



2010

Fish Assemblage Structure in Natural, Channelized, and Restored Sections of Nippersink Creek, Mchenry County, Illinois

Sarah Ann Zack
Loyola University Chicago

Follow this and additional works at: https://ecommons.luc.edu/luc_theses



Part of the [Terrestrial and Aquatic Ecology Commons](#)

Recommended Citation

Zack, Sarah Ann, "Fish Assemblage Structure in Natural, Channelized, and Restored Sections of Nippersink Creek, Mchenry County, Illinois" (2010). *Master's Theses*. 553.
https://ecommons.luc.edu/luc_theses/553

This Thesis is brought to you for free and open access by the Theses and Dissertations at Loyola eCommons. It has been accepted for inclusion in Master's Theses by an authorized administrator of Loyola eCommons. For more information, please contact ecommons@luc.edu.



This work is licensed under a [Creative Commons Attribution-Noncommercial-No Derivative Works 3.0 License](#).
Copyright © 2010 Sarah Ann Zack

LOYOLA UNIVERSITY CHICAGO

FISH ASSEMBLAGE STRUCTURE IN NATURAL, CHANNELIZED, AND
RESTORED SECTIONS OF NIPPERSINK CREEK, MCHENRY COUNTY, ILLINOIS

A THESIS SUBMITTED TO
THE FACULTY OF THE GRADUATE SCHOOL
IN CANDIDACY FOR THE DEGREE OF
MASTER OF SCIENCE

PROGRAM IN BIOLOGY

BY

SARAH A. ZACK

CHICAGO, IL

MAY 2010

Copyright by Sarah Ann Zack, 2010

All Rights Reserved

ACKNOWLEDGEMENTS

There are a lot of people who deserve thanks for helping me see this project through to the end. First and foremost, I'd like to thank my family. The field work never would have been completed without my dad and brother, Kent and Mike Olson, who helped me sample the river no matter how cold it was, or my mother, Sue Olson, who always packed the most fantastic picnic lunches so we wouldn't starve during those long days in the field. This thesis never would have been written without the support of not only my parents, but also my wonderful husband, Conrad Zack, who has always supported and encouraged me. Your help and support is the best gift you have ever given me, and I could not have finished this without all of you poking and prodding at me this past year! Thank you, I love you all. Knowing you all believed I could finish this thesis was all the encouragement I needed to keep plodding along.

To my thesis directors, Drs. Martin Berg and Terry Grande, thank you for your guidance and support over these long years. You never gave up on me, even when I felt like giving up on myself, and I will forever be grateful for that. Thank you for not only the knowledge you shared with me, but for helping me see this through to the end. When you both took me on as a student in the Fall of 2003, I doubt any of us thought it would take this long! Terry, thanks for letting me remain welcome your lab for so long; even when I wasn't always around I knew I belonged. I'll truly miss being an active member of the Grande Lab, although I hope I have a lifetime membership! Marty, your red pen

does wonders. Your prolific comments on the multiple drafts of my thesis were particularly helpful throughout this process...you made me make sense! I can't thank you both enough. To my committee members, Drs. John Dettmers and Jan Savitz, thank you for your helpful comments on my research throughout this process, and in particular during the beginning stages. I'd also like to thank Dr. Savitz for his encouragement and kind words over the past few years of seeing me around campus.

I would have been lost without the constant companionship and deviousness of Amanda Burdi, my partner in crime in the Grande Lab. Amanda, you made this whole process so much more fun, and I'm so glad we went through this experience together. In addition, former graduate student Jen Koshorek and undergraduate student Brian Schuetz were both incredibly helpful in the field during the early stages of my fieldwork. I probably never could have gotten the initial sampling effort underway without their help. I'd also like to thank the Biology Department at Loyola University for purchasing equipment to allow me to get underway. I'd like to give a special thank-you to James Schnulle and his guard dogs, of Woodstock, IL, who allowed us to trespass on his land to sample every month. Also, this study never could have been completed without the support and guidance from Brad Woodson and his colleagues at McHenry County Conservation District. Thank you for your help throughout this process and for allowing me access to Nippersink Creek in beautiful Glacial Park.

TABLE OF CONTENTS

ACKNOWLEDGEMENTS	iii
LIST OF FIGURES	vii
LIST OF TABLES	x
ABSTRACT	xi
INTRODUCTION	1
Channelization and Restoration	3
Research Goals.....	5
MATERIALS AND METHODS.....	6
Study Location	6
Site Selection	8
Data Collection	9
Data Analysis.....	12
Historical Data and Pre-Restoration Channel Data	18
RESULTS	22
Fish Assemblage Structure	22
Influence of Habitat Variables	37
Comparison to Historical Data.....	41
Comparison to Pre-Restoration Channel Data.....	46
DISCUSSION.....	52
Fish Community Structure and Integrity	52
Influence of Habitat Variables	55
Influence of Habitat on Common Species	59
Comparison to Historical Data.....	62
Effects of Channelization and Restoration	63
Future Directions	66

CONCLUSIONS.....	70
REFERENCES	72
VITA.....	78

LIST OF FIGURES

Figure 1. A) Map of Illinois, showing the location of McHenry County, IL; b) Map of Nippersink Creek, McHenry County, IL, in the study area; and c) Location of the pre-restoration channel in reference to the restored area of Nippersink Creek, McHenry County, IL. Site locations indicated as points. Abbreviations: U=upstream of the restored sites; R=within the restored area; D=downstream of the restored area. Map modified from Andrade 2006.....	7
Figure 2. Aerial photographs of the original meanders of Nippersink Creek through Glacial Park in 1939, and the channelized stream section in 1967. Reproduced from Shore 2001	8
Figure 3. Nine most abundant species collected at each 50 meter site in Nippersink Creek, McHenry County, IL (September 2004 – September 2005). Site abbreviations as in Table 1; missing sample dates as in Table 2	24
Figure 4. Mean number of fish sampled at each 50 meter site in Nippersink Creek, McHenry County, IL (September 2004 – September 2005). Site abbreviations as in Table 1; missing sample dates as in Table 2.....	25
Figure 5. Mean number of fish caught each month at all 50 meter sites combined in Nippersink Creek, McHenry County, IL (September 2004 – September 2005). Site abbreviations as in Table 1; missing sample dates as in Table 2.....	26
Figure 6. Species richness at each 50 meter site in Nippersink Creek, McHenry County, IL (September 2004 – September 2005). Site abbreviations as in Table 1; missing sample dates as in Table 2	27
Figure 7. Monthly species richness in Nippersink Creek, McHenry County, IL (September 2004 – September 2005). Site abbreviations as in Table 1; missing sample dates as in Table 2	28
Figure 8. Species richness at upstream, restored, and downstream sites in Nippersink Creek, McHenry County, IL (September 2004 – September 2005). Missing sample dates as in Table 2.....	30

Figure 9. Index of Biotic Integrity (IBI) scores at upstream, restored, and downstream sites in Nippersink Creek, McHenry County, IL (September 2004 – September 2005). Missing sample dates as in Table 2.....	31
Figure 10. Non-metric multidimensional scaling ordination plot of study site fish community structure similarity using square-root transformed fish abundance data. Site abbreviations as in Table 1; missing sample dates as in Table 2.....	32
Figure 11. Non-metric multidimensional scaling ordination plot of study site fish community structure similarity using presence/absence fish data. Site abbreviations as in Table 1; missing sample dates as in Table 2	34
Figure 12. Shannon Diversity at each site in Nippersink Creek, McHenry County, IL (September 2004 – September 2005). Site abbreviations as in Table 1; missing sample dates as in Table 2	35
Figure 13. Principal Components Analysis plot of number of species, number of individuals, number of families, and Shannon Diversity at each 50 meter site. Site abbreviations as in Table 1; missing sample dates as in Table 2.....	36
Figure 14. Non-metric multidimensional scaling ordination plot of study site similarity based on habitat data. Site abbreviations as in Table 1; missing sample dates as in Table 2	38
Figure 15. Non-metric multidimensional scaling ordination plot of study site similarity based on presence/absence fish data. Bubble sizes represent percent algal substrate cover at each study site. Site abbreviations as in Table 1; missing sample dates as in Table 2	42
Figure 16. Non-metric multidimensional scaling ordination plot of study site similarity based on presence/absence fish data. Bubble sizes represent percent gravel at each study site. Site abbreviations as in Table 1; missing sample dates as in Table 2	43
Figure 17. Non-metric multidimensional scaling ordination plot of study site similarity based on presence/absence fish data. Bubble sizes represent percent silt at each study site. Site abbreviations as in Table 1; missing sample dates as in Table 2	44

Figure 18. Index of Biotic Integrity scores calculated from samples collected in the current study (September 2004 – September 2005) and collected by the McHenry County Conservation District in past collections. Site abbreviations as in Table 1. Missing sample dates from current study as in Table 2 and sample dates from historical data as in Table 6 45

Figure 19. Non-metric multidimensional scaling ordination plot of study site similarity based on presence/absence historical fish data from McHenry County Conservation District. Site abbreviations as in Table 1; sample dates as in Table 6 47

Figure 20. Index of Biotic Integrity scores of pre-restoration channel sites on Nippersink Creek, McHenry County, Illinois. Scores calculated by the McHenry County Conservation District. Sample dates as in Table 7 48

Figure 21. Non-metric multidimensional scaling ordination plot of site similarity based on presence/absence fish data from McHenry County Conservation District pre-restoration channel samples. Sample dates as in Table 7 49

Figure 22. Non-metric multidimensional scaling ordination plot of site similarity based on presence/absence fish data from McHenry County Conservation District pre-restoration channel samples and July & August pooled data from the current study. Missing sample dates from current study as in Table 2 and sample dates from pre-restoration channel as in Table 7 51

LIST OF TABLES

Table 1. Study site numbers, codes, and locations, in upstream to downstream site order	9
Table 2. Study sites, dates, and reasons for missing fish and habitat samples in the current study of Nippersink Creek, McHenry County, IL (September 2004 – September 2005).	13
Table 3. Metrics used to assess biotic integrity of fish assemblages, based on the Index of Biotic Integrity. Modified after Karr 1981, Fausch et al. 1984, and Karr 1991	15
Table 4. Index of Biotic Integrity (IBI) score quality classes. Modified after IEPA 1989 and Karr 1981.	15
Table 5. Listing of habitat variable combinations used in BIO-ENV analysis.....	19
Table 6. Site codes and sample dates of data collections made in the current study and by the McHenry County Conservation District (MCCD) used for Index of Biotic Integrity and non-metric multidimensional scaling ordination plot comparisons	20
Table 7. Site names, sample dates, and location description of data collections made in the historical Nippersink Creek channel by McHenry County Conservation District (MCCD). Location descriptions courtesy of MCCD.....	21
Table 8. Fish species collected in Nippersink Creek, McHenry County, Illinois (September 2004 – September 2005). Common names as in Nelson et al. 2004	23
Table 9. BIO-ENV results and Spearman Correlation Coefficients relating fish assemblage structure to habitat variables. Presence/absence fish data were used to equally weight all taxa. The habitat variable combinations listed below are the best correlations to the fish assemblage structure according to the BIO-ENV procedure. The top three correlations are listed for each test run; significant values ($p < 0.05$) are bold.	40

ABSTRACT

Degradation of aquatic habitats and loss of biodiversity are growing concerns of natural resource managers and the general public. Channelization, the common historical practice of straightening streams and rivers for agricultural interests, has had profound detrimental effects on the biodiversity of lotic fish assemblages. Nippersink Creek, McHenry County, IL is a twenty-three mile stream that flows through an area valued for its fish, wildlife, and invertebrate biodiversity. Although a portion of the stream was channelized in the 1950's, restoration efforts by the McHenry County Conservation District have recently restored historical meanders. Nevertheless, efforts to restore streams and rivers to their natural conditions may also have unknown detrimental effects because the process of restoration is a disturbance to lotic fish assemblages. This project assessed and compared fish assemblage structure, habitat, and biotic integrity of historically channelized, restored, and natural sections of Nippersink Creek, utilizing data collected in the natural and restored areas of Nippersink Creek and data gathered by McHenry County Conservation District before restoration efforts began. Index of Biotic Integrity scores and species richness were low overall in comparison to historical data, but were as high or higher in the restored section of Nippersink Creek than in upstream natural and downstream natural areas, suggesting that the restoration effort was successful. An analysis of habitat variables found that percent silt, gravel, and algae substrate cover were most important in shaping the fish community, although a more

complete suite of habitat variables should be sampled in future studies to determine whether these variables are determinant. Findings from this study will contribute to a greater understanding of the effects of stream restoration on fish assemblages in Midwestern agricultural streams, and will be valuable in future stream restoration efforts within the Chicago area and throughout the United States.

INTRODUCTION

Streams and rivers face a multitude of anthropogenic disturbances, including direct channel modifications such as channelization and impoundments, as well as secondary effects from urbanization and agricultural land-use. These disturbances affect the quality of stream fish habitat by increasing erosion and siltation (Berkman and Rabeni 1987), straightening river bends (Scarnecchia 1988), and removing riparian vegetation, heterogeneous substrates, and instream woody debris (Hortle and Lake 1983, Scarnecchia 1988, Paller et al. 2000). Detrimental effects of habitat modification on stream fish communities result in a reduction of fish species diversity, family diversity, and abundance (Gorman and Karr 1978, Hortle and Lake 1983, Edwards et al. 1984, Raborn and Schramm 2003, Sullivan et al. 2004).

Stream habitat data are important when assessing fish community structure because the presence or absence of specific habitat characteristics influences fish species composition, abundance, and size/age structure (Gorman and Karr 1978, Schlosser 1982). Because the relationship between fish community structure and habitat quality is well-established, habitat and fish community data are essential in identifying, preventing, and reversing anthropogenic stream degradation, as well as measuring restoration success (Paller et al. 2000).

Although it is generally accepted that no one factor determines fish community composition (Gorman and Karr 1978, Schlosser 1982, Koehn et al. 1994), some habitat

factors that have been demonstrated to have a stronger influence on fish community composition than others. Koehn et al. (1994) found that stream depth and water velocity determined microhabitat use of fishes in a small Australian stream, whereas Gorman and Karr (1978) demonstrated the importance of depth, velocity, and substrate type to fishes in temperate disturbed and undisturbed streams, as well as a tropical undisturbed stream. Siltation, a function of both substrate type and bank erosion, has a strong influence on fish communities (Berkman and Rabeni 1987). Schlosser (1982) found substrate diversity and fish diversity to be positively correlated, whereas Feyrer and Healey (2003) found fish community structure to be most strongly influenced by water temperature and river discharge. Other habitat variables, such as riparian cover (Stauffer et al. 2000) and instream woody debris (Angermeier and Karr 1984, Wright and Flecker 2004) have been shown to be integral to a diverse fish community.

Specific habitat preference, i.e., fast or slow moving water, riffle or pool habitat, rocky or sandy substrates, of a fish species is generally consistent over its range (Gorman and Karr 1978), but each species may be considered microhabitat generalists or specialists (Wood and Bain 1995). Many factors, such as substrate or riparian zone composition, flow, predation, and/or competition, may interact to influence fish habitat selection (Koehn et al. 1994). However, stream fishes may need to use a variety of habitats to reproduce, forage, and avoid predators (Robertson and Winemiller 2003). Moyle and Baltz (1985) found that habitat and microhabitat use may vary within a species across sites due to habitat availability, temperature, food resource supply, and

competition, so subtle variations in habitat may strongly influence fish community structure.

Channelization and Restoration

Urban and agricultural demands on natural resources can lead to detrimental modifications of aquatic ecosystems. Channelization, the artificial straightening of streams and rivers, with resulting homogenization of instream habitat, is one such modification, and is one of the most destructive forms of anthropogenic stream disturbances (Detenbeck et al. 1992). Channelization results in the loss of woody debris and large substrate particles, which leaves fine, unstable substrates; a simplification of flow patterns; elimination of riparian canopy, which decreases allochthonous inputs to support aquatic food webs and increases stream water temperature; and ultimately, a reduction in fish habitat and shelter (Petersen et al. 1987, Gorman and Karr 1978). Most channelized stream channels lack well-defined riffle-pool-run sequences, so channelized sections of streams tend to be defined by higher current velocities, finer and less heterogeneous substrates, and an overall lack of suitable fish habitat (Hortle and Lake 1983, Scarnecchia 1988).

Channelized rivers generally lack snags or woody debris that serve as refugia from predation and high current velocities. Snags and woody debris also provide habitat for aquatic invertebrates, a primary food resource for many stream fish (Benke et al. 1985). Homogenization of stream channel substrate, often a result of channelization, also tends to decrease benthic invertebrate diversity (Fitzgerald et al. 1998). Angermeier and Karr (1984) found that fish and benthic invertebrates were more abundant in stream

reaches with woody debris than in areas where woody debris was lacking. Absence of overhanging vegetation or woody debris may account for lower fish abundance and species richness in channelized portions compared to unchannelized sections (Hortle and Lake 1983). Thus, it is not surprising that fish species diversity and density generally are lower in channelized reaches than in unchannelized portions of the same river (Hortle and Lake 1983, Edwards et al. 1984, Raborn and Schramm 2003).

The goal of many restoration programs in agricultural landscapes is to return streams and rivers to their pre-channelized state (with stable populations of native fishes) by adding meanders, increasing streambed heterogeneity, building artificial riffles and pools, and grading back steep banks. Nonetheless, fish species abundance in areas with artificial riffles typically is intermediate between that of natural and channelized areas (Edwards et al. 1984). Artificial riffles constructed to create pool habitat for game fishes tend to be minimally beneficial to nongame species (Fuselier and Edds 1995), and successful restoration of streams and the re-establishment of native fish assemblages must meet the habitat requirements of all native fish species (Trexler 1995).

The process of restoration, like channelization, is also a disturbance to stream fish assemblages, and its immediate effects on stream fauna are unknown (Muotka et al. 2002). Studies on recovery of fish communities from press disturbances such as restoration (habitat enhancement) and channelization indicate that from 5 to 52 years may be necessary for fish populations to completely recover from the disturbance (Detenbeck et al. 1992). Additional research documented recovery times of 3 to 8 years for a stream with close access to species sources such as lakes, river inputs, and deep pools (Lepori et

al. 2005). Furthermore, some studies have been incomplete in their efforts to determine the long-term consequences of channelization, and the effects of restoration efforts and timelines for full recovery of stream communities are often not thoroughly studied and/or documented (Scarnecchia 1988, Muotka et al. 2002, Detenbeck et al. 1992). The recovery process of restored streams must be monitored over time, as the endpoint for restoration success is often unclear (Detenbeck et al. 1992) and may be very site-specific.

Research Goals

The main objective of this research is to assess and compare sites within restored and unrestored areas of Nippersink Creek, McHenry County, IL. Habitat, fish assemblage structure, and biotic integrity of sites within the restored section of Nippersink Creek will be compared with historical data from the pre-restored channel and surrounding unrestored sites to determine how sites vary both within and across the different areas along Nippersink Creek. In addition, this research will determine the habitat variables most important in influencing fish communities in Nippersink Creek. Findings from this study will contribute to a greater understanding of the effects of channelization and stream restoration on fish assemblages in Nippersink Creek, and will be valuable in future stream restoration efforts both within the Chicago area and throughout the United States.

MATERIALS AND METHODS

Study Location

Nippersink Creek, in Glacial Park, McHenry County, Illinois, is the largest tributary of the Fox River, flowing from Wonder Lake in a northeasterly direction. The creek is 38.6 kilometers total length, with 30% (11.3 km) of the length occurring within Glacial Park, a protected area owned by the McHenry County Conservation District (MCCD) (Figure 1). Nippersink Creek is a fourth-order stream within the study area. Stream order is used by aquatic biologists to classify the size of streams. Headwater streams are first-order; two first-order streams meet to form a second-order stream, two second-order streams meet to form a third-order stream, and so on. For reference, the lower Mississippi River is a twelfth-order stream. In the early 1950s, a 5,000-meter reach of the naturally meandering creek was diverted into a linear 3,230-meter artificial channel, the original channel was filled with sand and gravel from nearby glacial kames, and adjacent wetlands were drained to serve local agricultural interests (Woodson, 2000). Fish surveys were performed periodically by the MCCD and historical data were compiled to document channelization effects on the stream, by comparing channelized and natural, unmanipulated stream reaches. In 1999, restoration efforts on Nippersink Creek by MCCD began by the digging of selected stream cross-sections and grading back of stream banks, using aerial photographs as guides (Figure 2). The artificial channel was successfully diverted into 4700 meters of newly-meandering stream (Woodson, 2000)

Figure 1. A) Map of Illinois, showing the location of McHenry County, IL; B) Map of Nippersink Creek, McHenry County, IL, in the study area; and C) Location of the pre-restoration channel in reference to the restored area of Nippersink Creek, McHenry County, IL. Site locations indicated as points. Abbreviations: U=upstream of the restored sites; R=within the restored area; D=downstream of the restored area. Map modified from Andrade 2006.

