# IMPORTANCE OF TOTAL SUSPENDED SOLIDS IN EXPLAINING FISH COMMUNITY STRUCTURE IN AGRICULTURAL HEADWATER STREAMS

by

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*for my family*

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# ABSTRACT

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Agricultural headwater streams in the Midwestern United States are subject to contaminants from fields, increased sedimentation, and degradation of natural habitat. Previous research has shown that physical instream habitat degradation better explained variation in fish community structure than water chemistry. However, these studies did not include total suspended solids (TSS), which are considered a major freshwater contaminant. The objective of this study is to determine whether total suspended solids better explains fish community structure than other variables in agricultural headwater streams. Mixed linear effects modeling was used to determine the set of independent variables that best predicts each of the fish response variables of species richness, Shannon diversity index, fish density, and index of biotic integrity. Standardized coefficients were used to determine which independent variable in each of the models had the largest influence on fish response metrics. The set of independent variables that best explained species richness were mean total suspended solids, imidacloprid, discharge, and substrate richness. Shannon diversity index was explained best by the combination of maximum total suspended solids, mean total suspended solids, atrazine, total nitrogen, and discharge. Fish density was explained best by the percentage of silt and clay, dissolved oxygen, the percentage of canopy cover, cover type richness, and discharge. IBI was explained best by the combination of the percentage of silt and clay, total phosphorus, mean total suspended solids, and dissolved oxygen. Total suspended solids was the most influential independent variable for fish species richness and Shannon diversity, however the percentage of silt and clay in benthic sediments was the most influential independent variable for fish density and IBI. Results also indicate discharge and total phosphorus as being influential to fish community metrics. The results from this study suggest that models containing a combination of different types of independent variables best explain fish community structure. This study supports the use of conservation and restoration practices that reduce total suspended solids and the amount of silt and clay present in bed sediments to increase fish community integrity of agricultural headwater streams of the Midwestern United States.

# INTRODUCTION

Headwater streams comprise approximately 70-80% of the total United States stream length (Nadeau, 2007; Richardson, 2007). They are found in the uppermost reaches of a watershed and are essential to conserving downstream waters, such as rivers and lakes (Colvin, 2019). Headwater streams are perennial, intermittent, or ephemeral, with perennial streams maintaining year-round flow, intermittent streams drying up during part of the year, and ephemeral streams only flowing in response to precipitation (Nadeau, 2007).

Headwater streams provide essential habitat for many organisms, which includes shallow depths, low velocity, sinuosity, riffle/run development, and larger substrates (Colvin, 2019; Lau et al., 2006; Meyer, 2007). Primary producers, such as algae, bryophytes, and other macrophytes rely on headwater streams for maintenance and transfer of propagules (Meyer, 2007). Many macrophytes require habitat with slower discharge, shallow water, and high solar radiation, which is characteristic of headwater streams having limited canopy coverage (Grinberga, 2010; Richardson, 2007).

Unique characteristics of headwater streams provide the habitat required for many macroinvertebrates to thrive. Increased macroinvertebrate richness has been associated with streams that are shallow, have sinuosity, and low velocity (Nakano, 2008). Other invertebrates, such as freshwater mussels inhabit headwater streams with some of these species being endangered (Meyer, 2007). Freshwater mussels rely on many headwater fish species for reproduction and are important components to the aquatic ecosystem since they facilitate oxygen exchange in sediments during burrowing and help filter the water during suspension feeding (Stoeckl et al., 2015; Tuttle-Raycraft et al., 2017).

Headwater streams have shallow depth, low velocity, and diverse substrates, which make them essential to many fish species. Some fishes, such as salmon and minnow species, use headwater stream habitat for breeding (Colvin, 2019; Meyer, 2007). These fishes travel to headwater streams to spawn and use the habitat for rearing young before moving on to larger streams and rivers (Meyer, 2007). Other fishes, such as darters and sculpin are only found in headwater streams due to their need for specific habitat (Meyer, 2007). Fantail darters (*Etheostoma flabellare*) and Orangethroat darters (*Etheostoma spectabile*) prefer habitats with larger substrates, shallow depths, and low velocity, which are characteristic of headwater streams (Pratt & Lauer, 2013). Overall, the unique habitat found in headwater streams is important to many types of aquatic species and loss of headwater habitat would be detrimental to those species.

The quality of headwater streams is being negatively impacted by many anthropogenic activities that alter habitat and introduce agrichemicals and sediments (Armstrong et al., 2012). Many streams in the Midwestern United States have been converted to drainage ditches for agriculture (Freeman et al., 2007; Mattingly et al., 1993). Headwater streams are channelized to improve drainage of agricultural fields, which reduces flooding that occurs during heavy rain events and removes excess water from the soil to improve crop yields for farmers (D'Ambrosio et al., 2014; Roley et al., 2012).

Channelization removes natural stream features such as sinuosity, riffle/run development, and larger substrates which are essential habitat characteristics for many aquatic species (Lau et al., 2006). The traditional channelized ditch is a trapezoidal-shaped design that makes stream banks more susceptible to erosion, which leads to repeated dredging to maintain the ideal flow and drainage.  Lau et al. (2006) found that the integrity of fish communities was of lower quality in channelized streams that lack natural meandering and development. Furthermore, dredging is an indiscriminate process so any organisms that are not able to escape such as plants, macroinvertebrates, and fishes will also be removed with the sediment and will perish (Shaw et al., 2015; Wenger et al., 2017). Overall, agricultural headwater instream habitat has been altered, which has a negative impact on the aquatic species that rely on headwater habitat for survival.

Headwater streams are subjected to contamination from agricultural fields (D'Ambrosio et al., 2014; Roley et al., 2012). Pesticides and nutrients that do not get absorbed by soil or plants can easily make their way into drainage ditches through ground water, tile drains, or surface runoff during rain events (Edwards, 2008; Williamson et al., 2014). These chemicals can have a negative impact on the aquatic communities through impacts such as, feminization of male frogs (Hayes et al., 2010), occurrence of testicular oocytes in fishes (Blazer et al., 2012), and oxidative damage to DNA of fishes (Iturburu et al., 2018).

Suspended solids are a known aquatic contaminant in Midwestern United States agricultural streams (Waters, 1995). Most agriculture requires disturbance of the soil through tilling and harvesting, which can lead to more sediments and other solid material entering adjacent streams. The amount of solids being introduced to the water column is increased during rain events because disturbed fields have little protection (i.e. plant cover) from loss of soil (Sciera et al., 2008). Increased solids in headwater streams can lead to higher turbidity, which can impact fish's vision and ability to find shelter or food (Prestigiacomo et al., 2007). Increased solids can also negatively impact filter feeding organisms by increasing the material they need to process. For example, researchers found decreased feeding in freshwater mussels exposed to high loads of suspended solids (Tuttle-Raycraft et al., 2017). Overall, suspended solids are a common contaminant in agricultural headwater streams and have negative impacts on aquatic biota.

It is important to determine what variables most strongly influence the integrity of aquatic communities. This information will allow conservationists and landowners to focus their investments for improving stream quality on the most beneficial impacts. Previous research within agricultural headwater streams in the Midwestern United States has found that physical habitat degradation was more important than water chemistry to the integrity of the fish communities (Sanders, 2012; Smiley et al., 2008) and water chemistry was more important in explaining amphibian community structure than instream habitat (Jordan et al., 2016). Studies have also found that land use, soil type, and channel morphology strongly influenced abundance and biomass of creek chubs (*Semotilus atromaculatus*), a common species found in midwestern agricultural headwater streams (Smiley et al., 2017). The studies mentioned above did not include total suspended solids (TSS) even though they have been identified as an influential contaminant in many freshwater systems (Waters, 1995).

Previous research has documented the negative impact of TSS to survival of eggs and juvenile smallmouth bass (*Micropterus dolomieu*) at concentrations of 100 mg/L and salmonid fishes at concentrations of 20-180 mg/L (Bilotta & Brazier, 2008; Suedel et al., 2017). Suspended solids also caused sub-lethal damage to the gill structure of cyprinids at concentrations of 100 mg/L (Sutherland & Meyer, 2007). Total suspended solids have been found to increase foraging time in fishes and cause avoidance of areas with high concentrations of TSS. The latter leads to an increase in percent of tolerant fish in assemblages (Schleiger, 2000; Wenger et al., 2017; Zimmerman et al., 2003). Many fishes rely on macroinvertebrates as a food source which, show increased drift, lower reproduction, and smaller population sizes when exposed to total suspended solid concentrations of 8 mg/L (Bilotta & Brazier, 2008).

