NATIONAL WATER RESEARCH INSTITUTE

INSTITUT NATIONAL de RECHERCHE sur les EAUX

# RIVERS HANDBOOK, VOLUME 2
## CHAPTER 3 : MONITORING PROGRAMMES
## SECTION 3.3 : BIOLOGICAL WATER QUALITY ASSESSMENT OF RIVERS BASED ON MACROINVERTEBRATE COMMUNITIES

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## MANAGEMENT PERSPECTIVE

Until recently, the monitoring, assessment, regulation and remediation of aquatic systems was largely based on chemical measures of water quality. Despite massive regulatory efforts, however, the quality of water resources in Europe and North American has continued to decline. It has now been recognized that the incorporation of biological assessment techniques into management policies is essential for the adequate protection of these resources.

There is a general consensus that benthic macroinvertebrate communities are among the most sensitive components of aquatic systems on which to base assessments of ecosystem health. Major advances in the development and application of benthic community assessment techniques have occurred over the last ten years in Europe and over the last five years in the United States. As a result, macroinvertebrate community assessment techniques are now influencing policy decisions concerning surface water management in both Europe and North America. In a landmark decision, the 1987 amendment of the Clean Water Act in the United States called for protecting the "biological integrity" of the Nation's waters. Individual states will be required to develop numerical biological criteria based on fish and macroinvertebrate communities by 1993.

Environment Canada is currently in the process of developing a new regulatory action plan for the pulp and paper industry where, again, existing environmental legislation has proven inadequate. Benthic macroinvertebrate community assessments are included among the "core" biological components of the proposed environmental effects monitoring program.

This contribution is intended for publication by Blackwell Scientific Publications, Oxford, in the two-volume "Rivers Handbook" (P. Calow & G.E. Petts, eds.) which will be a reference work on the application of ecologically-sound approaches, methods and tools in river management throughout Europe and North America. The chapter provides an account of the evolution of benthic bioassessment techniques, describing traditional, alternative and new approaches. The techniques which have been implemented in Great Britain and the United States are critically evaluated, and the approaches of other countries, including Belgium, The Netherlands and Spain are also discussed. This information is required for the informed selection of the most suitable techniques to be incorporated into the "Technical Guidance Manual for Aquatic Environmental Effects Monitoring at Pulp and Paper Mills", which is currently in preparation. Similar legislation is planned next for the regulation of the mining industry. It is critical that we take this opportunity to ensure that 1990s technology is applied to the management of these industries.

## SOMMAIRE À L'INTENTION DE LA DIRECTION

Jusque tout récemment, la surveillance, l'évaluation, la réglementation et l'assainissement des systèmes aquatiques reposaient en grande partie sur des mesures chimiques de la qualité de l'eau. Toutefois, malgré des efforts considérables en matière de réglementation, la qualité des ressources en eau en Europe et en Amérique du Nord a continué de diminuer. Il est maintenant reconnu que l'intégration de techniques d'évaluation biologique dans les politiques de gestion est essentielle si l'on veut protéger adéquatement ces ressources.

Il est généralement admis que les populations de macroinvertébrés benthiques figurent parmi les éléments les plus sensibles des systèmes aquatiques sur lesquels reposent les évaluations de la santé de l'écosystème. Les principaux progrès en matière de développement et d'application de techniques d'évaluation des populations benthiques ont été réalisés au cours des dix dernières années en Europe et au cours des cinq dernières années aux États-Unis. Il s'ensuit donc que ces techniques influent maintenant sur les décisions en matière de politique concernant la gestion des eaux de surface, tant en Europe qu'en Amérique du Nord. Dans une décision notoire, la modification de 1987 apportée au Clean Air Act des États-Unis exigeait que l'on protège "l'intégrité biologique" des eaux des nations. Chacun des états devra élaborer d'ici 1993 des critères biologiques numériques fondés sur les populations de poissons et de macroinvertébrés.

Environnement Canada élabore à l'heure actuelle un nouveau plan d'action en matière de réglementation pour l'industrie des pâtes et papiers domaine dans lequel, encore une fois, les dispositions législatives se sont révélées inappropriées. Les évaluations de populations de macroinvertébrés benthiques sont au nombre des éléments biologique "centraux" du programme de surveillance proposé des effets sur l'environnement.

Cet article sera publié par les Blackwell Scientific Publications (Oxford), dans l'ouvrage en deux volumes intitulé "Rivers Handbook" (P. Calow & G.E. Petts, eds.) et qui servira de référence sur l'application d'approches, de méthodes et d'outils écologiques en matière de gestion des cours d'eau en Europe et en Amérique du Nord. Le chapitre est un compte rendu de l'évolution des techniques d'évaluation biologique des organismes benthiques qui décrit des approches classiques, des approches de rechange et nouvelles. Les techniques employées en Grande-Bretagne et aux États-Unis sont évaluées de façon critique, et les approches adoptées par d'autres pays, notamment la Belgique, les Pays-Bas et l'Espagne, sont également abordées. Ces informations sont nécessaires afin de permettre un choix éclairé des techniques les plus appropriées qui seront incorporées dans l'ouvrage intitulé "Technical Guidance Manual for Aquatic Environmental Effects Monitoring at Pulp and Paper Mills", qui est en préparation. Des dispositions législatives similaires sont prévues en suite en vue de la réglementation de l'industrie minière. Il est important que nous profitions de cette occasion pour nous assurer que la technologie des années 1990 soit appliquée à la gestion de ces industries.

## ABSTRACT

This report provides a critical review of the history and current status of biological water quality assessment of rivers in Europe and North America, based on benthic macroinvertebrate communities. Traditional approaches to bioassessment (saprobic, diversity, biotic and community comparison indices), alternative approaches (functional feeding groups, reduced assemblages and ratio indices) and recent developments in the United Kingdom (multivariate approach) and the United States (U.S. EPA's Invertebrate Community Index and Rapid Bioassessment Protocols) are described, and their various applications discussed. Where the information is available, the performances of various indices and approaches are directly compared. New directions, which address multiple and non-point source stresses, the need for diagnostic tools to identify the characteristic responses of benthic communities to specific stresses, and recent modifications of existing techniques are also presented. The report provides information required by water resource managers for the informed selection of macroinvertebrate community monitoring techniques appropriate for the classification, assessment and protection of rivers and streams exposed to a wide variety of stresses under a range of environmental conditions.

## RÉSUMÉ

Le présent rapport est une étude critique des antécédents et de l'état actuel de l'évaluation de la qualité biologique de l'eau des cours d'eau en Europe et en Amérique du Nord, fondée sur les populations de macroinvertébrés benthiques. On décrit les approches classiques d'évaluation biologique (indices de comparaison saprobiotique, biotique, des populations et de la diversité), d'autres approches (groupes d'alimentation directe, assemblages réduits et indices des proportions) et les récents progrès réalisés au Royaume-Uni (approche à plusieurs variables) et aux États-Unis (U.S. EPA's Invertebrate Community Index et Rapid Bioassessment Protocols) et on aborde leurs différentes applications. Lorsque des informations sont accessibles, les performances des différents indices et approches font l'objet de comparaisons directes. Les nouvelles orientations, qui portent sur des agressions provenant de sources multiples et de sources diffuses, le besoin d'outils de diagnostic afin de déterminer les réactions caractéristiques des communautés benthiques à des agressions particulières, et les modifications récentes apportées aux techniques existantes sont également traités. Le rapport fournit des informations nécessaires aux gestionnaires des ressources en eau afin de leur permettre de faire un choix éclairé des techniques de surveillance des populations de macroinvertébrés qui conviennent à la classification, à l'évaluation et à la protection des cours d'eau et des ruisseaux exposés à des agressions très variées dans un vaste éventail de conditions environnementales.

## 3.3.1. INTRODUCTION

At present, the monitoring, assessment, regulation and remediation of aquatic ecosystems is largely based on chemical measures of water quality. Yet, chemical parameters alone do not provide adequate information for sound management of aquatic resources because they tell us little of the effects of pollution on living organisms.

Direct biological assessments of the health of biotic communities in receiving waters offer several important advantages over chemical-based approaches. For example, organisms integrate environmental conditions over time, whereas chemical data are instantaneous in nature and require large numbers of measurements for an accurate assessment (De Pauw & Vanhooren, 1983). Biological communities also integrate the effects of multiple stresses and demonstrate cumulative impact (Plafkin, Barbour, Porter et al, 1989). Biological studies can serve an early warning function by detecting intermittent pollution and subtle disruptions which would likely be missed by conventional chemical surveys (Howmiller & Scott, 1977; Reynoldson, 1984). Chemical monitoring programs are usually menu-driven, therefore, the possibility exists that the pollutants or factors responsible for environmental degradation will be excluded from consideration. Finally, it must be recognized that not all impacts are chemical in nature; only biological assessments can detect the impact of flow alterations, habitat destruction, overharvesting of biological resources, etc. (Karr, 1991).

As it is obviously impractical to conduct bioassessments on entire aquatic ecosystems, most workers have focused on a particular component. Hellawell (1977) and Reynoldson (1984) tabulated the advantages and disadvantages of all major groups, and a clear preference for using macroinvertebrates emerged. Benefits of using benthos include: (1) Macroinvertebrate communities are differentially sensitive to pollutants of various types and react to them quickly (Cook, 1976), and are capable of a graded response (Pratt & Coler, 1976). (2) Macroinvertebrates are present in most aquatic habitats, especially flowing water systems (Reynoldson, 1984), and are abundant and relatively easy and inexpensive to collect (Plafkin, Barbour, Porter et al, 1989). Furthermore, their taxonomy is well-established, although admittedly difficult at the species level for some groups (Reynoldson, 1984). (3) Benthic invertebrates are relatively sedentary, and are therefore representative of local conditions (Cook, 1976). (4) They have life spans long enough to provide a record of environmental quality (Pratt & Coler, 1976). (5) Finally, macroinvertebrate communities are very heterogeneous, with numerous phyla and trophic levels represented. The probability that at least some of these organisms will react to a particular change in environmental conditions is, therefore, high (France, 1990).

The use of macrobenthos in bioassessment has three major disadvantages: (1) They respond to seemingly minor changes in substrate particle size, organic content and even texture. As a result, discrimination between the effects of pollution and other environmental factors is often difficult (France, 1990). (2) Their life histories are complex and the results of bioassessments can vary with season (Hellawell, 1977). (3) Spatial heterogeneity is high, requiring considerable replication (Reynoldson, 1984).

This chapter is largely based on an earlier review of bioassessment techniques developed and applied in Europe (Metcalfe, 1989). The information presented here consists of an update of this review and also considers the North American literature and several new and alternative approaches to bioassessment.

## 3.3.2 TRADITIONAL APPROACHES TO BIOASSESSMENT

### 3.3.2.1 *The Saprobic System*

The term 'saprobia' refers to the dependence of an organism on decomposing substances as a food source (Persoone & De Pauw, 1979). The early research efforts of two German scientists, R.W. Kolkwitz and M. Marsson, led to the classic Saprobic System. It is best known through the saprobic index, which is based on the presence of indicator species (mainly bacteria, algae, protozoans and rotifers, but also some benthic invertebrates and fish) which have been assigned saprobic values based on their pollution tolerance. Values range from 0 to 8; the higher the value the more tolerant the organism. Pollution tolerances of individual species are determined by observations on their relative occurrence under specifically-defined conditions of water quality. According to the saprobic system, water quality is classified into one of ten categories ranging from the purest ground water to anaerobic sewage and industrial wastes (see Table 1 in Metcalfe, 1989). No single indicator species will be representative of only one saprobic zone; rather, its distribution will follow a normal curve over a range of zones reflecting its tolerance. The shape and area of this distribution curve defines the saprobic 'valency' of the species (Zelinka & Marvan, 1961), and the position of the apex is its saprobic value (Sladecek, 1979). Various lists of saprobic values have been published, all for European species. Most notable is that of Sladecek (1973), which contains information for approximately 2000 species.

