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Long-Term Trends of Stream Fish Community Assemblages in Southern Missouri with Contemporary Land Use Impacts

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LONG-TERM TRENDS OF STREAM FISH COMMUNITY ASSEMBLAGES IN SOUTHERN MISSOURI WITH CONTEMPORARY LAND USE IMPACTS

A Masters Thesis
Presented to
The Graduate College of
Missouri State University

In Partial Fulfillment
Of the Requirements for the Degree
Master of Science, Biology

By
Stephanie Marie Sickler
August 2018

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Biology

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Master of Biology

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ABSTRACT

Stream fish communities in the Ozarks are structured via a number of different mechanisms, including basin, stream size, and human land use. The purpose of this study was to understand the structuring mechanisms of stream fish communities in southern Missouri. I compiled 48 years of historical fish collections performed by the Ichthyology class at Missouri State University consisting of 140 sites. I resampled 45 of these sites in summer of 2016. First, I tested whether communities are different between basins and stream size. Next, I tested associations of land use at three spatial scales to local fish communities. Last, I used historical collections to determine occupancy of species through time. Contemporary fish communities were used to answer basin, stream size, and land use structuring questions. Bray-Curtis dissimilarities demonstrated that communities in the modern data were different between basin, but not different in relation to stream size. I used Bray-Curtis again to calculate dissimilarity of land use composition at each spatial scale, then compared that to the communities with Mantel tests. Mantel tests showed that differences in land use were associated with differences in fish communities at all three spatial scales. Historical data were used to create logistic regressions for occupancy of each species to determine if presence is increasing or decreasing. Logistic regressions showed many species in decline, especially darters and minnows. This points to a need to more fully understand how fish communities in the Ozarks are impacted by human activities.

KEYWORDS: fish, land use, communities, agriculture, urbanization, spatial scale

This abstract is approved as to form and content

Sean P. Maher, PhD
Chairperson, Advisory Committee
Missouri State University

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Approved:

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Julie Masterson, PhD: Dean, Graduate College

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I was going to dedicate this thesis to someone, but in the end, I just did it for the halibut.

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OVERVIEW

The geomorphic history of Missouri has fostered the creation of three ecoregions, the Temperate Plains in the northern half of the state, the Ozark Highlands in the southwest, and the Mississippi Alluvial Basin in the southeast. The history of the Missouri Ozarks in particular has facilitated high levels of species diversity (Berendzen et al., 2010; Sievert et al., 2016). Glaciers extended as far south as central Missouri during the Pleistocene, and the northern boundary of the Ozark highlands mark their farthest southern extent (King, 1973; Berendzen et al., 2010; Sievert et al., 2016). Glaciers also constricted northern species towards the south, many of which found refuge in the Ozarks (Mayden, 1985; Berendzen et al., 2003, 2010; April et al., 2012). These glacial remnant species persist in the coolwater streams common in the Ozarks, and this is the mechanism behind why some species with most of their range in the Appalachian Mountains of northeastern US have disjunct populations in the Ozarks (King, 1973; Mayden, 1985; Berendzen et al., 2010; April et al., 2012; Sievert et al., 2016). Retreating glaciers also allowed southern species to expand north into the Ozarks (Berendzen et al., 2003; Sievert et al., 2016).

Geologically, streams across the Ozarks are all fairly similar, with high gradients (i.e. a steep slope), gravel/cobble substrate, low turbidity, and considerable groundwater inputs (Sievert et al., 2016). Despite these similarities, the three main basins in the Ozarks are highly disconnected, with one draining to the south, one to the north, and one to the west, and all are bounded by large rivers (the Arkansas River, Missouri River, and Mississippi River; (Sauer, 1920; King, 1973).

Large rivers serve as barriers to movement for fish that are adapted to survive in smaller streams, and the longitudinal change in stream size can structure assemblages by restricting movements of some species (Allan et al., 1997; Jackson et al., 2001; Allan, 2004; Dauwalter et al., 2008; Hitt & Angermeier, 2011). This effect results in a large number of species whose ranges are delineated by basin and they share a recent common ancestor (Goldstein & Meador, 2004; Grenouillet et al., 2004; Hoeinghaus et al., 2007). The stippled darter once was considered a single species, but recently was separated into three distinct species determined by watershed. The same is true for the *Luxilus* shiners, Ozark/knobfin sculpin, and the Ozark/Black River madtom. The result is that there are different species pools within each basin, with a number of endemic species.

Sources of impairment in this ecoregion include dams, urbanization, and agriculture. Dams are one of the main impacts for stream fishes in this region due to their effectiveness at disconnecting populations and the switch from a lotic to a lentic system (Warren, Jr. et al., 2000). Other impacts include different land use types. Agriculture and urbanization are present to a lesser degree than in Missouri's two other subregions, with about 50% of the Ozarks still reported as forested land (Owen et al., 2011; Sievert et al., 2016).

Southeast Missouri is a separate ecoregion (the Mississippi Alluvial Basin) from the Ozarks and has its own unique assemblage of species (Pflieger, 1997; Sievert et al., 2016). These unique assemblages provide an opportunity to study how they may respond differently to similar types of impacts. This region is where the Mississippi Alluvial Basin begins and fish communities in this region more closely match those found further south (Sievert et al., 2016). Physical traits of streams in this region are distinctly different from streams in the Ozarks, further supporting a different assemblage of species as compared to the Ozarks subregion (Sievert et al., 2016). Streams in this region have low gradients and many are channelized into ditches for agriculture. The substrate varies based on flow conditions, with fine silt occurring in slow-flowing streams, and faster flowing streams have sand and fine gravel (Sievert et al., 2016). Impacts in this region are largely related to high levels of agriculture. (Sievert et al., 2016) reported 83% of the Mississippi Alluvial Basin in Missouri to be cultivated.

Another consideration when analyzing fish communities is the potential for long-term change in communities. A particular species may be absent in an area due to a prior cause that is no longer actively present. Some impacts can have legacy effects that continue to impact streams long after the activity causing them has ceased (Jacobson et al., 1997; Harding et al., 1998). For example, Appalachian streams are still impacted by extreme sedimentation that occurred during forest removal and farming, even though much of that area is reforested (Hooke, 2000; James, 2013). Dams also can change fish assemblages in ways separate from surrounding land use, and they continue to impact streams as long as they are still present (Marchetti & Moyle, 2001; Mims & Olden, 2013). My study attempted to provide an image of how fish community assemblages are changing over a long-term scale by looking at 48 years of fish collection data, and how contemporary land use is affecting their structure in the present day.

CONTEMPORARY FISH COMMUNITIES AND THE EFFECTS OF LAND USE IN THE MISSOURI OZARKS

Introduction

Influences to stream fish assemblages are complex and multifaceted. In addition to geologic influences, one of the most important considerations to understanding contemporary fish community assemblages is anthropogenic land use. Land use can have varying effects depending on type and intensity, from increased sedimentation and nutrient levels to lowered baseflows (Allan, 2004; Gido et al., 2010). Further, land use impacts streams differently at various spatial scales. Lowered baseflows and increased stormflows can have catchment-wide impacts, while loss of leaf litter inputs and woody debris have more localized impacts (Blair, 1996; Allan et al., 1997).

There are a variety of different families of fish in the Ozarks. Cyprinids are the most diverse members of Ozark streams. A number of species can be found schooling together, avoiding competition by compartmentalizing where and what they feed on (Pflieger, 1997). The high oxygen, and clear water of Ozark streams creates idealized habitat for darters. Most darters inhabit riffles exclusively and inevitably disappear from streams without this habitat (Gelwick, 1990; Pflieger, 1997). Mosquitofish, silversides, and topminnows all prefer backwaters and areas with little flow, and are not usually present in streams that lack these areas (Pflieger, 1997; Giam & Olden, 2016). Sunfish, including black basses, prefer pools with structure and little current (Pflieger, 1997; Jackson et al., 2001). This separation is based on microhabitat and predation, explaining why surrounding land use has such pervasive effects on stream communities (Gelwick, 1990; Jackson et al., 2001; Giam & Olden, 2016).

Different types of land use often can cause similar impacts on streams, but the intensity of the impacts often differ (Figure 1; (Bain et al., 1988, 2012; Allan et al., 1997; Allan, 2004; Buck et al., 2004; Cervantes-Yoshida et al., 2015). The biggest impact of urbanization is hydrologic shifts, resulting in changes in the flow of streams (Henshaw & Booth, 2000; Allan, 2004; Vondracek et al., 2005; Matono et al., 2013; Cervantes-Yoshida et al., 2015). Many urban streams are channelized; the straightened channels and armored banks result in increased flow velocity and decreased residency time (Henshaw & Booth, 2000; Allan, 2004; Vondracek et al., 2005; Matono et al., 2013; Cervantes-Yoshida et al., 2015). Stormwater flow is released directly from roads, pavement, and rooftops into urban streams without being allowed to saturate into the soil, causing increased stormflows (Henshaw & Booth, 2000; Allan, 2004; Vondracek et al., 2005; Matono et al., 2013; Cervantes-Yoshida et al., 2015). Reduced residency time in turn results in lowered baseflows and increased habitat homogeneity, which work to reduce stream fish diversity (Henshaw & Booth, 2000; Allan, 2004; Vondracek et al., 2005; Matono et al., 2013; Cervantes-Yoshida et al., 2015; Perkin et al., 2017).

The main impact of agriculture is increased nutrient and sediment inputs (Henshaw & Booth, 2000; Allan, 2004; Buck et al., 2004; Holden, 2013). Increased nutrient levels result in increased algal growth, which is compounded by riparian removal, resulting in increased light levels (Allan et al., 1997; Henshaw & Booth, 2000; Allan, 2004; Buck et al., 2004; Vondracek et al., 2005; Holden, 2013; Cervantes-Yoshida et al., 2015). This causes a shift from allochthonous to autochthonous energy sources as the algal biomass increases (Allan et al., 1997; Henshaw & Booth, 2000; Allan, 2004; Buck et al., 2004; Vondracek et al., 2005; Holden, 2013; Cervantes-Yoshida et al., 2015).

Ecological processes occur at different scales along a stream and within a catchment and are impacted in numerous ways by land use activities (Figure 2; (Allan et al., 1997; Allan, 2004; Buck et al., 2004; Bain et al., 2012; Blevins et al., 2014; Dala-Corte et al., 2016). There are three main spatial scales considered by most studies. The local scale is the land use occurring close to the area sampled, usually within a few hundred meters. Riparian scale includes the land immediately surrounding the entire length of the stream, and at the catchment scale all land draining into the stream is included. Some effects, such as increased temperature, primarily impact streams at the local scale, while other effects, like sedimentation, are most strongly impact at the riparian scale, and catchment-wide effects include hydrologic alterations, nutrient enrichment, and channel form (Allan et al., 1997; Allan, 2004; Buck et al., 2004).

Land use has far-reaching impacts on aquatic communities and causes changes in fish species assemblages by altering habitat quality and food webs (Allan et al., 1997; Allan, 2004; Foley et al., 2005; Vondracek et al., 2005; Cervantes-Yoshida et al., 2015; Kim et al., 2015). Changes in the timing and volume of high flows can exclude sensitive species, especially during susceptible life stages, often extirpating all but the most rapid dispersers (Bain et al., 1988; Allan et al., 1997; Henshaw & Booth, 2000; Allan, 2004; Buck et al., 2004; Vondracek et al., 2005; Holden, 2013; Cervantes-Yoshida et al., 2015). Loss of these sensitive species results in an increase in tolerant and nonnative species (Henshaw & Booth, 2000; Allan, 2004; Vondracek et al., 2005; Matono et al., 2013; Cervantes-Yoshida et al., 2015). Sedimentation fills interstitial spaces in gravel, which has the dual effect of removing habitat for benthic species, and eliminating important substrate for gravel spawners (Allan et al., 1997; Henshaw & Booth, 2000; Allan, 2004; Buck et al., 2004; Vondracek et al., 2005; Holden, 2013; Cervantes-Yoshida et al., 2015). One example is the loss of riparian shade and increased nutrients from agricultural runoff has increased algae growth in streams, resulting in a switch from the stream relying on allochthonous energy sources to autochthonous energy (Allan et al., 1997; Allan, 2004; Buck et al., 2004; Vondracek et al., 2005; Holden, 2013). This switch has allowed the expansion and increased abundance of stonerollers across much of their range.

The goal of this study was to bring clarity to how land use is associated with fish assemblage structure in the Missouri Ozarks. First, I predicted that basin would have an important control over fish species distributions due to the geologic history of the Ozarks. I also hypothesized that land use would impact fish community structure differently due to the differing intensity of effects between different types of land use and the ability of certain species to tolerate specific conditions of impairment. Finally, I tested whether spatial scale would act as an important structuring mechanism due to the differences in impacts over a range of spatial scales.

Methods

I compiled historical data from the Ichthyology class collections at Missouri State University, beginning in 1970 and continuing through 2016. There are a total of 140 sites stretching across southern Missouri, and all sites occurred at bridges or access points. Township and Range (T/R) coordinates and road names were given for each site and these were used to locate and determine latitude and longitude coordinates for each

(Table 1). Sampling transects were not used for this, but time spent seining was recorded for most samples, and I therefore used the average time spent seining (1 hour) to control for sample effort when I resampled sites. Backpack shockers were also not used when resampling in order to keep samples comparable. However, I was unable to attain land use data covering the historical samples and therefore limited my analysis to the resampled data.

In summer 2016, I resampled 45 of the historical sites (Table 1), focusing on sites in the Ozarks subregion in southwest Missouri (Figures 3 and 4). Collection methods were intended to match the collection methods used in the historical data. Each site was kick and haul seined for approximately 1 hour. Some qualitative habitat data were recorded, as well as any signs of human activities (gravel mining, dams, livestock access). Prior approval for this project was obtained from the Missouri State University Institutional Animal Care and Use Committee (IACUC; 11 May 2016; approval #16-026.0). The Missouri Department of Conservation (MDC) also provided me with a collector's permit for sampling (8 April 2016; permit #16855).

All individuals captured were counted and identified to species. A total of 58 species in 12 families were caught (Table 2). A Bray-Curtis dissimilarity index was calculated to compare pairwise differences in abundance of the fish community at each site using the vegdist function in the vegan package (version 2.4-4) in R (v. 3.4.1; (Matono et al., 2013; Dala-Corte et al., 2016; Oksanen et al., 2017; R Core Team, 2017). Values for Bray-Curtis range from 0 to 1 with identical communities having a value of 0 and communities with no shared species having a value of 1 (Cervantes-Yoshida et al., 2015). The vegan package (version 2.4-4) was used to perform Adonis and betadisper tests, as well as non-metric multidimensional scaling (NMDS) to determine if fish communities were separating by basin or stream order (Hitt & Angermeier, 2011; Oksanen et al., 2017; R Core Team, 2017).

Non-metric multidimensional scaling is useful for visualizing and analyzing multidimensional data in fewer dimensions, typically two, and has several benefits compared with other ordination procedures (Oksanen et al., 2017). Most ordination techniques calculate a large number of axes and then display a subset of them, while NMDS returns a limited number of axes for display. The benefit to this is that there are no axes of variation that are not included in the visualization of the data. NMDS also works iteratively, while other methods calculate only a single solution (Legendre & Legendre, 1998; Quinn & Keough, 2002). Another benefit of NMDS is that it is not an eigenanalysis technique, resulting in the axes that do not represent decreasing amounts of variance (axis 1 represents the greatest amount of variance, axis 2 the next greatest, etc.). Therefore, NMDS plots can be rotated, centered, or inverted to fit any chosen configuration (Legendre & Legendre, 1998; Quinn & Keough, 2002). NMDS is well suited to a broad variety of data, including any distance measure, because it makes few assumptions about the nature of the data included. NMDS is also non-parametric, therefore not requiring data to follow a normal distribution.

The main drawback of using NMDS on my data is that it can fail to find the true best solution if it gets stuck on local minima (Legendre & Legendre, 1998; Quinn & Keough, 2002). The solution to this issue is to have random restarts, allowing the iterations to run through the data many times in order to give it a better chance of finding the true best solution (Legendre & Legendre, 1998; Quinn & Keough, 2002). Stress is the score given after the NMDS has run through all of the tries to show the goodness of fit between the dissimilarity index values and the reduced dimensions (Oksanen et al., 2017). Lower stress values show a better fit, with results below 0.2 preferred and a score below 0.1 is even better (Legendre & Legendre, 1998; Quinn & Keough, 2002).

Adonis and betadisper tests work together to determine if differences seen in data, such as my NMDS plots, are significant (Anderson, 2006; Anderson et al., 2006). Adonis is a permutational multivariate analysis of variance (PerMANOVA) that uses distance matrices to evaluate variance. Betadisper is a multivariate test for homogeneity of dispersion, which is an assumption when using Adonis. A non-significant betadisper result indicates a significant Adonis result is due to differences in composition between groups, and not due to differences in composition among groups (Anderson, 2006; Anderson et al., 2006).

A stream network for Missouri was created in ArcMAP version 10.5.1 from a 60 m digital elevation model (DEM) downloaded from the Missouri Spatial Data Information System (MSDIS) website. Sample locations were then plotted over the network. The Hydrology toolbox in ArcMAP was used to determine Strahler stream order for the created network (Figure 5; (Shreve, 1966).

Buffers were created to represent three spatial scales at each site. The local scale buffer for each site extended from the point location to 500 m upstream of the sample site, as well as extending 100 m out to either side of the stream. Riparian buffers extended 100 m out to each side of the stream and extended through the entire upstream network. For the catchment scale buffers, I again used the watershed tool in the Hydrology toolbox to delineate the entire upstream catchment of each sample site.

