Alternative Methods for the Biological Classification of Rivers

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Statement of use
This report reviews the methods and philosophies supporting the Biological Classification of rivers. It is intended to raise awareness of Agency staff to alternative methods to those currently used within the Agency. This information will be of value in discussions relating to possible implementation of the Water Policy Framework Directive.

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EXECUTIVE SUMMARY

The changing nature of river management, recent research developments and environmental legislation combine to present the need and opportunity to review contemporary methods of biological classification of river quality. This report is based on an extensive review of published literature, returns from a widely circulated questionnaire and information from personal interviews and unpublished reports.

The report provides a comprehensive overview of the different approaches to bioassessment of rivers in current use in addition to the UK's predictive approach using macroinvertebrates. The principles and concepts underlying biological assessment are outlined and requirements and considerations for a broadly applicable system of classification are discussed. This provides a context for the subsequent review of the principle biological groups: algae (diatoms), macrophytes, macroinvertebrates and fish.

Diatoms are autotrophic and abundant in all water bodies given sufficient levels of light, water and nutrients. They can be sampled with low cost equipment but may require specialist resources for analysis. The global distribution of many species facilitates the direct transfer of river quality assessment methods over wide geographical areas. Assessment methods using diatoms have been devised to classify general river quality and to indicate specific impacts including organic pollution, eutrophication and acidification.

Macroinvertebrates are the biological group most frequently used in river classification. They are diverse and abundant in most river systems, can be collected using low cost equipment and are relatively easy to identify. Invertebrates are represented in most trophic guilds (detritivore, herbivore, predator) offering the potential for functional feeding group analysis. Their popularity has resulted in the development of a variety of assessment methods adapted for biological classification. Assessment methods using macroinvertebrates have been devised to classify general river quality and to indicate specific impacts including organic loading, acidification, habitat degradation and toxic pollution.

Fish are one of the most conspicuous and economically important organisms of river communities. Sampling typically requires more than one person and specialist sampling equipment, although other sources (e.g. Angling clubs) offer cheap and feasible alternatives for data collection. The fish fauna of the UK is comparatively impoverished but has representatives in most trophic guilds (detritivore, herbivore, predator). Most classification methods using fish have been developed in the USA, although many have been successfully applied elsewhere. Although classification systems using fish communities have only been applied to general river quality, they have been successful in indicating specific impacts including habitat degradation, organic, thermal and toxic pollution.

Most contemporary methods of classification are based on either biotic indices, predictive techniques or multimetric approaches. Whilst a number of studies have compared the effectiveness of these contrasting approaches, results cannot be considered absolute as evaluations may favour a particular approach due to geographic location, the cause(s) of degradation and/or methodology employed in comparisons. Aspects generally applicable to systems of river classification are discussed with reference to how contrasting approaches, using different biological groups and different assessment methods, have addressed these issues.

Current UK predictive approaches possess some distinct advantages over alternative methods of biological classification of river quality. However, with the benefit of recent advances in research and the experience of international applications, existing methods could be further developed and refined to increase the sensitivity of methods and in the long-term achieve the sustainable use of rivers.

Key Words:
Bioassessment, River Classification, Macrophytes, Macroinvertebrates, Fish, Diatoms, Multimetric, Predictive, Biotic Indices.

1. INTRODUCTION

From the king's wine taster and the miner's canary to RIVPACS and the Index of Biological Integrity, the use of biological organisms has long provided an understandable measure to evaluate environmental quality. Although chemical analyses have historically been viewed as providing a 'scientific' and definitive measure of quality, water samples typically have only an ephemeral relevance to riverine ecological communities and their chemical measures require tenuous biological interpretation to evaluate ecosystem quality. Chemical based assessment cannot holistically monitor aquatic resources, failing to assess degradation mediated through physical and biological means. Furthermore, chemical measurements frequently do not account for cumulative impacts, synergistic interactions or biological transformations in the action of contamination. However, stresses are shown by the biological organisms that aquatic ecosystems support and, the resident biotic community thus provides a direct measure of ecological health.

With its origins, at the turn of the century, in the Saprobien system of Kolkwitz & Marsson (1902; 1908; 1909) the evolution and development of biological river assessment has predominantly focused on organic pollution (Metcalfe, 1989) reflecting the historical legacy of (perceived) degradation. Whilst legislation and technology have acted to reduce damaging sewage inputs, the changing activities of human society and a greater understanding of river ecosystems have combined to increase the prominence of other impacts (Sweeting, 1994). Contemporary threats to river ecosystems include a diverse array of contaminants entering from both point and diffuse sources and are of acute, chronic and episodic nature. In addition ecological damage may also be caused by species introductions and habitat degradation. The wide range of impacts, often with cumulative effects, must involve monitoring using biological assessment.

A body of national and international policy and legislation has evolved in response to changing environmental problems and assists in guiding management strategies towards sustainable use. The European Union Water Framework Directive, currently in draft, requires member states to identify least impacted rivers representative of each European ecoregion as reference sites and to implement a program of monitoring for river ecosystems. In addition, national and international obligations such as the EEC Habitats Directive (92/43/EEC), those arising from the Earth Summit (UNCED) as well as the UK Environment Agency's own mandate to the environment, oblige management to evaluate rivers so that they can be used in a sustainable manner.

Natural variation in river ecosystems and the diverse and changing nature of management problems require a suitable system of bioassessment and classification to fulfil the following objectives;

* Identify sites of conservation value;
* Provide an early warning system to ecological impairment;
* Diagnose the causes of environmental problems;
* Evaluate approaches to managing water and habitat quality;
* Evaluate restoration and rehabilitation practices.

River classification in the UK has principally focused on plants (Holmes, 1983) and macroinvertebrates (Wright *et al.* 1984). While current practices have responded well to some issues (e.g. Nature Conservancy Council, 1989), the prominence of organic pollution indices limits their application to the wide range of contemporary management problems (Metcalfe-Smith, 1994). Other countries have developed assessment techniques using different ecological groups (e.g. diatoms - Prygiel & Coste, 1993a; fish - Lyons *et al.* 1995) and employing different evaluation parameters (e.g. EPA, 1997) that may have relevance to river managers in the U.K.

Together, the evolving legislation, recent developments in other countries and the changing needs of river management combine to present the need and opportunity to review contemporary methods of biological river classification. Methods are required that are most appropriate for UK rivers but consideration of international approaches will provide a comprehensive base for assessing the best available practices. This R&D Technical Report is intended to provide a comprehensive overview of the different National and international approaches currently used outside the Environment Agency in river classification. The second chapter outlines the principles and concepts underlying biological assessment and discusses the requirements and considerations for subsequent classification approaches. This provides a context for the following chapters that cover each biological group in widespread use. Each chapter outlines the relevant ecological information before discussing the different methodological approaches and assessment value of respective techniques with reference to contemporary practices and expert opinion. A bias towards European and North American classification procedures is acknowledged and reflects linguistic limitations of the authors, access to published literature and responses to information requests for unpublished "grey" literature.

A questionnaire was designed to canvass up-to-date information concerning contemporary bioassessment techniques and this is included in Appendix A. The questionnaire was sent to approximately 200 research institutions and circulated on 2 international electronic mailing lists; 61 replies were received, the names and contact numbers of the respondents are listed in Appendix B. It was designed to cover a diversity of assessment approaches and offered the opportunity for respondents to provide information about bioassessment techniques using biological groups. The broad range of assessment methods that the questionnaire was designed to cover limited the extent of information that could be communicated. Following a pilot study it was decided that most questions would be formulated using a 'closed' reply design (i.e. requiring respondents to tick an appropriate box) to enable quick, easy and unambiguous replies and facilitate data comparison between different assessment approaches.

Respondents were asked to make explicit statements about the methods, limitations, utility and time required to assess river quality using their respective approaches and were encouraged to generalize and approximate where specific answers were not possible. Nonetheless, responding practitioners often felt unable to complete some questions while other researchers, keen to defend the integrity of their approach, included qualifying statements of honesty such as *'Do you expect anyone to answer this question truthfully?'* included in their reply. The authors consider that variance in replies would scarcely stand up to the statistical rigour that the same researchers might expect from their ecological data.

Despite its limitations, the questionnaire, importantly, served to open up channels of communication. Where novel assessment approaches were identified practitioners were contacted and more detailed information about their approaches were obtained. In addition many researchers sent back their replies accompanied by volumes of both published and unpublished literature and provided a valuable source of data that would otherwise have been difficult to access.

2. CONCEPTS OF BIOLOGICAL CLASSIFICATION TO DEFINE RIVER QUALITY

2.1 Defining Ecological Quality

The variables that determine river quality for biological organisms were classified into five principle components by Karr *et al.* (1986; Table 2.1). Although the relative importance of these factors vary in time and space, they combine to structure the resident biotic community. From the perspective of pollution this definition acknowledges that impairment to river ecosystems can be mediated through chemical, physical, biological or energetic means. In the context of bioassessment, ecological data must be collected and interpreted by measurements representative of these parameters before anthropogenic effects can be assessed.

Table 2.1 The five principle components defining river quality for biological organisms with some of their respective elements (after Karr *et al.* 1986)

WATER QUALITY:	temperature, turbidity, dissolved oxygen, acidity, alkalinity, organic and inorganic chemicals, metals, toxic substances
HABITAT STRUCTURE:	substrate, depth/width, riparian character, geomorphology, in stream cover, stability, spatial and temporal complexity of physical habitat
FLOW REGIME:	volume, temporal distribution, velocity, land use, ground water
ENERGY SOURCE:	organic inputs, sunlight, nutrients, 1' and 2' production, seasonal variation
BIOTIC INTERACTIONS:	competition, predation, disease, parasitism, mutualism, feeding

2.2 General Aspects of Biological classification

The variable character of river systems and diversity of environmental problems present a wide range of circumstances to which biological classification may be applied. Nonetheless, some general aspects are relevant to all assessment approaches.

A system of biological classification must express data in a consistent, objective and comprehensible manner so that the classification is able to detect a change. The successful application of biological assessment requires explicitly defined aims and objectives to which standardized methods for data collection and data interpretation can be prescribed. Ideally all bioassessment approaches should incorporate the following criteria:

* Appropriate references for comparative evaluation of the test site;
* An appropriate selection of biological organisms to assess quality;
* Standardized methods of data collection and analysis;
* Quality assurance;
* Ecological measurements based on scientific understanding;
* Defined levels of acceptable/unacceptable deviation from desirable classification;
* Results easily understood by river managers and river users;
* Known costs and processing time;
* Detection of temporal change;

Defined objectives for a survey enable the tailoring of assessment methods, within the constraints of budget and time, to address particular issues. Planning allows technical and theoretical knowledge to identify and control weaknesses and to exploit strengths, thus maximising the information obtained. A general scheme for biological classification is outlined in Figure 2.1 which links the biological classification to standards and compliance. Sometimes these standards are written into legislation so that they can be enforced. The following sections outline some of the principle concepts of a general framework for bioassessment and classification.

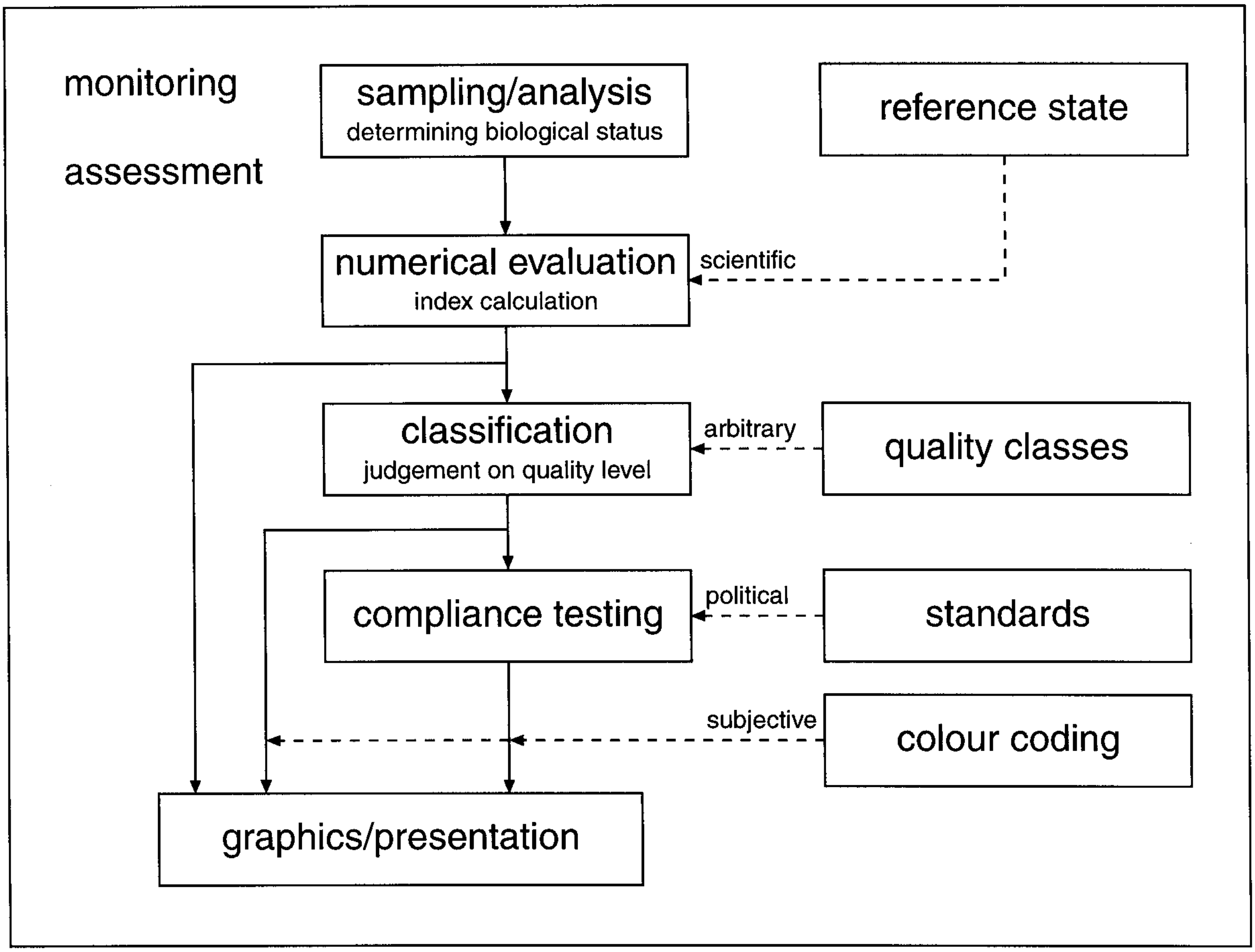

Figure:2.1 Elements of biological assessment methods (from UN/ECE Task force on monitoring and assessment, 1995)

2.3 Management Objectives

Clearly stated goals and objectives provide the benchmark for quality evaluation and a tangible target for management strategies. Such objectives need to be comprehensible scientifically, socially and politically and should be formulated by consultation of all potential resource users. Rivers have many uses and some may have ethical, religious or historical significance in addition to their ecological and economic value (Boon, 1992). In highly modified environments (e.g. urban areas) the ecological potential of a river may be limited. Assigning water quality objectives, therefore, requires a pragmatic approach and quality standards may differ accordingly.

The sustainability of rivers is an important issue with ever increasing demand for water abstraction, navigation, fisheries and hydroelectric power. Ratcliffe (1977) acknowledged the difficulties in assigning a comprehensive definition to ecological systems when using terms such as 'naturalness', 'typicalness', and 'richness' to distinguish features of conservation interest. The more broadly applicable expression 'ecological health' has been used for over 100 years to describe environmental well being or an absence of stress (Rapport, 1992; Scrimgeour & Wicklum, 1996). Despite the lack of consensus on a definition for ecosystem health (Calow, 1992) the concept facilitates easy communication and provides a convenient framework, from assessment to diagnosis and remedial action, (Rapport, 1992).

In the USA the term 'biological integrity' has been popular as a generally applicable, yet definitive reference of ecological status (see Davis & Simon, 1995). Most US researchers follow Karr & Dudley (1981) who define biological integrity as:

"...the ability of an ecosystem to support and maintain a balanced, integrated, adaptive community of organisms having a species composition, diversity and organization comparable to that of the natural habitat of a region. "

This definition carries an understanding of ecosystem dynamics and incorporates both structural and functional dimensions. Furthermore, it acknowledges a degree of resilience and allows for natural variation, thus retaining relevance where biological communities differ with geographic region (e.g. Corkum, 1989) and river character (e.g. Armitage *et al.* 1990). Karr

(1995) recently stressed the difference between health and integrity, arguing that integrity should strictly be reserved for pristine sites that have experienced little or no anthropogenic impact, whereas the more general term, health, is appropriate for modified rivers across a gradient of human disturbance, see Figure 2.2.

2.4 Ecological Theory Applicable to Biological Classification

The capacity to define river systems with regard to their complex, networked, continuum and an increased understanding of structural and functional components of biotic communities are key areas where theoretical developments have enhanced bioassessment techniques and facilitated new approaches to the biological classification of rivers.

Minshall (1988), emphasised the complex nature of river systems and discussed their spatial and temporal variability over 16 stream orders of magnitude. However, Frissel *et al* (1986) showed the correlation between the spatial and temporal dimensions, describing a physical river system as a series of hierarchically organised units with smaller spatio-temporal units nested as subsystems within higher units (Figure 2.3). Within each section (spatio-temporal unit) river processes are determined by proximate controls - intrinsic factors acting at the local scale, and ultimate controls - external factors acting at higher scales. For example, proximate controls at the riffle-pool scale refer to local features such as substratum characteristics, flow regime and marginal habitat structure which are determined by ultimate controls higher in the hierarchy such as slope and catchment characteristics. Similarly viewed at a higher spatial scale, slope and catchment characteristics become proximate controls that are determined, in turn, by ultimate controls (e.g. geology, tectonics) at next level up the hierarchy. Frissel *et* al (1986) discussed the hierarchical nature of these different spatial scales in the context of persistence over time. This hierarchical view of a physical river system provides a convenient framework for the definition of river sections and presents a spatial and temporal template relevant to biological communities that has been adopted by theoretical ecologists. (see Bayley & Li, 1992; Townsend & Hildrew, 1994; Hildrew & Giller, 1994).

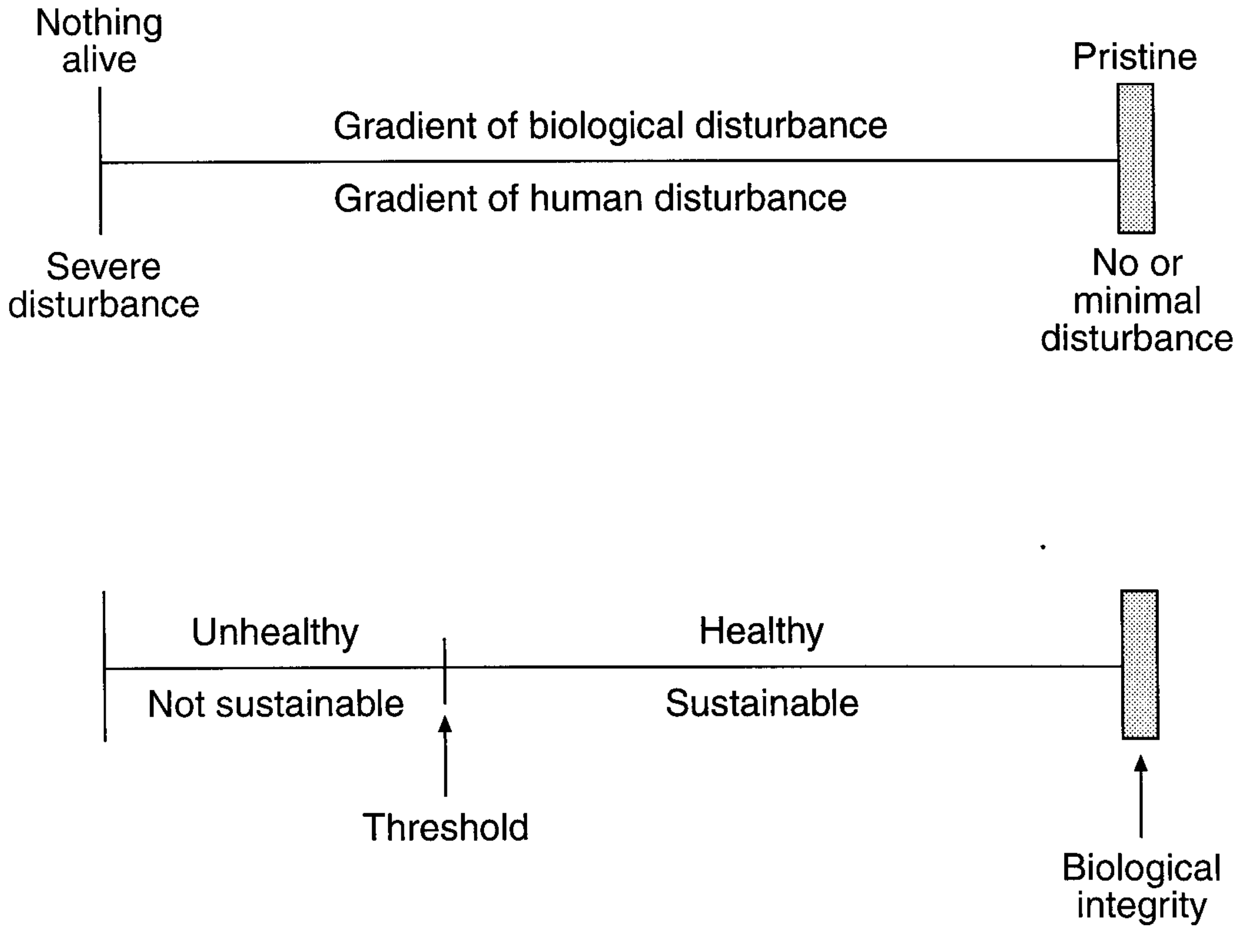

Figure:2.2 At one end of a continuum of human influence on biological condition, severe disturbance eliminates all life. At the other end of the gradient are "pristine" or minimmally disturbed, living systems (top); these systems possess biological integrity. A parallel gradient (bottom) from integrity towards nothing alive passes through healthy, or sustainable, conditions or activities. Below a threshold defined by specific criteria(see text), the conditions or activities are no longer healthy or sustainable in terms of supporting living systems. (adapted from Karr and Chu 1999).

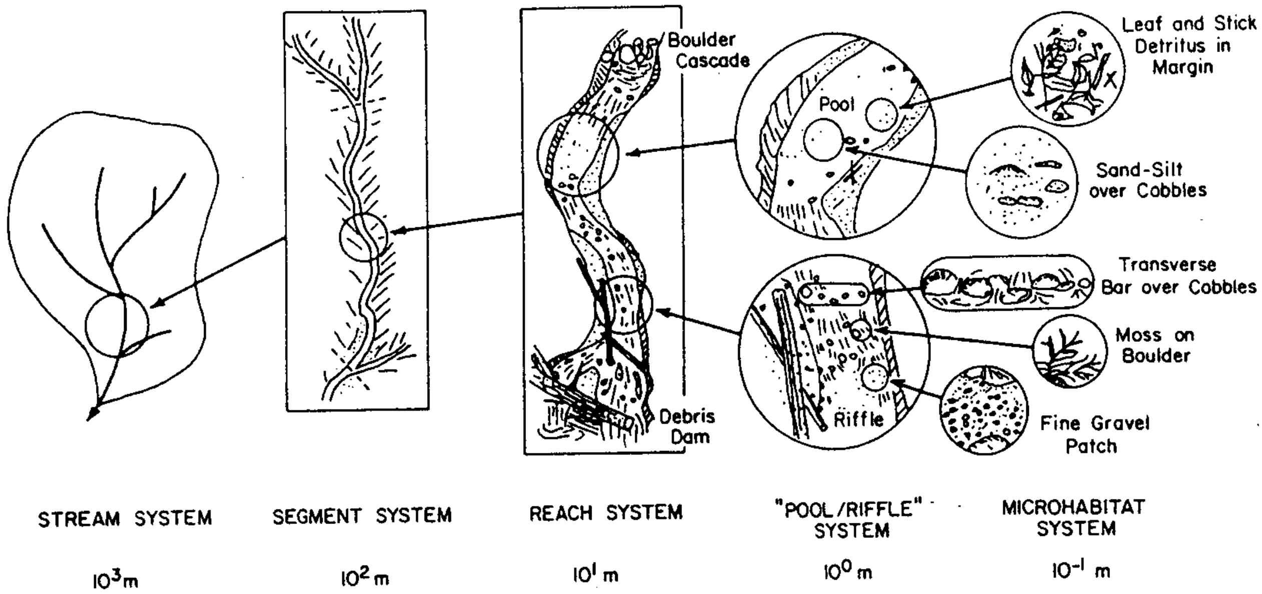

Figure 2.3. Hierarchical organization of a stream system and its habitat subsystems. Approximate linear scale, appropriate to second or third order mountain stream is indicated. (Frissel *et al.* 1986).

2.4.1 Models of river ecosystems

Vannote *et al.* (1980) synthesized contemporary theory and empirical evidence in a generalized model of a river ecosystem, describing the changing longitudinal nature of abiotic and biotic features in the River Continuum Concept (RCC). Conceived in North America, the RCC perceives headwaters as rising in shaded deciduous forests where allochthonous leaf litter provides the energy base. As a river increases in size environmental heterogeneity (e.g. discharge and temperature variability) increase, the shading effect of riparian vegetation is reduced and primary production within the river channel dominates the energy base. In the lower reaches of large lowland rivers, reduced transparency and increased water depth limit primary production and the energy base is primarily derived from depositions supplied by upstream sources.

Using macroinvertebrates, the RCC describes biological communities as serial processing units adapted to maximize available resources by adjustments in the relative dominance of functional feeding groups. Thus predictable biological assemblages are apparent in the transition from heterotrophic headwaters, autotrophic middle reaches and heterotrophic lowland rivers. Discontinuities to this longitudinal pattern can be cause by natural and anthropogenic influences such as sewage discharges (Rabeni *et al.* 1985) or the presence of lakes and impoundment's (Ward & Stanford, 1983) and require modified interpretation, e.g. the serial discontinuity concept (Ward & Stanford, 1983). Despite criticisms for lacking general application (e.g. Winterbourn *et al.* 1981) the RCC has proved useful in explaining longitudinal shifts in the structure and function of ecological communities (Cummins, 1988) and is central to process orientated bioassessment.

Whilst the RCC acknowledges the importance of riparian vegetation, it does not fully incorporate the lateral and temporal phenomena of the 'out of bank' flows experienced by unmodified rivers. Junk *et al. (1989)* conceptualised the periodic inundation of a rivers floodplain and the dynamics of the aquatic-terrestrial ecosystem in the flood pulse concept. This rhythmical event acts as an important environmental cue triggering biological cycles and driving the river productivity by its effect on nutrient cycling. Flood disturbance and the dynamic nature of marginal zones is considered to be a key factor in river biodiversity (Decamps & Tabacci, 1994).

Ward (1989) integrated the lateral and longitudinal components in describing rivers as four dimensional systems with longitudinal, lateral, vertical and temporal components. The additional vertical dimension includes ground water influences, the hyporheic zone and the 'dry' component, used by aquatic insects, plants and semi-aquatic organisms. The interstitial spaces of the hyporheic zone extend down vertically and out laterally and can, in some rivers, contain an invertebrate biomass that exceeds that of the channel (Stanford & Ward, 1988). The hyporheous may act as an important refuge for river organisms during disturbance events and is a key component in stream resilience (Sedell *et al.* 1990). Refugia can allow the rapid recovery of biological communities following disturbance which can thus influence biological assessment information. The temporal component refers to the dynamic nature of river environments including geomorphological and ecological changes occurring over time.

The influence of biotic and abiotic factors and the role of disturbance in structuring ecological assemblages is a central issue in community ecology (Wiens, 1984). Localised disturbances can reset ecological succession in discrete patches and thus, ecosystems can be viewed as a mosaic of such patches in different successional stages.

This patch dynamics perspective of ecosystems means that riverine communities can be expected to differ at a range of temporal and spatial scales (Pringle *et al.* 1988) and this has ramifications for sampling strategies related to bioassessment. Organisms differ in their response to such a patchy environment and their intrinsic ecological attributes govern their dynamics and determine the appropriateness for use in evaluation. Sampling at too small a scale may mistake normal variability (patch scale variability) for impairment, whereas sampling at a large scale may fail to identify insidious processes. Ross *et al.* (1985) monitored fish populations along two streams over 10 years; local, site specific data indicated that communities were unstable, however, data from a larger spatial scale, the entire streams, revealed comparatively stable populations. This illustrates the importance of scale in river classification programmes and why its scientific understanding is crucial to successful biological assessment.

In river systems there has been an inevitable focus on flow as an agent of disturbance (Stanford & Ward, 1983; Minshall & Peterson, 1985; Resh *et* al. 1988; Ward, 1989). Poff and Ward (1989) used long term hydrological data to classify river types in North America according to variability and predictability of flow, characterising river types ranging from unpredictable 'harsh intermittent' to stable 'mesic ground water'. This type of classification provides a suite of templates ranging in physical harshness and has been used to discuss and test the ecological response that is expected to be manifest in species traits (see Resh *et al.* 1988; Poff & Ward, 1989; Bayley & Li, 1992; Poff & Allen, 1995). Minshall and Peterson (1985) applied the Lotka-Volterra model of competition and predation to rivers communities which, other aspects being equal, views ecosystems as tending towards a stable equilibrium. This description of an ecosystem incorporates the familiar r and K strategies attributed to biological organisms that characterize communities at different successional stages. Following disturbance at a site, early successional stages are typified by populations of r-strategists: opportunistic, generalist species that are competitively weak with species traits including small body size, high fecundity, high dispersal abilities and short life spans. In later successional stages, reflecting more stable environmental conditions, populations are dominated by K-strategists that are competitively strong, niche specialists with larger body size, reduced fecundity and greater longevity. Thus, rivers with naturally high levels of disturbance (e.g. harsh intermittent) should posses ecological communities characterized by r-strategists, whereas environmentally benign rivers (e.g. mesic ground water) should be dominated by K-strategists. Therefore, although some river systems may be inherently unpredictable environments, their communities have been selected for these conditions and, as such, represent persistent, deterministic communities (e.g. Fisher *et al,* 1982). This stress species traits relationship has been widely observed in lentic freshwater ecosystems (Schindler, 1987), successfully employed in marine ecosystems (Warwick, 1986) and recently adopted and applied to river communities with promising results (Coeck *et al,* 1993; see Chapter 5.6). The importance of ecological theory, underpinning assessment approaches, is critical to successful bioassessment and subsequent classification.

For example hypothetical relationships for grossly polluted, moderately polluted and unpolluted systems are presented in Figure 2.4 In unpolluted systems, favouring K-selective species, the biomass curve should be above species abundance as most of the biomass is distributed amongst relatively few species (Figure 2.4a); in grossly polluted environments, favouring r-selective species, biomass curves should occur below the abundance curve as biomass is distributed across many species (Figure 2.4c); moderately polluted communities should be represented as an intermediate between these extremes (Figure 2.4b).

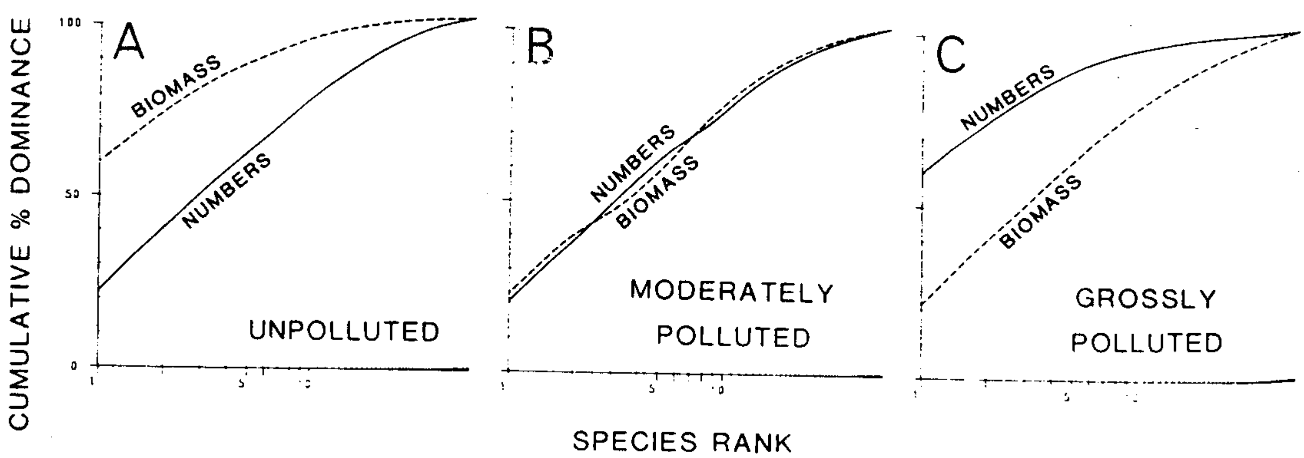

Figure 2.4. Hypothetical K-dominance curves for species biomass and numbers, showing, (a) unpolluted, (b) moderately polluted and (c) grossly polluted conditions (from Warwick, 1986).

2.4.2 Classification of biological attributes

Classification of the biological attributes of organisms, prior to assessment, provides a means of isolating different ecological processes and enables generalisations that can be used to compare different aspects of river quality. Functional feeding guilds, based on food type ingested and method of food acquisition, have been used to classify invertebrates (Cummins, 1973) and fish (Karr, 1981). Such trophic classification can be used to provide information on the energy and nutrient dynamics of a river system enabling assessment of subsequent changes in source or quality. Biological classification has also been applied to other biological aspects of organisms such as behaviour (Gorman, 1987) and spawning strategies (Balon, 1975) in fish, behaviour and habitat preference in invertebrate (see Merritt & Cummins, 1997) and morphology and method of attachment in diatoms (Molloy, 1992). Reducing the communities to these types of dimensions, facilitates the incorporation of theoretical knowledge of lotic ecosystems and allows a focus on specific aspects such as habitat, food resources and water quality.

2.5 Reference Conditions

In the development of bioassessment techniques, reference data have been used in two principal approaches:

1). to derive and calibrate biological measures (e.g. biotic indices), that directly evaluate the quality of a site.

Assessment measures that incorporate ecological knowledge of the tolerance of organisms used (e.g. biotic indices) provide a direct indication of the condition of a site. Ecological information used in such measures can be derived empirically, by collecting biological data from rivers that range in quality from poor to excellent, e.g. the DAIpo index using diatoms (Wanatabe *et al.* 1988; see Chapter 3.7), or can be derived subjectively using the opinion of expert ecologists e.g. the Biological Monitoring Working Party (BMWP) system using invertebrates (Armitage *et al.* 1983) (see Chapter 4.5).

