ASSESSMENT OF POTENTIAL AQUATIC HABITAT RESTORATION SITES IN THE BUFFALO RIVER AREA OF CONCERN

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EXECUTIVE SUMMARY

- The remedial action process for the Buffalo River Area of Concern (AOC) is at the stage that aquatic habitat restoration projects (including removal of contaminated sediment) are being considered. The objective of this study was to document the biological, water quality, and use characteristics of 10 promising habitat restoration sites located between Michigan Avenue and the river's confluence with Cazenovia Creek. All habitat restoration sites are located in shallow water areas near the shoreline and outside of the designated navigable channel. As part of the evaluation, the study developed a characterization matrix for each of the 10 candidate sites. The matrix was designed to serve as guidance for stakeholders and decision makers, allowing them to quickly review comprehensive assessments of the potential for effective habitat remediation.

- Larval fishes were sampled at the 10 study sites in June and August of 2003 and 2004 (four surveys total). Larval fish were collected at each site using two 0.5 m plankton nets with 560 μm mesh. The nets were towed at a speed of approximately 50 cm per second for 15 min in a circular pattern (shore to shore, but within the dredged channel of the river). One net was towed near the surface (depth of 1.0 – 2.5 m) and one closer to the river bottom (depth of 2.5 – 6.5 m).

- Sampling for juvenile and adult fish was carried out in June and August of 2003 and 2004 (four surveys total). Buffalo State College's 18' electrofishing boat, equipped with a Smith-Root type VI-A electrofishing unit, was used for each survey. At each site a single pass was made along both shorelines for a total of 300 seconds per site. Pulsed direct current was used at a pulse rate of 30-60 pps; output was maintained at approximately 3,000 watts for each survey.

- Benthos was sampled using a Ponar dredge at the 10 habitat restoration sites and at six sites within the dredged navigation channel. Samples were collected three times between mid-June and the end of October, 2003 and twice between the end of June and end of September, 2004.

- Presence/absence vegetation surveys were conducted at the 10 habitat sites in August, 2004 and 2005, while percentage of overhanging shoreline cover was estimated both from field observation and detailed digital satellite imagery.

- Water quality was evaluated principally through the use of Hydrolab Datasonde 4a's to measure dissolved oxygen, pH, turbidity, temperature, and conductivity. Hydrolabs were installed at three sites (two sites near the top of the AOC and Ohio St. Bridge) to continuously monitor these parameters from June through September of 2003 and 2004. Suspended sediment samples were collected once per week at the three Hydrolab sites. In addition, a Hydrolab Datasonde 4a was used to measure the same analytes at all 10 habitat sites, at three depths, 0.5 m below the surface; 1.0 m below the surface; and near the bed. This profiling was done once per week for 16 weeks in 2003 and 17 weeks in 2004. Finally, sampling was done for *Escherichia coli* analysis during a major runoff event and three dry days in September, 2004.

- A recreational use survey of the habitat sites and 15 other sites along the AOC was conducted by boat for a total of 73 days in 2003-04. The surveys were done during randomly selected time slots (7-9 am; 9am-12 pm; 12 pm-3 pm; 3-6 pm) on randomly selected days of the week.

- The larval fish sampling showed similar species diversity and abundance in 2003-2004 as compared to 1993 (8-10 species found). No site-specific trends were observed.The adult/juvenile fish sampling showed similar species diversity and abundance in 2003-2004 compared to 1993 (15-20 species across all sites). Lowest species diversity occurred at sites 1, 2, 5, and 10.

- DELT anomalies: varied greatly among species, with a low of 14% in pumpkinseed to a high of 87% in brown bullhead. For the river as a whole, DELT scores averaged 37%, which is much higher than what would be expected for a moderately impacted (2-5%) or unimpacted (<2%) river.

- Index of Biotic Integrity (IBI): Low site-specific species diversity and high DELT scores contributed to low IBI scores. Seven sites (#3, 4, 5, 7, 8, 9, and 10) would be rated "poor" and three (#1, 2, and 6) "very poor" using standard IBI criteria.

- Overall: Based on species diversity, IBI, and DELT scores, sites 3, 4, 7, and 8 tended to score higher in terms of fish community health while sites 1, 2, 5, 6, 9, and 10 tended to score lower.

- The Buffalo River AOC continues to be dominated by a low diversity benthic invertebrate community that is broadly tolerant of pollution and environmental degradation. High densities of tubificid oligochaeates (though lower than historical maxima), and their numerical dominance of the benthos, suggest poor environmental health. Oligochaete densities were higher in the channel than at shoreline habitat restoration sites. Fewer invertebrate families were collected in this study than in the early 1990's, possibly even indicating some reversal of biotic recovery. Substantially more families occurred at shoreline sites than in the channel, although the habitat restoration sites were still dominated by pollution-tolerant oligochaetes and chironomids. Likewise, chironomid taxonomic richness was markedly higher at habitat restoration sites than in the channel, but samples largely constituted pollution-tolerant species and genera. Chironomid mouthpart deformities remain very high at channel sites (as they were in 1990-93), but, interestingly, all of the rather limited number of larvae from shoreline sites had developed normally.

- More than 50 plant species were collected from the Buffalo River shoreline and herbaceous vegetation was well-developed at all sites. The 10 potential restoration sites differed considerably in their development of overhanging cover, ranging from 0 to 80%. Submerged macrophyte beds are not extensive, but are present at most sites. The presence of invasive plant species, including tree-of-heaven, Japanese knotweed, purple loosestrife, and submerged Eurasian watermilfoil degrades many of the sites and should be subject to eradication campaigns as part of habitat restoration efforts.

- Dissolved oxygen levels frequently were below state guidelines within the dredged portion of the AOC (representing all habitat sites except Site 1), while levels upstream of the dredged channel more frequently were above state guidelines. The low dissolved oxygen levels appear related to a combination of thermal stratification, system hydraulics, high sediment oxygen demand, and background biochemical oxygen demand. At the habitat sites, dissolved oxygen tended to be lower near the riverbed and higher near the surface. During dry periods, turbidity was relatively low (<20 NTU) in the upper 1m at all habitat sites, increasing to about 20-100 NTU near the bed. Turbidity increased during storm events, occasionally reaching values of 1,000 NTU. The levels of *E. coli* were high during the sampled storm event (up to 38,700 m.o./100 mL) and lower (50-2,200 m.o./100 mL) during dry periods. These results were consistent with past studies and re-emphasize the importance of the upper watershed as a source of bacteria.

- A total of 887 person-days of activity were observed through the recreational use survey. Fishing, boating, and "hanging out" in riparian areas were the most frequently observed activities (27%, 28%, and 22% of all activity, respectively). Swimming represented 3% of the observed activities. The observed level of 887 person-days underestimates actual activity because it only represents a three hour segment of each sample date. Adjusting the sampled person-day activity to reflect all daylight hours for the entire week, it is estimated that actual activity may have been on the order of 12,784 person-days in 2003-04. There was spatial variability in the frequency of activity, with habitat sites 3, 4, 5, 6, 8, and 10 having the lowest level of activity of all survey sites (≤8 person-days (unadjusted value) over the two year period).

- The site evaluation matrix was developed using an index approach for various biotic and abiotic categories. The benthic indices included the number of benthic families, oligochaete density, and the product of chironomid biotic score and number of chironomid taxa. The fish indices included species diversity, Index of Biotic Integrity, and DELT (Deformities, Eroded fins, Lesions, and Tumors). The vegetation indices were shading (% overhang) and macrophyte species diversity. The water quality indices were the National Sanitation Foundation Water Quality Index (dissolved oxygen, pH, turbidity) and the Canadian Council of Ministers of the Environment Water Quality Index (dissolved oxygen only). No single site scored consistently high in all indices. Based strictly on the aggregate matrix scores, habitat sites 4, 7, and 8 had the best biological/water quality health while sites 2, 5, 6, and 10 scored lower. Interestingly, based on a recent Corps of

Engineers study, habitat sites 7 and 8 had PAH values in sediment that exceeded probable effect level for benthic organisms.

- Ecological integrity, as reflected by biota and water quality, certainly has improved in the Buffalo River AOC, as compared to 1970's conditions. However, there does not appear to be any improvement since the early 1990's. Habitat restoration measures such as improved overhang cover, macrophyte plantings, eradication of exotic plant species, removal of old dock pilings, naturalization of shorelines, or removal of contaminated bed sediment could improve ecological integrity at selected sites. Constraints on ecological integrity that may prove more challenging to overcome include warmer water temperatures and low dissolved oxygen levels.

- Chapter 8 has been prepared by Buffalo Niagara Riverkeeper based on our own interpretation of the data reported by Buffalo State College and Youngstown State University, and is therefore outside of the Buffalo River Remedial Advisory Committee recommendations of required actions. Riverkeeper strongly supports the findings of the water quality, benthic, fishery, and vegetation analysis. Riverkeeper suggests a continuation of river usage surveys into the future in combination with a market analysis of the river corridor. Riverkeeper strongly supports the ranking and evaluation system that was created for the "Site Characterization Matrix," though Riverkeeper wants to emphasize that the ranking system is just one of many tools available to decision-makers when prioritizing sites for restoration.

- The next steps for the data generated from this study include: the application of the results in the USACE's Environmental Dredging Feasibility Study, and the use of the data during the development of the updated Buffalo River Remedial Strategy and Delisting Criteria/Restoration Targets. Riverkeeper will coordinate an effort to fully investigate sites 5 and 6 regarding its unexplained poor ratings and high deformities. In addition, Riverkeeper will coordinate with the local efforts dedicated to Inner Harbor revitalization in terms of obtaining additional user surveys and a market analysis of the AOC in the near future.

- Buffalo River stakeholders will use this site matrix to prioritize restoration efforts and to identify possible funding sources, generate local community support, and coordinate partnerships for the implementation of recommended remedial actions-as identified by the Buffalo River Remedial Advisory Committee.

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CHAPTER 1
INTRODUCTION

K.N. Irvine

1.1 Background to Study

The Buffalo River is a recovering riparian system. Its history of heavy industrial discharge resulted in poor water quality and badly contaminated sediments. The river was considered biologically dead as recently as the early 1970's (Buffalo Courier Express, 1974) and it was designated as a Great Lakes Area of Concern (AOC) in the mid-1980's (New York State Department of Environmental Conservation (NYSDEC), 1989). Impaired beneficial uses include degradation of benthos, fish tumors, loss of fish and wildlife habitat, degradation of fish and wildlife populations, the tainting of fish and wildlife flavor, and the presence of bird or animal deformities or reproductive problems. Combined sewer overflows and upstream pollutant inputs remain concerns, but historical sediment contamination and poor habitat opportunities persist as the major obstacles to recovery.

This project is part of a unique collaborative effort to plan and fund sediment remediation for the restoration of aquatic habitat within the AOC. The Buffalo District U.S. Army Corps of Engineers (USACE) (2003) recently completed its Sediment Reconnaissance Study of the Buffalo River, a first step in determining whether targeted environmental dredging would improve aquatic conditions. Subsequently, the USACE (2004) agreed to pursue a full Feasibility Study of sediment cleanup options. The Feasibility Study will be conducted over the period 2005-2008 and is subject to a 50% non-federal match. The Buffalo River Partnership, organized by the Friends of the Buffalo Niagara Rivers (now Buffalo Niagara Riverkeeper) and other nonprofit organizations, has taken the lead in raising the non-federal matching funds, and in particular, the NYSDEC will do an intensive sampling and analysis of bed sediment in select areas of the river in 2005-06.

While signs of biological recovery have been documented within the AOC (e.g. Buffalo Courier Express, 1974; Diggins and Snyder, 2003), much of the shoreline's natural cover and vegetation has been removed and bank slopes have been altered or eliminated. Habitat restoration efforts could greatly speed recovery. Therefore, the objective of this study was to document the biological, water quality, and use characteristics of 10 promising habitat restoration sites located between Michigan Avenue and the river's confluence with Cazenovia Creek. These data are valuable in their own right as a direct measure and update of biotic and limnological information for the river. The study also developed a characterization matrix for each of the 10 candidate sites. This matrix was designed to serve as guidance for stakeholders and decision makers, allowing them to quickly view comprehensive assessments of the potential for effective habitat restoration (including the need for sediment removal and/or immobilization).

1.2 The Buffalo River Watershed and Area of Concern

The Buffalo River drains an area of 1,155 km^2 (447 mi^2) and Cayuga, Buffalo, and Cazenovia creeks are the three major tributaries within the watershed (Figure 1.1). The Buffalo River watershed occupies two physiographic regions. The northern and western portion of the watershed is within the Erie-Ontario Lake Plain Province, while the southern part of the watershed is within the Alleghany Plateau Province. The Erie-Ontario Province formerly was a glacial lake bed and therefore has limited relief. The watershed consists primarily of 21 different soil series, but the majority of soils texturally are a silt loam (U.S. Department of Agriculture, 1986). The slopes of these soil units range between nearly level and 0.50, while the drainage classification ranges from very poorly drained to excessively drained (U.S. Department of Agriculture, 1986).

The climate of the Buffalo area is classified under the Koppen system as humid continental with a mild summer (Dfb) (Gabler et al., 1997). Annual total precipitation at the Buffalo Airport averages 98 cm (38.6 in), with February being the driest month (5.9 cm (2.32 in) of precipitation) and August being the wettest month (10.6 cm (4.17 in) of precipitation). The lowest monthly mean flow recorded at U.S. Geological Survey (USGS) gauge stations on each of the tributaries (Figure 1.1) typically occurs in July and August when evapotranspiration is highest (Cayuga Cr. - 0.70 m^3s^{-1} (24.6 cfs); Buffalo Cr. - 1.30 m^3s^{-1} (45.8 cfs); and Cazenovia Cr. - 1.35 m^3 s^{-1} (47.8 cfs)). Highest monthly mean flow on the three tributaries typically occurs in March (Cayuga Cr. – 9.68 m^3s^{-1} (342 cfs); Buffalo Cr. – 14.0 m^3s^{-1} (495 cfs); and Cazenovia Cr. – 15.6 m^3s^{-1} (551 cfs)) as the result of snowmelt and spring rainfall.

Simple summation of the daily mean flow at the three gauge stations cannot provide an accurate approximation of the inflow to the top of the AOC (Figure 1.1) because the USGS gauges do not represent the entire contributing area for the three tributaries. Meredith and Rumer (1987) used a proportional-area approximation to account for the ungauged portions of the watershed in calculating inflow to the top of the AOC:

$$Q_T = Q_G \bullet \left(\frac{A_T}{A_G} \right) \qquad [1.1]$$

where:
Q_T = daily flow from the tributary to the top of the AOC (cfs or m^3s^{-1})
Q_G = daily flow at the gauge on the tributary (cfs or m^3s^{-1})
A_T = total drainage area at the mouth of the tributary (mi^2 or km^2)
A_G = drainage area upstream of the gauge (mi^2 or km^2)

The drainage areas upstream of the Buffalo, Cayuga, and Cazenovia creek gauges are 144, 94.9, and 134 mi^2 (372.1, 245.2, 346.2 km^2), respectively. The total drainage areas for Buffalo, Cayuga, and Cazenovia creeks are 146.2, 124.4, and 135.4 mi^2 (377.8, 321.4, 349.9 km^2), respectively (Meredith and Rumer, 1987). The flow adjustment approach represented by equation 1.1 was used in this study to estimate daily mean flow to the top of the AOC.

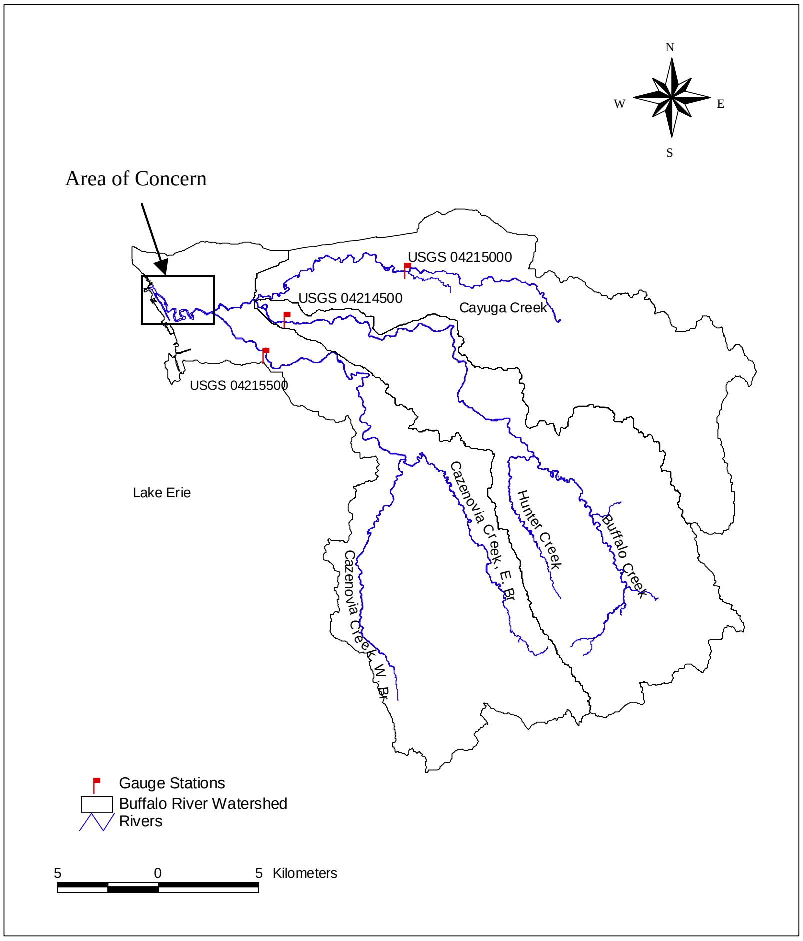

Figure 1.1 Buffalo River Watershed and USGS gauging stations

Land use within the watershed varies. Much of the upper portion of the watershed is characterized by woods and farmland, but prior to joining the Buffalo River the creeks also pass through several small communities and receive industrial, commercial, residential, and municipal discharges (Irvine and Pettibone, 1996). The lower Buffalo River historically has been highly urbanized and industrialized (Sauer, 1979; Rossi, 1995) and this appears to be principally why only the lower 9.6 km (6 mi) of the river was designated an AOC by the International Joint Commission. The change in the industrial composition of the AOC between 1929 and 1990 was documented and mapped by Irvine et al. (2003).

Much of the Buffalo River AOC is designated as a navigable channel and is maintained at a minimum depth of 7m (22 ft) by the Buffalo District USACE. This dredged reach is wider and deeper than the tributaries, but the bed slope is shallower. As a result of the changes in the hydraulic geometry, flow velocities within most of the AOC typically are less than those of the tributaries, producing local shoaling areas as sediment deposits. The Buffalo River Improvement Corporation (BRIC) was created in 1967 to supply industries along the Buffalo River with water for cooling and processing purposes. The water is pumped from Lake Erie and ultimately augments flows in the Buffalo River. The design operation of the BRIC system is 2.18 $m^3 s^{-1}$ (77 cfs) and during its early years of operation often contributed 90% of the total river flow in the drier summer months (Sauer, 1979). This flow augmentation helped to improve the water quality of the river at the time. As industry has declined along the river, so too has the BRIC pumping rate. Pumping rates in the early part of this decade averaged 0.66 $m^3 s^{-1}$ (23 cfs).

1.3 Habitat Assessment Sites and Study Approach

In consultation with the Friends of the Buffalo Niagara Rivers (now Buffalo Niagara Riverkeeper) and stakeholders (which included a boat tour of the AOC), and considering past research efforts (e.g. NYSDEC, 1993; Kozuchowski et al., 1994; Diggins and Stewart, 1998; Diggins and Snyder, 2003; U.S. Fish and Wildlife Service, undated; Irvine et al., 2003; 2005; 2005b) 10 promising habitat restoration sites were identified for study. The locations of these sites are shown in Figure 1.2 and a photo of each site is presented in Appendix 1.1. At each site, sampling was conducted for benthic organisms (identified to genus or species level and chironomid larvae deformity assessment), extent and species composition of submerged macrophytes, extent of shading, juvenile and adult fish species composition, conventional water quality parameters (pH, dissolved oxygen, temperature, turbidity, conductivity), anthropogenic use of the water (e.g. recreational activities), and adjacent property ownership. The specific approaches and results of the individual sampling efforts are presented in Chapters 2 through 6. The results subsequently are considered in a holistic way in Chapter 7 through the use of a site characterization matrix, with the intent of identifying the most promising sites for remediation work.

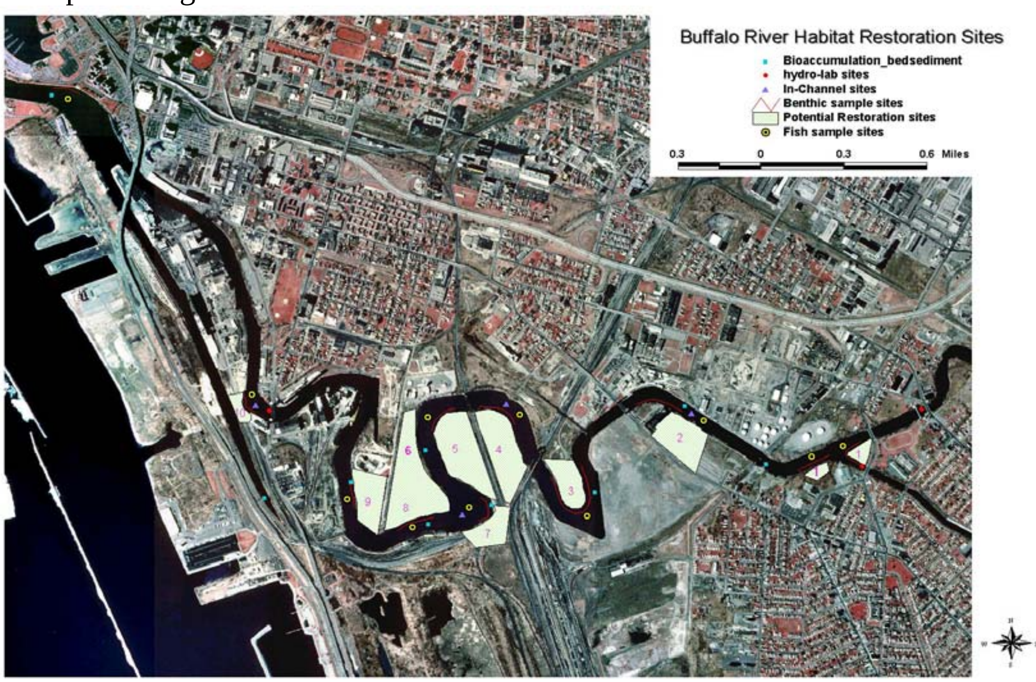

Figure 1.2 Study sample sites

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APPENDIX 1.1
HABITAT ASSESSMENT SITES

Site 1

Site 2

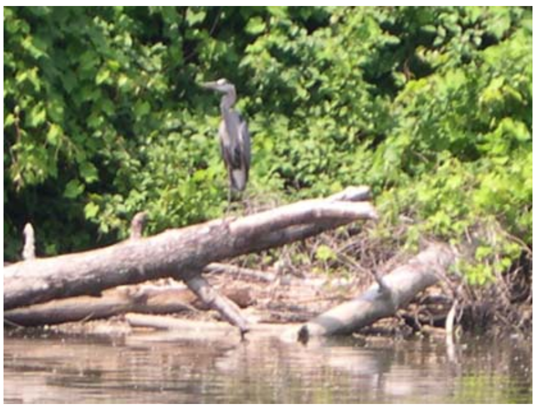
Site 3

Site 3 (Continued)

Site 4

Site 5

Site 6 **Site 6 (continued)**



Site 6 (continued) **Site 7**

Site 7 (Continued) **Site 8**

Site 8 (Continued)

Site 9

Site 9 (Continued)

Site 10

Site 10 (Continued)

CHAPTER 2
FISH SURVEY

R.J. Snyder

2.1 Introduction

Monitoring of biotic communities is an integral part of many aquatic habitat assessments. While chemical monitoring tends to provide a "snapshot" of conditions at the time of sampling, biological monitoring is based on the premise that biological communities are shaped by the long-term conditions of their environment and more accurately reflect the health of an ecosystem.

Fish have a number of advantages as indicators of ecosystem integrity. Fish are good indicators of long-term effects and broad habitat conditions because they are relatively long-lived and mobile (Karr et al. 1986). Typical fish assemblages represent a range of trophic levels and therefore reflect environmental health at a broad level, and moreover, fish are consumed by humans and are therefore critical in assessing impacts of environmental contamination. Finally, fish account for nearly half of the endangered vertebrate species and subspecies in the United States, making their monitoring and enhancement particularly important (Warren and Burr 1994).

To aid in identifying habitat sites for future rehabilitation, we examined fish community health at ten sites in the Buffalo River Area of Concern (AOC). We focused on measures of diversity and abundance, presence of DELT abnormalities, and Index of Biotic Integrity (IBI) scores to evaluate specific sites as well as the Buffalo River as a whole.

2.2 Larval Fishes

2.2.1 Methods

We assessed diversity of larval fishes at the ten study sites in June and August of 2003 and 2004 (four surveys total). Larval fish were collected at each study site using two 0.5 m plankton nets with 560 µm mesh. The nets were towed at a speed of approximately 50 cm per second for 15 min in a circular pattern (shore to shore, but within the dredged channel of the river). One net was towed near the surface (depth of 1.0 – 2.5 m) and one closer to the river bottom (depth of 2.5 – 6.5 m). Larval fish were fixed in 10% Wardsafe preservative and returned to the laboratory for identification and enumeration following Auer (1982). Results from shallow and deep tows at each site were combined for data summaries and analysis.

2.2.2 Results

In 2003-2004, a total of 10 species of larval fishes were collected across all study sites, which is similar to the larval diversity found in 1993 (Table 2.1). Abundance of larval fishes was highest in the June samples from both years and was much lower in the August samples (Figure 2.1). The large number of larval fishes collected at sites 8-10 in June 2004 may be due to fish moving into the river from nearshore lake habitats, since these sites are relatively close to the mouth of the river.

Table 2.1 Larval Fish Occurrences in the Buffalo River AOC (1993 and 2003-2004)

Species	1993	2003-2004
Alewife	X	X
Bluntnose minnow		X
Carp	X	X
Fathead minnow		X
Gizzard shad	X	X
Lepomis sp.	X	X
Logperch		X
Morone sp.	X	
Pomoxis sp.	X	X
Rainbow smelt	X	
Round goby		X
Yellow perch	X	X
Total # of species:	8	10

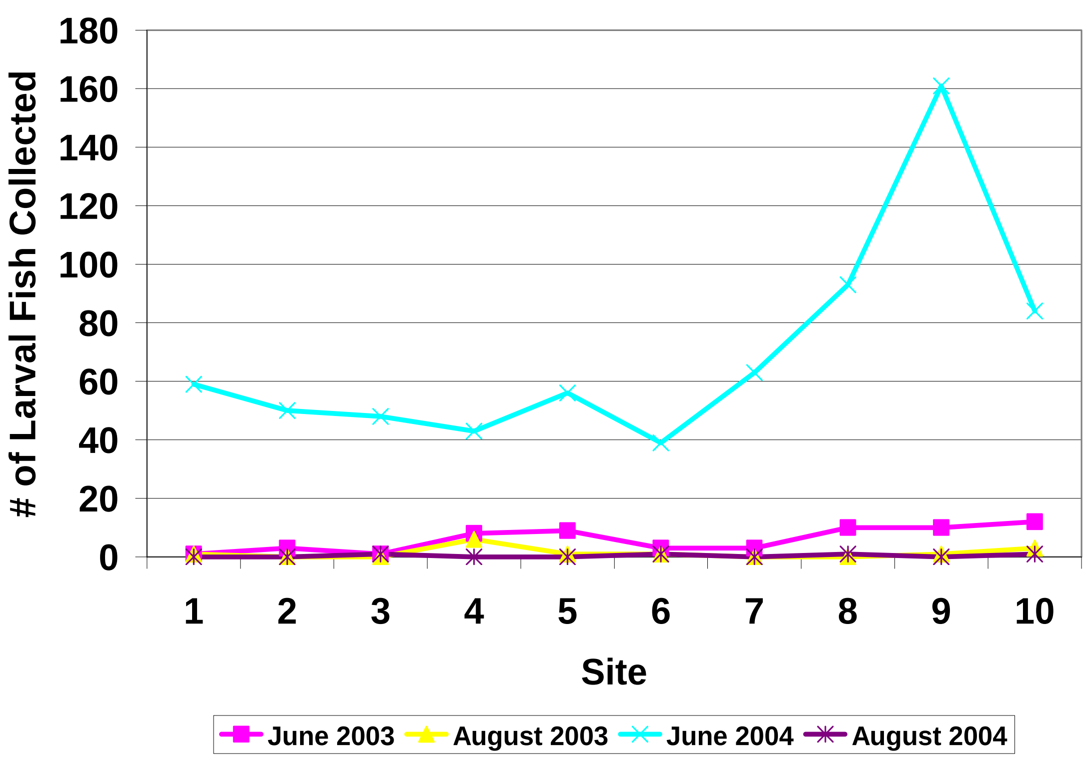


Figure 2.1 Number of larval fishes collected per site (2003-2004)

2.3 Juvenile and Adult Fishes

2.3.1 Methods

We assessed diversity of juvenile and adult fishes with electrofishing surveys, which were carried out in June and August of 2003 and 2004 (four surveys total). Buffalo State College's 18' electrofishing boat, equipped with a Smith-Root type VI-A electrofishing unit, was used for each survey. At each of the ten study sites, a single pass was made along both shorelines for a total of 300 seconds per site. Pulsed direct current was used at a pulse rate of 30-60 pps; output was maintained at approximately 3,000 watts for each survey.

All juvenile and adult fishes were temporarily immobilized by adding clove oil (dissolved in ethanol) to the aerated live well on board the electrofishing boat. Individual fish were then identified, measured for total length, and examined for DELT anomalies before being released.

2.3.2 Species Diversity

Diversity and species composition across all sites was similar in 2003 and 2004 (Table 2.2), ranging from 15-20 different species collected in the river on each sampling date. Species occurrences were similar in 2003-2004 compared to data collected in 1993 (Table 2.2), with a few exceptions. Carp x goldfish hybrids, common shiners, fathead minnows, hogsuckers, logperch, and rudd were taken in 2003-2004 but were not found in 1993. In contrast, rainbow trout, river chubs, white bass, white crappie, and white perch were collected in 1993 and were absent from sampling in 2003-2004 (Table 2.2). However, the data from 2003 and 2004 show significant variation in species occurrences from early to late season (i.e. from June to August) as well as from one year to the next, hence the differences in species composition from 1993 compared to 2003-2004 should be viewed cautiously.

To examine species diversity of juvenile and adult fishes per site, we calculated the average number of fish species occurring for the four sampling dates across both years (Figure 2.2). The average number of species occurring ranged from 3.5 – 7.5 per site. Sites 4, 8, and 9 had the highest average species diversity, while sites 1, 2, 5, and 10 had the lowest diversity (Figure 2.2).

Table 2.2 Juvenile and Adult Fish Occurrences From Electrofishing Surveys (1993 and 2003-2004)

Species	**May 1993**	**June 1993**	**July 2003**	**Aug 2003**	**June 2004**	**Aug 2004**
Bluegill	X	X	X	X	X	X
Bluntnose minnow	X		X			X
Brown bullhead	X	X	X	X	X	X
Carp	X	X	X	X	X	X
Carp x goldfish			X			
Common shiner				X		
Emerald shiner	X	X	X		X	X
Fathead minnow			X			
Freshwater drum	X	X	X	X	X	X
Gizzard shad	X	X	X		X	X
Golden shiner	X	X	X	X	X	X
Goldfish	X	X	X	X		
Hogsucker						X
Largemouth bass	X	X	X	X	X	X
Logperch						X
Northern pike		X	X	X		
Pumpkinseed	X	X	X	X	X	X
Rainbow trout	X					
Redhorse		X			X	
River chub	X					
Rock bass		X	X	X	X	X
Rudd			X			
Smallmouth bass	X	X	X	X	X	X
Spottail shiner	X	X	X	X		X
Walleye	X				X	
White bass	X	X				
White crappie		X				
White perch	X					
White sucker	X	X	X	X	X	X
Yellow perch	X	X	X	X	X	X
Total # of species:	**20**	**19**	**20**	**15**	**15**	**17**

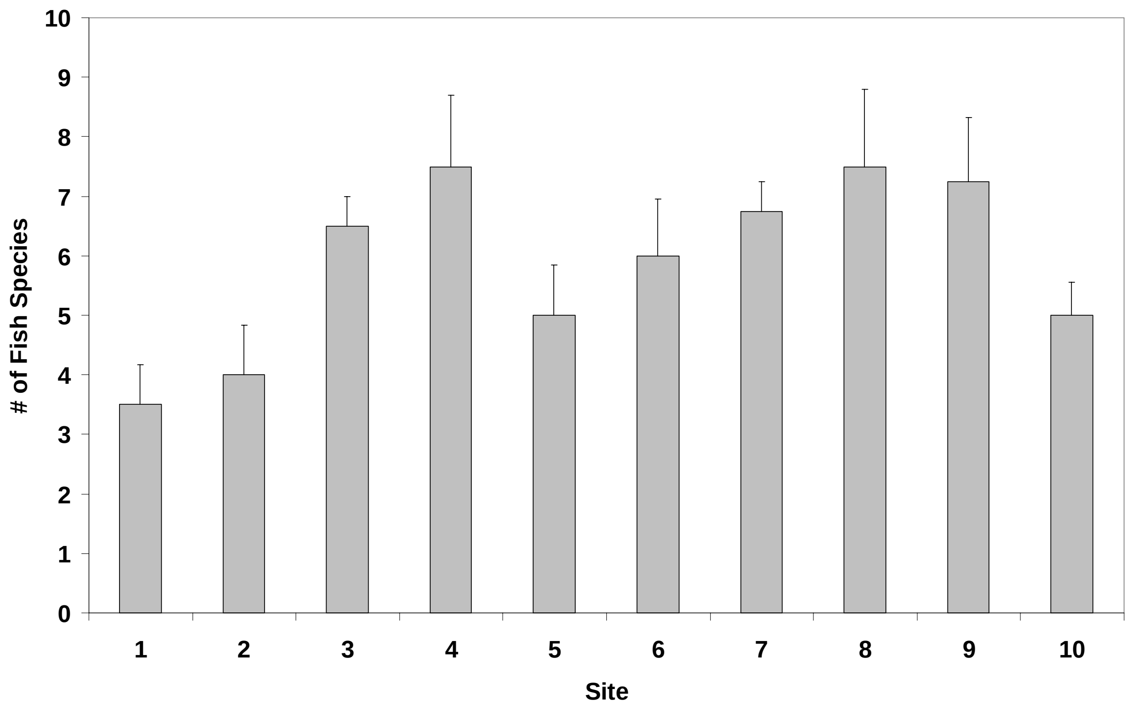


Figure 2.2 Mean number of fish species (± SE) collected per site (2003-2004)

2.3.3 Fish Health (DELT anomalies)

The frequency of occurrence of deformities, eroded fins, lesions, and tumors (i.e. DELT anomalies) depicts the health and condition of individual fish. These abnormalities occur infrequently or are absent from minimally impacted sites but occur frequently below point sources of pollutants and in areas where toxic chemicals are concentrated. The frequency of DELT anomalies provides an excellent measure of the subacute effects of chemical pollution and the aesthetic value of game and nongame fishes.

