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# Changes in Fish Assemblage Status in Ohio's Nonwadeable Rivers and Streams over Two Decades

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*Abstract.*—A systematic, standardized approach to monitor fish assemblages has been applied in Ohio's rivers since 1979. A primary objective is the assessment of changes in response to water pollution abatement and other water quality management programs. All major, nonwadeable rivers were intensively sampled using standardized electrofishing methods and a summer–early fall index period. Most rivers were sampled two or three times, before and after implementation of pollution controls at major point source discharges and best management practices for nonpoint sources. A modified and calibrated index of biotic integrity (IBI) was used to demonstrate and evaluate changes at multiple sampling locations in major river segments. An area of degradation value (ADV) and an area of attainment value (AAV) were also calculated from IBI results to demonstrate the magnitude and extent of changes in fish assemblage condition along segments and between sampling years. Positive responses in the IBI and the ADV/AAV were observed 4 to 5 years after implementing improved municipal wastewater treatment. Positive responses were much less apparent in rivers predominantly influenced by complex industrial sources, agricultural nonpoint sources, and extensive hydrologic modifications. The ADV/AAV showed incremental improvements in river fish assemblages, unlike pass/fail IBI thresholds, and tiered IBI biocriteria provided more appropriate benchmarks than chemical, physical, or qualitative biological criteria. The results show the value of standardized and intensive fish assemblage monitoring and the use of tools that reveal the extent and severity of impairments to determine the effectiveness of water pollution control programs.

## Introduction

The Ohio Environmental Protection Agency (EPA) initiated a comprehensive and standardized assessment of the fish assemblages in Ohio rivers in 1979. Its purpose was to provide information for establishing water quality standards (WQS), developing permit terms and conditions for pollutant dischargers, awarding grants for pollution control projects, and planning water quality projects (Yoder and Rankin 1995a; Yoder and Smith 1999). Fish assemblage assessments are one part of Ohio EPA biological and water quality surveys. These surveys are interdisciplinary monitoring efforts planned, coordinated, and conducted on specific water bodies and individual watersheds as part of a comprehensive statewide monitoring strategy. In main-stem rivers, these involve entire reaches, multiple and overlapping stressors, and tens of sampling sites.

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The aggregate database has supported research programs and projects, including the original work on ecoregions and regionalization (Hughes et al. 1986; Larsen et al. 1986; Omernik 1987), biological criteria (Larsen 1995; Yoder and Rankin 1995a, 1995b; Sanders et al. 1999; Thoma 1999; Barbour and Yoder 2000), water quality criteria (Ohio EPA 1999; Miltner and Rankin 1998), analysis of land use impacts (Yoder et al. 2000; Miltner et al. 2004), diagnosis of biological responses (Yoder and Rankin 1995b; Norton et al. 2000; Yoder and DeShon 2003), defining risk and application to management programs (Yoder 1998; Yoder and Rankin 1998; Cormier et al. 1999b; Barbour et al. 2000; NRC 2001; Karr and Yoder 2004; Erekson et al. 2005), and research to develop and validate physiological and genetic indicators (Silbiger et al. 1998; Cormier et al. 1999a).

Ohio EPA annually conducts biological surveys at 400–600 stream and river sampling sites. To date, fish assemblage sampling has included more than 8,000 sites in 1,750 rivers and streams since 1979. More than 100 reports, applied research papers, and technical publications have been developed by Ohio EPA and others (most reports are available at http://www.epa.state.oh.us/dsw/document_index/psdindx.html).

Standardized biological, chemical, and physical monitoring and assessment techniques are used to satisfy three baseline water quality management objectives: 1) determine the extent to which water body classifications assigned in the Ohio WQS are either attained or not attained; 2) determine if the classifications assigned to a given water body are appropriate and attainable; and 3) determine if any changes in ambient biological, chemical, or physical indicators have taken place over time, particularly before and after implementation of mandatory point source pollution controls or voluntary best management practices. Underlying all of the objectives is the identification of the causes and sources associated with impairments or threats identified by an integrated assessment (Yoder and DeShon 2003).

While there is not a single definition of a nonwadeable river, it functionally includes rivers that cannot be sampled effectively by wading techniques (Ohio EPA 1989a). The development of biological assessment tools for nonwadeable rivers, particularly those focused on assessments of condition and status, has lagged behind the development of wadeable methods in the United States. Biological assessments of great and large rivers have been conducted since the late 1940s, but few of these early efforts included fish. The routine assessment of fish assemblages is a comparatively recent addition and followed the development of effective electrofishing technologies. Single-gear electrofishing assessments include the pioneering work by Gammon (1973, 1976, 1980) and Gammon et al. (1981) in the Wabash River of Indiana. Other efforts followed and many were associated with studies of thermal effluents in response to Section 316[a] of the Clean Water Act (CWA) conducted mostly in the 1970s. Many of these studies lacked a conceptual framework for analyzing data and producing meaningful and consistent assessments. The development of the index of biotic integrity (IBI; Karr 1981; Fausch et al. 1984; Karr et al. 1986) provided a conceptual framework for the development of fish assemblage assessment approaches that can be applied to nonwadeable rivers. In this chapter, we report on changes in the fish assemblages of nonwadeable rivers in Ohio, before and after the implementation of pollution controls, using an IBI modified and calibrated for Ohio rivers. This was accomplished by assessing changes in the IBI and the area of degradation value (ADV; Yoder and Rankin 1995b) and an area of attainment value (AAV).

## Methods

All methods for capturing, identifying, and processing electrofishing samples follow procedures developed by Ohio EPA (Ohio EPA 1980, 1989a). The rationale for the development of these procedures is described in greater detail in Ohio EPA (1987a, 1987b, 1989a) and Yoder and Smith (1999).

### *Assessment Design*

The Ohio EPA rotating basin approach consists of surveys of specific river basins repeated at varying intervals depending on the need for information about spatial and temporal changes, but generally within 5–10 years. A spatially intensive design is employed to sample fish assemblages in a river reach to comprehensively assess major disturbances. The design requires multiple sampling sites in spatial proximity to suspected sources so that results can be analyzed and displayed in a longitudinal context. Sampling sites are established to represent all habitats including pools, runs, riffles, shoals, and backwaters as available. The sampling design and results interpretation relative to disturbance sources is based on the concepts originally described by Bartsch (1948) and Doudoroff and Warren (1951) to facilitate detection and quantification of varying pollution influences along a reach (i.e., pollution zones). Typically, we sample reaches upstream from major sources of disturbance, in areas of immediate impact and potentially acute effects, through zones of increasing and lessening degradation, and zones of recovery. We attempt to determine the role of specific sources as well as cumulative effects of multiple sources. Large rivers are treated as a single study unit to understand how changes take place along a longitudinal continuum with respect to both natural and anthropogenic influences. Important in the delineation of these study units are natural features and transitional boundaries (e.g., ecological and geological boundaries) and clusters of anthropogenic sources (e.g., major urban/industrial areas, impoundments, etc.). Some study reaches are up to 160 km long to capture all relevant influences, include zones of impact and recovery, and provide context for interpreting results within a localized reach or at a given location (e.g., Figure 1). This design yields a detailed assessment of status, the extent and severity of indicator responses in a particular river reach, and temporal changes (Figure 1). It produces assessments of the severity (departure from the desired state) and extent (lineal extent of the departures) of biological impairments in the various rivers.

When the assessment process described here was initiated in 1979, management programs were principally focused on major point source dis-

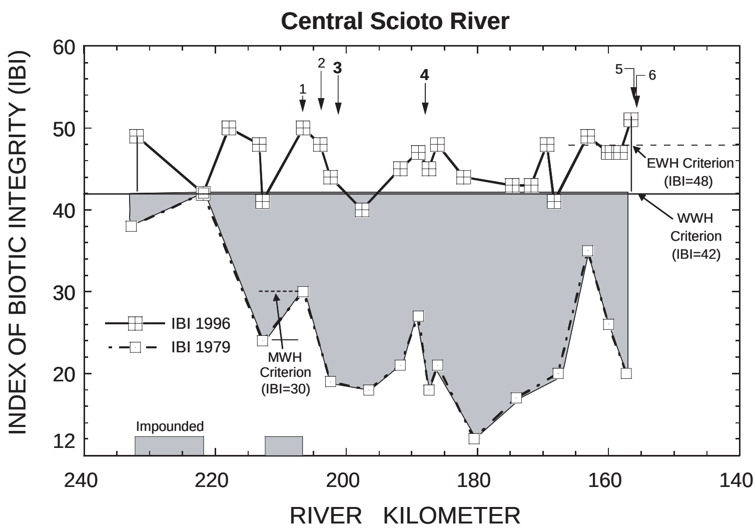


Figure 1.—IBI results from 2 years of electrofishing at multiple locations in a 64-km-long segment of the central Scioto River between Columbus and Circleville, Ohio. The biological criteria for the Warmwater Habitat (WWH), Modified Warmwater Habitat (MWH), and Exceptional Warmwater Habitat (EWH) use designations and major pollution sources are shown (1 = Whittier Street Combined Sewer Overflow; 2 = Techneglass; 3 = Jackson Pike Wastewater Treatment Plant; 4 = Columbus Southerly Wastewater Treatment Plant; 5 = Jefferson Smurfitt Corporation; 6 = Circleville Wastewater Treatment Plant; different font sizes indicate proportional pollutant loading). The shaded area below the WWH biocriterion yields the area of degradation value (ADV) and the unshaded area above the WWH biocriterion yields the area of attainment value (AAV).

charges such as municipal wastewater treatment plants (WWTPs) and heavy industries (steel making, petroleum refining, metal finishing, paper mills, and major manufacturing). This represented the focus of water quality management at that time, the administration of NPDES permits (National Pollution Discharge Elimination System, Section 402 of the Clean Water Act) and the municipal construction grants program. However, to meet the major objectives of the biological assessment program, we included the influence of other sources in the monitoring and assessment design. These included physical habitat alterations (dams, flow alterations, channelization), nonpoint sources (row crop agriculture, urbanization), and stressors resulting from land use changes (silt, nutrients, flow alterations). We characterized river status and apportioned impacts from various sources regardless of regulatory priority or status, precluding assessment bias towards a particular stressor. We focused on documenting changes in fish assemblages in Ohio rivers before and after implementation of mandatory wastewater treatment upgrades required to meet chemical water quality criteria and voluntary best management practices for the abatement of nonpoint sources of pollution. We chose these two time periods to demonstrate pre- to postpollution control changes, primarily upgrades in wastewater treatment at major municipal and industrial point sources. The principal regulatory driver was the EPA National Municipal Policy, which required wastewater treatment plants to attain effluent quality consistent with instream water quality criteria by July 1, 1988. The year 1990 was chosen as the boundary between "before" and "after" to reflect changes in the fish assemblage that allowed a 2–3-year period for achieving operational stability and initial biological recovery. The IBI and ADV/AAV were used to demonstrate the direction and degree of such changes in the subject rivers and to rank each in terms of their existing and potential quality and recovery status.

### *Training and Logistics*

Ohio EPA samples fish with field crews composed of one full time, professionally qualified crew leader and two technicians. Crew leaders have taxonomic and field sampling skills and are tested for this expertise during the interview and hiring process. New crew leaders undergo an apprenticeship with an experienced crew supervisor prior to performing crew leader duties on a routine basis (Ohio EPA 1989b). Technicians receive basic safety and sampling training prior to the field season.

### *Field Methods*

Fish are collected via standardized boat-mounted electrofishing (Ohio EPA 1987b, 1989b). Johnboats 3.6–4.8 m long are rigged with a hinged aluminum boom mounted on a bow platform. Four 6.25 mm diameter by 0.9 m long woven steel cables serve as anodes and are spaced evenly on a cross-member extending 2.0–2.75 m in front of the bow. Four 25 mm diameter by 1.8 m flexible stainless steel cathodes hang directly from the bow. Pulsed direct current is produced and transformed by Smith-Root type VI-A, 2.5, 3.5, and 5.0 GPP electrofishing units. Power output is varied depending on relative conductivity to produce 12–20 A.

Electrofishing is conducted during daylight June 15–October 15 each year, and for 0.5 km near shore (Ohio EPA 1989a; Yoder and Smith 1999). Time is recorded and a *minimum* of 2,000–2,500 s is specified to ensure sufficient intensity of sampling effort. All habitats (pools, runs, riffles, shoals, and backwaters) are thoroughly electrofished. The boat is maneuvered by outboard motors (9–15 hp) and/or manually by pushing in shallow water where motoring is not possible. Stunned and immobilized fish are netted by one person standing on the bow platform and placed in an aerated live well. This method of sampling is effective for a wide spectrum of river fish, including smaller benthic and riffle dwelling species, large bottom dwellers, pool dwellers, and deep, fast water inhabitants. Our sampling increased known ranges for numerous Ohio river fish species beyond those documented in Trautman (1981). Each sample typically produces 20–35 species (maximum = 50) and 250–500 fish (maximum > 1,000), provided chemical quality and physical habitat are not limiting.

Captured fish are enumerated by species, weighed, examined for external anomalies, and released or preserved in 10% formalin. Lengths are taken on selected species, otherwise species are classified as adults, 1+, or 0+. Fish less than 20–25 mm total length generally are not included in the samples, following the recommendation of Angermeier and Karr (1986).

A qualitative habitat assessment is conducted during each sampling pass using the Qualitative Habitat Evaluation Index (QHEI; Ohio EPA 1989a; Rankin 1989, 1995). The QHEI is a visual estimate of the quality, composition, amount, and extent of substrate, cover, channel, riparian, flow, pool/run/riffle, and gradient variables. The QHEI corresponds to key attributes of fish assemblage quality (Rankin 1989, 1995) and is an important tool in determining the appropriate and attainable use classification for Ohio rivers and streams (Rankin 1995; Yoder and Rankin 1995a). Those data are also entered and stored in the Ohio ECOS data management system.