2). to provide a method for comparative evaluation of test sites.

Despite lacking robustness, this approach is often used with biotic indices that are still widely used in biological classification in some countries (see Chapter 8). Classification of river quality (using biotic scores or otherwise) are typically derived by a test site's comparative performance against a reference that represents attainable or desired quality. In this case the selection of reference conditions are critical to the assessment process and a number of approaches have been used to derive the reference database:

1). historical data.

2). empirically derived reference data.

2.5.1 Historical data

Historical data are useful for site specific assessment and for monitoring temporal trends in particular, evaluating improvement or degradation over time. However, changes in the method of data collection can affect the quantity and quality of data recorded that may restrict the scope of analysis by violating requirements of statistical analysis (e.g. homogeneity of variances) Historical data must, therefore, be carefully evaluated before use in biological classification.

2.5.2 Empirical references data

Upstream - Downstream

Assessment of point source pollution often uses an upstream-downstream approach (sampling upstream and downstream of a pollutant) to evaluate ecological damage. This method uses the upstream site as a reference and assumes consistency in all variables affecting the biota, other than the point source discharge, to validate the comparison. The statistical robustness of this survey design has been criticised for psuedoreplication (Hulbert, 1984), but, this can be avoided in some cases by using the Before - After - Control - Impact (BACI) approach where there is a record of both reference and impacted site before and after the impact (Stewart-Oaten *et al.* 1986). Despite the limitations of this method an upstream reference is often most appropriate for assessing the potential for recovery (Milner, 1994).

Multiple Reference Sites

The use of multiple reference sites in different catchments to the test site to collectively define reference conditions allows a river to be viewed in a broader perspective, more appropriate for management concerns where potential impacts include combined point sources, diffuse pollution, physical disturbance and species introductions. However it is difficult in most countries to find unimpaired sites in terms of water quality or habitat. Hence Reynoldson *et al.* (1996) define the reference as *'regional reference condition: the condition representative of a group of minimally disturbed sites organized by selected physical, chemical and biological characteristics'*

Identifying 'minimally disturbed' or 'least impacted' sites to use as an evaluation yardstick inevitably includes a degree of subjectivity, requiring the professional judgement of experienced stream biologists. Hughes (1995) proposes four criteria that reference sites should fulfil:

1. representative of 'natural' conditions;

2. politically acceptable and understandable to all resource users;

3. generally applicable (so that separate references are not required for each system);

4. irrelevant boundaries (e.g. political) are not used to delineate systems.

Ecoregions for spatial scale of reference condition

The concept of ecoregions was developed in the U.S. to provide a more appropriate basis for reference classification (Ormernik, 1995). An ecoregion is a concept to define an area of expected biological similarity. Each ecoregion is viewed as a discrete system formed by the integration of geology, landform, vegetation, climate and human factors which, although their relative importance can vary within and between regions, they combine to present relatively homogenous ecological systems (Ormernik, 1995). However classification at the ecoregion level alone is unlikely to give appropriate weight to factors important in creating homogeneous data sets by comparing the biological condition of streams (Karr and Chu, 1999). Other factors, including topography, geological substrate, stream size and gradient may cause streams within an ecoregion to support contrasting biological communities (e.g Ormerod & Edwards, 1987).

***A priori* reference classification**

In the multimetric approach to bioassessment used in the USA (see below) reference conditions are determined by *a priori* classification using a hierarchy of environmental criteria. Initial classification uses land maps to distinguish lotic ecoregions on the basis that rivers reflect the environments they drain (Ormernik, 1995). The degree of subsequent definition within an ecoregion is a function of the size and variability within that region. Variables such as river size, temperature regime, altitude, riparian vegetation and land use have been effective in distinguishing broad ecological patterns and have been subsequently used for *a priori* reference classification.

Once an appropriate suite of physico-chemical parameters has been selected to group or classify reference conditions these same parameters are used to match a test site with its reference conditions for assessment measure comparison.

Reference conditions using a predictive approach

In the predictive approach (see 2.9.6) reference conditions are determined by statistical analysis of biological data from reference sites selected as minimally disturbed. In essence this is *a posteriori* approach. Statistical classification of biological data groups streams with similar communities together and ordination is used to place ecological communities, and thus classification groups, in statistical space (see Chapter 4.5.; Wright *et al.*, 1984). Further statistical techniques are employed to determine the physico-chemical variables that are most strongly associated with organism distributions in ordination (statistical) space. Thus, these physico-chemical variables can be regarded as providing reference co-ordinates for river classification groups in ordination space. These key physico-chemical variables are subsequently used to place a test site in ordination space, occupied by the reference data matrix, and thus assign it to its appropriate reference conditions. The predicted community (i.e. reference conditions) may be derived from a single classification group to which it is most similar i.e. closest in ordination space (e.g. BEAST; Reynoldson *et al.* 1996) or alternatively the reference conditions may be based on proportional contributions (based on relative distances) from all neighbouring classification groups (e.g. RIVPACS; Wright, 1995).

Whilst both *a priori* and *a posteriori* approaches use key physico-chemical parameters to identify their appropriate reference conditions, it is the classification of the reference sites into the collective 'reference conditions' that differs. In the *a priori* reference system classification is based on physicochemical variables whereas in the *a posteriori* system it is based on the biological community. Within these approaches there is debate whether impaired sites should be included in determining reference sites to create a gradient so as to discern the biological signal from the background of natural variation (Karr and Chu, 1999).

2.6 Habitat Assessment

Habitat is a major factor influencing biological communities in rivers and its degradation can influence community composition. As a consequence, habitat assessment has become an integral part of bioassessment and classification. For comparative evaluation against a reference, similar habitat conditions should be sampled to maximize consistency between reference and tests sites. This enables differences in ecological data to be attributed to other (e.g. chemical or biological) factors. Where similar conditions are not available the recording of habitat data is particularly important to aid interpretation of results.

In the UK most bioassessment methods record habitat variables to fulfil the requirements of their particular survey strategies (Wright, 1995; Kelly, 1998), and represents minimal habitat information. However, other assessment approaches have focused specifically on habitat evaluation itself, for example HABSCORE (Habitat Score) is used to relate standing crop or population data of a target species (salmonids) to habitat characteristics, identifying potential habitat limiting features (Milner *et al.* 1993). PHABSIM (Physical Habitat Simulation System) uses the Instream Flow Incremental Methodology (IFIM) relating information on species flow requirements at the microhabitat scale to flow regime data, generating information on habitat availability at various flows (Johnson, 1993a). Both these approaches are highly specific and, therefore, limited in their application.

2.6.1 The river habitat survey

The River Habitat Survey (RHS) recently developed by the Environment Agency presents a more generally applicable assessment of habitat quality. The RHS assesses river habitat over a 500m reach by classifying specific physical attributes of the channel (e.g. substratum, flow regimes etc.), bank (e.g. bank material, character) and riparian zone (vegetation structure) into categorical groups. These features are recorded at 10 spot checks, across defined width transects, 50m apart. In addition, a general 'sweep' over the 500m reach is used to summarize general and specific features relevant to the stream, encompassing additional features missed in spot checks whilst also substantiating spot check data (Raven *et al.* 1997).

The RHS was carried out nation wide with the intention of creating a database that would generate suites of river typologies. It was intended that these river types would then be used in predictive assessment (cf. RIVPACS) thus identifying the type and extent of anthropogenic impairment at a test site (Raven *et al.* 1997). Although many of the recorded variables equate with those used in fish, invertebrate and macrophyte analysis further testing is required before the RHS can be integrated into bioassessment techniques (Raven *et al.* 1997; Environment Agency, 1998).

2.6.2 Subjective habitat classification

In the USA, the Environmental Protection Agency (US EPA) (1997) recommend completing a standard habitat survey each time biological data is collected. Their suggested survey covers river habitat over a 100m reach and is based on subjective classification, designed to generate a habitat quality rating for the site. Features relating to general habitat characteristics are recorded (e.g. land use, riparian vegetation, substratum size) and a quality assessment is performed on 10 parameters focusing on channel micro-habitat (e.g. embeddedness, sedimentation), channel morphology (frequency of riffles/bends), bank (e.g. stability) and riparian (width) conditions. Each parameter is given a quality rating, scoring points from 1 - 20 divided into 4 categories (optimal, suboptimal, marginal, poor). In an attempt to minimise subjectivity, narrative descriptions accompany each category. Alternative assessments are suggested for upland and lowland rivers to account for natural differences thereby maximizing the relevance of data collected.

2.7 Sampling Logistics & Statistical Uncertainty

The principle aim of any sampling protocol is to optimize precision and accuracy. All surveys sample only a portion of the community, thus variability between collected samples is inevitable even if the community itself is identical.

The US EPA (1997) proposes that biological sampling protocols should aim to:

1. minimise between year variability;

2. maximize sampling equipment efficiency;

3. maximize accessibility of target assemblage.

To best achieve these criteria, ecological knowledge of the target assemblages and the optimum operating conditions of equipment should be considered. Although the timing of biological surveys is often dictated by assessment objectives (e.g. pollution response), where logistics permit, collections should be made under optimal sampling condition to reduce potentially confounding effects.

Methods of data analysis should be decided upon before a biological survey so that methods of data collection are consistent with data analysis. Rivers are heterogeneous environments contrasting between heterotrophic and autotrophic conditions over small spatial scales (e.g. pool - riffle) over which assessment results may differ (e.g. trophic based analysis, biotic indices). Similarly, if analysis requires quantitative data (e.g. abundance ratios), collection of ecological data must be based on quantitative methods. Stratified sampling of all habitats obtains a more holistic community and may be particularly important in the creation of reference databases for predictive techniques, as they cannot extrapolate beyond the range of their own database. Although proportional sampling of each sub-habitat provides a better characterisation of a site, it can make comparison with reference conditions more difficult because of intersite differences in habitat proportions that always exist (Resh & Jackson, 1993). Parsons and Norris (1996) support the 'riffles only' model for invertebrates as it reduces interhabitat variations and thus comparisons are more easily made between equivalent environmental units.

Elliott (1977) discusses how pilot studies can be used to determine desired levels of precision and accuracy and remains the standard text for lotic experimental design and statistical analysis. Hulbert (1984) provides an important discussion on experimental design and the problems of psuedoreplication, with a possible solution in the Before After Control Impact (BACI) design by Stewart-Oaten *et al.* (1986) as discussed in 2.5.2.

2.7.1 Precision or accuracy?

For the purposes of monitoring and surveillance it is essential to be able to attribute biological differences to differences in river quality as opposed to differences caused by natural variability and sampling bias. Thus sampling precision, i.e. reproducible results, are perhaps more important than accuracy, i.e. representation/truth (Norris & Georges, 1993). Where differences due to natural or sampling variance are not considered, sites may be erroneously identified as impaired when no impairment has occurred (often referred to as a 'type I' error) or conversely sites may be passed as acceptable when they are, in fact, suffering ecological impairment (a 'type II' error). Where possible data analysis should aim to quantify precision variance to indicate reliability of results.

The number of replicates required to confidently identify biological differences as opposed to sampling variance depends on the size of the population being measured, their degree of aggregation and the magnitude of difference the comparison is aiming to detect. A cost-benefit analysis to determine the optimum number of samples required to obtain the desired level of discriminatory resolution sufficient to detect differences is described by Ferraro's 'Power Cost Efficiency' formula (Ferraro *et al,* 1989).

2.8 Selection of Appropriate Biotic Approaches

The range of potential bioassessment applications (outlined in Chapter 1) indicates the diverse requirements for a generally applicable system of river classification. Within these broadly defined applications further distinctions are evident, for example 'identifying important resources' could mean evaluating commercially important fish stocks, measuring conservation value or assessing ecosystem ability to cope with an abstraction. Only after objectives have been explicitly defined can an appropriate suite of biological indicators be selected with which to judge river quality and evaluate management strategies (Cairns *et al.* 1993). Selection of indicators should be based on candidates that deliver the maximum information to assess management objects given the constraints of budget and time. Cairns *et al.*.(1993) present a detailed framework for indicator selection, identifying three distinct types: *Compliance* - to judge attainment and maintenance of ecological quality; *diagnostic* - to identify the cause of ecological damage; *early warning* - to identify a degrading impact before serious ecological damage. In the discussion of desirable qualities for these respective indicator types they highlight potential limitations and point out the inherent contradictory qualities required for a complete program of assessment. Some approaches to assess ecological health are discussed below.

2.9 Measuring Ecological Health

2.9.1 Single indicator species

The use of indicator, species regarded as sensitive, tolerant, nuisance, key stone, or integrator organisms can provide a relevant management focus and permit cheap and effective data collection. Single species monitoring often focuses on terminal trophic species (e.g. top predators) as they are considered to integrate conditions at lower trophic levels and, in addition, are generally popular with the public (Cairns *et al.* 1993). In addition, species high in trophic food chains are potentially more susceptible to insidious ecosystem problems such as biomagnification (Mason, 1996). Single species indicators enable simplistic communication of resource status and present a tangible target for management which may, for example, be indicative of conservation value (Boon, 1992) or of systems recovery (Milner, 1994).

Population studies of commercially valuable, endangered or nuisance species can provide detailed information about the effectiveness of management strategies. The presence of exotic species can have important bearing on ecological health by affecting the natural community structure directly (e.g. Meffe, 1983) or indirectly (e.g. Power *et al.* 1985). Furthermore introduced species are considered more likely to be successful under disturbed conditions (Harris, 1995), consequently their presence and abundance may indicate additional impacts.

However, indicator species are unlikely to be equally effective in responding to all impacts and may not be suited to all assessment aims; functional redundancy means that the indicator itself can be replaced without upsetting ecosystem processes. Cairns *et al.* (1993) point out that there is a greater chance of type I and type II errors (see above; Section 2.7.1) where monitoring programmes rely on a single species.

2.9.2 Individual condition

Analysis of the condition of individuals, referred to as biomarkers, uses external signs of poor condition, such as tumours, ulcers, deformities, parasites etc., to indicate ecosystem degradation. The use of biomarkers is based on the principle that the effects of malign influences (particularly chemical contamination) are initially shown at the lower levels of biological organization (i.e. genetic, cell, tissue) which are manifest in outward abnormalities. Thus individual condition provides the opportunity to assess more subtle problems and has potential value as an earlier warning signal (Simon & Lyons, 1995). Recently deformities in Chironomidae have become an innovative tool in water quality classifications (e.g Lenat, 1993)

2.9.3 Community structure

A more holistic picture of ecological status is provided by community structure and a variety of structural parameters can be measured for quality assessment. Many biological indices have been developed and a wide variety have been used for assessment purposes, all are designed to aid our understanding of community structure by simplifying ecological data into a single number. Indices can be divided into four groups: diversity indices, similarity indices, biotic indices and ratio indices. Washington (1984), Boyle *et* al.(1990) and Metcalfe-Smith (1994) provide comprehensive reviews and discussion of some indices commonly applied to aquatic ecosystems. However a brief overview of their key features is presented below.

Diversity indices

Perhaps the simplest overall measure of diversity is taxa richness. More sophisticated measures of diversity are given by diversity indices which incorporate abundance with taxa richness to gauge evenness. Thus, communities that are diverse but dominated by a few taxa will score less than communities with an equivalent taxa richness but a more equitable distribution. Measures of diversity provide a convenient and objective summary of community structure and this makes the diversity index the most popular index used in aquatic bioassessment (Reice & Wolhenberg, 1993). However, results from the questionnaire used in this R & D report indicated that taxon richness was the most popular single measurement, used in 82% of data analysis procedures, with only 40% of respondents calculating a diversity index.

The value of taxon richness and diversity indices in biological assessment have been questioned (e.g. Extence *et al.* 1987). The assumption that a diverse community is a healthy community may not be universally applicable as some unimpacted systems (e.g. nutrient poor, headwater streams) can naturally support low taxonomic diversity. Furthermore, anthropogenic impairment may actually increase diversity (Stanford & Ward, 1983). For example, in nutrient poor systems a certain level of organic enrichment could enhance benthic diversity Also, an impact may simply replace sensitive species by more tolerant species so their diversity remains constant or even increases (e.g. Pinder & Farr, 1987).

Biotic indices

Biotic indices incorporate ecological information about the organisms present, using the indicator concept by allocating a pollution tolerance (or sensitivity) rating to each taxon. Biotic indices can be based on simple presence/absence data or may incorporate abundance before combining all scoring taxa into an single overall value. The biotic index approach was pioneered by Kolkwitz and Marsson (1908; 1909) with their empirically derived Saprobic system. Saprobic indices use a variety of biological groups, although tend to focus on bacteria, protozoa and algae, measuring the degree of organic pollution by combining the abundance of an organism with its allocated tolerance score. A Saprobic rating for the site is determined by proportional contribution of scoring taxa. The questionnaire used in this report identified 4% of river classification based on the Saprobic System which, according to other sources, is still widely used, particularly in Eastern Europe (Whitton *et al.* 1991; Metcalfe-Smith, 1994; UN/ECE, 1995).

According to the questionnaire responses, biotic indices are the second most frequently used community measure (following taxon richness) used in 67% of data analysis. Some biotic indices increase as taxa richness increases and, therefore, increase with increasing sample size by accumulating additional scoring taxa. To adjust for this proportional relationship the score is calculated by dividing the total score by the number of scoring taxa, can be applied as a modification making scores independent of sample size, e.g average score per taxa (ASPT) (Armitage *et al.* 1983). However, unimpacted site scores vary according to sample location

(Wright *et al.* 1989), and this requires appropriate interpretation (e.g. according to habitat; Extence *et al.* 1987). Moreover, many biotic indices have been developed by the subjective allocation of tolerances (Hilsenhoff, 1987). Walley & Hawkes (1996) used objective statistical analysis of a large empirical data set to test the tolerance ratings of a widely used subjectively determined biotic scores (the BMWP) and found that many taxa were incorrectly rated in the original system. Finally, most biotic score and indicator systems have been developed for specific impacts, invariably organic pollution, and
their wider application is limited (Metcalfe-Smith, 1994).

Similarity indices

Similarity indices, unlike diversity indices or biotic indices, cannot be calculated from a single site because they measure similarity (or dissimilarity) between two sites for comparison. Similarity indices can differ in their structural comparisons by measuring general presence/absence of taxa between sites or using a more specific comparison using abundance of these taxa. Similarity indices were applied in bioassessment by 18% of respondents to the questionnaire.

Ratio indices

Ratio indices compare the relative abundance of one group of organisms to another at the same site. Ratio indices may be based on relative tolerance, comparing the abundance's of two groups of organisms with differential sensitivity to a particular pollutant (e.g. Whitehurst & Lindsey, 1990). They are, perhaps, more commonly employed in functional analysis, to compare the balance of a particular biological processes (e.g. Karr, 1981; see below). The diverse range of ecological relationships to which ratio indices may be applied make them a frequently employed assessment measure, used by 60% of questionnaire respondents.

2.9.4 Biological processes

Functional classification has been used to identify feeding strategies for macroinvertebrates (Cummins, 1973), as outlined earlier in the River Continuum Concept, and feeding strategies (Karr, 1981) and reproductive guilds (Balon, 1975) for fish. Knowledge of these taxonomic-functional relationships can be used to investigate river processes and assess ecosystem health

(US EPA, 1997). In process orientated analysis contemporary models of river ecosystems, such as the River Continuum Concept, are a valuable auxiliary to data interpretation, outlining expected patterns and identifying potential anomalies. Ecosystem processes relate directly to the ability of ecosystems to function effectively in the face of anthropogenic influences and provides measure of homeostasis (Odum, 1985; Rapport *et al.* 1985). However, the functional redundancy of species assemblages may limit the effectiveness of structural approaches to identify ecosystem breakdown. The questionnaire indicated that process orientated measures are used in 32% of assessment analysis procedures (predominantly by U.S researchers); of these most workers employ more than one process measurement.

2.9.5 Multimetric approach

Hellawell (1986), comparing the performance of a number of indices (saprobic, biotic, diversity and similarity), found no single index suitable for the range of biological changes experienced (cf. Washington, 1984; Boyle *et al.* 1990; Norris & Georges, 1993), suggesting that a combination of indices would improve biological classification of sites; they form the basis of a multimetric approach whereby a number of biotic measures or metrics are combined into a single summary score.

A metric can be defined as a term or enumeration representing some aspect of the biotic community structure, function or some other measurable characteristic that changes in some predictable way with ecological degradation. The rationale is that certain metrics are more responsive to specific stresses than others and by combining a number of metrics a wider range of impacts may be detected than one biotic index or individual metric. Originally developed using fish communities (Karr, 1981), the Index of Biological Integrity (IBI) incorporated 12 metrics from three broad biological categories; species richness, trophic composition and individual condition (see section 5.5.4).

In order to combine unrelated values from individual measures, metrics must be transformed to unitless entities. The combination of individual measures into an overall score provides a simple summary of ecological quality and has a clear communicative value to a variety of audiences, including non-scientists.

The multimetric approach has been widely applied to river classification in the USA with techniques for fish, invertebrates and algal communities (Plafkin *et al.* 1989; Karr, 1991; Rosen, 1995; US EPA, 1997). Individual metrics may be based on several of the ecological measures discussed above with the use of functional groups necessitating *a priori* classification of organisms into appropriate groups. Figure 2.5 illustrates the structural organisation of the multimetric approach. A variety of metrics have been proposed for potential use (US EPA, 1997), Barbour *et al.* (1995) provide a framework for the selection criteria of candidate metrics, Figure 2.6.

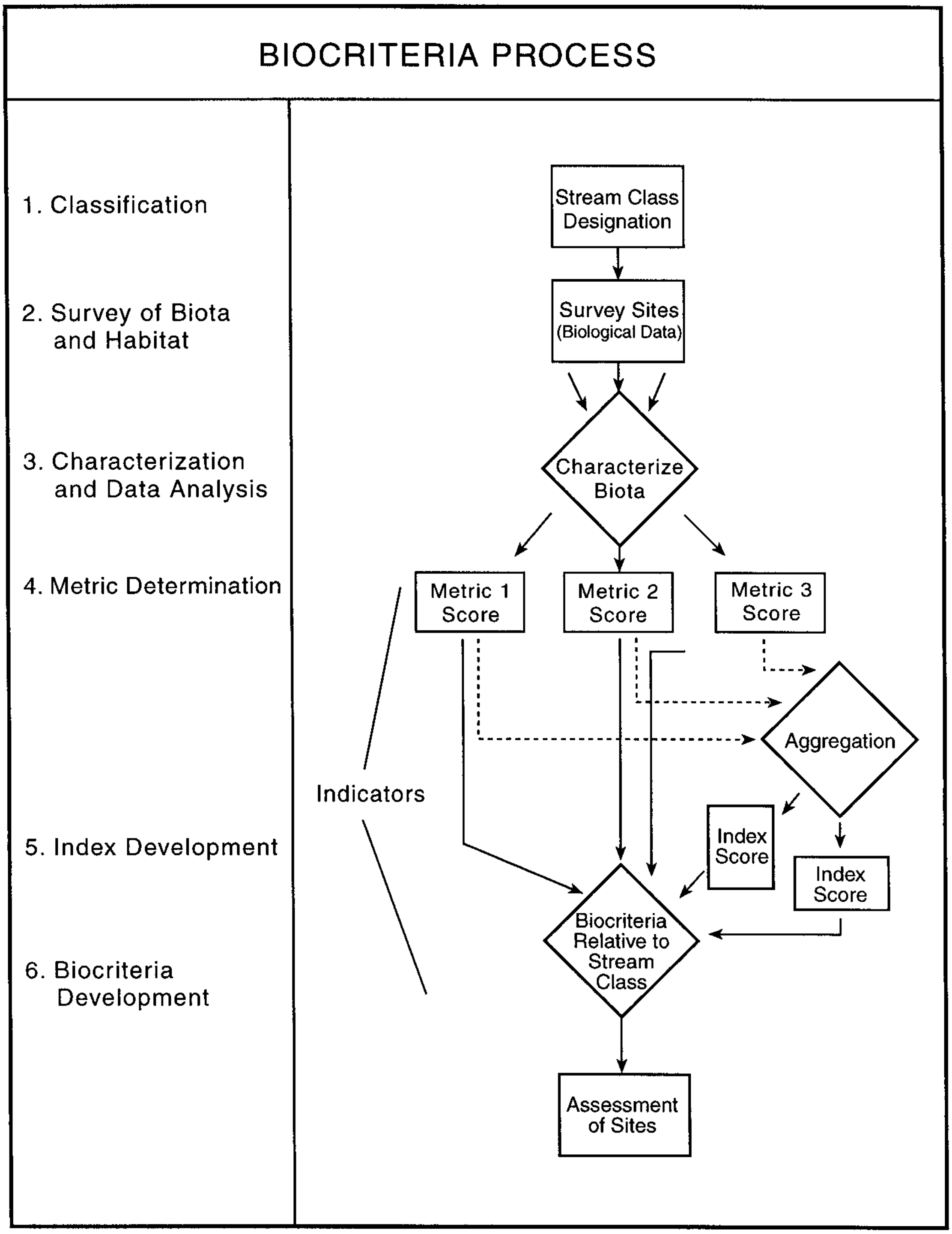

Figure:2.5 The conceptual process of proceeding from measurement to multimetric indicators of condition (modified from Paulsen et al. 1990)

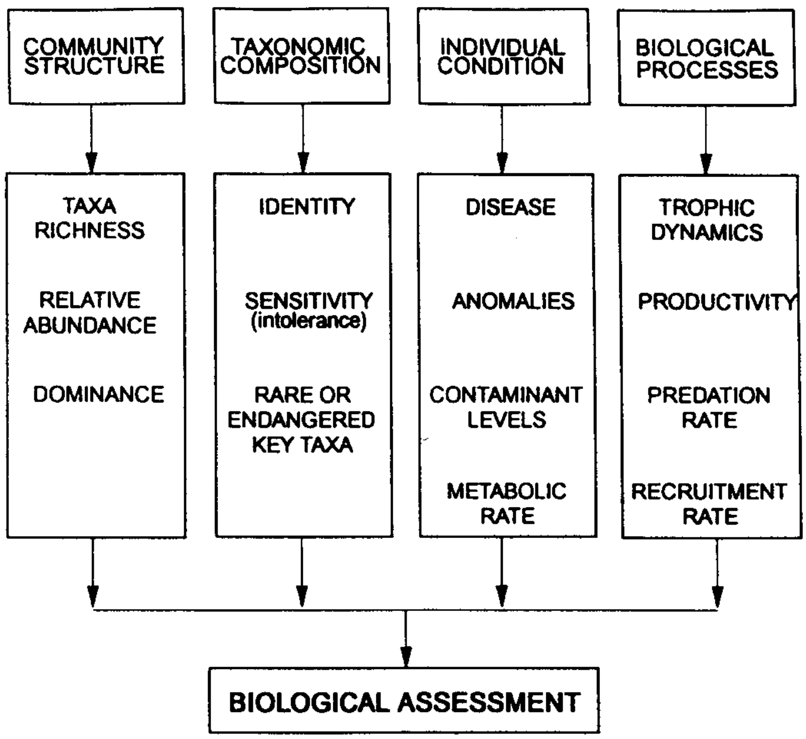

Figure 2.6 Schematic representation of the multimetric approach to river classification (from Barbour *et al.* 1995).

2.9.6 Predictive techniques

The predictive approach to river classification uses physico-chemical variables as input variables and predicts the biotic community either as a biotic score or as an expected taxa list that would be expected at a test site if no anthropogenic damage has occurred.

A site's expected fauna (E) can then be compared to the observed fauna (0), which can be summarized as an 0/E ratio. If the observed fauna is identical to predicted fauna the 0/E ratio will be 1, sites significantly lower than 1 indicate impairment has occurred. The capacity to predict site specific fauna provides an objective reference for assessing anthropogenic damage of any river. It can be used to identify sites of particular conservation interest (Wright *et al.* 1993) and has the potential to identify target species to assess recovery (Milner, 1994). An example of a predictive approach is RIVPACS, using macroinvertebrates, developed for use in British rivers (Wright *et al.* . 1997) and outlined in more detail in section 4.5.4.

2.10 Classification of River Quality

River quality can range from heavily degraded to unimpacted, with a quality measurement for a test site representing a point on this continuum. Classification reduces specific measured values into discrete quality categories; although information is lost through this collective grouping it provides an unequivocal statement of relative quality and is, therefore, helpful to management strategies (e.g. assisting in prioritization). The number of classification categories used and the boundaries that define category groupings may be arbitrary, varying for different assessment methods and in different countries and regions. Because biological classification has generally been introduced where an established river quality framework exists (typically based on chemical measures), most biological classifications systems define their categories in accordance with existing legislation. For example, in the USA Ohio EPA reduce their bioassessment measures into classification groupings representing exceptional, good, fair, poor and very poor quality, developed with corresponding narrative descriptions of quality and fitting in with pre-existing river quality legislation. Frequently these classes were those originally assigned using chemical tests. However, inappropriate levels of classification can lead to problems of failing to recognize important distinctions amongst sites. The key is to have the correct number of classes needed to cover the range of relevant biological variation in a region and the level appropriate for detecting the biological effect of human activity at that site (Karr and Chu, 1999). Chovanec *et al.* (in press) suggest a minimum of five classes to provide sufficient sensitivity to detect degradation.

3. DIATOMS

3.1 Introduction

Diatoms are the most abundant aquatic autotrophs, providing a major food source for protozoa, invertebrates and juvenile fish. Found in all water bodies where light, water and inorganic nutrients are sufficient, they are present on available surfaces (epiphyton) and in the water column (plankton). Their taxonomy and ecological requirements are better known than other algal groups (Kelly & Whitton, 1995) and their short cell cycles and rapid reproduction provides a rapid response to environmental perturbations. Diatom flora have a cosmopolitan distribution with globally common taxa demonstrating a consistent response to anthropogenic impacts throughout their geographic occurrence (Round, 1989). This has enabled the transfer and successful application of assessments techniques for use in the UK that were initially developed for use elsewhere (e.g. Petts *et al.* 1993).

Diatoms were initially used as a tool for river classification in 1908 forming a major part of the Saprobic system developed by Kolkwitz & Marsson (1908). Since these early days their utility as quality indicators has been refined and extended to include nutrient inputs (Schiefele & Kohmann, 1993), acidity (Pan *et al. 1996;* Schiefele & Schreiner, 1991), suspended solids (Chessman, 1986), metal toxicity (Rushforth *et al.* 1981) and salinity (van Dam & Mertens, 1993). Diatoms are a particularly valuable asset to biomonitoring projects where local conditions, such as channelised rivers or temporary water bodies, render the use of other biological groups (e.g. fish, invertebrates) unsuitable (e.g. Prygiel, 1991). Samples are readily collected using inexpensive equipment and their physical size and numeric abundance enables accurate, quantitative sampling (Round, 1993). Despite their potential value, diatoms have rarely been used for monitoring programs in the UK and most developments have come from studies in continental Europe, Japan and the USA (Round, 1993). In the questionnaire used in this R & D report, 26% of researchers use diatoms for bioassessment, however, only 15% (n=8) of completed forms were based on diatom analysis. The advantages and disadvantages of diatoms for bioassessment are summarized in Table 3.1.

Table 3.1 Advantages and disadvantages of diatoms for biological classification

Advantages
Ubiquitous presence in all rivers
Cosmopolitan distribution
Rapid cell cycles - responding quickly to environmental perturbations
Directly dependent on water or sediment for nutrient resources
Rapid, easy sampling; samples occupy a small volume
Immobile - cannot avoid impacts
Tolerance limits for many species are known
Remains often preserved - historical perspective
No ethical concerns
Sensitive to a range of water quality impacts
Relatively insensitive to physical environment
Cell counts by microscope rapid and accurate
Slides can be retained as permanent reference

Disadvantages
Resistant frustules can be transported to site, which are unrepresentative of local conditions
Influenced by substrate
Microscopic size requires experienced personnel and expensive equipment to identify flora
Biological effects of grazers
Response times unknown
Microhabitat differences

3.2 Ecology

The siliceous cell wall (frustule) around the cytoplasm forms the basis of diatom taxonomy and enables identification to species and sub-species level (e.g. Krammer & Lange-Bertalot, 1986-1991). The resistant frustule confer some unique advantages for bioassessment: diatoms are rarely damaged during sampling (unlike other periphytic algae); the frustule often remains preserved allowing reconstruction of past conditions by sediments analysis (e.g. Hall & Smol, 1992) or via herbarium plants (e.g. van Dam & Mertens, 1993) thus they are well suited to the creation of permanent records, a property invaluable for quality control, cross referencing, and retrospective analysis (Prygiel, 1991). Although independent of substrate effects the planktonic community is dominated by relatively few species and is spatially and temporally ephemeral. Furthermore, genuine plankton is restricted to relatively deep, slow flowing waters, is rarely present in upper and middle reaches of rivers and is consequently sparse in the comparatively short rivers of the UK. The benthic community are, therefore, more suited for site specific assessment and are favoured for biomonitoring.

Different types of rivers and different river habitats host distinct diatom assemblages as the relative importance of hydrology, substrata, light, water chemistry, temperature and biotic influences change. Communities are classified according to the habitat in which they are predominately found, viz.: diatoms populating rocks are referred to as the epilithon; populations on plants as the epiphyton; populations on sand as the epipsammon; and populations on mud as the epipelon. Further distinction is apparent in the contrasting growth forms that can be crudely classified as adnate (prostrate, strongly attached), pedunculate (attached by a stalk) and motile (free living) forms. Where turbulence and scour characterize benthic conditions firmly attached, adnate organisms dominate. However, as shear stress decreases an increasingly complex 'forest like' floristic structure develops with adnate, pedunculate and motile forms occurring together (Round, 1993). Flow regimes also affects colonization, with the rate of colonization substantially increased in slower flowing water (Reiter, 1986; Steinmann & McIntire, 1987).

Chemical interactions between substrate and diatom communities are thought to become increasingly important in nutrient poor waters (Cox, 1988; Mason, 1996). However, ambient water quality is considered of overwhelming importance (Round, 1993). In a study comparing different stone types Antoine & Benson-Evans (1985) found that sandstone was more rapidly colonized and supported a greater biomass than other stone types. Similarly Clifford *et al.* (1992) found rough tiles developed significantly more growth than smooth tiles during a 14 day colonization experiment. An apparent preference for rough surfaces has been attributed to greater heterogeneity providing greater diversity of microhabitats and increased protection from sloughing.

The effect of light intensity on diatom assemblages is unclear. Hodgkiss & Law (1985) and Robinson and Rushforth (1987) found riparian shading to significantly influence diatom assemblages, whereas other authors have found no difference between shaded and unshaded sites (e.g. Winterbourn, 1990). Wellnitz *et al.* (1996) identified a complex interactive effect between light intensity and grazer abundance in experimentally manipulated plots in a Colorado stream. However, the general consensus appears to be that differences due to riparian shade are mainly restricted to biomass with dominant indicator species represented in shaded and unshaded sites (Round, 1993).

