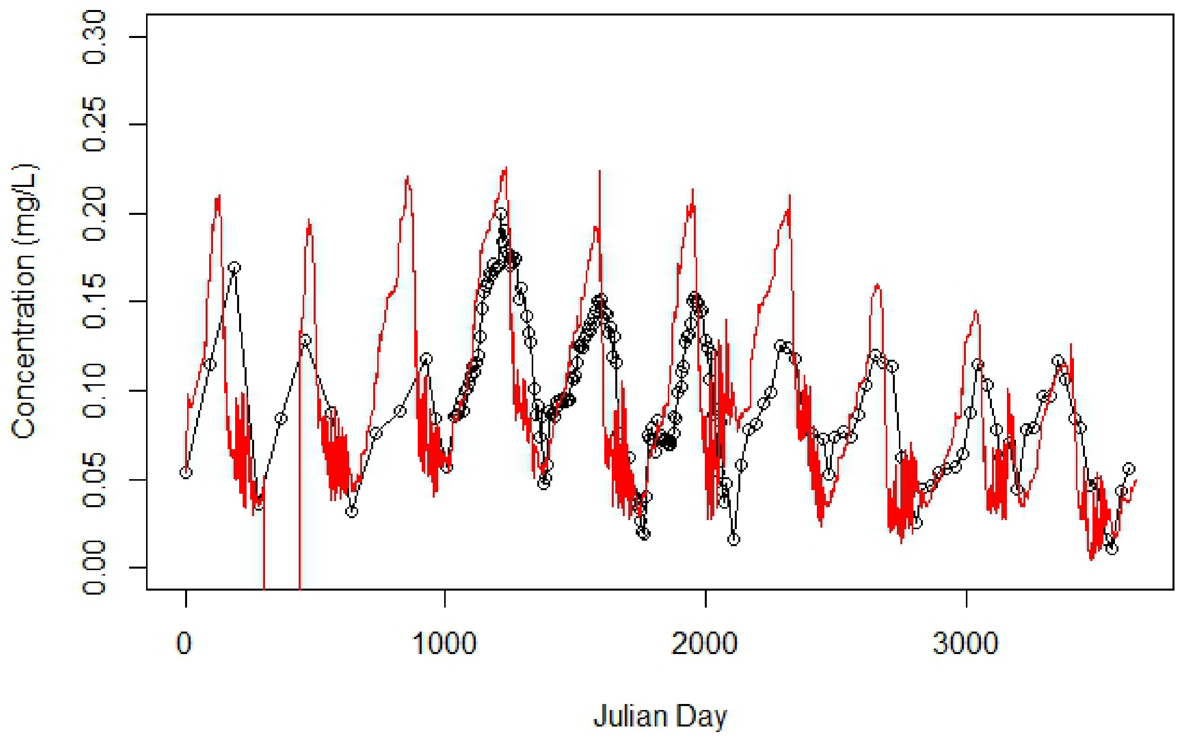

Figure A.9 Measured and modeled NO_3-N at the Cosgrove drinking water intake (2003-2012)

