Biological Assessment and Criteria

Tools for Water Resource Planning and Decision Making

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CHAPTER 13

Habitat Indices in Water Resource Quality Assessments

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1.0 INTRODUCTION

A key concept of the Clean Water Act is the protection of biological integrity of the streams and rivers of the United States. Basic to maintaining diverse, functional aquatic communities in surface waters is the preservation of the natural physical habitat of these ecosystems. As obvious and basic as this concept seems, regulatory and protective efforts regarding habitat have been minimal (Hughes et al. 1990; Karr 1991). As a result many thousands of miles of United States streams have been and continue to be degraded each year (Benke 1990; NRC 1992). This loss of habitat quality has resulted in extinctions (Williams et al. 1989), local extirpations (Karr et al. 1985), and population reductions (Trautman 1981; Ohio EPA 1992) of fish species and other aquatic fauna (e.g., Williams et al. 1993) in the United States. In contrast to many other human impacts, habitat loss can be essentially irretrievable over a human time frame.

The lack of consistent habitat-protective efforts is reflected in the extreme inconsistencies in reporting aquatic habitat problems across the nation. Habitat, both instream and riparian, can be the factor most limiting aquatic community potential in streams and rivers. Observed habitat conditions are usually the result of complex interplay between hydrogeomorphological factors and anthropogenic landscape alterations (Gregory et al. 1991; Hill et al. 1991). Surprisingly, water chemistry is often the only one of five major factors that affect biological integrity (Figure 1) assessed to determine aquatic life use attainment. While all states and territories collect fecal coliform, dissolved oxygen, and other chemical specific data, few states effectively monitor for habitat destruction and alteration or effectively integrate existing habitat and biosurvey work into surface water monitoring programs. Often, habitat impacts are only considered in a framework of gamefish management (Osborne et al. 1991). Figure 2, derived from the National Water Quality Inventory (USEPA 1992), illustrates this problem. Of the 47 states or territories reporting impairment data on streams and rivers for the 1992 report, 25 did not report habitat as a cause of problems.

Much of the regulatory emphasis of the Clean Water Act as interpreted by the United States Environmental Protection Agency (USEPA) has focused on point sources of pollution (e.g., wastewater treatment plants [WWTPs], and industries) because of the obvious threats to human health and the relative ease, from a regulatory viewpoint, of dealing with a discrete source of pollution. Unfortunately, many of the most serious remaining problems and threats to the biological integrity of ecological systems (e.g., habitat destruction, urbanization and suburbanization, mining, grazing, and agricultural impacts) do not fit well into a point source control conceptual framework. This chemical-specific load reduction framework is insufficient by itself to address most habitat and development-related impairments to ecosystems. The development of the "River Continuum Concept" (Vannote et al. 1980; Minshall et al. 1983) resulted in an abundant literature that described the connections among the landscape, habitat quality and water quality, and that support holistic approaches to water resource management.

Efforts to reduce habitat destruction will require us to examine multiple scales of impacts, from landscape ecosystem to microhabitat scales and a move away from reductionist approaches (Karr 1991).

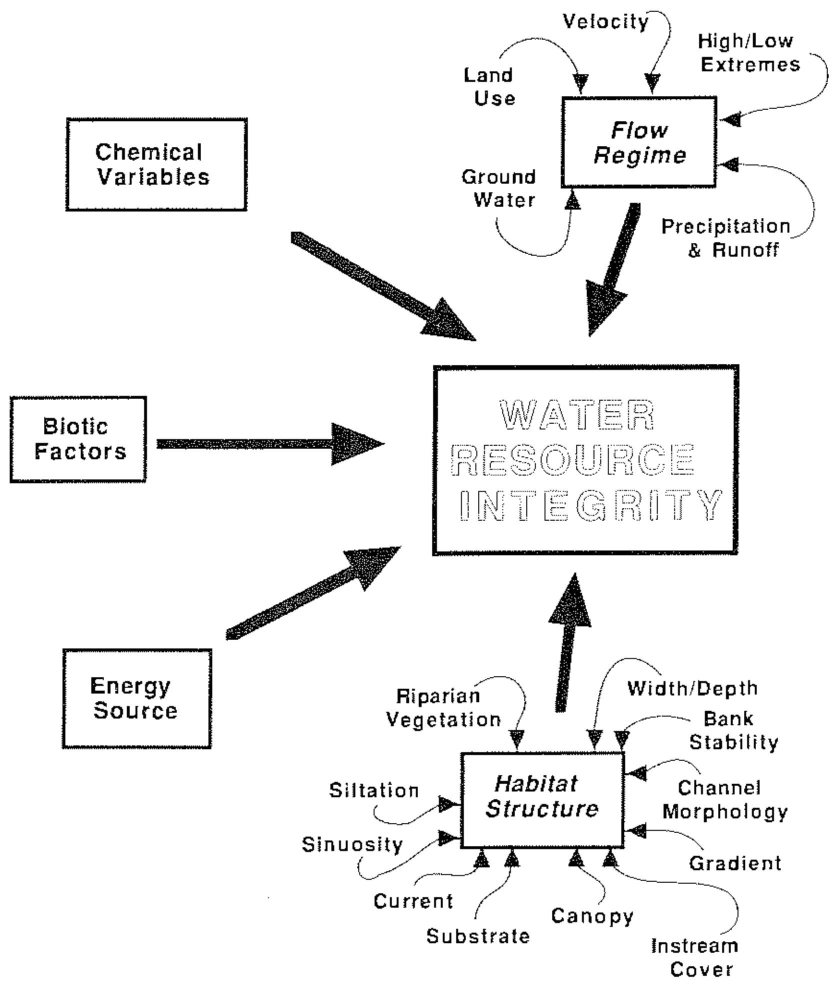

Figure 1. The five major factors that affect Water Resource Integrity with more detail on factors often used in habitat indices.

Such an effort will require states to improve their monitoring abilities to address and consider effects at each scale of impact. Monitoring data, including habitat assessments, can be used to rank physical and biological resource quality of streams, identify those streams and rivers threatened by anthropogenic changes, and to provide insight on possible remedies. Conversely, for areas that have had severe and essentially irretrievable (from a social and economic viewpoint) habitat losses, monitoring data can direct efforts towards areas where abatement efforts can be cost-effective and successful.

The objectives of this paper are to (1) review existing habitat assessment indices and methods, (2) explain how Ohio EPA incorporates its habitat index (the QHEI) in its integrated monitoring efforts, (3) delimit specific uses and limitations of habitat indices in state water resource quality management programs, and (4) encourage examination of water resource impacts at multiple spatial and temporal scales.

2.0 BACKGROUND

2.1 Approaches to Habitat Assessment

Various types of stream habitat indices and methodologies have been used in North America over the past 20 to 30 years (Table 1). The first, and most frequent use of habitat indices has been to relate the

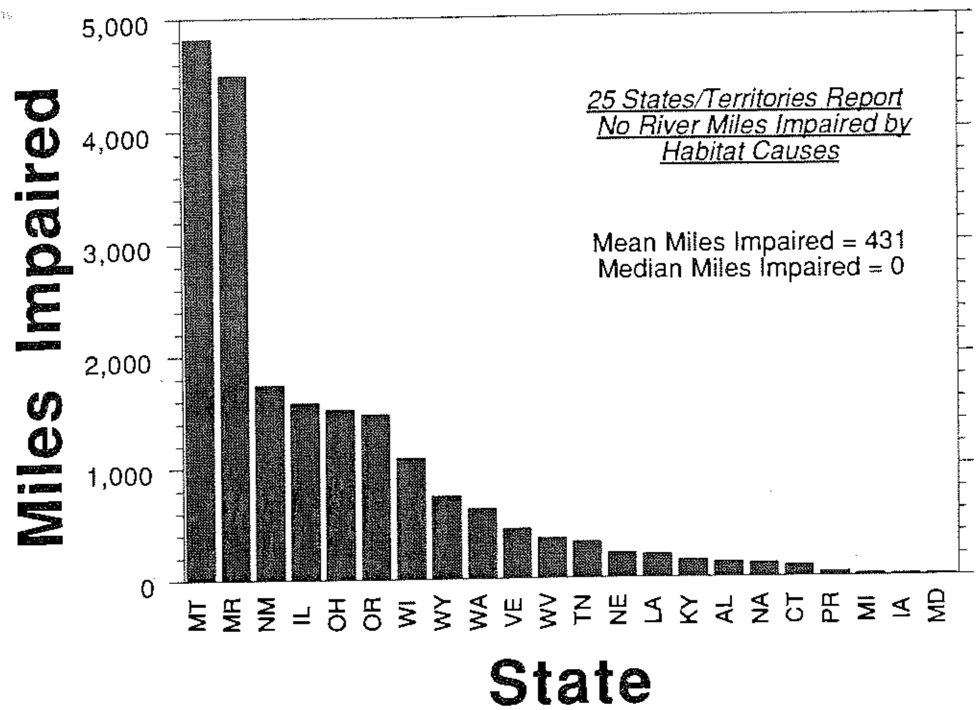

Figure 2. Miles of streams and rivers impaired by habitat causes, by state, summarized in the 1992 National Water Quality Inventory (USEPA 1992e). Data were reported by states to the USEPA through their 1990 305(b) reports.

standing crop or population of a target species, often a sport fish, to habitat characteristics in a stream. The usual goal of such work is to define limiting habitat factors to allow managers to manipulate stream habitat to enhance fishable populations. Most states use some type of transect or index method to accomplish such habitat assessments. Examples of this abound for salmonids (Binns and Eiserman 1979; Platts et al. 1983) as well as warmwater species (Layher and Maughan 1985; Layher and Brunson 1992). Another use of habitat indices, especially in the western United States is to determine the minimum or optimal stream flows that would protect habitat characteristics essential to the life history of one or more target species. The United States Fish and Wildlife Service (USFWS) has developed much of this work and it is summarized by Bovee (1982, 1986). Recent work by Hill et al. (1991) broadens this concept to include the importance of out-of-channel flow to habitat and riparian maintenance. This supports the call by Stalnaker (1990) for "progress beyond the minimal flow..., to focus on scientific principles in understanding riverine systems."

The third and most recent use of habitat indices, is as an integral part of water pollution control programs in states (Karr 1991). These states have recognized that the threats to biological integrity of streams are much more extensive than water quality threats alone. These monitoring programs use habitat indices to characterize the causes and sources of impacts and to help ascertain the potential for waters to support aquatic communities. Most of these habitat techniques focus on aquatic community responses rather than species-specific responses to changes in habitat quality, although the concepts are similar. The remainder of this paper will deal with the use of habitat indices in state water resource quality monitoring that focus on biological communities and biological integrity.

2.2 Habitat Indices in Use by States

Osborne et al. (1991) summarized habitat assessment programs in states of the American Fishery Society North Central Division (Midwest and Upper Midwest). Most of the habitat assessment efforts at

Table 1. A Selected Listing of Habitat Indices and Their Design Purpose Used in North America Over the Past 30 Years

Index/methodology	Purpose	Ref.
HEP/HSI	Relate habitat quality to single species carrying capacity	Terrell (1984); Layher and Maughan (1985)
HOI	Habitat as predictor of trout standing crop	Binns and Eiserman (1979)
BSC	Habitat quality used with IBI to determine biotic potential of a stream reach	Illinois EPA (1989); Hite (1988)
Transect methods	Assesses various aspects of stream habitat by taking measurements along transects in a reach	Dunham and Colotzi (1975); Platts et al. (1983); Armour et al. (1983); Duff et al. (1989)
Habitat diversity/ complexity	Shannon index application using substrate, depth, and velocity	Gorman and Karr (1978); Schlosser (1982)
HI	Missouri's habitat quality index based on ten components relating present status to pristine condition	Fajen and Wehnes (1982)
HCI	Habitat condition indicator for streambank and instream components	Duff et al. (1989)
BCI/DAT	Species diversity using habitat, species dominance, and taxa	Winget and Mangum (1979); Mangum (1986)
QHEI	Visual habitat method correlated with fish community condition (e.g., IBI)	Rankin (1989, 1991); Ohio EPA (1989)
IFIM	Used to determine flow needs of stream fish species	Bovee (1982, 1986)
RBP habitat qual.	Habitat evaluation based on stream classification guidelines for Wisconsin	Plafkin et al. (1989); Barbour and Stribling (1991); Ball (1982); Platts et al. (1983)

the time of the survey were directed towards game/sport fish management rather than towards broader efforts of protecting biointegrity or biodiversity. Four of these states used habitat assessments as a component of biological assessment, aquatic life use designations, or to identify reference reaches (IL, NE, OH, and WI). A more recent survey (Abe et al. 1992) of states in USEPA Region 5 (i.e., IL, OH, MI, IN, MN, and WI) has found that most states have begun or have initiated programs to address biological integrity within the framework of overall surface water protection programs. Other states around the country have reported on the use of the Rapid Bioassessment and Habitat Assessment Protocols of the USEPA (Primrose et al. 1991; Plotnikoff 1992; Hayslip 1993) or other similar methods incorporating the work of Ball (1982), Mangum (1986), Winget and Mangum (1979), and Platts et al. (e.g., Maret 1988 in Nebraska; Simonson et al. 1993 in Wisconsin). The burgeoning effort expended on assessment of biological integrity prompts consideration of some of the uses, limitations, and needs for habitat assessment protocols and a review of the existing literature.

3.0 METHODS

3.1 QHEI derivation

Many of the topics discussed in this chapter will be illustrated with examples from Ohio's experience with integrating the "Qualitative Habitat Evaluation Index" (QHEI) into its surface water monitoring program. The following is a brief summary of the calculation and scoring of the QHEI. A more detailed explanation is found in Ohio EPA (1988) and Rankin (1989).

