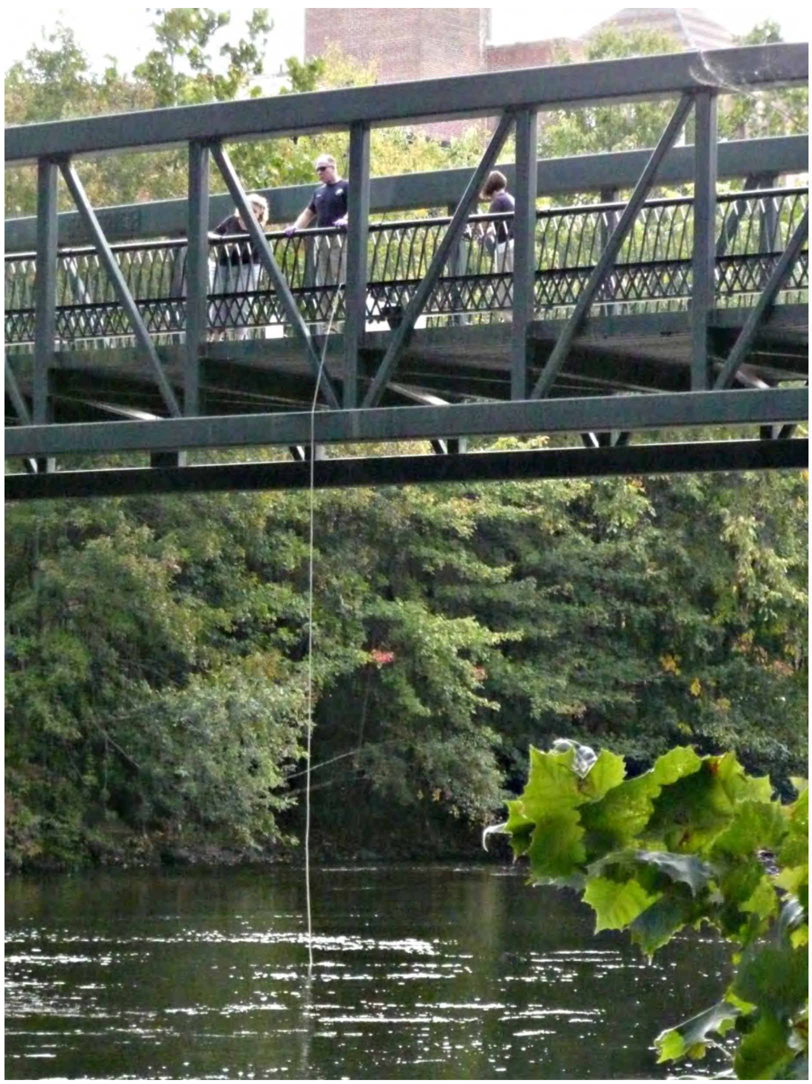
MA Water Resources Research Center

Annual Report 2010-2011

March 1, 2010 – June 30, 2011

Elizabeth Scotten Finn and Marie-Françoise Hatte



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Front cover: Sampling the Blackstone River. Photo by MF Hatte

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Introduction

This report covers the period March 1, 2010 to June 30, 2011[1], the 45^{th} year of the Massachusetts Water Resources Research Center (WRRC). The Center is under the direction of Dr. Paula Rees, who holds a joint appointment as Director of the WRRC and as Director of Education and Outreach of the Engineering Research Center for Collaborative Adaptive Sensing of the Atmosphere at the University of Massachusetts Amherst.

Several research projects were supported by the Massachusetts Water Resources Research Center. The Water Center funded a research project headed by Dr. Ellen Douglas of UMass Boston entitled "Developing a Physically-Based and Policy-Relevant River Classification Scheme for Sustainable Water and Ecosystem Management Decisions."

Four graduate student projects were also funded:

- *An Assessment Methodology for Differential Impact on Environmental Justice Populations of Releases of Industrial Toxics to Water in Massachusetts* under PI Dr. Michael Ash of UMass Amherst;
- *Impact of the Hemlock Woolly Adelgid on the Water Cycle in New England: Differences in Hydrologic Fluxes Between Hemlock and Deciduous Forest Stands* under PI Dr. Andrew Guswa of Smith College;
- *Monitoring and Modeling Chromophoric Dissolved Organic Matter in Neponset River and Boston Harbor Using GIS and Hyperspectral Remote Sensing* under PI Qian Yu of UMass Amherst; and
- *Surface Water-Groundwater Interactions on the Deerfield River* under PI Dr. David Boutt of UMass Amherst.

The *Acid Rain Monitoring Project*, led by WRRC Associate Director Marie-Françoise Hatte, was continued for another year in order to document trends in surface water acidification.

One Technology Transfer award supported the Eighth Annual Water Resources Conference, held April 7, 2011 and organized by the Water Center on the University of Massachusetts Amherst campus.

Other projects conducted at WRRC include the *Tri-State Connecticut River Targeted Watershed Initiative*, and the continued collaboration with UMass Extension on the *Stream Continuity Project*. The Center is also working on a stormwater clearinghouse project that enables users to search the web for stormwater Best Management Practices and to find innovative technologies available to treat stormwater. The *Blackstone River Water Quality Modeling* project continued.

Two other projects were the second year of a USGS 104G grant to Dr. David Boutt of UMass Amherst Geosciences: *Characterizing and Quantifying Recharge at the Bedrock Interface*, and an Army Corps of Engineers funded study of climate change impacts on the Great Lakes, to Casey Brown of UMass Civil & Environmental Engineering: *Evaluation of Adaptive Management of Late Superior amid Climate Variability and Change*.

[1] The USGS reporting year covers March 1 to February 28, while the University of Massachusetts and the Commonwealth of Massachusetts fiscal years run from July 1 to June 30. Projects funded by the State are reported for the period July 1 2010 - June 30, 2011.

WRRC was involved in three projects incorporating modern information technology into environmental research, teaching at the middle school and University levels, and public outreach in the Connecticut River watershed. All three use location-aware hardware and software technologies and handheld computers to enhance understanding of complex, place-based environmental issues.

Progress results for each project are summarized for the reporting year in the following sections.

Research Program

Eight research projects were conducted this fiscal year. One research project was funded through the USGS 104G program, and one research project received a no-cost extension for funds received through the previous USGS 104B program. Five new projects were funded through the 104B program and were completed this year. One research project was funded by the US Army Corps of Engineers for *Evaluation of Plausible Risk to Lake Superior Regulation and Upper Great Lakes Amid Climate Variability and Change* led by PI Dr. Casey Brown of UMass Amherst and administered by the Water Center.

1. Environmental Behaviors of Engineered Nanoparticles in Water (USGS 2007MA73B)

Principal Investigator: Dr. Baoshan Xing, UMass Amherst Plant, Soil, and Insect Sciences
Start Date: 3/1/2007
End Date: 5/31/2010
Reporting Period: March 1, 2010 – June 30, 2011
Funding Source: USGS (104B)
Research Category: Environmental and Water Chemistry
Focus Categories: Water Quality; Toxic Substances; Solute Transport
Descriptors: Nanoparticle; Sorption; Suspension; Phytotoxicity; Oxides; Natural Organic Matter; Environment

Problem and Research Objectives
Knowledge of engineered nanoparticles in water is critical for evaluating their environmental fate, exposure, risk and ecotoxicity. The research objectives for this project were:

1) To characterize colloidal and aggregation behavior of nanoparticles as affected by pH and organic matter
2) To examine the adsorption and desorption of organic chemicals and natural organic matter by nanoparticles.

Methodology
Batch sorption techniques, DSL, liquid scintillation counting, HPLC detection, TEM, AFM and SEM examinations.

Principal Findings and Significance
The colloidal stability of three structurally different humic acid (HA) coated Al_2O_3 nanoparticles (HAs-Al_2O_3 NPs) was studied in the presence of Ca^{2+}. HAs were obtained after sequential extractions of Amherst Peat Soil. Highly polar HA1-coated Al_2O_3 NPs exhibited strong aggregation in the presence of Ca^{2+}. HA3 and HA7-coated

NPs showed weaker aggregation due to their increased aliphaticity and low polarity. HA7-Al_2O_3 NPs displayed the weakest aggregation behavior even at relatively high Ca^{2+} concentration. The inverse stability ratio (α=1/W) was the lowest for HA7-Al_2O_3 NPs reflecting that strong steric stabilization enhanced colloidal stability. Atomic force microscopy (AFM) of pure Al_2O_3 NPs on Ca^{2+}-saturated mica clearly demonstrated significant aggregation following classical DLVO model for hard spheres. On the contrary, the weakly polar HA fraction produced an approximately 10 nm thick corona of adsorbed layer around each Al_2O_3 NP, thus stabilizing the coated nanoparticle suspension through a steric effect. Under alkaline conditions and at low ionic strength, adsorbed HA chains swelled, increasing their osmotic potential, which in turn resulted in stabilization of the colloids. Inherent structural variations of NOMs played a significant part in colloidal stability of the coated nanoparticles. Thus, development of sterically stabilized nanoparticles may have potential application for water remediation in marine and high salinity conditions.

Publications and Conference Presentations

Articles in Refereed Scientific Journals

Ghosh, S., Hamid Mashayekhi, P. Bhowmik and B. Xing, 2010. Colloidal stability of Al_2O_3 nanoparticles as affected by coating of structurally different humic acids. Langmuir, 26 (2): 873–879.

Dissertations

Ghosh, Saikat 2010. "Influence of Natural Organic Matter and Synthetic Polyelectrolytes on Colloidal Behavior of Metal Oxide Nanoparticles" Ph.D. Dissertation. University of Massachusetts, May 2010.

Conference Proceedings

Jiang, Wei, and B. Xing, 2010. "Behavior of Nanoparticles at the Bacteria-Water Interface." Poster presentation for the 7th Annual Water Resources Research Center, University of Massachusetts, Amherst, MA. April 8, 2010.

Student Support, Department of Plant, Soil and Insect Sciences

- Mr. Hamid Mashayekhi, PhD Candidate need departments for those
- Mr. Saikat Ghosh, PhD Candidate
- Ms. Wei Jiang, PhD Candidate

2. Bacterial Toxicity of Oxide Nanoparticles and Their Adhesion (USGS 2009MA177B)

Principal Investigator: Dr. Baoshan Xing, UMass Amherst
Start Date: 4/1/2009
End Date: 3/31/2010
Funding Source: USGS (104B)
Reporting Period: March 1, 2010 – Feb 28, 2011
Research Category: Water Quality
Focus Categories: Toxic Substances, Water Quality, Geochemical Processes

Problem and Research Objectives

Oxide nanoparticles (NPs) are widely used, and they are potentially toxic. The goal of this work was to evaluate the toxicity of several engineered oxide NPs to common bacteria species and the adhesion of NPs to the bacteria surface.

Methodology
Batch experiments, FTIR, Characterization of Nanoparticles, Toxicity evaluation, AFM and TEM imaging.

Principal Findings and Significance
Toxicity of nano-scaled aluminum, silicon, titanium and zinc oxides to bacteria (*Bacillus subtilis, Escherichia coli* and *Pseudomonas fluorescens*) was examined and compared to that of their respective bulk (micro-scaled) counterparts. All nanoparticles but titanium oxide showed higher toxicity than their bulk counterparts. Toxicity of released metal ions was differentiated from that of the oxide particles. ZnO was the most toxic among the three nanoparticles, causing 100% mortality to the three tested bacteria. Al_2O_3 nanoparticles had a mortality rate of 57% to *B. subtilis*, 36% to *E. coli*, and 70% to *P. fuorescens*. SiO_2 nanoparticles killed 40% of *B. subtilis*, 58% of *E. coli*, and 70% of *P. fluorescens*. TEM images showed attachment of nanoparticles to the bacteria, suggesting that the toxicity was affected by bacterial attachment. Bacterial responses to nanoparticles were different from their bulk counterparts; therefore nanoparticle toxicity mechanisms need to be studied thoroughly.

Publications and Conference Presentations
Conference Proceedings

Jiang, W. and B. Xing, 2010. "Behavior of nanoparticles at the bacteria-water interface" The 7th Annual Massachusetts Water Resources Conference, Amherst, April 8, 2010. Abstract book, p. 25.

Student Support

- Mr. Hamid Mashayekhi, Ph.D. UMass Amherst Plant, Soil & Insect Sciences
- Miss Wei Jiang, Ph.D., UMass Amherst Plant, Soil & Insect Sciences

3. Impact of Nanoparticles on the Activated Sludge Process Effects on Microbial Community Structure and Function (USGS 2009MA178B)

Primary Principal Investigator: Dr. Juliette N. Rooney-Varga, UMass Lowell
Other PIs: Deepankar Goyal
Start Date: March 1, 2009
End Date: Feb 28, 2010
Reporting Period: March 1, 2010 – June 30, 2011
Funding Source: USGS (104B)
Research Category: Basic Research
Focus Categories: Toxic Substances; Wastewater
Descriptors: Activated Sludge; Carbon Nanotubes; Nanomaterials; Emerging Contaminants; Molecular Microbial Ecology; Microbial Community Analysis, ARISA

Problem and Research Objectives
Nanotechnology, or the ability to create and use materials at the scale of 1 to 100 nanometers, is a rapidly expanding sector that is generating materials with unique physical and chemical properties. In particular, carbon nanotubes (CNTs) are known for their unique mechanical, electronic, and biological properties and have far-reaching potential applications as components of personal care products, pharmaceuticals, electronic devices, energy storage devices, stains and coatings, and new environmental clean-up technologies (Masciangioli and Zhang, 2003; Boczkowski and Lanone, 2007; Chen, 2007; Erdem, 2007; Rivas et al., 2007; Kislyuk

and Dimitriev, 2008; Prato et al., 2008; Theron et al., 2008). Massachusetts is poised to be a leader in nanotechnology research and development and this sector is expected to be a major component of the Commonwealth's economy for the foreseeable future. However, while the potential for nanotechnology is vast, relatively little is known about the health and environmental risks posed by nanomaterials (Colvin, 2003). Indeed, those watching the industry have commented that concern over unknown risks of nanomaterials is a major determinant of the future success of nanotechnology (Colvin, 2003).

Through nanomanufacturing and widespread use of nanomaterials, CNTs and other nanomaterials will inevitably be released into wastewater streams and enter wastewater treatment plants (Wiesner, 2006). All publicly owned wastewater treatment facilities rely on the 'activated sludge process,' which relies on controlled microbial degradation of waste materials to remove chemical and biological contaminants (Wagner et al., 2002). However, little is known about how CNTs will affect the complex microbial communities that are responsible for the activated sludge process and whether microorganisms are capable of removing them. Effluent from wastewater treatment plants ultimately is released to the environment, where it can impact aquatic ecosystems and drinking water. Any toxicity to microorganisms exhibited by CNTs has the potential to dramatically reduce the efficacy of the activated sludge process, resulting in the release of untreated sewage, pathogenic microbes, and CNTs into the environment. Shock-loading of other contaminants has shown that treatment performance can be affected for weeks or months, resulting in a reduction in treatment efficiency, environmental release of toxic contaminants, and operation problems that may require months to recover (Boon et al., 2003; Henriques et al., 2007). In addition, the ability of CNTs to strongly adsorb organic matter can reduce the bioavailability and, therefore, microbial degradation of organic pollutants, which would then effectively bypass the treatment process.

The composition and function of activated sludge microbial communities have received considerable attention, although the function of many specific phylogenetic groups and the factors that control them are not yet well understood. In broad terms, activated sludge contains microbial eukaryotes ("microeukaryotes"), including protozoa, fungi, and metazoans, as well as a wide diversity of bacteria responsible for varied metabolic functions, including oxidation of organic compounds and removal of nitrogenous pollutants and phosphates (Weber et al., 2007). Within this microbial community, many complex ecological interactions are thought to be necessary for the effective functioning of activated sludge. For example, ciliated protozoa and fungi have been found to form tree-like and filamentous colonies, respectively, that form a back-bone for bacterial colonization, resulting in the production of flocs, which readily settle out of the liquid phase and are collected for effective removal from treated wastewater (Weber et al., 2007). Both bacteria and microeukaryotes are likely to contribute to the formation of extracellular polymeric substances (EPS), which are high molecular weight compounds with adhesive properties that are critical to the formation and integrity of flocs (and biofilms more generally) (Raska et al., 2006; Weber et al., 2007). In addition, specific taxonomic groups of bacteria are known to carry out key functions in activated sludge. For example, members of the *Planctomycetes* are responsible for anaerobic ammonium oxidation; several lineages within the kingdom *Euryarchaea* produce methane; members of the genus *Nitrospira* oxidize nitrite to nitrate (Juretschko et al., 2002); *Actinomycetes* may contribute to the production of foam and reduce the quality of effluent; and members of the *Chloroflexi* have been associated with bulking events (Kragelund et al., 2007). Thus, analysis of microbial community composition can provide meaningful insight into various activated sludge functions, as well as the factors that control them (Liu et al., 1997; Forney et al., 2001).

Relatively little is known about the potential toxicity of CNTs to activated sludge microorganisms and studies on pure cultures or defined mixed cultures have yielded conflicting results. There is strong evidence for CNT toxicity to pulmonary cells, as well as potential toxicity to epithelial, brain, and liver cells (Lam et al., 2006; Smart et al., 2006; Warheit 2006). Single-walled CNTs (SWCNTs) have been reported to be highly toxic to *Escherichia coli* str. K12 cells that come in direct contact with them (Kang et al., 2007). Ghafari et al. (2008) found a moderate impact of SWCNTs on *E. coli*-gfp viability, although they did not differentiate between planktonic cells and those in contact with SWCNT aggregates. Interestingly, they found that *Tetrahymena thermophila,* a ciliated protozoan that is an important member of wastewater treatment microbial communities, ingested CNTs. As a result, the protozoan's ability to ingest and digest bacterial cells was impeded, suggesting a negative impact on an important function of these protozoa, namely bacterivory (Ghafari et al., 2008). Conversely, the CNTs may also have a positive impact on activated sludge processes, as they caused an increase in the production of exudates by ciliates, which may benefit floc formation and, therefore, sludge settleability.

While pure culture studies have shown that nanomaterials can act as antimicrobial agents, the complexity of the activated sludge community make it unlikely to respond to CNTs in the same manner as simple pure culture systems. Our objective was to use state-of-the-art molecular techniques to determine the impact of CNTs on microbial community dynamics in batch reactors that model the activated sludge process.

Methodology

Experimental set-up

Fresh activated sludge was collected from an aeration basin at the Lowell Regional Wastewater Treatment Facility, Lowell. This facility is designed to treat primarily municipal wastewater through conventional primary and secondary treatment processes. Whole unscreened samples were transported to the laboratory and processed within 30 minutes of sample collection. Experimental conditions for batch-scale reactor studies were previously described by Yin et al. (2009). In order to distinguish between effects of CNTs and potential toxic effects of impurities associated with them (such as amorphous carbon and metal catalysts), triplicate CNT-exposed reactors were compared to triplicate reactors exposed to impurities alone. CNTs used in the current study consisted of >90% pure CNTs (Sigma-Aldrich, Inc., St. Louis MO) characterized by Raman spectroscopy (Table 1). Reactors were filled with 2 L of fresh activated sludge, with an initial soluble chemical oxygen demand (sCOD) of 20 mg L^{-1} from the aeration basin effluent (Yin et al., 2009). The sludge was fed with peptone and aerated prior to and during the experiment as described by Yin et al. (2009). Sub-samples for microbial community analysis were taken aseptically immediately after adding CNTs or impurities (T_0), at 1.25 hr (T_1) after initial exposure, and at 5 hr (T_4). The samples were placed in cryovials, and stored at -80°C until further processing.

<u>DNA extraction and analysis</u>

Genomic DNA was extracted and purified from 400 µL sub-samples of sludge using the FastDNA Spin kit for Soil (MP Biomedicals Inc., Solon, OH). ARISA-PCR was performed as previously described (Fisher and Triplett 1999), with minor modifications. Reaction mixtures contained 1× AmpliTaq PCR buffer (Applied Biosystems, Inc., Carlsbad, CA), 2.5 mM $MgCl_2$, 400 ng μL^{-1} bovine serum albumin (BSA), 200 µM each dNTP, 400 nM each primer, 2.5 U of *Taq* DNA polymerase, and 1, 5, 10, or 20 ng of genomic DNA in a final volume of 50 µL. The primers used were

1392F (5'-G [C/A] ACACACCGCCCGT-3') and 23SR (5'GGGTT[C/G/T] CCCCATTC[A/G]G-3'). The 5'end of primer 1392F was labeled with 6-carboxyfluorescein (6-FAM). The following thermal profile was used for PCR: denaturation at 94°C for 3 min, followed by 30 cycles of amplification at 94° C for 30 s, 56° C for 30 s, and 72° C for 45 s, followed by a final extension of 72° C for 7 min. PCR products were analyzed by electrophoresis in 1% agarose gels (Ausubel et al., 1997) and were purified using QiaQuick PCR Purification Kits (Qiagen, Inc., Valencia CA).

20 ng each purified PCR product were lyophilized and subjected to automated capillary electrophoresis (CE) analysis in conjunction with a 50 – 1200 bp size standard labeled with LIZ™ (Applied Biosystems, Inc.) at the Center for AIDS Research, UMass Medical School, Worcester MA. ARISA conditions were optimized by comparing profiles generated from multiple DNA template amounts (1, 5, 10, or 20 ng per 50 µL PCR) and PCR product amounts (5, 10, or 20 ng PCR product per well). Comparison of these conditions indicated that the highest diversity (species richness and evenness) and signal to noise ratios were achieved using 1 ng DNA template DNA for PCR and 20 ng PCR product for CE analyses, which were used in subsequent analyses.

ARISA profiles were analyzed using PeakScanner software (Applied Biosystems Inc.) and processed as described by Brown et al., (2005). The program's Interactive and Automatic Binner were used to bin peaks, with a window size (WS) of 3 bp and a shift value (Sh) of 0.1 (Ramette, 2009). Peak areas were normalized to total peak area per sample and peaks representing <1% total peak area for a given sample were considered indistinguishable from background and removed from the analysis. Data visualization and ordination analyses were conducted using the packages Ecodist (Goslee and Urban, 2007) and Vegan (http://vegan.r-forge.r-project.org/) in the R statistical programming environment (Goslee and Urban, 2007). Pairwise Bray-Curtis distances between samples were calculated using the Ecodist package and a hierarchical clustering algorithm with average linkage clustering were used to construct a dendrogram depicting relationships among the samples' ARISA profiles. Correspondence analysis (CA), which assumes a unimodal relationship between relative abundance (i.e., normalized peak area) and ordination axes, was used to analyze relationships between samples. The R package Vegan was used to determine whether CA ordination axes were correlated with environmental variables. The latter included the experiment from which samples were analyzed (E1 for the experiment comparing CNTs to CNT-associated impurities, conducted on June 28, 2007; E2 for the experiment comparing CNT-associated impurities to a control conducted on July 19, 2007); time elapsed from the initiation of the experiment to sampling (0, 1.25, or 5 hours); and treatment (CNTs, associated impurities, or feed alone). "Dummy" variables were assigned for categorical variables and set to 0 or 1 depending on the presence of a given variable. The "envfit" goodness of fit test with 1000 permutations was used to assess the fit of environmental variables to ordination axes.

In order to determine the phylogenetic identity of dominant community members, as detected by ARISA, phylogenetic analysis of 16S rRNA genes contiguous with fragments analyzed in ARISA was used (Brown et al., 2005). DNA amplicons containing partial 16S rRNA genes and associated intergenic spacer (IGS) regions were generated from selected activated sludge genomic DNA samples using primers 338F and 23SR (5'GGGTT[C/G/T] CCCCATTC[A/G]G-3') (Amann et al., 1990; Brown et al., 2005). The resulting amplicons were cloned using the TOPO TA Cloning Kit for Sequencing with One Shot® TOP10 Chemically Competent *E. coli*, as described by the manufacturer (Invitrogen Corp., Carlsbad, CA). 90 cloned inserts were analyzed using ARISA, as described above, except that the template DNA for PCR consisted of *E. coli* clone cell lysates (obtained by suspending individual colonies in 0.1 M Tris-Cl, pH 8.0, and incubating them at 99° C for 2 minutes). ARISA peaks from cloned inserts were considered to match OTUs from environmental community ARISA patterns if their peak size was placed within the same 3 bp bin as a given OTU from environmental samples.

At least one cloned insert representative of each ARISA OTU was sequenced in both directions by Beckman Coulter Genomics Inc. (Danvers MA, USA) with M13 primers. Vector and primer sequences were trimmed, trimmed sequences were aligned to the Silva database, and phylogenetic relationships among aligned sequences and their 40 nearest neighbors in the Silva database were analyzed using ARB (Ludwig et al., 2004; Pruesse et al., 2007). Trimmed sequences were deposited in GenBank under accession numbers HM205112 - HM205114.

Principal Findings and Significance

Results

Effects of CNTs and their associated impurities

Analysis of ARISA profiles revealed several differences between bacterial community structure in batch reactors exposed to CNTs for five hours when compared to those exposed to associated impurities alone. For example, the relative peak areas of dominant OTUs represented by peaks 419, 794, and 839 bp were significantly different in communities exposed to CNTs vs. those exposed to CNT-associated impurities (Fig. 1). Similarly, a Chi-square goodness-of-fit test of correspondence analysis (CA) axes revealed that the effect of CNTs on community structure was significant (p =0.043), while exposure to impurities alone was not (p= 0.604). In order to assess the effect of CNTs without interference from the strong effects of time and experiment, CA ordination was repeated with only the time T4 samples from the experiment comparing CNTs to impurities alone (E1). A statistically significant effect of CNTs was observed (p<0.001), while a similar analysis of the effects of impurities alone (CA with experiment E2, time T4 samples) revealed no effect (p=0.316), as was also evident from direct inspection of ARISA profiles (Fig. 1). Samples taken after only 1.25 hours exposure (time T1) revealed no clear differences in ARISA profiles between either CNT- and impurities-exposed reactors or between reactors exposed to impurities and control reactors), indicating that exposure for 1.25 hours was insufficient for CNT effects to be detected via the approach used here.

Both hierarchical clustering and correspondence analysis (CA) of all samples revealed strong effects of the amount of time elapsed prior to sampling (0, 1.25, or 5 hours) and the date of the experiment (Fig. 2). Baseline (T_0) communities for E1 and E2 were fairly similar. However, these communities diverged substantially over the short experimental time period of five hours, with the resulting communities sharing only 14/29 total OTUs and 4/9 total "dominant" (considered here to be those with average relative peak areas > 5%) OTUs.

Three of the OTUs found in environmental samples were identified among the 90 cloned inserts analyzed here. These included peaks corresponding to 419, 740, and 812 bp (Fig. 1). Phylogenetic analysis placed these OTUs within the families *Sphingomonadaceae* (419 bp) and *Cytophagaceae* (740 bp), and the genus *Zoogloea* (812 bp). Two representatives of OTU 812 were sequenced and found to be identical. The closest relatives of the sequences representing OTUs 419, 740, and 812 were: an uncultivated *Sphingomonadaceae* bacterium from snow (97.1% similarity); an uncultivated *Cytophagaceae* bacterium from activated sludge (89.5% similarity); and *Zoogloea resiniphila*, a denitrifier isolated from activated sludge (99.8% similarity).

Discussion

While CNTs have the potential to be highly toxic to microbial cells, their impact under the complex abiotic and biological conditions found in environmental microbial communities remains poorly understood. The current study revealed changes in microbial community structure in activated sludge batch reactors exposed to CNTs, while no effects of CNT-associated impurities were detected. Yin et al. (2009) analyzed bulk parameters and performance from the CNT-exposed batch reactors described here and similarly found that CNTs, but not their associated impurities, had several effects on sludge performance. These effects included: increased organic carbon removal primarily through organic carbon adsorption; less negative surface charges of activated sludge flocs; and improved sludge settleability (Yin et al., 2009). Other parameters such as pH, dissolved oxygen, specific resistance to filtration, and relative hydrophobicity were not significantly impacted (Yin et al., 2009). These findings suggest that CNTs impacted community structure through toxicity to some community members, by reducing organic carbon bioavailability, and/or by altering floc properties.

The fact that CNT effects on microbial community structure were detected was especially interesting given that, unlike some previous studies, the experimental conditions used did not maximize CNT-cell interactions. For example, an assay for cytotoxicity developed by Kang et al. (2007) relies on drawing planktonic cells onto a filter that is coated with nanoparticles and observing the resulting effects on cellular membrane integrity over time. Under these conditions, direct cell-nanoparticle contact is artificially induced and CNTs demonstrated high levels of toxicity to Gram-negative (*Escherichia coli* and *Pseudomonas aeruginosa*) and, to a lesser extent, Gram-positive (*Staphylococcus epidermis* and *Bacillus subtilis*) cells (Kang et al., 2009). In contrast, here, CNTs were added to activated sludge bioreactors in suspension, making CNT-cell contact much less likely. In addition, the presence of extracellular polymeric substances (EPS) and high concentrations of DOC in the batch reactors used here may have mitigated CNT toxicity to some extent, as CNTs are likely to become embedded in EPS and thereby prevented from coming in direct contact with cell membranes (Neal, 2008; Luongo and Zhang, 2010). Lastly, the exposure time was kept short in order to avoid confounding effects of starvation and/or accumulation of waste products in closed-system batch reactors. Despite the use of short incubation times, changes in community structure with both CNT exposure and time over the course of the experiment were found (Fig. 1 and 2). Previous studies have shown that cellular inactivation increased with time of exposure (Kang et al., 2009), indicating that use of longer incubation times in continuous reactors may increase effects of CNTs on community structure.

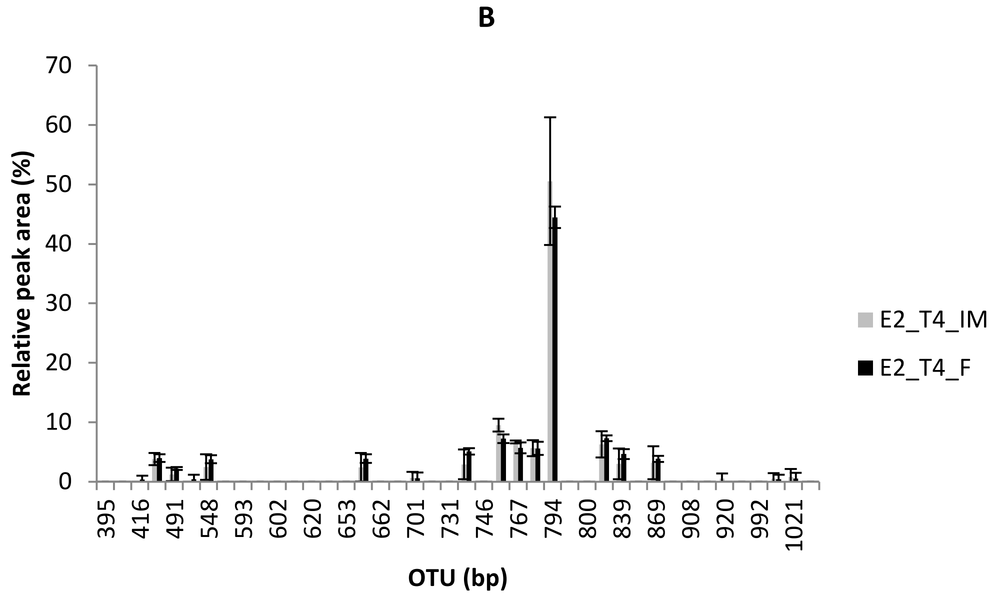
Phylogenetic analysis of cloned inserts that were matched to ARISA peaks revealed the presence of three phylogenetic groups that are responsible for important functions in activated sludge communities, including the members of the families *Sphingomonadaceae* (OTU 419) and *Cytophagaceae* (OTU 740) and the genus *Zoogloea* (OTU 812) (Manz et al., 1996; Neef et al., 1999; Juretschko et al., 2002; Wagner et al., 2002; Li et al., 2008). Of these, the sphingomonad (OTU 419) showed a trend of decreased relative peak intensity with exposure to CNTs (Fig. 1), indicating an adverse impact of CNTs on this group compared to other community members. Within wastewater treatment microbial communities, sphingomonads are thought to have wide metabolic diversity, are capable of degrading some xenobiotics, and contribute to the formation of flocs (Neef et al., 1999; Wagner et al., 2002). Although directly measuring these parameters was beyond the scope of the current study, the potential for negative impacts on CNTs on these microbial functions deserves further attention.

Table 1. Characteristics of CNTs used in the current study.

Purity	
Carbon nanotubes	>90%
Single-walled nanotubes	>50%
Impurities	
Amorphous carbon	<5%
Co	0.6%
Mg	1.2%
Mo	0.1%
Silicates	0.1%
Average outside diameter	1–2 nm
Density	1.7–2.1
Length	5–15 um
Specific surface area	>400 m2/g

Differences in the 'baseline' (T_0) community structure from one sampling date to another corroborate results obtained by Wittebolle et al. (2005), who observed that large community shifts occurred over a period as short as a few days in a given wastewater treatment plant and that community structure was related to performance of biological treatment. These findings underscore the need to analyze microbial community structure when assessing the effects of emerging contaminants on environmental systems, as differences in the starting community composition may alter the observed impacts on community performance.

In conclusion, our results indicate that the structure of activated sludge microbial communities is impacted by exposure to CNTs, even when such exposure is limited to a short time period, and that these effects were not due to impurities associated with CNTs. Community shifts found here indicated that CNTs differentially affect microbial species, as has been found under pure culture conditions (Kang et al., 2009). These results raise the concern of CNT impact on biological functions carried out by the activated sludge process.



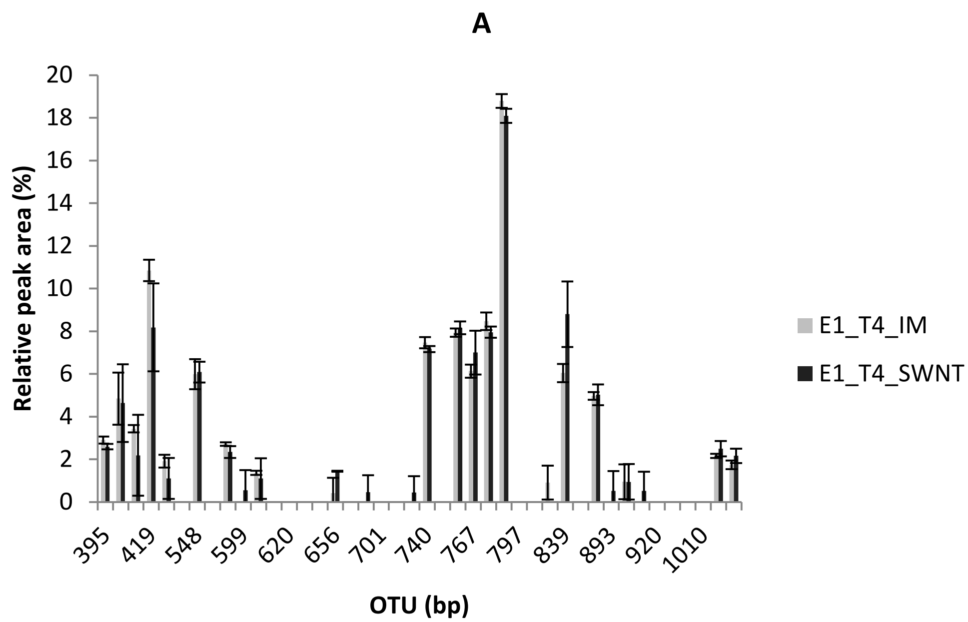


Figure 1. ARISA profiles of activated sludge bacterial communities exposed to CNTs, their associated impurities, or synthetic feed alone at the end of the experiments (T4). Comparisons were made between CNT- and impurities-exposed (IM) reactors during one experiment (designated E1; panel A) and between impurities-exposed and control reactors receiving feed alone (F) in a second experiment (E2; panel B). Means and standard deviations of relative peak areas from triplicate batch reactors are shown.

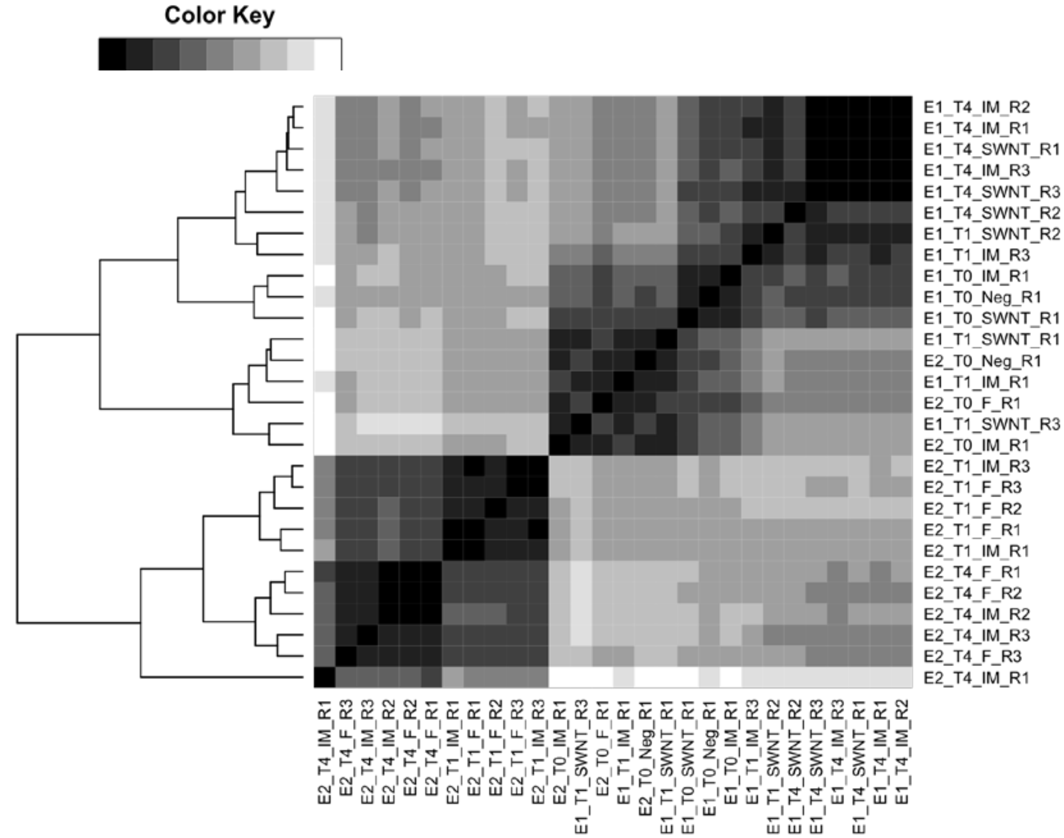


Figure 2. Hierarchical clustering analysis and heatmap of Bray-Curtis distances among samples taken from the first and second experiments (E1 and E2, respectively), at times 0, 1.25 hours, and 5 hours (T0, T1, and T4, respectively), and exposed to CNTs, impurities, or feed alone (SWNT, IM, or F, respectively).

Publications and Conference Presentations

Goyal, D., J. X. Zhang, J. N. Rooney-Varga, 2010. Impacts of single-walled carbon nanotubes on microbial community structure in activated sludge. *Letters in applied microbiology 51(4):428-35,* October 2010.

Student Support

Deepankar Goyal, M.S., UMass Lowell Biological Sciences
Gregory Doyle, B.S., UMass Lowell Biological Sciences

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4. Assessing the Transport and Fate of Effluent Organic Nitrogen in the Connecticut River and Long Island Sound Using Mass-Mapping Proteomics Technology (USGS 2009MA186B)

Principal Investigator: Dr. Chul Park, UMass Amherst
Start Date: 4/1/2009
End Date: 5/31/2010
Reporting Period: March 1, 2010 – June 30, 2011
Funding Source: USGS (104B)
Focus Categories: Water Quality, Acid Deposition, Nutrients
Descriptors: Water Quality

Problem and Research Objectives

Significant efforts have been made to reduce the nitrogen released from wastewater treatment plants (WWTPs) and this has been mainly achieved by upgrading the facility for enhanced nitrogen removal through nitrification and denitrification. Though these processes are effective for removing inorganic nitrogen (ammonia and nitrate), organic nitrogen remains little changed, presumably due to its recalcitrant nature, which leads to organic-N becoming a substantial fraction of the N in the final effluent. Thus, one major issue with effluent organic-N is whether it degrades and becomes bioavailable in receiving waters.

Our research group proposed a unique research plan that bases on proteomics analysis to characterize effluent proteins and to assess their fate in receiving waters. Better characterization of effluent proteins and better understanding of their fate in receiving water are critical, as proteins comprise a major fraction of effluent organic nitrogen. Furthermore, as proteins can be characterized at a molecular level, profiling of effluent proteins and tracking them in a well defined laboratory bioassay (that mimics receiving waters) will further enable us to determine the fate of proteins, thus a significant fraction of organic-N, in receiving waters. The specific objectives of this project are as follows:

- Determine and characterize proteins in wastewater effluents from major dischargers to the Connecticut River, thus to Long Island Sound.

- Perform a laboratory bioassay and apply proteomics analysis before and after the bioassay to evaluate the bioavailability of effluent proteins in receiving waters.

Methodology

This research was conducted in two phases: 1) collecting effluents samples and characterizing effluent proteins from various wastewater treatment plants and 2) performing a laboratory bioassay to investigate the fate of effluent proteins and organic nitrogen in receiving waters.

Collection of samples. Primary and secondary effluents were collected from three wastewater treatment facilities that discharge to the Connecticut River in Western Massachusetts. The Northampton and Amherst facilities use conventional activated sludge while the Springfield Regional Wastewater Treatment Facility uses the Ludtzac Ettinger process to treat their wastewater. Samples were collected in plastic containers kept on ice until processed later the same day. Total suspended solids (TSS), volatile suspended solids (VSS) and chemical oxygen demand (COD) measurements were taken the day of collection, while samples were frozen for later measurement of protein, total nitrogen, ammonium, nitrate and nitrite. The secondary effluent from each sample was also separated through a 0.45 µm filter.

Quantification of Proteins in Effluent. The Frølund adaptation of the Lowry method (1995) was mainly used to quantify proteins in wastewater effluents. This method can account for the interference of humic compounds in protein measurement. This method, however, often produced falsely negative protein concentrations for unconcentrated effluent samples. Thus, in this research the quantity of proteins in unconcentrated effluent was measured using the original Lowry method, while the protein concentration in ammonium sulfate concentrated samples were measured using the Frølund adaptation of the Lowry method. Light absorbance was measured with the Thermospectronic Genesys 10 UV Spectrophotometer (Thermo Spectronic, Madison, WI, USA) and concentrations calculated from a standard curve created from 0, 10, 25, and 50 mg/L BSA standards.

Ammonium Sulfate Precipitation. In order to visualize proteins using sodium dodecyl sulfate polyacrylamide gel electrophoresis (SDS-PAGE), secondary effluent and secondary filtered effluent were concentrated with 50% ammonium sulfate. The appropriate mass of ammonium sulfate was combined with 150 mL of primary effluent, 1.2 L of secondary effluent, and 2.2 L of 0.45 µm filtered secondary effluent, in 500 mL centrifuge bottles and one glass pyrex bottle (for 1 L of the 0.45 µm filtered secondary effluent). Precipitation procedures were conducted on ice for more than 12 hours, followed by centrifuging at 11,730 g for 45 minutes. The precipitate was re-suspended in a known volume in phosphate buffered saline (PBS: 10 mM NaCl, 1.2 mM $KH2PO_4$, 6.0 mM Na_2HPO_4), then dialyzed extensively in the same buffer with multiple changes at 4°C using a 6-8 kDa cellulose membrane. Sodium Dodecyl Sulfate – Polyacrylamide Gel Electrophoresis. The SDS-PAGE was performed according to the method of Laemmli (1970). Ammonium sulfate concentrated samples were prepared for size separation on polyacrylamide gels by incubating at approximately 95 °C for at least 10 minutes with a 3.3x sample buffer consisting of XT Mops sample buffer and a reducing agent (Bio-Rad, Hercules, CA, USA). Some samples were heat concentrated for up to one hour. Following heat concentration, samples were centrifuged at 12,000 rpm for 3 minutes and the supernatant was used for SDS-PAGE. Prepared samples were loaded onto pre-cast Criterion XT 4-12% gradient gels (Bio-Rad, Hercules, CA, USA) and separated on the gels by a current of 80 V for 20 minutes, followed by 100V for two hours. After electrophoresis, gels were stained with silver nitrate or coomassie brilliant blue using Bio-Rad's Silver Stain Kit or Bio-Safe stain (Bio-Rad, Hercules, CA, USA). Gel images were digitally recorded using a CanoScan 8800F desktop scanner (Canon, Tokyo, Japan).

Zymogram analysis. Samples were subjected to zymogram analysis to determine if they contained active proteolytic enzymes. Enzyme activity was determined by separating proteins using electrophoresis in a casein infused gel (Bio-Rad, Hercules, CA, USA). Before electrophoresis the samples were combined with zymogram buffer (Bio-Rad, Hercules, CA, USA) and centrifuged at 12,000 rpm for 3 minutes; the supernatant was used for the zymogram analysis. Gel images were digitally recorded using a CanoScan desktop scanner (Canon, Tokyo, Japan).

Chemical Analysis. Total protein in each of the effluents, both raw and concentrated with ammonium sulfate, was measured using the Lowry method (1951) and determined with a calibration curve generated with bovine serum albumin (Fisherbrand Scientific, Pittsburg, PA, USA). On the day of sample collection, COD, TSS and VSS were measured for primary and secondary effluents according to Standard Methods (2005). COD was measured for secondary effluent filtered through a 0.45 μm filter, as well. Light absorbances for COD tests were determined using the Thermospectronic Genesys 10 UV Spectrophotometer (Thermo Spectronic, Madison, WI, USA) and concentrations calculated from a standard curve using 0, 10, 75, and 150 mg/L KHP standards.

Nitrogen species. Total nitrogen concentrations in primary and secondary effluent, and 0.45 μm filtered secondary effluent were determined using the persulfate method (Hach, Loveland, CO, USA) and confirmed using a Shimadzu TN analyzer (Shimadzu TOC-VCPH with TNM-1, Shimadzu North America, SSI Inc., Columbia, MD, USA). Ammonium, nitrate and nitrite ions in the solution phase (<0.45 μm) of primary and secondary effluents were measured using a Metrohm ion chromatograph (Metrohm, Herisau, Sz). Organic nitrogen was estimated by subtracting the sum of the nitrogen ions from the total nitrogen.

Laboratory bioassay. Several incubation conditions have been tested in an effort to establish the final protocol of laboratory bioassay for this research. Some earlier incubation conditions included: 1) no mixing, 2) intermittent mixing, and 3) continuous mixing of the bioassay bottles. Each set of bioassay included a killed control set to make sure that changes in proteins and organic nitrogen during the bioassay were caused by biological activity. The earlier experiments also tested different dilution sets between effluents and the Connecticut River water at 1:9 and 5:5. For this laboratory bioassay, effluents from Springfield Regional Wastewater Treatment Facility were mainly used. The final bioassay protocol includes following conditions:

1) Filter river water using 100 μm filter.
2) Use 5:5 ratio for river water and secondary effluent for incubation.
3) Perform a separate bioassay on dissolved and whole fraction of secondary effluents.
4) Provide continuous mixing during the incubation.
5) Place the incubation bottles under natural sunlight conditions.

During this laboratory bioassay we also performed Tangential Flow Filtration (TFF) to effectively concentrate the sample before conducting all protein related analysis. Following the concentrating stage, proteins were separated by sodium dodecyl sulfate polyacrylamide gel electrophoresis (SDS-PAGE). Some effluent concentrate samples were sent to another laboratory for liquid chromatography tandem mass spectrometry (LC-MS/MS) analysis to identify the proteins. In addition, various effluent parameters such as TSS, total organic carbon (TOC), cations, anions, and inorganic nitrogen species, as described earlier, were also measured.

Principal Findings and Significance

The current project has revealed several important and new findings regarding effluent organic nitrogen and effluent proteins. The most important finding of this research is that facilities with more advanced N removal processes contain a greater amount of organic nitrogen with a higher diversity of proteins and active enzymes in their final effluents. This indicates that effluent organic-N in advanced wastewater treatment plants differs significantly than that found in conventional wastewater treatment systems. The full effects of released enzymes and proteins in the receiving ecosystem are unknown, but are thought to increase the bioavailability of natural organic matter and to modulate nutrient cycling in the receiving water. As advanced removal of N becomes mandatory in wastewater treatment, it is imperative that this process and potential unintended consequences be fully understood.

The consistent differences between effluent protein profiles from each of the treatment plants investigated further suggest that effluent proteins and enzymes could serve as "fingerprints" of distinct wastewater treatment works and provide a means to track their fate in receiving waters. This fingerprinting concept was employed in the bioassay during the later part of this research and partially used to track the fate of preselected proteins during the incubation. Other major findings and significance of this research can be further summarized as below.

- The research revealed that proteins are significantly correlated with organic nitrogen in effluent from each of the wastewater treatment facilities, demonstrating the significance of protein molecules in effluent organic nitrogen. We believe that there no previous study that has found this relationship or addressed the issue of proteins being an indication, or representative, of effluent organic nitrogen.
- We believe that this is also the first study showing changes in protein profiles, at a molecular level, across processes in a wastewater treatment plant. The results from this approach allowed direct evidence that some influent wastewater proteins persist through the wastewater treatment process and some of these proteins are actually active proteolytic enzymes.
- The research also showed that some bacterial proteins and enzymes that are generated during a biological treatment do indeed end up in the secondary effluent, as so-called soluble microbial products (SMP).
- The finding of active enzymes and proteins in filtered effluent samples is also important to note since the addition of a filtration process to a facility, such as microfiltration, is not likely to improve the capture of these potentially biological compounds.
- Because of these protein results, we gained new knowledge that different treatment works release different sets of proteins (thus, organic nitrogen) and proteolytic enzymes: this information could not be achieved by simple quantitative data or conventional size fractionation techniques. These results are important in that we do not know how these proteins and enzymes behave in the receiving water and what ecological and environmental impacts they may have. The study has provided us a chance to better characterize and identify effluent proteins and enzymes, which can be tracked thoroughly in well-defined laboratory bioassays or even directly in the receiving water.
- The bioassay that was designed to mimic the reaction of wastewater effluents in natural receiving waters requires natural sunlight and continuous and uniform mixing during the incubation.