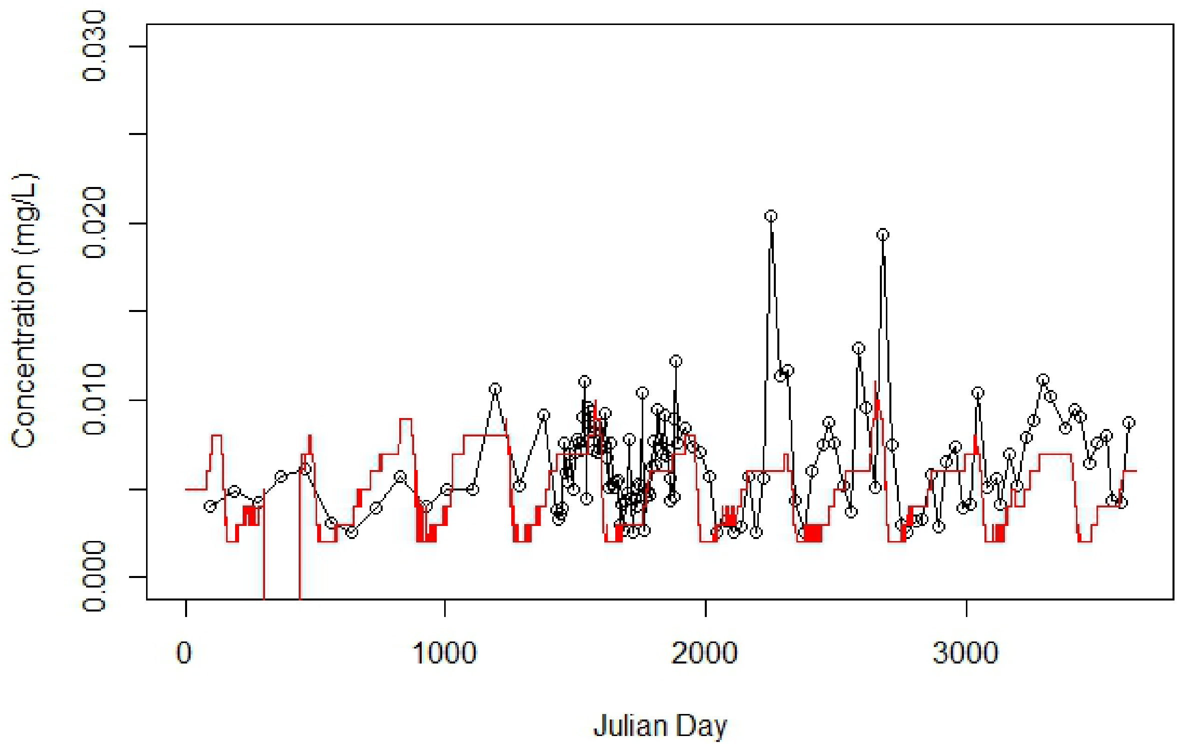

Figure A.10 Measured and modeled PO_4^{3-} at the Cosgrove drinking water intake (2003-2012)

REFERENCES

Adrian, R., O'Reilly, C.M., Zagarese, H., Baines, S.B., Hessen, D.O., Keller, W., Livingstone, D.M., Sommaruga, R., Straile, D., Van Donk, E., Weyhenmeyer, G.A., Winder, M., 2009. Lakes as sentinels of climate change, Limnol. Oceanogr. 54, 2283-2297.

Ahlfeld, D., Joaquin, A., Tobiason, J., Mas, D., 2003. Case study: Impact of reservoir stratification on interflow travel time, Journal of Hydraulic Engineering 129, 966-975.

Anandhi, A., Frei, A., Pierson, D.C., Schneiderman, E.M., Zion, M.S., Lounsbury, D., Matonse, A.H., 2011. Examination of change factor methodologies for climate change impact assessment, Water Resour. Res. 47.

Arhonditsis, G.B., Brett, M.T., DeGasperi, C.L., Schindler, D.E., 2004. Effects of climatic variability on the thermal properties of Lake Washington, Limnol. Oceanogr. 49, 256-270.

Bahadur, R., Samuels, W.B., 2014. Modeling the Fate and Transport of a Chemical Spill in the Elk River, West Virginia, J. Environ. Eng. 141.

Baker, M.B., Peter, T., 2008. Small-scale cloud processes and climate, Nature 451, 299-300.

Barry, M., Chiu, C., Westerhoff, P., 2016. Severe weather impacts on water quality in Central Arizona, JOURNAL - American Water Works Association 108, E221-E231. doi: 10.5942/jawwa.2016.108.0027.

Baum, R., Bartram, J., Hrudey, S., 2016. The Flint Water Crisis Confirms That US Drinking Water Needs Improved Risk Management, Environ. Sci. Technol.

Brown, C., Ghile, Y., Laverty, M., Li, K., 2012. Decision scaling: Linking bottom-up vulnerability analysis with climate projections in the water sector, Water Resour. Res. 48, W09537.

Brown, C., Wilby, R.L., 2012. An alternate approach to assessing climate risks, Eos, Transactions American Geophysical Union 93, 401-402.

