CHAPTER 7
SITE EVALUATION MATRIX

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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 ^oC 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:

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

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

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

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:

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

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:

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

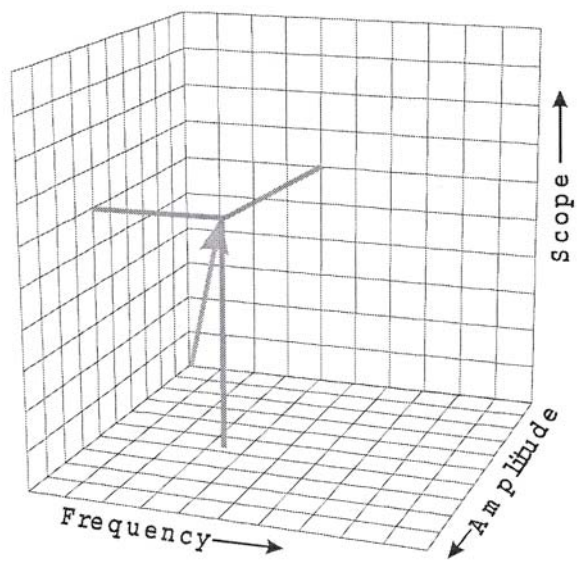

Figure 7.1 Conceptual model of the CCME WQI (from CCME, 2001a)

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.

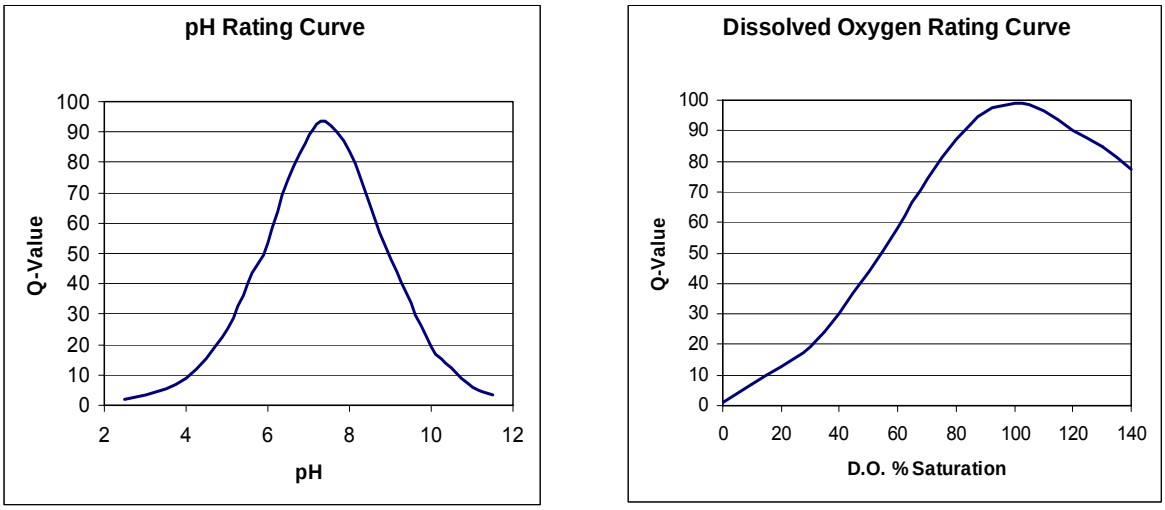

Figure 7.2 Q-value rating curves for NSF WQI

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.