- Concentrations of both inorganic nitrogen (ammonia and nitrate) and organic nitrogen changed greatly during the bioassay, indicating a degradation of organic nitrogen and release of newly generated organic nitrogen.
- After the incubation, new soluble protein bands were detected along with a substantial increase in algal biomass. This observation suggests that receiving waters utilized available effluent nitrogen, including organic nitrogen, and release of proteins from grown algal biomass contributed to a new set of dissolved organic nitrogen remaining in the bioassay.

Publications and Conference Presentations

<u>Articles in Refereed Scientific Journals</u>

Westgate, P. and Park, C. (In revision) Evaluation of proteins and organic nitrogen in wastewater treatment effluents. Environmental Science and Technology.

<u>Dissertations</u>

Westgate, Pamela. (2009) MS Thesis: Characterization of Proteins in Effluents from Three Wastewater Treatment Plants that Discharge to the Connecticut River, MS Environmental Engineering, Aug 2009

<u>Conference Proceedings</u>

Westgate, P. and Park, C. (2009) Evaluation of Effluent Proteins: Towards Characterization of Effluent Organic Nitrogen, Water Environment Federation 82nd Annual Technical Exhibition and Conference (WEFTEC 2009), October 2009, Orlando, FL.

Student Support

Primary funding was used to support Pamela Westgate for her MS research. Ms. Westgate defended her MS thesis in August 2009. Please see above.

Partial funding was also used to support the PI’s PhD graduate students, Meng Wang and Dong-Hyun Chon, who participated in this project by operating laboratory scale activated sludge systems and provided wastewater effluent for proteomic characterization for this research.

Notable Achievements and Awards

As discussed above, the current research has revealed very important findings regarding the release of nitrogen from advanced wastewater treatment systems and its impact on the receiving water ecosystem. Nitrogen in domestic wastewater effluents is a hot topic, garnering a lot of attention these days from researchers and plant operators as regulators increase pressure to reduce nitrogen discharges from wastewater treatment plants. Due to the significance of the findings of this research, we could submit a full journal manuscript to Environmental Science and Technology (currently, in revision process). Furthermore, Springfield Water and Sewer Commission (SWSC), who provided the matching funds to this project, will likely continue the funding on this research topic. Dr. Douglas Borgatti from SWSC and the PI will also present the findings of this research in annual New England Water and Environment Association Conference in January 2011 to discuss a regional nitrogen issue for wastewater treatment facilities.

5. Characterization of Flow and Water Quality of Stormwater Runoff from a Green Roof (USGS 2009MA199B)

Principal Investigator: Dr. Paul Mathisen, Worcester Polytechnic Institute
Start Date: 4/1/2009
End Date: 2/28/2011
Reporting Period: March 1, 2010 – June 30, 2011
Funding Source: USGS (104B)
Focus Categories: Hydrology; Water Quality; Water Quantity
Descriptors: Green Roof; Water Quality; Storm Water Runoff; NPDES Permit

Problem and Research Objectives

Low Impact Development (LID) techniques for site design are increasingly being utilized to mitigate the negative impacts associated with stormwater runoff, and green roofs are one such application. The ability of green roofs to reduce the total and peak volumes of stormwater runoff has been fairly well documented, but performance varies in different climate zones, and there is limited information available regarding green roof effectiveness in New England, a region whose weather patterns are notoriously variable from season to season and often even day-to-day. Additionally, there are questions regarding the impact that green roofs have on water quality. While there seems to be a general consensus that green roofs will leach phosphorus, and sometimes other contaminants into stormwater runoff within the first few years after installation, it is assumed that this phenomenon will not continue after the green roof vegetation has been established. However, it is still unclear whether or not this assumption is valid, and very few research projects have attempted to provide the necessary insight into the hydrologic and chemical processes that are contributing to this question.

Accordingly, the goals of this research were to provide insight into the hydrologic and geochemical processes that contribute to green roof performance. The specific objectives included the following:

- Determine the effectiveness of a green roof in attenuating stormwater flow
- Document a green roof's impact on water quality, specifically regarding phosphorus
- Identify the key components of the processes that are likely leading to the highest variability in observed water quality parameters – hence, the highest potential that a change in design could lead to significant improvements.

In addition to providing insight into green roof performance, these objectives are intended to provide a foundation for future research efforts to explore the behavior of phosphorus in soil solutions and its implications for stormwater treatment.

Methodology

The methodology for achieving the project objectives combined field monitoring and laboratory testing and analyses to characterize the quality of runoff associated with the Nitsch/Magliozzi Green Roof, an extensive green roof located on top of a new residence hall at WPI. This roof, which was donated to enhance the sustainability of the building and foster continued research and education, provided the context for the project. The research tasks included field monitoring of the roof drainage, laboratory testing of green roof panels under simulated rainfall conditions, bench-scale testing of phosphorus desorption from the growing medium, and laboratory analyses of water quality, soil characteristics, and plant phosphorus content. The methodology provided a basis for gaining a better understanding of the relationship between rainfall and runoff volumes, phosphorus sorption/desorption in the growing medium, and plant uptake processes.

The field monitoring program focused on the seasonal variations of water quality throughout a complete growing season. Two flow meters and sampling ports have been installed within the storm drain system of the residence hall: one to measure drainage from the green roof; and the other to measure drainage from the "non-green" portion of the roof. Using these sampling ports, a total of 25 grab samples from each roof were collected and analyzed between June 2009 and April 2010.

The laboratory testing and analysis program was developed to characterize both the stormwater retention performance and water quality characteristics of the green roof. For this program, two of the green roof panels were brought into a greenhouse maintained at WPI by WPI's Biology Department. A stand was constructed which allowed for the application of simulated rainfall and collection and measurement of runoff for each panel. For water quality monitoring, runoff from each panel was detoured through a flow-through device attached to a water quality monitoring sonde (Hach MS5 Hydrolab unit), and grab samples were collected at key points during the simulated storms. The Hach MS5 Hydrolab units, one of which was acquired using support from this grant, were important components of this system. Soil and plant samples were also collected, and additional bench-scale tests were completed to characterize the nature of the phosphorus desorption from the media. All samples of water, plant, and soil were analyzed in the water quality laboratory in WPI's Department of Civil and Environmental Engineering.

Principal Findings and Significance

In regards to storm-water flow attenuation, results from the greenhouse experiments showed that green roof performance was more effective for smaller storms, and was influenced by the soil properties (including field capacity and moisture content). Overall, these results are consistent with the published literature. For example, the reduced retention capacity observed during higher flow conditions is a common trend that has been reported for extensive green roof performance. At high rainfall intensities, the field capacity of the green roof panels is quickly exceeded, and the thin layer of the extensive green roof design does not provide much storage capacity. However, while the growing medium did not provide much storage during the heavier simulated rain event, the green roof vegetation's ability to rapidly uptake water when it becomes available did provide a stormwater retention benefit. The improved performance during the lower flow conditions was found to be more heavily influenced by the soil than by the plants. The highest retention rates in the simulated rain events were observed when the antecedent moisture content was low (9-11%). In contrast, for a light rain event, the moisture content of the soil at the beginning was the highest of all tests (26%), and the green roof panels retained only 38% of the influent volume, despite the fact this simulated storm used the smallest volume of water of all simulated events. Clearly the growing medium's field capacity is a critical design factor that is indicative of green roof performance.

In regards to water quality, phosphorus concentrations observed in runoff during greenhouse tests were similar in magnitude to the concentrations in samples collected from the green and white roofs, which were relatively high. These high concentrations were found to be primarily influenced by phosphorus in the growing medium, which quickly desorbs in response to flushing due to storm events. For all greenhouse tests, the phosphorus concentrations (and other constituents as well) showed up in the "first flush" runoff samples and continued to increase throughout the duration of the storm and after the simulated rainfall had stopped. This trend was consistently observed in all storms, regardless of their size or intensity. These results indicate that the desorption of phosphorus from the growing medium happens quickly, and the soil is not rapidly depleted of its phosphorus content. Also, the green roof panel whose soil was higher in phosphorus concentration (Stand B) also produced runoff with higher phosphorus concentrations than the other panel tested in the greenhouse (Stand A). Meanwhile, the growth of green roof plant material and its associated nutrient uptake processes did not appear to reduce the amount of phosphorus that ended up in the runoff. These results confirm that the growing medium is the source of phosphorus in runoff. However, while a bench-scale laboratory experiment indicated that phosphorus levels in runoff may decrease over time, the rate of desorption is not constant and cannot be easily predicted. Additional investigations will be needed in order to predict the long-term impact of a green roof on phosphorus loading.

With consideration to the design of green roofs, a number of key processes/factors were defined. First, this research showed that soil storage and soil moisture content are particularly important considerations with respect to green roof performance. Soil storage is heavily influenced by antecedent moisture content, and soil moisture content is a function of both weather, which cannot be controlled, and plant variety, which generally can be controlled. These results should help future designers determine whether the weather patterns in a particular location where a green roof is being considered will be hindrance to the effectiveness of a green roof. Areas experiencing significant amounts of rainfall that may keep the soil at field capacity would not be a good choice. However, selecting plant varieties that quickly uptake water, such as sedum and delosperma, will provide the ability to regenerate the holding capacity of the growing medium and will improve the performance of green roofs. Also, efforts should be taken to engineer new soil media that will maximize the field capacity of green roof designs. Second, the research showed that the leaching of phosphorus from the growing medium must be taken into consideration when designing a green roof. Previous studies have made assumptions that the leaching of phosphorus will decrease over time and many have predicted that the phenomenon will only occur for a few years after installation. However, the results of this study indicate that this assumption may not be valid. The long-term phosphorus loading resulting from a green roof may continue longer than previously assumed. Until additional investigations are conducted to develop a prediction model, the impacts of a green roof must be given careful consideration if being installed where phosphorus levels in stormwater are a concern. Further, it is recommended that phosphorus use be minimized in the growing medium. The typical green roof plant varieties, such as those studied here, do not appear to uptake very much of this nutrient, even in their first few establishment years.

In general, these results provide a basis for developing improved predictions of storm-water retention performance, gaining deeper insight into the transformations of phosphorus in the green roof panels, and developing a process by which continued, in-depth study could be performed under controlled laboratory and field conditions.

Publications and Conference Presentations

The results summarized in this summary report are also described in more detail in a Master of Science thesis prepared by Suzanne LePage in partial fulfillment of the requirements for her Master of Science degree at Worcester Polytechnic Institute. The results were also disseminated via a presentation at the EWRI Congress of the American Society of Civil Engineers (ASCE), and via a poster presentation at the Annual Water Resources Conference in Amherst, MA. The details of these items are included in the following listing:

<u>Dissertations/MS Theses</u>

LePage, Suzanne, 2010. An investigation of the hydrologic and geochemical processes contributing to green roof performance. MS Thesis, Worcester Polytechnic Institute, completed in May 2010.

<u>Other Publications and presentations</u>

LePage, Suzanne and Paul Mathisen, 2010. *An investigation of the hydrologic and geochemical processes contributing to green roof performance.* Presentation at the American Society of Civil Engineers (ASCE) World Environmental and Water Resources Congress 2010. Providence, Rhode Island. May 20, 2010.

LePage, Suzanne, 2010. *An Investigation into the Water Quality Impacts of a Green Roof*. Poster presentation at WPI Graduate Achievement Day, Worcester Polytechnic Institute, Worcester, MA on March 31, 2010.

LePage, Suzanne,2010. *An Investigation into the Water Quality Impacts of a Green Roof*. Poster presentation at the 7th Annual Water Resources Research Conference, Amherst, MA on April 8, 2010. 2nd Place Award.

Student Support

This project provided equipment that assisted the research program of 1 graduate student, Suzanne LePage, at Worcester Polytechnic Institute. The matching funds designated in this grant included the student time and effort as part of an independent study project (ISP) completed in the fall of 2009.can we find out her department?

Notable Achievements and Awards

2nd Place award for Poster entitled "*An Investigation into the Water Quality Impacts of a Green Roof*", which was presented at the Seventh Annual Water Resources Research Conference Poster Contest on April 8, 2010.

6. Characterizing and Quantifying Recharge at the Bedrock Interface (USGS 2009MA213G)

Primary Principal Investigator: Dr. David Boutt, UMass Amherst
Other PIs: Dr. Stephen B. Mabee
Start Date: 9/1/2009
End Date: 8/31/2012
Reporting Period: March 1, 2010 – June 30, 2011
Funding Source: USGS (104G)
Research Category: Groundwater Flow and Transport
Focus Categories: Groundwater; Water Supply; Water Quantity

Problem and Research Objectives

Evaluating the sustainability of fractured bedrock as a groundwater resource and understanding the environmental impacts of water withdrawals from the bedrock on nearby streams, wetlands, ecosystems and unconsolidated aquifer systems requires an estimate of the recharge and an understanding of the advective flux across the bedrock –overburden boundary. Few published studies address this issue with direct measurement (Rodhe and Bockgard, 2006, White and Burbey, 2007) while others use tracers (e.g. Rugh and Burbey, 2008) and numerical models to study the distribution of groundwater flow in the soil and bedrock (eg., Harte and Winter, 1995; Tiedeman et al., 1998). However, quantifying the flux of water between the overburden and bedrock remains one of the major sources of uncertainty in numerical models (Lyford et al., 2003). The only long term monitoring well that is screened in bedrock in Massachusetts (Figure 1) shows an interesting downward trend in hydraulic head over a period of almost 20 years while the hydraulic head from a nested piezometer in the surficial material above the bedrock piezometer does not show a similar trend. Understanding the dynamics of how systems like these interact is fundamental to improving our ability to manage and regulate these important resources.

The objectives of this project are to evaluate water flux across the overburden-bedrock interface under ambient and stressed conditions and to estimate its hydraulic conductivity in three typical hydrogeologic settings in the glaciated terrain of eastern Massachusetts. The hydrogeological conditions to be examined include thick till overburden, thin till-shallow bedrock and coarse-grained stratified deposits. The work isl being conducted in the Assabet River watershed because this watershed has previously been modeled by the USGS (DeSimone, 2004). The project complements past and ongoing work by the USGS in the New England region that evaluates water availability and the impacts of pumping on shallow aquifers and riparian systems (e.g.., DeSimone et al., 2002, DeSimone, 2004; Carlson et al., 2008). The project is also designed to complement a project underway by the USGS Water Science Center in Northborough, Mass. that is assessing the factors affecting bedrock well yields in the Nashoba terrane. Many of these projects benefited from several years of cooperation between the Office of the Massachusetts State Geologist, University of Massachusetts, USGS and the Mass. Department of Environmental Protection.

The expected outcome of this work is a clearer understanding of the groundwater flux across the overburden-bedrock boundary and how the coupled systems respond to seasonal changes, individual recharge events and potential stresses due to pumping. Acquisition of these data will provide a basis for calibrating numerical models that investigate the effects of groundwater withdrawals (both surficial and bedrock) on stream baseflows and ecosystems.

Methodology

Site Selection

A groundwater-monitoring site in Berlin, MA was chosen in the context of the existing USGS groundwater flow models. The site known as the Gates Pond Reservoir lies in areas where the Nashoba formation outcrops and subcrops and is covered by till or has exposed bedrock. The Town of Hudson Department of Public Works, the water supply agency that regulates the Gates Pond granted access to the site in May 2010. The Gates Pond Reservoir has four pre-existing monitoring wells that are 6 inches in diameter and terminate in bedrock at depths that range between 150 and 245 meters below surface level (bsl). The fractured bedrock intersected by these wells have been extensively characterized for their hydrologic properties as reported in Boutt et al., 2010.

The site was surveyed in June 2010 and maps were developed using resources provided by The Massachusetts Office of Geographic Information, the USGS. In addition to topographical surveys, 2D seismic refraction and electrical resistivity surveys (Figure 1) were conducted in the areas adjacent to the bedrock wells as well as till covered slopes to identify the location of the water table and bedrock interface.

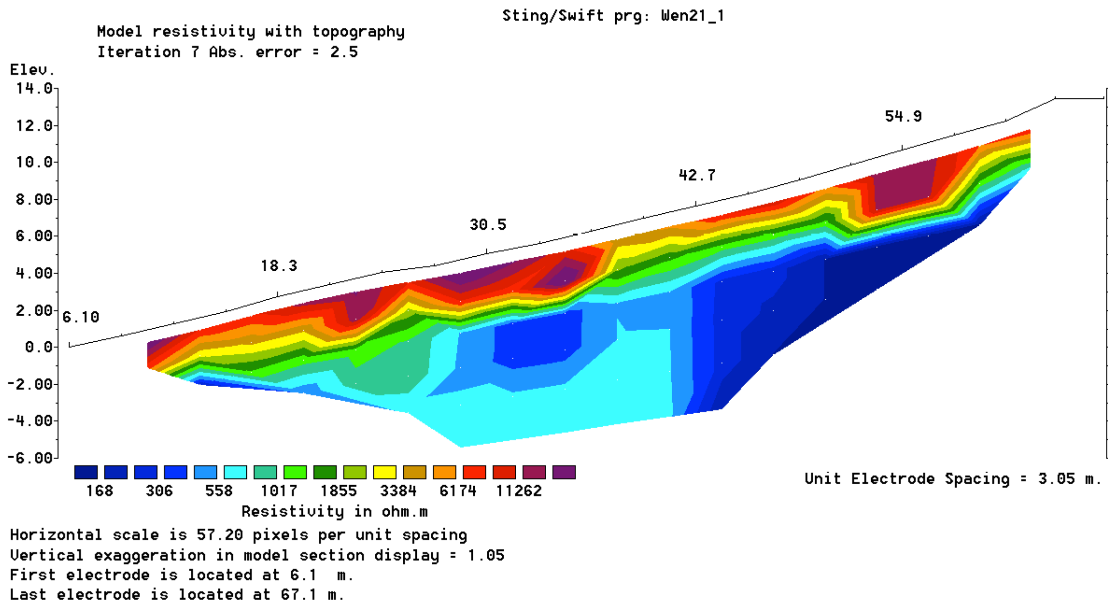


Figure 1. Inverse model of data collected from a East to West Wenner resistivity array on the eastern edge of the thick till (drumlin) deposit.

<u>Site Instrumentation</u>

Two bedrock wells (BMW-1 and BMW-2) were instrumented in May 2010 with Solinst LevelLoggers to measure time-series of temperature and pressure within each of the wells. To correct for barometric pressure influences on bedrock well water levels, a Solinst BaroLogger was installed above ground near BMW-1 to measure atmospheric pressure and air temperature. All four bedrock monitoring wells are instrumented to date with BMW3 and BMW4 monitoring beginning in Winter 2011. Two 1.75 inch diameter surficial wells were installed in both the thin till and thick till packages. The wells were installed to a depth of 1 and 2 meters respectively. Each of these wells was instrumented with a Solinst LevelLogger pressure and temperature sensor. Additionally, pond level was also measured with a LevelLogger. All LevelLoggers installed are logging pressure and temperature data at 5 minute intervals. In addition to the water level and temperature probes, Four Decagon Devices 5TM soil moisture probes were installed within both the thick and thin till. These probes are installed at the top of a slope and at the base of the slope at depths of .5 meters and 1.2 meters. The 5TM probe from Decagon Devices collects volumetric water content (VWC) by measuring and converting the soils dielectric permittivity into soil moisture content using the Topp equation:
VWC = 4.3x10-6 ea3 - 5.5x10-4 ea2 + 2.92x10-2 ea - 5.3x10-2

Data collected from the 5TM probes are logged and transmitted to an off site computer via a Decagon Devices EM50G wireless cellular datalogger. Precipitation measurements are taken from a USGS precipitation gage located in the nearby town of Sterling, Massachusetts. The USGS gage collects precipitation data at 15-minute intervals. Two distributed temperature sensing (DTS) fiber optic probes were built for both BMW1 and BMW2. These probes are deployed in the field and have only been implemented for limited time periods.

<u>Data Collection and Site Characterization</u>

In the spring of 2011 we performed a pump test at the proposed field site (Gates Pond) in the Nashoba terrane. Gates Pond consists of 4 bedrock wells with depths on the order of 250 meters. The wells are located in two NE-SW trending lineaments that are separated by a bedrock ridge, with wells 1 and 2 in the western-most lineament, and wells 3 and 4 the eastern-most lineament. Pumping was performed in well 1, and drawdown was observed in wells 2, 3 and 4. Well 2 displayed significant drawdown during pumping; however, wells 3 and 4 do not display significant drawdown. These results suggest that wells 1 and 2 are hydraulically connected, whereas wells 3 and 4 are not hydraulically connected to the pumping well. We interpret these observations to indicate a strong NE-SW hydraulic conductivity anisotropy. The orientation of this anisotropy is coincident with the orientation of Foliation Parallel Faults (FPF), suggesting that FPFs are playing a major role in permeability anisotropy between wells 1 and 2. Wells 3 and 4 are located in a lineament that is east of the pumping well, suggesting that E-W oriented fractures are less permeable than the NE-SW trending FPFs. What is more, the orientation of the regional maximum horizontal stress in New England is approximately NE-SW, and thus the maximum permeability orientation interpreted from the pump test is parallel to the maximum horizontal stress, and fractures oriented obliquely to the maximum horizontal stress are less permeable.

Data collection began in May 2010 in BMW1 and BMW2. The surficial well, pond, soil moisture measurements began in earnest September 2010. Currently at the site, 22 separate pieces of data (air temperature, barometric pressure, 4 soil moisture contents, 8 hydraulic heads, and 8 water temperatures) will be recorded at 5-minute intervals resulting in 6,336 pieces of data per day. The additional of the 15-minute precipitation measurements at the USGS Sterling, MA gage (96 per day), brings the total number of pieces of data collected per day to 6,432 pieces of data per day to be considered.

Water samples were taken for stable isotope analysis from all surficial and bedrock wells, as well as from Gates Pond Reservoir. Isotope samples may give some insight into subsurface fractionation processes that are expressed within recharge.

Soil cores were taken in the locations of the surficial wells and were analyzed for grain size, porosity, specific yield, total organic content and vertical saturated hydraulic conductivity via falling head permeameter testing.

In order to determine in-situ saturated hydraulic conductivity, reverse slug tests were performed in the surficial wells by removing a volume of water from the well and observing the rate of replenishment in the well via Solinst LevelLogger. Pumping tests were executed using a GeoTech submersible pump in BMW1 and BMW3 respectively and drawdown was monitored in all four bedrock-monitoring wells concurrently.

Data Reduction, Interpretation, Modeling

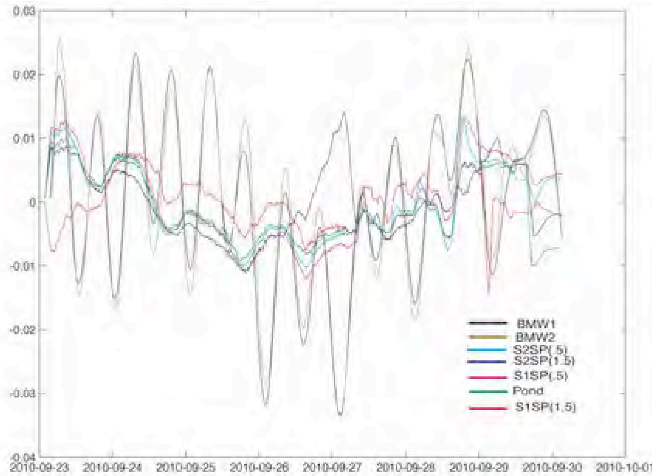
Data collected from the instruments on site is time synced to ensure proper analysis. A MATLAB code was developed to process all of the collected time series data from the deployed instruments. The data was normalized to an arbitrary datum that was established by topographical survey. Portions of the surficial well level data was detrended and cross correlated with the bedrock well level data in order to determine the lagged time response in the bedrock wells from a precipitation input (Lee et al., 2006). A cross correlation is essentially the plotted coefficients of determination (Rr) of a cross plot of the bedrock and surficial wells at a series of lagged sampling intervals. The higher the R2 value, the more likely there is a similarity in the well responses at that particular lag time. Currently, there is an effort being made to filter the detrended data in order to remove the periodic barometric loading effect. The barometric loading may be providing false correlations.

Synthesis and Modeling

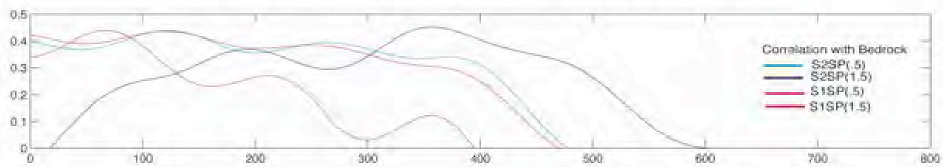
Observations of water and energy transport (i.e. temperature) will be summarized and compared to site characteristics such as till thickness, till composition, topographic setting, hydraulic heads of till and bedrock, bedrock hydraulic properties, distance to mapped structural features, and characteristics of fracture networks mapped in nearby outcrops. Recharge rates will be calculated and summarized for each site at monthly intervals. Relationships between the hydraulic conductivities determined at the bedrock-till interface and field observations will be made. Using the calibrated 1-dimensional models, we will perform some preliminary estimates of the impact of bedrock water withdrawals on the flux of water across the boundaries at the different sites. Fluid withdrawal will be simulated by modeling a pumping well placed into the bedrock at a specified depth. Results from the models will be used to infer the impact of further use of bedrock water supplies on surficial water supplies and recharge rates into the bedrock.

The hydraulic data together with the temperature data will be used to build 1-dimensional coupled saturated-unsaturated water flow and temperature models using the general finite-element method solver COMSOL multiphysics (Fleming, 2009). Hydraulic and heat boundary conditions will be provided by the field measurements. The integration of both head and temperature data into a model such as this reduces the degrees of freedom and will allow an estimation of the vertical flux across various boundaries (water table, till-bedrock interface) present in the model (Anderson, 2005) that is constrained by observations of head and temperature. A split data approach will be used to calibrate the model reserving the other half of the data to predict water flux under different hydraulic conditions throughout the data set. The hydraulic properties of the bedrock-till interface will be the main calibration parameter.

For each a site a detailed quantitative model will be developed to understand the movement of water from the surface through to the bedrock. Recharge rates to the till (where present) will be estimated and the amount of leakage (recharge) to the bedrock will also be determined using the data collected. Results from this analysis will yield a detailed set of water fluxes for the hydrologic periods during data collection. The hydraulic properties of the bedrock-till interface will be an important calibration and model result.



A



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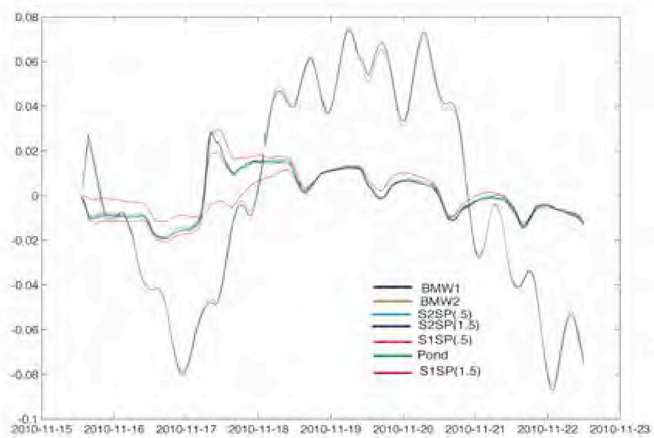

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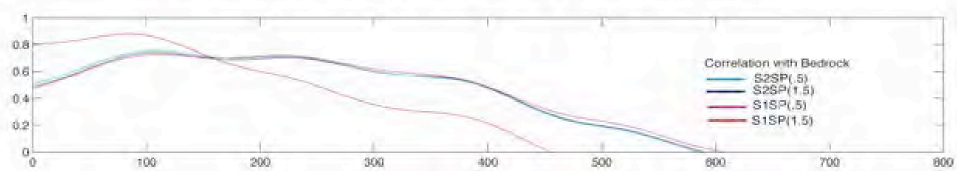

D

Figure 2. a) Detrended water level data from one week in September 2010. b) Cross correlated data between surficial and bedrock wells for the detrended data in 2a. c) Detrended water level data from one week in November 2010. d). Cross-correlated data between surficial wells for the detrended data in 2c. Note that the architecture of the correlation structure changes dramatically. There implies that the signals are responding to a common input. A low pass filter will need to be worked into the MATLAB code in order to refine the signal and eliminate the barometric loading bias that is apparent in all of the signals.

Preliminary Findings and Significance
Work on this project began on September 1, 2009 with site selection and characterization and continues today with time series analysis and aquifer hydraulic characterization. Figure 3 includes a site locus as site map showing the location of monitoring equipment as well as geophysical survey lines. Time series data collected from wells BMW1 and BMW2 have captured bedrock recharge timing and magnitude at the Gates Pond Site (Fig. 4).

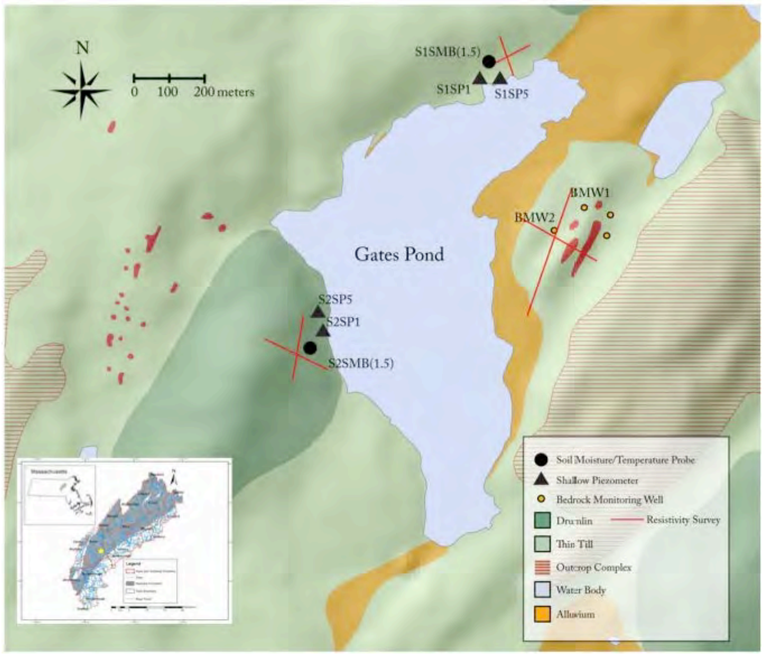

Figure 3. Site map of the Gates Pond Reservoir. In the lower left corner is a site locus showing the location of Gates Pond in relation ship to the Nashoba Formation in Massachusetts.

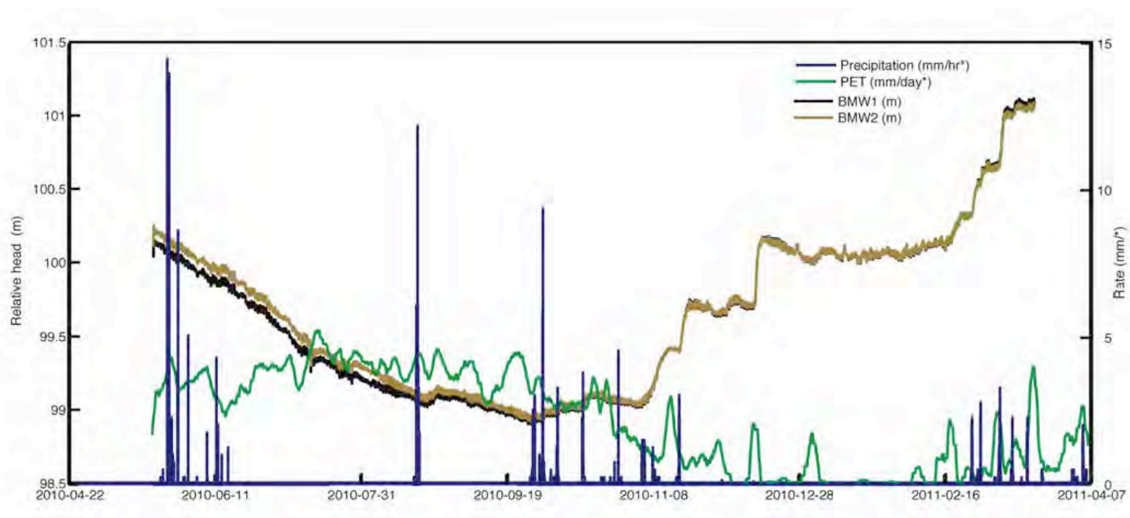

Figure 4. Time series of normalized head from BMW1 and BMW2. Potential evaporation rate as calculated using the Thorthwaite approximation and precipitation rate are also plotted.

Summer precipitation events do not to appear to have an appreciable effect on the trend of the bedrock head. Instead, bedrock recharge occurs during times with reduced potential evapotranspiration. The bedrock recharge occurs more rapidly as the Fall season progresses. During the earliest part of Fall, bedrock wells and the surficial wells respond with approximately the same magnitude until the middle of

Fall when the bedrock response to precipitation increases in its magnitude (Fig 5). This sudden change in how the bedrock system reacts suggests that there is a threshold in the system that must be met in order for there to be an appreciable amount of recharge to the bedrock. The investigators believe that the threshold is represented by the hydraulic conductivity in the till. While at lower soil moisture values, the tills have little ability to transmit water through the matrix. As soil moisture content increases (Figure 6), the till's hydraulic conductivity increases until an effective hydraulic conductivity is achieved. It is at this effective hydraulic conductivity that water may transmit water to deeper depths.

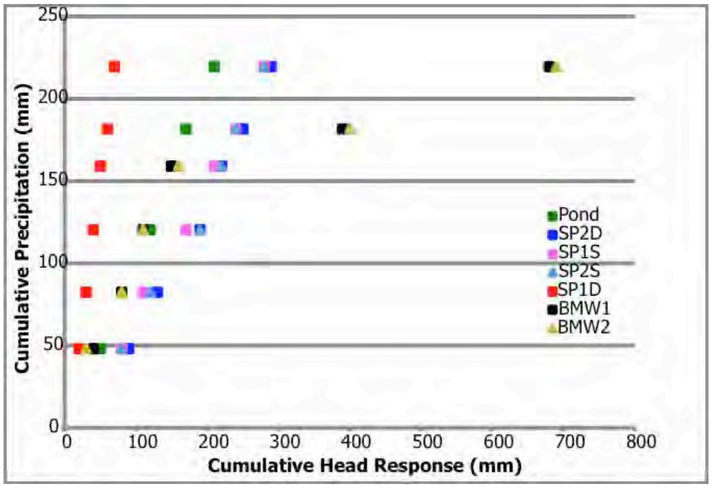

Figure 5. Cumulative plot of the early fall when recharge began to occur in the bedrock monitoring well. Note that the bedrock wells (BMW1 and BMW2) respond to precipitation events similarly to the surficial wells until a point after which the bedrock monitoring wells respond very differently.

There is also a steepening of the slope associated with the bedrock response to recharge that may be associated with an increase in the soil moisture content. Preliminary observations indicate that bedrock recharge is highly dependent upon soil moisture dynamics and overburden thickness.

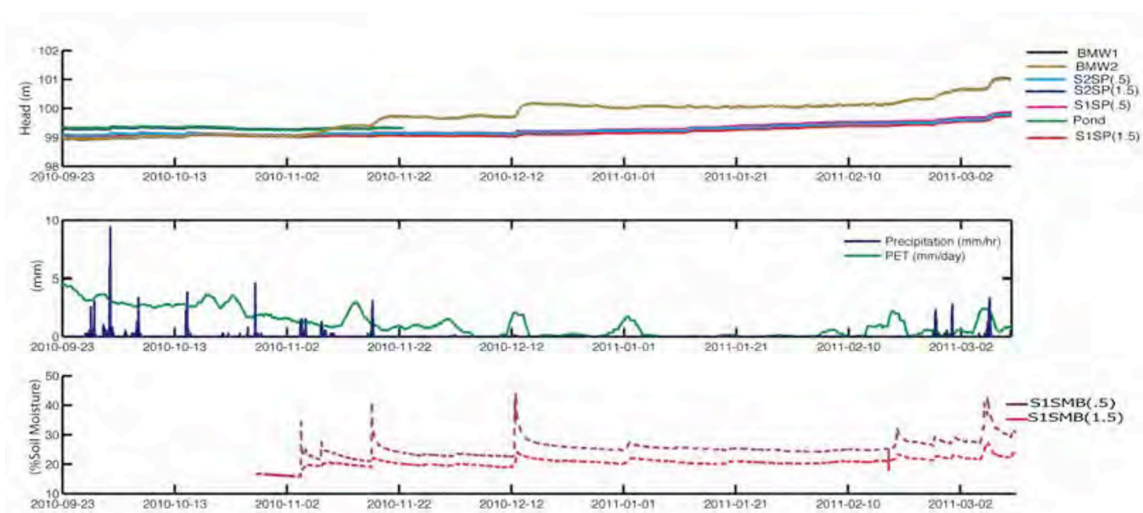

Figure 6. This figure illustrates the responses in the surficial water table compared to bedrock water table.

Soil hydraulic conductivities were taken at each site at two different depths. The slug test results, when solved with the Hvorslev method, show that a distinct non-linearity appears in late time during the test. This represents radial flow and can be attributed to the limited vertical connectivity to the aquifer due to the well construction techniques that were used. This non-linearity is shown in Figure 7.

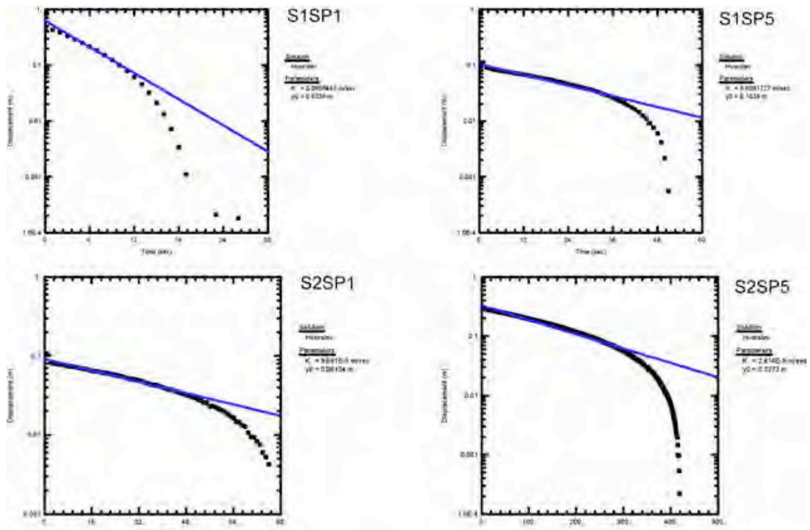

Figure 7. Slug test results as analyzed using the Hvorslev method. The non-linearity confirms the dominance of radial flow and therefore, results with a more appropriate solution (like Cooper et al.) can be used to get the horizontal hydraulic conductivity. This value can be compared to the falling head permeameter results to determine horizontal/vertical anisotropies in the system.

Coincidental to the bedrock recharge, the soil moisture begins to increase, suggesting that the hydraulic conductivity of the soil must also increase. Hydraulic conductivity varied between the thin and drumlin till sites and is summarized in Table 1 below.

Site	Slug Test	Falling Head
Thin Till		
S1SWS	8.4e-4 m/s	1.28E-7 m/s
S1SWD	1.7e-4 m/s	1.8E-7 m/s
Drumlin		
S2SWS	9.7e-5 m/s	3.1E-7 m/s
S2SWD	2.6e-5 ,/s	3.5E-7 m/s

Table 1. Hydraulic conductivities of surficial materials located at Gates Pond Reservoir.

Future Work

Plans are currently being developed to install an additional bedrock monitoring well at the Gates Pond site. While installing the well, the bedrock/till interface will be cored and characterized. The new bedrock well will be instrumented with fiber optic

DTS probes as well as instrumented to measure hydraulic head at multiple depths. The well will also be geophysically logged for resistivity, with a heat pulse flow meter, imaged and interpreted via optical televiewer and caliper. Time series data will continue to be collected and analyzed from all wells, soil probes and the pond. Stable isotope analysis will also be performed from regular sampling at the site. Stable isotopes will give the Investigators insight as to the origin of the water onsite and will allow the investigators to determine whether responses in bedrock wells are the result of advection across the bedrock/till interface or an expression of the pressure wave associated with hydraulic diffusion and surface loading of meteoric water mass.

Publications and Conference Presentations

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Characterizing Groundwater Recharge Across the Surficial/Bedrock Interface, 2010. Bevan, L.B., D.F. Boutt, S.B. Mabee. Massachusetts Water Research Resource Center Annual Conference. April, 2010. (Poster session).

Developing a Conceptual Model for Bedrock Recharge in the Glaciated Northeastern US, 2010. Bevan, L.B., D.F. Boutt, S.B. Mabee. American Geophysical Union Conference. December, 2010. (Poster session).

Developing a Conceptual Model for Bedrock Recharge in the Glaciated Northeastern US, 2011. Bevan, L.B., D.F. Boutt, S.B. Mabee. Massachusetts Water Research Center Annual Conference. April, 2011. (Poster session). First place poster submission.

Developing a Conceptual Model for Bedrock Recharge in the Glaciated Northeastern US, 2010. Bevan, L.B., D.F. Boutt, S.B. Mabee. Novel Methods for Subsurface Characterization Conference. May, 2011.(Poster session).

Student Support

Liam B. Bevan is fully supported by this project. He is pursuing an M.S. degree in geology in the Department of Geosciences at the University of Massachusetts, Amherst.

Evan Earnest-Heckler is partially supported by this project. He has been assisting with field work and developing a detailed characterization of the fractured bedrock of the site. He is pursuing a PhD in geology in the Department of Geosciences at the University of Massachusetts, Amherst.

Shakib Ahmed used data collected from this project for his Senior thesis. He recently completed his B.S. in geology in the Department of Geosciences at the University of Massachusetts, Amherst.

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7. Monitoring and Modeling Chromophoric Dissolved Organic Matter in Neponset River and Boston Harbor Using GIS and Hyperspectral Remote Sensing (USGS-2010MA231B)

Principal Investigators: Qian Yu, Weining Zhu, Department of Geosciences, UMass Amherst
Start Date: March 1, 2010
End Date: February 28, 2011
Reporting Period: March 1, 2010 – June 30, 2011
Funding Source: USGS (104B)
Descriptors: Water Quality, Hydrogeochemistry, Models

Problem and Research Objectives

(1) Problems

As one of the major components of dissolved organic matter (DOM), dissolved organic carbon (DOC) is a key factor for water quality. The organic molecules that make up the DOC pool in fresh and coastal waters come from natural sources, mainly from the decay of terrestrial and aquatic plants, algae and bacteria. Meanwhile, anthropogenic activities, such as sewer release and agricultural fertilization, also have strong impact on DOC contents and compositions in watershed regions with high population density. The watershed flux of DOC from terrestrial landscape to rivers has wide-ranging consequences for aquatic chemistry and biology. DOC affects the complexation, solubility, and mobility of metals as well as the adsorption of pesticides to soils, and is therefore a critical water quality parameter important for human health. In addition, DOC also plays an indispensable role in the cycle of terrestrial and atmospheric CO_2, and hence implies the climate change. Currently, due to physical and biological complexity, multiple scales of freshwater systems, and biogeochemical reactivity at the land-water interface, changes to DOC fluxes in response to terrestrial sources and climate change are not well-known, and so as to be hard to evaluate their potential impact on water quality. To quantify the seasonal or interannual variation of DOC flux, the first thing is to accurately estimate DOC amount in riverine and estuarine regions. This is not only crucial to analyze their sources and transport mechanisms, but also subsequently helpful for monitoring and controlling water quality and for understanding the regional and global carbon cycle.

Our study site is the Neponset River and Boston Harbor regions. The Neponset River is located in eastern Massachusetts, starting at the Neponset Reservoir and meandering generally northeast for approximately 29 miles to its mouth at Dorchester Bay of Boston Harbor. The Neponset River is fed by a watershed of approximately 130 square miles, including numerous aquifers, wetlands, streams, surrounding upland areas, and the watershed is home to about 330,000 people. The Neponset has been heavily polluted before, but at present it is clearer due to much remediation. Several water quality monitoring programs are running for this river, for example, the Massachusetts Water Resources Authority's harbor and river monitoring program monitors the impacts of combined sewer overflows (CSOs) on the Harbor and tributary rivers. The sampling crew measures temperature, bacteria, algae, water clarity, nutrients, and suspended solids in Boston Harbor and Neponset rivers. However, there are some shortcomings of current monitoring programs. First, DOM/DOC is not usually directly measured. The conventional water quality parameters, such as ions concentrations, pH values, and dissolved oxygen, are not appropriate to estimate DOC’s concentration. Second, even though DOM/DOC are sometimes measured, most of the measurements are only point sampling. Such field survey is insufficient or less reliable for evaluating the distribution and dynamic of organic matter at a large spatial scale. Third, our previous study shows DOC has strong seasonal and even daily variations. Frequent and synoptic DOC measurement is in great need for monitoring DOC flux spatial temporal variation in a large area.

Inversion of DOC concentration from satellite images has great potential to overcome the shortcomings of field surveys. The concentration of in-water components influences water’s absorption and backscattering coefficients and hence change the radiance received by satellite sensors. Therefore, based on satellite images, we can inversely predict those components. Although DOC can not be fully estimated from satellite images since part of it is not photoactive substances, the photoactive fraction of DOM, chromophoric dissolved organic matter (CDOM), could be used as the tracer of DOC. Many observations provide the evidence of a good correlation between CDOM and DOM/DOC loading across the different subcatchments, despite the absence of this co-variation in a few cases. CDOM absorbs primarily ultraviolet and blue light, and is fluorescent (350nm – 440nm). Together with the other two ocean-color components, chlorophyll and non-algal particles (NAP), CDOM has a significant contribution to the signals received by satellites and therefore is detectable by remote sensing.

(2) Research objectives

Our study focused on two aspects: 1) rapidly quantifying CDOM via remote sensing-based inversion and 2) watershed-based modeling to understand CDOM sources and degradation.

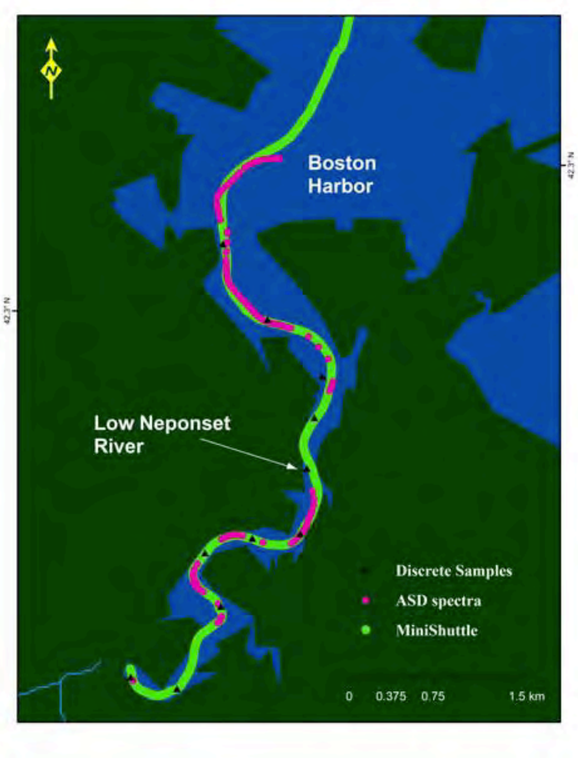
Remote sensing-based inversion is to observe CDOM concentrations in freshwater and coastal regions from in situ spectral data measured by ASD FieldSpec® 3 and satellite hyperspectral images EO-1 Hyperion. Hyperion sensor bears relatively high resolutions both in spectral (10nm) and spatial (30m), and hence provides a good platform for optical inversion of CDOM. Specifically, we aimed to improve and calibrate our algorithm, QAA-E (Extended Quasi Analytical Algorithm), to inverse the ag440 (the absorption coefficients of CDOM at 440nm, typically denoting CDOM concentrations in ocean-color science) from satellite images. Our previous study on algorithm development and test shows QAA-E is able to retrieve ag440 with acceptable accuracy in Mississippi River and Atchafalaya River plumes. This objective is the basis for further work of modeling.

Watershed-based modeling is to better understand the DOC sources and its relationship and coupling to a number of environmental factors associated with the watershed (vegetation coverage and density, topography, soil type, land cover and land use, etc.), hydrology (precipitation, flow, runoff, etc.), water quality (salinity, dissolved oxygen, etc.) and aquatic optical components (chlorophyll and sediments). Especially, the Vegetation-CO_2-DOC-CDOM chain relationship may give us a whole picture of DOC dynamic, transportation and cycle on a regional watershed ecosystem, as well as the possible reason and impact of anthropogenic activities on water quality. The SWAT (Soil and Water Assessment Tool) model will serve as a good tool to model the interactions between watershed features, environmental factors, and land use practice. This objective is challenging since the related systems are fairly complex. However, our goal is to build a model capturing the major factors controlling CDOM flux from Neponset River to Boston Harbor.

Methodology:

<u>(1) In situ measurement:</u>

Our field data collection was conducted on Sept. 25 and Nov. 04, 2009. The in situ CDOM concentration and spectral data were measured on the vessel R/V Neritic, cooperating with researchers from UMass Boston, over the low Neponset River and Boston Harbor (Fig. 1). The data acquisition activities included (1) continuous above-surface hyperspectral measurements, (2) continuous underwater measurements of the IOPs (attenuation and absorption coefficients), salinity, density, dissolved oxygen, UV radiance, CDOM fluorescence, chlorophyll fluorescence, and optical backscattering for suspended sediments, and (3) discrete water sampling and analysis in laboratory.



The water above-surface remote-sensing reflectance was measured by a portable spectroradiometer (ASD FieldSpec®), with a full spectral range (350 – 2500 nm).

The spectral sampling interval of output is 1 nm. In the entire cruise, we collected approximately 1,500 hyperspectral samples. The underwater measurements were carried out by the MiniShuttle, a towed, undulating vehicle based on the Nu-Shuttle manufactured by Chelsea Instruments. It is a synthetic function instrument with multiple devices, including a SeaTech fluorometer measuring the fluorescent dissolved organic matter. The resolution of our underwater measurements is very high and we consequently created a large dataset containing about 45,000 samples. In addition, about 25 discrete water samples were collected and analyzed to calibrate the real time measurements.

