A REDUCED PARAMETER STREAM TEMPERATURE MODEL (RPSTM) FOR FLUVIAL ECOSYSTEM FORECASTING

by

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To my son, husband, sister, dad and mom;

for your endless encouragement, support, and love to make this come true.

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CHAPTER I

INTRODUCTION

Water temperature in streams and rivers is an important attribute of water quality. It directly controls and often dominates the spatial and temporal distributions of most physical, chemical, and biological processes in rivers (Caissie *et al.*, 2005; Wehrly *et al.*, 2006). For example, stream temperature is universally recognized as a crucial variable affecting fish ecology and community dynamics (Diana, 1983, 1984; Bartholow, 1991; Bartholow *et al.*, 1993; Kapetsky, 2000; Borman & Larson, 2003; Wehrly *et al.*, 2003; Horne *et al.*, 2004; Wehrly *et al.*, 2006). Other examples, such as rate of evaporation and condensation, water density, viscosity, dissolved oxygen content, thermal pollution, chemical reactions, growth rate of aquatic organisms, and many others, are strongly tied with water temperature. Consequently, to examine the health of an aquatic ecosystem, it is critical to understand how water temperature changes by natural and anthropogenic stressors.

Stream water temperature changes naturally. It varies from headwaters to outlet because the water is in contact with the atmosphere and the streambed for cumulatively longer times (Caduto, 1990; Leopold, 2003; Allan, 2004). It can also be altered when tributaries from different geomorphic settings enter or when additional groundwater inputs recruit into the stream (LeBlanc *et al.*, 1997; Wehrly *et al.*, 2006). Seasonal variations in stream water temperature may reflect the changes in local weather condition, as well as the portion of groundwater contribution, while diurnal stream temperature variations are highly related to the air temperature and the receiving solar radiation (Sinokrot & Stefan, 1994; Bartholow, 2000). However, natural thermal regimes can be further altered by human activities. Urbanization, landuse change, water demand, and human-induced climate change can seriously affect stream water temperatures to a great extent.

Many different water temperature models have been developed. These models can be generally characterized as either being based on empirical methods or on numerically simulated heat balance equations. The empirical methods are almost always designed for specific locales (Baker *et al.*, 2005), making it inappropriate to use for capturing thermal dynamics for other rivers. Besides, these empirical models provide no basis for anticipating effects of future changes in landcover, hydrology, or climate.

The other type of temperature models is physically-based models. These are models that posit the complex dynamics of heat exchange at a particular location. Several problems associated with such models arise in the context of basin-wide ecological assessments. First, heat balance models are usually steady state models, which assume flows are essentially constant for the entire averaging period, and boundary conditions being simulated are homogeneous and constant (Bartholow, 2000). Second, most of them are based on the estimation and time averaged solutions of full energy budgets, including radiation, convection, conduction, evaporation, and advection. The parameters required for a full energy budget model are many, and most are difficult to estimate. Obtaining estimates of the parameters for a large number of sites is often considered simply impractical. Third, these models often ignore groundwater advection when modeling large rivers (Brown, 1969). However, a big river is almost always composed from several smaller headwater streams, and river valleys typically provide both substantial elevational head and conductive alluvial depths. Thus groundwater advection often has significant impacts on stream temperature under low flow conditions (O'Neal, 1997; Wiley *et al.*, 1997; Baker *et al.*, 2001, 2003). For the aforementioned reasons, and to overcome the difficulties of parameterization and coefficient estimation, a new water temperature model is desired.

The main purpose of my dissertation is to develop a new, simpler (more abstract) approach to modeling stream temperature which facilitates larger-scale modeling of an entire watershed network, and captures the dynamic hydrographic character typical of real river systems. Without sufficient long-term water temperature data to provide an empirical function of catchment properties to the predictions of water temperature across the Muskegon River Watershed, I chose the physically-based heat balance approach to accommodate changes in the energy balance that arise from changing environmental conditions, and transform to changes in water temperature (TVA, 1972; LeBlanc *et al.*, 1997). The thermodynamic principles are the same from other existing physically-based models. The calculation approach was quite different. I used only the dominant physical processes to simplify parameterization as much as possible. This helped reduce the complexity of parameterization that existed in most physically-based models, without sacrificing too much of the needed level of accuracy. My simplified stream temperature model required fewer site-specific parameters, such as riparian shading, wind speed, cloud cover, and humidity. More importantly, since my approach involves explicit coupling to dynamic surface runoff and groundwater models, it provides an obvious context for exploring the effects of ecological changes in both groundwater and surface runoff hydrology.

Here, I focus and discuss the development of the Reduced Parameter Stream Temperature Model (RPSTM) and its applications, including in the following five chapters:

Chapter II - The Reduced Parameter Stream Temperature Model:

This chapter consists of the theoretical development of the Reduced Parameter Stream Temperature Model (RPSTM), the calibration and validation results, and a sensitivity analysis.

Chapter III - Application I– Groundwater Withdrawal:

This chapter illustrates and quantifies patterns of downstream change in thermal regimes associated with water withdrawal in Michigan rivers, and examines the implications of such changes on the distribution of coldwater fish assemblages.

Chapter IV - Application II– Dam Removal:

This chapter compares the the impacts of dams and their removal on lower river thermal regimes, quantifies their impacts on the timing of the early life history stages of Great Lakes anadromous fishes, and explores the potential amounts of thermally suitable habitats above the dams for these fishes.

Chapter V - Application III– Climate Change:

This chapter presents a general methodology for water temperature prediction in the light of climate change scenario, provides a preliminary estimation of the impacts of climate change on thermal regimes, and assesses the potential shifts in early life stages of targeted game fish species (steelhead, walleye, and chinook salmon) in the Muskegon River Watershed.

Chapter VI Conclusion and Future Work:

This chapter briefly summarizes the RPSTM developed in this dissertation, and the findings from the three applications.

CHAPTER II

A REDUCED PARAMETER STREAM TEMPERATURE MODEL (RPSTM) FOR BASIN-WIDE SIMULATIONS

2.1 Abstract

Water temperature is a crucial predictor of species composition for fish and invertebrate communities in rivers. To help understand how changes in landscape hydrology and climate are likely to influence riverine communities, I constructed a Reduced Parameter Stream Temperature Model (RPSTM), a spatially explicit, easy to parameterize mechanistic heat balance model. Interfaced with dynamic hydrological models, it allows basin-wide, accurate estimation of stream temperatures under the impacts of changing climate, hydrologic, and land cover scenarios. In a validation test I used RPSTM to simulate daily water temperature at 119 locations in the Muskegon River Watershed from 1985 to 2005. A sensitivity analysis was also conducted. The results showed that RPSTM performed well (Maximum Absolute Error, MABSE, was generally about 1°C for most sites validated) in estimating stream temperature along the Muskegon river network. The sensitivity analysis revealed that RPSTM was most sensitive to the parameterization of local air temperature, hydraulic depth, and solar radiation, but relatively insensitive to rate of surface flow. RPSTM was moderately sensitive to rates of groundwater discharge, network travel time, and the parameterization of the heat transfer coefficient. This modeling approach is easily integrated into complex multi-modeling systems that are now being used to evaluate effects of long term changes in land use, climate change, and river management on river fish communities.

2.2 Introduction

Stream water temperature is a critical variable shaping the biology of riverine ecosystems. It exerts a fundamental control over distribution and abundance of fish and other assemblages through both physiological (rates of feeding, growth, and metabolism) and behavioral (preferential immigration and emigration) mechanisms (Blann & Nerbonne, 2002; Wehrly *et al.*, 2003; Brazner *et al.*, 2005; Cheng *et al.*, 2006; Wehrly *et al.*, 2006). Relatively small changes in temperature distributions can potentially radically alter the structure of fish populations (Diana, 1983, 1984; Cardwell *et al.*, 1996; Kapetsky, 2000; Borman & Larson, 2003; Wehrly *et al.*, 2003; Horne *et al.*, 2004; Brazner *et al.*, 2005; Cheng *et al.*, 2006; Wehrly *et al.*, 2006). Consequently, models linking hydrologic, channel and riparian conditions to the longitudinal distribution of temperature have played a key role in the study of river fish ecology (Mitchell, 1999; Wehrly *et al.*, 2009; Wiley *et al.*, 2010).

Stream temperature models can be generally characterized as either being based on empirical methods or on numerically simulated heat balance equations. Empirical relationships were based on time-series and/or geographic observations of stream temperature and both local and catchment properties. Although water temperature naturally varies with locations, geomorphic settings, and hydraulic routing, it almost always follows both diurnally and seasonally cyclic patterns. These patterns are mainly a response to temporal variation in solar radiation and air temperature (Leopold, 2003; Allan, 2004). Longterm stream temperature monitoring can provide empirical summary descriptions for a locale (Baker *et al.*, 2005), but it provides no basis for anticipating effects of future changes in landcover (which strongly influence hydrologic routing) or climate. Moreover, these empirical methods often assume similarity of the parametric influences to the past, precluding the consideration of future natural or human-induced impacts on stream temperatures. Since both land use and climate change seem an inescapably aspect of our near future, the need for more robust ways of predicting changing thermal environments in rivers is self-evident.

With the growing concern about ecological impacts of global warming, physically-based heat balance models are receiving renewed interest (Bartholow *et al.*, 2005; Chapra & Tao, 2006; Fujihara *et al.*, 2008). Heat balance models can accommodate changes in the energy balance that arise from changing environmental conditions (TVA, 1972; LeBlanc *et al.*, 1997). They also preserve each individual heat exchange term, which make quantifying single or multiple heat exchange on stream temperatures possible (Bartholow, 2000; Borman & Larson, 2003). Therefore, these physically-based heat balance models can be reasonably used to predict in hypothetical scenarios, such as the impacts of dam removal (Bushaw-Newton *et al.*, 2002; Bartholow *et al.*, 2005), in-stream flow manipulations (Zorn *et al.*, 2008), and anticipated climate change (Sinokrot *et al.*, 1995). Unfortunately, heat balance models involve a high level of complexity. The most widely used temperature models of this type depend on time averaged solutions of full energy budgets, and therefore, require detailed local parameterization to capture major heat fluxes into and out of streams (TVA, 1972; LeBlanc *et al.*, 1997; Bartholow, 2000; Borman & Larson, 2003; Chapra & Tao, 2006). These include: radiation, convection, conduction, evaporation, and advection. However, in many practical applications, there are insufficient data at the required scales to estimate parameter values needed in a spatially explicit way. As a result, their enormous data requirements for parameterizations have made them hard to apply across larger river networks and other regional-scale ecosystems (Crittenden, 1978).

In addition, most existing heat balance models use steady state solutions to simplify the calculations. These models assume flows are constant for the entire estimation period and boundary conditions are homogeneous and constant (Bartholow, 2000; Borman & Larson, 2003; Bartholow *et al.*, 2005). However, the temperature dynamics of streams and rivers are linked directly to dynamic water discharges, and it is difficult to capture this in steady state models.

As such, I developed a new Reduced Parameter Stream Temperature Model (RPSTM), which was conducted with the intent of capturing the correct parametric influences without the need for detailed site-specific data. It was designed to run explicitly in conjunction with hydrologic models' (e.g. HEC-HMS (Hydrologic Engineering Center's Hydrologic Modeling System)) simulations of daily flows through large stream networks and is currently being used in an existing watershed multi-modeling system (MREMS; Muskegon River Ecological Modeling System) (Wiley *et al.*, 2003a,b; Seelbach & Wiley, 2005; Wiley *et al.*, 2010). RPSTM provides a new, simpler (more abstract) approach to modeling stream temperature which facilitates large-scale implementation for an entire watershed network and allows dynamic thermal characterization typical of real river systems. Moreover, by combining several heat flux terms into one term (Lanini *et al.*, 2004; Sridhar *et al.*, 2004), it requires fewer data while still generating accurate stream temperature estimations. Most importantly, since this approach involves explicit coupling to dynamic surface runoff and groundwater models, it provides an obvious avenue for exploring the effects of changes in both groundwater and surface runoff hydrology.

This paper comprises of several parts. I first describe the basic heat balance equations that govern heat flux in river channel systems. Next I show the derivation of the simplified RPSTM heat balance equation from the full energy balance. Then I illustrate how the basic RPSTM approach can be implemented in a hydrologic modeling context; focusing on its implementation in the Muskegon River (a 7,057 km^2 catchment tributary to Lake Michigan). Finally I explore the results of the simulation in a series of validation tests and a sensitivity analysis.

2.3 Methods

2.3.1 General Basis: The Heat Balance Approach

Temperature is a measure of the amount of energy a system contains. Heat flux (dq/dt) in the full energy budget includes three standard mechanisms of heat transport: radiation, convection, and conduction.

In a stream system, energy flux related to radiative processes include solar radiation (i.e. shortwave radiation), long wave radiation, and back radiation. Consequently, physically-based in-stream temperature models treat the heat exchange processes as a combination of these major thermal processes (TVA, 1972; LeBlanc *et al.*, 1997; Bartholow, 2000; Borman & Larson, 2003; Chapra & Tao, 2006):

$$\frac{dq}{dt} = (SR + LR - BR) + C_v + C_d \tag{2.1}$$

where dq/dt is heat flux transferring through a unit surface over a specified unit of time (J/m^2h), SR is heat flux from solar radiation at the water surface (J/m^2h), LR is heat flux from longwave radiation (J/m^2h), BR is heat flux of back radiation from the water (J/m^2h), C_v is heat flux of convection (J/m^2h), C_d is heat flux of conduction (J/m^2h).

Shortwave solar radiation is typically the largest thermal input and strongly affects in-stream water temperature (TVA, 1972; LeBlanc *et al.*, 1997) , but its quantity is highly variable both within and between days. Latitude and longitude, as well as the attenuation rate, also affect the quantity of solar radiation reaching a stream. While latitude and longitude are easily incorporated into a model, attenuation of solar radiation affected by local variations in atmospheric transmission, cloud cover, reflections, and canopy shade, is extremely difficult to specify other than by local measurement.

Longwave radiation is the radiation emitted from nearby objects including the surrounding atmosphere and ground itself. It falls within the infrared portion of the spectrum (Adam & Sullivan, 1990). The downward flux of longwave radiation from the atmosphere to a stream can be calculated using the Stefan-Boltzmann Law:

$$LR = \sigma(T_{air} + 273)^4 \varepsilon_{air}(1 - R_L) \tag{2.2}$$

where T_{air} is air temperature (°C), σ is the Stefan-Boltzmann constant ($5.67 \times 10^{-8}\ W/m^2K^4$), ε_{air} is emissivity of the atmosphere, and R_L is the reflection coefficient, which is typically assumed to equal 0.03, and is negligible.

A stream also radiates back to the atmosphere, the ground near the stream, and the riparian vegetation. The amount of this back radiation from the water surface can also be approximated by the Stefan-Boltzmann law:

$$BR = \varepsilon\sigma(T_w + 273)^4 \tag{2.3}$$

where T_w is stream surface temperature (°C), and ε is emissivity of water.

Besides radiation, convection, including atmospheric convection and evaporation, is also important to the overall energy budget of a stream. Convection occurs mostly across the air-water interface when air and water temperature differs. As a consequence, the rate of the convective heat flux (C_v) can be computed as (Bowen, 1926; Kreith, 1973):

$$C_v = h_c(T_{air} - T_w) \tag{2.4}$$

where C_v is heat flux of convection (J/m^2h), T_w is stream surface temperature (°C), T_{air} is air temperature (°C), and h_c is a heat transfer coefficient ($J/m^2h°C$). The heat transfer coefficient (h_c) can be calculated as:

$$\frac{h_c}{k_e} = 1.5 \times 10^6 \tag{2.5}$$

where k_e is the evaporative coefficient for evaporation, and can be estimated as:

$$k_e = 1.74 \times 10^{-6} \times (1 + 0.72V_a) \tag{2.6}$$

where V_a is wind velocity (m/s).

2.3.2 RPSTM Theoretical Derivation

Streambed and stream water temperatures typically follow variations in air temperature, but are lagged with increasing depth (Acornley, 1999; Brown *et al.*, 2006). For example, water temperature in a stream responds to the atmospheric conditions in the time constants on the order of about 40 hours for every meter of water depth (Edinger *et al.*, 1979; Sinokrot & Stefan, 1993). This reflects mostly rapid adjustments to changes in net radiation flux and convective flux. Conduction is a much slower process in the time scale on the order of weeks (Sinokrot & Stefan, 1993). In shallow streams, measurements of streambed temperature profiles show that the temperature of the streambed surface is almost equal to the temperature of the stream water, implying that stream water temperature is not very sensitive to the streambed thermal conductivity (Sinokrot & Stefan, 1993; Lanini *et al.*, 2004; Sridhar *et al.*, 2004; Cadbury *et al.*, 2008). As a first step towards a more simplified heat flux representation, I assume the effect of streambed conduction on stream water temperature is small enough to ignore and drop it from the heat balance altogether.

In general, water temperature in a stream reaches a thermal equilibrium in balance with the environmental conditions. If there is temperature difference between stream water and its surroundings, they will exchange thermal energy (heat exchange) until a new thermal equilibrium is reached. To dynamically predict in-stream temperature, it is critical to calculate the total heat exchange ΔH (KJ) of a reach:

$$\Delta H = \Delta q \times A \tag{2.7}$$

where Δq is the amount of heat transferred through a unit area (KJ/m^2), and A is the total surface area of the reach. Once the heat exchange of a unit air-water interface is determined, one can calculate the temperature of a reach. Based on the definition of specific heat (c), the change in stream temperature (ΔT) is determined as:

$$\Delta T = \frac{\Delta H}{mc} \tag{2.8}$$

where m is the mass of water in the reach. Therefore, the continuous change in stream temperature (dT_w/dt) can be written as:

$$\lim_{\Delta t \to 0} \frac{\Delta T_w}{\Delta t} = \frac{dT_w}{dt} = \frac{1}{mc}\left(\frac{dH}{dt}\right) = \frac{1}{mc}\left[\frac{(dq \times A)}{dt}\right] \tag{2.9}$$

Assuming thermal uniformity within a channel both vertically and horizontally, as well as a constant interface area (A), the rate of temperature change can be simplified as:

$$\frac{dT_w}{dt} = \frac{A}{mc}(\frac{dq}{dt}) = \left[\frac{A}{(A \times d \times \rho) \times c}\right](\frac{dq}{dt}) = \frac{1}{d \times \rho \times c}(\frac{dq}{dt}) \tag{2.10}$$

where d is hydraulic depth, ρ is water density, and dq/dt is heat flux. Substituting heat fluxes from Equation (2.1), the rate of water temperature change can be written as:

$$\begin{aligned} \frac{dT_w}{dt} &= \frac{1}{d \times \rho \times c}(SR + LR - BR + C_v + C_d) \\ &= \frac{1}{d \times \rho \times c}\left[SR + \sigma(T_{air} + 274)^4 \varepsilon_{air}(1 - R_L) - \varepsilon\sigma(T_w + 273)^4 + h_c(T_{air} - T_w) + 0\right] \end{aligned} \tag{2.11}$$

The humidity of the atmosphere close to the water surface is usually large, so I used the emissivity of water in place of the emissivity of air. Equation 2.11 can then be rewritten as:

$$\frac{dT_w}{dt} = \frac{1}{d \times \rho \times c}\{SR + \sigma\varepsilon[(T_{air} + 273)^4] + h_c(T_{air} - T_w)\} \tag{2.12}$$

In Equation (2.12), the quadratic terms can be expanded as:

$$\begin{aligned} (T_{air} + 273)^4 - (T_w + 273)^4 &= T_{air}^4 + 4T_{air}^3 \times 273 + 6T_{air}^2 \times 273^2 + 4T_{air} \times 273^3 + 273^4 \\ &\quad - T_w^4 - 4T_w^3 \times 273 - 6T_w^2 \times 273^2 - 4T_w \times 273^3 - 273^4 \\ &= \left(T_{air}^4 - T_w^4\right) + 4 \times 273\left(T_{air}^3 - T_w^3\right) + 6 \times 273^2\left(T_{air}^2 - T_w^2\right) \\ &\quad + 4 \times 273^3\left(T_{air} - T_w\right) \\ &= (T_{air} - T_w) \times [4 \times 273^3 + 6 \times 273^2\left(T_{air} + T_w\right) \\ &\quad + 4 \times 273\left(T_{air}^2 + T_{air}T_w + T_w^2\right) + \left(T_{air}^3 + T_{air}^2 T_w + T_{air}T_w^2 + T_w^3\right)] \end{aligned} \tag{2.13}$$

Usually, $T_{air} + T_w$ (°C) is much smaller than 273 allowing Equation (2.13) to be simplified as:

$$(T_{air} + 273)^4 - (T_w + 273)^4 \approx 4 \times 273^3\left(T_{air} - T_w\right) \tag{2.14}$$

Consequently, Equation (2.12) can be simplified as:

$$\begin{aligned}
\frac{dT_w}{dt} &= \frac{1}{d \times \rho \times c}\left[SR + 4 \times 273^3 \sigma\varepsilon (T_{air} - T_w) + h_c (T_{air} - T_w)\right] \\
&= \frac{1}{d \times \rho \times c}(SR) + \frac{1}{d \times \rho \times c}\left(4 \times 273^3 \sigma\varepsilon + h_c\right)(T_{air} - T_w) \\
&= \frac{1}{d \times \rho \times c}(SR) + (-k)(T_w - T_{air})
\end{aligned} \tag{2.15}$$

where k is the heat exchange coefficient, which can be expressed as:

$$k \approx \frac{(4 \times 273^3 \sigma\varepsilon + h_c)}{d \times \rho \times c} \propto \frac{1}{d} \tag{2.16}$$

The solving of the first-order ordinary differential equation (Equation (2.15)) allows us to model stream temperature of a reach as:

$$T_{wo} = T_{air} + \frac{SR}{kd\rho c} + \left(T_{wi} - T_{air} - \frac{SR}{kd\rho c}\right) \times e^{-kt} \tag{2.17}$$

where T_{wo} is the stream temperature at the bottom of the stream reach prior to mixing with water from other reaches (°C), T_{wi} is the stream temperature at the start of the reach (°C), and t is travel time of water in each stream segment (h). This simplified equation makes the prediction of in-stream temperature from energy budget substantially easier. More than that, it can be used for a wide variety of conditions using only seven parameters, and can be easily estimated.

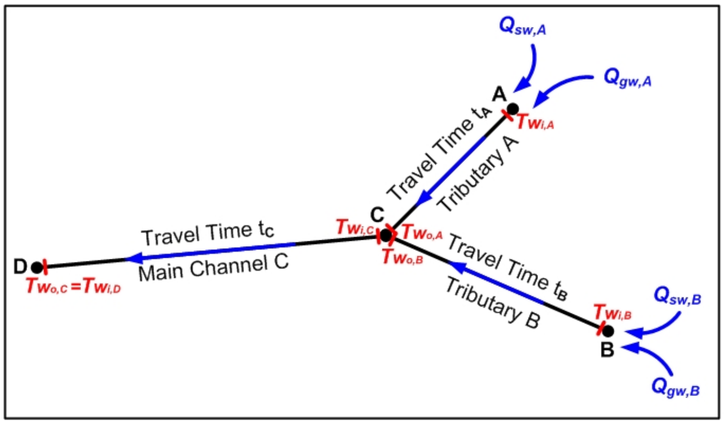
2.3.3 Implementation in RPSTM

1. Major Input Variables and Types

   RPSTM is implemented using a channel schematic framework similar to that used in HEC-HMS (Anonymous, 2001). In order to use the RPSTM approach, reach specific estimates of daily surface water and groundwater flows must be supplied by interfacing hydrological models. I use the outputs from surface water and groundwater models (ILHM; the Integrated Landscape Hydrology Model) (Hyndman *et al.*, 2007), routed in HEC-HMS to obtain channel flows and groundwater discharges (m^3/s), and reach-specific travel times (h) (Wiley *et al.*, 2010). Other required information includes catchment area (km^2), observed daily average solar radiation (kJ/cm^2), and observed daily average air temperature (°C). Solar radiation and air temperature data used in the simulations were developed by A. Kendall and D. Hyndman at Michigan State University from NOAA Doppler Radar data and local co-op weather gauges (Hyndman *et al.*, 2007). Additional necessary parameters include groundwater temperature (°C), the heat exchange coefficient (k), hydraulic depth (m), residence time of the reservoirs (h), and the solar coefficient. The parameterization processes will be discussed in section 2.3.4.

2. River Channel Configuration File

   In RPSTM, a river channel configuration file is used to route the calculations and encodes a schematic representation of the channel system. If a river is composed of two tributaries and a main channel, it is represented by 7 elements (i.e. 4 nodes and 3 lines), where lines AC and BC represent tributaries A and B, and line CD is the main channel C. In RPSTM, the head waters of tributaries (i.e., nodes A and B) consist of two components– surface water (Q_{sw}) and groundwater discharge (Q_{gw}). The node at the confluence of the two tributaries (i.e., node C) is defined as a junction (Figure 2.1). In the configuration file, I first construct a n by n table, where n is the number of nodes. At the outer array, tributaries and rivers are first arranged from up- to down-streams, and then from lower to higher orders where applicable. At the inner arrays, I use 0 and 1 to represent the absence and presence of a stream node. First order streams (i.e., nodes A and B) will always be presented as themselves, whereas junction and main channel nodes are presented by whatever is above or confluent into them (Figure 2.1. The mainstem temperature is simulated by fluxes from both upstream and several important tributaries. All groundwater inputs are assumed to occur at the head of each reach. To simplify, I assume conservation of mass such that after mixing at the junction, water travels to the next junction without considering



Configuration File:

	A	B	C	D
A	1	0	0	0
B	0	1	0	0
C	1	1	0	0
D	0	0	1	0

Figure 2.1: An example of a river composed of two tributaries and one main channel, and how to set up its configuration file before running RPSTM.

any evaporation loss.

3. Temperature Estimation

   To estimate the temperature of natural streams, it is critical to account for the more or less continuous mixing of water from different sources and temperatures (Adam & Sullivan, 1990). Since my model is driven by dynamic hydrological models, RPSTM takes the mixing of surface water and groundwater discharges into account to dynamically predict longitudinal variation in stream water temperature.

   For RPSTM to run, the initial stream temperature (T_{wi})(°C) is set to be the temperature at the start of each tributary, and is determined as:

$$T_{wi} = \frac{T_{sw} \times Q_{sw} + T_{gw} \times Q_{gw}}{Q_{sw} + Q_{gw}} \tag{2.18}$$

   where T_{sw} is head surface water temperature (°C), T_{gw} is groundwater temperature (°C), Q_{sw} is modeled surface water discharge (m^3/s), and Q_{gw} is modeled groundwater discharge (m^3/s). Precipitation, including rainfall and snowmelt, is the main source of head surface water. Head surface water temperature is assumed to be equal to the temperature of precipitation, which can be presented as local air temperature (Byers *et al.*, 1949). Surface water and groundwater discharge are presumably being mixed before water starts to travel downstream. For example in Figure 2.1, travel times of both tributaries (t_A and t_B) are pre-determined by HEC-HMS as the estimated time of concentration from the unit hydrograph (Wiley *et al.*, 2010). The output temperatures of nodes A and B at the bottom of the tributaries (Node C) before mixing is calculated by Equation 2.17. The input temperature at node C ($T_{wi,C}$) is determined by the temperatures of the two tributaries' output temperatures (i.e., $T_{wo,A}$ and $T_{wo,B}$) during the same time domain. Travel time of the tributaries A and B may not be the same. If the travel time of water in tributary A (i.e., t_A) is shorter than that of water in tributary B (i.e., t_B), then the input temperature at node C ($T_{wi,C}$) is the output temperature of node A ($T_{wo,A}$) at time t_A. At time t_B, the temperature at node C ($T_{wi,C}$) becomes a mixture of the output temperatures of nodes A and B ($T_{wo,A}$ and $T_{wo,B}$) as:

$$T_{wi,C} = \frac{T_{wo,A} \times (Q_{sw,A} + Q_{gw,A}) + T_{wo,B} \times (Q_{sw,B} + Q_{gw,B})}{(Q_{sw,A} + Q_{gw,A}) + (Q_{sw,B} + Q_{gw,B})} \tag{2.19}$$

where $Q_{sw,A}$ is modeled surface water discharge for node A, $Q_{sw,B}$ is modeled surface water discharge for node B, $Q_{gw,A}$ is modeled groundwater discharge for node A, and $Q_{gw,B}$ is modeled groundwater discharge for node B. The input temperature at node D ($T_{wi,D}$) is calculated by Equation (2.17) using the travel time of water in the main channel (t_C) and the output temperatures at node C ($T_{wo,C}$). If modeled temperatures fall below zero, the latent heat of water is considered in the temperature calculation to accommodate the phase change from water to ice.

To handle thermal stratification in lakes, residence time in the reservoir is in place of travel time to better estimate the thermal properties of reservoirs as heating sources for streams.

2.3.4 Application to A Large Complex River System: the Muskegon River

1. Study Area

The Muskegon River Watershed is located in the central lower Michigan, USA. The watershed lies within twelve counties: Kalkaska, Crawford, Clare, Lake, Mecosta, Missaukee, Montcalm, Muskegon, Newaygo, Osceola, Roscommon, and Wexford Counties. Its drainage basin encompasses about 7,057 km^2; with a mainstem length of 350 km. Originating in Higgins and Houghton Lakes, it flows southwest to the Muskegon Lake, which eventually drains into Lake Michigan (Figure 2.2).

Average annual precipitation ranges from 75 to 90 cm. Summer is usually a drier season. More than 85% of stream water volume in the tributaries of the Muskegon River is derived from groundwater (Boutt *et al.*, 2001; Pijanowski *et al.*, 2007). Thus groundwater fed tributaries are important in sustaining the cold- and cool-water fish species (Clapp *et al.*, 1990; Brazner *et al.*, 2004, 2005; Creque *et al.*, 2005; Wehrly *et al.*, 2006), which are common in much of the river system. Nonetheless, there is concern that a number of important fishes may become susceptible to thermal stresses (O'Neal, 1997; Wiley *et al.*, 2003a; DePhilip *et al.*, 2005) resulting from land use changes (e.g. urban sprawl) (Pijanowski *et al.*, 2006, 2007), climate change (Wiley *et al.*, 2010), and hydropower production (Horne *et al.*, 2004).

2. Specific Paramerterizations

The following variables were parameterized specifically for the Muskegon River–

(1) Groundwater Temperature

Groundwater temperature has been reported to be at a constant temperature as the annual average air temperature if originates from deeper groundwater, and to be more variable if it flows from shallow groundwater (Bundschuh, 1993). The shallow groundwater temperature is characterized by annual periodic temperature fluctuations, influencing by seasonal air temperature variations at the surface (Bundschuh, 1993; Lee & Hahn, 2006). In winter, shallow groundwater temperature can be 1-6 °C lower than the average annual air temperature, while in summer 2-4 °C higher (Lee & Hahn, 2006).

