Minnesota Pollution Control Agency

Request For Comments on Planned Amendments to Water Quality Standards to Incorporate a Classification System Based on the Application of Tiered Aquatic Life Uses. Minn. Rules Chapters 7050, 7052 and 7053.

NOTICE IS HEREBY GIVEN that the Minnesota Pollution Control Agency (MPCA) requests comments on planned changes to rules governing water quality standards, Minn. R. chapter 7050 (Waters of the State). The changes being considered will add a new framework called Tiered Aquatic Life Uses (TALU). TALU will add a biologic component to the existing standards for Class 2 (aquatic life and recreation) waters. Possible supporting changes to Minn. R. chapters 7052 (Lake Superior Basin), and 7053 (State Waters Discharge Restrictions) are also being considered. Comments should be submitted according to the **Public Comment** section below.

Subject of Rules: Minnesota's surface waters are currently classified according to a number of possible uses (e.g. drinking water, aquatic life, industrial use) and physical and chemical criteria apply according to those uses. Adoption of the TALU framework will not change the existing uses or physical/chemical criteria, but will more accurately categorize rivers and streams based on biological communities. TALU will change the current aquatic life use standards from a "one-size-fits-all" approach to one that protects rivers and streams based on their biological potential. This means:

- High quality or "Exceptional Use" rivers and streams will be given additional protection to maintain the condition of these habitats.
- Categories of "Modified" and "Limited Use" will be added to address some waters that cannot meet aquatic life use goals, either as a result of past practices such as ditching or channelization or because of technological limits.
- Assessment and enforcement of water quality standards will be more effective and transparent.

The changes being considered will include changes to Class 2 use classes and to the biological standards that are necessary to implement the TALU framework, but will not extend to changes to the chemical/physical standards (numeric and narrative) that already apply to Minnesota waters. The MPCA does not intend to change *Minnesota Rules* chapters 7052 or 7053 except as needed to correspond to the TALU changes being made to *Minnesota Rules* chapter 7050 or to make minor changes or corrections found in the course of the rulemaking process.

The state rulemaking process requires agencies to consider several specific topics as it develops rules. The MPCA is specifically asking for comment about the expected economic effect and cumulative impact[1] of the changes being considered (*Minnesota Statutes* §14.131), and also whether a local government may be required to adopt or amend an ordinance or other regulation in response to the changes. (*Minnesota Statutes* § 14.128).

Plain English Summary: This Request for Comments is the MPCA's legal notice of its intent to begin rulemaking. This is the first of several opportunities for public comment and input on this rulemaking. At this stage there is no draft rule available to review; we want your feedback to inform us about the ideas described under the Subject of Rules section above. If you have other ideas or information related to this topic that we need to consider, please provide them. Submitting your ideas and information to us at this early stage in rulemaking allows time to address issues and ensures informed decision-making.

[1] Cumulative impact means the impact that results from incremental impact of the proposed rule in addition to other rules, regardless of what state or federal agency has adopted the other rules.

Where to Get More Information: A web page has been established for this rulemaking. (Tiered Aquatic Life Use (TALU) framework) This page provides an explanation of the TALU system and rulemaking information will be posted in the ***Rulemaking documents*** and ***Schedule/public participation*** tabs. Draft rule language is not available at this time but a concept document that provides information about the changes being considered is available for review in the ***Rulemaking documents*** tab.

If you are interested in being notified when draft rules are available for review and of other activities related to this rulemaking, please register for GovDelivery at: https://public.govdelivery.com/accounts/MNPCA/subscriber/new. The checkbox for the TALU rule is located under the topic heading "Public Notices and Rulemaking".

Persons Affected: The TALU rulemaking potentially affects all persons working with or interested in the state Water Quality Standards, such as persons working in planning, natural resource management, soil and water conservation and environmental protection. Because the TALU system will also result in more accurate classification of waters and more effective assessment and implementation of Water Quality Standards, persons with a general interest in the quality of Minnesota's waters, such as fishing, tourism and agriculture, may also be affected.

Statutory Authority: *Minnesota Statute* § 115.03, subd. 1 grants the MPCA general authority to promulgate and/or revise the rules relating to the pollution of any of the waters of the state; and *Minnesota Statute* § 115.44 grants the MPCA authority to group designated waters of the state into classes.

Public Comment: Interested persons or groups may submit written comments or information on these planned rule amendments from August 25, 2014 until 4:30 p.m. on October 17, 2014. Comments or information should be submitted to Carol Nankivel at the address below. The MPCA does not plan to appoint an advisory committee to comment on the possible rules.

MPCA Contact Person: Written comments and requests for more information on these planned rule amendments should be directed to:

Carol Nankivel
Minnesota Pollution Control Agency
520 Lafayette Road North, St. Paul MN 55155-4194
Telephone: 651-757-2597 or **Toll-free:** 1-800-657-3864
TTY: 651-282-5332
E-mail: carol.nankivel@state.mn.us

Alternative Format: This information is available in an alternative format, such as large print, Braille, or audio through the contact person identified above.

Note: The comments received during this comment period will be considered by the MPCA in the development of the rules. However, if you submit written comments at this time and want the administrative law judge to review your comments, you must resubmit them when proposed rules are published for official comment.

8/14/2014
Date

John Linc Stine, Commissioner
Minnesota Pollution Control Agency

Exhibit 72 is not publicly posted on the MPCA web page due to copyright protection laws. However, the following link is provided for interested parties to access the document in accordance with the respective copyright restrictions. The document may also be available through your local library.

Angermeier P. L. & J. R. Karr. (1986) Applying an index of biotic integrity based on stream-fish communities: considerations in sampling and interpretation. *North American Journal of Fisheries Management* 6: 418-429.

http://www.tandfonline.com/doi/abs/10.1577/1548-8659%281986%296%3C418%3AAAAIOBI%3E2.0.CO%3B2?journalCode=ujfm20

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Karr J. R. & C. O. Yoder. (2004) Biological assessment and criteria improve total maximum daily load decision making. *Journal of Environmental Engineering* 130: 594-604.

http://ascelibrary.org/doi/abs/10.1061/(ASCE)0733-9372(2004)130%3A6(594)

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Lammert M. & J. Allan. (1999) Assessing biotic integrity of streams: effects of scale in measuring the influence of land use/cover and habitat structure on fish and macroinvertebrates. *Environmental Management* 23: 257-270.

http://link.springer.com/article/10.1007/s002679900184

Exhibit 75 is not publicly posted on the MPCA web page due to copyright protection laws. However, the following bibliographic citation is provided so that interested parties may acquire a copy of the document in accordance with the respective copyright restrictions. The document may also be available through your local library.

Bryce S. A. & R. M. Hughes (2002) Variable assemblage responses to multiple disturbance gradients: Oregon and Appalachia, USA, case studies. In: *Biological response signatures: Multimetric index patterns for assessment of freshwater assemblages* (ed T. P. Simon) pp. 539-560. CRC Press, Boca Raton, Florida.

United States
Environmental Protection
Agency

Office of Water
Regulations and Standards (WH-585)
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April 1990

EPA

Biological Criteria

National Program Guidance For Surface Waters

Printed on Recycled Paper

Biological Criteria

National Program Guidance for Surface Waters

Criteria and Standards Division
Office of Water Regulations and Standards
U. S. Environmental Protection Agency
401 M Street S.W.
Washington D.C. 20460

Contents

Acknowledgments

Development of this document required the combined effort of ecologists, biologists, and policy makers from States, EPA Regions, and EPA Headquarters. Initial efforts relied on the 1988 document Report of the National Workshop on Instream Biological Monitoring and Criteria that summarizes a 1987 workshop sponsored by the EPA Office of Water Regulations and Standards, EPA Region V, and EPA Environmental Research Laboratory–Corvallis. In December 1988, contributing and reviewing committees were established (see Appendix D). Members provided reference materials and commented on drafts. Their assistance was most valuable.

Special recognition goes to the Steering Committee who helped develop document goals and made a significant contribution toward the final guidance. Members of the Steering Committee include:

Robert Hughes, Ph.D.	*Chris Yoder*
Susan Davies	*Wayne Davis*
John Maxted	*Jimmie Overton*
James Plafkin, Ph.D.	*Dave Courtemanch*
Phil Larsen, Ph.D.	

Finally, our thanks go to States that recognized the importance of a biological approach in standards and pushed forward independently to incorporate biological criteria into their programs. Their guidance made this effort possible. Development of the program guidance document was sponsored by the U.S. EPA Office of Water Regulations and Standards and developed, in part, through U.S. EPA Contract No. 68-03-3533 to Dynamac Corporation. Thanks to Dr. Mark Southerland for his technical assistance.

Suzanne K. Macy Marcy, Ph.D.
Editor

In Memory of

James L. Plafkin, Ph.D.

Definitions

To effectively use biological criteria, a clear understanding of how these criteria are developed and applied in a water quality standards framework is necessary. This requires, in part, that users of biological criteria start from the same frame of reference. To help form this frame of reference, the following definitions are provided. Please consider them carefully to ensure a consistent interpretation of this document.

Definitions

- An **AQUATIC COMMUNITY** is an association of interacting populations of aquatic organisms in a given waterbody or habitat.

- A **BIOLOGICAL ASSESSMENT** is an evaluation of the biological condition of a waterbody using biological surveys and other direct measurements of resident biota in surface waters.

- **BIOLOGICAL CRITERIA,** or biocriteria, are numerical values or narrative expressions that describe the reference biological integrity of aquatic communities inhabiting waters of a given designated aquatic life use.

- **BIOLOGICAL INTEGRITY** is functionally defined as the condition of the aquatic community inhabiting unimpaired waterbodies of a specified habitat as measured by community structure and function.

- **BIOLOGICAL MONITORING** is the use of a biological entity as a detector and its response as a measure to determine environmental conditions. Toxicity tests and biological surveys are common biomonitoring methods.

- A **BIOLOGICAL SURVEY,** or biosurvey, consists of collecting, processing and analyzing representative portions of a resident aquatic community to determine the community structure and function.

- A **COMMUNITY COMPONENT** is any portion of a biological community. The community component may pertain to the taxomonic group (fish, invertebrates, algae), the taxonomic category (phylum, order, family, genus, species), the feeding strategy (herbivore, omnivore, carnivore) or organizational level (individual, population, community association) of a biological entity within the aquatic community.

- **REGIONS OF ECOLOGICAL SIMILARITY** describe a relatively homogeneous area defined by similarity of climate, landform, soil, potential natural vegetation, hydrology, or other ecologically relevant variable. Regions of ecological similarity help define the potential for designated use classifications of specific waterbodies.

- **DESIGNATED USES** are those uses specified in water quality standards for each waterbody or segment whether or not they are being attained.

- An **IMPACT** is a change in the chemical, physical or biological quality or condition of a waterbody caused by external sources.

- An **IMPAIRMENT** is a detrimental effect on the biological integrity of a waterbody caused by an impact that prevents attainment of the designated use.

- A **POPULATION** is an aggregate of interbreeding individuals of a biological species within a specified location.

- A **WATER QUALITY ASSESSMENT** is an evaluation of the condition of a waterbody using biological surveys, chemical-specific analyses of pollutants in waterbodies, and toxicity tests.

- An **ECOLOGICAL ASSESSMENT** is an evaluation of the condition of a waterbody using water quality and physical habitat assessment methods.

Executive Summary

The Clean Water Act (Act) directs the U.S. Environmental Protection Agency (EPA) to develop programs that will evaluate, restore and maintain the chemical, physical, and biological integrity of the Nation's waters. In response to this directive, States and EPA implemented chemically based water quality programs that successfully addressed significant water pollution problems. However, these programs alone cannot identify or address all surface water pollution problems. To create a more comprehensive program, EPA is setting a new priority for the development of biological water quality criteria. The initial phase of this program directs State adoption of narrative biological criteria as part of State water quality standards. This effort will help States and EPA achieve the objectives of the Clean Water Act set forth in Section 101 and comp y with statutory requirements under Sections 303 and 304. The *Water Quality Standards Regulation* provides additional authority for biological criteria development.

In accordance with priorities established in the *FY 1991 Agency Operating Guidance,* States are to adopt narrative biological criteria into State water quality standards during the FY 1991-1993 triennium. To support this priority, EPA is developing a *Policy on the Use of Biological Assessments and Criteria in the Water Quality Program* and is providing this program guidance document on biological criteria.

This document provides guidance for development and implementation of narrative biological criteria. Future guidance documents will provide additional technical information to facilitate development and implementation of narrative and numeric criteria for each of the surface water types.

When implemented, biological criteria will expand and improve water quality standards programs, help identify impairment of beneficial uses, and help set program priorities. Biological criteria are valuable because they directly measure the condition of the resource at risk, detect problems that other methods may miss or underestimate, and provide a systematic process for measuring progress resulting from the implementation of water quality programs.

Biological criteria require direct measurements of the structure and function of resident aquatic communities to determine biological integrity and ecological function. They supplement, rather than replace chemical and toxicological methods. It is EPA's policy that biological survey methods be fully integrated with toxicity and chemical-specific assessment methods and that chemical-specific criteria, whole-effluent toxicity evaluations and biological criteria be used as independent evaluations of non-attainment of designated uses.

Biological criteria are narrative expressions or numerical values that describe the biological integrity of aquatic communities inhabiting waters of a given aquatic life use. They are developed under the assumptions that surface waters impacted by anthropogenic activities may contain impaired aquatic communities (the greater the impact the greater the expected impairment) and that surface waters not impacted by anthropogenic activities are generally not impaired. Measures of aquatic community structure and function in unimpaired surface waters functionally define biological integrity and form the basis for establishing the biological criteria.

Narrative biological criteria are definable statements of condition or attainable goals for a given use designation. They establish a positive statement about aquatic community characteristics expected to occur within a waterbody (e.g., "Aquatic life shall be as it naturally occurs" or "A natural variety of aquatic life shall be present and all functional groups well represented"). These criteria can be developed using existing information. Numeric criteria describe the expected attainable community attributes and establish values based on measures such as species richness, presence or absence of indicator taxa, and distribution of classes of organisms. To implement narrative criteria and develop numeric criteria, biota in reference waters must be carefully assessed. These are used as the reference values to determine if, and to what extent, an impacted surface waterbody is impaired.

Biological criteria support designated aquatic life use classifications for application in standards. The designated use determines the benefit or purpose to be derived from the waterbody; the criteria provide a measure to determine if the use is impaired. Refinement of State water quality standards to include more detailed language about aquatic life is essential to fully implement a biological criteria program. Data collected from biosurveys can identify consistently distinct characteristics among aquatic communities inhabiting different waters with the same designated use. These biological and ecological characteristics may be used to define separate categories within a designated use, or separate one designated use into two or more use classifications.

To develop values for biological criteria, States should (1) identify unimpaired reference waterbodies to establish the reference condition and (2) characterize the aquatic communities inhabiting reference surface waters. Currently, two principal approaches are used to establish reference sites: (1) the site-specific approach, which may require upstream-downstream or near field-far field evaluations, and (2) the regional approach, which identifies similarities in the physico-chemical characteristics of watersheds that influence aquatic ecology. The basis for choosing reference sites depends on classifying the habitat type and locating unimpaired (minimally impacted) waters.

Once reference sites are selected, their biological integrity must be evaluated using quantifiable biological surveys. The success of the survey will depend in part on the careful selection of aquatic community components (e.g., fish, macroinvertebrates, algae). These components should serve as effective indicators of high biological integrity, represent a range of pollution tolerances, provide predictable, repeatable results, and be readily identified by trained State personnel. Well-planned quality assurance protocols are required to reduce variability in data collection and to assess the natural variability inherent in aquatic communities. A quality survey will include multiple community components and may be measured using a variety of metrics. Since multiple approaches are available, factors to consider when choosing possible approaches for assessing biological integrity are presented in this document and will be further developed in future technical guidance documents.

To apply biological criteria in a water quality standards program, standardized sampling methods and statistical protocols must be used. These procedures must be sensitive enough to identify significant differences between established criteria and tested communities. There are three possible outcomes from hypothesis testing using these analyses: (1) the use is impaired, (2) the biological criteria are met, or (3) the outcome is indeterminate. If the use is impaired, efforts to diagnose the cause(s) will help determine appropriate action. If the use is not impaired, no action is required based on these analyses. The outcome will be indeterminate if the study design or evaluation was incomplete. In this case, States would need to re-evaluate their protocols.

If the designated use is impaired, diagnosis is the next step. During diagnostic evaluations three main impact categories must be considered: chemical, physical, and biological stress. Two questions are posed during initial diagnosis: (1) what are obvious potential causes of impairment, and (2) what possible causes do the biological data suggest? Obvious potential causes of impairment are often identified during normal field biological assessments. When an impaired use cannot be easily related to an obvious cause, the diagnostic process becomes investigative and iterative. Normally the diagnoses of biological impairments are relatively straightforward; States can use biological criteria to confirm impairment from a known source of impact.

There is considerable State interest in integrating biological assessments and criteria in water quality management programs. A minimum of 20 States now use some form of standardized biological assessments to determine the status of biota in State waters. Of these, 15 States are developing biological assessments for future criteria development. Five States use biological criteria to define aquatic life use classifications and to enforce water quality standards. Several States have established narrative biological criteria in their standards. One State has instituted numeric biological criteria.

Whether a State is just beginning to establish narrative biological criteria or is developing a fully integrated biological approach, the programmatic expansion from source control to resource management represents a natural progression in water quality programs. Implementation of biological criteria will provide new options for expanding the scope and application of ecological perspectives.

Part I

Program Elements

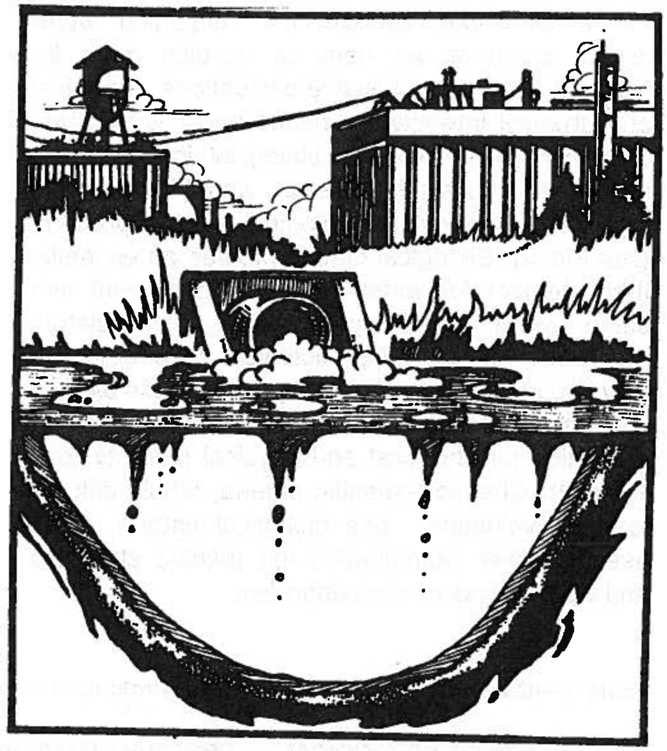
Chapter 1

Introduction

The principal objectives of the Clean Water Act are "to restore and maintain the chemical, physical and biological integrity of the Nation's waters" (Section 101). To achieve these objectives, EPA, States, the regulated community, and the public need comprehensive information about the ecological integrity of aquatic environments. Such information will help us identify waters requiring special protection and those that will benefit most from regulatory efforts.

To meet the objectives of the Act and to comply with statutory requirements under Sections 303 and 304, States are to adopt biological criteria in State standards. The *Water Quality Standards Regulation* provides additional authority for this effort. In accordance with the *FY 1991 Agency Operating Guidance*, States and qualified Indian tribes are to adopt narrative biological criteria into State water quality standards during the FY 1991-1993 triennium. To support this effort, EPA is developing a *Policy on the Use of Biological Assessments and Criteria in the Water Quality Program* and providing this program guidance document on biological criteria.

Like other water quality criteria, biological criteria identify water quality impairments, support regulatory controls that address water quality problems, and assess improvements in water quality from regulatory efforts. Biological criteria are numerical values or narrative expressions that describe the reference biological integrity of aquatic communities inhabiting waters of a given designated aquatic life use. They are developed through the direct measurement of aquatic community components inhabiting unimpaired surface waters.



Anthropogenic impacts, including point source discharges, nonpoint runoff, and habitat degradation continue to impair the nation's surface waters.

Biological criteria complement current programs. Of the three objectives identified in the Act (chemical, physical, and biological integrity), current water quality programs focus on direct measures of chemical integrity (chemical-specific and whole-effluent toxicity) and, to some degree, physical integrity through several conventional criteria (e.g., pH, turbidity, dissolved oxygen). Implementation of these programs has significantly improved water quality. However, as we learn more about aquatic ecosystems it is apparent that other sources of waterbody impairment exist. Biological impairments from diffuse sources and habitat degradation can be greater than those caused by point source discharges (Judy et al. 1987; Miller et al. 1989). In Ohio, evaluation of instream biota indicated that 36 percent of impaired stream segments could not be detected using chemical criteria alone (see Fig. 1). Although effective for their purpose, chemical-specific criteria and whole-effluent toxicity provide only indirect evaluations and protection of biological integrity (see Table 1).

To effectively address our remaining water quality problems we need to develop more integrated and comprehensive evaluations. Chemical and physical integrity are necessary, but not sufficient conditions to attain biological integrity, and only when chemical, physical, and biological integrity are achieved, is ecological integrity possible (see Fig. 2). Biological criteria provide an essential third element for water quality management and serve as a natural progression in regulatory programs. Incorporating biological criteria into a fully integrated program directly protects the biological integrity of surface waters and provides indirect protection for chemical and physical integrity (see Table 2). Chemical-specific criteria, whole-effluent toxicity evaluations, and biological criteria, when used together, complement the relative strengths and weaknesses of each approach.

Figure 1.—Ohio Biosurvey Results Agree with Instream Chemistry or Reveal Unknown Problems

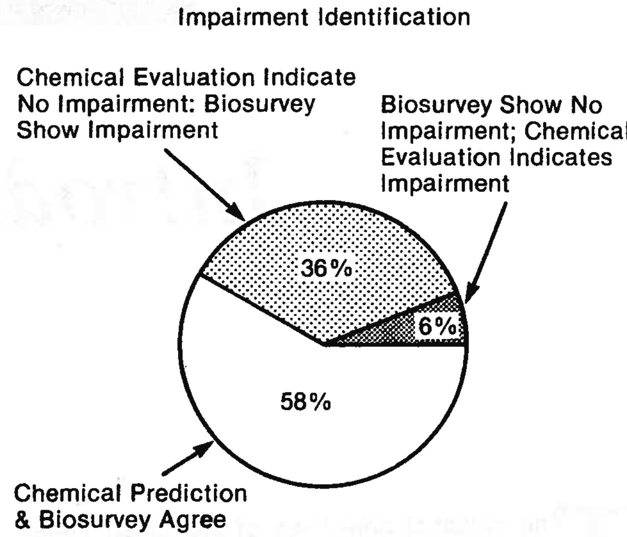


Fig. 1: In an intensive survey, 431 sites in Ohio were assessed using instream chemistry and biological surveys. In 36% of the cases, chemical evaluations implied no impairment but biological survey evaluations showed impairment. In 58% of the cases the chemical and biological assessments agreed. Of these, 17% identified waters with no impairment, 41% identified waters which were considered impaired. (Modified from Ohio EPA Water Quality Inventory, 1988.)

Biological assessments have been used in biomonitoring programs by States for many years. In this respect, biological criteria support earlier work. However, implementing biological criteria in water quality standards provides a systematic, structured, and objective process for making decisions about compliance with water quality standards. This distinguishes biological criteria from earlier use of biological information and increases the value of biological data in regulatory programs.

Table 1.—Current Water Quality Program Protection of the Three Elements of Ecological Integrity.

ELEMENTS OF ECOLOGICAL INTEGRITY	PROGRAM THAT DIRECTLY PROTECTS	PROGRAM THAT INDIRECTLY PROTECTS
Chemical Integrity	Chemical Specific Criteria (toxics) Whole Effluent Toxicity (toxics)	
Physical Integrity	Criteria for Conventionals (pH, DO, turbidity)	
Biological Integrity		Chemical/Whole Effluent Toxicity (biotic response in lab)

Table 1: Current programs focus on chemical specific and whole-effluent toxicity evaluations. Both are valuable approaches for the direct evaluation and protection of chemical integrity. Physical integrity is also directly protected to a limited degree through criteria for conventional pollutants. Biological integrity is only indirectly protected under the assumption that by evaluating toxicity to organisms in laboratory studies, estimates can be made about the toxicity to other organisms inhabiting ambient waters.

Table 2.—Water Quality Programs that Incorporate Biological Criteria to Protect Elements of Ecological Integrity.

ELEMENTS OF ECOLOGICAL INTEGRITY	DIRECTLY PROTECTS	INDIRECTLY PROTECTS
Chemical Integrity	Chemical Specific Criteria (toxics) Whole Effluent Toxicity (toxics)	Biocriteria (identification of impairment)
Physical Integrity	Criteria for conventionals (pH, temp., DO)	Biocriteria (habitat evaluation)
Biological Integrity	Biocriteria (biotic response in surface water)	Chemical/Whole Effluent Testing (biotic response in lab)

Table 2: When biological criteria are incorporated into water quality programs the biological integrity of surface waters may be directly evaluated and protected. Biological criteria also provide additional benefits by requiring an evaluation of physical integrity and providing a monitoring tool to assess the effectiveness of current chemically based criteria.

Figure 2.—The Elements of Ecological Integrity

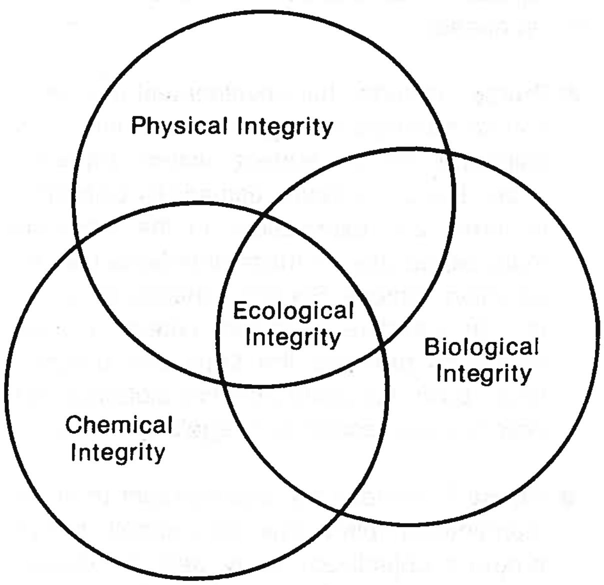


Fig. 2: Ecological Integrity is attainable when chemical, physical, and biological integrity occur simultaneously.

Value of Biological Criteria

Biological criteria provide an effective tool for addressing remaining water quality problems by directing regulatory efforts toward assessing the biological resources at risk from chemical, physical or biological impacts. A primary strength of biological criteria is the detection of water quality problems that other methods may miss or underestimate. Biological criteria can be used to determine to what extent current regulations are protecting the use.

Biological assessments provide integrated evaluations of water quality. They can identify impairments from contamination of the water column and sediments from unknown or unregulated chemicals, non-chemical impacts, and altered physical habitat. Resident biota function as continual monitors of environmental quality, increasing the likelihood of detecting the effects of episodic events (e.g., spills, dumping, treatment plant malfunctions, nutrient enrichment), toxic nonpoint source pollution (e.g., agricultural pesticides), cumulative pollution (i.e., multiple impacts over time or continuous low-level stress), or other impacts that periodic chemical sampling is unlikely to detect. Impacts on the physical habitat such as sedimentation from stormwater runoff and the effects of physical or structural habitat alterations (e.g., dredging, filling, channelization) can also be detected.

Biological criteria require the direct measure of resident aquatic community structure and function to determine biological integrity and ecological function. Using these measures, impairment can be detected and evaluated without knowing the impact(s) that may cause the impairment.

Biological criteria provide a regulatory framework for addressing water quality problems and offer additional benefits, including providing:

- the basis for characterizing high quality waters and identifying habitats and community components requiring special protection under State anti-degradation policies;
- a framework for deciding 319 actions for best control of nonpoint source pollution;
- an evaluation of surface water impairments predicted by chemical analyses, toxicity

testing, and fate and transport modeling (e.g., wasteload allocation);

- improvements in water quality standards (including refinement of use classifications);
- a process for demonstrating improvements in water quality after implementation of pollution controls;
- additional diagnostic tools.

The role of biological criteria as a regulatory tool is being realized in some States (e.g., Arkansas, Maine, Ohio, North Carolina, Vermont). Biological assessments and criteria have been useful for regulatory, resource protection, and monitoring and reporting programs. By incorporating biological criteria in programs, States can improve standards setting and enforcement, measure impairments from permit violations, and refine wasteload allocation models. In addition, the location, extent, and type of biological impairments measured in a waterbody provide valuable information needed for identifying the cause of impairment and determining actions required to improve water quality. Biological assessment and criteria programs provide a cost-effective method for evaluating water quality when a standardized, systematic approach to study design, field methods, and data analysis is established (Ohio EPA 1988a).

Process for Implementation

The implementation of biological criteria will follow the same process used for current chemical-specific and whole-effluent toxicity applications: national guidance produced by U.S. EPA will support States working to establish State standards for the implementation of regulatory programs (see Table 3). Biological criteria differ, however, in the degree of State involvement required. Because surface waters vary significantly from region to region, EPA will provide guidance on acceptable approaches for biological criteria development rather than specific criteria with numerical limitations. States are to establish assessment procedures, conduct field evaluations, and determine criteria values to implement biological criteria in State standards and apply them in regulatory programs.

The degree of State involvement required influences how biological criteria will be implemented. It is expected that States will implement these criteria in phases.

- **Phase I** includes the development and adoption of narrative biological criteria into State standards for all surface waters (streams, rivers, lakes, wetlands, estuaries). Definitions of terms and expressions in the narratives must be included in these standards (see the Narrative Criteria Section, Chapter 3). Adoption of narrative biological criteria in State standards provides the legal and programmatic basis for using ambient biological surveys and assessments in regulatory actions.
- **Phase II** includes the development of an implementation plan. The plan should include program objectives, study design, research protocols, criteria for selecting reference conditions and community components, quality assurance and quality control procedures,

Table 3.—Process for Implementation of Water Quality Standards.

CRITERIA	EPA GUIDANCE	STATE IMPLEMENTATION	STATE APPLICATION
Chemical Specific	Pollutant specific numeric criteria	State Standards · use designation · numeric criteria · antidegradation	Permit limits Monitoring Best Management Practices Wasteload allocation
Narrative Free Forms	Whole effluent toxicity guidance	Water Quality Narrative · no toxic amounts translator	Permit limits Monitoring Wasteload allocation Best Management Practices
Biological	Biosurvey minimum requirement guidance	State Standards · refined use · narrative/numeric criteria · antidegradation	Permit conditions Monitoring Best Management Practices Wasteload allocation

Table 3: Similar to chemical specific criteria and whole effluent toxicity evaluations, EPA is providing guidance to States for the adoption of biological criteria into State standards to regulate sources of water quality impairment.

and training for State personnel. In Phase II, States are to develop plans necessary to implement biological criteria for each surface water type.

- **Phase III** requires full implementation and integration of biological criteria in water quality standards. This requires using biological surveys to derive biological criteria for classes of surface waters and designated uses. These criteria are then used to identify nonattainment of designated uses and make regulatory decisions.

Narrative biological criteria can be developed for all five surface water classifications with little or no data collection. Application of narrative criteria in seriously degraded waters is possible in the short term. However, because of the diversity of surface waters and the biota that inhabit these waters, significant planning, data collection, and evaluation will be needed to fully implement the program. Criteria for each type of surface water are likely to be developed at different rates. The order and rate of development will depend, in part, on the development of EPA guidance for specific types of surface water. Biological criteria technical guidance for streams will be produced during FY 1991. The tentative order for future technical guidance documents includes guidance for rivers (FY 1992), lakes (FY 1993), wetlands (FY 1994) and estuaries (FY 1995). This order and timeline for guidance does not reflect the relative importance of these surface waters, but rather indicates the relative availability of research and the anticipated difficulty of developing guidance.

Independent Application of Biological Criteria

Biological criteria supplement, but do not replace, chemical and toxicological methods. Water chemistry methods are necessary to predict risks (particularly to human health and wildlife), and to diagnose, model, and regulate important water quality problems. Because biological criteria are able to detect different types of water quality impairments and, in particular, have different levels of sensitivity for detecting certain types of impairment compared to toxicological methods, they are not used in lieu of, or in conflict with, current regulatory efforts.

As with all criteria, certain limitations to biological criteria make independent application essential. Study design and use influences how sensitive biological criteria are for detecting community impairment. Several factors influence sensitivity: (1) State decisions about what is significantly different between reference and test communities, (2) study design, which may include community components that are not sensitive to the impact causing impairment, (3) high natural variability that makes it difficult to detect real differences, and (4) types of impacts that may be detectable sooner by other methods (e.g., chemical criteria may provide earlier indications of impairment from a bioaccumulative chemical because aquatic communities require exposure over time to incur the full effect).

Since each type of criteria (biological criteria, chemical-specific criteria, or whole-effluent toxicity evaluations) has different sensitivities and purposes, a criterion may fail to detect real impairments when used alone. As a result, these methods should be used together in an integrated water quality assessment, each providing an independent evaluation of nonattainment of a designated use. If any one type of criteria indicates impairment of the surface water, regulatory action can be taken to improve water quality. However, no one type of criteria can be used to confirm attainment of a use if another form of criteria indicates nonattainment (see Hypothesis Testing: Biological Criteria and the Scientific Method, Chapter 7). When these three methods are used together, they provide a powerful, integrated, and effective foundation for waterbody management and regulations.

How to Use this Document

The purpose of this document is to provide EPA Regions, States and others with the conceptual framework and assistance necessary to develop and implement narrative and numeric biological criteria and to promote national consistency in application. There are two main parts of the document. Part One (Chapters 1, 2, 3, and 4) includes the essential concepts about what biological criteria are and how they are used in regulatory programs. Part Two (Chapters 5, 6, and 7) provides an overview of the process that is essential for implementing a State biological criteria program. Specific chapters include the following:

Part I: PROGRAM ELEMENTS

- **Chapter 2, Legal Authority,** reviews the legal basis for biological criteria under the Clean Water Act and includes possible applications under the Act and other legislation.
- **Chapter 3, Conceptual Framework,** discusses the essential program elements for biological criteria, including what they are and how they are developed and used within a regulatory program. The development of narrative biological criteria is discussed in this chapter.
- **Chapter 4, Integration,** discusses the use of biological criteria in regulatory programs.

Part II: THE IMPLEMENTATION PROCESS

- **Chapter 5, The Reference Condition,** provides a discussion on alternative forms of reference conditions that may be developed by a State based on circumstances and needs.
- **Chapter 6, The Biological Survey,** provides some detail on the elements of a quality biological survey.
- **Chapter 7, Hypothesis Testing: Biological Criteria and the Scientific Method,** discusses how biological surveys are used to make regulatory and diagnostic decisions.
- **Appendix A** includes commonly asked questions and their answers about biological criteria.

Two additional documents are planned in the near term to supplement this program guidance document.

1. "*Biological Criteria Technical Reference Guide*" will contain a cross reference of technical papers on available approaches and methods for developing biological criteria (see tentative table of contents in Appendix B),
2. "*Biological Criteria Development by States*" will provide a summary of different mechanisms several States have used to implement and apply biological criteria in water quality programs (see tentative outline in Appendix C).

Both documents are planned for FY 1991. As previously discussed, over the next triennium technical guidance for specific systems (e.g., streams, wetlands) will be developed to provide guidance on acceptable biological assessment procedures to further support State implementation of comprehensive programs.

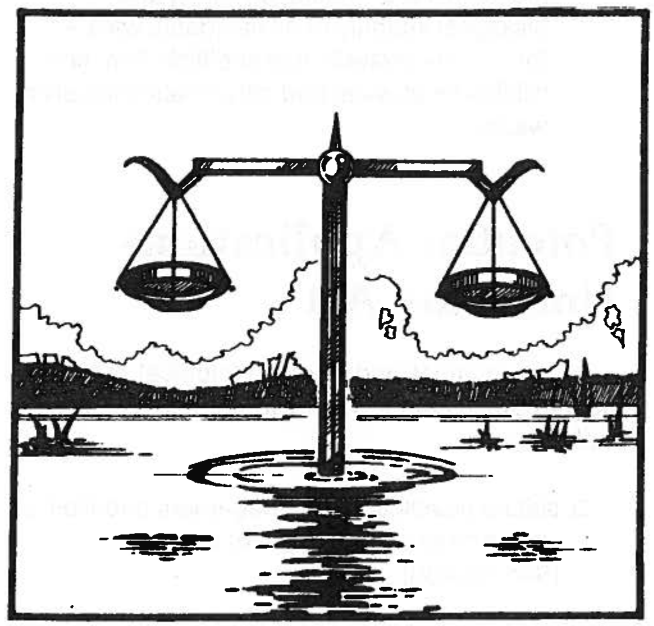
This biological criteria program guidance document supports development and implementation of biological criteria by providing guidance to States working to comply with requirements under the Clean Water Act and the Water Quality Standards Regulation. This guidance is not regulatory.

Chapter 2

Legal Authority

The Clean Water Act (Federal Water Pollution Control Act of 1972, Clean Water Act of 1977, and the Water Quality Act of 1987) mandates State development of criteria based on biological assessments of natural ecosystems.

The general authority for biological criteria comes from Section 101(a) of the Act which establishes as the objective of the Act the restoration and maintenance of the chemical, physical, and biological integrity of the Nation's waters. To meet this objective, water quality criteria must include criteria to protect biological integrity. Section 101(a)(2) includes the interim water quality goal for the protection and propagation of fish, shellfish, and wildlife. Propagation includes the full range of biological conditions necessary to support reproducing populations of all forms of aquatic life and other life that depend on aquatic systems. Sections 303 and 304 provide specific directives for the development of biological criteria.



Balancing the legal authority for biological criteria.

Section 303

Under Section 303(c) of the Act, States are required to adopt protective water quality standards that consist of uses, criteria, and antidegradation. States are to review these standards every three years and to revise them as needed.

Section 303(c)(2)(A) requires the adoption of water quality standards that "... serve the purposes of the Act," as given in Section 101. Section 303(c)(2)(B), enacted in 1987, requires States to adopt numeric criteria for toxic pollutants for which EPA has published 304(a)(1) criteria. The section further requires that, where numeric 304(a) criteria are not available, States should adopt criteria based on biological assessment and monitoring methods, consistent with information published by EPA under 304(a)(8).

These specific directives do not serve to restrict the use of biological criteria in other settings where they may be helpful. Accordingly, this guidance document provides assistance in implementing various sections of the Act, not just 303(c)(2)(B).

Section 304

Section 304(a) directs EPA to develop and publish water quality criteria and information on methods for measuring water quality and establishing water quality criteria for toxic pollutants on bases other than pollutant-by-pollutant, including biological monitoring and assessment methods which assess:

- the effects of pollutants on aquatic community components (". . . plankton, fish, shellfish, wildlife, plant life . . .") and community attributes (". . . biological community diversity, productivity, and stability . . ."); in any body of water and;
- factors necessary ". . . to restore and maintain the chemical, physical, and biological integrity of all navigable waters . . ." for ". . . the protection of shellfish, fish, and wildlife for classes and categories of receiving waters . . ."

Potential Applications Under the Act

Development and use of biological criteria will help States to meet other requirements of the Act, including:

- ❑ setting planning and management priorities for waterbodies most in need of controls [Sec. 303(d)];
- ❑ determining impacts from nonpoint sources [i.e., Section 304(f) "(1) guidelines for identifying and evaluating the nature and extent of nonpoint sources of pollutants, and (2) processes, procedures, and methods to control pollution . . ."].
- ❑ biennial reports on the extent to which waters support balanced biological communities [Sec. 305(b)];
- ❑ assessment of lake trophic status and trends [Sec. 314];
- ❑ lists of waters that cannot attain designated uses without nonpoint source controls [Sec. 319];
- ❑ development of management plans and conducting monitoring in estuaries of national significance [Sec. 320];
- ❑ issuing permits for ocean discharges and monitoring ecological effects [Sec. 403(c) and 301(h)(3)];
- ❑ determination of acceptable sites for disposal of dredge and fill material [Sec. 404];

Potential Applications Under Other Legislation

Several legislative acts require an assessment of risk to the environment (including resident aquatic communities) to determine the need for regulatory action. Biological criteria can be used in this context to support EPA assessments under:

- ❑ *Toxic Substances Control Act (TSCA) of 1976*
- ❑ *Resource Conservation and Recovery Act (RCRA),*
- ❑ *Comprehensive Environmental Response, Compensation and Liability Act of 1980 (CERCLA),*
- ❑ *Superfund Amendments and Reauthorization Act of 1986 (SARA),*
- ❑ *Federal Insecticide, Fungicide, and Rodenticide Act (FIFRA);*
- ❑ *National Environmental Policy Act (NEPA);*
- ❑ *Federal Lands Policy and Management Act (FLPMA).*
- ❑ *The Fish and Wildlife Conservation Act of 1980*
- ❑ *Marine Protection, Research, and Sanctuaries Act*
- ❑ *Coastal Zone Management Act*

- *Wild and Scenic Rivers Act*
- *Fish and Wildlife Coordination Act, as Amended in 1965*

A summary of the applicability of these Acts for assessing ecological impairments may be found in *Risk Assessment Guidance for Superfund-Environmental Evaluation Manual (Interim Final) 1989.*

Other federal and State agencies can also benefit from using biological criteria to evaluate the biological integrity of surface waters within their jurisdiction and to the effects of specific practices on surface water quality. Agencies that could benefit include:

- **Department of the Interior** (*U.S. Fish and Wildlife Service, U.S. Geological Survey, Bureau of Mines, and Bureau of Reclamation, Bureau of Indian Affairs, Bureau of Land Management, and National Park Service*),
- **Department of Commerce** (*National Oceanic and Atmospheric Administration, National Marine Fisheries Service*),
- **Department of Transportation** (*Federal Highway Administration*)
- **Department of Agriculture** (*U.S. Forest Service, Soil Conservation Service)*
- **Department of Defense,**
- **Department of Energy,**
- **Army Corps of Engineers,**
- **Tennessee Valley Authority.**

Chapter 3

The Conceptual Framework

Biological integrity and the determination of use impairment through assessment of ambient biological communities form the foundation for biological criteria development. The effectiveness of a biological criteria program will depend on the development of quality criteria, the refinement of use classes to support narrative criteria, and careful application of scientific principles.

Premise for Biological Criteria

Biological criteria are based on the premise that the structure and function of an aquatic biological community within a specific habitat provide critical information about the quality of surface waters. Existing aquatic communities in pristine environments not subject to anthropogenic impact exemplify biological integrity and serve as the best possible goal for water quality. Although pristine environments are virtually non-existent (even remote waters are impacted by air pollution), minimally impacted waters exist. Measures of the structure and function of aquatic communities inhabiting unimpaired (minimally impacted) waters provide the basis for establishing a reference condition that may be compared to the condition of impacted surface waters to determine impairment.

Based on this premise, biological criteria are developed under the assumptions that: (1) surface waters subject to anthropogenic disturbance may contain impaired populations or communities of aquatic organisms—the greater the anthropogenic

Aquatic communities assessed in unimpaired waterbodies (top) provide a reference for evaluating impairments in the same or similar waterbodies suffering from increasing anthropogenic impacts (bottom).

disturbance, the greater the likelihood and magnitude of impairment; and (2) surface waters not subject to anthropogenic disturbance generally contain unimpaired (natural) populations and communities of aquatic organisms exhibiting biological integrity.

Biological Integrity

The expression "biological integrity" is used in the Clean Water Act to define the Nation's objectives for water quality. According to Webster's New World Dictionary (1966), integrity is, "the quality or state of being complete; unimpaired." Biological integrity has been defined as "the ability of an aquatic ecosystem to support and maintain a balanced, integrated, adaptive community of organisms having a species composition, diversity, and functional organization comparable to that of the natural habitats within a region" (Karr and Dudley 1981). For the purposes of biological criteria, these concepts are combined to develop a functional definition for evaluating biological integrity in water quality programs. Thus, biological integrity is functionally defined as:

> *the condition of the aquatic community inhabiting the unimpaired waterbodies of a specified habitat as measured by community structure and function.*

It will often be difficult to find unimpaired waters to define biological integrity and establish the reference condition. However, the structure and function of aquatic communities of high quality waters can be approximated in several ways. One is to characterize aquatic communities in the most protected waters representative of the regions where such sites exist. In areas where few or no unimpaired sites are available, characterization of least impaired systems approximates unimpaired systems. Concurrent analysis of historical records should supplement descriptions of the condition of least impaired systems. For some systems, such as lakes, evaluating paleoecological information (the record stored in sediment profiles) can provide a measure of less disturbed conditions.

Surface waters, when inhabited by aquatic communities, are exhibiting a degree of biological integrity. However, the best representation of biological integrity for a surface water should form the basis for establishing water quality goals for those waters. When tied to the development of biological criteria, the realities of limitations on biological integrity can be considered and incorporated into a progressive program to improve water quality.

Biological Criteria

Biological criteria are narrative expressions or numerical values that describe the biological integrity of aquatic communities inhabiting waters of a given designated aquatic life use. While biological integrity describes the ultimate goal for water quality, biological criteria are based on aquatic community structure and function for waters within a variety of designated uses. Designated aquatic life uses serve as general statements of attained or attainable uses of State waters. Once established for a designated use, biological criteria are quantifiable values used to determine whether a use is impaired, and if so, the level of impairment. This is done by specifying what aquatic community structure and function should exist in waters of a given designated use, and then comparing this condition with the condition of a site under evaluation. If the existing aquatic community measures fail to meet the criteria, the use is considered impaired.

Since biological surveys used for biological criteria are capable of detecting water quality problems (use impairments) that may not be detected by chemical or toxicity testing, violation of biological criteria is sufficient cause for States to initiate regulatory action. Corroborating chemical and toxicity testing data are not required (though they may be desirable) as supporting evidence to sustain a determination of use impairment. However, a finding that biological criteria fail to indicate use impairment does not mean the use is automatically attained. Other evidence, such as violation of physical or chemical criteria, or results from toxicity tests, can also be used to identify impairment. Alternative forms of criteria provide independent assessments of nonattainment.

As stated above, biological criteria may be narrative statements or numerical values. States can establish general narrative biological criteria early in program development without conducting biological assessments. Once established in State standards, narrative biological criteria form the legal and programmatic basis for expanding biological assessment and biosurvey programs needed to implement narrative criteria and develop numeric biological criteria. Narrative biological criteria should become part of State regulations and standards.

Narrative Criteria

Narrative biological criteria are general statements of attainable or attained conditions of biological integrity and water quality for a given use designation. Although similar to the "free from" chemical water quality criteria, narrative biological criteria establish a positive statement about what should occur within a water body. Narrative criteria can take a number of forms but they must contain several attributes to support the goals of the Clean Water Act to provide for the protection and propagation of fish, shellfish, and wildlife. Thus, narrative criteria should include specific language about aquatic community characteristics that (1) must exist in a waterbody to meet a particular designated aquatic life use, and (2) are quantifiable. They must be written to protect the use. Supporting statements for the criteria should promote water quality to protect the most natural community possible for the designated use. Mechanisms should be established in the standard to address potentially conflicting multiple uses. Narratives should be written to protect the most sensitive use and support antidegradation.

Several States currently use narrative criteria. In Maine, for example, narrative criteria were established for four classes of water quality for streams and rivers (see Table 4). The classifications were based on the range of goals in the Act from "no discharge" to "protection and propagation of fish, shellfish, and wildlife" (Courtemanch and Davies 1987). Maine separated its "high quality water" into two categories, one that reflects the highest goal of the Act (no discharge, Class AA) and one that reflects high integrity but is minimally impacted by human activity (Class A). The statement "The aquatic life . . . shall be as naturally occurs" is a narrative biological criterion for both Class AA and A waters. Waters in Class B meet the use when the life stages of all indigenous aquatic species are supported and no detrimental changes occur in community composition (Maine DEP 1986). These criteria directly support refined designated aquatic life uses (see Section D, Refining Aquatic Life Use Classifications).

These narrative criteria are effective only if, as Maine has done, simple phrases such as "as naturally occurs" and "nondetrimental" are clearly operationally defined. Rules for sampling procedures and data analysis and interpretation should become part of the regulation or supporting documentation. Maine was able to develop these criteria and their supporting statements using avail-

Table 4.—Aquatic Life Classification Scheme for Maine's Rivers and Streams.

RIVERS AND STREAMS	MANAGEMENT PERSPECTIVE	LEVEL OF BIOLOGICAL INTEGRITY
Class AA	High quality water for preservation of recreational and ecological interests. No discharges of any kind permitted. No impoundment permitted.	Aquatic life shall be as naturally occurs.
Class A	High quality water with limited human interference. Discharges restricted to noncontact process water or highly treated wastewater of quality equal to or better than the receiving water. Impoundment permitted.	Aquatic life shall be as naturally occurs.
Class B	Good quality water. Discharges of well treated effluents with ample dilution permitted.	Ambient water quality sufficient to support life stages of all indigenous aquatic species. Only nondetrimental changes in community composition may occur.
Class C	Lowest quality water. Requirements consistent with interim goals of the federal Water Quality Law (fishable and swimmable).	Ambient water quality sufficient to support the life stages of all indigenous fish species. Changes in species composition may occur but structure and function of the aquatic community must be maintained.

able data from water quality programs. To implement the criteria, aquatic life inhabiting unimpaired waters must be measured to quantify the criteria statement.

Narrative criteria can take more specific forms than illustrated in the Maine example. Narrative criteria may include specific classes and species of organisms that will occur in waters for a given designated use. To develop these narratives, field evaluations of reference conditions are necessary to identify biological community attributes that differ significantly between designated uses. For example in the Arkansas use class <u>Typical Gulf Coastal Ecoregion</u> (i.e., South Central Plains) the narrative criterion reads:

> *"Streams supporting diverse communities of indigenous or adapted species of fish and other forms of aquatic life. Fish communities are characterized by a limited proportion of sensitive species; sunfishes are distinctly dominant, followed by darters and minnows. The community may be generally characterized by the following fishes: Key Species—Redfin shiner, Spotted sucker, Yellow bullhead, Flier, Slough darter, Grass pickerel; Indicator Species—Pirate perch, Warmouth, Spotted sunfish, Dusky darter, Creek chubsucker, Banded pygmy sunfish (Arkansas DPCE 1988).*

In Connecticut, current designated uses are supported by narratives in the standard. For example, under Surface Water Classifications, Inland Surface Waters Class AA, the Designated Use is: "Existing or proposed drinking water supply; fish and wildlife habitat; recreational use; agricultural, industrial supply, and other purposes (recreation uses may be restricted)."

The supporting narratives include:

> *Benthic invertebrates which inhabit lotic waters: A wide variety of macroinvertebrate taxa should normally be present and all functional groups should normally be well represented . . . Water quality shall be sufficient to sustain a diverse macroinvertebrate community of indigenous species. Taxa within the Orders Plecoptera (stoneflies), Ephemeroptera (mayflies), Coleoptera (beetles), Tricoptera (caddisflies) should be well represented (Connecticut DEP 1987).*

For these narratives to be effective in a biological criteria program expressions such as "a wide variety" and "functional groups should normally be well represented" require quantifiable definitions that become part of the standard or supporting documentation. Many States may find such narratives in their standards already. If so, States should evaluate current language to determine if it meets the requirements of quantifiable narrative criteria that support refined aquatic life uses.

Narrative biological criteria are similar to the traditional narrative "free froms" by providing the legal basis for standards applications. A sixth "free from" could be incorporated into standards to help support narrative biological criteria such as "free from activities that would impair the aquatic community as it naturally occurs." Narrative biological criteria can be used immediately to address obvious existing problems.

Numeric Criteria

Numerical indices that serve as biological criteria should describe expected attainable community attributes for different designated uses. It is important to note that full implementation of narrative criteria will require similar data as that needed for developing numeric criteria. At this time, States may or may not choose to establish numeric criteria but may find it an effective tool for regulatory use.

To derive a numeric criterion, an aquatic community's structure and function is measured at reference sites and set as a reference condition. Examples of relative measures include similarity indices, coefficients of community loss, and comparisons of lists of dominant taxa. Measures of existing community structure such as species richness, presence or absence of indicator taxa, and distribution of trophic feeding groups are useful for establishing the normal range of community components to be expected in unimpaired systems. For example, Ohio uses criteria for the warmwater habitat use class based on multiple measures in different reference sites within the same ecoregion. Criteria are set as the 25th percentile of all biological index scores recorded at established reference sites within the ecoregion. Exceptional warmwater habitat index criteria are set at the 75th percentile (Ohio EPA 1988a). Applications such as this require an extensive data base and multiple reference sites for each criteria value.

To develop numeric biological criteria, careful assessments of biota in reference sites must be conducted (Hughes et al. 1986). There are numerous ways to assess community structure and function in surface waters. No single index or measure is universally recognized as free from bias. It is important to evaluate the strengths and weaknesses of different assessment approaches. A multimetric approach that incorporates information on species richness, trophic composition, abundance or biomass, and organism condition is recommended. Evaluations that measure multiple components of communities are also recommended because they tend to be more reliable (e.g., measures of fish and macroinvertebrates combined will provide more information than measures of fish communities alone). The weaknesses of one measure or index can often be compensated by combining it with the strengths of other community measurements.

The particular indices used to develop numeric criteria depend on the type of surface waters (streams, rivers, lakes, Great Lakes, estuaries, wetlands, and nearshore marine) to which they must be applied. In general, community-level indices such as the Index of Biotic Integrity developed for midwestern streams (Karr et al. 1986) are more easily interpreted and less variable than fluctuating numbers such as population size. Future EPA technical guidance documents will include evaluations of the effectiveness of different biological survey and assessment approaches for measuring the biological integrity of surface water types and provide guidance on acceptable approaches for biological criteria development.

Refining Aquatic Life Use Classifications

State standards consist of (1) designated aquatic life uses, (2) criteria sufficient to protect the designated and existing use, and (3) an antidegradation clause. Biological criteria support designated aquatic life use classifications for application in State standards. Each State develops its own designated use classification system based on the generic uses cited in the Act (e.g., protection and propagation of fish, shellfish, and wildlife). Designated uses are intentionally general. However, States may develop subcategories within use designations to refine and clarify the use class. Clarification of the use class is particularly helpful when a variety of surface waters with distinct characteristics fit within the same use class, or do not fit well into any category. Determination of nonattainment in these waters may be difficult and open to alternative interpretations. If a determination is in dispute, regulatory actions will be difficult to accomplish. Emphasizing aquatic community structure within the designated use focuses the evaluation of attainment/nonattainment on the resource of concern under the Act.

Flexibility inherent in the State process for designating uses allows the development of subcategories of uses within the Act's general categories. For example, subcategories of aquatic life uses may be on the basis of attainable habitat (e.g., cold versus warmwater habitat); innate differences in community structure and function, (e.g., high versus low species richness or productivity); or fundamental differences in important community components (e.g., warmwater fish communities dominated by bass versus catfish). Special uses may also be designated to protect particularly unique, sensitive, or valuable aquatic species, communities, or habitats.

Refinement of use classes can be accomplished within current State use classification structures. Data collected from biosurveys as part of a developing biocriteria program may reveal unique and consistent differences among aquatic communities inhabiting different waters with the same designated use. Measurable biological attributes could then be used to separate one class into two or more classes. The result is a refined aquatic life use. For example, in Arkansas the beneficial use Fisheries "provides for the protection and propagation of fish, shellfish, and other forms of aquatic life" (Arkansas DPCE 1988). This use is subdivided into Trout, Lakes and Reservoirs, and Streams. Recognizing that stream characteristics across regions of the State differed ecologically, the State further subdivided the stream designated uses into eight additional uses based on regional characteristics (e.g., Springwater-influenced Gulf Coastal Ecoregion, Ouachita Mountains Ecoregion). Within this classification system, it was relatively straightforward for

Arkansas to establish detailed narrative biological criteria that list aquatic community components expected in each ecoregion (see Narrative Criteria section). These narrative criteria can then be used to establish whether the use is impaired.

States can refine very general designated uses such as high, medium, and low quality to specific categories that include measurable ecological characteristics. In Maine, for example, Class AA waters are defined as "the highest classification and shall be applied to waters which are outstanding natural resources and which should be preserved because of their ecological, social, scenic, or recreational importance." The designated use includes "Class AA waters shall be of such quality that they are suitable . . . as habitat for fish and other aquatic life. The habitat shall be characterized as free flowing and natural." This use supports development of narrative criteria based on biological characteristics of aquatic communities (Maine DEP 1986; see the Narrative Criteria section).

Biological criteria that include lists of dominant or typical species expected to live in the surface water are particularly effective. Descriptions of impaired conditions are more difficult to interpret. However, biological criteria may contain statements concerning which species dominate disturbed sites, as well as those species expected at minimally impacted sites.

Most States collect biological data in current programs. Refining aquatic life use classifications and incorporating biological criteria into standards will enable States to evaluate these data more effectively.

Developing and Implementing Biological Criteria

Biological criteria development and implementation in standards require an understanding of the selection and evaluation of reference sites, measurement of aquatic community structure and function, and hypothesis testing under the scientific method. The developmental process is important for State water quality managers and their staff to understand to promote effective planning for resource and staff needs. This major program element deserves careful consideration and has been separated out in Part II by chapter for each developmental step as noted below. Additional guidance will be provided in future technical guidance documents.

The developmental process is illustrated in Figure 3. The first step is establishing narrative criteria in standards. However, to support these narratives, standardized protocols need to be developed to quanitify the narratives for criteria implementation. They should include data collection procedures, selection of reference sites, quality assurance and quality control procedures, hypothesis testing, and statistical protocols. Pilot studies should be conducted using these standard protocols to ensure they meet the needs of the program, test the hypotheses, and provide effective measures of the biological integrity of surface waters in the State.

Figure 3.—Process for the Development and Implementation of Biological Criteria

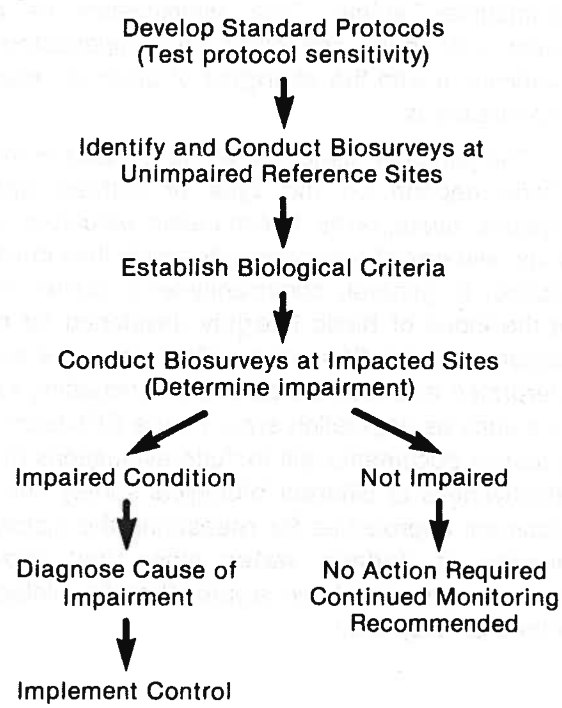


Fig. 3: Implementation of biological criteria requires the initial selection of reference sites and characterization of resident aquatic communities inhabiting those sites to establish the reference condition and biological criteria. After criteria development, impacted sites are evaluated using the same biosurvey procedures to assess resident biota. If impairment is found, diagnosis of cause will lead to the implementation of a control. Continued monitoring should accompany control implementation to determine the effectiveness of intervention. Monitoring is also recommended where no impairment is found to ensure that the surface water maintains or improves in quality.

The next step is establishing the reference condition for the surface water being tested. This reference may be site specific or regional but must establish the unimpaired baseline for comparison (see Chapter 5, The Reference Condition). Once reference sites are selected, the biological integrity of the site must be evaluated using carefully chosen biological surveys. A quality biological survey will include multiple community components and may be measured using a variety of metrics (see Chapter 6, The Biological Survey). Establishing the reference condition and conducting biological surveys at the reference locations provide the necessary information for establishing the biological criteria.

To apply biological criteria, impacted surface waters with comparable habitat characteristics are evaluated using the same procedures as those used to establish the criteria. The biological survey must support standardized sampling methods and statistical protocols that are sensitive enough to identify biologically relevant differences between established criteria and the community under evaluation. Resulting data are compared through hypothesis testing to determine impairment (see Chapter 7, Hypothesis Testing).

When water quality impairments are detected using biological criteria, they can only be applied in a regulatory setting if the cause for impairment can be identified. Diagnosis is iterative and investigative (see Chapter 7, Diagnosis). States must then determine appropriate actions to implement controls. Monitoring should remain a part of the biological criteria program whether impairments are found or not. If an impairment exists, monitoring provides a mechanism to determine if the control effort (intervention) is resulting in improved water quality. If there is no impairment, monitoring ensures the water quality is maintained and documents any improvements. When improvements in water quality are detected through monitoring programs two actions are recommended. When reference condition waters improve, biological criteria values should be recalculated to reflect this higher level of integrity. When impaired surface waters improve, states should reclassify those waters to reflect a refined designated use with a higher level of biological integrity. This provides a mechanism for progressive water quality improvement.

Chapter 4

Integrating Biological Criteria Into Surface Water Management

Integrating biological criteria into existing water quality programs will help to assess use attainment/nonattainment, improve problem discovery in specific waterbodies, and characterize overall water resource condition within a region. Ideally, biological criteria function in an iterative manner. New biosurvey information can be used to refine use classes. Refined use classes will help support criteria development and improve the value of data collected in biosurveys.

Implementing Biological Criteria

As biological survey data are collected, these data will increasingly support current use of biomonitoring data to identify water quality problems, assess their severity, and set planning and management priorities for remediation. Monitoring data and biological criteria should be used at the outset to help make regulatory decisions, develop appropriate controls, and evaluate the effectiveness of controls once they are implemented.

The value of incorporating biological survey information in regulatory programs is illustrated by evaluations conducted by North Carolina. In response to amendments of the Federal Water Pollution Control Act requiring secondary effluent limits for all wastewater treatment plants, North Carolina became embroiled in a debate over whether meeting secondary effluent limits (at considerable cost) would result in better water quality. North Carolina chose to test the effectiveness of additional treatment by conducting seven chemical and biological surveys before and after facility upgrades (North

To integrate biological criteria into water quality programs, states must carefully determine where and how data are collected to assess the biological integrity of surface waters.