Bryan, D.P., 2004. Investigation of Disinfection Byproduct Precursors in the Wachusett Reservoir Watershed, M.S. Thesis, Department of Civil and Environmental Engineering, University of Massachusetts Amherst.

Buttrick, D.R., 2005. Modeling Natural Organic Matter in an Unfiltered Surface Water Supply, M.S. Thesis, Department of Civil and Environmental Engineering, University of Massachusetts Amherst.

Camp, Dresser, McKee, 1995. Wachusett Reservoir Water Treatment Plan EIR Conceptual Design Task 2.3: Wachusett Reservoir Draft Modeling Report.

Carter, T., Parry, M., Harasawa, H., Nishioka, S., 1994. IPCC technical guidelines for assessing climate change impacts and adaptations.

Caverly, E., Kaste, J.M., Hancock, G.S., Chambers, R.M., 2013. Dissolved and particulate organic carbon fluxes from an agricultural watershed during consecutive tropical storms, Geophys. Res. Lett. 40, 5147-5152.

Chang, H., Evans, B.M., Easterling, D.R., 2001. The effects of climate change on stream flow and nutrient loading, JAWRA Journal of the American Water Resources Association 37, 973-985.

Chaudhury, R.R., Sobrinho, J.A.H., Wright, R.M., Makam Sreenivas, 1998. Dissolved oxygen modeling of the Blackstone River (northeastern United States), Water Res. 32, 2400-2412.

Chung, S., Gu, R.R., 2009. Prediction of the Fate and Transport Processes of Atrazine in a Reservoir, Environ. Manage. 44, 46-61.

Chung, S., Gu, R., 1998. Two-dimensional simulations of contaminant currents in stratified reservoir, Journal of Hydraulic Engineering 124, 704-711. doi: 10.1061/(ASCE)0733-9429(1998)124:7(704).

Clark, L.M., 2013. Using CE-QUAL-W2 to Model a Contaminant Spill Into the Wachusett Reservoir, M.S. Thesis, Department of Civil and Environmental Engineering, University of Massachusetts Amherst.

Clark, R.M., Grayman, W.M., Males, R.M., Kilgore, R., 1989. Identifying vulnerable surface water utilities, J. Am. Water Works Assoc. 81, 60-67.

Coats, R., Perez-Losada, J., Schladow, G., Richards, R., Goldman, C., 2006. The warming of lake Tahoe, Clim. Change 76, 121-148.

Cole, T.M., Wells, S.A., 2015. CE-QUAL-W2: A Two-Dimensional, Laterally Averaged, Hydrodynamic and Water Quality Model, Version 4.0 User Manual, U.S. Army Corps of Engineers. Washington, DC.

Cole, T.M., Wells, S.A., 2008. CE-QUAL-W2: A Two-Dimensional, Laterally Averaged, Hydrodynamic and Water Quality Model, Version 3.6, Department of Civil and Environmental Engineering, Portland State University, Portland, OR.

Correll, D.L., Jordan, T.E., Weller, D.E., 1999. Transport of nitrogen and phosphorus from Rhode River watersheds during storm events, Water Resour. Res. 35, 2513-2521.

Debele, B., Srinivasan, R., Parlange, J.Y., 2008. Coupling Upland Watershed and Downstream Waterbody Hydrodynamic and Water Quality Models (SWAT and CE-QUAL-W2) for Better Water Resources Management in Complex River Basins, Environmental Modeling and Assessment 13, 135-153.

Devonis, C.S., 2011. Wachusett Reservoir Contaminant Spill Modeling Using CE-Qual W2, M.S. Thesis, Department of Civil and Environmental Engineering, University of Massachusetts Amherst.

Dhillon, G.S., Inamdar, S., 2014. Storm event patterns of particulate organic carbon (POC) for large storms and differences with dissolved organic carbon (DOC), Biogeochemistry 118, 61-81.

Dhillon, G.S., Inamdar, S., 2013. Extreme storms and changes in particulate and dissolved organic carbon in runoff: Entering uncharted waters? Geophys. Res. Lett. 40, 1322-1327.

DiGiano, F., Grayman, W.M., 2014. Can we better protect vulnerable water supplies? Journal AWWA 106, 28.