Briefly, the Saprobic Index is calculated as follows:

$$S = \frac{\Sigma(s.h)}{\Sigma h}$$

where S = Saprobic Index for the site

s = saprobic value for each indicator species

h = frequency of occurrence of each species; rare: h = 1, frequent: h = 3, abundant: h = 5

The value of 'S' will normally range from 1 to 4 for ambient waters. Major criticisms of saprobic systems are: taxonomy is not far enough advanced for some groups and too controversial for others, intensive sampling is required, species lists and saprobic values will not be applicable to other geographic locations, the system cannot be confidently applied to other types of pollution (Persoone & De Pauw, 1979); pollution tolerances of species are very subjective, as they are based on observational rather than experimental data (Slooff, 1983); and each taxon is considered as a separate entity, therefore, no information on the community as a whole is provided (Jones, Tracy, Sebaugh, et al, 1981).

Two saprobic-based systems are currently in use: the Biologically Effective Organic Loading (BEOL) method in West Germany (Persoone & De Pauw, 1979) and the Quality-index, or K-index, in The Netherlands (Woodiwiss, 1980). The K-index is calculated as follows: approximately 60 indicator taxa are arranged in five groups, each of which is assigned a pollution factor (see Table 2 in Metcalfe, 1989). The percentage of the total number of animals in the sample belonging to each group is then multiplied by the appropriate factor, and the group values are summed into an index value ranging from 100 (very heavily polluted) to 500 (not polluted), as follows:

$K_{135}$ = (% Eristalsis + Chironomus-group) X 1 + (% Hirudinea-group) X 3 + (% Gammarus + Calopteryx-group) X 5.

In 1980, the Limburg Water Pollution Control Authority began to develop a biological classification system for rivers and streams in the province of Limburg, The Netherlands. Their purposes were to determine the extent of pollution by organic wastes, evaluate the effectiveness of enforced measures to reduce pollution, and define reference communities for different types of streams which could be used as water quality objectives. To select the most appropriate bioassessment method, Tolkamp (1985a,b) applied a variety of biotic and saprobic indices to a large data set from the drainage basin of the River Geul. None performed as well as the K-index, which was more sensitive to smaller changes in the middle range of the pollution scale. Most other indices underestimated water quality because they included indicator organisms which do not normally occur in these lowland streams. Vandelannoote, De Gueldre & Bruylants (1981) reported similar results for lowland streams in Belgium. The majority of biotic indices available were developed for small upland streams, and the assessment of lowland streams and rivers has generally been neglected.

#### 3.3.2.2 *Diversity Indices*

Diversity indices are mathematical expressions which use three components of community structure, namely, <u>richness</u> (number of species present), <u>evenness</u> (uniformity in the distribution of individuals among the species) and <u>abundance</u> (total number of organisms present), to describe the response of a community to the quality of its environment. Undisturbed environments are characterized by high diversity or richness, an even distribution of individuals among the species, and moderate to high counts of individuals (Ghetti & Bonazzi, 1977; Mason, Lewis & Weber, 1985). Organic pollution causes a decrease in diversity as sensitive organisms are lost, an increase in the abundance of tolerant organisms due to nutrient enrichment, and a decrease in evenness. In contrast, toxic or acidic pollution may cause a decrease in both diversity and abundance as sensitive organisms are eliminated and there is no additional food source for the remaining tolerant forms, and an increase in evenness (Kovalak, 1981).

By far the most widely used diversity index is the Shannon-Wiener index (H'), because it is stable in any spatial distribution and insensitive to rare species (Cairns & Pratt, 1986). Its formula is as follows (after Wilhm & Dorris, 1968):

$$H' = -\sum \frac{Ni}{N} \log_2 \frac{Ni}{N}$$

where: H' = index value

N = total number of individuals of all species collected

Ni = number of individuals belonging to the ith species

The higher the value of H', the greater the diversity and, supposedly, the cleaner the environment. The reader is referred to Washington (1984) for a critical review of the many diversity indices which have been applied to aquatic ecosystems. Diversity indices are considered to have the following advantages: they are strictly quantitative, dimensionless, and lend themselves to statistical analysis (Cook, 1976); most are relatively independent of sample size (Wilhm & Dorris, 1968); no assumptions are made about the relative tolerances of individual species, which may be very subjective (Pinder, Ladle, Gledhill et al, 1987).

France (1990) notes that 'Few subjects in applied ecology are as controversial as the use (and misuse) of diversity indices.' Many criticisms have been levelled at diversity indices, and only the major ones will be identified here. Their most serious problem is that they reduce individual species to anonymous numbers which disregard their environmental adaptations. This could result in equating, theoretically, a pollution-tolerant oligochaete/chironomid community with a pollution-sensitive mayfly/amphipod community (France, 1990). Secondly, not all undisturbed communities have inherently high diversity, therefore, it is not always possible to correlate certain values with ecological damage (Jones, Tracy, Sebaugh et al, 1981). For example, the oligotrophic offshore areas of large lakes (Howmiller & Scott, 1977) and the headwaters of streams fed by nutrient-poor groundwater (Pinder & Farr, 1987) are naturally low in productivity. Because wide variations in diversity index values have been reported for unpolluted conditions (Cook, 1976), standards set for the interpretation of values are not universally applicable. Thirdly, diversity indices may generate 'false negatives' under certain circumstances. Moderate pollution can cause an increase in abundance without excluding species, with the result that the index value actually goes up (Cook, 1976). Because H' is more sensitive to changes in evenness than diversity, its value may be high at sites heavily contaminated by toxic chemicals (Kovalak, 1981). Finally, many studies have shown that diversity indices are insensitive and give poor site discrimination, particularly over the moderate range of various types of pollution including nutrients (Jones, Tracy, Sebaugh et al, 1981), metals (Perkins, 1983) and pesticides (Webber, Bayne & Seesock, 1989). It is this last characteristic that limits the usefulness of diversity indices to assessing the impact of gross point source pollution of known chemical composition on relatively simple systems. They are of little use in complex systems affected by multiple and diverse stresses and/or basin-wide non-point source pollution. Some developing countries (e.g., Jhingran, Ahmad & Singh, 1989) continue to rely on diversity indices for assessing severe sewage pollution.

#### 3.3.2.3 *Biotic Indices*

The biotic approach to bioassessment, as defined by Tolkamp (1985b), is one which combines diversity on the basis of certain taxonomic groups with the pollution indication of individual species or higher taxa or groups into a single index or score. Numerous biotic index and score systems have been developed, most of them in Europe and the United Kingdom. The major indices and their relationships are illustrated in Fig.3.3.1. Only the most widely used systems and their recent modifications will be described. The reader is referred to Metcalfe (1989) and Washington (1984) for information on other biotic indices.

## TRENT BIOTIC INDEX

The Trent Biotic Index (TBI) was originally devised for use in the Trent River Authority area in England, but has since been adapted for use in many other countries and appears to form the basis for most modern biotic indices and scores (Persoone & De Pauw, 1979). Organisms are collected from all available habitats by means of a kicknet, then identified to Family, Genus or species depending on the type of organism, but they are not enumerated. The index is based on the sensitivity of key groups to pollution and on the number of component groups in a sample. Clean streams are given an index value of 10, and this value decreases with increasing pollution. The TBI was later extended to cover a wider range of water qualities (0 to 15 instead of 0 to 10) to improve sensitivity. This version, called the Extended Biotic Index (EBI), is shown in Table 3.3.1. One major drawback of these indices is that abundance is ignored. Therefore, the accidental presence of an organism in a sample (due to drift, for example), could drastically alter the value of the index (Cook, 1976).

## CHANDLER'S SCORE SYSTEM

This system was developed for upland rivers in Scotland (Cook, 1976). Chandler's Score is theoretically an improvement over the TBI because it includes an abundance factor and incorporates a more detailed list of macroinvertebrates. The Score is determined by identifying the organisms present, determining the abundance classification for each group, then selecting the appropriate points for that group (see Table 5 in Metcalfe, 1989). The points for all groups are added to give a site Score. Points scored increase with increasing abundance for sensitive groups and decrease with increasing abundance for tolerant groups, and the value of the site Score is unending. Criticisms of this system are that it is too complicated, the level of taxonomic identification is not uniform for all groups and also that it is geographically restricted due to the number of indicators identified to Genus. However, Cook (1976) found that modifications aimed at adapting Chandler's Score to local conditions did not significantly improve the performance of the system in a New York stream. The Score was superior to the diversity index H' in grading sections of a mildly polluted stream according to water quality. Armitage (1980) applied Chandler's Score to a zinc-contaminated river in Northern England and found that Score values were lowest at sites affected by high zinc levels, suggesting that it may be adaptable to other types of pollution.

## BIOLOGICAL MONITORING WORKING PARTY SCORE SYSTEM

The Biological Monitoring Working Party (BMWP) was set up in 1976 to develop a standardized system for assessing the biological quality of rivers in England, Scotland and Wales. Its terms of reference were '...to provide an overall view of the condition of rivers and canals and of the discharges to them and to show the effectiveness of pollution control policies.' (ISO, 1984). They developed a standardized score system which was a simplification of Chandler's Score, where all organisms were identified to Family for uniformity, families with similar pollution tolerances were grouped together, and the abundance factor was eliminated because it was time-consuming and had only a small effect on score value. The method for scoring is as follows, using Table 3.3.2: list the families present in the sample, ascribe the score for each Family, then add the scores together to arrive at a site score.

The 'average score per taxon' (ASPT) computation, which simply refers to dividing the total score by the number of scoring taxa, has frequently been applied to both the Chandler and BMWP Scores because it is independent of the number of taxa counted. Murphy (1978) compared the performance of Chandler's Score and ASPT, the TBI and several diversity indices over a range of polluted and unpolluted sites in Welsh rivers. The Chandler ASPT showed the least temporal variability, thus allowing the best spatial discrimination. While Score and TBI values were depressed at headwater sites, probably reflecting physical habitat properties, ASPT values were high, reflecting good water quality. Armitage, Moss, Wright et al, 1983) evaluated the BMWP Score vs. ASPT for classifying unpolluted sites with different physical and chemical characteristics. The results showed a steady decline in ASPT values from the upland to the lowland range of environmental features, while Score values varied. Predictive equations based on multiple regressions using physical and chemical parameters explained 65% of the variance in ASPT values as opposed to only 22% for Score values.

The performance of the Chandler and BMWP Scores and their ASPT versions were compared in a three-part study on a chalk stream in southern England (Pinder, Ladle, Gledhill et al, 1987; Pinder & Farr, 1987a,b). They found that the BMWP score stabilized after fewer replicates than Chandler's score, and that the ASPT versions of both were much less affected by sample size, season and habitat. The BMWP-ASPT was most sensitive to slight changes in pollution status, ranked sites similarly to both Scores and several diversity indices, and was also best correlated with direct chemical measures of water quality. In contrast, the Chandler-ASPT consistently failed to agree with other indices. Pinder & Farr (1987b) concluded that the BMWP-ASPT was the best biotic index available, although they recommended eliminating the Chironomidae (except Chironomus riparius) and Oligochaeta from the scoring system. Since both of these groups include many species which are tolerant of pollution and many which are sensitive, their inclusion without further discrimination had the effect of depressing the ASPT while contributing nothing to the score. After eliminating these groups from their calculations, Pinder & Farr (1987b) achieved better site discrimination.