Land use data for 2016 were obtained from the USDA CropScape database (USDA National Agricultural Statistics Service, 2017). These data are stored as a 30 m raster file with each cell representing the majority land use within that pixel. The buffers were overlayed on the land use data to determine the land use for each spatial scale at each site using the raster package (2.5-8) in R (Hijmans et al., 2016; R Core Team, 2017).

Bray-Curtis dissimilarity and non-metric multidimensional scaling plots were used to analyze differences in land use composition at each of the three spatial scales (Hitt & Angermeier, 2011; Oksanen et al., 2017). Bray-Curtis dissimilarity works in the same way for land use composition as it did for the fish community composition, with each different land use value being treated as a different ‘species’. A value of 1 represents no overlap in land use types between two sites, whereas a value of 0 represent identical land use between the sites. Twenty land use types were included in the local scale composition, 40 at the riparian scale, and 42 at the catchment scale.

Mantel tests with 999 permutations were used to determine if differences in fish community assemblage correlated with differences in land use composition at each spatial scale (Oksanen et al., 2017). Mantel tests are permutation tests that compare correlation structure between two distance matrices to assess whether the observed correlation is different than expected at random.

Results

Resampled sites were located in three separate basins: White River basin (22 sites), Neosho River basin (17 sites), and Osage basin (6 sites; Figure 3; Table 1). One site was a second order stream and 10 were third order. Fourth and fifth order streams were the most common with 14 and 16, respectively. Sixth order streams were represented by 4 sites and there were no higher order streams sampled (Table 1).

Minnows and darters (Cyprinidae and Percidae) were the most diverse groups captured. Stonerollers (*Campostoma spp.*) were the most widespread and abundant fish captured. The *Luxilus* shiner group, duskystripe (*L. pilsbryi*), bleeding (*L. zonatus*), and cardinal (*L. cardinalis*), also were common and abundant in their respective basin (White, Osage, and Neosho, respectively). The only darter species caught at the majority of sites was the orangethroat darter (*Etheostoma spectabile*) and the only common sunfish species was the longear sunfish (*Lepomis megalotis*).

Some fish were not widespread but locally abundant when found. Southern redbelly dace (*Chrosomus erythrogaster*) were uncommonly found, but when captured were often the most abundant species. Sculpins, both banded (*Cottus carolinae*) and knobfin (*Cottus immaculatus*), were uncommon but abundant in locations where they were present. The northern hogsucker (*Hypentelium nigricans*) was the only common sucker species captured. Both topminnows (*Fundulus catenatus, F. olivaceous, F. notatus*) and western mosquitofish (*Gambusia affinis*) were found only at sites with backwater and slack flow areas.

Some fish were widespread but never captured in large numbers. Smallmouth bass (*Micropterus dolomieu*) were caught at a number of sites, but typically only a single specimen per site. Logperch (*Percina caprodes*) and greenside darters (*Etheostoma blennioides*) were common in larger streams, but not in the high numbers that orangethroat darters were caught in. Whitetail shiners (*Cyprinella galactura*) were common in the White River basin but did not occur in great abundance. Chubs (*Semotilus atromaculatus, Nocomis biguttatus, N. asper*) were also caught in low numbers at several sites.

There were a number of species that were rarely captured, including all catfish species (madtoms and bullheads), but slender madtoms (*Noturus exilis*) and yellow bullheads (*Ameiurus natalis*) were the most common catfish species recorded. Both white suckers (*Catostomus commersonii*) and redhorse (*Moxostoma spp.*) were rare and only captured in small numbers. The only exception to this was Bear Creek site 1, where 26 young of the year *Moxostoma spp.* were captured.

The Bray-Curtis dissimilarity values for the fish communities varied from 0.115 to 1.000. The NMDS plot (stress = 0.19, dimensions = 2, restarts = 20, distance = Bray; Figure 6) and the Adonis tests showed that fish communities are different among watersheds ($P = 0.001$, adjusted $P = 0.003$, $R^2 = 0.182$, permutations = 999). Further, the Betadisper test for homogeneity of dispersion was not significant ($P = 0.196$, permutations = 999).

NMDS (stress = 0.188, restarts = 20, dimensions = 2, distance = Bray; Figure 7) plots of fish communities with classification by stream order showed a high amount of overlap overall, but there was low overlap between third order and sixth order streams. Adonis tests comparing the communities against stream order were not significant ($P = 0.119$, $R^2 = 0.086$, permutations = 999) and the betadisper test was significant ($P = 0.001$, permutations = 999).

Generally, the predominant land use for site at each spatial scale was one of three classes: urban, pasture, and forest. The only exception to this was the James River site 13

(JA-13) at the local scale. This site was at Tailwaters Access below the dam for Springfield Lake and the lake represented most of the land use values at the local scale. NMDS showed that predominant land use was an effective method for grouping sites (Figs. 8, 9, and 10). However, the classification of predominant land use varied by scale for some sites, most often where Forest became Grass/Pasture when moving to broader scales. Urban sites were consistently classified similarly regardless of scale examined.

Plotting the community NMDS against the local land use NMDS showed that communities were grouping by land use, particularly urbanized sites (Figure 11). Mantel tests between the fish community dissimilarity matrix and the differences in land use composition at the local scale were significant (Mantel r = 0.099, *P* = 0.037, permutations = 999). Similarly, plotting the community NMDS against the riparian scale NMDS showed groups of communities based on predominant land use (Figure 12). Fish community differences were correlated to differences in land use composition at the riparian scale (Mantel r = 0.1801, *P* = 0.001, permutations = 999). Fish communities from sites with a forested riparian zone are fairly clustered, showing a difference in community between streams with a forested riparian zone and those without. Mantel tests showed that differences in fish communities were correlated to differences in land use composition at the catchment scale *(*Mantel r = 0.160, *P* = 0.002, permutations = 999). There were few sites at the catchment scale that were primarily forest. The majority of sites had pasture as the predominant land use at this scale (Figure 13). Urbanized sites again were very separate from sites with other predominant land use types.

There were a number of sites whose predominant land use changed as spatial scale increased (Table 3). Forest was most common at the local scale, as 28 sites were forested at the local scale and 11 sites were forested at the catchment scale. Pasture became increasingly common as spatial scale increased, with 8 sites at the local scale and 26 sites at the catchment scale. Most of the sites with urbanization as the predominant land use did not change with increasing spatial scales. There was only one site (GA-2) that was forest at the local scale and urbanized at larger spatial scales, and two sites that were urban at the local scale and switched to a different land use at larger spatial scales (CO-1 and BT-1). Galloway Creek site 2 was a short distance downstream from the two other sites on Galloway Creek that I sampled and showed a more diverse community than the two upstream sites. The two sites that were urban at the local scale and not at larger spatial scales both had communities more similar to non-urbanized sites. Wilson's Creek site 2 had predominantly grass/pasture at the local scale due to its location next to a large park and was predominantly urban at the riparian and catchment scale yet didn't show the same difference in community as the forested Galloway Creek site.

Discussion

Fish community composition in the Missouri Ozarks differed between basins, which was expected considering the geologic history and the disconnected nature of Ozark basins (Sauer, 1920; King, 1973; Berendzen et al., 2010). There are some species that are only found in specific basins, and others that can be found in all of the basins I sampled. The Neosho basin to the west has several species of Fundulid that are found in the western plains states, as well as in the prairie subregion in the northern half of Missouri (Sauer, 1920; King, 1973; Pflieger, 1997). The White River basin has several species that only occur within that basin (such as the whitetail shiner), except in the far eastern portion (Black, Current, and eleven-point Rivers) which were connected to the Osage basin at one point in the past, and therefore have some species that are common to that basin (Sauer, 1920; King, 1973; Pflieger, 1997; Dauwalter et al., 2008). Further, several evolutionary lineages have been described as separate species based on drainage (such as the *Luxilus* shiners, stippled darter, and Ozark sculpin) (Pflieger, 1997; Dauwalter et al., 2008).

When using stream order to represent stream size, there were no apparent differences between communities. However, there was little overlap in the NMDS plot between third and sixth order streams, suggesting that higher and lower stream orders may have separate communities. The organization of stream orders across the first dimension of the NMDS plot also points to a longitudinal gradient of communities as streams become larger (Figure 7). Because most of my sites were located within fourth and fifth order streams, the overall high amount of overlap, and subsequent Adonis and betadisper tests, suggest that such differences in streams were not influencing fish communities at my sites.

The predominant land use at all of my sites fell within one of three land use types at all spatial scales, except James River site 13 at the local scale, despite there being a wider variety of land use types found within all of the buffers. Land use in the Ozarks largely consists of pasture and forest. Urbanization is clustered around cities, with most of my urbanized sites being in Springfield.

Fish communities are affected by factors related to land use operating at multiple spatial scales (Bain et al., 2012; Jacquemin & Doll, 2014). Local scale land use appears to have a significant impact on communities, especially when considering distance from urbanization (Allan, 2004). Understanding how land use affects streams at the local scale is important because these processes can have a major impact on fish community composition (Blevins et al., 2014).

My urban sites at the local scale showed a distinctly different fish assemblage than those sites that were not urbanized. All three samples taken in Galloway Creek were predominantly urban at the riparian and catchment scale, but Galloway Creek site 2 was primarily forest at the local scale and showed a very different community compared to the other two Galloway Creek sites that were sampled upstream (GA-2 distance = 0.453 and 1.000 compared to GA-1 and GA-3, respectively). Galloway Creek site 2 was in the Springfield Conservation Nature Center and the fish community there more closely reflected the communities at sites that were predominantly forest than those of other urbanized sites. Butler and Coon Creeks were predominantly urban at only the local scale, with Butler Creek becoming primarily forested at larger spatial scales and Coon Creek becoming primarily pasture. Yet the fish community at both of these sites exhibited a structure more similar to sites that were urbanized at all spatial scales (BT-1 mean distance = 0.807; CO-1 mean distance = 0.787; mean of urbanized sites = 0.855).

A riparian zone that is predominantly forested has been shown to improve stream health and increase fish and macroinvertebrate diversity (Harding et al., 1998; Allan, 2004; Buck et al., 2004; Bain et al., 2012; Blevins et al., 2014). At the riparian scale, forested sites had different communities than urbanized and pasture sites. This is likely due to the presence of the forested riparian buffer zones, which was associated with higher diversity of fish and macroinvertebrate communities in other locations (Allan et al., 1997; Harding et al., 1998; Allan, 2004; Buck et al., 2004). Forested riparian zones are also important for controlling several of the impacts of intensive land use that occur at the catchment scale, as they can capture sediment and nutrients before they enter a stream and are important for erosion control (Allan et al., 1997; Allan, 2004; Buck et al., 2004).

The catchment scale also showed different communities based on predominant land use, supporting findings of several studies that show catchment scale to be important to the formation of fish communities (Harding et al., 1998). Plotting the community NMDS against land use NMDS at the catchment scale showed that communities in sites with predominately pasture were similar to those in forested sites (Figure 13). This is could be due to the method with which I've delineated these sites. Predominant land use may not be the best method for looking at large spatial scales, as there a number of land use types that get ignored. Figures 14, 15, and 16 show that some sites have large gaps representing land use types that were not one of the three predominant types included. This is very likely where other land uses, especially intensive agriculture, have particular importance (Vondracek et al., 2005).

Fish diversity tends to be greater in streams with a forested catchment than those with an agricultural catchment (Harding et al., 1998). Forested riparian buffers are important for trapping sediments in agricultural catchments, but their ability is limited (Vondracek et al., 2005). The effects of forest fragmentation are as important to aquatic systems as they are to terrestrial systems and while the influence of forest fragment size has been well studied in terrestrial systems, it has been largely ignored when considering aquatic systems (Harding et al., 1998; Vondracek et al., 2005).

Sampling efforts and methods have been shown to influence recorded fish biodiversity, which presents limits on the inference that could be made from my data

(Cao et al., 2002; Kennard et al., 2006; Okamura et al., 2018). Using a transect length of a set distance would have ensured a more thorough and even sampling of each stream than simply timing how long we seined. Backpack shocking also would have been more efficient, but the results of my samples would not have been comparable to the historical data. Additionally, a full habitat analysis and record of individual health status would provide a better idea of the stream conditions and health of the community, allowing us to further partition variation in the community data.

Future directions for research would include more in-depth studies on how much riparian buffer zones can stabilize stream conditions before their ameliorating effects are overwhelmed. More studies on how the proportions of different land use types can affect streams differently would also be a logical choice. My tests accounted for differences in composition of land use at each spatial scale, however my sites were grouped by predominant land use type in order to more easily visualize them. The drawback to grouping my sites in this manner is that sites with lower proportions of land use (e.g. the dominant type was 40% forest) were grouped with those that had the same land use in higher proportions (such as 80% forest; Figures 14, 15, and 16). This also made it more difficult to understand the impact of more intensive agriculture, such as row crops. These types of agriculture did not predominate at any of my sites but due to their intensive nature they can have a disproportionate impact on streams when compared to less intensive land uses, such as pasture (Allan, 2004; Buck et al., 2004; Vondracek et al., 2005).

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Table 1. Sites sampled, in alphabetical order, including basin and order of each stream. Sample number was given in the order streams were sampled.

Stream	Location ID	Sample Number	Latitude	Longitude	Basin	Stream Order
Bear Creek	BA-1	29	37.63107	-93.6164	Osage	4
Beaver Creek	BV-1	19	36.95836	-92.7642	White	4
Bee Creek	BE-1	33	36.52331	-93.0891	White	4
Bryant Creek	BR-1	18	36.8713	-92.4718	White	6
Bull Creek	BU-1	10	36.8178	-93.1803	White	5
Bull Creek	BU-4	34	36.73123	-93.1933	White	5
Butler Creek	BT-1	14	36.55169	-94.5004	Neosho	3
Center Creek	CE-1	13	37.1755	-94.4548	Neosho	6
Coon Creek	CO-1	20	37.35149	-94.2987	Neosho	4
Crane Creek	CR-1	27	36.924	-93.5891	White	5
Dry Branch	DR-1	21	37.27027	-94.3065	Neosho	5
Dry Branch	DR-2	22	37.26949	-94.3249	Neosho	5
Elkhorn Creek	EH-1	15	36.68863	-94.2406	Neosho	3
Fassnight Creek	FA-1	6	37.18648	-93.3175	White	2
Flat Creek	FL-1	26	36.73249	-93.6704	White	6
Galloway Creek	GA-3	2	37.14532	-93.2385	White	3
Galloway Creek	GA-1	3	37.12984	-93.2344	White	3
Galloway Creek	GA-2	4	37.12474	-93.2416	White	3
Hickory Creek	HI-1	23	36.85685	-94.3353	Neosho	3
Honey Creek	HY-1	16	37.07818	-93.855	Neosho	5
Indian Creek	IN-1	24	36.81564	-94.1995	Neosho	3
James River	JA-2	38	37.19217	-93.1284	White	5
James River	JA-12	39	37.14992	-93.2032	White	5
James River	JA-13	40	37.10528	-93.2661	White	5
James River	JA-1	41	37.18128	-93.1654	White	5
Jordan Creek	JO-2	5	37.19721	-93.3187	White	3
Jordan Creek	JO-1	7	37.19002	-93.3243	White	3
Little Sac River	LS-3	42	37.30868	-93.3839	Osage	5
McCarty Creek	MC-1	28	37.74816	-94.1493	Osage	4
N Fork Spring River	SP-7	31	37.28421	-94.4884	Neosho	6
N Fork Spring River	SP-14	32	37.28544	-94.3427	Neosho	5
Niangua River	NI-2	44	37.51978	-92.9836	Osage	5
Panther Creek	PA-1	25	37.84109	-93.619	Osage	4
Pearson Creek	PE-2	1	37.17764	-93.1983	White	4
Pearson Creek	PE-1	45	37.17244	-93.1965	White	4
Shoal Creek	SH-1	35	36.81953	-94.0497	Neosho	5
Shoal Creek	SH-2	36	36.91634	-94.1336	Neosho	5

Table 1 continued. Sites sampled, in alphabetical order, including basin and order of each stream. Sample number was given in the order streams were sampled.

Stream	Location ID	Sample Number	Latitude	Longitude	Basin	Stream Order
Spring River	SP-8	37	36.97382	-93.7985	Neosho	4
Spring River	SP-1	43	36.94857	-93.7938	Neosho	4
Swan Creek	SW-1	11	36.78741	-93.0595	White	4
Turkey Creek	TU-1	30	37.51855	-93.5945	Osage	4
White Oak Creek	WO-1	12	37.19468	-94.0946	Neosho	4
William's Creek	WL-1	17	37.10026	-93.8653	Neosho	5
Wilson's Creek	WI-1	8	37.18679	-93.3315	White	3
Wilson's Creek	WI-2	9	37.18873	-93.3655	White	4

Table 2. This table shows species captured in each of the three main basins.

