# British Ecological Society

# Ecological Issues No.1

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# River Water Quality

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# Foreword

The Council of the British Ecological Society has instituted this series of Ecological Issues booklets with two major objectives: to inform members (and others interested) about what is and is not established scientifically concerning specific ecological issues of general importance and to point out to funding agencies priorities for future research.

The Society is dedicated to the pursuit of ecology as a science, but it is accepted that scientific considerations abut intimately on the economic, social and political. In producing the booklets the Society is not intending to take on the role of an environmental pressure group, but is trying to help those who make decisions on environmental issues by providing an independent statement on the scientific background.

Virtually all ecological issues are very complex, and several different approaches may prove useful in gaining an understanding. In many cases, the validity of competing hypotheses is not settled at the present time. The problem of producing accounts 'with authority' can, therefore, be difficult. The Society's approach has been to ask a group of about ten to fifteen recognized experts in the field to produce each booklet, and invite any member of the Society to send comments to the group at the time of their deliberation. The draft booklet is then considered by the Society's Policy Review Group and issued with their agreement. It is hoped that in this way a balanced treatment of any given issue will emerge.

I congratulate the group who have produced this first booklet, covering the whole range of approaches to the problem of water quality. This issue affects us all, whether we are interested primarily in the quality of water that we drink, in the effects on fish and birds, or in the effects on communities of less obvious animals, plants and micro-organisms. Because of limiting funding in the past, and because of the inherent difficulty of investigating rivers, our knowledge of their ecology is still very incomplete and priorities for new work are set out clearly at the end of the booklet.

Peter J. Grubb
*President: British Ecological Society*
*August 1990*

# RIVER WATER QUALITY

CONTENTS

# 1. INTRODUCTION

## *1.1 The problem*

Because they flow, rivers and streams have been used by man, throughout history, for the transport of materials, removal of wastes and supply of potable water (Figure 1.1). Their extended form also means that they are shared by many people, across local and national boundaries. They are, therefore, vulnerable to abuse, and a legal framework has had to be evolved to protect them and their users.

In the UK, the polluting effects of sewage have, until recently, formed the focus of attention and have driven legislation. Indeed, rivers have been classified mainly by the extent of organic pollution (Section 2.2). Now though, as Figure 1.1 suggests, the problems are considerably more diverse. There is an increasing array of new and toxic compounds whose polluting effect is essentially unknown.

It is also the case that legislative controls have had to be put in place in advance of a thorough ecological understanding of flowing-water systems, and of the way that they can be stressed and disturbed. Much of the existing legislation, therefore, is based at best on intuitive ecological judgements and at worst on no ecology at all. Yet, with the mounting pressure on flowing-water systems, legislators and water quality managers more than ever need sound ecological advice on which to base their decisions. Our aim here is to develop a framework for this advice. We begin, in this chapter, by developing a general definition of river water quality and by drawing attention to salient ecological features of flowing-water systems.

## *1.2 What is water quality?*

The wholesomeness of water to drink, its appearance, its odour, the abundance of fish and other living things, have all formed part of a subjective assessment of what we mean by "water quality". In this sense the concept is a management tool rather than having an objective, absolute meaning. Neither is it synonymous with water chemistry; for water chemistry, even in "pristine" systems, can vary, for instance between very nearly pure water to being very

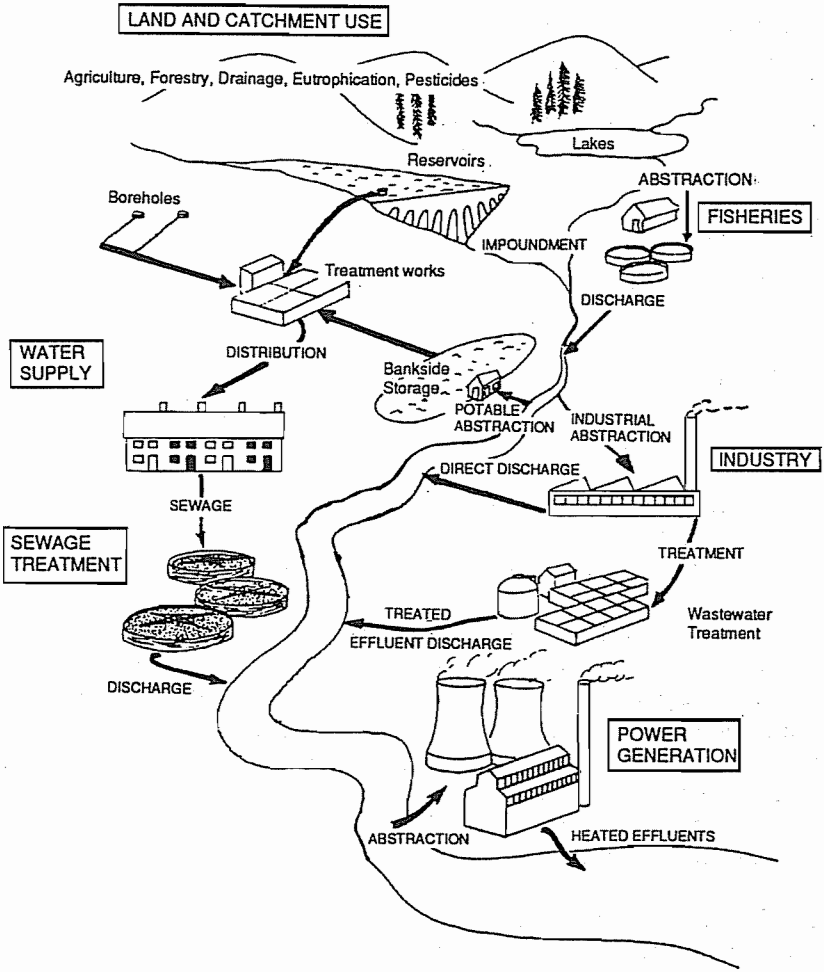


Fig. 1.1.
Diagrammatic representation of some of the uses of river quality
(after Hellawell, 1986)

heavily loaded with organic matter and anoxic. It would be hard to say what water quality "should" be in any particular circumstance other than by setting some target in management terms that is related to the use to which the system is put. For example, many urban rivers contain abandoned articles (e.g. scrap metal) that, whilst being unsightly, nevertheless provide microhabitats for biota and can increase diversity. Hence, in this case, water quality would be poor on human aesthetic grounds but unaffected or even enhanced in biological terms. Ecological science can assist river managers in the important task of assessing river water quality by predicting the ecological characteristics of undisturbed systems that would be expected in particular environments. These predictions must be based on a firm understanding of river ecology and should be cast in terms both of structural (species composition and relative abundance) and functional (energetics and nutrient dynamics) attributes of river communities. We take this up again in Chapter 4.

Environmental targets for water quality are implicitly biological in the broad sense. First, there are considerations of human health. Is water contact hazardous and how much treatment is necessary before it is safe to drink or use in various ways? Second, are fish and other "desirable" living things abundant? Is conservation furthered? Chemistry is important in that it enables environmental targets to be met and because various standard, chemical determinands have proved useful in surveillance because they are easy to measure and monitor. However, if water chemistry had no biological impact it would have little intrinsic importance in management. While this booklet is concerned with water quality in a more restricted ecological sense (we do not deal with direct impacts on health), the broader biological basis to the topic should be appreciated.

*1.3 Why is water quality important?*

Aquatic ecologists from the earliest days established correlations between the distribution and abundance of animals and plants and single chemical factors, such as calcium or oxygen concentration. It was realised that the relationships so derived rarely held when the study was extended into water bodies that differed in several other respects. Where a variety of factors differed, such as pH, oxygen,

calcium, temperature, substratum etc., consideration of single factors was uninformative. The availability of powerful computers and new statistical techniques, mainly borrowed from plant community ecology, meant that simultaneous multivariate analysis of whole communities and suites of environmental factors was possible. The results of several studies of invertebrates, on spatial scales ranging from groups of stations on one or a few rivers to many river systems, are now available in Britain. They are encouraging in that they implicate similar factors, e.g. alkalinity, plus some measure of distance from the source, in predicting community structure. Correlations between physicochemical variables and community structure will be considered further in Section 4.3; however, it is worth mentioning two caveats here. First, relatively little effort has been made to carry out essential experimental work to test the observed correlations. Second, features other than chemistry contribute to community structure and function and thus to the quality of our rivers. Our subsequent discussion is restricted to chemical factors simply due to limited space, and not because nothing else plays a role. For instance, fluctuations in flow are an ecological disturbance and may cause catastrophic changes in the density of river organisms. Flow also interacts with chemistry, and the concentration of many substances declines at peak discharge, whilst that of a few (particularly hydrogen ions and aluminium) may increase. For such determinands we should measure "flow-weighted" chemistry in order to characterize their effect. Channel structure, which may determine the availability of refugia for river animals during flow events, and temperature, modifiable by heated effluents and climatic change, are also important in determining river quality.