Carolina DNRCD 1984). Study results indicated that moderate to substantial in-stream improvements were observed at six of seven facilities. Biological surveys were used as an efficient, cost-effective monitoring tool for assessing in-stream improvements after facility modification. North Carolina has also conducted comparative studies of benthic macroinvertebrate surveys and chemical-specific and whole-effluent evaluations to assess sensitivities of these measures for detecting impairments (Eagleson et al. 1990).

Narrative biological criteria provide a scientific framework for evaluating biosurvey, bioassessment, and biomonitoring data collected in most States. Initial application of narrative biological criteria may require only an evaluation of current work. States can use available data to define variables for choosing reference sites, selecting appropriate biological surveys, and assessing the response of local biota to a variety of impacts. States should also consider the decision criteria that will be used for determining appropriate State action when impairment is found.

Recent efforts by several States to develop biological criteria for freshwater streams provide excellent examples for how biological criteria can be integrated into water quality programs. Some of this work is described in the *National Workshop on Instream Biological Monitoring and Criteria* proceedings which recommended that "the concept of biological sampling should be integrated into the full spectrum of State and Federal surface water programs" (U.S. EPA 1987b). States are actively developing biological assessment and criteria programs; several have programs in place.

Biological Criteria in State Programs

Biological criteria are used within water programs to refine use designations, establish criteria for determining use attainment/nonattainment, evaluate effectiveness of current water programs, and detect and characterize previously unknown impairments. Twenty States are currently using some form of standardized ambient biological assessments to determine the status of biota within State waters. Levels of effort vary from bioassessment studies to fully developed biological criteria programs.

Fifteen States are developing aspects of biological assessments that will support future development of biological criteria. Colorado, Illinois, Iowa, Kentucky, Massachusetts, Tennessee, and Virginia conduct biological monitoring to evaluate biological conditions, but are not developing biological criteria. Kansas is considering using a community metric for water resource assessment. Arizona is planning to refine ecoregions for the State. Delaware, Minnesota, Texas, and Wisconsin are developing sampling and evaluation methods to apply to future biological criteria programs. New York is proposing to use biological criteria for site-specific evaluations of water quality impairment. Nebraska and Vermont use informal biological criteria to support existing aquatic life narratives in their water quality standards and other regulations. Vermont recently passed a law requiring that biological criteria be used to regulate through permitting the indirect discharge of sanitary effluents.

Florida incorporated a specific biological criterion into State standards for invertebrate species diversity. Species diversity within a waterbody, as measured by a Shannon diversity index, may not fall below 75 percent of reference values. This criterion has been used in enforcement cases to obtain injunctions and monetary settlements. Florida's approach is very specific and limits alternative applications.

Four States—Arkansas, North Carolina, Maine, and Ohio—are currently using biological criteria to define aquatic life use classifications and enforce water quality standards. These states have made biological criteria an integral part of comprehensive water quality programs.

■ **Arkansas** rewrote its aquatic life use classifications for each of the State's ecoregions. This has allowed many cities to design wastewater treatment plants to meet realistic attainable dissolved oxygen conditions as determined by the new criteria.

■ **North Carolina** developed biological criteria to assess impairment to aquatic life uses written as narratives in the State water quality standards. Biological data and criteria are used extensively to identify waters of special concern or those with exceptional water quality. In addition to the High Quality Waters (HQW) and Outstanding Resource Waters (ORW) designations, Nutrient Sensitive Waters (NSW) at risk for eutrophication are assessed using biological criteria. Although specific biological measures are not in the regulations, strengthened use of biological monitoring data to assess water quality is being proposed for incorporation in North Carolina's water quality standards.

■ **Maine** has enacted a revised Water Quality Classification Law specifically designed to facilitate the use of biological assessments. Each of four water classes contains descriptive aquatic life conditions necessary to attain that class. Based on a statewide database of macroinvertebrate samples collected above and below outfalls, Maine is now developing a set of dichotomous keys that serve as the biological criteria. Maine's program is not expected to have a significant role in permitting, but will be used to assess the degree of protection afforded by effluent limitations.

■ **Ohio** has instituted the most extensive use of biological criteria for defining use classifications and assessing water quality. Biological criteria were developed for Ohio rivers and streams using an ecoregional reference site approach. Within each of the State's five ecoregions, criteria for three biological indices (two for fish communities and one for macroinvertebrates) were derived. Ohio successfully uses biological criteria to demonstrate attainment of aquatic life uses and discover previously unknown or unidentified environmental degradation (e.g., twice as many impaired waters were discovered using biological criteria and water chemistry together than were found using chemistry alone). The upgraded use designations based on biological criteria were upheld in Ohio courts and the Ohio EPA successfully proposed their biological criteria for inclusion in the State water quality standards regulations.

States and EPA have learned a great deal about the effectiveness of integrated biological assessments through the development of biological criteria for freshwater streams. This information is particularly valuable in providing guidance on developing biological criteria for other surface water types. As previously discussed, EPA plans to produce supporting technical guidance for biological criteria development in streams and other surface waters. Production of these guidance documents will be contingent on technical progress made on each surface water type by researchers in EPA, States and the academic community.

EPA will also be developing outreach workshops to provide technical assistance to Regions and States working toward the implementation of biological criteria programs in State water quality management programs. In the interim, States should use the technical guidance currently available in the *Technical Support Manual(s): Waterbody Surveys and Assessments for Conducting Use Attainability Analysis* (U.S. EPA 1983b, 1984a,b).

During the next triennium, State effort will be focused on developing narrative biological criteria. Full implementation and integration of biological criteria will require several years. Using available guidance, States can complement the adoption of narrative criteria by developing implementation plans that include:

1. Defining program objectives, developing research protocols, and setting priorities;

2. Determining the process for establishing reference conditions, which includes developing a process to evaluate habitat characteristics;

3. Establishing biological survey protocols that include justifications for surface water classifications and selected aquatic community components to be evaluated; and

4. Developing a formal document describing the research design, quality assurance and quality control protocols, and required training for staff.

Whether a State begins with narrative biological criteria or moves to fully implement numeric criteria, the shift of the water quality program focus from source control to resource management represents a natural progression in the evolution from the technology-based to water quality-based approaches in water quality management. The addition of a biological perspective allows water quality programs to more directly address the objectives of the Clean Water Act and to place their efforts in a context that is more meaningful to the public.

Future Directions

Biological criteria now focus on resident aquatic communities in surface waters. They have the potential to expand in scope toward greater ecological integration. Ecological criteria may encompass the ambient aquatic communities in surface waters, wildlife species that use the same aquatic resources, and the aquatic community inhabiting the gravel and sediments underlying the surface waters and adjacent land (hyporheic zone); specific criteria may apply to physical habitat. These areas may represent only a few possible options for biological criteria in the future.

Many wildlife species depend on aquatic resources. If aquatic population levels decrease or if the distribution of species changes, food sources may be sufficiently altered to cause problems for wildlife species using aquatic resources. Habitat degradation that impairs aquatic species will often impact important wildlife habitat as well. These kinds of impairments are likely to be detected using biological criteria as currently formulated. In some cases, however, uptake of contaminants by resident aquatic organisms may not result in altered structure and function of the aquatic community. These impacts may go undetected by biological criteria, but could result in wildlife impairments because of bioaccumulation. Future expansion of biological criteria to include wildlife species that depend on aquatic resources could provide a more integrative ecosystem approach.

Rivers may have a subsurface flood plain extending as far as two kilometers from the river channel. Preliminary mass transport calculations made in the Flathead River basin in Montana indicate that nutrients discharged from this subsurface flood plain may be crucial to biotic productivity in the river channel (Stanford and Ward 1988). This is an unexplored dimension in the ecology of gravel river beds and potentially in other surface waters.

As discussed in Chapter 1, physical integrity is a necessary condition for biological integrity. Establishing the reference condition for biological criteria requires evaluation of habitat. The rapid bioassessment protocol provides a good example of the importance of habitat for interpreting biological assessments (Plafkin et al. 1989). However, it may be useful to more fully integrate habitat characteristics into the regulatory process by establishing criteria based on the necessary physical structure of habitats to support ecological integrity.

Part II

The Implementation Process

The implementation of biological criteria requires: (1) selection of unimpaired (minimal impact) surface waters to use as the reference condition for each designated use, (2) measurement of the structure and function of aquatic communities in reference surface waters to establish biological criteria, and (3) establishment of a protocol to compare the biological criteria to biota in impacted waters to determine whether impairment has occurred. These elements serve as an interactive network that is particularly important during early development of biological criteria where rapid accumulation of information is effective for refining both designated uses and developing biological criteria values. The following chapters describe these three essential elements.

Chapter 5

The Reference Condition

A key step in developing values for supporting narrative and creating numeric biological criteria is to establish reference conditions; it is an essential feature of environmental impact evaluations (Green 1979). Reference conditions are critical for environmental assessments because standard experimental controls are rarely available. For most surface waters, baseline data were not collected prior to an impact, thus impairment must be inferred from differences between the impact site and established references. Reference conditions describe the characteristics of waterbody segments least impaired by human activities and are used to define attainable biological or habitat conditions.

Wide variability among natural surface waters across the country resulting from climatic, landform, and other geographic differences prevents the development of nationwide reference conditions. Most States are also too heterogeneous for single reference conditions. Thus, each State, and when appropriate, groups of States, will be responsible for selecting and evaluating reference waters within the State to establish biological criteria for a given surface water type or category of designated use. At least seven methods for estimating attainable conditions for streams have been identified (Hughes et al. 1986). Many of these can apply to other surface waters. References may be established by defining models of attainable conditions based on historical data or unimpaired habitat (e.g., streams in old growth forest). The reference condition established as before-after comparisons or concurrent measures of the reference water and impact sites can be based on empirical data (Hall et al. 1989).

Reference conditions should be established by measuring resident biota in unimpaired surface waters.

Currently, two principal approaches are used for establishing the reference condition. A State may opt to (1) identify site-specific reference sites for each evaluation of impact or (2) select ecologically similar regional reference sites for comparison with impacted sites within the same region. Both approaches depend on evaluations of habitats to ensure that waters with similar habitats are compared. The designation of discrete habitat types is more fully developed for streams and rivers. Development of habitat types for lakes, wetlands, and estuaries is ongoing.

Site-Specific Reference Condition

A site-specific reference condition, frequently used to evaluate the impacts from a point discharge, is best for surface waters with a strong directional flow such as in streams and rivers (the upstream-downstream approach). However, it can also be used for other surface waters where gradients in contaminant concentration occur based on proximity to a source (the near field-far field approach). Establishment of a site-specific reference condition requires the availability of comparable habitat within the same waterbody in both the reference location and the impacted area.

A site-specific reference condition is difficult to establish if (1) diffuse nonpoint source pollution contaminates most of the water body; (2) modifications to the channel, shoreline, or bottom substrate are extensive; (3) point sources occur at multiple locations on the waterbody; or (4) habitat characteristics differ significantly between possible reference locations and the impact site (Hughes et al. 1986; Plafkin et al. 1989). In these cases, site-specific reference conditions could result in underestimates of impairment. Despite limitations, the use of site-specific reference conditions is often the method of choice for point source discharges and certain waterbodies, particularly when the relative impairments from different local impacts need to be determined.

The Upstream-Downstream Reference Condition

The upstream-downstream reference condition is best applied to streams and rivers where the habitat characteristics of the waterbody above the point of discharge are similar to the habitat characteristics of the stream below the point of discharge. One standard procedure is to characterize the biotic condition just above the discharge point (accounting for possible upstream circulation) to establish the reference condition. The condition below the discharge is also measured at several sites. If significant differences are found between these measures, impairment of the biota from the discharge is indicated. Since measurements of resident biota taken in any two sites are expected to differ because of natural variation, more than one biological assessment for both upstream and downstream sites is often needed to be confident in conclusions drawn from these data (Green, 1979). However, as more data are collected by a State, and particularly if regional characteristics of the waterbodies are incorporated, the basis for determining impairment from site-specific upstream-downstream assessments may require fewer individual samples. The same measures made below the "recovery zone" downstream from the discharge will help define where recovery occurs.

The upstream-downstream reference condition should be used with discretion since the reference condition may be impaired from impacts upstream from the point source of interest. In these cases it is important to discriminate between individual point source impact versus overall impairment of the system. When overall impairment occurs, the resident biota may be sufficiently impaired to make it impossible to detect the effect of the target point source discharger.

The approach can be cost effective when one biological assessment of the upstream reference condition adequately reflects the attainable condition of the impacted site. However, routine comparisons may require assessments of several upstream sites to adequately describe the natural variability of reference biota. Even so, measuring a series of site-specific references will likely continue to be the method of choice for certain point source discharges, especially where the relative impairments from different local impacts need to be determined.

The Near Field-Far Field Reference Condition

The near field-far field reference condition is effective for establishing a reference condition in surface waters other than rivers and streams and is particularly applicable for unique waterbodies (e.g., estuaries such as Puget Sound may not have comparable estuaries for comparison). To apply this method, two variables are measured (1) habitat characteristics, and (2) gradient of impairment. For reference waters to be identified within the same waterbody, sufficient size is necessary to separate the reference from the impact area so that a gradient of impact exists. At the same time, habitat characteristics must be comparable.

Although not fully developed, this approach may provide an effective way to establish biological criteria for estuaries, large lakes, or wetlands. For example, estuarine habitats could be defined and possible reference waters identified using physical and chemical variables like those selected by the Chesapeake Bay Program (U.S. EPA 1987a, e.g., substrate type, salinity, pH) to establish comparable subhabitats in an estuary. To determine those areas least impaired, a "mussel watch" program like that used in Narragansett Bay (i.e., captive mussels are used as indicators of contamination, (Phelps 1988)) could establish impairment gradients. These two measures, when combined, could form the basis for selecting specific habitat types in areas of least impairment to establish the reference condition.

Regional Reference Conditions

Some of the limitations of site-specific reference conditions can be overcome by using regional reference conditions that are based on the assumption that surface waters integrate the character of the land they drain. Waterbodies within the same watershed in the same region should be more similar to each other than to those within watersheds in different regions. Based on these assumptions, a distribution of aquatic regions can be developed based on ecological features that directly or indirectly relate to water quality and quantity, such as soil type, vegetation (land cover), land-surface form, climate, and land use. Maps that incorporate several of these features will provide a general purpose broad scale ecoregional framework (Gallant et al. 1989).

Regions of ecological similarity are based on hydrologic, climatic, geologic, or other relevant geographic variables that influence the nature of biota in surface waters. To establish a regional reference condition, surface waters of similar habitat type are identified in definable ecological regions. The biological integrity of these reference waters is determined to establish the reference condition and develop biological criteria. These criteria are then used to assess impacted surface waters in the same watershed or region. There are two forms of regional reference conditions: (1) paired watersheds and (2) ecoregions.

Paired Watershed Reference Conditions

Paired watershed reference conditions are established to evaluate impaired waterbodies, often impacted by multiple sources. When the majority of a waterbody is impaired, the upstream-downstream or near field-far field reference condition does not provide an adequate representation of the unimpaired condition of aquatic communities for the waterbody. Paired watershed reference conditions are established by identifying unimpaired surface waters within the same or very similar local watershed that is of comparable type and habitat. Variables to consider when selecting the watershed reference condition include absence of human disturbance, waterbody size and other physical characteristics, surrounding vegetation, and others as described in the "Regional Reference Site Selection" feature.

This method has been successfully applied (e.g., Hughes 1985) and is an approach used in Rapid Bioassessment Protocols (Plafkin et al. 1989). State use of this approach results in good reference conditions that can be used immediately in current programs. This approach has the added benefit of promoting the development of a database on high quality waters in the State that could form the foundation for establishing larger regional references (e.g., ecoregions.)

Ecoregional Reference Conditions

Reference conditions can also be developed on a larger scale. For these references, waterbodies of similar type are identified in regions of ecological similarity. To establish a regional reference condition, a set of surface waters of similar habitat type are identified in each ecological region. These sites must represent similar habitat type and be representative of the region. As with other reference conditions, the biological integrity of selected reference waters is determined to establish the reference. Biological criteria can then be developed and used to assess impacted surface waters in the same region. Before reference conditions may be established, regions of ecological similarity must be defined.

Regional Reference Site Selection

To determine specific regional reference sites for streams, candidate watersheds are selected from the appropriate maps and evaluated to determine if they are typical for the region. An evaluation of level of human disturbance is made and a number of relatively undisturbed reference sites are selected from the candidate sites. Generally, watersheds are chosen as regional reference sites when they fall entirely within typical areas of the region. Candidate sites are then selected by aerial and ground surveys. Identification of candidate sites is based on: (1) absence of human disturbance, (2) stream size, (3) type of stream channel, (4) location within a natural or political refuge, and (5) historical records of resident biota and possible migration barriers.

Final selection of reference sites depends on a determination of minimal disturbance derived from habitat evaluation made during site visits. For example, indicators of good quality streams in forested ecoregions include: (1) extensive, old, natural riparian vegetation; (2) relatively high heterogeneity in channel width and depth; (3) abundant large woody debris, coarse bottom substrate, or extensive aquatic or overhanging vegetation; (4) relatively high or constant discharge; (5) relatively clear waters with natural color and odor; (6) abundant diatom, insect, and fish assemblages; and (7) the presence of piscivorous birds and mammals.

One frequently used method is described by Omernik (1987) who combined maps of land-surface form, soil, potential natural vegetation, and land use within the conterminous United States to generate a map of aquatic ecoregions for the country. He also developed more detailed regional maps. The ecoregions defined by Omernik have been evaluated for streams and small rivers in Arkansas (Rohm et al. 1987), Ohio (Larsen et al. 1986; Whittier et al. 1987), Oregon (Whittier et al. 1988), Colorado (Gallant et al. 1989), and Wisconsin (Lyons 1989) and for lakes in Minnesota (Heiskary et al. 1987). State ecoregion maps were developed for Colorado (Gallant et al. 1989) and Oregon (Clarke et al. mss). Maps for the national ecoregions and six multi-state maps of more detailed ecoregions are available from the U.S. EPA Environmental Research Laboratory, Corvallis, Oregon.

Ecoregions such as those defined by Omernik (1987) provide only a first step in establishing regional reference sites for development of the reference condition. Field site evaluation is required to account for the inherent variability within each ecoregion. A general method for selecting reference sites for streams has been described (Hughes et al. 1986). These are the same variables used for comparable watershed reference site selection. Regional and on-site evaluations of biological factors help determine specific sites that best represent typical but unimpaired surface water habitats within the region. Details on this approach for streams is described in the "Regional Reference Site Selection" feature. To date, the regional approach has been tested on streams, rivers, and lakes. The method appears applicable for assessing other inland ecosystems. To apply this approach to wetlands and estuaries will require additional evaluation based on the relevant ecological features of these ecosystems (e.g. Brooks and Hughes, 1988).

Ideally, ecoregional reference sites should be as little disturbed as possible, yet represent waterbodies for which they are to serve as reference waters. These sites may serve as references for a large number of similar waterbodies (e.g., several reference streams may be used to define the reference condition for numerous physically separate streams if the reference streams contain the same range of stream morphology, substrate, and flow of the other streams within the same ecological region).

An important benefit of a regional reference system is the establishment of a baseline condition for the least impacted surface waters within the dominant land use pattern of the region. In many areas a return to pristine, or presettlement, conditions is impossible, and goals for waterbodies in extensively developed regions could reflect this. Regional reference sites based on the least impacted sites within a region will help water quality programs restore and protect the environment in a way that is ecologically feasible.

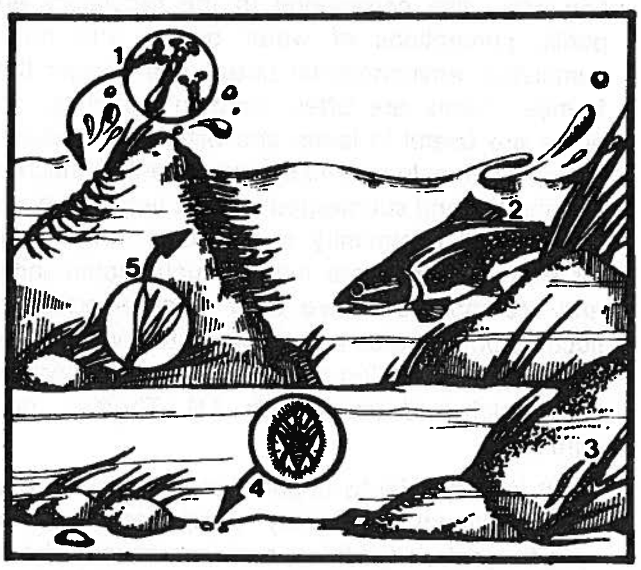
This approach must be used with caution for two reasons. First, in many urban, industrial, or heavily developed agricultural regions, even the least impacted sites are seriously degraded. Basing standards or criteria on such sites will set standards too low if these high levels of environmental degradation are considered acceptable or adequate. In such degraded regions, alternative sources for the regional reference may be needed (e.g., measures taken from the same region in a less developed neighboring State or historical records from the region before serious impact occurred). Second, in some regions the minimally-impacted sites are not typical of most sites in the region and may have remained unimpaired precisely because they are unique. These two considerations emphasize the need to select reference sites very carefully, based on solid quantitative data interpreted by professionals familiar with the biota of the region.

Each State, or groups of States, can select a series of regional reference sites that represent the attainable conditions for each region. Once biological criteria are established using this approach, the cost for evaluating local impairments is often lower than a series of measures of site-specific reference sites. Using paired watershed reference conditions immediately in regulatory programs will provide the added benefit of building a database for the development of regions of ecological similarity.

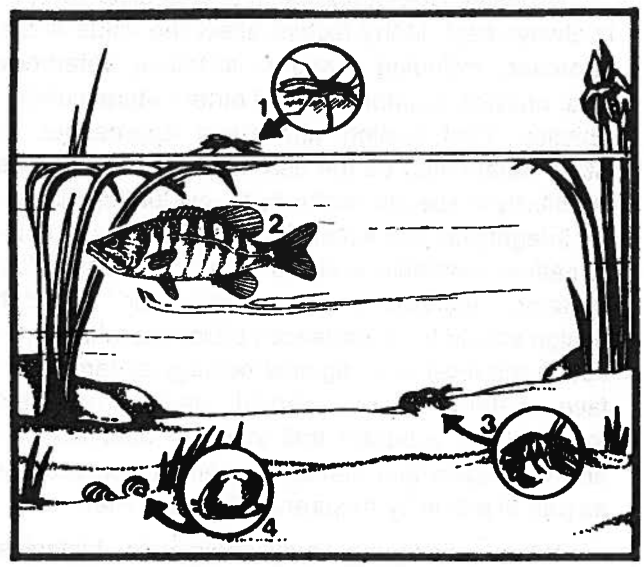
Chapter 6

The Biological Survey

A critical element of biological criteria is the characterization of biological communities inhabiting surface waters. Use of biological data is not new; biological information has been used to assess impacts from pollution since the 1890s (Forbes 1928), and most States currently incorporate biological information in their decisions about the quality of surface waters. However, biological information can be obtained through a variety of methods, some of which are more effective than others for characterizing resident aquatic biota. Biological criteria are developed using biological surveys; these provide the only direct method for measuring the structure and function of an aquatic community.



Different subhabitat within the same surface water will contain unique aquatic community components. In fast-flowing stream segments species such as (1) black fly larva; (2) brook trout; (3) water penny; (4) crane fly larva; and (5) water moss occur.



However, in slow-flowing stream segments, species like (1) water strider; (2) smallmouth bass; (3) crayfish; and (4) fingernail clams are abundant.

Biological survey study design is of critical importance to criteria development. The design must be scientifically rigorous to provide the basis for legal action, and be biologically relevant to detect problems of regulatory concern. Since it is not financially or technically feasible to evaluate all organisms in an entire ecosystem at all times, careful selection of community components, the time and place chosen for assessments, data gathering methods used, and the consistency with which these variables are applied will determine the success of the biological criteria program. Biological surveys must therefore be carefully planned to meet scientific and legal requirements, maximize information, and minimize cost.

Biological surveys can range from collecting samples of a single species to comprehensive evaluations of an entire ecosystem. The first approach is difficult to interpret for community assessment; the second approach is expensive and impractical. A balance between these extremes can meet program needs. Current approaches range between detailed ecological surveys, biosurveys of targeted community components, and biological indicators (e.g., keystone species). Each of these biosurveys has advantages and limitations. Additional discussion will be provided in technical guidance under development.

No single type of approach to biological surveys is always best. Many factors affect the value of the approach, including seasonal variation, waterbody size, physical boundaries, and other natural characteristics. Pilot testing alternative approaches in State waters may be the best way to determine the sensitivity of specific methods for evaluating biological integrity of local waters. Due to the number of alternatives available and the diversity of ecological systems, individuals responsible for research design should be experienced biologists with expertise in the local and regional ecology of target surface waters. States should develop a data management program that includes data analysis and evaluation and standard operating procedures as part of a Quality Assurance Program Plan.

When developing study designs for biological criteria, two key elements to consider include (1) selecting aquatic community components that will best represent the biological integrity of State surface waters and (2) designing data collection protocols to ensure the best representation of the aquatic community. Technical guidance currently available to aid the development of study design include: *Water Quality Standards Handbook* (U.S. EPA 1983a), *Technical Support Manual: Waterbody Surveys and Assessments for Conducting Use Attainability Analyses* (U.S. EPA 1983b); *Technical Support Manual: Waterbody Surveys and Assessments for Conducting Use Attainability Analyses, Volume II: Estuarine Systems* (U.S. EPA 1984a); and *Technical Support Manual: Waterbody Surveys and Assessments for Conducting Use Attainability Analyses, Volume III: Lake Systems* (U.S. EPA 1984b). Future technical guidance will build on these documents and provide specific guidance for biological criteria development.

Selecting Aquatic Community Components

Aquatic communities contain a variety of species that represent different trophic levels, taxonomic groups, functional characteristics, and tolerance ranges. Careful selection of target taxonomic groups can provide a balanced assessment that is sufficiently broad to describe the structural and functional condition of an aquatic ecosystem, yet be sufficiently practical to use on a daily basis (Plafkin et al. 1989; Lenat 1988). When selecting community components to include in a biological assessment, primary emphasis should go toward including species or taxa that (1) serve as effective indicators of high biological integrity (i.e., those likely to live in unimpaired waters), (2) represent a range of pollution tolerances, (3) provide predictable, repeatable results, and (4) can be readily identified by trained State personnel.

Fish, macroinvertebrates, algae, and zooplankton are most commonly used in current bioassessment programs. The taxonomic groups chosen will vary depending on the type of aquatic ecosystem being assessed and the type of expected impairment. For example, benthic macroinvertebrate and fish communities are taxonomic groups often chosen for flowing fresh water. Macroinvertebrates and fish both provide valuable ecological information while fish correspond to the regulatory and public perceptions of water quality and reflect cumulative environmental stress over longer time frames. Plants are often used in wetlands, and algae are useful in lakes and estuaries to assess eutrophication. In marine systems, benthic macroinvertebrates and submerged aquatic vegetation may provide key community components. Amphipods, for example, dominate many aquatic communities and are more sensitive than other invertebrates such as polychaetes and molluscs to a wide variety of pollutants including hydrocarbons and heavy metals (Reich and Hart 1979; J.D. Thomas, pers. comm.).

It is beneficial to supplement standard groups with additional community components to meet specific goals, objectives, and resources of the assessment program. Biological surveys that use two or three taxonomic groups (e.g., fish, macroinvertebrates, algae) and, where appropriate, include different trophic levels within each group (e.g., primary, secondary, and tertiary consumers) will provide a more realistic evaluation of system biological integrity. This is analogous to using species from two or more taxonomic groups in bioassays. Impairments that are difficult to detect because of the temporal or spatial habits or the pollution tolerances of one group may be revealed through impairments in different species or assemblages (Ohio EPA 1988a).

Selection of aquatic community components that show different sensitivities and responses to the same perturbation will aid in identifying the nature of a problem. Available data on the ecological function, distribution, and abundance of species in a given habitat will help determine the most appropriate target species or taxa for biological surveys in the habitat. The selection of community components should also depend on the ability of the organisms to be accurately identified by trained State personnel. Attendent with the biological criteria program should be the development of identification keys for the organisms selected for study in the biological survey.

Biological Survey Design

Biological surveys that measure the structure and function of aquatic communities will provide the information needed for biological criteria development. Elements of community structure and function may be evaluated using a series of metrics. Structural metrics describe the composition of a community, such as the number of different species, relative abundance of specific species, and number and relative abundance of tolerant and intolerant species. Functional metrics describe the ecological processes of the community. These may include measures such as community photosynthesis or respiration. Function may also be estimated from the proportions of various feeding groups (e.g., omnivores, herbivores, and insectivores, or shredders, collectors, and grazers). Biological surveys can offer variety and flexibility in application. Indices currently available are primarily for freshwater streams. However, the approach has been used for lakes and can be developed for estuaries and wetlands.

Selecting the metric

Several methods are currently available for measuring the relative structural and functional well-being of fish assemblages in freshwater streams, such as the Index of Biotic Integrity (IBI); Karr 1981; Karr et al. 1986; Miller et al. 1988) and the Index of Well-being (IWB; Gammon 1976, Gammon et al. 1981). The IBI is one of the more widely used assessment methods. For additional detail, see the "Index of Biotic Integrity" feature.

Index of Biotic Integrity

The Index of Biotic Integrity (IBI) is commonly used for fish community analysis (Karr 1981). The original IBI was comprised of 12 metrics:

- *six metrics evaluate species richness and composition*
  - *number of species*
  - *number of darter species*
  - *number of sucker species*
  - *number of sunfish species*
  - *number of intolerant species*
  - *proportion of green sunfish*
- *three metrics quantify trophic composition*
  - *proportion of omnivores*
  - *proportion of insectivorous cyprinids*
  - *proportion of piscivores*
- *three metrics summarize fish abundance and condition information*
  - *number of individuals in sample*
  - *proportion of hybrids*
  - *proportion of individuals with disease*

Each metric is scored 1 (worst), 3, or 5 (best), depending on how the field data compare with an expected value obtained from reference sites. All 12 metric values are then summed to provide an overall index value that represents relative integrity. The IBI was designed for midwestern streams; substitute metrics reflecting the same structural and functional characteristics have been created to accommodate regional variations in fish assemblages (Miller et al. 1988).

. Several indices that evaluate more than one community characteristic are also available for assessing stream macroinvertebrate populations. Taxa richness, EPT taxa (number of taxa of the insect orders Ephemeroptera, Plecoptera, and Tricoptera), and species pollution tolerance values are a few of several components of these macroinvertebrate assessments. Example indices include the Invertebrate Community Index (ICI; Ohio EPA, 1988) and Hilsenhoff Biotic Index (HBI; Hilsenhoff, 1987).

Within these metrics specific information on the pollution tolerances of different species within a system will help define the type of impacts occurring in a waterbody. Biological indicator groups (intolerant species, tolerant species, percent of diseased organisms) can be used for evaluating community biological integrity if sufficient data have been collected to support conclusions drawn from the indicator data. In marine systems, for example, amphipods have been used by a number of researchers as environmental indicators (McCall 1977; Botton 1979; Mearns and Word 1982).

Sampling design

Sampling design and statistical protocols are required to reduce sampling error and evaluate the natural variability of biological responses that are found in both laboratory and field data. High variability reduces the power of a statistical test to detect real impairments (Sokal and Rohlf, 1981). States may reduce variability by refining sampling techniques and protocol to decrease variability introduced during data collection, and increase the power of the evaluation by increasing the number of replications. Sampling techniques are refined, in part, by collecting a representative sample of resident biota from the same component of the aquatic community from the same habitat type in the same way at sites being compared. Data collection protocols should incorporate (1) spatial scales (where and how samples are collected) and (2) temporal scales (when data are collected) (Green, 1979):

- **Spatial Scales** refer to the wide variety of subhabitats that exist within any surface water habitat. To account for subhabitats, adequate sampling protocols require selecting (1) the location within a habitat where target groups reside and (2) the method for collecting data on target groups. For example, if fish are sampled only from fast flowing riffles within stream A, but are sampled from slow flowing pools in stream B, the data will not be comparable.

- **Temporal Scales** refer to aquatic community changes that occur over time because of diurnal and life-cycle changes in organism behavior or development, and seasonal or annual changes in the environment. Many organisms go through seasonal life-cycle changes that dramatically affect their presence and abundance in the aquatic community. For example, macroinvertebrate data collected from stream A in March and stream B in May, would not be comparable because the emergence of insect adults after March would significantly alter the abundance of subadults found in stream B in May. Similar problems would occur if algae were collected in lake A during the dry season and lake B during the wet season.

Field sampling protocols that produce quality assessments from a limited number of site visits greatly enhance the utility of the sampling technique. Rapid bioassessment protocols, recently developed for assessing streams, use standardized techniques to quickly gather physical, chemical, and biological quantitative data that can assess changes in biological integrity (Plafkin et al. 1989). Rapid bioassessment methods can be cost-effective biological assessment approaches when they have been verified with more comprehensive evaluations for the habitats and region where they are to be applied.

Biological survey methods such as the IBI for fish and ICI for macroinvertebrates were developed in streams and rivers and have yet to be applied to many ecological regions. In addition, further research is needed to adapt the approach to lakes, wetlands, and estuaries, including the development of alternative structural or functional endpoints. For example, assessment methods for algae (e.g. measures of biomass, nuisance bloom frequency, community structure) have been used for lakes. Assessment metrics appropriate for developing biological criteria for lakes, large rivers, wetlands, and estuaries are being developed and tested so that a multi-metric approach can be effectively used for all surface waters.

Chapter 7

Hypothesis Testing: Biological Criteria and the Scientific Method

Biological criteria are applied in the standards program by testing hypotheses about the biological integrity of impacted surface waters. These hypotheses include the null hypothesis—the designated use of the waterbody is not impaired—and alternative hypotheses such as the designated use of the waterbody is impaired (more specific hypotheses can also be generated that predict the type(s) of impairment). Under these hypotheses specific predictions are generated concerning the kinds and numbers of organisms representing community structure and function expected or found in unimpaired habitats. The kinds and numbers of organisms surveyed in unimpaired waters are used to establish the biological criteria. To test the alternative hypotheses, data collection and analysis procedures are used to compare the criteria to comparable measures of community structure and function in impacted waters.

Multiple impacts in the same surface water such as discharges of effluent from point sources, leachate from landfills or dumps, and erosion from habitat degradation each contribute to impairment of the surface water. All impacts should be considered during the diagnosis process.

Hypothesis Testing

To detect differences of biological and regulatory concern between biological criteria and ambient biological integrity at a test site, it is important to establish the sensitivity of the evaluation. A 10 percent difference in condition is more difficult to detect than a 50 percent difference. For the experimental/survey design to be effective, the level of detection should be predetermined to establish sample size for data collection (Sokal and Rohlf 1981). Knowledge of expected natural variation, experimental error, and the kinds of detectable differences that can be expected will help determine sample size and location. This forms the basis for defining data quality objectives, standardizing data collection procedures, and developing quality assurance/ quality control standards.

Once data are collected and analyzed, they are used to test the hypotheses to determine if characteristics of the resident biota at a test site are significantly different from established criteria values for a comparable habitat. There are three possible outcomes:

1. The use is impaired when survey design and data analyses are sensitive enough to detect differences of regulatory importance, and significant differences were detected. The next step is to diagnose the cause(s) and source(s) of impairment.

2. The biological criteria are met when survey design and data analyses are sensitive enough to detect differences of regulatory significance, but no differences were found. In this case, no action is required by States based on these measures. However, other evidence may indicate impairment (e.g., chemical criteria are violated; see below).

3. The outcome is indeterminate when survey design and data analyses are not sensitive enough to detect differences of regulatory significance, and no differences were detected. If a State or Region determines that this is occurring, the development of study design and evaluation for biological criteria was incomplete. States must then determine whether they will accept the sensitivity of the survey or conduct additional surveys to increase the power of their analyses. If the sensitivity of the original survey is accepted, the State should determine what magnitude of difference the survey is capable of detecting. This will aid in re-evaluating research design and desired detection limits. An indeterminate outcome may also occur if the test site and the reference conditions were not comparable. This variable may also require re-evaluation.

As with all scientific studies, when implementing biological criteria, the purpose of hypothesis testing is to determine if the data support the conclusion that the null hypothesis is false (i.e., the designated use is not impaired in a particular waterbody). Biological criteria cannot prove attainment. This reasoning provides the basis for emphasizing independent application of different assessment methods (e.g., chemical verses biological criteria). No type of criteria can "prove" attainment; each type of criteria can disprove attainment.

Although this discussion is limited to the null and one alternative hypothesis, it is possible to generate multiple working hypotheses (Popper, 1968) that promote the diagnosis of water quality problems when they exist. For example, if physical habitat limitations are believed to be causing impairment (e.g., sedimentation) one alternative hypothesis could specify the loss of community components sensitive to this impact. Using multiple hypotheses can maximize the information gained from each study. See the Diagnosis section for additional discussion.

Diagnosis

When impairment of the designated use is found using biological criteria, a diagnosis of probable cause of impairment is the next step for implementation. Since biological criteria are primarily designed to detect water quality impairment, problems are likely to be identified without a known cause. Fortunately the process of evaluating test sites for biological impairment provides significant information to aid in determining cause.

During diagnostic evaluations, three main impact categories should be considered: chemical, physical, and biological. To begin the diagnostic process two questions are posed:

- What are the obvious causes of impairment?
- If no obvious causes are apparent, what possible causes do the biological data suggest?

Obvious causes such as habitat degradation, point source discharges, or introduced species are often identified during the course of a normal field biological assessment. Biomonitoring programs normally provide knowledge of potential sources of impact and characteristics of the habitat. As such, diagnosis is partly incorporated into many existing State field-oriented bioassessment programs. If more than one impact source is obvious, diagnosis will require determining which impact(s) is the cause of impairment or the extent to which each impact contributes to impairment. The nature of the biological impairment can guide evaluation (e.g., chemical contamination may lead to the loss of sensitive species, habitat degradation may result in loss of breeding habitat for certain species).

Case studies illustrate the effectiveness of biological criteria in identifying impairments and possible sources. For example, in Kansas three sites on Little Mill Creek were assessed using Rapid Bioassessment Protocols (Plafkin et al. 1989; see Fig. 4). Based on the results of a comparative analysis, habitats at the three sites were comparable and of high quality. Biological impairment, however, was identified at two of the three sites and directly related to proximity to a point source discharge from a sewage treatment plant. The severely impaired Site (STA 2) was located approximately 100 meters downstream from the plant. The slightly impaired Site (STA 3) was located between one and two miles downstream from the plant. However, the unimpaired Site (STA 1(R)) was approximately 150 meters upstream from the plant (Plafkin et al. 1989). This simple example illustrates the basic principles of diagnosis. In this case the treatment plant appears responsible for impairment of the resident biota and the discharge needs to be evaluated. Based on the biological survey the results are clear. However, impairment in resident populations of macroinvertebrates probably would not have been recognized using more traditional methods.

In Maine, a more complex problem arose when effluents from a textile plant met chemical-specific and effluent toxicity criteria, yet a biological survey of downstream biota revealed up to 80 percent reduction in invertebrate richness below plant outfalls. Although the source of impairment seemed clear, the cause of impairment was more difficult to determine. By engaging in a diagnostic evaluation, Maine was able to determine that the discharge contained chemicals not regulated under current programs and that part of the toxicity effect was due to the sequential discharge of unique effluents (tested individually these effluents were not toxic; when exposure was in a particular sequence, toxicity occurred). Use of biological criteria resulted in the detection and diagnosis of this toxicity problem, which allowed Maine to develop workable alternative operating procedures for the textile industry to correct the problem (Courtemanch 1989, and pers. comm.).

During diagnosis it is important to consider and discriminate among multiple sources of impairment. In a North Carolina stream (see Figure 5) four sites were evaluated using rapid bioassessment techni-

Figure 4.—Kansas: Benthic Bioassessment of Little Mill Creek (Little Mill Creek = Site-Specific Reference) Relationship of Habitat and Bioassessment

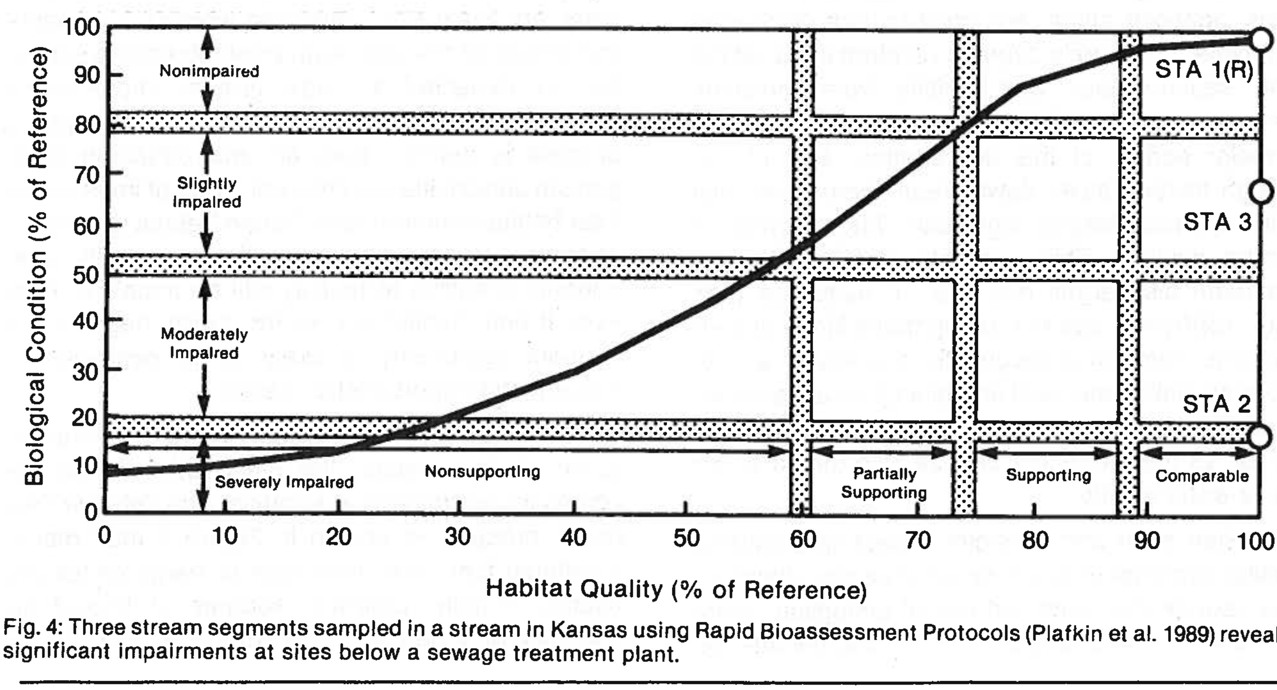


Fig. 4: Three stream segments sampled in a stream in Kansas using Rapid Bioassessment Protocols (Plafkin et al. 1989) revealed significant impairments at sites below a sewage treatment plant.

Figure 5.—The Relationship Between Habitat Quality and Benthic Community Condition at the North Carolina Pilot Study Site.

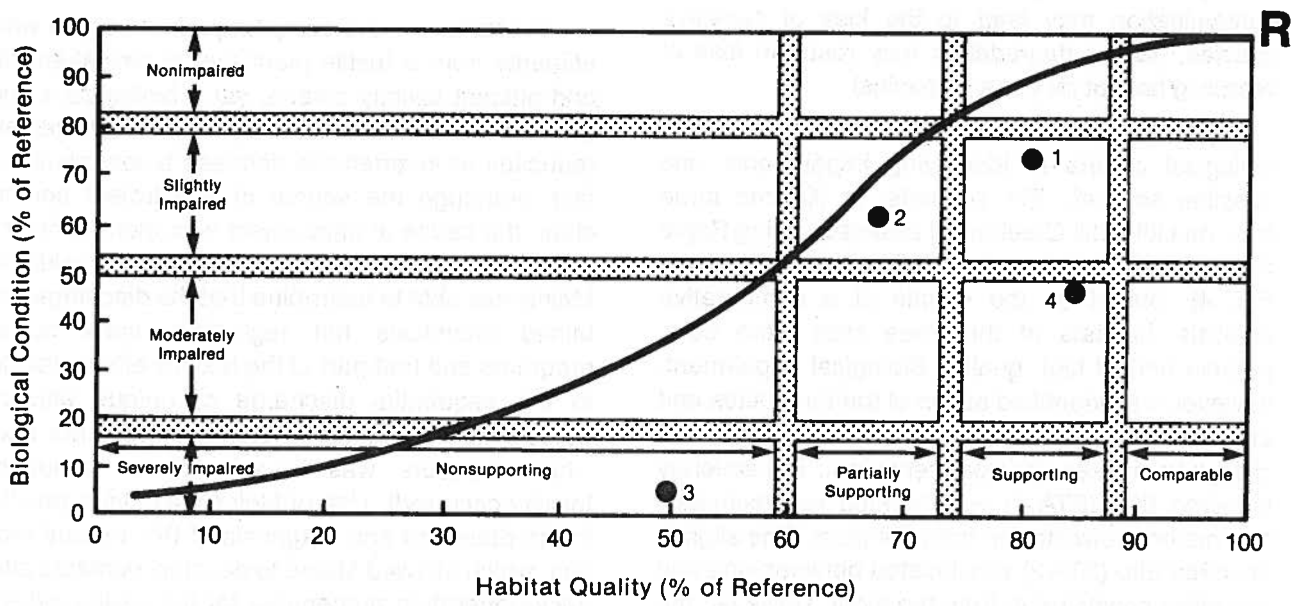


Fig. 5: Distinguishing between point and nonpoint sources of impairment requires an evaluation of the nature and magnitude of different sites in a surface water. (Plafkin, et al. 1989)

ques. An ecoregional reference site (R) established the highest level of biological integrity for that stream type. Site (1), well upstream from a local town, was used as the upstream reference condition. Degraded conditions at Site (2) suggested nonpoint source problems and habitat degradation because of proximity to residential areas on the upstream edge of town. At Site (3) habitat alterations, nonpoint runoff, and point source discharges combined to severely degrade resident biota. At this site, sedimentation and toxicity from municipal sewage treatment effluent appeared responsible for a major portion of this degradation. Site (4), although several miles downstream from town, was still impaired despite significant improvement in habitat quality. This suggests that toxicity from upstream discharges may still be occurring (Barbour, 1990 pers. comm.). Using these kinds of comparisons, through a diagnostic procedure and by using available chemical and biological assessment tools, the relative effects of impacts can be determined so that solutions can be formulated to improve water quality.

When point and nonpoint impact and physical habitat degradation occur simultaneously, diagnosis may require the combined use of biological, physical, and chemical evaluations to discriminate between these impacts. For example, sedimentation of a stream caused by logging practices is likely to result in a decrease in species that require loose gravel for spawning but increase species naturally adapted to fine sediments. This shift in community components correlates well with the observed impact. However, if the impact is a point source discharge or nonpoint runoff of toxicants, both species types are likely to be impaired whether sedimentation occurs or not (although gravel breeding species can be expected to show greater impairment if sedimentation occurs). Part of the diagnostic process is derived from an understanding of organism sensitivities to different kinds of impacts and their habitat requirements. When habitat is good but water quality is poor, aquatic community components sensitive to toxicity will be impaired. However, if both habitat and water quality degrade, the resident community is likely to be composed of tolerant and opportunistic species.

When an impaired use cannot be easily related to an obvious cause, the diagnostic process becomes investigative and iterative. The iterative diagnostic process as shown in Figure 6 may require additional time and resources to verify cause and source. Initially, potential sources of impact are identified and mapped to determine location relative

Figure 6.—Diagnostic Process

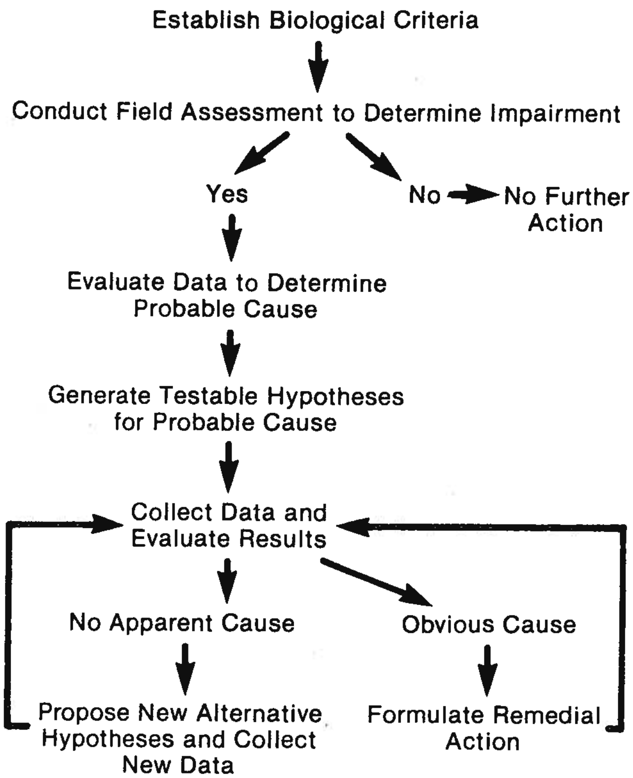


Fig. 6: The diagnostic process is a stepwise process for determining the cause of impaired biological integrity in surface waters. It may require multiple hypotheses testing and more than one remedial plan.

to the area suffering from biological impairment. An analysis of the physical, chemical, and biological characteristics of the study area will help identify the most likely sources and determine which data will be most valuable. Hypotheses that distinguish between possible causes of impairment should be generated. Study design and appropriate data collection procedures need to be developed to test the hypotheses. The severity of the impairment, the difficulty of diagnosis, and the costs involved will determine how many iterative loops will be completed in the diagnostic process.

Normally, diagnoses of biological impairment are relatively straightforward. States may use biological criteria as a method to confirm impairment from a known source of impact. However, the diagnostic process provides an effective way to identify unknown impacts and diagnose their cause so that corrective action can be devised and implemented.

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Appendix A

Common Questions and Their Answers

Q. *How will implementing biological criteria benefit State water quality programs?*

A. State water quality programs will benefit from biological criteria because they:

a) directly assess impairments in ambient biota from adverse impacts on the environment;

b) are defensible and quantifiable;

c) document improvements in water quality resulting from agency action;

d) reduce the likelihood of false positives (i.e., a conclusion that attainment is achieved when it is not);

e) provide information on the integrity of biological systems that is compelling to the public.

Q. *How will biological criteria be used in a permit program?*

A. When permits are renewed, records from chemical analyses and biological assessments are used to determine if the permit has effectively prevented degradation and led to improvement. The purpose for this evaluation is to determine whether applicable water quality standards were achieved under the expiring permit and to decide if changes are needed. Biological surveys and criteria are particularly effective for determining the quality of waters subject to permitted discharges. Since biosurveys provide ongoing integrative evaluations of the biological integrity of resident biota, permit writers can make informed decisions on whether to maintain or restrict permit limits.

Q. *What expertise and staff will be needed to implement a biological criteria program?*

A. Staff with sound knowledge of State aquatic biology and scientific protocol are needed to coordinate a biological criteria program. Actual field monitoring could be accomplished by summer-hire biologists led by permanent staff aquatic biologists. Most States employ aquatic biologists for monitoring trends or issuing site-specific permits.

Q. *Which management personnel should be involved in a biologically-based approach?*

A. Management personnel from each area within the standards and monitoring programs should be involved in this approach, including permit engineers, resource managers, and field personnel.

Q. *How much will this approach cost?*

A. The cost of developing biological criteria is a State-specific question depending upon many variables. However, States that have implemented a biological criteria program have found it to be cost effective (e.g., Ohio). Biological criteria provide an integrative assessment over time. Biota reflect multiple impacts. Testing for impairment of resident aquatic communities can actually require less monitoring than would be required to detect many impacts using more traditional methods (e.g., chemical testing for episodic events).

Q. *What are some concerns of dischargers?*

A. Dischargers are concerned that biological criteria will identify impairments that may be erroneously attributed to a discharger who is not responsible. This is a legitimate concern that the discharger and State must address with careful evaluations and diagnosis of cause of impairment. However, it is particularly important to ensure that waters used for the reference condition are not already impaired as may occur when conducting site-specific upstream-downstream evaluations. Although a discharger may be contributing to surface water degradation, it may be hard to detect using biosurvey methods if the waterbody is also impaired from other sources. This can be evaluated by testing the possible toxicity of effluent-free reference waters on sensitive organisms.

Dischargers are also concerned that current permit limits may become more stringent if it is determined that meeting chemical and whole-effluent permit limits are not sufficient to protect aquatic life from discharger activities. Alternative forms of regulation may be needed; these are not necessarily financially burdensome but could involve additional expense.

Burdensome monitoring requirements are additional concerns. With new rapid bioassessment protocols available for streams, and under development for other surface waters, monitoring resident biota is becoming more straightforward. Since resident biota provide an integrative measure of environmental impacts over time, the need for continual biomonitoring is actually lower than chemical analyses and generally less expensive. Guidance is being developed to establish acceptable research protocols, quality assurance/quality control programs and training opportunities to ensure that adequate guidance is available.

Q. *What are the concerns of environmentalists?*

A. Environmentalists are concerned that biological criteria could be used to alter restrictions on dischargers if biosurvey data indicate attainment of a designated use even though chemical criteria and/or whole-effluent toxicity evaluations predict impairment. Evidence suggests that this occurs infrequently (e.g., in Ohio, 6 percent of 431 sites evaluated using chemical-specific criteria and biosurveys resulted in this disagreement). In those cases where evidence suggests more than one conclusion, independent application applies. If biological criteria suggest impairment but chemical-specific and/or whole-effluent toxicity implies attainment of the use, the cause for impairment of the biota is to be evaluated and, where appropriate, regulated. If whole effluent and/or chemical-specific criteria imply impairment but no impairment is found in resident biota, the whole-effluent and/or chemical-specific criteria provide the basis for regulation.

Q. *Do biological criteria have to be codified in State regulations?*

A. State water quality standards require three components: (1) designated uses, (2) protective criteria, and (3) an antidegradation clause. For criteria to be enforceable they must be codified in regulations. Codification could involve general narrative statements of biological criteria, numeric criteria, and/or criteria accompanied by specific testing procedures. Codifying general narratives provides the most flexibility—specific methods for data collection the least flexibility—for incorporating new data and improving data gathering methods as the biological criteria program develops. States should carefully consider how to codify these criteria.

Q. *How will biocriteria fit into the agency's method of implementing standards?*

A. Resident biota integrate multiple impacts over time and can detect impairment from known and unknown causes. Biocriteria can be used to verify improvement in water quality in response to regulatory efforts and detect continuing degradation of waters. They provide a framework for developing improved best management practices for nonpoint source impacts. Numeric criteria can provide effective monitoring criteria for inclusion in permits.

Q. *Who determines the values for biological criteria and decides whether a waterbody meets the criteria?*

The process of developing biological criteria, including refined use classes, narrative criteria, and numeric criteria, must include agency managers, staff biologists, and the public through public hearings and comment. Once criteria are established, determining attainment\nonattainment of a use requires biological and statistical evaluation based on established protocols. Changes in the criteria would require the same steps as the initial criteria: technical modifications by biologists, goal clarification by agency managers, and public hearings. The key to criteria development and revision is a clear statement of measurable objectives.

Q. *What additional information is available on developing and using biological criteria?*

A. This program guidance document will be supplemented by the document *Biological Criteria Development by States* that includes case histories of State implementation of biological criteria as narratives, numerics, and some data procedures. The purpose for the document is to expand on material presented in Part I. The document will be available in October 1990.

A general *Biological Criteria Technical Reference Guide* will also be available for distribution during FY 1991. This document outlines basic approaches for developing biological criteria in all surface waters (streams, rivers, lakes, wetlands, estuaries). The primary focus of the document is to provide a reference guide to scientific literature that describes approaches and methods used to determine biological integrity of specific surface water types.

Over the next triennium more detailed guidance will be produced that focuses on each surface water type (e.g., technical guidance for streams will be produced during FY 91). Comparisons of different biosurvey approaches will be included for accuracy, efficacy, and cost effectiveness.

Appendix B

Biological Criteria Technical Reference Guide

Table of Contents (*tentative*)

Appendix C

Biological Criteria Development by States

Table of Contents (*tentative*)

Appendix D

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Minnesota River Assessment Project

1990-1992

A FISH COMMUNITY ANALYSIS OF THE MINNESOTA RIVER BASIN

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A FISH COMMUNITY ANALYSIS
OF THE
MINNESOTA RIVER BASIN

Table of Contents

INTRODUCTION

The streams of the Minnesota River basin provide important recreational resources for southern Minnesota. Canoeing and recreating in state and county parks are popular activities along the major watercourses. The Minnesota River from Lac qui Parle Dam downstream to Morton is a component of the Minnesota Wild, Scenic, and Recreational Rivers System.

Sport fishing is also an important activity in the basin. Walleye, channel catfish and northern pike support much of the sport fishery. Flathead catfish and sauger occur in the Minnesota River downstream of Granite Falls. Smallmouth bass are sought in several tributary streams and reaches of the mainstem Minnesota River. There are also 20 designated trout stream reaches on some of the smaller, spring-fed tributaries.

Minnesota Department of Natural Resources(MDNR) records, and a survey of resource managers have indicated impairments to the biological condition and recreational potential of the basin due to nonpoint source pollution (MDNR stream survey files, MPCA 1990). For example, smallmouth bass, a species that is sensitive to habitat and water quality perturbations has declined. A comprehensive fish and aquatic invertebrate survey conducted on the length of the Minnesota River showed smallmouth bass to be absent from 9 of 14 study sectors and to represent only 0.3% of the large fish[1.] species in the overall catch (Kirsch et al. 1985). Investigations of the Blue Earth and Cottonwood rivers, by the Minnesota Department of Conservation, indicated that siltation and flow fluctuations were seriously affecting smallmouth bass populations by the early 1940's (Kuehn 1948a, Kuehn 1948b). Recent efforts to restore a good smallmouth bass fisheries through stocking in these rivers have been largely unsuccessful because of nonpoint source

[1.] Excludes minnows, darters and other small fish species.

pollution (MDNR 1979, MDNR 1981, MDNR 1984, MDNR 1989).

Sedimentation and turbidity are the most obvious impacts of nonpoint source pollution. These problems are the result of cropland erosion, ditching, and field tiling with surface inlets. Removal of natural riparian vegetation by cropping and grazing, and extreme high flows resulting from the artificial drainage networks have caused excessive stream bank erosion which also has contributed to siltation and turbidity.

Recognition of the loss of resource quality has led state and local governments, educational institutions and private citizens to cooperatively seek solutions to these problems. The goal of the Minnesota River Assessment Program (MRAP) was to provide more definitive information regarding impairments occurring in the basin, and the causes and sources of these problems. The MRAP Biology Toxics Subcommittee and the Fish Work Group more specifically identified the need for quantitative evaluation tools and biocriteria to assess biological community health.

Chemical criteria and related chemical monitoring have been the traditional mechanism employed by water quality regulatory agencies for assessing the condition of aquatic life in waterbodies. Significant improvements in water quality have been made using this chemical approach. However, there are many nonchemical factors, such as habitat modifications (e.g., channelization, impoundments), water appropriations, and sedimentation, that cannot be assessed against chemical standards but are impairing aquatic life. In addition, chemical monitoring programs merely provide an instantaneous picture of water quality. Biological communities, on the other hand, are subjected to the cumulative impacts of all activities and are continually integrating the effects of environmental conditions over time. Biological criteria are needed as a benchmark for determining the extent and severity of impairments to biological health, and the effectiveness of remedial activities on improving water resources (USEPA 1987, Karr 1991).

Biological criteria provide an expression of what biological communities should look like under natural or least impacted conditions. In the MRAP study, the Index of Biotic Integrity (IBI) was chosen as the method for establishing biological criteria.

The IBI was originally developed for application in warmwater rivers and streams in the Illinois area (Karr 1981). Since its inception, the IBI has been modified and utilized by several state and federal agencies including US EPA(Simon 1991), Ohio EPA (1987), Illinois EPA (Bertrand and Hite (1989), Wisconsin DNR (Lyons 1992), National Park Service (Fausch 1986) and Tennessee Valley Authority (Saylor and Scott 1987).

The objectives of this study were to 1) adapt and calibrate the IBI for application in the basin, 2) evaluate the biological condition of streams within select first order minor watersheds using the IBI, and 3) assess the biological condition of stream reaches in the Redwood River and Blue Earth River watersheds using the IBI.

STUDY SITES

Fish community sampling was conducted at 116 sites. Study sites fell under three categories.

1) Potential reference sites were sampled to develop expected values for the IBI within the basin. Stream segments chosen were considered to represent the most natural or least impacted condition for the basin. Fifty six potential reference sites were sampled.

Stream segments were chosen where there was extensive riparian vegetation and the stream channel had natural morphological characteristics. Sites were avoided that were downstream of point sources of pollution. Sites were also located to ensure geographic coverage of the basin and selected to represent all stream sizes.

2) Stream segments in 1st order minor watersheds were sampled to determine the biological condition of these resources under low flows. The majority of these sites were selected in coordination with the MRAP land use committee. The sites sampled represent a wide spectrum of conditions in terms of habitat and watershed land use characteristics.

3) Sites were located on the Redwood and Blue Earth rivers and their tributaries to provide an analysis of the extent and severity of impacts to the fish communities within these watersheds. Sampling sites were chosen to represent extended reaches of river where physical characteristics were similar. Some sites were positioned to evaluate the possible impacts of specific dischargers or activities.

METHODS

Field Sampling - Fish

Sampling took place between the middle of June and the beginning of October in the years of 1990, 1991, and 1992. All fish sampling was conducted during daylight hours. Sampling occurred primarily during normal to low flows to provide for personnel safety and maximum gear efficiency. Low flow conditions are usually when aquatic communities are under the greatest stress due to high water temperatures and low dissolved oxygen. In 1992, however, flows throughout the basin remained unseasonably high during the entire summer. Although sampling in 1992 had to be completed during higher than normal flows, periods were avoided immediately after rain events when significant runoff occurred.

The principal method used to capture fish was electrofishing with adjustable, square wave, pulsed DC current. Four different types of gear were employed. The use of any one method was dependent primarily on stream size. Because of highly variable stream conditions, minor variations in electronic parameters were necessary to achieve maximum efficiency.

The gear types used in wadable streams were a battery powered backpack electrofisher (Smith-Root Type VII), an onshore generator/control unit (3000 watt Ac/Coffelt VVP-15) with the anode on a 300′ longline, and a stream shocker utilizing a 15 foot sport canoe with a generator/control unit (3000 watt Ac/Coffelt VVP-15). When the water was very shallow, the longline or backpack gear was used. The stream shocker was used in streams that were deep enough that the canoe could be pulled through the water. In the larger river segments, where depths made it impractical and unsafe for collectors to be in the water, a boom shocker was used. This unit was a Coffelt VVP-15 pulsed direct current boom shocker with two anode rings (20") and a 5000 watt generator.