Scientific Name	Common Name	White Basin	Osage Basin	Neosho Basin
Campostoma spp.	Stoneroller	1	1	1
Cyprinella galactura	Whitetail shiner	1	0	0
Cyprinella spiloptera	Spotfin shiner	0	1	0
Luxilus cardinalis	Cardinal shiner	0	0	1
Luxilus chrysocephalus	Striped shiner	1	1	1
Luxilus pilsbryi	Duskystripe shiner	1	0	0
Luxilus zonatus	Bleeding shiner	0	1	0
Lythrurus umbratilis	Redfin shiner	0	0	1
Nocomis asper	Redspot chub	0	0	1
Nocomis biguttatus	Hornyhead chub	1	1	0
Notropis boops	Bigeye shiner	0	1	0
Notropis nubilus	Ozark minnow	1	1	1
Notropis percobromus	Carmine shiner	1	1	0
Notropis telescopus	Telescope shiner	1	0	0
Chrosomus erythrogaster	Southern redbelly dace	1	1	0
Pimephales notatus	Bluntnose minnow	0	0	1
Semotilus atromaculatus	Creek chub	1	1	1
Hypentelium nigricans	Northern hogsucker	1	1	1
Moxostoma duquesnei	Black redhorse	0	0	1
Catostomus commersonii	White sucker	1	1	1
Moxostoma spp.	Redhorse sucker	0	0	1
Noturus exilis	Slender madtom	1	1	0
Noturus albater	Ozark madtom	1	0	0
Noturus miurus	Brindled madtom	0	0	1
Ameiurus natalis	Yellow bullhead	1	0	0
Oncorhynchus mykiss	Rainbow trout	1	0	0
Fundulus catenatus	Northern studfish	1	1	0
Fundulus olivaceous	Blackspotted topminnow	1	0	1
Fundulus notatus	Blackstripe topminnow	0	1	0
Gambusia affinis	Western mosquitofish	1	1	1
Labidesthes sicculus	Brook silverside	1	1	1
Cottus hypselurus	Ozark sculpin	0	1	0
Cottus immaculatus	Knobfin sculpin	1	0	0
Cottus carolinae	Banded sculpin	1	1	1
Ambloplites constellatus	Ozark bass	0	1	0
Lepomis macrochirus	Bluegill	1	1	1
Lepomis megalotis	Longear sunfish	1	1	1

Table 2 continued. This table shows species captured in each of the three main basins.

Scientific Name	Common Name	White Basin	Osage Basin	Neosho Basin
L. macrochirus X L. cyanellus	Bluegill X green sunfish	1	1	1
L. megalotis X L. cyanellus	Longear sunfish X green sunfish	1	0	0
L. macrochirus X L. megalotis	Bluegill X longear sunfish	0	1	1
Lepomis microlophus	Redear sunfish	1	0	0
Lepomis cyanellus	Green sunfish	1	1	1
Micropterus salmoides	Largemouth bass	1	1	1
Micropterus punctulatus	Spotted bass	1	1	1
Micropterus dolomieu	Smallmouth bass	1	0	1
Etheostoma flabellare	Fantail darter	1	1	1
Etheostoma blennioides	Greenside darter	1	1	1
Etheostoma spectabile	Orangethroat darter	1	1	1
Etheostoma caeruleum	Rainbow darter	1	0	1
Etheostoma punctulatum	Stippled darter	0	0	1
Etheostoma autumnale	Autumn darter	1	0	0
Etheostoma mihileze	Sunburst darter	0	1	0
Etheostoma juliae	Yoke darter	1	0	0
Etheostoma zonale	Banded darter	0	1	0
Percina caprodes	Logperch	1	0	1
Dorosoma cepedianum	Gizzard shad	1	1	0
Lepisosteus osseus	Longnose gar	1	0	0

Table 3. The predominant land use for each spatial scale at each site.

Stream	Location ID	Sample Number	Local Land Use	Riparian Land Use	Catchment Land Use
Bear Creek	BA-1	29	Forest	Pasture	Pasture
Beaver Creek	BV-1	19	Forest	Forest	Forest
Bee Creek	BE-1	33	Forest	Forest	Forest
Bryant Creek	BR-1	18	Forest	Forest	Forest
Bull Creek	BU-1	10	Forest	Forest	Forest
Bull Creek	BU-4	34	Forest	Forest	Forest
Butler Creek	BT-1	14	Urban	Forest	Forest
Center Creek	CE-1	13	Pasture	Pasture	Pasture
Coon Creek	CO-1	20	Urban	Pasture	Pasture
Crane Creek	CR-1	27	Forest	Pasture	Pasture
Dry Branch	DR-1	21	Pasture	Pasture	Pasture
Dry Branch	DR-2	22	Forest	Pasture	Pasture
Elkhorn Creek	EH-1	15	Pasture	Pasture	Pasture
Fassnight Creek	FA-1	6	Urban	Urban	Urban
Flat Creek	FL-1	26	Pasture	Pasture	Pasture
Galloway Creek	GA-3	2	Urban	Urban	Urban
Galloway Creek	GA-1	3	Urban	Urban	Urban
Galloway Creek	GA-2	4	Forest	Urban	Urban
Hickory Creek	HI-1	23	Pasture	Forest	Forest
Honey Creek	HY-1	16	Forest	Pasture	Pasture
Indian Creek	IN-1	24	Pasture	Pasture	Pasture
James River	JA-2	38	Forest	Forest	Forest
James River	JA-12	39	Forest	Forest	Pasture
James River	JA-13	40	Open Water	Forest	Pasture
James River	JA-1	41	Forest	Forest	Forest
Jordan Creek	JO-2	5	Urban	Urban	Urban
Jordan Creek	JO-1	7	Urban	Urban	Urban
Little Sac River	LS-3	42	Forest	Forest	Pasture
McCarty Creek	MC-1	28	Forest	Forest	Forest
N Fork Spring River	SP-7	31	Forest	Pasture	Pasture
N Fork Spring River	SP-14	32	Forest	Pasture	Pasture
Niangua River	NI-2	44	Forest	Forest	Pasture
Panther Creek	PA-1	25	Forest	Forest	Pasture

Table 3 continued. The predominant land use for each spatial scale at each site.

Stream	Location ID	Sample Number	Local Land Use	Riparian Land Use	Catchment Land Use
Pearson Creek	PE-2	1	Forest	Pasture	Pasture
Pearson Creek	PE-1	45	Forest	Pasture	Pasture
Shoal Creek	SH-1	35	Pasture	Pasture	Pasture
Shoal Creek	SH-2	36	Forest	Pasture	Pasture
Spring River	SP-8	37	Forest	Pasture	Pasture
Spring River	SP-1	43	Forest	Pasture	Pasture
Swan Creek	SW-1	11	Forest	Forest	Forest
Turkey Creek	TU-1	30	Forest	Pasture	Pasture
White Oak Creek	WO-1	12	Forest	Pasture	Pasture
William's Creek	WL-1	17	Forest	Pasture	Pasture
Wilson's Creek	WI-1	8	Urban	Urban	Urban
Wilson's Creek	WI-2	9	Pasture	Urban	Urban

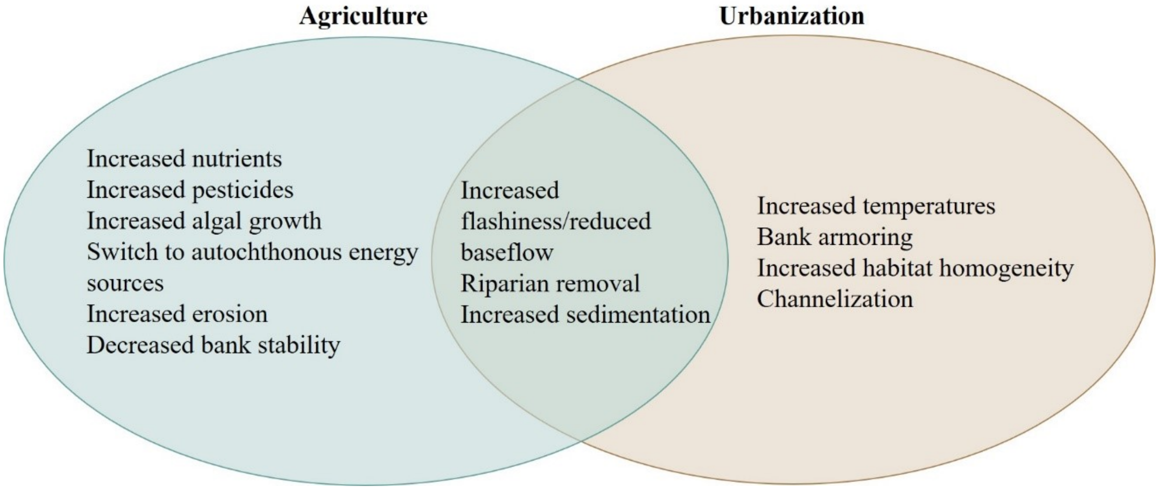

Figure 1. Different land uses can impact systems in very different, or very similar, ways. For example, even though increased flashiness is an issue for both agriculture and urbanization, flashiness is often the major issue related to urbanization, while the main concern with agriculture is typically increased nutrient loading (Allan et al., 1997; Allan, 2004; Buck et al., 2004; Bain et al., 2012; Cervantes-Yoshida et al., 2015).

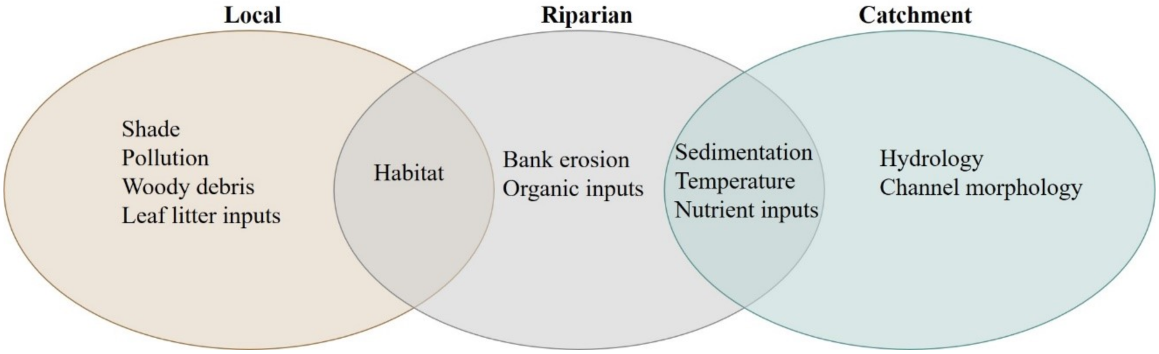

Figure 2. This figure shows how certain land use effects can cause impacts over different spatial scales. Some can impact streams at multiple spatial scales. The riparian scale serves as an important connector between local and riparian scale impacts.

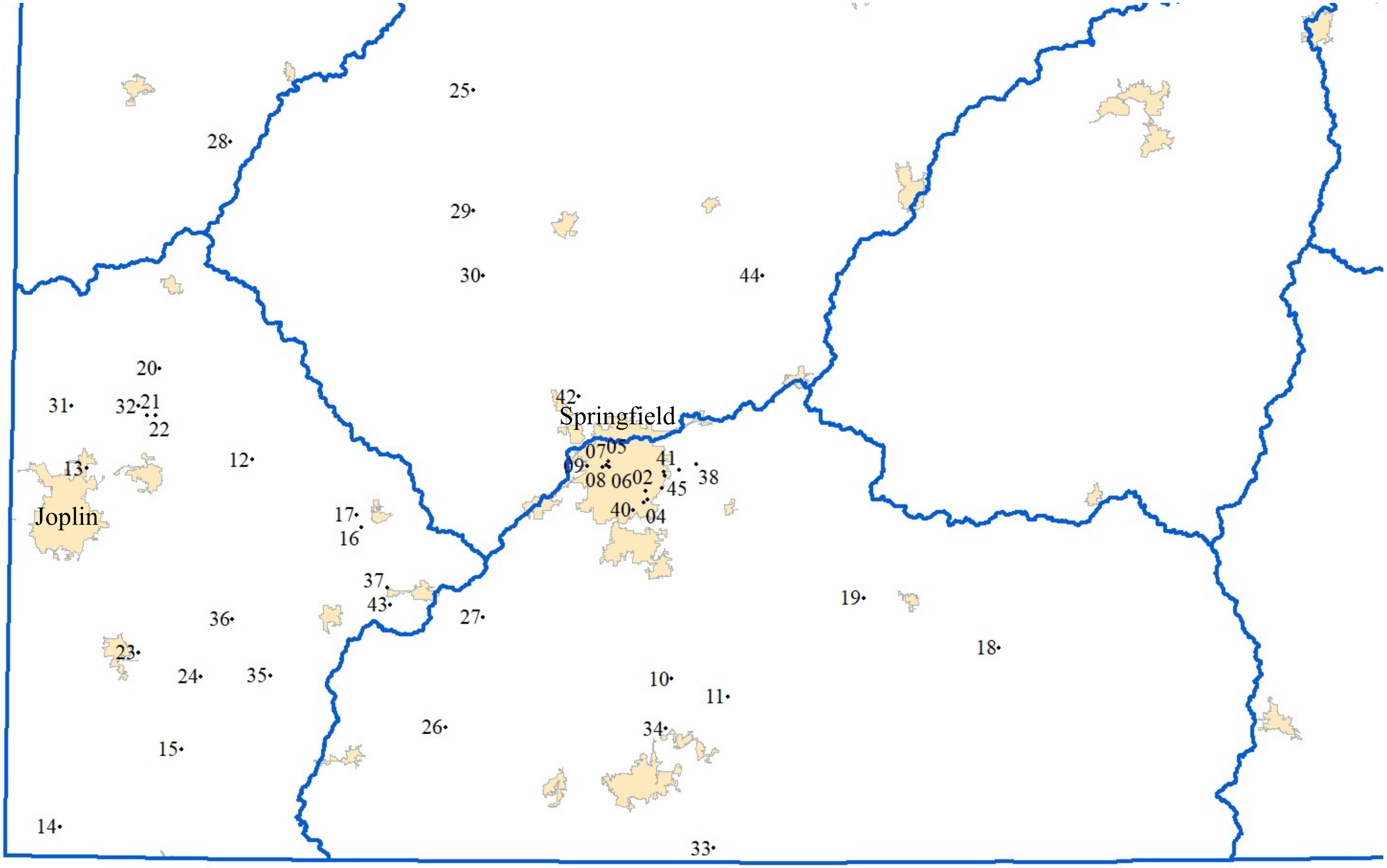

Figure 3. This map shows southwest Missouri with site locations marked as points. Basins are outlined in blue and towns shown in tan. Numbers match sample number in Tables 1 and 2.

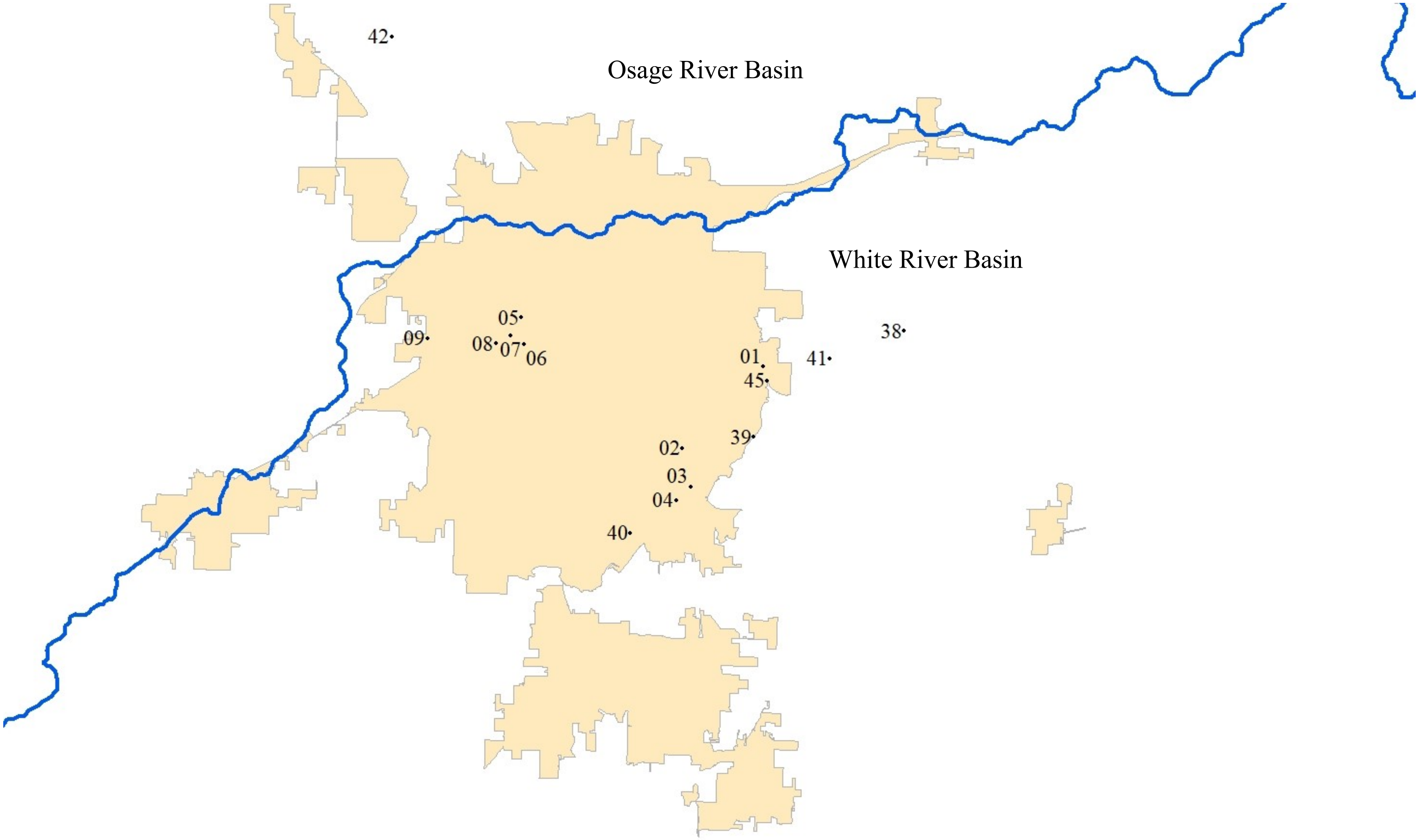

Figure 4. Map of sites within and around Springfield. The blue line represents the boundary between the Osage and White basins.

