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ABSTRACT

This paper reviews the history and development of biological water quality assessment using macroinvertebrates in Europe, and critically evaluates each of the principal approaches used. As the biotic approach incorporates the most highly regarded features of the saprobic and diversity approaches, it has received the most attention in recent years. Most modern biotic index and score systems have evolved from the Trent Biotic Index, through a series of refinements and adaptations (i.e. the Extended Biotic Index, Chandler's Score, Indice Biotique) into the two modern systems. These methods are the Biological Monitoring Working Party Score System, used mainly in Great Britain, and the Belgian Biotic Index Method. The results of these techniques are now influencing policy decisions concerning surface water management in Europe, where macroinvertebrate community assessments are being used as a planning tool for managing water uses, for ambient monitoring, and for evaluating the effectiveness of pollution control measures. New research directions aimed at improving the performance of bioassessment techniques are being explored. These include defining reference communities based on stream typology which can then be used to set water quality objectives, and applying these methods to the assessment of toxic pollution.

RÉSUMÉ

Le présent document étudie l'historique de l'évaluation de la qualité biologique de l'eau à l'aide des macro-invertébrés en Europe, et évalue de façon critique chacune des principales approches utilisées. Au cours des dernières années, on a porté une plus grande attention à l'approche biotique étant donné qu'elle incorpore les meilleures caractéristiques des approches saprophytiques et de diversité. La plupart des systèmes modernes de cotation et d'indices biotiques sont dérivés de l'Indice biotique Trent, à la suite d'une série de raffinements et d'adaptations (c.-à-d. l'Indice biotique extrapolé, le Chandler's score, l'Indice biotique) qui ont donné les deux systèmes modernes. Ces méthodes sont le "Biological Monitoring Working Party Score System" principalement utilisé en Grande-Betagne, et la Méthode de l'indice biotique belge. Les résultats de ces techniques sont désormais pris en considération dans les décisions relatives à la gestion des eaux de surface en Europe où les évaluations de la communauté des macro-invertébrés sont utilisées comme outil de planification pour la gestion des utilisations de l'eau, pour la surveillance ambiante et pour l'évaluation de l'efficacité des mesures de lutte contre la pollution. Il est également question des nouvelles orientations en matière de recherche visant à améliorer la performance des indices biotiques, à définir les communautés témoin basées sur la typologie des souches qui peuvent ensuite être utilisées pour établir des objectifs pour la qualité des eaux, et à appliquer ces techniques à l'évaluation de la pollution toxique.

MANAGEMENT PERSPECTIVE

Benthic macroinvertebrate community structure has long been recognized as a sensitive bioassessment technique for evaluating the ecological degradation occurring in rivers as a result of general organic pollution. The use of this technique in Canada has been sporadic, whereas standardized methods recently developed in Great Britain and Belgium are now influencing policy decisions concerning surface water management in Europe. This report reviews the history and development of biological water quality assessment using macroinvertebrates in Europe, and critically evaluates each of the major approaches and specific indices used. This information provides a knowledge base for the development of methods appropriate for the Canadian environment as well as for other countries. Although all of these techniques were originally developed for the purpose of assessing degradable organic pollution, the potential for their application to toxic pollution is currently being addressed in both Europe and North America. The Contaminants/Pesticides Project of the Rivers Research Branch, NWRI, is contributing to this new area of research in a study to evaluate benthic macroinvertebrate community structure as an indicator of pesticide contamination in the Yamaska River in Quebec. The European perspective presented in this review serves as a point of departure for the selection and development of methods appropriate for this application.

PERSPECTIVE-GESTION

La structure de la communauté des macro-invertébrés benthiques est reconnue depuis longtemps comme une technique sensible de bioessai pour évaluer la dégradation biologique en cours dans les rivières due à la pollution organique généralisée. L'utilisation de cette technique au Canada a été sporadique, tandis que des méthodes normalisées, récemment mises au point en Grande-Bretagne et en Belgique, sont un élément capital dans la prise de décisions politiques au sujet de la gestion des eaux de surface en Europe. Le présent rapport passe en revue l'historique et l'élaboration de l'évaluation de la qualité biologique de l'eau à l'aide des macro-invertébrés en Europe, et analyse de façon critique chacune des principales approches et chacun des indices utilisés. Ces informations fournissent des connaissances de base pour l'élaboration de méthodes propres aux conditions environnementales canadiennes et à celles d'autres pays. Bien que toutes ces techniques aient d'abord été mises au point dans le but d'évaluer la dégradation par la pollution organique, on étudie actuellement, en Europe et en Amérique du Nord, la possibilité de les appliquer à la pollution toxique. Le Projet des contaminants/pesticides de la Direction générale de la recherche sur les rivières (INRE) contribue à ce nouveau domaine de recherche avec une étude visant à évaluer la structure de la communauté des macro-invertébrés benthiques comme indicateur de la contamination par les pesticides de la rivière Yamaska au Québec. La perspective européenne, présentée dans ce rapport sert de point de départ à la sélection et la mise au point de méthodes appropriées à cette application.

INTRODUCTION AND SCOPE OF REVIEW

Ideally, the quality of running waters should be assessed on the basis of physical, chemical and biological characteristics, in order to provide the complete spectrum of information for proper water management. Biological assessments must be included as they offer important advantages over chemical measurements. For example, organisms integrate environmental conditions over long periods of time, whereas chemical data are instantaneous in nature and therefore require large numbers of measurements for an accurate assessment (DePauw & Vanhooren, 1983). Biological studies make important contributions under conditions of toxic, intermittent or mild organic pollution, where changes in water quality are not easily detected by chemical means (Chutter, 1972). According to Pratt & Coler (1976), as pollution control measures continue to reduce gross point source pollution, sensitive biological techniques will be required to detect the more subtle disruptions as well as non-point source pollution. Furthermore, they state: "Criteria restricted to chemical, physical, and bacteriological parameters no longer suffice when the value of water extends beyond its utilization for agricultural, domestic, and industrial ends to include aesthetic, recreational, and ecological dimensions." Finally, biological methods of water quality assessment measure actual effects on biota, whereas physical and chemical methods must eventually be interpreted on a biological basis.

In order to assess water quality on the basis of ecosystem health, it would be best to study the response of the entire aquatic community to stress. As this is obviously impractical, most workers have focussed on a particular sector of the ecosystem, such as periphyton, plankton, macrobenthos or fish. Of these, a clear preference for using macroinvertebrates has emerged for the following reasons: 1) Macroinvertebrates are differentially sensitive to pollutants of various types, and react to them quickly; macroinvertebrate communities are capable of a graded response to a broad spectrum of kinds and degrees of stress. 2) Macroinvertebrates are ubiquitous, abundant and relatively easy to collect. Furthermore, their identification and enumeration is not as tedious and difficult as that for microorganisms and plankton. 3) Benthic invertebrates are relatively sedentary, and are therefore representative of local conditions. 4) These organisms have life spans long enough to provide a record of environmental quality. 5) Finally, macroinvertebrate communites are very heterogeneous, consisting of representatives of several phyla. The probability that at least some of these organisms will react to a particular change in environmental conditions is, therefore, high (Cook, 1976; Pratt & Coler, 1976; Hellawell, 1977; De Pauw & Vanhooren, 1983). Other groups of organisms (fish, phytoplankton, etc.) possess some, but not all, of these important attributes.

This report traces the development of water quality assessment using macroinvertebrates, from its origin to present status, in Europe. New directions in the areas of increasing sophistication and sensitivity of the methods, adapting these techniques or developing new ones to assess chemical or toxic pollution in addition to degradable organic pollution, and using the information for water quality management are also addressed. The main countries involved in this research are identified and their approaches described. It is beyond the scope of this review to consider biochemical and physiological bioassessment (i.e. bioassays), and developments in macroinvertebrate community assessment in countries outside of Europe (mainly the United States).

HISTORY AND DEVELOPMENT OF BIOLOGICAL ASSESSMENT METHODS IN EUROPE

The history of surface water quality assessment based on biological indicators of pollution began before the turn of the century in Germany. Since then, more than 50 different methods have emerged (De Pauw & Vanhooren, 1983). These methods can be divided into two distinct groups. The Saprobic System, which is mainly based on the presence of microorganisms belonging to the plankton and periphyton communities, originated in Europe, while methods focussing on the presence or absence of macroinvertebrate indicators originated in the United States. Both groups of methods have evolved from qualitative to quantitative systems, yielding a long list of saprobic, biotic and diversity indices. By the mid 1970's, most European countries had rejected saprobic and diversity indices, for reasons which will be given later in this report, and had begun to concentrate on biotic index and score systems. Two exceptions are West Germany and The Netherlands, which continue to promote the saprobic-based Biologically

Effective Organic Loading, or BEOL (Woodiwiss, 1980), and Quality-index (Tolkamp, 1985), respectively.