Doran, P.T., McKay, C.P., Adams, W.P., English, M.C., Wharton, R., Meyer, M.A., 1996. Climate forcing and thermal feedback of residual lake-ice covers in the high Arctic, Limnol. Oceanogr. 41, 839-848.

Edinger, J.E., Brady, D.K., Geyer, J.C., 1974. Heat exchange and transport in the environment, 14.

Edinger, J.E., Duttweiler, D.W., Geyer, J.C., 1968. The response of water temperatures to meteorological conditions, Water Resour. Res. 4, 1137-1143.

Fang, X., Stefan, H.G., 2009. Simulations of climate effects on water temperature, dissolved oxygen, and ice and snow covers in lakes of the contiguous United States under past and future climate scenarios, Limnol. Oceanogr. 54, 2359-2370.

Fang, X., Stefan, H.G., 1999. Projections of climate change effects on water temperature characteristics of small lakes in the contiguous US, Climatic Change 42, 377-412.

Fang, X., Stefan, H.G., 1998. Potential climate warming effects on ice covers of small lakes in the contiguous US, Cold Reg. Sci. Technol. 27, 119-140.

Fang, X., Shrestha, R., Groeger, A.W., Lin, C., Jao, M., 2007. Simulation of Impacts of Streamflow and Climate Conditions on Amistad Reservoir, Journal of Contemporary Water Research & Education 137, 14-20. doi: 10.1111/j.1936-704X.2007.mp137001003.x.

Forman, R.T., Alexander, L.E., 1998. Roads and their major ecological effects, Annu. Rev. Ecol. Syst., 207-C2.

Garvey, E., 2000. Natural Organic Matter Fate and Transport in Quabbin Reservoir, Doctoral Dissertation, Department of Civil and Environmental Engineering, University of Massachusetts Amherst.

Garvey, E., Tobiason, J.E., Hayes, M., Wolfram, E., Reckhow, D.A., Male, J.W., 1998. Coliform transport in a pristine reservoir: Modeling and field studies, Water Science and Technology 37, 137-144. doi: 10.1016/S0273-1223(98)00048-1.

George, G., Hurley, M., Hewitt, D., 2007. The impact of climate change on the physical characteristics of the larger lakes in the English Lake District, Freshwat. Biol. 52, 1647-1666.

Grayman, W., Males, R., 2002. Risk-based modeling of early warning systems for pollution accidents, Water Science & Technology 46, 41-49.

Groisman, P.Y., Knight, R.W., Easterling, D.R., Karl, T.R., Hegerl, G.C., Razuvaev, V.N., 2005. Trends in intense precipitation in the climate record, J. Clim. 18, 1326-1350.

Groisman, P.Y., Knight, R.W., Karl, T.R., Easterling, D.R., Sun, B., Lawrimore, J.H., 2004. Contemporary changes of the hydrological cycle over the contiguous United States: Trends derived from in situ observations, J. Hydrometeorol. 5, 64-85.

Gu, R.R., Chung, S., 2003. A two-dimensional model for simulating the transport and fate of toxic chemicals in a stratified reservoir, J. Environ. Qual. 32, 620-632.

Gu, R., McCutcheon, S.C., Wang, P., 1996. Modeling reservoir density underflow and interflow from a chemical spill, Water Resour. Res. 32, 695-705.

Gullick, R.W., Grayman, W.M., Deininger, R.A., Males, R.M., 2003. Design of early warning monitoring systems for source waters, JOURNAL - American Water Works Association 95, 58-72.

Hagemann, M., Park, M., 2014. Trends in tributary water quality in a water supply reservoir, JOURNAL AWWA 106, 4.

Hagemann, M., Jeznach, L.C.; Park, M., Tobiason, J.E., 2016. Proactive modeling of water quality impacts of extreme hydrologic events. Part 1: Watershed loads. Submitted.

Henderson-Sellers, B., 1988. Sensitivity of thermal stratification models to changing boundary conditions, Appl. Math. Model. 12, 31-43.

Henderson-Sellers, B., 1991. Water Quality Modeling: Decision Support Techniques for Lakes and Reservoirs. CRC Press.