### *1.4 Fundamental features of river ecology*

Rivers are ecological systems in most respects like any others, sharing basic processes which occur throughout the biosphere. However, it is their peculiar features, along with their great practical importance, which lend them their particular significance. Rivers are extreme cases of "open" ecological systems, exchanging materials and energy with their surroundings to an enormous extent. They owe their existence to the excess of precipitation over evapotranspiration and it is these hydrological processes that link rivers so intimately with their catchments. This is why it is

inappropriate to apply the term "ecosystem" to rivers and streams themselves, for many of the most important pathways of energy flow and nutrient cycling occur across habitat boundaries, into the riparian strip or systems downstream. This dependence on external interactions influences processes at all levels of organization.

The "open" nature of river systems is very clearly reflected in the diverse pathways by which they receive energy. Primary production *in situ* is often restricted and research has stressed the role of energy imported from the catchment. This may be in the form of particulates, such as leaves from terrestrial plants, and, possibly, also dissolved organics, derived from groundwater. Therefore, much of the carbon in the living biomass of river systems has been fixed on land. Detritus of various kinds is of overwhelming importance and microbial communities provide the basis of most biological activity. In addition, much detritus may be exported, unmineralized, downstream, to be processed further or stored in flood-plains, estuaries, wetlands and shallow seas. Rivers truly have an ecological importance in the landscape extending beyond the small amounts of water stored in them.

Just as there can be tremendous chemical variations between rivers, there can also be considerable variation at specific locations within the same river through time (Fig. 1.2). "Average" chemistry is predominantly determined by the nature of the soils and geology in the catchment, and also by the proximity to the sea and by prevailing winds. Fluctuations in characteristics through time are due to a miscellany of processes, including seasonal shifts in biological activity in the catchment, and changes in the hydrological pathway by which run-off reaches the channel (for instance, as direct precipitation near the channel or as long-stored groundwater). Human activities now have a prime role in determining both long-term averages and fluctuations in water chemistry over much of the globe.

River organisms, therefore, are bathed in a medium of considerable chemical variability, and much of their evolution must have been driven by selection for tolerance to these conditions. Water chemistry is important at the ecophysiological level, affecting, for

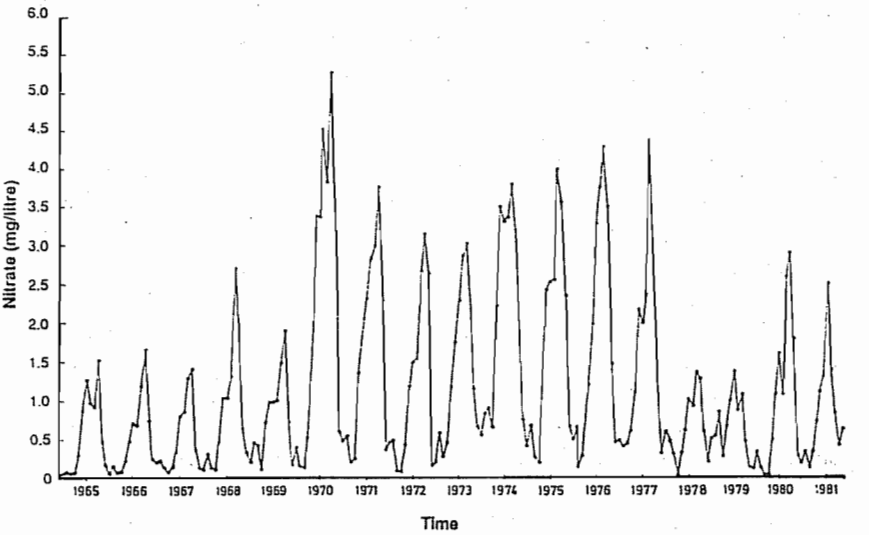


Fig. 1.2.
Monthly concentration of nitrate in stream water from Watershed 6
(after Likens, 1985)

instance, the energetic costs of ion balance, and its implications for individual growth, survival and reproduction are reflected in population processes and community structure and function. The chemical environment also has indirect "food-chain" effects at all ecological levels. For example, changes in hydrogen ion concentration influence the nature of the inorganic carbon supply to plants, through bicarbonate equilibria. This will affect primary producers, and, consequently, grazers and the rest of the food-web via "bottom-up" control. Similarly, loss of fish or other elements of the community by direct chemical effects, as in acidification, can have indirect effects elsewhere in the food-web. The relative importance of the direct, density-independent, intervention of environmental factors, including chemistry, into population and community processes in rivers is a controversial matter, reflecting a wider ecological debate.

Besides varying chemically, rivers are also dynamic in physical characteristics, particularly discharge. Communities within them have remarkable powers of recovery from physical disturbance; river organisms must be good colonizers. This is evident after spate when ecological structure and function can often recover very rapidly. Colonists can arrive by a variety of routes: downstream drift, upstream migration, vertical migration from the substratum and aerial invasion.

It may be fortunate, therefore, that because of features of their "normal" ecology, river communities can cope, to a limited extent, with chemical stress, physical disturbance and the processing of organic matter that has its origin outside the system - such as might arise from the effluents of sewage treatment works.

### *1.5 Objectives of this booklet*

Our overall objective in what follows is to point out the scientific basis of the three major aspects of river water management: i) the assessment of impact by pollution, ii) the prediction of hazard by some particular effluent, and iii) methods of controlling pollution and aiding recovery. To achieve this we have, in separate sections:

- Reviewed past and present techniques for measuring water quality.
- Presented an ecological assessment of current issues in water quality.
- Considered the potential role of ecological science in future developments in management. This role includes the provision of a stronger theoretical basis for legislation and water quality targets, and also of new or improved techniques and methods for the manager.
- Drawn specific conclusions and made recommendations aimed at policy makers, managers and ecologists.

# 2. PAST AND PRESENT TECHNIQUES OF MEASURING WATER QUALITY

*2.1 Monitoring programmes and ecotoxicology.*
In Great Britain there is an annual and 5-yearly national stock-taking of quality. The techniques used in these and more frequent, local exercises are similar but because of the wider scope of the national survey we concentrate on this.

Both local and national monitoring are concerned with judging the extent and seriousness of pollution, *after it has occurred.* There is another related but, nevertheless, distinct activity in which an attempt is made to assess the potential danger of releasing chemicals into the aquatic environment, so that appropriate constraints and safeguards can be applied to their production, distribution and release. Here, ecological information can be used

Table 2.1
River water quality classification scheme (DoE, 1986)

| Description | Class | Current potential use |
|---|---|---|
| Good quality | **1A**<br>BOD <1.5mg/l | Water of high quality suitable for potable supply abstractions; high class fisheries; high amenity value |
| | **1B**<br>BOD <2.0 mg/l | Water of less quality than Class 1A but usable for substantially the same purposes |
| Fair quality | **2**<br>BOD <5.0 mg/l | Waters suitable for potable supply after advanced treatment; supporting reasonably good coarse fisheries; moderate amenity value |
| Poor quality | **3** | Waters which are polluted to an extent that fish are absent or only sporadically present; may be used for low grade industrial abstraction purposes; considerable potential for further use if cleaned up |
| Bad quality | **4** | Waters which are grossly polluted and are likely to cause nuisance |

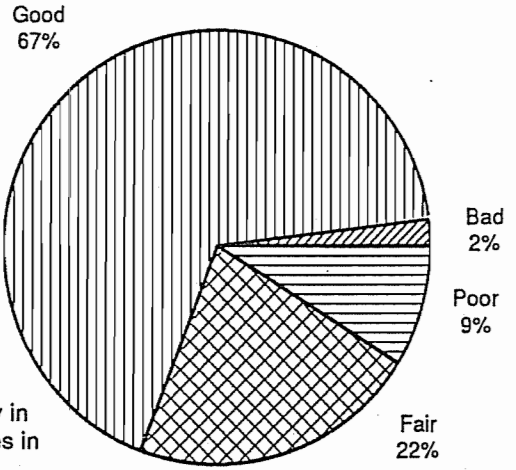


Fig. 2.1.
River water quality in England and Wales in 1985 (DoE, 1986)

predictively, to offer protection *before pollution has occurred*. This may be used in the registration of new chemicals, the establishment of environmental quality standards for existing chemicals and in the specification of legally enforced constraints (called "consents") on effluent discharges. This is the basis of ecotoxicology, about which much has been written (e.g. Moriarty, 1988) so we shall make only a few comments on this in Section 2.3.