In the first of two major efforts to advance research on biotic index and score systems, the Environment and Consumer Protection Service of the Commission of European Communities began, in 1975, a series of intercalibration studies and seminars which took place in West Germany, the U.K. and Italy. They recognized the need for harmonizing efforts in Europe and for standardizing methods. Apparently, more than 20 different methods were in use at the time and these involved macroinvertebrates, periphyton, and plankton collected from both natural communities and artificial substrates. At the conclusion of the seminars, the EEC adopted the Extended Biotic Index, which is derived from England's Trent Biotic Index, as a reference method (Woodiwiss, 1980).

The country which responded with the greatest effort was Belgium (De Pauw & Vanhooren, 1983), although studies have been conducted in Italy (Ghetti & Bonazzi, 1977) and more recently in Portugal (De Pauw et al., 1986), the latter in cooperation with Belgium. The Extended Biotic Index was rejected and the Indice Biotique, which had been developed in the late 1960's in France, was used as the basis for the development of the Belgian Biotic Index Method.

Partly as a result of the EEC initiative, the Biological Monitoring Working Party (BMWP) was set up in Great Britian in 1976 (ISO, 1979). It reported to the Standing Technical Advisory Committee on Water Quality, a joint Commission of the U.K. Department of the

Environment and the U.K. National Water Council. Its objectives were to coordinate efforts aimed at developing a system which would be suitable for the biological assessment of all rivers in the United Kingdom. The first version of their new system, which was based on the Trent Biotic Index used in England and Chandler's Score from Scotland, was produced in 1978. It was revised a year later and has since been subjected to testing in England (Pinder et al., 1987), England and Wales, (Armitage et al., 1983), and The Netherlands (Tolkamp, 1985). Thus, two major systems, one in Britain and one in Belgium, have developed essentially in parallel. The present state of the art in Europe appears, therefore, to be represented by the BMWP Score (as presented in Armitage et al., 1983) and the Belgian Biotic Index (De Pauw & Vanhooren, 1983).

It should be mentioned that the French have independently developed their own preferred system, the Indice Biologique Global, which was recently recommended as a standardized method to be used throughout France (AFNOR, 1985). It is a slight modification of the Indice Biologique de Qualité Genéralé of Verneaux et al. (1982) which, in turn, was derived from the Indice Biotique of Tuffery and Verneaux (1968). The most important biotic index and score systems and their chronological sequence are illustrated in Figure 1.

APPROACHES TO BIOLOGICAL ASSESSMENT

There are three principal approaches to biological assessment which utilize taxonomic and pollution tolerance data, these being the saprobic, diversity and biotic approaches. As previously mentioned, the most recent emphasis has been on developing the potential of the biotic scores and indices. Diversity indices are out of favour and although the saprobic system is still used, this is usually for comparative purposes. It appears to fare better in terms of sensitivity than the diversity indices, and in at least two cases (Tolkamp, 1985; Woodiwiss, 1980) refined versions compared very favourably with a number of biotic indices.

Each of the three approaches will now be described in detail, along with the major advantages and disadvantages of each. In the section on the biotic approach, all of the major systems will be presented in chronological order to demonstrate how the present systems have evolved.

Saprobic Approach

The term "saprobia" means the dependence of an organism on decomposing organic substances as a food source (Persoone & De Pauw, 1979). The early research efforts of two German scientists, R. 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 (Table 1) based upon such pollution-related parameters as BOD, bacterial counts, and concentrations of DO and H_2S. 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 (1973a) 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. The five major criticisms of saprobic systems, according to Persoone and De Pauw (1979), are:

1) Taxonomy is either not far enough advanced or is too controversial, especially for microorganisms.
2) The system implies more knowledge than actually exists - pollution tolerance limits for organisms are very subjective. As noted by Slooff (1983), pollution tolerances of species are based on ecological observations <u>not</u> confirmed by experimental studies.
3) Intensive sampling is required.
4) Species lists and saprobic values will not be applicable to other geographic locations.
5) The system cannot confidently be applied to other types of pollution, i.e. inorganic and organic, degradable and non-degradable toxic pollution, and radioactivity.

Chutter (1972) believes the saprobic system to be of limited usefulness because of its rigidity and because all indicator organisms, including those associated with severely polluted water, occur in natural waters. Jones et al. (1981) consider the system to be insufficient because each taxon is considered as a separate entity, and no information on the community as a whole is provided.

Two saprobic-based systems are currently in use. The BEOL (Biologically Effective Organic Loading) method was introduced in the mid-1950's by H. Knöpp of West Germany (Persoone & De Pauw, 1979) as a means of reducing large amounts of data on saprobic values of indicator species into a simple mathematical equation. The method is as follows:

- all species found are listed, and their relative frequency is estimated on a scale grading from 1 (one single individual) to 7 (abundant).
- the indicator value of each species is determined using classes 1 (oligosaprobic) through 4 (polysaprobic) from Table 1.
- the frequencies for all species in each class are then totalled, and the BEOL (expressed as a percentage) is calculated from the class totals using the formula:

$$\text{BEOL} = \frac{\epsilon \text{ (total frequencies for classes 3 \& 4)}}{\epsilon \text{ (total frequencies for classes 1, 2, 3 \& 4)}} \times 100\%$$

In the Third EEC Comparative Study on the Rivers Parma, Stirone and Po in Italy in 1978, the BEOL and the EBI (Extended Biotic Index) agreed well, having an almost linear relationship over most of their ranges (Woodiwiss, 1980).

The Quality-index, or K-index, was developed by Gardeniers and Tolkamp in 1976 in The Netherlands (Woodiwiss, 1980). Indicator species are arranged in five groups, each of which is assigned a pollution factor (Table 2). 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:

$$K_{135} = (\% \text{ Erist.} + \text{Chir. gr}) \times 1 + (\% \text{ Hirud. gr}) \times 3 + (\% \text{ Gam.} + \text{Calopt. gr}) \times 5.$$

$$K_{12345} = (\% \text{ Erist. gr}) \times 1 + (\% \text{ Chir. gr}) \times 2 + (\% \text{ Hirud. gr}) \times 3 + (\% \text{ Gam. gr}) \times 4 + (\% \text{ Calopt. gr}) \times 5.$$

The K_{135} index was found to be more sensitive to smaller changes in the middle range of the pollution scale than the saprobic index and a number of biotic indices, when tested on rivers and streams in the southern region of The Netherlands (Tolkamp, 1985). This is of benefit for the early detection of improvement or deterioration of water quality.

Diversity Approach

Diversity indices are mathematical expressions which use three components of community structure, namely richness (number of species present), evenness (uniformity in the distribution of individuals among the species) and abundance (total number of organisms present), to describe the response of a community to the quality of its environment. The assumption underlying the diversity approach is that undisturbed environments will be characterized by a high diversity or richness, an even distribution of individuals among the species, and moderate to high counts of individuals (Ghetti & Bonazzi, 1977; Mason et al., 1985). In environments stressed by degradable organic wastes, the community generally responds with a decrease in diversity as sensitive organisms are lost, an increase in abundance of the tolerant organisms which now have an enriched food source, and, of course, a decrease in evenness. In contrast, the response to non-degradable toxic or acidic pollution is a decrease in both diversity and abundance as the sensitive organisms are eliminated and there is no additional food source for the remaining tolerant forms. In fact, the natural food sources may be more limited than normal, and there may be sublethal stresses on the survivors which affect productivity (Mason et al., 1985; Persoone & De Pauw, 1979).

The most widely used measure of diversity is the Shannon-Wiener formula (after Wilhm & Dorris, 1968), but two others are included here for comparison: Simpson (after Pinder et al., 1987) and Margalef (after Wilhm & Dorris, 1968). These are given below:

Shannon-Wiener $$\bar{d} = -\Sigma \frac{Ni}{N} \log_2 \frac{Ni}{N}$$

Simpson $$\bar{d} = 1 - \frac{Ni\ (Ni - 1)}{N\ (N - 1)}$$

Margalef $$\bar{d} = \frac{S - 1}{\log_e N}$$

where: $\bar{d}$ = diversity

N = total number of individuals of all species collected

Ni = number of individuals belonging to the i^{th} species

S = # species

The higher the value of $\bar{d}$, the greater the diversity and, supposedly, the cleaner the environment. The Margalef formula differs from the other two in that it does not contain an evenness component.

Diversity indices are considered to have the following advantages:

1) They are strictly quantitative, dimensionless, and lend themselves to statistical analysis (Cook, 1976).
2) Most are relatively independent of sample size (Wilhm & Dorris, 1968; Pinder et al., 1987).
3) Unlike the saprobic index, no assumptions are made as to the relative tolerances of individual species, which may be very subjective (Pinder et al., 1987).
4) They can be applied equally well to measures of biomass which are less labour intensive than counts of individuals (Mason et al., 1985).

Many criticisms have been made against diversity indices. The major ones are presented below:

1) Values will vary considerably depending upon the equation used to calculate them, the method of sample collection, the extent of

identification (species diversity being greater than generic diversity), and the location and nature of the river being studied (Pratt & Coler, 1976).

2) While standards have been set for the interpretation of the index values (such as Wilhm & Dorris, 1968), the scales are not universally applicable. For example, not all undisturbed communities have inherently high diversity; therefore, it is not always possible to correlate certain values with ecological damage (Jones et al., 1981). Furthermore, wide variations in values have been reported for unpolluted conditions (Cook, 1976).

