# AN EXAMINATION OF RELATIONSHIPS BETWEEN CLIMATE AND FISH COMMUNITIES IN AGRICULTURAL HEADWATER STREAMS

by

**Darren J. Shoemaker**

**A Thesis**

*Submitted to the Faculty of Purdue University*

*In Partial Fulfillment of the Requirements for the degree of*

**Master of Science**

Department of Biology at Purdue Fort Wayne

Fort Wayne, Indiana

May 2021

**THE PURDUE UNIVERSITY GRADUATE SCHOOL**
**STATEMENT OF COMMITTEE APPROVAL**


**Dr. Robert Gillespie**
Department of Biology

**Dr. Peter Smiley Jr.**
USDA Agricultural Research Service, Soil Drainage Research Unit

**Dr. Jordan Marshall**
Department of Biology


**Approved by:**
Dr. Jordan M. Marshall

## ACKNOWLEDGMENTS

There are many people without whom this project would not have been possible. I first need to thank my primary advisory, Dr. Robert Gillespie, for his constant support and guidance during my time as his student. I also need to thank my committee members, Dr. Peter Smiley Jr. Dr. Jordan Marshall for their assistance, particularly with statistical analyses. I also need to mention fellow graduate student Tyler Shuman, for his mentorship from my time as a field tech through a graduate student. I need to thank the following organizations: 1) ARS-NSERL in West Lafayette, Indiana, through the USDA's CEAP program; and 2) the Purdue Fort Wayne Biology department for support and funding. I offer thanks to the many graduate and undergraduate students who assisted with collecting data in the field. Special thanks go to Kathy Gray for her unrelenting dedication to field work, who despite enduring torrential rains, 90+ degree heat, hostile swarms of mosquitos, and a conspicuous plot of poison ivy, kept showing up for work the next day.

# TABLE OF CONTENTS

# LIST OF TABLES

# LIST OF FIGURES

# ABSTRACT

Fish communities in agricultural headwater streams are known to be impacted by a variety of factors, including water chemistry, habitat modification, and hydrology. Little research has been conducted on how climate change influences these communities, yet the effects of climate on lake and river fish have been well documented. I hypothesized that fish community metrics would be reduced by the effects of climate change. I examined the effects of climate and hydrology metrics on fish communities at nine sites in the Saint Joseph River, Indiana and Michigan and at 18 sites in the Upper Big Walnut Creek, Ohio watersheds, from 2006 to 2019. Air temperature, water temperature, precipitation, water discharge, width, velocity, and depth metrics were calculated seasonally for each sampling year. Fish were examined seasonally with backpack electrofishing and seine netting and identified to species level. Principal component analyses were used to create axes which represented gradients of climate and hydrology metrics. Linear mixed effect and logistic regression modeling suggested that hydrology is a stronger predictor than climate, but that both influence fish communities. Percent Percidae, percent herbivore, and percent open substrate spawner were positively correlated with precipitation and water temperature. Presence herbivore was negatively correlated with precipitation and positively correlated with water temperature. My data only somewhat supported the hypothesis that climate would reduce fish community metrics. Gradients of hydrology were observed to be stronger predictors than gradients of climate. However, one must acknowledge relationships between climate and hydrology and the potential for climate to have indirect effects on fish communities through influences on hydrology. This study increases understanding of how fish communities in agriculturally dominated headwater streams are influenced, and emphasizes the need for further research on how these fishes will be impacted by a changing climate.

# CHAPTER 1. INTRODUCTION

Headwater streams are often heavily modified or completely constructed for agricultural benefit without regard for any habitat or organisms which live within them (Blann et al. 2009; McCall and Knox 1979; Scarnecchia 1988). Headwater streams are first through third-order streams which flow into or combine to form larger orders (Frissel et al. 1986; Peckham and Gupta 1999; Strahler 1957). Intense modification of riparian habitat, geomorphology, and water chemistry has had a negative influence on fish and macroinvertebrate communities within their waters (Freeman et al. 2007; Harrel et al. 1967). Examining the relationships between disturbance, modification, geologic structure, water chemistry, riparian habitat, instream habitat, other anthropogenic influences and organismal communities is critical for understanding headwater streams ecosystems (Colvin et al. 2019). Headwater streams are important both for their inherent value and for their influence on larger downstream rivers (Herzon and Helenius 2008). Existing research has been valuable for conservationists, land managers, regulators, and the agricultural community (Effert-Fanta et al. 2019; Tóth et al. 2019). In addition to agricultural modification, headwater streams are also susceptible to disturbance from climate change (Durance and Ormerod 2007; Wu et al. 2013). Headwater streams in Indiana, Michigan, and Ohio have been the subject of many previous studies (Jordan et al. 2016; King et al. 2009; Sanders et al. 2020; Shuman et al. 2020; Smiley et al. 2011; Wood et al. 2020). However, little is understood about how continued climate change will impact communities within headwater streams. I will attempt to identify relationships between climate and fish communities in headwater streams.

Headwater streams are often studied due to both their ecological importance and influence on larger downstream ecosystems. Modification and channelization of headwater streams have had negative influences on community structure (Karr et al. 1985; Scarnecchia 1988; Trautman 1939). Modification includes dredging to remove substrate, mowing and herbicide application to reduce riparian habitat, and channelization to remove natural sinuosity (Brookes 1987; Sullivan et al. 2004). These modifications intend to maximize drainage of excess water from agricultural land and do not consider ecological impact or habitat alteration (Sanders et al. 2020). The United States Geological Survey estimated in 2016 as much as 85 percent of headwater streams in the United States have been channelized or otherwise modified (Carlisle et al. 2017). Agricultural activity also has direct effects on headwater streams, including increased nutrient and pesticide loading. Previous studies suggest excess nutrient loading is the leading cause of reduced community diversity in 30 percent of all United States streams (Meador 2013). Studies have also reported pesticide residues detected in 90 percent of water samples collected from United States agricultural streams (Gilliom 2007). Chemical pollutants threaten both aquatic life and human health (Belden et al. 2007; Schwarzenbach et al. 2010).

Physical modification, effects from agricultural activity, and nutrient and pesticide loading render headwater streams more susceptible to the effects of climate change. Climate change influences biotic, chemical, and physical conditions in aquatic environments (Knouft and Ficklin 2017). Some estimates suggest that at a mean increase of two degrees Celsius, 37 percent of global land areas to experience an increase in extreme precipitation and streamflow, while 43 percent will experience a decrease in precipitation, leading to higher levels of drought (Krakauer et al. 2019). These studies also suggest these effects are nearly doubled with a mean global increase of four degrees Celsius (Asadieh and Krakauer 2017). Increased global air and water temperatures will cause dramatic and unpredictable alterations in streamflow conditions throughout headwater streams worldwide (Asadieh and Krakauer 2017). General circulation models suggest an average 13 percent increase in mean precipitation for every one degree Celsius increase in global air temperature (Lambert et al. 2008). Salmonid and trout species have been studied in greater detail than most fish species due to their high economic value. Studies of these taxa suggest that increased global air and water temperatures will negatively impact individual survival, growth, and life history (Ruesch et al. 2012; Wenger et al. 2011). Unfortunately, the detail of these studies has not been expanded to include the less economically valuable, but no less ecologically important, taxa which inhabit headwater streams. Despite numerous studies conducted on headwater streams, comparatively little research has been conducted on how climate change influences those communities.

Few issues within the scientific community receive as much attention as the issue of climate change. The idea of climate change is one of the most thoroughly supported scientific premises of modern times (U.S. EPA 2008; Wuebbles et al. 2017). Climate change influences conditions of natural environments at global, regional, and local geographic scales. Effects of climate change range from reduction of sea ice in the arctic, increasing average water temperatures in aquatic habitats, acidification of ocean waters, climactic alterations of weather patterns, increased prevalence of severe weather events, and altered migratory habits of certain animal species (USGCRP 2018). At the global scale, reports have shown average global air temperatures have increased by 0.5 to 1 degree Celsius in the last century (NOAA National Centers for Environmental Information 2020). Climate change more directly affects both human communities and natural ecosystems at the regional level. In Indiana: 1) since the 1920s, the average annual air temperature in Indiana has increased by 0.667 degrees Celsius; 2) the frost season has decreased by nine days; 3) average annual precipitation has increased by six to nine percent; 4) traditionally dry periods have become shorter (Höök et al. 2018; Wuebbles et al. 2017). Other qualitative effects have also been reported: 1) periods of snowfall have become shorter and periods of rainfall have become longer; 2) precipitation extremes, such as heavy rains, have become more frequent and variable; 3) short-duration droughts have become more common; 4) flooding has become less predictable (Mishra and Cherkauer 2010). Effects on biotic communities have also been documented: 1) forest compositions are altered as rising air temperatures push certain tree species north; 2) ranges and distribution of fish species are altered; 3) suitable habitat for fish species dependent on cold water is reduced; 4) increased water temperatures allow certain invasive aquatic species to move north, increasing competition and reducing resource availability for native species (Eaton and Scheller 1996; Pryor et al. 2014). The effects of climate change in Indiana and the Midwest have been well documented.

Aquatic ecosystems are uniquely vulnerable to impacts caused by climate change. Geographic and structural conditions in Midwest environments restrict the movements of many aquatic species. Flat topography, anthropogenic barriers, and lack of adequate aquatic habitats contribute to the sensitivity of organisms inhabiting these ecosystems to changes to water temperature and conditions (Hall 2012). Many animals attempt to migrate to colder climates in response to warmer air and water temperatures (Beever et al. 2017). This strategy is ineffective for most aquatic organisms. Systems of streams, rivers, and lakes create natural barriers to most aquatic species. Smaller streams are often unnavigable for larger lake fish, and smaller stream fish are vulnerable to predation when crossing large lakes. Additionally, artificial barriers such as dams and development further prevent relocation (Marschall et al. 2011). These obstacles leave freshwater organisms vulnerable to the effects of climate change (Shuter et al. 2012).

At the local level, climate change has been evaluated for impacts in lake and river ecosystems. Little effort thus far has been made to understand the impacts of climate change in headwater streams, with research generally grouping these environments with rivers and other larger waterways. The influences of climate change in Indiana are well-documented, as described in Höök (2018). Additionally, gaps in the literature appear when examining the influence of climate in headwater streams. Headwater streams have ecologic and economic importance because of their strong influence on larger downstream waters. Many reports, such as Höök, et al. (2018) only reference rivers and lakes, but exclude headwater streams from their analyses. Headwater streams provide water sources for all major rivers and lakes in the United States, and have strong influences on larger bodies of water (Alexander et al. 2007; Fritz et al. 2006). The importance of headwater streams is well understood but there is a critical lack of research on how climate change affects these ecosystems. One may extrapolate from studies on other freshwater ecosystems, but there is a need for focused study on headwater streams.

