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Biological Techniques of Still Water Quality Assessment.

Phase I Scoping Study.

Pond Action

R&D Technical Report E7

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Pond Action
c/o School of Biological and Molecular Sciences
Oxford Brookes University
Gypsy Lane Campus
Headington
Oxford OX3 0BP

Tel: 01865 483249 Fax: 01865 483282/483073

EA Project Leader
The EA's Project Leader for R&D Project i642 was:

Shelley Howard - EA Midlands Region

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Biological Techniques of Still Water Quality Assessment:
Phase 1 Scoping Study

Penny Williams, Jeremy Biggs, Lorna Dodds,
Mericia Whitfield, Antony Corfield, Gill Fox.

Research Contractor:
Pond Action

Environment Agency
Rivers House
Waterside Drive
Aztec West
Bristol
BS12 4UD.

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EXECUTIVE SUMMARY

This report describes the findings from Phase I of a three phase project to develop biological techniques for still water quality assessment. The main aim of this phase of the project has been to undertake a scoping study which:

(i) collates, and critically evaluates, information about the range of biological assessment methods used for monitoring still waters (lakes, ponds, canals, ditches, temporary and brackish waters),

(ii) considers broad approaches to biological assessment of still waters for use in General Quality Assessment and the establishment of Water Quality Objectives and recommends a method, or methods, for further evaluation and testing.

Evaluation of the Environment Agency's requirements for biological monitoring indicates the need for a two stage protocol:

Stage 1: General Ecosystem Assessment, which utilises a single, broad-based biological approach to evaluate the *net* effect of *all* impacts that degrade the integrity of freshwater systems.

Stage 2: Diagnostic Assessment, which employs one or more of an array of appropriate techniques (e.g. biological, chemical or historical data) to investigate the cause(s) of degradation.

The essential requirement for the development of a General Ecosystem Assessment method is that it should represent and summarise the overall *biological integrity* of a waterbody. Because biological integrity is a wide-ranging concept, an accurate measure of integrity is likely to be derived where *a number of biotic assemblages* and a *wide range of significant attributes* (e.g. measures such as species-richness, rarity, abundance) are used for assessment.

In order to assess their quality, sites should be compared with a baseline of minimally impacted reference sites. For biological data in general, the only viable option for the establishment of baseline conditions is comparison with 'least impacted' present day sites. In addition, all quality assessments need to be undertaken within the framework of a classification, which minimises the confounding effects of natural variation and allows degradation gradients to be identified more easily.

An important recent development in biological assessment methods has been the concept of *multimetric* evaluation in which ecosystem integrity is assessed on the basis of *multiple* attributes ('metrics') which are known to be related to ecosystem degradation. Using this system each attribute (e.g. factors such as taxa richness, percentage functional feeding groups, health etc.) is scored separately according to the extent to which it deviates from an undisturbed baseline condition. Metrics are then divided into simple 'rating' (e.g. 1-5) categories and summed to give a single index.

Matrix analysis was used to assess which biotic assemblages (aquatic macrophytes, zooplankton etc.) are likely to be practically feasible and cost-effective for still water monitoring within EA regions. The results indicate that no one assemblage is able to fully represent all major aspects of biotic integrity and to integrate all major effects of potential stresses. In practice, therefore, the reliability and validity of general ecosystem assessments will generally be enhanced by use of two biological assemblages. For lakes, which are both large waterbodies and prohibitively difficult to restore once degraded, monitoring on the basis of *at least* two biotic groups is likely to be necessary.

The best combination of two groups in most waters will generally be a combination of plant and animal taxa, since together these span a complimentary range of trophic levels, habitat niches and pollutant sensitivities. The assemblages which matrix analysis suggests are most suitable for still water quality assessment are:

(i) faunal assemblage - macroinvertebrates, (plus fish in permanent waters such as canals and lakes), and

(ii) floral assemblage - aquatic macrophytes or diatoms.

Of these assemblages, macroinvertebrate communities could be considered to be a relatively 'ideal' assessment group. Macrophytes are considered to be sub-optimal because their use is limited by poor temporal characteristics and the paucity of species found in naturally shallow, turbid and shaded waterbodies. Periphyton (particularly diatoms) and fish are both promising assemblages for water quality monitoring, but both require further investigation to assess their practical viability. Brackish waters and temporary waters are inherently species-poor habitats. This, combined with the paucity of information regarding their communities and impact sensitivity, makes it difficult to predict which (or how many) assemblages will have sufficient resolution to enable waterbody degradation to be adequately assessed.

Assemblages recommended above vary considerably in their potential for immediate development and testing as indicators of water quality. Macroinvertebrate-based methods could, for example, be developed quite quickly for pond or ditch assessment. In contrast, a diatom-based assessment method would require a prolonged set-up period during which the potential of the group, and appropriate sampling methodologies, were more fully evaluated.

Based on these findings we recommend a twin-track approach to further methodological development:

Track 1. Multimetric testing and development:

Test and begin development of a multimetric method based on macroinvertebrate assemblages in one or more of the following waterbodies: lakes, ponds, canals, ditches.

It is recommended that in order to develop the method proposed, a trial is set up based on a *regional data set.* Regional data could be collected specifically for the project or could be partly based on exiting data sets.

Track 2. Investigate the viability of other assemblages:

(i) investigate the comparative potential of diatom communities for application as a floral assemblage in lakes, ponds, canals, ditches.

(ii) use desk study information to investigate the potential for fish metrics to be developed for use in lakes and canals.

(iii) investigate the most appropriate combination of taxa to use in assessment of brackish and temporary waters.

1. INTRODUCTION

1.1 Project aims

This report describes the findings from Phase I of EA R&D Project i642 "*Biological techniques of still water quality assessment*". The project objective is to develop a biological assessment method (or methods) which will enable the EA to monitor the quality of still waters in England and Wales. The main aim of Phase I of this project has been to undertake a scoping study which:

(i) Collates information about the range of biological assessment methods used for monitoring still waters and critically evaluates these methods.

(ii) Considers broad approaches to biological assessment of still waters for use in General Quality Assessment and the establishment of Water Quality Objectives.

(iii) Recommends a method, or combination of methods, which fulfils the defined criteria sufficiently to warrant further evaluation and testing in Phases II and III of the project.

1.2 Background to the project

1.2.1 The importance of still waters

Still waters make up 65% of the total surface area of inland water in England and Wales (Barr *et al.* 1994), and encompass a wide variety of waterbody types, including natural and man-made lakes, reservoirs, permanent and temporary ponds, canals, ditches and brackish lagoons.

The most recent estimates show that at present there are about 157,000 ponds and lakes in England and Wales. More than 95% of these waters are between 0.0025 ha and 1 ha in area with only some 4300 larger lakes and reservoirs (Barr *et al.* 1994; Biggs *et al.* 1996). The number of very small (i.e. under 0.0025 ha) and temporary water bodies is not known, but is likely to be considerable (Pond Action, unpublished data). The total length of canals in England and Wales is estimated to be around 2500 km and Barnes (1989) has estimated that there are approximately 150 brackish lagoons skirting the coastline of Britain.

In part because of their abundance, still waters are an important natural resource, providing a public water supply, economically valuable fisheries, and facilitating a wide range of recreational activities. Considered as a whole, still waters also contribute significantly to UK biodiversity: over 90% of all British species of freshwater macrophytes, aquatic invertebrates, amphibians and freshwater fish occur in still waters, and these waters are second only to woodland in supporting species of conservation concern (BSG 1995).

1.2.2 The value of biological monitoring for assessing the quality of still waters

The tradition of biological and chemical monitoring

Modern concepts of biological water quality monitoring originated in Britain (Chadwick 1842), and have been developed world-wide for over 150 years. From the earliest days, and until relatively recently, biotic monitoring was mainly undertaken to indicate changes in the physical and chemical quality of water. In this respect biological methods have a number of well-known benefits which enable them to complement standard physical and chemical monitoring:

- Unlike chemical samples, which reflect water quality only at the moment of sampling, biological measurements reflect both current conditions and the cumulative effects of episodic impacts, which can go undetected in chemical monitoring programmes.
- Biological monitoring integrates the effects of chemicals in different environmental compartments. Chemical monitoring of the wrong compartment, or of just one compartment, may produce misleading information, leading to impacts being

overlooked. Indeed, it is almost by definition biological measures which alert those monitoring still waters to new impacts caused by environmental contamination.

- Chemical monitoring of the many thousands of microcontaminants which may pollute water is effectively impossible, and unlikely ever to be cost-effective. Biological techniques, which can integrate the effects of these many different compounds, provide a cost-effective method for detecting the net effect of these impacts. Where significant damage is recognised, either chemical or biological techniques may then be applied in diagnosis of problems.

A new role for biological monitoring

In recent years, the traditional view of biotic assemblages as merely 'good chemical monitors' has begun to change. This has in large part been due to an increasingly widespread recognition that the best way to sustain exploitation of water resources (whether potable water, a viable fishery or environmental recreation) is to ensure that the integrity of natural ecosystems is maintained.

This shift in approach is now evident in legislation, from the new EA imperative to consider environmental sustainability to the most recent EU Directives (for example, the Urban Waste Water Treatment Directive and the proposed Directive on Ecological Quality of Water) which focus on levels of ecosystem damage rather than on levels of chemical determinands. This change to a large extent reverses the traditional roles of chemical and biological monitoring. It is now the biological quality of the environment which is becoming of overriding concern.

As a means of monitoring ecosystem quality, assessments based on only physico-chemical parameters have considerable disadvantages. There are three main reasons for this:

- Our limited knowledge of the effect of environmental stressors on biota means that the effects of most chemical pollutants on ecosystem integrity cannot be predicted with any precision.
- Chemically-based water sampling does not take account of many anthropogenic perturbations, such as modifications of hydrological regimes and habitat degradation, which impair biological integrity.
- The biota itself can modify the physico-chemical environment and will initially buffer many changes, so that damaging impacts may be evident in the biota long before physical or chemical parameters show significant changes.

The limitations of a reductionist approach to environmental monitoring based on the assessment of a small range of physical and chemical parameters have also been documented in practice. For example, Ohio EPA compared numeric biocriteria with ambient chemical indications and showed that, out of the 645 waterbodies surveyed, biological impairment was evident in 49.8% of cases where there was *no* evidence of impairment from chemical water quality criteria alone (Courtmanche 1994).

In short, where the ultimate aim of water quality assessment is to protect the integrity of still water ecosystems, biological techniques must be the principle means of monitoring those ecosystems.

1.3 Definitions of terms used in the report

A brief set of definitions for terms which are critical to the project is given below.

1.3.1 'Biological techniques'

'Biological techniques' are taken to include all methods having a biological basis, including both *organism level* methods (such as field surveys of fish or laboratory toxicity tests of

Daphnia spp.) and *sub-organism level* methods (such as the *in vitro* enzyme-based ECLOX system).

Conceptually, all these techniques can be divided into two broad categories:

(a) methods which measure the *overall integrity* of biotic communities (using variables such as species richness, occurrence of sensitive species, population structure or levels of disease),

(b) methods which use organisms (or sub-organism level components) as *indicators* of physical and chemical degradation (e.g. macrophyte surveys to indicate nutrient status; ecotoxicological tests to indicate chemical toxicity).

This dichotomy, and its relevance to General Quality Assessment, is discussed in more detail in later sections of this report.

1.3.2 'Water quality'

The term ‘water quality’ is often used with reference to suitability for societal uses (such as irrigation, potable water supply or salmonid fisheries). Within the context of this project, however, the concern is with techniques which will allow water quality to be assessed for the biological window of the General Quality Assessment (GQA).

Following discussion at the Project Inception Meeting, it was agreed that ‘water quality' in this particular context should mean *biological water quality*, and that ‘biological water quality’ broadly implies the ability of waterbodies to support appropriate ecosystems (Royal Commission on Environmental Pollution 1992).

1.3.3 'Still waters'

For the purposes of the Scoping Study, the term 'still waters' includes the widest range of still-waterbody types likely to be appropriate to the current or future requirements of the EA (see Chapter 2). The extent of the EA's *current* responsibilities for still waters, as defined by the Water Resources Act (1991), is ambiguous. However, interpreted in its widest sense, this responsibility could include almost all standing waters, explicitly excluding only ponds which are both lined and off-stream.

In *future* the EA may be called upon to implement a monitoring programme which fulfils the requirements of the Directive on the Ecological Quality of Water. The Directive is still in draft form and is likely to be revised, but currently includes a requirement for Member States to undertake biological monitoring and classification of *all* surface waters. The main exemption is that: 'for practical reasons Member States should be authorised to exclude waters of insignificant size which have no significant effect on the quality of other waters’ (CEU 1994).

Combining the requirements of these areas of legislation, the range of still-waterbodies which the EA could potentially wish to monitor in the foreseeable future includes:

- lakes
- reservoirs
- permanent ponds
- temporary waters
- brackish ponds and lagoons
- canals
- ditches

Given the difficulties inherent in precisely defining these waterbody types, preliminary definitions are given in Table 1.1.

Identifying the boundary between still and flowing waters is particularly problematic. Ditches, for example, not only grade from still to fast-flowing, but may vary markedly in flow rate during the course of the year. In this early phase of the project, waterbody types which are on the borderline between still and flowing waters (in particular ditches and canals), are all included in the scoping study. In the course of Phases II and III, more precise definitions of the still-water/running-water boundary will need to be determined, by, for example, considering the limits of existing running-water assessment methods, such as the River Invertebrate Prediction and Classification System (RIVPACS).

Table 1.1 Definitions of still waterbody types included in the assessment

Lakes	A body of water greater than 2 ha in area (Johnes *et al.* 1994). Includes reservoirs, gravel pits, meres and broads
Permanent and semi permanent ponds	Waterbodies between 1 m^2 and 2 ha in area which usually retains water throughout the year (Collinson *et al.* 1995). Includes both man-made and natural waterbodies.
Temporary waters	Waterbodies with a predictable dry phase, usually in the order of 3-8 months (Ward 1992).
Brackish waters	Pools and lagoons containing between 500 and 30,000 mgl^{-1} sodium chloride (Allaby 1985).
Canals	Artificial channels originally constructed for navigation purposes.
Ditches	Man-made drainage channels. Includes drains and rhines.

1.4 Methods and sources

Four main sources of information were used in the preparation of this report:

(i) Published scientific literature which was accessed through online searches of the computer database maintained for UK Universities by Bath Information & Database Services (BIDS) (World-wide Web Reference: http://www.bids.ac.uk/). Databases maintained by Florida State University and US EPA were also accessed.

(ii) Information supplied by EA staff during discussion and through the use of a structured questionnaire which was circulated to Biology, Fisheries and Conservation staff in all regions.

(iii) Information supplied by water management agencies in Europe (including EU Member States and non-EU countries) on methods currently used to monitor still waters.

(iv) Discussions held with, and documentation provided by: (a) Dr James Karr (University of Washington, Seattle); (b) staff of the United States EPA; and (c) the University of Washington, during a visit to the US.

2. EA REQUIREMENTS FOR A BIOLOGICAL TECHNIQUE FOR STILL WATER QUALITY ASSESSMENT

2.1 Introduction

The EA's operational requirements are central to the development of a technique for assessing the biological quality of still waters. These requirements are therefore outlined at an early stage in this report. The following areas are covered:

- Water Resources Act (1991); Environment Act (1995).
- Additional UK and EU legislative requirements.
- UK biodiversity and sustainable development commitments.
- Additional EA operational requirements.

2.2 Requirements of the Water Resources Act (1991) and Environment Act (1995)

2.2.1 General legislative requirements for monitoring of the water environment

The EA's requirements for an effective monitoring programme are broadly determined by the framework provided by the Water Resources Act (1991), the Environment Act (1995) and EU Directives.

Under the Water Resources Act (1991), the NRA's main responsibilities were:

(i) To monitor the extent of pollution in controlled waters.

(ii) To promote the conservation of aquatic flora and fauna.

The Environment Act (1995) transferred these functions to the Environmental Agency (EA) in April 1996. The Act also gave the new Agency additional powers and responsibilities with respect to monitoring. Thus:

(i) *'It shall be the principle aim of the Agency'* .... *'in discharging its functions so to protect or enhance the environment, taken as a whole, as to make the contribution towards attaining the objective of achieving sustainable development mentioned in Subsection 3 (below).'* (Section 4/1).

(ii) The Agency is also required *'to take into account the likely cost and benefits'* when deciding to exercise its statutory powers (Section 39/1).

Through the provisions of the Environment Act (1995), the EA currently has both: (i) a *broad* responsibility to assess and monitor the quality of controlled waters in England and Wales, and (ii) a *specific* requirement to monitor waters to ensure industrial and other sector compliance with both UK legislation (e.g. Water Industry Act 1991, Land Drainage Act 1991) and EU directives (e.g. Nitrates Directive 1991, Urban Waste Water Treatment Directive 1991).

The EA largely aims to discharge both its general and specific legislative responsibilities through a nationwide programme of routine biological and chemical monitoring of controlled waters in England and Wales. This is supplemented by more limited reactive monitoring in response to specific pollution or other water protection and resource concerns.

2.2.2 The legislative requirement for biological monitoring of controlled still waters

With respect to monitoring the still waterbody types considered in this study (e.g. lakes, ponds, canals, ditches) the *precise* extent of EA responsibilities is open to interpretation.

The most explicit requirement arising from the Water Resources Act/Environment Act is a need for *pollution monitoring* of *controlled waters*. However, there is some ambiguity as to *which* still waters are included as 'controlled waters', the answer depending, in part, on the definition of the term 'discharge'. Relevant sections of the Water Resources Act state that controlled waters include:

> *'inland freshwaters, that is to say, the waters of any relevant lake or pond or of so much of any relevant river or watercourse as is above the freshwater limit'* (Subsection 104/1c),

where relevant lake or pond means:

> *'.....any lake or pond whether it is natural or artificial or above or below the ground which discharges into a relevant river or watercourse or into another lake or watercourse which is itself a relevant lake or pond'* (Subsection 104/3),

and relevant river or watercourse means:

> *'.....any river or watercourse (including an underground river or watercourse and an artificial river or watercourse) which is neither a public sewer or drain which drains into a public sewer'* (Subsection 104/3).

Controlled waters also specifically include both reservoirs (subsection 104/3) and relevant temporary waterbodies (subsection 104/2). In addition, the Secretary of State has the power to deem *any* waterbody a controlled water (subsection 104/4).

The EA has generally interpreted the term 'discharge' to include discharge to groundwater, as well as rivers and other controlled waters. In practice, therefore, since almost all still waterbodies discharge to at least groundwater, the EA's requirement for monitoring still waters potentially includes all standing-waterbody types, and only explicitly excludes waterbodies which are both lined and off-stream, such as butyl-lined farm irrigation reservoirs or garden ponds.

2.2.3 EA policy requirements for still water monitoring

The EA Water Quality Strategy (NRA 1993) commits the EA to:

(i) *'further develop the use of biological techniques for assessing the overall quality of waters and to assist in determining pollution sources'*;

(ii) *'review EA monitoring programmes to ensure a cost effective and consistent level of service for all controlled waters'*;

(iii) *'ensure that new approaches for the control and reporting for EC and other inter-governmental directives are met'*.

These commitments, as stated, indicate a requirement for biological methods to assess overall water quality. This may be taken to be a rather broader remit than a requirement to monitor the extent of pollution, as explicitly defined in legislation (see 2.2.1 above). It potentially includes changes in biological quality due to habitat degradation or the introduction of alien species (such as non-native crayfish or Zander).

2.2.4 Current extent of still water quality monitoring to fulfil Water Resources Act/Environment Act requirements

The results of the survey questionnaire completed by EA staff for this project (see Appendix 2) suggest that, in practice, the main types of still waters *currently* monitored routinely for biological water quality purposes are canals (70+ sites nationally per year), ditches (75+ sites) and lakes (c. 60 sites, including the Norfolk Broads)[1]. No ponds, temporary ponds or brackish waters are regularly monitored. In addition, a relatively small number of ditches, ponds, lakes and canals are *reactively* monitored (less than 80 sites in total per year[1]) in response to specific pollution concerns or incidents.

Apart from work undertaken to fulfil EC Directive requirements, most routine sampling uses macroinvertebrate methods, either based on RIVPACS assessment methods (in ditches and canals) or other techniques. Lakes (including some reservoirs) and ponds are also reactively monitored using a variety of plant, fish and invertebrate methods, for which there is no fully standardised inter-regional approach.

The EA regional survey returns suggest that, for still waters generally, no routine biological monitoring relating to the hydrological integrity of still waters or to the granting of abstraction and drainage licences is undertaken.

2.2.5 Implications for development of a biological monitoring method

Overall, existing UK legislation and policy, as well as existing operational practice, indicates that there is a need for still water biological monitoring methods which:

- are based on assessment of the existing quality of waterbodies,
- assess overall biological water quality, although perhaps with an emphasis on pollution,
- provide, where possible, additional information on pollutant causes,
- are standardised between regions and within waterbody types,
- are suitable for use in the General Quality Assessment and for detecting change,
- provide sufficient information to fulfil EC and other statutory requirements (see below),
- are cost-effective,
- can be used for monitoring of lakes, canals, ditches, ponds, temporary ponds, and brackish still waters.

Neither UK legislation or EA policy statements place any restriction on the *types* of methods, or indicator groups, which could be employed to fulfil these monitoring requirements.

2.3 EU Directives

EU Directives which relate to water quality are implemented in the UK either under the Acts and associated Statutory Instruments, or directly through the policy and powers of the EA.

The main Directive requirements which are likely to influence development of biological assessment method(s) for still waters are outlined, in date order, below.

2.3.1 Fisheries Directive (78/639/EEC)

The Fisheries Directive (1978) sets physico-chemical limits for designated salmonid and cyprinid waters. Compliance with the Directive is largely addressed through the chemical water quality monitoring programme. However, where there are failures for total ammonia, zinc or low pH, derogation may be granted at that time if there is a healthy fish population. Directive compliance may therefore require limited or occasional biological monitoring in waters (largely canals) designated under the Directive.

[1] These figures exclude monitoring of blue-green algae in response to toxicity concerns.

Current draft proposals in the Directive on the Ecological Quality of Water indicate that this Directive will replace the existing Fisheries Directive, if implemented (CEU 1994).

2.3.2 Environmental Assessment Directive (85/337/EEC)

The Environmental Assessment Directive (1985) makes Environmental Impact Assessment (EIA) mandatory where specific scales and types of developments are proposed. Where a project requires planning permission, an EIA may be required under the general regulations (SI 1199). A project not subject to planning regulation (e.g. work carried out under the terms of a General Development Order) may still require an EIA under SI 1217.

Within the EIA planning process, EA is a statutory consultee wherever preliminary screening indicates a water component to development proposals. Regional EA survey returns from staff indicate that the EA occasionally undertakes macrophyte and macroinvertebrate (including dragonfly) surveys on ditches and lakes (<20 per year) in order to identify the value of standing waters or to monitor development impacts during or after construction. More usually, the EA advises on, or sets standards for, survey techniques that developers/consultants should themselves use when undertaking assessment surveys.

Most river engineering/maintenance works are covered by Annex 2 of 85/337/EEC, for which EIA is discretionary rather than mandatory. However, the EA frequently undertakes an internal assessment of existing site value and sensitivity prior to works being carried out. Regional survey returns indicate that the EA mainly undertakes such environmental assessments on ditches (65+ km per year) and canal sections (3+ per year), largely using river-corridor and river-habitat type techniques.

2.3.3 Urban Wastewater Treatment Directive (91/676/EEC)

The aim of the Urban Wastewater Treatment Directive (UWWT Directive) is to protect the North Sea and other community waters from pollution likely to cause eutrophication, through control of discharge of urban waste water effluents. The Directive focuses on 'sensitive' (i.e. eutrophic or potentially eutrophic) waterbodies, with the aim of encouraging secondary treatment of effluents from larger urban and industrial agglomerations (generally >10,000 p.e.). Eutrophication, in the terms of the Directive, is defined as '*....enrichment of water by nutrients, especially compounds of nitrogen and/or phosphorous, causing accelerated growth of algae and higher forms of plant life to produce an undesirable disturbance to the balance of organisms present in the water and to the quality of water concerned*' (Article 2/11).

In undertaking still-water monitoring to ensure UK compliance with the UWWTD, the EA's objectives have principally been to identify relevant sites where poor urban wastewater treatment is likely to be demonstrably impacting aquatic communities and, in particular, to distinguish the effects of eutrophication from other toxic effects of urban wastewater.

A survey questionnaire to EA staff suggests that approximately 40 still waters (around 10 lakes, 6 ditch sites, and 23 canal sites) are currently monitored by EA in order to ensure compliance with the Directive. Assessment methods are not specifically defined, and either chlorophyll a or macrophyte methods are currently used.

2.3.4 Nitrate Directive (91/676/EEC)

The Nitrate Directive, as its full name indicates, concerns the 'Protection of Waters against Pollution caused by Nitrates from Agricultural Sources'. The Directive is currently still in its implementation stage. Compliance with the Directive is based on chemical monitoring of nitrate levels. As part of general data collation to indicate the existing vulnerability of surface waters to eutrophication, EA North West Region currently monitors chlorophyll a in approximately 20 lakes (mainly Cheshire/Shropshire Meres).

2.3.5 Habitats Directive (92/43/EEC)

The Habitats Directive (1992) concerns the designation of Special Areas of Conservation (SACs) for the protection of natural habitats and of wild fauna and flora. The Directive also lists priority habitat types and species for which provision should be made.

Priority standing-water habitats include:

- Natural eutrophic lakes with Magnopotamion or Hydrochariton-type vegetation.
- Hard oligotrophic water with benthic vegetation of *Chara* formations.
- Oligotrophic waters containing very few minerals of Atlantic sandy plains with amphibious vegetation: *Lobelia, Littorella* and *Isoetes*.
- Oligotrophic water in medio-European and perialpine areas with amphibious vegetation.
- *Littorella* or *Isoetes* or annual vegetation on exposed banks.
- Dystrophic lakes.

Priority species found in standing freshwaters include: *Austropotamobius pallipes* (the Atlantic Stream or White-clawed Crayfish) and *Triturus cristatus* (the Great Crested Newt).

The Directive specifies that '*a system should be set up for surveillance of the conservation status of the natural habitats and species covered by this Directive*'. The term 'conservation status' '*means the sum of the influences acting on a natural habitat and its typical species*'.

Much of the responsibility for ensuring compliance with the Habitats Directive lies with the statutory nature conservation agencies. However, under reciprocal agreements with English Nature EA has an agreement to provide monitoring information for freshwaters.

2.3.6 Draft Directive on the Ecological Quality of Water

In the next few years, the EA may be called upon to implement a monitoring programme which fulfils the requirements of the Directive on the Ecological Quality of Water (EQW). The Directive is, however, still in draft form, and there have been recent suggestions that the initial guidelines, published in 1995, may be modified. It currently appears that the EQW Directive will be included as a daughter directive of the framework 'Water Policy Directive', and that its requirements are likely to be less rigorous and/or wide-ranging than was originally proposed.

The Directive guidelines, as they stand, require all Member States to undertake a systematic ecological assessment and three-yearly monitoring programme for *all* surface waters. Exemptions are, however, likely to be authorised for '*....waters of insignificant size which have no significant effect on the quality of other waters*' (CEU 1995).

These can include:

(i) hydrologically isolated lakes or groups of interconnecting lakes less than 1 km^2 in surface,

(ii) fresh or brackish waters discharging less than 20 million m^3 annually into marine waters,

(iii) freshwaters (including lakes) which discharge less than 2 million m^3 annually into other freshwaters.

With respect to still waters within the EA's remit, the Directive may, therefore, apply mainly to canals, large lakes and reservoirs.

The draft Directive currently requires that operational targets should be set and reached for an extensive range of taxa, including plants, fish, invertebrates (planktonic and benthic), birds, amphibians and mammals. For each of these groups, assessments are required based on:

(i) diversity in comparison with the *undisturbed condition*, i.e. with insignificant anthropogenic disturbance,

(ii) *key species/taxa* normally associated with the undisturbed condition.

There are additional requirements that there should be (a) no evidence of excessive macrophytic or algal growth; (b) no elevated disease levels in animal life (including fish) and plant life; and (c) no artificial hindrance to migratory fish passage.

The draft Directive suggests that use and manipulation of data produced from monitoring should conform with technical specification drawn up by the Commission (current date: 1999) so as to ensure comparability of monitoring data and the determination of ecological water quality.

2.3.7 Summary of implications from EU Directive requirements for development of an EA biological monitoring method

The most specific biological monitoring requirements arising from implementation of EU Directives relate to the provisions of the UWWT Directive and, provisionally, the Ecological Quality of Water Directive.

1. The *UWWT Directive* requires use of nutrient monitoring and assessment methods which can be used to (i) prove environmental damage in areas relevant to the Directive and (ii) indicate that it is, specifically, nutrients (and not other pollutants or physical damage) which are causing adverse biological impacts. In comparison with rivers, the number of canals and lakes likely to be affected by the Directive is small but, nevertheless, monitoring to ensure compliance with the Directive is mandatory.

2. Unlike other directives or UK legislation, the draft *Ecological Quality of Water Directive* explicitly identifies the monitoring approach which should be employed, and the range of taxa which should be monitored. Full compliance with the Directive, as it stands, requires methods based on:

- development of an undisturbed state baseline for a wide variety of taxa in (at least) larger lakes and canals,
- monitoring, classification and targets using species diversity and key species across a broad range of waterbody habitats (benthic, planktonic etc.). Population disease levels also need to be addressed,
- technical specification for data manipulation which ensures comparability between member states.

Other requirements arising from EU Directives are relatively minor in terms of either the number of sites likely to be involved or the EA's apparent requirement to develop a monitoring method. This said, it is worth noting that although the EA has no specific requirement to develop monitoring methods to facilitate Environmental Impact Assessment monitoring in still waters, the development of such methods would be likely to promote higher standards of practice than is currently typical of EIAs.

2.4 UK biodiversity and sustainable development commitments

At the Earth Summit in Rio de Janeiro in 1992, the UK made commitments to the promotion of sustainable development and the protection of biodiversity. In 'Biodiversity: UK Action Plan', the UK Government has set as its overall goal the objective:

> *'To conserve and enhance biological diversity within the UK, and to contribute to the conservation of global biodiversity through all appropriate mechanisms'* (HMSO 1995).

The UK Biodiversity Action Plan lists as its key habitats: mesotrophic standing waters, eutrophic standing waters and aquifer-fed, naturally fluctuating, water bodies. It also identifies key species such as *Triturus cristatus* (the Great Crested Newt), *Myxas glutinosa* (the Glutinous Snail), *Segmentina nitida* (the Shining Ramshorn Snail), *Hirudo medicinalis* (the Medicinal Leech), *Damasonium alisma* (Starfruit), *Arvicola terrestris* (the Water Vole), and *Luronium natans* (Floating Water Plantain).

In considering issues which would make a major contribution to sustaining and enhancing biodiversity, the Action Plan also makes a recommendation that the UK should generally aim to create '*improved or maintained water quality or quantity*'.

Specific commitments to the Biodiversity Convention are largely discharged under the UK Action Plan programme, to which EA is a consultee and may play a contributing role.

2.5 Additional EA operational requirements

In addition to its main routine and reactive biological water quality monitoring programme the EA undertakes biological assessments in a number of other areas.

2.5.1 Public nuisance

The EA undertakes a relatively extensive programme of planned and reactive monitoring of blue-green algae in response to public health concerns. Over 330 sites are monitored annually, and in excess of 520 sampling visits made. Blue-green algal blooms are a problem associated with eutrophication, but although toxic blooms have been reported to affect some aquatic species (particularly fish), in practice this is largely a human nuisance issue.

2.5.2 General concern

EA receive public enquiries about a wide range of still-water issues of specific concern. These include, for example, issues such as colour change in water, falling water levels, fish mortality, botulism, 'red leg' in amphibians, and invasion of *Crassula helmsii* (New Zealand Swamp Stonecrop). Ponds, and to a lesser extent canals and lakes, are the main waterbody types concerned. Most enquires are dealt with through correspondence or phone calls, but some responses require a site visit.

2.5.3 General advice

Lakes and ponds are occasionally surveyed for macrophytes in order to provide 'weed control' advice to the public. Most types of still water (canals, ponds, lakes, brackish waters) are also monitored occasionally (about 30 sites in total annually) to provide management or conservation value information to aid management. Such monitoring, which largely employs macrophyte or macroinvertebrate (including dragonfly) assessments, is undertaken both for EA projects and for other independent conservation projects or sites.

2.5.4 Fisheries advice

On request, fish population surveys (i.e. numbers, biomass, diversity) are undertaken, as part of fisheries management service, in order to give advice on fish health and stock to anglers, clubs, fishery owners and managers. About 45 lakes and 25+ ponds are monitored. Invertebrates, in terms of 'food-for-fish', are also monitored in 25+ ponds, lakes and canals.

2.5.5 Specific projects

EA Anglian Region undertakes intensive surveys of phytoplankton, zooplankton, macrophytes and macroinvertebrates in 17-20 waters as part of regional research and development work, largely on the Norfolk Broads. Midlands Region are monitoring fish as part of a long-term study of the effect of Zander in canals.

2.6 Overall requirements for development of a biological monitoring method

Evaluation of legislation, relevant policies, and current EA practice suggests that, overall, the requirements for biological assessment methods are:

1. Development of standardised method(s) of general water quality assessment which can be used to *indicate the existing quality* of still-water sites, and, through regular monitoring, provide an assessment of quality change.

2. Development of assessment methods which are suitable for canals, lakes, ponds, ditches, temporary and brackish waters.

   Of these waterbody types, requirements to fulfil the UWWT Directive and, potentially, the Ecological Quality of Water Directive, currently suggest a monitoring focus on lakes, canals and perhaps ditches. Conservation/wildlife requirements and public enquiries more usually relate to monitoring of ponds and lakes. Brackish and, particularly, temporary waters, although rarely specified, are important in maintaining UK freshwater biodiversity, including an anomalously high proportion of rare and protected species.

3. Where possible, monitoring should aim to provide information on specific causes of pollution. There is a particular requirement for a method to distinguish nutrient impacts from other toxic effects of sewage in order to fulfil the UWWT Directive requirements.

4. There is a general requirement that monitoring methods should be:
   - cost effective,
   - compatible throughout EA regions and, where possible, with methods used in other EU Member States.

5. Finally, in addition to the above, if the relevant recommendations from the draft Ecological Quality of Water Directive are adopted, there will be a requirement for monitoring methods which involve:
   - development of undisturbed state baselines for a wide variety of taxa in larger lakes and canals,
   - monitoring, classification and targets, across a broad range of waterbody habitats, using species diversity, key species and disease parameters,
   - standardised reporting protocols.

3. A CONCEPTUAL FRAMEWORK FOR EA MONITORING OF STILL WATERS

3.1 Introduction

Biological techniques now play a central role in the monitoring of freshwater ecosystems and are used to evaluate water quality world-wide. In consequence, hundreds of assessment methods, using a variety of sampling procedures, analytical techniques, and taxonomic groups have been developed. Finding a path through this methodological maze to identify the most appropriate method for any given situation can be a daunting task.

The aims of this chapter are therefore two-fold: first, to provide an overview of the main methodological approaches to water quality assessment; and second, to provide a rationale for still-water monitoring which best fulfils the EA requirements (outlined in Chapter 2).

The end result of this evaluation process is the development of a conceptual framework for EA still-water monitoring in England and Wales (see Section 3.7). Within the context of this framework, assessment methods developed in the UK, Europe and North America are described (Chapter 4). In later chapters (Chapters 5 and 6) this framework provides the basis for more detailed method development.

3.2 What are biological assessment methods used for?

Over the last 25 years there has been a profound shift in thinking about the way in which water resources should be protected. Traditionally, freshwater ecosystems were managed by maintaining physical and chemical water quality for a specific use (such as abstraction for drinking-water supplies). More recently, there has been explicit recognition that long-term, sustainable exploitation of freshwater ecosystems is only viable if the overall biological integrity of ecosystems is maintained.

As described in Chapter 1, this shift in our approach to resource exploitation has also changed the role of biological monitoring. Previously, biological assessment was simply another way of detecting change in the physical and chemical environment. Now, biological monitoring provides the only practical way in which the benchmark of biotic integrity can be established, and the effects of resource use properly monitored.

3.2.1 Approaches to biological monitoring of water quality

Assessment of water quality involves one or more of three essentially different methodological approaches:

1. *General ecosystem quality assessment* to evaluate the current overall condition of the aquatic ecosystem.
2. *Diagnostic assessment* of the reasons for ecosystem degradation where it is observed.
3. Assessment for the purpose of providing an *early-warning* system, to identify impending damage before it becomes widely prevalent.

In the following sections each of these three approaches is discussed briefly. In the remainder of the chapter, general ecosystem assessment, which is the approach most relevant to EA requirements, is evaluated in more detail.

3.2.2 General ecosystem assessment

General ecosystem assessment methods can serve a variety of purposes and, for the EA, they form the core of the methods used to assess controlled waters for the biological window of the five-yearly General Quality Assessment (GQA).

Overall, general ecosystem assessment provides:

1. A *routine monitoring technique for general surveillance* of the quality of still waters which can be used to:
   - give an overall assessment of the status of still waters, providing data for policy makers concerned with catchment management issues,
   - provide an assessment of temporal trends in the condition of still waters which can be used to detect degradation of sites or to track the effectiveness of remedial measures,
   - enable identification of spatial differences in water quality, making it possible to locate specific problem areas for further attention and reactive control.
2. A means of fulfilling *mandatory monitoring requirements* (e.g. monitoring to ensure compliance with Directives or SWQOs).
3. A technique for *reactive monitoring* which can be used to assess the extent of pollution or other damage at specific sites.

The ultimate aim of any general ecosystem assessment method is to encapsulate and summarise the overall quality of the ecosystem under consideration. Methods which are likely to be appropriate for such a wide-ranging purpose will almost invariably need to be broadly based themselves in terms of the taxa and attributes that are measured. Community and ecosystem measures are therefore more likely to fulfil this condition than indicator species or ecotoxicological techniques. Suitable taxa for monitoring are likely to be those which span a number of trophic levels, occupy a variety of waterbody habitats (e.g. can be found in the littoral zone and open water) and are long-lived, so that they can provide a temporally and spatially integrated measure of the current ecosystem state.

The information derived from general ecosystem assessments must allow sites to be ranked according to their status ('classified' in EA terminology). Grading can be undertaken: (i) internally using the data collected from other similar surveys; or (ii) with respect to a fixed external benchmark. There is now an increasing consensus amongst freshwater ecologists, water managers and legislators, that the fixed external benchmark approach is preferable, and that the reference should be 'undisturbed' examples of the habitat type (EPA 1994, CEU 1994, Wright *et al.* 1984, Johnes *et al.. 1994*).

Where the range of monitoring sites assessed is varied in type, or covers an extensive geographical area, a classification is required to enable comparison with similar sites. Thus, in describing sites as 'poor' or 'good', it may be unreasonable to compare the number of species in a small waterbody with the number in a large site. Classification provides a framework in which like can be compared with like.

The essential requirements for general ecosystem assessment methods, which are needed for the biological window of the EA GQA, are therefore that:

1. they should be broad-based and holistic in approach;
2. an external benchmark is available, so that site quality can be compared and assessed 'objectively' against this criterion;
3. if the sites are physically or chemically varied, or if they occur over an extensive region, the results need to be assessed within the context of a classification.

3.2.3 Diagnosing the causes of damage

Biological methods have long been used as a means of diagnosing the causes of environmental degradation. Chlorophyll a trends are, for example, a simple and often reliable indicator of eutrophication, and loss of salmonids from lakes on base-poor substrates can help to indicate acidification.

For the EA, diagnostic assessments are likely to be necessary for a number of reasons, including:

1. a requirement to diagnose or monitor particular impact types (e.g. chlorophyll a measurements or macrophyte surveys to indicate levels of lake eutrophication) to fulfil legislative or internal policy requirements,
2. the need to further investigate the causes of an observed degradation in order to prevent further damage or stimulate restoration,
3. the potential to monitor the success of mitigation measures where a technique specifically related to the impact may be more useful than a general ecosystem assessment.

In practice, identifying the cause of environmental degradation is frequently considerably more difficult than merely observing that an impact has occurred. This is in part because ecosystems are often affected simultaneously by a number of impacts (physical, chemical and biological), but the problem is exacerbated because even a single impact type may have a number of knock-on effects. Nutrient enrichment, for example, may, in addition to its primary effects, increase deoxygenation of the water column and increase the release of sediment-stored toxins.

Because of their diverse and highly specific functions, the main characteristics of diagnostic assessment methods are very different to those of general ecosystem assessment methods. Diagnostic methods must single out causes rather than integrate them and are, typically, reductionist rather than broadly applicable. Ideal techniques are therefore more likely to be based on a limited range of indicator species or taxa, or on individual attributes (e.g. bioaccumulation, deformity levels etc.), rather than a range of taxa or attributes. Features which are particularly desirable are:

- a strong link to specific physical or chemical impacts,
- a high level of discrimination between potential impacts (e.g. the ability to distinguish nutrient impacts from organic effects).

Because of the considerable range of potential impacts and the required specificity of indicators, it follows that no one diagnosis indicator or method is likely to be applicable in all situations. Indeed, the complexity of impact effects means that, even at a single site, a variety of approaches may be required.

Overall, this indicates that in any water quality assessment programme, there will be a need for an array of complementary indicators which can be used to help diagnose the source(s) of degradation. The requirements for diagnostic indicators will vary: where specific needs exist, (e.g. compliance with legislation such as the UWWT Directive), individual methods which are highly tailored to a particular purpose may need to be developed. In other cases, where causes of degradation are unknown, flexibility is required and a range of diagnostic techniques may need to be employed. These may be biologically based (see Chapter 4) but other complementary approaches - desk studies of historical data, hydrological investigations or chemical monitoring - are likely to be equally, and additionally, relevant (EPA 1994).

3.2.4 Early-warning systems

Early-warning diagnostic indicators are a special category of methods which focus on *prevention* of an adverse effect. At present, these indicators are largely conceptual; there are no methods which have a proven track record, although a variety of techniques (e.g. ECLOX and various fish physiological biomarkers) have potential to be developed.

Whereas a general ecosystem assessment method can only document the occurrence of existing damage, the purpose of early-warning indicators is to identify impending problems before they have a substantial impact on the ecosystem. It follows from this that such methods will usually measure changes in the biota or biological processes which are *normally considered to be insignificant* (Cairns *et al.* 1993). In addition, the indicator must respond, be measured, be interpreted, and initiate management action in sufficient time to head off significant damage. In demanding a quicker response to stressors, early-warning indicators will usually measure changes which are of small magnitude and of little immediate significance.

Measurements performed on individuals or sub-organism material (e.g. enzyme analysis) will tend to be better diagnostic and early-warning indicators than measurements on communities or populations. For this reason, bioassays and biomarkers are increasingly being recognised as potential early-warning indicators.

There is a strong case to be made for undertaking early-warning monitoring as a supplement to general ecosystem monitoring. Predictive monitoring of this kind provides the best long-term solution to protection of aquatic resources. This is especially true of still waters, such as lakes, where many impacts are hard to reverse. By providing an indication of impending environmental degradation before damage is pervasive in a system, such methods not only prevent unnecessary damage, they also prevent the necessity for remediation.

Given the poor understanding and considerable costs of ecosystems restoration, early warning indicators may be a cost-effective addition to a total reliance on reactive management (Cairns *et al.* 1993).

3.2.5 Combining assessment methods: a protocol

It is clear from the sections above that different water quality requirements demand different methodological approaches: general ecosystem assessment requires integrative and holistic methods which assess ambient state; diagnosis requires a selection of indicators that can be flexibly tailored to specific impacts of interest; whilst early-warning methods need to measure the seemingly insignificant changes which precede and accurately foretell wider damage.

Under certain circumstances (though very few) it may be possible to combine more than one of these methodological functions into a single technique. Where waterbodies are impacted by a single environmental stressor, for example, it is possible to assess the quality of waterbodies purely in relation to that stress factor - providing both general ecosystem assessment *and* a diagnosis of damage caused by that impact. Such cases are, however, rare. Most waterbodies are impacted by a multiplicity of physico-chemical and sometimes biological impacts. Under these circumstances, combining general ecosystem assessment and diagnosis is likely to compromise the effectiveness of both.

The problems associated with mixing approaches can be clearly identified where specific biotic indices have been applied to general ecosystem assessment in existing EA GQA work. Thus, in rivers, the BMWP system specifically weights taxa according to oxygen tolerance. This weighting is not, however, equally applicable to damage caused by factors such as acidification, nutrient enrichment or habitat damage. Summed scores and indices will therefore misrepresent (usually underestimate or ignore) impact from other sources, and ultimately distort assessments describing the level of damage suffered by the ecosystem.