Since TSS has been found to be an influential contaminant to fish communities, it is important to determine how important TSS is in explaining fish community integrity of agricultural headwater streams in the Midwestern United States. The purpose of this research is to determine if total suspended solids can better explain variation in fish community metrics than any other variable in agricultural headwater streams. My hypothesis was that total suspended solids will best explain fish community structure in agricultural headwater streams of the Midwestern United States.

# METHODS

## Study sites

Sixteen sites in the Upper Big Walnut Creek Watershed (UBWC), Ohio, and nine sites in the Saint Joseph River Watershed (SJR), Indiana and Michigan, were selected as sampling sites. The watershed size for study areas ranged from 1.1 to 9.7 $km^2$ in UBWC and 1.4 to 278.8 $km^2$ in SJR. The percentage of agricultural land use ranged from 48 to 95 percent in UBWC and 54 to 83 percent in SJR. Land use data for SJR was quantified using the Great Lake Regional L-THIA modeling system and UBWC land use data was obtained from the Delaware, Ohio Soil and Water Conservation District.

## Instream and Riparian Habitat

Sampling locations consisted of 125 m segments. Instream and riparian habitat were measured at transects placed every 25 m throughout the site. Instream habitat was assessed at SJR and UBWC in May, July, and September of 2017 and at UBWC in May and July of 2018. Instream habitat was characterized by measuring wet width, and recording depth and velocity at 20, 40, 60, and 80% of the wet width along each transect. A substrate sample was collected at each of the four points along the transects and was assessed qualitatively for composition. The dominant substrate type was identified at each point as either silt, clay, sand, gravel, cobble, or boulder. The dominant instream cover type was identified at each point. Possible cover types included terrestrial vegetation, small woody debris, large woody debris, algae, aquatic plant, and leaf litter. Riparian habitat was surveyed at SJR in August of 2017 and September of 2018. UBWC riparian habitat was surveyed in September and October of 2017 and October of 2018. Percent canopy cover was measured at each transect on both banks and within the stream using a concave spherical densiometer.

## Water Chemistry

Water samples were collected on a weekly basis at SJR watershed sites from May 2017 through October 2017 and at UBWC watershed sites from May 2017 to October 2017 and May 2018 to August 2018. Water samples were collected from each of the sampling locations and were distributed into three bottles for analysis of turbidity, total suspended solids, pesticides and nutrients. Total suspended solids, pesticides, and nutrients of SJR were analyzed at the Agricultural Research Service, National Soil and Erosion Research Lab (NSERL). Nutrients for UBWC were analyzed by the Soil Drainage Research Unit (SDR) and nutrients were analyzed by NSERL. Nutrients and pesticides measured included total nitrogen (mg/L), total phosphorus (mg/L), atrazine (µg/L), imidacloprid (µg/L) and metalaxyl (µg/L). Total nitrogen is the sum of ammonia, organic nitrogen, reduced nitrogen, and nitrate-nitrite. Total suspended solids were measured using the gravimetric method for SJR sites. Turbidity samples were analyzed using a Hanna turbidity meter and were used in conjunction with established turbidity/TSS regression equations to predict TSS values for UBWC sampling sites. Temperature (°C), pH, specific conductivity (S/m), and dissolved oxygen (mg/L) were recorded during instream habitat sampling using a MS5 Hydrolab and/or YSI Pro 1020 meter in SJR and a YSI 556 multiparameter meter in UBWC.

## Fish surveys

Fish were sampled using a Halltech backpack electrofishing unit, once in each of May, July, and September of 2017 in SJR; in May and July, and October of 2017; and May and July of 2018 in UBWC. Electrofishing proceeded by moving downstream to upstream through the sampling location. Effort was made to collect fishes from all available microhabitat types Additionally, a 2 m seine net with 6 mm mesh size was used to collect fishes at one random point distributed every 25 m throughout the 125 m long site. Fishes were counted and identified to species at all sampling locations. Fishes were examined for abnormalities using protocols outlined in OEPA (2015) at SJR sites and four of the UBWC sites. All fish were released after identification.

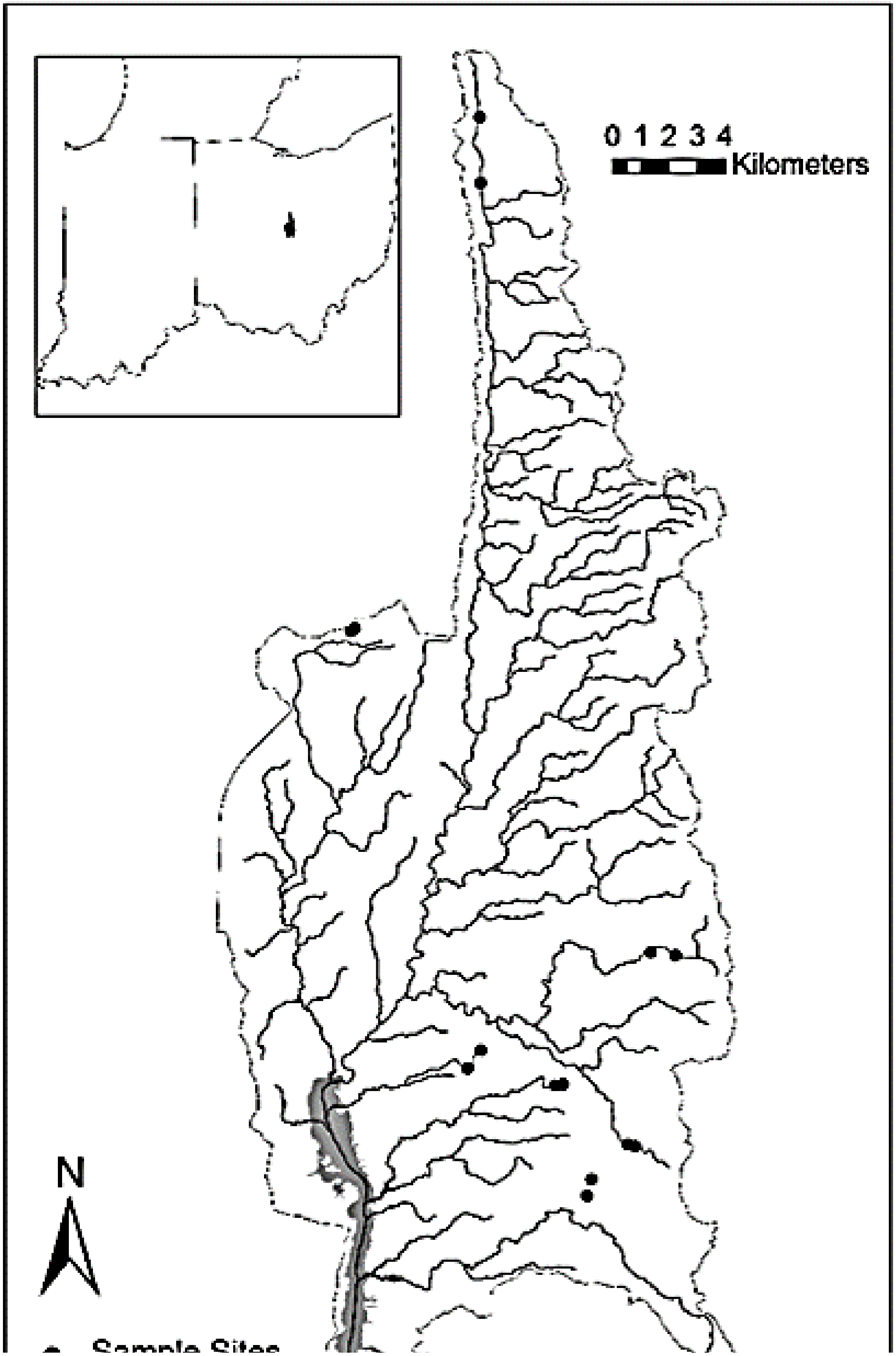


Figure 1: Map of Upper Big Walnut Creek (UBWC) study area, Ohio. Sampling locations are designated by the black dots. Shaded area represents the hoover reservoir.