3.3 Sampling

Although diatom populations are affected by seasonal change index values appear not to vary significantly (Kelly & Whitton, 1995). Thus, using most contemporary assessment methods, (see below) restricted seasonal sampling and season specific interpretation are not required.

Diatom samples are collected by 2 principle methods:

3.3.1 Direct sampling

According to questionnaire returns 88% of researchers collected diatom samples directly from natural habitats. For classification purposes the epilithon tends to be the most frequently sampled group as rocks are structurally similar, more suited to quantitative sampling and are often present over much of a river's length (Round, 1993). Round (1993) advocates the use of judgement in avoiding stones 'contaminated' with coatings of green algae or silt, advising preferential selection of stones with a visible brown coating, indicative of rich diatom growth.

Riffles are generally favoured as they provide consistent physical conditions and are an important habitat to diatom production. Kelly *et al.(1995)* found stone size influenced floral assemblages, suggesting this reflected differential susceptibility to disturbance, although no significant difference in assessment score were apparent between large and small stones. Nonetheless, Wanatabe *et al.* (1988) recommends that, where possible, samples should be restricted to the upper surfaces of flat stones with a water depth of approximately 30 cm and a current velocity of about 40 cm/s to ensure consistency between sites. Despite inevitable local variation due to microhabitat differences (stone orientation, roughness and composition) stone type is not regarded as critically important (Round, 1993). Five stones are generally considered to be sufficient to characterize a site and it is suggested that a more representative sample is obtained by collecting stones at intervals rather than those juxtaposed (Wanatabe *et al.* 1988; Round, 1993).

Where stones are unavailable, quantitative samples can be successfully collected from other surfaces such as macrophytes (e.g. van Dam & Mertens, 1993) or dead wood (e.g. Kelly *et al.* 1995). Most authors (e.g. Whitton *et al.* 1991) caution against collecting from depositing environments such as pools and margins where the resistant Frustules may have been carried for some distance before being deposited and may therefore confound evaluation. In channelised rivers or sites where large substratum size precludes removal, effective sampling can be obtained using a sealed tube with a brush attachment (Kentucky Department of Environmental Protection, 1993; US EPA, 1997).

Although techniques vary in their efficiency, removal by brushing with a stiff toothbrush, scrapping with a knife or rubbing off with a finger are all considered adequate for general assessment purposes (Round, 1993). However, unless a new brush is used for each site, knives or blades are preferred as they have a lower risk of contamination (Prygiel, 1996). Quantitative sampling can be attained by delimiting a defined area and various methods have been employed (e.g. cellophane, nail varnish). Removed diatoms are washed off with water and are fixed with a preservative such as formalin.

Contamination by unrepresentative flora can confuse site evaluation. Round (1993) recommend transporting stones back to the laboratory to view samples intact and therefore quantify this unwanted component. For epilithic analysis he also encourages a vigorous washing with a jet of water to remove the mobile 'silt' flora present leaving the remaining sample dominated by genuine epilithon. However, Kelly *et* al (1995) and Prygiel & Coste (1993a) argue that these 'additional' taxa are an important feature of the community often dominating where organic inputs occur and thus providing valuable information for assessment. Kelly *et al* (1995) consider that washing stones on site prior to sampling is sufficient to remove potentially contaminating lightly attached organisms.

3.3.2 Introduced substrata

Introduced substrata are rapidly colonized by diatoms and provide a convenient collection method for river bioassessment. The use of such 'artificial' substratum enables standardized sampling and reduces natural floral variability between meso- and micro- habitats where differences due to the habitat preferences of taxa and potential bias from differential sampling efficiency can occur. Moreover, artificial substrates can be positioned exactly in the river channel (e.g. at a point source discharge) where suitable natural substratum is unavailable or sampling of the 'natural' habitat otherwise problematic. The advantages and disadvantages of using artificial substrates are summarized in Table 3.2.

Table 3.2 Advantages and Disadvantages of using introduced substrata to sample diatoms.

Advantages

standardized sampling, time & area, no confounding effects of microhabitat
less contamination from macrophyte and other algal growth
substrates chemical character and surface structure controlled and consistent
dead cells are rarely included
positioned exactly in river.
sample in difficult locations

Disadvantages

only representative of flora present during immersion - limited integration
substrata position and length of time influence resultant community
biased to fast growers that can attach to specific substrata (i.e. often flat smooth surfaces)
apparatus must be held in stream bed - susceptible to loss and vandalism
additional cost
time delay (colonization period); 2 site visits required

Numerous materials have been used to provide a colonizing surface including microscope slides (Petts *et al.* 1993; Deniseger *et al.* 1986), plastic plates (Besch *et al.* 1972), ceramic tiles (Clifford *et al.* 1992) and clean sections of cut rocks (Antoine & Benson-Evans, 1985). Colonization times are influenced by local conditions (Section 3.2) and time to reach a 'climax' community may differ between stream types with eutrophic systems attaining a stable community quicker than oligotrophic systems (Rott, 1991). The duration of successional change in diatoms is unknown (Cox, 1991), however, available literature suggests that stable communities develop within 4 weeks, (Lamberti & Resh, 1985; Biggs, 1988; Cattaneo & Amireault, 1992). In the only questionnaire return based on artificial substrates the substrates were left for 30 days. Increased colonization rates can be achieved by manipulating substrates, for example, by coating with agar (Stevenson, 1984; Winterbourn, 1990).

A survey on the River Wissey found diatoms on glass slides bore little resemblance to adjacent flora on the channel bed indicating weak colonization by lateral migration and an assemblage primarily derived from the water column, thus originating from upstream (Petts *et al.* 1993). Such phenomena have lead to criticism of using introduced media for being unrepresentative of local fauna (Aloi, 1990). Whilst this may be an important consideration depending on the aims of an assessment program their value in reflecting extant environmental conditions is not diminished. Aloi (1990) and Cattaneo & Amireault (1992) provide reviews on the use of artificial substrates for diatom sampling.

3.4 Sample Processing

Preparation of samples is relatively simple although requires comparatively more time than preparation of other biological groups and involves potentially hazardous chemicals. Round (1993) provides an overview of required equipment and methods for diatom analysis and assessment.

The cosmopolitan distribution of diatom flora means that existing taxonomic guides can be used to identify most taxa to species worldwide (e.g. Kobayasi & Mayama, 1982; Juttner *et al.*, 1996). While it has been estimated that 8000 cells need to be identified to accurately estimate the species diversity of a stream (Patrick, 1954) this is seldom practical for bioassessment. In practice the number of cells identified varies depending on the object of the survey, the intended data analysis and the convictions of the investigator. Most studies use a fixed count of a predetermined number of cells. Descy (1979) advised the identification of 500 cells, and has been followed by a number of researchers (e.g. van Dam, 1982; Chessman, 1986; Lobo *et al.* 1995; Nather-Khan, 1990). Other surveys (e.g. Prygiel & Coste, 1993; Juttner *et al.* 1996) have used 400 taxa as suggested by the more recent taxonomic treatise of Krammer & Lange-Bertalot (1986-1991). For rapid evaluation reduced counts are favoured, for example Kelly *et al.* (1995) examined 200 cells to assess indices performance. Round (1993) considers the identification of as few as 50-100 organisms as sufficient to determine dominant taxa that characterize water quality, proposing that sites are generally dominated by very few taxa. Hence excessive counting increases numbers of rare taxa identified which is time consuming and merely increases representation of contaminant flora. Furthermore, rare taxa are often disregarded from analytical calculations (e.g. Pan *et al.* 1996). Although reduced counts clearly present a valuable time saving opportunity, experimental evidence suggests that counts often need to go as far as 400 to attain a stable index value (Prygiel, 1996; Prygiel & Coste, 1993b).

3.5 Assessment Approaches

The pioneering work of Kolkwitz & Marsson (1908) was based on the use of indicator species and, although there has been an increasing tendency to convert observed assemblages to numerical indices and a greater emphasis on statistically robust evaluation, data analysis methods have developed little beyond this early work (Round, 1993).

3.5.1 Diversity

Diversity indices are a measure commonly applied to biological communities in assessment projects (Section 2.9.3). Although some workers have found lower diversity at polluted sites (e.g. van Dam, 1982) the simple principle of degradation resulting in a decrease in diversity as a standard response is apparently not universal for diatom communities. Chessman (1986) studied 53 sites subject to various impacts including plantation forestry, agricultural development, urban and industrial wastes and found species richness peaked at intermediate pollution levels. Similarly Lobo *et* al.(1995) found species richness, Shannon diversity and evenness highest at sites with intermediate levels of organic and eutrophic pollution. These findings are substantiated by similar results from other studies (e.g. Stevenson, 1984; Nather-Khan, 1991; Juttner *et al.* 1996) Thus diversity indices alone are not considered suitable as bioassessment indicators using diatoms (Lange-Bertalot, 1979; Sullivan, 1986). Nonetheless, 50% of diatom based assessment employ the simplistic diversity measure of taxon richness, with 12.5% calculating a diversity index according to completed questionnaires.

3.5.2 Indicator species

Autoecological information for diatoms is well documented with ecological notes often provided in taxonomic keys (e.g. Krammer & Lange-Bertalot, 1986a-d) and numerous compendiums relating diatom tolerance to water quality (e.g. Descy, 1979; Leclercq &

Maquet, 1987; Sladecek, 1986; Descy & Coste, 1990). However, such information is usually incorporated into a statistically objective scoring system (see below).

3.5.3 Biotic indices

Biotic indices reduce floristic data to a numerical form and are generally derived from abundance data coupled with autoecological information. Biotic indices are the most frequently calculated ecological measure for diatom based assessment, used in 86% of community analysis (questionnaire results). Many of the assessment indices used for diatoms are derived from the same weighted average formula of Zelinka & Marvan (1961):

$$\text{index value} = \frac{\Sigma\, a_j s_j v_j}{\Sigma\, a_j v_j}$$

where: a_j = relative abundance of the species j

v_j = indicator value (relating to distribution)

s_j = pollution sensitivity.

The pollution sensitivity score (s_j) assigns organisms a rank value (typically ranging from 1 to 5) based on empirical data and this is the range of the index values. Descy's (1979) index, based on the above equation, is an example of diatom index used to evaluate river quality. Index values from 1-5 are delineated into scoring categories to provide a classification system (Table 3.3).

Table 3.3. Classification system for river quality based on diatom assessment using Descy's index (1979).

Index value	> 4.5	no pollution.
Index value	= 4.0 - 4.5	slight pollution, small change in community
Index value	= 3.0 - 4.0	moderate pollution or significant eutrophication, reduction of sensitive species
Index value	= 2.0 - 3.0	heavy pollution, increase in tolerant species.
Index value	= 1.0 - 2.0	very heavily polluted, tolerant species dominant.

While some indices are more detailed, including more taxa and/or a greater number of pollution sensitivity categories (i.e. intervals of 0.5), most variant indices based on this form differ simply in the values ascribed to v and *s* (e.g. Descy, 1979; Coste in CEMAGREF, 1982; Sladecek, 1986; Fabri, 1987; Leclercq & Maquet, 1987; Runeau & Coste, 1988; Schiefele & Kohmann, 1993). As a consequence correlation between individual indices is often high (Coste *et al*, 1991; Prygiel & Coste, 1993; Kelly & Whitton, 1995).

An index that has been developed using a different approach is the comprehensive Diatom Assemblage Index for organic pollution (DAlpo) developed by Wanatabe and colleagues (Wanatabe *et al.* 1990, references therein). Based on analysis of empirical data the DAIpo system uses 548 taxa derived from 134 sites in Japanese rivers, calibrating taxa's organic pollution rating by their statistical relationship with BOD (biological oxygen demand), COD (chemical oxygen demand), TOC (total organic carbon) and other variables (Wanatabe et *al* 1990).

Using this empirical data, taxa are then broadly classified as saprophilous (occurring in organically rich waters), eurysaprobic (present in waters that range widely in organic content) and saproxenous (organically poor waters) and given a DAlpo value between 1 and 100.

River quality is calculated by a proportional contribution equation:

$$\text{DAlpo} = 100 - \sum_{m}^{m} S_i - 1/2 \sum_{j=1}^{m} E_j$$

or

$$\text{DAlpo} = 50 + 1/2 \left(\sum_{k=l}^{p} X_k - \sum_{i=l}^{m} S_i\right)$$

where: $\sum_{i=1}^{m} S_i$: sum of relative abundance's (%) of saprophilous taxa from 1 to m.

$\sum_{j=1}^{m} E_j$: sum of relative abundance's (%) of eurysaprobic taxa from 1 to n.

$\sum_{k=1}^{p} X_k$: sum of relative abundance's (%) of saproxenous taxa from 1 to p.

The DAlpo system scores individual taxa over a greater range than Descy's index and thus presents tolerance values as more of a continuum as opposed to the categorical groupings that are effectively imposed by restricting tolerance values from 1 to 5 with intervals of 1. Wanatabe & Houki (1988) related DAlpo values, saprobic rating and BOD to describe seven classification groups, Table 3.4.

Table 3.4. Saprobic classification DAlpo value and BOD relationship (Wanatabe & Houki, 1988).

Saprobic system	DAlpo	BOD (mg/l)
Xenosaprobic zone	100 - 85	< 0.25
p - Oligosaprobic zone	84 - 70	0.25 - 1.25
x - Oligosaprobic zone	69 - 50	1.25 - 2.5
P -Mesosaprobic zone	49 - 35	2.5 - 5.0
a -Mesosaprobic zone	34 - 20	5.0 - 10.0
0 -Polysaprobic zone	15 - 5	10.0 - 20.0
a -Polysaprobic zone	84 - 70	> 20.0

An alternative approach, used by French workers, uses a hypothetical upstream-downstream transition of species (Coste,1984) and later expanded by Descy & Coste (1990) in a contract for the EEC to become the CEC index. The CEC index is based on a two dimensional grid entry system containing 223 taxa divided into 8 groups and 4 sub-groups (see Table 3.5). The columns range from sensitive (intolerant of pollution) to tolerant species, while the rows represent a theoretical upstream - downstream gradient of increasing mineralization. The CEC index value is calculated by:

$$\text{CEC index} = 12 - 2\text{MG}$$

or

$$\text{CEC index} = 12 \text{ (MGS and GM intersection value)}$$

where: MG = modal group (i.e. group containing the most abundant taxa)
MGS = modal sub-group (i.e. sub-group containing the most abundant taxa)

			G1	G2	G3	G4	G5	G6	G7	G8
			AMIN DTCA EPEC GANT HAAC MCIR MSIN	ALIB APEO CMIN DEHR DVUL FCAP MOIS	ALAN AAOS CCRE CPRO CSIN GMIN GOLI	DITE FCVA MVAR MLAN MFON MSOC AABB	AMMO MACI MJOU MPUP MIHU MPRE SBRE	GPAR MMIN GGLI MATO MIAR MMMU MSBM	HAMP MCOM MGOE MMLF MSEM MSMO MVEN	AVEN MACO MCPL MPAL MPAD MZSU MUMB
			G1	G2	G3	G4	G5	G6	G7	G8
SG1	AAUS ABIO ACLE ACOA AOET AEXI AFLE APEL APUS AVIT CCES CEHR CGRA CSLE DAMC DHME EARC ECUR EEXI EAHO ETUR FRSA FVIR GCLA GCLE GOMI MACO MBRY MEAF MGPE MHAN MPSL MRAO MAEI MRHY MSPO MSTL MTRI PMOB RGIB SLIN STAN	SG1	10	9	8	7	6	5	4	3
SG2	ANOR CAFF CBAC CCIS CCYM CELL CHEL CLAN CMIC CNAV CPED CPLA CSIL CSOL CTGL DOBL FCON FVUL GAFF GGRA GNOD GTRU GVAC GVAT MACU MCTE MIAM MINO MMEN MAEC MSBH MSBL MTPT MVER MZAG PGIB PSCA PVIR SACU SANG	SG2	9	8	7	6	5	4	3	2
SG3	ACON AEXG ASAX FBRE FLEP FPIN GANG GAUG GTER MAPI MCPR MCUS MDEC MORA MEXI MGRE MHEU MIFT MIGF MIGR MIRO MITE MKOT MLSU MLAT MLIN MSIO MTEN MVIR MVRO STKA SULM	SG3	8	7	6	5	4	3	2	1
SG4	ADEL ASPH BPAR CAMP CMEN GSCA MAUR MAMC MAMP MCIM MCLA MDEB MOUB MHAL MIFR MINC MINT MIPU MLEV MLVI MMUT MMVE MOBT MPHY MPRO MPVG MACS MSAL MSHA MSIG MSLE MTRV PMIC PMBA SIDE SOVI SPUL STAB	SG4	7	6	5	4	3	2	1	0

Table 3.5. Table for determining the 'CEC-index'; the groups (columns), include taxa with low indicator value ranked by increasing tolerance from Gl to G9; the 'sub-group' (lines) include taxa with high indicator value, ranked by increasing tolerance from SG1 to SG4 (from Descy & Coste, 1990).

3.5.4 Zonation

An approach proposed by Round (1993) stems from the distinct patterns in floral assemblages associated with increasing stream order (e.g. Maier & Rott, 1990). Viewing the river continuum as a series of zones (and subzones) that are characterized by a few typical flora. Round (1993) suggests the use of a reduced flora of about 20 indicator species, restricted to genuine epilithon, following a predictable hypothetical gradient of species from low pH, clean water taxa to increasingly eutrophic, 'poor water quality' taxa approximating to longitudinal downstream changes. (Table 3.6). Epilithic samples can then be used to determine the zone with the corresponding definition. Round (1993) considers this to be a preliminary 'broad brush' classification and stresses the need for further studies particularly in reactions classified into zone 4 (see Table 3.6).

Table 3.6. Classification of water quality representing a theoretical downstream gradient of increasing mineralization based on epilithic diatoms. (from Round, 1993).

ZONE 1: Clean water in uppermost reaches.

Dominants: *Eunotia exigua, Achnanthes microcephala*

ZONE 2: Nutrient richer and somewhat higher pH.

Dominants: *Hannaea arcus, Fagilaria capucina, Achnanthes minutissima*

ZONE 3: Nutrient rich

Dominants: *Achnanthes minutissima, Cymbella minutal Cocconeis placentula, Reimeria sinuata, Amphora pediculus.* Flora given in sequential order thus individual dominants can be considered as sub-zones.

ZONE 4: Eutrophic with restricted flora due to detrimental influx of materials

Dominants: *Gomphonema parvulum.* Too few taxa representative of this zone have been determined and further work is required

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ZONE 5: Flora grossly restricted by detrimental influx of materials.

Dominants: small *Navicula spp., small Nitzschia spp., Amphora veneta, Gomphonema augur, Gomphonema parvulum, Navicula accomoda, Navicula goeppertiana.*

3.5.5 A multimetric approach

Although multimetric assessment is not well developed for diatoms in comparison to other biological groups the US EPA (1997) have brought together regional and state information and outlined an approach for periphyton assessment (Table 3.7). The metrics are composed almost entirely on a structural basis with various diversity, biotic and community comparison indices combined with collective indicators (percent sensitive taxa, percent tolerant taxa). An additional functional metric, percent motile taxa, represents a easily calculated measure that has apparently been effective as a siltation index (Bahls *et al.* 1992).

Table 3.7. Summary of periphyton metrics suggested for use by the US EPA. (US EPA, 1997)

Total number of diatom taxa - estimate of species richness
Shannon diversity (for diatoms) - to measure richness and eveness
Percent community similarity of diatoms - comparison to reference site
Pollution tolerance index for diatoms - biotic index used by Kentucky DEP.
Percent sensitive diatoms - sum of relative abundance of all sensitive taxa
Percent motile diatoms - siltation index

Total number of non-diatom taxa - estimate of species richness
Indicator non-diatom taxa - based on published indicator values
Relative abundance of all taxa - % coverage
Number of divisions (of algae) represented all taxa - indicator of diversity
Chlorophyll a - estimate of biomass
Ash-free dry-weight - estimate of total organic material (on artificial substrate)

3.6 Assessment Performance

Various attempts have been made to evaluate different assessment systems for measuring general river quality. Coste *et al.* (1991) compared eight biotic indices and two diversity indices for diatoms using data sets collected from French, English and Spanish rivers. They found the SPI (Specific Pollution sensitivity Index) and CEC indices gave the best results. The poorer performance of the other methods were attributed to tolerance ranges being too broadly defined (e.g. Lange-Bertalot, 1979) or insufficient taxonomic coverage for the database tested (e.g. DAlpo, Wanatabe *et al.* 1988). Kelly *et* al.(1995) compared four indices and the zonation system devised by Round (1993) finding all four indices gave similar results, with SPI and GDI (Generic Diatom Index) performing best. Difficulties encountered classifying some sites into appropriate zones (for Round's 1993 system) led them to concluded that indices provided a simpler and more objective basis for biomonitoring.

According to the available literature comparative studies typically find a strong correlation between indices with the species based SPI and its simpler generic based GDI explaining a greater variation in water quality than other indices (Prygiel, 1991; Coste, 1991; Prygiel & Coste, 1993; Kelly *et al.* 1995). However, a bias to the geographical origin of indices is inevitable with the relatively localized testing apparent in these studies.

Although most diatoms indices have been developed to assess organic pollution many also respond to other impacts. The Diatom Assemblage Index for organic pollution (DAIpo) developed in Japan by Wanatabe and co-workers (1990) correlates strongly with conductivity, total nitrogen and total phosphate and has also identified salinity gradients in UK streams (Petts, G. *personal communication*). Prygiel & Coste (1993) found that SPI, CEC and GDI correlated well with ionic strength and eutrophication as well as organic loads. Whilst such findings indicate the potential for the wider application and greater diagnostic utility of diatoms for bioassessment they also highlight the difficulty of identifying the key pollutant using contemporary methods (Lobo *et al.*, 1995; Kelly *et al.*, 1995). The broad spectrum of impacts affecting river quality defines the need for management to distinguish between different types of degradation.

3.6.1 Impact specific indices

As organic pollution declines in prominence river managers have focused on other causes of river degradation and this has led to research directed towards other, impact specific indices using diatoms. In recognition of the potentially detrimental effects of eutrophication, Schiefele & Kohmann (1993) developed a Trophic Diatom Index for phosphate (TDI-P) and for phosphate and nitrate (TDI-NP) for use in German rivers. However, Kelly *et al. (1995)* found its overall performance in British rivers was poor, furthermore, it was unable to distinguish between eutrophication *per se* and eutrophication associated with organic pollution; an important distinction given the high cost of installing tertiary treatment (nutrient stripping) in sewage works. To address these shortcomings Kelly and Whitton (1995) devised an alternative Trophic Diatom Index (TDI) using a generic based framework (after the GDI of Runeau & Coste, 1988) with species identification for some key taxa. The TDI demonstrated a strong relationship with phosphorus concentrations and, by interpreting taxa present with respect to organic tolerance, the TDI can effectively distinguish the subtle difference between the two types of enrichment. Following national testing and some minor revisions to facilitate easier use and interpretation the Trophic Diatom Index provides a valuable bioassessment tool to assess eutrophication (Kelly, 1998).

Diatoms have also been used to measure the acidity of running waters although much information remains in unpublished reports. Coring (1996) summarizes the different approaches that have been taken to assess a rivers acidity to: calculation of a fixed value pH; calculation a range value of pH; classify a rivers acidity 'type'. All these approaches use empirical tolerance data for representative taxa. The Diatom Assemblage Type Analysis (DATA) represents the latter approach and has proved superior to other methods in evaluating German rivers (Coring, 1996). DATA provides a general ecological description of diversity with a species list including expected proportional representation (i.e. dominance patterns) of flora present. Using this information the site is classified into one of 6 strewn types (1 to 5 of decreasing acidity and 1 dystrophic classification for humic acidity).

Other degradation specific community changes have also been recorded for heavy metals (Rushforth *et al.* 1981), turbidity (Chessman, 1986), and salinity (van Dam & Mertens, 1993; Schiefele & Schreiner, 1991) demonstrating the further possibilities of diagnostic diatom indices although none appear to have been formalized into measurement indices or classification systems at present.

3.7 Future Developments

The cosmopolitan distribution of diatoms should facilitate their future use for the bioassessment of rivers over large geographical areas. For example, despite DAIpo's development for rivers in Japan more than 70% of diatoms collected from the River Wissey (UK) had DAIpo index values, demonstrating the potential for importing this index, with minor modifications, for successful evaluation of UK streams (Petts *et al. 1993).* Similarly Kobayasi & Mayama (1982) found that the classification system devised by Lange-Bertalot was effective in the assessment of a Japanese river despite its development in Germany. Strong floral similarities with European taxa are also apparent in the southern hemisphere (Reid *et al.,* 1995) and indicate the potential value of collating existing information on diatom flora.

With reference to the transfer of bioassessment methods, Coste *et al* (1991) stress the importance of consistent taxonomic knowledge, standardized sampling and the maintenance of quality control required for sensible integration of geographically separate data. Recent taxonomic publications such as the comprehensive treatises of Krammer & Lange-Bertalot (1986-1991) should help improve standards of identification and assists the potential for information transfer.

Large numbers of taxa in a sample can be problematic to effective biomonitoring. Restricted (habitat specific) sampling reduces identification required as assemblages are usually dominated by one or a few dominant taxa with a limited number of rare species (van Dam, 1982; Round, 1993). However, contamination presents a greater problem to site assessment, its potential magnitude is illustrated by the study of Hodgkiss and Law (1991) in their comparison of epibenthic flora across habitat types. In each case (episammon, epilithon, epipilon) they found that more than 2/3 of taxa recorded was anomalous contamination. Washing substrate removes many unattached organisms reducing this unwanted component, in addition Round (1993) suggests restricting identification to the most abundant species (Section 3.4). The fundamental problem of contamination stems from traditional identification methods using preserved cells, thus creating an inability to distinguish between viable and non-viable flora (i.e. live cells from dead, washed in, cells). However, Round (1993) recently published a taxonomic guide for the identification of live diatoms, representing an important advance that circumvents this problem with the potential to greatly improve their utility as tool for river bioassessment.

A higher level of identification

Higher levels of identification requires less expertise and facilitates rapid site assessment. Generic level based indices have compared well against indices based on species. For example Kelly *et al.* (1995) showed a strong connection between SPI scores (species identification) and GDI scores (generic identification) ($r^2 = 0.83$) in British rivers.

Coste *et al.* (1991) comparing the same indices on a larger data set (1500 sites) similarly found a strong correlation ($r^2 = 0.71$). Coste *et al* (1991) suggested that limited modification, such as the further division of genera containing species with a diverse range of tolerances, e.g. *Nitzsche,* would improve the assessment capabilities of the generic index (GDI) without greatly increasing identification required. The ambivalent goals of a broadly applicable assessment method and increased diagnostic capacity can be best achieved by the development of a reduced taxa index using simplified identification such as the GDI coupled with a limited number of additional, easily identifiable, indicator taxa to aid diagnosis of particular pollutants (Prygiel & Coste, 1993; Kelly *et al*, 1995).

Recent research efforts in France have been directed towards the development of a new 'practical index' currently being tested more intensively (Lenoir & Coste, 1996). This index was derived from a nation wide database of 1332 sites and is aimed at rapid assessment and wide scale application. A simplified taxonomic approach uses 209 key taxa with an additional 57 'matched' taxa (i.e. pairing taxa difficult and time consuming to identify in collective higher 'matched' taxonomic groups). Species and 'matches' are classified into abundance classes (low, average, high) determined separately for each species (thus circumventing a problems highlighted by Cox (1991) caused by size bias, where smaller taxa can be more numerically abundant by virtue of their small size) and ranked by probability occurrence against general water quality (defined by BOD, orthophosphate, nitrate, ammonia, chlorine and dissolved oxygen) based on empirical information. Initial tests have shown the practical index compares favourably against other indices (Lenoir & Coste, 1996). Kelly (1998) similarly made use of a reduced taxonomic assemblage in their development of the TDI developed from the GDI of Runeau & Coste (1988).

Biotic indices that include taxa tolerant to a broad range of conditions can present ambiguous results and confound a diagnostic interpretation of results (Round, 1993; Coste *et al*, 1991; Petts *et al.* 1993). Kelly *et al.* (1995) felt the inclusion of species, such as *Nitzschia palea* which thrives under high levels of both nutrients and/or organic pollution, is misplaced in a nutrient index precluding accurate evaluation. Whilst many flora will be sensitive to a range of impacts diagnostics require unequivocal stress-response patterns. The need to interpret Kelly's TDI (1998) alongside organic tolerance information highlights the difficulties of assessment using single indices and indicates the potential benefits of a multimetric approach. The ability of multimetrics to combine independent assessment measures (Section 2.9.5) could be used as an 'OK' signal to potential impairment with more detailed comparative analysis of the cordposite metric's (as used by Kelly) to diagnose specific problems. According to questionnaire returns, researchers using diatom based analysis are able to diagnose a wide range of pollutants using (their) contemporary assessment methods indicating the potential to develop a number of different diagnostic biotic indices.

A fundamental criterion to improve application of diatom based assessment was noted in the index comparison conducted by Kelly *et* al.(1995). Where sites consistently classified as 'good' quality (NWC rating) were compared, these authors identified a spurious decrease in index scores downstream concluding the need for a reference condition in the classification to be used for comparison.

Other benthic algae

Other algal groups have also been used for bioassessment purposes and are an integral part of the Saprobic system used in several European countries (see Whitton *et al.* 1991), however, at present taxonomic guides for these groups do not exist or are not in currently available in English (Kelly & Whitton, 1995) thus limiting their value for river classification of UK streams.

4. MACROINVERTEBRATES

4.1 Introduction

Macroinvertebrates predominately inhabit the benthos of rivers, they are readily visible to the naked eye and can be retained using a net with a mesh size of 200μm (Rosenberg & Resh, 1993). Macroinvertebrates are important to lotic ecosystem functioning, by occupying an intermediate position in energy transfer from algae and micro-organisms to fish and other vertebrates. Their variety of morphological and behavioural adaptations and diversity of trophic groups has promoted extensive research to describe and test theoretical concepts and advance our understanding of river ecosystems (e.g. Vannote *et al.* 1980).

With origins as a minor part of the Saprobic system of Kolkwitz & Marsson (1909) macroinvertebrates have a long history in river classification in both Europe and North America (Metcalfe, 1989; Cairns & Pratt, 1993). Their ubiquitous presence, high diversity and ease of collection have facilitated this role. Macroinvertebrates have been a primary focus of attention in river ecosystems (Cummins, 1992) and are the most frequently used group in bioassessment (Hawkes, 1979; Hellawell, 1986; Abel, 1989; Rosenberg & Resh, 1993; Davis *et al.* 1996). Of the completed questionnaires, 72% used macroinvertebrates for assessment purposes, with more than half (55%; n=33) of returns based on macroinvertebrate analysis.

Invertebrate communities respond in a predictable manner to range of impacts. Their autoecological requirements are comparatively well known and their response to pollutants has been studied at the biochemical, behavioural, individual (indicator), population and community level (Johnson *et al.* 1993). The strong relationship between invertebrates and the terrestrial environment marks their value as integrators of riparian characteristics (e.g. Ormerod & Tyler, 1993) and land use conditions (e.g. Corkum, 1990). However, their high abundance and comparatively sedentary nature have also enabled point source pollution incidents to be accurately located (e.g. Brittian & Saltveit, 1988). Macroinvertebrates are particularly well suited for assessment of shallow rivers, which are principally regarded as benthic systems (Reice & Wohlenberg, 1993), and their advantages for river classification have been highlighted by many authors (Hawkes, 1979; Hellawell, 1986; Abel, 1989;

Metcalfe, 1989; Rosenberg & Resh, 1993). Some advantages and disadvantages of using macroinvertebrates in river bioassessment are listed in Table 4. 1.

Table 4.1 Advantages and Disadvantages of using macroinvertebrates for river classification.

Advantages
Ubiquitous almost universally present, abundant
Diverse communities - differential sensitivities
Sedentary nature - spatial analysis
Sampling well developed, tested, and cost effective
Taxonomy of most groups well known
A variety of assessment methods for data analysis can be employed
Specific response established for many taxa, well suited to experimental studies - R&D potential
Close association with sediments which are repositories of nutrients and toxins

Disadvantages
Contagious distribution, which may affect sampling and interpretation
Quantitative sampling requires many replicates
Distribution influenced by variables other than water quality (e.g. velocity, depth, substrate), although could be an advantage if interested in habitat degradation
Seasonal variations in distribution according to life history
Drift of organisms
Not sensitive to some impacts (e.g. pathogens)
Taxonomic difficulties in some groups, potential loss of information
Biogeographical position may influence interpretation

4.2 Ecology

4.2.1 Patterns in space and time

Wide scale surveys examining faunal distribution between and within river catchments have found invertebrate assemblages to be related to pH and its correlates (aluminium, calcium, hardness etc.; Wright *et al.* 1984; Ormerod & Edwards, 1987; Rutt *et al.* 1989). However, large spatial surveys of circumneutral sites have been more clearly explained by biome (e.g. Corkum, 1989). Longitudinal models, from source to mouth, have described communities as a transition of trophic groups changing in response to a shifting nutritional base (Vannote et *al,* 1980) and in terms of morphological and behavioural adaptations in response to hydraulic force (Statzner *et al.* 1988). At the local, site scale distinct patterns are evident at the mesohabitat level, i.e. pools, riffles and margins (Ormerod, 1987; Jenkins *et al.* 1984) and at the micro scale, across individual stones of the substratum (Hildrew & Edington, 1979).

The contagious or clumped distribution of invertebrates (due to microhabitat preferences) and their temporal variability (due to life history factors, see 4.3.1) has important ramifications for sampling design and statistical treatment of data. At the local scale the presence of aquatic insects is initially determined by ovipostion. Complete life cycles or duration of discrete developmental stages (i.e. instars) can range from days to years both between and within taxonomic groups according to species traits and ambient conditions (Butler, 1984). Redistribution along the river length and in the benthos/hyporheous may be driven by biotic or abiotic influences and mediated by passive or active drift and migration.

Invertebrate drift is an important dispersal mechanism and means of colonizing downstream habitats, however, for river classification it can mean the occurrence of fauna in a non-viable environments and this may confound interpretation. Propensity to drift varies between and within invertebrate groups and has been related to a variety of physical, chemical and biological factors (see Brittain & Saltveit, 1988). With an estimated average of 3.6% of the total population entering the drift each day (Townsend & Hildrew, 1976) the phenomena of drift has important ramifications for biological assessment particularly where classification is based on simple presence/absence data. Typically drift distances are less than 20m, but for certain taxa it may be longer (Allan, 1995).