Therefore, where both general ecosystem assessment and diagnosis is required at a site, a rational approach is to consider these different processes as part of a two-stage protocol:

Stage 1 *General ecosystem assessment*, which evaluates the net effect of all forms of degradation;,

Stage 2 *Diagnosis*, a more detailed follow-up investigation, used where damage is evident, and employing one or more of an array of appropriate techniques.

A major benefit of this two-stage protocol is that it provides a general framework which can be used in any area and for any waterbody type, irrespective of the causes of damage.

It is worth noting, however, that separating out the different functions of biological techniques does not preclude re-analysis of data to fulfil more than one role. Thus, there is potential for data already collected for general ecosystem assessment, to be analysed separately, providing additional and independent indices relevant to specific impacts (eutrophication, biocides, acidification, habitat damage etc.). Where such an approach is realisable, this double use of data has the potential to provide a highly cost-effective method of monitoring and assessing water quality.

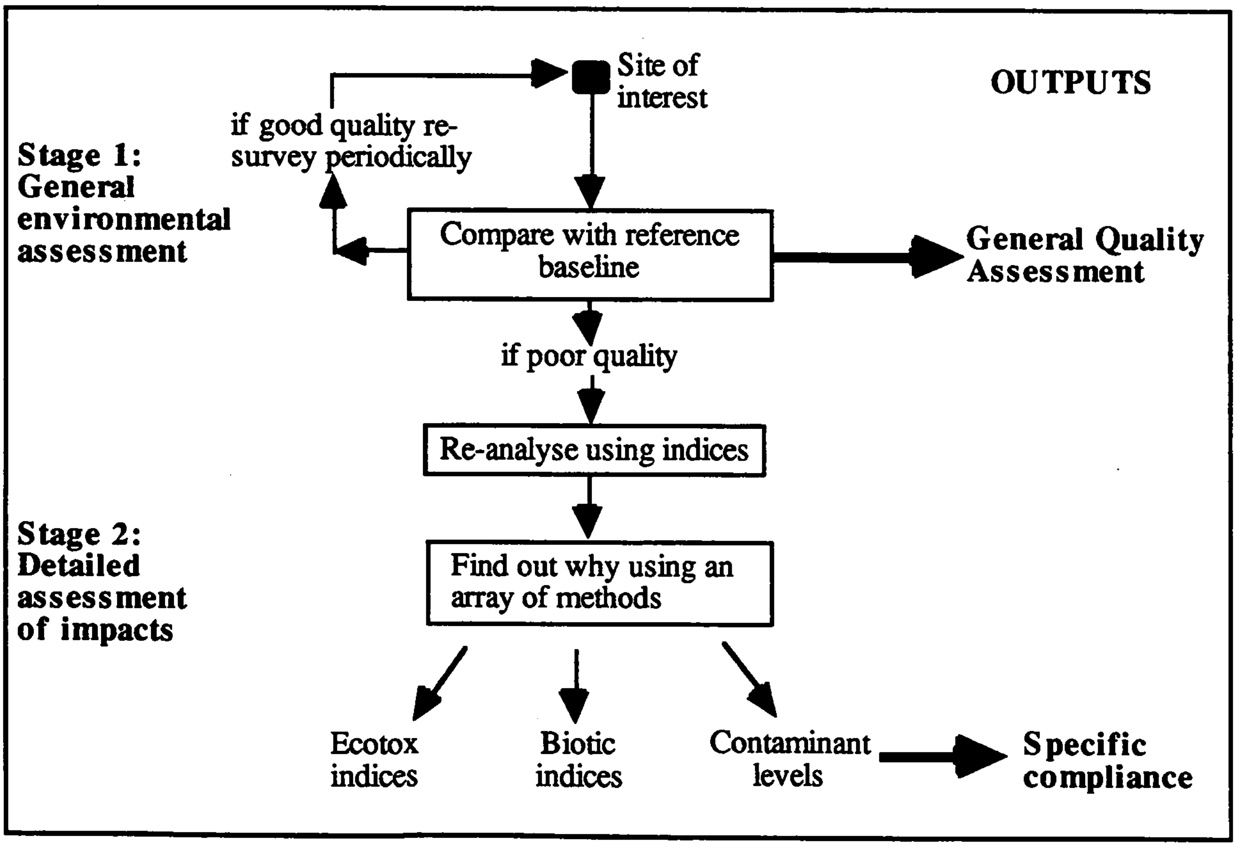


Figure 3.1 Biological assessment techniques: a framework

3.3 Undertaking general ecosystem assessments - more detailed discussion

3.3.1 Introduction

The underlying objective of R&D project 642 is to develop biological monitoring techniques which can be used for GQA and for setting still-water SWQOs. As outlined in Section 3.2.2, these requirements essentially demand the development of methods of general ecosystem assessment which can be used for surveillance of ambient water quality. The remainder of this chapter develops aspects of general ecosystem assessment in more detail.

3.3.2 Underlying principles of general ecosystem assessment

The essential requirement of a general ecosystem assessment in any waterbody is that it should represent and summarise the existing quality (often termed 'health') of that water body. Consideration of the multitude of elements which make up biological quality indicates that this is no small task.

A full assessment of quality, potentially, includes all elements which make up 'biodiversity', i.e. the *'variety of the earth's naturally occurring biological elements, which extend over a broad range of organisational scales from genes to populations, species, assemblages and landscapes'* (Karr and Dudley 1981) and 'biological integrity', i.e. *'the capability of supporting and maintaining a balanced, integrated, adaptive community of organisms having a species composition and functional organisation comparable to that of the natural habitat of the region'* (Karr and Dudley 1981), (see Table 3.1).

Full assessment of the biological integrity of any freshwater system on the basis of *all* its components is logistically impossible. General biological quality assessment relies on the premise that it is possible to select a more restricted set of biological variables that will reliably represent overall ecological integrity. Given that every measurable parameter has some value with regard to assessing environmental conditions, and that the number of potential indicators is virtually infinite, selection of the few 'best' indicators from this vast array is by no means a simple exercise.

The difficulty is compounded because suitable groups (taxa) and attributes (e.g. species richness, rarity) need to be 'suitable' not only in terms of their *scientific merits* (i.e. they need to work) but must be *practically viable* and *cost-effective*. In Chapter 5, these three methodological aspects - all of which must be present in any viable assessment method - are used as criteria to evaluate the most suitable taxonomic assemblages for use in water quality monitoring. In this framework chapter, however, it is the *scientific* credibility of assessment methods that is of primary concern and which is discussed further below.

3.3.3 Scientific validity of biological assessment methods

In theory, it should be possible to measure biological integrity using any or all of the components of integrity listed in Table 3.1: i.e. biodiversity elements, attributes and processes. In reality, it is widely acknowledged, even by those most interested in promoting biotic integrity concepts, that *process* components are highly complex and time-consuming to measure (Angermeier and Karr, 1994). As such, it is rarely practicable to include them in monitoring programmes.

General ecosystem assessments therefore revolve around monitoring the two remaining aspects of integrity: elements (genes, individuals, populations) and attributes (species-richness, trophic structure). Of these, the pragmatic requirements of monitoring programmes demand that biodiversity *elements* are principally monitored at species or higher taxa level - rather than considering aspects such as genes or population dynamics.

Table 3.1 Examples of the components of ecological integrity

Elements of biodiversity	Attributes of biodiversity	Processes relating to ecological integrity[1]
Gene	• Diversity • Relative abundance and dominance • Rarity • Purity (endemic/exotic)	• Mutations • Recombination
Individual, population	• Rarity • Disease • Size spectra • Biomass • Endemic/exotic etc.	• Metabolism: growth, reproduction • Population sources and sinks • Evolution/Speciation • Recruitment • Dispersal • Demography: age specific, birth and death rates
Assemblage, community, ecosystem	• Taxa richness • Rarity • Diversity • Relative abundance and dominance • Guild structure • Trophic structure/complexity	• Competition/predation • Diseases/parasitism • Energy flow • Nutrient cycles • Metapopulation dynamics
Landscape	• Size • Diversity • Rarity • Isolation/arrangement	• Population sources and sinks • Fragmentation • Water Cycle • Nutrient Cycles

[1]Ecological integrity = the sum of elements (biological diversity) and processes.

Modified from Angermeier and Karr (1994)

General ecosystem assessment techniques must essentially comprise methods based on one or more taxa or assemblages, measured using one or more attributes. Simply stated: choosing the best assessment method comes down to identifying which is the best taxonomic group (or groups) to use and which are the most appropriate means of measuring them (diversity, sensitivity, etc.).

In practice, choosing 'appropriate' taxa and attributes requires a combination of both empirical evidence and theoretical reasoning from ecological principles.

3.3.4 Reducing the number of elements and attributes to be monitored

Choosing the most suitable taxa

The most important property of any taxonomic group to be used as an indicator of environmental quality is that its responses should accurately reflect or predict those of other species or taxa in that waterbody (Williams & Gaston 1994).

Ideally, indicator taxa should be selected on the basis of empirical evidence. However, for most still-water systems, and particularly non-lake waterbodies, there are remarkably few studies which demonstrate the relationships between different biotic groups or indicate the ways in which different components of the biota respond to stressors (Kremen 1992, Angermeier and Karr 1994).

The alternative to empirical evidence as a basis for choosing indicator taxa is to select on theoretical grounds. In effect, this entails combining our existing knowledge of taxa characteristics with basic ecological principles. From such a basis, good indicator taxa are likely to be:

1. *Groups that are sufficiently well known taxonomically.*

   Use of taxa which are taxonomically obscure will make any assessment non-viable. Good indicator taxa are therefore likely to be groups which are well-studied and for which there is sufficient information with which to evaluate observed patterns (i.e. fish, birds, amphibians, zooplankton, zoobenthos, meiofauna, phytoplankton and periphyton).

2. *Groups sensitive to a wide variety of impacts.*

   For a general assessment method, the best indicators will be groups which respond to the widest possible range of relevant impacts i.e. impacts known to be associated with anthropogenic stress (see Table 3.2).

Since a general quality assessment is concerned with the overall status of waterbodies, the taxa monitored should be those which reflect the widest range of potential stress factors. This suggests that the following criteria are also relevant:

3. *Taxonomic groups which are naturally species-rich.*

   Groups rich in taxa inherently represent a greater proportion of total biodiversity than species-poor assemblages and are, in addition, likely to respond to a wider range of environmental stresses.

4. *Groups which span a number of trophic levels.*

   Groups which span a number of trophic levels are likely to integrate the effects of a wider range of impacts than trophic specialists.

5. *Groups which occupy a wide variety of waterbody habitats.*

   The wider the range of habitats in which a group is found, the more likely it is that the group will represent the total range of stress factors.

Overall, it is clear from the above that, given the range of stressors with the potential to affect waterbodies (see Table 3.2), monitoring of a wide range of taxa will normally be required to detect these impacts. It is likely that even a major assemblage (macroinvertebrates, macrophytes etc.) will not have the capacity to represent all ecosystem stresses alone.

The implication is that surveys targeting several assemblages may ultimately be necessary to detect a wide range of stresses and to protect the majority of the ecosystem (Kremen 1992).

Table 3.2 The main causes of anthropogenic impacts on freshwater systems

Energy source	Type amount and particle size entering waterbody, seasonal pattern of energy availability
Water quality	Temperature, turbidity, DO, acidity, alkalinity, organic and inorganic chemicals
Habitat structure	Substrate type, water depth and current, spatial and temporal complexity of physical habitat
Hydrological regime	Water volume and temporal distribution of flow
Biotic interactions	Competition, predation, disease, parasitism, mutualism

Adapted from Karr 1987

Reducing the number of attributes used

The most important characteristic of useful attributes is that they are direct correlates of ecosystem degradation and can, therefore, be used to clearly discriminate between sites of differing water quality.

The list of attributes which could potentially be used is extensive. Refining the choice of 'best attributes' is made easier, however, by data from a wide range of empirical studies examining degraded systems world-wide (e.g. Wilhm and Dorria 1968, Hughes and Noss 1992, Fausch *et al.* 1990, Margalef 1963, Gray 1989, Kelly and Harwell 1990). These, and many other studies, provide clear evidence that ecosystem degradation is associated with highly symptomatic changes in biotic communities. A compilation of the most frequently cited changes is given in Table 3.3 below.

This list can, however, only provide us with a set of likely candidates - a rough indication of the range of attributes which should be investigated further. For any waterbody type or any assemblage, knowledge of the attributes which will prove most useful in tracking degradation, can only be derived from real and appropriate data gathered from the region of interest.

Thus, although choice of 'best taxa' may be rationalised from a knowledge of assemblage characteristics, determining best ways to measure those taxa will ultimately rely on the collection and analysis of field data.

In principle, the attributes initially investigated in field trials should be as extensive as possible, spanning a wide range of community features and interactions (e.g. species/family richness, wet weight, disease, proportion of sensitive taxa).

Currently, most European monitoring methods use a relatively restricted set of biotic measures for water quality assessment - typically, diversity, relative abundance or taxa richness. Examination of Table 3.3 indicates that, in practice, there is the potential to use a much wider range of attributes. Thus, even from simple taxa lists it would be possible to derive measures such as percentage of exotics, proportion of functional feeding groups, ratio of predators/ herbivores and, possibly, rarity. Such measures where proven relevant in analysis, may have the potential to considerably broaden the assessment of ecosystem integrity with little extra resource requirement.

Table 3.3 Attributes associated with ecosystem degradation

- the number of native species declines
- the number of nationally uncommon or rare species declines
- the percentage of exotic or introduced species or stocks increases
- the number of generally intolerant or sensitive species declines, whilst the percentage of the assemblage comprising generally tolerant or insensitive species increases
- the percentage of trophic and habitat specialists declines, whilst generalists increase
- food-chain length decreases
- the incidence of disease and anomalies increases
- the percentage of large, mature or old-growth individuals declines
- reproduction of generally sensitive species decreases
- the number of size- and age-classes declines
- stability decreases, i.e. spatial or temporal fluctuations are more pronounced

Sources: Ulanowicz 1990, Kay; 1990, Wilhm and Dorria 1968, Hughes and Noss 1992, Fausch *et al.* 1990, Margalef 1963, Cairns *et al.* 1993, Williams & Gaston 1994, Gray 1989, Kelly and Harwell 1990, Resh and Jackson 1993, Angermeir and Karr 1994.

3.4 Establishing a baseline reference condition

3.4.1 Introduction

A central question in the development of any general biological assessment method is: 'How is good biological water quality to be defined?'. What bench marks or information do we use for comparison, and what are acceptable and unacceptable deviations from those bench marks? In this section, several ways of defining reference conditions are described, and those most appropriate for still waters are discussed.

It is now accepted by most ecologists, and by an increasing number of regulatory authorities (e.g. the EU and US EPA), that physical, chemical and biological conditions in all waterbodies should, where possible:

> *'resemble those of similar waterbodies with insignificant anthropogenic disturbance'* (CEU 1994).

Although this is a simple concept, in practice there are at least five different ways in which reference condition can be defined. These are:

(1) Comparisons with the best available present-day reference sites,
(2) Reconstruction of waterbody histories using paleolimnological techniques,
(3) Modelling approaches (including hindcasting),
(4) Historical data,
(5) Professional consensus.

In the United Kingdom, the first three methods of establishing reference conditions are well-known and biologists working in these areas have made important intellectual contributions to biomonitoring. These are: (i) RIVPACS; (ii) diatom-based lake history reconstruction; and (iii) chemical hindcasting (Johnes *et al.* 1994). In general, however, subsequent assessments against these baselines have been made using indices as indicators of physico-chemical conditions, rather than general biological indices of integrity.

3.4.2 Methods for establishing baseline reference sites

Present-day reference sites

The derivation of baselines using 'best available' present-day reference sites is the most commonly used approach in the United States, and in Britain has been the approach used for the development of RIVPACS (now being replicated in Europe) and the National Pond Survey. The principal advantage of this approach is that all aspects of physical, chemical and biological environment can, potentially, be measured or described. This is, therefore, the only method which can be used to describe biotic baselines for all still-water types (e.g. shallow ponds, temporary waters, canals, ditches and brackish lagoons). The approach assumes that, within the population of waterbodies, sufficient sites exist that are still minimally disturbed. Today few, if any, waterbodies remain unimpacted by anthropogenic stresses. Thus the greatest drawback to this method is that, in areas where all waterbodies are to some extent impaired, the baseline reference set may under-represent the extent of damage leading to misleading mediocre expectations for the area as a whole.

Paleolimnology

Paleolimnological methods use sub-fossil evidence preserved in waterbody sediments to reconstruct past, pre-impact communities and conditions. The advantage of this method is that any waterbody with an accurate sedimentary record can be a reference site, regardless of the severity of present-day pollution. Thus, a representative sample of waterbody reference sites can be established allowing direct comparison with present-day communities. The main disadvantage of paleolimnological referencing is that only a very limited fauna and flora is typically preserved and only diatoms are likely to be well represented. Even with diatoms, there may be reservations relating to the accuracy of the sub-fossil record in circum-neutral or alkaline waters, where differential dissolution of biogenic silica is likely (Reid *et al.* 1995). Community measures such as species richness, relative abundance and community structure have, therefore, to be used with caution. Other reservations with this method relate to the cost of coring and processing data, and more fundamentally, to the absence of an adequate sediment record in many waterbodies. This, in effect, limits the use of paleolimnological techniques to some lakes. All other waterbody types (ponds, ditches and canals) are typically too shallow (i.e. too well oxidised) or too frequently dredged to retain a suitable sediment record.

Modelling Approaches

Several modelling approaches can be used, including mathematical models (logical constructs following from first principles and assumptions), statistical models (built from observed relationships between variables) or a combination of the two. The degree of complexity of mathematical models used to predict reference conditions is potentially unlimited, with attendant increased costs and loss of predictive ability as complexity increases (Peters 1991). Several models to predict or hindcast *chemical* water quality in rivers, reservoirs and lakes have been quite successful (e.g. Kennedy and Walker 1990, Vighi and Chiaudani 1985; Vollenweider 1975), but they require a sufficiently large data base to test predictive relationships. This approach has recently been proposed as a basis for NRA/EA lake assessment by Johnes *et al.* (1994).

Table 3.4 Comparison of techniques for characterising reference conditions

	Present-day reference sites	Paleolimnology	Modelling	Historical data	Professional consensus
Strengths	• Applicable to all types of still water body. • All physical, chemical and biological characteristics can be measured.	• Provides historical time series data for diatom assemblages, chrysophytes, and, to a lesser extent, some crustaceans and some insects. • Water quality can be inferred from assemblage data.	• Can be used when no paleo-limnological or historical data are obtainable. • Works well for water quality.	• Gives actual historical information on status. • Inexpensive to obtain.	• Can be used when no data are obtainable. • Relatively inexpensive. • Can be better applied to biological assemblages than models. • Common sense and experience can be incorporated.
Weaknesses	• Even best sites subject to human impacts. • Inclusion of degraded sites can lower standard of reference sites.	• Restricted to sites with good sediment record • Preservation of fish, invertebrates, macrophytes, and non-diatom algae is poor. • Pre-settlement status might be unrealistic and unobtainable in a present-day context.	• Community and ecosystem models not useful. • Extrapolation beyond known data and relationships is risky. • Can be expensive. • Not testable	• Unlikely to be many sites with good data. • Data not usually collected for status monitoring so likely to be inappropriate. • Human impacts present in historical times were sometimes severe. • Difficulties in determining when the 'natural' state occurred.	• Qualitative descriptions of "ideal" communities. • Might be unrealistic and unobtainable. • Experts might have strong bias.

Models to predict *biological* conditions have been attempted in the United States and in Britain (Pond Action, work in progress with Zeneca Agrochemicals) but have not so far been used in an assessment or management context. Amongst the many difficulties of mathematical and statistical modelling approaches for creation of biotic reference levels, is that estimation of baseline conditions requires predictions from one model to be used as inputs for second and third models (e.g. using presettlement loadings to predict reference trophic state to predict reference biotic assemblages); the gross errors which arise from such a practice are untenable (EPA 1994). Just as problematic is the difficulty that the hypotheses produced are largely untestable (Osreskes *et al.* 1994; Peters 1991), since predictions for most components of the biota (macroinvertebrates, macrophytes, fish) cannot be confirmed by site observations.

Historical Data

Some waterbodies have extensive historical data bases from the early to mid-20th century, typically relating to fish and more rarely to macrophytes, diatoms and zooplankton. Applied carefully, this data may be used to augment present-day reference site data. More widespread use of this approach for establishing a baseline is prohibited by the patchiness of records and concerns relating to its representativeness. Historical data will not necessarily represent undisturbed conditions, for example. Similarly, sites for which historical data is available may be anomalous, having been selected for a specific purpose or reason (e.g. unique waterbodies, those near laboratories, water intake sites, etc.).

Professional consensus

When undisturbed reference sites are not available or appropriate in an area, informed consensus may be a workable alternative for establishing the expected reference conditions. Waterbodies for which such an approach might be appropriate include:

(i) Pumped storage reservoirs which have no natural analogues.

(ii) Waterbodies created or maintained for specific economic purposes (e.g. canals, gravel-pits dedicated to recreational use, farm irrigation ponds) where 'least impacted' biological targets may to be inappropriate.

Consensus under such circumstances might be used to delimit 'appropriate waterbody conditions' as the basis for assessment (e.g. a reference for canals based on 'normal' boat traffic) or could specify waterbodies where monitoring may not be appropriate (perhaps sewage lagoons).

Informed consensus can also be useful in supporting information and data interpretation derived from the other approaches.

3.4.3 Conclusion: development of baseline reference sites for still water monitoring

Of the approaches outlined above, the use of present-day reference sites for establishing baselines is, wherever possible preferred, because:

(i) this is the only method which is applicable to all waterbody types and all assemblages,

(ii) it is the only method which can be used to provide a full description of biological status for comparison with present-day sites.

Other approaches (e.g. hindcasting of chemical status, paleoecological techniques and historical data) are, however, also likely to have a role in aiding the selection of appropriate baseline sites, particularly where there is widespread impairment of existing waterbodies.

3.5 Classification

Still waters vary naturally in their physical, chemical and biological characteristics. The use of a single reference state which applies to all waterbodies (e.g. all lakes) is therefore misleading. Classification of baseline reference sites, and subsequent comparison of impaired sites within the framework of a classification is therefore an essential part of general ecosystem assessment.
The benefits of this approach are essentially two-fold: (i) by classifying sites, the *natural* variability of biological measures within classes is reduced, allowing comparison of 'like with like'; and (ii) the potential to identify significant impact gradients within each class is maximised.

Two main approaches have been used by freshwater biologists for waterbody classification. In the United States, classification invariably involves professional judgement to arrive at a workable system that separates clearly different ecosystems. In the UK, in contrast, there has been widespread use of multivariate statistical techniques for the classification of freshwater communities. Classifications have been developed for communities in lakes (Palmer *et al.* 1992), ponds (Pond Action 1994), and ditches (Alcock and Palmer 1985) as well as in river systems (Wright *et al.* 1984, Holmes 1983). The main advantage of such methods is that they provide a much more objective description of communities than the techniques used by North American biologists. Similar multivariate approaches have been extensively used in the Netherlands, but are rarely used elsewhere in Europe.

3.6 Conclusions: summary of the framework for GQA biological methods

This chapter has recommended an outline framework for biological assessment of still waters in England and Wales, and a theoretical rationale for the development of general ecosystem monitoring methods.

3.6.1 Conceptual framework

It is suggested that biological assessment of still waters should be considered as a two-stage process:

Stage 1. General ecosystem assessment

General ecosystem assessments aim to evaluate the net effect of all types of impact which degrade the integrity of freshwater systems. Such assessments should form the basis of surveillance monitoring programmes.

Stage 2. Diagnosis

Diagnosis assessments employ one or more of an array of appropriate techniques (biological, chemical, historical etc.). They may be applied as a follow-up investigation where damage is evident from general ecosystem assessments; or they may be necessary for specific compliance with EU Directives and other legislation. Diagnosis of the causes of damage at specific sites may be facilitated by re-use of data derived from general ecosystem assessment monitoring.

A third class of assessment technique, **early warning indicators**, is also recognised. These methods are as yet poorly developed but, where field-based, have the potential to be used along side general ecosystem assessments to aid prevention of ecosystem degradation.

3.6.2 A rationale for the development of general ecosystem assessment methods

1. The essential requirement of general ecosystem assessment in any waterbody is that it should represent and summarise the existing biological quality and integrity of that water body.
2. In practice, ecosystem integrity is measured in terms of biodiversity elements (e.g. genes, species) and attributes (e.g. taxa richness, trophic structure).
3. To assess *all* aspects of biological integrity is not an economically viable option. A narrower range of groups (taxa) or attributes (species-richness, rarity etc.) which, however, *still represents* the overall integrity of the system, therefore needs to be selected.

   A number of criteria (scientific, practical, economic) can be used to reduce the range of taxa and attributes measured, but the *validity* of the choice depends on the knowledge that the measures chosen *do* adequately reflect overall changes in the integrity of the system under stress.
4. An accurate measure of ecosystem integrity is most likely to be derived where a number of assemblages and a number of significant attributes are used for assessment providing an holistic, rather than reductionist, approach. This is for several reasons:
   (i) there is too little information about the relationships between taxonomic groups and the detailed effects of stressors to enable identification of specific taxa which are likely to be the best indicators of overall quality.
   (ii) analysing a *range* of groups/attributes overcomes the limitations and biases of any one taxonomic group or attribute in detecting the considerable number of stressors which may degrade ecosystem quality.
5. In order to assess their quality, sites need to be compared with a minimally impacted reference condition. In effect, this means 'least impacted' present-day sites. All quality assessments need to be undertaken within the framework of a classification, which minimises the confounding effects of natural variation and allows degradation gradients to be identified more easily.

4. ASSESSMENT METHODS USED IN BRITAIN, EUROPE AND THE UNITED STATES

4.1 Introduction

This chapter reviews techniques currently used for biological assessment of water quality, drawing on examples from Britain, continental Europe and North America. Although the review is primarily concerned with assessment of still waters, it remains true that water quality techniques are typically developed first for assessment of streams and rivers, and subsequently applied to standing waters. Methods currently used for running waters have therefore also been considered in this review where appropriate.

The aims of the review have been:

(i) to describe the contrasting *approaches* to water quality monitoring that have developed in Europe and North America and to compare these approaches with the EA monitoring framework developed in Chapter 3,

(ii) to describe *specific* methods of assessment and evaluate their applicability to EA requirements,

(iii) to assess the *potential* for each of the main biological assemblages (plants, birds, etc.) to provide a basis for water quality assessment.

The chapter concludes with a brief discussion of the applicability of existing approaches and methods to the development of still water assessment techniques in the UK.

4.2 The contrasting approaches to water quality monitoring in Europe and the United States

For the last 20 years European and North American practitioners have been following essentially different philosophies of general surveillance monitoring.

European methods have mainly been based on traditional single pollutant-based approaches which use biology essentially as a sophisticated proxy for monitoring water chemistry parameters. This approach has focused on the development of indices which addressed a specific water quality pollutant - the Saprobic system for organics, Trophic Scores for nutrients etc. Measures such as species-richness and rarity have often been seen as aspects of 'conservation value' which are either irrelevant or the responsibility of other, specifically conservation based agencies.

Until the mid-1970's the North American approach to water quality monitoring ran along similar lines to that in Europe (Davis 1995). But at the end of the decade, North American methods began to take a new course, as the approach to water quality monitoring gradually shifted from a focus on specific pollutants, to a much broader view of aquatic ecosystem protection.

The initial direction for this new approach was provided by a powerful and influential organisation, the US Environmental Protection Agency (US EPA). However its work was backed by important legislation in the form of the Clean Water Act (1976), which first embodied the principle of maintaining the biotic integrity of ecosystems.

Following the impetus of legislation, the scientific and technical principles of 'biological integrity' were rapidly developed and in 1981 the first Index of Biotic Integrity was proposed (Karr 1981). Over the subsequent 15 years, US biologists and managers have developed, tested and applied biotic integrity assessment methods as primary basis of freshwater monitoring programmes.

The concept of biotic integrity monitoring, now routinely used in the United States, is very close to the theoretical framework proposed for use by EA in England and Wales (see Chapter 3). The way in which the US concept has been realised in practice is therefore of considerable relevance. In the section below, US methodologies are described in more detail. Examples of current European methodologies, which are increasingly developing along similar lines are discussed in Section 4.4.

4.3 Multimetric monitoring: the US approach to GQA

In 1981, Jim Karr first introduced the concept of metrics and multimetric monitoring to the US. The basis behind this approach is that, since ecological systems are complex, adequate monitoring of an ecosystem must be achieved on the basis of a range of measures.

He also coined the term 'metric' to define 'a calculated term or numeration representing some aspect of biological assemblage, structure, function or other measurable characteristic that changes in some predictable way with increased human influence' (Karr 1995). In effect, therefore, metrics are predominantly attributes, such as those listed in Table 3.3, which are particularly associated with degradation and therefore provide a good measure of the effects of anthropogenic impacts.

The conceptual leap that Karr made in the early 1980's was the development of a *multimetric* integrity index which combines the use of many metrics, and therefore the effect of many stresses, as part of a single assessment. Metrics included in an integrity index can (and should) include attributes which span a wide range of structural and functional features; not just species or taxonomic richness but aspects of trophic organisation and species health.

For index calculation each metric is scored according to the extent to which it deviates from an undisturbed baseline condition. Metrics are then divided into simple 'rating' categories on a four or five point scale (e.g.: 5 = a site only slightly different from the reference condition; 3 = a moderate degree of difference from the baseline condition; 1 = a strong deviation from the baseline condition). To calculate the index value, these metrics are simply summed to give a single score which is used to represent the integrity of the community as a whole.

In addition to their ecological validity, multimetric methods have a number of advantages over more traditional assessment indices:

- multimetric indices are very flexible: any environmental feature from a given site can be included in an index, provided that it is relevant and a technique for establishing a baseline condition is available.
- new metrics can be added at any stage without undermining the entire concept.
- in contrast to many biotic indices, multimetric indices have statistical properties which allow use of standard statistical techniques, such as ANOVA, for hypothesis testing.

Karr originally developed his concept of an Index of Biotic Integrity (IBI) using stream fish communities and included a set of 12 metrics to summarise integrity (see Table 4.1) (Karr 1981). Subsequent workers have developed IBI's to describe all the main biotic assemblages (phytoplankton, sediment diatoms, macrophytes, invertebrates, fish and higher vertebrates) in both separate and combined indices. Statistical investigation of the properties of the IBI have widely tested, and proven to have sufficient statistical power to distinguish between five or six non-overlapping categories of biotic integrity (Fore *et al.* 1994).

Modified IBIs have since been developed in many regions of North America (Invertebrate Community Index; DeShon 1995, Biological Condition Score; Plafkin *et al.* 1989, and Mean Biometric Score; Shackleford 1988), and although initially applied to river ecosystems, they are increasingly being used for still waters, including a major multimetric programme currently proposed by the US EPA as the basis for a lake and reservoir bioassessment programme (EPA 1994).

Table 4.1 Metrics for US river fish communities used in the original Index of Biotic Integrity (Karr 1981)

Species composition and richness
- Number of species
- Presence of intolerant species
- Species richness and composition of darters [no. of species reduced with impacts]
- Species richness and composition of suckers [no. of species reduced with impacts]
- Species richness and composition of sunfish (except green sunfish)
- Proportion of green sunfish [a tolerant species]
- Proportion of hybrid individuals

Community structure
- Number of individuals in sample
- Proportion of omnivores
- Proportion of insectivorous cyprinids
- Proportion of top carnivores
- Proportion with disease, tumours, fin damage, and other anomalies

4.4 European General Quality Assessment methods

4.4.1 Introduction

As described above, the European ethos of water quality assessment through most of this century has remained focused on developing more sophisticated ways of monitoring specific pollutants. Thus improvements to the Saprobic system - the basis of practically all river based pollution monitoring, have involved adding more taxa to measure oxygen tolerance rather than expanding the system to consider a wider range of stressors. Other impacts such as eutrophication and habitat damage have been addressed (e.g. Trophic Ranking Score for nutrients, River Corridor Surveys for habitat quality) but these have remained independent assessments.

The last five years has begun to see a shift in emphasis in many areas however. For example, when applying the BMWP system, biologists in EA have made increasing use of taxon richness (which is a more holistic metric linked to most sources of degradation) as well as using BMWP and ASPT indices to track organic pollution. Although no method directly equivalent to the American IBI system has yet been proposed, an increasing number of holistic approaches to ecosystem monitoring are being developed. These inevitably have some aspects in common with multimetric IBIs.

Four different European approaches are discussed briefly below: the AMOEBA model, developed and used in the Netherlands (the closest of the European methods to IBIs), the Riparian, Channel and Environmental Inventory (Petersen 1992) and SERCON, the System for Evaluating Rivers for Conservation (Boon, pers. comm.). The lake classification system recently proposed by Johnes *et al.* (1994) also has many of the features of multimetric indices although, as a predominantly chemical assessment, it includes little biological data.

4.4.2 The AMOEBA model.

The AMOEBA model (AMOEBA is the Dutch acronym for "a general method of ecosystem description and assessment") was developed in the 1990's (ten Brink *et al.* 1991, ten Brink and Woudstra 1991). Conceptually the AMOEBA approach is similar to the multimetric IBIs developed in the United States in that it considers a number of ecosystem components simultaneously, rather than focusing on a single environmental stressor, such as organic pollution or eutrophication. The AMOEBA model is based on the concept that the closer an ecosystem comes to a reference condition which is 'not, or only slightly, influenced by human activity' (ten Brink *et al.* 1991) the more that three desirable ecosystem attributes will be maintained: (i) ecosystem yield and production (ii) species richness and (iii) ecosystem self-regulation.

Within the AMOEBA system up to 150 target variables can be considered relating to the physico-chemical environment (e.g. radioactivity, water depth, phosphate concentration), species or taxa (e.g. algae, macrophytes, otters), health of biotic assemblages (e.g. fish diseases and growth rates) and uses of the environment (e.g. water supply, groundwater abstraction). However, biological targets tend to be stated mainly in terms of species, rather than index values of community attributes which structure metrics. The model has been applied to both marine and freshwater systems and at a variety of scales. A distinctive (though not critical) feature of the AMOEBA system is its use of rose diagrams as a presentational device.

Under the AMOEBA system the following criteria are used to select target species:

- quantitative data on the species must be available,
- the species must be susceptible to human influence,
- the species must be accessible to easy and accurate measurement,
- the species should have some indicative value of the condition of the system,
- the species should, ideally, have some political and social appeal,
- species should be selected from all types of water-subsystems,
- species should be chosen from the benthos, water column, water surface and shores,
- species from high and low in the food web should be chosen, including plants and animals,
- locally extinct species may be included,
- sessile, migratory and non-migratory species should be included.

The principle drawback of the AMOEBA model is that target measures and variables are chosen subjectively, on the basis of popular or economic interest and 'general scientific agreement'. Reference conditions for each target are also subjective: it is recommended that they are based on a mixture of old inventories, ecological theory and comparative research (ten Brink *et al.*, 1991). The reference baseline can, therefore, vary between individual target species or families. Whilst IBIs may also consider economic interests, only those metrics which show a relationship with environmental degradation are included in the index.

4.4.3 Lake classification.

The NRA/EA proposed Lake Classification method developed by Johnes *et al.* (1994) is firmly based in multimetric principles and has many aspects in common with the US method including: (i) comparison of data with relatively unimpacted reference sites (ii) assessment on the basis of a number of normalised variables (metrics) linked to degradation (iii) combination of these metrics to give a single index. The main difference between the two systems is that the method proposed by Johnes *et al.* (1994) is, at present, predominantly based on physico-chemical data and cannot, therefore, provide an adequate measure of the ecological integrity of waterbodies (see also Chapter 5).

4.4.4 Riparian, Channel and Environmental Inventory (RCE).

The RCE proposed by Petersen (1992) is concerned with assessment of small streams but is notable for the inclusion of a wide range of metrics. It is also of interest as one of the few European applications of multimetric IBIs that is directly related to North American techniques, drawing heavily on the US Forest Service Stream reach inventory and channel stability evaluation. It has not, to our knowledge, been applied in the United Kingdom. The variables considered in the RCE are listed in Table 4.2.

Table 4.2 Variables considered in the Riparian, Channel and Environmental Inventory (RCE).

1. Fish
2. Macrobenthos
3. Aquatic vegetation
4. Width of riparian zone from stream edge to field
5. Completeness of riparian zone
6. Vegetation of riparian zone within 10m of channel
7. Land-use pattern beyond the immediate riparian zone
8. Detritus
9. Retention devices
10. Channel structure
11. Channel sediments
12 Stream-bank structure
13. Bank undercutting
14. Stony substrate; feel and appearance
15. Stream bottom
16. Riffles and pools, or meanders

Variables are scored on a four point scale from 1 to 30, from most degraded to least degraded, although Petersen does not specify the reference conditions against which these values are assessed.

4.4.5 SERCON (System for Evaluating Rivers for Conservation).

SERCON has been developed to provide a system for evaluating the conservation value of river systems (Boon *et al.* 1994) and is currently being developed and tested. SERCON has many similarities with multimetric IBIs, including the assessment of a wide variety of variables (see Table 4.3) and comparison relative to a reference baseline.

At present, however, reference conditions are mainly defined using professional judgement, rather than using more objective techniques. Only macroinvertebrate measures (which can be assessed in the context of the RIVPACS system) are related to an objectively established baseline. In addition, the measures proposed as indicators have not been statistically investigated to prove their relationship to environmental degradation.

Table 4.3 Conservation value measures used in SERCON

Physical diversity
- Number of natural substrate types present (9 categories)
- Number of natural fluvial features present (23 categories)

* Number of plant forms present (9 categories)

Naturalness
- %Channel artificial or realigned
- %Channel profile artificial
- Naturalness of flow regime (scale 0-5)
- Number of artificial structure/10 km
- %Bank affected by engineering or developments
- %Bank affected by engineering or developments
- %Bank with natural vegetation
- %Riparian zone with natural vegetation
- %Native aquatic and marginal macrophyte species
- Number of alien aquatic invertebrate species
- %Native fish species

  %Native breeding bird species

Representativeness
- %Artifical substrate
- %Artifical fluvial features
- Aquatic plant community: similarity to expected NCC community type
- Aquatic invertebrates: BMWP EQI from RIVPACS
- Number of fish species found of those expected
- Number of breeding bird species recorded of those expected

Rarity
- Number of EC Habitat Directive, Bern Convention, and W&C Act species found
- Number of Red Data Book/Nationally Scarce macrophyte species
- Number of Red Data Book/Nationally Scarce invertebrate species
- Number of Regionally rare macrophyte species

Species richness
- Number of aquatic and marginal macrophytes species
- Number of native aquatic invertebrate species/families
- Number of native fish species
- Number of native breeding bird species birds

Special features
- Influence of natural on-line lakes (scale 0-5)
- %Riparian zone grater than 5m wide
- Floodplain: recreatable water-dependent habitats
- Floodplain: unrecreatable water-dependent habitats
- Invertebrates of river margins and banks
- Amphibians
- Wintering birds on floodplain
- Mammals

Impacts
- Acidification
- Toxic/industrial/agricultural effluent
- Sewage effluent
- Groundwater abstraction
- Surface water abstraction
- Inter-river transfers
- Channelisation
- Management for flood defence
- Man-made structure
- Recreational pressures
- Introduced species

4.4.6 Comparison of US and European holistic approaches

Shortcomings of the European holistic approaches

The difference between most of the European holistic approaches and those developed in the US are fourfold:

1. US metrics are typically calculated, and arise from, analysis of ecosystem data, using information from both unimpacted and damaged sites. Potential metrics are validated statistically derived, following testing and calibration against known damage gradients. Efforts are made to ensure metrics are ecologically valid and to avoid redundancy between metrics. In contrast, European 'metrics' have tended to be picked 'ad hoc' on a subjective basis (typically in terms of public interest or scientific consensus).
2. In the few holistic European methods, the baseline (with which results are compared) is not fixed, and usually reflects a best guess.
3. Assessment is not generally been made within the confines of a classification, there is, therefore, no evidence that comparisons are really assessing damage - rather than natural differences between the site and the 'ideal'.
4. There is no concept of combining the results into a single index to give an easily comparable numerical measure that could, for example, be used for GQA.

The recently developed lake classification (Johnes *et al.*. 1994) is the only holistic method to avoid these pitfalls and, indeed the method shares many of the characteristics of multimetric IBIs. Its main drawback is, as noted above, the paucity of biological attributes which are incorporated, which limit its ecological validity as a general ecological monitoring technique for freshwaters.

Shortcomings of the American approach

From a European perspective, a shortcoming of the multimetric approach is that the more advanced techniques for defining and classifying biotic baseline conditions (e.g. the RIVPACS methodology and the lake diatom water chemistry reconstructions) are not routinely used in the United States. In effect, US practitioners guestimate which waterbodies should form the baseline conditions for any waterbody type. An attempt to minimise variation is made by only comparing sites within natural regions. However, this is not ideal; there is no proof that sites with similar environmental gradients are being compared.

In essence, the main difference between the two approaches is that North American biologists have paid more attention to the creation of ecologically realistic indices, whereas European biologists have spent more effort on developing methods for establishing baseline conditions, paying less attention to the wider application of these referencing techniques.

4.5 <u>Detailed descriptions of methods - an outline of the approach taken</u>

The remainder of this chapter provides an overview of the range of biological methods that have been developed for water quality assessment in Europe and the United States (see Table 4.4).

4.5.1 The approach adopted in the following sections of this chapter

Biological assessment methods can, essentially, be grouped in one of two ways:

(i) according to the type of method (e.g. biotic index, diversity index, ecotoxicological method),

(ii) according to the taxonomic group being surveyed (phytoplankton, macroinvertebrates, fish).

In the following sections, the approach adopted has been to group methods taxonomically.

This is because: (i) many methods are 'mixed', combining elements of different indexing methods, and cannot be simply categorised according to type (ii) grouping methods taxonomically facilitates comparison of the attributes of each assemblage group, a process that is important for later sections of the report.

Methods have, therefore, been summarised under the following headings:

- Phytoplankton
- Periphyton
- Aquatic macrophytes (vascular plants, mosses, liverworts)
- Microinvertebrates (zooplankton etc.)
- Macroinvertebrates
- Fish
- Amphibians
- Birds
- Mammals
- Habitat-based methods (e.g. River Corridor Survey)
- Rapid screening tests (e.g. enzyme-based methods)

4.5.2 Section content

1. Existing methods, which are currently in use or in development, are reviewed under the following headings:
   - *Community -based assessments indicating a single environmental stress* (e.g. macrophyte Trophic Ranking Scores for eutrophication; diatom reconstruction of lake pH; macroinvertebrate species richness). These methods may be used in a variety of ways. For example they can be (i) part of a multimetric index (e.g. species richness in the IBI) (ii) used to diagnose the reasons for damage (iii) used as an early warning system.
   - *Multimetric methods* describing general ecological quality (and integrating many different stresses). For each taxonomic group the current level of development of multimetric methods is described (e.g. IBIs or European equivalents). Multimetric methods often use very similar sampling methods to traditional 'single-issue' methods, differing mainly in their conceptual design and analytical approach.
   - *Bioassays* in which the responses individual species or taxa are used as a measure of ecosystem damage. These methods are used: (i) to diagnose the reason for change (e.g. fish health examinations, laboratory toxicological tests) or to provide an early warning system (biomarkers, fish biomonitors).
2. An assessment of the *potential* of each assemblage (realised or not) as the basis for general surveillance assessments, problem diagnosis techniques or early warning systems.