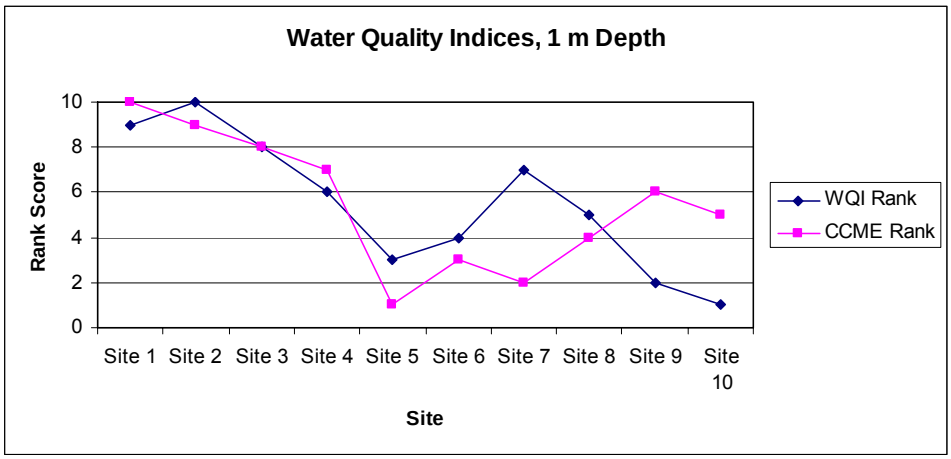

Figure 7.3 Water quality index rank score by site

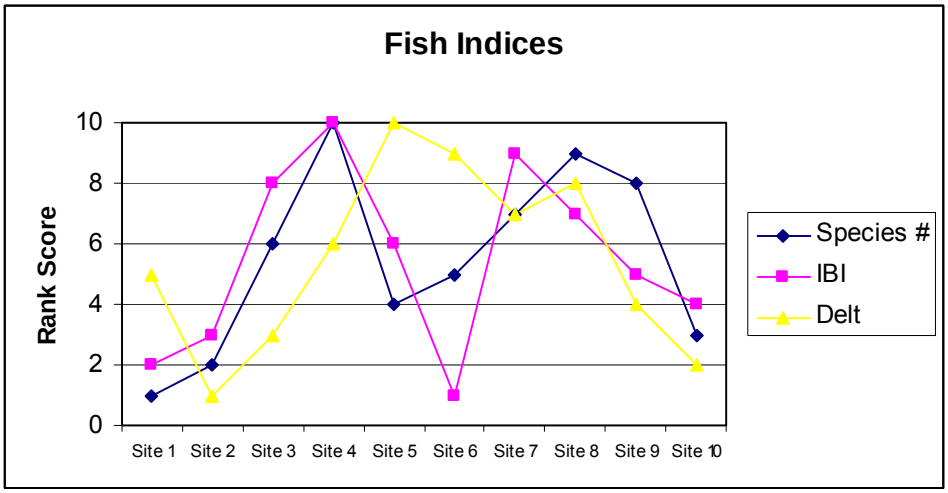

Figure 7.4 Fish indices rank score by site

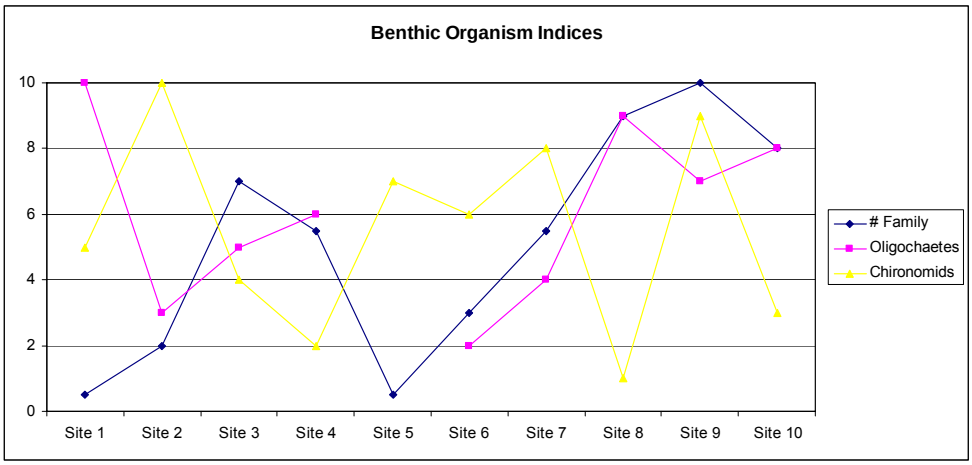

Figure 7.5 Benthic organism rank score by site

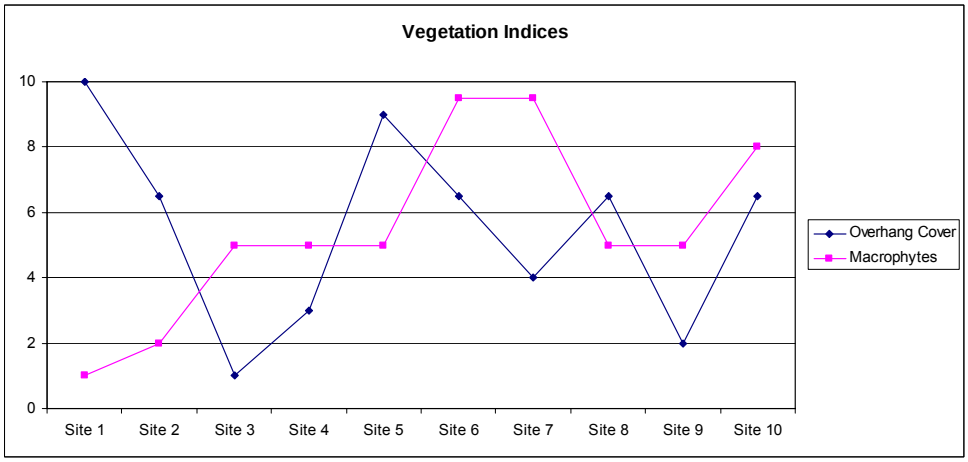

Figure 7.6 Vegetation (% overhang cover and number of macrophyte species) indices rank scores by site

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

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7.9 References

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