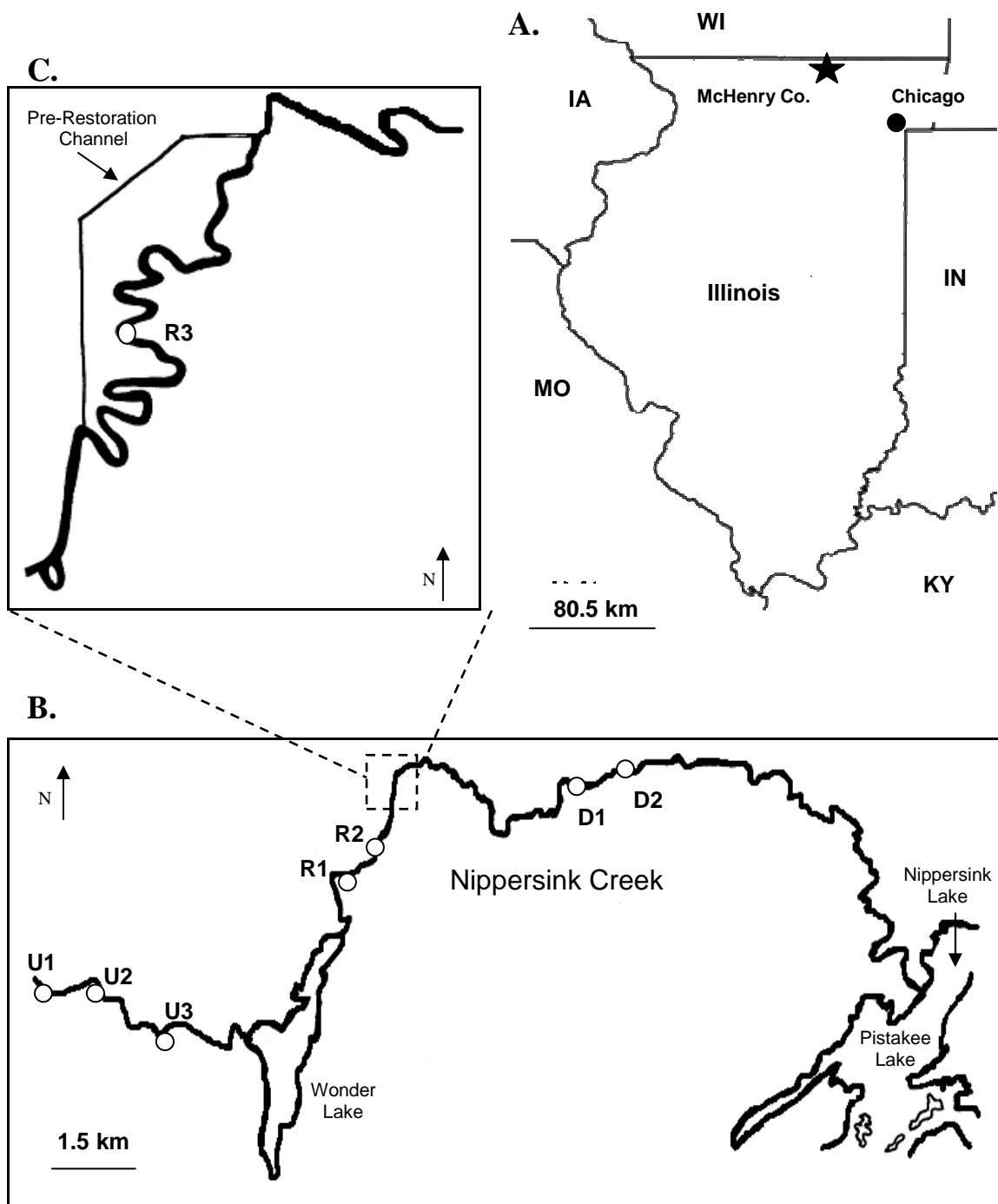
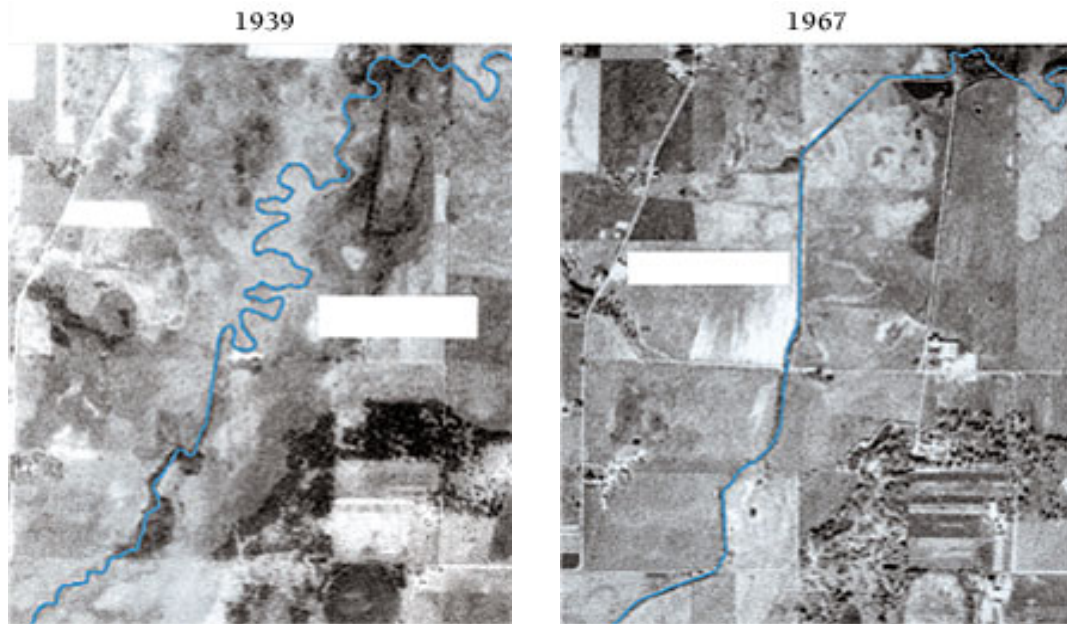


Figure 2. Aerial photographs of the original meanders of Nippersink Creek through Glacial Park in 1939, and the channelized stream section in 1967. Reproduced from Shore 2001.



and the MCCD continues to restore habitat for fish, wildlife, and invertebrate communities in Glacial Park.

Site Selection

Fish assemblages in Nippersink Creek were sampled monthly at eight 50 meter reaches over a thirteen month period from September 2004 to September 2005. Three study sites (U1, U2, and U3) were located in the natural area upstream from Wonder Lake, three sites (R1, R2, and R3) were located in the restored section of stream within Glacial Park, and two sites (D1 and D2) were located in natural areas downstream of Glacial Park and upstream of a chain of lakes including Pistakee, Nippersink, and Fox Lakes (Figure 1). Sites were numbered sequentially from upstream to downstream, and given codes to denote location type (Table 1). Sites were chosen based on habitat

Table 1. Study site numbers, codes, and locations, in upstream to downstream site order.

Site Number	Location Type	Site Code
1	Upstream Natural	U1
2	Upstream Natural	U2
3	Upstream Natural	U3
4	Restored	R1
5	Restored	R2
6	Restored	R3
7	Downstream Natural	D1
8	Downstream Natural	D2

diversity, i.e., all sites contained riffle, run, and to a lesser extent, pool habitat. Sites U1 and U2 were located in a mainly agricultural landscape, with a meadow/grassland riparian zone, and were narrow in width in comparison to all other sites. Site U3 was located on private land not used for agriculture, and was characterized by a mixed woodland/shrub riparian zone on one side and a meadow/grassland on the other, with a predominance of rubble/cobble substrate not seen at any other site. All three upstream natural sites begin and end upstream of a road crossing. The three restored sites were located within Glacial Park, with meadow/grassland riparian zones. Both D1 and D2 were located in wooded areas. Although the downstream natural sites were accessed via road bridges, the sites terminated over 100 meters downstream from the nearest road crossing.

Data Collection

Air and water temperature were recorded at each site with either a Fisher Scientific Accumet Dissolved Oxygen/°C/°F Data Meter or a mercury thermometer. A qualitative estimate of discharge (below normal, normal, or above normal) was made by

comparing the present water level to stream bank indicators (i.e., evidence of high water as shown by crushed vegetation, evidence of low water as shown by exposed bank). If areas of the stream bank were dry but looked as though they would normally be underwater, then a rating of below normal was assigned; if the stream was flowing over areas of terrestrial grasses or vegetation, a rating of above normal was assigned (Simonson et al. 1993).

Fish were sampled once a month using a 3-meter seine with 1.27 centimeter mesh. One seining pass of the entire 50 meter site was made and every effort was made to span the entire width of the site with the seine. Shallow riffle areas were kick-seined to collect fishes located within rubble/cobble substrates. Fish trapped in the seine were placed in holding buckets until the entire site was sampled. Fish less than 20 mm total length were not included in the study because they were too small to be effectively captured by the seine. All fish collected were identified to species and counted. Individual weights (g) and lengths (mm) of all adult game fish species were recorded, and aggregate weights were taken of all other species. Fish were given fresh water in the holding buckets if they were held longer than 15 minutes, and were promptly returned to the stream once the entire 50-meter site had been sampled and the data recorded, in accordance with MCCD permit regulations. If species identification could not be made in the field, a representative sample of 1-3 fish was placed in a screw top jar, labeled with the date and site number, anaesthetized in a solution of MS-222 (tricaine methanesulfonate, 200 g/L) and sodium bicarbonate buffer (500 g/L), preserved in a 10% formalin solution, and returned to the laboratory for further identification (Barbour et al.

1999; IDNR Permit Nos. A04.2086, A05.2086; Loyola University ACTS No. 78). Keys to fish species found in Smith (1979), Becker (1983), and Pflieger (1997) were used in fish identification. Preserved fishes were deposited in the Loyola University Chicago, Department of Biology, Fish Collection.

After fish were sampled and returned to the stream, stream habitat variables were measured. Habitat variables were visually assessed according to a modified version of the Wisconsin method (Simonson et al. 1993) and were expressed as percentages of the entire study site. Visual estimates are used as approximations for many variables when direct measurements are too time-intensive to make, and give precise results if performed by an experienced observer (Simonson et al. 1993). Habitat type was recorded as percentages of riffle, pool, and run that comprised the entire site. Substrate composition was assessed as the percentage of each component (bedrock, boulder, rubble/cobble, gravel, sand, silt, clay, detritus) that comprised the stream bed. Vegetation characteristics were recorded as the percentage of the site that contained either submerged, overhanging, or emergent macrophytes and the percentage of the substrate that was covered in algae. Riparian land-use was defined as the land from the edge of the water to a point 5 meters inland (Simonson et al. 1993), and was recorded as a percentage of the entire length of the site. Riparian land-use choices included cropland, pasture, barnyard, developed, meadow, shrubs, woodland, wetland, and exposed rock. All parameters were visually assessed by the primary investigator, and substrate composition and in-stream vegetation characteristic assessments were supplemented by feeling the substrate with hands and feet.

Due to inclement weather and stream conditions, certain sites were not sampled at particular times during the thirteen month study (Table 2), resulting in 90 total collections rather than 104. In the upstream natural section of Nippersink Creek Nippersink Creek, U2 was not sampled in either January or February 2005 due to heavy ice cover. In the restored section, R1 and R2 were not sampled in January 2005 because heavy snowfall created high snow drifts that prevented stream access. R1 was also not sampled in June and July 2005 due to low water and high amounts of macrophytes that made sampling impossible. In addition, R1 was not sampled in September 2005 due to heavy thunderstorms that prevented sampling for the remainder of the weekend and significantly altered stream flows during the rest of the month. R2 was not sampled in June 2005 because low water and heavy macrophytes prevented seining. Due to structural changes that occurred in Nippersink Creek in early summer 2005, site R2 was not sampled in July, August, or September 2005. The creation of a new riffle downstream caused stream flow to back-up in such a way that site R2 was drastically altered and became too deep to safely sample. In the downstream natural section of Nippersink Creek, D1 was not sampled in September 2005 due to thunderstorms that halted sampling and altered stream flow conditions later in the month. D2 was not sampled in January 2005 due to high snow drifts that blocked access to the stream, and also was not sampled in June 2005 due to extremely low flows that prevented seining.

Data Analysis

The Index of Biotic Integrity (IBI) was used to assess stream degradation and make comparisons between restored and natural stream sites. The IBI incorporates

Table 2. Study sites, dates, and reasons for missing fish and habitat samples in the current study of Nippersink Creek, McHenry County, IL (September 2004 – September 2005).

Site	Month	Year	Reason for Missing Sample
U2	January	2005	Ice Cover
U2	February	2005	Ice Cover
R1	January	2005	Snow Drifts
R1	June	2005	Low Water & Dense Macrophytes
R1	July	2005	Low Water & Dense Macrophytes
R1	September	2005	Thunderstorms
R2	January	2005	Snow Drifts
R2	June	2005	Low Water & Dense Macrophytes
R2	July	2005	High Water
R2	August	2005	High Water
R2	September	2005	High Water
D1	September	2005	Thunderstorms & High Water
D2	January	2005	Snow Drifts
D2	June	2005	Low Water

multiple attributes of stream fish assemblages to evaluate various anthropogenic effects (e.g. channelization, restoration, etc.) on streams and watersheds (Karr 1991). The IBI was developed to function as a relatively simplistic, easily communicable method to quantify and rate the biotic communities of Midwestern streams, and is based upon the principle that biological communities reflect environmental conditions (Karr 1981). Karr (1986) describes biological integrity as analogous to human health. Good health of a stream system is indicative of realized potential, stable condition, inherent resilience, and a minimum need of management (Karr 1986). There are advantages to using fish communities to assess stream integrity, or stream health: fish are easily identified and represent a variety of trophic levels, they live in all but the most degraded aquatic habitats and their environment is well-understood, and information about the condition of a fish community is easily communicated from natural resource managers to the general public

(Fausch et al. 1984). In addition, assessments in Illinois streams have found the IBI to identify known stream degradation more reliably than diversity measures such as the Shannon-Wiener diversity index (Angermeier and Schlosser 1987).

The IBI takes into account twelve key aspects (metrics) of fish assemblages, which are divided into three categories: Species Composition, Trophic Composition, and Fish Abundance and Condition (Table 3). Certain metrics, i.e. number of darter (benthic) species, number of sunfish (water column) species, are designed to test assemblage complexity by identifying fish species that occupy specific habitats within stream ecosystems (Karr 1981). A number rating (1, 3, or 5) is assigned to each metric based on how similar each metric is to the expected value for a pristine environment, with a rating of 5 assigned to those metrics closest to the expected value. The sum of the twelve metric scores is the total IBI score, which corresponds to a Biological Stream Characterization Category and a Biotic Integrity Class (Table 4).

IBI metrics have been modified slightly over time and for different geographic regions by natural resource managers, and small variations may be found in the range of scores that fall into a particular integrity class. As such, IBI scores for the current study were calculated using the same IBI version as MCCD, IEPA (1989), so that any differences noted would not be due to inconsistencies in the calculation method. Index of Biotic Integrity (IBI) values were calculated for each sampling event and integrity comparisons were made across sites, and over time using IBI scores from MCCD historical data collections from the existing stream as well as the pre-restoration channel. Average annual scores were used to make site comparisons in the current study. Means

Table 3. Metrics used to assess biotic integrity of fish assemblages, based on the Index of Biotic Integrity. Modified after Karr 1981, Fausch et al. 1984, and Karr 1991.

Metric	Scoring Criteria			
	<u>5</u>	<u>3</u>	<u>1</u>	
<u>Species Richness and Composition</u>				
1	Total number of native fish species	Metrics 1-5 vary with stream size		
2	Number of darter/benthic species			
3	Number of sunfish/water column species			
4	Number of sucker species			
5	Number of intolerant species			
6	Percentage of total as green sunfish	<5	5-20	>20
<u>Trophic Composition</u>				
7	Percentage of omnivores	<20	20-45	>45
8	Percentage of insectivorous cyprinids	>45	20-45	<20
9	Percentage of top carnivores	>5	1-5	<1
<u>Fish Abundance and Condition</u>				
10	Number of individuals in a sample	Metric 10 varies with stream size		
11	Percentage of hybrids	0	>0-1	>1
12	Percentage with physical anomalies	0-2	>2-5	>5

Table 4. Index of Biotic Integrity (IBI) score quality classes. Modified after IEPA 1989 and Karr 1981.

IBI Score Range	Biological Stream Characterization Class	Biotic Integrity Class
51-60	Unique Aquatic Resource	Excellent
41-50	Highly Valued Aquatic Resource	Good
31-40	Moderate Aquatic Resource	Fair
21-30	Limited Aquatic Resource	Poor
≤20	Restricted Aquatic Resource	Very Poor

of each site type (upstream, restored, and downstream) were generated for each month, and the mean of the monthly means was used to compare site types over the course of the study. In addition, IBI scores from the current study were used as a habitat factor in further statistical analyses. IBI scores were used to help assess ecological changes in Nippersink Creek since channelization and subsequent restoration.

Data were statistically analyzed with non-metric multidimensional scaling (nMDS) using the Primer 5 statistical package (Primer v5, Primer-E Ltd.). Non-metric multidimensional scaling is a multivariate ordination technique that produces a two-dimensional plot to show the relationship among samples. The nMDS procedure creates an among-sample similarity matrix of the data which, when plotted, may be interpreted in terms of relative similarity of samples to each other (Clarke and Warwick 2001). Non-metric multidimensional scaling produces a plot that is easy to interpret; points close together are more similar than those found farther apart (Clarke and Warwick 2001). All nMDS ordination was performed using square-root transformed abundance or presence/absence fish data, and the similarity matrix used to create the fish assemblage nMDS plots was generated using Bray-Curtis similarities. Habitat data were normalized to account for differences in scale, and the similarity matrix used to create the habitat composition nMDS plots was generated using normalized Euclidian distance and no data transformations. Bray-Curtis (dis)similarity is more appropriate for generating similarity matrices of biotic data, whereas Euclidian distance is preferred for abiotic and environmental similarity matrices (Clarke and Gorley 2001). Stress values are given for each plot as an indication of how strongly the relationships are represented by the plot.

Stress values ≤ 0.05 portray an excellent representation with no prospect of misinterpretation, values between 0.05 and ≤ 0.1 portray a good ordination with no real prospect of misinterpretation; values between 0.1 and ≤ 0.2 give a potentially useful representation, and stress values > 0.2 indicate random point placement and a representation that should be treated as arbitrary (Clarke and Warwick 2001).

The BIO-ENV procedure in the Primer 5 statistical package was used to match biotic to environmental patterns (Clarke and Gorley 2001, Clarke and Warwick 2001). This procedure was used to compare fish assemblage data to habitat composition data and analyze the extent to which the fish assemblage found at a given site is explained by its habitat. Prior to performing the BIO-ENV, draftsman plots (i.e., all possible pairwise scatter plots) of all habitat variables were generated to discern whether any variables were correlated. It is important to remove highly correlated variables (i.e., those with a correlation ≥ 0.95) before running the BIO-ENV procedure because including both variables in a highly correlated pair serves no useful purpose and may obscure relevant results. Highly correlated variable pairs were examined, and one variable from each highly correlated pair was removed from subsequent BIO-ENV analyses (Clarke and Ainsworth 1993, Clarke and Warwick 2001). The presence/absence transformed similarity matrix that was used to generate fish assemblage nMDS plots was compared to the remaining combinations of habitat variables. BIO-ENV generates a matrix of the best combinations of variables by creating increasingly complex groupings of variables. Spearman rank correlation was then used to represent the extent to which fish assemblage structure may be explained by habitat variables (Clarke and Warwick 2001). Community

structure was analyzed by performing the BIO-ENV procedure three times, using the full set of habitat variables and subsets of instream and riparian variables (Table 5). This is, however, a purely exploratory tool rather than a demonstration of causality, and no statistical significance is implied (Clarke and Gorley 2001).

Principal Components Analysis (PCA) was performed to examine what characteristics (Shannon diversity, number of families, number of species, number of fish) had the most influence on fish communities at each site. PCA is an ordination technique that attempts to examine similarities in community structure by ordering sample data in $n-1$ dimensional space, where n is equal to the number of variables input into the analysis (Clarke and Warwick 2001). Data were normalized to account for differences in scale. Results generated by PCA analysis were used to plot each sample point along a given axis. The percentage of variance that can be explained by a variable will increase until the total of the variances along all PC axes is equal to the total variance of the sample points (Clarke and Warwick 2001).

Historical Data and Pre-Restoration Channel Data

Comparisons of historical and current fish data can provide a way to detect changes in fish assemblages and declines in species abundance and distribution, and in particular, help investigators discover changes in habitat over time (Johnston and Maceina 2009). In their study of current and historical fish assemblages in Alabama streams, Johnston and Maceina (2009) found that fish assemblage changes were detected through historical comparisons that would have been missed by an index such as the IBI.

Table 5. Listing of habitat variable combinations used in BIO-ENV analysis.