*2.2 Monitoring water quality*

The water quality of rivers in England and Wales has been surveyed periodically since 1958 (quinquennial since 1970). The 1985 survey ranked water in several classes from Good Quality (Classes 1A and 1B) to Bad Quality (Class 4), largely on the basis of chemical criteria and particularly biochemical oxygen demand (BOD) (Table 2.1).

Results of the survey indicated that *c.* 90% of rivers surveyed (i.e. in terms of length; 38,896 km surveyed in total) were in the good-to-fair category (Fig. 2.1). However, for the first time since the survey had begun, there was a net deterioration in water quality. Surveys of Water Authorities carried out by Environmental Data Services (ENDS) showed a marked acceleration of the downward trend in 1986 (with a further net downgrading of 514 km), a slight

reversal in 1987 (net improvement of 140.5 km) (ENDS, 1988, Reports 162 & 166), but a continuing downward trend in 1989 (ENDS, 1990, Report 184). The next national survey is scheduled for 1990.

In Scotland, on the other hand, the 1985 survey showed an improving trend with 95% of Scotland's Rivers in Class 1 and 2 (HMSO, 1986). In Europe as a whole, a Water Research Centre (WRc) report prepared for the Department of the Environment (ENDS, 1987, Report 151) showed that about a quarter of the rivers are equivalent to Class 3/4. Eire, Scotland and Northern Ireland had the cleanest rivers; France, West Germany and Belgium were worse than England and Wales.

It is, of course, possible to argue about these statistics on various grounds. However, there are a number of important ecological problems with the classification schemes:

1. The chemical quality parameters used were those expected to be achieved in 95% of samples taken (i.e. 95 percentile values) over a calendar year (i.e. 1985). This is a tacit acceptance that one in twenty samples will exceed the prescribed standard. However, the occasional pulse at relatively high concentration may well have a disproportionately large adverse effect upon the biota.

2. Apart from information on fish, no biological data were used in the 1985 survey for England and Wales. The emphasis was on chemical assessment because this can provide a cost effective way of classifying rivers. However, there are several disadvantages in its use. First, there is strong reliance on the precision of the chemical data. Usually, these are based on spot samples taken relatively infrequently throughout the year at sites which can be considerable distances apart. However, variables such as BOD, dissolved oxygen ($O_2$) and ammonia ($NH_3$) can vary rapidly and considerably in time and space. Second, infrequent sampling often fails to detect intermittent pollution (see above). Third, chemical sampling will only detect those chemicals for which analyses are carried out. Fourth, low

concentrations of toxicants which may accumulate in biological systems will often be overlooked or fall outside the levels of sensitivity of existing analytical procedures. For example, some organotins and mothproofing agents are known to have toxic effects at concentrations below which they can be quantified accurately (WRc, 1988 a & b). Thus, the use of chemical criteria, more or less alone, to assess water quality may lead to over-optimistic classifications. Additionally, the emphasis on BOD, $O_2$ and $NH_3$ as criteria in the national water quality surveys, all of which are principally affected by changes in organic loading, implies that quality is being defined in relation to the effects of pollution by biodegradable organic matter. This will also tend to make the overall picture look better than it is.

All this argues for a greater emphasis on biological criteria in water quality assessments. Three main approaches have been adopted for water management purposes. The first is based on the differential sensitivity of species to a pollutant (biotic approach) and the second is based on changes in community diversity (diversity approach). The third, **R**iver **I**nvertebrate **P**rediction **A**nd **C**lassification **S**ystem (RIVPACS), combines an assessment in terms of both the types of species present and the relative abundances of families. Although these biological criteria could be applied to the whole community, it is usual for them to be applied only to part of it, e.g. algae or macrophytes or macroinvertebrates. The most favoured taxa, as far as rivers are concerned, are the benthic macroinvertebrates (see Section 4.3).

*Biotic indices*: The ones most frequently used in Britain have been the Trent Biotic Index (TBI), Chandler Biotic Score (CBS), and, more recently, Biological Monitoring Working Party (BMWP) score (reviewed in Metcalfe, 1989). All three are based largely on presumed relative tolerance of macroinvertebrates to organic pollution. The TBI and CBS require identification down to species whereas the BMWP requires animals to be identified only to family. The CBS alone takes abundance into account.

*Diversity indices*: The ones used most frequently (but in North America more than Britain) concentrate on two attributes of

community structure: species richness and equitability - the distribution of individuals amongst species. The inclusion of the equitability component is based on the view that communities with the most equitable distribution of individuals amongst species are the most diverse. A commonly used diversity index that combines these two attributes is the Shannon-Wiener diversity index (Wilhm, 1970).

*RIVPACS*: More recently, a different approach has been developed (Wright *et al.*,1989) that can be used to predict the fauna of a site using environmental variables (Fig. 2.2). The critical feature of this is that each prediction is specific to the environmental information which defines the site. The model predicts the type of community of macroinvertebrates that should occur at unperturbed sites. These provide "target communities" against which the combined effects of both physical and chemical stresses and subsequent biotic interactions on communities can be assessed. Predictions of the probability of species and family occurrence can be made in addition to predictions of relative abundance of families and the values of biotic indices. A comparison of observed values with predicted provides an environmental quality index.

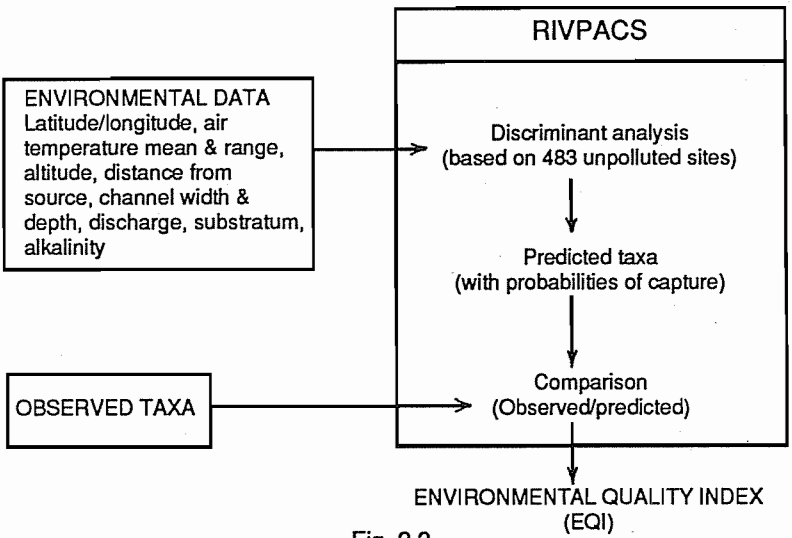


Fig. 2.2.
RIVPACS. A schematic representation

All the above techniques are concerned with assessments based on the structural properties of freshwater communities. In principle, at least, the state of these systems might also be assessed in functional terms, for example, by measuring primary and secondary production, respiration and nutrient cycling.

It should also be remembered that by accumulating chemicals, the bodies of various organisms can be good indicators of the presence of chemicals. Bioaccumulation studies have been carried out by many authors (see Hellawell, 1986) but their practical use in fresh waters has only recently started to become recognised (Kelly, 1989).

*2.3 Ecotoxicology*

The aim of ecotoxicology is to use the responses of particular biological test systems to chemicals to assess the ecological risks associated with them. Combined with information on their volume of production and use, and their stability in aquatic systems, it is possible to estimate the ecological hazard they would present. Which system should be used, though, and which tests should be carried out on them?

A recent survey of the aquatic ecotoxicological literature published over the last 20 years indicates that most of the work has been carried out on single "indicator" species in isolation, rather than on groups of interacting species (Maltby and Calow, 1989). Of the single species studies, *c.* 82% of the papers reported on invertebrates, 11% on fish, 4% on algae and only 3% on macrophytes. Of the work on invertebrates, tests involving cladocerans, particularly the water flea, *Daphnia*, dominated (Fig. 2.3).

Tests can either estimate concentrations required to kill a proportion (for statistical reasons usually 50%) of a test population (data summarized as $LC_{50}$), or the time required to kill a proportion (again usually 50%) of a test population ($LT_{50}$) or more subtle sublethal responses in terms of fecundity, developmental rates, and various physiological, cellular and molecular responses. It is likely that the sublethal effects are more ecologically relevant than the lethal responses. But, because of the speed and ease with which they can be carried out, the lethal tests have been employed most frequently (Fig. 2.4).

