CHAPTER 5

ECOLOGICAL INTEGRITY

Introduction

This chapter describes the ecological or biological integrity of investigated watersheds. Ecological integrity can be equated with chemical, physical, and biotic integrity. The natural patterns of river-riparian ecosystems are disrupted by a variety of stresses. As described in Chapter 2, Karr et al. (1986) defined five major classes of environmental factors that affect aquatic biota. Stresses on the biotic components of river and stream ecosystems arise from (1) challenges in the quality, quantity, and seasonal availability of food for organisms, (2) deterioration of water quality, (3) modifications of the habitat including the substrate, (4) water quantity or flow regime, and (5) biotic integrations. An evaluation of habitat quality is critical to any assessment of ecological integrity. Habitat is the principal determinant of biological potential and can be used as a general predictor of the biological condition. The habitat assessment has been included in ecological evaluation of water bodies.

The majority of water resources programs are oriented toward detection and monitoring of chemical contamination (pollutants) and are deficient in detecting other forms of perturbation causing pollution. There are various indices used to assess the effects of stressors on (aquatic) populations and communities. Biotic indices are generally specific to the type of pollution or the geographical area; they are used to classify the degree of pollution by determining the tolerance of an indicator organism to a pollutant. Indicator species are assigned scores for their tolerance level. Multi metric indices take advantage of several indicator groups rather than individual species.

Ecological indices have been analyzed to describe the overall state of biological integrity. Fish and macroinvertebrate communities have been sampled as well as habitat. Additional data have been acquired from Wisconsin Department of Natural Resources (DNR). The index of biotic integrity (IBI) has been calculated for both fish and macroinvertebrate samples. The DNR data have been used to analyze relationship between urbanization and biotic integrity.

Ecological Integrity Assessment Methodology

Fish IBI

Fish samples were collected at representative sites throughout the Menomonee River and Oak Creek watersheds in Southeastern Wisconsin during June and July 1999 and 2000. Fish were sampled by electroshocking, thereby allowing inspection, species identification and enumeration without significant mortality (Figure 5.1). Collections were made at three sites in 1999 and seven sites during 2000. A stream shocker comprised of a gas-powered generator towed in a small boat with two hand-held electrodes attached to long electrical cords allowed operators access to all stream habitats. A control box mounted on the boat allowed regulation of voltage and amperage under varying water conductivity conditions. Maintaining 4-6 amps and 150 -200 volts provided sufficient current to stun all sizes of fish yet allow quick recovery. Stunned fish were placed in a live well in the stream shocking boat and processed at the end of the station or at intermediate points if necessary. Each fish was examined for external abnormalities, measured (total length) and returned to the stream. Fish that could not be easily identified were preserved and taken to the lab for identification using an identification key (Eddy and Underhill, 1978).

Figure 5.1. Fish IBI sampling on the Oak Creek by the Wisconsin Lutheran College and Marquette University team

The length of each sampling station was 35 times the mean channel width at normal flow in order to provide a sufficient representation of that section of the stream (Lyons, 1992). Fish collections at each site were used to calculate a variety of metrics based on overall species composition and abundance of tolerant and intolerant species. These metrics were combined to provide an index of biotic integrity (Lyons, 1992) for each station.

The Wisconsin version of IBI consists of 10 basic metrics plus 2 additional metrics (correction factors) that affect the index only when they have extreme values. These metrics are:

SPECIES RICHNESS AND COMPOSITION

- total number of native species
- number of darter species
- number of sucker species
- number of sunfish species
- number of intolerant species
- percent (by number of individuals) that are tolerant species

TROPHIC AND REPRODUCTIVE FUNCTION

- percent omnivores
- percent insectivores
- percent top carnivores
- percent simple lithophilous spawners

FISH ABUNDANCE AND CONDITION

- number of individuals (excluding tolerant) per 300 m sampled
- percent with deformities, eroded fins, lesions, or tumors

Macroinvertebrate IBI

Macroinvertebrate communities were sampled in a variety of habitats at nine sites on the Menomonee River and five sites on Oak Creek during 1999-2000. Subsequent identification and enumeration of macroinvertebrates allowed calculation of metrics that were used to develop an index of biotic integrity for each station. Samples were collected with D-nets and direct removal of organisms with forceps from in-stream structures such as large rocks, tree limbs and an occasional tire. Organisms were picked from the D-net samples on site and preserved in 70% isopropyl alcohol. Sampling was continued until either 100 organisms were collected or one hour of effort with at least two individuals had been expended. Samples were taken from riffle, pool and run sections of the stream at those stations where these habitats existed. Samples were taken near the stream bank as well as throughout the channel. Each available substrate type was sampled at each station. Identification keys (Merritt and Cummins, 1996; Eddy and Hodson, 1961) were used in the lab to assist with identification to appropriate taxonomic category (Michigan DNR, 1991).

In this study a reference site approach for macroinvertebrate communities (Michigan DNR 1991) was used along with a macroinvertebrate Family Biotic Index (FBI) developed by Hilsenhoff (1988). The FBI is essentially weighted average of tolerance values assigned to individual organisms (species). The macroinvertebrate community analysis through the use of within watershed reference sites and the family biotic index allowed evaluation of changes along an urbanization gradient within the watershed.

The Multimetric Index of Biotic Integrity (MIBI) is comprised of 9 metrics:

- total number of taxa (TAXTOTAL)
- total number of mayfly taxa (TAXE)
- total number of caddisfly taxa (TAXT)
- total number of stonefly taxa (TAXP)
- percent mayfly (PERE)
- percent caddisfly (PERT)
- percent contribution of dominant taxon (PERDOM)
- percent isopods/snails/leeches (PERISOP)
- percent surface dependent (PERSURF)

Habitat Assessment

The habitat evaluation followed the protocol described in Rapid Bioassessment Protocol (Plafkin et al., 1989). The evaluation begins with the riparian zone (stream bank and drainage area) and continues through instream to sediment/substrate descriptions. Various habitat parameters are weighted to emphasize the most biologically significant parameters. This methodology was recently revised (Barbour et al., 1997) to include additional assessment parameters for high gradient streams and a more appropriate parameter set for low gradient streams.