Fish are particularly sensitive to changes in water temperature (Pauly 1980). Body temperature, metabolism, feeding behavior, reproduction, and growth of fishes are affected by the temperature of their environment. A study conducted in the Great Lakes Basin determined climate-induced increases in water temperature would significantly disrupt food web dynamics (Hill and Magnuson 1990a). Bioenergetics models used to estimate prey consumption and growth indicate that increases in water temperature would alter prey consumption and growth for several predatory fish species, including largemouth bass, *Micropterus salmoides* (Rice and Cochran 1984), lake trout, *Salvelinus namaycush* (Stewart et al. 1983). Additionally, these bioenergetic models suggest changes in water temperature would also alter interactions between these species (Kitchell and Breck 1980; Lyons and Magnuson 1986). Annual growth and predation generally increased with warmer water temperatures. Fish typically experience periods of growth beginning in the spring and extending through the warm summer months, and experience reduced growth during colder winter months (King et al. 1999). This life history strategy suggests a need to conserve energy during colder seasons. Warmer climates extend the length of this growing season, which previous research suggests causes fish to consume more prey, disrupting food webs and consuming resources faster than the ecosystem can replenish (Ficke et al. 2007). Fish physiology leaves them sensitive to warmer water, and climate change is increasing average global water temperatures. By examining fish communities in headwater streams, we can better understand how they will be affected by a warming climate.

Headwater streams have direct influence on conditions of larger water bodies downstream in the watershed. As primary water sources for rivers, headwater streams transport nutrients, chemicals, sediments, solutes, and organisms into rivers and lakes. Additionally, these streams affect hydrological processes including storage, flow, and residence of surface water (Alexander et al. 2007). The relatively small volume of headwater streams exacerbates these interactions and effects: less sediment, chemical, and nutrient input is required for significant impacts on stream conditions; smaller environments are often less resilient to disturbance; less energy is required to cause changes in water temperature; and, organisms may be more likely to interact and have fewer opportunities for avoidance (Wohl 2017). Physical modifications such as mining, plowing, draining, channelizing, groundwater withdrawal, and riparian removal disproportionately impact headwater streams more than other aquatic ecosystems, including (Petersen Jr et al. 1987). These factors emphasize the vulnerability of headwater streams and the importance of understanding both their own ecology and their influence on larger aquatic habitats.

At the watershed basin scale, land use may also explain some variation among fish community metrics between sites and years. Agricultural activity such as row crop agriculture, application of pesticides, herbicides, and fertilizer, and vegetation removal impacts headwater streams (Moore and Palmer 2005; Young and Huryn 1999). Land use is sometimes correlated with riparian disturbance (Dempsey et al. 2017). Riparian disturbance caused by agricultural activity is the greatest source of stream habitat degradation in the United States (U.S. Environmental Protection Agency 2016). Despite evidence supporting negative relationships between land use and riparian habitat, research remains inconclusive about the extent to which land use impacts fish community diversity, but land use is generally considered to be a poor predictor (Meador and Goldstein 2003; Tóth et al. 2019). The size and health of riparian buffer zones does have some effect on agricultural stream communities, though these effects are not necessarily consistent (Fischer et al. 2010). Fish abundance is sometimes higher at sites with shorter riparian buffers due to high numbers of chemically tolerant species, though at sites with wider riparian buffers, biotic integrity and habitat quality is often higher (Effert-Fanta et al. 2019).

The dynamic interactions among biological, chemical, and geologic factors create difficulty when examining relationships between individual fish community metrics, climate, and land use, as these factors likely affect one another in nature. Intersections between these variables makes isolating the effects of a single variable challenging (Dey and Mishra 2017). Connections between the structure and composition of headwater streams cannot be understated, and so understanding the extent to which climate change is influencing these streams is critical for management and regulatory efforts to preserve and restore watersheds in Indiana. These impacts may be exacerbated in the smaller, more isolated headwater streams flowing across the

Midwest agricultural landscape due to the sensitivity of smaller headwater streams. I will attempt to identify relationships between climate and fish communities within headwater streams. I hypothesize that climate change will significantly reduce fish community metrics in agricultural headwater streams in the Saint Joseph River and Upper Big Walnut Creek watersheds.

# CHAPTER 2. METHODS

## 2.1 Study Areas

Researchers with the Conservation Effects Assessment Project (CEAP) have been collecting data on headwater streams for nearly two decades. These data include analyses of habitat types, fish and macroinvertebrate species, water quality assessments, sediment composition, riparian, erosion, and geomorphology surveys, and bioassays. Northeastern Indiana and Central Ohio are two CEAP research areas from which these data are collected (Metz and Rewa 2019). These data are a robust source of information on fish communities in headwater streams. Fish community metrics were calculated from CEAP data collected from 2006 to 2016 and continued sampling efforts provided data for 2017-2019. Eight sites were sampled along Cedar Creek in the Saint Joseph River (SJR) watershed (Figure 1). A ninth site in Hillsdale Michigan is included in the Cedar Creek analysis. The Michigan site is on the eastern segment of the SJR and is used as a reference site for some CEAP analyses. Eighteen sites were sampled in Upper Big Walnut Creek (UBWC) watershed (Figure 2). Cedar Creek is the largest tributary of SJR. SJR flows into the Lake Erie Basin, while UBWC is a part of the Ohio River Basin.

Fish were sampled from the SJR and UBWC watersheds. Sampling efforts were made seasonally from 2006 to 2019. Nine sites were sampled in the SJR watershed and 18 sites were sampled in the UBWC watershed (Figures 1, 2). Seasons were defined for this analysis as spring (March, April, May), summer (June, July, August), and fall (September, October, November). Fewer than one percent of fish sampling efforts were completed out of season. Sampling delays which did occur were caused by heavy rainfall which restricted access to the field site.

## 2.2 Fish Communities and Habitat Variables

Fish communities in each watershed were surveyed using electrofishing and seining with consistent protocols throughout all sampling efforts (Smiley et al. 2009). Electrofishing was conducted along 125-meter sites, working upstream. Fishing crews consisted of one electrofisher and one to three netters, depending on the width and flow rate of the site. Fish were stunned at 150 V, 60 Hz, DC current and collected by netters. Crews sampled all habitats within each site to ensure as thorough and complete catch as possible. Additional sampling was conducted using seining at five points distributed throughout sites. Pools and deep water were sampled using seine hauls. Riffles and runs were sampled using kick seining. Fish were identified, measured, counted, and released. Fish which could not be identified onsite were measured, photographed, and either documented before being released, or euthanized in MS 222, fixed in formalin, preserved in ethanol for laboratory identification.

Measurements of instantaneous water temperature, midpoint wet width, water depth, and water velocity were collected concurrently with fish sampling. Water temperatures were measured using either a Hydrolab or YSI multiparameter meter. Wet widths were obtained using a tape measure, water depths were obtained using a meter stick, and water velocities were obtained using an electromagnetic flow meter. Four measurements of water depth and velocity were recorded at six transects per site, 25 meters apart. An additional transect with 10 equidistant measurements of water depth and velocity were taken at the midpoint of each site for the calculation of instantaneous discharge. At sites where an instantaneous discharge measurement was unavailable, discharge was calculated from wet width, depth, and velocity measurements. Instantaneous discharge calculated from six instream habitat transects was significantly and often strongly correlated with instantaneous discharge calculated from a single designated transect (Smiley Jr. 2020).

## 2.3 Statistical Analyses

### 2.3.1 Hydrology and Climate Predictor Variables

Four climate (Table 3) and four hydrology (Table 4) predictor variables were used for this study. All data except for water temperature were calculated by seasonal mean, as water temperature was only recorded once per season. Climate data include mean water temperature, mean precipitation, daily minimum air temperature, and daily maximum air temperature. Hydrology data include mean velocity, mean discharge, mean depth, and mean wet width. Air temperature and precipitation data were sourced from the NOAA National Centers for Environmental Information climate data (2020). Daily precipitation (cm) and minimum/maximum air temperatures (°C) were obtained from Auburn 0.8 NE, IN US (station #: GHCND:US1INDK0005, lat/long 41.370451, -85.045718), Hillsdale, Michigan NOAA station (station #: USC00203823, lat/long 41.9352, -84.6411), and the Westerville, Ohio NOAA station (USC00338951, lat/long 40.1268, -82.9441) for the time period of 1 January 2006 to 31 December 2019.

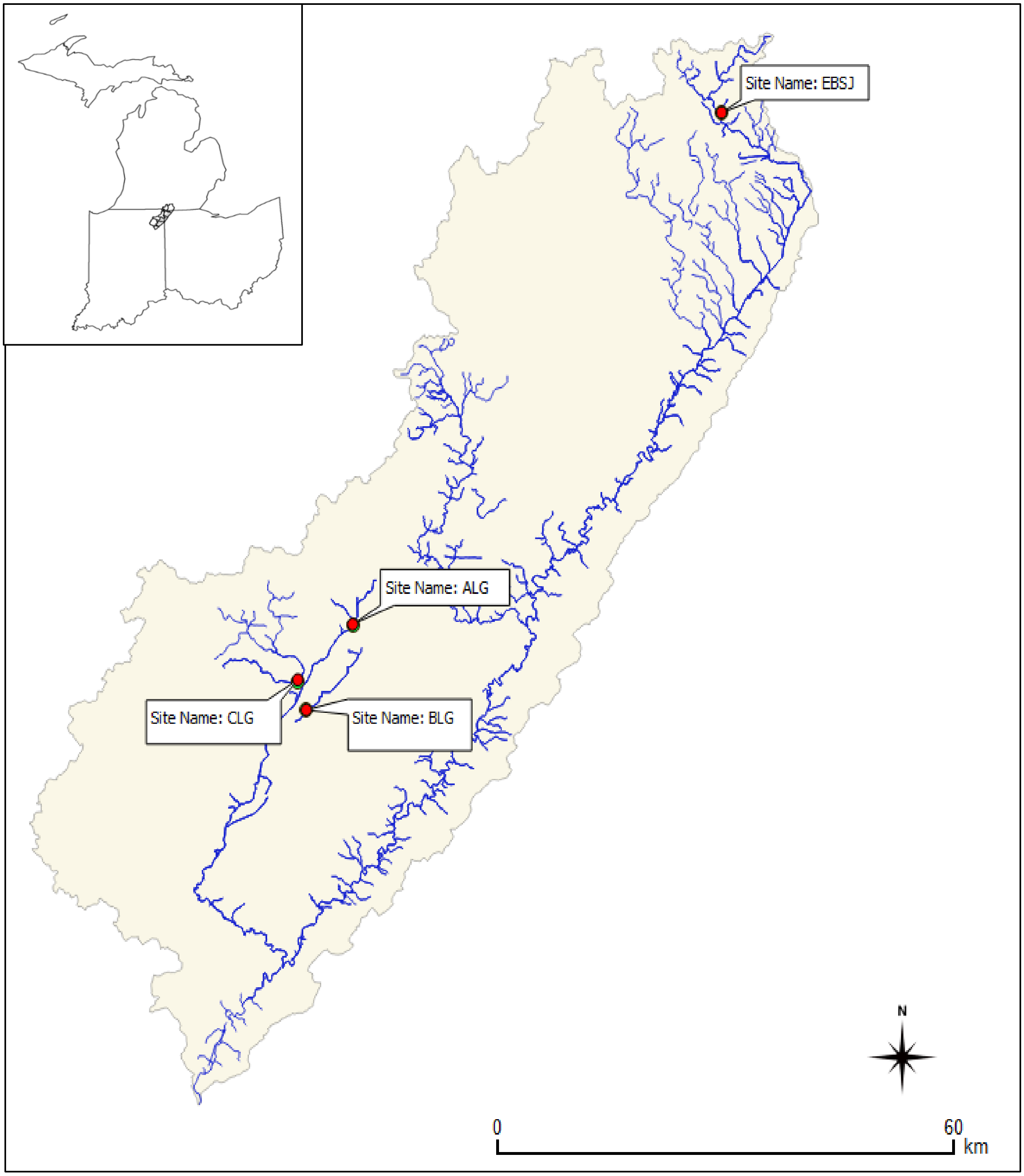

Figure 1. Map of Saint Joseph River Watershed. A stream, B stream, and C stream are in the Cedar Creek subwatershed. Map courtesy of Tyler Shuman.