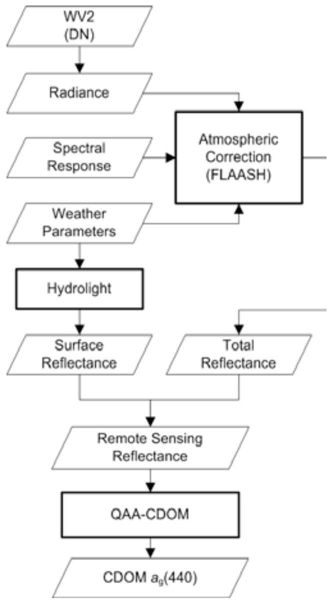
The EO-1 Hyperion images in the same regions were requested during fieldtrip dates. The image acquired on Nov. 04 is cloud free. Hyperion images provide a high spectral resolution 400 – 900 nm with 10 nm interval, and a high spatial resolution 30 m. We also acquired a latest WorldView 2 (WV2) image covering our study site. WorldView-2 is a multispectral sensor which bears 8 bands, including a coastal blue band (400 – 450 nm), and also is with very high spatial resolution (~1.8 m).

(2) Remote sensing inversion algorithm and processing:

Figure 1. Study site map and in situ measurement tracks of MiniShuttle and ASD, and locations of discrete samples in Low Neponset River and Boston Harbor regions.

The whole processing of CDOM remote sensing inversion from WV2 can be referred to in Fig. 2(a). Given a WV2 image, we need to convert the digital number (DN) to the radiance and then make an atmospheric correction, using FLAASH module provided by ENVI®, to obtain the total reflectance, and then compute the remote sensing reflectance, R_{rs}, using HydroLight simulation to remove the water surface reflectance, and finally input these R_{rs}' into remote sensing inversion model to derive CDOM concentration $a_g(440)$ (absorption coefficient at 440 nm). For Hyperion images, some additional preprocessing is needed, including replacing dark lines, destriping and denosing.

To retrieve CDOM's $a_g(440)$, we developed a QAA-CDOM algorithm (Fig. 2(b)). QAA-CDOM is based on Lee's QAA algorithm and its extension, QAA-E. QAA is a quasi-analytical level-by-level algorithm combining a series of empirical, semi-analytical, and analytical algorithms. QAA only requires R_{rs} at several wavelengths (410, 440, 490, 555, and 640 nm) as input data, and at different levels, it outputs r_{rs}, absorption and backscattering coefficients of water's total (a_t, b_{bp}), chlorophyll (a_{ph}, b_{ph}) and CDM (a_{dg}, b_d) (CDOM and NAP together) for all given R_{rs}' wavelengths. QAA has been tested and used in many applications. QAA's output $a_{dg}(440)$, however, has been proven too rough to represent $a_g(440)$ in estuarine and coastal turbid waters, so that QAA's extension, QAA-E, has been developed, in which $a_g(440)$ is exactly derived, using either a_d-based or a_g-based methods. Recently, QAA-E has been further improved to QAA-CDOM in which a QAA's original function and a few parameters has been optimized by integrating synthetic data, very high spatial resolution in situ data from our Mississippi cruise and other field data (NOMAD) collected globally during the last decades. QAA-CDOM has been tested with not only excellent inversion accuracy (<25%), but also suitable for high CDOM variability (0.01 – 13.3 m^{-1}).



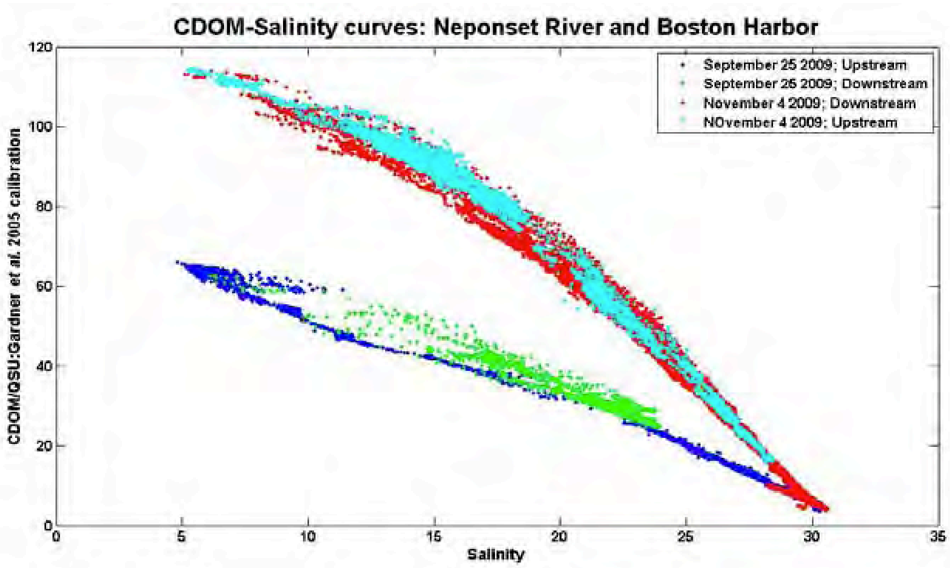
(a) (b)

Figure 2. The simple flow charts of (a) whole processing of WV2 satellite images and (b) QAA-CDOM algorithm.

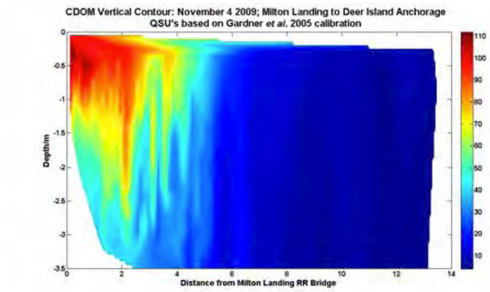
Principal Findings and Significance:

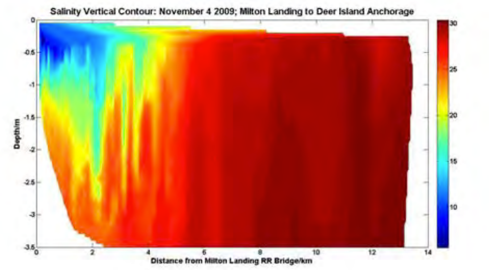
(1) In situ CDOM concentration

According to our measurements on Nov. 4, 2009, CDOM in the Neponset River and Boston Harbor regions ranged from 8.17 to 86.75 QSU, with mean value 34.7 QSU. This wide range shows CDOM in the Neponset River is highly varied complicated. Also, CDOM and salinity demonstrate high negative correlation (Fig. 3). However, their correlation coefficients change at different times (Sept. 25, 2009 vs. Nov. 04, 2009) but are similar at different locations (upstream vs. downstream), indicating that CDOM seasonal or temporal variations are more significant than its spatial variations.



(a)





(b) (c)

Figure 3. CDOM-Salinity relationship in the Neponset River and Boston Harbor regions. (a) CDOM-Salinity curves; (b) CDOM vertical contour; (c) Salinity vertical contour; (Figure courtesy of G. Bernard Gardner).

<u>(2) CDOM high-resolution remote sensing inversion</u>

CDOM distribution in the low Neponset River and Boston Harbor was mapped in very high spatial resolution (~ 1.8 m) from a WV2 image, see Fig. 4. This resultant image and statistical comparisons (Fig. 5) show that QAA-CDOM is able to invert CDOM absorption coefficients with excellent accuracy (RMSE = 0.11, R^2 = 0.73). Our results also indicate that $a_g(440)$ in the Neponset River and Boston Harbor is slightly underestimated, particularly for the fresh water in the upstream. According to our analysis, this underestimation is possibly due to the interference of very high concentrations of phytoplankton (chlorophyll) growing in the same regions.

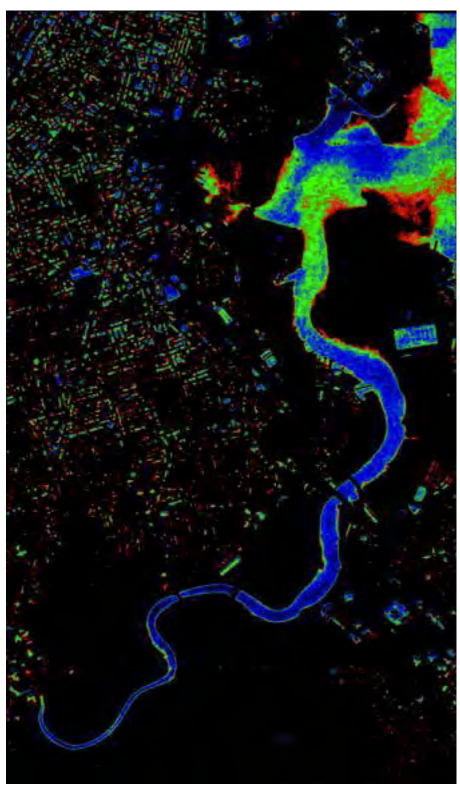


Figure 4. CDOM distribution in the Low Neponset River and Boston Harbor regions, derived from WV2 satellite images.

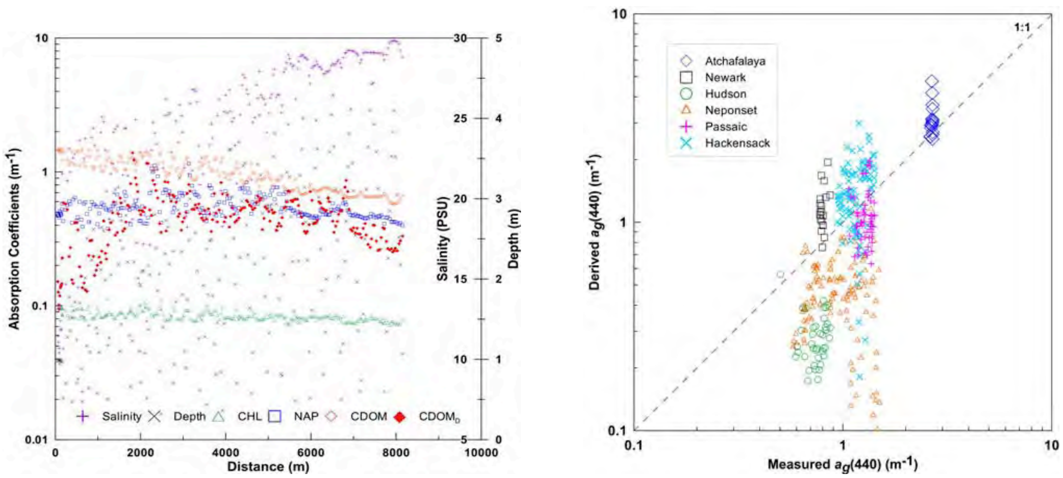


Figure 5. Derived CDOM and properties of other in-water components. (a) Satellite-derived CDOM and in situ measured concentration of CDOM, CHL, and non-algal particles, and water depth and salinity, along our tracks. (b) Comparisons between derived and measured CDOM in six locations, including the Neponset River.

(3) Uncertainty of in situ measurement and remote sensing inversion

The uncertainties related to the whole inversion processing were analyzed in three levels: uncertainties in the processing of in situ measurement (level 1), satellite image preprocessing (level 4) and remote sensing inversion model (level 5), respectively. We found that in level 1, the uncertainty of the in situ CDOM measurement in the Neponset River and Boston Harbor is approximately 1.547, which is about 10 times larger than our other study sites in the Mississippi River plumes and Hudson River estuary. This result indicates that the true CDOM distribution in the Neponset River is complex and highly varied even in a small water volume. The overall average level 1 uncertainty of our three study sites is 0.262%. This normalized uncertainty is reported in volts. If we convert it to quinine sulfate units (QSU) by multiplying by 30, then we get 7.86%. This value, closing to 0.1, implies that in QSU, keeping 1 decimal is significant for shuttle's measurement. If we further convert it to absorption coefficient, we obtain for ag(440) the normalized uncertainty of about 0.32%. Similarly, it indicates that in the unit of absorption coefficient (m^{-1}), the first two decimals are significant. If we exclude the Neponset data, the normalized uncertainties for volts, QSU, and absorption coefficient are 0.122%, 3.66%, and 0.22%, respectively.

The uncertainties of level 4 and level 5 are listed in the Table 1 and Table 2, respectively.

Table 1. Uncertainties of above surface spectrum. Star symbol indicates the results were calculated from strong wave areas. The values in the '*Err*' columns multiple 100 is the error percentage. The *Err*4 and *Err*49 are the mean value of the 4 bands (440, 490, 555, 640) and the 49 bands of Hyperion sensor.

Err_{49}	Err_4	Err (440)	Err (490)	Err (555)	Err (640)	U_{A49}	U_{A4}	U_A (440)	U_A (490)	U_A (555)	U_A (640)
0.128	0.654	1.121	0.655	0.482	0.356	0.009	0.014	0.021	0.013	0.014	0.009

Table 2. Uncertainty of QAA-CDOM remote sensing inversion model.

n	Measured			Derived			U_A	U_{A2}	$Err\%$ mean	$Err\%$ Abs mean	Err_{log} mean	RMSE	$RMSE_{log}$
	Min	Avg	Max	Min	Avg	Max							
1143	0.57	0.98	1.53	0.09	0.47	1.42	0.0179	0.0189	-0.488	0.494	-0.346	0.607	0.426

Publications and Conference Presentations:

Articles in Refereed Scientific Journals

Zhu, W.N., Q. Yu, Y.Q. Tian, R.F. Chen, and B.G. Gardner, 2011, Estimation of chromophoric dissolved organic matter in the Mississippi and Atchafalaya river plume regions using above-surface hyperspectral remote sensing, Journal of Geophysical Research-Oceans, 116, C02011

Yu, Q., Y. Q. Tian, R.F. Chen, A. Liu, G.B. Gardner and W.N. Zhu, 2010, Functional linear analysis for estimating riverine CDOM in coastal environment using in situ hyperspectral data, Photogrammetric Engineering and Remote Sensing, 76(10), 1147–1158. (Early Career Best Paper Award, AAG Remote Sensing Specialty Group)

Dissertations

Weining Zhu, Pending, University of Massachusetts Amherst, Dept. of Geosciences, Inversion And Analysis of Chromophoric Dissolved Organic Matter In Estuarine And Coastal Regions Using Hyperspectral Remote Sensing

Other Publications

Presentation, Weining Zhu and Qian Yu, 2011, Uncertainty analysis of remote sensing of colored dissolved organic matter: evaluations and comparisons for three rivers in North America, AAG Annual Meeting, Seattle, WA, April 12-16 2011.

Presentation, Yu, Q., W.N. Zhu, Y.Q. Tian and R.F. Chen, 2011. High resolution estimation of colored dissolved organic carbon in riverine and plume area, AAG Annual Meeting, Seattle, WA, April 12-16 2011.

Presentation, Tian, Y.Q., Q. Yu, and R.F. Chen, 2011. Sensitivity of terrestrial DOC export to climate change from urban landscape, AAG Annual Meeting, Seattle, WA, April 12-16 2011.

Presentation, Weining Zhu, Qian Yu, 2010. Inversion of chromophoric dissolved organic matter in coastal and estuarine regions using satellite hyperspectral remote sensing, AAG Annual Meeting, Washington, D.C., April 14-18 2010.

Presentation, Yu, Q., W.N. Zhu, R.F. Chen, Y.Q. Tian, and G.B. Gardner, 2010. Examining seasonal variation of CDOM concentration in coastal river using hyperspectral data, AAG Annual Meeting, Washington, DC, April 14-18 2010.

Presentation, Tian Y.Q., R.F. Chen, Q. Yu, K. Cialino, W. Huang, and G.B. Gardner, 2010. Variation of DOC exports from urban watershed to marine water in response to climate change: A case study for the Northeast of the United States, AAG Annual Meeting, Washington, DC, April 14-18 2010.

Student Support:

Weining Zhu, Ph.D. candidate, UMass Amherst Geosciences

Notable Achievements and Awards:

Using this grant as seed fund and preliminary study, we submitted a NSF proposal and it was successfully funded.

Qian Yu (PI), Co-PI: Anna Liu, Collaborated with Yong Tian and Robert Chen at

UMass-Boston, *Collaborative Research: Modeling DOC dynamics from landscapes to coasts: hydrological connectivity and estuary processes*, NSF Collaboration in Mathematical Geosciences (CMG), #1025547, $517,987 ($329,346 on Amherst Campus), Sept 2010 - Aug 2013.

8. Surface Water-Groundwater Interactions on the Deerfield River (USGS-2010MA237B)

Principal Investigator: Dr. David Boutt, UMass Amherst
Start Date: March 1, 2010
End Date: February 28, 2011
Reporting Period: March 1, 2010 – June 30, 2011
Funding Source: USGS (104B)
Descriptors: Surface Water, Groundwater, Methods, Ground-water Flow and Transport

Problem and Research Objectives
Near daily hydroelectric dam releases from Fife Brook Dam in Rowe, MA raise the stage of the river and alter downstream interaction between water in the river and that in the riparian aquifer. Several studies (Sawyer 2009, Arntzen 2006) have shown that these anthropogenic "flood" events in a typically gaining river setting can reverse the hydraulic gradient between the stream and the aquifer, causing the stream to temporarily lose water as the dam release flood-wave passes. Bank storage, the natural analogue of this hydraulic gradient reversal, is an important phenomenon for attenuating hydrograph spikes during natural floods. Past work on anthropogenic stage increases has typically focused on an individual site below a dam, thereby oversimplifying the downstream heterogeneity of stream and aquifer morphologies. In this study, the goal was to investigate how different geologic settings along a 20 km reach of river may influence the magnitude of dam-induced bank storage events. We hypothesized that areas of greater aquifer transmissivity would account for greater exchange of water between the river and adjacent riparian aquifer.

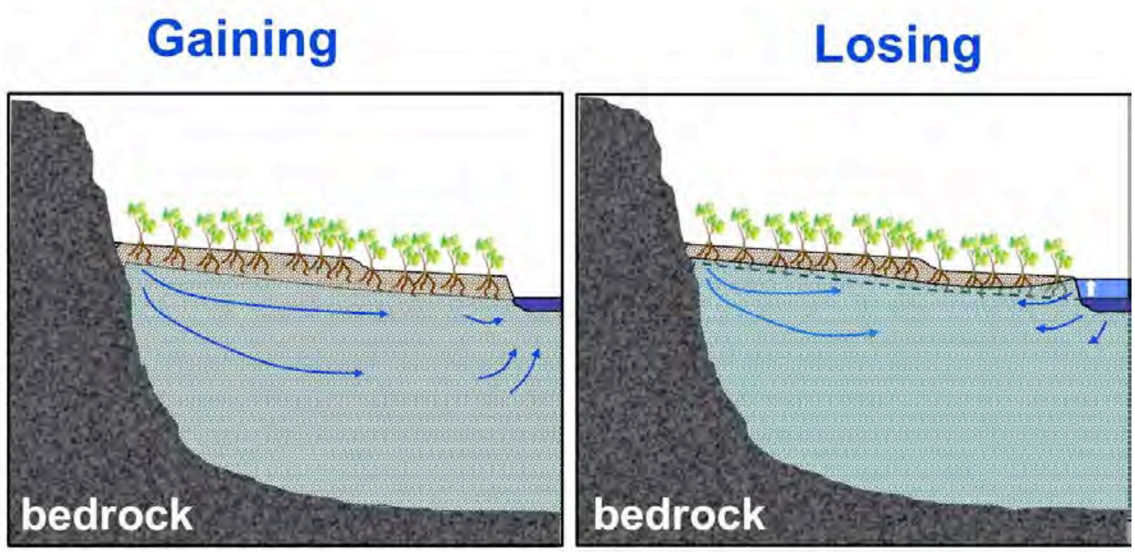


Figure 1 - At low stage (left) the water table slopes towards the river and the river is gaining. The hydraulic gradient is reversed during a dam release, causing the river to lose.

Methodology:
Three distinct strategies were employed to better understand the controls on dam-induced bank storage: 1) Several sites were selected for in-stream monitoring of water fluxes across the river bed. Direct Darcy-based methods with piezometers and pressure transducers provided direct observation of the hydraulic gradient between river water and groundwater. Vertical thermistor arrays collected temperature profiles of the subsurface to enable the use of river heat as a natural tracer, giving us a second means by which to observe fluxes across the riverbed. 2) Two-dimensional finite-element modeling of the riparian aquifer was used to simulate bank storage events to evaluate how different aquifer dimensions mediate surface water-groundwater exchange. 3) A water budget for the 20 km river reach was created by accounting for known inputs to and outputs from the river to evaluate how dam-induced bank storage affects the river at the reach scale (see Figure 2).

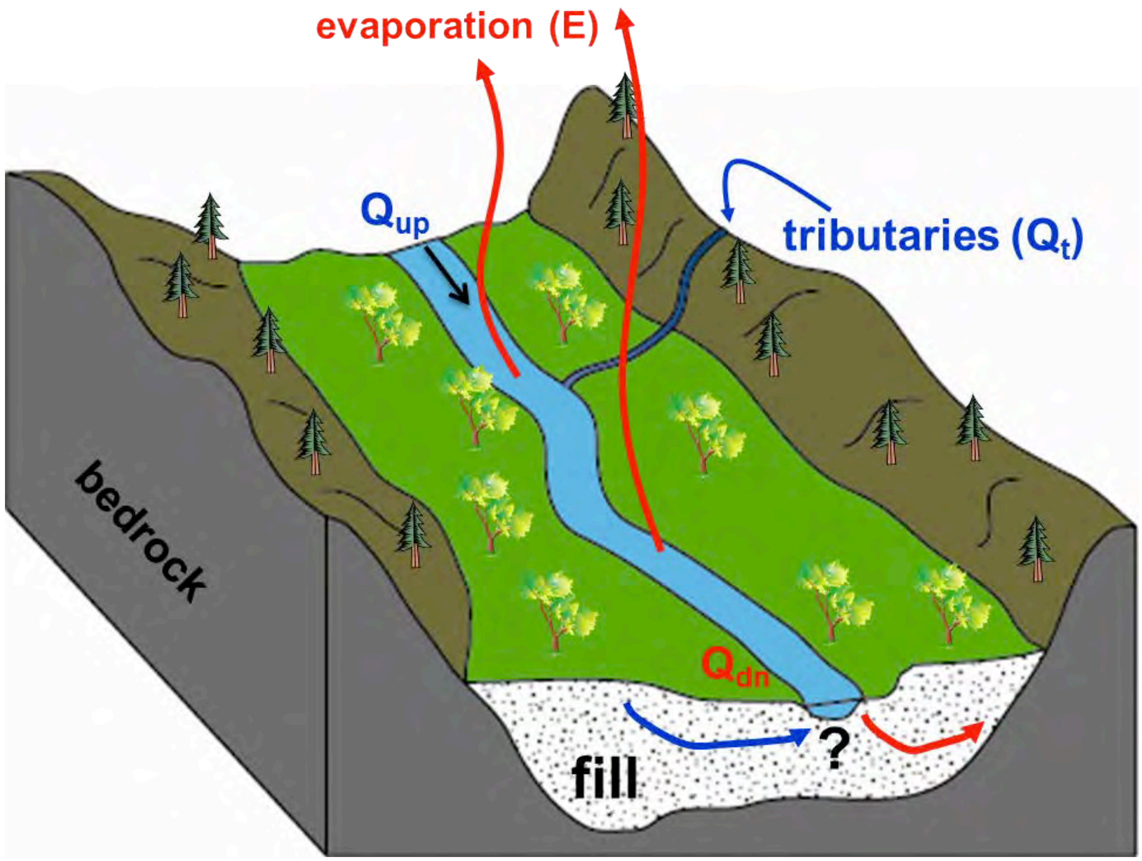


Figure 2 - The water budget for the 20 km river reach accounted for inputs from the upstream dam and tributaries, and outputs from the downstream end of the reach and direct evaporation.

Principal Findings and Significance:
From June through September, the 20 km study reach of the Deerfield River below Fife Brook Dam loses 5-10% of its discharge. Unregulated rivers in this region typically gain during all seasons. Water budget analysis of the entire reach, as well as direct observations from in-stream piezometers and vertical temperature arrays all indicate that the river loses water in a variety of geologic settings.

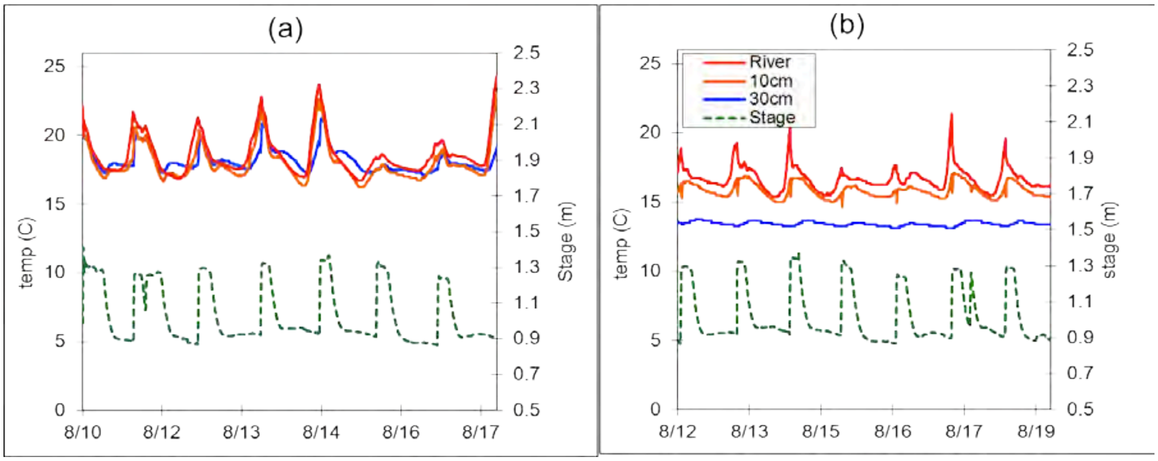


Figure 3 - Vertical temperature profiles at two sites in the Deerfield River study reach. Slug tests showed that the riverbed at site (a) had much higher hydraulic conductivity than that at site (b). The temperature of the subsurface therefore varies a lot at site (a) due to the close hydraulic connection between surface water and groundwater there.

Field data and two-dimensional simulations suggest that the width and conductivity of the valley aquifer have the greatest control on losses. We have identified a strong correlation between water loss during a given event and evaporative forcing, suggesting that water driven into the bank during high stage is made available to riparian vegetation and lost permanently due to transpiration. This finding has two major implications: 1) Hydroelectric operators with multiple facilities on a single river face a tradeoff when generating electricity on days with high evaporative forcing; 2) Minimum flow requirements for in-stream fauna may need to be adjusted considering that the discharge at some distance downstream can be reduced below that which is released from the dam.

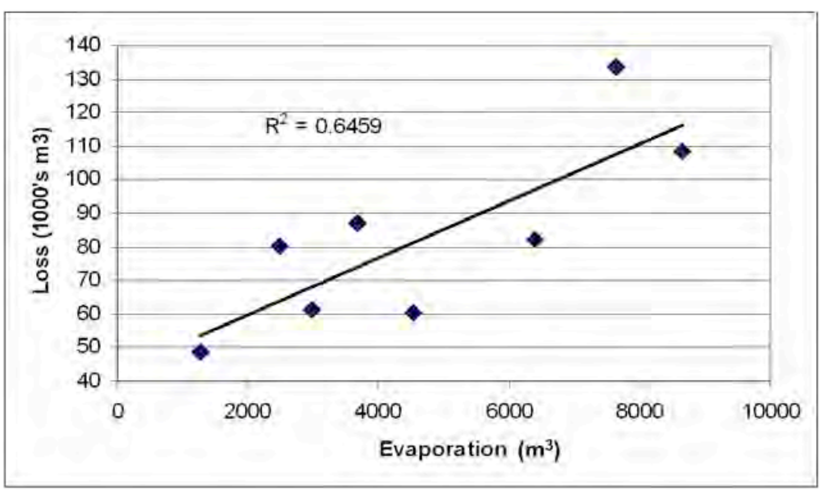


Figure 4 - Total direct evaporation from the river surface during an individual flood event is a good predictor of loss during that same event. Data from a nine-day period in July 2010 are plotted here.

Publications and Conference Presentations:

Yellen, Brian, David Boutt, 2011.,Can hydroelectric dams cause a river system to lose water? Water Resources Conference, Amherst, MA, April 7, 2011.

Yellen, Brian, David Boutt, 2010. A reach-scale study of dam-induced hyporheic exchange: controlling mechanisms and effects, Deerfield River, Massachusetts.

American Geophysical Union Annual Fall Meeting, San Francisco, CA, December 13, 2010.

Student Support
One full-time graduate student, Brian Yellen MS in Geosciences at UMass Amherst.

Citations

Arntzen, E.V. 2006, Effects of fluctuating river flow on ground water/surface water mixing in the hyporheic zone of a regulated, large cobble bed river, Wiley : Chichester, United Kingdom, United Kingdom.

Sawyer, A.H. 2009, Impact of dam operations on hyporheic exchange in the riparian zone of a regulated river, Wiley : New York, NY, United States, United States.

9. Impact of the Hemlock Woolly Adelgid on the Water Cycle in New England: Differences in Hydrologic Fluxes Between Hemlock and Deciduous Forest Stands (USGS- 2010MA241B)

Principal Investigator: Dr. Andrew J Guswa, Smith College, Northampton
Start Date: May 1, 2010
End Date: December 31, 2011
Reporting Period: March 1, 2010 – June 30, 2011
Funding Source: USGS (104B)
Descriptors: Hydrology, Invasive Species, Ecology, Climate and Hydrologic Processes

Problem and Research Objectives
Problem and Research Objectives:
Invasive pests, especially in conjunction with climate change, have the potential to transform the species composition of many forests. In the northeastern United States, the hemlock woolly adelgid (HWA) poses a significant threat to eastern hemlock (Tsuga canadensis). This pest arrived in western Massachusetts over the last ten to fifteen years, is steadily making its way into southern Vermont and New Hampshire, and often kills infested hemlocks within a few years (National Forest Service, 2009).

Replacement of hemlock forests by other species, such as birch, maple, and oak, may alter the hydrologic cycle and impact water resources. Changes to hydrologic fluxes include both the input of water, which is affected by canopy interception, and the uptake of water for transpiration. Canopy interception affects not only the mean input of water but also its distribution in space, which has implications for hydrologic and geochemical processes that are non-linear functions of soil moisture, such as drainage below the root zone and nitrification/denitrification. To better understand the impact of HWA invasion on the hydrologic cycle, we must understand how hemlock and deciduous forest stands differ with respect to canopy interception and water uptake.

This project aimed to build upon and complement these early findings to better understand differences in hydrologic fluxes between hemlock and deciduous forests. The objectives of this research project were to:

- Quantify the difference in average interception between hemlock and deciduous stands
- Quantify the spatial variability of throughfall in hemlock and deciduous stands

- Quantify differences in summertime water use for hemlock and deciduous stands.

Methodology:
During the summer of 2010, a cohort of three undergraduates engaged in a two-month field campaign to measure and characterize hydrologic fluxes in hemlock and deciduous forest stands. The fieldwork was carried out at the Ada and Archibald MacLeish Field Station, a 240-acre site maintained by Smith College and located adjacent to the primary reservoir that supplies drinking water to the City of Northampton. Ongoing monitoring at this site includes continuous measurements of precipitation, air temperature and pressure, relative humidity, solar radiation, and wind speed and direction.

We established two hemlock sites and two deciduous sites and instrumented them to measure throughfall. Additionally, we characterized the trees and vegetation in each plot (e.g., stem location, species). Our hemlock sites are situated within permanent vegetation plots that were established on the property in 2009. Within these 20 m x 50 m plots, all trees and saplings > 1.4 m have been tagged and measured for diameter at breast height (DBH); individual trees are being tracked for growth and survival in coming years. Tree coring is being used to reconstruct stand histories and to assess hemlock population dynamics over time.

At both the hemlock and deciduous sites, we deployed thirty stationary throughfall collectors over a ten-meter by ten-meter plot. In two sites (one hemlock and one deciduous), these collectors were arranged in a regular grid with 1.5-meter spacing; in the two other sites, the collectors were placed using a stratified random design. Throughfall volumes were measured following each precipitation event.

To assess water use, the students developed instrumentation to measure sapflux using the heat-pulse method. These sensors use the transport of a heat pulse to infer water velocity in the xylem. Due to unforeseen challenges in the development and use of these sensors, we got only as far as field-testing the sensors in a single tree, and we obtained no useable sapflux data during the summer of 2010. Three undergraduates will continue this work during the summer of 2011.

Principal Findings and Significance:
The measurement and analysis of throughfall during the summer of 2010 complements a prior throughfall study carried out in 2009. From 3 June through 25 July 2009, fourteen precipitation events generated 311 mm of rain; 2010 was much drier with eight rain events generating 148 mm of precipitation over the campaign.

In 2009, stand-average throughfall amounted to 276 mm (89% of precipitation) in the deciduous plot and 242 mm (78%) in the west hemlock stand; in 2010, stand-average throughfall totals were 129 mm (87%), 123 mm (83%), and 126 mm (85%) in the deciduous, west hemlock, and north hemlock stands, respectively. On an event-by-event basis, the throughfall fraction increases with precipitation amount, and representing interception as a threshold depth, D, provides a good fit. These threshold depths are lower (2.4 mm and 2.5 mm) for the deciduous stand than for the hemlock stands (5.0 mm, 3.5 mm, and 3.2 mm).

With the exception of very light events (i.e., less than 5 mm of precipitation), the spatial variability among collectors, as measured by the coefficient of variation or the ratio of the interquartile range to the median, is insensitive to precipitation amount. Additionally, wet and dry spots tend to persist over the season and even from year to year, and analysis of variance confirms that collector position is highly significant as a predictor of normalized throughfall amount. While persistent through time, the spatial patterns exhibited no discernable spatial correlation structure. Moment-based statistics of spatial variability are strongly influenced by extremes, and the deciduous stands show positive skewness among collector totals (i.e., some very wet spots), while the hemlock stands exhibit negative skewness (i.e., dry spots).

Publications and Conference Presentations:

Articles in Refereed Scientific Journals

Guswa, Andrew J. and C. M. Spence, 2011, in review. Effect of vegetation change on throughfall patterns and recharge: Application to hemlock and deciduous forests in western Massachusetts, submitted to Ecohydrology.

Dissertations

Spence, C. M., 2011. Biotic and abiotic factors affecting throughfall volume and spatial variability in a New England forest, undergraduate Honors Theses, Smith College.

Conference Proceedings

Spence, C. and A. J. Guswa, 2011. Biotic and abiotic factors affecting throughfall volume and spatial variability in a New England forest, MA Water Resources Research Center, 8th Annual Conference, University of Massachusetts, Amherst, MA, 7 April 2011.

Guswa, Andrew J., M. Mussehl '12, A. Pecht '12, and C. Spence '11, 2010. Spatial pulses of water inputs in deciduous and hemlock forest stands, Eos Trans. AGU, Fall Meeting Suppl., Abstract B24B-08. what's the date on this???

Student Support

Three students supported for summer research during the summer of 2010:
B.S. engineering, 2011
B.S. engineering, 2012
B.A. engineering and geosciences, 2012

Notable Achievements and Awards

Katelyn Gerecht , Undergraduate Student at Smith College received first prize for her poster on this project at the seventh Annual Water Resources Conference in Amherst April 8, 2010.

10. An Assessment Methodology for Differential Impact on Environmental Justice Populations of Releases of Industrial Toxics to Water in Massachusetts (USGS- 2010MA248B)

Principal Investigator: Michael Ash, UMass Amherst
Start Date: May 1, 2010
End Date: December 31, 2011
Reporting Period: March 1, 2010 – June 30, 2011
Funding Source: USGS (104B)
Descriptors: environmental justice, water toxics, drinking water

Problem and Research Objectives

Release of toxic chemicals into surface water by industrial facilities in Massachusetts puts the water of communities at risk and potentially impacts human health. Previous research on environmental justice (EJ) has found that low income and minority populations are disproportionately impacted by industrial air pollution and by the siting of hazardous waste facilities, but no methodology exists for assessing the differential impact of surface-water releases of industrial toxics on EJ populations. We examined social and economic characteristics of communities in proximity to facilities releasing toxics to surface water in Massachusetts. The methods and metrics developed for assessing community exposure and environmental justice in industrial water pollution will be applicable to the entire United States.

Methodology:

U.S. EPA's Risk Screening Environmental Indicators (RSEI) generates annual toxicity-weighted hazard measures for every industrial facility based on the mass and toxicity of releases into different media (air, water, ground) reported to U.S. EPA's Toxics Release Inventory (TRI). This study focuses on releases to surface water. We geo-code U.S. Census data to determine the proximity of each census tract to facilities identified as high hazard based on the surface-water releases. The data permit examination of both how sources of toxic releases differ in their disproportionate impacts and also how communities differ in their cumulative exposure to toxic releases. We generate facility-based estimates of toxic water pollution weighted by relative toxicity and examine the demographics of communities in proximity to each facility. By comparing the demographics of nearby communities to the demographics of the entire Commonwealth, we measure disparities in exposure to water toxic releases. Additionally, we model the probability that a census tract is near a toxic releasing facility based on its demographic characteristics.

This project draws on the Toxic 100 Water Polluters, an index developed by the Political Economy Research Institute at the University of Massachusetts Amherst in partnership with Food & Water Watch, a non-profit consumer organization.

U.S. EPA has recently revised the portion of the RSEI methodology that models the fate and transport of toxics released into surface water. The RSEI water geographic micro data, which reports the estimated concentration of every toxic release to surface water in every downstream flowline, has not previously been used in research. These data, which became available in September 2010, permit more specific measurement of community exposure to industrial toxic releases to surface water than previous methods based on simple proximity to releasing facilities.

Results Expected

The primary objective of this research is to develop measurements of the disproportionate impacts on EJ communities of toxics released into nearby surface water. We developed measurements based both on community proximity to toxic releasing facilities and of exposure to drinking water supplies. The results will provide a baseline assessment which will allow us to compare releases and impacts annually in order to determine if policies, enforcement efforts and community action are effective in reducing the impact of toxic releases on exposed populations.

Principal Findings and Significance:

The project successfully developed a methodology for assessing disproportionate impacts on EJ communities of toxics released into nearby surface water and applied the methodology to Massachusetts. First we present some results and then discuss the methodology. These results have not yet been peer reviewed and should be considered provisional.

There are roughly 88,500 Census Blocks in Massachusetts. Census Blocks correspond to city blocks in urban areas and can be somewhat larger in rural areas; the average Census Block in Massachusetts has 72 residents, but there is variation. Using our method to classify Census Blocks by proximity to waterways that were highly polluted by industrial releases in 2007, we identified 204 "extremely polluted Census Blocks" associated with the top five industrially polluted flowlines; 146 of these Census Blocks are in Suffolk County, 45 are in Middlesex County, and 13 are in Franklin County. We then analyzed the racial and ethnic composition and average household income in the extremely polluted Census Blocks and the rest of Massachusetts. The Census Blocks with extremely high impact from industrial toxic water releases are substantially less (non-Hispanic) white (56 percent versus 84 percent) and more Hispanic (35 percent versus 6 percent) than is the rest of the state. The percentage black is similar in highly impacted and other Census Blocks. The highly impacted Census Blocks are also very much poorer: the median household income is almost $20,000 less in the most impacted Census Blocks.

	All MA blocks (N= 88,307)		Extremely polluted blocks (N= 204)	
	Mean	Std. Dev.	Mean	Std. Dev.
percent white	83.9%		56.0%	
percent black	4.6%		2.6%	
percent hispanic	5.6%		35.4%	
median income	$55,090	$24,200	$38,450	$16,660

The table focuses on the most impacted Blocks and implies that there is significant disparity in water pollution exposure by race, ethnicity, and class.

Regression analysis can test whether the EJ relationship appears pervasively in the state and whether the race/ethnic and income relationships appear independently from each other. Specifically, we modeled how the racial, ethnic, and income composition of a Census Block is associated the likelihood of a Census Block appearing among the most impacted 10 percent, 5 percent, or 1 percent of Massachusetts Blocks that abut a waterway. We also examined the effect of socio-economic composition on the likelihood of any exposure and on a continuous measure of exposure, as well as other specifications. We present illustrative results here.

	top 10%	top 5%	top 1%
Percent Black	-0.257 ***	-0.205 ***	-0.0590 ***
	(0.0245)	(0.0151)	(0.00883)
Percent Hispanic	0.265 ***	0.319 ***	0.153 ***
	(0.0319)	(0.0291)	(0.0208)
Percent Asian	0.0756 *	0.0840 **	-0.0318 ***
	(0.0453)	(0.0365)	(0.0103)
Percent Native American	-0.462 ***	-0.291 **	-0.0825
	(0.160)	(0.116)	(0.0679)
Median income ($10K)	-0.0296 ***	-0.000134	0.000739
	(0.00408)	(0.00277)	(0.00158)
Median income squared	0.00111 ***	-0.000388 **	-0.000108
	(0.000241)	(0.000153)	(8.81e-05)
Constant	0.241 ***	0.0687 ***	0.0113 *
	(0.0155)	(0.0109)	(0.00650)
Observations	58,108	58,108	58,108
R-squared	0.027	0.029	0.020

These results indicate that compared to non-Hispanic whites, African Americans are substantially less likely to live in the most polluted Census Blocks near water. Hispanics are much more likely to live in the most polluted Census Blocks that abut water. A Block that is 100 percent Hispanic rather than 0 percent Hispanic is more than 30 percentage points more likely to appear in the top 5 percent of Census Blocks and 15 percentage points more likely to appear in the top 1 percent. (A randomly selected Block is by definition 5 percent likely to appear in the top 5 percent of Census Blocks; so the Hispanic effect is quite strong.) A high percent Asian increases the probability that a Block is among the top 10 percent or top 5 percent but not in the most polluted 1 percent. Because of the relatively small population in Massachusetts, the Native American results are estimated with limited precision but nonetheless indicate that percent Native American is associated with lower likelihood of high pollution exposure.

The income results imply a significant protective effect of income with respect to the likelihood of a Block being in the most polluted 10 percent of Blocks abutting water. Furthermore, it should be noted that the race and ethnicity results control for income. So the effect of high percentage Hispanic is not explained by income differences between Hispanics and non-Hispanics.

The results described above required the development of new methods for matching polluted stream reaches to Census geography and operationalization of the concepts exposure and differential exposure for industrial toxic releases to water. The key dataset is the U.S. EPA's Risk Screening Environmental Indicators, Geographic Microdata, for Water (RSEI-GMW) for 2007. This dataset provides the estimated concentration in every downstream flowline of every 2007 TRI release to surface water. The RSEI model also provides oral toxicity weights for each of the approximately 600 TRI chemicals that make it possible to add toxicity-weighted concentrations of different chemicals. Details of the RSEI model are available from U.S. EPA http://www.epa.gov/oppt/rsei/pubs/index.html and there is a summary of the RSEI model on the Corporate Toxics Information Project website at http://www.peri.umass.edu/ctip.

Because the aim is to provide inter-area assessment for very large areas (the Commonwealth of Massachusetts in this project, but the methods are scalable), GIS is not feasible for matching polluted reaches to Census geography. There are approximately 8 million Census Blocks in the United States, 3 million flowlines in the NHDPlus (EPA's augmented version of the National Hydrography Dataset), 187,000 impacted flowlines and 123 million distinct release-flowline impacts, or estimated concentrations in flowlines from industrial toxic releases, in RSEI-GMW. Although GIS data for the components exist, spatial joins are impractical or impossible on this scale. Instead, we extracted longitude and latitude data for every point on every flowline from the NHDPlus, converted each point to a corresponding 1-kilometer cell in the RSEI geography, and employed the existing crosswalk between RSEI geography and the Census. In this way, we were able to assign pollution concentration estimates, by specific source industrial facility and chemical, to each U.S. Census Block. (Blocks that do not abut water have zero concentration as do Blocks that abut water but are not affected by any upstream industrial releases.)

The extraction and matching methodology was carried out with perl and MySQL, and scripts are available on request. The scripts, in particular those that extract and process point data from the NHDPlus shapefiles, may prove useful for a variety of applications. Other scripts are more specifically focused on RSEI applications, and these too are available for researchers.

Student Support

Four (4) students received support as research assistants from matching funds associated with this project:
Grace Chang, Ph.D. student in Economics, UMass Amherst
Robin Kemkes, Ph.D. student in Economics, UMass Amherst
Helen Scharber, Ph.D. student in Economics, UMass Amherst
Owen Thompson, Ph.D. student in Economics, UMass Amherst

Notable Achievements and Awards

- The Corporate Toxics Information Project received the second year of a two-year grant from Food and Water Watch, a Washington, D.C.-based non-profit organization to further develop methodology for the assessment of population risk and environmental-justice disparities from toxic industrial water pollution reported in the U.S. EPA's Toxics Release Inventory and the Risk Screening Environmental Indicators model.
- James K. Boyce (PI) and Michael Ash (co-PI) submitted a successful proposal to the National Science Foundation in collaboration with researchers at the University of Southern California and the University of Michigan to continue research on environmental justice with the Risk Screening Environmental Indicators geographic microdata.
- Michael Ash (PI) submitted a proposal and received an allocation on TeraGrid, National Science Foundation's effort to build and deploy the world's largest distributed infrastructure for open scientific research, to use TeraGrid resources to manage the databases for the project.

11. Developing a physically-based and policy-relevant river classification scheme for sustainable water and ecosystem management decisions. (USGS-2010MA253B)

Principal Investigator: Ellen Marie Douglas, Bob Bowen UMass Boston
Start Date: March 1, 2010
End Date: February 28, 2011
Reporting Period: March 1, 2010 – June 30, 2011
Funding Source: USGS (104B)
Descriptors: dam removal, river restoration, ecosystem management

Problem and Research Objectives

One of the biggest human impacts on rivers has resulted from the building of dams. The number of dams worldwide has been estimated at 40,000 large (>15 meters in height) and more than 800,000 smaller ones (Petts, 1984; McCully, 1996). Dam operations have caused ecological changes in riparian ecosystems at all scales; how to balance the needs of aquatic and riparian ecosystems and humans remains one of the most important questions of our time (Nilsson and Berggren, 2000). There are 2,964 dams in the Massachusetts dam inventory database. Over half of these dams are privately owned and nearly a third is municipally owned. Most of the dams in Massachusetts are low head, "run-of-the-river" dams that no longer serve the purpose for which they were built. The presence of these dams has fragmented aquatic and riparian ecosystems, impeded fish passage and generally impacted the natural ecological and hydrological functioning of the streams in which they reside. Dam removal should be considered when a dam no longer serves its function. Facilitating dam removal is a major focus of the Massachusetts Division of Ecological Restoration (DER; see http://www.mass.gov/dfwele/river/programs/priorityprojects/projectlist.htm). The removal of a dam incurs many environmental benefits (i.e., enhanced fish passage, restoration of aquatic and riparian habitats, return to a more natural flow regime), but sometimes it can incur environmental costs as well (i.e., release of contaminants that were sequestered behind the dam). In many cases, dam removal is less costly than dam maintenance or upgrade, hence dam removal decisions tend to be based on purely monetary considerations, and the environmental costs or benefits associated with the dam are not fully considered. Furthermore, dam removal projects can be delayed or completely derailed by the perception that doing so will result in the loss of aesthetic, recreational and property values associated with the impoundment behind the dam. While dam removal is a high priority in Massachusetts as well as across New England, the true cost of these efforts, which include direct (economic), indirect (environmental) and cultural (recreation, aesthetic) costs, are not well understood and hence are usually not well quantified in dam removal decisions.

The main challenge of water resource management is to find a balance between the use of resources as a basis for human livelihood and the protection and conservation of the resource to sustain its ecosystem functions and benefits. In assessing ecological condition, whether it be in terms of ecosystem health, service value or integrity, it is necessary to consider economic, social and political influences, since these are often the driving factors in environmental protection and restoration decisions. One of the ultimate goals of this proposed research is to provide a framework for estimating the total ecosystem service value that any particular riverine habitat has to the humans and ecosystems that benefit from it. For this one year project, we focused on the dam removal. We developed a decision support framework that is both physically-based and policy-relevant, and useful to environmental managers and policy-makers in dam removal decisions. The specific objectives of this research were to:

- estimate the economic costs and benefits of dam removal, including estimates of value of ecosystems that are impacted.
- develop a comprehensive framework that incorporates both physical (construction-related) and non-physical (market-related and environmental) costs and benefits
- test this framework on one or more upcoming dam removal projects in Massachusetts that are being facilitated by the DER.

Methodology

This one year project was comprised of three tasks: 1) a survey to assess the current status of the dam removal decision process; 2) development of a conceptual framework for dam removal decisions that accounts for environmental costs and benefits and 3) applying this framework to a case study, namely the removal of two dams on the Ipswich River in northeastern Massachusetts.

<u>Task 1: Stakeholder Survey.</u> Before we could develop an improved dam removal decision-making framework, we first needed to assess how well the current decision process does or does not work and what stakeholders believe are the most important aspects of this process. We developed set of nine questions that were sent via SurveyMonkey (www.surveymonkey.com), a free on-line survey tool, to eleven stakeholders. The stakeholders, including dam owners, town managers, consultants, state and federal agencies and watershed associations, were chosen because they were involved in some aspect of the dam removal process in Massachusetts. Table 1 presents the survey questions and the range of response choices.

<u>Task 2: Conceptual Framework an ideal dam removal process.</u> The results of the stakeholder survey was summarized and analyzed. A conceptual framework for an ideal dam removal process and a comprehensive accounting of the costs and benefits of dam removal was developed. This was done using the information gained from the survey and by adapting a framework for preliminary economic assessment of dam removal on the Klamath River in California, presented by Kruse & Scholz (2006; with the authors approval). In addition, existing feasibility studies for dam removals in Massachusetts and Maine were perused for environmental benefits, which are typically qualitatively considered, but not quantitatively assessed.

<u>Task 3: Cost-Benefit Analysis for dam removal on the Ipswich River.</u> A cost-benefit analysis of removing two dams located within Ipswich River Watershed was performed. In this analysis, environmental and social costs and benefits are included, but are not yet fully quantified. The benefit transfer method was used, which takes an estimated value from another study and adopts it as the transfer value for the new analysis. Benefit Transfers is a less expensive and less time-consuming method. However, there will always be some error in the transfer application. Benefit transfer's limitations are related to a lack of accessible, unbiased information, difference in methods, and differences in geographic and socioeconomic conditions.