3) In the calculation of diversity indices, individual species are reduced to anonymous numbers which disregard their pollution tolerances. It is as important to know which species are present as it is to know how many. Diversity index values cannot tell us if the community is composed of pollution-tolerant or -intolerant species (Cook, 1976). Furthermore, diversity indices are ratios of two variables and, as such, have serious statistical implications. When variables are compounded into ratios, the variances of the numerator and denominator are ignored and the resulting ratio will have greater variability than either of the two variables from which it was derived (Green, 1979).

4) The response of a community to increasing pollution is not necessarily linear. In fact, there is evidence that moderate pollution can cause an increase in abundance without excluding species, with the result that the index value actually goes up (Cook, 1976).

5) Diversity indices have generally been applied to the extremes of the pollution scale, i.e. pristine vs. downstream of an effluent discharge. Not enough testing has been conducted in the middle range which represents most ambient waters of concern. Jones et al. (1981) compared the Shannon-Wiener diversity index with a biotic index in their abilities to differentiate among Missouri Ozark streams varying from clean to slightly enriched. They found that the streams could be ranked in order according to their pollution status using the biotic index, but the diversity index designated all sites as unstressed.

Several modifications to diversity indices have been proposed which may improve their performance. Pratt & Coler (1976) recommend employing a "hierarchial" diversity where, for example, the diversity of 30 species belonging to one insect order is considered to be less than if the 30 species were distributed among three orders. This refinement would begin to qualify the differences among the species. Hughes (1978) studied the influence of taxonomic level of identification on the value of Shannon's Index and found that index values increased steadily from the order to the species level. He warned that details of the taxonomic levels used for all organisms should accompany diversity values, otherwise the information generated is worthless from a comparative viewpoint. Perkins (1983) recommends using community comparison indices which are also strictly quantitative, but which differ in approach from diversity indices as follows: if one community is more polluted than another, then a comparison of the two communities should not be similar with respect to species composition. Their similarity should decrease as pollution in one increases, all else being equal. The author found that community indices could be used successfully to rank five streams in Texas according to their degree of impact due to contamination with copper. In contrast, Shannon's Index was too insensitive, giving false negatives for the two streams receiving the lowest concentrations of copper.

It is apparent from the above discussions that the "ideal" index should be one which combines a quantitative measure of species diversity (diversity approach) with qualitative information on the ecological sensitivities of individual species (saprobic approach) into a single numerical expression which can be statistically analyzed. This is, in essence, the biotic approach.

Biotic Approach

The biotic approach to biological assessment, as defined by Tolkamp (1985), 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 indices and biotic scores have been developed for use in Europe, the most important of which are shown in Figure 1. Most were initially developed for use in a particular country and have since been modified to allow for wider application either within the

United Kingdom or in continental Europe. The difference between a biotic score and a biotic index is that the former includes a measure of abundance. Score systems demand more effort and are less practical to use, but they may provide more information. Each of the major index or score systems are described below.

Trent Biotic Index and Extended Biotic Index

The Trent Biotic Index 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 either 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 (Table 3). Clean streams are given an index value of 10, and this value decreases with increasing pollution. Pinder et al. (1987), in their comparison of macroinvertebrate surveillance methods for assessing the water quality of a chalk stream in England, suggested that the Trent Biotic Index (TBI) would not be sensitive enough to detect any but major differences in water quality among streams because of its restricted range in values. In fact, much earlier criticisms of the same nature led to the TBI being extended to cover a range of water qualities from 0 to 15 instead of 0 to 10, and resulted in the Extended Biotic Index (Table 4). 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 originally designed specifically 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 for a particular site is determined by identifying the organisms present, determining the abundance classification for each group from the abundance chart which accompanies the score, then selecting the appropriate points for that group (Tables 5 and 6). The points for all groups are added to give a site score. Note that points scored increase with increasing abundance for sensitive groups and decrease with increasing abundance for tolerant groups, and also that the value of the site score is unending. Criticisms levelled at this system are that it is too complicated, requiring detailed taxonomic identification plus enumeration, that the level of taxonomic identification is not uniform (species for some, generic for others), that it is applicable to upland rivers only, and finally that it is geographically restricted because it makes use of indicator species identified to a high level. Some of these criticisms are unfounded. Cook (1976) found the score to be widely applicable both geographically and with studies on rivers in different area of England where lowland and upland sections yielded similar scores. In her own work on a stream in New York, she found that modifications aimed at adapting the score to local conditions (Table 7) did not significantly improve the performance of the index. Also in her study, Cook found the Chandler Score to be superior to the Shannon-Wiener Diversity Index in grading sections of a mildly polluted stream according to water quality. The Chandler Score was well correlated with variables normally associated with pollution, such as BOD and coliform count, while the diversity index tended to classify mildly polluted sites as unpolluted.

Biological Monitoring Working Party Score System

The Biological Monitoring Working Party (BMWP), set up in 1976, used the Chandler Score System as the starting point for developing a standardized biotic system for assessing the biological quality of rivers in England, Scotland and Wales (ISO, 1979). Members of the Party recommended the following changes: all groups would be identified to the Family level in order to have taxonomic uniformity, less variability due to misidentification of species, and a wider application; the abundance factor would be eliminated because it is too time-consuming and has only a small effect on score value. In these ways, they simplified the Chandler System. However, they also added a refinement in the form of a zonation factor which called for separate score systems for the eroding and depositing zones of rivers. They recommended sampling eroding zones wherever possible as the collection methods are simpler and the biota more sensitive to alterations in water quality. The earliest version of the BMWP Score System is presented in Table 8. The method for scoring is as follows: select the appropriate scale (depositing or eroding), list the families present, ascribe the score for each Family, then add the scores together to arrive at a site score. Score values for individual families reflect their pollution tolerance based on current knowledge of distribution and abundance. The system was modified the next year into that shown in Table 9. The changes were: the eroding zone scale was eliminated and the depositing zone scale was applied generally, the values ranged from 1-10 instead of 1-100, and the range was extended at the lower end to include two additional groups. No further modifications appear to have been made to the present day.

At this point, the "average score per taxon" (ASPT) computation, which has frequently been applied to both Chandler and BMWP scores, should be introduced. It simply refers to dividing the total score by the number of scoring taxa. The ASPT version is often preferred because it limits all values to within a scale of 1-10, because the value of the index is independent of the number of taxa counted and also, according to Pinder et al. (1987), ASPT is relatively independent of sample size, sampling technique and season, and therefore has many of the characteristics desirable in an index of water quality. Pinder et al. (1987), in their study on a chalk stream in England, compared the performances of four diversity indices and three biotic indices or scores (the Trent, Chandler and NWC (NWC = BMWP), and ASPT versions of the latter two) at a single site where sampling methods, substrate and level of identification were being tested. In their ranking of individual sets of samples, all methods were highly correlated with the exception of one of the diversity indices and the Chandler-ASPT. The reason for the failure of the latter is not known. The NWC-ASPT was recommended by the authors on the grounds that it was little affected by sample size, was simple to calculate, and required a limited degree of taxonomic expertise. Armitage et al. (1983) evaluated the performance of the BMWP-score and -ASPT at 268 sites on 41 rivers in Great Britain. These sites were unpolluted, but ranged from high altitude, low alkalinity, coarse substrate to low altitude, high alkalinity, fine substrate. They were attempting to identify natural communities which were representative of certain physical and chemical characteristics unrelated to pollution, in order to produce a biological classification of running-water sites in Great Britain. The results showed a steady decline in ASPT values from the upland to lowland range of environmental features, while score values varied. When predicting ASPT and score values from multiple linear regression equations using physical and chemical parameters as the independent variables, the predictive equations for ASPT were much better (explaining 65% of the variance as opposed to 22% for score values). Where deviations between predicted and observed values were observed, these sites proved, upon closer examination, to be influenced by sewage effluents. Predictive equations worked least well for sites with very high or very low scores, and the authors are investigating other environmental factors or mathematical techniques to improve this. Taxonomic identification was to the Family level, and they reviewed a number of studies which addressed the problem of selecting the most efficient level of taxonomic identification. The consensus was that, for routine monitoring, Family level was adequate. Tolkamp (1985) evaluated saprobic, diversity and biotic indices in an effort to classify streams in the province of Limburg, the Netherlands. The purpose was to set up reference communities for various stream types, so that disturbances could be adequately identified. Unlike Armitage et al. (1983), he found very little difference between the BMWP-score and -ASPT methods. Although the Chandler-ASPT appeared to be as sensitive, he rejected it because he felt the indicator list would have to be adapted for widespread use.

Indice Biotique

The Indice Biotique (IB) was developed by Tuffery and Verneaux (1968) for use in France. It is derived from the Trent Biotic Index (TBI), but differs in the following ways:

- The IB contains a greater number of specific indicator taxa than the TBI (Persoone & De Pauw, 1979).
- The two indices give different weights to some indicator groups. In the TBI, Nais is kept separate from the Naididae and Baetis is

kept separate from the other Ephemeroptera (Ghetti & Bonazzi, 1977), while in the IB, the Ecdyonuridae are kept separate from the other Ephemeroptera, and Trichoptera are divided into species with and without cases due to the greater sensitivities of the former groups in both instances (De Pauw & Vanhooren, 1983).