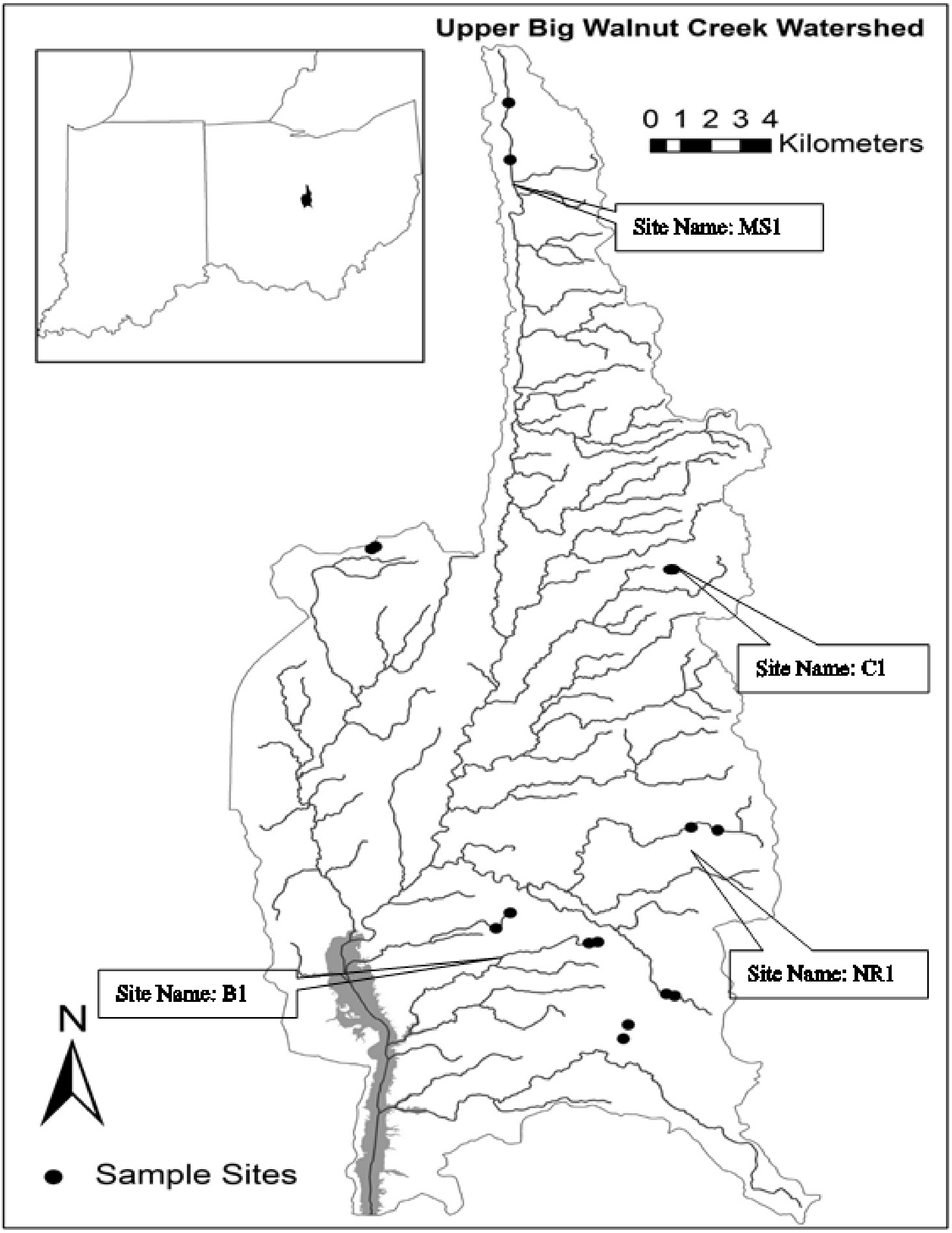


Figure 2. Map of Upper Big Walnut Creek Watershed. Map courtesy of Peter Smiley.

Pearson correlation tests ($\alpha = 0.05$) were conducted using the *cor.mtest* function from the stats package the corrplot package (Wei and Simko 2017) in R between independent variables to test for multicollinearity. A principal component analysis (PCA) based on a correlation matrix was conducted using the *prcomp* function from the stats package (R Core Team 2020) in R on each category to reduce the number of independent variables and address multicollinearity. The PCA axes served as independent variables in statistical tests and reduced the number of predictor variables in each category. Two hydrology PCA axes and two climate PCA axes were retained (Tables 4 and 5). PCA analyses were conducted with RStudio (RStudio Team 2020) in R 4.0.3 (R Core Team 2020).

### 2.3.2 Fish Response Variables

Twenty-two fish community metrics (Table 1) were calculated for each season for SJR. Diversity metrics (Shannon Diversity Index (H), species richness, abundance, sum of squares, evenness), percent thermal regime (warmwater, coolwater, coldwater), percent tolerance (tolerant, intermediate, intolerant), percent of two families (Cyprinidae, Percidae), percent feeding guild (invertivore, carnivore, herbivore, planktivore, detritivore), percent reproductive guild (nest-spawner, substrate-chooser, brood-hider, open-substrate-spawner), and percent headwater species (Pflieger et al. 1997). Thermal regimes were defined as species which are best adapted to or prefer certain water temperatures during the summer season - warmwater (greater than 25°C), coolwater (between 19 and 25°C), and coldwater (less than 19°C) (Becker 1983; Eakins 2020; Froese and Pauly 2008; Wisconsin Sea Grant 2020). Only 21 fish assemblage metrics were calculated for UBWC, since no coldwater or planktivorous species occurred in that watershed.

Response variables were tested for normality using a Shapiro-Wilk ($w \geq 0.9$) test using the *shapiro.test* function and transformed if variables did not meet a normal distribution. Arcsine square root transformations were applied to percent data and log transformations were applied otherwise. Pearson correlation tests with a significance level of 0.05 were ran between pairs of response variables to address redundancy. Pearson correlation tests were ran using the *cor.mtest* and *cor.test* functions from the stats package in R (R Core Team 2020). Variables which were significantly correlated (r coefficients > 0.60 or < 0.60 and p value < 0.05) were removed. Variables of particular ecological interest, i.e. species richness and Shannon Diversity, were retained in addition to variables which were not significantly correlated. Metrics which occurred at less than 20 percent of sites were also retained as presence-absence binary variables. A total of 18 response variables (Table 2) were retained. Normality and correlation analyses were conducted with RStudio (RStudio Team 2020) in R 4.0.3 (R Core Team 2020).

### 2.3.3 Mixed Effects Modeling

Linear mixed effects model analyses were completed to identify relative influence of climate variables on community metrics. Explanatory linear models are a common technique used to represent a simplified version of the environment to identify which independent variables have significant impacts on response variables (Heize, Wallisch, & Dunkler, 2017). The inclusion of random effects in modeling is necessary due to repeated surveys at same site during different seasons and years (Harrison et al. 2018). Site, year, and season were included as random effects. Each response variable was analyzed with three models: site as a random effect, site crossed with year as random effects, and site crossed with year, with season nested in year as random effects, and the best random effect was selected based on the Akaike Information Criterion (AIC).

Linear mixed effects model analyses were used to determine the strength of each of the four PCA axes at predicting each of the retained fish response metrics. Linear models were constructed for each response variable with the two climate and two hydrology PCA axes as fixed effects. Singularity of each model was evaluated using the *isSingular* function in R. The best random effect model for each response variable was chosen using AIC. A Shapiro-Wilk test ($w \geq 0.9$) was conducted using the *shapiro.test* in the stats package function in R to determine normality of residuals for each model (R Core Team 2020). Models for response variables with non-normal residuals were ran again with transformed data. A square root transformation was applied to abundance and arcsine square root transformations were applied to percent warmwater, percent coldwater, percent tolerant, percent intolerant, percent herbivore, percent planktivore, percent detritivore, percent substrate chooser, and percent open substrate spawner. Models which still had non-normal residuals after transformation are noted. Linear mixed effects model analyses were conducted using the *lmer* function from the lme4 package in R (Bates et al. 2015). A significance level of 0.05 was used for determining the significance of fixed effects.