The frequency of DELT anomalies varied greatly among species collected during this study. For the six most commonly encountered species during this study (species that were found at five or more sites on each sampling date), DELT frequencies ranged from a low of 14% in pumpkinseed to a high of 87% in brown bullhead across all sites and sampling years (Figure 2.3).

To examine the frequency of DELT anomalies on a per site basis, we used data from the six most commonly encountered species listed above. For each species, sites were ranked from 1-10, with 1 representing the site with the highest % DELT and 10 the lowest. We then summed the six ranked scores (one for each species) for each study site to assign each site a single composite fish health score. With this procedure, a high composite health score indicates a site with a greater percentage of healthy individuals while low scores indicate a site with a lower percentage of healthy individuals. The composite health scores ranged from a low of 21.5 for Site 2 to a high of 44 for Site 5 (Figure 2.4).

Figure 2.3 Mean percentage of individuals (± SE) with DELT anomalies in the six most commonly encountered species collected in 2003 and 2004

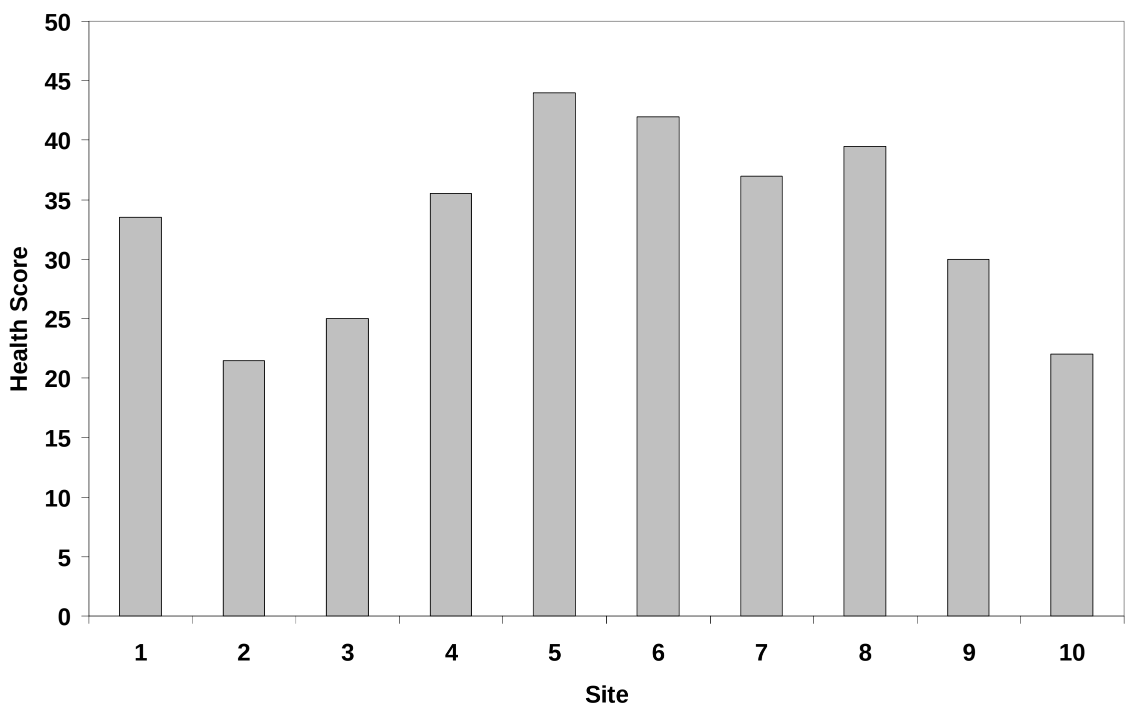


Figure 2.4 Composite fish health scores for each site based on DELT values for the six most commonly encountered species collected in 2003 and 2004. Higher values correspond to a larger percentage of healthy individuals at a particular site

2.3.4 Index of Biotic Integrity

The Index of Biotic Integrity (IBI) uses attributes of the fish association in a stream or river to index human effects on the drainage relative to regional and historical standards (Karr 1981). As originally developed, the IBI consisted of 12 metrics that could be grouped under five categories: species richness and composition, local indicator species, trophic composition, fish abundance, and fish condition. Ideally, these metrics provide information about a broad range of structural and organizational aspects of a river ecosystem, including habitat features of different types and sizes, food sources, productivity, predation, and parasitism (Steedman 1988). Each metric is scored against values that would be expected for an undisturbed stream or river in that particular region. The IBI consists of the sum of the values assigned to each metric in the index.

Many studies have confirmed the general usefulness of the IBI approach. However, since the IBI was originally developed for warmwater streams in the midwestern United States, it is common practice to modify the IBI to take into account local conditions, native fish assemblages, highly degraded habitats, or other factors (Steedman 1988). For this study, we have retained the basic structure of the original IBI (i.e. the same five categories of metrics have been used), but we have modified the metrics to better reflect local conditions (Table 2.3).

Table 2.3 IBI Metrics for the Buffalo River AOC

		Scoring Criteria		
Category	**Metric**	**5**	**3**	**1**
Species Richness	1. Total number of fish species	>15	8-15	<8
Composition	2. Total number of insectivore species	>7	3-7	<3
	3. Total number of sunfish and cyprinid* species	>7	3-7	<3
	4. Percent of individuals that are tolerant	<12%	12-22%	>22%
Trophic Composition	5. Percent of individuals that are omnivores	<20%	20-45%	>45%
	6. Percent of individuals that are insectivores	>65%	30-65%	<30%
	7. Percent of individuals that are top carnivores	>5%	1-5%	<1%
Fish Abundance	8. Total number of individuals caught	>250	75-250	<75
Condition	9. Percent of individuals with DELT	0-2%	2-5%	>5%

* Cyprinid species excluding carp and goldfish

We applied the following scale to convert the total IBI score (i.e. the sum of the scores for each metric) into a quality rating for each of the ten sites in this study: Excellent, 41-45; Good, 34-40; Fair, 27-33; Poor, 20-26; and Very Poor, 9-19. This scale is consistent with those developed and adopted for other studies in this geographic region (Kurtenbach 1994, Greer 2002).

We first calculated IBI scores for each site based on data from the four sampling dates; we then calculated a single IBI for each site as the average of these four values. The mean IBI scores ranged from a low of 15.5 (at sites 1 and 6) to a high of 23 (at sites 4 and 7)(Figure 2.5). Using the stream rating scale given above, all of the sites would be rated as either "poor" (sites 3, 4, 5, 7, 8, 9, and 10) or "very poor" (sites 1, 2, and 6).

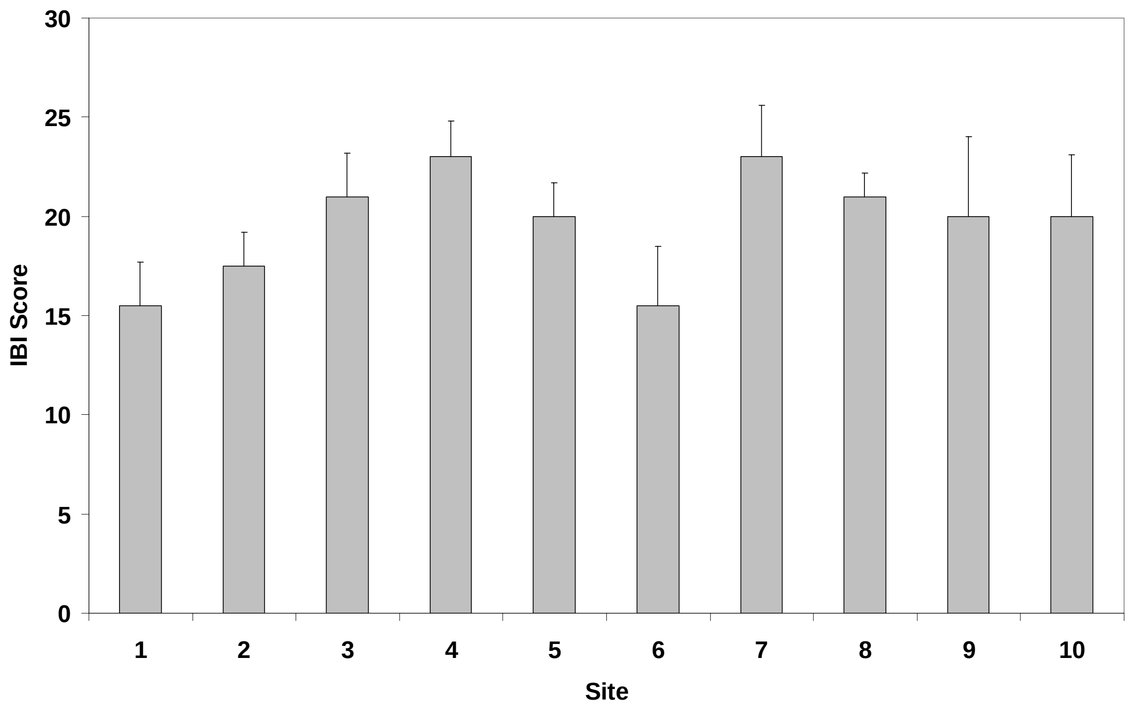


Figure 2.5 Mean IBI scores (± SE) for each study site using data from 2003-2004

2.4 Conclusion

The larval fish sampling showed similar species diversity and abundance in 2003-2004 as compared to 1993 (8-10 species found). No site-specific trends were observed. The large number of larval fish collected in June 2004 compared to the other sampling dates does not appear to correlate with any particular biotic or abiotic factor. However, this large variation in yield among sampling dates does highlight the need for more intensive, fine-scale sampling of larval fishes in future studies.

The juvenile and adult electrofishing surveys showed similar trends to those reported in previous studies from the early 1990's (NYS DEC 1993, Singer et al. 1994). Table 2.2 summarizes fish occurrences from the early 1990's compared to the present study, and overall species number and the specific fishes present are similar. With respect to recreational fisheries in the Buffalo River, the abundance and size of largemouth bass in the present survey is notable. Largemouth bass were more abundant in the 2003-2004 surveys compared to those from the early 1990's, and the mean size was greater as well. Although fish consumption advisories must be taken into account, the largemouth bass fishery seems to be improving in the Buffalo River AOC.

DELT anomalies varied greatly among species, with a low of 14% in pumpkinseed to a high of 87% in brown bullhead. For the river as a whole, DELT scores averaged 37%, which is much higher than what would be expected for a moderately impacted (2-5%) or an unimpacted (< 2%) river. While it is difficult to establish appropriate background levels for DELT anomalies in the Great Lakes region where virtually all streams and rivers are impacted by human development to some extent (see Premdas et al. 1995), the percentage of fish with DELT anomalies appears to be very high in the Buffalo River AOC. Although data on frequency of occurrence of DELT anomalies is lacking for many species of fishes encountered in this study, brown bullhead have been monitored frequently throughout the Great Lakes region for incidence of cutaneous and oral tumors (an important component of the DELT index). The overall incidence of dermal and oral tumors in bullhead from the Detroit River (an impacted site) was 10.2%, but in the oldest age classes the frequency of occurrence was as high as 100% (Maccubbin and Ersing 1991). Other studies document skin tumor frequencies in bullheads that range from 0% in unimpacted systems (Baumann et al. 1987) to over 90% in heavily impacted areas (Poulet et al. 1994). Given the strong correlation between skin cancers in bullheads and high concentrations of sediment pollutants (Black 1983), it would appear that current brown bullhead populations in the Buffalo River AOC are still being exposed to high levels of environmental contamination (most likely in the form of polynuclear aromatic hydrocarbons and/or metals).

Using standard criteria for interpreting the Index of Biotic Integrity scores, seven sites (#3, 4, 5, 7, 8, 9, and 10) would be rated “poor” and three (#1, 2, and 6) would be rated “very poor”. These scores are similar to those obtained by Greer (2002) for Cazenovia Creek, a tributary of the Buffalo River. The IBI scores for the Buffalo River AOC should be interpreted cautiously, since it can be difficult to apply the IBI to severely altered waterways for which little historical data exists. However, it is clear from the analysis that the low species diversity present in the river and the very high incidence of DELT anomalies remain obstacles to improvement of the fish community in the AOC. Restoration activities focused on increasing the diversity of fish habitats in the river should increase species diversity over time, and remediation of contaminated sediments may also decrease the occurrence of DELT anomalies and lead to overall improvements in fish health.

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APPENDIX 2.1

NUMBERS AND AVERAGE LENGTHS OF LARVAL FISHES COLLECTED AT EACH SITE AND COLLECTION DATE

Appendix 2.1 Numbers (N) and average total lengths (TL in mm) of larval fishes caught at each site and collection date (shallow and deep tows combined at each site).

	June 2003		August 2003		June 2004		August 2004	
	N	TL	N	TL	N	TL	N	TL
Site 1								
Alewife	-	-	-	-	10	5.3	-	-
Bluntnose minnow	-	-	-	-	4	5.8	-	-
Gizzard shad	1	4.2	-	-	39	4.3	-	-
Lepomis sp.	-	-	1	17.2	-	-	-	-
Yellow perch	-	-	-	-	6	6.4	-	-
Site 2								
Alewife	-	-	-	-	17	4.9	-	-
Bluntnose minnow	-	-	-	-	1	5.9	-	-
Gizzard shad	2	4.1	-	-	22	4.5	-	-
Logperch	-	-	-	-	2	8.7	-	-
Pomoxis sp.	1	3.5	-	-	2	4.4	-	-
Yellow perch	-	-	-	-	6	6.4	-	-
Site 3								
Alewife	-	-	-	-	39	5.0	-	-
Gizzard shad	-	-	-	-	8	4.4	-	-
Pomoxis sp.	-	-	-	-	1	3.9	-	-
Round goby	-	-	-	-	-	-	1	10.0
Yellow perch	1	6.7	-	-	-	-	-	-
Site 4								
Alewife	-	-	-	-	29	4.8	-	-
Bluntnose minnow	1	7.0	-	-	-	-	-	-
Gizzard shad	-	-	1	23.0	3	4.4	-	-
Lepomis sp.	-	-	4	9.9	-	-	-	-
Logperch	-	-	-	-	5	6.8	-	-
Pomoxis sp.	1	4.2	1	7.2	5	4.8	-	-
Yellow perch	6	7.8	-	-	-	-	-	-
Site 5								
Alewife	-	-	-	-	48	5.4	-	-
Bluntnose minnow	2	6.8	-	-	-	-	-	-
Fathead minnow	1	7.5	-	-	-	-	-	-
Gizzard shad	-	-	-	-	4	5.0	-	-
Lepomis sp.	-	-	1	10.8	-	-	-	-
Logperch	-	-	-	-	2	6.8	-	-
Pomoxis sp.	1	4.7-	-	-	2	5.2	-	-
Yellow perch	5	7.3	-	-	-	-	-	-

	June 2003		August 2003		June 2004		August 2004	
	N	TL	N	TL	N	TL	N	TL
Site 6								
Alewife	-	-	-	-	28	5.1	-	-
Bluntnose minnow	2	5.5	-	-	-	-	-	-
Gizzard shad	-	-	-	-	7	4.5	1	4.2
Logperch	-	-	-	-	2	6.5	-	-
Pomoxis sp.	1	4.4	1	7.1	2	4.4	-	-
Site 7								
Alewife	-	-	-	-	51	5.1	-	-
Fathead minnow	1	7.0	-	-	-	-	-	-
Gizzard shad	1	4.4	-	-	8	4.9	-	-
Logperch	-	-	-	-	1	6.4	-	-
Pomoxis sp.	-	-	-	-	3	4.2	-	-
Yellow perch	1	7.3	-	-	-	-	-	-
Site 8								
Alewife	-	-	-	-	75	5.3	-	-
Common carp	-	-	-	-	-	-	1	5.0
Gizzard shad	1	4.3	-	-	8	4.9	-	-
Logperch	-	-	-	-	5	6.9	-	-
Pomoxis sp.	1	4.4	-	-	5	4.4	-	-
Yellow perch	9	6.7	-	-	-	-	-	-
Site 9								
Alewife	-	-	-	-	124	5.4	-	-
Gizzard shad	1	4.8	-	-	24	4.7	-	-
Logperch	-	-	-	-	6	6.2	-	-
Pomoxis sp.	2	4.4	1	6.2	5	4.6	-	-
Yellow perch	7	6.8	-	-	2	7.0	-	-
Site 10								
Alewife	-	-	-	-	79	5.5	-	-
Common carp	-	-	-	-	-	-	1	6.6
Fathead minnow	1	7.2	-	-	-	-	-	-
Gizzard shad	1	4.4	-	-	1	4.6	-	-
Logperch	-	-	-	-	4	6.6	-	-
Pomoxis sp.	1	4.6	3	7.1	-	-	-	-
Yellow perch	9	6.7	-	-	-	-	-	-

APPENDIX 2.2

NUMBERS, LENGTHS, AND SIZE RANGES OF JUVENILE AND ADULT FISHES COLLECTED AT EACH SITE AND COLLECTION DATE

Appendix 2.2a Numbers (N), average total lengths (TL in cm), and size ranges (in cm) of juvenile and adult fishes caught by electrofishing at each site during 2003.

	June 2003			August 2003		
	N	TL	Range	N	TL	Range
Site 1						
Bluegill	2	12.3	11.5-13.0	-	-	-
Brown bullhead	2	33.0	31.0-35.0	-	-	-
Common carp	-	-	-	1	65.0	-
Emerald shiner	1	5.5	-	-	-	-
Golden shiner	-	-	-	1	12.5	-
Largemouth bass	-	-	-	1	37.0	-
Pumpkinseed	3	11.8	10.0-13.0	5	12.7	10.0-15.0
White sucker	2	22.5	22.0-23.0	-	-	-
Site 2						
Brown bullhead	3	29.7	28.0-32.0	1	24.5	-
Common carp	4	55.0	48.0-62.0	3	51.7	28.0-64.0
Emerald shiner	3	7.0	6.5-8.0	-	-	-
Gizzard shad	9	23.9	21.0-28.0	-	-	-
Largemouth bass	-	-	-	2	19.5	17.0-22.0
Pumpkinseed	2	13.0	13.0-13.0	4	13.1	8.0-17.0
White sucker	1	25.0	-	-	-	-
Yellow perch	-	-	-	1	13.0	-
Site 3						
Bluegill	1	13.0	-	-	-	-
Brown bullhead	4	33.8	33.0-35.0	1	32.0	-
Common carp	-	-	-	3	52.7	29.0-67.0
Common shiner	-	-	-	2	8.3	8.0-8.5
Emerald shiner	1	6.0	-	-	-	-
Golden shiner	1	12.5	-	3	13.0	11.0-15.0
Goldfish	-	-	-	2	29.0	28.0-30.0
Largemouth bass	2	24.0	23.0-25.0	4	23.8	20.0-26.0
Pumpkinseed	7	12.6	8.5-16.0	3	13.2	12.5-14.0
Smallmouth bass	1	7.5	-	-	-	-
Yellow perch	1	15.0	-	-	-	-
Site 4						
Bluegill	1	12.0	-	-	-	-
Brown bullhead	5	31.2	22.0-37.0	2	29.8	24.5-35.0
Carp x goldfish	1	20.5	-	-	-	-
Common carp	1	62.0	-	1	27.5	-
Emerald shiner	3	6.2	5.5-7.0	-	-	-
Gizzard shad	14	23.1	20.0-26.0	-	-	-
Golden shiner	-	-	-	2	14.3	11.5-17.0
Goldfish	-	-	-	1	28.0	-
Largemouth bass	8	23.6	16.0-31.0	4	27.5	19.0-35.0
Northern pike	1	57.0	-	-	-	-

	June 2003			August 2003		
	N	TL	Range	N	TL	Range
Site 4 (cont.)						
Pumpkinseed	18	13.9	10.5-18.0	2	12.3	8.5-16.0
Rock bass	-	-	-	1	2.5	-
Spottail shiner	1	8.0	-	-	-	-
White sucker	-	-	-	1	34.0	-
Site 5						
Bluegill	-	-	-	1	18.0	-
Brown bullhead	-	-	-	1	36.0	-
Common carp	-	-	-	1	57.0	-
Emerald shiner	4	6.9	6.0-7.5	-	-	-
Gizzard shad	6	25.7	23.5-28.0	-	-	-
Golden shiner	1	8.5	-	-	-	-
Goldfish	1	26.0	-	-	-	-
Largemouth bass	1	40.0	-	-	-	-
Pumpkinseed	5	13.5	11.0-16.0	-	-	-
Rock bass	1	17.0	-	-	-	-
White sucker	2	31.0	26.0-36.0	-	-	-
Site 6						
Bluntnose minnow	2	7.0	5.5-8.5	-	-	-
Brown bullhead	5	31.8	20.0-56.0	1	25.5	-
Common carp	-	-	-	2	58.0	57.0-59.0
Emerald shiner	1	6.0	-	-	-	-
Freshwater drum	1	40.0	-	-	-	-
Gizzard shad	7	23.2	20.5-25.0	-	-	-
Golden shiner	2	18.5	18.0-19.0	2	12.8	12.0-13.5
Goldfish	-	-	-	1	30.0	-
Largemouth bass	5	26.6	15.0-35.0	-	-	-
Pumpkinseed	3	10.0	5.5-13.0	1	17.0	-
White sucker	1	37.5	-	1	39.0	-
Site 7						
Bluegill	1	17.0	-	-	-	-
Brown bullhead	-	-	-	1	34.0	-
Common carp	1	69.0	-	1	57.0	-
Emerald shiner	36	6.9	5.5-9.0	-	-	-
Gizzard shad	5	20.2	15.5-25.0	-	-	-
Golden shiner	2	15.3	15.0-15.5	-	-	-
Largemouth bass	1	16.0	-	1	26.0	-
Pumpkinseed	11	15.2	11.0-17.5	1	6.5	-
Smallmouth bass	-	-	-	1	4.5	-
Spottail shiner	1	8.0	-	-	-	-
Site 8						
Bluegill	1	19.0	-	-	-	-
Bluntnose minnow	5	6.3	6.0-7.0	-	-	-
Brown bullhead	12	32.2	29.0-36.0	-	-	-

	June 2003			August 2003		
	N	TL	Range	N	TL	Range
Site 8 (cont.)						
Common carp	1	55.5	-	-	-	-
Fathead minnow	1	9.5	-	-	-	-
Freshwater drum	-	-	-	1	61.0	-
Gizzard shad	2	31.5	28.0-35.0	-	-	-
Golden shiner	1	20.0	-	1	7.5	-
Largemouth bass	2	30.0	26.0-34.0	1	24.0	-
Pumpkinseed	8	13.6	10.0-16.0	1	13.5	-
Rock bass	2	11.8	5.5-18.0	-	-	-
Spottail shiner	7	9.5	7.5-10.0	-	-	-
White sucker	1	21.0	-	1	30.0	-
Site 9						
Bluegill	1	14.0	-	-	-	-
Bluntnose minnow	1	8.0	-	-	-	-
Brown bullhead	4	32.0	29.0-34.0	1	33.0	-
Common carp	1	63.0	-	5	60.8	54.0-70.0
Common shiner	-	-	-	1	11.5	-
Gizzard shad	1	24.0	-	-	-	-
Golden shiner	2	18.5	18.0-19.0	4	12.6	11.0-14.5
Largemouth bass	2	33.0	32.0-34.0	-	-	-
Pumpkinseed	8	14.4	9.5-17.0	-	-	-
Rock bass	1	18.5	-	-	-	-
White sucker	1	43.0	-	-	-	-
Site 10						
Bluegill	2	15.5	14.0-17.0	-	-	-
Brown bullhead	2	30.0	28.0-32.0	3	30.7	26.0-36.0
Common carp	-	-	-	3	58.0	55.0-62.0
Gizzard shad	1	16.0	-	-	-	-
Largemouth bass	2	30.5	21.0-40.0	-	-	-
Pumpkinseed	2	16.5	16.0-17.0	-	-	-
Rudd	1	35.0	-	-	-	-
Smallmouth bass	2	21.5	8.0-35.0	1	7.0	-
White sucker	-	-	-	2	32.0	26.0-38.0

Appendix 2.2b Numbers (N), average total lengths (TL in cm), and size ranges (in cm) of juvenile and adult fishes caught by electrofishing at each site during 2004.

	June 2004			August 2004		
	N	TL	Range	N	TL	Range
Site 1						
Brown bullhead	1	30.0	-	-	-	-
Common carp	1	60.0	-	1	59.0	-
Gizzard shad	1	33.0	-	-	-	-
Pumpkinseed	1	13.0	-	-	-	-
Site 2						
Brown bullhead	2	32.8	32.5-33.0	-	-	-
Gizzard shad	2	32.5	32.0-33.0	-	-	-
Largemouth bass	2	27.8	20.0-35.5	-	-	-
Pumpkinseed	2	14.8	14.0-15.5	3	8.8	8.5-9.0
Site 3						
Bluegill	-	-	-	1	7.0	-
Brown bullhead	1	27.0	-	-	-	-
Common carp	2	71.0	56.0-86.0	1	63.0	-
Freshwater drum	1	48.0	-	-	-	-
Gizzard shad	3	31.5	27.5-34.5	-	-	-
Golden shiner	-	-	-	3	13.7	9.5-17.0
Largemouth bass	3	34.7	34.0-35.0	-	-	-
Logperch	-	-	-	1	9.5	-
Pumpkinseed	2	14.8	13.5-16.0	6	11.3	8.0-14.0
White sucker	1	29.5	-	-	-	-
Site 4						
Bluegill	-	-	-	2	11.0	7.5-14.5
Brown bullhead	2	34.0	33.0-35.0	-	-	-
Emerald shiner	-	-	-	2	7.0	7.0-7.0
Gizzard shad	-	-	-	2	20.3	19.5-21.0
Golden shiner	2	14.3	13.0-15.5	3	9.0	7.0-12.0
Largemouth bass	-	-	-	2	28.0	19.0-37.0
Logperch	-	-	-	1	10.0	-
Pumpkinseed	15	11.4	5.5-14.0	11	11.9	8.5-15.0
White sucker	-	-	-	1	17.0	-
Yellow perch	-	-	-	1	12.0	-
Site 5						
Brown bullhead	2	29.5	27.0-32.0	-	-	-
Gizzard shad	-	-	-	1	20.0	-
Golden shiner	-	-	-	1	8.0	-
Largemouth bass	2	35.8	30.0-41.5	3	17.2	16.0-18.0
Pumpkinseed	4	12.1	10.0-15.0	1	7.0	-
Rock bass	-	-	-	1	12.0	-
Smallmouth bass	2	33.0	28.0-38.0	-	-	-

	June 2004			August 2004		
	N	TL	Range	N	TL	Range
Site 6						
Brown bullhead	2	30.3	29.5-31.0	-	-	-
Common carp	1	58.5	-	4	68.6	65.0-72.0
Freshwater drum	1	44.0	-	-	-	-
Gizzard shad	5	30.1	16.0-35.0	-	-	-
Golden shiner	-	-	-	1	15.0	-
Largemouth bass	1	30.0	-	-	-	-
Rock bass	1	17.5	-	-	-	-
White sucker	-	-	-	1	43.0	-
Site 7						
Bluegill	12	16.7	14.5-19.0	-	-	-
Brown Bullhead	4	31.5	27.5-35.0	-	-	-
Common carp	-	-	-	4	58.5	54.0-65.0
Freshwater drum	1	49.0	-	-	-	-
Gizzard shad	1	38.0	-	-	-	-
Golden shiner	-	-	-	1	8.0	-
Largemouth bass	6	34.9	27.5-43.0	3	27.5	23.5-34.0
Logperch	-	-	-	1	10.0	-
Pumpkinseed	8	15.0	14.0-16.5	1	7.5	-
Rock bass	-	-	-	1	12.0	-
Smallmouth bass	1	35.0	-	-	-	-
Spottail shiner	-	-	-	1	7.5	-
Site 8						
Bluegill	2	16.0	15.0-17.0	1	7.5	-
Brown bullhead	5	30.9	30.0-32.0	-	-	-
Common carp	3	48.0	41.0-62.0	-	-	-
Emerald shiner	1	5.0	-	1	5.0	-
Gizzard shad	-	-	-	1	21.0	-
Golden shiner	1	13.0	-	6	9.6	7.5-12.0
Pumpkinseed	20	12.9	10.0-15.0	2	7.8	7.0-8.5
White sucker	1	11.5	-	-	-	-
Yellow perch	1	17.5	-	-	-	
Site 9						
Bluntnose minnow	-	-	-	2	8.0	8.0-8.0
Brown bullhead	1	28.5	-	-	-	-
Common carp	-	-	-	4	63.1	57.0-67.5
Emerald shiner	-	-	-	1	6.5	-
Golden shiner	-	-	-	5	9.2	8.5-10.5
Hogsucker	-	-	-	1	13.5	-
Largemouth bass	4	19.0	13.0-29.0	4	21.0	5.5-33.0
Logperch	-	-	-	1	10.0	-
Pumpkinseed	31	14.0	10.5-17.0	2	9.3	7.0-11.5
Rock bass	4	20.8	20.0-21.0	-	-	-
Smallmouth bass	4	32.1	20.5-43.0	-	-	-
Yellow perch	2	17.3	16.0-18.5	1	15.5	-

	June 2004			August 2004		
	N	TL	Range	N	TL	Range
Site 10						
Brown bullhead	1	34.0	-	-	-	-
Common carp	-	-	-	1	53.5	-
Emerald shiner	1	5.5	-	1	6.0	-
Golden shiner	-	-	-	3	12.3	9.0-15.5
Largemouth bass	1	30.0	-	1	16.5	-
Pumpkinseed	4	14.4	10.5-17.0	-	-	-
Smallmouth bass	2	37.0	31.0-43.0	-	-	-

CHAPTER 3
BENTHIC INVERTEBRATES

T.P. Diggins

3.1 Introduction

In the early 1960s much of the Buffalo River was considered biologically "dead", and few if any benthic (bottom-dwelling) organisms could be collected from its sediments (Blum 1964). By 1965 the Federal Water Pollution Control Agency listed the Buffalo River AOC as one of the three most polluted rivers in the United States (Sweeney 1973). With continuous recycling of water for industrial cooling, summer surface temperatures often exceeded 40°C, and discharged contaminants accumulated to shocking levels (Sweeney and Merckel 1972). Thick oil slicks covered the river's surface and caught fire on at least four occasions (Boyer 2002). Increased precipitation in the fall often flushed this grossly polluted water into the Niagara River in a concentrated "slug", causing widespread harm to wildlife downstream (Sweeney and Merkel 1972).

As environmental conditions grew intolerable even for commerce and industry, the City of Buffalo and the river's major industries established the Buffalo River Improvement Corporation to combat thermal pollution and contaminant accumulation (Oleszko 1977). Starting in 1967 a minimum of 400 million L of water daily (later reduced to ~ 60 million L following industrial closings) were pumped from Lake Erie to the river to provide cooling water and to augment low summer flows (Sweeney and Merckel 1972, Oleszko 1977).

Three of the river's major industries (Republic Steel, Donner-Hannah Coke, and Mobil Oil) have since closed or curtailed operations for economic reasons, decreasing industrial discharges. However, the Buffalo River continues to face environmental risks from residual sediment contamination (Stewart and Diggins 2002), and from combined sewer overflows (Loganathan et al. 1997), municipal wastewater treatment plants (Rossi 1995), smaller extant industries, leaking disposal facilities, and various non-point sources (NYS DEC 1989, Lee et al. 1991). The Buffalo River AOC currently suffers sediment and water quality impairments that have led to restrictions on recreation, fish consumption, water consumption, and to loss of wildlife habitat (NYS DEC 1989).