Habitat parameters pertinent to the assessment of habitat quality are separated into three principal categories. Primary parameters characterize the stream micro-scale habitat and have the greatest direct influence on the structure of the indigenous communities. The secondary parameters measure the macro-scale habitat such as channel morphology characteristics. Tertiary parameters evaluate riparian and bank structure. The number shows the range of scores assigned to the parameter.

PRIMARY- SUBSTRATE AND INSTREAM COVER

- Bottom substrate and available cover 0-20

•	Embeddedness	0-20
•	Flow/velocity	0-20

SECONDARY - CHANNEL MORPHOLOGY

•	Channel alternation	0-15
•	Bottom scouring and deposition	0-15
•	Pool/riffle, run/bend ratio	0-15

TERTIARY - RIPARIAN AND BANK STRUCTURE

•	Bank stability	0-10
•	Bank vegetation	0-10
•	Streamside cover	0-10

A total score is obtained for each biological station and compared to a site-specific control or regional reference station. The ratio between the score for the station of interest and the score for the control or regional reference provides a percent comparability measure for each station.

Morphological Changes by Urbanization

Morphological changes of streams by urbanization are as important as the changes in chemistry. Channels of natural streams are in an equilibrium with the flow. Leopold, Wolman and Miller (1994) documented that channels of rivers in eastern and Midwestern US have a channel capacity that can contain a flow (a bankfull stage) that has an approximate recurrence interval of about 1 ½ years as shown on Figure 5.2. This figure is essentially a channel rating curve normalized by the bankfull capacity flow and bankfull depth obtained from USGS and other observations in the eastern US. In their observations the authors noted that the recurrence interval of bankfull flows ranged from one to 1.9 years. Leopold, Wolman and Miller noted that "there is a remarkable similarity in the frequency of bankfull stage on a variety of rivers in diverse physiographic settings and differing greatly in size".

Analyses of the data from Western humid regions of the US (Oregon and Washington) by Williams (1978) more or less confirmed this finding but the author pointed out a large scatter of the recurrence intervals around the mean of 1 ½ years (80 percent of intervals were within the range from about 1.0 to 3 years with a log - mean of 1 ½ years). Consequently, floods, defined as a flow exceeding the bankfull flow occurs in natural channels on average once in two years.

Urban communities, in the past and today, have tried to restrict the enlargement of eroding channels by armoring and lining the banks or even the entire channel. In other places, channel straightening, diking and enlarging were implemented. In extreme, cases, urban streams were diverted into enlarged closed conduits and put away from the sight. While these practices may slow down or eliminated bank erosion and reduce flooding these practices not provide conditions for habitat preservation and propagation of aquatic biota.

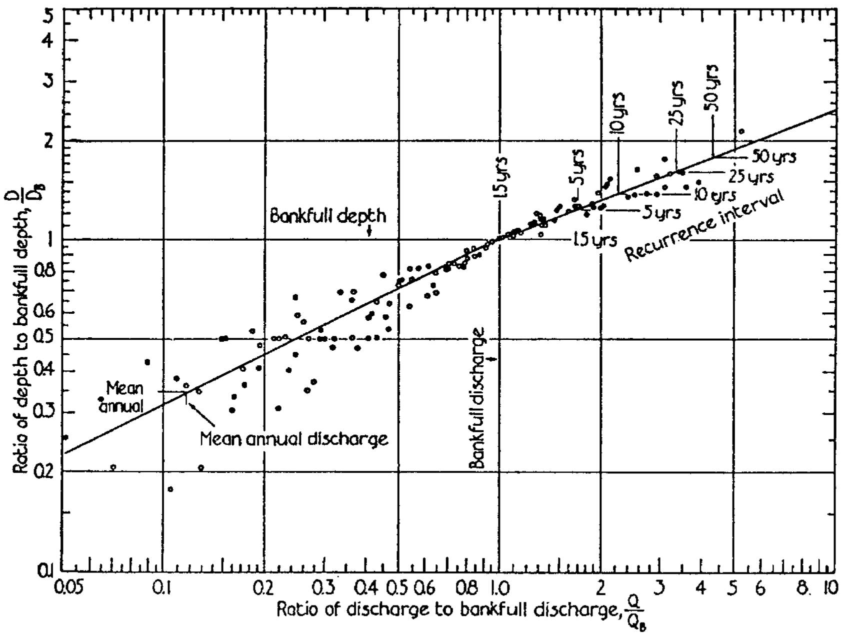

Figure 5.2. Relationship of flows to the recurrence interval. Flow and depths are normalized by the bankfull flows and depths (Leopold, Wolman and Miller, 1994).

Booth and Jackson (1997) analyzed channels stability in the Seattle (WA) metropolitan area east and south east of lake Washington. The watersheds ranged from 38 km^2 to 181 km^2 in area (14.7 to 70 sq mi). They noted that channels remain relatively stable with good habitat conditions when the degree of imperviousness of the watershed is less than 10 percent (Figure 5.3). When, as a result of urbanization, the impervious portion of the watershed exceeds the 10 percent threshold the channel become unstable and enlarging by accelerated bank erosion. They correlated the channel stability to a 10 year flow in a pre-development forested watershed and degree of imperviousness. Apparently, the forested watersheds of the Pacific Northwest are in an equilibrium with the 10-year flow while watersheds situated in eastern and Midwestern US and urban basins that lost their forests in the North-west are more in equilibrium with the 1½ to 2 year flows. Combined with the hydrological changes of increased flush runoff and diminishing base flow, urban stream have significantly larger dimensions than their rural counterparts. Because of diminished infiltration and shallow aquifer recharge, urban streams have either very little flow between events or base flow is supplied by infiltration into storm sewers.