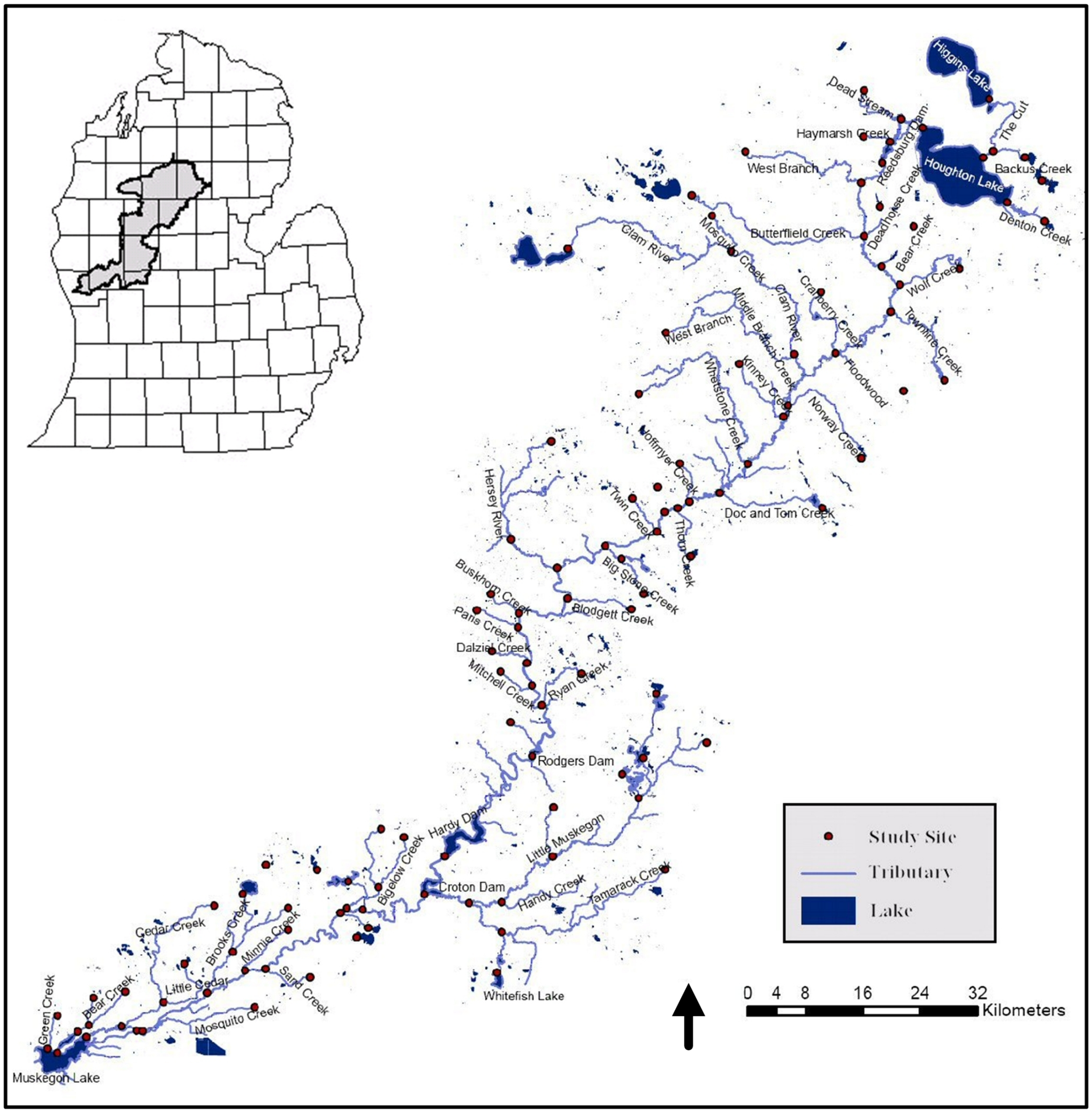


Figure 2.2: The location of the Muskegon River Watershed in Michigan (upper left corner), and the configuration of tributaries (blue lines) and study sites (red nodes).

In the Muskegon River Watershed, the springs come from shallow groundwater, or the mixing of shallow with deep groundwater. Given no direct measurements are available, and the average annual air temperature for this region was 8.9 °C from 1985-2005, estimates of groundwater temperature at 12.5°C from May 30 to October 7 (i.e. 150^{th} to 280^{th} days of the year), and 6°C for the rest days of the year were applied for the entire model.

(2) The Heat Exchange Coefficient (k)

From Equations (2.5) and (2.6), it is necessary to know the wind velocity in order to estimate the heat transfer coefficient (h_c). With an observed range of 0 to 8 m/s for the wind velocity in the Muskegon River Watershed, h_c ranges from 10.13 to 17.64, and therefore the theoretical value of k is somewhere between 0.006 and 0.019 (mh^{-1}) (Equation 2.16). I conducted several field and lab experiments to better estimate the heat exchange coefficients (k) applied in RPSTM. The k estimated from field and flume experiments in the lab agreed with theoretical values ranging from 0.006 to 0.019 h^{-1} at a depth of 1 meter for observed wind speeds in the Muskegon River watershed. In RPSTM simulations, 0.0092/depth (h^{-1}) for small channels, and 0.015/depth (h^{-1}) for large rivers and reservoirs, were applied for the entire model.

(3) Hydraulic Depth Estimates

Hydraulic depth for each node and reach in the Muskegon River was determined as:

$$d = e^{-1.446} \times D_r^{0.125} \times Q^{0.202} \tag{2.20}$$

where d is hydraulic depth, D_r is drainage area, and Q is the total channel flow, which is the summation of modeled surface water (Q_{sw}) (m^3/s) and groundwater discharges (Q_{gw}) (m^3/s) from the hydrological models. Depths were calibrated to the average depth in the low flow season from field records at approximately a hundred locations across the Muskegon River Watershed.

(4) Dam Simulation

Table 2.1: Epilimnion depths, volumes, and calculated hydraulic residence times of three major reservoirs on the mainstem Muskegon River.

Reservoirs	**Epilimnion Depth (m)**	**Flow** (m^3/d)	**Epi-Volume** ($10^6 m^3$)	**Residence Time (h)**
Rogers Dam	3	38.06	5.45	40
Hardy Dam	6	41.34	94.71	636
Croton Dam	2	52.99	9.78	51

In reservoirs, RPSTM estimates the water temperature in the epilimnion. As a result, I used epilimnion depths of the reservoirs in place of hydraulic depths (Table 2.1). Moreover, to handle thermal stratification in lakes, residence time in the reservoir was included to better estimate the thermal properties of reservoirs as heating sources for streams. The residence time of Rogers Dam, Hardy Dam, and Croton Dam were calculated by Epi-Volume/Flow (Table 2.1).

In order to offset the underestimated energy inputs in the epilimnion of hydropower reservoirs, I included the solar heating coefficient into RPSTM for calibrating modeled temperature at Rogers, Hardy, and Croton dams. The solar heating coefficient was used to reflect a reservoir's relatively large volume of water acting like an energy stock to absorb and store solar energy.

2.4 Results

Modeled stream temperatures were validated for the observed daily mean water temperatures at 10 monitoring locations. These locations represent a variety of fluvial types (reservoirs, small reaches, and main channels) and had relatively complete, multi-year data records.

Among the 10 locations, the correlations between predicted and observed daily mean water temperature ranged from 0.77 (Low Clam) to 0.99 (Big Rapids) (Table 2.2, and Figures 2.3 and 2.4). Correlation coefficient less than 0.9 occurred at two locations (Low Clam and Hersey River) in year 2000; each had with 116 days of observed daily mean

Table 2.2: Goodness-of-fit comparison between simulated and observed mean daily water temperatures (°C) at 12 locations in the Muskegon River.

Location	**Distance from mouth (*km*)**	**Year of observed data**	**No. of days of observed data**	**Correlation (*r*)**	**MABSE (°C)**
Low Clam	232.64	2000	116	0.77	1.1
Hersey River	155.12	2000	116	0.81	1.1
Little Muskegon	125.69	2000	149	0.97	1.1
Little Muskegon	125.69	1993-1994	351	0.98	1.0
Big Rapids	123.38	1998-2005	2623	0.99	1.1
Croton Dam	80.82	1995-1998	4008	0.98	1.0
Downstream of Croton Dam	69.56	1990	120	0.96	0.9
Henning	58.30	2000	118	0.90	0.7
Upper Cedar	31.71	2003-2004	656	0.90	2.0
Lower Cedar	15.17	2002-2004	578	0.95	1.6
Little Cedar	12.08	2003-2004	467	0.95	1.8

temperature data. The remaining 8 locations showed a much better relationship (correlation ranged from 0.9 to 0.99) between modeled and observed daily mean stream temperatures and much higher of observed days.

Mean absolute error (MABSE, °C) was calculated as the average of the absolute deviations of data points between modeled and observed temperatures. Overall, RPSTM demonstrates high predictive power. 10-year daily MABSEs were generally about 1 (°C) among the 10 locations. For locations in the mainstem of the Muskegon River, MABSEs were within 1.1 °C of the measured temperature, whereas for those in the tributaries, they were within 2°C. The largest MABSE (2.0 °C) occurred in Upper Cedar Creek, and the smallest (0.7 °C) occurred at Henning (Table 2.2, and Figures 2.3 and 2.4).

2.5 Sensitivity Analysis

In this paper, I performed a sensitivity analysis (Reckhow & Chapra, 1983; LeBlanc *et al.*, 1997) to investigate how simulated stream temperature was affected by the seven major parameters in RPSTM, including: air temperature (T_{air}), depth (d), the heat flux of shortwave solar radiation (SR), modeled groundwater discharge (Q_{gw}), heat transfer coefficient (h_c),

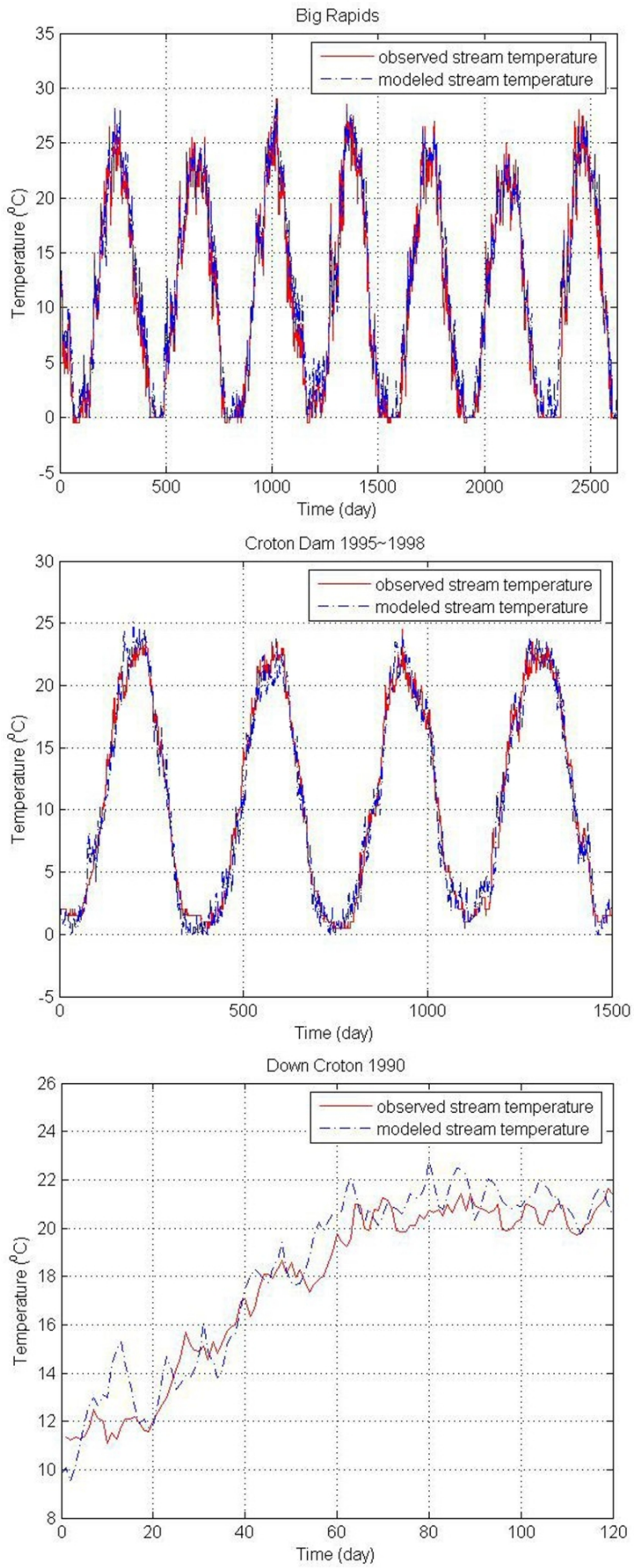


Figure 2.3: Comparison of simulated mean daily water temperature with observed data at Big Rapids, Croton Dam, and downstream of Croton Dam located in the mainstem of the Muskegon River.

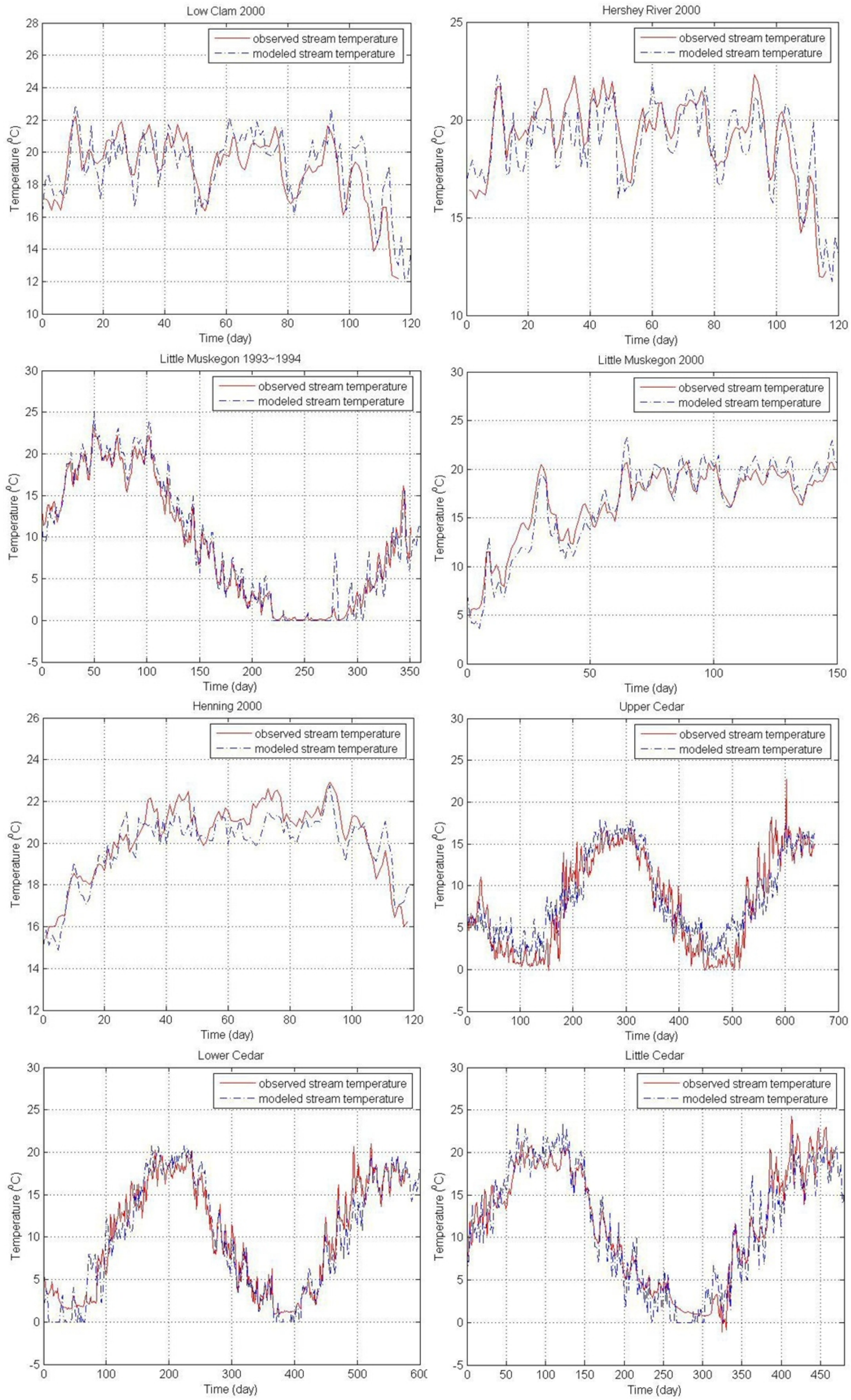


Figure 2.4: Comparison of simulated mean daily water temperature with observed data at the tributary locations of the Muskegon River.

modeled surface flow (Q_{sw}), and network travel time (t). I used 10-year July mean stream temperature as an indicator of sensitivity. It is of particular interest to us since summer stream temperature is one of the most important factors limiting the distributions of many cool- and cold-water fishes in this region (Wehrly *et al.*, 2003, 2006, 2007).

I further studied the impacts of these parameters on the major geomorphic settings in rivers. In the Muskegon River, I chose four locations representing upstream conditions (Big Rapids), largest tributaries (the Little Muskegon River), large reservoirs (Croton Dam), and small tributaries (Lower Cedar Creek confluence to the Muskegon River near its terminus). I independently evaluated the sensitivity to the seven key parameters. I adjusted values of each parameter by factors of 0.25, 0.5, 0.75, 1.25, 1.5, 1.75, and 2, and re-ran the model, evaluating changes in outputs. The results were then summarized by plotting these changes in estimated stream temperature at each site. These plots allowed us to identify the magnitude and direction (positive or negative) of a single parametric influence to stream temperatures, and to quantitatively understand their impacts on modeled temperatures.

The analysis results at four locations all showed that air temperature is the key thermal driver in my simulations (Figures 2.5 to 2.8). When adjusting air temperature by factor 2, it gave a value up to 60°C in summer when the original air temperature was about 30°C. Although this extremely high air temperature is not realistic, it was utilized to show the rate of change in water temperature by air temperature. The air temperature affected in-stream temperature linearly with sensitivity between 15.5°C/factor to 19.0°C/factor.

The second-most influential parameter is stream depth. In-stream temperature at all locations monotonically decreased while the factors of depth increased (Figures 2.5 to 2.8). The impact of depth on in-stream temperature was from −5.9°C/factor to −2.9°C/factor.

Shortwave solar radiation is the third most sensitive parameter to the in-stream temperature. Similar to air temperature, it was linearly related to in-stream temperature with sensitivity varying from 2.4°C/factor to 4.6°C/factor.

At all four locations, groundwater discharge showed an impact of −1.4°C/factor to

$-0.7°$C/factor. It was monotonically and inversely related to stream temperature. Heat transfer coefficient influenced in-stream temperature at 0.6°C/factor to 0.8°C/factor, except at the Little Muskegon River its influence (i.e. 2.2°C/factor) was much higher than the other three locations.

Surface flow appeared to be the least sensitive parameter, as it influenced in-stream temperatures among four locations at less than 0.25°C/factor in average. The influences of network travel time on stream temperature varied with location in the system as might be expected. The sensitivity analysis showed that network travel time was less sensitive to the estimated in-stream temperature at Big Rapids ($-0.1°$C/factor) and Little Muskegon River (0.7°C/factor); however, it was sensitive at Croton Dam ($-3.1°$C/factor) and Lower Cedar Creek ($-3.9°$C/factor). It is noted that at Croton Dam and Lower Cedar Creek, network travel time affected the modeled stream temperature to different levels at certain factors. For instance, from factors 0.25 to 0.5$\times$travel time at Croton Dam, network travel time on average affected in-stream temperature at $-8.6°$C/factor. At Lower Cedar Creek from factors 1 to 1.25$\times$travel time, it impacted in-stream temperature to $-23.4°$C/factor (Figures 2.5 to 2.8).

2.6 Discussion and Conclusion

RPSTM is a physical process model adopted from the energy balance equations. I further simplified the energy inputs and outputs into seven major parameters, and then translated the changes in energy balance into water temperature changes. Since the parameterization process was much reduced compared to most existing physically-based stream temperature models, it is relatively easy to apply RPSTM to large scale river watersheds. Large river watersheds usually involve more complex settings and interactions, thus their thermal conditions are more difficult to model from a highly parameterized model. Although RPSTM is more complex than a regression type model, it provides a more robust picture of stream temperatures in relation to potential changes in land use, climate, and hydrology.

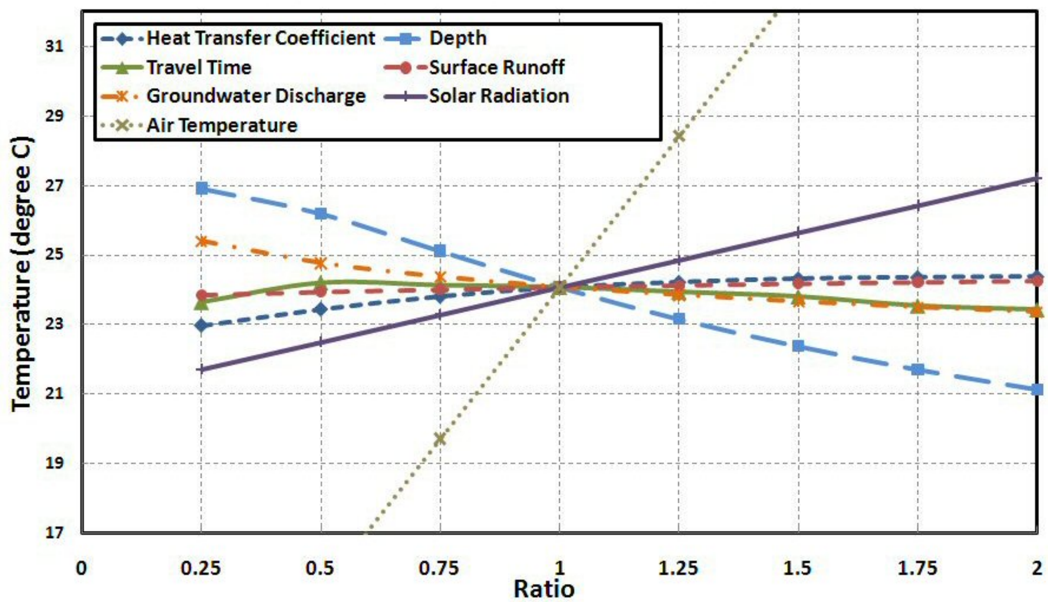


Figure 2.5: Sensitivity Analysis: Identify the influence of each parameter to the modeled July mean stream temperature by perturbing one parameter of factors ranging from 0.25 to 2, with 0.25 increments at a time at Big Rapids in the Muskegon River.

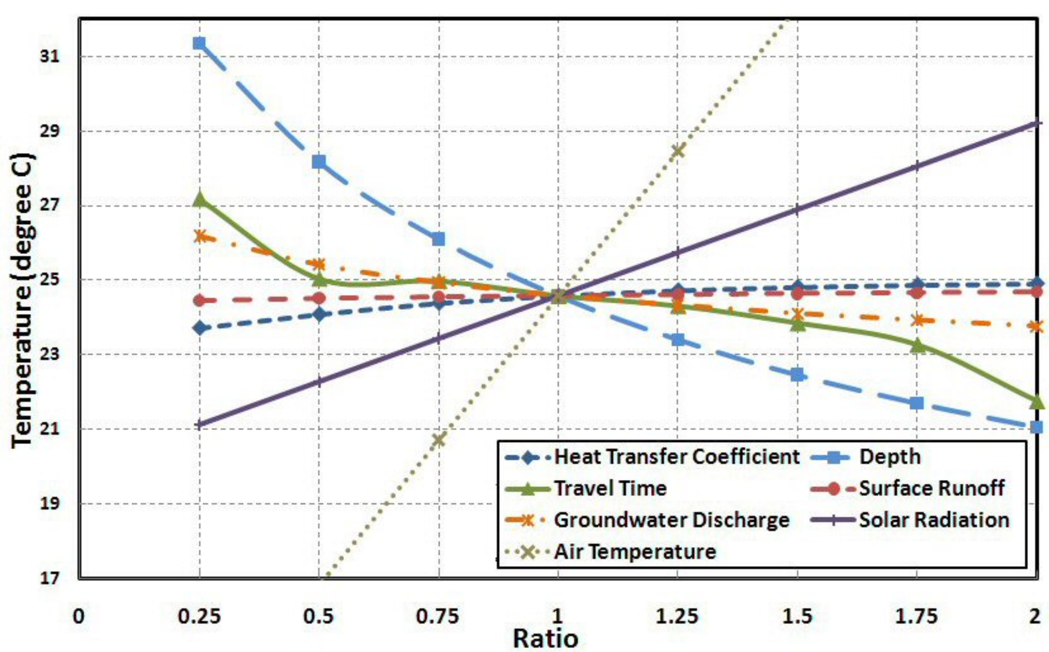


Figure 2.6: Sensitivity Analysis: Identify the influence of each parameter to the modeled July mean stream temperature by perturbing one parameter of factors ranging from 0.25 to 2, with 0.25 increments at a time at Croton Dam in the Muskegon River.

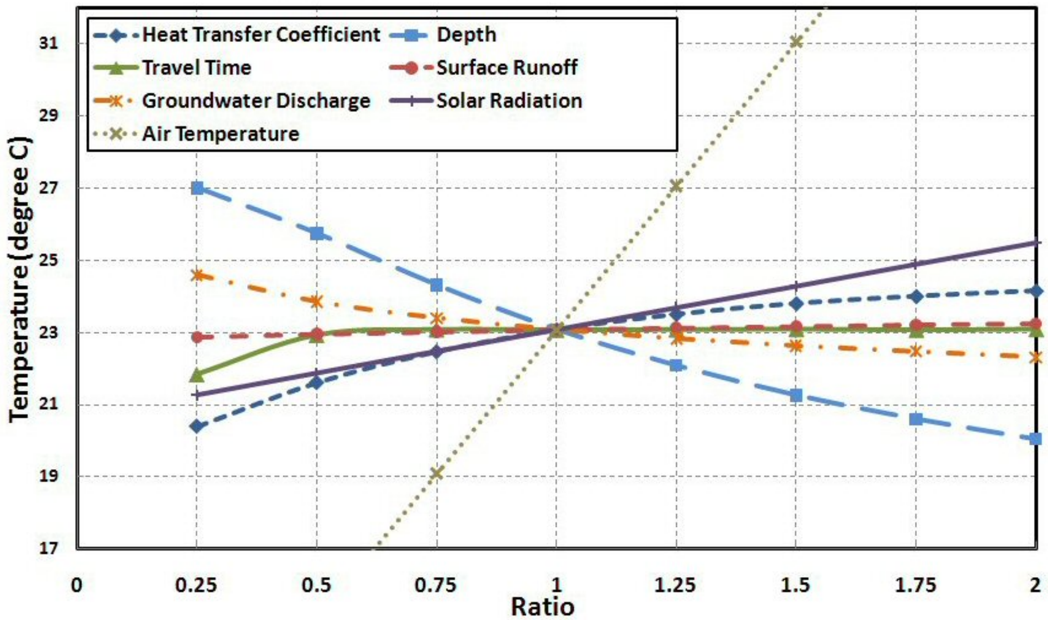


Figure 2.7: Sensitivity Analysis: Identify the influence of each parameter to the modeled July mean stream temperature by perturbing one parameter of factors ranging from 0.25 to 2, with 0.25 increments at a time at Little Muskegon in the Muskegon River.

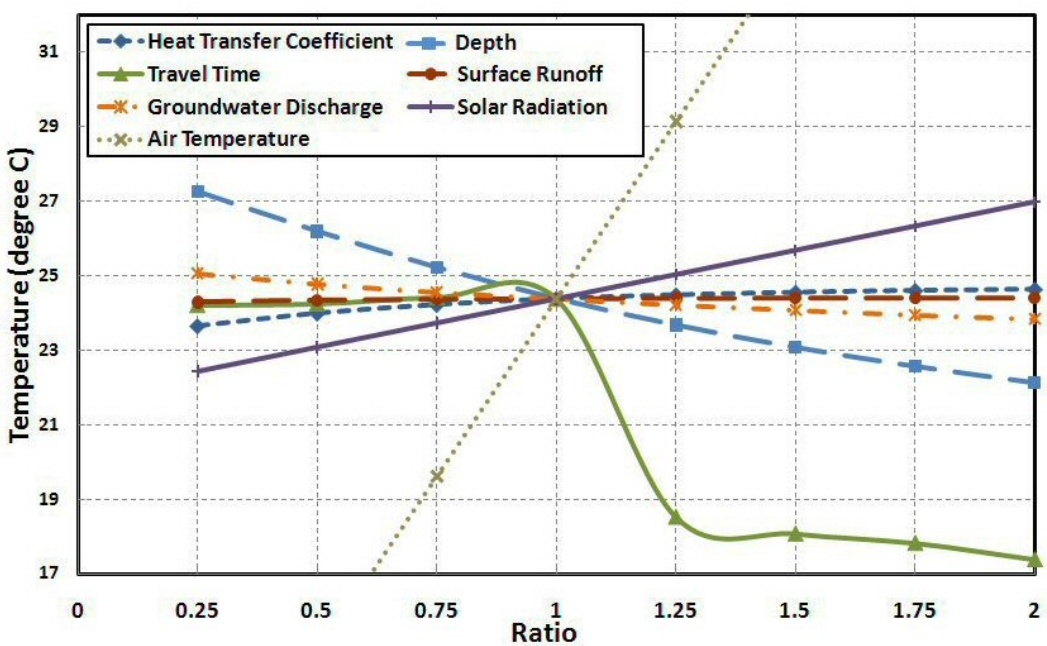


Figure 2.8: Sensitivity Analysis: Identify the influence of each parameter to the modeled July mean stream temperature by perturbing one parameter of factors ranging from 0.25 to 2, with 0.25 increments at a time at Lower Cedar Creek in the Muskegon River.

RPSTM was integrated with an existing watershed multi-model, MREMS (the Muskegon River Ecological Modeling System) (Riseng *et al.*, 2006; Wiley *et al.*, 2010), allowing examination of impacts of natural- or human-induced stressors on stream temperature. By manipulating changes in the environmental conditions, a comprehensive analysis of impacts can be easily investigated.

In this study, RPSTM was able to capture the river's daily and seasonal signals, and matched well with observed stream temperature data. Results showed that the prediction accuracy of RPSTM was within 1.1°C MABSE in the mainstem and within 2°C MABSE in the tributaries. Deviations from observed stream temperature data likely were the result of inaccuracies in both the hydrological model inputs and in parameter estimates.

One likely cause for simulation error is the inaccuracy inherent in air temperature and solar radiation time-series data. The sensitivity analysis showed that air temperature was the most influential parameter to the modeled stream temperature, as most heat exchange terms would involve local air temperatures. I found that shortwave solar radiation was also sensitive to the estimations of stream temperatures, because in the daytime without full shading, shortwave radiation serves as one of the major heating sources to the river water (Bartholow, 2000). With the increasing accuracy of the measuring techniques, errors causing by the measured instrument can be limited. Since air temperature data were measured at weather stations miles away from the Muskegon River watershed area, it cannot capture the real interactions in the local air-water interface. Shortwave solar radiation measurements were obtained at five sites by the river, but applied for the entire model. With the limitations of site-specific input data, RPSTM still provided useful and reasonable predictions with Pearson's product-moment coefficient (r) ranging from 0.77 to 0.99 over such long time period. More accurate simulations could potentially be achieved by using a higher density of weather data.

Another likely source of error could be hydraulic depth estimation. Depth is the second most sensitive parameter to in-stream temperature estimations. I determined depth by regression analysis (Equation 2.20) using data from the low flow season (summer). This would make the depth estimates in high flow seasons (winter) less accurate. This is especially true for estimations in small, shallow creeks with large groundwater inputs, because they are very sensitive to estimates of depth and groundwater advective heat flux. For example, temperature estimates in Cedar Creek showed large deviations from observed stream temperatures in winter. With a large amount of groundwater fed into Cedar Creek, errors from groundwater input, coupled with the errors from the depth estimation, probably both contributed to the observed error. This suggests that larger errors are more likely to occur in my modeling for small tributaries with ample groundwater inputs in winter months.

In addition, errors can arise from the network travel time estimates. Network travel time was modeled as a constant average value for each reach, whereas in reality it fluctuates with discharge, wind speed, and debris load. The sensitivity analysis revealed that travel time has highly variable impacts depending on locations. This is because network travel time is used to determine which equilibrium temperature (i.e., air temperature) should be applied to calculate stream temperature (i.e., a travel time of 2 days will result in an equilibrium temperature set to day (n+2)'s air temperature). Manipulating travel time in rivers in the sensitivity analysis would cause the model to compute equilibrium temperatures that are very different from initially computed equilibrium temperatures. Likewise, altering residence time in reservoirs would also result in the selection of very different equilibrium temperatures. This is especially evident in river junctions from extremely short travel time, such as small creeks, to long travel times, such as reservoirs or big channels. Junctions between small and large rivers may be more sensitive to estimates of travel times because the two rivers are equilibrating to the air temperatures from different days, and usually function differently. This is seen in the model, as sensitivity to network travel time varied markedly among locations (Figures 2.5 to 2.8).