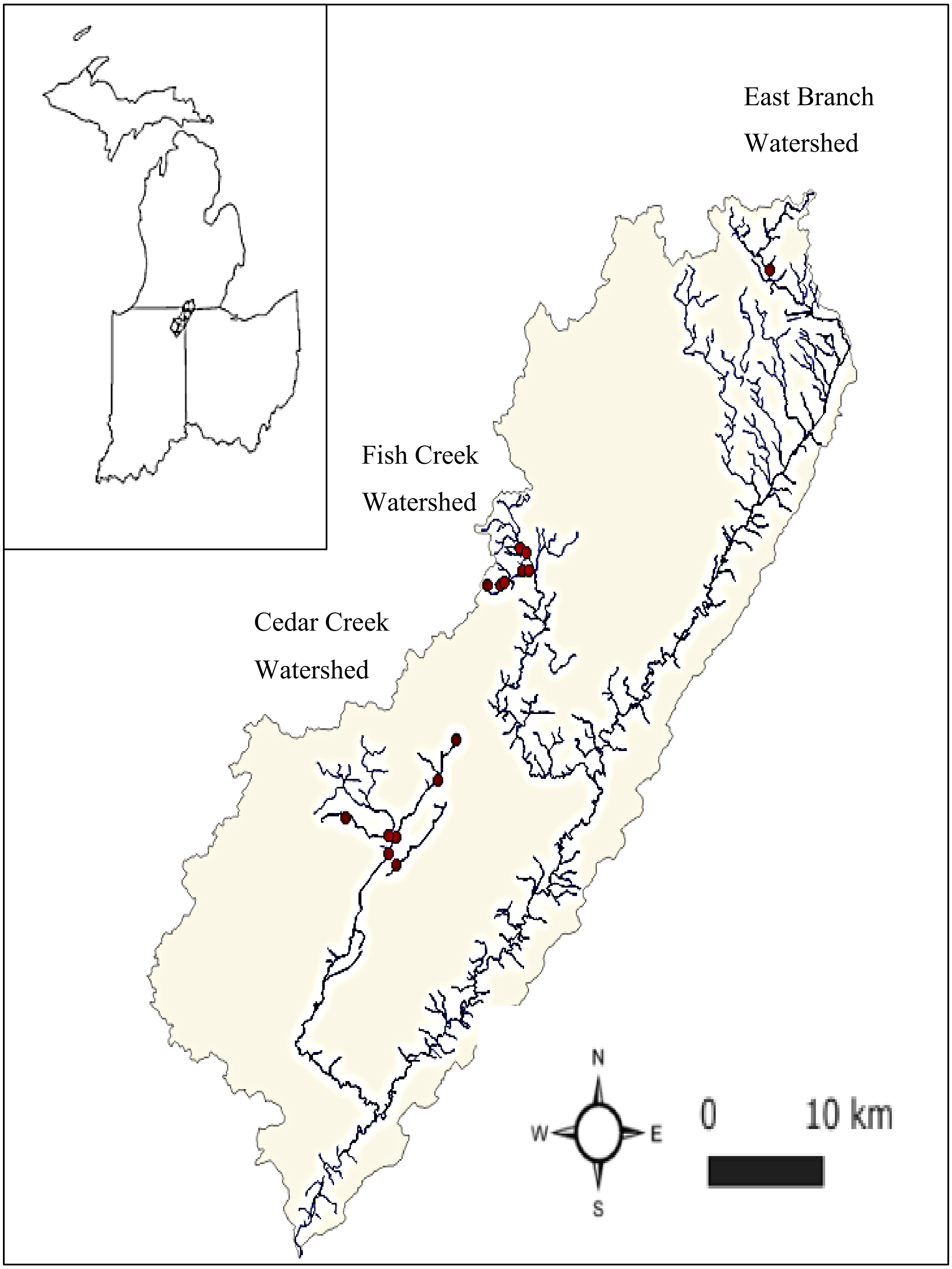


Figure 2: Map of Saint Joseph River Watershed (SJR) study area encompassing parts of Northeast Indiana and southern Michigan. Sampling sites were located in the sub watersheds of Cedar Creek, Fish Creek, and East Branch and are designated by the dots.

## Statistical analysis

### Fish response variables

Species richness, Shannon Diversity Index, fish density, and the index of biotic integrity (IBI) (Karr, 1987; OEPA, 1988) were calculated. Using criteria outlined in Karr (1987) and OEPA (1988) IBI metrics received a score of 5 if the metric was consistent with the range of a characteristically undisturbed sites, 3 if values suggest slight degradation, and 1 if values indicate severe degradation. Fish density response variable was not normally distributed and data was log transformed to satisfy normality.

### Habitat and water chemistry variables

From each site on each sampled day I calculated the proportion of each dominant substrate type and proportion of each dominant cover type. Substrate richness and cover type richness were also calculated for each sampling event. Substrate richness was the number of different types of substrate. Cover type richness was the number of different types of cover types present. Discharge was calculated for each fish sampling event from depth and velocity measurements collected on the day of fish sampling prior to fishes being collected in SJR and after fishes have been collected in UBWC (see Table 1). Instream habitat variables were selected by using correlation matrices, which helped avoid multicollinearity between independent variables. Independent variables with correlation coefficients greater than 0.6 were removed from the analysis. Independent variables selected for further analyses, included discharge, percentage of silt/clay, substrate richness, percentage of leaf litter, and cover type richness. Riparian habitat was represented by percent canopy cover. The percentage of agricultural land use represented a watershed metric.

Table 1: Equations and criteria used to calculate metrics

| Variable | Criteria/ Equation |
|---|---|
| Species Richness | total number of species |
| Shannon Diversity Index | $-\Sigma(P_i*\ln[P_i])$<br>$P_i$: Proportion of $i^{th}$ species |
| Fish Density | $\frac{number\ of\ fish\ caught}{area\ (m^2)}$ |
| IBI Score Subcomponents | Total number of species<br>number of darter species (include sculpin)<br>number of headwater species<br>number of minnow species<br>number of sensitive species<br>percent of tolerant species<br>percent of omnivorous species<br>percent of insectivorous species<br>percent of pioneering species<br>number of individuals/ 300m<br>number of simple lithophilic species<br>percent of DELT anomalies |
| Discharge | $\Sigma_i(Area_i*Velocity_i)$ |
| Canopy Cover | ((number of densiometer points covered/96)<br>*100) |

Water chemistry independent variables were selected for further analyses using correlation matrices to avoid multicollinearity. Independent variables with correlation coefficients greater than 0.6 were removed. Independent water chemistry variables selected for further analyses, included mean dissolved oxygen, mean and maximum total suspended solids, mean total nitrogen, mean total phosphorus, mean atrazine, mean imidacloprid, and mean metalaxyl. The mean of TSS, nutrients, and pesticides, as well as the maximum value of total suspended solids were calculated from data collected during the 28 days before and 28 days after fish data collection at each sampling location.

**Linear mixed effects modeling**

Linear mixed effects modeling was used since data were collected at the same sampling locations repeatedly. This analysis incorporated the use of random effects which can account for influences not fully under my control (Sokal & Rohlf, 2012). To avoid multicollinearity only one substrate variable, one cover type variable, one insecticide, one herbicide, and one fungicide were included in the linear mixed effects modeling analysis. These variables were identified by conducting Pearson correlation tests for all possible pairs of independent variables (Appendix A to Appendix D). The variable that was most often correlated with other variables within its category (substrate, cover type, insecticide, herbicide, and fungicide) was chosen to be included in the linear mixed effect modeling analysis. The random effects chosen were site, season, site and season interaction, watershed, and site nested within year. Akaike information criterion with correction (AICc) was used to determine the best random effect to include into the analysis for each response variable. Models were determined to differ significantly if the change in AICc values (ΔAICc) was greater than 2. Akaike weights (Wi) were used to determine the probability that the model was best at predicting the response variable (Symonds & Moussalli, 2011).