Table 1. Survey Questions

1.You represent:

Private sector

Public sector (choose one below)

- State agency

- Federal agency

Please indicate your affiliation:

2. Do you think that dam removal is a necessary process? Why is that?

3. In your opinion, the decision of removing a dam is based on:

Funding

Improving fish habitat

River restoration

Liability

Others (please, specify)

1-Very significant; 2-Significant; 3- Somewhat significant; 4- Not significant

4. In your opinion, what is the most important outcome of the dam removal?

5. If you were the only decision maker, how would you decide which dam to be removed first?

6. Do you think that the existing legislation regarding dam removal makes this process an efficient one? If this is not the case, what other regulations would you add or/and would you change if you had the power to do so?

7. Why do you think the dam removal process has grown so much over the last decade?

8. Who do you think should decide when and what dam to be removed? Why is that?

9. Would you consider being part of a future survey?

Principal Findings and Significance

Survey Results. Only seven of the eleven stakeholders have responded to date. The majority of the responders (six out of seven) ranked funding as a “very significant” factor in the decision making process. Improving fish habitat is the second most important factor, and it is also listed as the most important outcome of the dam removal by five out seven respondents. Three of the responders ranked all of the factors as very significant factors in the decision making process. Two responded that liability and river restoration are “significant” or “somewhat significant”. Only one respondent ranked river restoration as “not-significant”. While funding appears to be the most significant factor in the decision making process, river restoration and improving the fish habitat are the main outcomes of the removal. The biggest difference in opinion is reflected in the fact that all the public agencies agreed that dam removal is a necessary process, while the private sector does not always find it necessary. The lack of legislation is not seen as a challenge directly, it is more reflected in the permitting process and the lack of funding. All the responders agreed that the removals should be prioritized based on the highest potential hazard and the highest environmental outcomes (fish habitat and river restoration). One responder specified, however, that currently the removal is prioritized among the dams that already have been chosen for removal by the owners. All the answers indicated that the final decision should belong to the owners or to whoever pays for the removal.

Three of the responders added that the community should be part of the decision making process too. To summarize, the general perception of the survey appears to be that dam removal is a good alternative for improving the fish habitat and river restoration. However, the decision process itself as well as which dams to remove are topics that require improvement.

Conceptual Framework. Based on the survey results summarized above, an improved decision making process is presented as a five-step conceptual framework. This conceptual framework is a statement of an ideal dam removal process, which would create ways in which the dam removal could become a more controlled, efficient and transparent process. The steps of this conceptual framework are as follows:

1. **Institute a dam removal regulation:** This would include a general rule in which a high hazard dam or dams in disrepair would not be allowed to remain in place. Owners who could not afford to remove the dam could transfer the property rights to the state. The state, through a well-trained entity, would become the only decision-maker. This way the owner transfers the liability, and does not have to pay for the removal. Furthermore, this entity could now prioritize the removals throughout risk assessment and/or comprehensive cost-benefit analysis.
2. **Create a streamlined mechanism for ownership transfer:** a straight-forward mechanism for the transfer of ownership of a dam to the state.
3. **Establish a single agency responsible for dam removal with responsibility for reviewing and permitting projects.** Currently the permitting process alone can take more than a year, because there are numerous agencies involved. Some of these agencies (local, state and federal) have overlapping authority and reviewers assess essentially the same factors, making for a time consuming and confusing process.
4. **Prioritize the dam removal based on potential hazard and/or the potential environmental and social benefits.** The dam removal decision should be based on studies which indicate what dam has the highest hazard in case of failure and/or the greatest potential environmental and social benefits. For instance, if the main goal of the removal is to restore fish habitat, it would be efficient to start with the most-downstream dam and proceed sequentially in an upstream direction.
5. **Comprehensive valuation of all the costs and benefits, including the environmental and social externalities as part of the dam removal decision.** Neglecting to include externalities and common goods in an economic analysis can result in a failure to efficiently allocate all the resources related to dam removal. Externalities arise when the impact of an economic activity on outsiders is not considered. For example, in the case of a dam removal, the sediments from behind the dam released by the removal could settle downstream. The long term impact that the sediment deposits could have on downstream areas is rarely included in the cost-benefit analysis of the dam removal. These costs and benefits are listed in Table 2.

Table 2: Conceptual framework for a complete economic assessment of dam removal (adapted from Kruse & Scholz, 2006 with permission)

Costs	Benefits
Immediate costs: Final design; Sediment disposal Staging of materials; Constructions on site; Feasibility Study; Disposal of waste material; Permits;	Market goods: These are not applicable as they would be transferred from another location; hence including them would result in double counting.
Loss of Direct Services: Hydro-power Recreational value (use of the lake for fishing, boating) Flood control Irrigation abilities Water supply	Non-market goods / Environmental Benefits: Improving fish habitat; Fish passage Returning the river to the free-flow conditions; Recreational opportunities (fishing, boating, hiking, etc) Environmental aesthetics; Reduce flooding upstream of the dam; Cultural values (if this is the case)
External (indirect) impacts: - wetland change; - change in wildlife from the lake; - loss of "lake view"; - temporary sediment transport Infrastructure Cultural	

Case Study: Cost-Benefit Analysis. The conceptual cost-benefit analysis was applied to the removal of two dams located within Ipswich River Watershed: the South Middleton Dam and the Ipswich Mills Dam. The South Middleton Dam is currently under consideration for removal, but it is not necessarily the most viable decision because it is upstream of the Ipswich Mills Dam. One of the biggest challenges in this analysis is the lack of data. Dam removal in a new process in Massachusetts; hence the availability of records pre or post removal is minimal. Another problem is the lack of literature. Although the studies concerning dam removal are increasing dramatically, studies including environmental valuations with respect to the dam removal are sparse. In this analysis we focus on identifying all the environmental benefits associated with the removal of the Ipswich Mills and South Middleton Dams. Not all of these benefits have been quantified as yet, but they must be included in the cost-benefit analysis and will be quantified in the future. These benefits include: improving the fish habitats, returning the river to the natural flow, aesthetics, recreation (boating, freshwater fishing, walking), reducing the upstream flooding, liability, and cultural values. Using benefits transfer method, the benefits of returning to a free-flow river were quantified. The willingness to pay method (WTP) performed by Sanders, Walsh, and Loomis (1990) was used to quantify the benefits from returning the river to a natural flow regime in Colorado. This entailed a mail survey of WTP to preserve free-flowing rivers sent to Colorado households statewide. The mail survey had a 51% response rate of deliverable surveys. The annual WTP per household for option, existence and bequest value was $77 in 1983 dollars. In order to obtain the value per mile, the total amount was divided by the length of river being valued (555 miles). This resulted in a WTP of $0.21 per mile in 1983 dollars. If we account for inflation by using the ratio median incomes, a value of $77 in 1983 would be equivalent $163.70 in 2009 in Colorado (this is the most recent year that had median income information) or $0.29 per mile. Transferring this value to Massachusetts using the ratio of median incomes results in 0.305 per mile in 2009. Removal of the South Middleton and Ipswich Mill Dams would open 56 miles and 17 miles of river, respectively. In Middleton there are 2305 households and in Ipswich there are 5290 households. Multiplying $0.305 per mile by the potential free flowing river length then by the number of households in each community results in a benefit of $39,369 for removing the South Middleton Dam and 27,429 for removing the Ipswich Mill dam.

Dam deconstruction and removal costs were obtained from a report entitled "A Preliminary Site Reconnaissance and Cost Estimates for Ipswich River, Ox Pasture Brook, and Skug River Dams" by Woodlot Alternatives, Inc. determined a total value of deconstructing and removing the dams. In addition to these costs, the cost of reconsolidation of the two bridges located downstream of Ipswich Mill dam need to be added because there is a concern that the removal of this dam would affect the structure of these bridges. In January 2010, the California Department of Transportation developed a general guideline for structure type selections and their costs7. They determine the "bridge cost" based on the bridge type and the span range. They estimate that for reinforced concrete slab bridges with the span range between 16 and 19 feet, the cost range is $90-200 dollars per square foot. The bridges downstream of the dams are concrete structures measuring approximately 40 feet long (personal observations, October 2010) by 12 feet wide (http://www.ckollars.org/ipswich.html). With these considerations a rough estimation of the cost of repairing the affected bridges would be somewhere between $43,000 and 96,000. There is also major concern about the impact that removing the Ipswich Mill dam would have on the structure of the EBSCO Building located just upstream the dam (Ipswich River Watershed Association, personal communication, April 2010). However, this cost could not be estimated. With these assumptions, the estimated cost of the removal of the Ipswich Mill dam to be within a range of $608,000- $661,000. The removal cost of the South Middleton dam was estimated to be $1,310,000 using a similar method. Table 3 is a summary of preliminary cost and benefit estimates and a list of environmental benefits that still need to be quantified.

The ultimate goal of this research was to provide a useful tool that identifies ecosystem service value associated with a multitude of indicators of human influence. For this research, we focused our approach on dam removal, which is important to the restoration of aquatic and riparian ecosystems and fish passage in New England. The conceptual framework developed in this research will be available for decision makers, to address the complexities involved in ecosystem restoration, multi-objective decision-making and the optimal allocation of limited state and local resources.

Table 3: Summary of the costs and benefits associated with the two dam removals

South Middleton Dam	Ipswich Mills Dam
Cost of Removal	**Cost of Removal**
$1,310,000	$608,000- $661,000
Benefits of Removal	**Benefits of Removal**
Returning to free –flow river: $39,369 **Other benefits to be valued.** Improve Fish Habitat Support from public and private sector	Returning to free –flow river: $27,429 **Other benefits to be valued.** Improve Fish Habitat Reduces flooding upstream of the dam Reduce risk (The lack of exclusionary fencing and ease of public access)

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Publications and Conference Presentations:

a. Articles in Refereed Scientific Journals in preparation

Oglavie, D.R., E. M. Douglas, D. G. Terkla and B. Lambert, Does current policy help or hinder dam removal decision-making in Massachusetts: a survey of stakeholders, J. Environmental Management, anticipated submission, Jul 2011.

Oglavie, D.R., E. M. Douglas, D. G. Terkla and B. Lambert. A framework for valuing the environmental costs and benefits of dam removal, Water Resources Management, anticipated submission Sep 2011.

O'Brion, K., E. Douglas and A. Christian, Biomonitoring the Eel River headwaters restoration project, Plymouth, Massachusetts, Freshwater Biology, anticipated submission, Jan 2012.

b. Conference Presentations

A conceptual framework for estimating the environmental costs and benefits of dam removal (oral), National Conference on Engineering and Ecohydrology for Fish Passage, Amherst, MA June 27-29, 2011.

Freshwater fish and aquatic macroinvertebrate biomonitoring of the Eel River Headwaters Restoration sites in Plymouth, Massachusetts (oral), National Conference on Engineering and Ecohydrology for Fish Passage, Amherst, MA June 27-29, 2011.

Freshwater fish and aquatic macroinvertebrate biomonitoring of the Eel River Headwaters Restoration sites in Plymouth, Massachusetts (poster), North American Benthological Society Annual Meeting 2011, Providence RI, May 22-26, 2011.

Freshwater fish and aquatic macroinvertebrate biomonitoring of the Eel River Headwaters Restoration sites in Plymouth, Massachusetts (oral), 67th Annual Northeast Fish and Wildlife Conference, Manchester, NH, April 17-19, 2011.

A conceptual framework for estimating the environmental costs and benefits of dam removal (oral), 67th Annual Northeast Fish and Wildlife Conference, Manchester, NH, April 17-19, 2011.

Student Support
Two graduate students were supported with this grant:
- Kevin O'Brion, MS Env. Sci (summer 2010 support).
- Doina Oglavie, MS Env. Sci (summer 2010 through spring 2011).

12. Acid Rain Monitoring Project (USGS-2009MA211B)

Principal Investigator: Marie-Françoise Hatte, MA Water Resources Research Center, UMass Amherst
Start Date: March 1, 2009
End Date: February 28, 2010
Reporting Period: March 1, 2010 – June 30, 2011
Funding Source: USGS (104B) and MassDEP
Descriptors: Acid Deposition; Surface Water Quality; Volunteer Monitoring

Problem and Research Objectives
This report covers the period July 1, 2010 to June 30, 2011, the tenth year of Phase IV of the Acid Rain Monitoring Project. Phase I began in 1983 when about one thousand citizen volunteers were recruited to collect and help analyze samples from nearly half the state's surface waters. In 1985, Phase II aimed to do the same for the rest of the streams and ponds[2] in Massachusetts. The third phase spanned the years 1986-1993 and concentrated on a subsample of streams and ponds to document the effects of acid deposition to surface waters in the state. Over 800 sites were followed in Phase III, with 300 citizen volunteers collecting samples and doing pH and ANC analyses. In 2001, the project was resumed on a smaller scale: about 50 volunteers are now involved to collect samples from approximately 150 sites, 26 of which are long-term sites with ion and color data dating back to Phase I. In the first years of Phase IV (2001-2003), 161 ponds were monitored for 3 years. Between Fall 2003 and Spring 2010, the project monitored 151 sites, mostly streams, except for the 26 long-terms sites that are predominantly ponds. This year, reduced funding eliminated our October sampling and monitoring occurred on April 10, 2011 only. Some of the sites monitored changed, in order to revisit ponds that were monitored in 2001-2003.

[2] Note: The term stream in this report refers to lotic waters (from creeks to rivers) and the term ponds refers to lentic waters (lakes and ponds, but not marshes)

Goals

The goals of this project are to determine the overall trend of sensitivity to acidification in Massachusetts surface waters and whether the 1990 Clean Air Act Amendment has resulted in improved water quality.

Methods

The sampling design was changed this year to monitor both streams and ponds. In 2001-2003 mostly ponds were monitored. In Fall 2003 the sampling scheme switched to streams to evaluate their response to air pollution reductions. This year the site list was modified to include both ponds and streams. Half of the streams monitored since 2003 were kept, and half of the ponds monitored in 2001-2003 were added back. The streams that were removed were chosen randomly by county. Ponds that were reinstated on the sampling list were chosen at random by county and by ease of accessibility to replace the removed streams. Because those sites were not chosen with a preconceived plan, they can be considered picked at random.

Also different from previous years, only one collection took place this year, due to budget reductions. The April sampling date (April 10, 2011) was chosen rather than the October collection, because surface waters show lowest pH and ANC in the spring, and the project aims to document the worst case conditions.

This year the sampling location for Quabbin Reservoir, one of our long-term sites, was changed from within the Reservoir to the outlet, because availability of a boat to collect mid-reservoir ceased, and in April the reservoir is often still frozen, preventing the collection of a mid-pond sample. A previous ARM study established that sampling at the outlet does not yield significant changes from sampling mid-lake (Godfrey et al, 1996).

Methods were otherwise unchanged from previous years: Volunteer collectors were contacted a month before the collection to confirm participation. Clean sample bottles were sent to them in the mail, along with sampling directions, a field sheet/chain of custody form, and directions including GPS coordinates and maps to the sampling sites. Volunteers collected a surface water sample at their sampling sites either from the bank or wading a short distance into the water body. They collected one foot below the surface, upstream of their body, after rinsing their sample bottle three times with pond or stream water. If collecting by a bridge, they collected upstream of the bridge unless safety and access do not allow it. They filled in their field data sheet with date, time, and site code information, placed their samples on ice in a cooler and delivered the samples to their local laboratory right away. They were instructed to collect their samples as close to the lab analysis time as possible. In a few cases, samples were collected the day prior to analysis because the lab is not open on traditional "ARM Sunday." Previous studies by our research team have established that pH does not change significantly when the samples are refrigerated and stored in the dark.

Volunteer labs were sent any needed supplies (sulfuric acid titrating cartridge, electrode, buffers), two quality control (QC) samples, aliquot containers for long-term site samples, and a lab sheet one week to ten days before the collection. They analyzed the first QC sample in the week prior to the collection and called in their results to the Statewide Coordinator. If QC results were not acceptable, the volunteer analyst discussed possible reasons with the Statewide Coordinator and made modifications until the QC sample analysis gave acceptable results. On collection day or the day after, volunteer labs analyzed the second QC sample before and after the regular samples, and reported the results on their lab sheet along with the regular samples. Analyses were done on their pH-meters with KCl-filled combination pH electrodes. Acid neutralizing capacity (ANC) was measured with a double end-point titration to pH 4.5 and 4.2. Most labs used a Hach digital titrator for the ANC determination, but some used traditional pipette titration equipment. Aliquots were taken from the 26 long-term sites to fill two 60mL bottles and one 50mL tube per site for later analysis of ions and color. These aliquots were kept refrigerated until retrieval by UMass staff.

Aliquots, empty bottles, and results were collected by the ARM Statewide Coordinator a day or two after the collection. The Cape Cod National Seashore lab mailed those in, with aliquot samples refrigerated in a cooler with dry ice.

The Statewide Coordinator reviewed the QC results for all labs and flagged data for any lab results that did not pass Data Quality Objectives (within 0.3 units for pH and within 3mg/L for ANC). pH and ANC data were entered by one ARM staff and proofed by another. Data were entered in a MS excel spreadsheet and uploaded into the web-based database at http://umatei.tei.umass.edu/ColdFusionProjects/AcidRainMonitoring. Data were also posted on the ARM web page at http://www.umass.edu/tei/wrrc/arm/.

UMass Chemistry Department's Dr. Julian Tyson and his laboratory team of graduate students ran the Environmental Analysis Lab (EAL) and provided the QC samples for pH and ANC to all of the volunteer labs. EAL also provided analysis for pH and ANC for some samples from Hampshire and Franklin Counties.

Aliquots for 26 long-term sites were analyzed for color on a spectrophotometer within one day; anions within one month on an Ion Chromatograph; and cations within one month on an ICP at the Environmental Analysis Lab (EAL) on the UMass Amherst campus. The data was sent via MS Excel spreadsheet to the Statewide Coordinator who uploaded it into the web-based database.

The Statewide Coordinator and the Project Principal Investigator plotted the data to check for data inconsistencies and gaps. They then analyzed the April data from 1983 through 2011, using the statistical software JMP (http://www.jmp.com/software/) to run bivariate analyses of pH, ANC, ions, and color against date. This yielded trends analyses with a fitted X Y line, using a 95% confidence interval.

Results

1. There were 150 sites to be monitored, 77 ponds and 73 streams. Of those, 19 ponds and 7 streams are "long-term" sites that are sampled every year and analyzed for color and a suite of ions in addition to pH and ANC.

2. Sampling was completed for 144 sites (74 ponds and 70 streams), including all of our long-term sites.

3. Some quality control problems arose, mostly due to new volunteer staff performing lab analyses. This resulted in three labs failing quality control for pH and two labs failing quality control for ANC. Consequently, pH data was discarded for 29 sites, and ANC data was discarded for 10 sites.

4. The network of volunteers was maintained and kept well informed on the condition of Massachusetts surface waters so that they would be able to participate effectively in the public debate. This was accomplished by e-mail and telephone communications, as well as through updates via an internet list-serv.

79 volunteers participated in this year's collection. Several new volunteer collectors were recruited to replace ill or retiring volunteers via several internet listservs and by word of mouth.
There were 11 volunteer labs across the state, in addition to the EAL at UMass Amherst, in charge of pH and ANC analyses (Table 1).

Table 1: Volunteer Laboratories

Analyst Name	Affiliation	Town
Joseph Ciccotelli	Ipswich Water Treatment Dept	Ipswich
Nicole Henderson	UMass Boston Environmental Studies Program	Boston
Cathy Wilkins	Greenfield High School	Greenfield
Sherrie Sunter	MDC Quabbin Lab	Belchertown
Dave Bennett	Cushing Academy	Ashburnham
Holly Bayley	Cape Cod National Seashore	South Wellfleet
Robert Caron	Bristol Community College	Fall River
Bob Bentley	Analytical Balance Labs	Carver
David Christensen	Biology Dept. Wilson Hall WSC	Westfield
Jim Bonofiglio	City of Worcester Water Lab	Holden
Carmen DeFillippo	Pepperell Waste Water Treatment Plant	Pepperell
Chengbei Li	UMass Amherst Environmental Analysis Lab	Amherst

5. The ARM web site and searchable database were maintained and updated. 2011 pH, ANC, ions and color data that met data quality objectives were added to the web database via the uploading tool created in previous years. The database was evaluated for quality control and uploading errors were corrected.

6. The data collected was analyzed for trends in pH and ANC in April months only for 144 sites and for color and ions for 26 sites, using the JMP® Statistical Discovery Software (http://www.jmp.com/software/). Trend analyses (scatter plots, regression, and correlation) were run on pH, ANC, each ion, and color separately, predicting concentration vs. time.

Data Analysis Results
pH and ANC

Trend analysis for pH and ANC

Table 2 displays the number of sites out of a maximum of 144 that show a significant change over time for pH or ANC. If the difference was not statistically significant ($p>0.05$), the sites are tabulated in the 'No Change' category. Those results are also graphed in Figure 1.

This trend analysis indicates that for most sites, neither pH nor ANC changed significantly over time. However, for those sites that show a significant change, more show an increase than a decrease in value: about a quarter of the sites saw an increase in pH and ANC, more so for pH than ANC. This is consistent with previous years' results. It is interesting to compare ponds and streams this year and to note that statistical results are very similar for both types of water bodies, though pH and ANC increased for more streams than ponds.

Table 2: Trend analysis results for pH and ANC, April 1983 – April 2011

	All Sites		Ponds		Streams	
	pH	**ANC**	**pH**	**ANC**	**pH**	**ANC**
Increased	40	27	16	12	24	15
Decreased	3	2	1	0	2	2
No Change	98	111	55	59	43	52

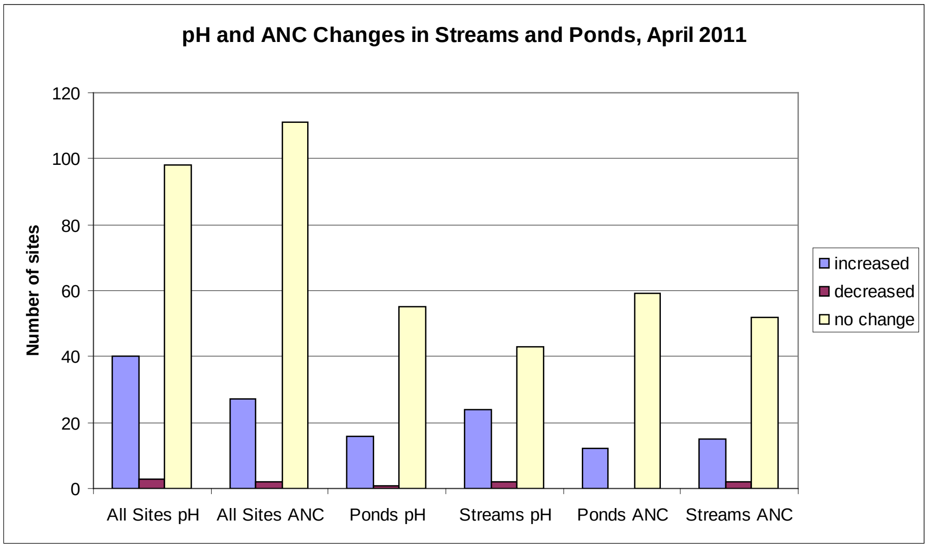


Figure 1. Changes in pH and ANC, from trend analysis

A visual check of the scatter plots for all data shows that for a number of ponds, the pH was clearly lower this year than in the past, see Figure 2 below with accompanying statistics. However, just one data point in 2011 compared to many data points before 2003 is not enough to create a statistically significant trend. Monitoring will need to continue for several years to establish whether such a trend is real or an anomaly due to late snowmelt or even an undetected laboratory error.

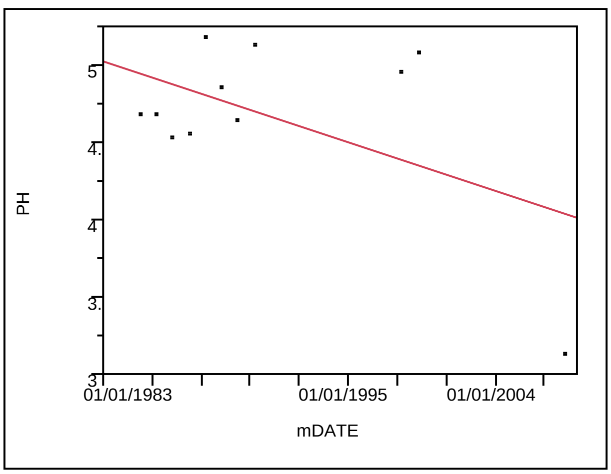


Figure 2: Example of Bivariate Fit of PH By DATE (Notch Pond, 72088.0001)

Linear Fit
PH = 7.7815024 - 1.1062e-9*mDATE

Summary of Fit

RSquare	0.25805
RSquare Adj	0.175611
Root Mean Square Error	0.512234
Mean of Response	4.668182
Observations (or Sum Wgts)	11

Analysis of Variance

Source	DF	Sum of Squares	Mean Square	F Ratio
Model	1	0.8213112	0.821311	3.1302
Error	9	2.3614524	0.262384	**Prob > F**
C. Total	10	3.1827636		0.1106

Parameter Estimates

Term	Estimate	Std Error	t Ratio	Prob>|t|
Intercept	7.7815024	1.766463	4.41	0.0017*
mDATE	-1.106e-9	6.25e-10	-1.77	0.1106

Ions and Color

Trend analyses were run for the 26 long-term sites that are analyzed for eleven ions and color.

Table 3 and Figure 3 show the results of the trend analysis for all parameters.

Table 3: Trend analysis results for ions and color

	April 1983 - April 2011		
	No Change	Increased	Decreased
Mg	21	1	4
Si	21	0	5
Mn	20	1	5
Fe	20	1	5
Al	18	2	6
Ca	13	13	0
Na	23	3	0
K	11	15	0
Cl	17	8	1
NO3	3	1	22
SO4	4	22	0
Color	21	1	4

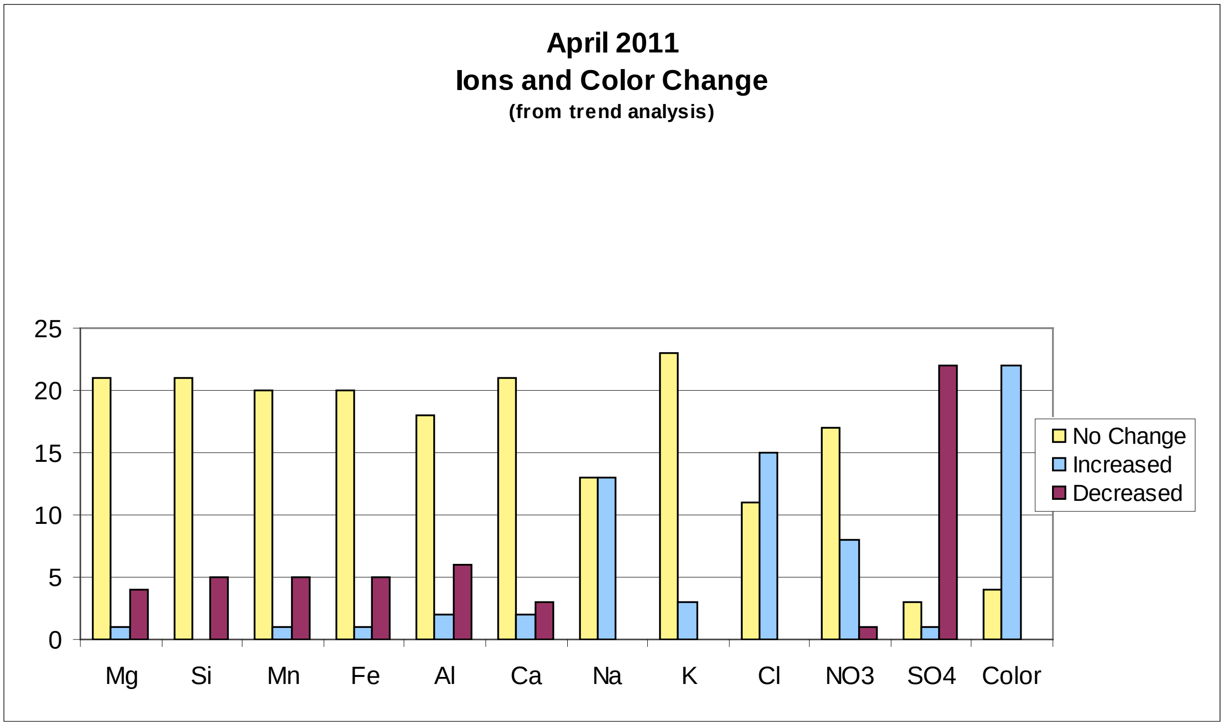


Figure 3: Results of trend analysis for ions and color for 26 long-term sites, April 2011

Most cations show no significant change over time for the 26 sites we are following. The exception, as in the past, is for sodium, which increased in half of the sites.

All anions show significant changes as well. Chloride never decreases with time, and increases for 58% of the sites. Nitrate's change is less definite, but it clearly increases for about a third of the sites and decreases for only one site. Sulfate shows the most dramatic change, a strong decrease for 85% of the sites. Color also continues to show a consistent increase over time, for 85% of the sites as well.

Discussion

Despite a heavier snowpack that melted later in the season, results for 2011 are very similar to those of previous years – pH and ANC still show more increases than decreases with time. As noted above, however, unless the change were drastic, it would not affect the statistical analysis with only one new data point.

Looking at ponds vs. streams, no noticeable difference can be detected between the two. Ponds continue to show an increase in both pH and ANC in some sites though more sites still show no statistically significant change with time.

The base cations calcium and magnesium do not show any sign of recovery, though sulfate continues to show a strong and significant decline. The increase in nitrate is still present, as is the increase in sodium and chloride, and color.

These results are consistent with the analysis performed last year (Hatte and Finn, 2010). Conclusions mirror those of 2010, namely that the increase in nitrate is thought to be caused by emissions from increased vehicular circulation, the increase in sodium and calcium is attributed to road salting, and the increase in color is due to decreased acidic inputs (see Hatte et al, 2010).

This year's data confirm that the 1990 Clean Air Act Amendment has resulted in modest improvements in water quality in Massachusetts surface waters.

It is our recommendation that monitoring these ponds and streams continue in order to document water quality trends and detect any changes that might occur due to climate change effects on surface waters.

Literature Cited

Godfrey, Paul G., Mark D. Mattson, Marie-Françoise Walk, Peter A. Kerr, O. Thomas Zajicek, Armand Ruby III, 1996. The Massachusetts Acid Rain Monitoring Project: Ten Years of Monitoring Massachusetts Lakes and Streams with Volunteers, Publication No. 171, Water Resources Research Center, University of Massachusetts Amherst. (http://www.umass.edu/tei/wrrc/WRRC2004/pdf/ARMfinalrpt.PDF)

Hatte, Marie-Françoise, Elizabeth Finn, 2010. Acid Rain Monitoring Project FY10 End of Fiscal Year Report, Water Resources Research Center, University of Massachusetts Amherst.
(http://www.umass.edu/tei/wrrc/arm/ARM%20FY10%20Annual%20Report.pdf)

Students Supported

1 BS student in Economics at UMass Amherst
1 BS student in Mathematics at UMass Amherst
1 BS student in Chemical Engineering at UMass Amherst
1 PhD student in Chemistry at UMass Amherst.

13. Evaluation of Adaptive Management of Lake Superior Amid Climate Variability and Change (USGS-Award No. G10AP00091)

Principal Investigator: Casey Brown, UMass Amherst Civil and Environmental Engineering
Start Date: April 30, 2010
End Date: March 31, 2012
Reporting Period: March 1, 2010 – June 30, 2011
Funding Source: USGS Supplemental
Descriptors: Management and Planning, Climatological Processes, Surface Water

Problem and Research Objectives

Background

The International Upper Great Lakes Study (IUGLS) began in 2007, as the International Joint Commission (IJC) established an independent study board composed of U.S. and Canadian members to review the operation of structures controlling Lake Superior outflows and to evaluate improvements to the operating rules and criteria governing the system. The Board is expected to publish recommendations in spring 2012 for near term changes to the regulation plan of Lake Superior, the largest managed freshwater body in the world (Clites and Quinn, 2003). The regulation of Lake Superior affects lake level, navigation and hydroelectricity production on Lakes Michigan, Huron and Erie, comprising an immense water resources system.

As a result of the considerable uncertainty associated with future climate and lake levels, as well as other sources of uncertainty such as ecosystem responses and the state of the navigation industry, a process of selecting the optimal plan based on a most probable future scenario was rejected. Instead, a bottom-up process for identifying vulnerabilities and assessing risk from climate change has been adopted.

Lake levels in the Upper Great Lakes levels exhibit a significant degree of natural variability in the historical record (Clites and Quinn, 2003). This variability has caused considerable challenges in the design of regulation plans for Lake Superior in the past with changes being implemented several times in the 20^{th} century. Given the lack of success in designing regulation plans that were robust to natural variability in the past and the additional uncertainty associated with climate change in the future, a change to the traditional regulation plan design is warranted. Underlying the process is the premise that we are limited in our ability to anticipate the future and therefore any recommended plan must perform well over a very broad range of possible futures. Of additional concern are surprises, low probability events that could have very large impacts. While incorporating unknown surprises into a regulation plan appeared infeasible, a strategy for managing their occurrence was prioritized. Finally, it was recognized that the identification of vulnerabilities must be led by those who understand the specific aspects of the lakes best, the stakeholders.

With these considerations, the Lake Superior regulation strategy incorporates an investigation of regulations plans that are robust to a wide range of climate conditions, definition of impacts in terms of lake levels by stakeholders, and an adaptive management process to manage uncertainty and to facilitate adaptation to changing climate, and other unanticipated changes. The uncertainty is due not only to changing climate but also due to the relatively poorly understood lake dynamics in response to changing conditions, the uncertainties associated with the definition of coping zones (described below) and other performance metrics and also the possibility of changing objectives for lake management.

The strategy consists of three primary parts:

1) Identification of vulnerabilities by stakeholders and definition of acceptable and unacceptable lake levels for each impact area, called coping zones
2) The development and assessment of a dynamic regulation plan for Lake Superior that could incorporate short range and long ranges forecasts
3) Assessment of plausible climate risk for the evaluation of regulation plans and to assess climate risk beyond what the regulation of Lake Superior outflows is able to manage.

In recognition of the short timeframe for the study and Study Board guidance regarding the extent to which adaptive management will be investigated, our recent work has focused on the identification and estimation of plausible risk posed by climate change.

The work in this report reflects an effort that is conducted in collaboration with a variety of U.S. and Canadian partners. The UMass Hydrosystems Research Group is engaged in various aspects of these tasks. This report is an attempt to describe our efforts in that regard but does not report on the considerable effort being made by others. The effort reported here is the result of collaboration of many partners and it is not intended to be interpreted as entirely our effort, although the results reported here have all been produced by our group.

<u>Part 1. Vulnerability Identification and Definition of Coping Zones</u>

In order to prioritize concerns for the regulation of Lake Superior, stakeholder experts were tasked with identifying the vulnerabilities of the system to climate changes and other changing conditions. Termed technical working groups, stakeholders and technical experts convened in the following impact areas: ecosystems, hydropower, commercial shipping, municipal and industrial water and

wastewater systems, coastal systems and recreational boating and tourism. A primary challenge was the quantification of vulnerabilities in commensurate units. To address this issue, our stakeholder groups represented by the Technical Working Groups (TWGs) were asked to define vulnerabilities in terms of lake levels, including the duration of the lake level. Lake levels were defined in three categories called "coping zones": A (acceptable), B (significant negative impacts, but survivable) and C (intolerable without policy changes). The TWGs are defining what combination of lake level and duration lead to the kind of impacts consistent with the coping zone descriptions. The definition of coping zones allows the evaluation of regulation plan performance to be conducted in terms that are comparable across impact sector and defined by the stakeholders.
At present we have evaluated the coping zones reported by all the Technical Working Groups. Table 1 shows the most conservative range of zones from those reported by Coastal TWG. The low coping zone C occurred 7 times during the 1918 – 2008 period, which seems frequent for a condition deemed intolerable. Figure 1 shows graphically the levels for each lake that define coping zones B and C. The Coastal TWG zones continue to be the most conservative. The occurrence rate for coping zones under historic and the 50,000 year stochastic NBS time series are shown in Figures 2 through 5. Figures for Michigan-Huron, Erie and Lake St. Clair are in the Appendix. The results indicate that low C occurrences are more frequent than high C, largely as a result of a more conservative definition on the low side.

Table 1. Coastal TWG coping zones and the number of occurrences based on monthly Lake Superior levels 1918 – 2008. Note that Coast TWG coping zones vary by season on the low levels.

Coping Ranges for Superior			Count
Upper C	183.95	183.95	0
Upper B	183.56	183.56	244
Lower B	183.06	182.89	10
Lower C	182.99	182.79	7
Valid Months	Jun-Nov	Dec-May	

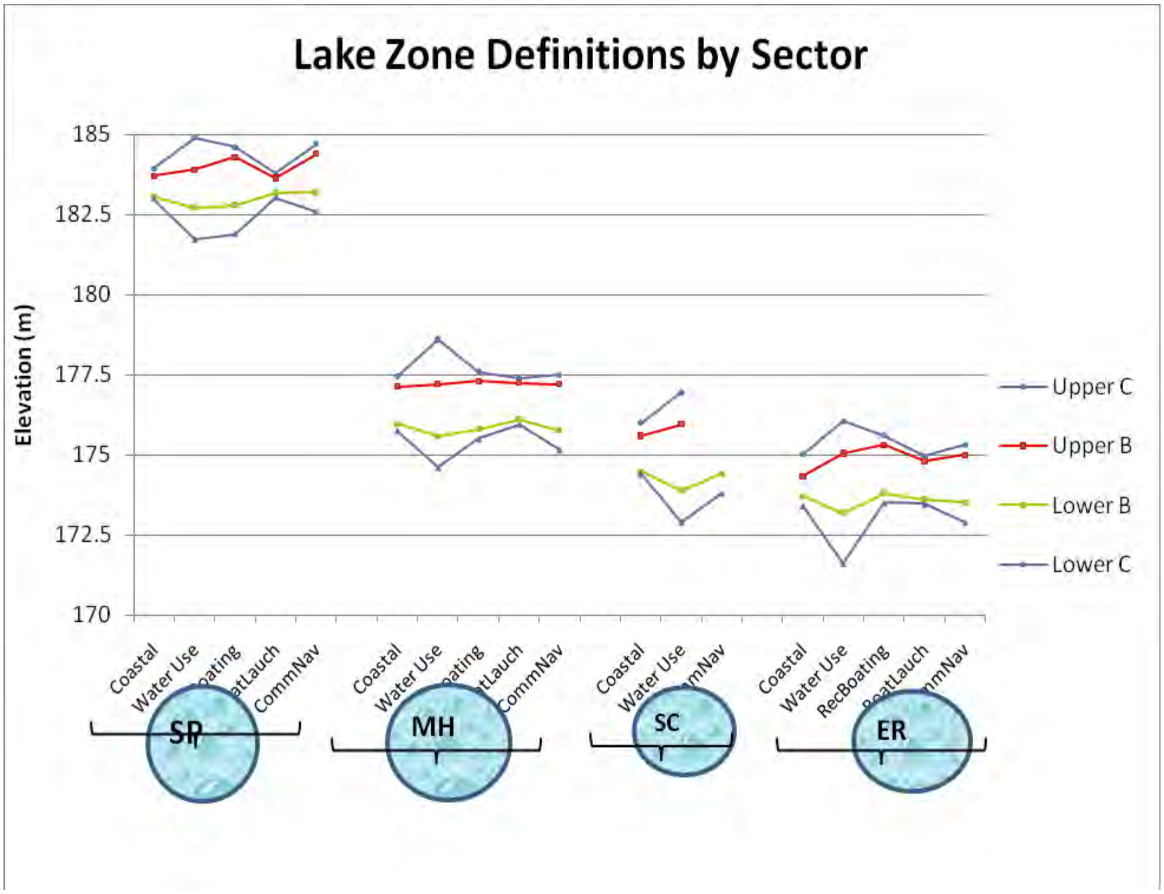


Figure 1: Zone Definitions by Lake and Sector

In a technical sense, the definition of the coping zones constitutes the definition of the "hazard," that is, the negative impacts that result from, say, a climate change. It does not take into account the probability of those impacts. Risk is defined as the product of the hazard and the probability of that hazard occurring. Our approach and progress in estimating risk is described in Part 3.

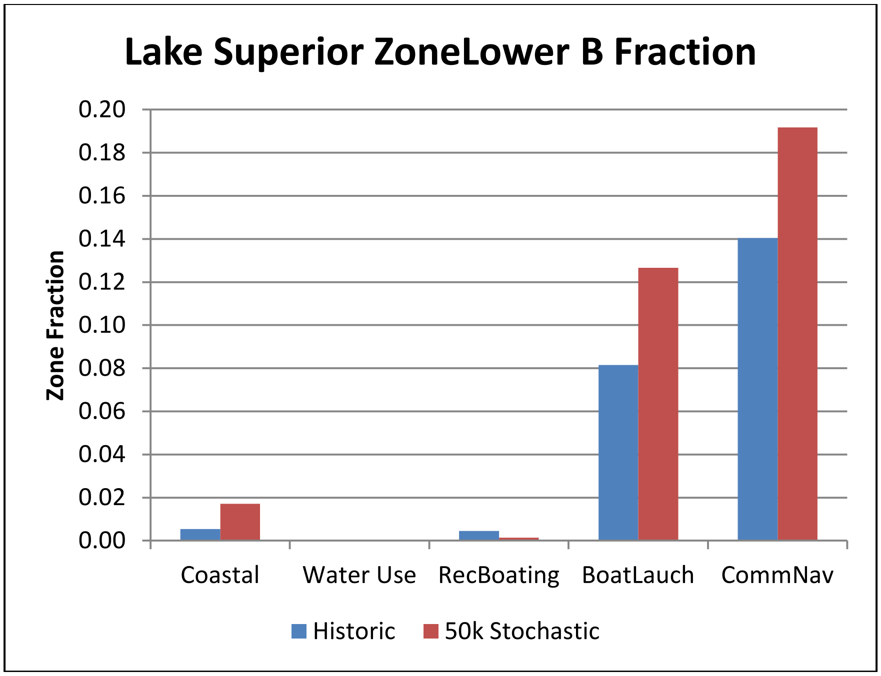


Figure 2. Lower Zone B Occurrences on Lake Superior

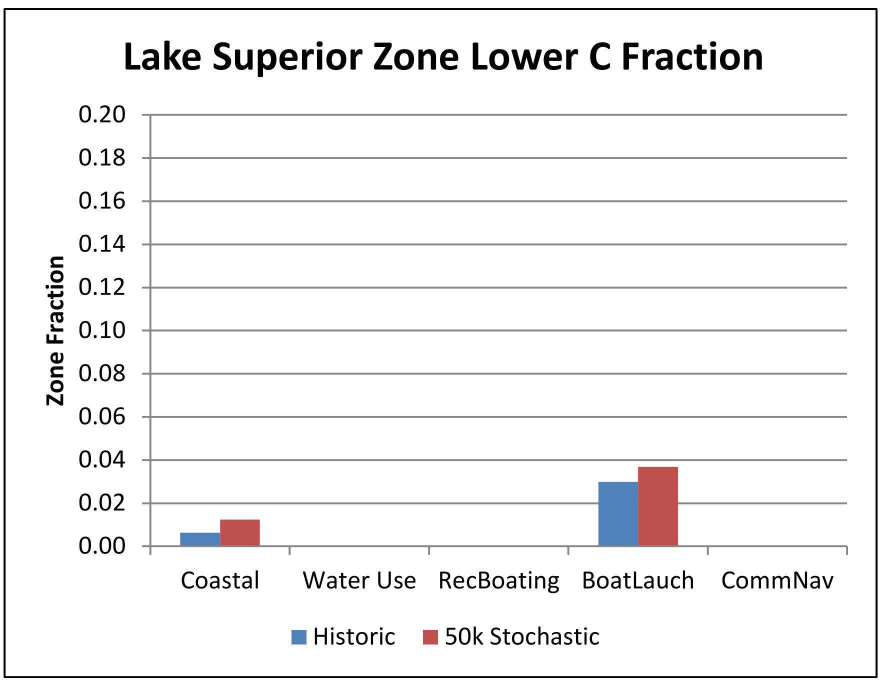


Figure 3. Lower Zone C Occurrences on Lake Superior

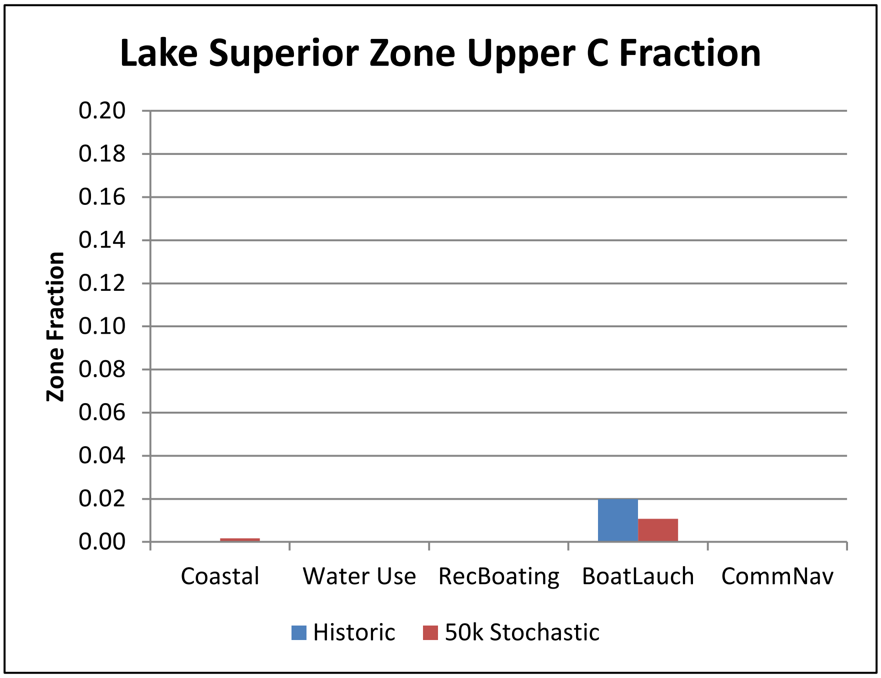


Figure 4. Upper Zone C Occurrences on Lake Superior

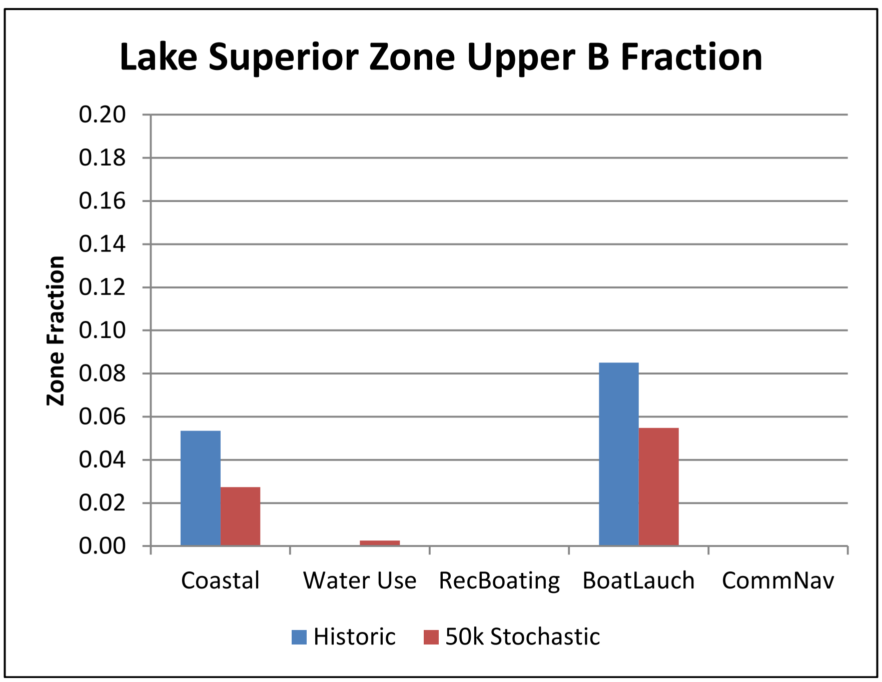


Figure 5. Upper Zone B Occurrences on Lake Superior

Part 2. Development and Assessment of a Lake Superior Regulation Plan

Traditional water resources planning often focuses on formulating an optimal design based on performance evaluated with a best estimate of future hydrologic conditions. This is an apt description for previous approaches to developing regulation plans for Lake Superior (see Clites and Quinn, 2003). Regulation plans for Lake Superior were modified approximately seven times during the regulated period of 1914 to the present. Changes to the regulation plans resulted not only due to hydrologic conditions but also due to evolving societal priorities for regulation, including increasing interest in hydroelectricity production and new emphasis on including impacts on downstream lakes.

The relatively frequent rate of adjustment of past regulations plans reinforced the emphasis of this study on robustness, defined as a regulation strategy that could perform acceptably over a wide range of climate conditions. In order to increase the range over which a regulation plan can perform acceptable, the concept of a dynamic regulation plan has been proposed. The theory is that a dynamic plan would have rules that vary based on the prevailing climate conditions and possibly a forecast of future climate conditions. Our conceptualization of a dynamic regulation plan has two aspects: 1) a short term change in releases based on a short term forecast (< 1 year ahead) and 2) a major change in regulation strategy based on evidence of a major climate shift. Because new regulation strategies are still in development, most are very similar to the current 77A plan, and there has been development of a potentially skillful forecast of Lake Superior NBS, we have focused efforts on the first aspect of dynamic regulation.