- A systematic unit represented by a single individual is not considered in the calculation of the IB, because its occurrence may be accidental; also, this helps to reduce fluctuations (Persoone & De Pauw, 1979).
- Finally, the TBI specifies that a handnet be used for sampling all habitats; all organisms collected are then combined. The IB calls for sampling two habitats - flowing, with a Surber sampler and quiet, with a grab. The samples are kept separate and two indices are calculated, I_{lo} and I_{le} (lotic and lentic, respectively). Both are used in the interpretation of the index (Persoone & De Pauw, 1979).

The index is calculated using Table 10 and the level of identification of systematic units is according to that in Table 11. Table 10 has both rows and columns, representing faunistic groups and systematic units, respectively. The seven faunistic groups in Column I 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 systematic units present (Column II). The row chosen from the table is the one corresponding to the presence of the most sensitive faunistic group in the sample. The vertical column chosen from Column

III depends on the number of systematic units present in the sample. The intersection of the appropriate row and column give the index value for the site. According to Persoone & De Pauw (1979), if the index is lower than 5, the site is considered polluted. If the I_{1o} and I_{1e} differ by more than 2 units, and if one is less than 5, then the site is considered polluted.

Indice Biologique de Qualité Générale and Indice Biologique Global

The Indice Biologique de Qualité Générale (IBG) was introduced by Verneaux et al. (1982) as a new method for assessing the quality of rivers and streams in France. It was considered to be more precise and sensitive than the Indice Biotique, from which it was derived, because of improvements in the sampling protocol and the use of a greater number of indicator taxa. The IBG method requires that eight different habitats, which are precisely defined on the basis of substrate and velocity conditions, be sampled at each site to be assessed. If a particular substrate is found at a variety of velocities, then the velocity with which it is most often associated is sampled. If the substrate is very homogeneous and the eight specified habitats can not be found, then eight areas differing as much as possible in velocity are to be sampled. Organisms are identified to a convenient level - usually to Family, but in some cases (e.g., Oligochaeta) to Class. A total of 135 systematic units are considered. Thirty-eight of these were believed to have specific indicator value, and were assigned to ten faunistic groups representing a range of pollution tolerances. The Indice Biologique Global, which was recommended as a standardized method for use through France a few years later (AFNOR, 1985), differed slightly from the original method in that nine faunistic groups were used and some of the indicator taxa were assigned to different groups (Table 12). In both cases, the index is calculated as follows:

- the total number of taxa present, (including those represented by only one individual), are determined using the list of 135 systematic units. This constitutes a measure of community diversity, which is divided into 12 categories (columns).

- the faunistic groups are ranked in order of increasing tolerance to pollution (rows). The row chosen from the table is the one corresponding to the most sensitive group which is represented by at least three individuals. The intersection of the appropriate row and column gives the index value for the sample.

There are several studies which compare the performance of the French indices (IB, IBG) with other biological assessment techniques. Tolkamp (1985) applied the TBI, IB, IBG, saprobic index, Quality-index, and Chandler's Score and -ASPT to samples from the

River Geul in The Netherlands, which has a well known pollution status confirmed by water chemistry. The TBI, IB and IBG were all rejected because they failed to rank the sites correctly. The reason for their poor performance is believed to be the so-called "insect-effect". These indices rely heavily on the presence/absence of a few insect orders in the higher quality classes without accounting for the great differences in pollution tolerance among individual species within these orders. Ghetti & Bonazzi (1977) compared various diversity indices, the saprobic index, the TBI and the IB in an effort to select the best method for assessing the water quality of the Torrente Parma in Italy. They found very little difference between the TBI and the IB and, in fact, the TBI, IB and saprobic index were all so highly correlated that they proposed a conversion scale for them. In contrast, the diversity indices were poorly correlated with the others.

Casellato et al (1980) applied the IB to benthic community data from the River Brenta in Northern Italy, and were unsatisfied with its performance. They were unable to sample, or even locate, lentic sites in the upland stretches of the river and lotic sites in the lowland stretches. As a result only one index, either the I_{lo} or the I_{le}, could be calculated for a given site. Index values for the upland sites, including those downstream of major sewage outfalls, were considerably higher than those for the lowland sites which received no additional pollution. Lower index values at the latter sites were unrelated to pollution; rather, they were attributable to the absence of high-scoring Plecoptera, Trichoptera and Ephemeroptera which do not colonize the muddy substrates characteristic of lowland rivers. Casellato et al (1980) suggested that a more meaningful upland-lowland comparison might be possible if separate tables, each based on the best achievable communities under lotic and lentic conditions, were used to calculate the I_{lo} and the I_{le}. The Italian investigators also criticized the subjectivity of the taxonomic level of identification recommended by the IB for several groups (Table 11), stating that index values based on different levels of organization cannot be compared.

The Belgian Biotic Index Method

The Belgian Biotic Index Method (BBI) combines the Indice Biotique from France with the sampling method used for the Trent Biotic Index in the U.K. As previously described, the sampling method for the TBI involves the use of a handnet to sample all available habitats, while the IB uses surbers and grabs and calculates a separate index for each habitat. In preliminary studies, the authors (De Pauw & Vanhooren, 1983) determined that samples collected by handnet contain a greater diversity of organisms because the handnet explores a larger array of habitats. They also introduced several minor modifications to the IB:

- Nematodes are excluded entirely, as most will not be caught in a 300-500µ mesh handnet.
- The Chironomidae are divided into two systematic units, those belonging to the thummi-plumosus group and those not.
- Certain levels of identification for Trichoptera, Mollusca, Diptera, Platyhelminthes and Hirudinea were set at the Family Level to avoid erroneous interpretations due to misidentification.

The BBI table is identical to the IB (Table 10); identification limits are given in Table 13. Calculation is as described for the IB. In order to visualize the biotic indices obtained for all rivers, streams and brooks in Belgium, index values are grouped into five classes which are assigned different colours, and these are then mapped.

This index and method have been declared highly successful (De Pauw & Vanhooren, 1983). Identification keys were standardized as were the specifications of the handnet, and intercalibration exercises with respect to sampling and identification gave satisfactory results. Results were reproducible over long periods of time in areas where no changes in pollution status occurred, and seasonal changes were minor. They advised that a single sampling in either early summer or fall was sufficient for a proper assessment. They identified several areas for further research, including: identification of reference communities for the different types of watercourses based on areas not yet polluted, the development of alternative sampling methods where the use of a handnet is impossible (i.e. large, deep rivers; canals) and the preparation of an index with indicator species suitable for coastal and brackish waters. The second problem, sampling methods for large rivers, has been addressed in a recent paper (De Pauw et al, 1986), where various types of artificial substrate samplers were tested in mountain streams in Portugal and lowland streams in Belgium. They concluded that there are still major drawbacks to artificial samplers, namely, the long periods of time needed to obtain representative samples, the necessity to visit each site twice for placement and retrieval, and unforeseen losses.

THE USE OF MACROINVERTEBRATE BIOASSESSMENT DATA IN WATER MANAGEMENT

Ultimately, the value of bioassessment techniques will be judged on the basis of their successful application to water management. The most valuable tools will be those which are efficient and cost-effective, accurate in their assessment and predictive abilities, precise (reproducible), sensitive to minor changes in water quality (in order to be useful under ambient conditions), and relevant. As discussed below, the application of macroinvertebrate bioassessment data for this purpose has differed somewhat between Belgium and Great Britain/The Netherlands.

Belgium

The impetus behind the water quality surveillance program in Belgium has largely been concerns about sanitation. Almost all watercourses in Belgium are exceedingly polluted (Dirk Roels, pers. comm.); therefore, human health takes precedence over ecosystem health at the present time. Indeed, the National Institute for Hygiene and Epidemiology in Brussels sponsors the biomonitoring program which 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 a 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 fast-flowing shallow streams and slow-running deep lowland rivers and canals. By 1985, over 30,000 km of watercourses had been surveyed and mapped using the BBI, and at present a four-year program is underway to assess all watercourses in Flanders, including the smallest brooks. When compared with chemical water quality indices, the BBI method has been shown to accurately reflect the general ecological degradation occurring in cases of organic as well as toxic pollution. The information generated by the BBI is now used extensively for policy decisions concerning surface-water management. The method has recently been shown to be applicable in other countries, including Spain, Algeria, Luxemburg, Portugal and Canada (De Pauw et al, 1986).

