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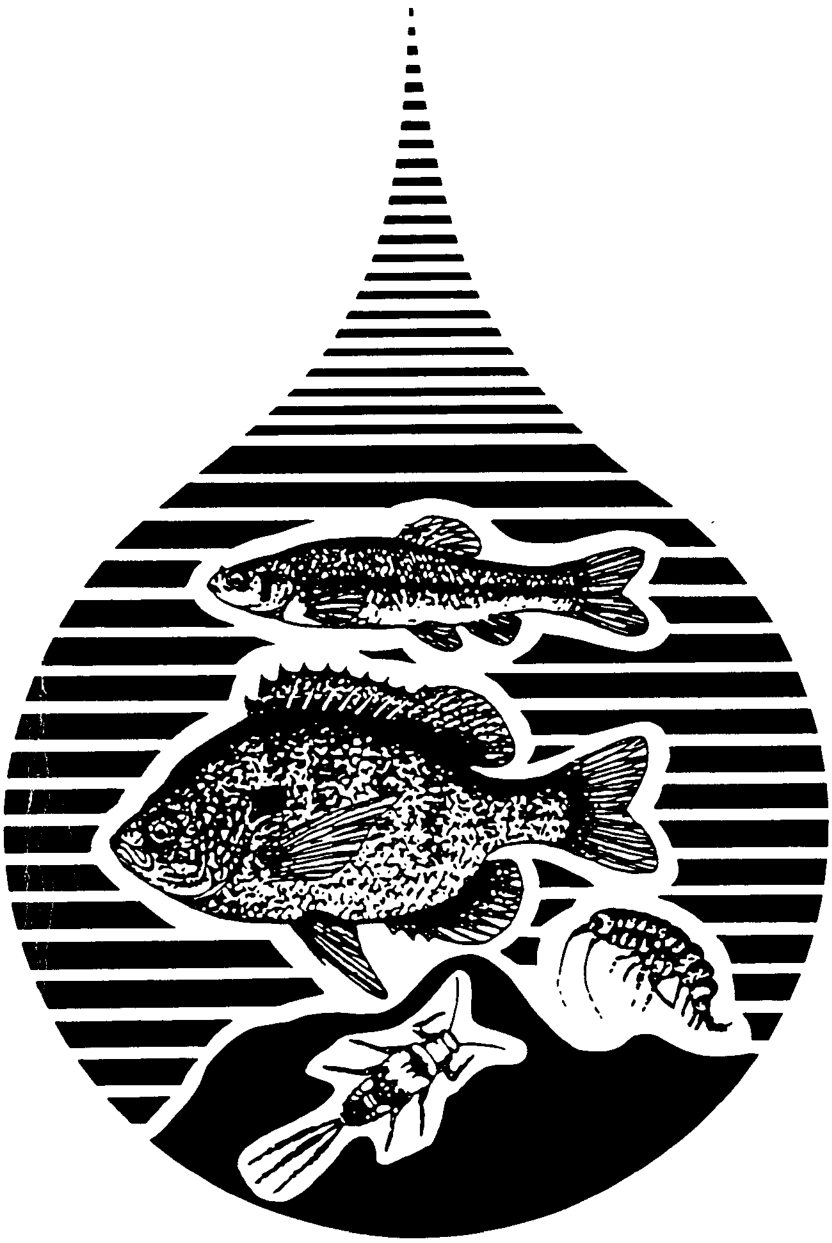

PROCEEDINGS OF THE 1990 MIDWEST

POLLUTION CONTROL BIOLOGISTS MEETING

held in

CHICAGO, ILLINOIS

April 10-13, 1990

Edited by:

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Sponsored by:

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Assessment and Watershed Protection Division
Washington, D.C. 20460

U.S. Environmental Protection Agency, Region V
Environmental Sciences Division
Instream Biocriteria and Ecological Assessment Committee
Chicago, IL 60604

The 1990 MPCB Meeting was dedicated to

James L. Plafkin
(1990)

The 1990 MPCB Proceedings is dedicated to

Michael J. Glorioso
(1980)

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FORWARD

A Historical Perspective on Regulatory Biology

After several years of debate, the United States Environmental Protection Agency (USEPA) has thoroughly recognized the benefit of using biological survey data to assist with the Agency's regulatory decisions for surface waters. The myriad of uses that biological survey data can provide were discussed in the recent "Biocriteria Program Guidance Document" (USEPA 1990a), and examples were presented in the "Biocriteria Development by States" (USEPA 1990b).

The use of biological survey data to assess the health of our rivers and streams was prompted by some early work conducted by the Illinois State Natural History Survey almost a century ago. This paper specifically reviews the circumstances regarding that first biological survey, and the evolution of more rigorous and objective assessment end-points. Washington (1984) provided a comprehensive review of the history and application of biotic indices, and the information presented here is intended to complement Washington's work.

"Dilution is the Solution"

In 1848, the Illinois and Michigan Canal (I&MC) was opened crossing the continental divide between the Great Lakes drainage system and the Mississippi River drainage. The I&MC connected the South Fork of the South Branch of the Chicago River near the tanneries and packinghouses of the famous Chicago Stockyards with the Upper Illinois River at LaSalle (Figure 1; U.S. Engineers Office 1924). Initially intended for navigation, the I&MC soon became an obvious outlet for the sewage created by a growing population and industrial base, and, beginning in 1869, an average of 167 cfs (maximum 400 cfs) of water was pumped from the Chicago River into the sluggish I&MC, reversing the normal flow of the South Branch into the Illinois River. More direct gravity flow from the Chicago River was provided in 1871 by deepening the summit of the canal, and, in 1884, an additional 1000 cfs was pumped into the river to facilitate the flow. The growth of Chicago's north side also required additional pumping to increase the southward flow of the water and 400 cfs was pumped from Lake Michigan to the Chicago River mainstem. The inadequacy of the I&MC to solve the city's sewage needs, and to keep the city's water intake supply in Lake Michigan from additional contamination, became apparent.

The Sanitary District of Chicago was created by the State of Illinois in 1889 to plan for the city's additional sewerage requirements. In 1892, construction of the 28-mile long Chicago Drainage Canal began adjacent to the much smaller I&MC, connecting the South Branch of the Chicago River (near the West Fork) with the Des Plaines River at Lockport. The Canal, opened in January 1900, was designed to divert up to 14,000 cfs of water from Lake Michigan, resulting in a reversal of the flow of the Chicago River and sewage flowing away from the drinking water supply.

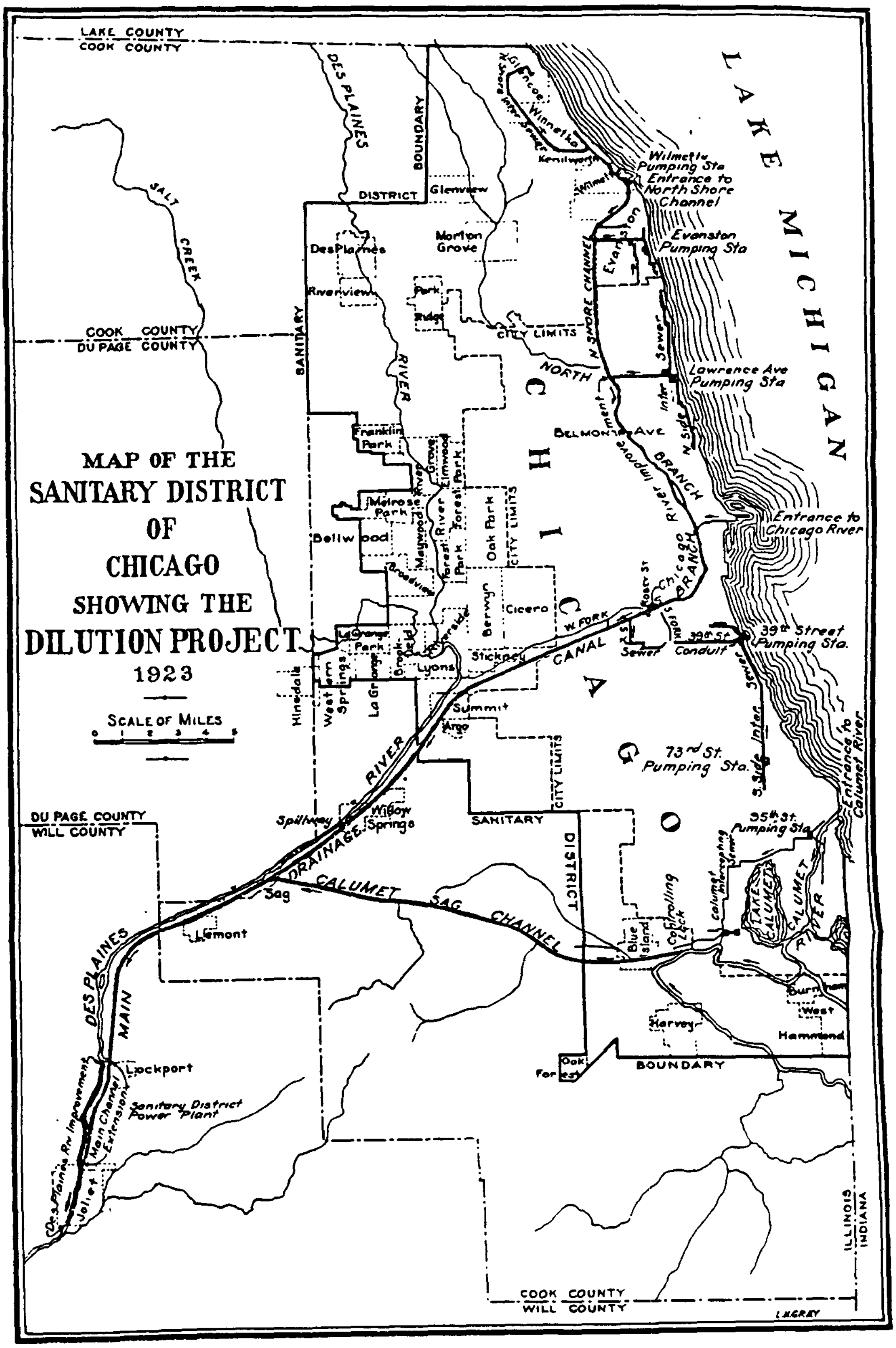

Figure 1 (from US Engineers Office 1924)

The population and industrial growth of the city required several pumping stations to be added to adequately "flush" the raw sewage downstream. Construction of the North Shore Channel from 1908-1910 which connected Lake Michigan with the Chicago River north of the city added about 1000 cfs. Sewers were soon built along the lakeshore communities eventually diverting all sewage in the Sanitary District from Lake Michigan into the Des Plaines River basin. The expansion of the far south side district resulted in the construction of the Calumet-Sag Channel, which was completed in 1922. The channel connected the Little Calumet River with the Chicago Drainage Canal 22 miles downstream, diverting an additional 500 cfs from Lake Michigan.

"Poisoning the Mississippi?"

As one can imagine, this diversion of raw sewage from the Chicago River system into the Des Plaines River did not please the downstream communities, although Chicago's drinking water supply was no longer contaminated. Before the Chicago Drainage Canal was officially opened, the State of Missouri brought suit against the State of Illinois and the Chicago Sanitary District on January 17, 1900 seeking an injunction from opening the Canal (Leighton 1907). The suit charged that the drainage of the "sewage matter from nearly the whole City of Chicago and a portion of Cook County" would "poison the waters of the Mississippi and render them unfit for domestic uses". The defendants claimed, however, that the Mississippi River water would actually be cleaner due to the dilution water from Lake Michigan (approximately 9:1).

Due to the interstate nature of this dispute which affected State sovereignty, the United States Supremè Court became involved. After years of gathering facts and assembling the nation's expert water quality scientists, engineers, and sanitarians, the testimony provided by these experts was deemed to be overwhelmingly in support of the State and Sanitary District's position. On February 19, 1906, the United States Supreme Court concluded that not only did the State of Missouri not prove it's case, but that the Chicago Drainage Canal (i.e. Sanitary and Ship Canal) substantially improved the quality of the nearby Illinois River (Leighton 1907).

"Bureaucracy in the 1800's"

Today's bureaucracy is painted with pictures of government required permits, licenses, and unnecessary delays, but such was the case even in the 1800's. In fact, a crucial permit was not obtained related to the building of the Chicago Drainage Canal because it was not thought to be necessary (U.S Engineer Office 1924).

A primary concern in the late 1800's was the defense of this country, under the authority of the War Department (now Department of Defense). One of the main mechanisms for defense mobilization was through navigable waterways, and their maintenance to serve in that capacity was of great importance. Maintenance of all navigable waterways in this country was the primary impetus for the Rivers and

Harbors Appropriation Act of 1899 (30 Stat. 1151 1899). Before any major alterations of a waterway were to be conducted, a Federal permit was required, but the Chicago Sanitary District did not apply for the permit until 1896, four years after construction was begun. The Federal government conceded the permit later that year, but stated that "the authority shall not be interpreted as approval of the plans of the Sanitary District of Chicago to introduce a current into the Chicago River." The Federal government also issued a temporary and revocable permit to open the canal in 1899 and eventually modified the permit in 1903 restricting the flow through the South Branch of the Chicago River to a maximum of 5833 cfs in winter when navigation is closed, and a maximum of 4167 cfs the rest of the year.

"Dilution was NOT the Solution"

Beginning in 1907, the Sanitary District made several permit applications for increasing the allowable flow in the river system through the diversion of water from Lake Michigan. Each application was denied by the War Department which stated in 1907 that diversion greater than the values permitted in 1903 would not be allowed. Despite the denial of Federal permits and even a Federal Court suit seeking to enjoin the District from constructing the Calumet-Sag Channel which would divert even more water, the construction of the North Shore and Calumet-Sag Channels continued, and was soon completed (U.S. Engineer Office 1924).

Arguments for both sides in this law suit were heard in Federal Court beginning in 1915, and in 1920, the United States District Court gave an oral opinion that the United States government was entitled to an injunction. After a motion of reconsideration was filed by the Sanitary District, the District Court issued a formal decree in 1923 supporting the injunction sought by the United States government. After an appeal to the Supreme Court, the decision made by the District Court was upheld on January 5, 1925.

The Supreme Court decision affected not only the illegal diversion of waters from Lake Michigan, but also the mitigation of adverse effects and actual damage caused by the diversion on the ecology of the Illinois River. To comply with both the need to dispose of sewage without additional diversion of water from Lake Michigan and to improve the poor ecological conditions of the Illinois River resulting from the now illegal diversion, the Sanitary District of Chicago committed over $125 million to construct/improve the sewer system and build the first wastewater treatment plants in the area. It was demonstrated that dilution alone no longer provided adequate protection of the streams and rivers and the era of physical and biological wastewater treatment arrived (Sanitary District of Chicago 1925).

"Early Biological Surveys"

It is likely that the immediate outcome of the law suits filed against the Sanitary District would not have adequately addressed the impacts on the Illinois River if it had not been for the work of Stephen Forbes, the Director of the Illinois State Laboratory of Natural History. In fact, the District Engineer from the U.S. Engineer Office documented the damage to the Illinois River based upon the work conducted and initiated by Forbes, and included that information in a 1924 report submitted to the War Department supporting the Federal government's case before the Supreme Court (U.S. Engineer Office 1924).

Forbes opened a permanent field biology station on the Illinois River in 1894 to document the effects on the stream biology as a result of the opening of the Chicago Drainage Canal in 1900. The planning for that station included investigating not only the effects due to periodic flooding of the river due to the increased flow, but also the direct effects from the pollution added from the Canal (Forbes 1928).

Between 1894 and 1899, Kofoid (1908) studied the river's plankton populations, life histories and how they were affected by environmental factors including sewage. This baseline study was later used by Forbes and Richardson (1913) and Purdy (1930) to document the river's assimilative capacity and decline of the plankton populations due to the opening of the Chicago Drainage Canal

After increasing the number of monitoring stations on the river, Stephen Forbes and Robert Richardson (1913) published their first report on the conditions of the Illinois River. They defined the degradation via pollutional zones (septic, polluted, contaminate, and clean water), similar to that of Kolkwitz and Marsson's (1908) Saprobic Index. However, the Saprobic Index was based upon bacteria and protozoa while Forbes and Richardson's zones were based on water chemistry, plankton, benthic macroinvertebrate and fish populations. Their surveys conducted between 1909 and 1911 documented 107 miles of water below the mouth of the Chicago Drainage Canal to be polluted before recovery fully occurred.

"Development of the First 'Biotic Index'"

In 1921(a), Richardson found 146 miles of the river to be polluted based on their work conducted between 1913 and 1915 and in 1920 concluded that 226 miles of the Illinois River was now polluted with 146 miles of near anoxic conditions (1921b). Based on their data collected over seven years, they reported between 8 and 16 additional river miles per year were classified as polluted. Richardson began to rely heavily on defining pollution zones based on pollution tolerances for the biota, focusing heavily on the benthic macroinvertebrate community due to their role in the food chain.

The last report on the pollution biology of the Illinois River conducted by Richardson was published in 1928, and proved to be the predecessor to the more recent biotic indices. The study added data collected between 1924 and 1925 and also marked the change of responsibility for the river surveys to the State Water Survey. In this classic report, he detected shifts in water quality based on observing the benthos alone, although he preferred to also use chemical data to better define the pollutional zones. He further refined the pollutional zones (called septic, pollutional, subpollutional, and clean water; Richardson 1925) based on "index values" of the benthos which used specific taxa as indicators.

"Development of Biological Assessment Methods"

Based on the work conducted by Forbes and Richardson, the use of benthic macroinvertebrates as indicator organisms was founded and the concept of biotic indices introduced. Although the work was similar in concept to the "Saprobic Index" developed by Kolkwitz and Marsson (1908), Forbes and Richardson used organisms at higher trophic levels. The work conducted by Forbes and Richardson forms the basis for some of our current regulatory biology programs supported by USEPA.

Richardson (1925,1928) decided that numerical abundances of each index group was not as significant as their relative abundances and overall occurrences. For instance, he reported the number of pollution tolerant Tubificid worms in the river to range from under 1000 to over 350,000 per square yard in pollutional zones, and Chironomids to range from zero to over 1,000 per square yard. Richardson also reported seasonal and habitat changes as responsible for much of the numeric variability at a given site, supporting the use of the occurrence of a species as the better index measure.

Wright and Tidd (1933) actually applied the numerical abundance of oligochaetes to assess the degree of pollution. They reported values of less than 1000 m^{-2} as indicating negligible pollution, between 1000-5000 m^{-2} as mild pollution, and over 5000 m^{-2} was severe pollution (Washington 1984). Washington felt this work was the "original index", apparently unaware of Richardson's earlier work with "index values" for benthos. Ruth Patrick (1950) developed a "histogram" based upon seven taxonomic groups and assigned stream classes of healthy, semi-healthy, polluted, very polluted, and atypical based upon the comparison of predominance of three of the taxonomic groups with the other four.

"Biotic and Diversity Indices"

It took several more years for an improved assessment end-point to be introduced. In 1955, Beck published a biotic index which produced a numerical end-point that could more easily be interpreted by biologists, as well as the engineering and management dominated discipline. Beck's index was based upon two classes of benthos: intolerant and facultative, assigned a weight value of 2 and 1, respectively. Therefore, the higher the index value, the healthier the stream is assumed to be. In the next two decades Washington reported several biotic indices, but Chutter's (1972) biotic index ushered in a new era for these indices.

Chutter studied South African streams and assigned specific tolerance values ranging from 0 to 10 for various taxa which accounted for both the number of individuals and the number of taxa. The results were also presented in a 0 to 10 fashion with an average biotic index value of 0-2 regarded as clean-water, 2-4 slightly enriched, 4-7 enriched, and 7-10 polluted. This index was the predecessor to the widely used Hilsenhoff Biotic Index (1977) in the midwestern United States which was initially based on a 0-5 scale which included many more taxa than Chutter's.

Hilsenhoff's index was based on taxa from Wisconsin using genus/species classifications of the aquatic insects. Hilsenhoff updated his index in 1982 and 1987 to revise the index values and include new taxa, and in 1988 he developed a very popular family-level biotic index.

The development and use of diversity indices for water pollution assessment was thoroughly reviewed by Washington (1984). Essentially, the use of diversity as an optimal measure of stream community response has been widely used since the 1960's, but the theoretical diversity which is based on calculating the evenness of the number of individuals among the assembled taxa was not developed, nor intended, for the application to natural systems.

One of the first diversity indices based on information was published by Shannon (1949) as H'. Washington (1984) stated that it was eventually termed the Shannon-Weiner Index because Weiner (1948) independently published a similar measure. Washington further explains that the confusion with erroneously calling the index the Shannon-Weaver index began when Shannon published his work in a book coauthored by Weaver.

Possibly the first use of diversity indices for assessing water quality, particularly the Shannon-Wiener Index, was by Wilhm and Dorris (1966) and further explained by Wilhm (1967) who described the ranges of H' associated with clean, moderately polluted, and substantially polluted streams.

"New Assessment End-Points"

The major asset of both diversity and biotic indices is that they both reduced the relatively complex interactions and pollution responses of an aquatic community into a single number for water quality management purposes. However, neither of these indices were successful in describing the overall "health" of the aquatic communities under a variety of conditions. Little information is available on whether both of these indices were widely used together, or if an attempt was made to develop a single end-point based on assigning a score to each index. However, it was clear that a better tool was need to more consistently and accurately characterize the aquatic communities.

In 1981, James Karr published the Index of Biotic Integrity (IBI), based on supporting the 1972 Clean Water Act's objective to restore and maintain the physical, chemical, and biological integrity of the nation's waters. The IBI was based on 12 individual indices, or metrics, based on fish species composition, trophic composition, and abundance and condition. Each metric was given a score (index value) based upon specific ecological expectations,and the twelve scores were added to provide a single site assessment end-point. The scores resulted in "integrity classes" for streams of excellent, good, fair, poor, very poor, or no fish (Karr et al. 1986).

The IBI can be termed a "composite index" because it combines several community attributes into a single index value. This overall concept, and the IBI in particular, has been demonstrated to be very successful for water quality, or water "resource" evaluations and is widely used by a number of regulatory agencies (Dodd et al. 1990; Cunningham and Whitaker 1989). It did not take long before this concept was successfully applied to benthic macroinvertebrates as well. Jeff DeShon at the Ohio Environmental Protection Agency (OEPA) developed the Invertebrate Community Index (ICI) in 1986 which is based on ten structural and functional metrics the OEPA biologists had subjectively used for a number of years (Ohio EPA 1987a-c).

Based on the enormous success of the IBI as an assessment end-point, USEPA independently began the development of an IBI-type index for benthos. In 1989, USEPA published another set of composite indices called Rapid Bioassessment Protocols (RBPs) for benthic macroinvertebrate and fish communities (Plafkin et al. 1989). The benthic community metrics were based on very general structural and trophic relationships that could be applied nationally, and the primary fish assessment method was the IBI. The RBPs are best used with regionally-defined (ecoregions) reference sites which can be validated for specific States by modifying the RBP metrics, as done by Bruce Shakelford (1988) in Arkansas.

Currently, the USEPA water quality standards program is requiring each State to adopt narrative biological criteria within the next three years, and eventually numerical biocriteria. Development and implementation of biocriteria would not be successful without these composite indices. These new assessment end-points have truly changed the way regulatory agencies can, and will, utilize biological survey data.

Wayne S. Davis
1990 MPCB Meeting Coordinator

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TABLE OF CONTENTS

Data Variability and the Use of Chironomids In Environmental Studies: The Standard Error of the Midge

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Abstract
Many aquatic insect taxa have been used as indicators of environmental quality. However, none has been used as extensively as chironomids (Diptera: Chironomidae). The successful use of midges in environmental studies relies on the integrity and reliability of chironomid databases with species-specific environmental requirements. Errors or inconsistencies within these databases may lead to inconclusive or erroneous results. Much of this can be attributed to methodological errors associated with using midges in environmental assessment. These include: 1) failure to identify to species, 2) inaccurate identifications, 3) inappropriate sampling designs, and 4) inadequate sampling, sorting and sample preparation techniques. These errors can have substantial impacts on estimates of species richness and diversity, on the detection of environmental impact and change, and on determinations of secondary production rates and energy flow in aquatic ecosystems. We refer to the common tendency to ignore or misuse chironomids in environmental assessment as "The Standard Error of the Midge."

Key Words: midges, sampling, taxonomy, data variability, environmental assessment

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Introduction

Aquatic insects long have been used as indicators of water quality (Hellenthal 1982). Fundamental to assessing environmental quality in aquatic habitats is the recognition of reliable indicator species or, preferably, indicator associations or assemblages. Historically, one of the most widely used groups of indicator organisms in both lotic and lentic ecosystems has been larvae of the dipteran family Chironomidae, or midges.

In lentic ecosystems, Thienemann (1922) used genera of chironomids, primarily Chironomus and Tanytarsus, as the basis for his lake typology system. In lotic systems, some of the earliest pollution studies recognized the usefulness of chironomids as indicators of impacted areas (Gaufin and Tarzwell 1952, Richardson 1921).

The reasons that chironomids have been used extensively in assessing water quality are sound. The family Chironomidae is a species-rich group with about 15,000 species worldwide and 1000-2000 species in North America (Coffman and Ferrington 1984). Midges are ubiquitous and frequently the numerically dominant insects in aquatic habitats, attaining densities in excess of 50,000 m^{-2} (Coffman and Ferrington 1984). Finally, the environmental requirements of many chironomid species are environmentally specific and well documented (Beck 1977, Dawson and Hellenthal 1986).

Unfortunately, these characteristics, which give chironomids such great potential in environmental assessment, also contribute to serious difficulties for environmental biologists. Because of their small size (mature larvae range from 2-30 mm) and because of their high densities, the collecting and sorting of larval chironomids is a tedious and time-consuming process. In addition, accurate species-level identifications of larval chironomids may be difficult for the untrained biologist. It is these species identifications, however, that are essential for effective use of chironomids to assess environmental quality. When combined, errors associated with the sampling and identification of chironomids result in highly variable and unreliable data.

Study Site

The data used to illustrate the common errors associated with using chironomids were collected from Juday Creek, a third-order stream in northern Indiana (41°43'N, 86°16'W, elevation = 206 m). Juday Creek is a tributary of the St. Joseph River that flows north into Lake Michigan. Mean annual discharge in Juday Creek is 0.26 m^3s^{-1} with a range of 0.09 m^3s^{-1} in August to 0.56 m^3s^{-1} in April (Schwenneker 1985). The study site was located 0.35 km upstream from the confluence in a natural woodland maintained by the Izaak Walton League of America. This section of stream has a moderate gradient of 1.3% and is primarily riffle habitat with occasional small pools and a pool:riffle ratio (Platts et al. 1983) of 0.1:1.

Results and Discussion

Sampling

One of the greatest sources of variability in using chironomids, particularly in stream studies, is the design of a sampling regime. Sampling must consider both spatial and temporal population characteristics. As with most stream insects, the distribution of chironomid larvae within a stream reach typically is heterogeneous. This heterogeneity must be considered in the design of sampling programs that attempt to detect changes in community structure or population densities. To minimize variability, researchers may have to choose between collecting many samples from a wide range of microhabitats or collecting fewer samples that are restricted to a particular microhabitat.

The distribution of chironomids, even within a single riffle area, is highly variable (Figure 1). Thus, studies that restrict sampling to a particular area of the stream, such as the center, still can result in highly variable density estimates. This variability, however, can be minimized

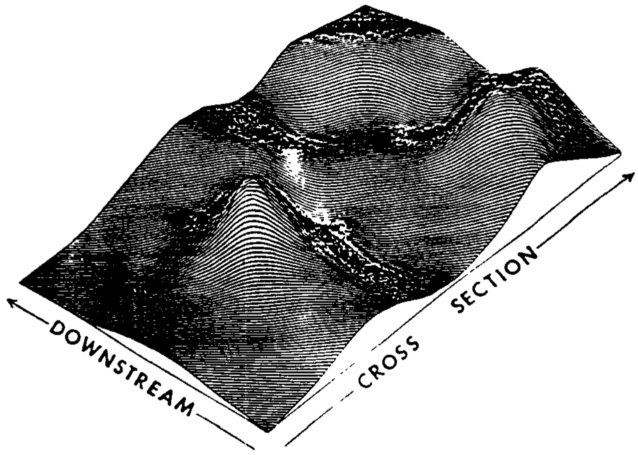

Figure 1. Three-dimensional response surface showing the distribution of the chironomid Pagastia (Oliver) (Diamesinae) during the winter within a single riffle area of Juday Creek.

by conducting a preliminary study to determine species-specific microdistributional patterns (Schwenneker and Hellenthal 1984). Results from this preliminary study then can be used to develop a more efficient sampling strategy designed to address the specific question being considered. For example, prior to conducting a study in Juday Creek, preliminary sampling was used in an attempt to minimize variability of chironomid density estimates. Based on results from this preliminary study, a sampling program was designed that resulted in density estimates with standard errors within 5% of the mean.

In addition to considering spatial variability in designing adequate sampling programs, temporal variability components also must be addressed. Since the Chironomidae is a diverse taxonomic family, often with more than 100 species found in a given habitat (Boerger 1981), a wide variety of life cycles commonly are represented. These can range from univoltine to asynchronous. Species with overlapping cohorts may make determination of life cycles difficult. Chironomid life cycles of three, four or more generations per year are common. As a result of these diverse life cycles, densities of chironomids can vary dramatically throughout the year (Figure 2). Densities of chironomids in Juday Creek range from 7500 m^{-2} in October to 90,000 m^{-2} in early May. Variations such as these must be taken into account in the design of adequate sampling programs. In Juday Creek, 50 samples would be required to detect a 100% change in total chironomid numbers during the summer while only 15 samples would be necessary during the winter. It is clear that a knowledge of species life histories is essential to design the most cost-effective and efficient sampling program. Knowledge of life histories also is essential to ensure that failure to collect a particular species is not misinterpreted as an effect of an environmental impact.

Instar-specific distributional patterns also may be an important source of data variability. The summer and winter distributions of Parametriocnemus lundbecki (Johannsen) (Orthocladiinae), in addition to showing substantial spatial heterogeneity within each season, also

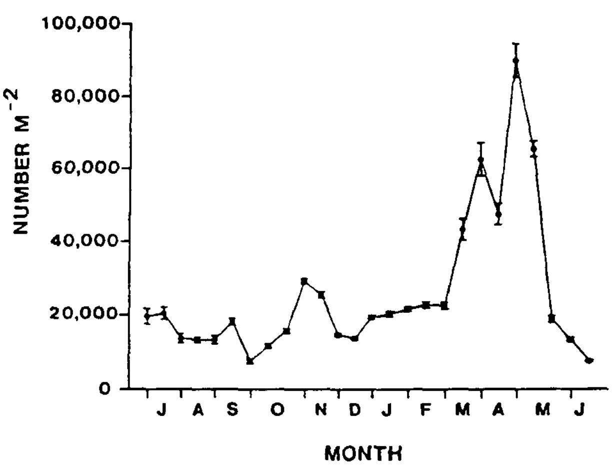

Figure 2. Annual variability in total larval chironomid density (mean density m^{-2} ± 95% CI) in Juday Creek.

demonstrate large interseasonal distribution differences (Figure 3). These differences are primarily the result of early instar larvae predominating along stream margins in the summer and moving toward the center of the stream as they mature during winter. The summer sample was collected early in the season at a time when most of the organisms were second instars. The winter sample, on the other hand, had a mixed assemblage of second, third and fourth instars. The high degree of intra-annual variability due to instar-specific and species-specific distributional patterns may limit the ability to detect significant changes in chironomid densities and species composition during the course of a year. This variability can be minimized by restricting sampling to a particular time of year. This decision should be based on the specific question being addressed. For example, in studies that attempt to detect changes in chironomid densities, sampling should be conducted at times when densities are most stable. Such an approach will result in a greater likelihood of detecting an environmental impact in addition to saving substantial time, manpower and money. Thus, failure to consider both spatial and temporal aspects of chironomid sampling can result in the inability to detect environmental change or an erroneous conclusion that an environmental change has occurred.

A second major source of variability is choice of sampling method. Hess and Surber samplers, which are among the most commonly used benthic samplers, typically have too coarse mesh sizes to retain most larval chironomids. The use of one of these samplers or a similar type of net, such as a kick-net, leads inevitably to the loss of many chironomids and, therefore, to a gross underestimation of chironomid densities. In addition, methods such as these also bias sampling in favor of larger taxa and may lead to serious underestimates of species richness and

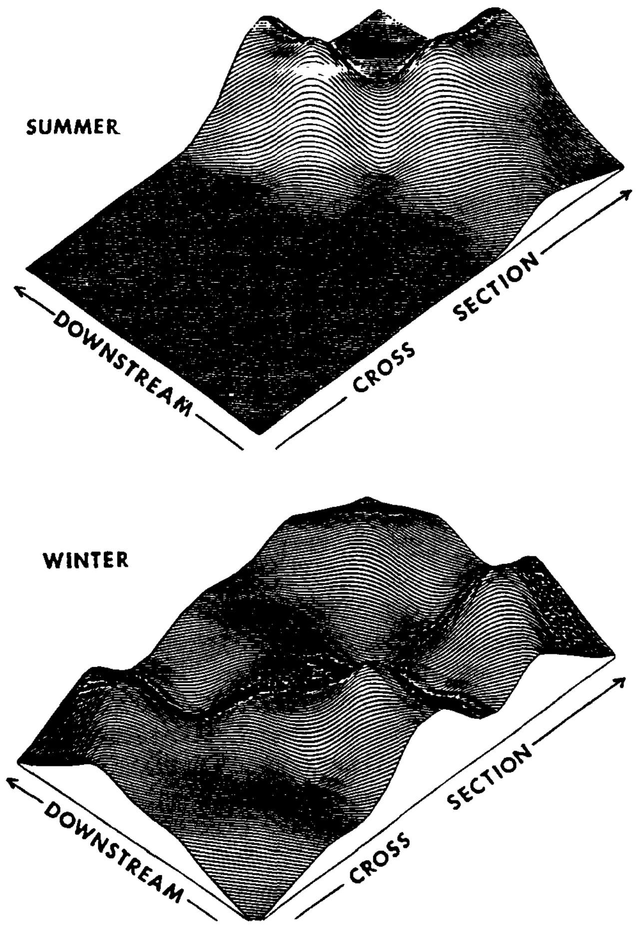

Figure 3. Three-dimensional response surface illustrating interseasonal variability in microdistributional patterns of the chironomid Parametriocnemus lundbecki (Johannsen) (Orthocladiinae) in Juday Creek.

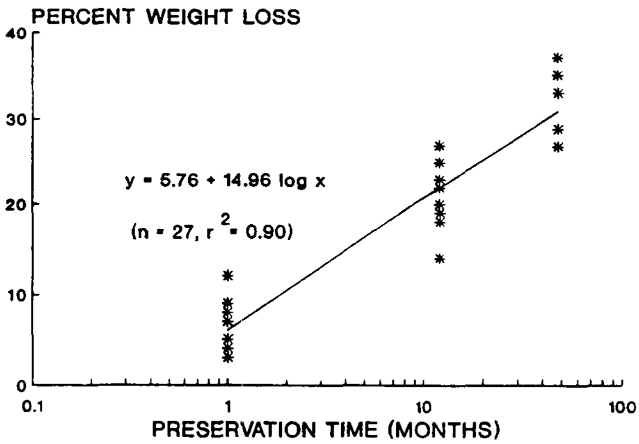

Figure 4. Replicated linear regression of percent weight loss of chironomid larvae versus length of time in an 80% ethanol preservative.

diversity. Similar errors occur when sampling involves scrubbing rocks, tiles or other substrates with brushes. Passing benthic samples through one or more sieves to facilitate sample processing and sorting also may cause the loss of substantial numbers of chironomids. In Juday Creek, even the use of a 250μm mesh sieve often resulted in the loss of as much as 80% of the larval chironomids. Samples with high densities should be subsampled repeatedly until a manageable density is attained. Analyses should then be conducted on the entire subsample.

Sampling errors also occur in attempts to measure standing stock biomass of chironomids, such as those in secondary production studies. Conflicting views can be found in the literature as to the effects of preservatives on biomass estimates. Some researchers have reported little or no weight loss upon preservation in formalin or ethanol (Wiederholm and Eriksson 1977) while others claim substantial losses in weight (Howmiller 1972). In a four-year study of the effects of an 80% ethanol preservative on dry mass of larval chironomids, we found that chironomid biomass decreased by 5% after the first month, 22% after 1 year (i.e. an additional 17% from month 1 to month 12) and 31% after 3 years (or an additional 9% from month 12 to month 48) (Figure 4). Percent weight loss was described by the linear regression:
$y = 5.76 + 14.96 \log x$ ($n=27$, $r^2=0.90$), where x = months in preservative. Weight loss using other types of preservatives also may occur. Thus,

the length of time in preservative can result in substantial differences in chironomid standing stocks observed between different studies as well as within a study. Understanding the relationship between duration in preservative and changes in biomass is essential in any study that relies on estimates of standing stocks or secondary production. The duration of these studies should correspond to the maximum length of time any one sample remains in preservative prior to biomass determination.

It is now clear that different methods can yield substantially different, and potentially conflicting, results. Thus, the high level of variability observed among studies using chironomids is not surprising given that results from studies using different methods often are compared.

Identification

Another major source of error in environmental studies using chironomids concerns larval identifications. Taxonomic keys for larval chironomids are based largely on conspicuous, or not so conspicuous, headcapsule characteristics of mature fourth instar larvae. To see these characters, it is necessary to sever the headcapsule and mount both headcapsule and body on a microscope slide. Thus, the identification of even a few larvae is an extremely time-intensive ordeal. It is not difficult to understand why many researchers have tried to find short cuts to avoid this whole procedure. The most common short cut is to group all chironomids at the family level and to deal with the midges as a single taxonomic entity. This approach invariably will result in the loss or obscuring of important information such as species diversity and species richness that could be used in assessing environmental quality. This is probably why the use of chironomids as environmental indicators has had varying success. If expertise in the identification of larval chironomids is lacking, it is essential to seek additional assistance so that the sensitivity of the results can be maximized.

Ideally, the identification of chironomids to the species level would be of greatest value since published environmental requirements are described for individual species. However in the case of some larval chironomids, the inability to make species-level identifications without rearing the larvae, combined with the high number of species collected in a given habitat, result in studies identifying chironomids to a taxonomic level higher than species, such as species group or subfamily. This approach obscures important ecological information since a high level of diversity exists within these groups with respect to species-specific environmental requirements. The different taxonomic levels to which midges are identified in different studies must be kept in mind. Failing to do so can result in the inability to assess the usefulness of chironomids in environmental evaluation.

An obvious source of error when using chironomids is the accuracy of the identification. Chironomid taxonomy has gone through major changes in the past decade and continues to change at a rapid pace. Relying on outdated handbooks that provide keys to midges can result in costly misinterpretations of community composition. These misinterpretations are perpetuated in the literature and inevitably result in erroneous or conflicting conclusions that cast doubt on the usefulness of midges in environmental research. The value of accurate and complete chironomid identifications can not be overstated. Confirmation of species identifications by qualified researchers and the maintenance of voucher collections are important steps to ensure the integrity of chironomid databases.

Given all of the variability associated with using chironomids and the

Table 1. Assumptions of the size-frequency method and effects on secondary production estimates if assumptions are violated.

Effect on Production Estimate	Assumption
All species have similar life cycles	Underestimate
All species attain same maximum size	Overestimate
Same length of time is spent in each size class	Overestimate or Underestimate

difficulty in working with them, the question that often arises is why do they even have to be considered? One way of assessing the importance of chironomids would be to examine their role in energy transfer and their contribution to overall stream insect secondary production. Previous studies that have attempted to examine chironomid secondary production have committed many of the same errors discussed above.

One of the most commonly used methods to calculate chironomid secondary production is the size-frequency method (Hynes and Coleman 1968). Since this method does not necessitate cohort separation, chironomids usually are pooled at the family level and production is calculated on the family as a whole. However, a series of assumptions associated with this method is invariably violated when chironomids are grouped into a single taxonomic group. These assumptions are: 1) all species have similar life cycles, 2) all species attain the same maximum size, and 3) the same length of time is spent in each size class. Violating these assumptions can either underestimate or overestimate secondary production or can have unpredictable effects on secondary production estimates (Table 1).

In a study conducted in Juday Creek, we estimated chironomid secondary production by following 48 species for one year and calculating secondary production on a species-specific and, usually, a cohort-specific basis without grouping midges at some higher taxonomic level. We found that the 15 numerically dominant chironomid species accounted for over 80% of the total stream insect secondary production (Figure 5). Thus, chironomids are an important energetic component in streams and must be considered in any rapid bioassessment or other environmental assessment program.

Conclusions

The effects of common methodological and taxonomic errors in chironomid studies have strong implications in many areas of aquatic ecology such as designing adequate sampling programs, the examinations of secondary production and seasonal patterns of energy flow, and the evaluation of stream diversity, as well as the ability to conduct successful environmental monitoring. If chironomids are to be used successfully in future environmental studies, reducing the level of non-impact related variability is crucial. This can be achieved best by reducing what we have called the "standard error of the midge."

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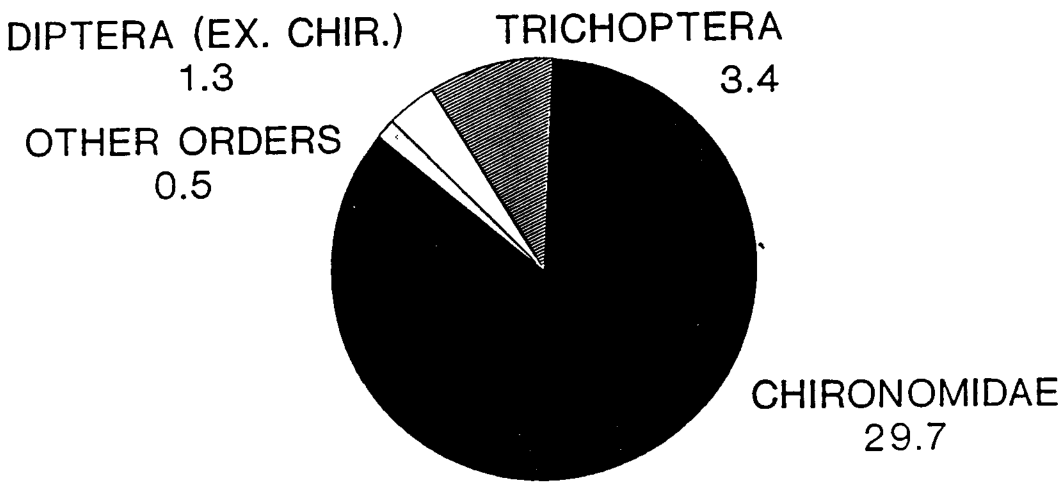

Figure 5. Comparison of chironomid and non-chironomid secondary production rates (g dry mass m^{-2} yr^{-1}) in Juday Creek.

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The Nature of Sampling Variability in the Index on Biotic Integrity (IBI) in Ohio Streams

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Abstract
The Index of Biotic Integrity (IBI) was examined from a number of Ohio streams to determine the amount of variation that can be expected among replicate samples within years and among rivers with various degrees of cultural impact. Biosurvey data have been collected with standardized, pulsed-DC electrofishing techniques by the Ohio EPA over the past 12 years as part of its surface water monitoring program. The variation among samples, as measured by the coefficient of variation (CV) is lowest in streams and rivers that are least impacted by pollution (CV values < 10-12%) and increases in streams as cultural pollution increases (CV values up to 30-40%) until impacts are so toxic that there are only minimal fish communities (IBI scores 12-15). Indeed, high variability among samples in a year was a characteristic of impacted waterbodies. Variability among sampling passes also increased with decreasing habitat quality as measured by the Qualitative Habitat Evaluation Index (QHEI). Precision in the IBI compared favorably to precision in toxicological studies and analytical chemistry results. Among these approaches, as a direct measure of aquatic life, the IBI will be the most accurate arbiter of aquatic life use attainment in most situations.

Introduction
With increasing use of biosurvey data in state water resource monitoring programs it is important to understand, define and control the sources of variation common to biosurvey data. The Ohio EPA has been collecting fish community data, in a standardized manner, in streams and rivers since 1979 and has amassed data on over 3600 sites. This data provides an opportunity to examine patterns of data variability in response to temporal, geographical, and anthropogenic factors.

Five important sources of variability in biosurvey data are: (1) temporal variability (e.g., seasonal, daily, and diurnal changes in community composition), (2) sampling variability (e.g., related to gear, training, and effort), (3) spatial variability (e.g., related to stream size, faunal changes), (4) analytical variability (e.g., related to choice of the appropriate analytic tool), and (5) anthropogenic variability (e.g., degradation of water quality, habitat, toxic impacts to aquatic communities). It is critical to minimize or partition temporal, sampling, and analytical variation in biosurvey data to maximize the ability to distinguish anthropogenic impacts and variation. The goal of this paper is to define the "background" variation in the Index of Biotic Integrity (IBI) in minimally impacted streams (to define temporal and sampling variability) for comparison with variability in streams impacted by anthropogenic activities (i.e., those with aquatic life use impairment).

Background and Methods
The Ohio EPA uses pulsed-DC electrofishing methods (Ohio EPA 1989a) to capture a representative sample of the resident fish community in Ohio streams and rivers. Temporal variability in fish communities composition is minimized by sampling during daylight hours during the summer-early fall months (June 15 - October 15). In most situations we collect three sampling passes on different days during this period to detect within

season (temporal) changes in the fish community. Recent work, however, in the largest Ohio rivers (Ohio River, lower Muskingum River) suggests that night sampling may provide more reliable results in these waterbodies (Sanders 1990) and we have excluded these rivers from this analysis.

Sampling variability is minimized through an extensive training program supported by a a detailed quality assurance manual (Ohio EPA 1989a) and the retention of experienced supervisory and field personnel (average experience > 10 years). Sampling equipment (longline, towboat, or boat mounted electrofishers) and methods and sampling effort are chosen to match the stream size and habitat (Figure 1). Effort is standardized on linear sampling distance which increases with stream size (Figure 1); minimum sampling times are defined for boat methods to ensure a minimum level of effort in large river habitats.

Macroinvertebrate community data and water column chemistry data are generally collected during the same time period as fish community data (June 15 - October 15). Field crews also perform habitat assessments with the Qualitative Habitat Evaluation Index (QHEI: Rankin 1989, Ohio EPA 1989a) within fish sampling zones. Water chemistry data, habitat data, knowledge of pollution sources, and biological response "signatures" (e.g., community response to different types of impacts) are used to determine the causes, sources, and magnitudes of impacts to aquatic life (Ohio EPA 1990). The Index of Biotic Integrity (IBI) is an analytical index used to assess fish community integrity; its applicability and derivation have been discussed elsewhere (Karr 1981, Fausch et al. 1984, Karr et al. 1986, Ohio EPA 1987a,b).

As a measure of variation we calculated the percent coefficient of variation (SD/Mean * 100) for the IBI at sites with three sampling passes

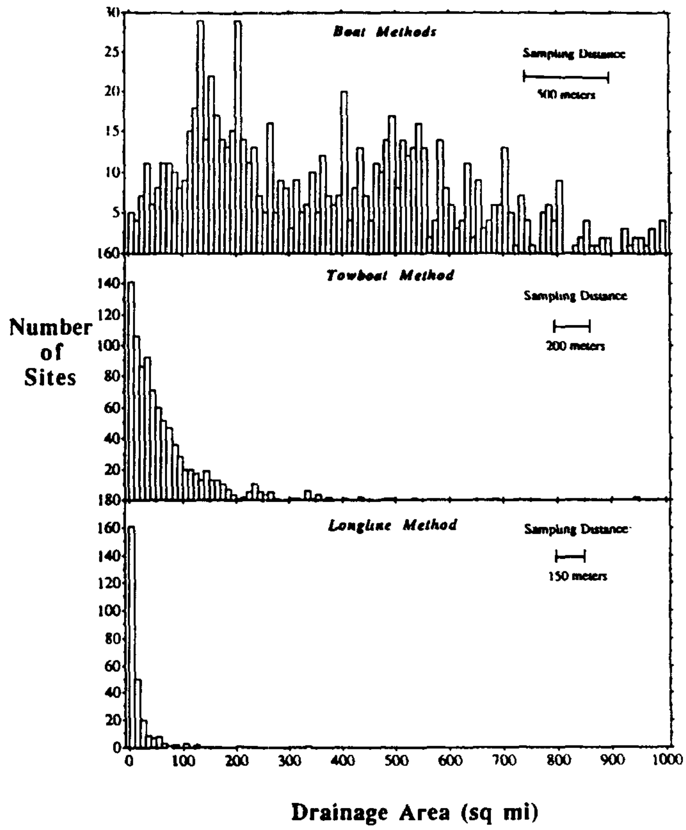

Figure 1. Range of stream sizes sampled by the Ohio EPA with boat, towboat, and longline pulsed-DC electrofishing methods. Sampling zone length for each method is included on each graph.

between June15 - October 15. The IBI, the Index of well-being (Iwb) for fish (Gammon et al. 1981) and the Invertebrate Community Index (ICI) for macroinvertebrates (Ohio EPA 1987b) comprise Ohio's biocriteria (Ohio Administrative Code 3745-1) and are the arbiter of aquatic life use impairment for Ohio's streams and rivers.

Although it is beyond the scope of this paper, one critical source of variation in water resource monitoring with biosurvey data is the appropriate choice of analytical tool. The advantages of broad-based, multimetric indices that have an ecological basis with both structural and functional components have been discussed by others (Karr 1981, Fausch et al. 1984, Karr et al. 1986, Karr 1990).

Results and Discussion

The median percent coefficient of variation (CV) at 1335 sites (1979-1989) with three sampling passes shows a distinct increase with decreasing IBI score except at the very lowest IBI range of 12-15 (Figure 2). Figure 2 is divided into IBI ranges that roughly correspond to Ohio's Exceptional Warmwater Habitat (EWH) aquatic life use IBI criteria, Warmwater Habitat (WWH) aquatic life use IBI criteria, and IBI scores that reflect impaired aquatic life uses. The median CV is generally less than 10% in EWH streams and 15% in WWH streams that achieve their respective IBI biocriteria. The distribution and range of CV values broadened significantly in streams with impaired aquatic life uses except at the very lowest IBI scores (12-15). By themselves increases in the variation of biosurvey data are an indication of impact to a stream. Cairns (1986) suggests that "...differences in variability rather than differences in averages or means might be the best measure of stress in natural systems".