4.2.2 Trophic guilds and river processes

The classification of macroinvertebrates into functional feeding groups was first described by Cummins (1973) and has since been further modified and refined (Merritt & Cummins, 1996). Functional feeding groups are based on gut contents analysis, mouth part morphology and behavioural studies (e.g food acquisition etc). Five principle categories have been defined:

1. shredders - feeding on course particulate organic matter (CPOM material > 1mm) principally derived from the terrestrial environment;
2. collectors - feeding on fine particulars organic matter (FPOM material < 1mm) subdivided to 'collector - filterers' obtaining FPOM from the water column and 'collector gatherers' obtaining FPOM by browsing surface deposits;
3. scrapers - feeding on attached periphyton;
4. piercers - feeding on macroalgae, piecing individual cells;
5. predators - feeding on prey, subdivided to 'engulfers' or 'piercers'.

Classification into trophic guilds facilitated studies of ecosystem processes and nutrient dynamics. Significant interactions are evident between functional groups, for example, Petersen & Cummins (1974) estimated that 30% of FPOM was derived from shredder processing of CPOM; such discoveries undoubtedly stimulated lotic ecology theory. Vannote *et al.* (1980) synthesized empirical and conceptual knowledge of river ecosystems in the River Continuum Concept (RCC) describing longitudinal shifts in the energy base and subsequent patterns in functional feeding groups of the invertebrate community (Section 2.4.1). According to the RCC shredders and collectors dominate headwater channels, collectors and grazers are codominant in mid-order rivers and collectors dominate the lower reaches with predators occurring at a constant relative abundance throughout the system.

4.3 Sampling

4.3.1 Variable assemblages

The timing of biological data collection is often constrained by the objectives of an assessment project (e.g. pollution response). In all cases sampling strategies should aim to maximize the efficiency of equipment (Section 2.7) whilst maintaining assessment consistency. In some locations temporal variability of invertebrate populations requires samples to be collected within defined temporal intervals (e.g. Ohio EPA, 1987) or interpreted with reference to season (e.g. Barbour *et al.,* 1996). Studies in the UK have found less extreme variation although spring is regarded as the optimum sampling time when diversity, and thus potential ecological information, is at a maximum (e.g. Furse *et al,* 1984; Ormerod, 1987; Pinder *et al,* 1987).

Invertebrate samples can be collected in two ways:

4.3.2 Natural habitat

Local conditions such as depth, current velocity and substratum affect the distribution of invertebrates and can restrict the use of some methods. Sampling techniques differ in their bias with respect to optimal operating conditions and the portion of the community they collect. However, selectivity can be exploited to increase efficiency when collecting specific faunal groups (e.g. Gislason & Gardarsson, 1988). Merritt *et al.* (1984), Peckarsky (1984) and Winterbourn (1985) review various methods for sampling macroinvertebrates.

Simple, low cost equipment that is easy to transport has made the kick net (or pond net) a favoured sampling device (at least 72% according to questionnaire returns; Wright *et al.* 1984; Resh & Jackson, 1993; Bailey *et al.* 1996; US EPA, 1997). Kick nets have compared favourably with other sampling methods obtaining a greater diversity of invertebrates than other methods (Extence, 1987). Furthermore, tests of operator variance have been found to be well within acceptable levels (Furse *et al.* 1981).

4.3.3 Quantitative sampling

Genuine quantitative sampling must be defined by both area and time, however, the logistics of a strictly defined sampling area can limit the versatility of equipment. Resh & McElravy (1993) provide a detailed assessment of the variety of methods that have been employed in quantitative classification projects. According to questionnaire returns 42% of methods used are quantitative (all of which retained separate replicate samples enabling quantitative analysis), 52 % are semi-quantitative and only 6% of respondents use non-quantitative sampling. Of these, 76% quantify sampling by restricting sampling time and 40% sample invertebrates from one habitat type; 24% of assessment techniques quantify by both time and area. This information contrasts with the published literature where most contemporary bioassessment programs employ qualitative sampling to reduce time and costs (Metcalfe-Smith, 1994). Semi-quantitative data can be readily obtained by imposing a time limit on sampling duration.

4.3.4 Sampling location

Samples should typically be collected away from potentially influential features (bridges, weirs etc.) and chosen from a site representative of the stretch under investigation. Protocols often advise sampling a single habitat in an effort to maximize consistency and improve precision (e.g. Biological Monitoring Working Party, 1978). The diversity of micro-habitats in riffles typically support the highest diversity and abundance of invertebrates (Hynes, 1970) and are the considered the most suitable single habitat for assessment purposes (Pinder *et al.* 1987). Applying general criteria such as substratum size or current velocity further enhances consistency. Clearly where riffles are unavailable other sites must be sampled, however, uncommon habitats potentially harbour unique fauna and their sampling may decrease assessment precision.

In contrast to single habitat samples stratified sampling is used to collect invertebrates from several habitats, and is considered by some researchers (e.g. Resh & McElravy, 1993) to reduce variability and increase sensitivity of biological data. Multiple habitat sampling typically includes a range of flow regimes and substrata from all available habitat types (riffles, runs, pools, margins, submerged macrophyte beds, submerged wood) and may involve modified collection techniques and vigilant inspection. Since many faunal groups have restricted distributions for total resource evaluation, such as conservation assessment, multiple habitat sampling is essential (Jenkins *et al* 1984). Moreover, from the perspective of creating reference databases, a holistic approach to sampling is an important consideration as predictive models cannot extrapolate beyond the limits of the input data. Of questionnaire respondents 30% collect invertebrates from a single habitat type; 34% sample from more than one specified habitat and 36% sample all available habitats.

4.3.5 Mesh size

For sampling devices using nets, choice of mesh size will affect the accuracy of population estimates. The proportion of the benthic fauna retained depends upon net mesh size and this is an important consideration for population estimates or in the collection of certain faunal groups such as chironomids (Storey & Pinder, 1985), early instars of other insects and Oligochaeta (Learner *et al.* 1978). Mesh size needs to be considered in the context of processing as there is little point in retaining animals that are likely to be missed during sorting. However, retention of more and smaller individuals increases evaluation time - both sorting and identification. Ultimately decisions regarding mesh size are a trade off between time and accuracy. In questionnaires returned, mesh size used for macroinvertebrate sampling ranged from 250µm to 1000µm although the most commonly used size was 500µm (45% of respondents). Most rapid bioassessment techniques tend to use a coarser mesh size as population estimates are rarely used (Resh & Jackson, 1993).

4.3.6 Artificial substrate

Artificial substrates have some note worthy advantages in bioassessment programs and can be especially valuable where direct sampling is difficult. Some types of artificial substrata (e.g. Dandy-Hess sampler) present consistent, identical conditions, providing exact quantitative samples and a high degree of precision. However, such devices often bare little resemblance to the natural substrata and tend to be selectively colonized by particular faunal groups (scrapers and collector-filterers). According to questionnaire returns, 27% of practitioners use artificial substrates to collect invertebrates, although the majority (88%) also collect from natural habitats.

Rosenberg & Resh (1982) consider that most disadvantages associated with artificial substrata can be minimized by following strict guidelines regarding placement and data interpretation. However, the US Science Advisory Board (1993) extends the caution of Rosenberg & Resh advising only substrata that are representative of the natural substratum (i.e. rock baskets etc.) be used as they are more likely to yield realistic results. Cairns (1982) gives an overview of information about the use of artificial substrates, Rosenberg and Resh (1982) and Plafkin *et al* (1989) discuss some of the advantages and disadvantages of their use, summarized here in Table 4.2.

Table 4.2 Advantages and disadvantages of using artificial substrates to sample macroinvertebrates

Advantages
Sample where other methods are restricted
Standardized, quantitative sampling, high precision
Less operator skill
Convenient
Non-destructive of the environment
Selectively samples faunal groups (scrapers, collector-filters, more mobile taxa).

Disadvantages
Incompletely known colonization dynamics (spatial, temporal uncertainty)
Long exposure time to sample
Potential loss of fauna on retrieval
Unforeseen losses of substrates
2 trips required
Selectively samples faunal groups (scraper, collector-filterers, more mobile taxa)
Little resemblance to natural habitat

4.4 Processing

Sorting is a time consuming and laborious part of sample preparation and a variety of methods have been employed to speed the process (Resh & McElravy, 1993). Some projects have incorporated live sorting of field sampling, using invertebrate movements to help distinguish them from organic matter (e.g. Chessman 1995; Friberg & Johnson, 1995). A number of techniques such as elutriation, floatation, sieving and centrifugation have been used to separate invertebrates from mineral and organic matter with varying degrees of success. Dyes (e.g Rose Bengal) may be applied to increase visibility of the organisms by contrasting them to their mineral/organic background, potentially improving sorting efficiency and accuracy. Although live animals take up dye faster, dyes can be added to preserved animals allowing longer for absorption. Caution needs to be exercised to avoid missing some animals, e.g. beetles with dark elytra and cased caddis larva, that appear unchanged externally. Dyes can be cleared by acidification if necessary for identification.

4.4.1 Sub-sampling

The ability of bioassessment techniques to evaluate site conditions and provide a rapid feedback of information to river management is one of the key advantages of biological assessment. Subsampling invertebrate samples has greatly facilitated the rapid processing of samples and helped reduced costs of contemporary methods. The Rapid Bioassessment Protocols (RBPS) described by Plafkin *et al.*(1989) has lead to the adoption of subsampling strategies in many state agencies in the USA (US EPA, 1997) and is currently an area of enthusiastic debate (see Courtemanch, 1996; Barbour & Gerritsen, 1996; Vinson & Hawkins, 1996). Based on returned questionnaires, 33% of invertebrates assessment approaches employ subsampling.

The two approaches to subsampling are:

1. sub-sampling a predetermined number of organisms e.g. 100, 200 or 300 etc. (fixed count);
2. sub-sampling a fraction of total sample, by weight, volume or area (fixed fraction).

4.4.2 Fixed count

Fixed count sub-sampling is advocated in the RBPs of Plafkin *et al.*(1989), is commonly used throughout the USA, and is the most common sub-sampling method (55% according to questionnaire returns; US EPA, 1997; Resh & Jackson, 1993). The desired number of organisms are typically obtained by randomly sub-sampling incremental fixed fractions of the total sample until the predetermined number of animals is attained (but see Chessingtion, 1995). For example, Marchant (1989) recommends a box (35x35x10 cms) divided into 100 equal cells in which the entire sample is placed and shaken to evenly distribute it, cells are then randomly sampled until the required number of organisms have been picked out. The random sorting of individual cells avoids any potential bias towards large or highly visible animals. Barbour & Gerritsen (1996) caution that the number of organisms collected in each cell during such cell by cell approach should not exceed 20% of the target.

4.4.3 Fixed fraction

Fixed fraction sub-sampling may use area, volume or weight depending on the composition of the sample and the preference and judgement of the operator. To use the above example of Marchants' box the same preparation applies but the end point is reached once a fixed fraction has been sorted i.e. a predetermined number of cells regardless of the number of animals collected. An advantage of fixed fraction sub-sampling is its quantitative nature thus, providing the initial sample was quantitative, estimates of population densities can be calculated by extrapolation and quantitative analysis can be employed.

4.4.4 Subsampling and loss of information

Although subsampling inevitably sacrifices some information, through loss of taxa, these tend to be rare species (Reynoldson & Rosenberg. 1996). Barbour *et al.* (1996) suggest this is seldom a problem because rare taxa are impractical as indicators and they are often omitted from data analysis to reduce noise (Faith & Norris, 1989). With regard to subsampling, decisions should be made taking into account the proposed data analysis. A cost-benefit analysis to determine the optimum number of samples (in this case the degree of subsampling) required to obtain the desired level of discriminatory resolution sufficient to detect differences is described by Ferraro's 'Power-Cost Efficiency' formula (Ferraro, 1989). However Cao *et al.*

(1998) suggest that rare species are critical to bioassessment and not detecting these species can damage the sensitivity of community based methods to detect ecological change. These authors contend that the sample size standard of 300 individuals used in EPA RBP's is too small to effectively use the information macroinvertebrate communities can provide and may greatly underestimate the differences between reference and impacted sites.

Similarly, quantitative data are often not required for biotic scores and are similarly not particularly important for analysis using classification and ordination as these methods show little difference when either quantitative (abundance) or binary (presence/absence) data is used (Furse *et al.* 1984).

4.4.5 Taxonomic level of identification

The level of identification to adequately assess aquatic resources is one of the enduring debates in bioassessment (see Resh & McElravy, 1993). Thirty three percent of questionnaire returns used species identification, with most using variable identification, typically leaving the time consuming Diptera at higher (i.e. family) level. Interestingly only 6% used family level identification despite the abundance of analytical methods based on family level data input. Identification to species level enables the incorporation of specific knowledge on physiological requirements and known tolerances providing the maximum information for data interpretation. At higher taxonomic levels application of autoecological information may be confounded by differences between individuals within related groups (Learner *et al.'* 1978; Pinder, 1989). The generalization necessary when applying tolerance levels across higher taxonomic groups has been regarded as one of the primary limitations of family level biotic indices (e.g. Walley & Hawkes, 1996).

Nonetheless, higher levels of identification (family and order) have demonstrated their effectiveness at identifying impacts. Pinder (1989) considers analysis by the simplification of data aggregated into scores and indices to be less sensitive to taxonomic information loss, an assertion substantiated in comparisons of biotic indices using family and species level data (Hilsenhoff, 1988). Similarly classification and ordination techniques have shown few differences where family level data is substituted for species level data, indeed predictive capacity is improved at family level (Furse *et al.* 1984). The enhanced performance of predictive techniques at family level has been attributed to the (taxonomic) aggregation acting to reduce the importance of patchy distributed species (Norris, 1996).

Ultimately the level of identification employed represents a compromise between the value of increased information and the cost (time and expertise) available. Difficulties in certain taxonomic groups and resource limitations frequently lead to the use of variable levels (Resh & McElravy, 1993) presumably enabling a focus on specific, readily identifiable, sensitive groups. However, a valuable note of caution is provided by the experience of Ohio's EPA research and development team which identified *Cricotopus spp.* as an important diagnostic indicator some ten years after the initial decision was made to identify midges to genus level (Yoder & Rankin, 1995).

4.4.6 Functional feeding guilds

The identification of functional feeding guilds (FFG) adds a further dimension to data interpretation. Analysis using trophic classification is widely used in the USA and consequently FFG are typically referenced alongside taxonomic identifications (e.g. Merritt & Cummins, 1996). However, some workers suggest that invertebrates cannot be assigned into functional feeding groups without undertaking gut analysis for initial classification (e.g. Winterbourn, 1981). Of questionnaires returned, 17% of macroinvertebrate assessment techniques involve functional feeding guild analysis.

4.5 Assessment Approaches

4.5.1 Biotic indices

Three groups of indices that summarize the entire macroinvertebrate community assemblage have been used in bioassessment; diversity, similarity and biotic indices. The first two measures are unitless measures and their general application discussed in detail in Section 2.9.3. Biotic indices are specific to particular biological groups and a wide range have been developed using macroinvertebrates. A review of their evolution and differences between some commonly used macroinvertebrate indices is given by Metcalfe-Smith (1994). These include the Trent Biotic Index (TBI), Chandler's score, Chutter's Biotic Index, the Belgian

Biotic Index (BBI), the Indice Biologique of France, the Biological Monitoring Working Party score system (BMWP) and the Hilsenoff Biotic Index (HBI). Most of these indices are modifications of the Trent Biotic Index.. These macroinvertebrate indices are typically characterised by summing tolerance values of (key) invertebrate taxa and in some indices dividing by the total number of scoring taxa or the total abundance to render the score independent of sample size. Although the questionnaire found that 38% of assessment methods involved a biotic index, only 6% used this approach as the sole method for evaluation. The application of biotic indices in different countries is reviewed in Chapter 8. Whilst many different biotic indices have been devised for macroinvertebrates, their limitations for river classification purposes are generally applicable and can be discussed using a single example; the BMWP system which is widely used in the UK.

The BMWP system of the U.K. was conceived by a committee of 11 members (the Biological Monitoring Working Party) who first met in 1976. Several meetings and almost two years later they produced a report outlining the BMWP scoring system based on subjectively selected scores. The BMWP requires family level identification (except for Oligochaeta) and scores taxa from 1-10 according to their pollution tolerance. The BMWP score is based on simple presence/absence data and is calculated by summing the scores of all families recorded. Since BMWP scores increase as diversity increases, by the addition of more scoring taxa, thus a score increases as sample size increases i.e. through replication or effort (Armitage *et al.* 1983). The Average Score Per Taxa (ASPT) is the product of dividing the total BMWP score by the number of taxa contributing to the score, thus expressing it in proportion to the number of contributing taxa and rendering it independent of sample size. Score expectation changes depending on sample location (riffle, margin or pool; Armitage *et al.* 1983). Nonetheless, score expectations also change with natural faunal variation associated with different river types (i.e. baseline water quality etc.) and longitudinal, downstream changes and need to be interpreted accordingly (Wright *et al.* 1989).

A frequent criticism of biotic scores is the subjective allocation of taxa to tolerance categories. For example, a recent study by Walley & Hawkes (1996) used a large data set and employed statistical analysis to generate objective BMWP scores. They compared these scores to the original subjectively allocated categories and found that the scores of many taxa were either too high or too low in the original system. Also, the collective grouping of species information to higher taxonomic levels (e.g. families) compromised index performance (Walley & Hawkes, 1996).

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This is particularly apparent for the low scores allocated to chironomids and oligochaetes which occur across a range of water quality (Pinder & Farr, 1987). Many indices were devised when organic pollution was the primary concern of water quality impairment. For example, the family biotic index (FBI) developed by Hilsenoff (1988), was based on family level identification (Hilsenhoff, 1988). The FBI defines invertebrates with a value of 0 as extremely sensitive to water quality degradation through to invertebrates with a value of 10 as insensitive (or tolerant) to organic pollution. The FBI score is calculated by multiplying the number in each family by the tolerance value allocated, summing the products and dividing by the total number of individuals in the sample:

$$FBI = \left(\sum_{n=l}^{n=j} A_j S_j\right) / n$$

where: A_j = number of taxa of family j

S_j = FBI tolerance value for family j

n = total number of arthropods in sample.

On the basis of these scores sites can be placed into water quality classification groups (Table 4.3)

Table 4.3. Classification scheme for river quality based on Hilsenhoff's FBI (from Hilsenhoff, 1988)

Family Biotic Index	Water Quality	Degree of Organic Pollution
0.00-3.75	Excellent	Organic pollution unlikely
3.76-4.25	Very good	Possible slight organic pollution
4.26-5.00	Good	Some organic pollution probable
5.01-5.75	Fair	Fairly substantial pollution likely
5.76-6.50	Fairly poor	Substantial pollution likely
6.51-7.25	Poor	Very substantial pollution likely
7.26-10.00	Very poor	Severe organic pollution likely

Many of these organic pollution indices are still widely used although their ability to address the wider problems of river assessment may be limited (Metcalfe-Smith, 1994). However, the FBI has found wide applicability in the US as it has proved sensitive to other forms of degradation (Resh & Jackson, 1993).

4.5.2 Condition measures

Invertebrates often display morphological deformities under environmentally stressful conditions. Lenat (1993) found mentum deformities in Chironomidae larvae to be indicative of toxic pollution, with severity of deformity proportional to degree of pollution.

4.5.3 Multimetric indices

Initially adapted to macroinvertebrates by the Ohio Environmental Protection Agency (Ohio EPA, 1987), the multimetric approach using macroinvertebrates became more widely used in the USA following the publication of the US EPA report by Plafkin *et al* (1989) entitled

"Rapid Bioassessment Protocols for use in Streams and Rivers: Benthic Macroinvertebrates and Fish". Eight metrics were used in these original Rapid Bioassessment Protocols (RBPs) (Table 4.4) and were used at three levels I, II, and III dependent on the level of sampling and identification. Among these metrics some measures such as species richness, the Hilsenoff FBI, percent dominant taxa, and the EPT index (number of taxa belonging to Ephemoptera, Plecoptera and Trichoptera), were evident explicitly or implicitly in the traditional assessment of single biotic indices. However the use of functional feeding groups added an additional dimension providing a perspective of ecosystem processes with diagnostic potential. For example the ratio of scrapers and filter feeding groups provides a reflection of the trophic dynamics and can, for example, detect increased organic loads (by increasing FPOM and thus increased abundance of collector gathers and collector filterers). Alternatively filter feeders may decrease where toxic material is bound to organic particles (Cummins, 1988). Unlike the fish Index of Biological Integrity (IBI), no individual condition metrics were included for macroinvertebrates However a number of metrics incorporated into RBP III were never adequately evaluated and further testing seems to indicate that they do not meet the standards necessary for accepting metrics (Kerans and Karr, 1994, Fore *et al.*, 1996).

Table 4.4 Eight original RBP metrics suggested by Plafkin *et al. (1989)*

1. Taxa Richness
2. Family Biotic Index of Hilsenhoff (1988)
3. Ratio of Scrapers to Collector - Filterers
4. Ratio of EPT and Chironomid Abundance's
5. % Contribution of Dominant family
6. EPT index (number of EPT taxa)
7. Community Similarity Index
8. Ratio of Shredders to Total Number of Invertebrates

Multimetric indexes measure the performance of a variety of structural and functional parameters of the invertebrate community, with individual scores being combined into an overall expression of ecological quality. Common to all multimetric approaches is the necessity to standardise metric scores into unitless values so that the range of scores for different metrics are comparable (Barbour *et al.* 1994). Two methods are common. The first approach suggested by Karr *et al.* (1986) uses both potential reference (minimally disturbed) and impaired sites and for each metric the 95% percentile is determined. Below this value the range of values is trisected or quadrisected and values of 5, 3, 1 (if trisected) given to the metric ranges (Figure 4.1a). In the second approach, Box and Whisker plots are determined for reference sites only and thus a lower percentile can be used e.g. the 25% percentile and used as the criterion to separate unimpaired from impaired (Figure 4.1b). Metric unitless scores are summed and if there are say 10 metrics then in the above example unimpaired sites would have a maximum value of 50 - this range would then be trisected to give two impaired categories.

Reference sites are typically selected by using ecoregion and appropriate sub-classes therein for different stream classes. For each metric, the sites are sorted by stream class and the metric values plotted for each class (Figure 4.2). Where individual metrics overlap, the same range will be applicable to a set of sites that represent different classes. Where metrics do not overlap (e.g. Class D and E from A and B in Figure 4.2) then separate classes are necessary to minimize the variability in each metric (Barbour *et al. 1995).*

Subsequent testing of the multimetric approaches using macroinvertebrates in states have been formulated by regional projects such as the Benthic Index of Biotic Integrity for south western Oregon (Fore *et al.* 1996), the Benthic Index of Biotic Integrity for the Tennessee Valley (Kerans and Karr, 1994), the Stream Condition Index for Florida (Barbour *et al.* 1996), the Multimetric Benthic Index for Wyoming (Barbour *et al.* 1994), and the Invertebrate Community Index of Ohio (DeShon, 1995). Many of the additional metrics contained in the above multimetric approaches have been developed to address river management issues specific to a region and to accommodate zoogeographic distributions and thus may not be applicable to other geographic regions.

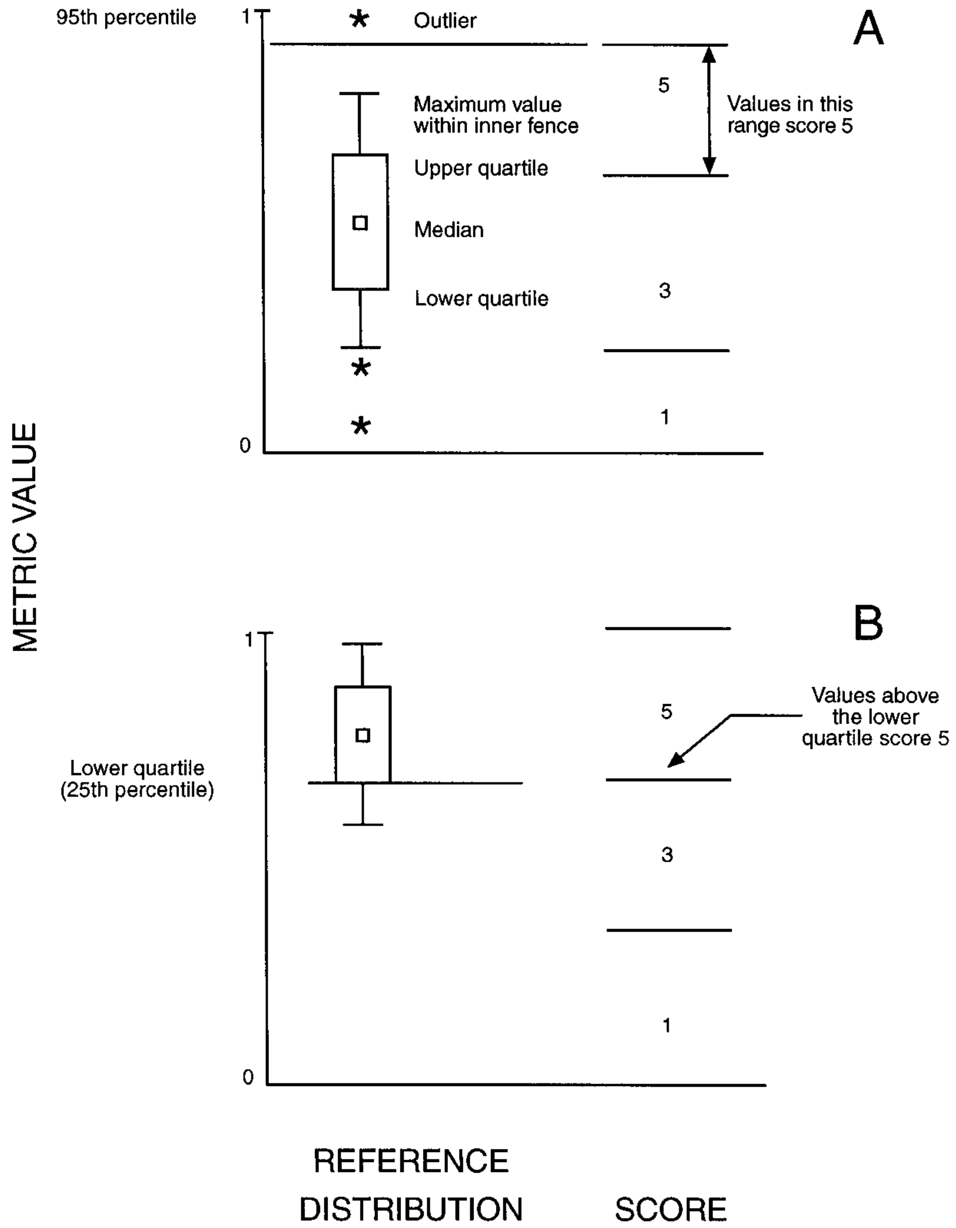


Figure:4.1 Methods for scoring metrics. Box plots show several attributes of the distributions of variables (Turkey 1977). The box marks the lower and upper quartiles of a distribution, and the length of the box is the interquartile range. The whiskers mark observations within "inner fences",defined as the quartiles ±1.5 x (interquartile range). The "outer fences" are the qualities ± 3 x (intermediate range). Observations beyond the inner fences are outliers, and observations beyond the outer fences are extreme values.

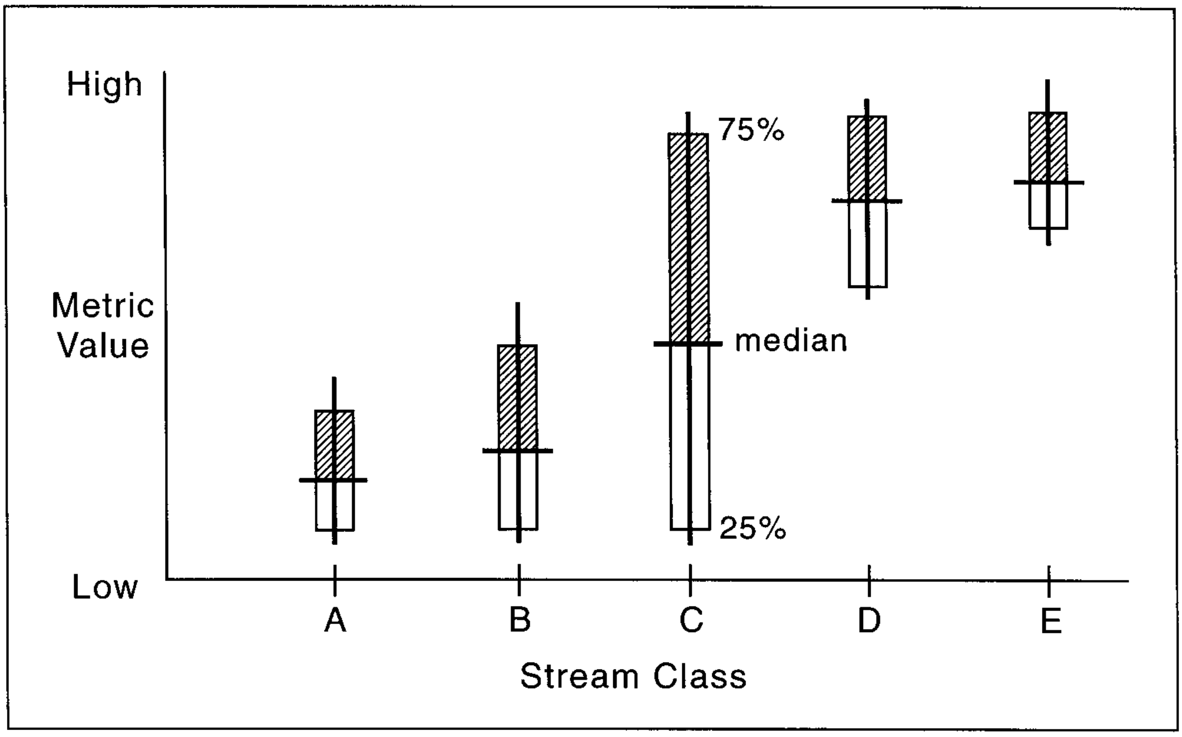


Figure:4.2 Box and whisker plots of metric values from hypothetical stream classes. (from Barbour et al. 1995)

An example of how multimetric techniques have been used for biological classification is provided through the Invertebrate Community Index (ICI) of Ohio. The ICI uses 10 metrics (Table 4.5). All metrics are structural rather than functional because of their accepted historical use, simple derivation and ease of interpretation (DeShon, 1995). Interestingly, and in contrast to other regional multimetric approaches, metrics 1-9 are derived from biological data collected from artificial substrates while metric 10 (number of EPT taxa) is based on a qualitative kick sample. Each metric receives a score of 6, 4, 2 or 0 points based upon comparison with a set of regional reference sites. Scatterplots of metric values plotted against drainage area (log transformed) are used to identify the relationship between metric score and river size and thus enable comparison against a realistic reference. The area beneath the curve is divided into quartiles that correspond to the classification scores (see Figure 4.3). Metric scores are then totalled to provide an overall quality measure with a range from 0-60. Sites can then be classified into one of four categories depending on the score (Table 4.6). The use of four categories was an arbitrary division.

Table 4.5 Metrics used in the calculation of the Invertebrate Community Index (ICI) of Ohio (DeShon, 1995).

1. Total number of taxa	6. Percent Trichoptera
2. Number of Ephemeroptera taxa	7. Percent tribe Tanytarsini midge composition
3. Number of Trichoptera taxa	8. Percent other Diptera and non-insects
4. Number of Diptera taxa	9. Percent tolerant organisms
5. Percent Ephemeroptera	10. Number of qualitative EPT taxa

Table 4.6. Scoring categories used to classify rivers based on their Invertebrate Community Index score. (DeShon, 1995)

	Excellent	Good	Fair	Poor
Invertebrate community index	>45	35-45	13-35	<13

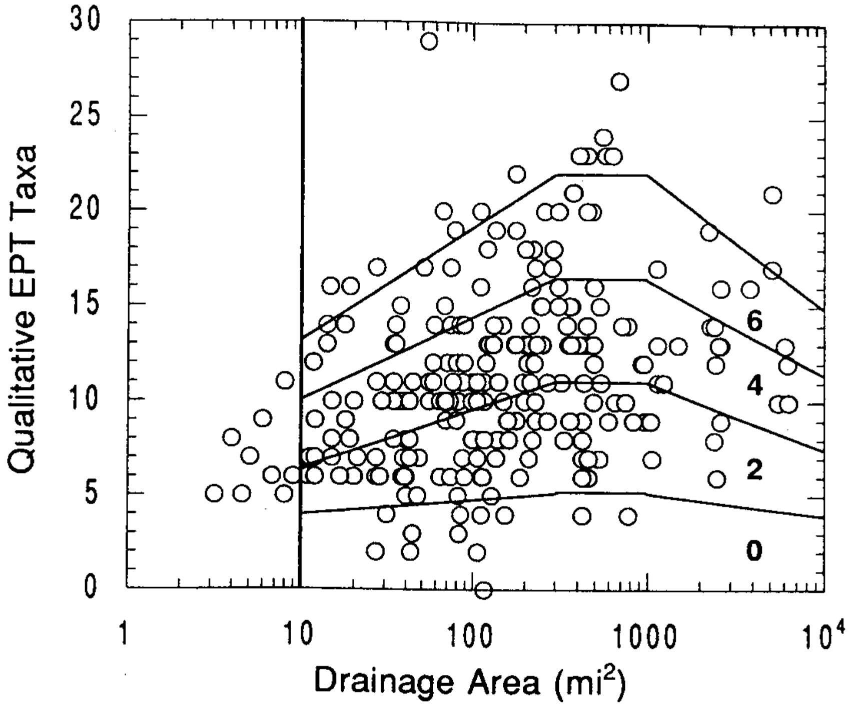


Figure 4.3. Scatter plot of qualitative EPT taxa against log transformed drainage area. Bold lines are quartiles and represent different classification scores (0,2,4,6). (from DeShon, 1995).

There has been an extensive revision of the US EPA multimetric approaches with the addition of large number of metrics to be viewed as possible candidate metrics for inclusion in protocols. These revisions are discussed further in Chapter 8.

4.5.4 Multivariate techniques and predictive approaches

Macroinvertebrates were used in a predictive assessment technique for Utah in the USA by Winget & Mangum (1979). Their simple statistical model enabled them to predict a biotic score, the Biotic Condition Index (BCI), based on substrate size, sulphate and alkalinity. The predicted BCI was then compared to a BCI calculated from a macroinvertebrate sample to assess river quality. Although this approach has been used in other states, it has not been widely applied in the USA.

