LANDSCAPE IMPACTS ON FISH COMMUNITY STRUCTURE AND FOOD CHAIN LENGTH IN PRAIRIE AND OZARK RIVERS.

By

Micaleila D. Desotelle

Submitted to the Department of Ecology and Evolutionary Biology and the Graduate Faculty of the University of Kansas in partial fulfillment of the requirements for the degree of Master of Arts

Committee:

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Chairperson

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Date Defended _____

The Thesis Committee for Micaleila D. Desotelle certifies that this is the approved version of the following thesis:

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Micaleila D. Desotelle M. A.

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University of Kansas

Rivers in the Ozark Highland ecoregion and Central Prairie ecoregion differ in land use and diversity, and these could impact food chain length. The primary factors controlling food chain length are not certain, but were considered. Fish and invertebrates were collected for stable isotope analysis and analyzed for trophic position. Land use was measured using remote sensing. Fish community structure was correlated to land use, but not necessarily to water quality. In particular, it appears that the amount of forest or agriculture is very important in determining fish and invertebrate stream community composition. Food chain length was related to neither the predicted hypotheses nor community structure. However, members of the family Cyprinidae were very common, and rivers where few cyprinids were captured had low food chain length. Food chain length is driven by many processes and the effects of landscape should be considered.

Key Words: trophic position, land use, fish

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Introduction

Land use changes and impaired water quality have led to changes in stream biodiversity, community structure, and food web properties such as greater connectance and, in some studies, trophic position (Thompson and Townsend 2005; Romanuk et al. 2006). Food chain length and trophic position are especially useful tools for comparing food webs in different landscapes, but theories that explain variation in food chain length often appear contradictory. Food chain length is a metric reflecting the number of energy transfers from the original food source through the food web to the top consumer (Post 2002a), while trophic position is a specific measure of an individual's location within the food chain (Vander Zanden and Rasmussen 1996). The trophic position of the same species of fish in different habitats can differ as a result of predator prey interactions, omnivory, and stream community composition (Vander Zanden and Rasmussen 1996, Beaudoin et al. 1999, Vander Zanden et al. 2000, Post and Takimoto 2007). The trophic position of the top predators in an ecosystem can be inferred to be the longest food chain and the total food chain length. Since food chain length integrates important energy flows, understanding the causes of variation in food chain length is important.

Variations in food chain length have been attributed to many factors including the total energy available to the food web (Jenkins et al. 1992, Kauzinger and Morin 1998, Townsend et al. 1998), disturbance in streams (Power 1992, Marks et al. 2000, Parker and Huryn 2006), and ecosystem size (Cohen and Newman 1991, Post 2000). Ecosystem size is thought to be an important factor because larger habitats tend to

have more species, as discussed in related theories of island biogeography (MacArthur and Wilson 1967, Holt 1996). Post (2000) and Thompson and Townsend (2005) concluded that ecosystem size and productivity best explained food chain length in lakes and streams. Power et al. (1996) felt that the intermediate disturbance would lead to the longest food chains. When disturbance and productivity were studied simultaneously (Townsend et al. 1998), only productivity was found to be important. Studies have not adequately explained why productivity, disturbance, or ecosystem size may be important in one system, but not in another. Post and Takimoto (2007) recently hypothesized that species additions, deletions, and omnivory are all proximate factors that can alter food chain length. In less species-rich systems, additions and deletions of taxa can cause noticeable changes in food chain length (Vander Zanden et al. 2000), but not be noticeable in streams (Quinn et al. 2003). These differences between ecosystems may lead to different conclusions. In addition to natural causes of differences among ecosystems in food chain length, many streams and rivers are heavily impacted by land use practices in the watershed in ways that can significantly alter the natural community composition (Wang et al. 2006). The use of hierarchy theory to place food webs in context with landscape features may help explain why certain theories explain food chain length in one system but not another.

Hierarchical theory can be used to understand how large spatial and temporal scale factors alter stream conditions. These environmental conditions act as filters that determine which life history traits enhance survival at a given site and affect

distribution and abundances of stream organisms (Southwood 1977, Frissell et al 1986, Townsend and Hildrew 1994). More recent changes to the landscape, such as the conversion of grassland and forest to agriculture, have led to major changes to stream communities (Quinn and Hickey 1990, Corkum 1991). Such large-scale landscape effects on stream communities have been hypothesized to be important for food web patterns (Woodward and Hildrew 2002).

In the present study, I compared two ecoregions (the Central Irregular Plains and the Ozark Highland Ecoregion) with very different geological features and vegetation to understand how the impacts of natural and altered landscape features affect food chain length and maximum trophic position. The fish communities in streams of these two ecoregions are distinct (Cross and Collins 1995, Pflieger 1997) as a result of differences in regional environmental characteristics and in stream conditions (Cross 1967, Smith and Fisher 1970, Marsh-Matthews et al. 2000). Large-scale patterns of fish diversity are impacted both by latitude, and fish diversity decreases from east to west as environmental conditions become harsher (Marsh-Matthews et al. 2000). Although the rivers I studied are tributaries of the Missouri and Mississippi Rivers and should thus have a common regional species pool, distribution of fish species within these rivers have been determined by past climatic and glacial events, zoogeography, drainage patterns and topographic limits (Cross 1970, Pflieger 1971, Matthews and Robison 1998 a, b, Marsh-Matthews et al. 2000). As a result of differences in climate, geology and glacial impacts, prairie streams

have low endemic diversity, while streams in the Ozark Plateau have high endemic diversity and indices of biological integrity (Cross 1970, Pflieger 1971).

The in stream conditions, that act as filters, are very different between the two ecoregions. The Central Irregular Plains were historically covered by tall grass prairies, with forests only in riparian habitats. This ecoregion has rolling topography as a result of glacially deposited soils. Streams flowing within the ecoregion are characterized by highly variable flow, sandy beds, and turbid conditions (Matthews 1988, Dodds 2004, Galat et al. 2005). The lack of rocks within these streams increases the importance of woody debris from riparian forests as hard substrates (cf. Benke et al. 1984, Hax and Galladay 1998, Quist et al. 2001). In stream conditions are considered to be challenging both historically and currently, creating assemblages of organisms with critical adaptations to high turbidity and large fluctuations in temperature and flow (Cross 1967, Matthews and Styron 1981, Bonner and Wilde 2000, Spranza and Stanley 2000). As is characteristic of streams with large variations in discharge (Poff and Allen 1995, Poff 1997), streams in the Great Plains tend to have more omnivores in their food webs.

Habitat conditions for stream organisms in the forested Ozark Highlands Ecoregion of southern Missouri differ considerably from streams in the Central Irregular Plains Ecoregion. The Ozark Highlands are characterized by underlying karst topography with steep mountains covered with deciduous forests. Streams within this ecoregion streams are famous for their clear water and cobble beds

(Pflieger 1971, Brown et al. 2005), which increases diversity (Gorman and Karr 1978).

Differences in landscapes must have led historically to very differently stream communities and food web properties in these two ecoregions, and the impacts of human land use have further magnified and/or altered these differences in community diversity and composition. Prairie streams have been strongly affected by the conversion of watersheds from grasslands to agriculture, water extraction (Matthews 1988, Dodds et al. 2004), and channelization (Vokoun and Rabeni 2003). Reservoirs have reduced turbidity and flow variability in many prairie streams which have allowed the invasion of lentic and exotic fish species (Quist et al. 2004, Falke and Gido 2006). Cyprinid species particularly adapted to turbidity and harsh conditions have become less common and have been replaced by red shiners (*Cyprinella lutrensis*) and emerald shiners (*Notropis atherinoides*) as a result of impoundments (Quist et al. 2004, Bonner and Wilde 2000). Conversion of native grasslands and forest to agriculture has occurred in both ecoregions and has decreased water quality, increased sediment loads, added nutrients, and enhanced algal productivity, all of which have promoted a shift in the community composition to pollution-tolerant species of fish and invertebrates (Quinn and Hickey 1990, Corkum 1991, Corkum 1996, Delong and Brusven 1998). For these reasons, it is important to understand how both historic and current landscape features interact to affect the nature of stream communities and their food webs.

If two streams differ in community composition and diversity, it seems likely that this should produce significant differences in their respective food webs. Few studies have been conducted looking at the effects of land use on food webs in the past (Thompson and Townsend 2005, Romanuk et al. 2006), but the effects of community structure on food webs have not been examined. As described above, the different landscape features act as filters on community composition in streams. In the prairie streams, harsh conditions create low diversity communities dominated by omnivores, whereas Ozark streams have high diversity and possibly a relatively smaller proportion of omnivores. I hypothesized that the better landscape conditions, greater diversity, and more stability in the Ozark streams will allow for higher trophic levels and longer food chain lengths.

I examined food webs of communities in eight rivers located in a multi-ecoregional (Richetts et al. 1999) grassland watershed (composed of: the Central and Southern Mixed Grasslands; Flint Hills Tall Grasslands; and Central Forest/Grassland Transitional Zone) and a forested landscape (Ozark Highlands) of the U. S. Central Plains (Fig. 1). The former includes the Central Prairie and Middle Missouri freshwater ecoregions, and the latter is within the Central Prairie and Ozark Highlands freshwater ecoregions (Abell et al. 2000). For purposes of my discussion, these rivers will be divided between grassland (Grand, Platte [in Missouri], Kansas, and Republican Rivers) and forested ecoregions (Current, Black, St. Francis, and Eleven Point Rivers). Within these eight rivers, I analyzed food chain length from fish and invertebrates data in reference to stream characteristics and the nature of the

terrestrial ecoregion and regional watershed conditions. Characteristics of the watershed were analyzed using several landscape measures from remotely sensed imagery.

Methods

Study Sites

Prairie Rivers in the Central Plains region of the U.S. Great Plains are relatively warm, turbid, and sandy. Reported average annual precipitation values for grassland ecoregion rivers were 61 cm for the Kansas River and 92 cm for the Grand River (Galat et al. 2005). Ozark streams watersheds are dominated by deciduous forests, and many are within the Mark Twain National forest. The geomorphic features of Ozark watersheds are typically characterized by uplifted limestone, sandstone, and both shale and limestone karst topography. Stream beds commonly contain large amounts of cherty limestone gravel, and the waters are less turbid than those in grassland rivers. Rivers of the Ozark ecoregion normally receive over 100 cm of precipitation per year (Brown et al. 2005).

Sample Collection

Stable isotope samples were collected from eight rivers in 2003 and four rivers in 2005. Invertebrates were collected using D-nets in rocky and snag habitats and were stored in jars on ice for transport to the lab. Invertebrates were left in aerated water tanks in the lab for 24 hours to allow their guts to clear before being frozen. Fish were collected by electrofishing or with seines for smaller fish. To eliminate the possibility that body size effects trophic position, only adult fish were used for isotope analyses. White muscle tissue was extracted behind the dorsal fin

using a tissue sampler (Fischer Catalog), stored on dry ice during transport to the lab, and later frozen in the lab until processed for stable isotope analysis.

Sample locations were picked to minimize human impact. However, some samples were collected below reservoirs in the case of the Grand and Kansas River, above a reservoir in the Republican River, and below an urban area in the Kansas River. Samples were collected from September to November 2003 and July to October 2005. Temperature, conductivity, pH, and dissolved oxygen were measured using Hydrolab® sonde in 2003. Turbidity and chlorophyll-*a* were measured using Turner® meters in 2004. Chemical and physical measurements were collected along the transects used for electrofishing. Samples in 2003 were collected as part of an EPA study of the effects of watershed condition on stream communities. Four reaches were either electroshocked for 600 seconds or seined in each of the eight rivers. Large specimens were identified in the field and released while small specimens were frozen on dry ice and taken to the lab to be identified in the lab. Invertebrates for the diversity study were collected with D-nets along similar reaches and later identified to the family level.

Lab Processing

Fish and invertebrate samples were thawed, dried in the lab at 50-60ºC for 40-48 hrs, ground to a fine powder using a Wiggle-L-Bug® , and weighed to the nearest 2 micrograms in tin capsules prior to isotope analysis. The invertebrate samples had been previously identified to family and rinsed with distilled water before drying.

Most were ground whole, but only the tail muscle was used for crayfish (which were identified to genus). Individuals of similar size were pooled together for all invertebrates. Whole snail and mussel samples were removed from shells and acidified to remove traces of inorganic carbon. Samples were packed into silver capsules and a drop of distilled H_2O was added. A Petri dish of 1 N HCl was placed in a desiccator with the samples and left for 24 hours. Samples were redried, ground, and weighed for analysis. Small fish were identified to species, and the whole dorsal muscle tissue was sampled for isotope analysis. As described earlier, tissue from the dorsal white muscle of larger fish species were extracted in the field and returned to the lab for processing.

The stable isotope ratios of all samples were determined at Kansas State University on a ThermoFinnigan Delta plus mass spectrometer with dual inlet and continuous flow. The precision levels were $\pm$ 0.3 per mil (1 sigma) and $\pm$ 0.2 per mil (1 sigma) for $\delta^{15}N$ and $\delta^{13}C$, respectively. Sample values for $\delta^{15}N$ ($^{15}N/^{14}N$) and $\delta^{13}C$ ($^{13}C/^{12}C$) were reported as parts per thousand (‰) in comparison to standards for atmospheric N and PeeDee Belemnite standard. Samples were not run in duplicate.

Data collected from the 2003 electrofishing surveys were categorized into trophic guilds and families based on Pflieger (1997) and Cross and Collins (1995) feeding descriptions. Diversity measures, number of fish caught (N), alpha diversity, species richness, and Simpson's-D were also calculated from this data. Non-carp cyprinids were separated into trophic groups as well. One final category was to arrange fish that typically are less than 15 cm at adult size into a group called small

fish to assess their importance to trophic position. This included many representatives

of Cyprindae (no large carp species), Fundulidae, Percidae (but not perch), Cottidae,

Gambusia affinis, and *Labidesthes sicculus.*

Trophic Levels

Food chain length can be measured by different methods, but stable isotopes

have proved to be a useful technique for measuring trophic position and maximum

food chain length of piscivorous fish in streams (Vander Zanden and Rasmussen

1996, Post and Takimoto 2007). The advantages of stable isotopes over other

methods, such as analysis of gut contents, are that the former is a measure of what has

been assimilated rather than what has been merely been consumed. Nonetheless,

trophic position calculated by stable isotope method agrees well with trophic position

computed from gut analysis (Vander Zanden et al. 1997). Studies have found that an

organism's $\delta^{13}C$ ($^{13}C/^{12}C$ ratio) reflects their food source, but there can be a shift of

1‰ when moving from one trophic level to the next (Post 2002, DeNiro and Epstien

1978). Fractionation differs within different tissues. Fractionation of $\delta^{13}C$ in muscle

(fish) was closer to 1 ‰, while whole organism (invertebrates and mussels) were

closer to 0.3 to 0.8 ‰ (Vander Zanden and Rasmussen 2001, McCutchan et al. 2003),

but $\delta^{13}C$ fractionation was not found to influence trophic position (Post 2002).

DeNiro and Epstein (1981) noted that the $\delta^{15}N$ ($^{15}N/^{14}N$ ratio) increased from one

trophic level to the next, but the amount of fractionation between levels varies

considerably among different trophic levels (Post 2002). However, an average

fractionation of 3.4 ‰ over the whole food web is an acceptable average (Vander

Zanden and Rasmussen 2001, Post 2002, Peterson and Fry 1987, Minagawa and

Wada 1984). I determined trophic position using models from both Vander Zanden

and Rasmussen 1996 and Post (2002b). In the Vander Zanden (1996) model trophic

position is equal to: ($\delta^{15}N_{consumer}$- $\delta^{15}N_{baseline}$)/3.4) +2. For this model $\delta^{15}N_{baseline}$ is

assumed to be from the same nitrogen sources as the $\delta^{15}N_{consumer}$ of fish. $\delta^{15}N_{baseline}$

was estimated by using snails (families Physidae or Pleuroceridae), which were found

in all rivers. Post (2002b) used both sestonic and benthic baseline sources to estimate

nitrogen determine trophic position. Post (2002b) model used a linear mixing model

to determine trophic position

$$\lambda + (\delta^{15}N_{sc} - [\delta^{15}N_{base1} \times \alpha + \delta^{15}N_{base2} \times (1-\alpha)])/3.4.$$

where α is the proportion of a food source and is calculated by:

$$(\delta^{13}C_{sc} - \delta^{13}C_{base2})/(\delta^{13}C_{base1} - \delta^{13}C_{base2}).$$

The $\delta^{15}N_{sc}$ or $\delta^{13}C_{sc}$ of the organism of interest was compared to baseline $\delta^{15}N$

or $\delta^{13}C$. The $\delta^{13}C_{base1}$ is from snails (families Physidae for Prairie rivers,

Plueroceridae for most Ozarks rivers except Valvidae for the Eleven Point River).

The second base is either the Asian clam (*Corbicula fluminea*) for the Ozark rivers or

filter-feeding caddisflies (Hydropsychidae) in the Kansas and Platte rivers. These

aquatic insects were used because neither *Corbicula* nor unionid mussels were

sufficiently abundant. Hydropsychidae species have been found to consume large

amounts of detritus and animal matter (Benke and Wallace 1980), but some

Hydropsychidae consume more algae in downstream reaches (Sheih et al. 2002).

Using Hydropsychidae as a base may not completely represent the nitrogen from algal sources, but is the only baseline found that was sufficiently abundant in the two grassland rivers. Only Corixidae were present in the Republican and Grand Rivers, and so these rivers lacked a second base. Only the Vander Zanden model was used for these rivers.   Best effort was made to select similar organisms in each river (Anderson and Cabana 2007).

Some fish or invertebrates were found to have different $\delta^{13}C$ than their baselines, and so the alphas were either greater than 1 or less than zero as calculated by the Post 2002b model. Samples that have a calculated alpha that exceeded these numbers are probably not well represented by the baselines, and the nitrogen source of their diets may have been from a different location.   Alphas were corrected by changing all those greater than one to one and those less than zero to zero (Post *personal communication*).   Organisms with alphas greater than 4 or less than -3 were considered to be too poorly represented by the nitrogen sources and were removed from analysis.

Another correction was for lipids in fish muscle tissue and whole invertebrates. The $\delta^{13}C$ of lipids tends to be depleted in comparison to muscle or other tissue, and is important if the organism contains high lipid content (DeNiro and Epstein 1977, Post et al. 2007). Lipids were corrected using Post's 2007 regression model for aquatic organism.

$$\delta^{13}C = -3.32 + 0.99 \text{ x C:N}$$

Where C:N is calculated by %C/%N as provided with stable isotope analysis.

Fish C:N was usually close to 3, but some fish had ratios of as high as 7. The C:N ratios of some invertebrates and baseline sources were greater than 7, but the highest C:N were less than 9. The highest C:N includes Hydropsychidae from the Platte River. Invertebrates with high C:N were not corrected using the chloroform extraction.