## THE BELGIAN BIOTIC INDEX METHOD

The Belgian Biotic Index Method (BBI) is derived from the French Indice Biotique (IB), which in turn is a modification of the Trent Biotic Index. The IB differs from the TBI in that it includes a greater number of indicator taxa (Persoone & De Pauw, 1979), separates the Ecdyonuridae from other Ephemeroptera and divides the Trichoptera into species with and without cases (De Pauw & Vanhooren, 1983), does not separate Nais from other Naididae and Baetis from other Ephemeroptera (Ghetti & Bonazzi, 1977) and does not consider a systematic unit represented by a single individual since its occurrence may be accidental. Also, the TBI specifies that a handnet be used for sampling all habitats, while the IB calls for lentic and lotic habitats to be sampled separately and for indices from both habitats to be used in interpreting water quality (Persoone & De Pauw, 1979). De Pauw & Vanhooren (1983) adapted the IB for use in Belgium, with the following modifications: sampling would be by handnet only, as this technique explores a larger arrays of habitats than other samplers; nematodes would be excluded from consideration, as most would not be caught in a 300-500 micron mesh handnet; the Chironomidae were divided into two systematic units, those belonging to the thummi-plumosa group and those not; level of taxonomic identification would generally be set at a higher level (Genus or Family) to avoid erroneous interpretations due to misidentification.

The BBI is calculated using Table 3.3.3., which has both rows and columns representing faunistic groups and systematic units (SUs), respectively. The seven faunistic groups are ranked in order of increasing tolerance to pollution. For groups 1-3, it is necessary to know whether there are 1, 2 or more SUs present. The row chosen from the table is the one corresponding to the presence of the most sensitive faunistic group in the sample. The column chosen depends on the number of systematic units present in the sample. The intersection of the appropriate row and column gives the index value for the site.

The surface water quality of Belgian rivers has been routinely surveyed by the BBI since 1978, and by 1985, over 30,000 km of watercourses had been surveyed and mapped using the BBI. The program is sponsored by the National Institute for Hygiene and Epidemiology in Brussels and is driven by the urgent need for a coordinated policy in the field of surface water sanitation and management (De Pauw & Vanhooren, 1983). Goals of the program are to obtain better insight into the self-purification of rivers and streams and to assist decision-makers in selecting sites for water purification plants and surface water reservoirs. Belgium requires a method which is equally applicable in fastflowing shallow streams and slow-running deep lowland rivers and canals. The BBI has been declared highly successful (De Pauw & Vanhooren, 1983) for this application. Identification keys and handnet specifications have been standardized, and intercalibration exercises with respect to sampling and identification gave satisfactory results. Results are reproducible over long periods of time in areas where no changes in pollution status occur, and seasonal changes are minor. A single sampling in either early summer or fall is sufficient for a proper assessment. Artificial substrates were found to be a valid alternative for deep, lowland rivers where handnet sampling was difficult. De Pauw, Roels & Fontoura (1986) found that three pooled replicates of a 2250 - 4500 $cm^3$ sampler yielded BI values equivalent to those generated by handnet collections.

De Pauw & Roels (1988) investigated the relationship between the BBI and various common chemical indicators of pollution, using data from a wide variety of polluted and unpolluted sites in Belgium, Italy, Portugal and the United Kingdom. They found that correlations between chemical variables and the BBI were consistently positive (dissolved oxygen) or negative (BOD, COD, $NH_4$, $PO_4$), but that the slopes of the regression lines varied considerably among watersheds. This indicated that the degree of stress associated with a particular chemical factor in one river was not necessarily of the same magnitude in another. They suggested, therefore, that '...biological assessments should be used as an early warning system and be the precursor of extensive chemical analyses, identifying the causes of biological stress.' The BBI has recently been shown to be applicable in other countries, including Spain, Algeria, Luxembourg, Portugal and Canada (De Pauw, Roels & Fontoura, 1986).

Bervoets, Bruylants, Marquet et al (1989) proposed several modifications to the BBI to improve reliability, save time and provide better correlation with water chemistry. BBI values were found to be higher if samples were sorted live, instead of preserved in formalin and washed through a series of sieves, and if SUs consisting of only 1 individual were included in the calculation of the index. With respect to sample replication, polluted sites could be accurately characterized in only two subsamples, whereas new single-individual SUs were often found in the 10th subsample from unpolluted brooks. They strongly recommended that sampling effort be standardized for area sampled rather than time, because the specified five minute period was insufficient for finding rarer individuals in the cleanest streams. Their modification was also more efficient, allowing more samples to be processed within a given time period.

The Indice Biologique Global is the standard method presently used in France (AFNOR, 1985). It is also derived from the IB, and its application appears to be restricted to that country. Further details on the development of the French indices are presented in Metcalfe (1989).

## CHUTTER'S BIOTIC INDEX

A biotic index for South African streams and rivers was developed by Chutter (1972). European indices such as the TBI could not be readily applied, because some of the key indicators were absent (e.g., Gammarus, Asellus) or of very restricted occurrence (Plecoptera) in South African rivers, while the Baetidae fauna was much richer. Chutter assigned Quality (tolerance) values of 1(clean) - 10(polluted) to all taxa collected, based on the literature. All 'pristine' species were assigned a value of 0. Each organism found in the sample was recorded at its quality value, then the values were summed for all taxa and divided by the total number of individuals. Because some taxa were extremely abundant, he included a 'sliding scale' of quality values which takes into account the abundance and diversity of these dominant taxa. Because of the instability of flows and river beds in the rainy season in South Africa, the index was not reliable in recently flooded areas. Although loosely based on the TBI, this approach is unique in that every individual organism contributes to the index value. According to Washington (1984), Hilsenhoff is the only worker to consider Chutter's index and adapt it for use in another country.

## HILSENHOFF'S BIOTIC INDEX

In 1977, Hilsenhoff introduced a biotic index for evaluating organic pollution in Wisconsin streams based on riffle-dwelling arthropod fauna (Hilsenhoff, 1987). To determine the index value for a given site, a sample of 100 or more arthropods was collected by means of a handnet and the organisms identified to Genus or species. Each taxon was assigned a tolerance value ranging from 0 (most sensitive) to 5 (most tolerant) based on information from 53 streams. As per Chutter's index, the average of the tolerance values for all individuals constituted the site index value. Hilsenhoff's biotic index was extensively tested by the Wisconsin Department of Natural Resources and the data generated was used to improve the index. Tolerance values were revised by comparing the original tolerance value assigned to a species with the average biotic index value of streams in which it most commonly occurred. To provide greater precision, the scale was expanded from 0 - 10. To date, tolerance values have been assigned to some 400 species or genera, and these are listed in Hilsenhoff (1987). The influence of current, temperature and seasonal factors on BI values have also been evaluated (Hilsenhoff, 1988b). BI values were found to be erroneously high in summer, when many sensitive species are in diapause, and in currents of less than 0.30 m/sec. Table 3.3.4. is a guide to the water quality of streams based on this index.

Hilsenhoff (1988a) also adapted his index for a rapid assessment by providing tolerance values for families. Because family tolerance values are averages, they are lower than the tolerance values for some species in the family and higher than the values for other species. The result should be a dampening effect on the performance of the index. This was confirmed in testing on second and third-order Wisconsin streams, where the family-level biotic index (FBI) tended to be higher than the BI at unpolluted sites and lower at polluted sites. The FBI was also more variable. However, an average of only 23 minutes was required to sample, sort and calculate an FBI in the field, as compared with at least 85 minutes to calculate a BI. Hilsenhoff (1988a) recommended using the FBI only for rapid assessment of the general status of organic pollution in streams, in essence, as a screening tool to identify problem areas. He noted that if all organisms were preserved, the BI could always be calculated later if needed.

Despite all of the effort which has gone into the development of biotic indices, they have serious limitations as bioassessment tools. First of all, '...they set the same target for all sites when it is clear that different physical and chemical regimes of fastflowing mountain streams and slow-flowing lowland rivers will support totally different faunal communities.' (Armitage, Pardo, Furse et al, 1990). Secondly, a primary weakness of biotic indices is the subjective approach which is often used to classify organism tolerance. Herricks & Cairns (1982) suggest that only where a long record of study is available (e.g., Sladecek, 1973; Hilsenhoff, 1987), is reliability high. They recommend replacing 'subjective tolerance estimates' with 'quantitative tolerance determinations', which require extensive correlations between species presence and water quality. Thirdly, biotic indices apply only to organic pollution, and their application to other types of pollution or perturbation is questionable at best and erroneous at worst. Clearly, a more diagnostic approach is needed.

#### 3.3.2.4 *Community Comparison Indices*

Community comparison indices (CCIs), which measure the similarity of the structure of two communities, have mostly been developed for use in terrestrial ecology and have not been extensively applied to aquatic ecosystems (Washington, 1984). CCIs require a clean water station for comparison, and have therefore mainly been used for upstream-downstream contrasts of the response to point source pollution. Like diversity indices, they are strictly quantitative and therefore provide no information about the actual composition of communities. Unlike biotic indices, they will respond to any perturbation that affects benthic communities, not just organic pollution. There is some evidence that CCIs may be more sensitive to subtle changes in community structure than diversity indices. For example, Perkins (1983) found that CCIs demonstrated a consistent decrease in similarity among benthic communities exposed to increasing concentrations of copper, while the diversity index H' gave false negatives for the lower concentrations. Washington (1984) called for further evaluation of CCIs, specifically for comparisons with both diversity and biotic indices.

A wide range of CCIs are currently available, all of which have somewhat different mathematical and ecological properties. To add to the confusion, some measure similarity and others dissimilarity. Because of their differing properties, various indices generate somewhat different results when applied to the same data set. Most require both validation under controlled conditions and careful consideration of their intrinsic properties to determine which index is most suited to the objectives of a particular study and the type of data available. Formulae for an exhaustive list of indices are presented in Perkins (1983) and Washington (1984). One example of the application of CCIs will be presented here.

Brock (1977) used the indices $P_{sc}$ and B to evaluate the effects of thermal effluents on zooplankton communities in a reservoir in central Texas. The formulae are as follows:

$$P_{sc} = 100 \div 0.5 \sum^{k} | a - b |$$

where: a and b are, for a given species, percentages of the total samples A and B which that species represents. The absolute value of their difference is summed over all species, k.

$$B = \frac{1}{k} \sum^{k} \frac{\min\,(Xia,\ Xib)}{\max\,(Xia,\ Xib)}$$

where: Xia and Xib = abundances of species i at Stations A and B, the smaller number being divided by the larger number for each species, and

k = total number of species observed between the two stations.

$P_{sc}$ is based on relative abundance while B is based on actual abundance, therefore, the two indices respond differently to certain changes in community structure. Brock (1977) found that sites with very similar proportions of taxa but very different total abundances registered a high degree of similarity according to $P_{sc}$ and less similarity according to B. The removal of a few rare species from a data set for two stations hardly affected the value for $P_{sc}$, whereas B indicated a definite change. Conversely, B was insensitive to the addition of a substantial number of individuals to a dominant taxon. Brock (1977) concluded that if B overemphasizes shifts in rare taxa and de-emphasizes changes among dominant species, then it is too sensitive to normal sampling error. It could, however, have an important application where the loss of rare or endangered species is of interest.

Camargo (1990) recently developed a new 'ecotoxicological index' for assessing the impact of a regulated and industrial area on the Duraton River in Spain. This index combines a measure of the percent difference between the number of species occurring above and below a disturbance point $[(A-B) \times \frac{100}{A}]$, with a measure of the species substitution between the two sites $[(A-C) \times \frac{100}{A}]$, where A = number of species occurring upstream, B = number of species downstream and C = number of species common to both sites, into a single index:

$$EI = \frac{(2A - B - C) \times 50}{A}$$

The values of the ecotoxicological index, or EI, range from 0 (no impact) to 100 (maximum impact).

