LONG-TERM CHANGES IN BENTHIC MACROINVERTEBRATE ASSEMBLAGES OF THE WEST FORK WHITE RIVER (1979-2015)

A THESIS SUBMITTED TO THE GRADUATE SCHOOL

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Acknowledgements

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Introduction

Benthic macroinvertebrates are frequently used for rapid bioassessment of water resources (Resh, Norris, and Barbour, 1995). Aquatic macroinvertebrate assemblage data provide assessment of biodiversity, water quality, hydrology and overall stream health. Due to the high variability among taxa in trophic functions and sensitivity, benthic macroinvertebrates are frequently used as water quality indicators (Cairns and Pratt, 1993). Temporal variation in macroinvertebrate assemblages is influenced by landuse, hydrologic alterations, pollution, and urbanization, resulting in long-term assemblage shifts (Usseglio-Polatera, and Beisel, 2002). Macroinvertebrate assemblages are also influenced through long-term and seasonal changes in precipitation and temperature (Pletterbauer, Melcher, and Graf, 2018), predator-prey interactions (Boelter et al., 2018), and habitat availability (Buss et al., 2004). Long-term temporal data provide insight into the intricate abiotic and biotic interactions resulting from natural or anthropogenic causes (Jackson and Füreder, 2006). For example, Stone and Wallace (1998) observed spatial and temporal variation within macroinvertebrate communities using taxonomy and trophic relationships following long-term changes in riparian and stream quality.

The White River, Indiana is a predominately agricultural watershed, but historic urban and industrial impacts resulted in poor water quality (Chauret et al, 2001). Throughout the early to mid-1900's a natural gas boom brought large scale industrialization to Muncie, Indiana (Lynd and Lynd, 1937). Increased industrial and municipal point source pollution was discharged from urban areas of the West Fork White River (Martin and Craig, 1990). Large volumes of municipal pollution reach the river through combined sewage overflows, which release raw sewage during heavy rain events (Martin et al., 1996). Other major sources of pollution to the river are from agricultural nonpoint sources (Martin et al. 1996).

Compared to other US river basins, the White River has high concentrations of pesticides at both urban and agriculture sites (Martin et al., 1996). Additionally, trace element concentrations in streambed sediments are generally high but rarely exceed the aquatic life criteria (Martin et al., 1996).

Prior to the Clean Water Act (CWA) in 1972 the West Fork White River was impacted by high concentrations of ammonia, cyanide, and heavy metals (Conrad, 2017). Post-CWA the Muncie Sanitary District's Bureau of Water Quality (BWQ) was tasked with monitoring water quality of the West Fork White River through Muncie, IN. In addition to monitoring water quality, the director John Craddock also monitored the biological community through annual fish and macroinvertebrate collections. The BWQ monitors multiple macroinvertebrate indices annually (Bowley, 2018). Annual monitoring surveys resulted in availability of these long-term benthic macroinvertebrate data.

The objectives of this study were to characterize the spatial and temporal variation in macroinvertebrate assemblages using taxonomy, trophic traits, and sensitivity traits during a 30-year period. We hypothesized that there was a major change in assemblage structure of macroinvertebrates in the West Fork White River following the 1972 CWA. We predicted that changes in water quality practices influenced and increased taxa richness and relative abundance of taxa. Spatial variation from upstream to downstream in taxon abundance was expected with land-use modifications and agricultural point/non-point source pollution. We expected urban and downstream communities to be different from upstream communities due to differences in land-use practices, and municipal pollutant discharge among river reaches. We predicted increased abundances of specialized trophic types and sensitive taxa due to decreases in heavy metal concentrations, following large-scale reductions in industrialization. Characterization of temporal and spatial variation among macroinvertebrate communities can provide an assessment of how large-scale changes in water quality practices from the CWA affected benthic macroinvertebrate communities in the West Fork White River in Muncie, Indiana.

Methods

Field Collections

Macroinvertebrates were collected by the Muncie Sanitary District's Bureau of Water Quality during monitoring surveys on the West Fork White River and its tributaries in Delaware and Randolph County, IN. Ten sites located on the mainstem of the West Fork White River were sampled annually from 1979-2015 (Figure 1). From 1979-2008 the BWQ used an in-house protocol for macroinvertebrate collections (Craddock, 1980). Minor modifications of the sampling procedure occurred through the period. After 2007, to gain consistency among local samples, the protocol from the Indiana Department of Environmental Management (IDEM) was used. We excluded data collected in 2008 as it was distinctly modified from other collection methods. In 2009, IDEM used the Rapid Bioassessment Protocol using d-nets, similar to previous sampling. In 2010, IDEM adopted a Multi-habitat Macroinvertebrate Collection Procedure (mHAB), similar to the BWQ original method, with the exception of using only one sampler. The mHAB method includes a one-min riffle kick, if there is no riffle a mid-stream kick is used, and a 12-min 50 m bank sample (Bowley, 2018). After a six-time elutriation, samples were poured through a #30 USGS sieve, and contents were subsampled for 15 min (Bowley, 2018). Macroinvertebrates were identified in a BWQ lab to lowest taxonomic level.