Some errors might also come from the hydrologic model's estimates of groundwater quantity, and my estimates of its recharging location and temperature. I modeled most groundwater recharging locations at upstream as most existing stream temperature models did, if they took groundwater convection into account. I realized that when groundwater is recharged in areas other than upstream, this assumption would make the model not be able to accurately incorporate advective thermal influence on water temperature. To reduce errors resulting from this, in any area, if groundwater advective heat flux was known to be significant, I manually relocated the groundwater recharging area and considered the mixing impacts (Equation 2.19) into the temperature estimation. As a result, errors from groundwater recharging locations can be limited. However, errors can still be from the estimation of groundwater quantity and temperature. The thermal impact of groundwater on the stream water temperature depends on the proportion of groundwater to surface water. The quantity of groundwater is dependent on the time of the year, geological factors, and the watershed area. The temperature of groundwater is more stable compared to the discharge fluctuations, but errors from the estimation of groundwater temperature could still cause in-stream temperature simulation errors. In RPSTM, groundwater temperature companied with air temperature were used to determine the initial condition of the water temperature at the beginning of each modeled upstream node. In this study, quantity of groundwater discharge was provided by the linked groundwater model (i.e., ILHM) for dynamic estimates at each node. Consequently, any errors from the assigned groundwater temperature or the simulated groundwater quantity would result in errors of initial stream temperature estimates.

Furthermore, RPSTM was not as accurate in reservoirs as in normal channel flow units. RPSTM assumed the temperature in epilimnion of reservoirs to be homogeneous, but in reality, it can vary with depth, season, and probably flow. Assuming a homogeneous temperature in epilimnion resulted in a greater discrepancy between modeled and observed segment temperature due to variations in heat exchange and storage in colder, deeper (i.e., hypolimnetic) layers. Although a solar heating coefficient was applied in reservoirs to better calibrate the outputs, RPSTM was not able to correctly locate the actual heat exchange fluxes in reservoirs, as the temperatures in reservoirs were often more extreme but also more stable than those in rivers. Since RPSTM was designed in a more abstract way mainly for river reaches, it does not consider depth dependent stratification and other complex processes a large lake typically has. Therefore, it can not provide as accurate predictions in reservoirs.

Finally, the estimates of the heat transfer coefficient (h_c) might cause certain errors to the simulation results. From Equations 2.5 and 2.6, I found that heat transfer coefficient varies with wind speed. Wind speed varies as a function of canopy density, season, and geography. When running RPSTM, the heat transfer coefficient at each node was estimated as a site-specific constant, resulting in a simpler model but likely decreasing model accuracy. However, Brown(1969) and Sinokrot and Stefan (1993) both pointed out that the impact of wind speed on stream temperature, either by evaporative cooling or by convective heat transfer, is much less important than the impact from exposure/isolation to the sun (Brown, 1969; Sinokrot & Stefan, 1993). Similarly, the sensitivity analysis showed that the heat transfer coefficient is not as sensitive as aforementioned parameters. Therefore, I believe errors associated with assuming a fixed wind speed are likely small.

RPSTM can be a useful approach for modeling large-scale temperature distributions across a big river network. Many other stream temperature models work well at few locations, or in the well studied rivers with relatively small drainage areas (Webb & Nobilis, 1994; Webb, 1996). For example, there have been a variety of stream temperature models developed for specific locations in rivers (Webb & Nobilis, 1994; Webb, 1996), that can only predict stream temperatures within the study sites which they are parameterized. RPSTM's general format allows applicability across wide range of river networks when basin-wide hydrologic modeling is available.

RPSTM offers the advantage of reduced need for parameterization while providing accurate stream temperature estimates. RPSTM shares many similarities with HEC-5Q, a temperature model developed by the US Army Corps of Engineers (1986), which also accurately predicts stream temperatures across whole river networks with MABSE between 1.63-2.23, (Bartholow *et al.*, 2005). However, RPSTM had similar performance (MABSE ranges between 0.71 to 2.0 °C) to HEC-5Q while requiring fewer parameters. HEC-5Q requires data on many meteorological conditions (i.e atmospheric pressure, air temperature, humidity, shortwave radiation, cloudiness, and wind speed) that are not readily available in most watersheds. Moreover, HEC-5Q is sensitive to estimates of topographic shading. RPSTM requires no shading inputs. In theory, shading impacts stream temperature by blocking incoming shortwave solar radiation, and influencing local air temperatures. In RPSTM, the effect of topographic shading on stream temperature has been absorbed via the exhibition of local air temperature and solar radiation as required input data. Therefore, there is no need to double count the impact of shading. As such, RPSTM is easier to use than HEC-5Q while providing similar accuracy in temperature estimation. The only atmospheric conditions required by RPSTM are air temperature and shortwave radiation, making it more practically applicable. If there are no observed air temperature data available, the General Circulation Models (GSMs) (Lau *et al.*, 1996) can provide estimates. Furthermore, the linearity relationship of solar radiation to modeled stream temperature implied that one can still use RPSTM for in-stream temperature prediction when there is no solar radiation data available, if assisted by calibration techniques.

The most important advantage of RPSTM is that it supports dynamic estimations of stream temperature. Accuracy is also enhanced due to explicitly considering travel time in stream temperature estimation. Moreover, using dynamic air temperature better incorporates the effects of local climate variation on stream temperature. Furthermore, it simulates process-based flow conditions by coupling with HEC-HMS. Flow is one of the controlling factors on stream temperature. By representing flow dynamically, one can predict stream temperature more accurately than a steady state flow model. In addition, RPSTM also has the ability to respond to dynamic changes in groundwater discharge. Existing models with reduced parameter requirements often ignore groundwater advection when modeling large rivers (Brown, 1969). However, big river segments almost always contain several smaller headwater streams, and large river valleys typically provide both substantial elevational head and conductive alluvial depths. As a result, groundwater advection often has significant impacts on stream temperature under low flow conditions (Bundschuh, 1993; Constantz *et al.*, 1994; O'Neal, 1997; Wiley *et al.*, 1997; Baker *et al.*, 2001, 2003; Lee & Hahn, 2006). RPSTM took groundwater advective impacts into account by considering their magnitude in the presumed mixing process with surface water discharge, and reflected in calculating the initial stream temperature at the start of each tributary. On the other hand, it might imply that without dynamic discharge information, either provided by field measurements or by modeling approach, it will be difficult to apply RPSTM.

In conclusion, RPSTM has been designed and developed to be a spatially distributed, easy to parameterize heat balance model. It can be integrated with dynamic hydrological models to provide dynamic stream temperature estimation under the variable influences of micro-climate, land cover, and hydrology. Furthermore, it is easy to apply to large watersheds that are difficult to model using extant models. Comparison of model outputs to observed data from a variety of streams in the Muskegon River Watershed reveal that RPSTM provides accurate stream temperature estimations. Given that RPSTM is a process-based model, it can be and has been used in dynamic simulations of past, current, and future watershed management scenarios (Wiley *et al.*, 2010).

CHAPTER III

DOWNSTREAM IMPACTS OF WATER WITHDRAWAL: A SPATIALLY EXPLICIT SIMULATION

3.1 Abstract

Groundwater plays an important role in shaping the ecological character of rivers. As humans search for water resources to satisfy increasing demand, pumped withdrawals pose potential threats to river water quality and biological integrity. In 2006, the State of Michigan enacted a new groundwater withdrawal law (2006 PA 34), which explicitly linked potential impacts on fishes to permitting. To better quantify potential withdrawal impacts, I used modeling experiments to evaluate the changes in thermal regimes associated with different levels and locations of water withdrawal. I also linked thermal regimes to potential impacts on coldwater fishes by calculating the changes in rates of potential lethal events across affected lengths of river channel. Results showed that withdrawal effects were highly location and intensity dependent, although increases in potential lethal events for coldwater fishes were in general strongly correlated with flow reductions. Since the impacts of a water withdrawal varies with location due to the spatial heterogeneity of groundwater loading. It is important to understand site-specific hydrologic routing when issuing water withdrawal licences. This study contributes to the discussion of how environmental protection standards and permitting requirements can best be developed to both provide for the needs of our citizens and to protect our waters and water-dependent resources.

3.2 Introduction

River flow is composed of surface runoff and groundwater. These two components have very different characteristics. Surface water flow is almost always faster, and warmer in summer and cooler in winter, than groundwater. Stream temperature varies spatially and temporally with latitudinal and longitudinal locations, day/night and seasonal fluctuations, global and local climate, geomorphic settings, and hydraulic routing. Stream width, depth, and surface area (size), length, and velocity are important in governing river temperature. Wide streams will have a large surface area compared to volume exposed to solar radiation and thus will be heated to a higher temperature than narrow streams (Farber *et al.*, 1998). Longer streams with slower velocity will have more time in contact with the atmosphere and expose to solar radiation, and therefore, will have higher temperatures (Adam & Sullivan, 1990). Often, the proportion of surface and groundwater entering a river channel determines its water temperature. Groundwater can directly affect river temperature through advective processes that buffer the water mass against source of heat input (Wehrly *et al.*, 2006). Therefore, the magnitude of groundwater delivery to river channels modulates downstream heat gain. In addition, the specific locations where groundwater exchanges occur are important in shaping river thermographs downstream. Given the relatively constant groundwater temperature (Bundschuh, 1993; Sinokrot *et al.*, 1995; Constantz, 1998), high groundwater flows typically (in Michigan) result in cooler river temperatures in summer and warmer temperatures in winter.

In Michigan, the landscape was heavily glaciated from 43-16 k BP. A protracted period of retreat (16-10 BP) led to ample coarse drift deposits and groundwater inputs to modern river systems. However, even in this region, increasing consumptive demand on surface and groundwater supplies are raising concern (Great Lakes- St. Lawrence River Basin Water Resources Compact and Agreement, 2005). Consumptive withdrawals can affect river water levels and (indirectly) temperature, potentially dramatically altering aquatic communities (Jansen, 2000; Turner *et al.*, 2005; Cott *et al.*, 2008). In 2006, to better manage river water quality and to protect river fishes from water withdrawal impacts, the State of Michigan enacted a new groundwater withdrawal law. Under this legislation, river flows cannot be decreased "such that the stream's ability to support characteristic fish populations is functionally impaired" (Senate Bill 0850, Michigan 2006). Later in 2007 and 2008, several additional Public Acts were enacted to (1) prohibit all withdrawals resulting in adverse resource impacts; (2) establish a water withdrawal permitting system; and (3) require several state departments to develop an assessment tool to determine and proposed withdrawal's potential impact. While it was understood, the thermal impacts of water withdrawal necessarily vary from site to site and generally permutate downstream (Cott *et al.*, 2008; Zorn *et al.*, 2008), many existing stream temperature models are either too simplistic to capture these longitudinal dynamics, or too complex to be easily applied (TVA, 1972; LeBlanc *et al.*, 1997; Bartholow, 2000; Borman & Larson, 2003; Bartholow *et al.*, 2005; Chapra & Tao, 2006).

Here, I apply the newly developed Reduced Parameter Stream Temperature Model (RP-STM; see Chapter 2) to explore the temporal and spatial dynamics of water temperature responses to typical water withdrawal scenarios. The objectives of this paper are: (1) to illustrate and quantify patterns of downstream change in thermal regimes associated with water withdrawal in Michigan rivers, and (2) to examine the implications of such changes on the distribution of coldwater fish assemblages.

3.3 Methods

In order to understand the impacts of simulated rates of water withdrawal on river ecosystems, it is necessary to explicitly consider linkages between hydrological and thermal regimes. To this end I chose three representative Michigan river systems as locales for my analysis. Hydrologic models were developed for each of the systems and then were used to simulate baseline flow (summer) conditions. My newly developed temperature model, RP-STM (see Chapter 2) was then used to model baseline thermal profiles in each river. Next, simulated flows were incrementally reduced from baseline levels at one or more locations in each of the modeled river networks, and RPSTM used to examine changes in temperature that resulted. Finally, resulting changes in thermal regime were then summarized as changes in thermal habitat quality by estimating rates of change in potential lethal events for coldwater fishes. "Potential lethal event" is defined here as a seven-day running average July and August stream temperatures greater than 22°C (Wehrly *et al.*, 2007). Presumably a single lethal event could substantially impact local fish populations.

3.3.1 Site Descriptions

The Muskegon River, Cedar Creek, and Mill Creek were chosen as representative systems because they represent a range of hydrologic types and sizes common in Michigan, and had sufficient existing data sets to support the required modeling simulations (Figure 3.1).

1. The Muskegon River Watershed:

   The Muskegon River Watershed is within the Lake Michigan/Huron basin, part of the Laurentian Great Lakes, in central lower Michigan, USA. Drainage basin encompasses about 7,057 km^2, which lies within 10 counties (Clare, Lake, Mecosta, Missaukee, Montcalm, Muskegon, Newaygo, Osceola, Roscommon, and Wexford counties). The main river runs about 350 *km*, making it the second longest river in Michigan. The main stem Muskegon River is a fifth order stream, arising in Higgins and Houghton lakes, and flowing southwest to Muskegon Lake and into Lake Michigan. Stream flow within the Muskegon River Watershed is dominated by groundwater sources (Hyndman *et al.*, 2006, 2007) and as a result supports a number of coldwater fishes (O'Neal, 1997; Wehrly *et al.*, 2006). The northern part of this watershed is currently dominated by forest, with agriculture dominant in the central region, and urban areas in the south (Pijanowski *et al.*, 2007). Peak flows in this system occur in the spring with associated snowmelt or large rain events (Hyndman *et al.*, 2006; Pijanowski *et al.*, 2007).

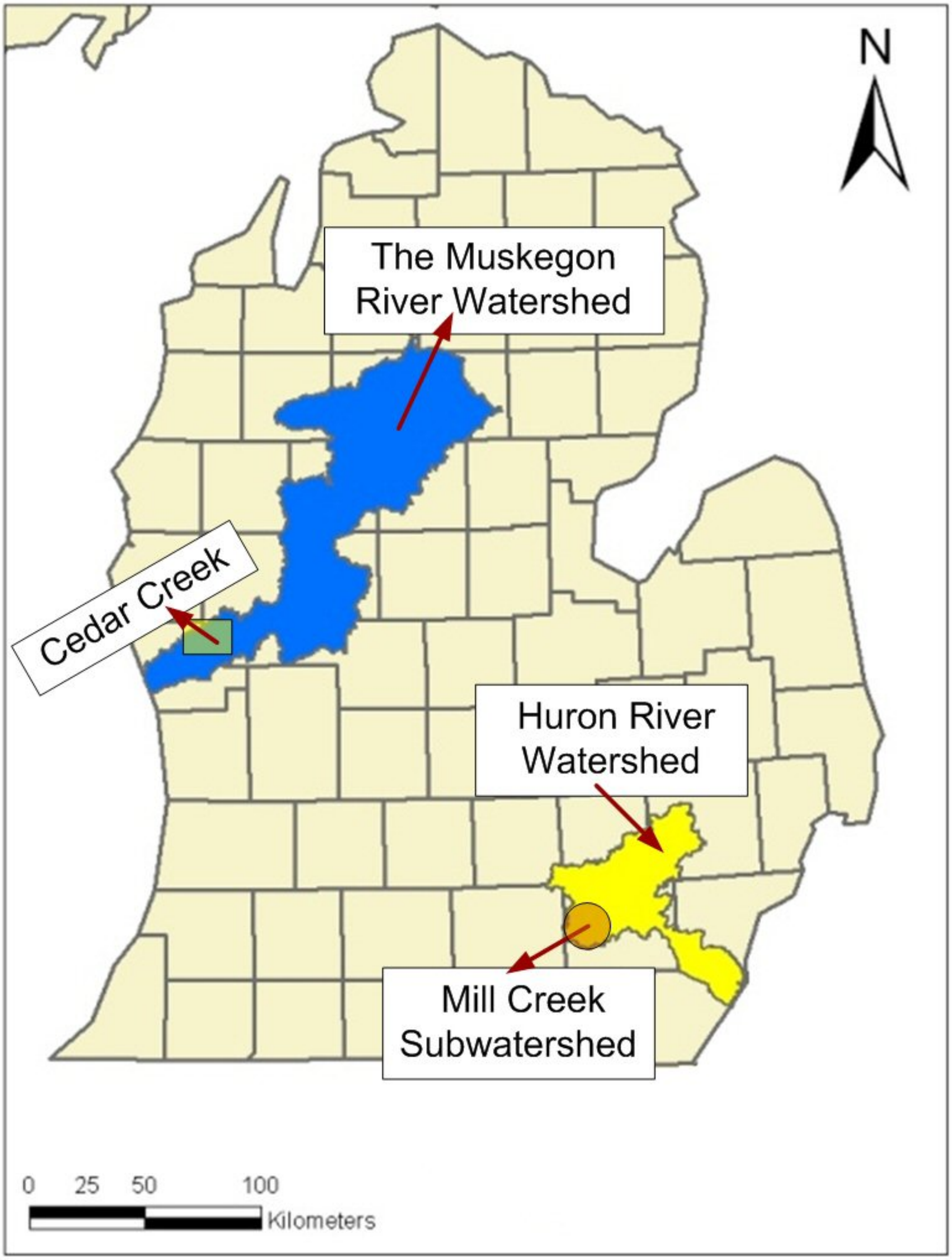


Figure 3.1: The locations of the Muskegon River Watershed, Cedar Creek (a tributary of the Muskegon River), and the Mill Creek Watershed in Michigan.

The Muskegon River's many groundwater fed tributaries sustains cold- and cool-water species, including walleye, northern pike, yellow perch, white sucker, sculpin, steelhead, brook trout, lake trout, coho, Chinook salmon, and brown trout (Clapp *et al.*, 1990; Brazner *et al.*, 2004, 2005; Creque *et al.*, 2005; Wehrly *et al.*, 2006). Nevertheless, it has been reported that a number of important sport fish populations appear to be under stress due to human-induced changes in the environmental conditions (O'Neal, 1997; Wiley *et al.*, 2003a).

2. Cedar Creek:

   The 147.89 km^2 Cedar Creek watershed is a sub-basin of the Muskegon River and is located in Muskegon and Newaygo counties. Cedar Creek is comprised by 3 small tributaries- Upper Cedar, Lower Cedar, and Little Cedar. The dominant landcover in the upstream portion of Cedar Creek is agriculture, while downstream landcover is almost entirely forested, which provides shade and helps moderate water temperatures. The lower portion of Cedar Creek is strongly influenced by groundwater (Hyndman *et al.*, 2006, 2007). Downstream shading and groundwater provide habitat for brook trout and Chinook salmon.

3. Mill Creek:

   Mill Creek drains the largest sub-basin of the Huron River, a medium-sized river tributary to Lake Erie. It flows through a bowl-shaped catchment (340 km^2) in an agriculture-dominated landscape. The geology of the Mill Creek catchment includes morainal ice contact hills and coarse moraines (north and west), finer texture moraines (east and south), alluvium and outwash (east), and till plain for most of the catchment area (Seelbach & Wiley, 1996). Mill Creek has four major branches- the Main, the North, the South and East branches. The largest branch is the Main Branch, which drains the southwest edge of the watershed. The Lower Mainstem carries the combined flow of the four upper branches into the Huron River at Dexter.

3.3.2 Hydrological Modeling Overview

1. The Muskegon River Hydrologic Model

   Flows for the Muskegon River Watershed were simulated using HEC-HMS (Hydrologic Engineering Center's Hydrologic Modeling System) linked with a 2-layer recharge and groundwater model developed by Dr. P.E. Richards, SUNY at Brockport, NY, as a part of the the Muskegon River Ecological Modeling System (MREMS) (see http://mwrp.net) (Wiley *et al.*, 2003a; Brenden *et al.*, 2008; Wiley *et al.*, 2010). The Muskegon HMS model represents this system as a series of 21 channel reaches originating in 40 sub-basins (a total of 118 nodes) (See Figure A.1 in Appendix A). It was run and ran with a daily time step, simulating the period from 1/1/1985 to 12/31/2005.

2. The Cedar Creek Hydrological Model

   Flow in Cedar Creek was modeled using ILHM (Integrated Landscape Hydrological Model) (Hyndman *et al.*, 2006) routed through a higher resolution HEC-HMS network. Baseflow simulations and runoff abstraction from ILHM were linked with the HEC-HMS to estimate an in-channel hydrograph. The hydrologic model of Cedar Creek was comprised of 20 sub-basins feeding 18 channel reaches with a total of 53 nodes (See Figure A.2 in Appendix A). Model simulation ran on an hourly time step; flows were simulated for the period from 12/19/2002 to 8/4/2004.

3. The Mill Creek Hydrological Model

   Flow in Mill Creek was modeled using HEC-HMS with empirically calculated monthly groundwater flows. The model represented Mill Creek as 10 sub-basins and 7 channel reaches with a total of 26 nodes (See Figure A.3 in Appendix A). The model ran on a daily time step from 1/1/2003 to 9/30/2004. Daily groundwater discharge was estimated as an average monthly loading from a regional regression model (Seelbach & Wiley, 1996).

3.3.3 Modeling Experiments

1. **Experiment 1: Response Variability to Basin-Wide Flow Reductions**

   In this Experiment, I incrementally two factors simultaneously reduced (a) total flow (defined as the sum of surface water and groundwater), and then (b) only groundwater, at each of the 40 modeled sub-basins. The effect was a basin-wide reduction in flows driven by either losses of groundwater, surface water, or both. My goal was to investigate how proportional losses might vary spatially within a single river system. Results for each reach were summarized as 10-year (from 1996 to 2005) July and August mean temperatures, because these two months usually consist of the highest water temperature of the year, which is most likely to threaten the coldwater fish species. I also calculated a 7-day moving average of the daily modeled temperature at each reach for every 7-day interval for the period from 1985 to 2005. Since summer weekly mean in-stream temperature below 22°C is a widely used thermal threshold for coldwater species in the midwest U.S. (Lyons, 1996; Lyons *et al.*, 1996; Wehrly *et al.*, 2003, 2007), when a 7-day average is above the 22°C thermal threshold occurred, I counted it as one potential lethal event (Wehrly *et al.*, 2007).

   The impact of water withdrawal on both temperature and habitat availability was compared for each withdrawal level to the baseline conditions in terms of the probability of potential lethal events. These were calculated by dividing the modeled number of potential lethal events by 1816 (the number of lethal events that would occur during the simulation if all summer temperatures exceed the 22°C threshold).

2. **Experiment 2: Spatial Variability in Response to Localized Flow Reductions in (a) a Cold and (b) a Warm Water System**

   My purpose in Experiment 2 was to investigate the impact of local withdrawal on downstream water temperatures, and to compare those impacts in a small coldwater (Cedar Creek) and a small warmwater (Mill Creek) stream. This experiment differs

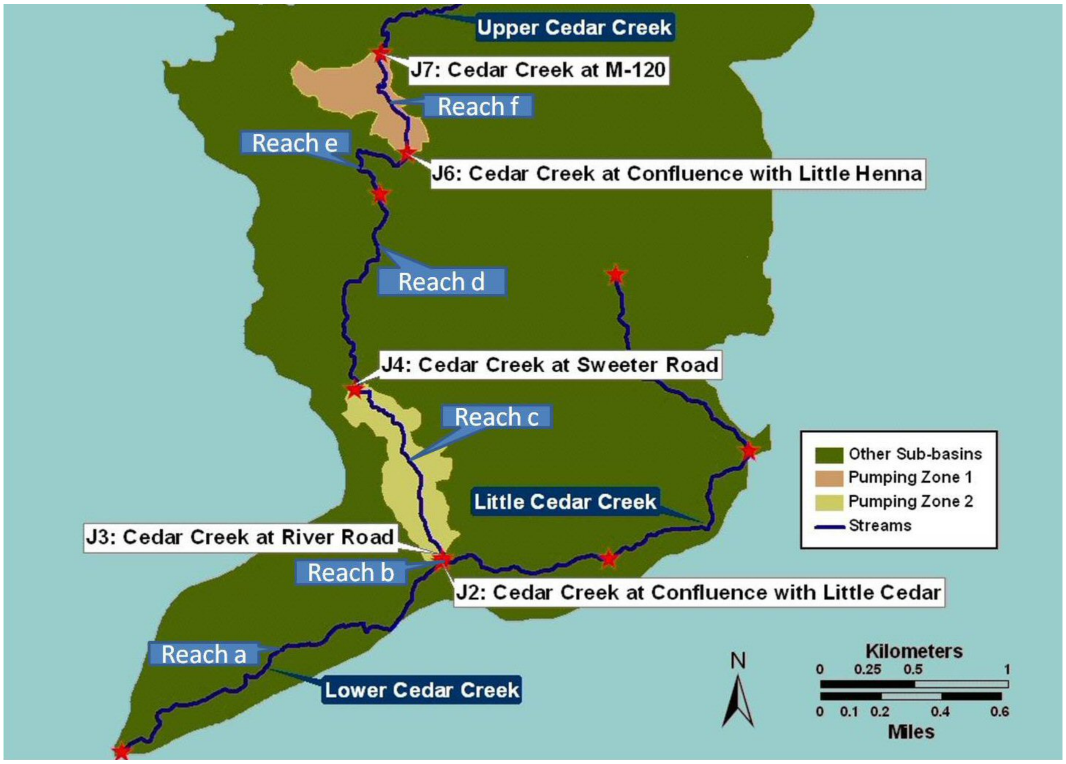


Figure 3.2: The location of 2 pumping zones used in simulations in experiment 2a for Cedar Creek, Michigan.

from Experiment 1 in that withdrawals were limited to a single location for each model run, although I explored spatial variability by examining 2 withdrawal locations in each stream system. Pumping Zone 1 extended from Cedar Creek at M-120 to its confluence with Little Henna Creek, and Pumping Zone 2 was from Sweeter Road to River Road (Figure 3.2). In a similar way, flow was reduced at 2 different locales in the Mill Creek simulation (2b). Pumping Zone 1 was in the upper Main Branch basin, and Pumping Zone 2 was in the upper North Brain basin (Figure 3.3). Simulated temperatures were again summarized by monthly mean temperatures in July and August, and potential lethal events. The cumulative effects of flow withdrawal on thermal regimes were evaluated and compared moving downstream to the mouth.

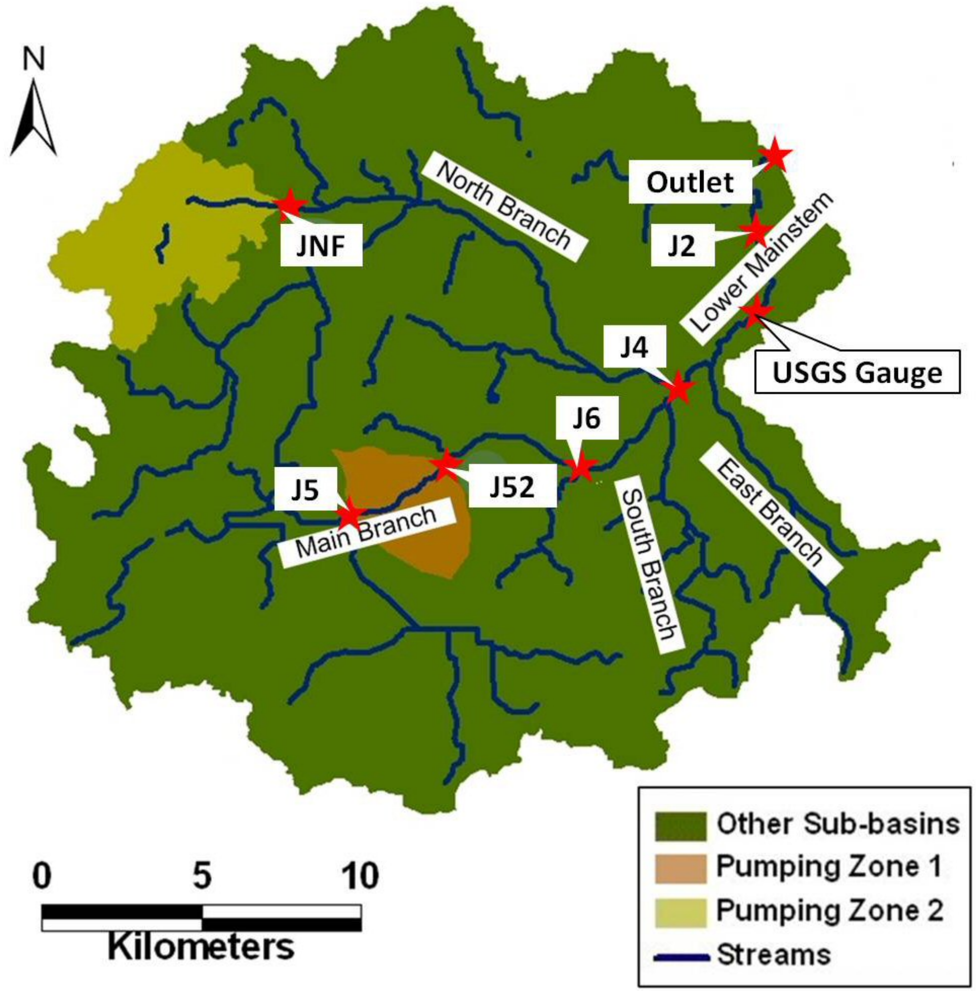


Figure 3.3: The location of 2 pumping zones used in simulations in experiment 2b for Mill Creek, Michigan.

3.4 Results

3.4.1 Experiment 1

1. Changes in Water Temperature

   Modeled water temperature exhibited large spatial variation throughout the Muskegon River watershed under baseflow conditions. Likewise, responses to experimental flow reductions were highly variable spatially. For example, in the mainstem river below Croton Dam, July mean temperature increased about 2.4°C from 24.6°C (i.e., baseline) up to 27.0°C with a 90% total flow reduction. In contrast, in the lowest reaches of Brooks and Cedar creeks, important cool- and cold-water tributary systems, July mean temperature only increased about 1.8°C to 2°C from 18.1°C to 19.8°C and from 18.9°C to 20.9°C with a 90% flow reduction respectively (Table 3.1).

   Reductions in sub-basin groundwater loading generally resulted in more rapid warming than did reductions in surface water loading. For example, a 90% groundwater reduction in Cedar Creek in July resulted in a 3.0°C warming, while only 2.0°C warming resulted from a 90% reduction in total flow. On the other hand, tributaries with very little groundwater input saw almost no change in temperature when groundwater alone was withdrawn. When similar percentages were withdrawn from both surface and groundwater, tributaries with little groundwater input warmed rapidly and more than from a withdrawal of groundwater alone. In this case, a much larger absolute water volume was taken by the simulated surface water than by groundwater pumping activities. For example, at the outlet to Muskegon Lake in August, a 0.6°C warming occurred with a 90% groundwater withdrawal, in contrast to 0.7°C warming from a 90% reduction in both surface and groundwater loading (Table 3.2).

   In general, Experiment 1 showed that increasing levels of water removal generally result in warmer stream temperatures during the summer. Impacts of pumping also

Table 3.1: Modeled 10-year (1996 to 2005) July and August mean water temperature resulting from different percentages of total flow reduction at different locations in the Muskegon River Watershed.

Site	**Middle Branch**	**Croton Dam**	**Little Muskegon**	**Brooks Creek**	**Cedar Creek**	**Outlet to the Muskegon Lake**
% **Flow Reduction**	Modeled 10-year July Mean Water Temperature (°C)					
0% (Baseline)	22.3	24.6	20.8	18.1	18.9	25.1
10%	22.4	24.7	20.8	18.2	19.0	25.1
30%	22.6	25.0	21.1	18.3	19.2	25.5
50%	22.9	25.3	21.3	18.5	19.5	25.4
70%	23.3	25.9	21.8	18.9	19.9	25.7
90%	24.0	27.0	22.7	19.8	20.9	26.1
% **Flow Reduction**	Modeled 10-year August Mean Water Temperature (°C)					
0% (Baseline)	22.0	24.9	20.4	17.9	18.6	24.6
10%	22.0	25.0	20.5	18.0	18.7	24.6
30%	22.2	25.3	20.7	18.1	18.9	24.7
50%	22.5	25.6	20.9	18.3	19.1	24.9
70%	22.8	26.1	21.3	18.7	19.5	25.0
90%	23.5	27.1	22.2	19.5	20.4	25.3

varied with location and the type of removal (total flow vs. groundwater).

2. Changes in Thermal Habitat Quality

   In Experiment 1, I simulated daily temperatures across the river network for a period of 21 years (1/1/1985 to 12/31/2005) and estimated probability of lethal thermal events for salmonids as a general biological metric for thermal habitat.