Table 4.4 Biological assessment methods reviewed in this report

Multimetric	
• Proposed US EPA 'multimetric' lake bioassessment methods	EPA 1994
• Index of Biotic Integrity	Karr 1981
• Invertebrate Community Index	DeShon 1995
• Biological Condition Score	Plafkin *et al.*. 1989
• Mean Biometric Score	Shackleford 1988
• Zooplankton biomonitoring program	Marmorek and Korman 1993.
• AMOEBA model	ten Brink *et al.* 1991
• SERCON (System for Evaluating Rivers for Conservation)	Boon *et al.* 1994
• River, Channel and Environmental Inventory (RCE)	Petersen 1992
• EA proposed lake classification	Johnes *et al.* 1994
Phytoplankton	
• Index of Chlorophyceae	Thunmark 1945
• Algal Quotients	Nygaard 1953
• Dominant Limnetic Algae	Rawson 1956
• Rating of organic pollution tolerant algae (Palmer's Pollution Index)	Palmer 1969
• EPA Lake Bioassessment Programme: phytoplankton metrics including species richness, percent contribution of dominant taxa, Shannon-Weiner Diversity	EPA 1994
• Indicator taxa: e.g. counts of blue-green algal genera; estimation of the abundance of filamentous algae	Many studies
• Ratios of algal divisions (e.g. blue-green algae: total) or other functional groupings (e.g. motile cells: total)	Many studies
• Pollution Tolerance Index, based on tolerance groups of Lange-Bertalot	Bahls 1993
• HPLC (High Pressure Liquid Chromatography) to determine the percentage contribution of five major algal groups	Wilhelm *et al.* 1995
• Diatom indicator species values for pH, salinity, nutrients, oxygen/saprobity, metals and moisture	Husted 1937-39, Battarbee and Charles 1987, van Damm *et al.* 1991, 1994, Dixit and Smol 1994, Reid *et al.* 1995.
• Diatom-based transfer functions for pH and phosphate	e.g. Reid *et al.* 1995; Bennion 1994
• Algal biomass (e.g. chlorophyll a concentration)	Many studies
• Carlson's Trophic State Index	Carlson 1977
• US Trophic State Index	Premazzi and Chiaudani 1992
• Algal Assay Procedure, based on the green alga *Selenastrum capricornutum*	Trainor 1984
• Algal Assay Procedure, based on the green alga *Selenastrum capricornutum*	Trainor 1984
• Algal Fluorescence Techniques	Munawar *et al.*, 1991
• Bioassays using adenylate energy charge (AEC), and ATP (adenosine triphosphate) concentrations with *S. capricornutum*	Johnson 1995

Table 4.4 Biological assessment methods reviewed in this report (continued)

Periphyton	
Indice Daitomique (Id)	Descy 1979
Indice de Polluosensibilité (SPI)	Coste *et al.* 1991
Organic Pollution Index	Steinberg and Schiefele 1988
Trophic Diatom Index (TDI)	Kelly *et al.* 1996
Diatom indicator value	van Dam *et al.* 1994
Macrophytes	
Trophic Ranking Score	Palmer *et al.* 1992
Mean Trophic Rank	Holmes 1995
Macrophyte Index Scheme	Caffrey 1987
Bioassay for heavy metals	Whitton *et al.* 1991
Bioassay for chlorinated hydrocarbons	Mouvet *et al.* 1985a
Bioassays for copper and cadmium	Mouvet 1984
Macrophyte multimetric assessment techniques	EPA 1995
Species richness and Species Rarity Index	Pond Action 1994a
Damage Rating value	Haslam 1990
UK National Vegetation Classification	Rodwell 1991, 1995
Microinvertebrates	
Saprobic system	Sládecek 1973
Zooplankton acidification monitoring	Marmorek and Korman 1993
Zooplankton multimetric assessment techniques	Fore and Karr 1994
Zooplankton multimetric assessment techniques	EPA 1995
Macroinvertebrates	
Environmental Index	
Benthic Quality Index	Wiederholm 1980
Acidification Index	Fjellheim and Raddum 1990
Species richness and Species Rarity Index	Pond Action 1994a, 1995
• Lake trophic status: oligochaete proportion and abundance of indicator species	Howmiller and Scott 1977; Lang and Lang-Dobler 1980; Wiederholm 1980
• Lake trophic status: chironomid indicator species and relative abundance	Saether 1979
• Chironomid Pupal Exuvial Technique	Wilson 1992
• Lake trophic status: ratio of chironomids to oligochaetes	Wiederholm 1980
• Lake trophic status: proportion and abundance of Mollusca as gastropods and bivalves	Mouthon 1993
• Lake trophic status: Corixidae as indicator species	Savage 1995
• Family Biotic Index	Hilsenhoff 1982
• Laboratory bioassays: behavioural impairment, enzyme activity, ion regulation, energy metabolism, respiratory metabolism	Various authors
• Chironomid head and mouthpart deformities	Johnson *et al.* 1983
• Behavioural impairment: net spinnng caddis etc.	Kitching *et al.* 1987
• 62 further macroinvertebrate indices are listed in Table 4.12.	Various authors

Table 4.4 Biological assessment methods reviewed in this report (continued)

Fish	
Fish biomass, length-weight relationships, growth rates	Many studies
Index of Biotic Integrity	Karr 1981
French Index of Biotic Integrity	Oberdorff and Hughes 1992
Index of gill damage	Poleksic and Mitrovic-Tutundzic 1994
WRc Fish Monitor	Seager *et al.* 1994
Fish biomarkers (experimental)	Many studies
Amphibians	
Species richness and species rarity	Swan and Oldham 1993
Birds	
National Wildfowl Counts	Owen *et al.* 1986
Birds of Waterways Survey	Bibby *et al.* 1993
Population studies of individual species	Many studies
Mammals	
Otter survey	NRA 1994
Physical habitat assessment methods	
US EMAP Lake Bioassessment Criteria	EPA 1995
Qualitative Habitat Evaluation Index	Rankin 1995
River Corridor Survey	NRA 1992
River Habitat Survey	NRA 1995
River, Channel and Environmental Inventory	Petersen 1992
Rapid screening tests	
Microtox	Johnson 1995
ECLOX	Johnson 1995
Daphnia magna in vivo enzyme inhibition	Johnson 1995
Algal/macrophyte fluoresence inhibition	Johnson 1995

4.6 Phytoplankton

4.6.1 The importance of phytoplankton

Phytoplankton are the free-floating component of the algal flora, drawn primarily from the following groups (terminology follows that used by Lund and Lund 1995): Cryptophyta (cryptophytes), Pyrrophyta (dinoflagellates), Chlorophyta (green algae), Euglenophyta (euglenoids), Bacillariophyta (diatoms), Chrysophyta and Haptophyta (yellow-green algae) and the Cyanophyta (blue-green algae). Phytoplankton are believed to represent a large proportion of the species diversity of many freshwater communities. The diversity of phytoplankton communities has been most thoroughly documented in large water bodies, where planktonic algal species richness is reported to be similar to the richness of littoral macroinvertebrates. Phytoplankton play a central role in the structure and functioning of many freshwater ecosystems. In many waters they contribute a large proportion of primary production and, with changes in the structure of ecosystems, may exert a profound influence on other ecosystem components (e.g. zooplankton, macrophytes, macroinvertebrates). Although individual algal species are rarely 'socially relevant' (sensu Cairns *et al.* 1993) toxic blooms of blue-green algae (for example, members of the genus *Microcystis* and *Aphanizomenon*) can create considerable nuisance problems in lake and more rarely canals and ponds.

4.6.2 Existing assessment methods using phytoplankton

Existing phytoplankton-based methods can be resolved into two main groups: those in which the taxonomic composition of the *community* is investigated and algal bioassays which record *biomass and primary production*. The greatest variety of algal methods are based on changes in taxonomic composition.

Community-based assessments indicating a single environmental stress

Algal community-based methods can themselves be divided into two broad categories: (i) sampling the contemporary community from the water column and (ii) sampling sedimented communities (particularly diatoms) derived from benthic samples or cores.

Contemporary phytoplankton community data has mainly been used to provide information on nutrient status. Early indices of trophic status included the Index of Chlorophyceae for Swedish lakes (Thunmark 1945), and Algal Quotients, developed for Danish ponds and lakes (Nygaard 1953). In these methods the proportion of species in different algal taxa was used to calculate a simple index of trophic status. Rawson (1956) proposed the Dominant Limnetic Algae method, developed in Canada, which considered only the dominant species as indicators of trophic status.

More recently a wider variety of analytical techniques have been tested for use with phytoplankton count data, although these still deal principally with trophic state (or eutrophication). Methods include:

(i) Measures of richness and diversity: e.g. taxa richness, percent contribution of dominant taxon and Shannon-Wiener Index, all of which have been proposed as metrics by the US EPA (1994) (see Table 4.4).

(ii) Indicator taxa: e.g. counts of blue-green algal genera; estimation of the abundance of filamentous algae, rating of organic pollution tolerant algae (Palmer 1969).

(iii) Indices and ratios including:

- Pollution Tolerance Index, based on tolerance groups of Lange-Bertalot (e.g. Bahls 1993),
- similarity indices, comparing the similarity in community composition to reference conditions (using simple similarity indices, such as Jaccards Index and multivariate techniques, such as CANOCO),

- Ratios of algal divisions (e.g. blue-green algae:total) or other functional groupings (e.g. motile cells:total).

For practical reasons few studies consider the less abundant algal species. It has been estimated, for example, that some 8000 algal cells need to be counted in a standard sample to obtain information on rarer species (Patrick 1951).

In recognition of the considerable time required to identify algal taxa, very recent techniques for assessing nutrient status have attempted to automate community assessments. HPLC (High Pressure Liquid Chromatography) developed recently in Germany (Wilhelm *et al.*, 1995), for example, uses algal taxon-specific xanthophyll pigments to determine the percentage contribution of five major algal groups (cyanobacteria, the green algae, the cryptoflagellates, the dinoflagellates and the diatoms) to total algal biomass.

One component of the phytoplankton community, the diatoms, has been more widely investigated in terms of the environmental tolerances of individual species. The wide range of studies undertaken have enabled 'indicator values' to be assigned to many species, in response to environmental variables/stresses. These include:

- pH (Husted 1937-39; van Damm *et al.* 1994, Nygaard 1956; Battarbee and Charles 1987),
- salinity (Kolbe 1927, van Damm *et al.* 1994),
- nutrients and trophic state: phosphorous, nitrogen, silica (van Damm *et al.* 1994),
- oxygen/saprobity (van Damm *et al.* 1994, Reid *et al.* 1995),
- metals
- moisture (Van Dam *et al.* 1991, 1994; Dixit and Smol, 1994).

In practice, the diatom component of phytoplankton assemblages derived from water column samples has rarely been used to assess conditions other than tropic status (and to a lesser extent acidification). This is probably a reflection of the traditional use of phytoplankton data, rather than an inherent lack of value. There may, therefore, be a greater potential for use of phytoplankton methods in lakes and other still waters.

Multimetric methods describing general ecosystem quality

There are currently no tested community-based multimetric approaches to still water assessment using phytoplankton. However, the US EPA have proposed metrics for lake and reservoir assessment based on aspects of trophic state, taxa richness, percent dominance and similarity indices. Three tiers of survey are proposed: screening, standard survey and diagnostic. All metrics are designed to be assessed from community information derived from mid lake water column samples identified to 'lowest practicable level' (usually order level counts). A list of the proposed metrics is given in Table 4.5 below.

Sediment record

Although sediment-derived communities can be used to describe recent phytoplankton assemblages, this is normally only done in conjunction with lake history reconstruction studies (Reid, 1995). In this context they provide a useful tool for obtaining site or region specific reference conditions for lakes (Smol 1992), although not all lakes are suitable for diatom reconstruction studies.

The use of sedimented diatoms is best documented for tracking pH trends (e.g. United Kingdom Surface Water Acidification Project (SWAP), (Mannion 1989) and trophic status trends (Dixit *et al.* 1992; Bennion 1994).

Table 4.5 Examples of US EPA lake phytoplankton metrics

Tier 1 metrics (screening)	Trophic State Abundance (cells/cm^{-3}) Relative abundance (%) of: • cyanobacteria • greens • diatoms • chrysophytes
Tier 2 metrics (standard)	Relative abundance (%) of: • *Anabaena, Aphanizomenon, Microcystis* • Volvocales (flagellated green • Centric diatoms (of total diatoms) • Pennate diatoms (of total diatoms) • Colonial greens • Euglenophyta • Dinoflagellates
Tier 3 metrics (experimental and diagnostic)	Taxa richness Diversity % dominance Lange Berthalot index (Pollution Tolerance Index [Bahls 1993]) Indicator taxa (presence or %)

Analytical assessment methods include:
1. use of specific diatom indicator species,
2. calculation of ratios using selected taxonomic groups,
3. use of simple or multiple regression techniques.

These techniques are used to derive transfer functions, which enable the sedimentary diatom assemblage to be used to predict historic water chemistry. In the United States, the EMAP (Environmental Monitoring and Assessment Programme) survey aims to use sedimentary diatom assemblages to evaluate the extent and rate of environmental and biological change (biotic integrity) in lakes (Hughes *et al.*, 1992), based on diatom assemblage metrics.

Bioassay approaches

Virtually all EU countries include some element of algal biomass assessment to estimate trophic state in their lake monitoring programmes. Although measurement of algal biomass is traditionally regarded as a 'community technique', its use as an indicator of eutrophication is in many ways, more like other bioassays. By far the most common measure used of algal biomass is chlorophyll a. The main advantage of chlorophyll-based techniques over other assessments of primary productivity (e.g. dry or wet weight, biovolume, particulate carbon, or Secchi transparency) is that it estimates photosynthetically active phytoplankton. It therefore distinguishes between physiologically active and disintegrating cells. Chlorophyll a levels are routinely measured using fluorometric or spectroscopic techniques. Concentrations may be used directly to assess overall trophic status of a waterbody, or converted to index values (e.g. Carlson's Trophic State Index, US Trophic State Index) (Premazzi and Chiaudani 1992).

A small number of more general bioassays using phytoplanktonic algae have also been developed. The Algal Assay Procedure, based on the green alga *Selenastrum capricornutum*, is used as a standard organism in laboratory-based bioassays world-wide. Its use is largely as a cultured inoculant monitored in terms of its growth response to an extensive range of toxic pollutants (Trainor 1984).

A more recent bioassay involves use of Algal Fluorescence Techniques (Munawar *et al.*, 1991). These measure toxin inhibition of photosynthesis and chlorophyll fluorescence and allow a visual examination of the impact of contaminants on individual cells/organisms under the microscope. The data can be used for rapidly screening of large numbers of environmental samples. Effects of toxicants on adenylate energy charge (AEC), and ATP (adenosine triphosphate) concentrations have also been proposed as bioassay techniques for use with *S. capricornutum* (Johnson 1995).

4.6.3 Potential suitability of phytoplankton for biological assessment

Phytoplankton are abundant, widespread and occur in all regions and in all water body types. This gives them considerable advantages in water quality assessments. In practice, however the communities of most standing waters (permanent and temporary ponds, canals and ditches), and even lake margins, are poorly known. In effect therefore, community-based assessments developed for almost all types of standing waters would be experimental

Although algae, and particularly diatoms, are known to be responsive to a wide range of impacts, algae are characterised by rapid reproduction rates and very short life cycles (hours to days; Reynolds 1984). This makes them valuable indicators of short term impacts, but poor integrators of long term conditions. In tandem with this high turnover of individuals is a pattern of rapid changes in community composition: algal community sucessional cycles are, in addition, only general, and their exact timing a composition are not predictable (Reynolds 1984).

As an assessment tool, this high temporal variability predicates against the use of phytoplankton for general ecosystem monitoring, and limits their use for impact diagnosis. Control of variability is possible through repeat sampling (typically weekly or monthly), with a minimum of 8-10 samples necessary to obtain either an annual average or a seasonal average (e.g. growing season, spring overturn, peak biomass) (Knowlton and Jones 1989a, EPA 1994, Johnes *et al.* 1994). Such assessments considerably increase sampling costs however.

Although temporal variability is a disadvantage for general ecosystem monitoring, their rapid reproduction rates and short life cycles make algae highly responsive to short term changes. This characteristic clearly gives them advantages for laboratory assessments of water quality and (potentially) as an early warning system for assessment of waters, such as effluents, which suffer fluctuating quality.

Although phytoplankton communities are often viewed as spatially homogeneous, they can be patchily distributed even in the open water of lakes (Brierley, in prep). There is little information available about the spatial distribution of algae in other water bodies (e.g. canals, ponds), or in lake littoral zones, but it seems unlikely that communities will be any less homogeneous than other assemblages e.g. periphyton or zooplankton.

4.6.4 Evaluation of the operational feasibility of phytoplankton-based techniques

Phytoplankton assemblages occur in all types of water body, are species-rich and easily sampled. However, *surveillance and compliance monitoring* methods are currently only available for lakes and would be experimental in all other waterbody types. In addition, the spatial and temporal variability in phytoplankton communities means that frequent sampling is needed if reasonable estimates of population variables are to be obtained (e.g. chlorophyll a, dominant species).

A well developed set of methods using phytoplankton is available for *diagnosis* of eutrophication and, to a lesser extent, acidification. The use of phytoplankton (particularly planktonic diatoms) to diagnose other impacts (e.g. heavy metals, biocides) is currently experimental At present, there are no practical methods which use phytoplankton to provide an early warning of impacts.

Table 4.6 Advantages and disadvantages of phytoplankton assemblages for bioassessment

	Advantages		Disadvantages
1.	Phytoplankton are widespread, occurring in all types of standing water.	1.	Phytoplankton have relatively short life cycles and can only indicate short term trends in water quality.
2.	The phytoplankton is a species-rich assemblage and is a significant proportion of the biodiversity of standing waters.	2.	Seasonal variation may complicate interpretations or comparisons.
3.	Many species (especially diatoms) are good indicators of a range of environmental stresses.	3.	The taxonomy of most groups is only moderately well known and genus/species level identification is highly skilled and time-consuming.
4.	Qualitative sampling techniques are well developed and can be done quickly using inexpensive equipment.	4.	Metrics are not well-developed or tested in most still waters.
5.	Some communities (e.g. open water) may be *relatively* homogeneous.	5.	Phytoplankton are mostly unknown to ordinary members of the public.
6.	Within water bodies, phytoplankton may be collected from all open water zones.	6.	There may be considerable spatial and temporal variability in phytoplankton communities, making it necessary to undertake regular sampling or integrate over years or areas.
7.	Phytoplankton have trophic links to fish and birds, and contribute to algal blooms, and therefore may of interest to many members of the public.	7.	Diatoms respond to a complex of variables that are highly interrelated; thus their separate effects cannot be identified easily (Yang and Dickman 1993).
8.	Many phytoplankton taxa are cosmopolitan, enabling data from one region to be extrapolated to others.	8.	Different authors record different autecological preferences for some algal species. For example, *Cyclotella glomerata* is referred to as an indicator of oligotrophic water by Stockner (1971) and as a eutrophic indicator by Brugam (1979).
9.	Phytoplankton survey methods vary from the inexpensive routine (e.g. chlorophyll a measurement) to the detailed investigative (e.g. diatom coring).		

4.7 Periphyton

The periphyton community comprises a diverse assemblage of bacteria, fungi, algae and protozoa growing attached to plants, rocks and other firm substrates. However, in practice most periphyton research data relates only to attached diatoms, so that for the purposes of biomonitoring 'periphyton' and 'diatoms' are largely synonymous. Both diatoms and other components of the periphyton occur widely in still waters including temporary and brackish sites (van Dam *et al.*, 1994). The exact number of species in the periphyton assemblage in Britain is not known since the group is relatively little surveyed and many species are taxonomically difficult. However, Frank Round (Round 1964) has noted that probably over nine tenths of all algal species grow in benthic habitats.

4.7.1 Existing assessment methods using periphyton

Periphyton assemblages are not yet widely used in the assessment of *still* water ecosystems, but have been widely applied, particularly in continental Europe, to river pollution monitoring (Whitton *et al.* 1991). At present, within the EU regular monitoring of still water periphyton assemblages is only undertaken in the Netherlands, where epiphytic diatom assemblages are monitored at about 650 sites, including lakes, ponds, canals and ditches (Roos *et al.* 1991).

Community-based assessment methods indicating specific environmental stresses
Diatom communities are widely used in river bioassessment as indicators of organic pollution (saprobity), and to monitor eutrophication, acidification and the impact of salt (Whitton *et al.* 1991). The relationships between individual diatom taxa (often species) and specific environmental factors are reasonably well-known and have enabled 'indicator values' to be proposed for a large number of diatom species, including many of still waters. For example, indicator values for the Netherlands diatom flora (948 diatom taxa including 776 species) have been prepared by van Dam *et al.* (1994), covering responses to pH, oxygen requirements, saprobity, trophic state, nitrogen concentrations and moisture (see Table 4.7). The sensitivity of individual diatom taxa is assessed on a 4 to 7 point scale, individual values being based on literature data and field studies.

Diatom indicator values of the type shown in Table 4.7 provide the basis for a number of biotic indices using diatoms to assess river pollution (Whitton and Kelly 1995). These include the Indice Diatomique (Id) and the Indice de Polluosensibilité (SPI), which measure organic pollution, and the Organic Pollution Index developed by Steinberg and Schiefele (1988), which considers both inorganic nutrient enrichment and organic pollution.

Although there is a long tradition of using diatoms for river water quality monitoring in continental Europe, in Britain it is only relatively recently that diatom-based assessment techniques have been introduced. The EA is currently testing a new assessment method, the Trophic Diatom Index (TDI) developed by Martyn Kelly and colleagues (Kelly *et al.* 1996). The TDI is intended to provide an alternative tool for assessing eutrophication in rivers, particularly with reference to the Urban Waste Water Treatment Directive.

The TDI uses just over 80 diatom genera, chosen for their indicator potential and ease of identification, to make an assessment of the degree of eutrophication. Epilithic diatom samples are taken from boulders of diameter greater than 265 mm although other substrates may need to be used in slow flowing rivers. At sites relatively free of organic pollution the TDI was more highly correlated with aqueous P concentrations than previous diatom indices. However, where there was heavy organic pollution it was difficult to separate the effects of eutrophication form other effects. To allow for this effect the TDI is supplemented by an indication of the sample that is composed of organic pollution tolerant taxa (Kelly and Whitton 1995).

Table 4.7 Classification of diatom ecological indicator values in the Netherlands (van Damm *et al.* 1994)

Environmental variable	Value	Description	Environmental characteristics
pH	1	Acidobiontic	Optimal occurrence at pH <5.5
	2	Acidophilous	Mainly occurring at pH <7
	3	Circumneutral	Mainly occurring at pH values about 7
	4	Alkaliphilous	Mainly occurring at pH >7
	5	Alkabiontic	Exclusively occurring at pH >7
	6	Indifferent	No apparent optimum
Oxygen requirements	1	Continuously high	(About 100% saturation)
	2	Fairly high	(above 75% saturation)
	3	Moderate	(above 50% saturation)
	4	Low	(above 30% saturation)
	5	Very low	(about 10% saturation)
Saprobity	1	Oligosaprobous	BOD_5 <2 mg l^{-1}
	2	ß-mesosaprobous	BOD_5 2-4 mg l^{-1}
	3	Alpha-mesosaprobous	BOD_5 4-13 mg l^{-1}
	4	Alpha-meso/ polysaprobous	BOD_5 13-22 mg l^{-1}
	5	Polysaprobous	BOD_5 >22 mg l^{-1}
Salinity	1	Fresh.	Cl^-: <100 mg l^{-1} Salinity: <0.2 °/oo
	2	Fresh brackish.	Cl^-: <500 mg l^{-1} Salinity: <0.2 °/oo
	3	Brackish fresh.	Cl^-: 500-1000 mg l^{-1} Salinity: <0.2 °/oo
	4	Brackish.	Cl^-: 1000-5000 mg l^{-1} Salinity: <0.2 °/oo
Trophic state	1	Oligotraphentic	
	2	Oligo-mesotraphentic	
	3	Mesotraphentic	
	4	Meso-eutraphentic	
	5	Eutraphentic	
	6	Hypereutraphentic	
	7	Oligo- to eutraphentic (hypereutraphentic)	
Moisture	1	Never, or very rarely, occurring outside water bodies	
	2	Mainly occurring in water bodies, sometimes on wet places	
	3	Mainly occurring on water bodies, also rather regularly on wet and moist places	
	4	Mainly occurring on wet and moist or temporarily dry places	
	5	Nearly exclusively occurring outside waterbodies	

Multimetric methods describing general ecosystem quality

The US EPA propose to use multimetric assessments of periphyton as a method for assessing lakes and reservoirs ecological integrity. However, as in Europe, the use of periphyton for still water assessment is still essentially an experimental procedure. A suggested survey method has been proposed, involving removal of periphyton at random locations around the lake edge. However, to date, no metrics have been suggested. Metrics of periphytic diatoms have shown promise for bioassessment, based on investigation of undisturbed reference lakes in Montana, but the actual response to pollution is not known (EPA 1995).

Bioassay approaches

Periphyton and attached diatoms have not been widely used in bioassays or ecotoxicological tests. This is probably because most bioassays are done over short periods and periphyton assemblages (and diatoms in particular) have been found hard to establish and culture reliably. However, an increasing number of flow-through studies are being undertaken which attempt to simulate the effects of pollutants on stream periphyton communities (e.g. Belanger *et al.* 1996, Mitchell *et al.* 1993, Schneider *et al.* 1995).

4.7.2 Potential for the development of periphyton assessments methods

The paucity of periphyton assessment methods and applied research data, means that most information relating to these assemblages is theoretical and relates to their potential rather than their proven value. A summary of the potential advantages and disadvantages of periphyton assessment methods is given below.

Advantages

In theory, periphyton may offer similar water quality monitoring and diagnostic potential to macroinvertebrates. Ten Cate *et al.* (1993), for example, suggest that diatoms explained a similar amount of environmental variation in standing and running waters in the Netherlands as did macroinvertebrate assemblages (Verdonschot 1992). Vos andOpdam (1993) found better general correlations between the regional water chemistry and diatom assemblages than between the water chemistry and the macrophyte vegetation. They suggested that this was because macrophytes were, in part, reflecting the qualities of sediments as well as water quality, whereas diatoms assemblages were influenced mainly by water column chemistry.

As a group diatoms, in particular, are ubiquitous. They occur in all still water body types including temporary and brackish sites (van Dam *et al.* 1994) and are unusual in that many individual species are also geographically widespread and appear to have similar environmental tolerances (Reid *et al.* 1995). As a result, use can often be made of taxonomic and ecological studies from many parts of the world when establishing the environmental indicator value of diatoms. Diatoms are also found in abundance across a wide variety of water quality types including clean and grossly polluted sites, brackish waters, alkaline and acid waters. Species diversity is, however, relatively low in highly acid waters. Diatoms are also numerically abundant in most aquatic environments.

In comparison with phytoplankton, periphyton diatoms have longer average cell cycles (periphyton: 1-100 days cf. phytoplankton: 1-50 days) (Rott, 1991). As a result they have a rather greater potential to integrate changes in ecosystem quality. They can also be surveyed all year round, although biomass and abundance is usually greatest in spring and summer.

Disadvantages

One of the major disadvantages of using diatoms for still water assessment is the lack of research data about potential assessment methods. A number of workers have shown that diatom assemblage composition varies between substrate types (Kelly *et al.* 1996), but there is as yet, little work to suggest whether such difficulties can be adequately overcome through appropriate survey techniques. Similarly, there is little information about temporal and between habitat variability. Methodological testing is therefore likely to be a pre-requisite to any attempt to develop periphytic diatoms techniques for routine water quality assessment.

A second disadvantage is lack of information relating to the relationship between diatoms and degradation at *family* level, which is the most realistic level of identification for routine monitoring. Linked with this are disadvantages relating to the relatively high set-up costs requiring high performance microscopes and the facility for scanning electron microscopy. Time requirements for sample preparation and identification at generic level, are similar to those for macroinvertebrate processing. However, species level identification requires a very high level of taxonomic competence.

4.7.3 Evaluation of the operational feasibility of diatom-based techniques

It is clear that a considerable amount of development work is required before periphyton methods could be put into use for still water assessments. Particular doubts relate to the extent to which inherent temporal, habitat and particularly substrate variability, reduces their practical viability.

For *surveillance and compliance monitoring,* diatoms have similar characteristics to macroinvertebrates. They are widespread, species rich and sufficiently abundant to make sampling straightforward. They may also have a similar potential in their response to degradation, and may prove complementary to invertebrates in the range of impacts to which they respond. However, comparison between the two groups is difficult given the absence of research data.

The results of the EA Trophic Diatom Index trials are likely to cast light on the practical viability of using diatoms at generic level for water quality assessments. Although this method has been developed for rivers, the methodological information is likely to be directly applicable to still waters.

Diagnosis and early warning
The strong environmental preferences which are reported for diatom species (acidification, eutrophication, metals etc.) suggests diatoms may have considerable potential as a diagnostic tool. However, again, much work is required to examine the viability of generic-level assessments for still waters.

Use of diatoms as an early warning indicator would rely on the potential for species to show strong discrimination of pollutant gradients, so that trends in degradation could be identified rapidly. Discrimination at this level will almost certainly require species-level identification of diatoms. Thus although they have considerable potential for early warning, in practice, the detailed analysis required may deter use of diatoms for this purpose.

4.8 Macrophytes

4.8.1 The importance of macrophytes

Macrophytes include the marginal and aquatic vascular plants, aquatic bryophytes (mosses and liverworts) and stoneworts (Charophyceae). Macrophytes occur in all still water habitats, although temporary waters sometimes lack truly aquatic species. Most vascular macrophytes occur in relatively shallow water (up to 5-6m), but mosses and charophytes may be found at depths of 50m or more in the clearest lakes (Hutchinson 1975).

The macrophytes are a moderately species rich assemblage with abut 450 species known from all freshwater wetland habitats in Britain. Macrophytes profoundly influence the structure and function of littoral still water communities, providing physical habitat for periphyton, invertebrates and fish. Macrophytes are also one of the most conspicuous assemblages in standing waters, widely recognised as significant by non-biologists. The taxonomy and distribution of macrophytes is well known in the UK and Red Data Books, listing vulnerable and sensitive species, are available for the vascular macrophytes and charophytes (Perring and Farrel 1983, Stewart and Church 1992).

4.8.2 Existing assessment methods using macrophytes

Rooted and floating macrophytes respond to a variety of environmental stresses including nutrient enrichment, metal contamination (particularly copper), micro-organic contamination (specifically herbicides), salinisation, turbidity (including dense periphyton and phytoplankton growth), acidification and water level changes.

However, in practice, most macrophyte-based assessment methods have used plants only to assess trophic status or eutrophication. More rarely, plant bioassays are used to assess the distribution in the environment of metals and micro-organics, although these techniques are not yet widely used in practical monitoring programmes.

Community based methods indicting specific environmental stresses

In *standing* waters in Britain, bioassessments using wetland plants (particularly submerged aquatic species) have mainly focused on lake trophic status and changes associated with eutrophication. These techniques have been most fully developed by Palmer *et al.* (1992) who prepared a botanical classification of lakes in Great Britain and developed the concept of Trophic Ranking Score. The lake classification is based on presence/absence data for aquatic macrophyte species and groups sites into 10 main (essentially trophic) categories. Comparisons between categories can be made on the basis of average plant species richness at a site, providing an approximate index of community quality.

The Trophic Ranking Score system, which was developed in parallel with the lake classification, is based on the presence of indicator species, weighted (1 to 10) according to the mean nutrient status of the waterbodies in which they typically occur (from dystrophic to highly eutrophic). The average Trophic Ranking Score from each site therefore gives an approximate assessment of the trophic status of the waterbody. There is, therefore, the potential to use this method for assessment of both acidification and eutrophication trends in still waters. The main difficulties with the application of this method in practice are lack of information relating to (i) sampling variability and (ii) the degree of sensitivity to changes in trophic state (or acidification).

The EA is currently developing a modified version of the trophic ranking, the Mean Trophic Rank, for use in monitoring of river eutrophication (Holmes 1995) This method is intended for use in compliance monitoring for the Urban Wastewater Treatment Directive, where it is necessary to distinguish nutrient pollution from organic pollution. The results of the testing phase of this project are likely to prove insightful for further development of trophic ranking methods in still waters. A similar approach, the Macrophyte Index Scheme, has been used for a number of years in the Republic of Ireland (Caffrey 1987). This places macrophyte taxa into one of four categories (sensitive, less sensitive, tolerant and more tolerant) according to their perceived sensitivity to eutrophication. The occurrence and abundance of the plants in these four categories is then used to place river sites into one of the five categories for the Quality (Q) Rating System.

Monitoring of macrophyte communities is common in still water bioassessment programmes throughout Europe, with species composition of the flora and vegetation abundance used as the main indicators of waterbody status (see below). For example, in Denmark, submerged macrophyte coverage is measured annually in 17 of the 37 lakes monitored nationally, with more detailed investigations every 5th year (Danish Environmental Protection Agency 1993).

Macrophyte bioassays for assessment of metals and micro-organics

A number of studies have suggested that bioaccumulation of metals and micro-organics by macrophytes could be used to provide an indication of intermittent or chronic pollution including the impacts of mercury (Mortimer 1985), various heavy metals (As, Cd, Cu, Fe, Pb, Zn and Cr) (Mouvet 1985, Whitton *et al.* 1991) and chlorinated hydrocarbons (Mouvet 1994). Although a number of the proposed techniques have undergone laboratory and field trials in Europe and the United States, most techniques remain speculative. All require more extensive testing to calibrate species specific uptake and contaminant levels under field conditions before they can be routinely used. The two main approaches which have been suggested are:

(i) Use of laboratory cultured plants, particularly bryophytes (which do not suffer the seasonality effects of higher plants). For example:

In France, Mouvet *et al.* (1985a) transplanted the aquatic moss *Fontinalis antipyretica* to monitor concentrations of copper and cadmium in a river receiving intermittent pollution. Leaves were stained and examined using electron microscopy.

In other work, Mouvet (1994) examined the use of an abundant river bryophyte *Cinclidotus danubicus* as a potential indicator of chlorinated hydrocarbon pollution, and tested the method in the rivers Saone and Durance (tributaries of the Rhone). Laboratory purified extracts were measured using gas chromatography techniques to establish pollutant levels.

(ii) Use of plants growing *in situ*

Whitton *et al.* (1991, 1995) have suggested that bioassays of river macrophytes growing *in situ* can be used to monitor heavy metals in UK rivers. *Elodea canadensis* and *Potamogeton pectinatus* were recommended as the assay species because of their widespread distribution and tolerance of organic pollution.

Multimetric methods for assessment of general ecosystem quality

Multimetric methods have not yet been widely applied to the monitoring of still water communities. However, US EPA is currently developing multimetric assessment techniques for lakes and reservoirs as part of the proposed national biomonitoring programme. The draft methodology has proposed seven metrics derived from surveys of aquatic (not marginal) macrophytes species. Field data is likely to be obtained from grapnel trawls perpendicular to the shore. The proposed metrics are:

- total vegetated area (% of littoral)
- percentage exotics or weedy species
- number of exotic species
- density or biomass in vegetated areas
- taxa richness
- percentage of dominant species (by weight)
- maximum depth of plant growth

Note that these are currently *proposed* metrics and have yet to be verified by field testing.

In Europe, assessments of general ecosystem quality are, conceptually, less well-developed and *still* water multimetric indices have only been used in a limited form. Pond Action, for example, used two commonly measured components of biotic integrity (species richness and species rarity) to assess the quality of ponds in regional and national surveys (Pond Action 1994a,b). However, macrophyte data which may be suitable for inclusion in multimetric IBIs is widely collected in the United Kingdom and elsewhere in Europe.

Although there are no specifically multimetric indices in use in Europe, Haslam (1990) has proposed what is, in effect, a multimetric index for river assessment, the Damage Rating system. This method has several features in common with the multimetric methods in that it compares the plant community of a river reach with the best available reference site using a number of measures of ecosystem integrity and is used within the context of a classification. In addition, the extent of ecosystem degradation is assessed in terms of six variables: species richness, percentage of pollution tolerant species and weightings for (i) nutrient tolerant species (ii) substrate (iii) vegetation abundance and (iv) community type compared to reference sites. The Damage Rating value (which Haslam has tested throughout western Europe) has not yet been widely adopted in practice, perhaps in part because the reference classification proposed (based on Haslam's colour banding system) is not sufficiently 'mainstream' (i.e. not based on a multivariate method such as TWINSPAN) to be credible. In addition to Haslam's Damage Rating value, Boon *et al.* (1996) have also included a number of measures of plant community integrity in the SERCON system in a way which is also analogous to a multimetric integrity index.

Other general ecosystem assessment methods include plant based classifications such as the UK's recently published National Vegetation Classification (NVC) for mires and aquatic plants (Rodwell 1991, 1995). These classifications provide a means of distinguishing plant community types on the basis of the presence or absence of indicator species. Descriptive comment is often also given summarising observed (and presumed) relationships between community types and aspects of degradation (e.g. eutrophication). However, the NVC has a number of important limitations for monitoring:

(i) there is no statistical assessment of the relationships between vegetation and environmental features,

(ii) there is no adequate means of analysing the significance of a change in community type if one is observed (i.e. is a change from NVC type A9, *Potamogeton natans* community to NVC type A8 *Nuphar lutea* community good or bad - and if bad, how bad?),

(iii) there is no indication whether groups differ due to natural influences (such as shade, region, water depth) or anthropogenic stresses,

The NVC cannot, therefore, be used as it is to provide a basis for the identification of undisturbed reference sites and, as it stands, has relatively little potential for monitoring the quality of freshwater plant communities.

River Corridor Surveys (RCS) are worth mentioning briefly in this section, because although primarily a type of habitat survey (see Section 4.13) they include aspects of a general plant community assessment. The RCS method, which was designed to provide practical guidance to land drainage engineers about the location of river channel features which should be protected, is based on surveys of 0.5 km river lengths. Macrophyte recording includes noting plant species rarity and species dominance. The method could be easily adapted for use in canals, ditches, lakes and ponds. The main drawback of the existing RCS technique for assessment work is the lack of survey consistency (e.g. species lists for each length, based on a reference list, are not consistently collected), which prevents the results from being used rigorously to compare lengths or assess temporal change.

4.8.3 Potential suitability of macrophytes assessments

Advantages and disadvantages

As a group used for monitoring and assessment purposes, macrophytes have a number of practical advantages:

(i) Within the growing season, macrophytes growing *in situ* are likely to provide a good temporal integration of ambient water quality.

(ii) The group is well known taxonomically, and, for most groups, quick to survey to species level. This enables parameters such as species rarity and species preferences to be used in rapid assessment.

(iii) Macrophytes field survey techniques are well-developed and relatively simple.

(iv) Macrophyte assemblages are naturally present in a wide range of still water body types (excluding naturally shaded sites).

The most important practical disadvantage of macrophytes is their seasonality. To obtain a good species list, or to estimate abundance reliably, surveys can only be made in a relatively short time in the summer, usually between July and October. In addition:

(i) There may be few macrophyte species present in temporary and, particularly shaded, sites.

(ii) The characteristics of *marginal* macrophyte communities may be highly variable depending, for example, on bank profile and management. The quality of marginal communities may also reflect the quality of the surrounds (Pond Action 1994a) rather than water quality.

(iii) Within water bodies macrophytes may be quite patchily distributed, and therefore time consuming to survey.

(iv) Many vascular macrophytes species are rooted in the sediment and reflect both water column quality and sediment quality (EPA 1995). Plants with a simpler morphology, such as bryophytes (and algae), are therefore likely to be better indicators of pollution, since the accumulation of toxicants is largely from the water rather than the sediment.

Evaluation of operational feasibility

General Ecosystem Monitoring. As a popular and structurally important group, there is a strong case for including macrophytes in any general ecosystem quality monitoring programme. The case is clearest for *aquatic* plants, which have a known relationship with waterbody trophic state and respond to other pollutant influences such as biocides, turbidity and salinity. The inclusion of *marginal* macrophytes is less likely to be necessary, because marginal plant community structure is, in part, reflecting the quality of the terrestrial environment.

Multimetric assessments tested on variables such as species richness, occurrence of sensitive species, abundance (and potentially Trophic Ranking Score) seem likely to provide a good basis for general ecosystem surveillance, with the benefits that macrophyte surveys are relatively inexpensive compared to other surveys carried out to the same taxonomic level (i.e. species level). In addition, the results of macrophyte surveys are relatively easy to convey to managers and non-specialists.

There is a considerable body of macrophytes survey data for many still waters, some of which might provide the basis for a national database of reference sites.

Diagnosis and early warning. Macrophyte-based trophic ranking and bioaccumulation of metals and other toxicants clearly provide potential methods for investigation of the causes of observed degradation.

Bioaccumulation might, in addition, provide the basis for an effective early warning system where there is a known potential for degradation by metal pollution (e.g. effluent emissions). However, the applicability of this technique to still waters is likely to be low because neither bioaccumulation or trophic ranking are ideal as *general* early warning systems. Bioaccumulation methods are likely to be inappropriate because they provide information on only a small range of pollutant types. Trophic Ranking is not likely to be suitable for early warning because it is based on assessment of 'after the event' changes in plant community.

4.9 Microinvertebrates

4.9.1 The importance of microinvertebrates

The microinvertebrates include a wide range of smaller invertebrate animals, with some 1300 species drawn from abut 10 phyla. In monitoring programmes they are largely synonymous with the zooplankton, there being much less information available about the assemblages of littoral and benthic microinvertebrates. Table 4.8 lists the main microinvertebrate phyla found in freshwaters, and the numbers of species known in the United Kingdom (with the exception of Protozoa).

Although the great majority of microinvertebrates are little known, the assemblage includes the most intensively studied of all freshwater organisms, cladocerans in the genus *Daphnia*. In a recent literature review it was found that 20% of all scientific papers dealing with freshwater invertebrates referred to members of this genus alone (Pond Action, unpublished data). Microinvertebrates occur in all freshwater habitats but their distribution patterns are poorly known and, consequently, no microinvertebrate group is either (i) listed in Red Data Books that identify sensitive species or (ii) specially protected in legislation.

Table 4.8 Numbers of species in the principal microinvertebrate groups (Maitland 1977, DOE unpublished)

Protozoa	Unknown	Tardigrada	42
Porifera (sponges)	8	Hydracarina	336
Cnidaria (*Hydra* and allies)	8	Cladocera	90
Playthelminthes	47	Ostracoda	88
Nematoda	81	Copepoda	112
Rotifera	476		
Gastrotricha (Hairybacks)	22	Total 1319	
Bryozoa	9		

4.9.2 Existing assessment methods using microinvertebrates

Community based methods indicating specific environmental stresses

A very wide variety of environmental factors are known to influence populations of microinvertebrates. However, with the exception of the Saprobic system (which is used in running waters), no microinvertebrate groups have been used as indicators of specific environmental stresses, primarily because of the taxonomic difficulties associated with most groups.

The Saprobic system, as originally conceived, included indicator values for a very wide range of microinvertebrates, including protozoans (Sládecek 1979, Friedrich 1990). In practice, however, all countries where the Saprobic system is used, survey the *macro*invertebrate component and do not routinely use microinvertebrates.

Marmorek and Korman (1993) reviewed the potential for use of zooplankton in acidification monitoring. Although there are consistent changes in zooplankton assemblages associated with acidification, the changes that do occur have not yet been incorporated into any indexing system. None of the methods proposed by these authors are widely applied, either in Europe or the United States, and all would require further development before they could be used to monitor environmental stress, either alone or as part of a multimetric method.

Multimetric methods using microinvertebrate assemblages

At present there are no fully developed multimetric methods for assessing the integrity of microinvertebrate populations. However, at least one method is being tested in the United States, based on zooplankton assemblages (Fore and Karr 1994). In addition the US EPA has suggested potential zooplankton metrics, although these have not yet been tested. These are:

- %large Daphnia spp. (>1 mm)
- Tax richness
- Percent dominance
- Percent large Daphnia
- Trophic structure metrics:
  - (i) no. trophic links
  - (ii) complexity measures
  - (iii) percent large predators
  - (iv) no. predator species

There are already a wide range of monitoring programmes in which zooplankton populations are monitored, and population parameters such as species richness, abundance and other structural attributes recorded. Zooplankton are routinely monitored in a number of European lake monitoring programmes. For example in Denmark zooplankton samples are collected from representative lakes in the national monitoring programme at roughly fortnightly intervals. Zooplankton sampling is also undertaken in national sampling programmes in Norway, Sweden, Netherlands, Luxembourg, Poland and Finland.

Ecotoxicology and bioassay

Microinvertebrates, particularly *Daphnia magna* and *Ceriodaphnia dubia*, are two of the most important ecotoxicological test organisms. However, despite the vast range of tests undertaken using these animals few are applicable to assessment of biological integrity, most being more relevant to diagnosis and, possibly, early warning methods. Johnson (1995) has recently recommended that a *Daphnia magna* in vivo enzyme inhibition test could be included in a suite (of four) effluent screening toxicity tests for use by EA (see Section 4.14).

4.9.3 The potential suitability of microinvertebrates for water quality monitoring

Advantages and disadvantages for bioassessment

Microarthropods, particularly zooplankton, are easy to collect and may be relatively homogeneously distributed in open water habitats. They occur in all freshwater habitats, can be found throughout the year and are often highly sensitive to environmental stress. Microinvertbrates represent an important component of freshwater biodiversity. Zooplankton, in particular, inhabit the water column where they may be exposed to water column pollutants. They also respond quickly to any changes in the environment.