In a review of mostly unpublished historical Buffalo River benthic invertebrate data (1964 – 1993), Diggins and Snyder (2003) documented marked recolonization and expansion of the benthos from the barren conditions seen in 1964. While these developments were encouraging, biological recovery was far from complete. New taxa often occurred as scattered individuals, and the benthic community remained 70 – 99% tubificid oligochaetes (very pollution-tolerant) in terms of abundance as recently as 1993 (Diggins and Snyder 2003). Also, Diggins and Stewart (1998) reported 10 – 46% occurrence of mouthpart deformities in the chironomid (midge) genus *Chironomus* during 1990 – 1993, far exceeding the Great Lakes reference condition of 2.15% (Burt et al. 2003).

The objective of this portion of our comprehensive assessment of the Buffalo River was to evaluate the condition of benthic invertebrate communities at potential habitat restoration sites. As relatively local and sedentary components of the biota, benthic invertebrates must tolerate water and sediment conditions (i.e., they do not readily move away), and so provide an integrated metric of environmental health. Benthic communities have been well studied at a number of Great Lakes AOCs (Thornley 1985, Hart et al. 1986, Krieger and Ross 1993), including the Buffalo River, where detailed historical data are available (Diggins and Snyder 2003).

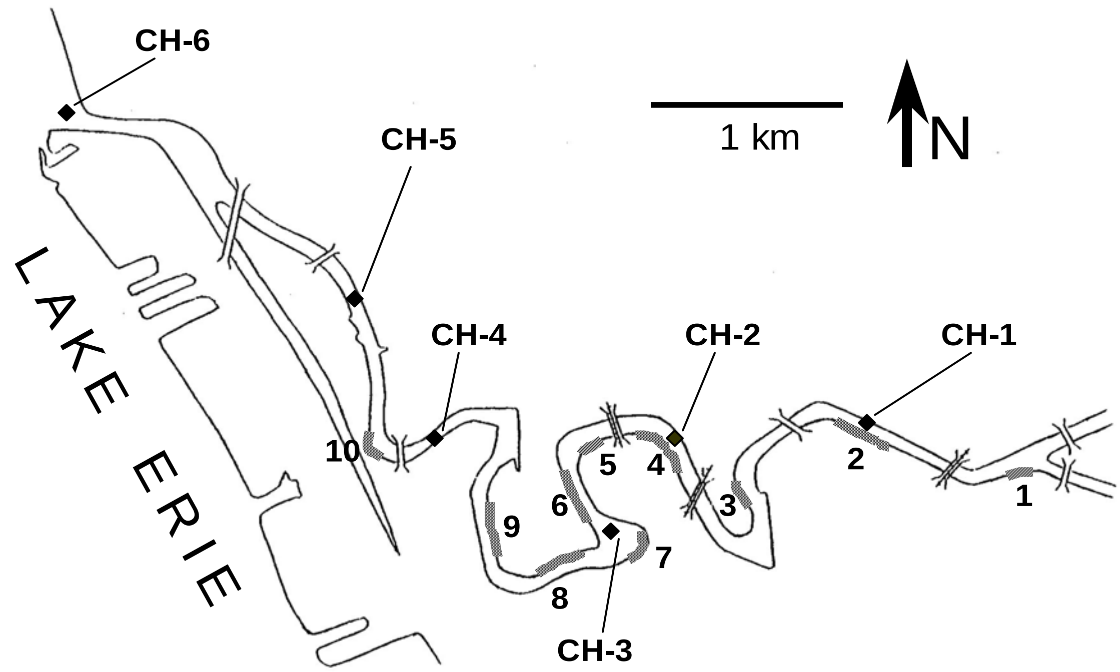

Figure 3.1 Location of shoreline habitat restoration (hatched lines) and mid-channel (diamonds) sites from which benthic invertebrates were sampled during 2003 – 2004.

3.2 Benthic sampling

In addition to the ten potential habitat restoration sites described in Chapter 1, benthos at six stations within the dredged navigation channel (Figure 3.1) were sampled for comparison with long-term trends that are better documented here than in the shallows (Diggins and Snyder 2003). Bottom sediments were sampled from a boat with a 15 x 15-cm Ponar grab. Nearshore habitat restoration sites were sampled at 0.5 – 2.0 m depth, and typically within 5 m of shore. In-channel sites were sampled at the 6 – 8-m depth that is maintained for navigation. Benthic samples were collected three times in 2003 (16 June, 18 August, and 30 October) and twice in 2004 (26 June and 27 September). Three replicate grab samples were taken at each site on each date. Samples were sieved in the field (500 micron), and retained material was preserved upon return to the lab (10% formalin and/or 70% ethanol). Habitat restoration site #5 was sampled only once, on 16 June 2003, with great difficulty. The decision was made henceforth to drop this site from the benthic sampling plan, especially in light of its maze of underwater steel pilings and cables that threatened to damage a boat on approach.

Invertebrates were identified to lowest practical taxon (always at least to family, but usually to genus) and enumerated. Gastropoda were identified following Jokinen (1992). Chironomid larvae were slide-mounted (Simpson and Bode 1980) for genus/species identification according to Simpson and Bode (1980), Peckarsky et al. (1990), and/or Merritt and Cummins (1996). Presence of mentum (mouthpart) abnormalities in larvae of the genus *Chironomus* was assessed as described by Diggins and Stewart (1993, 1998).

3.3 Data analysis

Due to their broad taxonomic and ecological diversity, benthic invertebrate communities are typically assessed by multimetric analytical approaches, e.g., as followed by Greer et al. (2002) in a study of the Buffalo River tributary Cazenovia Creek. Such analyses may incorporate measures of species richness, EPT (Ephemeroptera [mayfly], Plecoptera [stonefly], and Trichoptera [caddisfly]) richness, and one or more pollution tolerance-based biotic indices (e.g., Hilsenoff Biotic Index). In this study of the Buffalo River AOC we likewise followed a multimetric approach, but we have selected variables that are the most appropriate for the river's organically enriched, oxygen stressed, and likely still contaminated sediment environment. For example, EPT richness is not useful for comparison among Buffalo River sites, as few of these pollution sensitive organisms are found here, and all sites score uniformly low for this metric.

Also, we have explored a number of taxon-specific indicators, focusing on the Chironomidae (aquatic midges), which are typically the dominant insects in stressed systems (including the Buffalo River). In addition to the assessment of mouthpart deformities as mentioned above, we catalogued chironomid genus/species richness and applied a tolerance-based index of biotic integrity to the Chironomidae at the genus/species level. Diggins and Stewart (1998) found that during 1990 – 1993 such metrics were significantly associated with a gradient in trace metal contamination in the dredged channel of the Buffalo River AOC. These correlations between biotic health and sediment quality were not evident until detailed analyses of the Chironomidae were performed. Unfortunately, chironomids are too often reported only to the family level.

Most of the data reported here are presented both as figures, to allow visualization of spatial trends, and in tabular form, for possible inclusion in future biomonitoring efforts.

3.3.1 Benthic community metrics

1. Number of families per sample/site. The invasive Dreissenidae (zebra and quagga mussels) are excluded from this calculation to avoid characterizing their presence as an "improvement".
2. Oligochaete (nearly all family Tubificidae) density per m^2.
3. Chironomid density per m^2.
4. Percent contribution of tubificid oligochaetes to overall invertebrate density
5. Number of genera of Chironomidae per sample/site.
6. Genus/species Biotic Index for the Chironomidae, in which chironomid community pollution tolerance scores are generated.
7. Incidence of mouthpart deformities in larvae of the chironomid genus *Chironomus* (see Figure 3.2).

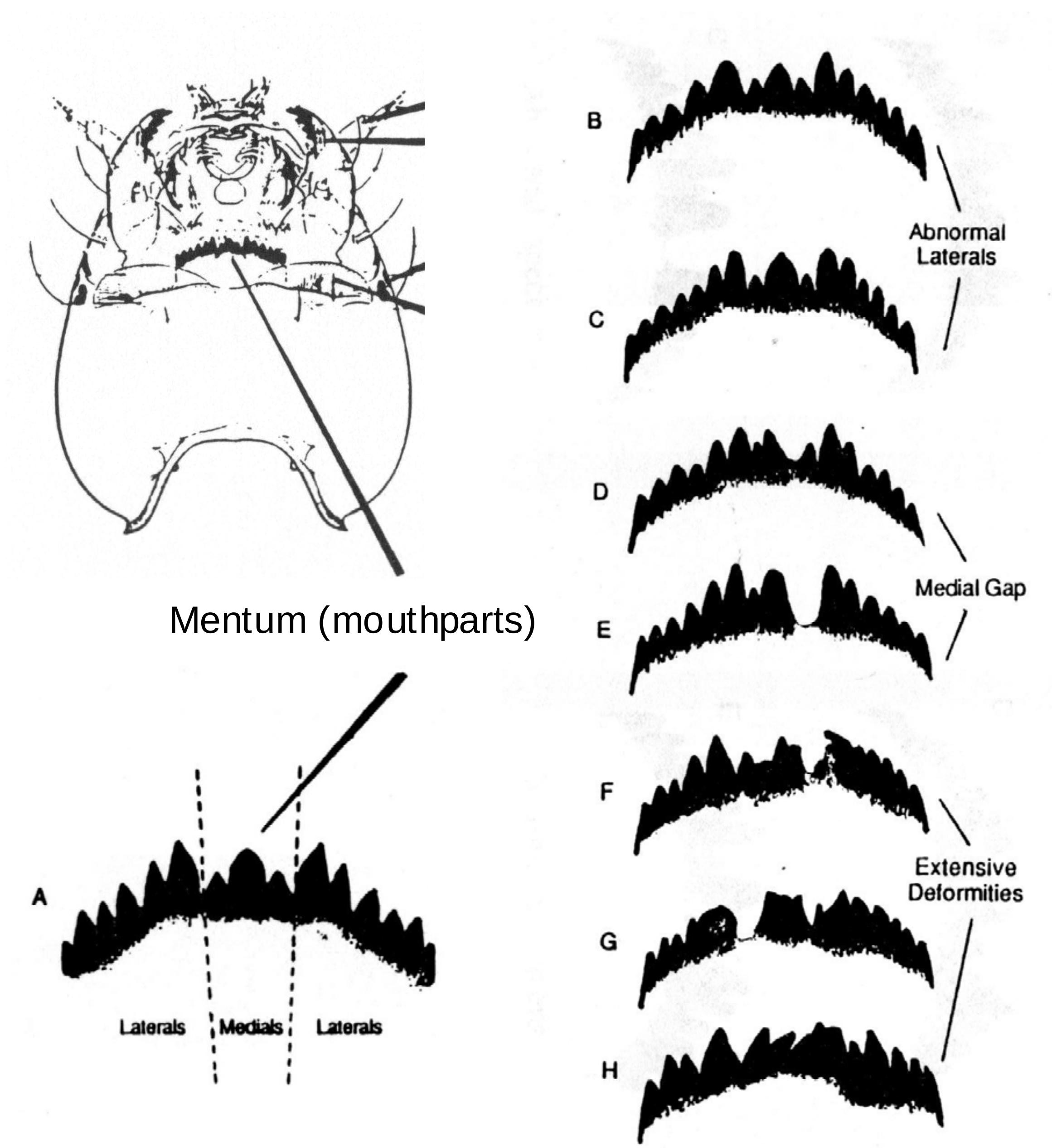

Figure 3.2 Larval head capsule morphology (modified from Oliver and Roussel 1983) and examples of mentum (mouthpart) deformities. A) Normal mentum. B – H) Deformities as observed in the Buffalo River during 1990 – 1993. Figure modified from Diggins (1997).

3.4 Results and Discussion

3.4.1 Benthic invertebrate families

Sixteen families of benthic invertebrates, including the invasive Dreissenidae, were collected from the Buffalo River during 2003 – 2004 (Table 3.1). Tubificidae (annelid oligochaete "sludge worms") and Chironomidae (insect "midge" larvae) dominated numerically. Other taxa consistently encountered included several families of gastropod mollusks, sphaerid "fingernail" clams, small leeches, and the dreissenids. Occurring as rare and scattered individuals were juvenile stages of several other insect groups usually found in streams with more heterogeneous sediments (i.e., sand and gravel in addition to mud and silt) than those of the Buffalo River. Notably absent were nymphs of mayflies and stoneflies, two groups generally considered pollution sensitive.

Nearshore habitat restoration sites consistently (with the exception of upstream site 1) yielded more invertebrate families than channel sites (Figure 3.3, Table 3.2). Up to 11 families were collected from nearshore sites, whereas only channel site CH-5 (above Michigan Avenue) yielded more than five. A plausible speculation is that river edge sediments may be more structurally heterogeneous, and oxygen stress may be less severe in shallower water, both of which could allow persistence of more invertebrate families. However, shoreline habitat restoration sites were still solidly dominated by oligocheates and chironomids (discussed below).

Disappointingly, invertebrate community richness during 2003 – 2004 was not only no better than during the early 1990s (Diggins and Snyder 2003), it had actually declined (Figure 3.4.A). In retrospect, collecting and enumerating the channel samples was very informative indeed, as it showed very clearly how historical trends of increasing taxonomic richness have reversed in the last decade. The reasons for this are not readily apparent, but this finding suggests that post-industrial biological recovery of the Buffalo River in its present state may remain stalled without active remediation.

Table 3.1 Occurrence of invertebrate families in the Buffalo River

A. Habitat restoration sites

Phylum	Class/Order/Suborder	Family	Common name	1	2	3	4	5	6	7	8	9	10
Annelida	Oligochaeta	Tubificidae	sludge worms	X	X	X	X	X	X	X	X	X	X
Annelida	Hirudina		leeches			X					X	X	X
Arthropoda	Insecta/Diptera	Chironomidae	midges	X	X	X	X	X	X	X	X	X	X
	Insecta/Diptera	Ceratopogonidae	biting midges										
	Insecta/Odonota/Zygoptera		dragonflies										
	Insecta/Trichoptera		caddisflies			X							
	Insecta/Coleoptera	Psephenidae	water penny beetles	X									
	Insecta/Coleoptera	Elmidae	riffle beetles								X	X	
Arthropoda	Amphipoda		scuds		X								
Mollusca	Gastropoda	Bythinidae	faucet snails				X	X	X	X	X	X	X
		Valvatidae	valve snails			X	X		X	X	X	X	X
		Planorbidae	rams horn snails										
		Physidae				X	X		X	X	X	X	X
Mollusca	Bivalvia	Sphaeridae	fingernail clams			X				X	X	X	X
		Unionidae	native clams		X		X						
		Dreissenidae	zebra/quagga mussels	Z	Z	Z/Q	Z		Z/Q	Z/Q	Z	Z	Z

B. Channel sites

Phylum	Class/Order/Suborder	Family	Common name	CH-1	CH-2	CH-3	CH-4	CH-5	CH-6
Annelida	Oligochaeta	Tubificidae	sludge worms	X	X	X	X	X	X
Annelida	Hirudina		leeches		X			X	X
Arthropoda	Insecta/Diptera	Chironomidae	midges	X	X	X	X	X	X
	Insecta/Diptera	Ceratopogonidae	biting midges		X				
	Insecta/Odonota/Zygoptera		dragonflies		X				
	Insecta/Trichoptera		caddisflies						
	Insecta/Coleoptera	Psephenidae	water penny beetles						
	Insecta/Coleoptera	Elmidae	riffle beetles						
Arthropoda	Amphipoda		scuds						
Mollusca	Gastropoda	Bythinidae	faucet snails			X		X	
		Valvatidae	valve snails					X	
		Planorbidae	rams horn snails					X	
		Physidae						X	
Mollusca	Bivalvia	Sphaeridae	fingernail clams					X	
		Unionidae	native clams						
		Dreissenidae	zebra/quagga mussels	Z	Z	Z	Z/Q	Z/Q	Z

Table 3.2 Site-mean benthic invertebrate parameters in the Buffalo River

A. Habitat restoration sites

	1	**2**	**3**	**4**	**5**	**6**	**7**	**8**	**9**	**10**
Families/sample	3	5	8	7	3	6	7	10	11	9
Oligochaetes per sq. meter	3802	12356	9758	8795		12390	11699	4811	8033	5951
Chironomids per sq. meter	370	573	291	351		281	583	178	522	262

B. Channel sites

	CH-1	**CH-2**	**CH-3**	**CH-4**	**CH-5**	**CH-6**
Families/sample	4	5	4	2	8	3
Oligochaetes per sq. meter	12217	14484	11230	15889	13970	12267
Chironomids per sq. meter	894	350	400	328	333	378

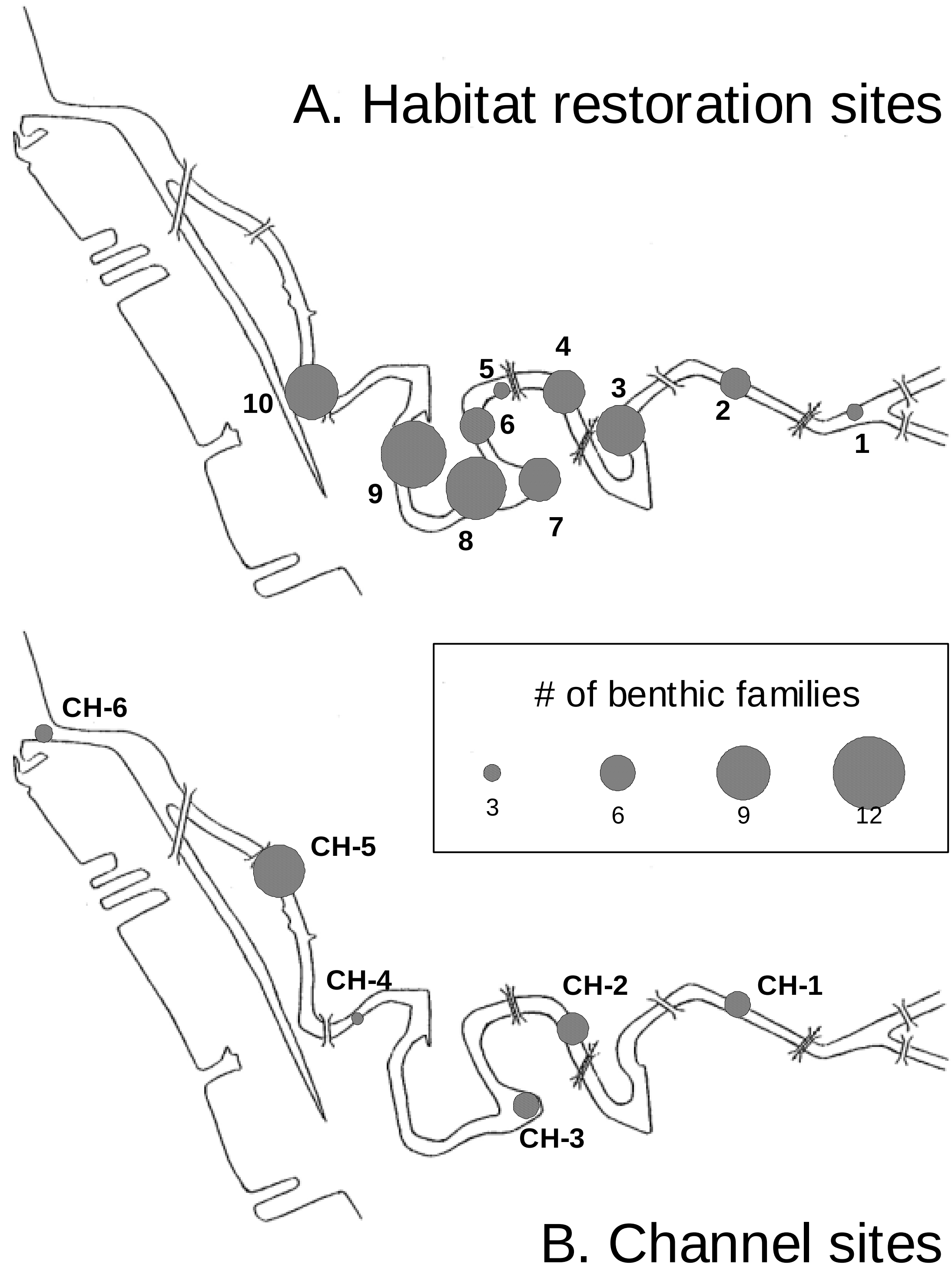

Figure 3.3 Site-mean richness of benthic invertebrate families during 2003 – 2004 at A) shoreline habitat restoration, and B) mid-channel sites.

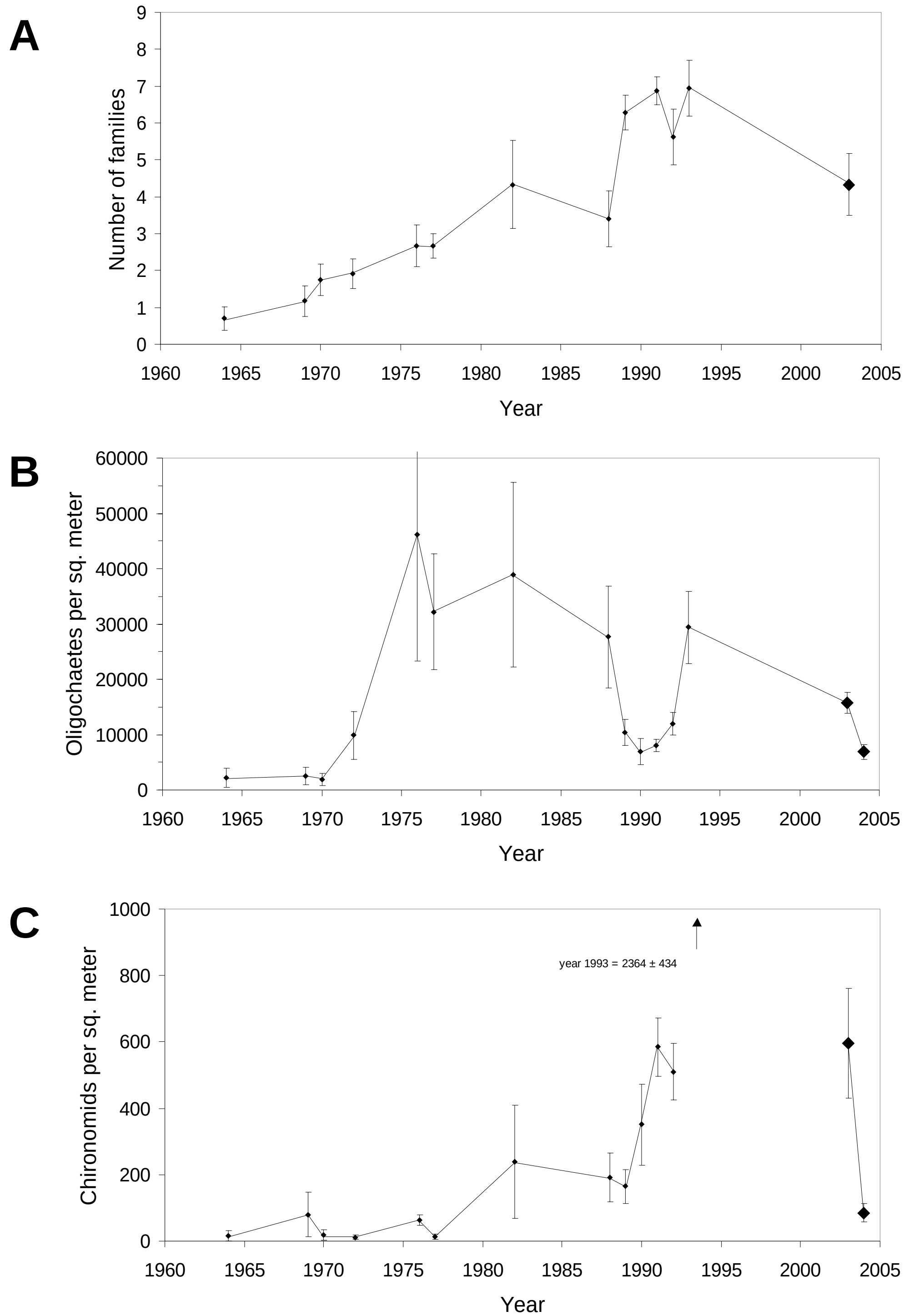

Figure 3.4 Whole river temporal trends in A) invertebrate family richness, B) Oligochaete density, and C) chironomid density.

3.4.2 Oligochaetes

Oligochaete density regularly exceeded 10,000/m^2 during 2003 – 2004, with only the habitat restoration sites 1 and 8 yielding average densities below 5000/m^2 (Table 3.2). Site-mean oligochaete densities were typically higher in the dredged channel than at shoreline sites (Figure 3.5). Oligochaete densities were also much lower in 2004 than in the preceding year – combined averages of 10,639 and 5153 for channel and shoreline sites, respectively, vs. 15,717 and 6893 during 2003. This may have been partially the result of the temporal proximity of our 27 September 2004 sampling date to the 09 September flood described in Chapter 5, which may have scoured bottom sediments. Oligochaete (and chironomid) from habitat restoration site 5 densities are not included in our site characterization matrix because sampling was not conducted here on the 27 September 2004 date, potentially biasing the data from this site.

Oligochaete densities during this study were generally similar to those recorded during the early 1990s (see Figure 3.4.B), with the exception of the very high densities from 1993 that were based on only one sampling date (Diggins and Snyder 2003). While these densities of 7 – 15,000/m^2 are much lower than the 30 – 50,000+/m^2 recorded during the late 1970s (Figure 3.4.B, also Diggins and Snyder 2003), tubificid oligochaete abundance still exceeds a long-held threshold of 5000/m^2 that signifies organic pollution (Nalepa and Thomas 1976). Additionally, the benthic invertebrate community was consistently >95% tubificid oligochaetes numerically at both channel and shoreline sites. Only at the habitat restoration sites 1 and 9 was the invertebrate community less than 90% oligocheates, and at 89% for each, barely so. Clearly, the entire Buffalo River AOC continues to be dominated by abundant and very pollution tolerant tubificid oligochaetes.

3.4.3 Chironomid densities

Site-mean chironomid densities during 2003 – 2004 ranged between 200 and 900/m^2 (Table 3.2), with no obvious trends along the river, or between channel and shoreline sites (Figure 3.6). River-wide chironomid densities during 2003 were in the range of those recorded in the early 1990s, again with the exception of a very high density from 1993's single sample date (Figure 3.4.C). As with the oligochaete data discussed above, chironomid densities were much lower in 2004 than in 2003 (585 vs. 85/m^2). Again, we speculate this may have resulted from bottom scouring during the 09 September 2004 storm event.

3.4.4 Chironomid richness and pollution tolerance

Twenty-two chironomid taxa (species or genera, depending on whether specific identification can be made based only on larval characteristics) were collected from the Buffalo River during 2003 – 2004 (Table 3.3). This is slightly fewer than the 27 taxa encountered during 1990 – 1993 (Diggins 2000). However, if only channel samples from 2003 – 2004 are considered (all 1990 – 1993 data are from the channel), chironomid richness appeared very poor during the present study – only six taxa. While the present study represents less than 1/6 of the sampling effort of studies in the early 1990s (Singer et al. 1994, Diggins and Stewart 1998, Diggins 2000), and thus may have missed rare species, 2003 – 2004 results still may reveal an actual decline in mid-channel chironomid richness over the past decade. Unfortunately, very few historical shoreline invertebrate data are available for comparison with present results from habitat restoration sites.

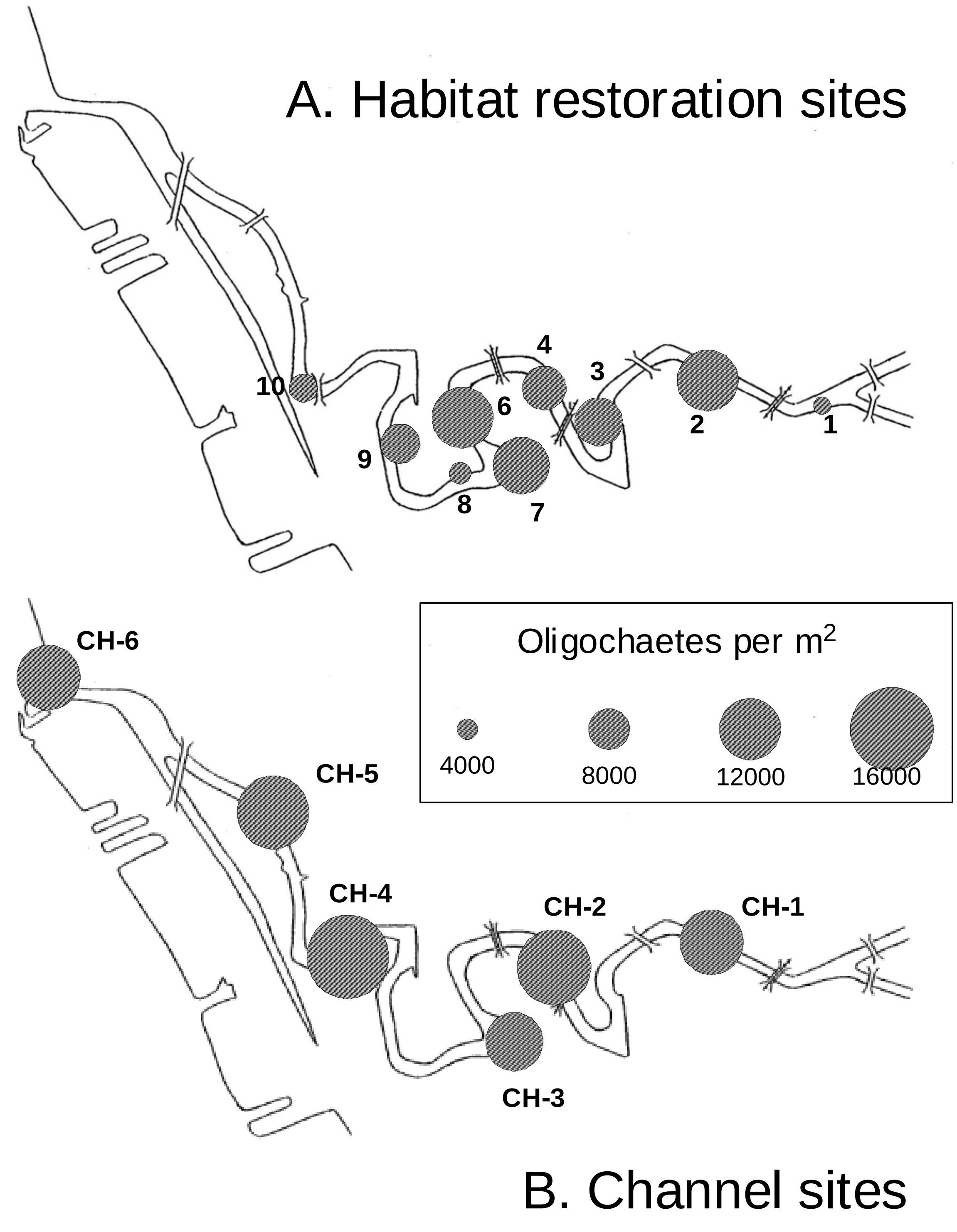

Figure 3.5 Site-mean densities of tubificid oligochaetes during 2003 – 2004 at A) shoreline habitat restoration, and B) mid-channel sites.

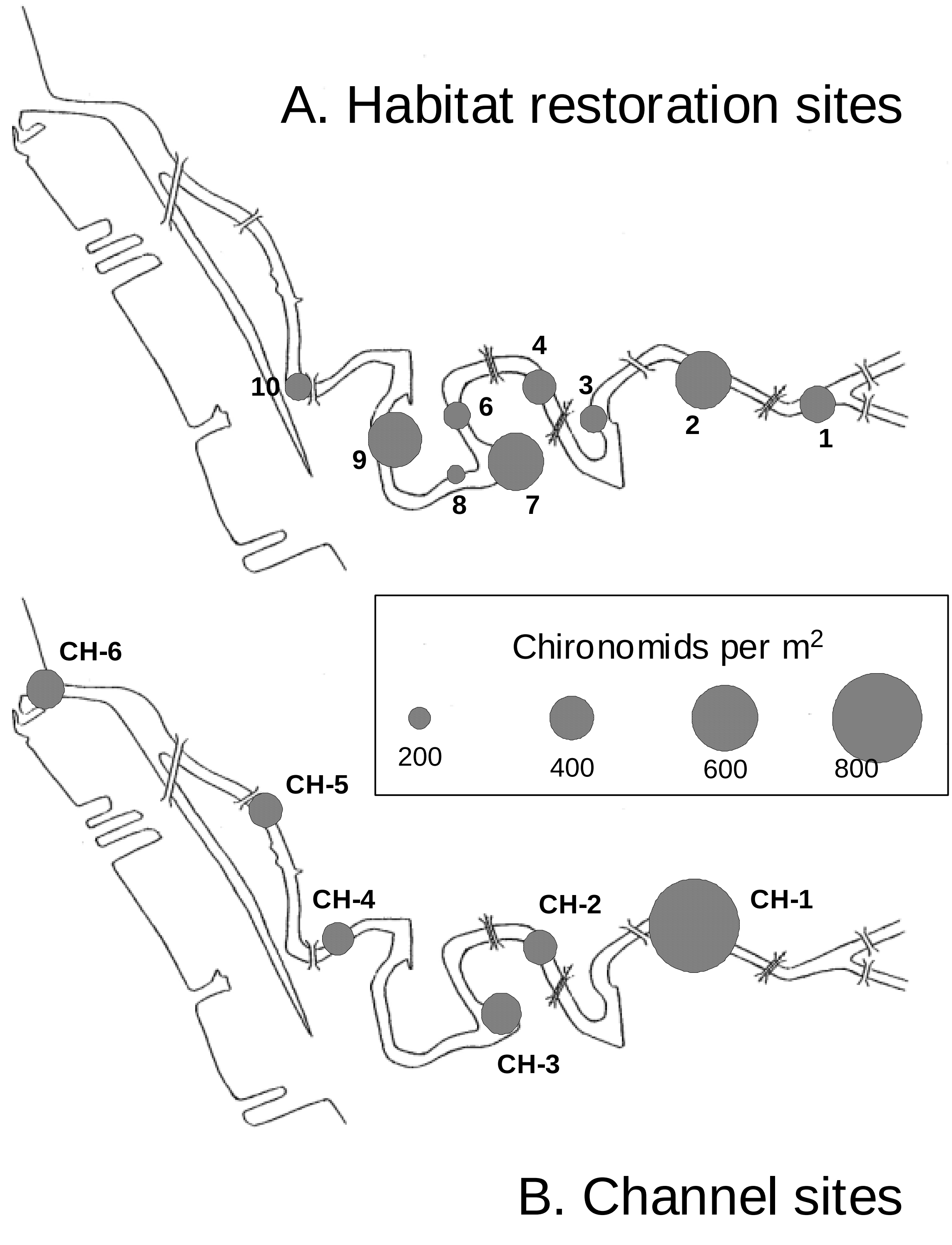

Figure 3.6 Site-mean densities of chironomid larvae during 2003 – 2004 at A) shoreline habitat restoration, and B) mid-channel sites.