Increases in variation are observed among streams affected by most types of impact (Figure 3). Ohio has no pristine, unimpacted streams. The "least impacted" streams in Ohio, however, such as the West Fork of Little Beaver Creek, Captina Creek, Rocky Fork of the Licking River, and the Kokosing River, have CV values of less than 5-10% and stable fish communities (as measured by the IBI). For example, the West Fork of Little Beaver Creek achieves an IBI of 50 or more (Ohio's EWH IBI criteria) in 25 of 27 sampling passes (Figure 4). This data spans five years and the two exceptions to this trend are due to a problem of recent origin.

Streams with impacted fish communities (IBI scores generally less than 40) had 75th percentile CV values of > 10-15% and as high as 30-40% (Figure 3). For example, the CV was negatively correlated with the QHEI (Qualitative Habitat Evaluation Index: Rankin 1989, Ohio EPA 1989), a measure of habitat quality (Figure 4). Low QHEI scores reflect low habitat quality that supports fewer habitat sensitive species and more tolerant individuals resulting in higher variability in catches and CPUE. Other impacts also resulted in increased variation in the IBI with toxic impacts among those associated with the highest IBI variation (Figure 3). Low species richness or low abundance of certain species, due to any impact type, increases the likelihood of IBI metrics being near scoring thresholds (1 vs 3 or 3 vs 5 points) and increases the variability in the IBI. Similarly, water quality impacts can reduce species numbers or affect trophic group composition through avoidance or mortality, and increase the variability of the IBI.

In contrast, extremely toxic impacts (IBI scores 12-15) were often characterized by little or no variation. In these situations few fish survive and metrics nearly always score a one. Exceptions are the downstream "edge" of a toxic effect (or episodic water quality impacts) which may shift the location of an impact over time, especially where there is migration from a nearby "refugia" with a healthy fish community. This situation was illustrated in Hurford Run near Canton Ohio (Figure 5). Upstream sections of Hurford Run had fish communities that were consistently very poor, but the fish community near the mouth fluctuated as tolerant, colonizing fish species (young-of-year green sunfish [Lepomis cyanellus], bluntnose minnow [Pimephales notatus], creek chub [Semotilus atromaculatus]) migrated from a mainstem "refugia".

The CV showed no regional pattern other than that which can be explained

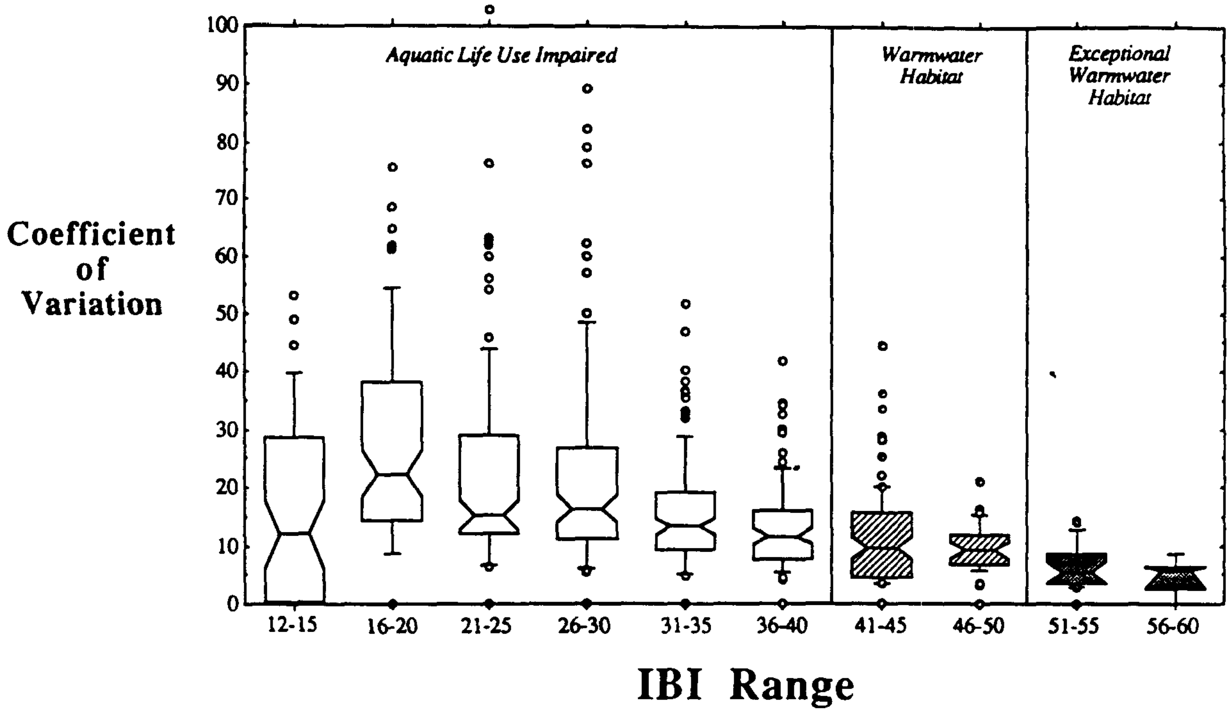

Figure 2. Median percent coefficient of variation (CV), 25th and 75th CV percentiles, CV range, and CV outliers (> 2 interquartile ranges from median) for the Index of Biotic Integrity (IBI) versus IBI range. CV values were calculated for sites with three sampling passes collected between June 15 - Oct 15. N = 1335 sites.

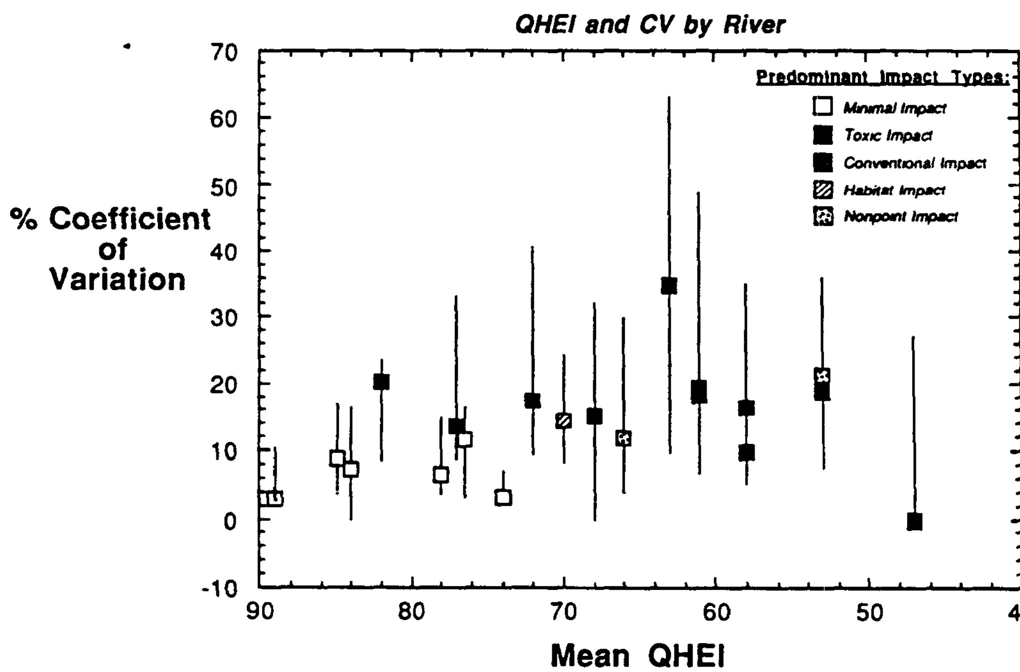

Figure 3. Median percent coefficient of variation (CV), and 10th and 90th CV percentiles for the Index of Biotic Integrity (IBI) versus QHEI (Qualitative Habitat Evaluation Index) for twenty Ohio streams and rivers. Shading of median values represents the predominant impact type in these streams.

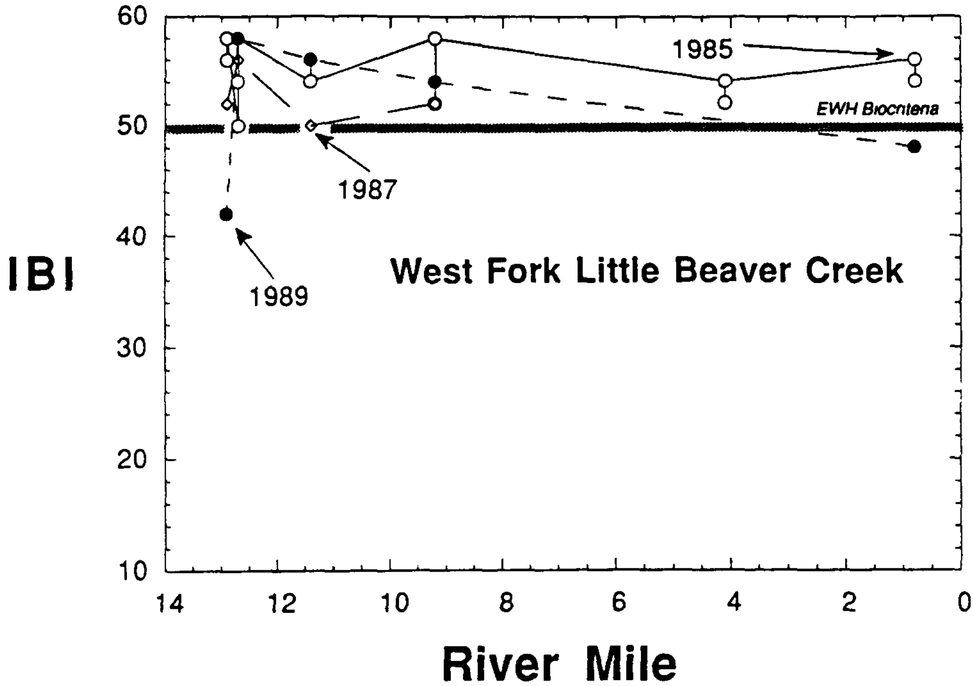

Figure 4. The Index of Biotic Integrity (IBI) versus river mile (upstream to downstream) for the West Fork of Little Beaver Creek (Columbiana Co., Ohio) for 1985 (N=3 passes), 1987 (N = 1 pass), and 1989 (N = 1 pass).

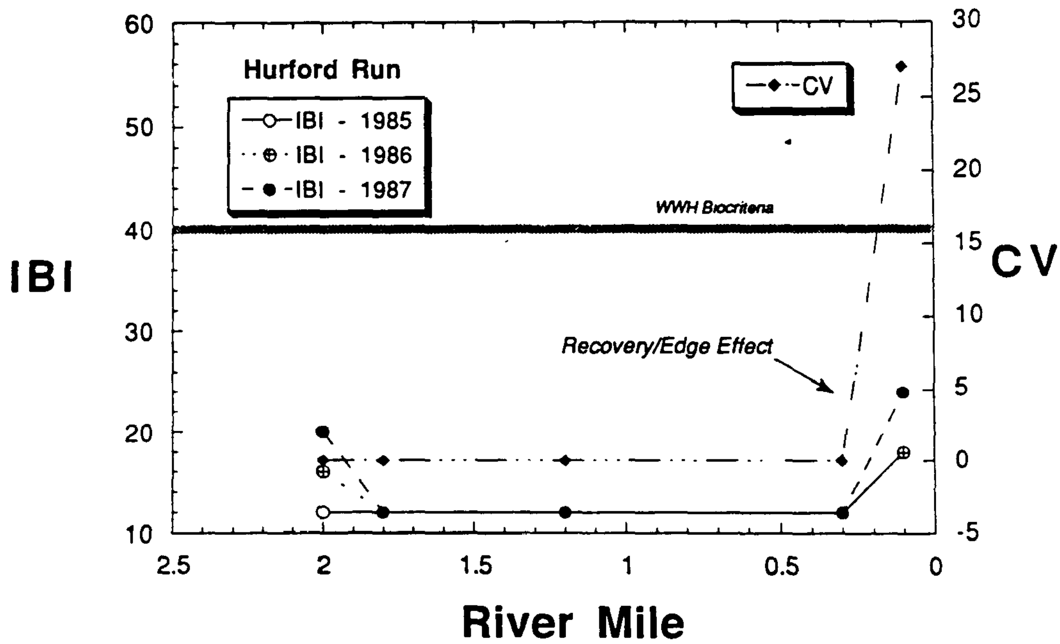

Figure 5. The Index of Biotic Integrity (IBI) and median percent coefficient of variation (CV) versus river mile (upstream to downstream) for the Hurford Run (Stark Co., Ohio) for 1985 (N=3 passes), 1986 (N = 1 pass), and 1988 (N = 1 pass).

by overall impacts within the ecoregions of Ohio (Figure 6). There was a slight trend in the upper threshold of variation in the IBI with stream size (Figure 7). Figure 7 represents the CV for streams in Ohio with IBI scores greater than 48 (i.e., the CV at these sites represents background variation due to inherent sampling variation and normal fluctuations in fish communities over time). The increase in the CV with stream size (Figure 7) most likely reflected the smaller proportion of the total community that was sampled in large versus small streams. Even in larger rivers, however, the CV was under 10-12% in the majority of situations.

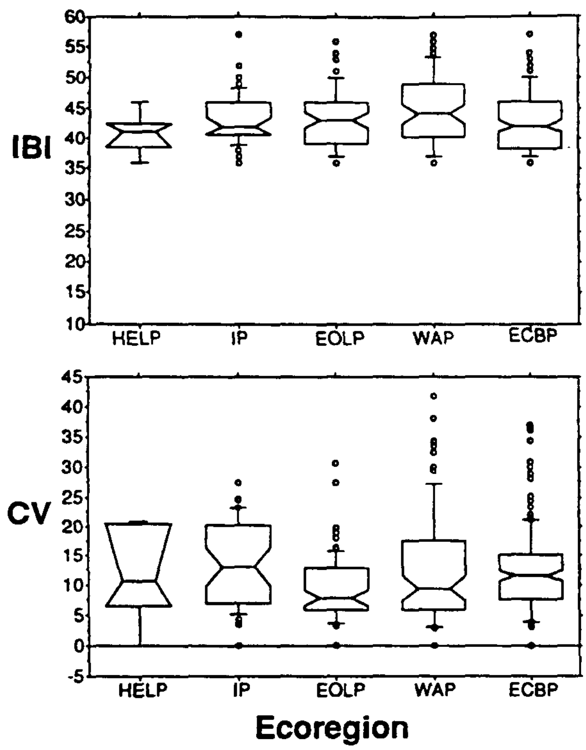

Figure 6. Boxplot of the median, 25th and 75th percentiles, range, and outliers (> 2 interquartile ranges from median) for the IBI (top panel) and percent coefficient of variation (CV, bottom panel) for sites in Ohio's five ecoregions. HELP: Huron Erie Lake Plain, IP: Interior Plateau, EOLP: Erie Ontario Lake Plain, WAP: Western Allegheny Plateau, ECBP: Eastern Corn Belt Plains.

Development of Ohio EPA "Significant Difference" in the IBI

Because we expected some background variation or "noise" in our samples we derived guidelines for detecting significant differences between IBI values from our intensive surveys and the regional reference sites used to derive our ecoregion-based biocriteria. We examined histograms of deviations in sample IBI values from mean IBI values at all locations where we had three sampling passes (Figure 8). We chose the 75th percentile value of this deviation from the mean as the limit of tolerable variation. This resulted in a guideline that the difference between a sample IBI and the ecoregion IBI biocriteria must be greater than 4 units to be classified as a significant departure. Because we used a mix of impacted and relatively unimpacted sites deviations of greater than 4 units probably reflects variation of anthropogenic origin. This is a protective criteria, however, because all available and applicable criteria for two organism groups (i.e., the modified iwb for fish in addition to the IBI and ICI for macroinvertebrates) must be met to fully attain an aquatic life use (Ohio EPA 1987b).

Detecting impacts and their underlying causes is more complex than simply determining significant departures from ecoregion biocriteria. For sites not attaining their aquatic life use the structural and functional characteristics of fish and macroinvertebrate communities provide information or "biological response signatures" about the type of impact that is affecting the aquatic life (Ohio EPA 1990a). Two sites that have similar IBI scores that indicate impaired communities may have very different community responses. The difference in the composition, function, and structure of the communities, in concert with chemical, toxicological, and physical data, provide clues to the cause or causes of impairment. Similarly, contrasts

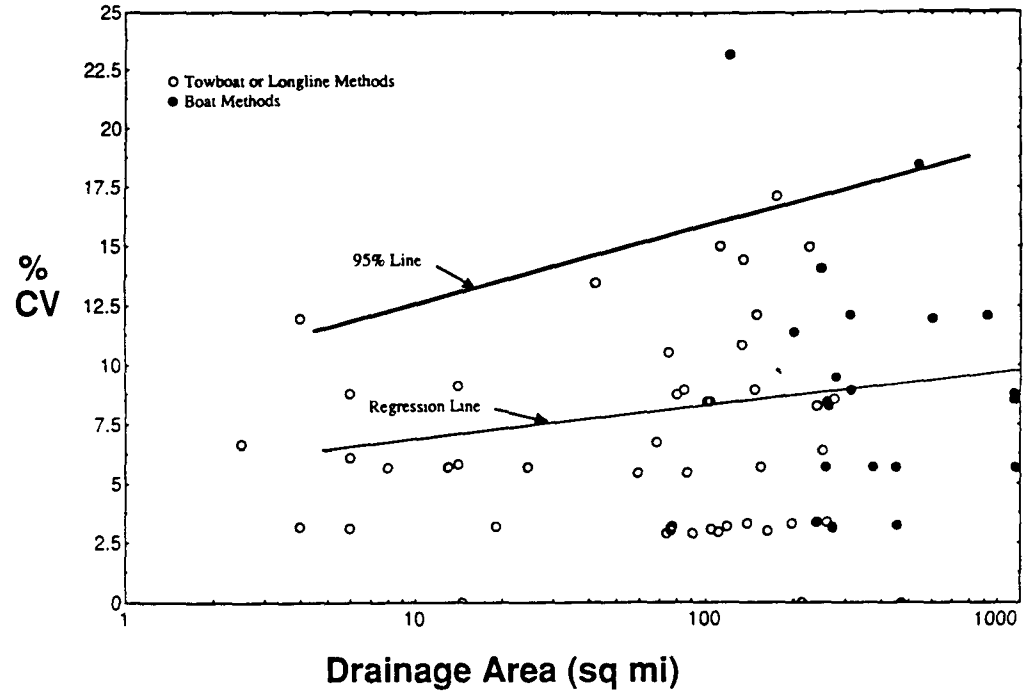

Figure 7. Median percent coefficient of variation (CV) versus drainage area for streams in Ohio with IBI scores > 48. The line on the graph represents an "upper threshold" and was drawn by eye through the upper 5% of the points.

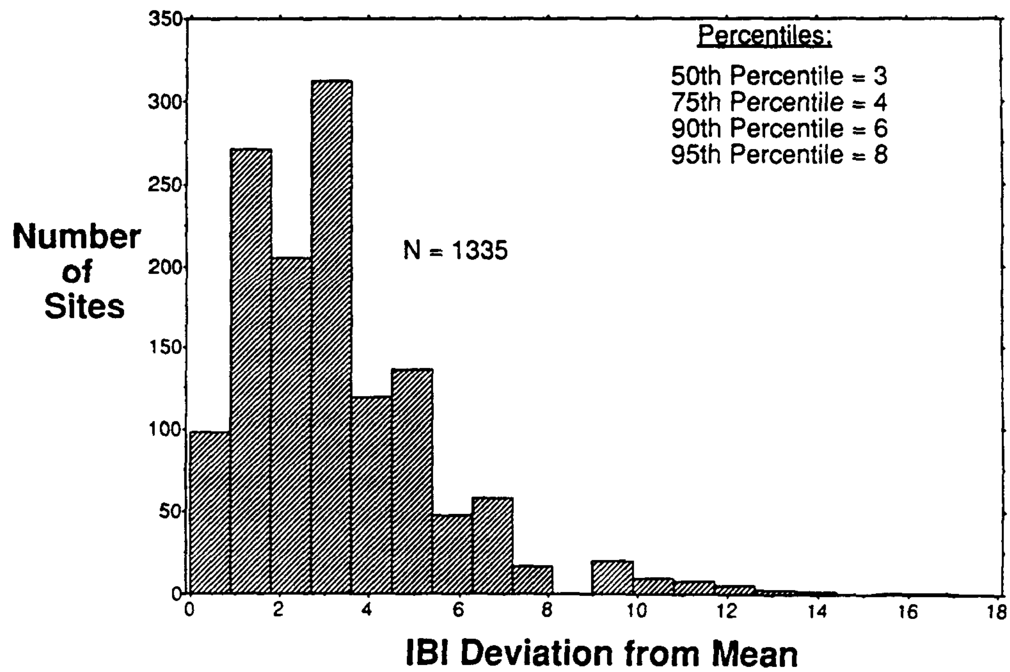

Figure 8. Frequency of the deviations of individual IBI passes from mean IBI values at stream sites with three sampling passes for all sites (solid bars) and reference sites (cross-hatched bars).

between the fish and macroinvertebrate community response are advantageous for detecting the type of impact. Work on formally classifying the responses of the biota to different types of impacts is in a developmental stage. New techniques, such as artificial intelligence (e.g., machine learning algorithms) may prove useful in this endeavor (David Davis, BBN Inc., personal communication).

Comparison of the CV values from biosurveys with other types of environmental monitoring data (e.g., water column chemistry, toxicity testing) provides additional perspective on the precision of the IBI. Mount (unpublished) compiled coefficient of variation values from a number of efforts to compare interlaboratory variability in toxicity testing and analytical water chemistry data. For organic and inorganic analyses most CV values were greater than 30% for the lower detection range of these parameters (e.g., mean of inorganic analyses = 125%). CV values, however, generally decrease when higher concentrations of compounds are analyzed (Turle 1990). The mean CV value (inter-laboratory variability) for toxicity tests (mostly LC50 values) was 30% (range: 0 - 66%; N = 16 CV values). Although replicate variability in the IBI was examined in this paper, the levels of interlaboratory variability associated with analytical chemistry data and toxicity testing are somewhat higher than the replicate biosurvey data. Though this interlaboratory variability is not strictly comparable to biosurvey replicate variability it does suggest that variability in biosurvey data is within or below the range of other, widely accepted environmental measurements.

CV values for replicate macroinvertebrate samples in a Wisconsin stream ranged from 6.2% to 43.6% (Szczykto 1989) depending on the index used in the analysis (all index scores were generated from the same data). Davis and Lubin (1989) calculated a CV of 20% for the Invertebrate Community Index (ICI) for all of the sites in Ohio EPA's regional reference site database. "Background" levels of precision are likely lower than 20% for replicate ICI scores for any given site because the reference sites are not homogeneous and represent a gradient of aquatic life performance. Nineteen replicate ICI scores at a relatively unimpacted test site in Big Darby Creek had a CV of 10.8%, which was lower than 8 of 9 of the index's underlying components. This CV value is similar to those found for the IBI in relatively unimpacted sites (see Figure 3).

Based on the data presented here the IBI scores collected by the Ohio EPA reflect low enough levels of sampling and natural variation to detect meaningful changes in biological integrity in streams. The precision of the IBI compares favorably with precision in analytical water chemistry methods and toxicity testing. However, this is not an effort to establish the "superiority" of one environmental measure over the other. Beyond considerations of precision, biosurvey data, water chemistry data and toxicity tests have specific applications where they are most appropriate and accurate. Our experience in Ohio has shown us that biosurvey, water chemistry, and toxicity testing are all necessary to completely and accurately define an impact to a stream in a complex situation, but that each is not necessarily independent of the other in all situations. There will be instances where one measure will carry more influence or weight than another. Unfortunately, this is not completely predictable at this point.

In the assessment of water resource impacts it is important to differentiate between accuracy and precision and to choose the appropriate "tool". Given an acceptable level of <u>precision</u>, emphasis should be put on environmental measures that accurately reflect water resource management goals (e.g., protection of aquatic life). For example, biological community data is free from assumptions and safety factors associated with laboratory derived data and accurately and directly reflects attainment of aquatic life uses (i.e., a high level of reality). Rankin and Yoder (1990) have shown that a reliance on water chemistry data and criteria alone under-estimated the impacts on aquatic life uses in Ohio in 49% of stream segments that were assessed. In contrast, only a small percentage of stream segments (< 3%) had biological communities that attained aquatic life uses, but violated chemical water quality criteria.

The IBI, when data collection methods are standardized, increases the accuracy of water resource assess-ments. Further work needs to: (1) identify biological response "signatures" for different types of impact, (2) identify situations where bio-survey data from multiple organism groups decreases the "variability" or increases the sensitivity of an assessment, (3) identify inter-laboratory variability in biosurvey data collection, and (4) compare variation between quantitative, standardized sampling methods (Ohio EPA approach described here) and more qualitative methods (e.g., Rapid Bioassessment Protocols, volunteer monitoring).

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Reducing Variability in Freshwater Macroinvertebrate Data

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Abstract
The benthic macroinvertebrate community is often used to evaluate stream water quality, but this efficiency of this process may be complicated by high data variability. This variability can be reduced by proper selection of sampling sites, collection methods, identification levels, and analysis metrics. Corrections also can be made to compensate for predictable changes associated with ecoregion, stream size and seasonality. Some evaluation should be made for the effects of antecedent flow, especially after droughts and high rainfall periods.

Key words: North Carolina, benthos, data variability, methods, identification.

Introduction
Environmental monitoring groups often use characteristics of freshwater macroinvertebrate communities to assess stream water quality. In cases of severe pollution, any kind of collection technique and/or any kind of data analysis can be used to demonstrate a water quality problem. In cases of "less than catastrophic" pollution, however, high data variability may obscure the effects of changes in water quality (Howmiller 1975). There are many different sources of variation for benthic macroinvertebrate data, including differences in collection efficiency, habitat, season of the year, and flow.

The problems of data variability can be greatly reduced by making corrections for any changes in habitat and season of the year, as well as through wise choices of identification levels, collection methods, and data analysis techniques. Erman (1981) has shown the frustrations in trying to compare studies with different collection techniques and identification levels. This paper will focus on North Carolina's experience with making these choices, and the ways we are developing seasonal and habitat-associated adjustments to our biocriteria. Some overlap with Lenat (1988) is inevitable, as both papers discuss the subject of taxa richness variability, but a large amount of new material has been included.

The North Carolina program was originally set up to deal with relatively simple between-station and between-date comparisons; the emphasis was on showing large changes in water quality or habitat quality. As the water quality program expanded, we began to look at more subtle water quality problems. Monitoring was required for all stream sizes (from temporary streams to large rivers) and we were asked to make collections during all months of the year and under a variety of flow conditions. To deal with these complicating factors, we are examining "normal" changes in the benthic macroinvertebrate community associated with differences in habitat, stream size, and seasonality.

North Carolina originally used quantitative collections (kick-net samples) to evaluate the benthic macroinvertebrate community. All samples were laboriously sorted in the lab. As our monitoring requirements expanded, we developed several new collection methods to collect reliable information in a more cost-efficient manner, including a new "rapid bioassessment" technique.

Much of the data presented in this paper is still in a preliminary stage of analysis, as North Carolina has just completed a four month effort to put all information (1983-present) into a large computerized data base. We have been using this data set to look at the spatial, temporal preferences of each taxa, as well as generating pollution tolerance data. We would like to use this paper as a means of soliciting opinions and advice concerning these analysis methods from other biomonitoring groups.

Results and Discussion

Collection and Identification Choices

The first step in reducing variability is to apply common-sense during sample collection. Stations should be chosen to be similar in habitat characteristics, and collections should not be made if high flow will interfere with collection efficiency. The collection method also should be suitable for the habitat being sampled. For example, dredge samples are rarely appropriate for shallow, fast-flowing streams.

There is considerable disagreement about the appropriate identification level and/or what groups should be identified (see Lenat 1988). North Carolina has chosen to use species or genus level identifications (where possible), including the infamous Chironomidae. It is clear that species level identifications increase the efficiency of site classifications (Resh and Unzicker 1975, Furse et al. 1981, Furse et al. 1984, Hilsenhoff 1982, Rosenberg et al. 1986), but with a cost of added identification time. I agree with Hilsenhoff (1982) that the added time required for species identifications is trivial compared to collection and sorting time. Many investigators elect to identify the Chironomidae to family (or subfamily) level, even if other groups are classified at a genus/species level. While the taxonomy of this group can be difficult, the information added by good chironomid data can be valuable in determining the nature of water quality problems.

Collection methods should be chosen which yield reliable data in the most cost-efficient manner. This choice will vary depending on the objectives of the study, especially on the need for precise estimates of species abundance. Abundance measurements will be required for life cycle studies and production studies, but are notoriously difficult to obtain. Our experience in water quality assessment is that we need a quantitative estimate of taxa richness and a qualitative estimate of abundance values (Rare, Common, Abundant). These requirements lead to the derivation of our standardized qualitative collection method (Lenat 1988).

All North Carolina collection methods utilize large composite, multiple-habitat, samples. The Standard qualitative method utilizes 10 samples, taken with 6 different collection methods. We have also developed an Abbreviated ("rapid bioassessment") collection method, which has become an important part of North Carolina's biomonitoring program. The latter method uses only 4 composite samples (kick-net, sweep-net, leaf-pack, and "visuals"), with collection and identification limited to the EPT groups. Note that the Abbreviated collection method produces a subsample of the Standard collection. We have recently compared Standard and Abbreviated samples collected independently at 30 sites (Larry Eaton, unpublished data). The 4-sample collections naturally collect fewer species than the 10-sample collections, but results from these two methods are highly correlated (Figure 1, r^2=0.96), allowing criteria to be developed for each. High variability is associated with a smaller sample size, but this is offset by the larger number of sites that may be sampled. A more detailed description of the Abbreviated method is in preparation.

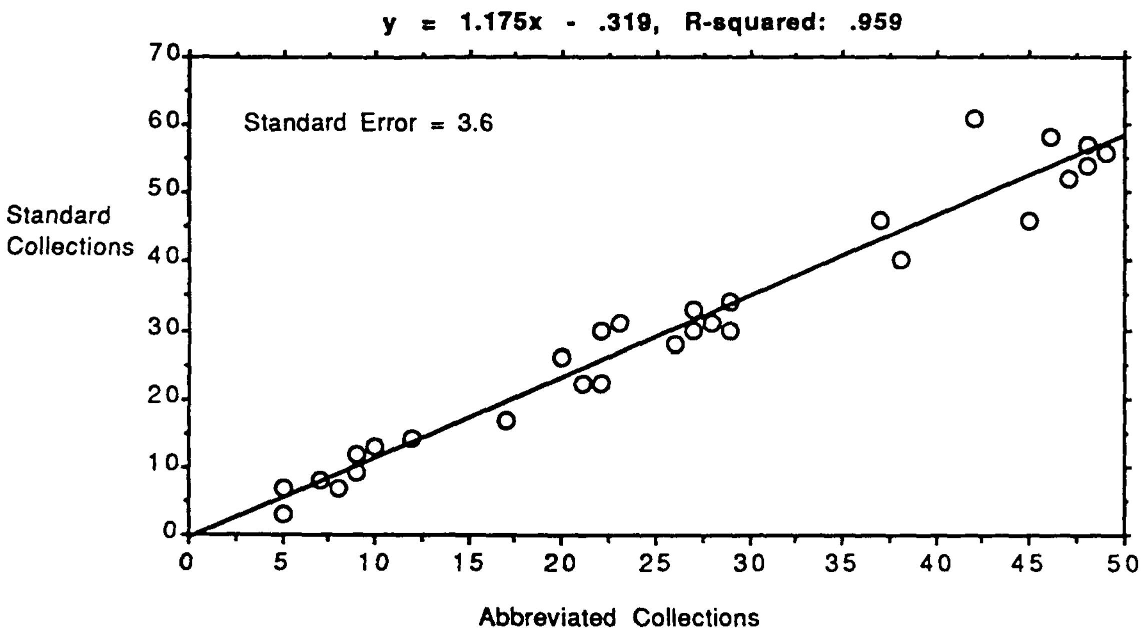


Figure 1. EPT taxa richness for standard (10-sample) collections vs. abbreviated (4-sample) collections.

Analysis Metrics
The choice of analysis metrics have a significant effect on the variability of your data or the reliability of site ratings. The ideal metric will be insensitive to normal habitat changes, but sensitive to changes in water quality. Many monitoring groups are trying to increase the confidence in their water quality evaluations by using several (relatively independent) ways of examining the benthic macroinvertebrate community. This latter technique has been borrowed from the Index of Biotic Integrity (Karr 1981) used by fisheries scientists.

Taxa Richness. The North Carolina methods tend to focus on taxa richness, especially taxa richness for the intolerant (EPT = Ephemeroptera + Plecoptera + Trichoptera) groups. Many investigators have shown that taxa richness (and related parameters) are more stable than abundance values (Godfrey 1978, Minshall 1981). Taxa richness values have been frequently associated with environmental stress (especially water quality), but this parameter is fairly stable in clean water habitats, even given some changes in habitat characteristics and/or flow (Patrick 1975, Bradt and Wieland 1981, Minshall 1981, Wagner 1984).

Biotic Indices. Another way to reduce variability is to use metrics which are (theoretically) independent of sample size. Diversity indices were derived with this in mind, but have proved to be unreliable in many types of pollution assessment (Godfrey 1978, Hughes 1978). Biotic indices have greater promise for water quality assessment (Hilsenhoff 1982), but their use in the Southeast has been hampered by the lack of a good data base on the environmental tolerances of benthic macroinvertebrates. Tolerance values have invariably been

Table 1. Preliminary information for deriving a North Carolina biotic index from existing bioclassifications. Mean abundance values vary from 0-10 and bioclassifications are coded 1-5. Percentile calculations are based on cumulative abundance values, starting from the Excellent bioclassification.

	Mean Abundance Values					Bioclass #			
Bioclassification:	Poor	Fair	Good-Fair	Good	Excellent	Mean	Percentiles		Converted[1]
Bioclass #:	1.0	2.0	3.0	4.0	5.0		75th	90th	75th
Percentile									
Intolerant Species									
Drunella wayah	-	-	-	0.2	0.3	4.5	4.6	4.3	0.6
Rhithrogenia spp.	-	-	0.1	0.2	0.8	4.5	5.0	4.0	0.0
Chimarra spp.	0.1	0.3	1.2	2.5	3.8	4.2	4.2	3.0	1.1
Micrasema wataga	-	0.1	0.8	0.8	1.0	3.9	3.7	3.2	1.9
Goera spp.	-	-	-	0.3	0.6	4.5	4.8	4.3	0.3
Brachycentrus chelatus	-	-	-	0.1	0.1	4.5	4.5	4.2	0.7
Pteronarcys dorsata	-	0.1	0.3	0.7	0.7	4.0	4.1	3.3	1.3
Acroneuria abnormis	0.2	0.6	2.3	5.4	8.0	4.2	4.2	3.0	1.1
					Means:	4.3	4.4	3.7	0.6
Facultative Species									
Stenonema modestum	1.5	7.0	8.4	7.8	8.3	3.5	3.0	2.2	2.9
Ephemerella catawba gr.	0.7	0.7	0.9	1.7	3.0	3.9	3.4	2.0	2.3
Eurylophella temporalis	0.1	0.4	0.9	1.3	1.3	3.8	3.5	2.7	2.1
Cheumatopsyche spp.	3.0	7.3	7.7	7.0	7.4	3.4	2.7	2.0	3.3
Hydropsyche venularis	0.7	1.9	3.4	2.5	2.7	3.5	3.1	1.9	3.0
Perlesta spp.	0.1	1.0	1.4	1.6	1.4	3.6	3.2	2.4	3.1
Ancyronyx variegata	0.9	2.1	2.2	1.5	0.9	3.1	2.5	1.8	3.6
Polypedilum convictum	0.5	1.6	2.8	2.0	1.8	3.4	3.0	2.0	2.9
					Means:	3.5	3.1	2.2	2.9
Tolerant Species									
Cricotopus bicinctus	3.6	3.3	3.2	2.0	1.1	2.8	1.9	1.4	4.4
C. tremulus gr (C/O sp. 5)	1.4	1.2	1.1	0.7	0.4	2.7	1.8	1.2	4.6
Chironomus spp.	3.8	2.2	1.5	0.9	0.5	2.4	1.6	1.2	4.9
Polypedilum illinoense	4.3	3.4	3.3	2.3	1.7	2.9	1.9	1.3	4.4
Physella spp.	3.4	2.7	2.3	1.8	1.1	2.8	1.8	1.3	4.6
Argia spp.	3.1	3.4	2.7	1.9	1.8	2.9	2.0	1.4	4.3
Limnodrilus hoffmeisteri	3.6	2.2	0.8	0.6	0.6	2.3	1.5	1.2	5.0
Asellus spp.	1.7	1.3	0.8	0.5	0.3	2.5	1.7	1.2	4.7
					Means:	2.6	1.8	1.3	4.6

[1]Numbers "flipped" so that a higher value reflects greater pollution tolerance: x = 6-y, range expanded (with regression equation) to a 0-5 scale: tolerance value = 1.43x - 1.43. Converted numbers are comparable to a Hilsenhoff-type index.

assigned based on best professional judgement, as was the case for North Carolina's existing biotic index.

North Carolina has initiated a program to more systematically derive invertebrate tolerance values, using our existing computerized data base. This data base currently has 1300+ individual collections, including samples from a broad range of water quality classifications, ecoregions, stream sizes and seasons. Table 1 presents some very preliminary data from our efforts to derive tolerance values. I present this information here in an effort to solicit comments and suggestions from readers, the final form of our biotic index may vary substantially from the concept presented here.

The initial step was to combine information on bioclassifications (based on EPT taxa richness), with abundance (0=Absent, 1=Rare, 3=Common, 10=Abundant) and frequency data. The first set of numbers in Table 1 are average abundance values (0-10) for each water quality class. The summary values are based on the water quality class (1-5), with a mean, 75th percentile and 90th percentile. Percentiles are based on the cumulative frequency distribution, starting from Excellent water quality (Class #5). Ideally, the tolerance values should show a large separation of tolerant and intolerant taxa, while still producing intermediate values for facultative taxa (near the median bioclass # of 3.0). The 75th percentile number was chosen as the summary statistic closest to these ideal characteristics, and was converted to a Hilsenhoff-type biotic index. The numbers were "flipped" so that a higher number reflects greater pollution tolerance. This produced a range of values similar to a Hilsenhoff index, but with a range of only 1.0 to 4.5. A simple regression equation was used to expand this range to 0-5, with the resulting numbers directly comparable to Hilsenhoff-type indices. If there is insufficient data to derive a tolerance value for some species, the original value (based on best professional judgement) can be retained. This approach to deriving a biotic index seems to show great promise as an alternate method of bioassessment. The next step would be to derive index criteria for both Standard and Abbreviated samples.

Collector Effects
Several investigators have examined the effect of the collector on the variability of benthic invertebrate data (Chutter and Noble 1966, Pollard 1981, Furse et al. 1981, Lenat 1988). While some differences can be found, most studies agree with Egglishaw (1964) that collector effects are "not large". The type of differences noted by Furse et al. (1981) argue strongly for standardization of collection methods.

Habitat Effects
Between-site and between-sample differences in habitat often contribute to data variability. If these differences are large, it may negate any attempt to look for changes in water quality. In many cases, the investigator can limit habitat differences, but these problems may be unavoidable for basin-wide surveys. Habitat differences can be considered for three distinct size scales: ecoregion, stream reach, and microhabitat.

Ecoregion. Ecoregion is rapidly becoming one of the most significant buzz-words of the 1990's. No government document can be released without at least one reference to the need for ecoregion reference sites. The ecoregion concept suggests that streams within a relatively uniform geographic areas will have similar faunas, or at least similar community structure (Hughes and Larsen 1988). This concept has been most fully developed for fish communities, but has also been shown to be applicable to stream invertebrates (State of Arkansas 1987, Lenat 1988). North Carolina has utilized three broad eco-

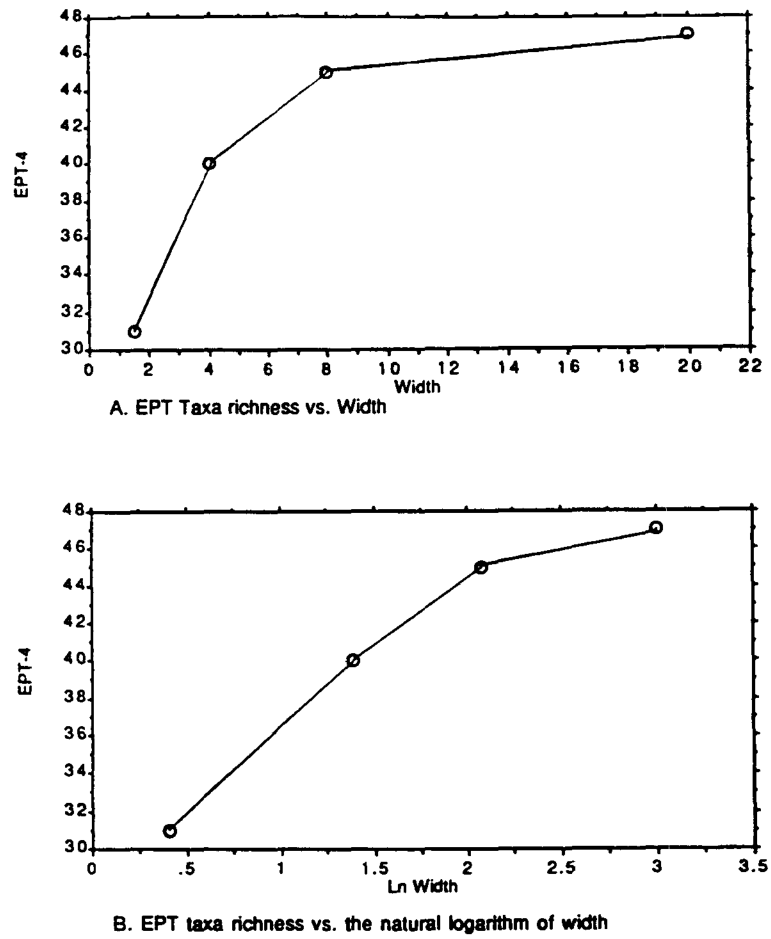


Figure 2. EPT taxa richness (abbreviated samples) vs stream width (m). Cataloochee Creek catchment, January 1990.

regions to develop bioclassification procedures, but there may be up to 12 different ecoregions in our state. Preliminary work indicates that at least 7 ecoregions will be needed to establish reliable site classifications, requiring up to 7 different sets of biocriteria. Some important factors in determining ecoregion include elevation/slope, soil type and permeability, geology, vegetation, and land use.

Stream Reach. At the next size scale, one must consider variability between stream reaches, especially in regard to stream size. Several studies have looked at the changes in the invertebrate community in relation to stream size, usually indicating an increase in taxa richness from first to fifth order streams, with a decline in higher order streams. Such studies usually look at average values per sample, rather than looking at changes in the entire stream community (Minshall et al. 1985 and Naiman et al. 1987). It is possible that a part of the decline in higher order streams is related to the smaller proportion of the stream that single (usually midstream) collections will sample in larger rivers. Gaschignard et al. (1983) found that the river fauna could be separated into two units: a mid-channel community, and a community found within 10 meters of the bank. In small streams, midchannel samples will include both assemblages. As stream size increases, however, there is a decreased probability that the bank assemblage will be included in single-habitat samples. Multiple-habitat samples may eventually produce a slightly different picture of stream size versus taxa richness.

All investigators agree that lower taxa richness is expected in small stream. This point is illustrated in Figure 2, showing a sharp drop in taxa richness in comparing a site 1.5 meters in width with a site 4.5 meters in width. Taxa richness vs. the natural logarithm of width (in this example) showed an almost linear relationship. Most biological criteria are derived from larger streams and rivers; a logical refinement would be to make some adjustment for different size classes.

The problem of classifying streams with taxa richness values is greatest for very small streams. These streams will have more limited habitat complexity, but the most important cause of reduced taxa richness in these systems is the periodic stress caused by drought/low flow conditions. Droughts may cause drastic reductions in current speed, often with an accompanying reduction in dissolved oxygen; some streams may dry up entirely.

What constitutes a "small stream" in North Carolina will vary with soil permeability. In well-drained soils (Sandhills ecoregion), permanent flow occurs in some streams less than one meter wide. In poorly drained soils, however, (Slate Belt Ecoregion) streams up to 15 meters wide may become temporary during extended droughts. In evaluating very small streams, it is important to evaluate prior flow/rainfall records.

Small pristine mountain streams also have been found to have reduced taxa richness and North Carolina is in the process of deriving special criteria for these areas. Preliminary analysis indicated that these criteria should be applied only to mountain streams with the following physical characteristics:

1. First or second order stream

2. Average width <4 meters

3. Largely closed canopy (70-100%)

4. No abundant Aufwuchs growths

Given these characteristics, we would define areas with an Excellent bioclassification based on EPT taxa richness (>27 for Abbreviated samples, >30 for Standard samples), ratio of EPT S/Total S (>0.5), Few Odonata, Coleoptera and Mollusca (<10% of total taxa richness), a biotic index value (still being derived) and the presence of species characteristic of small streams. A list of "small stream" taxa also is currently being developed from our data base. All of the above classification criteria are in review, and some minor changes are expected.

<u>Microhabitat</u>. Examination of individual samples has often indicated species with a "clumped" spatial distribution. This problem can be overcome by the use of larger samples, especially composite samples. This is the strategy implicit in "traveling kicks", many types of D-frame or pond-net collections, and North Carolina's composite collections. Our multiple-habitat semi-quantitative sampling should help to reduce microhabitat variations.

Jenkins et al. (1984) recommended sampling at least three habitats to adequately inventory the aquatic fauna, especially in relation to the "conservation" value of streams. Brooker (1984) also showed that the effects of habitat change (channelization, etc.) were not properly assessed by riffle-only collections. Cuff and Coleman (1979) showed that overall precision was increased by taking single samples from many stations, rather than by taking many replicates at a single site. This analysis would seem to support a multi-habitat sampling design.

Changes with Time

Seasonal Changes. Individual macrobenthic species are well known to exhibit marked seasonal changes in abundance (Hynes 1972). Overall seasonal changes in community structure are more difficult to form generalizations about, but we should expect considerable between-ecoregion and between-year differences, largely due to differences in seasonal temperature regimes. Spring and/or fall peaks in taxa richness have been observed at many of our North Carolina sites, with the spring peaks being the most pronounced. Seasonality changes are not predictable using a "standard" correction factor for each month. Different years may have quite different seasonal patterns, especially with regard to the onset of spring generations. We have also found that greatest seasonal variation occurs at sites with highest water quality, i.e., seasonal variation is reduced at severely polluted sites. Some of the "seasonal" change in slightly impacted streams may reflect a real change in water quality, not a change caused by temperature-related hatching or emergence. The latter is especially true in agricultural areas, where there may be a seasonal input of sediment, nutrients and/or pesticides.

The first step in making seasonal corrections in taxa richness is some knowledge of the life cycles of the invertebrates in each ecoregion (Table 2). Year-round species, or multi-voltine species with no resting stage, have little influence on seasonal changes in taxa richness. However, many species will be absent for a portion of the year, sometimes up to 9 months. Often spring peaks in EPT taxa richness are caused by the addition of many Plecoptera species. This pattern is illustrated in Table 3, comparing EPT taxa richness of single spring collections with average summer data. It is apparent from these examples that a large part of the spring taxa richness increase was caused by the appearance of many plecopteran taxa. In some cases, some adjustment also must be made for increases in Ephemeroptera. Simple subtraction of these species, rather than making the same proportional adjustment for all sites, appears to be the most reasonable means of seasonal adjustment. In all cases, the seasonal adjustment must be validated by comparison with summer data. We have not yet been able to come up with an adjustment scheme that does not require such test sites. The importance of control sites, especially ecoregion reference sites, cannot be overemphasized in making water quality assessments outside of the usual summer collection periods.

Flow. Some "seasonal" changes do not reflect normal shifts in populations, but irregular changes in water quality or habitat quality, often related to flow. Given adequate flow information, it may be possible to predict at least the direction of changes associated with floods and/or droughts. Note that high quality (daily/hourly) flow information is usually available from the United States Geological Survey's monitoring network.

Extreme variation in flow has been shown to have a catastrophic effect on the macroinvertebrate fauna of some streams (Gray 1981). Given some refuge from scouring, however, the invertebrate community can withstand more moderate changes in flow. Data from both King (1983) and Poole and Stewart (1976) indicate that the hyporheic zone may act as a partial refuge from the effects of elevated flow. The invertebrate community, however, seems to have much of its variability caused by changes in flow (Leland et al. 1986, McElravy et al. 1989); some seasonal minima may be more related to floods than to emergence (Chutter 1970).

The effects of drought and flood are often very site-specific, but can be

Table 2. Examples of variations in normal seasonal patterns.[1] Numbers are frequency of collection (0-1) x average abundance value when present (0-10), final values vary from 0-10. Underlining indicates periods of maximum abundance, bold-faced type used to show minima.