Habitat Combination	Included Variables
All Habitat Variables, Excluding Run and Meadow/Grassland	IBI Score, Riffle (%), Pool (%), Boulder (%), Rubble/Cobble (%), Gravel (%), Sand (%), Silt (%), Clay (%), Detritus (%), Woody Debris (%), Submerged Vegetation (%), Emergent Vegetation (%), Overhanging Vegetation (%), Overhanging Trees (%), Algae Substrate Cover (%), Shrubs (%), Woodland (%)
Instream Variables Only	Riffle (%), Pool (%), Run (%), Boulder (%), Rubble/Cobble (%), Gravel (%), Sand (%), Silt (%), Clay (%), Detritus (%), Woody debris (%), Submerged Vegetation (%), Emergent Vegetation (%), Algae Substrate Cover (%),
Riparian Variables Only	Overhanging Vegetation (%), Overhanging Trees (%), Meadow/Grassland (%), Shrubs (%), Woodland (%)

Two types of fish assemblage data and IBI scores were provided by the McHenry County Conservation District (MCCD) for comparison with the current study: historical data from the current stream channel (referred to as historical data) and historical data from the pre-restoration agricultural channel (referred to as pre-restoration channel data). IBI scores from MCCD historical data were compared to single-event IBI scores that occurred in the same calendar month as the MCCD collection, and every attempt was made to match sites from historical data to those from the current study in both place and time (Table 6). All MCCD fish data were presence/absence transformed to make direct comparisons with data from the current study. An attempt was made to only compare data collected in the same month, with one exception (Table 6). Historical data from site R3 were only available from the month of August. Site R3 was not sampled in August in the present study, so historical data were compared to the sample collected in September 2004, which provided the closest seasonal match. The decision to compare August data to September data was supported by Schlosser (1982, 1985), who

Table 6. Site codes and sample dates of data collections made in the current study and by the McHenry County Conservation District (MCCD) used for Index of Biotic Integrity and non-metric multidimensional scaling ordination plot comparisons.

Site Code	Current Study	MCCD Data
U1	August 2005	August 1994
U2	October 2004	October 1993
U3	August 2005	August 1992
R1	September 2004	September 1996
R2	August 2005	August 2001
R3	September 2004*	August 2002
D1	September 2004	September 1991
D2	August 2005	August 2002

*September data from the current study must be compared to August MCCD data for this site because site R3 was not sampled in August 2005.

categorized August and September as “late summer” in his studies on an Illinois warmwater stream.

In addition to historical data from the current study sites, fish data from four sites within the unrestored channel, collected prior to the start of the restoration project, were also made available for comparison (Table 7). Pre-restoration channel data were collected in July and August, so current data from those months were pooled to examine unrestored channel data without any seasonal bias. All historical data were analyzed for patterns & similarities to data from the current study using nMDS plots.

Table 7. Site names, sample dates, and location description of data collections made in the historical Nippersink Creek channel by McHenry County Conservation District (MCCD). Location descriptions courtesy of MCCD.

Site Code	Sample Date	Location Description
C1	7/20/1988	Ditched channel 3/5 kilometer north of Valley Road
C2	8/30/1993	Channelized section north of old bridge, south of Valley Road
C3	8/19/1998	Channelized section at Trail-of-History site and site south of Valley Road
C4	8/19/1998	Channelized section north of Valley Road and south of bridge

RESULTS

Fish Assemblage Structure

A total of 4,450 fish, representing 36 species and 10 families, were collected, identified, and counted during the study period (Table 8). The two most commonly collected species, *Cyprinella spiloptera* and *Notropis stramineus*, were each represented by > 1000 individuals. The nine most abundant species comprised 90.5% of all fish collected (Figure 3). The most abundant species represented four families: Cyprinidae (*Campostoma anomalum*, *Cyprinella spiloptera*, *Notropis stramineus*, *Pimephales notatus*, *Semotilus atromaculatus*), Catostomidae (*Catostomus commersonii*), Centrarchidae (*Lepomis macrochirus*), and Percidae (*Etheostoma nigrum*, *E. zonale*). Although these nine species were numerically dominant, the focus of this study was to examine fish assemblage structure; therefore, all fish were included in statistical analyses.

The greatest mean number of fish per site (79.8) was found at R3, and the fewest number (23.4) was captured at U3 (Figure 4; Standard Error 7.24). Mean fish per month was low during December 2004 and January – March 2005, whereas the mean was high during September 2004, and June, July, and September 2005 (Figure 5; Standard Error 11.39). Fish species richness was greatest (25) at R3, and lowest (14) at U3 (Figure 6). Species richness was lowest in March 2005, and greatest in September 2004 (Figure 7). Collectively, restored sites showed species richness values equal to or greater than

Table 8. Fish species collected in Nippersink Creek, McHenry County, Illinois (September 2004 – September 2005). Common names as in Nelson et al. 2004.

<u>Family</u>	<u>Latin Name</u>	<u>Common Name</u>
Cyprinidae	<i>Campostoma anomalum</i>	Central stoneroller
	<i>Cyprinella spiloptera</i>	Spotfin shiner
	<i>Cyprinus carpio carpio</i>	Common carp
	<i>Hybognathus hankinsoni</i>	Brassy minnow
	<i>Notropis atherinoides</i>	Emerald shiner
	<i>Notropis stramineus</i>	Sand shiner
	<i>Phenacobius mirabilis</i>	Suckermouth minnow
	<i>Pimephales notatus</i>	Bluntnose minnow
	<i>Pimephales promelas</i>	Fathead minnow
	<i>Pimephales vigilax</i>	Bullhead minnow
	<i>Rhinichthys cataractae</i>	Longnose dace
	<i>Semotilus atromaculatus</i>	Creek chub
	Catostomidae	<i>Carpiodes cyprinus</i>
<i>Catostomus commersonii</i>		White sucker
Ictaluridae	<i>Ameiurus melas</i>	Black bullhead
	<i>Ameiurus natalis</i>	Yellow bullhead
	<i>Ictalurus punctatus</i>	Channel catfish
	<i>Noturus flavus</i>	Stonecat
	<i>Noturus gyrinus</i>	Tadpole madtom
Umbridae	<i>Umbra limi</i>	Central mudminnow
Fundulidae	<i>Fundulus notatus</i>	Blackstripe topminnow
Atherinopsidae	<i>Labidesthes sicculus</i>	Brook silverside
Gasterosteidae	<i>Culaea inconstans</i>	Brook stickleback
Moronidae	<i>Morone mississippiensis</i>	Yellow bass
Centrarchidae	<i>Lepomis cyanellus</i>	Green sunfish
	<i>Lepomis gibbosus</i>	Pumpkinseed
	<i>Lepomis humilis</i>	Orangespotted sunfish
	<i>Lepomis macrochirus</i>	Bluegill
	<i>Lepomis microlophus</i>	Redear sunfish
	<i>Micropterus dolomieu</i>	Smallmouth bass
	<i>Micropterus salmoides</i>	Largemouth bass
	<i>Pomoxis nigromaculatus</i>	Black crappie
Percidae	<i>Etheostoma flabellare</i>	Fantail darter
	<i>Etheostoma nigrum</i>	Johnny darter
	<i>Etheostoma zonale</i>	Banded darter
	<i>Perca flavescens</i>	Yellow perch

Figure 3. Nine most abundant species collected at each 50 meter site in Nippersink Creek, McHenry County, IL (September 2004 – September 2005). Site abbreviations as in Table 1; missing sample dates as in Table 2.

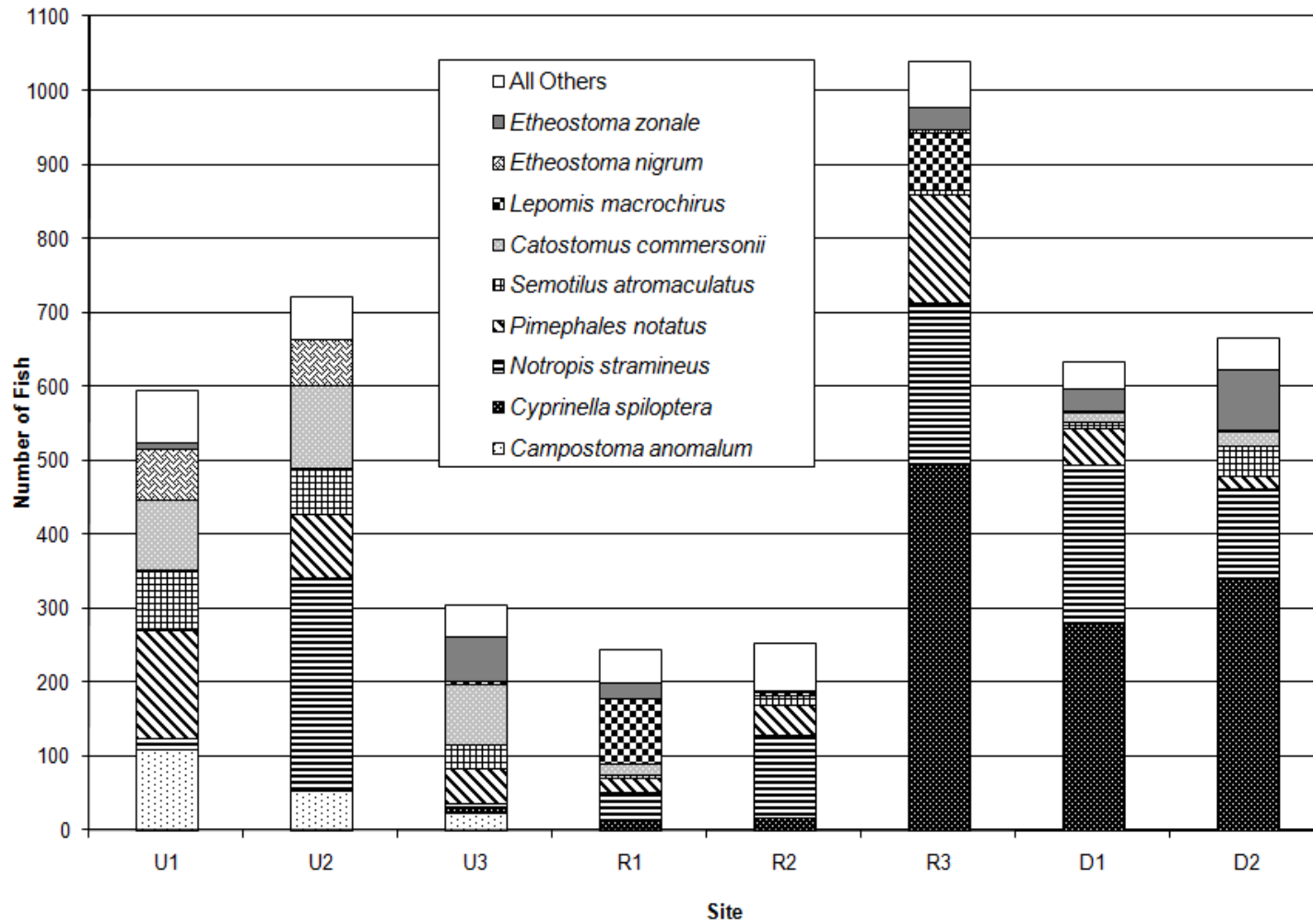


Figure 4. Mean number of fish sampled at each 50 meter site in Nippersink Creek, McHenry County, IL (September 2004 – September 2005). Site abbreviations as in Table 1; missing sample dates as in Table 2.

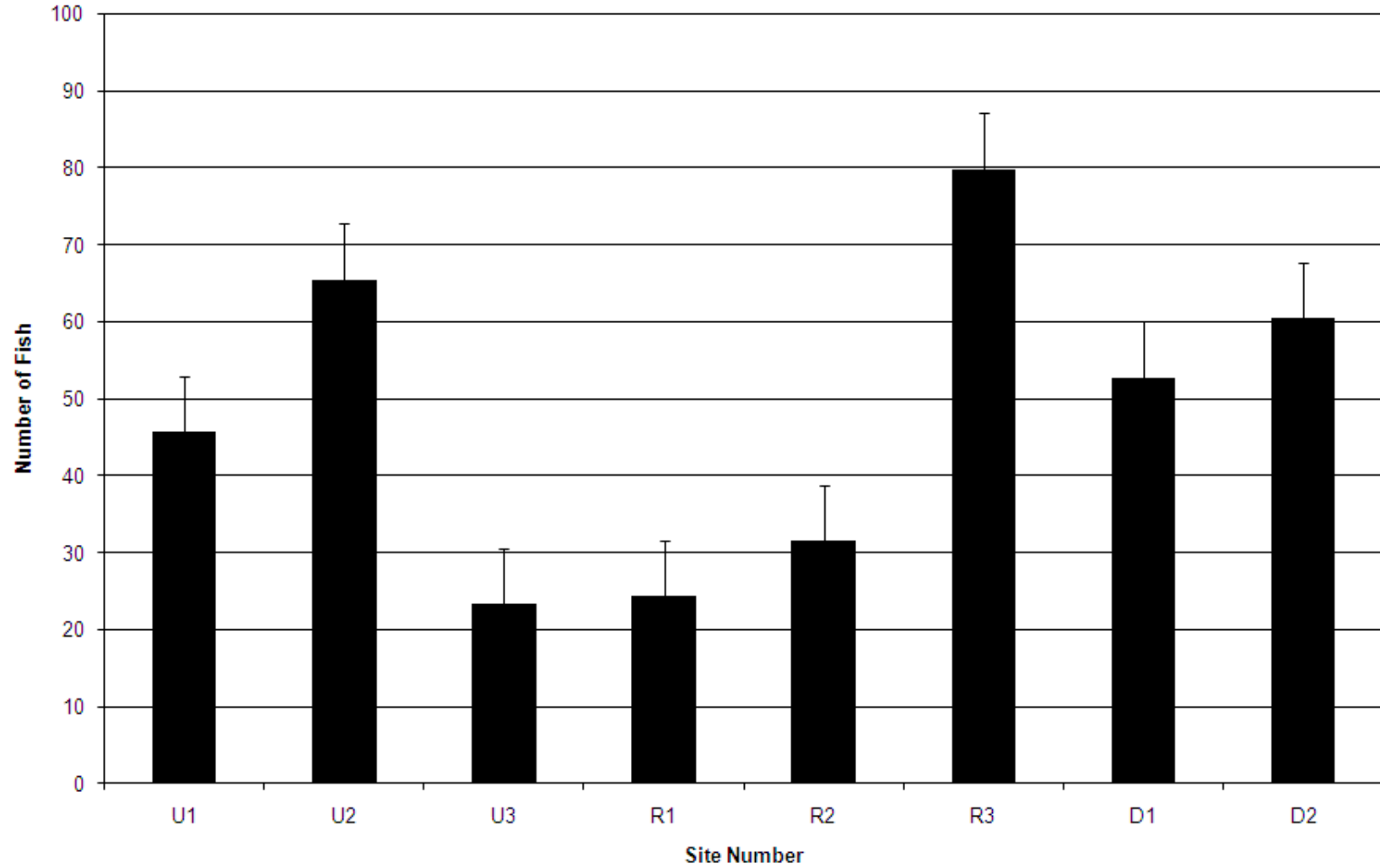


Figure 5. Mean number of fish caught each month at all 50 meter sites combined in Nippersink Creek, McHenry County, IL (September 2004 – September 2005). Site abbreviations as in Table 1; missing sample dates as in Table 2.

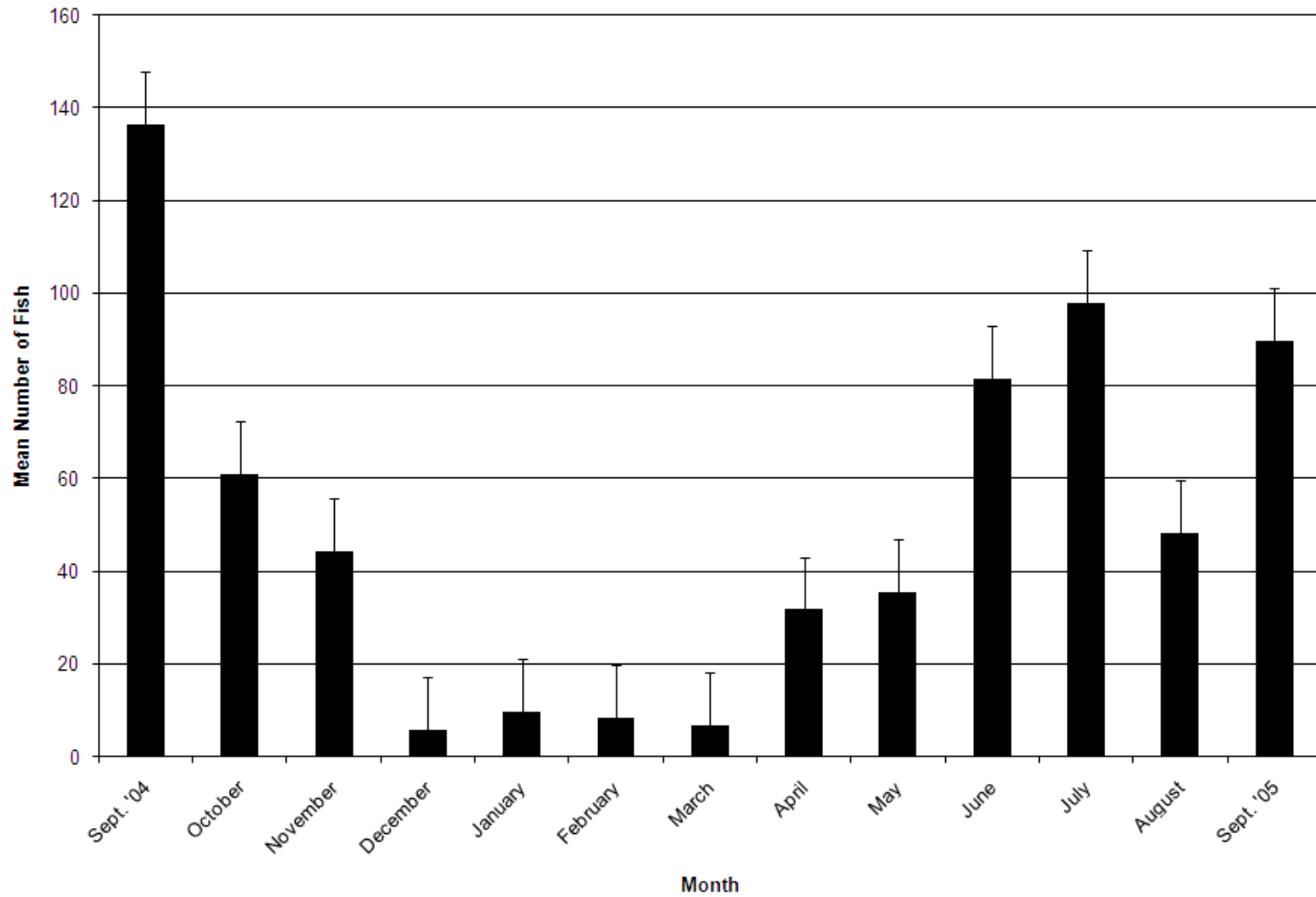


Figure 6. Species richness at each 50 meter site in Nippersink Creek, McHenry County, IL (September 2004 – September 2005). Site abbreviations as in Table 1; missing sample dates as in Table 2.