Unmodified streams respond to increased flows and frequency of floods due to urbanization by enlarging their cross-section until a new equilibrium is reached in the future. For urbanized watersheds, Leopold (1968) stated that the channels tend to enlarge proportionally to the increase in the 1½ year flow. If the 1½ year flow increases by a factor of four, the width of the channel is approximately doubled. The transitional period may last decades and, during this transitional period, the channel banks become unstable and eroding. The eroded materials from the banks and from the watershed may also silt the bottom of the streams. Consequently, fish and aquatic life habitat is diminished. Unstable banks and eroding channels do not provide good habitat conditions for aquatic biota. Fish need the banks for shelter; eroded stream bank material silt the river bottom and increase turbidity.

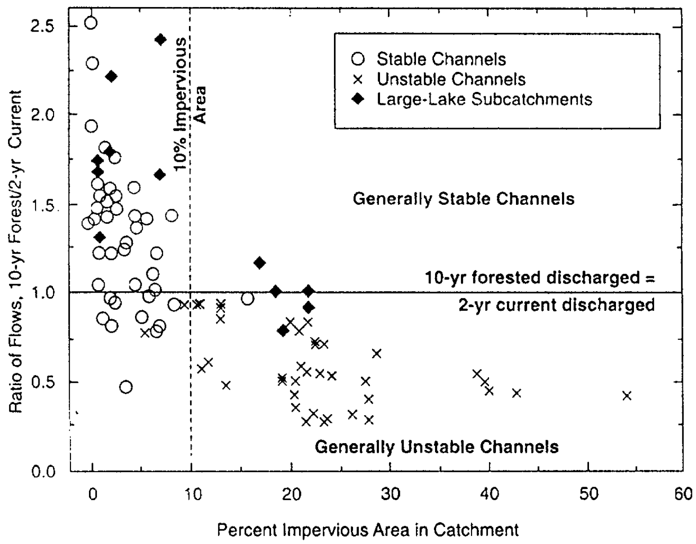

Figure 5.3. Channel stability as a function of imperviousness and flow increase in the Pacific North-west US (from Booth and Jackson, 1997)

However, degradation of habitat by stream bank erosion and hydrologic and hydraulic changes of urban stream are only a part of the causes of the overall degradation of the integrity of urban streams. Wang et al. (1997, 2000, 2001) correlated the Index of Biotic Integrity (Karr et al., 1986; Plafkin et al. 1989) to the percent of urban land cover and found that watersheds in Wisconsin with more than 20 percent urban land use (more than 10 percent impervious) had IBI ranking as poor to very poor although their habitat scores may have ranged from very poor to good. There appeared to be a threshold between 10 and 20 percent urban land use across which IBI scores dramatically declined. One could make an obvious conclusion that both physical degradation of habitat and toxic effects of pollutants in water and sediments may lead to declining IBI scores of urban streams.

Results and Discussion

Data Description

Marquette University Sampling

A combined total of 1,562 fish were examined in the Menomonee River and Oak Creek watersheds during the course of this study. These collections included 20 species of fish ranging from bottom feeding carp to insectivorous brown trout. Ten species were collected in Oak Creek and twenty species were collected in the Menomonee River watershed. The most abundant species was the mud minnow (*Umbra limi*). However, this species only occurred at the Freistadt Road site in the upper Menomonee River. The most commonly occurring species were creek chub (*Semotilus atromaculatus*), white sucker (*Catostomus commersoni*) and green sunfish (*Lepomis cyanellus*). These species occurred at every sampling site in both watersheds. Blacknose dace (*Rhinichthys atratulus*), a species often associated with riffles, was only found in the central and lower sections of the Menomonee River. Two species of darters, fantail (*Etheostoma flabellare*) and

Johnny (*Etheostoma nigrum*) along with northern pike (*Esox lucius*) were only found in the central and upper portions of the Menomonee River watershed.

Slightly more than 4,000 macroinvertebrates comprised of 34 different taxonomic families were collected in the Menomonee River and Oak Creek during June and July, 1999 and 2000. The Menomonee River watershed produced 30 families in 1999 and 26 families in 2000. Aquatic sow bugs (23%) and common net spinner caddisflies (22%) dominated Menomonee River samples. Small minnow mayflies (9%), scuds (8%) and midge larvae (7%) also comprised a large part of the Menomonee River samples. Oak Creek collections revealed 25 families in 1999 and 20 families in 2000. Midge larvae (23%) and aquatic sow bugs (20%) were the most abundant families collected in Oak Creek. Left-handed snails (*Physa*, 13%) and scuds (12%) were also major contributors to the samples. Aquatic sow bugs and midge larvae occurred at nearly all sampling sites in both the Menomonee River and in Oak Creek during 1999 and 2000. Although caddisfly larvae were very abundant, they only occurred in sites on the Menomonee River that had a rocky substrate. Both the midge larvae and left-handed snail, *Physa*, were far more abundant in Oak Creek than the Menomonee River.

DNR Dataset

The historic data on macroinvertebrate community have been acquired from Wisconsin DNR. The DNR master data set includes all macroinvertebrate evaluations completed in Wisconsin from 1978 to present. These data were posted and maintained by the University of Wisconsin - Stevens Point on their website (Aquatic Entomology Web, 2000). All data are in dBase IV format and can be manipulated by any spreadsheet or database program. Field and laboratory methods for these samples follow Wisconsin DNR protocols that are based on Hilsenhoff (1977, 1982, and 1987).