He reported that the new EI was significantly correlated (but inversely, due to their inverted scales) with H' and Margalef's diversity index. However, the range of values was much greater for the EI than for H', suggesting that the EI may be more sensitive to subtle changes. Interestingly, Margalef's index gave better site discrimination than H', a finding which was also report by Wilhm (1967).

## 3.3.3. ALTERNATIVE APPROACHES

### 3.3.3.1 *Functional Feeding Groups*

According to the River Continuum Concept (Vannote, Minshall, Cummins et al, 1980; Minshall, Cummins, Petersen et al, 1985), drainage networks form a predictable continuum of increasing channel size and associated biological characteristics. Stream morphology, current velocity, substrate composition, temperature and allochthonous vs. autochthonous food sources all interact to influence food availability to invertebrates, and these interactions vary systematically with stream order thereby regulating distribution patterns of invertebrate functional feeding groups (Hawkins & Sedell, 1981). These feeding groups are referred to as the scrapers, collector-filterers, collector-gatherers, predators and shredders (Cummins, 1974). Under unperturbed conditions, headwater areas are normally dominated by shredders and collectors due to the mainly allochthonous energy source, mid-sized streams are autotrophically driven and dominated by scrapers and collector-filterers, while communities in large rivers are mainly composed of collector-gatherers due to the accumulation of fine sediment from upstream (Rabeni, Davies & Gibbs, 1985; Cummins, 1988). Although this general pattern appears to hold worldwide, the exact nature and rate of change will vary from river to river depending on catchment characteristics and waters chemistry at the origin (Omerod & Edwards, 1987) and on the efficiency of retention of sediments and organic matter (Cummins, 1988). Cushing, McIntire, Cummins et al (1983) verified the changes in functional feeding groups predicted by the RCC in a study of 16 streams in Oregon, Idaho, Michigan and Pennsylvania. An important finding was that rates of change varied regionally, such that first order streams in Pennsylvania and Michigan were more like third order than first order streams in Oregon. As the RCC was developed in North America, rivers on other continents may show considerable divergence from the original model (e.g., Winterbourn, Rounick & Cowie, 1981 and Marchant, Metzeling, Graesser et al, 1985, for New Zealand and Australia).

Cummins and others (e.g., Cummins, 1974; Cummins & Klug, 1979; Merritt & Cummins, 1984; Cummins, 1988) developed a functional classification of stream invertebrates over the last 15 years in response to the inadequacies of systematic and trophic analyses, i.e., identification to the species level is still difficult and gut content analyses reveal that all invertebrates are omnivores. According to Cummins (1988), the functional view permits '...clustering of genetically and taxonomically diverse entities into groups, or guilds, which share fundamental properties - such as invertebrates having the same morphological-behavioural mechanisms of food acquisition.' Hawkins & Sedell (1981) point out that such a system reduces the variability associated with taxonomic complexity, allowing trends to be more easily recognized. This approach is also more universally applicable because local taxonomic differences do not seriously affect it. The only drawback to the functional feeding group (FFG) approach is that it is based on nutrient dynamics and can therefore only be used to assess the effects of organic enrichment.

Rabeni, Davies & Gibbs (1985) measured changes in benthic community structure and function in the Penobscot River, Maine, before and after pollution abatement technology had been implemented at all of the major point sources, including pulp mills and municipal sewage treatment plants. Since changes in community structure were consistent with the responses of benthos to organic enrichment, with toxicity playing a lesser role, they felt it appropriate to examine alterations in energy dynamics in order to determine how the communities changed functionally in response to enrichment. Characteristics of the study area, including the stream order, water clarity and general erosional substrate classified it as autotrophically-driven under the RCC scheme. At unperturbed sites, densities of scrapers exceeded those of collector-filterers and -gatherers combined, reflecting this normal autotrophic nature. As water quality degraded, the functional groups responded in a manner predicted by the RCC (e.g., the percent density of scrapers decreased from a high of 45% to less than 1% at the most polluted sites) and the system became heterotrophic. This change occurred in the absence of any longitudinal gradient, suggesting that organic pollution can 'reset' the normal sequence of feeding group shifts and convert an autotrophic system into a heterotrophic state that would normally be found further downstream in a much larger river. As water quality improved due to pollution abatement, the ratio of collector-filterers and scrapers to collector-gatherers increased, indicating that the river was returning to its normal condition. Rabeni, Davies & Gibbs (1985) concluded that the FFG approach is promising because it '...may reflect more ecologically significant attributes of streams and rivers than do structurally based water quality systems.'

### 3.3.3.2 *Reduced Assemblages and Ratio Indices*

Most macroinvertebrate bioassessment techniques are based on the response of the entire benthic community to pollution. However, focusing on a single component of the community (generally an Order or Family) has several attractive benefits. It can simplify the collection, sorting and identification of benthic samples, thereby reducing time and effort. Selective sampling techniques can be employed which provide more precise estimates of the diversity and abundance of the organisms under consideration. The resources saved by limiting the investigation to one group can then be redirected into more intensive or extensive studies and species-level taxonomic identification. The latter is an extremely important consideration. Indices which use species-level identification have better potential for site discrimination (Hilsenhoff, 1988a; Furse, Moss, Wright et al, 1984), since species have more precise environmental requirements than families and species belonging to a single group may have a wide range of susceptibilities to various pollutants (Slooff, 1983). Working at the species level also allows the identification of indicator species for certain types of perturbations. Observational data can then be verified by laboratory toxicity tests on these species, such as those conducted by Chapman, Farrell & Brinkhurst (1982a,b), in order to establish cause/effect links.

For a group to be a candidate for the reduced assemblage approach, it must be capable of representing the response of the community as a whole. Therefore, it must be a prominent group comprising a large proportion of the fauna; it must contain many ecologically different species; and individual species within the group must have a broad range of tolerances for different types of pollution. The groups which have been most successfully exploited are the oligochaetes, chironomids and caddisflies.

Oligochaetes have mainly been used in lakes because of their prominence in soft sediment communities. They have been used both for trophic classification (e.g., Saether, 1979, for nearctic and palaearctic lakes; Howmiller & Scott, 1977, and Lauritsen, Mozley & White, 1985, for the Laurentian Great Lakes) and for indicating different types and degrees of pollution (e.g., Lang & Lang-Dobler, 1979, for eutrophication and heavy metal pollution in Lake Geneva). Saether (1979) felt that oligochaete communities could not provide as distinct a classification system as chironomid communities because their environmental requirements are less restricted. However, Chapman, Farrell & Brinkhurst (1982a,b) have since demonstrated that oligochaete species have a broad range of tolerances to organic pollution and specific chemicals. Recently, there have been several studies on oligochaetes in lotic environments. Smith, Wyskowski, Brooks et al (1990) observed changes in species composition and dominance in oligochaete assemblages in low-order woodland streams in the Adirondack Mountains of New York, in response to the degree of acid pollution. Barton & Metcalfe-Smith (1991) found that variations in oligochaete densities in both the soft sediments and riffles in an agriculturally-polluted watershed in Quebec gave excellent site discrimination, were highly correlated with other indices and were temporally stable. It appears that oligochaetes deserve further consideration for river and stream applications. They may be particularly suitable for assessing sediment quality in large rivers. As Barton (1989) points out, a drawback to using oligochaetes is that they can only be identified if they are sexually mature whereas chironomids, for example, can be identified at any stage of maturity after the first instar.

Chironomids are an extremely diverse group of insects, frequently accounting for 50% of the total species diversity in benthic communities (Merritt & Cummins, 1984). They occur in a wide range of freshwater habitats, have representatives in all trophic groups (predators, herbivores, detritivores) and are important food items for fish (Rosenberg, Danks & Lehmkuhl, 1986). They have been successfully used in lake classification (e.g., Saether, 1979; Johnson, 1989, for Swedish Lakes), although Johnson (1989) commented that many genera and species have wide tolerance ranges and are therefore poor indicators of lake type. He recommended identifying and focusing on indicator species. In a study on the Scioto River basin in Ohio, Rae (1989) found that several groups of common chironomid genera were indicative of certain chemical conditions. Stictochironomus was an indicator of hard, clean, unpolluted water, while Pentaneura, Cricotopus and Tanytarsus were characteristic of sewage pollution (phosphates, low oxygen), Ablabesmyia and Tribelos were associated with soft water and general organic pollution, and Procladius and Dicrotendipes indicated moderate hardness and high agricultural runoff (nitrates and turbidity due to fertilizers and siltation, respectively). Thus, Rae (1989) succeeded in isolating indicator taxa which were extremely tolerant or intolerant of certain types of pollution. He makes a strong case for focusing on indicator taxa rather than studying the entire benthic community, by pointing out that little information is gained by examining the distributions of facultative organisms.

Trichoptera are also very diverse group of insects, occupying a wide variety of habitats and trophic levels. Caddisflies are also well-represented in all functional feeding groups (Cummins & Klug, 1979). Basaguren & Orive (1990) investigated Trichoptera as indicators of water quality in the River Cadagua basin in Spain. They identified 33 taxa from 12 families within the basin and observed a succession of species from the headwaters to the lowland reaches in relation to selected physico-chemical features. This succession was particularly evident in the Family Hydropsychidae among species of the genus Hydropsyche. In organically-polluted river sections, species substitutions occurred which deviated from those in unpolluted sections with the same habitat. In an earlier study on a nearby pristine river, the River Lea, Basaguren & Orive (1989) had identified 47 taxa from 14 families and 32 genera and described, using ordination techniques, zones characterized by different communities of caddisflies. The headwaters of the main river were distinct from those of the tributaries and a downstream sequence was indicated in the main river. The main river had a more diverse fauna, and diversity increased in a downstream direction with increasing river width, water level and substrate diversity. Although this is a characteristic pattern for unpolluted rivers, it is rarely observed today due to the counter-effects of cumulative pollution. Basaguren & Orive (1989) worked largely at the species level, and were able to describe a very detailed continuum of caddisflies in this system which should be extremely useful for assessments of similar systems in the Basque country. Higler & Tolkamp (1983) also found the Hydropsychidae to be useful for classifying watercourses in The Netherlands. Based on historical data, the distributions of ten native species in small streams to large rivers throughout The Netherlands were determined. A succession was again identified, and deviations from the expected pattern for a given area could be attributed to pollution sources. Although several species of Hydropsychidae were found to be tolerant of organic pollution, even these species were eliminated by severe agricultural runoff.

Ratio indices, which express the dominance of one group of organisms over another, have been used in conjunction with both taxonomic and functional feeding group data. They are simple to determine, but can be fairly specific and therefore useful in certain situations. Saether (1979) found that an increasing ratio of oligochaetes to chironomids was a good indicator of eutrophication in lakes. Winner, Boesel & Farrell (1980) assessed the response of aquatic insect communities to heavy metal pollution (Cu, Cr, Zn) in two metal-contaminated Ohio streams. They observed a good correlation between the degree of metal impact and the numerical dominance of chironomids in the insect community. They therefore proposed that the ratio of chironomids to total insects was indicative and should be pursued as an index of heavy metal pollution. Several ratio indices have been incorporated into the U.S. EPA's Rapid Bioassessment Protocols for river assessment (Plafkin, Barbour, Porter et al, 1989), which will be discussed in more detail in the next section. For example, a decrease in the ratio of scrapers to filtering collectors is used to indicate degradation from an abundance of diatoms (which are the primary food source for scrapers) to an abundance of filamentous algae and aquatic mosses (which provide attachment sites for filterers and accumulations of the fine particulate matter on which they feed). Also, communities having an even distribution of organisms among four key insect groups: the mayflies, stoneflies, caddisflies and chironomids, are considered healthy, while those skewed towards a disproportionate number of chironomids indicate environmental stress.