Remote Sensing and Landscape Measures

Different measurements of landscapes were considered in this study including remote sensing, land use and land cover and landscape pattern metrics (Table 1, Fig 2.). Watershed land use has major impacts on in stream characteristics and stream organisms (Roth et al. 1996, Wang et al. 1997, Wang et al 2006). Other researchers have concluded that riparian buffer condition was a better predictor than watershed land use (Carter 1996, Lammert and Allen 1999, Parsons et al. 2003). While buffers can influence amount of erosion and the amount of fine sediments, whole catchment geology and land use can impact stream morphology, habitat, and stream organisms (Richards et al. 1996). Watersheds with intense agriculture can overwhelm the influence of intact riparian buffers (Wang et al. 2006). Because many streams in the Great Plains are highly impacted by agriculture, watershed land use should have an impact on stream communities more than riparian land use. Therefore, this study does not examine the effects of riparian buffer. Instead, whole watershed land use and smaller subcatchments above the sample point were used to look at the effects of land use.

To determine the effects of the size of the subcatchment above the sample point, different size watersheds were created using 12 digit HUC watersheds aggregated together to form larger watersheds in Arc-MAP. The smallest extent included the remaining watershed above the sampling location and the next HUC above it (2 HUCs). The largest whole watershed for land use was the entire watershed above the sampling point. The largest watershed for remote sensing and landscape metrics were composed of 21 HUC's. These included both the river itself and smaller streams to form similar sized subwatershed basins. The next smaller aggregation was 11 HUCs, and then 4 which included the main channel and tributaries. The subwatershed HUCs were appended to the aggregations to have the correct area, and polygon borders were dissolved to form one polygon.   These were put in a mask to extract NDVI and VPM statistics.

Remote sensing imagery data was taken from the Advanced Very High Resolution Radiometer (AVHRR) satellite recorded over a 15-year period (1989-2003) to calculate vegetation greenness as a measure normalized difference vegetation index (NDVI) and vegetation phenology metrics (VPMs) (Reed et al. 1994). NDVI is the normalized ratio between the absorbance of the red wavelength of light by chlorophyll and the reflectance of the infrared by moisture content and structural components in the leaves (Myneni et al. 1995). Healthy green vegetation should have an index close to one. NDVI was selected because of its ability to monitor changes in vegetation over time, and its ability to be compared to biological processes (Kerr et al. 2003, Pettorelli et al. 2005). NDVI has correlated well with

climatic data, actual evapotranspiration rates, and net primary productivity (Goward et al. 1985, Box et al. 1989, Rundquist et al. 2000). Because of its ability to measure watershed vegetation, NDVI and its metrics have been found to have stronger correlations with water quality parameters such as nitrates, phosphorous, conductivity, and turbidity than using traditional land use and land cover data and landscape pattern metrics. NDVI is typically calculated more frequently than the often long-period land-cover maps, thus allowing seasonal changes in the watershed to be observed (Griffith et al. 2002a). NDVI as a remote sensing tool is useful because it reduces sun angle illumination differences, cloud shadows, atmospheric attenuation, and topographic noise. However, NDVI is sensitive to some atmospheric effects, and soil background, and it saturates at when vegetation is very thick (Goward et al. 1991, Jensen 2005). NDVI were calculated from biweekly composites of cloud free pixels. Vegetation phenology metrics (VPMs) were also calculated from the biweekly NDVI values, and include: date of onset of greenness and NDVI, rate of greenup, Max NDVI and date of maximum NDVI, rate of senescence, date of end of greenness and NDVI of end of greenness for 2003. The 15 year average included average maximum NDVI, average rate of green up, average rate of senescence, accumulated growing season NDVI and average growing season NDVI. VPMs were calculated using modified methods by Reed et al. (1994).

Watershed boundaries were delineated using Digital Elevation Models and the USGS Hydrologic Unit Codes HUCs into polygons. Polygons were reprojected in the Lambert azimuthal equal area projection, converted to raster format, and cell sizes

were converted to 1000 m by 1000 m grid so that AVHRR pixels matched with stream polygons. These watershed polygons were converted to a mask to extract NDVI. Percent land cover was calculated from the 1992 National Landcover Database (NLDC; Vogelmann 1998) at the 1000 m resolution from the original 30 pixels. The NLDC pixels were reclassified into percentages for forest, cultivation, grassland, urban, water and wetland. A binary method was used to determine the average percent of each land cover class in the 1000 m pixels.

Landscape pattern metrics (LPMs) were selected because the amount of fragmentation including how many patches, their size and shape may be just as important as the amount of land use on the watershed. For instance in urban (Kearns et al. 2005) and agricultural environments (Cifaldi et al. 2004), the number of patches and the shape of the patches were found to be very important measures of landuse change. Landscapes often change from simple landscapes to those that are more heterogeneous with smaller patch sizes (Cifaldi et al. 2004, Kearns et al. 2005). LPMs were calculated as Fragmentation Indices using Fragstat 2.0 from the 2001 NLCD dataset downloaded from the Environmental Protection Agency (NLCD) website. Metrics were calculated for 2 HUCs and the 21 HUCs. Metrics chosen were total core area (TCA), number of patches, mean patch size (MPS), and edge density (ED) for the dominant land cover types (agriculture, grassland, and forest). The number of wetland patches was only measured in the 2-HUC extent size. Average-weighted mean patch fractal dimension (AWMPFD), area-weighted mean shaped index (AWMSI), and interspersion and juxtaposition index (IJI) were selected as

measures of shape and distance from other patches. These variables were averaged

for all patch types available. These metrics were selected *a priori* based on the

literature emphasis on those that explain patterns in landscape change (Griffith et al.

2002b, Cifaldi et al. 2004; Kearns et al. 2005).

Variables explaining Food Chain Length

Trophic position was compared to theories of food chain length: productivity,

disturbance and ecosystem size. Productivity was estimated using sestonic

chlorophyll-*a* concentrations (μg/L), which is a proxy for algal biomass (Hauer and

Lamberti 1996). Disturbance was measured using coefficient of variation of mean

daily discharge (Colwell 1974, Poff 1997). Disturbance variables (CV of discharge)

were determined from discharge data. Discharge data were downloaded from the

United State Geological Survey's National Water Information System (NWIS) for the

sample location. Daily average discharge was averaged over a 25-yr period from Dec.

1, 1978 to Dec. 1, 2003 for the discharge variable. Variability of disturbance was

calculated using coefficient of variation of discharge. Ecosystem size was measured

using discharge (m^3/s), watershed size (m^2), dendritic length (m), and density.

Discharge was used from NWIS website and averaged over 25 years to account for

natural variation. Size was also the area of the whole watershed polygon above the

sampling location. Within the stream polygon, dendritic length (stream length) and

density were calculated. Total nitrogen and phosphorous were provided by Central

Plains Center for Bioassessment (CPCB) at the University of Kansas.

Statistical Analyses

Because data did not meet the normalcy and equal variance requirements for ANOVA, the Mann-Whitney test was used for multiple comparisons. Principal Components Analysis was used to reduce the number of variables using standardized variables and correlation matrices. Trophic guilds and family composition were used as percent of total community to weight rivers with large numbers of cyprinids or omnivores, but had few fish. In the whole watershed, forest, grassland, cultivation, wetland and urban areas were all variables included in the Principal Component Analysis because of their high correlation with each other (King 2005). The first two Principal Components were usually considered if the factor loadings were biologically meaningful. Pearson's correlation coefficient was calculated for trophic position, water quality, river characteristics, and watershed VPMs using SPSS.

Principal Component scores were not calculated for fragmentation metrics to understand each metrics role in explaining variance in community structure and food chain length. Pearson's correlation coefficient was calculated for trophic position, water quality, river characteristics, and watershed land use using SPSS v. 15.

Results

Landscape Differences Among Ecoregions

Landscapes of these two ecoregions were very different. Watersheds surrounding Ozark ecoregion rivers were dominated by forests with high NDVI levels, whereas watersheds in the Prairie ecoregion consisted mostly of grasslands and cultivated land with lower NDVI (Figs. 3-4). The Ozark ecoregion rivers were mostly forested, but some had more cultivation and grassland than was expected. Water quality conditions in both ecoregions were quite different as well. As expected, prairie rivers were much higher in turbidity, nutrients, chlorophyll-*a*, and temperatures, while Ozark rivers were clearer and colder (Table 2). These water quality conditions could be attributed to some degree to land cover. Nutrients, especially total phosphorous, increased with the amount of cultivation on the whole watershed (Pearson Correlation, $r = 0.843$, $p<0.05$). Total nitrogen data was available in too few rivers to correlate with land use. Concentrations of Chl-*a* were linked to higher nutrient values (total phosphorous, $r = 0.938$, $p < 0.01$) and to percent cultivation, wetlands and urban areas (Whole Watershed PC1) (Pearson Correlation, $r = 0.889$, $p <0.01$) (Table 3 and 4). Temperature, turbidity were also associated with whole watershed PC1 (Pearson Correlation, $r=0.767$ and 0.829 $p < 0.05$), Conductivity and pH (Waterquality PC2) were highly associated with whole watershed PC2, which was highly associated with grasslands (Pearson Correlation, $r = -0.884$ and -0.928, $p<0.01$). The $\delta^{15}N$ values of fish were related to land cover characteristics as well. Values of $\delta^{15}N$ in fish were strongly with NDVI, LPMs, and

land use (Pearson Correlation r<0.7, p<0.05). However, there was no detectable relationship between baseline $\delta^{15}N$ with percent land use though many relationships were close to significant (Pearson Correlation, r = 0.6, p>0.05). Mean $\delta^{15}N$ of fish and baselines in the prairie ecoregion rivers were significantly higher than in Ozark streams for both years (Table 2) (Mann-Whitney U test, Z = -13.1 and -9.6 for fish, and -6.1 and -4.1 for baselines p< 0.001). From a temporal perspective, $\delta^{15}N$ of fish did not change between years in either ecoregion, but the Kansas baselines were enriched in 2005 (Mann-Whitney U test, z = -0.178,   z = -0.833 for fish, and z=-1.18 and z=-1.98 for baselines p>0.05). As expected, watershed land cover and river water quality of the Prairie and Ozark ecoregions were different.

Possible Effects of Landscape Characteristics on Fish Communities

Land use also appeared to impact fish and invertebrate diversity. Increased whole watershed land use caused a reduction of species richness, alpha diversity, and Simpson's reciprocal D (Diversity PC1) at the whole watershed size extent (Pearson Correlation, r = -.875, p<0.01, Fig. 5a), and smaller watershed extents as well (-0.864 to -0.917, p<0.01). Diversity PC1 was also highly correlated with most of the other landscape measures (Table 4). No relationship existed between diversity and PC2, which was associated with grasslands. Diversity PC1 decreased when water quality PC1 increased, but the relationship was not significant (Pearson Correlation, r=-0.503, p>0.05) (Fig. 5b). Invertebrate diversity was also highly related to NDVI,

LPMs, and watershed land use. Lastly, no watershed measurement correlated with the number of fish caught (PC2).

Trophic and Fish Community Differences Among Ecoregions

Trophic guild structure was also related to landscape patterns. Omnivorous fish dominated the community within prairie streams, while invertivorous species constituted a greater percentage of the Ozark streams (Fig. 6). Looking at the δ^{15}N and δ^{13}C biplots between the ecoregions, fish within these trophic guilds in the Ozark ecoregion formed more distinct trophic levels than in the Prairie ecoregion, especially between piscivorous fish and invertivorous fish (Fig. 7). The range of δ^{13}C and δ^{15}N was greater for Prairie streams than Ozark streams, but the standard deviation of δ^{15}N of fish and invertebrates was low, especially in piscivorous fish. Omnivorous fish δ^{13}C and δ^{15}N was more variable and was less likely to fall into a distinct trophic level (Fig. 7-8). While trophic positions of all trophic guilds appeared to be higher in Prairie ecoregions, they were not necessarily significantly different (Fig. 8). The trophic position of piscivores was significantly higher than any other trophic group (Mann Whitney U, $p<0.05$). Trophic position of piscivores were significantly different between ecoregions (Mann Whitney U, $p<0.05$), but piscivores in the same ecoregion did not differ between years (Mann Whitney U, $p>0.05$). Ozark omnivores, invertivores and planktivores differed from Prairie trophic guilds in 2003 (Mann Whitney U, $p<0.05$), and only planktivores were different between years (Mann Whitney U, $p<0.05$). Other trophic guilds were not significantly different from each

other in the same ecoregion expect the omnivores from the insectivores in the Ozarks and the planktivores from the omnivores and insectivores in the Prairie ecoregion. (Fig. 8). Trophic guild structure also highly related to many of the landscape measurements (Table 4). Whole percent land use and increased cultivation were associated with increased the numbers of omnivores (Pearson Correlation r=-0.899, p<0.01), but the relationship between trophic guild composition and water quality was not significant (Pearson Correlation, r=-0.653, p>0.05). The fact that the number of metrics correlated with trophic guilds shows that land use has led to major differences between trophic guild communities that may be related more than simply degraded water quality.

Fish family structure was different between the two ecoregions as well (Fig. 9). Catfish in the family Ictaluridae and *Lepisosteus spp.* were more common in prairie rivers, while Centrarchidae, Cottidae, and darters in the Percidae family were more common in Ozark streams. The bass and sunfish family Centrarchidae was the second most abundant taxa, but less so in prairie rivers. Suckers in the family Catostomidae were important in all rivers, but were a smaller percentage. Most rivers sampled had a small percentage of *Aplodinotus grunnienss* and *Dorosoma cepedianum*. *Gambusia affinis* dominated the Kansas River. These differences in fish family community structure were also evident looking at family community structure PC1 and PC2 (Table 3, Fig. 9). While many landscape measures were related to diversity and trophic guild structure, very few measures were related to family community structure (Table 4). Water quality PC1 was strongly related to fish

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community structure PC1 (Pearson Correlation, r=0.818, p<0.05), but only whole watershed land use was related to fish community structure (Pearson Correlation, r=0.824, p<0.05, and other watershed sizes were not significant. Other aspects of landscapes besides simple land use were probably important.

The family Cyprinidae was the numerical dominants in all rivers (Fig. 9-10). Prairie streams had more *Cyprinus carpio*, but they also had a large number of minnows and shiners. *Cyprinella lutrensis* composed almost the entire cyprinid communities in the Republican, Kansas, Grand, and Platte Rivers (Fig. 9). *Notropis atherinoides* represented 5% of the cyprinid taxa in Prairie ecoregion. Other species composed a small percentage of the fish communities in the Kansas and Grand Rivers. Ozark rivers had many more species of cyprinids and no carp species (Fig. 10). Fifteen species of cyprinids were caught in the Current River. *Notropis atherinoides* formed 20-30% of the minnow community in the Black and St. Francis Rivers. Other important fish were the *Cyprinella venusta* in the Black River, *Campostoma* spp. and the *Notropis nubilus* in the Current River, *Luxilus zonatus* in the Eleven Point River, and the *Notropis boops* in the Saint Francis River. *Campostoma* spp. were found in all Ozark rivers.

Trophic Position

The two ecoregions were different in terms of trophic structure and family structure, and so trophic position of common species in different ecoregions and rivers were compared (Table 5). Fish with the highest $\delta^{15}N$ in the Ozark ecoregion

were *Micropterus salmoides, M. dolomieu*, *M. punctulatus*, and *Ambloplites ariommus*, and these fish also had high trophic positions (Fig. 7, Table 5). Fish in the Prairie ecoregion rivers with the highest $\delta^{15}N$ and trophic position were *Lepisosteus spp.*, *Pomoxis annularis, Pylodictis olivaris*, and *Aplodinotus grunniens*. *Pomoxis nigromaculatus* had the highest trophic position analyzed in the Black River, but was near trophic level 2. *Lepisosteus spp.* had the highest trophic position of the species caught in the Kansas River. The trophic position of *Micropterus salmoides* was the highest in Ozark rivers in this study. The Saint Francis and Current Rivers were not different from each other or did rivers differ between years. The trophic position of *Micropterus salmoides* in the Eleven Point was significantly higher than in the other Ozark ecoregion rivers. Fish from the Black and Platte Rivers were significantly lower than other rivers. *Moxostoma erythrurum* caught in the Black River were statistically lower than in the other Ozark streams. *Notropis atherinoides* and *Carpiodes carpio* in the Platte River had a lower trophic position than in other prairie rivers.

Non-piscivorous fish within the Prairie ecoregion had much higher $\delta^{15}N$ than expected, especially fish in the Grand River. Most fish in the Grand River had a trophic position of 4.5, including *Cyprinella lutrensis*, *Carpiodes carpio*, and *Aplodinotus grunniens.* These were significantly higher than the same fish in other rivers of the same ecoregion. Trophic positions of fish caught in the Republican River in 2005 were also near 4, and significantly higher than the Kansas River in 2005.

Some fish had lower than expected trophic levels such as *Dorosoma cepedianum* and invertebrates. Fish caught in the other rivers had trophic positions around 3.

Fish caught in the same river or ecoregion in different years often had trophic positions that were not statistically different (Table 5). Gar were significantly different between years, but the difference was only 0.2 of a trophic level. *Pylodictis olivaris* was not significantly different between ecoregions, whereas the omnivorous *Ictalurus punctatus* was different between ecoregions in 2003 but not in 2005. The invertivore *Lepomis macrochirus* and particulate feeding *Dorosoma cepedianum* and *Notropis atherinoides* were significantly different between ecoregions. While many fish differed between ecoregions or rivers, fish often kept the same trophic position between the two years studied.

Mean trophic position of fish was higher in rivers from the Prairie ecoregion than from the Ozark ecoregion for both years (Mann Whitney U z=-4.5 and z=-3.3; p<0.001); (Table 6). Trophic position was not different between years for the same ecoregion (Mann Whitney U z=-1.4; z=-1.3 p>0.05) and varied among rivers in general. (ANOVA, F=66.6, p<0.001). Trophic positions were similar in the Current and Saint Francis Rivers and also in the Kansas and Republican Rivers in 2003 (Mann Whitney U, z=-3.0, -1.2; p>0.05). Trophic positions of fish within the Black river were significantly lower than in other Ozark rivers (Mann Whitney U, p<0.001). Trophic positions in Platte River fish were lower than in Prairie rivers (Mann Whitney U, p<0.001), and the Grand was especially high (Mann Whitney U, p<0.001). These differences between the rivers could be important.

While there are differences of trophic position among rivers, the relationships between community structure and trophic position were weak. Trophic position diversity (PC1) and number of fish caught (PC2) increased, but the relationship was low and insignificant (PC1 r=-0.084, PC2 = 0.444, p>0.05) (Fig. 5 c). Considering that the most abundant family was Cyprinidae, it would seem that there would be an effect on trophic position, but no correlation was evident (Pearson correlation, p>0.05). Also, many other small fish species (i.e. *Gambusia affinis*) were present, and so the number of small fish of the community could be important, but the relationship was insignificant. (Pearson Correlation, r = 0.349, p>0.05) (Fig. 5d). Fish family community structure and trophic guild structure were different between ecoregions, but trophic position was not related to either (Pearson Correlation, p>0.10) (Fig. 5e-f). Even though diversity and fish trophic structure was impacted by land use, trophic position was not (Table 4) nor did watershed size alter this relationship. Maximum trophic position was not significantly related to water quality (Fig. 5g) nor with metrics of NDVI or LPMs at any watershed size scale.