The use of the QHEI is dependent on visual estimates of habitat features. Each of the habitat attributes assessed is summarized in Table 2. A sample data sheet is illustrated in Figure 3. Definitions and procedures for scoring the attributes in Table 2 are detailed in Ohio EPA (1988) and Rankin (1989); all staff using this index in Ohio go through a yearly training program. Scores for each category of the QHEI were originally assigned on the basis of a literature review of the response of warmwater fish species and communities to various habitat characteristics. These original scores were adjusted by examining the

Table 2. Physical Habitat Attributes Scored in Ohio EPA's Qualitative Habitat Evaluation Index (QHEI)

I. Substrate quality
  a. Two most predominate substrate types
  b. Number of substrate types
  c. Substrate origin (tills, limestone, etc.)
  d. Extensiveness of substrate embeddedness (entire reach)
  e. Extensiveness of silt cover (entire reach)
II. Instream cover
  a. Presence of each type in the reach
  b. Extensiveness of all cover in reach
III. Channel quality
  a. Functional sinuosity of channel
  b. Degree of pool/riffle development
  c. Age/effect of stream channel modifications
  d. Stability of stream channel
IV. Riparian quality/bank erosion
  a. Width of intact riparian vegetation
  b. Types of adjacent landuse
  c. Extensiveness of bank erosion/false banks
V. Pool/riffle quality
  a. Maximum pool/glide depth
  b. Pool/riffle morphology
  c. Presence of current types
  d. Average/maximum riffle/run depth
  e. Stability of riffle/run substrates
  f. Embeddedness of riffle/run substrates
VI. Local stream gradient (ft/mi) from 7.5' topographic map

Note: Habitat attributes are visually estimated over a 150 to 500-m reach that corresponds to a biological sampling reach.

response of the IBI, collected at a series of least impacted and habitat modified reference sites, to each of the QHEI habitat characteristics (Rankin 1989). These reference sites are the same as those used to derive Ohio's biological criteria, and are also discussed in Yoder and Rankin (Chapter 9) and DeShon (Chapter 15). This database, augmented with some newer data, will be used to illustrate many of the concepts discussed in this chapter.

Stream flow data (periodicities, peaks, minimums, etc.) are not an explicit part of the QHEI. The flow regimes to which a stream are subject, however, are a fundamental consideration when interpreting habitat and biological data. Incorporating the effects of flow on streams can be accomplished by stratifying streams according to their flow characteristics as has been proposed by Poff and Ward (1989). Extremely high, flashy flows can be limiting in certain small streams as can very low flows. However, in many situations in Ohio flow is not limiting and flow data are not always readily available; thus, it was excluded as part of the index. In addition, our habitat sampling is generally done in concert with biosurvey sampling which integrates and reflects the effects of past flow events. For some regions of the country measures of flow may be an essential component to include in an index.

4.0 RESULTS AND DISCUSSION

4.1 Regionalization of Habitat Approaches

A ecoregional approach to examining and managing surface waters has many advantages for organizing ecological data and interpreting man's impact on rivers and streams (Hughes et al. 1990). Ecologically pertinent stratification can simplify sampling approacbes (Gallant et al. 1989) and provide a conceptual and operational framework for defining biotic potential or biotic limitations (Hughes et al. 1990). Advantages for considering regional differences in habitat assessments are especially convincing. Habitat features, which often affect or limit biological communities, are a consequence of geomorphologic and other natural factors that are the basis of regionalization efforts such as ecoregions (Omernik 1987). For some areas of the United States the application of ecoregions to water resource components has been successfully demonstrated (Larsen et al. 1986; Rohm et al. 1987; Whittier et al. 1988).

Ohio EPA Qualitative Habitat Evaluation Index Field Sheet QHEI Score: ☐

Stream ____ RM ____ Date ____ River Code ____
Location ____ Scorers Name: ____

1] SUBSTRATE (Check *ONLY* Two Substrate *TYPE BOXES; Estimate %* or *note every type present*);

TYPE	POOL	RIFFLE	TYPE	POOL	RIFFLE
☐ ☐-BLDR /SLABS[10]	___	___	☐ ☐-GRAVEL [7]	___	___
☐ ☐-BOULDER [9]	___	___	☐ ☐-SAND [6]	___	___
☐ ☐-COBBLE [8]	___	___	☐ ☐-BEDROCK[5]	___	___
☐ ☐-HARDPAN [4]	___	___	☐ ☐-DETRITUS[3]	___	___
☐ ☐-MUCK [2]	___	___	☐ ☐-ARTIFICIAL[0]	___	___
☐ ☐-SILT [2]	___	___			

NOTE: (Ignore sludge that originates from point-sources; score on natural substrates)
NUMBER OF SUBSTRATE TYPES: ☐-5 or More [2] ☐-4 or Less [0]
COMMENTS ____

SUBSTRATE ORIGIN
Check ONE (OR 2 & AVERAGE)
☐ -LIMESTONE [1]
☐ -TILLS [1]
☐ -WETLANDS[0]
☐ -HARDPAN [0]
☐ -SANDSTONE [0]
☐ -RIP/RAP [0]
☐ -LACUSTRINE [0]
☐ -SHALE [-1]
☐-COAL FINES [-2]

SUBSTRATE QUALITY
Check ONE (OR 2 & AVERAGE)
SILT:
☐ -SILT HEAVY [-2]
☐ -SILT MODERATE [-1]
☐ -SILT NORMAL [0]
☐ -SILT FREE [1]
EMBEDDEDNESS:
☐ -EXTENSIVE [-2]
☐ -MODERATE [-1]
☐ -NORMAL [0]
☐ -NONE [1]

Substrate ☐ Max 20

2] INSTREAM COVER

TYPE: (Check *All* That Apply)
☐ -UNDERCUT BANKS [1]
☐ -OVERHANGING VEGETATION [1]
☐ -SHALLOWS (IN SLOW WATER) [1]
☐ -ROOTMATS [1]
☐ -DEEP POOLS> 70 cm [2]
☐ -ROOTWADS [1]
☐ -BOULDERS [1]
☐ -OXBOWS [1]
☐ -AQUATIC MACROPHYTES [1]
☐ -LOGS OR WOODY DEBRIS [1]
COMMENTS: ____

AMOUNT: (Check *ONLY* One or check 2 and *AVERAGE*)
☐ - EXTENSIVE > 75% [11]
☐ - MODERATE 25-75% [7]
☐ - SPARSE 5-25% [3]
☐ - NEARLY ABSENT < 5%[1]

Cover ☐ Max 20

3] CHANNEL MORPHOLOGY: (Check *ONLY* One PER Category OR check 2 and *AVERAGE*)

SINUOSITY	DEVELOPMENT	CHANNELIZATION	STABILITY	MODIFICATIONS/OTHER	
☐ - HIGH [4]	☐ - EXCELLENT [7]	☐ - NONE [6]	☐ - HIGH ["	☐ - SNAGGING	☐ - IMPOUND.
☐ - MODERATE [3]	☐ - GOOD [5]	☐ - RECOVERED [4]	☐ - MODERATE [2]	☐ - RELOCATION	☐ - ISLANDS
☐ - LOW [2]	☐ - FAIR [3]	☐ - RECOVERING [3]	☐ - LOW [1]	☐ - CANOPY REMOVAL	☐ - LEVEED
☐ - NONE [1]	☐ - POOR [1]	☐ - RECENT OR NO RECOVERY [1]		☐ - DREDGING	☐ - BANK SHAPING
				☐ - ONE SIDE CHANNEL MODIFICATIONS	

Channel ☐ Max 20

COMMENTS: ____

4]. RIPARIAN ZONE AND BANK EROSION - (*check ONE box per bank or check 2 and AVERAGE per bank*) ★River Right Looking Downstream★

RIPARIAN WIDTH
L R (Per Bank)
☐ ☐-WIDE > 50m [4]
☐ ☐ - MODERATE 10-50m [3]
☐ ☐-NARROW 5-10 m [2]
☐ ☐- VERY NARROW <5 m[1]
☐ ☐- NONE [0]

FLOOD PLAIN QUALITY (*PAST 100 FOOT RIPARIAN*)
L R (Most Predominant Per Bank)
☐ ☐-FOREST, SWAMP [3]
☐ ☐-SHRUB OR OLD FIELD [2]
☐ ☐-RESIDENTIAL,PARK,NEW FIELD [1]
☐ ☐ -FENCED PASTURE [1]
L R
☐ ☐-CONSERVATION TILLAGE [1]
☐ ☐ -URBAN OR INDUSTRIAL [0]
☐ ☐ -OPEN PASTURE,ROWCROP [0]
☐ ☐ -MINING/CONSTRUCTION [0]

BANK EROSION
L R (Per Bank)
☐ ☐ -NONE/LITTLE [3]
☐ ☐ -MODERATE [2]
☐ ☐ -HEAVY/SEVERE[1]

Riparian ☐ Max 10

COMMENTS: ____

5.]POOL/GLIDE AND RIFFLE/RUN QUALITY

MAX. DEPTH
(Check 1 ONLY!)
☐ - >1m [6]
☐ - 0.7-1m [4]
☐ - 0.4-0.7m [2]
☐ - 0.2- 0.4m [1]
☐ - < 0.2m [POOL=0]

MORPHOLOGY
(Check 1 or 2 & AVERAGE)
☐-POOL WIDTH > RIFFLE WIDTH [2]
☐ -POOL WIDTH = RIFFLE WIDTH [1]
☐-POOL WIDTH < RIFFLE W. [0]
COMMENTS: ____

CURRENT VELOCITY [POOL & RIFFLES!]
(Check *All* That Apply)
☐-EDDIES[1]
☐-FAST[1]
☐-MODERATE [1]
☐-SLOW [1]
☐-TORRENTIAL[-1]
☐-INTERSTITIAL[-1]
☐-INTERMITTENT[-2]

Pool/Glide ☐ Max 12

CHECK ONE OR CHECK 2 AND AVERAGE

RIFFLE /RUN DEPTH
☐ - GENERALLY >10 cm;MAX > 50 [4]
☐ - GENERALLY >10 cm; MAX < 50[3]
☐ - GENERALLY 5-10 cm[1]
☐ - GENERALLY < 5 cm [RIFFLE=0]
COMMENTS: ____

RIFFLE/RUN SUBSTRATE
☐-STABLE (e.g.,Cobble, Boulder) [2]
☐-MOD. STABLE (e.g.,Large Gravel) [1]
☐-UNSTABLE (Fine Gravel,Sand) [0]

RIFFLE/RUN EMBEDDEDNESS
☐ - NONE [2]
☐ - LOW [1]
☐ - MODERATE [0]
☐ - EXTENSIVE [-1]
☐ - NO RIFFLE [Metric=0]

Riffle/Run ☐ Max 8

6] GRADIENT (ft/mi): ____ **DRAINAGE AREA (sq.mi.):** ____ %POOL: ☐ %GLIDE: ☐ %RIFFLE: ☐ %RUN: ☐

Gradient ☐ Max 10

EPA 4520 4/7/92

Figure 3. Qualitative Habitat Evaluation Index (OHEI) field sheet used by Ohio EPA.

It is obvious from differences in geomorphology and anthropogenic impacts around the country that states or agencies need to tailor habitat monitoring methodologies that (1) reflect factors that are likely to be limiting in their jurisdiction, (2) are sensitive to the range of habitat disturbance likely to be encountered, and (3) provide for an assessment of conditions necessary to evaluate and maintain genetic biodiversity and viability of aquatic biota. Adjusting indices for local geomorphology, land use, and biota will likely result in more accurate predictions or at least a more useful tool for examining responses of the aquatic biota. Hayslip (1993), for example, held workshops in USEPA Region 10 where water quality personnel adjusted and field tested biological and habitat parameters of the Rapid Bioassessment Protocols. National efforts, by USEPA and other federal agencies, should focus on developing guidelines for ecoregionalization of methods and they should promote training and proper quality assurance/quality

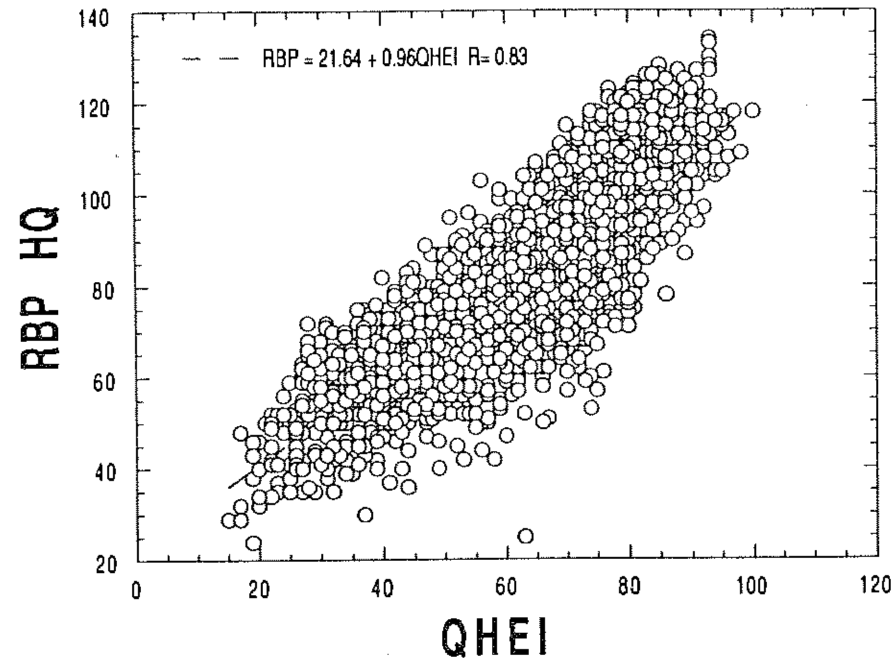

Figure 4. Qualitative Habitat Evaluation Index (QHEI) for all sites in Ohio EPA's database in relation to the Rapid Bioassessment Protocol Habitat Quality (RBP HQ) methodology derived from the same data.

control (QA/QC) procedures (e.g., as was done in Region 10 with regional modifications to the Rapid Bioassessment Protocols; Hayslip 1993). Common habitat attributes of streams, however, should have standard definitions as well as minimum standards for measurement (Armentrout 1981). A likely avenue for regionalizing habitat measures would be through the Environmental Monitoring and Assessment Program (EMAP; Paulsen et al. 1991; Kaufman 1993) in concert with national monitoring efforts of other agencies (USFWS Biomonitoring of Environmental Status and Trends [BEST] Program), USGS National Water-Quality Assessment [NAWQA] Program (Meador et al. 1993); and USFWS Aquatic Ecosystem Analysis Program (Mangum 1986a, 1986b).