Whitehurst & Lindsey (1990) compared the performance of a Gammarus:Asellus ratio index with traditional biotic indices (Chandler's Score, Extended Biotic Index, BMWP Score) for assessing mild sewage pollution in the River Adur in

Sussex, England. Gammarus is more sensitive to organic pollution than Asellus. Since these taxa are direct niche competitors, a reduction in the abundance of Gammarus due to pollution will result in a corresponding increase in the abundance of Asellus, thereby altering the ratio. Whitehurst & Lindsey (1990) found this ratio index to be more sensitive to mild organic pollution than the biotic indices. The reason for this may be as follows: if an impact is subtle and affects only certain organisms in the community, then the appropriate ratio index should provide good site discrimination. However, a biotic index could mask this response because it includes information on other organisms which are not affected by the impact. In a similar study, Olive, Jackson, Bass et al (1988) assessed the water quality of the upper Cuyahoga River in Ohio using a variety of diversity, biotic and ratio indices. The river is a designated Ohio Scenic River and water quality is generally high. Tested indices included the ratio of scrapers to detritivores and the ratio of amphipods to isopods, both of which decrease with increasing organic pollution. They found that the different indices did not always agree in their classification of certain sites, usually due to the confounding effects of habitat. They caution against using indices of community response interchangeably or relying on only one type of index for assessing the effects of pollution and conclude that ratio indices may be most useful when combined with other indices.

### 3.3.4. RECENT DEVELOPMENTS

The purpose of biomonitoring and assessment is to distinguish degraded sites from healthy ones, identify the cause of the impact, then monitor the response of the system to remediation. The situation is rarely simple. Rather, multiple stresses, which originate from both point and non-point sources and can be chemical or physical in nature, generally act in a cumulative manner on aquatic systems. Traditional approaches to bioassessment have been unsatisfactory for a variety of reasons. Diversity and community comparison indices respond to all types of perturbations and the normal range of environmental factors, such that except in clear-cut situations they cannot be easily interpreted. On the other hand, biotic and saprobic indices only respond to organic pollution and are geographically restricted.

Probably the major obstacle to incorporating bioassessments into water management policies, has been the lack of realistic targets with which to compare index values and to serve as water quality criteria. There has been a general recognition on both sides of the Atlantic of the need for identifying and characterizing reference sites in unimpaired locations in order to define attainable water quality objectives. The United Kingdom and the United States have addressed this problem differently, but both countries have succeeded in incorporating macroinvertebrate community assessments into the water resource management process.

#### 3.3.4.1 *The Multivariate Approach - United Kingdom*

In the early 1980s, The Freshwater Biological Association (FBA) began to computerize macroinvertebrate taxa lists and accompanying environmental data to explore the relationships between environmental parameters and macroinvertebrate communities by multivariate analysis techniques. The goal of the 'River Communities Project' of the FBA was to prepare a biological classification of all running-waters in Great Britain. The work was conducted in three phases, the results of which have appeared in a series of papers published over the last ten years. A good overview is presented by Wright, Armitage, Furse et al (1989).

Armitage, Moss, Wright et al (1983) examined the possibility of predicting 'expected' communities from physical and chemical variables unrelated to pollution, using data collected from 268 sites on 41 unpolluted rivers. Multiple linear regressions were computed using BMWP score or ASPT as the dependent variable and various physical and chemical parameters as the independent variables. The predictive equations for ASPT were superior; as previously noted, ASPT is less sensitive to sampling effort and seasonal change than is the Score. Approximately 70% of the variability was explained using both physical and chemical data, and 60% by physical data alone. Wright, Moss, Armitage et al (1984), in perhaps the 'keystone' paper of the series, used the same data set to develop a classification of running-water sites based on all macroinvertebrate taxa (not just BMWP Scores) and to predict community type from environmental data. Using ordination (detrended correspondence analysis) and a hierarchical clustering technique called TWINSPAN (two-way indicator species analysis), the sites were classified into 16 groupings using species lists generated from three seasons of sampling. Multiple discriminant analysis was then used to correlate the groupings with 28 physical and chemical variables (Table 3.3.5.). Using environmental data, 76.1% of the sites were predicted to the correct grouping. For a further 15.3% of the sites, the correct grouping was the second most probable.

Furse, Moss, Wright et al (1984) tested the influence of season and level of taxonomic identification on the performance of this system. They found that qualitative species-level data led to more reliable classifications and predictions than either quantitative or qualitative family-level data, because of the greater number of taxa and because individual species have more precise environmental requirements than families. The greatest accuracy was achieved combining the results from all three seasons, because species which were absent from one season's data due to life cycle, drought, etc., would have a good chance of being captured in another season.

Wright, Armitage, Furse et al (1985) noted that although the prediction of site groupings is useful for classification, it is only a step towards the prediction of species occurrence at sites with known environmental characteristics. To this end, Moss, Furse, Wright et al (1987) conducted field trials to test the accuracy of classification and prediction of 21 new unpolluted sites using Wright, Moss, Armitage et al's (1984) model, and to determine the probability that a certain species would occur at a given site. They based their analyses on combined seasons' species-level data, and compared the reliability of predictions using suites of 28, 11 and 5 physical and/or chemical variables. They found that reducing the number of environmental variables resulted in very little loss of predictive accuracy. For example, of all taxa predicted as having a >75% chance of occurring at a given site, using suites ranging from 5 physical features to 28 physical and chemical features, 87.0-89.7% actually did occur. Moss, Furse, Wright et al (1987) felt that the major use of their system would be to provide a 'target' community to be used as a standard for a given site when it is unpolluted. The magnitude of the difference between the expected and observed fauna gives a measure of the loss of biological quality due to pollution or other perturbations.

The most recent development in the River Communities Project has been the adaptation of the reference communities, and the predictive equations for group occurrence and probability of species occurrence, into software that can be run on a personal computer (Wright, Armitage, Furse et al, 1989). The software is called RIVPACS (River InVertebrate Prediction and Classification System), and is intended for analysis of combined three-seasons data obtained by standard procedures (Furse, Wright, Armitage et al, 1981; Wright, Moss, Armitage et al, 1984). Initially, the program permitted the prediction of fauna at one or all of the following taxonomic levels: species (qualitative), all families (log. categories of abundance), all families (qualitative), and the more restricted listing of BMWP families (qualitative) using one of four sets of environmental variables (11 or 5 physical and chemical variables or 11 or 5 physical variables only). The system also allowed for determination of the BMWP score or ASPT, which is widely used for rapid site appraisal. The predictive equations were revised after the data base was expanded to 370 sites on 61 rivers, including more small streams and lowland river sites. According to Wright, Armitage, Furse et al (1989), 'RIVPACS offers a site-specific prediction based on environmental features and can therefore set a target from which any loss of biological quality due to environmental stress can be measured by the 'observed/expected' ratio.'

The latest (Phase III) version of RIVPACS is based on 438 sites from 80 rivers, and permits a classification to either 10 or 25 TWINSPAN groups. Each group contains at least 6 reference sites, ensuring reliability of the system. It is more flexible in that data from one, several or all three seasons can be imported, and predictions of ASPT, BMWP score and number of scoring taxa can be obtained. However, for the purpose of standardization, a single group of 11 physical and chemical variables is used (Table 3.3.5.). Testing has now begun in polluted systems to demonstrate the utility of the prediction technique. Wright, Armitage, Furse et al (1988) compared the observed vs. expected occurrence of 75 BMWP families at sites above and below the input of a papermill effluent on a river in south-west England. As illustrated in Fig.3.3.2., the underrepresentation of families at the impact site compared to the control site is clearly evident.

The application of the BMWP score system and RIVPACS to the biological assessment of Spanish rivers is currently being evaluated. Armitage, Pardo, Furse et al (1990) applied the Phase II version of RIVPACS to the assessment of eighteen sites in two Galician rivers in northwest Spain, one receiving minor organic pollution and one unpolluted. The 5 physical and chemical variable option was selected, and the model generated print-outs consisting of lists of predicted families and their probability of capture, and the expected BMWP score and ASPT values for each site. The predicted faunal parameters could then be compared to the observed. An example print-out is shown in Fig.3.3.3. At this particular site, Plecoptera were conspicuously absent and this was attributed to low DO and an unsuitable substratum. Armitage, Pardo, Furse et al (1990) point out that the absence of certain predicted taxa and the presence of taxa with a low probability of occurrence may provide information on the type of environmental stress at a given site. Although this study indicated that RIVPACS could be successfully applied in Spain, the authors cautioned that the system may break down if applied to rivers in driers areas of Spain which have physical and chemical characteristics outside the range of the model. Ultimately, the most efficient application of the system in Spain, or in other countries for that matter, will require the acquisition of a data-based similar to that available in Great Britain. The BMWP system has already been modified slightly to accommodate the richer faunal complement in Spanish rivers. Rodriguez & Wright (1988, in Armitage, Pardo, Furse et al, 1990) compared the original and modified systems in three Basque Rivers in northeast Spain and found that the modified 'Iberian' scores were always higher than the British scores, although the overall trends in quality indicated by the two systems were similar.

Bargos, Mesanza, Basaguren et al (1990) used multivariate techniques to identify macroinvertebrate communities associated with polluted and unpolluted lotic conditions in Biscay (Basque Country, northeast Spain) and to determine the rates of change in rivers with differing types and degrees of pollution. The goal of the program was to create a data base which could be used for biological surveillance and pollution control purposes. They sampled 175 polluted and unpolluted gravel riffle sites on 3rd and 4th order streams in twelve basins. Their approach contrasts with that of the UK researchers in that it focuses on the effects of pollution and standardizes for habitat. They used correspondence analysis as the ordination technique, then correlated ordination scores on axis 1 with BMWP biotic index values with the aim of determining which of the methods best discriminated among sites, especially the less perturbed sites. All basins were similar at their headwaters. However, some changed only slightly along their courses due to light agriculture, some were altered only in the lower reaches, and some showed dramatic changes along their entire courses. Correlations between axis 1 scores and values of the BMWP index were good for rivers which showed a great deal of change along their length, but poor for rivers which showed smaller changes. Bargos, Mesanza, Basaguren et al (1990) concluded that ordination techniques worked best for determining the differences in fauna along the course of relatively unpolluted rivers, while the BMWP actually gave better discrimination among sites in very polluted rivers.

#### 3.3.4.2 *U.S. EPA Approach*

The quality and quantity of water resources in the United States continue to decline despite massive regulatory efforts (Karr, 1991). For example, 602 stream and river segments in the northwest U.S. are water-quality limited due to chemical contamination and 56% of degraded segments nationwide have reduced fishery potential due to pollution. Hunsaker & Carpenter (1990) state that twenty-seven percent of the fish fauna of the U.S. are endangered, threatened or of special concern, over 50% of the mollusc species of the Tennessee River system are either extinct or endangered, and 38 states reported fisheries closures, restrictions or advisories in 1985. Serious and widespread biological impairment is apparent, clearly indicating that existing monitoring and assessment programs, which are based on chemical criteria, are inadequate.

Under the most recent amendment (1987) of the Clean Water Act, the U.S. EPA is required to develop programs to '...evaluate, restore and maintain the chemical, physical and biological integrity of the Nation's waters' (U.S. EPA, 1990). The inclusion of biological integrity as a goal on equal footing with chemical water quality is a major step forward. For a review of the arduous process leading up to this legislation, see Karr (1991). Biological criteria based on structural and functional attributes of resident fish and benthic invertebrate communities are currently being developed and integrated with chemical-specific criteria and whole-effluent toxicity evaluations for a more holistic approach to water management.