## *2.4. Summary*

In this chapter we have briefly reviewed how monitoring programmes and ecotoxicological tests are carried out. We have drawn attention to a number of problems that will be addressed in more detail later - particularly in Chapter 4. These are:

- The reliance of monitoring programmes on chemical criteria.
- A lack of understanding of the relationship between chemical criteria, even BOD, and biological impact.

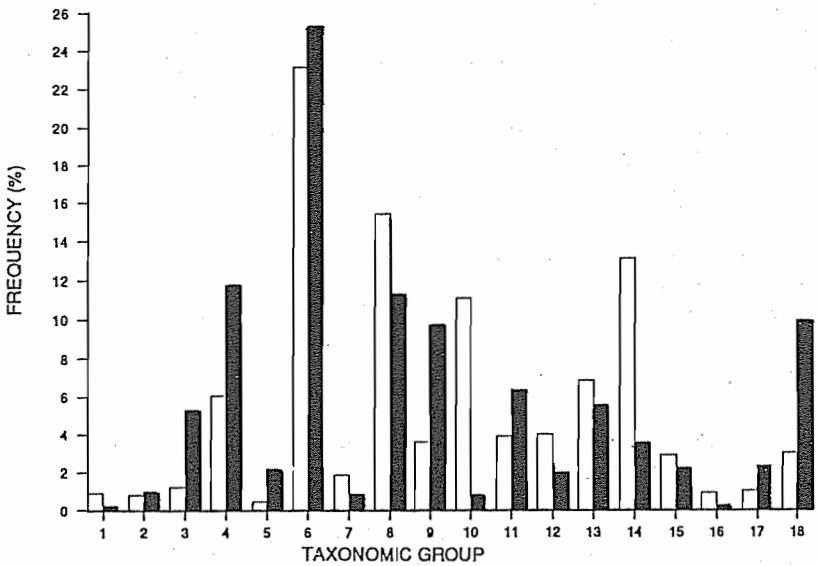


Fig. 2.3.

Frequency (%) of types of invertebrates used in single-species, lethal laboratory tests in papers published pre-1979 (n=909)(open blocks) and those published between 1979 and 1987 (n=707)(black blocks).

1=Protozoa; 2=Nematoda; 3=Platyhelminthes; 4=Oligochaeta; 5=Hirudinea; 6=Cladocera; 7=Copepoda; 8=Amphipoda; 9=Isopoda; 10=Decapoda; 11=Diptera; 12=Trichoptera; 13=Ephemeroptera; 14=Plecoptera; 15=Odonata; 16=Coleoptera; 17=Heteroptera; 18=Mollusca.

- The reliance of biological monitoring on: intuitive views about sensitivity of species; correlation analysis; structural (species composition) rather than functional attributes of river communities.
- The reliance of ecotoxicological tests on single-species, lethal tests involving a few, not necessarily representative, species.

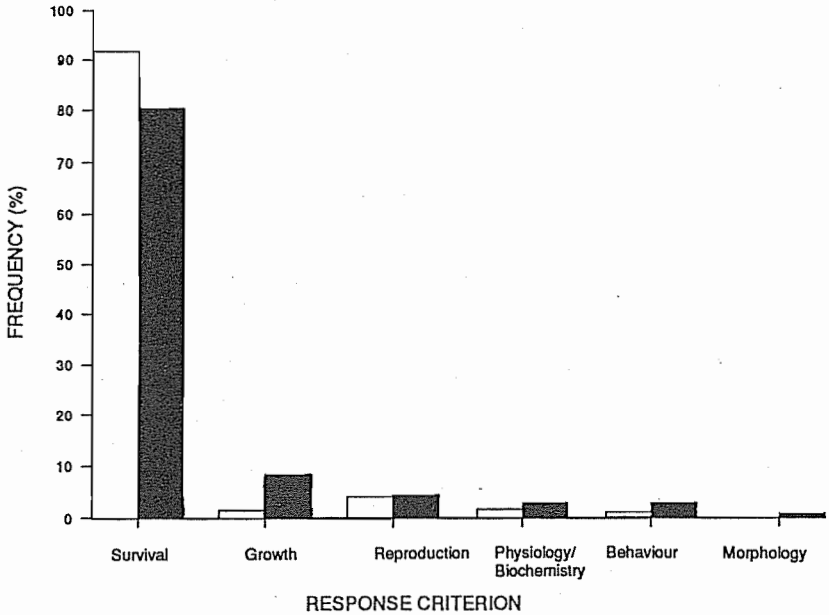


Fig. 2.4.
Frequency of criteria used in predictive, single-species laboratory tests in papers published pre-1979 (n=992)(open blocks) and those published between 1979 and 1987 (n=1175)(black blocks)

# 3. SOURCES OF WATER QUALITY DETERIORATION

## *3.1 Introduction*

As mentioned in Section 1.1, the types and ranges of pollutants to which flowing water systems can be exposed are now increasing with changes in industrial and agricultural practices. Here, we provide a basis for classifying these inputs and then give a brief description of some of the more important ones.

The origin of pollutants entering watercourses ranges from those that are predominantly point sources (e.g. effluent from sewage treatment works, fish farms and industry) to those that are mainly diffuse, such as agricultural run-off and acid deposition (Fig. 3.1). Whilst some are effectively continuous, there is an increasing realization that intermittent pollution events (episodes), exacerbated by heavy rainfall, may be equally damaging to ecosystems.

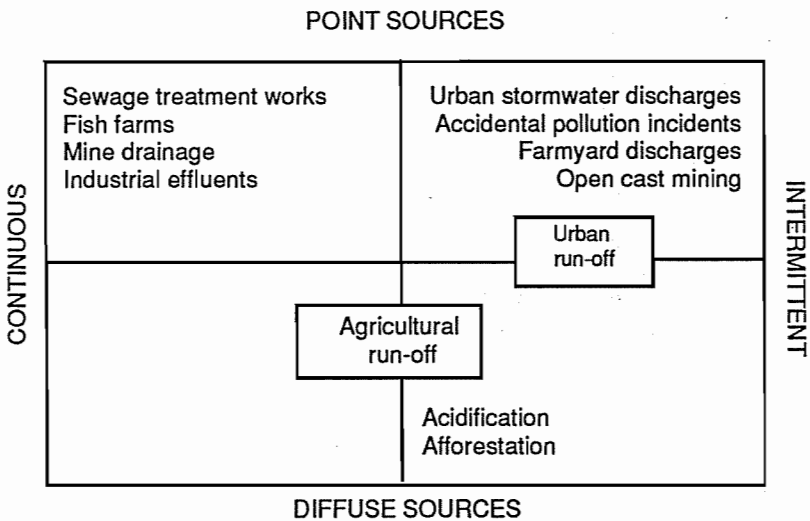


Fig. 3.1.
Diagrammatic representation of water pollution sources

*3.2 Sewage Treatment Works (STW)*

Although the quality of many major UK rivers has improved over recent years as a result of substantial capital investment in sewage treatment, water quality problems associated with old and overloaded STWs are still widespread. The impact of inadequately treated STW effluents on the ecology of receiving water has been extensively studied and comprehensively reviewed (Gray, 1989). Most studies have been concerned with the problems of organic enrichment and the characterization of biological communities associated with different degrees of organic pollution.

Conventional procedures for controlling STW effluent quality have involved the setting of limits on BOD and suspended solids. Other parameters, especially $NH_3$, may be included in these legal constraints (consents) depending on the size and importance of the works and the nature of discharges received by the sewerage system. BOD and suspended solids may provide an indication of the polluting "strength" of the effluent, but as already noted in the last chapter, insufficient is known about their ecological significance and this will be taken up later (Chapter 4).

Downstream of STW effluent discharges, population densities of certain organisms are often high, but diversity low. In general, there is an increase in biomass of those macroinvertebrates, such as chironomids and oligochaetes, that exploit fine particulate matter.

STW effluents often contain significantly increased nutrient concentrations, but limitations on these are not included in consents. Nutrient enrichment in standing fresh waters, particularly by nitrogen and phosphorus, has been the subject of extensive ecological investigation. However, relationships between nutrient loadings and ecosystem responses in rivers have not been established clearly.

*3.3 Urban stormwater discharges*

Storm-generated discharges in urban catchments are a persistent and widespread cause of poor river quality in the UK. The increasing extension of impermeable urban surfaces has resulted in rivers receiving run-off carrying a range of pollutants, including

toxic substances and solids. When stormwater enters combined sewer systems these may overflow and spill untreated sewage into rivers.