To illustrate potential downfalls of accepting a national monitoring tool without local adjustments, I compared Ohio's Qualitative Habitat Evaluation Index (QHEI) was compared to USEPA's rapid bioassessment protocols for habitat quality assessment (RBP HQ). USEPA urges users of the protocols to tailor them to regional conditions (Barbour and Stribling 1991); thus, this is not an effort to "validate" this methodology, but, rather to illustrate a loss in the power of a habitat index when not adjusted to regional conditions.

The major difference between the QHEI and the RBP habitat tool is not the types of variables examined, but rather in the weighting of these factors as to their influence on biological integrity. To illustrate this each of the "metrics" or categories of the RBP HQ was generated from data components also collected for the QHEI. Because the RBP HQ results were derived from the QHEI results the actual degree of relationship could differ if the two were scored independently. However, the pattern of results related to differences in score weightings discussed below should still be valid.

The QHEI and the RBP HQ are significantly correlated (Figure 4). When compared statewide with the IBI at Ohio EPA reference sites (least impact and physically modified), however, the QHEI explained more of the variation in the IBI than did the RBP HQ (Figure 5). The use of other response variables such as the number of sensitive species at a site and the percent of individuals captured that were tolerant showed similar patterns (Figure 6). The better performance of the QHEI compared to the RBP HQ is not attributed to some inherent superiority in the QHEI. Rather, the improved explanatory power is likely related to: (1) the fact that the QHEI was calibrated, and metric scores were weighted, based on both literature reports *and* observed correlations of the IBI with habitat characteristics at a series of least

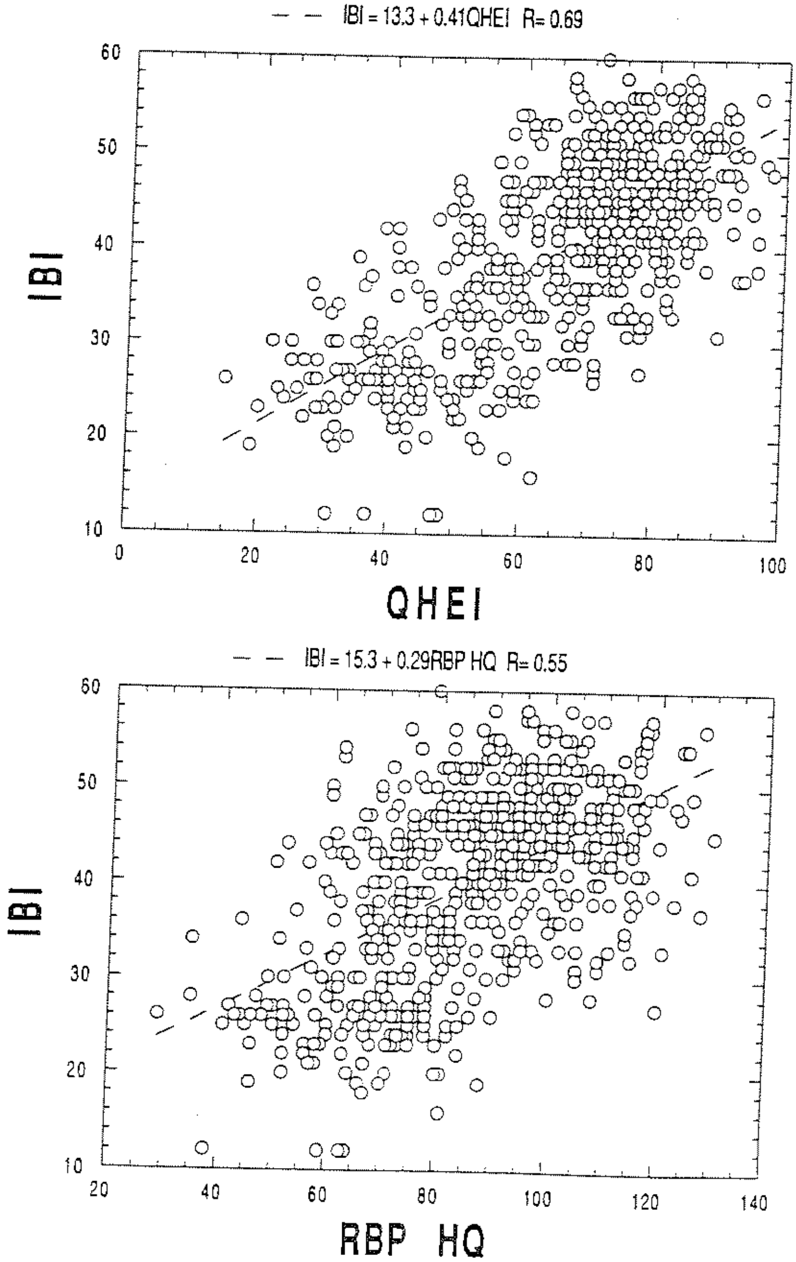

Figure 5. Top: Index of Biotic Integrity (IBI) values in relation to the QHEI at Ohio EPA's natural and modified reference sites (ecoregions differentiated by point type). Bottom: Index of Biotic Integrity (IBI) values in relation to the RBP HQ at Ohio EPA's natural and modified reference sites (ecoregions differentiated by point type).

impacted and physically modified references sites, (2) the QHEI is designed to measure those components most important to one organism group; fish, and (3) the relationships between QHEI and IBI most strongly reflect the anthropogenic disturbances common to Ohio (e.g., channelization and riparian destruction).

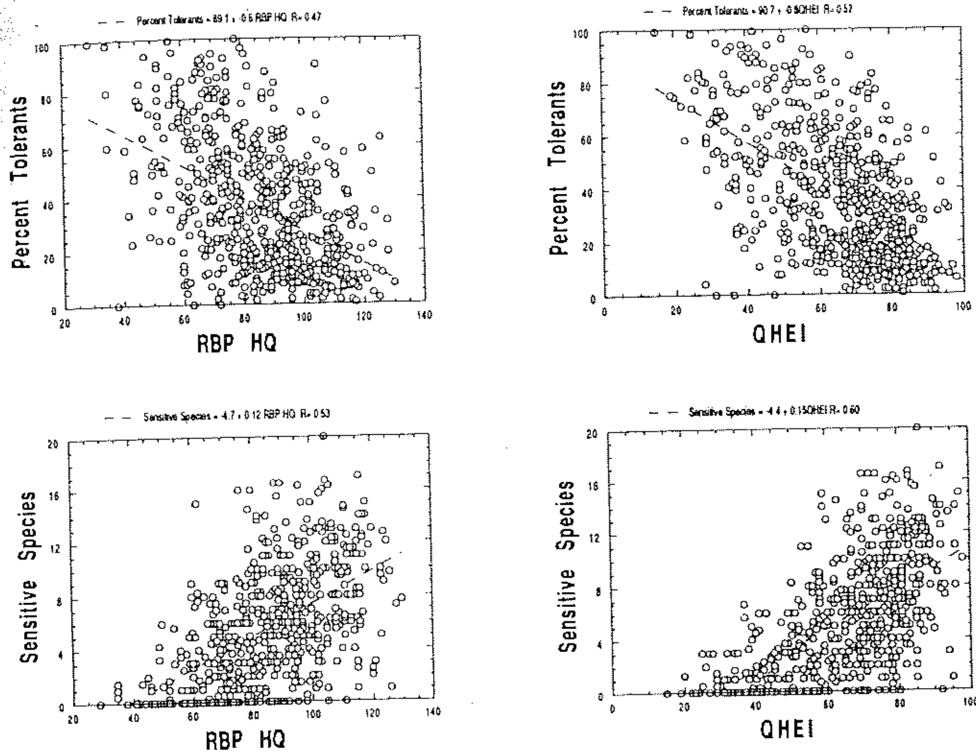

Figure 6. Top left: Percent tolerant individuals in relation to the RBP HQ at Ohio EPA's natural and modified reference sites. Top right: Percent tolerant individuals in relation to the QHEI at Ohio EPA's natural and modified reference sites. Bottom left: Number of sensitive species in relation to the RBP HQ at Ohio EPA's natural and modified reference sites. Bottom right: Number of sensitive species in relation to the QHEI at Ohio EPA's natural and modified reference sites.

4.2 Essential Components of any Habitat Index

Given that each habitat index needs to be calibrated regionally, preferably to a group of reference sites, are there components that should be common to any habitat quality index? The underlying effect of geomorphology on lotic ecosystems results in number of habitat characteristics that should be considered in any index. Calibrating habitat expectations upon reference conditions should account for the relative influence of geomorphology and stream energy (gradient) and important factors should become evident. Karr and Dudley (1981) summarize general characteristics of natural and modified streams in the eastern United States while Platts et al. (1983) provides a thorough analysis of many of the field techniques pertinent to all streams. The USEPA Rapid Bioassessment Protocols are largely based on the work of Platts et al. (1983) and Ball (1982). Bisson et al. (1981) provide excellent drawings and descriptions of common channel characteristics (pool/glide/riffle types). Other useful resources are Hynes' (1970) *The Ecology of Running Waters*, the various symposia that summarize many of the types of habitat assessment approaches used by workers across North America (Krumholz 1981; Armentrout 1982), and summaries of techniques and state programs by regional chapters of the American Fishery Society (e.g., Western Division, American Fishery Society 1985; Osborne et al. 1991). Thorough reading of some of the excellent geomorphology and hydrology texts that exist (e.g., Leopold 1964; Morisawa 1968; Gordon et al. 1992) will provide much insight to the types of forces that may be influential in various regions of the country. In addition, EMAP, which crosses political boundaries, should eventually provide regional data and useful field tests of habitat techniques in small streams and rivers. Preliminary discussions from an EMAP design workshop (Kaufman 1993) suggested potential stratifications for data analysis (ecoregion, size, and gradient) and eight categories of stream attributes (channel/riparian interaction, stream size, channel gradient, habitat type/distribution, channel substrate, riparian vegetation, and anthropogenic alterations) that are likely candidates for field data collection.

Barbour and Stribling (1991) describe four generic categories of stream types, mountain, piedmont, valley/plains, and coastal, for which the relative importance of habitat characters will differ. Much of the variability in habitat conditions among these stream types is related to the inherent energy in streams and the types of materials through which the streams flow (bedrock, sand, tills, alluvial deposits, etc.). Barbour and Stribling (1991) modified the RBP habitat procedures to provide separate methods for high gradient (riffle/run dominance) and low gradient (pool/glide dominance) streams. Mangum (1986a, b) describes the use of the Biotic Condition Index (BCI), which uses species diversity based on dominance and taxa to describe instream habitat conditions, and which can be used with or without RBP methodologies.

As mentioned earlier, most states have some group within their natural resource departments or universities that have some expertise in habitat assessment that should be involved in the development of procedures. For states beginning to incorporate habitat assessments into surface water protection programs the most conservative approach is to collect information on an array of habitat features and then refine an index on the basis of the desired response variable (e.g., IBI) from a series of reference sites (including physically impaired sites). The caveats of Platts (1981) to consider the accuracy and precision of assessment techniques should be addressed when choosing assessment tools. A sensible approach would be to include existing techniques where some idea of the variability of the measures has been assessed (e.g., Platts et al. 1983) and later modify these techniques if necessary rather than creating new methods.

Hawkins et al. (1993) suggest a hierarchical approach for classifying stream features, based on increasingly fine descriptions of stream morphology and hydrology within channel units. In such an approach, habitat measures may be made, for example, within a rapid (fine resolution), which is a subset of turbulent, fast water channel units (coarse resolution). Their categories for the classifications largely retain the nomenclature of Bisson et al. (1982) and Helm (1985). Use of a hierarchical approach such as this could be a catalyst towards some useful standardization of habitat type classification in streams (Hawkins et al. 1993).

Habitat altered sites, though they may be avoided by some biologists, should be sampled to provide a range of conditions under which to examine community responses to various impacts. The following section discusses habitat attributes that most indices should consider; useful (but not exhaustive) references for each attribute are included.

4.2.1 Substrate Type and Quality

All habitat indices should measure several characteristics of substrates. For most streams with all but the lowest gradient the type and quality of substrate conditions can be limiting. For most streams coarser substrates (gravels to boulders) are more likely characteristic of unaltered reference conditions. The addition of finer substrates via erosion is generally associated with land use changes and habitat modifications. As fines fill up the interstices of the larger substrates, the substrates are considered to be embedded. Some measure of the percent fines or degree or extent of embeddedness (Everest et al. 1981; Platts et al. 1983) is common to most habitat indices. Similarly, the degree to which substrates are covered by clayey-silts is also a common attribute of habitat indices. Sedimentation is widely held as responsible for degradation of fish communities in warmwater (Trautman 1981; Berkman 1987) and coldwater (Tappel and Bjornn 1983; Platts et al. 1989) streams, and many of the mechanisms of this degradation (loss of spawning habitat, lowering of interstitial dissolved oxygen, loss of habitat space, reduction in benthic production) have been well documented (Chapman 1988).