Table 3.3 Occurrence of chironomid taxa in the Buffalo River

A. Habitat restoration sites

Chironomid taxon	1	2	3	4	5	6	7	8	9	10
TANYPODINAE										
Ablabesmyia	X									
Procladius		X		X	X	X	X	X	X	X
ORTHOCLADINAE										
Cricotopus sp.		X								
Cricitopus bicinctus									X	
Cricotopus silvestris	X									X
Nanocladius						X				
Psectrocadius	X	X								X
TANYTARSINI										
Paratanytarsus		X				X			X	
Rheotanytarus exiguus		X								
Tanaytarsus glabrescens		X								
Tanytarsus guerlus		X			X	X	X		X	
CHIRONOMINI										
Chironomus	X	X			X		X			
Cladopelma		X	X	X	X	X	X		X	
Cryptochironomus		X			X		X		X	
Crytptotendipes									X	
Dicrotendipes neomodestus	X				X	X	X		X	X
Endochironomus subtendens						X	X			
Glyptotendipes	X									
Parachironomus aborticus										
Paratendipes									X	
Polypedilum	X	X	X		X	X	X		X	
Tribelos										

B. Channel sites

Chironomid taxon	CH-1	CH-2	CH-3	CH-4	CH-5	CH-6
TANYPODINAE						
Ablabesmyia						
Procladius	X	X	X	X	X	X
ORTHOCLADINAE						
Cricotopus sp.						
Cricitopus bicinctus						
Cricotopus silvestris						
Nanocladius				X		
Psectrocadius						
TANYTARSINI						
Paratanytarsus						
Rheotanytarus exiguus						
Tanaytarsus glabrescens						
Tanytarsus guerlus						
CHIRONOMINI						
Chironomus						
Cladopelma				X	X	
Cryptochironomus	X			X		
Crytptotendipes						
Dicrotendipes neomodestus						
Endochironomus subtendens						
Glyptotendipes						
Parachironomus aborticus						
Paratendipes						
Polypedilum			X			
Tribelos	X					

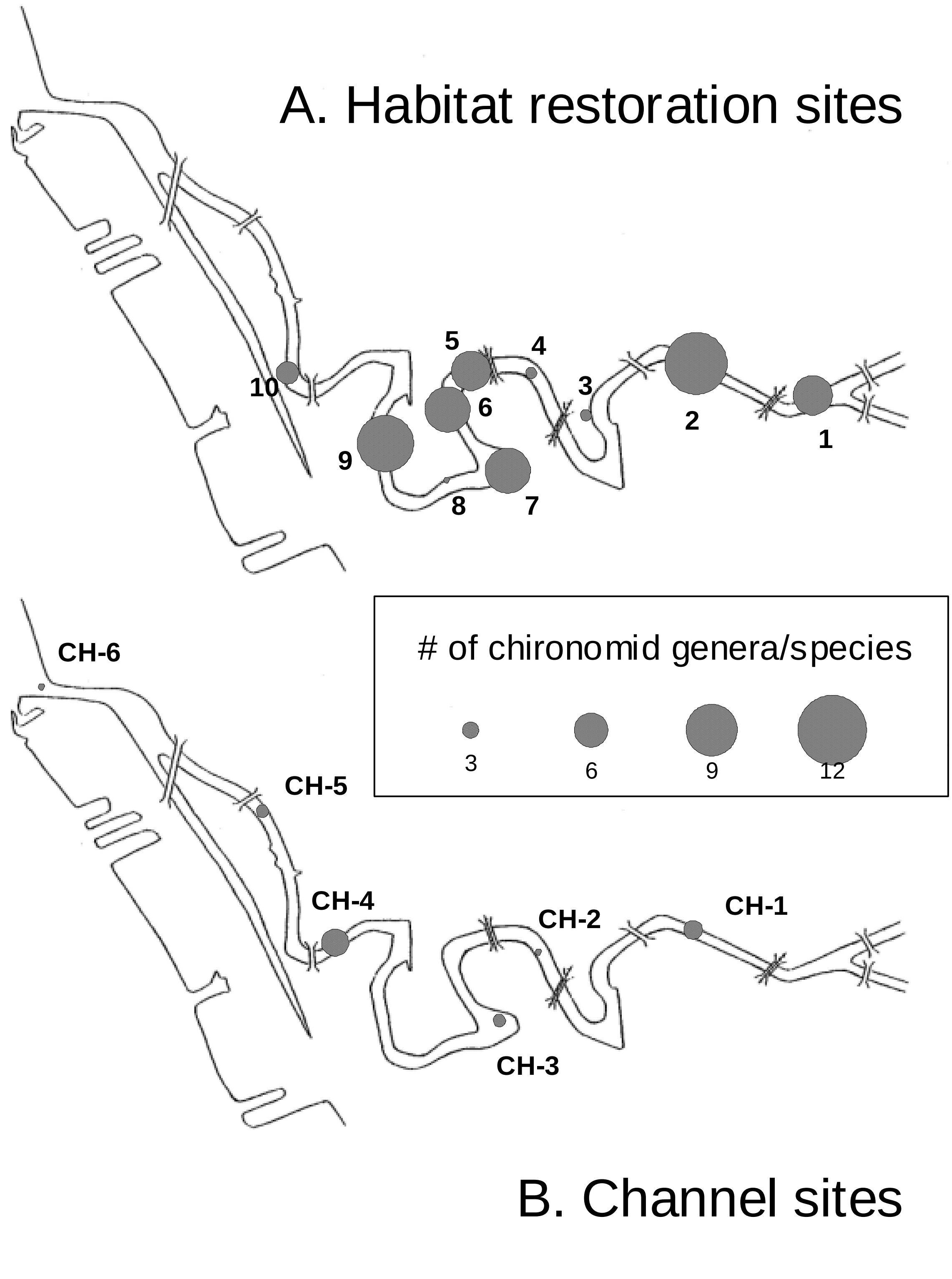

Figure 3.7 Number of chironomid genera/species collected during 2003 – 2004 at A) shoreline habitat restoration, and B) mid-channel sites.

The very pollution-tolerant genus *Procladius* (“Hilsenhoff” tolerance 10 [i.e., maximum]) was the most abundant chironomid collected during 2003 – 2004. Other abundant taxa included *Cladopelma* (tolerance 9), *Dicrotendipes neomodestus* (tolerance 8), *Tanytarsus guerlus* (tolerance 6), *Polypedilum* (tolerance 6), *Cryptochironomus* (tolerance 8), and *Chironomus* (tolerance 10). Tolerance values are reported from Mandaville (2002), summarizing a large number of sources.

Chironomid richness differed markedly between nearshore habitat restoration sites and channel sites (Figure 3.7), with none of the less diverse channel sites exceeding four taxa. Shoreline sites yielded up to 11 taxa, but were highly variable, with sites 3, 4, and 8 as species-poor as the channel sites.

Chironomid Biotic Index scores (i.e., tolerance score averaged among all individuals in a sample) at habitat restoration sites ranged from 6.60 (site 3) to 10.00 (site 8), with 10 representing a community composed entirely of the most pollution tolerant taxa. According to standards presented in Table 3.4, sites 2, 3, 5, 7, 9, and 10 were categorized as either “poor” or “impacted”, while sites 1, 4, 6, and 8 were “very poor”. All of the channel sites were characterized as “very poor”. It should be noted that habitat site 3, with its least tolerant Biotic Index, also yielded only two taxa, one of which (*Polypedilum*) happened to be only moderately tolerant. As such, it does not offer very convincing evidence of substantially greater environmental health than the five other “poor” sites.

Table 3.4 Ranges and interpretations of Biotic Index scores (modified from Mandaville 2002).

Biotic Index	Water/sediment quality	Degree of pollution
0.0 – 4.5	Excellent/very good	None to slight
4.51 – 6.5	Good/fair	Some to moderate
6.51 – 7.5	Impacted	Fairly substantial
7.51 – 8.5	Poor	Very substantial
8.51 – 10.0	Very poor	Severe

3.4.5 Chironomid mouthpart deformities

Mouthpart deformities (see Figure 3.2) in larvae of the chironomid genus *Chironomus* were not a useful metric of environmental health at habitat restoration sites, simply because we did not encounter sufficient numbers of this indicator genus in nearshore sediments to make site-to-site comparisons. However, more reliable deformity data were generated at channel sites, especially after taking extra grab samples dedicated to collecting large (>1 cm) red chironomids that often include *Chironomus*. Thus, we were at least able to compare 2003 – 2004 results with deformity frequency data from the early 1990s (Diggins and Stewart 1993, 1998). Unfortunately, this comparison yielded essentially the same trend as for other benthic invertebrate data – no improvement over the last decade, and some evidence of a decline in environmental health. In 2003 – 2004, 54.5% of *Chironomus* larvae (12 of 22) from in-channel sites displayed obvious mouthpart deformities. This percentage is at the high end of the range of deformity frequencies reported by Diggins and Stewart (1993, 1998) at channel sites during 1990 – 1993. (Reference populations at Great Lakes sites free of industrial impact display mouthpart deformities in only 2.15% of larvae [Burt et al. 2003].) Interestingly, however, *all* of the more limited number of *Chironomus* larvae (n = 12) collected at Buffalo River shoreline sites in 2003 – 2004 had normal mouthparts. This indicator genus was consistently less common in the Buffalo River during the present sampling than it had been a decade earlier, so an intensive study dedicated only to mouthpart deformities is recommended to help clarify the implications of this dichotomy between the shore and the channel. A preliminary, and tentative, interpretation is that shoreline chironomid larvae might not be exposed to or influenced by teratogenic (i.e., disrupting development) concentrations of sediment contaminants, whereas in-channel populations are.

3.5 Conclusions

Data collected during the present study indicate the Buffalo River AOC continues to be dominated by a rather low-diversity benthic invertebrate community broadly tolerant of pollution and environmental detrioration. High densities of tubificid oligochaetes (though lower than historical maxima), and their numerical dominance of the benthos, reveal poor environmental health. Oligochaete densities were higher in the channel than at shoreline habitat restoration sites. Fewer invertebrate families were collected during 2003 – 2004 than in the early 1990s, possibly even indicating some reversal of biotic recovery. Substantially more families occurred at shoreline sites than in the channel, although the habitat restoration sites were still dominated by pollution-tolerant oligochaetes and chironomids. Likewise, chironomid taxonomic richness was markedly higher at habitat restoration sites than in the channel, but samples largely constituted pollution-tolerant species and genera. Chironomid mouthpart deformities remain very high at channel sites (as they were during 1990 – 1993), but, interestingly, all of the rather limited number of larvae from shoreline sites had developed normally.

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CHAPTER 4
VEGETATION

T. P. Diggins, B. Sinn, C. F. Chuey

4.1 Introduction

Today's Buffalo River bears little resemblance to the stream that existed before the city was developed. The pre-settlement river was smaller and much shallower, and was fringed with extensive wetlands (Sweeney and Merkel 1972, Rossi 1996). However, the commercial value of the Buffalo River was realized as early as the 1820s, following the selection of Buffalo as the terminus of the Erie Canal. As extensive population growth and industrialization followed, the river was gradually channelized and deepened into its present form (Sweeney and Merckel 1972). While increases in channel width and depth facilitated industry and commerce, they also dramatically lengthened hydraulic retention time and exacerbated the ensuing decline in environmental health.

Environmental impacts of the Buffalo River's industrialization extend well beyond the water's edge. Much of the shoreline is "hardened" by mooring pilings, stone rip-rap, steel bulkheads, and structures built right to the river's edge. Submerged (macrophyte) and emergent (wetland) vegetation is limited because both above- and below-water riverbanks slope unnaturally steeply. Surrounding land is dominated by post-industrial "brownfields", often with highly disturbed soils and degraded vegetation. Invasive species abound.

The objective of this aspect of our Buffalo River assessment was to catalogue the structure and composition of submerged, emergent, and terrestrial vegetation communities at potential habitat restoration sites. It is essential to remediation efforts to assess the integrity of the flora, both as ecosystem components in their own right, and as structural components of wildlife habitat.

4.2 Methods

Presence/absence vegetation surveys were conducted at potential habitat restoration sites during August 2004 and August 2005. Included were woody and herbaceous terrestrial vegetation and submerged aquatic macrophytes. Materials were identified to species according to Gleason and Cronquist (1991) and Holmgren (1998). Plants were identified as invasive according to USDA (2005). Voucher specimens of all taxa are housed in the Herbarium of Youngstown State University. Percentage of overhanging shoreline cover was estimated in the field, and from the detailed satellite images presented in Figure 4.1.

4.3 Results and Discussion

More than fifty plant species (Table 4.1) were collected from Buffalo River shoreline sites, suggesting the potential for vigorous, productive plant communities to establish here. Perhaps not surprisingly, though, a number of taxa were invasives that exploit disturbed habitats.

Native black willow was often the dominant woody overstory plant, providing shade, habitat, and underwater structure when its brittle limbs fall into the river. Other native species such as eastern cottonwood, green ash, and silver maple occur at habitat restoration sites, but, unfortunately, so does the aggressive invasive tree-of-heaven. This species will readily displace native trees, and can grow under literally any soil condition, including cracks in pavement and/or masonry. Patches of tree-of-heaven are particularly well established at site 6, which is otherwise dominated by native black willow.

Habitat restoration sites differ markedly in their development of overhanging cover, ranging from near total (~80%) cover at site 1 (confluence with Cazenovia Creek) to 0% at site 3, the recently re-graded “Color Peninsula” (Figure 4.1, Figure 4.2).

Herbaceous vegetation is well developed at all sites, even where much of the shoreline is composed of stone rip-rap (e.g., sites 3 and 9). However, two nuisance invasives, purple loosestrife and especially Japanese knotweed, are also abundant. Japanese knotweed has established thick monospecific stands at several sites. Emergent wetland vegetation (cattails, reeds, etc.) is relatively limited, given the prevailing steepness of the riverbanks. Submerged macrophyte beds also are not extensive (again, due to the steeply sloping shoreline), but are present at most sites. Native American eelgrass and the pondweeds tend to dominate early in the season, while invasive Eurasian watermilfoil increases from August onward.

Invasive plant species stand as an impediment to habitat restoration along the Buffalo River, although all exotics may not pose equally great threats to community integrity. Many are relatively benign. Several, however, can seriously degrade a site both ecologically and in terms of aesthetics and recreation. (E.g., Japanese knotweed excludes all other vegetation, degrading the habitat, and also poses a physical obstacle to recreational access.) Therefore, we recommend that the serious nuisance invasives tree-of-heaven, Japanese knotweed, purple loosestrife, and submerged Eurasian watermilfoil should be subject to eradication campaigns as part of habitat restoration efforts. These species are difficult to eliminate completely, but they often can be controlled locally by diligent monitoring and periodic removal.

4.4 References

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Table 4.1. Occurrence (shoreline) of plant species in Buffalo River AOC. Asterisk (*) indicates non-native. Bold indicates nuisance invasive.

		Site									
Scientific name	**Common name**	**1**	**2**	**3**	**4**	**5**	**6**	**7**	**8**	**9**	**10**
Woody species											
Acer negundo L.	Ashleaf maple						X				
Acer saccharinum L.	Silver maple		X								
****Ailanthus altissima*** (Miller) Swingle	Tree of heaven				X		X				
Cornus sericea L.	Red osier dofwood								X	X	X
Fraxinus pensylvanica Marshall	Green ash					X					
**Lonicera morrowii* A. Gray	Morrow's honeysuckle									X	
Populus deltoides Marshall	Eastern cottonwood							X		X	X
Rhus typhina L.	Staghorn sumac		X						X		X
Salix nigra Marshall	Black willow	X	X		X	X	X	X	X	X	X
**Ulmus pumila* L.	Siberian elm		X								
Herbaceous species											
Circaea lutetiana L.	Enchanter's nightshade					X					
Cirsium arvense (L.) Scop.	Canada thistle								X		
**Coronilla varia* L.	Crown vetch	X									
Cuscuta gronovii Willd.	Dodder		X								
**Dipsacus slyvestris* Hudson	Teasel					X					
Elymus canadensis L.	Wild rye								X		
Equisetum arvense L.	Horsetail		X								
Eupatorium maculatum L.	Joepyeweed										X
Eupatorium perfoliatum L.	Joepyeweed							X			
Eupatorium purpureum L.	Joepyeweed									X	
**Euphorbia esula* L.	Leafy spurge							X			
Galeopsis tetrahit L.	Hempnettle							X			
Geum virginianum L.	Cream avens		X								
Helianthus tuberosus L.	Jerusalem artichoke				X	X					
**Hesperis matronalis* L.	Dames rocket						X				
Hypericum perforatum L.	St. Johnswort		X								
Impatiens capensis Meerb.	Jewelweed	X									
**Linaria vulgaris* Miller	Butter and eggs		X								
****Lythrum salicaria*** L.	Purple loosestrife		X					X	X	X	X
Medicago sativa L.	Alfalfa		X								
Mentha arvensis L.	Wild mint				X				X	X	
**Nepeta cataria* L.	Catnip										X
****Polygonum cuspidatum*** Sieb. & Zucc.	Japanese knotweed						X		X	X	X
Polygonum hydropiperoides Michx.	Swamp smartweed					X				X	X
Pontederia cordata L.	Pickerelweed			X							
Potentilla reptans L.	Creeping cinquefoil	X									
**Rumex obtusifolius* L.	Bitter dock				X						
Scrophularia marilandica L.	Carpenter's square					X		X			
**Sinapsis arvensis* L.	Charlock mustard	X			X						X
Solanum dulcamara L.	Climbing nightshade									X	
Solanum nigrum L.	Black nightshade							X			
Solidago gigantea Ait.	Giant goldenrod	X									
**Tussilago farfara* L.	Coltsfoot								X		
Verbena urticifolia L.	White vervain							X	X		X
Vitis riparia Michx.	Riverbank grape					X			X		
Aquatic species											
****Myriophyllum spicatum*** L.	Eurasian watermilfoil		X	X	X	X	X	X	X	X	X
**Potamogeton crispus* L.	Curlyleaf pondweed			X			X	X			
Potamogeton filiformis Pers.	Fine leaf pondweed				X	X		X			
Potamogeton sp.	Pondweed						X				X
Vallisneria americana L.	American eelgrass						X	X	X	X	X

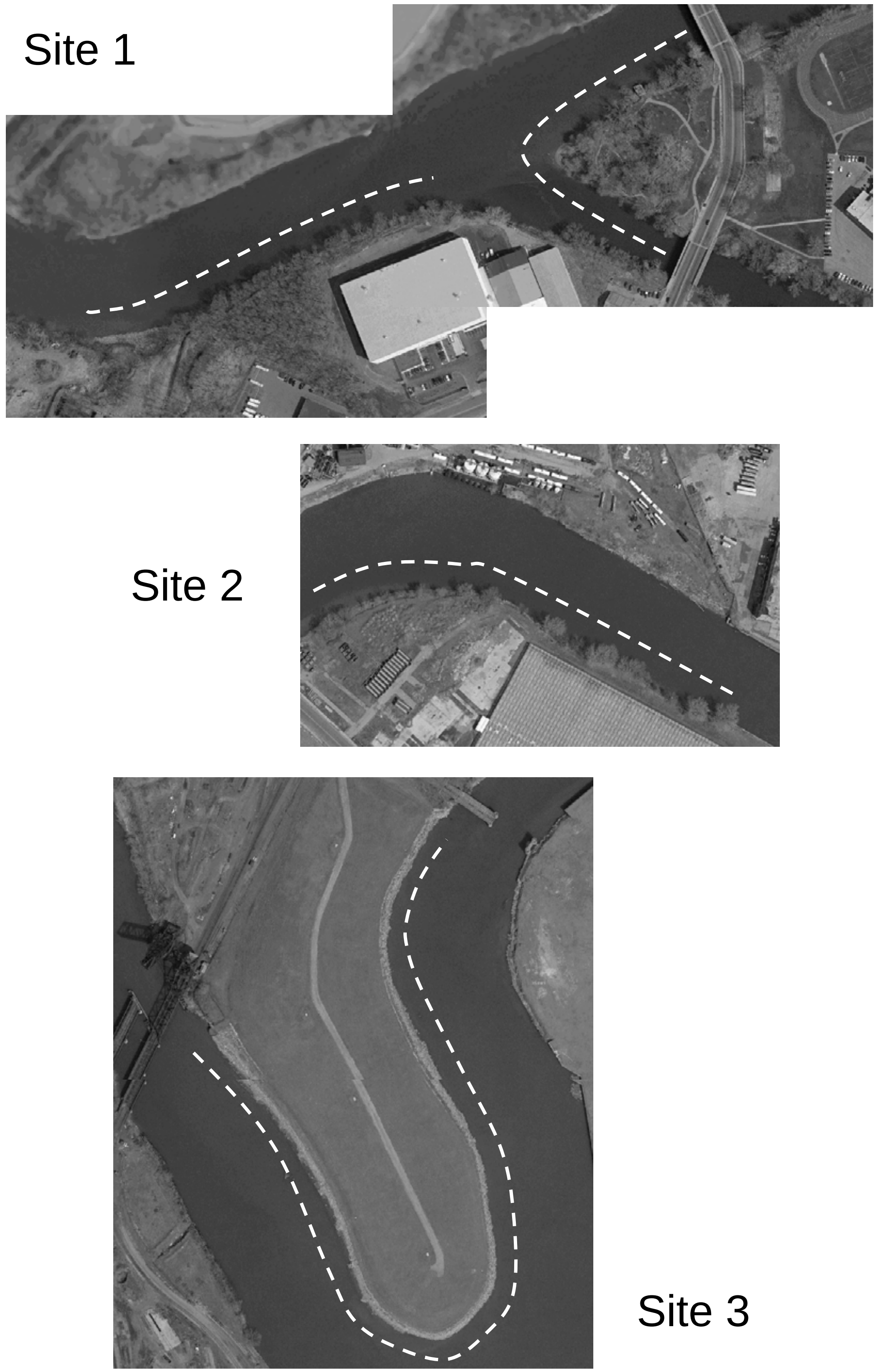
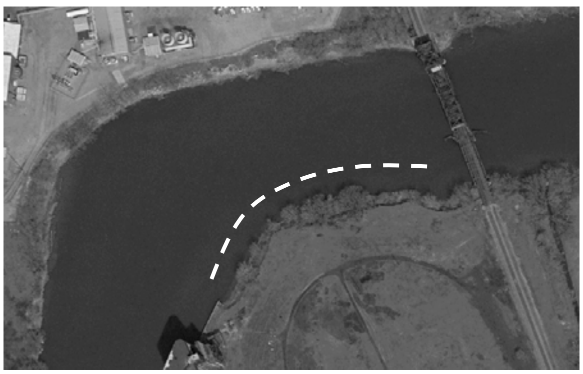
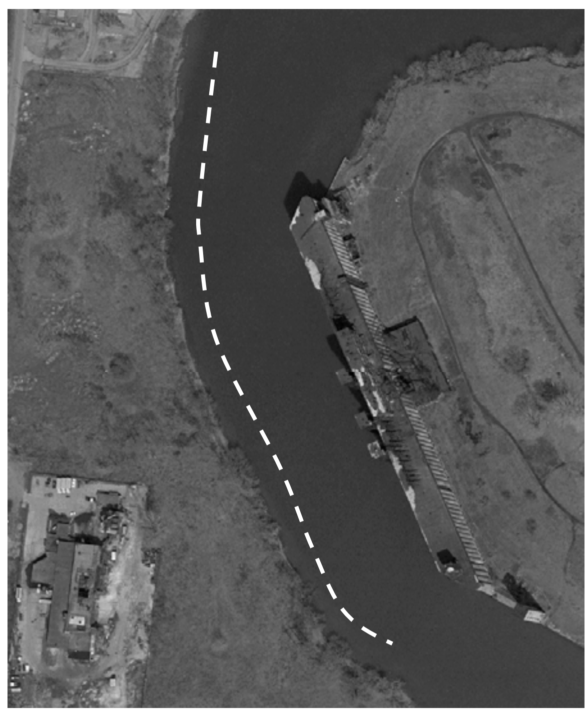


Figure 4.1 Aerial views (satellite ortho-imagery) of potential habitat restoration sites.

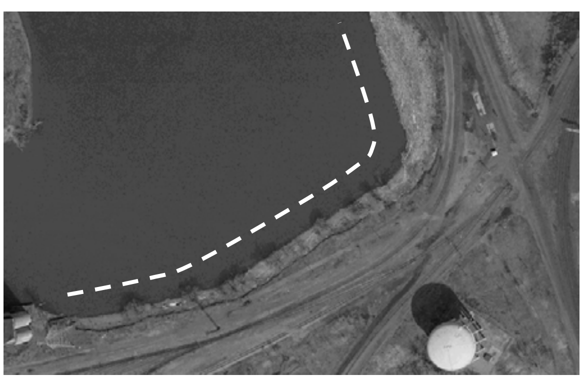
Site 4



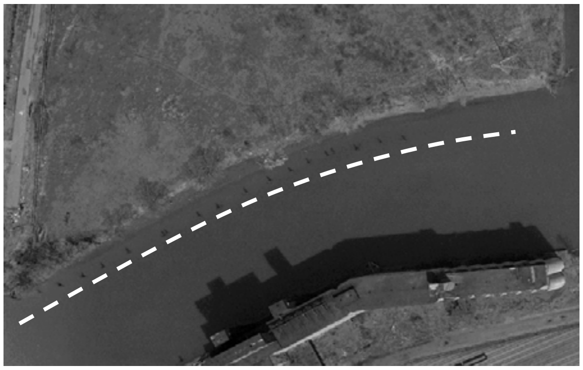
Site 5



Site 6



Site 7



Site 8

Site 9

Site 10

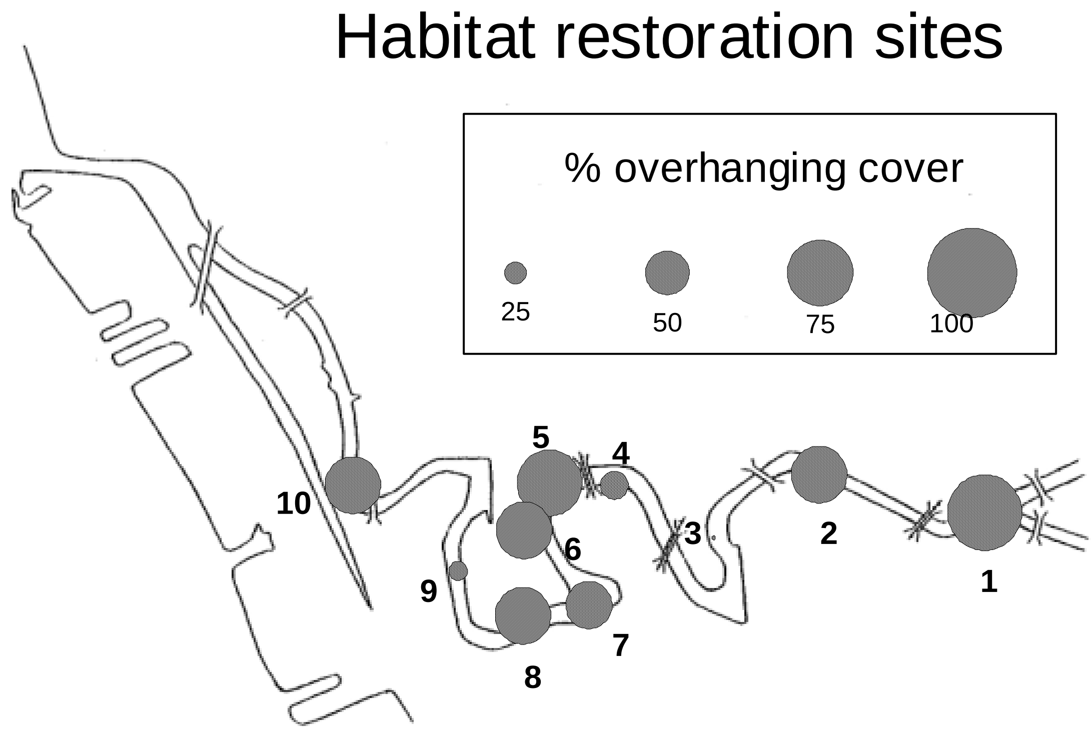


Figure 4.2 Percentage of shoreline with overhanging woody vegetation at potential habitat restoration sites.

CHAPTER 5
WATER QUALITY

K.N. Irvine

5.1 Introduction

Water quality evaluation for this project principally relied on measurement of conventional parameters, dissolved oxygen, turbidity, conductivity, temperature, and pH, using Hydrolab Datasonde 4a instrumentation. The Hydrolab Datasonde 4a's were used to continuously log these parameters at three fixed sites to gain an understanding of system dynamics during storm events and dry weather periods. In addition, a Datasonde 4a was used to measure these parameters through the water column at each habitat site, once per week. This latter sampling provided information both on vertical variability of the parameters and site-specific information for the habitat assessment.

Late in the project a new test method for *Escherichia coli* was identified, the Coliscan Easygel kit. Sampling for *E. coli* was not part of the original project scope of work, but it was decided to apply the Coliscan Easygel system for a limited number of tests to evaluate its utility for a citizen monitoring program.

5.2 Hydrolab Sample Methods

5.2.1 Continuous Logging

Hydrolab Datasonde 4a's were installed at the Seneca St. Bridge (Hydrolab Site 2), upstream of the confluence with Cazenovia Creek; at the mouth of Cazenovia Creek (Hydrolab Site 7); and at the Ohio St. Bridge (Hydrolab Site 4). The locations of these fixed sites are shown in Figure 5.1 and the specific installations are shown in Figures 5.2-5.4. The site numbers correspond to the numbers used in the Buffalo Sewer Authority Long Term Control Plan Study (e.g. Irvine et al., 2005), rather than the Habitat site numbers. The datasondes recorded pH, conductivity, temperature, dissolved oxygen, and turbidity at 15 minute time steps for the periods 6/4/03-10/6/03 and 6/2/04-9/29/04. The locations of these fixed sample sites were based on previous monitoring experience (Irvine et al., 2005) that showed they would provide a good representation of storm dynamics and dry weather flow. Furthermore, Sites 2 and 7 represent the upstream boundary conditions of the Area of Concern (AOC) impact area, while Site 4 represents the longitudinal mid-point of the federal navigable channel. Water quality monitoring also historically has been done at Site 4, so a maximum amount of additional data are available (e.g. NYSDEC, 1989; Atkinson et al., 1994; Irvine and Pettibone, 1996).

All Hydrolabs were installed so that they were contained within a capped PVC tube (Figure 5.3). The lower section of the PVC tube had holes drilled through it to allow the water to move freely past the Hydrolab sensors. The PVC tubes protected the Hydrolabs from damage due to floating storm debris in the river and the locked caps provided a level of security from tampering. At all sites the PVC tube was fixed to a stationary object (e.g. bridge abutment, rip rap) so that the sensors would be approximately 1.0 m below the March low water datum. Clearly, then, during storm events, the water depth above the sensors was greater.