None of the most prominent theories on food chain length proved useful in explaining the results found here for differences among rivers in trophic position. There was no significant relationship between maximum trophic position and Chl-*a* (r= -0.019, p>0.05) (Fig. 5g). Measures of ecosystem size, such as the amount of average stream discharge, did not relate to the maximum or mean TP (Fig. 5h). (Pearson Correlation r=0.310, and PC2=0.386 p>0.05). The relationship between maximum trophic position and disturbance (coefficient of variation) was not

significant (Pearson Correlation r=0.191, p>0.05) (Fig. 5i). Fish community structure weakly decreased with increasing disturbance, but was not statistically significant (Pearson Correlation, p>0.05) (trophic guild composition (r=-0.344), fish family composition (r=0.392), fish diversity PC1 (r=-0.692), invertebrate family richness (r=-0.513), and family EPT numbers (-0.414)). The prevailing theories of trophic position were not related to food chain length.

Effects of Baseline

Fish $\delta^{13}C$ should fall within the range of the $\delta^{13}C$ of the herbivorous baseline to adequately capture the nitrogen signature of the food source. Some river fish were well within the range of the baseline, while others appeared to not fall within the baselines as evident by the Post (2002b) alphas (Fig. 7, Table 6). The Current, Kansas, Eleven Point, and Saint Francis Rivers all had a mean alpha between one and zero, meaning that the baseline reflected the food source of consumers well. The average alphas for fishes in the Black and Platte Rivers were greater than one, indicating that they were not closely linked to the baseline values used in the calculations. Baseline $\delta^{15}N$ and $\delta^{13}C$ have important impacts on trophic position, and so it is important to address how well the baseline captured the source of nitrogen.

Effects of δ¹⁵N and Baseline on Trophic Position

The method used by Post (2002b) and Vander Zanden and Rasmussen. (1996) to calculate trophic positions for fish are only partially similar (Table 6). Using Post's method led to lower trophic position in the Eleven Point River in comparison to using Vander Zanden's model. Only Corixidae was available as a second baseline in the Grand and Republican Rivers, so it is not possible to compare the two methods. Using a Wilcoxon Signed Ranks Test for fish and invertebrates for both years showed that both methods were significantly different ($Z = -4.97$, $p<0.05$), and not correcting for lipids led to differences as well (-2.02, $p<0.05$) However, the differences between the Vander Zanden's method using snails and Post's (2002b) approach with corrected lipids led to small to moderate differences in trophic position. Not correcting for lipids often led to much lower trophic position for many rivers (Post 2002b, Post et al. 2007). While I cannot be sure of the exact trophic position of our samples, I have estimated it closely as possible.

Effects of landscape measurements and size of watershed

NDVI, landscape pattern metrics and percent land use were all compared to water quality conditions and stream community structure. As far as land pattern metrics, Ozarks had greater mean patch size and total core area, but more patches and great edge density. Shape (AWMPFD and AWMSI) were comparable for most rivers and IJI fell in the middle of the range of this metric (Table 7). While shape metrics did not appear to be very different, they were significantly related to many stream

community variables (Table 4). Those that were most important were total core area, number of patches and mean patch size and edge density were positively associated with many of the variables. Forest and cultivation patch types were important for stream diversity and community structure, but grassland patches were important for water quality.   For landscape pattern metrics, total core area, number of patches and mean patch size was important for both water quality PCs, but conductivity and pH were most frequently associated with LPMs for the 2 HUC and 21 HUC. Family community structure was related to total core area, number of patches, mean patch size, AWMSI and AWMPFD and IJI. Trophic guild structure and invertebrate diversity was highly related to nearly all LPMs. LPMs were able to capture the effects of land use on stream organisms.

NDVI was also effective. The onset of greenness often occurred earlier in the Prairie ecoregion than in the Ozark ecoregion, especially for the Republican River. NDVI was much higher in the Ozark ecoregion than the Prairie ecoregion (Fig. 4). NDVI measures such as start NDVI, maximum NDVI, accumulated growing season, average of the growing season and 15 year NDVI were those that correlated with the most variables (Table 4).   Conductivity and pH (Waterquality PC2) were correlated to more NDVI variables than temperature and dissolved oxygen (Waterquality PC1). Start week, Max NDVI and average growing season were important for conductivity (PC2), but max week was important for water quality PC1 for all HUCs. Fish δ^{15}N, was highly associated with most NDVI and VPMs, but there was no relationship with base δ^{15}N. Start NDVI, max NDVI, accumulated growing season, average growing

season, and 15 year NDVI PC 1 were all related to increased number of omnivores (Trophic Guild PC1) (Mann Whitney U, $p<0.05$). Invertebrate Family Richness, and Invertebrate EPT number were all highly associated with rate of green-up, accumulated growing season, average growing season, and 15-year average NDVI PC (Pearson Correlation, $p<0.05$). Looking at land use and land cover, all measures of community structure were highly correlated with land use PCs. Percent land use was best at explaining variation in diversity, trophic guild structure, $\delta^{15}N$ of fish and nutrients. Percent forest was more effective in explaining variation than grassland for all landscape measurements. Larger size watersheds were explaining more variation than smaller watersheds. These watershed variables all explained variation in water quality, fish and invertebrate diversity, trophic structure, and fish family structure, but not trophic position.

Discussion

Effects of Land Use on Diversity and Food Chain Length

Landscape differences have led to changes in water quality and animal diversity but not necessarily to changes in food chain length in this study. Water quality changes were often due to differences between forest and cultivated watersheds, but the amount of grasslands was less important. As found in other studies, increased cultivation was found to be related to higher nutrients, turbidity and chlorophyll. Diversity was strongly negatively correlated to increased cultivation (Quinn and Hickey 1990). Watershed landuse explained changes in diversity and fish trophic guild structure, but not the number of fish caught and fish community composition. In the Platte River, where the watershed had a large agricultural component, food chain length, fish abundance, and diversity were the lowest examined. Food chain length was very low in the Black River and few fish were also collected from the Black River in comparison to other Ozark ecoregion rivers, but its watershed was heavily forested and diversity was not necessarily lower than other rivers. Urban development was high in this river, but it was also very high in the Kansas River, which had long food chain lengths. Therefore, watershed land use impacted fish diversity and the trophic guild structure, but food chain length was not necessarily impacted by the loss of biodiversity as a result of land use.

A positive relationship between food chain length and species richness has been hypothesized (Bengtsson 1994). However, many other studies have also found that the loss of stream organisms as a result of water quality degradation did not

necessarily shorten food chain length. A study comparing small streams impacted by acid mine drainage found that impacted streams were less diverse but had similar mean food chain length compared to undisturbed streams (Quinn et al. 2003). A longitudinal study comparing river food webs from the headwaters in the mountains to forests to grasslands found that all systems had four trophic levels (Romanuk et al 2005). Even in lakes, land use altered species richness, but not food chain length (Lake et al. 2001). New Zealand watersheds that were predominately used for pastures had greater productivity, species richness, and mean food chain length in small streams than those covered primarily by native grassland or forests (Thompson and Townsend 1998, 2005). Losses of species diversity did not cause changes of food chain length in these cases. Only in large oligotrophic lakes did fish diversity explain a large amount of variation in food chain length (Vander Zanden 2000). It appears that while land use has demonstrable links with diversity and trophic guild community structure, the effects of land use on food chain length are less measurable.

Diversity itself then must not be a direct mechanism for variation in food chain length, and more complex processes must be occurring. Mechanisms for food chain lengths are a result of predator-prey interactions within the food chain. Fish are gape limited and prefer food that is less than the mouth width (Hambright 1991). Fish predators are opportunistic and feed at many trophic levels to optimize the amount of energy by feeding on the largest and most available prey (Werner and Hall 1974, Beaudoin et al. 1999, Sih and Christianson 2001), and most productive food chains

(Layman et al. 2005 a, b). The feeding behavior is related to the species of fish, and the types of fish present are determined by landscape features.

This study found that cyprinids were the most numerous fish family, especially shiners and minnows. Due to their size and numbers, it is hypothesized that this group represents an important energy flow in rivers. Cypriniformes were found to be important food sources in the gut contents of *Lepisosteus* spp., *Micropterus salmoides*, and *Pomoxis annularis* in the Kansas River (Cross et al. 1982). Rivers that lack many fish, such as the Black and Platte Rivers, were found to have the lowest trophic levels. While the relationship between the number of fish caught was not significant with trophic position, it appears that it is important. Small predator-prey body size ratios were important for longer food chain length in oceans (Jennings and Warr 2003). When few fish are present as a result of land use, invertebrates may be contributing to the diets of piscivorous fishes, lowering trophic levels. This has been found to occur with pike in lakes, where pikes feed on invertebrates when few fish are available (Beaudoin et al. 2001). It appears that the number and types of small fish available is important for variability in the food chain. The lack of significance between the number of fish caught may be related to trophic structure.

The trophic guild structure of cyprinids may explain that lack of relationship with the numbers of cyprinids caught. As found by Poff and Allen (1995) in northern streams, the hydrologically variable prairie ecoregion rivers were dominated by omnivorous minnows, while the more hydrologically stable rivers of the Ozark ecoregion were dominated by invertivorous minnows. The omnivorous *C. lutrensis*

was the most common cyprinid in this study and have been found to compose nearly 90% of fish communities in most Central Plains streams. These fish are particularly tolerant of prairie stream conditions and degraded streams (Cross 1967, Marsh-Matthews and Matthews 2000b). The omnivorous *C. lutrensis* was found to prey heavily on aquatic insects, but also consumed fish larvae (Ruppert et al. 1993), which may explain the high trophic level of *C. lutrensis* in this study. Another study found that the trophic position of *C. lutrensis* was 2.7 (Franssen and Gido 2006), which is much lower than this study. A high trophic position of a very common cyprinid would lead to a high trophic position in the top piscivorous fish. Therefore, even though the Grand River had fewer cyprinids than were present in Ozark rivers, the large number of *C. lutrensis* may have increased food chain length in this river. The Platte River had too few cyprinids to support piscivorous growth. Land use and other factors have favored the growth of the pollution tolerant *C. lutrensis*, and they are less likely to be lost if the stream is degraded, unlike more pollution intolerant cyprinids in the Ozark rivers.

Other fish that are common in all rivers feed lower in the food chain and could decrease food chain length. The planktivorous *Notropis atherinoides* are also common in many streams, though they have been found to consume stream insects, and so they frequently have trophic positions near 3 (Franssen and Gido 2006). Other important species within the Ozark ecoregion are the herbivorous stonerollers and various insectivorous cyprinids. *Campostoma* spp. consume algae and detritus, but are known to eat macroinvertebates (Evans-White et al. 2001). These fish appear to

feed lower on the food chain, decreasing overall food chain length of the Ozark ecoregions. In general, small cyprinids have trophic positions between 2 and 3 (Franssen and Gido 2006). The trophic guild designation of the most common cyprinid should be very important for the trophic position of the top piscivore, and should be considered. The composition of cyprinids is determined by landscape features. Cyprinids are more like to be herbivorous in the Ozark ecoregion, therefore lowering the food chain length. Prairie ecoregion cyprinids are omnivores or detritivores. Cyprinid trophic guild structure is especially determined by landscape features. Distributions of minnows within the Ozark ecoregion are limited by physiography (Matthews and Robison 1998), variable flow (Matthews and Styron 1981, Spanza and Stanley 2000) and turbidity (Bonner and Wilde 2000, Quist et al. 2004) in the prairie ecoregion. The number of fish has been hypothesized to be important, but the trophic position of the most common small fish is also hypothesized to be important. These factors complicate the understanding the drivers of food chain length.

Trophic positions of crayfish, which are important contributors to smallmouth bass diets, were between 2 and 3. Crayfish consume animal matter as juveniles but consume algae and detritus as they mature (Evans-White et al. 2001, Parkyn et al. 2001). Crayfish within prairie streams consumed a great deal of detritus and algae (Evans-White et al. 2001), but Ozark crayfish were found to consume more animal matter (Whitledge and Rabeni 1997). These differences may explain why they were closer to 2 in prairie streams but closer to 3 in Ozark streams. Crayfish are very

important for fish (Keast 1985), and so their trophic position is important to consider as well. Therefore the trophic position of crayfish is also likely to determine the trophic position of the top piscivore.

Landscapes have led to major differences of the piscivore fish community structure as well between these two ecoregions. Landscape features and watershed land use have impacted water quality conditions and zoobiogeography, which has created modern day fish communities (Cross 1970). Prairie streams had communities composed of *Lepisosteus* spp., Ictaluridae, and *Aplodinotus grunniens*. Ozark streams had greater numbers of centarchid and catostomid fishes. These different communities create different food web structures. *Lepisosteus* spp. are nearly completely piscivorous, consuming *Dorosoma cepedianum*, and cyprinids and usually have trophic positions near 4 (Cross et al. 1982, Williams and Trexler 2006). Centrarchids such as *Micropterus salmoides* are much more likely to be omnivorous, and consume both cyprinids and stream macroinvertebrates (Keast 1985), and have trophic positions between 3 and 4 (Vander Zanden et al. 1997, Franssen and Gido 2006). Smaller centrarchids such as *Ambloplites rupestris*, *Micropterus punctulatus* and *Poxomis* spp. are more likely to consume invertebrates than small fish (Keast 1985, Paterson et al. 2006). Differences in fish community structure then are important to consider as the top piscivore in the Prairie streams is much less likely be omnivorous than the centrarchids in the Ozark streams, and these differences are a result of landscape features.

Prevailing Theories

These direct explanations for food chain length are hypothesized to be important when current theories were not fully supported in this study. According to the prevailing ecological thought, food chain length should be longest where there is a large amount of energy, a large-sized ecosystem, and an intermediate level of disturbances (Power 1996, Townsend et al. 1998, Post 2000). The fact that food chain length, productivity, and nutrients were higher in the Prairie ecoregion than the Ozark ecoregion gives some support to this hypothesis, but no strong relationships were found. Algae in the Ozark ecoregion appeared to be nutrient limited (Lohman et al. 1991), which explains why the amount of chlorophyll, and the amount of energy available in the Ozark streams was less than Prairie streams. However, measuring chlorophyll alone may mislead the investigator because the phytoplankton may be dominated by less palatable and nutritious algal types (Bunn et al. 1999), thereby decreasing the potential amount of energy available to consumers. Prairie streams are also dominated by large omnivores such as carp, which can feed on algae. Because these fish rapidly reach a size allowing their escape from piscivorous fish, they help produce short but productive food webs with lower maximum trophic positions (Layman et al. 2005 a, b). This may be an explanation for the lack of relationship between productivity and food chain length. Higher productivity may be increasing the number of inedible species, and shortens food chain length.

There was also a lack of support for disturbance, measured in this case as the coefficient of variation of discharge. Disturbance is more frequent in rivers of prairie

ecoregions. However, animals from these rivers may be adapted such environments, and so the amount of variability of discharge may have not been sufficient to significantly affect food chain length (Resh 1988, Poff and Allen 1995). This also explains why variability of discharge did not strongly impact stream community structure. Lastly, this study tends to support arguments for the importance of ecosystem size, but again there was no strong correlation evident from my study because all sampled streams appeared to be large enough to support large piscivores. This is relevant because diversity tends to increase with stream order (Schlosser 1982, Fausch et al. 1984, Williams et al. 1996), and larger piscivores are found in larger aquatic habitats. Food chain length has been hypothesized to be impacted by environmental degradation that results in the loss of top predators and subsequent shortening of the food chain length (Odum 1985, Petchy et al. 2004). This loss of natural top predators may have been hidden by the fact that many of these streams in both ecoregions are stocked with game fish, most of which are at least partially piscivorous, and we cannot be certain if these rivers can naturally support longer food chains. Also, reservoirs allow other more lentic piscivores, such as *Micropterus dolomieu*, *Lepomis cyanellus*, and *Ambloplites rupestris*, to disperse as well into rivers (Quist et al. 2004). In both ecoregions, the additions of piscivores have increased the number of trophic levels artificially, and thereby hide our ability to determine if they streams are truly able to support additional trophic levels. This study is unable to find support for any of the prevailing hypotheses but cannot refute any as well. However, the strong differences between these two ecoregions give

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support for the differences in food chain length. Food chain length and trophic position are a result of complex interactions impacted by large-scale factors as hypothesized by Frissell et al. (1986), where landscape features significantly impacts habitat and stream communities.

Explanation for Errors

Some fish had unexpectedly high trophic levels, which also may be explained by landscape features. Sources of nitrogen for the food web are especially impacted by microbial process and landscape patterns. Microbial transformation of N from one form to another increases $\delta^{15}N$ (Caracos et al. 1998). Clearance of land for agriculture (Udy and Bunn 2001, Anderson and Cabana 2005, Vander Zanden et al. 2005) and sewage inputs (Steffy and Kilham 2004) have been found to increase $\delta^{15}N$ up to 10 ‰. Prairie $\delta^{15}N$ is comparable to sources of N in degraded watersheds and supports the effects of landscape effects on $\delta^{15}N$. This may explain the higher $\delta^{15}N$ in these rivers. Detritivorous fish consume decomposing matter, which through the process of microbial degradation increases $\delta^{15}N$ (Caracos et al. 1998). Consequently, *Cyprinella lutrensis, Cyprinus carpio*, and *Carpiodes carpio* in my study had very high $\delta^{15}N$ from this food source. This may be one explanation for their unexpectedly high trophic levels.   Also, fish move within home ranges greater than the area that I sampled baselines and may explain why baselines were not effective. Predators tend to move larger areas of kilometers, and smaller fish within an area of 500 m (Mundahl and Ingersoll 1989, Goforth and Foltz 1998, Smithson and Johnston 1999,

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Snedden et al. 1999, Paller et al. 2005).   Due to the variability of the sources and

movement of fish, it is difficult to know if the top piscivores are represented by the

baselines.   Using baselines that may not reflect the original food source may lead to

a 0.5 difference in trophic position (McKinney et al. 1999). Lengths of one half

trophic levels can be biologically important (Post 2003, Matthews and Muzumder

2005), but I cannot be confident about the differences in trophic position because of

errors associated with baseline and methods used to calculate trophic position. This

means choosing a baseline should encompass as many habitats as possible.   Better

measures of food web should include collecting data from the home range of the top

predator (Cousins 1996). The use of stable isotopes can capture food web processes,

but landscape variation can create results that are difficult to interpret.