Data Analysis

We used family-level taxonomy for macroinvertebrates because all data were not to the same resolution. Potential taxonomic errors were omitted from the data when we detected them (e.g., erroneous taxon names) and we deleted terrestrial macroinvertebrates. Collections were reported as abundance of taxa collected at sites. Taxa abundance at sites was summed to create annual abundance and a pearson correlation analysis was used to determine if taxa richness increased.

We used nonmetric multidimensional scaling (NMDS) in R (R Core Team, 2016) using the vegan package version 2.4-2 (Oksanen, 2017; ordiellipse and anosim functions) to summarize temporal and spatial relationships. Variation in macroinvertebrate family abundances were represented by pairwise Bray-Curtis distances in the NMDS and reduced to a two-dimensional configuration. Final configurations were calculated 20 times from a random starting arrangement, and the lowest stress for permutations was used. Stress ≤ 0.20 is considered useful for pattern analysis (Clarke, 1993). NMDS is useful to graphically represent large ecological datasets with fewer distribution assumptions (Kenkel and Orlció, 1986).

We defined river reaches as upstream, urban, and downstream of Muncie, Indiana for analyses. Upstream and downstream sites were beyond Muncie city limits, and urban sites were within city boundaries. We combined annual collections into decades to detect temporal variation that was not obvious in our pilot analyses of annual collections. For example, years 1980-1989 were grouped and the ordiellipse function created ellipses around data within the NMDS bi-plot that represented those years. The same process was used to plot data for spatial analyses. Anosim was used to test for statistical differences among decades and spatial trends.

Macroinvertebrate families were classified into trophic and sensitivity traits and tested for temporal variation using the same analyses as above. Trophic relationships were obtained from Merritt and Cummins (1996) and Hauer and Resh (2011). Sensitivity trait scores were from Barbour et al. (1999). We used a sensitivity scale of 0-10, with 0 the most sensitive and 10 the least sensitive. In long term literature ‘rarity’ has different meanings and implications, where taxa might be only temporally rare, but common in long-term analysis (Resh et al., 2005). We selected not to use the common 5% rarity cutoff for taxa because the resulting assemblage was reduced to only five families. We identified rare taxa as 0.05% of total abundance and excluded them from further analyses. We also log (x + 1) transformed abundance data prior to analyses as data had three orders of magnitude in abundance variation. Analyses were repeated with rare taxa and we found similar temporal and spatial trends.

Results

Our dataset consisted of 33 years of collections from 1979-2015. Data from 1989, 1990, 2003, and 2008 were omitted due to sampling or data inconsistencies, resulting in 77 families and 92,477 individuals. Taxa richness increased over the study ($r = 0.62$, Fig. 2). Our NMDS used 27 taxa with 87,569 individuals, after deleting rare taxa. Three phyla were present continuously throughout the data set. Annelida included *Glossiphoniidae*, and *Oligochaeta*. *Oligochaeta* was included in analyses based on abundance variation throughout the time period. Mollusca consisted of two families, *Cyrenidae* and *Physidae*. Arthropoda comprised 21 families. Arthropod taxa were in 8 orders, Amphipoda (1), Coleoptera (3), Diptera (2), Ephemeroptera (6), Hemiptera (5), isopoda (1), Odonata (2), and Trichoptera (1). The five taxa (insect families) with highest abundances were *Hydropsychidae* (Trichoptera), *Chironomidae* (Diptera), *Elmidae* (Coleoptera), *Coenagrionidae* (Ephemeroptera), and *Caenidae* (Ephemeroptera) that comprised ~55% of all individuals. *Hydropsychidae* comprised 19.7% of the total dataset. *Chironomidae* 12.2%, *Elmidae* 11.3%, *Coenagrionidae* 6.4%, and *Caenidae* comprised 5.4% of all individuals collected.

The NMDS identified high temporal variation as assemblage structure differed significantly among decades (stress: 0.14, R: 0.45, $p \leq 0.01$) (Fig. 3). Dominant taxa ($\geq$ 5% of collection) varied throughout the study. Dominant taxa during the 1980's were *Hydropsychidae, Coenagrionidae, Chironomidae, Elmidae,* and *Baetidae*. In the 1990's dominant taxa were slightly different with *Hydropsychidae*, *Elmidae*, *Chironomidae*, *Caenidae*, and *Hydrophilidae* in the highest abundance. The 2000's resulted in a change in dominant taxa with *Elmidae* (22% of collection) more dominant than *Hydropsychidae*. Additional changes in dominant taxa through the 2000's include *Leptohyphidae* that replace *Hydrophilidae*. Finally, dominant taxa in the 2010's consisted of *Hydropsychidae, Chironomidae*, *Elmidae*, *Veliidae*, and *Leptohyphidae* (Fig. 4). Major changes in abundance of several families occurred from 1980-2000 (e.g., *Elmidae* 16% increase, and *Coenagrionidae* 12.7% decrease). Additionally, *Elmidae* taxa abundance from 2000-2015 decreased 10.7%.