Response variables which occurred at less than 20 percent of sites were converted to presence-absence data due to their low percentages. Three response variables, percent coldwater, percent intolerant, and percent herbivore were also evaluated as presence coldwater, presence intolerant, and presence herbivore. Binary logistic regression analysis was conducted for these variables. These models were using the same three random effect structures as the original models and the best models were chosen using AIC. Binary logistic regression models were constructed using the *glmer* function from the lme4 package in R (Bates et al. 2015). All statistical analyses were performed with RStudio (RStudio Team 2020) in R 4.0.3 (R Core Team 2020)

Table 1. Fish response definitions

| Fish Response | Criteria or calculation |
|---|---|
| Abundance | Number of fishes recorded during sampling event |
| Species Richness | Number of fish species recorded during sampling event |
| Shannon Diversity | -∑(pi*ln[pi]) |
| Evenness | -∑(pi*ln[pi])/ln(richness) |
| Percent Warmwater | ([# warmwater fishes in site/N])*100 |
| Percent Coolwater | ([# coolwater fishes in site/N])*100 |
| Percent Coldwater | ([# coldwater fishes in site/N])*100 |
| Percent Tolerant | ([# tolerant fishes in site/N])*100 |
| Percent Intermediate | ([# intermediate fishes in site/N])*100 |
| Percent Intolerant | ([# intolerant fishes in site/N])*100 |
| Percent Cyprinidae | ([# cCprinidae fishes in site/N])*100 |
| Percent Percidae | ([# Percidae fishes in site/N])*100 |
| Percent Invertivore | ([# invertivorous fishes in site/N])*100 |
| Percent Carnivore | ([# carnivorous fishes in site/N])*100 |
| Percent Herbivore | ([# herbivorous fishes in site/N])*100 |
| Percent Planktivore | ([# planktivorous fishes in site/N])*100 |
| Percent Detritivore | ([# detritivorous fishes in site/N])*100 |
| Percent Nest Spawner | ([# nest spawner in site/N])*100 |
| Percent Substrate Choosers | ([# substrate choosers in site/N])*100 |
| Percent Brood Hiders | ([# brood hiders in site/N])*100 |
| Percent Open Substrate Spawner | ([# open substrate spawner in site/N])*100 |
| Percent Headwater Species | ([# headwater fishes in site/N])*100 |

Table 2. Retained fish response variables

| Fish Response | Criteria or calculation |
|---|---|
| Abundance | Number of fish recorded during sampling event |
| Species Richness | Number of fish species recorded during sampling event |
| Shannon Diversity | -∑(pi*ln[pi]) |
| Evenness | -∑(pi*ln[pi])/ln(richness) |
| Percent Warmwater | ([# warmwater fishes in site/N])*100 |
| Percent Coldwater | ([# coldwater fishes in site/N])*100 |
| Percent Tolerant | ([# tolerant fishes in site/N])*100 |
| Percent Intolerant | ([# intolerant fishes in site/N])*100 |
| Percent Percidae | ([# Percidae fishes in site/N])*100 |
| Percent Invertivore | ([# invertivorous fishes in site/N])*100 |
| Percent Herbivore | ([# herbivorous fishes in site/N])*100 |
| Percent Planktivore | ([# planktivorous fishes in site/N])*100 |
| Percent Detritivore | ([# detritivorous fishes in site/N])*100 |
| Percent Substrate Choosers | ([# substrate choosers in site/N])*100 |
| Percent Open Substrate Spawner | ([# open substrate spawner in site/N])*100 |
| Presence-Absence Coldwater | 1 = coldwater species present; 0 = coldwater species absent |
| Presence-Absence Intolerant | 1 = intolerant species present; 0 = intolerant species absent |
| Presence-Absence Herbivore | 1 = herbivorous species present; 0 = herbivorous species absent |

Table 3. Climate predictor definitions

| Climate Predictors | Criteria |
|---|---|
| Water Temperature (°C) | Instantaneous temperature of water at site |
| Maximum Air Temperature (°C) | Mean daily high temperature for sampling season |
| Minimum Air Temperature (°C) | Mean daily low temperature for sampling season |
| Precipitation | Mean daily rain/snowfall for sampling season |

Table 4. Hydrology predictor definitions

| Hydrology Predictors | Criteria |
|---|---|
| Discharge ($L^3/s$) | Output of water in cubic liters per second |
| Wet Width (cm) | Mean width of stream at each site for during each season |
| Velocity (m/s) | Mean velocity of water at each site during each season |
| Depth (cm) | Mean depth of water at each site during each season |

# CHAPTER 3. RESULTS

## 3.1 Fish Communities

A total of 162,811 fish were collected during this 14-year study (Table 5). A total of 46 species occurred in the SJR watershed and 37 species occurred in the UBWC watershed (Table 6). Abundance ranged from 0 to 1,963 individuals per sample. Mean abundance in SJR was 205 (SD ± 215), while mean abundance in UBWC was 119 (SD ± 174). Mean species richness in SJR was 9 (SD ± 5), while mean species richness in UBWC was 5 (SD ± 4). Mean Shannon Diversity in SJR was 1.33 (SD ± 0.68), while mean diversity in UBWC was 0.88 (SD ± 0.62). Mean percent coldwater species ranged from 0 to 70%, with a mean of 8% in SJR. No coldwater species occurred in UBWC. Percent intolerant species ranged from 0 to 37%, with a mean of less than 1%. Mean percent intolerant species in SJR was 2%, while mean percent intolerant species in UBWC was less than 1%. Percent Percidae ranged from 0 to 90%, with a mean of 13%. Mean percent Percidae in SJR was 13%, while mean percent Percidae in UBWC was 16%. Percent planktivore ranged from 0 to 75%, with a mean of less than 1%. No planktivorous species occurred in UBWC (Table 6).

## 3.2 Climate and Hydrology

Water temperature ranged from 1.52 to 36.21 ℃, with a mean of 17.53 ℃ (SD ± 5.23) across both watersheds (Table 7). Mean water temperature in SJR was 17.74℃ (SD ± 4.40), while mean water temperature in UBWC was 17.42℃ (SD ± 6.02). Mean maximum air temperature was 20.1℃ (SD ± 5.47) in SJR, while mean maximum air temperature was 22.2℃ (SD ± 4.98) in UBWC. Mean daily minimum air temperature in SJR was 8.3℃ (SD ± 5.36), while mean daily minimum air temperature in UBWC was 10.2℃ (SD ± 5.14). Mean daily precipitation in SJR was 3.1 cm (SD ± 1.01), while mean daily precipitation in UBWC was 0.3 cm (SD ± 0.10) (Table 7).

Discharge ranged from 0 l/s when a site was dry to 4875.84 l/s, with a mean of 139 l/s (SD ± 420.11). Mean discharge in SJR was 409 l/s (SD ± 660), while mean discharge in UBWC was 9 l/s (SD ± 17). Mean wet width in SJR was 367 cm (SD ± 273), while mean wet width in UBWC was 181 cm (SD ± 94). Velocity ranged from -0.24 when a site was dry, water levels were sufficiently low, or a negative velocity occurred, to 0.53 m/s, with a mean of 0.05 m/s (SD ± 0.08). Mean velocity in SJR was 0.13 m/s (SD ± 0.11), while mean velocity in UBWC was 0.02 m/s (SD ± .04). Mean depth in SJR was 26 cm (SD ± 12), while mean depth in UBWC was 12 cm (SD ± 8) (Table 7).

Table 5. Number and relative abundance of fish captured in Saint Joseph River watershed, Indiana and Michigan, and Upper Big Walnut Creek watershed, Ohio, from 2006 to 2019.

| Common Name | Species Name | Total Number | Percentage |
|---|---|---|---|
| Banded Darter | *Etheostoma zonale* | 4 | <0.1 |
| Black Bullhead | *Ameiurus melas* | 218 | 0.1 |
| Black Crappie | *Pomoxis nigromaculatus* | 14 | <0.1 |
| Black Stripe Topminnow | *Fundulus notatus* | 313 | 0.2 |
| Blacknose Dace | *Rhinichthys atratulus* | 9159 | 5.6 |
| Blackside Darter | *Percina maculata* | 27 | <0.1 |
| Blue Catfish | *Ictalurus furcatus* | 1 | <0.1 |
| Bluegill | *Lepomis macrochirus* | 4814 | 3.0 |
| Bluntnose Minnow | *Pimephales notatus* | 9338 | 5.7 |
| Bowfin | *Amia calva* | 1 | <0.1 |
| Brook Silverside | *Labidesthes sicculus* | 1 | <0.1 |
| Brook Stickleback | *Culaea inconstans* | 17 | <0.1 |
| Brown Bullhead | *Ameiurus nebulosus* | 5 | <0.1 |
| Brown Trout | *Salmo trutta* | 18 | <0.1 |
| Central Mudminnow | *Umbra limi* | 4571 | 2.8 |
| Central Stoneroller | *Campostoma anomalum* | 11200 | 6.9 |
| Channel Catfish | *Ictalurus punctatus* | 2 | <0.1 |
| Common Carp | *Cyprinus carpio* | 426 | 0.3 |
| Common Shiner | *Luxilus cornutus* | 1880 | 1.2 |
| Creek Chub | *Semotilus atromaculatus* | 54988 | 33.8 |
| Fantail Darter | *Etheostoma flabellare* | 1819 | 1.1 |
| Fathead Minnow | *Pimephales promelas* | 11223 | 6.9 |
| Golden Shiner | *Notemigonus crysoleucas* | 318 | 0.2 |
| Grass Pickerel | *Esox americanus vermiculatus* | 887 | 0.5 |
| Green Sunfish | *Lepomis cyanellus* | 7778 | 4.8 |
| Greenside Darter | *Etheostoma blennioides* | 445 | 0.3 |
| Hornyhead Chub | *Nocomis biguttatus* | 571 | 0.4 |

Table 5 continued.

| Common Name | Species Name | Total Number | Percentage |
|---|---|---|---|
| Johnny Darter | *Etheostoma nigrum* | 18049 | 11.1 |
| Largemouth Bass | *Micropterus salmoides* | 1249 | 0.8 |
| Logperch | *Percina caprodes* | 22 | <0.1 |
| Longear Sunfish | *Lepomis megalotis* | 2 | <0.1 |
| Madtom Tadpole | *Noturus gyrinus* | 17 | <0.1 |
| Mosquitofish | *Gambusia affinis* | 54 | <0.1 |
| Mottled Sculpin | *Cottus bairdii* | 6727 | 4.1 |
| Northern Hogsucker | *Hypentelium nigricans* | 580 | 0.4 |
| Orange Throat Darter | *Etheostoma spectabile* | 8388 | 5.2 |
| Pumpkinseed | *Lepomis gibbosus* | 234 | 0.1 |
| Quillback | *Carpiodes cyprinus* | 1 | <0.1 |
| Rainbow Darter | *Etheostoma caeruleum* | 100 | 0.1 |
| Redear Sunfish | *Lepomis microlophus* | 22 | <0.1 |
| Rockbass | *Ambloplites rupestris* | 318 | 0.2 |
| Sand Shiner | *Notropis stramineus* | 3 | <0.1 |
| Silver Shiner | *Notropis photogenis* | 1 | <0.1 |
| Silverjaw Minnow | *Notropis buccatus* | 480 | 0.3 |
| Spotfin Shiner | *Cyprinella spiloptera* | 2 | <0.1 |
| Spotted Bass | *Micropterus punctulatus* | 27 | <0.1 |
| Spotted Sucker | *Minytrema melanops* | 1 | <0.1 |
| Stonecat | *Noturus flavus* | 121 | 0.1 |
| Striped Shiner | *Luxilus chrysocephalus* | 126 | 0.1 |
| Suckermouth Minnow | *Phenacobius mirabilis* | 8 | <0.1 |
| Warmouth | *Lepomis gulosus* | 33 | <0.1 |
| White Crappie | *Pomoxis annularis* | 6 | <0.1 |
| White Sucker | *Catostomus commersonii* | 5761 | 3.5 |
| Yellow Bullhead | *Ameiurus natalis* | 437 | 0.3 |
| Yellow Perch | *Perca flavescens* | 4 | <0.1 |

Table 6. Minimum, maximum, average, and standard deviation of fish community metrics from the Saint Joseph River, Indiana and Michigan, and Upper Big Walnut Creek, Ohio watersheds, from 2006 to 2019.