Only subsets of all samples available from DNR were used in analysis. The study region was defined as southeastern Wisconsin and comprised of the following counties: Kenosha, Milwaukee, Ozaukee, Racine, Walworth, Washington, and Waukesha. Geomorphologic, soil, hydrological and ecological conditions are very similar among selected counties. All counties belong to Southeastern Wisconsin Till Plains ecoregion. The study area is part of two watersheds, the Lake Michigan and Mississippi River watersheds, a factor which may affect the analysis as every watershed may have its own specific biota. However, one may argue that macroinvertebrate community would be less affected than fish community would. Adult insects of most species can overcome relatively small distance between monitored streams.

Variables recorded by DNR can be categorized as descriptive, hydraulic, locational, substrate, habitat/riparian quality, and water physical properties. Sample was taken in one to five replicate samples. Hydraulic, substrate and water physical properties were then recorded as an average for the sampling site as well as separate value for each replicate sample. Variables describing habitat and riparian quality such as water quality, siltation, presence of septic, are categorical variables with three possible values. The value shows whether the presence of a feature is significant, insignificant, or a feature is not present.

Index of Biotic Integrity

Fish collections in the Menomonee river and Oak Creek watersheds resulted in very poor biotic integrity ratings at all but one station when compared to a composite of warm-water streams in Wisconsin. The more upstream (closer to head waters) sampling locations had somewhat better biotic integrity ratings than the sites further down stream (Figure 5.4). Comparisons between 1999 and 2000 showed similar species composition sampled each station. The majority of fish species sampled are considered tolerant to habitat and water quality degradation and exhibit omnivorous feeding behavior.

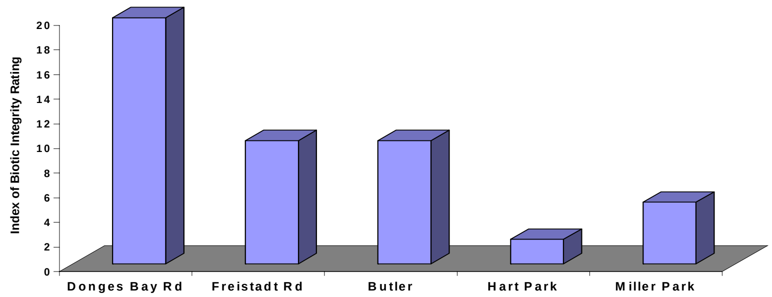

Figure 5.4. Fish IBIs in Menomonee River from headwaters to lower waters.

Multimetric indices of biotic integrity based on the Donges Bay Road reference site in the upper Menomonee River watershed indicated nearly all sampling locations were in the good to fair quality categories. Unlike the pattern seen for fish indices, the macroinvertebrate indices did not indicate better quality in the upper portions of either river system (Figures 5.5 and 5.6. The pattern seen for macroinvertebrate indices seemed to reflect habitat conditions at individual sampling locations rather than a continuum of degradation as the river or creek moved into more urbanized areas. The details of evaluation can be found in Technical Memorandum #2 (Anderson, 2001).

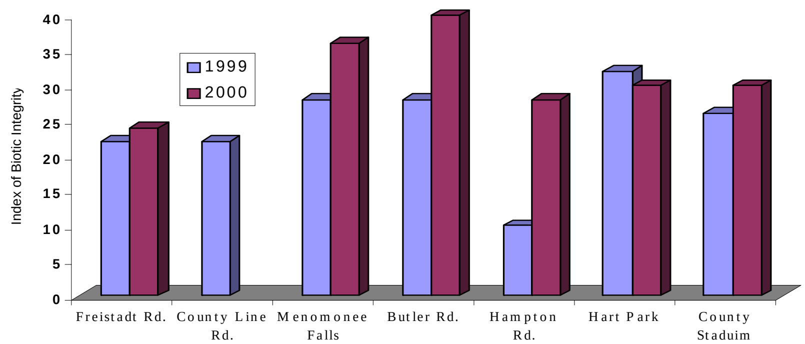

Figure 5.5. Macroinvertebrate IBIs in Menomonee River from headwaters to lower waters.

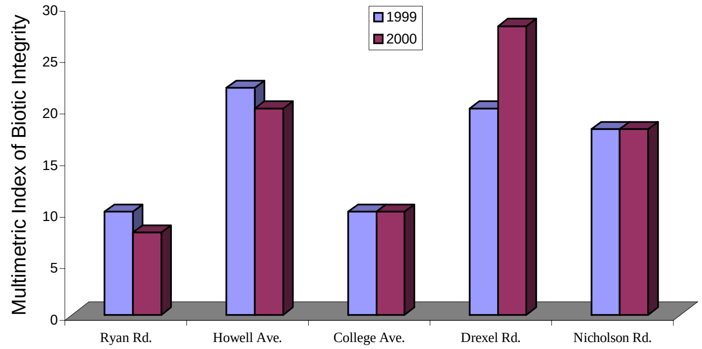

Figure 5.6. Macroinvertebrate IBIs in Oak Creek from headwaters to lower waters.

Effect of Urbanization on IBI

The DNR dataset has been used to evaluate the effect of urbanization on macroinvertebrate IBI. The data were collected over a period of years (1978-2000) together with basic description of habitat. ArcView GIS has been used to estimate corresponding urbanization from a map of historic urban growth.

The dataset has been separated into two sets based on urbanization: below and above 20%. This level of urbanization corresponds to about 10% imperviousness. Figure 5.7 shows differences in distribution of MIBI for urbanization below and above 20%. The MIBI values are normally distributed for low urbanization sites. Low MIBI values are significantly more frequent in sites affected by urbanization.