Spatial analyses indicated significant differences by river location (stress: 0.20, R: 0.32, $p \leq 0.01$) (Fig. 5). Exploratory analyses of spatial groups showed differences in relative abundance of families at upstream, urban, and downstream sites (e.g., *Chironomidae* and *Elmidae*) (Fig. 6). *Chironomidae* comprised 16% of all individuals at urban sites, 12% downstream, and 7% upstream. *Elmidae* varied by river location and comprised 11% of collection at urban sites, 7% downstream, and 15% upstream.

We detected five trophic traits: collectors, filterers, predators, scrapers, and shredders. Dominant trophic traits were collectors that comprised 53% of individuals, and predators at 33% of individuals. Filterers, scrapers, and shredders comprised ~14% of individuals collected. Sensitivity traits were distributed into seven values (2,4,5,6,7,8,9). Individuals in families that tended to be more sensitive to disturbance (scores 2-5) comprised ~52% of total abundance, and less sensitive families comprised ~48% of individuals. The highest abundance (42.7%) of individuals were scored with a sensitivity score of 4. The relative abundance of the five trophic traits among decades and river location had no noticeable changes. No major changes were found in sensitivity scores by decade or river location. NMDS and anosim analyses by trophic relationship and sensitivity did not result in significant temporal or spatial patterns.

Individual river locations by decade resulted in a similar significant decadal variation pattern for each location in NMDS analyses in Fig. 7 (downstream reach: stress: 0.15, R: 0.42, $p \leq 0.01$), (urban reach: stress: 0.15, R: 0.39, $p \leq 0.01$), (upstream reach: stress: 0.17, R: 0.41, $p \leq$

0.01. We found significant differences among river locations for macroinvertebrate assemblages by decade in NMDS analyses in Fig. 8 (1980s: stress: 0.14, R: 0.61, $p \leq 0.01$), (1990s: stress: 0.13, R: 0.54, $p \leq 0.01$), (2000s: stress: 0.14, R: 0.32, $p \leq 0.01$), (2010-2015: stress: 0.16, R: 0.14, $p \leq 0.05$).

Discussion

We detected major changes in macroinvertebrate assemblage structure of the West Fork White River in Muncie during a 30-year period. The variation that we observed was likely responses to decreased point source pollution following the Clean Water Act of 1972. Significant temporal and spatial trends (Figs. 3 and 5) were likely a result of changes in water quality practices. We found high variation in relative abundances of the dominant taxa during this period, that likely caused the temporal patterns in NMDS analyses. The largest increase in abundance was for *Elmidae* from 1980-2000, likely due to improved water quality practices and reductions in hydrologic alterations (Conrad, 2017). Other studies of benthic macroinvertebrates found that large changes in anthropogenic practices directly impacted community composition (Azrina et al., 2006). For example, *Elmidae* taxa require well-oxygenated flowing water (Elliott, 2008). Point source discharge was reduced throughout the White River basin after 1980 (Martin and Craig, 1990). These reductions likely provided increased habitat with suitable dissolved oxygen for taxa like *Elmidae*, and resulted in increased relative abundance after 1980. *Coenagrionidae* had the largest reduction in relative abundance during this period. Damselfly communities are strongly influenced by predator-prey interactions of fish communities (McPeek, 1998). Increased abundances of fishes that prey on benthic insects (largemouth bass *Micropterus salmoides*, golden shiner *Notemigonus crysoleucas*, white crappie *Pomoxis annularis*) after 1980 were observed by Holloway, Doll, and Shields (2018) in a long-term study of the White River, Indiana.

Additional modifications to the West Fork White River during this period included reductions in nutrient discharge, industrial pollutants, and hydrologic alteration (Martin et al., 1996). A major change in socioeconomic status of citizens in the West Fork White River watershed occurred simultaneously (Tamney and Johnson, 1983). Muncie was an industrial manufacturing city at the turn of the 20th century, but the majority of this industry is gone. Impacts from industry were primarily point-source pollution that included heavy metals and other wastes (Conrad, 2017). Holloway, Doll, and Shields (2018) reported significant reductions in heavy metal concentrations from 1980-2016 throughout the West Fork White River in Muncie, Indiana. Heavy metals including iron in industrial effluent alter macroinvertebrate abundance, species composition, and relative abundances of sensitive taxa (Nedeau et al., 2003). Other current and historic major human impacts to the White River include sewage discharge and agricultural landuse (Martin et al. 1996). The city of Muncie currently treats sewage prior to release, but nutrients and pharmaceuticals are still present (Bunch and Bernot 2011, Veach and Bernot 2011).