Chemical and physical water quality data are collected near each biological site by separate field crews dedicated to this type of sampling. Core parameters collected at each site include field measurements (temperature, dissolved oxygen, conductivity, and pH) and a baseline set of conventional parameters (nitrogen series, phosphorus, biochemical oxygen demand, suspended and dissolved solids, chlorides, sulfates, and hardness), and common heavy metals. Additional parameters (other heavy metals, organic chemicals) are added if these contaminants are suspected. Most sampling consists of grab samples collected 3–8 times from the water column during the summer–early fall index period. Composite samples, continuous data, and analyses of bottom sediments (metals and organics) are included as the complexity of the situation dictates. This provides the essential data on stressors and exposures against which the response by the fish assemblage is interpreted (Yoder and Rankin 1998; Yoder and DeShon 2003.)

### *Data Entry and Data Analysis*

Data are recorded on field sheets, and after verifying voucher specimens, entered into the Ohio ECOS data management system. Each crew leader and a data entry analyst proofread all entries before the data are considered valid. We use a modified IBI that was developed and calibrated for boat-mounted electrofishing (Ohio EPA 1987b; Yoder and Rankin 1995a; Yoder and Smith 1999). The IBI consists of 12 metrics (Table 1) and generally adheres to the original guidance of Karr et al. (1986). The IBI values are calculated for individual sampling passes by a program in the Ohio ECOS routine following the procedures in Ohio EPA (1987b, 1989b). Data are analyzed with box-and-whisker plots and graphical routines for IBI scores and ADV values. The IBI is used to determine the proportion of sampled reaches that attain designated aquatic life uses (Ohio Administrative Code 3745–1). A site is considered impaired if the sample result is more than four IBI units below the criterion. The statistical properties of this IBI were described by Ohio EPA (1987b), Rankin and Yoder (1990), and Fore et al. (1993).

The ADV (Yoder and Rankin 1995b) was originally developed to quantify the extent and severity of departures from biocriteria within a defined river reach. We have added an Area of Attainment Value (AAV) that quantifies the extent to which minimum attainment criteria are surpassed. The ADV/AAV correspond to the area of the polygon formed by the longitudinal profile of IBI scores and the straight line boundary formed by a criterion, the ADV below and the AAV above (Figure 1). The computational formula (after Yoder and Rankin 1995b) is

$$\mathrm{ADV/AAV} = \sum [(\mathrm{aIBIa} + \mathrm{aIBIb}) - (\mathrm{pIBIa} + \mathrm{pIBIb})] * (\mathrm{RMa} - \mathrm{RMb}), \text{ for } a = 1 \text{ to } n, \text{ where}$$

aIBI$a$ = actual IBI at river mile $a$,

aIBI$b$ = actual IBI at river mile $b$,

pIBI$a$ = IBI biocriterion at river mile $a$,

pIBI$b$ = IBI biocriterion at river mile $b$,

RM$a$ = upstream most river mile,

RM$b$ = downstream most river mile, and

$n$ = number of samples.

The average of two contiguous sampling sites is assumed to integrate fish assemblage status for the distance between the points. The intensive survey

Table 1.—Index of biotic integrity metrics and scoring criteria based on fish assemblage data collected with boat electrofishers at nonwadeable sites in Ohio (after Ohio EPA 1987b). All percent metrics are based on fish numbers. Species metric assignments are available in Ohio EPA (1987b).

| Metric | Scoring criteria 5 | 3 | 1 |
|---|---|---|---|
| Native species[a] | >20 | 10–20 | <10 |
| % Round-bodied suckers[b] | >38 | 19–38 | <19 |
| Sunfish species[c] | >3 | 2–3 | <2 |
| Sucker species | >5 | 3–5 | <3 |
| Intolerant species | >3 | 2–3 | <2 |
| % tolerant | <15 | 15–27 | >27 |
| % omnivores | <16 | 16–28 | >28 |
| % insectivores | >54 | 27–54 | <27 |
| % top carnivores | >10 | 5–10 | <5 |
| % simple lithophils | | | |
| ≤1,560 ha (600 $mi^2$) | >50 | 25–50 | <25 |
| >1,560 ha (600 $mi^2$) | | Varies by drainage area | |
| % DELT anomalies | <0.5[d] | 0.5–3.0[e] | >3.0 |
| Fish numbers (no./km)[f] | >450 | 200–450 | <200 |

[a] Excludes all introduced and alien fish species.
[b] Includes *Moxostoma, Hypentelium, Minytrema, Erimyzon,* and *Cycleptus*; excludes white sucker *Catostomus commersonii*.
[c] Excludes black basses (*Micropterus* sp.).
[d] Or > 1 individual at sites with < 200 total fish.
[e] Or > 2 individuals at sites with < 200 total fish.
[f] Excludes tolerant and alien species and all hybrids; metric scoring adjustments are made at < 50 and 50–200 fish/km (Yoder and Smith 1999).

design typically positions sites in close enough proximity to sources of stress and along probable zones of impact and recovery so that meaningful changes are adequately captured. We have observed fish assemblages as portrayed by the IBI to change predictably in proximity to major sources and types of pollution in numerous instances (Ohio EPA 1987a; Yoder and Rankin 1995b; Yoder and Smith 1999). Thus, the longitudinal connection of contiguous sampling points produces a reasonably accurate portrayal of the extent and severity of impairment in a specified river reach as reflected by the IBI (Yoder and Rankin 1995a). The total ADV/AAV for a specified river segment is normalized to ADV/AAV units/km for making comparisons between years and rivers.

The ADV is calculated as a negative (below the biocriterion) expression; the AAV is calculated as a positive (above the biocriterion) expression. Each depicts the extent and degree of impairment (ADV) and attainment (AAV) of a biological criterion, which provides a more quantitative depiction of quality than pass/fail descriptors. It also allows the visualization of incremental changes in condition that may not alter the pass/fail status, but are nonetheless meaningful in terms of quantitative change over space and time. In our analyses, the Warmwater Habitat (WWH) biocriterion for the IBI, which varies by use designation and ecoregion (Table 2), is used as the threshold for calculating the ADV and AAV. The WWH use designation represents the minimum goal required by the Clean Water Act for the protection and propagation of aquatic life, thus it is used as a standard benchmark for ADV/AAV analyses.

### *Integrated Assessments*

Data and information are analyzed in accordance with a stress-exposure-response sequence (Figure 2; Yoder and Rankin 1998; Karr and Yoder 2004). The fish assemblage data are used to characterize and quantify the biological response to accompanying chemical/physical data and disturbance data

Table 2.—Biological criteria for the index of biotic integrity that are applicable to boat electrofishing sites in Ohio (Ohio Administrative Code Chapter 3745-1).

| Ecoregion | Modified Warmwater Habitat (MWH)[a] | Warmwater Habitat (WWH) | Exceptional Warmwater Habitat (EWH) |
|---|---|---|---|
| HELP – Huron/Erie Lake Plain | 20/22 | 34 | 48 |
| EOLP – Erie/Ontario Lake Plain | 24/30 | 40 | 48 |
| IP - Interior Plateau | 24/30 | 38 | 48 |
| ECBP – East Corn Belt Plains | 24/30 | 42 | 48 |
| WAP – West Allegheny Plateau | 24/30 | 40 | 48 |

[a] MWH biocriteria for channelized/impounded sites.

such as pollutant loadings, spills, land uses, and other indicators of human activity. This process, first developed by U.S. EPA (1990, 1995), has been extensively described by Yoder and Rankin (1998), Yoder and Smith (1999), Yoder and Kulik (2003), and Karr and Yoder (2004) and it is routinely employed by Ohio EPA. Key to the process is the accurate identification and quantification of biological impairments and the association of these impairments with relevant chemical and physical indicator thresholds and criteria (Yoder and Rankin 1995b; Yoder and DeShon 2003).

## Results

### *Changes in Fish Assemblages*

We analyzed data from 1979 through 2001, when we collected 135 of the 172 fish species recorded for Ohio (Trautman 1981; Appendix A). Many species occurred sporadically and rarely (58 species occurred in >10% of samples after 1990, versus 43 before 1990). Most species increased in relative abundance after 1990, and 36 species more than doubled in abundance. Of these, 14 are considered

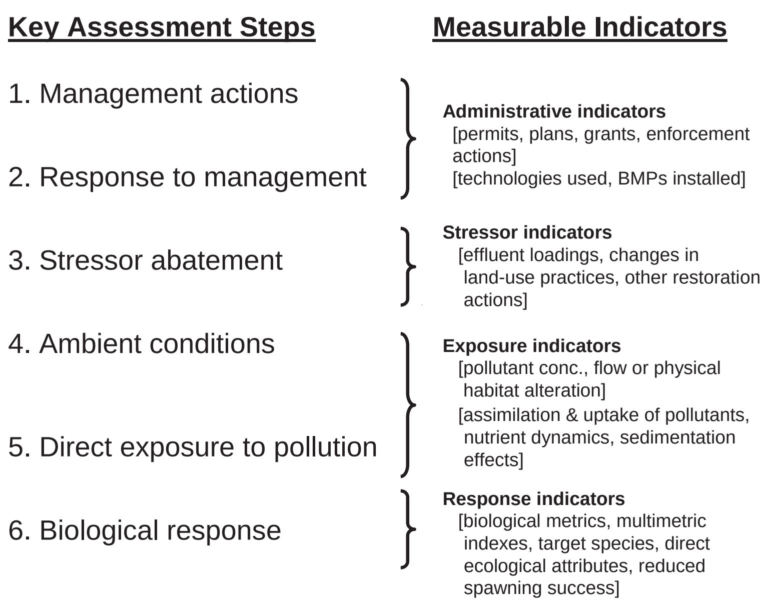


Figure 2.—Linkages between key steps of an adequate monitoring and assessment process, including their measurable indicators; the key assessment steps are sequential in a descending manner and comprise a feedback loop among and between steps (modified from U.S. EPA 1995, Yoder and Rankin 1998; and Karr and Yoder 2004).

highly intolerant, 5 are round-bodied suckers, 10 are darters, 5 are large river riffle species, and 5 are obligate large river species. One highly tolerant species, blacknose dace *Rhinichthys atratulus*, doubled in abundance but was rare in nonwadeable rivers and occurred in less than 5% of samples. Species declining at least 25% after 1990 were goldfish *Carassius auratus*, brown bullhead *Ameiurus nebulosus*, green sunfish *Lepomis cyanellus,* and white perch *Morone americana*, the first three of which are considered tolerant of poor water quality. Three other tolerant species declined slightly and six tolerant species increased slightly, but only one, bluntnose minnow *Pimephales notatus*, increased substantially. Our results indicated increased abundance and distribution of many Ohio fish species, with range extensions of several large river species collected by boat electrofishing (e.g., river redhorse *Moxostoma carinatum*, greater redhorse *M. valenciennesi*, smallmouth buffalo *Ictiobus bubalus*, black buffalo *I. niger*, sauger *Sander canadensis*, and gravel chub *Erimystax x-punctatus*). However, many other species remain well below their historical abundances and distributional ranges.

We ranked the status of 45 of Ohio's nonwadeable rivers including all major main-stem rivers and their largest tributaries draining more than 390 ha (Figure 3) using box-and-whisker plots of pre- and post-1990 IBI results. Rivers were ranked according to the post-1990 75th percentile IBI value, which better indicate assemblage condition *potential* than do medians or averages, and minimized the influence of reaches and sites that have not fully responded to pollution abatement practices and/or where those efforts were incomplete (Figure 4). Twelve of the top ranked 14 rivers were either wholly or partially classified as Exceptional Warmwater Habitat (EWH), which reflects a quality consistent with the 75th percentile of Ohio's least disturbed reference sites (Yoder and Rankin 1995a). The median IBI for 30 rivers met the Warmwater Habitat (WWH) IBI criterion, which is set at the 25th percentile of least disturbed reference sites and is the minimum restoration goal of the Clean Water Act in Ohio. Least disturbed ecoregional sites represent attainable background conditions and were used as reference sites for developing the Ohio biocriteria (Ohio EPA 1987b; Yoder and Rankin 1995a). Typically, this includes nonwadeable river reaches that are upstream or outside of the immediate influence of point sources and major habitat modifications. As such, these sites reflect attainable biological condition in terms of the concept of regional reference envisioned by Hughes et al. (1986).

We used changes in the median IBI and changes in the ADV/AAV to express the degree and significance of temporal changes for 38 river segments (Tables 3–5). Positive changes in median IBI values were significant (i.e., >4 units; Ohio EPA 1987b; Rankin and Yoder 1990) in 28 rivers, with changes of greater or equal to 10 or more units in 17 rivers (Table 3). There were no significant declines in the median IBI in any of the rivers. The percent change in median IBI values was positive in all but three rivers, exceeding 100% in three rivers (central Scioto, middle Great Miami, and Cuyahoga rivers) and more than 25% in 15 others (Table 4). Changes in the ADV/AAV were expressed as a real change in impairment (negative ADV), a real change in attainment (AAV), and the net change in both the ADV and AAV (Table 4). Reduced impairment occurred in all but seven rivers, and gains in attainment occurred in all but seven rivers. Net gains in ADV/AAV were observed in all but five rivers.

## *General Disturbance Types*

In terms of the relative ranking of the rivers using the 75th percentile IBI (Table 3), the highest ranked rivers received effluents from municipal WWTPs and runoff from agricultural nonpoint sources, followed by industrial, urban, and other sources. Seven rivers predominantly disturbed by complex toxics and acid mine drainage ranked at the bottom, even though incremental improvements were observed (Table 3). They reflected residual impacts from uncontrolled toxic impacts and contaminated benthic sediments that are not as amenable to conventional abatement practices.