Linear mixed effects model analyses were completed for each dependent variable (species richness, Shannon diversity index, fish density, and IBI). These models were constructed using all potential independent variables and any random effects if AICc indicated they had created a better model. Using a backward stepwise procedure, variables that did not significantly add to the model were removed manually yielding models for each dependent variable that only included those that added significantly to the model. This process included first removing variables with a p-value greater than 0.75 then rerunning the analysis. These steps were repeated with more-strict p-values (0.50, 0.25, 0.10, 0.05) until all variables left in the analysis were significantly adding to the model. If more than one significant model was found, they were compared using ΔAICc and Akaike weights (Wi), to determine the best model.

Response variables and all independent variables included in best final models for each fish metric were transformed to z-scores to obtain standardized coefficients. This transformation allowed coefficients to be compared as variables were inherently measured using different scales. The variable with the largest standardized coefficient in the model was assumed to be the greatest contributor to the overall model. Statistical analyses were conducted using RStudio version 1.1.463. Significance level for all tests was p-values less than 0.05.

# RESULTS

## Fish metrics

A total of 8,728 fish were caught in the SJR watershed. The most abundant fish species were creek chub (*Semotilus atromaculatus*), central mudminnow (*Umbra limi*), and mottled sculpin (*Cottus bairdii*). A total of 11,657 fish were caught in the UBWC watershed. The most abundant species were creek chub (*Semotilus atromaculatus*), Johnny darter (*Etheostoma nigrum*), and orangethroat darter (*Etheostoma spectabile*). Species richness ranged from 1 to 20 (median= 11) among sites in SJR. Species richness ranged from 1-14 (median= 7) among sites in UBWC. The Shannon Diversity Index was similar among watersheds (0.0-3.4), however, the median was slightly higher in SJR (1.72) than that of UBWC (1.3). Fish density ranged from 0.0 to 3.4 fish/m$^2$ (median= 0.4 fish/m$^2$) among sites in SJR. Fish density ranged from 1.0 to 7.2 fish/m$^2$ (median= 0.4 fish/m$^2$) among sites in UBWC. The IBI ranged from 16 to 46 (median= 32) among sites in SJR. The IBI had a range of 16 to 38 (median= 30) among sites in UBWC.

## Independent variables

Mean TSS had a range of 3.20 to 81.61 mg/L (median= 20.26 mg/L) among sites in SJR. Mean TSS had a range of 8.67 to 114.61 mg/L (median= 22.12 mg/L) among sites in UBWC. Maximum TSS had a range of 6.06 to 293.97 mg/L (median= 41.29 mg/L) among sites in SJR. Max TSS had a range of 14.03 to 829.29 mg/L (median= 80.82 mg/L) among sites in UBWC. The ranges of dissolved oxygen, total nitrogen, total phosphorus, imidacloprid, and metalaxyl measured among sites in SJR had a smaller range than the ranges measured in UBWC sites (Table 2). Atrazine concentrations among SJR sites and UBWC had a similar range (Table 2).

The ranges of percentage of silt and clay, percent leaf litter, cover type richness, percent canopy cover, and percentage of agricultural land use among sites in SJR were lower than the ranges measured in the UBWC sites (Table 2). The range measured for discharge was higher in the SJR sites than the UBWC sites (Table 2). The range for substrate richness was equal in both watersheds (Table 2).

Table 2: Median, minimum, and maximum values of fish community metrics and selected water chemistry and habitat variables from the Saint Joseph River Watershed, Indiana and Michigan and the Upper Big Walnut Creek Watershed, Ohio.

| | Saint Joseph River | | | Upper Big Walnut Creek | | |
|---|---|---|---|---|---|---|
| | Median | Min | Max | Median | Min | Max |
| Species Richness | 11.00 | 1.00 | 20.00 | 7.00 | 1.00 | 14.00 |
| Shannon Diversity | 1.81 | 0.00 | 2.65 | 1.25 | 0.00 | 3.45 |
| Fish Density (fish/$m^2$) | 0.30 | 0.01 | 1.43 | 0.38 | 0.01 | 7.20 |
| IBI | 30.00 | 20.00 | 40.00 | 30.00 | 16.00 | 38.00 |
| Dissolved Oxygen (mg/L) | 6.43 | 2.92 | 17.02 | 7.34 | 0.48 | 21.54 |
| Mean TSS (mg/L) | 20.26 | 3.20 | 81.61 | 22.12 | 8.67 | 114.61 |
| Max TSS (mg/L) | 41.29 | 6.06 | 293.97 | 80.82 | 14.03 | 829.29 |
| Total Nitrogen (mg/L) | 2.94 | 1.41 | 7.24 | 2.76 | 0.52 | 12.59 |
| Total Phosphorus (mg/L) | 0.01 | 0.00 | 0.12 | 0.13 | 0.05 | 0.81 |
| Atrazine (µg/L) | 0.70 | 0.01 | 7.82 | 0.47 | 0.00 | 7.74 |
| Imidacloprid (µg/L) | 0.06 | 0.01 | 0.52 | 0.03 | 0.00 | 26.47 |
| Metalaxyl (µg/L) | 0.01 | 0.00 | 0.04 | 0.01 | 0.00 | 16.09 |
| Discharge (cms) | 0.02 | 0.00 | 1.44 | 0.00 | 0.00 | 0.21 |
| %Silt/Clay | 35.42 | 0.00 | 91.67 | 33.33 | 0.00 | 100.00 |
| Substrate Richness | 3.00 | 2.00 | 5.00 | 3.00 | 1.00 | 4.00 |
| %Leaf Litter | 53.55 | 15.79 | 100.00 | 41.51 | 0.00 | 100.00 |
| Habitat Richness | 3.00 | 1.00 | 5.00 | 2.00 | 0.00 | 5.00 |
| %Canopy Cover | 0.00 | 0.00 | 22.92 | 4.72 | 0.00 | 88.31 |
| %Agriculture | 75.77 | 54.54 | 82.88 | 66.10 | 48.51 | 95.12 |

*Sample size SJR n=26; UBWC n=72.

The best identified random effect for the Shannon Diversity Index was site (Table 6). There were four significant models tested and two models best explained Shannon Diversity since their AICc values were lowest, however they were within 2 units of each other making them unable to be differentiated (Table 7). Both models included mean total suspended solids, maximum total suspended solids, total nitrogen, and atrazine, but one of the models also included the variable discharge. Based on Wi scores the model that was most likely to accurately predict the Shannon Diversity Index was model 1, which included five variables of maximum total suspended solids, mean total suspended solids, total nitrogen, atrazine, and discharge (Table 7). Model 1 had the highest K value and the lowest AICc score indicating it had a higher penalty imposed due to increased independent variables, but still had the greater AICc value (Table 7). Thus, I feel model 1 is a better model than model 2. In both of the models the highest contributing variable was maximum total suspended solids, which had a positive influence on the Shannon Diversity Index (Table 8). The standardized coefficient for mean total suspended solids was only 0.04 different from maximum total suspended solids meaning that it also was a high contributing variable that had a negative influence on the Shannon Diversity Index (Table 8).

The model that best explained fish density did not include a random effect (Table 9). There were ten significant models tested and based on AICc and Wi scores model 1 was best at explaining fish density (Table 10). Model 1 included discharge, the percentage of silt and clay, habitat richness, dissolved oxygen, and percent Canopy Cover. Model 1 had the highest k value indicating that it had higher penalties imposed due to a larger number of independent variables. Model 1 also had the lowest AICc score despite having higher penalties imposed. Thus, I feel that model 1 is the best model. The percentage of silt and clay was the highest contributing variable in the model for fish density and it had a negative influence (Table 11).

The model that best explained the IBI included the random effect of site (Table 12). Of the ten significant models tested, AIC scores indicated that model 1 best explained IBI (Table 13). This model included the percentage of silt and clay, dissolved oxygen, mean total suspended solids, and total phosphorus. Model 1 had the highest k value and lowest AICc score, which indicates model 1 as being the best model even though it was calculated with more severe penalties due to increased independent variables (Table 13). The percentage of silt and clay was the highest contributing variable to the model that best explained IBI (Table 14). The standardized coefficient for total phosphorous was 0.02 less than the percentage of silt and clay, which means that it is also a high contributing variable to the IBI response variable. The percentage of silt and clay and total phosphorus both had a negative influence on the IBI response variable (Table 14).