In the past, it has been assumed that the impact of stormwater will be offset by the increasing dilution capacity of the river during conditions of high storm flow. However, this has not been critically evaluated. A field study currently in progress by the WRc on Pendle Water, Lancashire, has suggested that episodic discharges from storm-sewer overflows have significant effects on benthic invertebrate communities. These seem to be both immediate, by removing invertebrates through physical disturbance during high flows, and persistent, following the settling of sewer-derived material.

*3.4 Industrial discharges*

The liquid effluents from industrial premises are disposed of by two main routes - either to a sewer for subsequent treatment and disposal through the local STWs, or by direct discharge after treatment on the factory site.

Discharges passing directly into rivers can have an immediate effect on water quality and aquatic life, and, in Britain, are controlled by the use of consenting procedures embodied within the Water Act 1989. This enables regulatory bodies, such as the newly established National Rivers Authority, to give consent to discharge and set conditions on the quality of discharges. Because the Authority has a statutory duty to protect aquatic life, it must satisfy itself that the consent conditions afford this protection.

In the past, conditions placed upon most industrial discharges were determined by reference to the effect that the main constituent(s) had on the watercourse. There is increasing concern, however, that most industrial discharges comprise a mixture of chemicals, some of which may act synergistically. It is, therefore, important to assess the toxicity to aquatic life of the whole effluent, and to take into account both lethal and sublethal effects. Moreover, in order to ensure that discharges comply with their consent conditions, on-site tests need to be developed.

*3.5 Farm wastes*

The last survey of river quality in England and Wales (Department of the Environment, 1986) showed pollution from farm wastes to be one of the commonest causes of water quality deterioration. Data published by the Water Authorities Association and the Ministry of Agriculture, Fisheries and Food (Water Authorities Association, 1988) indicated that between 1979 and 1988 the number of farm pollution incidents increased by 179%. Widespread problems with farm waste discharges have also been reported in Scotland and Northern Ireland. The most frequently reported causes are the release of animal slurry from stores, either deliberately or through structural failure, the discharge of silage effluents, and run-off from farmyards, particularly during heavy rainfall. Farm wastes have BOD and $NH_3$ concentrations many times those of sewage effluent and have caused widespread ecological damage in the upper reaches of many rivers which were previously of "good" quality.

Despite the seriousness of the problem, very little is known of how weather and specific farming practices affect the chemical and biological quality of receiving waters. However, a recent study by the WRc of a catchment affected by discharges from intensive dairy farms (Eastern Cleddau, North Wales) has demonstrated the marked influence of rainfall on receiving water quality. During storms, the washing of yards with inadequate drainage facilities results in significant peaks in $NH_3$ concentration and concomitant dissolved $O_2$ depressions in adjacent streams.

The consequences of intermittent exposure of aquatic biota to episodic pollution pulses have received little attention. A better understanding of how individual organisms and whole communities respond to and recover from these events is, therefore, an obvious research priority.

*3.6 Fish farming*

Production from freshwater fish farms in the UK has risen steadily over the past ten to fifteen years. In 1976, more than 2,000 tonnes of freshwater fish were produced; in 1988 the production of rainbow trout alone had risen to approximately 17,000 tonnes (Ministry of Agriculture, Fisheries and Food, personal communication).

Fish farms tend to be sited next to relatively unpolluted rivers, as a clean water supply is a prerequisite for successful fish rearing. Effluents from fish farms are qualitatively similar to those from STWs, with increased quantities of BOD, $NH_3$, suspended solids and nutrients resulting from the input of faecal matter and excess food. An important difference, however, is the presence of substances, such as fungicides, algicides, antibiotics, vitamins and dyes, that are used in husbandry. Quantitatively, the polluting effects of a fish farm can be far greater than a small STW. Solbé (1982) estimated that the BOD and suspended solids from an "average" farm were equivalent to the treated waste from 10,590 and 46,600 people respectively.

It is important to recognize that environmental concerns over fish farming embrace more than just water quality. The potential interaction between wild fish and those escaping from farms, the reduction in river flow between inlet and outlet points, and problems of disease transfer all require further attention.

*3.7 Acidification*

Many lakes and streams, particularly those receiving drainage from catchments with base-poor rocks in the upland regions of northern and western Britain, are now recognized as being acidified (United Kingdom Acid Waters Review Group, 1989). Awareness of the problem in the UK arose mainly from the need to make comparative surveys to those for acidified waters in Scandinavia. Prior to this, routine water quality surveys had largely failed to identify the problem, partly because pH had not always been measured, and also because some badly affected streams were too small to be included.

Examination of metal compounds, carbon particles and diatoms has enabled acidification histories to be reconstructed for lakes, and these have demonstrated the influence of acid deposition during the 20th century. Changes in land use may also lead to acidification and an important aim of current research is to quantify contributions from the various sources.

Many effects of acidification on stream chemistry, such as pulses of low pH, increased aluminium and heavy metals, have now been

established. These in turn affect the biota. For instance, invertebrate communities may be altered by the replacement of sensitive species with those more tolerant of low pH, or simply by species loss. Two main factors may be responsible for this. Water chemistry may be directly responsible through lethal or sublethal effects, or indirectly responsible through food-webs or by the absence of fish and reduced predation pressure (Sutcliffe and Hildrew, 1989).

The practice of liming as an antidote to acidification has been used extensively in Scandinavia and on a smaller scale in Britain. Although fish populations may be restored, it does not result in the restoration of the original plant and animal communities and if applied to the catchment may cause marked vegetational changes. The only satisfactory way to protect waters is to remove the causes of acidification by reducing the quantities of atmospheric pollutants. The United Kingdom Acid Waters Review Group (1989) has suggested that a reduction in acidic deposition of 90% from 1985 levels is required to return most surface waters to near pristine conditions, and a reduction of 30% would be required to prevent further deterioration.

*3.8 Forestry*

About 10% of the land in the United Kingdom has been planted with forest (Forestry Commission, 1985), including large tracts of coniferous species.

Conifer planting and forestry practice exert both physical and chemical influences on rivers and streams, again illustrating the importance of catchment processes on watercourse characteristics. The main areas of concern for water quality are increases in acidification, nutrients, particularly phosphorus, and sediments.

Deteriorating water quality in afforested catchments can have a severe impact on stream flora and fauna. There is evidence from a number of areas that water draining susceptible base-poor catchments with mature forest cover is more acid and contains higher concentrations of aluminium than that draining non-afforested catchments. This effect is probably due to the enhanced foraging of dry and occult (mist droplets) deposition from the atmosphere.

There is no evidence that forests significantly enhance acidification in areas of low acidic deposition, and neither is there likely to be much effect where dry or occult deposition contribute little to the total load. However, the potential effects of afforestation on water quality go beyond its implications for acidification. Increased sediment input as a result of deforestation may smother aquatic plants, invertebrates and spawning gravels, and when in suspension reduces light penetration and can cause increased scouring of bottom-dwelling organisms.

It is essential to manage existing forest plantations with maximum environmental sensitivity, and guidelines have been developed that apply to areas such as road construction, ploughing, ditching and the use of buffer strips alongside streams. However, the negative impacts of afforestation can best be avoided by the careful siting of new developments. Schemes such as that of the Welsh Water Authority (1987), in which limits to planting were recommended on the basis of the mean annual hardness of affected streams, require further testing and refinement.

### *3.9. Summary*

In this chapter we have sought to draw attention to, and classify, the main sources of water pollution. Again this underlines a series of problems that require ecological attention:

- Understanding the explicit link between organic loading (and hence BOD) and community effects.
- Recognizing that the impact of particular episodes of pollution can be considerable and persistent (again undermining the 95 percentile philosophy).
- Recognizing that effluents rarely contain single chemicals; rather they usually consist of complex, interacting mixtures.
- Recognizing that pollutants can interact in complex ways with their surroundings; e.g. adsorbing to sediments and releasing other toxic chemicals.
- Recognizing the "open" nature of river systems and the intimate ecological links between the river and its catchment.

# 4. FUTURE DEVELOPMENTS

## *4.1 Introduction*

Chapters 2 and 3 have been concerned with a description of past and present techniques for river water quality assessment and a review of some of the major types of water quality problems. The aim of this chapter is to indicate how ecology can be used to address some of these problems and to suggest ecologically-relevant tools for assessing and predicting the impact of changes in the water quality of freshwater systems.