A. Year-round taxa: Multiple species/Univoltine or multivoltine with no resting stage

	1	2	3	4	5	6	7	8	9	10	11	12
Stenonema modestum	6.5	6.6	6.3	**5.0**	6.8	6.7	8.1	8.2	6.7	8.2	**5.0**	7.6
Acroneuria abnormis	3.2	2.4	3.3	2.7	**2.1**	**2.0**	3.5	4.8	3.6	4.5	**2.4**	**2.4**
Stenacron interpunctatum	**0.4**	**0.8**	1.5	2.0	3.7	1.2	3.0	3.4	2.0	2.0	1.4	1.3
Isonychia spp.	3.4	1.9	2.7	2.0	3.0	4.3	5.3	6.4	3.3	4.1	2.4	2.3
Hydropsyche sparna	1.9	1.3	3.1	1.2	1.4	1.6	2.3	3.7	1.1	2.0	1.3	1.3
Cheumatopsyche spp.	4.7	4.0	6.0	3.7	7.4	5.7	7.4	8.0	5.5	6.3	4.5	9.8

B. Almost Year-round species, with periods of distinct absence or minima.

	1	2	3	4	5	6	7	8	9	10	11	12
Baetisca carolina	1.1	0.4	0.3	0.6	0.1	0.4	+	+	0.3	0.2	0.7	0.8
Caenis spp.	+	0.7	0.7	1.6	1.9	2.9	3.5	2.2	1.8	0.6	1.1	**0.2**
Serratella deficiens	0.3	0.7	0.3	1.2	2.5	0.9	1.6	2.3	**0.1**	**0.3**	**0.2**	0.4
Heptagenia marginalis	**0.2**	**0.1**	**0.3**	+	0.4	0.5	1.6	2.5	1.2	0.6	0.3	+
Eurylophella temporalis	1.2	1.8	3.4	1.5	3.2	1.0	**0.1**	**0.1**	**0.2**	0.5	0.6	1.5
Neoperla spp.	0.6	0.4	+	+	0.8	0.3	0.5	0.1	0.5	1.8	0.3	0.3
Perlesta spp.	**0.2**	0.8	1.0	3.0	5.4	3.1	1.1	0.4	+	**0.1**	**0.2**	0.6
Trianenodes tarda	**0.2**	**0.2**	+	0.7	0.4	1.0	1.4	1.0	0.6	1.2	**0.1**	0.7
Hydroptila spp.	**0.2**	**0.2**	**0.3**	0.6	0.3	0.9	1.1	1.5	0.3	**0.2**	**0.3**	0.6
Hydropsyche morosa	**0.2**	+	**0.2**	0.4	-	0.2	1.0	1.9	**0.2**	**0.3**	+	0.4

Fast (Short Life Cycle) Taxa: Univoltine with resting stage.

	1	2	3	4	5	6	7	8	9	10	11	12
Danella simplex	-	-	-	-	-	+	0.3	0.9	-	-	-	-
Drunella allegheniensis	-	-	-	-	-	+	0.3	0.9	-	-	-	-
Serratella serrata	-	-	-	-	0.2	0.2	0.1	0.4	+	-	-	-
Baetis pluto	0.2	+	+	0.8	1.8	1.5	1.0	2.0	1.8	3.5	0.7	0.1
Cinygmula subaequalis	-	-	0.7	1.9	0.1	0.3	-	-	-	-	-	-
Drunella walkeri	-	-	0.9	1.0	0.3	0.5	-	+	-	-	-	-
Agapetus spp.	0.4	0.1	0.3	0.4	0.6	0.4	+	+	-	-	-	-
Isoperla namata	1.0	1.0	3.9	1.2	-	0.1	-	-	-	-	-	0.5
Clioperla clio	1.2	1.7	0.3	-	+	-	-	-	-	0.6	1.2	1.5
Leptophlebia spp.	2.3	2.4	1.0	0.1	0.1	+	+	0.1	+	1.1	2.4	3.7
Apatania spp.	4.0	0.7	0.2	-	-	-	-	+	0.2	0.3	1.5	2.0
Strophopteryx spp.	5.1	3.7	0.9	+	-	-	-	-	-	-	0.9	2.6

[1]Numbers are derived from North Carolina's computer data base (1983-present, 1300+ collections), representing a wide range of water quality conditions, ecoregions, seasons, and stream sizes.

Table 3. Evaluation of EPT taxa richness, comparing summer vs.spring collections in three ecoregions of North Carolina.

A.Mountain
French Broad River at Rosman

	Summer Value Mean	(Range)	Spring Value		# Univoltine Taxa with (<6 month) Life Cycles Summer	Spring
Ephemeroptera	20.3	(19-23)	22	(No change)	8	8
Plecoptera	7.0	(6-8)	14	(+7)	0	11
Trichoptera	17.0	(12-20)	19	(No change)	4	3
Total	44.3		55	(+11)		

B. Upper Piedmont
Mayo River at Price

	Summer Value Mean	(Range)	Spring Value		Summer	Spring
Ephemeroptera	18.0	(-)	23	(+5)	9	11
Plecoptera	4.5	(3-6)	13	(+8)	0	8
Trichoptera	16.0	(-)	18	(No change?)	6	3
Total	38.5		54	(+15)		

C. Coastal Plain
Drowning Creek near Hoffman

	Summer Value Mean	(Range)	Spring Value		Summer	Spring
Ephemeroptera	6.5	(5-8)	11	(+4)	2	3
Plecoptera	6.5	(6-7)	12	(+5)	0	7
Trichoptera	16.5	(15-19)	17	(No change)	1	2
Total	29.5		40	(+10)		

broken down into a series of common sense questions:

1. Was there a substantial decline in current velocity that might eliminate high current species? (especially in small streams)

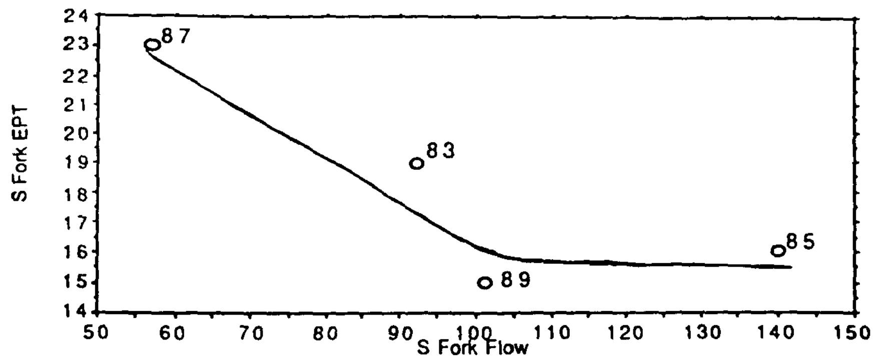
2. Was there a change in scour? (especially for extremely sandy streams with little or no refuge) Was there a refuge from scour and was this refuge included in the samples collected? Refuges include interstitial habitat (especially clean rubble/boulder substrate), snags above the bottom, river weed, etc.

3. Was there a change in dilution of a point source discharger, especially if organic loading was a problem? If there was a significant point source impact, was there a change in length of recovery zone? Note that recovery zones are often shorter under low flow conditions, but with more acute effects close to discharge point.

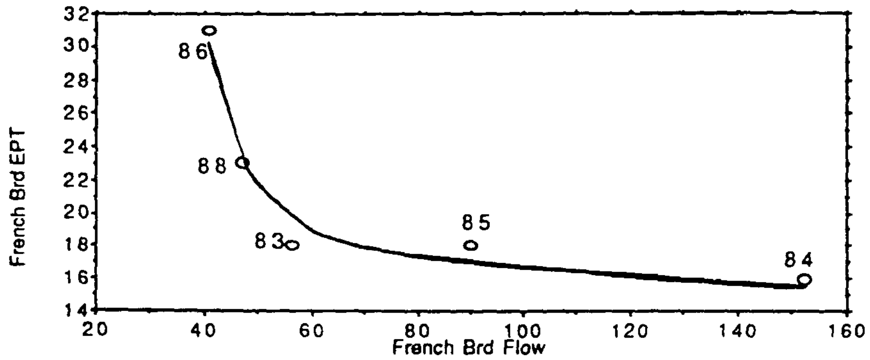
4. Was there a change in the amount of nonpoint runoff, especially if the catchment contains land-disturbing activities?

5. Was there a change in macrophyte growths or the Aufwuchs population caused by a change in transparency, scour, and/or nutrient concentration?

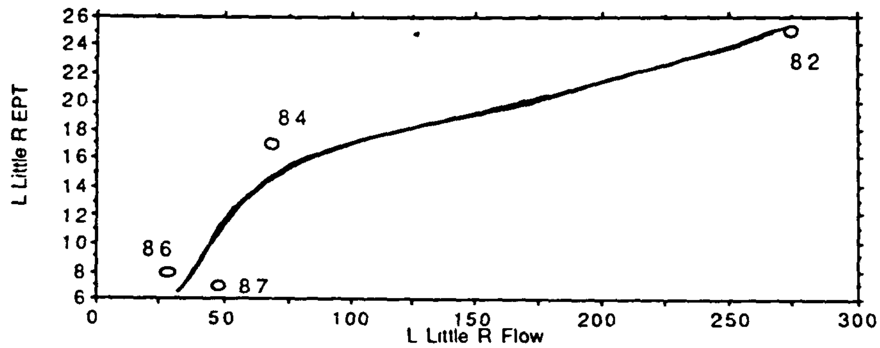
Separating out the possible effects of changes in flow regimes from real changes in water quality is the task of most trend monitoring networks.



A. South Fork Catawba River, 1983-1989



B. French Broad River, 1983-1988



C. Lower Little River, 1982-1987

Figure 3. Examples of flow (as % of normal) vs. EPT Taxa Richness: South Fork Catawba River at MacAdenville, French Broad River at Marshall and Lower Little River at Manchester.

North Carolina has had such a network in place since 1983, and samples have been taken after both drought and flood conditions. A few examples have been drawn from this data base toillustrate possible complications caused by between-years changes in flow.

Figure 3 shows flow (as percent of average flow) for three sites. Two of these sites (Figure 3A and 3B) illustrate results from catchments affected by nonpoint runoff. For both the French Broad River at Marshall and the South Fork Catawba River at McAdenville, there was an inverse

relationship between flow and EPT taxa richness. Low flows, especially during the summers of 1987-1988, were associated with an increase in EPT taxa richness, but it is unlikely that this changes represent a true long-term change in water quality.
The third site is the Lower Little River at Manchester. There is a municipal wastewater treatment plant above this station, with a permitted flow of 8.0 MGD. During high flow years, (1982, 1984) relatively high EPT taxa richness values were recorded. Low flow years, however, provided little dilution for the wastewater discharge, and EPT taxa richness declined sharply. Changes in flow probably contribute to the decline in taxa richness at the Lower Little River site, although this information does not preclude the possibility of an actual decline in water quality as well.

Summary
Many factors affect the variability of benthic macroinvertebrate data. Much of this variability can be reduced by appropriate choices of sample sites, collection method, identification level, and analysis techniques. Variability can also be reduced by making corrections for predictable changes associated with habitat characteristics (ecoregion, stream size) or the time of the year. In the absence of specific corrections methods, analyses should be supported by a comparison with ecoregion reference sites. The effects of changes in flow are less predictable that habitat associated changes, but the general trend can be evaluated based on land use, ecoregion, stream size and the presence of point source dischargers.

North Carolina is in the process using a computerized data base to correct biocriteria for predictable variation in taxa richness based on ecoregion, stream size and seasonal changes. Collections in very small streams or during spring months can be expected require some adjustment before applying biocriteria. Our data base is also being used to derive tolerance values for a Hilsenhoff-type biotic index.

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The Use and Variability of the Biotic Index to Monitor Changes in an Effluent Stream Following Wastewater Treatment Plant Upgrades

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Abstract
In 1982 the Madison Metropolitan Sewerage District began an intensive study of the Nine Springs wastewater treatment plant effluent stream Badfish Creek. The purpose of this study was to provide baseline data to monitor changes in the aquatic communities that may have occurred as a result of upgrades in the wastewater treatment plant that were completed in 1986.

The biotic index was determined for replicates of Kick net (6) and artificial substrate (3) samples at 4 sites along Badfish Creek at ca. 5 mile intervals from the headwaters progressing downstream. There was a definite improvement in water quality ratings at all stations from spring 1983 - spring 1988. Generally all sampling stations improved at least one water quality rating during this period and these improvements were probably due to upgrades in the wastewater treatment plant. There were some differences in spring and fall BI values, however these differences were not substantial. Artificial substrate samples generally had lower BI values and water quality ratings than the kick net samples taken at the same station and time. Approximately 39% of the comparisons of mean BI values of kick net and artificial substrate samples had different water quality ratings.

Kick net samples were overall slightly more variable (CV = 4.5%) than artificial substrate samples (CV = 2.7%). The standard deviation of the kick net samples was 0.31 which is comparable to other studies and the standard deviation of the artificial substrate samples was 0.19.

Introduction
The Madison Metropolitan Sewerage District (MMSD) began a detailed aquatic macroinvertebrate study in 1982 on Badfish Creek which is a receiving stream for the Nine Springs wastewater treatment plant (Fig. 1). The purpose of this study was to provide baseline data to monitor changes in the aquatic communities that may have occurred as the result of upgrades in wastewater treatment, that were completed in 1986. The District also anticipated that these biosurvey data might be an important tool in future years when examining necessary permit limits.

The MMSD treats wastewater from the City of Madison and surrounding communities, comprising ca. a 149 square mile service area, at the Nine Springs wastewater treatment plant. The plant is an activated sludge, advanced secondary treatment facility. The plant was upgraded in 1986 to gain advanced secondary treatment status which included: in-plant nitrification, larger plant size allowing longer retention time which lowered suspended solids and biological oxygen demand in the effluent, a switch from chlorination to ultraviolet disinfection, and bank stabilization (riprap) of three key sections of Badfish Creek.

Changes in effluent water quality due to the treatment plant improvements discussed above have been significant

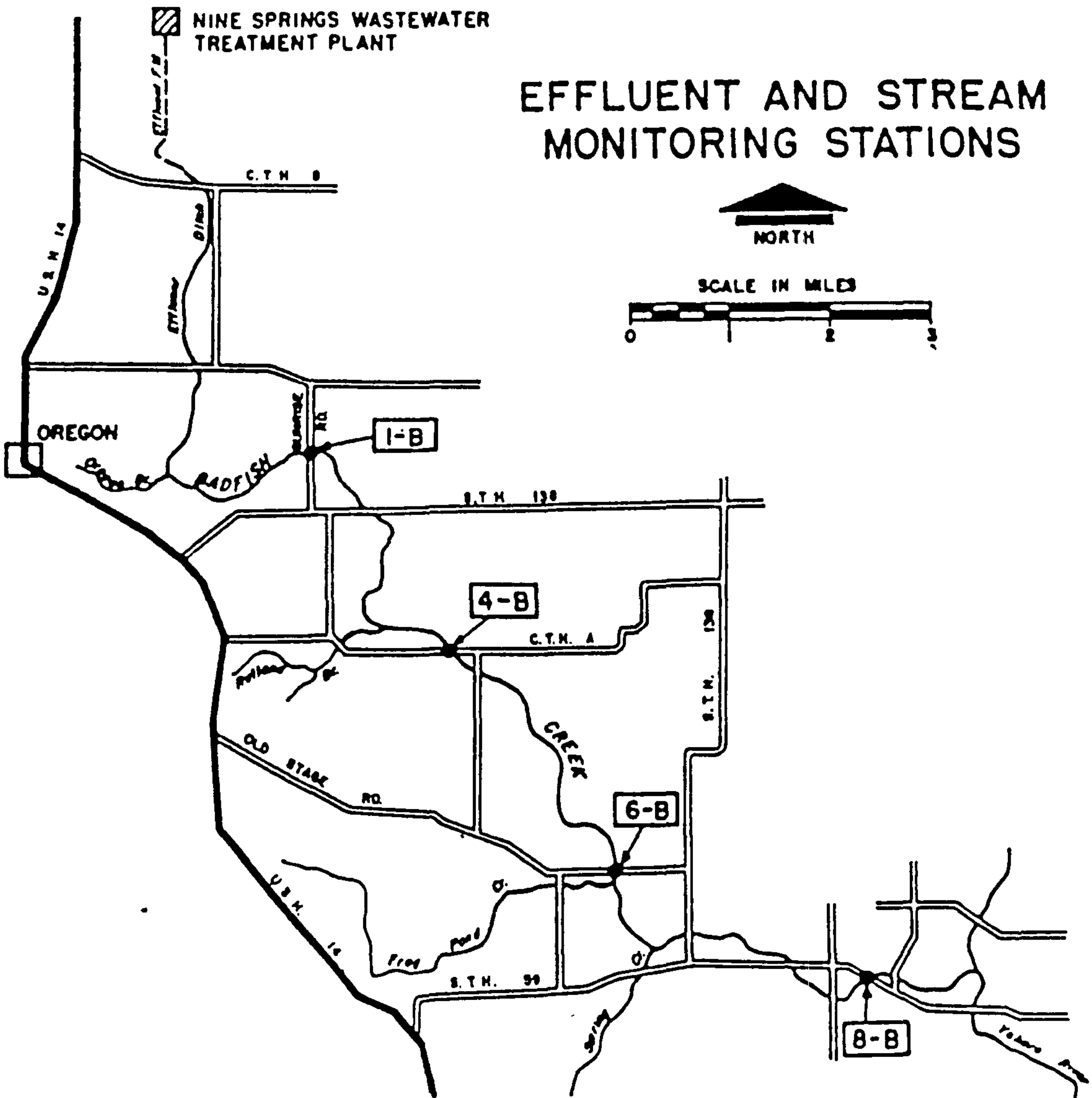

Figure 1. Biotic index sampling stations on Badfish Creek, Dane Co., Wisconsin.

and include a decrease in free ammonia from 9-15 ppm prior to 1985 to less than 0.2 ppm after April 1986. Total suspended solids also decreased substantially from 10-15 ppm prior to 1986 to ca. 5 ppm after April 1986. Biological oxygen demand decreased from 15-20 ppm prior to 1986 to 2-6 ppm after April 1986.

Treated wastewater from the Nine Springs facility has been discharged to the headwaters of Badfish Creek since December 1958. The effluent travels through an underground pipeline for ca. 5 miles to where it surfaces at the headwaters of Badfish Creek. The plant currently discharges 37 million gallons per day to the creek, which constitutes about 80% of the flow in the upper reaches. Inflows from 4 tributaries (Oregon Branch, Rutland, Spring and Frog Pond Creeks) and other surface runoff increases the flow to ca. 67 million gallons 20 miles downstream near the Yahara

River. The effluent constitutes ca. 50% of the flow prior to entering the Yahara River.

Materials and Methods

Four macroinvertebrate sampling sites were established on Badfish Creek which were positioned at ca. 5 mile intervals from the headwaters progressing downstream (Fig. 1). Biotic index (BI) samples (Hilsenhoff 1982, 1987) were taken in the spring (April) and fall (October) from spring 1983-spring 1988 (1988 fall samples were not included).

Aquatic macroinvertebrates were collected with a standard D-frame kick net which had a 12 in diameter opening and a 1 mm mesh bag. Samples were collected by kicking and disturbing the substrate upstream from the net for ca. one minute (Hilsenhoff 1982, 1987). Six kick samples were taken at each of the 4 sites using stratified random sampling procedures. One sample each was taken at the center and to the right and left of the center of the creek, 2 samples were taken near one bank and one sample was taken along the other bank for a total of 6 replicate kick net samples. After kick samples were taken they were vigorously washed in the net to remove fine sediments. The remaining debris and organisms were placed in labeled pint jars and preserved in 95% ethanol.

Artificial substrate samplers, similar to those described by Beak et al. (1973) were also used to collect BI samples. These samplers consisted of a 16 in dia., 1 in deep aluminum pizza pan with two expanded metal mesh inserts placed inside the tray one on top of the other, which served as the colonization substrates. The tray and inserts were attached to a cement anchor block that was placed on the stream bottom. These trays worked well because the problems of vandalism and organic debris build-up were minimized by their low profile. Three trays were placed across the creek at each sampling site. One tray each was placed at the center and near each bank at each sampling site 6 weeks prior to the fall and spring sampling dates to allow sufficient time for colonization to occur. A special retrieval top was used to retrieve the trays after the 6 week colonization period. This method ensured minimal loss of organisms during the retrieval process. The inserts and trays were rinsed into a 1 mm mesh soil sieve to remove fine sediments and retain the macroinvertebrates. The organisms and retained debris were then placed in labeled pint jars and preserved with 95% ethanol.

Aquatic macroinvertebrates were removed from debris in the laboratory and the contents of each sample were placed in a screen and rinsed to remove the alcohol and remaining fine sediments. The samples were then placed in a 10 in X 16 in white enamel pan and evenly spread over the pan bottom in water. A plexiglass grid (1.5 in high) was placed in the pan which partitioned the sample into 32 squares.

Squares were randomly selected for the kick net samples and all organisms were removed from each square until a total of 150 organisms were removed. If 150 organisms were removed before a square was completed the remaining organisms in the square were also included. Artificial substrate samples were also sorted in the enamel pan and 4 squares (ca. 1/8 of the sample) were randomly selected and all organisms in each square were picked. The grid insert was removed after the 4 squares were picked and the rest of the sample was sorted. The dominant organisms (over 30 count in the 4 squares) were not picked in the remaining sample but those picked from the 4 squares were multiplied by 8 to approximate the total numbers in the sample. All remaining non-dominant organisms were picked and counted.

Aquatic macroinvertebrates were identified to the lowest possible taxonomic unit. Chironomidae (Diptera) were slide mounted with Hoyers mounting media and allowed to clear to facilitate identification.

The BI was determined for each sample (Hilsenhoff 1982, 1987) and a mean BI was determined for each 6 replicate set of kick samples and 3 replicate set of artificial substrate samples. The BI was originally designed to detect problems with low dissolved oxygen caused by organic loading of putrescible wastes and it appears to work well for that purpose (Hilsenhoff 1977, 1982, 1987). It has been widely used by many state agencies and is the standard rapid bioassessment measure used by the WI Dept. of Natural Resources for water quality assessments. This index provides water quality ratings based on a numerical system of 0-10 with 0 indicating very good water quality and no organic pollution and 10 indicating very poor water quality and severe organic pollution (Table 1). The coefficient of variability (CV) and the standard deviation (STD) of the means were used to estimate data variability. The CV and STD were determined for each replicate set of kick net and artificial substrate samples for each sampling period.

Results and Discussion

Generally, BI values decreased and water quality ratings improved at all sampling stations from 1984 to 1988 and BI values increased at all stations from 1983 to 1984 (Table 2; Figs. 2-5). The most substantial improvements in water quality ratings occurred in the spring of 1985 and improvements continued through the spring of 1986 after which ratings for all stations appeared to stabilize (Table 3).

Station 1B had consistently the poorest water quality ratings of all stations (Table 3). This is not surprising since it has the greatest percentage of effluent of all stations and is closest to the MMSD wastewater treatment plant. Water quality at station 1B improved from very poor in 1983-1984 to fairly poor in the spring of 1988. Water quality at station 4B improved from a poor or very poor rating in 1984 to a fair rating from fall 1985 to spring 1988. Stations 6B and 8B improved from a general rating of fair - fairly poor in 1983-1984 to a fair - good rating in spring of 1988. Stations 48, 6B and 8B generally had similar water quality ratings after spring of 1985 (Table 3). These improvements in water quality from 1983-1988 are most likely due to the improvements and upgrades in the MMSD treatment plant including decreases in free ammonia, total suspended solids, and biological oxygen demand, and the change from chlorination to ultraviolet light for disinfection.

Fall samples generally had slightly lower mean BI values (determined from the replicate sets) than spring samples. Fifty five percent of the kick net samples and 65% of the artificial substrate samples had lower BI values in the fall than spring (Table 2; Figs. 6-9), however only 16% and 18% respectively of the kick net and artificial substrate sample comparisons of spring and fall data had different water quality ratings (Table 3). The absolute mean difference between spring and fall kick net samples was 0.47 ± 0.43 and 0.75 ± 0.77 for artificial substrate samples. Hilsenhoff (1988) recommended that BI samples be taken 60 days after the 440 degree day accumulation in warm-water streams and 45 days after the 1050 degree day accumulation in cold-water streams. Badfish Creek is classified as a warm-water stream and the fall samples were taken at least 45 days after the 440 degree day accumulation.

Approximately 39% (17) of the 44 comparisons of mean BI values from replicate sets of artificial substrate (3 replicates) and kick samples (6 replicates) taken at the same time and stations had different water quality classifications. Artificial substrate

Table 1. Water quality classifications for the biotic index (from Hilsenhoff 1987).

BIOTIC INDEX VALUES	WATER QUALITY CLASSIFICATION	DEGREE OF ORGANIC POLLUTION
0.00-3.50	**Excellent**	No apparent organic pollution
3.51-4.50	**Very Good**	Possible slight organic pollution
4.51-5.50	**Good**	Some organic pollution
5.51-6.50	**Fair**	Fairly significant organic pollution
6.51-7.50	**Fairly Poor**	Significant organic pollution
7.51-8.50	**Poor**	Very significant organic pollution
8.51-10.00	**Very Poor**	Severe organic pollution

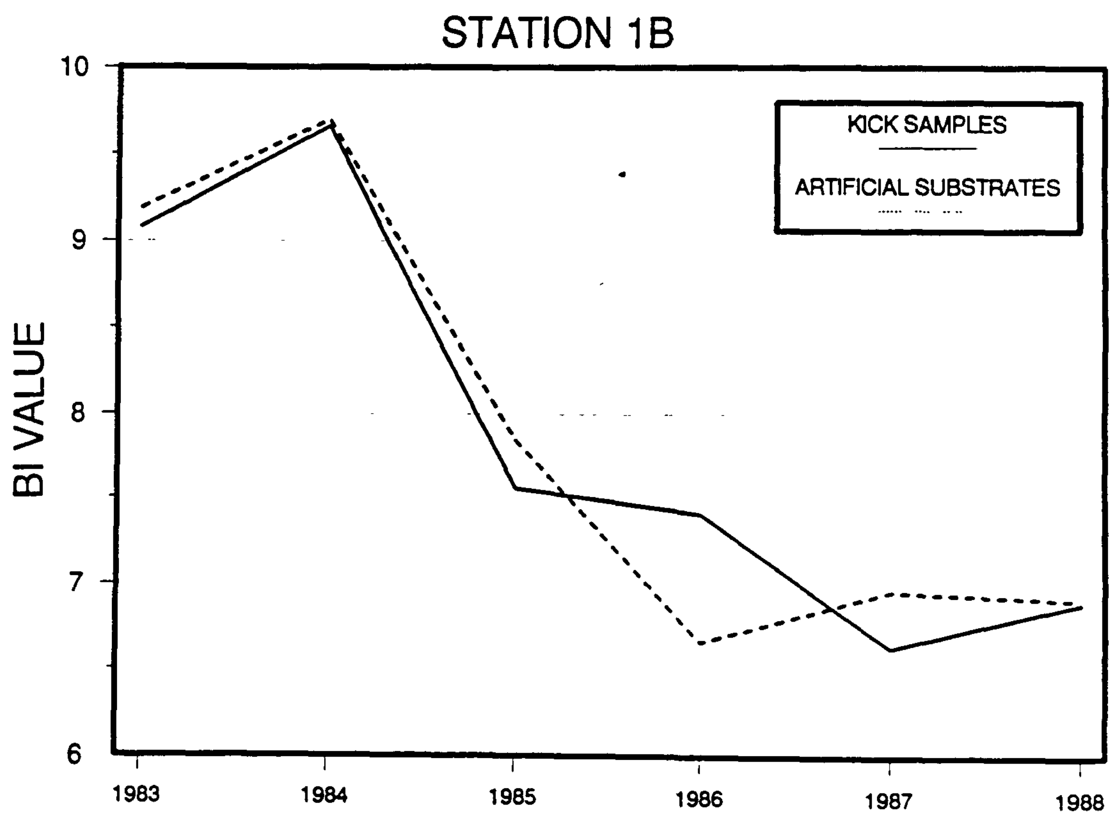

Figure 2. Mean BI values of the replicate set of kick net and artificial substrate samples (spring and fall data combined) from spring 1983 - spring 1988 for sampling station 1B from Badfish Creek, Dane Co., Wisconsin.

Table 2. Seasonal BI values by year and station (values are means and standard deviations of the replicate sample sets - 6 samples for kick net and 3 samples for artificial substrates).

Year/Season Type of sample	Sampling Stations 1B	4B	6B	8B
1983/Spring				
KS[1]	9.23 ± 0.32	7.12 ± 0.44	6.93 ± 0.16	6.57 ± 0.22
AS[2]	9.25 ± 0.30	6.37 ± 0.31	6.80 ± 0.02	6.47 ± 0.11
1983/Fall				
KS	8.92 ± 0.24	7.10 ± 0.19	6.84 ± 0.22	6.73 ± 0.39
AS	9.12 ± 0.12	8.03 ± 0.42	7.00 ± 0.10	6.09 ± 0.07
1984/Spring				
KS	9.51 ± 0.22	7.82 ± 0.83	7.43 ± 0.22	6.65 ± 0.30
AS	9.57 ± 0.32	7.09 ± 0.37	6.91 ± 0.09	6.35 ± 0.13
1984/Fall				
KS	9.81 ± 0.05	9.19 ± 0.12	8.42 ± 0.39	7.15 ± 0.27
AS	9.82 ± 0.03	9.73 ± 0.07	8.31 ± 0.83	6.46 ± 0.22
1985/Spring				
KS	8.40 ± 0.90	6.74 ± 0.57	6.69 ± 0.15	6.25 ± 0.12
AS	9.05 ± 0.21	6.32 ± 0.09	6.51 ± 0.41	6.10 ± 0.06
1985/Fall				
KS*	6.71 ± 0.10	6.47 ± 0.02	6.27 ± 0.27	6.68 ± 0.60
AS	6.62 ± 0.09	6.23 ± 0.10	6.37 ± 0.18	5.93 ± 0.12
1986/Spring				
KS	7.70 ± 0.62	6.35 ± 0.20	6.09 ± 0.32	6.38 ± 0.51
AS	6.79 ± 0.30	5.97 ± 0.09	6.09 ± 0.13	5.97 ± 0.16
1986/Fall				
KS	7.12 ± 0.40	6.33 ± 0.14	5.54 ± 0.23	6.17 ± 0.42
AS	6.52 ± 0.28	5.83 ± 0.20	5.70 ± 0.60	5.44 ± 0.17
1987/Spring				
KS	6.61 ± 0.09	6.30 ± 0.24	6.40 ± 0.30	6.16 ± 0.09
AS	6.22 ± 0.19	6.10 ± 0.02	6.27 ± 0.13	6.11 ± 0.07
1987/Fall				
KS	6.64 ± 0.14	6.12 ± 0.32	5.64 ± 0.18	5.72 ± 0.56
AS	7.69 ± 0.25	5.55 ± 0.24	5.13 ± 0.05	5.16 ± 0.06
1988/Spring				
KS	6.89 ± 0.46	6.20 ± 0.32	5.79 ± 0.18	5.98 ± 0.38
AS	6.91 ± 0.27	5.94 ± 0.18	5.41 ± 0.11	5.89 ± 0.22

[1] Kick net samples
[2] Artificial substrate samples

Table 3. Seasonal water quality ratings by year and station (water quality ratings from Hilsenhoff 1987).

Year/Season Type of sample	Sampling Stations 1B	4B	6B	8B
1983/Spring				
KS[1]	Very Poor	Fairly Poor	Fairly Poor	Fairly Poor**
AS[2]	Very Poor	Fair*	Fairly Poor	Fair*
1983/Fall				
KS	Very Poor	Fairly Poor	Fairly Poor	Fairly Poor
AS	Very Poor	Poor	Fairly Poor	Fair
1984/Spring				
KS	Very Poor	Poor	Fairly Poor*	Fairly Poor**
AS	Very Poor	Fairly Poor	Fairly Poor	Fair*
1984/Fall				
KS	Very Poor	Very Poor	Poor*	Fairly Poor
AS	Very Poor	Very Poor	Poor*	Fair*
1985/Spring				
KS	Poor*	Fairly Poor	Fairly Poor**	Fair
AS	Very Poor	Fair*	Fairly Poor*	Fair
1985/Fall				
KS	Fairly Poor	Fair*	Fair	Fairly Poor**
AS	Fairly Poor**	Fair	Fair**	Fair
1986/Spring				
KS	Poor	Fair*	Fair	Fair
AS	Fairly Poor	Fair	Fair	Fair
1986/Fall				
KS	Fairly Poor	Fair*	Fair**	Fair*
AS	Fairly Poor**	Fair	Fair**	Good*
1987/Spring				
KS	Fairly Poor**	Fair*	Fair*	Fair
AS	Fair	Fair	Fair	Fair
1987/Fall				
KS	Fairly Poor**	Fair	Fair**	Fair
AS	Poor**	Fair**	Good	Good
1988/Spring				
KS	Fairly Poor	Fair	Fair	Fair
AS	Fairly Poor	Fair	Good*	Fair

[1] Kick net samples

[2] Artificial substrate samples

* Ratings which missed the next poorer water quality rating by 0.20 BI units

** Ratings which missed the next better water quality rating by 0.20 BI units

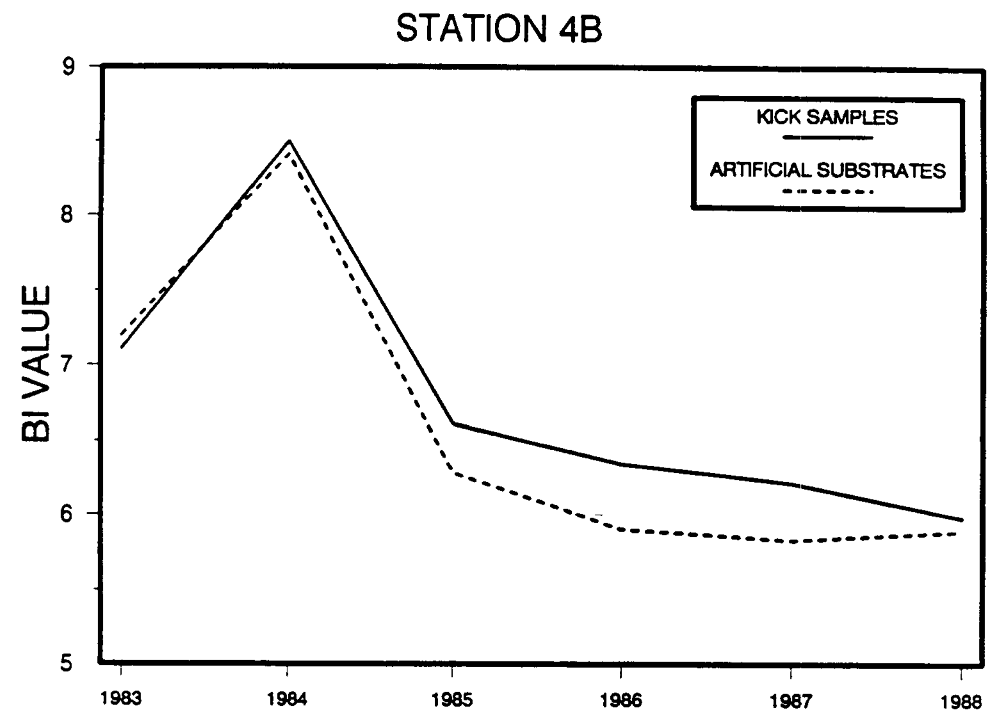

Figure 3. Mean BI values of the replicate sets of kick and artificial substrate samples (spring and fall data combined) from spring 1983 - spring 1988 for sampling station 4B from Badfish Creek, Dane Co., Wisconsin.

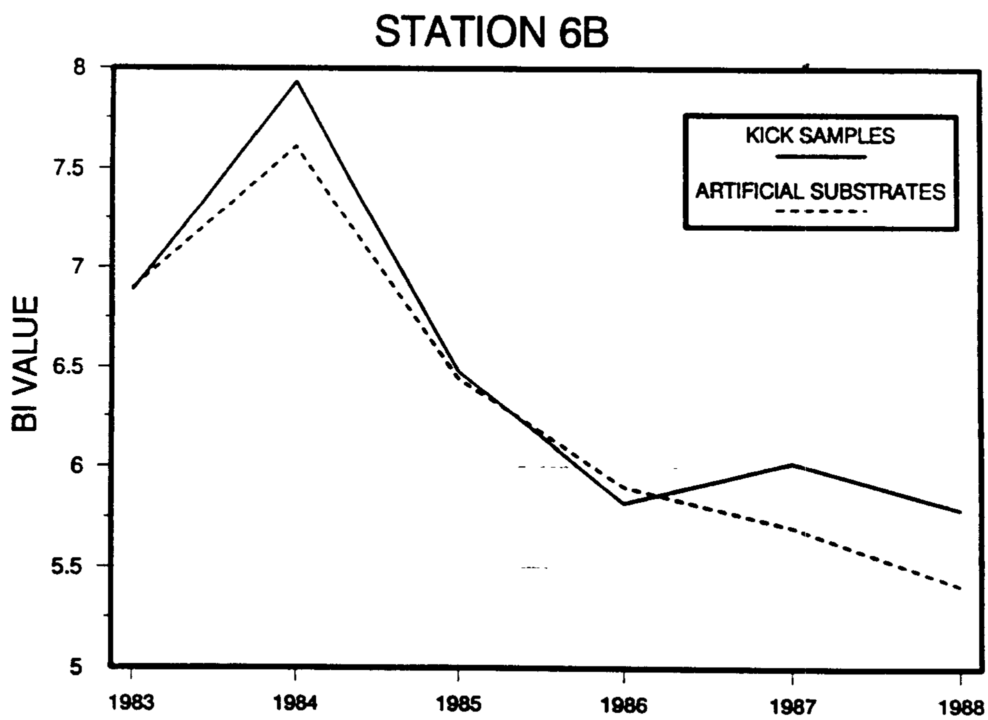

Figure 4. Mean BI values of the replicate sets of kick and artificial substrate samples (spring and fall data combined) from spring 1983 - spring 1988 for sampling station 6B from Badfish Creek, Dane Co., Wisconsin.

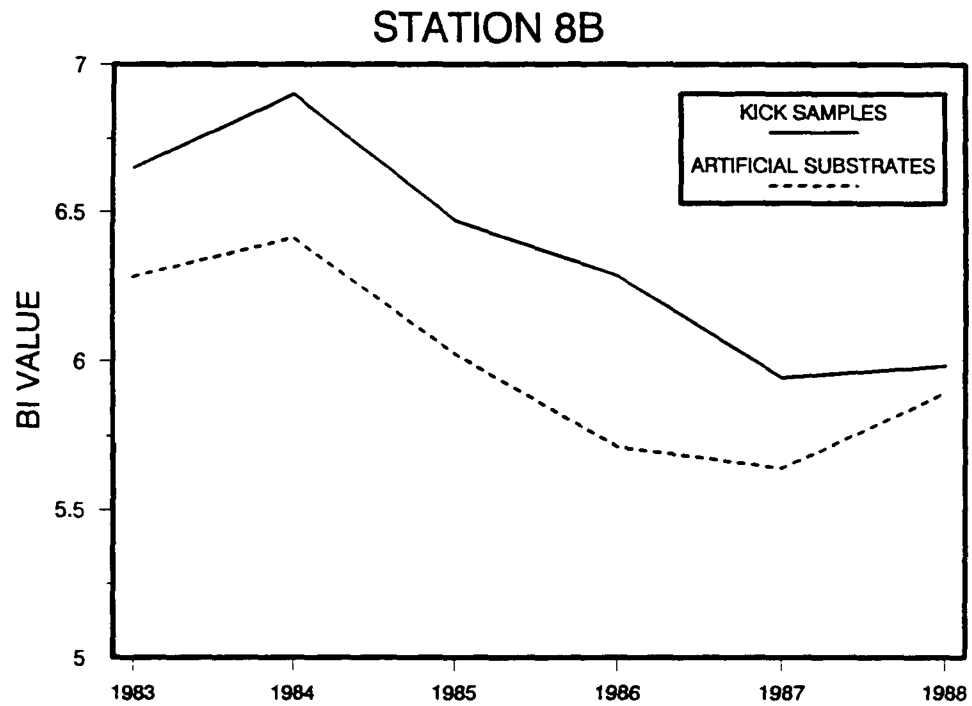

Figure 5. Mean BI values of the replicate sets of kick and artificial substrate samples (spring and fall data combined) from spring 1983 - spring 1988 for sampling station 8B from Badfish Creek, Dane Co., Wisconsin.

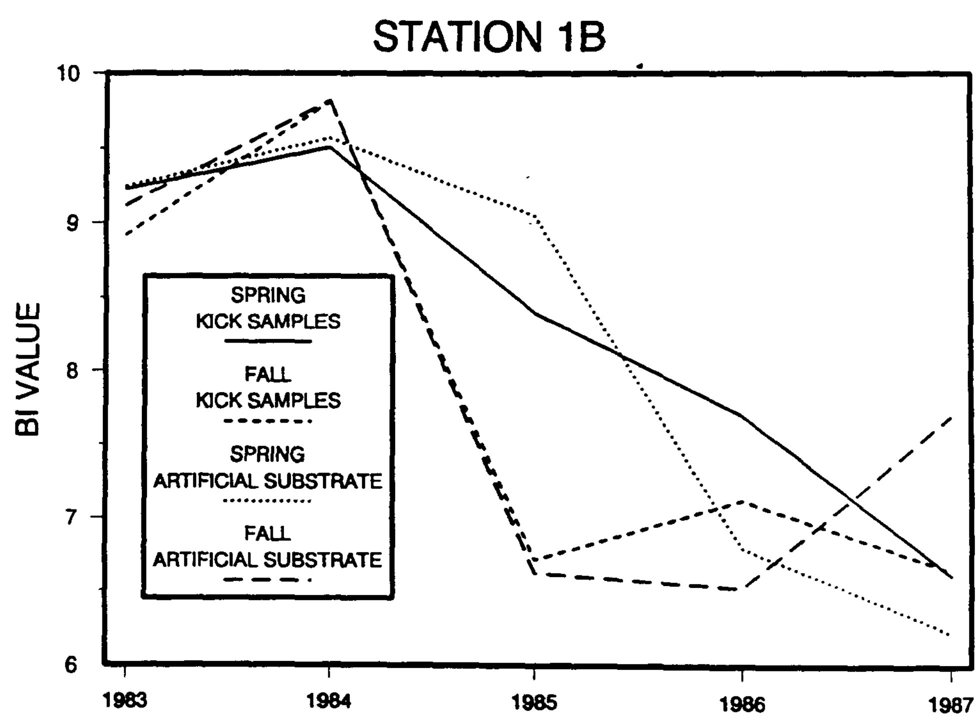

Figure 6. Mean BI values of the replicate sets of kick and artificial substrate samples for spring and fall from spring 1983 - spring 1988 for sampling station 1B from Badfish Creek, Dane Co., Wisconsin.

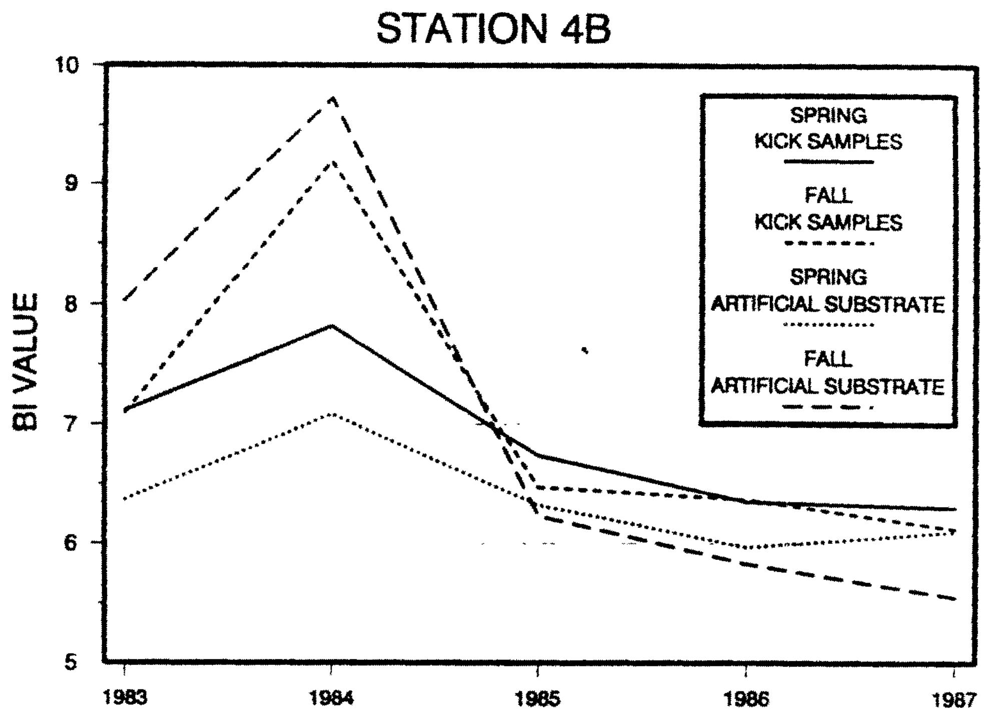

Figure 7. Mean BI values of the replicate sets of kick and artificial substrate samples for spring and fall from spring 1983 - spring 1988 for sampling station 4B from Badfish Creek, Dane Co., Wisconsin.

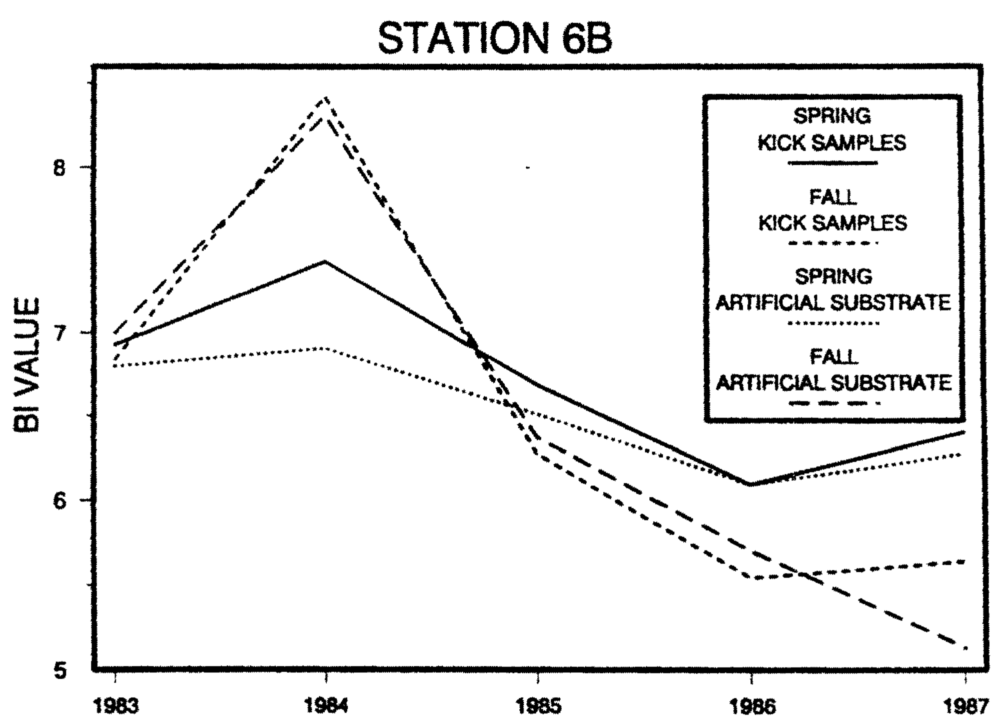

Figure 8. Mean BI values of the replicate sets of kick and artificial substrate samples for spring and fall from spring 1983 - spring 1988 for sampling station 6B from Badfish Creek, Dane Co., Wisconsin.

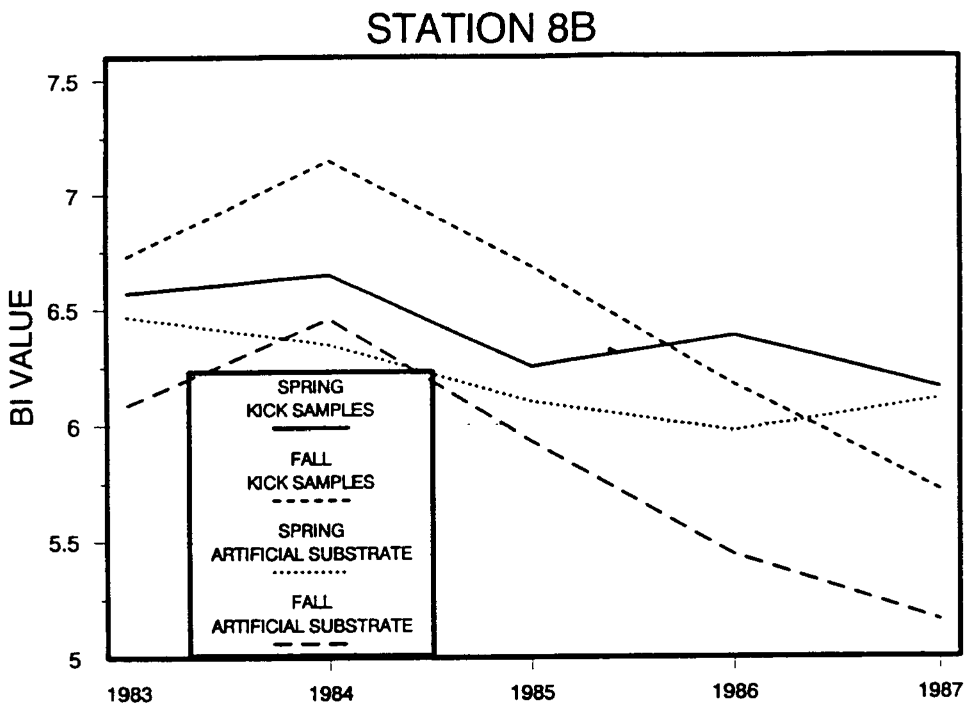

Figure 9. Mean BI values of the replicate sets of kick and artificial substrate samples for spring and fall from spring 1983 - spring 1988 for sampling station 8B from Badfish Creek, Dane Co., Wisconsin.

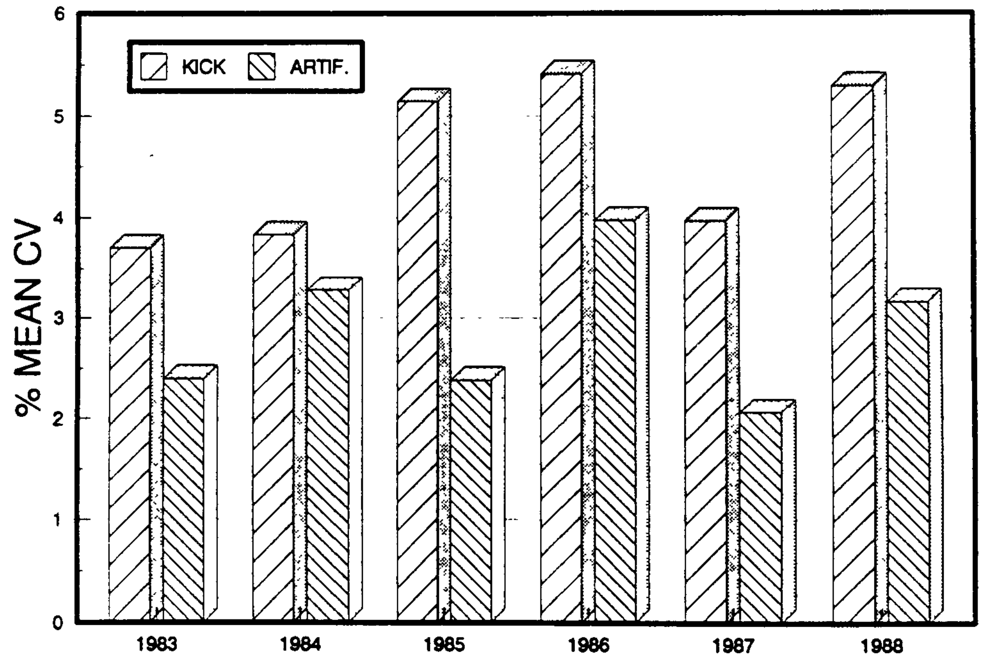

Figure 10. Annual mean coefficient of variation (CV) of the replication sets of kick net and artificial substrate samples (spring and fall and all station data combined) from spring 1983 - spring 1988 from Badfish Creek, Dane Co., Wisconsin.