Hodgkins, L., 1999. Measurement and characterization of biodegradable dissolved organic carbon in the MWRA drinking water supply and distribution system, .

Hondzo, M., Stefan, H.G., 1996. Long-term lake water quality predictors, Water Res. 30, 2835-2852.

Hondzo, M., Stefan, H.G., 1993. Regional water temperature characteristics of lakes subjected to climate change, Clim. Change 24, 187-211.

Horton, R., Yohe, G., Easterling, W., Kates, R., Ruth, M., Sussman, E., Whelchel, A., Wolfe, D., Lipschultz, F., 2014. Ch. 16: Northeast. Climate Change Impacts in the United States, in: Anonymous The Third National Climate Assessment, J. M. Melillo, Terese (T.C.) Richmond, and G. W. Yohe, Eds. U.S.Global Change Research Program, pp. 371-395.

Inamdar, S., O'Leary, N., Mitchell, M., Riley, J., 2006. The impact of storm events on solute exports from a glaciated forested watershed in western New York, USA, Hydrol. Process. 20, 3423-3439.

IPCC, 2013. Climate Change 2012: The Physical Science Basis. Contribution of Working Group I to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change.

Jankowski, T., Livingstone, D.M., Bührer, H., Forster, R., Niederhauser, P., 2006. Consequences of the 2003 European heat wave for lake temperature profiles, thermal stability, and hypolimnetic oxygen depletion: Implications for a warmer world, Limnol. Oceanogr. 51, 815-819.

Jeznach, L.C., Tobiason, J.E., Ahlfeld, D.P., 2014. Modeling Conservative Contaminant Impacts on Reservoir Water Quality, JOURNAL - American Water Works Association 106, E295-E306.

Jeznach, L.C., Tobiason, J.E., 2015. Future Climate Effects on Thermal Stratification in the Wachusett Reservoir, JOURNAL - American Water Works Association, 107, No. 4, pp. E197-E209.

Jeznach, L.C., Jones, C., Matthews, T., Tobiason, J.E., and Ahlfeld, D.A. (2016) "A framework for modeling reservoir contaminant impacts on water quality", Journal of Hydrology, Vol 537, pp. 322-333, doi: 10.1016/j.jhydrol.2016.03.041

Jeznach L.C., Hagemann, M., Park, M., Tobiason, J.E. "Proactive modeling of water quality impacts of extreme hydrologic events. Part 2: Drinking water reservoir". In preparation.

Joaquin, A.L., 2001. Modeling the Effect of Quabbin Transfers on Wachusett Reservoir Composition, M.S. Thesis, Department of Civil and Environmental Engineering, University of Massachusetts Amherst.

Jordan, M., Likens, G., 1975. An Organic Carbon Budget for an Oligotrophic Lake In New Hampshire, U. S. A, Verhandlungen Internationale Vereinigung Limnologie 19, 994-1003.

Jung, B., Lee, J., Kim, H., Park, J., 2014. Export, biodegradation, and disinfection byproduct formation of dissolved and particulate organic carbon in a forested headwater stream during extreme rainfall events, Biogeosciences 11, 6119-6129.

Kirshen, P., Fennessey, N., 1995. Possible climate-change impacts on water supply of metropolitan Boston, J. Water Resour. Plann. Manage. 121, 61-70.

Komatsu, E., Fukushima, T., Harasawa, H., 2007. A modeling approach to forecast the effect of long-term climate change on lake water quality, Ecol. Model. 209, 351-366.

Lee, H.W., Kim, E.J., Park, S.S., Choi, J.H., 2012. Effects of climate change on the thermal structure of lakes in the Asian Monsoon Area, Clim. Change 112, 859-880.

Livingstone, D.M., 2003. Impact of secular climate change on the thermal structure of a large temperate central European lake, Clim. Change 57, 205-225.

Livingstone, D.M., 1993. Temporal structure in the deep-water temperature of four swiss lakes: A short-term climatic change indicator? Preceedings-International Association of Theoretical and Applied Limnology 25, 75-75.