The crew size varied depending on the gear being employed. A two person crew was used with the backpack shocker. One person carried the unit and held the anode ring (12") and the other person netted the fish. Utilizing the longline, one person operated the generator/control box, one person held the anode ring (16"), and the third person netted. With the stream shocker, there were two anode rings (16"), and a five person crew. One person guided the boat and operated the generator/control unit, two crew held the anode rings, and two netted fish. There were three persons involved in boom shocking, the boat operator and two collectors. All persons in the water wore rubber gloves and waders. The netters wore polarized sunglasses to reduce glare.

Sampling in wadable streams was conducted in a zigzag fashion moving upstream. An effort was made to sample all available habitats with emphasis on important habitats such as undercut banks, snags and rootwads, and around boulders. At riffle areas, nets were posted below the anode rings. Electrode operators kicked loose substrates to mobilize fish. Boom shocking was done in a downstream direction with maneuvering into shore, near submerged structures and other types of habitat. In some locations, a hand held electrode connected to the boom shocker was utilized by wading in riffle areas.

All sites were sampled with a single pass. The length of station varied with size of the stream and diversity of the habitat. Where riffle/pool/run sequences occurred, site length was at least two sequences. Eleven of the sites were resampled at least once during the study to evaluate reproducibility of IBI scores.

All fish captured were identified to species and counted. Each fish was examined for external anomalies. Voucher specimens were collected for each species from each station except for game fish and large specimens where taxonomic identification was common knowledge. Voucher specimens and fish of uncertain identity were preserved and deposited in the University of Minnesota fish collection.

Field Sampling - Habitat Evaluation

Sites were evaluated qualitatively for habitat condition. The habitat survey was accomplished by completing the Qualitative Habitat Evaluation Index, QHEI (Appendix A). The QHEI rates the condition of the habitat in terms of surrounding land use, riparian vegetation, shade, bank condition, channel morphology and substrate, instream cover, water depth, and channel stability. To complete the QHEI, condition, channel morphology and substrate, instream cover, water depth, and channel stability. To complete the QHEI, the stream was walked and measurements of stream width, depth, and length by channel type were recorded, as well as observations of specific habitat attributes. At some sites, due to the depth of the water, it was difficult to thoroughly assess the instream channel, substrate and cover attributes. For these sites, which were the majority of boom shocking sites, a QHEI score was not determined.

Gradient and sinuosity were calculated from measurements made on USGS 1:24000 scale topographic maps. Drainage area of the watershed upstream from the site was determined from Minnesota Land Management Information System (MLMIS) files or from digitizing the watershed boundaries from the USGS maps for very small watersheds.

Flow, conductivity, dissolved oxygen, and temperature measurements were determined at many of the sampling locations when equipment was available.

<u>IBI Metric Development</u>

The IBI is a composite index that evaluates an array of ecological attributes of fish communities. The IBI as originally created by Karr(1981) is comprised of 12 fish community characteristics or metrics. These metrics assess species richness and composition, indicator taxa (tolerant and intolerant), trophic structure, fish abundance, and the incidence of hybridization and external body anomalies. Each of the 12 metrics has a range of sensitivity to environmental degradation. The composite IBI was designed, therefore, to detect differences in environmental quality through a spectrum of conditions from very high quality streams to very degraded streams(Karr et al. 1986).

In developing the IBI for the Minnesota River watershed, Karr's original metrics(1981) and other researchers' modifications were reviewed. These metrics were evaluated the metrics were based. Species known to be present in the basin were classified in regard to those feeding and spawning guilds used in the metrics. Species were also classified in regard to tolerance or intolerance to a wide range of degradation (Appendix B).

Fish species richness and composition vary with stream size. To adjust for this, the values for fish community characteristics calculated from each reference data set was plotted against log transformations of site drainage area. Drainage area is considered a reasonable measure of stream size(Hughes and Omernik 1981). The scatter plots were then examined to determine whether there was a positive relationship with drainage area for the metric, and over what range of stream sizes this relationship occurred. If a positive relationship occurred then a maximum species richness line(MSL) or composition line was drawn with slope fit by eye to incorporate approximately 95% of the data points. The area under the line was then divided equally into 3 sections. Where a positive relationship was not found, an alternative trisection method was used. For those metrics, a horizontal 5% and 95% line was determined and the area between them trisected. This procedure followed Ohio EPA's (1987) methods. Each section was then assigned a score of 5,3 and 1. The section assigned a 5 would be closest to the reference condition, the section assigned a 1 represents those sites that deviate most strongly from the reference conditions.

Development of the MRAP IBI required the utilization of a reference data base that indicates the highest attainable fish community characteristics for the basin. Forty regional reference sites sampled during this project provided the majority of this information. In addition to these regional reference sites, there were data sets obtained from over 650 historical fish collection records. The sources of these records were Professor James Underhill and his colleagues at the University of Minnesota, MDNR stream surveys files, and collections made by Konrad Schmidt.

Data from sixteen of the 56 potential reference sites were dropped for metric development because they were judged to be under significant impacts that made the fish community unrepresentative of a "least-impacted" condition. Several of the sites that were rejected as reference, were below dams that had concentrations of fish that biased the sample. Other sites were found to be significantly impacted by excessive sedimentation. Many of the historical records were excluded for IBI development where there was only information about game fish, or the site sampled was within five miles of the confluence with a lake or much larger river. This data was excluded because of the possible presence of lake species or large river species.

Site and Data Evaluation

Once the metric scoring plots were developed from the reference data base, IBI scores were determined for each site sampled during this study. Each of the 12 metrics were scored 5, 3, and 1, based on where the community characteristic value of a given site fell on each metric plot. The total IBI score was determined by summing all the metric scores together. The total IBI score range was from 12 to 60 or no fish.

A classification scheme was created to describe what the IBI score meant in terms of biological integrity. Such descriptive biological integrity classes provided a means of taking IBI scores and making their results comprehensible to decision makers and the public. Six classes (exceptional, good, fair, poor, very poor, and no fish) developed by Karr et al.(1986) were adapted for this study. The relationship between IBI scores and these narrative classes was determined by best professional judgement. The range and distribution of IBI scores for the reference sites and the entire data base was examined, and classes, with equal IBI score intervals(10 points) were constructed. These classes were then described in terms of what type of fish community was represented relative to the other sites sampled in the basin.

IBI scores were compared for large, midsized, and small streams. QHEI scores were compared between midsized and small streams. Comparisons were made using notched box plots. The notched part of the box indicates the confidence intervals on the median of the scores. If the notched area of a box does not overlap with another, then one can be confident at the 95% confidence level, that the population medians are different (McGill et al. 1978).

The relationship between the IBI and the QHEI for groupings of sites was examined by scatter plots and correlation analysis.

IBI METRICS AND EXPECTED VALUE RESULTS

Description of Metrics and Scoring from Reference Data

The IBI metrics chosen for use in the Minnesota River basin (Table 1) were modified from Karr's original metrics (1981) and were very similar to those adopted by Ohio EPA (1987).

The historical and regional reference site data bases were used to construct the expected values for Metrics 1-5. The historical data set was not used for constructing expected values for Metrics 6-11 because sampling techniques employed by the different collectors varied widely. These metrics are based on relative abundance and catch per unit effort which is greatly affected by sampling gear type and techniques used. Metric 12 was based on best professional judgement because there was limited quantitative information available on the occurrence of external anomalies.

Definitions of the metrics and their expected values are as follows.

Table 1. IBI metrics for the Minnesota River basin

Metric 1.	Total number of native fish species.
Metric 2.	Number of darter species
Metric 3.	Number of sunfish species Dropped for sites < 100 sq. mi.
Metric 4.	Number of sucker species(excluding white sucker) Number of minnow species(excluding common carp, creek chub, fathead minnow)-at sites < 100 sq. mi. drainage area.
Metric 5.	Number of intolerant species
Metric 6.	Proportion of individuals that are tolerant
Metric 7.	Proportion of individuals that are omnivores
Metric 8.	Proportion of individuals that are specialized insectivores
Metric 9.	Proportion of individuals that are top carnivores Dropped for sites < 100 sq. mi.
Metric 10.	Catch per unit effort(Time) by gear type
Metric 11.	Proportion of individuals that are simple lithophils
Metric 12.	Proportion of individuals with deformities, eroded fins, lesions, and tumors (DELT)

Metric 1. Number of native fish species

Species richness, the number of species in a community, is one of the most basic community characteristic measured in ecological studies. The assumption underlying the use of species richness in the IBI is that the number of species in warmwater streams is strongly associated with the complexity and quality of the environment. Environmental degradation whether it be from habitat destruction, flow alteration, or pollution, will be reflected in a simplification of the community structure and a decrease in number of species (Karr et al. 1986). Because certain exotic species can flourish in degraded conditions within the Minnesota River basin, they were excluded from this metric.

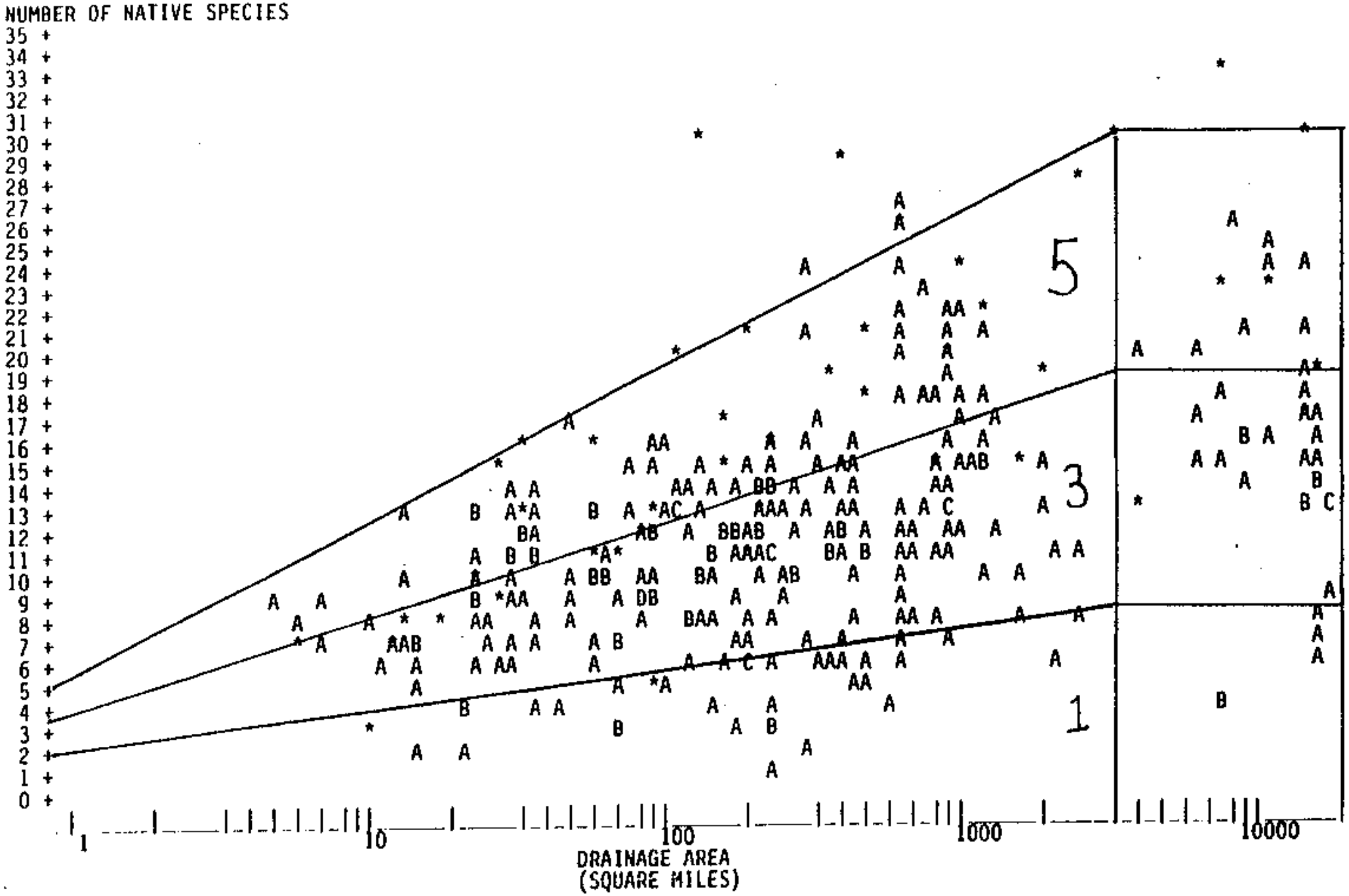
There is a strong positive relationship between the number of native species and stream size up to approximately 3,000 square miles (Figure 1).

Metric 2. Number of darter species

Darters are a group of fishes that belong to the perch family (Percidae). Many darters are habitat specialists, several being adapted to the coarse gravel and rubble substrates of stream riffles. The number of darter species is considered sensitive to changes in condition of water quality (dissolved oxygen, toxicants), as well as habitat quality (Karr et al. 1986; Kuehne and Barbour 1983). This metric responds most sensitively in fair to high quality waters.

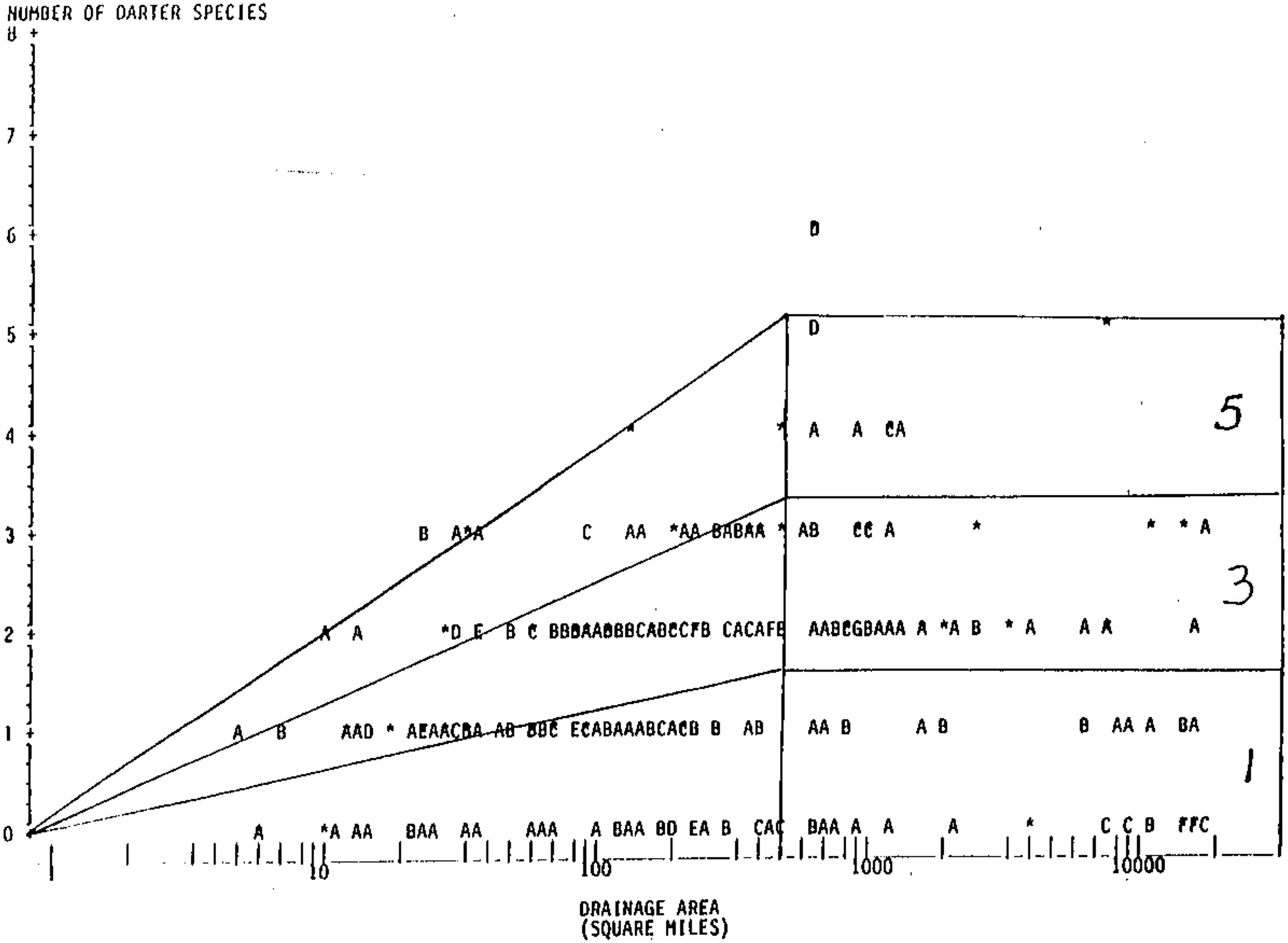
There are 15 species of darters found in Minnesota. Nine species have been recorded in the Minnesota River basin (Underhill 1989).

Plotting reference data against drainage area indicates that the number of darter species is positively correlated with basin size to approximately 500 sq. mi. (Figure 2).



Symbol '*' represents 1990 reference sites
Letters represent historical data where A= 1 obs, B= 2obs, etc.

Figure 1. Number of native species vs. drainage area for determining 5, 3, and 1 scoring.



Symbol '*' represent 1990 reference sites.
Letters represent historical data where A= 1 obs, 2= 2 obs, etc.

Figure 2. Number of darter species vs. drainage area for determining 5, 3, and 1 IBI scoring.

Metric 3. Number of sunfish species

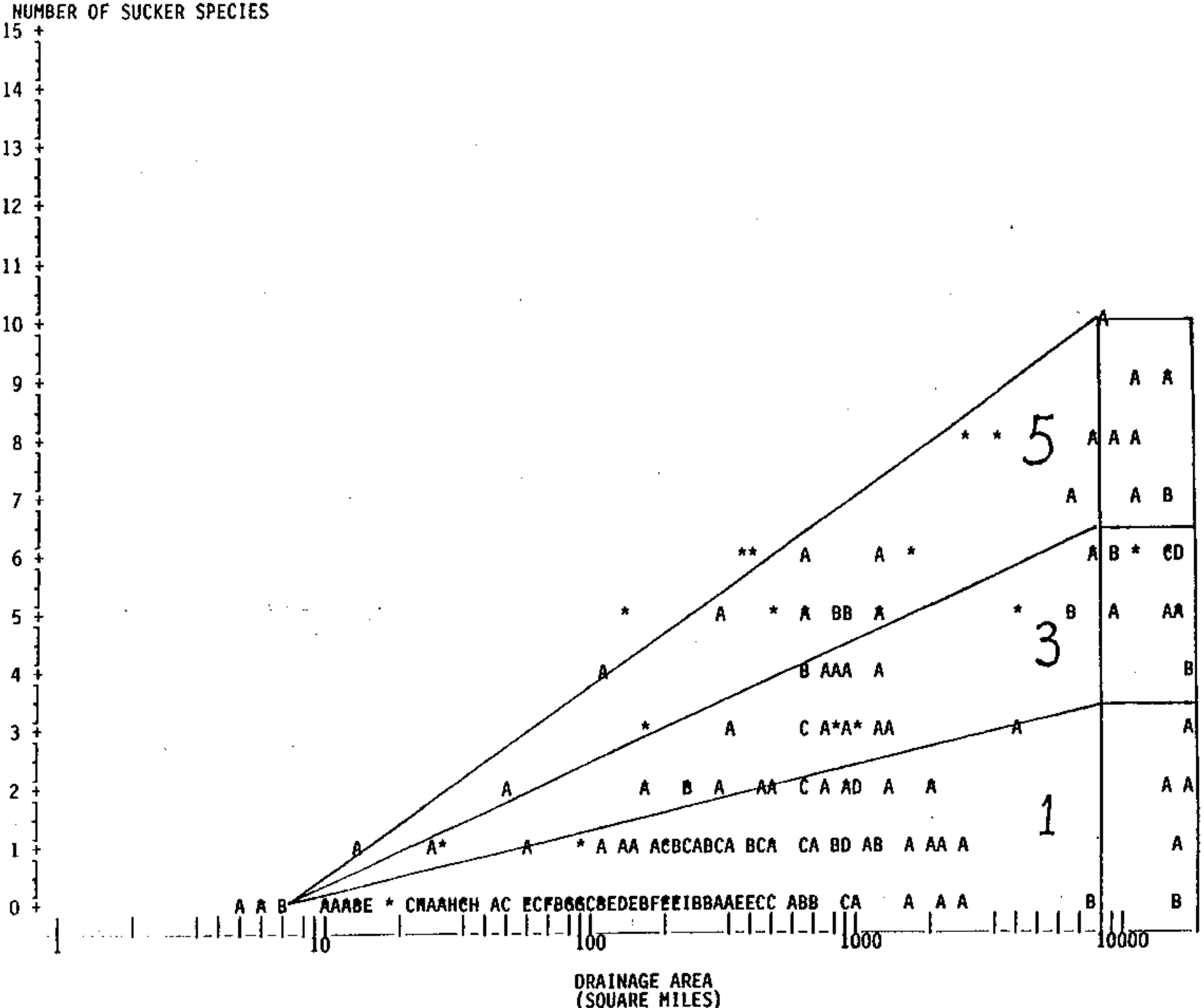
There are eleven sunfish species in Minnesota, nine of which are found in the Minnesota River basin(Underhill 1989). Members of the sunfish family (Centrarchidae) include the basses, crappies, and sunfish. As a group, sunfish species are considered sensitive to degradation of pool habitats and instream cover, and degradation of their preferred food items (Karr et al. 1986). This metric is considered most sensitive to degradation in fair to high quality waters.

The plot of number of sunfish species against drainage area indicated that sunfish were fairly uncommon in smaller streams (Figure 3). This stands to reason considering that the amount of pool habitat is naturally limited in smaller stream segments. For this reason, this metric was dropped for sites with drainage area of less than 100 square miles.

Metric 4. Number of sucker species
At sites > 100 sq. mi.
Number of minnow species
At sites < 100 sq. mi.

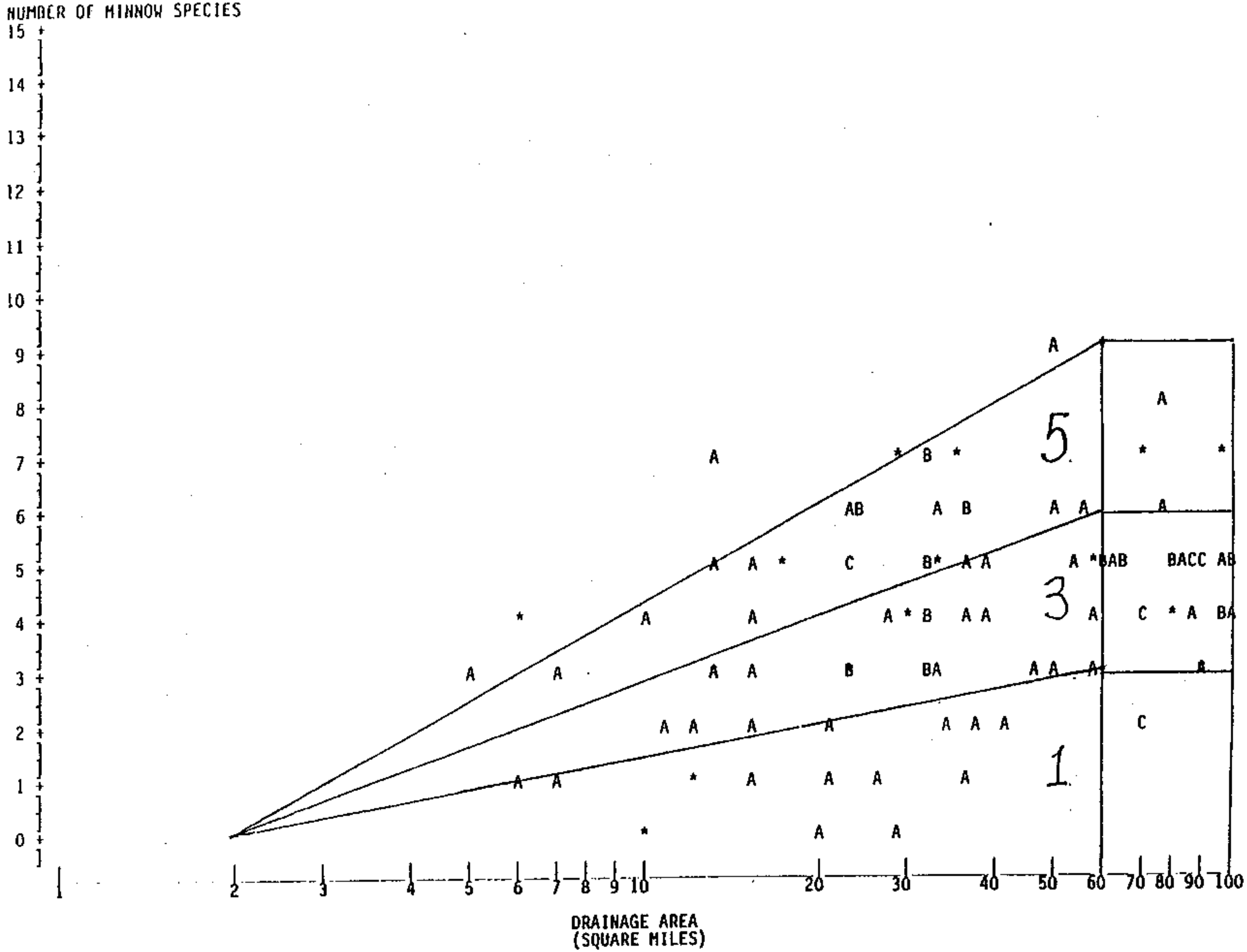
The sucker family (Catostomidae), which includes the redhorses and buffaloes, is the most abundant family in most Minnesota streams and rivers (Waters 1977). There have been thirteen species of suckers recorded from the Minnesota River basin (Appendix B). Many sucker species are considered to be intolerant to both habitat and water quality degradation. This metric is considered most sensitive to degradation in high quality areas. Suckers tend to live for a number of years so this metric also provides a long-term assessment of environmental conditions (Karr et al. 1986). Because the white sucker is considered highly tolerant, we excluded it from the count of sucker species.

There is a positive relationship between drainage area and



Symbol '*' represents 1990 reference sites
Letters represent historical data where A= 1 obs, B= 2 obs, etc.

Figure 4. Number of sucker species vs. drainage area for determining 5, 3, and 1 scoring.



Symbol '*' represent 1990 reference sites.
Letters represent historical data where A= 1 obs, 2= 2 obs, etc.

Figure 5. Number of minnow species vs. drainage area for determining 5, 3, and 1 IBI scoring.

percent occurrence of the species in the Minnesota River basin collections was also considered (Appendix B). In compiling this list, we tried to ensure representation of species that would be found in all sizes of streams. For this reason, our list exceeded the 10% guideline established by Karr et al. (1986).

The graph of number of intolerant species against drainage area indicated a stream size effect (Figure 6).

Metric 6. Proportion of tolerant individuals

This metric is considered most sensitive to degradation in streams that are in fair condition. It has been found that certain species tend to become dominant in streams where habitat degradation or poor water quality is occurring. These tolerant species can survive and thrive in more degraded conditions. For the Minnesota River basin, white sucker, common carp, fathead minnow, creek chub, and black bullhead were considered to be species that possess this characteristic. The combined relative abundance of these five species was used to determine this metric.

Plotting the relative abundance against drainage area indicates a negative relationship with stream size (Figure 7).

Metric 7. Proportion of individuals as omnivores.

Omnivores are defined as species that have a diet that includes at least 25% animal foods and 25% plant foods (Schlosser 1982). Because omnivores are flexible in regard to the food they eat, they generally do better than more specialized foragers in conditions where the food supply is disrupted or degraded. For this reason, omnivores can become dominant in degraded conditions (Karr et al. 1986). This metric is considered most sensitive to conditions in fair to poor quality resources. The combined relative

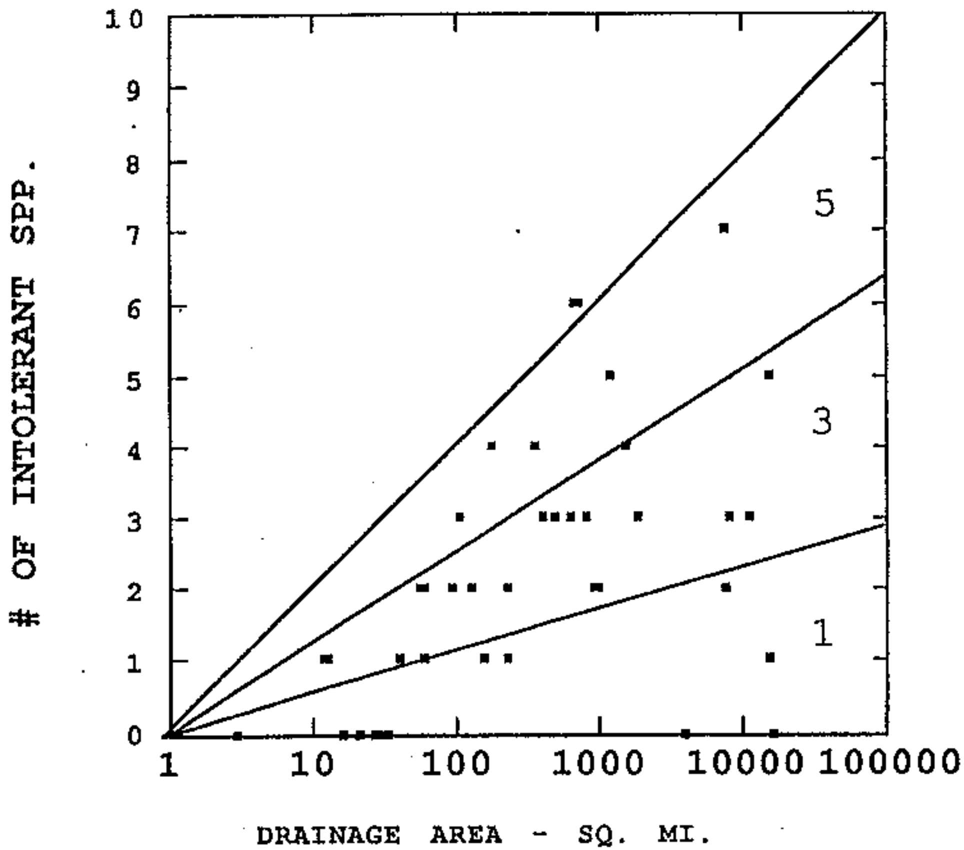


Figure 6. Number of intolerant species versus drainage area for determining 5, 3, and 1 scoring.

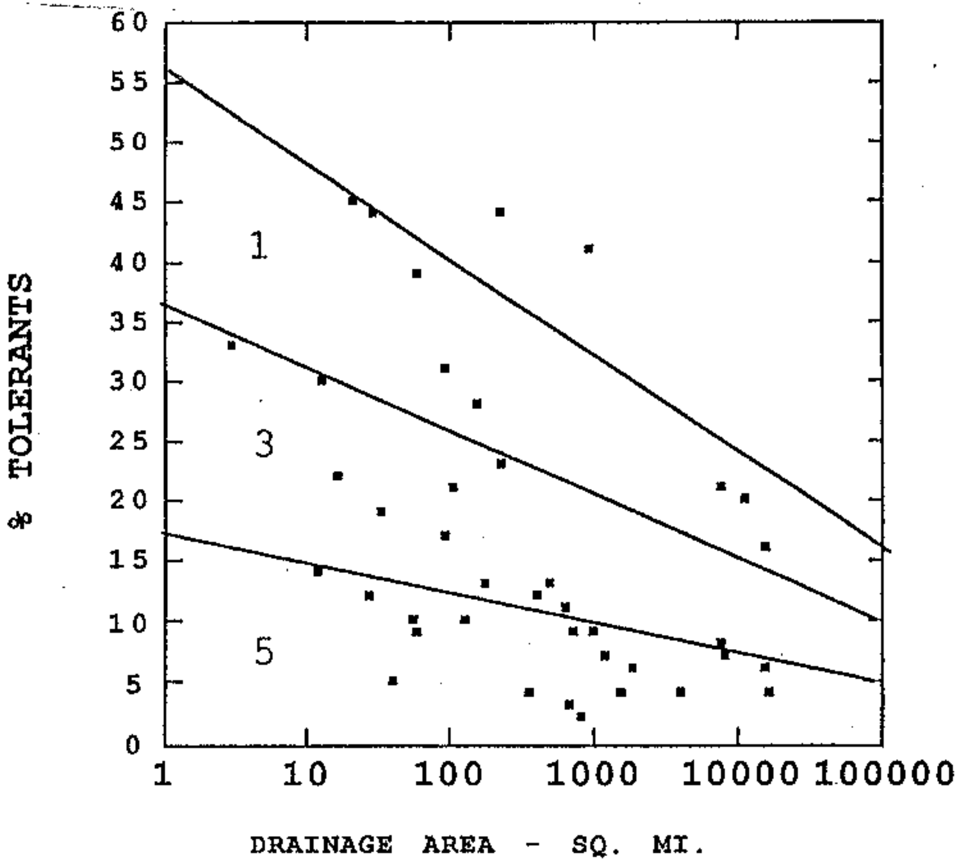


Figure 7. Percent tolerant species versus drainage area for determining 5, 3, and 1 scoring.

abundance of species considered omnivores (Appendix B) was used to construct this metric.

There was no relationship between drainage area and percent omnivores for reference sites. The alternative trisection method was used to determine 5, 3, 1 scores (Figure 8).

Metric 8. Proportion of individuals as specialized insectivores

Just as the proportion of omnivores will generally increase with degraded conditions, the proportion of insectivores, in many cases, will decrease. Fish species that are feeding specialists and restrict their diets to certain benthic insects will respond to changes in these insect populations. Populations of benthic insects can decline due to siltation, poor water quality, and a disruption in energy sources (Karr et al. 1986). All fish species that primarily eat benthic insects were used excluding those that eat large Dipterans (Appendix B).

The relative percent of specialized insectivores increased with drainage area (Figure 9).

Metric 9. Proportion of top carnivores

Top carnivores are those fish that feed on other vertebrates and crayfish (Karr et al. 1986). Relatively high percentages of top carnivore species are considered indicative of high biological integrity. Those species considered top carnivores are listed as such in Appendix B.

A positive relationship exists between percent top carnivores and drainage area (Figure 10). For sites under 100 square miles, the number of top carnivore species was dropped because of their absence or relatively low numbers at these small stream reference sites.

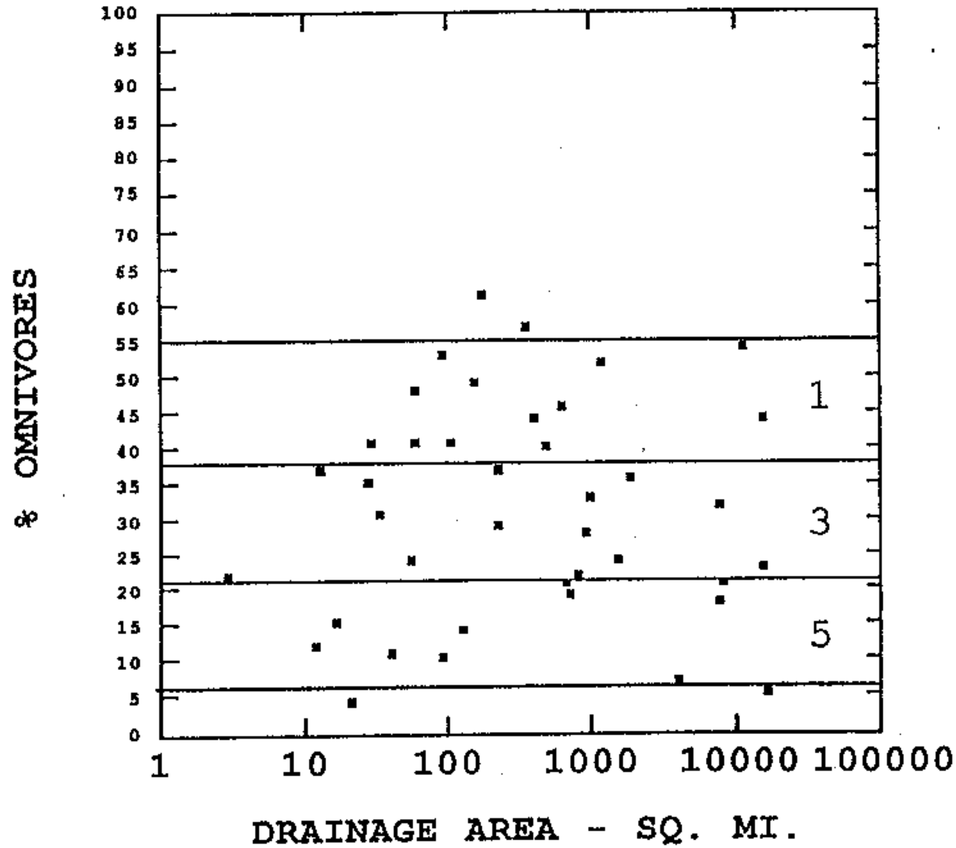


Figure 8. Percent omnivore species versus drainage area for determining 5, 3, and 1 scoring.

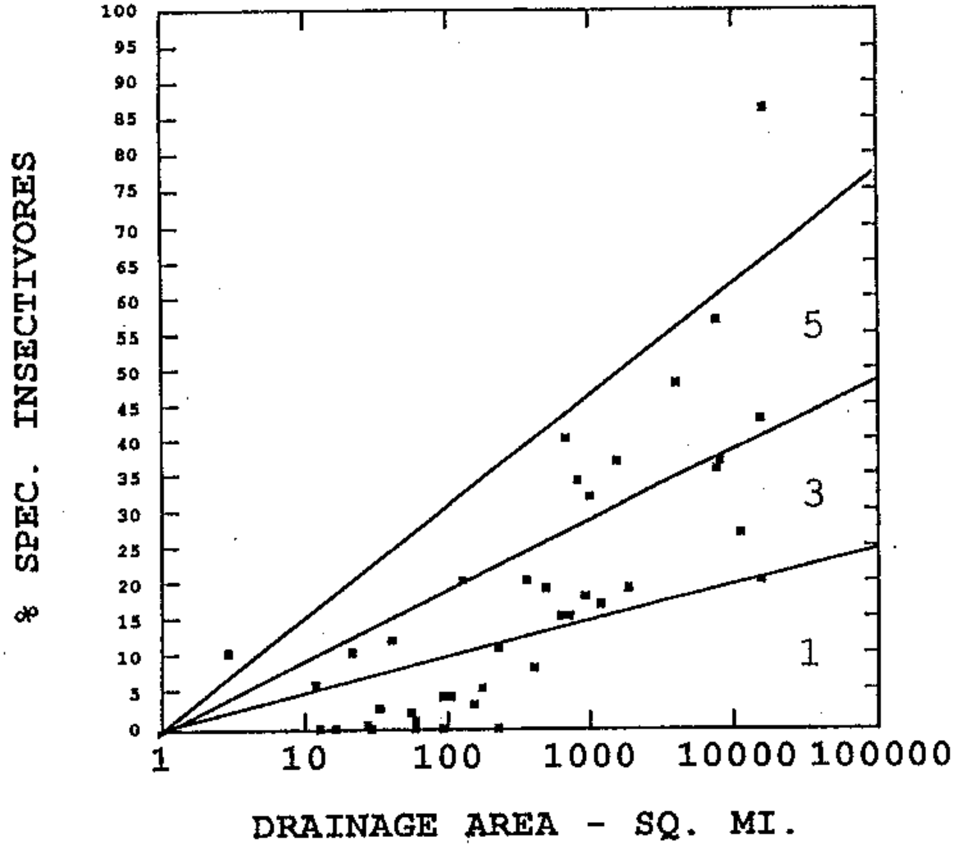


Figure 9. Percent specialized insectivores versus drainage area for determining 5, 3, and 1 scoring.

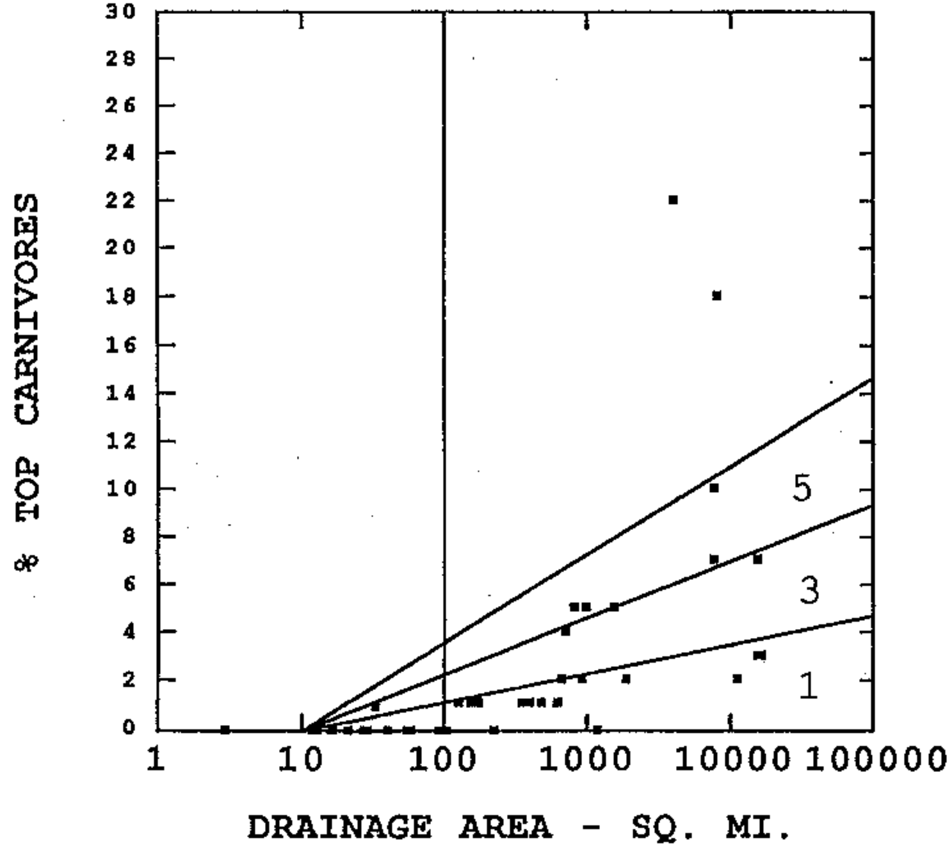


Figure 10. Percent top carnivore species versus drainage area for determining 5, 3, and 1 scoring.

100 square miles, the number of top carnivore species was dropped because of their absence or relatively low numbers at these small stream reference sites.

Metric 10. Number of individuals in a sample expressed as Catch Per Unit Effort (CPUE) - fish per hour

The basis for this metric is the assumption that sites that are degraded will yield fewer individuals than similar sites of higher quality (Karr et al. 1986). Certain perturbations such as channelization accompanied by canopy removal may increase the number of individuals due to an increase of tolerant individuals (Ohio EPA 1987). For this reason, the tolerant individuals are subtracted from the total for determining this metric.

The number of individuals collected at a site is a function of not only the site condition, but also the gear used and amount of effort. To standardize for these factors, reference sites were separated by the three main gear types (backpack, stream shocker or longline, and boom shocker). The CPUE was expressed in terms of the number of fish collected in an hour of effort. Catch rate was not influenced by drainage area for any of the gear types so the alternative trisection method was used (Figure 11, 12, 13).

Metric 11. Proportion of individuals as simple lithophils.

Ohio EPA (1987) introduced proportion of individuals as simple lithophils as a metric to evaluate spawning conditions. Simple lithophilic spawners are those species that exhibit simple spawning behavior and need clean gravel to boulder size substrate for successful reproduction. Berkman and Rabeni (1987) found that the presence of simple lithophils was negatively related to siltation in riffle areas. This metric has value for reflecting the occurrence of sedimentation. Species that are considered to be simple lithophils are indicated in Appendix B. This list followed

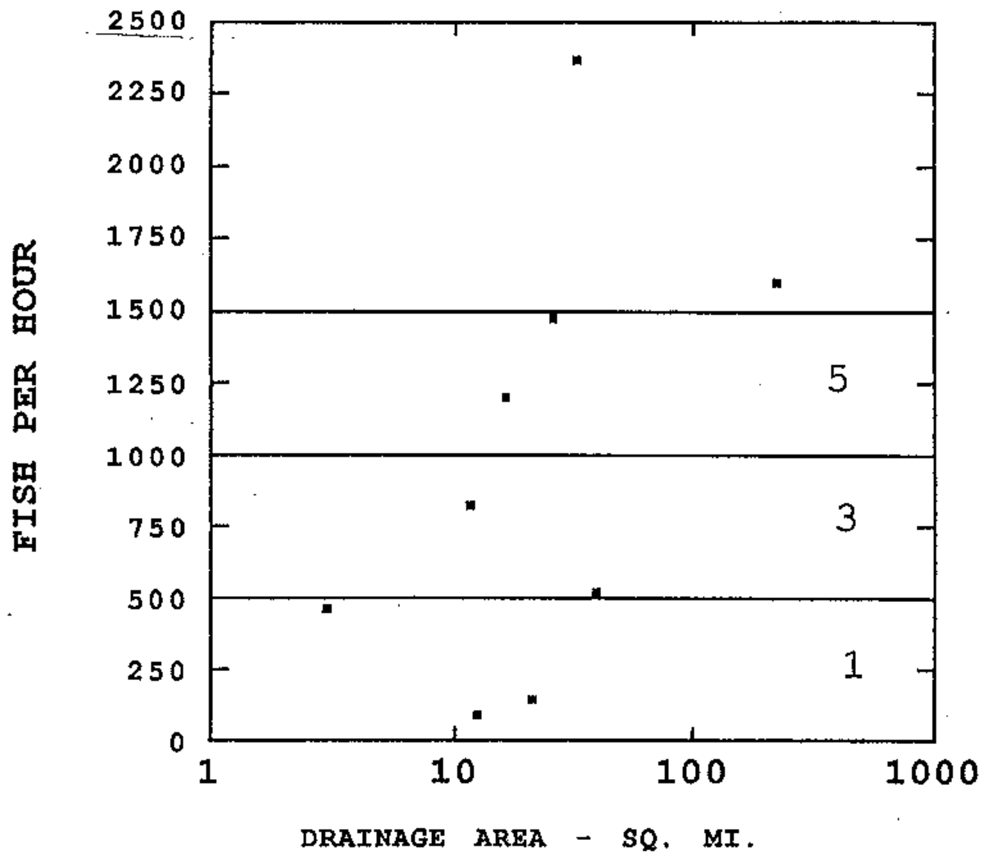


Figure 11. Number of fish per hour (minus tolerant) versus drainage area for determining 5, 3, and 1 scoring. (Backpack sites)

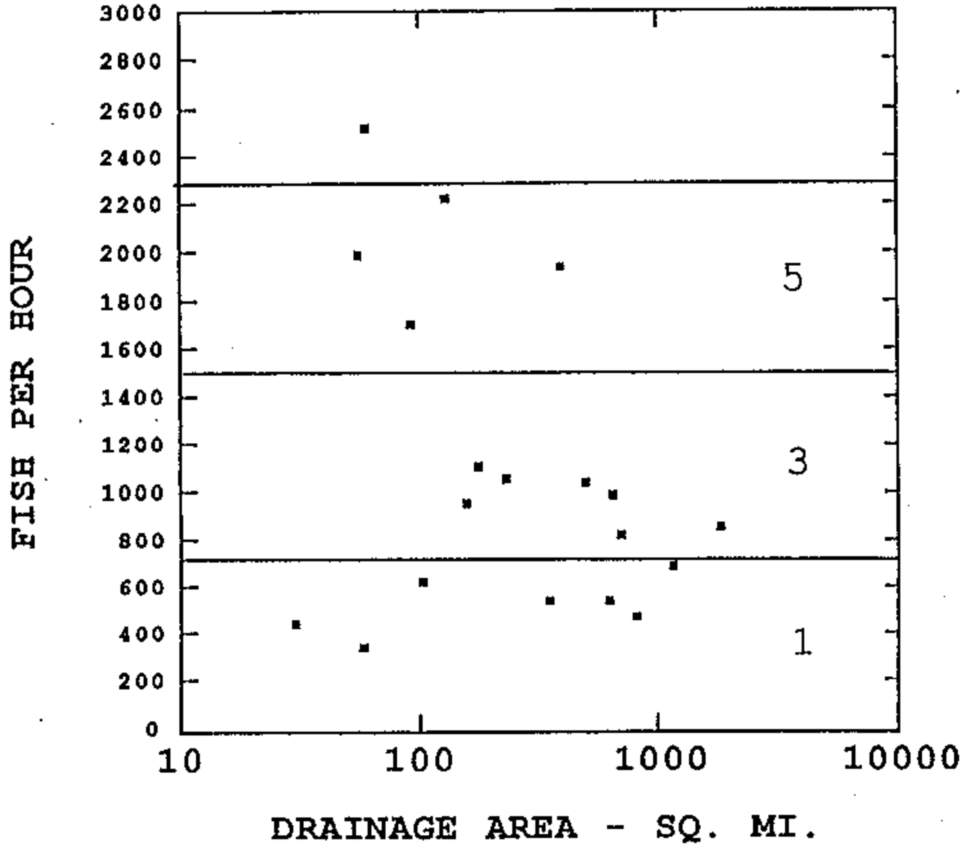


Figure 12. Number of fish per hour (minus tolerant) versus drainage area for determining 5, 3, and 1 scoring. (Stream shocker sites)

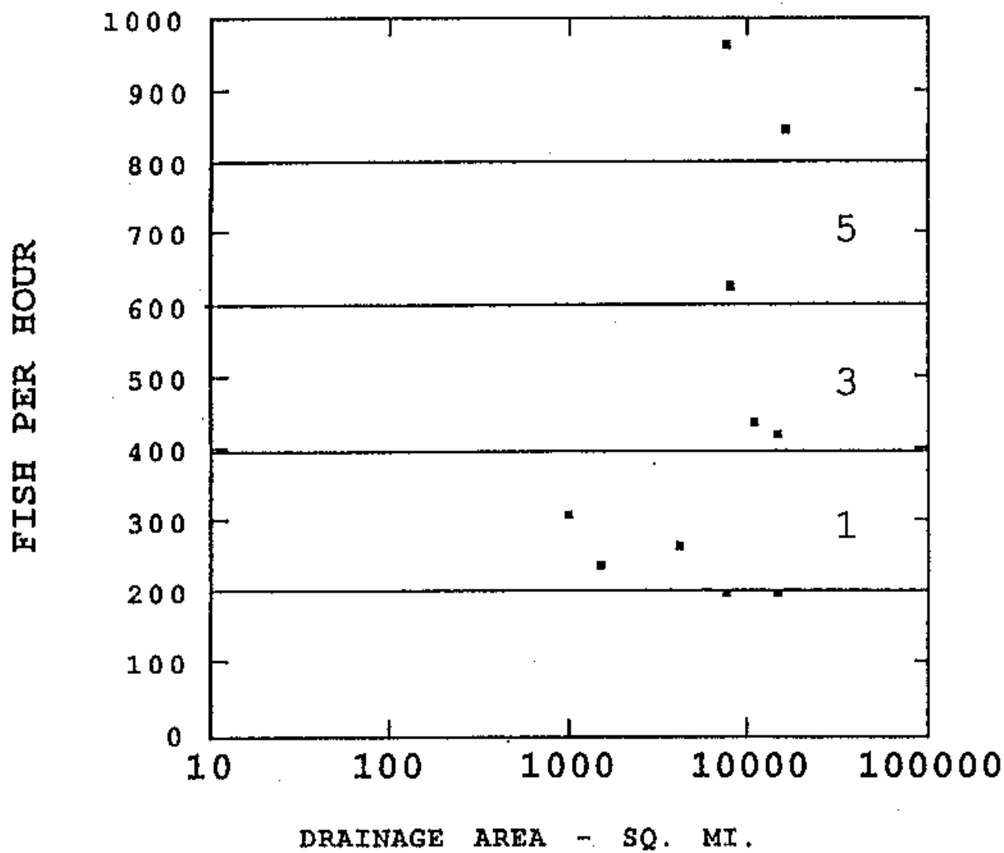


Figure 13. Number of fish per hour (minus tolerant) versus drainage area for determining 5, 3, and 1 scoring. (Boat sites)

indicated no stream size effect (Figure 14). The alternative trisection method was therefore used.

Metric 12. Proportion of individuals with deformities, eroded fins, lesions, and tumors (DELT)

In evaluating stream condition, this metric's primary sensitivity is in significantly degraded environments (Karr et al. 1986). In wild fish populations, the occurrence of most external anomalies is rare or present at low rates but may increase at sites under stress. The types of environmental conditions that are considered to cause an increase in anomalies include chemical pollution, excessive siltation, improper diet and overcrowding. In Ohio, the highest incidence of DELT have occurred downstream of industrial and municipal wastewater plants, and combined sewers and urban runoff (Ohio EPA 1987).

In the historical fish data for the Minnesota River basin, information concerning external anomalies has not generally been recorded. The only information available for metric development was from the sites sampled during this study and the experience of fish biologists consulted. It is generally thought that the incidence of DELT anomalies on Minnesota fishes is rare.

Individuals with external anomalies were found at only four of the reference sites. The proportion of individuals with DELT for these sites ranged from 0.7% to 1.2%. Correlation of percent DELT with stream size could not be determined.

Because any DELT would seem of some concern, a conservative scoring method was used (Fausch et al. 1984). Sites with zero DELT were scored a 5, 0-1% scored 3, and greater than 1% were scored a 1. Blackspot was excluded as a DELT anomaly because it may be a natural condition and not related to environmental degradation (Ohio EPA 1987; Whittier et al. 1987).

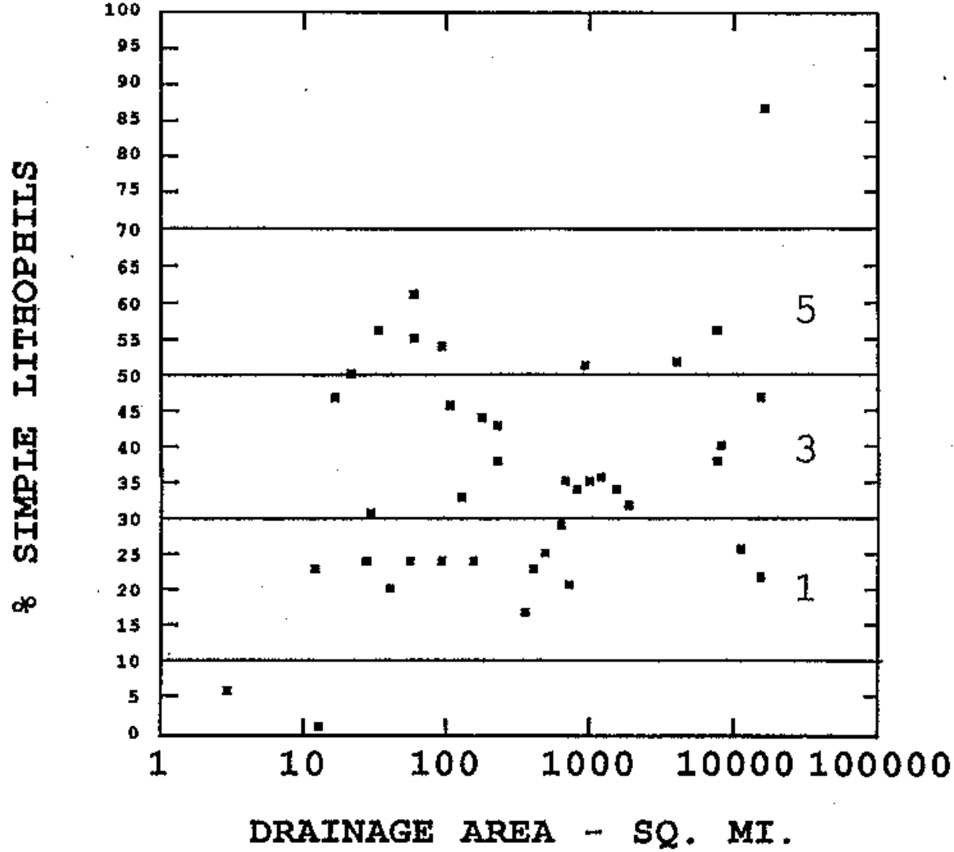


Figure 14. Percent simple lithophils versus drainage area for determining 5, 3, and 1 scoring.

Scoring Modifications

The scoring of a site involved determining the score for each metric from the plots and adding these scores together to get a total IBI score. Because two of the metrics were dropped for sites less than 100 square miles, the sum score of the 10 metrics was multiplied by a factor of 1.2 so that small stream scores would be comparable to other sites.

At some sites, particularly in small streams, there were only a few species and individuals collected. For these sites, the relative abundance metrics may not accurately reflect resource quality because the percent composition can be greatly influenced by the presence of a few individuals. To address this problem, for sites where Metric 1 scored a 1, Metrics 6,7,8, and 11 were also scored a 1.

BASIN STUDY RESULTS

All Sites

The field survey locations selected for this study represent a diverse array of stream conditions across the basin. Study sites varied widely in their respective upstream watershed areas, as well as in the type and quality of riparian vegetation, channel morphology, substrates and instream cover (Appendix C). Although land management practices have highly modified terrestrial vegetative communities and impacted aquatic habitats, a wide range of fish species occur in the basin.

A total of 68 species representing 17 families were collected during this three year study (Table 2). The study did not result in any new species discovered in the basin when compared to the list of 90 compiled from past collections (Appendix B). The shovelnose sturgeon was the only MDNR species of special concern collected during this study.

Table 2. Fish species sampled within the Minnesota River Basin, MRAP 1990-92.

Family	Species	Common name
Petromyzontidae - Lampreys		
	Ichthyomyzon castaneus	chestnut lamprey
	Ichthyomyzon unicuspis	silver lamprey
Acipenseridae - Sturgeons		
	Scaphirhynchus platorynchus	shovelnose sturgeon
Lepisosteidae - Gars		
	Lepisosteus platostomus	shortnose gar
Hiodontidae - Mooneyes		
	Hiodon alosoides	goldeye
	Hiodon tergisus	mooneye
Anguillidae - Eels		
	Anguilla rostrata	American eel
Clupeidae - Herrings		
	Dorosoma cepedianum	gizzard shad
Cyprinidae - Minnows		
	Campostoma anomalum	central stoneroller
	Campostoma oligolepis	largescale stoneroller
	Cyprinella spiloptera	spotfin shiner
	Cyprinus carpio	common carp
	Hybognathus hankinsoni	brassy minnow
	Luxilus cornutus	common shiner
	Nocomis biguttatus	hornyhead chub
	Notemigonus crysoleucas	golden shiner
	Notropis atherinoides	emerald shiner
	Notropis dorsalis	bigmouth shiner
	Notropis heterolepis	blacknose shiner
	Notropis hudsonius	spottail shiner
	Notropis rubellus	rosyface shiner
	Notropis stramineus	sand shiner
	Phoxinus eos	northern redbelly dace
	Pimephales notatus	bluntnose minnow
	Pimephales promelas	fathead minnow
	Rhinichthys atratulus	blacknose dace
	Semotilus atromaculatus	creek chub

Table 2. Continued.

Family	Species	Common name
Catostomidae - Suckers		
	Carpiodes carpio	river carpsucker
	Carpiodes cyprinus	quillback
	Carpiodes velifer	highfin carpsucker
	Catostomus commersoni	white sucker
	Hypentelium nigricans	northern hog sucker
	Ictiobus bubalus	smallmouth buffalo
	Ictiobus cyprinellus	bigmouth buffalo
	Moxostoma anisurum	silver redhorse
	Moxostoma erythrurum	golden redhorse
	Moxostoma macrolepidotum	shorthead redhorse
	Moxostoma spp.	redhorse yy
	Moxostoma valenciennesi	greater redhorse
Ictaluridae - Bullhead Catfishes		
	Ameiurus melas	black bullhead
	Ameiurus natalis	yellow bullhead
	Ictalurus punctatus	channel catfish
	Noturus flavus	stonecat
	Noturus gyrinus	tadpole madtom
	Pylodictis olivaris	flathead catfish
Esocidae - Pikes		
	Esox lucius	northern pike
Umbridae - Mudminnows		
	Umbra limi	central mudminnow
Salmonidae - Trouts		
	Salmo trutta	brown trout
Gasterosteidae - Sticklebacks		
	Culaea inconstans	brook stickleback
Percichthyidae - Temperate Basses		
	Morone chrysops	white bass
Centrarchidae - Sunfishes		
	Ambloplites rupestris	rock bass
	Lepomis cyanellus	green sunfish
	Lepomis humilis	orangespotted sunfish
	Lepomis macrochirus	bluegill
	Lepomis spp.	hybrid sunfish
	Micropterus dolomieu	smallmouth bass
	Micropterus salmoides	largemouth bass
	Pomoxis annularis	white crappie
	Pomoxis nigromaculatus	black crappie

Table 2. Continued.

Family	Species	Common name
Percidae - Perches		
	Etheostoma caeruleum	rainbow darter
	Etheostoma exile	Iowa darter
	Etheostoma flabellare	fantail darter
	Etheostoma nigrum	johnny darter
	Etheostoma zonale	banded darter
	Perca flavescens	yellow perch
	Percina maculata	blackside darter
	Percina phoxocephala	slenderhead darter
	Stizostedion canadense	sauger
	Stizostedion vitreum	walleye
Sciaenidae - Drums		
	Aplodinotus grunniens	freshwater drum

Community composition varied among large river, midsized, and small stream habitats. Sampling at large river sites, defined as sites with a drainage area of greater than 1,000 square miles, produced 54 species representing 15 families. Collectively the most abundant species at these sites in total numbers were emerald shiner, spotfin shiner, sand shiner, shorthead redhorse, and common carp comprising 18%, 13%, 12%, 10%, and 9% of the combined catch respectively. From midsized stream sites, those with drainage areas between 100 square miles and 1000 square miles, 65 species were sampled. The most abundant species in combined total numbers were spotfin shiner, common shiner, central stoneroller, sand shiner, and fathead minnow comprising 12%, 11%, 10%, 8% and 5% of the catch respectively. Forty-six species were sampled from small stream sites. These sites were defined as those less than 100 square miles. As a combined sample, the most abundant species were common shiner, fathead minnow, blacknose dace, central stoneroller, and creek chub comprising 16%, 11%, 10%, 10% and 9.4% of the catch. The total numbers of fish collected at each site and the percent composition and CPUE are reported by watershed in Appendix D.