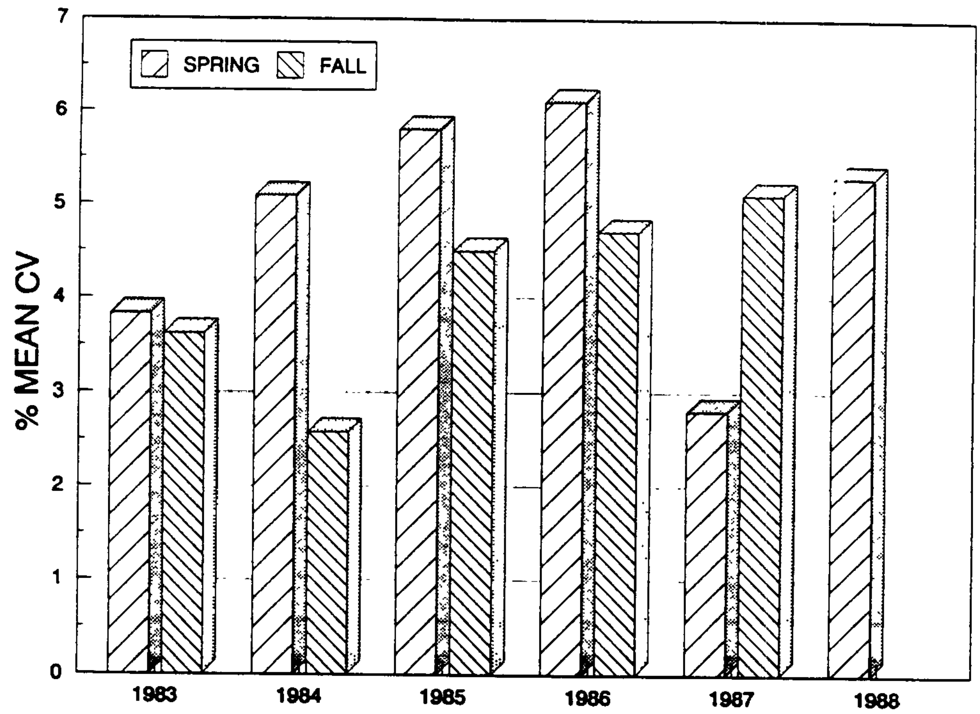

Figure 11. Seasonal mean coefficient of variation (CV) of kick net samples from spring 1983 - spring 1988 from Badfish Creek, Dane Co., Wisconsin.

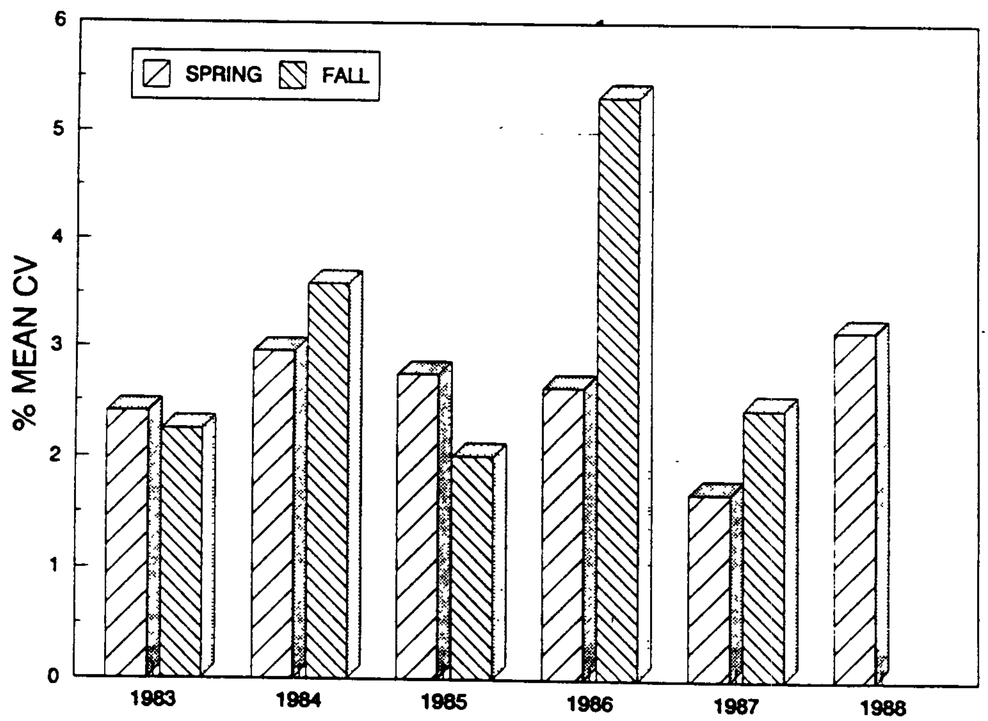

Figure 12. Seasonal mean coefficient of variation (CV) of artificial substrate samples from spring 1983 - spring 1988 from Badfish Creek, Dane Co., Wisconsin.

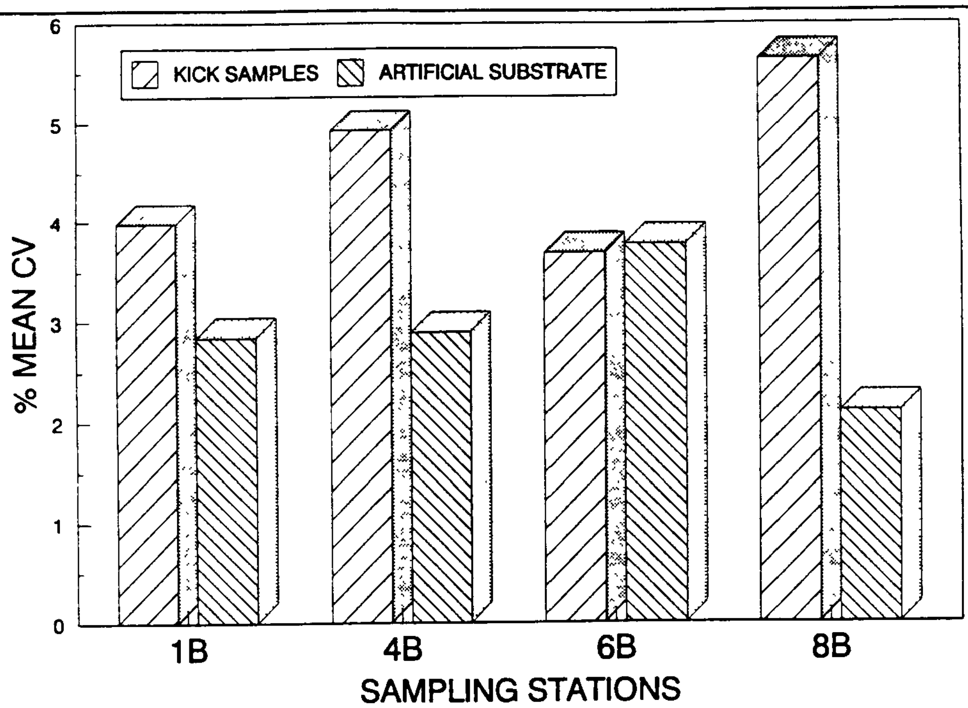

Figure 13. Mean coefficient of variation (CV) of the replicate sets of kick net and artificial substrate samples for each station (annual and seasonal data combined) from Badfish Creek, Dane Co., Wisconsin.

samples had a one range cleaner water quality classification than the kick samples in 15 of the 17 comparisons that were different. Only 2 of artificial substrate samples provide a poorer (one range) water quality rating than the kick samples (Table 3; Figs. 2-9). The absolute mean difference between the means of the BI determined from the replicate sample sets for kick net and artificial substrate samples taken at the same times and stations was 0.37 ± 0.29.

The overall variability of the kick samples (mean CV = 4.5%) was greater than the artificial substrate samples (mean CV = 2.7%) based on comparisons of the means of 6 and 3 replicate sets for kick and artificial substrate samples respectively from spring 1983-spring 1988 (Fig. 10). This slight difference in variability may have been due to the different number of replicates taken for artificial substrate and kick net samples, or to differences inherent in the types of samplers. The overall standard deviation of 6 replicate sets of kick samples was 0.31 which is comparable to values of 0.24 and 0.28 reported by Hilsenhoff (1988) and Szczytko (1988) respectively for biotic index samples. The overall standard deviation of the artificial substrate samples was 0.19.

The variability of biotic index values of replicate sample sets combined from 1983-1988 was slightly lower in the fall (CV = 4.10%) than spring (CV = 4.83%) for kick samples and greater in the fall (CV = 3.17%) than spring (CV = 2.62%) for artificial substrate samples (Figs. 11 & 12). These differences are small and were probably related to sampling and sorting techniques or to the heterogeneous distribution of the macroinvertebrates.

The CV among replicate sample sets was variable from 1983-1988 for both kick net and artificial substrate samples and no annual trends were apparent (Fig. 10). The CV ranged from 3.84-5.41% for kick samples and from 2.06-3.97% for artificial substrate samples during the course of this study. Sampling station 8 had the lowest combined (all years and seasonal data combined) variability (CV = 2.1%) for artificial substrate samples and station 6 had the lowest variability (CV = 3.7%) for kick samples (Fig. 13). There were no obvious trends in variability related to the sampling stations in this study.

Conclusions
There was a definite improvement in water quality ratings determined from BI values in Badfish Creek from spring 1983- spring 1988. Basically all sampling stations improved at least one water quality rating better during the course of this study. These improvements were most likely related to upgrades in the wastewater treatment plant discussed above and the water quality ratings observed in the spring of 1988 are probably the water quality ratings which will continue in the future. Station 1B had the lowest water quality ratings of all stations and station 4B, 6B and 8B had similar ratings by the end of this study.

Generally there were some differences in fall and spring BI values, however we do not view these differences as substantial since part of the variability can be explained by the standard deviation of the means. Also only 16% of the kick net and 23% of the artificial substrate sample comparisons between spring and fall had different water quality ratings and many of the water quality ratings that were different missed the water quality rating of the other season by only 0.20 BI units or less (Table 3).

Artificial substrate samples generally had lower BI values and water quality ratings than the kick net samples taken at the same time and sampling station. These differences were probably related to the different types of samplers and sorting techniques used. In many cases water quality ratings from kick or artificial substrate samplers missed the water quality rating of the other type of sampler by only 0.20 BI units or less (Table 3).

Kick net samples were slightly more variable than artificial substrate samples (mean CV determined from the sets of replicate samples). These differences in variability (1.8%) were small and were probably related to sampling and sorting techniques. The overall standard deviation of the kick net samples were comparable to other studies. The variability of BI samples is lower than most other benthic community metrics currently used for water quality assessment (Szczytko 1988).

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Data Variability in Arthropod Samples Used for the Biotic Index[1]

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Abstract
Factors that influence the reliability of the biotic index (BI) for evaluating the water quality of streams include sample size, substrate sampled, current, method of processing samples, water temperature, time of the year, and level of arthropod identification. Comparison of standard deviations of sample sizes of 50, 100, 150, and 200 indicated that a sample of 100 was adequate for most evaluations. By sampling only riffles, differences in substrate and current were minimized, and differences between riffles in the same stream were not substantial enough (SD=0.25) to alter evaluations made with the BI. Biases were found in samples picked in the laboratory as well as in those picked in the field, but these biases had little effect on the BI. By processing samples in the laboratory more valuable field time is made available. The greatest variability in BI evaluations resulted from seasonal differences in the fauna, with index values being abnormally high in late spring or summer. Much time can be saved by evaluating streams with a family-level biotic index, but precision is lost and the ability to discriminate between various levels of pollution is deminished.

Key Words: Biotic index, Water quality, Pollution, Arthropods, Insects, Data variability, Sampling, Sample bias, Streams

Introduction

In 1977 I recommended using a biotic index (BI) of the arthropod fauna to evaluate the water quality of streams. This index was based on a sample of 100 or more arthropods that were collected with a net from the riffle area of a stream (Hilsenhoff 1977). Species (or genera when species could not be identified) of stream arthropods were assigned tolerance values of 0 to 5, depending on their tolerance to organic pollution, with the most tolerant organisms having a value of five. The BI is the average of tolerance values for all species of arthropods in a sample. After five years tolerance values were revised and several studies relating to sampling procedure and data variability were completed (Hilsenhoff 1982). More recently data from more than 2,000 stream sites were used to further revise tolerance values and a 0 to 10 scale was introduced to increase precision (Hilsenhoff 1987). Since tolerance values of 0 to 10 are assigned to each species there are only 11 categories of arthropods that are use to calculate a BI. This results in less data variability than when several dozen different species are available for collection. A discussion of important factors that introduce variability into the BI follows.

Sample Size

Kaesler and Herricks (1976) found that a sample size of 100 was adequate for evaluation of stream samples with a diversity index. Two sets of six samples of 50 arthropods from Armstrong Creek, Wisconsin were combined in all possible ways to produce three replicated samples of 50, 100, 150, and 200 arthropods (Hilsenhoff 1982). As sample size was increased, standard deviations decreased (Table 1), but when evaluating streams with the BI the gain in precision from a sample of more than one-hundred probably does not justify the extra time needed to

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Table 1. Standard deviations of biotic index values in relation to sample size from two sets of six samples of 50 arthropods, combined in all possible ways to produce samples of 100, 150 and 200. Samples were collected from the same riffle in Armstrong Creek; one set was one picked in the field and the other was picked in the laboratory.

Sample		Standard Deviation	
Size	Number	Field-Picked	Lab-Picked
50	6	0.213	0.347
100	15	0.124	0.205
150	20	0.085	0.146
200	15	0.062	0.103

collect and process a larger sample. If greater precision is desired, replicated samples are recommended.

Sampling Site Differences

Three different riffles in each of six streams were sampled at two-week intervals from April through November in 1984 and 1985 (Hilsenhoff 1988a). The standard deviation of 32 sets of 3 samples from each stream was 0.25 and the 95% confidence limits were + 0.48 (Table 2).

Table 2. Standard deviations (SD) and confidence limits of biotic index (BI) values of samples collected from three different riffles in each of 6 streams on 32 dates over a two-year period.

			Confidence Limits	
Stream	BI	SD	95%	99%
Otter Creek	2.24	0.26	+0.50	+0.67
Trout Creek	2.29	0.33	+0.65	+0.85
Sugar River	4.91	0.26	+0.50	+0.67
Pecatonica R.	5.48	0.25	+0.49	+0.64
Narrows Creek	6.08	0.20	+0.38	+0.52
Badfish Creek	6.46	0.14	+0.28	+0.36
Average	4.58	0.25	+0.48	+0.63

Differences in substrate and current are most likely to affect the fauna; areas with slow currents, especially, tend to be inhabited by insects that are more tolerant of low dissolved oxygen levels and organic pollution. When the BI of three riffles in each of six streams was compared (Hilsenhoff 1988a), significant differences were found in four of the streams (Table 3), but these differences were not great enough to substantially alter the evaluation of any stream. Differences did not appear to be related to current since the riffle with the slowest current had the highest BI value in two of the streams and the lowest in the other two. Most riffles have currents in excess of 0.5 m/sec, which is sufficiently fast so that arthropods will not be stressed in well-oxygenated water. Variability of substrate was most likely responsible for significant differences in the BI of samples from some streams.

Bias in Sample Picking

When arthropods are picked from a sample there is always a distinct bias that favors certain species. In a set of 12 samples from the same riffle in Armstrong Creek that were alternately picked in the field or preserved and picked in the laboratory (Hilsenhoff 1982) distinct biases in picking were obvious (Table 4); at the time the samples were picked I believed that almost every arthropod had been removed from each sample. Active arthropods tend to be preferentially picked in field samples, and if cryptically colored they are difficult to find among the debris in preserved samples. Inactive, cryptically-colored arthropods are difficult to see in field samples, but many change color when preserved in alcohol and are easy to find in the laboratory. Larvae of Optioservus (Elmidae) are an excellent example. When preserved in alcohol they often become distended, exposing white intersegmental membranes that are easily seen. Fortunately these biases usually do not have much effect the BI. In a study of five streams in which alternate samples from the same riffle were picked in the field or in the laboratory (Hilsenhoff 1982) only

Table 3. Analysis of variance of biotic index values of arthropod samples from three riffles with varying currents at low flow in six Wisconsin streams. Samples were collected at 2-week intervals from 18 September to 13 November 1985. (Reproduced with permission from the Great Lakes Entomologist.)

	Current m/sec			Mean Biotic Index				
Stream	1	2	3	1	2	3	SD	F
Otter Creek	0.42	0.53	0.75	2.08	1.83	2.06	0.27	1.37
Trout Creek	0.87	0.81	0.59	3.83	3.33	2.73	0.46	7.23*
Sugar River	0.47	0.38	0.56	5.27	5.62	5.19	0.24	4.49*
Pecatonica River	0.68	0.61	0.65	5.41	5.92	5.82	0.16	14.68**
Badfish Creek	0.48	0.64	0.66	7.11	6.87	7.08	0.17	3.12
Narrows Creek	0.81	0.81	0.71	7.85	7.41	7.13	0.32	6.36*

* P = 0.05   ** P = 0.01

Table 4. The amount of bias in laboratory- and field-picked samples.

	Number of Arthropods			Bias[c]	Bias
Family or Order	Lab[a]	Field	Difference[b]	Ratio	Rank
Perlidae	96	142	+46	+1.48	6(+)
Baetidae	173	176	+3	+1.02	
Ephemerellidae	65	65	0	1.00	
Heptageniidae	40	57	+17	+1.43	8(+)
Other Ephemeroptera	38	38	0	1.00	
Odonata	12	14	+2	+1.17	
Brachycentridae	31	94	+63	+3.03	3(+)
Glossosomatidae	54	12	-42	-4.50	2(-)
Hydropsychidae	245	358	+113	+1.46	7(+)
Corydalidae	38	35	-3	-1.09	
Elmidae adults	38	84	+46	+2.21	4(+)
Elmidae larvae	264	36	-228	-7.33	1(-)
Athericidae	37	48	+11	+1.30	9(+)
Chironomidae	93	73	-20	-1.27	10(-)
Simuliidae	46	54	+8	+1.17	
Tipulidae	51	28	-23	-1.82	5(-)
Gammaridae	54	61	+7	+1.13	
Asellidae	200	202	+2	+1.01	

[a] Adjusted so that laboratory-picked totals equal field-picked totals.
[b] Laboratory-picked sample subtracted from field-picked sample.
[c] Bias ratio is a ratio of the largest number to the smallest.

in the Mecan River was there a significant difference in the BI (Table 5). Here Optioservus larvae (tolerance value of 4) predominated in laboratory-picked samples, while the active but cryptically colored Brachycentrus americanus, B. occidentalis, and Ceratopsyche sparna (all with a tolerance value of 1) were most numerous in field-picked samples. The BI varies most in very clean streams (Tables 2, 6), but since all values below 3.5 are considered to represent "excellent" water quality (Hilsenhoff 1987), these variations in the BI of clean streams are not important. Preserving samples and processing them in laboratory is

Table 5. Comparison of differences (t-test) between means of biotic index values of replicated field-picked and laboratory-picked samples from five streams. SD = standard deviation from the mean.

Stream	Mean Biotic Index Field	Lab.	df	t	SD
Armstrong Creek	2.22	2.08	10	0.84	0.29
Badfish Creek	7.16	7.22	4	0.15	0.50
Mecan River	2.01	3.15	4	14.51**	0.09
Milancthon Creek	3.71	3.80	4	0.28	0.40
Poplar River	5.01	5.08	4	1.76	0.15

** P = 0.01

Table 6. Comparison of differences (t-test) between means of the biotic index (BI) and the family-level biotic index (FBI) of three replicate samples from six streams in mid-April, late-June, early-September and mid-November in 1984 and 1985. SD = standard deviation from the mean. (Reproduced with permission from the Journal of the North American Benthological Society.)

		Mean			SD	
Stream	Year	BI	FBI	t	BI	FBI
Otter Creek	1984	2.43	2.77	4.65**	0.22	0.30
	1985	2.62	3.27	4.90**	0.27	0.37
Trout Creek	1984	2.23	2.52	4.41**	0.45	0.54
	1985	2.61	3.18	4.84**	0.35	0.39
Sugar River	1984	5.49	5.13	7.28**	0.28	0.33
	1985	5.44	4.83	8.73**	0.23	0.28
Pecatonica River	1984	6.31	6.31	0.06	0.19	0.21
	1985	5.81	5.76	0.34	0.20	0.23
Narrows Creek	1984	6.68	6.15	6.67**	0.20	0.34
	1985	6.36	5.83	10.76**	0.18	0.20
Badfish Creek	1984	7.05	6.71	2.20*	0.17	0.30
	1985	6.77	6.24	6.08**	0.15	0.36
All samples					0.24	0.32

* P = 0.05 ** P = 0.01

recommended because much valuable field time is saved.

Seasonal Variability

A recent study (Hilsenhoff 1988a) showed that the greatest variability in BI evaluations resulted from seasonal differences in the fauna (Fig. 1). BI values were highest in summer when water temperatures were warmest, currents were slowest, and species that were collected were those that are most tolerant of low dissolved oxygen. In warm-water streams a substantial rise (usually greater than 1.5) in the BI occurred in late May or June and lasted for about two months. In cold water streams this rise occurred in the summer and was of a lesser magnitude (about 1.0). The timing and magnitude of the late spring or summer elevation of the BI depends on spring temperatures and can be predicted by accumulation of degree days from a

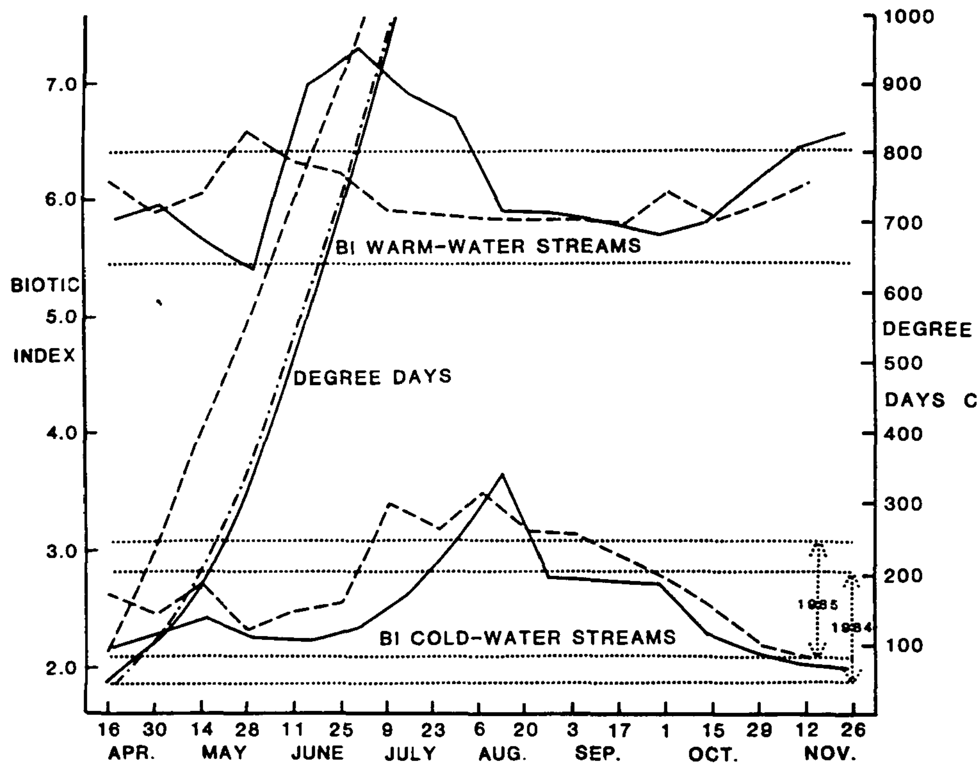

Figure 1. Mean biotic index values of four warm-water streams and two cold-water streams in 1984 (solid lines) and 1985 (dashed lines), with 95% confidence limits (dotted lines) for the mean of the lowest 75% of biotic index values. Comparison of degree day accumulations of mean air temperature above 4.5° C in 1984 (solid line) and 1985 (dashed line) with 1951-1980 average (dot-dash line). (Reproduced with permission from the Great Lakes Entomologist.)

base of 4.5° C (Hilsenhoff 1988a). Using the BI to evaluate streams during the summer months is not recommended.

Family-level Biotic Index
Evaluation of streams with a family-level biotic index (FBI) takes about one-fourth the time required for a BI evaluation that uses species and genera (Hilsenhoff 1988b). This saving of time, however, results in greatly reduced precision and there is a greater chance of making an erroneous evaluation. In organically polluted streams the FBI was substantially lower than the BI and in unpolluted streams it was higher (Table 6); standard deviations were always greater when using the FBI. However, if FBI samples are preserved, a BI evaluation can always be completed at a later date.

Summary
If samples of 100 or more arthropods are collected from rock or gravel riffles at the proper time of the year, sample variability will be held to a minimum and the BI can be used to accurately evaluate the degree of organic or nutrient pollution that has occurred in the stream. Use of the FBI will save considerable time, but the evaluation will be much less accurate.

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Results of Ohio River Biological Monitoring During the 1988 Drought[1]

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Abstract
The Ohio River Ecological Research Program is a long-term monitoring study sponsored by several electric utilities owning coal-fired power plants on the Ohio River (American Electric Power, Cincinnati Gas & Electric Company, Ohio Edison Company, Ohio Valley Electric Corporation, Tennessee Valley Authority). The 1988 drought created anomalous physicochemical conditions in the Ohio River; extremely low flows and elevated ambient water temperatures were observed at plant sites between RM 54-946. Despite potential limiting conditions, monitoring studies indicated diverse and healthy communities. Macroinvertebrate data indicated no consistent differences between upstream/downstream assemblages; substrate quality appeared to be more limiting than water quality at all plant sites. Record high densities of larval fish were observed at most sites in 1988, and total larval species richness was second highest of recent years. A record total 84 species of adult/juvenile fishes were collected throughout the river. Record number of species were collected at five of six plant sites; likewise total abundance of fishes was relatively high at all sites. Spatial differences in fish abundance/biomass were not consistent between upstream/downstream sites at individual plant sites. Drought conditions likely caused displacement of some fish species from inland waters into the Ohio River.

Key Words: Ohio River, Drought, Larval fish, Adult fish, Macroinvertebrates, Thermal effects.

Introduction
The Ohio River Ecological Research Program is a long-term study of aquatic life near once-through cooled power plants on the Ohio River. The purpose of the Program is to: (1) assess potential effects of wastewater discharges (principally once-through cooling water) on nearby aquatic communities; (2) define factors influencing spatial and temporal patterns of biological parameters; and (3) provide inferences on the status of Ohio River water quality based on biological parameters. As a continuation of the Program, biological and water quality data were collected at six coal-fired generating stations on the Ohio River during 1988: Ohio Edison Company's W. H. Sammis Plant (River Mile 54), Ohio Power Company's Cardinal Plant (RM 76.7), Ohio Valley Electric Corporation's Kyger Creek Plant (RM 260), Cincinnati Gas & Electric Company's W. C. Beckjord Plant (RM 453), Indiana Michigan Power Company's Tanners Creek Plant (RM 494), and Tennessee Valley Authority's Shawnee Plant (RM 946). Macroinvertebrates were collected near three plant sites (Cardinal, Kyger Creek, Tanners Creek Plant) whereas ichthyoplankton and juvenile/adult fishes were collected at all plant sites.

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[1] A publication of the Ohio River Ecological Research Program, sponsored by American Electric Power, Cincinnati Gas & Electric Company, Ohio Edison Company, Ohio Valley Electric Corporation, and Tennessee Valley Authority

[2] Chairman, Sponsor Group, Ohio River Ecological Research Program

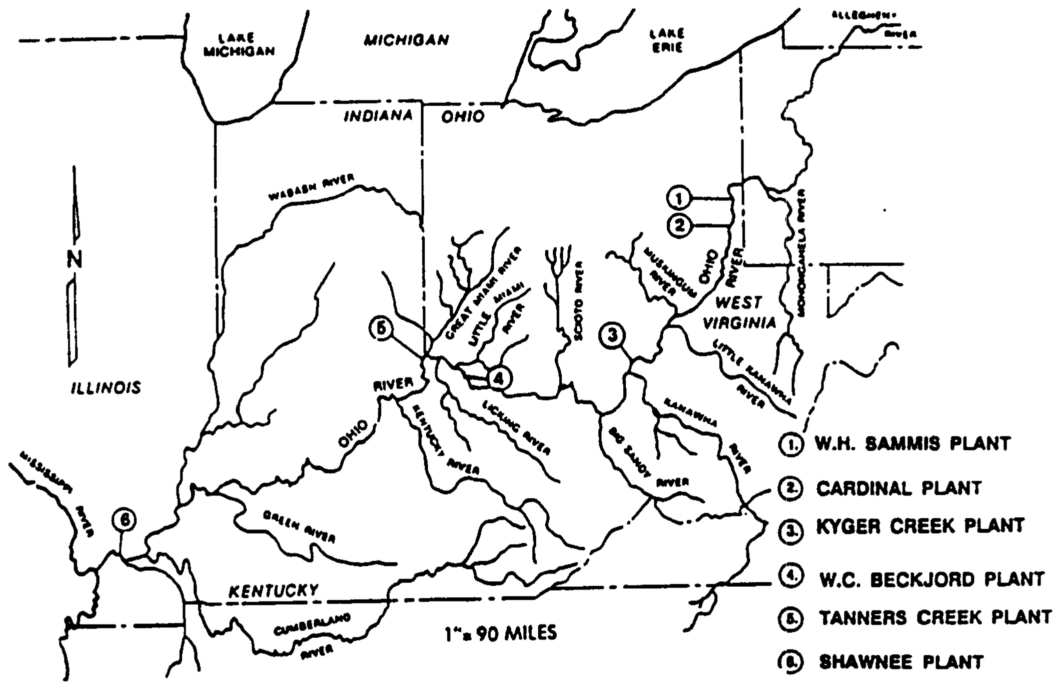

Figure 1. Location of power plant study sites for 1988 Ohio River Ecological Research Program.

During 1988 the Ohio River experienced two perturbations that, collectively, had the potential to cause long-term effects on the ecology of the river. On January 2, 1988 the Ashland Oil Company's Floreffe, Pennsylvania terminal had a release of approximately 750,000 gallons of crude oil into the Monongahela River which soon entered the upper Ohio River. Though long-term effects on the aquatic ecology of the upper river may never be determined, immediate and short-term impacts were generally less than expected.

The second perturbation experienced during 1988 was a prolonged drought. Low rainfall, elevated ambient air and water temperatures, and reduced flow rate all combined to produce potentially critical conditions in the Ohio River, and more crucially, in inland rivers and streams. During the anomalous meterological and hydrological conditions observed in summer 1988 the potential for deleterious power plant effects (e.g., once-through cooling water effects) was considerably increased. Because regulatory agencies issue permits for wastewater discharges based on protection of uses during critical low flows, results of biological monitoring during 1988 were crucial in providing data on the responses of aquatic communities to "worst-case" point source exposure. Likewise, biological monitoring during 1988 enabled dischargers and regulatory agencies the opportunity to assess the appropriateness of temperature criteria variances for Ohio River facilities allowed under Section 316(a) of the Clean Water Act.

Methods and Materials

Water quality, macroinvertebrate, ichthyoplankton, and adult/juvenile fish data were collected at six coal-fired power plant sites on the Ohio River during April through September, 1988. All sampling was conducted by a consultant, Environmental Science and Engineering, Inc. A longitudinal distance of 1,435 km (892 miles) separated the uppermost plant site (Sammis Plant; RM 54) and the furthest downstream site (TVA's Shawnee Plant; RM 946) (Fig. 1). The six plant sites encompass three distinct ecoregions within the Ohio River basin (Western Allegheny Plateau, Interior Plateau, and Interior River Lowland). The boundaries of these ecoregions approximate the traditional geographic delineation of upper, middle, and lower segments of the Ohio River (Pearson and Krumholz 1984, Omernik 1987) A brief description of methods and material for all sampling is given below. Detailed descriptions are given in ESE (1989).

Physicochemical and Flow Measurements
Two or more routine water quality variables (dissolved oxygen, water temperature, conductivity, Secchi disk depth) were measured during all sampling dates at all stations. During weekly ichthyoplankton collections dissolved oxygen and water temperature were measured at stations upstream of power plants (i.e., ambient measurements). All other variables were measured during ichthyoplankton beach seine sampling (semi-monthly) and adult fish sampling (once during May, July, and September).

River flow data were obtained from U.S. Army Corps of Engineers measurements at the following locations: New Cumberland Lock and Dam (Sammis Plant), Pike Island Lock and Dam (Cardinal Plant), Gallipolis Lock and Dam (Kyger Creek Plant), Meldahl Lock and Dam (Beckjord Plant), Markland Lock and Dam (Tanners Creek Plant), and Smithland Dam (Shawnee Plant). River stage data were also obtained, but river stage varied only slightly during summer and fall 1988.

Macroinvertebrates
Macroinvertebrates were sampled at three plant sites (Cardinal Plant, Kyger Creek Plant, and Tanners Creek Plant) during two seasonal surveys. Organisms were collected at two stations using Hester-Dendy artificial substrate samplers and ponar grabs. Sampling stations were located just upstream of the power plant and between 250-1,000 meters downstream of the once-through cooling water discharge. At each station, five replicate Hester-Dendy's were set. Three replicate ponar grabs were taken at the time of Hester-Dendy retrieval.

Two seasonal (temporal) collections of macroinvertebrates were taken at each station. The first colonization period was during mid-May through mid-June and the second period was during mid-July to mid-August.

Ichthyoplankton
Ichthyoplankton (larval fish and eggs) were sampled at all plant sites from April 19 through August 25 using plankton nets and a bag seine. Nighttime ichthyoplankton tows (using 500 u mesh nets having a 1-meter diameter mouth) were taken weekly at two transects upstream of all power plants. Duplicate surface tows and replicate bottom tows were taken at each transect, with a minimum of 50 m^3 water sampled for each tow. A total of 864 ichthyoplankton tow samples were collected in 1988. Bag seine samples were taken weekly from mid-April through July and once in August at all plant sites. A 560 u bag seine was used to sample larval fishes in shallow littoral areas at three stations along the plant shore. A total of 162 beach seine samples were collected.

Adult and Juvenile Fish
Adult and juvenile fishes were sampled using electrofishing, seining, trawling, hoop netting, and gill netting gear. Fishes were sampled during three seasonal surveys (May, July and September) at six stations per plant site. Three stations were located upstream of the plants and three were located downstream of the once-through cooling discharge. Details on field and laboratory processing for all methods are given in ESE (1989).

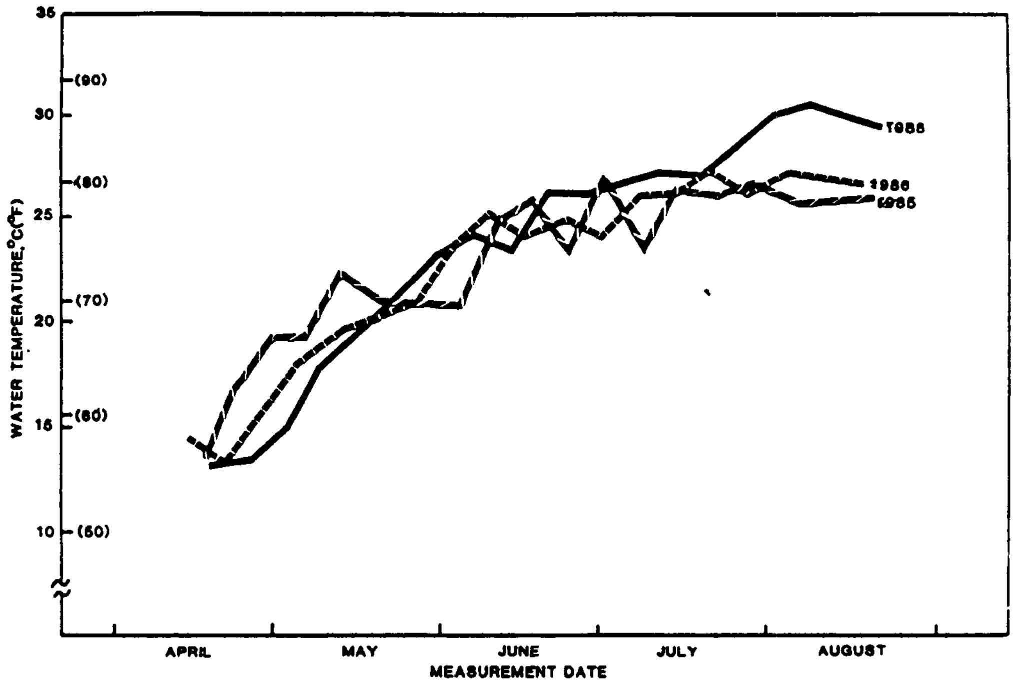

Figure 2. Ambient water temperature measurements taken upstream of Kyger Creek Plant during 1988, 1986 and 1985. Measurements were taken during weekly ichthyoplankton tows.

Results

Physicochemical and Flow Rate

In 1988, ambient water temperatures in the Ohio River approached historical mean values during the months of May and June. Ambient temperatures near 19°C have typically been associated with high densities of dominant larval fishes (gizzard shad, freshwater drum, carp, and carpsucker/buffalo) during previous years.

During 1988 temperatures near 19°C occurred during the week of May 16 in the upper Ohio River and during the week of May 9 in the middle and lower Ohio River, a trend observed in several previous years. Ambient water temperatures during July and August (the months following spawning for several species), however, were higher than historical means at all plant sites. As a site-specific example, July and August water temperatures upstream of Kyger Creek Plant were considerably higher than recent previous years, and temperatures exceeded 30°C during all measurements in August (Fig. 2).

During August at all plant sites, ambient temperatures exceeded maximum allowable stream temperatures as established by ORSANCO (ORSANCO 1987). At Shawnee Plant, ambient temperatures exceeded maximum ORSANCO criteria during several months. These observations indicate that ORSANCO temperature criteria were not derived to reflect anomalous meterological and hydrological conditions, and that generic temperature criteria for the upper, middle, and lower Ohio River may not be appropriate due to differing ambient temperature regimes in the lower and upper sections.

Dissolved oxygen (DO) concentrations were near saturation during all sampling occasions at all sites. The lowest DO concentration recorded during 1988 was 6.0 mg/L downstream of

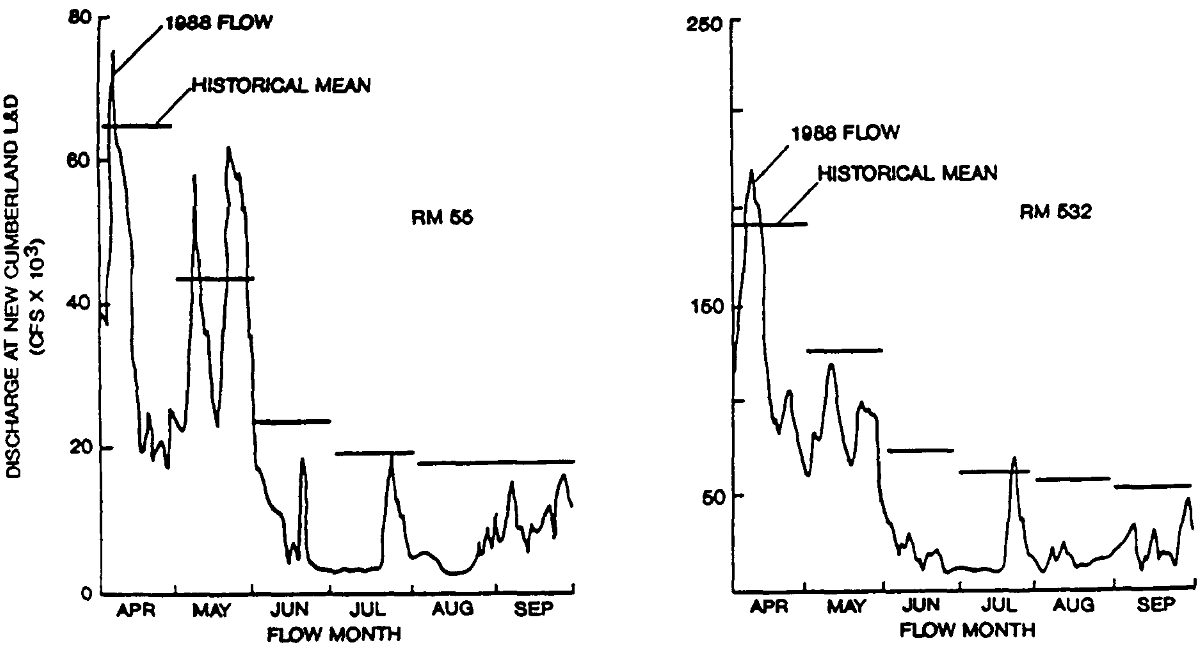

Figure 3. Flow rate measured at New Cumberland Lock and Dam (RM 55) (left) and Markland Lock and Dam (RM 532) (right), April through September, 1988. Historical mean flows indicated by horizontal line for each month.

Kyger Creek Plant in July. Upstream concentrations were just slightly higher on this date, however, averaging 6.4 mg/L. In general, concentrations in the Ohio River were not limiting during 1988 and downstream sites influenced by cooling water discharges at all plants had similar or only slightly lower DO levels.

Flow rates measured at proximal lock and dam locations indicated markedly lower flows in 1988 compared to historical means. Throughout the river, flow rate was highest in spring (April and May), lowest in late June and August, and somewhat higher in September compared to August. The magnitude of deviation of 1988 flows from historical means was related to longitude, and tended to increase downstream. Flow rates near Sammis Plant (RM 54) were below historical means during June through September whereas flow rates near Shawnee Plant (RM 946) were well below historical means for all months studied (April through September); deviation of flow rates from historical means near Tanners Creek Plant (RM 495) was intermediate compared to previously mentioned plant sites (Figs. 3, 4).

Benthic Macroinvertebrates

Combined Hester-Dendy and ponar grab collections (for both surveys) showed total macroinvertebrate densities of 1,459/m^2 at Cardinal Plant and 1,381/m^2 at Tanners Creek. In contrast, combined ponar and Hester-Dendy samples from Kyger Crek Plant had a mean total density of 892/m2. Although a lower mean density was observed at Kyger Creek, the total number of taxa collected was similar at all plant sites (Table 1). The benthic community near Cardinal Plant was dominated by an oligochaete-amphipod complex. An

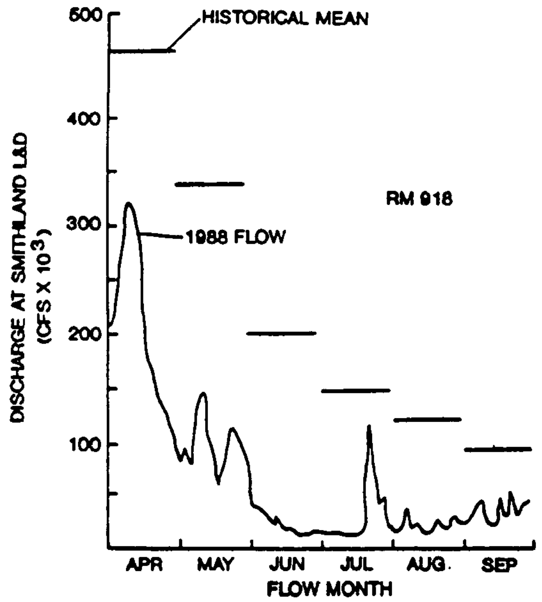

Figure 4. Flow rate measured at Smithland Lock and Dam (RM 918), April through September, 1988. Historical mean flow indicated by horizontal line for each month.

oligochaete-mollusk complex dominated at Kyger Creek plant whereas an oligochaete-amphipod-chironomid assemblage was numerically dominant near Tanners Creek Plant (Table 1).

Temporal variation in macroinvertebrate parameters between upstream and downstream stations was observed at all three plant sites. At Cardinal Plant, marcoinvertebrate parameters during the May-June Hester-Dendy survey suggested a more limited community at the downstream station. Upstream and downstream values (in parentheses) for number of taxa, total density (#/m2) and biotic index were 24(18), 1,199(569) and 4.19(4.18), repsectively. During the late summer survey, however, the upstream station showed a more limited community. Upstream and downstream values (in parentheses) for number of taxa, total density, and biotic index for the July-August survey were 17(20), 312(1,281), and 5.77(7.24).

A similar trend was observed at Kyger Creek Plant. During the first Hester-Dendy survey upstream and downstream values (in parentheses) for number of taxa, total density, and biotic index were 33(30), 665(460), and 4.36(6.18), respectively. For the July-August survey upstream and downstream values (in parentheses) for number of taxa, total density, and biotic index were 24(33), 618(801), and 7.07(6.66), respectivley. These data not only confirm the expected temporal variability of macroinvertebrate parameters in the Ohio River, but indicate that downstream benthic communities were not consistently less diverse and abundant than upstream communities.

At all plant sites, Hester-Dendy samples had consistently greater number of taxa compared to Ponar grab samples. This trend ws observed for both seasonal surveys. These results suggest that substrate characteristics were more limiting than potential water quality effects at all sites studied.

The collection of one macroinvertebrate species in 1988 deserves special mention. Medusae of the freshwater jellyfish (Craspedacusta sowerbyi) were collected in ichthyoplankton tows at all six plant sites. The presence of this species indicates low flow conditions in the Ohio River as this invertebrate is usually restricted to lentic systems (Pennak, 1978).

Ichthyoplankton

For combined tow and beach seine samples at all sites, a total of 492,365 larvae and eggs were collected during 1988. Seventy taxa (including 52 species) representing 13 taxonomic families were identified. This was the second highest total taxa since larval fishes were first collected in 1976. Taxa richness was highest from Shawnee Plant collections, where 47 taxa (34 species) were collected in 1988. Taxa richness was lowest at Tanners Creek Plant (32 taxa, 24 species) and Kyger Creek Plant (33 taxa, 25 species).

Table 1. Benthic macroinvertebrate sampling results at three Ohio River plant locations, May-August, 1988. Values given are for combined upstream and downstream stations and combined May-June and July-August surveys.

Macroinvertebrate Parameter	Cardinal RM 77	Kyger Creek RM 260	Tanner Creek RM 495
Mean density (Hester-Dendy)[a]	840	636	2,769
Mean density (ponar)[a]	2,077	1,148	954
Mean density (combined methods)[a]	1,459	892	1381
Most abundant taxa (combined methods)[b]	Imm. tubificids Gammarus sp. Aulodrilus sp. Limnodrilus sp. Dugesia sp.	Imm. tubificids Limnodrilus sp. Corbicula sp. Gammarus sp. Glyptotendipes sp.	Imm.tubificids Gammarus sp. Glyptotendipes sp. Cricotopus sp. Cyrnellus sp.
Total taxa	54	63	62
Shannon-Wiener diversity	2.49	3.14	3.24
Biotic index	6.18	6.49	7.40

a Densities given as #/m2.
b Most abundant taxa listed in descending order of relative abundance.

Taxa that were abundant at all plant sites included gizzard shad, carp, emerald shiner, carpsucker/buffalo, Morone sp./white bass, Lepomis sp., and freshwater drum. Spotfin shiner, sand shiner, mimic shiner, bluntnose minnow, channel catfish, logperch, and walleye were collected at all plant sites but in fewer numbers. These ubiquitous species have extensive geographic ranges and many can tolerate a wide range of water quality/habitat conditions.

Several taxa were restricted to specific regions of the Ohio River. Larval fishes collected exclusively in the upper ecoregion (Western Allegheny Plateau) were northern hog sucker, shorthead redhorse, rock bass, banded darter, and yellow perch. Larval species restricted to the middle and lower ecoregions included paddlefish, goldeye, speckled chub, bullhead minnow, striped bass, threadfin shad, blue sucker, blue catfish, and brindled madtom.

During 1988 larvae of four species were collected for the first time: pumpkinseed (RM 76), silver lamprey (RM 260), gravel chub (RM 453), and striped bass (three lower plant sites). The collection of a larval lamprey at Kyger Creek Plant was unexpected as ammocoetes of most lamprey species are typically confined to inland streams or rivers. The collection of this specimen may represent actual spawning in the Ohio River or displacement from streams having insufficient flow due to drought conditions.

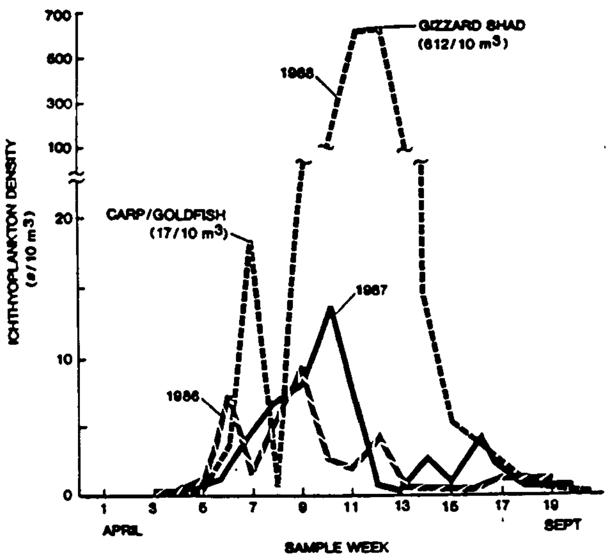

Figure 5. Weekly densities of ichthyoplankton sampled just upstream of W.H. Sammis Plant (RM 54), 1986-1988.