The principal disadvantage of microinvertebrates for bioassessment is that there has been too little research on the range of responses to environmental stress for them to be used as indicators.

In addition, zooplankton populations are:

(i) spatially and temporally varied, so that relatively large numbers of samples are required to obtain reasonable estimates of population variable,

(ii) taxonomically difficult (especially any group other than cladocerans),

(iii) strongly influenced by other assemblages, especially fish.

Evaluation of the operational feasibility of microinvertebrate techniques

General ecosystem quality monitoring. The lack of basic research on the responses of microinvertebrate communities to environmental stressors suggests that they are unlikely to provide information which cannot be obtained from other assemblages. The frequency of sampling, combined with the taxonomic problems of dealing with most microinvertebrates (apart from the Cladocera), suggests that microinvertebrates are not suitable as a general assessment technique at present.

Diagnosis and early warning. Microinvertebrates are likely to remain important for diagnostic investigations particularly laboratory studies where their ease of culture and sensitivity are important advantages. At present there are no practical techniques which are likely to be suitable for providing early warning methods, although with their short life cycles and rapid responses this might be an area that would be worthy of further development.

4.10 Macroinvertebrates

4.10.1 The importance of macroinvertebrates

Macroinvertebrates are a diverse group, collectively comprising ca. 1,500 freshwater species in Britain. Table 4.9 lists the number of species in the major groups. Macroinvertebrates occur in all still water habitats and for most UK species (apart from the Diptera) taxonomic and distribution data is reasonably good. Uncommon species are listed in Red Data Books and ten species are given special protection under the provisions of the Wildlife and Countryside Act 1981 (Biggs *et al.* 1995).

Macroinvertebrate assemblages respond to a very wide range of environmental stresses, although most monitoring is concerned with the effects of organic pollution and eutrophication. In general there is only moderate public interest in macroinvertebrates, reflecting a general concern for nature conservation and the protection of freshwaters.

Table 4.9 Numbers of species in the major macroinvertebrate groups (Maitland 1977, DOE unpublished)

Tricladida (Flatworms)	12	Neuroptera/Megaloptera	7
Hirudinea (Leeches)	16	Trichoptera	198
Gastropoda	44	Lepidoptera	5
Bivalvia	28	Coleoptera	273
Malacostraca	40	Diptera: Chironomidae	510
Ephemeroptera	48	Diptera: Tipulidae	227
Plecoptera	34	Other Diptera	428
Odonata	45		

4.10.2 Existing assessment methods using macroinvertebrates

Macroinvertebrates have long been recognised as important indicators of environmental quality and this is reflected in the large number of invertebrate-based bioassessment methods that have been developed. Although the great the majority of techniques have been developed for use in *running waters* many are conceptually relevant to still waters.

Macroinvertebrate methods which have been used to diagnose environmental stresses can be divided into two groups:

1. Community based assessments including:
   - methods for assessing the impact of a specific environmental stress (such as organic pollution),
   - methods for assessing *general* ecosystem quality.

2. Bioassays of individual species (often laboratory based)

Community based methods for the assessment of specific environmental stresses

To date a limited number of macroinvertebrate assemblage-based methods have been developed for use in still waters, the majority of which are concerned either with eutrophication or acidification. Several indexing methods for assessing trophic status, and changes in trophic status, have been proposed based on the composition of oligochaete and chironomid communities, such as the Environmental Index (Mozley and Howmiller 1977) and the Benthic Quality Index (Wiederholm 1980). A smaller number of studies have investigated the use of invertebrates for assessment of acidification, leading to the development of techniques such as the Acidification Index of Fjellheim and Raddum (Raddum and Fjellheim 1985, Fjellheim and Raddum 1990).

Lake invertebrate monitoring in Europe has mainly used benthic invertebrate communities. Samples are variably assessed in terms of their *overall* (i.e. community) species richness, biomass, number of individuals and, less frequently, size distribution (e.g. of mussels) (DEPA 1993, R.K.Johnson pers. comm., Premazzi and Chiaudani 1992). Invertebrate community assessments of ditches and canals are undertaken more rarely (H. Tolkamp pers. comm., Roos *et al.* 1991).

Most monitoring is undertaken on macroinvertebrate community assemblages as a whole. However, some workers have also advocated the use of specific taxonomic groups as the basis of monitoring techniques (mainly for lakes). Taxa recommended include: oligochaetes using proportion and abundance of indicator species (e.g. Howmiller and Scott 1977; Lang and Lang-Dobler 1980; Wiederholm 1980; Milbrink 1983); chironomids based on indicator species, relative abundance and biotic indices (e.g. Saether 1979; Wiederholm 1980; Courtemanche 1989, Wilson 1994, Koskenniemi and Sevola 1992), the ratio of chironomids to oligochaetes (Wiederholm 1980); Mollusca based on proportion and abundance of gastropods and bivalves (Mouthon 1993) and corixids based on the frequency of indicator species (Savage 1995).

The relatively restricted group of invertebrate attributes and taxa used for *still* water assessment belies a much more extensive list of community assessment techniques used for *streams and rivers*. To exemplify the range of approaches, Table 4.10 groups together six categories of method that are commonly used in Europe and the United States. Traditionally, European workers have advocated use of methods from the first four categories, that is:

(i) similarity indices (e.g. Jaccards index, and the conceptual extension of these methods, multivariate community analysis methods, such as DECORANA and CANOCO),

(ii) diversity indices (e.g. Shannon-Weaver H'; Simpson's D),

(iii) biotic indices (e.g. saprobic system, Chandler Score, BMWP/ASPT),

(iv) richness measures (e.g. species or taxa richness).

The two other categories are more normally associated with monitoring programmes in North America i.e.:

(v) enumerations (e.g. number of individuals, percent EPT taxa of total fauna),

(vi) functional measures (percent shredders, percent scrapers).

A brief summary of each of these broad assessment approaches is given below. The historical sequence of development of biotic indices is shown in Figure 4.12.

Similarity indices. Similarity indices describe how similar two samples are, usually in terms of the number of taxa they contain. Similarity indices are now largely superseded by multivariate methods, although their simplicity is sometime useful. Interestingly they are recommended as an integral part of environmental assessment work in marine ecology, although they figure little in practical bioassessment studies. The oldest (and simplest), Jaccards Index which was defined in the 1920s, is still used. It is simply the ratio of species in one sample compared to another.

Table 4.10 Examples of approaches used for assessment of macro-invertebrate assemblages in Europe and the United States

1. Similarity Indices
- Coefficient of Community Loss (Courtemanch and Davies 1987)
- Jaccards Index (Jaccard 1912)
- Pinkham-Pearson Community Similarity Index (Pinkham and Pearson 1976)
- Number of dominant taxa in common
- Number of taxa in common
- Quantitative Similarity Index (Barbour *et al.* 1992)
- % change in taxa richness
- Number of unique species per site
- Missing EPT taxa at study site (cf. reference site)
- Index of Community Integrity (Hayslip 1992)

2. Diversity indices
- Shannon's Index (Shannon 1948)
- Margalefs Index (Margalef 1951)
- Menhinick's Index (Menhinick 1964)
- Simpson's Index (Simpson 1949)
- Equitability (Hayslip 1992)

3. Biotic indices
- Trent Biotic Index (Woodiwiss 1964)
- Belgian Biotic Index (De Pauw and Vanhooren 1983)
- Biotic Condition Index (Plafkin *et al.* 1989)
- Biotic Index (Chutter 1972; Hilsenhoff 1982, 1987, 1988; Lenat 1993).
- BMWP score (Wright *et al.* 1988)
- Chandler Biotic Score (Chandler 1970)
- Florida Index (Ross and Jones 1979)
- Indicator-organism presence
- ISO score (ISO 1984)
- Community Tolerance Quotient (Winget and Mangum 1979)
- Saprobic Index (Zelinka and Marvan 1961)
- Dominance of tolerant groups (Plafkin *et al.* 1989)
- Indicator Assemblage Index (Shackleford 1988; Hayslip 1992).

4. Richness measures
- Number of taxa
- Number of families
- Number of species
- Number of Ephemeroptera, Plecoptera, Trichoptera (EPT) taxa
- Niche occupancy forms (Mason 1979)
- Number of Ephemeroptera, Plecoptera, Trichoptera, Diptera taxa considered individually
- Number of intolerant snail and mussel taxa
- Number of Chironomidae
- Number of Crustacea and Mollusca taxa

5. Enumerations
- Number of individuals (or biomass)
- %EPT individuals
- %Chironomidae individuals
- %Tribe Tanytarsini individuals
- Ratio of EPT/Chironomidae individuals
- Ratio of Hydropsychidae/Trichoptera
- %Individuals of numerically dominant taxa
- %Non-dipterans
- %Non-Chironomidae, Diptera and other insect individuals
- Relative abundance of different individuals
- Five dominant taxa in common
- Common taxa index (Shackleford 1988)
- Indicator groups (Hayslip 1992)
- Relative abundance of different groups
- %Ephemeroptera, Trichoptera, Tanytarsini, Chironomidae, other Dipterans and non-insects (considered individually)
- %tolerant groups (Ohio EPA 1987; Yoder and Rankin 1995

6. Functional measures
- % Shredders (Cummins 1988)
- % Scrapers (Cummins 1988)
- % Collector-filterers (Cummins 1988)
- % Filterers (Cummins 1988)
- % Strict predators (Kerans *et al.* 1992)
- % Omnivores and scavengers (Kerans *et al.* 1992)
- Ratio of scrapers/collector-filterers (Cummins 1988)
- Ratio of trophic specialists/generalists (Maine Department of Environmental Protection 1987)
- Types of functional-feeding groups (Cummins 1988)
- Functional group similarity (Cummins 1988)

Diversity indices. Diversity indices summarise the relationship between species number or richness (S) and species abundance (N). North American workers still use diversity indices quite frequently, particularly Shannon-Weiner index (*H'*) (Resh and Jackson 1993), despite extensive criticism of the method both in European and North American. For example, Bob May noted as long ago as 1975 (May 1975a) that the Shannon-Weiner *H'* was an insensitive measure of underlying abundance changes and that it would be better simply to describe the range of abundances of individual taxa, rather than condensing observations to a single number. Hawkes (1979) also noted that communities naturally poor in species, as well as polluted sites, could have low diversity scores, whilst Washington (1984) called diversity indices 'unsatisfactory owing to the lack of exploration of their biological relevance'.

Biotic indices. Biotic indices use organisms as indicators of specific environmental stresses. All macroinvertebrate biotic indices have their origin in the Saprobic system which was first conceived at the beginning of the 20th century. European biologists divided into two camps at mid-century, continental Europeans continuing to develop Saprobic index related methods, and biologists in Britain moving to apparently simpler systems which used smaller numbers of taxa.

Biotic indices are overwhelmingly popular in Europe still, although they are severely limited in referring primarily to organic pollution. Most countries in western Europe use one form of these indices or another (a full list is given by Premmazi and Chiaudani (1992)). Alternative biotic indices used in North America include the Florida Index (Ross and Jones 1979) and the Family Biotic Index (Hilsenhoff 1982), used in Wisconsin.

Richness measures. One of the simplest measures of community status is simply species richness. However, this fundamental parameter is likely to be of increasing interest as biodiversity assumes greater importance. Richness measures are inherent in all similarity, diversity and biotic indices.

Enumerations. Enumerations include counts of the number of individuals and of the proportions of particular taxa (e.g. %EPT - Ephemeroptera, Plecoptera, Trichoptera- taxa or other sensitive taxa). Enumerations are important because they convey information about the structure of the community and, because they are normally stated as proportions, can be derived from semi-quantitative data. Table 4.10 lists some of the wide range of enumerations than can be used. Enumerations are rarely used in European macroinvertebrate monitoring but are an integral part of bioassesment in North America.

The simplest enumeration measure is number of individuals and, as noted by May (1975a), it may be more useful to keep this piece of information separate for each taxon of interest than to conceal it within a diversity index. Similarly, rather than considering all members of the invertebrate fauna as in a BMWP score, in which there are many 'organic pollution-neutral' taxa (such as most water beetles and snails), the use of EPT taxa tends to focus attention on just the most sensitive groups. Other groups may also contain relevant information which can be swamped in a simple biotic index.

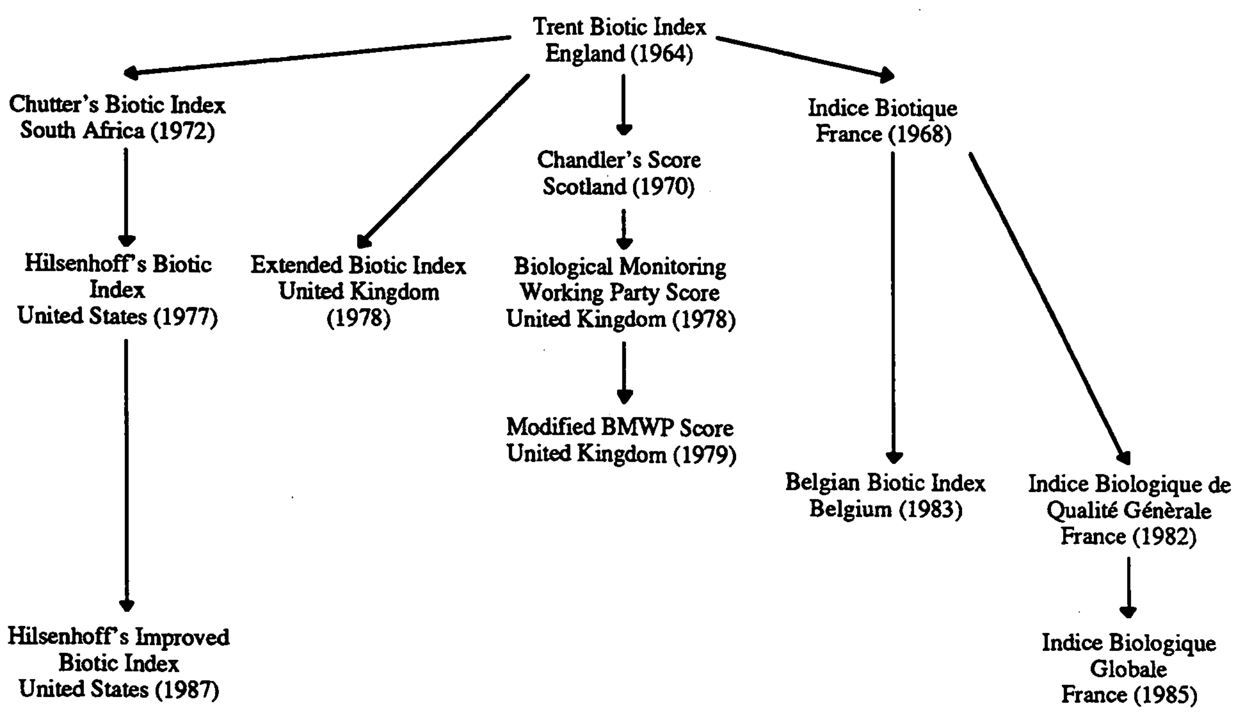
Simple measures such as %Dominant taxon may be expected to increase under stress whilst the mean number of individuals per taxon is either substantially higher *or* lower. It is quite likely that other simple enumerations could be useful. For example in pond ecosystems Pond Action (1994a) noted that the ratio of water beetle species to other taxa was correlated with water body permanence, one of the driving variables in still water habitats.

Functional measures. North American stream ecologists have used the concept of functional feeding groups in studies of invertebrates for some years, after they were first proposed by Cummins (1973). Table 4.11 shows the main categories of functional feeding groups. Studies have shown that proportions of functional feeding groups shift markedly in degraded streams. Describing changes of this sort in terms of functional groups focuses attention on stream function as well as simply on richness and loss of sensitive organisms. In this way attention may be drawn to impacts which are not simply pollution related, but also relate to stream structure. For example, large particulate shredders may be important in forested streams, and the absence of this group is characteristic of North American headwater streams that are unstable and poorly retentive of Coarse Particulate Organic Matter (CPOM) (Minshall *et al.* 1985). The use of functional groups has been criticised by some workers on the basis that it can be difficult to assign taxa accurately to functional groups.

Table 4.11 Macroinvertebrate functional feeding groups (Williams and Feltmate 1992).

Functional group	Subdivision based on feeding mechanism	Dominant food	Examples
Shredders	Chewers and miners	Herbivore	Trichoptera (Phryganeidae)
	Chewers and miners	Detritivore	Trichoptera (Limnephilidae)
Collectors	Filter or suspension feeders	Herbivore	Diptera (Simuliidae)
	Sediment or deposit (surface) feeders	Detritivore	Ephemeroptera (Caenidae)
Scrapers	Mineral scrapers	Herbivore	Ephemeroptera (Baetidae)
	Organic scrapers	Herbivore	Hemiptera (Corixidae)
Predators	Swallowers	Carnivore	Odonata
	Piercers	Carnivore	Hemiptera (Nepidae)

Table 4.12 The inter-relationships of biotic indices used in Europe and North America



Community based methods for assessing general ecosystem quality

In the United States invertebrate-based multimetric methods are widely used in stream assessment, based on a wide range of metrics such as: indicator groups, relative abundance of different individuals, percentage of tolerant groups etc. (Hayslip 1992; Ohio EPA 1987; Yoder and Rankin 1995). Multi-metric indices often draw on a variety of metrics from those listed in table 4.10.

Lake benthic metrics that are responsive to stresses are in general similar to stream invertebrate metrics (although clearly the precise taxa may vary). For example, the Tennessee Valley Authority has developed multimetric indices for reservoirs by (Cox 1994) which include the following metrics which have been demonstrated to be correlated with stresses:

- taxa richness
- number of long lived taxa (e.g. *Corbicula*, *Hexagenia*, mussels, snails)
- proportion of Chironomidae
- proportion of Tubificidae
- proportion as dominant taxa

Similarly, invertebrate metrics demonstrated to respond to stresses in Florida lakes include:

- taxa richness
- Shannon-Wiener diversity
- percent shredders
- percent suspension feeders
- Number of ETO taxa (Ephemeroptera, Tricoptera, Odonata) (EPA 1994).

The US EPA proposed Lake and Reservoir bioassessment programme has also put forward a similar set of potential metrics for macroinvertebrates for further testing. These are listed in Table 4.13.

Table 4.13 Lake and reservoir benthic metrics proposed by the US EPA

Proposed metric	**Response**
Number of taxa	reduced
Shannon-Wiener diversity	reduced
Mean number of individuals per taxon	substantially lower *or* higher
% contribution of dominant ton	elevated
% intolerant species	reduced
% oligochaetes	elevated under organic enrichment
ETO taxa	reduced under enrichment, D.O. stress
% non-insects	reduced
Crustacean + Mollusc taxa	reduced under acid low calcium stress
% Crustaceans and Molluscs	reduced under acid low calcium stress
Tolerance indices	reduced
% suspension feeders	reduced
% shredders	reduced under enrichment or in very large lakes

European multimetric methods. In Europe, the AMOEBA model, developed in the Netherlands (see Section 4.2) can include targets relating to specific still water taxa. Proposed river targets for examples include two Diptera families, a caddis family and a mayfly family (ten Brink *et al.* 1991).

As noted in Section 4.4, SERCON (System for Evaluating River Conservation) includes a number of measures of the biotic integrity of macroinvertebrate communities and is analogous to a true multimetric index. SERCON uses present-day best available sites and professional consensus to derive baseline conditions. Measures used in the index have not, however, been tested statistically.

Individual-based bioassessment methods

Laboratory based bioassays include assessments based on:

(i) behavioural impairment (Rand 1985),

(ii) enzyme activity in aquatic insects (mayflies, caddis, amphipods, bivalves) in response to effect of metals and biocides (Farris *et al.* 1988, 1989; Day and Scott 1990),

(iii) ion regulation: e.g. effect of acidification on crayfish and bivalves (Malley *et al.* 1988, Havas and Hutchinson 1983, Lechleitner *et al.* 1985),

(iv) energy metabolism in bivalves in response to heavy metals (Geisy *et al.* 1983),

(v) respiratory metabolism in a variety of benthic macroinvertebrates for a wide range of toxicants (Correa and Coler 1983, Darville and Wilhm 1984, Rockwood *et al.* 1990).

It has been suggested that a number of these assays (e.g. ion regulation, energy metabolism) could be applied to field situations for use as early warning systems for example. However most methods need considerably more testing and calibration before they could be used routinely.

Field based assays which are potentially far more useful for diagnosis and early warning monitoring include:

(i) morphological deformities, particularly in chronomid head and mouthparts in response to industrial and agricultural pollutants (Hamilton and Saether 1971).

(ii) behavioural impairment in flowing water species e.g. net spinning caddis, preference choices in stoneflies and movement and valve closure in molluscs (Schere and McNicol 1986, Petersen and Petersen 1984).

Bioaccumulation of metals (e.g. zinc, cadmium) in body tissues has been investigated in several taxa focusing on Oligochaeta, Mollusca, Crustacea and Chironomidae.

4.10.3 The potential suitability of macroinvertebrates for water quality monitoring

Advantages and disadvantages

The main advantages of macroinvertebrates for bioassessment are their ubiquity and diversity, the relative ease with which they can be sampled, and the wide range of interactions they have with other assemblages. Macroinvertebrate richness is related to plant species richness in ponds and probably also in other still water habitats. Macroinvertebrates can be collected at any time of the year (except in seasonal waters).

As indicators of environmental stress they are advantageous in that they are relatively long-lived and can integrate trends which occur over relatively long time periods. River invertebrates are often noted as integrating trends at sites, but it is not clear whether this would also be true of still water invertebrates.

Monitoring programmes using macroinvertebrates require relatively inexpensive equipment. Samples can be collected quite quickly in littoral habitats although they are more time consuming to collect from profundal habitats. Sample processing and identification time depends on the level of analysis; samples analysed to family or higher taxa level are relatively quick (overall taking no longer than collecting monthly chlorophyll a samples, or example). Intermediate levels of analysis (e.g. mixing species and higher taxa identification) is already undertaken by EA staff. Staff training for macroinvertebrate methods would largely overlap with the existing programme of river monitoring.

There are two main disadvantages associated with using macroinvertebrate for biomonitoring in still waters:

(i) Macroinvertebrates are mainly indicators of environmental stress and biodiversity in littoral zones. They are probably most representative of small water bodies which are primarily littoral habitats. Macroinvertebrates are probably less useful as indicators of environmental stress in open water habitats.

(ii) There are considerable season-season variations in the taxa likely to be recorded. Sampling the littoral zone may also be difficult beyond easily wadeable depths. Samples processed and analysed to identify species are time consuming and require considerable expertise.

In addition, although the taxonomy of macroinvertebrates is relatively well understood it is less well developed than the taxonomy of higher plant and vertebrates (e.g. fish, birds). Metrics describing the effects of environmental stress on macroinvertebrates are yet not well-developed in standing waters. This may constrain the development of effective macroinvertebrate based methods.

The main advantages and disadvantages of macroinvertebrates for bioassessment programme are listed in Table 4.14.

Evaluation of operational feasibility

Macroinvertebrates are probably the best indicator assemblage for monitoring *general ecosystem quality*, if balanced with a plant assemblage and probably fish. They can be found in all habitat types, are fairly easy to sample, are diverse enough to detect many impacts and not too obscure taxonomically.

Although relatively little developed at present for practical monitoring programmes, macroinvertebrates may also be useful for diagnosis and early warning.

Table 4.14 Advantages and disadvantages of macroinvertebrates for bioassessment

1. Macroinvertebrates are ubiquitous, occurring in all types of standing water	1. Quantitative sampling requires large numbers of samples because of spatial variability of benthic invertebrates
2. Macroinvertebrates represent a large proportion of the biodiversity of standing waters.	2 Factors other than water quality can affect the distribution of macroinvertebrates
3. Large numbers of species offer a spectrum of responses to perturbations	3. Seasonal variation may complicate interpretations or comparisons
4. The mainly sedentary nature of macroinvertebrates allows spatial effects of perturbations to be assessed	4. Some macroinvertebrate groups are not well-known taxonomically (e.g. chironomids)
5. The relatively long life cycles of macroinvertebrates allow effects of regular intermittent perturbations to be assessed	5. Benthic macroinvertebrates are not necessarily sensitive to all perturbations
6. Qualitative sampling techniques are well developed and can be done quickly using inexpensive equipment	6. Littoral sampling may be difficult (especially where water is too deep to wade)
7. The taxonomy of most groups is well known and there are good identification guides	7. Metrics are not well-developed or tested in lakes
8. Many methods of data analysis have been developed for macroinvertebrates	8. Most macroinvertebrates are unknown to ordinary members of the public and of little direct economic significance
9. Responses of many common species to some pollutants are reasonably well known	
10. Macroinvertebrates are reasonably known to members of the public with elementary experience of biology	
11. Macrophytes have trophic links to fish and birds, and therefore of interest to many members of the public	

4.11 Fish assemblage methods

4.11.1 The importance of fish

There are 42 species of freshwater fish which are regarded as indigenous to the British Isles (2 now extinct and 8 regarded as threatened), and 13 introduced species which have become established in the wild (Maitland and Campbell 1992, Maitland and Lyle 1992). Five native species are rare and are given special legal protection.

The national distribution and ecological characteristics of fish are well well-known, and life history information for most species is extensive. There is sufficient understanding of the taxonomy of fish for the preservation of genetically distinct populations of salmonids to be of concern in some areas (Ferguson 1989). Although the broad patterns of distribution of fish are well-known, there are relatively few still waters where detailed information on the whole fish community is available (Maitland and Lyle 1992).

Fish are of considerable public interest and economic importance, and recent estimates indicate that about 10% of households in England and Wales have members who participate in angling. In addition, fish are perhaps the most powerful symbol of the health of the aquatic environment.

4.11.2 Existing assessment methods using fish

Community-based methods for the assessment of specific stresses

Because of the considerable economic importance of fish there is an extensive body of data describing the effects of environmental stressors on fish populations. However, in Europe, fish assemblage data has mainly been used *specifically* to assess the condition of fisheries, and of fish populations, and has not normally been used to assess *general* environmental quality. In contrast, North American biologists have, since the early 1980s, made extensive use of fish population data in the monitoring of environmental degradation, using fish as *indicators* of pollution and habitat degradation, as well as monitoring the status of the populations as a fishery.

The use of fish for environmental monitoring is, however, a relatively recent development and the great majority of fish survey work has been focused on the measurement of population parameters important for angling and fisheries management. A wide variety of techniques are used to collect data of this type, with the precise methods chosen (e.g. traps, seine nets, gill nets, electrofishing, poisoning) mainly dependent on the physical characteristics of the water body being sampled.

All sampling techniques are affected by gear selectivity and fish mobility and considerable care must be taken to ensure that methods are comparable in different water body types and regions. In North America the EMAP Surface Water Northeast Lake Pilot Survey found electrofishing the single most effective single-gear technique (EPA 1994).

Since the focus of fisheries monitoring programmes has usually been stock assessment, individual fish species have not (until recently) been identified as indicators of specific conditions, and fish populations have not been used as the basis for biotic indices. However, standard fishery data contains all the necessary information to make these observations and it was with fishery data that Karr (1981) first established the concept of multimetric methods.

Methods for assessing general ecosystem quality

Fish assemblage data, used to assess the general quality of running waters, formed the basis of the original Index of Biotic Integrity (IBI) developed in the early 1980s (see Table 4.1). Regionally-based fish IBIs are now used routinely for *running waters* in the US and a European version of this index has been proposed by Oberdorff and Hughes (1992) for use in the Seine basin in France. The French IBI is based on three groups of metrics: (i) species richness and composition, (ii) trophic composition, and (iii) fish health and abundance (see Table 4.15).

In addition to the metrics above which might be described as 'biological', North America biologists and fisheries managers also explicitly consider 'fishability' criteria, including a variety of metrics which are used to address questions such, 'are there any large game fish?', 'do fish look edible?' and 'are fish safe to eat?'.

Table 4.15 Potential fish metrics for biological integrity assessment (Oberdorff and Hughes 1992)

Species richness and composition

1. Total number of species
2. Number of water column species
3. Number of benthic species
4. Number of intolerant species
5. % individuals as roach
6. Trout or pike year classes

Trophic composition

7. % of individuals as omnivores
8. % of individuals as 'invertivores'
9. % of individuals as top carnivores

Fish health and abundance

10. % of individuals as gravel spawners
11. % of individuals with anomalies (disease, tumours, fin damage)
12. Catch per minute of sampling

Individual-based bioassessment methods

In contrast to studies of fish assemblages, studies at the level of the individual have often been concerned with identifying ways in which fish can be used as indicator or sentinel organisms. Many of these techniques have potential for inclusion in multimetric indices.

Techniques which can be used to describe the condition of fish at the level of the individual can be broadly grouped into three categories: (i) biomarker studies (ii) bioaccumulation studies and (iii) bioassays, although the precise use of these terms is often inconsistent.

Biomarkers

Biomarkers are biological responses (normally at the sub-organism level) to xenobiotic exposure and can be any biochemical, histological and/or physiological alterations or manifestations of stress (Holdway *et al.* 1995). They can be based on observations of live animals or autopsy material, and include measures of stress proteins, liver enzymes, changes in blood chemistry, DNA adducts, DNA strand breakage, metallothionein levels, lipid peroxidation, endocrine and secondary stress responses, hind gut inflammation, damage to extremities (such as eyes, gills and pseudobranchs), mesenteric (visceral) fat deposits, spleen condition and state of maturity (Goede and Barton 1990).

An example of the way in which biomarkers may be developed is provided by the index of gill damage, *I*, proposed by Poleksic and Mitrovic-Tutundzic (1994) (see Table 4.16). Using

natural populations of barbel and chub and cultured mirror carp, these authors identify three stages in severity of gill damage, based on 26 different types of gill lesion. These were: (i) changes that can be repaired when water quality conditions improve (ii) changes which may be reversible but which can affect associated tissue function (iii) irreversible changes which lead to permanent gill damage, or mortality. Most changes are in the first category.
The index of gill damage proposed by Poleksic and Mitrovic-Tutundzic (1994) has the following *I* values: 0-10, functionally normal gills, 11-20 slightly to moderately damaged, 21-50, moderately to heavily damaged and >100 irreparable damage. There was no category 51-100 (no values were observed in this range in the trial) which, the authors argued, indicated the exponential nature of gill damage.

Poleksic and Mitrovic-Tutundzic (1994) recommended that:

(i) a minimum of 10 fish per location should be analysed and the mean value of *I* calculated,

(ii) the types of gill structure must always be uniformly assessed,

(iii) the values obtained for I should be presented together with the results of other methods of assessment,

(iv) the mode of life and type of feeding of the fish must be considered.

Table 4.16 Cyprinid gill lesions: measures used to assess index of gill damage, *I* (Poleksic and Mitrovic-Tutundzic 1994)

Stage 1: changes that can be repaired when water quality conditions improve

- Hypertrophy of respiratory epithelium
- Lifting of respiratory epithelial cells
- Leukocyte infiltration of gill epithelium
- Thinning of respiratory epithelium
- Focal hyperplasia of epithelial cells
- Hyperplasia from the base to approximately half of the length of the secondary epithelia
- Irregular (chaotic) hyperplasia of epithelial cells
- Fusion of the tips of secondary lamellae
- Fusion of several secondary lamellae
- Hypertrophy and hyperplasia of mucous cells
- Empty mucous cells or their disappearance
- Hypertrophy and hyperplasia of chloride cells
- Chloride cells present in secondary lamellae
- Lamellar telangiectasis
- Filament blood vessel enlargement
- Gill parasites

Stage 2: changes which may be reversible but which can affect associated tissue function

- Rupture and peeling of the lamellar epithelium
- Uncontrolled thickening of proliferated tissue
- Complete fusion of the secondary lamellae
- Haemorrhages with rupture of epithelium
- Stasis

Stage 3: irreversible changes which lead to permanent gill damage, or mortality

- Scar tissue - fibrosis
- Necrosis

Bioaccumulation studies
Bioaccumulation studies measure the levels of accumulated toxins, particularly heavy metals, in fish tissue. In contrast to other vertebrates (e.g. birds of prey), bioaccumulation has not been seen as a cause of major population declines in fish, and this assessment method has usually been applied in response to concerns over human health. However, body burdens of heavy metals also have the potential to be relevant to species feeding at upper trophic levels where bioconcentration is a risk.

Bioassays.
Unlike most individual condition-related assessments, which are largely based on general fish health, bioassays are more frequently used as measures of specific environmental pollutants. Fish have, for example, been extensively advocated as *in situ* biomonitors to assess fluctuating dissolved oxygen concentration, discharges from storm overflows (Seager *et al.* 1994) and a variety of specific toxicants (ammonia, phenol, paraquat, 2,4,6-Trichlorophenol, diesel oil) at concentrations lower than their respective LC_{50} values (Evans and Wallwork 1988). Biomonitors of this type have been installed in the UK (Evans and Wallwork 1988), South Africa (Morgan and Kuhn 1988) and Australia (Harris 1995), mainly for monitoring intake water quality at water treatment plants.

4.11.3 The potential suitability of fish for water quality monitoring

Advantages and disadvantages of fish populations for general quality monitoring

Advantages. Fish are capable of inhabiting most types of standing water, except seasonal water bodies which do not have intermittent connections to other waters, and have measurable responses to most types of environmental stress.

Fish are good 'integrators' of environmental conditions: they have relatively large natural ranges, so are less affected by small scale variations in habitat (Simon and Lyons 1995). They occur at all trophic levels and feed in a wide range of habitats. They are also present in the water at all times, providing good temporal integration of the physical, chemical and biological histories of water bodies.

Fish populations respond to many major environmental stressors, including acidification, deoxygenation, heavy metals, biocides (pesticides, herbicides etc.), changes in water regime, heat, physical damage/manipulation (e.g. modifying banks). Short-term responses to some stressors are known, (primarily in terms of fish physiology) as well as longer term trends. Fish may have the potential to function as early warning systems.

Disadvantages. Sampling fish requires relatively large teams of survey staff and equipment used is the most expensive of any employed for biological work. Equipment needs to be carefully calibrated if comparisons between water bodies are to be valid.

Fish represent a small component of total biodiversity and changes in fish stocks do not necessarily indicate changes in other plant and animal assemblages. Although fish occur at all seasons of the year, fisheries biologists often prefer not to sample during some seasons for reasons of animal welfare.

Fish may also be absent naturally from heavily shaded, acid and seasonal sites, and present at low diversity in very shallow waters (including late succession sites). There are also problems with establishing baseline conditions due to extensive modification of natural fish populations as a result of fisheries management and manipulation practices.

Evaluation of the operational feasibility of fish bioassessment methods

Fish assessment methods are well developed, and although barely used as a method for evaluating water quality in Europe, there is considerable potential for their application in all forms of assessment. Fish populations offer a number of advantages for general surveillance monitoring, in addition to their ecological relevance (see above and Chapter 5).
It is likely that the parameters needed for multimetric assessments could largely be derived

from data already collected in the UK. Potential environmental quality indicators such as biomass (g m^{-2}), density (ind. m^{-2}), species composition, growth rates and parasite loads, for example, all form part of current regular or routine assessments.

Biomarker and bioaccumulation methods may be of value as an early warning system as well as a diagnostic tool. In general, further experimental development of these techniques is generally needed before they can be routinely implemented.

Table 4.17 Advantages and disadvantages of fish assemblages for bioassessment (Simon and Lyons 1995, Harris 1995)

Advantages	Disadvantages
1. Fish communities represent various trophic classes.	1. Biotic integrity indices are not available for UK fish populations
2. Both acute toxicity (missing fish) and stress effects (depressed growth or reproductive effects) can be evaluated.	2. Sampling is time consuming and relatively expensive.
3. Fish are primarily influenced by 'macro-environmental factors' unlike algae and macroinvertebrates which respond to 'micro-' and macroenvironmental factors.	3. Fish populations have high spatial variability.
4. Communities are persistent and recover rapidly from natural disturbances.	4. Fish may be naturally absent from acid, seasonal, heavily shaded and late succession (i.e. organic rich) waters.
5. Fish have large ranges and are less affected by small-scale habitat variations.	5. Baseline and site assessment problems caused by intensive stocking and angling.
6. Most fish have long life spans (2-10+ years) and can reflect both short and long term change.	6. Performance of sampling gear varies greatly and is 'gear-specific'.
7. Fish continually inhabit the water so integrate stressors throughout year.	7. Sampling impact on fish and other communities.
8. Fish species have a broad range of tolerances.	
9. Fish are highly visible and well-known to public.	
10. Sampling frequency for trend assessment is lower than for other organisms.	
11. Taxonomy is well-known enabling many specimens to be identified in the field.	
12. Distribution is well known.	
13. Life history data is readily available.	

4.12 Amphibians

4.12.1 The importance of amphibians

Amphibians represent a small proportion of aquatic biodiversity in Britain with only six native species (a seventh, the pool frog, *Rana lessonae*, may also be native) (Snell 1994). Four of these species are common and widespread; the great crested newt is widespread, but local and the natterjack toad rare, being confined to about 20 sites. The pool frog is known from one site. Amphibians are conspicuous and familiar animals and approach garden birds in popularity with members of the public. However, the protection of amphibian populations has traditionally been seen as a conservation issue and has mainly been undertaken by the statutory nature conservation agencies, working with voluntary sector organisations.

4.12.2 Existing amphibian survey methods

Amphibian are not routinely used for monitoring of water quality in Europe or North America. This almost certainly reflects their relatively low species-richness and, the fact that amphibian communities are largely restricted to ponds (their main breeding habitat). The assessment below therefore considers the potential for use of amphibian communities for assessing water quality with particular reference to ponds.

Amphibians as indicators of water quality and environmental stress

It has been hypothesised by some herpetologists that amphibians are suffering a world-wide decline which in some way (currently unspecified) differs from the declines being seen in many other taxa due to habitat loss and degradation (Hedges 1993, Fellers and Drost 1993, Pound and Crump 1994, Pechman and Wilbur 1994). This has contributed to the view that amphibians may be unusually sensitive indicators of environmental stress but to date no evidence has been presented to substantiate the hypothesis (Pechmann and Wilbur 1994). In practice, there is relatively little information suggesting strong relationship between amphibians and water quality.

The clearest indication of water quality stress affecting amphibians is seen with acidification, amphibians (particularly eggs and young larvae) being sensitive to elevated H^+ concentrations (Sparling 1995). However, there is little evidence that amphibian populations are particularly sensitive to 'normal' levels of other common pollutants (such as organics, biocides, heavy metals) which affect populations of other aquatic organisms. In part this may be due to lack of research in this area, but recent investigations into the relationships between amphibians and nitrate levels, by R.S. Oldham and colleagues at DeMontfort University, have shown little evidence of significant damage even a high field concentrations (R.S. Oldham pers comm.). Indeed, the most widely reported impacts on amphibian communities are loss of breeding ponds and successional changes which make ponds less suitable for amphibians (Swan and Oldham 1993).

More speculatively, there is some evidence that amphibians are sensitive to ultra-violet radiation (Blaustein *et al.* 1994) although whether effects observed in the laboratory are significant in the field is not yet clear. In addition, Beebee (1995) has suggested that amphibian populations may be an indicator of global climate change. He has demonstrated that populations of common amphibians in the UK have been breeding increasingly early in the last 20 years.

Survey methods used

Most studies of amphibian populations use a suite of relatively simple methods intended either simply to record the presence of the animals at breeding sites or to measure population sizes. Most of this work is done at the breeding sites, there being relatively little information about amphibians in the terrestrial habitats. A number of surveys have recorded species richness and community variables for amphibians. However, there have been few measures of the functional components of amphibian populations.

Amphibians are normally surveyed during the breeding season when adults, eggs or larvae can be easily found at the breeding sites. Standard survey methods have been recommended by Swan and Oldham (1992), and used in a UK National Amphibian Survey. Survey methods generally involve a combination of counts of eggs, collection of larvae or counts of adults as they gather at the breeding ponds. The most accurate estimates of numbers are derived from interception of the adults as they move towards breeding ponds. Adults newts may also be trapped for counting. Individual adult amphibians are often marked or identified individually by distinctive skin patterns and there have been some experiments with radio tracking.

4.12.3 Potential suitability of amphibian survey methods for bioassessment

Advantages and disadvantages of amphibians population for general quality monitoring

The principal advantages of amphibian survey methods are the speed and relatively low cost of the simpler survey methods, the simplicity of amphibian taxonomy (although not all larval stages can be reliably separated) and their familiarity to members of the general public.

However, amphibian population parameters do not appear to be particularly well correlated with biodiversity of other aquatic groups and their relationships to environmental stresses appear to be either rather weak or poorly known. In addition, many water bodies naturally lack amphibians (often for reasons that are unclear).

Evaluation of the operational feasibility of amphibian survey methods

The advantages and disadvantages of using amphibians for bioassessment are summarised in Table 4.18. Overall, however, amphibians are relatively poor indicator of the integrity of aquatic ecosystems, partly because there are relatively few species and partly because much of the life cycle is spent out of the water. In addition, there is currently little information about how the way in which amphibian population parameters relate to other aspects of biotic integrity (e.g. is amphibian species richness related to the richness of other assemblages).

Table 4.18 The advantages and disadvantages of amphibians-based methods for bioassessment

Advantages.	Disadvantages.
1. Amphibians are well-known taxonomically and can easily be identified in the except (eggs and larvae of newts).	1. There are very few amphibian species.
2. Amphibians are well-known to the public and, in many ways, as symbolic of the health of freshwaters as fish.	2. Little is known about the factors affecting amphibian populations.
3. The semi-aquatic life histories of amphibians emphasises the inter-relatedness of land and water habitats.	3. Many water bodies do not have amphibians and they are not very well represented in large lakes or temporary ponds.
4. Basic presence/absence sampling is quick and cheap to obtain. Quantitative data, although time consuming to collect, is still relatively inexpensive.	4. Survey methods can be rather labour intensive; night-time sampling is often required.
5. Amphibian larvae integrate more than one trophic level.	5. Amphibians do not integrate annual water quality since they are only in the water for a few months at most.
6. Amphibians are relatively long-lived so may integrate broader changes in the catchments of water bodies.	6. Sampling is more-or-less confined to the spring and early summer.
	7. Abundance estimates can only be made at this time of the year.

4.13 Birds

4.13.1 The importance of birds

Birds are equal in popularity to fish and information about their population sizes and ecology is probably more extensive. The national distribution data is the best available for any group of plants or animals associated with freshwater and population sizes can be estimated for most species (Gibbons *et al.* 1993). Many important bird populations, particularly of waterfowl, are associated with standing waters, especially large lowland lakes and reservoirs.

There is a very wide range of life history information available about birds and the taxonomy of the group is highly developed (all stages from egg to adult can be identified in virtually all species). Birds receive a high degree of legal protection, all but pest species being protected by law. Special protection is given to 22 species which are associated with freshwater wetland habitats (RSPB, RSNC, NRA 1994).

4.13.2 Existing bird survey methods

Birds communities as indicators of environmental stress

As with other higher vertebrates, surveys of bird populations are primarily intended to provide information about specific bird communities or species, rather than general environmental degradation.

However, bird assemblages associated with aquatic ecosystems do show clear relationships with certain types of degradation. For example, there is some indication that bird communities respond to acidification, to changes in vegetation structure in individual water bodies, to lead abundance (lead shot and fishing weight) and possibly to fish abundance (Ormerod and Tyler 1993). Relationships with other environmental impacts, such as eutrophication and physical habitat damage, might be expected although they have not yet been clearly demonstrated.

Bird survey techniques fall into two broad categories:

(i) those concerned with studies of communities and assemblages,

(ii) those concerned with changes in the populations of individual species (e.g. studies of herons, dippers, grey wagtails).

Most relevant for general quality assessment are standard census schemes which provide information on the bird populations of individual sites. For example (i) counts of overwintering waterfowl (ii) counts of numbers of pairs of breeding aquatic species (e.g. great crested grebe, coot) and (iii) counts of numbers of breeding pairs of riparian species may all be used to assess the general status of bird populations. Standard count techniques for waterfowl have been developed for the UK National Wildfowl Counts scheme in which regular winter counts of birds are undertaken at between 1500 sites annually (Owen *et al.* 1986).

Breeding populations of individual bird species can be counted using a variety of techniques including direct observation of nest sites (e.g. moorhen) and counts of territorial males (e.g. shoveler, some waders) (Bibby *et al.* 1992). Riparian bird populations can be counted using habitat mapping techniques used for the Birds of Waterways Survey (RSPB, RSNC, NRA 1994).