Great Britain and The Netherlands

The approach taken by Great Britain and The Netherlands relates to the terms of reference of the Biological Monitoring Working Party, which 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, 1979). To satisfy the first objective, their earliest exercises involved sampling representative river reaches rather than the vicinities of specific discharges. The monitoring of ambient conditions was emphasized; therefore, the results of successive surveys were scrutinized on a percentage change basis. Armitage et al. (1983) also addressed this objective in their assessment of 268 sites on 41 rivers in England and Wales. All sites selected were of "good" or "fairly good" quality, as the main purpose of the study was to identify reference communities and the specific environmental factors which influence them. These communities would then be used to prepare a biological classification of all running-waters in Great Britain. The same approach has been taken in The Netherlands (Tolkamp, 1985), where the goals of the Limburg Water Pollution Control Authority are essentially the same as those of the BMWP. They are stated as follows:

1) To assess water quality on a biological basis in relation to water pollution by organic wastes, and to evaluate the effectiveness of enforced measures to reduce pollution.

2) To define reference communities which can then be used as a basis for ecological conservation, for managing water uses, and for identifying the "best achievable" communities for each type of running water, i.e. a water quality objective.

Although it is not stated in these papers, it is apparent that water management policies in these countries are currently based on chemical assessment methods. Biological assessment techniques require further research, testing and standardization before they can influence the policy-making process.

The need for developing better assessment methods which will better protect aquatic ecosystems from damage is in some cases quite urgent. For example, chalk streams in England have, until now, been little affected by pollution. However, their increasing use for domestic water supplies and fish farming, along with the economic value of their trout and salmon fisheries, may be incompatible with their increased use for the disposal of domestic and agricultural effluents. Efforts are being made (Pinder et al., 1987) to develop early warning systems for ecological damage in chalk streams.

NEW RESEARCH DIRECTIONS

Research into bioassessment techniques using macroinvertebrate communities is continuing in three main areas. Changes in methodology to improve accuracy, precision, and sensitivity, the definition of reference communities to aid in data interpretation, and the potential for applying these techniques to chemical or toxic pollution are all being addressed.

In an effort to improve sampling methodology, De Pauw & Vanhooren (1983) and Pinder et al. (1987) have begun to evaluate the effects of habitat, sampling technique, season, level of taxonomic identification, and replication on the performance of biotic indices. De Pauw et al (1986) have begun to develop a standard procedure with artificial substrates for use with the Belgian Biotic Index in situations where logistics prevent the use of the recommended handnet. Higler & Tolkamp (1983) are proposing the use of single species of the Family Hydropsychidae (net-spinning caddisfly larvae) as bioindicators for characterizing running waters in The Netherlands. This would greatly reduce the amount of time and effort normally expended on assessments involving the entire community. Results to date suggest that the scheme is suitable for small, fast-flowing streams, but that data on more members of the community are required for the classification of lowland streams.

One of the major obstacles to incorporating macroinvertebrate community assessment data into water management policies is the identification of reference communities with which monitored communities can be compared. The "best achievable" community which can occur under a particular set of physical, chemical, geological and geographical conditions must be known before the data on polluted sites can be interpreted in a meaningful way. Research into defining such reference communities has been recommended by Belgium (De Pauw &

Vanhooren, 1983) and The Netherlands (Tolkamp, 1985). As stated by Tolkamp, "It must be kept in mind that an optimal biological assessment can be achieved only through regional adaptations of methods, reflecting both biogeographical and biotypological differences between streams."

In the early 1980's, Great Britian began to computerize taxa lists and accompanying environmental data to explore the relationship between environmental parameters and macroinvertebrate communities by multivariate analysis techniques. Armitage et al. (1983) examined the possibility of predicting "expected" communities from physical and chemical data unrelated to pollution. Multiple linear regressions were computed using BMWP-ASPT data as the dependent variable and various physical and chemical parameters (Table 14) as the independent variables. The prediction of scores was relatively good, with 70% of the variability explained using both physical and chemical data and 60% explained using physical data alone. Wright et al. (1984) used multivariate techniques to classify unpolluted running-water sites and to predict community type from environmental data. Sites were classified into 16 groupings based on species lists generated from three seasons of sampling at 268 sites on 41 rivers. Multiple discriminant analysis was then used to correlate the groupings with 28 physical and chemical variables (Table 14). 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 one based on environmental criteria. The authors suggested that predictive accuracy could probably be improved by adding more environmental features to the analysis. Furse 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. Accuracy was also improved by combining the results from all three seasons, because species which were absent from one season's data due to life cycle, flood, drought, etc., would have a good chance of being captured in another season. However, the magnitude of the advantage of using species-level identification and data from all three seasons was not substantial.

Wright 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 et al. (1987) conducted field trials to test the accuracy of classification and prediction of 21 new unpolluted sites using Wright 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 numbers of environmental variables resulted in very little loss of predictive accuracy. For example, 87.0-89.7% of taxa predicted as having a $\geq$75% chance of occurring at a given site, using suites ranging from 5 physical features to 28 physical and chemical features, actually did occur. Moss 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 then gives a measure of the loss of biological quality due to pollution or other perturbations.

Unfortunately, the practical value of this system in the management of running-water ecosystems is presently unknown, as no studies on polluted sites have been conducted to date. However, Armitage et al. (1987) employed the refined system of Moss et al. (1987) to predict macroinvertebrate response to flow regulation below a set of upland reservoirs in Great Britain. Using only 5 physical and chemical factors and combined seasons' family-level data, they were able to identify families which responded to the conditions associated with flow regulation. For example, more deposit feeders were observed than predicted. This was believed due to the accumulation of fine sediment as a result of reduced flushing. From this example, we can evision the potential for applying the predictive model to other perturbations, including pollution.

In a somewhat different approach, De Pauw and Roels (1988) investigated the relationship between the Belgian Biotic Index and various common chemical indicators of pollution, using data from a wide variety of both 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.

All of the biological systems presented thus far were developed for the purpose of assessing degradable organic pollution, not for toxic or acidic pollution or radioactivity (Sladecek, 1973b). Nevertheless, there are some indications that they may be applicable to chemical pollution as well. The observation by Mason et al. (1985) that low macroinvertebrate diversity and high abundance are characteristic of the presence of organic wastes, while low diversity and low abundance are indicative of toxic or acidic pollution, could be used to distinguish between these two general types of pollution in routine monitoring. In a Canadian study, Chapman et al. (1982) tested the tolerances of 12 species of oligochaetes, which are important components of biotic indices, to a variety of pollutants and environmental factors. Their results both supported the use of oligochaete species to classify organically polluted waters and suggested that particular species assemblages could be used as indicators of specific chemical pollutants. Winner et al (1980) observed that the numerical dominance of chironomids in the aquatic insect communities of two streams heavily impacted with copper, zinc and chromium was highly correlated with the degree of contamination. They recommended that the ratio of chironomids to total insects be pursued as an indicator of metal pollution.

Despite these positive findings, there are still many problems to be resolved before the macroinvertebrate community approach can be confidently applied to the assessment of toxic pollution. According to Slooff (1983), laboratory toxicity data on the differences in susceptibilities of various invertebrate species to specific chemicals are required for a proper assessment. To provide such information, he compared the relative tolerances (lethal response) 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. Furthermore, species belonging to the same group often showed as much variability in susceptibility as species from different groups. The author suggests that the reliability of biological systems based on indicator species to classify surface waters into different degrees of chemical pollution will be poor until cause-effect relationships can be established. In a step towards this goal, Williams et al. (1984) reported a study in which toxicological and ecological data on the response of aquatic invertebrates to zinc agreed well. Two species, _Gammarus pulex_ and _Baetis rhodani_, were excluded from a river where zinc concentrations in the water exceeded those found to be lethal to these organisms after short term exposures in the laboratory.

LaPoint et al. (1984) found that community structure was not necessarily a good indicator of stressed conditions in American streams where metal concentrations exceeded aquatic life criteria. Confounding factors such as substrate composition, flow regime and nutrient concentrations seemed to overwhelm the effects of metals on resident fauna. They recommended the use of on-site bioassays in conjunction with biomonitoring to resolve this problem. In another Canadian study, Dance & Hynes (1980) investigated the effects of agricultural practices on stream macroinvertebrate communities. They found that, although this form of land use results in the input of pesticides and chemical fertilizers, an increase in suspended sediment loadings and temperature, and a decrease in allochthonous food sources for shredders and detritivores, the single most important factor affecting the invertebrate fauna was intermittent flow conditions due to changes in drainage patterns. In this situation, separating out the contribution made by one factor, such as the presence of a particular agricultural pesticide, would be very difficult.

SUMMARY AND CONCLUSIONS

This report describes the history and development of biological water quality assessment using macroinvertebrates in Europe. There are three principal approaches to assessing the response of macroinvertebrate communities to pollution, namely the saprobic, diversity and biotic approaches. The saprobic approach, which originated in Europe, is based on the pollution tolerances of indicator species from all components of the aquatic fauna, but mainly bacteria, algae, protozoans and rotifers. The diversity approach, which originated in the United States, uses three components of community structure - richness, evenness and abundance, to describe community response to environmental quality. Saprobic indices call for extensive sampling and the identification of all organisms to the species level. Their species lists and saprobic values are generally site-specific, and pollution tolerances are highly subjective. Diversity indices are quantitative measurements which can be analyzed statistically; however, index values vary considerably with sampling method, season and level of taxonomic identification. Classification of water quality on this basis is difficult, because wide variations in values have been reported for unpolluted conditions. Pollution tolerances, however subjective, are not considered in the diversity approach; therefore these indices cannot tell us whether a community is composed of pollution-tolerant or pollution-sensitive species.