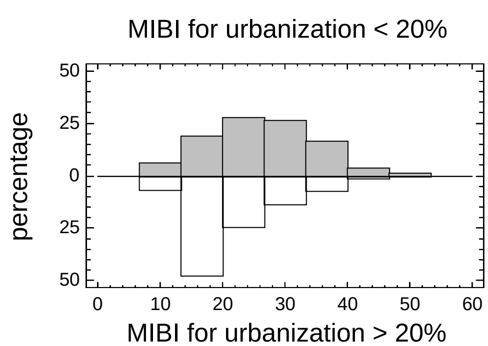

Figure 5.7. Comparison of MIBI distributions for urbanization below and above 20%.

Effect of urbanization on MIBI metrics is shown in Table 5.1. Almost all metrics show statistically significant difference in means (medians) for watersheds urbanized more than 20% and less than 20%. Slope of linear regression is also significantly different from zero, although the actual R^2_{adj} is rather small. The only exception is percent of surface dependent, which can be related to unusually small number of non zero values.

Correlation between urbanization and individual MIBI metrics is rather weak. However, there is a significant change in distribution of these values as urbanization increases. Figures 5.8, 5.9, and 5.10 shows scatter plots of total numbers of mayfly, caddisfly, and stonefly taxa against urbanization of watersheds. There is a clear distinction between values for watersheds affected by different levels of urbanization for these taxa. Number of particular taxa found on a site is also influenced by other local factors as well as natural variations.

Table 5.1. Effect of urbanization on MIBI metrics.

Variable	**Scale**	**Mean P-value**	**Variance P-value**	**Median P-value**	R^2_{adj} **[%]**	**Slope**	**Slope P-value**
Total number of taxa	Arit.	0.035	0.004	0.076	0.8	-2.914	0.016
	Log.				0.6	-0.096	0.030
Number of mayfly taxa	Arit.	0.000	0.000	0.000	5.5	-1.310	0.000
Number of caddisfly taxa	Arit.	0.000	0.000	0.000	2.5	-0.386	0.000
Number of stonefly taxa	Arit.	0.000	0.000	0.000	8.3	-1.910	0.000
Percent of mayflies	Arit.	0.003	0.000	0.000	2.2	-0.114	0.000
	Log.	0.006	0.256	0.008	1.7	-0.526	0.000
Percent of caddisflies	Arit.	0.000	0.000	0.000	2.5	-0.147	0.000
	Sq.rt.	0.000	0.101	0.000	4.1	-0.240	0.000
Percent of dominant taxon	Arit.	0.027	0.660	0.020	1.3	0.102	0.003
Percent snails, leeches, isopods	Arit.	0.000	0.000	0.000	4.4	0.225	0.000
	Log.	0.000	0.343	0.000	5.0	0.672	0.000
Percent surface dependent	Arit.	0.407		0.404	0.0	10^{-4}	0.671

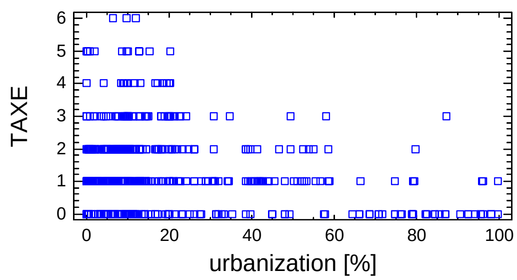

Figure 5.8. Changes in number of mayfly taxa with urbanization.

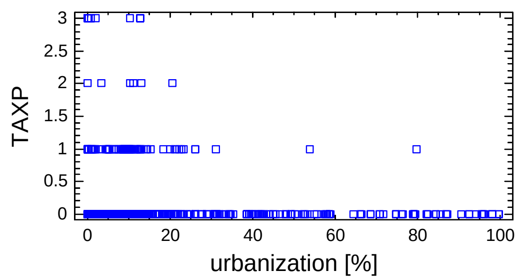

Figure 5.9. Changes in number of caddisfly taxa with urbanization.

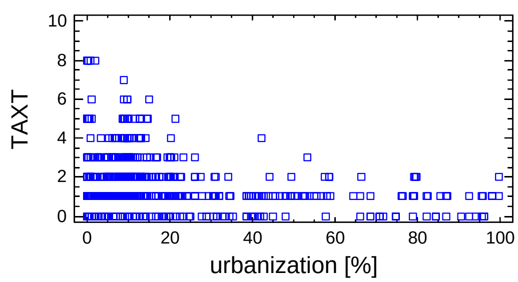

Figure 5.10. Changes in number of stonefly taxa with urbanization.

Effect of Habitat on IBI

Example: Number of Caddisfly Taxa

The methodology has been developed by Bartošová (2002). Total number of caddisfly taxa has been selected as an example to illustrate the analyses leading to habitat assessment methodology. Similar relationships can be found for other macroinvertebrate metrics such as total number of macroinvertebrate taxa, total number of mayfly taxa, or total number of stonefly taxa. Caddisfly taxa have been chosen as a representative metrics, showing significant change as well as having sufficient variability.

Substrate plays an important role in determining abundance of aquatic macroinvertebrates. Table 5.2 shows results of linear regression of total number of caddisfly taxa on substrate composition variables. Although p-value for slope is significant for almost all variables, value of R^2_{adj} is quite small.

Table 5.2. Effect of substrate composition on number of caddisfly taxa.

Variable	Scale	R^2_{adj} [%]	Slope	Slope P-value
% clay	Arit.	1.2	-0.025	0.000
% silt or smaller	Arit.	2.1	-0.013	0.000
% sand or smaller	Arit.	3.8	-0.013	0.000
% gravel or smaller	Arit.	4.3	-0.013	0.000
% rubble or smaller	Arit.	0.0	0.004	0.240
% boulder or smaller	Arit.	1.3	0.015	0.000

In spite of small correlation, scatter plots reveal significant relationship between total number of caddisfly taxa and percent substrate equal to or smaller than clay or rubble, respectively (Figures 5.11 and 5.12).