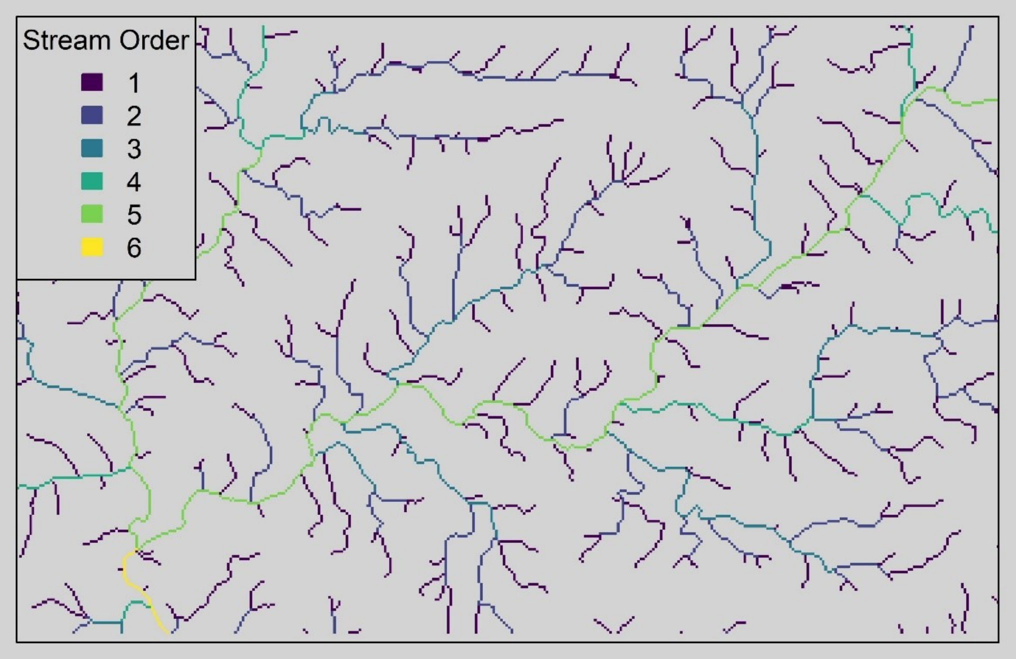

Figure 5. Part of the stream network, showing Strahler stream order that was created using the Hydrology toolbox in ArcMAP. Streams in the sample data ranged from second to sixth order.

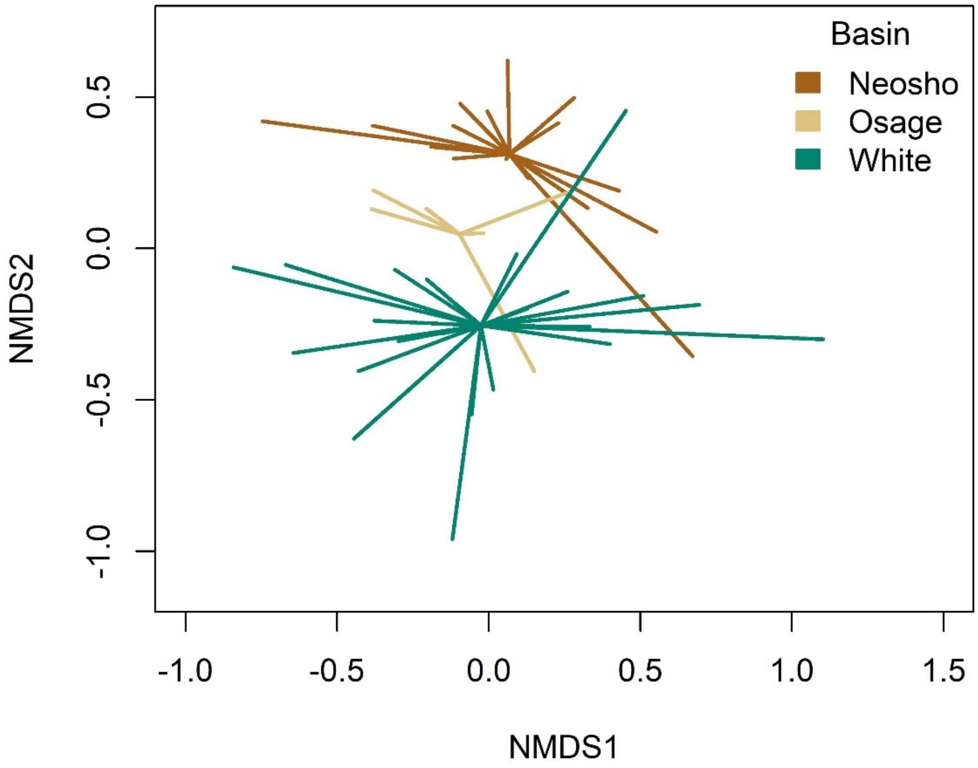

Figure 6. Fish communities in the Ozarks are different based on the basin sampled from (stress = 0.19, dimensions = 2, restarts = 20, distance = Bray). The second dimension of the NMDS plot shows the separation of the groups, and subsequent Adonis and betadisper tests supported this conclusion (Adonis *P* = 0.001, adjusted *P* = 0.003, R^2 = 0.182, permutations = 999, betadisper *P* = 0.196, permutations = 999).

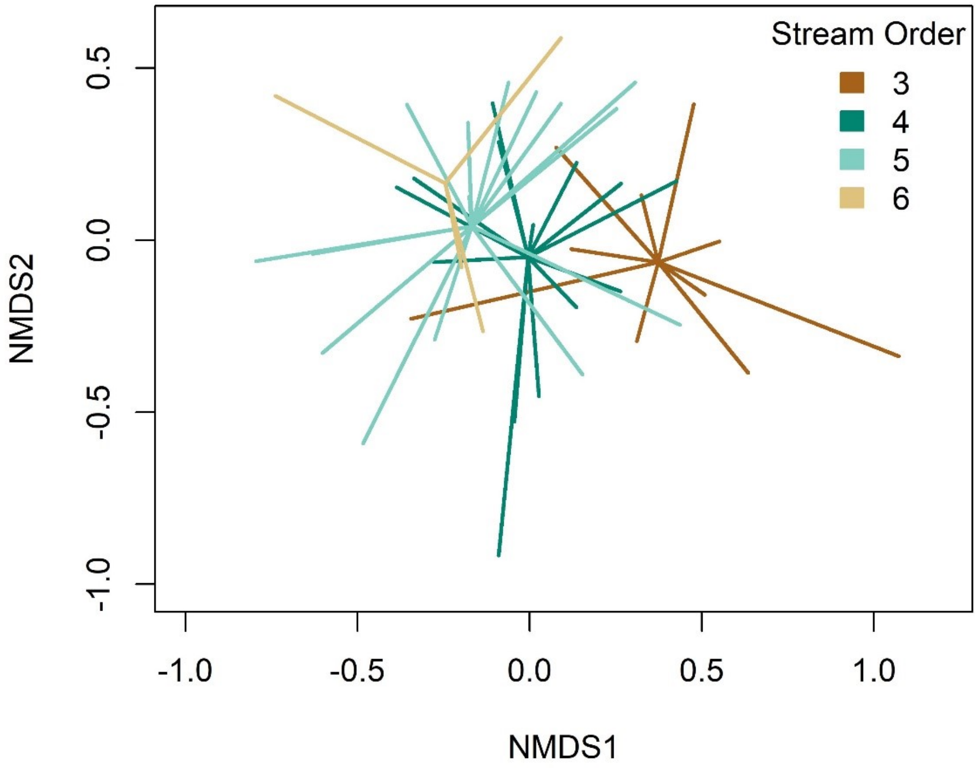

Figure 7. Overall the fish communities are not separating by stream size, but there is some separation between third and sixth order streams (stress = 0.19, dimensions = 2, restarts = 20, distance = Bray). The gradient of stream size groups across the NMDS plot follows the gradient of community shifts seen as streams get larger, but the small number of stream orders sampled in my data do not show a significant change (Adonis *P* = 0.119, R^2 = 0.086, permutations = 999, betadisper *P* = 0.001, permutations = 999).

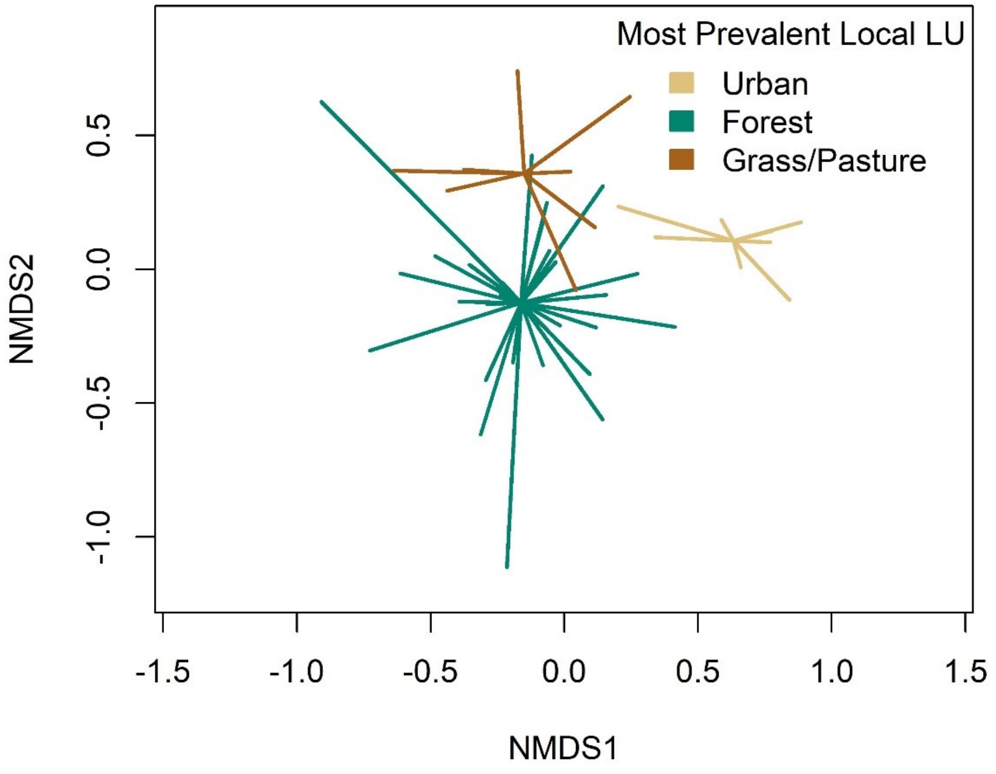

Figure 8. At the local scale sites clearly separated by most prevalent land use (stress = 0.21, dimensions = 2, restarts = 75, distance = Bray). JA-13 (open water) is not included on this plot.

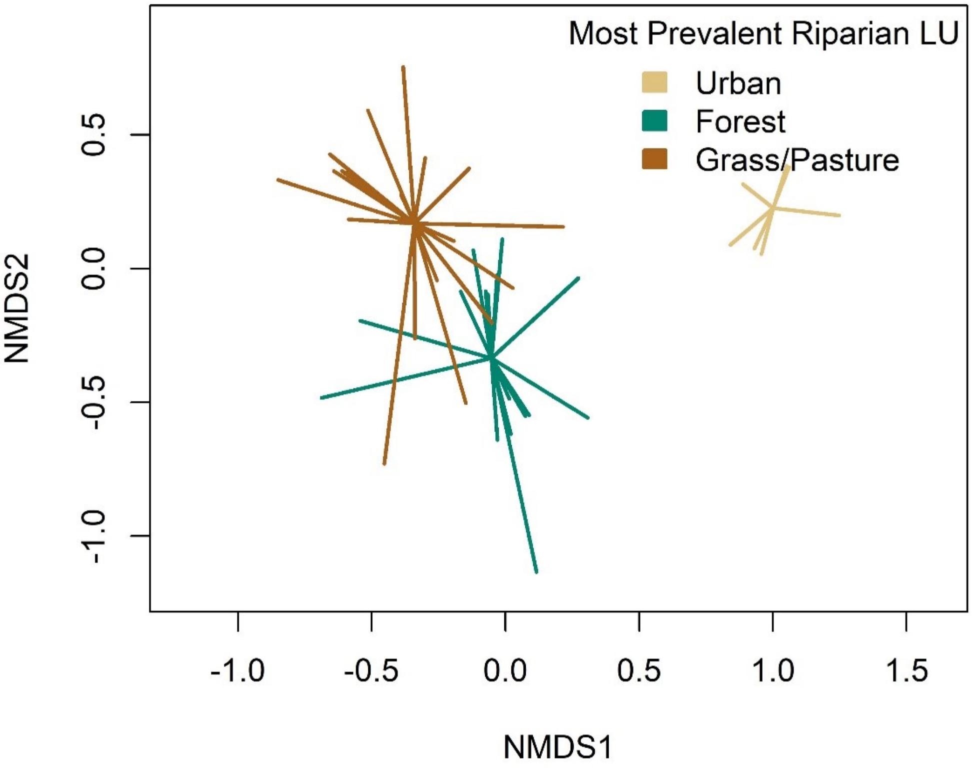

Figure 9. Land use at the riparian scale also separated by predominant land use (stress = 0.12, dimensions = 2, restarts = 20, distance = Bray). Urbanized sites are especially distinct.

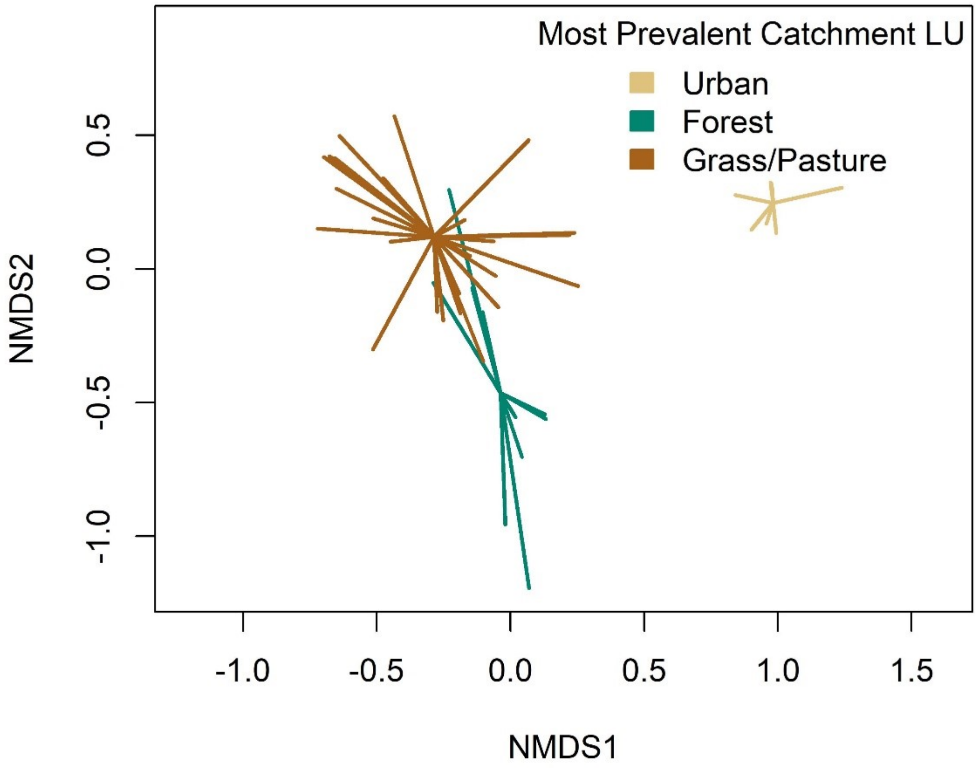

Figure 10. The catchment scale land use shows similar differences as seen at the riparian scale, with urbanized sites being more distinctly different than forested and pasture sites stress = 0.11, dimensions = 2, restarts = 87, distance = Bray).

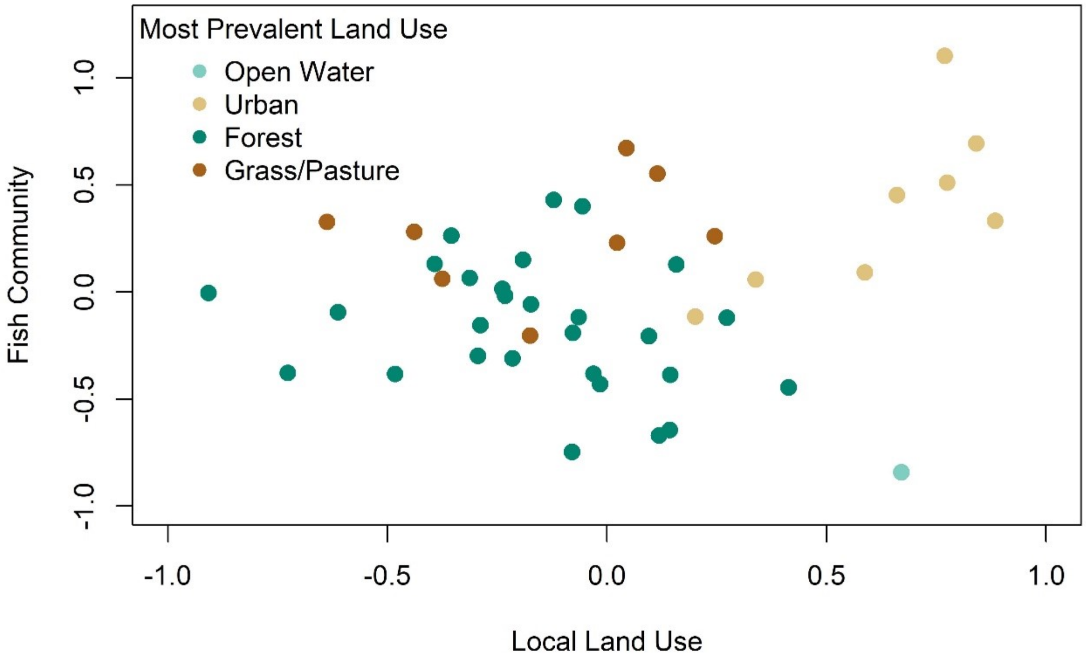

Figure 11. When plotting the fish community NMDS against the local land use NMDS there is a relationship between community and land use. Mantel tests show the fish community is significantly correlated with land use composition at the local scale *(P* = 0.037, r = 0.099, permutations = 999). The two sites showing predominately urban land use with communities more similar to forested and pasture sites are only urban at the local scale. James River at Tailwaters Access is the only site that was dammed at the sample location (predominately open water) and shows a different community to all other sites.