   In the baseline run a total of 4344 potential lethal events occurred within the entire watershed during the simulated 21 year period; this is equivalent to a basin average rate of approximately 1.7% per year (21 yrs * 138 VSEC-NHD segments * 86 events = 249228 possible events). This could be interpreted as meaning that over the 21 simulated summers, on average about 1.7% of the time temperatures in a single reach element (VSEC-NHD unit; Valley Segment Ecological Classification Unit-National Hydrography Dataset (Seelbach & Wiley, 2005; Seelbach *et al.*, 2006)) exceeded thresholds generally tolerable by coldwater species (Figure 3.4). Of course the spatial distribution was not uniform. The river above Rogers Dam was more favorable

Table 3.2: Modeled 10-year (1996 to 2005) July and August mean water temperature resulting from different percentages of groundwater withdrawal at different locations in the Muskegon River Watershed.

Site	**Middle Branch**	**Croton Dam**	**Little Muskegon**	**Brooks Creek**	**Cedar Creek**	**Outlet to the Muskegon Lake**
% **Withdrawal**	Modeled 10-year July Mean Water Temperature (°C)					
0%	22.3	24.6	20.8	18.1	18.9	25.1
10%	22.5	24.7	20.9	18.2	19.0	25.1
30%	22.8	25.0	21.2	18.4	19.3	25.2
50%	23.3	25.4	21.6	18.8	19.7	25.4
70%	24.0	26.0	22.3	19.5	20.3	25.6
90%	25.4	27.1	23.8	21.2	21.9	25.9
% **Withdrawal**	Modeled 10-year August Mean Water Temperature (°C)					
0%	22.0	24.9	20.4	17.9	18.6	24.6
10%	22.1	25.0	20.5	18.0	18.7	24.6
30%	22.5	25.3	20.8	18.3	18.9	24.7
50%	22.9	25.7	21.2	18.6	19.3	24.8
70%	23.7	26.3	21.8	19.3	19.9	25.0
90%	25.1	27.3	23.2	20.9	21.3	25.2

to coldwater species (lethal events probability of 11% to 20%) than the lower river (lethal events probability of 21% to 40%). Lower river tributaries, such as the Little Muskegon River, Bigelow Creek, Brooks Creek, and Cedar Creek also clearly provide good habitats for coldwater fishes (Figure 3.4). As water was removed from the sub-basins, the basin-wide total number of lethal events increased more rapidly with higher percentages of total flow reductions. When half of the total flow was withdrawaled, total number of lethal events could increase by about one third to the baseline (Table 3.3).

There was great spatial variability in terms of where these lethal events occurred. For example, there were zero potential lethal events in lower Brooks and Bigelow Creeks even at a 90% flow reduction. On the other hand, in the Lower Middle Branch, four additional (beyond baseline) potential lethal events occurred at a 70% flow reduction; in the Little Muskegon River, lethal events began to increase at a 30% flow reduction. Below Croton Dam and in the mainstem downstream of Cedar Creek, 30% flow reductions increased the frequency of lethal events by 63.2% and 22.4% respectively

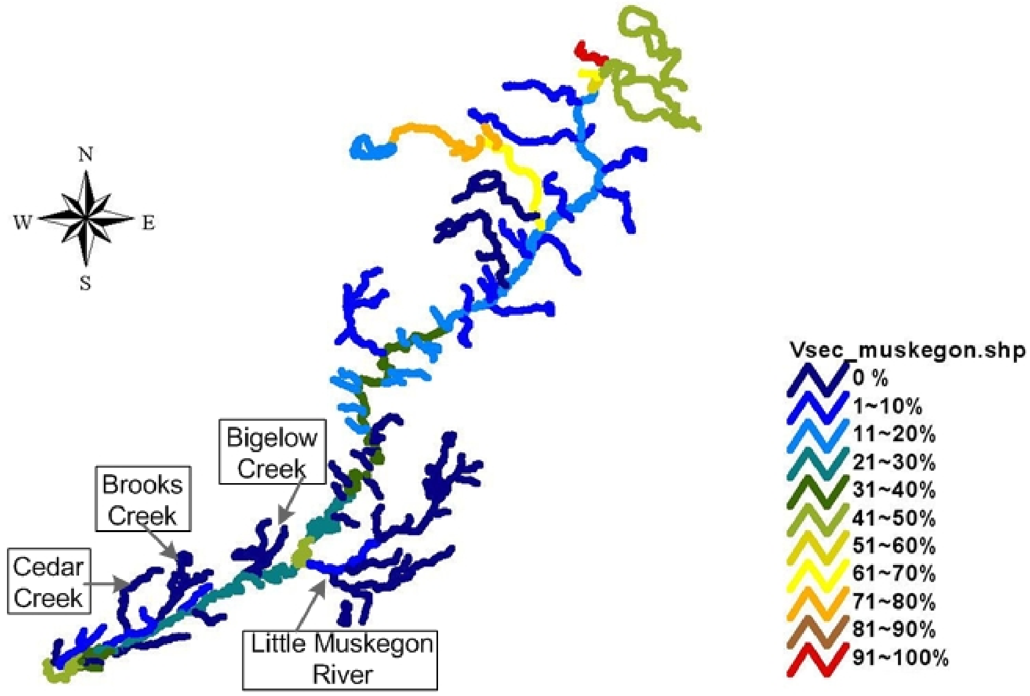


Figure 3.4: Baseline lethal events probability in over 21 years from 1985 to 2005 at each modeled locations throughout the entire Muskegon River Watershed under current flow condition.

Table 3.3: The total number of potential lethal events in the Muskegon River under different total flow withdrawals from 1985 to 2005.

Percent Total Flow Withdrawal	Lethal Events (Times)	Percent Increase
0%	4344	Baseline
10%	4556	4.9%
20%	4761	9.6%
30%	5005	15.2%
40%	5242	20.7%
50%	5532	27.3%
60%	5798	33.5%
70%	6126	41.0%
80%	6597	51.9%
90%	7365	69.5%

Table 3.4: Increment in potential lethal events for select sub-basin in the downstream of Middle Branch, Croton Dam Reservoir, downstream of the Little Muskegon River, downstream of Brooks Creek, down stream of Bigelow Creek, and downstream of Cedar Creek caused by different levels of total flow removal.

Location	**Middle Branch**	**Croton Dam**	**Little Muskegon**	**Brooks Creek**	**Bigelow Creek**	**Cedar Creek**
% Flow Reduction	**Number of Potential Lethal Events (Times)**					
0%	0	764	0	0	0	152
10%	0	819	0	0	0	175
20%	0	873	0	0	0	211
30%	0	935	1	0	0	248
40%	0	982	5	0	0	289
50%	0	1054	6	0	0	325
60%	0	1144	7	0	0	375
70%	4	1253	9	0	0	435
80%	6	1359	27	0	0	512
90%	37	1511	68	0	0	694

(Tables 3.4 and 3.5).

Using the thermal criterion of avoiding potential lethal events for coldwater fishes, I examined the overall patterns of thermal sensitivity in the different river segments to flow reduction and can identify those reaches most vulnerable to temperature change due to water withdrawal. A 10% modeled flow reduction was predicted to result in 1% to 5% increases in lethal events from the baseline in three areas (Clam River, Hersey River, and the lower part of the Little Muskegon River) and in almost all modeled reaches of the mainstem Muskegon River. The rest of the modeled tributaries and the very upstream main channels of the Muskegon River showed no change from the baseline. A 30% flow reduction caused 6% to 10% increases in lethal events probability in most of the mainstem channel, except reaches at the Hardy Dam and the lower parts of the Muskegon River which increased only 1% to 5% from the baseline. As water withdrawals were increased, lethal events occurred at more locations. At the 90% flow reduction rate, the upper main channels of the Muskegon River had 46% more events than at baseline; the middle parts of the river showed increases of 41% to 45%; and the lower river had increases from 25% to 16% (Figure 3.5). Reductions in groundwater alone showed similar trends, but with a more rapid increase

Table 3.5: Times and percentage increases of potential lethal events in the downstream of Cedar Creek caused and in the Croton Dam Reservoir by different level flow withdrawal upstream from the Muskegon River study during the period of 1985 to 2005.

Sites	**Downstream of Cedar Creek**		**Croton Dam**	
Percent Flow Reduction	**Event Increases**	**Percent Increase**	**Event Increases**	**Percent Increase**
0%	Baseline	Baseline	Baseline	Baseline
10%	23	15.1%	55	7.2%
20%	59	38.8%	109	14.3%
30%	96	63.2%	171	22.4%
40%	137	90.1%	218	28.5%
50%	173	113.8%	290	38.0%
60%	223	146.7%	380	49.7%
70%	283	186.2%	489	64.0%
80%	360	236.8%	595	77.9%
90%	542	356.6%	747	97.8%

in the rate of increase of lethal thermal events (Figure 3.5).

In terms of total length of the channel system affected by water withdrawal, I found that as more water was removed, the total adverse impact on coldwater fishes (kilometers of channel habitat lost) increased. For example, across the entire watershed, the total length of channel that experienced 1-week above at least 22°C thermal threshold ranged from 52.5*km* for a 10% total flow reduction to 332.5*km* for a 90% total flow reduction. In the worst case, more than 27 additional weeks above the threshold occurred when 90% of the total flow was removed. It is worth noting here that at a 30% reduction level, adverse effects for a 4- or a 5-week period began to occur. Clearly, the relationship between flow reduction and the relative adverse effect are complex, and no single description for all segments is possible. Nonetheless, I believe that the 30% flow reduction level represents a system level threshold at which impacts on coldwater fishes are great and will occur because of higher summer water temperature (Figure 3.6).

3. Experiment 2a: Confined Withdrawals in a Coldwater Stream

In the Cedar Creek modeling experiment, I simulated water withdrawal of both

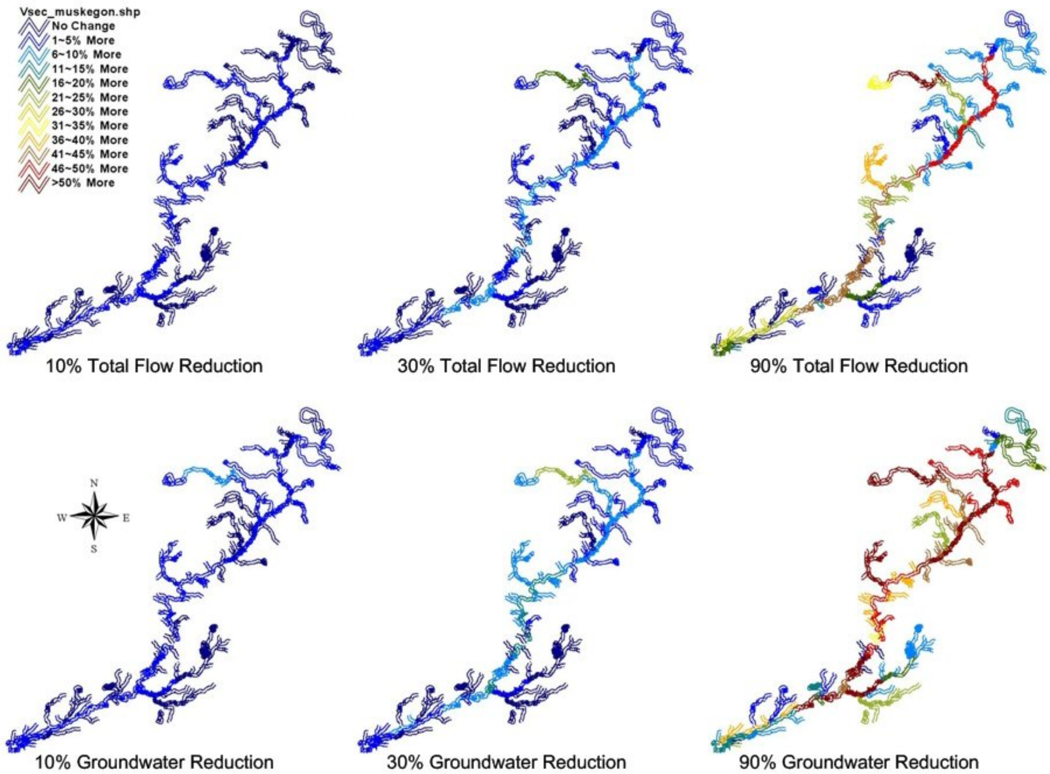


Figure 3.5: Changes in the possible lethal events probability with different level of total flow (i.e., surface water plus groundwater) or only groundwater withdrawn at every modeled location compared to the baseline within the entire Muskegon River Watershed from 1985 to 2005.

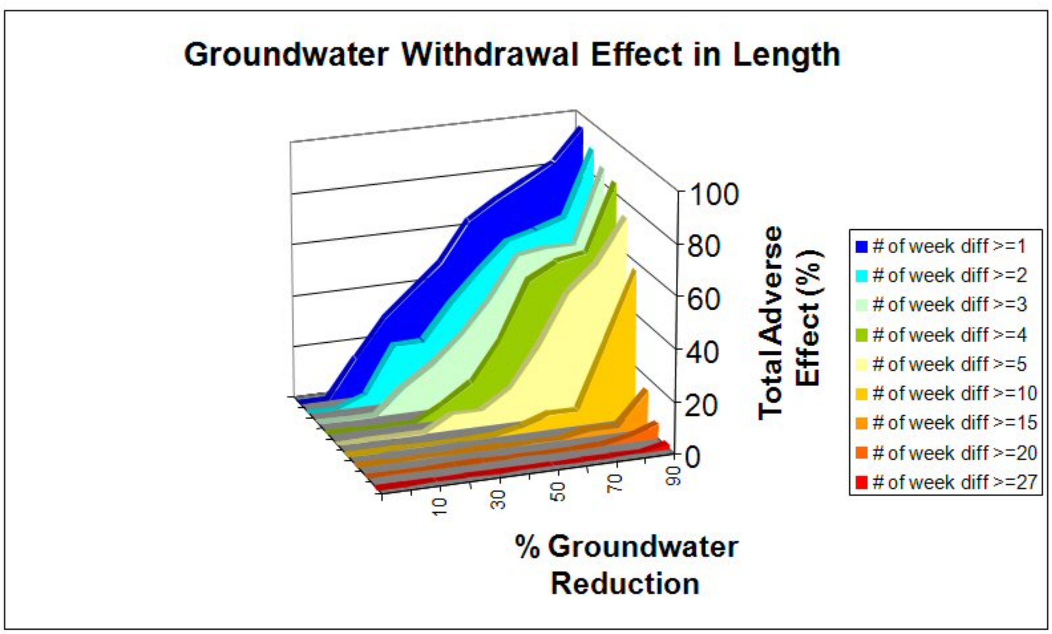


Figure 3.6: Changes in the total affected length of the modeled tributaries and mainstem Muskegon River in response to different flow reduction, and the resulted number of week differences to the baseline with potential lethal events across the entire Muskegon River Watershed from 1985 to 2005.

groundwater and surface water from a single locale (Zone 1 or 2) by manipulating surface water and groundwater elements in the model. Zone 1 contributed on average 51% of available flow at the mouth of the system and in this experiment pumping from Zone 1 simulated a local, medium-scale water withdrawal of the type common in areas under increasing development pressure. The modeled maximum percentage water withdrawal (90% reduction from these two elements) would be approximately 60% of the total available channel flow. Zone 2, a more upstream locale, simulated a full variety of scenarios, including large quantity water withdrawals (90% of the total available flow).

A 90% reduction in total flow from Zone 1 resulted in only a slight increase (0.23°C) in temperature 2.4 km downstream (Figure 3.7). Temperature increased by 1.64°C immediately downstream of pumping zone 1, from 14.4°C to 16.0°C with a 90% total flow reduction in July. Thus the impact of flow removal from zone 1 was limited to the immediate downstream reach, and was relatively benign given the already low baseline temperature in the lower half of this river.

Pumping in zone 2 had larger and more ecologically significant effects. Mean July stream temperature 0.1 km downstream increased by 2.5°C (from 19.8°C to 22.3°C) with a 90% total flow reduction; while August mean stream temperature increased 2.6°C, from 20.6°C to 23.2°C with a 90% total flow reduction (Figure 3.8). The magnitude of modeled temperature changes caused by removal at zone 2 was also similar when measured at 0.7 km below at the lowest reach.

Across the analysis period of approximately two years, pumping in zone 1 did not show any adverse impacts (increase in lethal events) to coldwater fish species. However, flow withdrawal in zone 2 had more substantial effects. Starting at the 30% total flow reduction, potentially lethal thermal events started to occur in the bottom reach of Upper Cedar Creek (reach b, 4.7 km below pumping zone 2). The next upstream reach (reach c, 2.3km below pumping zone 2) began to have lethal events occurring

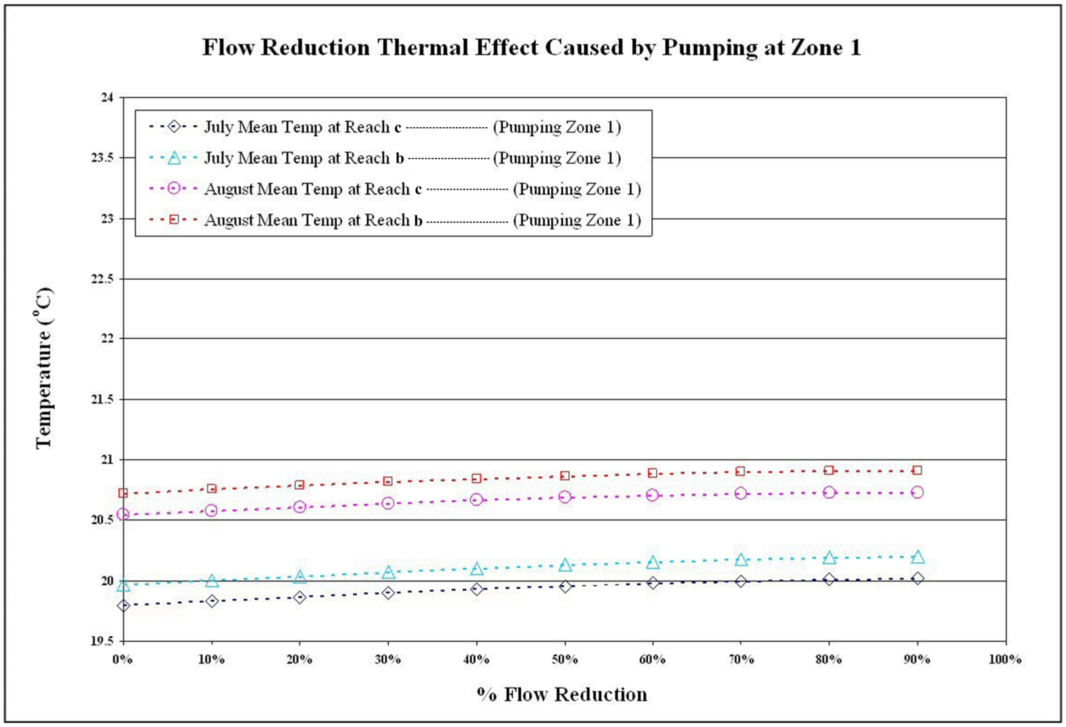


Figure 3.7: Experiment 2a (Cedar Creek): Modeled temperature response at downstream segments of Upper Cedar Creek (reaches b and c) as the total flow is reduced by pumping in Zone 1.

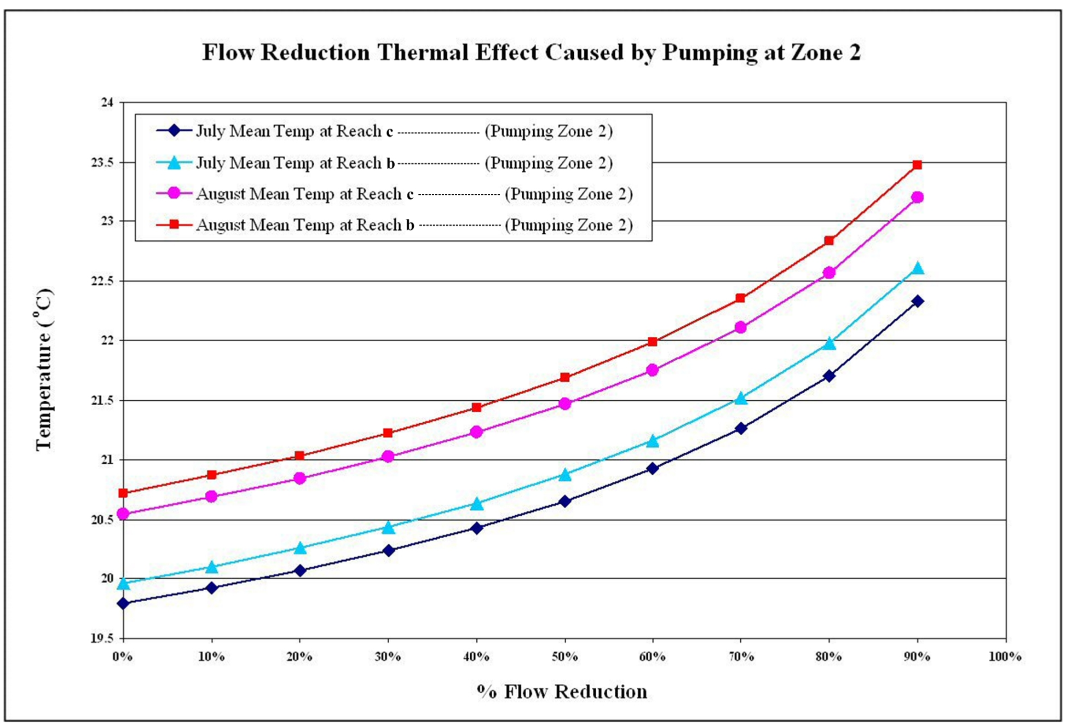


Figure 3.8: Modeled temperature response at downstream segments of Upper Cedar Creek (reache b (4.7 km below pumping zone 2) and reach c (2.3 km below pumping zone 2) as the total flow is reduced by pumping in Zone 2.

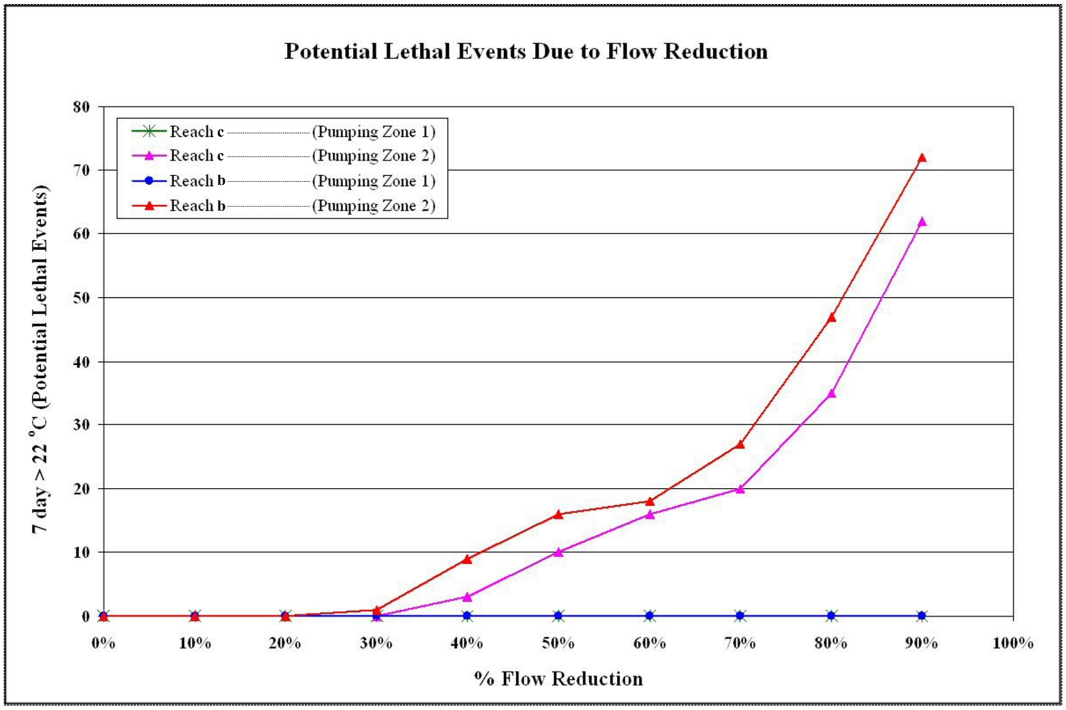


Figure 3.9: Modeled number of potential lethal events in a 21 month period on coldwater fish species using a seven-day average temperature greater than 22○C as a threshold.

at the 40% total flow reduction (Figure 3.9). With 90% total flow loss, lethal events were occurring 79%, and 68% of the time in July and August in reaches b and c, respectively.

4. Experiment 2b: Local Withdrawal from a Warm Water Stream (Mill Creek)

   In Mill Creek, simulated pumping in Zone 1 (upstream of the main branch) decreased the heating rate (defined as changes in water temperature per distance) of several downstream reaches; the effect differed depending on reach location. In the baseline run, July mean water temperature increased 1.28°C over a distance of approximately 20.5 *km* between upstream and the outlet of Mill Creek. In the Zone 1 pumping experiment, however, mean water temperature increased only 1.1°C, 0.8°C, 0.5°C, and 0.2°C over baseline temperatures given a 10%, 30%, 50%, and 70% total flow withdrawal respectively. With a 90% total flow withdrawal at Zone 1, modeled water temperature decreased 0.2°C. Trends of modeled temperature change at other lo-

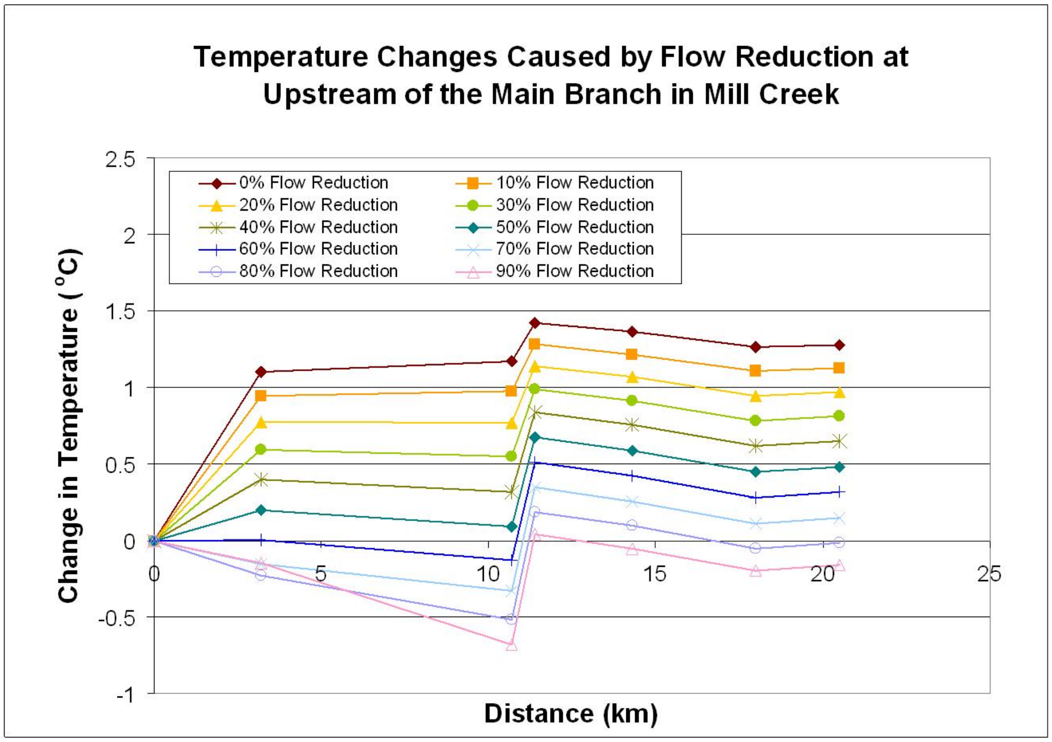


Figure 3.10: Changes in modeled 2003 July mean water temperature (y axis) at different distances (x axis) from J5 to the outlet caused by pumping at upstream of the main branch in Mill Creek.

cations along the lower main branch in Mill Creek were similar but with different magnitudes (Table 3.6).

When the pumping experiment was moved to Zone 2 (upstream of the north branch), I also observed that higher rates of flow reduction would decrease the downstream heating rate. The response was location dependent. Without pumping (i.e., baseline), 2003 July mean water temperature increased 1.04°C over a distance of approximately 14 *km* from the bottom of the Pumping Zone 2 to the confluence of the North and Main Branch. However, modeled 2003 water temperature with pumping in Zone 2 increased only 0.9°C, 0.7°C, 0.5°C, 0.2°C, and 0.0°C at 10%, 30%, 50%, 70%, and 90% total flow withdrawal level respectively. Without pumping (baseline), 2003 July mean water temperature increased 0.9°C over a distance of approximately 23.1 *km* from the bottom of the Pumping Zone 2 to the outlet of Mill Creek. Nonetheless, modeled 2003 July mean water temperature increased 0.8°C, 0.6°C, 0.3°C, 0.1°C at

Table 3.6: Modeled July and August mean water temperature change caused by different percentage of total flow reduction at upstream of the main branch in Mill Creek in years 2003 and 2004.

Reach	**J5-J52**	**J5-J6**	**J5-J4**	**J5-USGS Gage**	**J5-J2**	**J5-Outlet**
Distance (*km*)	3.2	10.7	11.4	14.3	18.0	20.5
Water Withdrawal (%)	Changes in 2003 July Mean Water Temperature (°C)					
0%	1.1	1.2	1.4	1.4	1.3	1.3
10%	0.9	1.0	1.3	1.2	1.1	1.1
20%	0.8	0.8	1.1	1.1	1.0	1.0
30%	0.6	0.6	1.0	1.0	0.8	0.8
40%	0.4	0.3	0.8	0.8	0.6	0.7
50%	0.2	0.1	0.7	0.6	0.5	0.5
60%	0.0	-0.1	0.5	0.4	0.3	0.3
70%	-0.2	-0.3	0.4	0.3	0.1	0.2
80%	-0.2	-0.5	0.2	0.1	-0.1	0.0
90%	-0.1	-0.7	0.0	-0.1	-0.2	-0.2
Water Withdrawal (%)	Changes in 2004 July Mean Water Temperature (°C)					
0%	1.1	1.1	1.4	1.3	1.2	1.3
10%	0.9	0.9	1.2	1.2	1.1	1.1
20%	0.7	0.7	1.1	1.0	0.9	0.9
30%	0.5	0.5	1.0	0.9	0.7	0.8
40%	0.3	0.3	0.8	0.7	0.6	0.6
50%	0.1	0.0	0.6	0.6	0.4	0.5
60%	-0.1	-0.2	0.5	0.4	0.2	0.3
70%	-0.2	-0.4	0.3	0.2	0.1	0.1
80%	-0.3	-0.6	0.2	0.1	-0.1	0.0
90%	-0.2	-0.7	0.0	-0.1	-0.2	-0.2
Water Withdrawal (%)	Changes in 2003 August Mean Water Temperature (°C)					
0%	1.0	1.1	1.3	1.2	1.2	1.2
10%	0.8	0.9	1.2	1.1	1.0	1.0
20%	0.7	0.7	1.0	1.0	0.9	0.9
30%	0.5	0.5	0.9	0.8	0.7	0.8
40%	0.3	0.3	0.8	0.7	0.6	0.6
50%	0.2	0.1	0.6	0.5	0.4	0.5
60%	0.0	-0.1	0.5	0.4	0.3	0.3
70%	-0.1	-0.3	0.3	0.3	0.2	0.2
80%	-0.2	-0.4	0.2	0.1	0.0	0.0
90%	-0.1	-0.5	0.1	0.0	-0.1	-0.1
Water Withdrawal (%)	Changes in 2004 August Mean Water Temperature (°C)					
0%	0.8	0.9	1.0	1.0	1.0	1.0
10%	0.7	0.7	0.9	0.9	0.9	0.9
20%	0.5	0.6	0.8	0.8	0.8	0.8
30%	0.4	0.5	0.7	0.7	0.7	0.7
40%	0.3	0.3	0.6	0.6	0.6	0.6
50%	0.1	0.2	0.5	0.5	0.4	0.4
60%	0.0	0.0	0.4	0.4	0.3	0.3
70%	-0.1	-0.1	0.3	0.3	0.2	0.2
80%	-0.2	-0.2	0.2	0.2	0.1	0.1
90%	-0.1	-0.3	0.1	0.1	0.0	0.0

Table 3.7: Modeled July and August mean water temperature change caused by different percentage of flow reduction at upstream of the north branch in Mill Creek, 2003 and 2004.