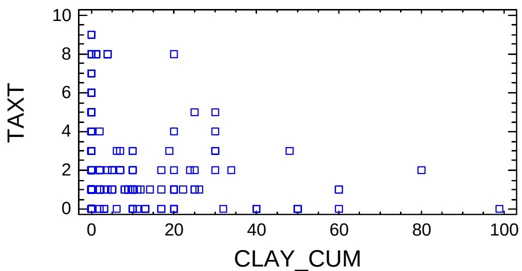

Figure 5.11. Changes in number of caddisfly taxa with percent clay in substrate.

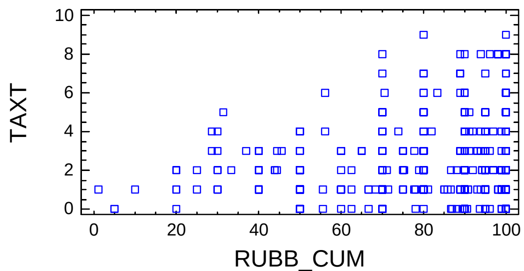

Figure 5.12. Changes in number of caddisfly taxa with percent rubble or smaller substrate.

Percentage of aquatic vegetation is another variable that does not show any correlation at all in spite of definite decrease in numbers with increasing percentage of aquatic vegetation. Figure 5.13 shows the decrease in number of taxa with increasing percentage of aquatic vegetation.

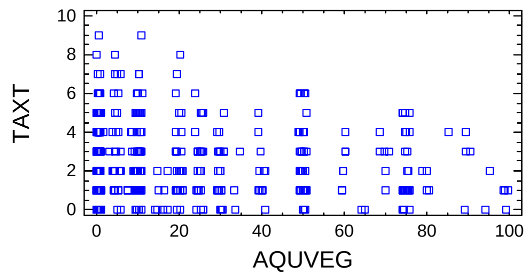

Figure 5.13. Changes in number of caddisfly taxa with aquatic vegetation.

Habitat Risk

The relationships described above for number of caddisfly taxa together with relationships for other macroinvertebrate metrics have been used to define the risk to aquatic biota. The methodology is based on Maximum Species Richness (MSR) concept (Lyons, 1992). The original MSR plots developed for fish species relate expected number of species to stream size. This concept has been extended for macroinvertebrates and their habitat (Bartošová, 2002).

Percent aquatic vegetation, percent of clay in substrate, and percent of rubble or smaller substrate have been selected as habitat indices having significant impact on macroinvertebrate metrics. The MSR plots have been converted into probability of taxa survival p_S by dividing the number of taxa by maximum number of taxa. Table 5.3 shows probability functions developed for selected macroinvertebrate metrics and habitat indices. Probability of taxa extinction p_E can be then calculated as $p_E = 1 - p_S$ for known habitat conditions.

Table 5.3. Taxa survival probability functions for habitat measures, $p_S^{(taxa|habitat)}$.

	% Aquatic vegetation (a)	**% Substrate ≤ clay (c)**	**% Substrate ≤ rubble (r)**
number of mayfly taxa	a<10%: 0.5 a>10%: 0.537 - 0.0037 a	0.5 - 0.0042 c	r<90%: 0.08 + 0.0046 r r>90%: 0.5
number of caddisfly taxa	0.5 - 0.044 a	0.5 - 0.005 c	0.055 + 0.0044 r
total number of taxa	0.26 - 0.0012 a	0.28 - 0.000022 c^2	0.035 + 0.0025 r

Probability of taxa survival has been calculated for each individual metrics and habitat measure. The joint probability of taxa extinction has been estimated assuming an independent effect of selected habitat measures as:

$$p_E^{(taxE)} = \left(1 - p_S^{(taxE|a)}\right)\left(1 - p_S^{(taxE|c)}\right)\left(1 - p_S^{(taxE|r)}\right) \quad \textbf{(5.1)}$$

$$p_E^{(taxT)} = \left(1 - p_S^{(taxT|a)}\right)\left(1 - p_S^{(taxT|c)}\right)\left(1 - p_S^{(taxT|r)}\right) \quad \textbf{(5.2)}$$

$$p_E^{(taxTotal)} = \left(1 - p_S^{(taxTotal|a)}\right)\left(1 - p_S^{(taxTotal|c)}\right)\left(1 - p_S^{(taxTotal|r)}\right) \quad \textbf{(5.3)}$$

where $p_E^{(taxE)}$ is the joint probability of mayfly taxa extinction, $p_S^{(taxE|a)}$ is the joint probability of mayfly taxa survival due to percent aquatic vegetation a, $p_S^{(taxE|c)}$ is the joint probability of mayfly taxa survival due to percent clay in substrate c,and $p_S^{(taxE|r)}$ is the joint probability of mayfly taxa survival due to percentof rubble or smaller substrate r. Similarly, $p_E^{(taxT)}$ is the joint probability of caddisfly taxa extinction and $p_E^{(taxTotal)}$ is the joint probability of total taxa extinction. The methodology is described in detail in Bartošová (2002).

Table 5.4 shows the results of calculation for those monitoring sites in the Oak Creek and Menomonee River watersheds where both habitat and biota were sampled. The risk ranges from 0.12 (mayfly taxa) to 0.44 (total taxa).

Table 5.4. Probability of taxa extinction due to selected habitat measures for sites in the Oak Creek and Menomonee River watersheds.