In summary, TSS and the percentage of silt and clay were the most influential independent variables to fish community integrity (Table 15). Discharge, which is present in three of the four models and total phosphorus was also influential to fish community metrics.

Table 3: Best identified random effect for fish species richness models from agricultural headwater streams in the Saint Joseph River Watershed, Indiana and Michigan, and the Upper Big Walnut Creek Watershed, Ohio. AICc, change in AICc, and Akaike weights are reported for each model. Bolded random effects indicated the best random effect having the lowest AICc scores.

| Random Effect | AICc | $\Delta_{AICc}$ | Wi |
|---|---|---|---|
| None | 504.02 | 14.34 | 0.00 |
| **Site** | **489.68** | **0.00** | **0.64** |
| Season | 507.03 | 17.35 | 0.00 |
| Watershed | 507.03 | 17.35 | 0.00 |
| Site, Season, and Watershed | 495.36 | 5.68 | 0.04 |
| Site and Season | 492.32 | 2.64 | 0.17 |
| Season and Watershed | 510.12 | 20.44 | 0.00 |
| Site and Watershed | 492.70 | 3.02 | 0.14 |
| Year/site | 499.85 | 10.17 | 0.00 |

Table 4: AICc, change in AICc, K, and Akaike weights for each model for species richness within agricultural headwater streams in the Saint Joseph River Watershed, Indiana and Michigan, and the Upper Big Walnut Creek Watershed, Ohio. Bolded models indicate those with the lowest AICc scores. Models with changes in AICc that are less than 2 cannot be distinguished.

| Models | Variables | AICc | $\Delta_{AICc}$ | Wi | K |
|---|---|---|---|---|---|
| **1** | **Mean TSS, Imidacloprid** | **472.20** | **1.32** | **0.33** | **5** |
| **2** | **Mean TSS, Imidacloprid, Discharge, Substrate Richness** | **470.88** | **0.00** | **0.64** | **7** |
| 3 | Imidacloprid | 482.52 | 11.64 | 0.00 | 4 |
| 4 | Mean TSS | 477.25 | 6.37 | 0.03 | 4 |

Table 5: Highest contributing independent variables for species richness. Coefficients and p-values for each independent variable within the two best models for fish species richness within agricultural headwater streams in the Saint Joseph River Watershed, Indiana and Michigan, and the Upper Big Walnut Creek Watershed, Ohio, 2017 and 2018. Bolded independent variables indicate the variable having the greatest standardized coefficient within each model.

| Models | Random Effect | Fixed Effects | Standardized Coefficients | p-value |
|---|---|---|---|---|
| **Species Richness Model 1** | Site | **Mean TSS** | **-0.231** | **0.0004** |
| | | Imidacloprid | -0.145 | 0.007 |
| **Species Richness Model 2** | Site | **Mean TSS** | **-0.278** | **<0.0001** |
| | | Imidacloprid | -0.147 | 0.006 |
| | | Discharge | 0.143 | 0.053 |
| | | Substrate Richness | -0.126 | 0.059 |

Table 6: Best identified random effect for Shannon Diversity Index models from agricultural headwater streams in the Saint Joseph River Watershed, Indiana and Michigan, and the Upper Big Walnut Creek Watershed, Ohio. AICc, change in AICc, and Akaike weights are reported for each model. Bolded random effects indicated the best random effect having the lowest AICc scores.

| Random Effect | AICc | ΔAICc | Wi |
|---|---|---|---|
| None | 174.32 | 28.36 | 0.00 |
| **Site** | **145.96** | **0.00** | **0.68** |
| Season | 177.33 | 31.37 | 0.00 |
| Watershed | 177.33 | 31.37 | 0.00 |
| Site, Season, and Watershed | 152.21 | 6.25 | 0.03 |
| Site and Season | 149.05 | 3.09 | 0.15 |
| Season and Watershed | 180.41 | 34.45 | 0.00 |
| Site and Watershed | 149.05 | 3.09 | 0.15 |
| Year/site | 161.96 | 16.00 | 0.00 |

Table 7: AICc, change in AICc, K, and Akaike weights for each model for Shannon Diversity Index within agricultural headwater streams in the Saint Joseph River Watershed, Indiana and Michigan, and the Upper Big Walnut Creek Watershed, Ohio. Bolded models indicate models with the lowest AICc scores. Models with changes in AICc that are less than 2 cannot be distinguished.
.

| Models | Variables | AICc | ΔAICc | Wi | K |
|---|---|---|---|---|---|
| **1** | **Max TSS, Mean TSS, Atrazine, Total nitrogen, Discharge** | **133.27** | **0.00** | **0.60** | **8** |
| **2** | **Max TSS, Mean TSS, Atrazine, Total nitrogen** | **134.37** | **1.10** | **0.34** | **7** |
| 3 | Max TSS, Mean TSS | 138.02 | 4.75 | 0.06 | 5 |
| 4 | Max TSS | 143.80 | 10.53 | 0.00 | 4 |

Table 8: Highest contributing variable for Shannon Diversity Index. Standardized coefficients and p-values for each independent variable within the two best models for Shannon Diversity Index within agricultural headwater streams in the Saint Joseph River Watershed, Indiana and Michigan, and the Upper Big Walnut Creek Watershed, Ohio, 2017 and 2018. Bolded independent variables indicate those having the greatest standardized coefficient within each model.

| Models | Random Effect | Fixed Effects | Standardized Coefficients | p-value |
|---|---|---|---|---|
| **Shannon Diversity Index Model 1** | Site | **Maximum TSS** | **0.592** | **<0.0001** |
| | | Mean TSS | -0.552 | 0.0003 |
| | | Atrazine | 0.202 | 0.008 |
| | | Total Nitrogen | -0.193 | 0.014 |
| | | Discharge | 0.156 | 0.054 |
| **Shannon Diversity Index Model 2** | Site | **Maximum TSS** | **0.549** | **0.0001** |
| | | Mean TSS | -0.504 | 0.0009 |
| | | Atrazine | 0.202 | 0.008 |
| | | Total Nitrogen | -0.184 | 0.02 |

Table 9: Best identified random effect for fish density models from agricultural headwater streams in the Saint Joseph River Watershed, Indiana and Michigan, and the Upper Big Walnut Creek Watershed, Ohio. AICc, change in AICc, and Akaike weights are reported for each model. Bolded random effects indicated the best random effect having the lowest AICc scores.

| Random Effect | AICc | $\Delta_{AICc}$ | Wi |
|---|---|---|---|
| **None** | **-89.17** | **0.00** | **0.60** |
| Site | -86.16 | 3.01 | 0.13 |
| Season | -81.72 | 7.45 | 0.01 |
| Watershed | -86.16 | 3.01 | 0.13 |
| Site, Season, and Watershed | -79.91 | 9.26 | 0.01 |
| Site and Season | -83.07 | 6.10 | 0.03 |
| Season and Watershed | -83.07 | 6.10 | 0.03 |
| Site and Watershed | -83.08 | 6.09 | 0.03 |
| Year/site | -83.54 | 5.63 | 0.04 |

Table 10: AICc, change in AICc, K, and Akaike weights for each model for fish density within agricultural headwater streams in the Saint Joseph River Watershed, Indiana and Michigan, and the Upper Big Walnut Creek Watershed, Ohio. Bolded models indicate those with the lowest AICc scores. Models with changes in AICc that are less than 2 cannot be distinguished.

| Models | Variables | AICc | ΔAICc | Wi | K |
|---|---|---|---|---|---|
| **1** | **%Silt/clay, Dissolved Oxygen, %Canopy cover, Discharge, Cover Type Richness** | **-107.19** | **0.00** | **0.80** | **7** |
| 2 | Discharge, %Silt/Clay, Cover Type Richness, Dissolved Oxygen | -101.78 | 5.41 | 0.05 | 6 |
| 3 | Discharge, %Silt/Clay, Cover Type Richness, %Canopy Cover | -98.15 | 9.04 | 0.01 | 6 |
| 4 | Discharge, %Silt/Clay, Dissolved Oxygen, %Canopy Cover | -101.96 | 5.23 | 0.06 | 6 |
| 5 | %Silt/Clay, Cover Type Richness, Dissolved Oxygen, %Canopy Cover | -101.67 | 5.52 | 0.05 | 6 |
| 6 | Discharge, %Silt/Clay, Cover Type Richness | -94.76 | 12.43 | 0.00 | 5 |
| 7 | Discharge, %Silt/Clay, Dissolved Oxygen | -98.99 | 8.20 | 0.01 | 5 |
| 8 | %Silt/Clay, Cover Type Richness, Dissolved Oxygen | -98.74 | 8.45 | 0.01 | 5 |
| 9 | Discharge, %Sit/Clay, %Canopy Cover | -94.54 | 12.65 | 0.00 | 5 |
| 10 | Cover Type Richness, %Canopy Cover | -94.63 | 12.56 | 0.00 | 4 |