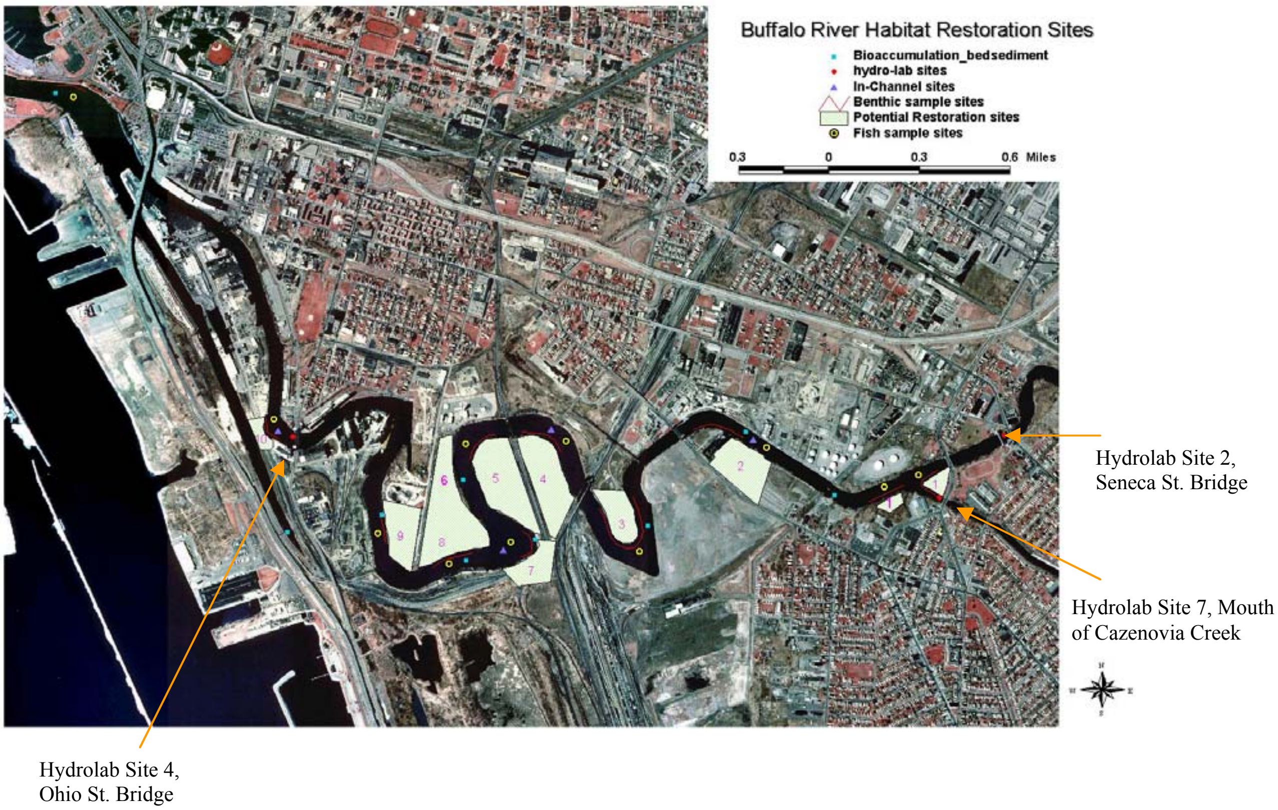


Figure 5.1 Location of sample sites; the fixed, logging Hydrolab locations are shown as red circles

Figure 5.2 Hydrolab Site 2, Seneca St. Bridge

Figure 5.3 Hydrolab Site 7, Mouth of Cazenovia Creek

Figure 5.4 Hydrolab Site 4, Ohio St. Bridge

Prior to installation for each field season, the Hydrolabs were calibrated and tested in the Black Rock Canal to help ensure the sensors were operating properly. If a problem was identified, the Datasonde was sent to Hydrolab (Loveland, CO) for factory correction. As recommended by the manufacuturer, a two minute warm-up was used for the dissolved oxygen and turbidity sensors. Although this increased battery wear, it provided the optimum time for the sensors to equilibrate.

Data from the Hydrolabs were uploaded to a laptop on a weekly basis. All data were managed and maintained in Excel spreadsheet format. During the weekly site visit to upload the data, all units were cleaned with Kimwipes and cotton swabs, and the general operation of each unit was checked. The dissolved oxygen sensors were calibrated each week using the 100% (air) saturation method as described by the manufacturer. The dissolved oxygen membranes and electrolyte were changed at the midpoint of each sampling season, or when the membrane appeared damaged. The pH sensors also were calibrated at the midpoint of each sampling season. All data were reviewed on a weekly basis in Excel format to identify any problem data. These data were flagged and discussions were held with the field crew maintaining the Datasondes to help identify and resolve the source of the problem. In general, the methodology for sampling and Datasonde maintenance followed that done previously for the Buffalo River (Irvine et al., 2005).

5.2.2 Hydrolab Profiling

A Hydrolab Datasonde 4a was used to collect pH, conductivity, temperature, dissolved oxygen, and turbidity data at all 10 habitat sites, at three depths, 0.5 m below the surface; 1.0 m below the surface; and near bottom. The profiling was done once per week for 16 weeks in 2003 and 17 weeks in 2004. The depth of the "near bottom" location was variable and depended on the site and total vertical depth at the particular time of year (Tables 5.1 and 5.2). Care was taken to ensure the Datasonde did not come into contact with the river bed, thereby resuspending sediment. If the Datasonde contacted the river bed, the instrument was moved slightly up-river and readings were taken five minutes later.

Table 5.1 Near Bottom Sample Depths (m) for Profiling, 2003

Site	1	2	3	4	5	6	7	8	9	10
Mean	2.4	4.8	5.5	5.1	6.1	5.0	4.7	5.4	5.0	6.1
Range	1.5-3.8	3.1-5.5	3.2-7.2	2.6-7.0	2.4-8.3	2.3-6.7	1.5-5.8	1.6-7.2	2.1-6.0	4.7-7.5

Table 5.2 Near Bottom Sample Depths (m) for Profiling, 2004

Site	1	2	3	4	5	6	7	8	9	10
Mean	2.8	5.8	6.5	4.9	7.8	4.5	4.5	6.0	5.1	6.2
Range	1.3-4.0	4.0-7.3	3.8-7.5	3.2-5.6	6.5-8.5	2.2-6.4	3.6-5.7	3.5-7.5	3.2-6.9	3.8-7.9

5.3 *E. coli* and Suspended Solids Sampling and Analysis

A total of four samples for *E. coli* analysis were collected at each of the three Hydrolab sites through the storm event of 9/9-10/04, while dry weather samples were collected at each of the three sites on 9/15/04, 9/22/04, and 9/29/04. Samples were collected from the mid-point of the bridges at each site by lowering a Teflon bailer into the water. Samples collected therefore represent conditions at the mid-point of the channel, near the surface (i.e. <0.5 m deep). To limit cross-contamination, a sample was collected in the bailer and discarded immediately prior to the collection of the sample used with the Coliscan Easygel kits.

Once the water sample was collected in the bailer, between 1 and 5 mL was withdrawn using a sterile, disposable pipette. The smaller volume was used for storm events and the larger volume was used for dry weather samples. The pipetted water was then placed in a plastic bottle of Coliscan Easygel media (Micrology Laboratories, LLC, Goshen, IN; http://www.micrologylabs.com/html/detecting_waterborne.html). The medium/inoculum mix was placed in a cooler on ice until returned to the Buffalo State Field Station for further processing.

At the Buffalo State Field Station, the bottles were swirled to distribute the inoculum and the medium/inoculum mixtures were carefully poured into labeled petri dishes. The lids were placed back on to the petri dishes and the poured dishes were gently swirled until the entire dish was covered with liquid. Each petri dish came pre-treated with a thin coating of material containing calcium ions. When the medium/inoculum is poured into the coated dish, the ions diffuse up through the medium and complex with the gelling agents, causing a solid gel to form. The system uses a temperature-independent gelling agent (low methoxyl pectins) that avoids the disadvantages of agar. The dishes were incubated at room temperature for 72 hours and the purple colonies were counted as *E. coli.*

Samples routinely were collected once per week at Sites 2, 4, and 7 for analysis of total suspended solids (TSS) concentration and additional samples were collected during storm events. Samples were filtered at the Buffalo State Field Station, following Standard Methods (APHA, 1985), although 0.45 μm Millipore filters were used rather than glass fiber filters.

5.4 Results and Discussion

5.4.1 Mean Conditions from Fixed Hydrolab Monitoring

The 15 minute data plotted on a weekly basis for each of the three fixed sites are provided on the attached CD. However, as a first step to summarizing the Hydrolab data, the weekly mean values of the sampled parameters for 2003 and 2004 are shown in Figures 5.5 and 5.6. The temperature and dissolved oxygen data in both 2003 and 2004 exhibited a level of seasonality. For example, temperature increased from early June, remained relatively stable from the end of June to early September and then decreased through September. Conversely, dissolved oxygen tended to be higher during the cooler water periods of early June and September. These seasonal trends were even more pronounced in 2000, when the monitoring period extended from 4/17/00 to 10/18/00 (30 weeks)(Irvine et al., 2005). Dissolved oxygen tended to be lower at Hydrolab Site 4 than Sites 2 and 7. This trend is discussed in more detail in Section 5.4.2. Diggins and Snyder (2003) reviewed dissolved oxygen data reported in several studies over a 30 year period, 1964-1993 and concluded that prior to 1970, levels within the AOC routinely were <1 mg L^{-1}. Dissolved oxygen levels increased to 1-4 mg L^{-1} by the early to mid-1970's and from 1982 to 1992 the levels stabilized in the 5-6 mg L^{-1} range. The pH values of between 7 and 8 were consistent with those recorded for the same sites in 2000 (Irvine et al., 2005). The pH values of around 9 at Hydrolab Site 2 for the first part of 2004 probably reflect a faulty pH sensor and should be disregarded. The mean conductivity values exhibit variability, but in general are consistent with those recorded for the same sites in 2000 (Irvine et al., 2005). The effects of larger storm events and associated increases in sediment concentration (and turbidity) due to erosion can be seen even at the weekly mean temporal scale, for example, in early June, 2003; nearly weekly events through July, 2004; end of August, 2004; and early September, 2004. Mean turbidity at all sites during dry weather was quite low, less than 20 NTU.

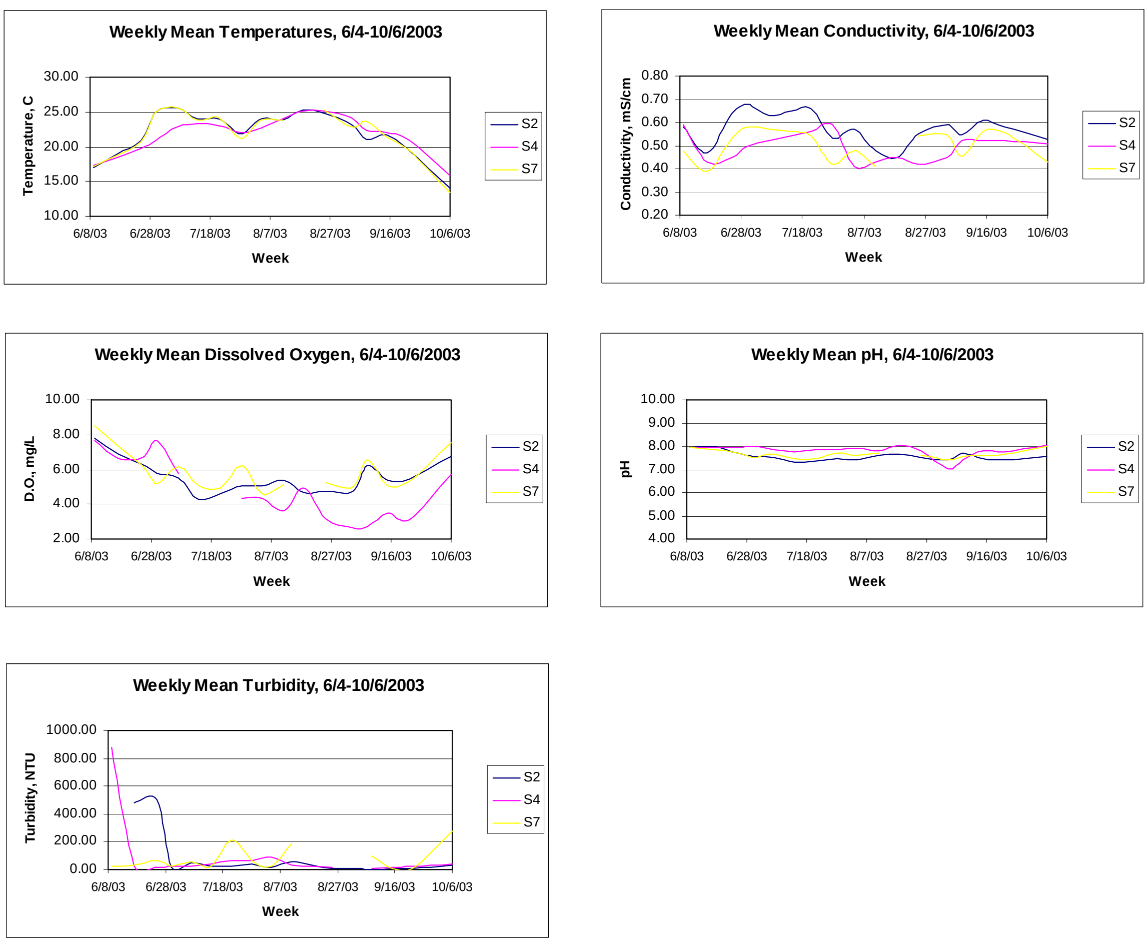


Figure 5.5 Weekly mean Hydrolab values, 2003

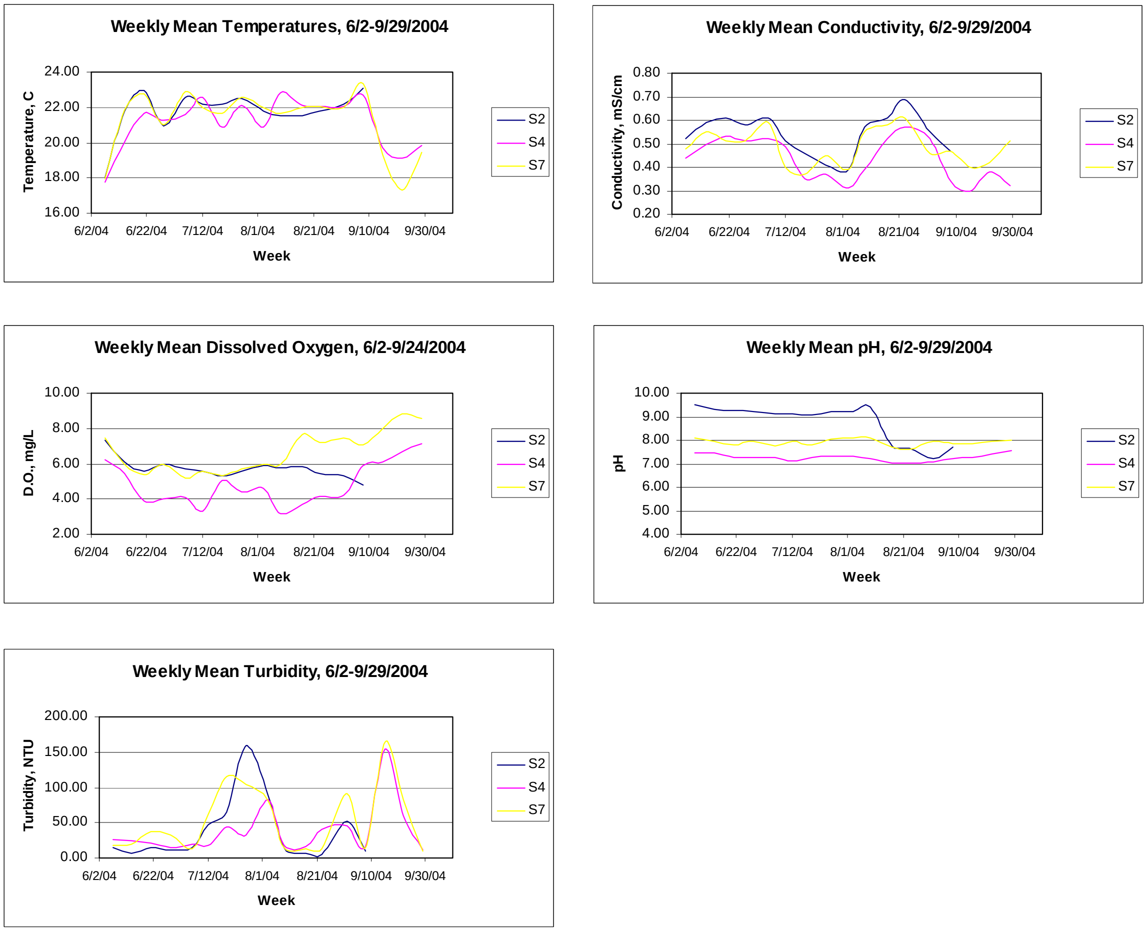


Figure 5.6 Weekly mean Hydrolab values, 2004

5.4.2 Dissolved Oxygen Guidelines

New York State guidelines for dissolved oxygen in class C, non-trout waters state "… the minimum daily average shall not be less than 5.0 mg L^{-1}, and at no time shall the DO concentration be less than 4.0 mg L^{-1}." Daily mean dissolved oxygen levels were calculated and the days for which the mean level was less than 5.0 mg L^{-1} were identified (Table 5.3). There was a higher percentage of non-compliance days at both upstream sites (Hydrolab Sites 2 and 7) in 2003 as compared to 2004, but in both years, Hydrolab Site 4 (Ohio St. Bridge) had a higher percentage of non-compliance days than either of the two upstream sites. Irvine et al. (2005) also found Site 4 had a higher number of non-compliance days (66 over the 30 week sample period) as compared to Sites 2 and 7. In general, there were fewer days of non-compliance at Hydrolab Sites 2 (0) and 7 (30) in 2000 as compared to 2003 and 2004 (Irvine et al., 2005).

The times during which the dissolved oxygen levels were <4.0 mg L^{-1} at each site also were identified and these are compared (as a percentage of time) to the total study time in Table 5.4. In 2000, the per cent time dissolved oxygen levels were <4.0 mg L^{-1} were reported as: Site 2 – 0%; Site 4 – 28%; Site 7 – 4.7% (Irvine et al., 2005).

In general, the spatial trend of higher dissolved oxygen levels near the top of the AOC and lower dissolved oxygen levels along the middle section was consistent in the data for 2000, 2003, and 2004. Irvine et al. (2005) noted that in 2000 between Sites 6 (Cazenovia Park) and 7, there was an increase in the proportion of days <5.0 mg L^{-1} (as well as an increase in the frequency of periods <4.0 mg L^{-1}) associated with storms. It is possible that this increase was related to the cumulative impact of the CSOs along the channelized section of Cazenovia Creek, although it is important to note that the creek also becomes wider and deeper in the channelized section between Sites 6 and 7. The Site 7 data were reviewed in more detail for six weeks in which multiple CSO events were recorded. Typically, it was not possible to visually identify a dissolved oxygen sag associated with an individual CSO event. In two cases, a small decrease in dissolved oxygen occurred at Site 7 after the third CSO event within the week. It is possible that the dissolved oxygen dynamics between Sites 6 and 7 were influenced by the change in channel characteristics (wider and deeper channel with lower velocity) and infrequently occurring hydrologic conditions (i.e. multiple CSO events in a short period of time).

Several dissolved oxygen modeling studies were completed through the 1990's for the Buffalo River (Blair, 1992; Wight, 1995; Hall, 1997) and these efforts were more recently expanded by Jaligama et al. (2004). These studies concluded that low dissolved oxygen, particularly in the upper to central portion of the Buffalo River, was related to a combination of stratification in the river at low flows that can reduce aeration, high sediment oxygen demand, together with long residence times due to system hydraulics (in particular, dredging increases channel cross-sectional area and residence time), and background biochemical oxygen demand (see Figure 5.7). The modeling efforts concluded that CSOs discharging to the river had minimal impact on dissolved oxygen.

Table 5.3 Number (and Per Cent) of Days when Daily Mean Dissolved Oxygen was <5.0 mg L^{-1} during the Periods 6/4/03-10/6/03 and 6/2/04-9/29/04

	Site 2, 2003	Site 4, 2003	Site 7, 2003	Site 2, 2004	Site 4, 2004	Site 7, 2004
# of Days <5.0 mg L^{-1}	42	70	36	12	72	12
% of Days <5.0 mg L^{-1}	34	59	31	12	61	10

Table 5.4 Per Cent of Time when Dissolved Oxygen was <4.0 mg L^{-1} during the Periods 6/4/03-10/6/03 and 6/2/04-9/29/04

	Site 2, 2003	Site 4, 2003	Site 7, 2003	Site 2, 2004	Site 4, 2004	Site 7, 2004
% of Time <4.0 mg L^{-1}	7.5	41	7.7	1.5	37	2.2

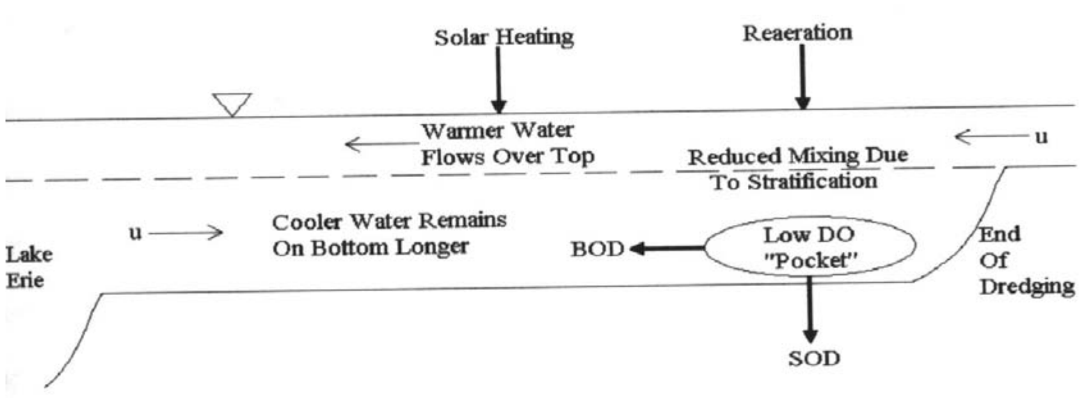


Figure 5.7 Factors influencing development of low dissolved oxygen levels in the AOC (from Hall, 1997)

5.4.3 Storm Event Dynamics

Averaging the Hydrolab data, as done in Section 5.4.1, provides information on general trends, but the averaging masks some of the important system responses to specific storm characteristics. A visual review of the weekly data plotted at the 15 minute time steps revealed some interesting trends, particularly for turbidity, conductivity, and dissolved oxygen.

Conductivity levels frequently exhibited a precipitous dip, in association with storm events (Figures 5.8-5.10). Constituents from chemical weathering of soils and bedrock may predominantly enter rivers in temperate, humid climates via groundwater inputs (Marsh, 1987; Morisawa, 1968). As such, conductivity and dissolved solids concentration would be greater during baseflow conditions, when the principal hydrologic input is groundwater and may become diluted by stormwater runoff (Walling and Webb, 1980). Tomlinson and De Carlo (2003) observed a dilution effect for conductivity during storms in their monitoring of streams in Hawaii, as did Irvine (2003) in a study of the Allegheny River, PA, and Irvine et al. (2005) for the Buffalo River monitoring of 2000. There can be some exceptions to this dilution pattern, as is shown for the storm event of 8/6/03 (Figure 5.11). Although conductivity dropped at the upstream Hydrolab Sites, 2 and 7, the dilution effect was dampened by the time the poorly-defined storm wave reached Hydrolab Site 4 (Figure 5.11).

Turbidity is an optical property that has a strong relationship with total suspended solids (TSS) concentration and therefore turbidity can be indicative of the suspended sediment transport dynamics. The relationship between turbidity and TSS is explored in more detail in Section 5.4.4. Turbidity always increased in response to storm events, as overland runoff introduced sediment from sheet and rill erosion and greater stream power resulted in increased bed and bank erosion. However, depending on the storm event, turbidity exhibited different between-site response. Frequently, for moderate-sized events, the upstream Hydrolab sites (2 and 7) had higher peaks, while the downstream site (4) had a lower peak that was offset (Figures 5.12 and 5.13). Daily mean flow entering the AOC from upper watershed (adjusted for ungauged area, see Section 1.2) was 3,697 cfs (105 m^3s^{-1}) for the event of 7/16/04 (Figure 5.12) and 4,227 cfs (120 m^3s^{-1}) for the event of 7/2704 (Figure 5.13). The lower peak turbidity at Hydrolab Site 4 for these events indicated that sediment was depositing in the downstream direction, which is consistent with the aggrading nature of the AOC and the need for dredging to maintain the depth of the navigable channel. The offset simply represents the lag of the storm wave movement downstream. Turbidity peaks for smaller storm events may be evident at the upstream sites, but sedimentation along the channel may reduce the peak to the extent that it is not particularly distinctive at the downstream (Site 4) location (Figure 5.14). Daily mean discharge entering the AOC (adjusted for ungauged area) for the event of 8/6/03 (Figure 5.14) was 1,165 cfs (33 m^3s^{-1}). Meredith and Rumer (1987) modeled sediment transport dynamics in the Buffalo River using HEC-6 and as part of the report, identified depositional areas within the AOC for moderate sized events ($6{,}000<Q<20{,}000$ cfs)($170<Q<566$ m^3s^{-1}) . The depositional area map developed by Meredith and Rumer (1987) is shown in Figure 5.15. Clearly, deposition is predicted to occur in the areas of the meander bends between sites 2 and 7 (upstream) and Site 4 (Ohio St. Bridge at transect 9671 in Figure 5.15). In general, the model results are consistent with the Hydrolab data.

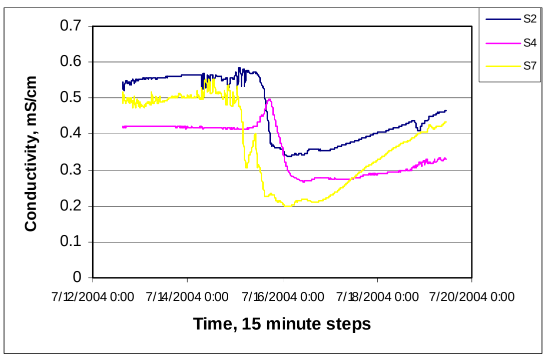


Figure 5.8 Dilution of conductivity, 7/16/04 event

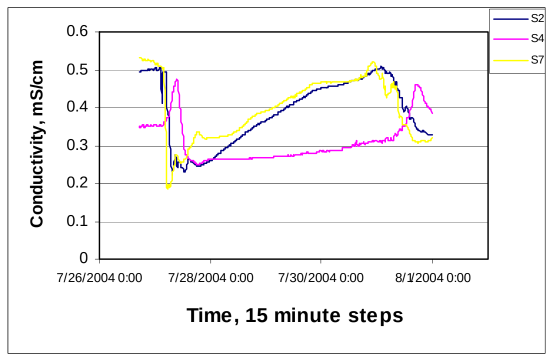


Figure 5.9 Dilution of conductivity, 7/27/04 event

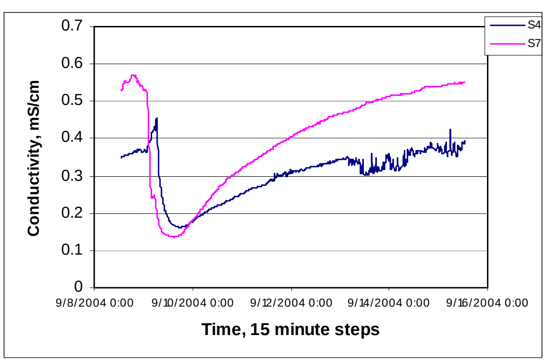


Figure 5.10 Dilution of conductivity, 9/9/04 event

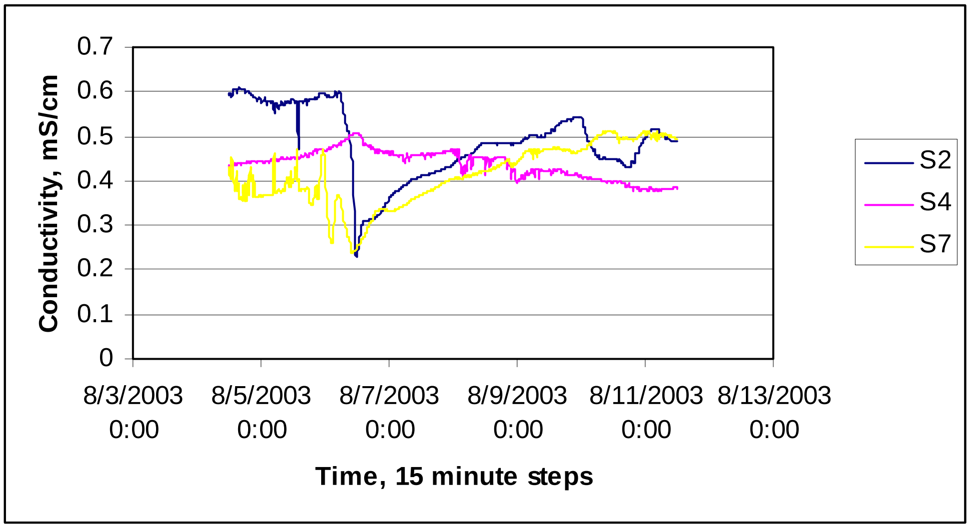


Figure 5.11 Storm event of 8/6/03. The dilution effect on conductivity is more apparent for the upstream sites (2 and 7) as compared to the downstream site (4)

Although the turbidity for a larger storm event (9/9/04) increased at the Hydrolab sites there was not the characteristic decrease in turbidity between the upstream and downstream sites (Figure 5.16). It should be noted that in this case the event was sufficiently large that it dislodged the Hydrolab at Site 2 and all data were lost. The "clipped" nature of the time series in Figure 5.16 occurred because 1,000 NTU is the maximum value that can be measured by the Datasonde 4a sensor. The *rise* in turbidity at Hydrolab Site 4 is lagged compared to Site 7 because of its downstream location (Figure 5.16). However, the falling limb of the event at Hydrolab Site 4 is not lower than Site 7, as was the case in Figures 5.12 and 5.13. It appears that the transport capacity associated with the storm of 9/9/04 (Figure 5.16) was sufficient to maintain sediment movement in this stretch of the river and there was minimal or no deposition. Peak flows for this event were approximately 9,500 cfs (269 m^3s^{-1}) at the USGS gauge on Buffalo Creek; 8,000 cfs (226 m^3s^{-1}) on Cayuga Creek; and 14,500 cfs (410 m^3s^{-1}) on Cazenovia Creek. A review of annual peak flows showed that since 1938, the Buffalo Creek peak has only been exceeded in five years; since 1937 the Cayuga Creek peak has been exceeded in 9 years; and since 1941 the Cazenovia Creek peak has never been exceeded. Photos of Hydrolab Sites 2, 4, and 7 during typical dry weather flow and the 9/9/04 storm are shown in Appendix 5.1 to provide a sense of the storm magnitude. Meredith and Rumer (1987) indicated that at flows in excess of 20,000 cfs "….. a significant amount of sand is being removed from the river bed and transported to the Buffalo harbor area…… (on the order of 7,000 tons per day)". Again, model results are consistent with the Hydrolab data, indicating that for larger storms deposition will be minimal and in fact, the system may experience net erosion.

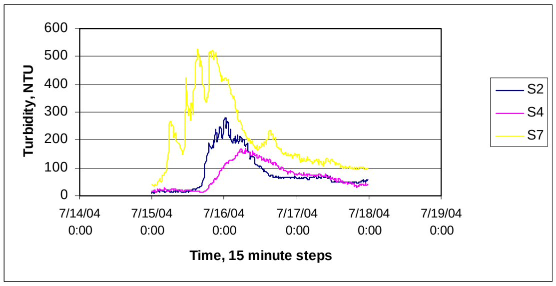


Figure 5.12 Storm event of 7/16/04

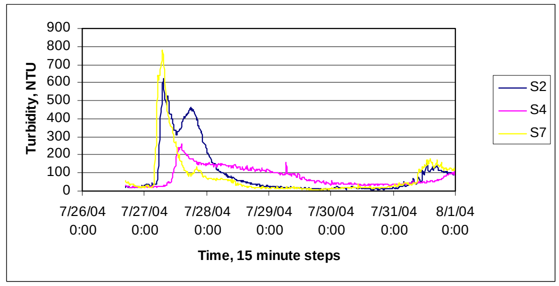


Figure 5.13 Storm event of 7/27/04

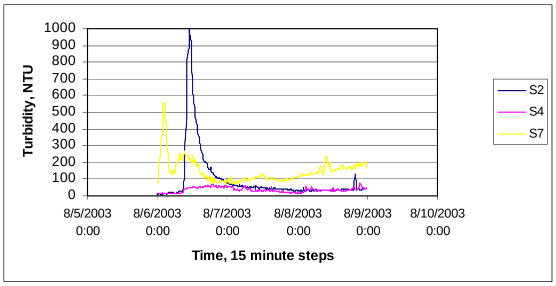


Figure 5.14 Storm event of 8/6/03

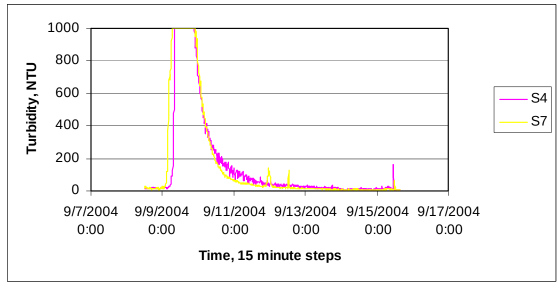


Figure 5.16 Storm event of 9/9/04

Irvine et al. (2005) conducted a detailed examination of dissolved oxygen levels associated with storm events as compared to dry weather periods for the Buffalo River AOC. For 19 storm events, Irvine et al. (2005) compared the mean 72-hour antecedent dissolved oxygen level with the storm event mean concentration and got mixed results. For 6 of 10 sample sites (including Hydrolab Site 2), the mean dissolved oxygen level was lower for storm events than dry weather periods. For 4 of 10 sample sites (including Site 7 and Site 4) the mean dissolved oxygen level was higher for storm events than dry weather periods. Irvine et al. (2005) also noted that for individual storms dissolved oxygen levels could be lower or higher than antecedent dry periods at a particular site.

It did appear that during the drier summer months when the dredged channel becomes nearly stagnant, dissolved oxygen might increase at Hydrolab Site 4 during storms, as part of the flushing associated with increased flow (e.g. Figure 5.17), although this was not always the case (e.g. Figure 5.18).