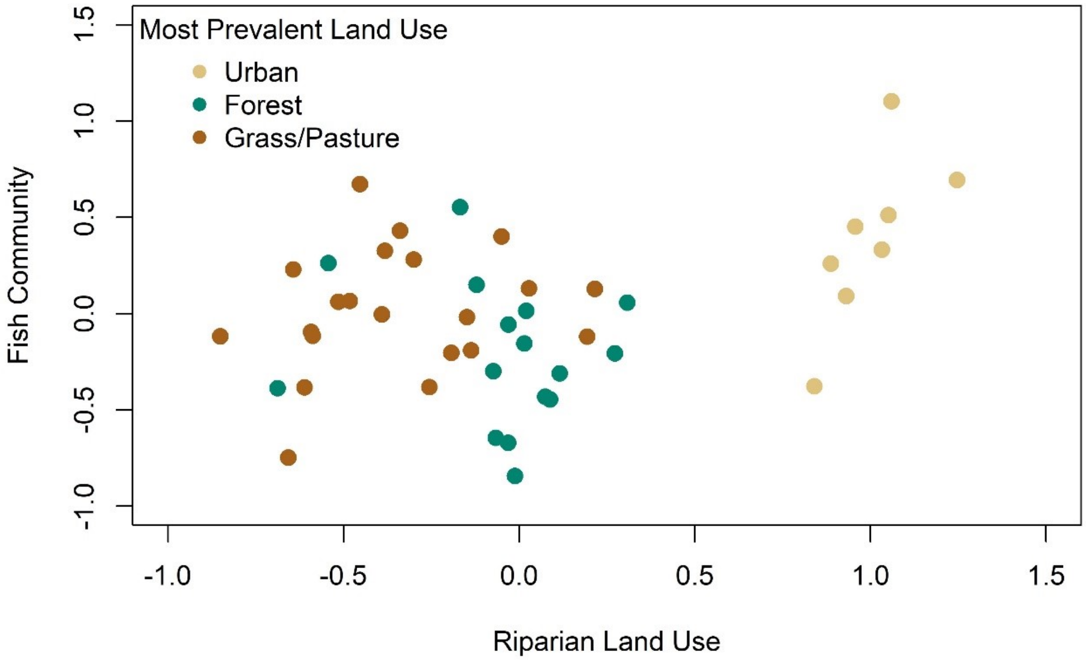

Figure 12. Plotting the community NMDS against the land use NMDS for the riparian scale also shows differences in community based on land use. Mantel tests show fish communities and land use at the riparian scale were significantly correlated *(P* = 0.001, r = 0.181, permutations = 999). The urban site showing a community more similar to forested sites is Galloway Creek at the Springfield Nature Center, which was forested at the local scale.

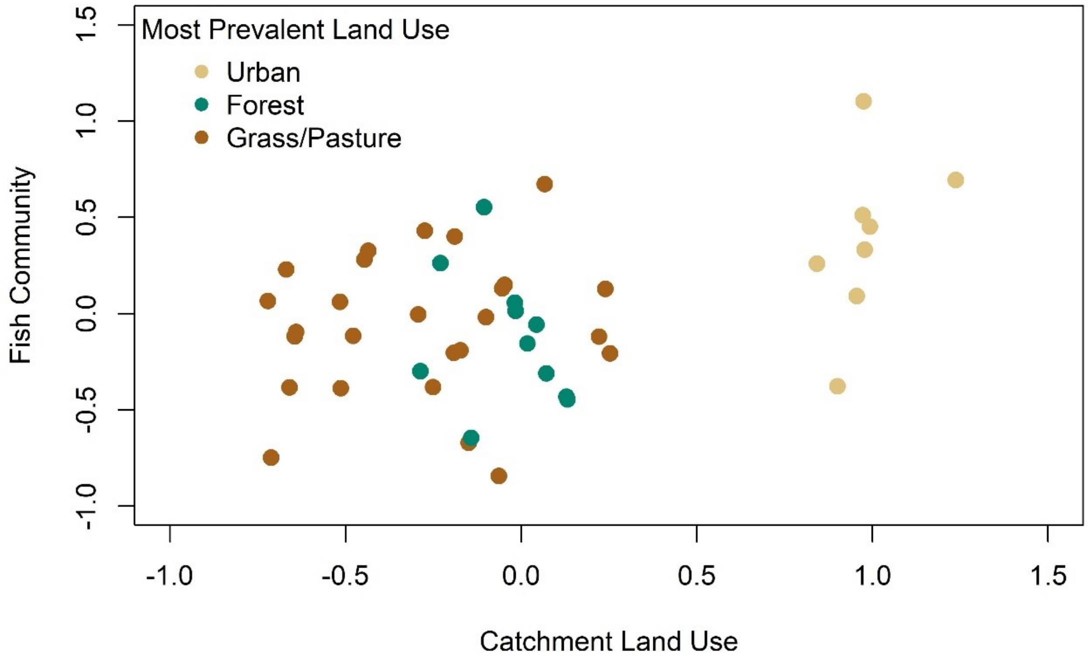

Figure 13. Comparing the fish community NMDS with the Catchment scale land use NMDS also showed differences between community and land use. Fish communities correlated significantly with land use composition at the catchment scale with Mantel tests ($P = 0.002$, r = 0.160, permutations = 999). Few sites at this scale were forested, and many sites included land use types other than the three predominant land uses at this scale (especially row crops).

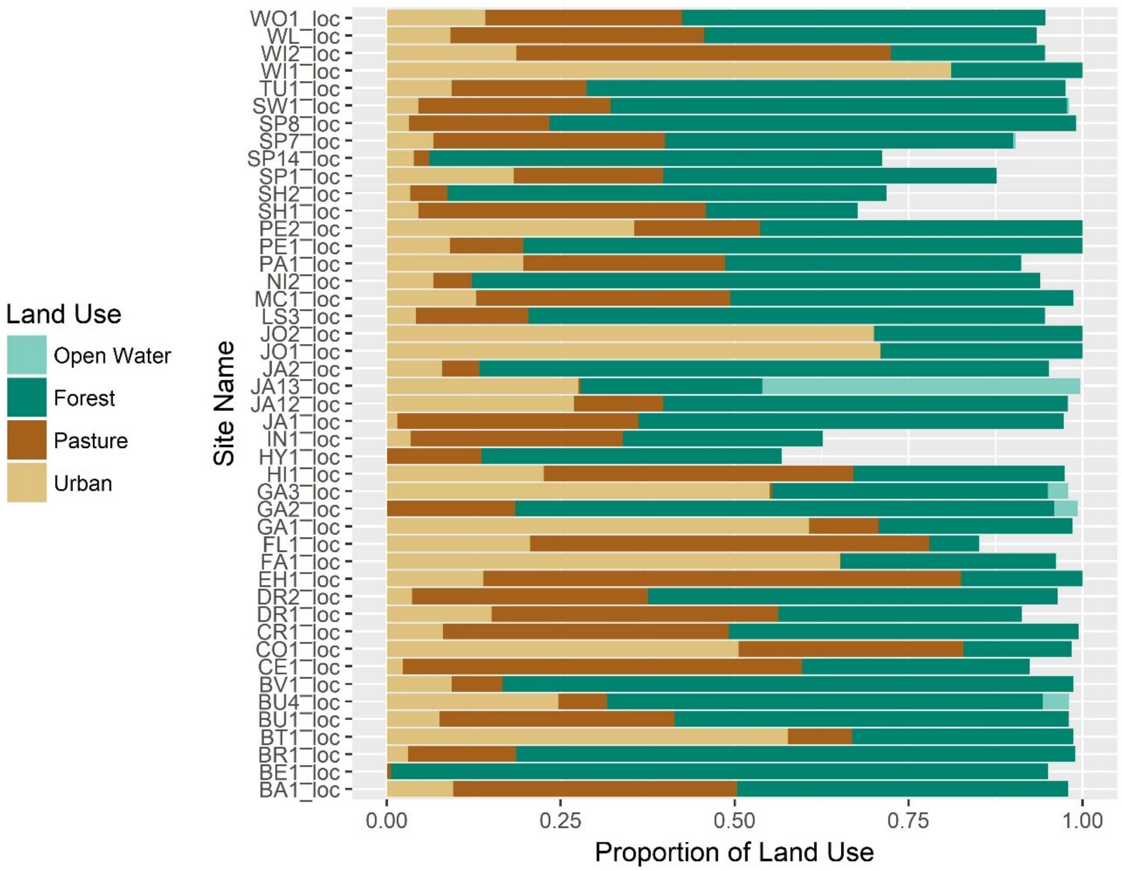

Figure 14. Proportions of the most dominant land use types at the local scale. Land use types not included in this plot were mainly intensive agriculture, such as row crops. The majority of sites at this scale are forested.

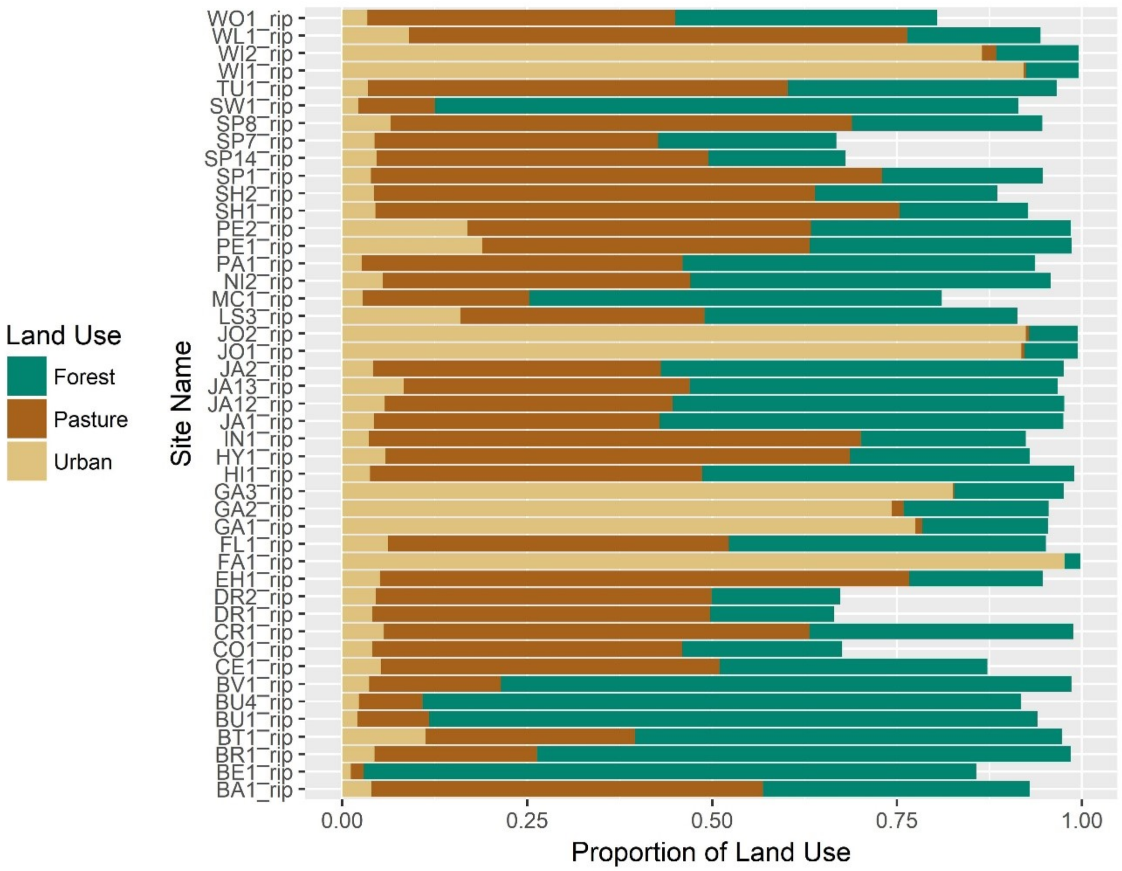

Figure 15. Proportions of the most dominant land use types at the riparian scale. Open water is no longer predominant for any site at this scale. Forest and pasture sites are more equal in proportions.

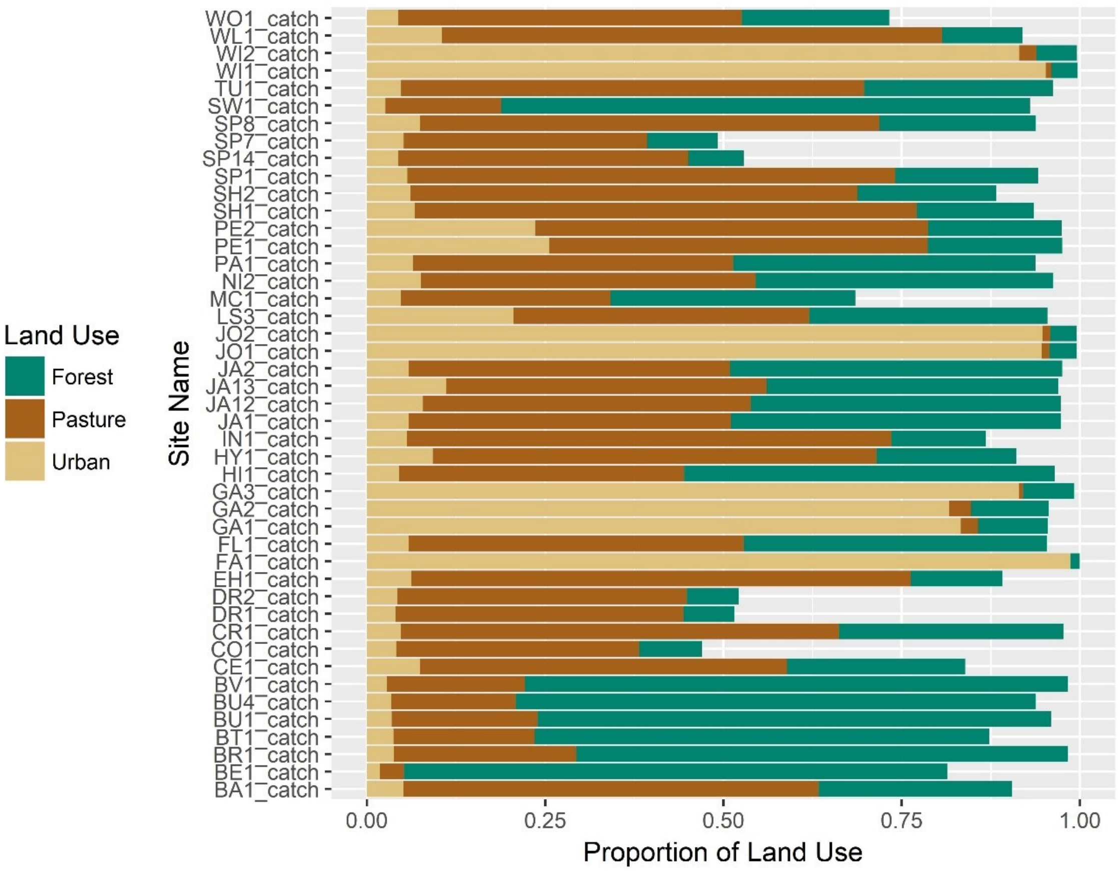

Figure 16. Proportions of the most dominant land use types at the catchment scale. Agriculture is more commonly found in sites at this scale and pasture is the predominant land use at the majority of sites. Urban sites have largely remained urban through all spatial scales.

LONG TERM TRENDS IN FISH COMMUNITIES IN SOUTHERN MISSOURI

Introduction

Aquatic ecosystems are some of the most under threat from human disturbance and changes in fish communities can provide insight into what efforts are needed to protect these systems (Adamski et al., 1995; Jacobson et al., 1997; Owen et al., 2011; Buckwalter et al., 2018). It is becoming more and more important to understand long-term trends in fish communities as aquatic systems become more heavily impacted by human disturbance (Warren, Jr. et al., 2000). Missouri has a range of impacts to streams, including dams, agriculture, and urbanization (Smart et al., 1985; Martin & Pavlowsky, 2011; Owen et al., 2011). These disturbances cause a variety of effects on streams, including sedimentation, hydrologic shifts, and changes in flow (Allan, 2004; Anderson et al., 2006; Bain et al., 2012). These changes in turn cause shifts in fish communities. Sedimentation can eliminate sensitive species and gravel spawning fishes (Zweig & Rabeni, 2001; Sutherland et al., 2002). Hydrologic shifts are especially capable of extirpating fish during sensitive life stages, such as larvae and eggs (Yang et al., 2008; Neufeld et al., 2018).