Table 11: Highest contributing variable for fish density. Standardized coefficients and p-values for each independent variable within the best model for fish density within agricultural headwater streams in the Saint Joseph River Watershed, Indiana and Michigan, and the Upper Big Walnut Creek Watershed, Ohio, 2017 and 2018. Bolded independent variables indicate the variable having the greatest standardized coefficient within each model.

| Model | Random Effect | Fixed Effects | Standardized Coefficients | p-value |
|---|---|---|---|---|
| **Fish Density Model 1** | None | **%Silt/Clay** | **-0.445** | **<0.0001** |
| | | Dissolved Oxygen | -0.289 | 0.001 |
| | | %Canopy Cover | -0.283 | 0.007 |
| | | Discharge | -0.252 | 0.007 |
| | | Cover Type Richness | -0.253 | 0.008 |

Table 12: Best identified random effect for Index of Biotic Integrity models from agricultural headwater streams in the Saint Joseph River Watershed, Indiana and Michigan, and the Upper Big Walnut Creek Watershed, Ohio. AICc, change in AICc, and Akaike weights are reported for each model. Bolded random effects indicated the best random effect having the lowest AICc scores.

| Random Effect | AICc | ΔAICc | Wi |
|---|---|---|---|
| None | 599.05 | 0.54 | 0.28 |
| **Site** | **598.51** | **0.00** | **0.36** |
| Season | 602.06 | 3.55 | 0.06 |
| Watershed | 602.06 | 3.55 | 0.06 |
| Site, Season, and Watershed | 604.76 | 6.25 | 0.02 |
| Site and Season | 601.60 | 3.09 | 0.08 |
| Season and Watershed | 605.14 | 6.63 | 0.01 |
| Site and Watershed | 601.60 | 3.09 | 0.08 |
| Year/site | 602.10 | 3.59 | 0.06 |

Table 13: AICc, change in AICc, and Akaike weights for each model for Index of Biotic Integrity within agricultural headwater streams in the Saint Joseph River Watershed, Indiana and Michigan, and the Upper Big Walnut Creek Watershed, Ohio. Bolded models indicate those with the lowest AICc scores. Models with changes in AICc that are less than 2 cannot be distinguished.

| Models | Variables | AICc | ΔAICc | Wi | K |
|---|---|---|---|---|---|
| **1** | **%Silt/Clay, Total Phosphorus, Mean TSS, Dissolved Oxygen** | **576.15** | **0.00** | **0.59** | **7** |
| 2 | %Silt/Clay, Mean TSS, Total Phosphorus | 578.40 | 2.25 | 0.19 | 6 |
| 3 | Dissolved Oxygen, Mean TSS, Total Phosphorus | 580.64 | 4.49 | 0.06 | 6 |
| 4 | %Silt/Clay, Mean TSS | 583.03 | 6.88 | 0.02 | 5 |
| 5 | %Silt/Clay, Total Phosphorus | 579.80 | 3.65 | 0.10 | 5 |
| 6 | Dissolved Oxygen, Total Phosphorus | 583.53 | 7.38 | 0.01 | 5 |
| 7 | Mean TSS, Total Phosphorus | 583.63 | 7.48 | 0.01 | 5 |
| 8 | %Silt/Clay | 587.14 | 10.99 | 0.00 | 4 |
| 9 | Mean TSS | 588.28 | 12.13 | 0.00 | 4 |
| 10 | Total Phosphorus | 585.33 | 9.18 | 0.01 | 4 |

Table 14: Highest contributing variable for Index of Biotic Integrity. Standardized coefficients and p-values for each independent variable within the best model for index of biotic integrity within agricultural headwater streams in the Saint Joseph River Watershed, Indiana and Michigan, and the Upper Big Walnut Creek Watershed, Ohio, 2017 and 2018. Bolded independent variables indicate those having the greatest standardized coefficient within each model.

| Model | Random Effect | Fixed Effects | Standardized Coefficients | p-value |
|---|---|---|---|---|
| **Index of Biotic Integrity Model 1** | Site | **%Silt/Clay** | **-0.303** | **0.007** |
| | | Total Phosphorus | -0.281 | 0.003 |
| | | Mean TSS | -0.195 | 0.03 |
| | | Dissolved Oxygen | -0.172 | 0.033 |

Table 15: Influence of independent variables on models that were best at explaining the response variables of species richness, Shannon Diversity Index, fish density, and Index of Biotic Integrity. Bolded terms indicate the independent variable that had the greatest effect on each response variable. Plus (+) and minus (-) signs indicate what direction of influence the variable had on the model.

| Fish Metric | Independent Variable | Influence |
|---|---|---|
| **Species Richness Model 2** | **Mean TSS*** | - |
| | Imidacloprid | - |
| | Discharge | + |
| | Substrate Richness | - |
| **Shannon Diversity Index Model 1** | **Maximum TSS*** | + |
| | Mean TSS | - |
| | Atrazine | + |
| | Total Nitrogen | - |
| | Discharge | + |
| **Fish Density** | **Percent Silt/Clay*** | - |
| | Dissolved Oxygen | - |
| | Percent Canopy Cover | - |
| | Cover Type Richness | - |
| | Discharge | - |
| **Index of Biotic Integrity** | **Percent Silt/Clay*** | - |
| | Total Phosphorus | - |
| | Mean TSS | - |
| | Dissolved Oxygen | - |

# DISCUSSION

In order to improve conservation and restoration efforts of agricultural headwater streams, it is important for resource managers to understand the most important contributors to degraded biotic communities. Many research studies have shown that high concentrations of total suspended solids negatively impact fish communities (Bilotta & Brazier, 2008; Sanders, 2012; Smiley et al., 2008; Waters, 1995), but few have evaluated the relative effects among other agricultural pollutants and degraded habitat. The objective of this study was to determine if total suspended solids can better explain variation in fish community metrics than any other variable in agricultural headwater streams.

It was expected that total suspended solids would better explain variation in fish community metrics than other variables in agricultural headwater streams. However, results from my study indicate that models containing different types of independent variables best explain fish community structure. A similar finding was documented by (Smiley et al., 2009) who found that fish community structure within channelized agricultural headwater streams in Indiana and Ohio were best explained by a combination of nutrient, pesticide, and physiochemical variables. My study suggests that TSS and bed sediment composition are most important in explaining fish community structure in these headwater streams. Results also indicate that discharge and total phosphorus are influential independent variables.

Total suspended solids were the most influential variable for species richness and Shannon Diversity Index and improved the strength of the model for IBI. Previous research has found that total suspended solids at concentrations of 20 mg/L lead to increased foraging time in salmonid fishes and that concentrations of 25 mg/L causes mortality in eggs and larvae (Bilotta & Brazier, 2008). Additionally, concentrations of 100 mg/L have been documented to cause moderate gill damage in cyprinidae (Sutherland & Meyer, 2007), while exposures as high as 500 mg/L led to increased mortality of salmonid juveniles (Bilotta & Brazier, 2008; Sutherland & Meyer, 2007). Mean TSS in streams of SJR and UBWC ranged from 3.2 to 114.6 mg/L, which falls within the range of concentrations that is known to cause negative impacts to fishes. However, the maximum concentration of total suspended solids recorded in UBWC was 829.3 mg/L, which could cause significant harm to fishes.