Magnuson, J.J., Robertson, D.M., Benson, B.J., Wynne, R.H., Livingstone, D.M., Arai, T., Assel, R.A., Barry, R.G., Card, V.V., Kuusisto, E., Granin, N.G., Prowse, T.D., Stewart, K.M., Vuglinski, V.S., 2000. Historical trends in lake and river ice cover in the northern hemisphere, Science 289, 1743-1746. doi: 8791 [pii].

Marti, C.L., Mills, R., Imberger, J., 2011. Pathways of multiple inflows into a stratified reservoir: Thomson Reservoir, Australia, Adv. Water Resour. 34, 551-561. doi: 10.1016/j.advwatres.2011.01.003.

Martin, P.H., LeBoeuf, E.J., Daniel, E.B., Dobbins, J.P., Abkowitz, M.D., 2004. Development of a GIS-based Spill Management Information System, J. Hazard. Mater. 112, 239-252. doi: 10.1016/j.jhazmat.2004.05.014.

Massachusetts Department of Conservation and Recreation, Division of Water Supply Protection Office of Watershed Management, 2013. 2013 Watershed Protection Plan Update.

Massachusetts Department of Conservation and Recreation, Division of Water Supply Protection Office of Watershed Management, 2011. Water Quality Report: 2010. Wachusett Reservoir and Watershed, 1-43.

Matthews, T.P., 2007. Modeling Fate and Transport of Fecal Coliform in Wachusett Reservoir, M.S. Thesis, Department of Civil and Environmental Engineering, University of Massachusetts Amherst.

Melillo, J.M., Richmond, T.T., Yohe, G., 2014. Climate change impacts in the United States, Third National Climate Assessment.

Metcalf, I., Eddy, H., 2003. Wastewater Engineering; Treatment and Reuse. McGraw-Hill, New York.

Meyer, J.L., Likens, G.E., 1979. Transport and transformation of phosphorus in a forest stream ecosystem, Ecology, 1255-1269.

Mohseni, O., Stefan, H.G., Erickson, T.R., 1998. A nonlinear regression model for weekly stream temperatures, Water Resour. Res. 34, 2685-2692.

Murdoch, P.S., Baron, J.S., Miller, T.L., 2000. Potential effects of climate change on surface-water quality in North America, JAWRA Journal of the American Water Resources Association 36, 347-366.

Narasimhan, B., Srinivasan, R., Bednarz, S., Ernst, M., Allen, P., 2010. A comprehensive modeling approach for reservoir water quality assessment and management due to point and nonpoint source pollution, Transactions of the ASABE 53, 1605-1617.

NOAA, 2013. State of the Climate Global Hazards, 2014.

Paerl, H.W., Paul, V.J., 2012. Climate change: Links to global expansion of harmful cyanobacteria, Water Res. 46, 1349-1363. doi: 10.1016/j.watres.2011.08.002.

Peeters, F., Livingstone, D.M., Goudsmit, G., Kipfer, R., Forster, R., 2002. Modeling 50 years of historical temperature profiles in a large central European lake, Limnol. Oceanogr. 47, 186-197.

Pica, J., 2012. Wicked problems: a look at climate change and water resources in New England, M.S. Thesis, Department of Civil and Environmental Engineering, University of Massachusetts Amherst.

Porter, K.G., Saunders, P.A., Haberyan, K.A., Macubbin, A.E., Jacobsen, T.R., Hodson, R.E., 1996. Annual cycle of autotrophic and heterotrophic production in a small monomictic Piedmont lake (Lake Oglethorpe): Analog for the effects of climatic warming on dimictic lakes, Limnol. Oceanogr. 41, 1041.

Roberts, M.G., 2003. NOM modeling for Quabbin Reservoir, M.S. Thesis, Department of Civil and Environmental Engineering, University of Massachusetts Amherst.

Sahoo, G., Forrest, A., Schladow, S., Reuter, J., Coats, R., Dettinger, M., 2015. Climate change impacts on lake thermal dynamics and ecosystem vulnerabilities, Limnol. Oceanogr. .

Sahoo, G., Schladow, S., 2008. Impacts of climate change on lakes and reservoirs dynamics and restoration policies, Sustainability Science 3, 189-199.