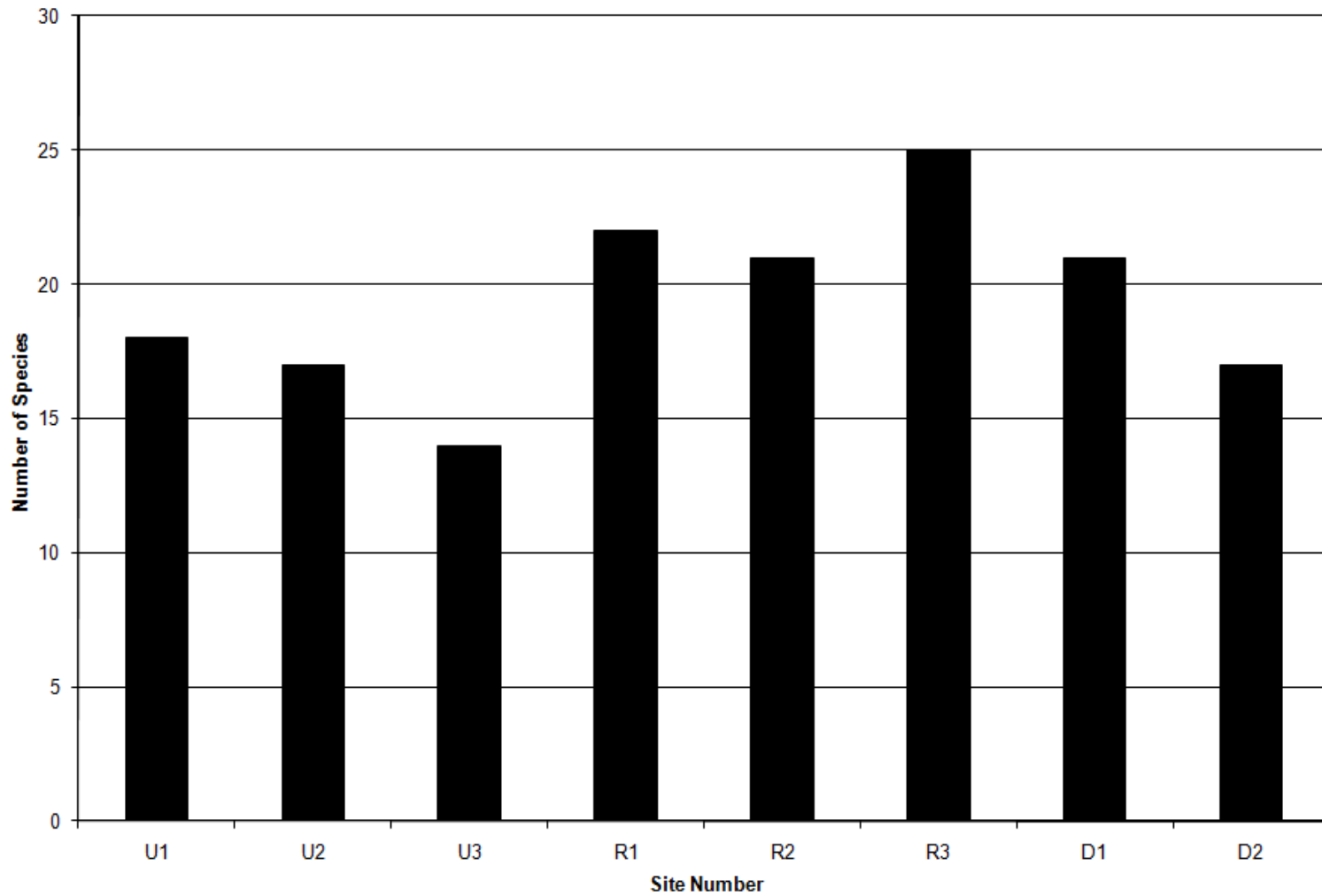
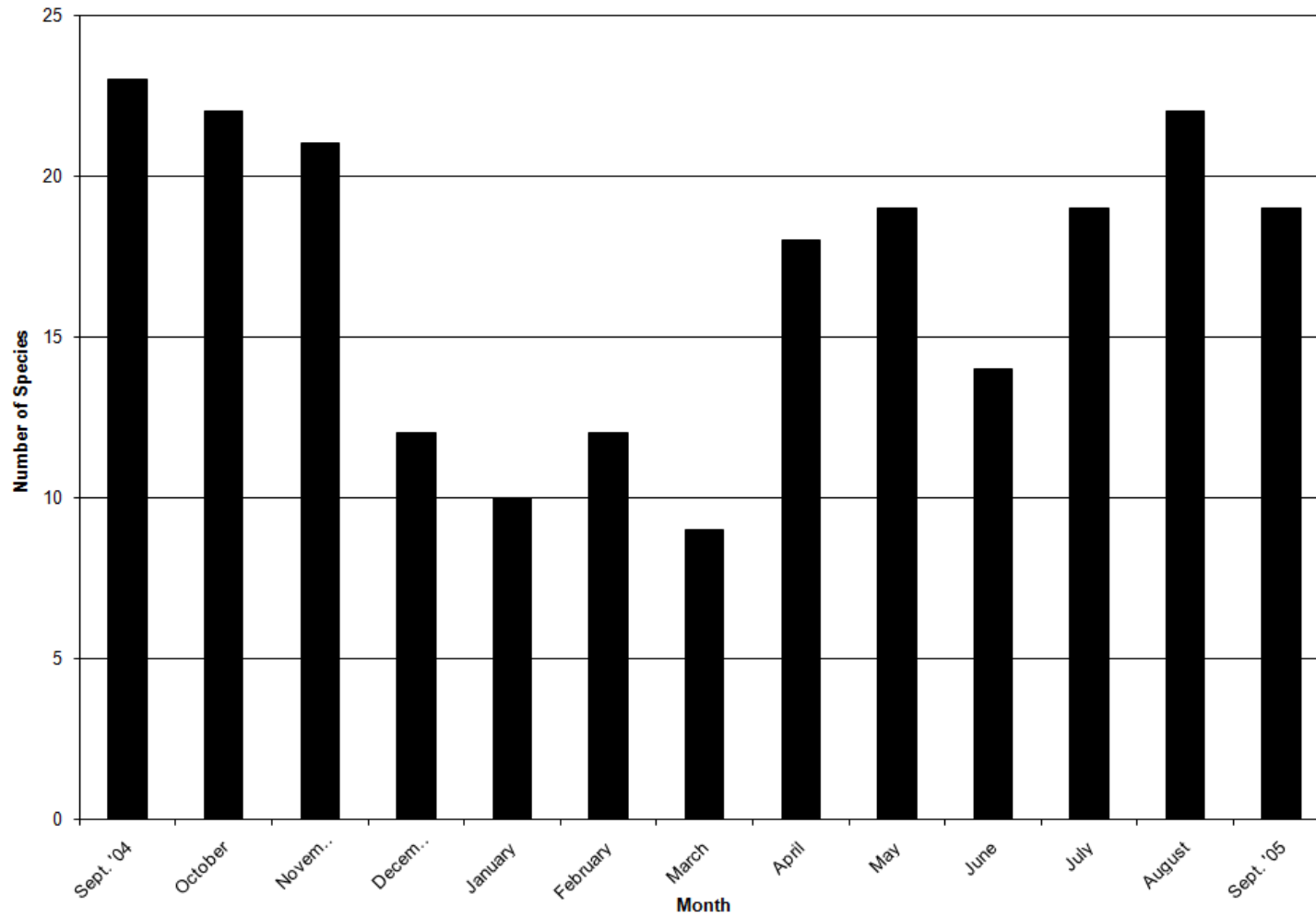


Figure 7. Monthly species richness in Nippersink Creek, McHenry County, IL (September 2004 – September 2005). Site abbreviations as in Table 1; missing sample dates as in Table 2.



upstream natural sites in 9 of the 13 months studied and equal to or greater than downstream natural sites in 12 of the 13 months studied (Figure 8). Index of Biotic Integrity (IBI) scores at the restored sites combined were as high or higher than upstream natural sites combined in 11 of the 13 months studied and were as high or higher than downstream natural sites combined in six of the 13 months studied (Figure 9).

The relationship among sites with respect to similarity of fish assemblages (abundance-based) revealed a clear separation of sites into two groups. This ordination had little prospect of misinterpretation (stress value = 0.07, Figure 10). The three upstream sites (U1, U2, and U3) were similar to each other in terms of fish communities and are characterized by higher numbers of *Campostoma anomalum*, *Rhinichthys cataractae*, and *Catostomus commersonii* than at other sites. Restored sites (R1, R2, and R3) and downstream sites (D1 and D2) form an ordination with a different fish community structure than upstream sites. *Cyprinella spiloptera* was one of the most common species found at these five sites. Collections of this species ranged from 14 individuals at R1 to 492 individuals at R3. *C. spiloptera* was rarely found in upstream sites, with 8 individuals found at U3 and no individuals at U1 and U2. The three most common species found in upstream sites, *C. anomalum*, *C. commersonii*, and *R. cataractae*, were absent or reduced in number at the restored and downstream sites.

Consistent with the ordination plot of abundance data, the relationship among sites with respect to the similarity of fish assemblages (presence/absence-based) revealed that fish communities at upstream natural sites were similar and that the ordination plot

Figure 8. Species richness at Upstream, Restored, and Downstream sites in Nippersink Creek, McHenry County, IL (September 2004 – September 2005). Missing sample dates as in Table 2.

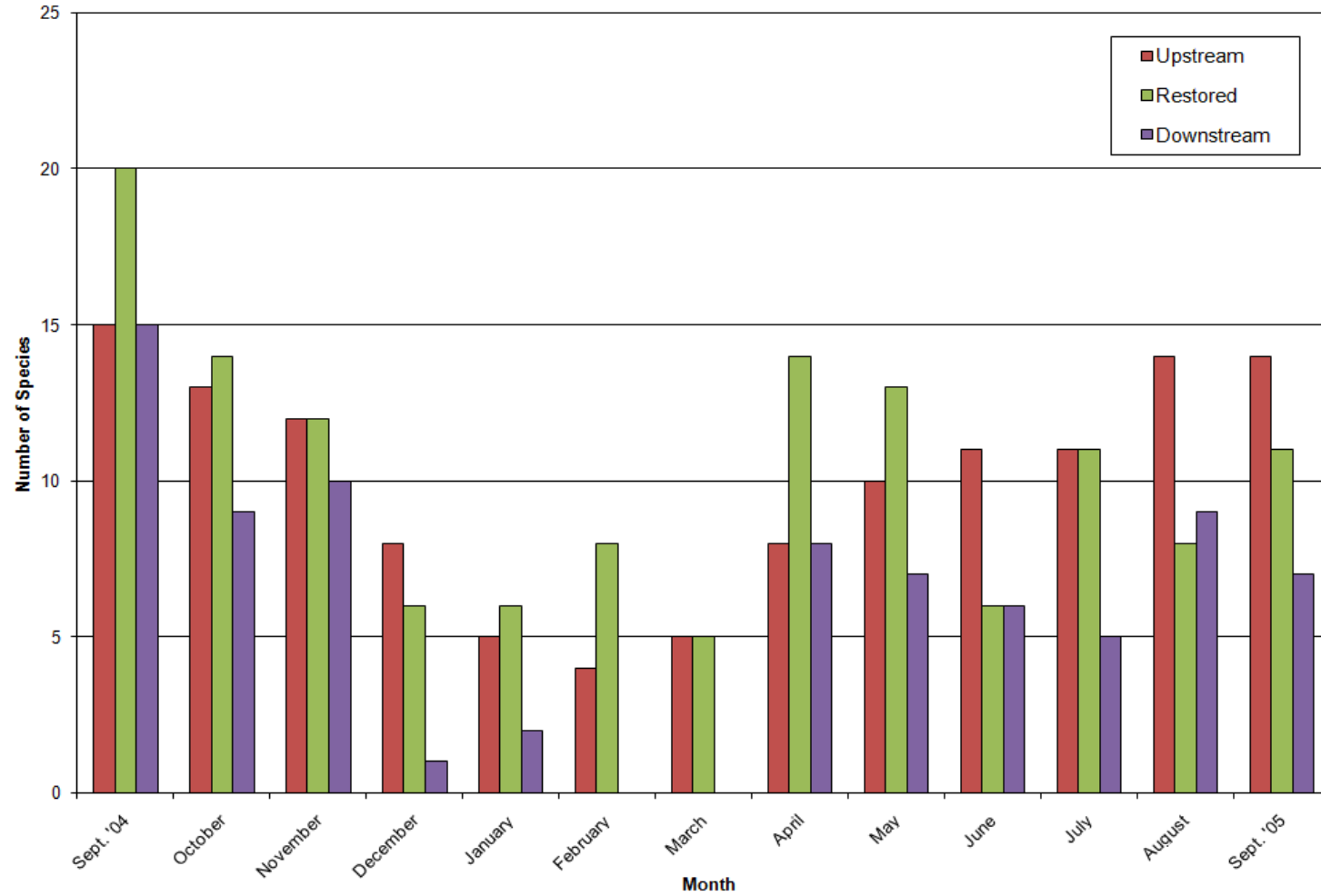


Figure 9. Index of Biotic Integrity (IBI) scores at Upstream, Restored, and Downstream sites in Nippersink Creek, McHenry County, IL (September 2004 – September 2005). Missing sample dates as in Table 2.

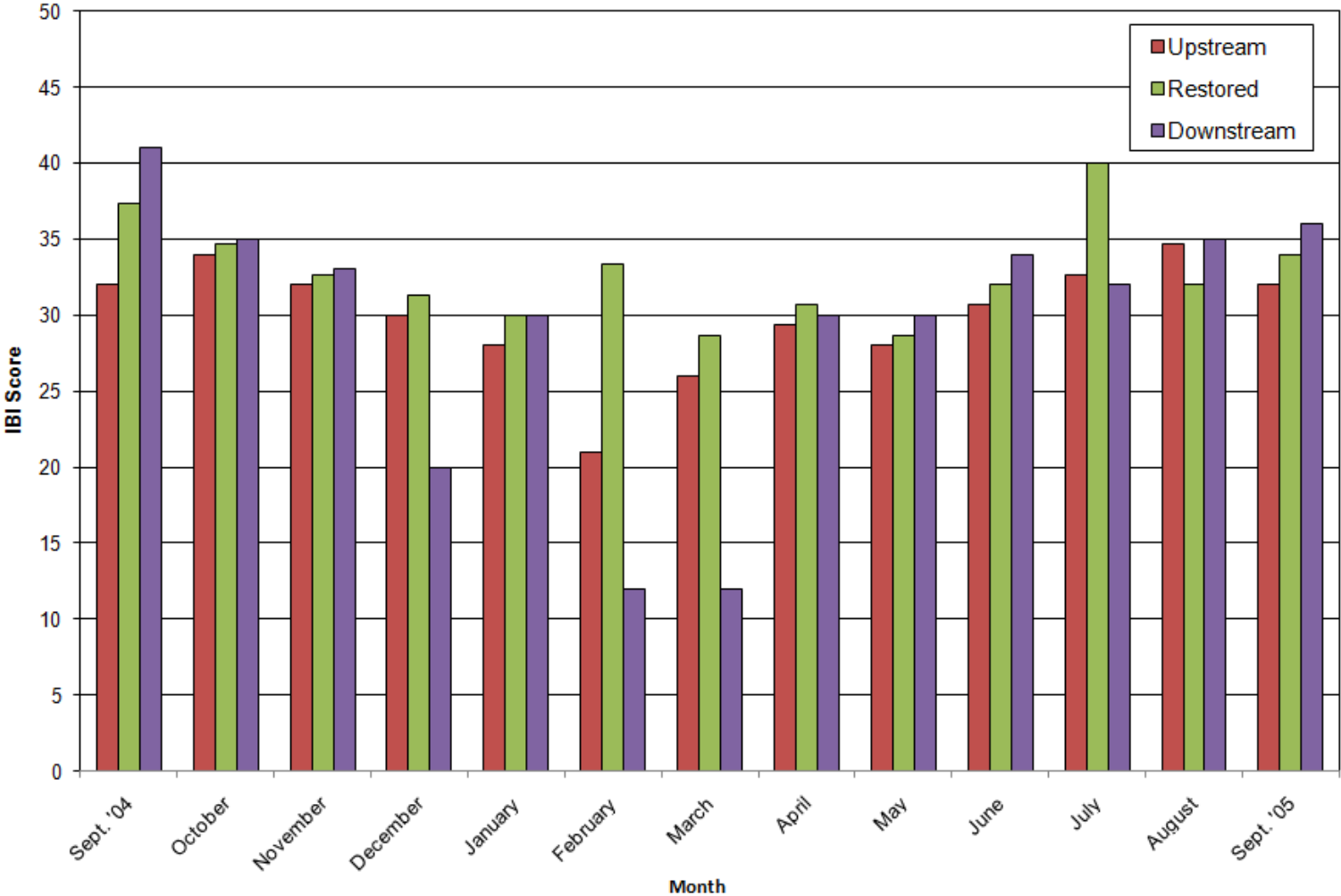
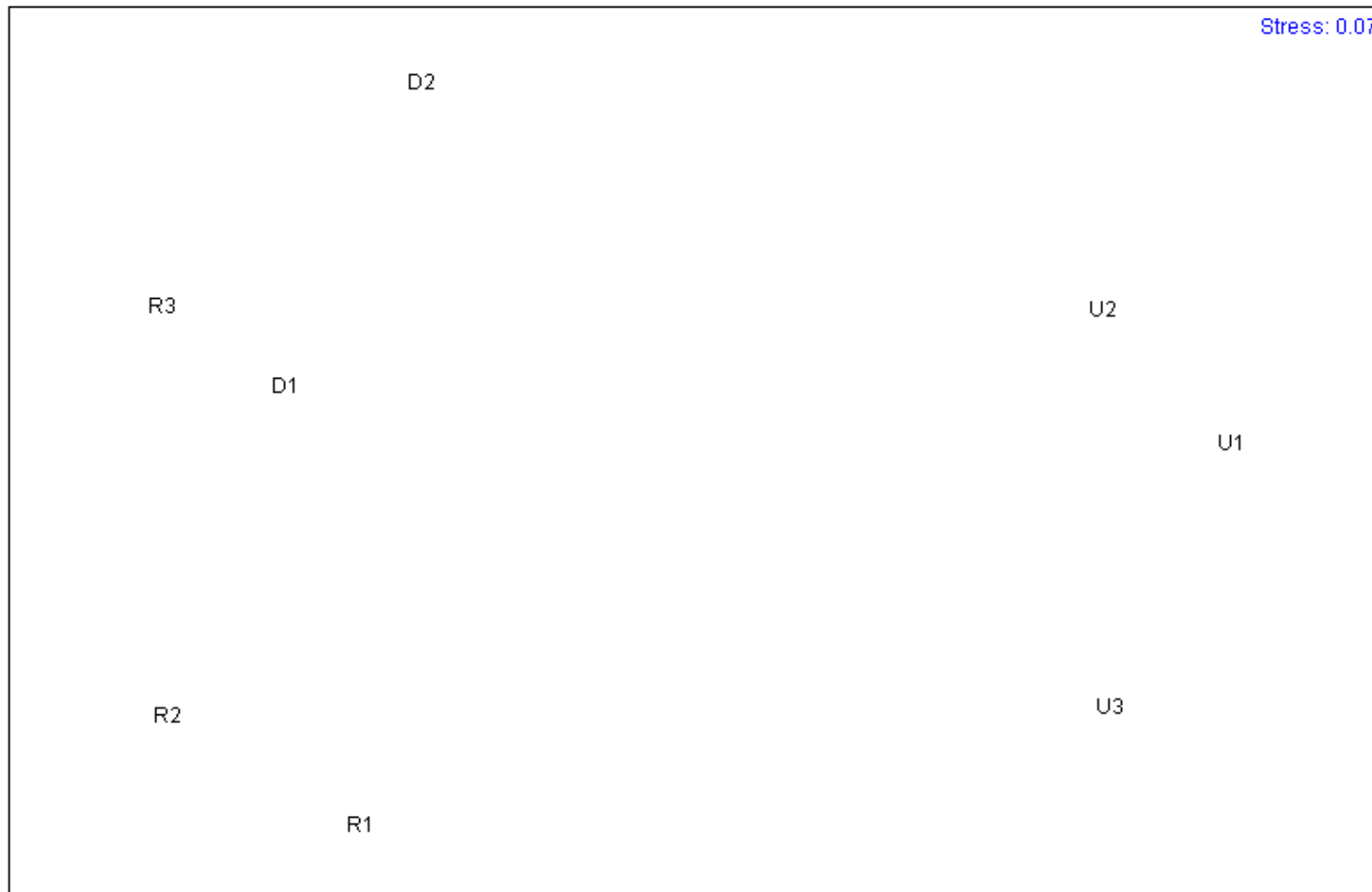


Figure 10. Non-metric multidimensional scaling ordination plot of study site fish community structure similarity using square-root transformed fish abundance data. Site abbreviations as in Table 1; missing sample dates as in Table 2.



had little prospect of misinterpretation (stress value = 0.06, Figure 11). The separation of upstream sites is partially due to the absence of *Labidesthes sicculus*, *Notropis atherinoides*, most *Lepomis* spp., and *Pimephales promelas*. However, unlike the plot of abundance data, fish communities of R1, R2, R3, and D1 formed a group, whereas site D2 had a fish community that is different from all other study sites. *Ameiurus natalis* and *L. sicculus* were found only at R1, R2, R3, and D1. The presence of *Hybognathus hankinsoni*, *Ictalurus punctatus*, *Noturus flavus*, and *Phenacobius mirabilis* as well as the absence of all *Lepomis* species and *Perca flavescens* distinguish site D2 from the remaining sites.

Shannon diversity was highest at R1, whereas the least diverse community was at D1 (Figure 12). In addition, Shannon diversity was lowest at the downstream sites, and highest at upstream and the restored sites. Principal components analysis (PCA) was performed using number of species, number of individuals, number of families, and Shannon diversity as dependent variables and suggests that 79.1% of the variability in the fish community can be explained by PC1 and PC2 axes, with number of fish and number of species most strongly influencing the ordination. The PC1 axis is represented by decreasing number of individuals and decreasing number of families, whereas the PC2 axis represents decreasing number of species and decreasing Shannon diversity (Figure 13). Restored sites and U1 are all found along the portion of the axis corresponding to high species richness and Shannon Diversity, even though R2 and R1 grouped along the portion of the axis representing lower numbers of individuals and families (Figure 13). Site D2 oriented along the axis of low numbers of species and Shannon Diversity,

Figure 11. Non-metric multidimensional scaling ordination plot of study site fish community structure similarity using presence/absence fish data. Site abbreviations as in Table 1; missing sample dates as in Table 2.