Macroinvertebrate assemblages that are upstream from Muncie likely have been different from urban and downstream sites due to watershed size differences, agricultural land use, and urban impacts. Watershed land use is predominately agriculture: the area of Delaware County is 253,000 acres and approximately 152,000 acres are corn and soybeans (Tedesco et al., 2011). Agriculturally influenced streams significantly impact the total number of EPT taxa and community composition (Richards, Host, and Arthur, 1993). We found that *Chironomidae* and *Elmidae* displayed the greatest spatial difference in relative abundance. *Chironomidae* abundance was higher at urban and downstream sites, likely because members of this family tent to have a high tolerance to changes in water chemistry parameters (Shimba, Mkude, and Jonah, 2018). Urban and downstream sites had greater influence from urban land-use, urban stream channelization, and current impacts from combined sewage overflows. *Elmidae* had the highest relative abundance at upstream sites. Decreased channelization at upstream sites may result in increased riffle-run development, and suitable habitat for *Elmidae*. Hydrologic regime variability provides necessary temporal and spatial variation in benthic macroinvertebrate assemblages (Resh et al., 1988).

We used trophic and sensitivity traits as an alternative to taxonomy, because we expected that organisms with similar trophic relationships and sensitivity to react similarly to human impacts. Floury et al. (2013) showed that long-term macroinvertebrate communities became dominated by taxa that are generalist for pollution-tolerance. In addition, Vaughan and Ormerod (2012) found that macroinvertebrate communities were resilient to long-term anthropogenic disturbances. However, we were limited to defining sensitivity and trophic relationships at the family level. Our results may differ if taxa were categorized by genus or species for sensitivity and trophic traits (Merritt and Cummins 1996). The West Fork White River fish assemblage in the early 1980's changed from being dominated by pollution-tolerant taxa to an assemblage currently dominated by sensitive taxa, likely following reductions in heavy metal concentrations (Holloway, Doll, and Shields, 2018). We predict a similar trend in macroinvertebrate assemblages with analyses at lower taxonomic levels.

We observed increased taxa richness that corresponded with increased water quality practices. However, Bowley (2018) reported decreased Shannon-Weiner Diversity Index (H') scores, decreased Macroinvertebrate Index of Biotic Integrity (mIBI), decreased intolerant taxa, and increased dipteran abundance, % non-insects, and % of collectors/filterers for 2017 collections. The improvements we detected from 1980-2015 might have reached an upper limit for this watershed.

We identified significant differences in temporal and spatial assemblages of benthic macroinvertebrates of the West Fork White River. Further analyses into the mechanisms influencing temporal and spatial variation could include detailed benthic macroinvertebrate community metrics collected by the BWQ. Analyses of these metrics with heavy metal concentrations and variation in the fish assemblages (Holloway, Doll, and Shields, 2018) may reveal details for unique temporal and spatial changes of macroinvertebrate assemblages.

Future restoration activities for the river have the potential to further modify to the ecosystem. The removal of low-head dams is predicted to restore some degree of natural hydrology, and thereby change the composition of the benthic macroinvertebrate assemblage (Poff et al. 1997). In addition, re-routing runoff during heavy rain events will mitigate the current impacts of combined sewage overflows. Reduction in sewage effluent has direct effects on community composition and distributions of benthic macroinvertebrates (Wright et al. 1995). In summary, we identified variation during a 30-year period in macroinvertebrate assemblages of the West Fork White River in Muncie, Indiana that corresponded to improvements in water quality.

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Fig. 1 Sample sites on the mainstem West Fork White River for benthic macroinvertebrate from 1979-2015. Sites are black dots and the ellipsis approximates Muncie city limits.