Management decisions should influence the choice of substrate assessment methods. Visual estimates of substrate types and conditions (Platts et al. 1983; Bain et al. 1985; Ohio EPA 1988; Plafkin et al. 1989) can be useful overall indicators of substrate quality; however, more specific objectives may call for more statistically rigorous methods (Everest et al. 1981; Platts et al. 1983) of substrate assessment. The sediment embeddedness criteria for salmonid spawning (Burton et al. 1991) discussed earlier provides a situation justifying very detailed field and lab measurements.

4.2.2 Instream Physical Structure/Cover

The presence of instream physical structure has a significant influence on aquatic organisms (Weshe 1980; Angermeier and Karr 1984; Weshe et al. 1987). Common attributes in habitat indices include the percent of a study reach with cover and recording the occurrence and extent of various types of cover. Types of physical structures or cover frequently measured or recorded include logs and woody debris, boulders, aquatic macrophytes, rootwads and rootmats, undercut banks, deep pools, and overhanging vegetation. Riparian forests are important contributors to instream cover (Murphy and Koski 1989). With the loss of riparian vegetation and the extensive dragging of woody debris from stream channels throughout much of the United States, streams are likely to have much less debris than has been historically present. For example, in old-growth streams in Oregon, Sedell et al. (1984) reported that woody debris "intervened" 16 to 18 times per 100 m of stream, a much higher rate than commonly found in the less-than-pristine streams of Ohio. Andrus et al. (1988) found that riparian trees must grow longer than 50 years to ensure an adequate, long-term supply of woody debris for stream channels. Similarly, in Delaware, analysis of stream recovery from modifications showed the most recovery at 30 years, roughly the time needed for trees to grow and start to fall into the stream channel (Maxted, unpublished data). In some low-gradient stream and river systems, physical structures such as logs and woody debris are the major source of invertebrate production (Benke et al. 1985; Benke 1990). The importance of instream structure has been documented for both coldwater and warmwater aquatic life. Physical structure can function to create pools and depth and velocity heterogeneity, to reduce export and increase processing of organic matter, as refuge from predation, as a substrate for prey organisms, as resting places from high velocity flows, and as spawning and nursery habitat (Angermeier and Karr 1984).

4.2.3 Channel Structure/Stability/Modifications

The natural channel morphology of stream and rivers is related to the geomorphology of an area, especially the energy of a stream (related to stream gradient) and the erodability of the material through which it flows. Streams in high-gradient areas generally flow "straighter" and erode less than streams in low-gradient areas that often meander through alluvial sediments that are more easily eroded. Unfortunately, hundreds of thousands of miles of streams and rivers have had their natural channel morphology altered significantly enough to impair both channel maintenance and the aquatic life in these systems. Alteration of the stream channel affects streamflow, the aquatic biota, and many of the characteristics measured by habitat indices (Emerson 1971; Trautman and Gartman 1974).

In Ohio most unaltered streams are sinuous or meandering. Some areas of the state were vast wooded wetlands (Black Swamp in northwestern Ohio) where stream channels were probably not well defined (Trautman 1981). The most common modifications in Ohio included channel straightening and deepening (channelization) for agricultural drainage and flood control. These activities destabilized streambanks and bottom substrates by increasing local gradients, increasing sedimentation, and exacerbating the peaks of storm flows. Such activities are also indicative of land use activities too close to stream channels that lead to increased sediment and pollutant runoff. Besides such physical changes, modified streams may alter recruitment of young fish to large rivers, resulting in fewer piscivores and insectivores and more omnivores and herbivore–detritivores. Thus, in Ohio we have habitat metrics that reflect ranges of sinuosity, channel modifications, and stream channel stability. A simple measure of sinuosity is the ratio of the stream path between two points and the straight line distance between these points (Leopold et al. 1964). The QHEI includes a functional correlate of sinuosity, pool formation on outside bends, to augment this habitat attribute.

Each state needs to determine the natural channel forms expected in a region and design its habitat indices so they detect important changes to the morphology and the other habitat attributes such changes may affect. It must be remembered that changes in channel morphology will export the effects downstream and sometimes upstream (e.g., head cutting). Other activities such as dams may reduce downstream deposition of alluvium and lead to bank erosion and changes in stream channel morphology (Johnson 1992). Thus, it is important to be able to detect the affects of upstream activities on channel morphology in the attributes of a habitat index.

4.2.4 Riparian Width/Quality

The quality and extent of the riparian vegetation is another critical component of a habitat index. More than other habitat components, however, the effects of removing or disturbing riparian vegetation often work at landscape scales (Gregory et al. 1991). While channel modifications have both immediate and downstream and upstream influences, the influence of riparian disturbance may be less evident in the immediate vicinity of a disturbance but become evident throughout a basin as riparian disturbances accumulate. In addition, the relative influence of riparian floodplain size to ecosystem function increases with stream size (Schlosser 1991). The importance of these large scale influences will be discussed later in this chapter.

Steedman (1988) examined the IBI in relation to land use and the existence of riparian zones near Toronto, Canada, and found significant correlations with both factors. He was able to generate a contour plot of qualitative IBI ratings as function of the percent urban land use and the proportion of upstream channels with intact riparian forest. Such studies can serve as models for the types of data needed to make habitat information a much more useful planning tool for preserving ecological integrity and riparian areas.

It is likely that no stream or river in Ohio has truly mature riparian forests that function as climax riparian forests functioned before European settlers arrived. In studies in "old-forest" areas of the Pacific Northwest, large logs were found to reside in a channel for a century or more (Sedell et al. 1984). Thus what could appear as a relatively undisturbed stream in Ohio could actually be an early successional stage with regard to riparian condition. In states with a great age variety in riparian forests, some estimate of forest maturity should be included in an index or accounted for in reference site stratification.

Habitat indexes often estimate the width of the riparian vegetation (i.e., trees, shrubs, and wetland) and the specific ages, stability, and species present. Because riparian tree species often require specific environmental conditions, such as out-of-bank flows, to germinate and grow (Hill et al. 1991), the composition of the vegetation can provide insight into previous environmental conditions. As riparian vegetation is degraded, lost functions include maintenance of narrow and deep channels (Platts and Rinne 1985), ineffective nutrient removal (Schlosser and Karr 1981; Lowrance et al. 1984; Peterjohn and Correll 1984), increased water temperatures (Karr and Schlosser 1977; Schlosser and Karr 1981), sedimentation and increased bank and bed erosion (Karr and Schlosser 1977, 1978), loss of terrestrial litter inputs and increased rate of organic export (Sedell et al. 1984), and loss of cover through woody debris "starvation" and loss of bank-related cover (e.g., undercut banks, rootwads, and rootmats) (Karr and Schlosser 1977). Work over the last decade by investigators, such as Bencala (1993), have emphasized the importance of riparian areas for maintaining the quality and function of the hyporheic zones of streams.

4.2.5 Bank Erosion

Bank erosion problems often occur hand-in-hand with riparian vegetation disturbance; however, bank erosion can occur in areas with "apparently" intact riparian vegetation. Stream channel alterations upstream in a watershed can drastically alter high flow characteristics making erosion problems more common downstream. Livestock grazing in riparian areas can also increase bank erosion and the formation of false banks. Typical modeling approaches to sediment runoff (e.g., Universal Soil Loss Equation, USLE) often do not account adequately for contributions from bank erosion or deposition (Schlosser and Karr 1981) nor the relative importance of particle types in runoff (e.g., clay vs. sand), which can have profound influences on aquatic community integrity (Ohio EPA 1992).

The streambank soil alteration rating and the streambank vegetative stability rating of Platts et al. (1983) and Duff et al. (1989) are widely used measures of bank erosion and the potential for bank erosion. The ability of a stream bank to erode will vary by region with the steepness of banks, bank materials (e.g., bedrock vs. alluvial soils) and stream gradient. As with the other measures, examination of reference sites will be useful in defining expectations for bank conditions.

4.2.6 Flow/Stream Gradient

Flow is not explicitly measured in Ohio's QHEI; however, as discussed earlier stream flow characteristics influence many of the habitat attributes of streams. Hill et al. (1991) examined four flow regimes that maintain physical and biological resources in stream ecosystems: (1) flood flows that form floodplain and valley features; (2) overbank flows that maintain surrounding riparian habitats, adjacent upland habitats, water tables, and soil saturation zones; (3) in-channel flows that keep immediate streambanks and channels functioning; and (4) in-channel flows that meet critical fish requirements. Streams in Ohio with the highest biological quality have natural flow regimes that include occasional flood flows that create and cleanse habitat, but which are not so frequent (as in urban streams) that they repeatedly scour bottom substrates and "reset" invertebrate communities (Matthews 1986). Work has shown that highly variable and unpredictable flow regimes (e.g., from urban runoff or controlled releases from dams) can have strong influences on fish assemblages (Bain et al. 1988). Extreme low flows are generally a problem in headwater areas (northwestern Ohio) where drainage activities have sped water off the landscape rather than slowly releasing water to stream channels. Small streams with such variable flows are dominated by tolerant and pioneering fish species that can withstand fluctuations in dissolved oxygen and temperature (Schlosser 1985; Matthews 1990; Schlosser 1990). Such broad landscape changes (e.g., draining most of NW Ohio) are responsible for the reductions in the distribution of many fish species across Ohio (Trautman 1981; Ohio EPA 1992; Yoder and Rankin, Chapter 9).

Each state will need to decide the advantages of explicitly including a measure of flow in a habitat index or using flow as an ancillary variable for interpreting symptoms of flow related problems that are observed in various habitat metrics and the biological communities from biosurvey data. One promising approach is to stratify streams by their flow characteristics. Poff and Ward (1989) examined streamflow characteristics across the United States, and on the basis of flow variability, flood regime patterns, and extent of intermittency distinguished nine stream types: harsh intermittent, intermittent flashy, intermittent runoff, perennial flashy, perennial runoff, snow melt, snow/rain, winter rain, and mesic groundwater. Examining biological performance and habitat conditions within such a conceptual framework, or for small regions a more finely divided framework, could improve explanations of patterns seen in aquatic assemblages and provide another useful form of stratification.

4.2.7 Riffle-Run/Pool-Glide Quality/Characteristics

Unaltered streams and rivers in the Midwest typically have fast, deep riffle/run complexes with large diameter substrates and deep pools with extensive physical structure. Even streams that, because of low gradient, lack riffles and runs often have a variety of flow regimes and depth heterogeneity associated with outside bends and meander patterns. Lobb and Orth (1991) examined a large warmwater stream in West Virginia and found five habitat-use guilds associated with the types of riffle/pool habitats: edge pool, middle pool, edge channel, riffle, and generalist. Degradation or loss of these types of habitats will eliminate or reduce abundance of the species in these guilds. Thus, it is important to measure the quality of these types of stream habitats.

Reference conditions can be used to determine the expected riffle/run and pool/glide types and their qualities for a region. In Ohio, stream channelization, siltation/sedimentation, and riparian destruction generally result in the loss of deep pools, the degradation of riffle habitat, and the predominance of shallow pool or glide habitat. The quality of riffles, runs, and pools is a direct result of the balance between erosion and deposition in natural systems. Many warmwater fish and macroinvertebrate species are habitat specialists and are eliminated as riffle and/or pool habitats are degraded. In Ohio, species associated with clean pool habitats appear to be especially vulnerable as relatively minor increases deposition of fine sediments has eliminated habitats and reduced distributions for many of the species over a wide area (e.g., sand darter, crystal darter, bigeye chub, and harelip sucker) (Trautman 1981). As sedimentation increases, even riffle habitats can become covered with fine substrates or more likely have large substrate interstices embedded with fine materials. Dunham and Collotzi (1975), Platts et al. (1983), and Duff et al. (1989) provide a rating for pool quality for small streams that incorporates pool morphology, stream depth, and instream physical cover and that could be modified for smaller or larger streams. Bisson et al. (1981) and Helm (1985), provide descriptions of various types of pools and riffles that are found in natural streams, and Hawkins et al. (1993) suggest a hierarchical framework for classifying such habitat types. As sediment delivery increases to a stream the morphology of the stream channel changes accordingly, often by becoming linear or convex in profile (Heede and Rinne 1990). Detailed measurements of stream channel morphology using cross-sectional transect techniques can provide statistics on changes in morphology, i.e., is it becoming wide and shallow or narrow and deep

(Olson-Rutz and Marlow 1992). Most index approaches, including the pool metric of the QHEI, which was derived from the Platts et al. (1983) and the (Habitat Condition Index HCI) from Duff et al. (1989), include some estimate of pool depth, morphology, instream cover, and sometimes velocity characteristics.

4.3 Importance of Reference Sites

A robust set of regional reference sites is critically important to accurately use biosurvey and habitat data. Single or multiple upstream control sites are important for interpreting longitudinal changes in the biota or habitat quality, but regional reference sites allow the quality of a stream to be placed in a broader perspective. In one sense, the use of single reference sites is a tie to the point source conceptual approach towards regulating water quality. For nonpoint or habitat problems, landscape-wide or broadscale land use problems that may be affecting habitat quality could likely be affecting the "control" condition as well. Anthropogenic changes may interact with the local geomorphology and have effects that may only be understood well when compared to regional patterns in biointegrity and habitat quality. Even when examining localized channel impacts such as bank erosion, the precipitating actions for such problems may originate from activities upstream in the basin.

The number of sites needed to accurately define baseline conditions will vary with the heterogeneity of the reference region and the variability inherent in the data. Yoder and Rankin (Chapter 9) consider this question for deriving biocriteria in Ohio (the 25th percentile of regional reference sites as a baseline for the Warmwater Habitat aquatic life use). To use habitat data effectively there also needs to be a suite of physically modified reference sites free from point source impacts. These modified reference sites should incorporate a broad range of habitat problems to allow sufficient resolution to document biological responses to multiple limiting factors. In Ohio, we have modified sites that reflect channel alterations, impounded streams, and nonacid, mine-related habitat impacts.