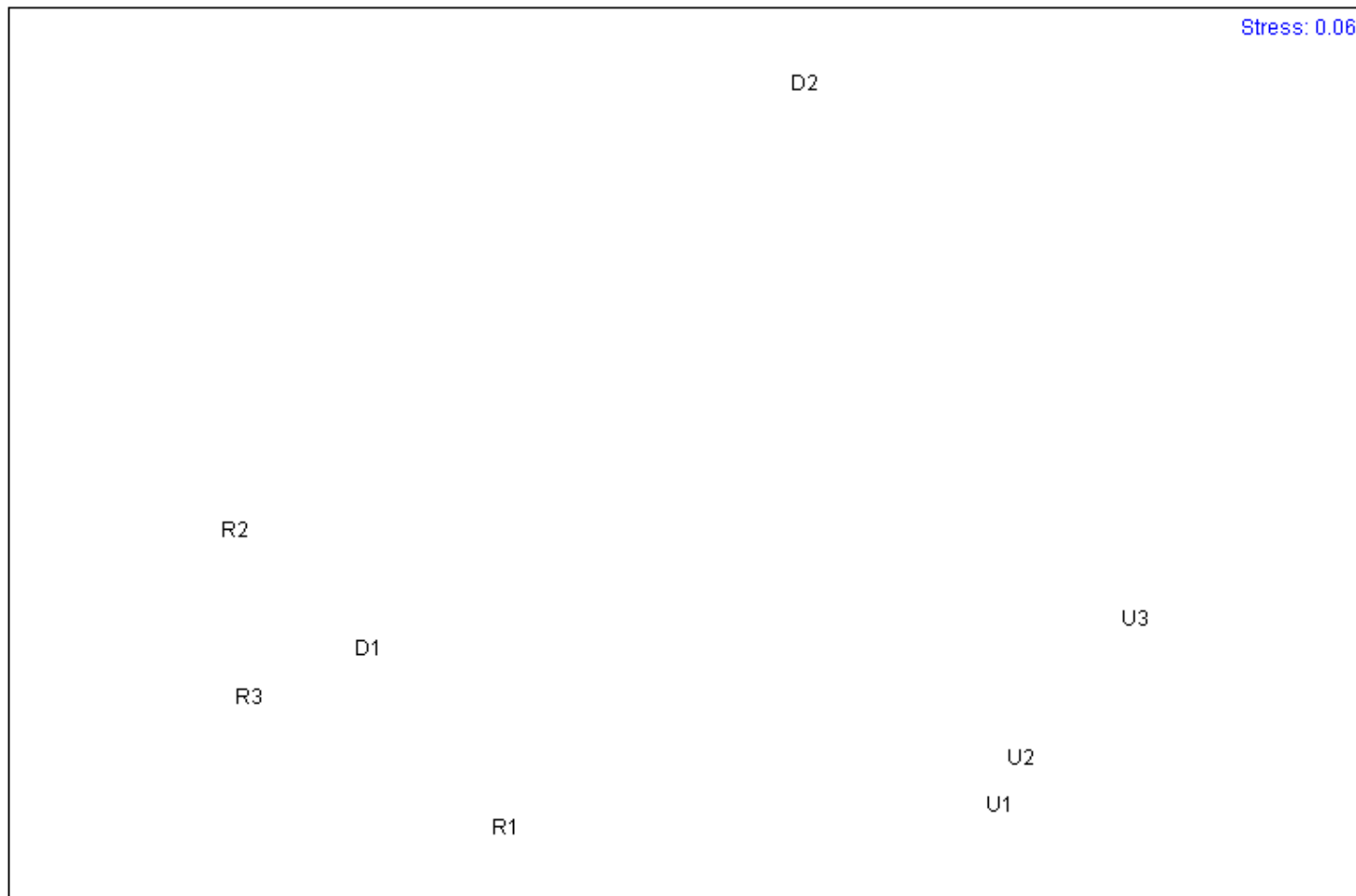


Figure 12. Shannon Diversity at each site in Nippersink Creek, McHenry County, IL (September 2004 – September 2005). Site abbreviations as in Table 1; missing sample dates as in Table 2.

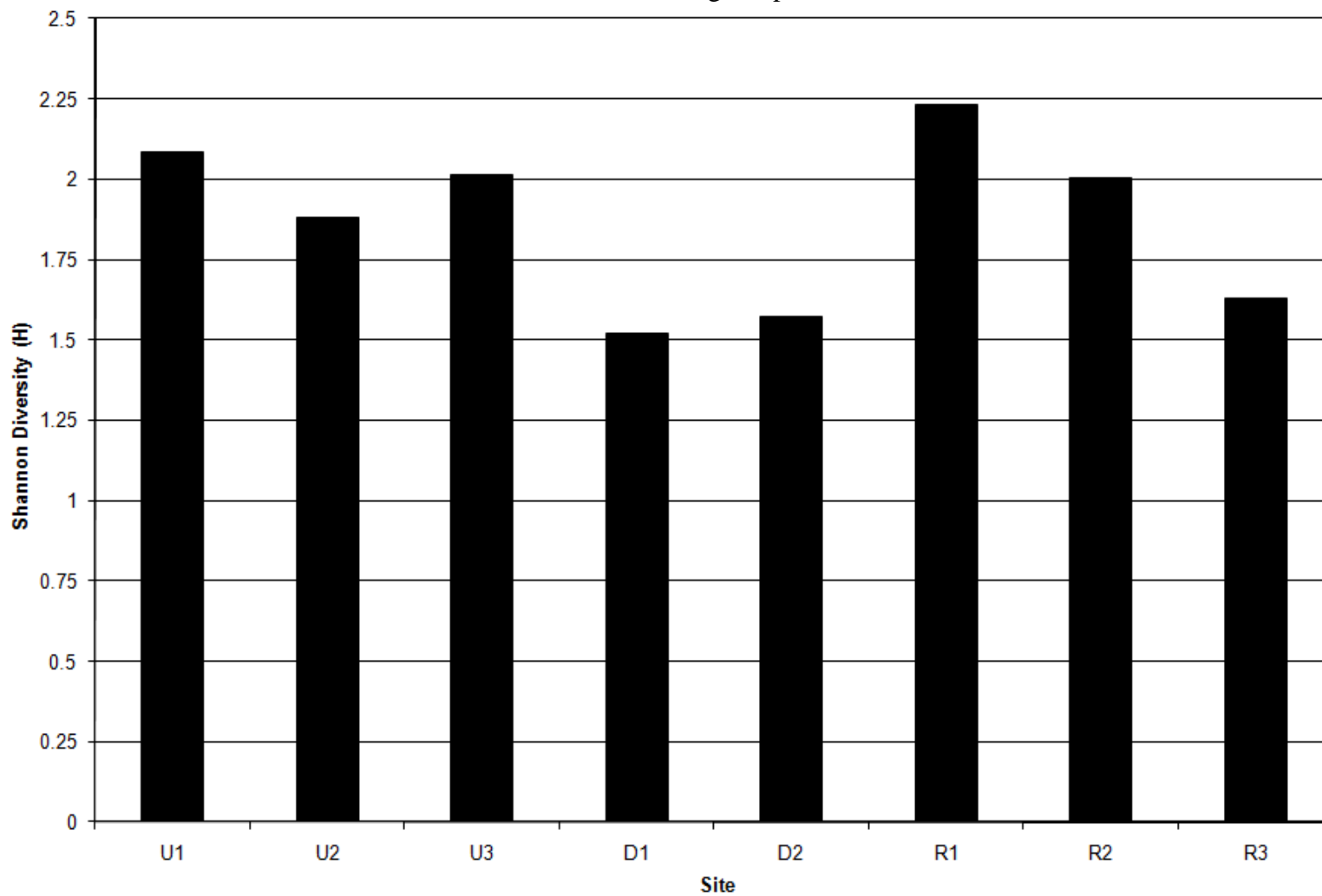
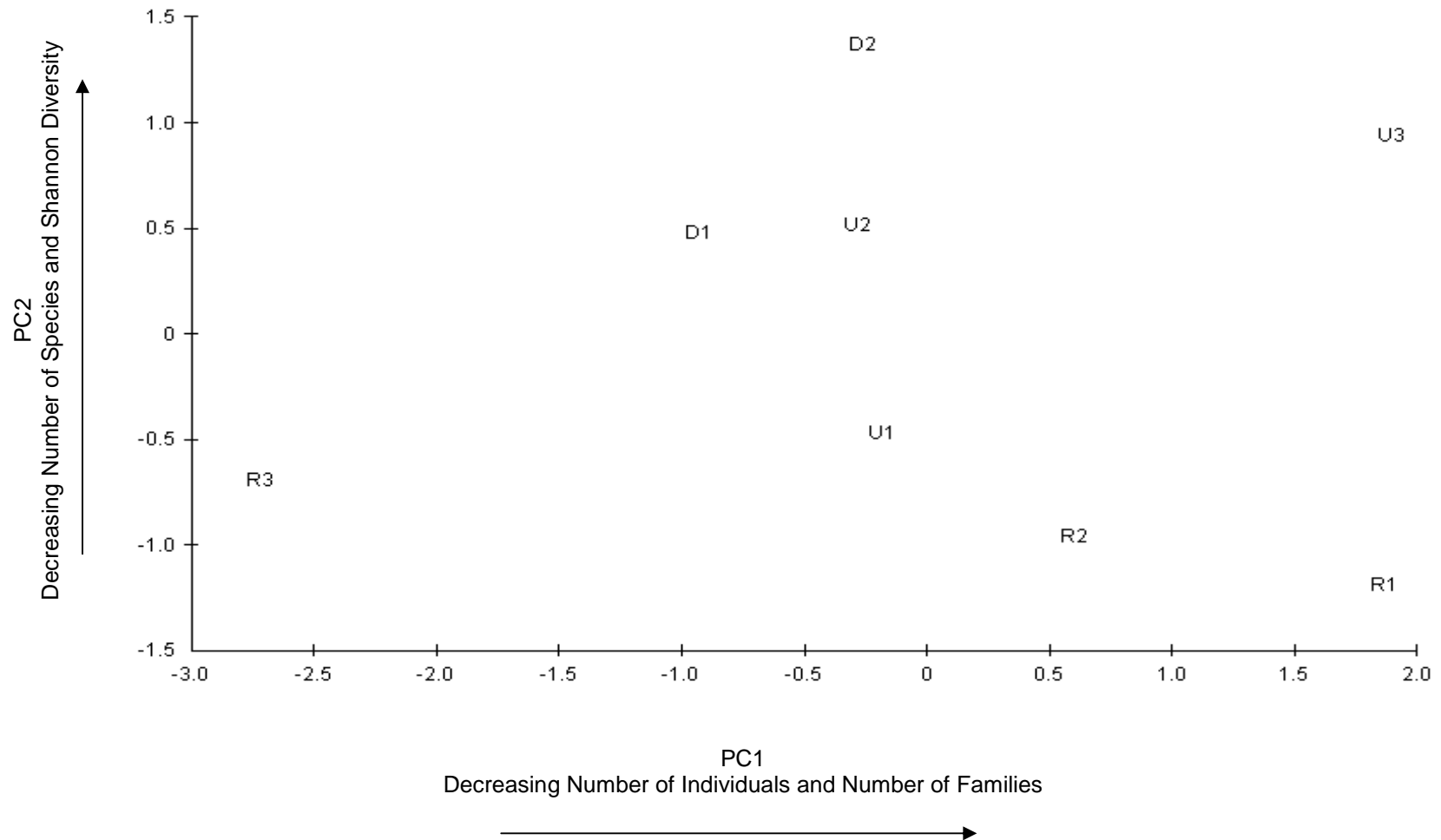


Figure 13. Principal components analysis plot of number of species, number of individuals, number of families, and Shannon diversity at each 50 meter site. Site abbreviations as in Table 1; missing sample dates as in Table 2.



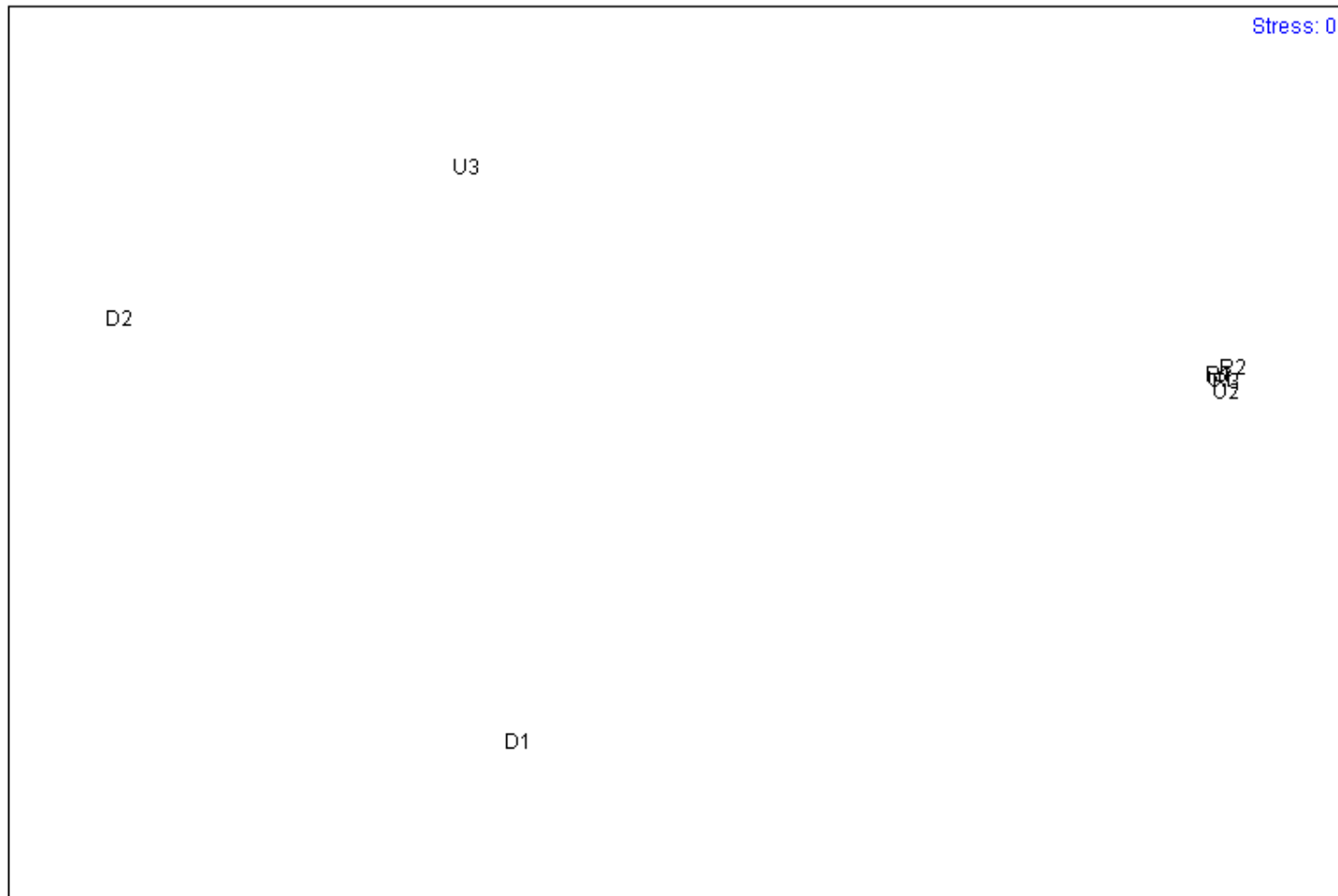
whereas sites U3 and R1 grouped along the axis represented by the lowest number of families and individuals. Sites D1, U1, and U2 occur near the middle of the plot, indicating intermediate numbers of species, individuals, families, and Shannon diversity.

Influence of Habitat Variables

The analysis of habitat variables yielded a much different ordination of sites compared to ordination plots of fish abundance data and presence/absence fish data, with a perfect representation and no prospect of misinterpretation (stress value = 0.00, Figure 14). The three restored sites, U1, and U2 form a tight cluster. These five sites have a meadow/grassland riparian zone, whereas the remaining sites have little-to-no meadow/grassland riparian composition. The sites in this tight cluster also, in general, contain a higher percentage of overhanging vegetation and a lack of overhanging trees, most likely due to the predominance of a meadow/grassland riparian zone. The Index of Biotic Integrity scores of the restored sites, U1, and U2 are slightly higher than the remaining sites. In contrast, D1, D2, and U3 appear dissimilar from each other and all other sites. This indicates four separate groups based on habitat composition. Site D1 is characterized by a high percentage of run habitat, and a low percentage of overhanging vegetation and algae substrate cover. Site D2 is characterized by a high percentage of riffle habitat, rubble/cobble and boulder presence, overhanging trees, and a completely woodland riparian zone. Site U3 is characterized by a high percentage of riffle habitat, rubble/cobble presence, overhanging vegetation, and a riparian zone comprised of a combination of meadow/grassland, shrubs, and woodland.

Draftsman plot analysis (i.e., all possible pairwise scatter plots) was performed to

Figure 14. Non-metric multidimensional scaling ordination plot of study site similarity based on habitat data. Site abbreviations as in Table 1; missing sample dates as in Table 2.



discern whether any variables were correlated; correlated variables may potentially obscure relevant results. Correlations among habitat data pairs included riffle and run (-0.998), riffle and rubble/cobble (0.983), run and rubble/cobble (-0.983), overhanging trees and meadow/grassland (-0.975), meadow/grassland and woodland (-0.988), and overhanging trees and woodland (0.996). It is redundant to keep both variables of a highly correlated pair in the BIO-ENV analyses, and thus, run and meadow/grassland were eliminated.

After removal of run and meadow/grassland, an analysis of community structure using BIO-ENV was conducted three times using 1) the complete set of habitat variables, 2) instream variables, and 3) riparian variables (Table 5) that revealed the three best habitat variable combinations for each trial (Table 9). The highest correlation between fish community structure and habitat is comprised of three variables: percent gravel, percent silt, and percent algae substrate cover (Table 9). However, examining only the instream habitat correlations reveals that the third best correlation to the fish community is provided by one habitat variable alone: algae substrate cover. The addition of other instream factors (i.e., percent gravel, percent silt, percent pool, and percent boulder) and riparian factors (i.e., percent overhanging trees, percent overhanging vegetation, and percent woodland) increase the correlation strength only slightly. In addition, an analysis of riparian variables produces weak correlations.

The three habitat variables that were best correlated with the fish assemblage were individually examined with the ordination plots of the presence/absence fish data. An examination of percent algae substrate cover superimposed on the plot of

Table 9. BIO-ENV results and Spearman correlation coefficients relating fish assemblage structure to habitat variables. Presence/absence fish data were used to equally weight all taxa. The habitat variable combinations listed below are the best correlations to the fish assemblage structure according to the BIO-ENV procedure. The top three correlations are listed for each test run; significant values ($p < 0.05$) are bold.

Spearman Correlation Coefficient	Habitat Variable Combination
All Habitat Variables, Excluding Run and Meadow/Grassland	
0.583	Gravel (%), Silt (%), Algae Substrate Cover (%)
0.557	Gravel (%), Silt (%), Overhanging Vegetation (%), Algae Substrate Cover (%)
0.546	Gravel (%), Silt (%), Overhanging Trees (%), Algae Substrate Cover (%)
Instream Variables Only	
0.583	Gravel (%), Silt (%), Algae Substrate Cover (%)
0.544	Pool (%), Boulder (%), Gravel (%), Silt (%), Algae Substrate Cover (%)
0.512	Algae Substrate Cover (%)
Riparian Variables Only	
0.166	Overhanging Vegetation (%), Woodland (%)
0.150	Overhanging Vegetation (%), Overhanging Trees (%)
0.139	Overhanging Trees (%)

presence/absence fish data reveals no clear gradient (Figure 15). There is, however, a spatial pattern: upstream natural sites, R1, and R2 have the most algae substrate cover, whereas R3, D1, and D2 have much lower algae substrate cover in comparison with the other four sites. When percent gravel is superimposed on the same ordination plot, no gradient is obvious, but there are clear delineations between certain site types (Figure 16). Sites U1 and U2 have higher percent gravel than both the restored and upstream natural sites. The amount of gravel at the remaining sites is relatively consistent. In superimposing percent silt on the same plot of presence/absence fish data, no clear pattern emerges (Figure 17). However, the plot reveals that U2, R1, and R2 are similar in terms of silt cover, whereas U3, which separates U2 and R1 geographically, is strikingly different, having a much smaller percentage of silt cover.

Comparison to Historical Data

A total of 5,393 fish representing forty-six species and 12 families were collected in the nine MCCD surveys used for comparison to current study sites.