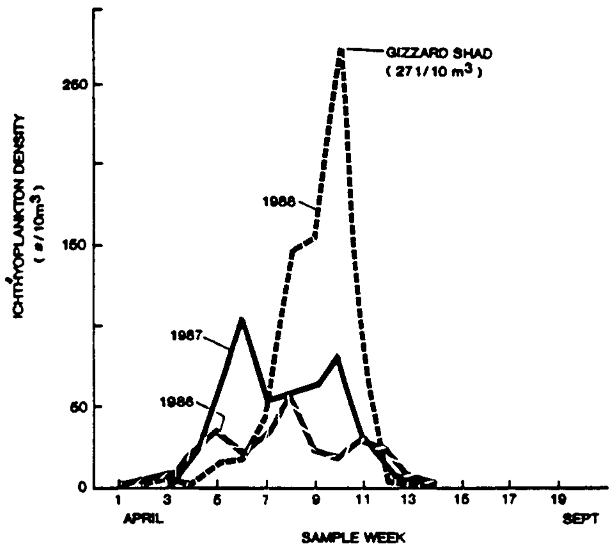

Figure 6. Weekly densities of ichthyoplankton sampled just upstream of Tanners Creek Plant (RM 495), 1986-1988.

Record high densities of ichthyoplankton were observed at five of six plant sites in 1988. Peak densities were highest at W. H. Sammis Plant (635 larvae/10 m3 on June 26) (Fig. 5). This peak density was the highest ichthyoplankton density observed in the history of the Program, and was comprised predominantly by gizzard shad larvae (612 larvae/10 m3).

Gizzard shad or combined herring taxa dominated the peak densities at all other plant sites. Gizzard shad comprised 98% of all larvae during peak densities at Tanners Creek Plant (Fig. 6), and herrings comprised 90% of all larvae during the peak density at Shawnee Plant (Fig. 7). Other taxa collected in considerably greater numbers during 1988 were carp, Morone sp., white bass, Lepomis sp., and Stizostedion sp.

In previous years, total mean density of ichthyoplankton was typically highest at middle or lower Ohio River plant sites. In 1988, however, total mean density was highest at W.H. Sammis Plant (upper river) and lowest at Beckjord Plant (middle river). The chance collection of numerous gizzard shad shortly after a major hatch near Sammis Plant is likely responsible for this observation.

Densities of ichthyoplankton in nearshore areas (beach seine collections) were highest at the two lower plant sites. Beach seine densities were highest at Shawnee Plant (mean density = 255/10 m3) and Tanners Creek Plant (mean density = 118/10 m3). Mean densities at other plant sites ranged between 23 - 70/10 m3.

<u>Adult and Juvenile Fish</u>

In 1988, a total of 90,710 individuals representing 94 taxa (84 species) were collected during adult and juvenile fish sampling. The 84 species collected in 1988 represents the highest species richness during the history of the Program. Forage species were numerically dominant throughout the river, as in previous years. Gizzard shad and emerald shiner accounted for 46% (41,638 individuals) and 27% (24,7470 individuals) of the total species catch, respectively. Channel

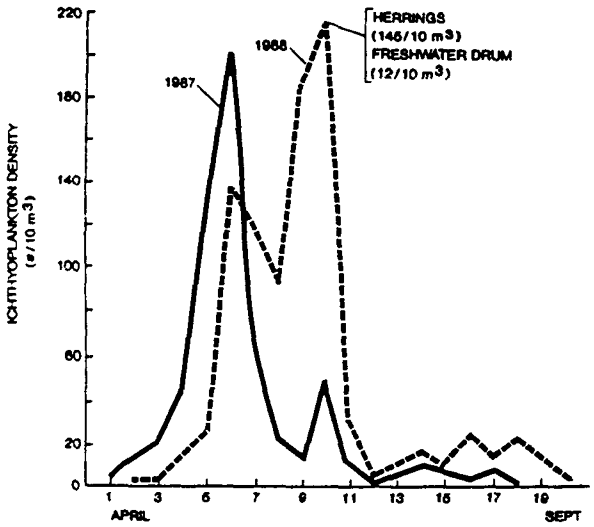

Figure 7. Weekly densities of ichthyoplankton sampled just upstream of Shawnee Plant (RM 946), 1987-1988.

catfish, white bass, bluegill, and freshwater drum were also abundant at all plant sites.

Species collected at plant sites located in the upper ecoregion only included brown trout, river chub, striped shiner, sand shiner, black redhorse, white sucker, rock bass, and several darter species. Paddlefish, shortnose gar, bowfin, threadfin shad, red shiner, and blue catfish were collected exclusively near the Shawnee Plant in 1988. Many of these region-specific collections have been documented in previous years. No longitudinal trends in species richness were evident during 1988, as in previous years. Species richness was highest at Sammis Plant (53 species) and Shawnee Plant (51 species). Species richness at other plant sites ranged from 39 to 47 species collected.

For all sites combined, seining was the most productive sampling gear with 51,400 individuals captured by seines in 1988. Electrofishing sampling resulted in the collection of 23,443 fishes, whereas gill netting and trawling each collected about 7,000 specimens. Hoop netting was the least productive sampling method (841 individuals). The relatively high catches in beach seines was due to utilization of near-shore littoral areas by several species, especially these using littoral zones for nursery areas.

Statistical analyses of total abundance and biomass data using ANOVA indicated significant temporal (i.e., seasonal) effects at all plant sites for two or more sampling methods. For example, gill netting and electrofishing in September produced higher catch rates than May and July samples at several plant sites. Seine collections produced higher catch rates in July at five of six plant sites.

In contrast, spatial (i.e., upstream versus downstream) effects were generally not observed during 1988. At Shawnee Plant, catch rates for gill netting and trawling were higher at upstream sites, whereas biomass of fishes was higher in electrofishing samples downstream of Kyger Creek Plant. Adult and juvenile fish sampling in 1988 indicated no trend of decreased catch rates at downstream sites. These results indicate that the abundance and biomass of fishes was similar upstream and downstream of the cooling water discharges, and potential thermal effects (expected to be exacerbated by low flow conditions) were not observed.

Discussion

Due to prolonged drought conditions, anomalous hydrological and physicochemical conditions were observed in the Ohio River during 1988. Elevated ambient temperatures and below normal flow rates appeared to profoundly influence the biological productivity of the entire river. The timing of elevated ambient temperatures appeared to be a crucial factor in promoting biological productivity, especially fish spawning success and larval survival. Water temperatures in June, July and August were warmer than historical means at all plant sites. These are the months following spawning of many Ohio River fishes. Sustained high temperatures likely enhanced the early spawning of some species, promoted larval growth rates due to the increased abundance of phytoplankton or zooplankton, and favored the increased duration of spawning for some species. Increased larval survival resulted in higher than normal ichthyoplankton densities, as was observed with gizzard shad. Because the flushing rate of the Ohio River was reduced considerably in 1988, larvae of pelagic spawning fishes (e.g., gizzard shad, freshwater drum, skipjack herring) were very abundant and appeared to have high survival.

Comparison of 1988 benthic macroinvertebrate data with previous years (1981 and 1984; Geo-Marine 1982, Geo-Marine 1986) indicates that no major changes in species composition have occurred at upper and middle river plant sites. An increase in the number of taxa present in 1988 was observed at Kyger Creek and Tanners Creek Plants; taxa richness at Cardinal Plant was similar to the number of taxa collected in 1984. Increases in taxa richness, however, may be attributable to increased ability to identify some taxa.

Total densities of macroinvertebrates were generally higher in 1988 compared to previous years. In addition, biotic index scores have generally decreased since 1981 at Cardinal and Kyger Creek Plants, suggesting improved water quality at these sites due to the presence of less intolerant communities. In contrast, biotic index scores at Tanners Creek Plant have not changed markedly since 1981. In summary, benthic macroinverterbrate data collected during the Ohio River Ecological Research Program suggest improved water quality, especially at sites in the upper river. Reinvasion or extensions of numerous fish species in the upper section have been recently noted (Pearson and Pearson 1989). These trends are consistent with temporal patterns of chemical-specific parameters in the upper river that indicate improvements in water quality (Cavanaugh and Mitsch 1989).

Adult and juvenile fish sampling in 1988 indicated that the longitudinal distribution of Ohio River fishes is related to factors associated with zoogeography, flow regime, and environmental tolerance. These and other factors have been discussed previously (Reash and Van Hassel 1988; Van Hassel et al. 1988). At plant-specific locations the abundance, biomass, and species richness of adult/juvenile fishes was not adversely affected by power plant discharges in 1988. Rather, the combination of habitat, water quality, and flow effects appear to be more important influences as significant temporal differences in fish community parameters were common, whereas upstream/downstream differences were rarely observed.

Acknowledgments

All field sampling and data analysis were conducted by the project consultant, Environmental Science and Engineering, Inc., St. Louis, Missouri. S. L. Foster typed the manuscript.

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Interpretation of Scale Dependent Inferences from Water Quality Data

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Abstract
A survey of 15 trout streams was conducted to evaluate spatial patterns of water quality and their relationship to biophysical processes on different scales within the driftless area of southeastern Minnesota. Results suggest that subsurface geology, surface landform and land-use patterns change significantly across this physiographic region. The Hilsenhoff Biotic Index, (%) EPT, EPT:Chironomidae, nitrate-N and specific conductance were all highly correlated with these regional biophysical characteristics. Stream discharge variance, pH, and (%) sediment on riffle sites varied signficantly with differences in watershed-level land use and morphology while alkalinity, (%) leaf substrate, (%) wood substrate, (%) shredders and the scraper:collector-filterer ratio varied with reach-level channel morphology and riparian management. Scale corrected classes of monitoring variables displayed different patterns of water quality. These results support theoretical claims that aquatic ecosystems are hierarchically structured and controlled by processes operating on multiple spatial and temporal scales. Water quality monitoring networks should be designed on a scale(s) defined by management objectives and the scale(s) upon which monitoring variables respond.

Key words: scale, water quality patterns, trout streams, driftless area, biomonitoring.

Introduction

Ecological phenomena are known to respond to processes which show hierarchical order (Koestler 1967, Allen and Starr 1982; O'Neill et al. 1986, Kolasa 1989, Wiens 1989, May 1990). Biophysical processes operating within the landscape provide a hierarchical set of constraints which define the observed characteristics and dynamics of our natural resources. Thus, processes operating at levels above those of interest constrain or limit processes at lower levels within the system (Allen and Starr 1982, O'Neill et al. 1986). These factors acting within their own holon (sensu Koestler 1967) or interacting between holons comprise the dynamic processes which define hydrologic regimes, soils and vegetation and influence physiological processes, life history characteristics and community composition and function of biota within aquatic ecosystems (Frissell et al. 1986, Cummins 1988, Delcourt and Delcourt 1988, Resh and Rosenburg 1989).

Factors controlling landscape dynamics and the structure and function of stream communities may manifest themselves at multiple spatial and temporal scales (Minshall 1988, Townsend 1989, Resh and Rosenburg 1989, Ward 1989). Geologic and climatic events exert control over landscape and watershed level characteristics on a spatial scale of 100's to 1000's of square kilometers and a temporal scale of 100,000 to 1,000,000 years. Vegetation dynamics and land management practices exert controls over processes operating over landscapes and watersheds on spatial scales of 10's to 100's of square kilometers and temporal scales of 100's to 1000's of years, while management and natural processes operating along the stream corridor determine inputs of organic matter, light energy and temperature regimes over spatial scales of 1 to 100 square meters and temporal scales of weeks to months (Frissell et al. 1986, Delcourt and Delcourt 1988, Minshall 1988).

Until recently, most efforts to define factors controlling benthic communities in streams have focused on watershed, reach and microhabitat level processes operating over temporal scales of days to months (Resh and Rosenburg 1989). In addition, these studies have focused on biological responses to processes operating on one spatial and/or temporal scale (e.g., Fisher 1987, Peckarsky 1986,1987). Despite excellent attempts to introduce hierarchy theory and the importance of scale to stream ecology (Frissell et al. 1986, Cummins 1988, Minshall 1988, Townsend 1989, Ward 1989), few attempts have been made to examine benthic data or controls over water quality monitoring variables across a number of spatial and temporal scales (except see Resh and Rosenburg 1989).

Scale phenomena also influence the design and inferences drawn from water quality investigations (Jeffers 1988). However, unlike the basic sciences, applied sciences like water quality are necessarily tied to the human perspective. This perspective (scale of human activities and influences) also operates hierarchically on multiple social and political scales and must be integrated with natural biophysical phenomenon to allow for proper monitoring and management of natural resources. In fact, it is preoccupation with the human perspective by the applied sciences which often limits the utility of monitoring data (Perry et al. 1984). Matching the scale of a natural phenomenon with the scale of a management objective is necessary to improve the efficiency and accuracy of our monitoring efforts (Schumm 1988).

The objectives of the work presented in this paper were to (1) define spatial patterns of water quality within a heterogeneous region, (2) determine the relationships between water quality monitoring variables and biophysical processes across multiple scales, and (3) compare patterns of water quality generated by variables operating on different spatial scales. We hypothesized that there would be discernible regional patterns in physical, chemical and biomonitoring variables throughout the "driftless area" (Winchell and Upham 1884). Furthermore, we hypothesized that different monitoring variables would respond at different scales of resolution (regional, watershed, reach, and riparian levels) within the driftless area, since processes controlling their dynamic behavior operate at different levels.

Study Area

Samples were collected from riffle sites on 15 randomly selected trout stream tributaries of the Root River Basin in southeastern Minnesota (Longitude 91°-93° W, Latitude 43°-44° N) (Fig. 1). This area of Minnesota was considered part of the "driftless area" by J.D. Winchell during his geological survey of the state (Winchell and Upham 1884) and falls within the "driftless area aquatic ecoregion" defined by Omernik and Gallant (1988). The climate of the study area is mid-continental with 72cm of precipitation per year, 66% occurring during the growing season (May-September). Mean air temperatures range from -10°C during the winter months to 22°C in the summer with extremes of -36°C in the winter and 36°C in the summer (Kuehnast 1972).

The Root River flows across the study area, downcutting into bedrock strata along its course to the Mississippi River (Fig. 1). Topography is largely determined by surface erosion into bedrock due to the lack of glacial till throughout much of the study area. Valley slopes exceed 35% near the Mississippi River, becoming more level along the western portion of the study area. Well drained, silty loam soils, derived from loess deposits, predominate throughout much of the study area (University of Minnesota 1973).

Natural vegetation within the region consists of maple-basswood forest in the eastern portion of the study area grading to open oak savannah in the west (Kratz and Jensen 1977). Dominant woody vegetation along stream corridors consists of willow (Salix spp.), elm (Ulmus americana L.), cottonwood (Populus deltoides Marsh) and birch (Betula spp.). Agriculture is a prominent feature of the landscape with production of corn, soybeans, alfalfa, swine and dairy cattle common in the uplands and along stream corridors (United States Department of Agriculture and Minnesota Department of Agriculture 1988).

Methods

Regional Variable

The mean elevation of spring sources above each site (ELEV) was determined from USGS topographic maps (scale 1:24000) for use as an independent variable in our analyses. ELEV served as a geology variable because changes in this characteristic imply different sources of water (i.e., geological strata). Five aquifers serve as the main sources of water for springs in the region. Streams draining the western portion of the study area originate from the karst limestone and dolomite aquifers of the Maquoketa/Dubuque and Galena formations, streams in the central portion of the study area drain from the sandstone and dolomite St. Peter and Prairie du Chien formations and streams in the eastern portion of the study area originate from sandstone aquifers of the Jordan and Franconia formations (Fig. 1). Springs were identified directly on each map or were inferred from the origin(s) of perennial stream flow on each map. ELEV data were standardized to the top of the Jordan aquifer using the information provided by Broussard et al. (1975) to correct for a westerly dip in the geological strata across the study area (Fig. 1).

Land-Use Data

Land-use information at three different spatial scales (watershed, reach, riparian) for each site was obtained from published sources and interpretation of low altitude, standard color aerial photographs. Watershed level land-use data was obtained from the Land Management Information Center (IMIC) of the Minnesota State Planning Agency (1971). The spatial resolution of this data is 40 acres (16.2 ha). Data obtained consisted of the percentage of 40 acre parcels within each section of a township which were dominated by cultivated (WCUL), pasture (WPAS) and forested (WFOR) land-use types. Aerial photographs taken during the 1987 growing season over 3 of the study watersheds were interpreted using IMIC procedures and revealed that 1969 data were satisfactory for watershed-level analyses.

Low altitude, standard color aerial photographs (1987 growing season) were obtained from county Agricultural Stabilization and Conservation Service offices to evaluate land-use at the reach and riparian levels using a modified version of the IMIC method. A twenty-five cell grid (total grid size 40 acres (16.2 ha), cell size 1.6 acres (0.65 ha)) was projected onto a color print of the section containing each site (Fig. 1). The entire grid was placed over a study site perpendicular to stream flow with the back edge of the middle cell corresponding with the location of the site. Dominant land-use in each cell was noted as was land use of the cells through which the stream flowed. Reach-level land use was estimated by calculating the percentage of cells dominated by cultivated (RECU), pasture (REPA) and forested (REFO) land-use types over the entire grid. Riparian-level land-use (RICU,RIPA, RIFO) was evaluated by calculating similar percentages for cells through which the stream flowed (Fig. 1).

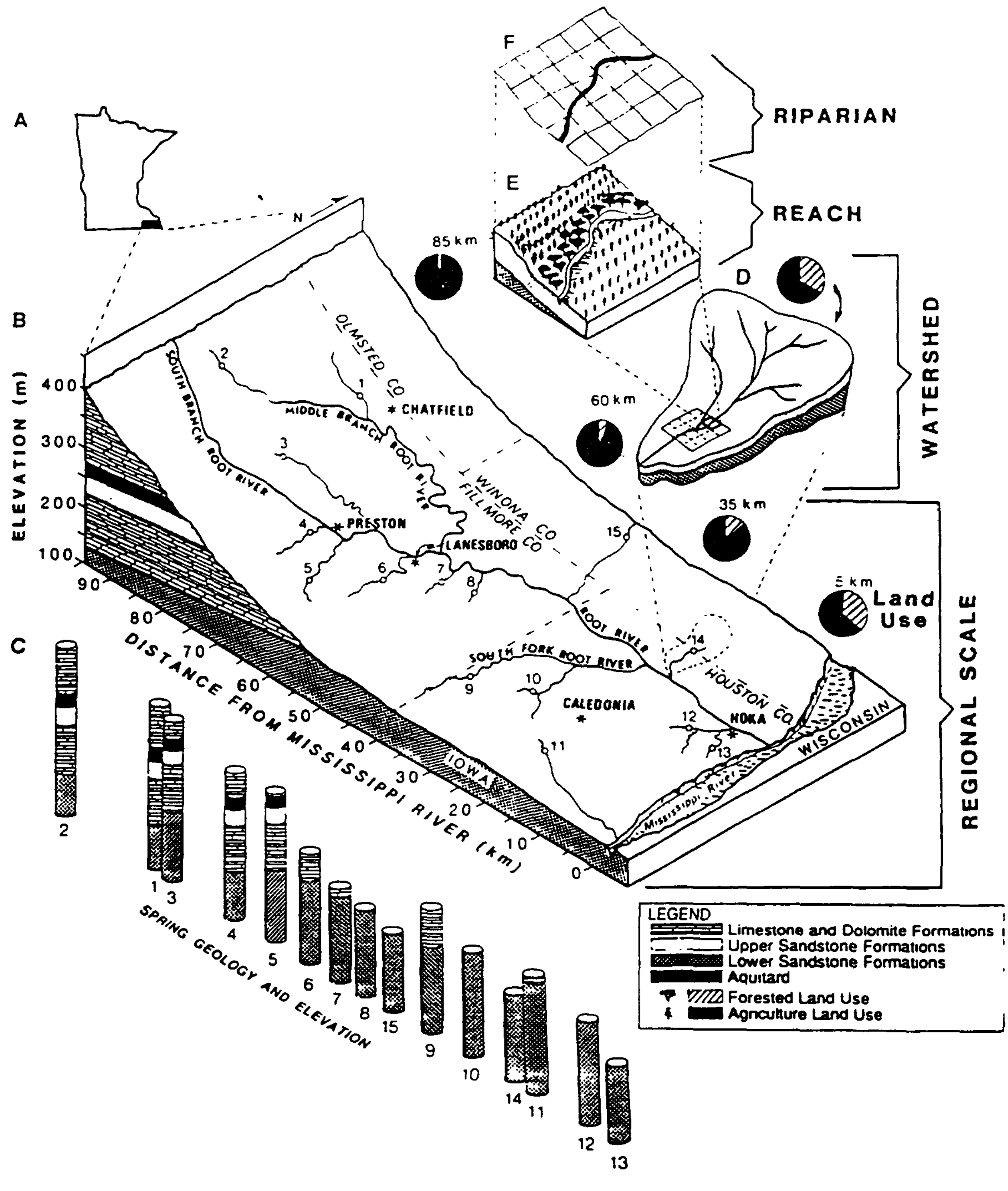

Figure 1. Diagram of study area in southeastern Minnesota showing locations of study sites, regional patterns in subsurface geology (cores) and land-use (pie-diagrams), and differences in biophysical perspective at regional, watershed, reach and riparian scales along the Root River Basin.

Table 1. Methods used in collection and analysis of water quality samples.

Variable[1]	Method	Source
NITR	Spectrophotometric	APHA[2] (1985)
ALKA	Titrimetric	APHA (1985)
PH	Electrometric	APHA (1985)
COND	YSI Model 33 S-C-T Meter	APHA (1985)
TEMP	YSI Model 33 S-C-T Meter	APHA (1985)
TURB	Hach Turbidimeter	APHA (1985)
FLCV	Sixth-Tenths-Depth Method	Platts, et al. (1983)
Substrate	% Occurrence along Transects	IN TEXT
HBIN	Duplicate, 1 min. Kicknet	Hilsenhoff (1988)
PEPT	Duplicate, 1 min. Kicknet	Plafkin et al. (1989)
EPTC	Duplicate, 1 min. Kicknet	Plafkin et al. (1989)
SHRD	Duplicate, 1 min. Kicknet	Plafkin et al. (1989)
SCCO	Duplicate, 1 min. Kicknet	Plafkin et al. (1989)

[1]Abbreviations as defined under Methods.
[2]American Public Health Association.

Accuracy of interpretations was calculated to be 95% based on quality control procedures.

Catchment and Channel Parameters
Watershed area (AREA) and channel gradient (GRAD) from headwaters to site were determined from USGS topographic maps (scale 1:24000). Watershed gradient was estimated by calculating the absolute gradient from headwaters to site (change in elevation over channel length) using information derived from the topographic maps. Mean channel width (WIDT), depth (DEPT) and current velocity (CURR) were determined from measurements taken in the field.

Monitoring Variables
Physical and chemical water quality characteristics were evaluated on 5 randomly chosen dates in the spring and fall of 1988 (i.e., 10 repeated measures at each site). Parameters evaluated included nitrate-N (NITR), specific conductance (COND), stream temperature (TEMP), turbidity (TURB) and coefficient of variation of flow measurements (FLCV). Methods used in the determination of these parameters are shown in Table 1.

The percent occurrence of five substrate categories were determined on each riffle on the first and last sampling dates in the spring and fall (i.e., 4 repeated measures at each site). A 10 meter chain was fitted with colored flags (10cm spacing) and laid across the stream at ten equally spaced longitudinal positions along the channel. Substrate observations were made at ten locations on the chain (0.10x channel width spacing) along each of the ten transects for a total of 100 determinations per riffle. The data translated directly to percent occurrence of rock (ROCK) (diameter > 4mm), wood (WOOD), leaf (LEAF), sediment (SEDI) (diameter ≤ 4mm) and macrophyte (MACR) on each riffle.

Invertebrates were sampled at each site on the first and last sampling dates in the fall of 1988. Biomonitoring metrics evaluated from these samples included the Hilsenhoff Biotic Index (HBIN), percent of Ephemeroptera, Plecoptera and Trichoptera (PEPT) invertebrates in each sample, the ratio of EPT to Chironomidae (EPTC), percentage of shredder invertebrates (SHRD), and ratio of scrapers to collector-filterers (SCCO) (Plafkin, et al. 1989). Averages of duplicate samples collected on each date were used in further analyses (i.e., 2 repeated measures at each site). In addition, the dominant taxon in kicknet samples from each site was identified and relative abundance of these taxa were compared between sites.

Data Analysis

Abiotic gradients and biophysical relationships across the study area were identified using graphical, multiple regression and principal components analysis techniques (NH Analytical Software, 1988). Date by date Spearman Rank Correlations were calculated from repeated measures of each monitoring variable versus biophysical characteristics of each site (n=15 sites). Seasonal (n=30; 15 sites x 2 seasons) and overall means (n=15 sites) for monitoring variables were used in regression analyses to define relationships between biophysical characteristics of each site and monitoring variables at different spatial scales. If season did not contribute significantly to a regression model (t-statistic, $p>0.05$), overall site means were used in regression analyses (n=15). Model selection was based on maximizing the coefficient of determination (R^2), minimizing the residual mean square and collinearity among predictors and achieving a null residual plot through transformation of the raw data (Weisberg 1985). Results of correlation and regression analyses were used to define scale corrected groups (classes) of monitoring variables. An agglomerative cluster analysis technique was used to identify site groupings based on scale corrected classes of monitoring variables. Site means of each monitoring variable were standardized for unit differences by calculating z-scores and clustered based on squared euclidean distances between centroids of each cluster with the software package SPSS (Norusis 1988).

RESULTS

Biophysical Characteristics of Sites

ELEV above each site varied significantly (Fig. 2a) with distance from the Mississippi River. These data confirm changes in aquifer sources to trout streams distributed across the study area (Fig. 1). In addition, GRAD (Fig. 2b) and WFOR (Fig. 2c) displayed significant regional patterns across the driftless area. Lower GRAD were observed in the western portion of the study area, reflecting regional changes in topography. WFOR also decreased logarithmically with distance from the Mississippi River. These data show the increase in intensity of agricultural land-use practices on a regional scale with distance from the Mississippi River and are consistent with Minnesota state records of agricultural land-use (United States Department of Agriculture and Minnesota Department of Agriculture 1987). AREA (Fig. 2d) did not vary significantly with distance from the Mississippi River as did the other regional biophysical variables. Thus, the observed regional trends in GRAD were not merely artifacts of sampling site location within study watersheds. Regional trends exhibited by ELEV, GRAD and WFOR clearly show heterogeneity in biophysical characteristics across the driftless area.

Additional information on the biophysical characteristics of these sites is provided by the results of principal components analysis (PCA). PCA created a new set of biophysical variables through linear combinations of the original 15 variables. Eigenvector loadings of monitoring variables on each principal component (PC) indicated that there were sets of biophysical characteristics operating together on different scales (Table 2). Highest loadings on PC1 were mainly from biophysical characteristics known to vary on a regional scale (ELEV, WCUL, WFOR, GRAD). Thus, PC1 explained 31.3% of the variability in the data set and seemed to represent large scale biophysical processes.

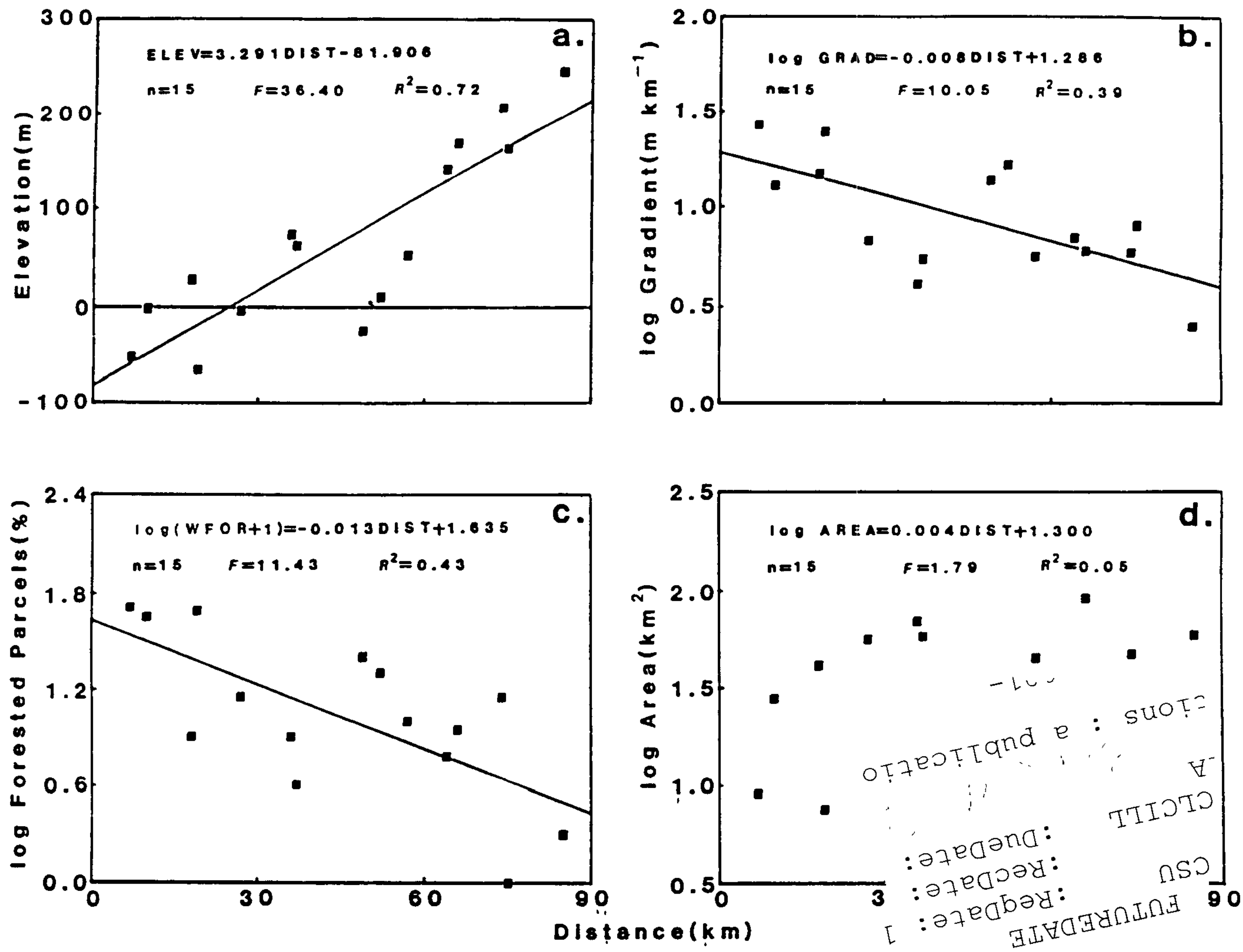

Figure 2. Regional patterns in biophysical characteristic the driftless area of southeastern Minnesota, (a) Rel spring elevation (ELEV) above each sampling site and ae that sampling site from the Mississippi River; (b) Relatio gradient (GRAD) and distance from the Mississippi River; (c the number of 40 acre parcels dominated by forested land- (WFOR) and distance from the Mississippi River; (d) ...etween watershed area (AREA) and distance from the Mississippi R

Highest loadings on PC2 were associated with reach and riparian management practices and channel morphology (RIFO, REFO, REPA, WIDT). PC2 explained an additional 26.8% of the variability in the data set and represented local scale processes. Loadings on PC3 were highest on reach and riparian management practices (RICU, RECU, REFO). Watershed level land-use (WPAS) and morphology (GRAD) were also highly correlated with this principal component. PC3 explained 15.9% of the variability in the biophysical data set. Loadings on PC4 were only significantly correlated with channel morphology (DEPT, WIDT) at the reach level. This principal component explained 8.7% of the variability in the data set. Strong collinearity was observed among land-uses on different scales, especially

Table 2. Eigenvalues, eigenvectors and variance (% Var) explained from principal components analysis of trout stream biophysical characteristics (n=15 sites) and scales represented by each principal component.

PC	Eigenvalue	% Var	Cumulative %
1	4.689	31.3	31.3
2	4.014	26.8	58.0
3	2.382	15.9	73.9
4	1.305	8.7	82.6

VARIABLE[1]	EIGENVECTOR	SCALE
	PC1	
ELEV	-0.39	
WCUL	-0.32	
WFOR	0.32	
DEPT	-0.29	
CURR	0.32	REGIONAL/
AREA	-0.27	WATERSHED
GRAD	0.31	
REPA	-0.30	
RIPA	-0.28	
	PC2	
REPA	0.34	
RIPA	-0.31	
REFO	0.37	REACH/
RIFO	-0.39	RIPARIAN
DEPT	0.23	
WIDT	-0.35	
	PC3	
WPAS	-0.28	
RECU	-0.54	
REFO	0.37	WATERSHED/
RICU	-0.54	REACH
CURR	-0.29	
GRAD	0.24	
	PC4	
DEPT	0.42	REACH
WIDT	0.48	

[1]Abbreviations defined under Methods

reach and riparian. High collinearity among predictors within a nested hierarchy would be expected since characteristics at one scale are part of the next higher scale.

Stream Water Quality Characteristics

Most of the monitoring variables displayed large ranges in values, typical of disturbed catchments over dissolution aquifers (Table 3). NITR and TURB values occasionally exceeded water quality standards for trout waters within the state (Minnesota Pollution Control Agency 1990) and flow variance was highest in streams of the karst area within the region (Table 3).

Most of the monitoring variables were significantly correlated with biophysical characteristics at multiple scales (Table 4). NITR and COND data displayed the expected trend of low values at sites draining diffuse sandstone aquifers of the Jordan and Franconia formations and high values from the Maquoketa-Dubuque and Galena formations. TURB values showed considerable variance due to the effects of a thunderstorm which influenced samples of sites 5, 14 and 15 on one date in the spring of 1988 and men working with farm equipment in the stream at site 3 on one date during fall sampling. Thus, COND seemed to respond on a regional scale with changes in subsurface geology and regional land-use. NITR and PH seemed to be influenced by watershed-level land-use characteristics while ALKA, TURB and TEMP were highly correlated with reach and/or riparian level management practices (Table 4). FLCV was not significantly correlated with any of the biophysical characteristics.

All five substrate types on riffles displayed large ranges in values (Table 3). ROCK was the dominant substrate at riffle sites across the study area followed by MACR (Table 3). WOOD was the least common substrate type and was highly correlated with the amount of reach and riparian forest above and adjacent to a site (Table 4). Similar observations were made of LEAF material as a substrate, except the occurrence of LEAF material was strongly seasonal (see below). SEDI was prevalent as a substrate type

Table 3. Overall summary statistics for monitoring variables evaluated in Southeast Minnesota Streams (n-number of repeated measures x 15 sites).

Variable[1]	n	Mean	Median	s.e.	Range
		Physical/Chemical			
NITR (mg L^{-1})	150	3.9	3.4	0.2	0.5-10.9
ALKA (mg L^{-1})	150	266	262	2	226-333
pH	150	8.01	8.02	0.03	7.05-8.74
COND (uS cm^{-1})	150	397	386	5	275-660
TEMP (°C)	150	11.7	11.0	0.4	4.2-25.0
TURB (NTU)	150	5.12	1.80	1.37	0.30-146.00
FLCV (%)	30	19.2	15.3	2.5	2.6-66.3
		Substrate			
ROCK (%)	60	51.8	56.0	3.2	0.0-89.0
WOOD (%)	60	2.5	1.0	0.5	0.0-13.0
LEAF (%)	60	8.2	1.0	1.5	0.0-48.0
SEDI (%)	60	8.4	4.0	1.5	0.0-56.0
MACR (%)	60	28.8	22.0	3.5	0.0-94.0
		Invertebrates			
HBIN	30	3.7	3.3	0.2	1.8-6.3
PEPT (%)	30	55.2	57.3	4.5	1.5-93.9
EPTC	30	9.1	5.8	2.2	0.02-63.7
SHRD (%)	30	12.1	7.2	2.8	0.0-67.3
SCCO	30	0.8	0.3	0.2	0.0-4.6

[1]Abbreviations as defined under Methods.

at many of the agricultural sites and was most highly correlated with riparian land management (Table 4). ROCK and MACR were most highly correlated with watershed and reach-level management practices. ROCK was more abundant at agricultural sites with high current velocities while MACR tended to be more abundant at forested sites with lower current velocity. Ranunculus aquatilis (Chaix), Veronica connata var. glaberrima (Pennell) and Nasturtium officinale (R. Br.) were the macrophyte species occurring most frequently at all sites across the study area.

HBIN values ranged from "fair" (6.25) to "excellent" (1.77) indicating that some of these trout streams were influenced by significant organic loading. The PEPT in kicknet samples ranged from 1.5 to 93.9% and EPTC in kicknet samples ranged from 0.02 to 63.67. These three metrics suggested differences in invertebrate community structure between sites and all three were highly correlated with regional changes in subsurface geology and watershed morphology (Table 4). Regional changes in benthic community structure were confirmed by examining the dominant invertebrate taxa at each site. Two western sites (1,5) were dominated by the chironomids Tanytarsus sp. (35%) and Rheotanytarsus sp. (54%), site 3 was dominated by the mayfly Baetis tricaudatus vagans (McDunnough) (42%), and site 4 was dominated by the caddisfly Cheumatopsyche spp. (17%). Empirically derived tolerance values to organic pollution for these taxa were 6, 6, 2 and 5 on a scale of 0-10 (Hilsenhoff 1987). The invertebrate communities of three centrally located sites (6, 8, 10) and one western site (2) were dominated by Optioservus fastiditus (LeConte). This

Table 4. Highest Spearman Rank correlation[1] between each monitoring variable and the most frequently correlated predictor at each scale based on date by date correlations of monitoring variables with biophysical characteristics.

Variable[2]	Regional[2]	Watershed[2]	Reach[2]	Riparian[2]
NITR	ELEV(0.79)	WFOR(-0.91)	CURR(0.77)	RIFOR(-0.64)
pH	ELEV(-0.56)	WPAS(0.83)	REPAS(-0.46)	RIFOR(0.39)
TURB	ELEV(0.50)	WPAS(0.58)	RECUL(0.74)	RIPAS(0.62)
ALKAL	ELEV(0.55)	GRAD(0.50)	RECUL(-0.69)	RICUL(-0.66)
COND	ELEV(0.81)	WFOR(-0.73)	REPAS(0.68)	RIPAS(0.67)
TEMP	ELEV(-0.67)	GRAD(0.58)	REFOR(-0.68)	RICUL(0.56)
FLCV	NO SIGNIFICANT CORRELATIONS WITH SITE VARIABLES			
ROCK	-	WPAS(0.71)	CURR(0.48)	RICUL(0.60)
WOOD	-	WFOR(0.52)	REPAS(-0.53)	RIFOR(0.52)
LEAF	ELEV(0.52)	GRAD(-0.49)	REPAS(-0.76)	RIFOR(0.41)
SEDI	ELEV(-0.38)	AREA(-0.60)	REFOR(-0.54)	RIFOR(0.62)
MACR	-	WPAS(-0.63)	RECUL(-0.69)	RICUL(-0.64)
HBIN	ELEV(0.62)	GRAD(-0.69)	CHNWD(0.55)	-
PEPT	ELEV(-0.64)	GRAD(0.53)	CHNWD(-0.55)	-
EPTC	ELEV(-0.56)	AREA(-0.68)	CURR(0.58)	-
SHRD	ELEV(0.50)	WFOR(-0.38)	RECUL(-0.74)	RICUL(-0.42)
SCCO	-	AREA(0.42)	REFOR(0.57)	RICUL(-0.61)

[1]Correlations presented in table statistically significant ($p<0.10$) based on quantiles for the Spearman's Test Statistic (Conover 1980).
[2]Abbreviations as defined under Methods.

elmid beetle contributed 19-27% of the cumulative number of invertebrates sampled at each of the sites and has a tolerance value of 4 on a scale of 0-10. Site 7 kicknet samples were dominated by the caddisfly Micrasema kluane (Ross and Morse) (50%) and the invertebrate communities of the eastern sites within the study area (sites 11-15) were all dominated by Brachycentrus occidentalis (Banks). This caddisfly contributed 21-55% of the cumulative abundance of all taxa collected on both dates at each of the eastern sites. Both of these brachycentrid caddisflies have tolerance values of 1 on a scale of 0-10. Thus, invertebrate communities in the western portion of the study area were dominated by taxa which were moderately tolerant to organic enrichment (except sites 2,3) while communities of eastern sites were dominated by taxa which exhibited low tolerance to high organic loadings.

While community structure seemed to be influenced primarily at the regional and watershed levels, invertebrate community function was more highly correlated with local reach/riparian level processes (Table 4). SHRD and SCCO were positively correlated with the extent of forest development at the reach and riparian-levels and negatively correlated with agricultural land-uses on these same scales.

Regression Relationships

Statistically significant relationships were observed for all monitoring variables with one or two biophysical characteristics (Table 5). However, the variance explained by these models was quite variable (range of $R^2= 0.17$ to 0.92). Six of the 17 monitoring variables displayed significant seasonal differences (i.e., PH, TURB, COND, TEMP, FLCV and LEAF). Of these six monitoring variables, only LEAF displayed higher values in fall samples. The remaining monitoring variables had significant amounts of variability explained by a combination of watershed, reach and riparian level biophysical site characteristics (Table 5). More than half of the regression equations explained 40% or more of the variance in the monitoring data.

Our monitoring variables can be divided into groups based on the predictors in each model (Watershed, Watershed/Reach, Reach/Riparian). Thus, 7 of the monitoring variables had significant amounts of their variance explained by biophysical characteristics on the watershed level (NITR, PH, FLCV, SEDI, HBIN, PEPT, EPTC), 2 monitoring variables by a combination of characteristics on the watershed and reach levels (COND, TEMP) and 8 variables by characteristics on the reach and riparian levels (TURB, ALKA, ROCK, WOOD, LEAF, MACR, SHRD, SCOO). Combining the results of correlation and regression analyses, we delineated scale corrected classes of monitoring variables (Table 6). Four classes (regional, watershed, reach, riparian) were defined.

Patterns at Different Scales

Scale corrected classes of monitoring variables (Table 6) were used to generate site groupings. Cluster dendograms were generated using monitoring variables which had the greatest amount of their variance explained by regional, watershed, reach and riparian-level biophysical characteristics. Dendograms produced by these analyses (Fig. 3) were compared to identify and interpret differences in spatial patterns of water quality within the study area.

Each dendogram portrays a different pattern of site groupings based on the relationship between monitoring variables used to generate the dendogram and biophysical characteristics of each site. Regional patterns (Fig. 3a) generated from NITR, COND, HBIN, PEPT and EPTC monitoring data show 5 distinct site groupings. An examination of site characteristics with respect to those monitoring variables allowed interpretation of the observed pattern. Cluster 1 (sites 8, 11, 12, 13) had low NITR and COND compared to the regional median. Cluster 2 (sites 2, 6, 15) had high NITR and COND values. Cluster 3 (sites 1, 4, 9, 10) had high HBIN, low PEPT and EPTC values compared to regional medians. In contrast, the cluster formed from the combination of clusters 1 and 2 (above) had low HBIN, PEPT and high EPTC values. The cluster formed from the agglomeration of sites 3 and 5 differed from the other monitoring sites due to low PEPT values. These two sites were located in the karst portion of the study area below heavily developed watersheds.

The watershed-level dendogram (Fig. 3b) delineates two major groups of sites; group 1 (Sites 2,4,6,8,9, 10,11,13,15) and group 2 (Sites 1,3,5, 7,12,14). These two groups can be distinguished by differences in SEDI and MACR substrate at riffle sites. Group 1 sites had lower SEDI and higher MACR than group 2 sites when compared to regional median values.

Reach and riparian-level analyses produced very similar site patterns (Fig. 3c,d). Two groups of sites are easily delineated from the dendograms produced by these analyses. All sites except 3,5,7 at the reach-level and 5, 7 at the riparian level belong to one large group of relatively similar sites. These outlier sites displayed poor water quality characteristics (i.e., higher NITR, COND, ALKA and SHRD and lower WOOD) as compared to regional median values for each monitoring variable.

Thus, cluster analysis on scale corrected monitoring data provided distinctly different site groupings which could be interpreted from monitoring variables operating on different scales.

Table 5. Regression models for monitoring variables and biophysical factors in Southeast Minnesota streams (n-number of observations; R^2-coefficient of determination; F(p)- F-statistic and probability value for regression; RMS-residual mean square for regression; Season- t-statistic and significance for season effect in regression (*-p<0.05, NS-p>0.05)).

Variable[1,2]	Predictor[1,2]	n[3]	R^2	F(p)	RMS	Season
NITR	-log(WFOR+1)	15	0.84	76.2 (<0.001)	1.039	0.62,NS
pH	-WCUL -GRAD	30	0.42	7.9 (<0.001)	0.042	-3.21,*
log TURB	RECU RIPA	15	0.37	5.0 (<0.016)	0.037	-1.52,NS
log ALKA	-RECU	15	0.17	3.9 (0.047)	0.001	0.29,NS
log COND	-WFOR REPA	30	0.70	23.0 (<0.001)	0.001	-5.27,*
log TEMP	log(WPAS+1) -log(REFO+1)	30	0.81	41.7 (<0.001)	0.003	-10.43,*
log FLCV	WCUL AREA	30	0.37	6.6 (<0.001)	0.051	-3.90,*
log Rock	CURR	15	0.25	5.7 (0.017)	0.044	-0.76,NS
arcs WOOD	RECU RIFO	15	0.52	8.7 (0.002)	0.004	-1.58,NS
log(LEAF+1)	log(RIFO+1)	29	0.92	172.9 (<0.001)	0.026	18.27,*
log(SEDI+1)	-log AREA	15	0.21	4.7 (0.029)	0.164	-0.60,NS
arcs MACR	-RECU	15	0.32	7.5 (0.007)	0.063	-1.20,NS
log HBIN	-log GRAD	15	0.42	11.2 (0.002)	0.011	-
arcs PEPT	GRAD	15	0.28	6.5 (0.011)	0.054	-
log(EPTC+1)	-AREA	14	0.53	15.4 (<0.001)	0.065	-
log(SHRD+1)	-RECU	15	0.38	9.55 (0.003)	0.086	-
log(SCCO+1)	REFO	15	0.28	6.4 (0.012)	0.034	-

[1]Monitoring and biophysical variable abbreviations as defined under Methods.
[2]Data transformations included arcs=$(\sin^{-1})^{1/2}$ and log=common logarithm.
[3]Seasonal means used in regression (n=30) or overall means (n=15).

Discussion

Data presented in this paper suggest that commonly used water quality monitoring variables respond to processes operating on several spatial scales within the driftless area. Regional patterns in NITR, COND, and invertebrate community structure were highly correlated with regional trends in subsurface geology and land-use patterns. Streams in the western portion of the study area drain karst limestone and dolomite aquifers (Winchell and Upham 1884, Broussard et

Table 6. Scale corrected classes[1] of monitoring variables based on correlation and regression analyses.

Variable[2]	REGIONAL	WATERSHED	REACH	RIPARIAN
LEAF			....	______________
WOOD		....	______________	______________
TURB		....	______________	______________
SHRD			______________	....
SCOO			______________	....
ALKA			______________	
MACR		....	______________	....
ROCK		....	______________	....
TEMP		______________	______________	....
COND	....	______________	______________	....
NITR	....	______________	....	....
PH		______________		
FLCV		______________		
EPTC	....	______________		
SEDI	....	______________		
HBIN	....	______________		
PEPT	....	______________		

[1] Class Membership Based on Regression Results- ___________
Class Membership Based on Significant Correlations- ....
[2] Abbreviations as defined under Methods.

al. 1975, Ojakangas and Matsch 1982, Singer et al. 1983). The combination of intensive agricultural land-use within these watersheds and karst subsurface geology promotes water quality problems (LeGrand 1973, Singer et al. 1982, St. Ores et al. 1982, Hallberg et al. 1985, Wall et al. 1989) which simplify the structure of invertebrate communities by reducing or eliminating intolerant taxa (Perry et al. 1988, Troelstrup and Perry 1989, Bartodziej and Perry MS, Wilton and Perry unpubl. data). In contrast, streams in the eastern portion of the study area originate from sandstone aquifers and drain watersheds with more woody vegetation. Despite steeper gradients, these streams have the lowest NITR, COND and HBIN values and the highest PEPT and EPTC values.

These regional biophysical characteristics serve as a template over which finer grain reach and riparian processes operate (Frissell et al. 1986, Minshall 1988, Townsend 1989, Ward 1989). Local responses of NITR, COND and PH were probably related to subsurface dynamics within the riparian zone. Agricultural land use adjacent to a stream is known to reduce denitrification and plant uptake of nitrogen and promote nitrification and nutrient export to the stream channel (Vitousek and Melillo 1979, Peterjohn and Correll 1984, Pinay and Decamps 1988). Higher NITR, COND and PH values would be expected adjacent to agricultural conditions where redox potentials and movement of soluble ions are high (Green and Kauffman 1989).