The overall distribution of IBIs and changes in the median IBIs between rivers predominantly disturbed by agricultural nonpoint sources, municipal WWTPs, and all other sources (industrial, urban, complex toxic, and acid mine drainage) showed

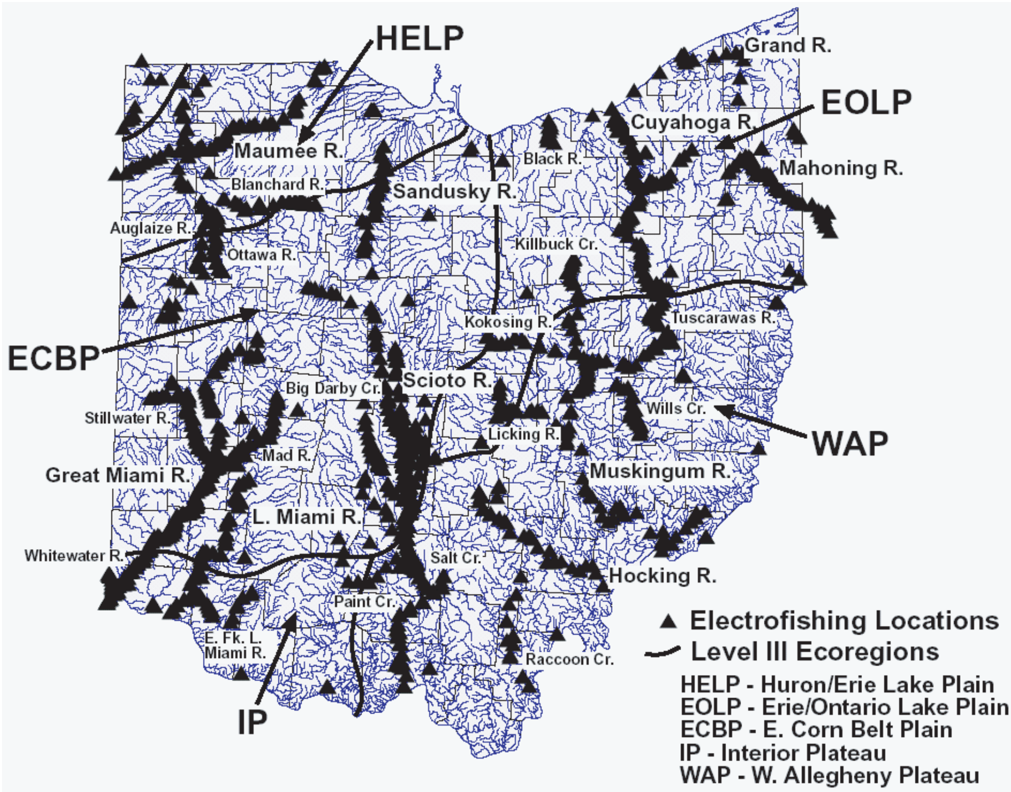


Figure 3.—Ohio EPA nonwadeable electrofishing locations sampled between 1979 and 2001. All sites drain greater or equal to 390 ha and were sampled with boat electrofishers per Ohio EPA (1989b). Sampled main-stem rivers and significant tributaries are labeled, and level-III ecoregion boundaries are shown.

improvements after 1990 (Figure 5). This improvement reflected the relative severity of each disturbance type, with the lowest IBI values associated with complex toxic and industrial sources, followed by WWTP and agricultural nonpoint sources. In terms of changes in median IBI values, this order was reversed. The greatest gains in the 25th–75th percentiles occurred with the complex toxic and industrial disturbance types, followed by WWTPs and agricultural nonpoint sources. This reflects, in part, the greater severity of these impacts and hence more potential for improvement. Nevertheless, half the rivers disturbed by complex toxic and industrial sources had IBIs that attained the WWH biocriterion after 1990, compared with no values meeting the biocriterion before 1990. No IBIs attained the EWH biocriterion for complex toxic and industrial sources before 1990 and only a few sites did so after 1990. Rivers with the least or no positive changes in the ADV or AAV were disturbed by complex toxics and acid mine drainage (Table 4). Agricultural nonpoint sources showed the least improvement in terms of net ADV + AAV increases, although the median IBI improved from marginal WWH to EWH quality. The WWTP impacted rivers showed slightly more improvement.

The largest increases in median IBIs consistently occurred in rivers that were predominantly impacted by municipal WWTPs and included the central Scioto (+26), upper Hocking (+22), Licking (+16), upper Great Miami (+14), and lower Great Miami (+12) rivers (Table 3). In each of these areas, agricultural row cropping was the predominant land use upriver of a major urban area that also included combined sewer overflows in all except one river. Areas of impairment associated with combined sew-

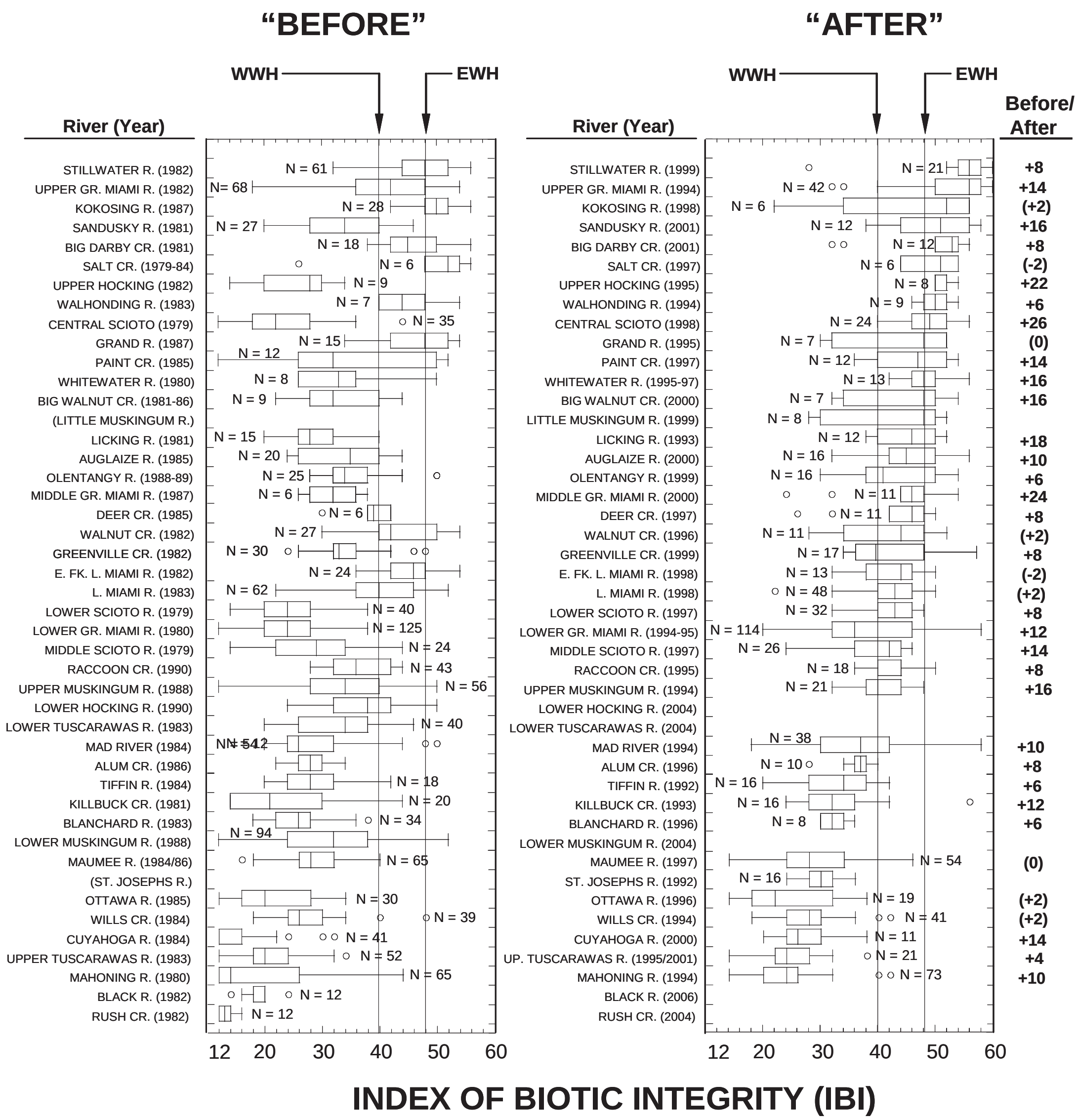


Figure 4.—Box-and-whisker plots of index of biotic integrity (IBI) results from 45 Ohio river segments sampled by boat electrofishers. Results from 1979 to 1990 appear in the left figure box and are labeled "Before." Results after 1990 appear in the right figure box and are labeled "After." Rivers are ranked according to the 75th percentile IBI value in the "After" period. The net change (±) in the median IBI from "Before" to "After" is listed under ΔBefore/After column; nonsignificant changes (< 4 IBI units) are listed in parentheses. The biological criteria for the Warmwater Habitat (WWH) and Exceptional Warmwater Habitat (EWH) use designations are indicated by vertical lines. N is sample size.

Table 3.— Fish assemblage status for boat electrofishing in 45 Ohio rivers comparing index of biotic integrity results for the earliest and latest Ohio EPA surveys during the period 1979 through 2001; rivers are ranked by their 75th percentile value for the latest ("After") period. Results prior to 1988–1990 ("Before") reflect conditions prior to the upgrades of major municipal wastewater treatment facilities and most industrial discharges; conditions after 1990 ("After") reflect reduced impacts. The IBI statistics are for the entire river segment included in the survey; maximum and minimum values exclude statistical outliers (>2 interquartile ranges [IQR] beyond the median). Principal disturbance types are listed in order of importance. Rivers in boldface are classified Exceptional Warmwater Habitat (EWH) over most of the surveyed segment, the remaining rivers are classified Warmwater Habitat (WWH). The IBI values that fail to meet numeric criteria for the IBI in each river are underlined; poor and very poor values are also in boldface.

| | Earliest ("Before") | | | | | | Latest ("After") | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| River | Year | N | Med. (±2IQR) | Max. | 75%ile | Min. | Year | N | Med. (±2IQR) | Max. | 75%ile | Min. | Median Change (±) | Disturbance type(s) |
| **Stillwater R.** | 1982 | 61 | 48 (1.34) | 56 | 52 | 32 | 1999 | 21 | 56 (3.96) | 60 | 58 | 52 | +8 | Ag. NPS, WWTPs |
| **Upper Gr. Miami** | 1982 | 68 | 42 (2.00) | 54 | 48 | 40 | 1994 | 42 | 56 (2.00) | 60 | 58 | 40 | +14 | WWTPs, Ag. NPS, Channelized |
| **Kokosing R.** | 1987 | 28 | 50 (1.46) | 54 | 52 | **18** | 1998 | 6 | 52 (10.0) | 56 | 56 | 22 | (+2) | WWTPs, Ag. NPS |
| Sandusky R. | 1981 | 27 | 34 (3.02) | 46 | 40 | **20** | 2001 | 12 | 50 (3.66) | 58 | 56 | 38 | +16 | Ag. NPS, WWTPs, Impounded |
| **Big Darby Cr.** | 1981 | 18 | 44 ((2.68) | 56 | 50 | 38 | 2001 | 12 | 52 (4.64) | 56 | 54 | 44 | +8 | Ag. NPS, WWTPs |
| **Salt Cr.** | 1979–1984 | 6 | 52 (9.12) | 56 | 54 | 48 | 1992 | 6 | 50 (4.06) | 54 | 54 | 44 | (-2) | Ag. NPS, Silviculture |
| Upper Hocking | 1982 | 9 | 28 (4.58) | 34 | 30 | **14** | 1995 | 8 | 50 (1.06) | 54 | 52 | 50 | +22 | WWTPs, Complex toxic, Chan. |
| **Walhonding R.** | 1983 | 7 | 44 (3.78) | 54 | 48 | 40 | 1994 | 9 | 50 (1.70) | 54 | 52 | 46 | +6 | Ag. NPS, Flow regulation |
| Central Scioto R. | 1979 | 35 | **22** (2.46) | 44 | 28 | **12** | 1998 | 24 | 48 (1.52) | 56 | 52 | 40 | +26 | WWTPs, CSOs, Urban, Ag. NPS |
| **Grand R.** | 1987 | 15 | 48 (3.22) | 54 | 52 | 34 | 1995 | 7 | 48 (7.26) | 52 | 52 | 30 | (0) | Ag. NPS, WWTPs |
| **Paint Cr.** | 1985 | 12 | 32 (7.24) | 52 | 50 | **12** | 1997 | 12 | 46 (3.44) | 54 | 52 | 36 | +14 | Industrial, Ag. NPS |
| **Whitewater R.** | 1980 | 8 | 32 (5.28) | 50 | 36 | 26 | 1995–1997 | 13 | 48 (2.18) | 56 | 50 | 42 | +16 | Ag. NPS, Flow reg. |
| **Big Walnut Cr.** | 1981–1986 | 9 | 32 (5.02) | 44 | 40 | 22 | 2000 | 7 | 48 (6.30) | 54 | 50 | 32 | +16 | Urban, WWTPs, Ag. NPS |
| **L. Muskingum R.** | (No survey before 1990) | | | | | | 1999 | 8 | 48 (6.66) | 52 | 50 | 28 | | Silviculture, Ag. NPS |
| Licking R. | 1981 | 15 | 28 (2.60) | 40 | 32 | **20** | 1993 | 12 | 46 (2.70) | 52 | 50 | 38 | +18 | WWTPs, Ag. NPS, CSO, Urban |
| Auglaize R. | 1985 | 20 | 34 (2.90) | 44 | 40 | **24** | 2000 | 16 | 44 (3.46) | 56 | 50 | 32 | +10 | Ag. NPS, WWTPs |
| Olentangy R. | 1988–1989 | 25 | 34 (2.42) | 44 | 38 | 28 | 1999 | 16 | 40 (3.48) | 54 | 50 | 30 | +6 | Urban, CSO, Impounded |
| Mid. Gr. Miami | 1988 | 27 | **22** (4.36) | 44 | 28 | **16** | 2000 | 11 | 46 (5.22) | 54 | 48 | 44 | +14 | Thermal, WWTPs, Urban |
| Deer Cr. | 1985 | 6 | 38 (3.62) | 42 | 42 | 38 | 1997 | 11 | 46 (5.50) | 50 | 48 | 42 | +8 | Ag. NPS, WWTPs |
| Walnut Cr. | 1982 | 27 | 42 (2.26) | 54 | 50 | 30 | 1996 | 11 | 44 (4.92) | 52 | 48 | 28 | (+2) | WWTPs, Ag. NPS |
| **Greenville Cr.** | 1982 | 30 | 32 (2.08) | 42 | 36 | **26** | 1999 | 17 | 40 (3.52) | 56 | 48 | 32 | +8 | WWTPs, Ag. NPS |
| **E. Fk. L. Miami** | 1982 | 24 | 46 (1.76) | 54 | 48 | 36 | 1998 | 13 | 44 (2.98) | 50 | 46 | 32 | (-2) | WWTPs, Flow reg. |
| **L. Miami R.** | 1983 | 62 | 40 (1.78) | 52 | 46 | 22 | 1998 | 48 | 42 (1.54) | 50 | 46 | 32 | (+2) | WWTPs, Ag. NPS, CSO |
| Lower Scioto R. | 1979 | 40 | **24** (1.86) | 38 | 28 | **14** | 1997 | 32 | 42 (1.54) | 48 | 46 | 32 | +18 | Industrial, Ag. NPS, Silviculture |