The IBI scores for the 116 sites ranged from 17 to 55 (Appendix C). At the 18 large river sites, the scores ranged from 30 to 55. The 39 midsized stream sites had IBI scores from 19 to 50. The greatest spread of IBI scores was found within the small stream grouping where the scores ranged from 17 to 55. A sampling at one small stream site produced no fish. Overall, the large river segments had higher scores than the other two size groupings (Figure 15).

Habitat condition, as measured qualitatively with QHEI, was determined at 93 sites (Appendix E). Measurements were not obtained at most of the large river sites. The QHEI is based on a scale of -5 to 100 with higher numbers indicating better habitat quality. QHEI scores for midsized streams ranged from 20 to 74 and at small stream sites from 13 to

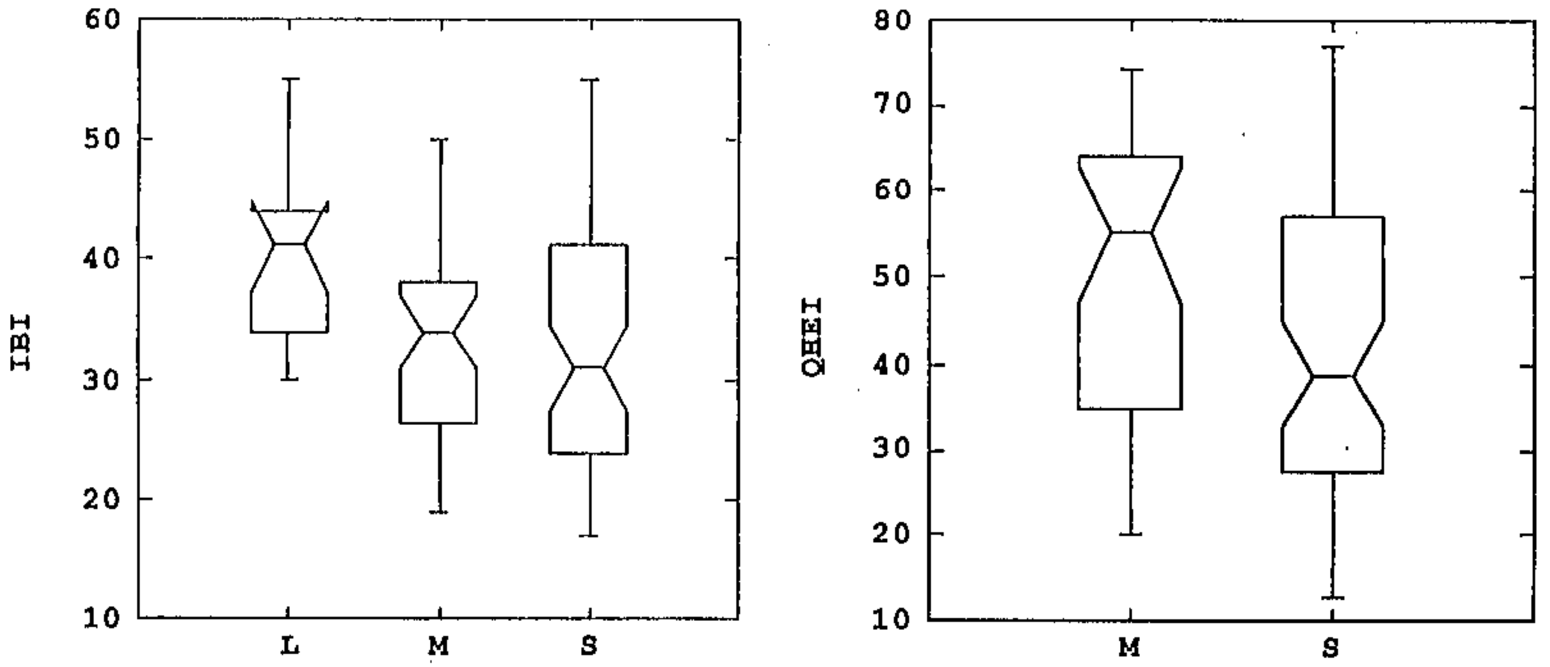


Figure 15. Notched box plots of IBI and QHEI scores of all sites by size groups. Notch overlap between size groups indicates that the median values are not significantly different (P<0.05). L=large, M=midsized, and S=small

77. The QHEI scores were higher overall at the midsized stream sites (Figure 15).

There was a strong positive relationship between the IBI and QHEI for the 32 midsized stream sites (Figure 16, Spearman Rank Correlation Coefficient=0.62, p<0.001). There was also a positive correlation between IBI and QHEI for the small stream sites (Figure 17, Spearman Rank Correlation Coefficient=0.45,p<0.001). However, there was a weaker relationship observed for these sites.

Reference Sites

The 40 sites that were considered reference sites (Appendix F, Map 1) had IBI scores that ranged from 24 to 54. The distribution of scores by stream size group indicated that the widest range in scores were found at the small stream reference sites. The median values for all size groupings were similar (Figure 18). QHEI scores ranged from 37 to 77. The spread of QHEI scores was greater for the small stream reference sites than the midsized stream sites (Figure 18).

IBI Variability

Eleven sites were visited more than once. Differences in IBI scores at these sites ranged from 0 to 10 points with a mean difference of 4 points (Table 3). For these sites, it was assumed that no significant change in environmental condition had occurred between sampling times. The fluctuations in score was therefore considered a reflection of sampling error and natural variability.

Biological Criteria and Biological Integrity Classes

The distribution of IBI scores for the reference sites and all sites by equal class intervals were determined (Figure 19). The majority of reference sites fell into the IBI 30-39 class. The majority of all sites sampled, which also includes the reference sites, fell within the IBI 30-39 class. Based on these ranges of scores, it appears reasonable to set a biological criterion at an IBI score of

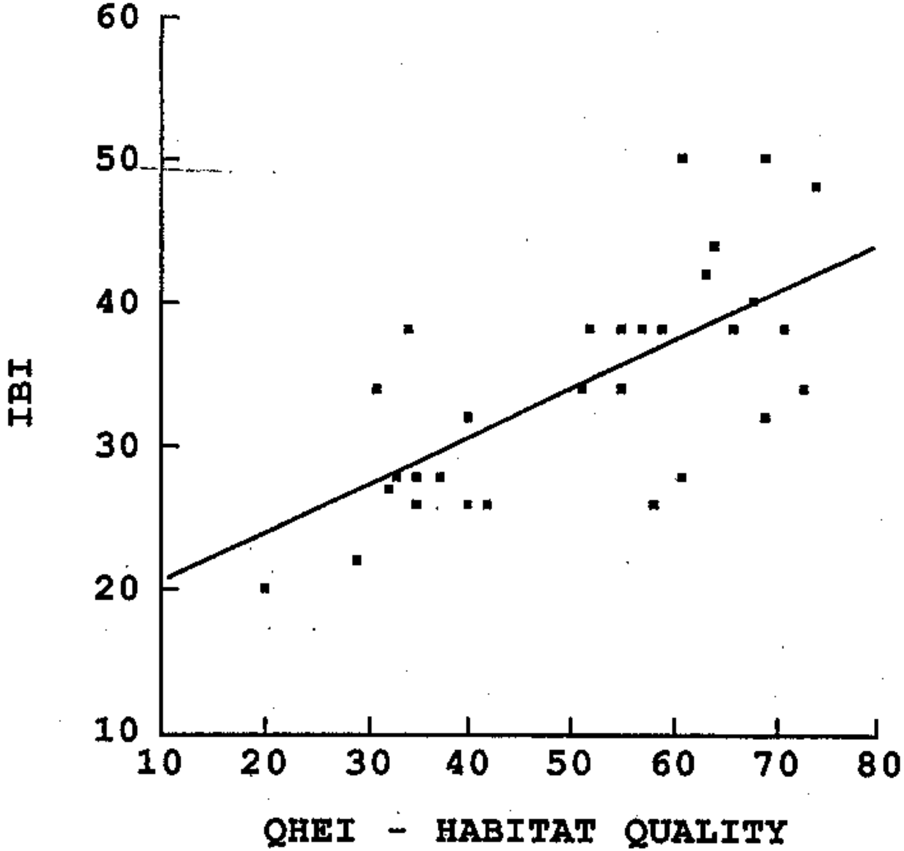


Figure 16. Plot of overall IBI score versus QHEI score for midsized stream sites.

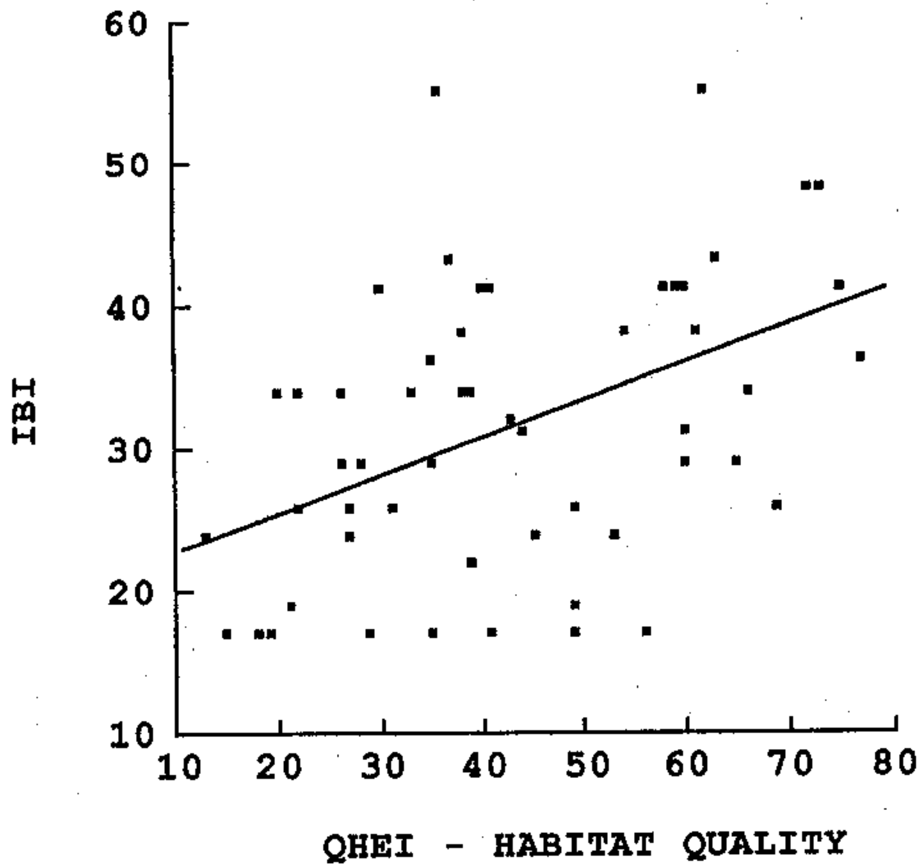


Figure 17. Plot of overall IBI score versus QHEI score for small stream sites.

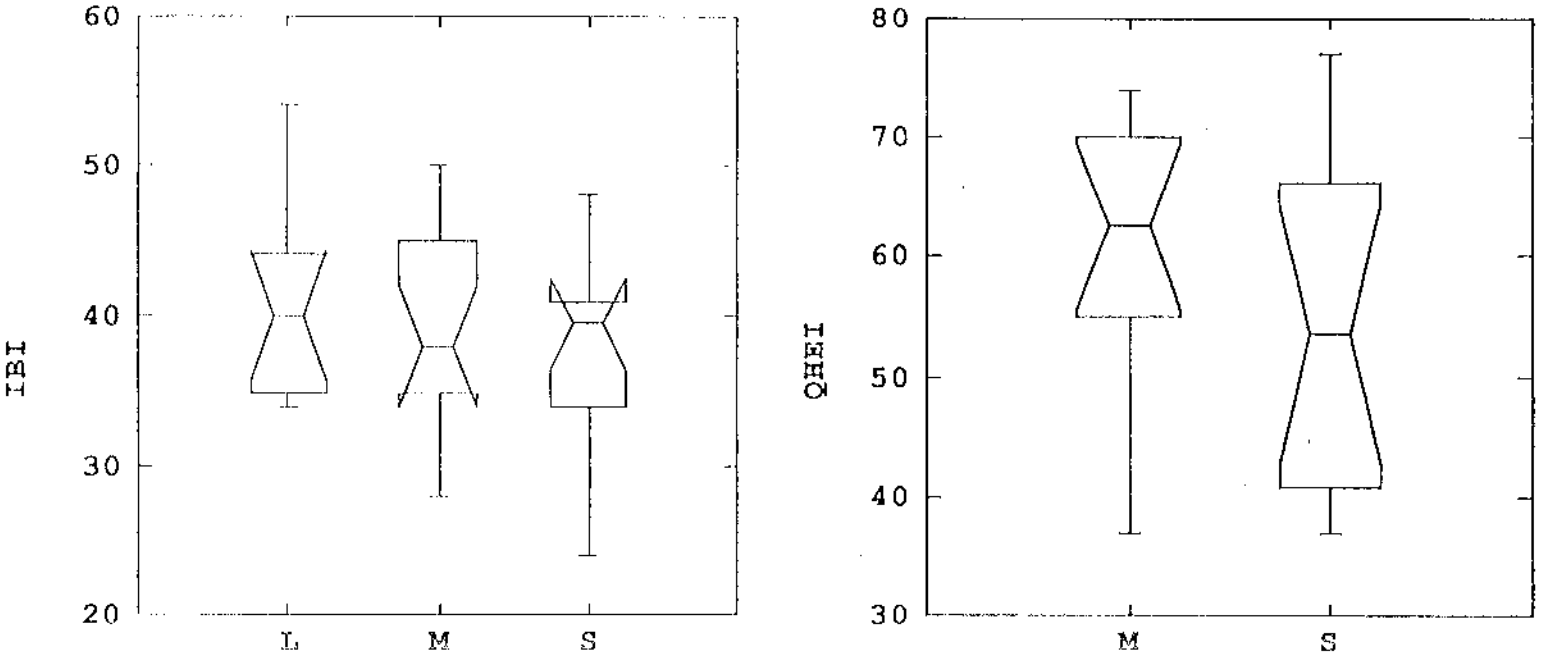


Figure 10. Notched box plots of IBI and QHEI scores of reference sites by size group. Notch overlap between size groups indicates that the median values are not significantly different (P<0.05). L=large, M=midsized, and S=small

Table 3. IBI scores from sites that were sampled more than once during the 1990-1992 project period.

Stream	Site Name	1990	Year 1991	1992	Difference in score
Redwood	RWR-81	38		40	2
Hawk	HWK-8	38		34	4
Blue Earth	BE-2	50		50	0
Blue Earth	BE-11	56		46	10
West Blue Earth	WBE-2	28		32	4
Credit	CRD-10	38		36/31	7
Camp Pope Creek	CAP-1		36	34	2
Mound Creek	MOU-17.4		48	53	5
Brush Creek	BRH-5	38		34	4
Buffalo Creek	BUF-1	41		22*/38	3
Silver Creek	SIL-3	41		36	5

* Score was determined from a sample that was taken under high flows. For this reason, this IBI score was not included in determining the mean difference.

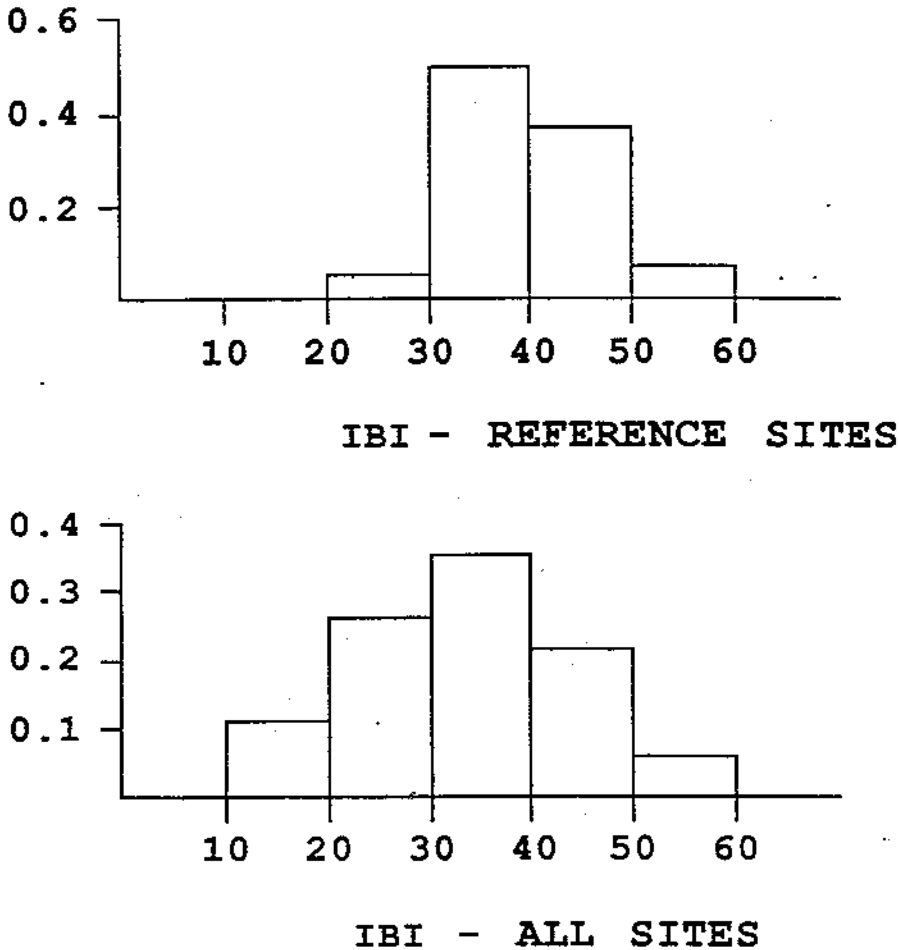


Figure 19. Proportion of sites in each biological criteria class.

30 for rivers and streams in the Minnesota River basin.

Sites scoring lower than 30 would represent stream reaches that deviate considerably from basin reference expectations. These sites, therefore, should be considered not supporting the biological criteria here established for the basin and should be considered not supporting the aquatic life goals of the Clean Water Act(US EPA 1992).

Based on knowledge of fish community characteristics of the sites that comprise each class, the classes themselves can be more narratively described for the basin (Table 4). According to this narrative classification scheme, 6.2% of all sites had excellent quality, 21.2% were rated good, 35.4% were fair, 25.7% had scores in the poor range, 10.6% were very poor, and 0.9% had no fish. Thirty-seven percent of the sites did not meet the IBI 30 criteria. The majority of these sites were in smaller watersheds (<100 sq. mi.).

SPECIFIC WATERSHED STUDY RESULTS

<u>Minor Watershed Study</u>

Thirty-five very small feeder streams were sampled in first order minor watersheds selected throughout the basin. The majority of these sites had drainage areas of less than twenty-five square miles. These sites were sampled in conjunction with other MRAP cooperators and the information from the fish community will be related with land use, water chemistry data and other biological information in another report.

A total of thirty-three species of fish were collected in these first order minor watersheds. The Cyprinidae family were the community dominants in these streams with 14 species represented. The white sucker was the only sucker species collected consistently. A shorthead redhorse was present in one sample. Five species of darters were recorded. The sunfish family was represented by the green

Table 4. Biological Integrity Classes

Integrity Class	IBI	Range Characteristics
Excellent	50-60	Comparable to the best sites in the region for that stream size; exceptional assemblage.
Good	40-49	Some decrease of species richness, especially of sensitive forms.
Fair	30-39	Decrease in sensitive species; trophic structure more skewed and omnivores more dominant.
Poor	20-29	Top carnivores and many other expected species absent or rare; omnivores and tolerant species dominant as do habitat generalists.
Very Poor	12-20	Few species and individuals present; tolerant dominant.
No Fish		Sampling finds no fish.

Modified from Karr et al. 1986.

sunfish, orangespotted sunfish and bluegill. Both the black bullhead and yellow bullhead were sampled as well as the stonecat. Single individuals of northern pike, walleye and yellow perch were found at a few sites. For these 36 sites, IBI scores ranged from 17 to 55 with a mean value of 30. At one site, no fish were collected.

The physical setting varied considerably among sites. Fifty-five percent of the sites had adjacent land use as row crop. Twenty-seven percent of the sites had adjacent land cover of more natural vegetation. The remaining sites had pastures or old field as the surrounding land use. Wide riparian corridors tended to be extensive in the natural areas. Sites located in cultivated areas had riparian corridors that were moderate (30′-150′) to very narrow (3′-15′). Many of the stream segments in the cultivated areas were county or judicial ditches that were channelized. These ditched segments generally had undifferentiated channel morphology and the substrates were dominated by silt and sand. In the more natural areas, pool and riffle sequences were present with coarser substrates typically found. The QHEI scores varied from 13 to 77 with a mean value of 38. There was a positive correlation of IBI to QHEI scores (Figure 20 Spearman Rank Correlation Coefficient = 0.63, p=0.001).

Redwood River

A total of 40 fish species representing nine families were collected in the Redwood River watershed. Families that had the greatest representation in terms of species included the minnows (13), the perch (8), the suckers (6), and the sunfish (5).

Fish were sampled from 15 main stem sites (Appendix D, Table E; Appendix F, Map 2). The fish community of the river was dominated in numbers by the common shiner (18%), central stoneroller (17%), spotfin shiner (8%), and fathead minnow (6%).

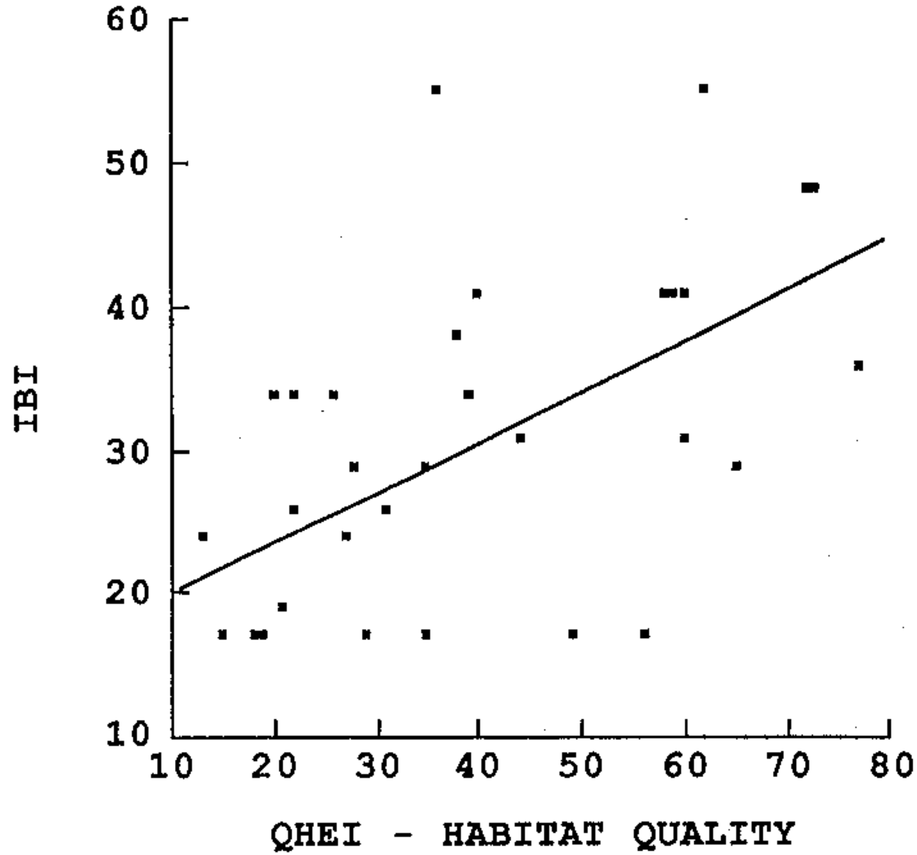


Figure 20. Plot of overall IBI score versus QHEI score for 1st order minor watershed sites.

The overall quality of the fish assemblage, as represented by the IBI, ranged from exceptional to poor (Figure 21). The only site considered exceptional was below Ramsey Creek in the lower Redwood River at river mile (RM 4). The site in Camden State Park (RM 81) had overall fish community integrity in the good range. Two other sites, RM 99 and RM 9, had IBI scores that rated in the good range. Downstream from Marshall (RM 61 to RM 13), the IBI scores from 8 sites indicated an overall fish community integrity of fair to poor.

The relative abundance of top carnivore species was very low at all sites on the main stem and only sites at RM4 and RM61 scored greater than 1 for the top carnivore metric. At several sites on the mainstem, the trophic structure appeared to be dominated by omnivores.

Several species considered intolerant to environmental degradation, however, were present at certain sites on the river. Greater redhorse were found in the lower portion of the river. Northern hog sucker were found at sites on the Coteau des Prairie and in the lower river. Smallmouth bass, rainbow darter, and slenderhead darter were sampled at the site below Redwood Falls.

Habitat quality in the mainstem Redwood River varied considerably with QHEI scores ranging from 31 to 69 (Figure 22). The highest scores were found at sites in the lower river from RM13 downstream (QHEI > 60), and at those stream segments through the Coteau des Prairie (RM71 to RM88). In these areas, the channel morphology was well developed with riffles and pools. The substrates in these segments were heterogeneous with larger particle sizes, including boulder, cobble and gravel being well represented. Instream cover was provided by boulders, woody debris, and some deeper pools. Sites in the mainstem river downstream from Marshall (RM61 to RM23) had generally poorer habitat quality. This was reflected in the QHEI scores that were

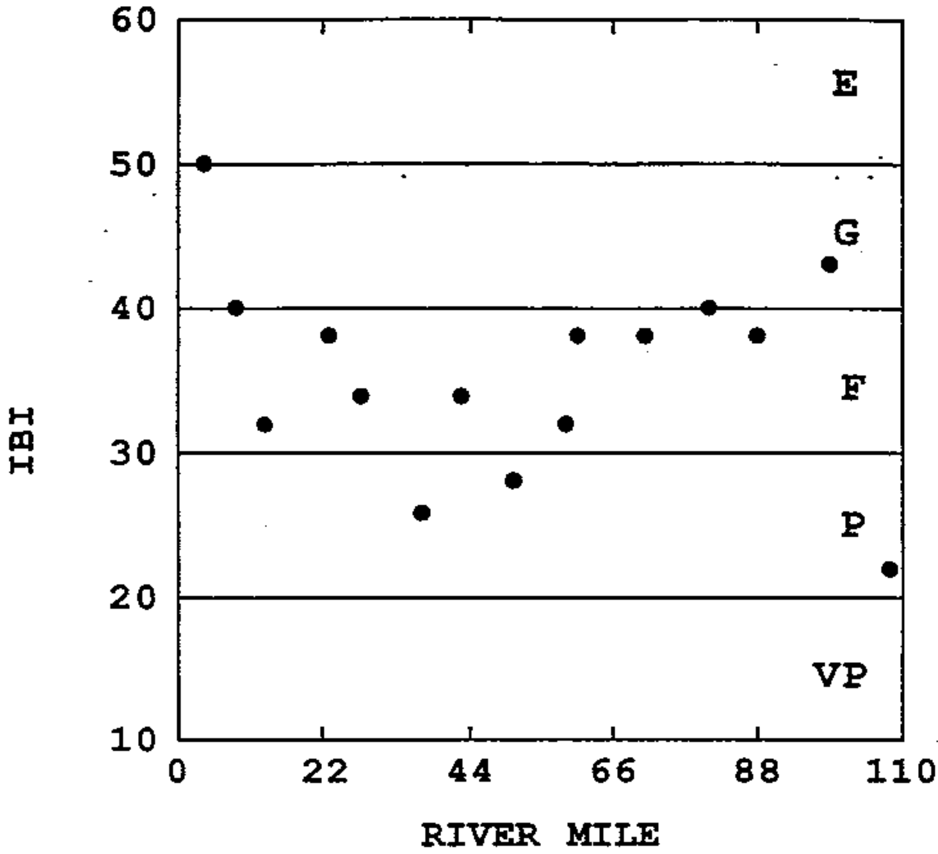


Figure 21. IBI scores for Redwood River main stem sites.
E= exceptional, G= good, F=fair
P= poor, VP = very poor

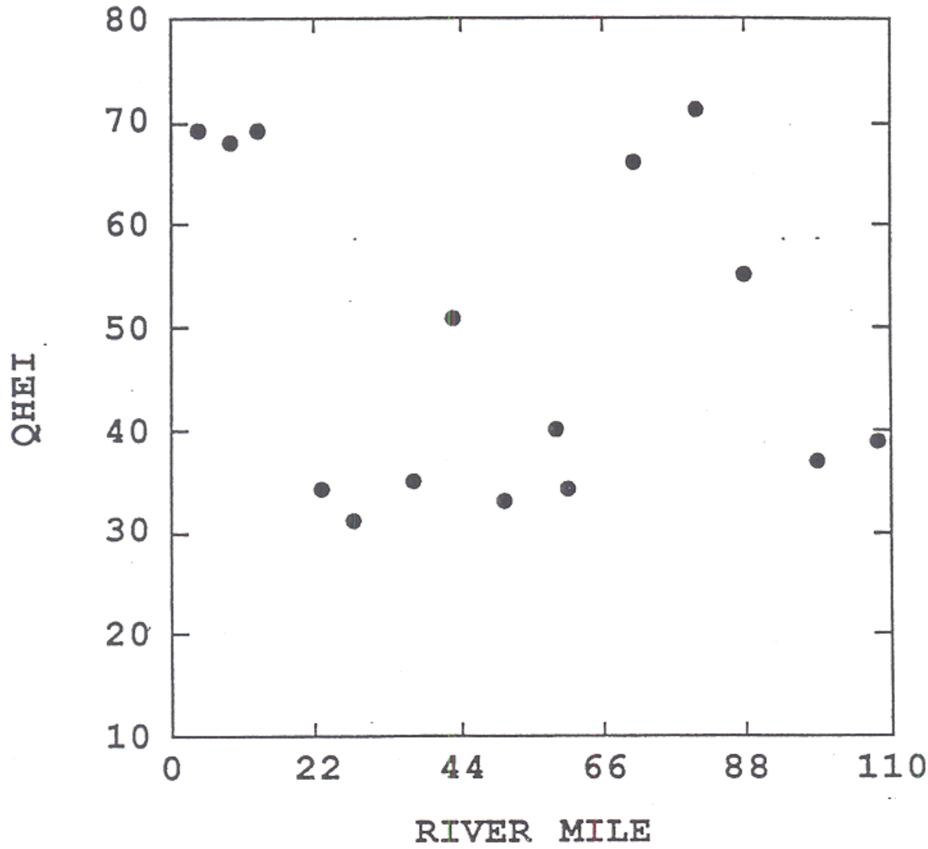


Figure 22. QHEI scores for Redwood River main stem sites.

generally less than 40. The larger particle size substrates of the river bottom were embedded by sand and silt.

Fourteen tributary sites were sampled in the Redwood River watershed (Appendix D, Table F; Appendix F, Map 2). The samples were dominated by minnow species with common shiner (17%), blacknose dace (13%), and creek chub (12%) being the most abundant. There were two sites considered to have very poor fish assemblages, four sites were rated poor, four sites rated fair, and four sites were rated good. There was no apparent trend in IBI scores with stream size (Figure 23). The habitat quality of the tributary sites varied considerably. The QHEI scores ranged from 22 to 75 with a mean of 44. The highest scores were found in the lower reach of Ramsey Creek (RAM-2, RAM-.2) where it enters the Minnesota River valley. In this section the stream had good riffle and pool development with abundant cover. The most downstream sites sampled on Tyler Creek (TY-2), Coon Creek (COO-3), and Clear Creek (CL-1) also had good channel development and diverse substrates.

The sites sampled in the upper segments of most of these streams (TY-6, CL-7, COO-24, NOW-2) were dominated by fine sediments. Tyler Creek, Clear Creek, and Ramsey Creek were also completely channelized in their upper portions as was County Ditch 33. The three sites sampled on Three Mile Creek were embedded with fines and had fair to poor channel development.

<u>Blue Earth River</u>

Fifty three species representing fourteen families were collected from the Blue Earth River watershed. Families that had the greatest representation of species included the minnows (12), the suckers (9), the perch (9), the sunfish (5) and the bullhead (5). This does not include sites within the LeSueur and Watonwan River watersheds.

Ten stations were sampled in the main stem of the Blue Earth

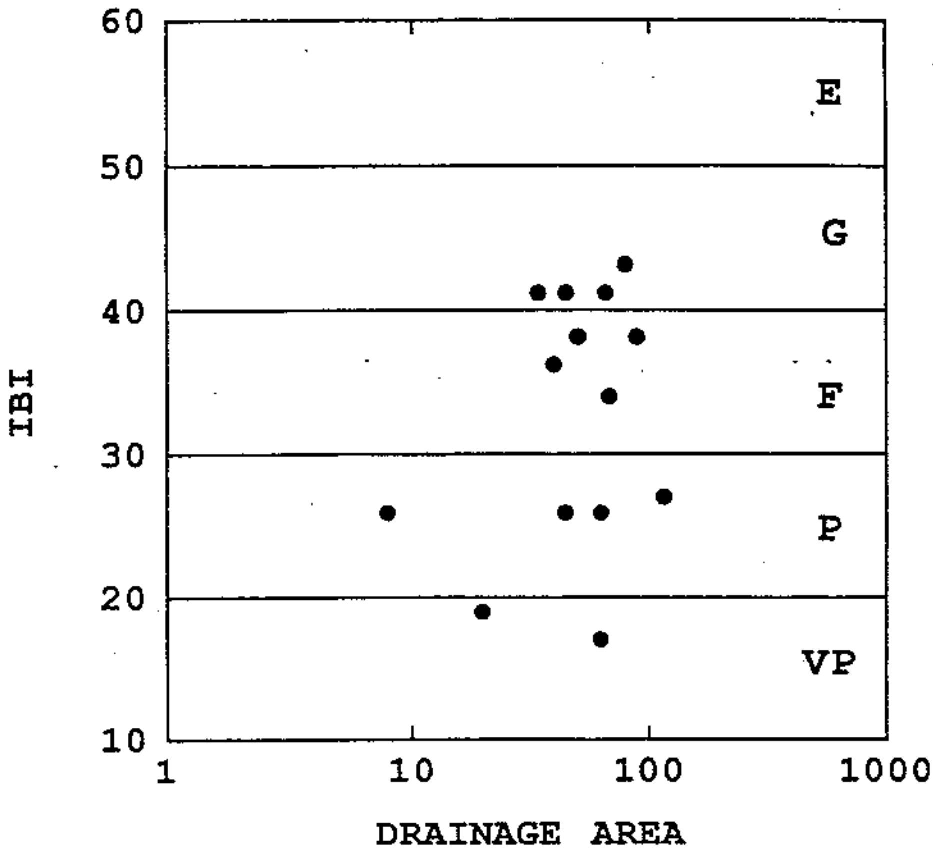


Figure 23. IBI score for Redwood River tributary sites.
E= exceptional, G= good, F= fair
P= poor, VP=very poor

River (Appendix D, Table I; Appendix F, Map 3). The total combined catch for these stations was dominated in numbers by sand shiner, spotfin shiner, common shiner, and shorthead redhorse, comprising 14%, 14%, 14% and 13% of the catch respectively. The fish assemblage condition, as measured by the IBI, ranged from poor to excellent (Figure 24). There was a definite trend in IBI scores with river mile, with exceptional scores found downstream of Rapidan dam (RM 11, RM 2) and poor scores found near the city of Blue Earth (RM98, RM93). At the most upstream sites, few darters, sunfish or intolerant species were collected (RM98 to RM56). However, several species of suckers were collected at most sites through this reach. The fish community in the lower reach of river was diverse with 20 to 30 species collected at each site. Suckers, darters and sunfish were well represented. Intolerant species collected included the smallmouth bass, slenderhead darter, highfin carpsucker, and northern hog sucker. The majority of sites on the main stem had a relatively high percent abundance of specialized insectivores and top carnivores. Although QHEI scores were not determined for the main stem of the Blue Earth, cursory observations of habitat quality were made. In general, the heterogeneity of bottom substrates appeared to increase from upstream to downstream. The frequency of riffle habitat also increased.

In total, forty species were collected at eighteen sites on the tributary streams of the Blue Earth River (Appendix D, Table J; Appendix F, Map 3). These sites had a combined catch that was dominated in numbers by fathead minnow, white sucker, creek chub, and bluntnose minnow which comprised 16%, 14%, 13% and 11% of the catch respectively. The IBI scores in the tributaries ranged from 17-38 and there was one site where no fish were taken. The mean IBI score was 27. Twelve sites sampled were rated as having poor fish assemblages, 2 very poor, and 4 sites were considered fair. There was no trend of improving biological quality with drainage area for these sites (Figure 25). The majority of

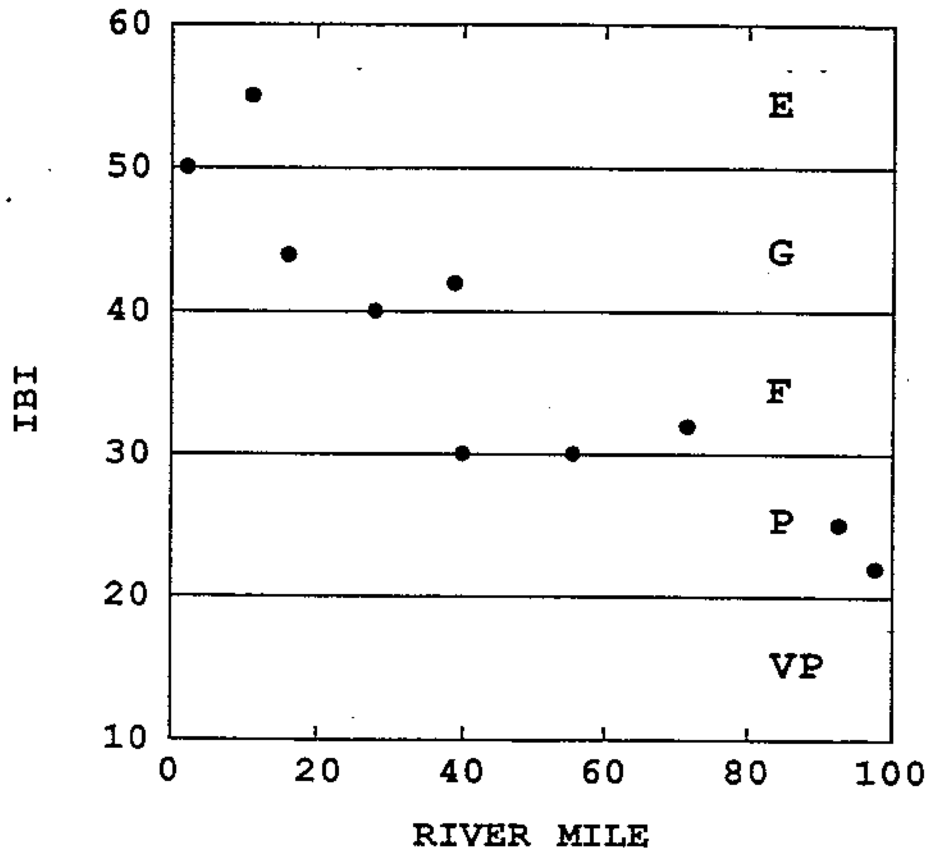


Figure 24. IBI scores for Blue Earth River main stem sites
E= exceptional, G= good, F= fair
P= poor, VP= very poor

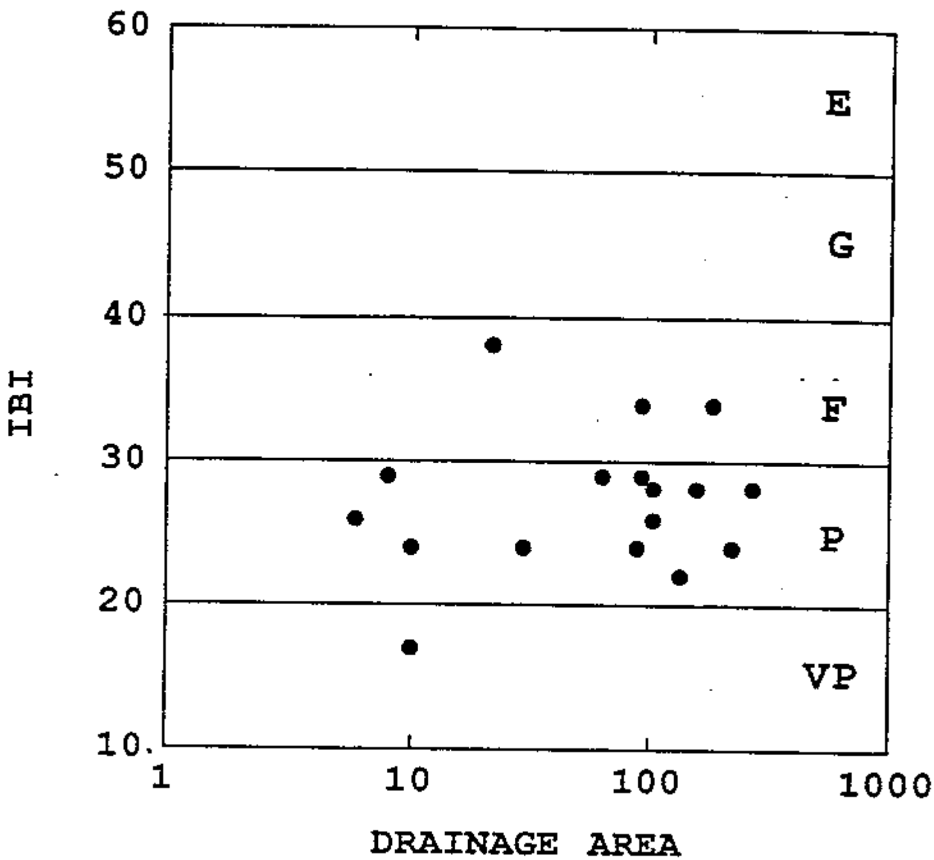


Figure 25. IBI scores for Blue Earth River tributary sites.
E= exceptional, G= good, F= fair
P= poor, VP=very poor

scores in the tributaries ranged from 17-38 and there was one site where no fish were taken. The mean IBI score was 27. Twelve sites sampled were rated as having poor fish assemblages, 2 very poor, and 4 sites were considered fair. There was no trend of improving biological quality with drainage area for these sites (Figure 25). The majority of sites sampled had IBI scores less than 30 and therefore would not meet biological criteria set in this report. Many sites had relatively high percentages of tolerant species and omnivore species. The QHEI scores ranged from 22 to 75. Many sites sampled were embedded with sand and silt. Poor to fair channel development was found at sites in the East Branch and its tributaries including Foster Creek and Brush Creek. Many of the tributaries entering the East Branch Blue Earth were channelized. Sites on the Middle Branch, West Branch, and Coon Creek had low QHEI score (29-37) with silt and sand dominating the bottom substrates. Sites on Elm Creek, Center Creek, and South Creek had more diverse bottom substrates, better cover and generally had higher QHEI scores.

DISCUSSION

Problem Identification

There are five principal factors (Figure 26) that have been identified as controlling biological integrity (Karr et al. 1986). Habitat structure is an extremely important factor and is considered to be the principal determinant of biological potential. For this reason, interpreting biosurvey results has to be done within the context of habitat quality (Plafkin et al. 1989).

Poor habitat quality from sedimentation and channelization was presumed to be the reason for low IBI scores at several of the river reaches sampled. Sites on the lower Redwood River having the lowest IBI scores were areas that had undergone channelization in the past. These sites had poor bottom substrates and little instream cover. Similarly, the

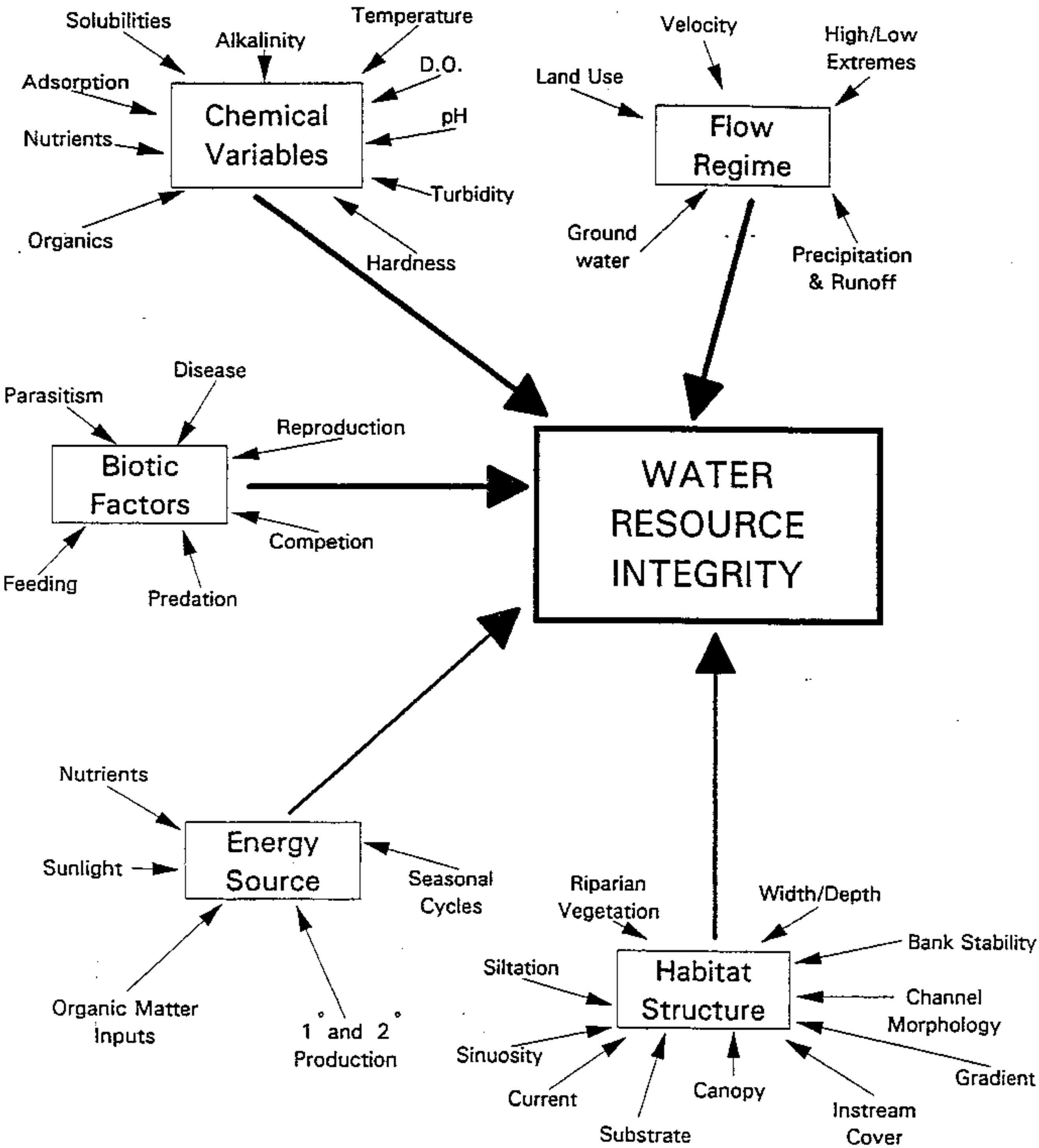


Figure 26. The five principal factors, with some of their important chemical, physical, and biological conponents that influence and determine the integrity of surface water resources (modified from Karr et al. 1986).

poorest IBI scores obtained on the Blue Earth River were in areas of the upper river where there was shifting substrate and little cover. The impacts of habitat modification and sedimentation appeared even more dramatic at sites in the smaller streams where the majority of very low IBI scores were observed. Several of the small stream sites appeared habitat limited by channel modifications, lack of cover, and sedimentation. Many of these sites had poor fish assemblages as measured by the IBI.

IBI scores of all groupings of sites analyzed had significant positive correlation with the QHEI. The relationship between habitat and fish community condition, however, was not obvious at every site sampled. At some sites in the Redwood River watershed, for example, there were high IBI scores found at some channelized stream sites. In some other areas the situation was different where apparent good habitat quality at a site was not supporting a good fish community. This appeared to be the case in several sites within tributaries of the upper Blue Earth River watershed. There are several possible explanations for these situations. Rankin (1991), in analyzing Ohio EPA QHEI and IBI data, found site QHEI scores insufficient for solely determining whether habitat was limiting. He contended that reach scale or watershed scale habitat determinations were more important. A watershed or river reach that has overall good habitat can provide a mitigating effect on biota in short sections of river that are degraded. On the other hand, Rankin (1991) contended reaches or watersheds that have overall poor habitat quality cannot maintain biological integrity in small oases of good habitat. This reasoning may explain the overall poor biological integrity seen in the sites sampled from the Blue Earth River watershed where much of the habitat, particularly in the tributary streams, has been degraded by siltation and channelization.

A further explanation of seeing higher IBI scores at some habitat degraded stream sites could also be partially explained by the weather conditions of 1992. The unseasonably cool temperatures and higher than normal flows, may have moderated these effects. These unseasonal conditions and high dilution factors may also explain the lack of obvious impacts observed below point dischargers. There were no discernible differences in IBI scores from sampling sites located upstream and downstream of these dischargers.

Water chemistry conditions in the basin, although not measurably affecting the biota due to the higher than normal flow conditions during much of the fish sampling, are implicated in degraded fish community conditions in the basin. The dominance of omnivores and species considered tolerant at many of the sites, and the low abundance of top carnivore species at most of the midsized stream sites suggests several areas where trophic structure has been altered over time. Such conditions are probably caused by a combination of factors including habitat degradation and organic enrichment in the smaller headwater systems (Karr et al 1983).

Concern over biological conditions in small streams is certainly warranted, even though these small streams do not typically support sport fish. It is vital to remember that these small streams are important components in river drainage networks because they are the entry point of many resources and pollutants from the terrestrial environment into the aquatic system. These small streams are also critical areas for spawning and juvenile fish of many fish species (Karr et al. 1983). Therefore, the condition of these small streams affects not only the local area but downstream reaches as well.

<u>IBI As A Monitoring Tool</u>

The IBI is a method of quantitatively evaluating the condition of fish communities in respect to reference expectations. It provides a criteria against which stream fish communities can be measured. The merits of the IBI lie in its ability to measure the impacts of water resource degradation on fish communities and discern spatial and temporal variations in these conditions. Our results also emphasize its reproducibility.

The positive correlation found between IBI scores and the independent measurement of habitat quality (QHEI), for various groupings of sites, indicates that the IBI can detect degradation in the Minnesota River basin caused by habitat impairment. Ohio EPA (1990) found strong positive correlations between habitat quality and IBI scores. Their research, along with the work of Karr et al. (1987), has provided considerable evidence that their versions of the IBI can not only reflect habitat quality perturbations but also various water quality problems. Ohio EPA (1990) has also demonstrated the value of the IBI as a tool for measuring improvement in biological conditions after pollution control measure have been put into place.

IBI Variability

The between-year sampling that was conducted at eleven sites (Table 4) also suggests that the IBI is fairly reproducible. The mean difference of 4 points in IBI scores is similar to results found in other studies that evaluated IBI variability. Karr et al. (1987) found in their original version of the IBI, a mean within-year difference of 7 points at 24 sites in two river systems. Lyon (1992) found a mean difference of 9 points at 6 sites over a four year period (IBI scored on a 100 point range as opposed to our 48 point range). In evaluating Ohio EPA's extensive IBI database (1,335 sites between 1979-1989), Rankin and Yoder (1990) used the percent coefficient of variation (CV) to evaluate variability at sites that had three sampling intervals within a given year. They found that the median CV was less than 10% at exceptional warmwater habitat sites, and 15% in their warmwater habitat streams that were in attainment with established criteria. Sites with lower biological integrity had, in general, higher CV% than sites with higher biotic integrity. For the Minnesota River basin, repeated sampling at select sites within years and between years should be done to further define the reproducibity of the IBI.

Future Needs

The IBI has been developed for midsized streams, modified for small streams and applied in large rivers (Blue Earth River; Minnesota River mainstem). The IBI version for these two resource types should be considered more provisional than for midsized streams. Small streams, particularly very small headwater streams, can undergo natural temporal changes.

Headwater stream habitat is usually shallow, and low or intermittent flows can occur for long periods of time. Most fish species that are found in headwater streams appear to have life history characteristics suited to this more unstable environment. Many headwater species have short life spans, small body size and early sexual maturity. They also appear able to rapidly recolonize after a disturbance such as drought (Schlosser 1990). For these reasons, natural variability of the IBI would be predicted to be high in smaller streams. Our results, from the limited repeat sampling conducted at some sites, does suggest that IBI scores for these sites are reproducible. However, continuing repeated sampling will provide greater information regarding the natural variability issue. For large rivers, the concerns are somewhat different. The ability to obtain a representative sample is more difficult due to water depth and the larger area. We considered the samples obtained to be fairly representative of the larger river sections evaluated. Additional work is necessary to evaluate the accuracy of the sampling techniques and the appropriateness of the metrics being used for large river sites.

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Index of Biological Integrity (IBI) Guidance for Coolwater Rivers and Streams of the Upper Mississippi River Basin

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St Paul, Minnesota
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I. BACKGROUND INFORMATION

Pollution control efforts have been largely successful in reducing point source pollution to rivers and streams. However, the emphasis on chemical contaminants has failed to address the more insidious consequences of landscape alteration and non-point source pollution on the quality of riverine environments. Watershed disturbances from urban, residential, and agricultural development (i.e., road building, stream channelization, alteration of the stream's riparian zone, and many others) contribute to an overall decrease in the physical, chemical, and biological quality of rivers and streams. To protect these environments we must strive to understand the relationship between human induced disturbances and aquatic resource quality. A focus on chemical indicators alone will not suffice. Instead, indicators that integrate the effects of physical, chemical, and biological stressors must be employed.

The condition of aquatic communities in any waterbody is the integrated result of many physical, chemical, and biological processes through time (Ohio EPA 1987a, Barbour et al. 1999). A healthy and diverse aquatic community is a reflection of the current condition of the resource, and an indication that the community has withstood and recovered from any short-term stresses that may have occurred previously. Many responses exhibited by aquatic communities when exposed to disturbance are predictable and measurable, and are therefore valuable indicators of aquatic ecosystem health.

Management activities that are intended to protect river and stream systems must be predicated upon a comprehensive monitoring strategy that can detect impacts from all types of human disturbance. A biologically focused monitoring strategy would ensure that management activities adequately address all factors that are limiting biological potential by revealing impairment from sources that are episodic, cumulative, and/or non-chemical (USEPA 1996). In addition, a more biologically focused monitoring strategy would provide the most comprehensive and direct measure of "biological integrity", one of the primary goals of the federal Clean Water Act.

THE INDEX OF BIOLOGICAL INTEGRITY

In recent years, methods have been developed to quantify and interpret the results of biological surveys. The new methods allow water-quality managers and policy makers to make informed decisions concerning the condition of aquatic resources. The advances can be attributed to the development of a practical definition of biological integrity, standardized assessment techniques, field and laboratory methods, and the regional reference site concept (Ohio EPA 1987a, USEPA 1996).

A definition of biological integrity proposed by Karr and Dudley (1981) helped form the basis for many later advances in biological survey methods and interpretation. Karr and Dudley (1981) said that biological integrity is the ability to support and maintain "a balanced, integrated, adaptive community, of organisms having a species composition, diversity, and functional organization comparable to that of natural habitat of the region.

The development of an Index of Biological Integrity (IBI: Karr 1981) provided a framework to interpret biological data to assess water quality. The IBI was developed in the early 1980's using attributes (termed metrics) of fish communities in moderate size wadeable streams of the Midwest. It has subsequently been modified for use throughout the country for a variety of assemblages in all types of waterbodies (Simon and Lyons 1995). Most IBI's developed for rivers and streams have used fish or macroinvertebrate communities. The sampling protocols and analysis procedures for fish and macroinvertebrates have been tested and refined over many years (Barbour et al. 1996; Barbour et al. 1999; Lyons 1992a; Meader et al. 1993; Bailey et al. 1994; Niemela et al. 1999; Niemela and Feist 2000; USEPA 1997).

Each metric in the IBI denotes a quantifiable attribute of a biological assemblage that changes in a predictable way with varying levels of human influence. An IBI usually includes 8-12 metrics. The metrics in a typical fish IBI fall into 3 broad categories: 1) species richness and composition, 2) trophic composition and reproductive function, 3) fish abundance and condition. Most IBI's include one or more metrics from each of these categories. The unitless scores assigned to each metric quantify how far any particular metric value deviates from a range of reference values. When the metrics are summed together the resulting IBI score characterizes the biological integrity or "health" of a site (Karr et al. 1986).

REGIONAL REFERENCE SITES

The MPCA uses a regional reference site approach to determine the biological condition of a waterbody within a given region of the state and within a given waterbody class. Properly defined reference conditions provide a benchmark for comparison to measure the level of aquatic resource impairment (Hughes 1995). The goal of regionalization and waterbody classification is to separate the relevant biological signals from noise caused by natural spatial variation. While it is widely recognized that the physical, chemical, and biological characteristics of individual waterbodies differ to some degree, grouping waterbodies by geographic region minimizes the variability due to factors such as climate, soil type, topography, etc.

Many state programs have adopted Omernik's ecoregions (Omernik and Gallant 1988) as a regional framework for IBI development. In Minnesota, versions of the IBI have been developed using Omernik's ecoregion delineation (Niemela et al. 1999) and the major basin framework (Bailey et al., 1994; Niemela and Feist, 2000). Both regional frameworks effectively reduce variability. However, both frameworks also have deficiencies that diminish their usefulness in Minnesota. The major basin framework does not take into consideration some intra-basin landscape differences that may affect the distribution of assemblages within a basin. On the other hand, the current ecoregion delineation developed for Minnesota is too broad and will require refinement before it can be used exclusively as a regional framework for IBI development.

The MPCA will continue to use the major river basin framework to develop IBI's, and at the same time consider the effects of substantial intra-basin landscape variation on species assemblages. Once a statewide data set is obtained, the MPCA will reevaluate the regional framework and make adjustments if necessary to further reduce regional variability.

The term "reference" denotes sites that are minimally impacted by human influence. Reference sites are not necessarily pristine, and in fact rarely are. Virtually all reference sites reflect at least a small degree of impairment resulting from over a century of settlement and land use. The goal is to find sites within the region that are minimally disturbed and likely to remain so (Hughes, 1995). Possible reference site locations may include old growth forests and woodlots, pockets of native prairie, minimally ditched or drained watersheds, roadless areas, preserves, refuges and wildlife management areas. Care must be taken to ensure that the reference sites do not have unusual characteristics or features. The MPCA selects reference sites following the eight-step procedure outlined by Hughes, 1995 (table 1). This process is iterative, in that early steps must often be revisited to refine the make up of the reference site list.

Table 1. Major steps in selecting regional reference sites from Hughes (1995)

1. Define areas of interest on maps
2. Delineate candidate reference catchments
3. Conduct aerial or photo evaluation
4. Conduct field reconnaissance
5. Subjectively evaluate quality of candidate reference sites
6. Determine number of reference sites desired
7. Evaluate biological health of candidate reference sites

STREAM CLASSIFICATION

The purpose of classifying waterbodies is to group similar systems, to prevent the comparison of inherently different systems. A regional framework alone does not sufficiently account for all of the factors that contribute to the natural variability of waterbodies. For example, we would obviously expect to find different aquatic assemblages in lakes than in rivers or wetlands. Moreover, within each of these major waterbody classes, other physical, chemical, and biological characteristics dictate the composition of aquatic assemblages.

It may be necessary to classify rivers and streams into separate classes based on differences in water temperature (coldwater vs. warmwater streams), size (large vs. small watershed drainage area) or other factors related to stream geomorphology (e.g. gradient). Factors such as barriers to migration, the proximity of a site to a larger body of water, or whether the waterbody is permanent or seasonally intermittent, are also considered.

IBI VALIDATION

The IBI concept has proven to be very adaptable (Karr and Chu 1999). Many metrics have been used successfully throughout different regions of the country in a variety of stream types (Simon and Lyons 1995). Metrics such as the total number of species or the percent of tolerant individuals within a sample are common to most IBI versions that have been developed for fish assemblages. However, Karr and Chu (1999) emphasize that each metric should be tested to ensure that the metrics in the IBI respond to disturbance. This is particularly true when developing an IBI for a new region or stream type, or when considering a new or unproven metric. The process we used to validate IBI metrics for the Upper Mississippi River Basin (UMRB) IBI is described in appendix 1.

APPLICATION OF THE IBI

Most of the work in IBI development has focused on moderate size, warm or coolwater, wadeable streams. Sampling methods for these streams have been developed that provide reliable and reproducible results. Additionally, aquatic communities within these systems have been extensively studied, particularly fish and macroinvertebrate assemblages. Recent promising applications of the multimetric concept have been developed to assess other environments including coldwater streams

(Lyons et al. 1996; Mundahl and Simon 1999), wetlands (Gernes and Helgen 1999; Helgen and Gernes 1999), great rivers (Simon and Emery 1995; Simon and Sanders 1999), lakes (Jennings et al. 1999; Minns et al. 1994; Whittier 1999, Drake and Pereira 2002), reservoirs (Jennings et al. 1995; McDonough and Hickman 1999), and terrestrial environments (James Karr, personal communication).

Many states have begun to develop multimetric indices for rivers and streams with the ultimate goal of developing biological criteria (narrative expressions or numerical values that describe the reference biological condition) for use within their own water-quality programs (USEPA 1996). Thirty-six states are using, or developing biological monitoring programs using multiple assemblages. Fish and macroinvertebrates are the assemblages most often used in state programs. The algal assemblage is used by a smaller number of states. The state of Ohio has taken the definitive lead by developing numeric biological criteria and using the information to guide management activities. Ohio Environmental Protection Agency uses the information from biological assessments in wastewater permitting, 305(b) assessments, 303(d) listing, 401 certification process, waste load allocation, and overall basin assessments.

THE MINNESOTA POLLUTION CONTROL AGENCY'S BIOLOGICAL MONITORING PROGRAM

Efforts in Minnesota to develop multimetric indices using fish communities began in 1990 with the initiation of the Minnesota River Assessment Project (Bailey et al. 1994). A subsequent interagency study conducted during 1994-1995 developed a fish-based IBI for the Lake Agassiz Plain ecoregion in the Red River of the North Basin (Niemela et al. 1999). In 1996, the MPCA adopted a monitoring strategy that centered on an integrated approach (i.e. fish and macroinvertebrate monitoring together with habitat and water chemistry measurements). About the same time the MPCA also adopted a major basin management framework. The MCPA has collected data to develop multimetric indices using the integrated biological approach from 4 basins in Minnesota; the St. Croix, Lake Superior, Upper Mississippi River, and the Minnesota River basins. A fish-based IBI for the St. Croix River Basin has been developed (Niemela and Feist 2000) and both fish and macroinvertebrate based IBI's are being developed for each of the other basins.