More recently, aided by computer technology, multivariate statistics have increasingly been used to describe patterns of invertebrates communities in relation to measured environmental parameters (Wright *et al.* 1984; Ormerod & Edwards, 1987; Corkum 1989; 1990). RIVPACS (River Invertebrate Prediction and Classification System) uses multivariate techniques to predict the expected invertebrate fauna using a suite of environmental variables by reference to a database of 614 'pristine' river sites (Wright *et al,* 1997). The capacity to predict site specific fauna provides an objective reference for river classification. The database was generated from invertebrates collected from all available habitats over 3 seasons (spring, summer and autumn) at the river sites. Seasonal data was combined and classified using TWINSPAN into 35 distinct groups with a minimum of 6 sites in each group (Wright et *al* 1997).

To predict faunal assemblages, 11 environmental variables are used to position a test site in ordination space - ordination space as defined, and occupied, by the reference data base - using multiple discriminant analysis (MDA). Once positioned in ordination space the proximity of a test site to adjacent TWINSPAN groups (measured to group centroids) and the size of respective TWINSPAN groups (number of group members) is used to determine the mathematical probability of its membership to that group (i.e. a proportional weighting). Membership probability is calculated for all neighbouring groups. Species prediction is calculated by combining the frequency of occurrence of each species within a (neighbouring) TWINSPAN group with the probability of a test sites membership of that TWINSPAN group (Wright, 1995). This procedure is used for each species common in all neighbouring groups. A percentage of probability occurrence for a species at a test site is calculated by the combined probabilities from all adjacent neighbouring groups. Results are presented by listing taxa alongside their respective probability occurrence. RIVPACS can also be used to predict BMWP and ASPT scores. These predicted taxa or scores can be compared with those observed at a test site. A ratio of O/E is determined where O = observed taxa or BMWP/ASPT values and E is expected taxa or BMWP/ASPT scores. In general the O/E ratio should be around unity given no impairment with lower scoring sites indicating a potential impact and high scoring sites as potentially of conservation interest or possible slight organic enrichment at nutrient poor upland sites.

Although RIVPACS was developed in the UK, similar approaches are being applied or modified for use in Australia and a number of European countries (see Chapter 8). In the

USA, the US Geological Survey (USGS) are responsible for a National Water-Quality Assessment (NAWQA) Program implemented in 1991. This involves past, ongoing or proposed investigations of 60 of the most important river basins in the country (study units) and more than two thirds of the Nation's freshwater use occur within these basins. National synthesis of data analyses is a major component of the programme (Wallace *et al.* 1997). This programme is discussed in more detail in Chapter 8. Ordination of invertebrate assemblages is used along with physical and chemical characteristics to identify community types that may be associated with water quality impairment and site groups as a result of physicochemical variables. Reynoldson *et al* (1995) use multivariate approaches to predict the biological state of sediment in the near shore areas of the Great Lakes.

4.6 Diagnostic Attributes of Macroinvertebrate Approaches

Although macroinvertebrate community analysis has demonstrated its ability to indicate a wide range of ecological impacts, few impact specific (diagnostic) indices have been developed. Most contemporary indices are based on general 'pollution' tolerance (e.g. BMWP) although they were devised at a time when organic effluents were the most prominent form of ecological impairment and they are perhaps most effective at detecting organic pollution. Nevertheless, some impact specific indices have been developed. Rutt *et al.* (1989) developed a dichotomous indicator system that could be used to classify rivers in relation to acidification and identify those rivers susceptible to acid impacts. Macroinvertebrates have also proved their value as indicators of toxic conditions, especially metal pollutants, although this work has generally focused on single species indicators (Johnson *et al.* 1993b), or chironomids (Lindegaard, 1995).

The functional approach of North American researchers in the analysis of trophic dynamics using FFG information provides a process orientated view of invertebrates communities. Functional analysis represents a hitherto disregarded aspect of community analysis which may offer additional diagnostic qualities. Significantly, the EU directive on the Ecological Quality of Surface Waters recognises functional quality as an integral component of river ecosystems indicating the importance of developing such functional analysis.

However, functional feeding group analysis for invertebrates has failed to respond as expected to specific impacts in recent tests and their value in bioassessment has been questioned (Karr, J. *personal communication;* Fore *et al,* 1996). The inconsistent response of such functional groups to anthropogenic impacts suggest that invertebrates may have been erroneously classified, this is perhaps most common where classifications have been based on taxonomic - trophic generalisations (Hawkins, C. *personal communication.),* and indicates the current, imprecise knowledge of invertebrate ecology (Fore *et* al., 1996). In addition, functional feeding group metrics may have been inappropriately applied, for example, increased sediment loads, arising from catchment activities, will only increase abundance's of filterers if sediment is of suitable nutritional quality and size (e.g. 500μm maximum for hydropsychids and simuliids) and thus they must be applied with caution.

Version III of RIVPACS recently increased its predictive capacity by increasing its database to 614 sites, re-appraising, and rejecting some, of the original sites (Wright *et al* 1997). The incorporation of the BMWP and its respective ASPT score in its data calculations provides an indication of general pollution status, however, the inclusion of other biotic indices or indicator taxa could be used to signal other types of impact and increase its diagnostic capacity. The performance of biotic scores can be greatly improved by taking advantage of computer technology and the ever expanding ecological data bases which present the opportunity to re-appraise and upgrade many indices in an objective, statistical manner (e.g. Walley & Hawkes, 1996) thus increasing their accuracy. An additional metric with potential diagnostic value rarely found in invertebrates indices (although commonly used in IBI of fish communities) is the condition metric.

5. FISH

5.1 Introduction

Fish are a popular and conspicuous element of river fauna being at the head of many lotic food webs providing an integration of river ecosystem processes. Their social and economic value to commercial and recreational fisheries have instigated many conservation and restoration programs and have provided the principle focus for past river management (Welcomme, 1992). Their contemporary political importance is reflected by their prominent role in narrative legislation which defines river quality according to the fish communities they support.

Fish kills are a dramatic indicator of an acute pollution event, the unequivocal and emotive presence of dead fish floating down river have assisted in achieving successful public prosecution of offenders (e.g. Brittian & Saltveit, 1988). However, their potential as more sensitive indicators of a range of degradation has long been recognized (Karr, 1981). Many species have been shown to respond in an predictable manner to a wide range of physical, chemical and biological impacts including habitat structure (Gorman & Karr, 1978) species introductions (e.g. Meffe *et al,* 1983) and water quality (Barnes & Hirst, 1995). As a tool for biomonitoring, fish can be analyzed at the molecular, cellular, organ, individual and population level, thus offering information over a range of temporal scales. As degradation of rivers ranges from discrete point source discharges to diffuse impacts at the catchments scale, biological damage can feature as acute or sub-lethal, chronic impacts. The ability of fish communities to reflect environmental quality over a broad range of spatial and temporal scales offers the potential for integrated assessment of river quality at a scale appropriate to management concerns.

The mobility of fish, enabling them to avoid pollutants by swimming, has caused scepticism concerning their ability to evaluate ambient quality. However, many species have a highly restricted home range and show a high degree of habitat selection (e.g. Gorman, 1987) with fish communities recovering rapidly from natural disturbance and maintaining stable populations over time (Ross *et al.* 1987; Doughty & Maitland, 1994). Furthermore, amongst the more mobile species migratory patterns are seasonally predictable with their extended spatial niche providing an opportunity to evaluate larger spatial scales and integrate catchment conditions (Plafkin, *et al.* 1989). Although there is comparatively little evidence for fish communities being used to explicitly to monitor European waters, fish community assessments in the USA form an integral part of bioassessment programs and are used in many states (US EPA, 1997). Of research institutions that return questionnaires, 18% used fish for bioassessment purposes with 12 % (n=7) of returned forms based on fish population analysis. The advantages of using fish in a bioassessment program are listed in Table 5.1.

Table 5.1 Advantages and disadvantages of using fish for river classification

Advantages
Conspicuous, with high socioeconomic value
Life histories well studied
Various trophic classes, many are high in the lotic food web - integrators of lower levels
Easy identification
Population data often readily available - historical information from fisheries
Communities are seasonally predictable, persistent and stable
Range of temporal scales (long and short life spans)
Range of spatial scales, sedentary and migratory species
Continually inhabit water.
Represent broad spectrum of tolerances to chemical, physical, biological degradation.
Fish form basis of narrative legislation - Water Quality Objectives.
Often used in laboratory studies.

Disadvantages
Difficulties in obtaining representative sample or accurate estimates of population
Variety of extraneous factors affect populations (e.g. fishing, predation)
Mobility and limited residence time of anadromous species in rivers
Relatively high manpower needed for field sampling

5.2 Ecology

5.2.1 Zoogeographic patterns

As with all bioassessment techniques the successful use of fish must be underpinned with an understanding of ecological processes that determine existing communities. The widespread success of introduced species indicates that conditions responsible for extant populations are of reduced significance in contemporary assemblages. The importance of a historical perspective is highlighted by divergent zoogeographic patterns. For example, the strong similarities in species richness, morphology and life history strategies apparent between European and western North American fauna contrast with the general ecological attributes of fish communities in eastern North America (Moyle & Herbold, 1987). This phenomena has been attributed to regional topography and subsequent conditions experienced during the Pleistocene (Moyle & Herbold, 1987). Understanding of such ecological phenomena is particularly important when considering the suitability of biomonitoring methods that have been developed in one particular zoogeographic region for use in another geographical location.

5.2.2 A spatio-temporal framework

Bayley and Li (1992) adapted the spatial and temporal framework presented by Frissel et *al.* (1986) (Section 3.4.1) as a convenient framework to consider fish ecology (Figure 5.1).

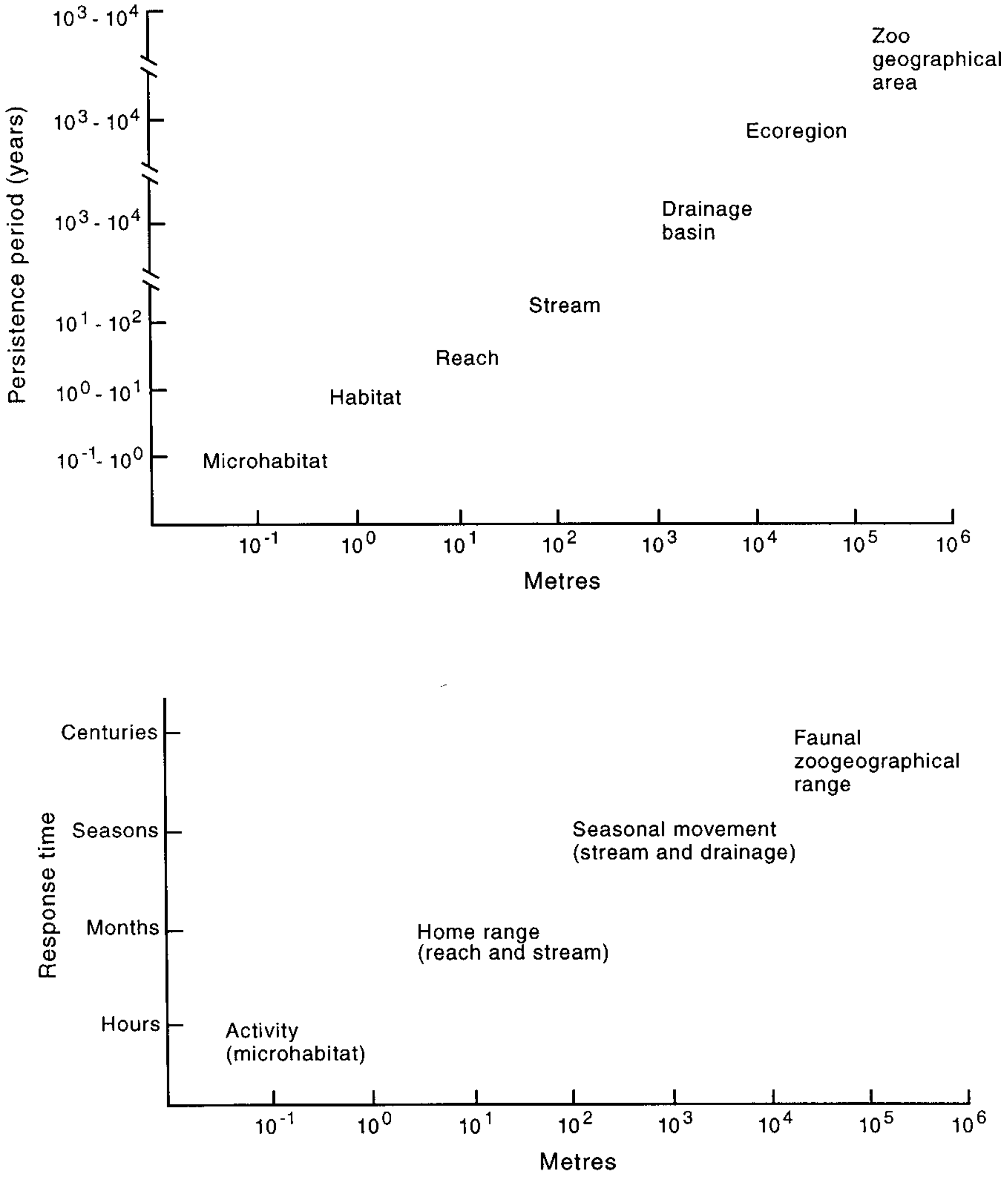


Figure:5.1 Spatio-temporal framework to consider fish ecology (from Bayley and Li, 1982)

Biological time scales pertinent to fish ecology can be measured within this spatiotemporal context, for example activity, such as feeding, generally occurs at a feeding station and thus relates to microhabitat environments and short temporal scales. At the other end of the spectrum zoogeographic range can be referred to at the catchment or biome scale and corresponds to time scales of thousands of years required for biological isolation and speciation.

5.2.3 Flow regimes

River flow is of fundamental importance to fish in river ecosystems (Hynes, 1970). The hydrological regime has a defining role in shaping channel geomorphology, habitat volume, current velocity, substratum composition, riparian vegetation and other structural features of direct importance to fish communities (Gorman & Karr, 1978; Schlosser, 1982; 1987; Milner, *et al.* 1985). The intensity and predictability of high and low flows defines the form of disturbance (Resh *et al* 1988) and directly influences fish communities (e.g. Milner et *al* 1981; Meffe, 1984; Bain *et al.* 1988). Thus different flow regimes, across different rivers, provide an environmental gradient in which biological communities exist *(sensu* Wiens, 1984). Poff and Ward (1989) classified rivers according to intensity and predictability of flow characterizing river types ranging from unpredictable 'harsh intermittent' to stable mesic ground water. Poff and Ward (1989) and Bayley and Li (1992) speculate that ecological patterns in fish communities such as increasing body size, greater diversity and a decreasing tendency to migration will occur as river discharge becomes increasingly predictable. Such ecological characteristics have important consequences for bioassessment projects affecting both measurements made and the scale of survey required for sensible evaluation. In a subsequent study, Poff and Allan (1995) examined fish communities over a range of hydrological regimes and found that these flow regime parameters were important determinants in controlling the structural and functional distribution of fish at a regional scale.

In large, unmodified rivers Junk *et al. (1989)* described the flood pulse as driving the river economy by its effect on production and nutrient cycling. They considered this seasonal inundation to be of particular importance to mobile fish communities that are able to take advantage of the ephemeral but abundant resources of the floodplain.

5.2.4 Delineating fish communities

Longitudinal gradients in physico-chemical variables and ecological communities in rivers have long been acknowledged and numerous attempts have been made to classify patterns from source to mouth (see Naiman *et al.,* 1992). Fish species have featured explicitly in some approaches, for example Huet (1959) classified rivers on the basis of slope and width into four zones (trout, grayling, barbel and bream), subsequent authors have expanded and refined this system (Hawkes, 1975). Although such an approach is clearly an oversimplification of a river continuum its serves as a useful first approximation where assessment requires a method of classification. As river size increases fish species assemblages change and taxon diversity increases (Angermeier & Schlosser, 1989). Recent work has shown that the position of a tributary in a network river system also affects species diversity. Small tributaries low down in the network system (i.e. close to a large rivers) have a higher species diversity than equivalent sized tributaries located in the headwaters (i.e. remote from a large rivers) (Osborne & Wiley, 1992). Delineation of river segments needs to incorporate such phenomena to ensure realistic interpretation and comparison of biological data.

5.2.5 Predictability in space and time

Knowledge of fish mobility is central to their use for bioassessment often explaining temporal variability at a site and assisting interpretation of site recovery monitoring by providing an understanding of colonization abilities i.e. mobility and tendency to migrate (Milner, 1994). In the UK many coarse fish species (e.g. bullheads, stoneloach, gudgeon) are known to have relatively sedentary lives and are likely to be useful indicators of site-specific conditions (Zweimuller, 1995). Others species such as chub, dace, barbel and brown trout realise larger spatial scales with spawning migrations that can take them to small tributaries some distance upstream. However, such migrations occur over limited distances and are seasonally predictable (e.g. Lelek, 1981). At the extreme end of the scale anadromous species (e.g. salmon and eels) may travel the entire length of a river provided their passage is free from chemical or physical barriers. Most studies have shown fish communities of unimpacted rivers to possess a high degree of temporal stability (Gorman & Karr, 1978; Schlosser, 1982) even when sites have experienced extreme flood (Moyle & Vondracek, 1985) or drought conditions, (Ross *et al.* 1985; Doughty and Maitland, 1994).

The importance of habitat structure to fish abundance and species diversity has been widely demonstrated (Gorman & Karr, 1978; Milner *et al,* 1985; Gorman, 1987). Some species show a high degree of habitat selection (e.g. Zweimuller, 1995) which can be strongly associated with age structure (Schlosser, 1982; Gorman, 1987). As a consequence, the relationships between fish communities and the habitat template is often employed in fisheries evaluation (Fausch et *al* 1988; Milner *et al.,* 1985).

5.2.6 Biological classification

Fish species demonstrate morphological, behavioural and trophic attributes that can be classified into functional categories. Numerous functional attributes have been classified into discrete units, for example, behavioural traits such as position in the water column can be identified as benthic or pelagic or may be segregated with greater exactness (e.g. Gorman, 1987). Similarly reproductive strategy such as bearing live young, simple lithophilous (spawning on exposed gravel), nest builder etc. have been used to separate reproductive traits (e.g. Balon, 1975). Fish species are also represented at a variety of trophic levels, although problems can occur with trophic generalists, but most species can be classified into general groups; detritivores, herbivores, invertevores and piscivores based on the origin and type of food being taken (Karr, 1981).

5.3 Sampling

Where possible sample sites should be away from natural features, such as tributaries and lakes, or unnatural modifications, e.g. culverts and dams, that may affect the 'true representation' of a fish community, unless the survey is evaluating the impact of such a feature. Season is also an important consideration. In general summer sampling reduces problems of fish movements as they tend to have a reduced home range (Lyons, 1992), furthermore, anthropogenic stresses are likely to be at a maximum as other (natural) controlling factors such as temperature and dissolved oxygen are at stress thresholds (Ohio EPA, 1987).

Sampled reaches should be inclusive of all habitat types (pools riffles, bends, backwaters, side channels, islands, debris dams). The US EPA (1997) protocol suggests that representative

100m reaches are generally sufficient to incorporate most features. Alternatively sampling area may be proportional to stream size defined by a predetermined number of stream widths long (e.g. 30-40 widths, Lyons, 1992; US EPA, 1997). Where natural barriers do not exist stop nets can be used to delineate fish populations and restrict fish movement thus with multiple samples an accurate estimation of population density can be calculated using catch depletion, regression analysis (e.g. Zippin, 1958). However, for general assessment purposes a single pass is considered sufficient to characterize a reach (US EPA, 1997).

5.3.1 General considerations

All methods of data collection are selective and the method of choice depends on the fish community under investigation, local site conditions and the objectives of the project. For anadromous species counts of fish returning to spawn, redd counts or numbers of decaying carcasses can all be used effectively although difficulties can occur due to the protracted nature of such events. In the US direct observation by snorkelling is becoming increasingly popular (MacDonald *et al.,* 1991). Seine netting and 'drag netting' a reach are also used but can be seriously limited in some rivers due to local conditions. Any survey that employs more than one method will likely yield more species than any single method, however, in general electrofishing is considered to be the single, most effective, non-destructive method for fish sampling (US EPA, 1997). All questionnaire respondents using fish assessment sampled by electrofishing.

Electrofishing efficiency is affected by depth, width, turbidity and general physical and chemical properties of the river. In addition, fish may vacate their normal habitat during high flows and sampling is, therefore, recommended only during normal flow conditions. Visibility is critically important during electrofishing; night fishing is discouraged and the use of Polaroid sunglasses to decrease surface reflection and assist visibility has also been suggested (US EPA, 1997). Although electrofishing is biased towards the collection of bigger fish this is not regarded as a problem as small fish are not considered in most data analysis methods (see below; Karr, 1986). However, these are the ones least likely to be stocked and therefore most likely to be useful in ecological assessment. Reynolds (1983) provides general considerations and precautions for electrofishing different river systems.

5.4 Data Collection

Information collected from sampled fish (e.g. biomass, fork length, condition etc.) depends on the objective of the study. Karr (1986) considered the collection of biomass data time consuming and a disadvantage to assessment approaches that require this information. However, subsampling and batch weighing techniques can provide biomass data can be readily obtained with little extra time and effort (Ohio EPA, 1989; ESE, 1996).

Fish data are usually collected along with comparatively detailed habitat information. Numerous methods have been employed to evaluate populations in the context of environmental conditions and several standardized approaches have been used to aid interpretation of fish community data e.g. Habitat Suitability Index (HSI; US Fish and Wildlife Service, 1980), Qualitative Habitat Evaluation Index (QHEI; Rankin, 1989) and HABSCORE (Milner, *et al.* 1985).

5.4.1 Fisheries data

Commercial and recreational fisheries can provide valuable information at little or no cost, such information can be approximately quantified using catch per unit effort and is routinely collected by most fisheries managers. This potentially provides valuable information on species representation, size distribution and seasonal and yearly trends. However, inconsistencies in data due to socioeconomic differences including skill or experience of fishermen, selective desire (i.e. fish preference), the use of selective fishing techniques and technological differences (i.e. equipment) must be identified and incorporated as error where information is used for river quality evaluation.

5.5 Data Analysis

5.5.1 Population studies

The social and economic importance of individual fish species has resulted in many river stretches being evaluated solely in terms of its quality as a fishery, typically focusing on one or a few species (e.g. salmonids). In such cases detailed assessment is achieved by recording length, mass and total abundance of the species of interest. This can be combined with age structure (from scale annuli or length structure) to provide information on recruitment success and to generate growth curves for the local population. This type of information is typically examined in relation to the other fisheries data to assess whether the maximum growth potential is being attained. Such information can then be used to guide management on stocking strategies or habitat modifications, to optimize its productivity as a fishery.

However, such assessment provides little information on overall ecological health and integrity and largely reflects the effectiveness of a hatchery. Indeed Lyons, *et al.* (1996) suggests that stocked fish should be excluded from river evaluation projects as a stocked river may support an excellent fishery but have a low biological integrity, conversely a river can be of poor fishery quality and have a high biological integrity. Where rivers containing stocked fish are evaluated, contributions of stocked fish can be determined by examination of size distribution and comparison with local stocking records and subsequently omitted from evaluations (Lyons *et al.* 1996).

5.5.2 Abundance, biomass and diversity

Simple presence/absence, species richness, and diversity indices have all been applied to fish populations as indicators of water quality (Langford & Howells, 1977; Portt *et al.* 1986). The advantages and disadvantages of these methods are discussed in Section 2.9 and their utility for riverine fishes is reviewed by Hendrick *et al.* 1980).

A more integrated evaluation based on those methods is the Index of Well Being (Iwb) developed by Gammon (1976) which incorporates 4 measures that have traditionally been used separately: number of individuals, biomass and the Shannon diversity indices (calculated twice) by numbers and by weight.

Index of well being (Iwb) = $0.5 \ln N + 0.5 \ln B + H'_n + H'_w$

where: N = relative numbers of all species

B = relative weight of all species

H'_n = Shannon index based on relative numbers

H'_w = Shannon index based on relative weight

The Iwb is based on the assertion that least impacted sites support a greater variety, abundance and biomass of fish than impacted sites. The combination of these parameters stabilizes assessment measures, individual Iwb components often range from 20-50% whereas the variability of the Iwb itself is around 7% (Gammon, 1976). The Shannon index calculations act to balance high abundance and/or biomass of one or two predominant species.

The Iwb has performed well in response to a variety of water quality parameters (Ohio EPA, 1987) Ohio EPA (1987) modified the Iwb by reducing contributions made by highly tolerant species and found it to be more sensitive to wide array of impacts, particularly where there was a shift in community composition without a significant change in abundance, species richness or biomass. The Iwb or modified Iwb score can be divided into scoring categories for river classification and has been used for this purpose by Ohio EPA (Table 5.2; Yoder & Rankin, 1995)

Table 5.2. Site classification based on scores of the modified index of well being (Iwb) as used by Ohio EPA (Yoder & Rankin, 1995).

Site type	Exceptional	Good	Fair	Poor	Very Poor
Boating sites	>9.6	9.5	6.4	5.0-6.3	<5.0
Wading sites	>9.4	9.3	5.9	4.5-5.8	<4.5

Abundance biomass comparison

The Abundance Biomass Comparison (ABC) is a simplistic method based on theoretical assumptions and was originally designed to assess environmental stress in marine systems (Warwick, 1988). Coeck *et al* (1993) adapted the ABC to freshwater fish communities and applied it to Belgian rivers. The ABC assumes that unstressed communities are dominated by k-selected fauna with a relatively stable population, close to the carrying capacity of the environment and possessing typical species attributes of large body size and long life spans where biomass is dominated by a few large species and numerical dominants are made up from smaller sized individuals. In contrast, stressed communities are dominated by r-selected, opportunistic species with relatively short life spans and a community dominated by small species with very few larger species present. When biomass is plotted against rank species abundance in unstressed systems the biomass curve should be above species abundance and in heavy stressed environments should be below the abundance curve (see Section 2.4.7). Coeck *et* al.(1993) also tested an ABC index proposed by Meire and Dereu (1990):

$$\text{ABC index} = \frac{B_i - A_i}{N}$$

where: B_i = % dominance of species i (ranked from highest to lowest biomass)

A_i = % dominance of species i (ranked most to least abundant)

N = total number of species

According to the above assumptions the ABC index should have a positive value for unstressed systems, negative for stressed systems and around 0 for intermediate sites. Although Coeck *et al.* (1993) only tested this method of analysis on a small number of lowland Belgium rivers, principally under physically modified conditions, they found that the ABC relationship was generally consistent with theoretical predictions and was a superior indicator compared to standing crop or abundance alone. Furthermore they found the ABC index was also significantly correlated with the index of overall water quality (QI).

5.5.3 Bioindicators/Biomarkers

Bioindicators are widely used in laboratories to assesses effluent samples and have also been used in the field for river quality evaluation by the examination of wild populations for ulcerations, tumors, fin rot, skeletal deformities, parasites (Adams *et al,* 1990; 1993; Karr, 1991). Although some studies have revealed a strong correlation between specific morphological lesions and the occurrence of specific pollutants, an epidemiological study in the UK of 3382 fish across 35 sites, ranging in water quality, failed to identify significant incidents of abnormalities in relation to a NWC rating (Barnes & Hirst, 1995). However, this may have been due to the fish species selected (primarily roach), the sensitivity of biomarkers examined or insufficient detail in the water quality data.

5.5.4 The index of biological integrity

The index of biological integrity (IBI) is based on a system initially developed for streams in the North American states of Illinois and Indiana by Karr (1981). The IBI includes several levels of biological hierarchy which are formulated into 12 measures (or metrics) of the fish community, these are combined to provide an overall expression of ecological health. Metrics used are based on various aspects of community structure, trophic dynamics, abundance and individual health this requires fish populations to be grouped *a priori* into separate taxonomic and process orientated, groups (e.g. feeding guilds, reproductive guilds etc.). The IBI employs the general principle that; degradation of environmental quality by toxicity, changes in the trophic base or an alteration of habitat structure, affect fish populations directly and indirectly, thus altering the abundance, diversity and structure of the fish community (Karr, *et al.* 1986). A listing of the 12 metrics that make up the IBI are provided in Table 5.3. The major groups of biological measure used in the IBI are outlined below.

Table 5.3 The 12 metrics of the Index of Biological Integrity (Simon & Lyons, 1995)

Species richness & composition

1. Number of native fish species
2. Number of Catostomidae species (suckers)
3. Number of Darter species (Percidae)
4. Number of sunfish species

Indicator species

5. Number of intolerant/sensitive species
6. Percent of individuals that are Centrarchidae.

Trophic function

7. Percent omnivores
8. Percent insectivores Cyprinidae
9. Percent top carnivores or piscivores

Reproductive function

10. Percent hybrids

Abundance and Condition

11. Abundance per unit effort
12. Percent diseased/deformed/parasites

Structural

General and specific aspects of community structure are presented in the species richness and composition metrics. The species diversity measure (metric 1) provides an overview of community structure and is commonly used in measures of environmental quality. Karr, *et al.* (1986) restricted diversity to the number of indigenous species. Other metrics may specifically identify the number of non-native species which represent an intrinsic biological impact. Furthermore, as exotic taxa are considered more likely to do well in degraded systems data on exotic species may be indicative of additional impacts (Harris, 1995).

Structural metrics are also used to exploit known sensitivities or tolerances of taxonomic groups to specific or general environmental conditions. For example, by quantifying the representation of fish that feed and reproduce in benthic habitats and thus are sensitive to siltation and deoxygenation of these environments, metric 3 (number of darter species) is essentially a evaluation of substratum conditions. In contrast metric 4 is based on the nektonic Sunfish and reflects quality of pool habitats and instream cover.

Trophic

Trophic composition metrics are process orientated providing an overview of trophic dynamics and enabling an assessment of the energy base of a river system. In general, healthy, stable communities are associated with increasing trophic connectivity and species interactions, in contrast generalized foraging is thought to increase as degradation increases (Section 2.4.7). According to these principles the proportion of top carnivores (metric 9) distinguishes systems of high to moderate integrity, whereas the more general measure of insectivorous cyprinids (metric 8), which form the bulk of fish communities in North America, only distinguishes low to moderate integrity. Metric 7 (percentage omnivores) identifies degraded rivers. Where no adults naturally occur the number of "yearlings" are considered to be a suitable substitute as most young fish feed omnivorously (US EPA, 1997).

Reproductive function

The percentage of lithophilous species is an assessment of substratum conditions. These fish lay their eggs on exposed course gravel and display no parental care, thus a decline in substratum conditions (siltation or deoxygenation) will result in a decline of lithophilous species. The proportion of hybrids is a metric used in some calculations of IBI and supplies a measure of reproductive isolation and suitability of reproductive habitat (Simon & Lyons, 1995).

Abundance and biomass

Fish abundance presents an indirect measure of recruitment and mortality and reflects ecosystem productivity. In general abundance decreases with ecosystem degradation. However some degradation such as mild nutrient enrichment, can increase some species abundance's that are tolerant to such impacts. In consequence, tolerant fish (e.g. bluntnosed minnow) have sometimes been excluded from this metric to increase performance (e.g Lyons, 1992).

Condition

Condition metrics, formed the basis of the biomarker principle discussed above, recording the number of tumours, lesions, vertebral deformities, fin erosion, parasites etc. This evaluates the health of individual fish and offers an assessment of biological integrity at lower levels of organization, thus providing an excellent measure of subacute and chronic effects.

Interpreting the IBI

The overall IBI score summarizes the biological integrity of a site and vary from 0 to 60; the higher the score the better the quality. Individual metrics offers potential diagnosis. For example, sunfish and top carnivores do best in deep pools with extensive cover, a low scoring metric based on these fish may indicate that these habitat features have been lost. Alternatively a skewed distribution of functional feeding groups suggest a disruption to the food web. Karr, *et al.* (1986), Berkman & Rabeni (1987), Ohio EPA (1988), Steedman (1988) and Yoder and Rankin (1995) provide examples of how metrics score can be interpreted to diagnose specific types of degradation.

Regional resource managers have employed a variety of modifications of existing metrics and devised new additional measures to address local assessment needs and zoogeographic distributions (Simon & Lyons, 1995). Once a suite of metrics have been tested locally and calibrated for use multimetrics scores can be formalized into classification classes of river quality. Examples are provided of how the IBI can be used as a classification system both on a station by station comparison (Figure 5.2) or to detect temporal changes at the same site (Figure 5.3).

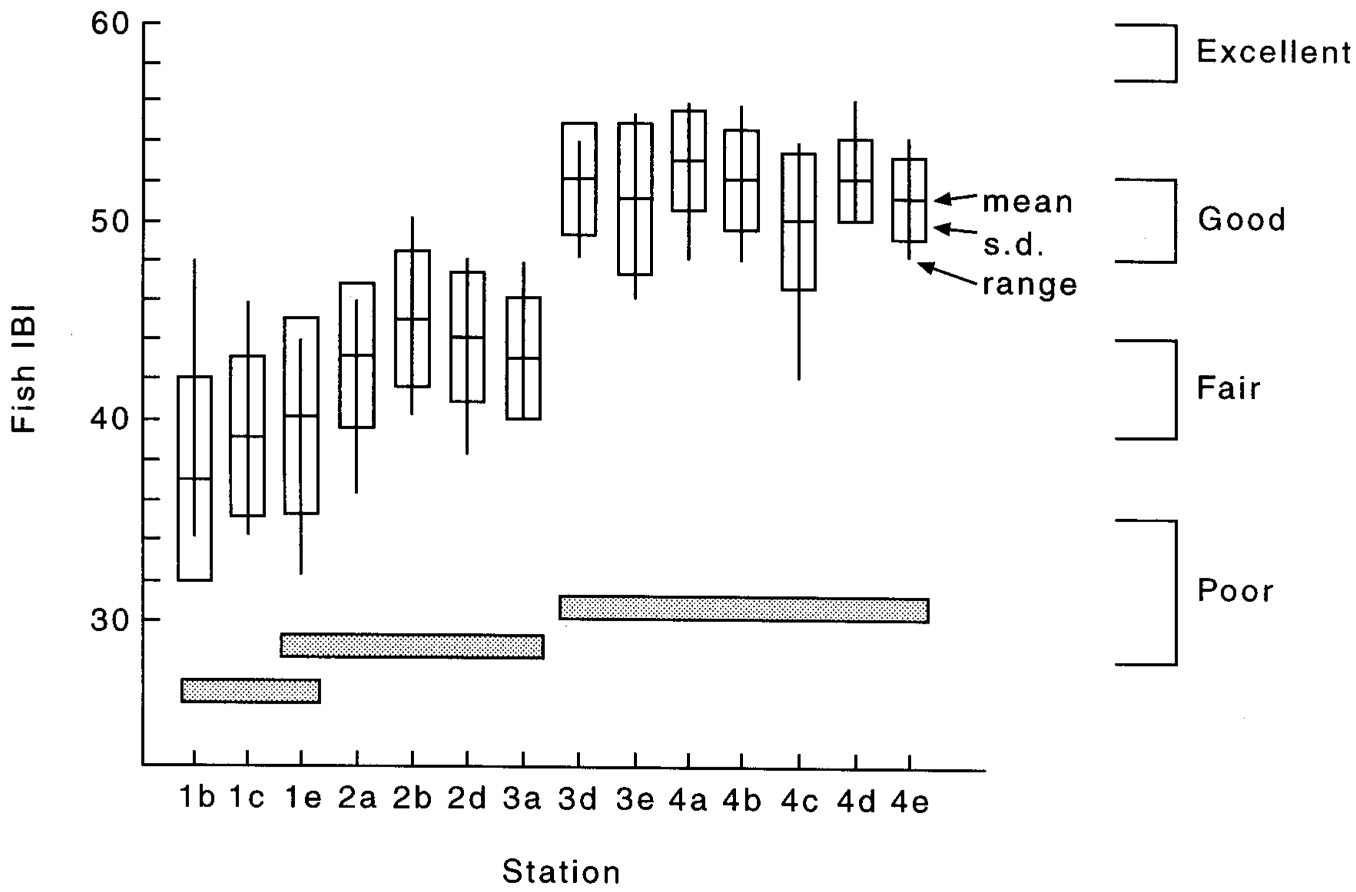


Figure:5.2 Fish IBI values for Jordan Creek, a first to third order stream in east-central illinois (from Karr et al. 1986). Higher values represent changes in the fish assemblage that reflect improved biological conditions from stations 1 through 4.