Index of Biotic Integrity (IBI) was used to compare MCCD historical data and data collected during this study (Figure 18). Historical IBI scores were greater than those from the current study at all but one site. The nine most abundant species found in historical collections were six cyprinids (*Campostoma anomalum*, *Notropis cornutus*, *N. stramineus*, *Cyprinella spiloptera*, *Pimephales notatus*, and *Semotilus atromaculatus*), one catostomid (*Catostomus commersonii*), one centrarchid (*Lepomis macrochirus*), and one percid species (*Etheostoma flabellare*). Seven of the nine most common species are

Figure 15. Non-metric multidimensional scaling ordination plot of study site similarity based on presence/absence fish data. Bubble sizes represent percent algal substrate cover at each study site. Site abbreviations as in Table 1; missing sample dates as in Table 2.



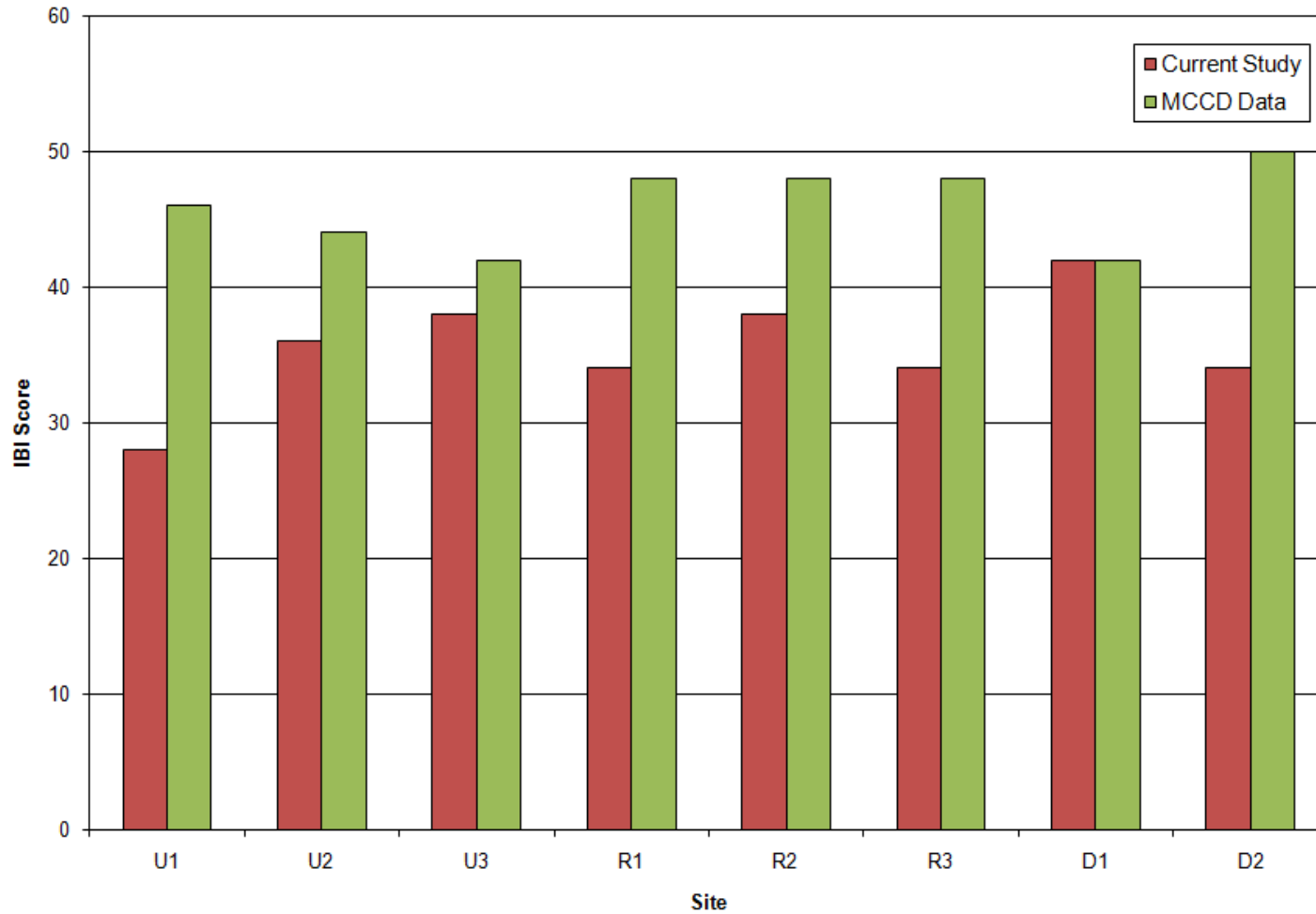
Figure 16. Non-metric multidimensional scaling ordination plot of study site similarity based on presence/absence fish data. Bubble sizes represent percent gravel at each study site. Site abbreviations as in Table 1; missing sample dates as in Table 2.



Figure 17. Non-metric multidimensional scaling ordination plot of study site similarity based on presence/absence fish data. Bubble sizes represent percent silt at each study site. Site abbreviations as in Table 1; missing sample dates as in Table 2.



Figure 18. Index of Biotic Integrity scores calculated from samples collected in the current study (September 2004 – September 2005) and collected by the McHenry County Conservation District in past collections. Site abbreviations as in Table 1. Missing sample dates from current study as in Table 2 and sample dates from historical data as in Table 6.



the same as in the current study: *C. anomalum*, *N. stramineus*, *C. spiloptera*, *P. notatus*, *S. atromaculatus*, *C. commersonii*, and *L. macrochirus*.

Analysis of the relationship of fish communities at historical sites produced a near perfect ordination that indicates D1, R3, and U1 have distinct fish communities (stress value = 0.01, Figure 19). The presence of *Labidesthes sicculus*, *Pomoxis nigromaculatus*, and *Esox lucius*, and the absence of *Catostomus commersonii*, and *Notropis cornutus* distinguished the fish community of D1 from other sites. Site R3 was characterized by the presence of *Aplodinotus grunniens* and the absence of *C. commersonii*, *Etheostoma nigrum*, and *Lepomis cyanellus*. Site U1 was distinct due to the presence of *Culaea inconstans*, and *Etheostoma flabellare*, and the absence of *Pimephales promelas*, *Cyprinella spiloptera*, and *E. zonale*. Sites U2 and U3 group together, indicating a similarity in their fish communities. The remaining sites (R1a, R1b, R2, and D2) form yet another similar group, which is characterized by the presence of *N. atherinoides*, *L. gibbosus*, *Carpiodes cyprinus*, *Ameiurus melas*, *Perca flavescens*, and *Sander vitreus*.

Comparison to Pre-Restoration Channel Data

McHenry County Conservation District provided Index of Biotic Integrity (IBI) scores for the pre-restoration channelized portion of Nippersink Creek. These scores ranged from 34-50 (Figure 20). Fish data from the pre-restoration channelized portion of Nippersink Creek were also analyzed, producing a good, non-misleading ordination that grouped sites into multiple distinct groups (stress value = 0.07, Figure 21). While all four sites appear to possess dissimilar fish communities, C1 is distinct from the rest of the

Figure 19. Non-metric multidimensional scaling ordination plot of study site similarity based on presence/absence historical fish data from McHenry County Conservation District. Site abbreviations as in Table 1; sample dates as in Table 6.

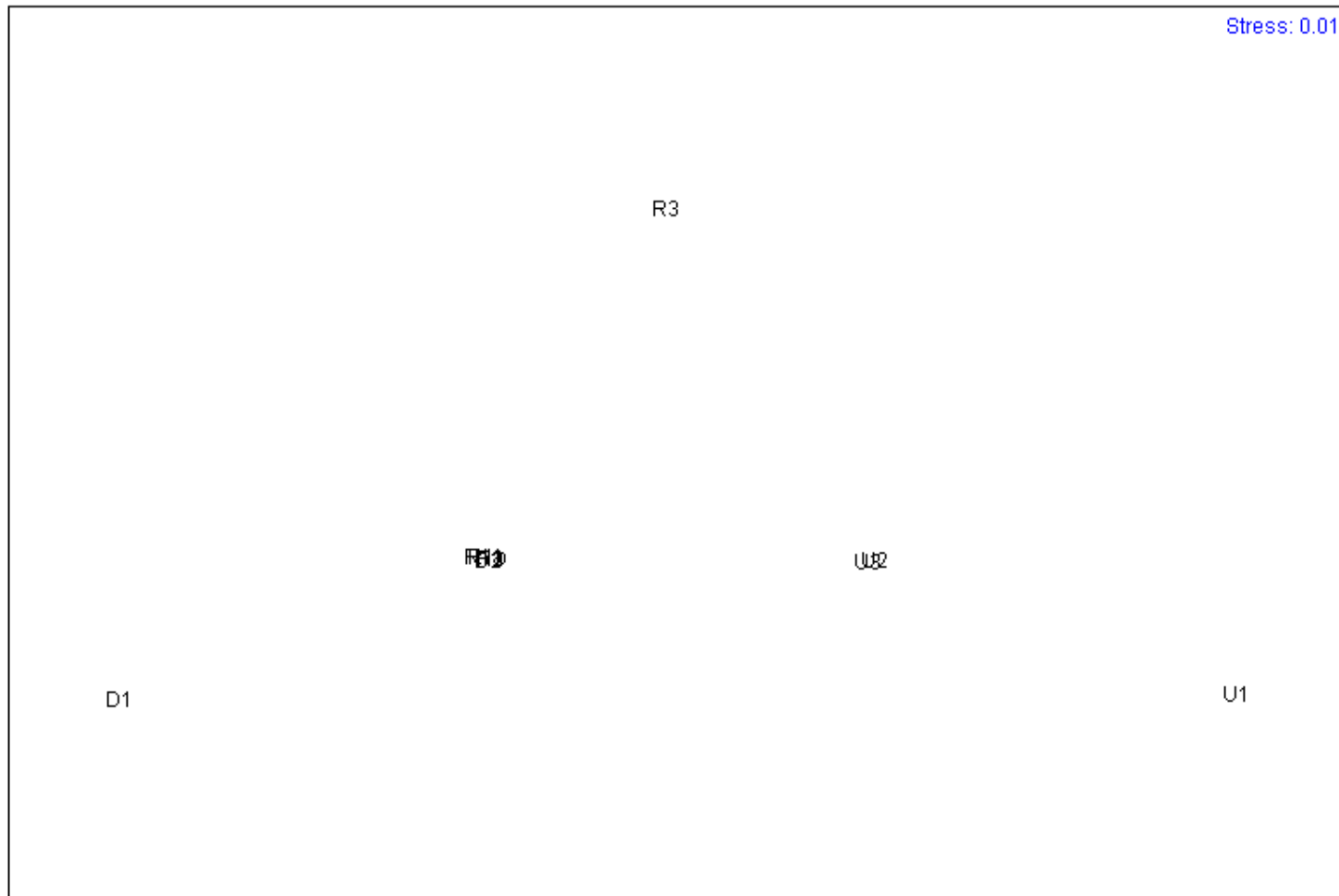


Figure 20. Index of Biotic Integrity scores of pre-restoration channel sites on Nippersink Creek, McHenry County, Illinois. Scores calculated by the McHenry County Conservation District. Sample dates as in Table 7.

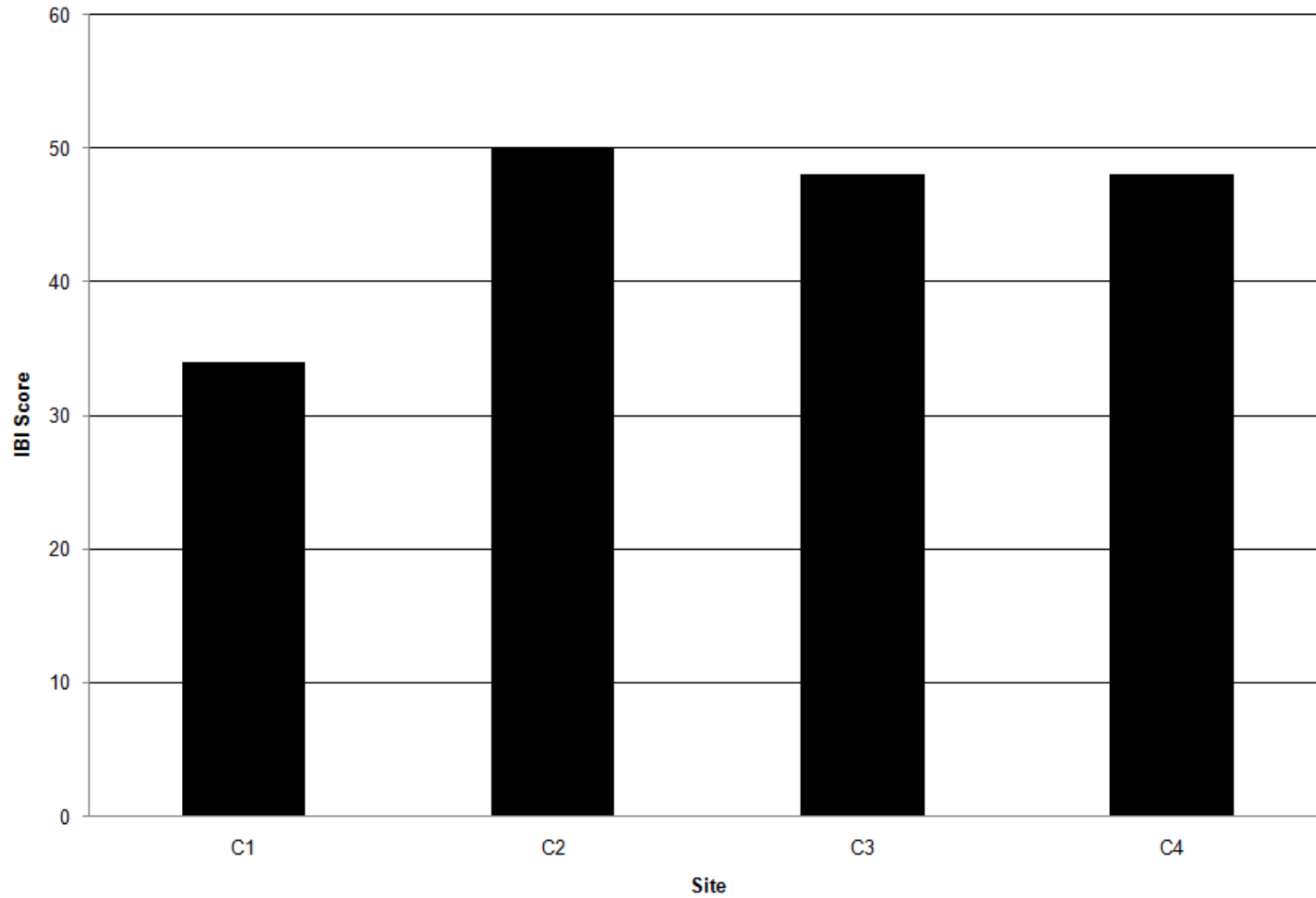
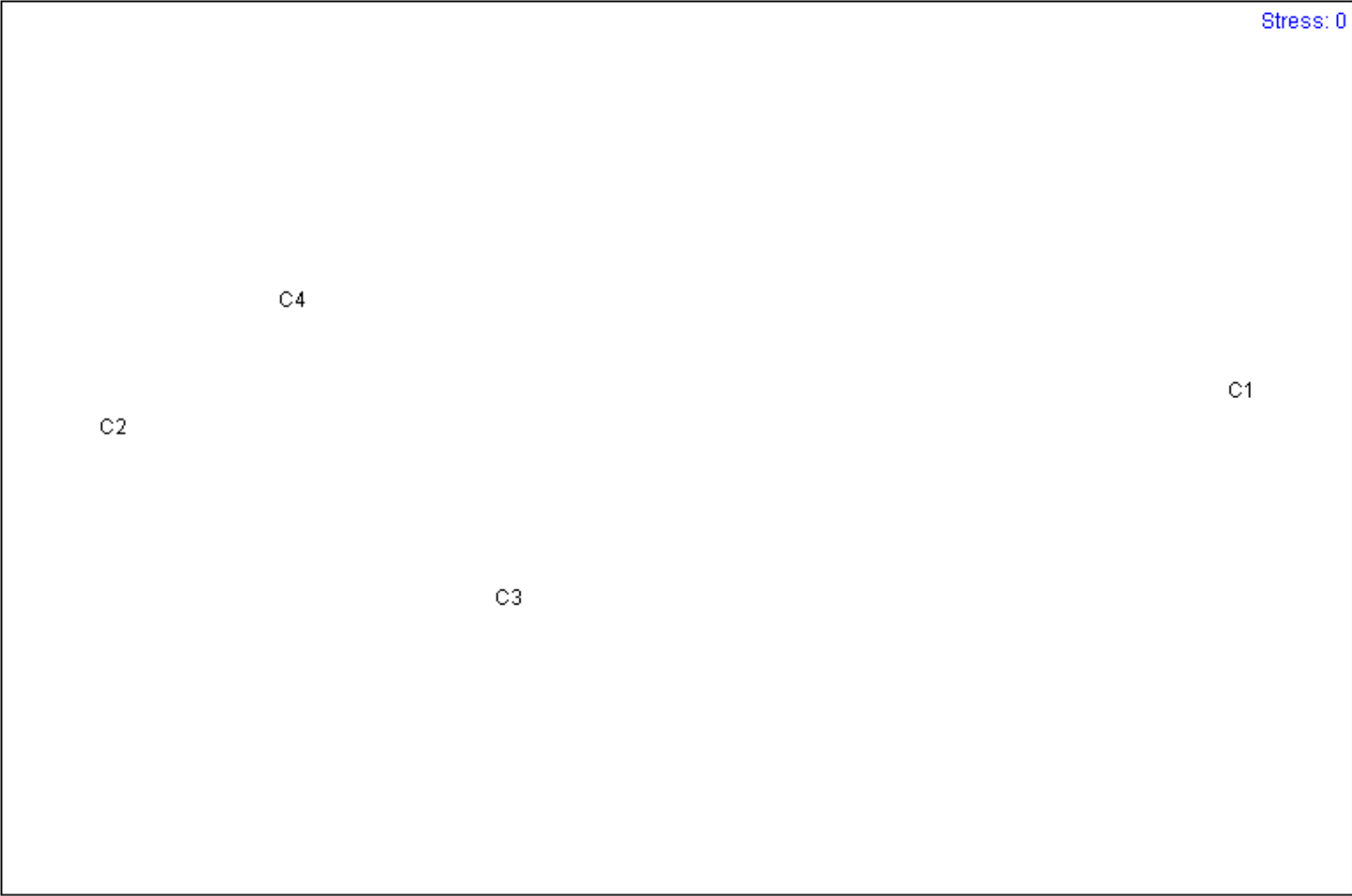
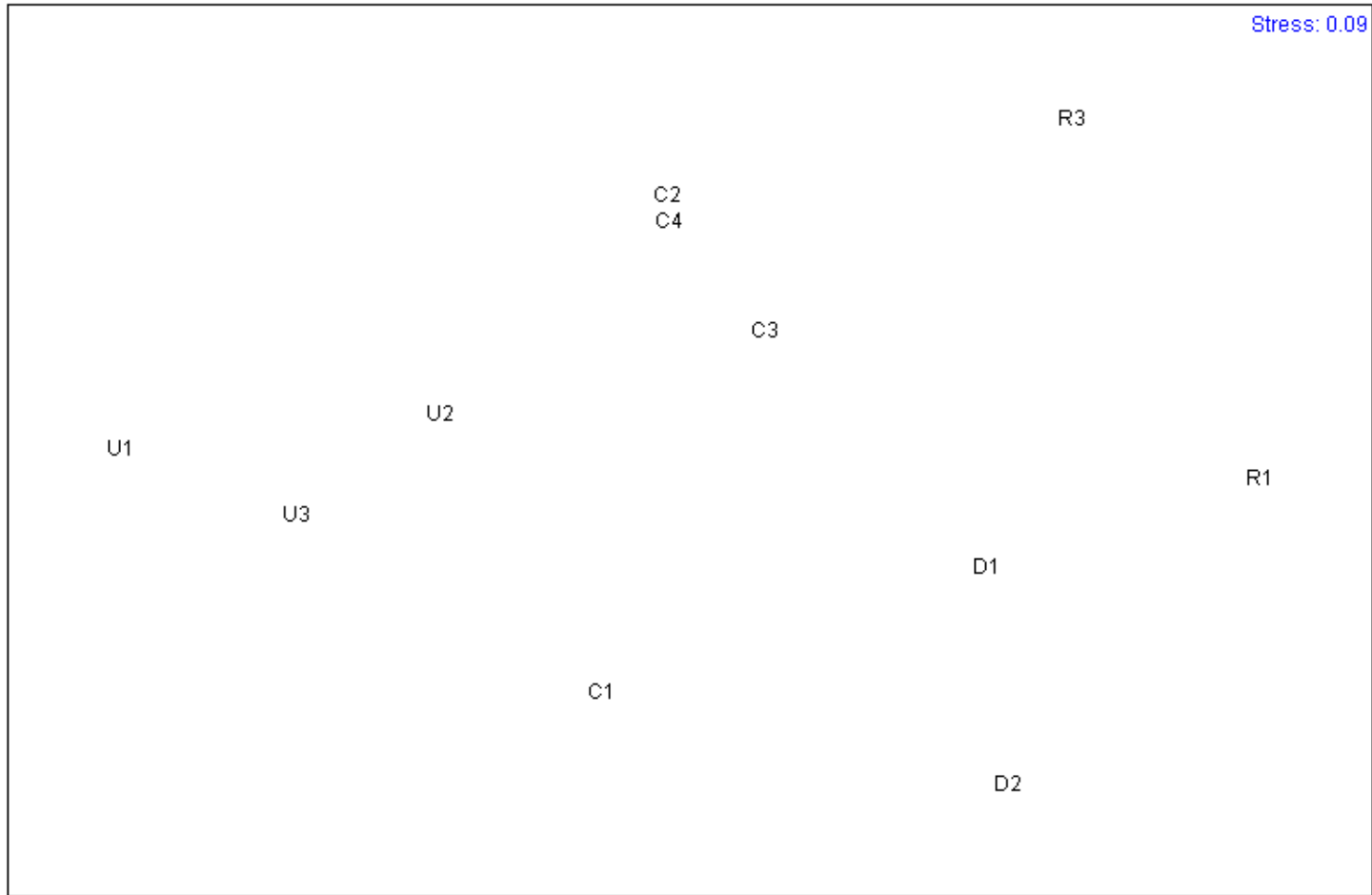


Figure 21. Non-metric multidimensional scaling ordination plot of site similarity based on presence/absence fish data from McHenry County Conservation District pre-restoration channel samples. Sample dates as in Table 7.



sites, while C2, C3 and C4 appear more similar. This could be due to the absence of *Cyprinus carpio*, *Ameiurus natalis*, *Lepomis* species, *Micropterus salmoides*, and *Perca flavescens*. In addition, *Notropis hudsonius* was the dominant *Notropis* species found at C1, while both *N. cornutus* and *N. spilopterus* were dominant at C2, C3, and C4. An ordination of summer pre-restoration channel data and data from the current study (July and August 2005) revealed patterns similar to the plot of the pre-restoration channel data (stress value = 0.09, Figure 22). Although the pattern of the pre-restoration channel sites was unchanged, by plotting channel data with summer data from the current study, the fish communities of the channel sites can be examined in the context of the fish communities found in the current study. Upstream natural sites; channelized sites C2 – C4; and R3 form three separate groups that appear to be very distinct from the remainder of the sites (i.e., C1, D1, D2, R1), which form one large group that is characterized by the absence of multiple *Notropis* and *Lepomis* species.