The MPCA has used the biological information to investigate the efficacy of wastewater permitting requirements, to determine use attainment status for streams in 305b assessments, to identify high quality reference streams, to identify impaired streams (e.g. 303(d) listed), and to identify and measure progress related to basin management activities.

The ultimate goal of the MPCA's biological monitoring program is to develop an IBI for each of Minnesota's nine major river basins with the intent of eventually developing statewide biological criteria. It is paramount to the development of biological criteria in Minnesota that we obtain fish community information statewide. There is currently a paucity of fish community data for rivers and streams in Minnesota, particularly those streams that have little potential to contain game fish. In fact, fish community information had not previously been obtained for many of the small streams sampled during the course of this study.

This report is the result of an effort to develop an IBI for permanent coolwater rivers and streams within the UMRB in Minnesota. The document is intended to provide guidance for those interested in conducting an IBI assessment. Readers interested in the theoretical underpinnings of biological monitoring and multimetric indices should refer to Karr and Chu (1999).

II. THE UPPER MISSISSIPPI RIVER BASIN

GENERAL BASIN CHARACTERISTICS

The UMRB encompasses all land draining into the Mississippi River upstream of the dam at Hastings, excluding the Minnesota River drainage. The basin is the largest of Minnesota's 9 major river basins, with a land surface area of 20,105 mi^2 (fig. 1). The flat to gently rolling topography was formed by glacial activity. The glacially derived land surface consists of till plains, morainal hills, lacustrine basins, and outwash plains (Omernik and Gallant 1988).

The pre-settlement vegetation was diverse ranging from coniferous forest in the north to native prairie in the southwest corner of the basin. Agriculture and urban development have significantly changed the vegetation throughout most of the basin. Hay, corn, and soybean fields now cover a significant amount of the land, particularly in the southern portion of the basin.

The predominant livestock raised within the basin are dairy cattle, beef cattle, and pigs. Poultry production is important primarily in the southern half of the UMRB.

Three ecoregions comprise over 99% of the land surface area in the basin (fig. 1). The Northern Lakes and Forests ecoregion makes up roughly the northern half of the basin (10,151 mi^2). Coniferous forest, hardwood forest and wetlands dominated the pre-settlement landscape in this region (fig. 2). Early settlers relied heavily on the vast forests of this ecoregion to support logging operations. In fact, logging still remains an important component of the economy in this region. The forest still constitutes a large percentage of the overall area but is now fragmented into a mosaic of forest and grass/cropland. The majority of wetland habitat within the basin lies within the Northern Lakes and Forests ecoregion.

Most of the southern portion of the basin (8,929 mi^2) lies within the North Central Hardwood Forest ecoregion. This is a transitional area between the forested ecoregions to the north and the more heavily agricultural regions to the south. Vegetation within the ecoregion was originally a mixture of hardwood forest and native prairie. Agricultural fields and pastures have now replaced much of the original forest and prairie landscape. Ditches and channelized streams are common throughout the North Central Hardwood Forests ecoregion.

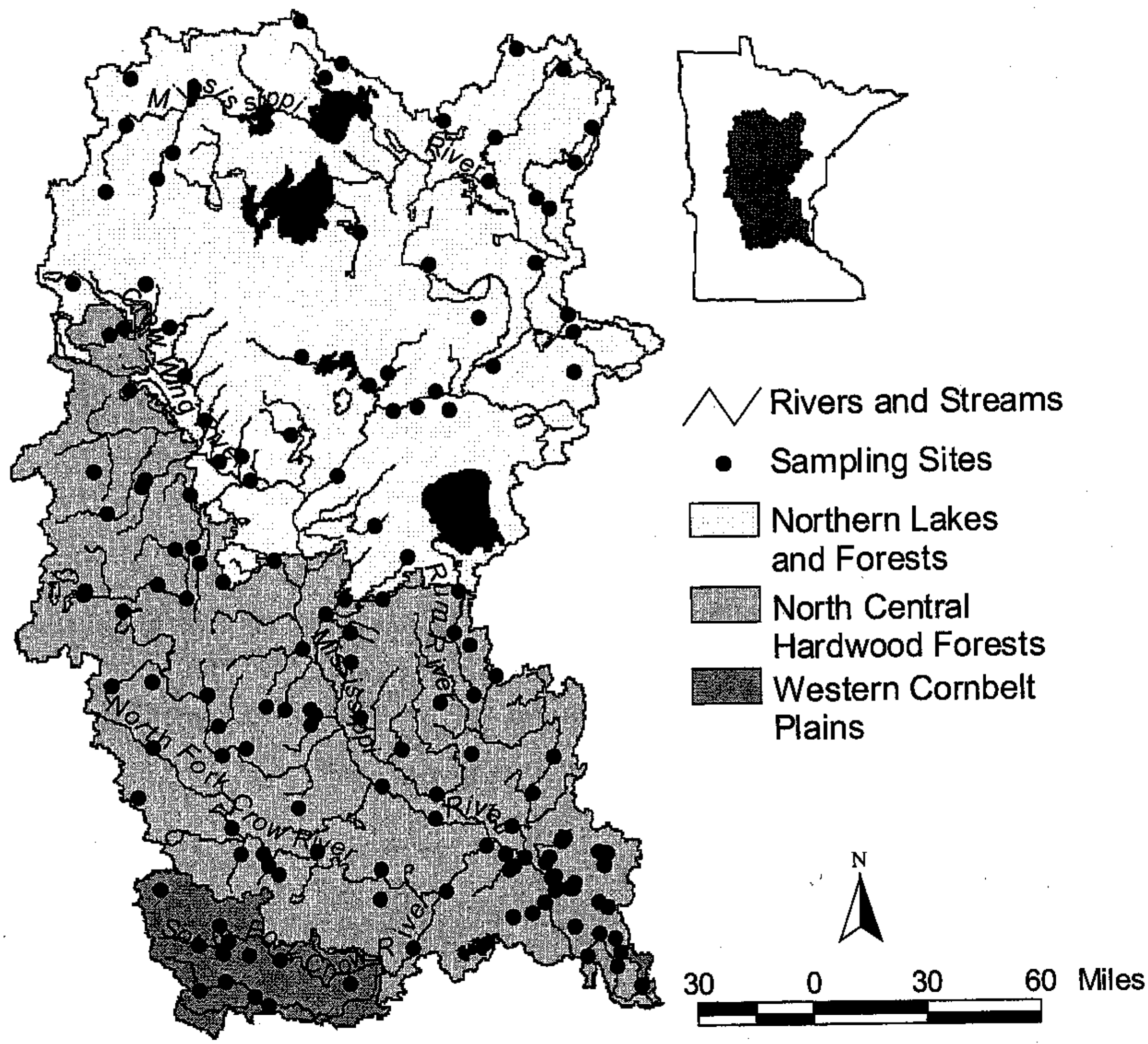

Figure 1. Map of the Upper Mississippi River Basin (UMRB) in Minnesota with major rivers, ecoregional boundaries and the location of each site used to develop the UMRB, Index of Biological Integrity (IBI).

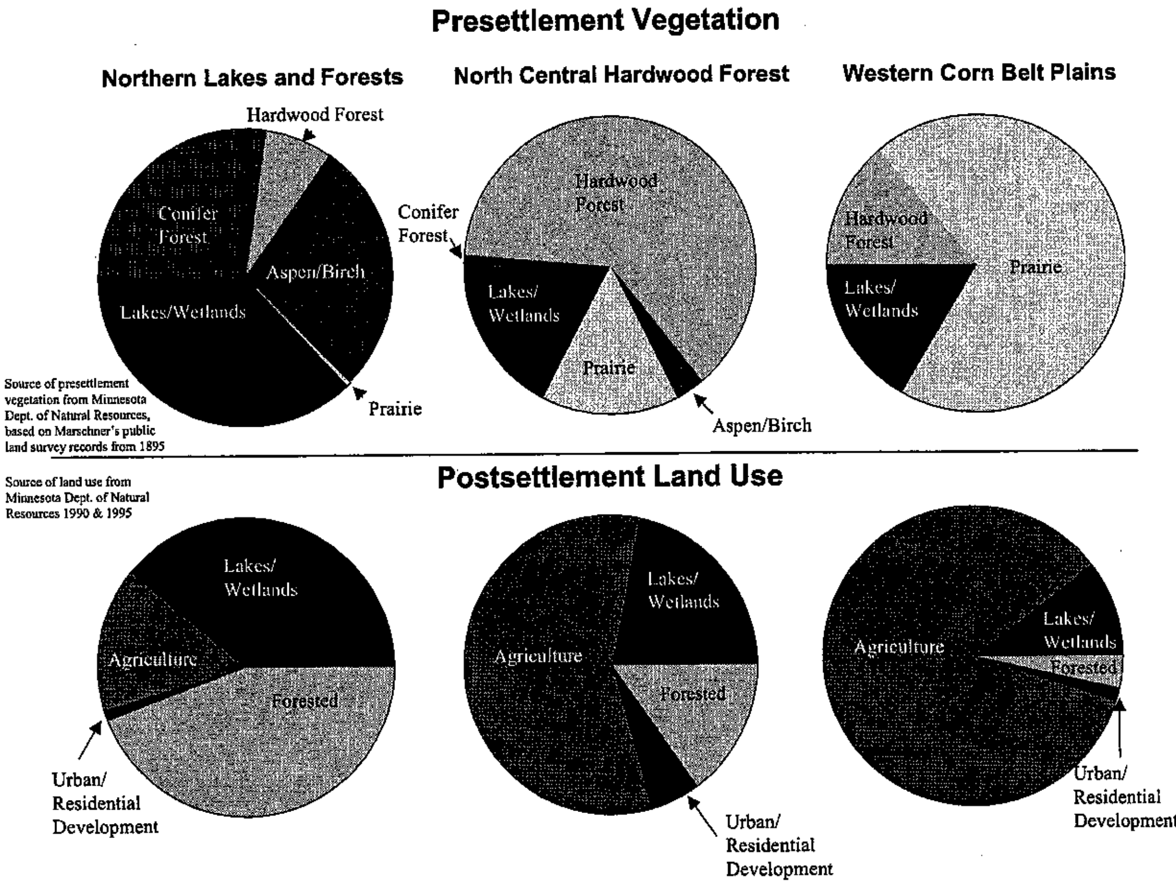

Figure 2. Presettlement vegetation and land use percentages in the Northern Lakes and Forests, North Central Hardwood Forests, and Western Cornbelt Plains ecoregion in the Upper Mississippi River Basin of Minnesota.

The Western Cornbelt Plains ecoregion (1,015 mi^2) lies within the southwestern most portion of the basin. Originally this area was dominated by native tall grass prairie but nearly all prairie vegetation has been removed. Row crop agriculture now dominates this portion of the basin. Many streams within this ecoregion have been channelized and much of the original wet prairie has been drained.

RIVERS AND STREAMS OF THE UPPER MISSISSIPPI RIVER BASIN

From its headwaters at Lake Itasca the Mississippi River winds its way generally east through the lake country of northern Minnesota before turning south into central and southeastern Minnesota. Along it's way the Mississippi River is joined by numerous large tributary streams including the Crow Wing River, Sauk River, Crow River, and Rum River. The segment of the Mississippi River between St. Cloud and Anoka, along with portions of the Rum and North Fork Crow Rivers are Outstanding Resource Value Waters (ORVW). ORVW waters have tighter restrictions on new or expanding discharges from point or non-point sources.

Streams in the southern half of the basin are strongly influenced by urban and residential development, and agricultural practices. The habitat in most headwater streams in the southern half of the basin has been altered through ditching, channelization and the subsequent removal of riparian wetlands.

These practices have allowed run off from fields and feedlots to become more pervasive. Consequently, turbidity levels are generally higher in the south and excessive levels of fecal coliform bacteria have been found in some streams.

Rivers and streams in the northern half of the basin have had less influence from human development. Because most headwater streams in the north have not been channelized, most of them have retained the morphological characteristics of a natural stream including the important connection to riparian wetland areas. Consequently, these streams are often less turbid and have a dark color due to the decomposition of plant material from the surrounding wetlands.

THE FISH ASSEMBLAGE

The UMRB supports a diverse fish assemblage. Eddy et al. (1963) identified 123 fish species below St. Anthony Falls and 64 species above the falls. The lock and dam system that was constructed at St. Anthony Falls in the 1960's has diminished its effectiveness as a barrier to fish migration. Currently, the next upstream dam, located at Coon Rapids, most likely impedes fish migration. The most recent fish distribution list identifies 74 fish species (17 families) that occur above St. Anthony Falls (Hatch and Schmidt 2001; appendix 2).

Most of the rivers and streams in the UMRB support fish that prefer warmwater or coolwater environments. Less than 2 percent of the stream miles in the basin are designated as coldwater streams that are capable of supporting trout. This IBI guidance is intended for all non-coldwater rivers and streams in the basin.

Minnesota does not list any of the fish species in the UMRB as endangered or threatened. However, the pugnose shiner (*Notropis anogenus*) and the shortnose gar (*Lepisosteus platostomus*) are considered special concern species.

WATER-QUALITY ISSUES

Development in the basin has raised many issues related to the quality of rivers and streams. Water-quality concerns in the UMRB include:

Population growth and development: Over 60% of Minnesota's 4.3 million people live within the basin (MPCA 2000). A high percentage of the population is located in the southeast corner of the basin in and around the Twin Cities metropolitan area. Other major communities within the watershed are the cities of Bemidji, Grand Rapids, Brainerd, and St. Cloud. Development within these highly populated areas results in impacts to rivers and streams from habitat destruction, increased point source discharges, and non-point run off. Increased levels of impervious surfaces in urbanized areas exacerbate non-point source run off. Impervious surface levels of 10% have been linked to significant changes in the diversity of aquatic systems (Wang et al 2000).

Wastewater treatment: Over 400 National Pollutant Discharge Elimination System (NPDES) or State Discharge System (SDS) permits have been issued to facilities that discharge pollutants to waters within the UMRB (MPCA 2000). The permit types include municipal, industrial, groundwater pump out and discharge, and stormwater. Each permit includes effluent limitations, monitoring requirements, and other conditions that are intended to protect water resources. There are also over 50 communities not served by a centralized sewer and wastewater treatment facility and many other developments, townships, and unincorporated areas remain unsewered (MPCA 2000). While these communities are not necessarily out of compliance with current regulations, their wastewater systems are not permitted through NPDES or SDS.

Soil erosion and sediment impacts: Soil loss from poorly managed land and inadequate riparian buffers impairs habitat for aquatic life. In many areas of the basin, the natural riparian zone along the stream has been eliminated and converted to cropland or pasture. Intact riparian buffers reduce soil loss from fields by providing a barrier between the open fields and the stream. Some soil conservation practices (e.g. conservation tillage, riparian buffer strips) are gaining acceptance within the farming community but these gains have been slow and soil loss from farm fields continues to be a problem.

Nutrient loading from agricultural run off: The primary sources of nitrogen and phosphorous from agricultural land are derived from fertilizer application and livestock manure. When improperly managed, excessive nutrients from these sources can lead to increased algae and plant growth, oxygen depletion, toxicity, and the presence of disease causing organisms (MPCA 2000).

Drainage and channelization: Ditching, stream channelization, and tile drainage compromise habitat and water quality. These drainage techniques are used throughout the basin but are much more prevalent in the southern half of the basin. Alteration of natural drainage pathways fundamentally alters the natural hydrologic cycle of streams causing changes in the flow regime and loss of habitat. Water that was once slowed by bends, pools, and woody debris in the water column is encouraged to move through the system faster by straightening the stream and removing obstructions. The faster flowing water erodes stream banks and carries with it sediment and nutrients, some of which is deposited in the downstream reaches.

III. IBI SAMPLING METHODS

WHEN TO SAMPLE

Fish community sampling is conducted during daylight hours from mid June through mid-September. All measurements are taken during base-flow conditions since flood or drought events can have a profound effect on fish community structure and sampling efficiency. Avoid sampling immediately following unusually high or low-flow periods.

REACH LENGTH DETERMINATION

It is very important to sample a sufficient length of stream to obtain a representative sample. A reach length that is too short may result in an inadequate sample size, and some uncommon species could be missed. Over-sampling a stream reach adds little to the interpretive capability of the data and places strains on limited monitoring budgets. Lyons (1992b) determined that an adequate reach length for wadeable streams in Wisconsin is 35 times the mean stream width. This level of effort helps to ensure that a representative sample of the fish community is obtained and all of the major habitat types are sampled. The MPCA has adopted this protocol with the added requirement that the reach length can be no less than 150 m and no more than 500 m. The mean stream width is determined prior to sampling by taking a minimum of 10 measurements of the wetted stream width (Simonson et al. 1994). These measurements are taken across the channel in all of the major habitat types found within the reach.

ELECTROFISHING TECHNIQUE

The MPCA uses four types of electrofishing gear to collect fish community information. Gear selection is dependent on stream size and type. However, there are a few procedures that are common to all of the gear types: 1) Net all fish regardless of the species or size of the fish. 2) Sample all available habitat types within the reach in the proportion that they occur. 3) Proper use of electrofishing gear requires extreme care and strict adherence to all recommended safety precautions. 4) Record the amount of time fished and the control box settings for each run. 5) Change holding water frequently to avoid stressing the fish.

Backpack electrofisher: This gear type is used in small, (usually <8 m wide) wadeable streams. Sampling proceeds in an upstream direction with one person carrying the electrofishing gear and collection bucket and the other person netting. In very small (<3m wide) streams it is possible to sample virtually all of the available habitats but in larger streams (>3m wide) it is often necessary to weave back and forth between habitat types.

Stream electrofisher: This gear type is used in larger, (usually >8m wide) wadeable streams and rivers. The stream electrofisher is a sport canoe or barge rigged for electrofishing with a generator, a control box to regulate the electrical output, and two anodes. Sampling proceeds in an upstream direction with a crew of five. Two members of the crew hold the anode poles; each accompanied by a netter. The fifth person pulls the sport canoe upstream, monitors the control box, and ensures team safety. It is usually necessary in these larger streams to weave back and forth between habitat types.

Mini-boom electrofisher: The miniboom electrofishing unit is used in small or hard to access unwadeable streams and rivers. This unit is a small jon-boat rigged for electrofishing with a generator, control box, and a single anode. One person drives the boat, monitors the control box, and ensures the safety of the single netter on the bow. Sampling proceeds downstream by weaving back and forth into different habitat types.

Boom Electrofisher: This gear is used in large, accessible rivers. The boom electrofisher is fished in a downstream direction in three separate runs; one run along each shoreline and a mid-channel run weaving across the stream channel. One person drives the boat, monitors the control box and ensures the safety of the two netters on the bow.

FISH PROCESSING

Fish are usually processed after the entire site has been sampled or after each run when using the boom electrofisher. In some cases, particularly in larger streams, it may be necessary to temporarily stop electrofishing activities during the run and process the larger fish to minimize fish mortality. Data from each run are recorded separately, then later pooled to yield one data set for the entire reach.

All fish are sorted into separate containers by species and enumerated. The minimum and maximum lengths and batch weights are also recorded for each species. Although the length and weight information is not used to calculate the IBI score, this additional information may provide evidence of a size/age imbalance that may be useful when evaluating a stream. Juvenile fish less than 25 mm are not included in the catch. Any deformities, eroded fins, lesions, or tumors (DELT anomalies) should be noted. Two specimens of each species from each site should be retained for later verification by an expert ichthyologist. All other fish should be released back into the stream.

IV. THE METRICS

We classified fish into metric groups by reviewing Wisconsin (Lyons 1992a) and Ohio EPA classifications as well as numerous ichthyological texts and papers (Balon 1975; Becker 1983; Etnier and Starnes 1993; Pflieger 1975). A list of the metric classifications for each fish species that is found in the UMRB is provided in appendix 2.

The metrics in the IBI were selected because they demonstrated a response to a gradient of human disturbance or were considered important in detecting change in the fish community at the most severe levels of degradation (e.g. the proportion of fish with deformities, eroded fins, lesions or tumors). The rationale for each metric used in the UMRB IBI is described below.

SPECIES RICHNESS AND COMPOSITION

Total number of species: This metric is common to almost every IBI developed in streams throughout the country. For warm or coolwater streams, species richness declines as environmental degradation increases (Karr et al. 1986; Leonard and Orth 1986). Hybrids and exotics are not included in this metric.

Number of darters, sculpins, and madtoms: Darters, sculpins, and madtoms are generally found in higher quality streams throughout the UMRB. They are generally considered sensitive to water-quality degradation. These species are considered benthic invertivores; they rely on undisturbed benthic habitats (i.e. clean, course substrates) to feed and reproduce. The madtoms in particular require an ample supply of aquatic macrophytes or woody debris for cover. The degradation of benthic habitats (e.g. channelization, siltation) will cause these species to decline.

Number of minnow species: Minnows are an important and diverse component of many aquatic communities in the UMRB. Because they exhibit a wide range of food and habitat preferences they should be sensitive to a wide range of environmental degradation. Minnow species classified as tolerant are not included in this metric.

Number of wetland species: Wetland habitats are effective stream buffers; filtering contaminants, trapping sediment, and mitigating flow extremes. Streams that have retained their connectivity to riparian wetlands are typically sinuous and slow moving with fine (sand or silt) substrates and a lot of instream and overhanging vegetative cover. Removal of riparian wetlands, ditching, and tiling will eliminate or destabilize these systems and reduce the amount of habitat available for aquatic communities. A number of species, such as the northern redbelly dace (*Phoxinus eos*) and finescale dace (*Phoxinus neogaeus*), are commonly found in headwater streams that have retained their connection to riparian wetlands. Species that were classified as tolerant are not included in this metric.

Number of intolerant species: Intolerant species are those that are known to be sensitive to environmental degradation. They are often the first species to disappear following a disturbance. Most intolerant species have a reduced distribution as a result of human influence. Their presence in a stream is an indication of a high quality resource.

Percent of individuals that are tolerant species: Tolerant species are known to persist in poor quality streams. They may become a dominant component of the fish community in streams that have been physically altered by channelization, siltation, or other hydrologic modifications. Tolerant species may also dominate in chemically altered streams with chronically low dissolved oxygen levels, high levels of ammonia, other toxic substances, or high turbidity (Lyons 1992a).

Percent of the dominant two species: In many degraded stream systems one or two species will tend to dominate the community while other species decline. Those species with the ability to capitalize on a physical or chemical change in their environment are usually tolerant species (Goldstein et al. 1994). This metric compliments the tolerant species metric by providing a measure of the degree in which two species dominate a particular environment. The percent dominance increases with a higher level of human disturbance.

TROPHIC COMPOSITION AND REPRODUCTIVE FUNCTION

Number of invertivore species: Invertivores are specialized feeders that are dependent upon a stable invertebrate food base. Disruptions in this food base through human disturbance can lead to a decrease in the number of invertivore species. Species classified as tolerant are not included in this metric.

Percent of individuals that are omnivores: Omnivorous fish species are those that have the physiological ability (usually indicated by the presence of a long coiled gut and dark peritonium) to digest both plants and animals (Karr et al. 1986). The ability to utilize multiple food sources allows the omnivore species to switch to another food source when one type of food is disrupted. A fish community dominated by omnivorous species indicates that there is an unstable food base.

Number of piscivore species: Piscivorous fish species are found in most undisturbed moderate size streams and rivers (>35 mi^2 drainage area) within the UMRB. The occurrence of a viable piscivore population indicates a healthy, trophically diverse fish community (Karr et al 1986).

Percent of individuals that are simple lithophilic spawners: Simple lithophilic spawners broadcast eggs over clean gravel substrates (Balon 1975). The metric is inversely correlated with habitat degradation due to excessive siltation (Berkman and Rabeni 1987).

FISH ABUNDANCE AND CONDITION

The number of fish per meter of stream sampled: This metric has been used to identify streams in which severe degradation has substantially reduced fish numbers. Lyons (1992a) found that the number of fish per meter is consistently low at highly degraded sites. We calibrated the metric so that only very low fish counts (< 5 fish per 100 meters of stream) would produce a poor metric score. Species classified as tolerant are not included in this metric.

Percent of individuals with Deformities, Eroded fins, Lesions, or Tumors (DELT): Like the number of fish per meter metric, the percent of individuals with DELT anomalies metric has been used to identify sites that have been severely degraded. In other parts of the Midwest DELT anomalies have been associated with environmental degradation primarily due to industrial pollutants (Sanders et al. 1999, Ohio EPA 1987b). DELT anomalies were not prevalent in fish from the UMRB. However, we feel it is important to retain the metric to identify streams that are severely degraded. Parasitic infestations are not included in this metric because parasitic burden does not necessarily correlate with environmental quality (Steedman 1991).

V. SPECIAL CONSIDERATIONS

LOW CATCH RATES

If the total number of individuals at a site is extremely low a few individuals can have a relatively large influence on the overall IBI score. In this case the IBI score may not be a true reflection of environmental quality. In our judgement, an IBI score should not be calculated for sites with less than 25 individuals. Rather, these sites should be rated as very poor since extremely low catch rates are almost always an indication of serious impairment in permanent, coolwater Minnesota streams.

INTERMITTENT STREAMS

Headwater streams pose a particular problem because of the need to distinguish between permanent and intermittent streams. Some headwater streams in the UMRB, particularly in the southern portion of the basin, tend to go dry during the summer. Our approach has been to consider the IBI a valid assessment tool if the stream does not go dry prior to sampling. This determination is often a judgment call based on habitat information gathered at the site along with information on precipitation during the weeks before sampling. We conduct an evaluation of the site at three different times during the season. An initial site reconnaissance is conducted during the spring. Fish, water chemistry, and habitat data are collected during a second visit in the summer. Finally, a macroinvertebrate sample is obtained during a third visit in the fall.

COLDWATER STREAMS

This IBI should not be used in coldwater streams. Many attributes of the fish community in coldwater streams differ significantly from warm or coolwater systems. Consequently, the IBI's that have been developed for coldwater systems bear little resemblance to their non-coldwater counterparts. For those interested in applying an IBI to Minnesota coldwater streams, Mundahl and Simon (1999) have developed an IBI for coldwater streams in the upper Midwestern United States. Also, Lyons (1996) has developed an IBI for Wisconsin coldwater streams that may be applicable in Minnesota.

NATURAL BARRIERS TO FISH MIGRATION

Barriers to fish migration may influence all streams; however, their effect on fish communities in headwater streams may be the most pronounced. Whereas larger streams usually offer some refuge during periods of stress (i.e., floods and droughts), there is an increased probability in headwater streams for the entire fish community to be extirpated. Therefore, the IBI may underrate headwater streams above fish barriers even though they are otherwise undisturbed. We recommend that the researcher use caution in applying the IBI if natural barriers to fish movement exist. A survey above and below the barrier may be useful in determining the effect of the barrier on the fish community.

STREAM MOUTHS

Sampling near the confluence of much larger streams, rivers, or lakes should be avoided because the larger waterbody may influence the fish community structure of the smaller stream. We recommend that sites should be located at least 1 mile from a significantly larger stream or lake. If it is necessary to sample closer, we recommend caution in interpreting the IBI score. Often times, a quick examination of the fish community data will provide evidence of the influence of the larger waterbody (e.g. a small stream dominated by emerald shiners *(Notropis atherinoides)*).

VI. CALCULATION AND INTERPRETATION OF THE IBI SCORE

A separate IBI has been developed for 4 stream size classes. The drainage area of the watershed (mi^2) upstream of the site was used as a measure of stream size. The size classes are: very small streams (<5 mi^2), small streams (5-35 mi^2), moderate streams (35-200 mi^2), and rivers (>200 mi^2). The size classes were chosen to minimize differences in maximum species richness within each size class (appendix 1).

The IBI score is determined by summing the metric scores for the appropriate stream size class (table 2, 4, 6, and 8). Each metric in the IBI represents a unique aspect of the fish community. A low metric score indicates that the fish community attribute deviates substantially from a minimally disturbed site. Conversely, a high metric score indicates that the fish community attribute approximates that of a minimally disturbed site. Many of the same metrics are used for each size class. However, a few metrics are unique to a single size class.

Scores of 0, 2, 5, 7 or 10 have been assigned for each metric (appendix 3). Once the metric scores have been obtained from the appropriate table they are added to produce a total IBI score ranging from 0 (lowest biological integrity) to 100 (highest biological integrity). A correction factor is needed for the very small stream size class (<5 mi^2) because only 7 metrics (instead of 10) are used to calculate the IBI. Multiply the IBI score by 1.43 to normalize the score to a 0 to 100 point scale for very small streams (<5 mi^2).

Narrative descriptions of the fish community within 5 integrity classes should be used as a guideline for interpreting the IBI score (table 3, 5, 7, and 9). There is considerable overlap in individual metric values between each integrity class. This illustrates the need for a multimetric approach to assess biological integrity. It is not possible to interpret the IBI score by knowing the value of a single metric. Indeed, if this were the case there would be no need to develop an index. A list of the sites sampled in the St. Croix River Basin and the IBI score for each site are provided in appendix 5.

Three factors; sampling error, natural variation, and human disturbance contribute to the variability of IBI scores. All users of this IBI must attempt to limit the first two sources of variation to detect the third. Sampling error results from a failure to accurately characterize the fish community. Natural variability results from regional and in-stream physical, chemical, and biological differences between streams. The guidance provided in this document along with rigorous adherence to the sampling protocols will limit the influence of sampling error and natural variation on the IBI score. The user will be able to detect changes in environmental condition due to human disturbance with a reasonable level of certainty (appendix 4).

CALCULATION OF THE WATERSHED DRAINAGE AREA

Calculating the drainage area of the upstream watershed is necessary to determine which IBI to use. We used the Minnesota Planning Land Management Information Center's (LIMC) Upstream program to identify all of the upstream minor watersheds. The minor watershed containing the site was picked from MDNR's 1995 minor watershed file (bas95ne3) using the latitude and longitude of the site. The minor watershed boundaries are nearly equivalent to the 14-digit hydrologic unit code (HUC) developed by the U.S. Geological Survey. Upstream additions were confirmed using the MDNR's 24K streams file (dnrstln3).

It may be necessary (particularly in very small streams) to edit the minor watershed containing the site so that the portion of the minor watershed downstream of the site is not included in the drainage area calculation. We edited the minor watershed containing the site using Geographic Information System (GIS), Arcview coverages. However, in most cases an estimate of the minor watershed area upstream of the site may be determined using U.S. Geological Survey (USGS) standard series, 1:24,000 topographical maps. The following methods were used in order of preference to edit the minor watershed containing the site:

a) using Arcview to delineate the drainage area with digital elevation models (DEM).
b) following the contour lines on digital raster graphics (DRG) from U.S. Geological Survey (USGS) standard series topographic maps.
c) or personal experience of watershed boundaries from visiting the site.

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Table 2. Scoring criteria for the 7 metrics used to calculate the IBI for very small streams (< 5 mi^2 drainage area) in the Upper Mississippi River Basin of Minnesota[1].

	Scoring Criteria				
Metric	**10**	**7**	**5**	**2**	**0**
Species richness and composition metrics					
Total number of species	9 or more	7 or 8	5 or 6	3 or 4	0 - 2
Number of wetland species[2]	2 or more		1		0
Percent tolerant species[3]	0-80	81-85	86 - 90	90-95	96-100
Percent dominant two species[3]	0-60	61-70	71 - 80	81-90	91-100
Trophic metrics					
Number of invertivore species[2]	2 or more		1		0
Fish abundance and condition metrics					
Number of fish per 100 meters[2]	5 or more				0-4
Percent DELT anomalies[3]	0-1		2 or 3		4 or more

[1]The sum of the 7 metrics for very small streams must be multiplied by 1.43 to obtain the final IBI score.
[2]Number of wetland species, number of invertivore species, and number of fish per 100 meters metrics do not include tolerant species.
[3]Round all percent metrics to the nearest 1 percent.

Table 3. Narrative guidelines for interpreting the overall IBI scores for very small streams (<5 mi^2) in the Upper Mississippi River Basin (modified from Karr 1981, and Lyons 1992a)

Overall IBI Score	Biological Integrity Rating	Fish Community Attributes
100-80	Excellent	Comparable to the best situations with minimal human disturbance; from 6 to 14 species typically present; the dominant two species and tolerant species comprise from 40 to 80 percent of the catch; non-tolerant individuals are abundant; 2 or more wetland and/or insectivorous species are typically present; a full array of age and size classes are represented
79-60	Good	Species richness somewhat below expectations, 4 to 8 species typically present; the dominant two species and the tolerant species comprise from 60 to 90 percent of the catch; non-tolerant individuals are common; 1 or more wetland and/or insectivorous species are typically present; size/age distributions may show signs of imbalance.
59-40	Fair	Decreased species richness, from 3 to 7 species present; the dominant two species and the tolerant species comprise from 70 to 100 percent of the catch; non-tolerant individuals are not common; typically, no wetland or insectivorous species, or if present there is typically only 1; size/age distributions may show signs of imbalance.
39-20	Poor	Decreased species richness, from 2 to 6 species present; the dominant two species and tolerant species comprise from 70 to 100 percent of the catch; non-tolerant individuals are very uncommon; no wetland or insectivorous species, or if present there is typically only 1; size/age distributions may show signs of imbalance; growth rates and condition factors sometimes depressed; hybrids sometimes common.
19-0	Very Poor	The community is indicative of an environment that is severely modified by human disturbance; very few species present, typically between 1 and 3; all are tolerant forms, wetland and insectivorous species are not present; hybrids, or exotics may be common; age/size distributions are abnormal; DELT fish (fish with deformities, eroded fins, lesions, or tumors) may be present in the most severely degraded environments.
No Score		Thorough sampling finds few or no fish; impossible to calculate IBI.

Table 4. Scoring criteria for the 10 metrics used to calculate the IBI for small streams (5 to 35 mi^2 drainage area) in the Upper Mississippi River Basin of Minnesota.

Metric	Scoring Criteria				
	10	**7**	**5**	**2**	**0**
Species richness and composition metrics					
Total number of species	14 or more	11-13	8-10	5-7	0-4
Number of wetland species[1]	3 or more		1 or 2		0
Number of minnow species[1]	5 or more	4	2 or 3	1	0
Number of intolerant species	2 or more		1		0
Percent tolerant species[2]	0-40	41-55	56-70	71-85	86-100
Percent dominant two species[2]	0-52	53-64	65-76	77-88	89-100
Trophic and reproductive function metrics					
Number of invertivore species[1]	5 or more	4	2 or 3	1	0
Percent simple lithophils[2]	49-100	37-48	25-36	13-24	0-12
Fish abundance and condition metrics					
Number of fish per 100 meters[1]	5 or more				0–4
Percent DELT anomalies[2]	0-1		2 or 3		4 or more

[1]Number of wetland species, number of minnow species, number of invertivore species, and number of fish per 100 meters metrics do not include tolerant species.
[2]Round all percent metrics to the nearest 1 percent.

Table 5. Narrative guidelines for interpreting the overall IBI scores for small streams (5 to 35 mi^2 drainage area) in the Upper Mississippi River Basin (modified from Karr 1981, and Lyons 1992a)

Overall IBI Score	Biological Integrity Rating	Fish Community Attributes
100-80	Excellent	Comparable to the best situations with minimal human disturbance; typically more than 13 species present; no more than 50% of the catch is comprised of tolerant species or dominated by two species; assemblage typically includes 1 to 3 intolerant species and 1 to 5 wetland, minnow, and invertivore species; simple lithophilic spawners comprise up to 50% of the community; a full array of age and size classes are represented
79-60	Good	Species richness somewhat below expectations, from 8 to 15 species possible, but more commonly 9 to 13; no more than 70% of the catch is comprised of tolerant species or dominated by two species; assemblage typically includes 1 to 3 intolerant species and 1 to 5 wetland, minnow, and invertivore species; simple lithophilic spawners make up to 50% of the community; size/age distributions may show signs of imbalance.
59-40	Fair	Decreased species richness, from 6 to 14 species possible, but more commonly 8 to 10; typically, from 50% to 90% of the catch is comprised of tolerant species or dominated by two species; assemblage typically includes no intolerant species, or if present, only 1 species; 1 to 3 wetland, minnow and invertivore species; simple lithophilic spawners comprise up to 30% of the community; size/age distributions may show signs of imbalance.
39-20	Poor	Decreased species richness; from 5 and 9 species possible, but more commonly 6 to 8; almost all are tolerant species (80-98%); the dominant two species comprise from 60 to 90% of the catch; assemblage typically includes 1 to 3 wetland, minnow, and invertivore species, no intolerant species, or if present, only 1 species; simple lithophilic spawners typically make up no more than 20% of the community; size/age distributions may show signs of imbalance; growth rates and condition factors sometimes depressed; hybrids sometimes common.
19-0	Very Poor	The community is indicative of a severely modified landscape and in-stream habitat; very few species present, typically between 1 and 6; almost all are tolerant forms (90-100%), the dominant two species comprise from 80 to 100% of the catch; wetland, minnow, intolerant, and invertivore species are not present, or if so, typically only 1 species present; simple lithophilic spawners comprise no more than 5% of the catch; hybrids, or exotics may be common; age/size distributions are abnormal; DELT fish (fish with deformities, eroded fins, lesions, or tumors) may be present in the most severely degraded environments.
No Score		Thorough sampling finds few or no fish; impossible to calculate IBI.

Table 6. Scoring criteria for the 10 metrics used to calculate the IBI for moderate size streams (35 to 200 mi^2 drainage area) in the Upper Mississippi River Basin of Minnesota.

	Scoring Criteria				
Metric	**10**	**7**	**5**	**2**	**0**
Species richness and composition metrics					
Total number of species	20 or more	16-19	12-15	8-11	0-7
Number of darter, sculpin, and madtom species	4 or more	3	2	1	0
Number of wetland species[1]	3 or more		1 or 2		0
Number of intolerant species	4 or more	3	2	1	0
Percent tolerant species[2]	0-35	36-50	51-65	66-80	81-100
Trophic and reproductive function metrics					
Number of invertivore species[1]	8 or more	6 or 7	4 or 5	2 or 3	0-1
Number of piscivore species	5 or more	4	2 or 3	1	0
Percent simple lithophils[2]	61-100	46-60	31-45	16-30	0-15
Fish abundance and condition metrics					
Number of fish per 100 meters[1]	5 or more				0
Percent DELT anomalies[2]	0-1		2 or 3		4 or more

[1]Number of wetland species, number of invertivore species, and number of fish per 100 meters metrics do not include tolerant species
[2]Round all percent metrics to the nearest 1 percent.

Table 7. Narrative guidelines for interpreting the overall IBI scores for moderate size streams (35 to 200 mi^2 drainage area) in the Upper Mississippi River Basin (modified from Karr 1981, and Lyons 1992a)

Overall IBI Score	Biological Integrity Rating	Fish Community Attributes
100-80	Excellent	Comparable to the best situations with minimal human disturbance; from 15 to 23 species possible, but more commonly 17 to 21; tolerant species typically less than 50% of the catch; assemblage typically includes 3 or more darter, sculpin, and madtom species, 3 or more wetland species; 6 to 8 invertivore species; 2 to 7 piscivore species, and 1 to 3 intolerant species; simple lithophilic spawners may comprise up to 70% of the catch; a full array of age and size classes are represented.
79-60	Good	Species richness somewhat below expectations, from 13 to 21 species possible, but more commonly 15 to19; tolerant species typically less than 50% of the catch; assemblage includes 1 to 3 darter, sculpin, and madtom species, 2 or more wetland species, 3 to 8 invertivore species, 1 to 5 piscivore species, and 1 to 2 intolerant species; Simple lithophilic spawners may comprise up to 70% of the catch; size/age distributions may show signs of imbalance.
59-40	Fair	Decreased species richness, from 6 to 17 species possible, but more commonly 10 to 14; tolerant forms typically less than 50% of the catch; assemblage includes 1 to 3 darter, sculpin, and madtom species, 1 or more wetland species, 3 to 6 invertivore species, up to 3 piscivore species, and 1 to 2 intolerant species; simple lithophilic spawners may comprise up to 70% of the catch; size/age distributions may show signs of imbalance.
39-20	Poor	Decreased species richness; from 6 to 13 species possible, but more commonly 8 to 11; tolerant forms typically over 50% of the catch; assemblage typically includes no more than 1 darter, sculpin, and madtom species; wetland, intolerant, and piscivore species may not be present or if so typically only 1; 2 to 5 invertivore species; simple lithophilic spawners may comprise up to 20% of the catch; size/age distributions may show signs of imbalance; growth rates and condition factors sometimes depressed; hybrids sometimes common.
19-0	Very Poor	The community is indicative of an environment that is severely modified by human disturbance; very few species present, typically less than 6; almost all are tolerant forms (90-100%); darters, sculpins, and madtom, wetland species, intolerant species, invertivore species, and piscivore species are not present, or if so, typically only 1 species present; simple lithophilic spawners comprise no more than 20% of the catch; hybrids, or exotics may be common; age/size distributions are abnormal; DELT fish (fish with deformities, eroded fins, lesions, or tumors) may be present in the most severely degraded environments.
No Score		Thorough sampling finds few or no fish; impossible to calculate IBI.

Table 8. Scoring criteria for the 10 metrics used to calculate the IBI for rivers (> 200 mi^2 drainage area) in the Upper Mississippi River Basin of Minnesota.

	Scoring Criteria				
Metric	**10**	**7**	**5**	**2**	**0**
Species richness and composition metrics					
Total number of species	28 or more	23-27	18-22	13-17	0-12
Number of darter, sculpin, and madtom species	4 or more	3	2	1	0
Number of intolerant species	4 or more	3	2	1	0
Percent tolerant species[2]	0-15	16-30	31-45	46-60	61-100
Trophic and reproductive function metrics					
Number of invertivore species[1]	14 or more	11-13	8-10	5-7	0-4
Percent omnivore species[2]	0-10	11-20	21-30	31-40	41 or more
Number of piscivore species	7 or more	6	4 or 5	3	0-2
Percent simple lithophils[2]	81-100	61-80	41-60	21-40	0-20
Fish abundance and condition metrics					
Number of fish per 100 meters[1]	5 or more				0
Percent DELT anomalies[2]	0-1		2 or 3		4 or more

[1]Number of invertivore species and number of fish per 100 meters metrics do not include tolerant species
[2]Round all percent metrics to the nearest 1 percent.

Table 9. Narrative guidelines for interpreting the overall IBI scores for rivers (>200 mi^2 drainage area) in the Upper Mississippi River Basin (modified from Karr 1981, and Lyons 1992a)

Overall IBI Score	Biological Integrity Rating	Fish Community Attributes
100-80	Excellent	Comparable to the best situations with minimal human disturbance; from 18 to 34 species possible but more commonly 21 to 30; tolerant species and omnivorous species typically less than 20% of the catch; assemblage includes from 2 to 6 darter, sculpin, and madtom species; 8 to 16 invertivore species; 4 to 9 piscivore species, and 1 to 4 intolerant species; simple lithophilic spawners comprise up to 95% of the catch (typically 20% to 60%); a full array of age and size classes are represented.
79-60	Good	Species richness somewhat below expectations, from 13 to 30 species possible but more commonly 16 to25; tolerant species typically less than 40% of the catch; less than 30% of catch are omnivorous; assemblage includes from 1 to 5 darter, sculpin, and madtom species, 6 to14 invertivore species, 3 to 7 piscivore species, up to 4 intolerant species; simple lithophilic spawners comprise up to 95% of the catch (typically 20% to 60%); size/age distributions may show signs of imbalance.
59-40	Fair	Decreased species richness, from 10 to 20 species possible, but more commonly 13 to 18; tolerant species typically less than 50% of the catch; less than 30% of catch are omnivorous; assemblage includes up to 3 darter, sculpin, or madtom species, 3 to 9 invertivore species, 2 to 5 piscivore species, up to 2 intolerant species; simple lithophilic spawners comprise up to 95% of the catch (typically 20% to 60%); size/age distributions may show signs of imbalance.
39-20	Poor	Decreased species richness, typically less than 17 species; over 50% are tolerant species; over 30% are omnivorous; typically no intolerant, or darter, sculpin and madtom species present or if so, typically only 1 species; typically, 3 to 7 invertivore species and up to 3 piscivore species may be present; simple lithophilic spawners comprise less than 20% of the catch; size/age distributions may show signs of imbalance; growth rates and condition factors sometimes depressed; hybrids sometimes common.
19-0	Very Poor	The community is indicative of an environment that is severely modified by human disturbance; few species present, typically less than 12; over 50% are tolerant species; over 30% are omnivorous; typically, no intolerant, or darter, sculpin and madtom species present or if so, typically only 1 species; up to 3 invertivore species, up to 3 piscivore species; simple lithophilic spawners comprise less than 20% of the catch; hybrids, or exotics may be common; age/size distributions are abnormal; DELT fish (fish with deformities, eroded fins, lesions, or tumors) may be present in the most severely degraded environments.
No Score		Thorough sampling finds few or no fish; impossible to calculate IBI.

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APPENDIX 1-METRIC VALIDATION

The following paragraphs outline the steps that were taken to validate metrics used in the UMRB.

SITE SELECTION

The UMRB IBI was developed with data collected during the 1999 and 2000 sampling seasons. A total of 161 sampling events from 148 sites were used in the analysis. A complete list of IBI scores for each site is provided in appendix 5.

Seventy three of the sites were selected specifically to develop the IBI. Karr and Chu (1999) recommend that the sites selected for development of an IBI should focus on multiple sites within similar environments, across a range of human disturbance from minimal to severe. The sites we selected represented a range of stream sizes, disturbances, and morphology types within the UMRB. Potential sites were selected by reviewing GIS coverages for land use, point source discharges, feedlots, and stream ditching. Prior to sampling, a field reconnaissance was conducted to corroborate the GIS based information, obtain landowner permission if necessary, locate and mark (with flagging) the exact sampling reach, and determine the appropriate fish sampling gear.

Seventy five additional sites were used in the analysis but were not selected specifically for the purpose of developing the IBI. Forty seven of these sites were randomly selected to monitor the condition of the basin using a statistically based approach developed by the U.S. Environmental Protection Agency, Environmental Monitoring and Assessment Program (EMAP) (Stevens (1997). Although these sites were not selected specifically for development of the IBI, they were important because they: 1) provided more data to use in the analysis, 2) helped to provide a better understanding of stream characteristics throughout the basin, 3) helped to provide an understanding of the type and extent of human disturbance throughout the basin.

HABITAT ANALYSIS

A habitat assessment was performed at each site to characterize the instream and riparian features of the stream reach. The habitat information was used to help classify streams and identify reference quality sites. In wadeable streams, we used a modified version of Wisconsin's quantitative habitat assessment procedure (Simonson et al. 1994). We also developed a qualitative habitat assessment for wadeable streams (table 10) similar to Wisconsin's Fish Habitat Rating System (FHR) to provide a summary of habitat data and compare the results of assessments between streams. The qualitative habitat assessment uses 6 variables that were the most highly correlated (Spearman rank correlation) with species richness (table 10). We required that there be at least one variable related to stream geomorphology, substrate, instream habitat, and riparian land use. The least disturbed sites were used to develop scoring criteria for the qualitative habitat assessment. Sites with less than 25 percent disturbed land use in the watershed, no obvious pollution sources immediately upstream, and no observable habitat alterations within the reach were considered least disturbed. Each habitat variable was assigned a rating of 2 (similar to least impacted sites), 1 (deviated somewhat from least impacted sites), or 0 (strong deviation from least impacted sites). The total score

Table 10. Habitat definitions and scoring criteria for a qualitative habitat index developed for wadeable rivers and streams within the Upper Mississippi River Basin. Habitat values used in the qualitative habitat index are derived from Wisconsin's habitat assessment guidance (Simonson et al. 1994).

Definitions:

Number of stream features: The number of major morphological features (riffles, runs, pools, and bends) per 100 meters of stream.

Number of substrate types: The number of substrate types found within the stream reach. Substrates include bedrock, boulders, rubble, gravel, sand, silt, clay, detritus, and other. This figure is calculated using the dominant substrate found at 4 equally spaced points along each of 13 transects.

Percent course substrate: The percent of substrate types that are gravel size or larger.

Percent cover for fish: Cover is quantified at 13 transects along the reach. Cover types include undercut banks, overhanging vegetation, woody debris, boulders, submergent macrophytes, emergent macrophytes, and other debris. Mean cover values for each cover type are added to obtain the percent cover for fish within the reach. Values over 100% are possible.

Mean thalweg depth: The depth of the thalweg (deepest area of the stream channel) is determined at 13 transects along the reach.

Sinuosity: Length of the stream reach divided by the straight-line distance between the upstream and downstream ends. Calculated from 1:24,000 USGS quadrangle topographical maps.

Mean bank erosion: The percent of stream bank within 5 meters of the stream that is eroded. Bank erosion is measured on each side of the stream at each of 13 transects. A total of 130 meters of stream bank are evaluated for erosion at each site (5 meters x 26 stream banks).

Percent land use disturbance within 30 meters of the stream: The percent of the riparian zone within 30 meters of the stream that is influenced by human disturbance. Human disturbance land use categories include cropland, pasture, barnyard, or developed. This figure is calculated using the dominant land use on each side of the stream bank at each of 13 transects.

Habitat Scoring Criteria

A.	small streams 0-35 mi^2 drainage area			moderate size streams 35-200 mi^2 drainage area			rivers >200 mi^2 drainage area		
Scoring	**2**	**1**	**0**	**2**	**1**	**0**	**2**	**1**	**0**
Habitat variable									
Number of stream features	≥ 6	4-5	≤ 3	≥ 3	2	1	> 1.5	0.5 to 1.5	<0.5
Number of substrate types	≥ 5	4	≤ 3	≥ 5	4	≤ 3	-	-	-
Percent course substrate types	-	-	-	-	-	-	>50	25-50	<25
Mean thalweg depth	> 50	25-50	< 25	> 70	35-70	< 35	-	-	-
Mean cover for fish	41-70	20-40 and 71-100	<20 and >100	≥30	15-29	≤15	≥30	15-29	≤15
Sinuosity	> 1.2	1.1-1.2	< 1.1	> 1.2	1.1-1.2	< 1.1	> 1.2	1.1-1.2	< 1.1
Mean bank erosion	-	-	-	-	-	-	0	>0-0.1	>0.1
Percent land use disturbance within 30 meters of stream	≤ 20	20-60	≥ 60	≤ 20	20 -60	≥ 60	< 20	20- 60	> 60

ranges from 0 (poor habitat) to 12 (excellent habitat).

In non-wadeable streams we used the Ohio Qualitative Habitat Evaluation Index (QHEI) (Rankin 1989). The QHEI rates the habitat based on substrate quality, in-stream cover, riparian zone quality and bank erosion, and pool/glide and riffle/run quality. The QHEI form takes only minutes to complete once the site has been surveyed.

QUANTIFYING HUMAN DISTURBANCE

Quantifying the overall disturbance level at a site in a large and diverse basin like the UMRB was difficult because there were many individual disturbances that could potentially act synergistically and/or cumulatively to influence the quality of the stream (Adams et al. 1996). We explored numerous avenues in attempting to define a disturbance gradient that accurately reflected the degree to which a site was modified by human disturbance including: 1) ratings based on GIS coverage's for land use, ditching, point source discharges, feedlots, and riparian zone quality, 2) general ratings of each site from excellent to poor based on our first hand knowledge of conditions at the site, 3) a rating based on the habitat score, and 4) a straight percentage of land that is used for agricultural, urban, residential development, or surface mining (i.e. disturbed land use).

The disturbance gradient that combined the GIS watershed rating and the habitat score tended to produce the strongest correlations with the majority of fish community attributes. The watershed/habitat rating incorporated the most prevalent types of disturbance at two different scales; within the watershed and at the reach. The watershed component of the rating was largely independent of natural factors that may affect the structure of a fish community. However, some components of the habitat score may have reflected changes due to human disturbance or natural factors (e.g. the percent fine substrate within the reach could be a reflection of human disturbance or natural geologic features within the watershed).

Eight disturbance factors were used in the watershed rating process (table 11). Each disturbance factor was assigned a value from 0 to 5 for a total of 40 points. A score of 0 represented the worst observable case within the basin. Conversely, a score of 5 represented the best observable case within the basin.

The rating process was somewhat subjective because of the need to consider differences in the size of each watershed, the perceived severity of the disturbance, and the proximity of the disturbance to each site. For example, are three small feedlots located a mile from a stream less or more of a concern than 1 large feedlot located 100 meters upstream? It was assumed that larger disturbances (i.e. larger towns, feedlots, permitted discharges, etc.) or more disturbances (i.e. more agricultural land use, ditches, etc.) had a greater potential to negatively influence stream health than smaller or fewer disturbances. Because of these types of concerns very few guidelines could be used to assign rating values to each site. Instead, the person rating the disturbances at a site needed to develop an overall perspective of the density, severity, and distribution of each disturbance factor within the basin. Then, using that knowledge and their knowledge of stream systems, consider the potential effect of each disturbance at each site.

To obtain the final disturbance gradient the results from the watershed rating process were combined with the habitat score (i.e.

the qualitative habitat score for wadeable streams or the QHEI score for non-wadeable streams) to form a single index. The watershed score and the habitat score were given equal weighting by normalizing the scores to 1 and summing them together. Normalized scores for the watershed rating, habitat score, and the total rating are provided in appendix 5.

Table 11. Watershed disturbance factors and scoring criteria used in the GIS based watershed rating process.

Disturbance Factor	Rating
1. Percent agricultural land use in the watershed	0-5
2. Percent urban land use in the watershed	0-5
3. Number and size of permitted industrial and municipal facilities within the watershed	0-5
4. Number and size of permitted feedlots within the watershed	0-5
5. Number of ditches or channelized streams within the watershed	0-5
6. Condition of riparian buffer within the watershed	0-5
7. Condition of riparian buffer at the site	0-5
8. Channelization at the site	0 or 5

Rating interpretation
0 = worst case
1 = close to the worst
2 = below average
3 = above average
4 = close to the best
5 = best case

Upstream land use in the watershed was characterized using 1990 vintage (MDNR filename: lulcxpy3) or 1995 vintage (MDNR filename lusatpy3) GIS land use coverages depending on which coverage was available for each site. The land use theme was overlaid onto the drainage area theme in Arcview and clipped producing a land use theme identical in shape and size to the drainage area theme. Land uses were then summed across the entire drainage area and divided by the total area to produce percentages for each land use. The percent watershed disturbance was calculated by adding the percentages of land uses that were agricultural, urban or residential, grassland associated with pastured areas, and mines. Basin wide coverages of municipal and industrial discharges and permitted feedlots were obtained through the MPCA permitting database. A MDNR stream trace coverage was used to assess the extent of ditching within the watershed. Aerial photographs (digital ortho quads) were used to assess riparian vegetation at the reach.

STREAM CLASSIFICATION

Proper stream classification is a very important component in IBI development. With too few stream classes it may be difficult to distinguish between natural stream variability and human induced variability (Karr and Chu 1999). On the other hand, the limited resources available to conduct biological monitoring may be wasted with too many stream classes. We considered water temperature, stream size, morphological type (riffle/run or glide/pool), and ecoregion as possible stream classification variables.

Stream temperature greatly influences the structure of the fish community and consequently, the metrics in an IBI (Lyons 1992a; Lyons et al. 1996; Mundahl and Simon 1999). We did not include stream reaches considered to be coldwater in this study. Therefore, any data from a stream that contained a significant population of trout was omitted from the data set. The distinction between warm and coolwater streams is not as easily defined. Lyons (1992a) provided a list of primary and secondary coolwater species for Wisconsin streams. Primary coolwater species are generally restricted to coolwater streams, while secondary coolwater species occur commonly in both cool and warmwater streams. Using these guidelines along with the fish community and water temperature data from this study, we concluded that the majority of unimpaired non-coldwater streams in UMRB have the temperature and fish community characteristics of coolwater streams.

Stream size greatly influences the structure of the fish community and consequently the metrics in an IBI. In the UMRB, there was a strong relationship (r^2=0.566, p<0.0001) between the drainage area ($\log_{10}$) and species richness across the full range of stream sizes (fig. 3). The UMRB IBI accounts for differences in species richness due to stream size by developing separate scoring criteria for 4 stream size classes. To determine size classification break points a scatter plot of watershed drainage area ($\log_{10}$) versus species richness was constructed using all available data (fig. 3). Size classes were then chosen to minimize differences in maximum species richness within each size class. For example, streams with watersheds of 5 to 35 mi^2 were placed into a size class because the maximum species richness within that range of stream sizes was similar. Stream size was not correlated with species richness within each of the four size classes (Spearman r_s, p>0.05).

We used the habitat information to help separate sites into riffle/run and glide/pool morphology classes. In our judgment, the most important habitat features used to distinguish between different stream morphological classes were the presence of riffles within the reach, stream gradient (riffle/run > glide/pool), and width-to-depth ratio (riffle/run > glide/pool). However, other physical stream characteristics such as dominant substrate type (coarse substrates in riffle/run, fine substrates in glide/pool), riparian vegetation (wooded or grass in riffle/run, wetland or wet meadow in glide/pool) and sinuosity (riffle/run < glide/pool) were also important considerations.

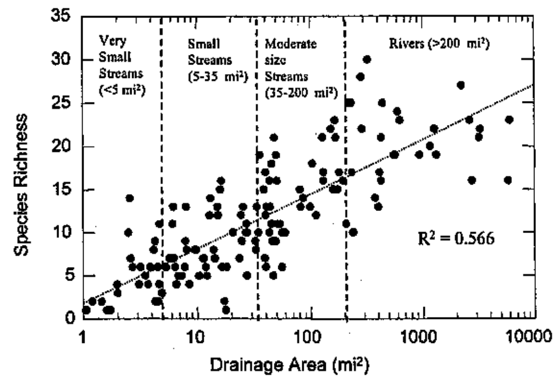


Figure 3. Species richness versus drainage area (mi^2). Vertical lines represent size class break points.

To determine if the stream morphology or ecoregion influenced a candidate metric we compared scatter plots of each candidate metric against the watershed/habitat rating for each ecoregion and stream morphology type (fig. 4). Karr and Chu (1999) refer to these scatterplots as ecological dose response curves. We plotted the residuals of the metric values from linear regressions of each candidate metric against the drainage area ($\log_{10}$) on the Y axis (fig. 4). By plotting the residual metric values it was possible to compare the distributions for each ecoregion or stream morphological type for the entire data set without dividing the data into size classes.

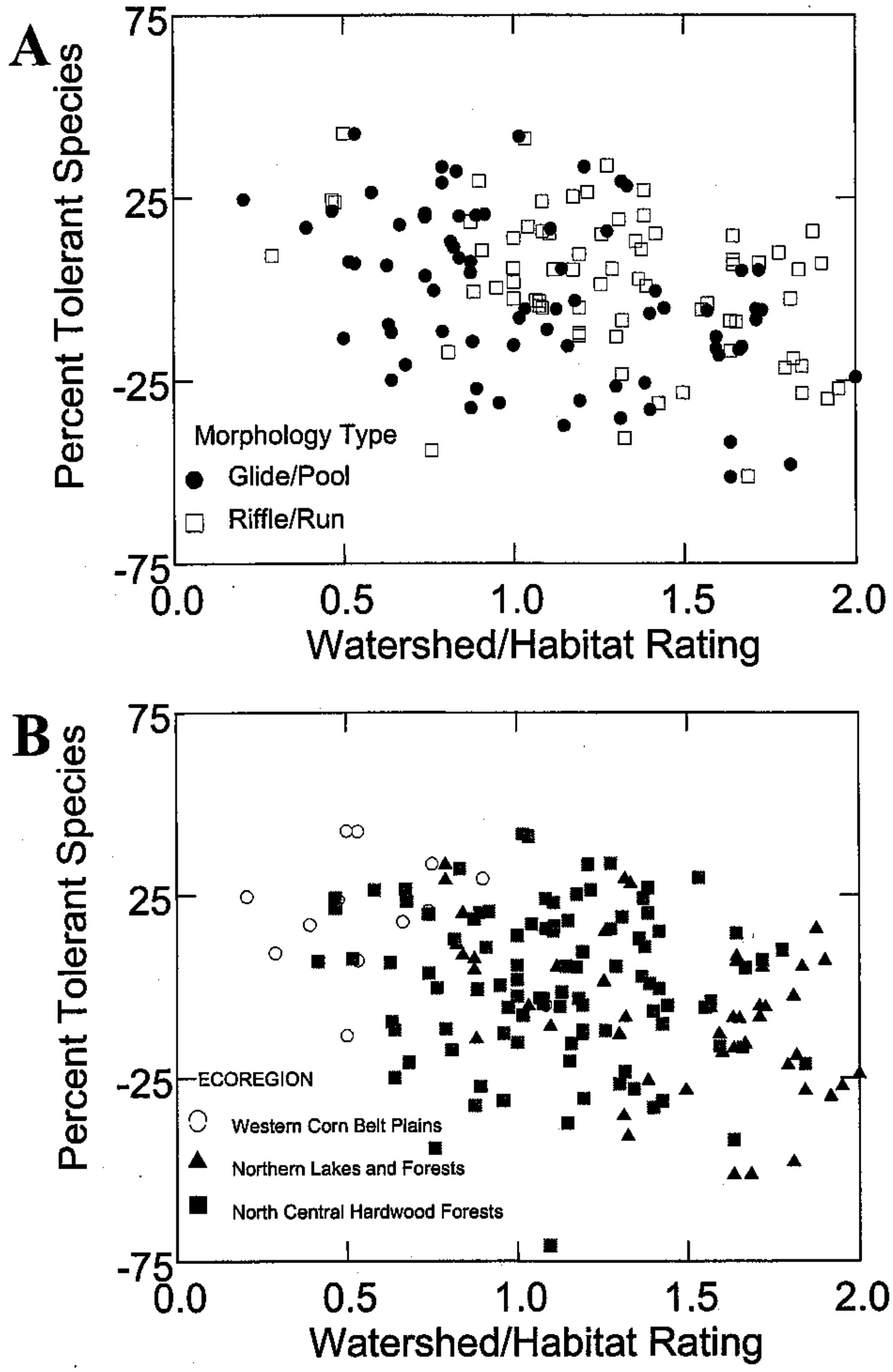


Figure 4. Example of the ecological dose response curves used to identify differences in metric expectations due to (A) stream morphological type and (B) ecoregion for the percent tolerant species metric. For each metric, residual values from a regression of the metric value vs. watershed drainage area (log_{10}) are plotted on the Y-axis against a watershed/habitat rating on the X-axis.

Two characteristics of the dose response curves were used to examine whether there were differences due to stream morphology or ecoregion; 1) a difference in the potential (maximum value) of the candidate metric due to stream morphology or ecoregion and, 2) a notable difference in the dose response due to morphology or ecoregion.