4.4 Need for Standardized Approaches and Quality Assurance Procedures

A call for "standardized" approaches to habitat assessment is not at odds with the call for regionalization of habitat assessment indices. Within a state or region, after one or more methodologies are selected, they need to be well defined, including specific purposes and objectives for each approach. For states or groups just beginning to develop or adopt habitat assessment procedures, effort to define QA/QC procedures is essential. The system, however, should be flexible enough to allow evolution in the specifics of each method. Sufficient regional reference sites should be sampled and habitat data compared to biosurvey results before a complete methodology is selected. As a result, it is advantageous to collect a broad spectrum of habitat data and examine multiple methodologies. Factors to consider when determining which individual habitat characters to measure include: (1) habitat characters with minimal between-user variation, (2) habitat characters that are clearly linked to biological responses, (3) the inclusion of habitat characters that measure each scale that can affect the biota (e.g., microhabitat to landscape), (4) characteristics likely to be affected by the major categories of stream alterations in a state, and (5) the time and effort required to measure/estimate the characteristic (i.e., cost-effectiveness).

Training is essential for reducing the between user error in habitat assessment methodologies. At Ohio EPA, we have yearly multiday training sessions consisting of classroom and hands-on field exercises. Data from these training sessions are used to refine the methods and to reduce user variation. In Ohio, we have found that it is extremely useful for "office" staff as well as field staff to be well versed in the methodologies and to know why examining habitat is important in protecting aquatic life uses. Incorporating these staff members into such training forces a clarification of the objectives of collecting such data.

Each state should develop a stream protection policy that clearly defines the need, mechanism, and technical justification for protecting stream habitat. In agencies still steeped in the point source conceptual framework of pollution control, it is important in making regulatory staff aware of the importance of habitat to ecological integrity. For example, it was our district water pollution control staff that, because of an awareness of habitat's significance to the biota, were able to identify that a permit-to-install (PTI) for a sewer line down the channel of a high-quality stream would have severe consequences for aquatic life and likely violate Ohio biological criteria (Ohio EPA 1992b).

4.5 Variability and Resolution of Habitat Indices

Data from habitat methods training sessions and special studies of data variation should be used to define the variability inherent in a habitat index. The degree of variability in an index will define the appropriate uses and limitations of an index and determine when more-detailed work is required. One way to examine between-user variability is to set up a set of test sites that users of the methods independently score for a given habitat technique. Data from these sites can be examined with quality control techniques and, potentially, users could be certified in the use of the tool.

Ideally, an index should minimize measurement error while maximizing the ability to distinguish important variation in habitat quality. For broad-based monitoring programs, cost-effectiveness must also be considered. An index that is cost-effective but does not provide the resolution needed to support decisions for an agency is useless, as is an index that provides resolution but is cost-prohibitive for general use.

Fortunately, indices such as the QHEI have sufficient resolution to support their desired uses under most circumstances, and are inexpensive to implement. Ohio uses the QHEI to explain changes in biological communities (as measured by the IBI); however, certain objectives may require more resource-intensive investigations, and it is important to recognize when an index like the QHEI is inappropriate. For example, in Idaho, salmonid spawning is a protected beneficial use for certain streams (Burton et al. 1991). The Idaho Department of Health and Welfare has developed protocols for assessment of dissolved oxygen and fine sediment in salmonid redds that affect salmonid embryo survival. Such a methodology is (1) sensitive to the types of impacts important in those streams, and (2) relates to a beneficial use of sufficient value to make a more intensive assessment justified.

Variation in the QHEI in Ohio is sufficiently low to make it useful for the objectives of our agency, which include: designating and protecting aquatic life uses, discerning causes and sources of impact in intensive surveys in watersheds, issuing PTIs for sewer lines and other construction, and supporting specific program activities such as 401/404 water quality certifications (i.e., certifying that dredge or fill operations in streams will not violate a state's water quality standards). Data from our training sessions, where QHEIs were generated independently by field staff and other trainees, found a strong, significant correlation between individual scoring and scoring of the instructor (Figure 7). Further confidence in decision making is provided by sampling multiple stations and incorporating other sources of data, especially fish community data. Both the habitat and biological data, including the biocriteria in our water quality standards, have been successfully court tested (*Northeast Ohio Regional Sewer District v. Shank* 1991).

4.6 Regional/Ecoregional Differences in Habitat

Habitat impacts that are likely to affect aquatic ecosystems around the country are often related to: (1) the geomorphology of a region and its effect on habitat diversity, and (2) land use activities typical to a region or ecoregion. For example, in Ohio agriculture is likely the most widespread activity that affects stream habitat and aquatic life (47% of Ohio's land surface is engaged in crop production, Hoorman et al. 1992). Hoorman et al. (1992) estimate that up to 55% of the agricultural area of Ohio needs "drainage improvements" to permit agricultural production. The effects and the need for such drainage vary regionally with relief and soil type. The Huron–Erie Lake Plain (HELP) ecoregion in Ohio, for example, suffers from both the need and effects of such drainage.

Figure 8 illustrates some of the differences between habitat characteristics related to ecoregions in Ohio streams. The most obvious difference between the HELP ecoregion and the other ecoregions is the overall low gradient of the HELP streams (Figure 8). Low-gradient streams are perhaps the most susceptible to sediment degradation and habitat destruction of all Ohio streams because the retention time of sediment is high and the stream energy, critical in developing and maintaining habitat diversity, is low. For all the habitat metrics shown here, the HELP had the most consistently poor habitat quality. For most of the other habitat parameters, the metric scores did not differ substantially with the exception of the interior plateau, which had more high-gradient streams (high relief and fewer large river sites) and more low-riffle scores. This small section of Ohio has many high-quality streams with good pools but riffles that may become intermittent during parts of the summer. Exploratory charting such as this is very useful for detecting regional patterns in data, especially when stratified by factors such as stream size, gradient, flow types, etc.

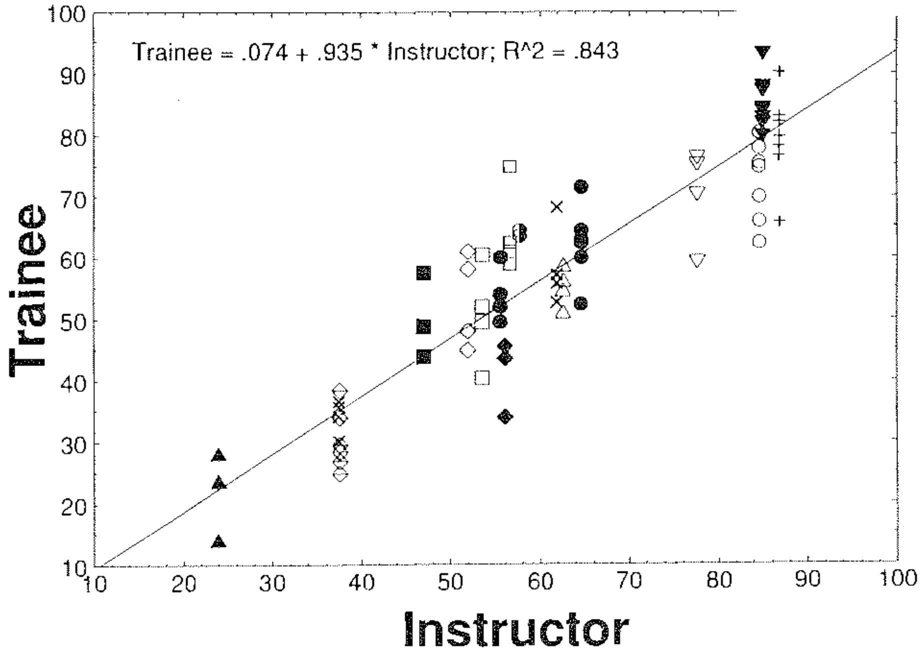

Figure 7. OHEI scores of individuals taking a QHEI training course in relation to QHEI scores generated by the instructor at small streams in Ohio. Data from five training sessions, one per area of Ohio (NW, NE, SE, SW, central). Common point symbols for a given instructor score indicate the same site.

4.7 Effect of Scale on Habitat Disturbance

As discussed earlier, habitat conditions are dependent on local geomorphology and anthropogenic influences on ecosystems. Much of the types of habitat assessments done in typical monitoring programs tend to focus on small scales of impact, usually at the microhabitat or, at most, the level of a several-hundred-meter reach. Assessments may focus on comparing a "site" (reach) to some reference "site" or sites. The "potential" of a study site to support aquatic life is then based on how close in quality this site is to a reference condition (Plafkin et al. 1989; Mangum 1990; Barbour and Stribling 1991). Such an approach, while useful in many cases, may not be sensitive to the effects of large-scale disturbances on stream ecosystems. In essence, impacts may not be totally predictable on the basis of site-specific habitat assessments alone because larger-scale disturbances are affecting the biota.

Frissel et al. (1986) and Gregory et al. (1991) advance a hierarchial conceptual framework of classifying stream habitat that incorporates various temporal (days to hundreds of years) and spatial (particle to stream network or subbasin) scales. The effects of anthropogenic changes on habitat should be considered at each of these scales (Schlosser 1991) when monitoring lotic systems. Such arguments are supported by Ohio EPA's monitoring data and in the theoretical ecological literature dealing with local extinctions and sources and sinks of individuals in ecosystems (Pulliam 1988).

Areas of Ohio that have had severe, large-scale landscape changes (HELP ecoregion) often have lower biological integrity and fewer species even in remaining areas of relatively good habitat. Presumably, local extinctions (e.g., 44% of original species in Maumee River drainage have declined or been extirpated; Karr et al. 1985) result from large, expansive areas of poor and modified habitat that act as "sinks" for production. In contrast, areas of Ohio with relatively intact landscapes and stream habitat often have high biological integrity. Here, short stretches of relatively poor habitat can often have much higher species richness and biological integrity than "predicted" from site-specific habitat assessments.

The effect of habitat disturbances at large scales can be seen in Ohio EPA fish community and QHEI databases. Subbasin-wide estimates of habitat quality in an area were estimated from average QHEI scores from each of 93 subbasins (all sites in our database; N = 2462) and plotted vs. average IBI scores

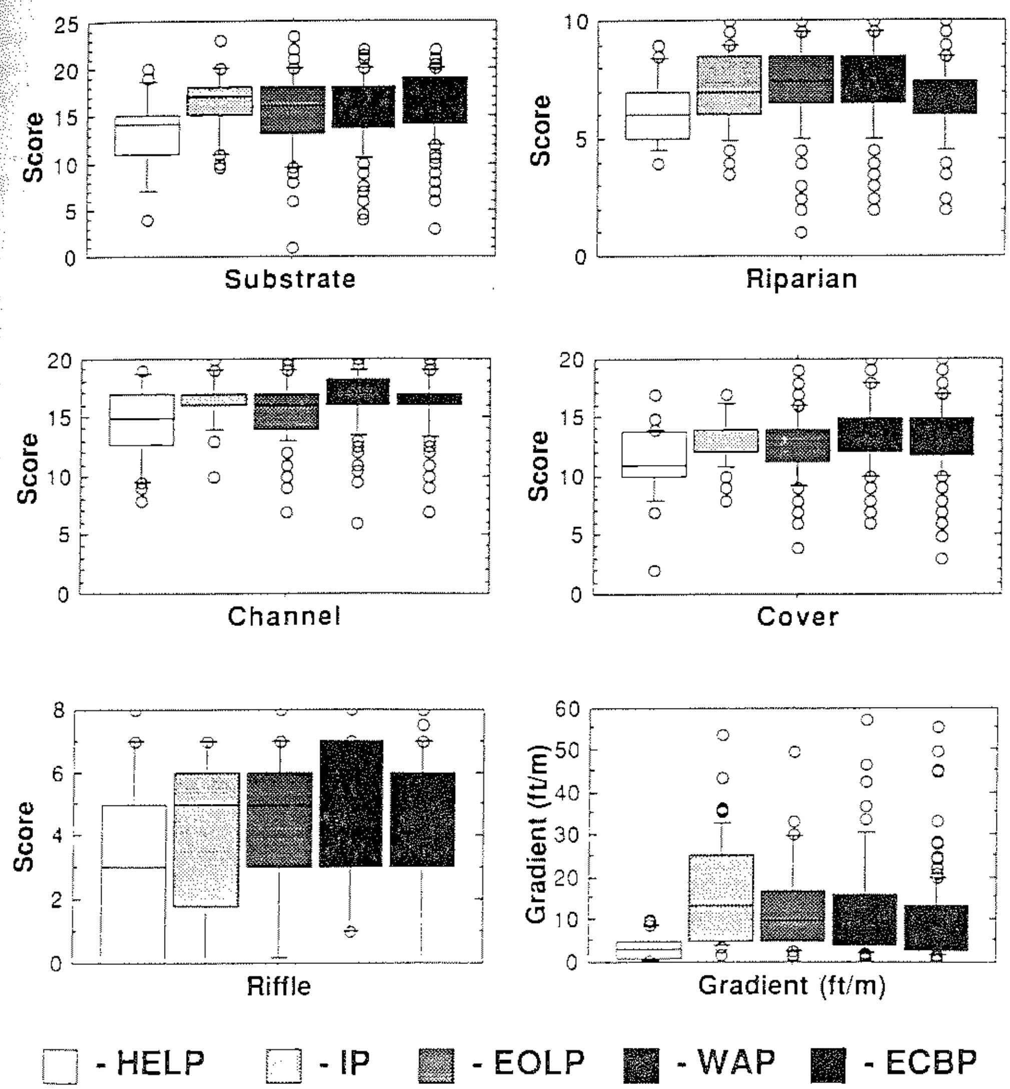

Figure 8. Box-and-whisker plots, by ecoregion, of QHEI metric scores for substrate score, riparian score, channel score, cover score, riffle score, and local gradient (ft/mi) for unmodified reference sites.

from our reference sites in these subbasins. There is a significant positive relationship between subbasin-wide estimates of habitat quality and reference site biological integrity (Figure 9). This pattern can also be seen when data from two individual subbasins (Little Auglaize River: habitat devastated; Twin Creek: habitat relatively intact) are examined (Figure 10). Although QHEI scores overlap (the highest QHEI scores in the Little Auglaize and the lowest in Twin Creek) the resulting IBI scores for a given QHEI differ substantially. Some evidence has suggested that high-quality oases of habitat could harbor sensitive species in areas that have been impacted by agriculture, urbanization, etc. (Luey and Adelman 1980). The Little Auglaize River in Ohio no longer has any oases of sensitive species. It is unknown how large such an oasis would need to be to remain viable and not subject to extirpations during "bottlenecks" of environmental stress (e.g., drought) in modified ecosystems. In most of Northwestern Ohio, headwater streams have been severely modified. Since headwater streams export problems downstream (e.g., sediment, flow, and tolerant species), species in a downstream "oasis" are likely to decline or be extirpated.