In addition to direct effects, TSS may significantly affect the health of fishes through indirect impacts. Over half of all sampling sites had populations of fishes that comprised at least 30 percent insectivore fishes. When exposed to TSS concentrations of 8 mg/L, many macroinvertebrates experience higher drift, reduced reproduction, and lower population sizes, suggesting that they may be more sensitive to TSS than fishes (Bilotta & Brazier, 2008). Research has also concluded that increased turbidity and suspended solids increase foraging time for fishes (Bilotta & Brazier, 2008; Zamor & Grossman, 2007) at concentrations that were found in our study area. The combination of decreased availability of food and increased foraging time could be responsible for extirpation of some fish species. Research has shown that fishes avoid areas with high levels of turbidity and suspended solids (Wenger et al., 2017). Avoidance may explain the negative relationship between mean TSS, species richness, Shannon Diversity Index, and IBI in my study. However, maximum TSS had a positive association with Shannon Diversity Index, which was surprising given the high concentrations of maximum TSS measured in my sampling sites and the negative impacts to fishes shown in previous studies. The maximum suspended solids were determined by the taking the value of the maximum concentration measured within an eight-week period, therefore the length of time the stream was subjected to the maximum concentration is unknown. Given the duration of exposure was not considered maximum TSS, as measured in my study, may not be a good predictor of fish response variables. It is also important to note that multicollinearity was present in the best model for Shannon Diversity Index due to the collinearity present between mean TSS and maximum TSS indicated by the Pearson correlation coefficient of 0.85 (see Appendix E). The presence of multicollinearity could yield spurious results, which could be indicated by the positive association between maximum TSS and Shannon Diversity Index. It is recommended that future analysis of Shannon Diversity index with fishes not include both mean and maximum TSS.

The percentage of silt and clay in sediments was the most influential independent variable for fish density and IBI. Increased siltation can have a negative influence on the survivorship of eggs and larvae of some fishes (Soulsby et al., 2001). Also, a higher percentage of lower quality substrates on the stream bottom, such as silt and clay are often associated with tolerant fishes, while higher percentages of higher quality substrates on the stream bottom, such as gravel and cobble have been associated with greater species richness and abundance (Bouska & Whitledge, 2014; Schlosser, 1982). Larger substrate particle sizes have been associated with IBI metrics that positively influence the IBI score, while smaller particle sizes are associated with more tolerant species, which negatively affects IBI (D'Ambrosio et al., 2009). Results from my study are consistent with this negative association between increased silt and clay composition and fish community integrity.

Discharge is present in three of four models explaining fish community structure. While it is not the most influential variable, its persistence in the models makes it worthy of discussion. The results for discharge are somewhat ambiguous, because it was positively associated with species richness and Shannon Diversity Index, but negatively associated with fish density. Although studies have found a positive relationship between relative abundance and richness of fishes with increased discharge (Franssen et al., 2006; Sagawa et al., 2007; Schlosser, 1995), I could not find research literature that suggested a negative relationship between fish density and discharge.

Total phosphorus, while not an important independent variable in explaining other response metrics, its standard coefficient is only 0.02 less than the most influential variable for IBI (silt and clay). Smiley et al. (2009) found that within channelized agricultural headwater streams in Indiana and Ohio total phosphorus was positively associated with the percent of guarder-nest spawners and negatively associated with the percent guarder-substrate spawners and percent Percidae. These results from Smiley et al. (2009) suggest total phosphorus loads have an impact on the type of fishes inhabiting agricultural headwater streams. My study found a negative association between IBI and total phosphorus. Total phosphorus has been positively associated with a greater percentage of tolerant species (Meador & Frey, 2018). Increased loads of total phosphorus have been negatively associated with IBI, with the decrease in IBI being due to an increase in tolerant species (Marshall et al., 2008). The inclusion of percent tolerant species in the IBI calculation may explain the negative association between IBI and total phosphorus in my study.

In conclusion, total suspended solids are an important variable in explaining fish community richness and diversity in agricultural headwater streams in SJR and UBWC. However, the percentage of silt and clay is most important in explaining fish density and IBI. Therefore, no single independent variable was best at explaining fish community structure of SJR and UBWC, a result reported by previous research conducted in SJR and UBWC. I suggest that total suspended solids and bed sediments are both very important influences on fish community structure. These results support the use of conservation and restoration practices that reduce TSS and the amount of silt and clay present in bed sediments within agricultural headwater streams to increase fish community integrity.

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# APPENDIX A

Pearson correlation matrix for choosing the substrate type most correlated with the other substrates. Bold values indicate significant correlation.

| Substrate Pearson correlation matrix | | | | | |
|---|---|---|---|---|---|
| | Clay | Sand | Gravel | Cobble | Boulder |
| Silt | **-0.36** | **0.17** | **-0.53** | **-0.21** | -0.04 |
| Clay | | **-0.45** | **-0.17** | **-0.27** | **-0.25** |
| Sand | | | **-0.39** | -0.12 | 0.13 |
| Gravel | | | | 0.2 | -0.08 |
| Cobble | | | | | **0.59** |

# APPENDIX B

Pearson correlation matrix for choosing the cover type most correlated with the other cover types. Bold values indicate significant correlation.

| Cover type Pearson correlation matrix | | | | | |
|---|---|---|---|---|---|
| | Terrestrial Vegetation | Algae | Aquatic Plant | Small Woody Debris | Large Woody Debris |
| Leaf Litter | **-0.39** | **-0.33** | **-0.23** | **-0.31** | -0.13 |
| Terrestrial Vegetation | | **-0.25** | -0.09 | **-0.27** | -0.1 |
| Algae | | | -0.06 | 0.03 | -0.02 |
| Aquatic Plant | | | | -0.09 | -0.03 |
| Small Woody Debris | | | | | -0.04 |

# APPENDIX C

Pearson correlation matrix for choosing the herbicide most correlated with the other insecticides. Bold values indicate significant correlation.

| Herbicide Pearson correlation matrix | | | | | | | |
|---|---|---|---|---|---|---|---|
| | Alachlor (µg/L) | Simazine (µg/L) | Atrazine (µg/L) | Metribuzin (ug/L) | 24D (µg/L) | Mesotrione (µg/L) | S_Metolachlor (µg/L) |
| Acetochlor (µg/L) | 0.15 | 0.17 | **0.48** | **0.18** | **0.36** | 0.02 | 0.18 |
| Alachlor (µg/L) | | **0.54** | -0.12 | **-0.15** | 0 | -0.07 | **0.53** |
| Simazine (µg/L) | | | **0.16** | 0.05 | **0.22** | **0.18** | **0.88** |
| Atrazine (µg/L) | | | | **0.27** | **0.38** | **0.33** | 0.08 |
| Metribuzin (µg/L) | | | | | **0.29** | 0.05 | 0.04 |
| 24D (µg/L) | | | | | | 0.06 | 0.02 |
| Mesotrione (µg/L) | | | | | | | 0.1 |

# APPENDIX D

Pearson correlation matrix for choosing the Insecticide most correlated with the other herbicides. Bold values indicate significant correlation.

| Insecticide Pearson correlation matrix | | |
|---|---|---|
| | Clothianidin | Imidacloprid |
| Malathion | -0.1 | **0.73** |
| Clothianidin | | 0.02 |

# APPENDIX E

Pearson correlation matrix of all independent variables. Independent variables with a correlation coefficient of greater than 0.6 were excluded. Bolded values indicate correlation coefficients that are greater than 0.6.