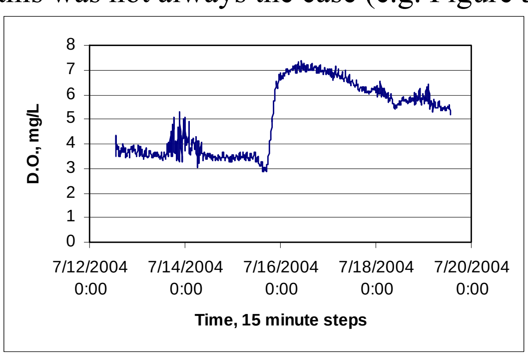


Figure 5.17 Example of increasing D.O. at Site 4, storm event of 7/16/04

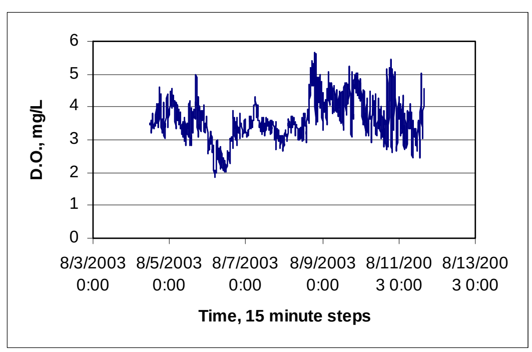


Figure 5.18 D.O. Site 4, event of 8/6/03

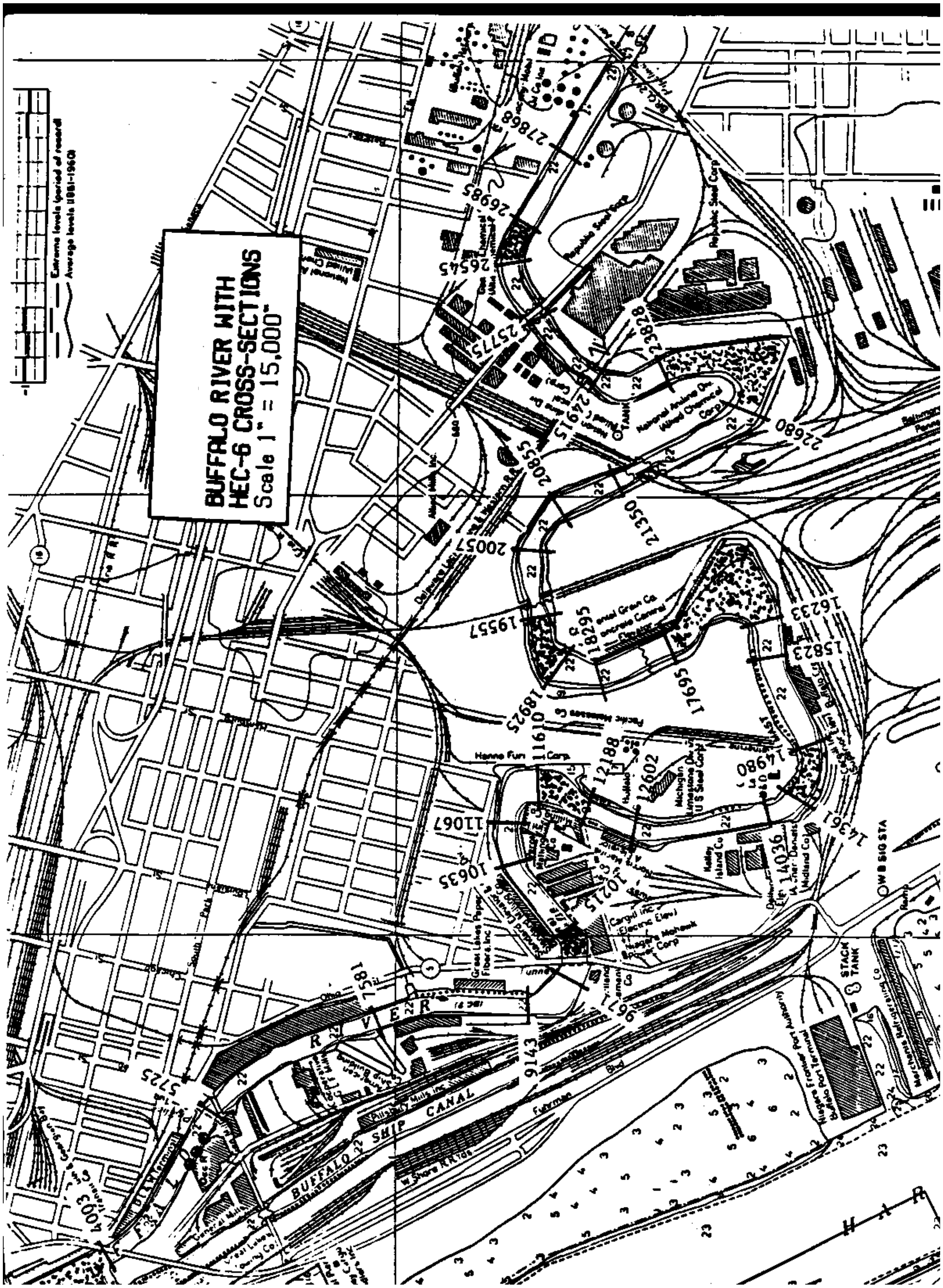


Figure 5.15 Depositional areas (shaded by dots) for moderate sized events (from Meredith and Rumer, 1987)

A diurnal fluctuation in dissolved oxygen was observed at Hydrolab Sites 2 and 7 (e.g. Figure 5.19), particularly during dry periods early in the sample season (June and early July). Algae and rooted aquatic plants can deliver oxygen to the water through photosynthesis (Mitchell and Stapp, 1995) so that dissolved oxygen levels rise from the morning and peak in late afternoon/early evening. At night, aquatic organisms continue to respire, consuming oxygen, and therefore the dissolved oxygen levels begin to decline through to the next morning. A diurnal fluctuation typically was not observed at Hydrolab Site 4.

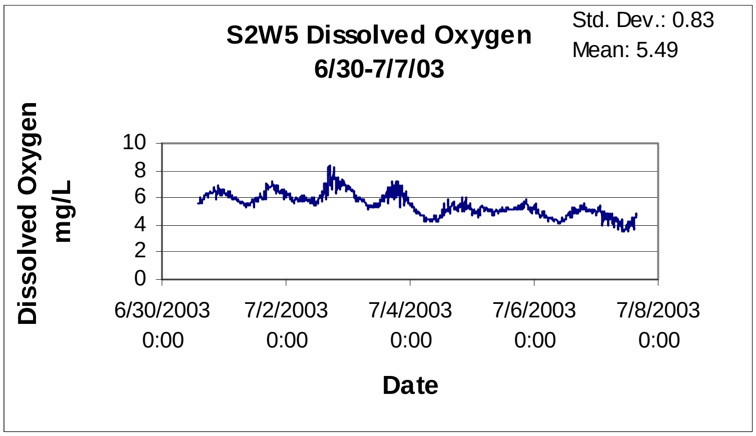


Figure 5.19 Example of diurnal pattern in dissolved oxygen, Site 2

5.4.4 Turbidity-TSS Relationships

Numerous studies have examined the relationship between turbidity and total suspended solids (TSS) in an effort to improve our ability to evaluate watershed-scale erosion and sediment transport dynamics (e.g. Lewis, 1996; Sun et al., 2001; Davies-Colley and Smith, 2001; Irvine et al., 2003). There can be several advantages to using automated turbidity measurements as a surrogate for TSS sampling in the examination of sediment erosion and transport. These advantages include the capability of providing fine time resolution measurements for extended periods, without having to rely on sampling teams to catch transient storm events with minimal notice (i.e. keeping teams “on call” to chase storms), and reduction of laboratory costs for the analysis of TSS. Ultimately, the success of using turbidity measurements in place of TSS sampling relies on the accuracy of the TSS-turbidity rating curve. Pfannkuche and Schmidt (2003) reported an r^2 of 60% between suspended sediment and turbidity, while others (e.g. Lewis, 1996; Davies-Colley and Smith, 2001; Tomlinson and De Carlo, 2003) have produced rating curves with higher r^2 values. Variability in the relationship may be related to a variety of factors, including changes in particle size, shape and composition, as well as the presence of humic acids.

Perrelli et al. (2005) used the rating curves developed by Irvine et al. (2003) to transform the 2000 turbidity time series into a TSS time series. Subsequently, the TSS time series was used to calibrate the erosion and sediment transport component of the HSP-F model for the watershed (e.g. Figure 5.20).

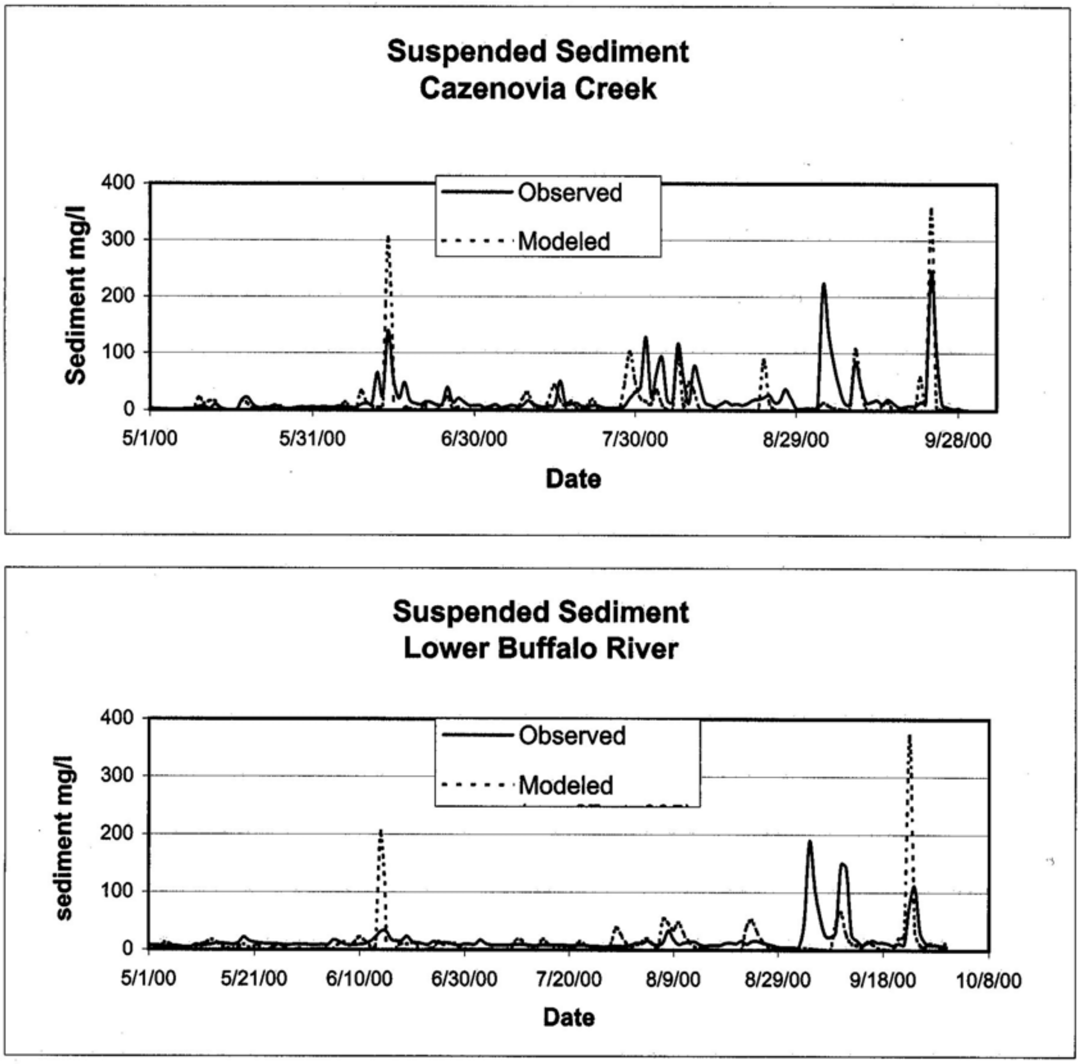


Figure 5.20 Model calibration results for suspended sediment estimates at Site 7 (top) and near Site 4 (bottom)(from Perrelli et al., 2005)

In the current study, the TSS concentrations were compared to turbidity measurements to develop a TSS-turbidity rating curve. To filter possible extraneous turbidity readings, the rating curve was developed using an average of the four turbidity measurements taken during the hour that the sample for TSS analysis was collected. Initial scattergrams and subsequent residual analysis indicated that a logarithmic transformation of the raw data was appropriate for the final form of each rating curve. The rating curves were developed using the combined 2003 and 2004 data for each site and the results are shown in Figure 5.21. The slopes of the regressions were significantly different from 0 (α=0.05) and the r^2 values are comparable to those reported in other studies. Irvine et al. (2003), working with a smaller data set from the 2000 sampling effort, found it was not necessary to logarithmically transform the data for Hydrolab Sites 4 and 7 and reported r^2 values of 74% and 72%, respectively. Sun et al. (2001) concluded that turbidity-TSS relationships may be both site and time-specific, so that a relationship is normally unique for a particular catchment and within a particular period of time. Certainly, Figure 5.21 indicates there is some spatial variability of the turbidity-TSS relationship (based on differences in the regression slopes and intercepts).

5.4.5 Habitat Site Water Column Profiling with Hydrolab Datasonde

The raw water column profiling data for all sample sites in 2003 and 2004 are included on the attached CD. Probably of greatest interest for this project are the dissolved oxygen and turbidity levels. The mean values of dissolved oxygen and turbidity at each sample depth for 2003 and 2004 are shown in Tables 5.5 and 5.6.

Table 5.5 Mean Dissolved Oxygen (mg L^{-1}) and Turbidity (NTU) Based on Weekly Samples, 6/11-9/24/03

Site	D.O., 0.5 m depth	D.O. 1.0 m depth	D.O., near bottom	Turbidity, 0.5 m depth	Turbidity, 1.0 m depth	Turbidity, near bottom
1	7.05	6.98	6.68	16.21	16.81	24.4
2	6.38	6.13	4.32	19.63	22.97	92.18
3	5.94	5.75	3.69	21.23	22.54	110.09
4	5.87	5.66	3.14	15.63	15.49	40.49
5	5.70	5.40	3.25	15.45	14.85	29.45
6	5.82	5.53	5.46	13.69	13.79	92.71
7	5.81	5.63	3.60	12.08	17.04	39.11
8	6.00	5.67	3.87	15.39	13.21	25.95
9	5.95	5.67	4.20	11.58	12.77	38.63
10	5.72	5.39	4.38	9.59	10.24	17.55

Table 5.6 Mean Dissolved Oxygen (mg L^{-1}) and Turbidity (NTU) Based on Weekly Samples, 6/25-9/24/04

Site	D.O., 0.5 m depth	D.O. 1.0 m depth	D.O., near bottom	Turbidity, 0.5 m depth	Turbidity, 1.0 m depth	Turbidity, near bottom
1	7.11	7.01	6.68	16.11	17.05	24.40
2	6.31	6.09	4.32	20.09	23.61	92.18
3	5.73	5.54	3.41	20.91	22.64	115.61
4	5.60	5.38	3.14	14.18	14.02	40.49
5	5.43	5.13	2.93	13.76	12.91	29.76
6	5.55	5.23	5.23	11.35	11.60	98.58
7	5.58	5.36	3.30	9.94	10.80	39.92
8	5.79	5.49	3.58	10.84	11.02	24.0
9	5.78	5.49	3.94	9.86	11.30	39.94
10	5.41	5.08	4.10	8.76	9.66	17.51

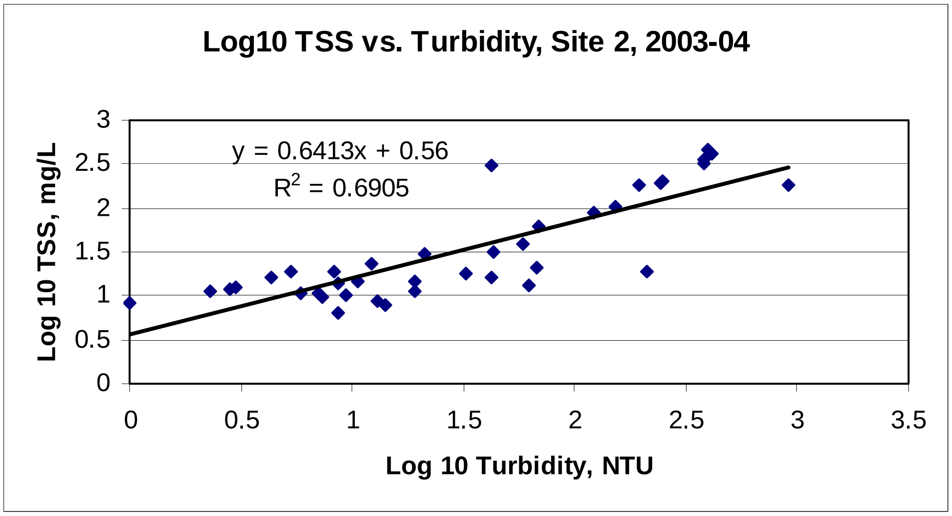


Figure 5.21

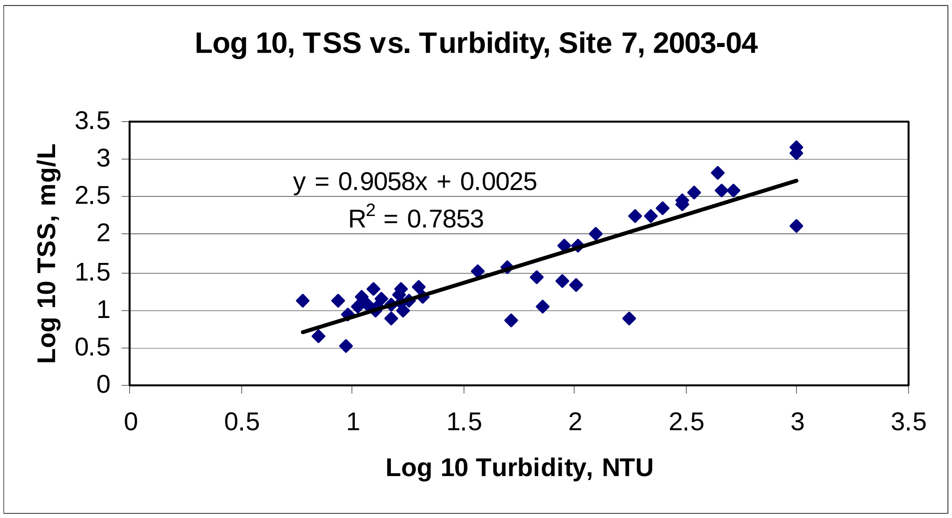
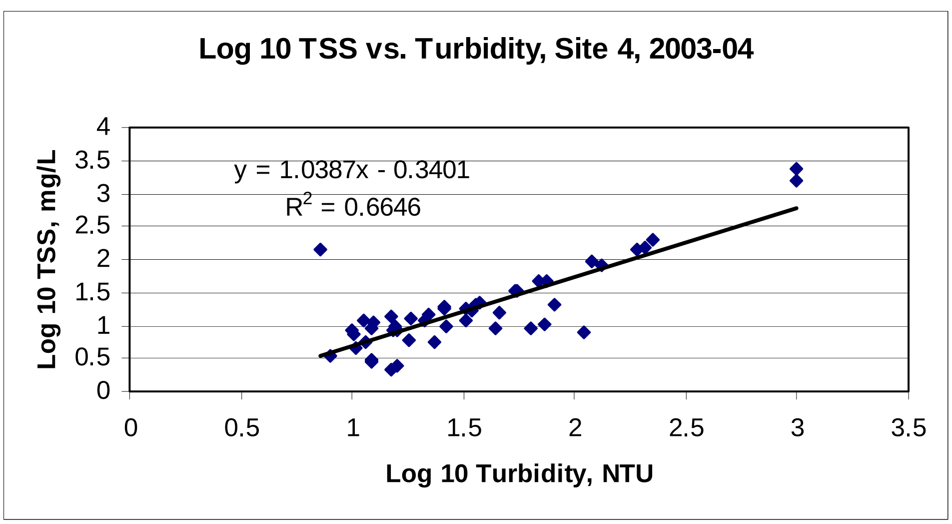


Figure 5.21



The data for each site are consistent between the two years, 2003 and 2004 (Tables 5.5 and 5.6). There is a general trend of decreasing dissolved oxygen and increasing turbidity with depth. The decrease in dissolved oxygen levels is consistent with the forcing factors described in Figure 5.7. In particular, the low dissolved oxygen near the bottom may be impacted by sediment oxygen demand (Jaligama et al., 2004). Suspended solids concentrations generally are higher near a river's bed, as a result of the combination of material settling and some near-bed saltation (temporary uplift and transport). Therefore, it is not surprising that turbidity readings were the highest near the bed. When considering the analysis of the data logged at fixed depths (Hydrolab Sites 2, 4, and, 7), it is important to keep in mind the potential for vertical variation. For example, the frequency of dissolved oxygen non-compliance (Tables 5.3 and 5.4) reflect a depth of 1-2 m; a greater frequency of non-compliance probably would occur in waters closer to the river bed.

Tables 5.5 and 5.6 show that, on average, Site 1 had the highest dissolved oxygen levels within the water column, as compared to the other habitat sites. Site 1 is upstream of the dredged navigable channel. Site 2, located in the upper portion of the dredged navigable channel, also had higher mean dissolved oxygen levels for the upper 1 m. However, dissolved oxygen levels decreased an average of 1.8-2.0 mg L^{-1} towards the riverbed at Site 2, a situation that seems to be well-explained by Figure 5.7. It is interesting that Site 6 did not exhibit the strong vertical trend in dissolved oxygen levels that were recorded at the other sites (Tables 5.5 and 5.6) and, on average for both years, the near-bed dissolved oxygen levels at this site were higher than the surrounding sites. A review of the data did not indicate that the higher levels of dissolved oxygen at this site were related to cooler water temperatures (e.g. due to shading from tree along the riverbank), but aquatic plant growth is apparent in this section of the river, which may contribute to the higher levels.

Mean turbidity typically was 20 NTU or less in the upper 1 m for all sites and there was a weak trend towards lower turbidity at Site 1, increasing turbidity at Sites 2 and 3, and then a general decrease in turbidity in the downstream direction from Site 3. The near bottom turbidity measurements exhibited a similar spatial trend, although as noted previously, the turbidity readings at this depth generally were higher, in the 20-100 NTU range, on average.

5.4.6 Profile vs. Continuous Logging Results

As noted in Section 5.2.1., Hydrolab continuous logging Sites 2 and 7 were selected to represent the upstream boundary conditions of the AOC, while Site 4 (Ohio St. Bridge) was selected to represent the longitudinal mid-point of the federal navigable channel. It was not intended that the logging sites necessarily would reflect the water quality conditions of the specific habitat sites. In fact, it was expected that there would be some spatial variation (e.g. Irvine et al., 2005). However, it is worth investigating whether the continuously logged data could be used to reflect conditions at the individual habitat sites. It was shown in Section 5.4.5 that variation was observed in the sample verticals at the habitat sites. Therefore, the habitat locations with greatest probability of being similar to the logging sites were those in close proximity, at a 1 m depth.

Figure 5.22 shows the relationship between the different water quality parameters at Hydrolab logging Site 4 (Ohio St. Bridge) and Habitat Site 9 (see Figure 5.1) and Figure 5.23 shows the relationship between the different water quality parameters at Hydrolab logging Site 7 (mouth of Cazenovia Creek) and Habitat Site 1 (see Figure 5.1). The relationships were developed by averaging the logged Hydrolab data within the three hour period that the water column profiling was done. Temperature, conductivity, and dissolved oxygen showed strong, positive trends between Hydrolab logging Site 4 and Habitat Site 9. The turbidity and pH unexpectedly showed negative, although weaker, relationships. It would be possible to consider the continuous data from logging Site 4 as reflecting the general conditions of Habitat Site 9 (1 m depth) for temperature, conductivity, and dissolved oxygen. The turbidity and pH values are in the same general range for logging Site 4 and Habitat Site 9, but the temporal trend is not consistent.

Weak positive trends were observed in Figure 5.23 for all parameters. Profiling was done at the downstream location of Habitat Site 1 (Figure 5.1), and the site, in fact, would represent a mix of conditions between Cazenovia Creek and the upper Buffalo River. It is not surprising that the relationship between Hydrolab logging Site 7 (mouth of Cazenovia Creek) and Habitat Site 1 are relative weak, because of the mixing at Habitat Site 1.

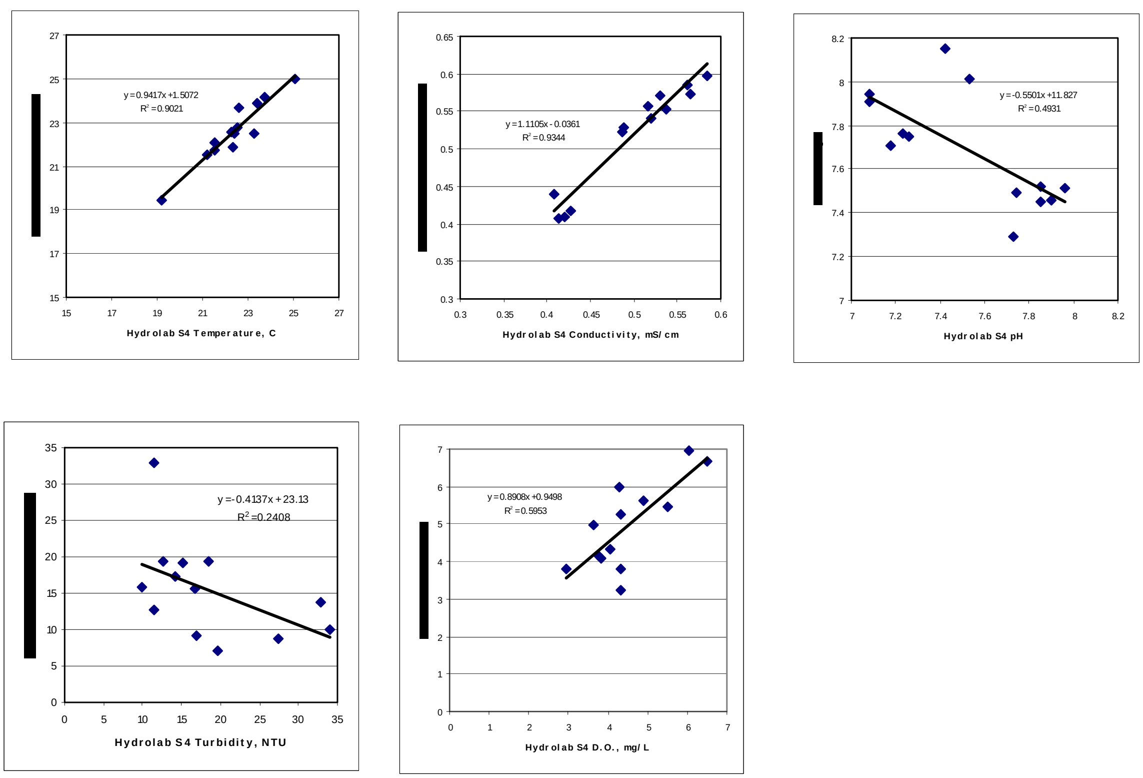


Figure 5.22 Hydrolab continuously logged data (Hydrolab Site 4) vs. water column profile results (1 m depth) at Habitat Site 9

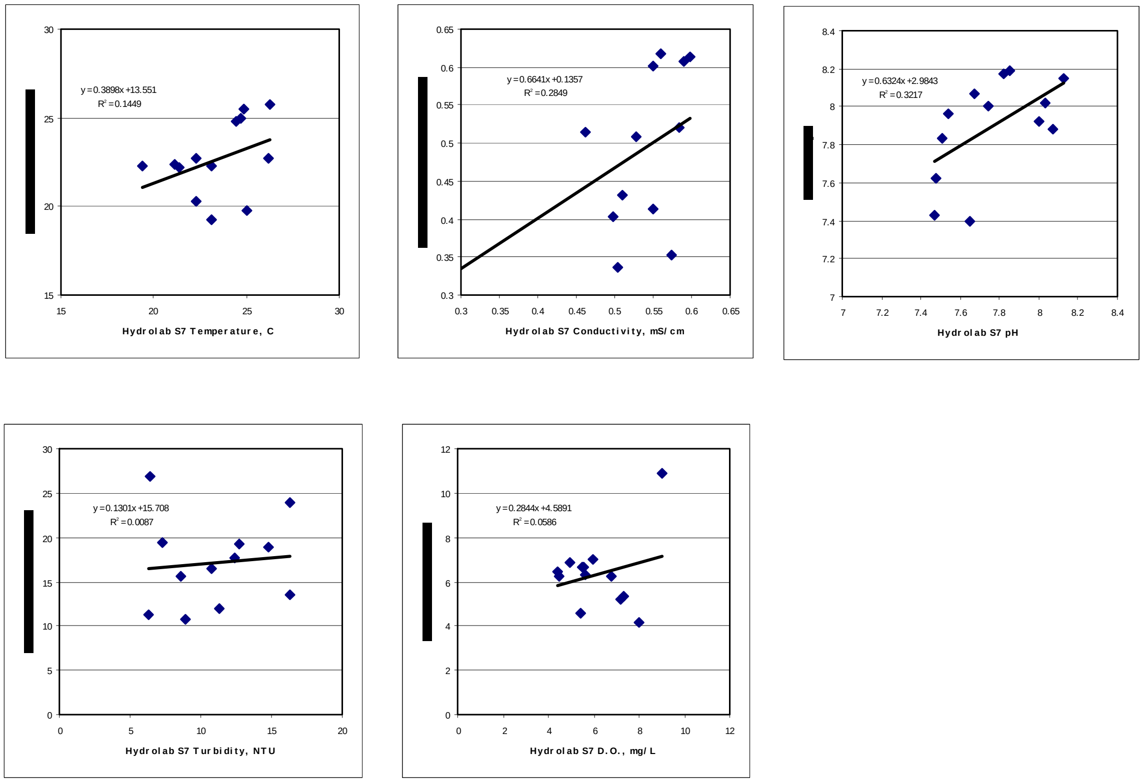


Figure 5.23 Hydrolab continuously logged data (Hydrolab Site 7) vs. water column profile results (1 m depth) at Habitat Site 1.

5.4.7 *E. coli* Levels

The results of the *E. coli* testing at Hydrolab Sites 2, 4, and 7 are shown in Table 5.7. The high levels associated with the storm event of 9/9/04 are consistent with the fecal coliform results reported by past studies (e.g. Irvine and Pettibone, 1996; Wills and Irvine, 1996; Irvine et al., 2005b). The *E. coli* levels on two dry weather dates were relatively low (50-500 m.o./100 mL), but the levels at all sites for the 9/22/04 sample date were elevated. A review the USGS gauge data for Cazenovia, Cayuga, and Buffalo Creeks indicated that flows on 9/22/04 were relatively low (34-88 cfs)(0.96-2.5 m^3s^{-1}) and it had been at least four days since a storm event had occurred.

Table 5.7 *E. coli* Levels per 100 mL, 2004

Date	Time	Site 2	Date	Time	Site 7	Date	Time	Site 4	Event/Dry
9/9/2004	11:10	36,100	9/9/2004	11:20	25,700	9/9/2004	10:55	32,800	E
9/9/2004	12:17	32,100	9/9/2004	12:30	25,200	9/9/2004	11:55	38,700	E
9/9/2004	14:15	24,700	9/9/2004	14:05	27,400	9/9/2004	13:50	30,200	E
9/10/2004	10:05	18,300	9/10/2004	10:10	12,400	9/10/2004	9:45	15,700	E
9/15/2004	11:55	500	9/15/2004	12:00	600	9/15/2004	11:00	50	D
9/22/2004	10:20	2,160	9/22/2004	10:25	1,820	9/22/2004	9:55	1,580	D
9/29/2004	11:00	100	9/29/2004	11:05	200	9/29/2004	11:50	50	D

Winkler (2005) sampled for *E. coli* on a daily basis at Hydrolab Sites 2, 4, and 7 between 3/24 and 4/10/04. Analysis was done using the Coliscan Easygel test, per the procedures outlined in Section 5.3. Water temperature ranged between 1.5-8.8 ^{o}C, 2.8-11.5 ^{o}C, and 1.8-8.7 ^{o}C at Sites 2, 7, and 4, respectively. River discharge conditions (to the top of the AOC) ranged between baseflow and approximately 6,500 cfs (184 m^3s^{-1}) (representing a combined rain and snowmelt event). The geometric mean *E. coli* levels were 31 m.o./100 mL, 9 m.o./100 mL and 112 m.o./100 mL at Sites 2, 7, and 4 respectively. Temperature and seasonal conditions may be limiting factors governing *E. coli* levels in the river. Although the higher end flow conditions would have been sufficient to move the bacteria to the river via overland runoff and also erode bed material (re-introducing bacteria to the water column), it seems that the bacteria had not yet become established (e.g. through re-inoculation of bed sediment) or multiplied under the protection of the sediment. Irvine and Pettibone (1996) found a similar seasonal trend for fecal coliform in the watershed. It seems likely that there is a critical time (and associated temperature range) at which the bacteria levels may be expected to increase.

5.5 Conclusion

Dissolved oxygen levels frequently were below state guidelines within the dredged portion of the AOC (representing all habitat sites except Site 1). Dissolved oxygen levels upstream of the dredged channel more frequently were above state guidelines, particularly in 2004. It appears that the low dissolved oxygen levels are related to a combination of stratification in the river at low flows that can reduce aeration, high sediment oxygen demand, together with long residence times due to system hydraulics (in particular, dredging increases channel cross-sectional area and residence time), and background biochemical oxygen demand (see Figure 5.7). Vertical variation in dissolved oxygen was observed at most sites, with lower levels being recorded near the bed of the river. This vertical trend was less apparent at Site 6, possibly due to the impact of aquatic vegetation.

During dry periods turbidity in the upper 1 m at all sites typically was <20 NTU (5-15 mg L^{-1}). Turbidity increased near the bed (ranging between approximately 20-100 NTU), generally as the combined result of sedimentation from above and near bed saltation. Turbidity during storm events was much greater, reaching 1,000 NTU (~300-600 mg L^{-1}). Unless the event was of extreme magnitude, such as the storm of 9/9/04, there was a trend of decreasing turbidity in the downstream direction as the result of sediment deposition.