| | Saint Joseph River | | | | Upper Big Walnut Creek | | | |
|---|---|---|---|---|---|---|---|---|
| | Min | Max | Mean | SD | Min | Max | Mean | SD |
| Abundance | 0 | 1381 | 205 | 215 | 0 | 1963 | 119 | 174 |
| Shannon Diversity | 0 | 2.65 | 1.33 | 1.33 | 0 | 2.29 | 0.88 | 0.62 |
| Species Richness | 0 | 22 | 9 | 5.12 | 0 | 15 | 5 | 3.54 |
| Evenness | 0 | 1 | 0.4 | 0.2 | 0 | 1 | 0.37 | 0.26 |
| % Warmwater | 0 | 100 | 18 | | 0 | 100 | 26 | 0.30 |
| % Coldwater* | 0 | 70 | 8 | | -- | -- | -- | -- |
| % Tolerant | 0 | 100 | 32 | | 0 | 100 | 26 | 0.26 |
| % Intolerant | 0 | 37 | 2 | | 0 | 5 | <1 | <0.01 |
| % Percidae | 0 | 90 | 13 | | 0 | 100 | 16 | 0.22 |
| % Invertivore | 0 | 100 | 82 | | 0 | 100 | 78 | 0.37 |
| % Herbivore | 0 | 100 | 8 | | 0 | 38 | 2 | 0.04 |
| % Planktivore* | 0 | 75 | < 1 | | -- | -- | -- | -- |
| % Detritivore | 0 | 100 | 5 | | 0 | 100 | 10 | 0.18 |
| % Substrate Choosers | 0 | 100 | 9 | | 0 | 100 | 4 | 0.13 |
| % Open Substrate Spawner | 0 | 100 | 11 | | 0 | 94 | 5 | 0.12 |

*Did not occur in Upper Big Walnut Creek Watershed

Table 7. Minimum, maximum, average, and standard deviation for hydrology and climate variables in Saint Joseph River, Indiana, and Upper Big Walnut Creek, Ohio.

| | Saint Joseph River | | | | Upper Big Walnut Creek | | | |
|---|---|---|---|---|---|---|---|---|
| Metric | Min | Max | Mean | SD | Min | Max | Mean | SD |
| Water Temperature (°C) | 3.33 | 30.54 | 17.74 | 4.40 | 1.52 | 36.21 | 17.42 | 6.02 |
| Daily Max Air Temperature (°C) | 13.5 | 29.8 | 20.1 | 5.5 | 16.7 | 30.4 | 22.2 | 5.0 |
| Daily Min Air Temperature (°C) | 0.7 | 17.0 | 8.3 | 5.4 | 4.6 | 19.0 | 10.2 | 5.1 |
| Daily Precipitation (cm) | 1.3 | 5.4 | 3.1 | 1.0 | 0.2 | 0.6 | 0.3 | 0.1 |
| Discharge (l/s) | 0.00 | 4875.84 | 409.17 | 659.64 | 0.00 | 168.23 | 8.72 | 17.03 |
| Wet Width (cm) | 0 | 1131 | 367 | 273 | 0 | 499 | 181 | 94 |
| Velocity* (m/s) | ~0 | 0.53 | 0.13 | 0.11 | ~0 | 0.23 | 0.02 | 0.04 |
| Depth (cm) | 0 | 87 | 26 | 12 | 0 | 56 | 12 | 8 |

*Limit of detection for water velocity measurements in Saint Joseph River watershed was 0.1 m/s and the limit of detection for water velocity measurements in Upper Big Walnut Creek watershed was -0.015 m/s.

### 3.3 PCA Results

Two climate and two hydrology axes were retained from each PCA analysis (Tables 8 and 9). A correlation analysis between climate and hydrology scores was conducted to test for multicollinearity using the *cor.test* function in R 4.0.3 (R Core Team 2020). No PCA axes showed multicollinearity.

### 3.4 Linear Mixed Effects Models

The best random effect for percent coldwater, percent intolerant, percent herbivore, and percent substrate chooser was site. The best random effect for abundance, evenness, percent warmwater, percent tolerant, percent Percidae, and percent invertivore was site crossed with year. The best random effect for Shannon Diversity Index, species richness, percent planktivore, percent detritivore, and percent open substrate spawner was site crossed with year, with season nested in year (Table 10). In 10 cases, AIC values from two different random effects models were within two AIC units. In these cases, both random effects are reported, but the random effect with the absolute lowest AIC was used as the final model for analysis.

Table 8. Loadings of PCA axes of climate metrics of headwater streams within the Saint Joseph River Watershed and Upper Big Walnut Creek watersheds. Bolded loadings are used in the primary interpretations for each axis.

| Climate Predictors | Climate Axis 1 | Climate Axis 2 |
|---|---|---|
| Water Temperature (℃) | 0.48 | **0.35** |
| Daily Maximum Air Temperature (℃) | 0.63 | **-0.14** |
| Daily Minimum Air Temperature (℃) | 0.61 | **-0.18** |
| Precipitation (cm) | -0.03 | **-0.91** |
| Percentage of Variance Explained by Axis | 59 | 28 |

Table 9. Loadings of PCA axes of hydrology metrics of headwater streams within the Saint Joseph River and Upper Big Walnut Creek watersheds. Bolded loadings are used in the primary interpretations for each axis.

| Hydrology Predictors | Hydrology Axis 1 | Hydrology Axis 2 |
|---|---|---|
| Discharge ($L^3/s$) | **0.51** | 0.20 |
| Wet Width (cm) | **0.50** | **-0.46** |
| Velocity (m/s) | 0.417 | **0.75** |
| Depth (cm) | **0.51** | **-0.43** |
| Percentage of Variance Explained by Axis | 72 | 13 |

PCA axes are gradients associated in either a positive or negative direction with the original independent variables. Climate axis 1 is a gradient of daily maximum and minimum air temperatures, where increasing positive site scores correspond to increases in daily maximum and minimum air temperatures. Climate axis 2 is a gradient of water temperature and precipitation where increasing positive sites scores correspond to increases in water temperature and decreases in precipitation. Conversely, decreasing negative site scores correspond to decreases in water temperature and increases in precipitation. Hydrology axis 1 is a gradient of discharge, water depth, wet width, and water velocity where increasing site scores correspond with increases in discharge, water depth, wet width, and water velocity. Conversely, decreasing site scores correspond with decreases in discharge, water depth, wet width, and water velocity. Hydrology axis 2 is a gradient of water velocity, wet width, and water depth, where increasing site scores correspond with increases in water velocity and decreases in wet width and water depth. Conversely, increasing site scores correspond with decreases in water velocity and increases in wet width and water depth.

No response variables exhibited significant effects with climate axis 1. Three response variables exhibited significant effects ($p < 0.05$) with climate axis 2 (Table 11). Percent Percidae, percent herbivore, and percent open substrate spawner exhibited significant positive relationships with climate axis 2 ($p < 0.05$). These models suggest percent Percidae, percent herbivore, and percent open substrate spawner increased with increasing water temperature and decreasing precipitation.

Eleven response variables exhibited significant effects ($p < 0.05$) with hydrology axis 1. The Shannon Diversity Index, species richness, evenness, percent warmwater, percent coldwater, percent tolerant, percent intolerant, percent invertivore, percent planktivore, percent detritivore, and percent substrate chooser exhibited significant positive effects ($p < 0.05$) with hydrology axis 1 (Table 11). These results suggest the Shannon Diversity Index, species richness, evenness, percent warmwater, percent coldwater, percent tolerant, percent intolerant, percent invertivore, percent planktivore, percent detritivore, and percent substrate chooser increased with increasing discharge, wet width, and water depth.

Eleven variables exhibited significant effects ($p < 0.05$) with hydrology axis 2. Percent coldwater, percent Percidae, and percent herbivore exhibited significant positive relationships ($p < 0.05$) with hydrology axis 2. Abundance, Shannon Diversity, species richness, evenness, percent warmwater, percent planktivore, percent detritivore, and percent substrate chooser showed significant negative relationships with hydrology axis 2 ($p < 0.05$). These results suggest percent coldwater, percent Percidae, and percent herbivore increased with increasing water velocity and decreasing wet width and water depth (Table 11). These results also suggest species richness, evenness, percent warmwater, percent planktivore, percent detritivore, and percent substrate chooser decreased with increasing water velocity and decreasing wet width and water depth.

### 3.5 Binary Logistic Regression

The best random effect for presence coldwater and presence intolerant was site (Table 10). The best random effects for presence herbivore were site and year. Presence coldwater and presence intolerant did not show significant effects with any PCA axes. Presence herbivore showed significant positive relationships with climate axis 1 and hydrology axis 1. These results suggest presence herbivore increased with increasing air temperature, wet width, and water depth.

Table 10. Best random effect for linear mixed effect model analysis of 16 fish community response variables from Saint Joseph River watershed, Indiana and Michigan and Upper Big Walnut Creek, Ohio from 2006 to 2019. + Indicates a crossed random effect and / indicates a nested model.

| Fish Community Response Variable | Random Effect(s) | AIC | Secondary Random Effects |
|---|---|---|---|
| Abundance | Site+Year | 6325.70 | -- |
| Shannon Diversity Index | Site+Year/Season | 1085.86 | Site+Year |
| Species Richness | Site+Year/Season | 4790.95 | -- |
| Evenness | Site+Year | -115.49 | -- |
| % Warmwater | Site+Year | 599.96 | Site+Year/Season |
| % Coldwater* | Site | -2454.07 | -- |
| % Tolerant | Site+Year | 463.65 | Site+Year/Season |
| % Intolerant* | Site | -3742.73 | -- |
| % Cyprinidae | Site | 214.10 | Site+Year |
| % Percidae | Site+Year | 92.89 | Site+Year/Season |
| % Invertivore | Site+Year | 991.14 | Site+Year/Season |
| % Planktivore | Site+Year/Season | -1092.8 | -- |
| % Herbivore* | Site | -3103.09 | -- |
| % Detritivore | Site+Year/Season | -261.73 | Site+Year |
| % Substrate Chooser | Site | -188.30 | Site+Year |
| Open Substrate Spawner | Site+Year/Season | -424.63 | -- |
| Presence Coldwater* | Site | 173.97 | -- |
| Presence Intolerant* | Site | 389.89 | -- |
| Presence Herbivore | Site+Year | 993.21 | -- |

*Models have nonnormal residuals

Table 11. Influence of climate and hydrology predictor variables on fish community metrics within agricultural headwaters streams in the Saint Joseph River watershed, Indiana and Michigan and Upper Big Walnut Creek watershed, Ohio from 2006 and 2019. Bolded p values are those <0.05 and having a significant effect on the fish community response variable. + indicates a positive correlation between the fish community response variable and the climate or hydrological PCA axis. – indicates a negative correlation between the fish community response variable and the climate or hydrological PCA axis.