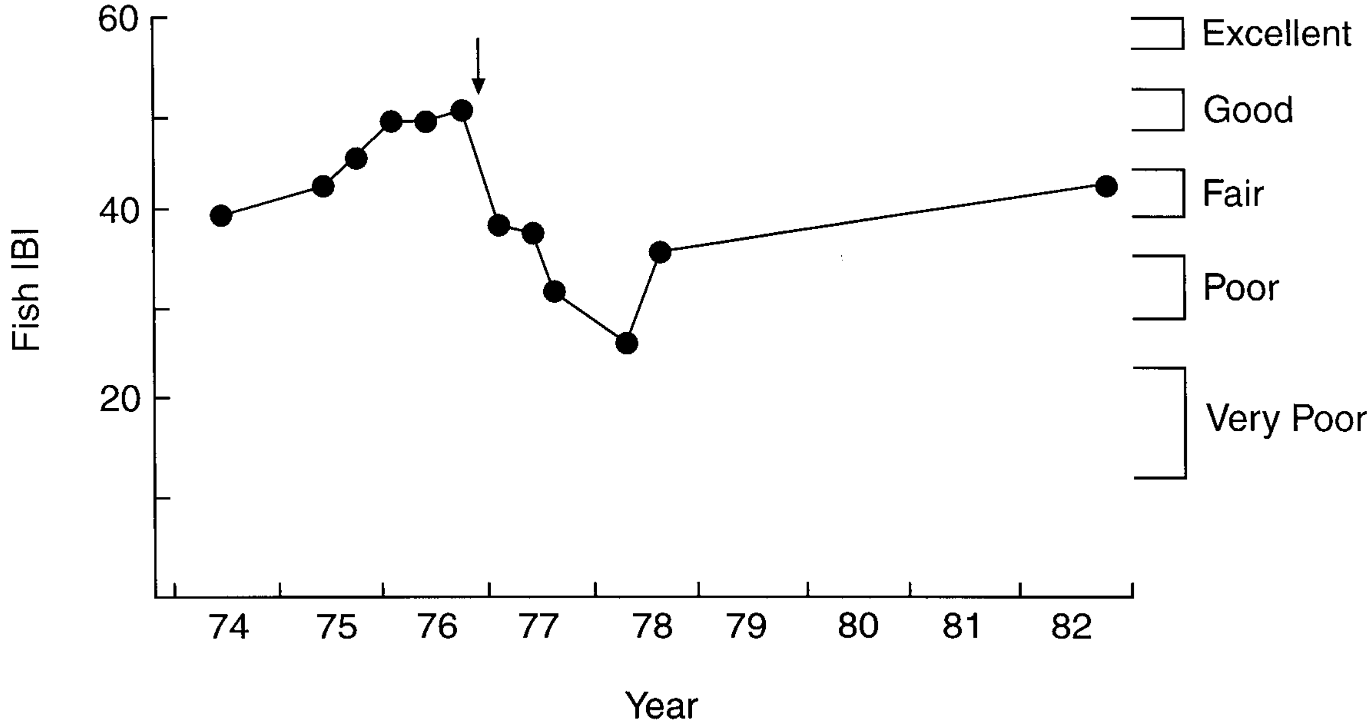


Figure:5.3 Changes in fish IBI values over time in Wertz Drain in Wertz Woods, Allen Country, Indiana. During 1974-76, Wertz Drain had relatively high IBI values for a first-order stream in an area of intensive agriculture. The channel was sinuous, pools and rifles were developed, and there were trees shading the channel. Although this site was not intentionally modified, a poorly executed bank stabilisation project upstream during 1976 transported sediment to the site. Consequently, habitat quality deterioted, as did the resident fish community. IBIs clearly trace the decline and slow improvement in stream condition over time.(from Karr and Chu, 1999).

Natural temporal variation confounds site assessment and is caused by a combination of natural environmental fluctuations and sampling error. Although the use of standardized procedures helps minimize the latter both must be acknowledged and incorporated as error in site assessment (Section 2.7). Variations in IBI score with no discernible change in biological integrity have been observed both within and between years. Typically average variations between year are low at around 10% (e.g.Lyons, 1992; Steedman, 1988; Karr *et al.* 1987), similar to within year average variance (e.g. Steedman, 1988; Karr *et al.* 1987) and are therefore not generally sufficient for a site to change its overall IBI rating. Where a site is sampled many times within a year sampling itself will impact the population causing significant mortality, emigration, skeletal deformities etc. that will reduce the IBI score.

Overall variation between and within years tends to be greater at sites with low biological integrity (Karr *et al.* 1987; Rankin & Yoder, 1990). Rankin & Yoder (1990) also found a greater variation in IBI score as stream size increased which they attributed to the greater error incurred sampling larger streams. Lyons (1992) considers a change in IBI score of 10% does not denote a significant change in biological integrity.

Where low numbers of fish are caught a reliable assessment of biological integrity cannot be made and its is advised that a river is rated as 'very poor' pending additional assessment (Karr *et al.* 1986; Lyons, 1992). Lyons *et al.* (1996) suggest that fewer than 25 individuals is insufficient to calculate IBI rating. Ohio EPA (1987) also recommend caution when interpreting IBI score based on 'moderately low' numbers of fish (i.e. less than 200) and developed a modified scoring system accordingly.

It recommended that calculations of the IBI should employ a minimum size class of fish greater than 20/25mm, due to difficulties of sampling and identification of small fish (Karr *et al.* 1986; Lyon 1992). Moreover, the high temporal variability of young fish may skew results and confuse bioassessment interpretation (Simon & Lyons, 1995). However, Simon (1989) proposed an lchthyoplankton Index specifically for young fish arguing that juveniles are particularly sensitive to certain types of degradation. Simon (1989) suggests that the proposed metrics based on taxonomic composition, abundance, reproductive guilds, known sensitivities and deformities could be used to assess nursery habitats in large rivers.

Modifications to the IBI

Individual metrics have been modified (or omitted) for use in different parts of the US to suit local zoogeographic coverage and according to differences of opinion as to fish sensitivities (Simon & Lyons, 1995). Further modification and fine tuning is also necessary to adjust for local environmental differences, for example species richness expectations change in relation to stream size. Additional delineation has used river width (Lyons, 1992), order (Karr, *et al.* 1986) or drainage basin (Ohio EPA, 1988). Also, proximity in relation a major river can be more important than local habitat structure (Osborne & Wiley, 1992; Section 6.2.4), similarly rivers that flow into lakes may have different community structure than isolated rivers (Lyons, 1992). Due consideration and appropriate modification should be given where such sites are included in comparative analysis.

The IBI was initially developed for use in warm water rivers (Max >24 ^{o}C) and applications to cool (<24 ^{o}C) and coldwater rivers (<22 ^{o}C) have generally been unsuccessful as scores tend to increased as degradation increases (Lyons, 1992). This has been attributed to the increase in temperature which accompanies most forms of degradation and presents an opportunity for the greater diversity of eurothermal and warm water species to colonize these, formally unviable, stretches. The increase in overall diversity increases in the numbers of trophic and reproductive specialists thus producing a higher IBI rating.

Consequently Lyons *et al.* (1996) propose an alternative IBI system specific for cold waters based on 5 metrics (Table 5.4). Although the reduction to 5 metrics is likely to be less sensitive to environmental change, the coldwater IBI used only metrics that where consistent across two ecoregions and were consistent regardless of strewn size. While such rigorous criteria indicate the potential for further refinement, the coldwater IBI represents a robust suit of metrics that show greater potential for use over a broad range of conditions and reflects the simpler nature of coldwater streams (Lyons *et al.* 1996).

Table 5.4. A IBI developed for use in cold (<22°C) and cool (<24°C) water rivers (after Lyons *et al.* 1996)

1. Number of intolerant species
2. Percent that are tolerant
3. Percent that are top carnivores
4. Percent that are stenothermal coolwater and coldwater species (native and exotic)
5. Percent of salmonid that are brook trout (if no salmonid captured than value = 0)

5.6 Future Developments

Despite the apparent divergence of IBI metrics in different regions and different river types the principles and concepts behind this approach are retained and their relationship to changes in water quality appear robust and flexible. River quality monitoring based on the IBI is widely used through the contrasting ecoregions of the US (US EPA, 1997) and has shown potential for use in other continents including Europe (Oberdoff & Hughes, 1992), Australia (Harris, 1995) and India (Karr, J. *personal communication*).

Despite the relatively impoverished diversity of fish fauna in the UK, resident fish populations include a range of morphological, behavioural and life history attributes and offer the possibility for a similar multimetric approach. The potential of importing such a technique is supported by apparent strong similarities in morphological and life history traits between north American and European fish populations (Moyle & Herbold, 1987) and the successful application of the IBI in France (Oberdoff & Hughes, 1992).

The life histories, spawning, feeding and habitat requirements of most species occurring in the British Isles is well know and water quality tolerance are also known for many species (e.g. Maitland & Campbell, 1992). Although there has been negligible effort in the UK to classify behaviour and trophic guilds much of the information exists in published literature and where information is lacking simple field surveys can provide additional data (e.g. Lyons *et al.* 1995). However a one off survey used to generate information may not detect important factors such as over harvesting, species introductions, blockages of migratory routes and episodic pollution thus weakening the bases of the data set. Given the diversity of ecoregions within the US and the 16+ years of IBI development and application, a large amount of information regarding the practicalities and considerations for applying this system of assessment in different biomes exists and would facilitate its development for UK rivers.

6. AQUATIC MACROPHYTES

6.1 Introduction

Macrophytes are an important component of riverine communities influencing geomorphology and water quality and providing structural habitat and food for a variety of aquatic and terrestrial animals. The value of river plants to conservation is reflected by their use to designate SSSI sites in the UK (NCC, 1989). Aquatic macrophytes are also becoming an increasingly employed tool for bioassessment in rivers (Fox, 1992). Table 6.1 summarizes the main advantages and disadvantages of using macrophytes for river bioassessment.

Table 6.1 Advantages and disadvantages of using macrophytes for river classification

Advantages
Non mobile
Bulk tissue available for analysis
Few ethical concerns in sampling
Valuable as bioaccumulators
Remote sensing potential - scale, access and time advantages

Disadvantages
Methodology less well developed than other groups
Restricted distribution for some species
Often removed for flood control, navigation etc.
Susceptible to abrasion and scouring from boat traffic and natural events like flooding
Rooted macrophytes interact strongly with sediments
Surveys restricted to certain seasons

6.2 Sampling

The best time of year to survey aquatic macrophytes in temperate climates is between mid June and September when low flows improve channel visibility and the presence of flowering structures facilitates accurate identification. Although difficulties can occur defining the vertical limits of a survey i.e. the inclusion of bank - riparian species, standard surveying methods have been published (Standing Committee of Analysts, 1987) and have been successfully employed for a variety of assessment purposes in the UK (Kelly & Whitton, 1995; Holmes, 1985). Advances in the field of remote sensing using aerial photography and satellite survey with increasing resolution, may provide a potentially interesting method that is rapid and useful in remote locations.

6.3 Data Analysis

6.3.1 Biomass

Biomass measures are frequently made using phytoplankton in lentic waters, or impounded rivers. However as the growth rate of plants is related to other factors such as flow (Westlake, 1967) and that plants can also become abundant in polluted streams in response to nutrient enrichment or a reduction in grazing pressure (Kelly & Whitton, 1995) the value of biomass as an assessment parameter is limited.

6.3.2 Index systems

A number of workers have devised indicator lists for higher plants similar to those developed for diatoms. Haslam (1987) grouped macrophytes on a scale of one to five in accordance to their tolerance to organic pollution. Haslam (1987) provides a macrophyte index based on species present, abundance, trophic status, diversity and the abundance of pollution tolerance species that combine in a calculation of 'damage rating'. All non-impairment factors, such as shade, cutting, trampling that otherwise affect macrophyte growth must be assessed in the calculation.

Harding (1987) presented a plant score based on a similar system as the BMWP for invertebrates. Plants are scored 1-10 according to their tolerance to nutrient enrichment and organic pollution and a sites quality rating is determined by summing the scores. He also advocated the ASPT version so that surveys of different sizes can be compared. A comparison of the plant score and BMWP invertebrates score correlated strongly, however, notably the plant score showed a greater potential to identify salt and zinc concentrations compared to the BMWP scores.

In the late 1970's the Nature Conservancy Council (NCC) in the UK commenced a national survey of the macrophyte flora with the aim of creating a working classification of rivers for conservation evaluation. Based on 100 rivers and over 500 sites Holmes (1983) used multivariate statistical analyses to identify distinct floral groups which then were related to environmental variables. For example, the initial separation grouped upland and lowland, acid soil sites together as lowland sites together. From these data Holmes (1995) derived Mean Trophic Rank (MTR) scores for each site. This was achieved by multiplying the Species Trophic Rank (STR) by the Species Cover Value (SCV) to give a Cover Value Score (CVS). The sum of the CVS is divided by the sum of the SCV's x 10 to give a MTR for the scale. MTR calculations have three suffixes to identify comparability of sites based on physical conditions, survey conditions and the number of scoring macrophyte species. Continuing research is determining what is a significant score difference and how this relates to water quality data determined by chemical sampling. Generally a lower MTR score is a lower quality site (see Table 6.2)

A generalized approach to the use of macrophytes to classify ecological quality in rivers and ponds of the EU is provided by Nixon *et al* (1994). This scheme depends upon scoring indicators of submerged macrophytes (bryophytes and angiosperms).

Theoretically a working classification system such as this should enable the appraisal of any river and compare its extant macrophyte community to that expected thus enabling the a quality evaluation of any river and the possibility of detecting impairment. However, because macrophytes impede flow, hinder navigation, increase sedimentation and restricting access higher plants are generally regarded as a nuisance to management and are often controlled (Wade, 1994), this makes interpretation of data difficult and restricts the potential value of macrophytes in river classification in some regions.

Potential classification of ecological quality in rivers and canals based on submerged macrophytes (bryophytes and angiosperms) (from Holmes 1995)

Indicator	High	Good	Fair	Poor	Bad
small rivers					
Presence / absence Key indicator species	present many	present some	present few	present none	absent —
medium rivers					
Presence / absense Key indicator species Depth distribution	present many >3m/MD	present some >1,0 m/MD	present few >0.5	present none <0.5	absent — —
Large rivers and canals					
Presence / absence Key indicator species Depth distribution	present many >m/MD	present none >0.5	absent — >0.5m	absent — <0.5m	absent — —

Notes MD: is maximum depth

Nevertheless, macrophytes have been used to detect enrichment in streams of France (Grasmuck *et al.* 1995; Robach *et al.* 1996)) and a rapidly applied technique for classifying streams in the Chesapeake Bay area (Small *et al.* 1996). Macrophytes may also be useful indicators of the effects of low flows (Kelly & Whitton, 1995).

7. OTHER GROUPS

7.1 Birds

Birds are highly conspicuous and relatively easily counted, potentially providing a valuable tool for river bioassessment. Although a variety of birds are strongly associated with rivers many have large ranges using lentic waters and estuaries and are migratory and are thus of reduced value in river biomonitoring (Ormerod & Tyler, 1993). Nonetheless the Dipper, *Cinclus cinclus,* is commonly found in fast flowing upland streams and occupies a restricted linear territory of 0.3-2 km long (Ormerod & Tyler, 1987) rarely moving more than 10-20 km in a generation (Tyler *et al.* 1990). Its feeding preference for mayfly nymphs and caddis larvae (Ormerod & Tyler, 1991) strongly bind it to the stream and provide a means of assessing local conditions.

Dippers have demonstrated their capacity as indicators of acidic conditions in upland Wales and Scotland (Ormerod & Tyler, 1991), and also as indicators of organic pollution showing a significant correlation with Chandlers score (Edwards, 1991 in Ormerod & Tyler, 1993). However, their restricted distribution limits their use in a broadly applicable system of classification.

7.2 Otters

Although otters are large, terminal trophic and highly charismatic animals making them ideal indicator species their decline in the 1960's was, paradoxically, largely unnoticed due to their nocturnal nature (Mason & MacDonald, 1986). Nonetheless, despite this behaviour populations can be reliably assessed by spraints (Mason & MacDonald, 1987) and their absence in otherwise expected locations has indicated acidic impacts (Mason & MacDonald, 1989), degradation of riparian vegetation and decline of water quality (Lunnon & Reynolds, 1991).

However, the absence of a species is not a reliable indicator (Cairns *et al.* 1993) and cannot be used to determine the type of impact. Indeed, many factors contributed to the decline of the otter (Mason, 1996).

8 INTERNATIONAL APPROACHES TO BIOLOGICAL CLASSIFICATION OF RIVER QUALITY

8.1 Introduction

This chapter summarizes the principal approaches for the biological classification of rivers employed by a variety of countries using the techniques outlined in the preceding chapters. A summary table has been produced (Table 8.1) which lists, where appropriate, the major biological group(s) used for classification together with any additional groups included in the final assessment. The typical number of categories produced in the classification for each country is included in this table. This review has focused principally on national monitoring programmes but more regional approaches are included where a overall national programme is not apparent (e.g. USA). In many cases, the number of classes used is arbitrary and follows older classifications using physicochemical variables or the Saprobien Index. Many systems of classification meet socio-political demands and clearly any class based system is going to lose information and may not be sensitive to all forms of impairment. Typically the number of classes designated is between five and nine, given that fewer classes may not be sufficiently sensitive enough to provide the necessary warning signal that management requires to detect degradation or improvement (Chovanec *et al.* in press). In Europe, the classification of rivers may be influenced by the draft European Union Council Directive establishing a Framework for Community Action in the field of water policy (EU Water Framework Directive 1998) which proposes classifying the ecological status of surface waters into five groups (high, good, moderate, poor, and bad). Many EU countries will have to adapt their present classifications to meet this new scheme if implemented.

Table 8.1 Summary of major biological groups and methods used in the classification of river quality for different countries

Country	Main form(s) of Water Quality Impairment	Main System	Main Group(s)	Type	Auxillary Groups	No. of Classes Described	Other Comments	References
Australia	Organic and Industrial Effluents	AUSRIVAS	Macroinvertebrates	P	-	4	Developing Predictive Models for fish and Diatoms	Smith et al. 1999
Austria	Agriculture/Organic	Ecological Integrity	Macroinvertebrates/Fish	M	Saprobic Index	7	Final class not average but lowest component	
Belgium: Flanders	-	Belgian Biotic Index	Macroinvertebrates	-	Chemical	6	-	De Pauw *et al (in press)*
Wallonia	-	Indice Biologique Global Normalise	Macroinvertebrates	I	Physical	6	Integrates IBGN with Physical structure	Vanden Bossche, (1998)
Denmark	Agriculture/Organic	Danish Faunal Index	Macroinvertebrates	I	-	7	Planning to use NORDPACS models	-
Finland	Acidification	Physico-Chemical Analysis	-	-	-	5	Planning to use NORDPACS models	-
France	Organic and Industrial Effluents	Indice Biologique Global Normalise	Macroinvertebrates	I	-	9	-	-
Germany	Organic and Industrial Effluents/Acidification	Saprobic	Macroinvertebrates	I	Diatoms	7	Additional indices for acidification	-
Italy	-	Indice Biotico Esteso	Macroinvertebrates	I	-	5	Index based on Indicator Taxa no additional measures	-
The Netherlands	Organic and Industrial Effluents	K Index	Macroinvertebrates	I	-		-	-
New Zealand	Organic and Industrial Effluents	Macroinvertebrate Community Index	Macroinvertebrates	I	-		May Develop Predictive Approach	-
Norway	Acidification	-	-	-	-		Use Raddum Acidification Index	Raddum *et al* (1988)
Sweden	Acidification	-	-	-	-		Developing NORDPACS	
U.S.A.	Multiple	U.S. EPA RBP	Macroinvertebrates	M	Diatoms; Fish	3-4	Regional Modifications	

Australia

Between 1994 and 1997, Australia developed an Australian River Assessment (AUSRIVAS) approach which is being incorporated into new national quality guidelines. The protocol involves a modified RIVPACS approach which samples a single habitat, riffles, in both spring and autumn. A composite kick sample over 10 m is taken with a 0.25 mm mesh net and the organisms are identified to family. There are 2000 reference sites used in AUSRIVAS but this is hoped to be increased to 6000 sites by the year 2000 if funding is available (Smith *et al.* 1999).

Models have been constructed for different habitats (edge and riffle) and seasons so there are 6 models for each region giving a total of 48 different RIVPACS models for Australia. These regions are not ecoregions but states and territories. The differences from RIVPACS are that (a) rare taxa are not included in the model and are eliminated if found at less than 10% of the sites, (b) 50% probabilities are used for classification, and (c) there are less impairment bands for the test sites. The width of the bands are determined by the range from the 10th to the 90th percentile of O/E scores for each of the outputs. These scores are either the taxa or the signal index (equivalent to ASPT).

For example bands for a model with 90% of reference O/E values (taxa or signal index) > 0.85

Band	Width	Level of Impact
X	> 1.15	Higher than reference
A	0.85 - 1.14	Equivalent to reference
B	0.50 - 0.84	Below reference
C	0.20 - 0.49	Well below reference
D	0 - 0.19	Impoverished

Sites allocated a band X are classed as a fail as the site could, for example, be nutrient poor but experiencing organic enrichment which has increased taxa diversity.

Austria

The assessment of ecological integrity or running waters in Austria involves in essence a form of a multidimensional approach using seven components of which four are biotic (Chovanec *et al.* in press);

1 surface and groundwater hydrology
2 habitat structures*
3 physico-chemical parameters*
4 macroinvertebrate assemblages*
5 fish assemblages*
6 (sapro-) biological water quality assessment*
7 ecological assessments

Those components marked with an asterisk can be ranked into one of seven classes to facilitate classification; the others must be incorporated by verbal description and discussion.

For the macroinvertebrate assemblage component, an ecological integrity "multimetric" index is determined using the following measures; species present, dominance, abundance and functional feeding groups/longitudinal zonation. These are subjectively compared with a site specific reference site as illustrated in Table 8.2 to provide seven classes from 1 to 4 with intermediate groupings e.g. 2-3. Similarly fish-ecological integrity classes are derived using three measures, species composition, abundance/dominance and population structure (see Table 8.3).

Following the example of Spindler (1996), Table 8.4 shows how rivers are classified using six of these seven components into one of these seven classes with the ecological integrity assigned to each class. The components are not averaged to give the overall rating but the most impaired component (i.e. closest to 4) provides the final assessment classification (see Table 8.5). Seven classes are the equivalent of the original Saprobien Index.

Table:8.2 Ecological integrity of Austrian running waters based on macroinvertebrate assemblages (from Chovanec et al. in press)

Class	Ecological integrity	Species	Dominance	Abundance	Long.zonation Funct.feed.g.
1	undisturbed	+	+	+	+
1-2	slightly disturbed	+	+/(-)	-	+/(+)
2	moderately disturbed	+	+/-	-	+/(+)
2-3	significantly disturbed	+/(-)	-	-	+/(+)
3	heavily disturbed	+/-	-	-	+/-
3-4	very heavily disturbed	-/(+)	-	-	(+)/-
4	completely disrupted	-	-	-	-

+ conformity with the site-specific reference site
(+) slight deviation from the site-specific reference state
(-) remnants of site-specific reference characteristics
- clear deviation from the site-specific reference state

Table:8.3 Fish-ecological integrity of running water in Austrian systems (from Chovanec et al. in press)

Class	Ecological integrity	Species composition	Abundance, dominance	Population structure
1	undisturbed	+	+	+
1-2	slightly disturbed	+/-	+	+
		+	+/-	+
2	moderately disturbed	+/-	+/-	+
		+	+	+/-
		+	+/-	+/-
		+/-	+	+/-
		+/-	+/-	+/-
2-3	Significatly disturbed	+/-	-	+/-
		-	+/-	+/-
		-	-	+/-
3	heavily disturbed	+/-	+/-	-
			+/-	-
3-4	very heavily disturbed	+/-	-	-
4	Completely disrupted	-	-	-

+ no or slight deviation from the reference state
+/- moderate deviation from the reference state
- large or very large deviation from the reference state

Table:8.4 Ecological integrity of the rivers of the Ill-Frutz-System. (from Spindler, 1996)

River	1	2	3	4	5	6	Ecological integrity	
Ill Ausleitungsstreke	4	3	2	3-4	3-4	2	4	completely disrupted
Ill Mündungsstrecke	3-4	3	2	3	3-4	2	3-4	very heavily disrupted
Seitenbach	3	1	1-2	3	3	2	3	heavily disturbed
Leimenbach	3-4	3	1-2	-	4	-	4	completely disrupted
Nafla Altenstadt	4	4	1-2	3	2-3	2	4	completely disrupted
Ehbachkanal	2	3-4	3	3-4	3	3	3-4	very heavily disturbed

1: surface and groundwater hydrology
2: habitat structures
3: physico-chemical parameters
4: macroinvertebrate assemblages
5: fish assemblages
6: (sapro-) biological water quality assessment

Table:8.5 Seven-class evaluation of the ecological integrity of running waters in Austria (from Chovanec et al. in press)

Class	Ecological integrity
1	undisturbed
1-2	slightly disturbed
2	moderately disturbed
2-3	significantly disturbed
3	heavily disturbed
3-4	very heavily disturbed
4	completely disrupted

Belgium

Before 1980, water quality monitoring in Belgium was the responsibility of the national government but subsequently Belgium was divided into three regions (Flanders, Brussels and Wallonia) and since 1991 water quality classification maps have been produced at the regional level. Flanders and Wallonia have been most active in this regard (De Pauw *et al.* in press). For biological monitoring, Belgium has used the Belgian Biotic Index (BBI) since 1984 which is a modification of the French Indice Biologique, itself evolved from the Trent Biotic Index (Metcalfe-Smith, 1994). Modifications mainly relate to the standardization of the sampling procedure using 3-5 minute kick samples with a pond net (300-500 µm mesh) and with identification to genus or family level. The index does not consider a single individual in a sample as a systematic unit as its occurrence could be accidental. The index is based on a scoring system using essentially the abundance of indicator taxa with most identification at the family level except for the insect orders Plecoptera, Ephemeroptera, Odonata, Megaloptera and Hemiptera which are taken to genus. Chironomidae of the thummi-plumosus group are also separated into a different systematic group. The value of the index varies from 0 to 10 with 0 the most impaired site and 10 the most unimpaired (see Table 8.6). The resulting value of the Belgian Biotic Index is assigned to one of five classes ranging from lightly or unpolluted to very heavily polluted as outlined in Table 8.7.

In Wallonia, the French 'Indice Biologique Global Normalise' (IBGN) is also calculated in addition to the BBI and although this index describes nine classes in France it has been adapted to fit into the five class designation of Belgium as outlined in Table 8.7. The IBGN is considered more precise than the BBI and thus better adapted to the local hydrological conditions of the Wallonia region (De Pauw *et al.* in press). It is proposed in this region to apply a method for integrated assessment which estimates the ecological quality of a site by combining the structure of the system with the IBGN values as outlined in Table 8.8. Results are presented on a map of the region using a three-sector circle for each site. The two joined sectors represent in the upper sector, the IBGN score and in the lower sector, the IBGN class designation while the separated third sector expresses the resultant ecological quality (Vanden Bossche, 1998) (see Figure 8.1).

Table:8.6 Standard table to calculate the Belgian Biotic Index (from De Pauw and Vanhooren, 1983)

I Faunistic groups	II	III Total numbers 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.M mentioned above are absent	3	4	5	6	7
5. Asellus or Hirudinea or Sphaeridae or Hemiptera (except Aphelocheirus)	0 all S.U.M mentioned above are absent	2	3	4	5	-
6. Tubificidae or Chironomidae of the thummi-plumosus group	0 all S.U.M mentioned above are absent	1	2	3	-	-
7. Eristalinae (=Syrphidae)	0 all S.U.M mentioned above are absent	0	1	1	-	-

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

Table:8.7 Water quality classes used in Belgium with corresponding biotic and chemical index values and colour codes (from De Pauw in press)

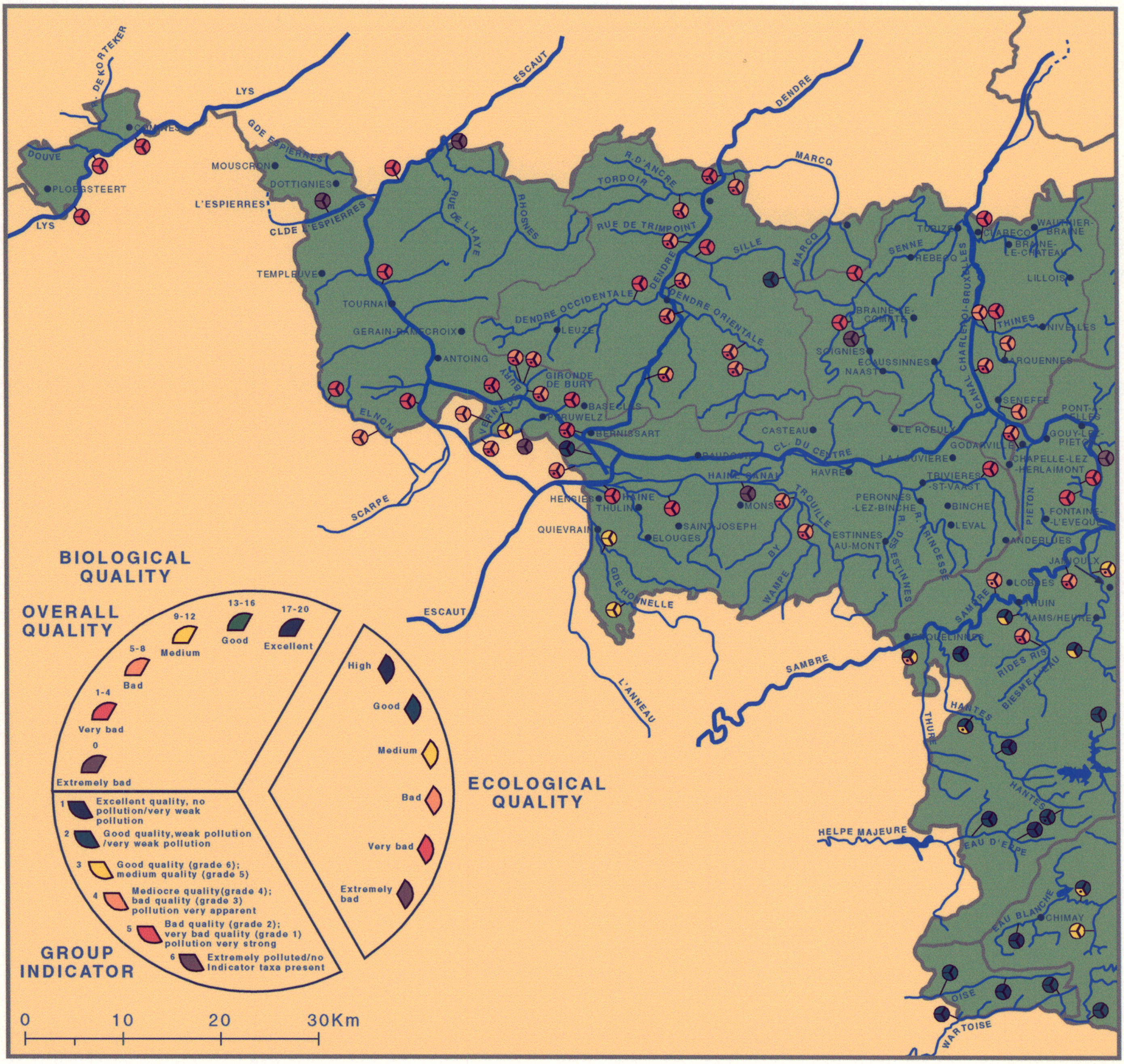
Class	Colour	Significance	BBI	IBG/IBGN[3]
I	Blue	Lightly or unpolluted[1] Very good / excellent[2]	10-9	17-20
II	Green	Slightly polluted Good / acceptable	8-7	13-16
III	Yellow	Moderately polluted Moderate / polluted	6-5	9-12
IV	Orange	Heavily polluted Bad / polluted	4-3	5-8
V	Red	Very heavily polluted very bad / heavily polluted	2-0	1-4

1 Significance of Belgian Biotic Index (BBI)
2 Significance of chemical indices (QI, BPI, IPO)
3 Significance of IBG/IBGN: see table 2b
4 Class VI refers to (B) PI values of more than 16 and is characterized as very heavily polluted

Table 8.8 Ecological quality of watercourses in Wallonia by combining system structure and IBGN values (from De Pauw *et al.* in press)

STRUCTURE OF THE ECOSYSTEM	NATURAL	SLIGHTLY MODIFIED	MODIFIED,NOT BUILT UP	PARTLY ARTIFICIAL	IRREVERSIBLY ARTIFICIAL
type of environment	deciduous forest, heath	coniferous forest, grassland	cultivation	discontinuous built-up area and/or presence of sewer	continuous built-up area, factories, navigation canal
banks	natural	mainly natural		mainly artificial	artificial
meanders	natural		natural or slightly straightened out	straightened out	straightened out
vegetation on banks	natural	not or slightly disrupted	moderately disrupted	disrupted	development not possible
BIOLOGICAL QUALITY (IBGN)	ECOLOGICAL QUALITY biological quality X structure of the ecosystem				
EXCELLENT 17-20 or indicator group 9	HIGH ECOLOGICAL QUALITY				
GOOD 13-16	GOOD ECOLOGICAL QUALITY				
MODERATE 9-12	MODERATE ECOLOGICAL QUALITY				
BAD 5-8	BAD ECOLOGICAL QUALITY				
VERY BAD 1-4	VERY BAD ECOLOGICAL QUALITY				
EXCESSIVELY BAD 0	EXCESSIVELY BAD ECOLOGICAL QUALITY				

The water course itself is not colour coded since it is considered that the classification is only valid for the sampling site. The background to the maps produced represents the predominant land use.