Levels of pH on average ranged between 7 and 8. These pH levels are related to the buffering capacity of the limestone and dolomite bedrock of the area and are not unusual for watersheds in this region. Conductivity varied in the range of about 0.3-0.7 mS/cm, but frequently showed a characteristic decrease during runoff events as the groundwater solute concentrations were diluted by stormwater.

The levels of *E. coli* were high during storm events (up to 38,700 m.o./100 mL) at upstream sites, as well as at the Ohio Street Bridge. This is consistent with past studies (e.g. Irvine and Pettibone, 1996; Irvine et al., 2005b) and again documents the importance of the upper watershed as a bacteria source. The *E. coli* levels were lower during dry periods, in the range of 50-2,200 m.o./100 mL. The upper end of this range still is high, but again, these results are consistent with past studies.

The Coliscan Easygel system was easy to use and provided results that were consistent with past studies of the river. Other citizen's groups throughout the U.S. successfully have used the Coliscan Easygel system in monitoring programs (e.g. Alabama Water Watch (1999); Virginia Citizen Water Quality Monitoring Program (2003); Texas Watch (http://www.texaswatch.geo.txstate.edu/Newsletters/98-04.pdf); Hoosier Riverwatch (http://www.in.gov/dnr//riverwatch/pdf/manual/Chap4.pdf); Alliance for the Cheasapeake Bay Citizens Monitor (http://www.acb-online.org/pubs/projects/deliverables-87-1-2003.pdf); University of Vermont (2003)). The Virginia Department of Environmental Quality approved the Coliscan Easygel method for screening purposes and independent testing (e.g. Alabama Water Watch, 1999; Deutsch and Busby, 2000) has shown that Coliscan Easygel results are comparable to standard methods.

5.6 References

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APPENDIX 5.1
HYDROLAB SITES DURING DRY WEATHER AND EVENT OF 9/9/04


Hydrolab Site 2, Seneca St. Bridge, dry weather conditions in summer


Hydrolab Site 7, Mouth of Cazenovia Creek, dry weather conditions in summer

Hydrolab Site 4, Ohio St. Bridge, dry weather conditions in summer


Hydrolab Site 2, Seneca St. Bridge, event of 9/9/04


Hydrolab Site 7, Mouth of Cazenovia Creek, event of 9/9/04


Hydrolab Site 4, Ohio St. Bridge, event of 9/9/04

6. USE SURVEYS

K.N. Irvine

6.1 Introduction

A variety of different recreational uses of Buffalo River water and riparian zones have been observed informally over the past 20 years, including swimming, canoeing, kayaking, power boating, fishing, walking along trails, and sitting at observation areas. However, the level of activity has never been quantified. This study provided a preliminary evaluation of activity level related to the 10 habitat sites, as well as other locations along the river, within the defined study area. This survey was not meant to be as detailed as those outlined, for example, by the Statewide Comprehensive Outdoor Recreation Plan (SCORP) prepared by the New York State Office of Parks, Recreation and Historic Preservation (OPRHP) (2002). The SCORP assesses both the supply and demand of recreation resources that includes consideration of geographic area, a variety of demographic information, use surveys, and impact on the environment. However, the survey reported here does provide insight as to the type of recreational uses that are prevalent, the level of activity, and the locations of highest activity along the river.

6.2 Methodology for Recreational Use Survey

It was necessary to conduct the recreational use survey by boat because many of the sites are more readily (and rapidly) observed from the water. The survey team consisted of two people, one to operate the 14 foot Boston Whaler and the second to complete the survey sheets. The surveys were done during randomly selected time slots (7-9 am; 9am-12 pm; 12 pm-3 pm; 3-6 pm) on randomly selected days of the week. All days and all time slots were sampled during the two year study. The survey was completed on 34 dates between June 18 and September 7, 2003 and on 39 dates between June 7 and September 3, 2004 (a total of 73 dates).

On each sample date, the survey was completed for 15 pre-determined sites. These sites were selected based on the author's 17 years of experience on the river. The 15 pre-determined sites were:

- Kotter Fireboat
- Great Lakes Fishing Club
- Ohio Basin Habitat Remediation Site and Canoe Launch
- Bison City Rod and Gun Club
- Foot of Hamburg St.
- Cargill's Grain Elevator
- Concrete Central Grain Elevator
- First CSX Railway Bridge
- Smith St. Habitat Remediation Site
- Smith St. CSO
- Second CSX Railway Bridge

- Third CSX Railway Bridge
- Boone St. CSO
- Old Bailey Woods
- Seneca Bluffs

A photo of each of these sites is provided in Appendix 6.1. The survey also was completed for each of the 10 habitat sites. The Old Bailey Woods site, listed above, is part of Habitat Project Site 1, but for the use survey it was considered separately because of its physical disconnect from Cazenovia Point. In addition to the 25 fixed sites, any activity that occurred at other locations within the study area was noted and the location was referenced with GPS.

6.3 Results and Discussion for Recreational Use Survey

A total of 887 person-days of activity were observed on the 73 sample days, 2003-04. Following the work done by Johns et al. (2003), this study defines a person-day as one person participating in an activity for a portion or all of a day. A summary of the activities observed in 2003-04 is provided in Figure 6.1. Clearly, fishing, boating, and "hanging out" were the predominant activities. In this case, boating includes, power boating, canoeing, kayaking, sailing, rafting, and rowing. "Hanging out" was a category used to classify general riparian activity that might include eating lunch, reading, talking with friends, walking trails, sunning, or relaxing (but *not* fishing).

The frequency of swimming, as shown in Figure 6.1, is lower than had been anticipated and there may be several explanations for this observation. Mean temperature data from the Buffalo Airport for 2003 and 2004 are shown in Figure 6.2, together with the 30-year norms (1961-1990). In both years, June and July were cooler than average, as was August, 2004. August, 2003 was warmer than average. Furthermore, there were no days greater than 90 ^{o}F (32.2 ^{o}C) in any of the surveyed months, 2003 or 2004. Historically, the mean number of days greater than 90 ^{o}F (32.2 ^{o}C) at the Buffalo Airport is 1 for June, 2 for July, 1 for August, and 0.5 for September. Monthly mean rainfall data from the Buffalo Airport for 2003 and 2004 are shown in Figure 6.3, together with the 30-year norms (1961-1990). June and August of both years were drier than average, while July and September of both years were wetter than average. In particular, July, 2004 had nearly twice the average monthly precipitation.

It might be argued that the study years were slightly cooler (both the mean temperature and days greater than 90 ^{o}F (32.2 ^{o}C)) and wetter at critical times (e.g. July) than average, which could negatively impact the frequency of swimming. Alternatively, because this is the first quantitative measure of swimming frequency, previous qualitative perceptions could be inaccurate and swimming frequency in fact may not have declined. If previous qualitative perceptions are correct and there has been a decrease in swimming activity, other possible explanations may include a shift in demographics (e.g. fewer children in the neighborhoods surrounding the Buffalo River), alternative available activities, or improved communications regarding the risk of swimming in the Buffalo

River (e.g. Buffalo Niagara Riverkeeper community outreach; posting of CSO locations). It is beyond the scope of this study to assess these alternative interpretations of the data.

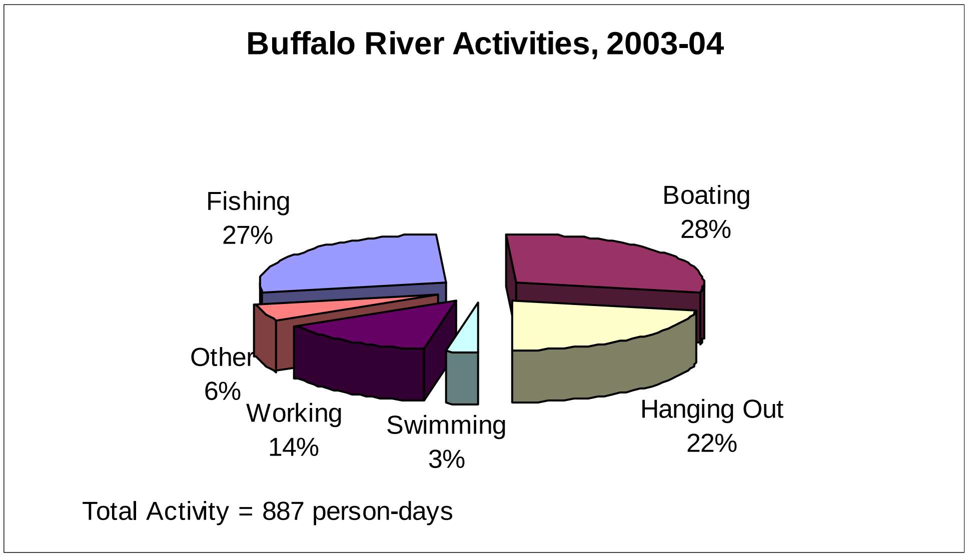


Figure 6.1 Summary of Buffalo River activities, 2003-04

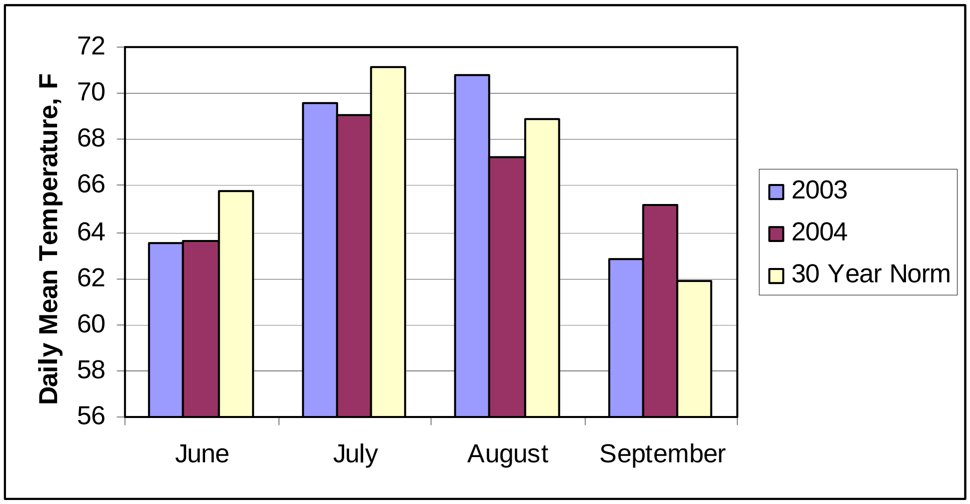


Figure 6.2 Daily mean temperature data from the Buffalo Airport

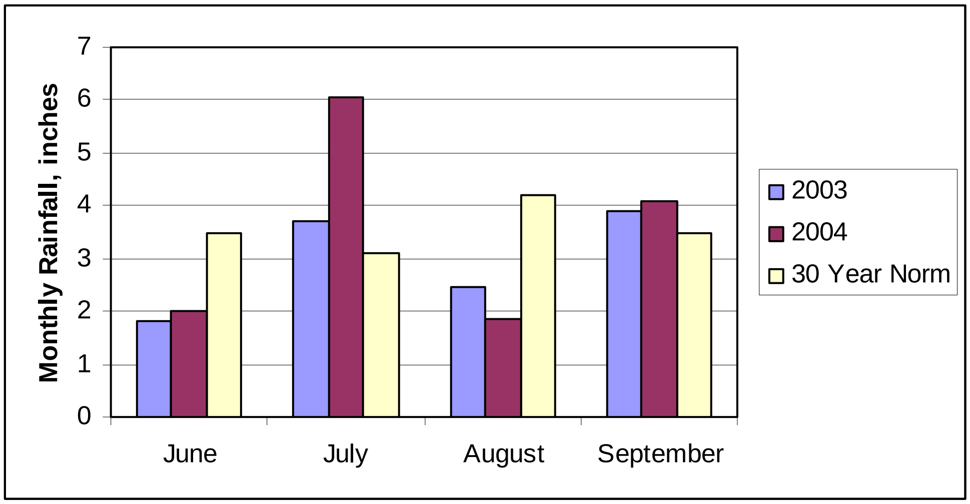


Figure 6.3 Monthly rainfall data from the Buffalo Airport

In 2003 there was a significant correlation (α=0.05) between time of sample and air temperature (r=0.53); time of sample and number of people observed in activity (r=0.60); and air temperature and number of people observed in activity (r=0.41). In 2004, correlations were weaker. The correlation between time of sample and air temperature was not significant (r=0.20) and neither was the correlation between number of people observed in activity and air temperature (r=0.002). The correlation between time of sample and number of people observed in activity was significantly correlated (r=0.32). Sunday tended to be the day of heaviest use in both 2003 and 2004; Saturday had the highest mean use of any day in 2003, but Saturday use was considerably lower in 2004. There appears to be some significant temporal trends in the level of activity, as use tends to increase through the day and, not surprisingly, is highest on weekends. The activity of fishing did not appear to attract people early in the morning.

The observed activity level of 887 person-days underestimates actual activity because it only represents a three hour segment on each of the 73 sample dates. Brother and Moore (1994) noted that samples of recreational activity should be adjusted to account for the entire period of activity. Adjustments to the estimates should consider the peak use periods, duration of the use, facility availability, resident or non-resident, and the turnover rate of the activity (Brother and Moore, 1994). Our survey data indicated that there were temporal trends in the level of activity. Given the types of activity recorded for the survey, we assumed that the turnover rate would be within the three hour time period of each survey. In our case, activity level also might have been adjusted for air temperature, but because the correlation was not significant in 2003, it was decided not to make this adjustment. A "representative" activity level for each survey time slot and each day was calculated from the observed data for each year. The representative level typically was calculated as the mean person-days from the observed data. These representative levels were used to adjust the estimate of 887 person-days for the period June through September 15 of each year. For time slots that had observed data, these were used in place of the representative level. Following this procedure, the adjusted activity level for 2003 was 6,862 person-days and for 2004 was 5,922 person-days. By way of comparison, Erie County parks totaled 120,000 visits in 2000 (i.e. Buffalo River activity was 5-6% of the Erie County park activity).

Both “formal” and “informal” spaces were used in the different activities represented in Figure 6.1. Generally, however, the “formal” spaces had the highest level of activity. Bison City Rod and Gun Club had the highest activity over the two year period, followed in order by Habitat Project Site 1 (fishing and hanging out at the “point” was popular), Ohio Basin Canoe Launch and Park, and Smith St. Habitat Remediation and Park. Habitat Project Sites 3, 4, 5, 6, 8, and 10 had the lowest level of activity of all survey sites, with each of these sites being 8 or less person-days (unadjusted numbers) over the entire two year period.

OPRHP (2002) calculated a relative index of needs on a county basis for different recreational activities observed within New York state. The index indicates the degree to which additional facilities are needed to meet future demand. A value of five indicates that for a given activity, the projected supply/demand ratio in the year 2020 will be at the statewide average. The scale ranges from 1 to 10. A value of one indicates a large availability of recreation resources relative to demand, with little or no crowding. A value of 10 indicates the opposite; most sites are heavily used. The relative index of needs for Erie County are: Swimming – 7; Walking – 7; Boating – 6; Fishing – 7. For these recreational activities OPRHP (2002) has indicated that Erie County will have pressure to meet the public demand. These activities already are observed for the Buffalo River. With improved habitat areas, the Buffalo River could have an increased capacity to meet this demand.

6.4 Land Ownership – Riparian Zone

Land ownership of the habitat sites are shown in Appendix 6.2. Clearly, some sites are entirely privately owned while other sites have mixed public and private ownership. It is unknown at this point whether any of the owners would be willing to consider a riparian restoration project on their property.

6.5 References

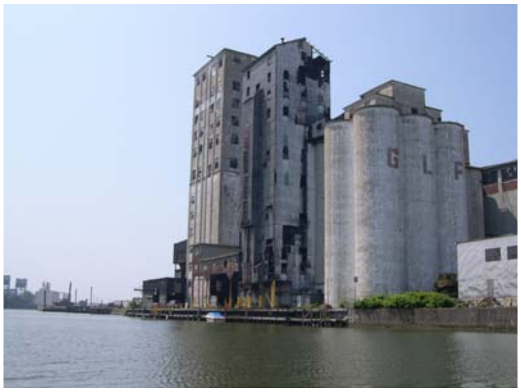
Brother, G. and Moore, R. 1994. Economic Impact of Assessment of Recreation Based Tourism. NCRPS Conference, Asheville, NC.

Johns, G.M., Leeworthy, V.R., Bell, F.W., and Bonn, M.A. 2003. *Socioeconomic Study of Reefs in Southeast Florida*, Report to NOAA.

Office of Parks, Recreation and Historic Preservation (OPRHP), 2002. *Final Statewide Comprehensive Outdoor Recreation Plan and Final Generic Environmental Impact Statement for New York State, 2003.* Albany, NY.

APPENDIX 6.1
FIXED RECREATIONAL USE SURVEY SITES

Kotter Fireboat



Great Lakes Fishing Club

Ohio Basin Habitat Remediation Site and Canoe Launch

Bison City Rod and Gun Club

Foot of Hamburg Street

Cargill's Grain Elevator

Concrete Central Grain Elevator

First CSX Railway Bridge

Smith St. Habitat Remediation Site

Smith St. CSO

Second CSX Railway Bridge

Third CSX Railway Bridge

Boone St. CSO

Old Bailey Woods

Seneca Bluffs

Seneca Bluffs (Continued)

APPENDIX 6.2
LAND OWNERSHIP IN THE BUFFALO RIVER RIPARIAN ZONE

CHAPTER 7
SITE EVALUATION MATRIX

K.N. Irvine, R.J. Snyder, and T.P. Diggins

7.1 Introduction

One of the objectives of this study was to develop a characterization matrix for each of the 10 potential habitat restoration sites. This matrix is intended to serve as a guidance tool for stakeholders and decision makers which would allow them to quickly review comprehensive assessments of the potential for effective habitat restoration. As can be seen from the preceding chapters, a large volume of data was generated in the field effort and it can be a challenge to provide an understandable summary of these data for stakeholders. One approach that has been used effectively in past studies to communicate and summarize large ecological data sets is the application of an index for specific ecological characteristics. In this chapter, we first review the different indices that have been applied in past studies and then develop a combination of indices that form the basis of our site evaluation matrix.

7.2 Water Quality Indices

Various water quality indexes have been developed in the past 40 years and House (1990) summarized the utility of a water quality index (WQI): i) volumes of water quality data are summarized in a single index value in an objective, rapid, and reproducible manner; ii) the numerical scale of an index facilitates evaluation of "within class" variations, thereby allowing identification of changes in water quality at a site that would not precipitate a change within the classification system; iii) the index values may be related to a "potential water use" classification scheme to help determine the ecological potential of the waterbody; iv) the index and associated waterbody classification scheme may be used in operational management to identify surface waters requiring priority action; and v) the index facilitates communication with the layperson, while maintaining the initial precision of measurement.

One of the earliest efforts to develop a WQI was done in association with the National Sanitation Foundation (NSF) (Brown et al., 1970; 1973). Brown et al. (1970) assembled a panel of 142 persons throughout the U.S.A with known expertise in water quality management. Three questionnaires were mailed to each panelist to solicit expert opinion regarding the WQI and the procedure incorporated many aspects of the Delphi method, an opinion research technique first developed by Rand Corporation. In the first questionnaire, the panelists were asked to consider 35 analytes for possible inclusion in a WQI and to add any other analytes they felt should be included. The panelists also were asked to rate the analytes that they would include on a scale from 1, (highest significance), to 5, (lowest significance).

The results from the first survey were included with the second questionnaire and the panelists were asked to review their original response. The purpose of the second questionnaire was to obtain a closer consensus on the significance of each analyte. Also included was a list of nine new analytes that had been added by some respondents in the first questionnaire. For the second questionnaire, the panelists were asked to list no more than 15 most important analytes for inclusion from the new total of 44.

From these first two responses, Brown et al. (1970) derived nine analytes for inclusion in the WQI. In the third questionnaire, the panelists were asked to draw a rating curve for each of the nine analytes on blank graphs provided. Levels of water quality (WQ) from 0 to 100 were indicated on the y-axis of each graph while increasing levels of the particular analyte were indicated on the x-axis. Each panelist drew a curve which they felt best represented the variation in WQ produced by the various levels of each parameter. Brown et al. (1970) then averaged all the curves to produce a single line for each analyte. Mitchell and Stapp, (1995) provide the best visual representation of each rating curve. Statistical analysis of the ratings enabled Brown et al. (1970) to assign weights to each analyte, where the sum of the weights is equal to 1. The nine parameters and their corresponding weights are listed in Table 7.1. The WQ value for each analyte then was calculated as the product of the rating curve value (also known as the Q-value) and the WQI weight.

Brown et al. (1973), as presented by Ott (1978), further assessed the validity of the WQI. A new panel of experts was assembled and polled using the same procedure as used in 1970. No significant differences were found between the quality rating curves from the original investigation and the new set of curves. According to Ott (1978), the NSF felt that the index developed by Brown et al. (1970; 1973) would help alleviate the limitations of previous efforts to develop a WQI and the index subsequently was ratified by the NSF in 1974. This index also was adopted for use by the NYSDEC in 1977 (Ott, 1978).

Table 7.1 NSF WQI Analytes and Weights

Analyte	WQI Weights
Dissolved oxygen	0.17
Fecal coliform (or *E. coli*)	0.15
pH	0.12
BOD_5	0.1
Nitrates	0.1
Phosphates	0.1
Δt ^{o}C from equilibrium	0.1
Turbidity	0.08
Total solids	0.08

Numerous water quality indices have been developed and applied throughout the world, although these often were variations of the NSF WQI (e.g. Yu and Fogel, 1978; Dunnette, 1979; Bhargava, 1983; House and Ellis, 1987; Dinius, 1987; Sharifi, 1990; Smith, 1990; Dojildo et al., 1994; Palupi et al., 1995; Wills and Irvine, 1996). Despite the apparent usefulness, application of non-specific WQI's such as the NSF WQI appeared to languish in the developed world during the 1980's and 1990's. According to Smith (1989), the main reason for the limited application of the non-specific WQI's is that during the data handling process, information can be "lost". For example, if eight of the analytes under the NSF WQI indicate pristine scores, but pH scores 0, a water body might have an index value of 85. This rates as a "good" score, but clearly, a water body with extreme high or low pH would not be capable of supporting certain aquatic life and may be unsuitable for recreation, drinking, or irrigation. Stoner (1978) suggested that specific water use indices may be more informative. House and Ellis (1987), for example, summarized three indices: one is general and similar to the NSF WQI; the second is an Aquatic Toxicity Index (ATI) that considers phenols and the dissolved or total concentration of various metals (including Cu, Zn, Cd, Pb, Cr, As, Hg, and cyanide); and the third is a Potable Sapidity Index (PSI) that includes some of the metals from the ATI as well as total PAHs and total pesticides.

More recently, the non-specific WQI's seem to have gained favor in applications for developing nations (e.g. Pesce and Wunderlin, 2000; Bordalo et al., 2001; Vermette et al., 2004). Furthermore, the state of Oregon has worked on updates of its original WQI, based on improved understanding about water quality behavior (Cude, 2001), while the Canadian Council of Ministers of the Environment (CCME) formalized a new approach to calculating a WQI (CCME, 2001a, b; Khan, 2004). The CCME approach was established because it was recognized that there were a number of agencies and institutions in Canada using some type of metric to assess water quality. The Water Quality Index Technical Subcommittee was formed by the Water Quality Guidelines Committee of the CCME in 1997 to assess different approaches to index formulation and to develop an index that could be used to simplify water quality reporting in Canada.

Ultimately, the CCME (2001a) decided on a three-factor approach that was similar to the index approach used in British Columbia. The three factors are scaled to range between 0 and 100 and Figure 7.1 shows the conceptual model for the index. The values of the three measures of variance from selected objectives for water quality are combined to create a vector in an imaginary "objective exceedance" space. The length of the vector is then scaled to range between 0 and 100 and subtracted from 100 to produce an index in which a number closer to 0 represents poorer water quality. The CCME approach emphasizes the use of water quality guidelines within the index in contrast to the Delphi and rating curve approach used in the NSF WQI.

The three factors defined in the CCME are:

Factor 1 (F_1) – Scope – the extent of water quality guideline non-compliance over the time period of interest:

$$F_1 = \left(\frac{\text{Number of failed variables}}{\text{Total number of variables}} \right) \times 100 \qquad [7.1]$$

Where *variables* indicates those water quality variables with objectives that were tested during the time period for the index calculation.

Factor 2 (F_2) – Frequency – represents the percentage of individual tests that do not meet objectives (i.e. "failed tests"):

$$F_2 = \left(\frac{\text{Number of failed tests}}{\text{Total number of tests}} \right) \times 100 \qquad [7.2]$$

Factor 3 (F_3) – Amplitude – represents the amount by which failed test values do not meet their objectives. F_3 is calculated in three steps:

Step 1: The number of times by which an individual concentration is greater than (or less than when the objective is a minimum) the objective is termed an "excursion" and is expressed as follows. When the test value must not exceed the objective:

$$excursion_i = \left(\frac{\text{Failed test value}_i}{\text{Objective}_j} \right) - 1 \qquad [7.3]$$

For the cases in which the test value must not fall below the objective:

$$excursion_i = \left(\frac{\text{Objective}_j}{\text{Failed test value}_i} \right) - 1 \qquad [7.4]$$

Step 2: The collective amount by which individual tests are out of compliance is calculated by summing the excursions of individual tests from their objective and dividing by the total number of tests (both those meeting objectives and those not meeting objectives). This variable, referred to as the normalized sum of excursions, or *nse*, is calculated as:

$$nse = \frac{\sum_{i=1}^{n} \text{excursion}_i}{\text{Number of tests}} \qquad [7.5]$$

Step 3: F_3 is then calculated by an asymptotic function that scales the normalized sum of the excursions from objectives (*nse*) to yield a range between 0 and 100:

$$F_3 = \left(\frac{nse}{0.01nse + 0.01} \right) \qquad [7.6]$$

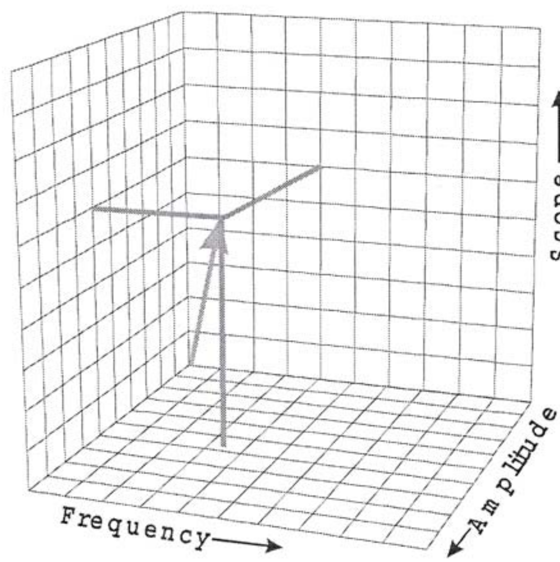


Figure 7.1 Conceptual model of the CCME WQI (from CCME, 2001a)

7.3 Benthic Macroinvertebrate Indices

Chapter 3 included analysis of benthic data using various indices. This section provides an additional review of index theory, specifically in relation to benthic rapid bioassessment and the application of multiple indices. The intent of benthic rapid bioassessment is to evaluate overall biological condition, optimizing the use of the benthic community's capability to reflect integrated environmental effects over time (Plafkin et al., 1989). The advantages to using benthic macroinvertebrate communities include: macroinvertebrate assemblages are good indicators of localized conditions because of their limited potential for movement; they integrate the effects of short-term environmental variations; degraded conditions often can be detected with only a cursory examination by an experienced biologist; macroinvertebrate assemblages constitute a broad range of trophic levels and pollution tolerances; sampling is relatively easy and inexpensive; macroinvertebrates serve as a primary food source for fish and are abundant in most streams; many state water quality agencies that routinely collect biosurvey data focus on macroinvertebrates (Barbour et al., 1999).

There are many different methods available to evaluate benthic macroinvertebrate communities that have been well documented in the literature (e.g. Hilsenhoff, 1977; Hilsenhoff, 1988; Novak and Bode, 1992; Bode et al., 1996; Diaz et al., 2004). More recently, there has been a movement towards integrating several different evaluations of the benthic community in an effort to provide more accurate assessments (e.g. Greer et al., 2002). The combination of indices based upon pre-determined thresholds is a multimetric approach (Norris, 1995). The multimetric approach involves defining several indices that provide information on diverse biological attributes; when integrated they give an overall indication of the condition of the biological community (Norris, 1995). In New York State, the NYSDEC employs a multimetric approach, using the indices described in the following sections (Bode et al., 1996)

7.3.1 Species Richness

Two primary approaches have evolved to assess species richness. The first is areal richness or density, which is expressed as the number of species in a unit area, while the second is a numerical richness as determined from fixed-count subsampling (Larsen and Herlihy, 1998). Numerical richness is expressed as the number of species in a unit count (e.g. number of species per 100 individuals enumerated). There are some fundamental differences in what is being measured with these two different methods. Areal richness expresses the number of species in a unit area regardless of their abundance; numerically abundant and rare taxa count equally (Larsen and Herlihy, 1998). In contrast, numerical richness expresses the number of species in a fixed-count; it is sensitive to the relative abundance of individuals. Larsen and Herlihy (1998) demonstrated that the two versions of species richness are correlated; the NYSDEC employs the fixed-count method.

7.3.2 EPT Richness

EPT denotes the total number of Ephemeroptera (mayfly), Plecoptera (stonefly), and Trichoptera (caddisfly) species found in a 100 organism subsample. Barbour et al. (1992) found a high correlation can exist between EPT richness and taxa richness, suggesting there may be a certain degree of redundancy when both metrics are used in the assessment of benthic communities. In regions where the number of mayfly, stonefly, and caddisfly taxa reflect a high diversity, it would be expected that the overall taxa richness also would be high. Therefore, the degree of redundancy between the two metrics is dependent on the community representation by the EPT taxa and can vary depending on the ecoregion (Barbour et al., 1992). Barbour et al. (1992) attribute the redundancy between the two metrics to the fact that the EPT taxa constituted a major portion of the total taxonomic composition. However, it is possible that the two metrics may not be redundant in all situations. As noted in Chapter 3, it was not expected that the EPT counts would be high within the Buffalo River AOC and the metric therefore was not employed in this study.

7.3.3 Hilsenhoff Biotic Index

The Hilsenhoff Biotic Index (HBI) is a measure of organic and nutrient pollution using benthic macroinvertebrate communities (Hilsenhoff, 1987). Organic and nutrient pollution cause lowered levels of dissolved oxygen, particularly after storm events, which in turn affects the ability of each species of arthropod to survive in a particular stream (Hilsenhoff, 1987). For the purpose of calculating the HBI, every species is assigned a tolerance value of 0-10, with 0 assigned to species most intolerant of organic pollution and 10 assigned to the most tolerant species (Hilsenhoff, 1987). In essence, the HBI represents the average tolerance for all individuals collected from a site. Hilsenhoff (1988) also developed a Family Level Biotic Index (FBI), which is an average of tolerance values of all arthropod families in a sample. The intent of the FBI was to eliminate the need for laboratory identification of specimens and reduce the time needed to process data, but ultimately Hilsenhoff (1988) concluded the loss of sensitivity in the FBI was not acceptable. Comparisons of HBI and FBI values indicated that the FBI overestimated impairment in moderately impaired waters and underestimated impairment in more severely impaired waters (Hilsenhoff, 1988). The HBI has been used widely in bioassessment programs (e.g. Hilsenhoff, 1987; Plafkin et al., 1989; Bode et al., 1996; Barbour et al., 1999) and is employed by the NYSDEC.

7.3.4 *Percent Model Affinity*

The Percent Model Affinity (PMA) index is intended to provide water quality information not entirely contained within the indices discussed in Sections 7.3.1 through 7.3.3 (Novak and Bode, 1992). It is based on the concept that the biological effects of pollutants can be measured by comparing the existing community with an expected community, a practice that many biologists carry out intuitively (Novak and Bode, 1992). The PMA accomplishes this quantitatively by establishing a model community comparison for a respective habitat type; affinity to that model is measured with a percentage similarity index (Novak and Bode, 1992). Novak and Bode (1992) concluded that the PMA was more accurate in detecting water quality changes than the HBI, particularly for streams that are impacted by non-organic pollution. Barton (1996) also found the PMA was able to effectively distinguish between minimally impacted headwater sites in Southern Ontario and downstream sites that were more heavily impacted by agriculture.

The use of the PMA index is one way to deal with complex or multiple impacts. The PMA is appropriate for these situations because it measures divergence from a reference condition, regardless of the stress or direction of the change. One of the challenges, however, in employing a PMA approach is the identification of an appropriate “expected” or “nonimpacted” community.

7.4 Fisheries Indices

A variety of indices have been developed to assess the health of a waterbody for fish, including the indicator species approach, species richness and diversity; the Index of

Well Being; and the Index of Biotic Integrity (IBI) (Simon, 1999). Of these, the most commonly used integrative approach is the IBI.

7.4.1 Index of Biotic Integrity

The IBI was first developed for use in small warmwater streams in central Illinois and Indiana (Karr, 1981). The original version had 12 metrics that reflected fish species richness and composition, number and abundance of indicator species, trophic organization and function, reproductive behavior, fish abundance, and condition of individual fish. Each metric received a score of 5, 3, or 1, based on its similarity to a fish community with little human influence. A score of 5 represents a minimally impacted community, 3 represents intermediate impacts, and 1 represents severe degradation (Karr, 1981). The total IBI score is the sum of the 12 metric scores and ranges between 60 (good) and 12 (poor).