| Fish Community Response Variable | Fixed Effects | p-value | Influence |
|---|---|---|---|
| Abundance | Climate axis 1 | 0.081 | |
| | Climate axis 2 | 0.111 | |
| | Hydrology axis 1 | 0.180 | |
| | **Hydrology axis 2** | **0.001** | **-** |
| Shannon Diversity | Climate axis 1 | 0.146 | |
| | Climate axis 2 | 0.055 | |
| | **Hydrology axis 1** | **<0.000** | + |
| | **Hydrology axis 2** | **<0.000** | **-** |
| Species Richness | Climate axis 1 | 0.300 | |
| | Climate axis 2 | 0.058 | |
| | **Hydrology axis 1** | **<0.000** | + |
| | **Hydrology axis 2** | **<0.000** | **-** |
| Evenness | Climate axis 1 | 0.226 | |
| | Climate axis 2 | 0.231 | |
| | **Hydrology axis 1** | **0.001** | + |
| | **Hydrology axis 2** | **0.043** | **-** |
| Percent Warmwater | Climate axis 1 | 0.111 | |
| | Climate axis 2 | 0.460 | |
| | **Hydrology axis 1** | **0.003** | + |
| | **Hydrology axis 2** | **0.019** | **-** |
| Percent Coldwater* | Climate axis 1 | 0.285 | |
| | Climate axis 2 | 0.558 | |
| | **Hydrology axis 1** | **<0.000** | + |
| | **Hydrology axis 2** | **0.001** | + |
| Percent Tolerant | Climate axis 1 | 0.538 | |
| | Climate axis 2 | 0.068 | |
| | **Hydrology axis 1** | **0.003** | + |
| | Hydrology axis 2 | 0.112 | |

Table 11 continued.

| Fish Community Response Variable | Fixed Effects | p-value | Influence |
|---|---|---|---|
| Percent Intolerant* | Climate axis 1 | 0.453 | |
| | Climate axis 2 | 0.809 | |
| | **Hydrology axis 1** | **<0.000** | + |
| | Hydrology axis 2 | 0.168 | |
| Percent Percidae | Climate axis 1 | 0.772 | |
| | **Climate axis 2** | **0.001** | + |
| | Hydrology axis 1 | 0.811 | |
| | **Hydrology axis 2** | **0.004** | + |
| Percent Invertivore | Climate axis 1 | 0.142 | |
| | Climate axis 2 | 0.275 | |
| | **Hydrology axis 1** | **<0.000** | + |
| | Hydrology axis 2 | 0.070 | |
| Percent Herbivore* | Climate axis 1 | 0.953 | |
| | **Climate axis 2** | **<0.000** | + |
| | Hydrology axis 1 | 0.909 | |
| | **Hydrology axis 2** | **<0.001** | + |
| Percent Planktivore | Climate axis 1 | 0.214 | |
| | Climate axis 2 | 0.617 | |
| | **Hydrology axis 1** | **0.002** | + |
| | **Hydrology axis 2** | **0.032** | - |
| Percent Detritivore | Climate axis 1 | 0.669 | |
| | Climate axis 2 | 0.533 | |
| | **Hydrology axis 1** | **0.004** | + |
| | **Hydrology axis 2** | **0.001** | - |
| Percent Substrate Choosers | Climate axis 1 | 0.162 | |
| | Climate axis 2 | 0.278 | |
| | **Hydrology axis 1** | **<0.000** | + |
| | **Hydrology axis 2** | **0.011** | - |
| Percent Open Substrate Spawner | Climate axis 1 | 0.663 | |
| | **Climate axis 2** | **0.018** | + |
| | Hydrology axis 1 | 0.159 | |
| | Hydrology axis 2 | 0.587 | |

Table 11 continued.

| Fish Community Response Variable | Fixed Effects | p-value | Influence |
|---|---|---|---|
| | Climate axis 1 | 0.473 | |
| Presence Coldwater* | Climate axis 2 | 0.269 | |
| | Hydrology axis 1 | 0.600 | |
| | Hydrology axis 2 | 0.913 | |
| | Climate axis 1 | 0.900 | |
| Presence Intolerant* | Climate axis 2 | 0.214 | |
| | Hydrology axis 1 | 0.096 | |
| | Hydrology axis 2 | 0.739 | |
| | **Climate axis 1** | **0.008** | + |
| Presence Herbivore | Climate axis 2 | 0.955 | |
| | **Hydrology axis 1** | **0.003** | + |
| | Hydrology axis 2 | 0.095 | |

*Models have nonnormal residuals

# CHAPTER 4. DISCUSSION

The SJR and UBWC watersheds are especially important due to their influence on the Great Lakes and Ohio River. Previous studies conducted in the SJR and UBWC watersheds have examined relationships between instream habitat, benthic sediment, water chemistry, pesticides, and riparian habitat on fish, crayfish, and aquatic macroinvertebrate communities (Sanders et al. 2020; Shuman et al. 2020; Smiley et al. 2011; Wood et al. 2020); however, no study thus far has examined relationships between climate and fish communities in these watersheds. As climate change increases in severity, understanding how climate interacts with these ecosystems becomes increasingly important. I hypothesized that increased water temperature, air temperature, and precipitation due to climate change would reduce fish community metrics in the SJR and UBWC watersheds.

The results of this study provide little support for the original hypothesis. Of the 18 fish community response variables analyzed in this study, climate showed significant effects on only four. Percent herbivore, percent Percidae, and percent open substrate spawner showed significant positive relationships with increasing water temperature and decreasing precipitation. Presence herbivore showed a significant positive relationship with water temperature. No other response variable showed any significant relationship with climate, particularly air or water temperature. Climate variables of air temperature, water temperature, and precipitation were poor predictors of fish community metrics in this study.

Unlike the climate variables described, these results suggest hydrology is a strong predictor of fish community metrics. Fifteen response variables showed significant positive relationships with hydrological variables, while only three response variables showed significant negative relationships with hydrological variables. These results are consistent with previous studies examining relationships of instream habitat and fish communities in both SJR and UBWC and other headwater streams (Poff et al. 1997; Sanders 2012; Smiley et al. 2008). Previous studies in the SJR and UBWC watersheds suggest hydrology has significant effects on fish communities (Sanders et al. 2020; Smiley et al. 2008). The results of this study are also consistent with literature from outside the SJR and UBWC watersheds (Bunn and Arthington 2002; Gorman and Karr 1978). Hydrology has also been shown to significantly influence water chemistry, which in turn has been shown to have weak, but still significant influences on fish communities (Smiley et al. 2009). Overall, these results suggest instream hydrology metrics are stronger predictors than climate metrics.

Intersectionality between headwater stream variables makes drawing definitive conclusions difficult. Hydrology metrics are often influenced by climate metrics (Christensen et al. 2004). Increases in precipitation directly influence water volume present in a watershed, which generally increase discharge, wet width, water velocity, and water depth (Karlsson et al. 2014). Increases in air temperature are correlated with increases in water temperature, though the relationship is not always linear (Morrill et al. 2005). In addition to direct quantitative relationships, climate change has also been linked to increased severe weather events, increased flooding, and more intense drought periods, all of which indirectly influence hydrology (Asadieh and Krakauer 2017). Nearly all chemistry, hydrology, and climate variables calculated for their respective studies have been shown to have some interaction with each other. Because of the relationships between climate and hydrology, it is difficult to draw definitive conclusions about direct relationships between fish community metrics and climate alone. These difficulties emphasize the importance of examining climate as a single factor within headwater streams, as all environmental factors in headwater streams are likely to be affected in some way by climate change.

Although this study suggests these climate metrics are not as strong as hydrology metrics at predicting fish community metrics, this does not mean air and water temperature have no effect on fish communities. Water temperature has been shown to have impacts on fish growth, distribution, feeding habits, and potential habitat range (Pauly 1980; Wenger et al. 2011). Salmonid species, which have been studied more thoroughly than other taxa due to their high economic importance, have notably low thermal tolerances and would be highly impacted by increases in water temperature (Knouft and Ficklin 2017). Distribution and phenology of fish species are frequently altered in response to climate change (Brown et al. 2016). Previous research strongly supports the idea that fish are highly influenced by changes in water temperature (Daufresne and Boet 2007; Hill and Magnuson 1990b; Wenger et al. 2011).

This study reported positive relationships between climate and three fish community metrics. A positive correlation between hydrological variables and percent Percidae is ecologically intuitive. The family Percidae includes both perch and darter species (Stepien and Haponski 2015). Larger perch species are commonly found in rivers, preferring relatively deep, open waters, while darter species prefer fast moving riffle habitats for feeding and predator avoidance (Greenberg 1991). These habitat and behavioral preferences are consistent with increased hydrological metrics of discharge, wet width, water velocity, and water depth, as well as increased precipitation. Deeper, fast flowing water should be more suitable for fish in the family Percidae. Warmer water temperatures have been linked to an increase in algae and plant life in aquatic ecosystems (Paerl et al. 2016), which may partially explain a positive relationship with herbivorous species, though more research is needed to fully understand this relationship.

Recent systematic review suggests the addition of habitat structure, including rock, gravel, and plant life, may be beneficial for open substrate spawning fishes (Taylor et al. 2019). The connection between warmer water, increase in plant and algae structure, and benefits to open substrate spawning fish is tenuous, and more research is necessary to fully understand these relationships. The underlying implication of these positive relationships and their potential ecological explanations is that changes in stream conditions due to climate change are likely not universally detrimental, and certain fish taxa such as darters may benefit from climate change. *Etheostoma nigrum* is the second most abundant species reported in this study, and additional study of fish communities and climate change in the SJR and UBWC watersheds could further clarify how climate change influences these taxa.

The lack of evidence for strong relationships between climate and fish variables was unexpected, given the current understanding of how climate change threatens not only aquatic ecosystems but all ecosystems worldwide (Butchart et al. 2010; Sala et al. 2000). Confounding factors may potentially explain the lack of significant relationships: 1) intense agricultural disturbance may more strongly influence fish communities than that from gradual climate change; 2) headwater streams in landscapes dominated by agriculture may only offer habitat suitable for highly tolerant fish species; 3) and, a 14-year study period may not be long enough to observe responses to climate change in fish communities.

By the end of the 19$^{th}$ century, farmland made up approximately 90 percent the state's acreage (Steinson 1994). Landscape modification for agricultural purposes, including stream channelization, has been shown to have dramatic impacts on geomorphology, hydrology, and fluvial processes, which have been shown to influence fish communities (Frothingham et al. 2002; Rhoads et al. 2016). Intense agricultural activity may be a more significant driver of fish community metrics in the SJR and UBWC watersheds, which may overshadow potential influences from climate change.

Fishes living in streams dominated by intense agricultural activity in the midwestern United States have been shown to be more tolerant of high turbidity, hypoxia, and hyperthermia (Matthews and Styron Jr 1981; Smale and Rabeni 1995). Conditions within those streams were also shown to be more favorable to generalist species than specialist species (Poff and Allan 1995). Fishes from the communities analyzed in this study may be relatively tolerant species, which are already subjected to harsh conditions due to agriculture, and therefore are unlikely to be as influenced by conditions brought on by climate change as more sensitive species in other habitats (Murdoch et al. 2020).