A review of the dose response curves suggested that stream morphology and ecoregion had little influence on the potential or response of each candidate metric to a gradient of disturbance. A lack of highly rated sites in the Western Corn Belt Plains ecoregion limited our ability to detect differences for this ecoregion (fig. 4). However, within the relatively narrow range of disturbance values encompassed by the Western Corn Belt Plains data set, the distributions were similar.

GRAPHICAL ANALYSIS OF CANDIDATE METRICS

Methods

We referred to a list of metrics compiled by Simon and Lyons (1995) to select candidate metrics for inclusion into the UMRB IBI. Many of the metrics listed by Simon and Lyons (1995) have been used successfully in IBI's throughout the Midwest.

Ecological dose response curves were used to select and validate metrics. Attributes of the fish community were plotted against the watershed/habitat rating, or in some cases components of the watershed/habitat rating, to yield an ecological dose response curve for each fish community attribute within each size class of stream. Two properties of the dose response curves were used as criteria for validating the relationship between the fish community attribute and the watershed/habitat rating: 1) The association (correlation) between the watershed/habitat rating and each fish community attribute, and 2) the difference between the attribute values from the 10 highest rated sites and the 10 lowest rated sites. Attribute values for the highest rated sites should separate from the lowest rated sites along the Y-axis of the dose response curve. Attributes of the fish community that demonstrated a response using either of the methods were retained for further consideration.

Spearman r_s correlation coefficients were used to determine the significance of the dose response relationship for each metric value, metric score, and final IBI score against the disturbance gradient. Spearman r_s correlation coefficients were also used to examine the correlation of each metric with one another. A Wilcoxon signed rank test was used to test for significant differences ($p<0.05$) in the metric values between the highest and lowest rated sites. Coefficients of determination from linear regression analyses were used to examine the relationship between the IBI score and each major component of the rating (i.e. the watershed rating and habitat score).

Results

Each of the selected metrics except the DELT anomalies metric demonstrated a detectable dose response when the metric values were plotted against a disturbance gradient. All of the metrics except for the DELT anomalies metric were significantly correlated with the watershed/habitat rating in very small streams, small streams, and rivers (Spearman r_s; $p<0.05$; table 12).

The DELT anomalies metric was not expected to elicit a dose response because the metric is designed to respond to changes in the fish community in only the most degraded streams. The metric has proven useful within other regions of the Midwest as an indicator of industrial pollution (Ohio EPA 1987b). Should human activities within the UMRB intensify the metric will become more valuable.

Seven of the 10 metrics for moderate streams were not statistically correlated with the watershed/habitat rating. The same pattern was observed when the metric values for the least disturbed sites were compared to the most disturbed sites (Wilcoxon signed rank test; $p<0.05$; table 13).

The dose response relationship for moderate streams improved when the habitat score was removed from the disturbance gradient. Five of the 10 metrics were significantly correlated with the watershed component of the rating (Spearman r_s; $p<0.05$; table 14). The dose response relationships also improved when the metric values were converted to scores (Spearman r_s; $p<0.05$; table 15). The DELT anomalies metric in all stream size classes and the number of darters, sculpins, and madtoms, percent tolerant species, and number of fish per 100 meters metrics in the moderate stream size class were the only metric scores not statistically correlated with one of the disturbance gradients. A more refined regional framework, waterbody classification system, or disturbance gradient may improve the dose response relationships and help to explain the relatively weak response observed in moderate size streams.

Table 12. Spearman rank correlation coefficients (r_s) and significance values (p) for each metric and total IBI score against a watershed/habitat rating within each size class.

	Very small streams (< 5 mi^2) (n=24)		Small streams (5-35 mi^2) (n=53)		Moderate streams (35-200 mi^2) (n=42)		Rivers (>200 mi^2) (n=42)	
Metric	correlation coefficient (r_s)	significance value (p)	correlation coefficient (r_s)	significance value (p)	correlation coefficient (r_s)	significance value (p)	correlation coefficient (r_s)	significance value (p)
Species richness and composition metrics								
Total number of species	.519	<.02	.419	<.002	.280	<.1	.314	<.05
Number of wetland species	.442	<.05	.410	<.005	.530	<.001		
Number of minnow species			.437	<.002				
Number of darter, sculpin and madtom species					.255	<.2	.385	<.02
Number of intolerant species			.296	<.05	.061	>.5	.608	<.001
Percent tolerant species	-.487	<.01	-.500	<.001	-.004	>.5	-.512	<.001
Percent dominant two species	-.517	<.02	-.414	<.005				
Trophic and reproductive function metrics								
Number of invertivore species	.485	<.02	.403	<.005	.274	<.1	.333	<.05
Percent omnivore species							-.497	<.002
Number of piscivore species					.354	<.05	.368	<.02
Percent simple lithophils			.357	<.01	.630	<.001	.544	<.001
Fish abundance and condition metrics								
Number of fish per 100 meters	.419	<.05	.424	<.002	.109	<.5	.510	<.001
Percent DELT anomalies	-.324	<.2	.060	>.5	-.041	>.5	-.263	<.1
Total IBI score	.513	<.02	.578	<.001	.513	<.001	.614	<.001

Table 13. Median, inter-quartile range, and significance values (Wilcoxon signed rank test; $p<0.05$) for each metric within each stream size class at the 10 highest rated sites (good) and the 10 lowest rated sites (poor).

		Very Small Streams (<5 mi^2 drain. area)			Small streams (5-35 mi^2 drain. area)			Moderate streams (35-200 mi^2 drain. area)			Rivers (>200 mi^2 drain. area)		
		median	quartiles	*p*	median	quartiles	*p*	median	quartiles	*p*	median	quartiles	*p*
Species richness and composition metrics													
Total number of species	Good	6.0	6.0 - 9.0	.03	9.0	7.0 - 13.0	.02	15.0	13.0 – 16.0	.10	22.5	21.0 – 25.0	.04
	Poor	3.0	2.0 - 4.0		6.0	4.0 - 7.0		15.0	12.0 – 18.0		16.5	11.0 – 19.0	
Number of darter, sculpin, & madtom species	Good							2.0	1.0 – 3.0	.39	3.0	3.0 – 4.0	.01
	Poor							1.0	1.0 – 3.0		1.5	1.0 – 2.0	
Number of wetland species	Good	2.0	1.0 - 3.0	.13	2.0	1.0 - 3.0	.01	3.0	3.0 – 3.0	<.01			
	Poor	0.5	0.0 - 1.0		0.0	0.0 - 1.0		0.5	0.0 – 1.0				
Number of minnow species	Good				3.0	2.0 - 4.0	.01						
	Poor				0.0	0.0 - 1.0							
Percent dominant two species	Good	78.5	59.5 - 84.3	.04	66.5	51.4 - 71.5	.01						
	Poor	92.3	86.8 - 100.0		91.0	86.6 - 96.5							
Number of intolerant species	Good				0.5	0.0 - 1.0	.10	0.0	0.0 – 1.0	.58	2.5	2.0 – 4.0	.01
	Poor				0.0	0.0 - 0.0		0.0	0.0 – 1.0		0.0	0.0 – 0.0	
Percent tolerant species	Good	73.6	56.8 - 88.0	.03	58.1	32.9 - 70.3	.01	58.5	24.7 – 65.5	.87	11.9	8.5 – 19.6	.01
	Poor	94.4	84.0 - 100.0		97.4	89.2 - 100		43.5	31.6 – 59.3		50.8	29.3 - 66.4	

Table 13. (continued)

		Very Small Streams (<5 mi^2 drain. area)			Small streams (5-35 mi^2 drain. area)			Moderate streams (35-200 mi^2 drain. area)			Rivers (>200 mi^2 drain. area)		
		median	quartiles	*p*	median	quartiles	*p*	median	quartiles	*p*	median	quartiles	*p*
Trophic and reproductive function metrics													
Number of invertivore species	Good	1.0	1.0 - 2.0	.05	2.0	1.0 – 3.0	.03	5.5	3.0 – 7.0	.17	11.0	9.0 – 12.0	.03
	Poor	1.0	0.0 - 1.0		0.0	0.0 – 1.0		3.5	3.0 – 5.0		7.0	3.0 – 8.0	
Percent omnivore species	Good										7.2	2.2 – 10.0	.03
	Poor										31.9	12.7 – 53.7	
Number of piscivore species	Good							2.5	1.0 – 3.0	.01	5.0	4.0 – 7.0	.01
	Poor							1.0	0.0 – 1.0		3.0	2.0 – 4.0	
Percent simple lithophils	Good				33.4	1.6 – 40.7	.02	48.7	16.5 – 54.0	.02	46.8	36.6 – 57.6	.03
	Poor				0.2	0.0 – 0.7		1.5	0.7 – 3.7		11.8	7.2 – 26.1	
Fish abundance and condition metrics													
Number of fish per 100 meters	Good	13.2	6.5 - 27.6	.20	6.7	2.5 – 20.1	.02	2.4	2.0 – 10.5	.87	113.7	57.4 – 144.4	.20
	Poor	1.6	0.0 - 10.1		1.3	0.0 – 6.0		6.1	2.0 – 7.7		20.5	15.3 – 38.8	
Percent DELT anomalies	Good	0.0	0.0 – 0.0		0.0	0.0 – 0.1	.60	0.3	0.0 – 1.1	.95	0.1	0.0 - 0.5	.28
	Poor	1.3	0.0 – 6.9	.04	0.0	0.0 – 0.4		0.4	0.0 – 1.0		0.6	0.0 – 0.9	
Total IBI score	Good	75.7	60.0 – 85.8	.02	60.5	52.0 – 68.0	<.01	65.0	48.0 – 79.0	.01	79.5	78.0 – 82.0	<.01
	Poor	18.5	14.3 – 55.7		21.0	10.0 – 29.0		39.5	35.0 – 54.0		39.0	22.0 – 62.0	

Table 14. Spearman rank correlation coefficients (r_s) and significance values (p) for moderate size stream metrics and the total IBI score against a watershed rating, habitat score, a combined watershed/habitat rating, and percent disturbed land use.

	Watershed rating (n=42)		Habitat score (n=42)		Watershed/habitat rating (n=42)		Percent disturbed land use (n=42)	
Metric	correlation coefficient (r_s)	significance value (p)	correlation coefficient (r_s)	significance value (p)	correlation coefficient (r_s)	significance value (p)	correlation coefficient (r_s)	significance value (p)
Species richness and composition metrics								
Total number of species	.356	<.02	.160	<.5	.280	<.1	-.340	<.05
Number of wetland species	.546	<.001	.416	<.02	.530	<.001	-.355	<.05
Number of darter, sculpin and madtom species	.277	<.1	.246	<.2	.255	<.2	-.266	<.1
Number of intolerant species	.014	>.5	.146	<.5	.061	>.5	-.067	>.5
Percent tolerant species	-.178	<.25	.171	<.5	-.004	>.5	.264	<.1
Trophic and reproductive function metrics								
Number of invertivore species	.363	<.02	.173	<.5	.274	<.1	-.288	<.1
Number of piscivore species	.494	<.002	.115	<.5	.354	<.05	-.572	<.001
Percent simple lithophils	.567	<.001	.590	<.001	.630	<.001	-.276	<.1
Fish abundance and condition metrics								
Number of fish per 100 meters	.177	<.25	.020	>.5	.109	<.5	-.220	<.2
Percent DELT anomalies	.130	<.5	-.197	<.5	-.041	>.5	-.210	<.2
Total IBI score	.569	<.001	.377	<.02	.513	<.001	-.468	<.005

Table 15. Spearman rank correlation coefficients (r_s) and significance values (p) for each metric score against a watershed/habitat rating or estimate of disturbed land use (marked with asterisk) within each size class.

	Very small streams (< 5 mi^2) (n=24)		Small streams (5-35 mi^2) (n=53)		Moderate streams (35-200 mi^2) (n=42)		Rivers (>200 mi^2) (n=42)	
Metric	correlation coefficient (r_s)	significance value (p)	correlation coefficient (r_s)	significance value (p)	correlation coefficient (r_s)	significance value (p)	correlation coefficient (r_s)	significance value (p)
Species richness and composition metrics								
Total number of species	.501	<.02	.409	<.002	-.328*	<.05	.334	<.05
Number of wetland species	-.563*	<.005	.514	<.001	.546	<.001		
Number of minnow species			.504	<.001				
Number of darter, sculpin and madtom species					-.268*	<.1	.378	<.02
Number of intolerant species			.364	<.01	.471	<.005	.683	<.001
Percent tolerant species	.434	<.05	.499	<.001	-.197*	<.5	.570	<.001
Percent dominant two species	.473	<.05	.435	<.002				
Trophic and reproductive function metrics								
Number of invertivore species	.483	<.02	.487	<.001	-.325*	<.05	.393	<.02
Percent omnivore species							.519	<.001
Number of piscivore species					.448	<.005	.386	<.02
Percent simple lithophils			.262	<.1	.558	<.001	.502	<.001
Fish abundance and condition metrics								
Number of fish per 100 meters	.500	<.02	.459	<.001	-.180*	<.5	na	na
Percent DELT anomalies	.083	>.5	-.027	>.5	.014	>.5	.105	>.5

The final IBI scores for all four stream size classes were significantly correlated with the watershed/habitat rating (Spearman r_s; $p<0.02$; table 12). Also, the IBI scores at the least disturbed sites were significantly different from the most disturbed sites (Wilcoxon signed rank test; $p<0.05$; table 13).

The relationship between the watershed/habitat rating and the IBI score was strongest in rivers followed by small streams, very small streams, and moderate size streams (fig. 5).

There was also a significant positive linear relationship between IBI scores and the 2 components of the disturbance gradient; the watershed rating and the habitat score (table 16). The relationship was the strongest when the watershed rating and the habitat score were combined except for moderate streams where the relationship was slightly weaker. Stauffer et al. (2000) found that watershed level disturbance factors as well as near stream factors influence fish community composition in streams of the Minnesota River Basin. Similarly, this study demonstrates that both spatial scales

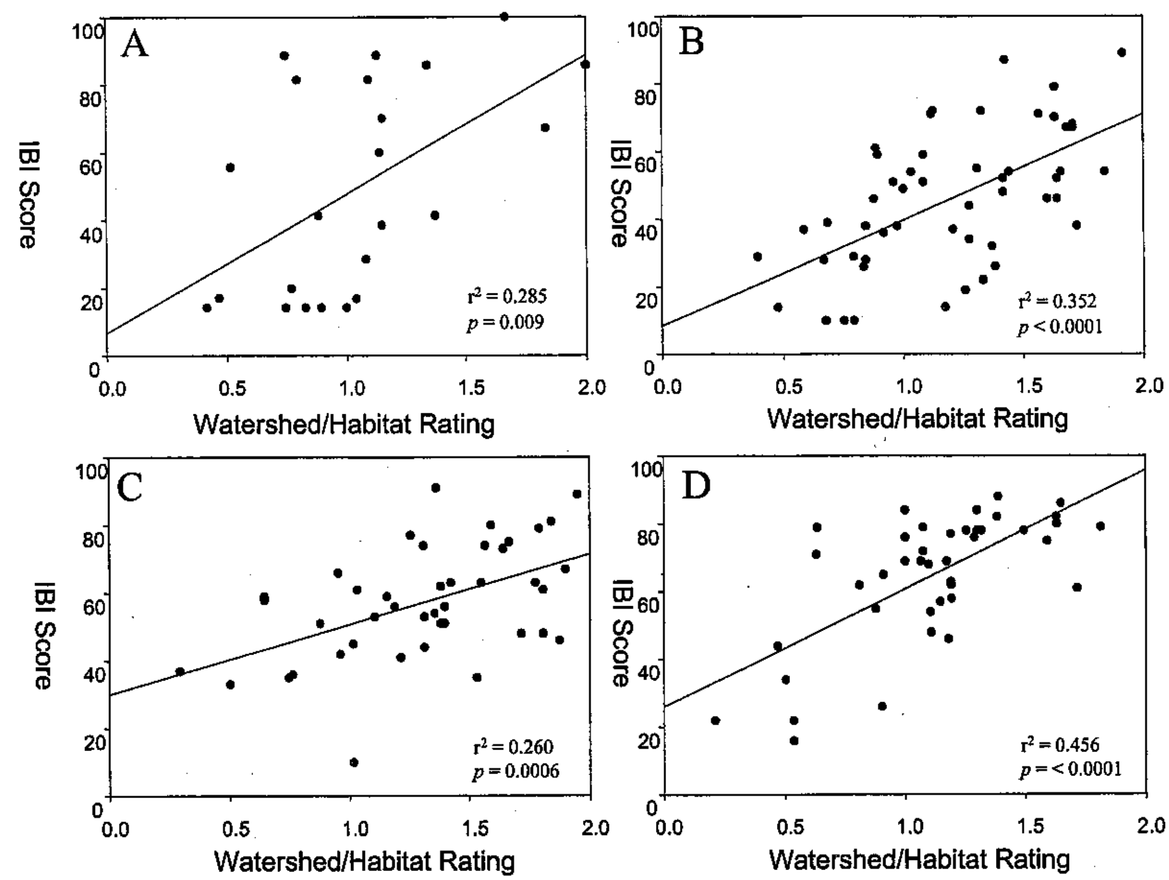


Figure 5. Index biological integrity (IBI) scores plotted against disturbance (an index based on watershed land use and in-stream habitat) for (A) very small streams (<5 mi^2 drainage area), (B) small streams (5-35 mi^2 drainage area), (C) moderate size streams (35-200 mi^2 drainage area), and (D) rivers (>200 mi^2 drainage area).

Table 16. Coefficients of determination and (*p*) values for the relationship between IBI scores and various measures of disturbance including: (A) a watershed rating, (B) a habitat score, and (C) the combined watershed/habitat rating. Streams are separated into size classes based on drainage area (mi^2).

	Watershed Rating	Habitat Score	Watershed/ Habitat Rating
Stream Size by drainage area (mi^2)	r^2 (*p*)	r^2 (*p*)	r^2 (*p*)
Very small (0-5)	0.250 (0.0127)	0.197 (0.0339)	0.285 (0.0014)
Small (5-35)	0.230 (0.0003)	0.252 (0.0001)	0.352 (<0.0001)
Moderate (35-200)	0.321 (<0.0001)	0.123 (0.0225)	0.260 (0.0006)
Rivers (>200)	0.413 (<0.0001)	0.307 (0.0002)	0.456 (<0.0001)

are important in streams of the UMRB.

Six out of 156 possible metric pairs were highly correlated with each other ($r_s>0.8$; table 17). Three of the 6 highly correlated metric pairs involved the species richness metric. For example, the species richness metric was highly correlated with the number of invertivore species in moderate size streams and rivers. We chose not to reduce the number of metrics in the IBI based solely on their statistical correlation with other metrics. In each case the biological basis for including the metric was sufficient to warrant the inclusion of both correlated metrics in the IBI. Furthermore, given a different set of environmental conditions (i.e. different types of disturbance) each metric may respond in a non-parallel manner.

The metrics were highly correlated with IBI scores for each size class with the exception of the number of intolerant species in moderate size streams which was marginally significant ($p< 0.06$), and the DELT anomalies metric where there was no correlation for any stream size class (Spearman r_s; $p<0.05$; table 18). The lack of a statistical correlation with the DELT metric is not surprising for the reasons mentioned previously. The metric was retained because of its use in detecting severe impairment.

Table 17. Metric pairs with the highest Spearman rank correlations (>0.8 r_s) for each stream size class.

Metric Pairs		Spearman Correlation (r_s)
Very Small Streams		
Total number of species	Percent dominant two species	-0.943
Number of wetland species	Percent tolerant species	-0.817
Percent tolerant species	Number of fish per 100 meters	-0.887
Small Streams		
Number of minnow species	Number of fish per 100 meters	.824
Moderate Streams		
Total number of species	Number of invertivore species	.840
Rivers		
Total number of species	Number of invertivore species	.886

Table 18. Spearman rank correlation coefficients (r_s) and significance values (p) between IBI score and individual metrics for each stream size class.

	Very small streams (< 5 mi^2) (n=24)		Small streams (5-35 mi^2) (n=53)		Moderate streams (35-200 mi^2) (n=42)		Rivers (>200 mi^2) (n=42)	
Metric	correlation coefficient (r_s)	significance value (p)	correlation coefficient (r_s)	significance value (p)	correlation coefficient (r_s)	significance value (p)	correlation coefficient (r_s)	significance value (p)
Species richness and composition metrics								
Total number of species	.874	<.001	.824	<.001	.793	<.001	.778	<.001
Number of wetland species	.866	<.001	.674	<.001	.623	<.001		
Number of minnow species			.642	<.001				
Number of darter, sculpin and madtom species					.615	<.001	.682	<.001
Number of intolerant species			.518	<.001	.293	<.06	.764	<.001
Percent tolerant species	-.881	<.001	-.827	<.001	-.329	<.05	-.470	<.005
Percent dominant two species	-.819	<.001	-.736	<.001				
Trophic and reproductive function metrics								
Number of invertivore species	.839	<.001	.816	<.001	.824	<.001	.743	<.001
Percent omnivore species							-.559	<.001
Number of piscivore species					.553	<.001	.636	<.001
Percent simple lithophils			.314	<.05	.623	<.001	.332	<.05
Fish abundance and condition metrics								
Number of fish per 100 meters	.909	<.001	.796	<.001	.653	<.001	.720	<.001
Percent DELT anomalies	-.002	>.5	.204	<.2	.075	<.5	-.040	>.5

APPENDIX 2 – UPPER MISSISSIPPI RIVER BASIN FISH ASSEMBLAGE* AND IBI CLASSIFICATION

Scientific name	Common name	IBI Classification[a]		
		Taxa	Trophic status	Reproductive guild
Lepisosteidae	**Gars**			
Lepisosteus platostomus	Shortnose gar**		Pi	
Amiidae	**Bowfins**			
Amia calva	Bowfin		Pi	
Cyprinidae	**Minnows**			
Campostoma anomalum	Central stoneroller	Mi		
Cyprinella spiloptera	Spotfin shiner	Mi	Ins	
Cyprinus carpio	Common carp	Mi To	Om	
Hybognathus hankinsoni	Brassy minnow	Mi		
Luxilus cornutus	Common shiner	Mi		SI
Margariscus margarita	Pearl dace	Mi We	Ins	
Nocomis biguttatus	Hornyhead chub	Mi Int	Ins	
Notemigonus crysoleucas	Golden shiner	Mi To We		
Notropis anogenus	Pugnose shiner***	Mi Int		
Notropis atherinoides	Emerald shiner	Mi	Ins	SI
Notropis dorsalis	Bigmouth shiner	Mi	Ins	
Notropis heterodon	Blackchin shiner	Mi Int	Ins	
Notropis heterolepis	Blacknose shiner	Mi Int	Ins	
Notropis hudsonius	Spottail shiner	Mi Int	Ins	
Notropis ludibundus	Sand shiner	Mi	Ins	
Notropis texanus	Weed shiner	Mi Int		
Notropis volucellus	Mimic shiner	Mi Int	Ins	
Pimephales notatus	Bluntnose minnow	Mi To		
Pimephales promelas	Fathead minnow	Mi To We	Om	
Pimephales vigilax	Bullhead minnow	Mi		
Phoxinus eos	Northern redbelly dace	Mi We		
Phoxinus neogaeus	Finescale dace	Mi We	Ins	
Rhinichthys atratulus	Blacknose dace	Mi To		SI
Rhinichthys cataractae	Longnose dace	Mi Int	Ins	SI
Semotilus atromaculatus	Creek chub	Mi To		
Catostomidae	**Suckers**			
Catostomus commersoni	White sucker	To	Om	SI
Hypentelium nigricans	Northern hogsucker	Int	Ins	SI
Ictiobus cyprinellus	Bigmouth buffalo	To	Om	
Moxostoma macrolepidotum	Shorthead redhorse		Ins	SI
Moxostoma anisurum	Silver redhorse		Ins	SI
Moxostoma valenciennesi	Greater redhorse	Int	Ins	SI
Ictaluridae	**Catfishes**			
Ictalurus punctatus	Channel catfish		Pi	
Noturus gyrinus	Tadpole madtom	DSM We	Ins	
Noturus flavus	Stonecat	DSM Int	Ins	
Pylodictis olivaris	Flathead catfish		Pi	
Ameiurus melas	Black bullhead	To We	Om	
Ameiurus natalis	Yellow bullhead	We	Om	
Ameiurus nebulosus	Brown bullhead	We	Om	

APPENDIX 2. (continued)

Scientific name	Common name	IBI Classification[a] Taxa	Trophic status	Reproductive guild
Esocidae	**Pikes**			
Esox lucius	Northern pike	We	Pi	
Esox masquinongy	Muskellunge	Int	Pi	
Umbridae	**Mudminnows**			
Umbra limi	Central mudminnow	To We	Ins	
Osmeridae	**Smelts**			
Osmerus mordax	Rainbow smelt			
Salmonidae	**Trouts**			
Coregonus artedii	Cisco (lake herring)			
Coregonus clupeaformis	Lake whitefish		Ins	
Oncorhynchus mykiss	Rainbow trout		Pi	
Salmo trutta	Brown trout		Pi	
Salvelinus namaycush	Lake trout		Pi	
Salvelinus fontinalis	Brook trout	Int	Pi	
Percopsidae	**Trout-perches**			
Percopsis omiscomaycus	Trout-perch		Ins	
Gadidae	**Codfishes**			
Lota lota	Burbot		Pi	Sl
Atherinidae	**Silversides**			
Labidesthes sicculus	Brook silverside		Ins	
Fundulidae	**Killifishes**			
Fundulus diaphanus	Banded killifish		Ins	
Gasterostidae	**Sticklebacks**			
Culaea inconstans	Brook stickleback	To We	Ins	
Pungitius occidentalis	Ninespine stickleback		Ins	
Cottidae	**Sculpins**			
Cottus cognatus	Slimy sculpin	DSM Int	Ins	
Cottus bairdi	Mottled sculpin	DSM Int	Ins	
Centrarchidae	**Sunfishes**			
Ambloplites rupestris	Rock bass	Int	Pi	
Lepomis cyanellus	Green sunfish	To		
Lepomis macrochirus	Bluegill		Ins	
Lepomis gibbosus	Pumpkinseed		Ins	
Lepomis megalotis	Longear sunfish	Int	Ins	
Lepomis humilis	Orangespotted sunfish		Ins	
Micropterus dolomieui	Smallmouth bass	Int	Pi	
Micropterus salmoides	Largemouth bass		Pi	
Pomoxis nigromaculatus	Black crappie		Pi	

APPENDIX 2. (continued)

Scientific name	Common name	IBI classification[a] Taxa	Trophic status	Reproductive guild
Percidae	**Perches**			
Etheostoma nigrum	Johnny darter	DSM	Ins	
Etheostoma exile	Iowa darter	DSM Int We	Ins	
Etheostoma microperca	Least darter***	DSM Int	Ins	
Perca flavescens	Yellow perch		Ins	
Percina caprodes	Logperch	DSM	Ins	Sl
Percina maculata	Blackside darter	DSM	Ins	Sl
Stizostedion vitreum	Walleye		Pi	Sl

[a] Taxa- DSM=darters, sculpins, and madtom species, Mi=minnows, Int=intolerant, To=tolerant, We=wetland
Trophic status- Ins=invertivore, Om=omnivore, Pi=piscivore
Reproductive guild- Sl=simple lithophil
* Fish species list is from Hatch, J.T. and K. Schmidt (2001)
** Fish species not collected in Upper Mississippi River Basin since 1975
*** Minnesota listed special concern species

APPENDIX 3 - SCORING METRICS

DEFINING SCORING LINES

Ecological dose response curves (Karr and Chu 1999) were used to score each metric (fig. 6). Scoring was accomplished by drawing a horizontal line through the dose response curve so that approximately 5 percent of the observations were above the line. This line is referred to as the Maximum Species Richness (MSR) line. Four equally spaced horizontal lines were then drawn below the MSR line to divide the graph into five separate sections. A score of 10 was assigned to the area of the graph immediately below the MSR line followed by a 7, 5, 2, and finally a 0 value in the lowest section of the graph. For metrics that responded negatively to disturbance the scoring process was just the opposite, with the MSR line defining the lower portion of the graph.

There are a few notable exceptions to this process:

1) *Number of fish per 100 meters*: The graph was divided into 2 sections and assigned a score of 0 for values of 5 or less fish per 100 meters or a score of 10 for values greater than 5 fish per 100 meters. Therefore, this metric will receive a score of 10 unless the number of fish (not including tolerant species) collected at the site is extremely low.

2) *Percent DELT anomalies*: Scored a 10 if the percent occurrence of DELT anomalies was 1% or less, 5 if the occurrence was between 1% and 3%, and 0 if the percent occurrence was 4% or greater.

3) *Number of wetland, invertivore, and intolerant species*: These metrics could not always be divided into 5 scoring categories because the maximum number of species was sometimes less than 4, particularly in the smaller stream size classes. In this situation, metrics were divided into 3 scoring categories and assigned scores of 0, 5 or 10.

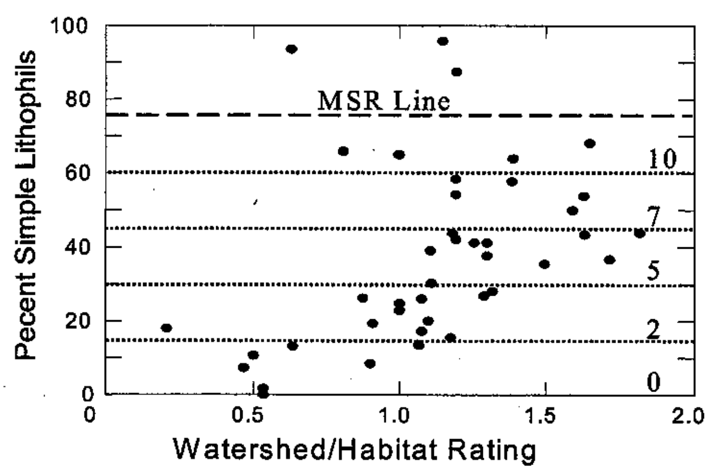


Figure 4. An ecological dose response curve and scoring criteria for the percent simple lithophilic species metric.

APPENDIX 4 - IBI VARIABILITY

We sampled 13 sites twice within a single sampling season to examine the variability of IBI scores (fig. 7). The sites ranged in size (6 to 35,593 mi^2 drainage area) and level of watershed disturbance (0.3% to 86%). The repeat sampling events occurred from June through September. The repeat samples were taken 6 to 50 days (mean=26 days) from the initial visit. IBI scores between repeat visits were not significantly different from each other (Student's paired t-test; p=.249). The mean (±SE) difference in IBI scores was 8.38±2.68(n=13). A difference of ±13 IBI points represented a 95% confidence limit for any given IBI score.

The variability of IBI scores was less at sites that were reference quality. The 2 reference quality sites had very consistent IBI scores between sampling periods, differing only by 1 and 6 IBI points. Scores at the 11 non-reference sites were more variable ranging from no difference between sampling periods to 35 IBI points between sampling periods. Niemela and Feist (2000) found that IBI scores were also more variable in disturbed streams in the adjacent St. Croix River Basin. Fish communities in more disturbed environments are usually less stable because they haven't developed the adaptive strategies (physiological or behavioral) that enable them to cope with the additional sources of stress in their environment (Fore et al 1994).

The most variable site, Mayhew Creek (field number = 00UM042), was channelized and heavily influenced by non-point pollution. Differences in IBI scores at this site may have been related to fluctuating dissolved oxygen levels. During the first visit in July the dissolved oxygen was 4.0 mg/l and the IBI score was 10. In August, the dissolved oxygen was 9.5 mg/l and the IBI score had risen to 45.

Niemela and Feist (2000) noted that the IBI scores from rivers (>270 mi^2 drainage area) in the adjacent St. Croix River Basin were more variable than the stream sites (<270 mi^2 drainage area). This did not appear to be the case in the UMRB where repeat samples from streams (<200 mi^2) differed by a mean (±SE) of 10.33±5.73 (n = 6) IBI points and from rivers (>200 mi^2) by 6.71±1.47 (n = 7) IBI points.

The variability in IBI scores between years was not measured during the UMRB study. However, in the St. Croix River Basin (Niemela and Feist 2000) the year to year variability was lower at sites that were likely to experience very little disturbance between sampling periods. In the St. Croix River Basin, the minimum difference in IBI score that was necessary to detect a change in biological integrity 95 % of the time was 10 IBI points.

The replicate sites selected in the UMRB did not include very small streams (<5 mi^2 drainage area). Future work should focus on obtaining an adequate number of stations in each of the 4 size classes to examine within and among year variation in IBI scores.

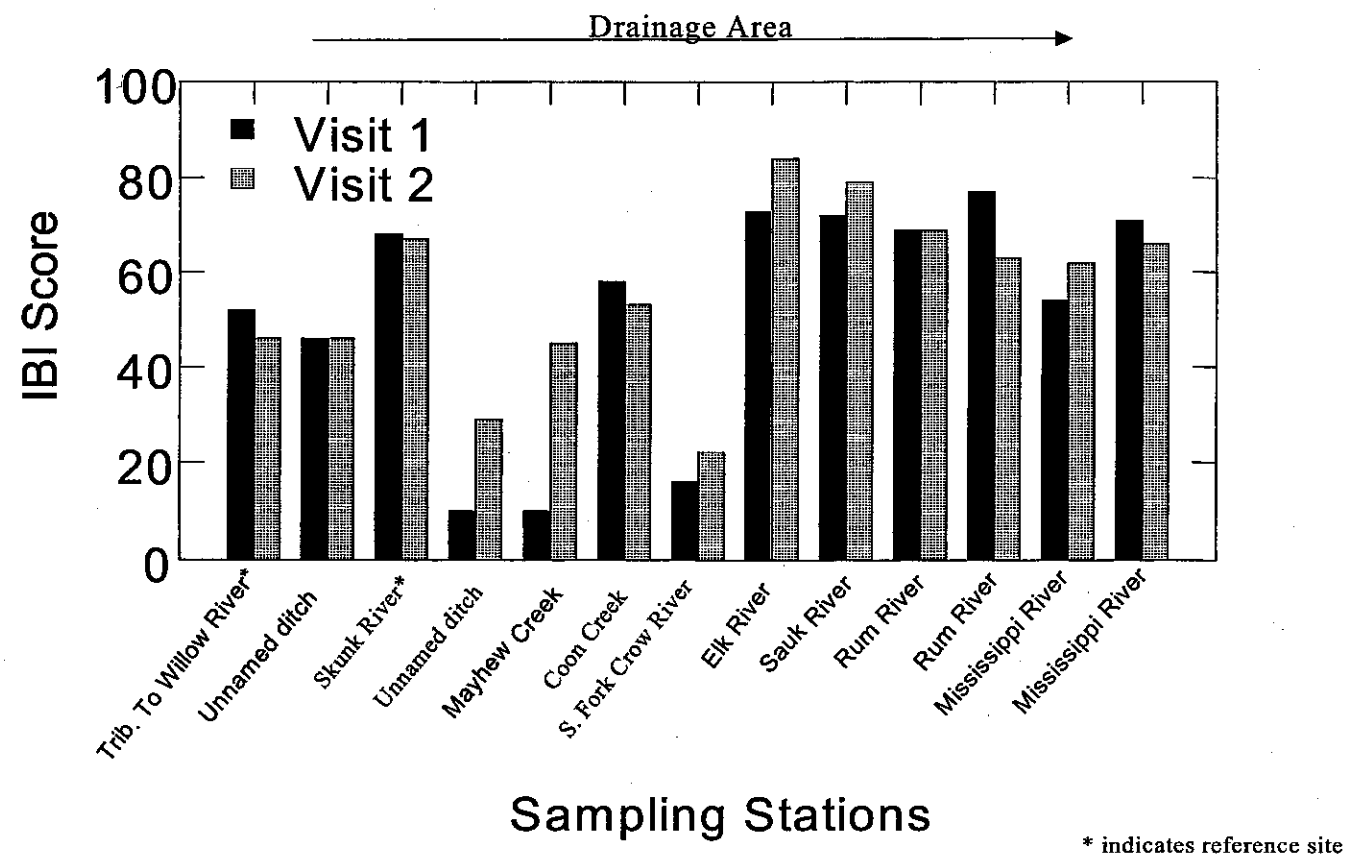


Figure 7. Index of biological integrity (IBI) scores from replicate samples taken during the 1999 and 2000 study period. An * indicates that the site is of reference quality.

APPENDIX 5 – UPPER MISSISSIPPI RIVER BASIN SAMPLING SITES

Stream Name	Sample Date	Drainage Area (mi^2)	Field Number[1]	County	Location	Latitude[2]	Longitude	IBI[3]	Land Rate[4]	Habitat Rate[5]	Total Rate[6]	Land Use %[7]
					Very Small Streams (< 5 mi^2 drainage area)							
unnamed creek	7/9/99	0.10	99UM024	Anoka	2.5 mi. N. of Lino Lakes	45.20462	93.08264	14	0.50	0.50	1.00	80.54
trib. to Cty Ditch 17(SE)	10/13/99	0.22	99UM109	Anoka	Spring Brook Nature Center	45.12175	93.27281	82	0.73	0.37	1.09	68.85
ditch to Mississippi R	7/28/99	0.60	99UM065	Crow Wing	2.5 mi. N. of Iron Hub	46.55338	93.85298	41	0.75	0.13	0.88	1.81
Sarita Creek	9/23/99	1.06	99UM093	Ramsey	U of MN St. Paul campus	44.97917	93.18077	14	0.35	0.07	0.42	98.99
unnamed ditch	6/22/99	1.20	99UM015	Aitkin	~3.0 mi. S.W. of Palisade	46.67187	93.50588	ns	0.45	0.33	0.78	69.84
Phalen Creek	6/12/99	1.45	99UM072	Ramsey	downstream of Swede Hollow Park	44.95972	93.07667	ns	0.60	0.33	0.93	98.20
County Ditch #23	8/17/99	1.60	99UM040	Meeker	~3 mi. N.W. of Cosmos	44.97914	94.71110	ns	0.35	0.17	0.52	99.32
unnamed creek	6/24/99	1.71	99UM007	Wadena	2.0 S.W. of Sebeka	46.59601	95.11783	14	0.58	0.25	0.83	71.45
trib. To Mississippi	8/21/00	1.97	00UM086	Anoka	downstream of C.R. 10 in Blaine	45.13124	93.26583	20	0.35	0.42	0.77	86.92
unnamed creek	7/1/99	2.00	99UM005	Stearns	2.0 mi. NE of Watkins	45.33592	94.37907	ns	0.40	0.42	0.82	64.82
trib to County Ditch 17 (NW)	10/13/99	2.48	99UM110	Anoka	Spring Brook Nature Center	45.12398	93.27750	86	0.63	0.72	1.34	82.35
trib. to Bluebill Lake*	**7/19/00**	**2.57**	**00UM005**	**Itasca**	**downstream of C.R. 52**	**47.62830**	**93.39102**	**100**	**1.00**	**0.67**	**1.67**	**2.51**
unnamed creek	7/21/99	2.60	99UM004	Anoka	@Ham Lake ~1 mile E. of Hwy. 65	45.24389	93.21799	82	0.38	0.42	0.79	49.69
trib. to Sauk River	7/8/99	2.70	99UM029	Stearns	~2.0 mi. W. of St. Martin	45.49662	94.70526	29	0.50	0.58	1.08	97.65
trib. To Bassett Creek	8/31/00	3.08	00UM094	Hennepin	@ 32nd Avenue in Crystal	45.02089	93.36128	17	0.38	0.67	1.04	92.12
Pigeon River	**7/12/00**	**3.19**	**00UM008**	**Itasca**	**downstream of culvert off F.R. 2382**	**47.58834**	**94.18702**	**67**	**1.00**	**0.83**	**1.83**	**0.71**
unnamed creek	6/29/99	3.50	99UM002	Ottertail	9 mi. E. of Henning	46.35034	95.26299	60	0.48	0.67	1.14	68.46
trib. to Zuleger Creek	6/30/99	3.70	99UM043	Benton	4 mi. N.W. of Rice	45.78353	94.14326	56	0.35	0.17	0.52	89.81
Nicollet Creek	**8/2/00**	**3.88**	**00UM002**	**Clearwater**	**Itasca State Park**	**47.19315**	**95.23087**	**86**	**1.00**	**1.00**	**2.00**	**0.00**
Unnamed ditch	7/13/99	4.15	99NF012	Itasca	~2.0 miles NE of Philbin	47.18167	93.31114	56	0.73			8.94
Willow Creek	6/20/99	4.28	99UM080	Ramsey	downstream of CR "D"	45.03445	93.04167	89	0.50	0.63	1.13	78.39
unnamed creek	6/30/99	4.30	99UM033	Mille Lacs	1.0 N.W. of Estes Brook	45.66159	93.75188	14	0.48	0.42	0.89	85.73
trib. to N. Fork Crow R.	7/7/99	4.50	99UM025	Wright	1.0 mi. W. of Rassat	45.15433	94.01125	41	0.63	0.75	1.38	92.20
Valley Creek	6/17/99	4.50	99UM078	Dakota	upstream of Marie St.	44.89056	93.12889	39	0.65	0.50	1.15	69.71
trib. to South Fork Crow R.	6/28/00	4.54	00UM055	Meeker	E. of 590th St.	44.89638	94.58439	14	0.33	0.42	0.74	97.65
Fish Creek	6/14/99	4.65	99UM073	Ramsey	upstream of CR 161	44.89944	92.99306	70	0.68	0.48	1.15	69.95
County Ditch #4	6/30/99	4.70	99UM013	Mille Lacs	2.0 mi. S.E. of Pease	45.67929	93.60711	89	0.33	0.42	0.74	97.32
trib. to North Fork Crow R.	6/28/00	4.90	00UM057	Meeker	downstream of C.R. 11	45.13884	94.45798	17	0.30	0.17	0.47	97.86

APPENDIX 5. (continued)

Stream Name	Sample Date	Drainage Area (mi²)	Field Number[1]	County	Location	Latitude[2]	Longitude	IBI[3]	Land Rate[4]	Habitat Rate[5]	Total Rate[6]	Land Use %[7]
					Small Streams (5 to 35 mi² drainage area)							
trib. to Watab River	6/26/00	5.21	00UM036	Stearns	upstream of C.R. 4	45.64065	94.32416	34	0.78	0.50	1.28	49.99
Little Buffalo Creek	7/6/00	5.35	00UM015	Crow Wing	downstream of Hwy. 371	46.34243	94.20192	19	0.68	0.58	1.26	58.68
Moose Creek	**6/23/99**	**5.70**	**99UM001**	**Itasca**	**4.5 mi. NW of Alvwood**	**47.71598**	**94.37437**	**46**	**0.85**	**0.75**	**1.60**	**15.49**
trib. to Willow River	**7/18/00**	**6.01**	**00UM014**	**Cass**	**10 mi. E of Remer**	**46.98470**	**93.79281**	**46**	**0.98**	**0.67**	**1.64**	**0.30**
trib. to Willow River	**8/2/00**	**6.01**	**00UM014**	**Cass**	**10 mi. E of Remer**	**46.98470**	**93.79281**	**52**	**0.98**	**0.67**	**1.64**	**0.30**
trib to Sauk River	6/30/99	6.10	99UM064	Stearns	0.5 mi. W. of Farming	45.51737	94.60640	48	0.50	0.92	1.42	95.47
trib. to Shell River	**7/14/99**	**6.20**	**99UM047**	**Becker**	**Smoky Hills State Forest**	**46.91625**	**95.36414**	**67**	**0.85**	**0.83**	**1.68**	**21.80**
Spring Brook	**7/7/00**	**6.26**	**00UM029**	**Todd**	**E. of C.R. 29 on gravel road**	**46.01968**	**94.71226**	**54**	**0.93**	**0.92**	**1.84**	**31.31**
tributary to Swan River	**6/22/99**	**6.40**	**99UM056**	**Itasca**	**~1.5 mi. N.E. of Warba**	**47.15015**	**93.25523**	**38**	**0.98**	**0.75**	**1.73**	**5.33**
tributary to Vadnais Lake	9/23/99	6.56	99UM094	Ramsey	upstream of Koehler Road	45.05447	93.08056	10	0.53	0.15	0.68	66.83
trib. to South Fork Crow R.	6/27/00	6.83	00UM047	Kandiyohi	downstream of 60th St. SE	45.08829	94.96299	14	0.23	0.25	0.48	94.70
trib. to Medicine Lake	8/7/00	7.29	00UM068	Hennepin	downstream of 26th Ave. N.	45.00664	93.44457	14	0.43	0.75	1.18	80.32
unnamed ditch	7/12/00	7.77	00UM001	Beltrami	1.5 mi. N of Sorway	47.53950	95.12639	38	0.43	0.42	0.84	42.60
tributary to Bear Creek	6/29/99	7.80	99UM012	Todd	2.0 mi. S.E. of Hewitt	46.30532	95.05601	36	0.25	0.67	0.92	86.75
Briggs Creek	**7/7/00**	**7.87**	**00UM043**	**Sherburne**	**upstream of C.R. 48**	**45.51623**	**93.92420**	**49**	**0.78**	**0.67**	**1.44**	**60.02**
County Ditch # 37	6/27/00	8.03	00UM046	Kandiyohi	6 mi. N.W. of New London	45.36334	95.06058	72	0.63	0.50	1.13	56.90
Island Lake Creek	**6/22/99**	**8.10**	**99UM036**	**Itasca**	**~6.0 mi. N.E. of Deer River**	**47.41504**	**93.72507**	**72**	**0.83**	**0.50**	**1.33**	**5.42**
Diamond Creek	8/14/99	8.48	99UM085	Hennepin	Elm Creek Park Reserve	45.19833	93.47305	10	0.68	0.00	0.68	61.47
County Ditch #13	7/19/99	9.50	99UM020	McLeod	~2.5 mi. of Glencoe	44.81221	94.15025	29	0.23	0.17	0.39	92.67
unnamed ditch	7/28/99	9.70	99UM051	Aitkin	~6 mi. N.E. of Waukenabo	46.81848	93.57098	46	0.63	0.25	0.88	0.99
unnamed ditch	8/4/99	9.70	99UM051	Aitkin	~6 mi. N.E. of Waukenabo	46.81848	93.57098	46	0.63	0.25	0.88	0.99
Battle Creek	6/14/99	10.34	99UM076	Ramsey	upstream of Roth St.	44.94278	93.01361	32	0.68	0.70	1.37	75.35
Little Rock Creek	7/15/99	11.20	99UM058	Morrison	~3 mi. S.W. of Buckman	45.87271	94.14576	26	0.55	0.83	1.38	86.78
Mike Drew Brook	**7/5/00**	**11.40**	**00UM031**	**Mille Lacs**	**5 mi. N of Milaca**	**45.83505**	**93.61943**	**54**	**0.83**	**0.83**	**1.66**	**41.21**
unnamed creek	6/22/99	11.80	99UM041	Aitkin	~ 2.5 mi. S.W. of Jacobson	46.98552	93.32022	22	0.75	0.58	1.33	7.86
Welcome Creek	7/19/00	12.13	00UM004	Itasca	downstream of Hwy 169 in Keewatin	47.39099	93.07008	28	0.43	0.42	0.84	45.81
Skunk River	**6/28/99**	**12.80**	**99UM067**	**Morrison**	**2.0 mi. S.E. of Sulivan**	**46.09823**	**93.89825**	**68**	**0.88**	**0.83**	**1.71**	**33.13**
Skunk River	**7/29/99**	**12.80**	**99UM067**	**Morrison**	**2.0 mi. S.E. of Sulivan**	**46.09823**	**93.89825**	**67**	**0.88**	**0.83**	**1.71**	**33.13**
Bogus Brook	7/1/99	13.10	99UM018	Mille Lacs	3.0 mi. S.E. of Bock	45.73833	93.51109	55	0.73	0.58	1.31	58.45

APPENDIX 5. (continued)

Stream Name	Sample Date	Drainage Area (mi²)	Field Number[1]	County	Location	Latitude[2]	Longitude	IBI[3]	Land Rate[4]	Habitat Rate[5]	Total Rate[6]	Land Use %[7]
					Small Streams (continued)							
Bear Creek	9/5/00	13.89	00UM096	Todd	So. of Hwy 210	46.32430	95.04318	37	0.25	0.33	0.58	88.75
Judicial Ditch # 29	6/28/00	14.22	00UM054	Meeker	2.5 miles S of Cosmos on Hwy 4	44.89855	94.69692	28	0.25	0.42	0.67	99.04
Sand Creek	7/17/00	15.04	00UM065	Anoka	upstream of Olive St.	45.18856	93.28525	61	0.55	0.33	0.88	78.41
County Ditch # 4	7/26/00	15.19	00UM050	Renville	downstream of 490th St.	44.81192	94.69040	51	0.25	0.83	1.08	98.73
Arvig Creek	7/28/99	15.90	99UM042	Cass	~2 mi. S.E. of Pine River	46.70580	94.36327	71	0.70	0.42	1.12	28.40
County Ditch # Twelve	**6/26/00**	**16.29**	**00UM035**	**Stearns**	**upstream of C.R. 4**	**45.62373**	**94.30029**	**87**	**0.68**	**0.75**	**1.43**	**60.24**
unnamed ditch	8/4/99	16.40	99UM030	Aitkin	~1.5 mi. N.W. of Tamarack	46.65208	93.15937	54	0.70	0.33	1.03	25.69
unnamed ditch	6/28/99	17.20	99UM035	Aitkin	~1.5 mi. N. of Pine Knoll	46.59786	93.76498	10	0.63	0.17	0.79	14.41
unnamed ditch	7/28/99	17.20	99UM035	Aitkin	~1.5 mi. N. of Pine Knoll	46.59786	93.76498	29	0.63	0.17	0.79	14.41
trib. to Mississippi River	9/22/99	17.64	99UM092	Washington	Cottage Grove Ravine Regional Park	44.80125	92.90199	10	0.60	0.15	0.75	85.36
Hoboken Creek	6/27/00	17.97	00UM037	Stearns	S. of Hwy 28	45.71507	95.00683	26	0.25	0.58	0.83	98.49
Eagle Creek	6/28/00	20.53	00UM058	Meeker	2 mi. NE Kingston	45.20627	94.29387	59	0.50	0.58	1.08	50.47
County Ditch # 6	8/24/00	23.31	00UM073	Pope	11 mi. W. of Sauk Centre	45.70177	95.18167	39	0.35	0.33	0.68	91.05
Daggett Brook	**7/5/00**	**24.24**	**00UM016**	**Crow Wing**	**12 mi. SW of Garrison**	**46.19203**	**94.04243**	**71**	**0.90**	**0.67**	**1.57**	**29.70**
Hay Creek	**6/22/99**	**24.50**	**99UM061**	**Itasca**	**E. of Swan Lake, 0.2 mi. E. of Hwy 12**	**47.28507**	**93.14539**	**70**	**0.80**	**0.83**	**1.63**	**31.39**
West Savanna River	**8/2/00**	**25.29**	**00UM021**	**Aitkin**	**@ Savanna Portage State Park**	**46.82736**	**93.18047**	**89**	**1.00**	**0.92**	**1.92**	**1.56**
South Two River	7/7/99	26.80	99UM044	Stearns	~3 mi. N.E. of Albany	45.64559	94.51517	59	0.50	0.39	0.89	86.46
tributary to N Fork Crow R.	7/1/99	27.10	99UM055	Meeker	~ 5 mi. N. of Litchfield	45.20048	94.52947	44	0.53	0.75	1.28	88.10
Hardwood Creek	9/28/99	27.21	99UM103	Anoka	upstream of CR 21	45.20075	93.03952	38	0.63	0.35	0.97	59.49
Shingle Creek	8/7/00	27.41	00UM069	Hennepin	upstream of Queen Ave. bridge	45.05065	93.31174	49	0.50	0.50	1.00	82.34
Jewett Creek	9/6/00	32.33	00UM097	Meeker	1.5 mi. N.E. of Litchfield	45.16097	94.50340	37	0.38	0.83	1.21	82.56
Battle Brook	7/9/99	32.60	99UM028	Sherburne	~4.0 mi. No. of Zimmerman	45.50139	93.61548	51	0.68	0.28	0.96	67.82
Kettle Creek	**7/11/00**	**33.42**	**00UM009**	**Becker**	**upstream of C.R. 119**	**46.76514**	**95.20550**	**79**	**0.80**	**0.83**	**1.63**	**52.15**
Spunk Creek	6/26/00	34.48	00UM040	Stearns	4 mi. N of Avon	45.63873	94.43555	52	0.75	0.67	1.42	58.20

APPENDIX 5. (continued)

Stream Name	Sample Date	Drainage Area (mi^2)	Field Number[1]	County	Location	Latitude[2]	Longitude	IBI[3]	Land Rate[4]	Habitat Rate[5]	Total Rate[6]	Land Use %[7]
					Moderate size streams (35 to 200 mi^2 drainage area)							
Moran Creek	**7/24/00**	**35.70**	**00UM077**	**Todd**	**5 mi S.W. of Staples**	**46.28296**	**94.85652**	**80**	**0.68**	**0.92**	**1.59**	**56.32**
Home Brook	**7/14/99**	**38.50**	**99UM027**	**Cass**	**4.0 mi. SW of Lake Shore**	**46.46906**	**94.40754**	**46**	**0.88**	**1.00**	**1.88**	**22.73**
Clearwater Creek	8/17/00	39.31	00UM084	Anoka	in Centerville	45.16425	93.05321	36	0.43	0.33	0.76	51.37
Turtle Creek	7/24/00	40.01	00UM078	Todd	3 mi E. of Browerville	46.07755	94.80560	ns	0.55	0.83	1.38	70.93
Hillman Creek	**7/15/99**	**40.10**	**99UM023**	**Morrison**	**1.0 mi. W. of Center Valley**	**45.97072**	**94.00387**	**63**	**0.80**	**0.75**	**1.55**	**40.21**
Mosquito Creek	**7/6/00**	**40.11**	**00UM013**	**Cass**	**3.5 mi. N of Motley**	**46.39992**	**94.62869**	**48**	**0.80**	**0.92**	**1.72**	**26.39**
Day Brook	**7/19/00**	**41.15**	**00UM006**	**Itasca**	**14 miles N of Nashwauk**	**47.56683**	**93.19076**	**67**	**0.90**	**1.00**	**1.90**	**23.81**
Birch Creek	**7/12/00**	**43.24**	**00UM011**	**Hubbard**	**on C.R. 4 in Yola**	**47.23312**	**95.01148**	**43**	**0.98**	**0.83**	**1.81**	**11.41**
Blueberry River	**7/11/00**	**43.43**	**00UM025**	**Wadena**	**upstream of C.R. 16**	**46.78451**	**95.14922**	**63**	**0.78**	**1.00**	**1.78**	**44.94**
Trott Brook	8/17/00	43.90	00UM067	Anoka	upstream of C.R. 5 in Ramsey	45.28201	93.44155	54	0.53	0.83	1.36	61.89
Grove Creek	7/19/99	45.00	99UM045	Meeker	3.0 mi. NE of Grove City	45.19817	94.62801	35	0.33	0.42	0.74	87.74
Getchell Creek	6/27/00	45.07	00UM039	Stearns	@ C.R. 176, 14 miles SW Albany	45.58553	94.72670	66	0.20	0.75	0.95	93.41
Twelvemile Creek	7/7/99	45.70	99UM060	Wright	~3.0 mi. E. of Howard Lake	45.06223	94.01806	53	0.40	0.92	1.32	84.53
Mayhew Creek	7/7/00	46.53	00UM042	Benton	5 miles E of Sauk Rapids	45.61270	94.10610	10	0.35	0.67	1.02	86.21
Mayhew Creek	8/2/00	46.53	00UM042	Benton	5 miles E of Sauk Rapids	45.61270	94.10610	45	0.35	0.67	1.02	86.21
Bradbury Brook	**7/5/00**	**47.92**	**00UM033**	**Mille Lacs**	**5 mi. S of Onamia**	**45.99742**	**93.66522**	**74**	**0.90**	**0.67**	**1.57**	**10.88**
South Fork Watab River	6/26/00	49.07	00UM041	Stearns	2 mi. N of St. Joseph	45.58987	94.31866	51	0.65	0.75	1.40	59.10
Wing River	**7/10/00**	**49.72**	**00UM023**	**Otter Tail**	**upstream of C.R. 42**	**46.22554**	**95.21056**	**75**	**0.75**	**0.92**	**1.67**	**61.32**
Coon Creek	7/17/00	50.21	00UM059	Anoka	downstream of Hwy 65	45.23314	93.23592	58	0.48	0.17	0.64	37.76
Coon Creek	8/21/00	50.21	00UM059	Anoka	downstream of Hwy 65	45.23314	93.23592	54	0.48	0.17	0.64	37.76
Rush Creek	7/16/99	50.71	99UM081	Hennepin	Elm Creek Park Reserve Group Camp	45.14861	93.45917	35	0.58	0.96	1.53	71.86
Farnham Creek	7/14/99	53.70	99UM022	Wadena	@C.R. 30, ~10.0 mi. N. of Staples	46.50712	94.79366	44	0.65	0.67	1.32	22.50
Crooked Lake Ditch	8/24/00	55.61	00UM072	Douglas	4 mi. N. of Osakis	45.92931	95.13734	51	0.38	0.50	0.88	86.91
Buffalo Creek	7/25/00	55.93	00UM049	Renville	upstream of 440th St.	44.78795	94.79429	33	0.25	0.25	0.50	98.73
Eagle Creek	7/11/00	59.31	00UM075	Todd	in Browerville on Cr 89	46.11954	94.91873	41	0.55	0.67	1.22	80.29
Little Pine River	**7/6/00**	**80.71**	**00UM017**	**Crow Wing**	**7 mi. S of Emily**	**46.65651**	**93.97946**	**89**	**0.95**	**1.00**	**1.95**	**4.00**
Third River	**7/12/00**	**81.86**	**00UM007**	**Itasca**	**upstream of F.R. 2171**	**47.54456**	**94.26144**	**61**	**0.98**	**0.83**	**1.81**	**4.18**
Elm Creek	8/6/99	86.16	99UM082	Hennepin	upstream of USGS gauge	45.16222	93.43555	53	0.65	0.46	1.11	68.32

APPENDIX 5. (continued)

Stream Name	Sample Date	Drainage Area (mi²)	Field Number[1]	County	Location	Latitude[2]	Longitude	IBI[3]	Land Rate[4]	Habitat Rate[5]	Total Rate[6]	Land Use %[7]
					Moderate size streams (continued)							
Elm Creek	8/18/00	86.17	00UM085	Hennepin	upstream of bridge on Elm Creek Road	45.16235	93.43614	56	0.65	0.75	1.40	68.31
Judicial Ditch # 15	7/26/00	99.20	00UM051	Renville	downstream of 550th St.	44.76638	94.55767	37	0.13	0.17	0.29	98.54
Coon Creek	7/17/00	103.98	00UM064	Anoka	in Erlanson Nature Center	45.17204	93.30096	62	0.55	0.83	1.38	54.12
N. Fork Crow River	7/15/99	112.10	99UM050	Stearns	~4.4 mi. N. of Belgrade	45.51745	95.00312	56	0.53	0.67	1.19	89.23
Little Elk River	**6/29/99**	**127.90**	**99UM003**	**Morrison**	**1.0 mi. NE of Randall**	**46.08582**	**94.48820**	**73**	**0.73**	**0.92**	**1.64**	**48.83**
Mississippi River	**8/1/00**	**130.21**	**00UM010**	**Hubbard**	**Stumphges Access**	**47.39754**	**95.14623**	**81**	**0.93**	**0.92**	**1.84**	**5.61**
SchoolCraft River	**6/23/99**	**130.30**	**99UM026**	**Hubbard**	**5.5 mi. SE of Becida**	**47.31309**	**94.94706**	**79**	**0.88**	**0.92**	**1.79**	**8.78**
Rice Creek	8/17/00	151.65	00UM083	Ramsey	upstream C.R. 10 @ Moundsview	45.09450	93.18966	61	0.45	0.58	1.03	53.64
Rice Creek	10/8/99	159.45	99UM105	Ramsey	upstream of CR "I"	45.11003	93.18570	42	0.53	0.43	0.96	51.16
Platte River	7/6/00	165.12	00UM030	Morrison	along C.R. 255, 3 mi. W of Pierz	45.97299	94.17345	91	0.70	0.67	1.37	56.78
Rice Creek	10/11/99	167.59	99UM107	Ramsey	downstream of old Hwy 8	45.09192	93.20202	77	0.65	0.61	1.26	52.73
Prairie River	**8/22/00**	**174.40**	**00UM020**	**Aitkin**	**@ Balsam Town Hall off C.R. 64**	**46.77539**	**93.15551**	**74**	**0.90**	**0.41**	**1.31**	**13.14**
Rice Creek	8/3/00	179.93	00UM060	Anoka	downstream of University Ave.	45.09004	93.26711	59	0.58	0.58	1.16	58.25
Rice Creek	10/12/99	194.43	99UM108	Anoka	Locke County Park in Fridley	45.09547	93.25513	63	0.60	0.83	1.43	56.87

APPENDIX 5. (continued)

Stream Name	Sample Date	Drainage Area (mi²)	Field Number[1]	County	Location	Latitude[2]	Longitude	IBI[3]	Land Rate[4]	Habitat Rate[5]	Total Rate[6]	Land Use %[7]
					Rivers (> 200 mi² drainage area)							
South Fork Crow River	6/29/00	206.67	00UM048	Kandiyohi	along 210th Ave. SE	44.92114	94.80447	16	0.20	0.33	0.53	87.13
South Fork Crow River	7/25/00	206.67	00UM048	Kandiyohi	along 210th Ave. SE	44.92114	94.80447	22	0.20	0.33	0.53	87.13
Fish Hook River	8/24/99	219.10	99UM031	Hubbard	At Park Rapids DNR office	46.91932	95.05259	78	0.80	0.70	1.50	17.88
Long Prairie River	7/20/00	232.61	00UM076	Douglas	1/2 mile east of Carlos	45.98158	95.30352	84	0.55	0.75	1.30	59.77
Buffalo Creek	7/25/00	233.76	00UM052	Renville	2 miles N of Stewart on 580th St.	44.74244	94.50008	34	0.25	0.25	0.50	97.34
Long Prairie River	8/23/00	236.79	00UM089	Douglas	near Carlos @ WWTP	45.99029	95.29223	57	0.63	0.52	1.15	60.21
South Fork Crow River	6/29/00	241.98	00UM053	Meeker	at Hwy 7 in Cosmos	44.93334	94.67450	22	0.13	0.08	0.21	88.76
Elk River	7/6/99	284.70	99UM038	Sherburne	~ 3.5 mi. N.W. of Big Lake	45.37821	93.76975	71	0.50	0.50	1.00	78.33
Elk River	7/21/99	284.70	99UM038	Sherburne	~ 3.5 mi. N.W. of Big Lake	45.37821	93.76975	84	0.50	0.50	1.00	78.33
Boy River	**7/19/00**	**289.30**	**00UM012**	**Cass**	**9 mi. NW Remer**	**47.07895**	**94.10055**	**79**	**0.90**	**0.92**	**1.82**	**6.42**
North Fork Crow River	6/28/00	326.10	00UM056	Meeker	11.5 miles N of Grove City on Hwy 4	45.27840	94.66102	79	0.30	0.33	0.63	86.71
Long Prairie River	7/8/99	376.50	99UM039	Todd	~ 2.25 mi. E. of Clotho,	46.01292	94.99060	46	0.55	0.63	1.18	66.13
Platte River	8/23/99	401.50	99UM048	Morrison	1.5 mi. N.W. of Vawter	45.92666	94.25674	58	0.65	0.54	1.19	59.01
Long Prairie River	7/24/00	413.64	00UM074	Todd	Long Prairie @ public access	45.97383	94.86837	62	0.48	0.33	0.81	67.24
South Fork Crow River	8/11/99	426.80	99UM070	McLeod	4.0 mi. S.W. of Hutchinson	44.87840	94.45300	26	0.40	0.50	0.90	88.31
Prairie River	**8/23/00**	**429.61**	**00UM003**	**Itasca**	**10 miles NE of Grand Rapids**	**47.36418**	**93.49267**	**82**	**0.95**	**0.43**	**1.38**	**8.07**
Sauk River	6/27/00	442.08	00UM038	Stearns	C.R. 168, in Melrose	45.68155	94.77174	72	0.58	0.50	1.08	82.64
Sauk River	8/16/00	442.08	00UM038	Stearns	C.R. 168, in Melrose	45.68155	94.77174	79	0.58	0.50	1.08	82.64
Long Prairie River	8/23/00	556.38	00UM079	Todd	upstream of CR 62 bridge	46.12211	94.84327	48	0.50	0.61	1.11	70.67
Rum River	**7/5/00**	**569.58**	**00UM032**	**Mille Lacs**	**8 mi. N of Milaca @ C.R. 16 bridge**	**45.86849**	**93.69054**	**61**	**0.80**	**0.92**	**1.72**	**14.48**
Shell River	**8/1/00**	**600.10**	**00UM027**	**Wadena**	**@ Shell City Landing**	**46.79255**	**94.94645**	**78**	**0.80**	**0.50**	**1.30**	**34.77**
Pine River	**10/5/99**	**631.50**	**99UM037**	**Crow Wing**	**2.5 mi N.W. of Mission**	**46.61597**	**94.06820**	**75**	**0.88**	**0.72**	**1.59**	**13.41**
Crow Wing River	**7/31/00**	**938.76**	**00UM026**	**Wadena**	**Upstream of bridge in Nimrod**	**46.64277**	**94.88039**	**80**	**0.80**	**0.83**	**1.63**	**33.54**
S. Fork Crow River	8/16/99	1170.80	99UM010	Carver	2.0 mi. N. of Mayer	44.91912	93.87605	44	0.30	0.17	0.47	91.05
Rum River	9/12/00	1272.74	00UM044	Isanti	downstream of C.R. 5 in Isanti	45.49079	93.26490	69	0.50	0.57	1.07	41.71
Rum River	9/18/00	1272.74	00UM044	Isanti	downstream of C.R. 5 in Isanti	45.49079	93.26490	69	0.50	0.67	1.17	41.71
Rum River	7/27/00	1324.62	00UM066	Anoka	downstream of C.R. 24 in St. Francis	45.38421	93.35741	77	0.53	0.67	1.19	42.25
Rum River	8/28/00	1324.62	00UM066	Anoka	downstream of C.R. 24 in St. Francis	45.38421	93.35741	63	0.53	0.67	1.19	42.25

APPENDIX 5. (continued)

Stream Name	Sample Date	Drainage Area (mi²)	Field Number[1]	County	Location	Latitude[2]	Longitude	IBI[3]	Land Rate[4]	Habitat Rate[5]	Total Rate[6]	Land Use %[7]
					<u>Rivers (continued)</u>							
Crow Wing River	**6/13/00**	**2232.26**	**00UM024**	**Wadena**	**~ 5 mi. NW of Motley**	**46.38150**	**94.72912**	**81**	**0.65**	**1.00**	**1.65**	**48.87**
Crow River	7/26/00	2632.80	00UM080	Wright	downstream of Hwy 55 @ Rockford	45.08879	93.73171	65	0.33	0.58	0.91	84.89
Crow River	8/14/00	2750.96	00UM081	Hennepin	4 mi. S. of Elk River	45.22643	93.55566	50	0.38	0.50	0.88	84.30
Crow Wing River	**8/24/99**	**3198.60**	**99UM062**	**Cass**	**~ 3 mi E.S.E. of Motley**	**46.32580**	**94.58757**	**78**	**0.63**	**0.63**	**1.26**	**52.62**
Mississippi River	8/23/00	3283.19	00UM090	Itasca	At hwy. 169 in Grand Rapids	47.23287	93.52285	82	0.68	0.96	1.63	9.05
Mississippi River	**8/22/00**	**5837.86**	**00UM087**	**Aitkin**	**upstream of CR 1 north of Aitkin**	**46.54223**	**93.70599**	**68**	**0.75**	**0.35**	**1.10**	**11.40**
Mississippi River	**8/22/00**	**6060.31**	**00UM088**	**Crow Wing**	**Hwy. 6 N. of Crosby**	**46.54215**	**93.95911**	**78**	**0.78**	**0.54**	**1.32**	**11.80**
Mississippi River	**8/24/00**	**11729.79**	**00UM091**	**Morrison**	**downstream of Royalton landing**	**45.82337**	**94.35502**	**88**	**0.65**	**0.74**	**1.39**	**25.26**
Mississippi River	8/17/99	13905.90	99UM034	Sherburne	Downstream of Clearwater	45.40453	94.01215	76	0.55	0.74	1.29	34.83
Mississippi River	8/29/00	14031.96	00UM092	Wright	Public Access @ Monticello	45.30466	93.78224	54	0.48	0.63	1.11	35.15
Mississippi River	9/14/00	14031.96	00UM092	Wright	Public Access @ Monticello	45.30466	93.78224	62	0.48	0.72	1.19	35.15
Mississippi River	9/13/00	19040.88	00UM098	Anoka	below 169 bridge in Anoka.	45.18595	93.39033	69	0.33	0.67	1.00	44.35
Mississippi River	8/12/99	35593.30	99UM017	Washington	~1 mi. S.W. of Newport	44.85958	93.00777	71	0.33	0.30	0.63	77.42
Mississippi River	10/1/99	35593.30	99UM017	Washington	~1 mi. S.W. of Newport	44.85958	93.00777	66	0.33			77.42

[1] Field number assigned to each station to designate a unique sampling location.