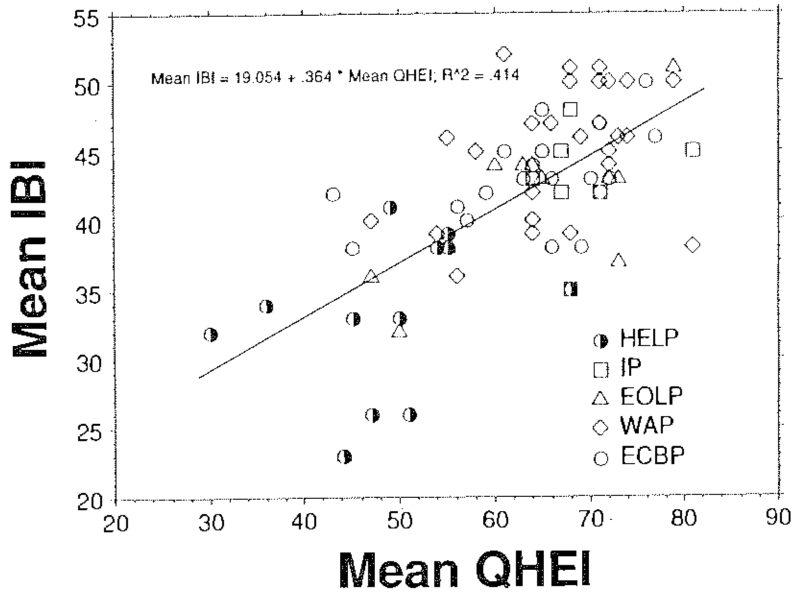

Figure 9. Average habitat quality in Ohio subbasins estimated by QHEI scores (all data from database) and average IBI scores at unmodified reference sites in Ohio subbasins. Ohio is divided into 93 subbasins.

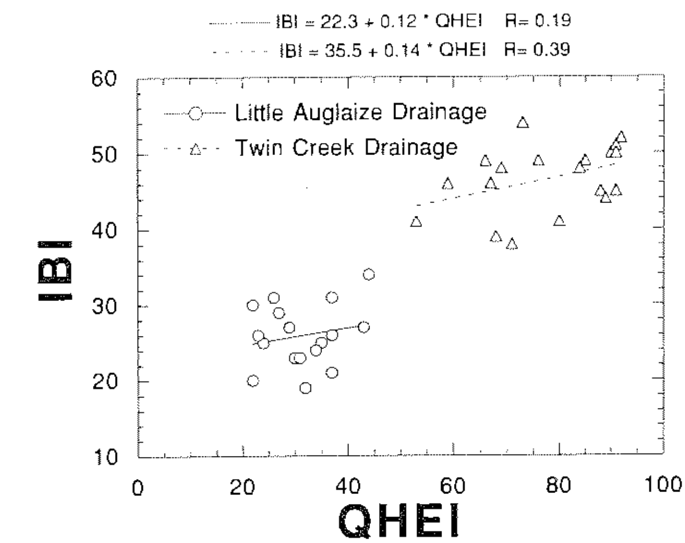

Figure 10. IBI in relation to the QHEI at two subbasins in Ohio, the Little Auglaize River in northwestern Ohio and Twin Creek in southwestern Ohio. The Little Auglaize River basin is a habitat poor, highly modified subbasin while Twin Creek has higher quality habitat with much less stream habitat disturbance.

The varying affects of different scales of habitat impacts on aquatic life and the importance of considering streams as open systems has important consequences for regulatory agencies charged with protecting streams. Too narrow a focus on specific sources of impacts (e.g., point sources) to the exclusion of other important factors (habitat and nonpoint) leads to the underprotection of streams. Similarly, there is often a focus on a study "site" and its impacts and aquatic potential rather than on a study reach or some larger scale. In day-to-day activities of a regulatory agency it is important to point out to clients (e.g., dischargers) that short stretches of modified stream do not preclude application of stringent water quality rules. Similarly, regulatory agencies need to protect against piecemeal degradation of stream habitats that would eventually result in large-scale devastation to aquatic life.

5.0 APPLICATION OF HABITAT INDICES IN WATER RESOURCE QUALITY ASSESSMENTS

In Ohio, habitat assessments are an integral part of our intensive survey program (Yoder 1991). Important uses of habitat assessment information include aquatic life use designations and as a tool in our intensive watershed surveys. The following sections provide examples of these uses in Ohio.

5.1 Habitat Indices in Stream and Basin-Intensive Surveys in Ohio

Habitat assessments (QHEIs) are done at all stream sites by the same field crew and during the same time period in which fish community data are collected. Besides its function in designating the proper aquatic life use, the QHEI assessment is used to explain causes and sources of impacts to the aquatic life. Although the final QHEI score is useful in interpreting habitat effects, we rely heavily on the component habitat characteristics to explain community impacts. Data from the QHEI and the fish communities at our reference sites and physically modified reference sites were used to derive habitat attributes that are characteristic of least-impacted or physically modified streams (Rankin 1989). Chi-square statistics were used to classify those attributes most often associated with low IBI or high IBI values (Rankin 1989). As the number of modified habitat attributes increase, the likelihood of having IBI scores similar to reference conditions decreases (Figure 11). These patterns of community response from our reference sites and the personal experience of our biologists are the basis of our interpretation of the patterns observed in intensive survey data. Patterns in biological response between the biota, water column chemistry, sediment chemistry, effluent characteristics, and land use patterns are all combined with the basic habitat condition to isolate the factors likely responsible for aquatic life impairment. Yoder (1994) provides some explanation on how biological responses ("signatures") can help in the interpretation of complex environmental data.

The following example will illustrate the use of the QHEI in intensive surveys. Data presented below have been summarized in an Ohio EPA biological water quality report (Ohio EPA 1991).

5.1.1 Hocking River

The Hocking River, located in southeastern Ohio, is a medium-sized river (1,197 mi^2 drainage area) of about 100 mi in length. The Hocking River headwaters are in glacial deposits of Fairfield County southeast of Columbus, Ohio, and it flows southeasterly through unglaciated, rugged topography to the Ohio River (Ohio EPA 1991). Since early in this century the river has been affected by industrial discharges and municipal and combined sewer discharges, especially in its headwaters near Lancaster, Ohio, and by acid mine drainage, nonacid mine effects (sediment), agricultural polluted runoff, severe bank erosion and sedimentation, and channelization (Shurrager 1932; Ohio EPA 1991). Shurrager (1932) surveyed the fish communities of the Hocking River in 1931-32 and the Ohio EPA surveyed the upper section of the river (RMs 73 to 95) in 1982 and the entire river in 1990 (Ohio EPA 1991).

In 1982, the Lancaster WWTP and industrial discharges to the WWTP were responsible for severe impacts to fish and macroinvertebrate assemblages throughout most of the study area (Figure 12). As a result of the 1982 survey, the City of Lancaster upgraded their WWTP in the late 1980s and an effective pretreatment plan was initiated. Improved treatment led to the elimination of exceedances of Ohio Water Quality criteria for ammonia-N, total zinc, and dissolved oxygen (Ohio EPA 1991). Macroinvertebrate data reflected the vastly improved water quality, recuperating from minimum Invertebrate Community Index values (ICI) near the WWTP during 1982 to exceptional levels of biotic condition in 1990 (Figure 12). Ohio EPA uses Hester-Dendy artificial substrates to collect macroinvertebrates (Ohio EPA 1988; DeShon, Chapter 15). Macroinvertebrate communities collected this way respond largely to changes in water quality constituents and do not usually reflect macrohabitat impacts to streams, especially if

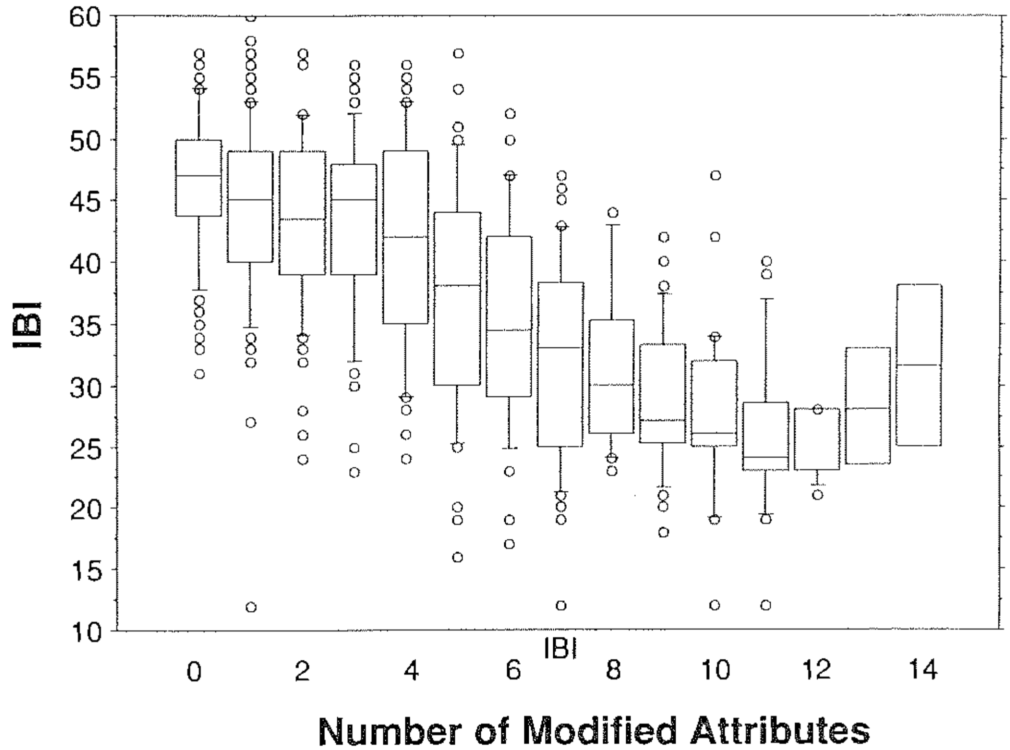

Figure 11. Boxplots of the IBI in relation to the number of modified warmwater habitat attributes at natural and modified reference sites in Ohio.

macrohabitat impacts are localized. Fish communities, however, respond strongly to both water quality and habitat disturbances. Like the macroinvertebrates, the fish assemblages were severely impaired by water quality impacts in 1982. In the 1990 survey the extensive habitat impacts were "unmasked" by improving water quality and are now largely limiting the fish community in the Hocking River. The results of the survey of the entire river in 1990 indicated that throughout the length of Hocking River habitat impacts, including channelization, riparian removal and bank erosion, mine-related sedimentation, and severe substrate embeddedness, significantly impact fish assemblages. Table 3 summarizes for the Hocking River important habitat attributes that commonly affect fish assemblages; reference sites from the Hocking River are included for comparison. This table format is a useful tool for examining the cumulative affects of habitat destruction on streams.

The inclusion of biosurvey and habitat data in the assessment of the Hocking River was indispensable for determining impacts and setting priorities for future stream improvements (e.g., restoring riparian forests and reducing bank erosion). If habitat data were not collected or if only a single organism group was surveyed, the relative magnitude of the various impacts would likely have been distorted. Our survey experiences in Ohio have often shown us that our presurvey, study-plan-derived perceptions of the impacts in a basin may be false. The consequences of not using integrated, intensive monitoring approaches in water management programs are inaccurate diagnoses of the causes and sources of impairments and water resources left underprotected. Data compiled nationally (see Figure 2) strongly suggest habitat is overlooked as a major limiting factor to biological integrity.

5.2 Habitat Assessment Techniques and Use Designations

In Ohio, instream biocriteria are the arbiters of aquatic life use designations; however, habitat assessment data plays an integral role. The achievement of the ecoregional biocriteria for a stream assures it of *at least* the associated designated use, regardless of the score of the QHEI. However, in many

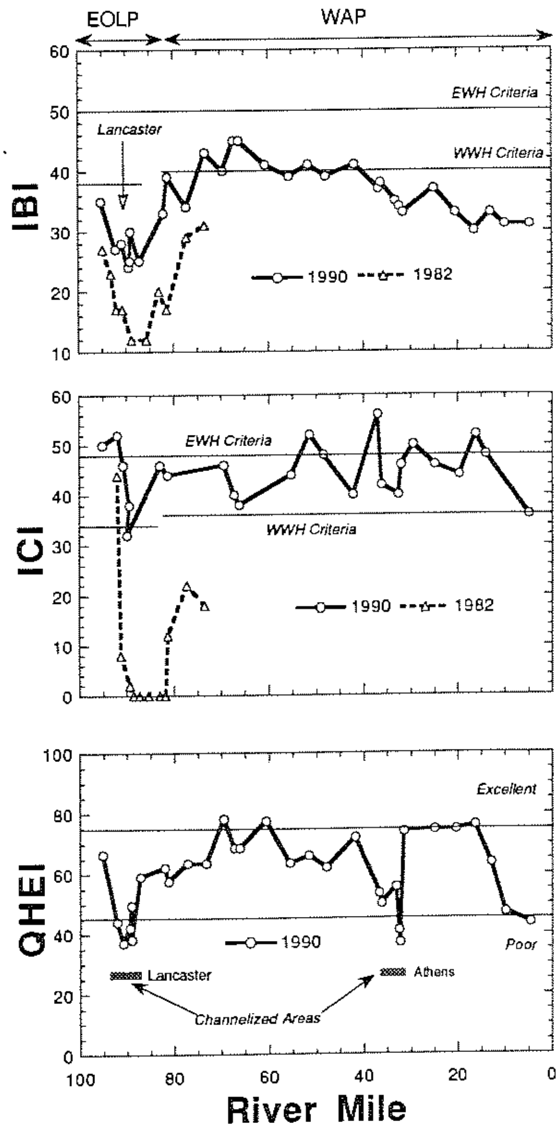

Figure 12. IBI, ICI, and QHEI in relation to River Mile in the Hocking River near Lancaster, Ohio in 1982 and 1990.

situations streams being designated or redesignated have water quality impacts that preclude the use of the biota alone to define the aquatic life potential of a stream. In these situations we rely heavily on the habitat assessment information.