Specifically, an alteration to plan 77A has been developed and evaluated based on a decision rule that relies on logic related to lake level and a forecast of Lake Superior NBS developed by Vincent Fortin and others at Environment Canada. The decision rule, simply stated, adjusts the plan 77A release either higher (or lower) when the Lake Superior level is in the upper tercile (or lower tercile) and the forecast is for above average NBS (or below average NBS). The decision rule is shown in Table 2 along with the results of the analysis. In an initial analysis, the release was adjusted by 5 to 10% in accordance with the decision rule and the resulting lake levels were assessed using Werick's Excel version of the 77A and the lakes routing model. These results showed that for certain years, a correct forecast could lower high lake levels by 3 – 6 cm and increase low lake levels by 2 – 5 cm on Lake Superior while having minimal impact on downstream lakes. Because the Werick Excel version of 77A does not react dynamically to changing lake levels, only single year results are obtained. These are shown in the figures B1 – B4 in the Appendix. The decision rule used for evaluation of the forecasts is purposely designed to be conservative, only using forecasts when a "wrong forecast" will have little impact. That is, it is presumed that when at a high tercile the impact of increasing releases will be small even if the forecast for high NBS does not come true. To test this hypothesis we explicitly evaluated lake levels in the actual years the Fortin forecast was not correct. The results are shown in Figure B5 and B6.The results proved promising enough to warrant evaluation with a full 77A and lakes routing model, which the UMass Hydrosystems Research Group has recently developed in Matlab. A comparison of the Matlab model results with the existing Fortan model results are shown in Figure B7. The results are very positive with a correlation of 0.9987. The model will also be useful in the evaluation of plausible risk of plan 77A, modified 77A, and new regulation plans as they are developed. Using Matlab 77A, the dynamic response of 77A to the cumulative effects of the modified releases could be evaluated. The results showed that the use of perfect forecasts has a compressing effect on Lake Superior levels and minimal effect on Lakes Michigan, Huron and Erie. The distribution of Lake Superior levels with and without the perfect forecast is shown in Figure B8. More detailed results, including the effects of operation forecasts are available but have not been included here in the interest of conciseness.

Table 2. Decision Rule for modified 77A releases from Lake Superior based on lake level and NBS forecast. The table shows the changes in the releases and also the effects of those changes based on our evaluation. Of particular interest are the effects of "big misses" and false alarms, as discussed in the text. The results show that these do not have large impacts due to the conservative decision rule.

Lake Level (Tercile)	Forecast (Tercile)	Proposed AM Action	Observed NBS (Tercile)	Comments	Impact
High	High	Increase Outflow by X%	High	Correct Forecast and Action	Reduce Highs by 3 - 6 cm (5-10% increase)
		Increase Outflow by X%	Medium	False Alarm	Reduce Highs by 3 - 6 cm (5-10% increase)
		Increase Outflow by X%	Low	False Alarm	Reduce Highs by 3 - 6 cm (5-10% increase)
	Medium	No Action	High	Miss	
		No Action	Medium	Correct Forecast and Action	
		No Action	Low	Miss	
	Low	No Action	High	Big Miss	Highs 3-6 cm higher than with AM
		No Action	Medium	Correct Action	
		No Action	Low	Correct Forecast and Action	
Medium	High	No Action	High	Correct Forecast and Action	
		No Action	Medium	Correct Action	
		No Action	Low	Correct Action	
	Medium	No Action	High	Correct Action	
		No Action	Medium	Correct Forecast and Action	
		No Action	Low	Correct Action	
	Low	No Action	High	Correct Action	
		No Action	Medium	Correct Action	
		No Action	Low	Correct Forecast and Action	
Low	High	No Action	High	Correct Forecast and Action	
		No Action	Medium	Correct Action	
		No Action	Low	Big Miss	Lows 2-5 cm lower than with AM
	Medium	No Action	High	Miss	
		No Action	Medium	Correct Forecast and Action	
		No Action	Low	Miss	
	Low	Decrease Outflow by X%	High	False Alarm	Increase lows by 2-5 cm (5-10% decrease)
		Decrease Outflow by X%	Medium	False Alarm	Increase lows by 2-5 cm (5-10% decrease)
		Decrease Outflow by X%	Low	Correct Forecast and Action	Increase lows by 2-5 cm (5-10% decrease)

Next Steps: The use of forecasts with plan 77A shows some promise for improving the range of acceptable performance of the plan and may also aid other candidate regulation plans. Our next analyses would focus on the use of a mock forecast developed by Vincent Fortin and applied to long records, stochastic NBS series and GCM based-series. To improve plan performance, the decision rule could be experimented with in terms of the % alteration of release and the use of lake level in the decision rule. At present, the focus of our work is on climate risk estimation. However, if time and interest permit, we could pursue investigation of the incorporation of forecasts into Lake regulation.

Part 3. Assessment of Plausible Climate Risk

The assessment of plausible climate risks consists of three steps. The first is the identification of climate hazards, which is conducted without regard to how plausible a hazard may be. The identification of hazards is described in Part 1. The second step consists of relating climate hazards to the climate conditions that cause them. We do so by "data mining" the 50K stochastic NBS time series and summarize the findings through the development of the climate response function. This is described below in section 3.1. The final step is the estimation of plausible climate risk, which is completed by using the climate response function to estimate the impacts of climate conditions derived from a variety of sources of climate information, including output from GCM and stochastic series, including paleodata-based. This step is described below in section 3.2.

3.1 Development of the Climate Response Function

Climate conditions on the Great Lakes can be summarized in terms of Net Basin Supplies (NBS), where NBS is the sum of precipitation, runoff, releases, inflows and diversions and evaporation (negative). The identification of risks first depends on the description of risk. In this process, risks are quantified in terms of coping zones as described above. The coping zones are used to quantify risk level by counting occurrences of coping zones B and C. Next, in order to relate coping zones to the estimation of plausible climate risk, the coping zones occurrences must be related to the climate conditions that cause them. The identification of climate risks for a particular regulation plan is conducted by summarizing the climate conditions in terms of the statistics of NBS and tracking the occurrence of zone B and C occurrences for each climate scenario. Note that because plausibility must be assessed in terms of climate conditions, i.e., the long term statistics of weather, the risks must be quantified in terms of climate conditions (e.g., 30 year mean annual NBS instead of a particular annual value of NBS).

Figures 6 through 9 display the relationship between climate conditions (in terms of NBS statistics) and the occurrences of coping zone levels on Lakes Superior and Michigan-Huron, as indicated by the symbols and their size. These figures are generated by "data mining" the 50K stochastic time series. Climate conditions for 30 year windows are calculated and the zone occurrences tracked. The figures indicate that zone occurrences are much more common under conditions of reductions in mean NBS and increases in variability. High zone levels occur only infrequently and under fairly extreme conditions.

Relationships between the explanatory climate conditions and the number of occurrences are then developed. The results indicate that no single statistic completely explains the occurrence of the zones. However, the mean, standard deviation and serial correlation of NBS over the 30 year climate window have significant influence on them. These results form the basis for the climate response function.

The development of the climate response function is the key step that will relate climate hazards to their plausibility in order to estimate risk. The function is used to predict the number of zone occurrences that will occur under a given set of climate conditions or climate change. It is designed to be able to directly link the climate change projections to the information encapsulated in the climate response function. A major benefit of this approach is that since the climate response function was developed on 50,000 years of NBS, the results yield a much more robust estimate of risk than would be attained by simply running the 30 year GCM time series through the lake routing model.
The underlying hypothesis is that statistics of climate variables can explain or predict zone occurrences on the Upper Great Lakes. Three climate statistics based on annual NBS values were analyzed. During the study, 100 year, 50 year, and 30 year windows were considered. 30 year analysis windows were used because it allows direct comparison with GCM models output. The three climate statistics are the normalized mean annual NBS, the normalized standard deviation of the annual NBS and the serial correlation of the annual NBS. The mean annual NBS and the standard

deviation of the annual NBS were normalized based on the historic NBS series mean and standard deviation.

$$\overline{NBS} = \frac{1}{n}\sum_{i=1}^{n} NBS_i \tag{1}$$

$$X_1 = \overline{NBS}^* = \frac{\overline{NBS} - \overline{NBS}_{hist}}{\overline{NBS}_{hist}} \tag{2}$$

$$s_{NBS} = \left(\frac{1}{n-1}\sum_{i=1}^{n}\left(NBS_i - \overline{NBS}\right)^2\right)^{\frac{1}{2}} \tag{3}$$

$$X_2 = s_{NBS}^* = \frac{s_{NBS} - s_{NBS,hist}}{s_{NBS,hist}} \tag{4}$$

$$X_3 = r = \frac{1}{n-1}\sum_{i=1}^{n-1}\frac{\left(NBS_i - \overline{NBS}\right)^2\left(NBS_{i+1} - \overline{NBS}\right)^2}{S_{NBS}^2} \tag{5}$$

The relationship between the percent change in average annual NBS, the standard deviation of the annual NBS, and the relative occurrence of zone C is shown in Figure 2 for Lake Superior and in Figure 3 for Lakes Michigan-Huron. Figure 4 shows the relationship between percent change in NBS, serial correlation and zone occurrences for Lakes Superior and Figure 5 for Michigan-Huron. The number of zone occurrences is proportional to the size of the marker.

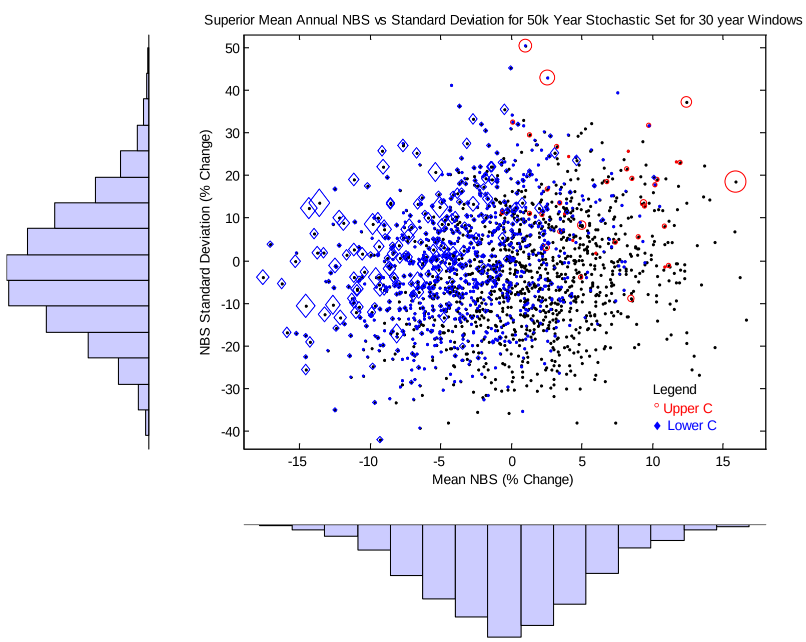


Figure 6: Lake Superior Mean NBS versus NBS Standard Deviation with Upper and Lower Zone C Occurrences for 30 Year Windows from the 50k Year Stochastic Data Set.

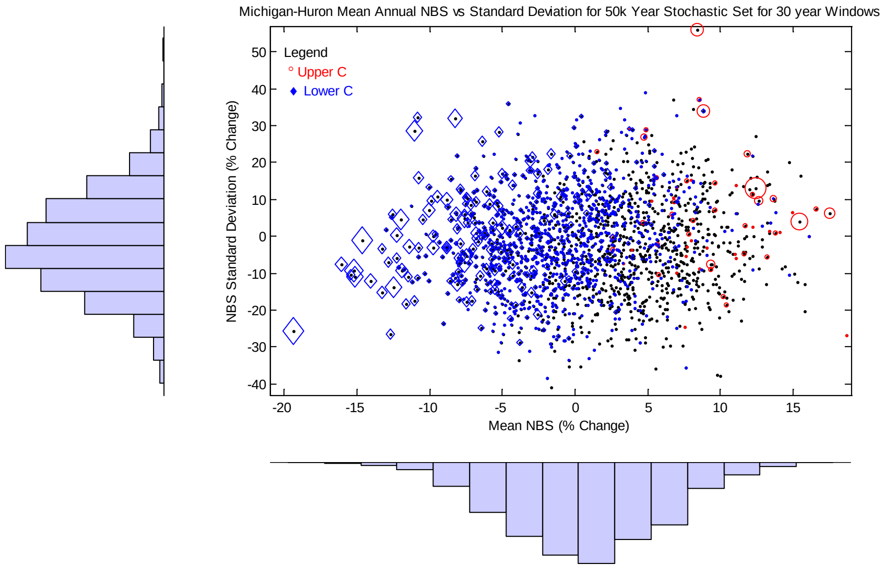


Figure 7: Lakes Michigan-Huron Mean NBS versus NBS Standard Deviation with Upper and Lower Zone C Occurrences for 30 Year Windows from the 50k Year Stochastic Data Set.

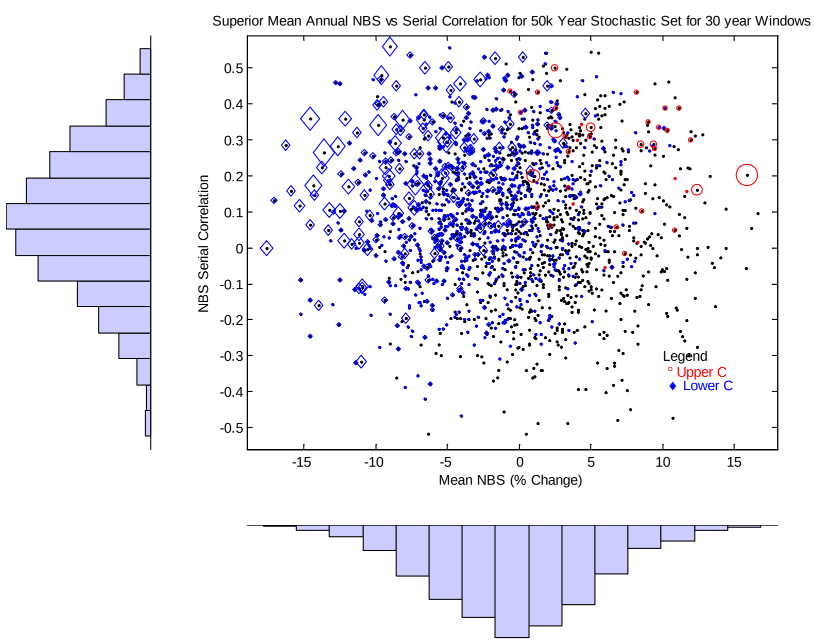


Figure 8: Lake Superior Mean NBS versus NBS Serial Correlation with Upper and Lower Zone C Occurrences for 30 Year Windows from the 50k Year Stochastic Data Set.

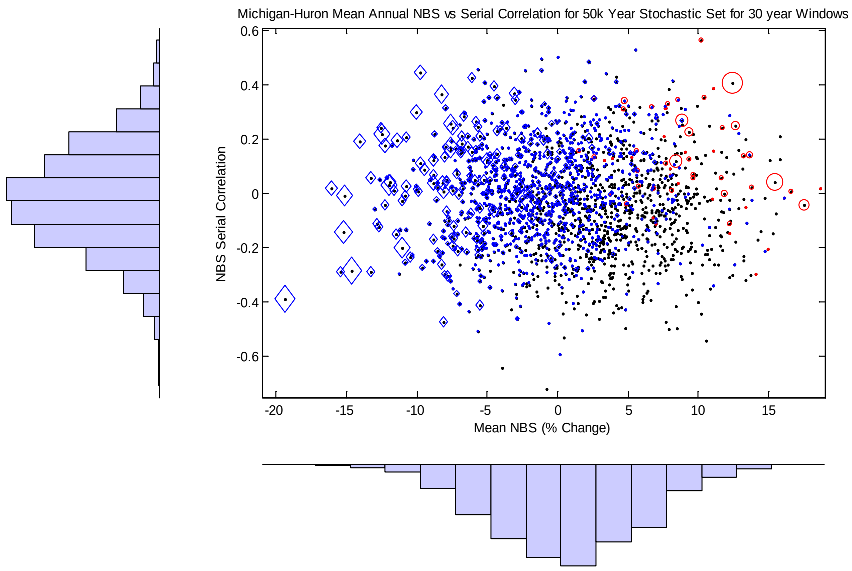


Figure 9: Lakes Michigan-Huron Mean NBS versus NBS Serial Correlation with Upper and Lower Zone C Occurrences for 30 Year Windows from the 50k Year Stochastic Data Set.

The figures demonstrate the relation between the climate variables and the zone occurrences. This led to the formulation and testing of statistical models of the relationship between the climate statistics and the zone occurrences. The number of zone occurrences can be treated as fractional data or as non-negative discrete data. This analysis treats the number of zone occurrences in the Lower C, Lower B, Upper B, and Upper C as independent. The occurrences are not, strictly speaking, independent and the total number of zone occurrences in Lower C, Lower B, A, Upper B and Upper C is equal to the number of months in the analysis period. In this statistical analysis, the model's ability to estimate the number of zone occurrences per zone was assessed. This model did not constrain the total number of predicted months.

Based on data screening, an exponential model was selected for the deterministic function. The data screening demonstrated a correlation between all three predictor climate statistics and the zones, with X_1 = percent change in mean NBS showing the strongest correlation. The candidate exponential models in Equations 6a through 6d were considered.

$$E(Y) = A\ exp(B\ X_1) \tag{6a}$$

$$E(Y) = A\ exp(B\ X_1 + C\ X_2) \tag{6b}$$

$$E(Y) = A\ exp(B\ X_1 + D\ X_3) \tag{6c}$$

$$E(Y) = A\ exp(B\ X_1 + C\ X_2 + D\ X_3) \tag{6d}$$

where Y is the expected number of Lower C, Lower B, Upper B or Upper C zone occurrences. The Maximum Likelihood Estimate method was used to determine the model parameters. Unlike the ordinary least squares method, this method requires an explicit assumption of the stochastic error component of the model. There are several stochastic error models appropriate for a non-negative, integer predictand. The two primary models that considered were the Poisson distribution and the negative binomial distribution. The Poisson distribution has a single parameter, λ, and has the following properties:

$$P_P(y = Y) = \frac{e^{-\lambda}\lambda^y}{y!} \tag{7a}$$

$$E[P_P] = \lambda \tag{7b}$$

$$s_{P_P} = \lambda \tag{7c}$$

The parameterization for the Poisson model allows us to substitute the deterministic model component in for λ. Substituting equation 6a into 7a yields:

$$Y \propto \frac{e^{A\,exp(B\,X_1)}(A\,exp(B\,X_1))^y}{y!} \tag{8}$$

The advantages of using a Poisson stochastic model include that it is bounded to non-negative integers and that it does not add additional model parameters. The model also allows for a standard deviation that changes with the expected value.

The negative binomial model has two parameterizations. The most common is the mechanistic parameterization where *p* represents the per trial probability of success and *n* represents the number of successes awaited. In this parameterization, *y* is the number of failures until *n* successes. The alternative parameterization is μ which is the mean and *k* which is an overdispersion parameter. In this case, *y* does not have a mechanistic interpretation.

$$P_{NB}(y = Y) = \frac{\Gamma(\mathbf{k}+\mathbf{y})}{\Gamma(\mathbf{k})\mathbf{y}!}\left(\frac{\mathbf{k}}{\mathbf{k}+\boldsymbol{\mu}}\right)^{\mathbf{k}}\left(\frac{\boldsymbol{\mu}}{\mathbf{k}+\boldsymbol{\mu}}\right)^{\mathbf{y}} \tag{9a}$$

$$E[P_{NB}] = \mu \tag{9b}$$

$$S_{P_{NB}} = \mu + \frac{\mu^2}{k} \tag{9c}$$

The advantage of the negative binomial model is that the parameter *k* can capture the greater standard deviation, or dispersion, of the predictand relative to the Poisson model.

The Maximum Likelihood Estimate method was used to estimate parameters. This method seeks the parameters that provide the greatest likelihood, given the data. For a data point, y_i, the probability of y_i given the input variables $\boldsymbol{X}_i$ and the parameters $\boldsymbol{C}$ can be expressed as the likelihood of the data and is given in Equation 10. Note that $\boldsymbol{X}$ and $\boldsymbol{C}$ can be vectors from X_1 to X_n and C_1 to C_m.

$$P(y_{\downarrow}i \mid X_{\downarrow}i, C) = L\{ [\![data]\!]_{\downarrow}i \mid X_{\downarrow}i, C\} \tag{10}$$

Assuming that the data points are independent, or that each 30 year sequence is independent of the next, then the likelihood of the entire data series is the product of the likelihood of each data point, as shown in Equation 11.

$$L\{data \mid X, C\} = \prod_{i=1}^{N} \left(P(y_i \mid X_i, C)\right) \qquad (11)$$

The product of small numbers gets small very rapidly, so it is common to take sum logarithm of the likelihoods. The maximum value of log likelihood occurs at the MLE estimates of the parameters, ***C***. Since many search algorithms are written to find minimum values, it is normal to search for the parameters, ***C***, that produce the smallest negative log likelihood, NLL.

$$NLL\{data \mid X, C\} = \sum_{i=1}^{N} \left(P(y_i \mid X_i, C)\right) \qquad (12)$$

The absolute value of the NLL does not have a probabilistic or physical meaning, but the relative value does convey information about the relative support for the model given the data. Competing models can be compared using their NLL values based on the best fit parameterization using information criteria. The most common of these is the Akaike Information Criteria (AIC), which is calculated using Equation 13. The principle of parsimony implies that simple models are better. Nearly any model can have an improved fit with more parameters or input variables, but the AIC penalizes the more complex model based on the number of parameters used, m.

$$AIC = 2\ NLL + 2\ m \qquad (13)$$

The relative AIC value is then used to distinguish between competing models. Models with a Δ AIC < 2 are indistinguishable; with 4 < Δ AIC < 7 they are distinguishable; and models with Δ AIC > 10 are definitely different.

Table 3: Model comparison for five competing models to predict Lake Superior Lower C zone occurrences using a 30 year analysis window over the 50k stochastic data set.

Deterministic Model	Stochastic Model	NLL	Parameters	AIC	Δ AIC
A exp(B X_1)	Poisson	7,050	2	14,104	6,554
A exp(B X_1 + C X_2)	Poisson	6,304	3	12,614	5,064
A exp(B X_1 + D X_3)	Poisson	6,379	3	12,764	5,214
A exp(B X_1 + C X_2 + D X_3)	Poisson	5,858	4	11,724	4,174
A exp(B X_1 + C X_2 + D X_3)	Negative Binomial	3,770	5	7,550	0

As shown in Table 1, the negative binomial stochastic model with an exponential model including three predictor variables has the lowest NLL value and the lowest AIC value. The Δ AIC values indicate that this model has significantly more support, given the data. Similar analysis of Lower B, Upper B and Upper C zones, all show that the exponential model with a negative binomial stochastic distribution is the consistent best fit model. Figures 10 through 13 show the expected number of zone occurrences with a 95% confidence interval for Lower C, Lower B, Upper B and Upper C on Lake Superior. Figures 14 through 17 show the same for Lakes Michigan-Huron. In each case, the only predictor variable shown is X_1, the percentage change in mean NBS, since it has the highest correlation.

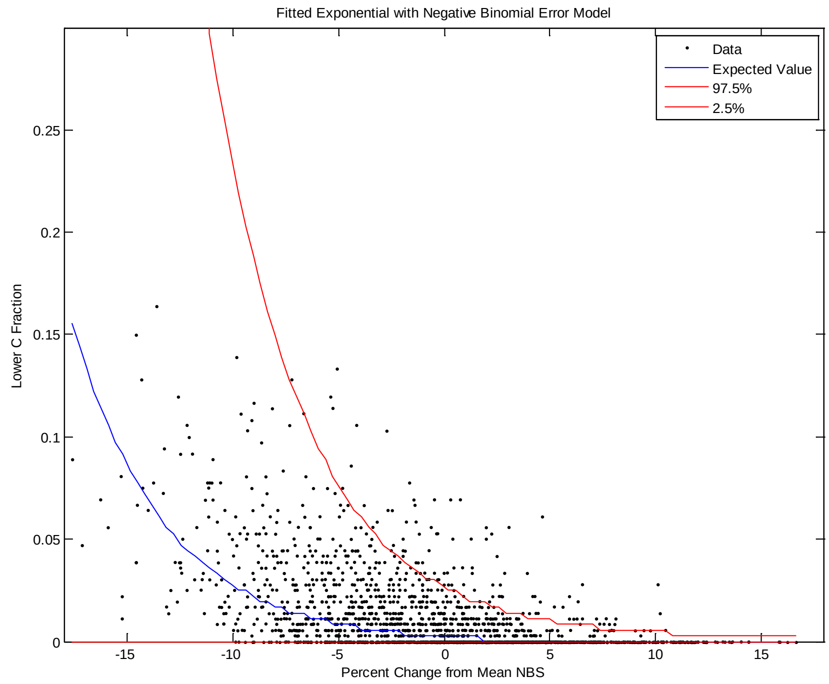


Figure 10: Lake Superior Mean NBS versus Lower Zone C Occurrences for 30 Year Windows from the 50k Year Stochastic Data Set with an exponential deterministic and negative binomial stochastic model.

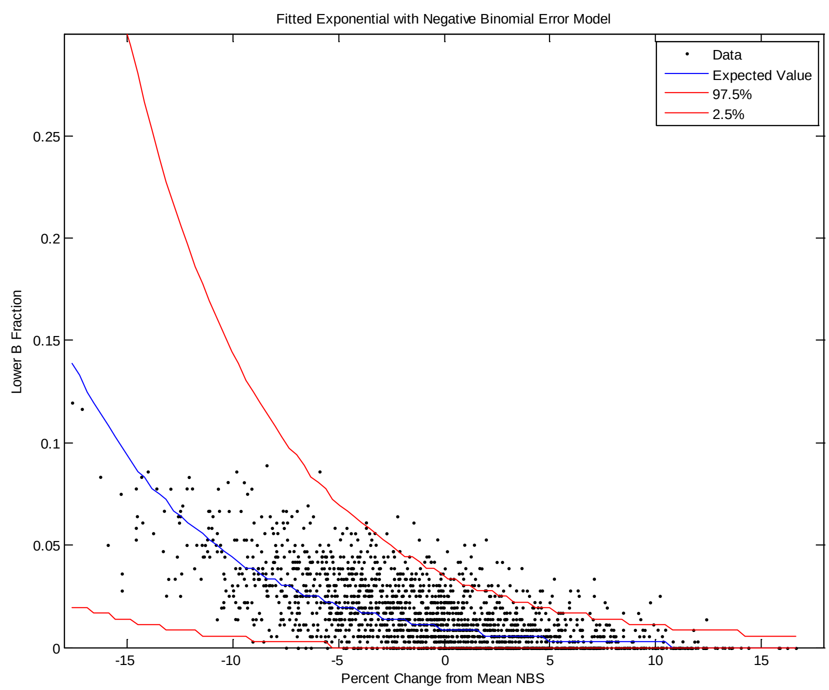


Figure 11: Lake Superior Mean NBS versus Lower Zone B Occurrences for 30 Year Windows from the 50k Year Stochastic Data Set with an exponential deterministic and negative binomial stochastic model.

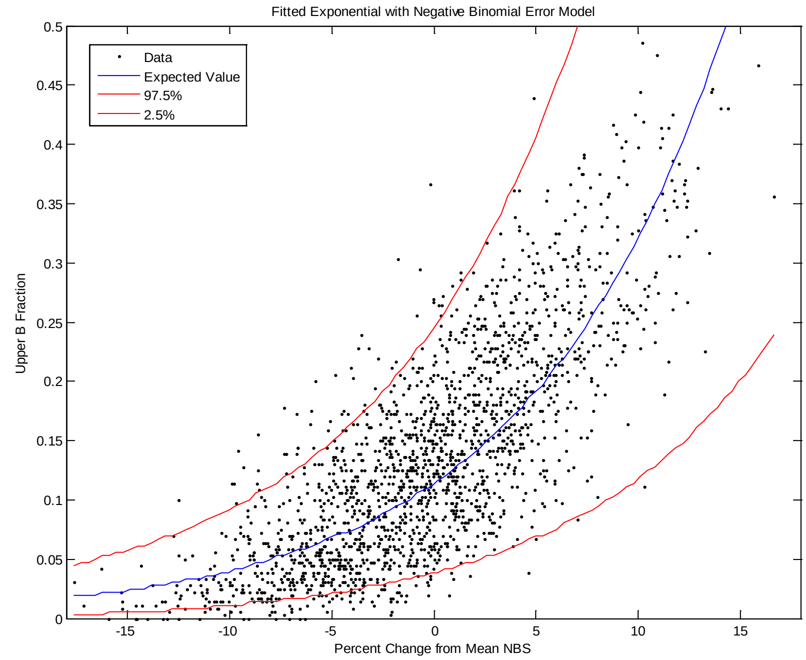


Figure 12: Lake Superior Mean NBS versus Upper Zone B Occurrences for 30 Year Windows from the 50k Year Stochastic Data Set with an exponential deterministic and negative binomial stochastic model.

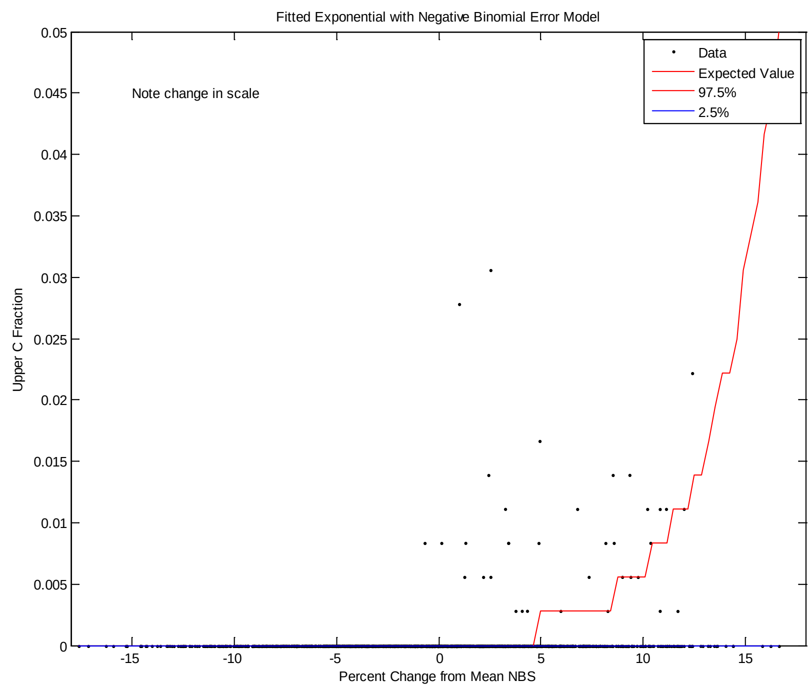


Figure 13: Lake Superior Mean NBS versus Upper Zone C Occurrences for 30 Year Windows from the 50k Year Stochastic Data Set with an exponential deterministic and negative binomial stochastic model.

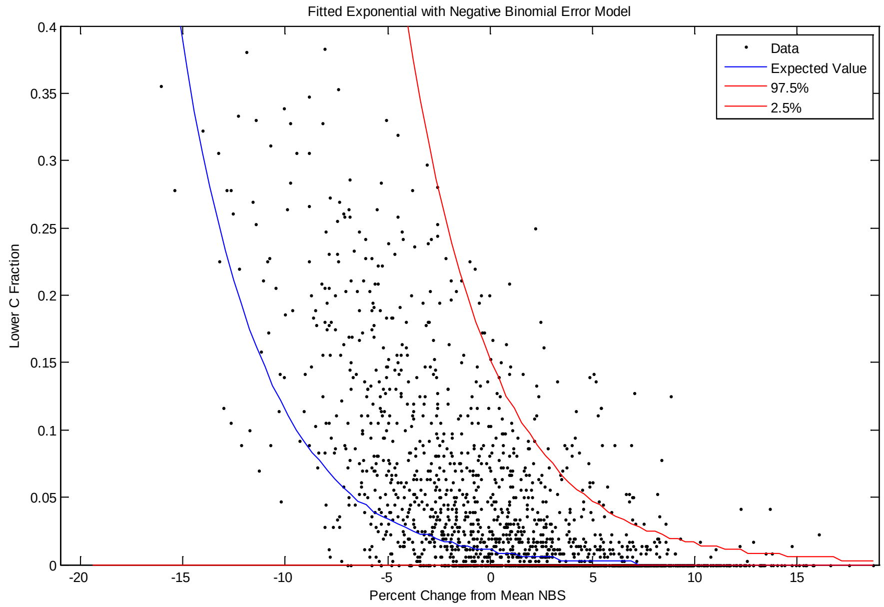


Figure 14: Lakes Michigan-Huron Mean NBS versus Lower Zone C Occurrences for 30 Year Windows from the 50k Year Stochastic Data Set with an exponential deterministic and negative binomial stochastic model.

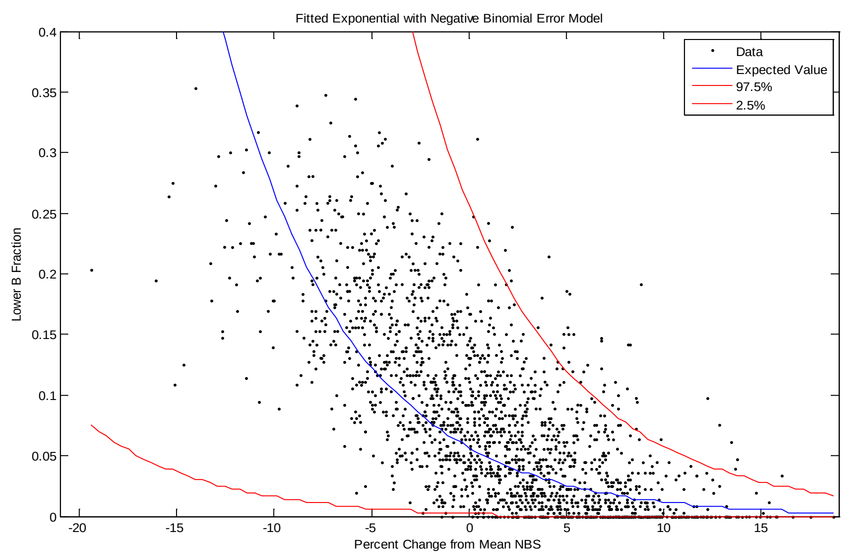


Figure 15: Lakes Michigan-Huron Mean NBS versus Lower Zone B Occurrences for 30 Year Windows from the 50k Year Stochastic Data Set with an exponential deterministic and negative binomial stochastic model.

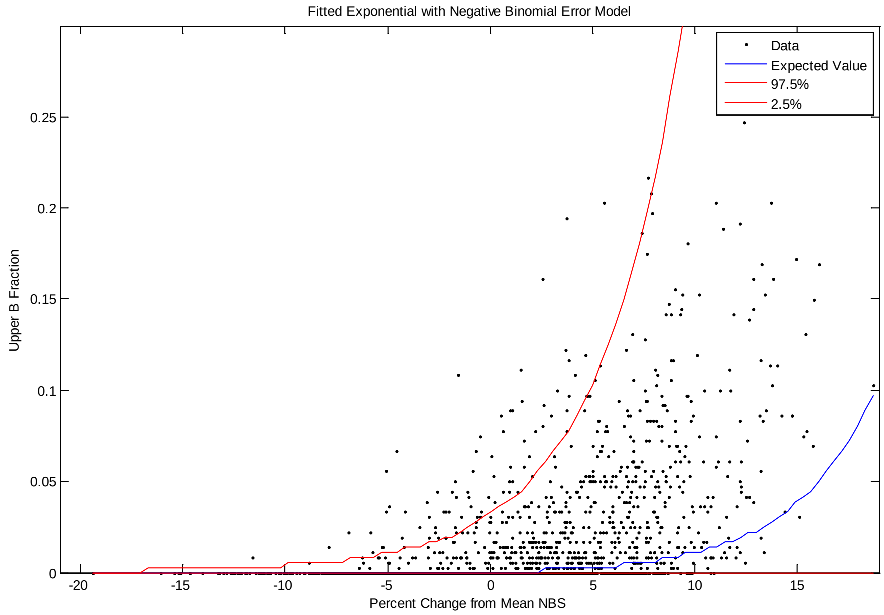


Figure 16: Lakes Michigan-Huron Mean NBS versus Upper Zone B Occurrences for 30 Year Windows from the 50k Year Stochastic Data Set with an exponential deterministic and negative binomial stochastic model.

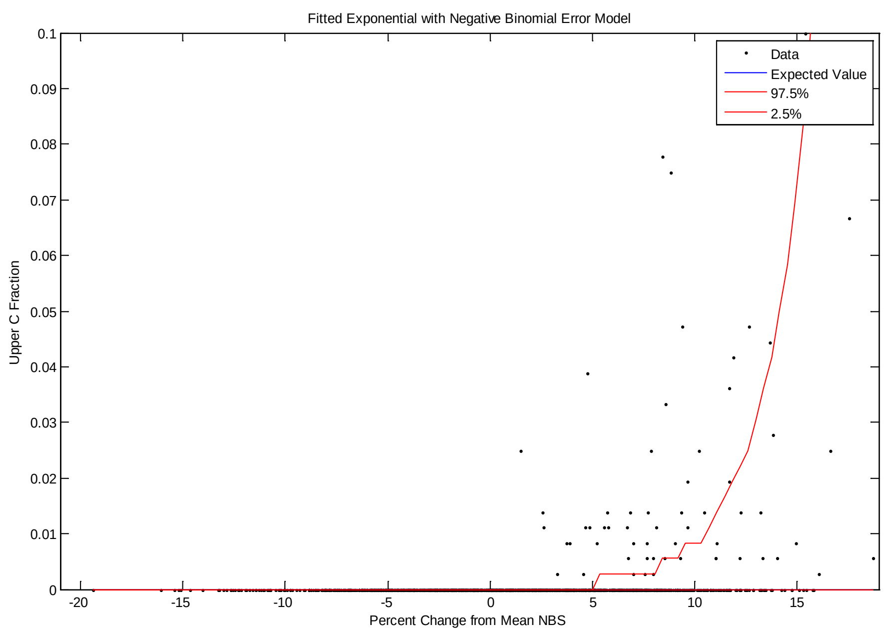


Figure 17: Lakes Michigan-Huron Mean NBS versus Upper Zone C Occurrences for 30 Year Windows from the 50k Year Stochastic Data Set with an exponential deterministic and negative binomial stochastic model.

The sum of the number of expected zone occurrences provides an indication of the climate conditions which would put the Upper Great Lakes at the highest risk given the current regulation plan. The impact is measured by an excessive number of zone occurrences. Figures 18 and 19 show a surface where the surface height, *z*, is equal to the fractional number of expected Lower C, Lower B, Upper B and Upper C occurrences on Lake Superior. Figure 18 shows the zone occurrences as a function of percent change mean NBS versus percent change NBS standard deviation while holding serial correlation constant. Figure 19 shows the relation between NBS percent change and serial correlation to zone occurrences while holding percent change standard deviation constant. Where the sum of the expected zone occurrences is greater than the number of months, the fraction is capped at 1. Figures 20 and 21 show the same relationships for Lakes Michigan-Huron.

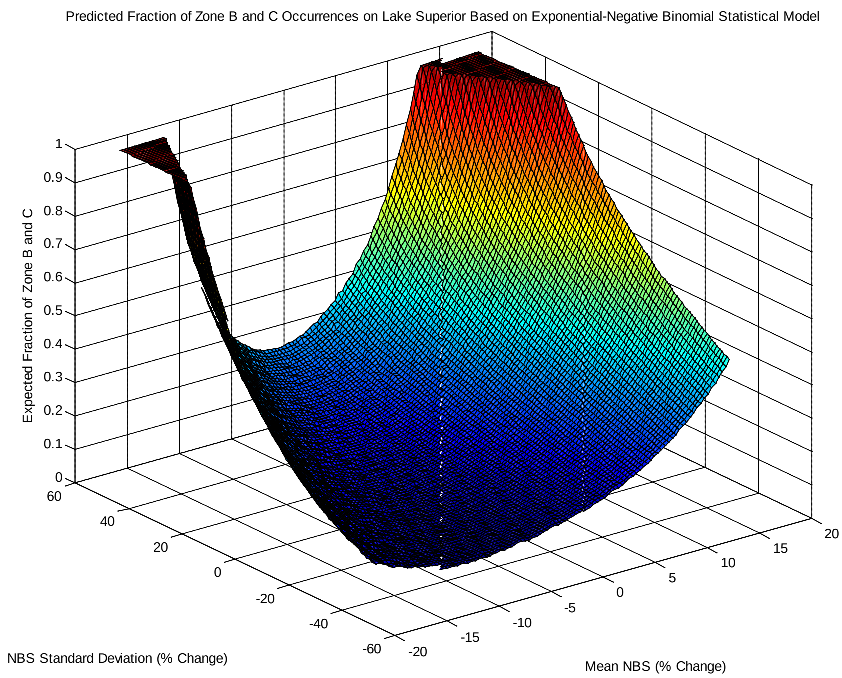


Figure 18: Lake Superior Expected Number of Zones as a Function of Mean NBS and NBS Standard Deviation Based on the Fitted Statistical Model.

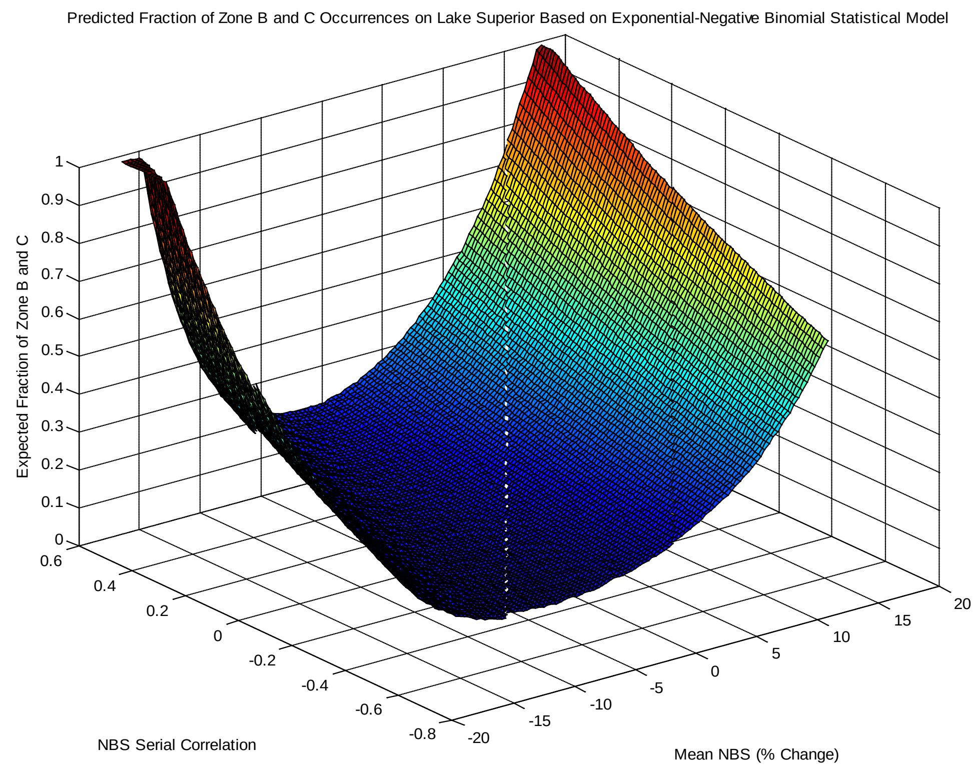


Figure 19: Lake Superior Expected Number of Zones as a Function of Mean NBS and NBS Serial Correlation Based on the Fitted Statistical Model.

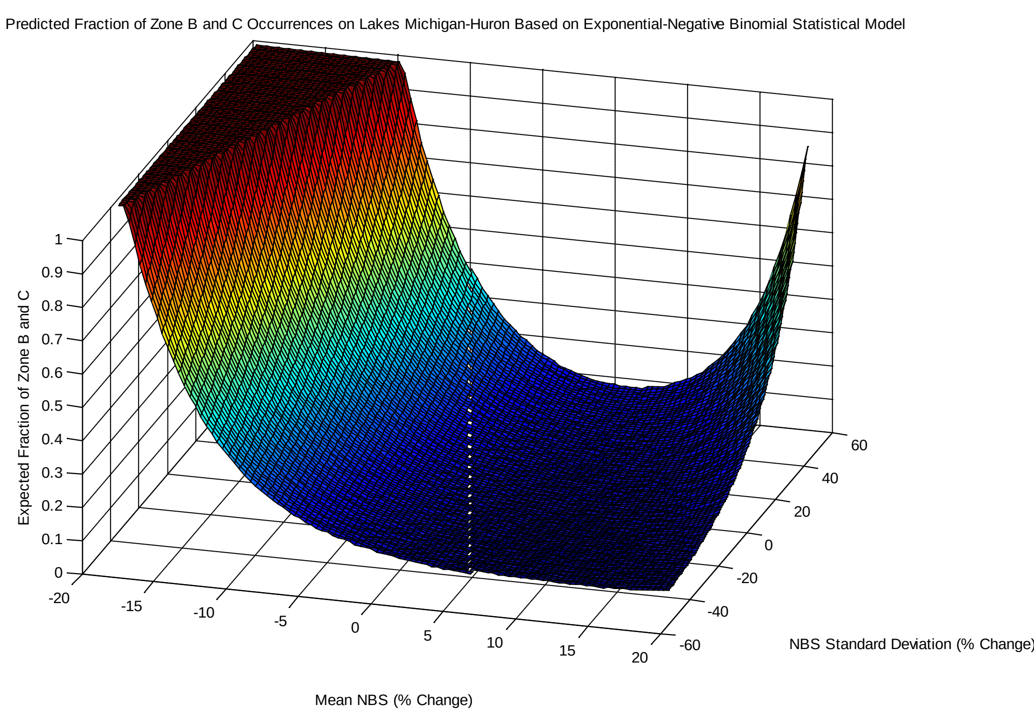


Figure 20: Lakes Michigan-Huron Expected Number of Zones as a Function of Mean NBS and NBS Standard Deviation Based on the Fitted Statistical Model.

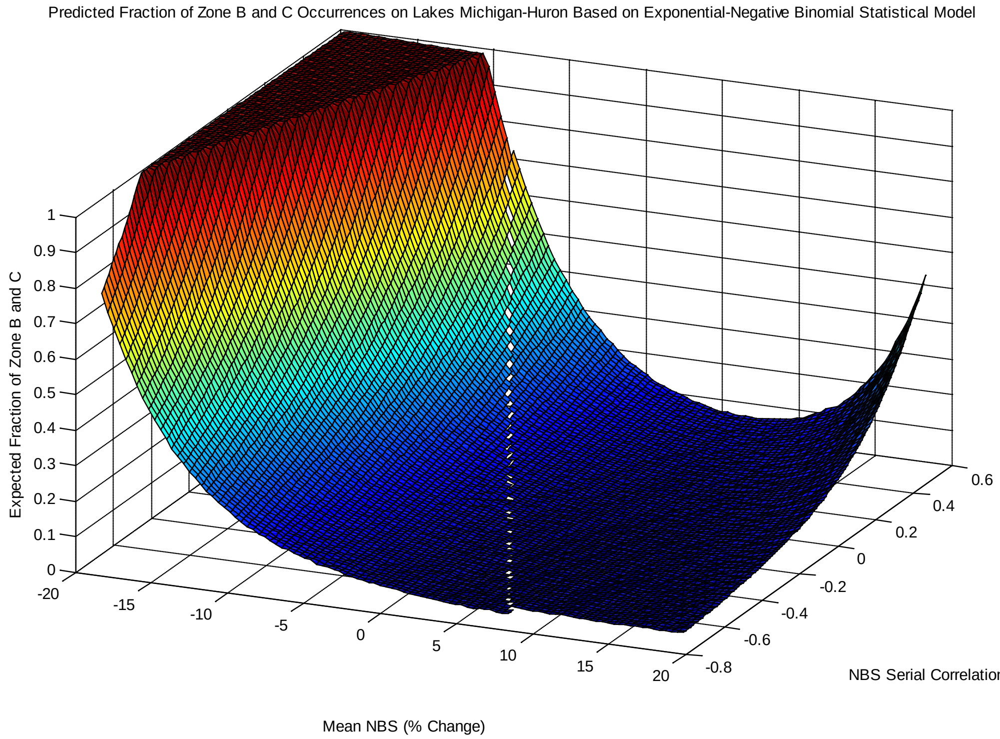


Figure 21: Lakes Michigan-Huron Expected Number of Zones as a Function of Mean NBS and NBS Serial Correlation Based on the Fitted Statistical Model.

Another way to consider the impact of climate on zone occurrences is to look at the histogram and predicted distribution of zone occurrences based on the percent change in mean NBS. Figure 22 shows the histogram and fitted distribution of Lower C Zone occurrences based on the range of mean NBS percent change. The top graph is for a -10% or greater mean NBS change and the bottom graph is for a +10% or greater mean NBS change. The shift in distribution and variability over the range of percent change of mean NBS is clearly evident. Figure 23 shows the same graph for Lower B Zone, Figure 24 shows the graph for Upper B Zone and Figure 25 shows the graph for Upper C Zone. Figure 21 also demonstrates the relatively low occurrence of Upper Zone C. Even at high values for percent change of mean NBS, there are still very few Upper Zone C occurrences.