Although ultimately interested in the effect of changes in water quality on whole ecosystems, we acknowledge that to have an effect at this level many other changes must have taken place at the community, population and individual levels. The types of effects that can occur at these different levels are illustrated in Figure 4.1. As we move up this hierarchy, the time between pollutant input and an effect being observed increases, from minutes or days for biochemical responses to months or years for ecosystem-level responses. Partly because of this, ecotoxicological studies designed to predict impact, tend to concentrate on responses of individual organisms (Section 2.3). In contrast, monitoring studies mainly concentrate on community-level effects. There is an obvious need to study interactions between the levels of organization. How are the sublethal effects of pollutants on individuals expressed at the population level or in communities? Conversely, could population or community-level changes brought about by pollution "cascade" down the hierarchy to affect the physiological ecology of individuals of one or more species? Community ecologists studying lakes have begun to explore such indirect or complex interactions (Kerfoot and Sih, 1987). If ecological science is to play an informed role in river management, we need to know if such interactions are important in running waters as well.

Two examples will illustrate the point. One of the major effects of organic pollution is a decrease in oxygen concentration. There are obvious changes in community structure below discharges from, for example, STWs, which presumably begin by killing susceptible individuals or by restricting their activity. Yet, surprisingly, the

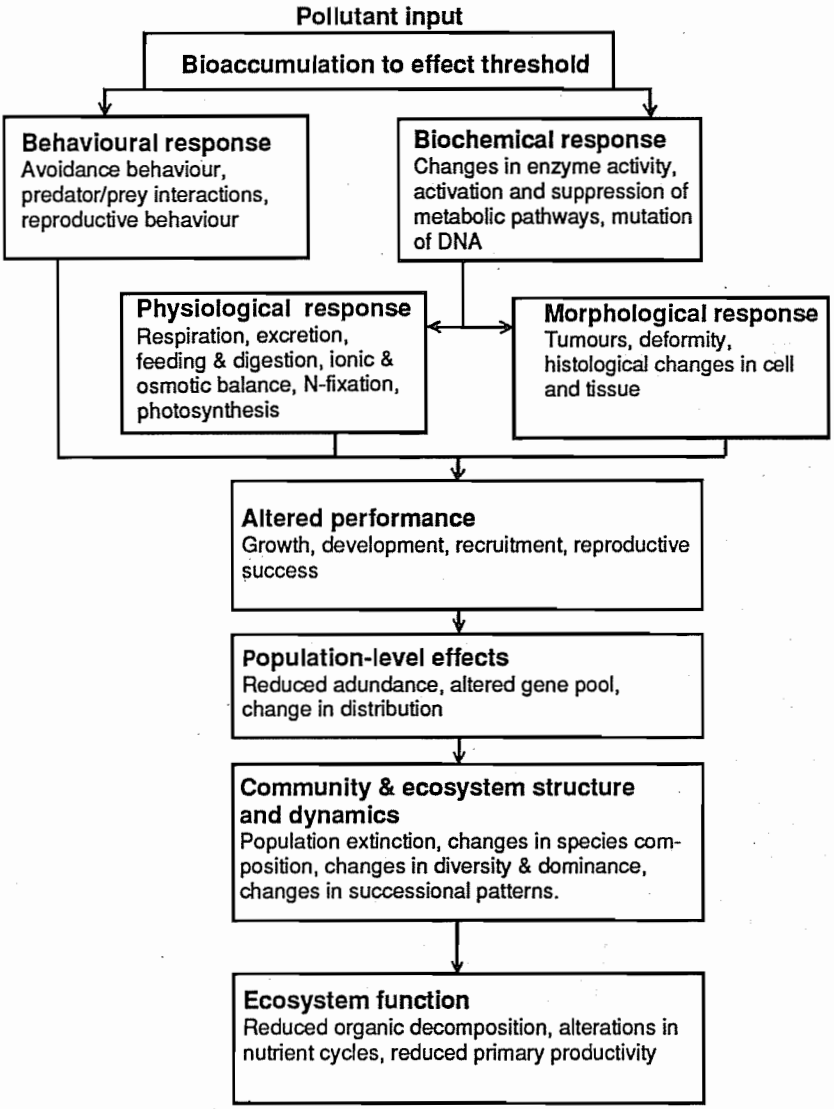


Fig. 4.1.
Flow-diagram of induced effects following exposure to toxic pollutants (after Sheehan *et al.*, 1984)

mechanism by which this leads to wholesale community change has not yet been demonstrated (p.19). The same is true for the effects of sediment. Sediments have trophic effects by increasing turbidity, cutting down light penetration and thus primary production, or by encouraging sediment feeding animals. They also have individual effects by acting as physical pollutants or by carrying sediment-bound toxicants. The relative importance of these mechanisms is poorly understood.

*4.2 Ecosystem-level research*

The study of the ecosystem as a unit in pollution studies is based on the view that the system as a whole possesses properties which are more than a simple summation of its component populations and processes. The impacts of pollutants on a single species may have a negligible effect on the system as a whole. This inertia (i.e. capacity of the system to resist change) is a function of several basic properties including: the tolerance of the organisms to environmental fluctuations; the extent to which one function is performed by many groups; the capacity of ecological systems for self-purification and their chemical buffering capacity.

Therefore, to understand fully the effect of changes in water quality on running water ecosystems it is necessary to conduct ecosystem studies of the type carried out on the Hubbard Brook Forest catchment, New England, U.S.A. (Likens, 1985). The objectives of this study have been to identify and quantify patterns of ecosystem behaviour for an undisturbed forest catchment. Armed with this basic information, anthropogenic impacts, either caused by changes in land management (clear-felling) or water quality (acidification), have been evaluated experimentally. This has provided many useful insights into the structure and functioning of riverine systems and their responses to perturbations; for example, they have established the critical dependency of water quality and biota on the nature of the catchment, its geology, soils, vegetation and land-use. One of the most important findings is that fluctuations in water chemistry, particularly in the major cations and anions, are much less in undisturbed than in disturbed catchments. Two reasons for the success of the Hubbard Brook Study are that the approach taken has been multidisciplinary and long-term (>20 years).

In Britain, ecosystem studies have been performed on the Llyn Brianne (Wales), Loch Fleet and Loch Dee (Scotland) catchments (Burns *et al.*, 1984; Stoner *et al.*, 1984; CEGB, 1985; Ormerod *et al.*, 1987). These are all upland catchments where the major ecological problems have been related to acidification and afforestation. Although these studies have provided useful information on the effects of these particular problems in upland regions, what is required is a series of studies on a range of river ecosystems, both lowland and upland. These would provide basic information on the factors important in the functioning of such systems and how they interact with the catchment.

To perform ecosystem studies that experimentally investigate the impact of changes in land use on water quality, and establish limits to background variation, a range of replicated treatments in different catchments is required. However, such a scheme is probably impractical, especially for lowland systems where the investigator would require some degree of control over the whole catchment. We therefore propose two alternatives:

a) Establishment of suites of sites for long-term monitoring covering a range of land uses and river types. Such a monitoring programme would provide a large database consisting of physical, chemical and biological information that could be used to tease out those factors that are important in determining freshwater communities. In addition, these studies would provide information on the "natural" variation in ecosystems as well as providing a measure of the response time of river communities to catchment changes. Such a scheme has been initiated in Norway (Saltveit *et al.*, 1989).
b) The above approach will not provide unequivocal information on cause-effect relationships. Such insights can only be gained by designing experiments to test particular hypotheses. Odum (1985) made a number of predictions concerning stressed ecosystems, some of which are summarized in Table 4.1. These could form the basis of such an experimental analysis. To carry this out effectively we propose that there should be a national mesocosm facility consisting of a large number of replicated channels which can be used for such experiments.

Table 4.1.
Trends expected in stressed ecosystems (Odum, 1985)

| | |
|---|---|
| | **Energetics** |
| 1. | Community respiration increases |
| 2. | P/R (i.e. production/respiration) ratio becomes unbalanced |
| 3. | Exported or unused primary production increases |
| | **Nutrient cycling** |
| 4. | Nutrient turnover increases |
| 5. | Horizontal transport increases and vertical cycling of nutrients decreases |
| 6. | Nutrient loss increases |
| | **Community structure** |
| 7. | Size of organisms decreases |
| 8. | Life spans decrease |
| 9. | Food chains shorten |
| 10. | Species diversity decreases and dominance increases |
| | **Ecosystem-level trends** |
| 11. | Ecosystem becomes more "open" |
| 12. | Successional trends reverse |
| 13. | Efficiency of resource use decreases |
| 14. | Functional properties are more robust than are structural ones |

Such facilities are available in the USA and Europe and have proved useful tools for both pure and applied research (Kosinski, 1989).