Integral to Ohio's biocriteria is a "tiered" system of aquatic life uses which define the *baseline* expectations for the aquatic life of a stream. Yoder and Rankin (Chapter x) defines these aquatic life uses in more detail. About 10% of Ohio's streams are designated as Exceptional Warmwater Habitat (EWH), which encompasses most of the streams that harbor endangered or threatened species and the most diverse assemblages. The majority of streams are designated as Warmwater Habitat (WWH) and contain balanced, reproducing assemblages of fish and macroinvertebrate communities. Ohio also has two "non-fishable" aquatic life uses: Modified Warmwater Habitat (MWH) and Limited Resource Water (LRW). Limited Resource Waters are those extremely small (<3 mi^2), ephemeral, and highly modified (often urban) streams that are not likely to support any semblance of a natural community. If any organisms are

Table 3. Warmwater and Modified Warmwater Habitat Attributes at Sites in Streams in the Hocking River Basin

															MWH attributes																						
				WWH attributes											High influence							Moderate influence															
River mile	QHEI	Gradient (ft/mi)	Ref. site	No channelization or recovered	Boulder/cobble/gravel substrates	Silt-free substrates	Good/excellent development	Mod/high sinuosity	Extensive/moderate cover	Fast current/eddies	Low/normal embeddedness	Max depth >40 cm	Low/no riffle embeddedness	Total WWH attributes	Channelized or no recovery	Silt/muck substrates	Low Sinuosity	Sparse/no cover	Max depth <40 cm (WD, HW)	Total (high influence)	MWH attributes	Recovering channel	Heavy/mod. silt cover	Sand substrates (BT)	Hardpan origin	Fair/poor development	Low/no sinuosity	Only 1-2 cover types	Intermittent and poor pools	No fast current	High/mod. embeddedness	Ext./mod. riffle embeddedness	No riffle	Total (moderate influence)	MWH attributes	MWH (High)/WWH attributes	MWH (mod.)/WWH attributes
(01-001)–Hocking River																																					
Year: 90																																					
95.2	66.0	13.1		■	■		■	■	■	■		■		7							0		▲								▲	▲	▲		4	0.13	0.63
92.2	44.0	4.7	M							■		■		2	●			●			2		▲			▲	▲				▲	▲			5	1.00	2.67
90.8	37.0	4.7			■					■		■		3	●			●			2		▲			▲	▲				▲	▲			5	0.75	2.00
89.4	41.5	4.7			■					■		■		3	●			●			2		▲			▲	▲				▲	▲			5	0.75	2.00
89.1	49.0	4.7		■	■				■			■		4							0		▲			▲	▲			▲	▲		▲		6	0.20	1.40
89.0	37.5	4.6			■					■		■		3	●			●			2		▲			▲	▲				▲	▲			5	0.75	2.00
87.1	58.5	4.6		■			■	■	■	■		■		6							0		▲								▲	▲			3	0.14	0.57
82.0	62.0	6.6		■	■		■	■	■	■		■		7							0		▲								▲	▲			3	0.13	0.50
81.2	57.5	4.6		■					■	■		■		4							0		▲	▲	▲	▲	▲	▲			▲	▲			8	0.20	1.80
77.2	63.5	3.1		■			■	■	■	■		■		6							0		▲	▲	▲						▲	▲			5	0.14	0.86
73.2	63.5	3.1		■			■	■	■	■		■		6							0		▲	▲				▲			▲	▲			5	0.14	0.86
69.5	77.5	3.7		■	■		■	■	■	■		■		7							0										▲	▲			2	0.13	0.38
67.3	68.0	3.4			■				■	■		■		4							0	▲	▲			▲	▲				▲	▲			6	0.20	1.40
66.2	68.0	3.4		■	■		■	■	■	■		■		7							0		▲	▲		▲					▲	▲			5	0.13	0.75
60.6	77.5	2.3		■	■		■	■	■	■		■		7							0		▲	▲						▲	▲	▲			5	0.13	0.75
55.7	63.5	6.4		■	■			■	■	■		■		6							0		▲			▲		▲			▲	▲			5	0.14	0.86
51.6	65.5	1.9		■	■			■	■	■	■	■		7							0		▲	▲		▲		▲				▲			5	0.13	0.75
47.9	62.0	1.9		■			■	■	■			■		5				●			1		▲	▲		▲				▲	▲		▲		6	0.33	1.33
41.8	72.0	1.9		■	■		■	■	■			■		6							0		▲	▲						▲	▲	▲			5	0.14	0.86
36.7	53.5	1.9		■	■			■	■			■		5		●		●			2		▲		▲	▲	▲			▲	▲		▲		7	0.50	1.67
36.2	50.0	1.9			■				■			■		3							0	▲	▲	▲		▲	▲			▲	▲	▲			8	0.25	2.25
33.1	55.5	1.6	M		■				■			■		3							0	▲	▲	▲		▲	▲			▲	▲	▲			8	0.25	2.25

32.5	41.0	1.6			■							■		2	●			●		2		▲	▲		▲	▲	▲	▲	▲	▲		8	1.00	3.67
32.3	37.0	1.6										■		1	●			●		2		▲	▲		▲	▲	▲	▲	▲	▲	▲	9	1.50	6.00
31.5	74.0	1.6		■	■		■	■	■	■		■		7						0		▲	▲					▲	▲	▲		5	0.13	0.75
25.0	74.0	1.6		■	■		■	■	■			■	■	7						0		▲	▲					▲	▲			4	0.13	0.63
20.4	74.0	2.2		■	■		■	■	■	■		■	■	8				●		1		▲	▲						▲			3	0.22	0.56
16.3	75.5	2.2		■	■		■	■	■	■		■	■	8						0		▲	▲						▲			3	0.11	0.44
12.9	63.0	0.1		■	■		■	■	■			■		6						0		▲	▲		▲			▲	▲		▲	6	0.14	1.00
9.8	47.0	0.1		■					■			■		3		●		●		2		▲	▲		▲	▲		▲	▲		▲	7	0.75	2.50
4.6	43.0	0.1		■					■			■		3		●		●		2		▲	▲		▲	▲		▲	▲		▲	6	0.75	2.25
(01-037)–Scotts Creek																																		
Year: 78																																		
8.9	76.0	14.2	R	■	■	■	■	■	■		■	■		8						0								▲			▲	2	0.11	0.33
8.1	69.5	14.2	R	■	■		■	■	■		■	■		7						0								▲			▲	2	0.13	0.38
(01-100)–Federal Creek																																		
Year: 90																																		
11.4	81.5	8.3	R	■	■		■	■	■	■	■	■	■	9						0												0	0.10	0.10
Year: 84																																		
1.3	74.0	2.3	R	■	■		■	■	■	■	■	■	■	9						0		▲										1	0.10	0.20
Year: 90																																		
1.3	75.5	2.3	R	■	■		■		■	■	■	■	■	8						0						▲						1	0.11	0.22
(01-400)–Clear Creek																																		
Year: 82																																		
16.3	55.0	6.1	M					■	■			■		3			●			1	▲	▲			▲	▲		▲	▲	▲		7	0.50	2.25
14.2	48.0	8.7	M		■			■	■					3					●	1	▲	▲			▲	▲		▲	▲	▲		7	0.50	2.25
7.3	89.5	9.0	R	■	■		■	■	■	■	■	■	■	9						0												0	0.10	0.10
Year: 90																																		
2.0	77.5	4.1	R	■	■		■	■	■	■	■	■	■	9						0		▲										1	0.10	0.20
(01-420)–Muddy Prairie Run																																		
Year: 82																																		
0.7	85.0	15.3	R	■	■	■	■	■	■	■	■	■	■	10						0												0	0.09	0.09
(01-430)–Dunkle Run																																		
Year: 82																																		
0.1	69.0	7.0	R	■	■		■	■	■			■		6				●		1		▲			▲			▲	▲	▲		5	0.29	1.00
(01-510)–Durbin Run																																		
Year: 82																																		
0.4	31.0	1.7	M								■			1	●	●	●	●	●	5		▲		▲	▲	▲		▲			▲	6	3.00	6.00
(01-520)–Turkey Run																																		
Year: 82																																		
1.4	67.0	10.8	R	■	■		■	■	■		■	■	■	8						0				▲				▲				2	0.11	0.33

Reference site codes: R — unmodified reference site; M — reference site with modified physical habitat.

present they are generally pioneering species that are exceptionally tolerant to poor water quality, extremely modified habitat, and intermittent stream flow. The MWH use is a relatively new use that is designed to protect those streams that will not attain the WWH use because of extensive habitat modifications, but that have a significant permanent assemblage of tolerant organisms that would not be adequately protected by the LRW use. Presently, MWH streams have the same water quality criteria as WWH streams except for dissolved oxygen and ammonia-N, of which the typical assemblages in MWH stream are tolerant.

By examining the preponderance of various modified habitat attributes and unmodified habitat attributes at multiple sites in a stream we can determine the likelihood that a stream will or will not be able to achieve a particular aquatic life use (see Figure 11). The flowchart in Figure 13 summarizes and simplifies an example of assigning an aquatic life use to a stream that is a candidate for either a MWH aquatic life use or a WWH aquatic life use. The first fork in the flowchart diverges on whether biosurvey data are available (Figure 13). If only habitat data are available, aquatic life use designations will only be made in very simple and obvious situations, such as small HELP ecoregion streams. The HELP ecoregion of Ohio has been so extensively ditched and drained that many of the small streams in this region are incapable of supporting a WWH use. In most other cases, however, aquatic life use decisions are made with biosurvey and habitat data.

For the present example, illustrated in Figure 13, if biosurvey data are available we will examine them for attainment of the appropriate WWH ecoregion biocriteria (Ohio EPA 1988). If the stream achieves the criteria it will assigned the WWH use regardless of habitat scores. If the WWH use is not achieved then it is considered for the MWH use only if there have been extensive physical alterations. With no physical disturbances the stream is assigned an WWH use or, in some circumstances (e.g., a wetland stream), becomes a candidate for site-specific biocriteria modification.

A stream with extensive habitat modifications becomes a candidate for the MWH if the modifications are substantial enough to preclude a WWH use and the likelihood of recovery of habitat conditions to support a higher use is low. The habitat attributes considered are listed in Figure 13. It is important to note that numerous sites along a stream are examined before making a use decision. We will, under some situations, assign different uses to different segments of a river where the potential of a stream obviously differs between segments (e.g., WWH segment in rural area, MWH within urban area). The situation in Figure 13 largely applies to MWH streams related to channel activities, but we also have MWH criteria that apply to nonacid mine-affected streams and impounded streams.

A large and robust set of reference sites (both least impacted and physically modified) is indispensable when designating aquatic life uses. By linking our aquatic life uses directly to our biocriteria we have made a direct connection between uses and the methods for assessing use attainment. Our large data set gives us a knowledge base for interpreting the potential of streams where the proper use may not be obvious.

The importance of tiered aquatic life uses to appropriate designation of those uses cannot be overstated. In Ohio, the combination of stream size stratification (headwater, wading, and boat types), ecoregions, and tiered aquatic life uses provides a flexible and workable approach for defining aquatic life potential. The incorporation of the MWH use has provided a needed intermediate use between the WWH use and a RW. This additional "tier" provides increased protection to assemblages that may have otherwise been designated as LRW. As illustrated in the flowchart (Figure 13) ecoregions can play a significant part in determining which habitat influences are most important to aquatic life. The HELP ecoregion is unique in Ohio because its low gradient and high water table led to extensive drainage that allows clays, silts, and other sediments to accumulate instream.

QHEI scores alone do not always reflect the potential of a stream. Streams can have a single attribute limiting the biota yet have relatively high QHEI scores. The coal-bearing WAP ecoregion, for example, can be devastated by extreme sediment plumes from unreclaimed surface mining while still retaining relatively intact channel and riparian conditions and relatively high QHEI scores. In other ecoregions larger streams can have intact channel and riparian characteristics, but headwater streams with modified channels can deliver sediment to the mainstems at extremely high rates, which can accumulate in pools. In Figure 5 these are generally the sites that are furthest below the regression lines.