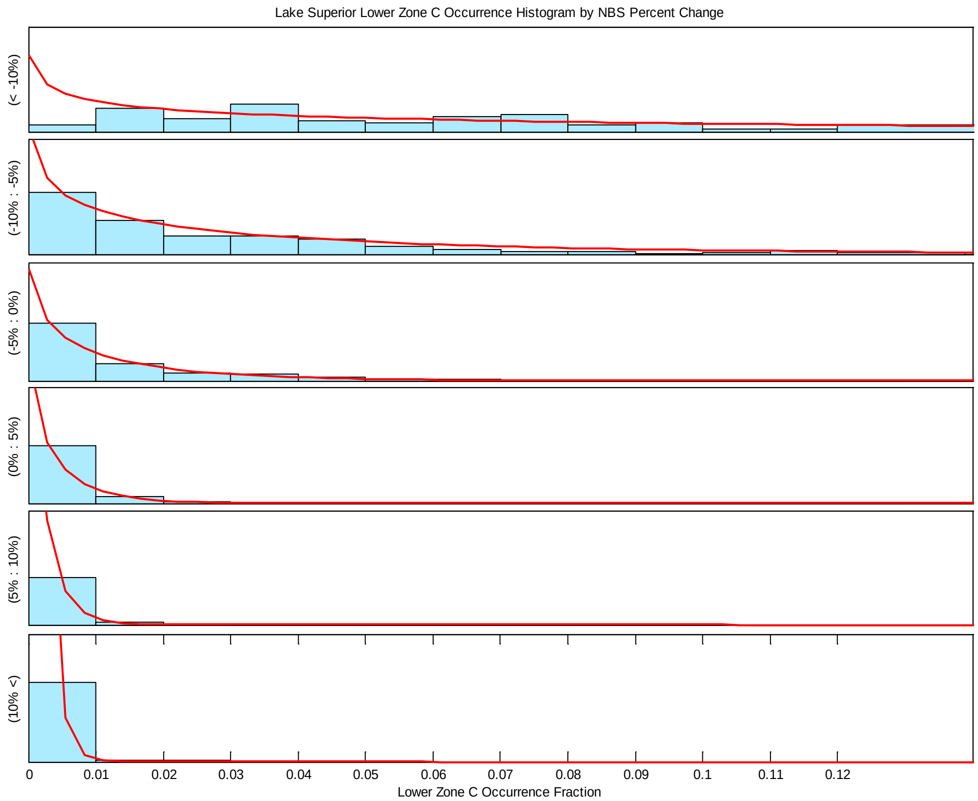


Figure 22: Lake Superior Lower C Zone Occurrence by NBS Percent Change

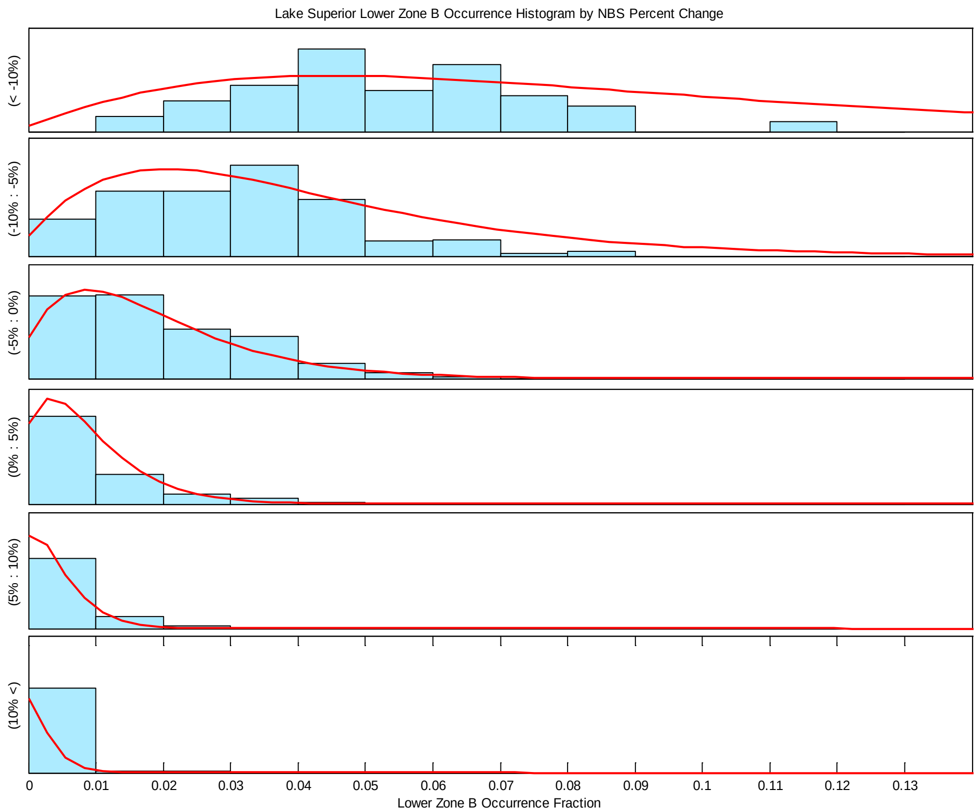


Figure 23: Lake Superior Lower B Zone Occurrence by NBS Percent Change

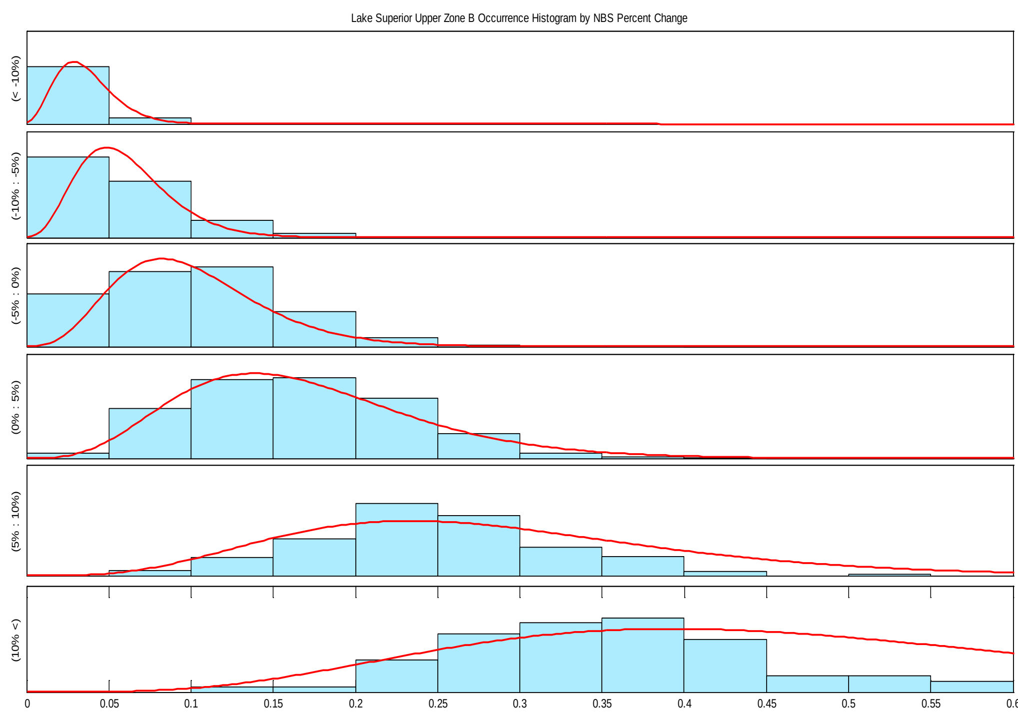


Figure 24: Lake Superior Upper B Zone Occurrence by NBS Percent Change

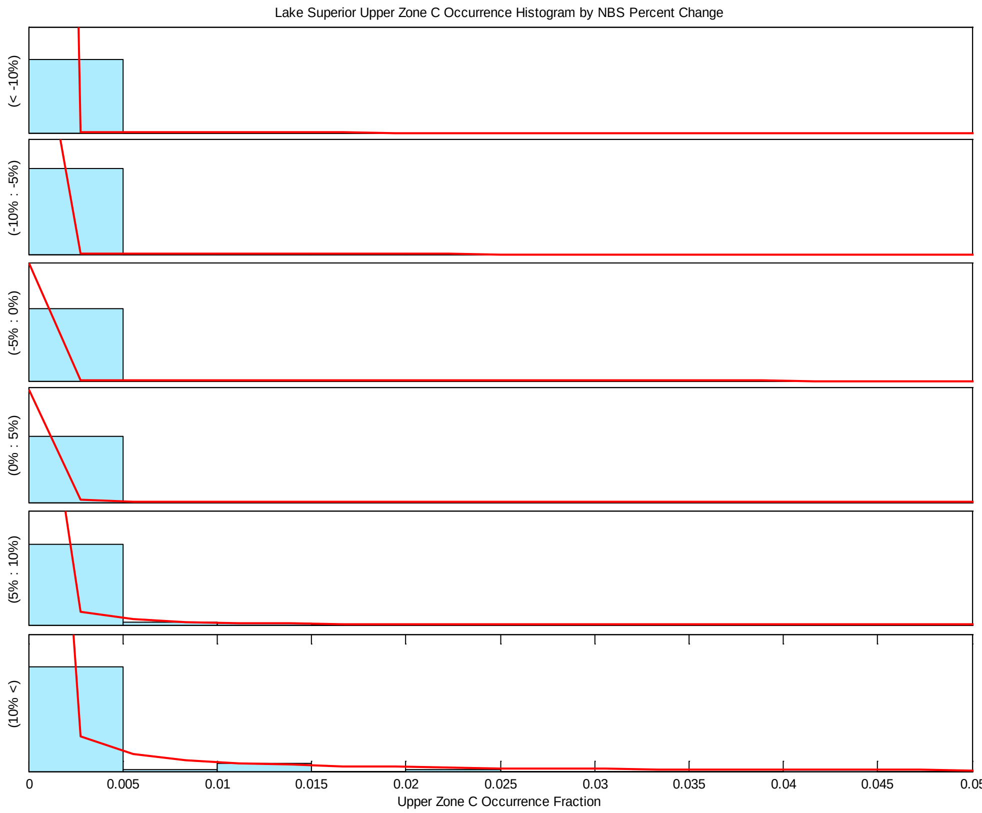


Figure 25: Lake Superior Upper C Zone Occurrence by NBS Percent Change

The relationships developed in the course of this analysis are helpful at identifying climate conditions that result in unfavorable lake conditions. Continued analysis of the GCM based climate projections and the paleo based climate will then help to identify plausible climate conditions. Plausible climate conditions that cause the current operations plan to fail can then be addressed to determine lake management decisions to mitigate the climate effects. The analysis should consider acceptable levels of failure based on zone occurrences.

There are several limitations to the current statistical model. The model treats the zone occurrences as independent and does not constrain the total number of zone occurrences to the number of months in the analysis period. The significance of the limitations increases as the percent change in mean NBS or the percent change in annual NBS standard deviation increases. Unfortunately, this corresponds to the areas in climate space that are the most interesting due to their impact. The UMass Hydrosystems Research Group is examining model alternatives that address these shortcomings.

3.2 Estimation of Climate-informed Plausible Risk

In the application to the Great Lakes, a multi-model, multi-run ensemble of GCM projections is used in combination with stochastically generated timeseries, including those informed by paleodata to describe probabilities. Given the uncertainties associated with the estimation of probabilities, the term "plausibility" has been adopted in its place. The concept of plausibility is best described as a stakeholder developed, subjective ranking of the probability of specific climate states. The concept borrows from the practice of shared vision modeling, in that the estimation of probabilities is not a black box process, but rather a tool for discussion and ranking relative uncertainties during the planning process.

The plausibility of a climate state is generally based on the frequency of occurrence of that state in the climate simulations. In addition, the source of the simulation is considered. For example, climate state that occurs in many runs from multiple GCMs and also occurs in the paleodata-based stochastic simulation is more plausible than a climate state that occurs rarely in a small number of sources. Where the relative plausibility is less clear cut, a discussion of the different sources of the occurrence of the climate states (e.g., specific GCMs) and relative merits of those sources can be discussed among the decision makers (Board members, other experts) facilitated by the AM and PFEG teams. The goal is to use a wide range of climate information in a transparent manner to facilitate comfort for the decision makers in the use of that information for decisions. The decision makers, in this case the Study Board, will be presented with plausibility estimates of climate states associated with each regulation plan and the sources of information that assigned probability to that state. The plausibility estimates may be adjusted based on different comfort levels of the Board members with the various climate information sources.

Access to 160 GCM runs, taken from 18 different models originated all across the globe, was made possible with the help of Dr. James Angel and Dr. Kenneth Kunkel. All 160 GCM runs, of 30yrs windows each (centered around 2050), were compared to a GCM Base Case (centered around 1985) to get the Mean NBS Percent Change suggested by these GCMs' models. All runs are established under the emission scenario A_2.

$$MeanNBS\%Change = \frac{\frac{1}{n}\sum_{i=1}^{n} X_{GCM_i} - \frac{1}{n}\sum_{i=1}^{n} X_{Base70-99_i}}{\frac{1}{n}\sum_{i=1}^{n} X_{Base70-99_i}} * 100\%$$

Where, X_{GCMi} = NBS monthly values for GCM runs centered around 2050 ; $X_{Base70\text{-}99i}$ = NBS monthly values for GCM Base Case (centered around 1985) ; n = the number of months in each timeseries; *MeanNBS%Change* = change in NBS, proposed by the GCM runs

A histogram is created to show the models' suggested changes. Figure 26 shows a range for NBS changes at Lake Superior of -28% to 19%, and at Lake Michigan-Huron from -28% to 24%. Distribution for mean NBS percent changes seem to follow a normal distribution, with the range falling outside of the range seen in the 50k stochastic set, meaning that GCMs are suggesting larger extreme scenarios than what the stochastic model is. Now, a study on model consistency and an examination of any discrepancy among the models is completed as followed.

When analyzing plausibility proposed by GCMs, a consideration of the models' resolution is warranted. Previous results had shown a fairly robust difference in the climate change signal for NBS based on models considered to be more or less coarse in resolution. To investigate the effect of resolution on plausible climate changes inferred from the GCMs, conditional histograms of the climate changes from fine resolution models (smaller than 2.5° grids) and from coarse resolution models (bigger than 2.5° grids) were created. Histograms in figure 27 and 28 show the distributions of mean NBS percent changes for both resolution categories. There is a clear disparity in the suggested mean NBS percent changes between the fine resolution models and the coarse resolution models. The reason for such inconsistency is still unknown. The processes by which these have been downscaled, or perhaps the particular source for each model should be considered suspects for such inconsistencies.

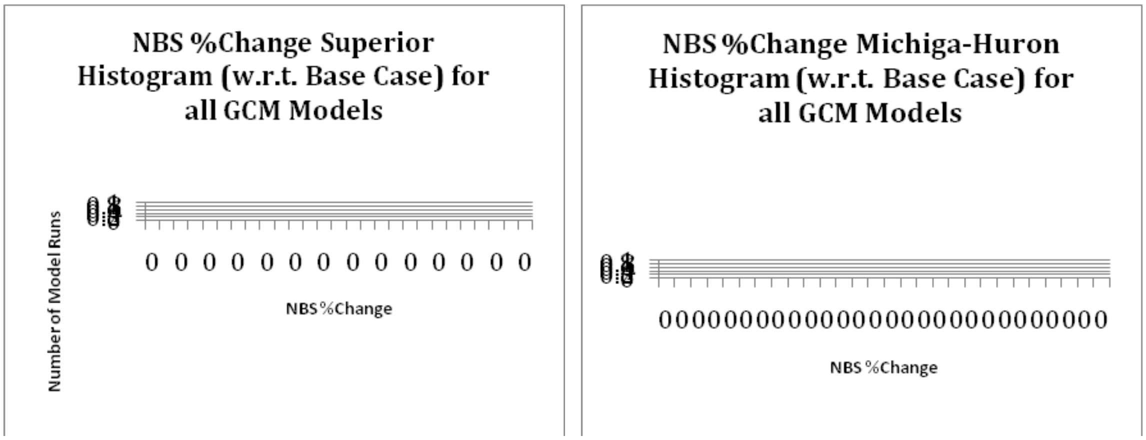


Figure 26: Histogram of NBS %Changes for all GCM Models

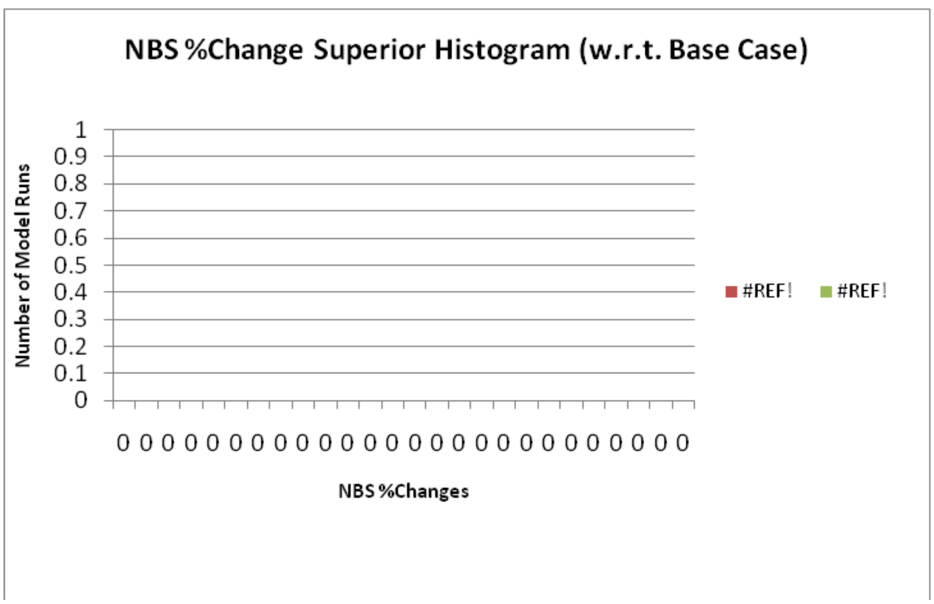


Figure 27: Lake Superior's Histogram of NBS %Changes for all GCM Models, split up into Fine and Coarse Resolution

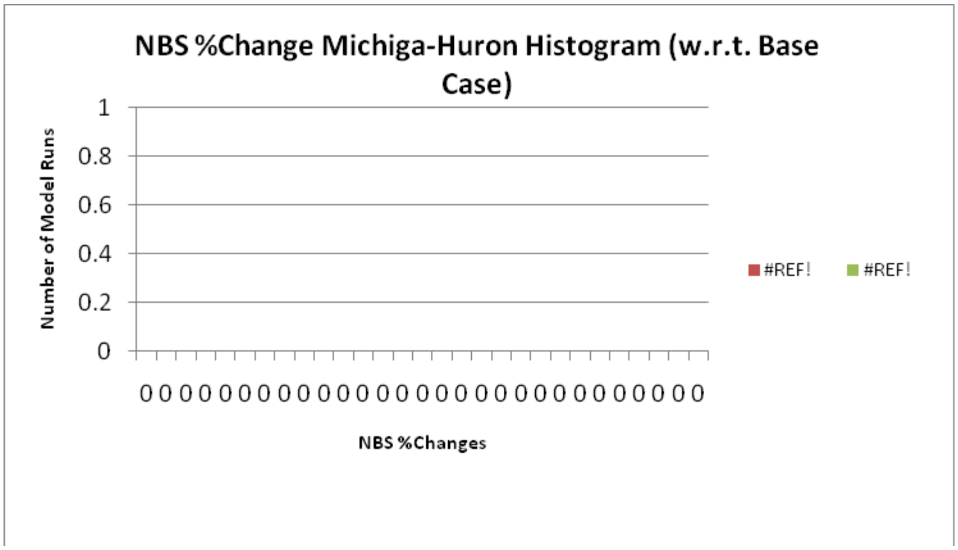


Figure 28: Lake Michigan-Huron's Histogram of NBS %Changes for all GCM Models, split up into Fine and Coarse Resolution

For a better understanding of what these models are actually proposing, the data is then fitted into a normal distribution. This fitting is done for three different datasets: All GCMs, Fine Resolution GCMs, and Coarse Resolution GCMs. The mean value and the standard deviation are calculated for each dataset, and based on these numbers a normal distribution is then created:

$$\mu_{dataset} = \frac{1}{n}\sum_{i=1}^{n} X_i$$

where, $\mu_{dataset}$ = mean value for dataset; X_{seti} = mean NBS percent change for each run

$$\sigma_{dataset} = \sqrt{\frac{1}{n}\sum_{i=1}^{n}(X_i - \mu_{dataset})^2}$$

where, $\sigma_{dataset}$ = standard deviation for dataset.

Tables 3 and 4 present the normal distribution values for each of the datasets, highlighting four specific mean NBS percent changes (-10, -5, 5, 10), and their respective probabilities. The exceedence probability is the probability of a variable being exceeded given the statistical distribution, while the cumulative probability is the probability of a variable being greater than or equal to a random sample from the same distribution. The probability density function (PDF) is also calculated and placed into a column next to the exceedance probability. Figures 25 and 26 illustrate the PDF curve for each of the GCM datasets.

Table 3: Lake Superior's Exceedence probabilities for the normally distributed series focused in the -10%, -5%, 5%, 10% NBS Change

Base Case Normal Distribution - ALL MODELS				
Mean	-4.771	n = 160	95% Confidence Internval	
Stand Dev	9.332		0.025	0.975
			-23.062	13.5192
% Base Change	Cumm. Probability	Excedence	PDF	
-30	0.003	0.997	0.001	
-28	0.006	0.994	0.002	
-26	0.011	0.989	0.003	
-24	0.020	0.980	0.005	
-22	0.032	0.968	0.008	
-20	0.051	0.949	0.011	
-18	0.078	0.922	0.016	
-16	0.114	0.886	0.021	
-14	0.161	0.839	0.026	
-12	0.219	0.781	0.032	
-10	0.288	0.712	0.037	
-8	0.365	0.635	0.040	
-6	0.448	0.552	0.042	
-5	0.490	0.510	0.043	
-4	0.533	0.467	0.043	
-2	0.617	0.383	0.041	
0	0.695	0.305	0.038	
2	0.766	0.234	0.033	
4	0.826	0.174	0.027	
5	0.852	0.148	0.025	
6	0.876	0.124	0.022	
8	0.914	0.086	0.017	
10	0.943	0.057	0.012	
12	0.964	0.036	0.009	
14	0.978	0.022	0.006	
16	0.987	0.013	0.004	
18	0.993	0.007	0.002	
20	0.996	0.004	0.001	

Base Case Normal Distribution - FINE RESOLUTION				
Mean	-10.191	n = 63	95% Confidence Internval	
Stand Dev	8.135		0.025	0.975
			-26.1366	5.75392
% Base Change	Cumm. Probability	Excedence	PDF	
-30	0.007	0.993	0.003	
-28	0.014	0.986	0.004	
-26	0.026	0.974	0.007	
-24	0.045	0.955	0.012	
-22	0.073	0.927	0.017	
-20	0.114	0.886	0.024	
-18	0.169	0.831	0.031	
-16	0.238	0.762	0.038	
-14	0.320	0.680	0.044	
-12	0.412	0.588	0.048	
-10	0.509	0.491	0.049	
-8	0.606	0.394	0.047	
-6	0.697	0.303	0.043	
-5	0.738	0.262	0.040	
-4	0.777	0.223	0.037	
-2	0.843	0.157	0.030	
0	0.895	0.105	0.022	
2	0.933	0.067	0.016	
4	0.959	0.041	0.011	
5	0.969	0.031	0.009	
6	0.977	0.023	0.007	
8	0.987	0.013	0.004	
10	0.993	0.007	0.002	
12	0.997	0.003	0.001	

Base Case Normal Distribution - COARSE RESOLUTION				
Mean	-0.164	n = 82	95% Confidence Internval	
Stand Dev	8.366		0.025	0.975
			-16.559925	16.2327
% Base Change	Cumm. Probability	Excedence	PDF	
-30	0.000	1.000	0.000	
-28	0.000	1.000	0.000	
-26	0.001	0.999	0.000	
-24	0.002	0.998	0.001	
-22	0.005	0.995	0.002	
-20	0.009	0.991	0.003	
-18	0.016	0.984	0.005	
-16	0.029	0.971	0.008	
-14	0.049	0.951	0.012	
-12	0.079	0.921	0.018	
-10	0.120	0.880	0.024	
-8	0.174	0.826	0.031	
-6	0.243	0.757	0.037	
-5	0.282	0.718	0.040	
-4	0.323	0.677	0.043	
-2	0.413	0.587	0.047	
0	0.508	0.492	0.048	
2	0.602	0.398	0.046	
4	0.691	0.309	0.042	
5	0.731	0.269	0.039	
6	0.769	0.231	0.036	
8	0.835	0.165	0.030	
10	0.888	0.112	0.023	
12	0.927	0.073	0.017	
14	0.955	0.045	0.011	
16	0.973	0.027	0.007	
18	0.985	0.015	0.005	
20	0.992	0.008	0.003	

Table 4: Lake Michigan-Huron's Exceedence probabilities for the normally distributed series focused in the -10%, -5%, 5%, 10% NBS Change

Base Case Normal Distribution - ALL MODELS				
Mean	-5.508	n = 160	95% Confidence Internval	
Stand Dev	9.788		0.025	0.975
			-24.692	13.676
% Base Change	Cumm. Probability	Excedence	PDF	
-30	0.006	0.994	0.002	
-28	0.011	0.989	0.003	
-26	0.018	0.982	0.005	
-24	0.029	0.971	0.007	
-22	0.046	0.954	0.010	
-20	0.069	0.931	0.014	
-18	0.101	0.899	0.018	
-16	0.142	0.858	0.023	
-14	0.193	0.807	0.028	
-12	0.254	0.746	0.033	
-10	0.323	0.677	0.037	
-8	0.400	0.600	0.039	
-6	0.480	0.520	0.041	
-5	0.521	0.479	0.041	
-4	0.561	0.439	0.040	
-2	0.640	0.360	0.038	
0	0.713	0.287	0.035	
2	0.778	0.222	0.030	
4	0.834	0.166	0.025	
5	0.858	0.142	0.023	
6	0.880	0.120	0.020	
8	0.916	0.084	0.016	
10	0.943	0.057	0.012	
12	0.963	0.037	0.008	
14	0.977	0.023	0.006	
16	0.986	0.014	0.004	
18	0.992	0.008	0.002	
20	0.995	0.005	0.001	

Base Case Normal Distribution - FINE RESOLUTION				
Mean	-9.344	n = 63	95% Confidence Internval	
Stand Dev	9.721		0.025	0.975
			-28.397	9.709
% Base Change	Cumm. Probability	Excedence	PDF	
-30	0.017	0.983	0.004	
-28	0.027	0.973	0.007	
-26	0.043	0.957	0.009	
-24	0.066	0.934	0.013	
-22	0.096	0.904	0.018	
-20	0.136	0.864	0.023	
-18	0.187	0.813	0.028	
-16	0.247	0.753	0.032	
-14	0.316	0.684	0.037	
-12	0.392	0.608	0.040	
-10	0.473	0.527	0.041	
-8	0.555	0.445	0.041	
-6	0.635	0.365	0.039	
-5	0.673	0.327	0.037	
-4	0.709	0.291	0.035	
-2	0.775	0.225	0.031	
0	0.832	0.168	0.026	
2	0.878	0.122	0.021	
4	0.915	0.085	0.016	
5	0.930	0.070	0.014	
6	0.943	0.057	0.012	
8	0.963	0.037	0.008	
10	0.977	0.023	0.006	
12	0.986	0.014	0.004	
14	0.992	0.008	0.002	
16	0.995	0.005	0.001	

Base Case Normal Distribution - COARSE RESOLUTION				
Mean	-1.791	n = 82	95% Confidence Internval	
Stand Dev	9.013		0.025	0.975
			-19.457	15.874
% Base Change	Cumm. Probability	Excedence	PDF	
-30	0.001	0.999	0.000	
-28	0.002	0.998	0.001	
-26	0.004	0.996	0.001	
-24	0.007	0.993	0.002	
-22	0.012	0.988	0.004	
-20	0.022	0.978	0.006	
-18	0.036	0.964	0.009	
-16	0.057	0.943	0.013	
-14	0.088	0.912	0.018	
-12	0.129	0.871	0.023	
-10	0.181	0.819	0.029	
-8	0.245	0.755	0.035	
-6	0.320	0.680	0.040	
-5	0.361	0.639	0.042	
-4	0.403	0.597	0.043	
-2	0.491	0.509	0.044	
0	0.579	0.421	0.043	
2	0.663	0.337	0.041	
4	0.740	0.260	0.036	
5	0.774	0.226	0.033	
6	0.806	0.194	0.030	
8	0.861	0.139	0.025	
10	0.905	0.095	0.019	
12	0.937	0.063	0.014	
14	0.960	0.040	0.010	
16	0.976	0.024	0.006	
18	0.986	0.014	0.004	
20	0.992	0.008	0.002	
22	0.996	0.004	0.001	
24	0.998	0.002	0.001	
26	0.999	0.001	0.000	

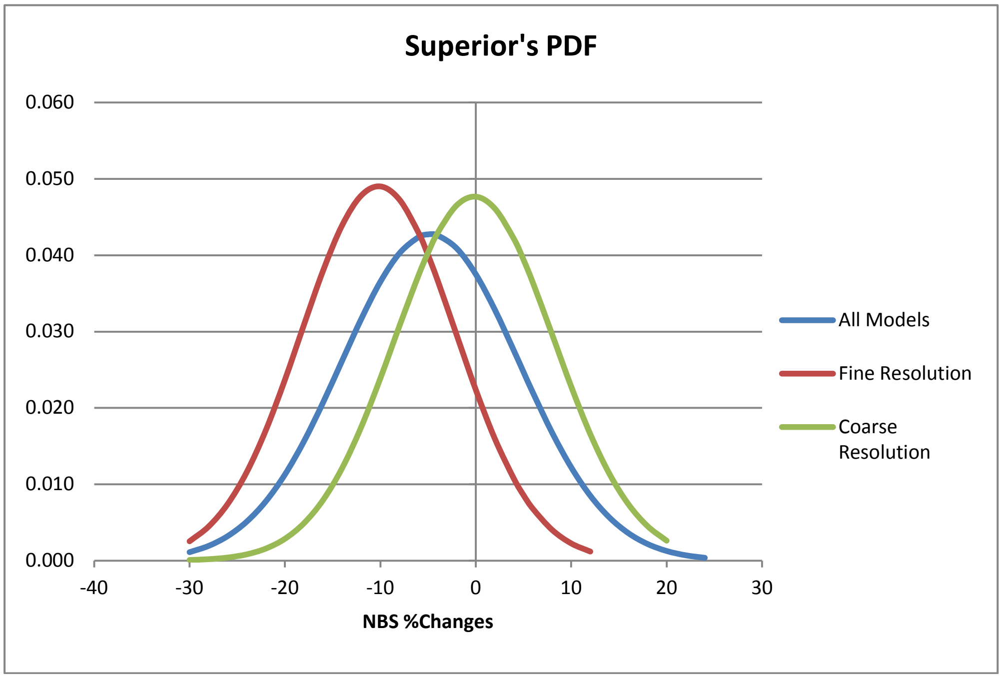


Figure 29: Lake Superior's Normal Distribution graph for all GCM models, for Fine Resolution models, and for Coarse Resolution models

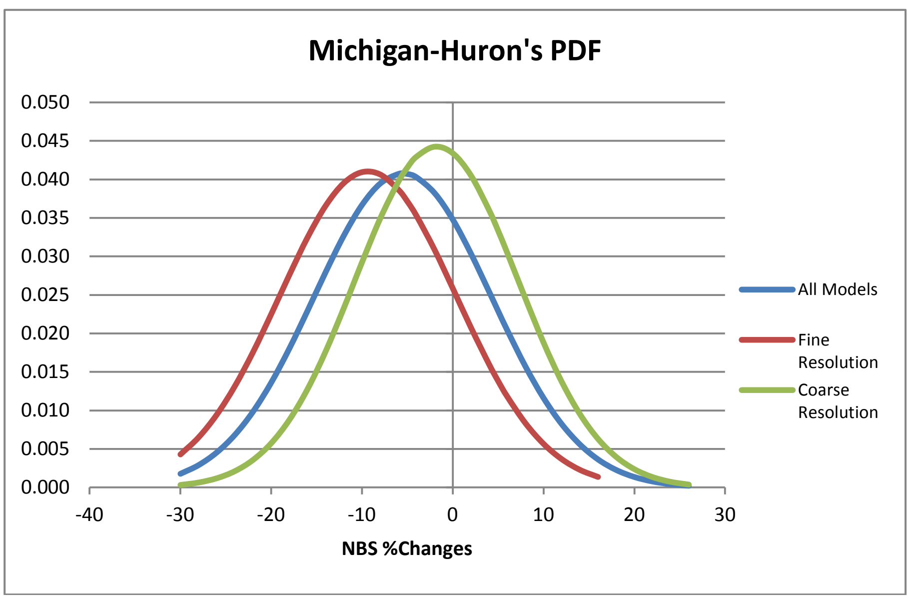


Figure 30: Lake Michigan-Huron's Normal Distribution graph for all GCM models, for Fine Resolution models, and for Coarse Resolution models

The T-test is a statistical hypothesis test appropriate for determining if two data sets are sampled from the same population. The null hypothesis is that the data sets are from the same population while the alternate is that they are from different populations with different means. With a P value of less than 0.05, the null hypothesis can be rejected with a 95% confidence. This means that with a 95% confidence it can be said that the two datasets in question are in fact not coming from the same population. Tables 5 and 6 exhibit some P values for different pairs of datasets, all coming from within the 160 GCM runs. The implications are that the coarse resolution and fine resolution results are statistically significantly different. However, it is unclear if the differences are due to the GCMs themselves, or due to differences in downscaling methods that were applied to the different models by Angel and Kunkel.

Table 5: Lake Superior's P-values for T-test

P values for Lake Superior's T-TEST						
Models	All	Fine Res.	Coarse Res.	European	American	Aus/Asian
All		3.4E-05	1.4E-04	0.408	0.982	0.208
Fine Res.	3.4E-05		2.6E-11			
Coarse Res.	1.4E-04	2.6E-11				
European	0.408				0.442	0.131
American	0.982			0.442		0.256
Aus/Asian	0.208			0.131	0.256	

P < 0.05 'Can regret Null Hypothesis at 95% confidence
P > 0.05 'Can not regret Null Hypothesis at 95% confidence

Table 6: Lake Michigan-Huron's P-values for T-test

P values for Lake Mich-Huron's T-TEST						
Models	All	Fine Res.	Coarse Res.	European	American	Aus/Asian
All		9.2E-03	3.6E-03	0.905	0.176	0.210
Fine Res.	9.2E-03		4.6E-06			
Coarse Res.	3.6E-03	4.6E-06				
European	0.905				0.503	0.564
American	0.176			0.503		0.040
Aus/Asian	0.210			0.564	0.040	

P < 0.05 'Can regret Null Hypothesis at 95% confidence
P > 0.05 'Can not regret Null Hypothesis at 95% confidence

It is useful to estimate climate impact from the GCM series to help establish risk. Using the Climate Response Function developed in our group, one can translate GCM climate statistics into lake impacts. The Climate Response Function requires three inputs: mean NBS percent change, annual standard deviation percent change, and annual serial correlation. The Climate response function was generated using percent change from the historic NBS series. To evaluate the GCMs using the Climate Response Function, a bias correction needs to be applied. This corrects the GCM percent change from the GCM base case to the historic NBS case using the historic NBS window corresponding to the base case. The following adjustment was applied to the mean NBS percent change, and to the standard deviation percent change:

$$\Delta GCM_H\% = \frac{X_{GCM} - [X_{Base70-99} - (X_{Hist70-99} - X_{Hist})]}{X_{Base70-99} - (X_{Hist70-99} - X_{Hist})} * 100\%$$

where, X_{GCM} = GCM statistic (centered around 2050) ; $X_{Base70-99}$ = GCM statistic for Base Case (centered around 1985) ; $X_{Hist70-99}$ = statistic for historic data (from 1970-99) ; X_{Hist} = statistic for historic data (from 1900-2006) ; $\Delta GCM_H\%$ = statistic percentage change suggested by GCM with respect to historic values. With the adjustment to the mean and standard deviation, the statistics from the 160 GCM runs can be used in the Climate Response Function. An initial attempt at this has been completed and the figures are shown in Figures 31 to 34 for Lakes Superior and Michigan-Huron.

Next Steps: The next step is to calculate probabilities associated with the risk levels through use of the climate response function with the various sources of climate information. We will also investigate the different approaches to weighting different sources. That work is currently being accomplished.

Task 4. Adaptive Management

The use of a dynamic regulation plan is envisioned to produce a robust regulation strategy for a broad range of future climates. However, it is well known that there are other uncertainties, including faulty assumptions and unforeseen surprises, which threaten the success of the regulation plan. For this reason, an adaptive management process is being incorporated into the regulation of Lake Superior. The process consists of long term monitoring of regulation plan performance and mechanisms for implementing changes when needed. Figure C1 in the Appendix illustrates the historical approach to management of Lake Superior regulation in comparison to the proposed adaptive management strategy (Figure C2).

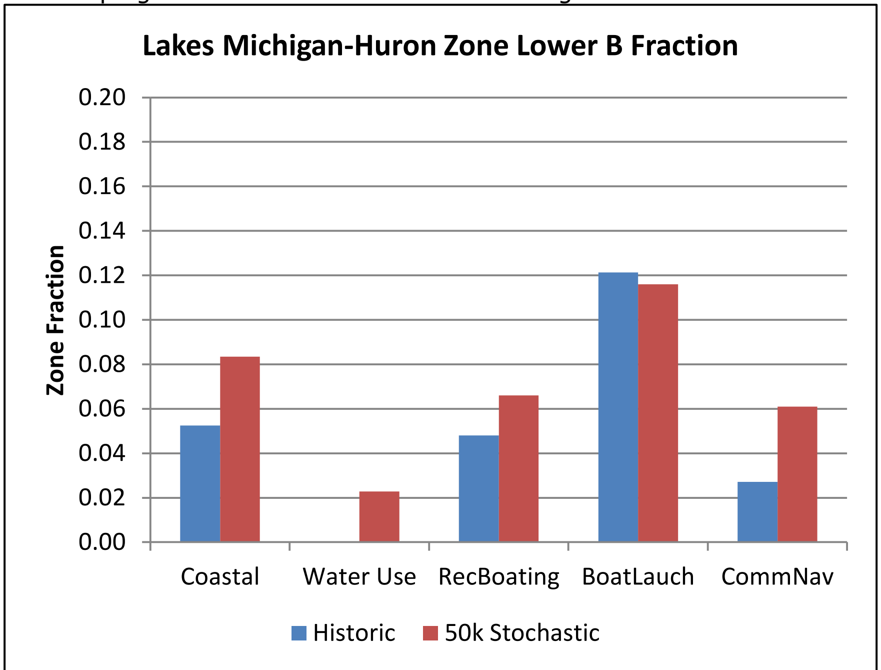
For any adaptive management process, monitoring is critical. The data gathered through carefully designed monitoring allows evaluation of the performance of the regulation plan and the need for changes, including regulation rule changes, changes to plan objectives or other possibilities that we cannot anticipate. The observations will provide direct feedback on plan performance. In addition, monitoring will be designed to evaluate the degree to which the coping zones are effective in estimating plan performance. Since there is uncertainty in the estimation of the coping zones by the working groups, it is possible that significant negative impacts may be accumulating for a stakeholder group despite lake levels remaining out of zone C. Adjustment to the zones themselves may be necessary.

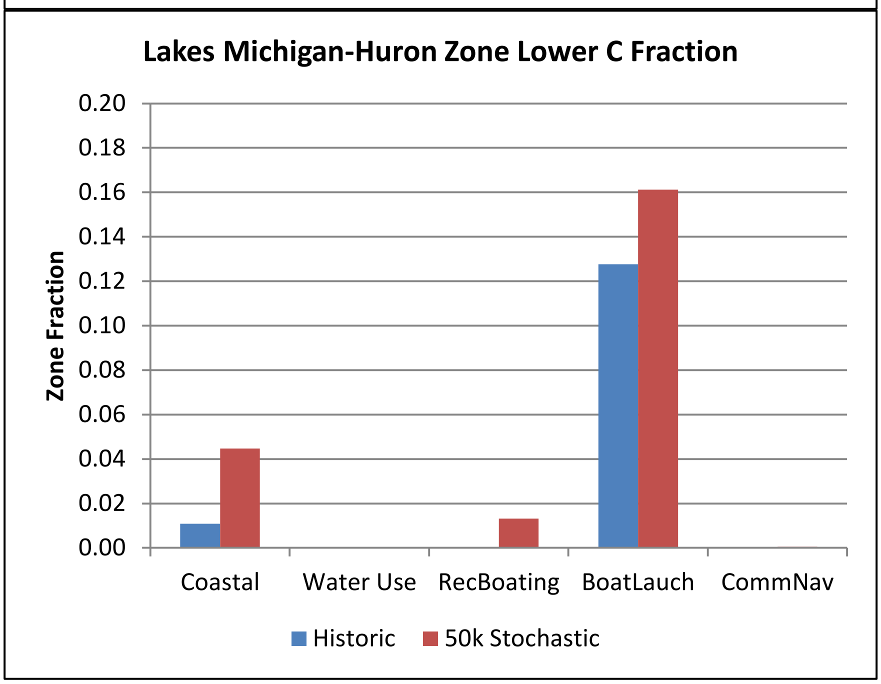
In order to sustain monitoring and provide mechanisms for use of the collected data in decision making, an institutional framework for the adaptive management process is required. Previous studies have shown that adaptive management often fails do to a lack of institutional buy in (Walters, 2007; Allan and Allan, 2005). The IUGLS study board is committed to implementing an adaptive management process. The AM leadership is conducting an institutional analysis that will investigate how the process will be funded, who would be responsible for each element of the plan, and how decisions will be made and implemented. The study board will recommend adaptive management to the IJC, and the common assumption is that a number of U.S. and Canadian agencies would agree to carry out different elements of the plan. This will not guarantee that adaptive management will occur even if these tasks are done well. But the adaptive management process has been designed to improve the odds of successful implementation.

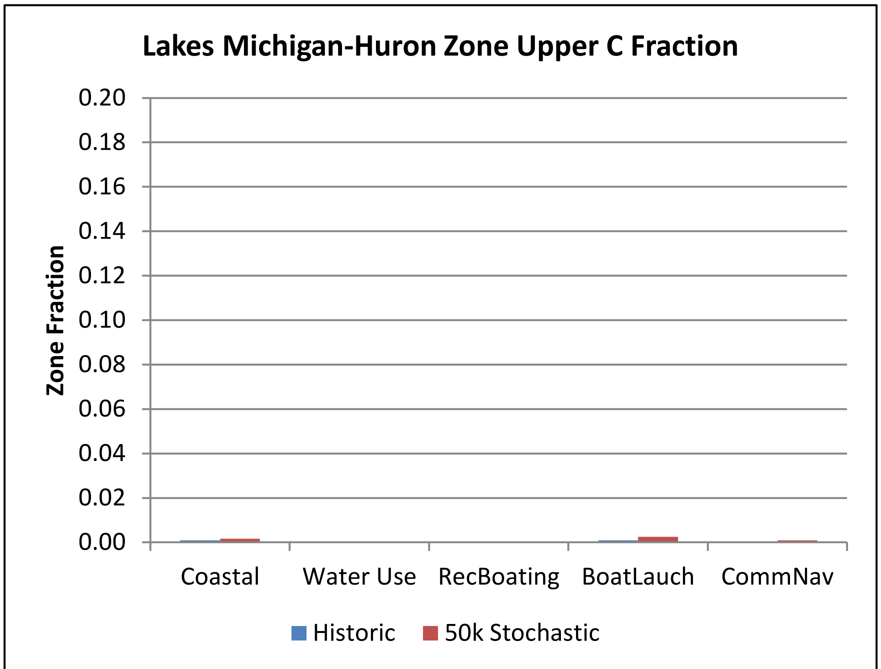
Next Steps: Support the AM leadership with results, especially estimates of plausible risk and risk beyond regulation, that can be used to build the institutional case for the need for adaptive management and to identify and enlist partners in the effort to improve the adaptive capacity of the Great Lakes stakeholders and affected communities.

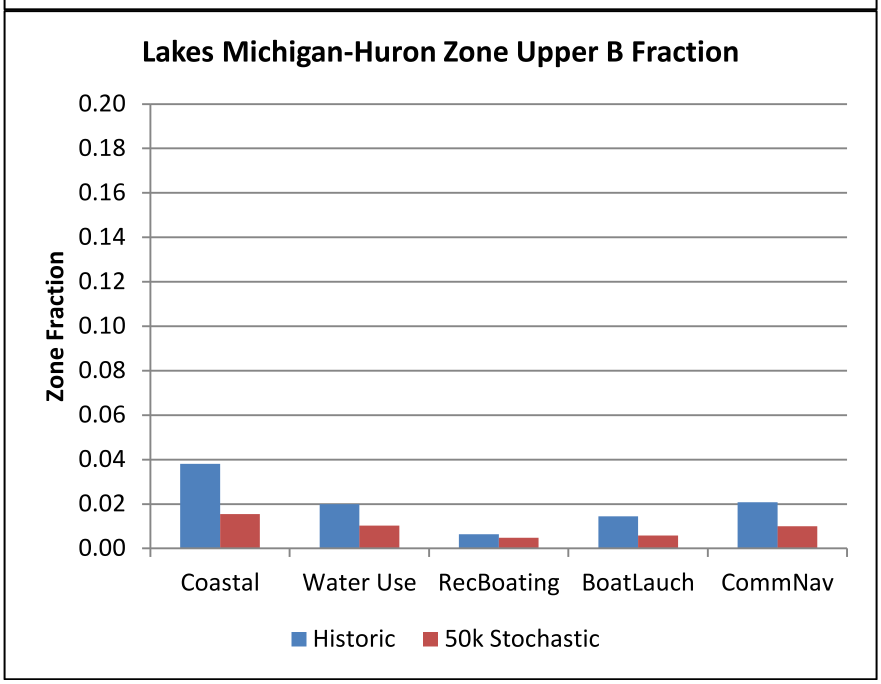
APPENDIX

Figures A1 - A4. Coping Zone occurrences on Lakes Michigan-Huron.

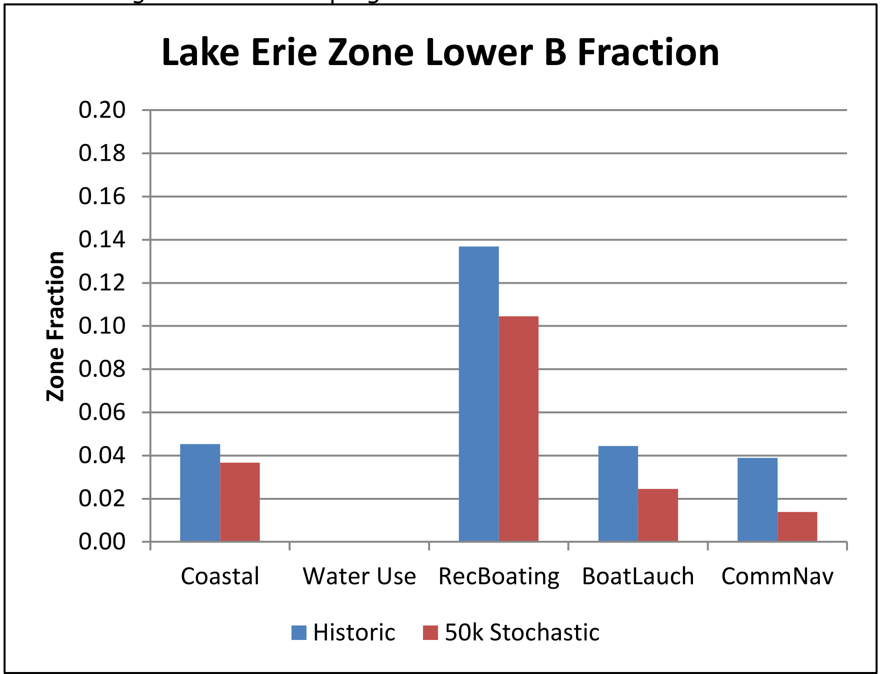


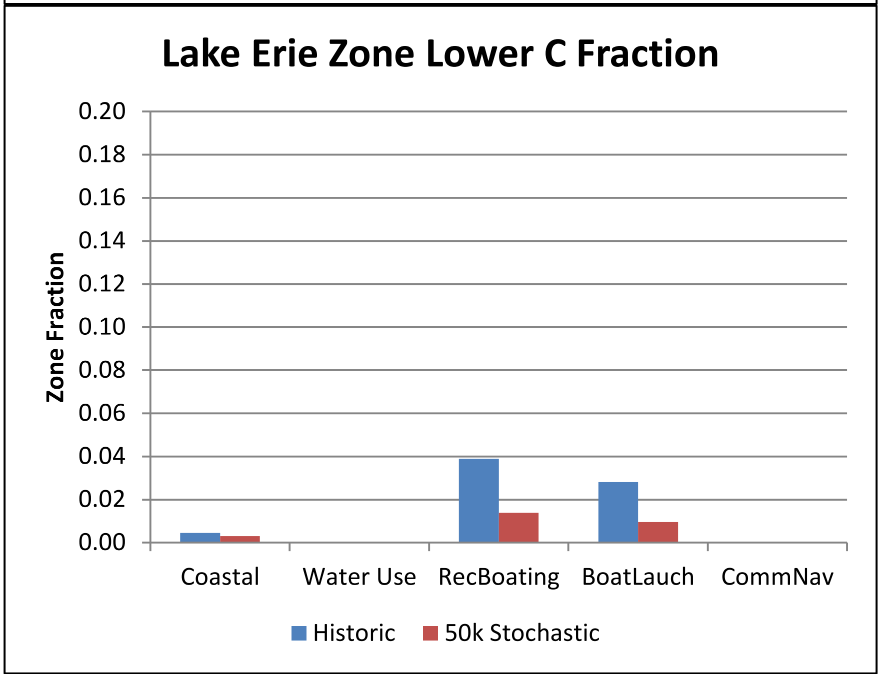


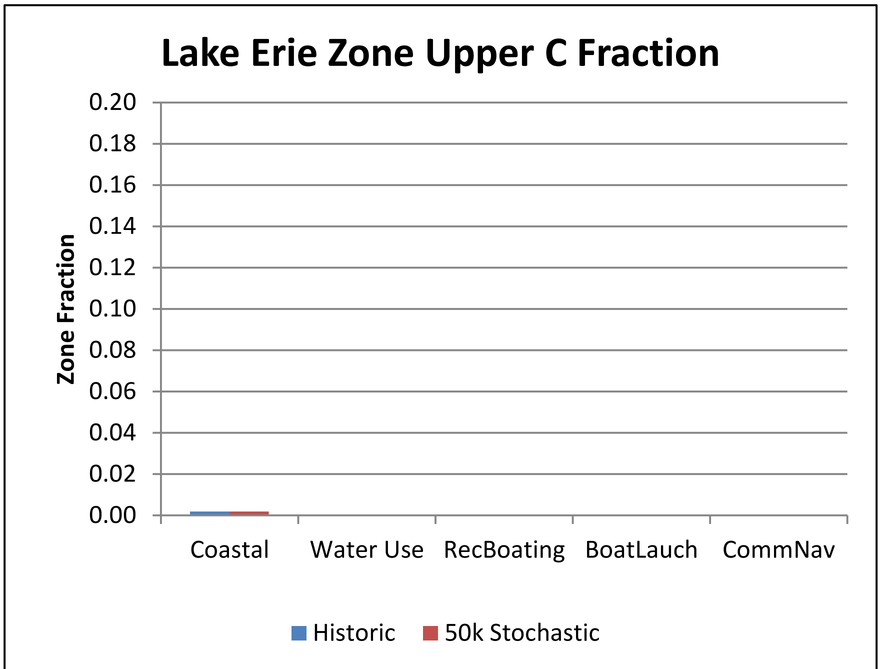

Lakes Michigan-Huron Zone Upper C Fraction
Zone Fraction
0.20
0.18
0.16
0.14
0.12
0.10
0.08
0.06
0.04
0.02
0.00
Coastal
Water Use
RecBoating
BoatLauch
CommNav
Historic
50k Stochastic

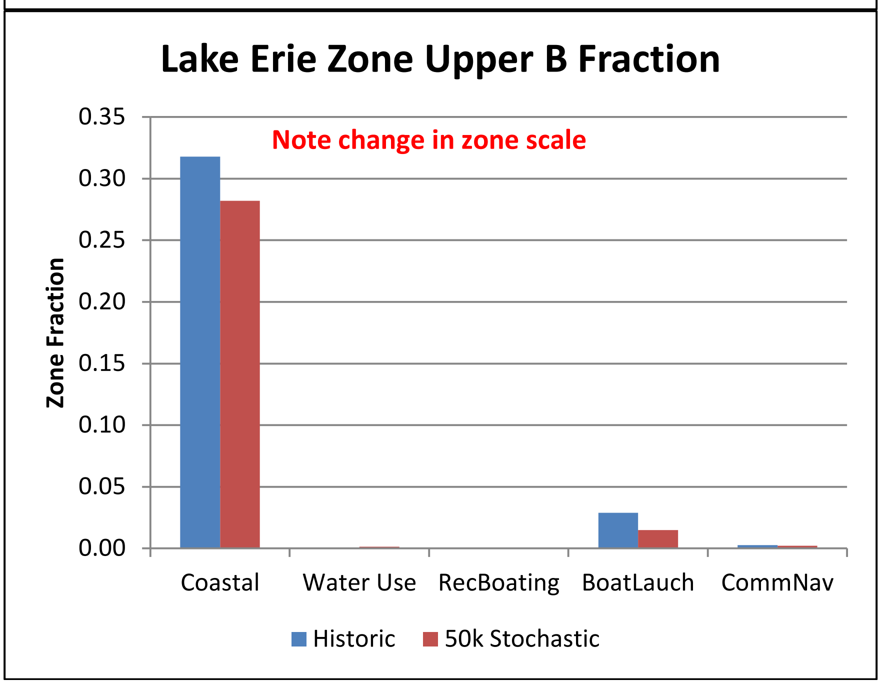

Lakes Michigan-Huron Zone Upper B Fraction
Zone Fraction
0.20
0.18
0.16
0.14
0.12
0.10
0.08
0.06
0.04
0.02
0.00
Coastal
Water Use
RecBoating
BoatLauch
CommNav
Historic
50k Stochastic

Figures A5-A8. Coping Zone occurrences on Lake Erie.