Sahoo, G., Schladow, S., Reuter, J., Coats, R., 2011. Effects of climate change on thermal properties of lakes and reservoirs, and possible implications, Stochastic Environmental Research and Risk Assessment 25, 445-456.

Sahoo, G., Schladow, S., Reuter, J., Coats, R., Dettinger, M., Riverson, J., Wolfe, B., Costa-Cabral, M., 2013. The response of Lake Tahoe to climate change, Clim. Change 116, 71-95.

Samal, N.R., Matonse, A.H., Mukundan, R., Zion, M.S., Pierson, D.C., Gelda, R.K., Schneiderman, E.M., 2013. Modeling potential effects of climate change on winter turbidity loading in the Ashokan Reservoir, NY, Hydrol. Process.

Samal, N.R., Pierson, D., Schneiderman, E., Huang, Y., Read, J., Anandhi, A., Owens, E., 2012. Impact of climate change on Cannonsville reservoir thermal structure in the New York City Water Supply, Water Qual. Res. J. Can. 47, 389-405.

Sansalone, J.J., Buchberger, S.G., 1997. Partitioning and first flush of metals in urban roadway storm water, J. Environ. Eng. 123, 134-143.

Sharpley, A.N., Chapra, S., Wedepohl, R., Sims, J., Daniel, T.C., Reddy, K., 1994. Managing agricultural phosphorus for protection of surface waters: Issues and options, J. Environ. Qual. 23, 437-451.

Sojkowski, B., 2011. 2D Spill Modeling in the Wachusett Reservoir With CE-QUAL-W2 for Years 2003-2006, M.S. Thesis, Department of Civil and Environmental Engineering, University of Massachusetts Amherst.

Stanford, B.D., Wright, B., Routt, J.C., Debroux, J.F., Khan, S.J., 2014. Water quality impacts of extreme weather-related events, 4324.

Stauber, C.L., 2009. Contaminant Spill Modeling for Wachusett Reservoir, M.S. Thesis, Department of Civil and Environmental Engineering, University of Massachusetts Amherst.

Sung, W., 2003. Incorporating Reservoir Transfer Into Treatment Optimization Decisions, Journal of Environmental Engineering 129, 377-381.

Tobiason, J.E., Ahlfeld, D.P., Joaquin, A., Mas, D., 2002. Water Quality in MDC Reservoirs. Project 1: Wachusett Reservoir Water Quality Modeling.

Tobiason, J.E., Male, J.W., Reckhow, D.A., Garvey, E., Hayes, M., and Wolfram, E., 1998. Contaminant Transport Modeling and the Impact of Watershed Management Activities on Water Quality in Quabbin Reservoir, Report to Metropolitan District Commission, Division of Watershed Management.

Verburg, P., Hecky, R.E., 2009. The physics of the warming of Lake Tanganyika by climate change, Limnol. Oceanogr. 54, 2418.

Wauchope, R., 1978. The pesticide content of surface water draining from agricultural fields—a review, J. Environ. Qual. 7, 459-472.

Whelton, A.J., McMillan, L., Connell, M., Kelley, K.M., Gill, J.P., White, K.D., Gupta, R., Dey, R., Novy, C., 2014. Residential Tap Water Contamination Following the Freedom Industries Chemical Spill: Perceptions, Water Quality, and Health Impacts, Environ. Sci. Technol. .

Whitehead, P., Wilby, R., Battarbee, R., Kernan, M., Wade, A.J., 2009. A review of the potential impacts of climate change on surface water quality, Hydrological Sciences Journal 54, 101-123.

Willems, P., Arnbjerg-Nielsen, K., Olsson, J., Nguyen, V.T.V., 2012. Climate change impact assessment on urban rainfall extremes and urban drainage: Methods and shortcomings, Atmos. Res. 103, 106-118.

World Meteorological Organization (WMO), 2014. Statement on the status of the global climate in 2013, WMO-No. 1130.

Yates, D., Miller, K., 2010. Climate Change in Water Utility Planning: Decision Analytic Approaches, 3132.

Yoon, B., Raymond, P., 2012. Dissolved Organic Matter Export From a Forested Watershed During Hurricane Irene, Geophysical Research Letters 39, L18402.