Effects of Watershed Area and Landscape Measurements

This study found that percent land use, as related to the amount of forest and

cultivation in the watershed was the landscape metric that related to the most

variables and with the strongest Pearson correlations. NDVI and LPMs were found to

be important as well. However, percent land use was very effective at capturing

effects of landscapes on stream community structure. These results were unlike

Griffiths et al. (2002 a, b) study, which found that NDVI was more effective at

correlating with water quality than LPM or land use and land cover. NDVI variables

in this study that were important were: onset of greenness NDVI, maximum NDVI,

accumulated growing season NDVI, average of growing season NDVI, rate of green

up, and rate of senescence. The week when phenology occurred and end NDVI were less important. Griffith et al (2002 a and b, 2000) found that VPMs of vegetation such as onset of greenness correlated with stream concentrations of nitrogen, phosphorous, and turbidity respectively as a result of the differences in reflectance of the response of corn and winter wheat to fertilization. Forests tend to green up earlier than agricultural crops (Griffith 2002a), and grasslands have low mean NDVI while forested landscapes have higher mean NDVI (Riera et al. 1998, Guerschman and Paruelo 2005).   VPMs measured spatial and temporal differences in regional NDVI values, which reflected phenological changes of vegetation, and differences in the amount of irrigated row crops and native grasslands. Fertilized and irrigated row crops green up later and have higher NDVI than natural grassland vegetation (Paruelo et al. 2001, Griffiths et al. 2002a, Guerschman and Paruelo 2005). Therefore, the level of NDVI was very effective at capturing differences of the two ecoregions. Many of the LPMs and land use correlated with water quality variables and stream organism diversity, which was unlike the Griffiths et al. (2002b) study. The amount of intact patches, as measured by total core area and size of patches was greater in the Ozark ecoregion, which probably was important. However, what was most important was the percent forest or cultivation on the watershed. Therefore, simple whole watershed land use was able to measure effects of land cover on stream organisms.

This study did not measure effects of geology and other aspects of landscapes other than land use. It was able to differentiate these communities because the ecoregions were so very different. Many studies have found that the ecoregion

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approach does not differentiate fish or macroinvertebrate communities (Hawkins and Vinson 2000). The ecoregion approach does work well for watersheds that are large and have very different topography (Feminella 2000, Hawkins et al. 2000), and has worked well for comparing macroinvertebrate communities between the Central Irregular Plains and Ozark Highland ecoregions in the past (Matthews and Robison 1988, Rabeni and Doisey 2000). Therefore, while this study has only considered land use as a measure of landscape, it has worked well because the geology and topography are so different. Comparing food webs in ecoregions that are more similar may require measuring variables that consider landscape properties as well as vegetation.

Conclusions

This study supports the conclusions that food chain length is primarily determined by productivity, disturbance, and ecosystem size, but also by complex patterns of organism community structure that are determined by large-scale factors. Species additions, losses, and increased omnivory have been hypothesized to be important drivers of food chain length (Post and Takimoto 2007). Food chain length increases until omnivory shortens the food chain at high levels of productivity (Diehl and Feissell 2000, Post and Takimoto 2007). Post (2000) found that larger lakes supported longer food chain lengths than smaller lakes. However, the landscape and biogeography can limit the ability of fish to access the lake (Hershey et al. 1999, Post 2002a). This implies that the current theories are only supported if organism can

access the additional primary productivity, the animals themselves are edible, and if landscape features allow piscivores to migrate to the habitat (Persson et al. 1996). Species redundancy has been hypothesized to be important maintainers of food chain length. It has been hypothesized that different genera of macroinvertebrates can maintain the food chain even when species diversity is lost (Quinn et al. 2003). This also may explain why rivers that would appear to be degraded and have evidently lost diversity are still able to maintain the same maximum trophic position. Prairie streams have lost some native cyprinid taxa as a result of reservoirs (Quist et al. 2004) and *Cyprinella lutrensis* has come to dominate the cyprinid community (Marsh-Matthews and Matthews 2000). Homogenization of cyprinids species has been measured in many streams (Scott and Helfman 2001, Scott 2006) and food chain length would not necessarily decrease. Even if other species are lost, *Cyprinella lutrensis* can support the food chain. It appears that food chain length does not decrease until stream is heavily degraded and the overall numbers of any prey decrease. While this study has found interesting patterns, too few rivers were sampled to definitively know the effects of the prevailing theories and the effects of land use, water quality, and community structure. To truly understand this, experiments are needed to understand the role of the number and trophic composition of small fish. Consequently, the use of food chain length as an indicator of landscape change may be appropriate only to indicate that the ecosystem has been heavily degraded or when it is partially recovered from the previous degradation (Kelly and Harwell 1990). Therefore, when studying food web properties, it is important to

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consider not the prevailing theories but also the effects of the landscape on community structure.

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APPENDIX A. TABLES

TABLE 1 Vegetation phenological metrics (VPMs) and landscape pattern metrics (LPMs). Modified from Reed (1994) and Griffith et al. (2002 a & b).

Metric	Description
Temporal metrics	
Date of onset of greenness (start week)	week when photosynthetic activity increases
Date of end of greenness (end week)	week when photosynthetic activity is low
Date of maximum greenness (max week)	week of maximum photosynthesis
NDVI	
NDVI at onset of greenness (start NDVI)	NDVI when photosynthetic activity increases
NDVI at end of greenness (end NDVI)	NDVI at end of photosynthetic activity
NDVI of maximum week (max NDVI)	NDVI at maximum photosynthetic activity
Accumulated growing NDVI season NDVI	net primary production
Rate of green up	acceleration of increasing photosynthesis
Rate of senescence	acceleration of decreasing photosynthesis
Average growing season	mean photosynthetic activity
Landscape Pattern Metrics	
Total Core Area (TCA) (ha)	Total area of a patch class that is a specified distance from the perimeter.
Number of patches (NUMP)	Number of patches of a class on the landscape
Mean Patch Size (MPS)	Area of class patches/ number of patches
Edge Density (ED) (m/ha)	Length of class patch edges/ total patch area
Area-weighted Shape Index (AWMSI)	Mean patch shape complexity measure. Equals 1 when patches are circular
Area-weighted Mean Patch Patch Fractal Dimension (AWMPFD)	Patch shape complexity measure. Equals 1 when patches are circular
Interspersion and Juxtapostion Index (IJI) (%)	Length of adjacent patch types/ total length of of two class types edges.

TABLE 2

Discharge is from the USGS National Water Information System daily average discharge averaged for 25 years. Mean water quality data collected from 2003 for temperature, dissolved oxygen, conductivity, pH, and 2004 for turbidity and chlorophyll-*a*. Median total nitrogen and total phosphorous concentrations were provided by the Central Plains Center for Bioassessment (CPCB) at the University of Kansas. Fish and stable isotope data collected from each river in 2003.

River	Discharge (m^3s^-)	Temp (^oC)	DO (mg/L)	Cond (mS/cm)	pH	TN (mgL^{-1})	TP (mgL^{-1})	Chl-a (μgL^{-1})	Turb (NTU)	N	Fish Richness	Invert EPT	$\delta^{15}N$ fish	$\delta^{15}N$ baseline
Central Plains														
Grand	134.2	28.5	5.88	0.266	7.38	1.46	0.20	13.69	51	198	17	11	14.5	6.3
Kansas	223.0	27.9	7.60	0.711	8.36	2.36	0.33	13.54	71	864	21	10	14.5	9.9
Platte	49.1	27.2	5.85	0.294	7.39	NA	NA	18.19	103	72	10	9	14.0	14.3
Republican	24.0	23.3	7.92	0.685	8.36	1.70	0.28	18.92	23	365	13	10	16.4	11.8
Ozark Highland														
Black	42.1	23.7	7.29	0.242	7.68	NA	0.04	0.83	4	448	41	13	11.0	7.7
Current	85.5	23.4	6.33	0.324	7.72	0.34	0.02	0.89	2	847	43	22	9.8	5.8
Eleven Point	22.9	20.2	8.48	0.376	7.75	0.65	0.02	0.94	9	1041	35	19	10.1	4.6
St. Francis	36.1	25.8	7.04	0.216	7.55	NA	0.05	1.55	2	845	26	15	11.1	7.5

TABLE 3 Principal Component Analysis of fish species diversity measures, trophic guild structure and fish community structure.   Discharge is from the USGS National Water Information System daily mean discharge averaged for 25 years. NDVI variables calculated from 15-year Advanced Very High Resolution Radiometer.

Diversity	PC1	PC2
Eigenvalues	**3.025**	**0.838**
N	0.58	0.811
Richness	0.989	0.066
Alpha	0.954	-0.237
Simpson's		
Reciprocal D	0.895	-0.346

Trophic Guild	PC1	PC2
Eigenvalues	**2.43**	**1.22**
Herbivores	0.522	-0.790
Invertivores	0.942	-0.081
Omnivores	-0.995	0.072
Planktivores	0.530	0.767

Fish Community	PC1	PC2
Eigenvalues	**3.98**	**1.42**
Lepisosteus spp.	0.948	0.218
Herring	-0.302	0.153
Cyprinidae	0.237	-0.950
Catostomidae	-0.773	0.549
Ictaluridae	0.954	0.261
Centarchidae	-0.718	0.027
A. grunniens	0.955	0.271

River size	PC1	PC2
Eigenvalues	**2.75**	**0.91**
Discharge	0.362	0.932
Watershed Area	0.966	-0.132
Dendritic Length	0.966	-0.150
Dendritic Density	0.865	-0.075

Water quality	PC1	PC2
Eigenvalues	**2.75**	**2.54**
Temperature	0.880	0.134
Conductivity	-0.162	0.976
pH	-0.417	0.874
Dissolved Oxygen	-0.829	0.432
Turbidity	0.828	0.400
Chl-a	0.637	0.678

Table 3 (cont'd)
15 Year NDVI

2 HUC	**PC1**	**PC2**
Eigenvalues	**4.536**	**0.437**
Average NDVI	0.961	-0.272
Rate of Greenup	0.991	0.086
Rate Senescence	0.834	0.550
Accumulated Growing Season	0.973	-0.220
Average Growing Season	0.995	-0.068

4 HUC	**PC1**	**PC2**
Eigenvalues	**4.47**	**0.43**
Average NDVI	0.968	-0.245
Rate of Greenup	0.976	0.009
Rate Senescence	0.808	0.587
Accumulated Growing Season	0.973	-0.137
Average Growing Season	0.992	-0.095

11 HUC	**PC1**	**PC2**
Eigenvalues	**4.34**	**0.57**
Average NDVI	0.959	-0.281
Rate of Greenup	0.976	0.051
Rate Senescence	0.742	0.666
Accumulated Growing Season	0.963	-0.198
Average Growing Season	0.995	-0.084

21HUC	**PC1**	**PC2**
Eigenvalues	**4.20**	**0.70**
Average NDVI	0.957	-0.287
Rate of Greenup	0.970	0.085
Rate Senescence	0.661	0.745
Accumulated Growing Season	0.956	-0.22
Average Growing Season	0.994	-0.09

Table 3 (cont'd)
Percent Land use/ Land Cover

2 HUC	**PC1**	**PC2**
Eigenvalues	**2.88**	**0.177**
Forest	0.966	0.260
Grassland	-0.998	0.037
Cultivation	0.975	-0.220

4 HUC	**PC1**	**PC2**
Eigenvalues	**2.78**	**0.21**
Forest	0.933	0.359
Grassland	-0.994	0.070
Cultivation	0.959	-0.277

11 HUC PC1	**PC1**	**PC2**
Eigenvalues	**2.89**	**0.096**
Forest	0.972	0.230
Grassland	-0.994	0.021
Cultivation	0.976	-0.207

21HUC PC1	**PC1**	**PC2**
Eigenvalues	**2.86**	**.13**
Forest	0.965	0.261
Grassland	-0.996	0.019
Cultivation	0.969	-0.241

Whole	**PC1**	**PC2**
Eigenvalues	**3.23**	**1.43**
Forest	-0.918	0.392
Grassland	0.367	-0.920
Cultivation	0.986	-0.035
Wetland	0.767	0.522
Urban	0.837	0.396

TABLE 4

Results of Pearson Product Moment Correlation. Mean fifteen-year NDVI, water quality, and fish community structure variables (diversity, fish community structure, trophic guild structure, invertebrate family richness, and EPT number (number of aquatic insects in the families within the orders of Ephemeroptera, Plecoptera and Trichoptera)) were reduced using principal component analysis (see Table 3; see Table 1 for descriptions of NDVI and LPMs). Results include Pearson Product Moment correlation r, p-value (*p<0.05, **p<0.01, ***p<0.001), and PC1(1) or PC2 (2) of dependent variables.

	TP	Diversity	Fish Community Structure	Trophic Guild	Water Quality	$\delta^{15}N$	Nutrients	Invertebrate Family Richness	EPT
2-HUC									
Start week	NS	NS	NS	NS	-0.732* (2)	NS	NS	NS	NS
Start NDVI	NS	0.860**(1)	NS	0.977***(1)	NS	-0.920*** (fish)	-0.861* (TP)	NS	
Max week	NS	NS	NS	NS	0.812* (1)	NS			
Max NDVI	NS	0.717*(1)	NS	0.834**(1)	-0.720* (2)	-0.922*** (fish)	-0.811* (TP)	NS	NS
End Week	NS	NS	NS	NS	NS	NS	NS	NS	NS
End NDVI	NS	NS	NS	NS	NS	NS	NS	NS	NS
Rate of Green Up	NS	0.723*(1)	NS	NS	NS	-0.737* (fish)	NS	0.849**	0.723*
Rate of Senescence	NS	-0.835**(1)	NS	NS	NS	0.731* (fish)	NS	NS	NS
Accumulated growing season	NS	0.844**(1)	NS	0.962***(1)	NS	-0.987*** (fish) -0.715* (base)	-0.918** (TP)	0.778*	0.826*
Average growing s season	NS	0.833**(1)	NS	0.960***(1)	-0.756* (2)	-0.971*** (fish)	-0.896** (TP)	0.717*	0.768*
15 year PCA	NS	0.849**(1)	NS	0.979***(1)	NS	-0.974*** (Fish)	-0.888** (TP)	0.775*	0.804*

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Table 4 (cont'd)

	TP	Diversity	Fish Community Structure	Trophic Guild	Water Quality	$\delta^{15}N$	Nutrients	Invertebrate Family Richness	EPT
4-HUC									
Start week	NS	NS	NS	NS	NS	NS	NS	NS	NS
Start NDVI	NS	-0.869*(1)	-0.711*(1)	0.970***(1)	NS	-0.903** (fish)	-0.844* (TP)	NS	NS
Max week	NS	NS	NS	NS	0.843** (1)	NS	NS	NS	NS
Max NDVI	NS	0.816*(1)	NS	0.911***(1)	-0.744* (2)	-0.949*** (fish)	-0.831* (TP)	NS	0.716*
End Week	NS	NS	NS	NS	NS	NS	NS	NS	NS
End NDVI	NS	NS	NS	NS	NS	NS	NS	NS	NS
Rate of Green Up	NS	0.757*(1)	NS	NS	NS	-0.786* (fish)	NS	0.943***	0.863**
Rate of Senescence	NS	-0.803*(1)	NS	NS	NS	NS	NS	NS	NS
Accumulated Growing Season	NS	0.844**(1)	NS	0.962***(1)	-0.732* (2)	-0.977** (fish) -0.760* (base)	-0.947*** (TP)	0.742*	0.826*
Average growing season	NS	0.844**(1)	NS	0.968***(1)	-0.759* (2)	-0.955*** (fish)	-0.900** (TP)	NS	0.770*
15 year PCA	NS	0.858* (1)	NS	0.985***(1)		-0.971*** (fish)	-0.908** (TP)	0.759*	0.809*

Table 4 (cont'd)

	TP	Diversity	Fish Community Structure	Trophic Guild	Water Quality	$\delta^{15}N$	Nutrients	Invertebrate Family Richness	EPT
11-HUC									
Start week	NS	NS	NS	NS	- 0.754* (2)	NS	NS	NS	NS
Start NDVI	NS	0.876**(1)	-0.731*(1)	0.981***(1)	NS	-0.922*** (fish)	-0.890** (TP)	0.746*	0.789*
Max week	NS	NS	NS	NS	0.863** (1)	NS	NS	NS	NS
Max NDVI	NS	0.749*(1)	NS	0.870**(1)	-0.852** (2)	-0.955*** (fish)	-0.911** (TP)	NS	0.713*
End Week	NS	NS	NS	NS	NS	NS	NS	NS	NS
End NDVI	NS	NS	NS	NS	-0.711*	-0.733** (fish)	-0.760* (TP)	NS	NS
Rate of Green Up	NS	0.773*(1)	NS	0.838*(1)	NS	-0.969*** (fish)	-0.931** (TP) -0.893* (TN)	0.860**	0.882**
Rate of Senescence	NS	-0.783(1)	NS	NS	0.745* (2)	0.722* (fish)	NS	NS	NS
Accumulated growing season	NS	0.824(1)*	NS	0.952***(1)	-0.737* (2)	-0.990*** (fish)	-0.934** (TP)	0.771*	0.817*
Average Growing Season	NS	0.834(1)**	NS	0.964***(1)	-0.780* (2)	-0.977*** (fish)	-0.933** (TP)	0.728* `	0.797*
15 year PCA	NS	0.853** (1)	NS	0.972***(1)	-0.717* (2)	-0.981*** (fish)	-0.919** (TP)	0.772*	0.811*

Table 4 (cont'd)

	TP	Diversity	Fish Community Structure	Trophic Guild	Water Quality	$\delta^{15}N$	Nutrients	Invertebrate Family Richness	EPT
21-HUC									
Start week	NS	0.745*(1)	NS	NS	NS	NS	NS	NS	NS
Start NDVI	NS	0.883**(1)	-0.734*(1)	0.979***(1)	NS	-0.918*** (fish)	-0.911** (TP)	0.731*	0.796*
Max week	NS	NS	NS	NS	0.776*(1)	NS	NS	NS	NS
Max NDVI	NS	0.769*(1)	NS	0.863**(1)	-0.850** (2)	-0.954*** (fish)	-0.911** (TP)	NS	NS
End Week	NS	NS	0.784*(1)	NS	0.741* (1)	NS	NS	NS	NS
End NDVI	NS	NS	NS	NS	-0.849**(2)	-0.829* (fish)	-0.883** (TP)	NS	NS
Rate of Green Up	NS	NS	NS	0.772*(1)	NS	-0.878** (fish)	-0.847**	0.804*	0.760*
Rate of Senescence	NS	-0.855**(1)	NS	NS	NS	0.745* (fish)	NS	NS	NS
Accumulated Growing Season	NS	0.796*(1)	NS	0.934***(1)	-0.783*(2)	-0.989***(fish)	-0.952***(TP)	0.757*	0.815*
Average Growing Season	NS	0.852**(1)	NS	0.955***(1)	-0.793*(2)	-0.978*** (fish)	-0.940** (TP)	0.722*	0.795*
15 year PCA	NS	0.861** (1)	NS	`0.962***(1)	-0.745* (2)	-0.983*** (fish)	-0.931** (TP)	0.756*	0.804*

Table 4 (cont'd)