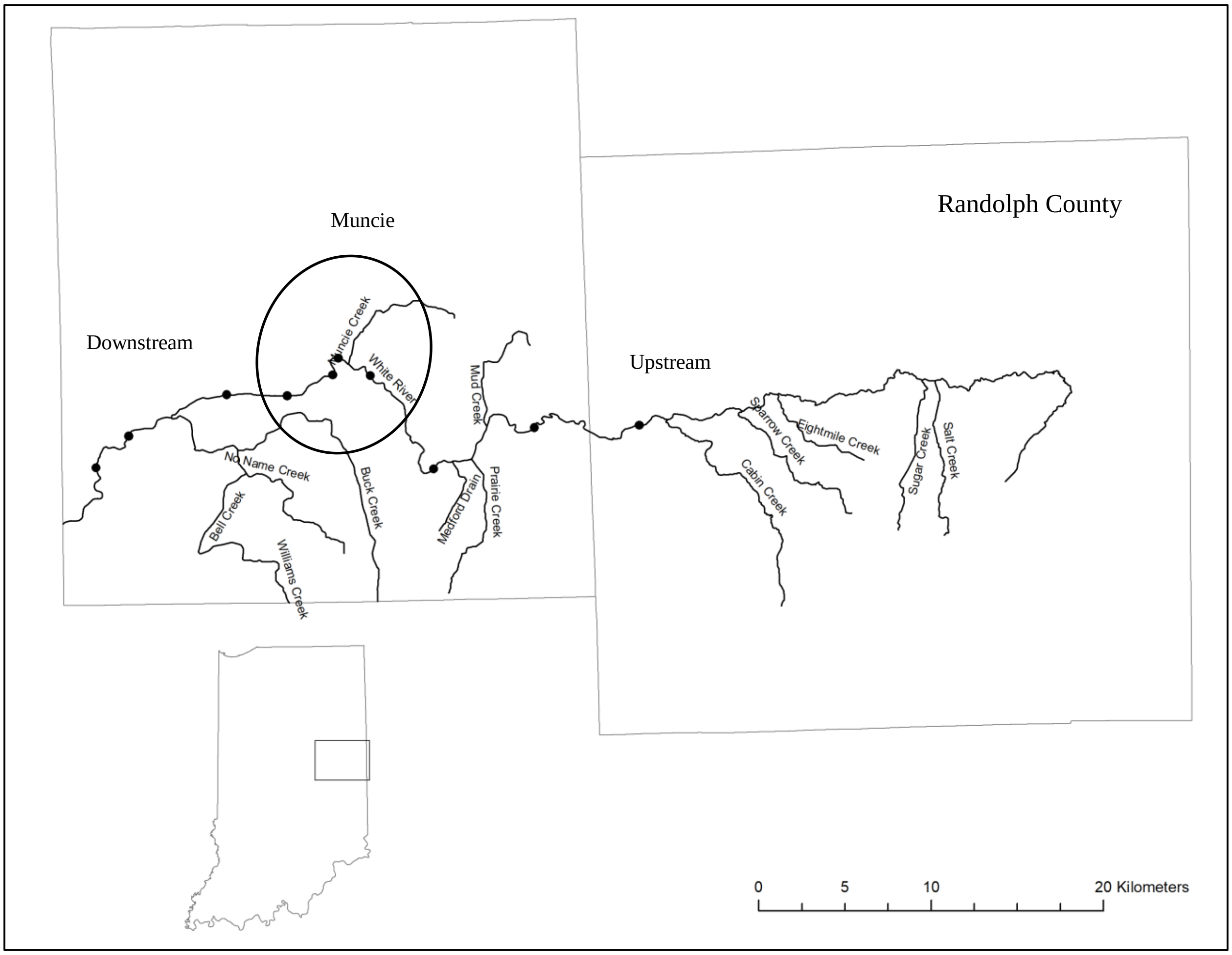

Fig. 2 Taxa richness of benthic macroinvertebrates in annual collections on the mainstem West Fork White River 1979-2015.

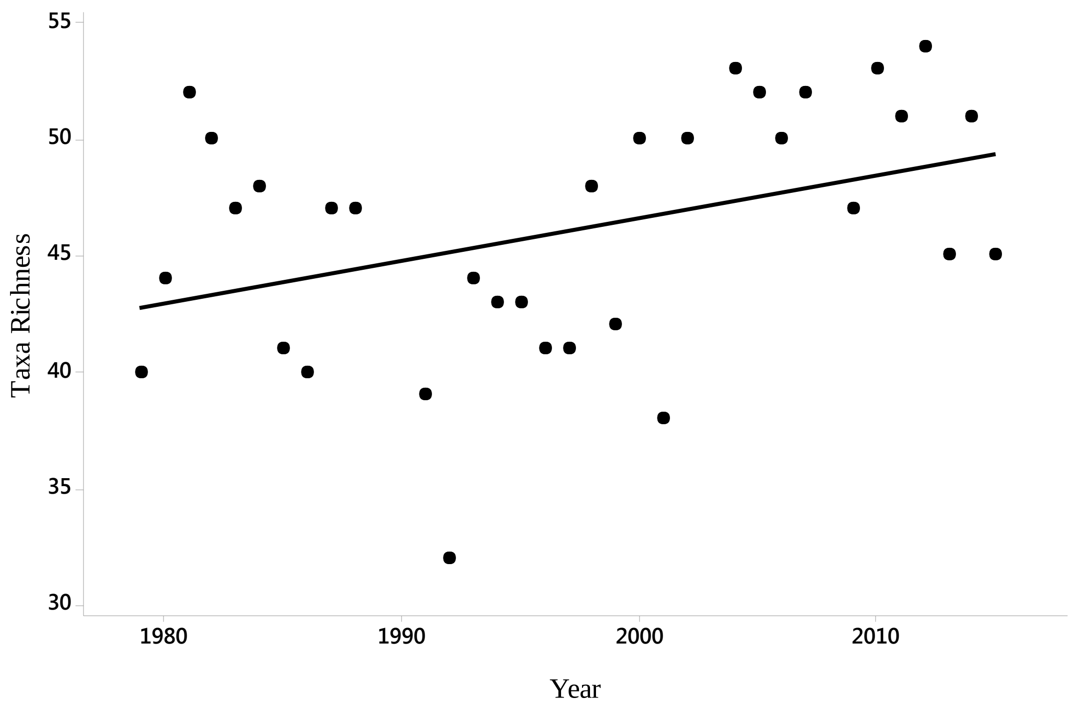

Fig. 3 Non-metric multidimensional scaling biplot of temporal trends in macroinvertebrate assemblages for 1979-2015 West Fork White River. Decades are represented by ellipses, families are indicated by red crosses. Highest loading score taxa are indicated on axes.