Figure 22. Non-metric multidimensional scaling ordination plot of site similarity based on presence/absence fish data from McHenry County Conservation District pre-restoration channel samples and July & August pooled data from the current study. Missing sample dates from current study as in Table 2 and sample dates from pre-restoration channel as in Table 7.



DISCUSSION

Fish Community Structure and Integrity

Results of this study indicate that fish communities of restored sites were generally as healthy or healthier (*sensu* Karr 1986) compared to upstream and downstream natural sites. Sites with the greatest fish abundance (Restored Site 3 [R3]) and highest Shannon diversity (Restored Site 1 [R1]) were in the restored section of the stream, and species richness was as high as or higher in restored sites than in upstream and downstream natural sites throughout most of the study. In addition, three of the four highest Index of Biotic Integrity (IBI) scores were also found in the restored section. This suggests that the restored section of Nippersink Creek supports a diverse fish community that is comparable to or higher in quality than fish communities in natural upstream and downstream reaches of Nippersink Creek.

Non-metric multidimensional scaling analysis revealed that fish communities in restored sites and Downstream Site 1 (D1) are similar. Sites D1 and R3 are close to one another, and this geographic proximity may explain the similarity. Also, Wonder Lake separates the upstream natural study area and the restored study area, and this may explain why upstream natural sites differ from restored sites and downstream natural sites. Influence of the lake on fish communities in Nippersink Creek was not specifically examined in this study, however, as Nippersink Creek flows out of Wonder Lake into

Glacial Park, it becomes noticeably deeper and, at times, wider than upstream natural sites, potentially creating new habitat that supports a different fish community. Larger habitats support more species than smaller habitats, and studies have found fish species richness, habitat area, and habitat diversity to be significantly correlated (Angermeier and Schlosser 1989, Arunachalam 2000, Cianfrani et al. 2009). Thus, the increase in habitat that could occur as Nippersink Creek flows out of Wonder Lake will enable the support of additional fish species.

The fish community composition of Downstream Site 2 (D2) is quite different than D1 and restored sites, which was somewhat unexpected given the proximity of the sites to one another. However, D2 is comprised of more than twice the area of riffle habitat and nearly twice the amount of rubble/cobble substrate of all other sites except Upstream Site 3 (U3). Fish community analyses found that D2 was different than the restored sites and D1 in terms of both number of fish and fish family richness over the course of the study. Habitat and microhabitat use varies among or across species and from juvenile to adult (Moyle and Baltz 1985, Fore et al. 2007), so it is plausible that habitat at D2, which is unique to this study, supports a different fish community than other sites.

IBI scores, used to assess the health of Nippersink Creek and to compare sites, were low throughout the study, indicating that Nippersink Creek is a limited aquatic resource per IEPA (1989). According to Karr (1981), the limited aquatic resource category corresponds to a poor integrity class, which is characterized by a dominance of omnivores, generalists, and tolerant fishes, with few top carnivores present. Previous

electrofishing work performed by the McHenry County Conservation District (MCCD) prior to restoration found higher IBI scores, on average, than the current study. One assumption of the IBI is that the fish sample used to calculate a score is representative of the entire fish community (Karr 1981). The low IBI scores found throughout this study may be due to undersampling of Nippersink Creek's fish community. Karr (1981) recommends the use of seines for small streams and a change to electricity-based sampling gears (electric seine, backpack electroshocker, pram, etc.) as stream size increases. It is possible that the size of the stream throughout the restored and downstream natural areas warranted the use of an electroshocker rather than a seine. Another potential explanation for the low IBI scores is that pool habitat was not well represented at the sites. Schlosser (1982) found that both species diversity and species richness increase with the addition of pool habitat. Even though the IBI scoring system adjusts for the maximum species richness value based on stream size (i.e., a small stream can only support so many species, and accordingly will not be scored poorly), an increase in species richness could increase IBI scores (Karr 1991).

IBI scores can be influenced by the length of reach sampled and sampling area due to the species-area relationship (Paller et al. 1996). Paller et al. (1996) found that 50-meter sites were not sufficient to always calculate precise IBI scores, as all microhabitats were not necessarily represented. Also, IBI scores in the present study were averaged over thirteen months to compare sites, and often, no fish were collected in winter months, resulting in an IBI score of 12. These low scores likely resulted in low averages. The

low scores found in the current study may very well be a function of the low number of fish collected in winter rather than actual stream quality.

Influence of Habitat Variables

Non-metric multidimensional scaling analysis of habitat variables grouped study sites much differently than analysis of fish assemblage structure. Although D1, D2, and U3 all showed distinct habitat structure, all restored sites were grouped with the two most upstream natural sites, indicating that, at least in terms of habitat, the restored sites are similar to undisturbed sites. However, the restored sites grouped with the only two study sites that were located in an agricultural area, Upstream Site 1 (U1) and Upstream Site 2 (U2). Although they have intact meadow/grassland riparian zones, portions of Nippersink Creek at U1 and U2 flow through farmland and/or cattle pasture.

Numerous studies have reported lower species diversity in agricultural streams due to increased siltation and subsequent reduction of riffle habitat (Berkman and Rabeni 1987, Walser and Bart 1999) or riparian removal and channelization (Sullivan et al. 2004). This is consistent with the results from this study, in which species richness was lower at U1 and U2 than at the restored sites and D1. It appears that the habitat at the upstream natural sites is, in fact, of low quality and/or disturbed, but IBI scores at U1 and U2 are higher than the downstream natural sites and R1. This finding is consistent with Snyder et al. (2003) and Wang et al. (2000), who found a positive relationship between agricultural land uses and IBI scores. One of the metrics of the IBI is the number of native species, and Fitzgerald et al. (1998) reported that even when agricultural land-use was dominant, as long as riparian zones remained undisturbed (as they are at sites in this

study), the overall fish community was supportive of native fishes. This conclusion is supported by the IBI scores calculated for U1 and U2.

However, the fact that IBI scores at the upstream natural sites are slightly higher might not be noteworthy. IBI scores for all sites range from a rating of Limited Aquatic Resource to Moderate Aquatic Resource (IEPA 1989), which indicates a shift from a balanced to deteriorating trophic structure and a reduction in intolerant species and top predators (Karr 1981). Further, the integrity class IBI score ranges used by Karr (1986) categorize Nippersink Creek as poor integrity, which indicates a dominance of tolerant, generalist fish species. Therefore, although the IBI scores at U1 and U2 are higher than the downstream natural sites, they are still very low and indicate that the quality of Nippersink Creek could be improved. That said, there are drawbacks to the use of the IBI, primarily the fact that the index distills down 13 metrics into a single number; a lot of biological information may be lost in calculating an IBI score.

Results from the BIO-ENV analysis indicated that percent algae substrate cover and, to a lesser extent, percent gravel and percent silt, exerted the most influence on fish communities. Although percent riparian vegetation was one of the most influential characteristics in grouping sites based on habitat, it had almost no direct influence on fish community structure at each site. These data suggest that the riparian zone has little influence on fish community composition, a fact that is not supported by previous research. Sullivan et al. (2004) showed that streams in agricultural landscapes in Indiana with altered or removed riparian vegetation had less complex fish communities. Other work supports the conclusion that complex habitat, including the presence of established

riparian zones, supports diverse fish communities (Gorman and Karr 1978, Schlosser 1982, Berkman and Rabeni 1987, Angermeier and Schlosser 1989, Arunachalam 2000, Cianfrani et al. 2009; Stauffer et al. 2000).

Stauffer et al. (2000) found that streams with wooded riparian zones had higher IBI scores than those with open riparian zones, which indicates that wooded riparian zones create better conditions for fish than the meadow/grassland riparian zones typical of the restored sites in this study. This contrasts with Murphy et al. (1981) who demonstrated that open-canopied sections of a trout stream, i.e., those without a wooded riparian zone, support greater fish production due to the limitations of shade on algal-dependent systems. Results from this study support Murphy et al. (1981), in that the three sites with primarily wooded riparian zones (U3, D1, and D2) had the lowest IBI scores. However, Nippersink Creek is not a trout stream, so results should support the conclusions of Stauffer et al. (2000). Nevertheless, the importance of algae substrate cover to fish communities of Nippersink Creek may be one reason why results of this study support those of Murphy et al. (1981).

Although Gorman and Karr (1978) showed that a combination of depth, flow, and substrate were the most important determining factors in predicting fish species diversity, Berkman and Rabeni (1987) found in their study of agricultural streams in Missouri that substrate, primarily the amount of silt, is of greater importance. Talmadge et al. (2002) found species diversity to be positively correlated with all substrate types except silt. Diana et al. (2006) found sedimentation to be negatively correlated to IBI score, with high scores being determined by the amount of exposed gravel. This supports the

findings of this study, in which percent gravel and percent silt were major determining factors in structuring fish communities in Nippersink Creek. In addition, the most important habitat factor in this study, percent algae substrate cover, is not necessarily a substrate component, but is potentially a function of the substrate found. Percent algae substrate cover is greater at the three upstream sites, R1, and Restored Site 2 (R2), i.e., the five most upstream study sites. Site R3 does not support this trend due to the presence of a high percentage of clay substrate. Because D1 and D2 both have primarily wooded riparian zones, increased stream shading likely inhibited an extensive benthic algal community (Murphy et al. 1981).

The suite of variables examined in this study is not all-inclusive, and potentially important habitat variables may have been omitted. Diana et al. (2006) found the stability of stream discharge to be positively correlated to IBI scores in agricultural streams. Substrate components were major habitat variables included in this study, however, depth and current velocity were not examined. Although percent silt was a substrate component of this study, an examination of bank slope and bank erosion could lend further insight into overlying causes of siltation (Talmadge et al. 2002, Iwata et al. 2003), which may directly affect fish community composition (Berkman and Rabeni 1987). Murphy et al. (1981) and Stauffer et al. (2000) found canopy cover to be an important habitat factor that influenced fish communities. Canopy cover should be assessed in future studies on Nippersink Creek because of its influence on algae substrate cover.

Influence of Habitat on Common Species

Habitat selection by a fish species is generally consistent over the range of the species (Gorman and Karr 1978), and each species may be considered microhabitat generalists or specialists (Wood and Bain 1995). Many biotic and abiotic factors, i.e., substrate and riparian composition, predation, and competition, affect and interact to influence habitat selection by stream fish (Koehn et al. 1994).

Although the most common fish species collected during this study represented four families, similarities in their habitat requirements and life history may lend insight into the results of this study. *Campostoma anomalum* is a silt-intolerant, algivorous cyprinid found throughout the Midwest in clear or turbid small, fast-flowing streams, with a clean gravel or sand-gravel bottom (Smith 1979). In the current study, *C. anomalum* were abundant only in upstream natural sites. Sites U1 and U2 are in an agricultural area, which could experience increased sedimentation (Berkman and Rabeni 1987), but the presence of *C. anomalum* in such large numbers suggest that siltation might not be an issue, at least in the stream stretches sampled in this study. Berkman and Rabeni (1987) found a decrease in *C. anomalum* as siltation increased, which they attributed to decreased algal production in turbid conditions.

Campostoma anomalum is well-adapted to scrape algae from rocks, logs, and bottom debris by using a hardened protuberance on the lower jaw (Becker 1983). The strong relationship between *C. anomalum* and algal communities has been well documented (Power et al. 1985, Gelwick and Matthews 1992), and it is unlikely that *C. anomalum* would be found in habitats without an abundant algal food resource. Sand,

gravel and rubble/cobble substrates predominated in upstream sites, and algae substrate cover was common, suggesting good conditions for this particular minnow. Increased runoff is common in agricultural areas (Walser and Bart 1999), and Stauffer et al. (2000) found higher IBI scores at stream sites with low runoff potential. Agricultural runoff adds additional sediment and nutrients to streams, possibly stimulating increased primary production that could support high numbers of algivorous fish, like *C. anomalum*, and macroinvertebrates (Talmadge et al. 2002). On the other hand, agricultural runoff may also add pesticides and an overabundance of nutrients and silt that may decrease primary production (Talmadge et al. 2002). Since agricultural runoff and turbidity data were not collected, further study is necessary to determine the influence of agriculture on primary production.

Other species often collected at upstream sites include *Catostomus commersonii*, a species that prefers clear water and sandy, gravelly substrates (Smith 1979), and *Semotilus atromaculatus*, which is considered tolerant, but prefers clear, warm, slow flowing waters with hard substrates (Becker 1983). Sites U1 and U2 had the highest percentages of gravel found at any sites. This helps to create optimal spawning conditions for these two species in particular, as both species require gravelly substrates to spawn.

Restored and downstream natural sections of Nippersink Creek are characterized by a deeper, wider channel than the upstream natural sites, with a higher percentage of silt substrate, and, in the case of R3, a high percentage of clay. A transition from meadow/grassland to a wooded riparian zone occurs in between the restored and

downstream natural section. The change in habitat from upstream to downstream coincides with a change in the numerically dominant species, which have different habitat requirements than the numerically dominant species in the upstream area. Even though *Cyprinella spiloptera* is not found in high numbers throughout the restored section, it is the most abundant cyprinid found in this study, and is numerically dominant in R3 and both downstream natural sites. *C. spiloptera* can tolerate silty conditions and polluted waters, and can be the most numerous cyprinid in turbid waters (Becker 1983). It is a habitat generalist, and occurs throughout streams and rivers, in clear to turbid waters, with or without vegetation, and often shows little preference for soft or hard substrates (Mueller and Pyron 2009). The absence of *C. anomalum* and the dominance of a tolerant minnow species such as *C. spiloptera* highlight the change in habitat composition and perhaps quality from an upstream to downstream gradient in Nippersink Creek.

Etheostoma zonale, a darter species that requires clean, clear waters and rocky substrates, was relatively common at R3 and throughout the downstream natural sites. This pattern seems to contradict the change in habitat composition and quality as evidenced by the absence of the *C. anomalum* and numerical dominance of *C. spiloptera*. In addition, *Notropis stramineus*, the second most abundant fish in the study, was numerous at U2, R2, R3, D1, and D2. *Notropis stramineus* is a habitat generalist, but is relatively intolerant and prefers sandy, gravelly, clear streams (Becker 1983, Mueller and Pyron 2009). It is unlikely that these two species would be found, particularly in such high numbers, in a stream with poor habitat.

Comparison to Historical Data

Ordination of historical fish data resulted in site groupings that differed from the present study. The most likely reason for the differences between historical and current nMDS analyses is the addition of multiple species not collected in the current study. The addition and removal of various species contributed to the differences in grouped sites.

Index of Biotic Integrity (IBI) scores from historical data ranged from 42-50 and classify Nippersink Creek as a highly valued aquatic resource. Highly valued aquatic resources are considered good for gamefish, with species richness values only slightly less than that expected in pristine conditions (IEPA 1989). This classification is in contrast to the IBI scores calculated in the current study, which classified Nippersink Creek between a limited/moderate aquatic resource to a highly valued resource (IEPA 1989).

One of the possible reasons for differences in IBI scores could be sampling methodology. Historical samples were collected by electrofishing, using either an electric seine or backpack electroshocking unit, whereas samples from the current study were collected using a hand seine. There are benefits and drawbacks to both seining and electrofishing sampling methods (Barbour et al. 1999). Onorato et al. (1998) compared the two methods in an Alabama stream and found that bass, catostomids, darters, and sunfish were captured in greater numbers by electroshocking; more cyprinids were captured by seining. This may explain the dominance of minnow species and lack of large fish species in the current study. Gear selectivity could have a major impact on IBI scores if entire families of fish are not represented. Thus, a possible explanation for

lower IBI scores in the current study may be due to the lack of large fish caught using the hand seine, as large fish can see and avoid a slow-moving seine in clear waters (Onorato et al. 1998).

Effects of Channelization and Restoration

Pre-restoration channel IBI scores indicate the channelized portion of Nippersink Creek was a moderate to highly valued aquatic resource (IBI scores ranged from 34-50). This contrasts with the results of the present study, in which the restored sites, on average, were a moderate aquatic resource (IBI scores ranged from 31-34). Higher pre-restoration IBI scores were unexpected because of the well-documented negative effects of channelization (Petersen et al. 1987, Gorman and Karr 1978), and previous research demonstrating that fish species diversity and density are lower in channelized reaches than in unchannelized portions of the same river (Hortle and Lake 1983, Edwards et al. 1984, Raborn and Schramm 2003).

It is possible that the high IBI scores in the pre-restoration channel of Nippersink Creek could be a function of the length of time that Nippersink Creek had been channelized. Scarnecchia (1988) found no difference in fish species diversity when comparing unchannelized reaches of a northern Iowa stream with reaches that had been channelized for roughly sixty years. Because Nippersink Creek was channelized approximately 50 years prior to restoration, it is possible that fish species had recolonized and adapted to channelized conditions as hypothesized by Scarnecchia (1988).

The process of stream restoration, like channelization, can be a disturbance with unknown effects to stream fish assemblages (Muotka et al. 2002). Because the present

study was initiated only four years after completion of the restoration project in August 2000, it is possible that the fish community did not have adequate time to respond to restored conditions before sampling began. Detenbeck et al. (1992) found that the type of disturbance had a strong effect on fish recovery rates, and stream fish communities may take as long as 5 -52 years to recover from press disturbances such as habitat enhancement or channelization. Lepori et al. (2005) suggest that a time period of 2 to 8 years could be sufficient for recovery if the restored area is surrounded by lakes, pools, or rivers that could act as potential sources of fish species. This idea is supported by other river restoration work (Hortle and Lake 1983, Scarnecchia 1988, Muotka et al. 2002), and is a potential topic for future research on Nippersink Creek.

The lack of instream structure for fish can have a strong effect on the fish community in Nippersink Creek. Woody debris and snags are an important component of riverine habitats that provide fish with refugia from currents, increased food availability, and cover (Angermeier and Karr 1984). The presence of boulders results in increased heterogeneity of substrate, depth, cover, as well as current velocity variability (Van Zyll De Jong et al. 1997). Although channelized rivers and streams typically lack instream structures such as fallen trees and snags, these structures were present in the pre-restoration channel of Nippersink Creek (Zack, personal observation). In contrast, the restored sites were completely devoid of woody debris and snags (most likely due to the meadow/grassland riparian zone), and had few boulders that could provide cover for fish.