LPM

	TP	Diversity	Fish Community Structure	Trophic Guild	Water Quality	δ¹⁵N	Nutrients	Invertebrate Family Richness	EPT
2 HUCs									
TCA	NS	0.760*41(1)	-0.707*41(1)	0.975***41(1) -0.822*82(1)	0.710*71(2)	-0.875**41(f) 0.908**(82)(f)	-0.858*41(P) 0.838*82(P)	-0.732*82	NS
NUMP	NS	0.735*A(1)	-0.721A(1)	-0.828**82(1)	NS	0.790*41(f) 0.807*82(f)	0.842*41(P) 0.882**22(P) 0.933*22(N)	-0.716*41 -0.807*82	-0.727*82 -0.709*22
MPS	NS	NS	NS	0.921 ***41(1) -0.745* 82(1) 0.729*A(1)	0.863*71(2) -0.801* A(1)	-0.854**41(f) 0.743*71(f) 0.887**82(f)	-0.829*41(P) 0.754*71(P) 0.804*82(P)	0.708*41 -0.711*82	0.770*41 -0.712*82
ED	NS	0.927***41(1) -0.833**82(1) -0.707*A(2)	NS	0.783*41(1) -0.957***82(1)	0.771*71(2) 0.769**81(1)	-0.828*41(f) 0.935***82(f) 0.737*82(b)	0.798*82(P)	0.765*41 -0.791*82	0.734*41 -0.805*82
AWMSI	NS	0.827* (1)	-0.727*(1)	NS	NS	NS	NS	NS	NS
AWMPFD		NS	-0.795*(1)	NS	NS	NS	NS	NS	NS
IJI		-0.806*(2)	0.720*(2)	NS	NS	NS	NS	-0.773*	-0.728*

	TP	Diversity	Fish Community Structure	Trophic Guild	Water Quality	$\delta^{15}N$	Nutrients	Invertebrate Family Richness	EPT
21 HUCs									
TCA	NS	0.870**41(1) -0.730*82(1)	NS	0.983***41(1) -0.815*82(1)	0.769*71(2)	-0.942***41(f) 0.891**82(f)	-0.932**41(P) -0.878*41(N)	0.731 *41	0.802*41
NUMP	NS	-0.813*82(1) 0.727*AVE*(1)	-0.709*AVE(1)	-0.908**82(1)	0.754*41(2) 0.745*71(2) 0.764*82(1)	0.868**82(f) 0.724*22(f)	0.947***82(P) 0.939*82(N) 0.782*22(P)	-0.714*82 -0.784 *A	-0.830*82
MPS	NS	0.893**41(1)	-0.720*AVE(1)	0.845**41(1)	0.921***71(1)	-0.795*41(f) 0.797*71(f) 0.813*82(f)	-0.786*41(P) 0.831*71(P)	NS	NS
ED	NS	0.880**41(1) -0.869**82(1)	NS NS	0.931***41(1) -0.966***82(1)	0.840*71(1)	-0.983***41(f) 0.946***82(f)	-0.877*41(P) 0.814*82(P)	0.830*41 -0.763*82	0.822`*41 -0.764*82
AWMSI	NS	0.816*(1)	NS	0.737*(1)	-0.812*(1)	-0.752*(f)	-0.770*(P)	0.808*`	0.866**
AWMPFD		0.829*(1)	-0.724*(1)	0.771*(1)	NS	NS	NS	NS	NS
IJI	NS	NS	0.731*(2)		NS	0.866**(1)	NS	NS	-0.764*

Table 4 (cont'd)

	TP	Diversity	Fish Community Structure	Trophic Guild	Water Quality	$\delta^{15}N$	Nutrients	Invertebrate Family Richness	EPT
Percent Watershed									
2-HUC	NS	-0.864** (1)	NS	-0.975***(1)	NS	0.978*** (fish)	0.918** (TP)	-0.775*	-0.813*
4-HUC (PC2)	NS	-0.881** (1)	NS	-0.959***(1)	0.737* (2)	0.949*** (fish)	0.897** (TP)	-0.719*	-0.771*
11-HUC (PC2)	NS	-0.894** (1) 0.754*(2)	NS	-0.977***(1)	0.725* (2)	0.960*** (fish)	0.916** (TP)	-0.734*	-0.804*
21-HUC (PC2)	NS	-0.917***(1) 0.784*(2)	NS	-0.963***(1)	0.723* (2)	0.952*** (fish)	0.918** (TP)	-0.728*	-0.800*
Whole Watershed (PC2)	NS	-0.875**(1)	0.824*(1)	-0.899**(1)	0.842**(1) -0.842**(2)	0.812*(fish)	0.843* (TP)	-0.717*	-0.782*

TABLE 5 Mann-Whitney U comparisons of fish species' trophic position from eight rivers. Comparisons are made between ecoregions (Ozark and Prairie), rivers, or years (2003 and 2005).

Fish Common Name	Scientific Name	Comparison	Mean TP (N)	Statistic
Piscivores				
Gar	*Lepisosteus spp.*	Year		
		Kansas 2003 and 2005	4.09 (4), 3.81(5)	Z=-2.2 p<0.05
Largemouth Bass	*Micropterus salmoides*	River		
		Current, St Francis	3.68 (4), 3.38 (5)	Z=-1.2,-.98 p>0.05
		Eleven Point and other rivers	4.20 (4)	Z=-2.3, -2.4 p<0.05
		Year		
		Current 2003 and 2005	3.68 (4), 3.51 (4)	Z=-.87, p>0.05
		St. Francis 2003 and 2005	3.38 (5), 3.37 (5)	Z=-1.1, p>0.05
Flathead Catfish	*Pylodictis olivaris*	Ecoregion		
		Ozark 2005 and Prairie 2005	3.28 (3), 3.39 (5)	Z=-.45, p>0.05
		Year		
		Kansas 2003 and Kansas 2005	3.49 (4), 2.95 (3)	Z=-1.8, p>0.05
Omnnivores				
Red Shiner	*Cyprinella lutrensis*	River		
		Kansas 2003 and Grand 2003	3.26 (4), 4.47(5)	Z=-2.4, p<0.05
		Year		
		Kansas 2003 and Kansas 2005	3.26 (4), 2.39 (3)	Z=-2.1, p=0.05
Channel Catfish	*Ictalurus punctatus*	Ecoregion		
		Ozark 2003 and Prairie 2003	2.60 (10), 3.18 (12)	Z=-2.6, p<0.01
		Ozark 2005 and Prairie 2005	3.31 (4), 3.85(8)	Z=0.0, p>0.05
		River		
		Black and Saint Francis 2003	2.41 (5), 2.79 (5)	Z=-2.6, p<0.01
		Kansas and Republican 2003	3.30 (4), 3.64(5)	Z=-2.0, p>0.05
		Kansas and Republican 2005	3.02 (4), 4.68 (4)	Z=-2.4, p<0.05
		Year		
		Saint Francis 2003 and 2005	2.79 (5), 3.31 (4)	Z=-2.4, p<0.01
		Kansas 2003 and 2005	3.30 (4), 3.02 (4)	Z=-1.7, p>0.05
		Republican 2003 and 2005	3.64 (5), 4.68 (4)	Z=-2.4, p<0.05
River Carpsucker	*Carpiodes carpio*	River		
		Grand, Kansas and Platte	4.37 (5), 3.12 (5), 1.87 (5)	Z=-2.6, p<0.01(all)
		Kansas and Republican 2005	3.01 (4), 4.72 (5)	Z=-2.3 , p<0.05
		Year		
		Kansas 2003 and 2005	3.12 (5), 3.01 (4)	Z=-0.98 , p>0.05
Common Carp	*Cyprinus carpio*	Year		
		Kansas 2003 and 2005	2.63 (4), 2.44 (5)	Z=-0.49, p>0.05
Insectivores				
Bluegill Sunfish	*Lepomis macrochirus*	Ecoregion		
		Ozark 2005 and Prairie 2005	3.13 (9), 3.81 (6)	Z=-2.1, p>0.05
		River		
		Kansas and Republican 2003	3.07 (5), 3.15 (5)	Z=-.52, p>0.05
		Current and Saint Francis 2005	3.15 (5), 3.11 (5)	Z=-.74, p=0.05
		Year		
		Republican 2003 and 2005	3.15(5), 4.03 (5)	Z=-2.0, p<0.05
Freshwater Drum	*Aplodinotus grunniens*,	River		
		Grand and Kansas 2003	4.66 (5), 3.44 (5)	Z=-2.61, p<0.01

Table 5 (cont'd)

Time				
		Kansas 2003 and 2005	3.44 (5), 3.27 (3)	Z=-0.745, p>0.05
Golden Redhorse	*Moxostoma erythrurum*	River		
		Black and Current	2.51(5), 2.98(5)	Z=2.6, p<0.01
		Black and Eleven Point	2.51(5), 3.23(5)	Z=2.6, p<0.01
		Black and Saint Francis	2.51(5), 3.00(5)	Z=2.6, p<0.01
		Current and Eleven Point	2.98(5), 3.23 (5)	Z=-2.2, p<0.05
		Current and Saint Francis	2.98(5), 3.00 (5)	Z=-.104, p>0.05
		Year		
		Current 2003 and 2005	2.98 (5), 2.88 (5)	Z=-.731, p>0.05
		Saint Francis 2003 and 2005	3.00 (5), 2.99 (5)	Z=-.104, p>0.05

Planktivore

Gizzard Shad	*Dorosoma cepedianum*	Ecoregion		
		Ozark 2005 and Prairie 2005	2.85 (10), 3.58 (8)	Z=-2.5, p<0.05
		River		
		Current and Saint Francis 2005	2.90 (5), 2.80 (5)	Z=-0.94, p>0.05
		Kansas and Republican 2005	3.13 (5), 4.34 (3)	Z=-2.2. p<0.05
Emerald Shiner	*Notropis atherinoides*	Ecoregion		
		Ozark and Prairie 2003	2.76 (11), 2.57 (10)	Z=-0.423, p>0.05
		River		
		Black and Saint Francis 2003	2.49(5), 3.00 (4)	Z=-2.4, p<0.05
		Platte and Republican 2003	1.96 (5), 3.18 (5)	Z=-2.6, p<0.01
		Year (Ozarks)		
		Saint Francis 2003 and 2005	3.00 (4), 2.98 (5)	Z=0.0, p>0.05

Herbivore

Stoneroller	*Campostoma spp.*	River		
		Black and Current	2.45 (5), 2.39 (5)	Z=-1.4, p>0.05
		Black and Saint Francis	2.45 (5), 2.77 (5)	Z=-1.4, p>0.05
		Black and Eleven Point	2.45 (5),2.85 (5)	Z=-2.0, p=0.05
		Current and Eleven Point	2.39 (5), 2.85 (5)	Z=-2.6, p<0.01
		Current and Saint Francis	2.39 (5), 2.77 (5)	Z=-2.6, p<0.01
		Eleven Point and Saint Francis	2.85 (5), 2.77 (5)	Z=-1.1 p>0.05

Invertebrate Predator

Gomphidae		River		
		Current and Eleven Point	2.09 (5), 2.13 (5)	Z=-0.73, p>0.05
		Saint Francis (from other Ozark rivers)	1.32 (5)	Z=-2.6, p<0.01
		Kansas and Republican 2005	1.66 (4), 3.03 (4)	Z=-2.3, p<0.05
		Time		
		Republican 2003 and 2005	2.23 (3), 3.03 (4)	Z=-2.1, p>0.05
Ceoargionidae		River		
		Platte and Republican 2003	2.37 (5), 2.75 (5)	Z=-1.4, p>0.05
		Kansas and Republican 2005	1.89 (3), 3.05 (5)	Z=-2.2, p<0.05
		Time		
		Current 2003 and 2005	2.37 (5), 2.03 (4)	Z=-2.4, p<0.05
		Republican 2003 and 2005	2.75 (5), 3.05 (5)	Z=-2.2, p<0.05

Invertebrate Omnivore

Orconectes spp.		Ecoregion		
		Ozark 2005 and Prairie 2005	2.11 (6), 2.47 (7)	Z=0.57, p>0.05
		River		
		Current and Saint Francis 2003	2.10 (4), 1.65 (5)	Z=-1.5, p>0.05
		Kansas and Republican 2005	1.72 (3), 3.04 (4)	Z=-2.1, p>0.05
Year				
		Saint Francis 2003 and 2005	1.65 (5), 2.08 (4)	Z=-1.7, p>0.05
		Kansas 2003 and 2005	2.80 (5), 1.72 (3)	Z=-2.2, p<0.05

TABLE 6 Maximum (from piscivorous fish) and mean trophic position of fish as calculated by Post 2002b. VZ TP calculated by Vander Zanden's (1996) model. Mean TP without Post's 2007 Lipid correction. Alpha from Post's 2002 model for calculating trophic position.

River	max TP	mean TP	VZ TP	TP not lipid corrected	Alpha
2003					
Central Plains					
Grand (a)	4.52(0.047)		4.42(0.071)		
Kansas (b)	3.79(0.097)	3.27(0.060)	3.75(0.059)	3.52(0.060)	0.29(0.028)
Platte (c)	2.64(0)	2.18 (0.14)	2.06(0.14)	1.17(0.23)	1.33(0.23)
Republican (a)	3.36(0.071)		3.36(0.069)		
Ozark Highland					
Black (d)	2.78(0.051)	2.61(0.038)	2.47(0.038)	2.19(0.060)	1.04(0.070)
Current (e)	3.55(0.033)	3.15(0.061)	3.11(0.065)	3.09(0.058)	0.63(0.18)
Eleven Point (f)	3.93(0.066)	3.44(0.090)	3.85(0.086)	3.53(0.094)	0.096(0.019)
St. Francis (e)	3.43(0.084)	3.08(0.052)	3.08(0.052)	3.08(0.052)	0.55(0.024)
2005					
Central Plains					
Kansas (b)	3.49(0.18)	3.02(0.094)	3.24(0.093)	3.09(0.091)	0.30(0.10)
Republican(g)	4.43(0.14)		4.30(0.092)		
Ozark Highland					
Current (e)	3.51(0.13)	3.07(0.058)	3.01(0.057)	3.00(0.059)	0.84(0.14)
St. Francis (e)	3.34(0.058)	3.10(0.036)	3.08(0.038)	3.07(0.036)	0.77(0.054)

TABLE 7 Landscape pattern metrics calculated using ArcGIS FRAGSTAT. Description of metrics in Table 1.

2-HUC	Total Core Area (Forest)	Ave Number of Patches	Ave Mean Patch Size	Edge Density	AWMSI	AWMPFD	IJI
Black	8547	160	9.76	18.3	5.33	1.196	54.3
Current	6088	114	8.24	17.9	4.53	1.174	44.5
Eleven Point	8306	75	16.53	12.8	3.37	1.158	44.4
Grand	502	87	6.11	12.8	3.33	1.159	53.2
Kansas	2188	87	7.04	14.1	3.55	1.163	48.6
Platte	284	44	4.49	18.7	2.64	1.134	53.3
Republican	177	91	8.75	15.0	3.54	1.176	48.7
Saint Francis	9916	79	14.29	11.2	3.28	1.160	45.5

21-HUC	Total Core Area (Forest)	Ave Number of Patches	Ave Mean Patch Size	Edge Density	AWMSI	AWMPFD	IJI
Black	128,736	1150	15.10	11.5	8.04	1.191	49.6
Current	126,523	1384	12.92	13.1	10.13	1.187	45.4
Eleven Point	114,972	1342	7.97	12.3	8.90	1.173	41.5
Grand	3158	767	6.93	13.0	3.62	1.170	55.7
Kansas	15,183	1396	6.35	15.6	3.94	1.169	56.8
Platte	6521	817	6.04	14.7	3.72	1.169	55.8
Republican	2403	893	11.24	11.7	6.45	1.170	47.1
Saint Francis	126882	1090	11.21	11.2	6.09	1.183	50.6

FIGURE 1. Watersheds sampled within the terrestrial ecoregion. Rivers are the Saint Francis (A), Black (B) Current (C), Eleven Point (D), Lower Grand (E); Platte (F), Kansas (G), and Republican (H) Rivers.
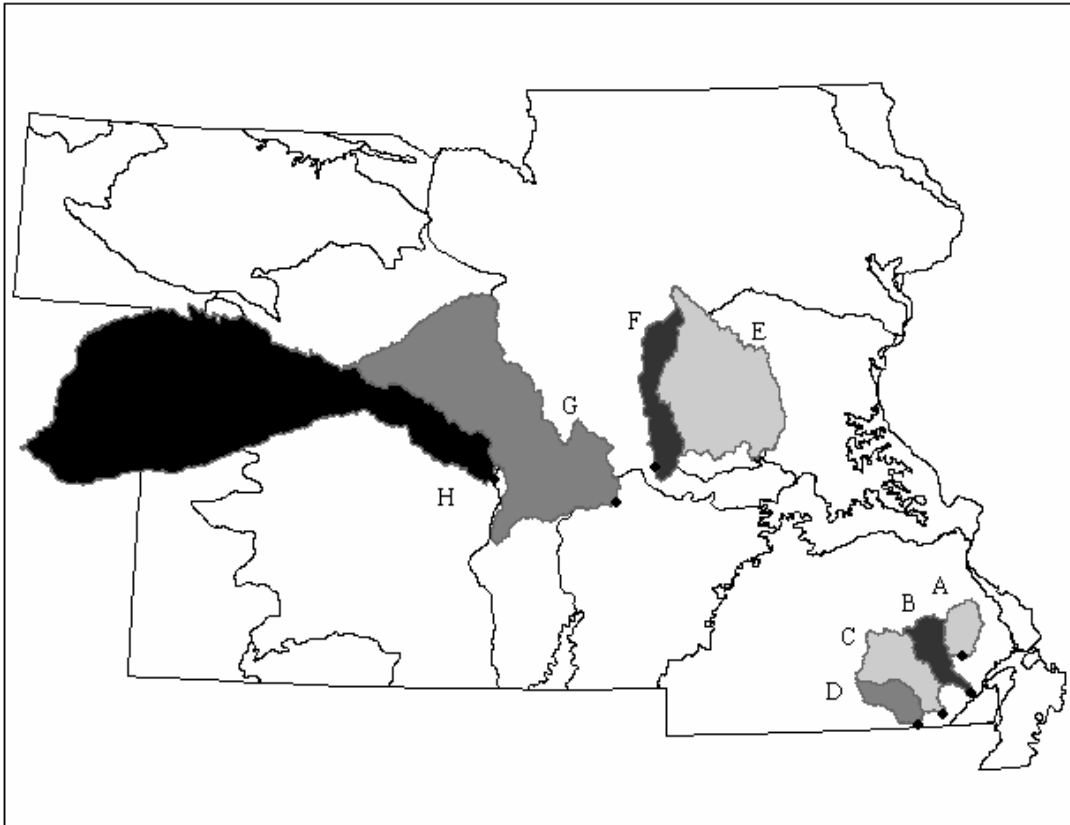
FIGURE 2.