TURB, ALKA and TEMP were also observed to vary with reach and riparian management in this study. These parameters are influenced directly by loss of riparian vegetation and subsequent bed and bank erosion in and adjacent to the stream channel. TURB levels increase dramatically in response to livestock grazing and cultivation adjacent to the stream

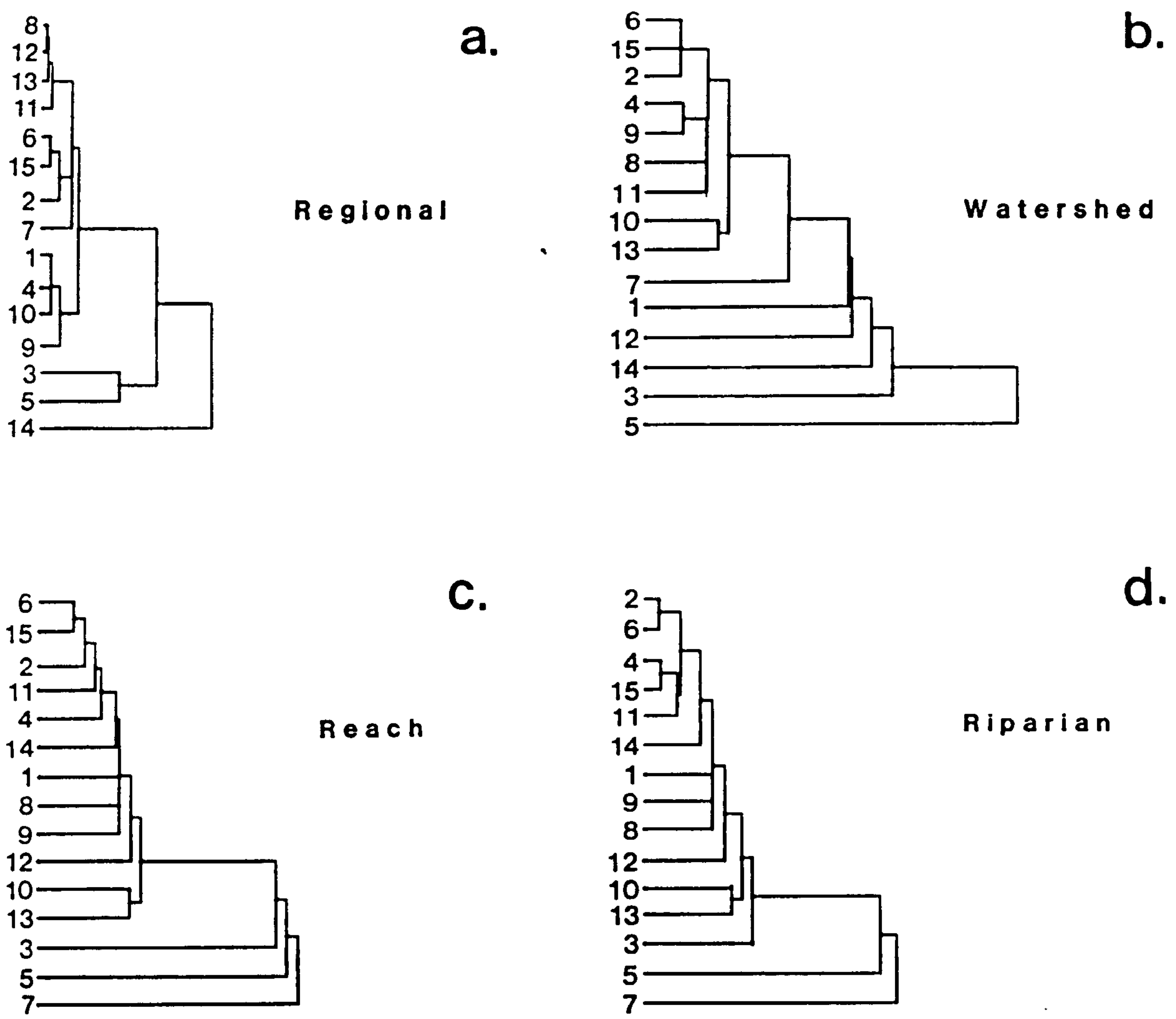

Figure 3. Results of clustering sites on scale corrected classes (see Table 6) of monitoring variables, (a) dendogram generated from regional class, (b) dendogram generated from watershed class, (c) dendogram generated from reach class, (d) dendogram generated from riparian class (Numbers adjacent to each dendogram refer to site locations as defined in Figure 1).

bank (Woodall and Wallace 1972, Karr and Schlosser 1978, Bratton et al. 1980, Menzel et al. 1984). These activities in close proximity to the stream may also increase channel width and reduce channel depth due to sedimentation of sloughed material from the stream bank (Clifton 1989). Numerous studies have noted increases in mean temperature and greater ranges in temperature regimes adjacent to agricultural areas (Karr and Schlosser 1978, Bratton et al. 1980, Dance and Hynes 1980, Menzel et al. 1984, Smart et al. 1985).

Substrate characteristics in driftless area streams are probably controlled by a combination of hydrologic processes operating on the watershed level and light and mesoscale hydrodynamics on the reach and riparian levels. ROCK and MACR substrate types were negatively correlated with one another and highly correlated with current regimes and reach and riparian management. Occurence of SEDI was negatively correlated with watershed area and positively correlated with reach and riparian management. The flashy nature of karst streams (LeGrand 1973, Hallberg et al. 1985) and prevalence of sedimentation in agricultural reaches (Karr and Schlosser 1978, Dance and Hynes 1980, Lenat 1984) would seem to explain observed patterns of these substrate types throughout the study area. WOOD material in the channel and LEAF material on the stream bottom were negatively correlated with reach pasture land and positively correlated with riparian forested land. Seasonal patterns in LEAF abundance associated with autumn abscission were also observed, suggesting that spatial and temporal patterns of availability govern the dynamics of these substrate types. Agricultural streams flowing through open riparian canopies have been shown to harbor extensive algal communities (Menzel et al. 1984, Smart et al. 1985, Bachmann et al. 1988). Thus, management of the riparian zone may directly influence functional characteristics of a stream by altering detrital inputs and primary production on a local scale within a watershed (Hynes 1975, Swanson et al. 1982).

Biomonitoring has been promoted as a useful tool in evaluating support of designated uses in water quality investigations (Lenat et al. 1980, Hilsenhoff 1982, Lenat 1988, Plafkin et al. 1989). Corkum and Ciborowski (1988) and Corkum (1989) were successful in delineating broad scale invertebrate community patterns associated with biophysical characteristics in northwestern North America. Invertebrate community structure proved sensitive to regional patterns in geology, surface land form and land-use in the driftless area. We observed low PEPT and EPTC and high HBIN values from streams draining dissolution aquifers through agricultural watersheds. Others have observed similar large scale patterns in community structure using these metrics (Welch et al. 1977, Bratton et al. 1980, Dance and Hynes 1980, Hilsenhoff 1982, Lenat 1984, Menzel et al. 1984, Hite and Bertrand 1989).

We observed shifts in the relative abundance of different feeding guilds within the invertebrate communities at our sites. Higher SHRD and SCOO values were observed adjacent to forested riparian zones. If food were a limiting resource to these insects, SHRD abundances would be expected to track the availability of LEAF material (Hynes 1975, Swanson et al. 1982, Cummins et al. 1989) while SCOO abundances would track the availability of benthic algae in the stream (Dance and Hynes 1980, Karr and Dudley 1981, Menzel et al. 1984). Ross (1963) provided evidence of regional patterns in the distribution of caddisflies (Trichoptera) related to the predominant terrestrial vegetation. Within his large scale framework, our data suggest that functional characteristics of invertebrate communities may be tightly tied to local biophysical characteristics which influence inputs and types of organic material to the stream (Hynes 1975, Swanson et al. 1982, Cummins et al. 1989).

The results of this study provide evidence of significant variability in biophysical characteristics across the driftless area in southeastern Minnesota. Subsurface geology, land surface form and land-use all vary significantly with distance from the Mississippi River. These trends in biophysical characteristics explain a significant amount of the variability in physical (COND), chemical (NITR) and biological (BIND, PEPT, EPTC) water quality monitoring variables. Regional patterns, together with local variance in reach and riparian characteristics, provide a mosaic of biophysical factors which vary at multiple spatial and temporal scales. This presents a tremendous challenge to the "aquatic ecoregion" concept which has been tested widely (Hawkes et al. 1986, Rohm et al. 1987, Hughes et al. 1987, Larsen et al. 1988, Whittier et al. 1988, Lyons et al. 1989) and implemented by state water quality agencies (e.g., Hieskary et al. 1987).

Aquatic ecoregions were originally defined to address national and large scale regional water quality issues (Omernik 1987). However, several investigators have noted problems in implementing this approach to water quality monitoring and management. Omernik and Griffith (1986) observed differences in alkalinity between seepage and drainage lakes within the same ecoregion. Hawkes (1986) found that fish ecoregions in Kansas showed little similarity to the aquatic ecoregions of Hughes and Omernik (1981). Lyons (1989) found that local habitat characteristics associated with reach and riparian management were better predictors of fish community characteristics than membership within an aquatic ecoregion in Wisconsin. Whittier et al. (1988) found poor separation of ecoregions in Oregon using periphyton and invertebrate data from streams across the state. In particular, large within-region variability occurred when valley and mountain streams occupied the same ecoregion. This challenge has been met by generalizing the "aquatic ecoregion" concept to a "regionalization" concept (Gallant et al. 1989). Under this approach, regions may be defined at any scale. Thus, heterogeneous regions may be broken up into smaller more homogeneous units. This approach has great promise if state water quality agencies can secure funding to increase the number of monitoring stations necessary for such stratification.

Properly designed water quality monitoring programs operate from well defined objectives, utilize monitoring variables which relate directly to those objectives, provide spatial and temporal information necessary to address those objectives and optimize resources to account for natural variability in measured parameters (Schaeffer et al. 1985, Perry et al. 1985). Consideration of which variables to measure is an important step in the planning process of these efforts. Many state and federal water quality agencies evaluate a list of variables at monitoring sites distributed over a sociopolitical area (state or county) and sample on a regular temporal frequency (e.g., 1 month) (Perry et al., 1984). Our data suggest that different monitoring variables are controlled by processes operating on different scales (regional, watershed, reach, riparian) and that different sets of variables may be more effective for detecting water quality problems at different scales. The variable/scale associations identified in this study may not be appropriate for other physiographic regions. In addition, this study focused on spatial patterns of water quality. Temporal dynamics in biophysical characteristics are equally important when designing a monitoring program (Wiens 1989). However, the methods used to derive scale relationships in this study could be used in other regions. Thus, scale corrected groups of monitoring variables could be identified which would (1) provide more sensitivity to a problem on a particular scale and (2) provide greater security against conflicting results due to scale incompatibility with management objectives.

Current ecological theory suggests that natural systems are hierarchically structured. The characteristics of any level of a natural system are thought to be controlled by rate processes operating on higher spatial and temporal scales (Koestler 1967, Allen and Starr 1982, O'Niell et al. 1986, O'Neill 1988). Examples of controlling processes operating within the landscape include faulting, volcanism, climatic changes, pedogenesis, weathering and erosion. These processes result in the development of controlling factors in the landscape which influence the characteristics of water resources and the biota inhabiting aquatic ecosystems. Clearly, interpretation of structural and functional patterns in nature is scale dependent and our ability to monitor and manage those resources depends on our understanding of that hierarchical structure and its dynamics.

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Aquatic Vegetation and Habitat Quality in the Lower Des Plaines River: 1985-1987

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Abstract
In the early 1980's, locally abundant populations of aquatic vegetation were observed in the lower Des Plaines River after being virtually absent for nearly 20 years. This study was conducted in 1985-1987 to characterize the aquatic vegetation community in a 13-mile reach (river miles 273-286) of the Des Plaines River and to assess aspects of habitat quality. Twenty species of aquatic macrophytes were identified; vegetation cover, estimated from ground-truth surveys and low-altitude color aerial photographs, reached 60 ha in August 1987. The most heavily vegetated areas were river miles 285.5, 279.5, 277, and 273.5. Sediments at these locations contained high levels of As, Cd, Cr, Cu, Fe, Hg, Pb, and Zn and aquatic macrophytes had high levels of Al, Ba, Cr, Co, Sn, V, Zn, and PCBs. At senescence, these accumulated substances can remain in macrophyte tissues or be released into the water or sediments, affecting overall habitat quality. Therefore, interactions between rooted vascular plants and toxic substances, particularly those in sediments, should be considered when assessing habitat quality.

Key words: aquatic macrophytes, Des Plaines River, Illinois, vegetation cover, aerial photography, heavy metals, PCB's

Introduction

The Illinois River and its bottomland lakes have been virtually devoid of submersed and floating-leaved vegetation since the early 1960's (Bellrose et al. 1983, Havera et al. 1980). This decline has been linked to the release of wastewater, industrial effluents, and runoff. In the early 1980's, locally abundant populations of aquatic vegetation were observed in the Lower Des Plaines River. Because macrophytes modify and diversify habitat and fuel secondary production by producing oxygen, cycling nutrients, and providing cover for fishes and substrate for fish food organisms (Barko et al. 1986, Bennett 1971, Engel 1985, Raschke 1978, Wright et al. 1981), recent increases in aquatic vegetation should improve water and habitat quality (e.g., reduced turbidity, increased oxygen levels, larger and more diverse invertebrate populations, and an improved fishery).

Macrophytes also modify sediment and water chemistry (Dawson et al. 1978, Hutchinson 1975, Sculthorpe 1967, Westlake 1973), often by substance uptake and release (Hill 1979, Jaynes and Carpenter 1986, Smith and Adams 1986). Accumulated substances, both mineral nutrients and toxic substances, may remain in roots and rhizomes or be translocated to other plant parts (i.e., acropetal translocation). During plant senescence, these substances may associate with decomposing particulate matter or leach into the water column. Thus, substances concentrated from deeper sediments can be moved into the water column and top sediments (Campbell et al. 1985, Everard and Denny 1985, Gabrielson et al. 1984, Howard-Williams and Lenton 1975, Kraus et al. 1986, McIntosh et al. 1978, Smith and Adams 1986, Welsh and Denny 1976).

Sediments of the Lower Des Plaines River are characterized by the presence of toxic substances, and rooted aquatic macrophytes are capable of mobilizing these sediment-bound substances. The purpose of this study was to assess habitat quality in the

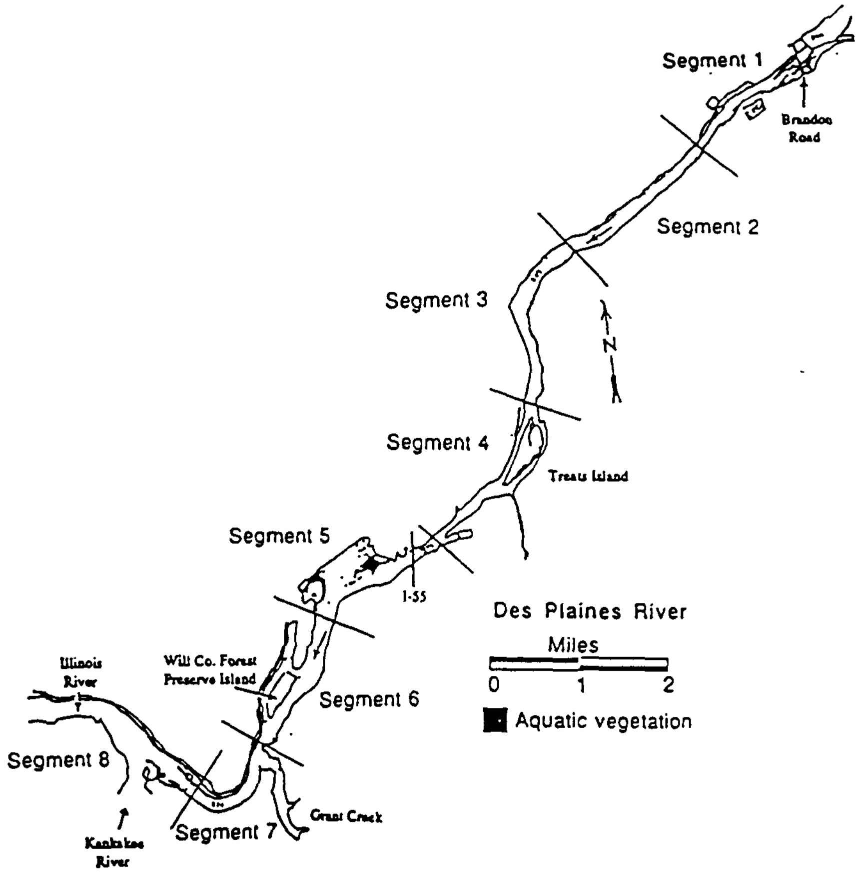

Figure 1. Location and extent of aquatic vegetation beds in the lower Des Plaines River (river miles 273-286) in August 1987.

Lower Des Plaines River by (1) documenting the extent and species composition of the aquatic macrophyte community, (2) chemically analyzing water, sediments, and macrophyte tissues for heavy metals, PCBs, and organic pesticides, and (3) examining toxic substance interactions between sediments and macrophytes.

Study Site
The study site, in Will and Grundy counties, Illinois, includes a reach of the Des Plaines River from Brandon Road Lock and Dam (RM 286) to the confluence of the Des Plaines and Kankakee rivers (RM 273) (Fig. 1). The tributary Grant Creek enters the Des Plaines River near RM 274; Mobil Oil, AMOCO, Olin Matheson, Commonwealth Edison, and Rexall Chemical are located along this reach. Treated effluents released into the Sanitary and Ship Canal by the Metropolitan Sanitary District of Greater Chicago enter the Des Plaines River 4 miles upstream of the study reach. Toxic sediments have been identified in the North Branch of the Chicago River and the Des Plaines River (Blodgett et al. 1984, Illinois Environmental Protection Agency 1984).

The study reach was divided into eight segments of approximately equal size (Fig. 1). Segment boundaries were delimited without separating heavily vegetated areas.

Materials and Methods
Low-altitude, natural-color aerial photography and ground-truth surveys (Motorola Mini-Ranger III System, transect methods, and hand mapping) were used to document location, extent, and species composition of

Table 1. Macrophytes collected in the Des Plaines River for chemical analyses in 1986 and 1987. Water and sediments were collected at all locations. RM = river mile.

Location	Macrophytes	1986	1987
Segment 1	Eleocharis acicularis	x	x
(RM 285.5)	Myriophyllum sp.		x
	Potamogeton cripsus		x
	Potamogeton nodosus		x
	Potamogeton pectinatus		x
Segment 4	Sagittaria latifolia	x	x
(RM 279.5)	Typha sp.	x	x
Segment 5a	Myriophyllum sp.	x	
(RM 277.5)	Potamogeton cripsus	x	
	Potamogeton nodosus	x	
	Potamogeton pectinatus	x	
	Vallisneria americana	x	x
Segment 5b	Myriophyllum sp.		x
(RM 276.8)	Potamogeton nodosus		x
	Potamogeton pectinatus		x
Segment 8	Potamogeton pectinatus		x
(RM 273.5)	Vallisneria americana		x

aquatic macrophytes in June or July, 1985-1987. Voucher specimens were collected, identified (Beal 1977, Correll and Correll 1972, Fassett 1967), and archived in the Illinois Natural History Survey Herbarium (ILLS). Data were recorded on base maps, digitized, and entered into a Geographic Information System (ARC/INFO) (Sparks et al. 1986, Tazik and Sparks 1987, Tazik 1988).

Eight macrophyte species (two emersed and six submersed) were collected and chemically analyzed in 1986 (Table 1). In 1987, macrophytes, water, and sediments samples were collected from five locations for chemical analysis. Prior to chemical analysis, macrophytes were divided into above-ground (shoots) and below-ground (roots) parts. Each sample analyzed was a composite or homogenate of several subsamples to assure thorough representation of the water, sediments, and plant species at each location. Samples were chemically analyzed for total cations using standard methods (Tazik 1988). Substances measured at or below detection limits for all sample sites or macrophyte species are not reported here.

Correlation analyses were used to examine the association between substance levels in sediments and plant tissues. Average linkage cluster analyses were used to identify similarities in sediments from the five locations, similarities in macrophyte species, and similarities between sediments and macrophyte tissues. (Pielou 1984, Sokal and Rohlf 1969, Wilkinson 1987). Variances were equalized prior to cluster analysis to prevent swamping of uncommon elements or those in lower concentrations by abundant elements (i.e., having higher means and variances) (Pielou 1984). For details of chemical analyses, mineral nutrient concentrations, macroinvertebrate communities, and

Table 2. Aquatic macrophytes collected in the Des Plaines River, 1985-1987. Growth forms are rooted (R), submersed (S), emersed (E), aquatic (A), terrestrial (T), floating (F), and floating-leaved (FL).

Scientific name	Common name	Growth form
Calamagrostis	Reed bentgrass	R T
Ceratophyllum demersum L.	Coontail	F A
Dianthera americana L.	Water willow	R E A
Eleocharis acicularis (L.) R. & S.	Needle rush	R E A
Elodea canadensis (Michx.) Planchon.	American elodea, waterweed	R S A
Gramineae	Grass family	R T
Lemna spp.	Duckweed	F
Lythrum salicaria L.a	Purple loosestrife	R E A
Myriophyllum sp.	Water milfoil	R S A
Nelumbo lutea (Willd.) Pers.	American lotus	R FL A
Nymphaea tuberosa Painea	White water lily	R FL A
Phragmites communis Trin.	Reed grass	R E A
Polygonum sp.	Smartweed	R T
Potamogeton crispus L.	Curlyleaf pondweed	R S A
Potamogeton pectinatus L.	Sago pondweed	R S A
Potamogeton zosteriformis Fernald.	Flatstem pondweed	R S A
Potamogeton nodosus Poirb	American pondweed	R FL A
Sagittaria latifolia L.	Common arrowhead	R E A
Scirpus fluviatilis (Torr.) Gray	River bulrush	R E A
Scirpus validus Vahl.	Soft-stem bulrush	R E A
Typha angustifolia L.	Narrowleaf cattail	R E A
Typha latifolia L.	Common cattail	R E A
Vallisneria americana (Michx.)	Eelgrass	R S A

a New taxa in 1986
b New taxa in 1987

macrophyte standing crops, see Sparks et al. (1986), Tazik and Sparks (1987), and Tazik (1988).

Results

Species Composition and Cover

Twenty macrophyte species were collected from the study reach from 1985 to 1987 (Table 2). The total vegetated area (46 ha) was nearly identical in 1985 and 1986 (Table 3). There were slight differences in the amount of cover of individual species and within segments, but overall there was little change between the 2 years. Total vegetative cover increased to 60 ha in 1987, primarily due to an increase in submersed macrophytes in Segment 5 (Table 3). The areas most heavily vegetated in all years were Segments 1 (RM 285.5), 4 (RM 279.5), 5 (RM 277), and 8 (RM 273.5) (Table 3, Figs. 2-5). Potamogeton spp., Myriophyllum sp., and Vallisneria americana accounted for approximately 70% of the total vegetated area in all years, with the most extensive cover in Segments 1, 5, and 8. Emersed vegetation, primarily Sagittaria latifolia, covered over 10 ha of the study reach, primarily in Segments 2, 3, and 4 (RM 279-284).

Chemical Analyses

Water samples from all sites contained low levels of nearly every element measured. Concentrations of 16 of 26 elements measured were at or below detection limits; all remaining elements were within quality criteria established for aquatic life (US Environmental Protection Agency 1976).

Table 3. Coverage (ha) of macrophyte species in study segments in 1986 and 1987.

Macrophyte	Segment 1	2	3	4	5	6	7	8	Total
				1986					
Ceratophyllum demersum	-	-	-	-	-	-	0.01	-	0.01
Myriophyllum sp.	0.82	0.02	-	-	-	0.02	-	-	0.86
Potamogeton pectinatus	-	0.02	-	-	-	0.02	-	-	0.04
Potamogeton crispus	0.19	-	-	-	-	-	0.02	-	0.21
Potamogeton zosteriformis	-	-	-	-	-	0.03	-	-	0.03
Potamogeton nodosus	0.17	-	-	-	-	-	-	-	0.17
Potamogeton spp. mix	0.34	-	-	-	-	-	-	-	0.34
Submersed species mix	8.77	-	-	-	14.62	0.09	0.09	8.55	32.12
Nymphacea tuberosa	-	-	-	-	-	-	0.12	-	0.12
Lythrum salicaria	-	-	-	-	-	-	0.02	-	0.02
Phragmites communis	-	-	0.05	0.06	-	-	-	-	0.11
Sagittaria latifolia	0.32	0.78	1.80	6.91	0.07	0.07	0.04	0.06	10.05
Typha spp.	-	-	0.10	1.48	0.18	-	0.26	0.11	2.13
Emersed species mix	-	-	-	-	-	-	0.04	-	0.04
Total	10.61	0.82	1.95	8.45	14.87	0.23	0.60	8.72	46.25
				1987					
Ceratophyllum demersum	-	-	-	-	-	-	-	-	-
Myriophyllum sp.	0.29	-	-	-	-	-	-	-	0.29
Potamogeton pectinatus	-	-	-	-	-	-	-	-	-
Potamogeton crispus	-	-	-	-	-	-	-	-	-
Potamogeton zosteriformis	-	-	-	-	-	-	-	-	-
Potamogeton nodosus	0.09	-	-	-	-	-	-	-	0.09
Potamogeton spp. mix	-	-	-	-	-	-	-	-	-
Submersed species mix	9.00	-	-	-	24.57	0.47	0.32	8.06	42.42
Nymphacea tuberosa	-	-	-	-	-	-	0.13	-	0.13
Lythrum salicaria	-	-	-	-	-	-	-	-	-
Phragmites communis	-	-	-	0.03	-	-	-	-	0.03
Sagittaria latifolia	0.48	3.07	3.67	7.05	0.07	0.10	-	0.09	14.53
Typha spp.	-	-	-	1.64	0.14	-	0.12	0.18	2.08
Emersed species mix	-	-	-	-	-	-	0.20	0.06	0.26
Total	9.86	3.07	3.67	8.72	24.78	0.57	0.77	8.39	59.83

Sediments samples in 1987 (Table 4) contained As, Cd, Cr, Cu, Fe, Hg, Pb, and Zn at highly elevated or extreme levels, the two highest categories of the Illinois Stream Classification System (Illinois Environmental Protection Agency 1984, Kelly and Hite 1984). Dieldrin and heptachlor epoxide were not detected and PCBs in the sediments were generally <1 ppm. All but a few substances were found in higher concentrations in sediments than in plant tissues (Table 5).

Substance concentrations in macrophyte tissues were generally comparable with those measured in other studies, although Zn levels were often higher in our plants (Campbell et al. 1985, Cowgill 1974, DiGiulio and Scanlan 1985). PCBs, Co, and Mn were consistently accumulated in greater amounts in macrophyte tissues than in sediments, while Zn and Ni were frequently present in amounts comparable to those in the sediments (Tables 4 and 5). Levels of Co, Cr, Se, Sn, and V were generally higher in plant roots than in shoots. Conversely, shoots had higher levels of PCBs, Zn, and Mn levels than did roots. Eleocharis acicularis, Myriophyllum sp., and

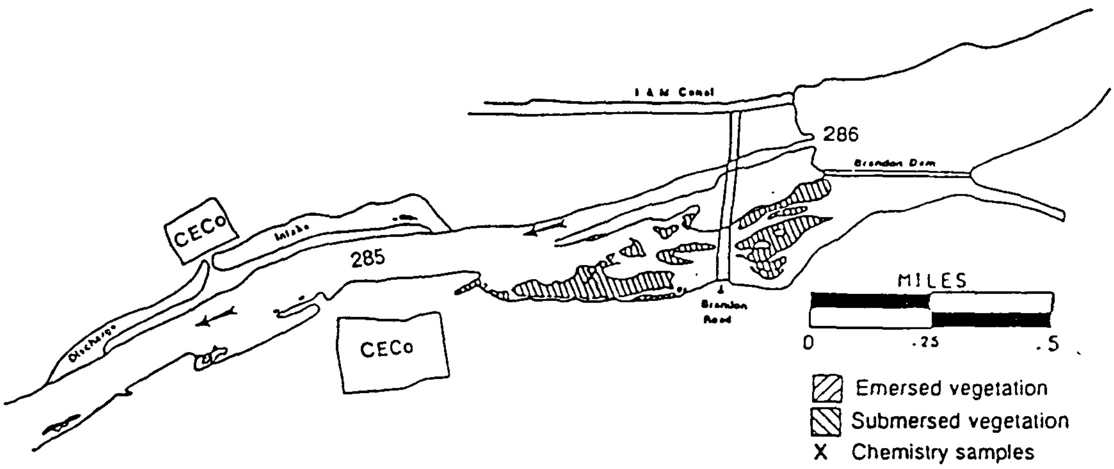

Figure 2. Location and extent of aquatic vegetation beds, Segment 1, lower Des Plaines River, August 1987.

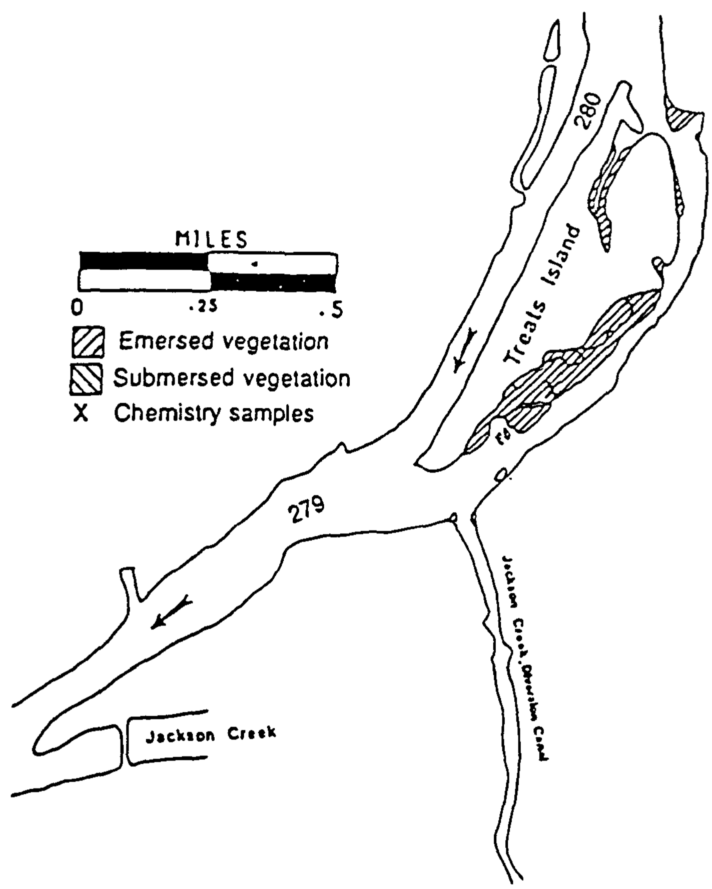

Figure 3. Location and extent of aquatic vegetation beds, Segment 4, lower Des Plaines River, August 1987.

Figure 4. Location and extent of aquatic vegetation beds, Segment 5, lower Des Plaines River, August 1987.

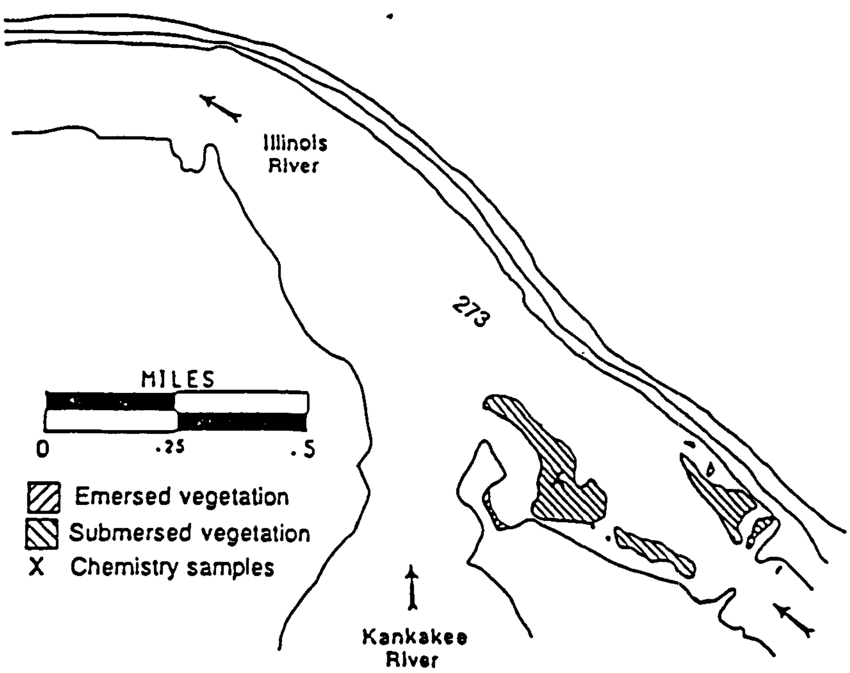

Figure 5. Location and extent of aquatic vegetation beds, Segment 8, lower Des Plaines River, August 1987.

Table 4. Concentration of minerals, metals, and PCB's in sediments collected from the Des Plaines River (RM 273-286), 1987. All concentrations are reported in ppm except Hg (ppb). Detection limits are in parentheses, concentrations less than detection limits are noted as <DL.

	Segments				
	1	4	5a	5b	8
Al (31.0)	35800	20300	26000	32000	41700
As (30.0)	44.5	<DL	<DL	34.5	47.0
B (2.80)	94.5	45.0	45.0	88.0	149.0
Ba (0.000)	221	376	320	714	602
Cd (3.80)	14.5	32.0	24.0	37.0	50.0
Cr (19.2)	40	188	134	255	344
Cu (3.40)	<DL	147.0	76.5	277.0	342.0
Fe (19.6)	40600	47100	29000	35800	37400
Hg (5.00)	91	1850	679	4180	3780
Mn (5.00)	1160	1450	1080	1040	1210
Ni (6.20)	69.5	121.0	46.0	71.0	76.0
Pb (11.6)	<DL	99	<DL	284	249
Sn (27.2)	<DL	41.5	<DL	56.5	119.0
V (27.2)	72.5	60.0	56.0	59.5	123.0
Zn (8.60)	197	1020	620	123	157
Total PCB (0.0001)	0.060	1.650	0.415	0.289	0.557

Vallisneria americana effectively accumulated Ba, Cd, Co, Cr, Ni, Zn, and PCBs (Table 5).

There were no significant correlations between substance levels in sediments and plant tissues. Cluster analyses showed no strong groupings of substance levels in sediments from the five locations. The strongest plant species groupings were identified in a cluster analysis of Cd, Cr, Hg, and Zn levels in plant roots; Potamogeton spp. grouped together, indicating that substance levels in roots were similar for species of that genus, and Vallisneria and Eleocharis grouped with Myriophyllum sp. in a cluster distinct from Potamogeton spp. (Fig. 6).

Discussion

A wide variety of submersed and emersed vegetation now inhabit the Lower Des Plaines River; all are typical temperate, riverine species (Clark et al. 1983, Donnermeyer and Smart 1985, Sparks 1984). Vegetation was abundant in four areas in all years; vegetation cover appeared to be increasing in Segments 2, 3, and 5.

Most trace metals entering aquatic systems eventually become incorporated in the sediments (Miller et al. 1983, Tessier and Campbell 1987). The Des Plaines River has been subjected to considerable pollution and its sediments are characterized by high levels of As, Cd, Cr, Cu, Hg, Pb, and Zn (Blodgett et al. 1984, Commonwealth Edison Co. 1986, Illinois Environmental Protection Agency 1984). The presence of toxic substances, and their availability to the biota, impacts habitat quality. Analysis of total cations is just the first step in understanding potential interactions of pollutants with biotic and abiotic components.

Many factors influence rates and patterns of substance concentration and translocation by aquatic macrophytes. Patterns observed may have been affected by (1) combining roots and rhizomes for analysis, (2) analyzing plants of varying age, (3) variation in local edaphic factors,

Table 5. Concentration of minerals, metals, PCBs, dieldrin, and heptachlor epoxide in macrophytes collected from the Des Plaines River (RM 273-286), 1986 and 1987. Samples designated by plant part were collected in 1987; other samples were collected in 1986. All concentrations are reported in ppm except Hg (ppb). Detection limits are in parentheses; concentrations less than detection limits are denoted <DL.

Macrophyte	Al (3.1)	B (0.28)	Ba (0.00)	Cd (0.38)	Co (0.30)	Cr (1.92)	Hg (5.00)	Mn (0.40)	Ni (0.62)
Segment 1									
Eleocharis acicularis	13500	43.4	168.0	14.40	12.4	73.6	347.0	4980	144.0
roots	7950	18.9	93.6	12.30	29.0	25.1	128.0	2430	87.7
shoots	1110	<DL	57.5	1.50	23.5	<DL	45.2	4840	68.1
Myriophyllum sp.									
roots	11200	39.7	72.9	9.05	31.3	30.8	110.0	887	71.5
shoots	1160	50.8	41.3	1.35	13.1	8.9	161.0	2340	44.4
Potamogeton crispus									
roots	3440	33.8	92.8	3.75	22.9	10.3	66.8	2710	64.2
shoots	1210	12.1	27.3	1.30	5.0	6.8	44.2	512	24.5
Potamogeton nodosus									
roots	6870	43.5	69.9	5.40	28.4	14.9	69.3	1230	58.8
shoots	1010	9.0	31.0	1.25	10.1	4.2	48.3	1400	40.6
Potamogeton pectinatus									
roots	1900	45.5	41.8	2.55	14.5	3.0	84.0	1000	29.1
shoots	1080	477.0	29.2	1.65	11.3	8.3	131.0	1550	47.3
Segment 4									
Sagittaria latifolia	4560	46.7	122.0	6.95	1.0	32.7	276.0	1300	44.1
roots	1020	17.1	106.0	3.40	6.5	8.3	124.0	195	11.8
shoots	529	3.3	9.2	0.75	1.3	6.5	66.6	105	4.9
Typha sp.	575	22.9	15.6	1.35	<DL	5.2	67.2	156	7.6
Segment 5a									
Myriophyllum sp.	6000	76.3	204.0	6.25	7.7	35.9	204.0	1360	44.0
Potamogeton cripsus	3900	43.9	102.0	5.20	3.6	28.7	242.2	614	40.8
Potamogeton nodosus	3340	35.1	113.0	4.35	4.0	28.1	236.0	958	52.0
Potamogeton pectinatus	3720	176.0	94.7	3.65	9.5	27.0	138.0	1120	100.0
Vallisneria americana	4560	46.7	122.0	6.95	1.0	32.7	276.0	1300	44.1
roots	4670	53.9	390.0	17.30	32.8	70.8	931.0	1120	55.8
shoots	2620	11.3	58.9	7.10	34.0	18.5	174.0	2740	88.5
Segment 5b									
Myriophyllum sp.									
roots	3450	47.4	509.0	6.70	39.8	<DL	81.5	1510	18.2
shoots	4410	69.0	89.7	1.60	8.7	2.3	38.1	2720	9.5
Potamogeton nodosus									
roots	1960	32.4	337.0	3.65	14.7	2.2	64.4	1200	8.4
shoots	2200	21.6	58.5	1.30	7.4	3.6	47.3	7700	16.3
Potamogeton pectinatus									
roots	2440	66.0	370.0	3.40	12.9	<DL	74.1	1410	6.0
shoots	3040	229.0	112.0	1.60	12.3	<DL	46.1	9300	19.0
Segment 8									
Potamogeton pectinatus									
roots	2850	79.0	215.0	4.60	13.2	19.3	156.0	1190	21.9
shoots	7650	241.0	141.0	4.85	29.5	10.4	61.6	5680	73.9
Vallisneria americana									
roots	3110	60.3	447.0	12.70	28.1	19.2	201.0	707	27.2
shoots	3030	7.8	51.4	5.95	19.0	14.4	106.0	2610	119.0

Table 5 (concluded).

Macrophyte	Pb (1.16)	Se (2.42)	Sn (2.72)	Total V (2.72)	Zn (0.860)	PCB (0.0001)	Dieldrin (0.0001)	Heptachlor epoxide (0.0001)
				Segment 1				
Eleocharis acicularis	18.8	<DL	20.00	28.0	508.0	21.700	0.1000	<DL
roots	<DL	12.0	8.95	24.0	306.0	1.220	<DL	0.0072
shoots	13.7	<DL	<DL	<DL	184.0	1.470	<DL	<DL
Myriophyllum sp.								
roots	<DL	18.8	7.50	31.9	178.0	1.100	0.0092	0.0096
shoots	6.8	6.2	<DL	2.8	168.0	1.340	0.0036	0.0050
Potamogeton crispus								
roots	<DL	4.5	<DL	5.0	156.0	0.997	0.0078	0.0088
shoots	<DL	<DL	<DL	8.3	195.0	1.100	0.0054	0.0078
Potamogeton nodosus								
roots	<DL	12.4	5.60	15.8	115.0	0.722	0.0040	0.0063
shoots	<DL	<DL	4.10	<DL	165.0	2.270	<DL	0.0087
Potamogeton pectinatus								
roots	<DL	3.3	5.85	4.1	82.0	0.866	0.0057	0.0072
shoots	<DL	3.4	<DL	<DL	327.0	1.130	0.0022	0.0052
				Segment 4				
Sagittaria latifolia	20.2	<DL	3.90	15.4	298.0	1.064	<DL	<DL
roots	<DL	4.8	<DL	5.6	93.3	0.557	<DL	0.0084
shoots	<DL	<DL	<DL	<DL	47.7	1.230	0.0050	<DL
Typha sp.	2.7	<DL	2.18	<DL	58.2	0.244	0.0016	<DL
				Segment 5a				
Myriophyllum sp.	29.3	<DL	2.80	18.0	262.0	3.020	<DL	<DL
Potamogeton crispus	9.4	<DL	9.00	9.8	268.0	1.250	<DL	<DL
Potamogeton nodosus	17.0	<DL	6.55	8.5	206.0	2.110	<DL	<DL
Potamogeton pectinatus	14.3	<DL	<DL	10.4	244 0	0.908	<DL	<DL
Vallisneria americana	16.7	<DL	7.35	24.7	557.0	1.800	<DL	<DL
roots	48.6	18.0	8.63	26.7	467.0	0.653	0.0031	0.0066
shoots	6.2	4.7	4.90	9.4	535.0	0.900	<DL	<DL
				Segment 5b				
Myriophyllum sp.								
roots	<DL	21.6	3.05	8.7	66.4	0.593	0.0042	0.0119
shoots	<DL	5.5	<DL	6.9	68.8	0.565	<DL	0.0064
Potamogeton nodosus								
roots	<DL	7.4	<DL	12.6	44.0	0.852	0.0038	0.0046
shoots	<DL	2.5	3.35	7.5	53.9	1.340	0.0127	<DL
Potamogeton pectinatus								
roots	<DL	11.7	5.50	12.4	28.9	0.922	0.0064	0.0088
shoots	<DL	3.6	<DL	6.3	41.5	0.630	0.0125	0.0100
				Segment 8				
Potamogeton pectinatus								
roots	<DL	8.0	5.70	18.2	115.0	1.300	0.0100	<DL
shoots	<DL	9.9	4.70	15.3	206.0	1.130	0.0040	0.0046
Vallisneria americana								
roots	<DL	19.1	7.50	24.5	222.0	0.854	0.0030	0.0064
shoots	<DL	4.1	7.50	4.3	524.0	0.837	0.0108	<DL

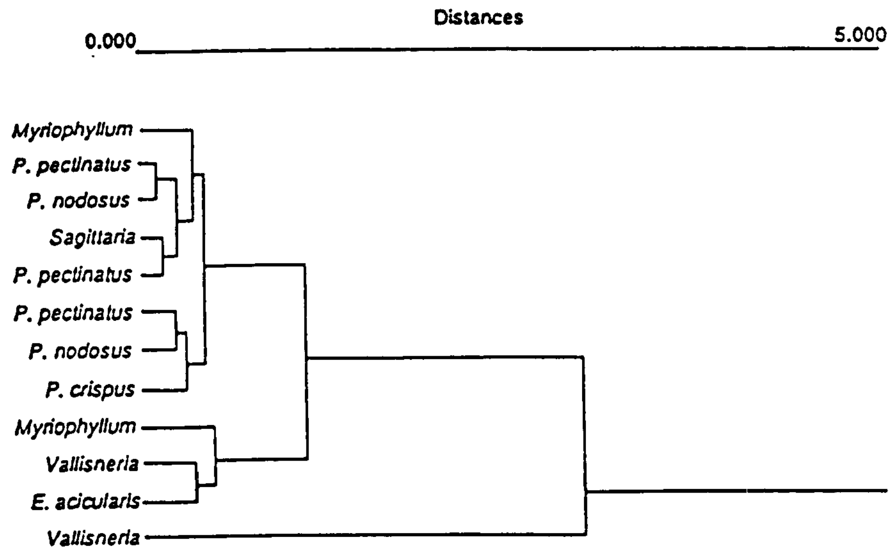

Fig. 6. Dendogram of cluster analysis of cadmium, chromium, mercury and zinc concentrations in macrophyte roots. The average linkage method is used, distance is measured in Euclidean distance (SYSTAT™).

and (4) differential uptake of substances (Campbell et al. 1985, Kraus et al. 1986, Kraus 1988, Miller et al. 1983). These data are not sufficient to establish a statistically significant relationship between substance concentrations in sediments and in plants; 65% of 105 cases examined by Campbell et al. (1985) showed no relation between these parameters. Nor do these data define individual macrophyte uptake and translocation processes. Nonetheless, substantial quantities of toxic substances were identified in sediments and macrophyte tissues. Once substances are concentrated by macrophytes, they may be stored in macrophyte tissues or released into the environment. For Co, PCBs, and Hg, which were concentrated by macrophytes to levels that exceeded those in sediments, release into the environment could pose a serious threat to other biota. Conversely, harvesting contaminated plants could be used as part of a rehabilitation plan. Cd, Co, Cr, V, Sn, and Se which are concentrated primarily in roots and rhizomes could also be removed from the system by harvesting macrophytes; moreover, this removal would not be complicated by seasonal senescence of shoots.

In conclusion, there is now locally abundant aquatic vegetation in the Lower Des Plaines River (RM 273-286). As macrophyte populations increase, water and habitat quality should improve (e.g., reduced turbidity, increased oxygen levels, larger and more diverse invertebrate populations, and an improved fishery). Toxic substances are clearly a part of this aquatic system, and interactions of aquatic plants and toxic substances, particularly those in sediments, can affect habitat quality. Rooted macrophytes can move toxic substances from deeper sediments to the water column and top sediments via uptake and acropetal translocation.

Conversely, substances that are concentrated and remain in belowground plant parts are unavailable to other biota, at least until those parts senesce. Removal of macrophytes that have accumulated toxic substances may provide a mechanism for rehabilitating polluted aquatic systems.

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Use of Acute and Chronic Bioassays to Assess the Applicability of Selected Advanced Wastewater Treatment Technologies for the Green Bay Metropolitan Sewerage District

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Abastract
Several state-of-the-art advanced wastewater treatment technologies were evaluated during pilot studies. All treatment endpoints received extensive chemical analysis as well as whole effluent bioassays. Results indicated that effluent from the existing carbonaceous treatment is toxic most of the time, whereas nitrified effluent streams showed no acute or chronic bioassay failures. Subtle effects on Ceriodaphnia were, however, observed. Tertiary treatment typically reduced these effects, though one treatment system introduced another source of toxicity inherent to the chemical process. Interpretation of bioassay results were further complicated by inconsistencies within the chronic test statistics procedure. These observations support the need to review all data generated during bioassays, such as mean growth rates or reproduction, rather than chronic "pass/fail" endpoints alone.

Introduction

The Green Bay Metropolitan Sewerage District (GBMSD) provides wastewater treatment for nine communities and two large pulp and paper industries. The existing facilities were placed into operation in 1975 and were designed to meet the community's needs through the year 1990.

The District's operation has consistently maintained compliance with EPA categorical effluent limits of 30 mg/l BOD and TSS, and a 1.0 mg/l total phosphorus limit as established by an International Joint Commission (IJC) Agreement between the U.S. and Canada.

The treatment plant effluent discharges to the Fox River at the mouth of Green Bay, Lake Michigan. Historical water quality problems of the Fox River and lower Green Bay have been well documented (Bertrand et al. 1976; Day 1978; Howmiller and Beeton 1971; Patterson et al. 1975; Peterman et al. 1980; Smith et al. 1988; Sullivan, and Delfino 1982). The lower Fox River/ Green Bay area has been designated as one of the 42 "Areas of Concern" by the IJC.

In 1987 the Wisconsin Department of Natural Resources (WDNR) issued notice of its intent to reissue a Wisconsin Pollutant Discharge Elimination System (WPDES) discharge permit to the GBMSD. The permit was to contain new and more stringent limits for CBOD, chlorine residual, fecal coliform and whole effluent toxicity, along with recommendations pertaining to future monitoring and effluent limits for certain toxic compounds, including ammonia, heavy metals and residual organics.

The Wisconsin DNR has recently developed new administrative codes NR105 and 106 for the control of toxics from point sources. These codes address over 100 toxic compounds, and also enable the WDNR to place a bioassay effluent limit or monitoring requirement in a WPDES permit.

Existing effluent data indicated the possibility of noncompliance with future permit conditions. Whole effluent bioassays performed in 1987 showed both acute and chronic failure to fathead minnows and Ceriodaphnia (Buttke and Rades 1987a,b).

GBMSD therefore commissioned a facility plan to address these and other issues. A major component of the plan included extensive pilot studies using state-of-the-art Advanced Wastewater

Treatment (AWT) technologies. The pilot studies were designed to evaluate alternative AWT processes likely to ensure compliance with the proposed and potential future permit requirements.

Methodology
Pilot studies were conducted in 1987 and 1988, with the majority of work occurring between November, 1987 and March, 1988. Processes investigated include:

- Single-stage nitrification (identified as B12)
- Powdered Activated Carbon (PACT™) nitrification
- Alum/sulfide treatment
- High-lime treatment
- Filtration
- Carbon adsorption

Four pilot study tests evaluated single-stage nitrification followed by the AWT systems. Four more tests evaluated the PACT™ process followed by the AWT systems.

Each location within the pilot system identified as a possible treatment endpoint was sampled intensively for chemical parameters and whole effluent bioassays, including: activated sludge nitrified effluent (B12); B12 after chlorination/dechlorination; PACT™ secondary effluent; PACT™ filter effluent; alum/sodium sulfide filter effluent; alum/sodium sulfide carbon column effluent; high lime recarbonation clarifier effluent; high lime filter effluent; high lime carbon column effluent; and existing carbonaceous effluent (B15). A total of eight 7-day bioassays were performed on eight effluents and a Green Bay dilution water control. All tests were conducted in strict accordance with EPA protocol (Horning and Weber 1985). The acute bioassay measures percent survival in 100% effluent. The chronic bioassay determines any sublethal effects, measured as reduction in growth for fathead minnows, or a decrease in Ceriodaphnia reproduction. The chronic "pass/fail" endpoint is based on observed sublethal effects at a particular effluent concentration, termed the "Instream Waste Concentration" (IWC). This concentration is defined as the percentage composition of the effluent in the receiving water stream assuming a stream flow of 25% of the historical minimum 7-day flow expected once in ten years (7-day Q_{10}). The chronic test statistics calculate a "No Observed Effect Concentration", or N.O.E.C. The GBMSD IWC was calculated to be 34%. Therefore, the N.O.E.C. calculated from a given test must be at least 34% to pass the chronic criterion.