### *4.3. Community-level research*

Assessment of water quality in Britain is based on routine monitoring programmes. Past and present monitoring techniques have concentrated on the impact of changes in water quality on community structure and the data from such surveys are expressed as either a biotic or diversity index (Section 2.2).

Several biotic indices have been developed and used in the past by Water Authorities. In calculating these, taxa are allocated a score related to their sensitivity to pollutants. However, these scores are generally assumed rather than experimentally derived. Moreover, sensitivity rankings are assumed to apply across a range of toxicants even though laboratory experiments have shown this not to be the case (Slooff, 1983). Such assumptions can lead to the misinterpretation of water quality. One good example of this is the false impression given by the presence of stoneflies in acidified streams. Stoneflies score highly in all biotic indices and are assumed to be very sensitive to pollutants, but some species are in fact tolerant of acidification (Sutcliffe, 1983). Therefore, if the use of biotic indices is to be continued, more research is needed into the actual sensitivity of organisms to a range of toxicants. In addition, the importance of intraspecific variation in susceptibility due to acclimation, adaptation or age-specific effects needs to be addressed.

Shifts from sensitive to tolerant forms has been used as an indicator in its own right (Iserentant and Blancke, 1986). However, indicator species can only be used to show worsening conditions; tolerant species will frequently grow under "clean" conditions, although abundance may change.

The rationale for using diversity indices to assess water quality is that a decrease in water quality causes loss of sensitive species and therefore a reduction in diversity. However, many factors influence community structure and as diversity indices do not take account of the species present, their usefulness as a measure of water quality can be questioned. There are two further problems with the use of diversity indices:

(i) There is still an unresolved debate on how to measure diversity (which indices?); indices do not always correlate with each other (Magurran, 1988). Also, which taxonomic group and level should be considered?
(ii) It is not clear how diversity should respond to pollution; e.g. diversity of plankton appears to reduce continuously with organic enrichment, but increases, then decreases for benthic invertebrates (Calow, 1984). Often diversity measures do not

change monotonically across environmental gradients and sometimes there are step changes (Roff and Kwiatowski, 1977).

Diversity and biotic indices are based on changes in community structure. As community structure varies temporally as well as spatially, the problem is to decide whether an observed change represents a deviation caused by a pollutant, or whether such changes are part of the "natural" fluctuations inherent in the system. In most cases, the detailed baseline data necessary to determine the magnitude of inherent fluctuations are not available and, therefore, classical statistical procedures cannot be used to decide whether an observed change is significant or not. In such cases, the investigator is left with the difficult problem of deciding when a change indicates a toxic effect without knowing how the system behaves in the absence of a pollutant. There is a need for carefully planned studies of communities in a number of unpolluted sites in order to gain insight into the "normal" behaviour of such systems.

RIVPACS (see Section 2.2) provides an alternative approach. The deviation between observed and predicted taxa is used as a measure of impact. In addition, the presence and absence of species in the observed and predicted taxa lists can provide clues as to how the habitat has changed (Armitage *et al.*, 1987). However, there is unlikely to be a simple relationship between observed/predicted values and level of disturbance, particularly at the edges of ecological ranges. These ratios need combining with judgements about relative sensitivity of species that are either present or absent. Moreover, RIVPACS predictions are based on correlations between species assemblages and sets of physicochemical variables. The determination of fundamental process changes that take place in polluted/stressed communities is necessary to understand which of the environmental variables are causing these changes. This reemphasises the need for detailed studies into the factors controlling community structure and the impact of changes in water quality upon them.

Freshwater communities include a range of different organisms covering all the major taxonomic groups. Despite this diversity,

past techniques for assessing water quality have tended to concentrate on benthic macroinvertebrates (p13), although monitoring studies based on macrophytes have been proposed (Haslam, 1982). The reasons for this are; (a) macroinvertebrates include a large number of species giving a wide range of sensitivities to pollutants and hence a high definition response to pollution, (b) the species are relatively easy to identify, and (c) they have relatively long life spans and limited mobility, so providing historical evidence of intermittent pollution. However, to understand fully the effect of water quality on communities we must look more widely than the macroinvertebrates. For example, microorganisms play an important role in the food webs of many freshwater systems. They not only provide a food source in their own right but also condition organic material making it available to detritivores such as the amphipod, *Gammarus pulex*, and the stonefly, *Nemurella pictetii*. Stress-induced changes in microbial communities may have implications for both the structure and functioning of other components of the ecosystem (Groom and Hildrew, 1989). This need to widen the basis of water quality research has been identified by the Natural Environmental Research Council which has, as one of its 1990 Corporate Plan Targets, the objective of incorporating botanical attributes into RIVPACS (NERC, 1990).

*4.4. Population / Species level*

Relatively little information is available on the population dynamics of most freshwater organisms. However, it is crucial that we understand what factors are important in determining the size of populations if we are to interpret toxicity data. The majority of laboratory toxicity tests are designed to provide $LC_{50}$ values (Section 2.3). Such data are then used by regulatory bodies to set standards and consents. However, does it matter that 50% of individuals in a population are killed by the toxicant? If the population is under density-dependent control, then this mortality may be compensated for by a reduction in density-dependent mortality resulting in no net change in population size. Similarly, a toxicant-induced reduction in growth rate and fecundity may or may not have an effect at the population level.

Populations can either be controlled by density-dependent or independent factors and either may exert "top-down" (predator/grazer)

or "bottom-up" (productivity) control (Section 1.4). The evidence for "top-down" control in river systems is equivocal and there is a need for more research to assess the problem fully. The relative importance of density-dependent and -independent factors may vary between sites. Some ecologists (e.g. Peckarsky, 1984) have suggested that biological interactions may be relatively unimportant in flowing water systems if harsh physical conditions eliminate predators and maintain prey populations at low numbers. As streams become more benign, biological interactions may increase in importance as a result of the release from physical limitations on species distributions (Hildrew and Townsend, 1987).

Reductions in density and the complete extinction of sensitive populations resulting from pollution stress forms the basis of biotic and diversity indices. However, such effects cannot be totally attributed to direct toxic mortality, but may be due also to reductions in the abilities of organisms to function successfully in competitive and trophic interactions, or may be the result of increased emigration or reduced immigration, or avoidance of the contaminated environment. The impacts of these indirect effects on localized species extinctions (and possible subsequent recolonization) remains largely unexplored.

Due to the episodic nature of many pollution events (e.g. urban stormwater discharges (Section 3.3), agricultural pollution (Section 3.5)) the persistence of a population will be dependent upon its ability to recover from such events. This may happen in one of two ways: either by production within the site or by recolonization. The degree of recovery shown by a population will therefore be dependent upon the time of year - populations containing reproductively active individuals will recover faster than those that are not reproductive - and availability of suitable refugia to act as sources of colonizers. Again, more work is required on recolonization after controlled denudation in various circumstances to understand fully the importance of all these components.

Finally, when considering the effect of pollutants on populations, intra-specific variation in response has to be taken into account. There have been several studies illustrating how organisms acclimate and even adapt to pollutants (e.g. Harding and Whitton,

1976; Maltby *et al.*,1987). Selection for tolerant genotypes can occur over apparently short distances and short time scales.

*4.5. Individual-level*

Adverse effects of toxic compounds have most frequently been considered in terms of their lethal impact under conditions of continuous exposure to chemicals in isolation. However, it is ecologically more relevant to consider the toxic effect of specific compounds in mixtures (p.20) and under conditions of low concentration or intermittent exposure (p.21). Moreover, it is probably more important for the "well being" of the ecosystem to detect sublethal stress quickly and accurately. Early detection allows corrective action to be applied before irreparable damage has occurred.

Sublethal effects will occur once the accumulation of the pollutant in organisms has reached a threshold level. Bioaccumulation has been used as a monitoring tool (p.15) and although such an approach is particularly useful in detecting low-level or intermittent discharges, there are a number of factors that must be borne in mind in doing this. The rate at which accumulation occurs depends upon the availability of the chemical, environmental conditions and the organism's ability to assimilate it. Accumulation also depends on excretion, detoxification and storage by the organism. If populations adapt to the presence of the pollutant by, for example, developing or enhancing detoxifying or storage mechanisms, then interpreting the ecological consequences of a given tissue loading becomes problematic. Therefore, if bioaccumulation is to be used to indicate more than just the fact that a pollutant is present, more information needs to be acquired on exactly how body/tissue loadings relate to changes in performance (growth, reproduction, survival etc.) and how variable these relationships are between populations. The organisms used for such studies also deserve attention. Most information will be gained by using a range of organisms which "sample" different parts of the environment. One group of organisms that has proved useful in detecting intermittent pollution, especially of heavy metals and radioactive elements, are bryophytes (Say, Harding and Whitton, 1981). Measurements can either be made on naturally occurring or transplanted moss (Kelly, 1989).