The original version of the IBI quickly became popular and has been used by many investigators to assess warmwater streams throughout the central U.S. (Simon, 1999). Since the IBI's inception, many have explored the sampling protocols and effectiveness of the original version in different regions and different types of waterbodies (e.g. Miller, 1988; Faush et al., 1990; Halliwell et al., 1999). As the IBI became more widely used, different versions were developed for different regions and ecosystems (e.g. Miller, 1988; Halliwell et al., 1999; Thoma, 1999). The new versions have a multimetric structure, but differ from the original version in the number, identity, and scoring metrics (Miller, 1988). In particular, new versions developed for streams and rivers in eastern and western U.S. and Canada tend to have a different set of metrics, reflecting the substantial differences in fish faunas between these regions and the central U.S. (e.g. Miller, 1988; Faush et al., 1990; Halliwell et al., 1999; Thoma, 1999; Greer et al., 2002).

7.5 The Components of the Site Evaluation Matrix

To help assess the potential for habitat rehabilitation at each of the 10 study sites, a site evaluation matrix was developed. An important component of the evaluation matrix was the application of biotic and abiotic indices to provide simple, but objective, decision support. In addition, information such as sediment chemistry, land ownership, and frequency of land and water use were considered. The components of the site evaluation matrix are summarized in Table 7.2 and in the subsequent sections, the specific methodologies used to determine each component are discussed.

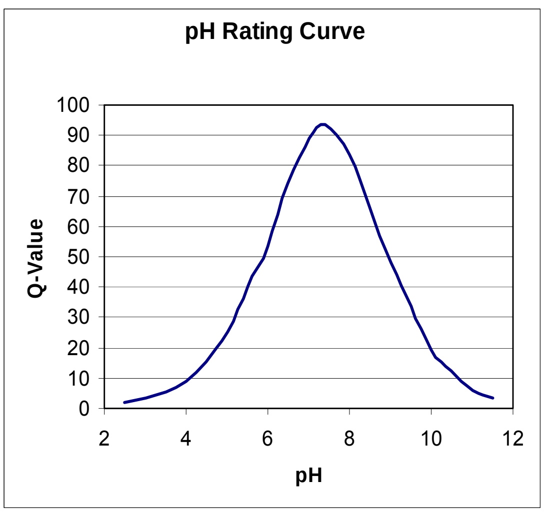
7.5.1 Water Quality Indices

Two different index approaches were used to assess the relative water quality at each site. The first index approach was based on the NSF WQI. Because the Hydrolabs were used as the principal tool to monitor water quality in this project, only the dissolved oxygen, pH, and turbidity components of the NSF WQI were calculated. Temperature was monitored in this project, but the NSF WQI evaluation specifically targets changes in temperature that might be related to point source discharges. Therefore, temperature was not included in the calculations for this study.

Table 7.2 Components of the Site Evaluation Matrix

Benthics	Fish	Vegetation	Abiotic
Number of benthic families (Species diversity)	Species diversity	Shading (% Overhang)	NSF WQI (dissolved oxygen, pH, turbidity)
Oligochaete density	Index of Biotic Integrity	Macrophyte species diversity	CCME WQI (dissolved oxygen)
Product of Chironomid biotic score and number of Chironomid taxa	DELT		

The first step in calculating the NSF WQI-based component was to determine Q-values (Quality values) for dissolved oxygen, pH, and turbidity. The data used for these calculations were the Hydrolab measurements collected at each site at 1 m below the surface, and separately, near the river bed. The Q-value rating curve graphs (e.g. Figure 7.2) presented in Mitchell and Stapp (1995) were converted into equations using a least squares approach in Excel. The raw data for each site were entered into the appropriate Q-value rating curve equation and the result was multiplied by the appropriate weighting factor, as shown in Table 7.1. Finally, the weighted Q-values were summed to provide an index value. This set of calculations was done for each site and for all the weeks of sampling from the two years, combined (a total of 29-30 weeks of data, depending on the depth of sample). The mean index value of the 29-30 weeks of data (at each depth) was used in developing the evaluation matrix.



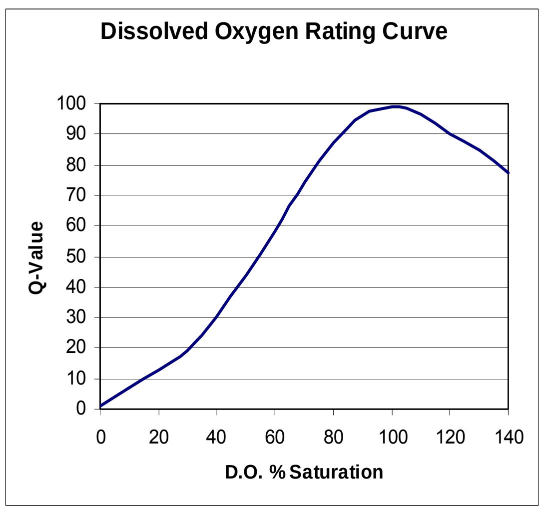


Figure 7.2 Q-value rating curves for NSF WQI

The second water quality index approach used in this study was based on the methodology established by the Canadian Council of Ministers of the Environment (CCME, 2001a, b). It was decided to only include dissolved oxygen in this calculation since New York State does not have well-defined numerical guidelines for temperature or turbidity in Class C rivers and generally, pH is not a concern for the river. The guideline used for the calculations was that at no time should dissolved oxygen be less than 4.0 mg/L. Because only one variable was considered for this index, factor F_1 (Scope) was not calculated, but factors F_2 (Frequency) and F_3 (Amplitude) were calculated (see equations 7.2 through 7.6). The final form of the CCME WQI therefore was:

$$CCMEWQI = 100 - \left(\frac{\sqrt{F_2^2 + F_3^2}}{1.414} \right) \qquad [7.7]$$

The factor 1.414 is a scaling factor that arises because each of the individual index factors can range as high as 100 (CCME, 2001a).

7.5.2 Benthic Organism Indices

It was decided to use three benthic organism indices for the site characterization matrix. The first index was the number of benthic organism families. The second index was the oligochaete density (number/m^2). Because oligochaetes are pollution tolerant, a high density is interpreted here as being an indication of poorer habitat conditions. The third index was the product of the Chironomid Biotic Index scores (i.e., tolerance score averaged among all individuals in a sample) for each site (see Chapter 3) and the number of chironomid taxa, where a higher value is indicative of better habitat conditions.

7.5.3 Fish Indices

Three fish indices were used for the site characterization matrix: species diversity; Index of Biotic Integrity (IBI); and incidence of Deformities, Eroded fins, Lesions, and Tumors (DELT). The IBI is discussed in greater detail in Chapter 2 and in Section 7.4.1.

7.5.4 Vegetation Indices

Two vegetation indices were applied in the site characterization matrix. The first index was an estimate of the percentage overhanging coverage (see Chapter 4). This index represents the amount of shading that might be expected at the site, where a higher value indicates better habitat conditions. The second index was the number of macrophyte species observed at the site (i.e. a measure of species richness).

7.6 Calculation of the Site Characterization Matrix

Because the different indices (water quality, benthic organisms, fish, vegetation) are expressed on different scales, it was decided that the simplest way to compare the results of the indices between sites was to rank the scores of each index for the 10 sites. Therefore, the site with the highest score for a particular index would be given rank 10, while the site with the lowest score would be given rank 1. This ranking process was done for all water quality, benthic, fish, and vegetation indices. As such, when all indices were summed across the sites, those sites with the highest scores represent the most healthy ecological conditions. The rank sum score reflects the aggregate influence of the benthics, fish, vegetation, and water quality indices, in which each index value is given the same individual weight. However, because there are three indices for benthic organisms and three indices for fish, in combination these categories exert a larger influence on the matrix total. The rank scores for all indices at all sites are shown in Table 7.3, as are the aggregate scores for each site.

Table 7.3 Site Characterization Matrix (Rank Scores)

Site	NSF WQI*	CCME WQI	Fish Species #	Fish IBI	Fish DELT	Benthic Family #	Benthic Olig. #	Benthic Chirn. Index	Vegetation Overhang	Macrophyte Species #	Total
1	9	10	1	2	5	0.5	10	5	10	1	53.5
2	10	9	2	3	1	2	3	10	6.5	2	48.5
3	8	8	6	8	3	7	5	4	1	5	55
4	6	7	10	10	6	5.5	6	2	3	5	60.5
5	3	1	4	6	10	0.5	**	7	9	5	45.5
6	4	3	5	1	9	3	2	6	6.5	9.5	49
7	7	2	7	9	7	5.5	4	8	4	9.5	63
8	5	4	9	7	8	9	9	1	6.5	5	63.5
9	2	6	8	5	4	10	7	9	2	5	58
10	1	5	3	4	2	8	8	3	6.5	8	48.5

*Calculated from data at the 1 m depth

**As noted in Chapter 3, sampling for this index was not done at site 5 and the total value for this site therefore is artificially low

7.7 Interpretation of Site Matrix

This section evaluates the results of the characterization matrix (Table 7.3) in terms of between-site comparisons of the total scores and results of categories of indices (fish, benthics, vegetation, water quality). Other factors that could (qualitatively) affect decision-making regarding prioritization of habitat restoration also are explored.

Based strictly on the total matrix scores, sites 4, 7, and 8 have the best aggregate ecological health of the ten sites evaluated. As noted in the previous section, the total value for site 5 is artificially low because benthic oligochaete density was not determined. Results for the individual indices at site 5 are discussed in more detail, below.

The rank scores for the water, benthic, fish, and vegetation indices are summarized in Figures 7.3-7.6. Qualitatively, several spatial trends emerge from Figures 7.3-7.6. Site 1 has a low rank (poor health) for two of the three fish indices, a low to moderate rank for two of the three benthic indices and a low rank for one of the two vegetation indices. Interestingly, both water quality indices rank site 1 as being of the highest water quality. There must be other factors besides water quality that are negatively impacting the biota at site 1. One possible explanation is the heavy use the site experiences (see use surveys in Chapter 6). Two of the three fish indices, one of two benthic indices, and both water quality indices rank site 5 as having amongst the poorest ecological conditions. Site 6, immediately across the river from site 5, also had relatively poor water quality, fish, and benthic organism results. There appears to be a general improvement in ecological conditions (based on the fish, benthic, and water quality ranks) moving downstream from sites 5 and 6 to sites 7, and 8. Site 10 tends to score lower in water quality, all fish indices, and oligochaete density.

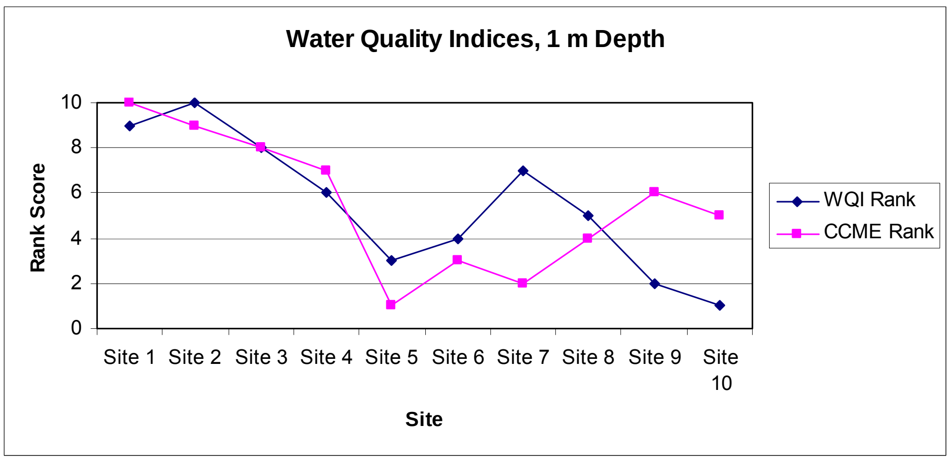


Figure 7.3 Water quality index rank score by site

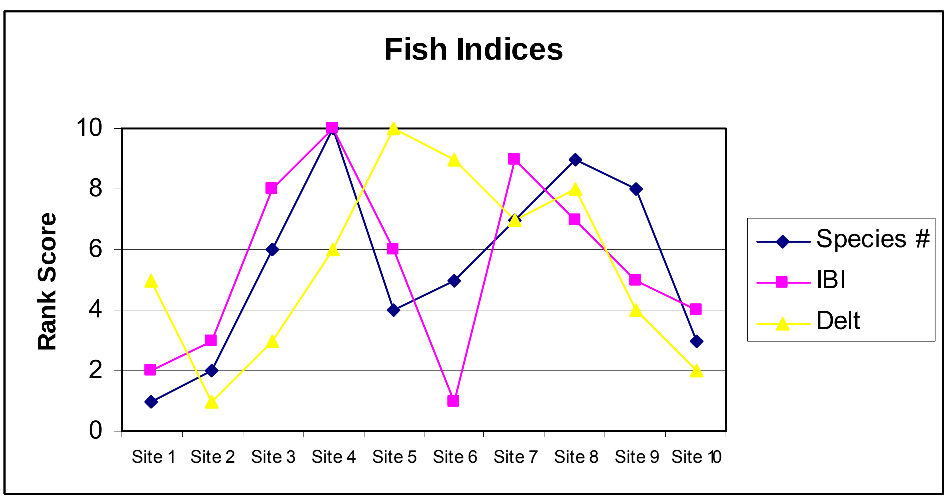


Figure 7.4 Fish indices rank score by site

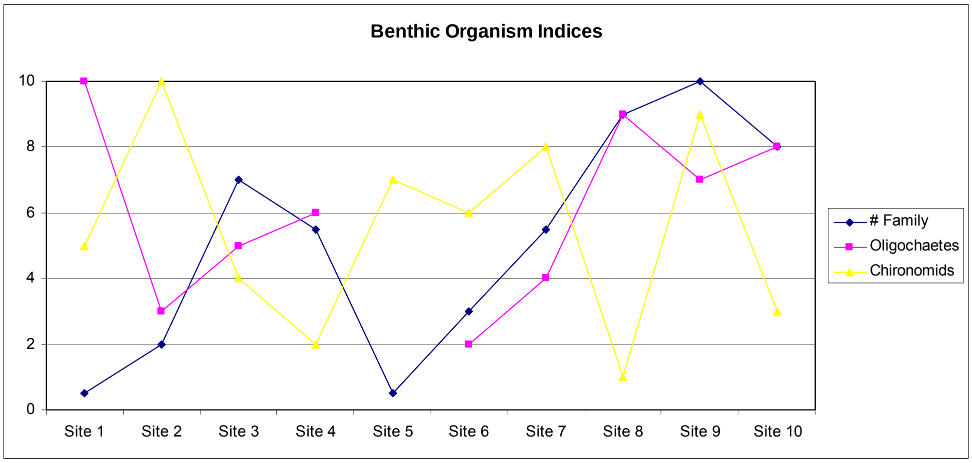


Figure 7.5 Benthic organism rank score by site

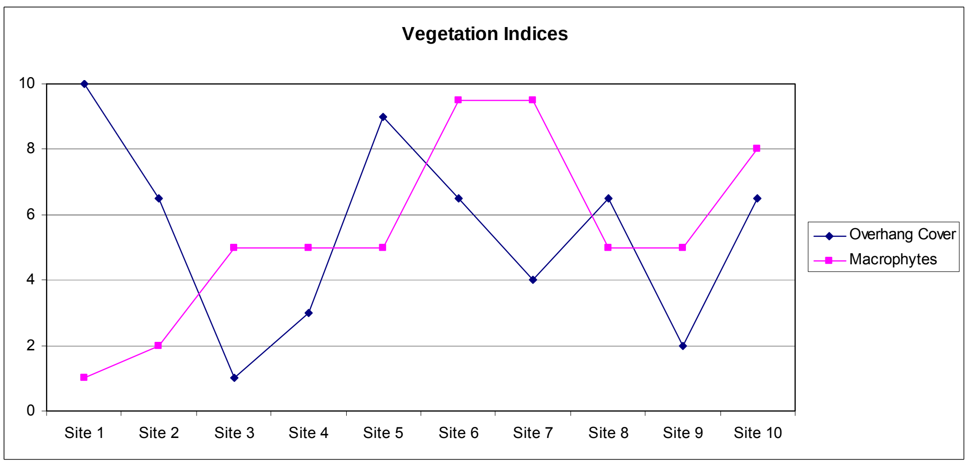


Figure 7.6 Vegetation (% overhang cover and number of macrophyte species) indices rank scores by site

7.7.1 Other Considerations

The U.S. Army Corps of Engineers, Buffalo District, conducted an exploratory study of sediment chemistry and biological uptake in the benthic organism *Lumbriculus variegates* for samples collected in the Buffalo River at 10 sites in 2003 (Karn et al., 2003). Several of the sediment sample sites were common to the habitat sites of this study and the results for PCBs, selected PAHs and metals are shown in Table 7.4.

Table 7.4 Sediment Chemistry for Habitat Sites

Analyte	Habitat Site 3 (Karn et al. Site 3)	Habitat Site 7 (Karn et al., Site 7)	Habitat Site 8 (Karn et al., Site 4)	Habitat Site 9 (Karn et al., Site 8)	Habitat Site 10 (Karn et al., Site 10)
PCB 1248 (µg/kg)	10.2	16.7	214*	136	109
PCB 1260 (µg/kg)	8.03	<8.94	60.7	34.2	41.4
Naphthalene (µg/kg)	42.2	220	221	138	88.3
Fluorene (µg/kg)	57.8	380	337	105	117
Phenanthrene (µg/kg)	365	**2750**	**1060**	606	**758**
Fluoranthene (µg/kg)	875	**6860**	1560	747	1670
Pyrene (µg/kg)	750	**6900**	**1650**	754	**1700**
Chrysene (µg/kg)	518	**4640**	859**	389	**951**
Benzo[a]anthracene (µg/kg)	331	**4320**	**745**	332	**731**
Benzo[b]fluoranthene (µg/kg)	397	3210	516	272	742
Benzo[k]fluoranthene (µg/kg)	308	3140	514	224	555
Benzo[a]pyrene (µg/kg)	321	**3410**	564	272	638
Cd (mg/kg)	0.51	0.63	1.36	0.659	0.829
Cr (mg/kg)	20.8	26.7	43.6	16.8	29.6
Cu (mg/kg)	31.2	37.1	47.8	25.4	52.8
Pb (mg/kg)	30.2	44.4	102	43.9	70.4
Hg (mg/kg)	0.066	0.09	0.37	0.12	0.17
Zn (mg/kg)	99.3	137	193	146	181

Bolded numbers exceed Probable Effect Level on benthic organisms, from Ingersoll et al. (2000)
Ingersoll et al. (2000) did not present a guideline value for benzo[k]fluoranthene
*approaches Probable Effect Level of 277 µg/kg if the Aroclors are summed (214+60.7=247.7 µg/kg)
**approaches Probable Effect Level of 862 µg/kg

Sediment quality guidelines for total PCBs and most of the PAHs shown in Table 7.4 have been developed by the New York State Department of Environmental Conservation (NYSDEC, 1998). The guidelines were developed using an equilibrium partitioning approach that estimates biological impact based on the contaminant's affinity to sorb to organic carbon in the sediment. As such, the guideline level is adjusted for the organic carbon content (g/kg) of the sediment sample (NYSDEC, 1998). Ingersoll et al. (2000) concluded that normalization of sediment quality guidelines for PAHs or PCBs to total organic carbon did not improve prediction of toxicity in field-collected sediment and therefore presented guidelines for dry-weight sediment. The probable effect level and severe effect level guidelines identified by Ingersoll et al. (2000) were used for comparison purposes in Table 7.4. None of the PCB or PAH values reported in Table 7.4 exceeded the severe effect levels, while several of the PAHs at Habitat Sites 7, 8, and 10 exceeded probable effect level. Furthermore, PCB levels at Habitat Site 8 were close to the probable effect level. Karn et al. (2003) also reported sediment chemistry for five other sites throughout the AOC and none of the Habitat sites (Table 7.4) had the highest contaminant levels, with the exception of PCBs for Habitat Site 8.

The organics and metals levels suggest that Habitat Sites 7, 8, and 10 have the potential for improvement through sediment remediation. Clearly, however, before any remediation is done, further sediment testing should be conducted to evaluate the spatial extent of the contamination. The current efforts of the NYSDEC in support of the U.S.

Army Corps of Engineers, Buffalo District, sediment feasibility study, represents one step towards defining the spatial extent of contamination.

The frequency and type of activity at each habitat site was discussed in Chapter 6. Habitat site 1 had the second highest use rate of any location regularly observed along the river. While the site provided good water access and has good water quality, fish, benthic organism, and macrophyte species richness were the lowest for the site. Sites 3, 4, 5, 6, 8, and 10 had the lowest activity level of all survey sites. Land ownership of the habitat sites was summarized in Chapter 6. It is unknown at this point whether any of the owners would be willing to consider a riparian restoration project on their property.

A qualitative summary of habitat considerations for each site is provided in Table 7.5. This summary includes the issues of potential sediment contamination and human activity.

Table 7.5 Summary of Habitat Positives and Deficiencies

Site	Positive Aspects of Site	Site Deficiencies
1	Water quality; shade	Poor fish results; mixed benthic organism results; poor macrophyte community; high human use
2	Water quality	Poor fish results; fairly poor benthic organism results; poor macrophyte community; moderate shading
3	Fish and benthic communities moderately good; water quality moderately good; low human activity	Poor shading
4	Good fish community; moderately good benthic community; moderately good water quality; low human activity	Poor shading; only fair macrophyte community
5	Good shade; moderately good macrophyte community; low human activity	Water quality; generally poor fish and benthic organism results
6	Moderate shading; good macrophyte community; low human activity	Poor fish results, except for DELT; poor benthic community; poor water quality
7	Fair water quality; good fish, benthic, and macrophyte communities; low human activity	Poor shading; poor sediment quality
8	Fair water quality; good fish results; good benthics except chironomids; low human activity	Moderate shading and macrophyte community; poor sediment quality
9	Moderate water quality; moderate to good fish community; good benthic community	Poor shading and macrophyte community
10	Moderate to good benthic community; good macrophyte community	Water quality; fish community; poor sediment quality; location is disturbed by docking of lake-going ships

7.8 Acknowledgement

Sections 7.3 and 7.4 of this chapter drew heavily from Michael Greer's Master's Thesis (Greer, 2001).

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CHAPTER 8
BUFFALO NIAGARA RIVERKEEPER INTERPRETATION

J. Jedlicka and J. Barrett O'Neill

8.1 Introduction

Buffalo Niagara Riverkeeper (formerly Friends of the Buffalo Niagara Rivers) submitted this project application to NYSDEC in order to obtain useful and up to date information on the biological and physical characteristics of the Buffalo River. The resulting data and information will now be used by numerous agencies, organizations and local decision makers to help guide the remedial strategy and priorities of the Buffalo River AOC.

As of 2003, Buffalo Niagara Riverkeeper has served as the coordinator of the Buffalo River Remedial Action Plan (RAP) and the Buffalo River Remedial Advisory Committee (RAC). This chapter, however, has been prepared by Riverkeeper based on our own interpretation of the data reported by Buffalo State College and Youngstown State University, and is therefore outside of the RAC recommendations of required actions.

8.2 About Buffalo Niagara Riverkeeper

Buffalo Niagara Riverkeeper (formerly Friends of the Buffalo Niagara Rivers) is a not-for-profit organization dedicated to promoting, preserving and protecting the natural and historical environments of the Buffalo and Niagara Rivers and their environs for the benefit of the local community. Riverkeeper's mission is to improve waterfront access, restore watershed ecology, conserve river heritage, and cultivate river stewardship.

8.3 Water Quality

The water quality evaluation using the Hydrolabs at three fixed sites gave a comprehensive overview of the river's dynamics, and used in conjunction with the weekly observations at the 10 potential habitat restoration sites, Riverkeeper feels confident about the adequacy of the data.

Dissolved oxygen has long been known to be a major cause of use impairments of the Buffalo River, and it has been only recently that the complicated relationship between stratification, system hydraulics, SOD and BOD defined the problem. The results of this study support previous findings that DO levels will continue to fluctuate and frequently drop below state guidelines, unless additional, man-made controls are implemented. Some of these suggested controls include the cessation of navigational dredging and allowing the dredge channel to fill in, the implementation of an artificial aeration system within the dredge channel, and even utilizing the existing infrastructure of the BRIC system to increase flows during low-flow periods.

The increased turbidity levels near the river bed are also consistent with previous findings, and will continue to be a problem due to inputs from the upper watershed. Because of the lack of riparian and aquatic vegetation (i.e.: wetlands) to filter particulates, as well as the shoreline erosion and surface water run-off generated in the upper watershed, turbidity will continue to regularly exceed recommended levels. Additional efforts must be made on a watershed level, possibly through the implementation of TMDLs, to address this issue.

Though not in the original work plan for this study, testing E. coli levels proved very useful in experimenting with the user-friendly and inexpensive Coliscan Easygel system. The results were consistent with earlier findings, and therefore demonstrate how this system can be utilized by citizens or other user-groups in the future as part of ongoing river monitoring. The data supports the hypothesis that the majority of bacterial contamination is generated from the upper watershed. Whether the contamination comes from a combination of CSO and SSO outfalls, surface water run-off, or leachate from faulty septic systems, because primary contact and bathing continues to be popular uses of the river by local residents, bacterial contamination remains a high priority in the development of an update remedial strategy. Much more attention and resources need to be dedicated to identify and control the sources from the upper watershed such as failing residential septic systems.

8.4 Fisheries

Riverkeeper believes that fish diversity and health has not improved over the last decade based on the data obtained in 2003-04, and compared to data available from fish surveys of the early 1990s. A non-AOC reference community has not been identified yet to allow a comparison of the DELT anomalies rate, however Riverkeeper strongly believe that “a range of 14-87% frequency for the six most commonly found species” is not a natural condition. The 87% rate for brown bullhead is of special concern because this species lives in contact with bottom sediments. These observations continue to support the belief that fish health is degraded by the presence of contaminated sediments throughout the AOC impact area. The 35% DELT rate for largemouth bass, a species that is often caught and eaten by anglers along the Buffalo River, also raises serious health concerns.

Riverkeeper believes that the conversion of the IBI score into a quality rating for the ten habitat sites is a useful tool for comparing habitat sites. Again, by evaluating these ten sites using the stream rating score, all sites have been identified as being “poor” or “very poor.” While these determinations are based on current conditions and the IBI score, we should emphasize that these results alone should not preclude any of the sites from being considered for restoration efforts. Of special concern is the observed “drop” in IBI score for sites 5 and 6. Both sites lie between the two main meanders of the stream with little active industry or known contaminated sites in the adjacent areas, and therefore it would be expected that these sites should score higher. Further investigation of these two sites may be warranted.

Because the electrofishing surveys were conducted in June and August of 2003 and 2004, the results cannot be easily compared with the species composition observed in May-July of 1993. Regardless of the ability to compare the surveys directly, Riverkeeper feels that the surveys conducted adequately represent the diversity seen in the Buffalo River AOC. Follow-up surveys in the near future would be useful to observe the impact of NYSDEC's Walleye Restoration Project. The data generated from the fish surveys will be used to help establish and monitor delisting criteria and restoration targets for fish populations and fish deformities within the Buffalo River.

The only data set that was not obtained as a part of this project for the fisheries of the Buffalo River was tissue sampling for contaminants. According to NYSDEC, a variety of species that could be consumed were last tested in 1993-94 (including walleye, bass, bluegill, perch, eels, and pike) and determined not to pose a threat to human health. Though fish consumption advisories still exist for carp in the Buffalo River and carp is tested periodically by NYSDEC, Riverkeeper believes it appropriate to re-test other consumed species on a regular basis. This analysis can be conducted in association with angler surveys to confirm if there is a tainting of fish flavor, and if the current fish consumption advisories are adequate.

8.5 Benthic Macroinvertebrates

Degradation of benthos continues to be a major beneficial use impairment of the Buffalo River. Based on the data obtained from the benthic sampling and analysis, Riverkeeper believes that there has been no improvement in macroinvertebrate diversity and health during the last decade. Of great concern is the data that shows in-channel community richness decreasing. Riverkeeper agrees with the assertion by Youngstown researchers that the "post-industrial recovery of the Buffalo River in its present state may remain stalled without active remediation."

Of special concern is the low species richness observed at sites 5 and 6. In addition, site 6 had very high density of the pollution tolerant species (tubificid oligochaetes) with very low densities of chironomids. Viewed independently of other data these sites would not be suspect, but combined with the fish survey results and water quality analysis, Riverkeeper feels that further investigation of these two sites is warranted.

Much of the Buffalo River continues to have low species diversity and is dominated by pollution-tolerant species (oligochaetes), particularly at the sites within the navigation channel. In addition, the *Chironomus* larvae sampled within the navigation channel and analyzed for mouthpart deformity was a shockingly high 54.5%. (Just as surprising and even encouraging is that <u>all</u> of the limited *Chironomus* larvae sampled from shoreline sites had normal mouthparts; however, we recommend additional benthic sampling at the shoreline habitat restoration sites to verify the observed 0% deformity rate). Riverkeeper strongly suspects that the ongoing disturbance of contaminated sediments associated with navigational dredging is a major factor in the high occurrence of benthic deformities in the channel.

In sites where this data supports contaminated sediment remediation, the information generated will prove useful to the ongoing Feasibility Study for Environmental Dredging. In addition, the data will assist the Buffalo River Remedial Advisory Committee in identifying quantitative restoration targets for benthos as well as defining an updated remedial strategy for the Buffalo River.

8.6 Vegetation

The data collected for the vegetation survey will be useful to the ongoing habitat assessment and study of impervious surfaces in the Buffalo River AOC. The findings were not any different from what has been known about the ecosystem for over a decade, and that is the dominance of invasive and non-native species. Where all invasive species in the AOC need to be addressed, of special concern is the dominance of Japanese knotweed. Not only is the knotweed out-competing the other native vegetation, it is a continuously growing physical barrier to shoreline and aquatic habitat restoration efforts. Riverkeeper has identified invasive species in the Buffalo River AOC as a priority and is investigating pilot programs for phyto-remediation and other eradication efforts at selected sites.

8.7 Use Surveys

The use survey was the first time that researchers have attempted to quantify recreational uses of the Buffalo River. Though Riverkeeper generally agrees with the survey methods and adjustments, we still believe that the primary contact use of the river by local residents has been underestimated (i.e.: swimming estimated at 3% of total activity). Humans can come into direct contact with water through other activities such as wading, fishing, and boat launching. For nearly 20 years, the local communities and residents have communicated to Riverkeeper that swimming in the River occurs on almost a daily basis during the warm weather months.

Since very little historical or baseline information exists regarding recreational use of the waterways, it would be useful to continue the survey process in the future. Recreational usage of the Buffalo River is extremely relevant to the recent economic redevelopment efforts for the Inner Harbor and Ohio Street. Much investment in redevelopment and restoration projects is based on economic impacts, or return on investment. By quantifying angler use, boating use, birding, etc., local decision-makers would have a more accurate picture of the benefits that could arise from the redevelopment and restoration of the Buffalo River. Riverkeeper strongly recommends additional surveys on a much larger scale. Combined with a market analysis, additional surveys will help to accurately depict the level of all current and potential recreational activity within the AOC. The market analysis would clarify current recreational conditions and associated economic impacts of recreational activity; identify opportunities for improving and increasing recreational opportunities; and help develop a market-based strategy.

8.8 Site Matrix

Riverkeeper strongly supports the ranking and evaluation system that was created for the "Site Characterization Matrix." Because the data and information will be examined by the scientific community, local leaders, and average citizens, the project partners feel justified in simplifying the ranking system for quick and easy interpretation. However, Riverkeeper wants to emphasize that the ranking system is just one of many tools available to decision-makers when prioritizing sites for restoration. Many parameters have not been considered as part of this project, including local community support, upland land use, contaminated sediments, and resources available.

The final scoring for the 10 sites was not without a few surprises. It was expected that most of the sites within the two main meanders of the river (Sites 3, 4, 7, 8 and 9) were found to have the highest potential for restoration and ranked as the top priority areas. However, Riverkeeper is greatly concerned about the low ranking of sites 5 and 6 (adjacent to Concrete Central and the Katherine Street peninsula), that are also located within the two main meanders of the river. Although sites 5 and 6 have high DO levels and high fish diversity, they also have the highest fish deformity rate, lowest benthic rankings and lowest overall water quality scores. These results can not be explained through the data that is currently available from this study, and therefore Riverkeeper strongly suggests continued investigation in and around these sites which include; sediment analysis, water quality testing for contaminants, SPDES permits investigation, the possible impact of noise pollution or other unknown physical disturbance.

Overall, the matrix gives us a strong set of data to review when prioritizing site restoration. In addition, the break-out of site "positives" and "deficiencies" helps us to begin to identify resources needed as well as remedial options available on a case by case basis.

8.9 Next Steps

The data generated from this study will be immediately analyzed and evaluated by USACE as part of the ongoing Feasibility Study for Environmental Dredging. In addition, Riverkeeper will refer to the final study results as it facilitates the Remedial Advisory Committee's efforts to establish delisting criteria/restoration targets and an updated remedial strategy for the Buffalo River.

Riverkeeper will coordinate an effort to fully investigate sites 5 and 6 regarding its unexplained poor ratings and high deformities. In addition, Riverkeeper will coordinate with the local efforts dedicated to Inner Harbor revitalization in terms of obtaining additional user surveys and a market analysis of the AOC in the near future.

The site matrix has now given local decision-makers another tool in developing priorities for restoration of the Buffalo River. The next step is to identify possible funding sources, generate local community support, and coordinate partnerships for the implementation of recommended actions- as identified by the Buffalo River Remedial Advisory Committee.