Pearson correlation matrix of all independent variables.

| | %Silt/Clay | Substrate Richness | %Leaf Litter | Cover Type Richness | Temperature (°C) | Dissolved Oxygen (mg/L) | pH | Specific Conductivity (S/m) | Mean Turbidity (NTU) | Max Turbidity (NTU) |
|---|---|---|---|---|---|---|---|---|---|---|
| Discharge | -0.24 | 0.25 | -0.02 | 0.09 | -0.01 | 0.04 | 0.3 | -0.01 | -0.07 | -0.12 |
| %Silt/Clay | | -0.48 | 0.24 | 0.33 | -0.09 | 0.01 | -0.06 | 0.22 | 0.07 | -0.04 |
| Substrate Richness | | | -0.28 | -0.19 | 0.02 | 0.2 | 0.25 | -0.32 | -0.3 | -0.28 |
| %Leaf Litter | | | | 0.04 | -0.18 | -0.07 | -0.12 | 0.2 | 0.23 | 0.16 |
| Cover Type Richness | | | | | -0.13 | -0.03 | -0.02 | 0 | -0.13 | -0.19 |
| Temperature (°C) | | | | | | 0.11 | 0.12 | 0.27 | 0.06 | 0.06 |
| Dissolved Oxygen (mg/L) | | | | | | | **0.68** | -0.2 | -0.13 | -0.22 |
| pH | | | | | | | | 0.17 | -0.24 | -0.28 |
| Specific Conductivity (S/m) | | | | | | | | | 0.26 | 0.29 |
| Mean Turbidity (NTU) | | | | | | | | | | **0.92** |

| | %Canopy Cover | Catchment Size (km2) | %Agriculture | Mean TSS (mg/L) | Max TSS (mg/L) | Total Nitrogen (mg/L) | Total Phosphorus (mg/L) | Atrazine (µg/L) | Imidacloprid (µg/L) | Metalaxyl (µg/L) |
|---|---|---|---|---|---|---|---|---|---|---|
| Discharge | -0.04 | **0.78** | -0.02 | -0.06 | -0.13 | 0.09 | -0.24 | 0.27 | -0.07 | -0.07 |
| %Silt/Clay | -0.51 | -0.31 | 0.19 | 0.15 | -0.02 | 0.16 | 0.16 | 0.01 | 0.12 | 0.04 |
| Substrate Richness | 0.26 | 0.17 | -0.19 | -0.27 | -0.28 | -0.13 | -0.26 | 0.14 | -0.16 | 0 |
| %Leaf Litter | -0.18 | 0.03 | 0.14 | 0.28 | 0.16 | 0.08 | -0.03 | 0.15 | 0.27 | 0.14 |
| Cover Type Richness | -0.32 | 0.09 | 0.27 | -0.01 | -0.07 | 0.16 | -0.09 | 0.18 | -0.17 | -0.15 |
| Temperature (ºC) | 0.05 | 0.06 | 0.06 | 0 | -0.01 | -0.04 | -0.06 | -0.12 | 0.13 | 0.13 |
| Dissolved Oxygen (mg/L) | -0.07 | -0.01 | 0.01 | -0.09 | -0.22 | 0.25 | -0.07 | 0.19 | 0.1 | -0.03 |
| pH | 0.04 | 0.18 | -0.08 | -0.22 | -0.3 | 0.11 | -0.28 | 0.19 | 0.08 | -0.02 |
| Specific Conductivity (S/m) | -0.37 | 0 | 0.51 | 0.03 | 0.02 | 0.25 | 0.08 | -0.1 | 0.39 | 0.09 |
| Mean Turbidity (NTU) | 0.02 | -0.11 | -0.04 | **0.78** | **0.68** | 0.1 | 0.34 | 0.03 | -0.03 | 0.01 |
| Max Turbidity (NTU) | 0.08 | -0.13 | -0.09 | **0.61** | **0.7** | 0.02 | 0.32 | -0.09 | -0.01 | 0 |
| %Canopy Cover | | -0.03 | -0.41 | 0.13 | 0.29 | -0.32 | -0.12 | -0.14 | -0.03 | -0.03 |
| Catchment Size (km2) | | | 0 | -0.11 | -0.13 | 0.05 | -0.27 | 0.19 | -0.08 | -0.08 |
| %Agriculture | | | | -0.1 | -0.17 | 0.3 | -0.06 | 0.28 | 0.08 | -0.06 |
| Mean TSS (mg/L) | | | | | **0.85** | 0.04 | 0.2 | 0.06 | 0.02 | 0.02 |
| Max TSS (mg/L) | | | | | | -0.04 | 0.21 | -0.09 | 0.05 | 0.05 |
| Total Nitrogen (mg/L) | | | | | | | 0.2 | 0.45 | 0.28 | 0.21 |
| Total Phosphorus (mg/L) | | | | | | | | -0.16 | 0.03 | 0.17 |
| Atrazine (µg/L) | | | | | | | | | -0.06 | -0.11 |
| Imidacloprid (µg/L) | | | | | | | | | | 0.36 |

# APPENDIX F

Fish Community Composition

| Species | Family | Common Name | Trophic Classification | Tolerance | SJR | UBWC |
|---|---|---|---|---|---|---|
| *Labidesthes sicculus* | Atherinopsidae | Brook Sillverside | Insectivore | Intermediate | x | |
| *Catostomus commersonii* | Catostomidae | White Sucker | Detritovore | Tolerant | x | x |
| *Hypentelium nigricans* | Catostomidae | Northern hog sucker | Insectivore | Intolerant | x | |
| *Ambloplites rupestris* | Centrarchidae | Rock bass | Piscivore | Intermediate | x | |
| *Lepomis gibbosus* | Centrarchidae | Pumpkinseed | Insectivore | Intermediate | x | x |
| *Lepomis cyanellus* | Centrarchidae | Green Sunfish | Insectivore | Tolerant | x | x |
| *Lepomis macrochirus* | Centrarchidae | Bluegill | Insectivore | Intermediate | x | x |
| *Lepomis microlophus* | Centrarchidae | Redear sunfish | Insectivore | Intermediate | | x |
| *Micropterus punctulatus* | Centrarchidae | Spotted bass | piscavore | Intermediate | | x |
| *Micropterus salmoides* | Centrarchidae | Largemouth bass | Piscivore | Intermediate | x | x |
| *Cottus bairdii* | Cottidae | Mottled sculpin | Insectivore | Intolerant | x | |
| *Campostoma anomalum* | Cyprinidae | Central stoneroller | Herbivore | Intermediate | x | x |
| *Cyprinus carpio* | Cyprinidae | Common carp | Omnivore | Tolerant | x | x |
| *Ericymba buccata* | Cyprinidae | Silverjaw minnow | Insectivore | Intermediate | x | x |
| *Luxilus chrysocephalus* | Cyprinidae | Striped shiner | Insectivore | Intermediate | | x |
| *Luxilus cornutus* | Cyprinidae | Common shiner | Insectivore | Intermediate | x | |
| *Nocomis biguttatus* | Cyprinidae | Hornyhead chub | Insectivore | Intolerant | x | |
| *Notemigonus crysoleucas* | Cyprinidae | Golden Shiner | Omnivore | Tolerant | x | x |
| *Pimephales notatus* | Cyprinidae | Bluntnose minnow | Omnivore | Tolerant | x | x |
| *Pimephales promelas* | Cyprinidae | Fathead Minnow | Omnivore | Tolerant | x | x |
| *Rhinichthys atratulus* | Cyprinidae | Blacknose dace | Generalist Feeder | Tolerant | x | x |
| *Semotilus atromaculatus* | Cyprinidae | Creek chub | Generalist Feeder | Tolerant | x | x |
| *Esox americanus* | Esocidae | Grass pickerel | Piscivore | Intermediate | x | x |
| *Fundulus notatus* | Fundulidae | Blackstripe topminnow | Insectivore | Intermediate | x | |

| | | | | | | |
|---|---|---|---|---|---|---|
| *Ameiurus melas* | Ictaluridae | black bullhead | Insectivore | Intermediate | x | x |
| *Ameiurus natalis* | Ictaluridae | Yellow bullhead | Insectivore | Tolerant | x | x |
| *Ameiurus nebulosus* | Ictaluridae | Brown Bullhead | Insectivore | Tolerant | x | |
| *Noturus gyrinus* | Ictaluridae | tadpole madtom | Insectivore | Intermediate | x | |
| *Etheostoma flabellare* | Percidae | Fantail darter | Insectivore | Intermediate | x | x |
| *Etheostoma blennioides* | Percidae | Greenside darter | Insectivore | Intermediate | x | |
| *Etheostoma caeruleum* | Percidae | Rainbow darter | Insectivore | Intermediate | | x |
| *Etheostoma nigrum* | Percidae | Johnny darter | Insectivore | Intermediate | x | x |
| *Etheostoma spectabile* | Percidae | Orangethroat darter | Insectivore | Intermediate | x | x |
| *Perca flavescens* | Percidae | Yellow Perch | Insectivore | Intermediate | x | |
| *Percina caprodes* | Percidae | Logperch | Insectivore | Intermediate | x | |
| *Gambusia affinis* | Poeciliidae | Mosquitofish | Insectivore | Tolerant | | x |
| *Umbra limi* | Umbridae | Central Mudminnow | Insectivore | Tolerant | x | x |