Sublethal effects fall into a number of categories: for example, behavioural, biochemical, physiological, whole-organism performance.

Exposure to pollutants, even at very low concentrations, can elicit behavioural responses. The use of these in assessing the impact of changes in water quality on natural populations is dependent upon defining and quantifying normal behaviour patterns so that a change can be demonstrated. However, behaviour is difficult to quantify due to variability between individuals and through time. Perception and avoidance of pollutants is an important behavioural response for a species exposed to contamination. Although many species show avoidance behaviour and could, therefore, be used to detect pollutants, the ecological significance of such behaviour under specific circumstances has been questioned. It has been suggested that, although avoidance behaviour may be beneficial in that individuals can move away from contaminated areas, it may be detrimental to the populations if it means that individuals cannot reach a spawning or breeding ground. Behavioural effects are probably of greater ecological significance when they are related to functions such as feeding, mating and escaping from predators.

Several biochemical responses have been used to assess the effect of stress in contaminated ecosystems. For example, inhibition of key enzymes has been used to detect heavy metal and pesticide pollution. Certain pollutants can also lead to genetic abnormalities as a result of chromosomal damage or direct changes in DNA. Such responses appear to be sensitive to pollution stress and are often easily associated with the toxic mechanism. However, extrapolation to the overall "well-being" of organisms in polluted ecosystems is, in general, poorly understood. If such measures are to be used in future monitoring programmes, this must be rectified.

The effect of pollutants on a number of physiological processes has been investigated. One of these involves the automated monitoring of ventilatory patterns in "caged" fish. Such a system has proved useful in detecting episodic pollution events (Seager and Maltby, 1989). Others include: feeding, metabolism, osmotic and ionic balance and photosynthetic activity. Several of these processes

(food intake, metabolism, excretion) can be integrated to give an index of the energy-status of the organism. The difference between the energy an organism absorbs and that lost by respiration and excretion is a measure of energy available for production (gametic or somatic). This energy difference has been termed "scope-for-growth" and has been used to assess the effect of environmental stress on a range of marine invertebrates (Bayne *et al.*, 1985). A similar approach is now being used to develop an *in situ* bioassay based on the freshwater amphipod *Gammarus pulex* (Maltby *et al.*, 1990).

The advantage of sublethal measures of stress is that they have a relatively short response time. Moreover, they tend to be easier to standardize and replicate. However, in order to be of use to ecotoxicologists, they must be ecologically relevant; i.e. use species that are relevant to the communities in question and use response criteria that can be related, preferably mechanistically, to changes at higher levels of biological organization. This translation of effect from one level of biological organization to another is one of the major challenges for ecotoxicology.

### *4.6 Summary*

Several general and persistent themes emerge from this chapter:

- Long-term ecological sites should be established on streams and rivers in a variety of landscapes. These could be integrated profitably into monitoring networks designed to assess long-term changes in acidification and climate.
- More research is needed on episodic as compared with continuous pollution events.
- River biologists should move away from approaches that simply correlate pollution with ecological effects, to ones based on sound ecological understanding.
- This will need the specification of mechanisms whereby pollutants enter ecosystems and then impact the functioning and taxonomic composition of the community.
- Models will be required that link "molecular" and "ecological" levels and that express the relative importance of "bottom-up" and "top-down" effects.

- This will only be achievable through an experimental programme that allows specific predictions to be tested in a properly controlled and replicated design - something that will require experimental mesocosm facilities.
- More information concerning the mechanisms and rates of recolonization is needed.
- Relevant indicators have to be identified for both monitoring and ecotoxicological tests. Should these be key taxa in the functioning of river ecosystems or generally-sensitive taxa? Do either of these categories exist in the river biota?
- Monitoring presents certain sampling challenges; in particular relating to the need to distinguish effects from "noise".

# 5. CONCLUSIONS

*5.1 The important topics for research.*
It is self-evident that we cannot effectively manage systems that we do not understand. Yet, despite the development of substantial and complex legislation and management practices for protecting the quality of flowing-water systems, one of the major conclusions from this booklet is that our understanding of their basic ecology remains rudimentary. Pragmatism, such as the emphasis on chemical criteria of water quality and on the effects of sewage pollution, has been understandable and important, but should not be mistaken for a sound scientific basis for regulation and monitoring.

Issues that, in our view, need urgent attention are:

- The stability of flowing-water populations and communities. What, in other words, is the "normal" variance associated with them, and how can perturbation effects be distinguished from this "noise"? This touches on the more general question of the relative importance of density-dependent and -independent regulation in flowing-water populations.
- That the composition of biota in particular stretches of streams and rivers can be predicted broadly from a few habitat variables - such as distance from source and alkalinity - now seems established; but what are the causal pathways involved in these associations, and in their perturbation by pollutants ?
- There are a limited number of good studies on recovery after experimental and accidental perturbation. More needs to be known here about general principles; are the rates and patterns of recovery pollutant specific or are there some general features that are independent of the nature of the disturbance ?
- There is a bias in the examples cited in this booklet towards macroinvertebrates. However, this reflects a general emphasis on the use of these organisms as "indicators", at least within Britain. There is some justification for this (p. 32), but the approach needs to be broadened, and we certainly need to pay more attention to the identification of ecologically "key" and sensitive species - if they exist. Certainly, we need to know much more about the microbial communities at the base of

detritus food chains that are so crucial for many flowing-water systems. How ecologically important are they? How sensitive are they to pollutants ?

- Much of the water quality monitoring has, as already noted, been based implicitly, if not explicitly, on experience from relatively continuous organic pollution from point sources i.e. from sewage treatment works. It seems likely, though, that for the future, complex chemical effluents from industrial sources, and low-level organic inputs that diffuse into rivers from the general catchment will become more important. Discontinuous inputs are likely to be at least as important as continuous ones. It is not legitimate to presume that the ecological effects of all these will be similar. Yet, given the sheer complexity of the chemistry behind mixed, micro-concentration and intermittent effluents, it is certain that biological monitors will be more dependable than chemical ones. So the ecological responses need documenting and investigating at all relevant levels.

*5.2 Funding*

The main responsibility for the funding of basic research on flowing-water systems within Britain lies with the Natural Environment Research Council (NERC). During 1988/89 *c.*£2.3M of its Science Budget (i.e. <2% total expenditure) was used to research flowing-waters, but from both a basic and strategic point of view. The National Rivers Authority (NRA) also has a research and development programme which, because it addresses the NRA's statutory duties, is largely concerned with applied research. For 1989/90, the budget was *c.*£7.5M and this is planned to rise to £9-10M for 1990/91. Despite the emphasis on applied R&D, there is the intention to invest some resource in 1991/92 in Research Fellowships to sustain basic research in its areas of interest. About £0.5M funding that came previously from the Department of the Environment for R&D in freshwater quality has now been transferred to the NRA. Some more focussed R&D in conservation comes from the Nature Conservancy Council and in fisheries research from the Ministry of Agriculture, Fisheries and Food and the Department of Agriculture, Fisheries and Food Scotland. There is some, even more focussed, funding from the industrial sector and there are likely to be opportunities of funding through

the European Commission (e.g. the Science and Technology for Environment Protection Programme).

However, current spending on *basic* research in flowing-water systems in Britain is almost certainly less than £5M p.a. Given their importance as a national resource and their vulnerability, this is a surprisingly small investment that should give cause for concern. One recommendation must, therefore, be that the level of funding, particularly from the NERC, should be increased. Even if this happens, though, it seems likely that the level of funding is going to be limited and to come from diverse sources. We strongly recommend, therefore, that there should be co-ordination and strategic planning in the use of this resource at all levels; e.g. between the NERC and the NRA, and also between research teams and individual scientists.

### *5.3 The Future*

The logic of our recommendations in the last section argues for the development of a networked community programme of basic, but applicable, research that addresses issues highlighted in Section 5.1. Throughout the booklet we have drawn attention to features of this, but in particular we would emphasise:

- The establishment of sites for long-term monitoring.
- The establishment of a national experimental mesocosm facility (p.28).
- A co-ordinated group of scientists working at different levels and on different taxa.

The British Ecological Society can help in this by providing an appropriate forum for discussion - but ultimately co-ordination must come from the NERC and the NRA.

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