Table 3.—Continued

| River | Earliest ("Before") Year | N | Med. (±2IQR) | Max. | 75%ile | Min. | Latest ("After") Year | N | Med. (±2IQR) | Max. | 75%ile | Min. | Median Change (±) | Principal impact type(s) |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Lower Gr. Miami | 1980 | 125 | 24 (1.04) | 38 | 28 | 12 | 1994–1995 | 114 | 36 (1.62) | 58 | 46 | 20 | +12 | WWTPs, Industrial, Urban |
| Middle Scioto R. | 1979 | 24 | 28 (2.70) | 44 | 34 | 14 | 1997 | 26 | 42 (2.00) | 46 | 44 | 24 | +14 | Industrial, Ag. NPS, Chan. |
| Raccoon Cr. | 1990 | 10 | 36 (3.54) | 44 | 42 | 28 | 1994–1995 | 18 | 44 (1.78) | 50 | 44 | 36 | +6 | AMD, Silviculture |
| Upper Muskingum | 1988 | 43 | 34 (2.84) | 50 | 40 | 12 | 1994 | 21 | 40 (2.12) | 48 | 42 | 32 | +16 | Industrial, Thermal, WWTPs |
| Lower Hocking R. | 1990 | 56 | 38 (1.66) | 50 | 42 | 24 | (First survey after 1990 planned for 2004) | | | | | | | AMD, WWTPs, Ag. NPS |
| Lower Tuscarawas | 1983 | 40 | 34 (2.20) | 46 | 38 | 20 | (First survey after 1990 planned for 2004) | | | | | | | Industrial, Ag. NPS, WWTPs |
| Mad R. | 1984 | 54 | 26 (1.90) | 44 | 30 | 20 | 1994 | 38 | 36 (2.88) | 58 | 42 | 18 | +10 | WWTPs, Ag. NPS, Channelized |
| Alum Cr. | 1986 | 12 | 28 (1.90) | 34 | 30 | 22 | 1996 | 10 | 36 (2.10) | 40 | 38 | 34 | +8 | Urban, CSOs, WWTPs, Flow reg. |
| Tiffin R. | 1984 | 18 | 28 (2.96) | 42 | 32 | 20 | 1992 | 16 | 34 (3.24) | 42 | 38 | 20 | +6 | Ag. NPS, Channelized |
| Killbuck Cr. | 1981 | 20 | 20 (3.68) | 44 | 30 | 14 | 1993 | 16 | 32 (3.78) | 42 | 36 | 24 | +12 | WWTP, Ag. NPS, Channelized |
| Blanchard R. | 1983 | 34 | 26 (1.60) | 36 | 28 | 18 | 1996 | 8 | 32 (1.52) | 36 | 34 | 30 | +6 | WWTP, Ag. NPS, CSO |
| Lower Muskingum | 1988 | 54 | 32 (2.06) | 52 | 38 | 12 | (First survey after 1990 planned for 2004) | | | | | | | Industrial, Thermal, Impounded |
| Maumee R. | 1984–1986 | 65 | 28 (1.24) | 40 | 32 | 18 | 1997 | 54 | 28 (2.02) | 46 | 34 | 14 | (0) | Ag. NPS, WWTPs, Impounded |
| St. Josephs R. | (No survey before 1990) | | | | | | 1992 | 16 | 30 (1.70) | 36 | 32 | 24 | | Ag. NPS, WWTPs, Channelized |
| Ottawa R. | 1985 | 30 | 20 (2.14) | 34 | 30 | 12 | 1996 | 19 | 22 (3.50) | 38 | 32 | 14 | (+2) | Complex toxic, Ind./WWTP, CSO |
| Wills Cr. | 1984 | 39 | 26 (1.82) | 34 | 30 | 18 | 1994 | 41 | 28 (1.60) | 36 | 30 | 18 | (+2) | AMD, WWTP, Ag. NPS, Silvic. |
| Cuyahoga R. | 1984 | 41 | 12 (1.54) | 22 | 16 | 12 | 2000 | 11 | 26 (3.14) | 38 | 30 | 20 | +14 | Complex toxic, WWTP/Ind., CSO |
| Upper Tuscarawas | 1983 | 52 | 20 (1.50) | 32 | 24 | 12 | 1995/01 | 21 | 24 (2.34) | 32 | 28 | 14 | +4 | Comp. toxic, WWTP/Ind., Chan. |
| Mahoning R. | 1980 | 65 | 14 (2.18) | 44 | 26 | 12 | 1994 | 73 | 24 (1.38) | 32 | 26 | 14 | +10 | Com. Tox., WWTP/Ind., CSO, Imp. |
| Black R. | 1982 | 12 | 20 (1.42) | 20 | 20 | 16 | (First survey after 1990 planned for 2006) | | | | | | | Complex toxic, WWTP/Ind., CSO |
| Rush Cr. | 1982 | 12 | 12 (0.90) | 16 | 14 | 12 | (First survey after 1990 planned for 2004) | | | | | | | AMD, Ag. NPS |

**Impact type abbreviations**: WWTPs = Municipal wastewater treatment facilities; Industrial = Industrial wastewater treatment facilities; Ag. NPS = Agricultural nonpoint sources (mainly row cropping); Chan. = channelization of main channel; Imp. = impoundment of main channel by run-of-river low head dams; Silv = Silvicultural practices; Complex Toxic = Complex mixture of toxic sources including industrial, WWTP, and NPS; CSO = Combined sewer overflows; Flow = flow regulation; AMD = Acidic mine drainage.

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Table 4.—Fish assemblage results from earliest and latest Ohio EPA surveys for 38 river segments comparing changes in median index of biotic integrity (IBI) values, area of degradation value (ADV) and the area of attainment value (AAV). Segments are arranged in the same order as Figure 4. Changes are expressed as the increase (+) or decrease (–) in the % median IBI, ADV/km, AAV/km, and net change in ADVand/or AAV/km. The IBI values that fail to meet numeric criteria for the IBI in each river are underlined; poor and very poor values are also in boldface. Declining net ΔADV+ AAV results are in boldface and underlined.

| | "Before" | | | | "After" | | | | Changes | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| River | Year | Median IBI (±2IQR) | ADV | AAV | Year | Median IBI (±2IQR) | ADV | AAV | %ΔMed. IBI (±) | Δ ADV | Δ AAV | Net Δ ADV + AAV |
| Stillwater R. | 1982 | 48 (1.34) | –2.0 | +41.1 | 1999 | 56 (3.96) | –4.6 | +118 | +17% | **–2.6** | +76.9 | +74.3 |
| Upper Gr. Miami | 1982 | 42 (2.00) | –17.8 | +25.5 | 1994 | 56 (2.00) | –0.1 | +115 | +33% | +17.7 | +89.5 | +158 |
| Kokosing R. | 1987 | 50 (1.46) | 0 | +90.7 | 1998 | 52 (10.0) | 0 | +88.7 | +4% | 0 | –2.**0** | –**2.0** |
| Sandusky R. | 1981 | 34 (3.02) | –32.4 | +6.7 | 1999 | 50 (3.66) | 0 | +85.9 | +47% | +32.4 | +79.2 | +112 |
| Big Darby Cr. | 1981 | 44 (2.68) | 0 | +71.1 | 2001 | 52 (4.64) | 0 | +90.3 | +18% | 0 | +19.2 | +19.2 |
| Salt Cr. | 1984 | 52 (9.12) | 0 | +80.0 | 1992 | 50 (4.06) | –0.2 | +50.6 | -3.8% | **–0.2** | **–29.4** | **–29.6** |
| Upper Hocking R. | 1982 | 28 (4.58) | –108 | 0 | 1995 | 50 (1.06) | 0 | +144 | +79% | +108 | +144 | +252 |
| Walhonding River | 1983 | 44 (3.78) | –5.5 | +3.2 | 1994 | 50 (1.70) | 0 | +70.8 | +14% | +5.5 | +67.6 | +73.1 |
| Central Scioto R. | 1979 | **22** (2.46) | –193 | 0 | 1998 | 48 (1.52) | 0 | +76 | +118% | +193 | +76 | +269 |
| Grand River | 1987 | 48 (3.22) | –32.9 | +39.6 | 1995 | 48 (7.26) | –3.9 | +72.0 | 0% | +29 | +32.4 | +61.4 |
| Paint Cr. | 1985 | 32 (7.24) | –61.6 | 0 | 1997 | 46 (3.44) | 0 | +56 | +44% | +61.6 | +56 | +118 |
| Whitewater River | 1980 | 32 (5.28) | –119.5 | 0 | 1995 | 48 (2.18) | –0.3 | +51.3 | +50% | +119 | +51.3 | +171 |
| Big Walnut Cr. | 1981–1986 | 32 (5.02) | –14.9 | 0 | 2000 | 48 (6.30) | –2.5 | +70.9 | +50% | +12.4 | +70.9 | +83.3 |
| Licking R. | 1981 | 28 (2.60) | –84.4 | 0 | 1993 | 46 (2.70) | 0 | +101 | +64% | +84.4 | +101 | +185 |
| Auglaize R. | 1985 | 34 (2.90) | –10.4 | +35.7 | 2000 | 44 (3.46) | –1.9 | +126 | +29% | +8.5 | +90.3 | +98.8 |
| Olentangy R. | 1988–1989 | 34 (2.42) | –35.0 | +11.3 | 1999 | 40 (3.48) | –0.4 | +67.8 | +18% | +34.6 | +56.5 | +91.1 |
| Mid. Gr. Miami R. | 1988 | **22** (4.36) | –147 | +1.5 | 2000 | 46 (5.22) | 0 | +129.1 | +100% | +147 | +128 | +275 |
| Deer Cr. | 1985 | 38 (3.62) | –69.6 | 0 | 1997 | 46 (5.50) | –4.7 | +16.5 | +21% | +64.9 | +16.5 | +81.4 |
| Walnut Cr. | 1982 | 42 (2.26) | –0.7 | +55.3 | 1996 | 44 (4.92) | –11.5 | +54.4 | +4.7% | **–10.8** | **–1.1** | **–11.9** |
| Greenville Cr. | 1982 | 32 (2.08) | –85.2 | +1.0 | 1999[1] | 40 (3.52) | –22.3 | +8.6 | +25% | +62.9 | +7.6 | +70.5 |
| E. Fk. L. Miami | 1982 | 46 (1.76) | –6.9 | +17.1 | 1998 | 44 (2.98) | –11.3 | +5.8 | -4.3% | **–4.4** | **–11.3** | **–15.7** |
| L. Miami R. | 1983 | 40 (1.78) | –54.5 | +3.1 | 1998 | 42 (1.54) | –22.8 | +8.3 | +4.7% | +31.7 | +5.2 | +36.9 |
| Lower Scioto R. | 1979 | **24** (1.86) | –117 | 0 | 1997 | 42 (1.54) | –0.3 | +61.7 | +75% | +116 | +61.7 | +178 |
| Lower Gr. Miami | 1980 | **24** (1.04) | –131 | 0 | 1994-5 | 36 (1.62) | –17.8 | +25.5 | +50% | +113 | +25.5 | +139 |
| Middle Scioto R. | 1979 | 28 (2.70) | –104 | 0 | 1997 | 42 (2.00) | –9.2 | +31.1 | +50% | +94.8 | +31.1 | +126 |
| Raccoon Cr. | 1990 | 36 (3.54) | –10.0 | 0 | 1994-5 | 44 (1.78) | 0 | +62.8 | +22% | +10.0 | +62.8 | +72.8 |
| Upper Muskingum R. | 1988 | 34 (2.84) | –12.2 | +13.4 | 1994 | 40 (2.12) | –3.0 | +31.2 | +18% | +9.2 | +17.8 | +27.0 |
| Mad R. | 1984 | **26** (1.90) | –109 | +1.7 | 1994 | 36 (2.88) | –37.1 | +18.5 | +38% | +71.9 | +16.8 | +88.7 |
| Alum Cr. | 1986 | 28 (1.90) | –103 | 0 | 1996 | 36 (2.10) | –40.7 | 0 | +29% | +62.3 | 0 | +62.3 |

Table 4.—Continued.

| River | Year | Earliest Median IBI (±2IQR) | ADV | AAV | Year | Latest Median IBI (±2IQR) | ADV | AAV | Changes %ΔMed. IBI (±) | Δ ADV | Δ AAV | Net Δ ADV + AAV |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Tiffin R. | 1984 | <u>28</u> (2.96) | –21.8 | +5.0 | 1992 | 34 (3.24) | –6.6 | +25.4 | +21% | +15.2 | +20.4 | +35.6 |
| Killbuck Cr. | 1981 | **<u>20</u>** (3.68) | –136 | 0 | 1993 | <u>32</u> (3.78) | –20.6 | +40.1 | +60% | +115 | +40.1 | +155 |
| Blanchard R. | 1983 | **<u>26</u>** (1.60) | –46.0 | 0 | 1996 | <u>32</u> (1.52) | 0 | +17.0 | +23% | +46.0 | +17.0 | +63.0 |
| Maumee R. | 1984–1986 | <u>28</u> (1.24) | –7.2 | +23.8 | 1997 | <u>28</u> (2.02) | –33.2 | +36.9 | 0 | -26.0 | +13.1 | –**<u>12.9</u>** |
| Ottawa R. | 1985 | **<u>20</u>** (2.14) | –146 | +0.8 | 1996 | **<u>22</u>** (3.50) | –116.0 | +3.5 | +10% | +29.8 | +2.7 | +32.5 |
| Wills Cr. | 1984 | **<u>26</u>** (1.82) | –94.0 | 0 | 1994 | <u>28</u> (1.60) | –72.0 | 0 | +7.6 | +22.0 | 0 | +22.0 |
| Cuyahoga R. | 1984 | **<u>12</u>** (1.54) | –207 | 0 | 2000 | <u>26</u> (3.14) | –56.3 | +3.9 | +100% | +151 | +3.9 | +155 |
| Upper Tuscarawas | 1983 | <u>14</u> (1.50) | –131 | 0 | 1995/01 | **<u>24</u>** (2.34) | –36.4 | +4.0 | +20% | +94.6 | +4.0 | +98.6 |
| Mahoning R. | 1980 | **<u>24</u>** (2.18) | –185 | +0.2 | 1994 | **<u>24</u>** (1.38) | –126 | 0 | 0 | +59.0 | –**<u>0.2</u>** | +58.8 |

[a] Data collected with wading methods; included for comparison purposes only.