Descriptions of NDVI vegetation phenology curve and metrics (based on graphs from Griffith et al. 2002*a* and Reed et al. 1994)
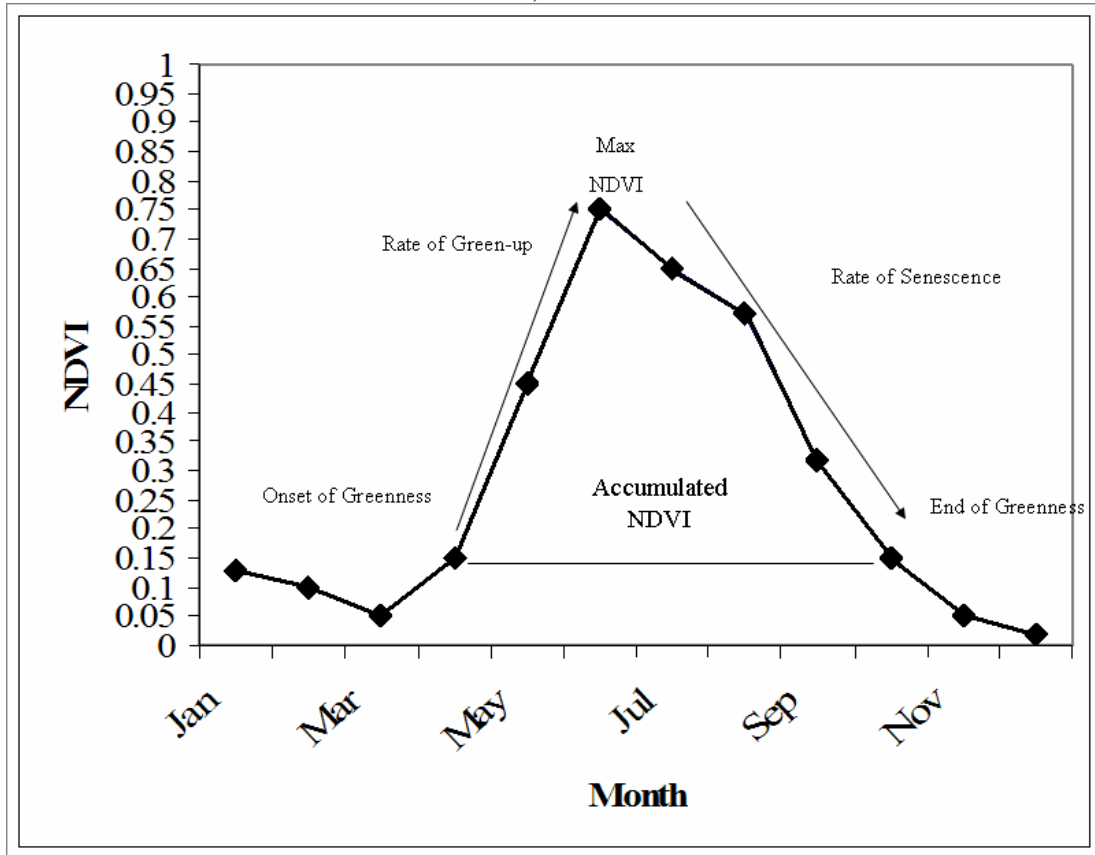
FIGURE 3. Percent land use of rivers from the Ozark and Prairie ecoregion. The lowest size subwatershed is 2 USGS Hydrologic Unit Code sub-watersheds above the sampling location are aggregated together. An aggregation of 2-21 HUCs are shown: (a) percent cultivation; (b) percent forest; (c) percent grassland. Ozark ecoregion rivers include the Black, Current, Eleven Point, and Saint Francis Rivers. Prairie ecoregion rivers include the Grand, Kansas, Platte, and Republican.

a).

b).

c).

——●——	Black River
·····○·····	Current River
--▼--	Eleven Point River
--△---	Grand River
——■——	Kansas River
--□--	Platte River
--◆--	Republican River
——◇——	Saint Francis River
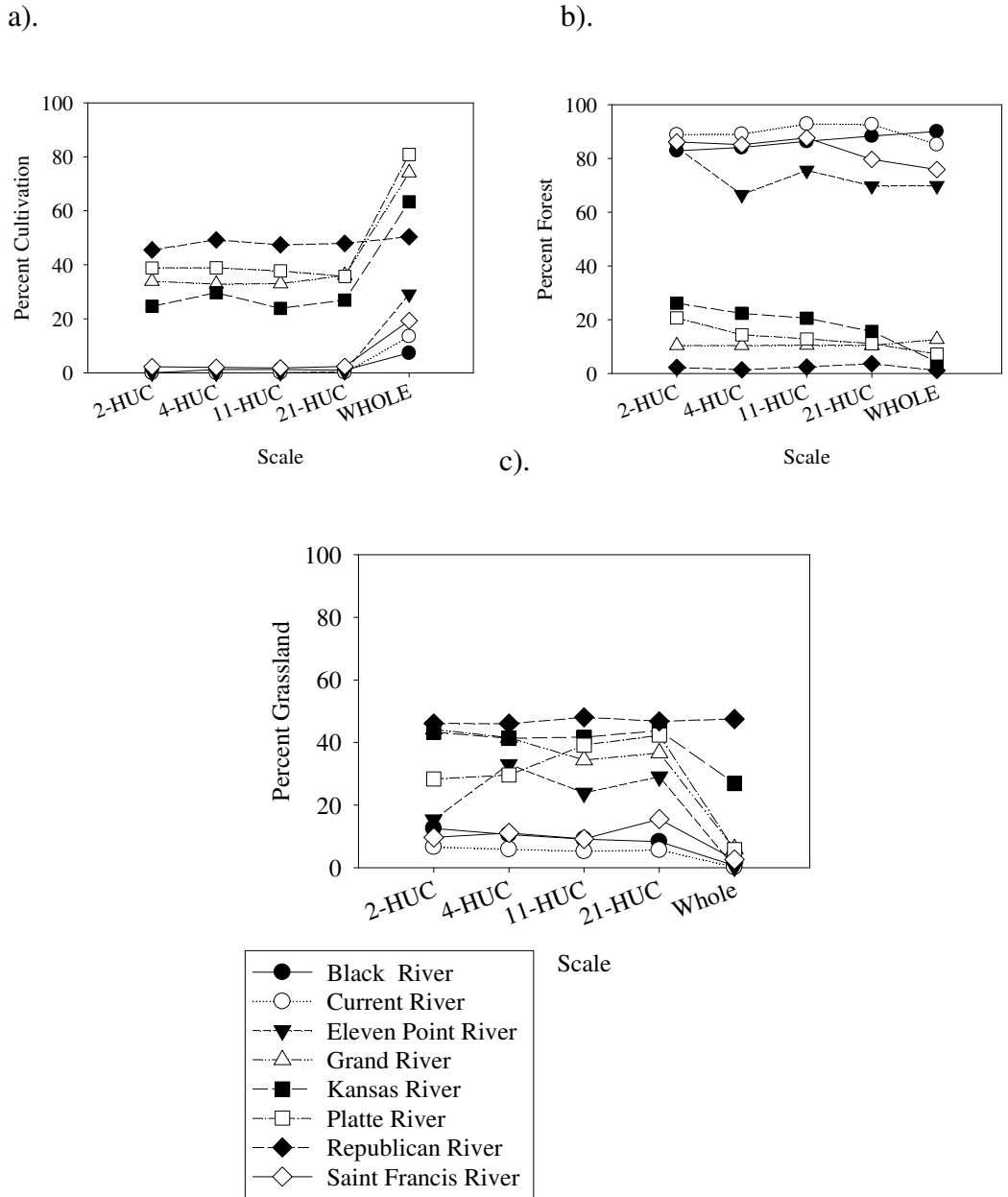
FIGURE 4. Normalized difference vegetation index (NDVI) phenology curves for eight rivers. Each data set is calculated from biweekly composites. Shown are subwatersheds consisting of : (a) 2 USGS HUCs and (b) 21 USGS HUCs.

a).

b).
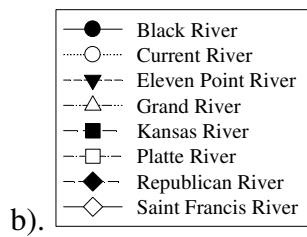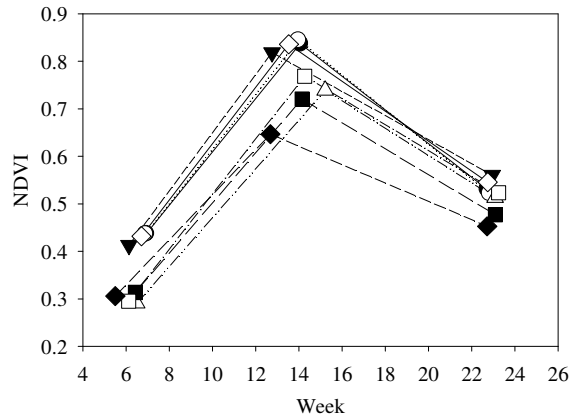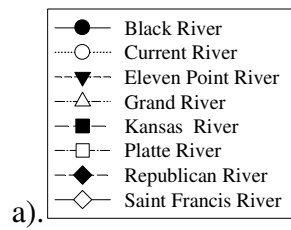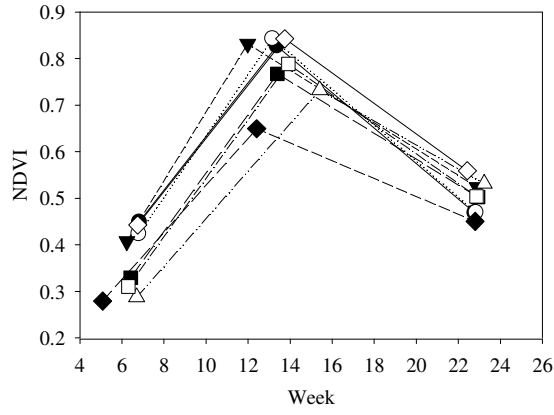
FIGURE 5. Correlations between TP, diversity, and principal component factors.

PC1
High percent of cultivation,
urban areas and wetland.
Low percent of forest.

a.

PC1
High temperature,
low dissolved oxygen
and high turbidity

PC2
High conductivity
High pH

b.

PC1
High fish species richness,
alpha diversity
and Simpson's Reciprocal D

PC2
High fish abundance (N)

c.

79

d.

PC1
High percent of *Lepisosteus* spp.,
Ictaluridae and *Aplodinotus grunniens*
Low percent Catostomidae and
Centarchidae.

PC2
Low percent Cyprinidae

e.

Trophic Position vs Trophic Guild Proportions Principal Component Factor Scores

● PC1	
○ PC2	

PC1
High percent of Invertivores,
Low percent of Omnivores.

PC2
Low percent of Herbivores.
High percent of Planktivores

f.

Water Quality Principal Component Factor Scores

● PC1	
○ PC2	

PC1
High temperature,
low dissolved oxygen
and high turbidity

PC2
High conductivity
High pH

g.

PC1
High watershed area,
dendritic length
and dendritic density

PC2
High discharge

h.

i.

FIGURE 6. Fish trophic structure calculated as proportion of each trophic guild to the number of fish caught. Samples collected in 2003.
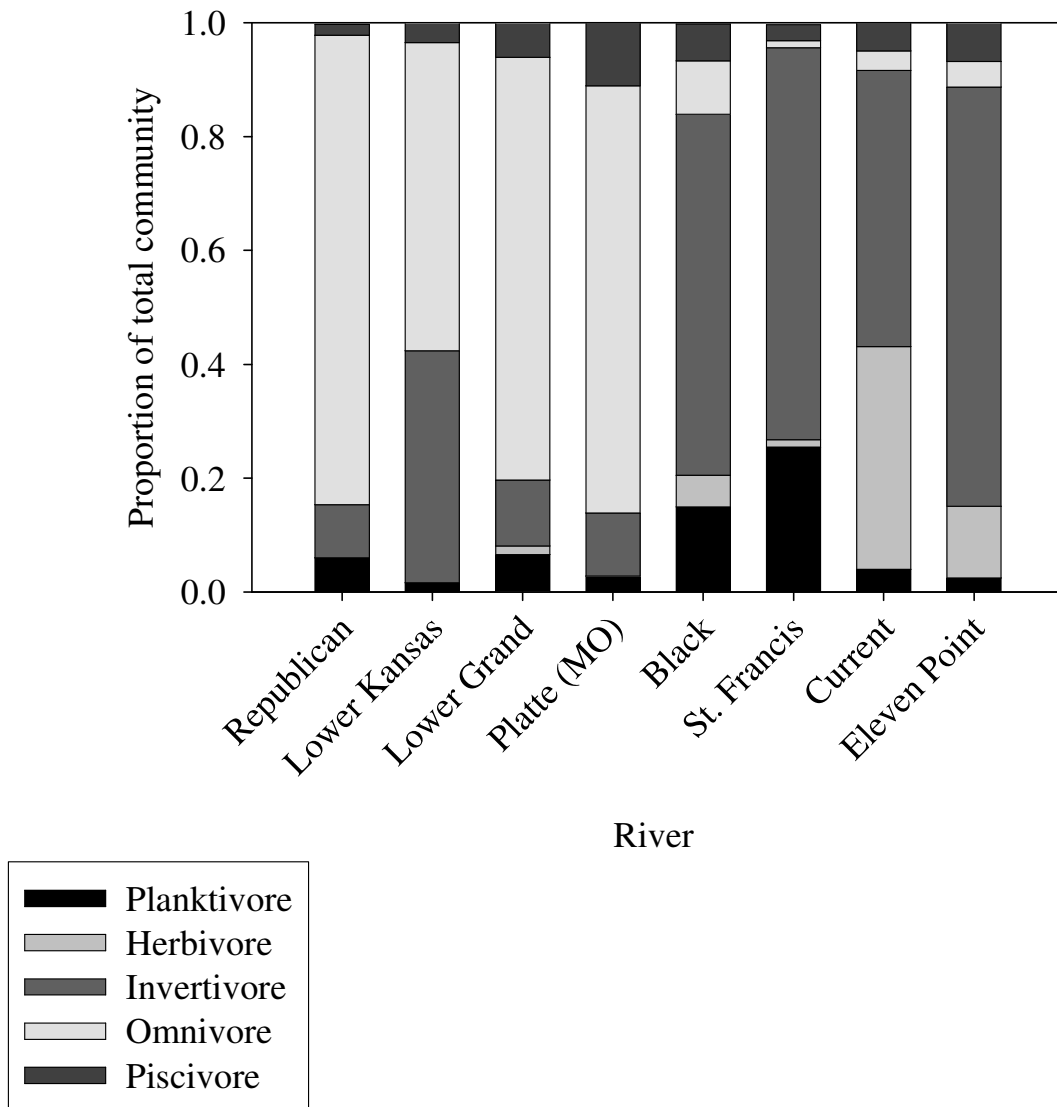
FIGURE 7. The $\delta^{15}N$ and $\delta^{13}C$ of fish and invertebrates caught within rivers. Bars represent one standard deviation. Rivers are the Black River (a), Current (b), Eleven Point River (c), Grand River (Missouri) (d), Kansas River (e), Platte River (f), Republican River (g), Saint Francis River (h), Kansas River 2005 (i),Republican River 2005(j), Saint Francis River 2005 (k), Current River 2005 (l).

a.

b:

c:

d.

e.

f.

g.

87

h.

i.

j.

k.