Results and Discussion
A total of eight 7-day test periods were utilized for bioassay analysis. Four of these included full-scale nitrification as the effluent source to the tertiary systems, while four runs reflect use of the PACT™ pilot plant effluent to drive the tertiary systems. Each 7-day run is identified by the "mode" of nitrification, e.g. "B12" or "PACT™", followed by the chronological run number.

Table 1 presents an overall summary of the 63 total acute and chronic bioassays performed. A total of six acute failures were noted; five in the existing carbonaceous effluent (B14/15), and one in the alum filter effluent. The B14/15 mortalities were presumably due to high ammonia levels, while residual sulfide was believed to be responsible for the alum mortality.

Referring to Table 1, a total of 12 chronic failures were noted during the entire study. Six of these corresponded to the B14/15 effluent. Of the remaining six failures, three were in the alum filter, two in the lime filter effluent, and one in the

Table 1. Number of acute and chronic bioassay failures observed during the GBMSD Pilot Study.

	Acute Bioassay Results				Chronic Bioassay Results			
	Fathead Minnow		Ceriodaphnia		Fathead Minnow		Ceriodaphnia	
Effluent	N	Failures	N	Failures	N	Failures	N	Failures
Full Scale Nitrification								
B12	4	0	4	0	3	0	4	0
B12 (chlor/dechlor)	2	0	2	0	2	0	2	1
Alum Filter Eff.	4	0	4	1	3	0	4	0
Alum CC Eff.	4	0	4	0	3	0	4	0
Lime Recarb Eff.	4	0	4	0	3	0	5	0
Lime Filter Eff.	4	0	4	0	3	1	3	1
Lime CC Eff.	4	0	4	0	3	0	4	0
PACT™ Nitrification								
PACT™ Secondary Eff.	4	0	4	0	4	0	4	0
PACT™ Filter Eff.	4	0	4	0	4	0	4	0
Alum Filter Eff.	4	0	4	0	4	0	4	3
Alum CC Eff.	4	0	4	0	4	0	4	0
Lime Recarb Eff.	4	0	4	0	4	0	6	0
Lime Filter Eff.	4	0	4	0	4	0	2	0
Lime CC Eff.	4	0	4	0	4	0	4	0
Carbonaceous								
B14/15	8	5	8	0	7	5	8	

B12-1 (chlorination/dechlorination) effluent. Figure 1 presents a graphical depiction of chronic bioassay results for run B12-3. Results are displayed for both fathead minnow and Ceriodaphnia. All graphs contain results from the 100% effluent analyses only. Each summary graph for fathead minnow data includes percent survival (bar graphs represent actual percent survival) and mean final weights including 95% confidence interval. N.O.E.C. values, as calculated by WDNR computer programs, are listed in parentheses above each bar or mean weight interval. Results for Ceriodaphnia include percent survival (again shown by bar graphs) and mean number of offspring per individual including 95% confidence interval. N.O.E.C. values are also listed above each bar or mean number interval. The following paragraphs discuss bioassay results in more detail, grouped by treatment system.

Carbonaceous Effluent (B14/15)
Significant detrimental effects were noted on all B14/15 bioassays. For fathead minnows, five of seven chronic tests yielded failures. One out of eight Ceriodaphnia tests failed. As previously noted, ammonia is thought to be the main source of toxicity. Fathead minnows are known to be highly

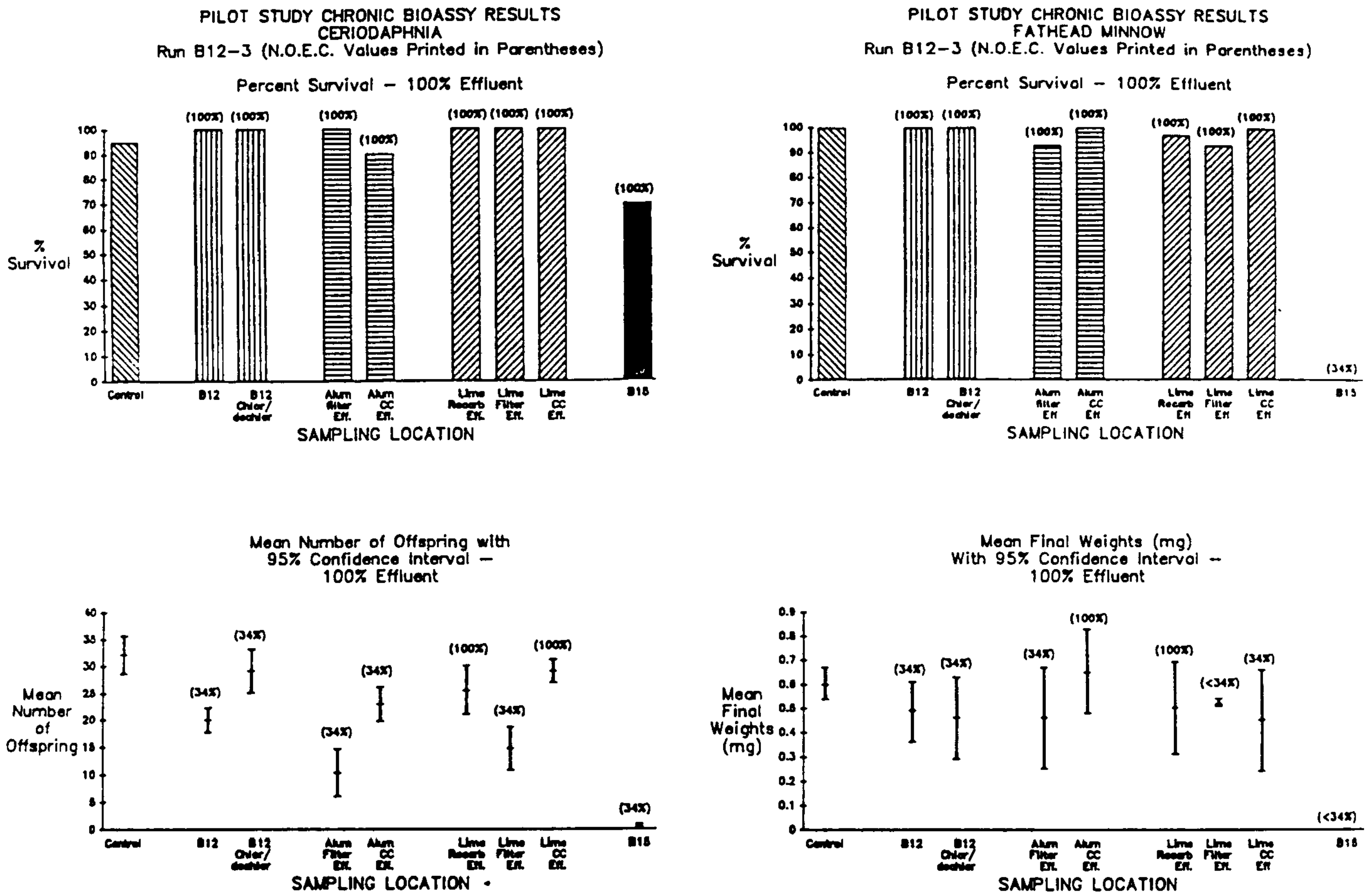

Figure 1. GBMSD pilot study bioassay results, run B12-3.

sensitive to ammonia, and this is reflected in the test results.

Full Scale Nitrification

Full scale nitrification yielded an effluent which passed all bioassays during the test period. Ammonia levels were generally low (less than 1.0 mg/l), further supporting the proposition that ammonia is the primary toxicant in the GBMSD effluent.

Fathead minnow results for all four B12 runs showed no significant difference to the control (all N.O.E.C. values 34% or greater). Some effect in Ceriodaphnia reproduction was noted in runs B12-1, B12-3 and B12-4, even though all N.O.E.C. values were 34% or above.

The WDNR had expressed interest in including bioassay analysis of a chlorinated/dechlorinated effluent to determine whether this process might in itself cause a toxicity problem. Therefore, simulation of chlorination/dechlorination was performed in the laboratory on B12 effluent during runs B12-2 and B12-3. Samples were first chlorinated at an approximate dosage of 10 mg/l for 30 minutes, then dechlorinated with sodium thiosulfate. These samples were then used for bioassay analysis. Fathead minnow results were identical to unchlorinated B12 effluent, but Ceriodaphnia reproduction was significantly reduced during run B12-2. Therefore, the dechlorination procedure was modified so that concentrations of excess sodium thiosulfate were minimized.

Samples from run B12-3 showed identical Ceriodaphnia results to unchlorinated B12 effluent.

PACT™ Nitrification
Bioassays were performed on two effluents from the Zimpro PACT™ pilot plant: secondary effluent; and, filter effluent. No chronic failures were noted throughout the four PACT™ runs. Fathead minnow results were very similar to the control. Ceriodaphnia reproduction was slightly effected during runs PACT™-3 and PACT™-4.

Alum AWT
Bioassays were performed on two effluents from the alum AWT system: alum filter effluent; and, alum carbon column effluent.

Fathead minnow results showed no significant difference from the control on all eight runs except for run B12-3, when the filter effluent showed a slight effect (N.O.E.C. of 34%).

Ceriodaphnia results, however, showed significant toxicity-related effects. Recall that run B12-1 showed an acute toxicity failure for Ceriodaphnia in alum filter effluent. This problem was believed to be related to residual sulfide from the alum/sodium sulfide treatment, in conjunction with the short detention time of the pilot system. A secondary aeration/holding tank was incorporated into the system hoping to drive off any residual sulfide, and no more acute failures were observed. However, chronic effects were noted throughout the remaining bioassays. Three of the eight alum filter effluent Ceriodaphnia bioassays failed the reproduction test. However, even the five tests which passed showed obvious detrimental effects. It was further noted that during four out of the eight runs, the carbon column treatment step improved conditions to the point that the bioassay results were not significantly different from the control. The remaining four runs showed improvement, but to a lesser degree.

It was thought, at first, that the added aeration step had alleviated the sulfide problem, as the next few bioassays yielded no failures. It later became apparent that a toxicity problem in the alum system was still present. In order to verify the effectiveness of the aeration tank, five grab samples of filter effluent were collected during run PACT™-4. Results showed a range of 41 to 74 $\mu g/l$ (ppb) residual sulfide. One sample was split prior to analysis, with one aliquot receiving an extra hour of vigorous aeration in the laboratory prior to analysis. This extra aeration step reduced the residual sulfide level from 41 to 35 $\mu g/l$.

Residual sulfide levels at these concentrations could be the source of toxicity in the alum system. The EPA "Gold" book criterion for undissociated H_2S for fish and aquatic life (in fresh and marine water) is 2.0 $\mu g/l$. Residual sulfide levels found in the alum system, however, are not identical in form to undissociated H_2S. Sulfide exists in three forms in water; H_2S, hydrosulfide (HS-) ions or sulfide (S=) ions. The proportion of each is controlled primarily by pH. As pH drops below 9.0, the proportion of undissociated H_2S (and therefore the toxicity) increases. The aeration step which was added to the alum pilot system increased the alum filter pH from approximately 7.1 to 8.0. This aeration-induced pH increase may have served to reduce the sulfide toxicity, rather than reducing the sulfide concentration.

To further investigate the sulfide toxicity problem, the pilot system was operated again in May, 1988. Ceriodaphnia bioassays were performed on alum filter effluent, both before and after a chlorination/dechlorination procedure. The chlorination process was suggested as a possible means of oxidizing any residual sulfide. The chlorination/dechlorination procedure was identical to that which was used on earlier bioassays, using sodium thiosulfate to dechlorinate.

Results of sulfide analyses indicated that the chlorination process did remove approximately half of the residual sulfide, though daily variability was considerable. Sulfide levels after chlorination ranged from <2 μg/l to 78 μg/l. Bioassay results were very similar to the previous eight tests, showing significant effect on Ceriodaphnia reproduction. The chlorination/dechlorination process reduced the level of toxicity, but to only a minor degree.

High Lime AWT

Bioassays were performed on three effluents from the high lime AWT system: recarb clarifier effluent; lime filter effluent; and, carbon column effluent.

Fathead minnow results showed no significant difference from the control on five of the eight (total) runs. Run B12-2 showed a significant effect for all three effluents, presumably caused by very low (approximately 20 mg/l) alkalinity concentrations observed in the lime system during this run. Lime system alkalinity values measured during the other pilot runs were all above 30 mg/l. It is thought that the lower than average operating load from one of the two paper mill influent streams is the reason for the low alkalinities seen during run B12-2. Alkalinity values close to 20 mg/l have the effect of increasing the toxicity of heavy metals and other compounds. It is believed that the B12-2 run results reflect this phenomenon.

In order to prove that the observed toxicity was alkalinity related, a duplicate series of lime system effluents with added $NaHCO_3$ were tested during the next bioassay. However, alkalinities in the lime system returned to the 30-40 mg/l range, and no toxicity was observed in either sample series.

The high lime system consumes alkalinity during the chemical reactions involved during treatment. Even though the other seven bioassays showed no repeat of this occurrence, it should be considered a potential problem for future full-scale application.

The lime filter effluent sample failed the fathead minnow growth test on run B12-3 (shown on Figure 1 as <34%). However, it appears that this failure is more related to a statistical error than to toxicity. The confidence interval around the mean weight value is extremely small, normally an indication of high data reliability. This narrow range of variability, however, allows the WDNR statistics program to detect very small differences when compared to the control. In effect, if the replicate weight variations had been greater, the N.O.E.C. would have been much higher, even if the mean value remained the same. Realistically, therefore, this test should not be considered a "fail".

An unusual event occurred with the fathead minnow bioassays during run PACT™-4. Relatively high mortalities were observed in the lime system samples for one day of the test, corresponding to effluent collected on February 18, 1988. The number of mortalities decreased as treatment advanced (i.e. most mortalities in recarb clarifier effluent, least in carbon column effluent). No further mortalities were observed. Currently, there are no obvious explanations for this occurrence. Review of chemical data shows no obvious problems, and no operational difficulties were noted. Even so, no acute or chronic test failures were observed for the run.

Ceriodaphnia results from high lime system bioassays indicate a subtle recurring effect on reproduction, particularly in the lime filter

Table 2. Ammonia concentrations measured in bioassay samples during the GBMSD Pilot Study. (Weekly average value followed by daily range.)

	Ammonia-Nitrogen (mg/l)	
Run	**B14/15**	**B12**
B12-1	17	<1
B12-2	13	<1
B12-3	22	2.3 (<1-5.4)
B12-4	14	<1
	B14/15	**PACT™**
PACT™-1	16	1.8 (<1-5.6)
PACT™-2	13	3.0 (<1-12.1)
PACT™-3	18	<1
PACT™-4	14	1.1

effluent. There were no chronic Ceriodaphnia failures observed in seven out of eight runs, but often times the N.O.E.C. values decreased through the system, an obviously anomaly. Lime filter effluent failed the reproduction endpoint on run B12-2, but it appears the previously discussed concerns regarding statistical evaluation of a narrow confidence interval may again be the reason for the failure.

An alteration to the Ceriodaphnia bioassay regime was made during the last three runs: PACT™-3, PACT™-4 and B12-4. This change entailed collecting a lime clarifier effluent sample (normally at pH = 11.2) and using CO_2 gas to adjust the pH to 7.0 prior to use in a bioassay. (The pilot system normally used sulfuric acid to adjust pH.) Therefore, Ceriodaphnia bioassays were performed on recarb clarifier effluent, lime clarifier effluent W/CO_2, and lime carbon column effluent during these runs. The recarb clarifier and lime clarifier W/CO_2 samples should have been identical except for the method of pH adjustment.

Results from this assessment were inconclusive. For runs B12-4 and PACT™-3, the two effluents had identical results. For run PACT™-4, the N.O.E.C. value for clarifier effluent W/CO_2 was lower than the recarb clarifier effluent value.

It appears, therefore, that some system-related chemical reaction or other factor may be exerting a small but measurable effect on the Ceriodaphnia chronic bioassay.

Ammonia Concerns

Historical bioassay results had indicated that ammonia was thought to be a major source of toxicity observed in GBMSD effluent. The GBMSD Facilities Plan intends to address ammonia, and so it was hoped to perform all pilot study bioassays on ammonia-free effluent in order to identify any other toxicity causing compounds. However, due to fire-related operational problems at one of the paper mills, influent ammonia concentrations sometimes exceeded nitrification capacities resulting in ammonia bleed-through to the tertiary treatment systems.

Table 2 presents ammonia concentrations throughout the pilot system during the study. Ammonia values exceeded 1.0 mg/l (weekly average) on three out of the eight runs. The highest daily value was reported during run PACT™-2, at 12.1 mg/l. The weekly average for this run was 3.0 mg/l. Bioassay results from PACT™-2 and B12-3 indicate a slight effect noted for fathead minnow growth, though no test failures occurred.

Weekly average ammonia values for existing carbonaceous effluent (B14/15) are also included in Table 2. It is interesting to note that for an average ammonia concentration of approximately 16 mg/l (entire study), failure rates for acute and chronic bioassays were 63% and 71%, respectively.

Bioassay Procedure Concerns

The 7-day chronic bioassay test procedures, as conducted during the GBMSD pilot study, have been developed primarily by the EPA. Several changes in techniques and procedures have been made during recent years, and even today, the methods appear to be in a state of continuing evolution.

The GBMSD experience with the test methods, themselves, was mostly positive. Overall, the tests appear to be credible and repeatable. It is interesting to note that the two organisms seem to respond quite differently to differing toxic agents, thus supporting their selection as complementary test animals.

Bioassay results have identified possible toxicity problems affiliated with some of the treatment systems tested, even when results of chemical analysis do not clearly show such evidence. However, during review of multiple data sets, several inconsistencies were noted relating to the statistical program which calculates final N.O.E.C. values.

For example, Green Bay dilution water used during the first three runs of the pilot study caused significant mortality to fathead minnows, both in the control samples themselves and in the 34% effluent samples. The problem appeared to be excessive numbers of bacteria or fungi in the bay water. Fish that died were observed to have fungus-like growths in their gut, and a mat-like layer developed on the bottom of the 34% effluent beakers each day. (This problem was eliminated by changing the water collection site from the east shore to the west shore.) The statistics program responded to this condition by lowering the "standards" of the test, in one case assigning a N.O.E.C. value of 100% to an effluent that achieved only 57% survival in 100% effluent. A later test, with 100% control survival, calculated a N.O.E.C. value of 34% for an effluent that achieved 87% survival in 100% effluent. Therefore, it appears that it is to the dischargers advantage to conduct effluent bioassays using dilution water that is mildly toxic. Clearly, improvements in the statistical analysis program would seem appropriate.

Another inconsistency involves the previously discussed situation where replicate variability is very low, allowing the statistics to detect very small differences between mean values of the control and the effluent. This means that the statistics seem to expect variability between replicates, and that a high degree of confidence regarding the actual test data may actually result in a lowered N.O.E.C. value. It would seem prudent, therefore, to review all bioassay results, such as that included in Figure 1, rather than to judge the test based strictly on N.O.E.C. values.

The GBMSD experience regarding EPA/WDNR bioassay test procedures was acceptable, though some inconsistencies with the statistics program were noted. Our experience tends to support the need to review bioassay results from a biological as well as a mathematical perspective. Bioassay data generated were useful in the final selection process within the GBMSD Facilities Plan. Figure 2 contains a comparison of bioassay results between existing carbonaceous effluent

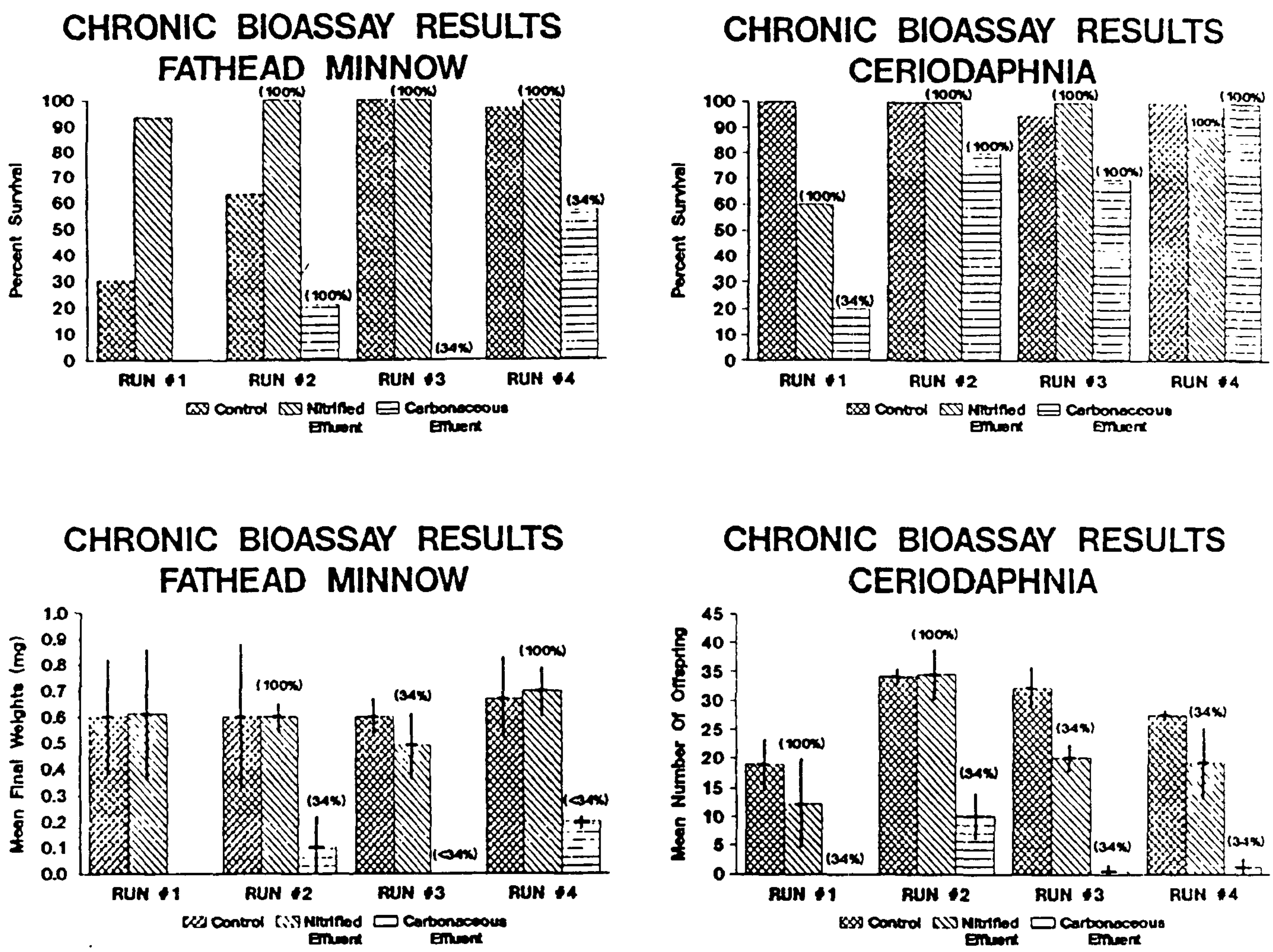

Figure 2. Comparison of bioassay results between GBMSD carbonaceous effluent, nitrified effluent, and Green Bay control water.

and full scale nitrification. Results from bioassays performed on nitrified effluent show a significant improvement over carbonaceous effluent. In fact, nitrified effluent results show only minimal variation from the receiving water controls.

Summary

A total of eight 7-day chronic bioassays were performed on eight effluents during the GBMSD pilot study. Excluding existing carbonaceous effluent (B14/15), only one acute and six chronic failures were observed throughout the entire test period.

Effluent from the full scale nitrification quadrant (B12) passed all acute and chronic bioassays. Fathead minnow results from all runs showed no significant difference from the control. A slight effect was noted in Ceriodaphnia reproduction, though not to the level of test failure.

Results from the PACT™ pilot plant effluent were very similar to B12 effluent with no apparent effect noted on fathead minnows, but a slight effect noted in Ceriodaphnia reproduction.

Results of the bioassay program show concern regarding residual sulfide levels in the alum/sodium sulfide treatment system. Significant reductions in Ceriodaphnia reproduction were noted, including one acute and three chronic failures. Subsequent testing indicated that residual sulfide is a definite problem with this form of treatment, though it is not known how the pilot scale results would apply to a full-scale operation.

Three separate effluents of the high lime system were analyzed. The effluent from this treatment system is characteristically low in alkalinity. Bioassay results have shown that the effluent alkalinity must be maintained at 30 mg/l or more in order to minimize increased toxicity from various compounds. A slight reduction in Ceriodaphnia reproduction was noted in the lime system effluents and may be related to the system itself. Bioassays on effluent using CO_2 gas for pH adjustment, instead of sulfuric acid, showed no apparent improvement.

High lime system results are difficult to assess completely, as the nitrified effluent which fed the lime system was already relatively nontoxic. However, if bioassay results from the influentto the lime system showed subtle effects as compared to the controls, the lime treatment typically improved the results. As with the alum system, the carbon column polishing step significantly improved the Ceriodaphnia bioassay test results if the influent stream showed depressed reproduction.

Several inconsistencies were noted relating to the statistical program which calculates final N.O.E.C. values. Results obtained during the GBMSD pilot study support the need to review all bioassay results, such as graphical plots of actual data, rather than to judge the test based strictly on N.O.E.C. values.

Acknowledgements

Credits

The Institute of Paper Chemistry, Appleton, Wisconsin, was contracted to conduct the bioassay analyses. George Buttke served as Project Officer, while Dave Rades served as Project Administrator. The pilot study was a joint effort involving all divisions with the GBMSD. CH2M Hill was the Facilities Plan consultant. This paper was accepted for presentation at the 62nd Annual WPCF Conference in San Francisco, but was cancelled due to the earthquake of October 17, 1989.

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Land Use Influence on Fish Communities in Central Indiana Streams

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Abstract
In recent years a number of large rivers including the Ohio and Wabash Rivers have experienced better environmental conditions and fish communities principally because of improvements in point-source discharges. Further progress is not likely to occur until NPS sources are reduced. The trends in smaller streams in agricultural landscapes are less encouraging. The fish communities in several stream systems in the Wabash River drainage have undergone sharp changes in character during the past decade or two. The sequence of change is a sudden loss of darters, followed by the disappearance of centrarchids, and then smallmouth bass. In extreme cases this is followed by a loss of Moxostoma species and a variety of minnows as more and more of the watershed is converted to tilled fields. Sporadic spills of fertilizer and feed lot wastes no doubt accelerate and confound the overall trends. The changes are not gradual and linear. The presence of "good" refugial tributaries permits a natural "reseeding" during benevolent years. The trends observed suggest that streams which have only recently lost their smallmouth bass populations may be rehabilitated with relatively modest effort and expense.

Introduction
The influence of agricultural on Indiana streams can be roughly categorized into (a) point source influences such as animal feed lots and fertilizer spills and (b) non-point source (NPS) influences including tilled fields and pastures.

Agriculture occupies 70% of Indiana lands and 78% of this land is devoted to rowcrop agriculture (55% of Indiana), mostly corn and soybeans. Of the estimated 108 million tons of soil annually eroded from Indiana land, nearly four-fifths (79%) originates from tilled fields that occupy slightly more than half of Indiana. Perhaps the Illinois rule-of-thumb estimate states the problem most succinctly: for every bushel of corn harvested, two bushels of soil are lost.

It is difficult to convince most people, including farmers, politicians, and engineers, that soil is a pollutant although they might readily agree that pesticides, herbicides, and fertilizers do pollute water. It is also extremely difficult to document the chronic effects of NPS pollution apart from the sporadic fish kills caused by specific agricultural activities.

In recent years, a number of large rivers including the Ohio and Wabash Rivers have experienced better environmental conditions and improved fish communities principally because of refinements in waste treatment of point-source discharges. Further progress is unlikely until NPS contributions are reduced. In the Wabash River the number of catchable game fish is more than 10 times as great as it was in the 1970's and the non-game species have similarly increased. Furthermore, the size of fish is bigger. About the only species which have declined are carp and gizzard shad. For the first time in over 20 years, predator fishes in the Wabash River are numerous enough to control gizzard shad.

Why this rather sudden improvement? It is partly because of improved waste treatment of towns and industries within the basin. Decomposable organic matter (BOD) today is only 2.5 to 3.0 mg/l today compared to 4.5 to 5.0 mg/l 15 years ago (Gammon 1989). That translates into better oxygen condi-

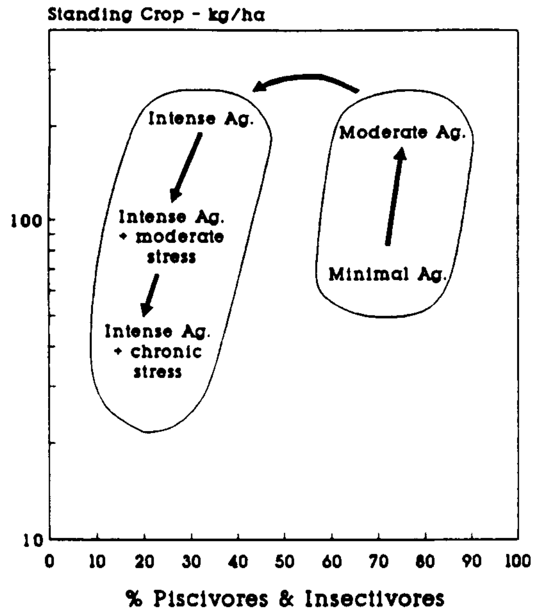

Figure 1: Changes in the fish communities of central Indiana streams under agricultural development.

tions through reduced dissolved oxygen deficits. An additional contributing factor might well be the 1983 PIK program, a year during which 25% less corn and soybeans were grown with concomitant reductions in applications of herbicides and pesticides.

In order to further improve river ecosystems nutrient delivery to the river must be reduced. Half of the carbonaceous BOD entering the middle Wabash is estimated to come from agriculture (HydroQual 1984). Phytoplankton, mostly diatoms, colors the Wabash River brown during the summer with as much as 100,000 algal cells per ml and chlorophyll-a concentrations exceeding 150 ug/l. High densities of algae produce two undesirable ecological effects: (a) the dissolved oxygen level is depressed at night and (b) the turbidity interferes with the ability of predator fish such as bass and walleye to locate food. In one segment of the Wabash it has also caused fishkills on at least two occasions (Parke 1985, Parke and Gammon 1986). There is also an undesirable recreational effect. Few people care to swim, canoe, or fish in turbid rivers which are perceived as being "dirty".

The trends of fish communities in smaller streams in Indiana's agricultural landscapes are quite different and not nearly as encouraging. Some are known to have undergone sharp changes in character during the past decade or two. It is likely that others are going to follow suit or, perhaps, have already done so without our knowledge.

The pattern of change was demonstrated during a Model Implementation Project study of three central Indiana stream systems (Gammon et. al. 1983). Moderate agricultural development of a watershed may only result in an increase in fish standing crop with no measureable change in the darter, sunfish, and bass components of the community (Figure 1). Further agricultural expansion, however, ultimately results in a sudden loss of darters, centrarchids, and possibly catostomids with expansions of omnivores and detritivores. Further changes occur through the loss of Moxostoma species (redhorse) and a variety of minnows as more and more of the watershed is converted to tilled fields. The changes occur first in smaller streams and then progress downstream.

Methods and Materials

The streams discussed in this paper are located in rural areas of Indiana and are not influenced strongly by industry, mining, or municipalities (Figure 2). The fish communities of most of the included streams have been intensely examined at multiple stations over several years during the past 12 years. Stream segments measurably influenced by towns or industries have been excluded from the analyses.

Methods of collecting fish vary with size of stream. A seine and backpack electrofisher were used in most small second and third order streams. Larger streams were electrofished with an electric seine, a backpack electrofisher carried in a boat, and/or a seine in shallows areas.

Data on agricultural landuse was obtained variously from (a) estimates by Soil Conservation Service personnel, (b) detailed computer analysis of LandSat imagery (Hyde, Goldblatt, and Stolz 1982), and (c) conventional analysis of enlarged LandSat infrared photographs taken during early summer, as described below.

Using topographic maps of the stream or tributary of concern, the drainage area perimeter was determined and drawn onto rescaled drainage maps (Hoggatt 1975). An infrared LandSat photograph taken on June 10, 1978 provided good contrast of permanent vegetation as grass or trees (red) from tilled field (tan and black). After establishing the best darkness setting the watershed of interest was xeroxed to produce an acceptable dark copy of the red portions in contrast to the lighter portions.

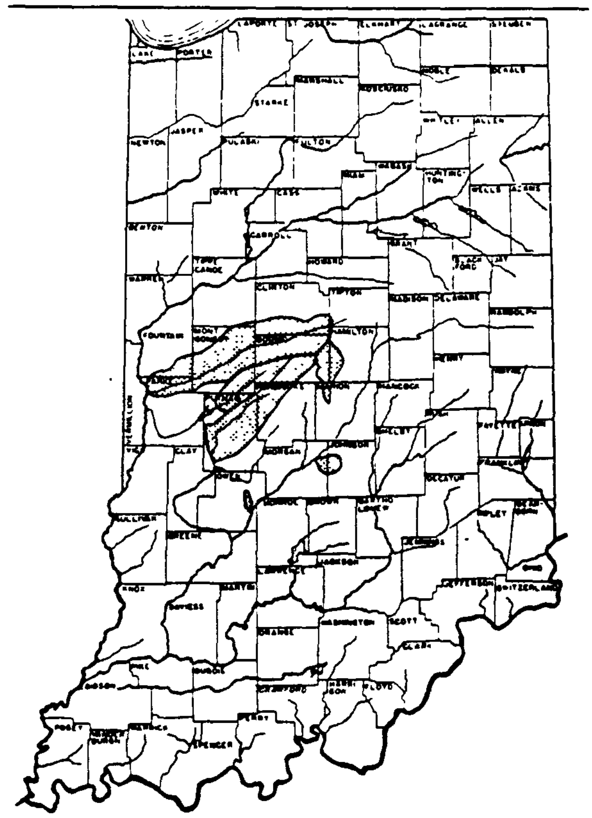

Figure 2: Location of Study Streams.

This xerox copy was superimposed over the scaled drainage maps on a light table, the drainage basin boundary was traced onto the xerox copy, and enlarged to 150%. This copy was placed over a fine transparent grid on a light table. If single grids contained more than 50% vegetation a mark was made. The total number of grids marked in relation to the total number of grids provided an estimate of the percentage of the drainage basin area in rowcrop agriculture.

Results

Data on the IBI of fish communities and the percent of watershed devoted to rowcrop agriculture are summarized in Table 1. A majority of streams flowed through watersheds with more than 65% of the area in row crop agriculture.

Discussion

The IBI should function well in assessing the degree to which stream fish communities are influenced by non-point source pollution because 5 of the 12 metrics include species sensitive to sediment pollution. The data from Table 1 was divided into two parts: (1) smaller streams (Orders I and II), and (2) larger streams (Orders III and IV). IBI values for the larger streams generally exceeded those for smaller streams, but there was considerable overlap. Studies in Ohio and Illinois indicate a direct relationship between stream order and IBI values (Ohio E.P.A. 1988, Hite and Bertrand 1989).

The IBI of the fish community and the percent of the watershed in rowcrop agriculture is summarized in Figure 3. The few watersheds having 50% or less of their areas in rowcrops contained fish communities with IBIs of 50 or greater. There was a statistically significant correlation (Spearman) at the 0.05 level between percent rowcrop

Table 1: Drainage basin area, agricultural land-use, and fish communities of central Indiana streams.

Stream	Stream Order	Drainage Basin Area km2	(mi²)	% Rowcrop	IBI	Number of Species Dar.	Sun.	Bass
			Sugar Creek System					
Mainstem								
Above Darlington	III	829	(320)		47.1[a]	2.0	0.9	0.5
Darl. to C-ville	IV	1318	(509)	75	49.7[b]	3.0	1.2	1.0
C-ville to mouth	IV	2100	(811)	60	48.0[c]	2.7	1.2	1.3
Tributaries								
Rush	I	42.2	(16.3)	64	44	2	0	0
Sugar Mill	II	197.4	(76.2)	69	42	1	0	2
Indian	II	65.5	(25.3)	70	38	1	3	1
Rattlesnake/	III	81.3	(31.4)	59	52	2	3	2
Offield	II			59	42	2	1	1
Black	II	90.4	(34.9)	66	40	3	2	1
Walnut Fork	II/III	117.3	(45.3)	71	42	3	3	2
Little Sugar	II/III	117.6	(45.4)	69	47	3	3	1
Lye	III	203.8	(78.7)	82	36.5	3	2	2
Wolf	II	65.8	(25.4)	74	52	4	5	1
Prairie	III	127.9	(49.4)	70	28	3	1	1
			Big Raccoon Creek System					
Mainstem								
Montogomery Co.	III	251.0	(96.9)	80	42[d]	1	3	1
Ramp Crk. to Putnam line	III	365.2	(141)	71	43.1[e]	1.42	1.62	0.82
Tributaries								
Cornstalk	II	52.6	(20.3)	72	41	2	2	1
Haw	II	72.5	(28.0)	73	42	1	3	1
Ramp	III	85.7	(33.1)	62	52	5	1	1
			Big Walnut Creek System					
Mainstem								
Above US 36	IV	357.6	(138)	81	50.2[f]	3.0	1.9	1.2
US 36 to G-castle	IV	575.0	(222)	67	48.5[g]	1.7	2.2	1.5
			Eagle Creek System					
Mainstem - upper	III	74.1	(28.6)	74.4	48	4	5	2
Tributaries								
School Branch	I	22.7	(8.7)	73.6	46	4	3	1
Fishback	II	53.8	(20.8)	65.3	42	5	3	1
Little Eagle	II	75.9	(29.3)	72.4	46	4	3	1
Finley	I	25.2	(9.8)	72.1	48	4	2	0
Mount's Run	II	41.2	(15.9)	59.7	48	4	5	2
			Stotts Creek System					
Mainstem	IV	155.6	(60.1)	58.4	48	3	3	2
North Fork								
lower	III	56.7	(21.9)	55.0	54	5	3	2
upper	II				43	5	3	2

Table 1 concluded.

South Fork							
lower	III	87.3 (33.7)	53.4	50	5	2	0
upper	II			44	2	2	0
Miscellaneous Streams							
Rattlesnake Creek	III	65.2 (25.2)	15?	53[h]	5	4.5	1.5
Stinking Fork	III	70.7 (27.3)	40?	50[i]	3.3	3	1.3

[a] Mean of 7 stations above Darlington.
[b] Mean of 4 stations between Darlington and Crawfordsville.
[c] Mean of 12 stations between Crawfordsville and the mouth.
[d] Mean of 3 stations.
[e] Mean of 8 stations over 8 years from 1981 through 1989.
[f] Mean of 8 stations from 1979 through 1984.
[g] Mean of 8 stations from 1979 through 1987.
[h] Mean of 2 stations.
[i] Mean of 4 stations

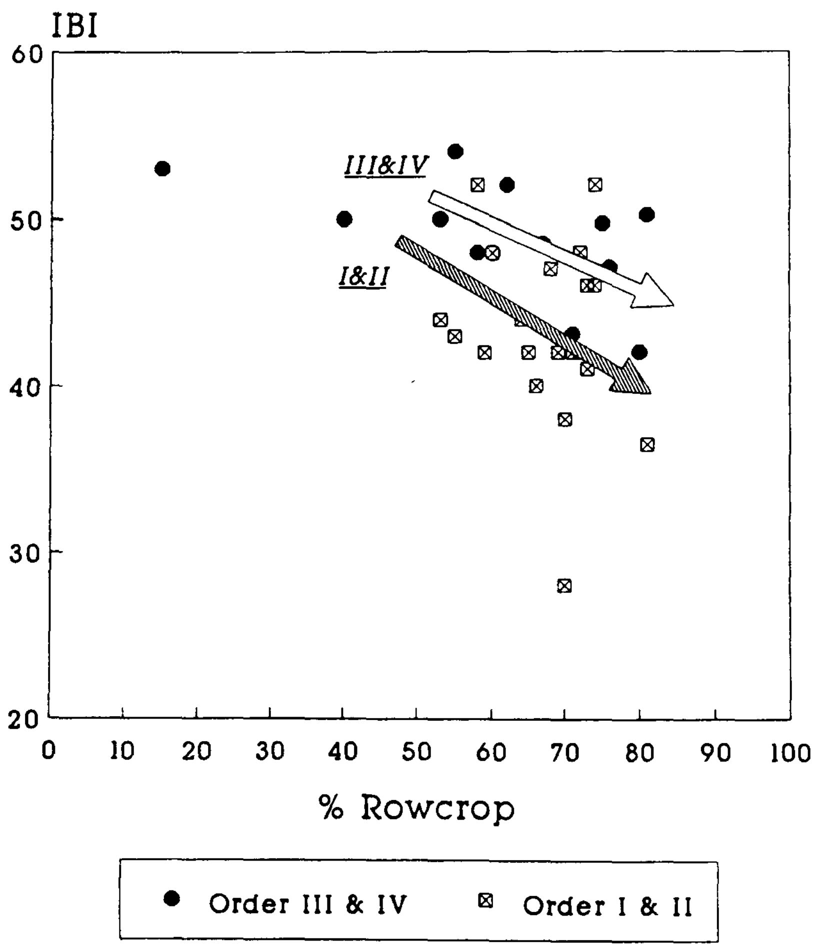

Figure 3: IBI values of fish communities as a function of rowcrop agriculture of the watersheds.

in the watershed of third and fourth order streams and the IBI. The IBI values decline steadily as the percentage of rowcrops increases, although there is much scatter among the data points. The general trends for smaller and larger streams, indicated by arrows, suggest that smaller streams are more strongly affected by progressively greater agricultural development. They also may be negatively influenced at lower rates of development, although watersheds in the 40% to 50% range are lacking.

Four of the stream systems originate in the Tipton Till Plain of Boone County, Indiana and flow in a generally south or southwesterly direction. Within 20 or so miles Sugar, Big Raccoon, and Big Walnut Creeks cut deeply into the plain creating a highly dissected landscape for varying distances. These portions of the watersheds are covered by a mature deciduous forest and are poorly suited for agriculture. The riparian protection thus afforded may be responsible in part for the continued maintenance of reasonably good fish communities in Sugar Creek.

All of the above streams, together with Eagle Creek, once supported healthy populations of smallmouth bass, sunfish, and darters. Sugar Creek still harbors them today (Gammon and Riggs 1983, Gammon et. al. 1990), but Big Walnut Creek and Eagle Creek contain only marginal populations (Benda and Gammon 1965, Fisher and Gammon 1981, 1982).

Big Raccoon Creek and some of its tributaries supported good populations 25 years ago (Gammon 1965), but darters, sunfish, and bass were lost sometime prior to 1981. From 1981 through 1989 three electrofishing collections each at eight stations were made for purposes of biologically monitoring a landfill (Gammon 1990). The landfill has had no measurable effect on the fish community, but agriculture has certainly impacted it.

This data set is interesting because it demonstrates community changes in agricultural watersheds as affected by natural weather and flow patterns.

Table 2 summarizes IBI values for each station and year of study. Mean IBI values for the most downstream station are lower, perhaps because of occasional spring inundation by Mansfield Reservoir downstream. The other stations are remarkably similar to each other, but variability over time is quite striking with mean IBIs lowest in 1981 (IBI = 36.5) and highest in 1988 (IBI = 50.5).

The low IBI values from 1981 through 1984 probably resulted from poor reproduction and survival during unusually high water in the summers of 1979, 1981, and 1982. Darters, sunfish, and bass were virtually absent during those years (Figure 4) and a special seining effort in 1984 aimed at collecting darters also indicated very low population densities. The very high IBI values found in 1988 were associated with extremely low flows and a prolonged drought. Fish were undoubtedly concentrated and more vulnerable to capture.

Over the period of study the IBI values steadily increased, and so did the mean frequency of darters, sunfish, and bass. Figure 5 shows that while the mean IBI increased from less than 35 to more than 50 the mean number of darter, sunfish, and bass species captured per station increased from near zero to more than 2 in a linear fashion.

The weather and regime of stream flow are obviously influential. A succession of years with poor reproduction may decimate species populations which are merely marginal during good years. Conversely, a run of years favoring good reproduction may lead to the appearance of recovery. Generalizations concerning the "health" of a fish population based on investigations conducted during a single year

Table 2: IBI values based on three series of electrofishing collections at 8 stations from Big Raccoon Creek from 1981 to 1989.

Year	F1	F2	F3	F4	F5	F6	F7	F8	Mean
1981	36	38	38	36	38	34	36	36	36.5
1982	40	44	42	40	40	40	38	34	39.8
1984	48	44	42	46	44	40	40	40	43.0
1985	46	50	48	53	50	50	48	44	48.6
1986	40	40	40	38	42	38	40	36	39.3
1987	44	46	50	40	43	44	40	34	42.6
1988	52	48	50	52	50	50	52	52	50.5
1989	42	48	46	44	44	50	42	40	44.5
Mean	43.5	44.8	44.5	43.6	43.9	43.5	42.0	39.0	**43.1**

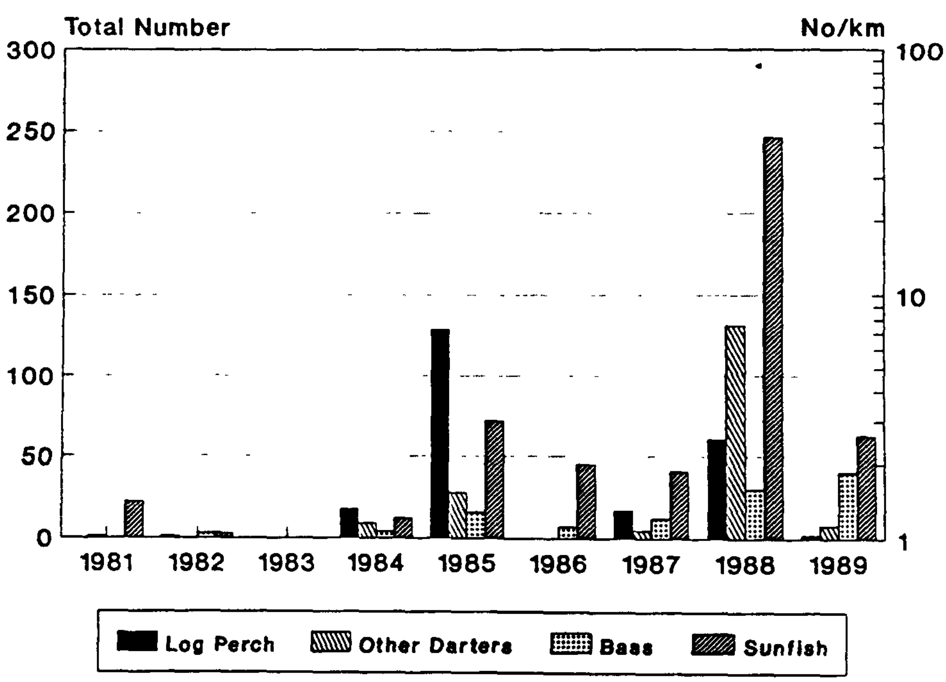

Figure 4: Total numbers of logperch, darters, sunfish, and bass collected in Big Raccoon Creek from 1981 through 1989.

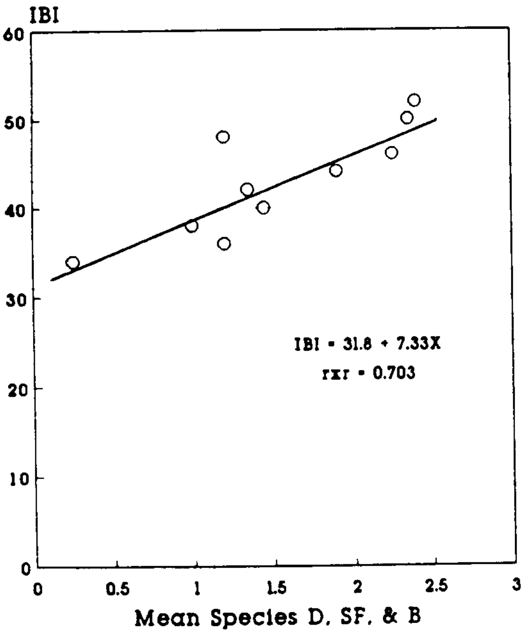

Figure 5: Relationship of catches of darters, sunfish, and bass to IBI values: Big Raccoon Creek 1981 through 1989.

would be unwise and tenuous. Unlike point-source influences which are more sustained and constant, agricultural nonpoint-source pollution is much more sporadic.

Rowcrop intensity has been used as a general measure of agricultural influence. The actual overall pattern of changes in fish communities is obscured and/or influenced by many factors other than agricultural land-use. Sporadic spills of fertilizer and feed lot wastes accelerate the process. Towns and industries may likewise reinforce the process through point-source contributions of wastes.

The pattern of fields relative to streams is no doubt of considerable importance. Streams which are well-protected by riparian vegetation are probably less susceptible to change than streams with tilled fields which extend to the stream banks and heavy lateral erosion. Lower Sugar Creek is strongly degraded by the effects of lateral erosion (Gammon and Riggs 1983), but not during all years (Gammon et. al. 1989).

Other environmental attributes unrelated to agriculture modify and/or influence the overall process in whole stream systems. The drainage pattern of the stream system is probably influential. Systems which are strongly linear with mostly small, low-order tributaries are likely to be more susceptible than more dendritic systems in which one or more less disturbed tributaries may serve as refugia which periodically replenish or restock a degraded mainstem during favorable periods.

Some of the agriculturally degraded tributaries of Sugar Creek appeared to contain better fish communities than they should have, probably because of the presence of good populations of fish in the mainstem and their migration during favorable periods. Upper Big Walnut Creek also contains a fairly good fish community despite being heavily rowcropped. All areas need to be examined for the pattern of permanent vegetation. Extensive agriculture may not be incompatible with good fish communities if adequate protection is afforded by a riparian buffer system.

Agriculture as an influence on streams has not received sufficient attention. There is a great need for programs to assess landuse activities throughout the state by GIS or comparable methodologies. Many of Indiana's streams have already been degraded by agriculture and even the better streams are in danger. We need to eliminate the pasturing of farm animals directly in streams. We need to develop programs for the enhancement of riparian buffers and stabilization of eroding banks.