Table 5.—Rank order of selected rivers for before and after changes (Δ) in median index of biotic integrity (IBI), % median IBI, area of degradation value (ADV), area of attainment value (AAV), and net ADV + AAV.

| ΔMedian IBI | %ΔMedian IBI | Δ ADV | ΔAAV | ΔNet ADV + AAV |
|---|---|---|---|---|
| Central Scioto R. (+26) | Central Scioto R. (+118) | Central Scioto R. (+193) | Upper Hocking R. (+144) | Middle Gr. Miami R. (+275) |
| Upper Hocking R. (+22) | Middle Gr. Miami R. (+100) | Cuyahoga R. (+151) | Middle Gr. Miami R. (+128) | Central Scioto R. (+269) |
| Lower Scioto R. (+18) | Cuyahoga R. (+100) | Middle Gr. Miami R. (+147) | Licking R. (+101) | Upper Hocking R. (+252) |
| Sandusky R. (+16) | Upper Hocking R. (+79) | Whitewater R. (+119) | Auglaize R. (+90.3) | Licking R. (+185) |
| Big Walnut Cr. (+16) | Lower Scioto R. (+75) | Lower Scioto R. (+116) | Upper Gr. Miami R. (+89.5) | Lower Scioto R. (+178) |
| Licking R. (+16) | Licking R. (+64) | Killbuck Cr. (+115) | Sandusky R. (+79.2) | Whitewater R. (+171) |
| Upper Gr. Miami R. (+14) | Killbuck Cr. (+60) | Lower Gr. Miami R. (+113) | Stillwater R. (+76.9) | Upper Gr. Miami R. (+158) |
| Paint Cr. (+14) | Whitewater R. (+50) | Upper Hocking (+108) | Central Scioto R. (+76) | Cuyahoga R. (+155) |
| Whitewater R. (+14) | Big Walnut Cr. (+50) | Middle Scioto R. (+94.8) | Big Walnut Cr. (+70.9) | Killbuck Cr. (+155) |
| Middle Gr. Miami R. (+14) | Lower Gr. Miami R. (+50) | Upper Tuscarawas R. (+94.6) | Walhonding R. (+67.6) | Lower Gr. Miami R. (+139) |
| Cuyahoga R. (+14) | Middle Scioto R. (+50) | Licking R. (+84.4) | Raccoon Cr. (+62.8) | Middle Scioto R. (+126) |
| Lower Great Miami R. (+12) | Sandusky R. (+47) | Mad R. (+71.9) | Lower Scioto R. (+61.7) | Paint Cr. (+118) |
| Middle Scioto R. (+12) | Paint Cr. (+44) | Deer Cr. (+64.9) | Olentangy R. (+56.5) | Sandusky R. (+112) |
| Auglaize R. (+10) | Mad R. (+38) | Greenville Cr. (+62.9) | Paint Cr. (+56) | Auglaize R. (+98.8) |
| Greenville Cr. (+10) | Upper Gr. Miami R. (+33) | Alum Cr. (+62.3) | Whitewater R. (+51.3) | Upper Tuscarawas R. (+98.6) |
| Mad R. (+10) | Auglaize R. (+29) | Paint Cr. (+61.6) | Killbuck Cr. (+40.1) | Olentangy R. (+91.1) |
| Killbuck Cr. (+10) | Alum Cr. (+29) | Mahoning R. (+59) | Grand R. (+32.4) | Mad R. (+88.7) |
| Mahoning R. (+10) | Blanchard R. (+23) | Blanchard R. (+46) | Middle Scioto R. (+31.1) | Big Walnut Cr. (+83.3) |
| Stillwater R. (+8) | Raccoon Cr. (+22) | Olentangy R. (+34.6) | Lower Gr. Miami R. (+25.5) | Deer Cr. (+81.4) |
| Big Darby Cr. (+8) | Deer Cr. (+21) | Sandusky R. (+32.4) | Tiffin R. (+20.4) | Stillwater R. (+74.3) |
| Raccoon Cr. (+8) | Tiffin R. (+21) | L. Miami R. (+31.7) | Big Darby Cr. (+19.2) | Walhonding R. (+73.1) |
| Alum Cr. (+8) | Upper Tuscarawas R. (+20) | Ottawa R. (+29.8) | Up. Muskingum R. (+17.8) | Raccoon Cr. (+72.8) |
| Walhonding R. (+6) | Big Darby Cr. (+18) | Grand R. (+29) | Blanchard R. (+17) | Greenville Cr. (+63.0) |
| Olentangy R. (+6) | Olentangy R. (+18) | Wills Cr. (+22) | Mad R. (+16.8) | Blanchard R. (+62.3) |
| Deer Cr. (+6) | Upper Muskingum R. (+18) | Upper Gr. Miami R. (+17.7) | Deer Cr. (+16.5) | Alum Cr. (+72.8) |
| Upper Muskingum R. (+6) | Stillwater R. (+17) | Tiffin R. (+15.2) | Maumee R. (+13.1) | Grand R. (+61.4) |
| Tiffin R. (+6) | Walhonding R. (+14) | Raccoon Cr. (+10) | Greenville Cr. (+7.6) | Mahoning R.(+58.8) |

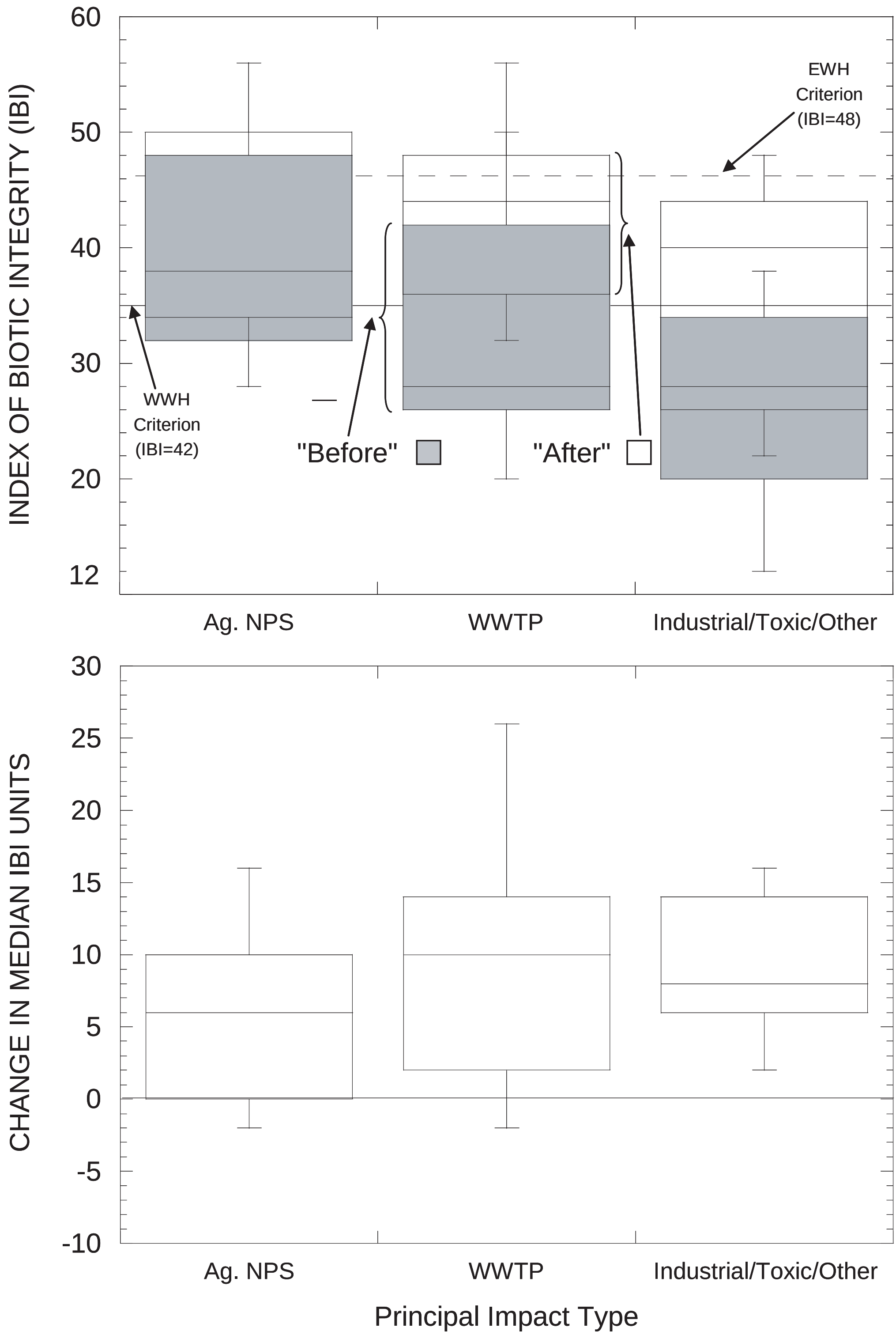



Figure 5.—Distribution of median IBI values (upper panel) and net change in median IBI (lower panel) between the "Before" and "After" periods aggregated by the three predominant disturbance types of Yoder and Rankin (1995b).

ers were observed in some urban areas, but were localized compared to the larger reaches affected by the WWTP effluents.

Notable improvements also occurred in some rivers disturbed by industrial, complex toxic, urban, and mine drainage sources, and included the lower Scioto River (industrial; +18), Big Walnut Creek (urban; +16), Paint Creek (industrial; +14), Cuyahoga River (complex toxic; +14), and middle Scioto River (industrial; +14). Improvements of a similar magnitude were less common in rivers receiving agricultural nonpoint sources, but included comparable changes in the Sandusky River (+16), Whitewater River (+16), and Auglaize River (+10). Many of the highest quality rivers in Ohio were predominantly disturbed by agricultural nonpoint sources and exhibited very good to exceptional quality before 1990. As a result, none of these rivers exhibited significant increases in median IBIs after 1990. Declines in median IBIs occurred in two rivers (East Fork Little Miami River and Salt Creek), but the changes were not significant (i.e., <4 IBI units). However, both are designated EWH and any decline is noteworthy. The East Fork Little Miami River was disturbed by municipal WWTPs that have recently approached treatment capacity. Salt Creek is disturbed by nonpoint sources, especially by row cropping that has increasingly encroached on the riparian zone.

### *Magnitude and Extent of Changes*

A comparison of the ranking of selected rivers for the median IBI and ADV/AAV statistics (Figure 4; Table 5) shows consistency in some, but not all rivers. The most impaired rivers before 1990 included the Cuyahoga, central Scioto, Mahoning, Ottawa, upper Tuscarawas, lower Great Miami, Whitewater, and lower Scioto rivers, all with ADV greater than 100 units/km and zero AAV/km. After 1990, impairment was substantially reduced in the central Scioto (ADV/km = 0), Whitewater (–0.3), lower Scioto (–0.3), and lower Great Miami (–17.8) rivers, but remained high in the Mahoning (–126) and Ottawa (–116) rivers. The reduced impairments were associated with improved wastewater treatment and included all rivers with large gains in median IBI values (Table 4). Those rivers with the greatest impairments and least improved IBIs after 1990 are disturbed by complex toxics.

Only 12 rivers had pre-1990 AAVs greater than 10 units/km, and 17 had values of zero. AAVs increased markedly after 1990 with three rivers exceeding 100 units/km and only three rivers with values of zero (Table 4). Rivers showing the largest gains included the upper Hocking River (recovery from WWTP and complex toxic impacts), middle Great Miami River (recovery from acute thermal impacts), Auglaize River (recovery from agricultural nonpoint source impacts), Stillwater River (existing high quality), and upper Great Miami River (recovery from WWTP impacts). In terms of net gains made in both ADVs and AAVs, 7 of the top 10 rivers were disturbed by WWTPs and increased as much as three- to fourfold over other sources (Table 5).

The greatest reduction in impairment occurred for the mix of other sources (industrial, urban, complex toxic, and acid mine drainage) and WWTP. Reductions were less for agricultural sources, primarily because the IBI departures before 1990 were less severe (Figure 6). Agricultural nonpoint sources and WWTPs showed greater changes in the AAV, although some rivers with other pollution sources showed the greatest gains.

### *Case Studies*

It is useful to examine three rivers in different natural settings that had differing disturbance and rehabilitation histories, both to better understand historical changes and future possibilities.