Site	JNF-J4		JNF-USGS Gage		JNF-J2		JNF-Outlet	
Distance (*km*)	14		16.9		20.6		23.1	
Time	**July 03**	**July 04**	**July 03**	**July 04**	**July 03**	**July 04**	**July 03**	**Jul 04**
% Withdrawal	**Changes in Water Temperature(°C)**							
0%	1.0	1.0	1.0	1.0	0.9	0.9	0.9	0.9
10%	0.9	0.9	0.9	0.9	0.8	0.7	0.8	0.8
20%	0.8	0.8	0.8	0.7	0.7	0.6	0.7	0.6
30%	0.7	0.7	0.6	0.6	0.5	0.5	0.6	0.5
40%	0.6	0.6	0.5	0.5	0.4	0.4	0.4	0.4
50%	0.5	0.4	0.4	0.4	0.3	0.3	0.3	0.2
60%	0.3	0.3	0.3	0.3	0.2	0.1	0.2	0.1
70%	0.2	0.2	0.2	0.2	0.0	0.0	0.1	0.0
80%	0.1	0.1	0.0	0.0	-0.1	-0.1	-0.1	-0.1
90%	0.0	0.0	-0.1	-0.1	-0.2	-0.2	-0.2	-0.2
Time	**Aug 03**	**Aug 04**	**Aug 03**	**Aug 04**	**Aug 03**	**Aug 04**	**Aug 03**	**Aug 04**
% Withdrawal	**Changes in Water Temperature(°C)**							
0%	0.9	0.8	0.9	0.8	0.8	0.7	0.8	0.7
10%	0.8	0.7	0.8	0.7	0.7	0.6	0.7	0.6
20%	0.7	0.6	0.7	0.6	0.6	0.5	0.6	0.5
30%	0.6	0.5	0.6	0.5	0.5	0.5	0.5	0.5
40%	0.5	0.4	0.5	0.4	0.4	0.4	0.4	0.4
50%	0.4	0.4	0.4	0.3	0.3	0.3	0.3	0.3
60%	0.3	0.3	0.3	0.3	0.2	0.2	0.2	0.2
70%	0.2	0.2	0.2	0.2	0.1	0.1	0.1	0.1
80%	0.1	0.1	0.1	0.1	-0.1	0.1	0.1	0.1
90%	0.1	0.1	0.1	0.1	-0.1	0.0	-0.1	0.0

10%, 30%, 50%, and 70% total flow withdrawal level respectively. Again, at the 90% total flow withdrawal, modeled water temperature even decreased 0.1°C. At 90% total flow reduction level, predicted July mean water temperature decreased the most at the downstream end of the lower mainstem of Mill Creek (i.e., J2; about 0.2°C decrease on average), and then warmed up a little at the outlet (about 0.02°C warmer than J2).

Mill Creek is not a suitable habitat for coldwater fish species;no analysis of potential lethal events for coldwater species was performed.

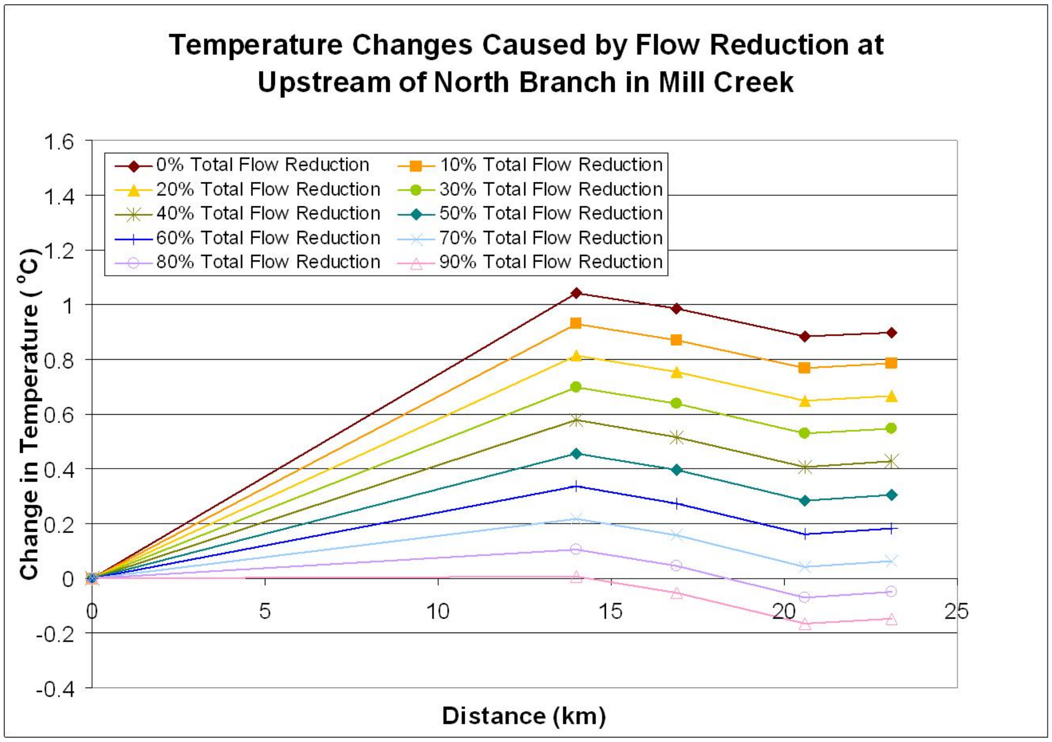


Figure 3.11: Changes in modeled 2003 July mean water temperature (y axis) at different distances (x axis) from JNF to the outlet caused by pumping at upstream of the North Branch in Mill Creek.

3.5 Discussion and Conclusion

In Experiment 1, the modeled summer stream temperatures were sensitive to flow reductions, but that impact was highly variable in space and depended strongly on upstream hydrology, flow volume, and position of the reach in the drainage network (i.e., time of travel). At a given location, the thermal impacts of total flow reduction on stream temperature were often similar to those of groundwater withdrawal. However, in places where groundwater dominated the hydrology, groundwater reductions had a stronger impact than total flow reduction. Conversely, in a surface-water-dominated channel reach, the thermal impacts from total flow withdrawal were relatively slight.

In Experiment 2 (a and b), where flow reductions were restricted to specific zones, the overall result was consistent with those observed in Experiment 1. Again, the amount of groundwater downstream was an important factor controlling the magnitude of change in downstream heating rate. Travel time was another critical factor. In the Mill Creek (Experiment 2b), water withdrawal upstream actually reduced the heating rate downstream by reducing the volume-weighted travel time in some segments below a confluent cooler water reach. This could happen because groundwater is generally very limited in the mainstem of Mill Creek, but a few tributaries of the North Branch carry an unusually high groundwater contribution. As more total flow was withdrawn upstream, it enhanced the cooling effect of downstream groundwater inputs on the remaining water volume. In addition, the distance below the withdrawal strongly affected the modeled response, since longer distances provided greater exposure to solar energy and larger influences from air temperature. In Mill Creek, traveling a greater distance also provided additional cooling from the relatively larger amounts of downstream groundwater. As a result, although water warmed immediately below the pumping zone, modeled temperature at some downstream locations dropped, not increased, in response to pumping. This surprising spatial variation illustrates the potential complexity of thermal regime response in river networks and the desirability of detailed thermal modeling as a part of the permitting process.

Interpreting the thermal regimes through the biological filter of potential lethal events also provided evidence for a strong correlation between flow reduction and declining coldwater habitat quality at reaches adjacent to pumping activities. However, the magnitude of the effect varied markedly between experiments and among sites. My modeling experiments suggest that it not only depends on pumping location, but also river hydrology (network structure), rate of withdrawal, and spatial patterns of flow accrual downstream. Although it is fair to generalize that in coldwater systems, the higher the groundwater withdrawal, the more the potential for impacts on coldwater fish species survival.

The State of Michigan enacted a new groundwater withdrawal law in 2006. Later in 2008, Michigan ratified the Great Lakes-St. Lawrence River Basin Water Resources Compact for the increased protection of the waters of the Great Lakes Basin. Public Acts 179 through 189 of 2008 were amended to bring the State's water withdrawal registration and permitting system into conformity with the Compact. A water-withdrawal assessment process and Internet-based screening tool were then developed to evaluate if proposed new or increased high-capacity water withdrawals will cause an adverse resource impact, and affect the ability of a stream to support the characteristic fish population (Reeves *et al.*, 2009).

This legislation is especially critical in rivers with regional groundwater sources as the results showed very different thermal response to water withdrawal. Based on my findings, large regional groundwater loading downstream may, but also may not, ameliorate the heating effects from water withdrawal upstream. For instance, in Experiment 1, Brooks and Bigelow Creeks both did not experience additional lethal events even at 90% flow reduction. In the Middle Branch, a few lethal temperature events started to appear at 70% flow reduction, while in the Little Muskegon River, lethal temperature events started to appear with 30% or more flow reduction. On the other hand, Experiment 2a showed that with ample groundwater sources, thermal impacts due to localized upstream pumping could still occur. My results suggested that 30% water withdrawal might be a removal threshold in terms of new legislative rule. A similar threshold was also recommended by Michigan Groundwater Conservation Advisory Council for their evaluated 11 river types (Anonymous, 2007).

Although a 30% withdrawal threshold might be a reasonable boundary to protect streams and their supporting fish fauna from the impacts of large-scale water withdrawal, it is important to understand that thermal regime responses to water withdrawal are highly dependent upon location and specific hydrologic context. Moreover, flow-biotic relationships are highly variable among species and river types (Arthington *et al.*, 2006; Zorn *et al.*, 2008). Zorn et al. (2008) indicated that "20 to 50% of the summer base flow could be removed without serious adverse impact to fish assemblages in all river types except the cold transitionals" (Zorn *et al.*, 2008). As a result, a site-specific analysis has been included into the Michigan's implementation legislation (Public Act 181 of 2008) for withdrawals falling into certain categories. This revision reflected the understanding of the aforementioned complexity and the possibly large range of variation in withdrawal impacts.

In conclusion, this study used a series of modeling experiments to demonstrate that (1) both surface water and groundwater removals have potentially serious impacts on river thermal regimes, and (2) that there is an inherently large spatial variability in thermal responses to unit flow removal. The magnitude, and sometimes direction of thermal impact, appears to be location specific shaped by patterns of local groundwater flux, mixing ratios, and travel times. Wells or water extractions closer to the river are likely to have larger impacts on thermal regime. Yet, large groundwater loading downstream could ameliorate local groundwater losses. More importantly, it might alleviate the heating effects from consumption of upstream local groundwater and/or surface water flow. Therefore, sensitivity to water extraction is likely to be spatially variable and site specific, and modeling heat transport implications will likely be necessary to evaluate ecosystem sensitivity. In this context, the simplified modeling approach of RPSTM can be a useful tool for explicit analysis of impacts. These modeling experiments were requested and performed under the contract with the Michigan Groundwater Advisory Council and have already provided useful information for the discussion of how environmental protection standards can best be developed to provide for the needs of our citizens without threatening our natural resource heritage.

CHAPTER IV

SIMULATING THERMAL IMPACTS OF DAM REMOVAL ON THE EARLY LIFE HISTORY OF GREAT LAKES ANADROMOUS FISHES AND THEIR DISTRIBUTION IN THE MUSKEGON RIVER WATERSHED IN MICHIGAN, U.S.A.

4.1 Abstract

The Muskegon River is one of the most popular fishing rivers in Michigan. It provides high quality nursery habitats for several highly-valued game fish species, including steelhead (*Oncorhynchus mykiss*), walleye (*Stizostedion vitreum*), and chinook (*Oncorhynchus tshawytschain*). However, three major hydropower dams located in the middle of the river's mainstem do not allow passage for fish and make the potentially valuable upstream habitats for spawning and nursery inaccessible.

On the Muskegon River proponents of dam removal have argued for clear benefits to the three above-mentioned fish species. They have argued that removal will both make additional upstream habitat available and improve the thermal habitats of the lower river.

I applied my newly developed RPSTM (Reduced Parameter Stream Temperature Model) to estimate stream temperature under "with dams" and "dams removed" scenarios at 20 sites in the Muskegon River from 1996 to 2005. Under the circumstances of "dams removed", the simulated stream water temperature at sites right at or below dams would be lower all year round, compared with the modeled temperatures when dams were present. Taking only thermal effects into account, the simulated results have revealed that dam removal

would cause delays of 8 to 10 days in initiation of spawning for steelhead and walleye, but no significant delays for chinook. At the same time, timing of fry emergence would be delayed for all three modeled fish species, although length of delay varied between species (7 to 15 days).

4.2 Introduction

There are more than 75,000 dams (with storage of greater than 50 acre-feet), and more than perhaps hundreds of thousands of smaller dams (<15 acre-feet) across the United States (Anonymous, 1999), affecting in various ways the physical and ecological characteristics of the waterways in which they are found. Many of these dams in the United States have reached their designed life expectancies, and are now under relicensing review by the Federal Energy and Regulatory Commission (FERC) (Bartholow *et al.*, 2005) or other state and local authorities. With a growing awareness of the importance of environmental preservation and the protection of fish species and other forms of wildlife (Doyle *et al.*, 2002), dam removal offers a viable alternative for river management, should the dams' licenses fail to be renewed. As part of a larger trend, the past 30 years have seen the removal of more than 600 dams in the U.S.; most of these were small, low-head, and run-of-the-river (Maloney *et al.*, 2008). A number of studies have indicated that top-draw hydropower operations would increase summer water temperatures and reduce winter water temperatures, which may dramatically affect the productivity of salmonids and alter the timing of their life history (Horne *et al.*, 2004; Tayler & Rutherford, 2007).

Anadromous rainbow trout, also known as steelhead (*Oncorhynchus mykiss*), chinook salmon (*Oncorhynchus tshawytscha*), and walleye (*Sander vitreus*) are among the most important sport fisheries in the Great Lakes. The timing of these fishes life history events is strongly tied to water temperature (Newcomb & Coon, 1997; Köster *et al.*, 2003; Horne *et al.*, 2004; Wehrly *et al.*, 2006). Water temperature in streams and rivers influences species distributions, individual growth and metabolism, and species composition (Bartholow, 1991; Bartholow *et al.*, 1993; Wiley *et al.*, 1997; Kapetsky, 2000; Borman & Larson, 2003; Wiley *et al.*, 2003a).

The Muskegon River Watershed lies in the central western part of Michigan's Lower Peninsula. The large number of groundwater-fed tributaries makes this river an important spawning habitat for cool- and cold-water fishes, with significant annual production of natural runs of steelhead, chinook salmon, and walleye (*Sander vitreus*) for regional sport fisheries (O'Neal, 1997; Wiley *et al.*, 2010). Steelhead and chinook salmon were introduced into the Great Lakes in the late 1800s (Biette *et al.*, 1981; Bartron & Scribner, 2004; Horne *et al.*, 2004). Due to their high recreational values, management agencies around the Lake Michigan basin have made an effort to sustain their populations, mostly by stocking. Unlike steelhead and chinook salmon, walleye are native to the Great Lakes. In the Muskegon system, walleye may move freely between the Muskegon River, Muskegon Lake and Lake Michigan throughout the year (Hanson, 2006). However, scientists have found that walleye are unable to sustain themselves in the Muskegon River without human assistance. Thus, the Michigan Department of Natural Resources and Environment (MDNRE) has launched a program of harvesting walleye eggs and stock at sites right below Croton Dam, the most downstream dam on the Muskegon River.

In addition to Croton Dam (constructed in 1907), Rogers Dam (constructed in 1906) and Hardy Dam (constructed in 1931) located within the lower half of river's mainstem, have posed problems for steelhead, chinook salmon, and walleye populations, by blocking access to abundant upstream habitat. On one hand, these water reservoirs provide important ecosystem services, including hydropower generation, irrigation, municipal and industrial water supply, recreational use, and flood control (Caduto, 1990; McCully, 1996; Hadley & Emmett, 1998; Bushaw-Newton *et al.*, 2002). On the other hand, a wealth of existing scientific evidence shows that large mainstem dams lead to alterations in river flow, water quality, sedimentation process, and the subsequent degradation of riverine ecosystems (Caduto, 1990; Hadley & Emmett, 1998; Graf, 1999; Pringle *et al.*, 2000; Bushaw-Newton *et al.*, 2002; Doyle *et al.*, 2002; Pohl, 2002; Horne *et al.*, 2004; Finger *et al.*, 2006; Jager & Smith, 2008; Maloney *et al.*, 2008).

In 1994, each of the three hydropower dams on the Muskegon River received a 40-year relicensing from FERC. Meanwhile, FERC also mandated a modification of the Muskegon River from peaking flows to run-of-river flows for the purpose of increasing habitat for, and production of, early life stages of salmonids and other fishes (Weisberg & Burton, 1993; Travnichek *et al.*, 1995). Although the dams on the Muskegon River were issued 40-year licences, dam removal could have been an outcome of the process. Given the fact that the removal of dams is not only costly but also controversial, future decision makers will need scientific evidence of the many costs and benefits of a dam's removal. The thermal impacts of dam removals are complex, poorly understood, and apparently highly variable. The majority of documented case studies to date have focused on relatively small dams. However, each dam varies in its hydrologic and hydraulic characteristics, associated watershed and location, and so the thermal and ecological effects of its removal are likely to vary (Pohl, 2002; Velinsky *et al.*, 2006). Understanding how changes in temperature associated with either dam construction or dam removal impact both species of interest and the ecosystem as a whole is a crucially important aspect of modern river management. Clearly, there is a need to understand the potential net thermal and ecological beneficial impacts of dam removal before a decision can be made.

With the hope of informing future management of the steelhead, chinook, and walleye fisheries in the Muskegon River, I applied my newly developed RPSTM (Reduced Parameter Stream Temperature Model) to dynamically predict stream temperature under "with dams" and "dams removed" scenarios. My objectives in this paper are: 1) to compare the the impacts of these dams and their removal on lower river thermal regimes, 2) to quantify their impacts on the timing of the early life history stages of steelhead, chinook, and walleye, and 3) to explore the potential amounts of thermally suitable habitats above the dams for steelhead, chinook, and walleye.

4.3 Methods

This study integrated hydrological, stream temperature, and biological/fishery models to examine and explore the impacts of hypothetical dam removals. As dams are primarily known to alter flow regimes, I coupled process-based hydrological models to RPSTM to dynamically predict the associated changes resulting from changes in surface water or groundwater discharge. The simulated water temperatures were used to analyze potential timing changes of steelhead, chinook, and walleye early life stages. As dam removal will allow salmonids access to potential spawning and nursery habitats upstream, simulated water temperatures were summarized as July mean temperatures for searching thermally suitable habitats above dams.

4.3.1 Site Description

The Muskegon River is the second longest river in Michigan. At 350 *km* in length, the Muskegon River flows from Higgins and Houghton Lakes, southwest to the Muskegon Lake, which drains into Lake Michigan. The drainage basin encompasses about 7,057 km^2 (Figure 4.1). The Muskegon River contains many groundwater fed tributaries, which help to sustain cold- and cool-water species (Clapp *et al.*, 1990; Brazner *et al.*, 2004, 2005; Creque *et al.*, 2005; Wehrly *et al.*, 2006). The Muskegon River has also been an impounded river system over 150 years. Today, there are three major dams lying on the river (Table 4.1). The most upstream dam is Rogers Dam, which was built in 1906. It is located on the Muskegon main stem 133.9 *km* above the mouth. Hardy Dam is in the middle of the three dams. It was built in 1931, and is located 40.0 *m* below Rogers Dam. Croton Dam, built in 1907, is at downstream about 9.4 *km* of Hardy Dam. These dams have been operated to schedule hydroelectric power for about a century. Besides the three major hydropower dams, there remain many other unregistered small dams located on tributary streams (O'Neal, 1997). Clearly, these impoundments provide human swimming, fishing, wildlife, and aesthetic benefits. However, harmful consequences of dams also exist in

Table 4.1: Locations of the modeled stream temperature sites in the Muskegon River.

Impoundment Characteristics	**Rogers**	**Hardy**	**Croton**
Location (km from the mouth)	133.9	93.9	84.5
Catchment Area (km^2)	4744.8	5008.9	6034.1
Residence Time (d)	1.8	46.5	5.9
Mean Depth (m)	3	6	2
Average Flow (cms)	38	41	53

terms of fisheries. For instance, dams present impassable barriers to adfluvial fish (such as salmonids, suckers, and walleye) spawning migrations. Moreover, the operation of dams results in changes of flow and stream temperature conditions, which strongly influence the distribution of fish populations (Tayler & Rutherford, 2007). The new licenses for these three hydropower dams issued by FERC will expire in 2033. More understanding of the relationship between existing dams or dam removals, and their impacts on riverine fisheries will be useful for the next licensing decision.

4.3.2 Thermal Impact Assessment

I modeled the stream temperature at 119 locations in the Muskegon River Watershed. Twenty of these locations are in the main stem of the Muskegon River from the mouth to Higgins Lake– 8 are downstream from the mouth to Croton Dam, 9 are above Rogers Dam, and 3 are in the major hydropower dam pools (i.e., Croton, Hardy, and Rogers) (Table 4.2). All other locations are in tributaries of the river (Figure 4.1).

The hydrological data (daily average surface water discharge, daily average groundwater discharge, and water travel time for each reach) required by RPSTM were estimated by HEC-HMS and ILHM (Hyndman *et al.*, 2006). Daily average solar radiation (kJ/cm^2) and daily average air temperature (°C) were also required to implement RPSTM. These climate data were observed from the Michigan Automated Weather Network (MAWN) (Anonymous, 2008) and from 9 commercial airport weather stations. Only one of the MAWN weather stations was located within the watershed, but two were near it.

Solar coefficients were estimated at the locations at reservoirs to assist in calibration

Table 4.2: Locations of the modeled stream temperature sites in the Muskegon River.

Site Name	**Location**
Site 1	Mouth of the Muskegon River to the Muskegon Lake
Site 2	North Channel of the Muskegon River
Site 3	Muskegon River at Mill Iron
Site 4	Muskegon Wastewater Treatment Plant
Site 5	Muskegon River at B31 Bridge
Site 6	Muskegon River at Bridgeton Bridge
Site 7	Muskegon River at Newaygo Bridge
Site 8	Muskegon River at Downstream Croton Dam (Below USGS Gauge)
Site 9	Muskegon River in Croton Dam Pond
Site 10	Muskegon River in Hardy Dam Pond
Site 11	Muskegon River in Rogers Dam Pond
Site 12	Muskegon River at Big Rapids (USGS Gauge)
Site 13	Muskegon River at Junction of the Hersey River
Site 14	Muskegon River at Evart (USGS Gauge)
Site 15	Muskegon River at Junction of Middle Branch
Site 16	Muskegon River at Junction of the Clam River
Site 17	Muskegon River at Down Reedsburg Dam
Site 18	Muskegon River at Reedsburg Dam
Site 19	Muskegon River at Houghton Lake
Site 20	Muskegon River at Higgins Lake

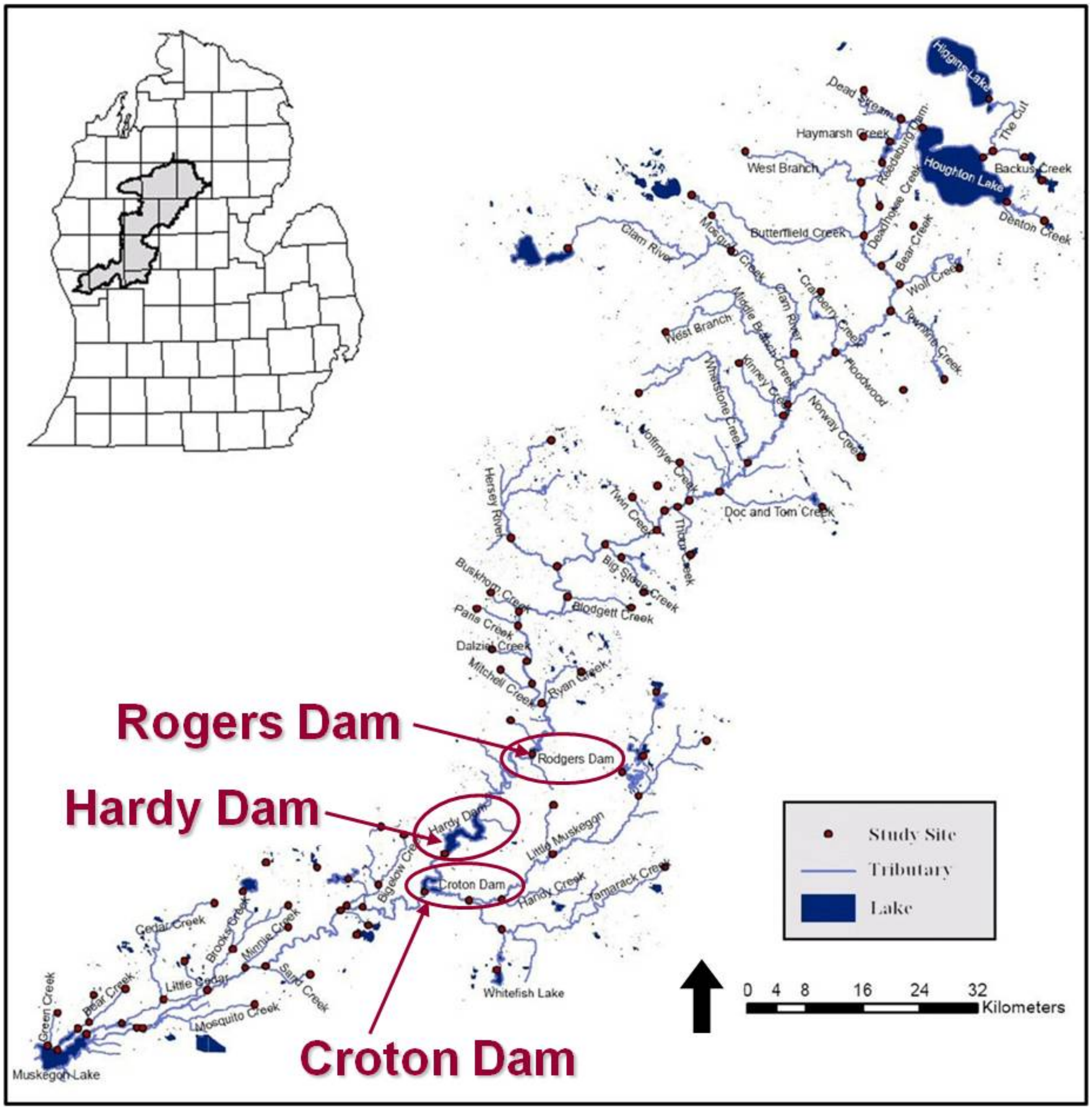


Figure 4.1: The locations of three major dams (Rogers, Hardy, and Croton) in the Muskegon River Watershed in Michigan (upper left corner), and the configuration of tributaries (blue lines) and the 119 modeled locations (red nodes).

of water temperature in the reservoirs. Other parameters needed for RPSTM implementation included the heat exchange coefficient, groundwater temperature, and hydraulic depth. The heat exchange coefficients were assumed 0.0092/depth (h^{-1}) for small channels, and 0.015/depth (h^{-1}) for large rivers and reservoirs, and were applied to the entire model. These values were derived from theory, and further confirmed by field and laboratory experiments (see Chapter II). Groundwater temperature was parameterized by calibrating modeled stream temperature to observed temperature in Cedar Creek, a tributary to the Muskegon River with many groundwater sources. The hydraulic depth (*m*), and residence time of the reservoirs (*h*), were parameterized by regression models (see Chapter II). Estimated hydraulic depth were calibrated to the observed data in the low-flow condition at approximately one hundred locations in the Muskegon River Watershed.

After calibration, RPSTM was applied to simulate daily average stream temperature given the current climate with predicted hydrologic data of dams in place in the HEC-HMS from 1996 to 2005. For the "dams removed" scenario, I restored the hydraulic depths and set the residence time of the dams into zero to reflect a free-flowing river if dams were absent. Then, by removing dam elements in the the HEC-HMS, I applied RPSTM with the modeled no-dams hydrologic data, and the aforementioned parametric changes, to predict daily average stream temperature for the same modeled period. The simulated temperature results were summarized into 10-year (from 1996 to 2005) monthly mean temperatures for comparison purposes.

4.3.3 Thermal Impacts on Fishes

1. Timing Shifts of Early Life History Stages

   The timing of fish life history stages are strongly tied into the stream temperature conditions. To examine impacts on early life cycle timing due to water temperature changes in relation to hypothetical dam removal, I applied simple timing and temperature criteria from literature reviews of three fish species: steelhead, chinook salmon, and walleye; all are important game fish in the Muskegon River. Locations examined were restricted to present fish spawning sites in the main Muskegon River below Croton Dam, including nearshore Muskegon Lake, the Muskegon main stem from Newaygo to Bridgeton, and in the reach immediately downstream from Croton Dam (O'Neal, 1997). When modeled stream temperature and time fell within the reported ranges, then that day was counted as a possible spawning date. I then estimated fry emergence date for that spawning day by accumulating temperature-units (TU; 1 TU = 1°C above freezing for 24 hours) until the reported total TU requirement was reached. Ten-year averages of the estimated early season spawning and fry emergence timing were calculated and compared to examine impacts of dam removal.

In the Muskegon River, the first spawning steelhead are found in March or April, when stream temperature reaches 4.4°C. The peak of the spawning activities are in mid-April, possible continuing into May, with a daily average stream temperature warms to 12.8°C (Biette *et al.*, 1981; Bell, 1991; Newcomb & Coon, 1997; O'Neal, 1997; Horne *et al.*, 2004; Richter & Kolmes, 2005). To include all possible spawning activities, I applied 4.4 to 12.8 °C as the criteria for daily average stream temperature to filter the modeled stream temperatures at each location from March to May. After spawning, I assumed that fry emerged after an accumulation of 230 to 340 TU (Myers & Horton, 1982; Murray & McPhail, 1988). I used 230 TU for early- and 340 TU for late-emergence.

Although a variety of stream temperature ranges have been reported for chinook salmon in different regions (Murray & Rosenau, 1989; Mullan *et al.*, 1992; Myers *et al.*, 1998; Murphy & Heard, 2001), I used a daily average temperature range from 7.2 to 14.5°C starting in September, as the criteria for fall chinook spawning in Michigan (Piper *et al.*, 1982; Connor *et al.*, 2002, 2003a,b). These values were particularly well matched to our observations for the timing and water temperature for fall spawning chinook in the river. I then estimated emergence by summing daily mean temperature from the first predicted spawning date until they accumulated 1000 TU (Seymour, 1956).

Walleye migrate up the river to spawn every spring. I used a daily average temperature in a range of 4.4 to 8.3°C from March to May as a thermal criterion (Baker & Manz, 1971; Roseman *et al.*, 1996; DePhilip *et al.*, 2005) for migration and spawning up river. Following spawning, I used 115 TU (Roseman *et al.*, 1996) as the threshold for fry emergence.

2. Thermally Suitable Upstream Habitats

   The upper Muskegon River contains many groundwater fed tributaries that are considered potential suitable habitats for salmonids. If dams were absent, these habitats would become available. In assessing these potentially suitable habitats, I applied the modeled alternative stream temperature along with a commonly used lethal temperature criterion for salmonids: a seven-day running average July and August stream temperatures greater than 22°C (Wehrly *et al.*, 2007). If a seven-day running average water temperature in a reach was modeled to be greater than 22°C, it was counted as a "potential lethal event". Presumably a single lethal event could substantially impact local fish populations. When a tributary contains no potential lethal event, it is considered a thermally suitable habitat.