Fourteen years may be inadequate to detect long term effects of climate change (Callahan 1984; Hasselmann et al. 2003). Since 1895, Indiana's annual mean air temperature warmed by 0.72 ℃ (Widhalm and Dukes 2020). From 2006 to 2019, mean annual air temperature anomalies increased by 0.38 ℃, producing mean annual air temperatures 1.13 ℃ higher than those from 1991-2000 (NOAA National Centers for Environmental information 2021). Another potential explanation for lack of strong relationships is that climate change has already impacted headwater streams in the SJR and UBWC watersheds to a degree that these habitats are no longer available to temperature sensitive species. Although hydrology exhibited more significant relationships with fish community metrics, climate may still have indirect effects on fish communities in headwater streams. Further analysis will examine climate data used for this study to determine how much significant change in hydrology, air temperature, water temperature, and precipitation occurred between 2006 and 2019.

This is the first 10+ year study on fish communities in the Saint Joseph River and Upper Big Walnut Creek watersheds and the first to examine climate as a potential influence. Although these results only somewhat supported the hypothesis of my study, this report has interesting implications for future research in this watershed. Riparian habitat, water chemistry, geomorphology, sediments, total suspended solids, nutrient loads, bioassays, crayfish, fish, and macroinvertebrate communities, fish length and biomass, and now climate have all been examined through the Conservation Effects Assessment Project in these watersheds. By exploring these ecological factors, we may be able to understand this headwater ecosystem more completely, which could provide valuable insight on how to best manage agricultural headwater streams. In conclusion, this study suggests that hydrology is a stronger predictor of fish community metrics than climate, although the potential relationships with climate are still important. The methods employed here may not have detected climate change within our watersheds, but additional research may offer additional insight into how climate interacts with fish communities in headwater streams. Although this study reported few significant relationships between our climate metrics and fish community variables does not indicate that climate change has had no effect on fish communities in Indiana and Ohio.

# REFERENCES


Alexander RB, Boyer EW, Smith RA, Schwarz GE, Moore RB. 2007. The role of headwater streams in downstream water quality 1. JAWRA Journal of the American Water Resources Association. 43:41-59.

Asadieh B, Krakauer NY. 2017. Global change in streamflow extremes under climate change over the 21st century. Hydrol Earth Syst Sci. 21:5863-5874.

Bates D, Mächler M, Bolker B, Walker S. 2015. Fitting linear mixed-effects models using lme4. Journal of Statistical Software. 67:1-48.

Becker GC. 1983. Fishes of wisconsin. Madison: University of Wisconsin Press.

Beever E, Hall LE, Varner J, Loosen A, Dunham J, Gahl M, Smith F, Lawler J. 2017. Behavior flexibility as a mechanism for coping with climate change. Frontiers in Ecology and the Environment. 15:299-308.

Belden JB, Gilliom RJ, Lydy MJ. 2007. How well can we predict the toxicity of pesticide mixtures to aquatic life? Integrated Environmental Assessment and Management: An International Journal. 3:364-372.

Blann KL, Anderson JL, Sands GR, Vondracek B. 2009. Effects of agricultural drainage on aquatic ecosystems: A review. Critical Reviews in Environmental Science and Technology. 39:909-1001.

Brookes A. 1987. Restoring the sinuosity of artificially straightened stream channels. Environmental Geology and Water Sciences. 10:33-41.

Brown CJ, O'Connor MI, Poloczanska ES, Schoeman DS, Buckley LB, Burrows MT, Duarte CM, Halpern BS, Pandolfi JM, Parmesan C. 2016. Ecological and methodological drivers of species' distribution and phenology responses to climate change. Global change biology. 22:1548-1560.

Bunn SE, Arthington AH. 2002. Basic principles and ecological consequences of altered flow regimes for aquatic biodiversity. Environmental management. 30:492-507.

Butchart SH, Walpole M, Collen B, Van Strien A, Scharlemann JP, Almond RE, Baillie JE, Bomhard B, Brown C, Bruno J. 2010. Global biodiversity: Indicators of recent declines. Science. 328:1164-1168.

Callahan JT. 1984. Long-term ecological research. BioScience. 34:363-367.

Carlisle D, Grantham TE, Eng K, Wolock DM. 2017. Biological relevance of streamflow metrics: Regional and national perspectives. Freshwater Science. 36:927-940.

Christensen NS, Wood AW, Voisin N, Lettenmaier DP, Palmer RN. 2004. The effects of climate change on the hydrology and water resources of the colorado river basin. Climatic change. 62:337-363.

Colvin SAR, Sullivan SMP, Shirey PD, Colvin RW, Winemiller KO, Hughes RM, Fausch KD, Infante DM, Olden JD, Bestgen KR et al. 2019. Headwater streams and wetlands are critical for sustaining fish, fisheries, and ecosystem services. Fisheries. 44:73-91.

Daufresne M, Boet P. 2007. Climate change impacts on structure and diversity of fish communities in rivers. Global Change Biology. 13:2467-2478.

Dempsey JA, Plantinga AJ, Kline JD, Lawler JJ, Martinuzzi S, Radeloff VC, Bigelow DP. 2017. Effects of local land-use planning on development and disturbance in riparian areas. Land use policy. 60:16-25.

Dey P, Mishra A. 2017. Separating the impacts of climate change and human activities on streamflow: A review of methodologies and critical assumptions. Journal of Hydrology. 548:278-290.

Durance I, Ormerod SJ. 2007. Climate change effects on upland stream macroinvertebrates over a 25-year period. Global Change Biology. 13:942-957.

Eakins RJ. 2020. Ontario freshwater fishes life history database.

Eaton JG, Scheller RM. 1996. Effects of climate warming on fish thermal habitat in streams of the united states. Limnology and oceanography. 41:1109-1115.

Effert-Fanta EL, Fischer RU, Wahl DH. 2019. Effects of riparian forest buffers and agricultural land use on macroinvertebrate and fish community structure. Hydrobiologia. 841:45-64.

Ficke AD, Myrick CA, Hansen LJ. 2007. Potential impacts of global climate change on freshwater fisheries. Reviews in Fish Biology and Fisheries. 17:581-613.

Fischer J, Quist M, Wigen S, Schaefer A, Stewart T, Isenhart T. 2010. Assemblage and population-level responses of stream fish to riparian buffers at multiple spatial scales. Transactions of The American Fisheries Society - TRANS AMER FISH SOC. 139:185-200.

Freeman MC, Pringle CM, Jackson CR. 2007. Hydrologic connectivity and the contribution of stream headwaters to ecological integrity at regional scales 1. JAWRA Journal of the American Water Resources Association. 43:5-14.

Frissel C, Liss W, Warren C, Hurley M. 1986. A hierarchical framework for stream habitat classification: Viewing streams in a watershed context. Environmental Management 10:199-214.

Fritz K, Johnson B, Walters D. 2006. Field operations manual for assessing the hydrologic permanence and ecological condition of headwater streams. Washington DC: U.S. Environmental Protection Agency, Office of Research and Development.

Fishbase. 2008. World Wide Web electronic publication; [accessed]. www.fishbase.org.

Frothingham KM, Rhoads BL, Herricks EE. 2002. A multiscale conceptual framework for integrated ecogeomorphological research to support stream naturalization in the agricultural midwest. Environmental management. 29:16-33.

Gilliom RJ. 2007. Pesticides in us streams and groundwater. Environmental Science & Technology. 41.

Gorman OT, Karr JR. 1978. Habitat structure and stream fish communities. Ecology. 59:507-515.

Greenberg LA. 1991. Habitat use and feeding behavior of thirteen species of benthic stream fishes. Environmental Biology of Fishes. 31:389-401.

Hall K. 2012. Climate change in the midwest: Impacts on biodiversity and ecosystems. The Nature Conservancy, Great Lakes Project.

Harrel R, Davis B, Dorris T. 1967. Stream order and species diversity of fishes in an intermittent oklahoma stream. The American Midland Naturalist. 78:428-436.

Harrison XA, Donaldson L, Correa-Cano ME, Evans J, Fisher DN, Goodwin CED, Robinson BS, Hodgson DJ, Inger R. 2018. A brief introduction to mixed effects modelling and multi-model inference in ecology. PeerJ. 6:e4794-e4794.

Hasselmann K, Latif M, Hooss G, Azar C, Edenhofer O, Jaeger CC, Johannessen OM, Kemfert C, Welp M, Wokaun A. 2003. The challenge of long-term climate change. Science. 302(5652):1923-1925.

Herzon I, Helenius J. 2008. Agricultural drainage ditches, their biological importance and functioning. Biological Conservation. 141(5):1171-1183.

Hill D, Magnuson J. 1990a. Potential effects of global climate warming on the growth and prey consumption of great lakes fish. Transactions of the American Fisheries Society. 119:265-275.

Hill DK, Magnuson JJ. 1990b. Potential effects of global climate warming on the growth and prey consumption of great lakes fish. Transactions of the American Fisheries Society. 119(2):265-275.

Höök T, Foley C, Collingsworth P, Dorworth L, Fisher B, Hoverman JD, LaRue E, Pryon M, Tank J, Widhalm M et al. 2018. "Aquatic ecosystems in a shifting indiana climate: A report from the indiana climate change impacts assessment. West Lafayette: Aquatic Ecosystems Reports.

Jordan MA, Castañeda AJ, Smiley PC, Gillespie RB, Smith DR, King KW. 2016. Influence of instream habitat and water chemistry on amphibians in channelized agricultural headwater streams. Agriculture, Ecosystems & Environment. 230:87-97.

Karlsson I, Sonnenborg T, Jensen K, Refsgaard J. 2014. Historical trends in precipitation and stream discharge at the skjern river catchment, denmark. Hydrology and Earth System Sciences. 18:595-610.

Karr JR, Toth LA, Dudley DR. 1985. Fish communities of midwestern rivers: A history of degradation. BioScience. 35:90-95.

King JR, Shuter BJ, Zimmerman AP. 1999. Empirical links between thermal habitat, fish growth, and climate change. Transactions of the American Fisheries Society. 128:656-665.

King KW, Smiley PC, Fausey NR. 2009. Hydrology of channelized and natural headwater streams / hydrologie de cours d'eau recalibrés et naturels de tête de bassin. Hydrological Sciences Journal. 54:929-948.

Kitchell J, Breck J. 1980. Bioenergetics model and foraging hypothesis for sea lamprey (petromyzon marinus). Canadian Journal of Fisheries and Aquatic Sciences. 37:2159-2168.

Knouft JH, Ficklin DL. 2017. The potential impacts of climate change on biodiversity in flowing freshwater systems. Annual Review of Ecology, Evolution, and Systematics. 48:111-133.

Krakauer NY, Lakhankar T, Hudson D. 2019. Trends in drought over the northeast united states. Water. 11:1834.

Lambert FH, Stine AR, Krakauer NY, Chiang JCH. 2008. How much will precipitation increase with global warming? Eos, Transactions American Geophysical Union. 89:193-194.