Figure 14 illustrates IBI and QHEI data for four streams in Ohio, one that has an EWH designation, one with a MWH designation, and two with WWH designations, one in the HELP and another in the ECBP ecoregion. The similarity between the biological and habitat scores for each of these streams, with

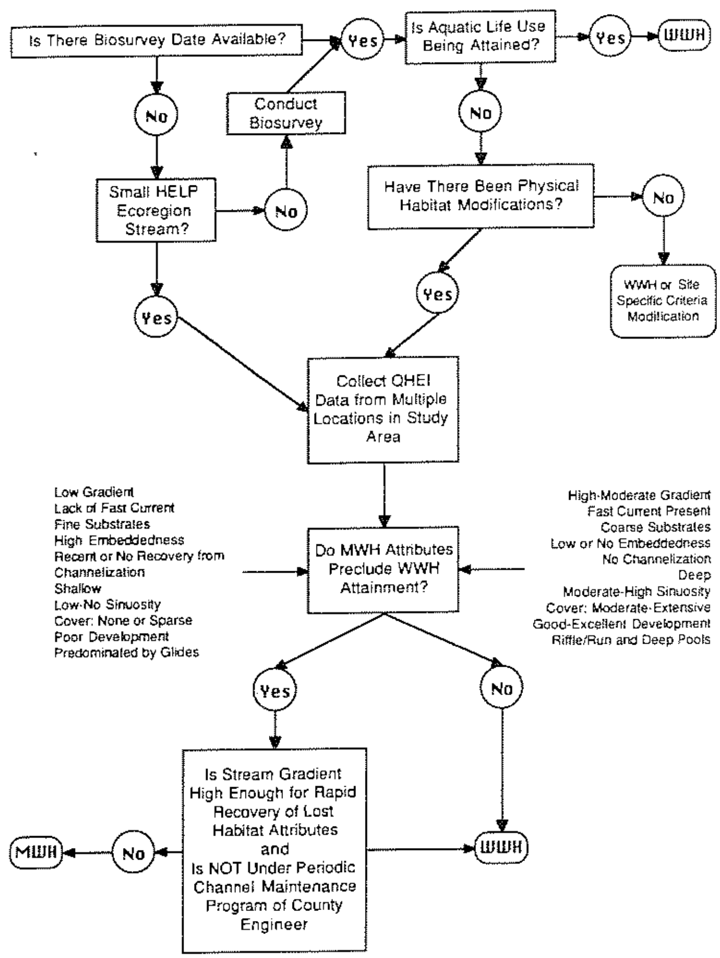

Figure 13. Flowchart summarizing the decision criteria used when assigning an aquatic life use to a stream that is a candidate for a Warmwater Habitat use (WWH) or Modified Warmwater Habitat use (MWH).

the exception of the IBI in Flatrock Creek (the WWH stream in the HELP ecoregion), is obvious. In Ohio the HELP ecoregion has much lower "expectations" for biological performance than other ecoregions in the state (Ohio EPA 1988b). Because this region of the state has been so extensively modified by stream drainage and channelization activities, reference sites meeting the "least impacted" definition for other regions of the state could not be found. Instead, the reference benchmark here was "best attainable" and derived from the 90th percentile of all sites in the region. Flatrock Creek, even though relatively unmodified and with a riparian zone comparable to other areas of the state, is severely affected by sediment and flow originating from headwater streams almost totally stripped of riparian vegetation, channelized, and drained. The Little Auglaize River, a MWH stream also in the HELP ecoregion, has had its mainstem and its headwater tributaries severely modified so that even the reduced expectations of the HELP WWH aquatic life use are unattainable.

The Kokosing River in the EOLP and WAP ecoregions is an obvious contrast to the Little Auglaize and Flatrock Creek drainages. Both the Kokosing River and its headwater streams are relatively intact and have sufficient gradient to continually flush excess sediment originating from agricultural land use practices. The proportion of land use in forest is much higher than in the HELP ecoregion. The increased relief also reduces the likelihood of riparian encroachment and the need for extensive drainage work.

Fourmile Creek is a WWH stream in the ECBP ecoregion in southwestern Ohio. The land use in this area is primarily row-crop agriculture; however, the relief is greater than in northwestern Ohio, especially in the lower part of the drainage. In contrast to Flatrock Creek, Fourmile Creek has had less of its

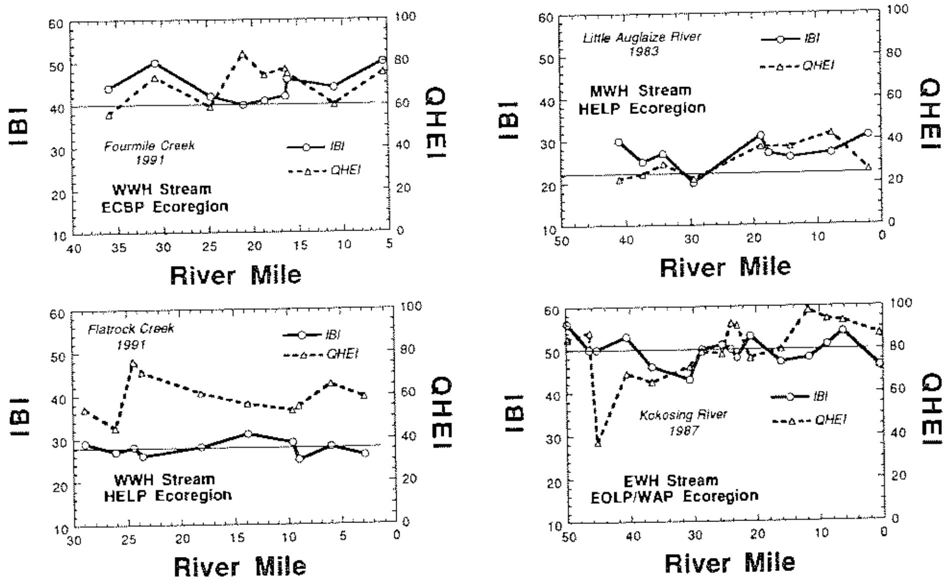

Figure 14. Longitudinal patterns in IBI and QHEI scores for a MWH stream (Little Auglaize River), an EWH stream (Kokosing River), and two WWH streams, one in the HELP (Flatrock Creek) and the other (Fourmile Creek) in the ECBP ecoregion.

headwaters modified by drainage but has been impacted from a WWTP. Riparian encroachment and extensive agriculture in the upper part of this drainage and a WWTP and an impoundment in the middle part of this stream may be masking the potential for this stream to reach the EWH aquatic life use. Improvements in the WWTP operation and recent plans to install nonpoint Best Management Practices (BMPs) in the upper basin may make this stream a candidate for a reassessment of its use when it is next assessed in Ohio's five-year basin monitoring approach.

5.3 Habitat Data in 401 Water Quality Certifications

The 401 water quality certification program is an important user of stream habitat data. The 401 program provides the states the ability to comment on and deny certain stream activities that deal with the fill of material into the nations waterways. States can deny or ask for modification of projects that could impair the state's water quality standards, including biocriteria. Thus, inclusion of biocriteria into water quality standards provides a direct link between habitat and water quality standards. In Ohio, numerous projects have been denied or modified because the activities would have degraded those habitat attributes upon which the biota is dependent. The process in these cases often follows those outlined for the intensive surveys. Some cases are complex and require biosurvey data and habitat data to arrive at a decision for denial or modification, while others are simple and can be denied or approved based upon what we have learned from our reference sites and the types of modifications that are planned.

Without biocriteria and without the collection of biosurvey and habitat data, there is a increased likelihood that destructive activities could occur because: (1) they do not usually violate chemically based water quality standards, or (2) biosurvey and habitat data are rarely integrated into regulatory decision making. Even with biocriteria, streams can still be degraded where the activities are covered by the nationwide 401 permit (e.g., in Ohio modifications on less than 1000 ft of stream) or are allowed by other, often conflicting regulations (e.g., state drainage laws). These concerns are similar to those of Schaeffer and Brown (1992) who reported that the "plethora of regulations" has not been successful in stopping the destruction of riparian habitats. They attributed this lack of success to federal statutes that protect water quality and quantity rather than wildlife habitat or biotic integrity. Federal rules and regulations (e.g., the Clean Water Act) need to promote ecosystem integrity rather than narrowly focusing on water quality in order to provide more practical, comprehensive, and cost-effective protection of water resources.

6.0 SUMMARY AND CONCLUSIONS

After Ohio instituted biosurvey and habitat assessment techniques in its water resource monitoring programs it became obvious that the state was not adequately protecting its streams and rivers and, as a result, biocriteria were incorporated into its water quality standards. In Ohio's 1988 305(b) report, its summary of the status and trends in the state's water quality for the U.S. Congress, the inclusion of biocriteria in addition to chemical criteria led to a significant increase in the numher of miles reported as impaired (Ohio EPA 1988). The identification of habitat and other nonchemical impacts to aquatic life was responsible for much of this change in the assessment. In subsequent reports (Ohio EPA 1990, 1992) substantial improvements in water quality related to improved wastewater treatment, have been documented; however, we have also documented more habitat disturbance and little or no habitat restoration. Existing regulations, such as Section 404 of the Clean Water Act (33 U.S.C. § 1344), do not provide a broad enough approach to adequately protect streams from habitat degradation (Schaeffer and Brown 1992).

The impacts we observe in Ohio are not limited to Ohio or even the Midwest. Recent work by Benke (1990), Karr (1991), and Allan (1993) and even articles in the *New York Times* (Stevens 1993) have reported on the widespread degradation to the nation riverine resources. The opening paragraphs from William Steven's article in the *New York Times* (Stevens 1993) summarizes the problem well:

> Two decades of Federal controls have sharply reduced the vast outflows of sewage and industrial chemicals into America's rivers and streams, yet the life they contain may be in deeper trouble than ever.
>
> The main threat now comes not from pollution but from humans' physical and ecological transformation of rivers and the land through which they flow. The result, scientists say, is that the nation's running waters are getting biologically poorer all the time and that entire riverine systems have become highly imperiled.

The recent report by the National Research Council (NRC 1992) also documented the impoverished condition of the nation's rivers and recommended that: (1) erosion control programs should be accelerated for soil conservation and environmental restoration, (2) grazing practices should be changed to minimize damage to river–riparian ecosystems, (3) erosion control, where feasible, should favor "soft" engineering over "hard" engineering (e.g., channelization) approaches, (4) nonfunctional or non-cost-effective dikes and levees should be breached to reestablish hydrological connections between riparian habitats and rivers, and (5) riparian areas should be classified, in land-use and wetland systems, on the basis of their connections to rivers. This committee also set a goal of restoring 400,000 mi of riparian–river ecosystems (12% of total U.S. rivers and streams) within the next 20 years (NRC 1992).

Given that the destruction of habitat is of major importance nationwide it seems essential that states have the tools to assess the extent and magnitude of these impacts and the tools to eliminate and reverse habitat destruction. Recent reviews of game fish habitat restoration efforts have resulted in recommendations of integrated management of land use, particularly in riparian areas (Lyons et al. 1988; Lyons and Courtney 1990).

All biological sampling protocols require some form of habitat assessment to permit accurate interpretation of results. Since the USEPA requires states to have narrative biocriteria in their state water quality standards many states will be instituting biosurvey and habitat assessment programs (USEPA 1990). In this chapter we argue that, to be effective, states must have a program of sufficient size to allow repeated sampling of natural and physically modified reference sites. These reference sites will be the basis for biological criteria and for developing habitat assessment techniques tailored to specific regions of the country. Regional efforts to define reference conditions and develop region-specific habitat indices should be organized by USEPA regions in concert with states and other institutions that have a stake in such collaborative efforts (e.g., USFWS, USGS, and local agencies and groups). Such groups need to coordinate stream policies that will ensure adequate habitat protection across state and political boundaries.

As states begin to amass habitat data, many of the techniques reviewed here should be refined. States should remain open to advances in habitat assessment techniques to ensure a reversal of the present slide in habitat and ecological integrity of the nations rivers. Some work has been done in using more advanced statistical techniques to analyze biological and habitat data including expert systems and machine learning techniques (Anderson et al. 1991). However, there has been little support for ecological and habitat research by USEPA *relative* to water chemistry/toxicological research. Unfortunately, national efforts for habitat protection and nonpoint source control have been meager and expenditures are still dominated by research and management priorities skewed towards "toxic" chemicals and point sources of pollution. Point sources are still serious problems in the United States, but USEPA-sponsored efforts to rank relative risks suggest more emphasis must be placed on protecting ecological systems and reversing habitat destruction. Until spending is increased or, more likely, spending priorities changed, biological and physical evidence suggests the quality of the nations rivers will continue to deteriorate (Benke 1990; NRC 1992).

With the sad state of federal support for habitat and ecological monitoring, why should states add another "fiscal" burden by spending essential resources on such monitoring? Simply put, the amount of money spent on monitoring is dwarfed by the amounts states require cities and industry to spend on treatment of effluents. Analyses performed in Ohio suggest that without biosurvey and habitat data there is a high risk of missing nonchemical and chemical impacts to streams (Ohio EPA 1990). There is a smaller but still significant risk of "finding" a water quality impact where one really does not exist when monitoring data are insufficient. This could result in a regulatory action that might cost hundreds of thousands of dollars or more to an entity, with costs passed along to consumers. Biosurvey and habitat data allow states to rank areas on the basis of need for remediation or protection. The interpretation and use of monitoring data in an ecoregional framework rather than in a political or hydrological framework can also lead to more accurate estimations of problems on state or national scales (Omernik and Griffith 1991).

The specter of millions of dollars being misspent on environmental controls without strong evidence of the efficacy of the treatment, indicates that money spent on high-quality monitoring programs is money well spent. Initiatives under discussion, such as pollution trading, will likely fail, or at least fail to control many of the factors limiting river integrity, if these initiatives are not based on accurate environmental information, including an effort to quantify habitat quality.

ACKNOWLEDGMENTS

This work would not have been possible without the staff of the Ecological Assessment Section of the Ohio EPA. The development of Ohio's biocriteria and habitat assessment techniques are the result of a team effort by many individuals. Drafts of this paper were reviewed by Chris Yoder, Marc Smith, Roger Thoma, Dave Altfater, Bernie Counts, and three anonymous reviewers.