   As fishes are not only constrained by water temperature, other habitat characteristics also influence the distribution of fish species. I further linked simulated water temperature along with modeled alternative flow and water quality outputs from other modeled portions of MREMS, to a modified version of Classification and Regression Tree (CART) (Steen *et al.*, 2006, 2010). CART, part of the Great Lakes Gap Analysis Program (GLGAP), is a regional fish habitat model that predicts presence and absence of a specific fish species for river segments. Linkage with CART provided more accurate and more useful prediction in quantifying suitable fish habitats.

Predictions were mapped into a VSEC-NHD (Valley Segment Ecological Classification Unit-National Hydrography Dataset) based map (Seelbach & Wiley, 2005; Seelbach *et al.*, 2006), which was used as the underlying spatial framework, for assessment of suitable habitats.

4.4 Model Results

4.4.1 Changes in Stream Temperature

In my simulations, locations close to and in reservoirs (i.e., Sites 9, 11, 17, 18, 19, and 20), as well as at the mouth (i.e., Site 1) of the Muskegon River, experienced the warmest stream temperatures in summer (Table 4.3 to 4.6). Simulated removal of Rogers, Hardy, and Croton dams lowered water temperatures for all sites below Rogers Dam pond from January to December in all years (Table 4.3 to 4.6). Not surprisingly, modeled temperatures at locations upstream of Rogers Dam showed negligible change between "with dams" and "dams removed" scenarios. For example, at Site 12, the modeled daily average temperatures were almost identical between "with dams" and "dams removed" scenarios. As a result, I restrict my discussion below to sites below Rogers Dam.

Simulations suggested that the largest thermal change would occur in the dam ponds (the impounded segments). The 10-year monthly mean temperature decreased by 0.26 to 0.94°C each month from January to December at Rogers Dam (Table 4.3), while it decreased from 0.22 to 0.69°C, and from 0.39 to 1.02°C, at Hardy Dam and Croton Dam, respectively (Table 4.4 and 4.5). The effect declined with distance away from the dams. Variation in change observed among sites ranged from 0.01 to 0.94°C. Site 1 (i.e., site at the mouth) was the least affected site by dam removal. The 10-year monthly average water temperature was lowered from 0.01 to 0.16°C from January to December at the mouth.

The impacts of dam removal also varied seasonally. In general, the smallest changes in water temperature occurred in January; although there were a few exceptions. In contrast,

Table 4.3: Comparison of modeled 10-year monthly mean stream temperature (°C) under "with dams" and "dams removed" scenarios at Rogers Dam.

Site: Rogers Dam			
Month	**With Dams**	**Dams Removed**	**Deviation**
1	1.5	1.2	-0.3
2	2.4	2.0	-0.4
3	3.1	2.6	-0.6
4	9.3	8.5	-0.7
5	14.4	13.7	-0.7
6	21.0	20.2	-0.8
7	24.7	23.8	-0.9
8	24.2	23.4	-0.9
9	21.8	20.9	-0.9
10	14.5	13.8	-0.7
11	7.9	7.4	-0.5
12	3.6	3.1	-0.4

the largest deviation of 10-year monthly mean temperature occurred in April for all the mainstream sites below Croton Dam (Table 4.6). For instance, water temperature below the dams fell 0.94°C at Sites 7 and 8 in April; while at Site 1, it fell only 0.16°C (Table 4.6). In currently impounded sites, the largest change in water temperature occurred in summer and early fall. For example, at Rogers and Croton dams, the largest 10-year monthly mean temperature deviations resulting from dam removals occurred in June or July; while at Hardy Dam, it occurred in September.

4.4.2 Changes in the Timing of Spawning and Fry Emergence

Fishes are sensitive to changes in water temperature. Modeling results indicated variable shifts for the timing of fishes early life history stages.

(1) Spawning Activities at Down Croton (Site 8)

At Site 8, an important current spawning location, modeling predicted that steelhead would spawn on average from March 25 to May 24, with year to year variation started as early as March 9 and as late as May 31. Without dams, modeling predicted a 10 day delay in the onset of spawning activity, which on average ranged from April 4 to May 24. Variation

Table 4.4: Comparison of modeled 10-year monthly mean stream temperature (°C) under "with dams" and "dams removed" scenarios at Hardy Dam.

Site: Hardy Dam			
Month	**With Dams**	**Dams Removed**	**Deviation**
1	2.1	1.5	-0.5
2	2.2	1.8	-0.4
3	2.7	2.2	-0.5
4	7.4	7.1	-0.3
5	12.7	12.4	-0.3
6	18.7	18.5	-0.2
7	23.2	22.9	-0.3
8	23.7	23.2	-0.5
9	21.8	21.3	-0.5
10	15.8	15.1	-0.7
11	9.0	8.4	-0.6
12	4.7	4.0	-0.6

Table 4.5: Comparison of modeled 10-year monthly mean stream temperature (°C) under "with dams" and "dams removed" scenarios at Croton Dam.

Site: Croton Dam			
Month	**With Dams**	**Dams Removed**	**Deviation**
1	1.4	1.0	-0.4
2	1.9	1.5	-0.4
3	2.6	1.9	-0.7
4	8.2	7.3	-0.9
5	13.8	12.9	-0.9
6	20.0	18.9	-1.0
7	24.6	23.6	-1.0
8	24.9	23.9	-1.0
9	22.7	21.8	-0.9
10	16.4	15.5	-0.8
11	9.1	8.5	-0.6
12	4.2	3.8	-0.5

Table 4.6: Comparison of modeled 10-year monthly mean stream temperature (°C) under "with dams" and "dams removed" scenarios at Site 1 to Site 8.

Site	**1**	**2**	**3**	**4**	**5**	**6**	**7**	**8**
Month	**With Dams (°C)**							
1	1.0	1.0	1.2	0.7	1.3	1.7	1.7	1.6
2	2.8	2.7	1.6	1.1	1.6	2.1	3.2	2.0
3	3.9	3.5	2.4	2.0	2.4	2.7	4.0	2.7
4	9.7	8.7	8.5	8.5	7.9	7.5	8.5	8.0
5	15.4	14.3	14.0	14.4	13.2	12.6	13.6	13.5
6	22.0	20.8	20.4	20.8	19.2	18.5	19.9	19.6
7	25.1	24.4	24.0	24.6	23.8	23.3	23.6	24.0
8	24.6	24.3	23.9	24.3	24.1	23.8	23.7	24.4
9	21.2	21.3	21.1	21.1	22.2	22.2	21.5	22.3
10	13.6	14.2	14.2	14.2	16.1	16.4	15.0	16.1
11	7.1	7.7	7.8	7.4	9.3	9.7	8.6	9.0
12	2.6	3.0	3.2	2.7	4.3	4.8	4.2	4.3
Month	**Dams Removed (°C)**							
1	1.0	1.0	1.2	0.6	1.2	1.5	1.4	1.1
2	2.8	2.7	1.5	0.9	1.3	1.8	2.8	1.4
3	3.8	3.4	2.1	1.7	1.9	2.2	3.5	1.9
4	9.5	8.4	8.1	8.1	7.2	6.6	7.6	7.1
5	15.2	14.1	13.7	14.1	12.6	11.9	12.9	12.7
6	21.9	20.6	20.1	20.5	18.6	17.8	19.0	18.7
7	24.9	24.2	23.7	24.2	23.2	22.6	22.8	23.4
8	24.5	24.0	23.7	24.0	23.6	23.2	23.0	23.8
9	21.1	21.1	20.8	20.8	21.7	21.6	20.9	21.7
10	13.5	13.9	13.9	13.9	15.7	15.9	14.5	15.6
11	7.0	7.5	7.6	7.2	9.0	9.4	8.2	8.6
12	2.5	2.9	3.1	2.5	4.0	4.5	3.8	3.9
Month	**Deviation = (Dams Removed) - (Current) (°C)**							
1	0.0	-0.1	-0.1	-0.1	-0.2	-0.2	-0.4	-0.4
2	0.0	-0.1	-0.1	-0.2	-0.3	-0.4	-0.4	-0.6
3	-0.1	-0.1	-0.2	-0.3	-0.5	-0.5	-0.6	-0.8
4	-0.2	-0.3	-0.3	-0.4	-0.7	-0.8	-0.9	-0.9
5	-0.1	-0.3	-0.3	-0.3	-0.6	-0.7	-0.8	-0.7
6	-0.1	-0.2	-0.2	-0.3	-0.6	-0.7	-0.9	-0.9
7	-0.1	-0.2	-0.3	-0.3	-0.6	-0.7	-0.8	-0.7
8	-0.1	-0.2	-0.3	-0.3	-0.5	-0.6	-0.7	-0.6
9	-0.1	-0.2	-0.3	-0.3	-0.5	-0.6	-0.6	-0.5
10	-0.2	-0.3	-0.3	-0.3	-0.5	-0.5	-0.5	-0.5
11	-0.1	-0.2	-0.2	-0.2	-0.3	-0.3	-0.4	-0.4
12	-0.1	-0.1	-0.2	-0.2	-0.3	-0.4	-0.4	-0.4

over the ten years was large with steelhead spawning occurring as early as March 10 and ending as late as May 31. With the dams in place, walleye spawning ranged from March 25 to April 26, varying over the ten years from March 8 to May 19. Simulated removal of the dams caused walleye's spawning activities to be delayed 8 days, and on average began on April 3 and ended on May 2, varying over the ten years from March 9 to May 20.

For chinook salmon, modeling suggested current spawning activities would occur on average from October 11 to November 27, varying over the ten years from October 8 to December 22. Removal of the dams did not cause chinook's spawning activity to change much. The timing of chinook spawning activity shifted only one day earlier– from October 10 to November 25 on average. Variation over the ten years for the timing of chinook spawning activity following dam removals ranged from October 4 to December 22 (Figure 4.2).

(2) Fry Emergence Immediately Downstream of Down Croton (Site 8)

At Site 8 when dams were present, steelhead hatched by April 29 to June 15 on average, varying over the ten years from April 12 to June 22. With dams absent, on average fry emergence were delayed from May 6 to June 16. , varying over the ten years from April 15 to June 23. At Site 8 when current dams-in, walleye fry emergence would take place from April 15 to May 7, with variation over the ten years ranged from March 26 to May 28. Without the dams, walleye emerged from April 22 to May 13, varying over the ten years from March 28 to May 29. As for fall spawning chinook salmon, with dams in place, fry emerged from April 26 to June 6, varying over the ten years from March 15 to June 15. If dams were removed, chinook fry emerged on average from May 6 to June 15, with variation over the ten years started from April 8, and ended on June 23 (Figure 4.2).

4.4.3 Potential Upstream Habitat

Based on temperature and access only, with dams in place, 10.4% (a length of 36.4 *km*) of the Muskegon Rivers' total channel miles are thermally suitable for fish that can tolerate

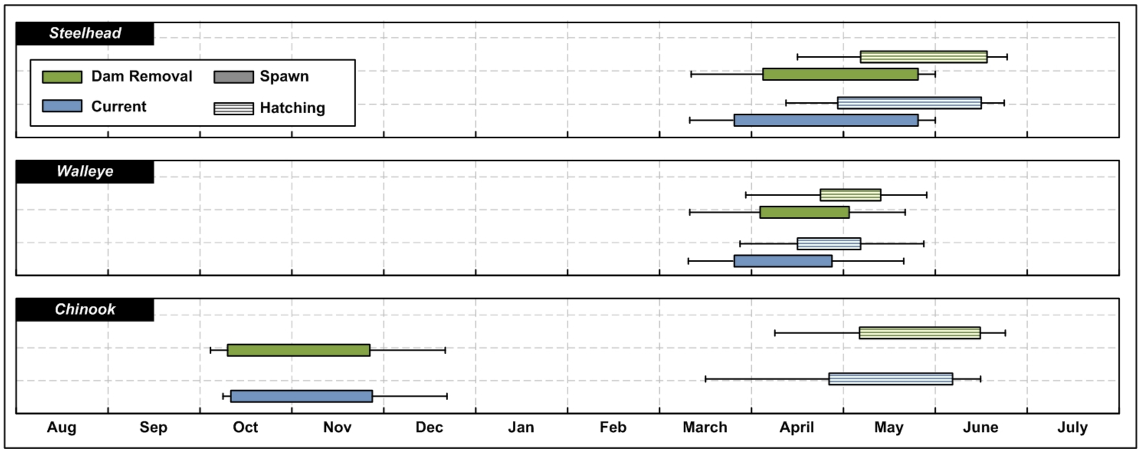


Figure 4.2: Predicted 10-year average timing shifts of spawning and hatching activities for steelhead, walleye, and chinook salmon at down Croton (Site 8). Solid lines indicate earliest to latest yearly variations.

7-day water temperature up to 22°C. If dams were absent, thermally suitable habitat would increase to 47.0% (a length of 164.5 *km*) of the Muskegon Rivers' total channel miles (Figure 4.3).

Using my thermal predictions to drive the Steen et al. (2010) macrohabitat model, I again found very large increases in potentially suitable habitat for each of the targeted sport fishes. In particular, if dams were removed, steelhead and walleye habitat would increase 200% and 250% respectively. Chinook habitat in the Muskegon River system would be increased by a more modest 75% (Figure 4.4).

4.5 Discussion and Conclusion

In this study, RPSTM was integrated with other MREMS models, providing more accurate estimates of water temperatures that would accompany the physical changes in hydraulic depth, residence time, and discharge under the circumstances of dam removal. My model predicted that dam removals would lower monthly stream temperature all year round. Collectively, the results have shown that series of dams in the Muskegon River had made the downstream waters slightly warmer in average on a monthly basis than the results under the "dams removed" scenario. These results were different than the prediction from a hypothetical dam removal study in the Manistee River in Michigan (Horne *et al.*, 2004). Horne

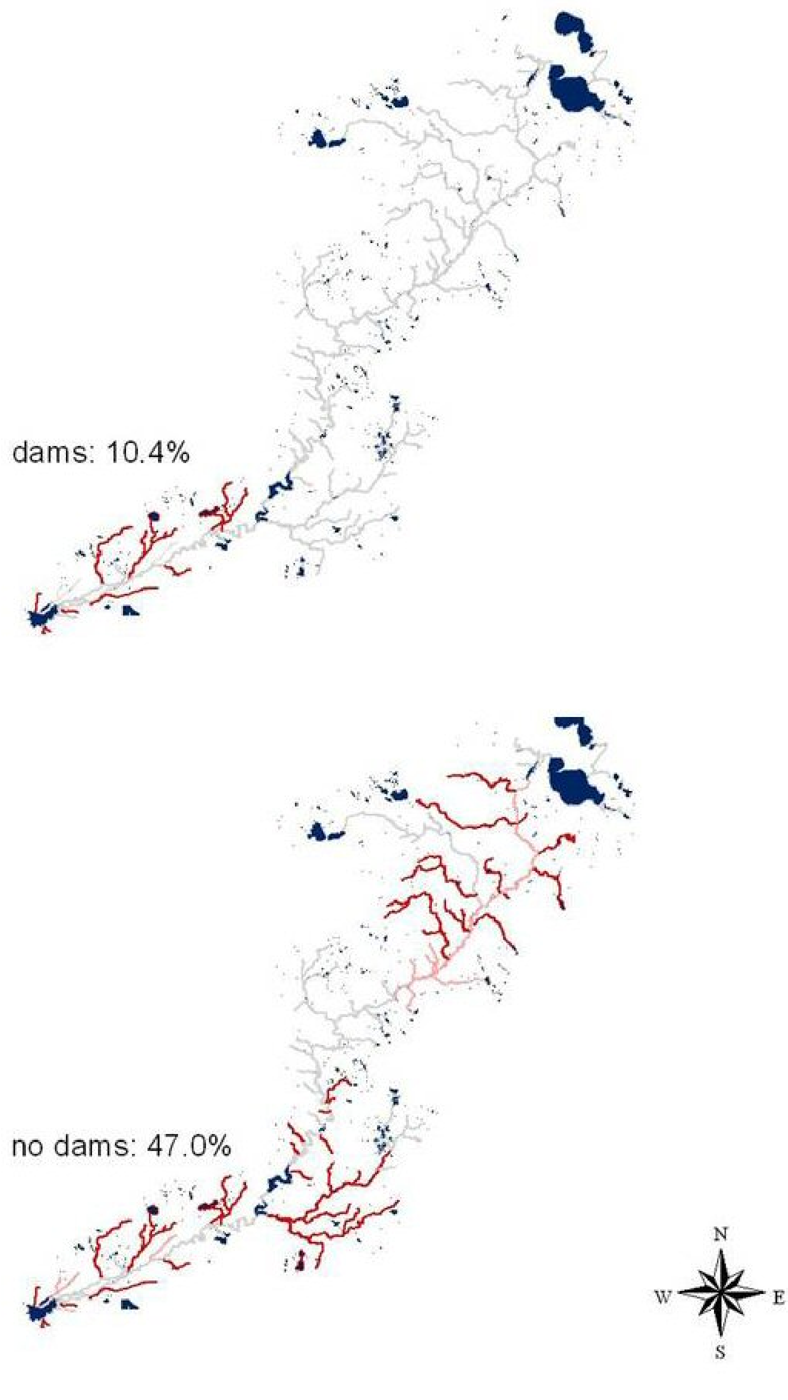


Figure 4.3: Model simulations of currently available habitat (dams) for fish that can tolerate 7-day water temperature up to 22°C, and of habitat that would be made available if Rogers, Hardy, and Croton dams were removed (no dams).

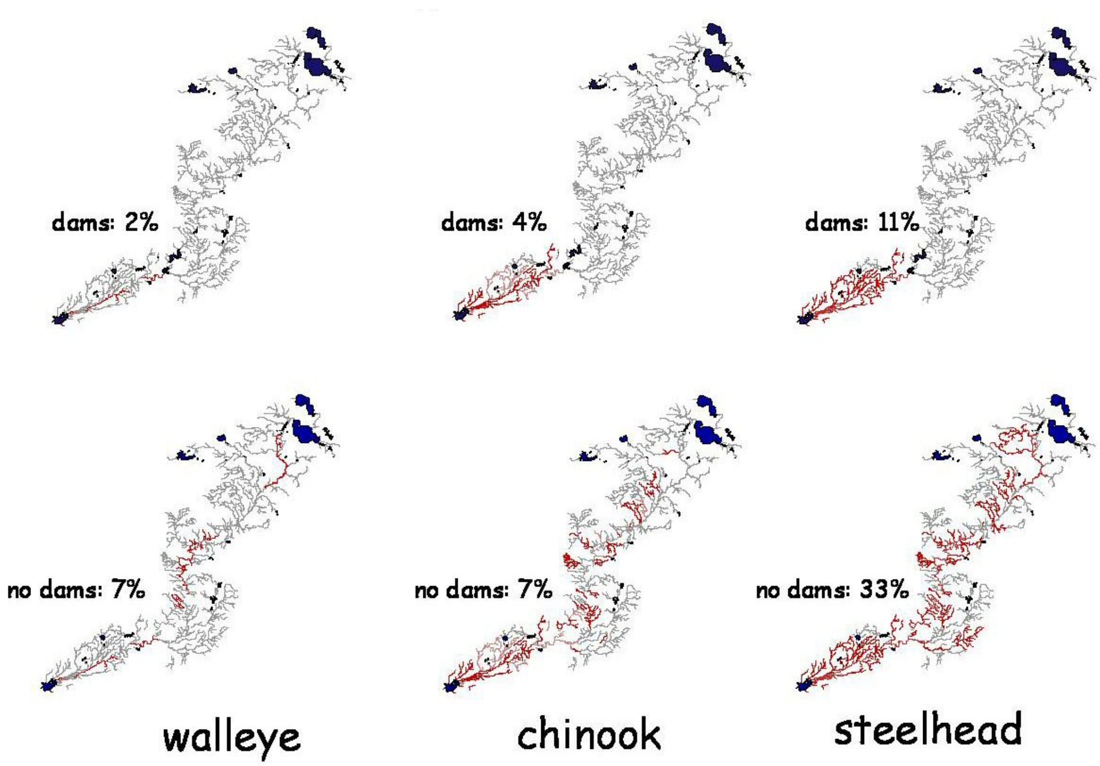


Figure 4.4: Model simulations of currently available habitat for steelhead, walleye, and chinook salmon (above), and of habitat that would be made available if Rogers, Hardy, and Croton dams were removed (below).

et al. (2004) found that removal of Manistee River dams would warm up sooner in spring and cool faster in fall than when impounded. The discrepancies of different direction of temperature changes between these two studies could possibly due to different approaches in hydrological estimation. Horne et al. (2004) did not as explicitly consider dynamic changes in hydrology in dam removal scenario. The hydrological models integrated with RPSTM predicted that without dams, total flow and hydraulic depths would be decreased at the reservoirs. Decreased water volume and shallower hydraulic depths could eliminate the thermal energy storage effects of impoundments, which in turn cause lower downstream temperatures. In addition, the hydrological models also found that dam removal would strengthen the accrual of ambient groundwater sources, which could further cool down the stream temperature in summer. Most importantly, when a river is under its natural hydraulic configuration after hypothetically removing the dams, a new temperature equilibrium would be reached due to adjustments of width, depth, and flow (Horne *et al.*, 2004; Velinsky *et al.*, 2006; Jager & Smith, 2008). As a result, different methods in simulating the changes of hydrology can potentially affect the estimation in stream temperatures.

The results were consistent with previous research findings, predicting that removing dams would result in the decrease of summer stream temperatures at the sites below the dams in Michigan rivers (Newcomb & Coon, 1997; Horne *et al.*, 2004). In particular, daily average temperatures in summer at sites right at or below dams were always lower than those with dams in place. My model predicted an average of 1°C decrease in summer water temperatures below the dams in the Muskegon River. Horne et al. (2004) also found that average summer water temperatures would decrease from 0.9 to 1.0°C without dams in the Manistee River. This is because both the Muskegon River and the Manistee River receive a large volume of groundwater input under the between the impoundments. With dams in place, the cooling impact from groundwater in summer is overwhelmed by the large warmer mass from impoundments' large water volume (Horne *et al.*, 2004). Removal of dams returns the river back to its natural condition, and allows groundwater inputs play sufficient cooling to the water.

My model further predicted that the magnitude of changes in stream temperature following dam removal varied temporally and spatially, depending on the distance to the dams, the hydrology, and the year-to-year climate conditions. In general, sites closer to the dams would experience greater temperature changes. As for sites farther from the dams, the thermal footprints of dams were diminished due to the longer distance of air-water interaction. Impacts of dam removal can also be varied due to the different geometry and geographic location of river network (Newcomb & Coon, 1997). In the Batsie River, a tributary of Lake Michigan, removing a dam from the upper reaches would have limited effect below dams because the inputs of warmer surface water from a headwater lake (Newcomb & Coon, 1997). Apart from the spatial changes, modeling results also varied year by year due to annual weather variation. The temporal variations were possibly caused by the influence of climatic cycle on air temperature, which eventually led to a similar trajectory on water temperature.

It is an important result in terms of fishery management in the Muskegon River that dam removal could effectively lower summer stream temperature and allow more habitat and time in the mainstream of the Muskegon River for coldwater species. For example, migrating adult steelhead would be less thermally stressed and find more thermal refuge in cooler tributaries in summer. Yet, the magnitude of the lowered summer stream temperature from dam removals varied temporally and spatially, depending on the distance to the dams, hydrology, and year-to-year climate conditions.

Analysis results supported the contention that removal of the three hydropower dams would bring large areas of usable habitats for the three targeted fish. In particular, the Little Muskegon River, a major tributary adjoining to the mainstem at where Croton Dam located, is currently fulfilling the 7-day 22°C criterion threshold. However, the existence of Croton Dam hinders possible fish movement. Dam removal would open the way and make this thermally suitable tributary accessible. CART analysis further confirmed that the Little Muskegon would be suitable for both steelhead and chinook, but not for walleye, due to other habitat characteristics, such as substrate and flow condition, instead of water temperature.

Apart from the adjacent Little Muskegon River, dam removals would in general bring a lot more usable habitat upstream for the targeted three migratory fish. My model predicted a doubled increase of usable habitats for steelhead and walleyes, and a 75% increase for Chinook salmon. In contrast, Creque (2002) estimated usable habitats would become tripled for steelhead, and 5-fold more for Chinook salmon above dams. Although both studies use similar variables in estimating usable habitats above dams for steelhead and Chinook salmon, hierarchies of the importance each variable has to the consideration of usable habitats are very different. For example, although both studies include water temperature and flow criteria of usable habitats for steelhead, different regression models can produce very different outputs. Besides, calculation of total usable area was very different in both studies. Difference in these modeling methods all contribute to the discrepancies. Nonetheless, results both suggested that suitable habitats above the dams in the Muskegon River would be large. Therefore, identifying suitable habitat for steelhead, walleyes, and Chinook salmon will be important to provide useful information for fishery managers to plan their stocking strategies.

The changes in modeled stream temperature following dam removal were certainly sufficient to affect the early life stages of steelhead, walleyes, and Chinook salmon. The modeling results revealed that the largest decrease in stream temperature would occur in April if dams were removed. This is a particularly sensitive time of year in terms of fish reproduction. Delays in spawning and hatching activities for steelhead and walleye were predicted. In contrast, fall chinook spawning moved slightly ahead; however, fry emergence for chinook was predicted to be delayed by one month on average. At the same time, the simulations revealed a longer incubation period for Chinook salmon eggs following dam removals. As for steelhead and walleyes, the thermally suitable hatching periods remain similar. Prolonged hatching time for Chinook salmon would therefore overlap more with the hatching activities of steelhead and walleye. This might imply potential inter-species competitions. For example, walleye eggs and fry currently do not sustain well in the Muskegon River because its often too cold temperature in spring (Ivan *et al.*, 2010), which implies a worse situation for walleyes after dam removal due to further decreased water temperature in spring. However, this study considered only water temperature in estimating the timing shifts in the early life stages. This could resulted in errors when predicting the changes in life history timing shifts. Day light and flow velocity are important factors affecting timing of fish spawning. In addition to water temperature, fish also choose where to spawn based on flow and substrates. Changes in timing of life history stages could also result in shifts of the structure for individual fish population and their community distribution. As dam removals might also cause dramatic changes in each species' major life history stage, not just in spawning and hatching, one might ask how the changes in one life history stage would affect other stages or other species. For example, the delayed timing of steelhead spawning might shorten hatching and emergence time for developing eggs (Burger *et al.*, 1985; Bartholow *et al.*, 2005). On the other hand, improved thermal conditions would make steelhead fry less stressed and more capable of dealing with other stressors, which would enhance their survival rate (Godby *et al.*, 2007). If survival rate were improved, and if juvenile growth were also increased, along with the appearance of more suitable thermal habitats and longer period of preferred stream temperature, a more complex intra- and inter-species competition may occur. Besides, since migrating adult steelhead would be less stressed by the summer stream temperature, their ability would be enhanced in coping with other stressors. This strengthened ability of adult steelhead could lead to an adverse effect on cool- and warm-water species. These factors were not considered in my study. Thus, a more detailed study of the causes and potential consequences of life history timing shifts will be required.

In conclusion, detailed daily water temperature modeling with RPSTM has provided useful information on the basis of physical changes in flow and water temperature and on the impacts to the fishery, allowing fishery managers to better plan for the future in advance. Apart from the species studied here, the Muskegon River has also been previously documented as important habitat for other river spawning species, such as the endangered lake sturgeon, and several species of suckers (O'Neal, 1997). Dam removal could also release usable spawning and nursery habitats upstream for other important riverine fish species. Furthermore, if air temperature is expected to increase further in the future under the impact of global climate change (Soloman *et al.*, 2007), dam removal might help in mitigating the negative impacts on the coldwater fish community. Consequently, this modeling approach will be useful in examine complex riverine systems for the future. A thorough cost-benefit evaluation of the operation/removal of hydroelectric facilities, including the accompanying social, physical, and ecological changes, will be needed to ensure that the negative impacts to human and riverine organisms that use these lotic habitats will be minimized.

CHAPTER V

IMPACTS OF GLOBAL WARMING ON THE EARLY LIFE HISTORY OF GREAT LAKES ANADROMOUS FISHES: A MODELING CASE STUDY

5.1 Abstract

Climate change is a serious issue both in the Great Lakes region and globally. To explore long-term, large-scale potential thermal impacts on Great Lakes fishes, I used a new reduced parameter heat balance model (RPSTM) and simulated detailed changes in seasonal temperature patterns for the Muskegon River, an important tributary of Lake Michigan. Predictions from the 4th IPCC A1B Scenario were used to drive a multi-modeling system which provided required inputs for RPSTM. Modeling suggested that the Muskegon River may experience 2 to 4.5°C warming in monthly water temperature. The greatest increases in modeled water temperature were predicted to occur in spring, especially during warmer than average years. Moreover, groundwater fed tributaries experienced greater water temperature change than large channels/reservoirs in winter. However, this spatial distinction was reduced during the summer. These changes in water temperature are likely to significantly alter the timing of steelhead, walleye, and Chinook salmon early life history. This integrated multi-modeling approach could be useful to fishery managers interested in planning management adaptations to cope with climate change.

5.2 Introduction

Reports of observed and potential impacts from a changing climate are accumulating rapidly (Soloman *et al.*, 2007). Reported impacts range from mountains (Lopez-Moreno *et al.*, 2008; Minder, 2010; Logan *et al.*, 2010) to oceans (Goberville *et al.*, 2010; Ruhland & Krna, 2010), from plants (Hou *et al.*, 2010; Sun *et al.*, 2010) to animals (Kissling *et al.*, 2010; Sauter *et al.*, 2010), and from individual species (Nardone *et al.*, 2010; Steen *et al.*, 2010) to the entire ecosystem (Mooij *et al.*, 2009; Wiley *et al.*, 2010). Although many reports are still controversial, long-term observations are helping scientists understand how climate is changing in space and time. For example, global warming is now known to result from increases of atmospheric concentrations, especially carbon dioxide and other green house gases. One of the principal ecological effects of climate change is the warming of earth's oceans, lakes, and rivers (Mohseni *et al.*, 1999).

Water temperature is a key driving variable governing the overall biological structure of riverine ecosystems (Bartholow *et al.*, 1993; Wehrly *et al.*, 2003). In particular, fish ecology and community dynamics are strongly tied to water thermal conditions (Diana *et al.*, 2004; Horne *et al.*, 2004; McRae & Diana, 2005; Meeuwig *et al.*, 2005; Wehrly *et al.*, 2006; Diana & Smith, 2008). In salmonids, for instance, life history staging is often triggered by changes in water temperature (Mullan *et al.*, 1992; Newcomb & Coon, 1997; McCullough, 1999; McCullough *et al.*, 2001), and due to their relatively narrow temperature tolerance, species distribution is extremely sensitive to the changes in water temperature (Jager *et al.*, 1997; McCullough *et al.*, 2001; Steen *et al.*, 2010)

Since temperature is such a crucial factor in shaping ecology of a riverine system, it is important to understand how climate-induced change may alter the natural spatial and temporal thermal regimes of aquatic ecosystems. Human-induced and natural changes in hydrology and climate may both contribute to future shifts in riverine thermal regimes (Webb & Clack, 1996; Brown *et al.*, 2006). North America is projected in the future to have warmer and wetter winters (Jonsson & Jonsson, 2009; Wiley *et al.*, 2010), with annual average air temperatures being 1.7°C to 4.4°C higher (Soloman *et al.*, 2007). Clearly, future water temperature regimes are likely to be very different and this will necessarily have strong impacts on the biology of river fishes.

To assess the potential changes of stream temperature in relation to future climate change, a modeling approach is required to link climate projections to changes in river thermal regimes. Currently, stream temperature models can be grouped into two types: empirical methods (Mohseni *et al.*, 1999; Mohseni & Stefan, 1999) and physically-based heat balance models (US Army Corps of Engineers, 1986; Sinokrot *et al.*, 1995; Bartholow, 2000). Mohseni and Stefan (1999) applied a simple regression function of air temperature to estimate weekly stream temperature under a warmer climate scenario. However, this empirical approach cannot be applied to shorter time scales (e.g. daily). More importantly, this kind of statistical relationship provides limited power for anticipating effects of future changes in climate since it incorporates no mechanism of heat fluxes associated with these climatic changes.