The biotic approach incorporates desirable features of the saprobic and diversity approaches, combining a quantitative measure of species diversity with qualitative information on the ecological sensitivities of individual taxa into a single numerical expression which can be statistically analyzed. This approach is favoured in Europe at present. Of nine studies which directly compared the performance of a diversity or saprobic index, or both, with a biotic index, none advised using a diversity index for pollution assessment and only three recommended the use of a refined saprobic index - usually in conjunction with a biotic index. The reasons generally given for rejection in both cases were lack of sensitivity and/or falsely classifying stressed communities as unstressed.

The Trent Biotic Index, which was devised in the mid-1960s for use in England, appears to be the origin for most modern biotic indices and score systems. Through a series of refinements and adaptations, the two most recent systems have emerged. These systems are the Biological Monitoring Working Party Score System, applied mainly in the U.K., and the Belgian Biotic Index Method used in Belgium. As they have never been directly compared with one another, a single "best" system cannot be selected at present.

During the last five to ten years, there has been a renewal of interest and an increase in research efforts into macroinvertebrate bioassessment techniques in Europe. At the beginning of the 1980s, more than 20 different methods involving periphyton, plankton and macroinvertebrates were in use, and the need for harmonizing efforts and standardizing methods was recognized. Countries which have been the most active are Belgium, England and Wales, and The Netherlands, and more recently Italy and Portugal. The objectives behind these renewed efforts include: to gain insight into the self-purification process in rivers and streams, to assist water managers in selecting sites for water-purification plants and surface water reservoirs, to provide an overview of the conditions of running-waters in Europe, and to evaluate the effectiveness of enforced measures to reduce pollution. All countries have recognized the need for defining reference communities based on chemical, physical, geological and geographical parameters unrelated to pollution, as a first step towards identifying the "best achievable" communities for each type of running-water which can then be used as a water quality objective. This task would require the application of sophisticated multivariate techniques. Whether such an undertaking would be cost-effective remains to be seen, as it would involve collecting biological, physical and chemical data from large numbers of unpolluted sites, then coding, computering and subjecting it to statistical analyses. Major opportunities for modelling in this area of research are anticipated, both to arrive at "expected" communities for a particular set of environmental conditions and to predict the consequences of inaction, further disturbances and ameliorative activities.

In addition to the continuous modifications aimed at improving the efficiency, accuracy, precision, sensitivity and predictive ability of biotic systems in assessing degradable organic pollution, the potential for applying these techniques to toxic and acidic pollution is also being addressed. This is a logical step forward, but also a major challenge. For example, organisms which are intolerant of organic pollution in general are sometimes very tolerant to specific toxicants, and vice versa. Also, chemical pollution problems are usually very complex as we are seldom dealing with single toxicants in the environment. As a result, it is extremely difficult to separate the effects of one chemical from another and in turn to separate these effects from the influence of other environmental factors either related or unrelated to pollution.

There have been suggestions that macroinvertebrate community structure is not sensitive enough to distinguish among various types and degrees of pollution. It is possible, however, that it is our method of detecting the response rather than the response itself which is insensitive. Theoretically, small changes in water quality should lead to alterations in the structure of a community which is already in a delicate balance. Even a seemingly minor change will upset this balance if it results in a survival advantage of one species over another. Minor differences in susceptibilities among species could translate into competitive disadvantages, decreased resistance to predation, lowered reproductive success, etc., for the more sensitive organisms. Even if it is not possible to identify representative communities for specific types of chemical pollution, community structure is still a useful tool for water management. Regardless of the cause, it identifies areas where ecosystem health is poor and where, therefore, investigative efforts should be focussed.

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Table 1. Classification of water quality according to the Saprobic System (adapted and reprinted with permission from Sladecek, 1973a)

Name	Rating	Description of Status With Reference to Organic Pollution
Katharobic	-1	Drinking water, purest water
Xenosaprobic	0	Pure water, unpolluted, biologically poor zone
Oligosaprobic	1	Clean, Healthy, not adversely affected by pollution, game fish zone
Beta-mesosaprobic	2	Mild to moderate pollution, lower recovery zone, fertile zone
Alpha-mesosaprobic	3	Distinctly polluted, upper recovery zone, active decomposition, intermediate bacterial decomposition zone
Polysaprobic	4	Heavily polluted, degradation, active bacterial decomposition
Isosaprobic	5	Active decomposition, degradation, sewage
Metasaprobic	6	Septic, decaying sewage, H_2S zone
Hypersaprobic	7	Strong decomposition, putrefaction, industrial liquors
Ultrasaprobic	8	Abiotic, lag-phase prior to degradation, lifeless liquors

TABLE 2. Species groups and pollution factors used in calculating the Quality-index (Woodiwiss, 1980).

	GROUP	Pollution Factor	
		K_{135}	K_{12345}
Eristalis-group	Eristalis (Diptera, Syrphidae) Culicidae s.s. (Diptera) Spercheus emarginatus ? (Coleoptera)	1	1
Chironomus-group	Cf. Tubificidae (Oligochaeta) Chironomus (Diptera, Chironomidae) Psectrotanypus varius (Diptera, Chironomidae)	1	2
Hirudinea-group	Volwassen Erpobdella octoculata (Hirudinea) Asellus aquaticus (Isopoda) Helobdella stagnalis (Hirudinea) Glossiphonia: 2 spec. (Hirudinea) Juveniele Erpobdella octoculata (Hirudinea) Cf. Lumbriculus variegatus (Oligochaeta) Macropelopia nebulosa (Diptera, Chironomidae) Conchapelopia melanops (Diptera, Chironomidae)	3	3
Gammarus-group	Prodiamesa olivacea (Diptera, Chironomidae) Asellus meridianus (Isopoda) Gammaraus pulex (Amphipoda) Corixidae-larven (Heteroptera) Dicranota (Diptera, Limnobiidae) Nemoura cinerea (Plecoptera) Odonata?, behalve Calopteryx Cloeon (Ephemeroptera) Baetis (Ephemeroptera) Anabolia nervosa (Trichoptera) Limnephilus ? rhombicus (Trichoptera) Phryganea ? (Trichoptera) Athripsodes ? (Trichoptera) Polycentropodidae (Trichoptera) Volwassen Hydracarina ? Laccophilus ? (Coleoptera) Gyrinus-larven (Coleoptera) Gobio gobio (Pisces) Nemacheilus barbatulus (Pisces) Procloeon pseudorufulum (Ephemeroptera)	5	4
Calopteryx-group	Cottus gobio ? (Pisces) Deronectes (Coleoptera) Helmidae (Coleoptera) Orectochilus (Coleoptera) Calopteryx (Odonata) Heptagenia ? (Ephemeroptera) Ephemera ? (Ephemeroptera) Halesus (Trichoptera) ? Potamophylax (Trichoptera) Goera pilosa (Trichoptera) Atherix ? (Diptera, Rhagionidae) Lampetra planeri (Cyclostomata)	5	5

Table 3. Trent Biotic Index (reprinted with permission from Chandler, 1970).

KEY GROUPS FOR THE TRENT RIVER BOARD BIOTIC INDEX

A "group" consists of:	Common Name
Each family of Trichoptera larvae	Caddis flies
Each family of Coleoptera larvae and adults	Beetles
Each family of Diptera (except blood worms)	True flies
Each family of Annelida Oligochaeta	Worms
Each genus of Plecoptera nymphs	Stoneflies
Each genus of Ephemeroptera nymphs	May-flies
Each species of Annelida Hirudinea	Leeches
Each species of Mollusca	Snails, limpets, etc.
Each species of Crustacea	Shrimps, water hoglice
Each species of Megaloptera larvae	Alder flies
Chironomus thummi	Blood worms

TRENT RIVER BOARD BIOTIC INDEX

			Total Number of Groups Present				
			0-1	2-5	6-10	11-15	16+
			Biotic Index				
Clean	_Plecoptera_ Nymphs Present	More than 1 Species	-	VII	VIII	VIII	IX
		One species only	-	VI	VII	VIII	IX
Organisms in order of tendency to disappear as degree of pollution increases.	Ephemeroptera nymphs present (excl. _Baetis_)	More than 1 Species	-	VI	VII	VIII	IX
		One species only	-	V	VI	VII	VIII
	Trichoptera larvae or _Baetis_ present	More than 1 Species	-	V	VI	VII	VIII
		One species only	IV	IV	V	VI	VII
	Gammarus present	All above species absent	III	IV	V	VI	VII
	Asellus present	All above species absent	III	IV	V	VI	VII
	Tubificid worms and/or red _Chironomid_ larvae present	All above species absent	I	II	III	IV	-
Polluted	All above species absent	Some organisms such as Eristalis tenax not requiring dissolved oxygen may be present	-	I	II	-	-

Table 4. 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
		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
Organisms in order of tendency to disappear as degree of pollution increases.	<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 5. Chandler's Biotic Index by the "Score System" (reprinted with permission from Chandler, 1970).