Site ID	Description	Mayfly taxa	Caddisfly taxa	Total taxa
RI-09	Menomonee River at Hart Park	0.168	0.179	0.435
RI-21	Menomonee River at Butler	0.141	0.152	0.390
MU-C	Mitchell Field Drainage Ditch at College Ave.	0.126	0.285	0.397
RI-24	Oak Creek at Howell Ave.	0.131	0.285	0.392
RI-26	Oak Creek at Nicholson Ave.	0.199	0.215	0.438
MU-R	Oak Creek at Ryan Rd.	0.149	0.332	0.406
RI-22	Menomonee River at Hampton Ave.	0.141	0.209	0.387
RI-10	Menomonee River at County Stadium	0.126	0.139	0.394

Ecological Risks and IBI

Figure 5.14 shows the effect of combined water column risk (i.e., acute plus chronic) on IBI values (see Chapter . The variation in IBI cannot be explained by water column risk only. The correlation is insignificant.

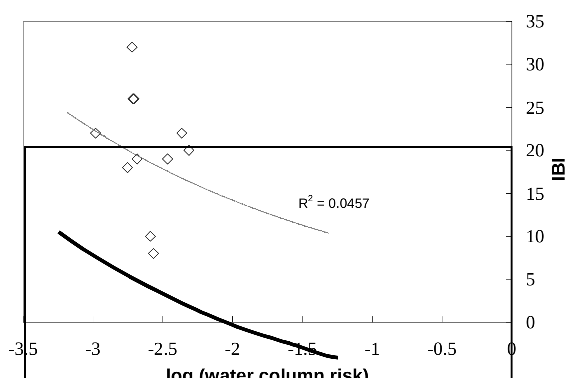

Figure 5.14. Effect of water column risk on MIBI.

However, the range of water column risk is quite narrow. With water quality comparable between sites, other factors determine the biotic integrity.

The IBI values are plotted against the cumulative sediment risk for copper and lead (Figure 5.15). There is a trend of decreasing IBI scores as cumulative sediment risk increases. The IBI scores do reflect some variation in addition to that caused by contaminated sediment. As sediment risks decreases, the effects on aquatic biota decrease and other factors become more important.

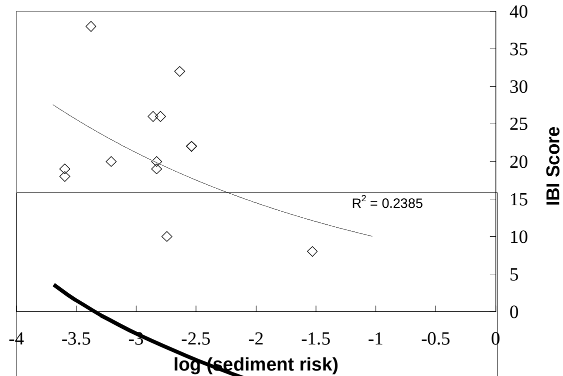

Figure 5.15. Effect of sediment risk on IBI.

Comparison of total risk, i.e. total water column risk plus total sediment risk, with IBI again shows the same trend of decreasing IBI scores as total (cumulative) risk increases, but there is more correlation than that for total sediment risk and IBI (Figure 5.16). This indicates that cumulative risk explains variation in IBI better than single component risk. From the three components investigated, sediment risk shows the highest correlation to IBI and is the driving force in determining the effect of chemical contamination on biotic integrity.

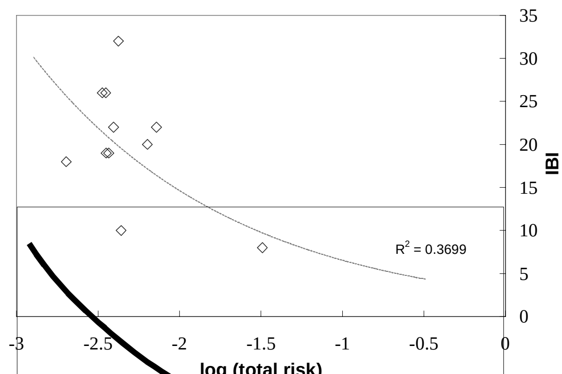

Figure 5.16. Effect of total risk (sediment and water column) on IBI.

Figures 5.17, 5.18 and 5.19 show the effect of individual habitat risks on IBI. There is a clear relationship between risk to caddisfly taxa and IBI. As the risk increases, the IBI linearly decreases. The relationship for habitat risks to mayfly taxa or total number of taxa is not so clear.

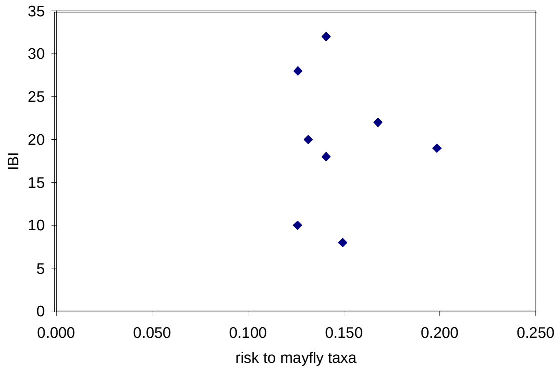

Figure 5.17. Effect of habitat risk to mayfly taxa on IBI.

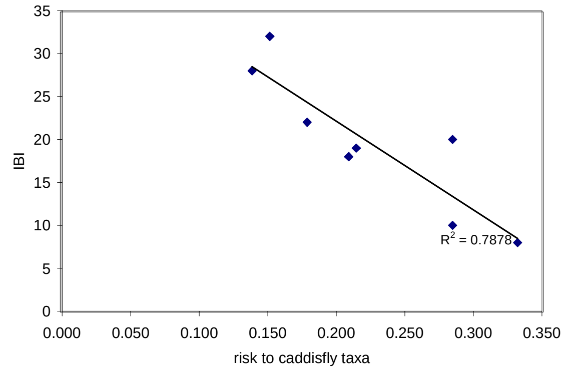

Figure 5.18. Effect of habitat risk to caddisfly taxa on IBI.