Multimetric methods using bird assemblage data

There are currently no multimetric techniques used to describe the biotic integrity of bird populations. However, the US EPA EMAP programme is currently developing riparian bird survey techniques for use in the lake monitoring programme which will use multimetric criteria (EPA 1995).

Bioassay approaches: environmental stresses indicated by individual bird species

Many aquatic birds might be expected to be vulnerable to toxins (pesticides, heavy metals, PCBs etc.) and studies have shown that some species do carry significant body burdens of these substances. However, trends in contaminant burdens have not been so closely connected with breeding success or population sizes in aquatic birds as in birds of prey (Ormerod and Tyler 1993). Standard laboratory bioassay techniques are used for analysis of body tissues.

4.13.3 The potential suitability of bird assemblages for biological assessment

Advantages and disadvantages of bird populations for general quality monitoring

The principal advantages of bird survey methods for general quality assessment are the ease of obtaining data (particularly compared to mammals), the straightforward taxonomy of the group and the ability of birds assemblages to integrate stresses over time and space. Bird assemblages may be especially useful as indicators of physical habitat quality.

However, birds are very mobile and populations sizes can change rapidly for a variety of reasons unrelated to environmental quality (such as cold weather migration). In addition, the effects of many common environmental stresses on bird populations are surprisingly poorly understood (e.g. nutrient enrichment). Since many species are only partially dependent on aquatic habitats they may reflect the quality of surroundings as much as the quality of waterbodies. Many small, temporary water bodies and acid upland waterbodies have few or no birds. Populations are also very different at different times of the year so that surveys can only be undertaken over quite short periods (effectively March-May for breeding birds and November-February for overwintering species).

Evaluation of the operational feasibility of bio-assessment methods using birds

Birds are an important assemblage in which there is great public interest. To date, however, their potential for demonstrating general environmental quality has been little explored. However, apart from physical habitat quality, bird assemblages may be useful indicators of general environmental quality.

4.14 Mammals

4.14.1 The importance of mammals

Three native species of mammals, and one introduced species, are closely associated with water in the UK: the otter, water vole, water shrew and mink. In addition several species of bats are also strongly associated with freshwater ecosystems, including still waters. The taxonomy of mammals is well known but their cryptic nature means that distributions are poorly known. There is only limited information about population sizes.

Mammals (apart from bats) are popular with members of the public and water voles are relatively easily observed. The specially protected otter is closely identified with the need to protect freshwater habitats. It is also widely perceived by many freshwater biologists as an important symbol of river conservation.

4.14.2 Existing mammal survey methods

Mammals as indicators of environmental stress

None of the native mammal species are currently used to assess the status of still water habitats although all undoubtedly use them. At present the most comprehensive programmes of survey work are undertaken for the otter, which is widely seen as indicative of general river quality (and perhaps indirectly the quality of other habitats). In fact, otters are probably mainly responding to levels of pesticides and other toxins (PCBs, heavy metals) still present in the environment.

In general mammals are not good indicators of environmental stress because too little is understood about their relationships with pollutants and habitat damage.

Mammals are normally cryptic and surveys frequently rely on indirect methods of observation, only the most intensive studies involving direct observation of individual animals. Mammals are most commonly surveyed by observing signs (e.g. otter sprints and tracks), by live trapping (e.g. water shrews, mink) and by inspection of dead animals (road casualties, for example). Radio tracking is often used in the most intensive studies. Relatively standardised survey techniques are available only for the otter, but they are primarily intended for use on rivers.

4.14.3 Potential for development of bio-assessment methods using mammals assemblages

Advantages and disadvantages of mammals for bioassessment

In some respects mammals are very good indicators of general environmental quality because they integrate across water bodies and time. Like other higher vertebrates, some species (e.g. otter) are especially vulnerable to the bioaccumulation of persistent environmental toxins, such as heavy metals. However, the practical disadvantage associated with surveying mammal assemblages (such as difficulty of observing animals) are probably too great to make them suitable for general quality assessment.

The main advantages of using mammal assemblages for bioassessment are:

(i) They are well-known taxonomically and there is detailed life-history data available,

(ii) Mammals are probably good indicators of the integrity of ecosystems because they cover a wide range of trophic levels and are found in very wide range of habitats (potentially all still waters in an area may be used),

(iii) Otter (and mink) may integrate stresses over large areas,

(iv) Mammals are 'high profile' animals, even though rarely seen by members of the public,

(v) Mammals are tolerant of a wide range of conditions, so can be used to asses habitats in different regions.

The main disadvantages of mammal assemblages for bioassessment are that:

(i) Mammals are difficult to survey and field methods require extensive staff training,

(ii) There are too few species for a wide variety of stresses to be integrated,

(iii) They do not integrate features of water bodies very well because they are semi-aquatic and may be away from waterbodies for long periods,

(iv) Biotic integrity indices have not been developed.

Evaluation of operation feasibility

Surveillance and compliance monitoring. Although popular and high profile animals, mammals are not suitable for general surveillance and compliance monitoring. This is mainly because (i) they are semi-aquatic, so may be reflecting broader environmental trends (ii) they are difficult to survey (ii) their responses to many environmental stresses are poorly understood.

Diagnosis and early warning surveys. Levels of bioaccumulating toxins may be an important early warning indicator of ecosystem degradation. Tissue analysis, particularly of fish eating mammals, may provide important information which is difficult to obtain by any other method.

4.15 Physical habitat assessment methods

4.15.1 Introduction

Physical habitat diversity plays an important role in shaping plant and animal assemblages in freshwater ecosystems, particularly running waters. Extensive removal of natural features from rivers has been widely recognised as damaging, affecting all levels of the biota and altering ecosystem function. The importance of physical habitat has led to the development of a number of habitat assessment methods which provide an objective description of the physical diversity of freshwater ecosystems. They have been extensively developed for use with rivers, and are particularly intended to assess the extent of degradation due to channel and catchment management (Rankin 1995). To date, physical habitat indices have not been extensively developed for still water habitats but work is currently in progress in the US EPA to develop a series of measures for the EMAP Lake Bioassessment Programme.

4.15.2 Existing habitat-based survey methods

Habitat assessment methods as indicators of environmental stress

Habitat assessment methods are concerned with describing two types of variable:

(i) classification variables, which are those features that are an intrinsic part of the system and relatively unaffected by human activity (such as geology, lake morphology),

(ii) assessment variables which may be influenced by human activity (e.g. shoreline urbanisation, abundance of emergent vegetation).

In the United States a variety of habitat indices are available, usually relating to specific regions of the country (Rankin 1995). Table 4.19 lists the features recorded in a typical index the Qualitative Habitat Evaluation Index (QHEI) used in Ohio (Rankin 1995). This index has been subject to fairly intensive testing with the Index of Biotic Integrity.

Habitat indices are less extensively used in Europe, although in England and Wales the NRA has made extensive use of the River Corridor Survey technique which includes both physical and biological features. RCS was originally conceived as a management tool intended to provide land drainage engineers with guidance about which channel features should be retained during river management operations. Like the North American indices RCS is based conceptually on the general relationship between the variety of physical habitats and biological diversity (particularly species richness). In the RCS physical features of the river (channel, banks, trees, distribution of main vegetation stands) are sketched onto a map and observation made about uncommon species and other biota of interest (NRA 1992a). RCS is a relatively informal technique and has not been subject to detailed testing. There is no overall assessment index used with the method and no analysis of critical parameters, resulting in much redundancy of data (NRA 1992b).

The EA is currently developing a new River Habitat Survey technique which has more in common with North American habitat indices. It is based on a classification of stream channel features and seems likely to supersede the RCS in due course as a survey technique.

The River, Channel, Environmental Inventory proposed by Petersen is similar to RCS and RHS but also has some similarities with a multimetric technique (Petersen 1992). See Section 4.4.4 for a description of the RCE methodology.

4.15.3 Potential suitability of habitat-based survey methods for bioassessment

In general habitat assessment methods are relatively poor indicators of overall environmental stress because they provide little or no information about stresses due to chemical impacts, particularly eutrophication, acidification (unless plants are included), organic pollution and biocides.

Table 4.19 Qualitative Habitat Evaluation Index (QHEI) (Rankin 1995)

I. Substrate quality
- a. Two most predominant substrate types
- b. Number of substrate types
- c. Substrate origin (tills, limestone, etc.)
- d. Extensiveness of substrate embeddedness (entire reach)
- e. Extensiveness of silt cover (entire reach)

II. Instream cover
- a. Presence of each type in the reach
- b. Extensiveness of all cover in reach

III. Channel quality
- a. Functional sinuosity of channel
- b. Degree of pool/riffle development
- c. Age/effect of stream channel modifications
- d. Stability of stream channel

IV. Riparian quality/bank erosion
- a. Width of intact riparian vegetation
- b. Types of adjacent landuse
- c. Extensiveness of bank erosion/false banks

V. Pool/riffle quality
- a. Maximum pool/glide depth
- b. Pool/riffle morphology
- c. Presence of current types
- d. Average/ maximum riffle/run depth
- e. Stability of riffle/run substrates
- f. Embeddedness of riffle/run substrates

VI. Local stream gradient (ft/mi) from 7.5' topographic map

Note: Habitat attributes are visually estimated over a 150 to 500-m reach that corresponds to a biological sampling reach.

Advantages

The main advantages of habitat-based methods for bioassessment are:

(i) They are quick and, therefore, relatively cheap. Staff training costs are also fairly modest.

(ii) They integrate the effects of physical stress throughout the year.

(iii) Sampling frequency can be low because changes in physical habitat occur fairly slowly.

Disadvantages

The main disadvantages of habitat based methods are:

(i) They contain little or no biological information so rely entirely on the relationship between physical diversity and other aspects of biodiversity. Although this relationship is broadly true, testing of habitat based methods has mainly been confined to relationships with species richness. Some sensitive or rare species may be associated with rather damaged sites, and only by detailed survey can these aspects of biodiversity be assessed adequately.

(ii) Relationships with biotic integrity not well understood.

(iii) Habitat surveys can be difficult to replicate (although they are perhaps not notably worse than most biological techniques in this respect).

Overall further development would be needed if habitat-based methods were to be used as a general method for assessing the biological integrity of still waters. However, description of certain aspects of the physical integrity of still waters is likely to be highly desirable.

4.16 Rapid screening tests

4.16.1 Introduction

Screening tests are a special category of ecotoxicological test used in the laboratory or field which can quickly provide a rough indication of the relative toxicity of effluents, or receiving waters, to biological systems.

Screening tests contrast with the two other standard types of ecotoxicological test:

(i) Confirmatory toxicity tests which enable a dose-response curve to be derived and key ecotoxicological parameters to be measured, such as LC_{50}, EC_{50}, LOEL (lowest observed effect level) or NOEL (no observed effect level).

(ii) Model aquatic ecosystem tests, using microcosms, limnocorrals, littoral enclosures or pond mesocosms to enable the effects of toxicants to be assessed in a more realistic setting (Kennedy *et al.* 1995).

This section considers screening tests which are likely to be of most relevance to GQA procedures. Confirmatory toxicity tests and the role of model ecosystems are considered, where relevant, in the previous chapters dealing with specific assemblages.

4.16.2 Rapid screening tests: methods

The great majority of ecotoxicological tests have been developed for laboratory testing of toxicity under controlled conditions. However, a number of screening tests are being introduced that are intended to provide a rapid indication of the toxicity of effluents or receiving waters, prior to more detailed investigations. The NRA recently prepared a shortlist of potentially suitable techniques which identifies four screening tests as suitable for further development:

(i) Microtox,

(ii) ECLOX,

(iii) *Daphnia magna* in vivo enzyme inhibition,

(iv) Algal/macrophyte fluorescence inhibition.

Each of these is briefly reviewed below.

Screening tests may be especially useful in General Quality Assessment since some tests could be included as a metric in biotic integrity indices (although to date no regulatory body in the United States or Europe has used these tests in this way).

'Microtox'. The Microtox test is the most developed and widely used bioluminescence assay. The test measures the effect of pure substances, formulations, effluents, leachates and receiving waters on the light output of the marine bacterium *Photobacterium phosphoreum* (Johnson 1995). The bacteria emit light as a by-product of metabolism and the luminescence response is easily calibrated with a photometer. When the bacterium is exposed to toxins there is a reduction in light production which is proportional to the toxins ability to inhibit respiration. The Microtox test has been standardised by (i) extensive testing (ii) some inter-laboratory replication (iii) most workers using the same materials and procedures (iv) the use of the same freeze-dried cultures and (v) standardised equipment. The test is reproducible, sensitive and precise (Mayfield 1993).

ECLOX. The Enhanced Chemical Luminescence Oxygen Reaction (ECLOX) test is a rapid *in vitro* chemiluminescent screening test involving horse radish peroxidase mediated conversion of luminol to light. The test measures the effects of substances over a period of four minutes. It can be used with heavy metals, polar narcotics (e.g. phenol), cholinesterase inhibitors (e.g. malathion), respiratory blockers (e.g. cyanide) and photosynthesis inhibitors (e.g. atrazine) and has been shown to react to effluents from sewage treatment works, textile and dye works, abattoirs, engineering works, metal electroplating works, chemical plants, food and drink processing plants, dairy plants and farm wastes (Johnson 1995).

Daphnia magna in vivo enzyme inhibition. Toxicant-induced effects on B-galactosidase activity in crustaceans is the basis for a 1 hour test using the presence/absence of fluorescence as the endpoint. Neonate *Daphnia magna* ingest 4-methylumbelliferyl-B-D-galacto-side (MUF-galactoside) which is metabolised by the enzyme B-galactosidase to produce B-galactose and fluorescent 4-methyl umbelliferol. Toxicants inhibit this reaction which is based in the gut of the animal Experimental development indicates that changes in luminescence are due to inhibition of the enzyme rather than changes in the rate of feeding. Animals are exposed to toxicants for 1 hour after which the MUF-galactoside solution is added. After 15 minutes incubation the number of fluorescent test animals is counted using a UV light source. The test is sensitive to heavy metals, polar narcotics (e.g. phenol) and cholinesterase inhibitors (e.g. malathion).

Algal/macrophyte fluorescence inhibition. These tests did not score particularly highly in the NRA analysis of screening tests (Johnson 1995) but were included because the three tests listed above were insensitive to photosynthesis inhibitors.

The range of sensitivities shown by these methods is given in Table 4.20 with comparative acute and chronic toxicity values for *Daphnia magna* and rainbow trout.

Table 4.20 Comparison of sensitivity of rapid screening tests compared to two standard laboratory toxicity tests (Johnson 1995)

Substance	Toxicity data IC_{50} or EC_{50} in mg l^{-1}				
	ECLOX	Microtox	In vivo enzyme inhibition (*Daphnia magna*)	48hr EC_{50} /LC_{50} for *Daphnia magna* immobilisation test (Johnson 1995)	Acute 96hr LC_{50} rainbow trout
Cadmium	>16.0	14.0	**0.2**	0.03	2.6 (0.001 for parr in soft water)
Mercury	>16.0	0.05	**0.015**	0.038	0.042
Zinc	550	**0.2**	0.98	1100	4.5
Phenol	**1.0**	20.0	37.5	10.0	-
Pentachlorophenol	-	0.52	1.0	0.55	-
Malathion	-	3.0	**0.022**	0.001	-
Cyanide	**<0.08**	2.8	-	-	-

IC_{50} 50% Inhibition concentration
EC_{50} 50% Effective concentration
LC_{50} 50% Lethal concentration

4.16.3 Potential suitability of ecotoxicological screening tests for bioassessment

Advantages and disadvantages

Controlled experimental studies are an essential part of water quality control but are not relevant to General Quality Assessment work. However, screening tests offer a number of important advantages for GQA work.

Advantages. Screening tests can be applied to all kinds of waterbody and provide comparable results in all waterbodies. Screening tests measure toxicants which are normally expensive and difficult to measure, requiring many different analytical methods to measure. Screening tests are amongst the quickest of bioassessment methods. They are quick to undertake and require little training time for staff, making them very cost effective. The tests are potentially highly sensitive, and may provide early warning of impacts.

Disadvantages. The main disadvantages of screening techniques are that they lack ecological reality and have not yet been widely applied in bioassessment programmes. Screening tests do not directly measure any elements of biodiversity. In effluent testing it is not usually possible to determine what the response is being made to, except the general 'cocktail' of potential toxicants.

Screening tests do not integrate variations in toxicant pulses (which may be especially important in water bodies experience intermittent toxic impacts). Results are also dependent on chemical concentrations so are likely to have greater variability than methods which integrate environmental stressors.

Screening methods which use *Daphnia* species suffer from the standard difficulties associated with culturing cladocerans for toxicological work, *Daphnia* spp. being very sensitive organisms which require careful culture measures (Burton and MacPherson 1995).

Table 4.21 Advantages and disadvantages of screening tests

1.	Provide a rough measure of toxicants which are normally difficult and expensive to measure	1.	Screening tests cannot discriminate between different test chemicals
2.	Measures biochemical processes which are not normally looked at except in experimental studies	2.	The use of screening methods in biotic integrity indices is not yet developed
3.	Screening tests are potentially the quickest of all bioassessment methods	3.	Results depend on toxicant concentrations being measured, so may be variable as normal water chemical sampling
4.	Easy to undertake, needing little training for staff	4.	Screening tests have a low level of ecological reality.
5.	Screening tests can be applied to many different kind of receiving water and effluents	5.	

4.17 Conclusions: adopting the principles of multimetric assessment

A major conclusion from this literature review is that the concept of multimetric assessment of ecological integrity - now used routinely in the United States - has many similarities with the theoretical framework proposed for use by EA in Chapter 3 of this report.

The key feature of the multimetric system is that ***each assemblage attribute which is shown to be relevant to ecosystem degradation (i.e. each metric) is scored separately according to the extent to which it deviates from an undisturbed baseline condition. Metrics are then divided into simple 'rating' categories and summed to give a single index.***

The principle benefit of this system is that it enables a wide range of ecosystem measures to be combined in a single index. This considerably increases the potential for biological assessments to represent the integrity of the community as a whole. In addition, multimetric indices are very flexible in that new metrics can be added at any stage without undermining the entire concept.

In effect the multimetric methodology provides a simple and convenient means of putting into practice the precepts of the rationalised protocol developed for EA monitoring in Chapter 3. ***It is therefore recommended that a multimetric approach is adopted for EA use in biological monitoring of still waters.***

As noted earlier in this Chapter, the main shortcoming of the multimetric approach as practised in the US, is that the classification groups used for any water body type are arrived at subjectively (although within the context of natural regions). The more advanced multivariate statistical techniques now routinely used in Britain (e.g. the RIVPACS methodology) have not yet been applied in the United States. Uniting the two approaches has the potential to give the best of both worlds.

Diagnosis

Most biological assessment methods developed to date (for both running and still waters) have been intended to assess the effects of specific forms of environmental stress (e.g. organic pollution, eutrophication), *inter alia*, most techniques have been *diagnostic* methods.

Good diagnostic indicators need to show a strong and discriminatory relationship with a particular form of stress. They may be based on:

(i) assemblage characteristics e.g. pollutant scores and indices such as the Trophic Diatom Index, Saprobic index etc.),

(ii) characteristics of individual species or taxa (e.g. indicator species),

(iii) aspects of the ecology or physiology of individual species (e.g. deformities, behavioural changes or physiological responses).

The physiological and ecological characteristics of different plant and animal assemblages predispose them to respond preferentially to certain types impact, making them more effective indicators of these impacts. Consequently, no one group is equally responsive to all stresses, and no one approach (field-based, lab-based, indicator species, assemblage index etc.) is likely to be appropriate for all situations.

EA requirements for impact diagnosis are highly varied - from compliance against legislation to investigation of reduced fish stocks. The requirements in any situation will therefore be varied, and are best satisfied by the development of specific tools for specific jobs. The need for such an approach has, in effect, been recognised already by EA, in the use of fish and invertebrate methods for general river monitoring and the development of new macrophyte and diatom indices for specifically assessment impacts of nutrient pollution in the context of the Urban Waste Water Treatment Directive.

'Early warning' methods

'Early warning' methods are, in theory, highly desirable and worth developing. Preventing damage before it occurs is the essence of sustainable development but at present few methods are capable of providing useful warning of environmental damage. Techniques identified in this review indicate that more refined early warning indicators could be based on:

(i) very sensitive indicator species with high discriminatory powers - so that trends can be identified early on (e.g. diatoms),

(ii) individual-based attributes which occur *prior* to death (e.g. deformity, disease),

(iii) broad-based methods such as ECLOX which have the potential to screen for a considerable number of chemical parameters, offering a wide-ranging assessment of chemical risk factors.

All early warning methods need to be developed further and ideally they should be used in tandem with general ecosystem monitoring. This requires methods which can either: (i) use information which could be quickly assessed from field samples already collected for GQA or (ii) uses rapid field or laboratory based screening assessments such as ECLOX.

5. CHOICE OF ASSEMBLAGES FOR ASSESSING THE QUALITY OF EACH WATERBODY TYPE

5.1 Introduction

Previous chapters have recommended the adoption of a multimetric method of water quality monitoring based on assessment of biotic integrity and using a range of assemblage attributes.

In this chapter each assemblage is evaluated in more detail in order to identify those most appropriate for use in assessing biological water quality in each waterbody type.

The chapter is divided into three sections:

(i) the results of the matrix analysis of each taxonomic assemblage,

(ii) discussion of which taxa are likely to provide the best assessment of overall quality in each waterbody type,

(iii) the implications of choosing specific assemblages for further development in terms of (a) EA requirements, and (b) harmonisation of survey techniques.

5.2 Choosing the most appropriate assemblages using matrix analysis

Identifying the assemblages most suitable for water quality monitoring involves evaluation of many (sometimes conflicting) variables. Matrix analysis was used to facilitate objective comparisons of the major taxonomic assemblages, with separate matrices completed for each of the main waterbody types (lakes, ponds, temporary ponds, ditches, canals, brackish waters).

5.2.1 Choice of criteria for evaluation of methods

Matrix evaluation was undertaken for ten major taxonomic groupings: phytoplankton, periphyton, marginal macrophytes, submerged aquatic macrophytes, microinvertebrates, macroinvertebrates, fish, amphibians, birds and mammals. Habitat-based assessments and screening ecotoxicological tests have not been considered further, for the reasons outlined in Chapter 4. Macrophytes were split into emergent and aquatic forms because of the considerable differences in the responses of these two groups to environmental stresses. Further divisions of the assemblages on the basis of habitat (i.e. divisions into littoral or benthic assemblages) and taxonomic groupings (e.g. microinvertebrates divided into Cladocera, rotifers etc.) were considered. However, following initial trials, these sub-divisions provided too little new information to justify the increase in complexity of the matrices.

The suitability of each taxonomic group was assessed using sixteen criteria (see Table 5.1). These were selected following an extensive literature review and were grouped under three broad headings:

1. *The ecological relevance of the group*

   Fifteen criteria were used to assess: (i) the extent to which the group is representative of overall biodiversity; and (ii) how well each assemblage is likely to respond to, and integrate, the wide range of anthropogenic stresses which may affect waterbody integrity (see Chapter 3). Public perception of the importance of each assemblage was also addressed.

2. *The practical suitability of the group*

   Five criteria were used to assess the practical suitability of using each assemblage as the basis for monitoring. This included questions relating to 'catchability' i.e. the abundance of individuals in waterbody types, and consideration of whether taxa are *naturally* found in all physico-chemical variants of each waterbody type. This is important since, clearly, if a group is often naturally absent (e.g. amphibians in highly acid waters), then it has significant disadvantages as a survey tool which can be universally applied. Questions of temporal and spatial variability within waterbodies were also addressed on the basis that groups with low intra-waterbody variability provide a considerable methodological advantage (i.e. representative samples can be collected at a small number of sampling stations and in a small number of sampling visits).

3. *The cost of collecting and analysing data for each assemblage*

   Costs were assessed using three criteria: (i) the cost of equipment and consumables (ii) the time require to undertake field surveys, laboratory work and data analysis, and (iii) the time required to train staff to become proficient in the use of methods.

   Unlike the other two areas of evaluation (ecological relevance and practical suitability), taxa can only be assessed on cost basis by considering specific methods. Costs were therefore estimated using examples of standard field collection methods undertaken to varying levels of taxonomic resolution (e.g. family and species level).

5.2.2 Matrix scores

Each of the sixteen criteria assessed in the matrix analyses was given a numerical score and assessed as follows:

(i) Evaluation of ecological validity and practical relevance was undertaken using a simple ranking system on a five-point scale (e.g. 0 = very poor to 4= very good). The scoring system used for each assessment criterion is given in Appendix 5.

(i) Costs were estimated and entered in monetary terms, and were therefore assessed independently.

Matrix scores were not weighted in any way, although there was considerable discussion relating to the advantages of weighting taxa on the basis of their ability to determine damage caused by the main environmental stressors (e.g. nutrients and acidification in lakes). Ultimately, weighting was not used in this context because it is essentially inappropriate in general quality assessment; this is because:

(i) If a pollutant does *indeed* cause wide ranging ecosystem damage this will, inevitably, be strongly tracked by taxa which integrate waterbody conditions as a whole.

(ii) Weighting factors that we subjectively *believe* to be important may lead to an underestimate of other significant influences.

5.3 Overview of matrix analyses results

Matrix results based on ecological relevance, practical suitability and cost are discussed below. The completed matrices for each still-water type are given in Appendices 5 and 6. A summary of the results is given in Table 5.2.

5.3.1 Ecological relevance

Matrix analysis indicates that none of the assemblages assessed is an ideal indicator of waterbody condition, able to provide a fully integrated response to all anthropogenic stresses which could affect still waters.

In general, however, and looking across all waterbody types, macroinvertebrates are the assemblage which most *consistently* give high scores. Fish score similarly, or slightly higher, in lakes, canals and ditches.

Table 5.1 Criteria used to evaluate suitability of plant and animal assemblages for GQA monitoring

Ecological relevance

Species-richness in the waterbody type

Range of trophic levels at which the group occurs

Range of waterbody habitats that the group occupies

Extent to which the group reflects aquatic/wetland (as opposed to terrestrial) influences

General interest in, and concern about, the group (ecological, conservation, public)

The ability of the group to integrate the environmental quality spatially

The ability of the group to integrate the environmental quality temporally

The responsiveness of the group to anthropogenic impacts including:

- Nutrient enrichment
- Acidification/pH changes
- Deoxygenation
- Biocides and other micro-organics
- Metals
- Turbidity
- Water level changes
- Physical habitat damage
- Biological impacts e.g. nuisance spp.

Practical suitability

How well is the taxonomy of the group known?

Does the group occur throughout the range of water chemistry regimes naturally present in the waterbody type?

Does the group occur throughout the range of physical variants naturally present in the waterbody type?

The typical abundance of individuals

The extent to which the group shows:

- temporal persistence in the waterbody
- intra-season stability in community types
- intra-habitat homogeneity within the waterbody

Costs

Cost of equipment items and consumables

Time required for staff training

Time required to undertake field surveys, laboratory work and data inputting

The major advantages inherent in both fish and macroinvertebrate assemblages are that these groups occupy a wide range of microhabitat types and a number of trophic levels. In addition, individuals show relatively long temporal persistence (i.e. long life-cycles), giving them the facility to represent the net effects of pollutants over time. In habitats where they are present, fish scored particularly well because of their ability to spatially integrate environmental conditions through mobility. In addition, many species are of considerable public interest.

The matrix scores for micro-floral and faunal assemblages, although relatively high, were consistently lower than for fish and invertebrates. This partly reflected the more limited ability of these groups to integrate conditions either temporally or spatially, and their narrower range of trophic interactions.

As might be expected, the relevance of macrophytes changed with waterbody type. Thus, *aquatic* macrophytes were most important for deeper waters, whilst *emergent* macrophytes became progressively more relevant as indicators in shallower waters. In temporary ponds, which are often strongly influenced by the quality of their sediments and surrounds, emergent macrophytes are likely be an important assemblage for quality assessment.

Amphibians, mammals and birds generally had the lowest matrix scores. This reflected the fact that all three groups are only partly dependent on the aquatic environment and, consequently, integrate aquatic environmental stresses less effectively than entirely aquatic taxa.

5.3.2 Practical feasibility

In terms of practical feasibility, most groups scored well. Only amphibians had comparatively low scores, resulting from their poor temporal persistence and low abundance (giving poor 'catchability').

Again, macroinvertebrate scores were consistently high for all still-water types, and whilst the matrix totals for periphyton and zooplankton were slightly lower, the scores indicate that they could also prove suitable as a basis for water quality assessments on practical grounds.

The practical suitability of some groups changed with waterbody type, however. In lakes and canals, most plant and animal groups (with the exception of amphibians) scored highly. Scores for ditches, ponds, temporary ponds and brackish waters were more variable, ultimately reflecting the smaller size and inherent natural variability of these ecosystems. Thus, although fish occur widely in lakes and canals, they are considerably less widespread (and therefore have lower viability) in smaller and more isolated waterbodies, due to their intolerance of the shallow, acid and naturally organic rich conditions which may prevail in these habitats. Aquatic and marginal macrophytes are similarly disadvantaged through their intolerance of heavy shade, which can exclude them from small or narrow waters.

Table 5.2 Summary of matrix analysis scores based on ecological relevance and practical suitability

	Phyto-plankton	Peri-phyton	Aquatic macro phytes	Emergent macro phytes	Micro inverte brates	Macro inverte brates	Fish	Amphib ians	Birds	Mammals
Lakes										
Ecological relevance	63	62	62	52	61	62	73	46	59	53
Practical suitability	69	74	77	74	74	77	74	66	74	69
Combined score	66	68	70	63	68	73	75	56	67	61
Ponds										
Ecological relevance	59	65	65	54	63	73	70	51	57	53
Practical suitability	66	74	71	69	74	77	63	54	60	57
Combined score	62	69	68	62	69	75	66	53	58	55
Temporary ponds										
Ecological relevance	58	70	59	62	67	71	0	53	0	54
Practical suitability	51	71	54	74	71	69	0	49	0	46
Combined score	55	70	57	68	69	70	0	51	0	50
Ditches										
Ecological relevance	56	62	62	54	62	71	71	48	55	55
Practical suitability	71	74	69	71	71	71	57	51	60	63
Combined score	64	68	65	63	67	71	64	50	58	59
Canals										
Ecological relevance	59	65	62	53	63	71	73	46	57	53
Practical suitability	71	74	74	71	74	77	77	57	60	60
Combined score	65	69	68	62	69	74	75	52	61	57
Brackish waters										
Ecological relevance	52	61	61	50	59	69	68	44	50	53
Practical suitability	66	74	66	74	74	77	63	54	57	54
Combined score	59	68	63	62	67	73	65	49	54	53

The scores for each assemblage are the percentage of the maximum score, which could be attained by each taxonomic group. Scores have been corrected for unknowns.

5.3.3 Combining ecological relevance and practical suitability

Table 5.3 shows the taxonomic groups with the highest matrix scores in each waterbody type when ecological and practical criteria are combined. The table suggests that macroinvertebrate-based methods may be appropriate for all still-water types. Periphyton and microinvertebrates also had consistently high scores. The relevance of fish, phytoplankton, aquatic and marginal macrophyte assemblages varied between waterbody type.

In summary, therefore, the matrix results suggest (in the absence of cost constraints), that the most viable taxonomic groups for each waterbody type are:

Lakes	Fish, macroinvertebrates, aquatic macrophytes (borderline: zooplankton groups and periphyton).
Canals	Fish, macroinvertebrates, periphyton, microinvertebrates (borderline: aquatic macrophytes).
Ditches	Macroinvertebrates, microinvertebrates (borderline: fish, aquatic macrophytes and periphyton).
Ponds	Macroinvertebrates (borderline: aquatic macrophytes, periphyton and microinvertebrates).
Temporary ponds	Macroinvertebrates, periphyton, microinvertebrates and emergent macrophytes.
Brackish waters	Macroinvertebrates and periphyton (borderline: microinvertebrates).

Table 5.3 Summary of taxonomic groups with the highest matrix scores based on ecological viability and practical relevance

	Lakes	Canals	Ditches	Ponds	Temporary ponds	Brackish waters
Phytoplankton		-	-	-	-	-
Periphyton	[*]	[*]	*	[*]	*	*
Aquatic macrophytes	*	[*]	[*]	[*]	-	-
Emergent macrophytes,	-	-	-	-	*	-
Microinvertebrates	[*]	[*]	*	[*]	*	[*]
Macroinvertebrates	*	*	*	*	*	*
Fish	*	*	[*]	-	-	-
Amphibians	-	-	-	-	-	-
Birds	-	-	-	-	-	-
Mammals	-	-	-	-	-	-

* = within the top 25% of the range of matrix scores

[*] = borderline i.e. within top 30% of the range of matrix scores

5.3.4 Cost matrix results

Methods

The costs of a biotic assessment method depend, crucially, on the details of the method used. Variables such as the number of annual samples used, the number of community attributes recorded and the taxonomic level to which groups are identified may alter the costs of sampling for any group by an order of magnitude. This presents a difficulty for accurate cost estimation, because for most groups there has been too little method evaluation in still waters to allow confident approximation of method details. With diatoms (and many other groups) for example, there is insufficient understanding of spatial and temporal sampling variability to be sure of the number of sampling visits which would be necessary in any year for reliable water quality assessment. Neither can we be sure that family level identification will be sufficient to characterise the integrity of relevant communities - although the presumption is that it will be.

To contend with these methodological unknowns, a number of survey options have been costed for each assemblage. These typically include: (i) the cost of a single 'standard' sample, (ii) the cost of a realistic number of annual samples, bearing in mind the known temporal variability and integrative potential of the group, and (iii) the cost of a more detailed level of survey (e.g. species-level) where appropriate. The specific sampling scheme used for each group (e.g. sampling and analysis times) has been based on a variety of sources including current EA practice in flowing waters, methods under development (e.g. Johnes 1994) and methods used or proposed for still-water assessment in the United States and in Europe.

As a rule, the methods chosen for costing have been those with the potential to provide a *representative and broadly based community sample* which could be flexibly analysed using a variety of potential metrics such as richness, community structure, aspects of health, age structure etc.

For the reasons outlined in Chapters 3 and 4, there has been a presumption against methods which focus on limited taxonomic groups (e.g. gastropods rather than littoral invertebrates) or which aim to distinguish specific pollutants. Estimates for the cost of sampling chlorophyll a have, however, been included to enable comparisons with this more 'chemical' approach to water quality assessment.

Appendix 6 gives an outline of each of the methods that has been costed, and an explanation of the assumptions used in calculation. Costs were estimated independently for each waterbody type. However, matrix tables for: (i) ditches and canals and (ii) ponds and brackish waters proved identical, and these data tables were combined.

Costs for each group of waterbodies were calculated in three areas:

1. Equipment and consumables (per year)

Costs of equipment and consumables are calculated for one person (or team), and include the costs of all survey and analytical equipment required to provide raw data for analysis. Costs are given as annual estimates, with the cost of large capital items (boats, sonar, analytical equipment) estimated as the total equipment cost averaged over 5 years.

2. Staff training (per assessment team)

Training time was based on an assessment of the period required for a new graduate staff member to gain sufficient methodological skills to undertake field surveys and analysis efficiently. This is, inevitably, a difficult estimate to make since, in practice, much of the learning process occurs 'on the job'. Assumptions made in calculating training costs for the trainer and trainee are given in Appendix 6.

3. Survey time (per site)

Estimates of survey costs include time taken to: (i) travel to, and access sites, (ii) undertake the survey, (iii) complete any additional laboratory analysis, and (iv) log the data. Where field surveys require teams of two or more staff, survey time is multiplied accordingly. To simplify the analysis, it has been assumed that data analysis and reporting costs are the same for all assemblages.

Staff costs were estimated at £20 per hour for survey work (including overheads) and £25 per hour where a training input is required from more senior staff.

Results

Tables showing the results of cost assessments for each waterbody type are given in Appendix 6. The findings are discussed below.

1. Equipment and consumables

Matrix results indicate that, at c. £15,000 - £20,000 per year, the equipment costs for fish surveys are at least three times higher than those for any other group. Equipment costs were lowest for amphibians and birds (c. £150 - £550 per year), essentially because surveys are based on simple field observations and require little other equipment. The equipment required for most other taxa lay in the range £1,500 - £4,500, with the majority of this being accounted for by the requirement for a microscope and, in lakes and large ponds, a boat.

2. Training time

An assessment of the costs involved in staff training indicates that, not surprisingly, gaining expertise in species-level identification of 'difficult' taxa (algae, zooplankton, macroinvertebrates) requires a considerable investment of time. Species-level training costs for these groups are estimated to be in order of £20,000 - £30,000 per staff member, compared to c. £7,000 for identification at family/genus-level.

Fisheries survey training costs are relatively low per person, since much training is undertaken 'on the job'. However, the total training cost is increased by the requirement to train four staff for each fisheries survey team. Total costs are therefore c. £12,600 per team, and would be considerably greater if one team member were to be trained in the use of acoustic surveying.

Per person costs for assessment of bird communities were the lowest at £2,600, but this relied on the assumption that it would be possible to employ staff who are already familiar with bird identification.

3. Survey costs

The matrix results indicate that, in general, the macroinvertebrates, zooplankton, and phytoplankton methods have similar costs, i.e. c. £100 - £150 per site for family-/genus-level assessments of sites where no boat-work is needed, and c. £150 - £270 per site for lakes where a boat is routinely used. Macrophyte surveys also appear cost-effective, especially on smaller waters. Fish survey costs are high per visit (c. £900 - £1500, depending on the use of sonar); again, this reflects the greater number of staff required in each survey team.

Survey costs are almost always considerably higher for lakes than for any other waterbody type. This reflects:

1. The greater area and depth of lakes, which increases the complexity of sampling (with the need to collect samples from deep water) and the amount of on-site travel if samples are taken from more than one location in a lake.
2. The requirement for a boat, which increases preparation time and, in most cases, doubles the number of survey staff needed.

It is evident from the data tables that survey costs are considerably increased where there is a requirement for multiple sampling visits each year. This gives an inherent advantage to those groups with longer life-cycles (e.g. macroinvertebrates and fish), which can temporally integrate and represent waterbody conditions through all life stages. In theory, such groups may only need to be assessed once in each survey year, as long as the sampling index period (e.g. season) remains constant between years, or seasonal variation in index values is taken account of in the design of the method (cf. RIVPACS).

5.3.5 Overall costs

For all waterbody types, with the exception of lakes, the cost of a biological survey appears to compare favourably with the cost of chemical surveys. Johnes *et al.*. (1994), for example, suggest that to obtain estimates of lake chemical variables (including chlorophyll a) with an accuracy of about 50% of the annual mean, approximately eight water samples per year would be necessary. The travel and field survey time involved in collecting so many replicates considerably increases the cost of such a method. Thus, total survey costs for an annual chlorophyll a survey (based on the estimates provided by Johnes *et al.*), are similar to those estimated for two seasons of invertebrate sampling (at family level) *plus* a survey of aquatic plants. Costs of phytoplankton and periphyton surveys are each broadly similar to those of macroinvertebrates. A single fish survey, however, costs considerably more than either of these options - approximately half as much again.

In lakes, the larger waterbody size and necessity for boat use considerably increase the cost of most biological survey methods. The costs of family level invertebrate surveys are, for example, at least doubled by the extra survey time and requirement for two staff members in the field. Under these circumstances, chemical sampling (if samples are taken at an outflow and without recourse to a boat), give costs which are comparable to a single invertebrate sample. Even in this case, however, littoral invertebrate surveys alone would still cost considerably less. In *shallow* lakes the cost of a fish survey is similar to the cost in other waterbodies, but increases in deeper waters where there is now the potential for use of sonar.

5.3.6 Summary of matrix findings

Overall, matrix analysis suggests that macroinvertebrates are a rational choice for monitoring in most waterbody types. They compare favourably with other methods on scientific, practical and cost grounds, are universally applicable to all still waterbody types, and when identified at family level survey costs are moderate. The only exception is in lakes, where there may be a need to collect sublittoral or benthic samples, resulting in considerably increased costs.

Fish are highlighted as a valuable survey assemblage, on scientific and practical grounds, in lakes and canals. Although fish survey costs are high per survey in all waterbodies, their relative cost decreases in lakes, where a single annual fish survey begins to compare more favourably with the cost of biannual invertebrate sampling.

Aquatic and marginal macrophytes differ in ecological and practical suitability between waterbody types. Aquatic macrophytes are more relevant in deeper waterbodies, and marginal species in shallower waters. The relative cost of both methods is low to moderate because of the absence of major laboratory processing costs.

Periphyton, and to a lesser extent zooplankton, are consistently high-scoring in ecological and practical suitability matrices, and have similar costs to macroinvertebrates. In lakes, however, periphyton *may* have an advantage, in that littoral samples could be collected without boat use. Phytoplankton generally appear unsuitable as a survey assemblage, because of their short generation times (hours to days). This makes them poor temporal integrators, and consequently expensive to survey, since multiple samples are required in any year or season.

Wetland birds, amphibians and mammals, do not score highly in most matrix analyses, although birds attain moderate ecological matrix scores, and both birds and amphibians have low survey costs. Birds, in particular, might prove a valuable survey group in lakes where populations can be of some importance and interest.

5.4 Evaluation of methods appropriate to each waterbody type

This section provides a more detailed evaluation of the biotic assemblages likely to be appropriate for monitoring each waterbody type. The evaluation is based on the results of matrix analysis but includes a number of additional ecological and practical considerations.

5.4.1 Some general points and assumptions

It is clear from all sections of the matrix analyses that no one assemblage is able to fully represent all aspects of biotic integrity and to integrate the effects of all possible stresses. The logical conclusion from this, must be that waterbody assessments should ideally include data from a range of assemblages.

Advantages of assessment using a number of taxonomic groups

Using more than one taxonomic group to characterise a waterbody has a number of major advantages:

- *It eliminates problems with inevitable false positives and negatives*

  It is important in any monitoring programme to minimise the inaccuracies and uncertainties involved. False positives or false negatives are inevitable with any sampling strategy - clearly, multiple lines of evidence are more likely to protect against erroneous conclusions by allowing validation of results (Cairns *et al.* 1993).
- *It provides strong corroborative evidence for damage where it occurs*

  Whether providing empirical evidence of damage for legislative purposes, or providing the basis for management decisions, there is a very considerable value in having two assemblages which show the same result (Yoder and Rankin 1995).
- *The limitations inherent in any one assemblage can be reduced*

  As stated above, no one assemblage can provide all the information needed to give a thorough assessment of ecological integrity. Using a number of carefully chosen assemblages can minimise the weaknesses shown by any group individually, and thus integrate sources of damage that might have been missed in an evaluation based on a single assemblage. This approach is somewhat analogous to the use of a fish species and an invertebrate species as standard bioassay test organisms (Barbour *et al.* 1995).

Means of combining groups

There are three options for choosing a combination of assemblages:

1. Use the 'best' groups (e.g. the groups with the highest matrix scores such as fish and macroinvertebrates in canals).
2. Choose groups which together reflect the widest range of environmental conditions, for example in terms of: (i) trophic levels (ii) range of habitats occupied (iii) range of impacts to which the groups are sensitive.
3. Combine approaches 1 and 2 (i.e. use high-scoring assemblages which are complimentary in their sensitivities).

As a rule, a combination of approaches (Option 3), is likely to bring the greatest benefits.

Choosing complimentary taxa

Appendix 5.2 (lowest sections) provide matrix data which describes the occurrence of each taxonomic group in terms of (i) trophic levels occupied (ii) the range of habitats used (iii) the range of impacts to which the groups are sensitive.

The results suggests that, in general, the best combination of any two groups would be one plant and one animal group. Together these assemblages have the potential to span a wide range of trophic levels and habitat types. And, whilst both assemblage groups are broadly responsive to major pollutant impacts (e.g. acidification and eutrophication), they are typically complementary in their sensitivity to other forms of degradation such as micro-organics (e.g. herbicides/pesticides), turbidity, deoxygenation, habitat damage etc.

More taxonomic groups for groups for assessing larger waterbodies.

In general, evidence suggests that the larger and the more complex a waterbody the more taxonomic groups are likely to be required for assessment. Ohio EPA, for example, compared 1300 rivers assessed on the basis of fish and invertebrate communities using multimetric criteria. They showed that for small streams (with catchments of <50 square miles) there was agreement between all metrics in 75% of sites. However, on large rivers (with catchments >500 square miles), metrics agreed at only 45% of sites (Yoder and Rankin 1995).

The implication is that there is a greater imperative to adopt a multi-assemblage approach for monitoring large waterbodies. In practice, this means lakes should ideally be surveyed using at least two assemblages, and perhaps more in large lakes.