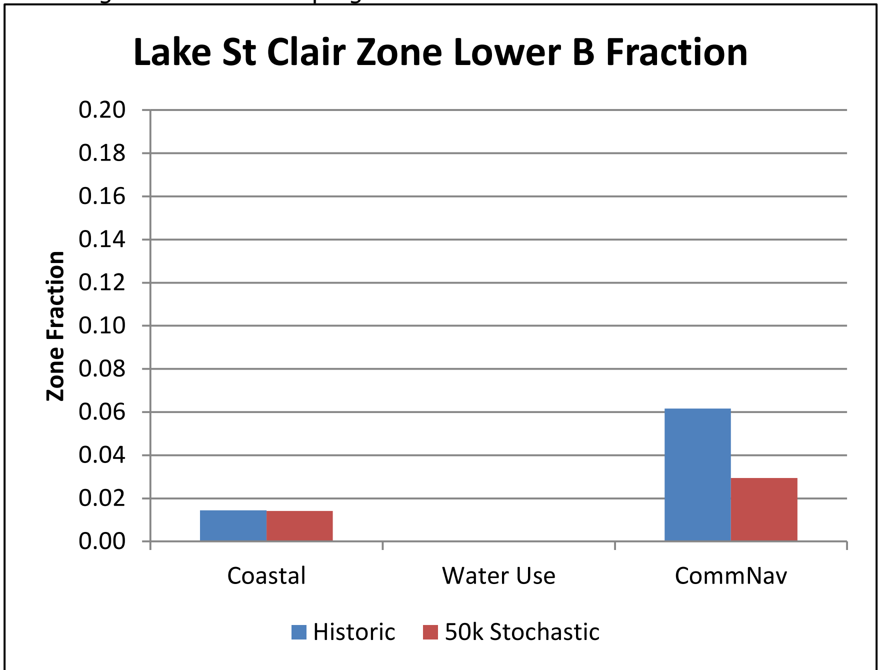


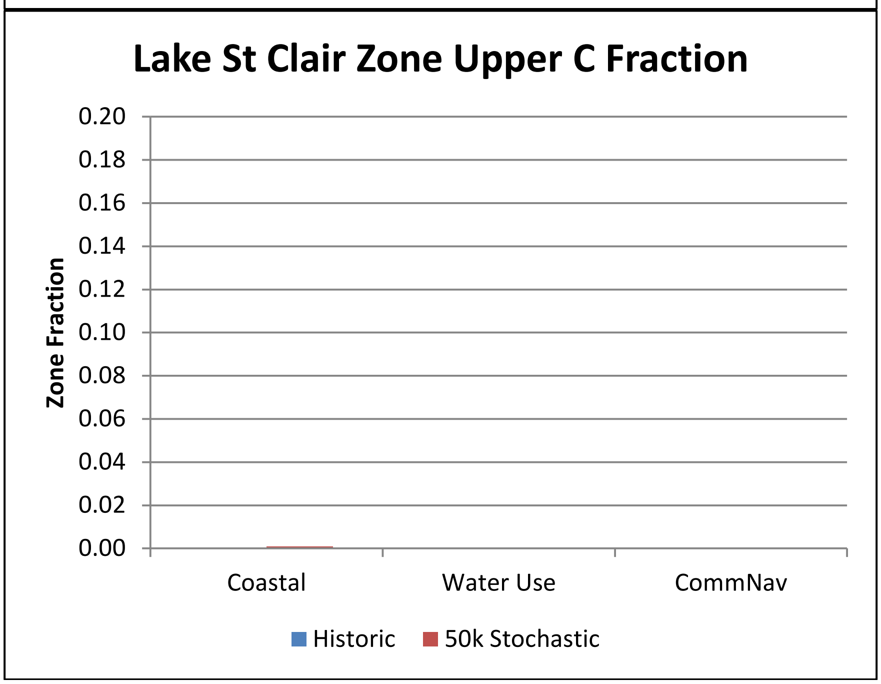


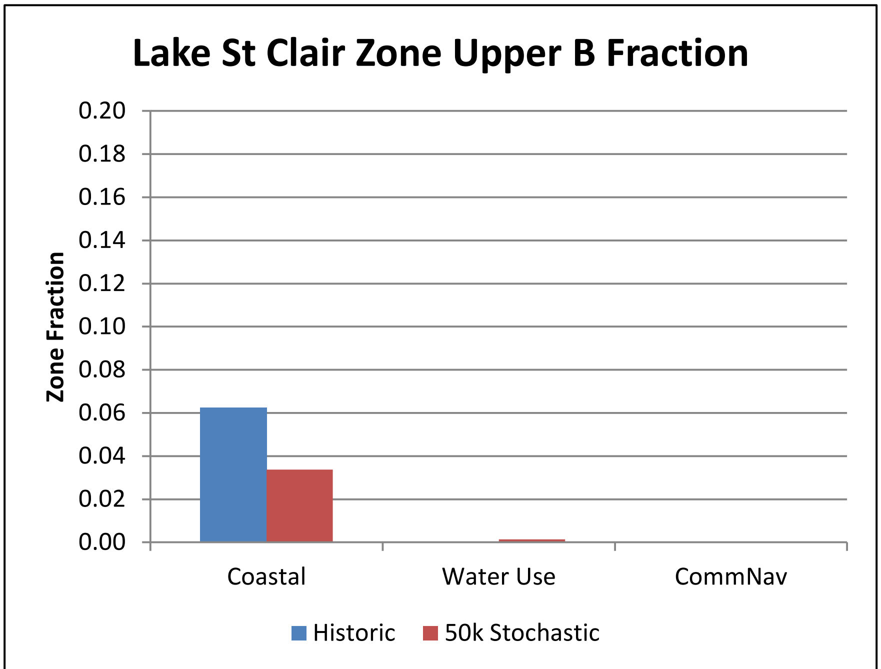

Lake Erie Zone Upper C Fraction
Zone Fraction
0.20
0.18
0.16
0.14
0.12
0.10
0.08
0.06
0.04
0.02
0.00
Coastal
Water Use
RecBoating
BoatLauch
CommNav
Historic
50k Stochastic


Lake Erie Zone Upper B Fraction
Note change in zone scale
Zone Fraction
0.35
0.30
0.25
0.20
0.15
0.10
0.05
0.00
Coastal
Water Use
RecBoating
BoatLauch
CommNav
Historic
50k Stochastic

Figures A9-A12. Coping Zone occurrences on Lake St. Clair.






Lake St Clair Zone Upper B Fraction
Zone Fraction
0.20
0.18
0.16
0.14
0.12
0.10
0.08
0.06
0.04
0.02
0.00
Coastal
Water Use
CommNav
Historic
50k Stochastic

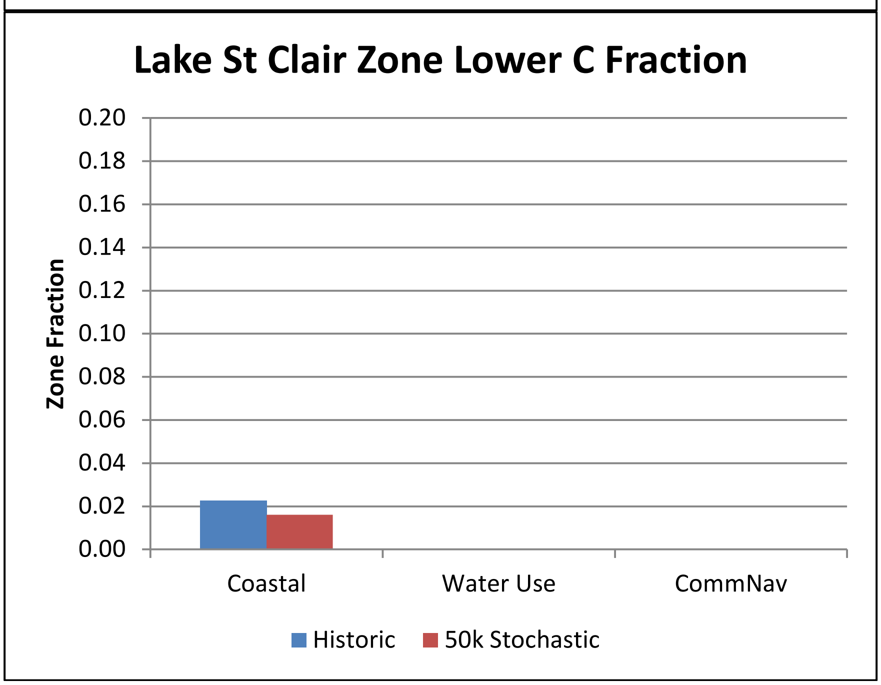

Lake St Clair Zone Lower C Fraction
Zone Fraction
0.20
0.18
0.16
0.14
0.12
0.10
0.08
0.06
0.04
0.02
0.00
Coastal
Water Use
CommNav
Historic
50k Stochastic

Figure B1. Lake Superior levels based on a 5% increase in 77A Lake Superior outflow from Jan through Aug based on a high lake level with a high NBS forecast. Note that AM refers to 77A with forecast adjustment.

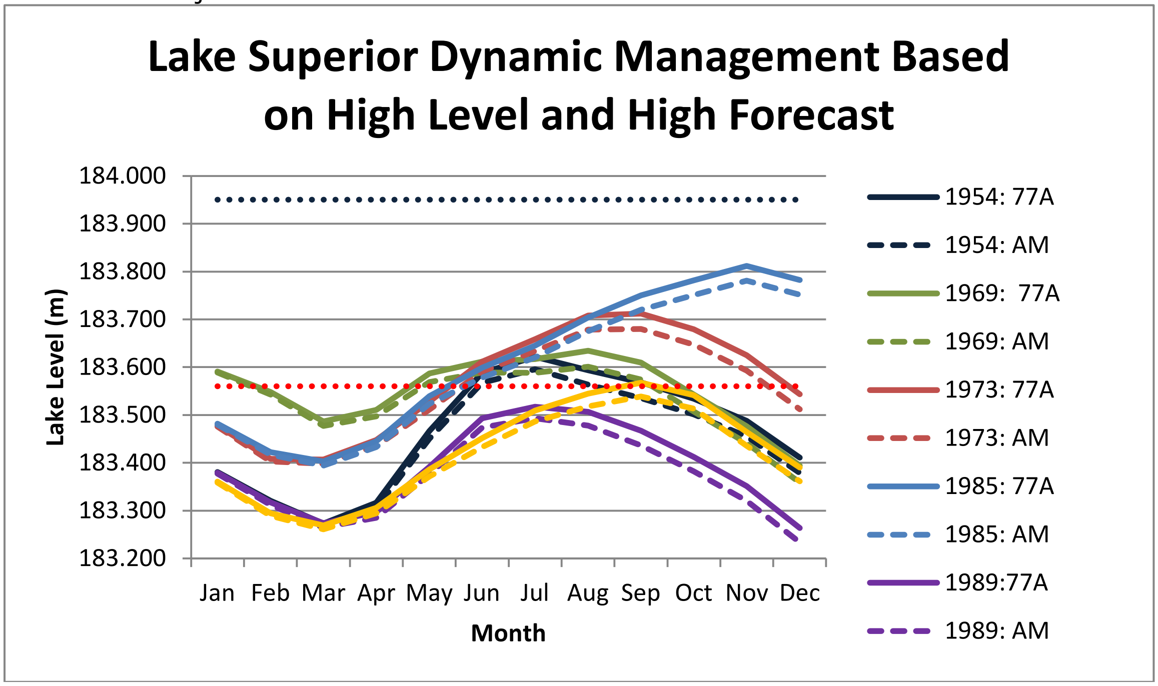


Figure B2. Results for a correct forecast based on 1969 showing the forecast-based release rule results in a lower high level and a shorter duration in coping zone B.

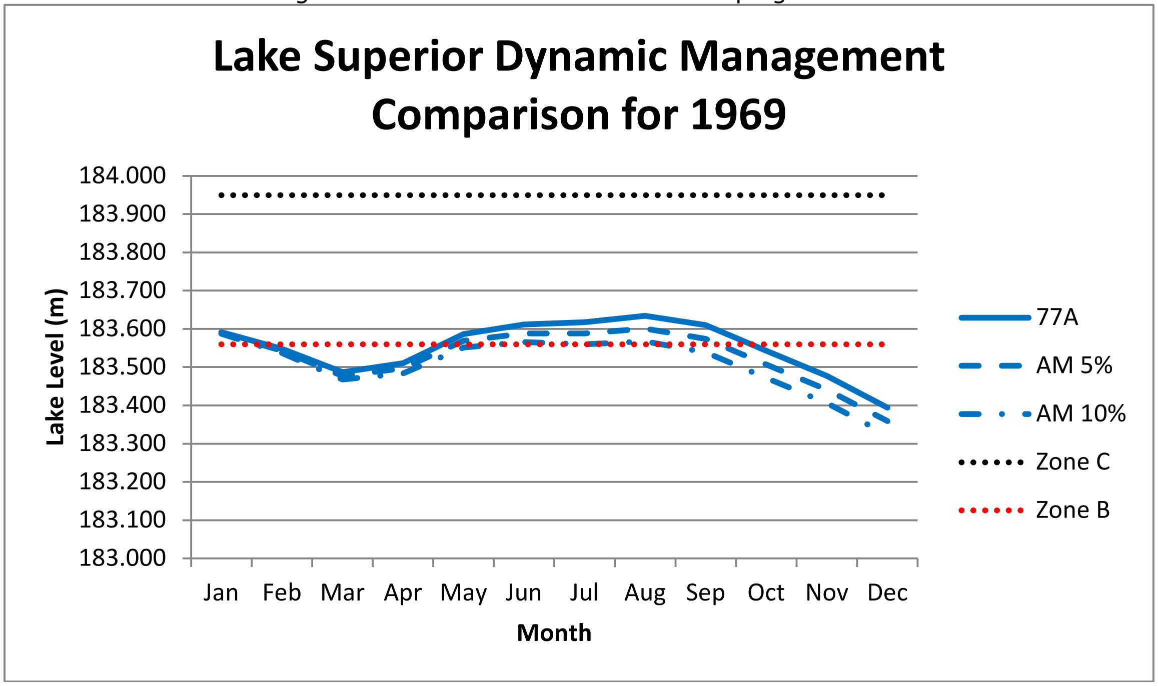


Figure B3. Impact of a 5% reduction in 77A Lake Superior outflow from Jul through Jun based on a low year with a low forecast.

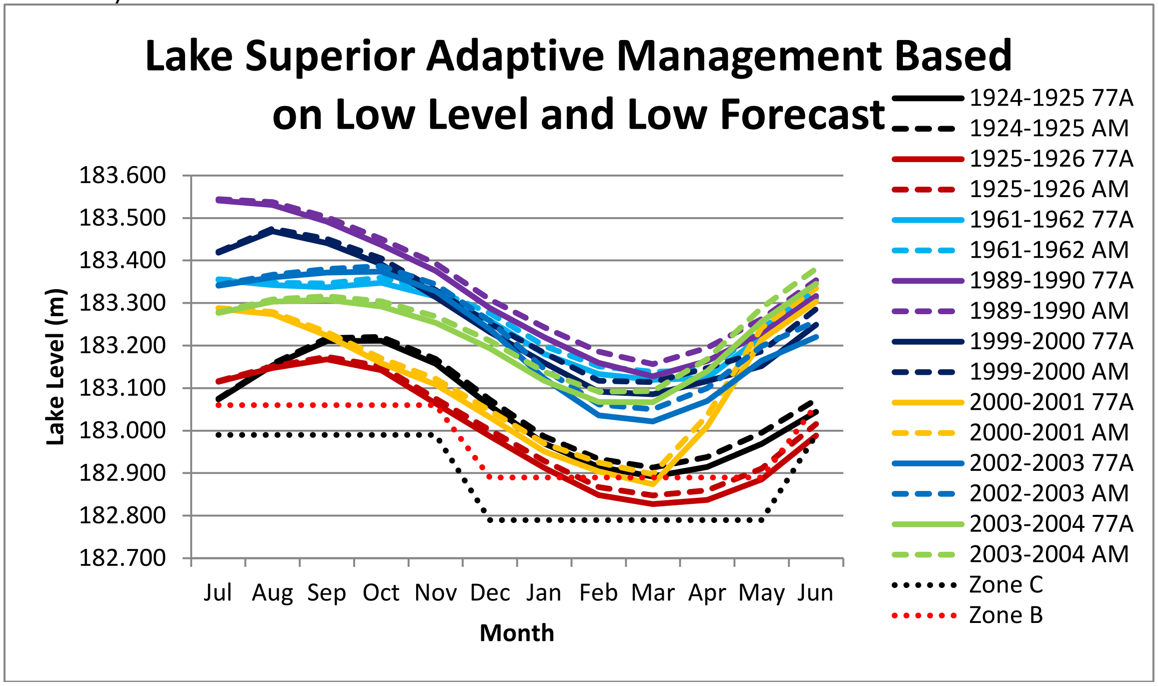


Figure B4. Results for a correct forecast based on 1925-1926 showing the forecast-based release rule results in a higher low level and a shorter duration in coping zone B.

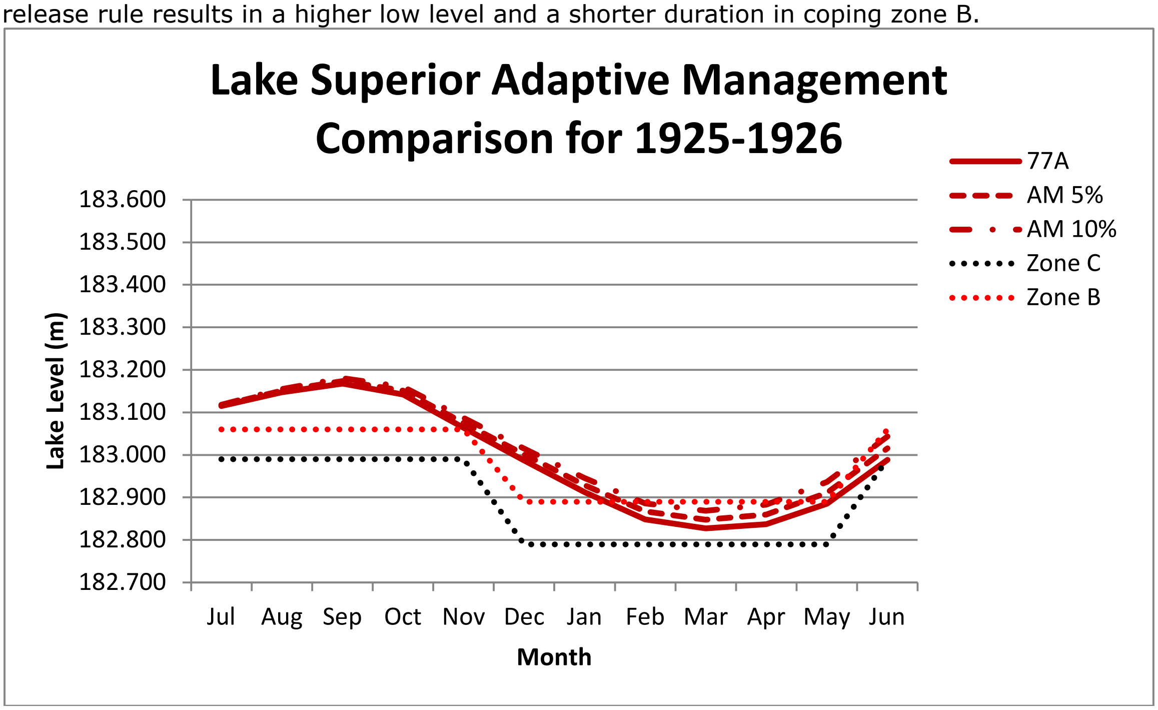


Figure B5. The impact of false alarm when low NBS is predicted but high NBS is observed. The results show that there is no negative impact of the false alarm.

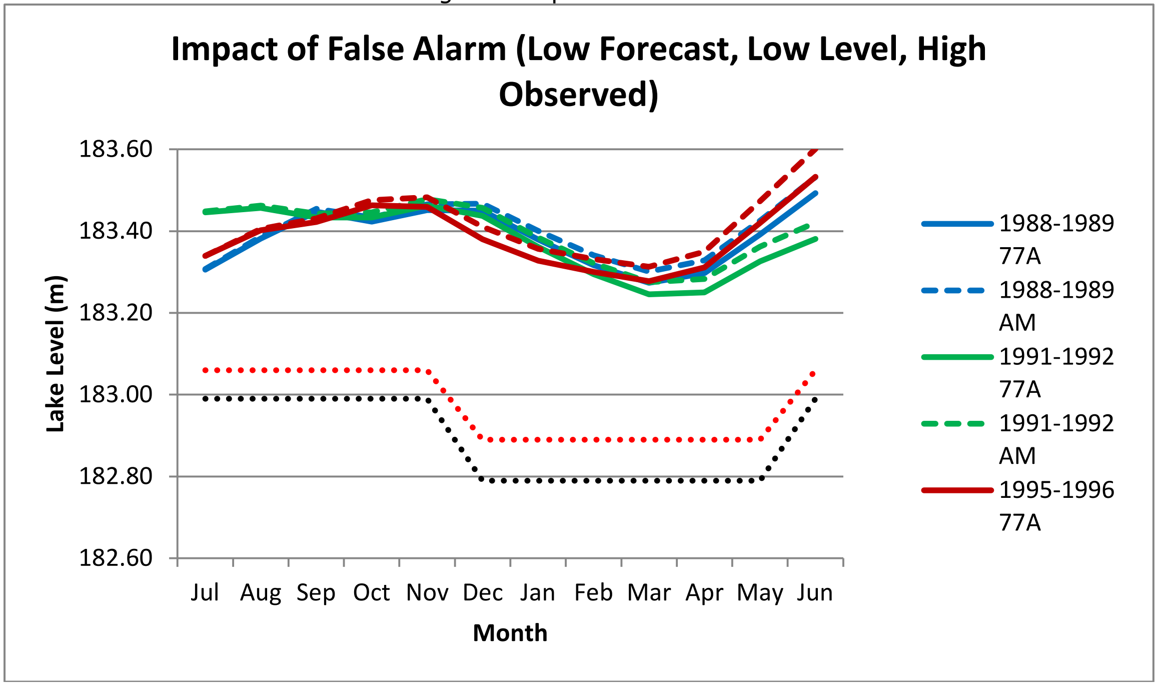


Figure B6. Figure B6. Comparison of Matlab 77A model compared with Fortran 77A.

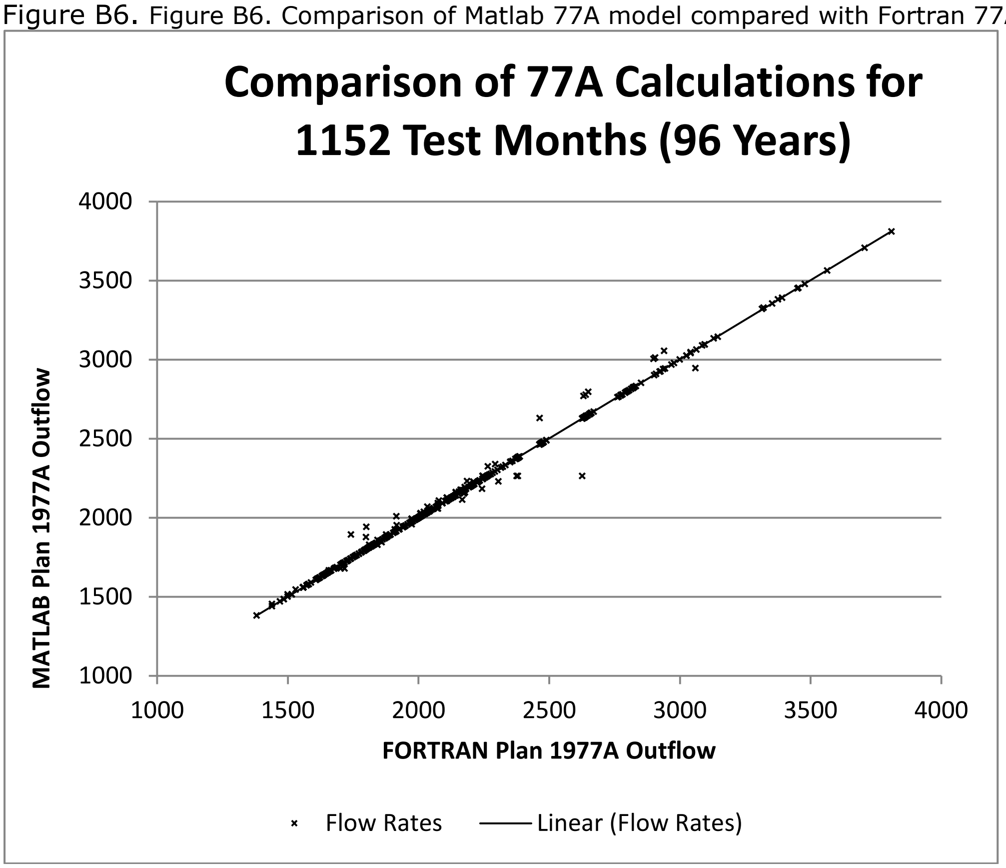


Figure B7. Results of dynamic 77a with forecast based releases and perfect forecasts based on the Vincent Fortin operation forecast period. Results show reduction in occurrences of high lake levels and high zone B and general compression of lake levels. Levels on Michigan-Huron were minimally impacted.

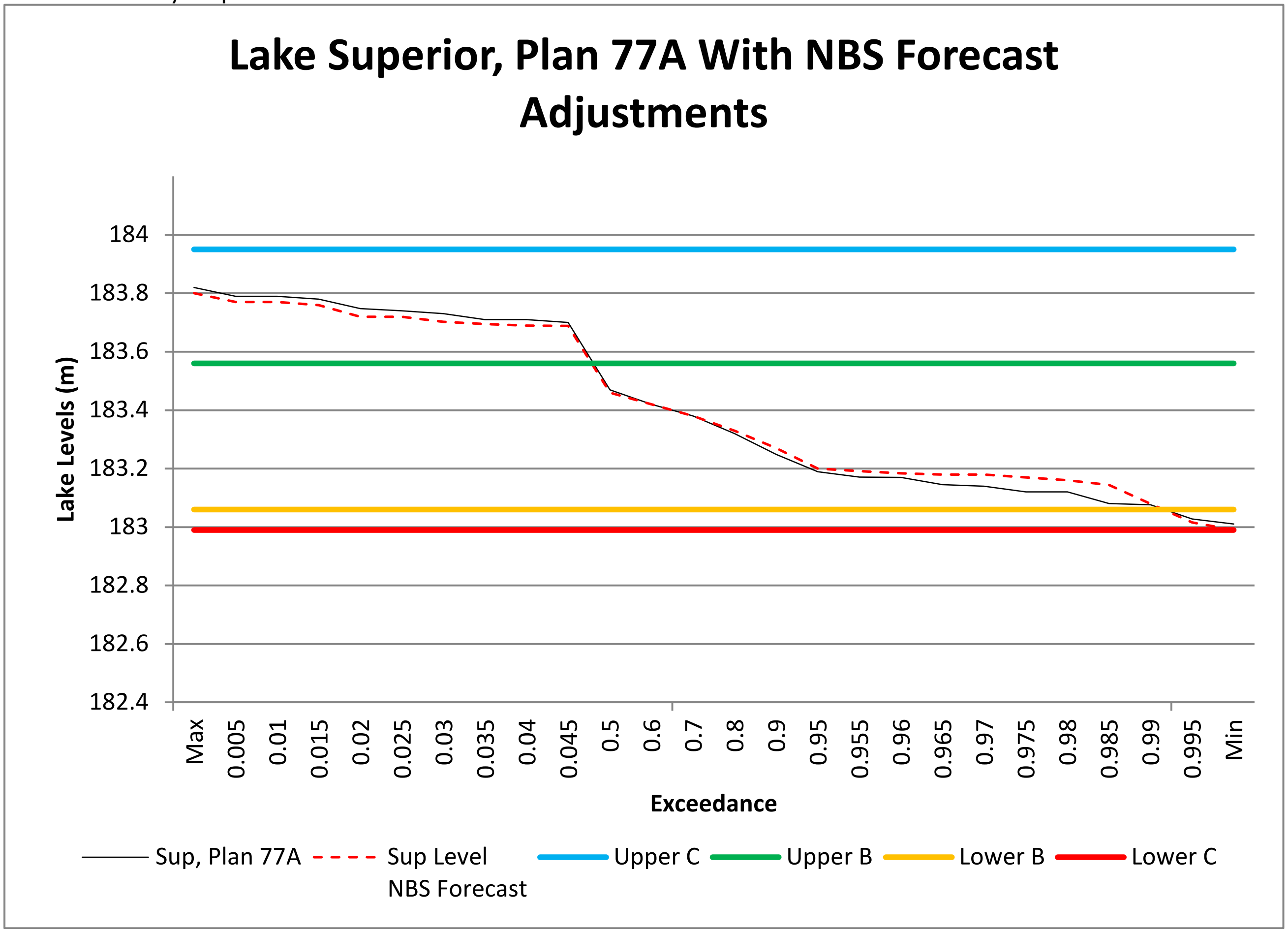


Figure C1. The current approach to management of Lake Superior: a regulation plan is enacted on the lake system which is subjected to exogenous factors resulting in the observed lake levels. The lake levels result in impacts that are primarily observed by stakeholders. Feedback is provided to the International Joint Commission often as complaints from the public.

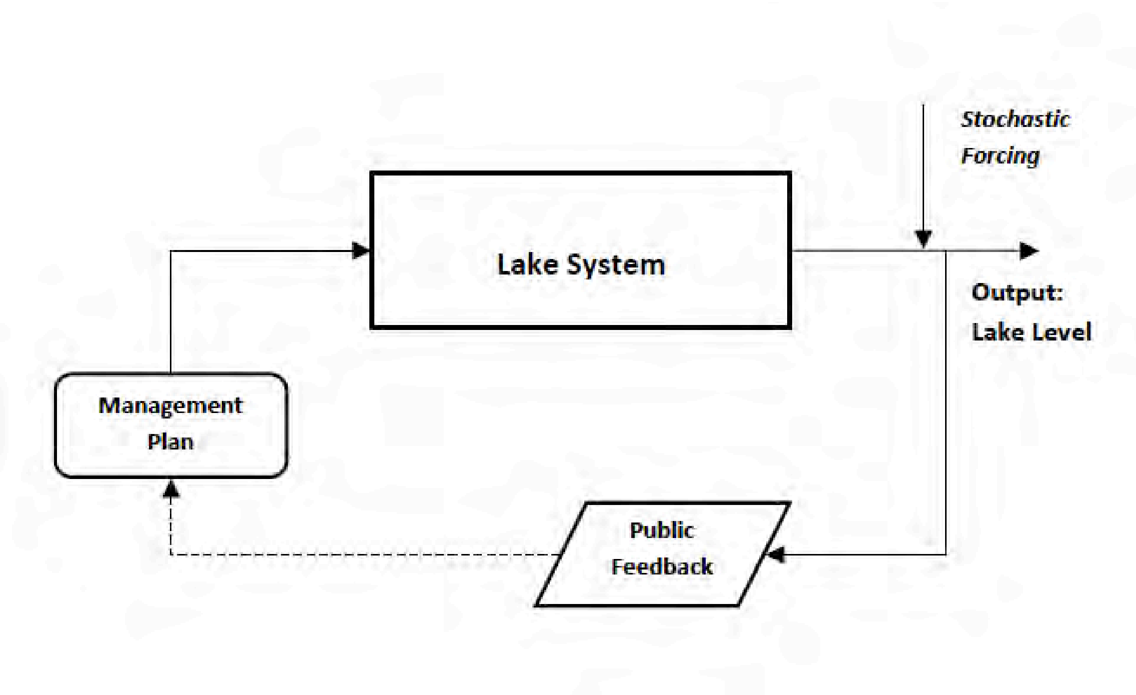
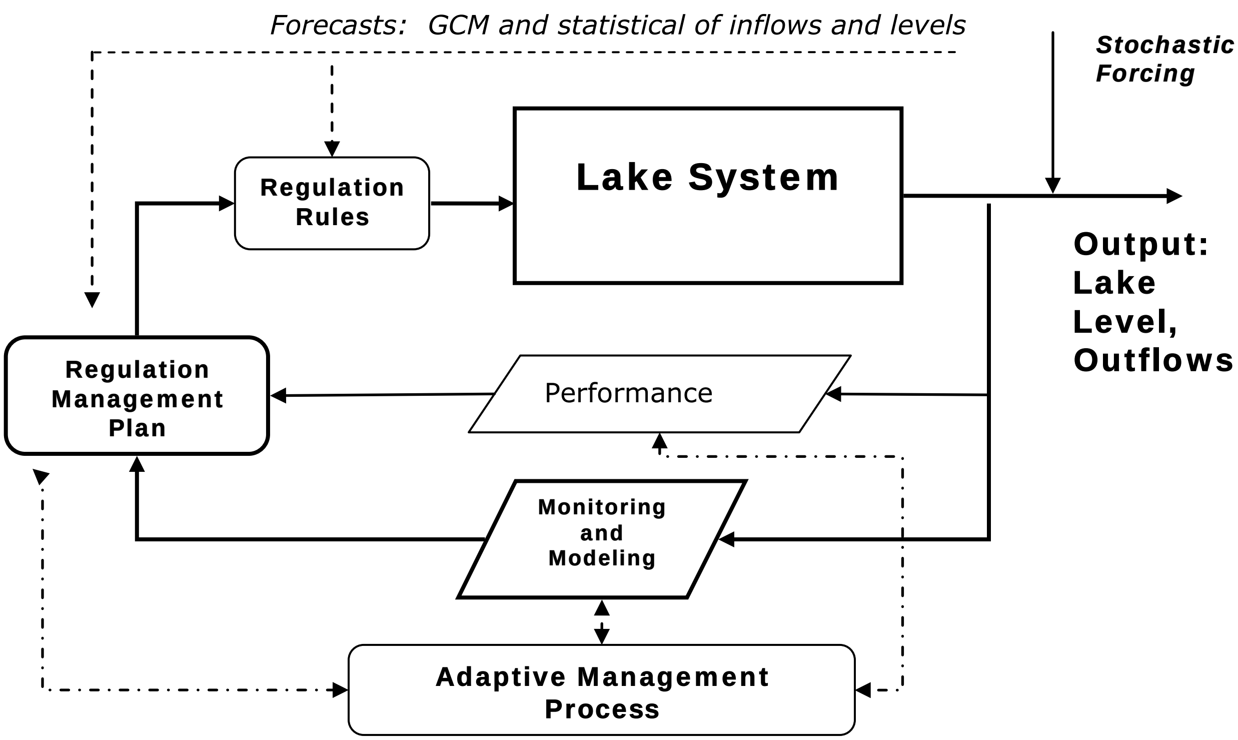


Figure C2. The proposed hierarchical adaptation strategy for the regulation of Lake Superior will utilize a dynamic regulation plan to select among several regulation approaches depending on the plan performance and the observed climate conditions. Feedback is provided via a monitoring program and ongoing evaluation of performance metrics related to coping zone status. At the highest level of the hierarchy, the performance of the dynamic regulation plan, including the performance metrics themselves and the monitoring program, is evaluated and when necessary, improved through an adaptive management process



14. Tri-State Connecticut River Targeted Watershed Initiative

Principal Investigator: Jerry Schoen, MA Water Resources Research Center, UMass Amherst
Start Date: December 1, 2007
End Date: June 30, 2013
Reporting Period: July 1, 2010 – June 30, 2011
Funding Source: USEPA Targeted Watershed Initiative Grant Program
Descriptors: Connecticut River; Water Quality; Volunteer Monitoring; Information Technology

Problem and Research Objectives

The Connecticut River has been described as "the Pioneer Valley's Boston Harbor" because the river still has significant water quality problems. In New Hampshire and Vermont, water quality is impaired due to erosion, sedimentation, and combined sewer overflows, and mercury and PCBs render fish consumption unsafe. From the Holyoke Dam south to Connecticut, water quality standards are not supported (Class B fishable/swimmable) due to pathogens and suspended solids, primarily from urban runoff and combined sewer overflows. According to the USGS, bacteria levels in the lower Connecticut River, which can measure as high as 10,000 fc/100ml, are among the highest found in southern New England rivers. In addition, the Connecticut Department of Environmental Protection concluded that nitrogen loads from the Connecticut River to Long Island Sound must be reduced by 58% in order to reverse eutrophication. Similar to Boston Harbor, clean-up costs are very high, estimated at $325 million for CSOs in Springfield, Chicopee, and Holyoke alone. This is an environmental justice issue, as many low-income residents in the Holyoke-Springfield reach use the river for fishing and swimming. The benefits of cleaner water will also be enormous due to the popularity of the river for recreation and riverfront economic development.

The Tri-State Connecticut River Targeted Watershed Initiative addresses the most significant water quality problems of the Connecticut River watershed: major bacterial pollution from combined sewer overflows and urban stormwater; extensive streambank erosion; threats to public water supplies; and nutrient loading from agricultural runoff. It is funded under a $953,000 Targeted Watershed Initiative grant from the U.S. Environmental Protection Agency, matched by $458,000 in local funding commitments.

WRRC organized and conducted a Water Quality Monitoring program to document E. coli bacteria levels at 26 sites along the river in the project's first 2 years, as described in the Center's 2009-2010 annual report.

Outreach and Education / Technology Transfer

WRRC continues to maintain data from this project to the Connecticut River website established for the Tri-State Connecticut River Targeted Watershed Initiative (http://www.umass.edu/tei/mwwp/ctrivermonitoring.html). This site is currently hosted and maintained by the Water Resources Research Center at UMass Amherst.

Information Technology

Project Website with Virtual Watershed Tour

WRRC and CESD continued to work on and improve the project web site: http://www.cesd.umass.edu/twi/. The web site includes a multimedia *virtual tour* of the watershed. Maps, photographs, graphs, audio and text illustrate valuable recreational, economic, and public health resources provided by the river along its length from source to sea. Site users are able to virtually fly over the watershed and zoom in to selected locations to learn of popular boating, fishing, swimming areas, water supply, agricultural lands, etc. The site contains links to other sites about natural and cultural history of the Connecticut River.

Mobile Story Tours

WRRC and CESD worked together to adapt several of the virtual tours for use on handheld computing devices, to enable users to learn of events and activities occurring at project site areas by playing back information about the project on mobile devices, while walking through the sites.

Electronic Field Guide

WRRC developed an electronic field guide to observational water quality indicators (e.g. pictures and narrative description of pollution or sediment plumes, degraded stream banks, when foam is or is not likely to be a sign of pollution). This was created for the web site and formatted to be viewable on handheld computers. The guide advises users how to document suspected pollution problems with photos, description and location information, how to submit observations to project staff. Project staff will follow up to validate observations and contact local authorities as appropriate

Notable Achievements and Awards

The Pioneer Valley Planning Commission used the results from this project to successfully apply to the federal 604(b) program for continued monitoring of nine Connecticut River Mainstem sites and of numerous tributary sites in Massachusetts, to locate sources of bacterial pollution. WRRC was used as a subcontractor in this project.

The information technology practices used in this project were cited in a successful proposal by UMass Amherst and Boston campuses to the National Science Foundation's Course, Curriculum and Laboratory Improvement Program. WRRC is collaborating with the UMA Center for Educational Software Development, the Landscape Architecture Program, and the Biology Department of both campuses on this two-year project, beginning in July 2010 and described elsewhere in this report.

Publications and Conference Presentations

Jerry Schoen, 2010. Rapid Response Water Quality Monitoring and Public Awareness Final Report. Water Resources Research Center, University of Massachusetts, Amherst MA 01003.

15. Connecticut River Water Quality Monitoring Project

Principal Investigator: Jerry Schoen, MA Water Resources Research Center, UMass Amherst
Start Date: August 1, 2009
End Date: June 30, 2011
Reporting Period: July 1, 2010 – June 30, 2011
Funding Source: USEPA 604B grant program
Descriptors: Connecticut River; Bacteria Monitoring

Problem and Research Objectives

The project continues an on-going volunteer based bacteria monitoring program in the Connecticut River watershed in Franklin, Hampshire and Hampden Counties. The project involves the collection of bacteria samples along the main stem of the river, new collection of baseline bacteria data on tributaries suspected to be sources of bacteria but where little or no data exists to document the problem, and to perform new monitoring and field reconnaissance at specific locations for bacteria source tracking. Data collected is shared with the public, DEP, municipal officials, and other stakeholders through posting the data to an established web site targeting recreational river users as well as outreach through local media and forum outlets.

Methods

The Pioneer Valley Planning Commission (PVPC) partnered with the Connecticut River Watershed Council (CRWC) and the Water Resources Research Center at UMass Amherst to perform the scope of work as described below.

QAPP

WRRC, in coordination with PVPC and CRWC, wrote Quality Assurance Project Plan (QAPP) that stipulates measures to be taken to assure data quality for the project water quality sampling program. The QAPP was approved by joint EPA/MassDEP review.

Volunteer Coordination and Training

PVPC coordinated volunteers in Hampshire and Hampden Counties and CRWC coordinated volunteers in Franklin County. Volunteers collected water quality samples during the high-use summer recreation months (May-October). PVPC and CRWC staff served as Regional Coordinators to oversee sample event preparation, activities of volunteer field samplers, sample transport to laboratories, and communication with labs relative to volunteer field samplers.

WRRC performed two volunteer-monitor training sessions (one in the upper reach and one in the lower reach) in May 2010.

Sampling sites were broken out into three tiers: Tier 1, Tier 2, and Tier 3 sites. Tier 1 sites included 9 sites along the main stem of the Connecticut River in Franklin, Hampshire and Hampden Counties. These are a subset of the 15 sites involved in the EPA Targeted Watershed Initiative monitoring program. Samples were collected at these sites one day per week for bacteria analysis.

Tier 2 monitoring sites are on Connecticut River tributaries that are suspected to be contributing bacteria loading to the main stem based on the land uses within the watershed and /or documented water quality impairments. Tier 2 tributaries were identified based on the bacteria levels at the main stem sites, guidance from the Advisory Committee, and DEP’s bacteria source tracking team. Up to 30 Tier 2 sites on tributaries along the entire main stem of the Connecticut River in Massachusetts were monitored two times per month for 6 months (total) for bacteria “screening level” sampling.

Tier 3 monitoring sites were identified specifically for bacteria source tracking along those Tier 2 tributaries where bacteria screening results indicated bacteria levels significantly higher than nearby Tier 2 sites. Tier 3 monitoring sites were to be sampled once per month for six months. Additional funding secured by PVPC allowed these sites to continue to be monitored beyond the end of the grant conclusion and this reporting period.

Outreach and Education / Technology Transfer

WRRC continues to maintain and post data from this project to the Connecticut River website established for the Tri-State Connecticut River Targeted Watershed Initiative (http://www.umass.edu/tei/mwwp/ctrivermonitoring.html). This site is currently hosted and maintained by the Water Resources Research Center at UMass Amherst. Data continues to be posted to the website within 24 hours of completed laboratory analysis to alert recreational users to water quality conditions. This website has been operational since 2008.

Other Publications and presentations

Newspaper Coverage

The Republican
"Connecticut River water quality monitoring project will keep river users informed of E. coli levels". Published: Sunday, June 06, 2010, 1:00 AM Updated: Monday, June 07, 2010, 7:29 AM. John Appleton, The Republican.

http://blog.masslive.com/breakingnews/print.html?entry=/2010/06/connecticut_river_water_qualit.html

Television Coverage

Channel 22 News
"Monitoring the quality of the CT River: Monitors water quality at a number of sites" Published: Friday, 04 Jun 2010, 5:17 PM EDT
http://www.wwlp.com/dpp/news/local/monitoring-the-quality-of-the-ct-river

Channel 22 News
"Water quality monitored on Conn. River: Volunteers will collect water samples this summer" Published: Sunday, 06 Jun 2010, 10:52 AM EDT
http://www.wwlp.com/dpp/news/massachusetts/water-quality-monitored-on-conn-river

16. Blackstone River Water Quality Modeling Study

Principal Investigator: Dr. Paula Rees, MA Water Resources Research Center, UMass Amherst
Start Date: 2/26/2004
End Date: On-going
Reporting Period: July 1, 2010 – June 30, 2011
Funding Source: Upper Blackstone Water Pollution Abatement District
Descriptors: Blackstone River; Water Quality Monitoring; Water Quality Modeling; Watershed Management
Focus Categories: Nonpoint Pollution; Hydrology; Water Quality; Management & Planning

Problem and Research Objectives

The purpose of this study is to assess existing water quality conditions, identify sources, quantify pollutant loads to the river, develop modeling tools for determining the fate and transport of nutrients along the river, and utilize these tools to evaluate the effectiveness of various management strategies (both point- and nonpoint source controls) for improving water quality and ecosystem health along the Blackstone River. The questions to be answered include:

- Based on existing data, particularly including data collected since completion of the Blackstone River Initiative, what is the current status of the Blackstone in terms of water quality conditions and ecosystem health during both dry and wet conditions along its various reaches?
- How have water quality and ecosystem health changed over time and how may they be expected to change in the future?
- How does water quality compare to that of other watersheds, both developed and non-developed, and thus what concentrations are feasible to attain along the Blackstone for a variety of water quality parameters?
- What are the sources of pollutants most negatively affecting Blackstone River water quality and ecosystem health?

- What are the relative contributions of pollutants from these sources?
- What strategies to improve water quality have been successfully implemented in similar watersheds?
- What are the most effective methods (feasible, reasonable, equitable, and economic) for improving water quality and ecosystem health of the Blackstone River?
- How are improvements in water quality and ecosystem health along various reaches of the Blackstone River anticipated to affect downstream receiving waters?
- What critical gaps are there in our understanding of loading and important processes impacting nutrient concentrations along the Blackstone River?
- What critical gaps are there in our understanding of the effects of point and nonpoint source pollution on ecological health within the Blackstone.

Background

The Blackstone River (Figure 1) originates at the confluence of Middle River and Mill Brook in Worcester, Massachusetts. It flows southeast for 46 miles into Rhode Island where it joins the Seekonk and Providence Rivers, which discharge to Narragansett Bay. The Blackstone River watershed, shown in Figure 1, has an area of approximately 480 square miles. The mainstem of the Blackstone River is joined by six major tributaries: Quinsigamond River, Mumford River, West River, Mill River, Peters River, and Branch River, as well as many smaller tributaries. The watershed consists of over 1,300 acres of lakes and ponds including the largest, Lake Quinsigamond. Several reservoirs in the northwest portion of the basin are used in conjunction with out-of-basin sources for the City of Worcester water supply.

The Blackstone is known as the "Birthplace of the American Industrial Revolution." During its 46 mile run toward Narragansett Bay, the river drops 438 ft (Shanahan, 1994; BRNHC, 2006), a steeper gradient than the Colorado River (Arizona Humanities Council, 2006). Recognizing the hydraulic potential of the river, Samuel Slater built the first mill in America at the outlet of the Blackstone in 1793. Others followed suit, and at one point the river had almost one dam for every mile of river along its run. The active and remnant dams along the river strongly influence water quality along the river, in part through their impacts on sediment and associated contaminant transport as well as travel time.

The historical significance of the Blackstone River has been widely recognized. In 1986, by an Act of Congress, the John H. Chafee Blackstone River Valley National Heritage Corridor (BRNHC) was established. This designation provides the region an organizational framework with the ability to preserve the unique and significant value of the Blackstone Valley and to celebrate the role it played in the development of the nation (BRNHC, 2006). In 1998, the Blackstone received American Heritage River status, opening up the potential for making more efficient and effective use of existing federal resources and cutting red-tape, without any new regulations on private property owners or state and local governments (USEPA, 2006).

In 2002, the Blackstone was one of eight rivers named to the Urban Rivers Restoration Pilot Study conducted by the United States Environmental Protection Agency (USEPA) and the United States Army Corps of Engineers (ACOE). The Urban Rivers Restoration Pilot Study is aimed at promoting collaboration among the EPA, ACOE, local businesses and the non-profit community within the watershed in order to advance pollution prevention, water quality improvements and restoration of wildlife habitat within the watershed (USEPA and ACOE, 2006).