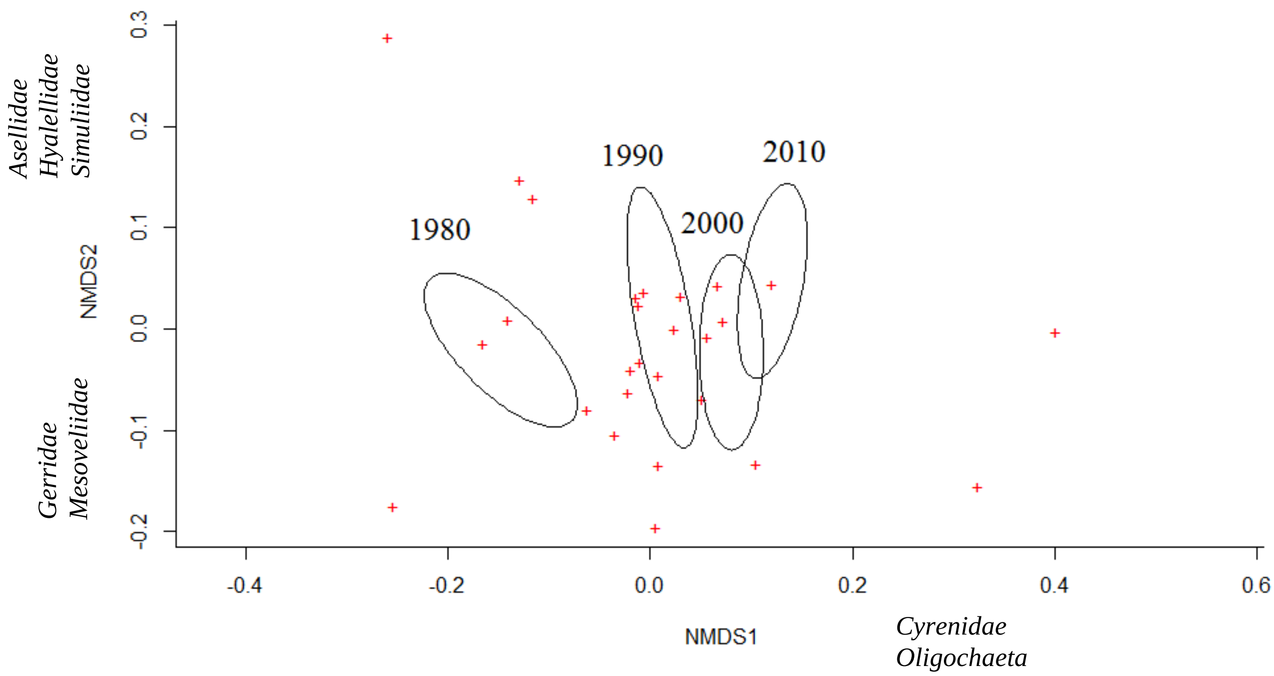

Fig. 4 Decadal time series plot of mean relative abundance of dominant benthic macroinvertebrate families of the West Fork White River.

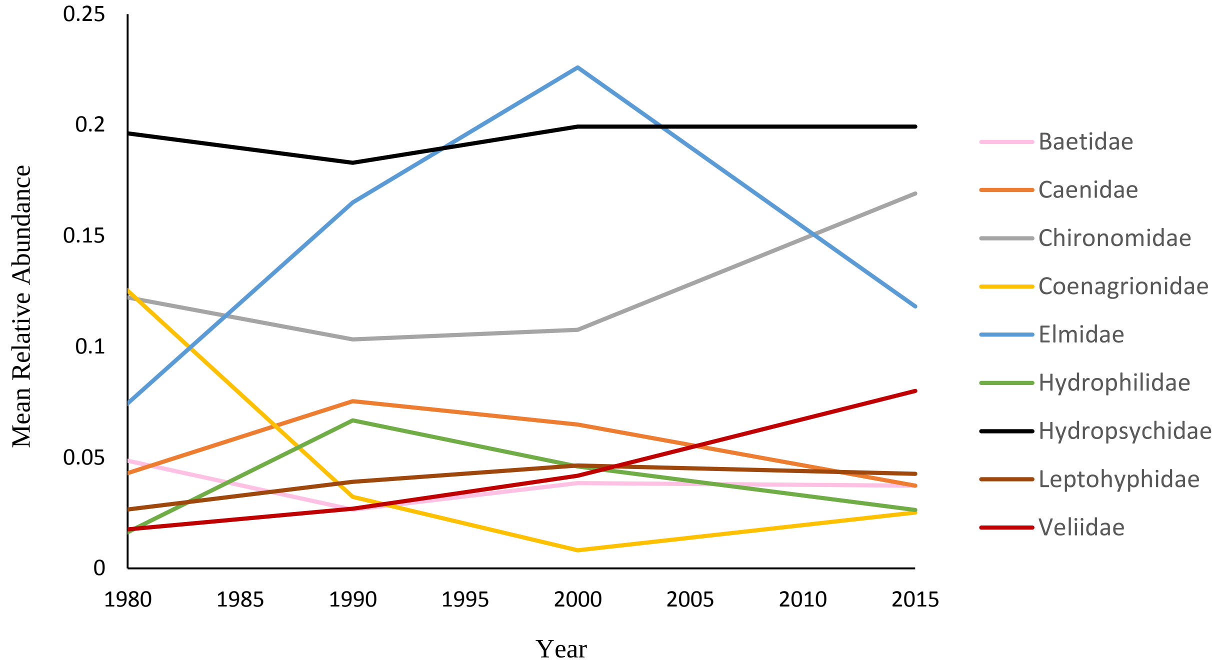

Fig. 5 Non-metric multidimensional scaling biplot of macroinvertebrate assemblages of the West Fork White River 1979-2015. Families are indicated by red crosses and ellipses represent different river locations. Highest loading score taxa are indicated on axes.