5.5 Methods for assessing lakes

5.5.1 Introduction

Lakes (i.e. waterbodies greater than 2 ha in area) are the most difficult still waterbody type to monitor cost-effectively. Their often considerable size and depth increases the range of habitats which need to be considered (adding for example, deep sub-littoral and benthic habitats). Most methods require use of a boat and some involve more complex sampling procedures which considerably increase staff time. In addition, lakes support important biotic communities, particularly birds (and too a lesser extent mammals and fish), which may need to be considered in their own right.

The results of matrix analysis suggest that, in terms of ecological relevance and practicability, fish, macroinvertebrate and aquatic macrophyte methods are likely to be most suitable for lake monitoring. However, three other groups scored almost as highly: phytoplankton, periphyton and microinvertebrates. Interestingly, this suite of six groups coincides with the list of taxa proposed in current (draft) guidelines for lake monitoring by the US EPA. Note however, that for each lake the EPA expect to survey all these assemblages using a survey team of 4-5 people over a period of 1 week to assess each lake! (EPA 1994).

The suitability of each of these six taxonomic groups for lake bioassessment is discussed briefly below.

5.5.2 Macroinvertebrates

Macroinvertebrates rank highly as a potential lake assessment assemblage through their ubiquity and their ability to integrate temporal stresses effectively. In a critique of lake assessment methods for the NRA, Johnes *et al.* (1994) doubted the viability of macroinvertebrates as a useful assemblage on a presumption of anomalous spatial heterogeneity. In practice, however, there is no reason to expect lake invertebrate communities to be significantly more variable than they are in rivers and ponds where they have proved to be a viable and sensitive assemblage for bioassessment.

Studies of gravel pit lake macroinvertebrate communities in Southern England confirm this. In a survey of ca. 20 lakes (up to 100 ha), surveyed using 3 minute hand-net samples it was found, for example, that even very short-duration sub-samples (ca. 12 seconds) taken from different parts of the same lake (e.g. bulrush stands on opposite margins of a 20 ha lake) typically grouped together in TWINSPAN analyses. The implication of this is that samples taken from even very limited portions of a lake may be adequately representative of the whole.

In North America, where multimetric assessments are applied routinely, macroinvertebrates have been used for monitoring lakes in both national programmes, such as the US Environmental Monitoring and Assessment Programme (EMAP), and in several states

(Florida, Oklahoma, North Dakota). They also form part of the proposed US Environment Protection Agency lakes and reservoir monitoring programme (EPA 1994).

One of the main questions relating to use of macroinvertebrates in lake assessment is the choice of survey areas. The US EPA and EMAP programmes propose *sublittoral* sampling. However the relative advantages of benthic and littoral sampling should also be considered. Littoral sampling is likely to be of particular interest if marginal degradation as well as chemical quality is of concern. Littoral subsamples in the Pond Action gravel-pit lake data set, for example, clearly showed the effects of degradation caused by wave wash from motor boats (Pond Action 1989).

Other concerns relate to sampling costs, which rise if a number of macroinvertebrate samples are taken and a boat is used. This said, a two seasons family level *littoral* invertebrate survey costs about the same as undertaking an annual set of chlorophyll a samples. A two season mixed benthic and littoral survey would cost perhaps half a much again as a chemical survey, but would provide a greater range of information relating to environmental quality.

5.5.3 Fish

Fish score highly in terms of their ecological relevance in lakes because of their considerable ability to integrate temporal and spatial environmental stresses, and their range of trophic niches. They are also of considerable public interest.

The major ecological disadvantage of fish is that their populations are often strongly influenced by stocking and fisheries management. In practice, this may make it difficult to identify natural community baselines with which other sites can be compared. Further investigation is required to identify the potential for baseline development and classification using ambient communities and/or historical data and professional experience.

A further difficulty with fish surveys is their expense. In practice, however, a wide variety of still water sites are already visited by EA and, were assessment methods to be developed, it might be possible for data from fish stock and other surveys to form the basis of a multimetric assessment. Potential metrics such as species richness, relative abundance, proportion of tolerant/intolerant species, growth rates and health can, for example, all be derived from lake data currently gathered by EA staff.

5.5.4 Aquatic macrophytes

Aquatic macrophytes are effective indicators of a range of stresses relevant to lake quality, and within the growing season, will integrate those stresses temporally. However, macrophytes are typically absent from deep waters and suffer from the disadvantages of strong seasonality. Existing lake surveys (e.g. Palmer *et al.* 1992) have already shown that lake macrophyte communities can be classified and used to provide information on quality and trophic state. In combination with another taxonomic groups, such as fish or invertebrates, it therefore seems likely that aquatic macrophytes could prove a useful assemblage.

5.5.5 Planktonic groups: phytoplankton and microinvertebrates

Although phytoplankton and zooplankton are the most studied lake organisms, their short life spans prevent them from being ideal long-term integrators of environmental conditions. Sampling costs are therefore high because of the requirement for a greater number of samples. In theory, planktonic samples taken from the open water areas of a lake offer considerable potential for assessing *net* lake water quality. In practice, open water areas often support impoverished planktonic communities with, therefore, relatively little discrimination ability. Littoral communities are typically richer, but in these areas other, more generally useful groups (such as invertebrates and periphyton), are more appropriate options.

5.5.6 Periphyton

Periphyton could provide a useful assemblage in lake assessment. Like macrophytes, the periphyton community is likely to be an effective indicators of a range of stresses relevant to lake quality. In addition, periphyton assemblages should be available for sampling over a more extended period of the year. There are, however, still many unknowns relating to the viability of this assemblage, including the spatial and substrate variability of periphyton, the optimal habitats for sampling (littoral, epibenthic etc.) and the ability of periphyton to track degradation (i.e. the ability to discriminate) where sample identification is only undertaken to generic level. Further investigation in these areas would seem to be justified.

5.5.7 Mammals, birds, amphibians and marginal macrophytes

Mammals, birds, amphibians and marginal macrophytes had the lowest scores in the lake matrix analysis, in part because none are entirely dependent on the lake environment. Thus, even though macrophytes, birds and amphibian surveys were amongst the least expensive methods, their viability as assessment tools is compromised by their inability to adequately reflect waterbody conditions.

5.5.8 Lake survey strategy recommended for further testing

Consideration of lake size suggests that ideally two (or three biotic) assemblages should be used to characterise the quality of lakes. On the basis of the findings above, it is recommended that these should be: (i) macroinvertebrates, (ii) fish and (iii) either macrophytes or periphyton.

In practice, there is currently insufficient data to recommend detailed survey methodologies for any of these groups. Thus, prior to the development of any multimetric assessment method there is a need for a method development phase. The information most immediately required is:

Fish: (i) a desk-study assessment of the potential for an 'adequate' unimpaired baseline, and (ii) the potential for development of cost effective sampling methods, taking existing survey strategies into account.

Macroinvertebrates: (i) the relative value of benthic, sub-littoral and littoral samples, and (ii) spatial variability considerations, including the number of subsamples required at any site.

Periphyton: (i) the effectiveness of littoral vs. other (e.g. epibenthic) samples, and (ii) investigation of temporal, spatial and substrate variability.

Macrophytes: details of survey methodologies e.g. number, length and locations of transacts, and the resultant variability.

Finally it should be noted that if the (draft) Directive on the Ecological Quality of Water is implemented in its current form, its remit requires surveys of three additional assemblages (where relevant): birds, amphibians and mammals. A desk study, followed by specific surveys aimed at relevant groups in the large waterbodies targeted by the Directive, would probably be the most cost effective means of sampling these groups.

5.6 Ponds

5.6.1 Choice of assemblages

In ponds, macroinvertebrate communities attained the highest matrix suitability scores. In common with most other assemblages, macroinvertebrate survey cost in these small and shallow waterbodies are relatively low. There is evidence to suggest that macroinvertebrate communities can be reliably sampled using standard techniques (e.g. Pond Action 1994).

Periphyton and aquatic macrophytes also score quite highly, in terms of both ecological relevance and practical suitability. Techniques for monitoring aquatic macrophytes in ponds are well developed, and there is evidence that their communities are significantly affected by degradation of water quality. The main disadvantage of macrophytes is that there are often relatively few species naturally present in small and shaded ponds. As a relatively rich assemblage, periphyton communities offer considerable potential for water quality assessment in ponds. However a considerable amount of work is required to assess the viability of this group for small waterbody assessment.

Pond microinvertebrate communities are likely to be species rich. However, little is known about the potential of this assemblage as an indicator of water quality. It is likely that microinvertebrates will be strongly influenced by fish communities, which themselves may be highly manipulated.

Fish communities are not recommended as a basis for monitoring ponds. Surveys have high associated costs and fish are frequently absent (or present but with very low species richness) in small, shallow, shaded and acid sites. In addition the natural fish population of many ponds is considerably compromised by management and casual introductions.

5.6.2 Baseline considerations

Although many ponds are man-made, ponds are essentially a semi-natural habitat type which has been perpetuated by human activity. It is quite feasible, therefore, to locate sites that are relatively unimpacted and reasonably straightforward to identify baseline conditions. In practice baseline conditions in ponds may be more variable than for lakes and canals, with quite localised influences (such as succession and shading) influencing community structure. A greater number of sites may therefore be required to characterise the baseline.

5.6.3 Recommendations

It is recommended that for ponds, macroinvertebrates should form the basis of quality assessment methods. The addition of a plant group, either aquatic macrophytes or periphyton, is also recommended. Of these two, aquatic macrophytes are currently the most readily monitored. However, periphyton communities may have considerable potential in the future.

5.7 Temporary ponds

Macroinvertebrates, microinvertebrates and periphyton and macrophytes are all viable assemblages for assessment of temporary pond communities. Choosing between these assemblages is, however, an operation of dubious merit. Temporary ponds have a naturally low species-richness but often support highly specialised and sensitive taxa. There is, therefore, some concern about the use of only one or two groups (especially if identified to family level) to adequately indicate environmental quality. In addition there is very little information relating to British temporary pond communities and the stresses acting upon them. A trial survey based on all four assemblages is therefore recommended prior to the further development of an assessment method.

5.8 Canals

5.8.1 Advantages and disadvantages of different taxonomic assemblages

Fish and macroinvertebrates both score highly in terms of ecological relevance and practical suitability for assessment of canals. Other groups which also score well are periphyton, aquatic macrophytes and microinvertebrates.

Fish have a high public profile in canals and can be easier to sample here than in most other still water habitats. As in lakes, however, there may be difficulties in setting minimally impacted baseline conditions due to the manipulation of fish populations.

Macroinvertebrates are already used to assess the quality of canals in some EA regions, and although there is concern as to whether a RIVPACS/BMWP approach is appropriate for very slow flowing waters, it is clear that invertebrate surveys are a viable approach to the monitoring of canals.

Use of aquatic macrophytes as a monitoring group is made difficult by the generally high levels of turbidity which are associated with canals, even at quite low frequencies of boat movement. Consequently, even in the absence of water pollution and other physical impacts, aquatic plant diversity in canals is usually low. This suggests that, although aquatic plants have relatively high matrix scores for canal assessment, they may not be ideal for monitoring in practice. Periphyton communities in canals are currently very little known but probably experience similar impacts to those affecting the macrophytes. However, the greater inherent community richness within the periphyton assemblage may make it a better option for canal monitoring.

Canal microinvertebrate communities are very poorly known but these taxa generally show relatively poor temporal integration and, as in ponds, the zooplankton component is likely to be highly influenced by managed fish populations. Further work would be required to fully judge the potential of microinvertebrate assemblages for canal quality monitoring, but there is currently a presumption against them, on the basis that animal communities can already be represented by better known taxa such as macroinvertebrates.

5.8.2 Establishing a baseline

In comparison with other waterbody types, canals are relatively homogeneous in terms of their physical characteristics (width, water depth, profile, flow). The number of baseline reference sites required to characterise canal assemblages is therefore likely to be relatively low, and is mainly needed to reflect regional differences or, in unused sections, succession.

Determining what constitutes 'appropriate' baseline reference conditions for canals may, however, present some initial difficulties. A few canals do support outstandingly rich plant and animal assemblages but such sites typically have no, or very little, boat traffic. Since canals were constructed specifically *for* boat traffic and have no natural analogues, it may be considered inappropriate to use such sites as a baseline. Ultimately a pragmatic compromise may be the answer, where a 'normal' baseline comprises sites impacted by moderate boat traffic but with minimal additional chemical (i.e. 'pollution') degradation.

5.8.3 Canal survey strategy recommended for further testing

Matrix analysis and existing operational practice indicate that a macroinvertebrate assemblage should form the basis of canal assessment techniques. However, as with other waterbody types, a better measure of biotic integrity is likely to be obtained if assessments are based on both plant and animal groups. In practice, canal aquatic macrophyte communities are likely to prove species poor where boat traffic is present and may not be suitable. The viability of this group therefore depends on the choice of baseline. Periphyton assemblages have potential but, as yet, the extent of this is undetermined. As with lakes, it would be worth investigating the potential for use of fish community data, particularly if information could be derived from data already collected.

The overall recommendation for canals is therefore to initially develop assessments on the basis of macroinvertebrates alone, but to investigate diatom and, potentially, fish assemblages with a view to developing a second assessment group.

If the current requirements of the Ecological Directive on Standing Waters are implemented, more detailed monitoring of canals may be required. In practice (only) fish, invertebrates, plants and to a lesser extent mammals (e.g. water vole) and water birds are relevant to canals. these assemblages could be easily incorporated into biotic integrity indices, but the cost of survey would clearly be high.

5.9 Ditches

5.9.1 Advantages and disadvantages of different taxonomic assemblages

Ditches are a highly varied and frequently highly stressed waterbody type: they may be temporary, still or flowing, and may vary through the year. In addition, even where seasonally dry, ditches are frequently connected to permanent streams and may quickly regain taxa such as fish. Ditch communities are often heavily impacted by pollutants from the land that they drain. Because they are narrow and often have hedges alongside them, they are often also heavily shaded. Many ditches are regularly managed by dredging.

This combination of stresses inevitably means that the best groups to use in ditch monitoring are tolerant and cosmopolitan groups (i.e. macroinvertebrates, periphyton, macrophytes and microinvertebrates). Existing studies have shown links between waterbody quality and macroinvertebrate communities in ditches (e.g. Foster *et al.* 1991) and there is a published methodology for assessing ditch quality based on macrophyte assemblages (Palmer and Alcock and Palmer 1985). In addition, ditches have an acknowledged importance in terms of their macrophyte communities in many parts of the country. As in other still waters, periphyton communities show potential for ditch monitoring, especially where 'natural' conditions are not ideal for macrophytes. Microinvertebrates may also have potential since a range of taxa (Cladocera, copepods etc.) are tolerant of the stressed conditions which may often prevail. In practice, however little is known of zooplankton and other microinvertebrate communities in ditch habitats, and their choice as a basis for monitoring would be highly speculative.

5.9.2 Baselines

Setting a baseline for ditches may be difficult, in that they are varied, usually wholly artificial and by their nature, usually associated with intensive forms of land management. This said, even in the most intensive landscapes, it is possible to find high quality ditch communities. In Eastern England, for example, Foster *et al.* (1991) discovered exceptionally rich invertebrate communities in groundwater-fed ditches, amongst otherwise impoverished drains.

5.9.3 Recommendations

The recommendation for ditches is similar to that for other shallow waterbodies: to base assessment on macroinvertebrate communities, with the addition of a plant assemblage. For ditches, the interest in associated plant communities would seem to make aquatic macrophytes a logical choice. However, again, diatom communities may prove a viable group, especially for sites highly stressed by changes in water depth and permanence.

5.10 Brackish waters

Like temporary waters, brackish waterbodies typically support relatively specialist and species-poor communities. At a national level, little is known of the composition of these communities, nor the principle stresses that degrade them. On this basis, choosing between assemblages is not simple. From present knowledge, macroinvertebrates, periphyton, zooplankton (and possibly macrophytes) would all be feasible groups on which to base an assessment programme.

Overall, the best course of action is, as in temporary ponds, to undertake a trial survey based on all four of the potential assemblages prior to the further development of an assessment method.

5.11 Other considerations in choosing appropriate methods and assemblages

The first part of this chapter evaluated biotic assemblages in order to determine the most appropriate and cost effective assemblages to survey in each waterbody type. The remainder of this chapter briefly discusses another important consideration relevant to method development: the potential for integration with other projects and methodologies within the EA and abroad.

5.11.1 Facilitating harmonisation and integration of still water methods

There are clear advantages to be gained from integrating different water quality assessment methods. Potential benefits include:

- the ability to directly compare results, facilitating greater understanding,
- greater clarity and coherence of methods, objectives and results - facilitating greater comprehension, acceptance and use,
- the potential for mixing data sets - giving greater analytical and investigative powers,
- economic efficiency in method testing and evaluation.

The rationale for integration is strengthened where different assessment programmes (e.g.. chemical vs. biological, fish vs. macrophytes) are applied to the monitoring of the same waterbody type. Thus, chemical and biological assessment methods need to be complimentary, working together as part of a coherent programme which ultimately aims to sustain the natural resource.

With respect to the development of still water assessment methods, two approaches, in particular, need to be considered early on in method development to allow maximum potential for integration. These are:

(i) the EA's proposed lake classification,

(ii) the RIVPACS method developed for general quality assessment of running waters.

5.11.2 EA Lake Classification

The EA Lake Classification technique developed by Johnes *et al.* (1994), and currently being trialled, essentially compares predominantly chemical variables with a hindcast state, calculated theoretically from catchment and waterbody characteristics. In its use of multiple variables, each ranked and normalised against a baseline state, the approach is directly comparable with the multimetric IBI methods developed by Karr in the early 1980s. It is also the approach adopted in this report for assessing ecosystem integrity based on biotic assemblages.

The main differences in the approaches are (i) the means of selecting the baseline, and (ii) the range of parameters used for assessment.

Baseline selection
The facility of the Lake Classification method to estimate the levels of chemical parameters expected in a relatively undisturbed catchment has many advantages, particularly in areas of Britain where undisturbed sites are rare. Unfortunately, as discussed in Chapter 3, the hindcast modelling approach is not one that can be easily applied to most components of the biota. In practice, there may be some potential for a combination of approaches. For example, where hindcast methods indicate that a currently eutrophic lake should be oligotrophic, it may be possible to adjust biotic scores, using knowledge from other waterbodies. The extent to which this 'mixed' approach is practicable or desirable needs to be investigated further.

Parameters used for assessment
In its current configuration, the Lake Classification essentially aims at a diagnostic assessment - i.e. it principally addresses two major causes of lake pollution - eutrophication and acidification (Johnes *et al.* 1994). This is an appropriate use of an essentially chemical monitoring approach which capitalises on the investigative potential of chemical determinands. The approach is, however, unlikely to provide an adequate assessment of the integrity of lake systems as a whole. There are two major reasons for this:

1. Predominantly chemical hindcasting cannot give an integrated assessment of general ecological quality and so cannot, for example, detect any other major causes of damage, e.g. changes associated with species introductions or disease, the effects of biocides, heavy metals or other chemical pollutants, damage to margins caused by drawdown or poor management practices etc.

2. Chemical parameters can be insensitive to ecological change. This is particularly true for standing waters, where residence time is high and water chemistry monitoring essentially looks at end-of-the-line determinand levels after biotic systems have compensated for, and buffered against, ecosystem degradation. Thus in Loughs Conn and Mak in Ireland, chlorophyll a and total phosphate concentration remained constant whilst algal blooms occurred, arctic char declined and brown trout biomass increased (McGarrigle and Champ 1996). These changes were later diagnosed and found to be due to increased phosphate loading of the lakes. This change was not detectable by water column chemical monitoring however.

   Thus whereas biotic assessments can provide a relatively early warning of damage *as it begins to occur*, reliance on purely chemical parameters runs a considerable risk that detrimental trends may not be identified until widespread, and often irreversible, damage to the system has occurred.

Overall it is suggested that:

1. Although the concept of a minimally impaired baseline is common to both the Lake Classification, and biotic assessment method proposed here, there are inherent differences in the approaches which will make full integration difficult. There may however, be an opportunity for utilisation of chemically derived baseline condition data, in informing the choice of unimpaired sites for biotic monitoring.
2. In terms of identifying ecosystem integrity as a whole, chemical parameters may support biological monitoring and can certainly aid in problem diagnosis. However, as stated in Chapter 1, ecosystem integrity can only be adequately monitored using biological criteria as a base. Too heavy a reliance on chemical parameters has a considerable risk for long term ecosystem protection.

5.11.3 RIVPACS

There are considerable conceptual similarities between the original development of RIVPACS (Wright *et al.* 1984) and the still-waters assessment method proposed in this report. Both assessments are based on the comparison of existing, minimally impacted, reference sites within the context of a classification.

A major difference, however, is that RIVPACS, as currently configured to predict BMWP scores, is specifically weighted to assess the effect of one pollutant on invertebrate communities. In contrast, the still water method proposed here aims to assess general water quality on the basis of a number of community measures, from one or more biotic assemblages, which together measure overall levels of environmental degradation.

It seems likely however, that further development of RIVPACS will lead to departures from the current emphasis on organic stress, a process which has already started with the recent incorporation of taxon-richness into the GQA classification. Current research on the use of neural network methods to investigate spatial and temporal relationships evident in existing survey data is likely to stimulate further developments in this direction.

In the light of the similarities between (i) the RIVPACS approach and the methods proposed here for still water assessment, and (ii) the proposal for adopting macroinvertebrate-based assessments in many still waters, there is a clear case for ensuring that the methods used to collect new invertebrate data are as compatible as possible. In particular, there should be an initial presumption that protocol details for aspects such as survey timing, waterbody areas surveyed and sample processing procedure should be directly comparable, unless there are critical disadvantages in doing so.

5.11.4 Harmonisation with methods used in continental Europe

There is currently an awareness of the need to promote harmonisation of monitoring methods across Europe, and a number of European programmes currently operate to investigate and facilitate this process (CEU 1995). The potential for methodological integration between EA and other European approaches is therefore of interest.

As noted in Chapter 4, assessment methods throughout continental Europe are generally dominated by diagnostic approaches based on specific pollutant problems. In this respect our holistic approach to overall degradation does not integrate well.

However, the draft Directive on the Ecological Quality of Water suggests that the focus of European thought is, itself, changing. Current Directive recommendations, for example, require collection of data relating to:

(i) unimpaired baseline states,

(ii) holistic assessments based on a wide range of taxa,

(iii) use of diversity and health parameters for assessment.

These recommendations suggests a Europe-wide trend towards the integrity-based ecosystem monitoring approach recommended in this report.

5.12 Conclusions and recommendations

For each still waterbody type considered in this report there are several candidate indicators which could be used to gauge water quality and integrity. Evaluation of the relative performance of each assemblage shows that no single group is consistently superior to all others. Given inevitable limitations to the use of any single assemblage for monitoring ecosystem conditions, it is recommended that at least two biological assemblages are used to provide a robust index of ecosystem quality and integrity. The assemblages recommended for each waterbody type are:

Lakes[1]	Macroinvertebrates + Aquatic macrophytes / Diatoms + Fish
Ponds	Macroinvertebrates + Aquatic macrophytes / Diatoms
Canals[1]	Macroinvertebrates + Diatoms / Fish
Ditches	Macroinvertebrates + Aquatic macrophytes / Diatoms
Temporary waters	(Macroinvertebrates, Microinvertebrates, Macrophytes, Diatoms)
Brackish waters	(Macroinvertebrates, Microinvertebrates, Macrophytes, Diatoms)

The assemblage common to, and favoured in, all waterbodies is that of macroinvertebrates. It is therefore recommended that macroinvertebrate communities form the core for water quality assessment in all still waters.

Using a multimetric approach, it should be possible to provide a broad assessment of biological water quality based on macroinvertebrate community metrics alone. However, the reliability and validity of assessments would inevitably be enhanced by addition of a second assemblage. For lakes which are both large waterbodies, and prohibitively difficult to restore once degraded, monitoring on the basis of at least two biotic assemblages is considered essential

A combination of two taxonomic groups would, ideally, comprise macroinvertebrates plus a floral assemblage. In lakes and (to a lesser extent) canals fish provide an additional choice. The main difficulty in selecting a second (or third) complementary assemblage is that there is no completely satisfactory candidate. Macrophytes and periphyton (particularly diatoms) are the principle options, but both have disadvantages as monitoring tools. Surveys of macrophytes are limited to the summer months and communities may be species-poor in naturally shallow, turbid and shaded waterbodies. Diatoms have rather better temporal attributes but are more time consuming (expensive) to process and identify. In addition, their

[1] Note that if the Directive on the Ecological Quality of Water is adopted in its present format, a wider range of taxa should be surveyed in the large lakes and canals to which the Directive applies. This would include mammals, birds and amphibian communities where relevant.

value as a monitoring assemblage is still partly assumed. More research is required in order to investigate areas of uncertainty, particularly: (i) likely spatial and substrate variability, (ii) optimal habitats for sampling, and (iii) likely discrimination of waterbody quality where sample identification is only undertaken to generic level. Further investigation in these areas would seem justified.

Fish are a difficult assemblage to evaluate in terms of their potential Theoretically, natural communities are likely to be an excellent indicator of the condition of permanent waters, particularly in lakes and canals. Their viability is compromised, however, on ecological grounds by frequent manipulation of their populations, and on practical grounds by their cost. Set against this, fish are already monitored in a variety of permanent still waters, and this data has potential for use as part of a multimetric water quality assessment. Overall, therefore, the recommendation is that a desk study is initiated to further investigate the potential of fish communities. Such a study could be undertaken by EA fisheries staff.

Brackish waters and temporary waters support communities which are inherently poor in taxa. Lack of detailed knowledge of the stresses acting upon these communities, combined with the paucity taxa, makes it difficult to predict whether one or two assemblage groups will have sufficient resolution to enable waterbody degradation to be adequately assessed. For these waterbody types an initial trial is recommended in order to investigate the most appropriate combination of taxa for multimetric method development.

Overall, recommendations for further development of the project are:

1. For lakes, ponds, canals and ditches, aim to develop multimetric methods based on macroinvertebrate communities and supplemented by a second group.
2. Initiate trials (for any or all of these waterbodies) based on macroinvertebrate assemblages.
3. Investigate the comparative potential of macrophyte and diatom communities for application as a second assemblage in these habitats.
4. Use desk study information to investigate the potential for fish metrics to be developed for use in lakes and, less importantly, canals.
5. Investigate the most appropriate combination of taxa for development of a multimetric method in brackish and temporary waters.

6. A STRATEGY FOR DEVELOPING A BIOASSESSMENT METHODOLOGY TO MEET EA REQUIREMENTS

6.1 Introduction

The rationale which has been developed in previous chapters provides a single unified approach, which can be consistently and flexibly applied to any waterbody type, using the range of taxa and community measures which are most appropriate to that system.

This chapter describes the methodological development which is necessary to create a fully-functioning general assessment method in any waterbody type. A step-wise approach which minimises initial outlay and risk is suggested.

6.2 Full method development

Full development of the multimetric approach for any waterbody type is a five stage process which comprises:

1. Choice of sites and survey techniques for the creation of a minimally impacted baseline dataset.
2. Collection of data and classification of unimpaired reference sites.
3. Collection of survey data for a range of variably impaired sites.
4. Identification of viable metrics.
5. Testing.

Each of these stages is considered below.

6.2.1. Choice of sites and survey techniques for the creation of a minimally impacted baseline dataset

For any waterbody type (pond, canal etc.), a minimally impacted baseline data set needs to be created for assemblage groups to be used in the assessment (see Section 3.4).

The characterisation of reference conditions can be undertaken using any or all of the five techniques described in Chapter 3 (i.e. minimally impacted present day sites, paleolimnology, historical data, modelling or professional consensus). The preferred method, however, is to use minimally impacted reference sites because they represent the most detailed and consistent source of comparative data, and can be used to set realistic, achievable goals. In addition these reference sites have the potential for further monitoring to investigate the importance of natural temporal variation. The major concern in selection of these reference sites is to ensure that they are as unaffected as possible by major anthropogenic influences, and not moderately disturbed, producing mediocre expectations.

For waterbody types or regions where minimally impacted reference sites are inappropriate or impossible to determine, other approaches, including expert opinion and modelling will inevitably be necessary to modify reference site choice. Such an approach is, for example, likely to be required in determining baseline states for man-made freshwater systems such as canals and reservoirs which are created and used for specific societal purposes.

Selection of reference sites, on whatever basis, needs to consider the principle natural chemical, physical and biotic parameters likely to be acting upon each waterbody type (e.g. longitude and latitude, geology, watershed characteristics, depth, shade). An initial desk study is therefore required in order to: (i) assess the results of previous work on the waterbody type; and (ii) identify a number and location of minimally impacted sites which will adequately reflect this variation. A good knowledge of existing survey data and literature is necessary to inform this process.

The number of regional reference sites chosen should be a function of regional variability and the desired level of detectable change. In practice, the ideal also needs to be balanced against budget realities. Hughes *et al.* (1992) re-analysed fisheries data collected by the Ohio EPA (based on several thousand collections over a 10 year period) and estimated that in the with 50 sites in each US EPA region, variation in the Index of Biotic Integrity of 10% should be detectable.

In addition to decisions relating to the location and numbers of reference sites to be selected, it is vital that the methods used to undertake either a trial or full survey are well designed and tested. Methods used to collect and analyse the reference data, will inevitably form the basis of subsequent methodologies (as was the case with RIVPACS, for example). Poor choices at this stage will, therefore, be perpetuated in all future surveys. In addition to time, habitat and sampling efficiency consideration, initial method testing needs to consider the range of metrics that will ultimately be tested.

6.2.2 Collection of data and classification of unimpaired reference sites

Selection of reference sites is followed by:

- Collection of appropriate biological data from these sites, together with sufficient physical and chemical information to characterise them.
- Classification of biological communities based on this data to minimise natural variation and give better within-class impairment resolution.
- Analysis to identify the natural environmental parameters which characterise (i.e. can be used to predict) each community type.

6.2.3 Survey data for a range of variably impaired sites

Surveys of *impaired* sites (good to very poor) are also essential in order to determine degradation gradients for metric discrimination. This survey may be undertaken consecutively with or following collection of baseline data set. There may also be potential for using existing data from 'impacted' sites, where they exist, providing data is fully compatible in terms of survey methodology and quality.

6.2.4 Identification and development of viable metrics

To determine the discriminatory power of metrics within a waterbody class potential metrics are chosen for assessment. The list of potential metrics should initially be extensive, and include parameters relating to a wide range of community interactions and health (e.g. species/family richness, proportion of functional feeding groups, wet weight, proportion of sensitive taxa etc.).

These variables are tested against the range of best quality and impaired data to identify parameters which show a significant relationship with damage. Clearly, metrics which show a strong monotonic gradient to degradation are likely to be the most effective in accurately expressing degradation through the range of impact intensities. Metrics are rejected if they:

- show high variability in response to *natural* environmental stress,
- the EA decide that implementation would be cost prohibitive,
- have superior measures.

All successful metrics are normalised against the baseline sites and divided to give simple scoring categories (i.e. 1= good, 2=fair, 3=poor, 4=very poor). The process of normalisation provides a mean of combining scores across metrics despite their initially dissimilar values. The division of sites on what is, in reality, a quality continuum, can be undertaken in a number of ways (simple division of the frequency distribution of data into percentiles; proportion of maximum levels etc.).

6.2.5 Combining metrics

Use of the metric data in practice involves combining the normalised metric results to give a single score which represents the overall integrity of the system. This score can be derived from the metrics of a single assemblage, or from the combined results of a number of taxonomic groups. Individual metrics may be weighted if appropriate.

Since metrics are not combined until the final analysis, new metrics or new assemblages can be developed independently, over different timescales, and added into the system as they become available. This gives a very flexible methodology which can be improved and refined without undermining the rationale for the method as a whole.

6.2.6 Testing

A trial phase, during which metrics are tested against new sites, is required to validate and refine the methodology. If there is evidence of poor performance this is most likely to indicate that the initial data set was not adequate to reflect natural variability and will suggest a need to collect further data.

6.3 Using the data to provide additional information on the causes of degradation

The approach described above provides a means of general ecosystem assessment, which can indicate whether biological integrity has been impaired. The method does not aim to determine the specific causes of degradation, although clearly the assessment will suggest factors which may be important. Investigating the cause(s) of degradation is, as discussed in earlier sections of this report, conceptually a separate stage, which is likely to require application of a wide array of methods to disentangle the complexities of causation.

This stated, it is clear that the data already gathered for multimetric analysis may have additional potential in providing clues to the causes of impairment. Thus, component parameters can be examined for their individual effects on the aggregated values providing further insight into the factors responsible for degradation.

In addition, there is considerable potential for correlation of individual metrics with specific pollutants or other data from impaired sites (collected either during biological surveys, or from other EA sampling programmes). The results of such analysis (e.g. development of trophic ranking scores etc.) may offer a considerable diagnostic capability.

As use of the RIVPACS method in rivers has shown, the process of routine monitoring for GQA itself generates large amounts of consistently collected data over a number of years. This information provides an important data resource which can, in later years, be used to refine the assessment methods. In association with physical and chemical data, as described above, this biotic data from routine monitoring may itself be used to increase the potential for diagnosis of the cause of degradation at a site.

6.4 Increasing the cost-effectiveness of method development

Clearly, the processes of setting up a full multimetric assessment system for all still waters would have considerable resource implications for the EA. However, as noted above, the multimetric approach is very flexible, and can be developed incrementally for any waterbody type.

There are at least six options which may be used to reducing the costs and risks, associated with the early development and trialling of the method. Not all of these options are recommended, however.

Recommended options

- Test the approach using a single waterbody type,
- Use existing survey data as the basis for baseline or metric development,
- Test in a single region.
- Test the approach using a single taxonomic assemblage (i.e. macroinvertebrates or fish),

Not recommended

- Starting by collecting only a minimally impacted baseline (cf. RIVPACS data set),
- Undertaking the baseline survey using family/genus level data (as opposed to species-level ID).

The advantages and disadvantages of these options are discussed briefly below.

6.4.1 Option 1: Test the approach using a single waterbody type

The most obvious option for reducing the initial cost of multimetric method development for still waters is to focus on one or a small number of waterbody types for method testing. Choosing the waterbody(s) most appropriate is ultimately a strategic decision for the EA, but factors which may influence this choice include:

- **Legislative requirements**

The main legislative requirements for monitoring of still waterbodies relate to the proposed Directive on the Ecological Quality of Water. Current documentation (CEU 1995) suggests that mandatory monitoring may be restricted to large waterbodies (lakes and potentially canals) and will involve a considerable range of taxa. Note, however, that these requirements may be considerably modified in the near future, and it would be politic to await new proposals before embarking on a method testing programme which aims to specifically fulfil EU obligations.

- **EA Policy**

Current policy commits the EA to review *'monitoring programmes to ensure a cost effective and consistent level of service for all controlled waters'* (NRA 1993). On this basis, waterbody types which are currently little monitored should be a target for method development. Of the more 'significant' still water types (lakes, ponds, canals, ditches) the current level of service is lowest for ponds, in that the EA undertakes no routine monitoring of these small and numerous controlled waters.

Lakes, ditches and canals, all receive some level of routine monitoring: lake monitoring is currently being addressed through the EA Lake Classification project (although this is broadly a chemical approach). Canals are currently monitored, with what appears to be moderate effectiveness, using RIVPACS methods, as are larger ditches.

- **Waterbody threats**

All freshwater waterbody types are under threat from a diverse range of impacts. Because of their limited size and buffering capacity, smaller waterbodies (ponds, ditches, temporary ponds) are likely to be the most rapidly impacted by pollutants and physical damage. Pollutant damage, in particular, is not evidenced so rapidly in lakes, but, once impacted lakes are prohibitively difficult to restore. There is therefore a considerable imperative for lake monitoring to ensure their protection.

- **The value of the resource (size, number and biodiversity/conservation value)**

It can be argued that the rational for choosing waterbody types for trialling should consider the extent of the standing water resource. This can be assessed purely in terms of their physical attributes (size, number) and also in terms of the biodiversity resource that they provide.

In terms of number, ponds are much the most numerous type of still waterbody (97% of still standing waters are less than 1 ha, Barr *et al.* 1994). They are also a particularly species-rich habitat type - as rich or richer than rivers (Pond Action unpublished data).

By area, rather than number, lakes are the most extensive waterbody type, representing an estimated 75% of permanent still water in the UK. Lakes too can be rich habitats, especially in their littoral zone. Because of their greater size they support fish, bird, invertebrate and plant species not found in other habitats. However, they also lack many specialists associated with small waterbodies.

The number of Britain's temporary ponds has never been estimated. However, it is clear from observations alone that, although temporary ponds are relatively uncommon in agricultural landscapes, in semi-natural areas they can be exceptionally abundant and may out number permanent ponds by an order of magnitude (Pond Action unpublished observations). Temporary ponds are typically relatively species-poor habitats but they are critical in terms of biodiversity; supporting many uncommon species and an important proportion of Britain's most endangered freshwater plants and animals (Biggs *et al.* 1995, Collinson *et al.* 1995).

There are approximately 2500 km of canals in England and Wales, many of which are highly visible and intensively used for recreation. The total length of wet ditches is unknown but is probably many thousands of kilometres. Some ditches and canals which retain good water quality, or are located in relict wetlands areas, support exceptional plant and animal communities (Foster *et al.* 1991). For the most part however, canals and ditches drain agricultural or urban areas and their communities are highly modified in consequence.

Brackish waters support distinctive, though often relatively species-poor communities, including a number of very uncommon plant and invertebrate species. The number and area of these waters has never been recorded in Britain, however, in comparison with other waterbody types it will be tiny. As such, justification of method development on these waters is difficult.

Conclusions
The discussion above indicates that there is a good case for developing methodologies for most still waters. Only temporary and brackish waterbodies could be considered minor still water types. On balance, the argument for method development seems greatest for lakes, but the EA Lake Classification is, in part, addressing this area. Of the other waterbodies, ponds are currently the least well served.

6.4.2 Option 2: Use existing survey data

Many organisations hold still-water data sets. Where this data is of sufficient relevance and quality, it may be cost effective to utilise existing data sets as part of method development. Note, however, that there are considerable resource implications associated with investing in data sets which subsequent analysis proves not to be relevant to the final method.

The following programmes, organisations and individuals currently hold datasets relating to England and Wales with still water ecosystems information at regional or national level: British Waterways, Countryside Council for Wales, Environment Agency, English Nature, DOE (Surface Water Acidification Programme, Countryside Survey), Pond Action, University College London, Liverpool John Moores University, Foster and colleagues (e.g. Foster *et al.* 1990).

However, of these, only a few hold regional or national data sets with information which is both:

(i) directly applicable to the waterbody types, assemblages and methods highlighted as relevant in Chapter 5 and 6,
(ii) collected in a systematic and repeatable manner.

These data sets are reviewed briefly below.

Environment Agency

Lakes. The most comprehensive data held by EA relating to still waters comes from surveys undertaken in the Norfolk Broads and at individual standing water sites (e.g. Rutland Water). The Norfolk Broads dataset covers water chemistry, phytoplankton, zooplankton, macroinvertebrates, macrophytes and fish. Similar data is available from Rutland Water from studies of the effects of ferric iron dosing. Chemical data from approximately 90 lakes throughout England and Wales, collated by Johnes *et al.* (1994), is also held by EA.

Canals and ditches. Family level invertebrate survey data is available for canals and ditches for (predominantly) impacted sites. The largest dataset is held by Severn-Trent region (120 sites) but other regions hold smaller amounts of data, including Anglian Region, Northumbria & Yorkshire Region and Southern Region.

English Nature

Lake macrophyte survey. The principal data set held by English Nature is from lake macrophytes surveys undertaken in the 1970s and 1980s by Margaret Palmer and colleagues (Palmer *et al.* 1992). The database covers approximately 1100 sites although the majority of these are in Scotland. The dataset covers macrophytes and basic water chemistry.

West Midlands Meres. English Nature has sponsored investigations of the West Midlands Meres and data is available for up to 23 sites. Survey work has been undertaken mainly by Brian Moss and colleagues. The dataset includes water chemistry, phytoplankton, zooplankton and macrophytes. Fish data, collected by gill netting, is available for 10 sites.

DOE Surface Water Acidification Programme (Environmental Change Research Unit, University College, London)

Surface Water Acidification Programme. University College London ECRU holds data collected from the national Surface Water Acidification Programme. This covers sites throughout Britain. This includes information on fish, macroinvertebrates, macrophytes, phytoplankton and sedimented diatoms (and other sub-fossil remains). This dataset also has a good range of water chemistry.

ECRU also holds two much larger datasets related to surface water acidification, the SWAP data set and the UK Acid Water Monitoring Programme.

CCW Welsh Lake Survey (Environmental Change Research Unit, University College, London)

The Welsh Lakes Survey covers sites throughout Wales. It includes data on water chemistry, phytoplankton, zooplankton, sedimented diatoms, periphyton, macroinvertebrates, macrophytes and fish. This data set is one of the most complete in its coverage.

Pond Action

Oxfordshire Pond Survey (OPS). The OPS is a detailed regional survey of 35 pond sites with 2 seasons of species level macroinvertebrate data, aquatic macrophyte and detailed physical and chemical data. Species level macroinvertebrate data is available from a further 100 sites.

National Pond Survey. This dataset includes 3 season species level macroinvertebrate data, macrophyte data and a wide range of physical and chemical data from 180 relatively unimpacted reference sites in England, Wales and Scotland. There are no other extensive lake or pond invertebrate datasets identified at this taxonomic level using standard methods.

Ongoing projects

Several projects are currently in progress which could be of relevance to this project. These are the Pond Action ROPA Project, the current extension and testing of the EA Lake Classification Project and the DOE Pond Survey 1996.

Pond Action ROPA Project. Pond Action will be surveying 200 pond sites in the wider countryside during 1996 and 1997 as part of a project on the impacts of pesticides on small water bodies funded by the Natural Environment Research Council. The survey will follow standard National Pond Survey methods and will include species-level macroinvertebrate data and aquatic macrophytes, with a wide range of physical and chemical parameters.

Environment Agency Lake Classification: Phase 2. The EA is about to start collecting data from a large number of lakes to test the proposed lake classification. This project is collecting data on phytoplankton and macrophytes. It may be possible to incorporate further biological sampling at some site to make this dataset suitable for testing biological methods on lakes.

DOE Pond Survey 1996. The DOE Pond Survey 1996 is currently being planned and methods developed with advice from Pond Action and ITE (Biggs *et al*. 1996). It is expected that the survey visit sites predominantly in lowland Britain and will collect biological data relating to aquatic macrophytes. Pond physical features will probably be described using National Pond Survey standard methods. The survey is likely to cover 150-300 waterbodies.

6.4.3 Option 3: Trial multimetric methods in a single region

A viable means of trialling the multimetric approach for any assemblage would be to undertake a pilot study in one or more regions of England and Wales. The main advantage of a regional study is that data set variability can be reduced, allowing a smaller number of sites to be used to trial the method.

6.4.4 Option 4: Test the approach using one taxonomic assemblage

It is recommended that for all still waters at least two biotic assemblages should be used to assess waterbody integrity. However, as noted in Section 6.3, it is quite feasible to test and develop multimetric methods based on a single taxonomic group and extend the method to other groups when appropriate. If only one taxonomic group is tested this should, ideally, be the macroinvertebrate assemblage or, if necessary, the aquatic macrophyte assemblage. Assessment methods based on fish and diatoms, the two other survey assemblages which have been shown to be potentially suitable, are currently relatively speculative and would require an initial investigative phase before testing.

6.4.5 Option 5: Start using only a minimally impacted baseline

It would be possible to stage the testing phase so that first the traunch of data collection was used to develop a national classification based on minimally impacted sites (cf. RIVPACS) and later stages used to collect impacted data for metric development. Our preference, however, would be to initially test the whole scheme regionally, rather than invest first in a national classification.

Note that this option is not recommended (see begin of Section 6.4).

6.4.6 Option 6: Undertake the baseline survey of reference sites using family/genus level data

In most of the taxonomically demanding groups (e.g. macroinvertebrates and periphyton) a considerable increase in resources is required to go from family-level to species-level identification. If it is certain that the method (e.g. the metrics) developed will be ultimately be used at family/genus level, then identifying the *initial* data sets at this level might be a cost effective option.

The main disadvantages of this approach are:

(i) there is no flexibility to mix identification levels where if appropriate (e.g. identify 'difficult' taxa such as Coleoptera to family level, but common snails to species level). This would reduce the number of potential metrics which could be developed,

(ii) it reduces the analytical power of the data and confidence in its precepts. It would not be possible, for example, to confirm that higher taxa (e.g. family level) indices *do* actually reflect species level data - an important step in ensuring that ecological integrity is adequately represented),

(iii) it reduces the potential to develop species level identification traits (e.g. species richness, rarity) which could be used for diagnosis or other purposes.