Van Zyll De Jong et al. (1997) found that salmonid populations increased in treatment streams with the addition of boulders and snags, and Angermeier and Karr

(1984) found that reduced woody debris caused a reduction in habitat complexity and a loss of deep pools, both required for gamefish to thrive. Wright and Flecker (2004) found that the loss of woody debris in pools resulted in fewer individuals and fewer species of fish than pools with woody debris. Angermeier and Karr (1984) reported that larger fish (age 2+ and older) avoid stretches of streams that lack instream woody debris. Talmadge et al. (2002) advised including boulder and woody debris as two of the six main habitat components during stream restoration. The presence of in-stream structure in the pre-restoration channel may be an explanation as to why IBI scores were higher than in the restored area where instream structure was absent.

Successful restoration of streams and the re-establishment of native fish assemblages must meet the habitat requirements of native fish species (Trexler 1995). Edwards et al. (1984) reported that fish species abundance in areas with artificial riffles was intermediate between that of natural and channelized areas, whereas Fuselier and Edds (1995, 1996) found that artificial riffles provide useful, natural habitat for nongame species, sometimes after just one year. The lack of agreement among studies may be due to the stage of restoration of a given stream. For example, Paller et al. (2000) suggested that rivers and streams in intermediate stages of recovery may be characterized by atypical fish assemblages and that the IBI was only sensitive to early stages of restoration and should not be used to assess intermediate restoration effects. Many stream studies have been incomplete in their efforts to determine the true effects of disturbances such as channelization and restoration, and the effects of restoration efforts on stream communities are understudied (Scarnecchia 1988, Muotka et al. 2002), so further

monitoring of Nippersink Creek will be the only way to determine if restoration has been a success.

Relatively undisturbed streams have more diverse habitat that supports more species, some of which are rare (Raborn and Schramm 2003, Cianfrani et al. 2009). Channelization often reduces fish species richness, but distributes the individuals of the remaining species more evenly (Raborn and Schramm 2003). As such, this reduction in richness and increase in evenness is a pattern that could be seen in any altered system, including a recently restored system. Results of this study indicate that fish species richness and diversity progressively increased downstream in the restored section and suggests that stream restoration efforts will, over time, be successful in Nippersink Creek.

Future Directions

The goal of restoration biology is to return an altered system to its original, pristine state, to the condition that existed prior to an anthropogenic disturbance, or to a condition similar to a nearby undisturbed area. In the case of river and stream restoration, this may involve adding meanders, increasing streambed heterogeneity, building artificial riffles and pools, planting riparian vegetation, and grading back steep stream banks. Restoration of altered systems is a fundamental necessity for long-term preservation of native biotic diversity, and is increasing in use as a conservation strategy. To evaluate the success of a restoration effort, data from the restored system collected prior to disturbance, or collected from a reference site at any time, must be compared to data collected during or after the recovery process. This is, however, one of the imperfections of restoration ecology: the endpoint for success is user-defined and often

not explicitly stated in many restoration projects. Most natural resource managers have an explicit idea of the result they would like to obtain in a restoration program, e.g., the return of native fish species to a restored habitat, but the target condition may or may not be the pristine state. The question then remains as to whether or not restoration reveals the true potential in systems that have been subject to anthropogenic disturbances.

The restoration effort of Nippersink Creek was successful, and continued monitoring will elucidate whether Nippersink Creek continues to improve. Restoration and rehabilitation of aquatic systems in general is a worthwhile pursuit, even if only for future conservation of biodiversity. From an aesthetic standpoint, many restoration projects serve to improve the appearance of aquatic systems. In the case of Nippersink Creek, the addition of meanders and grading back of steep stream banks transformed sections of the stream into a system much like what existed prior to the advent of agriculture in the watershed. However, measuring the success of a restoration project is difficult, because the definition of success is dependent upon the *a priori* goals. Because of the restoration efforts of MCCD, the entirety of Glacial Park may return to the state that existed prior to any anthropogenic disturbance.

The foundation of this study was based on the idea that fish assemblages can be used to measure restoration success because they are influenced by stream habitat (Paller et al. 2000). To that end, the results of this study indicate that restoration efforts have improved Nippersink Creek, as demonstrated by IBI scores and species richness values that are as high or higher than natural areas upstream and downstream of Glacial Park. Further, this study determined that upstream natural sites, found in an area primarily used

for agriculture, are possibly unaffected by the increased siltation and removal of riparian vegetation that are often detrimental consequences of agricultural land-use. This was demonstrated by the dominance of *Campostoma anomalum*, an algivorous, silt-intolerant cyprinid at upstream sites, which further highlights the importance of algal substrate cover and by association, riparian zone composition (or amount of shade and sunlight), to fish assemblages in Nippersink Creek. In addition, the lack of instream woody debris may be hindering further improvement in Nippersink Creek, as demonstrated by the high IBI scores of the pre-restoration channel compared to restored sites. However, because Nippersink Creek was channelized for roughly 50 years, time may be the major factor determining the recovery of Nippersink Creek now that progress has been seen.

Annual monitoring of fish in Nippersink Creek is recommended to measure additional progress as the stream continues to recover from the disturbance of the restoration efforts. However, there are also a variety of other areas of study that would lend further insight into not only restoration effort success, but Nippersink Creek as an important source of biotic diversity. This study dealt with adult and late-stage juvenile fish because they are useful in indices that determine restoration success. An examination of larval fish in Nippersink Creek would provide information on how Nippersink Creek is used by adult fish for spawning purposes, particularly given the proximity of lakes both upstream and downstream of Glacial Park. These data also could help in future restoration efforts, as spawning requirements are well-known for most fish species. The number of gamefish captured in the current study was limited. A future study examining sport fish in Nippersink Creek could benefit not only local fishers, but

also demonstrate how Nippersink Creek is used by both lake and river fish species. This is particularly important given the proximity to upstream and downstream lakes. A study of the distribution of benthic macroinvertebrates could help explain the pattern of fish species presence/absence found in the current study, and in particular the fish species found in Nippersink Creek that are invertebrate specialists. Data collected from these additional studies could be used to monitor and guide the continued recovery of Nippersink Creek.

CONCLUSIONS

The original goals of the Nippersink Creek restoration project involved more than simply improving fish habitat. The goals included reducing bank erosion and stream velocity, as well as improving the surrounding wetlands (Woodson 2000). McHenry County Conservation District (MCCD) hopes that this project will not only restore habitat for fish, mussels, birds, and plants, but will also help to improve recreational activities for visitors using Glacial Park (Woodson 2000).

One of the ways MCCD planned to measure the success of this project was to use the Index of Biotic Integrity (IBI) to monitor the fish community of Nippersink Creek (Woodson 2000). The results of this study found that IBI scores rate Nippersink Creek as limited to moderate aquatic resource, which was a lower rating than scores from historical MCCD collections in Nippersink Creek and the pre-restoration channel. However, IBI scores from the restored section of Nippersink Creek were slightly higher than those from natural areas located upstream and downstream of Glacial Park, indicating that the restored section is as healthy as or healthier than the upstream and downstream natural areas. A variety of factors could have contributed to the lower overall scores found in this study, including, but not limited to sampling method, time since channelization and restoration, and a lack of instream cover in the restored section. Further monitoring of the fish community is necessary to determine whether this downward shift in IBI scores is not a trend.

Due to the possible inefficiency of the sampling method used for generating IBI scores, the results of this study would be best viewed as part of a long-term monitoring effort. In addition, because winter sampling was sporadic due to ice and snow cover, future work should focus on summer and early fall collections. Use of a more effective sampling technique (i.e., backpack electroshocking, electric seining, etc.) during an intensive summer/fall collection period should be sufficient to effectively monitor the fish community of Nippersink Creek in the future.

This study found that percent silt, gravel, and algae substrate cover were the most important factors shaping the fish community of Nippersink Creek. Various other habitat factors that were not sampled (i.e., current velocity, depth, suspended sediment) could be have a strong effect on the fish community, and future work should include a larger suite of habitat variables. Although the results of this study are valuable in assessing restoration efforts in Nippersink Creek, the omission of key habitat factors suggests that future work is necessary to fully understand the habitat factors shaping fish communities in this Midwestern stream. Such studies are imperative to successful, long-term restoration of the stream and re-establishment of native fishes.

REFERENCES

- Andrade, P.C.N. 2006. The use of benthic macroinvertebrate functional feeding groups to assess stream ecosystem attributes. M.S. Thesis, Loyola University Chicago. 70 pp.
- Angermeier, P.L., and J.R. Karr. 1984. Relationships between woody debris and fish habitat in a small warmwater stream. *Transactions of the American Fisheries Society* 113:716-726.
- Angermeier, P.L. and I.J. Schlosser. 1987. Assessing biotic integrity of the fish community in a small Illinois stream. *North American Journal of Fisheries Management* 7:331-338.
- Angermeier, P.L. and I.J. Schlosser. 1989. Species-area relationships for stream fishes. *Ecology* 70(5):1450-1462.
- Arunachalam, M. 2000. Assemblage structure of stream fishes in the Western Ghats (India). *Hydrobiologia* 430:1-31.
- Barbour, M.T., J. Gerritsen, B.D. Snyder, and J.B. Stribling. 1999. *Rapid Bioassessment Protocols for Use in Streams and Wadeable Rivers: Periphyton, Benthic Macroinvertebrates, and Fish*, Second Edition. EPA 841-B-99-002. U.S. Environmental Protection Agency; Office of Water; Washington, D.C.
- Becker, G.C. 1983. *Fishes of Wisconsin*. University of Wisconsin Press. Madison, WI.
- Benke, A.C., R. L. Henry, D. M. Gillespie, and R.J. Hunter. 1985. Importance of snag habitat for animal production in southeastern streams. *Fisheries* 10:8-13.
- Berkman, H.E. and C.F. Rabeni. 1987. Effect of siltation on stream fish communities. *Environmental Biology of Fishes* 18:285-294.
- Cianfrani, C.M., S.M.P. Sullivan, W. C. Hession, and M.C. Watzin. 2009. Mixed stream channel morphologies: implications for fish community diversity. *Aquatic Conservation: Marine and Freshwater Ecosystems* 19:147-156.

- Clarke, K.R. and M. Ainsworth. 1993. A method of linking multivariate community structure to environmental variables. *Marine Ecology Progress Series* 92:205-219.
- Clarke, K.R. and R.N. Gorley. 2001. PRIMER v5 User Manual/Tutorial. PRIMER-E. Plymouth, U.K.
- Clarke, K.R. and R.M. Warwick. 2001. Change in marine communities: an approach to statistical analysis and interpretation, 2nd edition. PRIMER-E. Plymouth, U.K.
- Detenbeck, N.E., P.W. DeVore, G.J. Niemi, and A. Lima. 1992. Recovery of temperate-stream fish communities from disturbance: a review of case studies and synthesis of theory. *Environmental Management* 16:33-53.
- Diana, M., J.D. Allen, and D. Infante. 2006. The influence of physical habitat and land use on stream fish assemblages in southern Michigan. *American Fisheries Society Symposium* 48:359-374.
- Edwards, C.J., B.L. Griswold, R.A. Tubb, E.C. Weber, and L.C. Woods. 1984. Mitigating effects of artificial riffles and pools on the fauna of a channelized warmwater stream. *North American Journal of Fisheries Management* 4: 194-203.
- Fausch, K.D., J.R. Karr, and P.R. Yant. 1984. Regional application of an index of biotic integrity based on stream fish communities. *Transactions of the American Fisheries Society* 113: 39-55.
- Feyrer, F. and M.P. Healey. 2003. Fish community structure and environmental correlates in the highly altered southern Sacramento-San Joaquin Delta. *Environmental Biology of Fishes* 66:123-132.
- Fitzgerald, D.G., E. Kott, R.P. Lanno, and D.G. Dixon. 1998. A quarter century of change in the fish communities of three small streams modified by anthropogenic activities. *Journal of Aquatic Ecosystem Stress and Recovery* 6:111-127.
- Fore, J.D., D.C. Dauwalter, and W.L. Fisher. 2007. Microhabitat use by smallmouth bass in an Ozark stream. *Journal of Freshwater Ecology* 22:189-199.
- Fuselier, L. and D. Edds. 1995. An artificial riffle as restored habitat for the threatened Neosho Madtom. *North American Journal of Fisheries Management* 15: 499-503.
- Fuselier, L. and D. Edds. 1996. Seasonal variation of riffle and pool fish assemblages in a short mitigated stream reach. *The Southwestern Naturalist* 41:299-306.

- Gelwick, F.P. and W.J. Matthews. 1992. Effects of an algivorous minnow on temperate stream ecosystem properties. *Ecology* 73:1630-1645.
- Gorman, O.T., and J.R. Karr. 1978. Habitat structure and stream fish communities. *Ecology* 59:507-515.
- Hortle, K.G. and P.S. Lake. 1983. Fish of channelized and unchannelized sections of the Bunyip River, Victoria. *Australian Journal of Marine and Freshwater Resources* 34: 441-450.
- Illinois Environmental Protection Agency (IEPA). 1989. Biological Stream Characterization (BSC): A Biological Assessment of Illinois Stream Quality. Special Report No. 13 of the Illinois State Water Plan Task Force. IEPA Division of Water Pollution Control. Springfield, IL
- Iwata, T., S. Nakano, and M. Inoue. 2003. Impacts of past riparian deforestation on stream communities in a tropical rain forest in Borneo. *Ecological Applications* 13:461-473.
- Johnston, C.E., and M.J. Maceina. 2009. Fish assemblage shifts and species declines in Alabama, USA streams. *Ecology of Freshwater Fish* 18:33-40.
- Karr, J.R. 1981. Assessment of biotic integrity using fish communities. *Fisheries* 6:21-27.
- Karr, J.R., K.D. Fausch, P.L. Angermeier, P.R. Yant, and I.J. Schlosser. 1986. Assessing Biological Integrity in Running Waters: A Method and Its Rationale. Illinois Natural History Survey Special Publication 5. Illinois Natural History Survey. Champaign, Illinois.
- Karr, J.R. 1991. Biological Integrity: A long neglected aspect of water resource management. *Ecological Applications* 1:66-84.
- Koehn, J.D., N.A. O'Connor, and P.D. Jackson. 1994. Seasonal and size-related variation in microhabitat use by a southern Victorian stream assemblage. *Australian Journal of Marine and Freshwater Research* 45:1353-1366.
- Lepori, F., D. Palm, E. Brannas, and B. Malmqvist. 2005. Does restoration of structural heterogeneity in streams enhance fish and macroinvertebrate diversity? *Ecological Applications* 15:2060-2071.
- Moyle, P.B. and D.M. Baltz. 1985. Microhabitat use by an assemblage of California stream fishes: developing criteria for instream flow determinations. *Transactions of the American Fisheries Society* 114:695-704.

- Mueller Jr., R. and M. Pyron. 2009. Substrate and current velocity preferences of spotfin shiner (*Cyprinella spiloptera*) and sand shiner (*Notropis stramineus*) in artificial streams. *Journal of Freshwater Ecology* 24: 239-245
- Muotka, T., R. Pavola, A. Haapapa, M. Novikmec, and P. Laasonen. 2002. Long-term recovery of stream habitat structure and benthic invertebrate communities from in-stream restoration. *Biological Conservation* 105: 243-253.
- Murphy, M.L., C.P. Hawkins, and N.H. Anderson. 1981. Effects of canopy modification and accumulated sediment on stream communities. *Transactions of the American Fisheries Society* 110:469-478.
- Nelson, J.S., E.J. Crossman, H. Espinosa-Pérez, L.T. Findley, C.R. Gilbert, R.N. Lea, and J.D. Williams. 2004. *Common and Scientific Names of Fishes from the United States, Canada, and Mexico, Sixth Edition*. American Fisheries Society, Special Publication 29. Bethesda, MD.
- Onorato, D.P., R.A. Angus, and K.R. Marion. 1998. Comparison of a small-mesh seine and a backpack electroshocker for evaluating fish populations in a North-Central Alabama stream. *North American Journal of Fisheries Management* 18: 361-373.
- Paller, M.H., M.J.M. Reichert, and J.M. Dean. 1996. Use of fish communities to assess environmental impacts in South Carolina coastal plain streams. *Transactions of the American Fisheries Society* 125:633-644.
- Paller, M.H., M.J.M. Reichert, J.M. Dean, and J.C. Seigle. 2000. Use of fish community data to evaluate restoration success of a riparian stream. *Ecological Engineering* 15:S171-S187.
- Petersen Jr., R.C., B.L. Madsen, M.A. Wilzbach, C.D. Magadza, A. Paarlberg, A. Kullberg, and K.W. Cummins. 1987. Stream management: emerging global similarities. *Ambio* 16:166-179.
- Pflieger, W.L. 1997. *The Fishes of Missouri*. Missouri Department of Conservation. Jefferson City, MO.
- Power, M.E., W.J. Matthews, and A.J. Stewart. 1985. Grazing minnows, piscivorous bass, and stream algae: dynamics of a strong interaction. *Ecology* 66:1448-1456.
- Raborn, S.W., and H.L. Schramm Jr. 2003. Fish assemblage response to recent mitigation of a channelized warmwater stream. *River Research and Applications* 19:289-301.

- Robertson, M.S., and K.O. Winemiller. 2003. Habitat associations of fishes in the Devils River, Texas. *Journal of Freshwater Ecology* 18:115-126.
- Scarnecchia, D.L. 1988. The importance of streamlining in influencing fish community structure in channelized and unchannelized reaches of a prairie stream. *Regulated Rivers: Research and Management* 2:155-166.
- Schlosser, I.J. 1982. Fish community structure and function along two habitat gradients in a headwater stream. *Ecological Monographs* 52:395-414.
- Schlosser, I.J. 1985. Flow regime, juvenile abundance, and the assemblage structure of stream fishes. *Ecology* 66:1484-1490.
- Shore, D. 2001. Born again river: Bringing the Nippersink back to nature, backhoe-style. *Chicago Wilderness Magazine*. Available on the web at: <http://chicagowildernessmag.org/issues/winter2001/toc.html>.
- Simonson, T.D., J. Lyons, and P.D. Kanehl. 1993. Guidelines for Evaluating Fish Habitat in Wisconsin Streams. Gen. Tech. Rep. NC-164. St. Paul, MN: U.S. Department of Agriculture, Forest Service, North Central Forest Experiment Station. 36 pgs.
- Smith, P.W. 1979. *The Fishes of Illinois*. University of Illinois Press. Urbana, IL.
- Snyder, C.D., J.A. Young, R. Vilella, and D.P. Lemarié. 2003. Influences of upland and riparian land use patterns on stream biotic integrity. *Landscape Ecology* 18:647-664.
- Stauffer, J.C., R.M. Goldstein, and R.M. Newman. 2000. Relationship of wooded riparian zones and runoff potential to fish community composition in agricultural streams. *Canadian Journal of Fisheries and Aquatic Sciences* 57:307-316.
- Sullivan, B.E., L.S. Rigsby, A. Berndt, M. Jones-Wuellner, T.P. Simon, T. Lauer, and M. Pyron. 2004. Habitat influence on fish community assemblage in an agricultural landscape in four central Indiana streams. *Journal of Freshwater Ecology* 19:141-148.
- Talmadge, P.J., J.A. Perry, and R.M. Goldstein. 2002. Relation of instream habitat and physical conditions to fish communities of agricultural streams in the northern Midwest. *North American Journal of Fisheries Management* 22:825-833.
- Trexler, J.C. 1995. Restoration of the Kissimmee River: a conceptual model of past and present fish communities and its consequences for evaluating restoration success. *Restoration Ecology* 3:195-210.

- Van Zyll De Jong, M.C., I.G. Cowx, and D.A. Scruton. 1997. An evaluation of instream habitat restoration techniques on salmonid populations in a Newfoundland stream. *Regulated Rivers: Research & Management* 13:603-614.
- Walser, C.A. and H.L. Bart. 1999. Influence of agriculture on in-stream habitat and fish community structure in Piedmont watersheds of the Chattahoochee River System. *Ecology of Freshwater Fish* 8:237-246.
- Wang, L., J. Lyons, P. Kanehl, R. Bannerman, and E. Emmons. 2000. Watershed urbanization and changes in fish communities in southeastern Wisconsin streams. *Journal of the American Water Resources Association* 36:1173-1189.
- Wood, B.M., and M.B. Bain. 1995. Morphology and microhabitat use in stream fish. *Canadian Journal of Fisheries and Aquatic Science* 52:1487-1498.
- Woodson, B. 2000. The Nippersink re-meandering project. McHenry County Conservation District. Woodstock, IL.
- Wright, J.P. and A.S. Flecker. 2004. Deforesting the riverscape: the effects of wood on fish diversity in a Venezuelan piedmont stream. *Biological Conservation* 120:439-447.

VITA

Sarah A. Zack graduated from University of Wisconsin-Madison in 2001 with Bachelor of Science degrees in Zoology and Biological Aspects of Conservation.

Following a position at the Wisconsin Department of Natural Resources, where she first started working in streams and rivers, Sarah decided to pursue a Master of Science degree in Biology at Loyola University Chicago, focusing on aquatic ecology. She is currently working as a Teaching Instructor at Loyola University Chicago and as a Scientist and Lab Manager at EA Engineering, Science, and Technology, an environmental consulting firm in the Chicagoland area.