These effects hold especially true for areas with high species diversity or extensive amounts of human disturbance (Blair, 1996; Warren, Jr. et al., 2000). A number of other ecological communities have shown changes in response to disturbance. Bird communities along an urbanization gradient showed decreasing species diversity at high levels of urbanization, while moderate urbanization had the dual effect of both increasing overall diversity and reducing native species diversity (Blair, 1996). This was a result of increasing resources (water sources, ornamental plants, etc.) that allowed nonnative species to thrive while natives declined. Rocky intertidal zones showed a similar trend, with human disturbance at a moderate level increasing species diversity while extreme human disturbance resulted in species loss (Addessi, 1994). It seems reasonable to expect similar trends to occur in these fish communities due to the diversity of fishes in Missouri and the number and intensity of different human disturbances.

Dams are one of the main causes of disturbance for stream fishes in the Ozarks ecoregion (Warren, Jr. et al., 2000). Both dams and road crossings directly limit dispersal and reproduction by disrupting upstream movements that many fish undertake for spawning (Warren & Pardew, 1998; Porto et al., 1999; Santucci et al., 2005; Dugan et al., 2010). They also indirectly affect fish by altering stream flow, creating lentic conditions that often support nonnative species (Lessard & Hayes, 2003; Santucci et al., 2005; Anderson et al., 2006). Agriculture and urbanization are not as prevalent in the Ozarks compared to the Northern Plains and Mississippi Alluvial Basin, with about 50% of the Ozarks still reported as forested land (Owen et al., 2011; Sievert et al., 2016). Agriculture leads to run-off, which affects water quality by increasing flashiness, sediment and nutrient inputs (Buck et al., 2004; Vondracek et al., 2005; Dala-Corte et al., 2016). However, past high levels of agriculture are likely still impacting streams through legacy effects, especially sedimentation (Owen et al., 2011). Excessive sedimentation gets deposited along stream floodplains and change the form of the stream (Owen et al., 2011). Zinc and lead mining have been extremely common in areas of the Missouri Ozarks. Studies have shown that abandoned mines still have impacts on organisms via acid mine drainage, sedimentation build-up in the channel, and release of toxic heavy metals (Gray, 1997; Mol & Ouboter, 2004; Boudou et al., 2005; Schmitt et al., 2007; Allert et al., 2009).

Fish assemblages in the southeastern corner of Missouri are different from those found in the Ozarks, and more closely matching those found in the rest of the Mississippi Alluvial Basin (Pflieger, 1997; Sievert et al., 2016). These streams have been much more strongly impacted by agriculture, with over 80% of the region being cultivated (Sievert et al., 2016). Lead and zinc mining have also had strong impacts on streams in this region (Schmitt et al., 2007; Allert et al., 2009). Urbanization does not have a strong impact in this region, and makes up a very low percentage of the area included in the Missouri portion of the Mississippi Alluvial Basin (Sievert et al., 2016).

Historical collection data are important in understanding long-term trends in community assemblages (Graham et al., 2004; Szabo et al., 2010; Breed et al., 2012; Barnes et al., 2015). Lists of species are easy to collect, and can be recorded by citizens, thus increasing the number of people reporting observations and the area covered by samples (Szabo et al., 2010; Breed et al., 2012; Barnes et al., 2015). Depending on taxa, recorders often need little gear other than a notebook and a pencil (Barnes et al., 2015). Natural history collections are important as they provide vouchered specimens and represent a longer historical reference than citizen science data typically do (Graham et al., 2004; Hoeksema et al., 2011; Miller-Rushing et al., 2012; Zu Ermgassen et al., 2012). Both natural history collections and citizen science data allow unique opportunities for studies covering long time spans and large spatial scales (Hoeksema et al., 2011; Miller-Rushing et al., 2012).

The goal of this study is to examine changes in fish communities over recent time. Using lists of fish taxa obtained during field trips, I assessed whether there were apparent temporal changes in species presence. I predicted that a number of species, especially those considered sensitive, will exhibit declines. I also predicted that tolerant and nonnative species will increase through time.

Methods

I compiled 47 years of Missouri State University Ichthyology course field trip collection data. These data consisted of 560 samples at 140 wadeable stream locations across southern Missouri. Early years in the data had 15-20 sites sampled per year, while later years reduced to 4 or 5 (Table 4). Sites were sampled at random through the timespan, with some sampled nearly every year and others only sampled once or twice. All sites occurred at bridges or access points (Figures 17 and 18). Township and Range (T/R) coordinates and road names were included for each site, allowing me to pinpoint each location and obtain latitude and longitude coordinates.

Sampling techniques varied temporally as original effort included only seine nets, and later samples sometimes included backpack shockers. Counts of individuals were not recorded in the historical data, such that collections were simple lists of species present. Because a standardized transect was not recorded but time sampled was often included, time was used as a measure of sample effort when resampling sites.

I resampled 45 of these sites in southwest Missouri in the summer of 2016. Individual counts of fish captured were recorded to species level, along with time spent sampling and qualitative habitat data. Sites outside southwest Missouri were excluded from resampling to keep all resampled sites within the Ozarks ecoregion in order to more easily analyze differences in contemporary samples (Chapter 1). These collection data were then reduced to presence/absence to match the historical data. Logistic regressions were used to create a prediction of the probability of occurrence through time for each species using the glm function in R (R Core Team, 2017). Prior approval for this project was obtained from the Missouri State University Institutional Animal Care and Use Committee (IACUC; 11 May 2016; approval #16-026.0). The Missouri Department of Conservation (MDC) also provided me with a collector's permit for sampling (8 April 2016; permit #16855).

Results

The historical data consists of 158 species, 58 of which were captured when resampling. Minnows and darters were the most diverse and common groups. Samples from eastern sites in the St. Francis drainage and Mingo Wildlife Refuge were uncommon in the dataset due to eastern sites being sampled in only four of the years covered by the dataset. Further sampling in this region, as well as the number of samples done each year, declined with time (Table 4; Figure 19).

Logistic regressions yielded a set of models with either significant or non-significant relationships between presence and year (Table 5), but the majority of relationships were not different from zero. There are some species (n = 52) that did have a significant trend with time, of which 8 were positive and 44 were negative.

Several species of conservation concern were captured in the historical samples but were not captured in contemporary samples. This includes darters, such as the

Arkansas darter (*Etheostoma cragini*, Estimate = -0.036, $P = 0.001$, $Z = -3.292$), and several minnow species, including the slim minnow (*Pimephales tenellus*, Estimate = -0.052, $P = 0.042$, $Z = -2.032$). The Plains topminnow (*Fundulus sciadicus*) was also captured in several historical samples, but not found in any of the more contemporary samples (Estimate = -0.103, $P = <0.001$, $Z = -3.837$).

The most common species captured through time were in the Cyprinidae family. Stonerollers (*Campostoma* spp., Estimate = 0.020, $P = 0.019$, $Z = 2.341$) were caught in nearly every sample and had increased occupancy through time. Shiners in the *Luxilus* genus were very common across all sites in contemporary samples. The duskystripe shiner (*L. pilsbryi*) in the White basin showed an increasing trend over time (Estimate = 0.028, $P < 0.001$, $Z = 4.317$). The cardinal shiner (*L. cardinalis*) and bleeding shiner (*L. zonatus*) in the Neosho and Osage basins, respectively, did not show significant trends. Southern redbelly dace (*Chrosomus erythrogaster*) show an increasing trend over time (Estimate = 0.015, $P = 0.024$, $Z = 2.250$). Red shiners (*Cyprinella lutrensis*, Estimate = -0.040, $P < 0.001$, $Z = -4.304$), redfin shiners (*Lythrurus umbratilis*, Estimate = -0.058, $P < 0.001$, $Z = -5.422$), carmine shiners (*Notropis percobromus*, Estimate = -0.034, $P < 0.001$, $Z = -4.793$), bigeye shiners (*N. boops*, Estimate = -0.023, $P = 0.006$, $Z = -2.731$), and bluntnose minnows (*Pimephales notatus*, Estimate = -0.035, $P < 0.001$, $Z = -5.418$) all showed a decreasing trend.

The creek chub (*Semotilus atromaculatus*) is the only chub species that showed increased presence through time (Estimate = 0.023, $P < 0.001$, $Z = 3.569$). Several shiner species showed a reduced presence through time. Interestingly, the redspot chub

(*Nocomis asper*) showed a declining trend (Estimate = -0.036, *P* = 0.003, *Z* = -2.999) while the closely related hornyhead chub (*N. biguttatus*) showed no change.

Most sucker species were rare captures throughout the sample period, and there was an overall positive or negative trend of presence. However, redhorse suckers (*Moxostoma* spp.) did show a declining trend through time (Estimate = -0.050, *P* = 0.006, *Z* = -2.763). Generally, pickerel (*Esox* spp.) species were rare throughout the sample period, and grass pickerels (*Esox americanus*) were the most commonly captured pickerel in Missouri and showed a declining trend through time (Estimate = -0.037, *P* = 0.012, *Z* = -2.509).

No Ictalurids were commonly caught during the sample period. The slender madtom (*Noturus exilis*) were fairly common in the early samples but declined through time (Estimate = -0.020, *P* = 0.008, *Z* = -2.662). Stonecats (*Noturus* flavus) however, showed an increasing trend through time (Estimate = 0.039, *P* = 0.007, *Z* = 2.704). Other madtom species were very uncommon or rarely captured. Black bullheads (*Ameiurus melas*) also showed a declining trend through time (Estimate = -0.078, *P* < 0.001, *Z* = -4.692).

Rainbow trout (*Onorynchus mykiss*) were the only nonnative species captured in 2016. Other nonnatives listed in the historical data include common carp (*Cyprinus carpio*) and goldfish (*Carassius auratus*), but they were captured sporadically. Some Missouri natives have been spread to other basins they are not native to (such as the northern studfish, *Fundulus catenatus* into the Elk River), but these introductions typically occurred before the span of my historical data and did not affect any of the trends of introduced species.

Along with the plains topminnow, the blackstripe topminnow (*F. notatus*) showed declines over the sample period (Estimate = -0.049, $P < 0.001$, $Z = -4.031$). No other topminnow species showed clear trends. Brook silversides (*Labidesthes sicculus*) had a strong decline through years and are much less present now than they were in 1970 (Estimate = -0.023, $P < 0.001$, $Z = -3.651$).

Knobfin sculpin (*Cottus immaculatus*) have increased in presence in recent years (Estimate = 0.056, $P < 0.001$, $Z = 4.289$), which has been mirrored by a smaller increase in Ozark sculpin (*C. hypselurus*) in the Osage, Black, and Gasconade systems (Estimate = 0.064, $P = 0.017$, $Z = 2.394$). The mottled sculpin (*C. bairdii*) showed strong declines through time and has never been very common (Estimate = -0.064, $P < 0.001$, $Z = -4.539$).

Centrarchids showed a variety of trends. The Ozark bass (*Ambloplites constellatus*) had an increasing trend (Estimate = 0.047, $P < 0.001$, $Z = 3.756$). Green sunfish and bluegill (*Lepomis cyanellus*, and *L. macrochirus*) especially exhibited a decline through time (Estimate = -0.044, $P < 0.001$, Z -6.691; Estimate = -0.015, $P < 0.05$, $Z = -2.520$, respectively). Of the black basses, the largemouth bass (*Micropterus salmoides*) had a declining trend through time (Estimate = -0.032, $P < 0.001$, $Z = -4.768$).

Nearly all darter species showed declining trends and there were none showing an increase through time (Table 5). The fantail darter (*Etheostoma flabellare*), described as one of the most abundant darters in Missouri by (Pflieger, 1997) , showed a trend of decline in the logistic regression and was not commonly captured in contemporary samples (Estimate = -0.043, $P < 0.001$, $Z = -6.034$). Orangethroat darters (*E. spectabile*) and banded darters (*Etheostoma zonale*) were also common, yet declined through time

(Estimate = -0.014, $P = 0.022$, $Z = -2.282$ and Estimate = -0.020, $P = 0.003$, $Z = -2.946$, respectively). Least darters and Johnny darters (*E. microperca, E. nigrum*) also showed a trend of decline (Estimate = -0.062, $P < 0.001$, $Z = -4.579$ and Estimate = -0.060, $P < 0.001$, $Z = -3.925$, respectively). The sunburst darter *(Etheostoma mihileze*) had a declining trend through time (Estimate = -0.024, P =0.012, $Z = -2.520$) but this was not reflected in the closely related autumn and stippled darters (*E. autumnale*, *E. punctulatum*). *Percina* darters also showed declines. Logperch, channel darters, and slenderhead darters (*Percina caprodes*, *P. copelandi*, and *P. phoxocephala*) all also showed negative trends through time (Estimate = -0.020, $P = 0.002$, $Z = -3.078$; Estimate = -0.069, $P = 0.009$, $Z = -2.628$; Estimate = -0.050, $P = 0.002$, $Z = -3.090$, respectively).

Discussion

The majority of species with a significant trend in this study were in decline (declining = 44, increasing = 8). This is particularly true of darters and Cyprinids. All darters and many of the Cyprinids in decline are considered sensitive, which may point to pervasive threats to stream systems in the Ozarks (Pflieger, 1997; Barbour et al., 1999). Of species showing increasing trends there were four Cyprinids, two sculpins, the Ozark bass, and the stonecat. Of these the dace, Ozark bass, and stonecat are worth additional attention because they are considered sensitive species. The creek chub is considered tolerant and all other species with increased occupancy are not considered to be either tolerant or sensitive (Pflieger, 1997; Barbour et al., 1999).

For this analysis, I assume that an increase in occupancy means an increase in population, and that a decrease would mean a population in decline. Potential causes and mechanisms of positive and negative trends through time are likely specific to the species under consideration. For instance, some species listed by the state of Missouri as "conservation concern" are expected to show declining trends because they may be in decline in the state. The Arkansas darter is known to be impacted by Animal Feeding Operations (AFOs) and urbanization, both of which are expected to continue to increase in Missouri (Pflieger, 1997). The slim minnow and the plains topminnow both appear to have been impacted by dams (Pflieger, 1997).

It was not surprising to see an increasing trend in stonerollers because they have been expanding and increasing in abundance in association with increased primary productivity related to agricultural nutrient runoff (Pflieger, 1997; Allan, 2004). However, one study found that stoneroller abundance was related to the presence of hard substrate for periphyton attachment, and they had reduced abundance in streams with heavy siltation (Stauffer et al., 2000). The duskystripe shiner was another common cyprinid with an increasing trend. In my contemporary samples this species was by far the most commonly captured fish in agricultural streams. However, the lack of a similar trend in the closely related cardinal and bleeding shiners points to a potential for this trend to be related to the gradual increase of samples in the White River basin over time and the subsequent reduction of samples in other basins. The only chub with an increasing trend was the creek chub. This species is considered tolerant to many of the impacts of agriculture and urbanization, which may explain its increase over time (Barbour et al., 1999). Southern redbelly dace also showed increasing occupancy through time. These dace often occur sporadically but dominate the assemblage in streams where they are found. In contemporary samples, they were the only common minnow found in the heavily urbanized streams in Springfield, Missouri (Chapter 1).

Several cyprinids had a negative trend with time. Interestingly the red shiner and the redfin shiner are known to spawn over green sunfish nests, and it is possible that declines of these two shiners are related to declines in green sunfish. Carmine shiners are considered to be widespread and stable, but were extirpated from areas in the White River basin in the mid-1940s, as a result of the construction of Table Rock and Bull Shoals reservoirs (Pflieger, 1997). However, I captured seven individuals in Bee Creek, a tributary that flows into Bulls Shoals Reservoir, during the summer 2016 sampling, suggesting that this species may be able to recover from areas where they were previously extirpated. The redspot chub is considered a common minnow in the Neosho basin and (Pflieger, 1997) states that its distribution and abundance have not changed over 50 years, making it difficult to determine why it showed a decreasing trend in my analysis.

The decline of slender madtoms is likely related to water quality issues. Madtoms in general are considered sensitive to pollution (Barbour et al., 1999).

Declines of blackstripe topminnow and brook silverside likely are related to hydrologic changes that cause reduction in pool and backwater formation, such as channelization in urban streams. Both topminnows and silversides prefer shallow backwaters with little current as these provide important protection from aquatic predators and good foraging opportunities for insects at the surface (Pflieger, 1997). This also may be an explanation for the declines of green sunfish, bluegill, and largemouth bass, as they all prefer deep pools with structure (Pflieger, 1997). It is possible that declines of blackstripe topminnow are related to the introduction of the blackspotted topminnow into the Neosho basin (Pflieger, 1997).

The knobfin and Ozark sculpin are closely related, only recognized as separate species in 2010 (Kinziger & Wood, 2010). It is possible that these two species are responding similarly to pressures. The decline of mottled sculpins may have two other potential causes. This species is only found in the Osage basin, which it shares with the Ozark sculpin. Due to the similarity in appearance of these two species it is likely that the graduate students performing the later collections were not able to properly identify this species. Fish in earlier collections were identified by Dr. Taber, the Ichthyology professor. The second possibility is that the number of samples in the Osage basin declined through time, potentially affecting the results of the logistic regression.

The trend of decline shown for the green sunfish is unexpected as they are considered tolerant and common (Pflieger, 1997). (Pflieger, 1997) states that the bluegill is more widespread and abundant now than it was 50 years ago due to the creation of impoundments and stocking them as pond fish. It is not clear why they had a declining trend in my analysis. The same is true for largemouth bass, which I expected to show an increase due to the popularity of sport fishing bass and the management of many bodies of water to support this fishery.

Many darters are considered sensitive to water quality (Barbour et al., 1999), and it, therefore was not surprising to see declines in so many of them. Specific causes of decline are more difficult to determine, however. It is known that gravel mining in Haw Creek extirpated the least darter from that stream, but explanations for the many other species exhibiting declines are not clear (Pflieger, 1997). In Missouri the channel darter is only common in the North Fork of the Spring River, which is the furthest portion of the Spring River from Springfield (Pflieger, 1997). Decline of this species could well be due to a lack of sampling in its range. The fact that many darter species were considered stable and common in Pflieger's book *The Fishes of Missouri* and now appear to be in decline is potentially a cause for concern and further studies should be conducted to ascertain the true extent of declines.