*Central Scioto River (Management of Municipal Wastewater Pollution).*—The central Scioto River includes the main stem between Columbus and Circleville, Ohio, a distance of approximately 75 km. The main-stem Scioto River flows through the greater Columbus metropolitan area, with a human population exceeding one million. Wastewater treatment is provided by the Jackson Pike and Southerly WWTPs, with daily capacities of 283,875 and 454,200 $m^3$ of treated wastewater, respectively. These discharges collectively comprise 90–95% of the normal summer–fall flow in the Scioto River

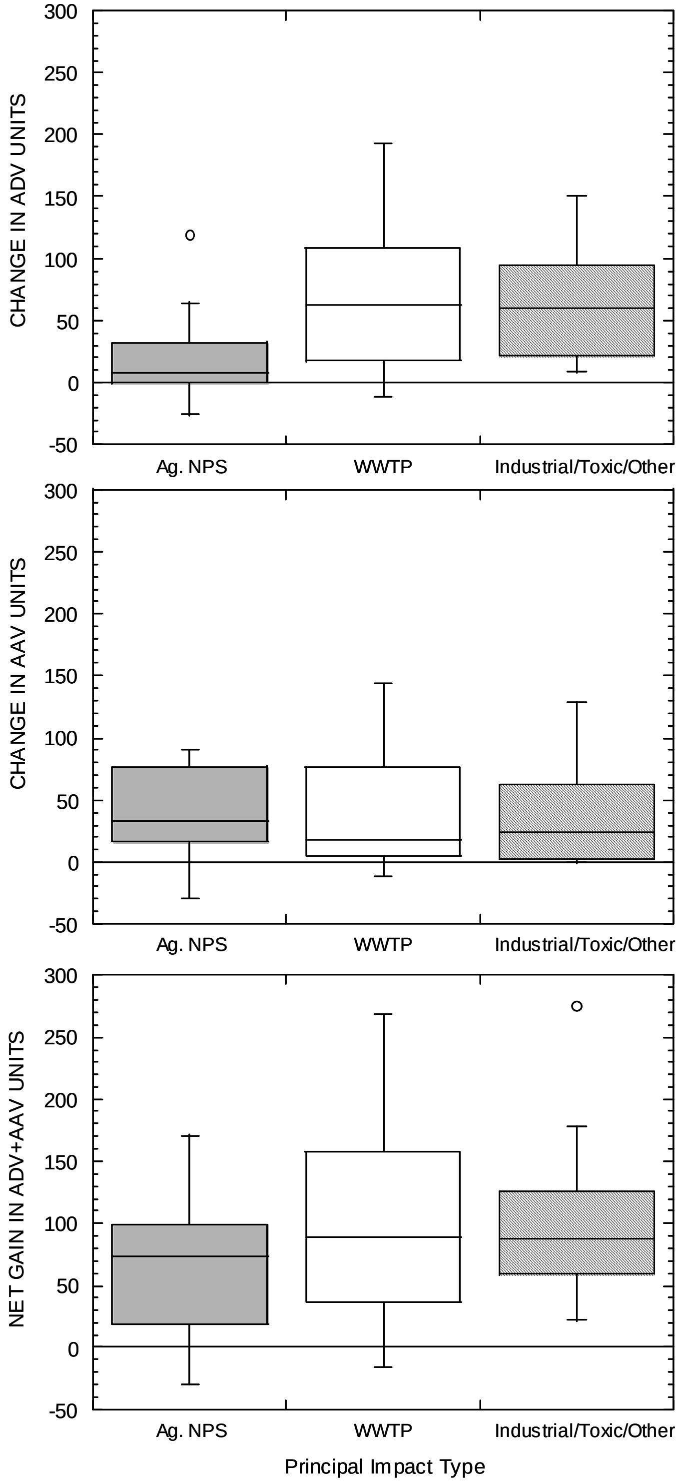



Figure 6. —Changes in area of degradation value (ADV, upper panel) and area of attainment value (AAV, lower panel) between "Before" and "After" periods and aggregated by the three predominant disturbance types.

down river from Columbus. At the turn of the 20th century, the Scioto River was grossly polluted and only six fish species were found (Trautman 1981). By the 1960s, the acute impacts from point sources extended from Columbus to the Ohio River, a distance of 200 km. The landscape and environmental setting is typical of many other municipalities throughout Ohio, which includes an agricultural watershed, an urbanized area including combined sewer overflows, run-of-river impoundments, and WWTPs that dominate the summer–fall flow regime down river from the city.

Fish assemblage data were collected 1979–1981 and annually since 1985 to better understand year-to-year changes and the management of large pollution sources (Figure 7). The ADV reflected incremental changes corresponding to the full suite of pollution controls implemented at the Columbus WWTPs since the early 1980s. The longitudinal IBI results show that improvements occurred through time and represent incremental recovery at most sampling locations impacted by the WWTPs. The 1996 results show most sites above the WWH biocriterion and some sites attaining the EWH biocriterion, indicating that a change in classification of the lower 13 km to EWH is warranted.

The improved fish assemblage corresponded with reduced pollutant loads and lower pollutant concentrations in the river (Figure 8). For example, ammonia-N loads were reduced from 1,000s of kg/d to less than 25–50 kg/d following implementation of advanced wastewater treatment (AWT) in 1988. These improvements in effluent and chemical water quality were followed by improvements in biological quality as indicated by the fish assemblage results (Figure 7). This exemplifies the process outlined in Figure 2 in which management actions and responses were followed by changes in chemical/physical indicators that produced a positive biological response in the receiving environment. Similar results have since been documented in other rivers disturbed by WWTP discharges that have been subjected to the same sequence of pollution controls.

*Upper Great Miami River (Response of a High Quality Resource).*—The upper Great Miami River is located in western Ohio between Indian Lake and Dayton, a distance of 130 km. Land use is predomi-

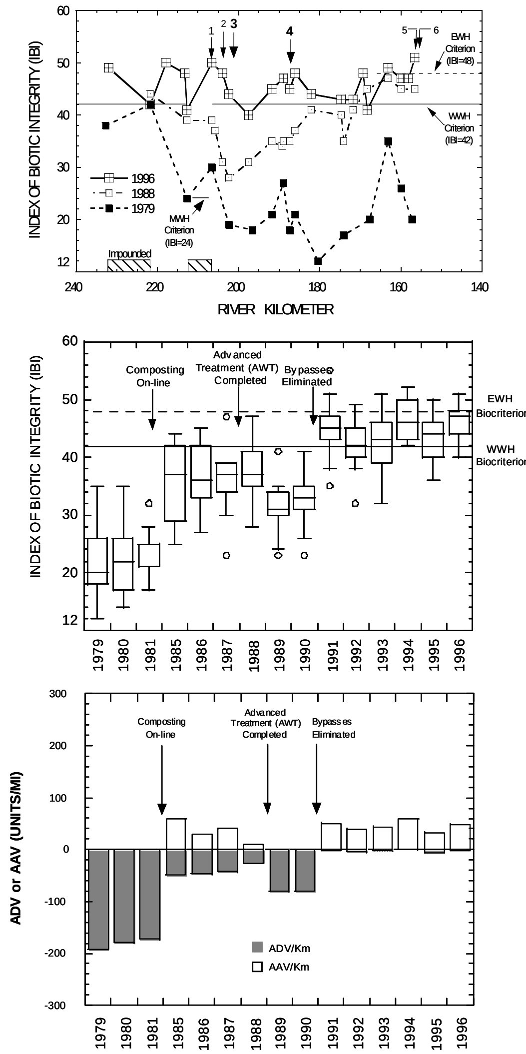



Figure 7.—Longitudinal profile of IBI scores in the central Scioto River main stem in and downstream from Columbus, Ohio in 1979, 1988, and 1996 (upper panel; WWH = Warmwater Habitat; EWH = Exceptional Warmwater Habitat; 1 = Whittier Street Combined Sewer Overflow; 2 = Techneglass; 3 = Jackson Pike Wastewater Treatment Plant; 4 = Columbus Southerly Wastewater Treatment Plant; 5 = Jefferson Smurfitt Corporation; 6 = Circleville Wastewater Treatment Plant). Annual IBI results from the central Scioto River main stem between 1979 and 1996 (middle panel), and ADV and AAV/km during the same period (lower panel). Significant changes in the operation of the Columbus sewage treatment system are noted on each panel.

nantly row crop agriculture, and there are major WWTPs at Sidney, Piqua, and Troy. While none of these dominate the flow of the main stem, each requires water quality-based effluent limitations to meet water quality criteria for common pollutants such as ammonia-N and biochemical oxygen demand. In 1982, the IBI results indicated attainment of the WWH biocriterion for the majority of the

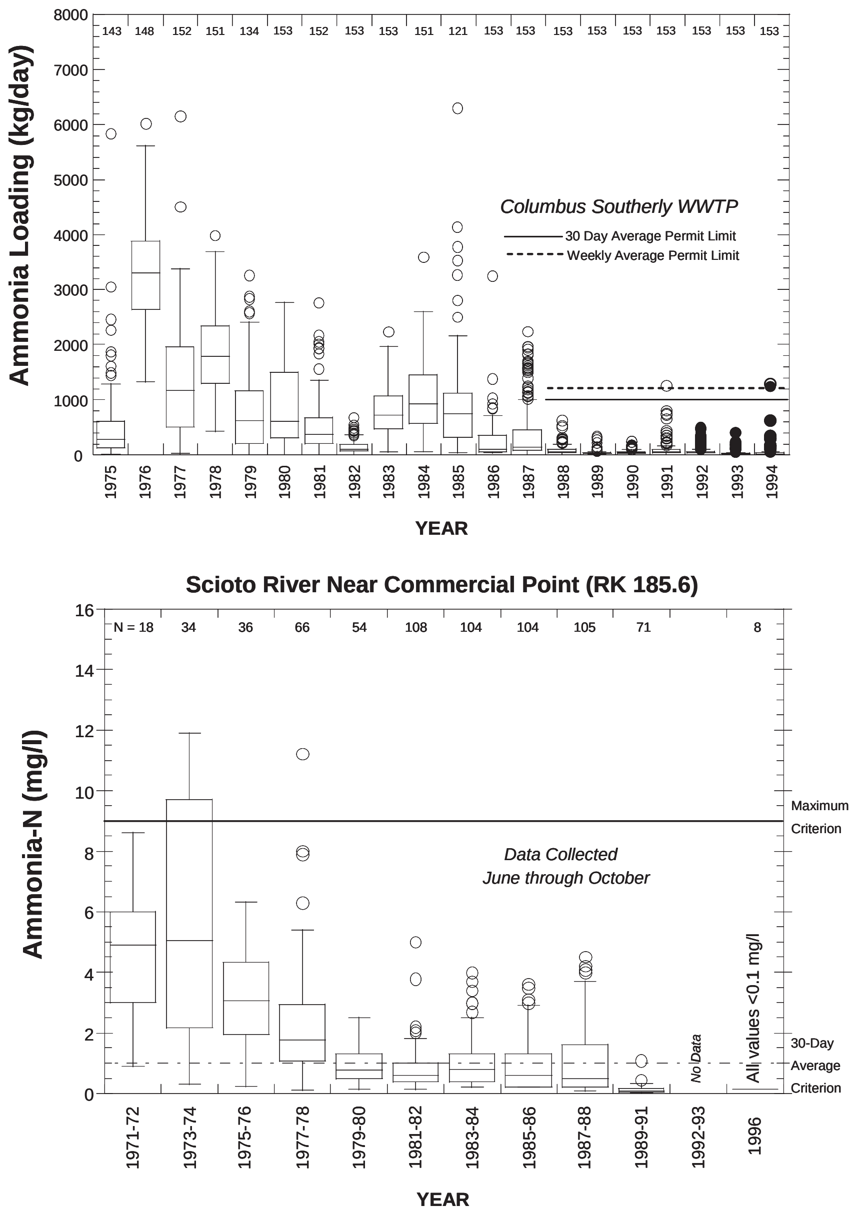



Figure 8.—Box-and-whisker plots of ammonia-N loads (kg/d) discharged annually by the Columbus Southerly WWTP between 1975 and 1994 based on frequent measurements of flow and ammonia-N concentration (mg/L) taken from the final effluent (upper panel). June–October concentrations of ammonia-N (mg/l) measured in the Scioto River 4.6 km (2.9 mi.) downstream from the Columbus Southerly WWTP between 1971 and 1996 (lower panel). Effluent and instream water quality criteria are indicated in each panel, and sample size is indicated at the top of each panel.

river reach (Figure 9). Localized zones of degradation and impairment were observed downstream from the WWTPs, and abatement measures were based on meeting water quality criteria for the WWH classification. The implementation of advanced wastewater treatment common to municipal WWTPs with water quality-based effluent limitations resulted in significant reductions in ammonia-N loads. Loads of greater than 200–300 kg/d were reduced to less than 10–20 kg/d following implementation of advanced wastewater treatment in 1987 and 1988 (Figure 9). This was followed by IBI improvements in 1994 that surpassed the minimum biocriterion for the EWH classification. This resulted in a redesignation of the upper main stem between Sidney and Dayton to EWH, thus increasing the level of protection for one of the highest quality river segments in Ohio. The channelized area in the upper portion of this reach showed insufficient improvement between 1982 and 1994, and this segment remained classified as WWH.

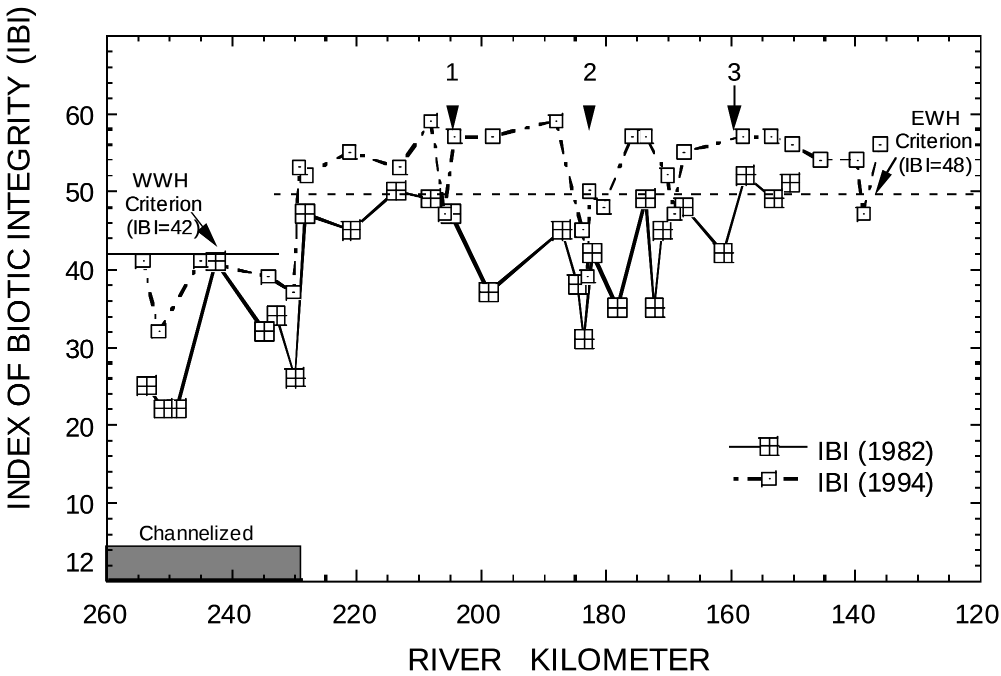
This example is particularly noteworthy in that the implementation of pollution controls resulted in improvements that went beyond the minimum goal of the CWA. Furthermore, the existence of a system of tiered use classifications in the Ohio WQS resulted in a higher level of protection for this reach in accordance with its ecological attributes. Most state WQS consist of single uses that do not include the multiple classification tiers demonstrated here. Such WQS leave high quality waters vulnerable to unintended degradation or lower levels of protection. That would have been especially true in this case since this reach lacks many of the ecological attributes (i.e., rare, threatened, or endangered species) that are usually required to draw special attention in single use systems. The exceptional IBIs were sufficient evidence of an outstanding and exceptional resource that merited a higher level of protection than CWA minima. The problems with single use classifications were highlighted by the NRC (2001) and extensively described by Karr and Yoder (2004).