Under Section 303 of the Act, individual states will be required to develop narrative biological criteria by 1993, with numerical criteria to follow. Biocriteria will be used to improve water quality standards, identify impairment of beneficial uses, assist in setting program priorities and detect problems which might otherwise be missed or underestimated. Biological criteria are especially suited for the detection of non-point source, cumulative and episodic pollution, as well as physical changes such as habitat deterioration, none of which would be detected using traditional chemical assessment methods. It has also been demonstrated that biocriteria are more sensitive than chemical criteria; in Ohio, an evaluation of instream biota indicated that 36% of biologically impaired sites had not been identified using chemical criteria (Ohio EPA, 1987).

Twenty states now use some form of biological assessment and five (Florida, Arkansas, North Carolina, Maine, Ohio) are currently using biological criteria to define aquatic life use classifications and enforce water quality standards (U.S. EPA, 1990). The application of biocriteria differs somewhat among the various states. Florida has a legal criterion based on macroinvertebrate diversity whereby impact is defined as a reduction of more than 75% below established reference values. Maine uses macroinvertebrate community data to assess attainment or non-attainment of standards for designated water uses (e.g., drinking water supply, fish and wildlife habitat, recreational use, agriculture, industrial use). Oklahoma uses biotic information to assess long-term trends in water quality.

Ohio's program is the furthest advanced, and serves as an example of the successful incorporation of biological criteria in water quality regulations. Biological criteria were developed for Ohio rivers and streams using the 'ecoregion' approach, which has been adopted nation-wide as a framework for defining attainable water quality objectives. An ecoregion is '...a relatively homogeneous area defined by similarity of geography, hydrology, land use, or other ecologically relevant variable.' (U.S. EPA, 1990). Sites within a given ecoregion have similar ecological potentials, and '...attainable quality is then based on assessment of conditions in minimally impacted reference sites that characterize the region' (Hughes & Larsen, 1988). In Ohio, parts of five ecoregions occur, these being the Eastern Corn Belt Plains, the Huron/Erie Lake Plain, the Erie/Ontario Lake Plain, the Western Allegheny Plateau and the Interior Plateau (Ohio EPA, 1987).

The Ohio EPA has used biological criteria based on fish and macroinvertebrates for quantitatively determining attainment/non-attainment of designated aquatic life uses (warmwater habitat, exceptional warmwater habitat, cold water habitat, seasonal salmonid habitat and limited resource waters) since 1980. For invertebrates, early criteria were narrative and based on simple structural measures of diversity, abundance and biomass. These have been recently been replaced by the Invertebrate Community Index (ICI). The ICI is derived from the IBI (Index of Biotic Integrity), which is based on attributes of fish communities and is described in detail in Karr (1991). The ICI consists of ten essentially structural community metrics (Table 3.3.6.), each with four scoring categories of 6, 4, 2 and 0 points which correspond to exceptional, good, fair and poor community condition. The scoring categories were calibrated using data from 232 reference sites in the five ecoregions. Individual metric scores are summed to generate a site ICI score which could theoretically range from 0 to 60. Quantitative sampling is conducted using multiple-plate artificial substrate samplers. However, additional qualitative sampling of the natural fauna from all available habitat types is also conducted. Metrics 1-9 are based on artificial substrate samples, while metric 10 is based on the natural fauna. Attainable values for the ICI were determined for each of the five ecoregions based on median values for the reference sites in these regions. Within each ecoregion, the lower (25%) and upper (75%) percentile values are used to determine attainment or non-attainment of criteria for warmwater habitat and exceptional warmwater habitat, respectively. To determine the performance of the ICI, index values were determined for 431 sites sampled between 1981 and 1984, including 279 which had been classified as 'good', 76 'fair' and 76 'poor' quality sites based on best professional judgement. The results indicated a wide segregation of the good and excellent sites from the fair sites and the poor sites. It was concluded that the ICI provides an '...objective, quantifiable, and standardized means of evaluating biological integrity.' (Ohio EPA, 1987).

The US EPA has also supported the development of a hierarchy of methods for biological monitoring which is referred to as the Rapid Bioassessment Protocols (Plafkin, Barbour, Porter et al, 1989). The document presents a tiered approach to using fish and macroinvertebrate communities in biological assessment. Three protocols are present for invertebrates, each one progressively more intensive and rigorous than the previous one. Protocol III is similar to the ICI, but includes some functional metrics (Table 3.3.7.). It was intended for riffle/run habitats in wadable streams. The original protocols were designed as inexpensive screening tools for determining if a stream is supporting or not supporting a designated aquatic life use. However, they were also found to be useful for discovering and determining the cause and severity of impairment, evaluating the effectiveness of control actions, determining attainability of aquatic life uses and trend monitoring.

### 3.3.5. NEW DIRECTIONS

Because biological systems are complex and the causes of degradation in rivers and streams are multiple, not all measures of community structure or function will be useful under all circumstances. Instead of rejecting the more specific approaches because they cannot be generalized, Karr (1991) favours integration of the various indices and metrics to create a more robust approach to biological monitoring and assessment. What is needed most now is information on the differential responses of benthic communities to the wide range of polluted conditions and physical perturbations which occur. To date, only the response to organic pollution (mainly domestic sewage) has been adequately documented.

It is unlikely that indices developed for assessing organic pollution will be useful for detecting toxic pollution or habitat alterations, because organisms do not respond in the same way to different stresses. For example, Chapman, Farrell & Brinkhurst (1982a) exposed nine species of freshwater oligochaetes to selected chemical pollutants under bioassay conditions. They observed that eutrophic species were, as expected, the most tolerant of sewage and anoxia. However, when specific pulp mill constituents, heavy metals and pentachlorophenol were tested, the relative tolerances of the various species were found to be pollutant-specific. This suggests that assemblages used to indicate organic enrichment are not appropriate for indicating other types of pollution. Similarly, Slooff (1983) compared the relative tolerances of 12 invertebrates from various taxonomic groups to 15 chemicals as well as surface-water concentrates from the Rhine River, The Netherlands. His most interesting finding was that organisms considered to be intolerant of organic pollution in general were sometimes very tolerant to specific toxicants, and vice versa.

Although much of the information is scattered at present, it may soon be possible to describe macroinvertebrate assemblages which are characteristic of broad categories of impacts, such as agricultural activities, heavy metals, acidic pollution and river regulation. Acidic pollution will be discussed as an example.

Several independent studies have collectively provided a profile of the characteristic responses of benthic communities to acid pollution. As previously noted (section 3.3.2.2.), communities affected by toxic or acidic pollution are typically depauperate in comparison with those influenced by organic pollution. Reash, Van Hassel & Wood (1988) described the benthos of a southern Ohio stream draining a coal strip-mining area (pH 4.5). Oligochaetes, Odonata (Pantala, Ishcura), Heteroptera (Trichocorixa, Notonecta), the alderfly Sialis, the diving beetle Laccophilus, the mosquito Anopheles and chironomids, all in relatively low numbers, were found. Sialis and the odonates were absent from the most impacted site. Tomkiewicz & Dunson (1977) investigated a tributary of the West Branch of the Susquehanna River in Pennsylvania, which was also polluted by acid strip mine drainage. The normal pH of the system was 6.0, that of the acid feeder stream was 3.2, and that of the stream below the feeder was

4.7, recovering to 5.0. Only a chironomid, Sialis and the caddisfly Ptilostomis inhabited the acid feeder. Populations of Coleoptera, Ephemeroptera and Trichoptera showed little or no recovery as the acid pollution abated over 3 downstream sites. Heptageniid and baetid mayflies, and pteronarcyid and peltoperlid stoneflies, which were present at the control site, did not reoccur. However, Diptera and leutrid and nemourid stoneflies showed a decided recovery at pH 5.0. Peterson & van Eeckhaute (1990) determined the distributions of 30 stonefly and 100 caddisfly species in relation to stream acidity in the Maritime provinces of Canada. Stonefly taxa most sensitive to acidic conditions were primarily the large Perlidae, but also the Chloroperlidae, Pteronarcidae and Perlodidae. Least sensitive were the Nemouridae, Leutridae and Capniidae. The Trichoptera were more difficult to categorize because several families contained both sensitive and tolerant species. The F. Hydropsychidae in particular had representative species in all categories (sensitive, tolerant, ubiquitous), confirming other reports of the utility of this group for pollution assessment (e.g., Higler & Tolkamp, 1983; Basaguren & Orive, 1990). As per Tomkiewicz & Dunson (1977), Peterson & van Eeckhaute (1990) also found the presence of Ptilostomis to be indicative of acid pollution.

While these descriptive studies are informative, a more precise and quantitative method for defining characteristic communities is needed. Smith, Wyskowski, Brooks et al (1990) employed multiple regression models to determine the response of benthic communities to acidity in low-order woodland streams in the Adirondack Mountains in New York. Benthic organic material and pH explained most of the variability in invertebrate densities and composition of taxa, and a number of Plecoptera, Coleoptera and Trichoptera species which were indicative of low and high pH sites were identified. The findings of this study were generally in agreement with those described earlier, but the data analysis technique permitted a more precise definition of the environmental requirements of the indicator species. For example, Cynigmula (Heptageniidae), Baetis (Baetidae), Elmidae (Coleoptera) and Perlidae occurred at pH 6.2, but not at pH 5.8; mayfly densities increased steadily from pH 4.9 to pH 6.2; Ephemerella was excluded at pH 4.9, but not at pH 5.3; nemourid and leutrid stoneflies were found at sites ranging from pH 4.9-6.2. It is worth noting that the indicator value of the mayfly Baetis with respect to acidity is exactly the opposite of its value as an indicator of organic pollution. This genus features largely in almost all biotic indices due to its extreme tolerance of organic pollution, yet it is very sensitive to low pH. It is abundantly clear that the application of traditional biotic indices to acidic pollution would give completely erroneous assessments.

Reynoldson & Zarull (1989) demonstrated that combined analysis of physical, chemical and biological data could be used to link cause and effect between sediment-associated contaminants and benthic communities in a study on the Detroit River in Canada. The Detroit River connects Lake St. Clair with Lake Erie and is one of the most industrialized regions in the Laurentian Great Lakes. Using clustering techniques applied to benthic community data, they were able to identify five distinct benthic communities mainly based on densities of tubificids, mayflies and amphipods. These 5 groups were correlated individually against 18 chemical and physical characteristics of the sediment in which they live. Site groups were found to be highly correlated with Hg, Cr, Zn, Ni, hexachlorobenzene and phosphorus and also significantly correlated with 5 other factors, including concentrations of Pb, Cd and Cu. They then used multiple discriminant analysis to determine the ability of these physico-chemical factors to predict benthic community structure. When all 11 environmental variables were included, every site was assigned to its correct group. Using this integrated strategy, Reynoldson & Zarull (1989) were able to associate concentrations of contaminants with specific levels of biological impact. The main variables responsible for degradation of the system were identified and could be singled out for remedial action. The method also defined a site-specific target (the cleanest group) to be used as an objective for cleanup. This approach would be particularly suitable for large lowland rivers with significant accumulations of fine sediment, where sediment toxicity is probably more significiant and limiting than water quality.