Acknowledgements

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Indiana, and Heritage Environmental Services made long-term studies possible. The Environmental Protection Agency supported the MIP studies. Other support has come from the Indiana Department of Natural Resources and the Dana Foundation.

Dedication

The oral presentation of this research followed by six hours the birth of Robert Wayne Gammon-Pittman. May he and his entire generation enjoy clean rivers in the future.

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Indiana's NPS Program

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Abstract
Although a variety of existing programs have helped curb non-point source (NPS) water pollution in Indiana, the effects have often been only coincidental to their primary goals. State conformance with Section 319 of the Clean Water Act has recently resulted in development of an integrated, multi-disciplinary NPS control plan that refocuses many programs on the issue of water quality, and has established a number of new initiatives. This effort has been significantly enhanced by national attention on the topic, with shifts toward a water quality emphasis by federal agencies such as the Department of Agriculture. Indiana is now able to address NPS water pollution in a much more unified fashion, guided by a comprehensive plan. Indiana's new Lake Enhancement Program is a part of "T by 2000," a statewide strategy for dealing with soil erosion and sedimentation problems. The goal for lake enhancement is to control the flow of sediment and associated nutrients into public lakes. Toward that goal, Indiana Department of Natural Resources' (IDNR) Division of Soil Conservation is providing technical and financial assistance for lake enhancement needs in accordance with guidelines set by the State Soil Conservation Board, the policy-making body for the division. The Lake Enhancement Program's policies and procedures will be reviewed and specific examples of how the program operates will be discussed.

Key words: Indiana, nonpoint sources, water quality, pollution

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Our environment looks and is cleaner than it was twenty years ago when the first Earth Day was celebrated. That's not to say that there isn't a great deal more to be done....but progress has been made, and that progress has been the result of the cooperative efforts of many federal and state agencies, local governments, and concerned citizens who value our water resources.

Up until now, the majority of our work in protecting water quality has been in cleaning up point source discharges by setting water quality standards, enforcing NPDES permit limitations, and promoting construction and upgrading of wastewater treatment facilities. Indiana is committed to continuing that work on <u>point</u> sources so that all communities and industries discharging to state waters will be in compliance with the Clean Water Act.

Review of surface water quality data for Indiana shows that pollution coming from point sources has declined significantly in twenty years. However, analysis has also shown that <u>nonpoint</u> source pollution continues to degrade water quality and that use impairments are often caused by NPS pollution, either by itself, or in combination with point sources. Studies of Indiana's public lakes reveal that they are particularly vulnerable to certain types of NPS pollution, which currently threaten the designated uses of many of them.

Traditionally, in many respects, NPS water pollution control has been secondary to regulation of point sources in Indiana as well as at the national level. This can be attributed largely to the difficulty and expense involved in identifying and monitoring many of the nonpoint pollutant sources...but it can also be attributed to the pervasive nature of the problems, and tacit acceptance arising from the belief that resolution of the problems was not economically feasible. However, it's become obvious that state water quality goals will never be attained without reduction of NPS pollution.

Indiana state government has sponsored some long-standing programs that have partially addressed certain categories of NPS pollution, such as disposal of waste from confined animal production facilities and control of agricultural erosion, for example, but none of the efforts have been adequate to fully address the problems. In addition, there have been some areas which have received only minimal attention, such as evaluating the effects that storm sewers and atmospheric deposition have on water quality. So, there have been obvious voids, then, in Indiana's overall ability to gauge and control the various NPS pollution, problems that exist in the state.

Section 319 of the Clean Water Act provided the impetus for the state to develop a comprehensive plan which would integrate all aspects of NPS control. In response to requirements of Section 319, a multiagency Task Force was formed which provided a strategy development forum for the state's resource professionals. By bringing a variety of program directors together as a group, a climate of increased cooperation was created in which NPS pollution control could be more thoroughly addressed. The Task Force included representatives of nine different organizations and was responsible for two major accomplishments:

- production of NPS "Assessment Report" which summarized available information regarding NPS-impacted water bodies and the causes of the problems, and;
- development of a NPS "Management Program" describing categorical NPS problems and their proposed solutions.

I should say a few words about the Assessment Report, since during its preparation one of the things we discovered was just how little actual scientific information was available regarding NPS impacts to public waters, and how difficult it was going to be to obtain the information in the future. So, although the rationale for preparing the report was to somehow quantify the extent of NPS pollution in the state, we were only able to assemble the data that were then available, which describe merely a fraction of the state's waters. This has left us in the position of needing to acquire an enormous amount of additional information if we're to be able to truly assess statewide NPS effects, in order to prioritize problem areas for treatment.

We are proceeding slowly in that direction, having begun to develop a variety of biological monitoring programs, since those appear to be the most cost effective and practical methods of evaluating impacts to aquatic systems, but our resources for pursuing such an initiative are limited.

The Management Program itself is based on five premises which must be supported in order for the program to be successful:

(1) financial assistance must be made available to fund recommended activities;

(2) activities involving different organizations must be well coordinated;

(3) research and monitoring must be continued which will provide information on water quality trends that will guide future program needs;

(4) information and education efforts must be an integral component of the overall program; and,

(5) in addition to financial and technical assistance, regulatory alternatives must also be considered

for the resolution of some types of problems.

It will take a great deal of money to implement all of the Management Program's recommendations, and Indiana has recently received assistance in this regard through EPA's granting of Section 319 funds to the state. Let me briefly highlight some of the work that we will be doing with the money.

Portions of the money will be used to finance projects demonstrating the elimination of acid runoff from abandoned mine land and reduction of erosion from a commercial timber harvesting operation.

Part of the money will be used to fund a NPS evaluation and prioritization project in an industrialized urban watershed.

A state university will use some of the grant to develop computer software that can be distributed to local health departments, enabling them to evaluate the adequacy of proposals for on-site disposal systems.

Another university will be paid to evaluate the effects of BMP implementation on particular lake watersheds.

And there will be a number of other uses for the money, with the most interesting being a survey of the Eel River to determine how NPS pollution is affecting the aquatic biota.

The continued support and involvement of federal, state, and local governments in the control of NPS pollution is essential. We're hopeful that current efforts, in combination with our Manage-ment Program's actions and federal assistance through Section 319, will eventually allow our streams and lakes to regain their former vitality.

Instream Water Quality Evaluation of the Upper Illinois River Basin Using the Index of Biotic Integrity

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Abstract
The twelve stations sampled within the Upper Illinois River drainage revealed that the best water quality as indicated by the Index of Biotic Integrity (IBI) was found in the Kankakee, Fox, DuPage, Des Plaines and Chicago River sub-basins, respectively. These areas were correlated with degree of dominant land use, e.g. agriculture and sparse residential areas in the Kankakee and Fox drainages and heavy urban and industrial in the Des Plaines, Chicago, and DuPage drainages. Principal concerns within each of these basins indicate that bank erosion from the lack of a stable riparian zone, combined sewer overflow and street runoff and point sources of pollution contribute greatly to the lower water quality observed.

The stations sampled in each drainage varied in number, however, the overall objective was to provide a quantitative approach to categorizing the biological integrity of the sub-basins on a long-term basis. The IBI was able to rapidly estimate water quality and provide interpretation of water quality without the long-term exercise of measuring water chemistry on a weekly or monthly basis. Yet the amount of similarity based on the two data sets are comparative since the biological fauna assimilates all past and present conditions.

Keywords: IBI, fish, Kankakee River, Des Plaines River, Upper Illinois River Basin, Chicago River and Canals, Fox River

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Introduction

The Upper Illinois River has a rich history of biological information dating back to the late 1800's (Mills et al. 1966; Steffeck and Striegl 1988). Information regarding fish distribution and documenting impacts incurred from once through cooling of industries and municipalities has added greatly to the body of data from this region with over 200 published papers and reports. The greatest handicap one has in interpreting this body of information lies in its relevance to the Upper Illinois River as it exists today. Collections on the River have been conducted for a variety of reasons, including but not limited to: species-specific population estimation, general distribution, length-weight ponderal indices, and fisheries management strategy development. The evaluation of water quality within the River has been one of immense concern, however, the implications of such varied collection techniques and study objectives has practically made the historic data base uninterpretable.

As part of the National Water Quality Assessments (NAWQA) Pilot Survey the Environmental Science Division's Central Regional Laboratory (CRL) of the Environmental Protection Agency (USEPA), Region V surveyed twelve stations in the basin to evaluate instream biological quality of the Upper Illinois River. Karr's index of biotic integrity (1986) was used to evaluate water quality based on fish communities.

Fish sampling was conducted at twelve stations within the Upper Illinois River basin which has the mainstem initiate at the junction of the Des Plaines and Kankakee Rivers, Will

County, Illinois and for this study terminated just below the junction with the Fox River at Ottawa, Illinois.

Sampling in the Upper Illinois River basin began during late July 1989 and was completed by late August. Water conditions were stable and close to normal conditions following drought conditions observed during summer 1988.

Study Area

The Upper Illinois River is considered a seventh order tributary in the vicinity of Ottawa, Illinois (IEPA 1988). The River is comprised of five sub-basins comprising the Kankakee, Chicago, Des Plaines, DuPage, and Fox Rivers. The general flow of the River is from northeast to southwest, with the most northerly sub-basin originating in Waukesha County, Wisconsin (Fox River) and the most easterly sub-basin in St. Joseph County, Indiana (Kankakee River). The River is primarily contained within the Central Cornbelt Plains ecoregion with a minor portion of its headwaters occurring in the Southeastern Wisconsin Till Plains ecoregion (Omernik 1987). The dominant land use in both ecoregions is cropland, however, soil constituents and the urban area of Chicago are the major differences. Low to moderate flows occur within the Rivers.

The study area borders the shores of Lake Michigan and is a primary drainage of the upper Mississippi River. The River has a series of navigation impoundments on the mainstem which has made the River more homogeneous, turning it into a series of pools. Each of the sub-basins has a series of low-head dams or flood control dams. The Kankakee River has a single dam on its entire length, and the entire stretch within Indiana has been ditched. The Chicago River, which previously was considered a Lake Michigan drainage tributary, was included in the current study because of its connection with the Upper Illinois River through the Sanitary and Ship Canal.

Station Locations

A total of twelve stations were sampled from July 26 to August 24, 1989 (Fig. 1). All stations occur in the State of Illinois unless otherwise noted. The furthest station downstream in the Upper Illinois River basin was the Illinois River at Marsailles (station 1), LaSalle Co., downstream of the dam from Central Illinois Power (R.M. 246.4) to Delbridge Creek (R.M. 245.5), T 33N R 4E S 15/16.

The first sub-basin was the Fox River which included three stations, one mainstem and two tributaries. Station 2, was the Fox River near the Rt. 62 bridge, Algonquin, Algonquin Township, downstream of the dam influence at a crossover walk bridge but upstream of the Algonquin STP (T 43N R 8E S 30). Station 3 was at Indian Creek, LaSalle Co, 11 mi N Ottawa, Freedom Township, at E 1553 and E 16th Street bridge intersection (T 35N R 3E S 1/2). Station 4 was at Honey Creek, Walworth Co., WI, Himelbaugh Road bridge, 7 mi N Burlington (T 4N R 18E S 25).

The second sub-basin was the DuPage River which included two stations. Station 5 was the DuPage River, Will Co., 1 mi N Shorewood, downstream of Black Road bridge at Hammel Woods DuPage River access (T 35N R 9E S 10). Station 6, was the East Branch of the DuPage River, Will Co., off Royce Road, 2-1/2 mi NW Bolingbrook, 1/2 mi E of Naperville Road intersection, (T 37N R 10E S 5/8).

Sub-basin 3 was the Des Plaines River which included two mainstem stations and a tributary station. Station 7, Des Plaines River at Brandon Road Lock and Dam, Will Co., was 3 mi S Joliet and sampled in a backwater area on the east side of the Navigation channel, (T 5N R 10E S 20). Station 8, Des Plaines River at Riverside, Cook Co., was accessed at a Cook County Forest

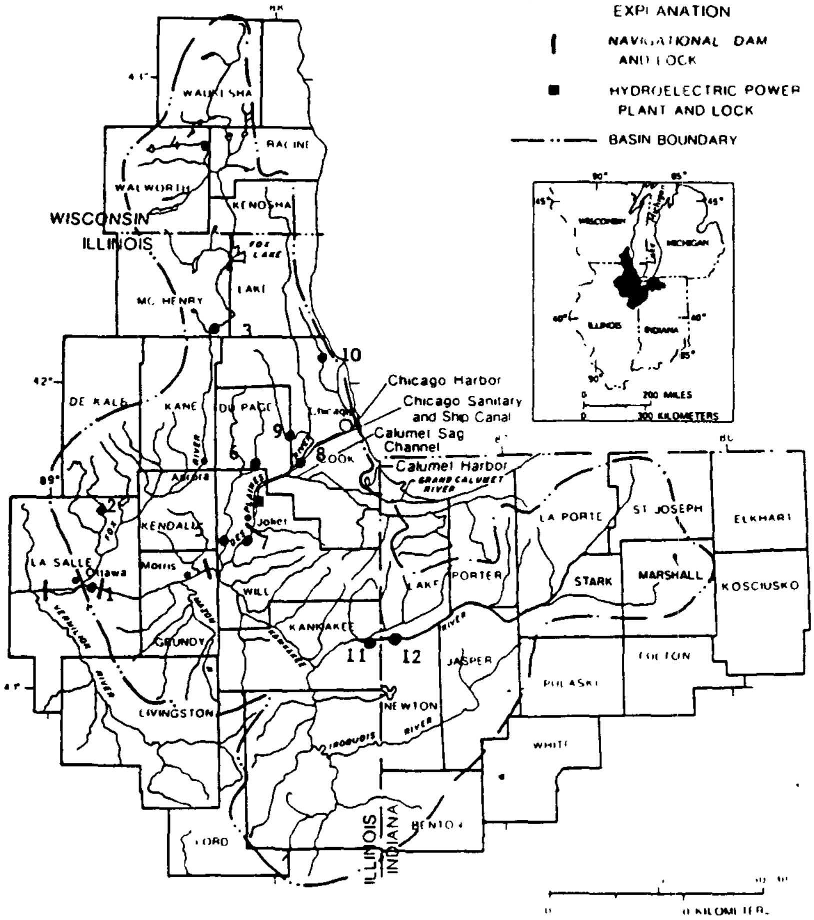

Fig. 1. Station locations for fish collected in the Upper Illinois River basin during 1989.

Preserve off 40th Street (T 39N R 12E S 36). Station 8 was downstream of the confluence of Salt Creek. Station 9, Salt Creek, Cook Co., at Beamis Woods footpath 1/2 mi N Western Springs off Wolf Road and Ogden Road (Rt. 34) (T 39N R 11E S 31).

Sub-basin 4 included the Chicago River basin and the canal system. A single station was sampled in this basin. Station 10, North Branch Chicago River, Cook Co., at Touhy Avenue bridge, 1.5 mi S Niles (T 42N R 12E S 15).

Sub-basin 5 was the Kankakee River basin which included two mainstem locations. Station 11, was the Kankakee River at Momence, Kankakee Co., off E 1050N, 1 mi from Rt. 114 bridge, T 31N R 13E, S 22/23. Station 12, Kankakee River, Newton Co., Indiana, In Rt. 55 bridge, 1 mi S Shelby, Eagle Creek Township, T 32N R 8W S 33/34.

Materials and Methods

Fish Sampling

The sampling protocols for fish follows that documented in the USEPA, Environmental Science Division's, Central Regional Laboratories Standard Operating Procedure for Rapid Assessment using fish (1988).

The following collection techniques were applied to obtain a representative sample from each of twelve stations within the Upper Illinois River basin. All habitats that were present were sampled including riffle, pool, and run. No samples were taken in the vicinity of bridges, or in the mouths of tributaries entering large rivers, lakes or reservoirs since they tend to be more similar to larger-order habitats than the one under consideration (Fausch et al. 1984).

Seines were considered by Karr et al. (1986) the best collection tool for obtaining an unbiased sample in small streams. As stream complexity increased a 50 ft. bag seine with 1/8 in mesh was utilized for collection and a boat mounted pulsed DC electroshocker was selectively included at appropriate sites. The seine was able to get an excellent representation of the species present, since low water levels allowed easy access for most portions of the Rivers. Likewise, adult species which only use the stream in a transitory manner would be excluded from this analysis (e.g. adult salmonids, eels). Young-of-the-year species less than 20 mm total length were excluded following the recommendations of Angermeier and Karr (1986). Distances between 100 to 500 m were sampled at each site and included similar levels of effort (usually 1 hr of intensive sampling per 100 m) within all available habitats.

Each sampling period consisted of a single site visit under normal to moderate flow conditions. During field collection, all larger specimens were identified to species, smaller specimens of minnows and darters were preserved in 10% formalin, and returned to the laboratory. At the completion of the study, voucher specimens were deposited into the fish collection repository at the Field Museum of Natural History.

The ambient environmental data was evaluated using the Index of Biotic Integrity (IBI; Karr et al. 1986). The IBI relies on multiparameters based on community concepts, to evaluate a complex system. It incorporates professional judgement in a systematic and sound manner, but sets quantitative criteria that enables determination of what is poor and excellent based on species richness and composition, trophic constituents, and fish abundance and condition. The twelve IBI metrics reflect insights from several perspectives and cumulatively are responsive to changes of relatively small magnitude, as well as broad ranges of environmental degradation (Table 1).

Since the metrics are differentially sensitive to various perturbations (e.g. siltation or toxic chemicals), as well as to various levels within the range of integrity, conditions at a site can be determined with considerable accuracy. The interpretation of IBI numerical scoring is provided in six narrative categories that have been tested in Region V (Karr 1981; Table 2).

Several of the metrics are drainage size dependent and require selection of numerical scores. The ecoregion approach developed by USEPA-Corvallis, OR was utilized to compare least impacted zones within the region (Omernik 1986). Extensive work within the Central Corn Belt Plain ecoregion by Illinois EPA (1988) and documentation in Karr et al. (1986), were used to determine "excellent" or control conditions for scoring the metrics based on stream sizes equivalent to the various sub-basins in the Upper Illinois River basin (Table 3).

Habitat Evaluation

A habitat quality evaluation assessment was completed in conjunction with fish collection. The QHEI, quality habitat evaluation index, developed by Ohio EPA (1986) provides numerical assignments for six criteria to assess riffles and pools. The criteria were modified to include only five of Ohio's criteria and were adjusted to reflect the same equivalent total score. Scoring was based on 100 total points and incorporates substrate quality, instream cover, channel morphology, riparian zone and bank erosion, and pool and riffle quality based on drainage area.

For station comparisons of the fish samples to be considered valid, the

TABLE 1. Scoring criteria for 12 IBI metrics for low to moderate gradient streams within the Northeastern Region of Illinois for the Upper Illinois River basin (Karr et al. 1986).

Metrics	Scores 1	3	5
1. Number of total species	Stream size dependent		
2. Number of darter species	Stream size dependent		
3. Number of sucker species	Stream size dependent		
4. Number of sunfish species (excluding Micropterus)	Stream size dependent		
5. Number of intolerant species	Stream size dependent		
6. Proportion of individuals as Green Sunfish	>20%	5-20%	< 5%
7. Proportion of individuals as omnivores	>45%	20-45%	<20%
8. Proportion of individuals as insectivorous minnows and darters	<20%	20-45%	>45%
9. Proportion of individuals as piscivores	<1%	1-5%	>5%
10. Catch rate (number/100 m)	Varies with gear and stream size		
11. Proportion of individuals with poor condition or disease	>5%	>2-5%	<2%
12. Proportion of individuals as hybrids	>5%	1-5%	<1%

stations must be capable of supporting the same type of communities. A stream section habitat evaluation was used to determine if all sample sites had similar habitat types for comparisons.

In order for a station to be comparable habitat scores from the QHEI had to be within 90% to be comparable and at least 75% to be supportive (Plafkin et al. 1989).

Results

Quality Habitat Evaluation Index

Flow, bank erosion, and warmer water temperatures varied the habitat within the Upper Illinois River basin between the various sub-basins (Table 4). Habitat criteria was developed for each site based on the quality of the site for promoting biological diversity. The highest QHEI score during the current study was 89.3. Comparing all other scores to this value resulted in seven stations being equal in available habitat, two stations being comparable, and three stations not meeting the 75% criteria. The station with the best overall habitat score was Honey Creek (Fox River sub-basin) station 4. The Upper Illinois River at Marsailles

Table 2. Biotic integrity classes used in assessing fish communities along with general descriptions of their attributes (Karr et al. 1986).

Class	Attributes	IBI Range
Excellent	Comparable to the best situations without influence of man; all regionally expected species for the habitat and stream size, including the most intolerant forms, are present with full array of age and sex classes; balanced trophic structures.	58-60
Good	Species richness somewhat below expectation, especially due to loss of most intolerant forms; some species with less than optimal abundances or size distribution; trophic structure shows some sign of stress.	48-52
Fair	Signs of additional deterioration include fewer intolerant forms, more skewed trophic structure (e.g., increasing frequency of omnivores); older age classes of top predators may be rare.	39-44
Poor	Dominated by omnivores, pollution-tolerant forms, and habitat generalists; few top omnivores; growth rates and condition factors commonly depressed; hybrids and diseased fish often present.	28-35
Very Poor	Few fish present, mostly introduced or tolerant forms; hybrids common; disease, parasites, fin damage, and other anomalies regular.	12-22
No Fish	Repetitive sampling fails to turn up any fish.	

Table 3. Metric scores for Illinois Northeast region surface waters of various stream orders for calculating the Index of Biotic Integrity (criteria shown is for the score of 3, values greater than that listed receive a 5 and lower a 1; IEPA 1988).

Metric	Stream Order					
	2	3	4	5	6	7
1. Total number of species	6-10	7-12	8-14	9-16	10-18	11-20
2. Number of Darter species	2	2-3	2-3	3-4	3-5	3-5
3. Number of Sunfish species	1	2	2-3	2-3	3-4	3-4
4. Number of Sucker species	2	2-3	2-3	3-4	3-5	3-5
5. Number of Intolerant species	2-3	2-3	3-4	3-5	3-5	4-6

(station 1), Fox River at Algonquin (station 2), DuPage River at Shorewood (station 5), East Branch DuPage River near Bolingbrook (station 6), the Des Plaines River at Brandon Road (station 7), and Des Plaines River at Riverside (station 8) were habitat equal, while Indian Creek (station 3), and Kankakee River at Momence (station 11) were habitat compatible. Habitat limited were Salt Creek at Beamis Woods (station 9), North Branch Chicago River (station 10), and Kankakee River at Shelby, IN (station 12). The primary causes of habitat degradation was channelization, siltation and embeddedness.

Fish - Index of Biotic Integrity

Illinois River-Mainstem : River conditions at Marsailles indicated "fair to poor" conditions from the upstream headwater drainage (Table 5). Sampling techniques used at this station consisted of 50 ft bag-seining, and electrofishing for 500 m of river reach. Habitat sampled included 70% riffle, 20% run and 10% pool.

Poor metric scores contributing to reduced station scoring included number of darter, sunfish, and sucker species, number of intolerant species, and number of individuals in the sample. Excellent scores were achieved for proportion of green sunfish, proportion of omnivores and carnivores, number of hybrids, and disease factor.

Dominant taxa within the site included emerald shiner (77.96%), gizzard shad (9.82%), and freshwater drum (3.61%). Intolerant taxa included three taxa. A stable level of carnivores were found in the drainage including smallmouth bass, flathead catfish, channel catfish, and white bass. Bullhead minnow, flathead catfish and white bass were collected exclusively at this station. Larval specimens of emerald shiner and gizzard shad were abundant along the margins of the River.

Fox River Basin: The Fox River basin obtained the highest IBI score among all Upper Illinois River sampling with a score of 52 at Indian Creek. A rating of "good" at Indian Creek was similar to the high score in the Kankakee basin. The Fox River at Algonquin rated "fair" and Honey Creek a "poor". Sample distances collected at the Fox River at Algonquin, Indian Creek and Honey Creek were 250 m, 100 m, and 150 m, respectively. Habitat sampled in the Fox River consisted of 70% run, and 15% each of pool and riffle. The primary collection technique was a 50 ft bag seine and a 10 ft common minnow seine. Indian Creek sampling consisted of common minnow seining within habitat composed of 30% each of pool and riffle, and 40% run. Honey Creek was likewise seined using a 10 ft common minnow seine within habitat composed of 15% pool, 40% riffle and 45% run.

The mainstem Fox River was classified "fair" due to low scores for number of darter, sucker, and intolerant species; and proportion of carnivores (Table 6). Sampling downstream of the walk bridge but upstream of the Algonquin STP resulted in high scores for total number of taxa, number of sunfish species, proportion of green sunfish, proportion of omnivores and insectivores, number of individuals in the sample, lack of hybrids, and disease. Indian Creek scored very high in most categories except number of sunfish species, while Honey Creek scored poorly in number of darter and sunfish taxa, proportion of carnivores, number of individuals and diseased individuals. A high proportion of individuals had black spot indicating environmental stress at Honey Creek.

Taxa unique to the Fox River basin included yellow bass at Fox River at Algonquin, and rainbow and fantail darters at Indian Creek. The downstream pool and riffle habitat had several intolerant taxa including two taxa at the Fox River proper, eight taxa at Indian Creek, and two taxa at Honey Creek. The most dominant taxa at

Table 4. Quality Habitat Evaluation Index scores for twelve stations sampled in the Upper Illinois River basin, during 1989.

Character	Illinois River	Fox River	Indian Creek	Honey Creek	DuPage River	E. Branch DuPage River
predominate substrate	boulder/ sand	sand/ cobble	sand/ gravel	sand/ cobble/ gravel	sand	sand/ gravel
Silt covered						
Area affected	none	none	none	none	none	none
Instream Cover						
Relative %	extensive	sparse	sparse	sparse-moderate	sparse	sparse
Channel Morphology						
sinuosity	none	moderate	high	high	moderate	moderate
development	good	fair	fair	good	fair	fair
channelization	recovered	none	none	none	none	none
stability	moderate	high	low	high	moderate	moderate
Riparian Zone						
Zone width	moderate	narrow	very narrow	wide	wide-extensive	extensive-narrow
Quality	forest	residen-tial	open pasture	forest	park/ forest	forest/ park
Bank erosion	little	little	moderate	little	moderate	little
QHEI score	83.1	89.0	74.3	89.3	88.1	87.0

Table 4. (continued)

Character	Des Plaines Brandon	Des Plaines Riverside	Salt Creek	N. Branch Chicago R.	Kankakee Momence	Kankakee Shelby
predominate substrate	muck/ bedrock	sand/ bedrock	sand	sand	sand/ gravel	sand
Silt covered						
Area affected	pools	none	none	none	none	none
Instream Cover						
Relative %	moderate	moderate	sparse	sparse-moderate	moderate	sparse
Channel Morphology						
sinuosity	low	moderate	low	none	low	low
development	good	good	fair	good	fair	fair
channelization	none	recovered	recovered	recovered	none	recovered
stability	low	low	moderate	moderate	high	moderate
Riparian Zone						
Zone width	narrow	wide	moderate	moderate/ narrow	very narrow	narrow
Quality	forest/ old field	forest	forest	forest	residen-tial	residen-tial
Bank erosion	moderate	heavy	little	little	none	moderate
QHEI score	81.5	83.7	59.2	57.6	74.5	61.7

Table 5. Fish collected, length range, number and relative percent composition from the Illinois River mainstem collected during July and August,1989.

Species	Illinois River at Marsailles N	%	Range mm
Gizzard shad	49	9.8	100-128
Quillback	1	0.2	300
Carp	2	0.4	345-688
Emerald shiner	389	77.9	59-110
Spottail shiner	1	0.2	100
Sand shiner	2	0.4	52-54
Spotfin shiner	10	2.0	36-51
Golden shiner	1	0.2	100
Bluntnose minnow	4	0.8	45-52
Bullhead minnow	1	0.2	38
Flathead catfish	1	0.2	650
Channel catfish	4	0.8	350-475
White bass	5	1.0	200-300
Bluegill	6	1.2	68-250
Smallmouth bass	5	1.0	300-385
Freshwater drum	18	3.6	375-500
IBI score	36		

Algonquin were spotfin shiner (63.1%), brook silverside (23.94%), and orangespotted sunfish (4.87%). Taxa dominant at Indian Creek included spotfin shiner (34.14%), sand shiner (21.72%), and common shiner (10.0%), while at Honey Creek dominant taxa included sand shiner (39.13%), spotfin shiner (31.3%), and golden shiner (18.26%). Unique taxa collected in the Fox River included yellow bass.

Larval fish collected or observed on the Fox River at Algonquin downstream of the walk bridge along run habitat included cyprinids and centrarchids. Downstream along the pool margins green sunfish and brook silverside were collected. Few larval fish were collected from Indian Creek with those collected being sand shiners. No larval fishes were collected from Honey Creek.

DuPage River Basin: Two stations were sampled in the DuPage River basin. The furthest downstream station, DuPage River proper at Shorewood (station 5), was seined for 200 m of stream reach and included 45% run, 45% riffle and 10% pool habitat. The East Branch of the DuPage River (station 6), was seined using a 10 ft common minnow seine for 100 m. The habitat sampled included 40% each of run and riffle, and 20% pool. A disjunct collection was obtained at this location with the majority of sampling being conducted in the run and riffle habitat along the margins of the park. Additional sampling was conducted upstream of the primary site, in the tree line around the bend in the River.

The mainstem DuPage River scored an IBI rating of "fair to good", while the East Branch station rated "poor" (Table 7). Contributions to reduced metric scores at Shorewood were low numbers of darter species (metric 2) and catch per unit effort (metric 10). High scores were observed for total number of species, proportion of green sunfish, proportion of omnivores, insectivores, and carnivores, lack of hybrids and diseased individuals. The East Branch of the DuPage River had reduced scores because of the lack of benthic and intolerant taxa (metrics 2, 4, and 5), reduced catch per unit effort (metric 10), and high proportion of diseased individuals (mostly black spot). High scores were observed for proportion of green sunfish, proportion of insectivorous cyprinids, and no hybrids.

Salt Creek. At Brandon Road, 60% riffle, 25% pool, and 15% run habitat was sampled for 300 m of stream reach. Riverside sampling consisted of 55% riffle, 25% run and 20% pool habitat being sampled for 400 m. This location consisted of a disjunct collection with 300 m sampled in the primary location (the long run and riffle) and 100 m sampled upstream in the River bend. Salt Creek was sampled for 100 m with 100% of the habitat consisting of run habitat.

Table 6. Fish collected, length range (range measured in mm), number and relative percent composition from the Fox River sub-basin of the Upper Illinois River collected during July and August, 1989.

	Location								
	Fox River at Algonquin			Indian Creek			Honey Creek		
Species	N	%	Range	N	%	Range	N	%	Range
Northern pike	1	0.1	875						
Carp	2	0.1	300-575						
Common stoneroller				2	0.7	49-92			
Emerald shiner	1	0.1	69						
Rosyface shiner				10	3.5	40-67			
River shiner				1	0.3	91			
Bigmouth shiner							21	18.3	32-71
Sand shiner	1	0.1	30	63	21.7	30-70	45	39.1	30-66
Mimic shiner				21	7.3	52-65			
Spottail shiner	1	0.1	38						
Spotfin shiner	1310	63.1	30-84	99	34.1	28-80	36	31.3	45-74
Common shiner				29	10.0	94-104			
Golden shiner	32	1.5	48-144				1	0.9	65
Bluntnose minnow	1	0.1	32	10	3.5	35-70	8	7.0	24-65
Fathead minnow	12	0.6	28-60						
Suckermouth minnow				2	0.7	49-55			
Hornyhead chub				2	0.7	40-100			
Creek chub				1	0.3	66			
White sucker							1	0.9	61
Quillback	10	0.5	59-95	2	0.7	60-66			
Smallmouth buffalo	9	0.4	69-114						
Northern hogsucker				4	1.4	55-65			
Silver redhorse							1	0.9	54
Brook silverside	497	23.9	21-77	3	1.0	44-49			
Yellow bass	7	0.3	64-275						
Black bullhead	5	0.3	200-275						
Largemouth bass	1	0.1	44						
Smallmouth bass				26	9.0	45-180			
Bluegill	71	3.4	20-83						
Green sunfish	3	0.1	22-27	7	2.4	30-114			
Pumpkinseed	1	0.1	100						
Orangespotted sunfish	101	4.9	22-82						
White crappie	6	0.3	50-72						
Black crappie	4	0.2	51-70						
Johnny darter				1	0.3	49	2	1.7	44-61
Rainbow darter				2	0.7	35-43			
Fantail darter				1	0.3	25			
Banded darter				4	1.4	47-56			
IBI score	44			52			32		

Table 7. Fish collected, length range (range measured in mm), number and relative percent composition from the DuPage River sub-basin of the Upper Illinois River collected during July and August, 1989.

	Location					
	DuPage River at Shorewood			East Branch DuPage River		
Species	N	%	Range	N	%	Range
Gizzard shad	3	1.1	69-95	19	25.3	47-85
Carp	1	0.4	350			
Common stoneroller	1	0.4	50	4	5.3	51-119
Bigmouth shiner				6	8.0	58-64
Sand shiner	12	4.5	25-69	6	8.0	57-66
Spotfin shiner	184	68.7	40-58	29	38.7	49-94
Common shiner	2	0.8	46-55	1	1.3	102
Golden shiner	1	0.4	55			
Bluntnose minnow	19	7.1	25-69	3	4.0	56-73
Creek chub				1	1.3	134
Quillback	2	0.8	250-325			
Smallmouth buffalo	1	0.4	195			
Blackstripe topminnow	3	1.1	30-67			
Largemouth bass	3	1.1	55-138	1	1.3	82
Smallmouth bass	12	4.5	45-325			
Bluegill				4	5.3	53-81
Green sunfish	1	0.4	95	1	1.3	127
Longear sunfish	12	4.5	64-89			
Orangespotted sunfish	3	1.1	38-66			
IBI score	46			32		

Dominant taxa at the DuPage River at Shorewood included spotfin shiner (68.66%), bluntnose minnow (7.09%), and equal dominance of sand shiner, smallmouth bass, and longear sunfish (4.48%). Dominant taxa on the East Branch included spotfin shiner (38.67%), Gizzard shad (25.33%), and equal numbers of sand and bigmouth shiners (8.0%). Four intolerant taxa were collected at Shorewood and one on the East Branch.

Few larval taxa were observed in the DuPage basin. Spotfin shiner larvae were the only taxa observed and only at the DuPage River at Shorewood.

Des Plaines River Basin: Three stations were sampled in the Des Plaines River basin, including the Des Plaines River at Brandon Road (station 7), Des Plaines River at Riverside (station 8), and Salt Creek (station 9). Seining and electrofishing techniques were used at Brandon Road, while only seining was conducted at Des Plaines River at Riverside and IBI scores at Brandon Road and Riverside rated "poor" with equivalent scores of 34 (Table 8). At Salt Creek a score of 30 rated the site "poor" (Table 8). Reduced metric scores at Brandon Road were observed for five metrics with low scores for number of darter and sucker species, number of intolerant taxa, proportion of omnivores, and reduced catch per unit effort. Low scores at Riverside were likewise a result of five metrics, including numbers of darters, suckers and intolerant species, proportion of carnivores, and reduced catch per unit effort. Six metrics scored poorly for Salt Creek with low scores for total number of species, number of darter, sucker, and intolerant taxa, proportion of carnivores, and catch per unit effort.

Dominant taxa at Brandon Road include bluntnose minnow (39.74%), emerald shiner (25.64%), and carp (10.26%). At Riverside dominant taxa included sand shiners (44.16%), bluntnose minnow (19.05%) and spotfin shiner (14.72%). Salt Creek was dominated by spotfin shiner (50.0%), bluntnose minnow (28.57%), and green sunfish (10.71%). Intolerant taxa included two taxa at Brandon Road, and a single taxa at Riverside and Salt Creek. Unique taxa collected in the Des Plaines River included mosquitofish at Riverside.

Many of the fish collected during the collections in the Des Plaines River were young of the year specimens. An abundance of tolerant taxa, i.e. green sunfish, bluntnose minnow, and fathead minnow, indicated degraded conditions for most of this basin from the urban and industrial areas of Chicago.

Chicago River and Canal Sub-Basin:
A single station was sampled in the Chicago River basin. The North Branch of the Chicago River at Touhy Avenue (station 10) was sampled for 100 m and consisted of entirely run habitat.

This station, although rigorously sampled, did not produce any fish. Several crayfish and a large snapping turtle were collected and released. This station scored the poorest of all 1989 Upper Illinois River stations with a score of zero (no fish).

Kankakee River Basin: Two stations were sampled in the Kanakakee River basin, including the Kankakee River at Momence (station 11) and Kankakee River at Shelby, Indiana (station 12). Sampling at Momence consisted entirely of seining for 300 m and included 50% each of run and riffle habitat. At Shelby, channelization of the Kankakee River resulted in both seining and electrofishing methods needing to be conducted. Over 500 m of reach was sampled by electrofishing and 50 m was sampled seining. The habitat within this reach consisted of 65% run and 35% pool.

The Kankakee River basin consistently scored the highest of all sub-basins collected during 1989. A rating of "good" was observed at Momence with a score of 52, and a rating of "fair" at Shelby with a score of 44 (Table 9). The only low score at Momence was for catch per unit effort, while at Shelby low scores were given for number of darter species, proportion of carnivores, and catch per unit effort.

Dominant taxa at Momence included spotfin shiner (41.04%), sand shiner (18.66%), and orangespotted sunfish (13.43%). At Shelby dominant taxa included spotfin shiner (87.45%), sand shiner (4.73%), and carp (2.47%). The number of intolerant taxa at Momence was seven taxa and five taxa at Shelby. At Momence unique taxa collected included spotted sucker, blackside darter, and mimic shiner.

Discussion

Water quality characterization of twelve stations within the Upper Illinois River basin provided expected results based on known water chemistry, areas of dominant land use, habitat and known point source dischargers (Fig. 2).

Increased biological integrity, as it relates to water quality, was observed from an upstream to downstream direction for the all of the various sub-basins. Reasons for these trends in index of biotic integrity rating depended on a variety of factors. In the Fox River sub-basin, at two similar sized streams, water quality was considered "good" at Indian Creek and "poor" at Honey Creek. Habitat quality was the reverse, with Honey Creek consisting of superior riparian zone and habitat cycles, while Indian Creek was in the center of a cow pasture. The Fox River proper was sampled only at a single site and was

Table 8. Fish collected, length range (range measured in mm), number and relative percent composition from the Des Plaines River sub-basin Upper Illinois River collected during July and August, 1989.

	Location								
	Des Plaines River Brandon Road			Des Plaines River Riverside			Salt Creek		
Species	N	%	Range	N	%	Range	N	%	Range
Northern pike	1	1.3	250						
Grass pickerel	1	1.3	134						
Gizzard shad				22	9.5	59-105			
Carp	8	10.3	300-500	3	1.3	372-750			
Goldfish	1	1.3	35						
Emerald shiner	20	25.6	53-76						
Spottail shiner				4	1.7	41-60			
Bigmouth shiner				3	1.3	39-46			
Sand shiner				102	44.2	22-65	1	3.6	33
Spotfin shiner	1	1.3	42	34	14.7	22-65	14	50.0	35-58
Bluntnose minnow	31	39.7	23-38	44	19.1	21-60	8	28.6	25-43
Fathead minnow				12	5.2	22-32	1	3.6	24
Smallmouth buffalo	1	1.3	300						
Blackstripe topminnow	5	6.4	65-72						
Mosquitofish				3	1.3	26-37			
Tadpole madtom	1	1.3	34						
Largemouth bass									
Bluegill	6	7.7	35-61	3	1.3	22-66	1	3.6	22
Green sunfish				1	0.4	84	3	10.7	65-125
White crappie	1	1.3	115						
Black crappie	1	1.3	75						
IBI score	34			34			30		

intermediate in quality between the upper and lower tributary segments. A rating of "fair" was scored because of a lack of benthic species (e.g. darters and suckers), however, a number of catfish were collected including various bullheads. Reasons for a decline at Algonquin were due to the uniformity of habitat (e.g. mostly run), and the lack of riparian buffer zone along the mostly residential shoreline. Input of nutrients from septic systems and runoff of fertilizers have probably contributed to degradation along this stretch of the River. Upstream of the site was a dam and downstream was the Algonquin STP. Both of these may act as barriers to recolonization of fish species from upstream and downstream refugia. The lack of darters was surprising since suitable riffle habitat was present at this site.

The DuPage River sub-basin, indicated that the East Branch of the River was "poor" and probably a result of upstream perturbations. The East Branch has undergone a series of building projects in many of the towns which line the River upstream. The lack of a substantial fish population at this station is indicative of areas with organic enrichment. The preponderance of green sunfish and the increase of black spot disease affecting individuals of the insectivorous trophic guild (e.g.

Table 9. Fish collected, length range (range measured in mm), number and relative percent composition from the Kankakee River sub-basin of the Upper Illinois River collected during July and August, 1989.

Species	Kankakee River at Momence N	Kankakee River at Momence %	Kankakee River at Momence Range	Kankakee River at Shelby N	Kankakee River at Shelby %	Kankakee River at Shelby Range
Grass pickerel	2	0.8	119-192			
Carp				12	2.5	432-628
Common stoneroller				1	0.2	51
Rosyface shiner				1	0.2	68
Sand shiner	50	18.7	37-68	23	4.7	40-65
Mimic shiner	4	1.5	52-76			
Bigmouth shiner				4	0.8	36-65
Spotfin shiner	110	41.0	36-70	425	87.5	35-85
Common shiner	20	7.5	94-176	3	0.6	40-52
Bluntnose minnow	2	0.8	54-63			
Creek chub				2	0.4	42-44
Hornyhead chub	1	0.4	82			
Northern hogsucker	1	0.4	212	4	0.8	125-250
Shorthead redhorse	1	0.4	143	3	0.6	152-314
Spotted sucker	5	1.9	130-167			
Brook silverside	7	2.6	33-49	3	0.6	32-37
Blackstripe topminnow	3	1.1	55-59			
Rock bass	12	4.5	70-93	1	0.2	151
Largemouth bass				2	0.4	115-135
Bluegill	3	1.1	80-131			
Green sunfish				1	0.2	91
Longear sunfish	5	1.9	78-101			
Orangespotted sunfish	36	13.4	46-78			
Black crappie	1	0.4	115	1	0.2	103
Johnny darter	1	0.4	39			
Banded darter	3	1.1	45-50			
Blackside darter	1	0.4	86			
IBI score	52			44		

spotfin shiner) are usually a result of fertilizer runoff and muck or soft substrates. The lack of a riparian zone probably has contributed greatly to this problem. The downstream location at Shorewood, on the mainstem DuPage River had an IBI rating of "fair to good". This particular station had a high proportion of smallmouth bass and insectivorous cyprinids which usually indicate increased water quality. A variety of sunfishes and other species typical of good pool habitat were present, as well as, specimens of herbivores and other trophic guilds. This particular station has potential for increased water quality scores in future sampling events.

The Des Plaines River sub-basin was rated the second poorest of the Upper Illinois River sub-basins. The River typically scored "poor" with downstream areas scoring higher than upstream locations. Brandon Road Lock and Dam was a surprise since the majority of the backwater habitat possessed an abundance of aquatic macrophytes and soft muck sediments. A high proportion of intolerant taxa were present including a number of game species, e.g. black and white crappie, northern pike, and bluegill. Tadpole madtom and smallmouth buffalo, both benthic species indicated that conditions were improving over historic conditions at this location. A nice firm riffle over a bedrock substrate may provide adequate habitat for Moxostoma species and other larger river species. Although redhorses were not collected during the present collection, further sampling will probably indicate their presence. The site at Riverside was equally impressive with a nice riffle/run along the margin of the Cook County Forest Preserve. A high amount of erosion does affect this site, since it too possessed few benthic species. The presence of both bigmouth and sand shiners, and a high proportion of insectivorous cyprinids indicates that it has potential to improve in water quality. Salt Creek the furthest upstream location sampled in the Des Plaines sub-basin had limited habitat diversity. Species collected at the station were expected for the site, due to the lack of true pool and the limited, basically non-existent, riffle habitat. Species using these types of habitats would be excluded from the site. Darters, suckers and madtoms were not present since they require clear clean riffles, and centrarchids and black basses were absent because there was no pool habitat. Further downstream pool habitat was present but was considerably disjunct from the location sampled. The major contributor of poor conditions has to be traced to the high degree of urbanization surrounding the basin. Sewer overflows, point source dischargers, flow fluctuations, and salt runoff from street cleaning contribute greatly to the inability of this basin to achieve its potential. Habitat within the basin is adequate to support more species than what currently occurs.

The Chicago River and its supporting canal system have been modified so that their flow is away from Lake Michigan and into the Upper Illinois River. This was done to protect the City of Chicago's water supply from contamination and send waste products down the Illinois River. Better treatment processes and increased water quality regulation have reduced the amount of waste going to the system, thereby, having a dramatic effect on the Upper Illinois River mainstem. However, the station at Touhy Avenue on the North Branch Chicago River indicates that other urban affects have reduced the biological integrity of the water. Straightening of the River channel by channelization, runoff of road salts used for winter, and severe bank erosion have all but eliminated use of the River for aquatic life. Although the presence of luxuriant aquatic macrophyte beds, debris piles, and firm substrates should attract fish species, no fish were collected after extensive sampling at Touhy Avenue.

Overall, the Kankakee River sub-basin possessed the best water quality among Upper Illinois River sub-basins. Channelization of the River within Indiana has improved flow rates and increased flushing rates into Illinois. Water quality at Momence was observed to possess a "good" rating with an abundance of intolerant species, including: three species of darters, rock bass, various redhorse and sucker species, and cyprinids. The River at Momence has a shallow wide topography with some islands and vegetation along the stream margins. A nice selection of habitat diversity occurs within the area, however, no pool habitat was located during collection. Additional predators, such as northern pike, smallmouth and largemouth bass would have been located if these pools existed further improving the location score. At Shelby, the Kankakee River although still of good water quality had a reduction in IBI rating because of the effect of channelization. The lack of shallow shore margins, the preponderance of sand substrate, and lack of heterogenous habitat has precluded many fish species from using the area. The lack of darter species and catfish were probably the most noticeable absent species.

Overall scoring indicated that the Kankakee River, followed by the Fox River, were the two best sub-basins in the Upper Illinois River System, while the DuPage, Des Plaines, and Chicago Rivers were respectively the next best. This presents an interesting comparison since the primary land use within the Kanakakee and Fox River sub-basins include agricultural and less-concentrated residential uses, while the DuPage, Des Plaines, and Chicago River sub-basins are heavily populated urban and industrial areas. A distinct difference between the non-continuous non-point source of diffuse pollution compared to constant discrete point source input suggests that the water quality in these five sub-basins suffer from upstream inputs from the City of Chicago and industrial suburbs. The riparian buffer zone and amount of allochthonous input shows that a increase in degradation is apparent as one gets closer to the metropolitan areas of Chicago. Increases in bank stabilization, improvement in combined sewer overflow and road runoff, and other non-point source influences will greatly improve the resiliency of these Rivers.

Similar results were observed between streams of equal size, third order streams, e.g. Honey Creek, East Branch DuPage River, and Salt Creek, were all considered "poor" by IBI standards. However, Indian Creek and the North Branch Chicago River were outliers representing the best and worst case scenarios for the same order streams.

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Appendix A. Fish metrics used to score specimens collected from the Sub-basins of the Upper Illinois River during July and August, 1989.

Species	Native	Endangered	Tolerance	Feeding Guild	Habitat Preference S	C	R
Gizzard shad	N			Omni	X		X
Alewife	I						X
Skipjack herring	N			Carn	X		X
Northern pike	N			Carn	X		X
Grass pickerel	N			Carn	X	X	
Carp	I		Tolerant	Omni	X	X	X
Goldfish	I		Tolerant	Omni		X	X
Common stoneroller	N			Herb	X	X	
Rosyface shiner	N		Intolerant	Insect	X	X	
Emerald shiner	N			Insect	X		X
River shiner	N			Insect			X
Mimic shiner	N		Intolerant	Omni	X	X	X
Sand shiner	N			Insect	X	X	
Bigmouth shiner	N			Omni	X	X	
Spottail shiner	N			Insect	X		X
Spotfin shiner	N		Intolerant	Insect	X	X	X
Common shiner	N			Insect	X	X	
Golden shiner	N			Omni	X		X
Bluntnose minnow	N			Omni	X	X	X
Fathead minnow	N			Omni	X	X	X
Bullhead minnow	N		Intolerant	Omni	X		X
Suckermouth minnow	N			Insect	X		X
Creek chub	N			Insect	X	X	X
Hornyhead chub	N			Insect	X	X	X
White sucker	N				X	X	X
Shorthead redhorse	N		Intolerant		X		X
Silver redhorse	N		Intolerant		X		X
Quillback	N			Omni	X		X
Smallmouth buffalo	N				X		X
Spotted sucker	N						X
Northern hogsucker	N		Intolerant		X	X	X
Brook silverside	N				X	X	X
Blackstripe topminnow	N				X	X	X
Flathead catfish	N			Carn			X
Channel catfish	N			Carn			X
Yellow bullhead	N				X		X
Black bullhead	N				X		X
Tadpole madtom	N				X	X	
Stonecat	N		Intolerant		X	X	X
White bass	N			Carn			X
Yellow bass	N			Carn			X
Rock bass	N		Intolerant	Carn	X	X	X
Largemouth bass	N			Carn	X	X	X
Smallmouth bass	N		Intolerant	Carn	X	X	X
Bluegill	N				X		X
Green sunfish	N		Tolerant		X	X	X

Appendix A (continued)

Longear sunfish	N	Intolerant		X		X
Pumpkinseed	N			X		X
Orangespotted sunfish	N			X		X
White crappie	N		Carn	X		X
Black crappie	N		Carn	X		X
Yellow perch	N		Carn	X		X
Johnny darter	N			X	X	X
Rainbow darter	N	Intolerant		X	X	
Fantail darter	N	Intolerant		X	X	
Banded darter	N	Intolerant		X	X	
Blackside darter	N			X	X	X
Logperch	N			X	X	X
Slenderhead darter	N	Intolerant		X		X
Freshwater drum	N			X		X
Mottled sculpin	N	Intolerant		X	X	

S - Streams and smaller rivers
C - Creeks and brooks
R - Larger rivers
Omni - Omnivore
Insect - Insectivorous cyprinid
Carn - Carnivore