*Auglaize River (Best Management Practices for Nonpoint Sources).*—The Auglaize River is a major tributary in the Maumee drainage of northwestern Ohio. The sampled nonwadeable segment extends for approximately 64 km from near Wapakoneta to Ft. Jennings and is disturbed primarily by row crop agriculture. Most small tributaries and the headwaters were channelized to enhance surface and subsurface drainage, which makes row cropping sustainable in this glaciated lake plain. The 2000 biological survey revealed significant increases in IBI values in this reach compared to 1985 and 1991 (Figure 10). The increase in the 2000 IBI was not explained by improvements in WWTP effluent quality as this was not a major disturbance in this river. Myers et al. (2000) noted that the implementation of tillage practices that leave crop residues on the soil surface increased markedly in this region in the mid 1980s through the 1990s (Figure 10) and corresponded with reduced concentrations of suspended sediments in the Auglaize River near Ft. Jennings. These reductions in suspended and deposited sediments coincided with higher IBI scores. The metrics most involved in the improved IBI scores (increased proportion of intolerant species, insectivores, simple lithophils and reduced tolerant species and omnivores) are those expected to respond to reduced fine sediments.

The improvement in IBI scores did not adequately communicate the quantity of improvement in the fish assemblage, thus the ADV and AAV were used to better document these changes (Figure 10). Much of the gain made in the Auglaize River fish assemblage was in the extent of attainment (as shown by the AAV), much more so than declines in impairment (shown by the ADV). This would not have been apparent in a conventional focus on simple pass/fail thresholds.

## Discussion

Ohio's nonwadeable rivers have been subjected to multiple impacts from human activities ranging from untreated discharges of human and industrial waste, deforestation, extensive changes in land use, and major hydrological modifications during the past 200 years. These impacts began in the early 19th century and peaked in the early to mid-20th century. Some rivers were so polluted that fish were virtually absent for significant distances (Trautman 1981). The Clean Water Act amendments passed in



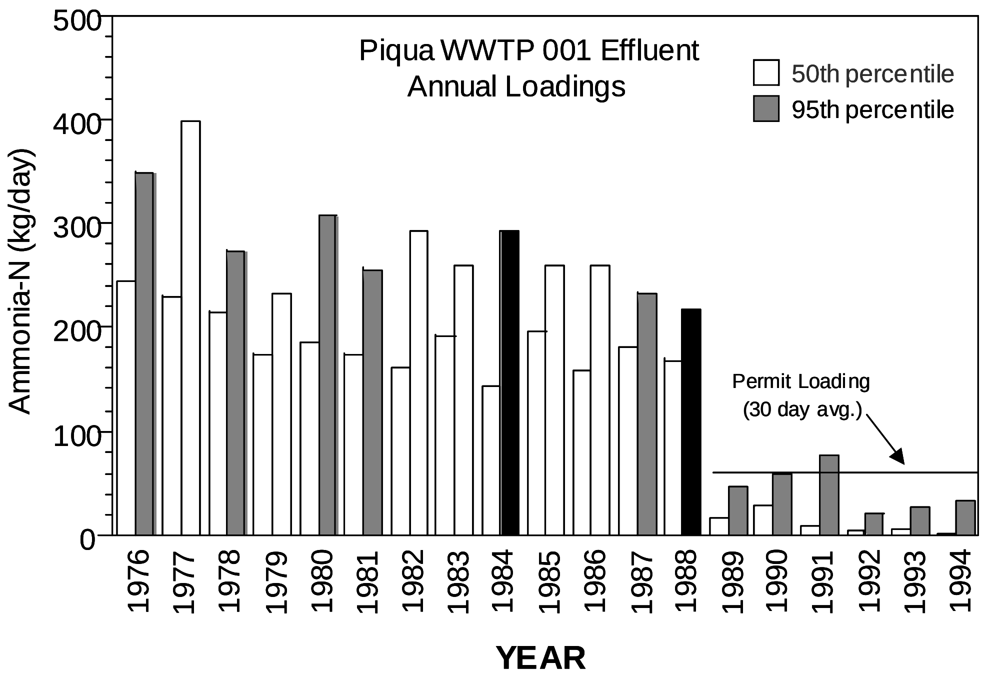




Figure 9. —Longitudinal IBI profile in the upper Great Miami River main stem between Indian Lake and Dayton, Ohio in 1982 and 1994 (upper panel; WWH = Warmwater Habitat; EWH = Exceptional Warmwater Habitat; 1 = Sidney WWTP, 2 = Piqua WWTP, 3 = Troy WWTP). Annual median and 95th percentile loads (kg/d) of ammonia-N discharged by the Piqua WWTP between 1976 and 1994 (lower panel).

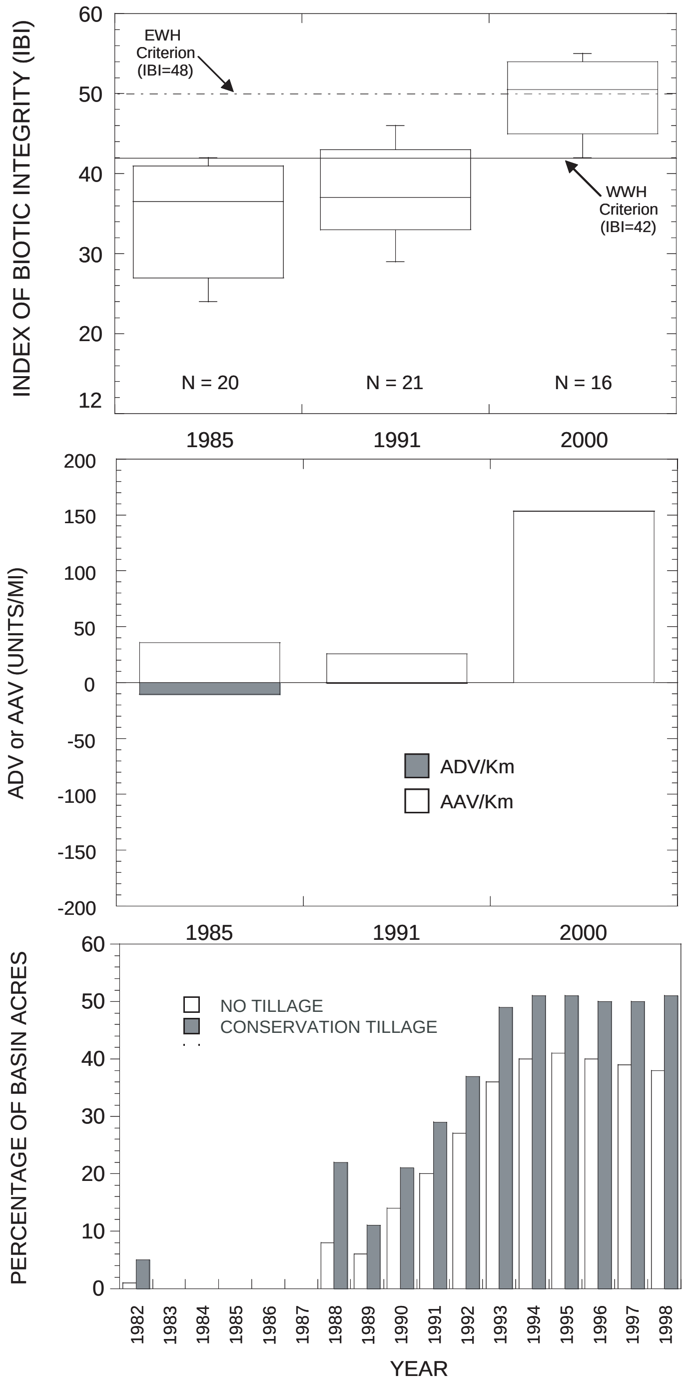



Figure 10. —Box-and-whisker plot of IBI values in the Auglaize River main stem between Wapakoneta and Ft. Jennings, Ohio in 1985, 1991, and 2000 (upper panel; WWH = Warmwater Habitat; EWH = Exceptional Warmwater Habitat; N = number of samples). ADV and AAV/km for the same segment and years (middle panel). Percent of conservation tillage and no till acres in northwestern Ohio between 1982 and 1998 (lower panel; modified from Myers et al. 2000).

1972 required improved wastewater treatment and basin planning, both aimed at reducing the gross impacts of decades of pollution. The Ohio EPA biological monitoring and assessment program was initiated to directly determine the effectiveness of CWA pollution control programs and to develop new and improved tools to better protect and manage water resources. As the gross pollution problems of the preceding decades were better controlled, new and less well understood issues were identified and required new and improved technologies and management practices. In addition, the costs of treatment and management programs made documentation of improvements in ecological and water quality resources more important (GAO 2000). Biological assessment such as that described here became increasingly important to document the ecological outcomes of water quality management programs (Karr and Yoder 2004).

All variables we used portray changes in fish assemblages before and after 1990 and indicated marked improvements in most Ohio rivers. In contrast, the declines observed in some rivers were comparatively small, but nonetheless meaningful. Lack of improvement or smaller than those observed elsewhere indicates remaining or increased pollution. The three case studies clarified the linkages between management programs and biological responses, the values of tiered uses, and the need for considering incremental improvements beyond pass/fail criteria.

We have demonstrated the value and benefits of operating a systematic, standardized approach to assess fish assemblage quality on a statewide basis. Not only is such an approach cost-effective, it is essential for accurate and proportionate assessments of surface water quality and management program effectiveness. By using a tool like the IBI, we have fulfilled one of its important purposes to "monitor biotic integrity at specific sites... screening a large number of sites in order to identify those that require attention, and for assessing trends over time"...and "...to interpret large amounts of data from complex fish communities when the objective is to assess biotic integrity" (Karr et al. 1986).

Sampling Ohio's nonwadeable rivers for more than 20 years has demonstrated improvements in fish assemblage quality that can be related to efforts to manage and improve water quality through CWA driven regulation and policy. In most states, waters are typically assessed for Clean Water Act purposes by comparing chemical, physical, and biological sampling results to chemical and physical criteria in simple pass/fail assessments. The availability of information rich tools like the IBI makes more sophisticated, quantitative, and meaningful assessments possible. When used within an appropriate survey design, sufficient biological data are generated to more quantitatively assess aggregate changes in biological quality and condition over spatially meaningful areas and relate them directly to water pollution abatement efforts. The pass/fail system presently emphasized in various EPA reporting venues (e.g., impaired waters listings, integrated reports) does not sufficiently recognize incremental changes that do not result in full compliance with pass/fail benchmarks and criteria. Nor does it recognize the extent and severity of departures below such benchmarks or by how much they are surpassed. When used with tools like the ADV and AAV, the approach documented here can produce incremental assessments and potentially provide a more integrative method of tracking progress than the conventional approaches that rely on surrogate endpoints such as chemical concentrations and pollutant loadings. Incremental changes not recognized by pass/fail approaches can be quantified and used as feedback for documenting, validating, and improving pollution abatement practices.

The IBI is not satisfactory for all aquatic resource management applications, such as single species management (Lyons et al. 2001). However, in terms of assessing progress towards important water quality management goals (Karr and Yoder 2004), it has performed well and represents an appropriate tool (Yoder and Kulik 2003), particularly where the administrative burden of managing hundreds and thousands of sources over wide geographical areas is required. More detailed assessment of specific sites, sources, and reaches is possible (and without the need to collect extensive new data) within the approach outlined here and is practiced routinely as part of the Ohio EPA integrated assessment program. However, success in this approach requires a robust derivation and calibration database, which requires several years to develop. It is therefore imperative that state biological monitoring programs incorporate quality assurance evaluations of sampling, sample processing, and data analyses; reference condition assessments; and index development as part of baseline efforts. Without these initial investments in data collection and data analyses, a poorly derived and calibrated indicator will result, leading to erroneous assessments.

In our experience the calibrated and modified IBI performed as described by Steedman (1988) as being "based on simple, definable ecological relationships that are quantitative as ordinal, if not linear measures, that respond in an intuitively correct manner to known environmental gradients. Further, when incorporated with mapping, monitoring, and modeling information it is invaluable in determining management and restoration requirements." In Ohio, we have been able to document which pollution control practices have worked and which sources and rivers have seen the most success in terms of attaining the goal of the CWA, the protection and propagation of balanced, indigenous populations of fish, shellfish, and wildlife. The two decades of effort described above provide a baseline for judging future changes and modifying and adapting existing practices in response to these changes.