The incorporation of pollution variables into multivariate analyses to determine the cause of community change would be an improvement to the British bioassessment system. At present, RIVPACS can predict the expected community at a given location in the absence of pollution and provide a measure of the <u>degree</u> of impact (i.e., deviation from normal). However, it cannot determine the type of impact. River regulation may be an exception since it primarily affects variables such as flow rate, substrate characteristics and suspended particulate loadings, a wide range of which are included in the available models. Armitage, Gunn, Furse et al (1987) used the prediction model of Moss, Furse, Wright et al (1987) to examine the effects of regulation on a set of upland reservoirs in Great Britain. They observed more deposit feeders than expected, due to the accumulation of fine sediment as a result of reduced flushing. In contrast, certain mayflies were adversely affected by increased siltation and algal growth, and predatory stoneflies were inhibited by the relatively constant rate of discharge. Moss, Furse, Wright et al (1987) suggest that by simulating likely changes in the physical and chemical characteristics of an unregulated site, it is possible to determine the impact of a proposed regulation strategy in advance of its implementation. They note, however, that the inclusion of more factors such as discharge pattern and epilithon characteristics into the predictive model would greatly improve the accuracy of these predictions.

More studies on relatively simple systems with well-known specific impacts are needed in order to define the characteristic responses of benthic communities to different types of pollution and habitat alterations. These community profiles or 'fingerprints' could then be included in models such as RIVPACS for comparison with observed communities, in order to identify the probable cause of deviation from normal conditions. The information could also be used to develop new biotic indices which respond to polluted conditions other than enrichment and for the development of new specific metrics for inclusion in the ICI or RBPs. Furthermore, characteristic community profiles could be used to identify the 'worst offender' in multiple stress situations.

It will probably not be feasible to develop biotic indices suitable for large-scale application. This task would require the generation of extensive data-bases on species occurances and tolerances, such as those prepared by Sladecek (1973) and Hilsenhoff (1987), which would still be geographically restricted in their application. However, specialized biotic indices could be very useful for managing systems at the local level. For example, Rabeni, Davies & Gibbs (1985) developed a biotic index which was specific to the Penobscot River in Maine. Tolerance values were assigned by direct observation of the species occurring in the system, and best achievable communities were defined as those occurring at the cleanest sites. As they state, the index was '...specific for the watershed, and specific in response to pulp and paper mill effluents and municipal sewage, and therefore sensitive to improvements in water quality.'

It is perhaps too early to suggest improvements to the ICI and RBPs, since they (particularly the latter) have not been extensively tested. However, Karr (1991) maintains that the goals for the future should be to develop suites of metrics that integrate taxa (fish, invertebrates, diatoms, microorganisms) and consider biological responses to stress at all levels of organization (individual, population, community, ecosystem).

Many different approaches to bioassessment have been presented and compared in this chapter. Because the most recent systems have not yet been directly compared, it is difficult to determine whether sophisticated multivariate techniques or suites of simple, yet specific, metrics are the way of the future for large-scale monitoring and assessment applications. Regardless of the technique selected, however, its performance fundamentally depends on the quality of the raw data to which it is applied. Improvements in sampling methodology to increase accuracy, precision and sensitivity must be continually sought. Sampling benthic invertebrate communities is difficult due to their aggregated and highly variable distributions. As a result, sampling precision is one of the major problems limiting quantitative studies of benthos (Clements, Van Hassel, Cherry et al (1989). Voshell, Layton & Hiner (1989) bemoan the fact that the significant advances made by basic aquatic ecologists in field techniques for studying stream macroinvertebrates have largely not been employed by applied biologists for hazard assessment. The reader is referred to their extensive review, which includes many recommendations for improving precision and accuracy in benthic sampling.

Once environmental stresses in rivers and streams have been identified and remedied, recovery may be a slow process. Gammon, Johnson, Mays et al (1983) studied the responses of Indiana streams to improvements in agricultural practices. Over a three-year period, no discernable improvement in the aquatic communities was observed. This implies, they state, that biotic response '...is non-linear in that an increment of improvement in water quality does not necessarily result in an increment of improvement in the aquatic community.' Rather, the pattern suggests that environmental quality must reach a critical level, after which a sudden transformation would occur. Fuchs & Statzner (1990) also investigated the time scales for recovery of river communities after restoration. The impetus behind their study was an announcement in 1988 by German politicians that the Rhine River would be restored sufficiently to support self-sustaining salmon populations by the year 2000. To challenge this claim, they physically restored two small streams draining agricultural lands and followed the recovery progress. Generally, lotic systems are very resilient, but recovery depends on their proximity to sources of potential colonizers or 'inocula'. In the stream which had communities of colonizers both above and below the affected stretch, recovery occurred within one year. In the other stream, which was isolated, significant recovery had not occurred within five years; of 49 species typical of lowland streams in the region, only three had recolonized. Fuchs & Statzner (1990) concluded that the recovery of large Central European rivers such as the Rhine, which are much more isolated than small streams and have lost large numbers of their former species, will require considerably more than twelve years.

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Table 3.3.1. Extended Biotic Index (reprinted with permission from Persoone & De Pauw, 1979).

| | | | Total Number of Groups Present | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Extended Biotic Index | | | 0-1 | 2-5 | 6-10 | 11-15 | 16-20 | 21-25 | 26-30 | 31-35 | 36-40 | 41-45 .. |
| Trent Biotic Index | | | Total number of groups present | | | | | | | | | |
| | | | 0-1 | 2-5 | 6-10 | 11-15 | 16+ | | | | | |
| | | | Biotic Indices | | | | | | | | | |
| Clean | Plecoptera nymphs present | More than 1 species | – | 7 | 8 | 9 | 10 | 11 | 12 | 13 | 14 | 15 |
| Organisms in order of tendency to disappear as degree of pollution increases. | | One species only | – | 6 | 7 | 8 | 9 | 10 | 11 | 12 | 13 | 14 |
| | Ephemeroptera nymphs present (excluding <u>Baetis rhodani</u>) | More than 1 Species | – | 6 | 7 | 8 | 9 | 10 | 11 | 12 | 13 | 14 |
| | | One species only | – | 5 | 6 | 7 | 8 | 9 | 10 | 11 | 12 | 13 |
| | Trichoptera larvae or <u>Baetis rhodani</u> present | More than 1 Species | – | 5 | 6 | 7 | 8 | 9 | 10 | 11 | 12 | 13 |
| | | One species only | 4 | 4 | 5 | 6 | 7 | 8 | 9 | 10 | 11 | 12 |
| | <u>Gammarus</u> present | All above species absent | 3 | 4 | 5 | 6 | 7 | 8 | 9 | 10 | 11 | 12 |
| | <u>Asellus</u> present | All above species absent | 2 | 3 | 4 | 5 | 6 | 7 | 8 | 9 | 10 | 11 |
| | Tubificid worms and/or red Chironomid larvae present | All above species absent | 1 | 2 | 3 | 4 | 5 | 6 | 7 | 8 | 9 | 10 |
| Polluted | All above species absent | Some organisms such as <u>Eristalis tenax</u> not requiring dissolved oxygen may be present | 0 | 1 | 2 | – | – | – | – | – | – | – |

Table 3.3.2 The Modified BMWP Score System (reprinted with permission from Armitage, Moss, Wright et al., 1983).

| Families | Score |
|---|---|
| Siphlonuridae Heptageniidae Leptophlebiidae Ephermerellidae Potamanthidae Ephemeridae Taeniopterygidae Leuctridae Capniidae Perlodidae Perlidae Chloroperlidae Aphelocheiridae Phryganeidae Molannidae Beraeidae Odontoceridae Leptoceridae Goeridae Lepidostomatidae Brachycentridae Sericostomatidae | 10 |
| Astacidae Lestidae Agriidae Gomphidae Cordulegasteridae Aeshnidae Corduliidae Libellulidae Psychomyiidae Philopotamidae | 8 |
| Caenidae Nemouridae Rhyacophilidae Polycentropodidae Limnephilidae | 7 |
| Neritidae Viviparidae Ancylidae Hydroptilidae Unionidae Corophiidae Gammaridae Platycnemididae Coenagriidae | 6 |
| Mesoveliidae Hydrometridae Gerridae Nepidae Naucoridae Notonectidae Pleidae Corixidae Haliplidae Hygrobiidae Dytiscidae Gyrinidae Hydrophilidae Clambidae Helodidae Dryopidae Eliminthidae Chrysomelidae Curculionidae Hydropsychidae Tipulidae Simuliidae Planariidae Dendrocoelidae | 5 |
| Baetidae Sialidae Piscicolidae | 4 |
| Valvatidae Hydrobiidae Lymnaeidae Physidae Planorbidae Sphaeriidae Glossiphoniidae Hirudidae Eropobdellidae Asellidae | 3 |
| Chironomidae | 2 |
| Oligochaeta (whole class) | 1 |

Table 3.3.3. Standard table to determine the Belgian Biotic Index (reprinted with permission from De Pauw & Vanhooren, 1983).

| I<br>Faunistic Groups | II | III<br>0-1 | Total Number of Systematic Units Present<br>2-5 | <br>6-10 | <br>11-15 | <br>16 and more |
|---|---|---|---|---|---|---|
| | | Biotic Index | | | | |
| 1. Plecoptera or Ecdyonuridae (=Heptageniidae) | 1 several S.U.* | - | 7 | 8 | 9 | 10 |
| | 2 only 1 S.U. | 5 | 6 | 7 | 8 | 9 |
| 2. Cased Trichoptera | 1 several S.U. | - | 6 | 7 | 8 | 9 |
| | 2 only 1 S.U. | 5 | 5 | 6 | 7 | 8 |
| 3. Ancylidae or Ephemeroptera except Ecdyonuridae | 1 more than 2 S.U. | - | 5 | 6 | 7 | 8 |
| | 2 2 or < 2 S.U. | 3 | 4 | 5 | 6 | 7 |
| 4. Aphelocheirus or Odonata or Gammaridae or Mollusca (except Sphaeridae) | 0 all S.U. mentioned above are absent | 3 | 4 | 5 | 6 | 7 |
| 5. Asellus or Hirudinea or Sphaeridae or Hemiptera (except Aphelocheirus) | 0 all S.U. mentioned above are absent | 2 | 3 | 4 | 5 | - |
| 6. Tubificidae or Chironomidae of the <u>thummi-plumosus</u> group | 0 all S.U. mentioned above are absent | 1 | 2 | 3 | - | - |
| 7. Eristalinae (=Syrphidae) | 0 all S.U. mentioned above are absent | 0 | 1 | 1 | - | - |

*S.U.: number of systematic units observed of this faunistic group.

Table 3.3.4. Evaluation of water quality using biotic index values of samples collected in March, April, May, September, and early October (reprinted with permission from Hilsenhoff, 1987).

| Biotic Index | Water Quality | 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 |

**Table 3.3.5. Various environmental factors considered in the ordination and classification of running-water sites in Great Britain, and the prediction of community type (after Wright, Moss, Armitage et al, 1984 and others).**

| Physical variables | Chemical variables |
|---|---|
| Distance from source* | pH |
| Slope* | Dissolved oxygen |
| Altitude* | Total oxidized nitrogen* |
| Discharge | Chloride* |
| Mean channel width* | Dissolved orthophosphate |
| Mean channel depth* | Total alkalinity* |
| Surface velocity (max., min., median/mode) | |
| Mean substratum size* | |
| Dominant particle size (max. min., median/mode) | |
| Substratum heterogeneity | |
| % macrophyte cover (max., min., mean) | |
| Air temp. range* | |
| Mean air temp.* | |

* used in latest verion of RIVPACS (Wright, Armitage, Furse et al, 1989).

**Table 3.3.6. Metrics for calculating the Invertebrate Community Index (ICI) (after Ohio EPA, 1987).**

1. Total number of taxa.
2. Total number of mayfly taxa.
3. Total number of caddisfly taxa.
4. Total number of dipteran taxa.
5. Percent mayfly composition.
6. Percent caddisfly composition.
7. Percent Tribe Tanytarsini midge composition.
8. Percent other dipteran and non-insect composition.
9. Percent tolerant organisms (oligochaetes and selected midges, limpets and pond snails).
10. Total number of qualitative EPT (Ephemeroptera, Plecoptera, Trichoptera) taxa.