Groups Present in the Sample	Increasing Abundance				
	Present	Few	Common	Abundant	Very Abundant
	Points Scored				
Each species of Planaria alpina Taeniopterygidae, Perlidae, Isoperlidae, Chloroperlidae	90	94	98	99	100
Each species of Leuctridae, Capniidae and Nemouridae (exd. Amphinemura)	84	89	94	97	98
Each species of Ephemeroptera (excluding Baetis)	79	84	90	94	97
Each species of cased Caddis, Megaloptera	75	80	86	91	94
Each species of Ancylus	70	75	82	87	91
Each species of Rhyacophila (Trichoptera)	65	70	77	83	88
Genera of Dicranota, Limnophora	60	65	72	78	84
Genera of Simulium	56	61	67	73	75
Genera of Coleoptera, Nematoda	51	55	61	66	72
Genera of Amphinemura (Plecoptera)	47	50	54	58	63
Genera of Baetis (Ephemeroptera)	44	46	48	50	52
Genera of Gammarus	40	40	40	40	40
Each species of uncased Caddis (excluding Rhyacophila)	38	36	35	33	31
Each species of Tricladida (excluding P. alpma)	35	33	31	29	25
Genera of Hydracarina	32	30	28	25	21
Each species of Mollusca (excluding Ancylus)	30	28	25	22	18
Each species of Chironomids (excluding Ch. riparius)	28	25	21	18	15
Each species of Glossiphonia	26	23	20	16	13
Each species of Asellus	25	22	18	14	10
Each species of leech, (excluding Glossiphonia, Haemopis)	24	20	16	12	8
Each species of Haemopis	23	19	15	10	7
Each species of Tubifex sp.	22	18	13	12	9
Each species of Chironomus riparius	21	17	12	7	4
Each species of Nais	20	16	10	6	2
Each species of air breathing species	19	15	9	5	1
No animal life	0	0	0	0	0

TABLE 6. Levels of abundance for calculating Chandler's Score (reprinted with permission from Chandler, 1970).

AQUATIC MACRO-INVERTEBRATES

Level	Number of Individuals per 5-min Sample	Remarks
Present	1 to 2	May be drift fauna from upstream
Few	3 to 10	Probably indigenous, but rare.
Common	11 to 50	
Abundant	51 to 100	
Very Abundant	more than 100	

Table 7. Biotic Index for Chandler's Score as adapted for Ox Creek, New York (reprinted with permission from Cook, 1976).

Groups Present in the Sample	Increasing Abundance				
	Present	Few	Common	Abundant	Very Abundant
			Points Scored		
Each species of Perlidae, Perlodidae, Chloroperlidae, Taeniopteryginae	90	94	98	99	100
Each species of Nemouridae (excl. Taeniopteryginae), Astacidae	84	89	94	97	98
Each species of Ephemeroptera (excl. _Baetis_)	79	84	90	94	97
Each species of Cased caddis, Megaloptera, _Agrion_ (Zygoptera)	75	80	86	91	94
Each species of _Ancylus_	70	75	82	87	91
Each species of _Rhyacophila_ (Trichoptera)	65	70	77	83	88
Genera of Dicranota, Limnophora, Tipulidae	60	65	72	78	84
Genera of _Simulium_, _Pristina_	56	61	67	73	75
Genera of Coleoptera (excl. _Stenelmis_), Nematoda	51	55	61	66	72
Genera of Ceratopogonidae	47	50	54	58	63
Genera of _Baetis_ (Ephemeroptera), Anisoptera, _Stenelmis_ (Coleoptera)	44	46	48	50	52
Genera of _Gammarus_	40	40	40	40	40
Each species of Uncased caddis (excl. _Rhyacophila_), Zygoptera (excl. _Agrion_)	38	36	35	33	31
Each species of Tricladida	35	33	31	29	25
Genera of Hydracarina	32	30	28	25	21
Each species of Mollusca (excl. _Ancylus_)	30	28	25	22	18
Each species of Chironomids (excl. _C. riparius_)	28	25	21	18	15
Each species of _Glossiphonia_	26	23	20	16	13
Each species of _Asellus_	25	22	18	14	10
Each species of leech, (excl. _Glossiphonia_, _Haemopis_)	24	20	16	12	8
Each species of _Haemopis_	23	19	15	10	7
Each species of _Tubifex_ _sp_.	22	18	13	12	9
Each species of _Nais_	20	16	10	6	2
Each species of air breathing species	19	15	9	5	1
No animal life			0		

Table 8. The Biological Monitoring Working Party Score System (ISO, 1979).

Families	Score	
	Eroding Zone	Depositing Zone
Siphlonuridae Heptagenidae Leptophlebiidae Ephermerellidae Potamanthidae Ephemeridae Taeniopterygidae Leuctridae Capniidae Perlodidae Perlidae Chloroperlidae Aphelocheiridae Phryganeidae Molannidae Beraeidae Odontoceridae Leptoceridae Goeridae Lepidostomatidae Brachycentridae Sericostomatidae	80	100
Astacidae Lestidae Agriidae Gomphidae Cordulegasteridae Aeshnidae Corduliidae Libellulidae Psychomyiidae Philopotamidae	60	80
Caenidae Nemouridae Rhyacophilidae Polycentropodidae Limnephilidae	50	70
Neritidae Viviparidae Ancylidae Hydroptilidae Unionidae Corophiidae Gammaridae Platycnemididae Coenagriidae	40	40
Mesoveliidae Hydrometridae Gerridae Nepidae Naucoridae Notonectidae Pleidae Corixidae Haliplidae Hygrobiidae Dytiscidae Gyrinidae Hydrophilidae Clambidae Helodidae Dryopidae Eliminthidae Chrysomelidae Curculionidae Hydropsychidae Tipulidae Simuliidae Planariidae Dendrocoelidae	30	30
Baetidae Sialidae Piscicolidae	20	20
Valvatidae Hydrobiidae Lymnaeidae Physidae Planorbidae Sphaeriidae Glossiphoniidae Hirudidae Eropobdellidae Asellidae	10	10

Table 9. The Modified BMWP Score System (reprinted with permission from Armitage 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 10. Standard table to determine the Indice Biotique and the Belgian Biotic Index (reprinted with permission from De Pauw & Vanhooren, 1983).

I Faunistic Groups	II	III Total Number of Systematic Units Present				
		0-1	2-5	6-10	11-15	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 thummi-plumosus 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 11. Limits of taxonomic identification of systematic units for the Indice Biotique (reprinted with permission from Persoone & De Pauw, 1979).

Orders	Limits
Plecoptera	genus
Trichoptera	family or genus depending on the cases
Ephemeroptera	genus
Odonata	genus
Coleoptera	family
Mollusca	genus or species depending on the cases
Crustacea	family
Megaloptera	genus
Hemiptera	genus
Diptera	family, sub-family or tribe depending on the cases
Planaridae	genus or species depending on the cases
Hirudinea	genus or species depending on the cases
Oligochaeta	family
Nematoda	presence
Hydracari	presence

TABLE 12. Indices Biologique Global (reprinted with permission from AFNOR, 1985).

Faunistic Groups	Total Diversity	12	11	10	9	8	7	6	5	4	3	2	1
		≥40	39–37	36–34	33–30	29–26	25–22	21–18	17–14	13–10	9–7	6–4	3–1
Chloroperlidae Perlidae Perlodidae Taeniopterygidae	9	20	19	18	17	16	15	14	13	12	11	10	9
Capniidae Brachycentridae Odontoceridae Philopotamidae	8	19	18	17	16	15	14	13	12	11	10	9	8
Leuctridae Glossosomatidae Goeridae Leptophlebiidae	7	18	17	16	15	14	13	12	11	10	9	8	7
Nemouridae Lepidostomatidae Sericostomatidae Ephemeridae Heptageniidae	6	17	16	15	14	13	12	11	10	9	8	7	6
Hydroptilidae Limnephilidae Rhyacophilidae Polymitarcidae Potamanthidae	5	16	15	14	13	12	11	10	9	8	7	6	5
Leptoceridae Polycentropodidae Psychomyidae Ephemerellidae	4	15	14	13	12	11	10	9	8	7	6	5	4
Hydropsychidae Baetidae Caenidae Triclades	3				11	10	9	8	7	6	5	4	3
Elmidae Odonates Gammaridae Mollusques	2					9	8	7	6	5	4	3	2
Chironomidae Asellidae Achètes Oligochètes	1						7	6	5	4	3	2	1

TABLE 13. Limits of taxonomic identification for the Belgian Biotic Index (reprinted with permission from De Pauw & Vanhooren, 1983).

Taxonomic Group	Determination Level of Systematic Units
Platyhelminthes	genus
Oligochaeta	family
Hirudinea	genus
Mollusca	genus
Crustacea	family
Plecoptera	genus
Ephemeroptera	genus
Trichoptera	family
Odonata	genus
Megaloptera	genus
Hemiptera	genus
Coleoptera	family
Diptera	family
	Chironomidae _thummi-plumosus_
	Chironomidae non-_thummi_-_plumosus_
Hydracarina	presence

TABLE 14. Various environmental variables considered in the ordination and classification of running-water sites in Great Britain and the prediction of community type (Armitage et al., 1983; Wright et al., 1984; Furse et al., 1984; Wright et al., 1985; Moss et al., 1987).