Note that this option is not recommended (see begin of Section 6.4).

6.5 Recommendations for methodological development

Based on the considerations developed in the previous section it is recommended that the next stage of the project should be to test the concept of multimetric assessment on a *major regional still water data set*, using a *macroinvertebrate assemblage* with the potential for addition of aquatic macrophytes. Where possible method development should capitalise on existing data, as long as this data has been collected using methods which are of direct relevance to final method development.

The choice of waterbody type on which the method is tested is an EA strategic decision. Our recommendation would be for either lakes, ponds or canals to be chosen for the test. Note however, that for a lake invertebrate survey to be undertaken, an initial investigative phase is required in order to develop appropriate sampling techniques.

6.5.1 The data required for trial methodology development

We would recommend a trial based on a 'regional' data set, (e.g. an extensive geographic area which includes a range of different topographies, landuses and geologies) using either newly collected data and/or existing data sets where available. The main stages in the multimetric method trial are:

1. Collect a 'regional' data set

The sites chosen should consist of:

- a range of minimally impacted sites, for the development of a baseline classification (e.g. allowing up to 5 end groups with 5 sites in each.
- sites experiencing a range of impacts from minimal to severe (e.g. giving the potential for 5-9 sites in each end group of the classification).

Because of the physical uniformity of canals, it is likely that fewer sites would be needed for the development of a classification within any major geographic region.

Data collected from each site needs to include:

i) A range of physical, chemical and biotic (e.g. shade) variables to assist in the classification of minimally impacted sites.

ii) Biological data from a number of assemblages, ideally including:

- invertebrate species level data, compatible with IFE RIVPACS samples (i.e. 3 minute hand-net samples),
- a second assemblage, preferably either aquatic macrophytes and/or diatoms,
- fish as an additional option for surveys of lakes (and potentially canals).

It would be advantageous if data was collected in more than one season or with replicated samples from each site, but for a trial this is not essential

2. *Analyse field data to produce a working method*

Following the collection of the data the following analytical steps are needed:

- Multivariate classification and delimitation of end group characteristics to form the basis of a classification.
- Assignment of impacted sites to classification end groups.
- Identification of a wide range of potential metrics for testing.
- Evaluation and normalisation of metrics.

3. *Future stages*

The course of future method development stages will inevitably depend on the outcome of the initial trials. However, assuming that the project demonstrates successfully that multimetric methods can be used on a regional data set, the method could be extended to include a wider range of sites, waterbody types or taxonomic groups as appropriate.

6.5.2 Future development needed for each waterbody type

This section outlines the preliminary work required to be able to undertake the trial assemblages recommended. The three waterbody types which could be used are considered in turn, with the options for undertaking that trial

The extent to which assemblage sampling methods can be applied 'off the peg' to each water body differs, and as a result there are differing preliminary set-up costs for each waterbody type. Thus in pond and ditches, the sampling methodologies which would be applied to surveying invertebrate and plant assemblages are already fairly clear cut, and it would be possible to initiate a trial of the multimetric approach with little preliminary work. In lakes, however, the details of sampling methodologies are not clear, and much more preliminary work is required.

The initial set-up requirements for developing fish, and particularly diatom, survey techniques are considerable and it is suggested that the initial testing of multimetric assessment methods does not await their development.

Choice of next-stage options are made more complex by the potential to build the method development phase on existing data. The potential use of relevant data sets is briefly outlined.

6.5.3 Lakes

Methods for sampling invertebrate, fish, diatom and to a lesser extent macrophyte assemblages are poorly developed in lakes, and a preliminary development phase is likely to be required before any of these could be used in a multimetric trial There are also a number of options for testing the method using existing data sets.

Preliminary work required prior to method development

Macroinvertebrates. An investigation of appropriate sampling techniques is required. This should include an assessment of the lake areas which should be sampled (e.g. littoral, sub-littoral, benthic), sampling techniques for deep water and sampling variability. Existing data may be available to facilitate this investigation. There should be a presumption that, as far as possible, data collection methods should be compatible with other habitats.

Aquatic macrophytes. A limited amount of methodological development is required to develop methods which could be widely applied (e.g. to other waterbodies, to ensure the potential for future compatibility).

Fish. A desk study is required to investigate the potential for using fish assemblages as an assessment tool in lakes. This should include (i) the potential to develop appropriate minimally impaired baselines, (ii) appropriate methodologies and (iii) the potential for use of data already routinely collected.

Diatoms. Diatom methods are the least well developed and a detailed investigation of assemblage viability and methodological techniques is required. Ideally a small research project is needed to investigate substrate and habitat variability, and the potential for useful assessments to be made using generic-level identification.

Summary of the steps required to trial a multimetric method on lakes

The two phases of development of a lake based trial would be:

Phase 1: Develop the sampling methodologies for one or more biotic assemblages as outlined above.

Phase 2: Collect a trial regional data set (unimpaired and impaired sites) for one or more assemblages and analyse to develop a multimetric assessment.

Alternative options

- Trial the multimetric method using regional CCW data for lakes (epilithic diatoms, littoral macroinvertebrates, macrophytes, fish). Extend the number of sites as necessary. Note however that the data may not be collected in optimal ways and that there are considerable resource implications associated with investing in data sets which subsequent analysis proves not to be relevant to the final method.
- Modify and extend the EA Lake Classification to include a greater number of biotic components. Note, however, that this would require modification of the hind casting baseline approach, since biotic parameters must be compared with current-day minimally impacted (or equivalent) sites.
- Develop fish methodologies in association with EA fisheries staff.
- Investigate the potential for using English Nature's lake macrophyte data as a basis for metric development (note however that many sites were located in Scotland).

6.5.3 Ponds and ditches

Methods for sampling invertebrate, and to a lesser extent, macrophyte communities are well developed and tested in ponds and ditches, so that a multimetric trial could be initiated with a very short set-up time. Despite the number of regional pond surveys, little macroinvertebrate data has been collected consistently using methods appropriate to the EA There are a therefore only a limited number of options for testing pond methods using existing data sets.

Preliminary work required prior to method development

Macroinvertebrates. Methods for collecting and analysing data are well developed, and can be based on hand net sampling techniques which are compatible with those used for RIVPACS. A small amount of development work is needed to ensure that there is close compatibility between river and still water methods (e.g. habitats sampled etc.).

Aquatic macrophytes. A small amount of methodological development is required to ensure that methods used are widely applicable (e.g. to other waterbodies, to ensure the potential for future compatibility).

Diatoms. Detailed investigation of assemblage viability and methodological techniques is required. As with the lake assessment, ideally a small research project is needed to investigate substrate and habitat variability, and the potential for useful assessments to be made using generic-level identification.

Steps required to trial a multimetric method

The two steps required for the testing of multimetric methods in ponds are:

Phase 1: brief development of sampling methodologies for macroinvertebrates (and potentially macrophytes) as outlined above,

Phase 2: collect a trial regional data set (unimpaired and impaired sites) based on macroinvertebrate (and potentially macrophyte) assemblages and analyse to develop a multimetric assessment.

Alternative options

There are three additional options which may be considered:

- Trial macroinvertebrate and macrophyte pond assemblages using Pond Action's regional data set for Oxfordshire (ca. 35 sites including physical and chemical data) supplementing with ca. 25 additional sites.
- Alternatively, use the National Pond Survey data set (at a national or regional level) as a basis for trial This data set currently comprises macroinvertebrate and macrophyte data (plus physical and chemical data) for ca. 180 minimally impaired ponds. 100 new 'impaired' sites will be added in summer 1996 and second 100 in 1997.
- For ditches it might be possible to supplement macroinvertebrate data with existing EA data already collected in routine invertebrate sampling programmes (note, however, that this is family level data). A more appropriate option might therefore be to use this information for testing the results of trial method development.

6.5.4 Canals

Preliminary work required prior to method testing

Macroinvertebrates. Methods for collecting and analysing macroinvertebrate data in canals are well developed, and can be based on hand net samples which are compatible with those used for RIVPACS. A minimal amount of development time is required to ensure maximal compatibility between river and still water methods (e.g. habitats sampled etc.).

Diatoms. Detailed investigation of assemblage viability and methodological techniques is required. As noted above further research is needed to develop diatom methods fully.

Fish. A desk study is required to investigate the potential for using fish assemblages as an assessment tool in canals. This should include (i) the potential to develop appropriate minimally impaired baselines, (ii) appropriate methodologies and (iii) the potential for use of data collected in routine monitoring programmes.

Steps required to trial a multimetric method

Phase 1: A very brief development phase for sampling methodologies for macroinvertebrates (and potentially macrophytes) as outlined above.

Phase 2: Collect a trial regional data set (unimpaired and impaired sites) based on macroinvertebrates assemblages and analyse to develop a multimetric assessment.

Alternative options

Options for method development are more limited than those for other habitat types and include:

- Use EA (family level) invertebrate data from routine monitoring of canals as the basis for assessment.
- Use EA canal data to test the results of method development trials.

6.5.5 Temporary and Brackish waters:

Preliminary work required prior to method development

Further detailed investigation of temporary and brackish waters is needed to: (i) assess the most viable combination of taxonomic groups with which to assess biological water quality (ii) appropriate survey methods for relevant taxa. This requires a preliminary comparative study of macroinvertebrate, microinvertebrates, macrophytes and periphyton, prior to method development

There appears to be very little existing data which could be used to aid in these assessments or to form the basis of method trials.

7. CONCLUSIONS AND RECOMMENDATIONS

7.1 General concept

The principle recommendation of this report is that the EA should develop a multimetric approach to monitoring still waters, using biotic assemblages to assess the general ecological quality and integrity of these systems.

In summary the multimetric approach involves:

1. Comparing selected biotic assemblages with least impacted present-day reference conditions.
2. Assessing the extent to which sites deviate from reference conditions using a variety of metrics (e.g. taxon richness, percentage sensitive groups, functional feeding groups).

   Together these metrics aim to summarise the integrity of the freshwater system.
3. Normalising metric data against the baseline and dividing it into simple scoring categories (1 = very poor to 5 = good)
4. Combining individual metric values to give a site integrity score. This score provides the basis for water quality assessment.

Diagnosis of the specific reasons for degradation, if recorded, is seen as a subsequent stage in the assessment process, and may employ any combination of an array of techniques (biological, chemical, historical) which are appropriate to specific waterbodies or specific legislative requirements.

7.2 Use of the scheme by the EA

The multimetric assessment approach proposed above fulfils all major EA operational and policy requirements for a biological assessment method for use in still waters. In particular:

1. The scheme is flexible, it can be applied across any region or area and adapted for use on any still waterbody type.
2. The wide range of parameters used to assess water quality can be summed, without loss of information, to give a single score which can form the basis for GQA assessment and the establishment of Water Quality Objectives.
3. Founded, as the method is, on principles of biodiversity and sustainability, the scheme addresses both the EA's pollution monitoring responsibilities and its general duty to have regard to the conservation of aquatic flora and fauna.
4. In terms of legislative requirements, the methodology can be applied to fulfil all biotic components of the draft Ecological Quality of Water Directive, including the requirement for comparisons with minimally impacted baseline conditions.

The objective of the method proposed, is to assesses the *overall* condition of freshwater ecosystems. The system does not, in itself, aim to provide a diagnosis of the cause of degradation. Indeed it is considered inappropriate for a general quality assessment method to be biased towards evaluation of a single or small number of pollutant impacts.

However, there is considerable potential for data which is collected using this scheme to be re-interpreted to diagnose the causes of degradation. This may be achieved both by inspection of individual metrics which make up the total integrity score, or by reanalysis to give pollution indices, such as trophic scores or acidification indices.

Reuse of data in this way, to provide information which will fulfil multiple end points, has the potential to make the scheme highly cost-effective. In addition, the method can be built up incrementally, minimising risk and initial costs in method development.

7.3 Developing the scheme in practice

A key decision required in order to implement the scheme is - which taxa should be monitored?

Recommended taxa for monitoring each waterbody type

The results of matrix analysis indicate that there are several candidate assemblages which could be used to gauge water quality and integrity. However, no one assemblage is able to fully represent all aspects of biotic integrity and to integrate the effects of all possible stresses. In general, the reliability and validity of assessments would therefore be enhanced by use of two biological assemblages. For lakes, which are both large waterbodies and virtually prohibitively difficult to restore once degraded, monitoring on the basis of *at least two* biotic assemblages is considered essential

In general, the best combination of two taxonomic groups in most waters is likely to be:
(i) a faunal assemblage - preferably invertebrates, but possibly fish in permanent waters and
(ii) a floral assemblage - either aquatic macrophytes or diatoms. Together these groups span a complimentary range of trophic levels, habitats and pollutant sensitivities and can effectively represent the integrity of the ecosystem.

The assemblages specifically recommended as a basis for monitoring in each waterbody type are:

Lakes	Macroinvertebrates	+	Aquatic macrophytes (Diatoms + Fish)[1]
Ponds	Macroinvertebrates	+	Aquatic macrophytes (or Diatoms)
Canals	Macroinvertebrates	+	(Diatoms or Fish)
Ditches	Macroinvertebrates	+	Aquatic macrophytes (or Diatoms)
Temporary waters	(Macroinvertebrates, Microinvertebrates, Macrophytes, Diatoms)		
Brackish waters	(Macroinvertebrates, Microinvertebrates, Macrophytes, Diatoms)		

In practice, of these assemblages, only macroinvertebrate communities could be considered to be an 'ideal' assessment group. Macrophytes are considered to be sub-optimal because their use is limited by poor temporal characteristics and the paucity of species found in naturally shallow, turbid and shaded waterbodies.

Periphyton (particularly diatoms) and fish are both promising assemblages for assessing biotic integrity, but both require further investigation to ensure their practical viability.

Brackish waters and temporary waters are inherently species-poor habitats. This combined with the paucity of information regarding their communities and impact sensitivity, makes it difficult to predict which (or how many) assemblages will have sufficient resolution to enable waterbody degradation to be adequately assessed.

Options for Phases II and III - a twin track approach

It is clear that the groups recommended above vary considerably in their potential for immediate development and testing. Thus macroinvertebrates assemblages could be rapidly applied as a basis for pond or ditch assessment. In contrast, a diatom-based assessment would require a prolonged set-up period during which the potential of the group was more fully evaluated.

Based on these findings we would recommend a twin-track approach to further methodological development.

[1] Assemblages in parenthesis are those for which methodological viability has not yet been fully established.

TRACK 1. Multimetric method testing and development

Test and begin development of a multimetric method based on macroinvertebrate assemblages in one or more of the following waterbodies: lakes, ponds, canals, ditches.

TRACK 2. Investigate the viability of other assemblages

(i) Investigate the comparative potential of macrophyte and diatom communities for application as a second assemblage in lakes, ponds, canals and ditches.

(ii) Use desk study information to investigate the potential for fish metrics to be developed for use in lakes and canals.

(iii) Investigate the most appropriate combination of taxa to use in assessment of brackish and temporary waters.

Other options

Many organisations hold still-water data sets. Where this data is of sufficient relevance and quality, it may be cost effective to utilise existing data sets as part of method development. The range of data which is currently available may influence the choice of waterbody and assemblages which are tested. Note, however, that there are considerable resource implications associated with investing in data sets which subsequent analysis proves not to be relevant to the final method.

The main data sets which are known to be directly relevant to the waterbodies and assemblages highlighted above and which have been collected in a systematic and repeatable manner include:

Countryside Council for Wales	Welsh Lake Survey: (water chemistry, phytoplankton, surface sediment diatoms, periphyton, zooplankton, macrophytes, macroinvertebrates and fish).
English Nature	West Midland Meres: 23 sites (water chemistry, phytoplankton, zooplankton, macrophytes, fish[1]).
Environment Agency	Norfolk Broads: 15 sites (water chemistry, phytoplankton, zooplankton, macrophytes, macroinvertebrates, fish).
	West Midlands canals: 120 sites (water chemistry, macro-invertebrates).
Pond Action	Oxfordshire Pond Survey: 35 sites (water chemistry, macroinvertebrates, macrophytes).
	National Pond Survey: 180 sites, rising to 400 1996/97 (water chemistry, macroinvertebrates, macrophytes, amphibians).

Testing the method

It is recommended that in order to develop the method, a trial is set up based on a major 'regional' dataset covering a variety of geological, topographic and landuse types. Regional data could be collected specifically for the project or could be partly based on exiting data sets such as those listed above.

It is suggested that at least 100 sites may be necessary to provide an adequate range of minimally impacted and impaired sites for method testing of lakes, ponds and ditches. The number required for canals is likely to be less.

[1] 10 sites only in the West Midlands Meres study have fish data.

The stages required to set up and implement a 'regional' trial for any waterbody are as follows:

1. Set up phase

(i) A detailed rationalisation of survey techniques, to ensure adequacy and compatibility with other methods (e.g. RIVPACS sampling techniques). This may only require a desk study (e.g. macroinvertebrates surveys in ponds or canals) or may necessitate field testing (e.g. macroinvertebrates in lakes)

(ii) Selection of potential metrics for later testing, and integration of this with survey technique rationalisation

(iii) Selection of physical and chemical variables to be measured

(iv) Selection of unimpaired and impaired reference sites including:

- consideration of the basis for selection (e.g. minimally impaired, use of informed opinion)
- basis for selection
- identification of the number and location of sites to adequately reflect natural and anthropogenic influences.

2. Collection of field data

Collection of biological, physical and chemical information from minimally impaired and impaired sites

3. Classification of unimpaired reference sites

Multivariate classification of biological communities from minimally impaired sites and analysis to identify the natural environmental parameters which characterise (i.e. can be used to predict) each community type. Testing using impaired sites and/or other data.

4. Development of viable metrics

Analysis of total data set within the context of the classification to (i) identify metrics which are effective in accurately expressing degradation and (ii) reject inappropriate metrics. Normalisation of metrics against the baseline sites and division into scoring categories

5. Testing

Validation using existing site data (e.g. EA held data sets) or collection of new data.

6. Recommendation for further work

Recommendations for the next phase of development: i.e. collection of further site data, collections of data on additional assemblages.

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APPENDICES

Appendix 1. Definitions of still waterbody types included in the assessment

Lakes	A body of water greater than 2ha in area (NRA 1994). Includes reservoirs, gravel pits, meres and broads
Permanent and semi permanent ponds	Waterbodies between $1m^2$ and 2ha in area which usually retains water throughout the year (Collinson *et al.* 1995). Includes both man-made and natural waterbodies.
Temporary waters	Waterbodies with a predictable dry phase, usually in the order of 3-8 months (Ward 1992).
Brackish waters	Pools and lagoons containing between 500 and 30,000 mgl^{-1} sodium chloride (Allaby 1985).
Canals	Artificial channels originally constructed for navigation purposes.
Ditches	Man-made drainage channels. Includes drains and rhines.

Appendix 2. Contacts made during Phase 1 Scoping Study

Europe

Name	Country	Address	Response
Georg Janauer	Austria	Inst. Plant Physiology, Univ. Vienna. Althustr, 14, A-10909 Vienna, Austria. Tel: + 313 361486 Fax: + 313 361776.	✔
Karel De Brabander	Belgium	van De Maelestraat 96, 9320 Erembodegem. Tel: + 053 72 62 11, Fax: + 053 71 10 78.	✔
Dr. P. Herman	Belgium	Minestere de la Region Wallone, DGRNE, Avenue Prince de Liege, B-5100 NAMUR (JAMBES), Belgium. Tel: + 32 81 32 56 09, Fax:+ 32 81 32 59 84.	
Prof. Niels De Pauw	Belgium	Department. Applied Ecology & Environmental Biology, J. Plateaustraat 22, B-9000 Gent, Belgium.	
Vladimir Korinek	Czech Republic	Hydrobiology, Charles University, Vinicna 7, Praha 2, CZ 12844. Fax: +42 299713, email: HYDROB@EARN. CVUT.C2.	✔
Mgr. Romana Zelenkova	Czech Republic	VUV, Podbabska 30, 160 00 PRAHA 6-Dejvice, Czech Republic.	✔
Mr T. Moth Iversen	Denmark	NERI, Vejlsoverj 25, DK-8600 Silkeborg. Tel: + 45 89 20 14 00, Fax:+ 45 89 20 14 14.	✔
Erik Mortensen	Denmark	Ministry of the EnvironmentNational Environmental Research Institute, PO Box 314, DK 8600 Silkeburg, Denmark. Tel: + 4589201400, Fax: + 4589201414.	
Merete Wichfeld	Denmark	Baunegaardesvej 73 DK-2900 Hellerud, Denmark.	
Pertti Heinonen	Finland	Finnish Environment Agency, PO Box 140, FIN-00251 Helsinki, Finland. email: pertti.heinonen@vyh.fi, Fax:+358 0 40 300 391, Tel:+ 358 040 327.	✔
Jirpa Herve	Finland	Central Finland Regional Environment Centre, PO Box 110, FIN-40101 JYVASKYLA, Finland.	
Esa Koskenniemi	Finland	WFREC, PO Box 262, FIN-6501 VAASA, Finland.	✔

Name	Country	Address	Response
Liisa Lepisto	Finland	Finnish Environment Agency, PO Box 140, FIN-00251, Helsinki, Finland.	✔
Marsa Ruoppa	Finland	Finnish Environment Agency, PO Box 140, FIN-00251 Helsinki, Finand.	
Yannick Galvin	France	Ministere de l'Environment/DE, 20 Avenue de Segure 75302, Paris 07 SR, France.	
M. J. Weber	France	Institut Francais de l Environnement, 17 rue des Hugeonots, F- 45058 Orleans Cedex 1, France.	
Dr. Jurgen Bohmer	Germany	Universiat Hohenheim, Instituet for Zoology, D-70593 Stuutgart, Germany.	
Ulrich Braukmann	Germany	Laudesanstalt fur mweltshultz, Baden - Wurttenberg, PO Box 210752, 76157 Karlstruhe, Germany.	
Ms. B. Clark	Germany	Umweltbundesamt, Postfach 33 00 22, 14191 Berlin, Germany.	
Dr. H. Engel	Germany	Bundenesanstalt fur Gewasserkunde, Kaiserin Augusta- Anlagen 15-17, Postfach 309, D-56068 KOBLENZ, Germany. Tel: + 49 261 13 06 229, Fax: + 49 261 13 06 280.	
Mrs. Aravantinou	Greece	Ministry of the Environment, 147 Patission Street, GR-11251 Athens, Greece.	
Mr. T. Ibsen	Iceland	Vonarstreati 4, IS-150 Reykavik, Iceland.	
Mr. L. Stapleton	Ireland	Environmental Protection Agency, Ardvacan, Wexford, Ireland.	
Prof. C. Gibson	N. Ireland	AERD, Dept. of the Agriculture, NI, New Forge Lane, Mallone Road, Belfast, N. Ireland. Tel: 01232 255509, Fax:01232 382244, email: c.gibson@uk.ac.qub.	✔
Peter Hale	N. Ireland	IRTU, 17 Antrim Road, Lisburn Co., Antrim, N. Ireland, BT28 3AL.	
Dr. Mike Meharge	N. Ireland	Dept. of the Environment, Countryside & Wildlife Division, Commonwealth House, 35 Castle Street, Belfast BT1 1GU, N. Ireland. Tel: 01232 661166/651165.	✔
Renato Baudo	Italy	CNR Istituto Italiano du Idribiologa, 28048 Verbiniss, Pallita, Italy.	

Name	Country	Address	Response
Mr. C. Pera	Italy	SINA- MInistero Dell'Ambiente, Via Della Ferratella in Laterano 33, I-00184 , Roma, Italy.	
M. J. Feltgen	Luxembourg	Ministere de l'Environnement, Montee de la Petrusse, L-2918 Luxembourg.	✔
Prof. Lucien Hoffmann CRP- CU	Luxembourg	Centre de recherche Public-Centre Universitaire, 162a, avenue de la Faiencerie, L-1511 Luxembourg. Tel: +4702611, Fax: + 470264.	✔
Mady Molitar	Luxembourg	Direction des Eaux at Forets, Boite postale 411, L 2014 Luxembourg. Tel: + 40 22 01. Fax: + 48 59 85.	✔
Ms. B. Kvavn	Norway	State Pollution Control Authority, PO Box 8100 Dep., N-0032 Oslo, Norway. Tel: 47 22 57 34 00, Fax: 47 22 67 67 06.	✔
Dr. Danuta Kedulska	Poland	Institute of Environmental Protection, Lake Protection Division, Kolektorska 4, 01-0692 Warsaw, Poland. Tel: 48 (22) 33 42 41 ext. 20, Fax 48 (22) 33 69 28.	✔
Jerzy Zerbe	Poland	Poznan University, Department of Water and Soil Analysis, Drymaly 24, 60613, Poland.	✔
Mrs. M. Gomes	Portugal	Minestero do Ambiente e dos Recursos Natuis, Direcao geral de Qualidide do Ambiente (SINAIA), Av Alm Gago Coutinho, 1000 Lisboa, Portugal.	✔
Dr. S. P. Klapwijk	The Netherlands	STOWA, P. O. Box 8090, NL-3503 RB, Utrecht, The Netherlands.	
Mr. A. Minderhoud	The Netherlands	National Institute of RIVM, Antonie van Leeuwenhoeklaan 9, Postbus 1, NL-3720 BA Bilthoven, The Netherlands.	
Dr. Harry. H. Tolkamp	The Netherlands	Zuiveringschap Limburg, Postbus, 314-6040 AH Roermond, The Netherlands. Tel: + 0475 394444, Fax: + 0475 311605.	
Abraham bij de Vaate	The Netherlands	Institute for inland water management and waste water treatment (RIZA), PO Box 17, NL-8200 AA, Lelysrad, The Netherlands. Tel: +313 20298701, Fax: + 313 20249218.	✔

Name	Country	Address	Response
Sr. A. Herrero	Spain	MOPTMA, Direccion de Politica Ambiental, Paseo de la Castellana 67, 2807, Madrid, Spain.	
Goran Dave	Sweden	SIS, Dept. of Zoophysiology, Hedicinaregartean 18, 41390, Goteburg, Sweden.	
Mr. E. Kvist	Sweden	National Focal Point to EEA, Swedish Environmental Protection Agency, S-106 48, Stockholm, Sweden. Tel + 46 8 698 1247, Fax + 46 8 698 1585, email: ebbe@environ.se.	✔
Dr. R. K. Johnson	Sweden	Swedish University of Agricultural Sciences, Dept. of Environmental Assessment-Biodiversity section, Sweden.	✔

North America

Name	Country	Address	Response
Prof. Norm Anderson	USA	Entomology Department, Oregon State University, Corvallis, OR.	✔
Dr. Sushil S. Dixit	Canada	Dept. of Biology, Queens University, Kingston, Canada, K7L 396, Tel: + 613 545-6160, Fax: + 613 545 6617, email: dixits@biology.queensu.ca.	✔
Tom Edmundson	USA	University of Washington, Civil Engineering Department, Seattle, WA. Tel: + 206 543 1669.	
Gretchen Hayslip	USA	United States Environmental Protection Agency (USEPA), 1200 Sixth Avenue (ES-097), Seattle, WA 98101.	✔
Dr. Paul Jepson	USA	Entomology Department, Oregon State University, Corvallis, OR.	✔
Mary Kantula	USA	USEPA, Corvallis, Oregon. Tel: + 754 4478.	
Dr. James R. Karr	USA	Institute for Environmental Studies, P.O. Box 352200,University of Washington, SeattleWA 98195, Tel: + 206 685 4784, Fax: + 206 543 2025, email:jrkarr@u. washington.edu.	✔
Arny Litt	USA	Civil Engineering Department, University of Washington, Seattle, WA. Tel: + 206 543 1623.	
Ken Ludwa	USA	Surface Water Management Division, Dept. of Public Works, 700 - 5th Ave, Suite 2200, Seattle, WA 98104. Tel: + 206 296 1911, Fax: + 206 296 8033.	✔
Dr. R. Naiman	USA	Centre for Streamside Studies University of Washington, Seattle, WA.	✔
Elissa Ostergaard	USA	Surface Water Management Division, Dept. of Public Works, 700 - 5th Ave, Suite 2200, Seattle, WA 98104, Tel: + 206 296 1911, Fax: + 206 296 8033, email: elissa@pwd.metrokc.gov.	✔
Gene Welch	USA	University of Wasington, Civil Engineering,Department. Tel: + 206 543 2632.	

Appendix 3. NRA biological monitoring of still waters

Anglian Region

Assemblage	No. of sites	Survey frequency	Monitoring strategy	Reason for monitoring	Notes
Lakes and reservoirs					
Bateriological	13	4 yr	Routine	Surface Water Directive	-
Blue-green algae	ca. 50 waters	ca. 1	Reactive	Public nuisance/Policy	-
Phytoplankton	20	12-52 yr	Routine	R&D/Regional Project	Species richness & abundance data
Zooplankton	17	ca. 18 yr	Routine	R&D/Regional Project	Species richness & abundance data
Macrophytes	17	1 yr	Routine	R&D/Regional Project	Species richness & abundance data
Macroinvertebrates	20	4 or 1 yr	Routine	R&D/Regional Project	Benthic samples, some littoral
Fish	20-30	1 yr	Reactive	By request	Enclosed waters only
Chlorophyll a	20	12-52 yr	Reactive	By request	-
Ditches					
RCS	-	3 yr rolling programme	-	-	-
Macrophytes	ca. 6	1-2 yr	Routine	UWWTD	-
Macroinvertebrates- for RIVPACS	ca. 6	2 yr	Routine	NRA policy	-
Macroinvertebrates - others	ca. 6	1 yr or 1 every 5 yrs	Routine	NRA policy? AMP	STW Monitoring/OSO's
Fish	-	1 every 3 yrs rolling programme	Routine	Strategic Fisheries Management	All major drainage systems - not small ditches

Anglian Region (continued)

Assemblage	No. of sites	Survey frequency	Monitoring strategy	Reason for Monitoring	Notes
Canals					
River corridor surveys	-	-	-	-	None-main rivers only
Macroinvertebrate - for RIVPACS	12	2 yr	Routine	NRA policy	-
Fish	All canals	1 every 3 years rolling programme	Routine	Strategic Fisheries Management	-
Permanent ponds					
Macrophytes	ca. 10	ca. 1	Reactive	Herbicide advice	Qualitative
Macroinvertebrates	ca. 5	ca. 1	Reactive	Public request	Marginal macroinvertebrates

Northumbria & Yorkshire Region

Assemblage	No. of sites	Survey frequency	Monitoring strategy	Reason for Monitoring	Notes
Lakes and reservoirs					
Blue-green algae	57	1 yr or as required	Reactive	Public nuisance	Weather dependent blooms Algal species counts
Phytoplankton	9	1 yr/ 1 fortnightly	Reactive or planned response	Public concern	-
Macrophytes	23	1 yr	Reactive or planned response	Often for weed control advice	-
Macroinvertebrates	4	1 yr	Reactive or planned response	Public concern or development control	-
Fish	5	1 yr	Reactive or planned response	Response to enquiries	Species composition, age, growth rates, disease, effects of pollution
Ditches					
River corridor surveys	10	2 yr	Reactive	Flood defence dredging; development control.	-
Macrophytes	2	1 yr	Planned response	Development control	-
Macroinvertebrates - for RIVPACS	16	2 yr	Routine	NRA policy	-
Macroinvertebrates - others	15	1 yr or as required	Reactive	Water quality requirements, pollution incidents, etc.	-

Northumbria & Yorkshire Region (continued)

Assemblage	No. of sites	Survey frequency	Monitoring strategy	Reason for Monitoring	Notes
Canals					
River corridor surveys	1	2 yr	Reactive	Development control Flood Defence work	-
Macrophytes	8	1 or 2 yr	Planned response	Legislative weed control	UWWTD
Macroinvertebrate - for RIVPACS	10	2 yr	Routine	NRA policy	-
Macroinvertebrates - others	4	1 yr	Reactive/Planned response	Water Quality/Conservation value	Pollution incident
Blue-green algae	5	1 yr	Reactive	Public complaint	-
Permanent ponds					
Macrophytes	70	1 yr	Reactive/Planned response	Development control; weed control advice	-
Macroinvertebrates	10	1 yr	Reactive/Planned response	Pollution; fisheries management	Occasional surveys to support planning applications, fisheries management advice etc.
Blue-green algae	8	1 yr	Reactive/Planned response	Public complaints	Algal species counts
Brackish waters					
Wansbeck Amenity Lake		10 days work during 96/97	Planned response	Public nuisance	Improve recreation use

North West Region

Assemblage	No. of sites	Survey frequency	Monitoring strategy	Reason for Monitoring	Notes
Lakes and reservoirs					
Blue-green algae	38	N/A/	Reactive	NRA policy	
Phytoplankton	8	2 yr	Routine	Development of management plans	
Fish	10 routine and lots reactive	6 as necessary	Routine and Reactive	Surveillance: Public	
Others (please state)	30	5 continuous others 2 yr	Routine	UWWTD; nitrate Directive	
Chlorophyll a					

Ditches. No data

Permanent ponds. No NRA/EA survey work. NRA funds the PondLife Project (Liverpool John Moores University)

Severn Trent Region

Assemblage	No. of sites	Survey frequency	Monitoring strategy	Reason for Monitoring	Notes
Lakes and reservoirs					
Bacteriological	24 15 canals	4	Routine	Statutory, operational requirements	-
Blue-green algae	110	100 x 1-2 yr 10 x 12 yr	Reactive Routine	Public nuisance/NRA policy	-
Macrophytes	-	-	Planned response	Background to management plans for NRA-owned sites	Winthorpe Lake, Dunham Lake
Macroinvertebrates	20	5 x 2 yr 15 x 1 yr	Reactive Planned response	Requests from fisheries; conservation/policy	Coed-y-Dinas Lake: development of invertebrate population on new gravel pit nature reserve
Fish	1 + Misc. lakes and pools	1 yr	Annual survey programme	Long term study into fish stocks and zander effects	Coombe Abbey Lake (LS Area)
Ditches					
Macrophytes	-	-	Planned response	-	Hatfield Ditches, presence absence, background for conservation project
Macroinvertebrates - others	44	x 2	Routine	GQA NRA policy	-

Severn Trent Region (continued)

Assemblage	No. of sites	Survey frequency	Monitoring strategy	Reason for Monitoring	Notes
Canals					
Bacteriological	15	4 yr	Routine	Statutory, operational requirements	-
River corridor surveys	2	-	Reactive and Routine	Strategic RCS	Beeston Canal Stroudwater Canal
Macrophytes	1	-	Planned response	Long term management data	Blunts Hill Canal
Macroinvertebrate - for RIVPACS	120	1-2 yr	Routine	NRA policy; GQA	-
Fish	Staff. and Worc. Canal (11 sites)	1 yr (1994/1995)	Planned response	Derogation of Fisheries Directive and post pollution assessment	Hydro acoustic techniques in use
	Glos-Sharpness Canal	Most years	Planned response	Derogation of fisheries Directive, fisheries work	-
	Oxford Canal	1990/1991	Annual survey programme	Zander removal and stock assessment	-
	Grand Union Canal (13 sites)	-	Planned response	Complaints of poor fishing; zander removal	-
	Other canals	-	-	Complaints of poor fishing	-
Amphibians	15	12 yr	-	Background to help determine long term management	Blunts Hill Canal
Chlorophyll a	15	-	Routine	UWWT Directive	Shropshire Union

Severn Trent Region (continued)

Assemblage	No. of sites	Survey frequency	Monitoring strategy	Reason for Monitoring	Notes
Permanent ponds					
Macrophytes	-	-	Planned response	Background for conservation project	Trent Valley Wetlands
				NVC assessment of over abstraction problem	Oxton Bogs
Macroinvertebrates	-	-	Special Study	To determine development of invertebrate population on new nature reserve and food availability for wildfowl and waders	Penarth Sewage Works Nature Reserve
Dragonflies	2	1	Planned response	Monitoring habitat creation	Four Ashes; Hadley Brook Pools
Brackish Waters			Planned response	Conservation request	Upton Warren SSSI

South Western Region

Assemblage	No. of sites	Survey frequency	Monitoring strategy	Reason for Monitoring	Notes
Lakes and reservoirs					
Blue-green algae	-	-	Reactive and routine	-	-
Macroinvertebrates	-	-	Reactive	Environmental Impact Assessment	Blashford Lakes
Fish	ca. 10 sites	3-5 yr	No formal sampling programme - sporadic sampling.	Fish stock assessments and fish quality assessments	Fisheries management
Dragonflies		'One-offs'	Reactive	Environmental Impact Assessment	Blashford Lakes
Ditches					
Macroinvertebrates - for RIVPACS	-	-	Reactive	Pollution incidents	-
Canals					
Fish	ca. 10 sites Tiverton (Grand Western) Canal and Exeter Canal	'One-offs'	Planned response	Perceived problem and as part of a fisheries management service offered to angling clubs	Objectives of surveys are for quantitative information on fish stocks i.e. numbers, biomass, species diversity, growth rates
Macroinvertebrate - for RIVPACS	-	-	Reactive	Pollution incidents	-

Permanent ponds: Reactive monitoring of macroinvertebrates

Southern Region

Assemblage	No. of sites	Survey frequency	Monitoring strategy	Reason for Monitoring	Notes
Lakes and reservoirs					
Blue-green algae	10-20	Once yr	Reactive	Requests from public or direct by Water Quality officers	-
Ditches					
River corridor surveys/RHS	50-60 km	Once yr - when stretch is in Flood defence dredging plans	Planned response	To develop conservation sensitive dredging plan and misc. others	-
Macroinvertebrates - for RIVPACS	Pevensey Levels	GQA years 2 p.a. (90, 95 etc.)	Routine	NRA policy - statutory	Many of GQA sites are ditches but usually with some flow
Canals					
Macroinvertebrates - for RIVPACS	11	GQA years 2 p.a. (90, 95 etc.)	Routine	NRA policy - statutory	-
Permanent ponds					
River corridor surveys/RHS	-	-	-	-	Ponds included if on line of a watercourse
Blue-green algae	ca. 5	1 yr	Reactive	Requests from public or direct by Water Quality officers	-

Thames Region

Assemblage	No. of sites	Survey frequency	Monitoring strategy	Reason for Monitoring	Notes
Lakes and reservoirs					
Bacteriological	-	-	Reactive	Public nuisance	In response to 'swimmers itch'
Blue-green algae	-	-	Reactive and routine	-	-
Macrophytes	-	Single 'one-off' sample	Reactive	Development proposals	-
Macroinvertebrates	-	Single 'one-off' sample	Reactive	Development proposals	-
Fish	20	Single 'one-off' sample	Reactive	Identification of cause of problem	-
Canals					
Macroinvertebrates	33	Single 'one-off' sample	Reactive	Water Quality Assessments	-
Chironomid pupal axuviae technique					
Ditches					
Macrophytes	-	Single 'one-off' sample	Reactive	Development proposals	-
Macroinvertebrates - for RIVPACS	-	Single 'one-off' sample	Reactive	Development proposals	-
Permanent ponds					
Macrophytes	-	-	Reactive	Pollution incidents	-
Macroinvertebrates	-	-	Reactive	Pollution incidents	-

Welsh Region

Assemblage	No. of sites	Survey frequency	Monitoring strategy	Reason for Monitoring	Notes
Lakes and reservoirs					
Blue-green algae	15 in SW area 20 in N area	ca. 1 yr	Reactive	Public nuisance NRA policy	
Phytoplankton	Varies	-	Reactive	-	
Zooplankton	Varies	-	Reactive	-	
Macrophytes	1 in N area	-	Reactive	Long term fisheries/liming project	None unless reactive. Area permissive project (Gamallt)
Macroinvertebrates	1 in N area	-	Reactive	Long term project	As above
Fish	(1 in N area)	1 yr	Reactive	Assessment of stocks	
Ditches					
Macroinvertebrates for RIVPACS	-	1 yr	-	-	Some GQA sites in the Lugg catchment
RCS/RHS	-	-	-	-	General Visual Assessments
Canals					
RCS/RHS	-	-	Reactive	Conservation advice to NRA staff & external	General Visual Assessments
Macroinvertebrate - for RIVPACS	-	-	Reactive	Pollution assessment	
Fish	-	-	Reactive	Pollution assessment	
Permanent ponds				To assess impact of pollution and to assess abundance of 'food for fish' for restocking of fish	

Appendix 4. Annex I and II from *Memorandum concerning the proposal for a Directive on the Ecological Quality of Water 1994*

Annex I: Ecological Water Quality - Working definitions

The ecological quality of water systems is determined by the state of those representative elements from the following list which are relevant to the individual waters concerned:

1. Dissolved oxygen
2. Concentrations of toxic or other harmful substances in water, sediment and biota.
3. Levels of disease in animal life, including fish, and in plant populations due to anthropogenic influence.
4. Diversity of invertebrate communities (planktonic and bottom-dwelling) and key species/taxa normally associated with the undisturbed condition of the ecosystem.
5. Diversity of aquatic plant communities, including key species/taxa normally associated with the undisturbed condition of the ecosystem, and the extent of macrophyte or algal growth due to elevated nutrient levels of anthropogenic origin.
6. The diversity of the fish population and key species/taxa normally associated with the undisturbed condition of the ecosystem. Migratory fish passage, insofar as it is influenced by human activity.
7. The diversity of the higher vertebrate community (amphibians, birds and mammals).
8. The structure and quality of the sediment and its ability to sustain the biological community in the ecosystem.
9. The riparian and coastal zones, including the biological community and the aesthetics of the site.

Annex II: Good Ecological Water Quality - specifications

Member States shall, based on the precautionary principle, fix the operational targets to be reached in accordance with this directive within the framework of representative elements from the following list which are relevant to the individual waters concerned.

1. Dissolved oxygen should allow survival and reproduction of indigenous animals.
2. Concentrations of toxic or other harmful substances in water, sediment and biota should not go beyond levels which have been demonstrated to pose no threat to aquatic species and should not prevent the normal uses of the water body.
3. There should be no evidence of elevated levels of disease in animal life, including fish, and plant life due to anthropogenic influence
4. The diversity of invertebrate communities (planktonic and bottom dwelling) should resemble that of similar water bodies with insignificant disturbance. Key species/taxa normally associated with the undisturbed condition of the ecosystem should be present.
5. The diversity of aquatic plant communities should resemble that of similar water bodies with insignificant anthropogenic disturbance. Key species/taxa normally associated with the undisturbed condition of the ecosystem should be present. There should be no evidence of excessive macrophytic or algal growth due to elevated nutrient levels of anthropogenic origin.
6. The diversity of fish communities should resemble that of similar water bodies with insignificant anthropogenic disturbance. Key species/taxa normally associated with the undisturbed condition of the ecosystem should be present. There should be no significant artificial hindrance to the passage of migratory fish.
7. Higher vertebrate life (amphibians, birds and mammals) should reflect that of similar water bodies with insignificant anthropogenic disturbance. Key species/taxa normally associated with the undisturbed condition of the ecosystem should be present.
8. Sediment structure and quality should allow the occurrence of biological communities typical of the region.
9. The status of riparian and coastal zones should, in non-urban areas, reflect either the absence of any significant influence by human activity, or care for the preservation of the biological community and for the aesthetic of the site.