**Table 3.3.7. Rapid Bioassessment Protocol III (after Plafkin, Barbour, Porter et al, 1989).**

1. Taxon richness.
2. Hilsenhoff's Family Biotic Index.
3. Ratio of scrapers to filtering collectors.
4. Ratio of EPT and chironomid abundances.
5. Percent contribution of dominant family.
6. EPT index.
7. Community similarity index.
8. Ratio of shredders/total organisms.

## FIGURE CAPTIONS

**FIG. 3.3.1.** Development of the most widely used biotic index and score systems. (Modified and reproduced with permission from Metcalfe, 1989).

**FIG. 3.3.2.** Observed and expected BMWP family accretion curves for an upstream control site and an impact site which receives papermill effluent. 'Expected' curve accumulates the probabilities of occurrence of 75 BMWP families using the sequence given in the prediction. 'Observed' curve accumulates the number of families captured, based on the same sequence of taxa. (Reprinted with permission from Wright, Armitage, Furse et al, 1988).

**FIG. 3.3.3.** Example of a print-out showing predicted families with their probability of capture for site 5-Eidos, on the Rio Louro. Taxa actually observed at the site are indicated with a ✓. (Reprinted with permission from Armitage, Pardo, Furse et al, 1990).

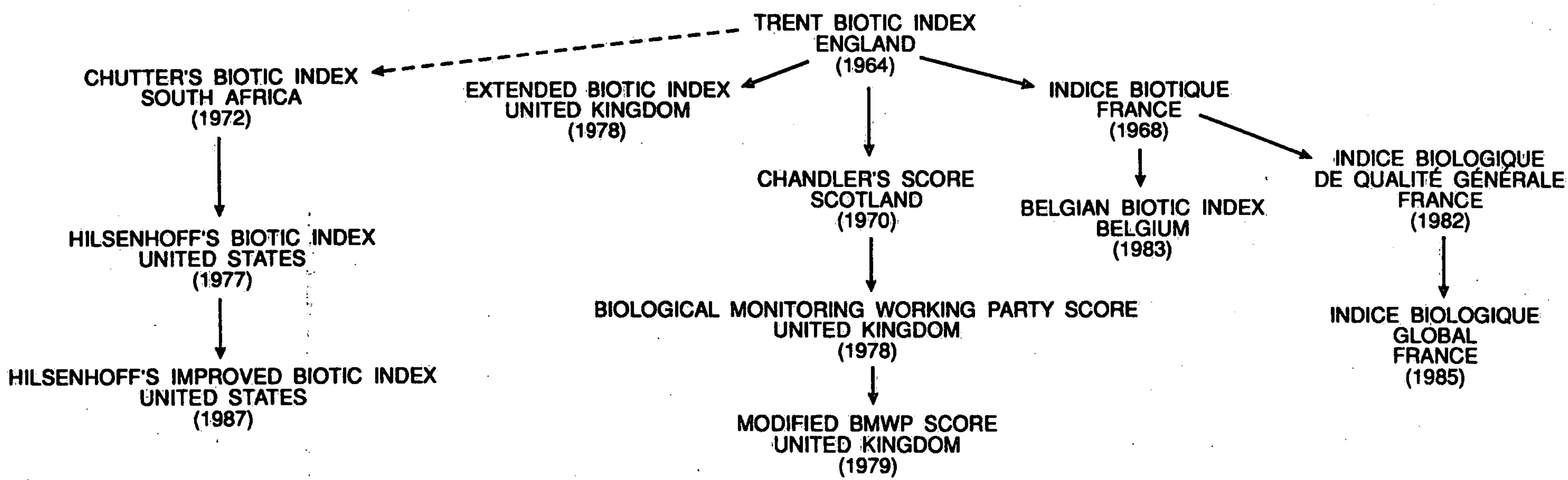
TRENT BIOTIC INDEX
ENGLAND
(1964)
CHUTTER'S BIOTIC INDEX
SOUTH AFRICA
(1972)
EXTENDED BIOTIC INDEX
UNITED KINGDOM
(1978)
INDICE BIOTIQUE
FRANCE
(1968)
CHANDLER'S SCORE
SCOTLAND
(1970)
BELGIAN BIOTIC INDEX
BELGIUM
(1983)
INDICE BIOLOGIQUE
DE QUALITÉ GÉNÉRALE
FRANCE
(1982)
HILSENHOFF'S BIOTIC INDEX
UNITED STATES
(1977)
BIOLOGICAL MONITORING WORKING PARTY SCORE
UNITED KINGDOM
(1978)
INDICE BIOLOGIQUE
GLOBAL
FRANCE
(1985)
HILSENHOFF'S IMPROVED BIOTIC INDEX
UNITED STATES
(1987)
MODIFIED BMWP SCORE
UNITED KINGDOM
(1979)

FIG. 3.3.1

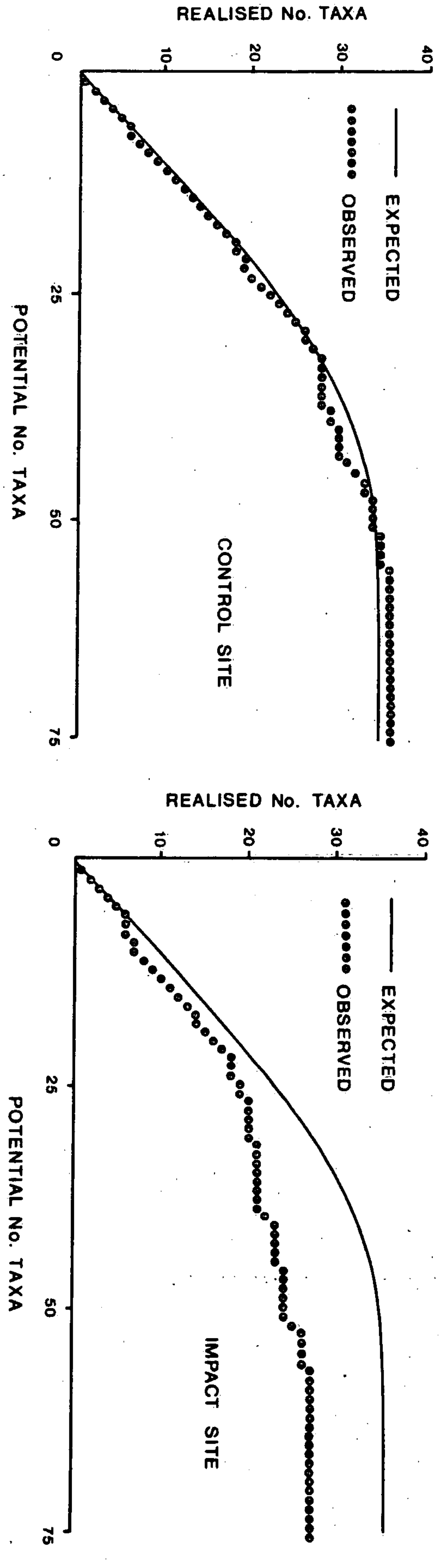
REALISED No. TAXA
0
10
20
30
40
EXPECTED
OBSERVED
25
50
75
POTENTIAL No. TAXA
CONTROL SITE
REALISED No. TAXA
0
10
20
30
40
EXPECTED
OBSERVED
25
50
75
POTENTIAL No. TAXA
IMPACT SITE

FIG. 3.3.2.

River : LOURO Site : 5 EIDOS

Environmental data used :

| | | |
|---|---|---|
| Substratum composition | | |
| Boulders+cobbles (%) | | 0.00 |
| Pebbles+gravel (%) | | 50.00 |
| Sand (%) | | 30.00 |
| Silt+clay (%) | | 20.00 |
| Mean substratum (phi) | 0.57 | |
| Dist. from source (km) | 18.70 | |
| Total oxidised N (ppm) | 1.51 | |
| Alkalinity (ppm CaCO3) | 31.42 | |
| Chloride (ppm) | 28.95 | |

Groups predicted from MDA with 5 physical and chemical variables

| | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|
| 42 | 42.2% | 23 | 28.5% | 41 | 8.8% | 44 | 5.1% | 37 | 5.0% |
| 38 | 3.3% | 40 | 3.1% | | | | | | |

Predicted taxa, in decreasing order of probability

| | | | |
|---|---|---|---|
| 99.7% | Oligochaeta ✓ | 48.9% | Psychomyiidae |
| 99.7% | Chironomidae ✓ | 48.2% | Goeridae |
| 99.7% | Elminthidae ✓ | 27.2% | Planorbidae ✓ |
| 99.3% | Baetidae | 24.9% | Brachycentridae |
| 99.2% | Simuliidae | 23.9% | Aphelocheiridae |
| 98.7% | Ephemerellidae ✓ | 22.2% | Coenagriidae |
| 96.9% | Tipulidae | 21.5% | Odontoceridae |
| 96.7% | Leuctridae | 21.1% | Perlidae |
| 95.4% | Sphaeriidae ✓ | 18.0% | Cordulegasteridae ✓ |
| 94.2% | Gammaridae | 17.1% | Gerridae |
| 93.9% | Hydropsychidae | 12.9% | Beraeidae |
| 93.9% | Limnephilidae | 12.3% | Helodidae |
| 93.1% | Perlodidae | 11.9% | Piscicolidae |
| 90.4% | Dytiscidae | 11.3% | Dryopidae |
| 90.1% | Polycentropodidae ✓ | 8.4% | Physidae ✓ |
| 89.8% | Heptageniidae | 7.3% | Neritidae |
| 89.8% | Hydrobiidae ✓ | 6.2% | Capniidae |
| 87.7% | Erpobdellidae ✓ | 5.9% | Phryganeidae |
| 87.6% | Rhyacophilidae | 5.7% | Hydrometridae |
| 86.2% | Leptoceridae | 4.7% | Nepidae |
| 86.2% | Nemouridae | 4.7% | Siphlonuridae |
| 85.6% | Sericostomatidae | 3.7% | Dendrocoelidae |
| 84.8% | Lymnaeidae | 3.6% | Philopotamidae |
| 84.6% | Gyrinidae | 3.4% | Valvatidae |
| 80.7% | Ancylidae | 2.1% | Astacidae |
| 78.8% | Hydrophilidae | 1.1% | Molannidae |
| 78.4% | Caenidae | 0.7% | Unionidae |
| 77.4% | Glossiphoniidae ✓ | 0.4% | Hirudidae |
| 76.7% | Ephemeridae | 0.3% | Viviparidae |
| 75.3% | Leptophlebiidae ✓ | 0.2% | Notonectidae |
| 73.3% | Hydroptilidae | 0.0% | Platycnemididae |
| 73.0% | Planariidae | 0.0% | Aeshnidae |
| 72.3% | Chloroperlidae | 0.0% | Libellulidae |
| 65.0% | Sialidae | 0.0% | Corophiidae |
| 62.6% | Lepidostomatidae | 0.0% | Pleidae |
| 58.6% | Corixidae ✓ | | |
| 56.7% | Asellidae | | |
| 56.5% | Haliplidae | | |
| 56.3% | Taeniopterygidae | | |
| 51.9% | Agriidae | | |

A) Predicted number of families to 50% = 33.2
B) Observed number of families to 50% = 11
C) Total observed families = 14
D) Observed BMWP score = 66
E) Observed ASPT = 4.714
F) Faunal index 11/33.2
G) BMWP score index 66/234
H) ASPT index 4.714/5.827

Expected taxa P >= ,75 = 26.9. with P >= .5 = 33.2
Predicted BMWP score = 234
Predicted average BMWP score per taxon = 5.827

FIG 3.3.3.

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