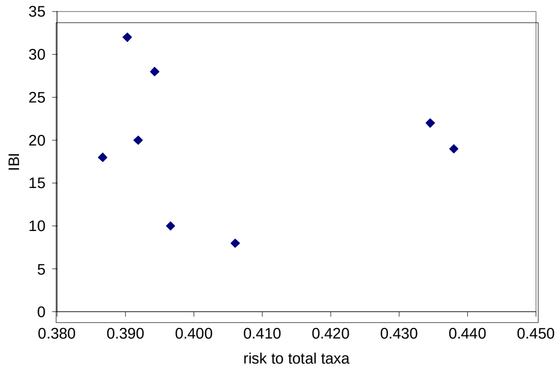

Figure 5.19. Effect of habitat risk to total taxa on IBI.

The sites with lowest IBIs hindering the relationship are located in the Oak Creek watershed (College Avenue and Ryan Road). The site at Ryan Road showed extremely high risk from contaminated sediment. The site at College Avenue drains part of Mitchell Field International Airport, Milwaukee. The contamination by heavy metals is not significantly higher than that for other sites, but other contaminants in the airport runoff degrade the water and sediment quality causing the decrease in IBI. When these sites are excluded, the general decreasing trend of IBI with increasing habitat risk appears.

The multiple regression was run on the individual components of ecological risk to determine weights of individual factors: acute toxicity due to water column contamination (*WQ_ac*), chronic toxicity due to water column contamination (*WQ_cc*), chronic toxicity due to sediment contamination (*Sed*), and risk due to habitat impairment in two components (risk to mayfly taxa, $p_E^{(taxE)}$, and geometric mean of all habitat risk components, $p_E^{(gavg)}$). The habitat risk components have been selected after running separate regression model selection procedure. The habitat risk components are inter-correlated. Although risk to caddisfly taxa showed the single highest correlation with IBI in selected watersheds, risk to mayfly taxa with geometric mean showed similar correlation and include the individual effects of all components. This result broadens applicability of the model for other watersheds. The results of regression are reported in Table 5.5. Only chronic toxicity due to water column contamination is not statistically significant on significance level $\alpha = 0.10$.

Table 5.5. Regression analysis. Effect of individual risk components on IBI.

Dependent variable: IBI

Parameter	Estimate	Standard Error	T Statistic	P-Value
CONSTANT	153.296	7.40861	20.6916	0.0023
$p_E^{(taxE)}$	245.714	14.4736	16.9767	0.0035
$p_E^{(gavg)}$	-444.209	18.4211	-24.1141	0.0017
WQ_ac	12.7881	1.2027	10.6328	0.0087
WQ_cc	-0.417349	1.14387	-0.364859	0.7502
Sed	4.55296	0.50236	9.06314	0.0120

R-squared = 99.7956 percent
R-squared (adjusted for d.f.) = 99.2848 percent
Standard Error of Est. = 0.685482
Mean absolute error = 0.249298

Table 5.6 then shows the final model after excluding insignificant variable:

$$IBI' = a\ p_E^{(taxE)} + b\ p_E^{(gavg)} + c\ WQ\_ac + d\ Sed + e \qquad \textbf{(5.4)}$$

where *IBI’* is the estimate of index of biological integrity, *a*, *b*, *c*, and *d* are the model coefficients determining the weight of individual risk components, and *e* is the model constant. The correlation is quite high: the R^2_{adj} reaches 99.5% showing almost perfect fit. The relationship between predicted and observed variables (IBI) can be seen on Figure 5.20. The plot confirms validity of the model. However, caution should be exercised when extrapolating the model to other watersheds. There are many variables influencing the model coefficients. Site specific data should always be used to estimate local model coefficients.

Table 5.6. Regression analysis. Effect of individual risk components on IBI.

Dependent variable: IBI

Parameter	Estimate	Standard Error	T Statistic	P-Value
CONSTANT	154.073	5.98329	25.7506	0.0001
$p_E^{(taxE)}$	246.54	12.0543	20.4525	0.0003
$p_E^{(gavg)}$	-447.617	13.3875	-33.4355	0.0001
WQ_ac	12.539	0.834857	15.0193	0.0006
Sed	4.55363	0.423603	10.7497	0.0017

R-squared = 99.782[illegible] percent
R-squared (adjusted for d.f.) = 99.4914 percent
Standard Error of Est. = 0.57802
Mean absolute error = 0.260396

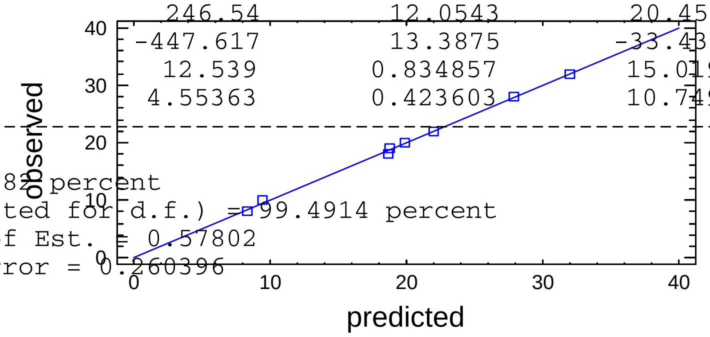

Figure 5.20. Plot of observed IBI versus IBI predicted by the final regression model.

The methodology was tested using data from the Menomonee River and Oak Creek watersheds. Applicability to a regional data base (Wisconsin) could not be tested accurately because of data gaps in the regional data base.

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