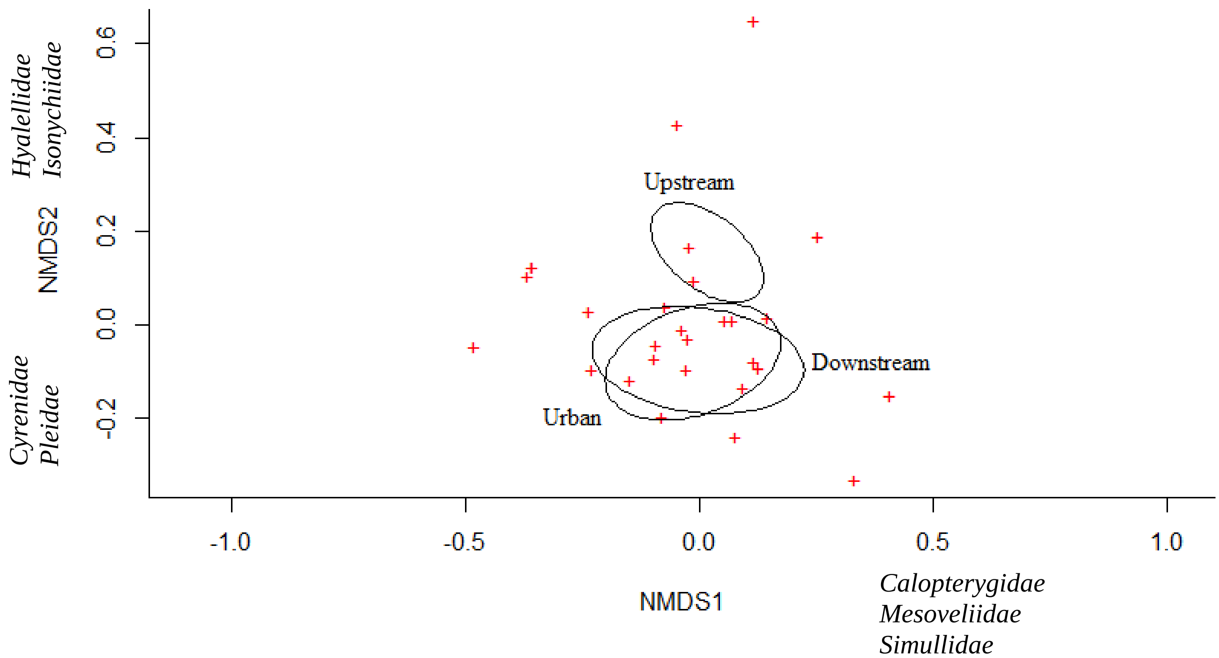

Fig. 6 Mean relative abundance of *Elmidae* and *Hydropsychidae* by river reach for samples collected 1979-2015.