[2] Latitude and longitude are formatted in WGS 84 decimal degrees.

[3] IBI score is the overall IBI score assigned to the site. Scores range from 0 (lowest biological integrity) to 100 (highest biological integrity). ns = no score due to insufficient sample size.

[4] normalized (maximum value = 1) watershed rating based on GIS coverages for land use, point sources, feedlots, and ditched streams.

[5] normalized (maximum value = 1) habitat score based on the quantitative habitat assessment or QHEI.

[6] sum of watershed and habitat rating.

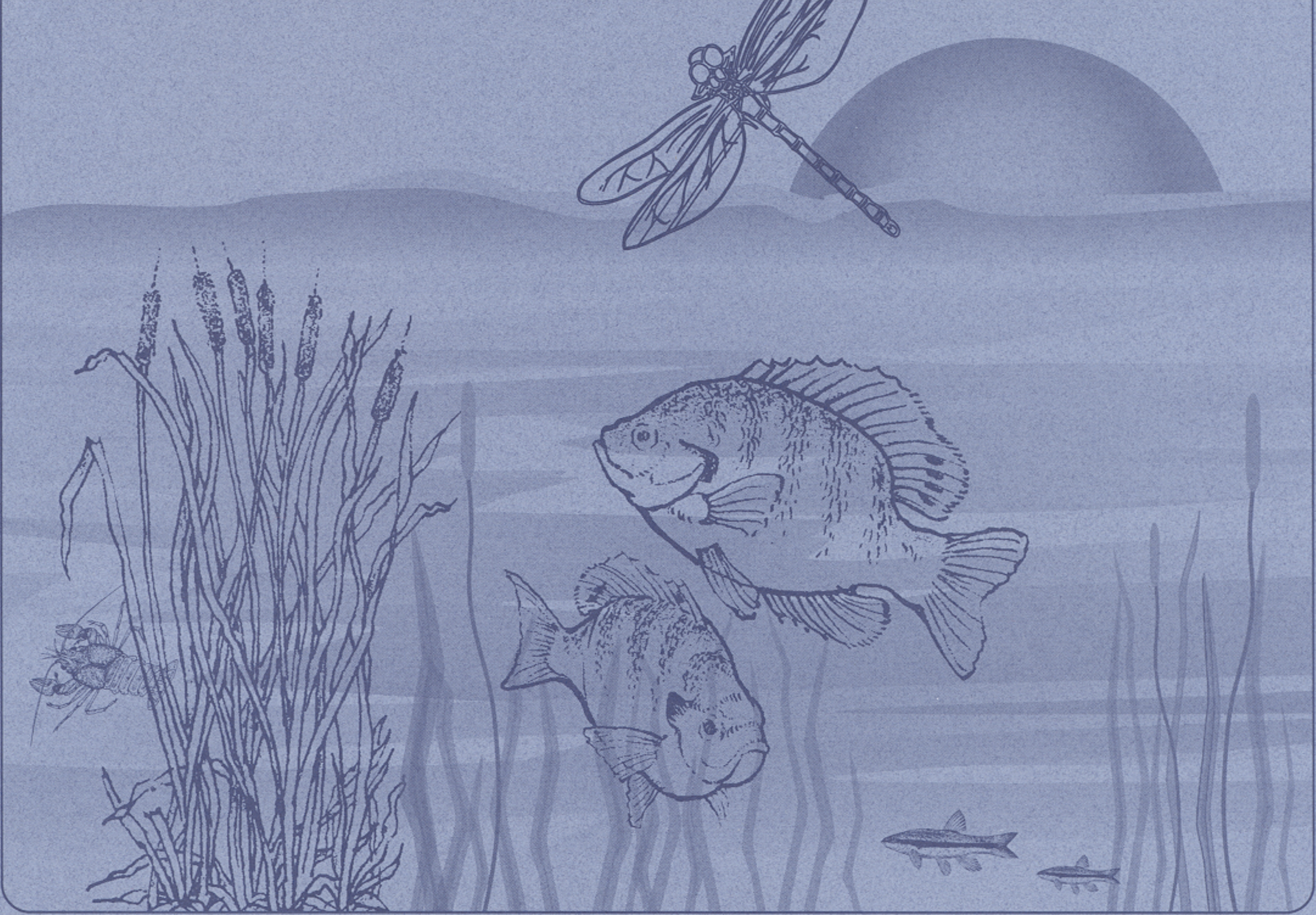
[7] Land use expressed as a percent of the watershed that has been altered by human development. It includes disturbance from agricultural, residential, urban, and mining land uses.

* Sites in bold text were selected as reference sites based on land use, habitat, and biology.

Minnesota Pollution Control Agency

Biological Monitoring Program

Index of Biotic Integrity Guidance for Coolwater Rivers and Streams of the St. Croix River Basin



Index of Biotic Integrity (IBI) Guidance for Coolwater Rivers and Streams of the St. Croix River Basin in Minnesota

By Scott Niemela and Michael Feist

Minnesota Pollution Control Agency
Biological Monitoring Program

St Paul, Minnesota
2000

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Index of Biotic Integrity (IBI) Guidance for Coolwater Rivers and Streams of the St. Croix River Basin in Minnesota

I. BACKGROUND INFORMATION

Rivers and streams serve many functions in today's society by providing a source of food and water, a mode of transportation for many of our crops and material goods, and as a recreational and aesthetically pleasing resource for many people. However, the innumerable functional and aesthetic qualities of rivers and streams create pressures on the resource, which are exacerbated by an ever-increasing human population. Watersheds that were once mainly forested have been altered for the social and economic benefit of today's society, often at the expense of the river's health. The degradation of Minnesota's rivers and streams can be traced to a multitude of sources including: chemical pollutants from municipal and industrial point source discharges; agricultural runoff of pesticides, nutrients, and sediment; hydrologic alteration from stream channelization, dams, and artificial drainage; and habitat alteration from agricultural, urban, and residential encroachment. To ensure the integrity of rivers and streams in Minnesota, we must understand the relationship between these human induced disturbances and their effect on aquatic resources.

For many years we have attempted to manage human impact on rivers and streams by restricting the amount and kinds of chemicals that enter them. Federal and state government agencies have developed and enforced water-quality standards to ensure that chemical concentrations in our streams do not exceed certain limits. But, while we have been largely successful in reducing chemical pollution point sources, in many respects we have failed to recognize the more insidious effects that landscape alteration and non-point pollution have on river and stream quality. Watershed disturbances from urban, residential, and agricultural development contribute to an overall decrease in the biological integrity in many of our rivers and streams (i.e., road building, stream channelization, alteration of the stream's riparian zone, and many others). It is increasingly apparent that monitoring activities cannot focus solely on chemical indicators but must instead focus on indicators that integrate the effects of both physical and chemical stressors. Proper management of river and stream systems must be predicated upon a comprehensive monitoring strategy that is able to detect degradation in rivers and streams due to human disturbance.

In recent years, scientists have developed methods to quantify and interpret the results of biological surveys, allowing water-quality managers and policy makers to make informed decisions concerning rivers and streams. There are many advantages to using aquatic organisms, such as fish, in a water quality monitoring program. Aquatic organisms are responsive to the cumulative affects of both physical and chemical disturbances. They are easily sampled with the proper equipment. They are sensitive to human induced changes over time, and the public recognizes them as being important indicators of a healthy environment (Karr 1981).

THE INDEX OF BIOTIC INTEGRITY (IBI)

At the forefront of this effort has been the development of a multimetric framework for biological data interpretation known as the Index of Biotic Integrity (IBI: Karr 1981). The IBI was first developed in the early

1980's using attributes of fish communities in moderate size wadeable streams of the Midwest. It has subsequently been modified for use throughout the country for a variety of assemblages in all types of aquatic systems (Simon and Lyons 1995). Each metric in the IBI denotes a quantifiable attribute of a biological assemblage that changes in a predictable way with different levels of human influence. Typically, 8-12 metrics are combined to form a single index or IBI. The metrics in a typical fish IBI fall into 3 broad categories: 1) species richness and composition, 2) trophic composition and reproductive function, 3) fish abundance and condition. A well-rounded IBI will include 1 or more metrics from each of these broad categories.

REGIONALIZATION AND STREAM CLASSIFICATION

If the IBI is to detect human induced changes in resource integrity it is necessary to identify and partition the factors that contribute to the natural variability of streams so that changes caused by humans may be detected. On a broad regional scale, differences in climate, topography, geology and other geophysical characteristics of an area dictate species distributions. Thus, an IBI developed for predominantly agricultural areas in the Midwest should not be applied to the mountainous regions of the western U.S. The ecoregion concept (Omernik and Gallant 1988) has been the most common regional framework for developing the IBI. In Minnesota, versions of the IBI have been developed using an ecoregional (Niemela et al. 1999) and basin framework (Bailey et al. 1993).

Rivers and streams in Minnesota are physically, chemically and biologically diverse. They range in size from small headwater streams that are less than 1 meter wide, to large navigable rivers such as the main stem of the Mississippi River. The majority of streams in Minnesota are considered warm or coolwater, but coldwater streams are also present, particularly in the northeastern and southeastern regions of the state. Riffles are an important feature of many higher gradient streams. However, in many of Minnesota's lower gradient streams there are few or no riffles. Within a stream reach, variables such as stream size, gradient, and water temperature influence the type of aquatic assemblage present. An IBI should account for reach level differences as well as regional differences through proper stream classification.

Once a stream classification framework is developed to account for the natural variation in the fish community structure, each metric within the IBI must be selected (based on the metrics response to a gradient of human disturbance) and calibrated (i.e. adjusted) to account for differences in metric expectations between each stream class. For example, calibration of each metric is necessary because we would expect to collect less fish species from a first order stream than from a third order stream. While it is almost always necessary to calibrate the IBI scoring system to account for differences due to stream size, it is also possible that metrics will need to be calibrated to account for stream morphological or ecoregional differences.

IBI VALIDATION

The IBI concept has proven to be very adaptable (Karr and Chu 1999). Many of the same IBI metrics have been used successfully throughout different regions of the country in a variety of stream types (Simon and Lyons 1995). Metrics such as the total number of species or the percent of tolerant individuals within a sample are common to most IBI versions that have been developed for fish assemblages. However, Karr and Chu (1999) emphasize that "no metric should become part of a regional multimetric index before it is thoroughly and systematically tested and its response has been validated across a gradient of human influence." This is particularly true when developing an IBI for a new region or stream type, or when considering a new or unproven metric. The process we used to validate IBI metrics for the St. Croix River Basin IBI is described in appendix 1.

SUCCESSFUL APPLICATION OF THE IBI

Many states have begun to develop multimetric indices for rivers and streams with the ultimate goal of developing biological criteria (narrative expressions or numerical values that describe the reference biological condition) for use within their own water-quality programs (U.S. EPA 1996). The state of Ohio has taken the definitive lead by developing numeric biological criteria and using the information to guide management activities. Ohio EPA uses the information from biological assessments in wastewater permitting, 305(b) assessments, 401 certification process, waste load allocation, and overall basin assessments. Other state programs in which multimetric biological assessments are integrated into water-quality programs include the programs of North Carolina, Florida, and Maine.

Most of the work in IBI development has focused on moderate size wadeable streams. Sampling methods for these streams have been developed that provide reliable and reproducible results. Additionally, aquatic communities within these systems have been extensively studied, particularly fish and macroinvertebrate assemblages. Recent promising applications of the multimetric concept have been developed to assess wetlands (Gernes and Helgen 1999; Helgen and Gernes 1999), large rivers (Simon and Emery 1995; Simon and Sanders 1999), lakes (Jennings et al. 1999; Minns et al. 1994; Whittier 1999, Drake and Pereira 2000), reservoirs (Jennings et al. 1995; McDonough and Hickman 1999), and terrestrial environments (James Karr, personal communication). However, many of these applications are still in the early stages of development.

THE MINNESOTA POLLUTION CONTROL AGENCY'S BIOLOGICAL MONITORING PROGRAM

Efforts at the state level, largely by the Minnesota Pollution Control Agency (MPCA) and Minnesota Department of Natural Resources (MDNR), to develop multimetric indices began in 1990 with the initiation of the Minnesota River Assessment Project (MRAP). A subsequent interagency study conducted during 1994-1995 focused on the Lake Agassiz Plain ecoregion within the Red River of the North Basin. In the mid-1990's the MPCA adopted a monitoring strategy and management framework centered on the idea of managing watersheds. The strategy included a plan to monitor the condition of each basin using a random site selection process (Stevens 1997) to provide a basin-wide assessment of water quality in streams. This monitoring program was supported by long term legislative funding for biological monitoring and biological criteria development.

The goal of the MPCA's biological monitoring program is to develop an IBI for each of Minnesota's nine major river basins with the intent of developing statewide biological criteria in the future. It is paramount to the development of biological criteria in Minnesota that we obtain fish community information statewide. There is currently a paucity of fish community data for coolwater streams in Minnesota, particularly those streams that have little potential to contain game fish. In fact, fish community information had not previously been obtained for many of the small streams sampled during the course of this study.

This report is the result of an effort to develop an IBI for all permanent coolwater rivers and streams within the St. Croix River Basin in Minnesota. The document is intended to provide guidance for those interested in conducting an IBI assessment. Readers interested in the theoretical underpinnings of multimetric indices in general should refer to Karr and Chu (1999).

II. THE ST. CROIX RIVER BASIN

The St. Croix River Basin includes 7650 mi^2 of flat to gently rolling terrain in Minnesota and Wisconsin (fig. 1). Historically, the basin was almost entirely vegetated by a variety of forest types including the Great Lakes pine forest which was typified by vast stands of mature white and red pines (Fago and Hatch 1993). Logging and agricultural land use practices have almost entirely eliminated large pine stands. A diverse mixture of second growth mixed-hardwood forests, open fields, and cropland now dominates the basin (fig. 2). An ecoregional divide running roughly through the center of the basin in an east-west direction separates the Northern Lakes and Forests ecoregion in the north from the North Central Hardwood Forest ecoregion in the south. Today, the mixed forests that are found in the nutrient poor soils of the Northern Lakes and Forests ecoregion provide a contrast to the more agricultural landscape of the North Central Hardwood Forests ecoregion. The amount of forest cover within the entire basin is currently about 44% (fig. 2). However, the majority of the remaining forest is confined to the northern half of the basin. Residential development is a concern, primarily in the southern portion of the basin around the Twin Cities metropolitan area.

RIVERS AND STREAMS OF THE ST. CROIX RIVER BASIN

Rivers and streams within the St. Croix River Basin are arguably some of the most scenic in Minnesota. The federal government recognized the importance of the St. Croix system in 1968 when the Upper St. Croix River (above Taylors Falls) and its main tributary, the Namekagon River, were included as one of eight initial stream reaches in the National Wild and Scenic Rivers System. In 1972 the Lower St. Croix River (from Taylors Falls to its confluence with the Mississippi) was added to the national system (Fago and Hatch 1993).

Headwater streams within the basin often originate from peat lands, resulting in dark, tannic acid stained water. These streams are usually low gradient streams that lack riffles and have a glide/pool type of stream morphology. In addition they are typically sinuous, with fine substrates and have a riparian zone comprised of wetland vegetation. The Snake and Kettle Rivers, the two largest tributaries to the St. Croix River in Minnesota, originate in wetlands. However, as these streams progress towards their confluence with the St. Croix River their morphology changes. Lower reaches of the Snake and Kettle Rivers, like many other larger streams in the St. Croix River Basin, have a riffle/run/pool stream morphology with a variety of substrate types and a wooded riparian zone.

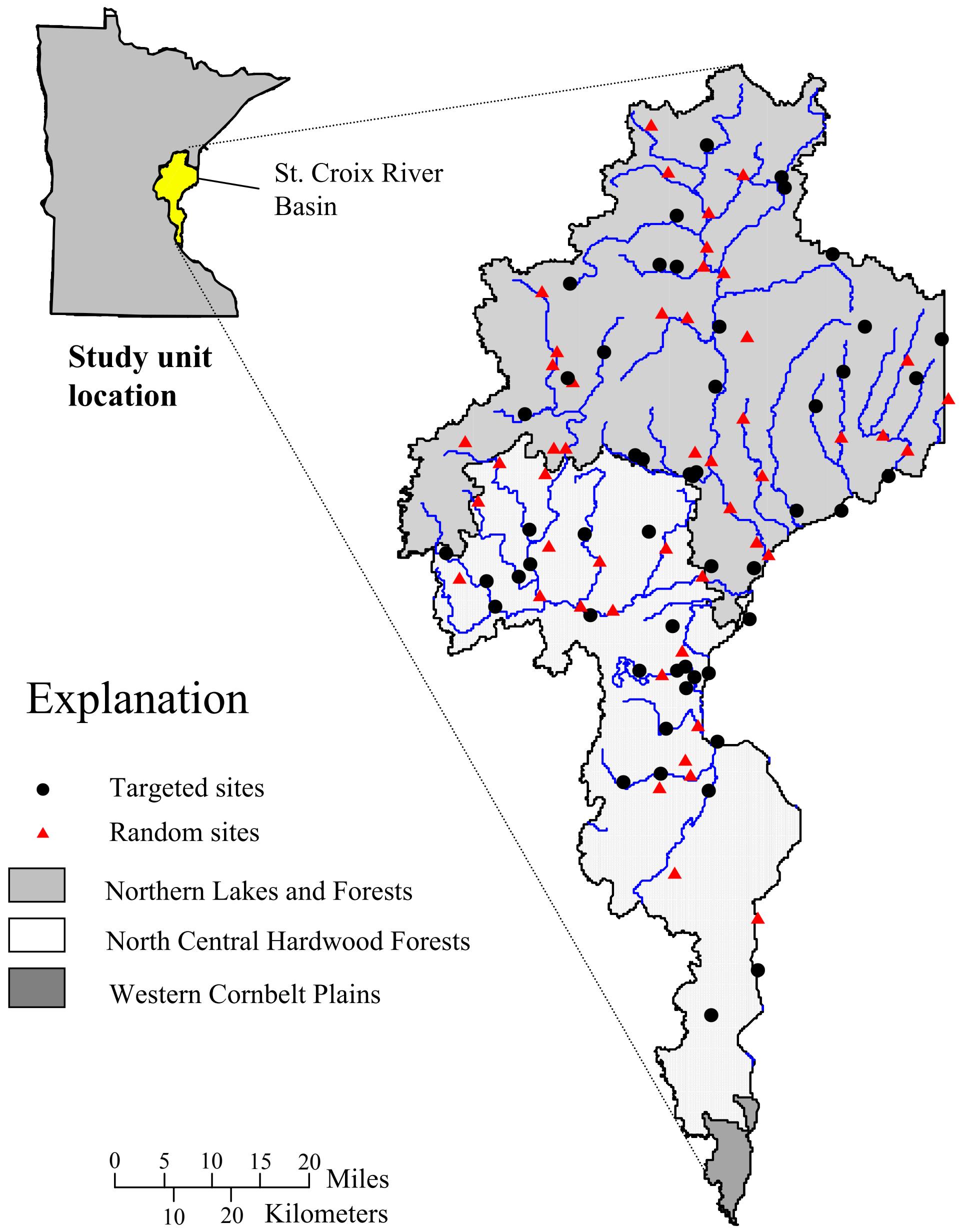

Figure 1. Map of the St. Croix River Basin in Minnesota with major rivers, ecoregional boundaries and the location of each site used to develop the St. Croix River Basin, Index of Biological Integrity (IBI).

THE FISH ASSEMBLAGE

The St. Croix River Basin supports a diverse fish assemblage. Fago and Hatch (1993) list 110 species of fish representing 24 families occurring in the St. Croix River Basin (appendix 3). A dam at St. Croix Falls has been a barrier to fish migration for over 80 years. One hundred and three fish species have been reported from the lower portion of the basin below the falls, compared to 84 above St. Croix Falls dam (Fago and Hatch 1993). Fago and Hatch (1993) list 7 species that have not been collected within the basin since 1974.

Minnesota does not currently list any of the fish species in the St. Croix River Basin as endangered. However, the paddlefish (*Polydon spathula*) is considered threatened and 9 other species known to occur within the basin are considered special concern (appendix 3).

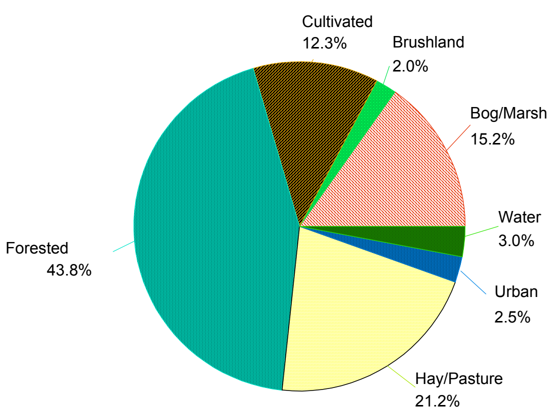

Figure 2. Land use percentages within the St. Croix River Basin of Minnesota.

III. IBI SAMPLING METHODS

The MPCA sampling procedures for wadeable streams are modeled after Wisconsin's warmwater stream guidance (Lyons 1992a). The MPCA sampling procedures for large unwadeable reaches follow U. S. Geological Survey (USGS) National Water-Quality Assessment (NAWQA) guidance (Meador et al, 1993b).

WHEN TO SAMPLE

Sampling should be conducted during daylight hours from mid June through September. All measurements should be taken during base-flow conditions since flood or drought events can have a profound effect on fish community structure and sampling efficiency. Also, an effort should be made to avoid sampling immediately following unusually high or low-flow periods.

REACH LENGTH DETERMINATION

It is vitally important to sample the appropriate reach length. A reach length that is too short may result in an inadequate sample size, and some uncommon species could be missed. Over-sampling a stream reach adds little to the interpretive capability of the data and places strains on limited monitoring budgets. Lyons (1992b) determined that an adequate reach length for Wisconsin wadeable streams is 35 times the mean stream width. This reach length is usually sufficient to obtain a representative sample of the fish community and include the major macrohabitat types (Lyons, 1992b). The mean stream width should be determined prior to sampling by taking a minimum of 10 measurements of the wetted stream width (Simonson et al. 1994). These measurements should be taken across the channel in all of the major macrohabitat types found within the reach.

ELECTROFISHING TECHNIQUE

Four types of electrofishing gear are used to collect fish community information. Selection of electrofishing gear is dependent on stream size and type. However, there are a few procedures that are common to all of the gear types: 1) It is important to net all fish that have been stunned by the electrofishing unit regardless of the species or size of the fish. 2) Sample all available habitat types in the proportion that they occur within the site. 3) Proper use of electrofishing gear requires extreme care and strict adherence to all recommended safety precautions. 4) The amount of time fished and the control box settings should be recorded for each run. 5) Water in the holding bucket or tank should be changed frequently to avoid stressing the fish.

Backpack electrofisher: This gear type is used in small, (usually <8 m wide) wadeable streams. Sampling proceeds in an upstream direction with one person carrying the electrofishing gear and collection bucket and the other person netting. In very small (<3m wide) streams it is possible to sample virtually all of the available habitats but in larger streams (>3m wide) it is often necessary to weave back and forth between habitat types.

Stream electrofisher: This gear type is used in larger, (usually >8m wide) wadeable streams and rivers. The stream electrofisher is a sport canoe rigged for electrofishing with a generator, a control box to regulate the electrical output, and two anodes. Sampling proceeds in an upstream direction with a crew of five. Two members of the crew hold the anodes; each accompanied by a netter. The fifth person pulls the sport canoe upstream, monitors the control box, and ensures team safety. It is usually necessary in these larger streams to weave back and forth between habitat types.

Mini-boom electrofisher: The miniboom electrofishing unit is used in small or hard to access unwadeable streams and rivers. This unit is a small jon-boat rigged for electrofishing with a generator, control box, and a single anode. One person drives the boat, monitors the control box, and ensures the safety of the single netter on the bow. Sampling proceeds downstream by weaving back and forth into different habitat types.

Boom Electrofisher: This gear is used in large, accessible rivers. The boom electrofisher is fished in a downstream direction in three separate runs; one run along each shoreline and a mid-channel run weaving across the stream channel. One person drives the boat, monitors the control box and ensures the safety of the two netters on the bow.

FISH PROCESSING

Fish are usually processed after the entire site has been sampled or after each run when using the boom electrofisher. In some cases, particularly in larger streams, it may be necessary to temporarily stop electrofishing activities during the run and process the larger fish to minimize fish mortality. Data from separate runs should be pooled to yield one data set for the entire site.

All fish are sorted into separate containers by species and enumerated. A minimum and maximum length and batch weight is recorded for each species. Juvenile fish less than 25 mm are not included in the catch. Any deformities, eroded fins, lesions, or tumors (DELT anomalies) should be noted. Two specimens of each species from each site should be retained for later verification by an expert ichthyologist. All other fish should be released back into the stream.

IV. THE METRICS

We classified fish into metric groups by reviewing Wisconsin (Lyons 1992a) and Ohio EPA classifications as well as numerous ichthyological texts and papers

(Balon 1975; Becker 1983; Etnier and Starnes 1993; Pflieger 1975). A list of the metric classifications for each fish species found in the St. Croix River Basin is provided in appendix 3.

Most of the metrics in the IBI were selected because they demonstrated a response to a gradient of human disturbance (see appendix 1). A few metrics did not demonstrate a response to human disturbance but were included in the final IBI because of their importance in detecting change in the fish community at the most severe levels of degradation (e.g. the proportion of fish with deformities, eroded fins, lesions or tumors).

For rivers (drainage area > 270 mi^2) there was not a sufficient gradient of human disturbance within the basin to elicit a response. Therefore, the IBI for rivers should be considered tentative (see appendix 2). The rationale for each metric used in the St. Croix River Basin IBI is described below.

SPECIES RICHNESS AND COMPOSITION

Total number of species: The species richness metric is common to almost every IBI developed in streams throughout the country. For coolwater streams, species richness declines as environmental degradation increases (Leonard and Orth 1986). Hybrids, subspecies and exotics are not included in this metric.

Number of darter species: Darters are commonly found in riffle habitats throughout the St. Croix River Basin. Many darters are considered sensitive to water quality degradation (appendix 3). Because darters require clean coarse substrate materials in order to thrive, they tend to disappear in streams that have been affected by siltation or channelization.

Number of minnow species: Minnows are an important and diverse component of aquatic communities in the St. Croix River Basin. Many minnow species are considered sensitive to water-quality degradation (appendix 3). In general they are found in slack water habitats. Therefore, accumulating silts and toxins pose a direct threat to their ecological sustainability. Minnow species classified as tolerant are not included in this metric.

Number of headwater species: Flow rates and other physical and chemical parameters of headwater streams can change dramatically in a short time period. However, many headwater systems in the St. Croix River Basin retain some permanence of fish habitat in all but the most severe climatic situations, particularly those headwater streams that have retained their connection to wetlands. Certain human disturbances (e.g., watershed urbanization, and channelization) have the affect of exacerbating the fluctuations and reducing the amount of available habitat. Certain species of fish have evolved adaptive strategies in response to naturally occurring fluctuations in headwater streams but may be unable to compensate for higher levels of disturbance caused by humans. Species such as the northern redbelly dace (*Phoxinus eos*) and finescale dace (*Phoxinus neogaeus*) are commonly found in headwater streams that still retain their connection to wetlands. Species classified as tolerant were not included in this metric.

Number of intolerant species: Intolerant species are those that are sensitive to environmental degradation. They are often the first species to disappear following a disturbance or whose distribution has diminished as human influence has increased. Therefore, their presence in a stream is an indication of a high quality resource.

Percent individuals that are tolerant species: Tolerant species are known to persist in poor quality streams. They may become a dominant component of the fish community in streams that have been physically altered by channelization, siltation, or hydrologic modification. Tolerant species may also dominate in chemically altered streams with chronically low dissolved oxygen levels, high levels of ammonia, other toxic substances, or high turbidity (Lyons 1992a).

Percent of the dominant two species: In many degraded stream systems one or two species will tend to dominate the community while other species decline. Those species with the capacity to capitalize on a physical or chemical change in their environment are usually tolerant species. This metric compliments the tolerant species metric by providing a measure of the degree in which two species dominate a particular environment. Percent dominance increases with a higher level of human disturbance.

TROPHIC COMPOSITION AND REPRODUCTIVE FUNCTION

Number of invertivore species: Invertivores are specialized feeders dependent upon a stable invertebrate food base. Disruptions in this food base through human disturbance leads to a decrease in the number of invertivore species. Species classified as tolerant were not included in this metric.

Number of benthic invertivore species: Darters, suckers, madtoms and some minnows are benthic invertivores. Benthic invertivore species rely on undisturbed benthic habitats to feed and reproduce. Many benthic invertivores require clean course substrates and an ample supply of aquatic macrophytes or woody debris for cover. Degradation of benthic habitats (e.g. channelization, siltation) will cause benthic invertivore species to decline. Species classified as tolerant were not included in this metric.

Number of omnivore species: Omnivorous fish species are those that have the physiological ability (usually indicated by the presence of a long coiled gut and dark peritonium) to digest both plants and animals (Karr et al. 1986). Their dominance within a fish community indicates an unstable food base. The ability to utilize multiple food sources allows the omnivore species to switch to another food source when one type of food is disrupted.

Percent individuals that are piscivores: In moderate size streams and rivers (>54 mi^2 drainage area) within the St. Croix River Basin, the occurrence of a viable piscivore population indicates a healthy, trophically diverse fish community. This metric was not used in small streams because piscivores usually make up an insignificant component of the fish community in these streams.

Percent of individuals that are simple lithophilic spawners: Simple lithophilic spawners broadcast eggs over clean gravel substrates (Balon 1975). The metric is inversely correlated with habitat degradation due to excessive siltation (Berkman and Rabeni 1987).

FISH ABUNDANCE AND CONDITION

The number of fish per meter of stream sampled: This metric has been used to identify streams in which severe degradation has substantially reduced fish numbers. Lyons (1992a) included this metric as a correction factor even though he did not find a strong relationship between fish abundance and an environmental quality measure. We calibrated the metric so that only very low fish counts (< 11 fish per 100 meters of stream) would produce a poor metric score. Species classified as tolerant are not included in this metric.

Percent of individuals with Deformities, Eroded fins, Lesions, or Tumors (DELT): Like the number of fish per meter metric, the percent of individuals with DELT anomalies metric has been used to identify sites that have been severely degraded. In other parts of the Midwest DELT anomalies have been associated with environmental degradation primarily due to industrial pollutants (Sanders et al. 1999, Ohio EPA 1988). DELT anomalies were not prevalent in fish from the St. Croix River Basin. However, we feel it is important to retain the metric to identify streams that are severely degraded. Parasitic infestations are not included in this metric because parasitic burden does not necessarily correlate with environmental quality (Steedman 1991).

V. SPECIAL CONSIDERATIONS

LOW CATCH RATES

If the total number of individuals at a site is extremely low a few individuals can have a relatively large influence on the overall IBI score. In this case the IBI score may not be a true reflection of environmental quality. In our judgement, an IBI score should not be calculated for sites with less than 25 individuals. Rather, these sites should be rated as very poor since extremely low catch rates are almost always an indication of serious impairment in permanent, coolwater Minnesota streams.

INTERMITTENT STREAMS

This IBI is intended for use in permanent coolwater streams throughout the St. Croix River Basin. Headwater streams with drainage areas <20 mi^2 pose a particular problem because of the need to distinguish between permanent and intermittent streams. Unless we have additional information suggesting otherwise, our approach has been to consider the stream permanent if it does not go dry during the year that we take the sample. We conduct an evaluation of the site at three different times during the season. An initial site reconnaissance is conducted during the spring. Fish, water chemistry, and habitat data are collected during a second visit in the summer, and a macroinvertebrate sample is obtained during a third visit in the fall. The IBI should not be applied if, during any of these site visits the stream is dry, or if any other information suggests that the stream is intermittent.

COLDWATER STREAMS

This IBI should not be used in coldwater streams. Structural and functional attributes of fish communities in coldwater streams differ significantly from warm or coolwater systems. Thus, IBI's developed for coldwater systems bear little resemblance to their warm or coolwater counterparts. For those interested in applying an IBI to Minnesota coldwater streams, Mundahl and Simon (1999) have developed an IBI for coldwater streams in the upper mid-western United States. Also, Lyons (1996) has developed an IBI for Wisconsin coldwater streams that may be applicable in Minnesota.

NATURAL BARRIERS TO FISH MIGRATION

Barriers to fish migration may have an effect on all streams; however, their effect on fish communities in headwater streams may be the most pronounced. Whereas larger streams usually offer some refuge during periods of stress (i.e., floods and droughts), there is an increased probability in headwater streams for the entire fish community to be extirpated. Therefore, the IBI may underrate headwater streams above fish barriers even though they are otherwise undisturbed. We recommend that the researcher use caution in applying the IBI if natural barriers to fish movement exist. A survey above and below the barrier may be useful in determining the effect of the barrier on the fish community.

STREAM MOUTHS

At the confluence of two streams, fish community structure may be influenced by both stream systems. In such cases the smaller stream may have some fish community characteristics of the larger stream into which it flows. For this reason sampling near the mouths of streams should be avoided, particularly at the confluence of a much larger stream.

VI. CALCULATION AND INTERPRETATION OF THE IBI SCORE

A separate IBI has been developed for 4 stream size classes. The drainage area of the watershed (mi^2) upstream of the site was used as a measure of stream size. The size classes are: very small streams (<20 mi^2), small streams, (20-54 mi^2), moderate streams (55-270 mi^2), and rivers (>270 mi^2). The size classes were chosen to minimize differences in maximum species richness within each size class (appendix 1).

To calculate the watershed area of the sampling sites, we used the Minnesota Planning Land Management Information Center's (LIMC) Upstream program. The MDNR minor watershed containing the site was picked from MDNR's 1995 minor watershed file (bas95ne3) using the latitude and longitude of the site. The MDNR minor watershed boundaries are nearly equivalent to the 14-digit hydrologic unit code (HUC) developed by the U.S. Geological Survey. Upstream additions were confirmed using the MDNR's 24K streams file (dnrstln3).

It may be necessary (particularly in very small streams) to edit the minor watershed containing the site so that the portion of the minor watershed downstream of the site is not included in the drainage area calculation. We edited the minor watershed containing the site using Geographic Information System (GIS), Arcview coverages. However, in most cases an estimate of the minor watershed area upstream of the site may be determined using U.S. Geological Survey (USGS) standard series, 1:24,000 topographical maps. The following methods were used in order of preference to edit the minor watershed containing the site:

a) using Arcview to delineate the drainage area with digital elevation models (DEM).
b) following the contour lines on digital raster graphics (DRG) from U.S. Geological Survey (USGS) standard series topographic maps.
c) or personal experience of watershed boundaries from visiting the site.

The biological integrity of the site is determined by summing the metric scores for the appropriate stream size class. Each metric in the IBI represents a unique and important aspect of the fish community. A low metric score indicates that the fish community attribute deviates substantially from a minimally disturbed site. Conversely, a high metric score indicates that the fish community attribute approximates that of a minimally disturbed site. Many of the same metrics are used in each IBI. However, a few metrics are unique to a single size class. For very small streams refer to table 1, for small streams refer to table 2, for moderate size streams, and rivers in the Northern Lakes and Forests ecoregion refer to table 3, and for rivers in the North Central Hardwood Forests ecoregion refer to table 8.

Scores of 0, 2, 5, 7 or 10 have been assigned for each metric (appendix 4). Once the metric scores have been obtained from the appropriate table (table 1, 2, 3 or 8) they are added to produce a total IBI score ranging from 0 (lowest biological integrity) to 100 (highest biological integrity). A correction factor of 1.11 must be applied if the drainage area of the stream is less than 55 mi^2 because only 9 metrics (instead of 10) are used to calculate an IBI score for the two smallest stream size classes. Narrative descriptions that describe characteristics of the fish community within certain IBI scoring ranges should be used as a guideline for interpreting the IBI score (Lyons 1992a) (table 4). A list of the sampling sites and the IBI score for each site is provided in appendix 5.

Three factors; sampling error, natural variability, and human disturbance, may contribute to the variability of IBI scores. All users of this IBI must attempt to limit the first two sources of variation to detect the third. Sampling error results from a failure to accurately or precisely characterize the fish community (Lyons 1992a). Natural variability occurs because of climatic fluctuations, biological interactions, or any other factor that cannot be attributed to human disturbance (Lyons 1992a). Proper study design and rigorous adherence to sampling protocol can limit the effects of sampling error and natural variation on the IBI score.

The IBI methodology described in this report will allow the user to detect changes in environmental condition due to human disturbance with a reasonable level of certainty. A 10 point difference in streams (<270 mi^2 drainage area) represents a real difference in biological integrity that can be attributed to a change in the level of disturbance (appendix 6). IBI scores for rivers (>270 mi^2 drainage area) are more variable. A difference of 30 IBI points represents a significant change in biological integrity for rivers.

ACKNOWLEDGEMENTS

This report could not have been completed were it not for the inspired efforts of many dedicated professionals. Konrad Schmidt provided a great deal of field help as well as serving as an invaluable teacher in the identification of Minnesota fishes. The quality of this data is due, in large part, to Konrad's thorough knowledge of Minnesota's streams and the fish species that live in them. Thomas Simon was instrumental in getting this research off the ground by sharing his ideas on IBI development and working to obtain partial funding through USEPA. James Karr's advice on IBI development and data analysis has led to a much better final product. His invaluable guidance greatly influenced the development of this IBI. Each summer a new group of student interns and numerous MPCA employees made it possible to accomplish the fieldwork including: Erin McMahon, Lynn Bergquist, Tammy Kelly, Russ Swanson, Leah Class, Josh Fye, Lane Urtel, Regina Cesario, Dori Chretin, Doug Peterson, Marney Milsten, Chandra Carter, Chris Scharenbroich, Nicole Trifilette, Sandy Bissonnette, and Carolyn Voelkers, In addition to assisting in the field, Louise Hotka, Robert Murzyn , and David Christopherson provided help in the project design, database management, and data analysis phases of the work. We thank the Minnesota Department of Natural Resources, particularly Jack Enblom and David Wright and Phil Talmage for their guidance and assistance throughout this work. The collaboration between MPCA and MDNR has contributed greatly to the success of the stream biological program. Finally, we thank the reviewers, Robert Goldstein, Don Pereira, Melissa Drake, Tom Simon, and Phil Talmage for providing helpful insight and suggestions.

Table 1. Scoring criteria for the nine metrics used to calculate the IBI for very small streams (< 20 mi^2 drainage area) in the St. Croix River Basin of Minnesota[1].

	A. Scoring Criteria				
Metric	**10**	**7**	**5**	**2**	**0**
Species richness and composition metrics					
Total number of species	10 or more	8 or 9	6 or 7	4 or 5	0 - 3
Number of headwater species[2]	3 or more		1 or 2		0
Number of minnow species[2]	5 or more	4	2 or 3	1	0
Percent tolerant species[3]	0-60	61-70	71 - 80	81-90	91-100
Percent dominant two species[3]	0-52	53-64	65 - 76	77-88	89-100
Trophic composition and reproductive function metrics					
Number of invertivore species[2]	5 or more	4	2 or 3	1	0
Percent simple lithophils[3]	49-100	37-48	25-36	13-24	0-12
Fish abundance and condition metrics					
Number of fish per 100 meters[2]	11 or more				0-10
Percent DELT anomalies[3]	0-1		2 or 3		4 or more

[1]The sum of the nine metrics for headwater streams must be multiplied by 1.11 to obtain the final IBI score.
[2]Number of headwater species, number of minnow species, number of invertivore species, and number of fish per 100 meters metrics do not include tolerant species.
[3]Round all percent metrics to the nearest 1 percent.

Table 2. Scoring criteria for the nine metrics used to calculate the IBI for small streams (20 to 54 mi^2 drainage area) in the St. Croix River Basin of Minnesota[1].

	B. Scoring Criteria				
Metric	**10**	**7**	**5**	**2**	**0**
Species richness and composition metrics					
Total number of species	15 or more	12-14	9-11	6-8	0-5
Number of intolerant species	4 or more	3	2	1	0
Number of minnow species[2]	6 or more	5	3 or 4	2	0 or 1
Percent tolerant species[3]	0-40	41-55	56-70	71-85	86-100
Percent dominant two species[3]	0-44	45-58	59-72	73-86	87-100
Trophic composition and reproductive function metrics					
Number of benthic invertivore species	4 or more	3	2	1	0
Percent simple lithophils[3]	49-100	37-48	25-36	13-24	0-12
Fish abundance and condition metrics					
Number of fish per 100 meters[2]	11 or more				0–10
Percent DELT anomalies[3]	0-1		2 or 3		4 or more

[1]The sum of the 9 metrics for headwater streams must be multiplied by 1.11 to obtain the final IBI score.
[2]Number of minnow species, and number of fish per 100 meters metrics do not include tolerant species.
[3]Round all percent metrics to the nearest 1 percent.

Table 3. Scoring criteria for the ten metrics used to calculate the IBI for moderate size streams (55 to 270 mi^2 drainage area) in the St. Croix River Basin and rivers (>270 mi^2 drainage area) in the Northern Lakes and Forests ecoregion portion of the St. Croix River Basin in Minnesota. See appendix 2 for scoring criteria for rivers in the North Central Hardwood Forests ecoregion.

	Scoring Criteria				
Metric	**10**	**7**	**5**	**2**	**0**
Species richness and composition metrics					
Total number of species	23 or more	20-22	17-19	14-16	0-13
Number of darter species	5 or more	4	3	2	0 or 1
Number of intolerant species	8 or more	7	4-6	3	0-2
Percent tolerant species[1]	0-20	21-40	41-60	61-80	81-100
Trophic composition and reproductive function metrics					
Number of benthic invertivore species	9 or more	7 or 8	5 or 6	3 or 4	0-2
Number of omnivore species	0 or 1	2	3	4	5 or more
Percent piscivore species[1]	25-100	19-24	13-18	7-12	0-6
Percent simple lithophils[1]	61-100	46-60	31-45	16-30	0-15
Fish abundance and condition metrics					
Number of fish per 100 meters[2]	11 or more				0–10
Percent DELT anomalies[1]	0-1		2 or 3		4 or more

[1]Round all percent metrics to the nearest 1 percent.
[2]Number of fish per 100 meters metrics does not include tolerant species.

Table 4. Guidelines for interpreting overall IBI scores (from Lyons 1992)

Overall IBI Score	Biotic Integrity Rating	Fish Community Attributes
100-65	Excellent	Comparable to the best situations with minimal human disturbance; all regionally expected species for habitat and stream size, including the most intolerant forms, are present with a full array of age and size classes; balanced trophic structure
64-50	Good	Species richness somewhat below expectations, especially due to the loss of the most intolerant forms; some species are present with less than optimal abundance's or size/age distributions; trophic structure may show signs of imbalance.
49-30	Fair	Signs of additional deterioration include decreased species richness, loss of intolerant forms, reduction in simple lithophils, increased abundance of tolerant species, and/or highly skewed trophic structure (e.g., increasing number of omnivore species and less specialized feeding species); older age classes of top carnivores rare or absent.
29-20	Poor	Relatively few species; dominated by tolerant forms, habitat generalists, and omnivores; few or no top carnivores or simple lithophilic spawners; growth rates and condition factors sometimes depressed; hybrids sometimes common.
19-0	Very Poor	Very few species present, mostly tolerant forms, hybrids, or exotics; few large or older fish; DELT fish (fish with deformities, eroded fins, lesions, or tumors) sometimes common.
No Score		Thorough sampling finds few or no fish; impossible to calculate IBI.

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APPENDIX 1-METRIC VALIDATION

Karr and Chu (1999) state that "no metric should become part of a multimetric index before it is thoroughly and systematically tested and its response has been validated against a gradient of human influence". The following paragraphs outline the steps that were taken to validate metrics used in the St. Croix River Basin IBI.

SITE SELECTION

The St. Croix River Basin IBI was developed with data collected during the 1996 through 1999 sampling seasons. One hundred and thirty four sampling events were conducted at 100 sites throughout the basin. A complete list of IBI scores for each site is provided in appendix 5.

The sites selected for development of an IBI should focus on multiple sites within similar environments, across a range of human disturbance from minimal to severe (Karr and Chu 1999). We selected 50 sites to represent a range of stream sizes, disturbances, and morphology types within the basin. Least disturbed sites were selected by assessing habitat and land use within the watershed. Disturbed sites were selected by examining land use, point source discharge, feedlot, and stream ditching coverages to locate stream reaches where the cumulative effects of multiple stressors were likely to be the greatest.

Fifty additional sites were used in the analysis but were not selected specifically for the purpose of developing the IBI. Rather, these sites were chosen randomly to monitor the condition of rivers and streams throughout the St. Croix River Basin (fig. 1). These sites were important in the process of IBI development because they helped to provide a better understanding of stream characteristics throughout the basin, the magnitude of human disturbance throughout the basin, and the types of human disturbance that appeared to influence biological integrity.

HABITAT ANALYSIS

A quantitative habitat assessment was performed at each site to characterize the instream and riparian features of the stream reach. The habitat information was used to classify streams and delineate excellent quality sites from poor quality sites. We used a modified version of Wisconsin's quantitative habitat assessment procedure (Simonson et al. 1994). We also developed a qualitative habitat assessment (table 5) similar to Wisconsin's Fish Habitat Rating System (FHR) to provide a summary of habitat data and compare the results of assessments between streams. We selected six variables that were the most highly correlated (Pearson correlation coefficient) with species richness and included components of stream geomorphology, substrate, instream habitat, and riparian land use. Least impacted (i.e. candidate reference) sites were used to develop scoring criteria for the qualitative assessment. Sites with less than 25 percent land use disturbance in the watershed, no obvious pollution sources immediately upstream, and no observable habitat alterations within the reach were considered least impacted. Each habitat variable was assigned a rating of 2 (similar to least impacted sites), 1 (somewhat deviate from least impacted sites, or 0 (strong deviation from least impacted sites). The total score ranged from 0 (poor habitat) to 12 (excellent habitat).

QUANTIFYING HUMAN DISTURBANCE

At any given point along a stream, resource integrity may be affected by the interaction of many human activities within the watershed. This is particularly true in a

Table 5. Habitat definitions and scoring criteria for a qualitative habitat index developed for streams within the St. Croix River Basin. Habitat values are derived from Wisconsin's habitat assessment guidance (Simonson et al. 1994).

Definitions:

Number of stream features: The number of major morphological features (riffles, runs, pools, bends) per 100 meters of stream.

Percent course substrate types: The percent of the substrate that is gravel size or larger. This figure is calculated using the dominant substrate found at 4 equally spaced points along each of 13 transects

Number of substrate types: The number of substrate types (silt, sand, cobble, etc.) within the stream reach.

Coefficient of variation of the depth: The coefficient of variation of the thalweg depth measurements taken at each transect.

Sinuosity: Length of the stream reach divided by the straight-line distance between the upstream and downstream ends. Calculated from 1:24,000 USGS quadrangle topographical maps

Percent land use disturbance within 30 meters of the stream: The percent of the riparian zone within 30 meters of the stream that is influenced by human disturbance. Human disturbance land use categories include cropland, pasture, barnyard, or developed. This figure is calculated using the dominant land use on each side of the stream bank at each of 13 transects.

	Scoring criteria for glide/pool streams			**Scoring criteria for riffle/run streams**								
	0-54 mi^2 drainage area (n=5)			very small streams 0-20 mi^2 drainage area (n=9)			small streams 20-54 mi^2 drainage area (n=4)			moderate size streams 55-270 mi^2 drainage area (n=10)		
Scoring	**2**	**1**	**0**	**2**	**1**	**0**	**2**	**1**	**0**	**2**	**1**	**0**
Habitat variable												
Number of stream features	> 3	2-3	< 2	> 10	6-10	< 6	> 4	3-4	< 3	> 2	1-2	0
Percent course substrate types	> 17	8-17	< 8	> 66	33-66	< 33	> 80	40-80	< 40	> 75	37-75	< 37
Number of substrate types	> 4	4	< 4	> 4	4	< 4	> 4	4	< 4	> 4	4	< 4
C.V. of depth	> 22	11-22	< 11	> 50	25-50	< 25	> 53	26-53	< 26	> 36	18-36	<18
Sinuosity	> 1.2	1.1-1.2	< 1.1	> 1.2	1.1-1.2	< 1.1	> 1.2	1.1-1.2	< 1.1	> 1.2	1.1-1.2	< 1.1
Percent land use disturbance within 30 meters of stream	< 10	10-50	> 50	< 10	10-50	> 50	< 10	10-50	> 50	< 10	10-50	> 50

river basin like the St. Croix where a variety of land use activities occur. No single variable can completely represent human disturbance because of the complex and dynamic nature of the disturbances. We explored numerous avenues in attempting to define a disturbance gradient that accurately reflected disturbance within the basin including: 1) general rankings of each site from excellent to poor based on our first hand knowledge of conditions at the site, 2) rankings based on GIS coverages for land use, ditching, point source discharges, feedlots, roadways etc. 3) identification of variables from the habitat assessment (i.e., percent fines, percent embeddedness, percent of disturbed riparian area) that may reflect human disturbance. We chose a GIS based watershed characterization of disturbance because it could be calculated easily using GIS land use coverages, it could not be confused with naturally occurring factors (for example, the percent fine substrate within the reach could be a reflection of human disturbance or natural geologic features within the watershed), and it is understandable conceptually: That is, the more the watershed is altered, the higher the probability the rivers and streams within the watershed will be impaired.

Upstream land use in the watershed was characterized using 1990 vintage (MDNR filename: lulcxpy3) or 1995 vintage (MDNR filename lusatpy3) GIS land use coverages depending on which coverage was available for each site. The GIS land use theme was overlaid in Arcview onto the drainage area theme and clipped producing a land use theme identical in shape and size to the drainage area theme. Land uses were then summed across the entire drainage area and then divided by the total area to produce percentages for each land use. The percent watershed disturbance was calculated by adding the percentages for the land use themes that were indicative of human disturbance. This included all agricultural and urban themes, grassland that was most often associated with pastured areas, and mines and open pits. The vast majority of disturbed land use in the St. Croix River Basin was agricultural in nature.

STREAM CLASSIFICATION

Proper stream classification is a very important component in IBI development. With too few stream classes it may be difficult to distinguish between natural stream variability and human induced variability (Karr and Chu 1999). On the other hand, the limited resources available to conduct biological monitoring may be wasted with too many stream classes. We considered water temperature, stream size, morphological type (riffle/run or glide/pool), and ecoregion as possible stream classification variables.

Stream temperature greatly influences the structure of the fish community and consequently, the metrics in an IBI (Lyons 1992; Lyons et al. 1996; Mundahl and Simon 1999). We did not include stream reaches considered to be coldwater in this study. Therefore, any data from a stream that contained a significant population of trout was omitted from the data set. The distinction between warm and coolwater streams is not as easily defined. Lyons (1992) provided a list of primary and secondary coolwater species for Wisconsin streams. Primary coolwater species are generally restricted to coolwater streams, while secondary coolwater species occur commonly in both cool and warmwater streams (Lyons 1992). Using these guidelines along with the fish community and water temperature data from this study, we concluded that the majority of non-coldwater streams in the St. Croix River Basin have, or at one time had, the temperature and fish community characteristics of coolwater streams.

The St. Croix River Basin IBI accounts for differences in metric expectations due to stream size by developing separate scoring criteria for 4 stream size classes. To determine size classification break points a scatter plot of watershed drainage area (log_{10}) versus species richness was constructed using all available data including replicate samples (fig. 3). As expected, species richness was significantly correlated with drainage area (log_{10}) across the full range of stream sizes (Spearman r_s, $p<0.05$). Size classes were then chosen to minimize differences in maximum species richness within each size class. For example, streams with watersheds of 0 to 20 mi^2 were placed into a size class because the maximum species richness within that range of stream sizes was similar. Stream size was no longer correlated with species richness when sites were separated into four size classes (Spearman r_s, $p>0.05$).

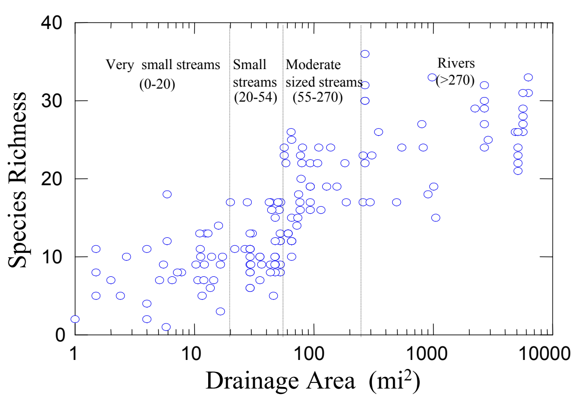


Figure 3. Species richness versus drainage area (mi^2). Vertical lines represent size class break points

We categorized sites as either riffle/run or glide/pool streams based on habitat information collected using Wisconsin's habitat assessment guidance (Simonson et al 1994). In our judgement, the most important habitat features used to distinguish between different stream morphological classes were the presence of riffles within the reach, stream gradient, and width-to-depth ratio. However, other physical stream characteristics such as dominant substrate type and riparian vegetation were also important considerations (table 6).

To determine if stream morphology or ecoregion had any affect on the candidate IBI metrics, we plotted each candidate metric against percent watershed disturbance for each size class (fig. 4). Karr and Chu (1999) refer to these graphs as ecological dose response curves. Two characteristics of the dose response curves were used to examine whether there were differences due to stream morphology or ecoregion: 1) a difference in the potential (maximum value) of the candidate metric due to stream morphology or ecoregion and, 2) a notable difference in the dose response due to morphology or ecoregion.

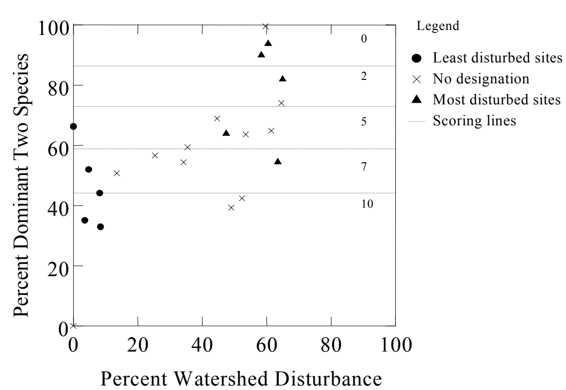


Figure 4. An ecological dose response curve and scoring criteria for the percent dominant two species metric

It was difficult to evaluate each of the 12 potential stream classes (3 stream size classes, 2 morphology types, 2 ecoregion types) with only 100 total sites. However,

Table 6. Guidelines for classifying stream reaches into a morphological type, listed in order of importance from top to bottom. Habitat variables used to classify streams by morphological type were collected using Wisconsin's habitat assessment guidance (Simonson et al. 1994).

Stream Characteristics	**Riffle/Run**	**Glide/Pool**
Prevalence of riffles	Riffles usually present within the stream reach	No riffles within the stream reach
[1]Width-to-Depth ratio	Usually > 12	Usually 12 or less
[2]Stream gradient	Usually > 1.0 m/km	Usually < 1.0 m/km
Substrate type	Course substrates usually prevalent	Course substrates not a significant component of stream bottom
Riparian zone type	In least impacted streams the dominant riparian vegetation is usually forest	In least impacted streams the dominant riparian vegetation is usually wetland, grass, or shrubs.

[1]Width-to-depth ratio is obtained by dividing the average stream width by the average thalweg depth in runs and pools.
[2]Stream gradient was obtained using 1:24,000 USGS topographic maps.

careful scrutiny of the dose response curves indicated that stream morphology and ecoregion differences were not large enough to warrant separating streams into different ecoregion or morphology classes. Although fish species sometimes differed between riffle/run and glide/pool streams, the response and expectations (i.e. number of fish species) of all of the final metrics were similar. A few of the candidate metrics we tested appeared to be influenced by stream morphology (i.e. number of simple lithophilic species). However, we did not include any of the candidate metrics influenced by stream morphology differences in the final version of the IBI.

The ecoregion concept is the most common geographical framework used to develop IBI's throughout the country. However, ecoregional differences were not an important factor in developing an IBI for very small to moderate size streams of the St. Croix River Basin in Minnesota. Ecoregions may, however, be a more important factor in rivers of the St. Croix River Basin (appendix 2).