Methodology

In 2004, the Upper Blackstone Water Pollution Abatement District (UBWPAD) initiated the Blackstone River Water Quality Study in order to develop a watershed management tool for the Blackstone River basin that could be used to evaluate the impacts of the plant effluent, the effectiveness of point source control versus non-point source management, and the effectiveness of alternative management strategies on downstream river quality. The study was initiated to enhance the overall understanding of flow and water quality characteristics of the river. Specifically, the study was designed to:

- Conduct a field-sampling program to provide wet and dry weather water quality data in headwater and mainstem locations in the Blackstone River watershed (see Mangarillo, 2006 and 2009; Patterson, 2007).
- Evaluate and model dynamic water quality conditions incorporating daily, monthly, seasonal and inter-annual variability.
- Incorporate explicitly into the modeling analysis point source (e.g. waste water treatment facilities) and non-point source (stormwater runoff) loads to the river.

To support this effort, the Blackstone River HSPF Water Quality Model was developed by the Massachusetts Water Resources Research Center in collaboration with CDM. This model is based on an existing water quantity model of the Blackstone River watershed, which was developed by the United States Geological Survey (USGS) (Barbaro and Zariello, 2006). The University of Massachusetts Amherst (UMass Amherst) and CDM extended the simulation period of the model through 2007 and employed HSPF modules for simulating water quality in addition to water quantity. The water quality model was officially released to the public for comment and review in June 2008. Model calibration to field measurements collected from 1997 through 2007 by a number of different agencies was documented in the *Blackstone River HSPF Water Quality Model Calibration Report* (CDM and UMass, 2008). This version of the model was utilized in the development of the *Blackstone River HSPF Model Scenario Report* (MAWRRC, 2008).

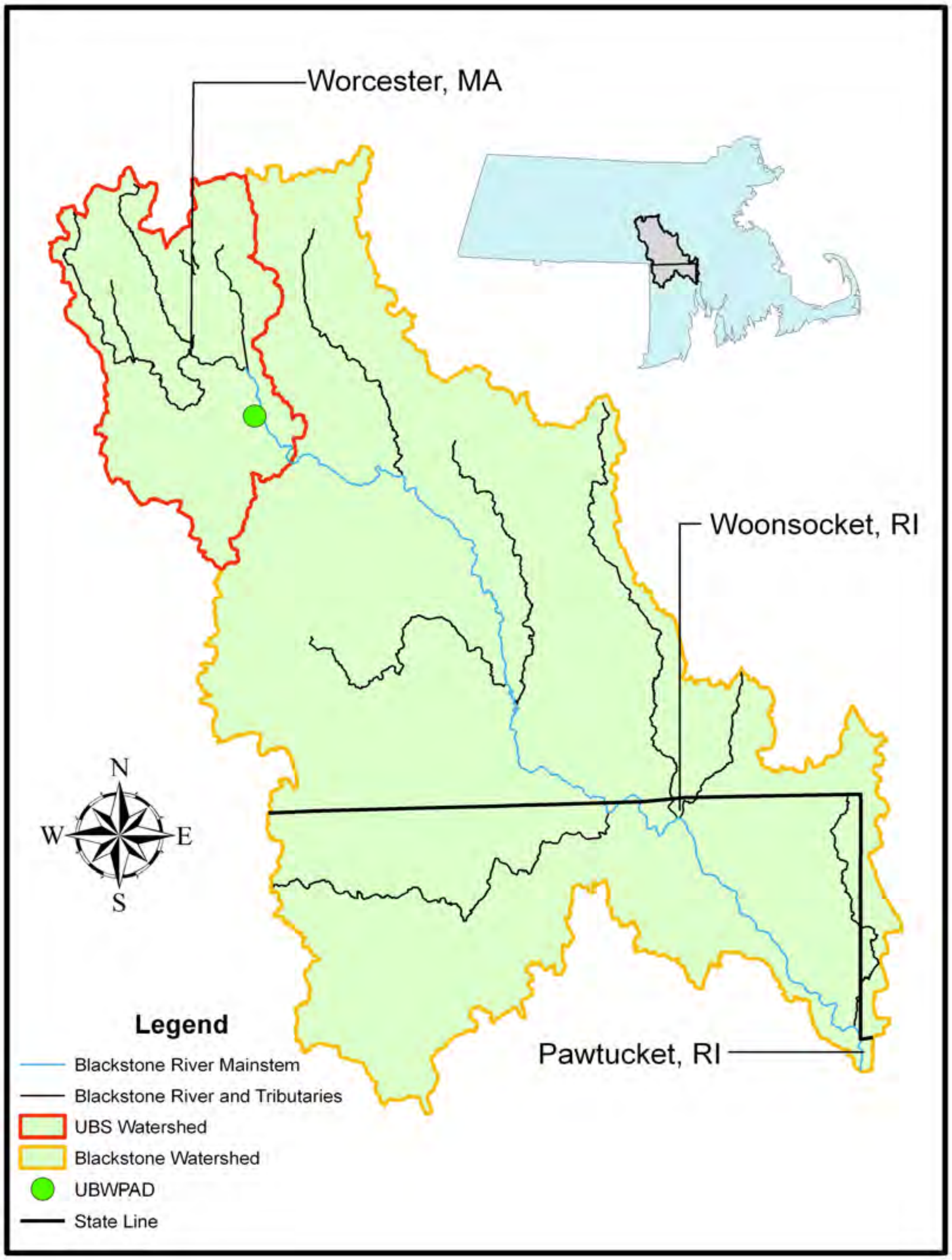


Figure 1. Blackstone Watershed

Work under this reporting period

Between July 1, 2010 and June 30, 2011, modeling work focused on the following:

- Review of the recent modeling work by our Technical Advisory Committee,
- Refinement of the hourly dissolved oxygen (DO) calibration and time of travel comparison,
- Evaluation of re-calibration impacts on the scenario results,
- Completion of a model validation study,
- Initiation of a new monitoring program,
- Participation in a watershed workshop.

Each of these activities is described in more detail below.

Technical Advisory Committee Meeting
In 2007, the technical advisory committee (TAC) was formed to provide technical review and guidance for the Blackstone River water quality modeling performed by UMass Amherst and CDM. The most recent meeting with the TAC occurred August 30, 2010. The objective of the August 30th TAC meeting was to review the model calibration and simulations completed since 2008 which was when the TAC last reviewed and provided comment on the model. During the August 30th meeting several additional tests and analyses were identified to further evaluate or improve the model calibration. These included generation of additional model calibration/goodness of fit statistics, more extensive modeling work to validate the model time of travel results, and additional work on model DO simulation.

DO Calibration and Time of Travel

In response to comments by the Technical Advisory Committee (TAC) during the August 30, 2009 meeting, the Blackstone River HSPF Model was extended through 2009 to simulate the USGS Time of Travel Study conducted in September 2009. This simulation provides a more accurate depiction of travel time in the model than previous estimates calculated using the stage-volume-discharge curves (FTABLES) rather than through a dynamic simulation.

Input time series were used to define the hour and location of each of the three dye injections. An initial dye injection of 10,000 kg over one hour was arbitrarily used for each phase to ensure the simulated concentrations were large enough to be reported by the model. Because the actual mass of dye injected during the study is not known at this time, the actual simulated concentrations of dye were not intended to be accurate. For the purpose of this comparison, only the shape and movement of the tracer plume was of interest.

In summary, the time of travel simulations using a conservative tracer agree well with USGS Sept 2009 TOT study, except for over prediction of TOT within downstream impoundments.This is likely due to short-circuiting, which cannot be fully captured by the 1-dimensional HSPF model. The HSPF model appears to capture longitudinal advection and dispersion (spreading) of the dye injections adequately for its intended uses.

One of the comments from the TAC review was that there appeared to be a shift in the diurnal signal of modeled DO relative to the measured DO. The modeled diurnal DO in many cases showed the highest DO occurring in the early morning hours (e.g., 6 am), which is typically when the lowest DO may be expected. A number of model adjustments were made to the model to shift these patterns to better match observed data, including development of a new solar radiation disaggregation methodology, adjustments to the water temperature simulation, adjustment of some of the phytoplankton and benthic algae growth, respiration and death parameters, and adjustment of some reaeration and SOD rates. DO results were significantly improved as a result.

2010 Scenario Report
During the past year, a study was conducted to assess the impacts of the 2010 HSPF Model Calibration on the 2009 scenario study results. Specifically, we wished to identify changes in scenario simulation results resulting from the 2010 calibration and subsequently determine if the findings of the *2009 HSPF Scenario Report* still hold.

Ten scenarios (Table 1) were simulated using the HSPF water quality model developed to assess the effectiveness of point source load reductions at UBWPAD, nonpoint source pollution reduction, and impoundment management on downstream water quality conditions. Water quality metrics (Table 2) for Chlorophyll *a*, total phosphorous, dissolved oxygen and total nitrogen, derived from *Total Maximum Daily Load For Nutrients in the Upper/Middle Charles River, Massachusetts - Draft* (CRWA and Numeric Environmental Services, 2009), were used to compare scenarios. While these metrics offer a useful point of reference, natural variability in base-line conditions warrants the development, in coordination with regional regulatory agencies, of metrics specific to the unique characteristics of the Blackstone River. Graphs and tables of simulated instream concentrations summarize the scenario comparison in terms of the selected metrics. The scenarios and water quality metrics utilized for comparison are completely analogous between the 2009 and 2010 calibration results.

Table 1: Summary of model scenarios

Scenario	UBWPAD	Woonsocket	NPS Reduction	Dam Conditions
Historical	Historical	Historical	0	Existing
UP1	2001 Design	2008 Permit	0	Existing
UP2	2008 Permit	2008 Permit	0	Existing
ZeroUB	No Load (Flow Only)	2008 Permit	0	Existing
UP1_NPS20	2001 Design	2008 Permit	20%	Existing
UP1_NPS60	2001 Design	2008 Permit	60%	Existing
UP1_FERC	2001 Design	2008 Permit	0	FERC Dams Only
UP1_NoDams	2001 Design	2008 Permit	0	No Dams
UP1_NPS60_FERC	2001 Design	2008 Permit	60%	FERC Dams Only
UP2_NPS60_FERC	2008 Permit	2008 Permit	60%	FERC Dams Only
Pristine	No Load	No Load	Forest	Existing

Table 2: Metrics utilized in study

Parameter	Summary Statistic	Target Concentration
Chlorophyll-*a*	Summer Mean	≤ 10 mg/L
	Summer Peak (90th Percentile)	≤ 20 mg/L
Total Phosphorus	Summer Peak (90th Percentile)	≤ 0.1 mg/L
	Summer Mean	≤ 0.1 mg/L
Dissolved Oxygen	Minimum Instantaneous (10th Percentile)	≥5 mg/L

	7-day Minimum	≥5 mg/L
	Maximum Instantaneous (90ttPercentile)	≤ 125% saturation
Total Nitrogen	Summer Mean	≤ 0.94 mg/L
	Summer Peak (90th Percentile)	≤ 0.94 mg/L

Adjustments to the calibration generally increased the number of scenarios exceeding DO and Chlorophyll *a* metrics. Improvements were observed in the Percent Saturation DO values (e.g., additional scenarios meet the metric based on the 2010 calibration). Results for TN and TP were virtually indistinguishable between the two calibrations. Differences between scenario results for the two calibrations are summarized as follows:

- In comparison with the 2009 HSPF Model Calibration results, Summer Peak (e.g., 90th percentile) Chlorophyll *a* results for the 2010 HSPF Model Calibration are shifted higher for all scenarios by 5 to 30 μg/L. This shift is higher in Rhode Island for some of the scenarios. Differences between scenarios become more discernable in Rhode Island than previous results.

- Summer Peak (e.g., 90th percentile) Chlorophyll *a* results for the 2010 HSPF Model Calibration show some distinct pairings in the Rhode Island portion of the river not observed in the earlier results. These pairings are the results of differential shifts in Peak Chlorophyll *a* results between the scenarios. For example, 2009 calibration results in Rhode Island were less impacted by the UP1_NPS60_FERC scenario (e.g., higher values previously) and more impacted by the UP1, UP2 and UP1_NPS60 scenarios (e.g., lower values previously) compared to the 2010 results.

- Summer Mean Chlorophyll *a* results for the 2010 HSPF Model Calibration also tend to be higher.

- Along stream minimum DO [both the Minimum Instantaneous (10th Percentile) and 7-day Minimum] low points tend to decrease, resulting in more excursions below the 5 mg/L metric. For example, the range in along stream 7-day Minimum DO values across all scenarios is 5 – 9 mg/L based on the 2009 calibration results and 2 – 8 mg/L based on the 2010 calibration results.

- Plots of along stream Summer Mean DO (not included in Table 9) are very similar between the 2009 and 2010 calibration results.

Differences between scenario results for the two calibrations had little impact on the overall conclusions in the earlier report. While combined management strategies are successful in lowering concentrations below guidance values for some parameters, no single or combined management scenario evaluated to date results in water quality improvements which address all of the metrics utilized in this study.

Several factors limit or influence instream response to nutrient reduction including phosphorus abundance, travel and residence time, and sedimentation processes. When impoundments are present to retard the downstream transport of phosphorus, even slightly elevated levels of phosphorus can result in higher Chlorophyll *a* concentrations along the river. If no impoundments are present, there is insufficient time for algae to take advantage of increased phosphorus levels. Removal of dams along the river has a significant impact on travel time and Chlorophyll *a* concentrations along the river. However, while load reductions paired with dam management have the combined potential to result in Chlorophyll *a* values below the target values, the resulting TP concentrations along the river remain above the target values. If phosphorus levels in the river are only of concern due to their influence on algal growth, higher target values may be acceptable if the hydrodynamics of the river are also managed.

The following is a list of the key findings from this study based on the 2010 HSPF Calibration Scenario Results:

- The simulated scenarios that resulted in reduction of peak Chlorophyll *a* concentration in the Massachusetts portions of the river *downstream of the UBWPAD WWTF* to achieve study metrics include UP1_NoDams, UP1_NPS60_FERC, UP2_NPS60_FERC and UP1_NPS60. The UP1_FERC scenario is very close to meeting the metric.

- Simulation results suggest that only in the scenario where all the dams are removed (UP1_NoDams) do simulated concentrations in both the Massachusetts and Rhode Island portions of the river meet the Chlorophyll *a* water quality metric.

- Travel time appears to be an important factor influencing algal growth, and thus peak Chlorophyll *a* concentration, in the river.

- Simulated Chlorophyll *a* values in Rhode Island are fairly insensitive to changes in phosphorus load alone. (???? Not sure if this one still holds???)

- Simulation results suggest that peak TP water quality metrics for both the Massachusetts and Rhode Island portions of the river are not achieved by any of the river management scenarios included in the study.

- Simulated instream TP concentrations along the Massachusetts portion of the river are higher than in Rhode Island, with observable increases occurring downstream of most WWTFs.

- Simulation results suggest that summer peak and mean TN water quality metrics for both the Massachusetts and Rhode Island portions of the river are not achieved by any of the river management scenarios included in the study.

- When impoundments are present to retard the downstream transport of phosphorus, even slightly elevated levels of phosphorus can result in higher Chlorophyll a concentrations along the river. However, if no impoundments are present, there is insufficient time for algae to take advantage of increased phosphorus levels.

- Model simulations suggest that plausible load reduction alone will not achieve both TP and Chlorophyll *a* metrics.

- Less nutrient load reduction may be necessary to achieve Chlorophyll *a* targets if the hydrodynamics of the river are managed, here explored through dam removal.

- If phosphorus levels in the river are only of concern due to their influence on algal growth, higher TP target values may be acceptable if the hydrodynamics of the river are also managed.

Decreases in nutrient inputs to the watershed do not necessarily translate into equivalent decreases in fluxes out of the basin due to internal processes such as sedimentation, algal growth, and other water column – bed dynamics.

Validation report
In Spring 2010, MassDEP released a copy of the Draft Access Database from the on-going USGS Sampling Project conducted on the Blackstone River from 2007 to present. This version of the database included nitrogen and phosphorus concentration data for instream sampling sites monitored in 2007 and 2008. MassDEP also provisionally released to the project the results of the MassDEP 2008 Blackstone River Watershed Sampling season. A robust model will perform equally well when compared against data not utilized in calibration, when model parameter values are set. Such a comparison (e.g. between simulated data and a new data set) is typically referred to as model validation. During this reporting period, these data were utilized as a model validation data set.
Simulated and observed data were then compared both graphically and statistically. These results were compared against similar results for the calibration period.

The USGS study was designed to take integrated instream samples that would be flow-weighted and collected over a period of days. Roughly bi-weekly composite samples were collected. Depending on flow conditions, subsamples during the bi-weekly period were pumped either to a stormflow or a baseflow sample bottle. These composite samples were then sent to the lab for testing. Thus for each sampling location, roughly bi-weekly concentration data are available for each parameter as well as for stormflow and baseflow conditions, assuming both occurred during the time period. Typically 5 subsamples (e.g., aliquots) per day were collected and split into baseflow or stormflow as appropriate. In addition to the integrated or composite samples, the USGS collected a limited number of point and equal-width-increment (EWI) samples for QA/QC purposes at each sampling location. These samples are analogous to 'instantaneous' point samples collected under other studies. EWIs are concurrent with 1 pump sample. EWIs were collected in 2007 and 2008 and varied by time and season; there are approximately 3 to 5 EWI data points per station.

A total of thirteen in-stream sampling locations were included in the USGS study, ten located along the mainstem of the Blackstone River in Massachusetts as well as three tributary locations, Figure 1. Integrated instream sampling, however, was only conducted at six of the mainstem and three tributary sampling locations. While additional parameters were monitored, the nutrient data are of primary interest for validation of the HSPF model. Nutrient parameters included in the study include TAM, NO2, NO3, TN, PO4 (filtered), TP, DO and TSS, Table 1. Discharge data were collected continuously (15-minute intervals) at each sampling location. DO data, collected with sondes, are not yet available for comparison. In addition, TN and TP were analyzed for both filtered and unfiltered samples, thus total dissolved and total dissolved plus particulate data are available.

In total, data are available over the approximate period from March 2007 through August 2008. Infrastructure to refrigerate the samples and protect sampling equipment was built at each sampling location. These sampling "houses" were built one at a time and data collection at each began as they were finished. The stateline house was built first and therefore has the longest data set.

Table 1: Summary of nutrient data collected during the USGS and MassDEP studies. Note that naming conventions utilized in the respective studies is retained, but USGS PO4 is equivalent to MassDEP DRP.

Parameters Included	
USGS	***MassDEP***
TN (filtered and unfiltered)	TN
NO23	NO23*
NO2	--
TAM	TAM
TP	TP
PO4 (filtered)	TSS
TSS	DO
DO	DRP*
	TRP*

* Collected at only a sub-set of sites.

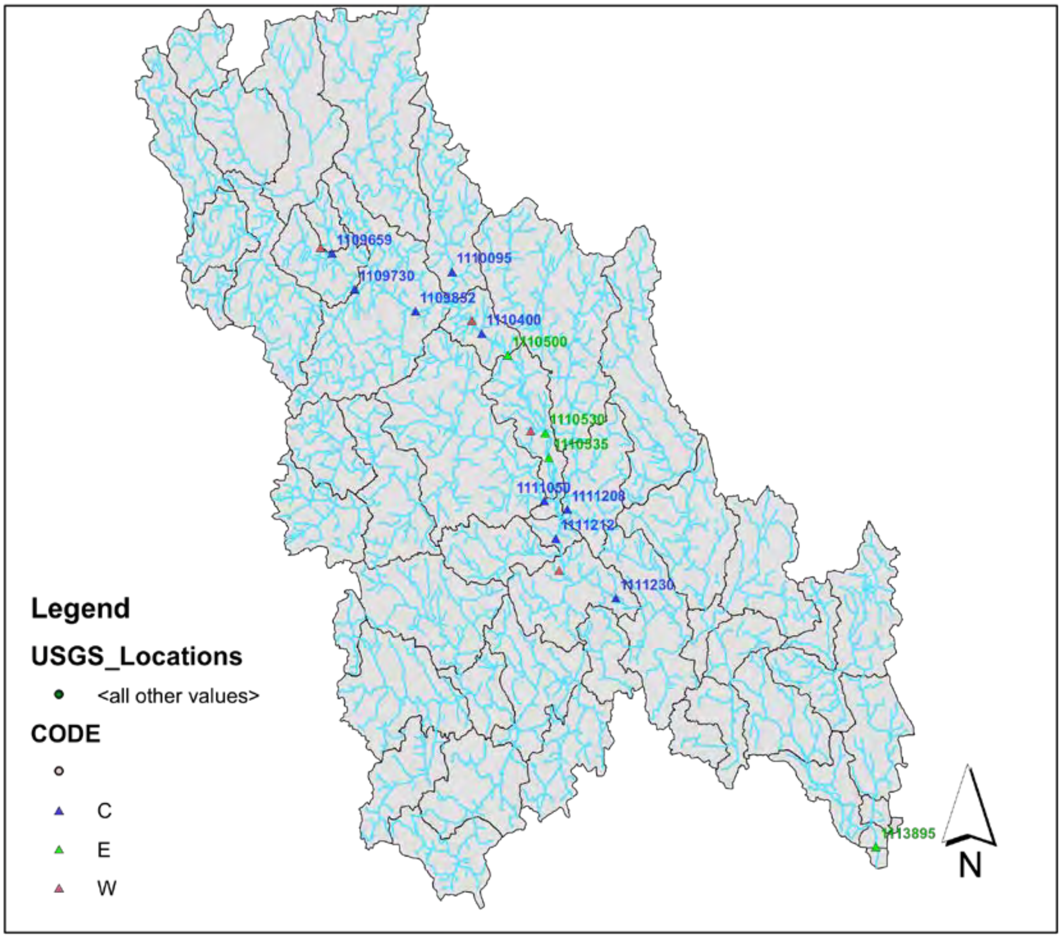


Figure 1: 2007-2009 USGS study monitoring locations in the Blackstone watershed. Color coding is used to indicate type of sampling (Blue – composite samples plus EWI, Green – EWI only, Brown – WWTF's).

During the summer of 2008, MassDEP monitored a total of 72 river stations (including water quality, fish population, metals, unattended deployment, and biomonitoring) and one lake in the Blackstone River watershed. In terms of nutrients, TN, NO23, TAM, TP and TSS data are available. Nutrient data are available for 17 mainstem locations. Additional data are available for tributaries, including the Middle River in the Blackstone headwaters. DO sondes were installed at 9 locations for three to six day periods in May, June, and August (early and late). The sondes provide temperature, dissolved oxygen, and percent saturation information at 30-minute intervals. The MassDEP sampling locations are detailed Figure 2.

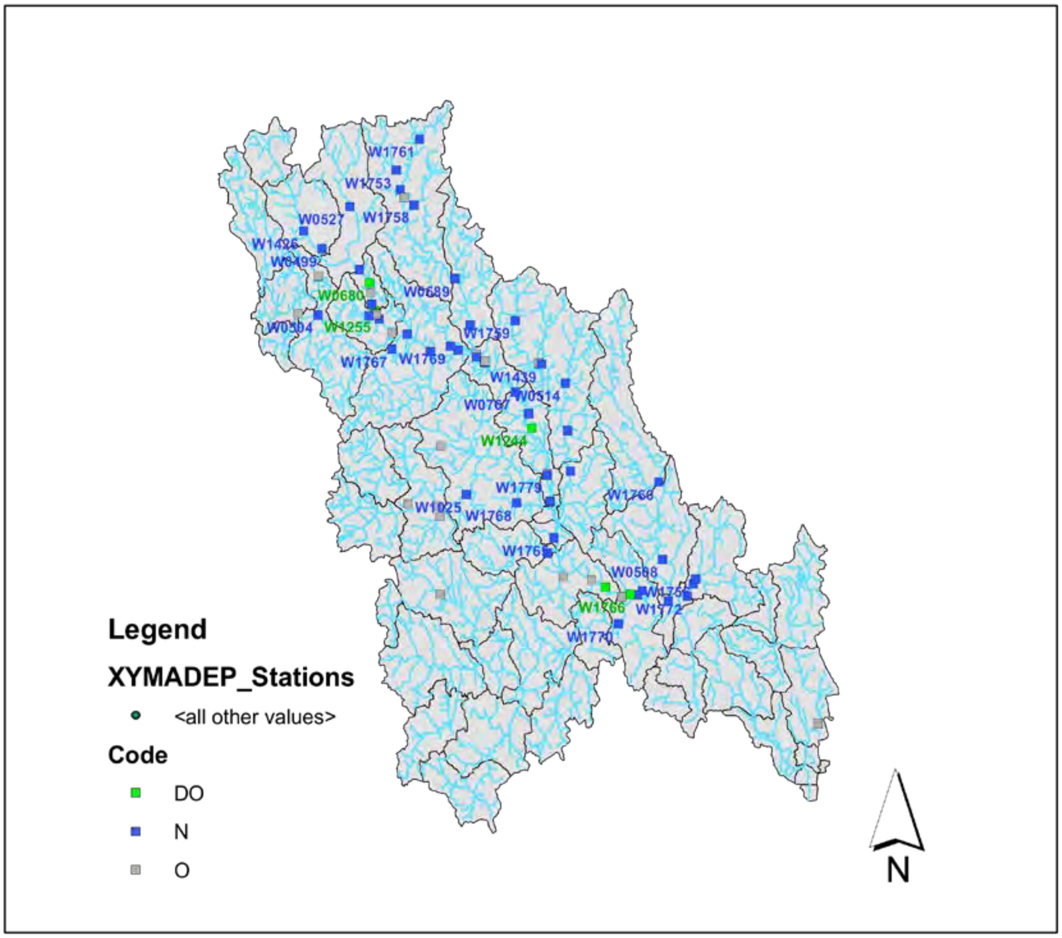


Figure 2: 2008 MassDEP study monitoring locations in the Blackstone watershed. Color coding is used to indicate type of smpling (Green – DO sondes and nutrients, Blue – nutrients, Grey – other, and no station identifier provided).

During the summer of 2008, MassDEP also conducted a river algal biomass and Chlorophyll *a* study of the river. Algae samples were collected from the Blackstone River benthos and water column and analyzed for Chlorophyll *a* content. These samples were also examined to determine the dominant genera composing the algal community. The sampling plan initially specified monthly sampling during periods of low flow. The summer of 2008 was unusually rainy, resulting in river flows that remained high for much of August. Because of these conditions, data were not collected in August. Macrophyte density, biovolume and selected species location maps were prepared for 5 selected impoundments, including Singing Dam, Fisherville, Farnumsville, Riverdale, and Rice City Pond based on mid-August conditions. The dates and ten locations where samples for water column chlorophyll a were collected are listed in Table 2. Samples were either collected from shore by attaching a sample bottle to a pole or, in the case of impoundments, from a boat.

Table 2: 2008 DEP Chlorophyll a data availability

Reach	Location	Water column Chlorophyll *a* (river stations)			Water column Chlorophyll *a* (impoundments)		
		Jun 30	Sep 2	Sep 4	Sep 11	Sep 15	Sep 16
384	Blackstone Bikeway below Rte. 146, Millbury	X					
402	Approx. 60 ft. upstream from confluence of UBWPAD discharge and Blackstone River, Millbury			X			
400	Below confluence UBWPAD and Blackstone River, Millbury		X				
NA	Kettle Brook, bypass upstream of bike path, Millbury			X			
400	Upstream McCracken Rd. Millbury	X		X			
378	Singing Dam Impoundment, Sutton	X		X		X	
378	Singing Dam, along right hand shore, Sutton	X		X			
392	Central Cemetary, downstream Waters St. , Millbury	X	X	X			
370	Depot St., Sutton		X				
326	Rice City Impoundment, Uxbridge				X		
358	Farnumsville Dam, Sutton					X	
362	Fisherville Impoundment, Grafton						X
342	Riverdale Impoundment, Northbridge						X

Data from the two validation data sets were compared against model output for six mainstem reaches (290, 308, 362, 372, 390, and 402) and three tributary reaches (7, 21, and 24) where bi-weekly baseflow and composite data were collected. Five time series plots were generated for each reach and parameters NO2, NO23, TAM, TN filtered, TN unfiltered, and TP. Model and observed phosphate concentrations were not compared as the model output is for total orthophosphate and only dissolved orthophosphate was monitored. In addition, mainstem along stream plots comparing simulated and observed baseflow and stormflow concentrations were generated for 16 bi-weekly periods when baseflow composite data were available and 13 bi-weekly periods when stormflow composite data were available. An overall assessment of model calibration for each parameter was based on the overall relative error, calculated as the median absolute error divided by the average observed concentration, for both the calibration (1996-2007) and validation (2008) periods.

The following conclusions may be drawn from the analysis of model validation results:

- The model captures mainstem baseflow and stormflow TN and TP well. Model performance is typically slightly better for baseflow than stormflow.

- Model performance along the mainstem during the validation period either exceeds or matches model performance during the calibration period. Although the model may not simulate individual species, the validation confirms that the TP and TN simulations are in the good/very good category for the mainstem.

- Tributary baseflow and stormflow statistics suggest the model is likely over-predicting nutrient concentrations in tributary reaches, particularly during baseflow conditions.

New Water Quality Monitoring Along the River
UB plant upgrades based on the 2001 permit are now fully operational, and water quality monitoring will help assess response of the river to reduced nutrient concentrations in the effluent. In addition, these data will help evaluate the ability of the model to capture this response. Towards this end we have begun the following:

- Monthly (May through December) sampling of the Massachusetts portions of the river, timed to coincide with sampling conducted by Narragansett Bay Commission along the Rhode Island portion of the river for nutrients and chlorophyll-a.

- Additional sampling each month (roughly in between the monthly trips) along the mainstem of the river to develop a roughly bi-weekly chlorophyll-a data set. During this sampling run, qualitative data about algae blooms and macrophyte coverage along the river and in impoundments is also collected.

A photo-journal of sampling sites is being created, along with planimetric maps describing macrophyte and periphyton coverage along select portions of the river.

Bay and Watershed Symposium
On June 16th, the Narragansett Bay Commission sponsored a daylong workshop to provide an opportunity to discuss the latest water quality data collected and research conducted in Narragansett Bay. Entitled *A Day on the Upper Bay: Current Monitoring, Research Source Reduction Progress & Future Challenges*, the workshop brought together the Blackstone River and Narragansett Bay water quality experts, regulators and other stakeholders to discuss their most recent findings, source reduction successes and future monitoring and research needs. The workshop also brought together regulators for Rhode Island and Massachusetts to provide the states' regulatory perspectives. The workshop was an opportunity for all stakeholders, scientists and regulators to exchange the latest information and improvements in the Upper Bay, and discuss the future path toward achieving and surpassing water quality standards. The WRRC provided an overview of the Blackstone River HSPF Water Quality modeling efforts.

Students Supported
Over the course of the project, 1 PhD (Jim Mangarillo) and 2 M.S. (Jim Mangarillo and Megan Patterson) students have been supported as well as numerous hourly students and two undergraduate summer researchers (Ryan Leblanc, Noam Perlmutter).

During this reporting period, the project provided hourly support for one student, Michelle Kinney.

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Information Transfer Program

A significant portion of 104B funds retained at the Center supports the information transfer objective of 104B.

Our main information transfer tool is the Annual Water Resources Conference, initiated in 2003 by then Director David Reckhow. The conference provides an interdisciplinary forum for scientists, practitioners, and policy makers to discuss current critical water research, foster greater collaboration among scientists and practitioners, and strengthen the connection between research, education, and policy. Participants include researchers, stakeholders, and managers of water resources from academia, government, non-profits, and the private sector. The 8th Annual Water Resources Research Center Conference is described in the subsequent section. The Center publishes programs from all of our conferences on our website (http://www.umass.edu/tei/wrrc/WRRC2004/WRRCconferences.html).

The Center relies heavily upon the Internet for information transfer. Several of the Center's projects have significant Internet information transfer elements that are still in existence and utilized today. One of these funded through 104B is the Acid Rain Monitoring Project (ARM) (http://umatei.tei.umass.edu/ColdFusionProjects/AcidRainMonitoring/). Another information transfer relationship we are cultivating is with the Consortium of Universities for the Advancement of Hydrologic Science, Inc. (CUAHSI), in order to make data more readily available in a "data clearinghouse" used by local, regional, and national organizations.

1. Innovative Stormwater Technology Transfer and Evaluation Project

Principal Investigator: Jerry Schoen, MA Water Resources Research Center, UMass Amherst
Start Date:
End Date:
Reporting Period: July 1, 2010 – June 30, 2011
Funding Source: MassDEP
Descriptors: Stormwater; Water Quality; Nonpoint Pollution

The Massachusetts Dept of Environmental Protection (MassDEP) awarded WRRC a two and a half year grant to continue a previous project WRRC staff had contributed to in FY'05 and FY'06. The goal of this project is to provide technology transfer information about innovative stormwater Best Management Practices (BMP) to MassDEP, conservation commissions, local officials, and other BMP Users. The project maintains and updates the database already in place (www.mastep.net), by adding information on new studies performed on BMPs previously listed on the MASTEP web site as well performance data on BMPs to the site. These pertain both proprietary BMPs and Low Impact Development BMPs. The MASTEP web site now has approximately 80 technologies profiled. WRRC staff participated in meetings of the Massachusetts Stormwater BMP working group, gathered to generate recommendations for a volume to flow conversion method for purposes of sizing flow-based stormwater treatment BMPs. MassDEP has taken the group's recommendations under advisement. WRRC staff made a presentation on MASTEP, BMP performance, and stormwater regulations to an audience of 100 at a June 10 2011 stormwater seminar in Westborough. The project continues into FY 2012 with additional presentations and web site updates planned. A proposal to continue MASTEP into FY 2012 and 2013 is now under consideration by MassDEP.

2. Stream Continuity Project

Principal Investigator: Scott Jackson, Environmental Conservation, UMass Amherst
Start Date: Spring 2000
End Date: Ongoing
Reporting Period: July 1, 2010 – June 30, 2011
Funding Source: UMass Extension
Descriptors: Stream Crossings; Water Quality; Fish Passage

Under a memorandum of understanding with UMass extension, WRRC staff worked to coordinate volunteers and manage the database for the Stream Continuity Project, a study looking at stream crossings and their status at creating barriers for fish and wildlife passage.

In 2005, three of the organizations/agencies that were key players in initiating and implementing the project joined to create the River and Stream Continuity Partnership. Founding members of the Partnership include:

- UMass Extension (University of Massachusetts Amherst)
- Massachusetts Riverways Program (Mass Department of Fish and Game)
- The Nature Conservancy

Members of the Partnership have made a commitment to the ongoing implementation of the River and Stream Continuity Project, including updates and revisions to the Mass River and Stream Crossing Standards, coordination and implementation of volunteer assessments, management of the Continuity database, and projects to upgrade or replace substandard crossing structures.

Representatives of Partnership organizations as well as other agencies and organizations that have been providing input and advice to the project make up the River and Stream Continuity Advisory Committee.

In this reporting year, WRRC staff reviewed entered data for inconsistencies and to account for a change in field measurements. We also wrote a manual for survey coordinators, and continued acting as database coordinator.

3. Other Information Transfer/Outreach

Jerry Schoen assisted the UMass IT Minor Program and 5 College Inc. in planning and organization of the Information & Communication Technology Summit, held at the Campus Center on March 31, 2011. The summit explored collaborative paradigms in pedagogy, and how ITC can cultivate an environment for a first class education. http://www.umass.edu/itprogram/ictsummit2011.html

4. 2011 Water Resources Conference

Principal Investigators: Dr. Paula Rees, Marie-Françoise Hatte, MA Water Resources Research Center
Start Date: 3/1/2010
End Date: 2/28/2011
Reporting Period: March 1, 2010 – June 30, 2011
Funding Source: USGS 104B
Descriptors: Water Quality, Water Quantity, Climatological Processes
Focus Categories: Education; Management & Planning; Climatological Processes

The Water Resources Research Center organized the eighth annual Water Resources Conference on the UMass Amherst campus on April 7, 2011. While the conference took place in April 2011, most of the work for this conference was accomplished in the reporting period. The Cooperative State Research, Education, and Extension Service New England Regional Program again cooperated in planning the conference. Four additional co-sponsors helped underwrite the cost of the conference.

Thirty-two posters were presented and there were 30 platform presentations in three concurrent sessions. The presentations were grouped into the following 9 sessions:

- Climate Change and Stream Crossings in the Northeast
- Monitoring and Detecting Harmful Algal Blooms
- Nutrients Management in Water

- Climate Change Adaptation Implementation Strategies
- Fish Passage and Stream Continuity
- Findings of the Connecticut River Targeted Watershed Initiative
- Climate Change Adaptation and Decision Making
- Tools for Water Management in the Connecticut River Basin
- Stormwater and Low Impact Development

There were three Plenary Addresses at the beginning of the conference:

- "The University Perspective" by Rick Palmer, Professor and Department Head, Dept. of Civil and Environmental Engineering, UMass Amherst
- "The State of Massachusetts Perspective" by Vandana Rao, Assistant Director for Water Policy, Mass. Executive Office of Energy and Environmental Affairs
- "The New England Regional Perspective" by Jessica Cajigas, Environmental Analyst, New England Interstate Water Pollution Control Commission.

The Keynote Address was given by Dr. Richard Vogel, Professor of Civil and Environmental Engineering and Director of the Graduate Program in Water: Systems, Science and Society, Tufts University, on "Water Resources Planning in a Changing World."

181 people registered for the event, representing 14 colleges and universities, 23 companies, 15 governmental agencies, 4 non-profit organizations, and 13 municipalities.

Twenty-four students (from 6 different institutions) participated in the Best Student Poster Competition, evaluated by 14 judges. Liam Bevan of UMass Amherst Geosciences (and a WRIP research project awardee this fiscal year) and Barbara DeFlorio of UMass Amherst Veterinary & Animal Sciences tied for first place. Bevan's poster was entitled "Water Flux at Till/ Bedrock Interfaces in Central Massachusetts." DeFlorio's poster's title was: "Optimizing Vegetative Filter Strips Treating Runoff from Turf."

Students supported by project
1 BS student in Mathematics at UMass Amherst
1 BS student in Chemical Engineering at UMass

5. Information Technology

WRRC is involved in three projects using information technology for environmental research, teaching and outreach.

1) The Center's work on the EPA-funded Tri-State Watershed Initiative in the Connecticut River watershed is described above.
2) The Center collaborated with the UMass Center for Educational Software Development, the Biology Departments of the Boston and Amherst campuses, and the Landscape Architecture and Regional Planning Program on a successful proposal to the National Science Foundation's Course, Curriculum and Laboratory Improvement Program (CCLI, now renamed to Transforming Undergraduate Education in Science, Technology, Engineering and Mathematics; TUES). This project, begun in July 2010 and named SeeTrees, is based on developing content and testing software and hardware options to enhance learning and scientific discovery in field courses. This is being done to support courses with a significant field component in which the students do

field projects on plants. The essence of the project is to improve the way students learn to see and communicate their observations among their peers and to the broader community. New instructional methods are being introduced into the classes which involve mobile technologies including digital cameras, PDAs and cell phones in conjunction with open source Internet tools. These are used in the field and laboratory to (1) familiarize students with plants, plant characters and habitats, (2) identify species and habitats, (3) record observations that are automatically geo-tagged and time stamped, and (4) review and synthesize field observations. Jerry Schoen of the Center is the project manager.

3) The Center is collaborating with the Center for Educational Software Development, Department of Environmental Conservation, Trout Unlimited and other partners to develop and raise funds for The River's Calendar, a citizen-science climate change monitoring project focused on impacts to phenology of riparian areas in coldwater fisheries, and consequent recreational and economic impacts.
4)

Other Activities

1. Environmental Analysis Laboratory

Reporting Period: July 1, 2010 – June 30, 2011

The Environmental Analysis Laboratory (EAL) was created in 1984 by WRRC to assist the Acid Rain Monitoring Project (ARM) by analyzing more than 40,000 samples for a suite of 21 parameters. Since 1988, the Lab has provided services to a wide range of off-campus and on-campus researchers. EAL provided chemical analysis of water, soils, tissue, and other environmental media for University researchers, public agencies, and other publicly supported clients. The EAL conducts a wide variety of analyses to support environmental research, management, and monitoring activities. EAL provides high quality analytical services for inorganic substances in water including nutrients, inorganic anions, and metals and has especially distinguished itself in the analysis of trace levels of phosphorus.

In this past year, EAL continued to provide laboratory support for the Acid Rain Monitoring Project, including a quality-control program for pH and alkalinity and analytical determinations for a suite of 15 parameters. The quality-control program for volunteer-monitoring groups continued for pH, alkalinity and dissolved oxygen. Analytical services were provided for four university researchers, primarily for anions and cations. Collaboration with Dr. Julian Tyson of the Chemistry Department continues and his lab has been responsible for sample analyses and for new methods development.

The management structure of the lab offered a unique opportunity to provide both the campus community and others with specialized methods development as well as basic analytical services.

Students Supported
1 PhD Student, Chemistry Department.

2. Fish Passage Conference

WRRC staff cooperated on a conference, the National Conference on Engineering & Ecohydrology for Fish Passage, organized by UMass Amherst Civil and Environmental

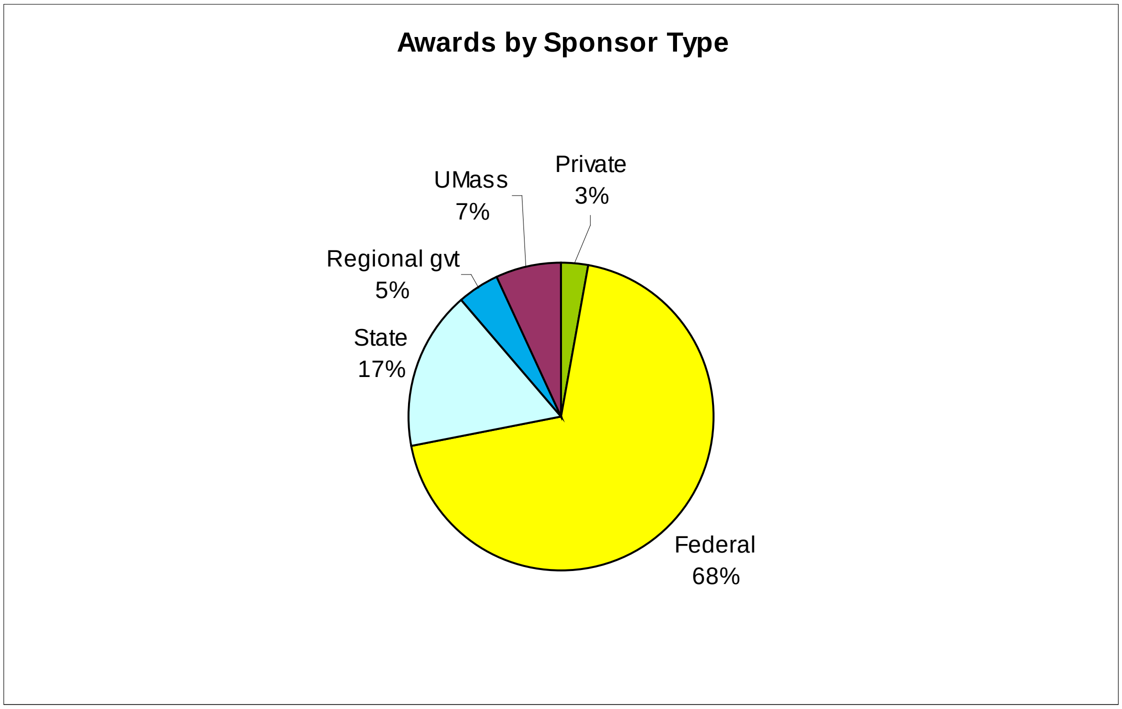
Engineering. We consulted with CEE at various steps on how to plan and run an conference, created and managed a web site, created a call for abstracts and managed submissions. The conference took place at UMass Amherst June 27-29, 2011.

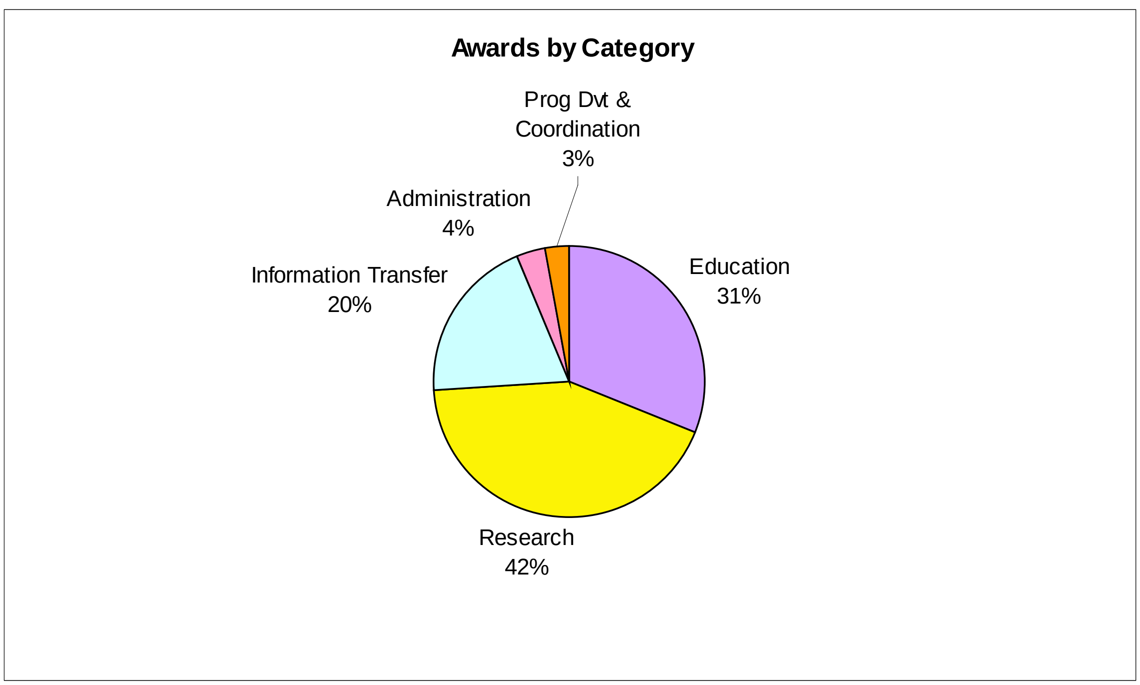
Financial Overview

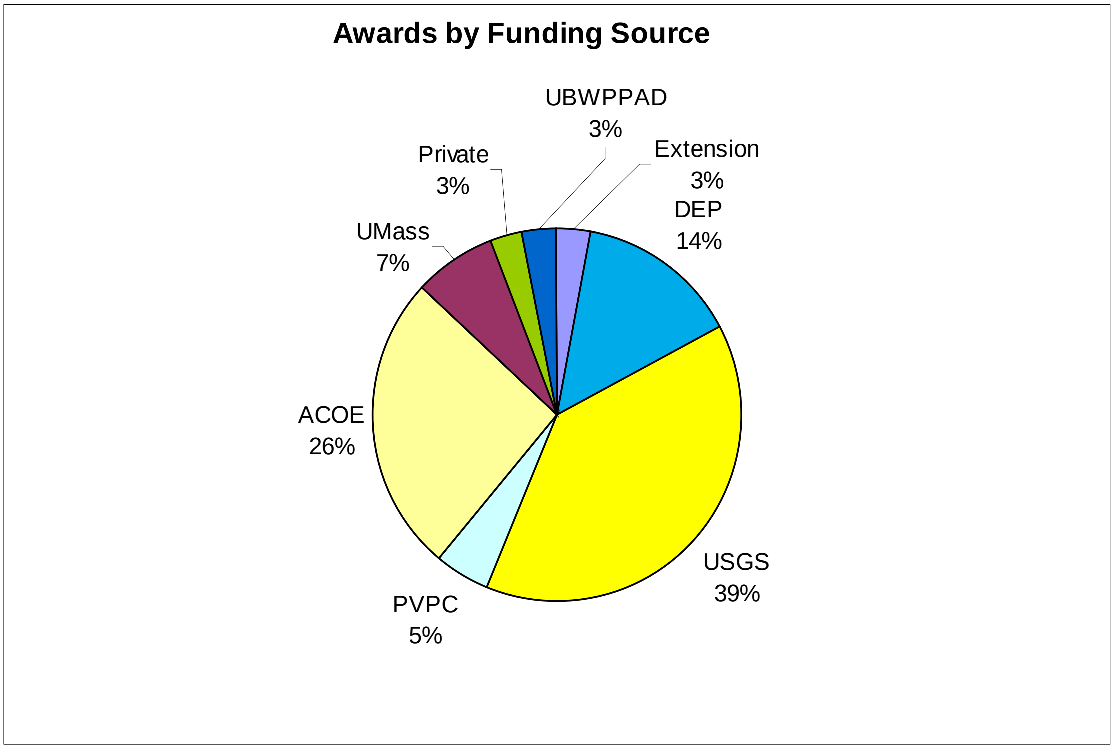
Center revenues come strictly from grants and contracts. The University of Massachusetts contributes 20% of the salary for a half-time Director and also provides physical facilities for the WRRC.

Total revenues amounted to $455,394:

ACOE	$116,433
USGS 104G:	$ 79,676
USGS 104B:	$ 92,335 broken down as follows:
	$29,937 Douglas research project
	$20,473 Conference
	$16,736 Administration
	$5,191 ARM Project
	$5,000 Yu research project
	$4,998 Boutt research project
	$5,000 Ash research project
	$4,5,000 Guswa research project
UMass (Director)	$ 25,628
Extension	$ 13,737
ARM Project	$ 25,000
CT River TWI	$ 20,671
CT River 604(B)	$ 9,533
MASTEP	$ 28,461
ICT Conference	$ 5,873
SeeTrees	$ 11,745
Blackstone River	$ 13,010
Conference Revenues	$ 8,800
EAL	$ 4,402
Fish Passage	$ 14,582


Awards by Sponsor Type
UMass
7%
Private
3%
Regional gvt
5%
State
17%
Federal
68%


Awards by Category
Prog Dvt &
Coordination
3%
Administration
4%
Information Transfer
20%
Education
31%
Research
42%


Awards by Funding Source
UBWPPAD 3%
Extension 3%
DEP 14%
Private 3%
UMass 7%
ACOE 26%
USGS 39%
PVPC 5%