Urbanization, hydrologic alteration, groundwater extraction, and climate change are anthropogenic impacts known to cause species declines and range interruptions (Allan, 2004; Foley et al., 2005; O'Gorman et al., 2012; Mantyka-Pringle et al., 2014; Sievert et al., 2016; Kuczynski et al., 2018). The Missouri Ozarks are not free of these impacts and have been strongly affected in some areas (Adamski et al., 1995; Jacobson et al., 1997; Owen et al., 2011). However, caution should be used when looking at the trends seen in this study. There are several potential influences that may cause the logistic regression to produce a declining trend where one does not actually exist.

Several difficulties often accompany the use of species lists. One is that these may sometimes consist of lists of species that were observed at a location, without individual counts or other recorded ancillary data (Szabo et al., 2010; Breed et al., 2012). Also, most analyses capable of handling large long-term datasets require additional data, such as survey type and sampling method (Szabo et al., 2010; Breed et al., 2012). Finally, there is often little control of sampling effort in this type of data, making it difficult to compare samples (Szabo et al., 2010; Breed et al., 2012). Keeping these difficulties in mind, we can point out several potential issues with analyzing the dataset in this study.

First, there likely was variability in sampling effort during collections in part because there were no transects or set sampling areas determined for sites and no indication of how many people were sampling. The collections were performed by Ichthyology classes, but class sizes ranged from 5–6 students up to 20–30. The best indicator of sampling effort is time sampled, which was recorded for most of the samples and averaged about an hour per sample. If sampling effort changed over time it would affect fish captures since increased sample effort is known to increase the likelihood of capturing any particular species (Walther et al., 1995; Martinez et al., 1999; Gotelli & Colwell, 2001; Bady et al., 2005).

Another consideration is which sites were sampled over the duration of the study. This is especially important as my analysis (Chapter 1) showed that species assemblages in 2016 were different based on basin. Missouri State University is located in Springfield, Missouri and the samples taken in the eastern half of the state would have required an overnight stay and would have to have been completed on a weekend, which students may find difficult to attend. These sites were abandoned over time due to the difficulty in travelling that distance and could yield the declining trends shown for any species found in those sites. The same can be considered for sites in western Missouri, where further sites were abandoned in favor of closer sites that were easier to reach in the time limit of a class. Increasing samples done at closer sites, made it appear as though species captured at those sites are becoming more common. A good example of this would be the three *Ambloplites* species found in Missouri. The Ozark bass (*A. constellatus*) is found in the White basin and appears to be increasing in presence over time. The rock bass (*A. rupestris*) is found in the Neosho and Osage basins, while the shadow bass (*A. ariommus*) is found in the eastern part of the White basin and in the Mississippi Alluvial Basin, and both species appear to be in decline over time. This difference may well be due to not sampling in those basins as much as the White basin. Sampling in the White basin increased through time since Springfield lies in this basin. At the same time, all collections in the southeastern part of Missouri were discontinued partway through the collection records and while the Neosho and Osage basins were still sampled, they were not sampled as often as the White basin in later collections.

Finally, the ability of the collectors to accurately identify species also needs to be considered (Szabo et al., 2011; Barnes et al., 2015). The professor teaching Ichthyology, who was an expert at fish identification, oversaw the early samples. Graduate students with varying degrees of identification experience oversaw later samples. The subtle difference between many species of fish may make it difficult to properly identify some of the species captured, especially minnows, darters, and young-of-the-year sunfishes (Pflieger, 1997).

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Table 4. Mean distance traveled each year and number of samples occurring in western and eastern basins.

Year	Mean Distance Traveled (km)	Number of Samples in Western Basins	Number of Samples in Eastern Basins
1970	143.36	3	3
1971	60.99	15	0
1972	81.65	27	6
1973	53.98	9	0
1974	44.24	20	0
1975	46.93	9	0
1976	43.84	24	0
1977	44.64	17	0
1978	47.27	16	0
1979	54.45	21	0
1980	40.21	11	0
1981	49.63	24	0
1982	108.48	14	5
1983	61.23	24	0
1984	110.86	10	4
1985	46.32	9	0
1986	45.96	9	0
1987	38.81	8	0
1988	39.20	8	0
1989	41.15	14	0
1990	39.20	8	0
1991	39.41	17	0
1992	35.11	13	0
1993	32.84	11	0
1994	34.70	10	0
1995	29.12	12	0
1996	43.26	4	0
1997	29.10	13	0
1998	31.63	9	0
1999	33.36	9	0
2000	33.13	10	0
2001	30.42	9	0
2002	31.84	6	0
2003	27.98	6	0
2004	8.78	5	0
2005	17.19	9	0

Table 4 continued. Mean distance traveled each year and number of samples occurring in western and eastern basins.

Year	Mean Distance Traveled (km)	Number of Samples in Western Basins	Number of Samples in Eastern Basins
2006	17.19	9	0
2007	16.24	10	0
2008	16.32	10	0
2009	22.66	6	0
2010	22.16	3	0
2011	31.76	4	0
2012	27.02	5	0
2013	31.76	4	0
2014	26.80	5	0
2016	53.36	45	0
2017	8.07	4	0

Table 5. Year coefficients from the logistic regression for each species in the historical data.

Common Name	Scientific Name	Estimate	Standard Error	Z Statistic [1]
Cyprinidae				
Stoneroller	*Campostoma spp.*	0.020	0.009	2.341^{*}
Bluntface Shiner	*Cyprinella camura*	-0.051	0.028	-1.863
Whitetail Shiner	*Cyprinella galactura*	0.008	0.009	0.918
Red Shiner	*Cyprinella lutrensis*	-0.040	0.009	-4.304^{***}
Spotfin Shiner	*Cyprinella spiloptera*	-0.001	0.030	-0.048
Blacktail Shiner	*Cyprinella venusta*	-0.010	0.019	-0.550
Steelcolor Shiner	*Cyprinella whipplei*	-0.050	0.095	-0.533
Streamline Chub	*Erimystax dissimilis*	-0.014	0.024	-0.562
Ozark Chub	*Erimystax harryi*	0.040	0.025	1.618
Gravel Chub	*Erimystax x-punctatus*	0.0005	0.019	0.026
Mississippi Silvery Minnow	*Hybognathus nuchalis*	-0.519	0.534	-0.971
Cardinal Shiner	*Luxilus cardinalis*	-0.006	0.007	-0.869
Striped Shiner	*Luxilus chrysocephalus*	0.005	0.006	0.864
Duskystripe Shiner	*Luxilus pilsbryi*	0.028	0.006	4.317^{***}
Bleeding Shiner	*Luxilus zonatus*	0.002	0.007	0.226
Redfin Shiner	*Lythrurus umbratilis*	-0.058	0.011	-5.422^{***}
Redspot Chub	*Nocomis asper*	-0.036	0.012	-2.999^{**}
Hornyhead Chub	*Nocomis biguttatus*	0.003	0.007	0.459
Golden Shiner	*Notemigonus crysoleucas*	-0.054	0.012	-4.610^{***}
Bigeye Chub	*Notropis amblops*	-0.025	0.047	-0.545
Bigeye Shiner	*Notropis boops*	-0.023	0.008	-2.731^{**}
Ghost Shiner	*Notropis buchanani*	-0.058	0.038	-1.532
Wedgespot Shiner	*Notropis greenei*	0.038	0.031	1.218
Taillight Shiner	*Notropis maculatus*	-0.519	0.534	-0.971
Ozark Minnow	*Notropis nubilus*	-0.005	0.006	-0.865
Carmine Shiner	*Notropis percobromus*	-0.034	0.007	-4.793^{***}
Sand Shiner	*Notropis stramineus*	-0.063	0.019	-3.309^{***}
Telescope Shiner	*Notropis telescopus*	0.007	0.009	0.784
Mimic Shiner	*Notropis volucellus*	-0.061	0.036	-1.687
Pugnose Minnow	*Opsopoeodus emiliae*	-0.089	0.044	-2.028^{*}

[1] * significant at $P < 0.05$; ** significant at $P < 0.01$; *** significant at $P < 0.001$.

Table 5 continued. Year coefficients from the logistic regression for each species in the historical data.

Common Name	**Scientific Name**	**Estimate**	**Standard Error**	**Z Statistic [1]**
Suckermouth Minnow	*Phenacobius mirabilis*	-0.056	0.027	-2.042^{*}
Southern Redbelly Dace	*Chrosomus erythrogaster*	0.015	0.007	2.250^{*}
Bluntnose Minnow	*Pimephales notatus*	-0.035	0.006	-5.418^{***}
Fathead Minnow	*Pimephales promelas*	-0.040	0.020	-1.944
Slim Minnow	*Pimephales tenellus*	-0.052	0.026	-2.032^{*}
Bullhead Minnow	*Pimephales vigilax*	-0.034	0.085	-0.404
Creek Chub	*Semotilus atromaculatus*	0.023	0.006	3.569^{***}
Goldfish	*Carassius auratus*	0.008	0.040	0.194
Common Carp	*Cyprinus carpio*	-0.032	0.014	-2.383^{*}
Carp X Goldfish	*C. carpio X C. auratus*	-0.050	0.095	-0.533
Red Shiner X Blacktail Shiner	*C. lutrensis X C. venusta*	-0.004	0.073	-0.055
Bleeding Shiner X Ozark Minnow	*L. zonatus X N. nubilus*	-0.071	0.109	-0.650
Rosyface Shiner X Ozark Minnow	*N. rubellus X N. nubilus*	-0.057	0.057	-0.995
Duskystripe Shiner X Ozark Minnow	*L. pilsbryi X N. nubilus*	-0.034	0.085	-0.404
Notropis Hybrid	*Notropis spp.*	-0.068	0.054	-1.275
Catostomidae				
Quillback	*Carpiodes cyprinus*	-0.042	0.089	-0.470
Highfin Carpsucker	*Carpiodes velifer*	0.006	0.070	0.088
Smallmouth Buffalo	*Ictiobus bubalus*	0.023	0.049	0.473
Bigmouth Buffalo	*Ictiobus cyprinellus*	-0.043	0.032	-1.337
Northern Hog Sucker	*Hypentelium nigricans*	0.008	0.007	1.181
River Redhorse Sucker	*Moxostoma carinatum*	-0.024	0.033	-0.744
Black Redhorse	*Moxostoma duquesnei*	-0.001	0.007	-0.215
Western Creek Chubsucker	*Erimyzon claviformis*	-0.058	0.030	-1.929
White Sucker	*Catostomus commersonii*	0.009	0.008	1.056

[1] * significant at $P < 0.05$; ** significant at $P < 0.01$; *** significant at $P < 0.001$.

Table 5 continued. Year coefficients from the logistic regression for each species in the historical data.

Common Name	Scientific Name	Estimate	Standard Error	Z Statistic [1]
Silver Redhorse	*Moxostoma anisurum*	-0.028	0.019	-1.476
Pealip Redhorse	*Moxostoma pisolabrum*	-0.046	0.016	-2.926^{**}
Golden Redhorse	*Moxostoma erythrurum*	-0.007	0.009	-0.758
Sucker	*Moxostoma spp.*	-0.050	0.018	-2.763^{**}
Spotted Sucker	*Minytrema melanops*	118.980	67.894	1.752
Ictaluridae				
Checkered Madtom	*Noturus flavater*	0.014	0.040	0.358
Slender Madtom	*Noturus exilis*	-0.020	0.007	-2.662^{**}
Tadpole Madtom	*Noturus gyrinus*	-0.121	0.076	-1.587
Ozark Madtom	*Noturus albater*	-0.002	0.012	-0.197
Black River Madtom	*Noturus maydeni*	-0.519	0.534	-0.971
Stonecat	*Noturus flavus*	0.039	0.014	2.704^{**}
Brindled Madtom	*Noturus miurus*	-0.044	0.024	-1.875
Madtom	*Noturus spp.*	-0.722	0.519	-1.391
Black Bullhead	*Ameiurus melas*	-0.078	0.017	-4.692^{***}
Yellow Bullhead	*Ameiurus natalis*	0.002	0.008	0.258
Brown Bullhead	*Ameiurus nebulosus*	-0.121	0.076	-1.587
Channel Catfish	*Ictalurus punctatus*	-0.025	0.014	-1.782
Flathead Catfish	*Pylodictus olivaris*	-0.029	0.024	-1.198
Esocidae				
Grass Pickerel	*Esox americanus*	-0.037	0.015	-2.509^{*}
Chain Pickerel	*Esox niger*	-0.121	0.076	-1.587
Pickerel	*Esox spp.*	-0.050	0.095	-0.533
Salmonidae				
Rainbow Trout	*Oncorhynchus mykiss*	0.012	0.011	1.130
Brown Trout	*Salmo trutta*	-0.333	0.350	-0.952
Fundulidae				
Northern Studfish	*Fundulus catenatus*	-0.008	0.007	-1.115
Blackspotted Topminnow	*Fundulus olivaceous*	-0.003	0.006	-0.568
Starhead Topminnow	*Fundulus dispar*	-0.122	0.063	-1.946
Blackstripe Topminnow	*Fundulus notatus*	-0.049	0.012	-4.031^{***}
Plains Topminnow	*Fundulus sciadicus*	-0.103	0.027	-3.837^{***}
Topminnow	*Fundulus spp.*	0.013	0.049	0.275
Poeciliidae				
Mosquitofish	*Gambusia affinis*	0.009	0.006	1.434

[1] * significant at $P < 0.05$; ** significant at $P < 0.01$; *** significant at $P < 0.001$.

Table 5 continued. Year coefficients from the logistic regression for each species in the historical data.

Common Name	**Scientific Name**	**Estimate**	**Standard Error**	**Z Statistic** [1]
Atherinopsidae				
Brook Silverside	*Labidesthes sicculus*	-0.023	0.006	-3.651***
Inland Silverside	*Menidia audens*	-0.211	0.167	-1.263
Cottidae				
Ozark Sculpin	*Cottus hypselurus*	0.064	0.027	2.394*
Knobfin Sculpin	*Cottus immaculatus*	0.056	0.013	4.289***
Banded Sculpin	*Cottus carolinae*	-0.012	0.007	-1.789
Mottled Sculpin	*Cottus bairdii*	-0.064	0.014	-4.539***
Centrarchidae				
Rock/Shadow/Ozark Bass	*Ambloplites*	-0.134	0.082	-1.628
Shadow Bass	*Ambloplites ariommus*	0.006	0.070	0.088
Ozark Bass	*Ambloplites constellatus*	0.047	0.013	3.756***
Rock Bass	*Ambloplites rupestris*	-0.082	0.014	-6.000***
Flier	*Centrarchus macropterus*	-0.101	0.051	-1.989*
Warmouth	*Lepomis gulosus*	-0.052	0.027	-1.940
Orangespotted Sunfish	*Lepomis humilis*	-0.087	0.017	-5.021***
Bluegill	*Lepomis macrochirus*	-0.015	0.006	-2.520*
Longear Sunfish	*Lepomis megalotis*	-0.005	0.006	-0.818
Bluegill X Green Sunfish	*L. macrochirus X L.cyanellus*	-0.015	0.012	-1.233
Longear X Green Sunfish	*L. megalotis X L. cyanellus*	0.021	0.040	0.520
Bluegill X Longear Sunfish	*L. macrochirus X L. megalotis*	0.068	0.044	1.523
Green X Orangespotted Sunfish	*L. cyanellus X L. humilis*	-0.133	0.116	-1.150
Green Sunfish X Rock Bass	*L. cyanellus X A. rupestris*	-0.241	0.264	-0.913
Bluegill X Orangespotted Sunfish	*L. macrochirus X L. humilis*	-0.077	0.081	-0.957
Redear Sunfish	*Lepomis microlophus*	-0.018	0.039	-0.467
Green Sunfish	*Lepomis cyanellus*	-0.044	0.007	-6.691***
Lepomis sp.	*Lepomis spp*	-0.009	0.053	-0.179
Spotted Sunfish	*Lepomis punctatus*	-0.519	0.534	-0.971

[1] * significant at $P < 0.05$; ** significant at $P < 0.01$; *** significant at $P < 0.001$.

Table 5 continued. Year coefficients from the logistic regression for each species in the historical data.

Common Name	**Scientific Name**	**Estimate**	**Standard Error**	**Z Statistic** [1]
Bantam Sunfish	*Lepomis symmetricus*	-0.149	0.080	-1.865
Largemouth Bass	*Micropterus salmoides*	-0.032	0.007	-4.768^{***}
Spotted Bass	*Micropterus punctulatus*	-0.013	0.008	-1.670
Smallmouth Bass	*Micropterus dolomieu*	0.005	0.007	0.681
Black Crappie	*Pomoxis nigromaculatus*	-0.034	0.018	-1.905
White Crappie	*Pomoxis annularis*	-0.071	0.015	-4.665^{***}
Elassomatidae				
Banded Pygmy Sunfish	*Elassoma zonatum*	-0.210	0.089	-2.360^{*}
Percidae				
Darter	*Etheostoma spp.*	-0.186	0.123	-1.512
Fantail Darter	*Etheostoma flabellare*	-0.043	0.007	-6.034^{***}
Greenside Darter	*Etheostoma blennioides*	-0.011	0.006	-1.833
Orangethroat Darter	*Etheostoma spectabile*	-0.014	0.006	-2.282^{*}
Rainbow Darter	*Etheostoma caeruleum*	-0.008	0.006	-1.322
Stippled Darter	*Etheostoma punctulatum*	-0.017	0.018	-0.974
Autumn Darter	*Etheostoma autumnale*	0.027	0.017	1.593
Sunburst Darter	*Etheostoma mihileze*	-0.024	0.009	-2.520^{*}
Yoke Darter	*Etheostoma juliae*	-0.005	0.010	-0.520
Banded Darter	*Etheostoma zonale*	-0.020	0.007	-2.946^{**}
Missouri Saddle Darter	*Etheostoma tetrazonum*	-0.015	0.008	-1.787
Arkansas Darter	*Etheostoma cragini*	-0.036	0.011	-3.292^{***}
Least Darter	*Etheostoma microperca*	-0.062	0.013	-4.579^{***}
Johnny Darter	*Etheostoma nigrum*	-0.060	0.015	-3.925^{***}
Niangua Darter	*Etheostoma nianguae*	-0.013	0.019	-0.722
Cypress Darter	*Etheostoma proeliare*	-0.244	0.120	-2.046^{*}
Arkansas Saddled Darter	*Etheostoma euzonum*	-16.075	1083.589	-0.015

[1] * significant at $P < 0.05$; ** significant at $P < 0.01$; *** significant at $P < 0.001$.