Denmark

Since 1993, Danish counties have employed the Danish Faunal Index which is a modification of a Saprobic System using macroinvertebrates and involves standardized sampling approaches and data analysis. However there is considerable variation in how these approaches are applied by the various counties within Denmark. The index allocates rivers into seven water quality classes. These seven categories are I, I-II, II, II-III, III, III-IV and IV and originally class I were unpolluted and IV the most impaired but recently these categories have been reversed. Macroinvertebrate samples are collected using standardized 12 kicks in 3 transects across all habitats with a 0.5 mm mesh net in both spring and autumn. A greater percentage of the water quality impairment in Denmark involves organic pollution.

Finland

Water quality monitoring programmes in Finland have concentrated principally on physicochemical analyses although biological monitoring has been included in some programmes focused on macroinvertebrates in riffles. Classification of rivers is typically into five categories; poor, passable, satisfactory, good and excellent. The water quality of rivers in Finland often deteriorates naturally due to the high content of humus. Finland are in co-operation with other Nordic countries to use predictive models (NORDPACS) based on reference sites in Sweden.

France

In France the French 'Indice Biologique Global Normalise' (IBGN) is used for classification based on macroinvertebrates. This index is very similar to the Belgian Biotic Index using a list of 9 fauna groups ranked in order of increasing tolerance to water quality impairment in rows (Table 8.9). The row selected corresponds to the most sensitive group which is represented by at least three individuals. The 12 categories or columns are determined based on the number of systematic units which may then be represented by only one individual in a sample. The index ranges from 1 to 20 depending upon the degree of impairment and sites are allocated into one of nine classes.

Table:8.9 Table to calculate the Indice Biologique Global Normalise (IBGN) of France. (from AFMOR, 1992).

Classe de variété		14	13	12	11	10	9	8	7	6	5	4	3	2	1
Taxons Indicateurs	Σt GI	> 50	49 45	44 41	40 37	36 33	32 29	28 25	24 21	20 17	16 13	12 10	9 7	6 4	3 1
Chloroperlidae *Perlidae* *perlodidae* *Taeniopterygidae*	9	20	20	20	19	18	17	16	15	14	13	12	11	10	9
Capniidae *Brachycentridae* *Odontoceridae* *Philopotamidae*	8	20	20	19	18	17	16	15	14	13	12	11	10	9	8
Leuctridae *Glossosomatidae* *Beraeidae* *Goeridae* *Leptophlebidae*	7	20	19	18	17	16	15	14	13	12	11	10	9	8	7
Nemouridae *Lepidostomatidae* *Sericostomatidae* *Ephemeridae*	6	19	18	17	16	15	14	13	12	11	10	9	8	7	6
Hydroptilidae *Heptageniidae* *Polymitarcidae* *Potamanthidae*	5	18	17	16	15	14	13	12	11	10	9	8	7	6	5
Leptoceridae *Polycentropodidae* *Psychomyidae* *Rhyocophilidae*	4	17	16	15	14	13	12	11	10	9	8	7	6	5	4
Lymnephilidae 1) *Hydropsychidae* *Ephemerellidae 1)* *Aphelocheiridae*	3	16	15	14	13	12	11	10	9	8	7	6	5	4	3
Baetidae 1) *Caenidae 1)* *Elmidae 1)* *Gammaridae 1)* *Mollusques*	2	15	14	13	12	11	10	9	8	7	6	5	4	3	2
Chironomidae 1) *Asellidae 1)* *Achetés* *Oligochétes*	1	14	13	12	11	10	9	8	7	6	5	4	3	2	1
1) Taxons représentés par au moins 10 individus - Les autres par au moins 3 individus															

Germany

Since the late 1970's the Sabropien System has been used, initially with the species indicator group of Sladecek (1973) but more recently with a revised list limited to macroinvertebrates based on statistical analysis of long term data sets. The values of the Saprobic Index are then translated into 7 water quality classifications as illustrated in Table 8.10.

Table 8.10. The seven classes of the Saprobien system used in Germany.

Class	Degree of impairment	Colour
1	virtually no impact	dark blue
2	little impact	light blue
3	medium impact	dark green
4	clear impact	light green
5	weak impairment	yellow
6	strong impairment	orange
7	severe impairment	red

Furthermore Germany has developed a classification system for the degree of acidification based on the tolerance of 200 macrozoobenthic species and is developing a classification of the trophic state of streams based on indicator values of diatoms. Recently Germany has developed a method for the ecological classification of rivers by assessing the composition of the functional feeding types, the habitat preference types, the flow preference types, the locomotion types and the composition of regions indicated by organisms (Schmedtje, U. *personal communication*).

Italy

Italy uses a modification of the Trent Biotic index called the Indice Biotico Esteso (IBE - but until 1997 termed the EBT) and scores are used to classify rivers into one of 5 categories with Class 1 being unimpaired and Class V being very poor quality. Macroinvertebrates are collected with a hand net into a pooled sample collected continuously across a transect of the stream. Stream sites are monitored every three years with samples being collected once at low flow and once at high flow. Macroinvertebrates are the only group employed in biological classification. Plecoptera and Trichoptera are identified to genera while other taxa are identified to family. A score from 0 to 14 is allocated to the IBE according to the abundance of certain key indicator taxa - for example if 26-30 individuals of *Leuctra* are collected across a transect sample then the IBE score is 12. If between 6 - 10 *Asellus* are found then the IBE value is 4. IBE values of 1, 2 and 3 translate to a class of V, 4 -5 = Class IV, 6-7 = Class Ill, 8-9 = Class II, and 10-12 = Class I. There is no comparison to reference sites with this classification system. This index was 10 years in the development process and, although a predictive approach was considered, it was not adopted.

The Netherlands

The Quality Index or K Index is used whereby indicator species are arranged in five groups *(Eristalis group, Chironomus* group, Hirudinea group, *Gammarus* group and the *Calopteryx* group), each of which is assigned a tolerance or pollution factor. The percentage of the total numbers of animals in the sample belonging to each group is then multiplied by the pollution factor and the group values are summed into an index which can range from 100 (significantly impaired) to 500 (unimpaired).

New Zealand

There is no national monitoring scheme for New Zealand and currently regional councils monitor streams and rivers locally but there is no national standard methodology. Some councils use a macroinvertebrate community index (MCI) which is a modification of the Trent Biotic Index. New Zealand are considering a trial of a RIVPACS type approach in one region but there is a lack of an extensive database for reference sites with which to build a model (Winterbourn, M. *personal communication.*). Additional reference sites will therefore have to be established.

In New Zealand the Macroinvertebrate Community Indice (MCI) and its quantitative varient the QMCI are Widely used, and a version for use with a semi-quantitative data (the SQMCI) has recently been developed. These indices were proposed by Stark (1998) to evaluate organic enrichment of stony streams, and were originally developed for use on the Taranaki floodplain. The MCI group of indices are based on the UK BMWP score system which allocates tolerance values to taxa, but cannot be used in New Zealand because of major differences in faunal composition.

A limitation of the MCI is its apparent unsuitability for use in many non-stony streams and rivers. A major reason for this is that macroinvertebrates that live naturally in fine sediments (mud, silt, sand) including such substrata in otherwise stony streams, are predominantly the same taxa that occur in increased abundance where water habitat and quality are reduced by organic pollutants. Notable in this respect are the Oligochaeta (segmented round worms) and some groups of midge larvae (Diptera: Chironomidae), taxa with low MCI scores.

Norway

Although national monitoring was common in the late 1970's and the 1980's presently there is no national scale monitoring programmes. Macroinvertebrates are the most frequently used biological parameter in monitoring Norwegian rivers which are principally impacted by acidic rainfall particularly in the south and west of Norway. The assessment is generally based on species list and the presence and relative abundance of indicator taxa although in acidic streams assessment is determined using the Raddum acidification Index placing sites into 1 of 5 groups (Raddum *et al.* 1988). Macroinvertebrate samples are collected using a composite 3 minute kick sample collected from all types of substrate in the spring.

Sweden

Recent revisions to the national monitoring programme have placed additional emphasis on the use of biological indicators to monitor water quality. Nevertheless chemical criteria are still primarily used for the classification of surface water quality. However a recent focus has been the development of predictive models (NORDPACS) using macroinvertebrates for classification along the lines of RIVPACS (Johnson, R. *personal communication*). The database comprises between 600-700 headwater streams. These predictive models may also be applicable to parts of Denmark, Finland and Norway and may include fish and macropyhte assemblages.

Switzerland

In Switzerland there is no national biomonitoring scheme and in the French speaking part of the country the French IBGN is used.

U.S.A.

There is no national scheme for the biological classification of water quality in the U.S.A. and decisions on the use of biotic approaches are made at the State level, usually by that state's Department of Environmental Conservation. Forty seven states employ some form of biological monitoring and benthic macroinvertebrate assemblages form all or part of the sampling programmes. Of these states, 25 also monitor fish assemblages but periphyton is only used regularly in a small number of states (Southerland and Stribling, 1995) (see Table 8.11.). Most states employ some form of the multimetric approach using macroinvertebrates as discussed in section 4. based on the US EPA Rapid Bioassessment Protocols (Plafkin *et al.* 1989).

Table:8.11 Status of Biological Assessment Programs (Based on the Target Assemblage Used) and Biological Criteria Programs (based on How Bioassessment Results are Used) in the United States. (from Southerland and Stribling, 1985)

	Bioassessment Target assemblage			Use of Bioassessment As Biocriteria			
State	Fish	Benthos	Periphyton	Used in Regulations	Used in Water Resource Management	Biocriteria in Development	None or Unknown
AL		✔			✔		
AK		✔				✔	
AR	✔	✔			✔		
AZ		✔				✔	
CA		✔				✔[2]	
CO		✔				✔	
CT		✔			✔		
DE		✔		✔[1]	✔		
DC		✔				✔	
FL		✔		✔		✔	
GA		✔				✔	
HI							✔
ID	✔	✔			✔		
IL	✔	✔				✔	
IN	✔	✔				✔	
IA	✔	✔				✔	
KS		✔				✔	
KY	✔	✔	✔		✔		
LA	✔	✔				✔	
ME		✔					
MD		✔				✔	
MA	✔	✔			✔		
MI	✔	✔			✔		
MN	✔	✔			✔		
MS		✔			✔		
MO		✔				✔	
MT		✔			✔	✔[3]	
NE	✔	✔			✔		
NV							✔
NH		✔				✔	
NJ		✔			✔		
NM		✔				✔	
NY		✔			✔		
NC	✔	✔			✔		
ND	✔					✔	
OH	✔	✔					
OK	✔	✔				✔	
OR	✔	✔			✔		
PA	✔	✔			✔		
PR						✔[2]	
RI	✔	✔				✔	
SC	✔	✔				✔	

Initially this was sparked by the publication of Rapid Bioassessment Protocols (RBPS) by Plafkin *et al* (1989) and in these original protocols three levels of approach (I, II and III) were proposed. Level I involved limited data generation and was essentially to discriminate obviously impacted and non-impacted sites from potentially affected sites that may require further investigation. At levels II and III, sites were classified into 3 and 4 biological condition categories respectively using percent comparability of the study site to reference site(s). Level II represented field sorting and identification to family level only. Level III encompassed laboratory sorting and identification to genera for the EPT taxa. The four categories (with % comparison to reference score) were non-impaired (>83%), slightly impaired (54-79%), moderately impaired (21-50%) and severely impaired (<17%), see Figure 8.2.

Site-Specific Study

Sampling & Analysis

Metric	Biological Condition Scoring Criteria 6	4	2	0
1. Taxa Richness[(a)]	>80%	60-80%	40-60%	<40%
2. Hilsenhoff Biotic Index (modified)[(b)]	>85%	70-85%	50-85%	<50%
3. Ratio of Scrapers/Filt. Collectors[(a,c)]	>50%	35-50%	20-35%	<20%
4. Ratio of EPT and Chironomid Abundances[(a)]	>75%	50-75%	25-50%	<25%
5. % Contribution of Dominant Taxon[(d)]	<20%	20-30%	30-40%	>40%
6. EPT Index[(a)]	>90%	80-90%	70-80%	<70%
7. Community Loss Index[(e)]	<0.5	0.5-1.5	1.5-4.0	>4.0
8. Ratio of Shredders/Total[(a,c)]	>50%	35-50%	20-35%	<20%

(a) Score is a ratio of study site to reference site x 100
(b) Score is a ratio of reference site to study site x 100
(c) Determination of Functional Feeding Group is independent of taxonomic grouping.
(d) Scoring criteria evaluate actual percent contribution, not percent comparability to the reference station.
(e) Range of values obtained. A comparison to the reference station is incorporated in these indices.

BIOASSESSMENT

% Comp. to Ref. Score[(a)]	Biological Condition Category	Attributes
>83%	Nonimpaired	Comparable to the best situation to be expected within an ecoregion. Balanced trophic structure. Optimum community structure (composition and dominance) for stream size and habitat quality.
54-79%	Slightly impaired	Community structure less than expected. Composition (species richness) lower than expected due to loss of some intolerant forms. Percent contribution of tolerant forms increases.
21-50%	Moderately impaired	Fewer species due to loss of most intolerant forms. Reduction in EPT index.
<17%	Severely impaired	Few species present. If high densities of organisms, then dominated by one or two taxa.

(a) Percentage values obtained that are intermediate to the above ranges will require subjective judgement as to the correct placement. Use of the habitat assessment and physiochemical data may be necessary to aid in the decision process.

Recommendations

Figure:8.2 Flowchart of bioassessment approach advocated for rapid bioassessment protocol III.(from plafkin et al.1989)

Three states (Ohio, Maine and Florida) currently have biological criteria incorporated into their state's water quality standards and thus these criteria are used in the enforcement of legislation. These states have a relatively long history of biomonitoring established before the US EPA Rapid Bioassessment Protocols were published and thus have developed different approaches than multimetric indices. For example in Maine "aquatic life standards" have been developed from their extensive databases which incorporate taxonomic equality, numerical quality, and the presence of pollution-intolerant indicator taxa. In Ohio, the Invertebrate Community Index, is applied as outlined in section 4.5.3. Other states, like North Carolina, have used extensive biological monitoring programmes as a basis for establishing biological criteria but these are not integrated into state water quality standards.

The US EPA RBP's are presently being revised and are under review (US EPA, 1997). The revisions include an extended list of 58 macroinvertebrate metrics suggested for possible inclusion in a multimetric approach. A number of these metrics have been those that have been regionally developed. It is now recommended that macroinvertebrates be collected with a D net using a multi-habitat approach whereas the original 1989 RBP's suggested a riffle only model. In addition metrics are suggested for fish and periphyton. Periphyton techniques involve both diatom metrics and non-diatom metrics.

Core biotic metrics are selected to provide a strong and predictable relationship with stream condition habitat and physicochemical state and to avoid redundancy. Discriminatory ability of metrics can be evaluated by comparing the distribution of each metric at a set of reference sites with the distribution of metrics from a set of known impaired sites within each site class. If there is minimal or no overlap between the distributions, then the metric can be considered to be a strong discriminator between reference and impaired conditions. Finally the scoring criteria for each metric (within each site class) from the appropriate metrics is determined from the appropriate percentile of the reference condition. The multimetric index for a site is a summation of the scores of the metrics and has a finite range within each stream class depending upon the number of metrics included and the number of divisions for each metrics - thus if 10 metrics are used with values 0 to 4 then the multimetric index can range from 0 to 40 - this range can then be subjectively subdivided into any number of categories corresponding to various levels of impairment. In most instances this is four.

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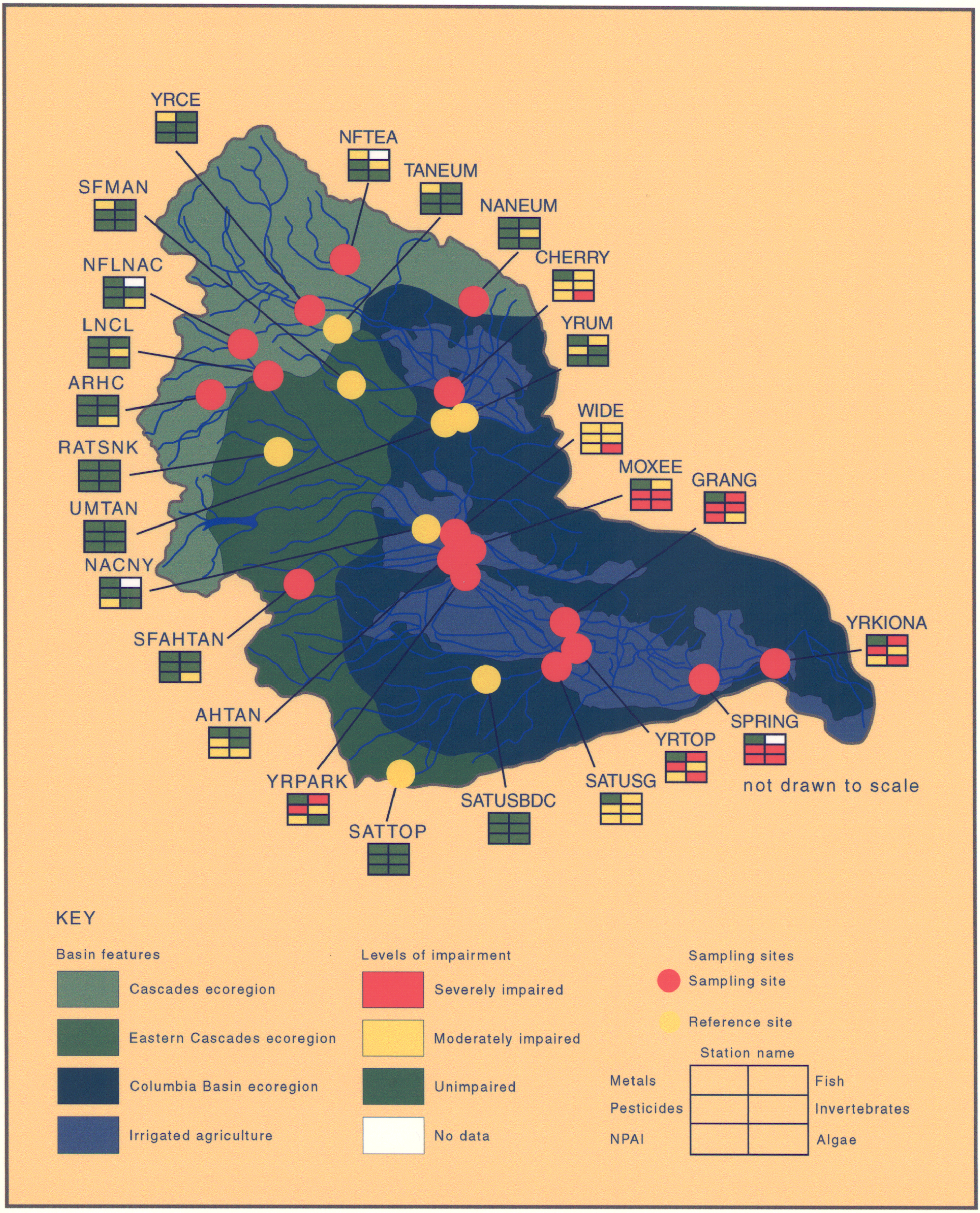

9. SYNTHESIS

9.1 Introduction

This document has comprehensively reviewed biomonitoring techniques using different groups of organisms, and discussed how these data are analysed to provide measures that distinguish different degrees of water and habitat quality which can then be used to classify sites. The aim of this chapter is to synthesize the main points from this review and examine some of the concerns and limitations of the classification approaches outlined.

9.2 Biomonitoring Approaches

This review has highlighted three main approaches to biomonitoring in the classification of river sites:

* Biotic indices;
* Multimetric indices;
* Predictive techniques.

From the sources of information available, macroinvertebrates are the biotic group most widely used in biomonitoring, followed by fish, diatoms and macrophytes. Many countries still use single indices which are essentially modifications of the Trent Biotic Index (e.g. Denmark and Italy) but a number of countries are now applying or investigating an integrated approach to site assessment and classification incorporating different biotic groups, physicochemical variables and habitat assessment (e.g. Austria and the Wallonia region of Belgium). In the context of the human health analogy, measuring a number of attributes is similar to a doctor taking a variety of medical measurements (e.g pulse, blood pressure, urine sample etc.) to maximize the available information before diagnosing the state of health of a patient. Whilst the combination of contrasting information is relatively recent to ecological science it has a established history and is accepted practice within other disciplines such as economics e.g. the FT 100 index, Dow-Jones Industrial Average. Although current multimetric techniques have typically been limited to individual biological groups (e.g. fish, invertebrates or algae), if properly calibrated the same methods could be extended to integrate information from different groups to provide an integrated ecological index.

For those countries deciding to adopt another technique than a single biotic index, then most are favouring a RIVPACS type predictive approach; e.g. Australia with AUSRIVAS and Scandinavia with NORDPACS. A comparison of multivariate approaches like RIVPACS and multimetric approaches is outlined in section 9.2.1 below.

Biomonitoring is typically undertaken at 2 levels:

1 an indication of water and habitat quality and provide a red flag to stations that may warrant further investigation with no diagnostic abilities as to the causes of impairment;
2 provide a diagnostic indication of the source of contamination or habitat impairment a broad scale e.g. acidification, organic pollution, heavy metals.

Some approaches using the same biological group may have the ability to operate at both levels (e.g. multimetric indices) but frequently the two levels are identified by different approaches or require further investigation and interpretation of the database. The use of different ecological groups complement and improve evaluation by providing greater spatio-temporal coverage and by increase assessment sensitivity and diagnostic potential. Thus a more realistic extrapolation to overall ecosystem health is achieved.

9.2.1 Multimetric v Multivariate approaches

Recently there has been extensive debate in the literature as to the comparative merit of these two now common macroinvertebrate approaches in the classification of rivers (e.g. Gerritsen, 1995, Norris, 1995).

One concern with the US EPA multimetric approach is that the division of the metric values into classes is arbitrary and subjective (trisecting or quadrasecting percentiles) and is not based on empirical data of different definitive levels of water quality impairment and no statistical analyses are applied to test for difference between classes. Karr and Chu (1999) suggest that the key is ecological dose-response curves where values of an individual metric or a multimetric index are correlated with a number of diverse anthropogenic influences (e.g. timber harvest, agriculture and urbanization) for a range of geographic areas. Karr & Chu (1999) provide examples of the relationship between four invertebrate metrics and percent area logged (Figure 9.1).

The combination of unrelated measurements requires individual metrics to be 'normalized' into unitless quantities. In most multimetric systems this is achieved by dividing scores into discrete bands. In rivers, measured quality represents a continuum of potential ecological health and the rounding errors involved in such simplification of data may result in a loss of information. Whilst classification is a helpful technique for presentation of final data, employing this method for many individual scores means rounding errors may be greatly amplified. More representative results are obtained by expressing metrics as a continuous fraction (see below).

Metric value = (Xi - Xmin)/(Xmax - Xmin)

where Xi = measured value

By retaining individual metrics as a continuous score and reserving classification for final presentation such rounding errors will be considerably reduced.

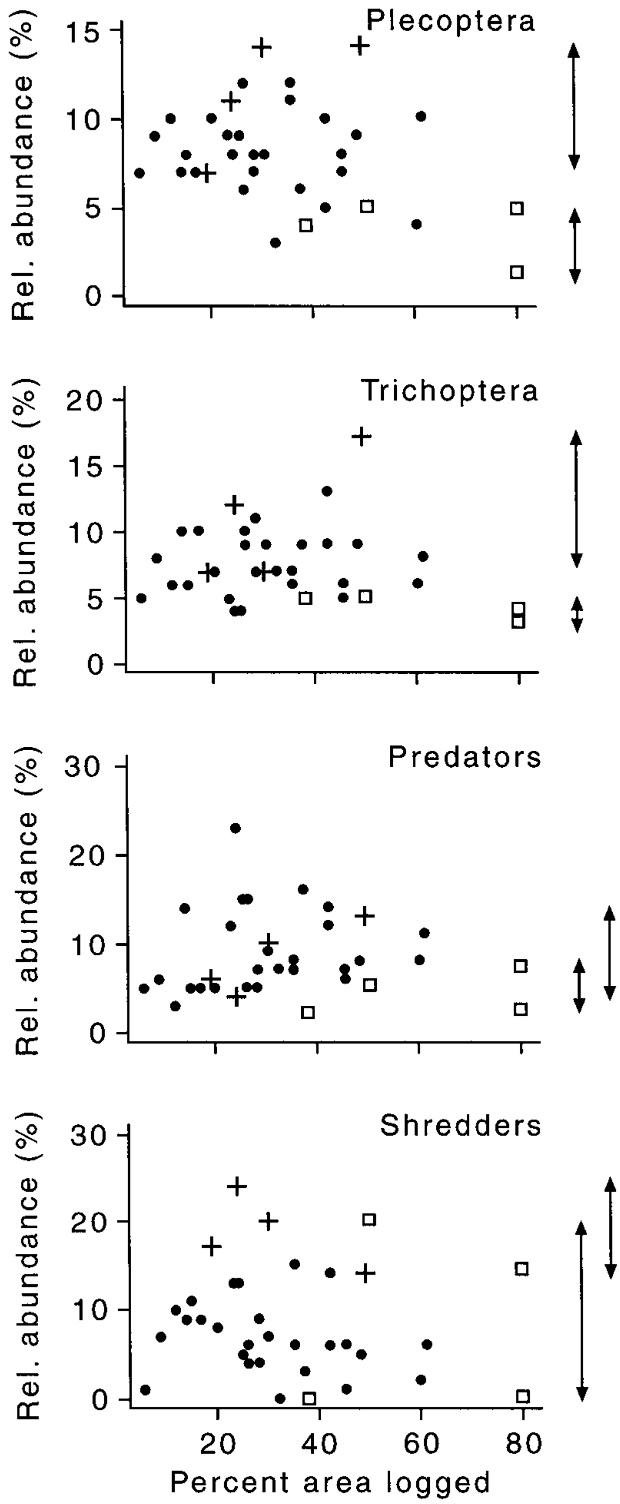

Figure:9.1 Relationship between invertebrate metrics and percent area logged. Metrics that clearly distinguish least degraded (+) from most degraded stream sites (□) are used in B-IBI. Moderately degraded sites are also shown (•). Two taxa richness metrics (upper panels) distinguished the two groups of sites. Trophic structure metrics, such as percent of predators and shredders (lower panels), failed to distinguish the groups of sites. Arrows to the right of each plot indicate the range of metric scores for good and poor sites. (from Karr and Chu, 1999)

Multimetric approaches frequently use ratio measures that are based on the comparative abundance of 2 different groups. Fore *et al.* (1996) recently criticised the use of such ratio measures because both the numerator and denominator vary together. For example, high numbers of scrapers and filterers will have the same ratio value as low numbers, although large values of these groups may indicate entirely different conditions at a site. In addition, the variance of 2 variables combined tends to be higher than the variance of either variable alone (Sokal & Rohlf, 1981). Fore *et al.* (1996) therefore recommend that if 2 attributes of an ecological community are potentially important they should be expressed individually as percentages or relative abundance. Some critics (e.g. Suter, 1993; Norris & Georges 1993; Reynoldson *et al.* 1997) suggest that information is lost by combining metrics into a single index. Karr and Chu (1999) dispute this and ascertain that no information is lost, rather that multiple measures increase the probability of an accurate diagnosis over a single measure.

The multimetric approach of combining ecological measures attributes an equal weighting to each score contributing to the final overall, index value. Redundancy in such a system occurs when component measures are effectively measuring the same ecological variables. The inclusion of redundant measurements where all scores are combined and overall quality is based on their collective value, means redundant measures make a disproportionate contribution and skew the final score, amplifying it in their direction. Thus redundancy carries a serious danger of mis-classification. Barbour *et al.* (1992) recommend the use of pair-wise correlation of candidate metrics to ascertain redundancy through correlation analysis and selecting an threshold 'r' value (e.g. $r > 0.7$) which signifies redundancy.

A primary concern suggested of multivariate analyses in biological classification is that decisions are based on the statistical properties of the data rather than the biological information provided by metrics in how animals feed, reproduce, use the habitat or respond to human activities (Fore *et al.* 1996). Gerritsen (1995) suggests that multimetric indices have been specifically designed for bioassessment and are thus more easily understood by management whereas multivariate methods are more involved and complex to implement and may be more difficult to convey to managers. In addition using multivariate techniques inevitably involves a large database of reference sites to build a predictive model (e.g. initially RIVPACS was about 300) before any test sites can be evaluated. The multimetric technique requires less initial investment before site analysis can occur and this may have been one of the reasons why this approach was initially proposed in the USA as it would have been easier to sell to funding managers than a predictive approach, even though the predictive approach had already been published (Wright *et al.* 1984).

Another concern of the predictive approach is streams with low temporal community persistence at the reference sites (Bunn, 1995). In most cases the model is established from a database sampled in one year (although seasons may be amalgamated) and thus if the taxa composition of the streams changes markedly from year to year, species prediction of the community given no impairment may not be accurate and a Type I error may occur when compared with a test site unless all sites change temporally in the same manner. Persistence of communities over time may depend upon the availability of refugia to avoid disturbance which could vary significantly from stream to stream.

The need for these evaluations to interpret ecological data with reference to more than one measurement highlights the potential diagnostic benefits of a multimetric approach to data analysis and classification. Whilst diagnostics assessment will clearly be enhanced where an expected taxa list is available - either by historical records or predictive techniques - by enabling diagnostic analysis of organisms both present and absent, the scope for analytical community measurements is reduced where predictive ability is limited to presence/absence data.

Multivariate predictive models have been shown to be superior than multimetric indices in detecting differences between test and reference sites potentially impaired by timber harvest (Hawkins, C. *personal communication*). Reynoldson *et al.* (1997) found that two predictive approaches had higher precision and accuracy than multimetric indices in discriminating reference sites. Nevertheless both multimetric and multivariate analyses are considered superior to single indices (Norris & Georges, 1993; Fore *et al.* 1994) but Gerristen (1995) considers that neither has been shown to be superior to either. The two approaches are not mutually exclusive and it is possible that the inclusion of metrics into established multivariate may yield more powerful predictive tools for biological classification.

9.2.2 Limitations of biomonitoring approaches

Macroinvertebrates are specious and ecologically diverse thereby providing a broad array of responses to stresses, nonetheless, this group represents a finite 'window' onto the ecosystem. Many macroinvertebrates possess behavioural and life cycle characteristics that enable them to endure and avoid impacts at a particular spatial and temporal scale. The physical, chemical and temporal refuges (e.g. hyporheous, terrestrial phases, dormancy) employed, both actively and passively by invertebrate communities, are spatially and temporally different to those used by other organisms such as fish (e.g. dead zones, flood plain waters, adjacent tributaries). Thus the spatio-temporal scale of information provided by invertebrate analysis is not necessarily representative of conditions endured by organisms of the entire riverine community.

In addition, a limited ecological focus may bias assessment results by failing to acknowledge that biological groups differ in their susceptibility and response to anthropogenic impacts. For example, pesticide pollutants affect the invertebrate community most strongly, whereas diatoms appear the most sensitive organisms to eutrophication and heavy metals. Similarly, features of the physical habitat differ in their importance to respective ecological groups; assessment of the diatom community may indicate a healthy system where river waters flow through concrete conduits, whereas assessment using fish or invertebrate classification would likely indicate the radical compromise to ecological integrity.

Many bioassessment techniques are based on interpreting species present in relation to their pollution tolerance, paying little attention to the dynamics of lotic ecosystems. The type, scale and timing of an anthropogenic impact determine the magnitude of ecological impairment. Bioassessment is based on taxa present at a site which represent not only the ability of an organism to withstand disturbance (resistance) but also its ability to subsequently recover (resilience). Resistance and resilience are determined by characteristics of both the ecological community and the river habitat. Key ecological characteristics governing resilience of an organism are mobility, fecundity, voltinism and vagility, which influence species traits (Townsend *et al.* 1997). Such characteristics are the theoretical r- and K- strategies attributed to ecological organisms as discussed in Section 2.4. These species traits have been successfully applied to river communities demonstrating the relevance of resilience (Section

5.5) to bioassessment and the value of incorporating theoretical concepts into river classification approaches.

In addition river characteristics (disturbance regime, deadwater zones, marginal habitat etc.) and the relative location of the assessed site (position in the river network, distance from tributaries, presence of lakes, dams, falls etc.) will contribute to community resistance and recovery processes by affecting colonizing sources and routes for organisms initiating recovery. The importance of such features highlight the value of a comprehensive habitat assessment.

9.2.3 Diagnostic abilities of bioassessment approaches

For specific assessment (e.g. acidification, organic pollutants) the differential sensitivities of ecological groups can be used to indicate what factors are responsible for the degradation. Diagnostics refers to the ability of an assessment method to identify the specific cause of ecological impairment. Diagnostic assessment is based on a comprehensive knowledge of the ecology and autoecological requirements of individual organisms and employing this information in data interpretation. Although experienced freshwater ecologists may use professional judgement to suggest the possible cause of ecological impairment, this information needs to be formalized into measurements that are widely comprehensible by non-specialists and managers of water resources.

Specific biotic indices (e.g. DAlpo index for organic pollution; Wanantabe, *et al.* 1988) and simple indictor species analysis (e.g Weatherley & Ormerod, 1990 for acidification) are examples where pollutant tolerance data has been used to develop a management tool for diagnostic assessment. Whereas species indicator analysis identifies a basic pass/fail classification, biotic indices indicate the degree of impairment presenting the possibility for more detailed classification. In addition species indicator analysis, based on a few key organisms, is more likely to be geographically restricted in its application to zoogeographic regions where the selected indicator species occur, thus biotic indices are superior management tools.

An example of how directed research and data analysis can provide diagnostic features is provided by the development of the Trophic Diatom Index used to identify eutrophication

(TDI; Kelly, 1998). The initial index, developed by Schiefele & Kohman (1993), had proved unsuccessful in practice as it was unable to distinguish between eutrophic enrichment and organic pollution, which is often associated with eutrophication, because several of the 'indicator' taxa it used thrive under both conditions (Kelly & Whitton, 1995). However, following research, Kelly's (1998) TDI is able to distinguish between these two types of impairment by interpreting the TDI index alongside information about taxa's organic tolerance and thus identify the type of enrichment occurring. Similarly, by examining several contrasting community measures together, Ohio EPA have been able to identify some distinct 'biological response signatures' to specific impacts (see Yoder & Rankin, 1995). The use of metrics that employ functional feeding groups can offer diagnostic potential as increases in shredders within the community can be indicative of acidified conditions (Sutcliffe & Hildrew, 1989). Increases in numbers of collector filters may be indicative of enhanced levels of fine particulate organic matter from organic pollutants.