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Appendix A.— Fish species and their average relative abundance (numbers/km) at boat electrofishing sites draining >390 ha (150 mi.[2]) before (n = 3471 samples) and after 1990 (n = 2176 samples). Species are ranked by post-1990 relative abundance; the percentage of sampling locations at which each occurred is also shown.

| | Before 1990 | | After 1990 | |
|---|---|---|---|---|
| Species | No./km | %Occurrence | No./km | %Occurrence |
| gizzard shad *Dorosoma cepedianum* | 63.9 | 74.0 | 70.8 | 77.8 |
| spotfin shiner *Cyprinella spiloptera* | 33.7 | 71.2 | 48.3 | 80.9 |
| golden redhorse *Moxostoma erythrurum* | 21.9 | 59.7 | 37.5 | 70.2 |
| bluntnose minnow *Pimephales notatus* | 19.9 | 58.1 | 36.2 | 74.8 |
| northern hog sucker *Hypentelium nigricans* | 10.3 | 42.4 | 26.5 | 65.3 |
| common carp* *Cyprinus carpio* | 24.3 | 91.4 | 25.7 | 94.5 |
| bluegill sunfish *Lepomis macrochirus* | 16.2 | 64.7 | 24.3 | 71.2 |
| emerald shiner *Notropis atherinoides* | 13.5 | 23.1 | 23.8 | 37.5 |
| longear sunfish *Lepomis megalotis* | 18.7 | 47.2 | 22.6 | 57.8 |
| white sucker *Catostomus commersonii* | 16.6 | 45.5 | 16.3 | 35.8 |
| smallmouth redhorse *Moxostoma breviceps* | 2.83 | 23.8 | 15.5 | 37.0 |
| smallmouth bass *Micropterus dolomieu* | 8.67 | 50.4 | 15.3 | 67.0 |
| central stoneroller *Campostoma anomalum* | 4.90 | 20.7 | 14.8 | 40.9 |
| green sunfish *Lepomis cyanellus* | 22.7 | 64.2 | 14.2 | 60.8 |
| sand shiner *Notropis stramineus* | 2.97 | 15.7 | 10.2 | 36.8 |
| freshwater drum *Aplodinotus grunniens* | 3.46 | 29.5 | 9.54 | 48.7 |
| black redhorse *Moxostoma duquesnei* | 4.47 | 22.2 | 9.27 | 30.0 |
| channel catfish *Ictalurus punctatus* | 5.23 | 41.9 | 9.24 | 62.7 |
| suckermouth minnow *Phenacobius mirabilis* | 1.27 | 11.1 | 8.50 | 24.2 |
| river carpsucker *Carpiodes carpio* | 4.22 | 24.9 | 7.39 | 31.5 |
| gravel chub *Erimystax x-punctatus* | 0.16 | 2.4 | 6.82 | 19.7 |
| striped shiner *Luxilis chrysocephalus* | 3.92 | 20.0 | 6.26 | 28.7 |
| rock bass *Ambloplites rupestris* | 6.38 | 46.2 | 5.97 | 41.2 |
| greenside darter *Etheostoma blennioides* | 0.91 | 12.7 | 5.27 | 36.2 |
| silver shiner *Notropis photogenis* | 4.45 | 16.1 | 5.05 | 22.2 |
| silver redhorse *Moxostoma anisurum* | 3.45 | 30.6 | 4.98 | 47.9 |
| logperch *Percina caprodes* | 1.31 | 16.1 | 4.78 | 38.7 |
| orangespotted sunfish *Lepomis humilis* | 4.27 | 27.1 | 4.45 | 24.3 |
| largemouth bass *Micropterus salmoides* | 4.81 | 48.1 | 4.42 | 43.6 |
| creek chub *Semotilus atromaculatus* | 4.79 | 17.0 | 4.37 | 14.5 |
| spotted sucker *Minytrema melanops* | 2.32 | 22.8 | 4.28 | 23.1 |
| quillback *Carpiodes cyprinus* | 2.50 | 30.0 | 3.97 | 44.5 |
| steelcolor shiner *Cyprinella whipplei* | 1.29 | 11.0 | 3.96 | 21.6 |
| spotted bass *Micropterus punctulatus* | 3.98 | 20.6 | 3.62 | 30.2 |
| common shiner *Luxilis cornutus* | 0.81 | 3.97 | 3.41 | 3.91 |
| pumpkinseed *Lepomis gibbosus* | 3.76 | 23.2 | 3.39 | 21.0 |
| mottled sculpin *Cottus bairdii* | 0.18 | 1.30 | 3.28 | 4.73 |
| river chub *Nocomis micropogon* | 0.78 | 5.07 | 3.17 | 14.8 |
| bullhead minnow *Pimephales vigilax* | 0.65 | 5.56 | 2.91 | 19.9 |
| smallmouth buffalo *Ictiobus bubalus* | 0.71 | 1.34 | 2.88 | 27.8 |
| banded darter *Etheostoma zonale* | 0.35 | 5.01 | 2.61 | 21.8 |
| white crappie *Pomoxis annularis* | 2.64 | 30.6 | 2.43 | 26.3 |
| golden shiner *Notemigonus crysoleucas* | 2.20 | 18.5 | 2.14 | 11.9 |
| black crappie *Pomoxis nigromaculatus* | 1.27 | 20.2 | 1.81 | 26.6 |
| rainbow darter *Etheostoma caeruleum* | 0.20 | 3.60 | 1.75 | 17.3 |
| rosyface shiner *Notropis rubellus* | 0.66 | 3.63 | 1.58 | 11.5 |
| goldfish* *Carrasius auratus* | 3.84 | 18.1 | 1.49 | 12.1 |
| white bass *Morone chrysops* | 1.34 | 16.5 | 1.44 | 21.4 |

Appendix A.—Continued.

| Species | Before 1990 No./km | Before 1990 %Occurrence | After 1990 No./km | After 1990 %Occurrence |
|---|---|---|---|---|
| yellow bullhead *Ameiurus natalis* | 0.97 | 19.4 | 1.78 | 21.5 |
| blacknose dace *Rhinicthys atratulus* | 0.20 | 1.15 | 1.15 | 4.04 |
| sauger *Sander canadensis* | 0.53 | 12.9 | 1.15 | 21.9 |
| fathead minnow *Pimephales promelas* | 0.89 | 4.47 | 1.02 | 4.91 |
| flathead catfish *Pylodictis olivaris* | 0.57 | 12.6 | 1.02 | 23.7 |
| river redhorse *Moxostoma carinatum* | 0.34 | 6.71 | 0.92 | 13.6 |
| mimic shiner *Notropis volucellus* | 0.24 | 2.45 | 0.89 | 6.71 |
| black buffalo *Ictiobus niger* | 0.08 | 2.25 | 0.84 | 13.1 |
| brook silverside *Labidesthes sicculus* | 0.48 | 6.31 | 0.83 | 11.2 |
| yellow perch *Perca flavescens* | 0.55 | 7.63 | 0.73 | 9.10 |
| white perch *Morone americana* | 0.98 | 5.24 | 0.70 | 3.35 |
| blackside darter *Percina maculata* | 0.31 | 6.91 | 0.68 | 10.3 |
| shorthead redhorse *Moxostoma macrolepidotum* | 0.52 | 4.02 | 0.67 | 3.89 |
| variegate darter *Etheostoma variatum* | 0.06 | 1.01 | 0.66 | 6.00 |
| ghost shiner *Notropis buchanani* | 0.08 | 1.12 | 0.61 | 2.30 |
| stonecat *Noturus flavus* | 0.10 | 2.56 | 0.60 | 13.1 |
| johnny darter *Etheostoma nigrum* | 0.37 | 6.17 | 0.59 | 11.9 |
| spottail shiner *Notropis hudsonius* | 0.28 | 2.62 | 0.59 | 2.67 |
| longnose gar *Lepisosteus osseus* | 0.28 | 8.04 | 0.55 | 14.8 |
| streamline chub *Erimystax dissimilis* | 0.11 | 1.50 | 0.48 | 4.23 |
| redfin shiner *Lythrurus umbratilis* | 0.44 | 3.63 | 0.47 | 4.60 |
| silverjaw minnow *Notropis buccatus* | 0.19 | 2.59 | 0.39 | 2.80 |
| black bullhead *Ameiurus melas* | 0.24 | 4.75 | 0.39 | 3.54 |
| bigmouth buffalo *Ictiobus cyprinellus* | 0.21 | 5.47 | 0.39 | 7.12 |
| skipjack herring *Alosa chrysochloris* | 0.05 | 1.09 | 0.38 | 5.19 |
| bluebreast darter *Etheostoma camurum* | 0.01 | 0.20 | 0.37 | 2.90 |
| highfin carpsucker *Carpiodes velifer* | 0.21 | 5.50 | 0.35 | 6.85 |
| brown bullhead *Ameiurus nebulosus* | 0.71 | 10.9 | 0.32 | 4.41 |
| greater redhorse *Moxostoma valenciennesi* | 0.07 | 0.98 | 0.30 | 3.45 |
| hornyhead chub *Nocomis biguttatus* | 0.05 | 0.63 | 0.30 | 1.33 |
| trout-perch *Percopsis omiscomaycus* | 0.36 | 4.49 | 0.29 | 3.03 |
| warmouth *Lepomis gulosus* | 0.45 | 5.82 | 0.29 | 6.11 |
| dusky darter *Percina sciera* | 0.03 | 0.86 | 0.27 | 4.37 |
| blackstripe topminnow *Fundulus notatus* | 0.16 | 3.26 | 0.24 | 4.43 |
| central mudminnow *Umbra limi* | 0.02 | 0.72 | 0.23 | 0.51 |
| walleye *Sander vitreus* | 0.14 | 3.66 | 0.23 | 5.24 |
| slenderhead darter *Percina phoxocephala* | 0.05 | 0.89 | 0.22 | 5.88 |
| fantail darter *Etheostoma flabellare* | 0.10 | 1.53 | 0.22 | 5.79 |
| grass pickerel *Esox americanus* | 0.42 | 9.39 | 0.20 | 5.15 |
| Tippecanoe darter *Etheostoma tippecanoe* | <0.01 | 0.09 | 0.19 | 2.25 |
| brown trout* *Salmo trutta* | 0.07 | 0.40 | 0.15 | 0.74 |
| redear sunfish *Lepomis microlophus* | 0.02 | 0.54 | 0.14 | 1.93 |
| channel shiner *Notropis wickliffi* | 0 | 0 | 0.14 | 0.60 |
| northern pike *Esox lucius* | 0.12 | 4.12 | 0.13 | 4.46 |
| mooneye *Hiodon tergisus* | 0.05 | 1.47 | 0.12 | 3.26 |
| rainbow trout* *Oncorynchus mykiss* | 0.20 | 0.72 | 0.11 | 1.47 |
| bowfin *Amia calva* | 0.05 | 1.35 | 0.09 | 1.75 |

Appendix A.—Continued.

| Species | Before 1990 No./km | Before 1990 %Occurrence | After 1990 No./km | After 1990 %Occurrence |
|---|---|---|---|---|
| scarlet shiner *Lythrurus fasciolaris* | 0.09 | 1.73 | 0.06 | 1.19 |
| tonguetied minnow *Exoglossum laurae* | <0.01 | 0.14 | 0.05 | 0.65 |
| mountain madtom *Noturus eleutherus* | 0 | 0 | 0.05 | 0.46 |
| silver lamprey *Ichthyomyzon unicuspis* | 0.01 | 0.46 | 0.05 | 2.21 |
| silver chub *Macrhybopsis storeriana* | 0.04 | 0.75 | 0.05 | 1.42 |
| bigeye chub *Hybopsis amblops* | 0.02 | 0.32 | 0.05 | 0.41 |
| muskellunge *Esox masquinongy* | <0.01 | 0.20 | 0.05 | 1.75 |
| orangethroat darter *Etheostoma spectabile* | <0.01 | 0.09 | 0.04 | 1.56 |
| tadpole madtom *Noturus gyrinus* | 0.03 | 0.66 | 0.04 | 0.87 |
| American brook lamprey *Lampetra appendix* | 0.04 | 0.84 | 0.04 | 0.92 |
| brindled madtom *Noturus miurus* | 0.02 | 0.84 | 0.04 | 1.61 |
| eastern sand darter *Ammocrypta pellucida* | 0.01 | 0.20 | 0.03 | 0.74 |
| grass carp* *Ctenopharyngodon idella* | <0.01 | 0.03 | 0.02 | 0.74 |
| popeye shiner *Notropis ariommus* | 0 | 0 | 0.02 | 0.10 |
| blue sucker *Cycleptus elongatus* | <0.01 | 0.03 | 0.02 | 0.60 |
| least brook lamprey *Lampetra aepyptera* | 0.01 | 0.46 | 0.02 | 0.60 |
| river shiner *Notropis blennius* | 0.02 | 0.49 | 0.01 | 0.14 |
| alewife* *Alosa pseudoharengus* | <0.01 | 0.12 | 0.01 | 0.32 |
| threadfin shad *Dorosoma petenense* | 0 | 0 | 0.01 | 0.23 |
| striped bass* *Morone saxatilis* | 0 | 0 | 0.01 | 0.18 |
| northern madtom *Noturus stigmosus* | 0 | 0 | 0.01 | 0.18 |
| redside dace *Clinostomus elongatus* | <0.01 | 0.03 | <0.01 | 0.14 |
| shortnose gar *Lepisosteus platostomus* | <0.01 | 0.12 | <0.01 | 0.18 |
| spotted darter *Etheostoma maculatum* | 0 | 0 | <0.01 | 0.10 |
| paddlefish *Polyodon spathula* | 0 | 0 | <0.01 | 0.10 |
| American eel *Anguilla rostrata* | <0.01 | 0.12 | <0.01 | 0.10 |
| river darter *Percina shumardi* | <0.01 | 0.09 | <0.01 | 0.10 |
| coho salmon* *Oncorhynchus kisutch* | <0.01 | 0.14 | <0.01 | 0.05 |
| round goby* *Neogobius melanstomus* | 0 | 0 | <0.01 | 0.05 |
| white catfish* *Ameirus catus* | 0 | 0 | <0.01 | 0.05 |
| rainbow smelt *Osmerus mordax* | 0 | 0 | <0.01 | 0.05 |
| goldeye *Hiodon alosoides* | <0.01 | 0.06 | <0.01 | 0.05 |
| brook stickleback *Culaea inconstans* | <0.01 | 0.06 | 0 | 0 |
| Iowa darter *Etheostoma exile* | <0.01 | 0.09 | 0 | 0 |
| western mosquitofish *Gambusia affinis* | <0.01 | 0.03 | 0 | 0 |
| banded killifish *Fundulus diaphanus* | 0.2 | 0.43 | 0 | 0 |
| pugnose minnow *Opsopoedus emiliae* | <0.01 | 0.03 | 0 | 0 |
| creek chubsucker *Erimyzon oblongus* | <0.01 | 0.03 | 0 | 0 |
| sea lamprey* *Petromyzon marinus* | <0.01 | 0.06 | 0 | 0 |

* alien to Ohio