1.
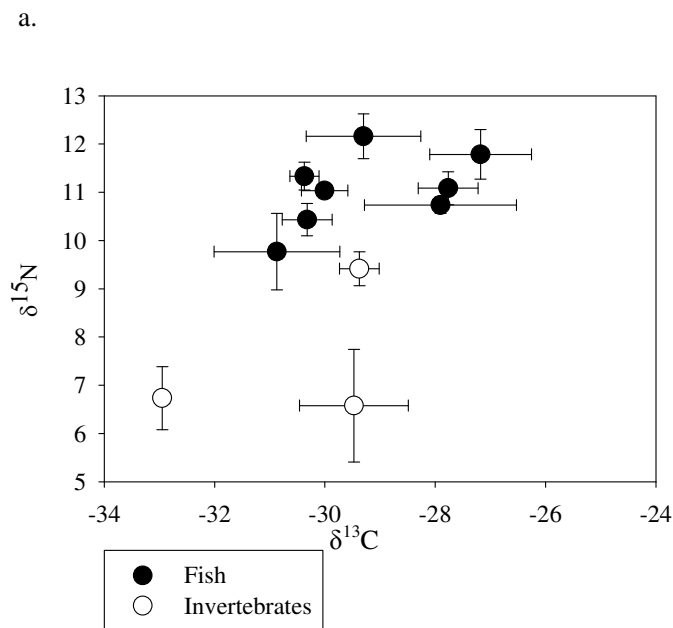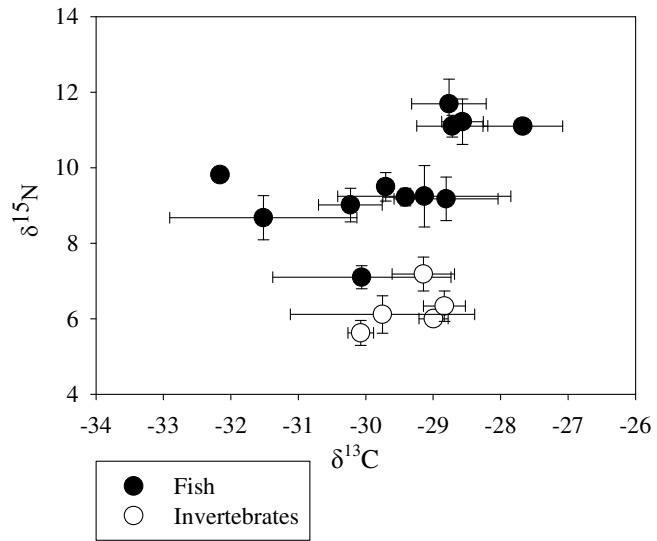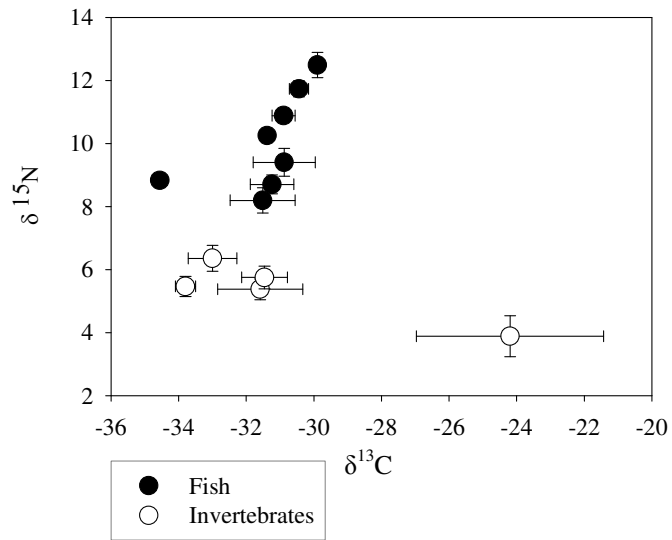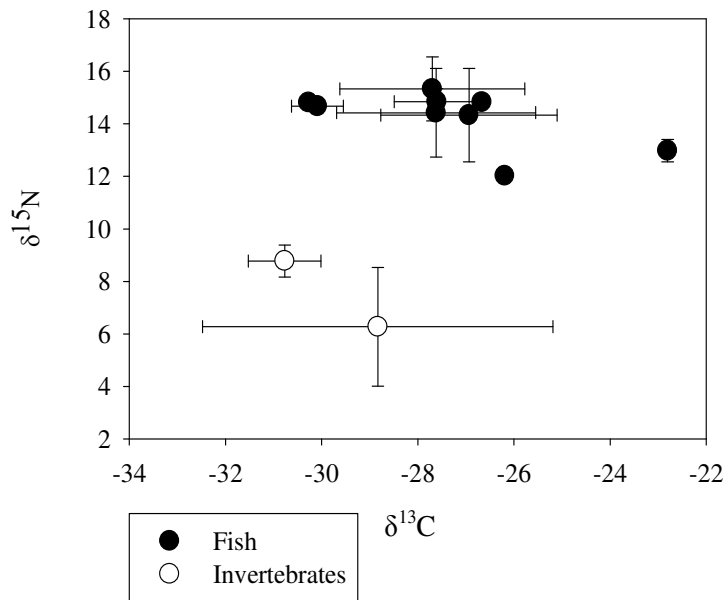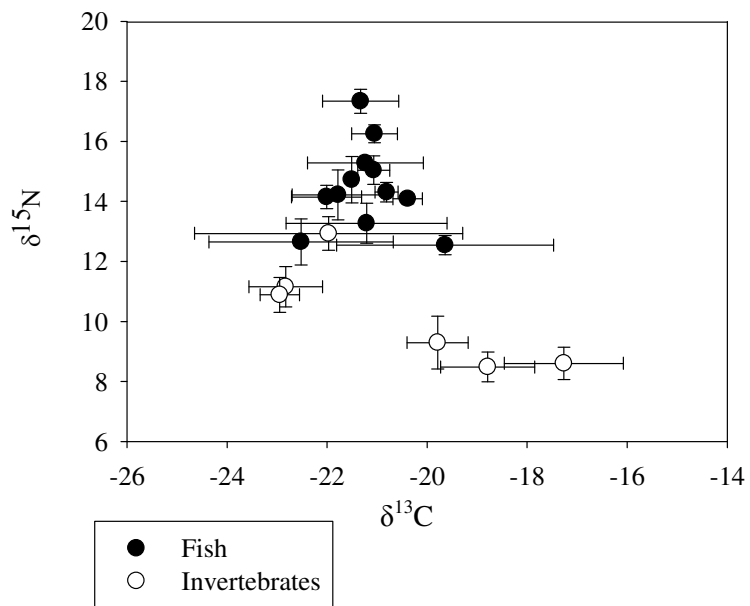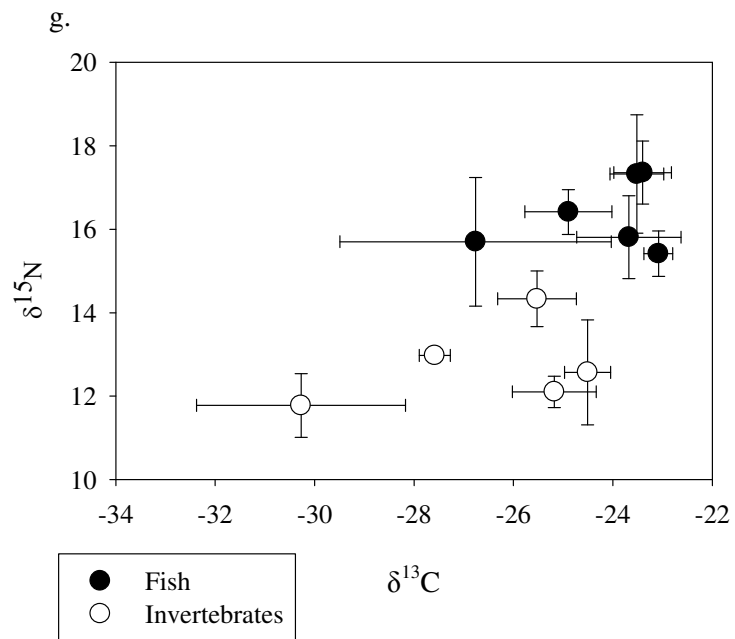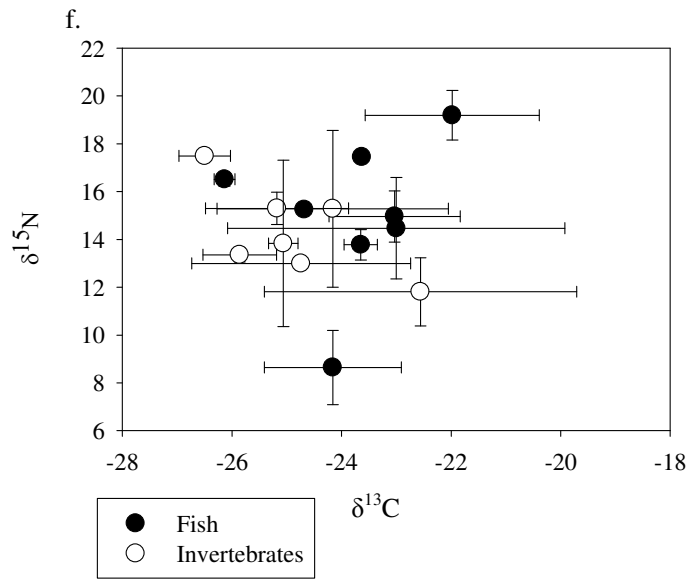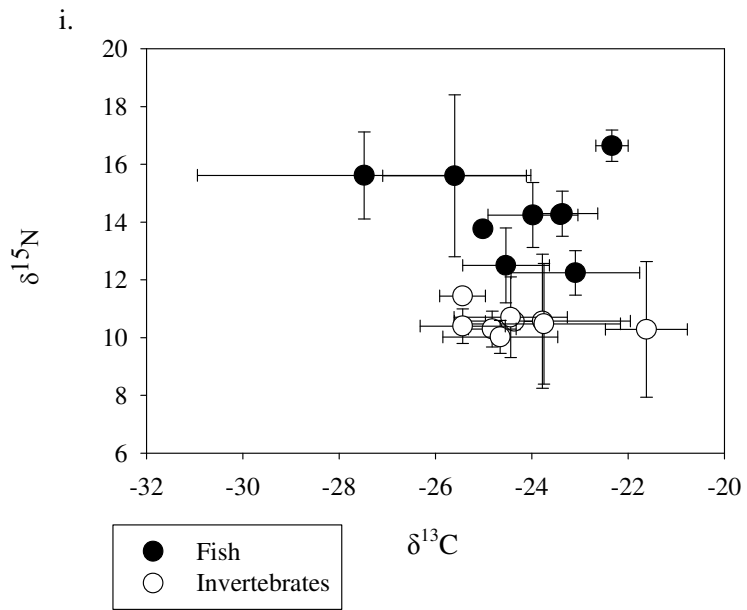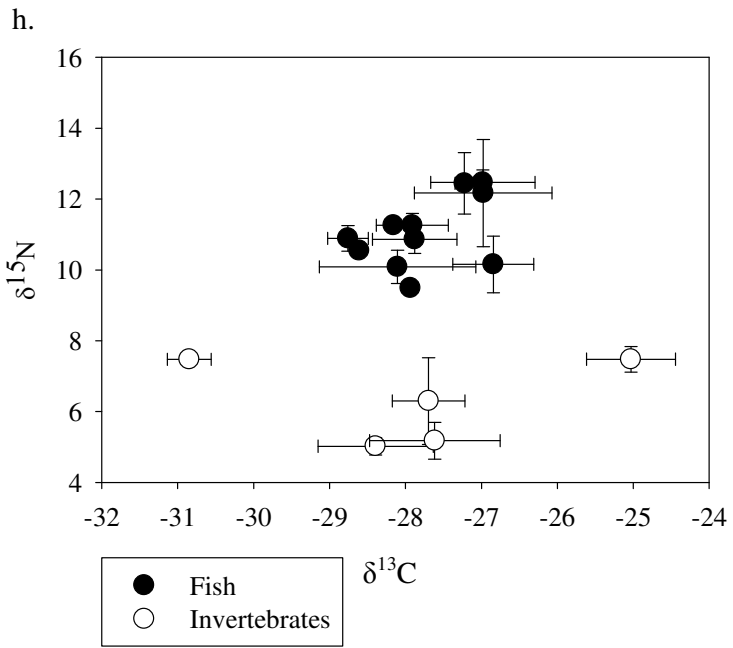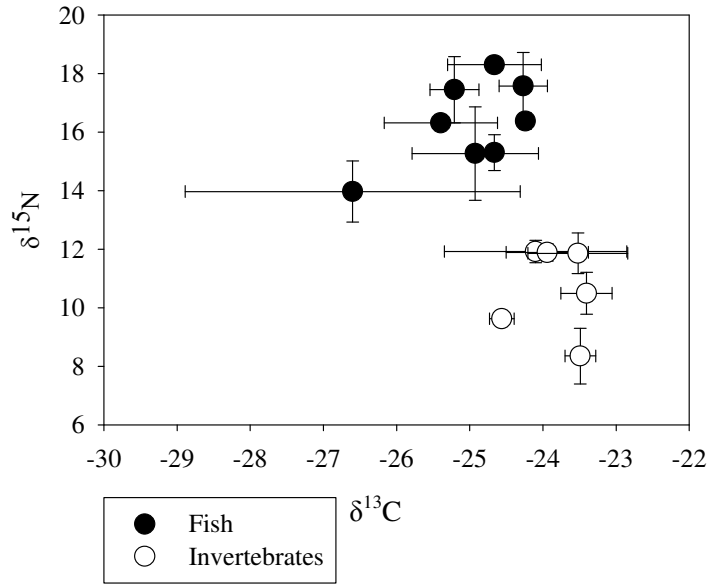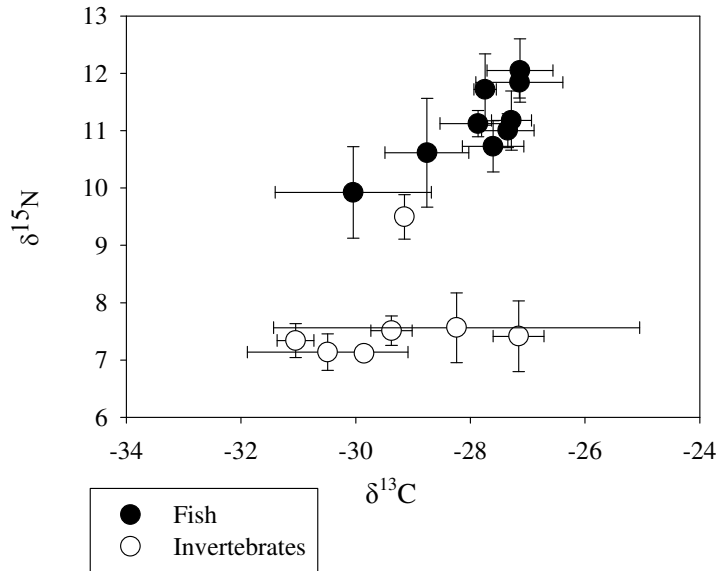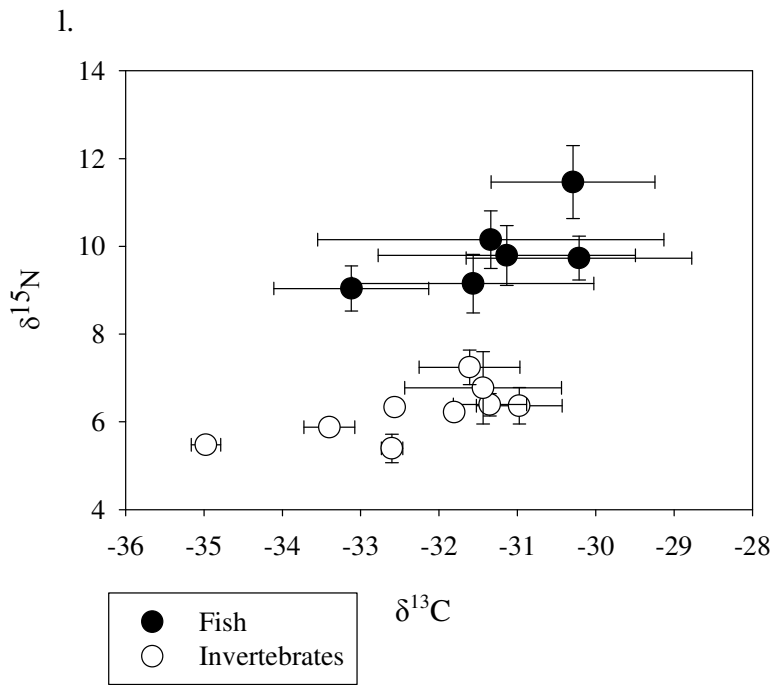
FIGURE 8. The mean trophic position of fish within trophic guilds. Bars represent one standard error.
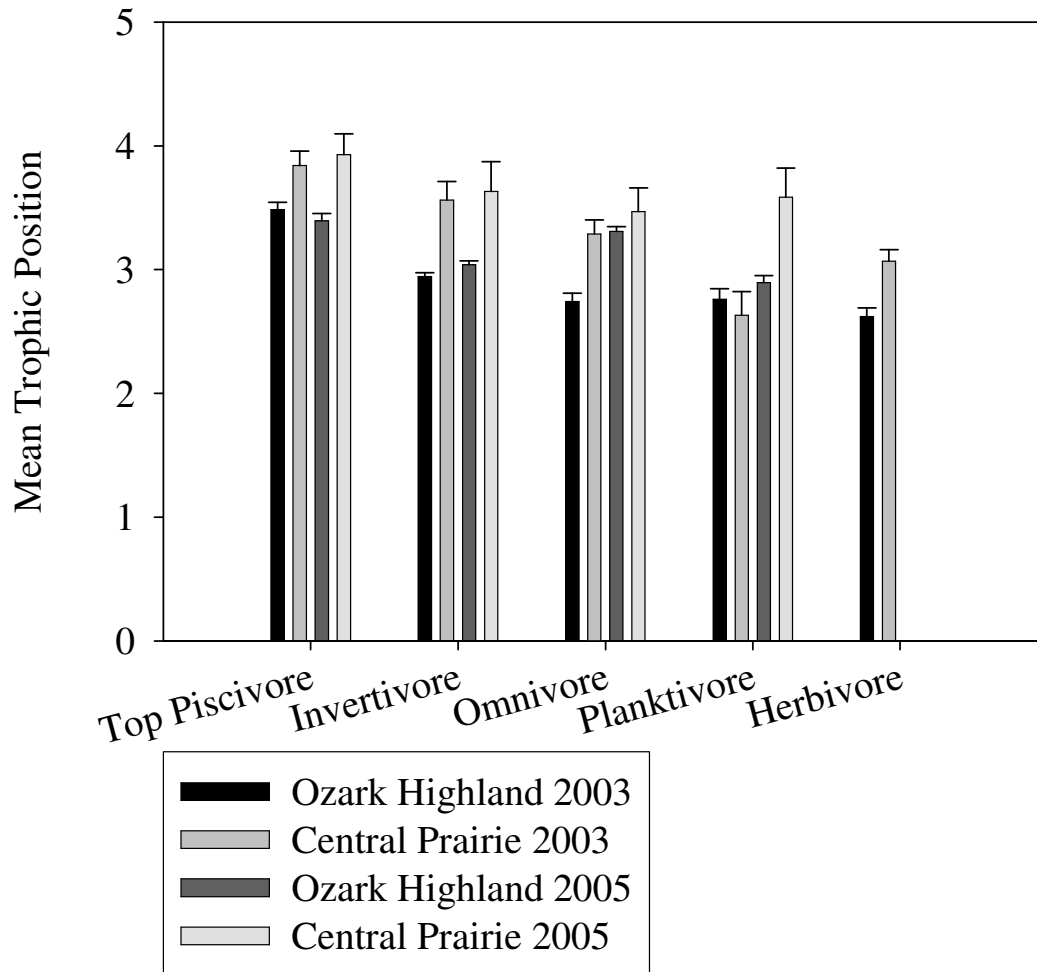
FIGURE 9 Proportion of the number of fish caught within each fish family by the number of fish caught. Fish collected in 2003.
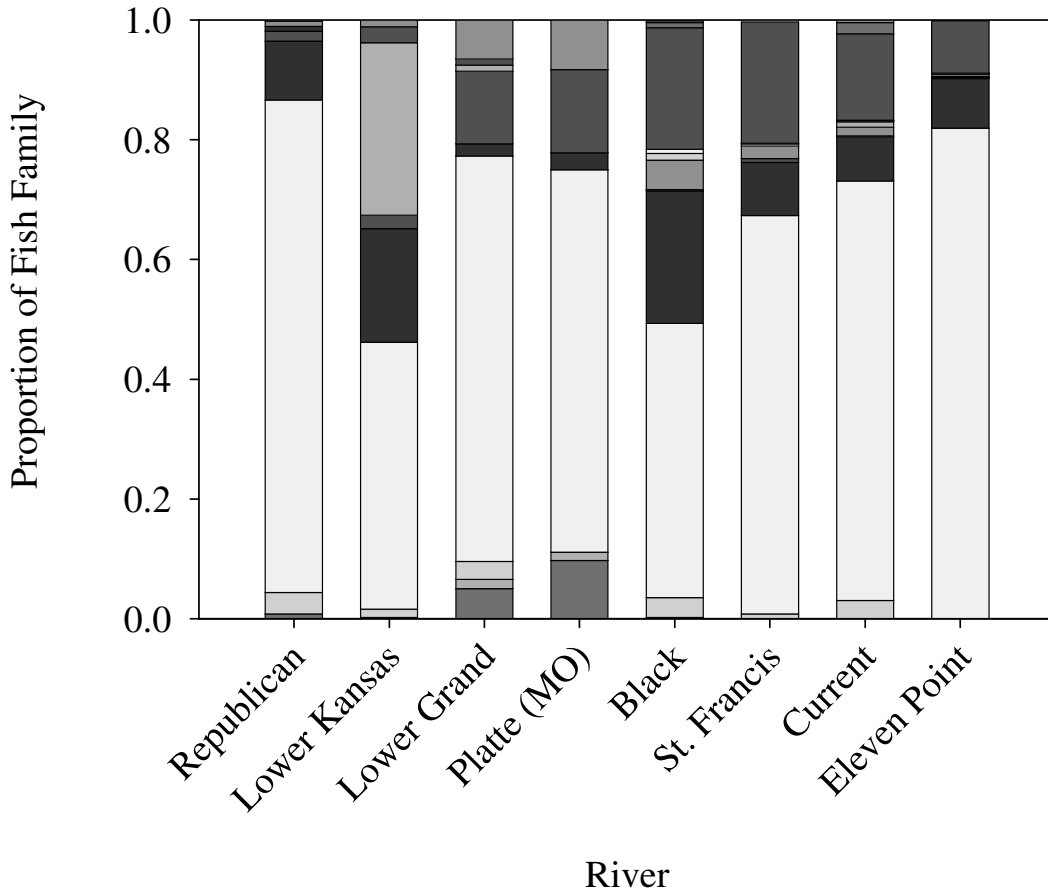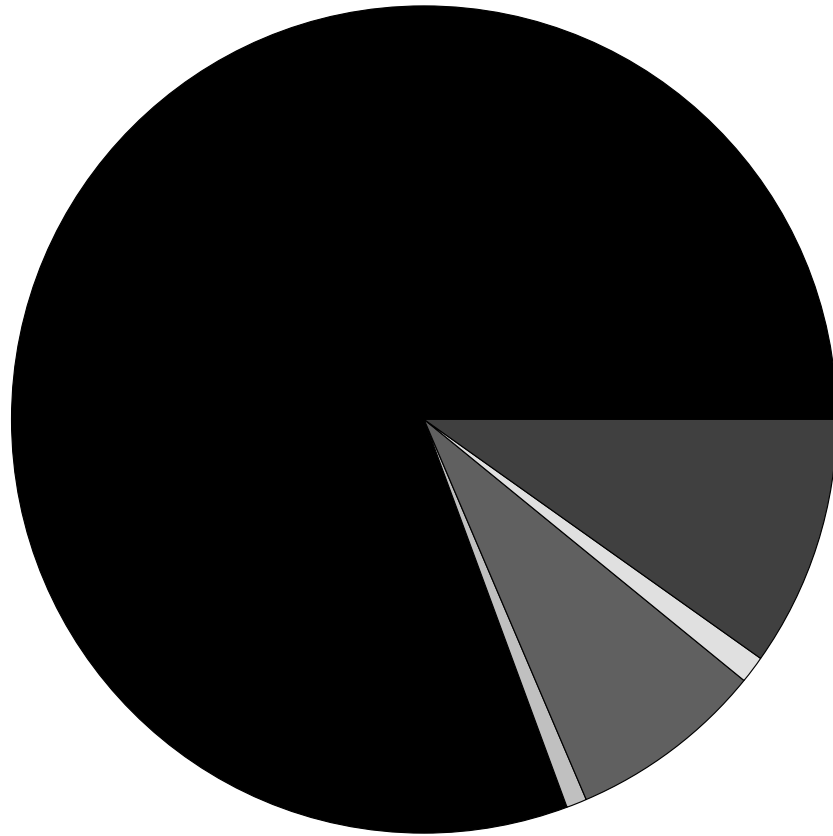
FIGURE 10. Percent of fish species within the Cyprinidae family excluding *Cyprinus carpio.* Rivers are the Republican River (a), Kansas River (b), Grand River (c), Platte River (d), Black River (e), Saint Francis River (f), Current River (g), Eleven Point River (h).
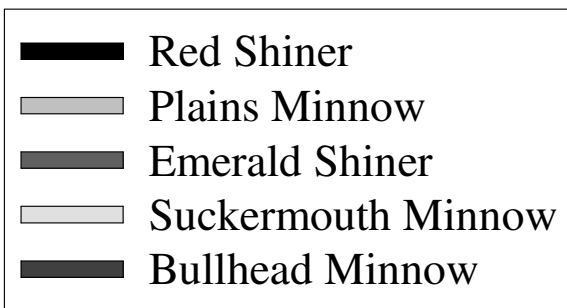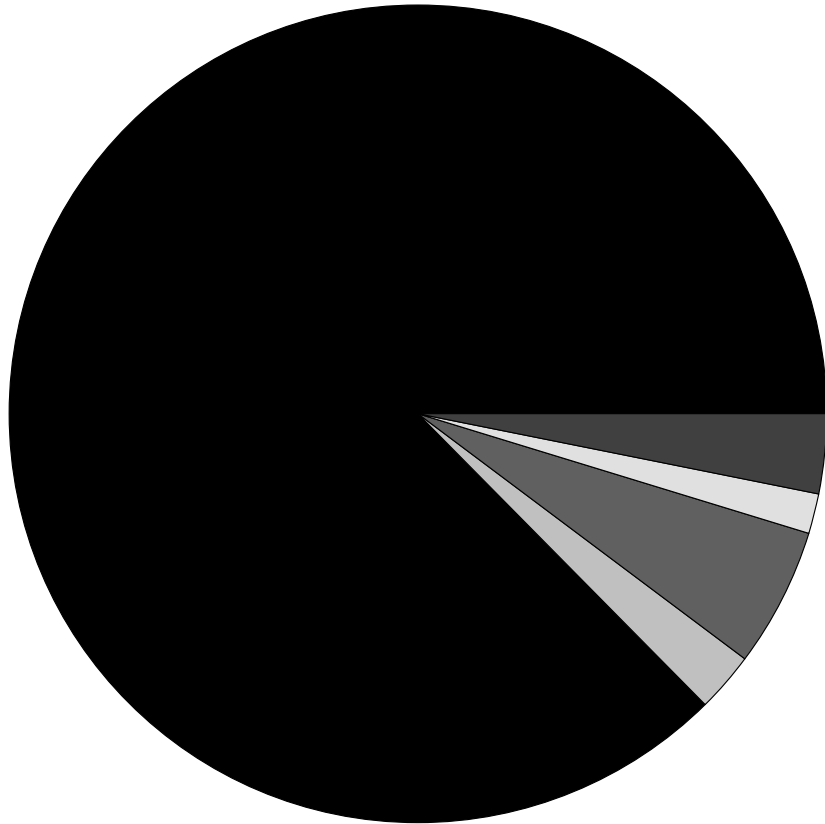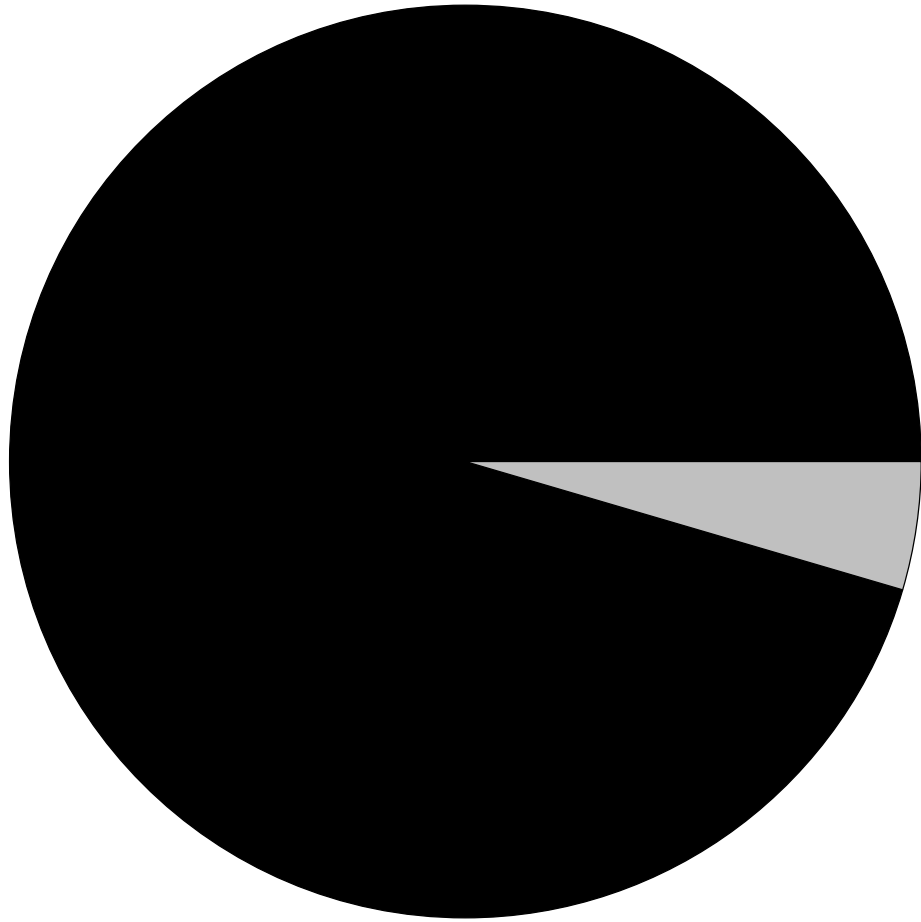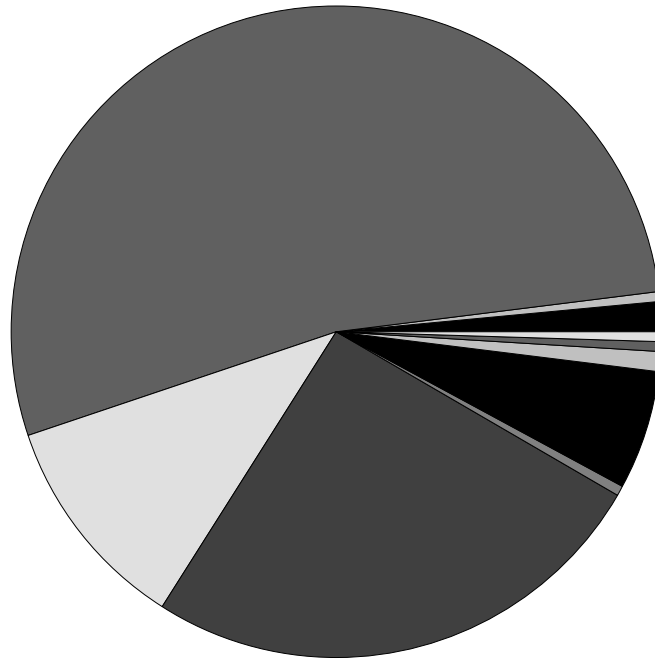
a.