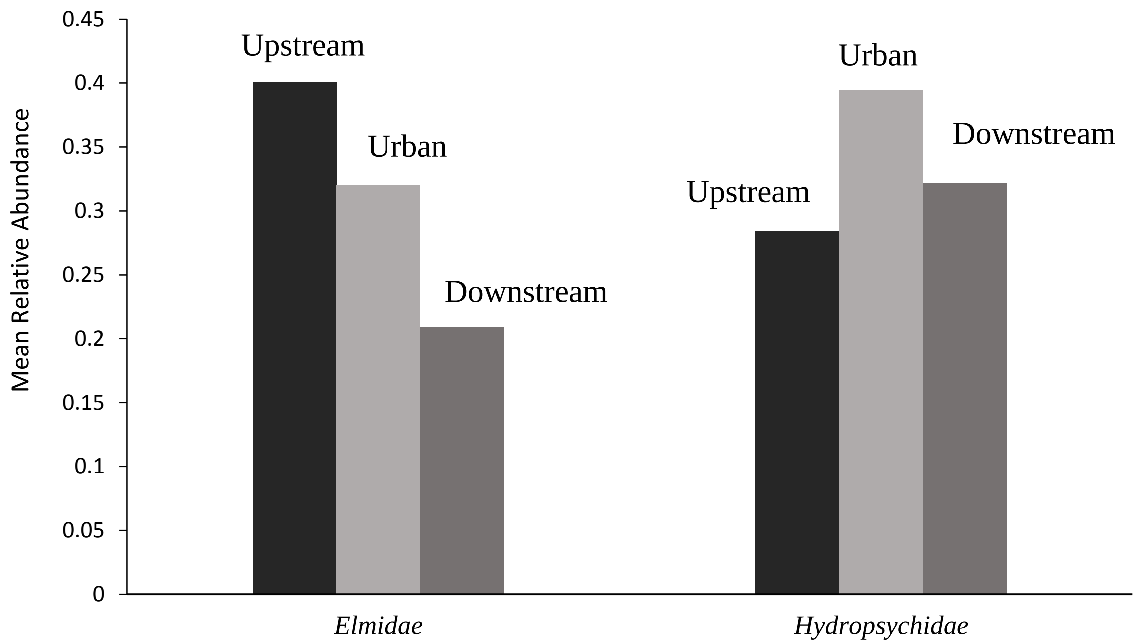

Fig. 7 Non-metric multidimensional scaling biplots of temporal trends by river reach for benthic macroinvertebrates of the West Fork White River. Highest loading score taxa are indicated on axes.

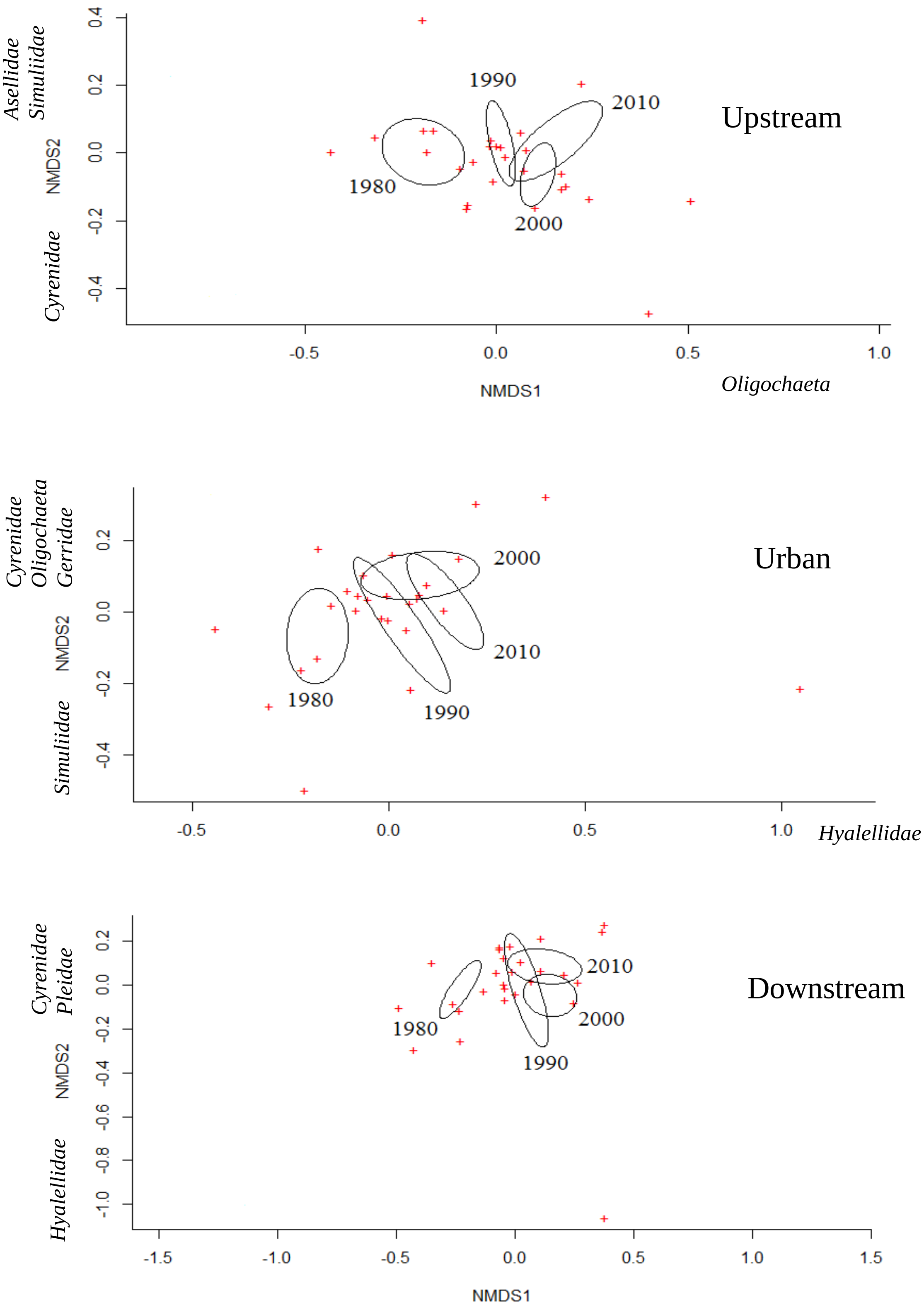

Fig. 8 Non-metric multidimensional scaling biplots of spatial trends by decade for benthic macroinvertebrate collections for the West Fork White River. Highest loading score taxa are indicated on axes.