GRAPHICAL ANALYSIS OF CANDIDATE METRICS

Methods

Simon and Lyons (1995) have compiled a comprehensive list of metrics that has been used successfully throughout the country. We referred to this list to select candidate metrics for possible inclusion into the St. Croix IBI. Many of the metrics listed by Simon and Lyons (1995) have been used successfully in IBIs throughout the Midwest.

Ecological dose response curves were used to select and validate metrics. Attributes of the fish community were plotted against percent watershed disturbance to yield an ecological dose response curve for each fish community attribute within each class of stream. Two properties of the dose response curves were used as criteria for validating the dose response relationship between the fish community attribute and percent watershed disturbance: 1) The association (correlation) between percent watershed disturbance and the fish community attribute and 2) the difference between the attribute values from the 5 least disturbed and the 5 most disturbed sites. The five least disturbed sites within each size class were selected by determining which sites had the least watershed disturbance and the best qualitative habitat rankings. Conversely, the five most disturbed sites within each size class were those that had the most watershed disturbance and worst qualitative habitat rankings (appendix 5). Attribute values for the least disturbed sites should separate from the attribute values from the most disturbed sites along the Y-axis of the dose response curve. Attributes of the fish community that demonstrated a response using either of the methods were retained for further consideration. Spearman r_s values were calculated to test for significance of the dose response relationship (table 7). A Mann-Whitney U test was used to test for significant differences ($p<0.05$) between the most and least disturbed sites (fig. 5). Correlation matrixes were constructed to examine the correlation of each metric to the IBI score and the redundancy between each metric.

Results

Most metrics in the IBI were significantly correlated with disturbance (Spearman r_s, $p<0.05$), (table 7) or the metric values from the least disturbed sites were significantly different from the most disturbed sites (Mann-Whitney U, $p<0.05$), (fig. 5). Some metrics were correlated with human disturbance for 1 or 2 of the size classes, but

Table 7. Spearman rank correlation coefficients and significance values for each metric and total IBI score against percent watershed disturbance within each stream size class.

	Very small streams (< 20 mi^2)		Small streams (20-54 mi^2)		Moderate streams (55-270 mi^2)	
Metric	correlation coefficient (r_s)	significance value (*p*)	correlation coefficient (r_s)	significance value (*p*)	correlation coefficient (r_s)	significance value (*p*)
Species richness and composition metrics						
Total number of species	-.316	.0864	-.473	.0297	.190	.3639
Number of headwater species	-.252	.1754				
Number of minnow species	-.399	.0292	-.484	.0251		
Number of darter species					-.294	.1435
Number of intolerant species			-.290	.1922	-.399	.0423
Percent tolerant species	.451	.0119	-.128	>.5	.306	.1261
Percent dominant two species	.318	.0845	.599	.0042		
Trophic and reproductive function metrics						
Number of invertivore species	-.402	.0277				
Number of benthic invertivore species			-.188	.4182	-.169	.4168
Number of omnivore species					.725	<.001
Number of piscivore species					-.347	.0741
Percent simple lithophils	-.457	.0105	-.364	.0977	-.299	.1362
Fish abundance and condition metrics						
Number of fish per 100 meters	-.296	.1151	-.304	.1740	.227	.2706
Percent DELT anomalies	-.163	.3991	-.039	>.5	-.077	>.5
Total IBI score	-.476	.0077	-.529	.0131	-.415	.0388

not all of them. Such was the case with the percent tolerant species metric which was strongly correlated with disturbance in the very small streams (0-20 mi^2) but was not as strongly correlated with disturbance in the larger size streams. A few of the metrics would not be expected to elicit a dose response in the St. Croix River Basin where agricultural and industrial activities are relatively light in comparison to other areas of Minnesota. For example, the percent of fish with DELT anomalies and the number of fish per 100 meter metrics are designed to respond to changes in the fish community in the most degraded streams. The DELT anomalies metric in particular has proven useful within other regions of the Midwest as an indicator of industrial pollution (Ohio EPA 1988). Should human activities within the St. Croix River Basin intensify these metrics will become more valuable.

For many of the metrics there was a clear separation along the Y-axis of the dose response curve for the 5 least disturbed sites and the 5 most disturbed sites even though the correlation between watershed disturbance and the fish community attribute was not statistically significant (Mann-Whitney U, $p<0.05$), (fig. 5). For example, the percent tolerant species and number of piscivore species metrics in moderate streams were not significantly correlated with disturbance but there was a significant difference between the 5 most disturbed and 5 least disturbed sites.

All metrics used in the IBI were examined to detect redundancies between metrics. Mundahl and Simon (1999) eliminated metrics that were the weakest discriminator between reference and impaired sites if they were highly correlated with each other (Spearman $r_s > 0.80$). Using this approach, we did not find any significant redundancies between metrics in small or moderate size streams. A few of the metrics were significantly correlated with each other in very small streams. For example, there was a Spearman r_s of 0.86 between the total number of species and the number of minnow species metrics, and a Spearman r_s of 0.87 between the percent tolerant species and dominant two species metrics. However, we did not eliminate any metrics based on their statistical correlation with other metrics, opting instead to include 9 to 10 metrics in each IBI.

IBI scores in very small streams were correlated with watershed disturbance (Spearman r_s, $p<0.05$) (table 7). However, the least and most disturbed sites were not significantly different from each other (Mann-Whitney U test, $p<0.05$) (fig. 6). IBI scores for least disturbed sites in very small streams ranged from 37 to 84. The wide range in IBI scores in least disturbed sites suggests that a better understanding of these systems is needed to properly classify them. For example, the West Branch of the Kettle River had an IBI score of 11 even though the percent disturbance in the watershed was only 15% (appendix 5). This site was essentially a low gradient glide/pool system that flowed through a large wetland. The dissolved oxygen concentration in the middle of the afternoon was <3mg/l. Most likely this was a naturally occurring phenomenon. This suggests that streams of this nature may need to be separated into a distinct class and a new IBI developed specifically for this type of stream; an IBI that takes into account the unique physical and biological characteristics of these streams. Two sites with a high percentage of watershed disturbance and poor habitat had relatively high IBI scores. In spite of the disturbance present in these streams other factors (e.g. a more intact riparian zone along streams throughout the watershed, less intensive agriculture, better feedlot waste management, etc) may have played a role in protecting the biological integrity of these streams.

IBI scores for small and moderate streams were significantly correlated with disturbance (Spearman r_s, $p<0.05$) (table 7) and the IBI scores for the least disturbed sites were significantly different from the most disturbed sites (Mann-Whitney U test, $p<0.05$) (fig. 6). The range in IBI scores for least disturbed sites in the small and moderate stream classes was less than in very small streams (68-86 in small streams, 60-97 in moderate streams). The range in watershed disturbance for small and moderate streams was also less, rarely exceeded 65%. The relatively narrow range of watershed disturbance values in small and moderate size streams complicated metric validation because the dose response relationship between watershed disturbance and the fish community attribute was not as pronounced.

IBI scores were correlated with metric values for each size class with few exceptions (Spearman r_s, $p<0.05$), (table 8). There was no correlation between IBI score and the DELT anomalies metric for any stream size class. This is not surprising for the reasons mentioned previously. The number of fish per 100 meters metric was strongly correlated with IBI score ($p<0.05$) for all size classes except the moderate size class. The metric was retained because of its use in detecting severe impairment. The percent tolerant species metric was not significantly correlated with IBI scores in small streams (20-54 mi^2) but was correlated with IBI scores in other size classes. In very small streams the percent of simple lithophils metric was not significantly correlated with IBI scores, although nearly so ($p=.062$).

A

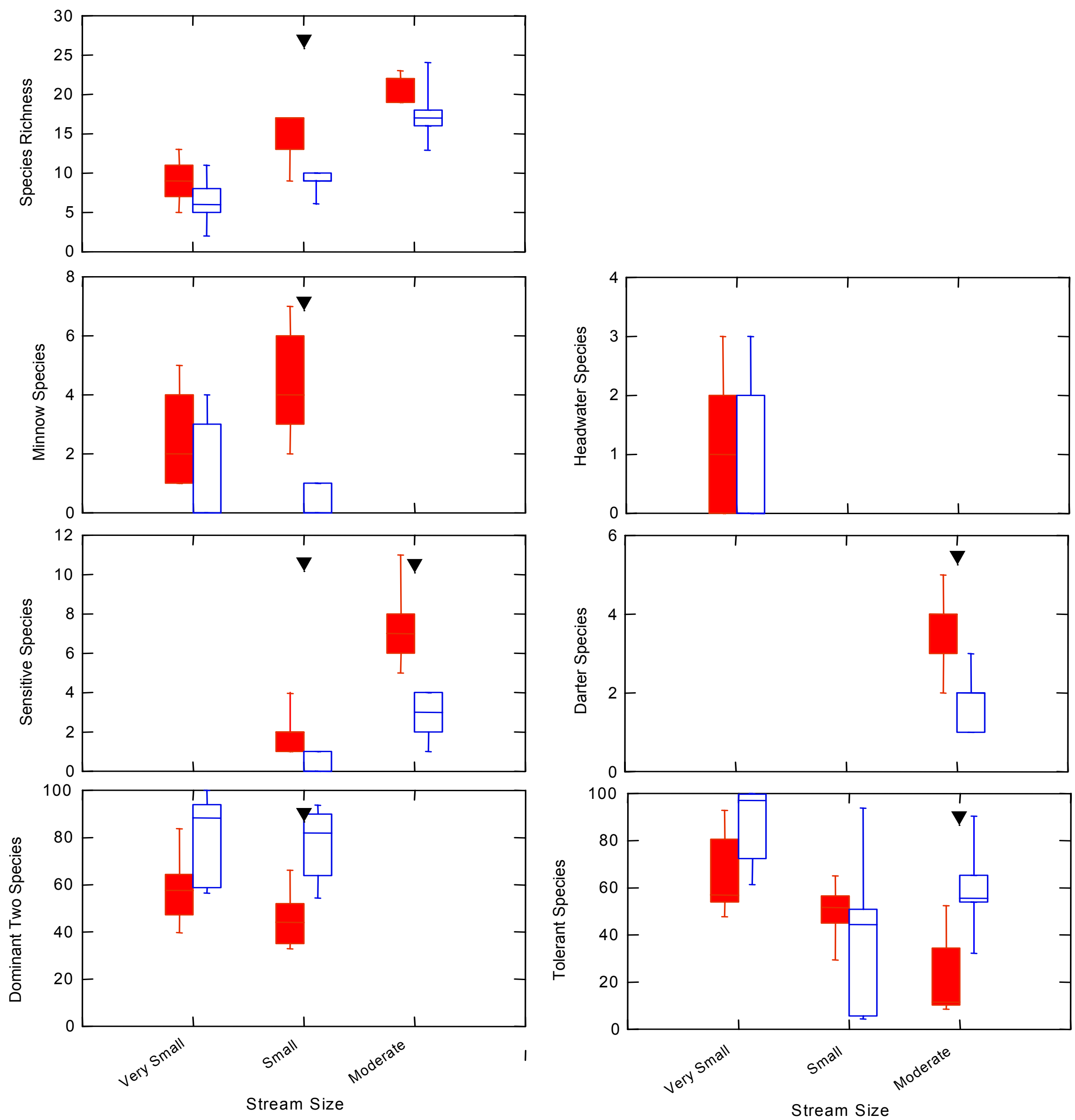
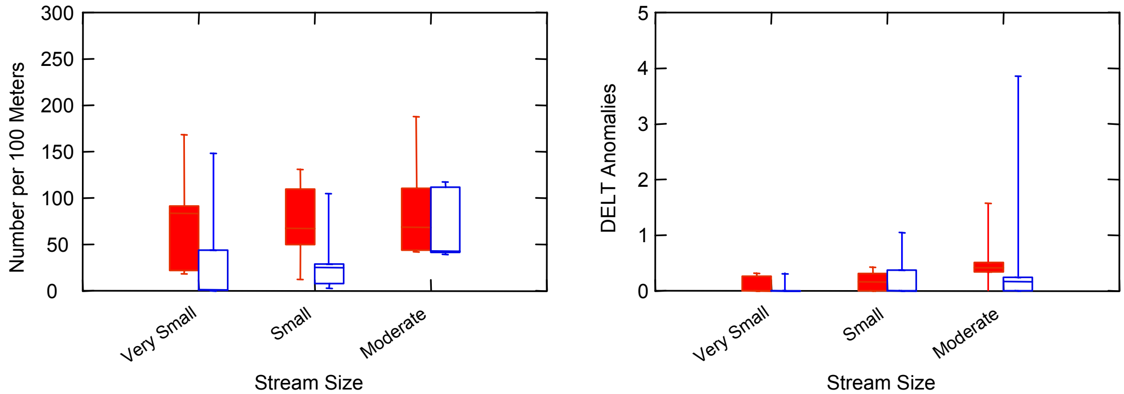


Figure 5. Box and whisker graphs of each metric by stream size showing the observed range (whisker boundaries) and 25th and 75th percentiles (box boundaries) for the five least disturbed sites (filled boxes) and the five most disturbed (no fill). The metrics are grouped by (A) species richness, (B) abundance and condition, and (C) trophic composition and reproductive function,. Downward arrows indicate a significant difference (Mann-Whitney U, $p<0.05$) exists between the least disturbed and most disturbed sites within a particular size class.

B



C

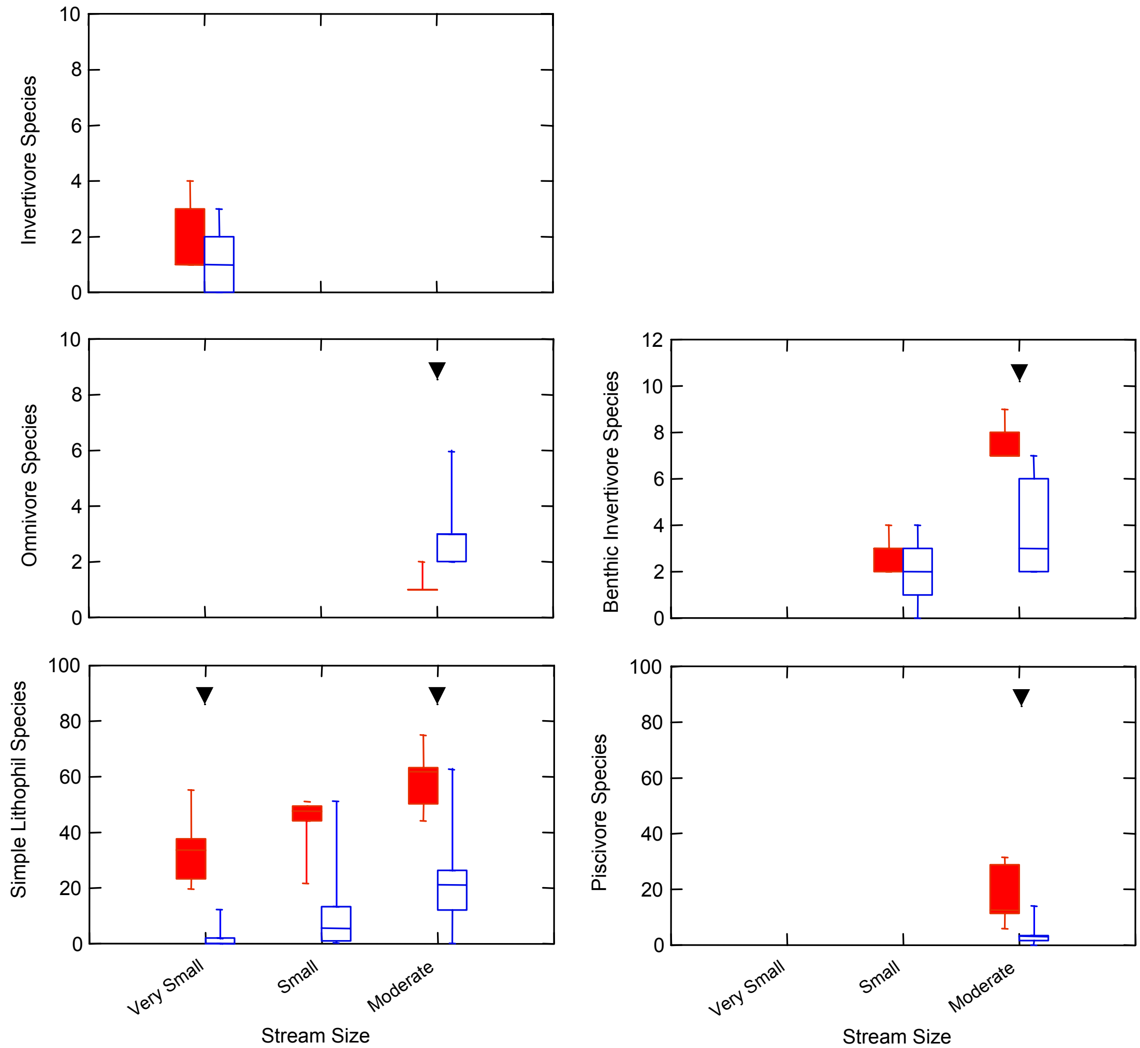


Figure 5 continued.

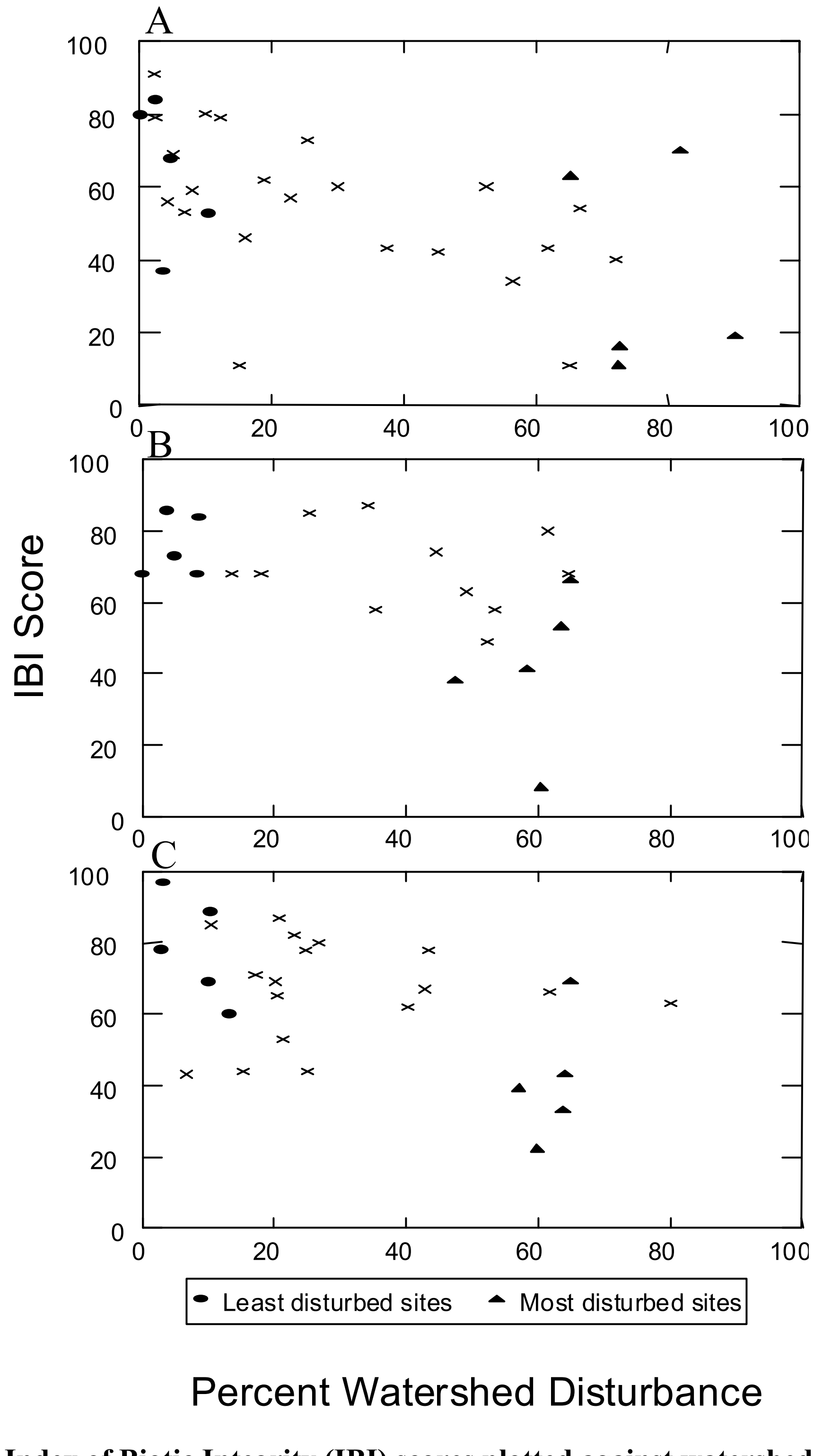


Figure 6. Index of Biotic Integrity (IBI) scores plotted against watershed disturbance for (A) very small streams (<20 mi^2 drainage area), (B) small streams (20-54 mi^2 drainage area), and moderate size streams (55-270 mi^2 drainage area).

Table 8. Spearman rank correlation coefficients (r_s) and significance values (p) between IBI score and individual metrics for each stream size class.

Metric	Very small streams (< 20 mi^2) correlation coefficient (r_s)	Very small streams (< 20 mi^2) significance value (p)	Small streams (20-54 mi^2) correlation coefficient (r_s)	Small streams (20-54 mi^2) significance value (p)	Moderate streams (55-270 mi^2) correlation coefficient (r_s)	Moderate streams (55-270 mi^2) significance value (p)
Species richness and composition metrics						
Total number of species	.865	<.001	.828	<.001	.478	.0112
Number of headwater species	.742	<.001				
Number of minnow species	.922	<.001	.665	<.001		
Number of darter species					.795	<.001
Number of intolerant species			.701	<.001	.655	<.001
Percent tolerant species	-.895	<.001	-.213	.3386	-.762	<.001
Percent dominant two species	-.828	<.001	-.450	.0394		
Trophic and reproductive function metrics						
Number of invertivore species	.811	<.001				
Number of benthic invertivore species			.717	<.001	.726	<.001
Number of omnivore species					-.597	<.001
Percent piscivore species					.584	.0015
Percent simple lithophils	.332	.062	.816	<.001	.849	<.001
Fish abundance and condition metrics						
Number of fish per 100 meters	.516	.0026	.470	.0309	-.363	.0603
Percent DELT anomalies	-.071	>.5	-.172	.4545	.191	.353

APPENDIX 2–RIVER METRICS

RIVERS (>270 mi²)

We did not attempt to validate metrics for rivers (drainage area > 270 mi²) because there was not a sufficient gradient of disturbance to validate each metric. Comparable rivers outside of the St. Croix River Basin may be needed to develop dose response relationships between disturbance and attributes of the fish community.

We used the metrics developed for the moderate streams to develop an IBI for rivers. The IBI scores for rivers should be considered tentative until each metric is validated against a gradient of human disturbance. Scoring was determined using the same techniques as in the smaller streams (appendix 4). Species expectations were generally higher for the lower portion of the St Croix and Snake River systems than the upper St. Croix and Kettle River systems. The differences corresponded roughly with ecoregion boundaries. Therefore, IBI scoring criteria for rivers in the Northern Lakes and Forests ecoregion are identical to scoring criteria for moderate size streams, (table 3) while scoring criteria for rivers in the North Central Hardwood Forests ecoregion have been adjusted upward (table 9).

Table 9. Scoring criteria for the ten metrics used to calculate the IBI for rivers (>270 mi² drainage area) in the North Central Hardwood Forests ecoregion portion of the St. Croix basin of Minnesota. See table 3 for river scoring criteria in the Northern Lakes and Forests ecoregion.

	Scoring Criteria				
Metric	**10**	**7**	**5**	**2**	**0**
Species richness and composition metrics					
Total number of species	29 or more	24-28	19-23	14-18	0-13
Number of darter species	5 or more	4	3	2	0 or 1
Number of intolerant species	8 or more	7	6	5	0-4
Percent tolerant species[1]	0-20	21-40	41-60	61-80	81-100
Trophic composition and reproductive function metrics					
Number of benthic invertivore species	11 or more	9 or 10	7 or 8	5 or 6	0-4
Number of omnivore species	0 or 1	2	3	4	5 or more
Percent piscivore species[1]	25-100	19-24	13-18	7-12	0-6
Percent simple lithophils[1]	61-100	46-60	31-45	16-30	0-15
Fish abundance and condition metrics					
Number of fish per 100 meters[2]	11 or more				0–10
Percent DELT anomalies[1]	0-1		2 or 3		4 or more

[1]Round all percent metrics to the nearest 1 percent.
[2]Number of fish per 100 meters metric does not include tolerant species.

APPENDIX 3 - ST. CROIX RIVER BASIN FISH ASSEMBLAGE* AND IBI CLASSIFICATION

Common name	Scientific name	IBI Classification[a]		
		Taxa	Trophic status	Reproductive guild
Lampreys	**Petromyzontidae**			
American brook lamprey	*Lampetra appendix*	He In		
Chestnut lamprey	*Ichthyomyzon castaneus*	In	Pi	
Northern brook lamprey***	*Ichthyomyzon fossor*	In		
Southern brook lamprey***	*Ichthyomyzon gagei*	In		
Silver lamprey	*Ichthyomyzon unicuspis*	In	Pi	
Sturgeons	**Acipenseridae**			
Lake sturgeon***	*Acipenser fulvescens*		Bi In	Sl
Shovelnose sturgeon**	*Scaphirhynchus platorynchus*		Bi In	Sl
Paddlefishes	**Polyodontidae**			
Paddlefish****	*Polydon spathula*	In		Sl
Gars	**Lepisosteidae**			
Longnose gar	*Lepisosteus osseus*		Pi	
Shortnose gar	*Lepisosteus platostomus*		Pi	
Bowfins	**Amiidae**			
Bowfin	*Amia calva*		Pi	
Freshwater eels	**Anguillidae**			
American eel	*Anguilla rostrata*		Pi	
Herrings	**Clupeidae**			
Skipjack herring** ***	*Alosa chrysochloris*		Pi	
Gizzard shad	*Dorosoma cepedianum*			
Mooneyes	**Hiodontidae**			
Goldeye**	*Hiodon alosoides*	In	In	
Mooneye	*Hiodon tergisus*	In	In	
Trouts	**Salmonidae**			
Cisco (lake herring)	*Coregonus artedii*			
Rainbow trout	*Oncorhynchus mykiss*		Pi	
Brown trout	*Salmo trutta*		Pi	
Lake trout	*Salvelinus namaycush*		Pi	
Brook trout	*Salvelinus fontinalis*	In	Pi	
Pikes	**Esocidae**			
Northern pike	*Esox lucius*		Pi	
Muskellunge	*Esox masquinongy*	In	Pi	
Mudminnows	**Umbridae**			
Central mudminnow	*Umbra limi*	To	In	

APPENDIX 3. (continued)

Common name	Scientific name	IBI Classification[a] Taxa	Trophic status	Reproductive guild
Minnows	**Cyprinidae**			
Common carp	*Cyprinus carpio*	To	Om	
Brassy minnow	*Hybognathus hankinsoni*	Mi		
Golden shiner	*Notemigonus crysoleucas*	To		
Creek chub	*Semotilus atromaculatus*	To		
Blacknose dace	*Rhinichthys atratulus*	To		Sl
Longnose dace	*Rhinichthys cataractae*	Mi In	Bi In	Sl
Hornyhead chub	*Nocomis biguttatus*	Mi In	In	
Spottail shiner	*Notropis hudsonius*	Mi In	In	
Pallid shiner** ***	*Notropis amnis*	Mi In	In	
Emerald shiner	*Notropis atherinoides*	Mi	In	Sl
Sand shiner	*Notropis stramineus*	Mi	In	
Weed shiner**	*Notropis texanus*	Mi In	In	
Mimic shiner	*Notropis volucellus*	Mi In	In	
Pugnose shiner***	*Notropis anogenus*	Mi In	In	
River shiner**	*Notropis blennius*	Mi	In	Sl
Bigmouth shiner	*Notropis dorsalis*	Mi	In	
Blackchin shiner	*Notropis heterodon*	Mi In	In	
Blacknose shiner	*Notropis heterolepis*	Mi In	In	
Central stoneroller	*Campostoma anomalum*	Mi		
Largescale stoneroller	*Campostoma oligolepis*	Mi		
Bluntnose minnow	*Pimephales notatus*	To		
Fathead minnow	*Pimephales promelas*	To	Om	
Northern redbelly dace	*Phoxinus eos*	He Mi	In	
Finescale dace	*Phoxinus neogaeus*	He Mi	In	
Spotfin shiner	*Cyprinella spiloptera*	Mi	In	
Common shiner	*Luxilus cornutus*	Mi		Sl
Speckled chub	*Macrhybopisis aestivalis*	Mi In	Bi In	
Silver chub	*Macrhybopsis storeriana*	Mi	Bi In	
Pearl dace	*Margariscus margarita*	He Mi	In	
Pugnose minnow	*Opsopoeodus emiliae*	Mi In	In	
Suckers	**Catostomidae**			
White sucker	*Catostomus commersoni*	To	Om	
Quillback	*Carpiodes cyprinus*		Om	
Highfin carpsucker	*Carpiodes velifer*	In	Om	
Shorthead redhorse	*Moxostoma macrolepidotum*		Bi In	Sl
Silver redhorse	*Moxostoma anisurum*		Bi In	Sl
River redhorse	*Moxostoma carinatum*	In	Bi In	Sl
Golden redhorse	*Moxostoma erythrurum*		Bi In	Sl
Greater redhorse	*Moxostoma valenciennesi*	In	Bi In	Sl
Northern hogsucker	*Hypentelium nigricans*	In	Bi In	Sl
Smallmouth buffalo	*Ictiobus bubalus*		Om	
Bigmouth buffalo	*Ictiobus cyprinellus*	To	Om	
Spotted sucker	*Minytrema melanops*		Bi In	Sl

APPENDIX 3. (continued)

IBI classification[a]

Common name	*Scientific name*	Taxa	Trophic status	Reproductive guild
Catfishes	**Ictaluridae**			
Blue catfish	*Ictalurus furcatus*		Pi	
Channel catfish	*Ictalurus punctatus*		Pi	
Tadpole madtom	*Noturus gyrinus*		Bi In	
Stonecat	*Noturus flavus*	In	Bi In	
Flathead catfish	*Pylodictis olivaris*		Pi	
Black bullhead	*Ameiurus melas*	To	Om	
Yellow bullhead	*Ameiurus natalis*		Om	
Brown bullhead	*Ameiurus nebulosus*		Om	
Trout-perches	**Percopsidae**			
Trout-perch	*Percopsis omiscomaycus*		Bi In	
Codfishes	**Gadidae**			
Burbot	*Lota lota*		Pi	Sl
Killifishes	**Cyprinodontidae**			
Banded killifish	*Fundulus diaphanus*		In	
Silversides	**Atherinidae**			
Brook silverside	*Labidesthes sicculus*		In	
Sticklebacks	**Gasterostidae**			
Brook stickleback	*Culaea inconstans*	To	In	
Sculpins	**Cottidae**			
Slimy sculpin	*Cottus cognatus*	He In	Bi In	
Mottled sculpin	*Cottus bairdi*	He In	Bi In	
Temperate basses	**Percichthyidae**			
White bass	*Morone chrysops*		Pi	
Sunfishes	**Centrarchidae**			
Rock bass	*Ambloplites rupestris*	In	Pi	
Green sunfish	*Lepomis cyanellus*	To		
Warmouth	*Lepomis gulosus*		Pi	
Bluegill	*Lepomis macrochirus*		In	
Pumpkinseed	*Lepomis gibbosus*		In	
Longear sunfish	*Lepomis megalotis*	In	In	
Smallmouth bass	*Micropterus dolomieui*	In	Pi	
Largemouth bass	*Micropterus salmoides*		Pi	
White crappie	*Pomoxis annularis*		Pi	
Black crappie	*Pomoxis nigromaculatus*		Pi	
Perches	**Percidae**			
Johnny darter	*Etheostoma nigrum*	Da	Bi In	
Mud darter**	*Etheostoma asprigene*	Da	Bi In	
Rainbow darter	*Etheostoma caeruleum*	Da In	Bi In	Sl

APPENDIX 3. (continued)

Common name	Scientific name	IBI classification[a] Taxa	Trophic status	Reproductive guild
Perches (continued)	**Percidae**			
Iowa darter	*Etheostoma exile*	Da In	Bi In	
Fantail darter	*Etheostoma flabellare*	Da He	Bi In	
Least darter***	*Etheostoma microperca*	Da In	Bi In	
Yellow perch	*Perca flavescens*		In	
Logperch	*Percina caprodes*	Da	Bi In	Sl
Gilt darter***	*Percina evides*	Da In	Bi In	Sl
Blackside darter	*Percina maculata*	Da	Bi In	Sl
Slenderhead darter	*Percina phoxocephala*	Da In	Bi In	Sl
River darter	*Percina shumardi*	Da	Bi In	Sl
Walleye	*Stizostedion vitreum*		Pi	Sl
Sauger	*Stizostedion canadense*		Pi	Sl
Crystal darter***	*Ammocrypta asprella*	Da In	Bi In	Sl
Western sand darter	*Ammocrypta clara*	Da In	Bi In	Sl
Freshwater drum	**Sciaenidae**			
Freshwater drum	*Aplodinotus grunniens*		In	

[a] Taxa- Da=darters, He=headwater, Mi=minnows, In=intolerant, To=tolerant
Trophic status- Bi=benthic invertivore, In=invertivore, Om=omnivore, Pi=piscivore
Reproductive guild- Sl=simple lithophil
* Fish species list is from Fago and Hatch (1993)
** Fish species not collected in St. Croix River basin since 1974
*** Minnesota listed special concern species
**** Minnesota listed threatened species

APPENDIX 4 - SCORING METRICS

DEFINING SCORING LINES

Ecological dose response curves (Karr and Chu 1999) were used to score each metric (fig. 4). Scoring was accomplished by drawing a horizontal line through the dose response curve so that approximately 5 percent of the observations were above the line. This line is referred to as the Maximum Species Richness (MSR) line. Four equally spaced horizontal lines were then placed below the MSR line to divide the graph into five separate sections. A score of 10 was assigned to the area of the graph immediately above the MSR line followed by a 7, 5, 2, and finally a 0 value in the lowest section of the graph. For metrics that respond negatively to disturbance the scoring process is just the opposite, with the MSR line defining the lower portion of the graph and the highest score (10) defining the portion of the graph below the MSR line.

There are a few notable exceptions to this process:

1) **Number of fish per 100 meters:** The graph was divided into two sections and assigned a score of 0 for values of 10 or less fish per 100 meters or a score of 10 for values greater than 10 fish per 100 meters. Therefore, this metric will receive a ten unless the number of fish collected at the site is extremely low.

2) **Percent DELT anomalies:** Scored a 10 if the percent occurrence of DELT anomalies was less than 1 percent, 5 if the occurrence was between 1 and 3 percent, and 0 if the percent occurrence was greater than 3 percent.

3) **Number of headwater species:** Because the maximum number of headwater species was three, this metric could not be divided 5 ways. A score of 0 was assigned if no headwater species were present, a score of 5 was assigned if one or two headwater species were present, and a score of 10 was assigned if the three or more headwater species were present.

APPENDIX 5 - ST. CROIX RIVER BASIN SAMPLING SITES

Stream Name	Sample Date	Drainage Area (mi^2)	Field Number[1]	County	Location	Latitude[2]	Longitude	IBI Score[3]	Land Use %[4]	Habitat Score [5]
Very Small Streams (<20 mi^2 drainage area)										
Trib. to Little Hanging Horn Lake	07/16/96	0.2	96SC062	Carlton	2 mi. E. of Barnum	46.49293	92.6606	56	4.12	10
ditch to Hay Creek**	07/23/96	1	96SC016	Chisago	2 mi. N.E. of North Branch	45.53867	92.9333	11	72.51	2
tributary to Burnam Creek	07/17/96	1.5	96SC044	Pine	2 mi. S. of Ellson	46.28559	92.9872	66	7.85	9
tributary to Chelsey Brook*	08/20/96	1.5	96SC051	Aitkin	Near C.S.A.H. 23, 3 mi. S.W. of Giese	46.17344	93.1756	37	3.52	11
W. Fork Redhorse Creek*	08/06/96	1.5	96SC073	Pine	@ Chengwatana State Forest	45.8573	92.7687	80	0.00	11
county ditch #7	07/08/96	2	96SC027	Chisago	1.5 mi. S. of North Branch	45.48991	92.991	43	61.84	6
tributary to Snake River	08/20/96	2.4	96SC049	Aitkin	3.5 mi. S. of McGrath	46.20026	93.2542	46	15.84	9
Squib Creek	08/08/96	2.7	96SC080	Pine	Rd. btn. S 28/33, 2.5 mi. W. of Cloverton	46.17207	92.3746	80	9.84	9
tributary to Spring Lake	07/09/96	4	96SC005	Kanabec	Near C.R. 71, 2 mi. N.E. of Mora	45.89621	93.2604	11	65.12	5
tributary to Dead Moose R	07/16/96	4	96SC036	Carlton	Rd. btn. S 27/34, 2 mi. E. of Automba	46.52159	92.9718	73	25.32	8
Wolf Creek	08/07/96	4	96SC075	Pine	2 mi. N. of Sandstone	46.16224	92.86	34	56.53	10
judicial ditch #1	06/24/98	5.1	98SC017	Pine	4 mi. N.W. of Hinckley	46.046	93.0248	43	37.44	7
Deer Creek	07/18/96	5.5	96SC054	Pine	4 mi. N.E. of Hinckley	46.05361	92.8817	57	22.95	12
Bear Creek	08/06/96	6.5	96SC068	Pine	@ C.S.A.H. 10, 4 mi. N.E. of Pine City	45.85946	92.8695	54	66.62	5
tributary to Rock Creek**	07/01/98	7.2	98SC014	Pine	In town of Rock Creek	45.75742	92.9637	70	81.93	2
tributary to Kettle River	06/24/98	7.8	98SC012	Pine	1 mi. E of Rutledge	46.2597	92.8466	42	45.29	8
Chelsey Brook	08/07/96	10.3	96SC077	Aitkin	@ S.H. 18, 1 mi. W. of Giese	46.21754	93.1302	69	4.95	7
Cane Creek	07/11/96	10.7	96SC045	Pine	@ C.S.A.H. 33, 4 mi. N. of Askov	46.24622	92.7816	60	30.13	6
E. Fork Crooked Creek*	08/08/96	11.1	96SC079	Pine	@ C.S.A.H. 32, 11 mi. E. of Askov	46.18695	92.5496	84	2.39	10
judicial ditch #4**	07/27/98	11.2	98SC006	Isanti	8 mi. SE of Cambridge	45.49891	93.0784	63	65.22	4
W. Fork Crooked Creek	07/25/96	11.3	96SC064	Pine	@ C.S.A.H. 30, 5 mi. W. of Duxbury	46.12927	92.6172	53	6.84	7
Hay Creek**	07/01/98	11.6	98SC016	Pine	9 mi. NW of Rock Creek	45.77863	93.1324	16	72.69	4
Cowan's Brook	07/09/96	12	96SC061	Aitkin	5.5 mi. S.W. of Giese	46.17407	93.2158	62	18.72	8
Spring Brook	08/07/96	12.1	96SC078	Kanabec	1 mi. E. of Mora	45.86176	93.2739	40	72.12	5
Hay Creek	08/05/96	12.4	96SC067	Pine	@ Kingsdale	46.23876	92.3095	82		9
Hay Creek	08/07/96	13	96SC076	Kanabec	@ S.H. 27, 2 mi. W. of Woodland	46.11535	93.3194	79	12.16	6
Browns Creek**	07/31/96	13.6	96SC066	Washington	@ C.R. 68, 4 mi. N.W. of Stillwater	45.10778	92.8744	19	90.13	5
Knife River	07/02/96	13.9	96SC008	Mille Lacs	C.S.A.H. 27, 5 mi. S. of Isle	46.06915	93.4677	60	52.49	5
Gillespie Brook*	07/16/96	14.5	96SC042	Carlton	Near C.R. 135, 5 mi. N. of Moose Lake	46.52108	92.792	53	10.21	10
Redhorse Creek	08/06/96	15.9	96SC072	Pine	@ Chengwatana State Forest	45.85687	92.7666	91	2.14	7
W. Branch Kettle River	09/11/96	16.5	96SC039	Carlton	Near C.S.A.H. 22, 6 mi. N. of Automba	46.60099	93.0138	11	15.06	
Snake River*	08/05/96	16.5	96SC069	Aitkin	C.S.A.H 2, 2.5 mi. E. of Pliny	46.33351	93.2102	68	4.65	12
Lower Tamarack River	08/08/96	17.2	96SC082	Pine	Rd. btn. S 28/33, 8.5 mi. S.E. of Bruno	46.26003	92.4966	79	2.36	9

APPENDIX 5. (continued)

Stream Name	Sample Date	Drainage Area (mi²)	Field Number[1]	County	Location	Latitude[2]	Longitude	IBI Score[3]	Land Use %[4]	Habitat Score [5]
Small Streams (20-54 mi² drainage area)										
Little Ann River*	07/03/96	20	96SC004	Kanabec	@ S.H. 47, 4 mi. N. of Ann Lake	45.96881	93.4282	86	3.60	12
Bear Creek	09/24/96	21.8	96SC055	Pine	Near C.S.A.H. 30, 3 mi. E. of Sandstone	46.11138	92.7909	49	52.38	
S. Branch Grindstone River	07/24/96	26.5	96SC063	Pine	Rd. btn. S 17/18, 4 mi. N.W. of Hinckley	46.03819	93.0345	58	35.48	10
E. Fork Crooked Creek*	09/11/96	27.7	96SC058	Pine	4 mi. S.W. of Duxbury	46.07914	92.555	73	4.77	11
Mission Creek**	08/06/96	29.3	96SC013	Pine	1 mi. S.W. of Beroun	45.89328	92.9803	8	60.47	11
Mission Creek	08/26/96	29.3	96SC013	Pine	1 mi. S.W. of Beroun	45.89328	92.9803	13	60.47	8
Mission Creek	06/26/97	29.3	96SC013	Pine	1 mi. S.W. of Beroun	45.89328	92.9803	24	60.47	7
Mission Creek	07/27/98	29.3	96SC013	Pine	1 mi. S.W. of Beroun	45.89328	92.9803	32	60.47	8
Birch Creek	08/07/96	29.3	96SC074	Pine	Rd. btn. S 21/22, 2 mi. W. of Denham	46.36694	92.9919	68	13.51	10
Birch Creek	08/29/96	29.3	96SC074	Pine	Rd. btn. S 21/22, 2 mi. W. of Denham	46.36694	92.9919	68	13.51	10
Birch Creek	08/13/97	29.3	96SC074	Pine	Rd. btn. S 21/22, 2 mi. W. of Denham	46.36697	92.9924	70	13.51	11
Birch Creek	06/24/98	29.3	96SC074	Pine	Rd. btn. S 21/22, 2 mi. W. of Denham	46.36694	92.9919	68	13.51	9
Mud Creek**	06/24/98	29.6	98SC018	Kanabec	Upstream of SNH 23 on SE of Quamba	45.91266	93.1757	38	47.46	6
McDermott Creek*	09/12/96	30.5	96SC038	Pine	4.5 mi. N.W. of Cloverton	46.20675	92.3947	68	0.04	10
Birch Creek	06/24/98	33.2	98SC020	Pine	Upstream of CSAH 40 in town of Denham	46.36224	92.9508	68	18.09	6
Rush Creek**	06/23/98	35.3	98SC001	Chisago	1.5 mi W of Rush City	45.68372	93.0137	41	58.35	6
Rush Creek	08/13/98	35.3	98SC001	Chisago	1.5 mi W of Rush City	45.68372	93.0137	43	58.35	8
Willow River*	08/08/96	36.6	96SC083	Pine	@ C.S.A.H. 48, 1 mi. N.W. of Durquette	46.38127	92.5722	68	8.19	10
Groundhouse River*	08/05/96	42.4	96SC070	Kanabec	@ Rum River State Forest	45.88155	93.5069	84	8.46	10
Rush Creek**	06/23/98	43.3	98SC002	Chisago	2 mi E of Rush City	45.6854	92.9542	53	63.50	8
Bear Creek	07/18/96	43.5	96SC034	Pine	@ S.H. 48, @ Cloverdale	46.01359	92.7449	87	34.19	9
Pokegama Creek	07/02/98	44.4	98SC015	Pine	3.5 mi. W. of Beroun	45.91702	93.0213	74	44.62	10
Rush Creek	06/28/96	45.9	96SC015	Chisago	I 35 @ Rush City	45.67968	92.9891	17	59.68	10
Rush Creek**	06/23/98	47.2	98SC003	Chisago	8 mi E. of Rush City	45.68958	92.9344	66	65.00	7
Goose Creek	08/09/96	47.5	96SC084	Chisago	@ C.S.A.H. 30 in Harris	45.58751	92.9764	58	53.53	9
Goose Creek	08/19/96	47.5	96SC084	Chisago	@ C.S.A.H. 30 in Harris	45.58751	92.9764	51	53.53	10
Goose Creek	06/16/97	47.5	96SC084	Chisago	@ C.S.A.H. 30 in Harris	45.58812	92.9761	60	53.53	10
Goose Creek	06/16/98	47.5	96SC084	Chisago	@ C.S.A.H. 30 in Harris	45.5881	92.9761	79	53.53	9
Split Rock River	08/28/96	50.1	96SC086	Carlton	C.S.A.H. 17, 9 mi. W. of Moose Lake	46.44727	92.9504	85	25.39	9
S. Fork Groundhouse River	07/06/98	51.2	98SC011	Kanabec	4 mi. SE of Ogilvie	45.78992	93.3887	80	61.46	10
Rush Creek	08/09/96	52.3	96SC081	Chisago	@ C.S.A.H. 5, 2 mi. E. of Rush City	45.67386	92.9112	68	64.60	8
Rush Creek	08/19/96	52.3	96SC081	Chisago	@ C.S.A.H. 5, 2 mi. E. of Rush City	45.67386	92.9112	57	64.60	8
Rush Creek	06/26/97	52.3	96SC081	Chisago	@ C.S.A.H. 5, 2 mi. E. of Rush City	45.67386	92.9112	48	64.60	8
Rush Creek	06/23/98	52.3	96SC081	Chisago	@ C.S.A.H. 5, 2 mi. E. of Rush City	45.67386	92.9112	61	64.60	8
Mud Creek	07/19/96	52.7	96SC011	Pine	Near C.S.A.H. 11, 1 mi. W. of Henriette	45.87203	93.1351	63	49.04	9

APPENDIX 5. (continued)

Stream Name	Sample Date	Drainage Area (mi^2)	Field Number[1]	County	Location	Latitude[2]	Longitude	IBI Score[3]	Land Use %[4]	Habitat Score [5]
Moderate size streams (55-270 mi^2 drainage area)										
Rush Creek**	06/23/98	56.6	98SC004	Chisago	3 mi S.E. of Rush City	45.65458	92.9007	69	64.97	9
Groundhouse River*	06/27/96	58.3	96SC017	Kanabec	Near C.R. 53, 2 mi. N.W. of Ogilvie	45.84109	93.4474	60	13.20	10
Groundhouse River	07/02/98	60.9	98SC005	Kanabec	Upstream of SNH 23, .1 mi E of Ogilvie	45.83268	93.4096	44	15.32	8
N. Branch Sunrise River**	07/27/98	61	98SC008	Chisago	5 mi E of North Branch	45.51322	92.9638	33	63.83	8
N. Branch Sunrise River	06/18/98	61	98SC008	Chisago	5 mi E of North Branch	45.51322	92.9638	36	63.83	8
Rock Creek	07/31/96	64.6	96SC022	Chisago	Near C.S.A.H. 3, 3 mi. N.E. of Rush City	45.7189	92.9107	63	80.08	11
Ann River	09/03/96	65.2	96SC021	Kanabec	Near C.S.A.H. 12, 2 mi. W. of Mora	45.87219	93.3436	65	20.43	10
Snake River	07/16/98	65.2	96SC050	Aitkin	Near C.S.A.H. 2, 1 mi. S.W. of Pliny	46.32376	93.2762	43	6.66	2
Snake River	07/26/96	65.2	96SC050	Aitkin	Near C.S.A.H. 2, 1 mi. S.W. of Pliny	46.32376	93.2762	49	6.66	3
Snake River	06/27/97	65.2	96SC050	Aitkin	Near C.S.A.H. 2, 1 mi. S.W. of Pliny	46.32376	93.2762	41	6.66	5
Snake River	08/28/96	65.2	96SC050	Aitkin	Near C.S.A.H. 2, 1 mi. S.W. of Pliny	46.32405	93.2765	57	6.66	5
Ann River	07/02/98	72.3	98SC019	Kanabec	4 mi. SW of Mora	45.84157	93.3309	44	25.16	8
Kettle River	08/27/96	73.4	96SC085	Carlton	@ C.S.A.H. 14, 6 mi. N. of Kettle River	46.56601	92.8802	69	20.23	9
N. Branch Sunrise River**	08/19/96	74.5	96SC025	Chisago	S.H. 95, 4 mi. E. of North Branch	45.51219	92.8928	43	64.01	8
Goose Creek**	07/30/96	76.5	96SC023	Chisago	@ Wild River State Park	45.59389	92.8998	39	57.02	9
Knife River	06/25/96	76.8	96SC006	Kanabec	Near C.S.A.H. 15, 6 mi. S.W. of Warman	46.03528	93.38	53	21.32	10
Moose Horn River	08/29/96	77.4	96SC087	Carlton	1 mi. N. of Barnum	46.5137	92.6985	71	17.17	9
Grindstone River	06/22/98	78.3	98SC009	Pine	N. side of CR 140, 3 mi. E. of Hinckley	46.01487	92.924	62	40.19	10
Grindstone River	06/22/98	79.4	98SC010	Pine	N. side of CR 140, 1 mi. E. of Hinckley	46.01733	92.9062	67	42.97	9
Grindstone River	08/17/98	80.4	98SC013	Pine	2 mi. E of Hinckley	46.01062	92.8868	78	43.52	9
Upper Tamarack River	08/13/96	93.4	96SC037	Pine	Primitive Rd., 2 mi. S.E. of Cloverton	46.14239	92.2942	74		7
Upper Tamarack River	08/27/96	93.4	96SC037	Pine	Primitive Rd., 2 mi. S.E. of Cloverton	46.14239	92.2942	74		7
Upper Tamarack River	07/28/97	93.4	96SC037	Pine	Primitive Rd., 2 mi. S.E. of Cloverton	46.14237	92.2942	68		8
Upper Tamarack River	06/25/98	93.4	96SC037	Pine	Primitive Rd., 2 mi. S.E. of Cloverton	46.14239	92.2942	63		7
Knife River	09/18/96	107.6	96SC097	Kanabec	@ C.R. 77, 3 mi. N. of Mora	45.92043	93.3082	80	26.65	9
Pine River	09/04/96	109.9	96SC043	Pine	3 mi. N.W. of Rutledge	46.28046	92.9279	78	24.69	11
Sunrise River**	07/29/96	114.6	96SC024	Chisago	Near C.R. 84, 1 mi. E. of Wyoming	45.34659	92.9589	22	59.87	8
Lower Tamarack River*	08/14/96	128	96SC056	Pine	@ St. Croix State Forest	46.07938	92.4277	78	2.89	11
Sand Creek	09/05/96	138.5	96SC090	Pine	@ St. Croix State Park	45.95387	92.6669	87	20.83	10
Snake River*	07/25/96	155.9	96SC052	Aitkin	Near S.H. 18, 2 mi. S.E. of McGrath	46.22278	93.2419	69	9.84	9
Lower Tamarack River*	08/14/96	182.3	96SC029	Pine	@ St. Croix State Forest	46.05412	92.3962	97	3.06	9
Kettle River	08/21/96	187	96SC040	Carlton	5 mi. W. of Moose Lake	46.45578	92.8735	82	23.01	9

APPENDIX 5. (continued)

Stream Name	Sample Date	Drainage Area (mi^2)	Field Number[1]	County	Location	Latitude[2]	Longitude	IBI Score[3]	Land Use %[4]	Habitat Score[5]
					Moderate size streams (continued)					
Snake River	07/24/96	258.3	96SC002	Kanabec	Near C.S.A.H. 24, 3 mi. E. of Warman	46.06192	93.2197	85	10.39	7
Snake River*	07/24/96	258.3	96SC003	Kanabec	Near C.S.A.H 24, 3 mi. E. of Warman	46.06017	93.2204	89	10.39	8
Sunrise River	06/23/97	268	96SC065	Chisago	Downstream of Kost Dam County Park	45.48178	92.8741	66	61.64	11
Sunrise River	08/26/96	268	96SC065	Chisago	Downstream of Kost Dam County Park	45.48178	92.8741	68	61.64	12
Sunrise River	06/17/98	268	96SC065	Chisago	Downstream of Kost Dam County Park	45.48179	92.8741	67	61.64	10
Sunrise River	07/30/96	268	96SC065	Chisago	Downstream of Kost Dam County Park	45.48178	92.8741	68	61.64	9
					Rivers (270 mi^2 drainage area)					
Kettle River	08/21/96	296.2	96SC047	Pine	3 mi. N.W. of Sturgeon Lake	46.39804	92.8796	82	22.78	
Snake River	09/25/96	305.7	96SC007	Kanabec	S.W. of Warman	46.0179	93.2399	68	13.06	
Kettle River	08/20/96	348.5	96SC046	Pine	Near C.S.A.H. 52, 3 mi. N. of Willow River	46.36692	92.8609	84	23.65	
Kettle River	08/22/96	493.6	96SC048	Pine	2.5 mi. N. of Willow River	46.35389	92.8398	70	24.28	
Snake River	06/26/96	545	96SC018	Kanabec	3.5 mi. S. of Mora	45.81261	93.2799	78	22.60	
Snake River	07/09/96	803.2	96SC019	Kanabec	2 mi. W. of Grasston	45.79363	93.1802	78	31.10	
Snake River	07/17/96	824.2	96SC010	Pine	2 mi. E. of Grasston	45.79031	93.1069	63	32.10	
Kettle River	07/25/96	903.8	96SC053	Pine	4 mi. N.E. of Hinckley	46.03673	92.872	51	26.16	
Snake River	08/01/96	978.8	96SC012	Pine	4 mi. E. of Pine City	45.84358	92.8896	78	35.90	
Kettle River	07/24/96	1010	96SC032	Pine	7 mi. S.E. of Hinckley	45.96045	92.8234	66	27.68	
Kettle River	08/15/96	1049.9	96SC033	Pine	@ Kennedy Brook in St. Croix State Park	45.90154	92.731	86	27.63	
St. Croix River	09/17/96	2236	96SC096	Pine	@ S.H. 48, E. of Hinckley	46.00894	92.4438	89		
St. Croix River	09/04/96	2680	96SC089	Pine	@ St. Croix State Park	45.95089	92.5563	57		
St. Croix River	09/17/96	2680	96SC089	Pine	@ St. Croix State Park	45.95089	92.5563	79		
St. Croix River	09/15/97	2680	96SC089	Pine	@ St. Croix State Park	45.95089	92.5563	76		
St. Croix River	08/13/98	2680	96SC089	Pine	@ St. Croix State Park	45.95089	92.5563	72		
St. Croix River	09/19/96	2886	96SC030	Pine	Kettle River Slough	45.88046	92.7294	92		
St. Croix River	09/12/96	4863	96SC094	Pine	@ S.H. 70, S.E. of Pine City	45.77148	92.7808	83		
St. Croix River	09/16/96	5120	96SC095	Chisago	E. of Rush City @ Ferry Crossing	45.68222	92.8764	91		
St. Croix River	09/23/96	5120	96SC095	Chisago	E. of Rush City @ Ferry Crossing	45.68222	92.8764	84		
St. Croix River	08/21/97	5120	96SC095	Chisago	E. of Rush City @ Ferry Crossing	45.68199	92.8773	83		

APPENDIX 5. (continued)

Stream Name	Sample Date	Drainage Area (mi²)	Field Number[1]	County	Location	Latitude[2]	Longitude	IBI Score[3]	Land Use %[4]	Habitat Score[5]
					Rivers (continued)					
St. Croix River	10/08/98	5120	96SC095	Chisago	E. of Rush City @ Ferry Crossing	45.68222	92.8764	92		
St. Croix River	09/03/96	5635	96SC088	Chisago	Downstream of Sunrise River mouth	45.56667	92.8548	84		
St. Croix River	09/18/96	5635	96SC088	Chisago	Downstream of Sunrise River mouth	45.56667	92.8548	81		
St. Croix River	07/31/97	5635	96SC088	Chisago	Downstream of Sunrise River mouth	45.56667	92.8548	68		
St. Croix River	08/27/98	5635	96SC088	Chisago	Downstream of Sunrise River mouth	45.56667	92.8548	97		
St. Croix River	08/02/96	6240	96SC028	Washington	McLeods Slough	45.26937	92.763	73		
St. Croix River	09/06/96	6240	96SC091	Washington	Downstream of Marine on St. Croix	45.18148	92.7621	81		

[1] Field number assigned to each station to designate a unique sampling location.
[2] Latitude and longitude are formatted in WGS84 decimal degrees.
[3] IBI score is the overall IBI score assigned to the site. Scores range from 0 (lowest biological integrity) to 100 (highest biological integrity).
[4] Land use expressed as a percent of the watershed upstream of the sampling location that has been altered by humans. It includes disturbance from agricultural, residential, urban, and mining land usage.
[5] Habitat score is a ranking of habitat based on 6 metrics (see Table 6). Scores range from 0 (poorest fish habitat) to 12 (best fish habitat).
* Sites that were designated as being of excellent quality based on land use and habitat.
** Sites that were designated as poor quality based on land use and habitat.

APPENDIX 6 - IBI VARIABILITY

We sampled 11 sites twice within a single sampling season to examine the variability of IBI scores within a single year. The sites ranged in size (29-5635 mi^2 drainage area) and level of disturbance (7 to 65% watershed disturbance). The repeat sampling events occurred from June through September. The repeat samples were taken 7 to 51 days (mean=22) from the initial visit. IBI scores from each site visit were not significantly different from each other (paired t test, p=.516). The mean difference in IBI scores was 4.18 (C.V.=.806). At the 5 least disturbed sites the mean difference of IBI scores was lower (mean = 3.12, C.V.=1.022).

We conducted repeat sampling at 10 of the above mentioned sites during a 4 year period to examine the variability of IBI scoring between years. We placed each site into one of 3 categories to examine the variance associated with stream size and human disturbance (table 10). The categories included streams (<270 mi^2 drainage area) with < 15% watershed disturbance, streams with > 50% watershed disturbance, and rivers (>270 mi^2 drainage area).

As expected, sites with little human disturbance experienced the least change in IBI scoring over a 4 year period. IBI scores in streams with a high level of human disturbance varied more with the exception of the Sunrise River site (table 10). At this site, wetlands within the watershed and an impoundment upstream of the site may have actually stabilized the stream by regulating flow and acting as a sediment trap. We attribute the relatively large variation in IBI scores at streams with a high degree of watershed disturbance to disturbance within the watershed. Farming practices, agricultural pesticide usage, the amount of municipal or industrial effluent discharged into the watershed, and countless other land use practices that occur within a watershed may vary temporally and spatially. This may cause fish communities in streams with a high degree of human disturbance to undergo periods of stress followed by periods of recovery. Fish community structure may be more variable because the strategies (physiological or behavioral) fish have developed to adapt to natural sources of stress (eg. floods, temperature extremes, etc.) may not sufficiently compensate for higher levels of human disturbance (Fore et al 1994).

IBI scores from the three river sites were more variable than the stream sites. This variability may result from natural or human induced changes in disturbance or be an artifact of our sampling methodology. Rankin and Yoder (1990) attributed higher coefficients of variation in IBI scores in large streams to a greater degree of sampling error. It is interesting to note that the two lower sites (St. Croix at the Ferry Landing and St. Croix below the Sunrise River) had the highest variability, whereas the upper most site (St. Croix River at St. Croix State Park) was much more consistent from year to year. This suggests that the lower sites may be too large to sample effectively using the sampling protocol we have used for rivers in other basins such as the Minnesota and Red River Basins. However, it is also possible that the difference in IBI scores between sampling periods is related to natural or human induced change. Although the St. Croix River is one of the most pristine rivers in Minnesota its watershed is not undisturbed, particularly in the lower reaches.

The replicate sites did not include very small streams (0-20 mi^2 drainage area). The within and among year variability in IBI scores may have been higher if streams in this size class had been included. Future work should focus on obtaining an adequate number of stations in each of the 4 size classes to examine within and among year variation in IBI scores.

By collecting samples at different times at sites experiencing no new human influences Karr et al. (1986) were able to detect five quality classes ranging from excellent to very poor within the IBI scoring. The width of each class is an indication of the level of confidence the user should have in an IBI score. We were able to classify streams with drainage areas < 270 mi^2 into 10 quality classes and rivers with drainage areas > 270 mi^2 into 3 quality classes. For streams with drainage areas less than 270 mi^2 a difference of 10 IBI points represents a statistically valid change in integrity. A difference of 30 IBI points represents a statistically significant change in integrity for rivers. The assignment of quality classes is based on the assumption there is no significant difference in the level of disturbance between sampling periods. This may not be a valid assumption for the lower portion of the St. Croix River.

Table 10. IBI scores and summary statistics for replicate samples taken over a 4 year period (1996-1999). Sites have been grouped into stream (<270 mi^2 drainage area) and river (>270 mi^2 drainage area) classes. The streams are further divided into sites with little watershed disturbance (<15%) and high watershed disturbance (>15%).

	Streams <270 mi^2							Rivers (>270 mi^2)		
	IBI score for sites with little watershed disturbance			IBI score for sites with high watershed disturbance				disturbance varies from upstream (lowest) to downstream (highest)		
Year	Birch Creek	Snake River	Tamarack River	Mission Creek	Rush Creek	Goose Creek	Sunrise River	St. Croix State Park	St. Croix Ferry	St. Croix Sunrise
1996	68	49	74	8	68	58	71	79	91	84
1997	70	41	68	24	48	60	68	76	83	68
1998	68	43	63	32	61	79	72	72	92	97
1999	66	51	66	18	42	73	68	73	67	62
mean score	68	46	68	21	54	68	70	75	83	78
range	4	10	11	24	26	21	4	7	25	35
C.V.	.024	.103	.069	.493	.217	.150	.030	.042	.139	.204
% disturbance	13.51	6.66	Est. <5	60.47	64.6	53.53	61.64	-	-	-