Lyons J, Magnuson J. 1986. Effects of walleye predation on the population dynamics of small littoral-zone fishes in a northern wisconsin lake. Transactions of the American FIsheries Society. 116:29-39.

Marschall EA, Mather ME, Parrish DL, Allison GW, McMenemy JR. 2011. Migration delays caused by anthropogenic barriers: Modeling dams, temperature, and success of migrating salmon smolts. Ecological Applications. 21:3014-3031.

Matthews WJ, Styron Jr JT. 1981. Tolerance of headwater vs. Mainstream fishes for abrupt physicochemical changes. American Midland Naturalist. 105:149-158.

McCall JD, Knox RF. 1979. Riparian habitat in channelization projects Washington D.C.: U.S Department of Agriculture, Forest Service.

Meador MR. 2013. Nutrient enrichment and fish nutrient tolerance: Assessing biologically relevant nutrient criteria1. JAWRA Journal of the American Water Resources Association. 49:253-263.

Meador MR, Goldstein RM. 2003. Assessing water quality at large geographic scales: Relations among land use, water physicochemistry, riparian condition, and fish community structure. Environmental Management. 31:0504-0517.

Metz LJ, Rewa CA. 2019. Conservation effects assessment project: Assessing conservation practice effects on grazing lands. Rangelands. 41:227-232.

Mishra V, Cherkauer K. 2010. Assessment of drought due to historic climate variability and projected future climate change in the midwestern united states. Journal of Hydrometeorology. 11:46-68.

Moore AA, Palmer MA. 2005. Invertebrate biodiversity in agricultural and urban headwater streams: Implications for conservation and management. Ecological Applications. 15:1169-1177.

Morrill JC, Bales RC, Conklin MH. 2005. Estimating stream temperature from air temperature: Implications for future water quality. Journal of Environmental Engineering. 131:139-146.

Murdoch A, Mantyka-Pringle C, Sharma S. 2020. The interactive effects of climate change and land use on boreal stream fish communities. Science of the Total Environment. 700:134518.

NOAA National Centers for Environmental Information. 2020. State of the climate: Global climate report for annual 2019.

Climate at a glance: Global time series. 2021. [accessed 2021 February 18]. https://www.ncdc.noaa.gov/cag/.

Paerl HW, Gardner WS, Havens KE, Joyner AR, McCarthy MJ, Newell SE, Qin B, Scott JT. 2016. Mitigating cyanobacterial harmful algal blooms in aquatic ecosystems impacted by climate change and anthropogenic nutrients. Harmful Algae. 54:213-222.

Pauly D. 1980. On the interrelationships between natural mortality, growth parameters, and mean environmental temperature in 175 fish stocks. ICES Journal of Marine Science. 39:175-192.

Peckham S, Gupta V. 1999. A reformulation of horton's laws for large river networks in terms of statistical self-similarity. WATER RESOURCES RESEARCH. 35:2763-2777.

Petersen Jr RC, Madsen BL, Wilzbach MA, Magadza CH, PAARIBERG A. 1987. Stream management emerging global similarities. Ambio. 16:166-179.

Pflieger W, Sullivan M, Taylor L. 1997. The fishes of missouri (revised edition). Jefferson City: Missouri Department of Conservation.

Poff NL, Allan JD. 1995. Functional organization of stream fish assemblages in relation to hydrological variability. Ecology. 76:606-627.

Poff NL, Allan JD, Bain MB, Karr JR, Prestegaard KL, Richter BD, Sparks RE, Stromberg JC. 1997. The natural flow regime. BioScience. 47:769-784.

Pryor SC, Scavia D, Downer C, Gaden M, Iverson L, Nordstrom R, Patz J, Robertson GP. 2014. Midwest. Climate change impacts in the united states: The third national climate assessment. In: Melillo, JM; Richmond, TC; Yohe, GW, eds National Climate Assessment Report Washington, DC: US Global Change Research Program: 418-440.418-440.

R Core Team. 2020. R: A language and environment for statistical computing. Vienna, Austria: R Foundation for Statistical Computing.

Rhoads BL, Lewis QW, Andresen W. 2016. Historical changes in channel network extent and channel planform in an intensively managed landscape: Natural versus human-induced effects. Geomorphology. 252:17-31.

Rice J, Cochran P. 1984. Independent evaluation of a bioenergetics model for largemouth bass. Ecology. 65:732-739.

RStudio Team. 2020. Rstudio: Integrated development for r. Rstudio. PBC: Boston, MA: RStudio.

Ruesch AS, Torgersen CE, Lawler JJ, Olden JD, Peterson EE, Volk CJ, Lawrence DJ. 2012. Projected climate-induced habitat loss for salmonids in the john day river network, oregon, u.S.A. Conservation Biology. 26:873-882.

Sala OE, Chapin FS, Armesto JJ, Berlow E, Bloomfield J, Dirzo R, Huber-Sanwald E, Huenneke LF, Jackson RB, Kinzig A. 2000. Global biodiversity scenarios for the year 2100. science. 287:1770-1774.

Sanders KE. 2012. Relative importance of water quality and habitat to fish communities in streams influenced by agricultural land use in the cedar creek watershed, indiana. Citeseer.

Sanders KE, Smiley Jr. PC, Gillespie RB, King KW, Smith DR, Pappas EA. 2020. Conservation implications of fish–habitat relationships in channelized agricultural headwater streams. Journal of Environmental Quality. 49.

Scarnecchia DL. 1988. The importance of streamlining in influencing fish community structure in channelized and unchannelized reaches of a prairie stream. Regulated Rivers: Research & Management. 2:155-166.

Schwarzenbach RP, Egli T, Hofstetter TB, von Gunten U, Wehrli B. 2010. Global water pollution and human health. Annual Review of Environment and Resources. 35:109-136.

Shuman TC, Smiley PC, Gillespie RB, Gonzalez JM. 2020. Influence of physical and chemical characteristics of sediment on macroinvertebrate communities in agricultural headwater streams. Water. 12:2976.

Shuter B, Finstad A, Helland I, Zweimüller I, Hölker F. 2012. The role of winter phenology in shaping the ecology of freshwater fish and their sensitivities to climate change. Aquatic Sciences. 74:637-657.

Smale MA, Rabeni CF. 1995. Hypoxia and hyperthermia tolerances of headwater stream fishes. Transactions of the American Fisheries Society. 124:698-710.

Smiley Jr. PC. 2020.

Smiley P, Gillespie R, King K, Huang C-h. 2009. Management implications of the relationships between waterchemistry and fishes within channelized headwater streams inthe midwestern united states. Ecohydrology.294-302.

Smiley PC, Gillespie RB, King KW, Huang C-h. 2008. Contribution of habitat and water quality to the integrity of fish communities in agricultural drainage ditches. Journal of Soil and Water Conservation. 63:218A-219A.

Smiley PC, King KW, Fausey NR. 2011. Influence of herbaceous riparian buffers on physical habitat, water chemistry, and stream communities within channelized agricultural headwater streams. Ecological Engineering. 37:1314-1323.

Steinson BJ. 1994. Rural life in indiana, 1800–1950. The Indiana Magazine of History. 90:203-250.

Stepien CA, Haponski AE. 2015. Taxonomy, distribution, and evolution of the percidae. Biology and culture of percid fishes. Dordrecht, Netherlands: Springer. p. 3-60.

Stewart D, Weininger D, Rottiers D, Edsall T. 1983. An energetics model for lake trout, salvelinus namaycush: Application to the lake michigan population. Canadian Journal of Fisheries and Aquatic Sciences. 40:681-698.

Strahler A. 1957. Quantitative analysis of watershed geomorphology. Transactions of the American Geophysical Union. 38:913-920.

Sullivan BE, Rigsby LS, Berndt A, Jones-Wuellner M, Simon TP, Lauer T, Pyron M. 2004. Habitat influence on fish community assemblage in an agricultural landscape in four east central indiana streams. Journal of Freshwater Ecology. 19:141-148.

Taylor JJ, Rytwinski T, Bennett JR, Smokorowski KE, Lapointe NW, Janusz R, Clarke K, Tonn B, Walsh JC, Cooke SJ. 2019. The effectiveness of spawning habitat creation or enhancement for substrate-spawning temperate fish: A systematic review. Environmental Evidence. 8:1-31.

Tóth R, Czeglédi I, Kern B, Erős T. 2019. Land use effects in riverscapes: Diversity and environmental drivers of stream fish communities in protected, agricultural and urban landscapes. Ecological Indicators. 101:742-748.

Trautman MB. 1939. The effects of man-made modifications on the fish fauna in lost and gordon creeks, ohio, between 1887-1938. The Ohio Journal of Science. 39:275-288.

U.S. Environmental Protection Agency. 2016. National rivers and streams assessment 2008-2009: A collaborative survey. Washington D.C.: Office of Water and Office of Research and Development.

U.S. EPA. 2008. Climate change effects on stream and river biological indicators: A preliminary analysis (final report). Washington D.C.: Global Change Research Program, National Center for Environmental Assessment.

USGCRP. 2018. 2018: Impacts, risks, and adaptation in the united states: Fourth national climate assessment, volume ii: Report-in-brief. Washington D.C.: U.S. Global Change Research Program.

R package "corrplot": Visualization of a correlation matrix. 2017. 0.84. [accessed]. https://github.com/taiyun/corrplot.

Wenger SJ, Isaak DJ, Luce CH, Neville HM, Fausch KD, Dunham JB, Dauwalter DC, Young MK, Elsner MM, Rieman BE et al. 2011. Flow regime, temperature, and biotic interactions drive differential declines of trout species under climate change. Proceedings of the National Academy of Sciences. 108:14175-14180.

Widhalm M, Dukes JS. 2020. Introduction to the indiana climate change impacts assessment: Overview of the process and context. Climatic Change. 163:1-11.

Wisconsin fish id. 2020. Madison, WI: University of Wisconsin; [accessed]. https://www.seagrant.wisc.edu/fish-id/.

Wohl E. 2017. The significance of small streams. Frontiers of Earth Science. 11:447-456.

Wood TC, Smiley PC, Gillespie RB, Gonzalez JM, King KW. 2020. Injury frequency and severity in crayfish communities as indicators of physical habitat quality and water quality within agricultural headwater streams. Environmental Monitoring and Assessment. 192:227.

Wu JY, Thompson JR, Kolka RK, Franz KJ, Stewart TW. 2013. Using the storm water management model to predict urban headwater stream hydrological response to climate and land cover change. Hydrol Earth Syst Sci. 17:4743-4758.

Wuebbles DJ, Fahey DW, Hibbard KA, Dokken DK, Stewart BC, Maycock TK. 2017. Climate science special report: Fourth national climate assessment, volume i. Washington D.C.: USGCRP.

Young RG, Huryn AD. 1999. Effects of land use on stream metabolism and organic matter turnover. Ecological Applications. 9:1359-1376.