More mechanistic models follow deterministic heat transport theory to calculate changes in-stream water temperatures. For these "process-based" models, prediction of water temperature is frequently accurate within 1°C compared to observed field data. However, the existing physically-based heat balance models require a large number of spatially variable parameters and large amounts of local calibration data. For example, MNSTREM (Sinokrot & Stefan, 1993) and HEC-5Q models (US Army Corps of Engineers, 1986) both require many site-specific variables, including stream morphology, stream hydrology, meteorological variables, and stream bank vegetation types. While some parameter values may be easily obtained, many, such as local (reach scale) relative humidity, wind speed, and cloud cover, are not always available, especially in large regional studies.

Parameterization will be inherently risky under a climate change scenario due to uncertainties in future climate outcomes. The magnitude and even direction of parameter response to climate change are difficult to verify because of scaling and interaction concerns. Local weather conditions are very likely to differ from global or regional patterns predicted by General Circulation Models (GCM; (Deardorf, 1972; Sellers *et al.*, 1986; Pirtle *et al.*, 2010; Sinha *et al.*, 2010)) of future climate scenarios. In addition, the interactions of a warming climate with stream hydrology, geomorphology, and plant communities are likely complex. They might interact with each other in different relationships than empirical models, and therefore, the parameterization process will likely be more difficult and variable in regard to global warming.

The developed Reduced Parameter Stream Temperature Model (RPSTM) provides accurate water temperature estimation that requires fewer parameters, and supports dynamic, and spatially explicit hydrologic simulations. Here I use this temperature modeling approach to examine ways in which climate change may alter life history and distributional patterns of several important fishes in the Muskegon River Watershed, Michigan, USA.

The purpose of this study was to: (1) present a general methodology for water temperature prediction in the light of a climate change scenario; (2) provide a preliminary estimation of the impacts of climate change on thermal regimes; and (3) assess the potential shifts in early life stages of targeted game fish species (steelhead, walleye, and Chinook salmon) in the Muskegon River Watershed as a result of climate change.

5.3 Methods

5.3.1 Site Description

This study examines climate change impacts on the Muskegon River Watershed. The watershed is located in the west-central part of the Lower Peninsula of Michigan, USA. The drainage basin encompasses 7,057 km^2, of which 33.4 % is agriculture, 9.6 % urban, 22 % forested, 14 % wetlands, and 21 % covered by other land use type (Anonymous, 2004).

The Muskegon River is the second longest river in Michigan. Its mainstem is 350 km long; from the head waters to Lake Michigan, it drops about 175 m. The river originates in Higgins and Houghton lakes, and flows southwest to the Muskegon Lake, which eventually drains into Lake Michigan. The river presently has 4 major water impoundments, including Reedsburg (constructed in 1940), Rogers (constructed in 1906), Hardy (constructed in 1931), and Croton dams (constructed in 1907). Reedsburg Dam is located at the headwaters of the river, just below Houghton Lake. The other three are large hydropower dams with large impoundments, located in the middle portion of the mainstem (O'Neal, 1997).

The average annual precipitation in the Muskegon River Watershed is 90 *cm* (Pijanowski *et al.*, 2006). Peak flows tend to occur in the spring either following snowmelt or associated with large rain events. Water quality is good in most parts of the watershed (O'Neal, 1997). Presently, the Muskegon River supports 77 native fish species and an additional non-native 12 species that colonized through constructed channels or were introduced directly (O'Neal, 1997). Many groundwater fed tributaries help to sustain cold- and cool-water species (Clapp *et al.*, 1990; Brazner *et al.*, 2004, 2005; Creque *et al.*, 2005; Wehrly *et al.*, 2006).

I focused the assessment of fish early life history of adfluvial steelhead (*Oncorhynchus mykiss*), Chinook salmon (*Oncorhynchus tshawytscha*), and Great Lake walleye (*Sander vitreus*) (O'Neal, 1997) on the lower part of Muskegon River, ranging from Croton Dam to Muskegon Lake. Access for migratory fishes to the upper river is restricted by the three large hydropower dams. The length of the lower Muskegon River is approximately 70 *km* (Steen *et al.*, 2010), and provides the primary spawning areas of the species discussed here.

5.3.2 Model Linkage

To explore the impacts of future climate change on thermal regimes of the Muskegon River, RPSTM was applied to estimate both present (baseline) and future water temperatures. RPSTM was linked to a suite of interacting sets of climate, hydrology, and landuse models in MREMS (The Muskegon River Ecological Modeling System) (Seelbach & Wiley, 2005; Wiley *et al.*, 2010). Participating component models of MREMS were developed and executed independently. They communicated by exchanging inputs/outputs via a shared data directory referenced by VSEC (Valley Segment Ecological Classification) spatial framework (Anonymous, 2000; Seelbach *et al.*, 2006). Models were structured and executed in a proper hierarchy that represented how climate condition influenced local hydrologic, water quality, and ultimately biological consequences in real river ecosystems (Wiley *et al.*, 2010).

Typically, the weather and landuse models drive all other dependent models. Current (i.e., years 1996 to 2005) climate data were developed by Kendall and Hyndman at Michigan State University from NOAA Doppler Radar data and local cooperative weather gauges. The climate change scenario was based on projections of the Intergovernmental Panel on Climate Change (IPCC) A1B climate modeling in the 4th Assessment Report (Meehl *et al.*, 2007). Projected monthly anomalies of average daily air temperature and total daily precipitation (Soloman *et al.*, 2007) were used to adjust historic data from 1996 to 2005 (Wiley *et al.*, 2010). Future landcover was simulated by a Land Transformation Model (LTM) (Pijanowski *et al.*, 2000). LTM uses neural network simulation algorithms that apply the historical patterns of urban and forest growth into the future (Pijanowski *et al.*, 2006, 2007). Groundwater flux was predicted by the Integrated Landscape Hydrology Model (ILHM) (Hyndman *et al.*, 2007). In addition to weather data, ILHM required outputs from the landcover model. The simulated groundwater discharge was then accumulatively operated and transformed by HEC-HMS through its embedded programing in basin and channel routing. Later, these hydrological outputs (i.e., surface water and groundwater discharge), along with climate data were used as inputs to execute RPSTM. Water temperature predictions were then summarized into daily thermographs. At the end, predicted changes in water temperature were used to determine the associated changes in fish habitats (Figure 5.1).

The distributed multi-modeling environment of MREMS allowed me to provide a consistent and coherent picture of how the Muskegon River ecosystem might functionally respond to changing climate. Both MREMS and RPSTM were successfully verified from

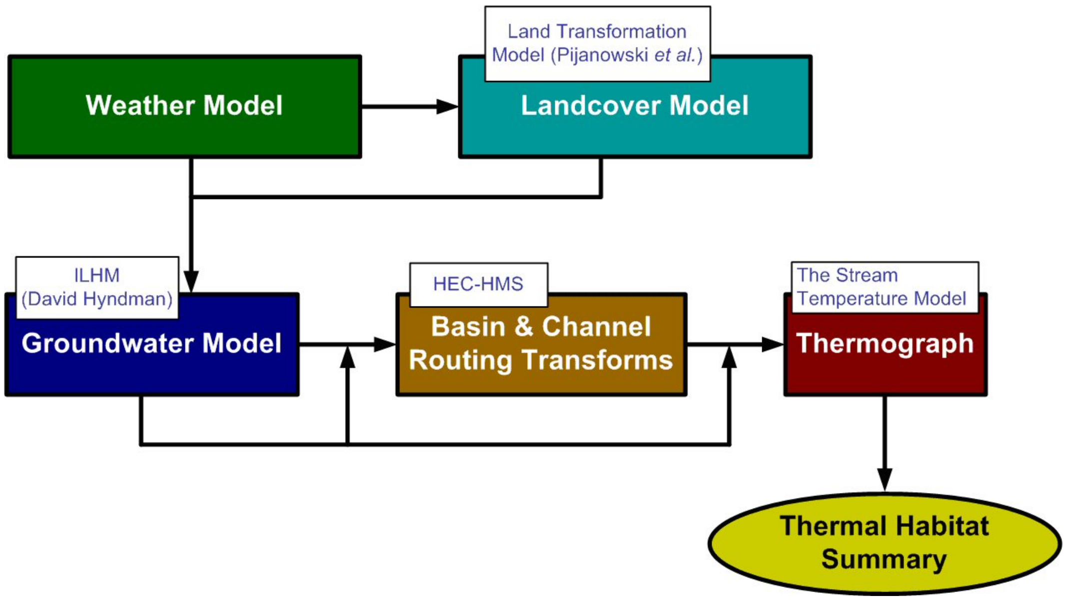


Figure 5.1: Flow diagram of how RPSTM linked to the interacting sets of models in MREMS for predicting impacts of climate change on water temperature and fish thermal habitat in the Muskegon River Watershed, Michigan, USA.

a variety of monitoring locations representing different fluvial types, including reservoirs, small reaches, and main channels (see Chapter 2) (Wiley *et al.*, 2010).

5.3.3 Simulating Effects of Climate Change

Based on IPCC (2007), monthly anomalous rise for average daily air temperature was predicted to rise between 3.5 to 4°C in North America based on the conservative A1B scenario (Soloman *et al.*, 2007). In comparison to other IPCC scenarios, A1B scenario was based on a modest reduction of greenhouse gas emissions to the end of this century. IPCC clearly stated out in the report that reliably simulating and attributing observed temperature changes at smaller scales remains difficult, due to relatively larger natural climate variability and uncertainties at local scales (Soloman *et al.*, 2007). To apply IPCC's projection without losing higher resolutions at local scales, I added IPCC's monthly predictions of air temperature rise into the current daily weather profiles. These data were then used as "future climate data" to drive the entire integrated modeling system. In this way, I preserved more realistic rain and temperature distributions across the catchment.

Table 5.1: Locations of the modeled stream temperature sites in the Muskegon River.

Site Name	Location
Site 1	Mouth of the Muskegon River to the Muskegon Lake
Site 2	North Channel of the Muskegon River
Site 3	Muskegon River at Mill Iron
Site 4	Muskegon River at Muskegon Wastewater Treatment Plant
Site 5	Muskegon River at B31 Bridge
Site 6	Muskegon River at Bridgeton Bridge
Site 7	Muskegon River at Newaygo Bridge
Site 8	Muskegon River at Downstream of Croton Dam (Below USGS Gauge)
Site 9	Muskegon River in Croton Dam Pond
Site 10	Muskegon River in Hardy Dam Pond
Site 11	Muskegon River in Rogers Dam Pond
Site 12	Muskegon River at Big Rapids (USGS Gauge)
Site 13	Muskegon River at Junction of the Hersey River
Site 14	Muskegon River at Evart (USGS Gauge)
Site 15	Muskegon River at Junction of Middle Branch
Site 16	Muskegon River at Junction of the Clam River
Site 17	Muskegon River at Down Reedsburg Dam
Site 18	Muskegon River at Reedsburg Dam
Site 19	Muskegon River at Houghton Lake
Site 20	Muskegon River at Higgins Lake

Stream temperature predictions were compiled at 20 locations in the main stem of the Muskegon River; 8 locations were from the mouth to Croton Dam, 9 locations were above Rogers Dam, and 3 locations were at the major hydropower dams (i.e., Croton, Hardy, and Rogers Dams) (Table 5.1).

Stream temperature from “current” and “future” climate scenarios were estimated in daily time steps from 1996 to 2005. Inputs accounted for in the climate change alteration include changes in daily air temperature, daily surface runoff, daily groundwater discharge, travel time, hydraulic depths, and groundwater temperature. Future groundwater temperature was assumed to increase from 12.5°C to 16°C in summer, and from 6°C to 9.6°C for other seasons. This is because groundwater temperature responds closely to ground surface temperatures (Taylor & Stefan, 2009). It typically follows regional average annual air temperature (Bundschuh, 1993; Lee & Hahn, 2006). As future air temperatures was projected to increase from 3.5 to 4°C, groundwater temperature is expected to rise at similar magnitudes (Andrushchyshyn *et al.*, 2009). Some existing studies also indicated that groundwater temperature can be from 2 to 5 °C warmer in response to global warming, depending on the regions and types of land use (Andrushchyshyn *et al.*, 2009; Taylor & Stefan, 2009).

Simulated daily water temperature results were summarized into monthly values to examine the impacts of changing climate on stream water thermal conditions.

5.3.4 Estimating Impacts of Climate Change on Steelhead, Walleye, and Chinook Salmon

Steelhead, walleyes, and Chinook salmon are three of the most popular game fish species in the Great Lakes region. Will the warming global climate reduce the thermal habitats of these fishes in this important river? If air temperature continues to rise, how will the spawning and hatching activities be impacted?

Simple timing and temperature criteria from literature reviews were applied to translate modeled temperature into shifts in the early life timing of steelhead, walleyes, and Chinook salmon. These spawning thermal criteria (see below), along with time window criteria ,were used to determine when spawning activities would begin and end in every simulated year. Fry emergence was dependent on spawning date and predicted spawning activities were then projected by accumulating temperature-units (TU; 1 TU = 1°C above freezing for 24 hours). A 10-year average of the estimated timing for spawning and fry emergence was calculated and compared to examine impacts from climate change.

For steelhead, spawning usually starts in March or April, typically when stream temperature reaches 4.4°C. The peak of the spawning activities are in mid-April, possibly continuing into May, with a daily average stream temperature ranging from 10 to 12.8°C (Biette *et al.*, 1981; Bell, 1991; Newcomb & Coon, 1997; O'Neal, 1997; Horne *et al.*, 2004; Richter & Kolmes, 2005). To include spawning seasons from the beginning to the end, I used a temperature range of 4.4 to 12.8 °C from March to May as a threshold for spawning. Fry emergence criteria for steelhead fry typically occurs after the accumulation of 230 to

340 TU (Myers & Horton, 1982; Murray & McPhail, 1988). I used a range of 230 and 340 TU for early and late emergence, respectively.

Walleyes are unable to reproduce successfully in the Muskegon river without human assistance, but they do migrate up the river to spawn every spring. Eggs and sperm are typically fertilized and raised in the hatchery every year. Then, the 5-cm-long walleyes originating in the Muskegon River are stocked back in the river in late May. To assess the impact of climate change on the timing of walleye spawning activities, I used a spawning range of from 4.4 to 8.3°C from March to May as the thermal criterion (Baker & Manz, 1971; Roseman *et al.*, 1996; DePhilip *et al.*, 2005). I used 115 TU as the threshold for emergence (Roseman *et al.*, 1996).

A variety of stream temperature ranges have been reported to trigger spawning of Chinook salmon in different regions (Murray & Rosenau, 1989; Mullan *et al.*, 1992; Myers *et al.*, 1998; Murphy & Heard, 2001), I used a daily average temperature ranges of 7.2 to 14.5°C starting in September, as the criteria for fall spawning in Michigan (Piper *et al.*, 1982; Connor *et al.*, 2002, 2003a,b). This particularly matched well to our observations of fall spawning Chinook salmon in the Muskegon River. I then used 1000 TU (Seymour, 1956) as the criterion for fry emergence.

5.4 Model Results

5.4.1 Water Temperature Regimes

My model projected increases in average water temperature of 2 to 4.5°C each month across the Muskegon River Watershed given a monthly anomalous air temperature increase of 3.5 to 4°C in the area. Increase in water temperature was predicted to occur across the entire Muskegon River Watershed with almost no exceptions throughout the continuous 10 modeled years. However, the magnitude of changes in water temperature varied seasonly and spatially.

Location played an important role in controlling the magnitude of increase in water temperature with regard to climate change. Relative sensitivity to air temperature change also varied seasonally. In general, in the mainstem, the greatest changes in stream temperature were projected to occur in spring, and occasionally the month July in warm years, followed by fall and summer, and the smallest changes occurred in winter. Average seasonal changes were from 3.2 to 4.5°C each month in spring, from 3.2 to 4.4°C each month in fall and summer, and from 2.2 to 4.3°C each month in winter. On average, January was the least affected month in the mainstem. However, in upstream locations where small streams receive large groundwater inputs, the greatest water temperature changes were predicted to occur in winter (i.e., 4.2 to 4.3°C increase each month from December to February), and the smallest change in summer (i.e., 3.7 to 4.1°C increase each month from June to August).

Moreover, elevated groundwater temperature shows more effective in warming under the climate change scenario. Consequently, a greater discrepancy occurred in winter when groundwater fed tributaries experienced greater water temperature changes than those at main channels and reservoirs. For example, during December to February, Cedar and Brooks creeks (both have ample groundwater inflows) encountered about 4.3°C anomalous rise in these month; while during the same period, Muskegon Lake increased over 2.3 to 3.4°C, and Hardy Dam Pond increased over 3.1 to 3.9°C. The greater temperature increases in winter in the tributaries might provide over-wintering shelter for fishes. Spacial variation in the impact on climate change in water temperature was less in summer. For example, Hardy Dam Pond showed higher increment (4°C) than Brooks Creek (3.8°C) in July, while the Muskegon Lake was 3.4°C similar (the lowest average increment).

Change in daily average water temperature was greater than monthly average change, and varied from 1 to 5°C with occasional events below 1°C or above 6°C. However, seasonal variance was more pronounced. When averaging daily changes by month, April through July was the period of greatest change. The reach downstream immediately of Croton Dam, had the highest change across the entire watered: $>=$ 7°C daily change was found in April 29, 2000, and the averaged 10-year monthly increases of 4.5°C were found in April and May. In June and July, the daily anomalous rise was above 3.7°C. Conversely, December through February were the periods with the least temperature change, ranging from 2.1 to 3.6°C downstream of Croton Dam. In the main channel, spring months and July were still the two major periods that water temperature was impacted by climate change the most.

5.4.2 Changes in the Timing of Spawning and Fry Emergence

1. Steelhead:

   Predictions of the impact of climate change on the timing of steelhead spawning and hatching activity were quite different at different locations. In general, below Croton Dam appeared to be the place most affected by climate change. Based on the 10-year averaged simulations, steelhead were predicted to start their spawning activities on March 25, and continue spawning until May 24 under current climate. In the climate change scenario, steelhead spawned from March 4 to May 2, about 3 weeks earlier. A warming climate would also result in an earlier fry emergence. Under current climate, fry were predicted to emerge on average from April 29 to June 15 (yearly variation ranges from April 12 to June 22). With climate change, fry were predicted to emerge on average from April 6 to May 24, ranging from as early as March 26 to as late as June 6. Overall, if air temperature warms up 3.5 to 4°C, fry emergence will be ahead about three weeks, similar to the impact on spawning activities (Figure 5.2).

2. Chinook salmon:

   In the Muskegon River, Chinook salmon begin moving into the river in August and remain through December (Piper *et al.*, 1982; Connor *et al.*, 2002, 2003a,b). Peak spawning occurs in mid Oct, but can shift to late October, even early November, if the fall is warm. According to the model prediction for current climate conditions,

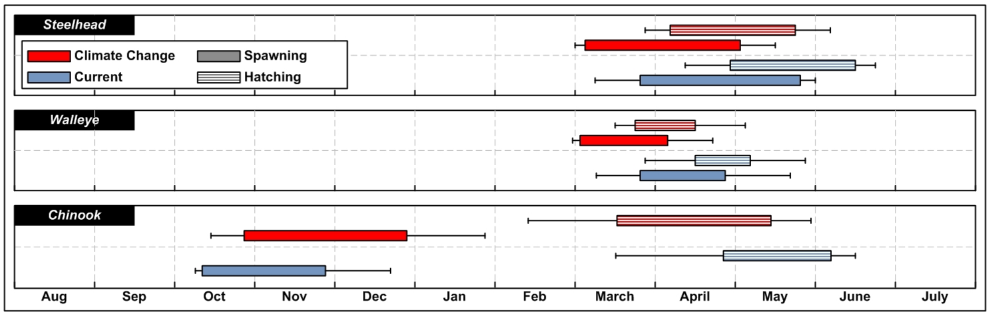


Figure 5.2: Projected 10 year averaged shifts of spawning and hatching activities for steelhead, walleye, and Chinook salmon at site 8, with error bars indicate yearly variations.

Chinook salmon spawn below Croton Dam on average from October 11 to November 27, with variation ranging from October 8 to December 22. Given projected climate, my model predicted Chinook salmon will delay their spawning from late October (October 27) through late December (December 28), but yearly variation existed and ranged from October 14 to January 27. On average, warming climate caused the early spawning cohort to be delayed about 2 weeks, and the late one about one month.

Using the predicted spawning time as a start to accumulate the necessary energy for fry emergence, model results suggest that global warming will induce an earlier emergence– on average by one month. Under current climate settings, the model predicted Chinook salmon fry emerge from April 26 to June 6. While with global warming climate, Chinook salmon will emerge on average from March 16 to May 14 (Figure 5.2).

3. Walleye:

   Walleye, a native species in Michigan, is another popular sport fish species in the Muskegon River. Below Croton Dam, given the current climate, my model predicts that on average spawning should extend from March 25 to April 26. Under a warming climate, spawning activities would on average extend from March 2 to April 5, about three weeks earlier. For current climate, fry would emerge from April 15 to May 7, versus March 22 to April 16 for warming climate. However, large yearly variations in dates exist in the model runs driven by annual climate variation. For example, across the 10 year period, the early run fry emergence occurred 11 to 32 days earlier in response to the climate change scenario, and late season run fry emergence was 9 to 53 days earlier (Figure 5.2).

5.4.3 Impacts of Climate Change on Fish Species at Different Locations

There are several distinct but distinct spawning locations for river fishes in the lower Muskegon River (O'Neal, 1997). Modeling suggests that the impacts of climate change vary spatially across locations.

For steelhead, the predicted warmer water temperature in spring shifted spawning 25 days earlier than present immediately below Croton Dam, and also downstream in the reach from Newaygo to Bridgeton. Similar to the impacts on steelhead, climate change was estimated to shift walleye spawning activity at Bridgeton 24 days earlier, whereas immediately below Croton Dam and at Newaygo, it was only 22 days earlier. Conversely, the predicted warmer water temperature in fall resulted in a delayed spawning activity for Chinook salmon. Minor differences occurred between spawning locations: immediately below Croton Dam spawning was estimated to be delayed 16 days, Bridgeton 15 days, and Newaygo 14 days (Figure 5.3).

For fry emergence, impacts of climate change were found to affect the location at Bridgeton the most for all three species. Averaged modeling results suggest that steelhead and walleye fry will emerge about 25 days earlier at Bridgeton, 23 days below Croton Dam, and downstream in the reach at Newaygo about 20 days ahead. Chinook salmon fry emerged from 41 to 42 days earlier below Croton Dam and also downstream in the reach from Newaygo to Bridgeton (Figure 5.4).

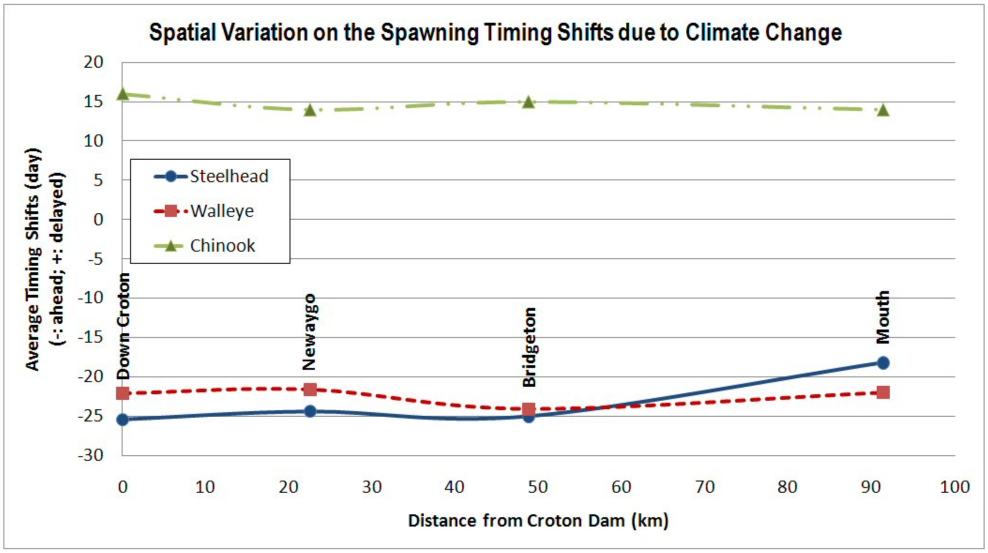


Figure 5.3: The impact of climate change on the timing shifts of spawning activities varied spatially at down Croton, Newaygo, and Bridgeton. Timing shifts were calculated as the differences of early season spawning activities from current and climate change scenarios, and averaged from 10-year daily simulations.

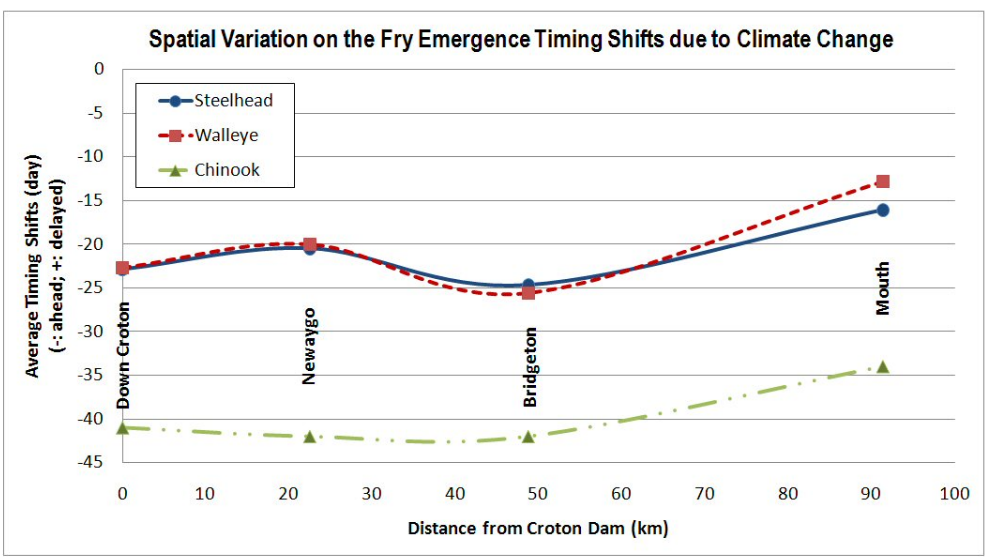


Figure 5.4: The impact of climate change on the timing shifts of fry emergence varied spatially at down Croton, Newaygo, and Bridgeton. Timing shifts were calculated as the differences of early season hatching activities from current and climate change scenarios, and averaged from 10-year daily simulations.

5.5 Discussion and Conclusion

Prediction results indicated that the maximum changes in monthly water temperature will occur in spring, followed by fall and summer, and the minimum in winter at locations in the mainstem and reservoirs. This projection is consistent with Mohseni et al. (1999) who used a regression model on a weekly time scale in 993 stream gaging stations throughout the contiguous United States (Mohseni *et al.*, 1999). They found that maximum change in weekly stream temperatures should occur in spring (March to June), and the minimum changes in winter (December and January) and summer (July and August) (Mohseni *et al.*, 1999). However, I found that in smaller tributaries, seasonal trends of water temperature change are very dependent on groundwater inputs. Projections of water temperature change in the tributaries are very different than those in the mainstem. In general, locations with ample groundwater-fed tributaries experienced the greatest water temperature change in winter, and the smallest change in summer. This was due to temperature buffering effects by significant groundwater inflow. In winter, as air temperatures were elevated by climate change projections, higher air temperatures, more rapid snow melt, and more rain events, all of which contributed to a higher groundwater inflow rate in ILHM predictions.

In addition, I found that projected water temperature followed the cold- and warm-year oscillations, but the impacts of climate change were not significant by categorizing into cold- and warm- years. Annual average air temperature was projected to increase 3.5°C, which caused 40.8% and 42.1% increments to the annual average air temperature to the current weather in years 2000 and 2002. This, in turn, drove annual average water temperatures to increase 33.0% and 33.6% in those respective years. The 1.3% difference in annual average air temperature only raised annual average water temperature by 0.6%. As warmer years typically have higher daily air temperature, and less daily total flow, the net heat transported into stream water will be compensated by heat losses in evaporation and transpiration.

Changes in water temperature in spring resulted on average in a predicted three-week shift in spawning and fry emergence of steelhead and walleyes. Meanwhile, elevated stream water temperature also shortened incubation time for spring spawners. In contrast, projected elevated water temperature in fall led to an average of a one to two month delay in spawning activities of Chinook salmon, with a shorter development time and earlier fry emergence. However, this study considered no other factors but only water temperature in estimating the early life stage timing shifts. Other than water temperature, tributary hydrology, flow stability, and day length also determine spawning and nursery habitats for fish (Creque *et al.*, 2005; Godby *et al.*, 2007; Ivan *et al.*, 2010). Without thoroughly consideration of these important factors, results could show a broader range of the timing in spawning and fry emergence. This was found in this study as Chinook salmon migration to the Muskegon River currently occur from late September, which is about one to two weeks earlier than my prediction. Also, Chinook salmon fry currently emerge from late March to late May, about one month earlier than the predictions from my model.

Moreover, physical habitat conditions, such as flow velocity and substrates are important to fish in choosing their spawning habitats. These factors are critical in determining absence and presence of fish in an area, which could potentially lead to a more complex inter- and intra-interactions. In this study, I did not consider these complex interactions among species. For example, shorter incubation time could potentially decrease the risk of eggs being eaten by animals, and might thus decrease mortality. However, if many species are so affected, it could result in a general increase in complex and interesting interspecies competition. Increasing competition interactions may alter expectations based on climate change alone. As Steen et al. (2010) predicted, climate change will cause a decline of coldwater fish, and significant increases of cool- and warm-water species in the Muskegon River Watershed (Steen *et al.*, 2010). As a result, a more detailed study of the causes and potential consequences of life history timing shifts will be required.

In regions other than Michigan, global climatic changes have also been predicted to be a great threat to many riverine fishes (Casselman, 2002; Field & Francis, 2002). However, the impacts of climate change could vary with river ecosystems and species. They are dependent upon the changes of rate and magnitude relative to historical and present hydrothermal regimes for each watershed, and the adaptive ability and capacity of species (Palmer *et al.*, 2009). The relations between a warming climate and the changes in life-history found in this study are different than those found in other geographical areas. For example, in Pacific salmon, projected warmer sea-surface temperature can influence migratory timing (Jonsson & Jonsson, 2009). Hodgeson et al. (2006) reported that sockeye salmon *Oncorhynchus nerka* returned early to the rivers in south-west Alaska, but late to the rivers in British Columbia, Canada due to a relatively warmer sea-surface water temperature (Hodgeson *et al.*, 2006). In addition, climatic variability in the North Pacific, the Pacific decadal oscillation (PDO), also greatly impacts salmon for both the spawning success of adults and for growth and survival of juveniles during their freshwater residency (Hilborn *et al.*, 2003).

It is important to note that my projections focused only on the influence of water temperature, and did not account for impacts from other factors, nor account for longterm genetic changes that might occur in response to climate change. For example, spawning sites are not rigidly controlled by temperature alone but are the product of a dynamic biological process influenced by physical habitat conditions such as flow and substrate. These limitations are important to acknowledge because the environmental changes can also lead to population-level changes, such as the life-history, phenology, and morphological traits of many species (Ozgul *et al.*, 2010). These changes could occur as concurrent responses to both environmental change and changing selection pressures or natural selection on heritable traits or drift (Ozgul *et al.*, 2010).

An important challenge to modeling the impact of climate change on stream water temperature lays in the great uncertainty about estimates of groundwater temperature. Even though there has been tremendous progress in hydrothermal research, there are major gaps in our basic understanding of the influence of climate change on groundwater temperature.