9.3 Classification designations and spatial scale

From this review the number of river station classes designated typically falls between five and nine, although in the USA three or four classes are applied. Some authors (e.g. Chovanec *et al.* in press) suggest that below five is not sufficient to discriminate between lower degrees of degradation or improvement. Where seven classes are identified, particularly in Europe, the number of classes has evolved from the original Saprobic System.

However it must be remembered that the number of classes used is arbitrary and frequently selected as a result of professional opinion or for management reasons. There is frequently no statistical treatment attempted to validate what is "truth" regarding the number of classes of impairment identified. If a particular index or measure is, for example, less than 75% of reference it is implied but not certain that the site is impaired; and the degree of impairment compared to a site less than 50% of reference is typically unknown even though designations such as "slightly impaired" or "moderately impaired" are frequently attached to these derivations from reference. In a study of the effects of urbanization on streams in Anchorage, Alaska, Milner and Oswood (in press) used both "clean" and "dirty sites" to evaluate the number of classes apparent using TWINSPAN classification and found three clear groups but no further classes were clearly defined. Jaccard's Coefficient analyses showed these three groupings had the closest similarity in taxa and hence functional attributes than other groupings. Hence in this example a system that would have identified more classes would not have been justifiable.

It seems rather inconsistent to use sophisticated multivariate techniques to determine differences in biological communities and detect change but then arbitrarily decide classes without further analyses. Karr and Chu (1999) recommends the use of dose response curves by plotting metric or index values against different types of anthropogenic disturbance (e.g area logged or area paved) (see Figure 9.1). This is more difficult with predictive approaches but O/E ratios could be plotted.

In the same way that assessment methods using different biological groups require habitat parameters of different scales, analysis of their respective communities will provide information of river quality at different scales. Despite broad overlap between taxonomic groups, ecological generalisations, with reference to space and time, are apparent and are related to intrinsic ecological attributes. Diatoms, vascular plants, invertebrates, fish and other aquatic/semi-aquatic organisms respond differently to degradation (both within and between groups), with different spatial and/or temporal scales of refugia, re-colonization, reproduction groups), with different spatial and/or temporal scales of refugia, re-colonization, reproduction rate's and other ecological attributes. Although difficult to define, an statement of scale (estimates of upper and lower range limits) would provide valuable quantitative information on the limitations of bioassessment results and offer a more explicit evaluation with meaningful units (e.g. classifying biological quality for approx. x km of river length). Despite the difficulties in developing these criteria, such efforts would offer a more definitive statement of river quality and would focus attention on the suitability of the evaluation technique(s) being employed.

Where degradation is detected it is important to determine the extent of biological impairment, i.e. what length of river does that site represent?. To make informed decisions management must be able to quantify both the severity and longitudinal magnitude of the impact. To address this issue Ohio EPA developed the Area of Degradation Value (ADV; Yoder & Rankin, 1995). Quantifying damage in this manner enables a comparative evaluation of impacts so that management action can be prioritised. Ohio's ADV system is based on graphical plots of bioassessment results, with the x-axis representing the longitudinal nature of the river, and the index value of acceptable quality as a constant providing the benchmark for degradation measurement.

The magnitude of degradation is defined as the area of the graph beneath the constant value (see Fig 9.2). The ecological group(s) used for assessment and the characteristics of the river (presence of tributaries etc.) determine the frequency of sampling sites required to map ecological damage and recovery. The ADV system also has potential value in determining appropriate compensation for pollution by assessing penalties on the basis of actual damage (Yoder & Rankin, 1995).

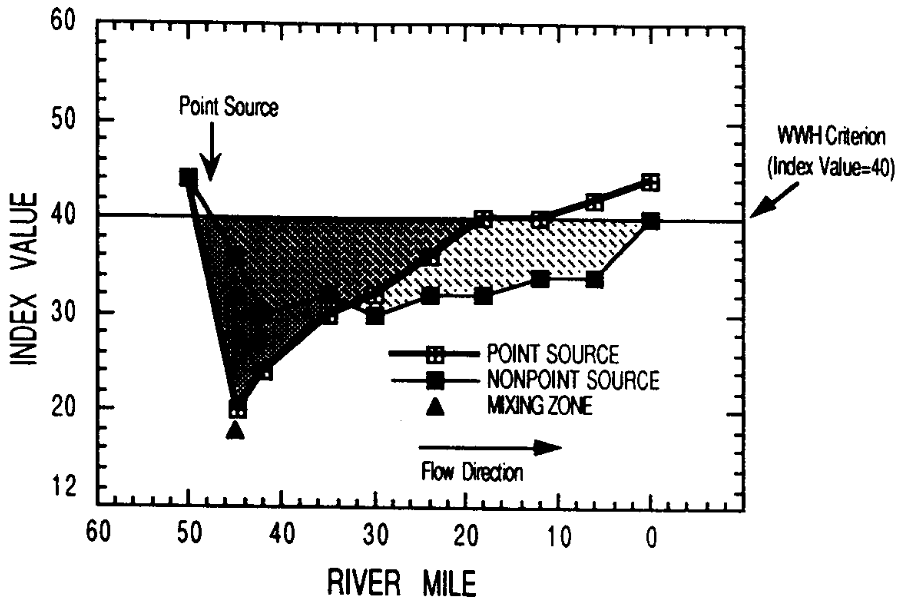

Figure 9.2. Illustration of the Area of Degradation Value (ADV) concept used to estimate and quantify the severity of ecological impairment.

Zoogeographical differences in the distribution of biological communities (zoogeographic distributions) creates a problem when biological indices are indiscriminately applied (a problem even more emphatic with indictor based assessment). Regional differences in both the distribution of organisms and susceptibility to human impacts emphasize the benefit of an ecoregion approach to river management. Identifying regional differences before evaluation enables assessment of an ecological community to be expressed in terms of their own potential, thus facilitating direct comparison of results from contrasting environments. This allows a meaningful comparison of relative quality across large, contrasting geographical areas and has particular significance in the light of imminent European legislation (The Water Framework directive). The EU is likely to call for a harmonised five class system for wider quality designation (The Water Framework Directive)

However many of these indices could be adapted to fit within the five classes proposed; for example the Danish Faunal Index (range 7 categories) would operate as follows depending upon river size;

	Small	**Medium**	**Large**
Bad	**I**	**I**	**I**
Poor	**I - II /II**	**I - II**	**I - II**
Fair	**II - III**	**II/ II - III**	**II**
Good	**III/III-IV**	**III**	**II/III /III**

9.4 Application of reference conditions

The reference condition, against which many assessment measures are compared, defines the parameters for biological quality classifications and is thus a very important consideration . Although the type and use of reference conditions is highly variable (Section 2.5) this discussion is based on their use for test site comparisons and is restricted to a comparison of multimetric and predictive approaches to establishing reference conditions. Both multimetric and predictive techniques use regional reference sites (Reynoldson *et al.* 1997) with similar 'least impacted' sites classified into collective groups to create generalized 'reference conditions'. The biological attributes of a test site are compared against the biological attributes derived from the suite of reference sites that together define the reference condition (Section 2.5). Both approaches also use measurements of key physicochemical parameters to refer a test site to its corresponding reference conditions. However, the two systems differ in the manner used to classify the reference site into their respective reference conditions.

In a predictive approach ecological data from reference sites are used, employing multivariate statistical techniques, to objectively classify streams and then position them in statistical space (see Chapter 2.5; Wright *et al.* 1984) Thus streams classified together have similar ecological communities. Further statistical techniques are used to determine the physico-chemical parameters that are most strongly associated with organism distributions and, therefore, also associated with the classification of reference sites. These key parameters are subsequently used to place a test site in the statistical (ordination) space occupied by the reference data matrix and thus assign it to its appropriate reference conditions. Predictive techniques may assign reference conditions from a single classification group (form the reference database) to which it is most similar i.e. closest in statistical space (e.g. BEAST; Reynoldson *et al.* 1995) or alternatively the reference conditions may be based on proportional contributions (based on relative distances) from all neighbouring classification groups (e.g. RIVPACS; Wright, 1995).

The multimetric approach determines biological reference conditions by *a priori* classification, using a hierarchy of environmental criteria. Thus reference sites are typically classified on the basis of physico-chemical parameters, as opposed to the biological community. Using the multimetric approach a test site and its corresponding reference conditions are established requiring no statistical analysis. In addition the multimetric approach recommends sampling across all habitat types but there is no mechanism for calibrating index results where differences are due to varying habitat types that occur naturally thereby influencing biotic community structure, and not due to anthropogenic impairment.

Where sufficient resources exist the establishment of a predictive approach, using the biological community to objectively identify reference conditions, clearly enables a more appropriate and robust basis for evaluating the biological community. Furthermore, predictive methods enable biological objectives or criteria to be set from habitat characteristics, bringing an ecological focus to river management and facilitating the building block approach to river management discussed by Harper *et al.* (1995). Within the predictive approach, RIVPACS offers further advantages over the BEAST system as the reference conditions for test site comparison are never a compromise by having to rely on a group of reference sites that are a close approximation as is the case with BEAST. RIVPACS uses a continuous data base, interpolating data between discrete classification groups, thus offering the unique possibility of site specific reference conditions.

The difficulties of selecting suitable reference sites present an acknowledged limitation to any assessment methods (Wright *et al.* 1997). However, the criteria used to determine least impacted conditions may fail to address other insidious river degradation problems (e.g. catchment land use, acidification). Evaluation biases may also be introduced by the parameters used to assign a test site to its reference conditions, for example, RIVPACS uses substrate as a predictor variable and is therefore less likely to detect ecological impairment resulting from substrate degradation. The dynamic nature of river ecosystems mean that evaluations cannot be based on a static model, requiring reference sites to be periodically resurveyed; however, this creates an assessment paradox confounding the detection of widescale degradations such as acidification and global warming.

9.5 Importance of linking biomonitoring to habitat assessment

Habitat quality is a major factor influencing the riverine community, its degradation can impair ecosystem health and its assessment is becoming an increasingly important part of bioassessment and classification. Despite the existence of standard methodologies for particular assessment techniques there is generally little consistency in habitat information gathered between groups because the scale of habitat features relevant to each ecological group are different. Whilst such an approach suits the needs of individual group evaluation and optimizes efficiency of data collection, it presents problems for cross examination of results between groups and limits more holistic assessment (see above, Section 9.2). An overall habitat assessment that enabled a focus on sub-units of an appropriate scale to be measured with the resolution required for a particular biological group, whilst remaining nested within a broad scale description of habitat (of a scale relevant other ecological groups), would provide a valuable framework for integrated analysis. Such a framework would bring an ecological focus to the hierarchical scaling apparent in the physical environment (Frissel *et al.* 1986; see Section 2.4), would facilitate the incorporation of community resilience, and provide insight to the 'functional habitats' for the whole community (see Harper *et al.* 1995 for invertebrates).

Much of the infrastructure for a more holistic approach towards habitat assessment already exists. Based at the catchment and sub-catchment scale, SERCON (System for Evaluating Rivers for Conservation) evaluates river quality using ecological and river habitat attributes measured in relation to their conservation value (Boon *et al.* 1997). SERCON delineates reaches into Evaluated Catchment Sections (ECS) typically between 10-30 km in length defining boundaries by a dramatic change in river morphology i.e. slope, major tributary etc. (see Frissel *et al.* 1986). It has been developed in parallel with the River Habitat Survey (RHS) which records many of the physical features relevant to SERCON which, in turn, relies on RHS data for much of its input (Boon *et al.* 1997).

The RHS records habitat information over a 500m reach providing a consistent, objective method for habitat evaluation (Section 2.6). Although the RHS has been devised to address a wide variety of river management issues (Raven *et al.* 1997) it has been successfully used. opportunistically, in assessment of ecological communities (e.g. Buckton & Ormerod, 1997;

Brewin *et al.* in press). However, in its current form the scale of habitat parameters recorded in the RHS are not directly comparable to those used in established bioassessment techniques for fish, invertebrates and macrophytes (e.g. RIVPACS, HABSCORE, PHABSIM/IFIM) although much of the general information is the same (Raven *et al.* 1997). Nonetheless the objectives, techniques and utility of these surveys are closely related and research, currently underway (Raven *et al.* 1997), to more effectively integrate their assessment approaches will have valuable benefits for ecosystem assessment.

For bioassessment purposes the utility of the RHS could be increased by focusing on the scale and detail of selected recorded parameters whilst remaining essentially an objective survey. However, additional information, relevant to ecological quality, would probably require value judgements. Paradoxically, the objective nature of the RHS is one of its key strengths in river broad scale description of habitat (of a scale relevant other ecological groups), would provide a valuable framework for integrated analysis.

The RHSs' ability to classify a river into a discrete 'type' may have additional benefits for comparative assessment. Characterisation of river type is currently an area of interest for theorists who have used the river as a template to describe ecological communities (Resh *et al.* 1988; Bayley & Li, 1992; *Freshwater Biology Special Edition,* 1994). Given the importance of river character in determining levels of ecological disturbance (Poff & Ward, 1989) and community characteristics (e.g. Poff & Allen, 1995) the identification of river types could be a useful discriminatory tool, increasing assessment sensitivity. Moreover, current efforts directed towards creating a predictive reference system (c.e RIVPACS) so that rivers can be compared to a 'pristine' standard (Raven *et al.* 1997) offers a particularly valuable tool for bioassessment. Measuring habitat quality using this system would posses all the advantages of predictive techniques (discussed above), providing an objective and robust habitat quality value that could be used in biological classification.

10 RECOMMENDATIONS

Although this review indicates that current UK Environment Agency practices of using the predictive RIVPACS approach possesses some distinct advantages over alternative methods for the biological classification of rivers, further refinements of existing techniques could increase the utility of assessment approaches. These suggestions follow;

1. River classification based on more than a single biological group increases holistic evaluation and the diagnostic potential of assessment thereby permitting a more accurate estimate of ecosystem quality. Thus the development of predictive approaches using diatoms and possibly fish would be of significant value in this respect.

2. The inclusion of the BMWP and ASPT scores in RIVPACS demonstrates the scope for incorporating diagnostic analyses into current practices within one biological group. Other metrics (e.g. functional measures) could be included for prediction that would broaden the application of RIVPACS to different types of impairment.

3. Habitat information is rarely used in detail to interpret biological data. Developing habitat assessment into a more generally applicable, integrated habitat survey would facilitate community comparisons across biological groups and increase potential for ecosystem evaluation. In addition, more detailed habitat assessment would help identify differential ecological resilience resulting from differences in stream type (e.g. refugia and sources for recovery), and thus increase assessment precision.

4. The biogeographical distribution of communities within rivers require differential regional management. Identification of ecoregion, and sub-ecoregion differences will enable a focus on management problems that may be locally acute and allow ecological quality to be evaluated in terms of a rivers own potential. It is possible that different

classification approaches should be used depending upon the scale of investigation and the type of impairment. For example macrophytes and fish may be more appropriate for impairment at the catchment scale whereas algae are better indicators at the reach or patch scale. Although clearly preferable, the "one technique fits all" may not be appropriate at these different spatial scales.

5. Defining appropriate reference conditions is critical to accurate assessment and subsequent biological classification; a standard approach to reference criteria may not be suitable for all assessment tasks. The strength of RIVPACS is that it predicts site specific reference conditions for comparison. However the use of a dynamic model (i.e. resurveying and updating reference conditions to account for year to year variation) is important for the accurate assessment of ecological systems and to avoid Type I errors. 'Pristine' reference rivers may not be the most appropriate for the management of highly modified rivers (e.g. urban rivers) by failing to provide sufficient resolution on their residual ecological quality and thus failing to most effectively manage and enhance their quality.

6. Classification approaches must recognise that current assessment methods are limited by our incomplete understanding of river ecosystems. Assessment techniques need to be flexible so that they can be upgraded as knowledge of river systems and ecological communities (and our ability to measure their quality) increases. Assessment techniques would benefit from the further application of available ecological knowledge. Functional processes are widely recognised as an important component of ecological quality and can be analysed through functional attributes of biological organisms. Also, consideration of the ecological strategies of organisms would assist diagnostic interpretation of results and help the selection of the most appropriate assessment approach for a particular management issue.

The Water Framework directive requires member states to implement a program of river classification to monitor the ecological quality of river ecosystems. An auxiliary document (The Harmonised Monitoring and Classification of Surface Water in the European Union) acknowledges that individual member states have vested interests in their respective approaches to bioassessment and describes how a common system of classification (using 5 quality classes) can be applied to individual assessment methods. The intention of a common classification scheme is to enable comparisons of river quality across EU countries. This means that different assessment methods (e.g. predictive techniques, biotic scores, multimetrics) using different biological groups (e.g. diatoms, macroinvertebrates, fish) will be used to classify river quality in different EU countries.

As discussed in Section 9.3 the classification of bioassessment results represent an arbitrary division of measured ecological quality (in this case 5 divisions). Although member states will be using the same number of divisions to present their results, the accuracy of their classifications will depend on the method of bioassessment used to measure ecological quality. Thus, although quality class categories will equate numerically for all member states, comparing of river quality between EU countries (that use different assessment methods) will be like comparing apples with oranges.

Current UK Environment Agency practices include some of the best available techniques for evaluating and classifying river quality. The further development of these approaches will improve assessment calibre and increase management's potential to achieve sustainable river use. However, EU member states using more accurate and sensitive bioassessment (and, therefore, river classification) methods may, in consequence, impose stricter legislation on themselves compared to member states using cruder assessment approaches.

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13. APPENDICES

13.1 Appendix A

Bioassessment Questionnaire

The Environment Agency/University of Birmingham
Environment Agency National Research and Development Project 712:
Alternative Methods in Biological Classification

This questionnaire is part of a review of contemporary methods of bioassessment. Its aim is to identify methods that are currently being used and to evaluate the techniques employed by identifying problems and benefits associated with their use; inclusion of any further information that may help this assessment is welcome. Your help in completing this questionnaire is greatly appreciated. Please return completed forms by April 15, 1997 to:

Kieran Monaghan
School of Geography
University of Birmingham
Edgbaston
Birmingham B15 2TT
United Kingdom

Please fill in your name, address and email/fax numbers;

Name ....

Address ....

....

....

....

Email ....

Fax ....

General

1.(a) Which group of organisms have you used to assess water quality in the last five years?

(tick each box as appropriate)

	code	
microbial	(MIC)	[ ]
diatoms	(DIA)	[ ]
plankton	(PLK)	[ ]
bryophytes	(BRY)	[ ]
macrophytes	(MAC)	[ ]
invertebrates	(INV)	[ ]
fish	(FIS)	[ ]
birds	(BIR)	[ ]
other *(please specify)*	(OTH)	[ ] _________

1.(b) Please indicate the purpose for which you use these organisms in bioassessment

(tick each box as appropriate)

purpose	**biological group (code)**								
(see below)	MIC	DIA	PLK	BRY	MAC	INV	FIS	BIR	OTH
survey									
monitoring									

survey: define/identify patterns and variations in space

monitoring: ensure compliance, assess impact of pollution incident, identify trends, strategy verification

1.(c) What aspects of water quality are you assessing? *(tick each box as appropriate)*

potential pollutant	**biological group (code)**								
	MIC	DIA	PLK	BRY	MAC	INV	FIS	BIR	OTH
nonspecific									
inert solids									
heavy metal									
land disturbance									
acidification									
thermal									
eutrophication									
drought/abstraction									
pesticides									
oil									
salinity									
organic									
other _ _ _ _ _									

With reference to ONE of the above biotic groups and one bioassessment purposes (e.g. macroinvertebrates & monitoring) complete the rest of the questionnaire.

Completion of separate questionnaires for each biological group used for bioassessment would be appreciated.

BIOLOGICAL GROUP: _ _ _ _ _ _ _ _ _ _ _ _   ASSESSMENT PURPOSE: _ _ _ _ _ _ _ _ _

Sampling

2. How often is sampling undertaken? *(if variable please state why)*

3. Are (routine) biological samples collected with additional information?

NO [ ]...........*go to question 4*

YES [ ]...........*variables recorded: (tick each box as appropriate)*

channel dimensions	[ ]
current velocity	[ ]
substrate type	[ ]
bank characteristics	[ ]
slope	[ ]
land use	[ ]
riparian vegetation cover/type	[ ]
temperature	[ ]
conductivity / pH	[ ]
specific chemical analysis *(please specify)*	[ ] _ _ _ _ _ _ _ _ _ _ _ _ _ _ _ _ _
other *(please specify)*	[ ] _ _ _ _ _ _ _ _ _ _ _ _ _ _ _ _ _

4. What is the geographical scale of your investigation? *(tick each box as appropriate)*

local (single site)	[ ]
regional *(please name region)*	[ ] _ _ _ _ _ _ _ _ _ _ _
national *(please name country)*	[ ] _ _ _ _ _ _ _ _ _ _ _
international *(please name countries)*	[ ] _ _ _ _ _ _ _ _ _ _ _

5. How are samples collected? *(tick each box as appropriate)*

from natural habitat [ ].............*go to question 6*

using artificial substrate / traps [ ].............**How long is the artificial substrate / traps left before collection?**

_ _ _ _ _ days

6. Please describe sampling device(s) (e.g. Surber sampler, artificial substrate)

N.B. If a net is used please specify mesh size and depth of the bag.

7. What river features (other than accessibility) limit effective sampling using this method? *(for each box ticked please state the range over which sampling is effective)*

none [ ]

substratum type [ ] _ _ _ _ _ _ _ _ _ _ _ _

water depth [ ] _ _ _ _ _ _ _ _ _ _ _ _

flow regime [ ] _ _ _ _ _ _ _ _ _ _ _ _

channel vegetation [ ] _ _ _ _ _ _ _ _ _ _ _ _

turbid water [ ] _ _ _ _ _ _ _ _ _ _ _ _

time of year (season) [ ] _ _ _ _ _ _ _ _ _ _ _ _

other*(please specify)* [ ] _

8. Are samples collected from one habitat type (e.g. riffle) or stratified to include others?
(tick one box only)

one habitat [ ]..........*go to question 9*

more than one habitat [ ] *please state which habitats and approx. contribution from each*

_ _

_ _

9.(a) Is the sampling: *(tick each box as appropriate)*

qualitative [ ]................*go to question 10*

quantitative [ ]...............*answer (b)*

semi-quantitative [ ]...............*answer (b)*

9.(b)Quantification by: *(tick each box as appropriate)*

area/volume [ ] *please specify* _ _ _ _ _ _ _ _ _ _ _ _ _ _ _ _

time [ ] *please specify* _ _ _ _ _ _ _ _ _ _ _ _ _ _ _ _

10. Are replicate samples collected at each site? *(tick one box only)*

NO [ ]

YES [ ] *please specify how many*_ _ _ _ _

Are replicates pooled together?

YES [ ]

NO [ ]

11. What sort of formal training is given to new personnel before field work is undertaken?

(tick each box as appropriate)

none [ ]

oral instruction *(please specify)* [ ] _

written instruction *(please specify)* [ ] _

practical instruction *(please specify)* [ ] _

12. Average time (approx.) to sample an individual site?

_ _ _ _ minutes

Post Sampling Procedures

13. Are samples sorted: *(tick each box as appropriate)*

dead/preserved [ ].....*go to question 14*

live [ ].....**Are organisms returned after processing?**

YES [ ]

NO [ ]

14. Are complete samples used for analysis or are subsamples taken? *(tick one box only)*

complete sample [ ]

subsample [ ] *please state the method of subsampling employed (i.e. what fraction of original is used and/or total organism count)*_ _ _ _

_ _

_ _

_ _

15.(a) Is preparation (e.g. separating organisms from mineral/organic matter, slide mounting) necessary before identification? *(tick one box only)*

YES [ ] *please specify* _ _ _ _ _ _ _ _ _ _ _ _ _ _ _ _ _ _ _

NO [ ]

15.(b) Are any techniques used to assist preparation? (e.g. floatation, staining)

YES [ ] *please state* _

NO [ ]

15.(c) Average (approx.) time to prepare a sample from an individual site for analysis?

_ _ _ _ minutes

16. What level of identification is used for data analysis? *(tick one box only)*

order [ ]

family [ ]

genus [ ]

species [ ]

other (eg morphology) *please specify* [ ] ___________

variable [ ] *which groups are identified to which level?*

____

____

____

17. Is accurate identification sometimes a problem?

NO [ ]

YES [ ] *What affects accurate identification?* __________________

____ ___________________________________

____

18. Average time (excluding preparation) to analyse samples for an individual site?

____ minutes

Data analysis

19. Which of the following measurements are used in assessment:

(tick each box and specify measurement where appropriate)

taxon richness [ ] ________________________________

diversity index [ ] ________________________________

relative abundance ratios [ ] ________________________________

trophic status (feeding groups) [ ] ________________________________

biotic score/index [ ] ________________________________

key taxa (e.g. exotic, indicator) [ ] ________________________________

population denstiy [ ] ________________________________

biomass [ ] ________________________________

condition (e.g. disease, deformities) [ ] ________________________________

other [ ] ________________________________

none [ ]

20. Do you experience any problems in producing these measurements from your samples?

NO [ ]

YES [ ] *(please specify)* ________________________________

____

21.(a) For data comparison of the study site do you use: *(tick each box as appropriate)*

a subjectively ascribed value [ ]................*go to question 22*

a data base from the same site (i.e. historical info.) [ ]................*go to question 21(d)*

a single reference site [ ]................*answer (b), (c) & (d)*

a group of reference sites [ ]................*answer (b), (c) & (d)*

21.(b) Is/are the reference site(s): *(tick each box as appropriate)*

within the same river (upstream/downstream) [ ]

a different river within the same catchment [ ]

from a river in a different catchment [ ]

21. (c) What criteria are considered in selection/identification of the reference site(s)?

(tick each box as appropriate & indicate order of importance if relevant)

	used	*order of importance (1=most important)*
ecoregion	[ ]	[ ]
geographical location	[ ]	[ ]
channel dimensions	[ ]	[ ]
substrate type	[ ]	[ ]
temperature regime	[ ]	[ ]
flow regime	[ ]	[ ]
water depth	[ ]	[ ]
base line chemical characteristics	[ ]	[ ]
bank / riparian character	[ ]	[ ]
land use	[ ]	[ ]
other *(please specify)* ______________		[ ]

21.(d) How often are reference sites re-calibrated?

____ years

22. Statistical tests used to detect difference between sites:

ANOVA	[ ]
t-test	[ ]
Chi square	[ ]
Regression, correlation	[ ]
Similarity index *(please specify)*	[ ]
Kruskal-Wallace	[ ]
Mann-Whitney	[ ]
classification *(please specify)*	[ ] _
ordination *(please specify)*	[ ] _
other *(please specify)*	[ ] _
none	[ ]

Monitoring and Surveillance

This section is only applicable to those using bioassessment for purposes of monitoring.

23. Are sites identified as impacted when no impact is occurring? *(tick one box only)*

never (0%)	[ ]
rarely (1-5%)	[ ]
sometimes (5-15%)	[ ]
frequently (>15%)	[ ]

24. Are sites passed as unimpacted when impact is occurring? *(tick one box only)*

never (0%)	[ ]
rarely (1-5%)	[ ]
sometimes (5-15%)	[ ]
frequently (>15%)	[ ]

25. Please describe any further analysis undertaken after bioassessment when a site fails to meet the assessment criteria

26. What problems do you encounter with the application of your bioassessment data?

(tick each box as appropriate)

seasonal variability can mask impact [ ]

level of sampling precision can mask impact [ ]

comprehensible only to biologists [ ]

non-diagnostic [ ]

mixtures of pollutants limit diagnostic capacity [ ]

not legally recognized (i.e. cannot be used to prosecute polluter) [ ]

processing time too long [ ]

other *(please specify)* [ ] _______ _____

27. What forms of pollution can you diagnose using this method of bioassessment?

(tick each box as appropriate, Please state method of diagnosis for each box ticked)

none (only indicates impact) [ ] ____________________________

organic [ ] ____________________________

heavy metal [ ] ____________________________

land disturbance [ ] ____________________________

acidification [ ] ____________________________

thermal [ ] ____________________________

eutrophication [ ] ____________________________

drought stress/abstraction [ ] ____________________________

pesticide [ ] ____________________________

oil [ ] ____________________________

salinity [ ] ____________________________

other *(specify)*________ [ ] ____________________________

28. What criteria did you consider in selecting this method of bioassessment?

(tick each box as appropriate)

suitable for local conditions [ ]

suitable over a broad range of conditions [ ]

low cost of equipment [ ]

rapid form of assessment [ ]

appropriate biological time scale (i.e. life cycle) [ ]

little variation between different sampling operators [ ] low level of expertise required for analysis [ ]

specific response to a particular impact [ ]

diverse response to range of impacts [ ]

results recognized by non biologists [ ]

other(s) *(please specify)* [ ] ________________

____

29.(a) Have your methods of biological assessment changed over time? (e.g. level of identification, methods of sampling, processing or analysis, sampling frequency/season)

not changed [ ]

changed [ ] *How?* ____________________________

____

____

____

29.(b) Please indicate the reason for changes listed above *(tick each box as appropriate)*

more suitable sampling methods [ ]

cheaper equipment [ ]

faster method of assessment [ ]

greater sensitivity to impacts [ ]

more appropriate biological time scale (i.e. life cycle) [ ]

less expertise required for analysis [ ]

more suitable for geographic region (i.e. biogeographic coverage) [ ]

other *(please specify)* [ ] ____________

30. Which aspects of your bioassessment methods do you think need to be developed to increase its utility?

THANK YOU FOR ANSWERING THIS QUESTIONNAIRE

13.2 Appendix B

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14. GLOSSARY

Allochthonous: "Energy" input derived from outside a system, such as leaves of terrestrial plants falling into a stream.

Anadromous: moving from the sea to freshwater for reproduction.

Autochthonous: Energy input derived from within a system , such as organic matter within a stream resulting from photosynthesis by aquatic plants.

AUSRIVAS: **AUS**ralian **RIV**er **AS**sessment. AUSRIVAS is a modified RIVPACS approach. It uses samples from a single habitat, riffles. There are 2000 reference sites throughout Australia at present.

Bioassessment: Similar to biomonitoring although it may be more restricted to structural components of the aquatic community.

Bioassays: The use of test organisms to assess water quality either in the laboratory or in the field, by comparing the effect of a contaminant on living organisms, with the effect of a standard preparation on the same type of organisms. Organisms usually used in isolation i.e one type of organism for a specific type of pollutant.

Biocriteria: Numeric values or narrative expressions that describe the reference biological integrity of aquatic communities and are set as achievable goals for the biotic community.

Biomarkers: These are physical characteristics related to the condition of individuals within populations, that may be useful for assessing chemical contaminants. Physical characteristics in individual condition that may be useful in such assessment would result from microbial or viral infection, or some sort of teratogenic or carcinogenic effects being caused in individuals by contaminants. Such characteristics are usually categorized under: diseases, anomalies or metabolic processes.

Biomass: combined weight of all living organisms in a given area.

Biome: a major type of ecological community e.g. grassland

Biomonitoring: The systematic use of biological responses to evaluate changes in the environment with intent to use this information in a quality control programme.

Biological classification: The use of metrics, index scores or community structure to classify rivers into different categories of water quality.

Biotic index: There have been numerous biotic indices developed e.g Trent Biotic Index, BMWP score system etc. However, all biotic indices are essentially numeric expressions which combine diversity, on the basis of certain taxonomic groups, with the pollution indication of individual species/higher taxa /groups, into a single index or score.

Control: see reference condition

Detritivore: An organism that feeds on decaying organic matter (detritus).

Diatom: Any of various minute, one celled or colonial algae of the class Bacillariophyceae, having siliceous cell walls consisting of two overlapping symmetrical parts.

Diversity: In the context of this paper diversity can essentially be regarded as the relative degree of abundance of a species, function, community, habitat, or habitat feature per unit of area. Diversity indices are mathematical expressions which use 3 components of community structure, namely richness (Number of species present), eveness (Uniformity in the distribution of individuals among the species) and abundance (Total number of organisms present). The most widely used diversity index is the Shannon Information index.

Ecoregion: Area defined by expected similarity of biological communities. Relatively homogenous ecological system formed by the combined integration of geology, landform, vegetation, climate and anthropogenic factors.

EPT Taxa: The number of qualitative EPT taxa is a metric of the Invertebrate Community Index (ICI) which is generated by the qualitative sampling occurring in conjunction with the artificial substrate sampling that is also part of the ICI. The metric consists of the taxa richness of Ephemeroptera (mayflies), Plecoptera (stoneflies) and Trichoptera (caddisflies).

Functional feeding group: Groups of organisms that obtain energy in similar ways. e.g Autotrophic plants fix energy from sunlight. Shredders chew large particles like tree leaves.

Guild: Group of organisms that exhibit similar habitat requirements and that respond in a similar way to changes in their environment.

Hyporheic: Subsurface zone of channel substrate. Many characteristics of which are determined by hydrological conditions. Can provide an important refugia for invertebrates.

Indicator Taxa: A taxa that reacts to a specific or narrow range of pollutants, and can therefore be used as an indicator to the quality of the river with regard to the specific pollutant. Indicator taxa may provide diagnostic potential to bioassessment.

Instar: An insect or other arthropod that is between molts.

Measure: similar to metric

Metric: Provides information on biological attributes of the riverine community that changes in some predictable way with human influence - this response must be discriminated from natural variation.

Multimetric indices: A multimetric index is a single index that combines several community metrics into one value.

NORDPACS: A predictive model similar to RIVPACS that is being developed by a coalition of nordic countries including Denmark, Norway, Finland and Sweden. Reference sites for the model are at present in Sweden only.

Predictive: see RIVPACS.

Resistance: Ability of a community to initially resist a disturbance.

Resilience: Ability of a community to recover to a pre-disturbance state.

Reference condition: A site (or sites) that represents important aspects of "natural" river conditions, in fact it typically represents minimal disturbance as there are few pristine areas still left. Used for direct comparison with test site to determine degree of impairment; or to initially calibrate a scoring scheme for classification (e.g Hilsenoff's Family Biotic Index).

RIVPACS: **R**iver **I**n**V**ertebrate **P**rediction **A**nd **C**lassification **S**ystem. RIVPACS is a software package with a database based on 438 reference sites from 80 rivers throughout Great Britain. It can predict the expected invertebrate community at a given location in the absence of pollution, and provide a measure of the degree of impact (i.e deviation from normal). However, it cannot determine the type of impact.

Salmonid: A fish of the family Salmonidae, which includes trout and salmon.