Appendix 5. Methods assessment: ecological relevence and practical suitability

Appendix 5.1 Criteria for matrix anaysis: ecological relevence and practical suitability

Criteria for assessing method	***How measured***
Ecological relevance	
Species-richness in the waterbody type	Number of British species: 1=very low, 2=low 3= moderate, 4=high, 5=very high.
Does the group span a wide range of trophic levels?	1=ususally one level, 2=two levels, 3= three levels 4=four levels.
Is the group present in a wide range of waterbody habitats?	Presence of the taxa in range of waterbody habitats 1=restricted to one habitats , 2=small number of habitats 3=moderate number of habitats, 4=widespread 5=ubiquitous.
Extent to which the group reflects aquatic/wetland as opposed to terrestrial influences	1=predominantly terrestrial , 2=largely terrestrial 3=equal, 4=largely aquatic 5=totally aquatic.
General interest in and concern about the group (ecological, conservation and public)	1=little interest, 2=moderate, 3=moderately good, 4=good, 5= v. strong interest.
How well does the group integrate the environmental quality of the waterbody spatially?	1=v. poorly (eg affixed groups with specialised habitat requirements), 2= poorly 3=moderately, 4= well, 5= v.well (ie mobile groups which occur homogeneously).
How well does the group integrate environmental quality temporally?	1=v. poorly (eg either too slowly or too rapidly), 2= poorly 3=moderately, 4= well, 5= v.well (ie seasonally to annually).
Method/taxa currently believed to be indicative of the types of impact in still waters:	
• Nutrient enrichment	0=no, 1=little, 2=moderate, 3=moderately good, 4=good, 5= v.good, x=unknown
• Acidification/pH	0=no, 1=little, 2=moderate, 3=moderately good, 4=good, 5= v.good, x=unknown
• deoxygenation	0=no, 1=little, 2=moderate, 3=moderately good, 4=good, 5= v.good, x=unknown
• Biocides and other microorganics	0=no, 1=little, 2=moderate, 3=moderately good, 4=good, 5= v.good, x=unknown.
• Metals	0=no, 1=little, 2=moderate, 3=moderately good, 4=good, 5= v.good, x=unknown.
• Turbidity	0=no, 1=little, 2=moderate, 3=moderately good, 4=good, 5= v.good, x=unknown.
• Water level changes	0=no, 1=little, 2=moderate, 3=moderately good, 4=good, 5= v.good, x=unknown.
• Physical habitat damage	0=no, 1=little, 2=moderate, 3=moderately good, 4=good, 5= v.good, x=unknown.
• Biological impacts eg nuisance spp.	0=no, 1=little, 2=moderate, 3=moderately good, 4=good, 5=v.good, x=unknown.

Practical feasibility	
Are the taxa used for the method taxonomically well known? ie species-level identification is possible at most major life stages.	1= relatively little known, 2=moderate to poor 3= fairly good, 4=good knowledge, 5=relatively very well known.
How 'wide-ranging' is the group in terms of:	
• occurrence across the natural range of water chemistry types found in waterbodies eg acid/alkaline, high/low base levels	Range of water chemistry types: 1=v..limited range of water chemistry types 2=poor range, 3=moderate range, 4= good range 5=found across all water chemistry variants.
• occurrence across the natural range of physical variations found in the waterbody type eg shade, depth, altitude, area, sucessional stage.	1= v. poor range ie group is absent from a large number of important waterbody variants (eg in shaded *and* small ponds), 2= poor range, 3= moderate range, 4=good range 5= groups occurs in all waterbody variants even at the extremes of the range eg totally shaded pools).
Likely numerical abundance of individuals/taxa used in the environment	Typically and relatively: 1=very few, 2=small number 3=moderate, 4=abundant, 5= very abundant.
• seasonal occurrence	Surveys can be undertaken over what period of the year: 1=very restricted period (eg <1 month), 2=part of the year (1-6months), 3=most of the year (6-11months), 4=throughout the year but not always optimal, 5= throughout year.
• within season stability	Similar taxa throughout a sampling season: 1=highly variable seasonally, 2=variable 3=moderately variable, 4=stable 5=very stable.
• within waterbody uniformity	Heterogeneity of species/taxa within major waterbody zones (eg margin, littoral, benthic, planktonic): 1=taxa highly varied within waterbody zones, 2=taxa varied 3=taxa 3=moderately varied, 4=taxa fairly uniform, 5=taxa homogeneous throughout.

Appendix 5.2 Matrix analysis: ecological relevence and practical suitability

Lakes

Criteria for assessing method	Phytoplankton	Periphyton	Aquatic macrophytes	Emergent macrophytes	Micro-invertebrates	Macro-invertebrates	Fish	Amphibians	Birds	Mammals
Ecological relevance and importance of the taxon										
Species-richness in the waterbody under consideration	4	4	2	3	4	4	2	1	2	1
Does the group span a wide range of trophic levels?	1	1	1	1	2	2	3	3	3	3
Is the group present in a wide range of waterbody habitats?	3	4	3	2	4	4	4	2	5	4
How much is the group likely to reflect aquatic/wetland (as opposed to terrestrial) influences?	5	5	5	4	5	5	5	2	2	3
General interest and concern: ecological, conservation,public (not inc. introduced and nuicance spp.)	2	1	3	3	1	2	5	1	4	2
How well does the group integrate the environmental quality of the waterbody spatially?	3	2	3	1	3	3	4	3	3	3
How well does the group integrate environmental quality temporally?	1	3	3	3	2	5	5	3	3	3
How reponsive are the group to stressors (directly or indirectly):										
Nutrient enrichment	5	5	5	3	4	4	4	2	3	x
Acidification/pH	4	4	4	3	4	4	4	3	3	2
Deoxygenation	2	2	2	1	4	4	5	x	x	x
Biocides and other microorganics	3	3	3	3	3	3	3	3	3	3
Metals	2	3	3	2	3	3	3	x	x	x
Turbidity	5	5	5	2	3	3	3	2	2	2
Water level changes	2	2	2	3	2	2	2	2	2	2
Physical habitat damage	2	2	2	4	2	3	3	2	3	3
Biological impacts eg nuisance spp.	3	3	3	3	3	3	3	3	3	3
SCORE (% of maximum corrected for unknowns)	63%	62%	62%	52%	62%	68%	73%	46%	59%	53%

Lakes

Criteria for assessing method	Phytoplankton	Periphyton	Aquatic macrophytes	Emergent macrophytes	Micro-invertebrates	Macro-invertebrates	Fish	Amphibians	Birds	Mammals
Practical suitability of taxon (excluding time constraints)										
Is the group taxonomically well known?	2	2	4	4	2	3	4	4	5	5
Does the group occurr across the natural range of water chemistry types found in the waterbody type?	5	5	5	5	5	5	5	5	5	5
Does the group occurr across the natural range of physical variation found in the waterbody type?	5	5	5	5	5	5	4	5	5	5
What is the typical abundance of individuals?	5	5	3	3	5	4	2	1	1	1
How 'wide-ranging' is the group in terms of:										
• temporal occurance	3	4	3	3	4	5	5	2	3	3
• intra-season stability	1	3	5	4	2	3	4	4	2	3
• intra-habitat homogeneity	3	2	2	2	3	2	2	2	2	2
SCORE (% of maximum corrected for unknowns)	69%	74%	77%	74%	74%	77%	74%	66%	66%	69%
COMBINED SCORE FOR ECOLOGICAL RELEVENCE AND PRACTICAL USE	66%	68%	70%	63%	68%	73%	74%	56%	63%	61%

Lakes

Criteria for assessing method	Phytoplankton	Periphyton	Aquatic macrophytes	Emergent macrophytes	Micro-invertebrates	Macro-invertebrates	Fish	Amphibians	Birds	Mammals
Which tropic levels does the group occupy?										
Primary producer	**	**	**	**	-	-	-	-	-	-
Primary consumer	-	-	-	-	**	**	**	*	*	*
Secondary consumer	-	-	-	-	**	**	**	*	-	-
Tertiary consumer/top predator	-	-	-	-	-	-	**	-	*	*
Which waterbody microhabitats does the group occupy:										
Marginal wetland	-	*	*	**	-	*	-	**	**	**
Littoral	**	**	**	**	**	**	**	**	*	*
sublittoral	**	**	**	*	**	**	**	-	*	*
profundal	*	*	?	-	**	**	**	-	-	-
Water column	**	**	-	-	**	**	**	-	-	?

Ponds

Criteria for assessing method	Phytoplankton	Periphyton	Aquatic macrophytes	Emergent macrophytes	Micro-invertebrates	Macro-invertebrates	Fish	Amphibians	Birds	Mammals
Ecological relevance and importance of the taxon										
Species-richness in the waterbody under consideration	2	4	2	3	4	4	2	1	2	1
Does the group span a wide range of trophic levels?	1	1	1	1	2	2	3	3	3	3
Is the group present in a wide range of waterbody habitats?	2	4	3	3	3	4	4	2	5	4
How much is the group likely to reflect aquatic/wetland (as opposed to terrestrial) influences?	5	5	5	4	5	5	5	2	2	3
General interest and concern: ecological, conservation,public (not inc. introduced and nuicance spp.)	1	1	4	4	2	4	3	4	3	2
How well does the group integrate the environmental quality of the waterbody spatially?	4	2	3	2	3	3	3	3	3	3
How well does the group integrate environmental quality temporally?	1	3	3	3	2	5	5	3	3	3
How reponsive are the group to stressors (directly or indirectly):										
Nutrient enrichment	5	5	5	3	4	4	4	2	3	x
Acidification/pH	4	4	3	3	4	4	4	3	3	2
Deoxygenation	2	2	2	1	4	4	4	x	x	x
Biocides and other microorganics	3	3	3	3	3	3	3	3	3	3
Metals	2	3	3	2	2	3	3	x	x	x
Turbidity	5	5	5	2	3	3	3	2	2	2
Water level changes	2	3	3	2	3	3	3	2	1	2
Physical habitat damage	2	3	3	4	3	4	3	2	3	3
Biological impacts eg nuisance spp.	3	3	3	3	3	3	3	3	3	3
SCORE (% of maximum corrected for unknowns)	59%	65%	65%	54%	63%	73%	70%	51%	57%	53%

Ponds

Criteria for assessing method	Phytoplankton	Periphyton	Aquatic macrophytes	Emergent macrophytes	Micro-invertebrates	Macro-invertebrates	Fish	Amphibians	Birds	Mammals
Practical suitability of taxon (excluding time constraints)										
Is the group taxonomically well known?	2	2	4	4	2	3	4	4	5	5
Does the group occurr across the natural range of water chemistry types found in the waterbody type?	5	5	5	5	5	5	3	3	5	4
Does the group occurr across the natural range of physical variation found in the waterbody type?	4	5	3	3	5	5	2	3	3	2
What is the typical abundance of individuals?	4	5	3	3	5	4	2	1	1	1
How 'wide-ranging' is the group in terms of:										
• temporal occurance	4	4	3	3	4	5	5	2	3	3
• intra-season stability	1	3	5	4	2	3	4	4	2	3
• intra-habitat homogeneity	3	2	2	2	3	2	2	2	2	2
SCORE (% of maximum corrected for unknowns)	66%	74%	71%	69%	74%	77%	63%	54%	60%	57%
COMBINED SCORE FOR ECOLOGICAL RELEVENCE AND PRACTICAL USE	62%	69%	68%	62%	69%	75%	66%	53%	58%	55%

Ponds

Criteria for assessing method	Phytoplankton	Periphyton	Aquatic macrophytes	Emergent macrophytes	Micro-invertebrates	Macro-invertebrates	Fish	Amphibians	Birds	Mammals
Which tropic levels does the group occupy?										
Primary producer	**	**	**	**	-	-	-	-	-	-
Primary consumer	-	-	-	-	**	**	**	*	*	*
Secondary consumer	-	-	-	-	**	**	**	*	-	-
Tertiary consumer/top predator	-	-	-	-	-	-	**	-	*	*
Which waterbody microhabitats does the group occupy:										
Marginal wetland	-	*	*	**	-	*	-	**	**	**
Littoral	**	**	**	**	**	**	**	**	*	*
sublittoral	**	**	**	*	**	**	**	-	*	*
Water column	**	**	-	-	**	**	**	-	-	?

Temporary ponds

Criteria for assessing method	Phytoplankton	Periphyton	Aquatic macrophytes	Emergent macrophytes	Micro-invertebrates	Macro-invertebrates	Fish	Amphibians	Birds	Mammals
Ecological relevance and importance of the taxon										
Species-richness in the waterbody under consideration	1	3	1	3	3	3	0	1	0	1
Does the group span a wide range of trophic levels?	1	1	1	1	2	2	0	3	0	3
Is the group present in a wide range of waterbody habitats?	2	4	1	3	4	4	0	3	0	2
How much is the group likely to reflect aquatic/wetland (as opposed to terrestrial) influences?	5	5	5	4	5	5	0	2	0	3
General interest and concern: ecological, conservation,public (not inc. introduced and nuicance spp.)	1	2	1	3	2	3	0	1	0	1
How well does the group integrate the environmental quality of the waterbody spatially?	4	2	3	2	3	3	0	3	0	3
How well does the group integrate environmental quality temporally?	1	3	3	3	2	4	0	3	0	3
How reponsive are the group to stressors (directly or indirectly):										
Nutrient enrichment	5	5	5	4	4	4	0	2	0	x
Acidification/pH	4	4	4	4	4	4	0	3	0	2
Deoxygenation	x	x	x	x	x	x	0	x	0	x
Biocides and other microorganics	3	3	3	3	3	3	0	3	0	3
Metals	2	3	3	3	3	3	0	x	0	x
Turbidity	5	5	3	2	3	3	0	2	0	2
Water level changes	2	4	4	4	4	4	0	2	0	2
Physical habitat damage	4	4	4	4	4	4	0	3	0	4
Biological impacts eg nuisance spp.	x	x	x	x	x	x	0	x	0	x
SCORE (% of maximum corrected for unknowns)	58%	70%	59%	62%	67%	71%	0%	53%	0%	54%

Temporary ponds

Criteria for assessing method	Phytoplankton	Periphyton	Aquatic macrophytes	Emergent macrophytes	Micro-invertebrates	Macro-invertebrates	Fish	Amphibians	Birds	Mammals
Practical suitability of taxon (excluding time constraints)										
Is the group taxonomically well known?	2	2	4	4	2	3	0	4	0	5
Does the group occurr across the natural range of water chemistry types found in the waterbody type?	3	5	2	5	5	4	0	2	0	1
Does the group occurr across the natural range of physical variation found in the waterbody type?	2	5	1	4	5	4	0	2	0	1
What is the typical abundance of individuals?	3	4	2	4	4	3	0	1	0	1
How 'wide-ranging' is the group in terms of:										
• temporal occurance	4	4	3	3	4	5	0	2	0	3
• intra-season stability	1	3	5	4	2	3	0	4	0	3
• intra-habitat homogeneity	3	2	2	2	3	2	0	2	0	2
SCORE (% of maximum corrected for unknowns)	51%	71%	54%	74%	71%	69%	0%	49%	0%	46%
COMBINED SCORE FOR ECOLOGICAL RELEVENCE AND PRACTICAL USE	55%	70%	57%	68%	69%	70%	0%	51%	0%	50%

Temporary ponds **Criteria for assessing method**	Phytoplankton	Periphyton	Aquatic macrophytes	Emergent macrophytes	Micro-invertebrates	Macro-invertebrates	Fish	Amphibians	Birds	Mammals
Which tropic levels does the group occupy?										
Primary producer	**	**	**	**	-	-	-	-	-	-
Primary consumer	-	-	-	-	**	**	**	*	*	*
Secondary consumer	-	-	-	-	**	**	**	*	-	-
Tertiary consumer/top predator	-	-	-	-	-	-	**	-	*	*
Which waterbody microhabitats does the group occupy:										
Marginal wetland	-	*	*	**	-	*	-	**	**	**
Littoral	**	**	**	**	**	**	**	**	*	*
Water column	**	**	-	-	**	**	**	-	-	?

Ditches

Criteria for assessing method	Phytoplankton	Periphyton	Aquatic macrophytes	Emergent macrophytes	Micro-invertebrates	Macro-invertebrates	Fish	Amphibians	Birds	Mammals
Ecological relevance and importance of the taxon										
Species-richness in the waterbody under consideration	2	4	2	3	4	4	2	1	2	1
Does the group span a wide range of trophic levels?	1	1	1	1	2	2	3	3	3	3
Is the group present in a wide range of waterbody habitats?	2	4	3	3	3	4	3	2	5	4
How much is the group likely to reflect aquatic/wetland (as opposed to terrestrial) influences?	5	5	5	4	5	5	5	2	2	3
General interest and concern: ecological, conservation,public (not inc. introduced and nuicance spp.)	1	1	4	4	1	2	3	2	2	3
How well does the group integrate the environmental quality of the waterbody spatially?	4	2	3	2	3	3	4	3	3	3
How well does the group integrate environmental quality temporally?	1	3	3	3	2	5	5	3	3	3
How reponsive are the group to stressors (directly or indirectly):										
Nutrient enrichment	5	5	5	3	4	4	4	2	3	x
Acidification/pH	4	4	3	3	4	4	4	3	3	2
Deoxygenation	2	2	2	1	4	4	5	x	x	x
Biocides and other microorganics	3	3	3	3	3	3	3	3	3	3
Metals	2	3	3	2	2	3	3	x	x	x
Turbidity	3	3	3	2	3	3	3	2	2	2
Water level changes	2	3	3	2	3	3	3	2	1	2
Physical habitat damage	2	3	3	4	3	4	3	2	3	3
Biological impacts eg nuisance spp.	3	3	3	3	3	3	3	3	3	3
SCORE (% of maximum corrected for unknowns)	56%	62%	62%	54%	62%	71%	71%	48%	55%	55%

Ditches

Criteria for assessing method	Phytoplankton	Periphyton	Aquatic macrophytes	Emergent macrophytes	Micro-invertebrates	Macro-invertebrates	Fish	Amphibians	Birds	Mammals
Practical suitability of taxon (excluding time constraints)										
Is the group taxonomically well known?	2	2	4	4	2	3	4	4	5	5
Does the group occurr across the natural range of water chemistry types found in the waterbody type?	5	5	5	5	5	4	2	3	4	4
Does the group occurr across the natural range of physical variation found in the waterbody type?	5	5	2	4	5	4	2	2	4	4
What is the typical abundance of individuals?	5	5	3	3	5	4	1	1	1	1
How 'wide-ranging' is the group in terms of:										
• temporal occurance	4	4	3	3	4	5	5	2	3	3
• intra-season stability	1	3	5	4	2	3	4	4	2	3
• intra-habitat homogeneity	3	2	2	2	2	2	2	2	2	2
SCORE (% of maximum corrected for unknowns)	71%	74%	69%	71%	71%	71%	57%	51%	60%	63%
COMBINED SCORE FOR ECOLOGICAL RELEVENCE AND PRACTICAL USE	64%	68%	65%	63%	67%	71%	64%	50%	58%	59%

Ditches

Criteria for assessing method	Phytoplankton	Periphyton	Aquatic macrophytes	Emergent macrophytes	Micro-invertebrates	Macro-invertebrates	Fish	Amphibians	Birds	Mammals
Which tropic levels does the group occupy?										
Primary producer	**	**	**	**	-	-	-	-	-	-
Primary consumer	-	-	-	-	**	**	**	*	*	*
Secondary consumer	-	-	-	-	**	**	**	*	-	-
Tertiary consumer/top predator	-	-	-	-	-	-	**	-	*	*
Which waterbody microhabitats does the group occupy:										
Marginal wetland	-	*	*	**	-	*	-	**	**	**
Littoral	**	**	**	**	**	**	**	**	*	*
sublittoral	**	**	**	*	**	**	**	-	*	*
Water column	**	**	-	-	**	**	**	-	-	?

Canals

Criteria for assessing method	Phytoplankton	Periphyton	Aquatic macrophytes	Emergent macrophytes	Micro-invertebrates	Macro-invertebrates	Fish	Amphibians	Birds	Mammals
Ecological relevance and importance of the taxon										
Species-richness in the waterbody under consideration	2	4	2	3	4	4	2	1	2	1
Does the group span a wide range of trophic levels?	1	1	1	1	2	2	3	3	3	3
Is the group present in a wide range of waterbody habitats?	2	4	3	3	4	4	4	2	5	4
How much is the group likely to reflect aquatic/wetland (as opposed to terrestrial) influences?	5	5	5	4	5	5	5	2	2	3
General interest and concern: ecological, conservation,public (not inc. introduced and nuicance spp.)	1	1	2	3	1	2	4	1	3	2
How well does the group integrate the environmental quality of the waterbody spatially?	4	2	3	2	3	3	4	3	3	3
How well does the group integrate environmental quality temporally?	1	3	3	3	2	5	5	3	3	3
How reponsive are the group to stressors (directly or indirectly):										
Nutrient enrichment	5	5	5	3	4	4	4	2	3	x
Acidification/pH	4	4	3	3	4	4	4	3	3	2
Deoxygenation	2	2	2	1	4	4	5	x	x	x
Biocides and other microorganics	3	3	3	3	3	3	3	3	3	3
Metals	2	3	3	2	2	3	3	x	x	x
Turbidity	5	5	5	2	3	3	3	2	2	2
Water level changes	2	3	3	2	3	3	3	2	1	2
Physical habitat damage	2	3	3	4	3	4	3	2	3	3
Biological impacts eg nuisance spp.	3	3	3	3	3	3	3	3	3	3
SCORE (% of maximum corrected for unknowns)	59%	65%	62%	53%	63%	71%	73%	46%	57%	53%

Canals

Criteria for assessing method	Phytoplankton	Periphyton	Aquatic macrophytes	Emergent macrophytes	Micro-invertebrates	Macro-invertebrates	Fish	Amphibians	Birds	Mammals
Practical suitability of taxon (excluding time constraints)										
Is the group taxonomically well known?	2	2	4	4	2	3	4	4	5	5
Does the group occurr across the natural range of water chemistry types found in the waterbody type?	5	5	5	5	5	5	5	5	5	5
Does the group occurr across the natural range of physical variation found in the waterbody type?	5	5	4	4	5	5	5	2	5	2
What is the typical abundance of individuals?	5	5	3	3	5	4	2	1	1	1
How 'wide-ranging' is the group in terms of:										
• temporal occurance	4	4	3	3	4	5	5	2	3	3
• intra-season stability	1	3	5	4	2	3	4	4	2	3
• intra-habitat homogeneity	3	2	2	2	3	2	2	2	2	2
SCORE (% of maximum corrected for unknowns)	71%	74%	74%	71%	74%	77%	77%	57%	66%	60%
COMBINED SCORE FOR ECOLOGICAL RELEVENCE AND PRACTICAL USE	65%	69%	68%	62%	69%	74%	75%	52%	61%	57%

Canals

Criteria for assessing method	Phytoplankton	Periphyton	Aquatic macrophytes	Emergent macrophytes	Micro-invertebrates	Macro-invertebrates	Fish	Amphibians	Birds	Mammals
Which tropic levels does the group occupy?										
Primary producer	**	**	**	**	-	-	-	-	-	-
Primary consumer	-	-	-	-	**	**	**	*	*	*
Secondary consumer	-	-	-	-	**	**	**	*	-	-
Tertiary consumer/top predator	-	-	-	-	-	-	**	-	*	*
Which waterbody microhabitats does the group occupy:										
Marginal wetland	-	*	*	**	-	*	-	**	**	**
Littoral	**	**	**	**	**	**	**	**	*	*
sublittoral	**	**	**	*	**	**	**	-	*	*
Water column	**	**	-	-	**	**	**	-	-	?

Brackish waters

Criteria for assessing method	Phytoplankton	Periphyton	Aquatic macrophytes	Emergent macrophytes	Micro-invertebrates	Macro-invertebrates	Fish	Amphibians	Birds	Mammals
Ecological relevance and importance of the taxon										
Species-richness in the waterbody under consideration	2	3	2	3	3	3	1	1	1	1
Does the group span a wide range of trophic levels?	1	1	1	1	2	3	3	3	3	3
Is the group present in a wide range of waterbody habitats?	2	4	3	3	4	4	4	2	5	4
How much is the group likely to reflect aquatic/wetland (as opposed to terrestrial) influences?	5	5	5	4	5	5	5	2	2	3
General interest and concern: ecological, conservation,public (not inc. introduced and nuicance spp.)	1	1	2	2	1	2	2	1	1	1
How well does the group integrate the environmental quality of the waterbody spatially?	4	2	3	2	3	3	4	3	3	3
How well does the group integrate environmental quality temporally?	1	3	3	3	2	5	5	3	3	3
How reponsive are the group to stressors (directly or indirectly):										
Nutrient enrichment	4	4	4	2	3	3	3	1	2	x
Acidification/pH	x	x	x	x	x	x	x	x	x	x
Deoxygenation	2	2	2	1	4	4	5	x	x	x
Biocides and other microorganics	3	3	3	3	3	3	3	3	3	3
Metals	2	3	3	2	2	3	3	x	x	x
Turbidity	5	5	5	2	3	3	3	2	2	2
Water level changes	2	3	3	2	3	3	3	2	1	2
Physical habitat damage	2	3	3	4	3	4	3	2	3	3
Biological impacts eg nuisance spp.	3	3	3	3	3	3	3	3	3	3
SCORE (% of maximum corrected for unknowns)	52%	61%	61%	50%	59%	69%	68%	44%	50%	53%

Brackish waters

Criteria for assessing method	Phytoplankton	Periphyton	Aquatic macrophytes	Emergent macrophytes	Micro-invertebrates	Macro-invertebrates	Fish	Amphibians	Birds	Mammals
Practical suitability of taxon (excluding time constraints)										
Is the group taxonomically well known?	2	2	4	4	2	3	4	4	5	5
Does the group occurr across the natural range of water chemistry types found in the waterbody type?	5	5	3	5	5	5	3	3	4	3
Does the group occurr across the natural range of physical variation found in the waterbody type?	4	5	3	5	5	5	2	3	3	2
What is the typical abundance of individuals?	4	5	3	3	5	4	2	1	1	1
How 'wide-ranging' is the group in terms of:										
• temporal occurance	4	4	3	3	4	5	5	2	3	3
• intra-season stability	1	3	5	4	2	3	4	4	2	3
• intra-habitat homogeneity	3	2	2	2	3	2	2	2	2	2
SCORE (% of maximum corrected for unknowns)	66%	74%	66%	74%	74%	77%	63%	54%	57%	54%
COMBINED SCORE FOR ECOLOGICAL RELEVENCE AND PRACTICAL USE	59%	68%	63%	62%	67%	73%	65%	49%	54%	53%

Brackish waters

Criteria for assessing method	Phytoplankton	Periphyton	Aquatic macrophytes	Emergent macrophytes	Micro-invertebrates	Macro-invertebrates	Fish	Amphibians	Birds	Mammals
Which tropic levels does the group occupy?										
Primary producer	**	**	**	**	-	-	-	-	-	-
Primary consumer	-	-	-	-	**	**	**	*	*	*
Secondary consumer	-	-	-	-	**	**	**	*	-	-
Tertiary consumer/top predator	-	-	-	-	-	-	**	-	*	*
Which waterbody microhabitats does the group occupy:										
Marginal wetland	-	*	*	**	-	*	-	**	**	**
Littoral	**	**	**	**	**	**	**	**	*	*
sublittoral	**	**	**	*	**	**	**	-	*	*
Water column	**	**	-	-	**	**	**	-	-	?

Appendix 6 Survey methods used for costing

This appendix itemises the costs of sampling methods used in matrix analysis in Chapter 5. For each taxa-based method three types of costs are estimated:

(i) Equipment costs
Estimates are made of the cost of large capital items and consumables required for a staff member (or team) to undertake the survey. Costs are given as annual estimates which include all survey and analytical equipment. For large capital items annual costs are estimated as the total equipment cost averaged over 5 years.

(ii) Training time
Training time is assessed for each of the appropriately sized team, and envisages training an appropriate graduate. This is, however, difficult to estimate for a number of reasons: Where there is more than one member of a survey team, training is generally 'on the job' and requires considerably less real time/cost than in single-sampler training where staffing is essentially doubled-up. In addition, for specialist work, there may be a need for training courses, and a prolonged learning period during which the time taken to undertake tasks (e.g. identification of algae taxa) will be considerably increased. An attempt has been made to take all of these factors into account in the final estimate. Training is assumed to be 'on-the-job'. Trainer time during field work is therefore calculated as the extra time taken to explain methodologies whilst sampling is being undertaken (double, as a rule of thumb). Trainee times are reduced where field work would already require two staff to be present.

(iii) Staff time
Staff time includes the time taken to travel to sites, undertake the survey, complete any additional laboratory analysis, log and evaluate the data. Where field survey requires teams of two or more staff, survey time is multiplied accordingly.

A6.1 Chlorophyll a

Method

Chlorophyll a concentrations derived from a fixed volume sample taken at 0.5m. An annual average of 8 samples taken near lake/pond outlets (Cf. Johnes 199x) or from the bank (canals, ditches). Analysis with flurometric or spectrophotometric techniques.

Equipment costs

Large capital items: Sample storage (£1,000), Centrifuge (£2,000), Spectrophotometer etc. (£20,000). Total averaged over 5 years = £5,000/yr.

Small capital items and consumables : Life jackets etc., Sampling equipment, bottles, filtering equipment, gloves, acetone etc. Total = ca. £500/yr.

Staff time (per waterbody)

Field survey, including 2 hours average travel time = 3 hours. Laboratory analysis (inc. bottle washing etc.), data inputting and analysis = 1hour.

Skills training

Training in sampling and analysis skills: Trainee = 1 week. Trainer = 2.5 days.

A6.2 Phytoplankton

Survey method

Epilimnion grab samples taken at 0.5m depth from a number of samples taken along a transect at the approximate maximum depth of the waterbody. Microscopic identification and enumeration of cells to lowest viable taxonomic level. Screening level monitoring comprises a single sample taken in mid summer. More detailed monitoring comprises a seasonal or annual average of 8 samples (March to October). Greater survey time is required for larger waterbodies where a boat is needed (i.e. lakes and 50% of ponds).

Equipment costs

Large equipment items: Boat and motor (for lakes and large ponds) (£10,000). Inverted microscope (£10,000). Total averaged over 5 years = £4,000.

Small equipment items and consumables : Life jackets etc., Samplers, bottles, filtering equipment, gloves, acetone, laboratory consumables etc. Total = ca. £500/yr.

Staff time (per waterbody)

Field survey (including 2 hours average travel time): 3 hours for most waterbodies, 6 hours for two staff members if a boat is required. Laboratory analysis, data inputting and analysis for 2-3 samples per site: 7 hours.

Skills training

Field work: Trainee = 5 days. Trainer = 2.5 days.

Laboratory sorting, ID and data inputting for *family level* work: Trainee = 5weeks. Trainer =2 weeks. For *species* level ID: Trainee = 8 months. Trainer = 1 month.

A6.3 Periphyton

Survey method

Periphyton samples scraped from natural substrates. Spatial variation minimised by analysing a composite from sub-samples taken from a number of natural substrates at several random sampling points along a number of transects (number based on waterbody size and judgement). Samples taken during fixed annual seasons to minimise temporal variation. Samples (preserved-for non diatoms) mounted and proportional count made at family or lowest practicable taxonomic level.

Equipment costs

Large equipment items: Microscope (£10,000). Total averaged over 5 years = £2,000.

Small equipment items and consumables : Life jackets etc., Samplers, bottles, filtering equipment, gloves, acetone, laboratory consumables etc. Total = ca. £500/yr.

Staff time (per waterbody)

Field survey (including 2 hours average travel time): 3.5 hours for most waterbodies, 6 hours for large waterbodies. Laboratory analysis, data inputting and family level analysis for 3-4 samples per site = 4 hours. Species level ID = 12 hours per site.

Skills training

Field work: Trainee = 5 days. Trainer = 2.5 days.

Laboratory sorting, ID and data inputting for *family level* work: Trainee = 5weeks. Trainer =2 weeks. For *species* level ID: Trainee = 8 months. Trainer = 1 month.

A6.4 Submerged macrophytes

Survey method

Sampling undertaken by (i) grapnel trawls along transects through the littoral and sublittoral zone perpendicular to the shore (ii) walked transects along the shore line looking at shallow-growing aquatics and strandline debris. Information gathered relating to: species richness/rarity, percentage cover and wet weight. Sampling period restricted to the summer months.

Equipment costs

Large equipment items: Boat and motor (for lakes and large ponds) (£10,000), Binocular microscope (£5,000). Total averaged over 5 years = £3,000.

Small equipment items and consumables: Safety and sampling equipment, taxonomic keys etc. Total = ca. £400/yr.

Staff time (per waterbody)

Field survey (including 2 hours average travel time): 4.5 hours for most waterbodies, 7 hours for two staff members if a boat is required. Laboratory checking, data inputting and analysis 1.5 hours.

Skills training

Field, lab and data entry skills: Trainee = 3 weeks (2.5 weeks for lakes i.e. with boat work which already requires staff doubling-up). Trainer = 1week.

A6.5 Marginal macrophytes

Survey method

Surveys undertaken along 100m x 10m transects noting structure and species composition of the margins (or per waterbody for smaller waters). Sampling period restricted to the summer months.

Equipment costs

Large equipment items: Binocular microscope (£5,000). Total averaged over 5 years = £1,000.

Small equipment items and consumables: Safety and sampling equipment, taxonomic keys etc. Total = ca. £400/yr.

Staff time (per waterbody)

Field survey (including 2 hours average travel time): 4 hours for smaller waterbodies, 6 hours for lakes. Laboratory checking, data inputting and analysis 1.5 hours.

Skills training

Field, lab and data entry skills: Trainee = 5 weeks. Trainer = 1week. Plus a Training Course.

A6.6 Zooplankton

Method

Samples taken using a vertical tow plankton net taken through water column at mid waterbody. Sampled mid summer, or two to three times during the growing season. Individuals preserved and a subsample enumerated and identified to genera or species level.

Equipment costs

Large equipment items: Boat and motor (for lakes and large ponds) (£10,000), Microscope (£10,000). Total averaged over 5 years = £2,000 - £4,000.

Small equipment items and consumables : Life jackets etc., Samplers, bottles, filtering equipment, gloves etc., laboratory consumables etc. Total = ca. £500/yr.

Staff time (per waterbody)

Field survey (including 2 hours average travel time): 3 hours for most waterbodies, 6 hours for two staff members if a boat is required. Laboratory analysis, data inputting and analysis for 1-3 samples per site: 14 hours.

Skills training

Field work: Trainee = 5 days. Trainer = 2.5 days.

Laboratory sorting, ID and data inputting for *family level* work: Trainee = 5weeks. Trainer =2 weeks. For *species* level ID for groups with good taxonomic keys: Trainee = 8 months. Trainer = 1 month.

A6.7 Macroinvertebrates

Sampling method

The sampling method used for lake surveys separates littoral and benthic samples. For the other shallower and more easily accessible waterbody types littoral and benthic samples are combined.

Littoral/sublittoral samples: timed hand net samples taken in the littoral and sub-littoral zone. Spatial variation minimised by bulking a number of samples from waterbody microhabitats. Temporal variation minimised by sampling within fixed seasonal periods. Large waterbodies will require use of a boat. Samples laboratory sorted and preserved. Identified to family or species level.

Lake benthic samples: composite of multiple grab samples taken from different locations (along random transects) during fixed seasonal periods. Samples laboratory sorted and preserved. Identified to family or species level. Deeper ponds and some lakes would require use of a boat.

Equipment costs

Large equipment items: Boat and motor (for lakes and large ponds), (£10,000), Microscopes (£5,000 - £15,000 depending on taxonomic level), Total averaged over 5 years = £1,000 - £3,000.

Small equipment items and consumables: Safety and sampling equipment, taxonomic keys etc. Total = ca. £500/yr.

Staff time (pèr waterbody)
Field survey (including 2 hours average travel time): 3.5 hours for smaller waterbodies, 7 hours for a lake including benthic samples. Ponds are assumed to comprise 80% small waterbodies and 20% large waterbodies. Family level sorting and identification, laboratory checking, data inputting and analysis = 5 hours. Species level sorting and identification, data entry etc. = 14 hours.

Skills training

Field work: Trainee = 5 days. Trainer = 2.5 days.

Laboratory sorting, ID and data inputting for *family level* work: Trainee = 5weeks. Trainer =2 weeks. For *species* level ID for groups with good taxonomic keys: Trainee = 8 months. Trainer = 1 month.

A6.8 Fish

Sampling method

Electrofishing (5-10 transects) to estimate health and/or species size, abundance, biomass and composition. Could also use fykes, seines, gill net, traps or sonar in deep water. Details of species composition, biomass etc. are likely to require substantial field effort. Sampling gear is highly selective and, since no single method is appropriate for all depths and types of waterbody, fixed effort surveys using standard methodologies for similar waterbody types need to be applied.

For fish health (e.g. pathological anomalies) concentrate on a few species of bottom feeding fish.

Equipment costs

Boat and motor (£10,000), nets (£30,000), microscopes (£5,000 - 15,000), electrofishing boom (£10,000), sonar (£40,000). Total averaged over 5 years = £12,000 - £20,000.

<u>Small equipment items and consumables</u> : Life jackets etc., weighing and handling gear etc., laboratory consumables etc. Total = ca. £1,000/yr.

Staff time (per waterbody)
Small waterbody (e.g. canal section, small shallow lake) surveyed by sonar and or seine nets = equivalent of 1 day each for a four person team. Large waterbody surveyed by sonar and other techniques = 2 days for a four person team. Laboratory analysis, for gill net catches and health check = 1 day. Extra cost for scale analysis to assess age/growth rate. Data inputting and analysis for 2 hours.

Skills training

Field work training assuming that most fieldwork is undertaken on the job, that trainees already have a good knowlege of fish, that two staff members are trained at a time and four are trained altogether. Trainee time = 4 weeks each. Trainer =1.5 weeks for each pair.

Sonar skills for one team member: Trainee = 12 weeks. Trainer =1 week (only relevent to lake surveys).

A6.9 Amphibians

Survey methods

Netting plus observations of egg masses along the shoreline in spring.

Equipment costs

Small equipment items and consumables: Waterproofs and footwear, net, notebooks etc. Total = ca. £150/yr.

Staff time (per waterbody)

Field survey (including 2 hours average travel time): smaller waterbodies - 3.5 hours, lakes - 5 hours. Data entry and analysis 1 hour.

Skills training

Trainee = 2.5 weeks. Trainer = 1 week.

A6.10 Wetland birds

Survey methods

Transects along the shore in spring, recording birds seen or heard at set points. In large lakes, a canoe may be required to minimise disturbance.

Equipment costs

Large equipment items: Boat (i.e. canoe for large lakes) £1,000, binoculars and telescope: £2,000. Total averaged over 5 years = £400 - £600.

Small equipment items and consumables : Waterproofs and footwear, guides, notebooks etc. Total = ca. £150/yr.

Staff time (per waterbody)

Field survey (including 2 hours average travel time): 4 hours for smaller waterbodies, 7 hours for lakes. Data entry and analysis 1.5 hours.

Skills training

Assumes that a competent ornithologist is employed. Trainee = 2 weeks. Trainer = 1 week.

A6.11 Mammals

Survey methods

Trapping (with mandatory visits twice each day to check and empty traps) plus use of hair tubes for aerial (e.g. reed climbing) species combined with observation of spraints etc. Surveys undertaken in late spring.

Equipment costs

Large equipment items : 50 Longworth traps and hair tubes £3,000. Microscope £5,000. Total of £1,700/yr. over 5 years.

Small equipment items and consumables : Bait, bedding, scales, bags etc. Waterproofs and footwear, notebooks etc. Total = ca. £600/yr.

Staff time (per waterbody)

Method 1: Field trapping over 4 days plus laboratory identification of hair tube data, and data logging. Total = ca. 5 days.

Skills training

Field and laboratory skills including identification of hair tube samples. Trainee = 5 weeks. Trainer = 2 weeks.

Lakes	Capital items and consumables (pr yr)	Staff training time	Survey of one site (field and lab. work plus data inputting)
Chlorophyll a: 8 samples per year (bankside sample)	£ 5,500	£1,300	£731
Phytoplankton: single mid summer sample (species level ID)	£4,500	£30,900	£434
Phytoplankton: 8 samples	£4,500	£30,900	£3,474
Periphyton: single sample (family level)	£2,500	£7,300	£229
Periphyton: 3 samples (species level)	£2,500	£30,900	£1,234
Macrophytes: submerged plant transects	£3,400	£3,000	£320
Macrophytes: marginal plant survey	£1,400	£5,400	£160
Zooplankton: single mid summer sample (genus level ID)	£4,500	£7,300	£366
Zooplankton: 3 samples (ID to genus level)	£4,500	£7,300	£1,097
Zooplankton: 3 samples (ID to sp. level)	£4,500	£30,900	£1,783
Macroinvertebrates: littoral family level ID (single season)	£3,500	£7,300	£251
Macroinvertebrates: littoral family level ID (two seasons)	£3,500	£7,300	£503
Macroinvertebrates: littoral species level ID (single season)	£3,500	£30,900	£457
Macroinvertebrates: benthic family level ID (single season)	£3,500	£7,300	£434
Macroinvertebrates: benthic family level ID (two seasons)	£3,500	£7,300	£869
Macroinvertebrates: benthic species level ID (one season)	£3,500	£30,900	£640
Fish: electrofishing / netting (for a shallow lake)	£15,000	£12,600	£914
Fish : electrofishing / netting + sonar (for a deep lake)	£20,000	£23,200	£1,646
Amphibians: egg/lava search	£150	£3,000	£137
Wetland birds: transects	£550	£2,600	£160
Mammals: mixed methods	£1,700	£6,000	£800

Ponds	Capital items and consumables (pr yr)	Staff training time	Survey of one site (field and lab. work plus data inputting
Chlorophyll a: 8 samples per year (bankside sample)	£5,500	£1,300	£731
Phytoplankton: single mid summer sample	£4,500	£30,900	£251
Phytoplankton: 8 samples	£4,500	£30,900	£1,829
Periphyton: single sample (family level)	£2,500	£7,300	£171

Periphyton: 3 samples (species level)	£2,500	£30,900	£1,063
Macrophytes: submerged plant transects	£3,400	£3,400	£160
Macrophytes: marginal plant survey	£1,400	£5,400	£91
Zooplankton: single mid summer sample (genus level ID)	£4,500	£7,300	£160
Zooplankton: 3 samples (ID to genus level)	£4,500	£7,300	£480
Zooplankton: 3 samples (ID to spp. level)	£4,500	£30,900	£1,166
Macroinvertebrates: littoral and benthic sample. Family level ID (single season)	£3,500	£7,300	£194
Macroinvertebrates: littoral and benthic sample. Family level ID (two seasons)	£3,500	£7,300	£389
Macroinvertebrates: littoral and benthic sample. Species level ID (single season)	£3,500	£30,900	£400
Fish: electrofishing / netting (for a shallow lake)	£15,000	£12,600	£914
Amphibians: egg/lava search	£150	£3,000	£103
Wetland birds: transects	£550	£2,600	£91
Mammals: mixed methods	£1,700	£6,000	£800

Canals and ditches	Capital items and consumables (pr yr)	Staff training time	Survey of one site (field and lab. work plus data inputting
Chlorophyll a: 8 samples per year	£5,500	£1,300	£731
Phytoplankton: single mid summer sample	£2,500	£30,900	£251
Phytoplankton: 8 samples	£2,500	£30,900	£1,829
Periphyton: single sample (family level)	£2,500	£7,300	£171
Periphyton: 3 samples (species level)	£2,500	£30,900	£1,063
Macrophytes: submerged plant transects	£1,400	£3,400	£103
Macrophytes: marginal plant survey	£1,400	£4,600	£91
Zooplankton: single mid summer sample (genus level ID)	£2,000	£7,300	£160
Zooplankton: 3 samples (ID to genus level)	£2,000	£7,300	£480
Zooplankton: 3 samples (ID to spp level)	£2,000	£30,900	£1,166
Macroinvertebrates: littoral and benthic sample. Family level ID (single season)	£1,500	£7,300	£194
Macroinvertebrates: littoral and benthic sample. Family level ID (single season)	£1,500	£7,300	£389
Macroinvertebrates: littoral and benthic	£1,500	£30,900	£400

sample. Species level ID (single season)			
Fish: electrofishing / netting	£15,000	£12,600	£914
Amphibians: egg/lava search	£150	£3,000	£103
Wetland birds: transects	£550	£2,600	£91
Mammals: mixed methods	£1,700	£6,000	£800

Temporary ponds	Capital items and consumables (pr yr)	Staff training time	Survey of one site (field and lab. work plus data inputting
Chlorophyll a: 8 samples per year	£5,500	£1,300	£731
Phytoplankton: single mid summer sample	£2,500	£30,900	£251
Phytoplankton: 8 samples	£2,500	£30,900	£1,829
Periphyton: single sample (family level)	£2,500	£7,300	£171
Periphyton: 3 samples (species level)	£2,500	£30,900	£1,063
Macrophytes: submerged plant transects	£1,400	£1,300	£103
Macrophytes: marginal plant survey	£1,400	£5,400	£91
Zooplankton: single mid summer sample (genus level ID)	£2,000	£7,300	£160
Zooplankton: 3 samples (ID to genus level)	£2,000	£7,300	£480
Zooplankton: 3 samples (ID to spp. level)	£2,000	£30,900	£1,166
Macroinvertebrates: littoral and benthic sample. Family level ID (single season)	£1,500	£7,300	£194
Macroinvertebrates: littoral and benthic sample. Family level ID (two seasons)	£1,500	£7,300	£389
Macroinvertebrates: littoral and benthic sample. Species level ID (single season)	£1,500	£30,900	£400
Amphibians: egg/lava search	£150	£3,000	£103
Wetland birds: transects	£550	£2,600	£91