The few existing model predictions are quite variable (Andrushchyshyn *et al.*, 2009; Taylor & Stefan, 2009; Vicca *et al.*, 2009), and in some cases controversial. Groundwater temperature varies with soil depth, geological setting, and landcover type (Andrushchyshyn *et al.*, 2009; Taylor & Stefan, 2009; Vicca *et al.*, 2009). Taylor and Stefan (2009) indicated that both urbanization and climate warming will contribute to a warmer groundwater temperature (Taylor & Stefan, 2009). When transforming a land from "undeveloped" to "fully urbanized" area, along with a doubled carbon dioxide emission climate scenario, groundwater temperature is likely to raise by about 5°C (Taylor & Stefan, 2009). Vicca et al. (2009) predicted a +2, +4, and +6°C increase in the shallow groundwater temperature in response to climate change in their ciliate communities study (Vicca *et al.*, 2009). Indeed, the uncertainties in the groundwater system in response to climate change makes the prediction of stream water temperature very challenging. To better understand the effects of this uncertainty on my model results, I conducted a sensitivity analysis by manipulating the magnitude of change in groundwater temperature. Rerunning my analysis I found that a 5°C increase in groundwater temperature produced an additional from 1.2 to 2.2°C increases in predicted water temperature (Figures 5.5 to 5.8). Increases were dependent upon travel distance and time, and the amount of groundwater inflow. This is coincident with my conclusions that water temperature within a free-flowing river typically increases gradually in a downstream direction (Vannote et al. 1980), but can vary at the microscale as a result of groundwater inflow (Geist *et al.*, 2002). It is not difficult to imagine that in different regions, these discrepancies will be even larger. Therefore, it seems that studies of potential climate change impacts on soil and water temperature are critical to understanding climate impacts on the thermal regimes of rivers. Such studies are needed to provide a more through basis for modeling impacts of climate change on groundwater-influenced riverine and wetland ecosystems.

In conclusion, linking RPSTM to MREMS multi-modeling provided a useful approach to forecasting how climate change may alter riverine thermal habitats and cause shifts in the

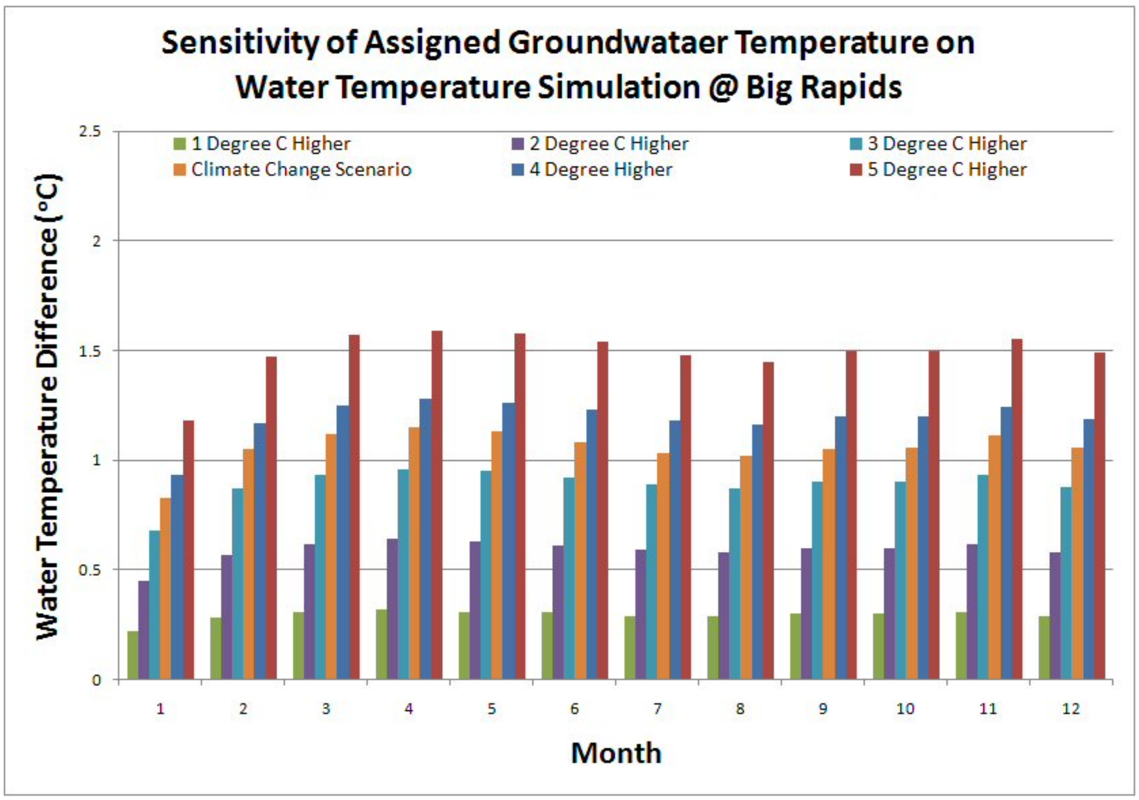


Figure 5.5: The sensitivities of different magnitude of elevated groundwater temperature at Big Rapids (a big channel) to the projected water temperature in response to climate change.

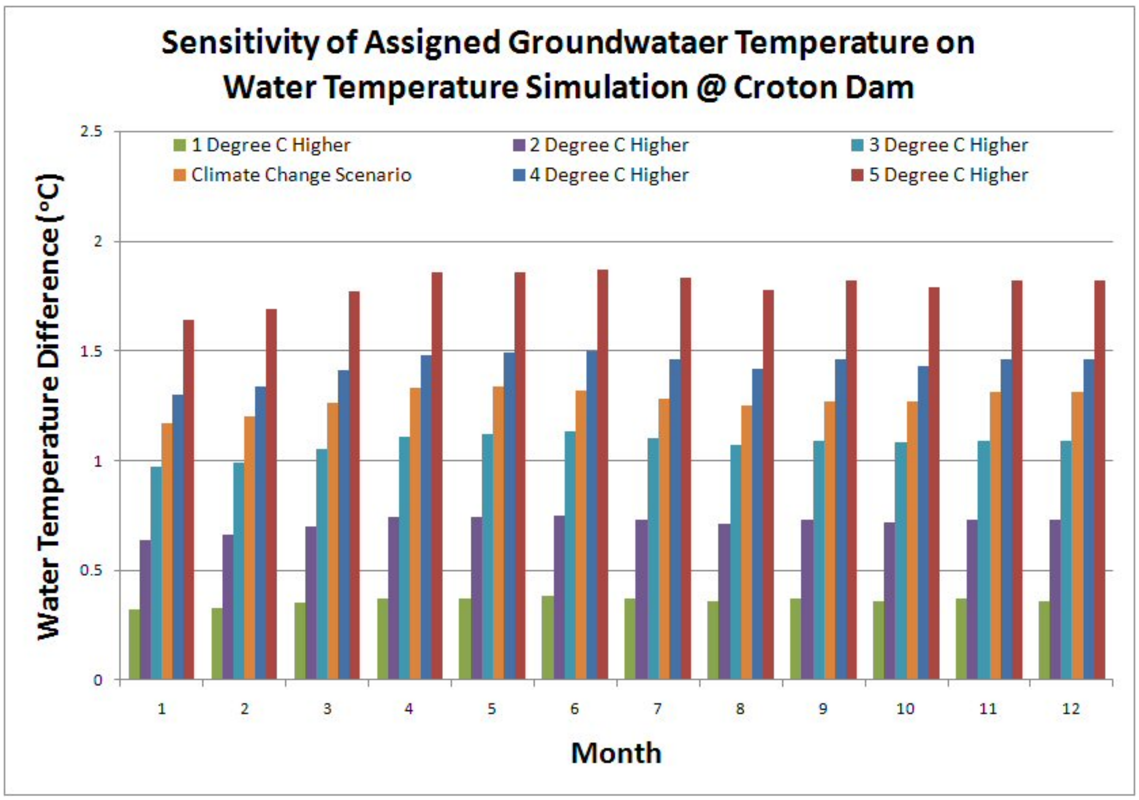


Figure 5.6: The sensitivities of different magnitude of elevated groundwater temperature at Croton Dam (a reservoir) to the projected water temperature in response to climate change.

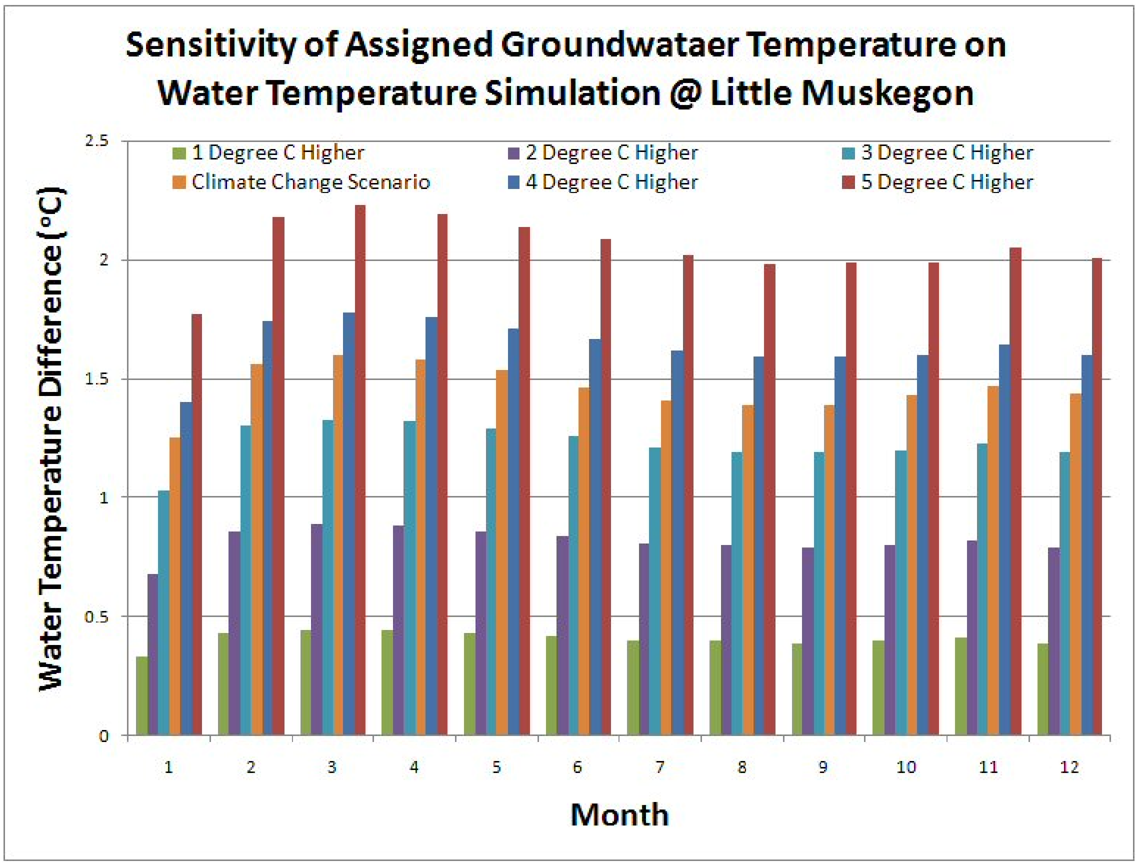


Figure 5.7: The sensitivities of different magnitude of elevated groundwater temperature at Little Muskegon (a major tributary to the Muskegon River) to the projected water temperature in response to climate change.

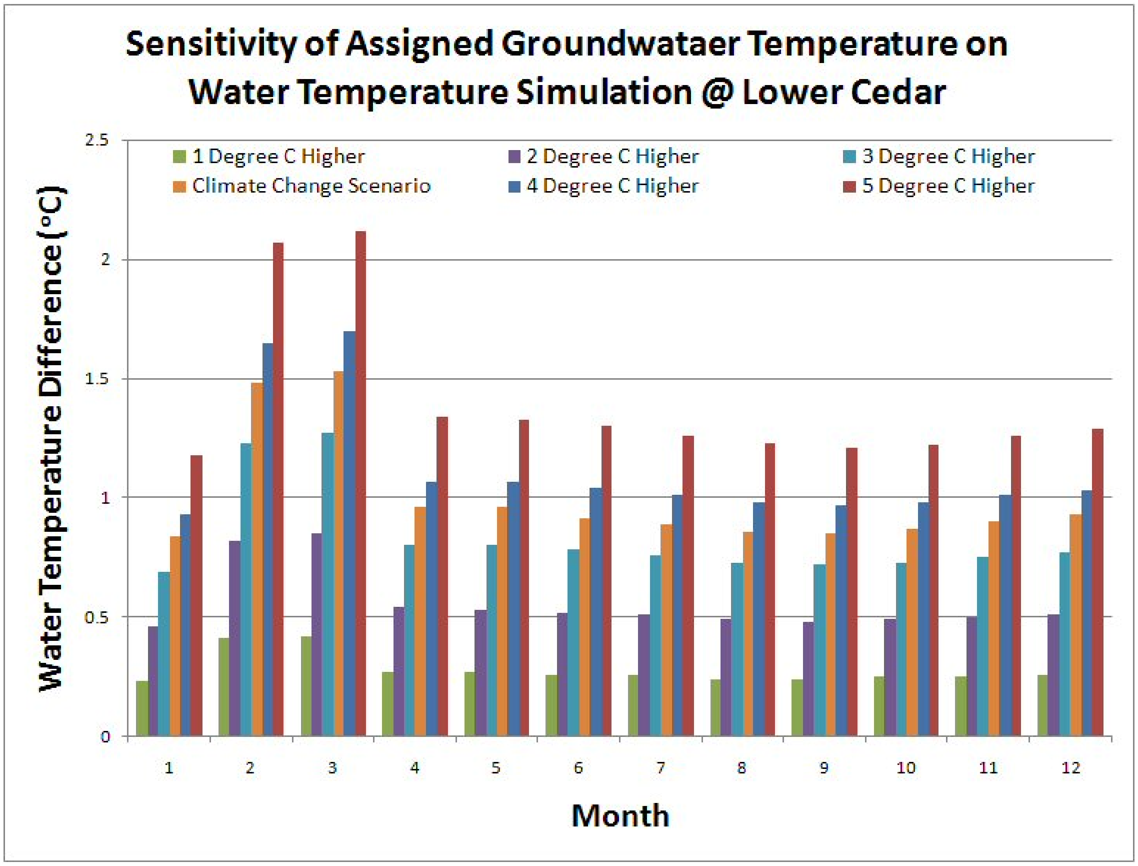


Figure 5.8: The sensitivities of different magnitude of elevated groundwater temperature at Lower Cedar (a small groundwater fed tributary) to the projected water temperature in response to climate change.

timing of Great Lakes anadromous fishes early life history. A predicted 2 to 4.5°C increase in monthly water temperature will induce earlier spawning and fry emergence activities for steelhead and walleyes, and delay spawning and accelerate fry emergence for Chinook salmon. These predictions should be useful to fishery managers interested in planning for adaptation strategies to cope with climate change.

CHAPTER VI

CONCLUSION AND FUTURE WORK

6.1 Summary of Chapter Findings

In Chapter 2, I described a newly designed Reduced Parameter Stream Temperature Model (RPSTM). RPSTM offers significant implementation and computational advantages over existing heat balance models when simulating stream temperatures across large watersheds. Moreover, the model is processed-based, increasing the confidence in the output of dynamic simulations of past, current, and future scenarios. The results showed that RPSTM performed well in estimating stream temperature along the Muskegon river network. The sensitivity analysis revealed that RPSTM is most sensitive to the parameterization of local air temperature, hydraulic depth, and solar radiation, but relatively insensitive to rate of surface flow. RPSTM is moderately sensitive to rates of groundwater discharge, network travel time, and the parameterization of the heat transfer coefficient. This modeling approach is easily integrated into complex multi-modeling systems that are now being used to evaluate effects of long term changes in land use, climate change, and river management on river fish communities.

In Chapter 3, I used RPSTM to evaluate the thermal impacts of water withdrawal. The results demonstrated that (1) both surface water and groundwater removals have potentially serious impacts on river thermal regimes, and (2) that there is an inherently large spatial variability in thermal responses to unit flow removal. The magnitude, and sometimes direction of thermal impact, appears to be location specific shaped by patterns of local groundwater flux, mixing ratios, and travel times. Wells or water extractions closer to the river are likely to have larger impacts on thermal regime. Yet, large groundwater loading downstream could ameliorate local groundwater losses. More importantly, it might alleviate the heating effects from consumption of upstream local groundwater and/or surface water flow. Therefore, sensitivity to water extraction is likely to be spatially variable and site specific, and modeling heat transport implications will likely be necessary to evaluate ecosystem sensitivity.

In Chapter 4, I examined the impacts of dam removal and found that dam removal could effectively lower summer stream temperature and allow more space and time in the mainstream of the Muskegon River for coldwater species. Yet, the magnitude of the lowered summer stream temperature from dam removals varied temporally and spatially, depending on the distance to the dams, the hydrology, and the year-to-year climate conditions. Under the circumstances of "dams removed", the simulated results have revealed that dam removal would cause delays (8 to 10 days) in the timing of spawning for steelhead and walleyes, but have no significant impact on Chinook salmon. At the same time, timing of fry emergence would be delayed for all three modeled fish species, although length of delay varied between species (7 to 15 days). Analysis results also supported the contention that removal of the three hydropower dams would in general bring a lot more usable habitat upstream. As the total area of the Muskegon River was large, quantifying suitable habitat via the GIS technology for steelhead, walleye, and chinook provides useful information for fishery managers to plan their stocking strategies.

Lastly in Chapter 5, I applied RPSTM along with the predictions from the 4th IPCC A1B Scenario to drive a multi-modeling system, and evaluated the impacts of climate change on the early life history of Great Lakes anadromous fishes. Modeling suggested that the Muskegon River may experience a 2 to 4.5°C warming in monthly water temperature. The greatest increases in modeled water temperature occurred in spring, especially during warmer-than-average years. Moreover, groundwater fed tributaries experienced greater water temperature change than large channels/reservoirs in winter. However, this spacial distinction was reduced during the summer. These changes in water temperature are likely to significantly alter the timing of steelhead, walleyes, and Chinook salmon early life history. This integrated multi-modeling approach could be useful to fishery managers interested in planning management adaptations to cope with climate change.

6.2 Importance of Hydrologic Modeling to Water Temperature

Many factors affect local stream water temperature. These include basin geography, global and local climate, geomorphic setting, and details of hydraulic routing (see Chapter 2). River width, depth, surface area (size), length, and velocity are also important factors. For example, wider streams heat more rapidly than narrower streams due to the increased surface-to-volume ratio, which results in greater exposure to solar radiation (Farber *et al.*, 1998). Likewise, water in long, slow velocity channels also has greater exposure to the surrounding atmosphere and more time to equilibrate with warmer air temperatures (Adam & Sullivan, 1990). These two facts alone require that inside a particular river system water temperature will continuously vary spatially from headwaters to terminus.

The physical, chemical, and biological systems of a river are closely and intensely linked together, making the predictions for a river ecosystem very complex. The temperature of rivers draining a landscape is driven by interactions of energy inputs from diverse sources. Linking to a multi-modeling systems helps to more accurately capture the influence of dynamic changes from landscape, climate, and hydrology.

Among all these potentially influential factors, the magnitude of groundwater discharge is a major controlling factor for stream temperature in the upper Midwestern U.S. (LeBlanc *et al.*, 1997; Wehrly *et al.*, 2006). Both distribution and timing of groundwater inputs are important in shaping the river thermograph downstream. High groundwater yields typically result in cooler than air stream temperatures in summer and warmer than air temperatures in winter (Bundschuh, 1993; Sinokrot *et al.*, 1995; Constantz *et al.*, 1994; Constantz, 1998). Therefore, the magnitude of groundwater delivery to river channels modulates downstream heat gain, and creates thermal environments that are relatively cold and stable which favor coldwater fish species (Wehrly *et al.*, 2003).

As such, the hydrological modeling plays a much more important role in the prediction of stream water temperatures. In this study, RPSTM required information of stream network, discharge rate, travel time, and hydraulic depth from the outputs of the hydrologic modelings– ILHM (the Integrated Landscape hydrology Model Groundwater; (Hyndman *et al.*, 2007)) and HEC-HMS (the Hydrologic Engineering Center's Hydrologic Modeling System; (Anonymous, 2001)). These two models process watershed discharge coming from precipitation to land surface and water body, take evapotranspiration into account, infiltrate to soils and groundwater aquifer, and then combine with water body as discharge of surface runoff and groundwater. The models describe what happens as water that has not infiltrated or been stored on the watershed moves over or just beneath the watershed surface (the surface runoff part), and simulates the slow subsurface drainage of water from the system into the channels (the groundwater part). They also support various aspects of hydrologic estimation, such as: peakflow, flow duration curves, runoff, and baseflow, for any kind of land development scenario.

In addition, the results from the water withdrawal application clearly demonstrated that changing the volume of streamflow and/or groundwater also altered thermal regimes. It provides clear evidence of how important it is to know the magnitude of hydrologic loadings to the prediction of water temperature. Without hydrologic models, a greater uncertainty might emerge from areas with large underground aquifers. Surface runoff information is usually easier to obtain either by online USGS gauge data, or by discharge measurements in a stream. Groundwater information, on the other hand, is more difficult to acquire, since the magnitude, character, and extent of groundwater contributions are not as easy to access as those in river flow. As a result, a groundwater model is very important in rivers with groundwater inputs.

However, long-term water temperature records may provide information on both the location and the magnitude of the groundwater source. This is because temperature of groundwater is relatively colder in summer, and warmer in winter than the air temperature. Even without knowing how much or what temperature groundwater is, inputs of groundwater into a reach will make the reach's thermograph different than a reach with no groundwater source within the same stream network. Therefore, if the influences from other factors are known (by running RPSTM), the actual water temperature measurements can reflect the influence of groundwater. From this point of view, RPSTM has the potential to identify groundwater inputs when longterm water temperature records are available.

6.3 Reduce Parameterization of a Model

As Albert Einstein said: "It can scarcely be denied that the supreme goal of all theory is to make the irreducible basic elements as simple and as few as possible without having to surrender the adequate representation of a single datum of experience." This statement clearly expressed the intent of my research– to develop a simple stream temperature model, which can generally be applied to any river system. Compared to most physically-based heat balance models, RPSTM does require much less parameterization and makes it possible to predict stream water temperature across a large river network. Another purpose of this study is to evaluate the effects of human-induced thermal changes on fish distributions and their early life history, and results have shown that RPSTM indeed provided needed accuracy for this purpose.

RPSTM might be reduced too much in terms of using it in the impoundments. It provides less accurate predictions in large impoundments, simply because RPSTM was designed in a more abstract way mainly for river reaches, and did not consider depth dependent stratification and other complex processes a large lake typically has. However, it is always a necessary trade-off that a simple model is more generalizably applicable, while a complex model is usually more specifically applicable. To solve this problem, a more complex version of RPSTM will be needed to consider the important mechanisms for large impoundments which were ignored in the current version of RPSTM.

6.4 Future Work

1. Dam Removal Plus Climate Change Scenario:

   As stated in Chapter 4, removing dams would potentially decrease the current stream temperatures by about 1°C. It is an interesting question if dam removal could help mitigate the negative impact of global climate change on the coldwater fish community. As a result, a scenario comprised "dam removal plus climate change" may be able to answer this question.

2. Error Propagation in Multi-Modeling Approach:

   It is not clear in this study how errors propagated in this multi-modeling system. It cannot be denied that through series of model runs, it is very difficult to determine whether errors (including specification error, parameterization error, or computational error) (Wiley *et al.*, 2010) cancel each other out or accumulate. Quantifying errors from each sub-model will be a necessary step towards analyzing a system's error propagation. This might help identify uncertainty about the magnitude and the direction in response to future forecasting using this multi-modeling approach.

APPENDICES

APPENDIX A

HYDROLOGICAL MODELING (HEC-HMS) SETUP FOR THE MUSKEGON RIVER, CEDAR CREEK, AND MILL CREEK

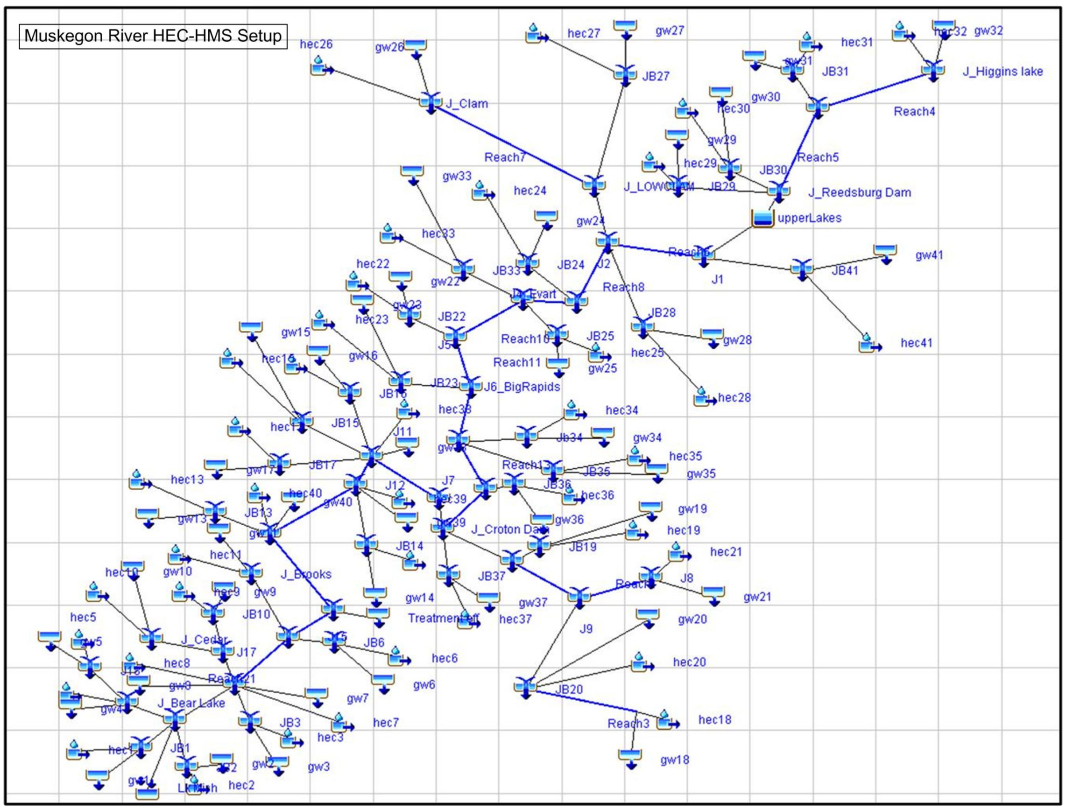


Figure A.1: Channel schematic for the Muskegon River Watershed, Michigan.

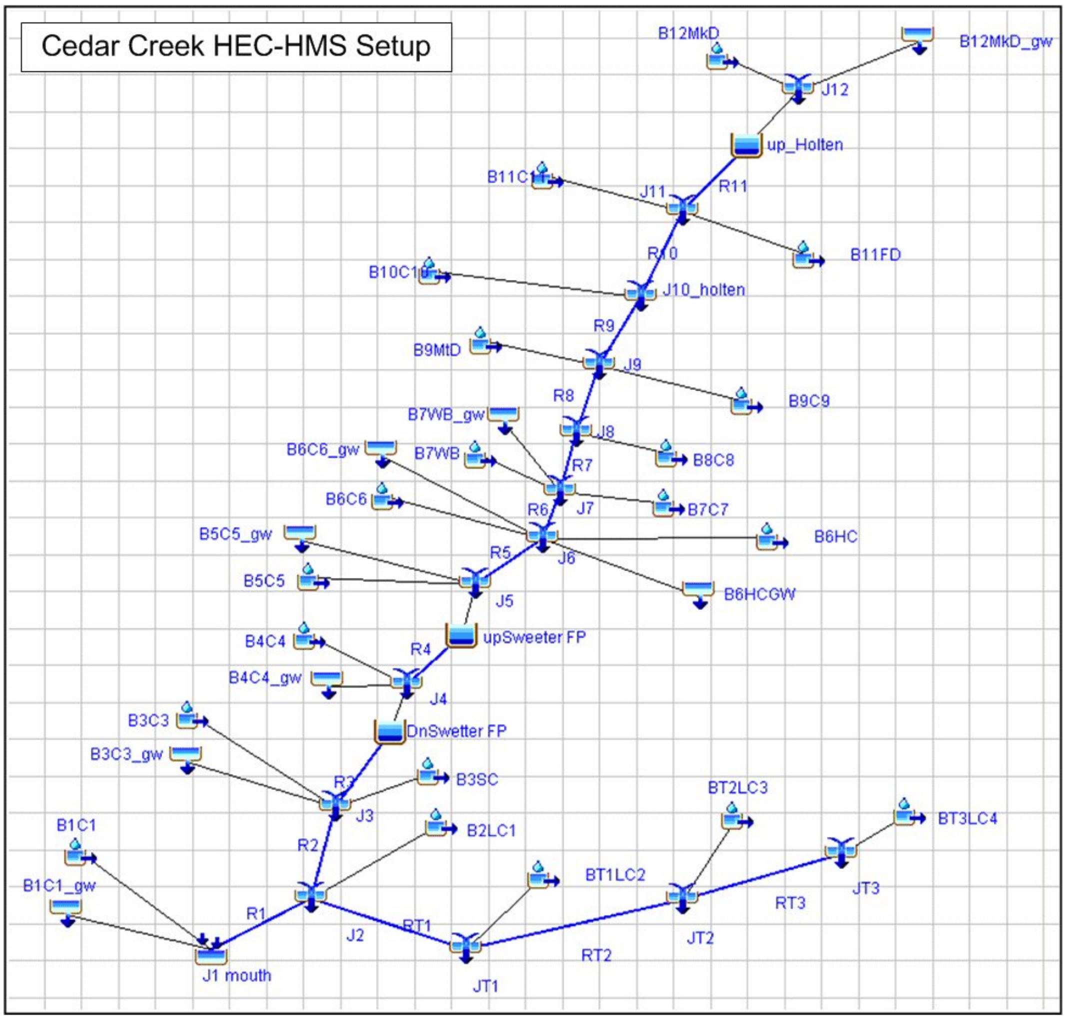


Figure A.2: Channel schematic for Cedar Creek, Michigan.

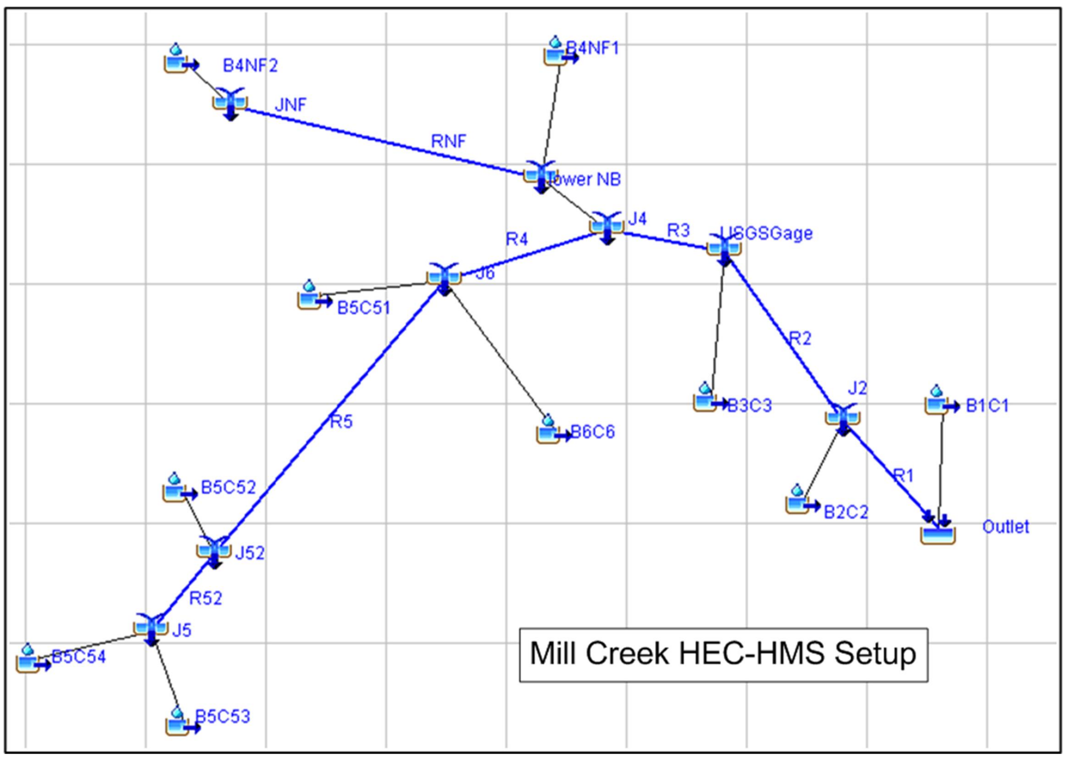


Figure A.3: Channel schematic for Mill Creek, Michigan.

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