Red Shiner
Emerald Shiner

b.

	Red Shiner
	Emerald Shiner
	Sand Shiner
	Suckermouth Minnow
	Bullhead Minnow

c.

Red Shiner
Plains Minnow
Emerald Shiner
Suckermouth Minnow
Bullhead Minnow

d.

Red Shiner
Emerald Shiner

e.

	Central Stoneroller
	Whitetail Shiner
	Blacktail Shiner
	Mississippi Silvery Minnow
	Emerald Shiner
	Bigeye Shiner
	Wedgespot Shiner
	Sabine Shiner
	Bluntnosed Minnow
	Bullhead Minnow

f.

Central Stoneroller
Steelcolor Shiner
Ozark Chub
Striped Shiner
Emerald Shiner
Bigeye Shiner
Bluntnosed Minnow

g.
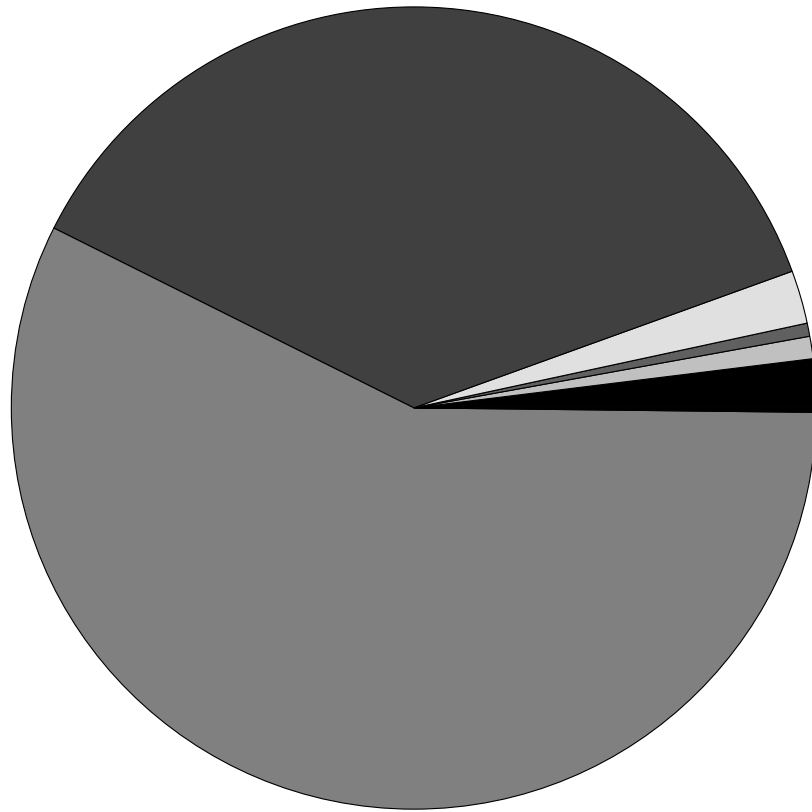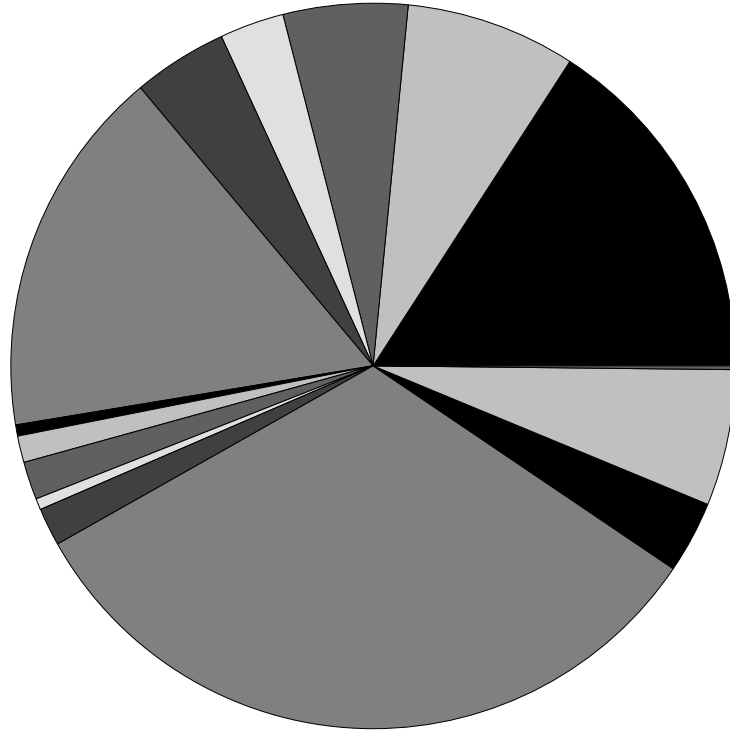

Largescale Stoneroller
Central Stoneroller
Whitetailed Shiner
Ozark Chub
Striped Shiner
Bleeding Shiner
Hornyhead Chub
Bigeye Chub
Emerald Shiner
Bigeye Shiner
Wedgespot Shiner
Ozark Minnow
Rosyfaced Minnow
Telescope Shiner
Bluntnosed Minnow

h.

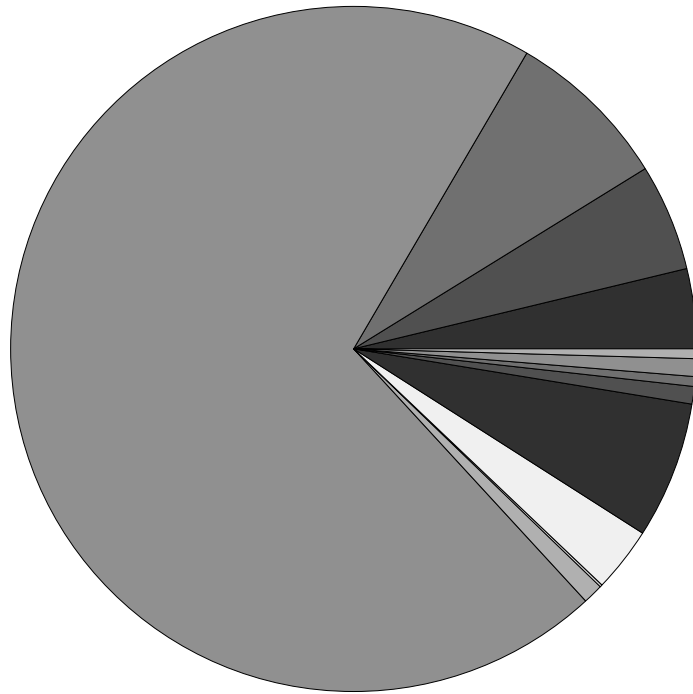|	Largescale Stoneroller
	Central Stoneroller
	Striped Shiner
	Bleeding Shiner
	Hornyhead Chub
	Bigeye Chub
	Emerald Shiner
	Ozark Minnow
	Rosyfaced Minnow
	Telescope Shiner
	Bluntnose Minnow
	Bullhead Minnow

TABLE i. Sizes of watersheds

Sizes of extent	2-HUC (ha)	21-HUC (ha)	Whole Watershed (ha)
Black	17,362.53	204,158.70	311,291.11
Current	12,469.05	212,572.89	531,661.76
Eleven Point	12,824.64	256,135.69	256,467.34
Grand	15,078.78	130,128.12	1,963,263.39
Kansas	15,435.63	193,398.30	3,606,287.87
Platte	4397.31	119,286.63	614,444.69
Republican	14,968.62	218,139.48	6,385,525.10
Saint Francis	14,542.38	216,620.55	242,813.26

FIGURE ii: Trophic position of species caught.

Common name	N	Scientific name	Trophic Guild	Trophic Position
Black crappie	4	*Pomoxis nigromaculatus*	Piscivore	2.89
Bleeding shiner	5	*Luxilus zonatus*	Invertivore	3.01
Bluegill sunfish	25	*Lepomis macrochirus*	Invertivore	3.29
Bluntnose shiner	4	*Pimephalus notatus*	Omnivore	2.90
Channel catfish	34	*Ictalurus punctatus*	Omnivore	3.18
Common carp	11	*Cyprinus carpio*	Omnivore	2.78
Drum	14	*Aplodinotus grunniens*	Invertivore	3.75
Dusky darter	3	*Percina sciera*	Invertivore	2.62
Emerald shiner	26	*Notropis antherinoides*	Planktivore	2.73
Fantailed darter	1	*Etheostoma flabellare*	Invertivore	2.59
Flathead catfish	13	*Pylodictis olivaris*	Piscivore	3.48
Gar (Long and short)	11	*Lepisosteus* spp.	Piscivore	4.11
Gizzard shad	20	*Dorosoma cepedianum*	Planktivore	3.15
Golden Redhorse	30	*Moxostoma erythrurum*	Invertivore	2.93
Goldeneye	3	*Hiodon alosoides*	Invertivore	4.33
Green sunfish	5	*Lepomis cyanellus*	Piscivore	3.36
Hornyhead chub	1	*Nocomis biguttatus*	Omnivore	3.02
Largemouth bass	23	*Micropterus salmoides*	Piscivore	3.64
Logperch	21	*Percina caprodes*	Invertivore	2.94
Longear sunfish	9	*Lepomis megalotis*	Invertivore	3.04
Mississippi silvery minnow	3	*Hybognathus nuchalis*	Herbivore	3.07
Mosquito fish	2	*Gambusia affinis*	Invertivore	2.88
Rainbow darter	1	*Etheostoma caeruleum*	Invertivore	3.11
Red shiner	14	*Cyprinella lutrensis*	Omnivore	3.46
Redear sunfish	3	*Lepomis microlophus*	Invertivore	3.05
River carpsucker	24	*Carpiodes carpio*	Omnivore	3.43
Rock bass	4	*Ambloplites rupestris*	Piscivore	3.53
Rosyface shiner	2	*Notropis rubellus*	Insectivore	3.01
Shadow bass	4	*Ambloplites ariommus*	Piscivore	3.66
Smallmouth bass	12	*Micropterus dolomieu*	Piscivore	3.68
Spotted bass	15	*Micropterus punctulatus*	Piscivore	3.22
Spotted sucker	2	*Minytrema melanops*	Invertivore	3.34
Stoneroller	20	*Campostoma* spp.	Herbivore	2.62
White Crappie	10	*Poxomis annularis*	Piscivore	4.14