Table 5 continued. Year coefficients from the logistic regression for each species in the historical data.

Common Name	**Scientific Name**	**Estimate**	**Standard Error**	**Z Statistic** [1]
Bluntnose Darter	*Etheostoma chlorosoma*	-0.114	0.060	-1.912
Harlequin Darter	*Etheostoma histrio*	-0.121	0.108	-1.121
Speckled Darter	*Etheostoma stigmaeum*	-0.138	0.097	-1.420
Highland Darter	*Etheostoma teddyroosevelt*	-0.036	0.020	-1.757
Slough Darter	*Etheostoma gracile*	-0.092	0.042	-2.181^{*}
Redfin Darter	*Etheostoma whipplei*	-0.071	0.029	-2.423^{*}
Logperch	*Percina caprodes*	-0.020	0.006	-3.078^{**}
Channel Darter	*Percina copelandi*	-0.069	0.026	-2.628^{**}
Bluestripe Darter	*Percina cymatotaenia*	-0.021	0.026	-0.786
Blackside Darter	*Percina maculata*	-0.050	0.095	-0.533
Slenderhead Darter	*Percina phoxocephala*	-0.050	0.016	-3.090^{**}
Dusky Darter	*Percina sciera*	-0.050	0.095	-0.533
Stargazing Darter	*Percina uranidea*	-0.050	0.095	-0.533
Walleye	*Sander vitreus*	-0.036	0.043	-0.844
Clupeidae				
Gizzard shad	*Dorosoma cepedianum*	-0.034	0.008	-4.043^{***}
Threadfin Shad	*Dorosoma petenense*	-0.121	0.108	-1.121
Lepisosteidae				
Spotted Gar	*Lepisosteus oculatus*	-0.121	0.108	-1.121
Longnose Gar	*Lepisosteus osseus*	-0.016	0.020	-0.812
Shortnose Gar	*Lepisosteus platostomus*	-0.134	0.082	-1.628
Sciaenidae				
Freshwater Drum	*Aplodinotus grunniens*	-0.014	0.025	-0.550
Petromyzontidae				
Lamprey	*Ichthyomyzon spp.*	-0.093	0.052	-1.784
Least Brook Lamprey	*Lampetra aepyptera*	-1.086	1.008	-1.077
Moronidae				
White Bass	*Morone chrysops*	-0.022	0.017	-1.265
Aphredoderidae				
Pirate Perch	*Aphredoderus sayanus*	-0.137	0.056	-2.453^{*}
Amiidae				
Bowfin	*Amia calva*	-0.100	0.095	-1.060

[1] * significant at $P < 0.05$; ** significant at $P < 0.01$; *** significant at $P < 0.001$.

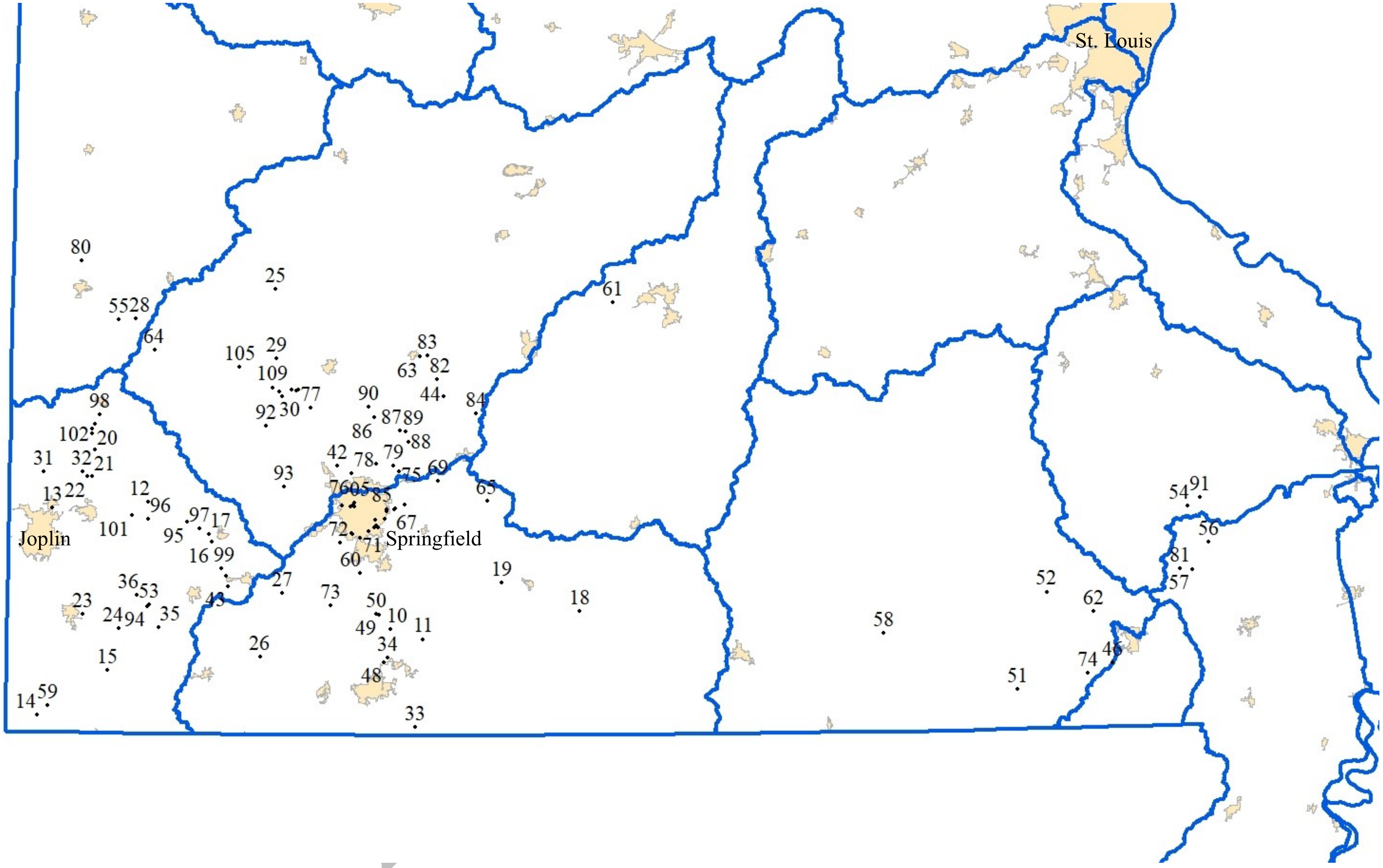

Figure 17. Map of study sites. Basins are outlined in blue and cities are shown in tan.

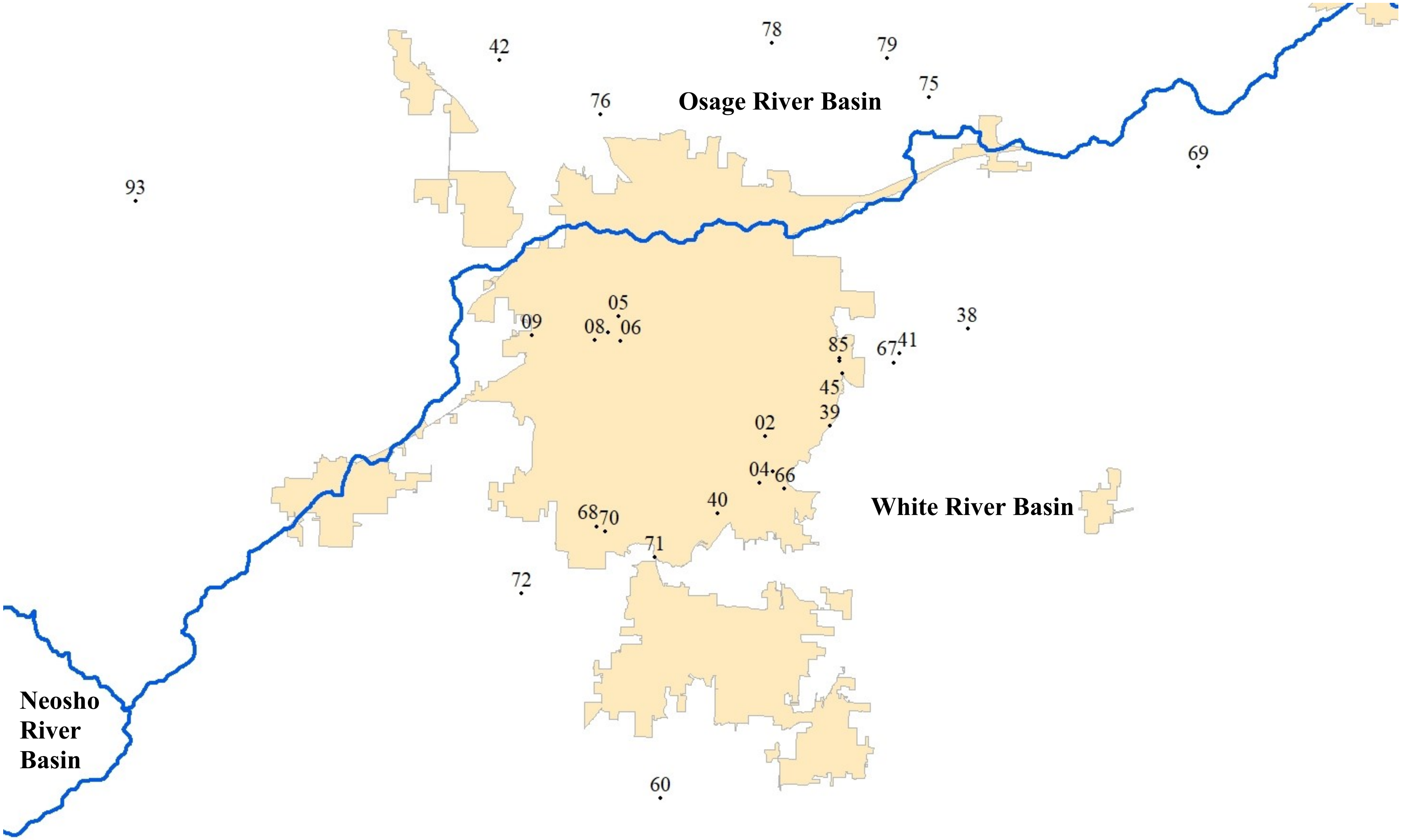

Figure 18. Sites located in and around Springfield, Missouri. Springfield is shown in tan and basin boundaries are in blue.

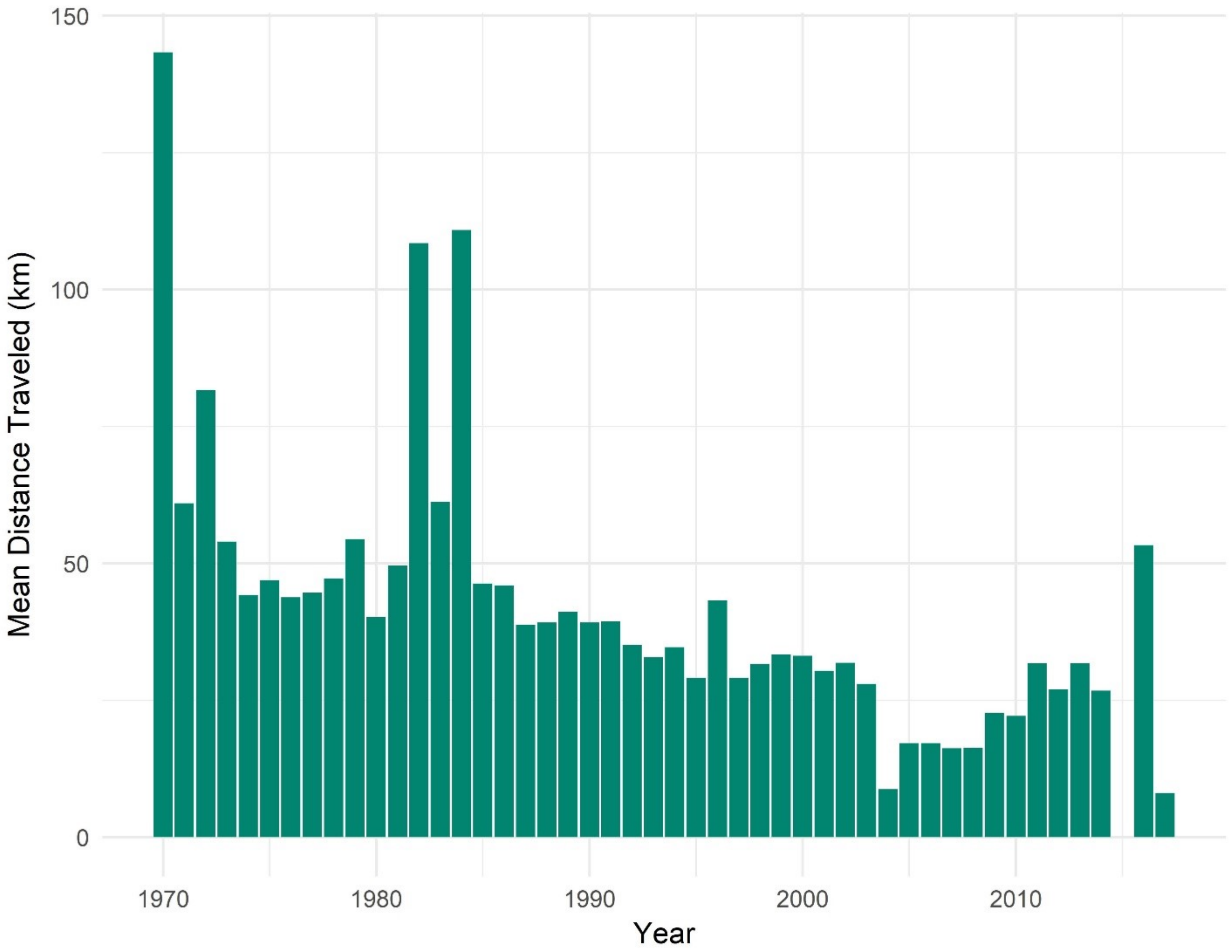

Figure 19. Mean distance traveled for sampling through time. Large distances in the early 1970s and 1980s show years when eastern sites were sampled. Distance travelled each year became much lower in later years.

SUMMARY

By combining a contemporary analysis of land use and a long-term analysis of historical data we have been able to bring some clarity to fish communities in the Missouri Ozarks. Overall, many of the species included in the study appear to be stable. However, the large number of species that showed declines, as well as all significant trends for darters showing decline is cause for concern. Many of the species showing declines are sensitive and this may be a sign of increasing species loss in the future.

There are a number of concerns that need to be addressed when dealing with list data. Lack of sample effort is the most pressing concern related to my long-term dataset. It is hard to say for certain if trends seen in the long-term data are independent of or related to sample effort. Presence/absence data can also be difficult to work with. Abundance information can sometimes provide better insight into declining and increasing trends than simple presence/absence data. Overall, this helps to highlight the importance of recording precise data, even if the main objective is to teach a class.

Human disturbance seems to be the most profound cause of these declines. Dams and roadways block fish passage and disconnect populations. Dams also increase lentic habitat and water temperature while reducing available oxygen (Porto et al., 1999; Santucci et al., 2005; Dugan et al., 2010). Acid mine drainage and metal toxicity are main issues associated with lead and zinc mining in southern Missouri (Gray, 1997; Mol & Ouboter, 2004; Boudou et al., 2005; Schmitt et al., 2007; Allert et al., 2009). Urbanization accounts for only a small proportion of land use in southern Missouri yet has a disproportionate effect on fish communities (Wang et al., 2001; McKinney, 2002;

Cervantes-Yoshida et al., 2015). Channelization, hydrologic shifts, increasing temperatures, and increased sedimentation make it difficult for all but the hardiest of species to survive in urban streams. Agriculture uses the largest amount of land in southern Missouri by far, with nearly 50% of the Ozarks being cultivated (Sievert et al., 2016). This causes sedimentation, increased nutrients, increased flashiness, and erosion (Stauffer et al., 2000; Vondracek et al., 2005; Dala-Corte et al., 2016). Removal of riparian vegetation also causes a number of effects, including erosion, increased temperatures, and a switch to autochthonous energy (Jones III et al., 1999; Stauffer et al., 2000).

These disturbances often occur together. Riparian vegetative removal often occurs alongside both agriculture and urbanization, removing an important protective barrier for streams. Leaving riparian vegetation intact is very important to ameliorate the effects of human activities on streams. Dams are often built near cities to provide hydroelectric power and water. My study serves to highlight the importance of considering how human activities will affect streams. Using best management practices in agriculture and rerouting city stormflow into rain gardens and other holding structures can provide increased protection for streams.

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