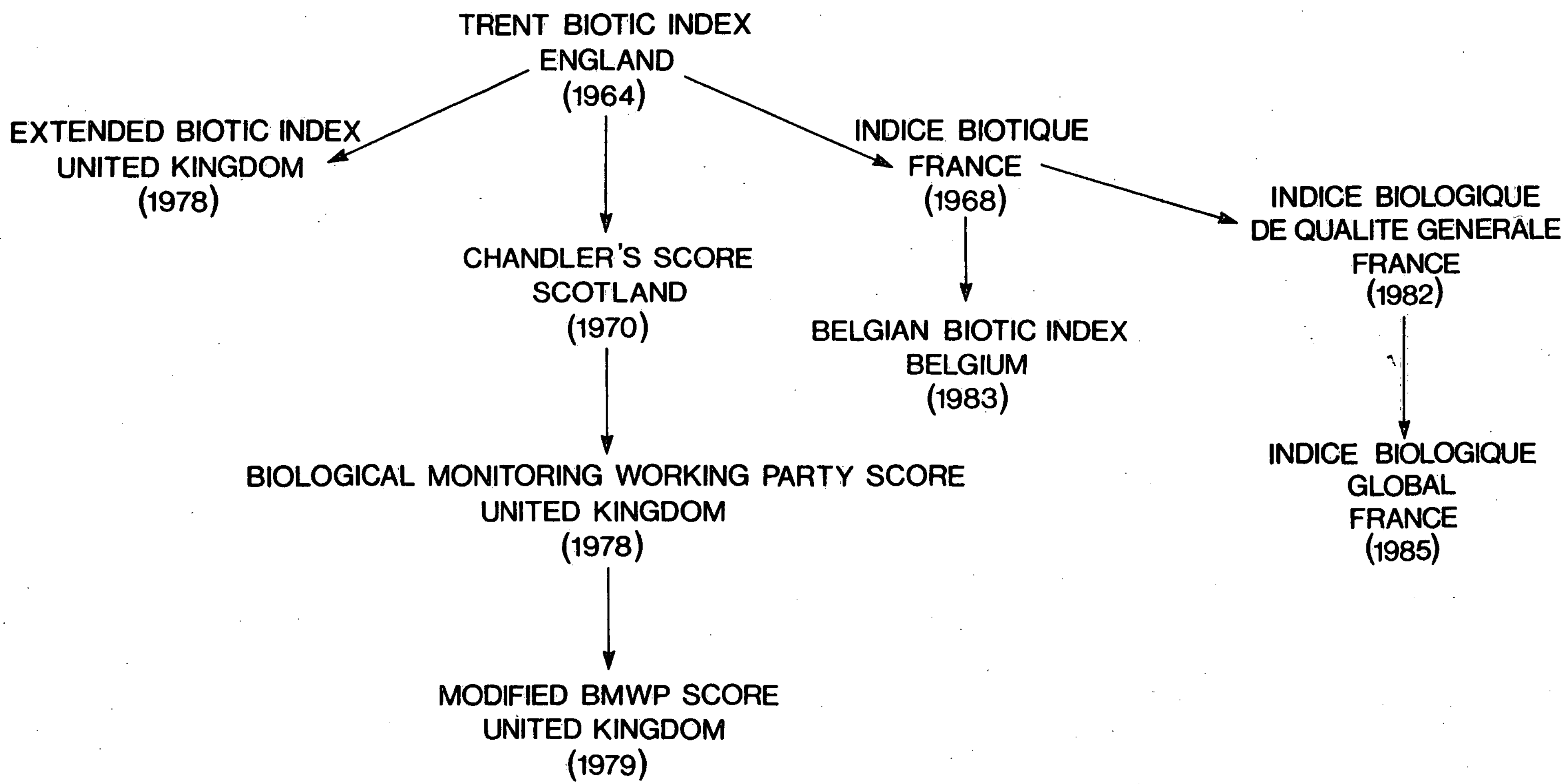
Physical variables	
Latitude	Sampling date
Longitude	Width of water
Altitude	Depth of water
Air temperature	Surface velocity
Discharge	Substratum heterogeneity
Slope	Dominant substratum particle size
Distance from source	Percent macrophyte cover
Chemical variables	
pH	Chloride
Dissolved oxygen	Dissolved orthophosphate
Total oxidized nitrogen	Alkalinity

LEGEND TO ILLUSTRATION:

Fig. 1. Development of the most important biotic index and score systems in Europe, in chronological order.

ABBREVIATED RUNNING TITLE:

Macroinvertebrate Bioassessment of Running Waters in Europe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(1978)
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(1968)
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DE QUALITE GENERALE
FRANCE
(1982)
CHANDLER'S SCORE
SCOTLAND
(1970)
BELGIAN BIOTIC INDEX
BELGIUM
(1983)
BIOLOGICAL MONITORING WORKING PARTY SCORE
UNITED KINGDOM
(1978)
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FRANCE
(1985)
MODIFIED BMWP SCORE
UNITED KINGDOM
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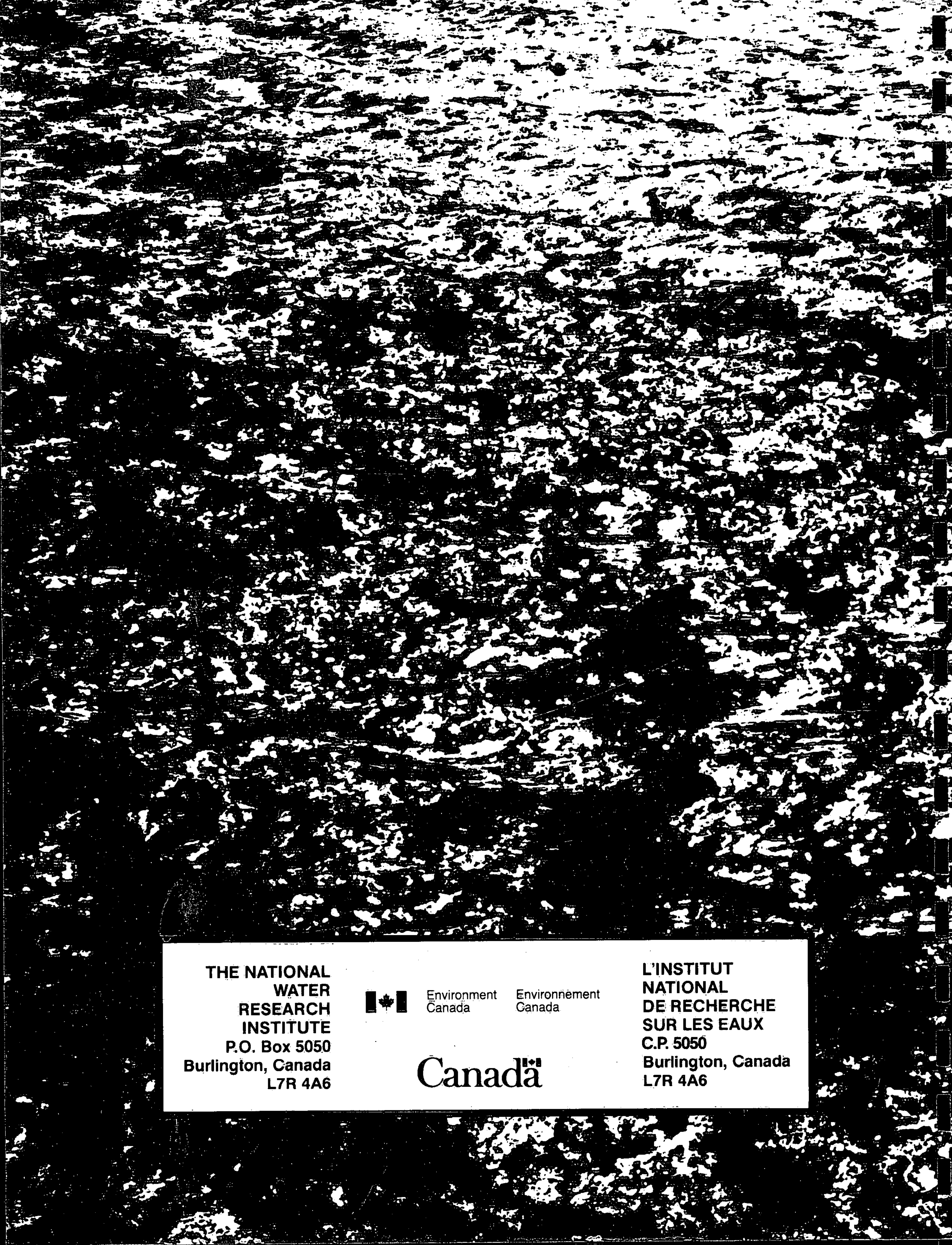

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