WATER QUALITY AND STRESS INDICATORS IN MARINE AND FRESHWATER ECOSYSTEMS: LINKING LEVELS OF ORGANISATION (INDIVIDUALS, POPULATIONS, COMMUNITIES)

Edited by

DAVID W. SUTCLIFFE

Published by the Freshwater Biological Association

Invited papers from a joint Associations specialised conference

held at

Napier University, Edinburgh on 6-7 September 1993

by

The Freshwater Biological Association,

The Ferry House, Far Sawrey, Ambleside, Cumbria LA22 0LP, England

and

The Marine Biological Association of the United Kingdom,

The Laboratory, Citadel Hill, Plymouth PL1 2PB, England

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The Scottish Association for Marine Science,

PO Box 3, Oban, Argyll PA34 4AD, Scotland

ISBN 0-900386-54-1

Use of macroinvertebrate communities to detect environmental stress in running waters

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In this paper we present an account of some current uses of RIVPACS (River Invertebrate Prediction and Classification System), a software package developed by the Institute of Freshwater Ecology. Background information is also given on the unique data-set on which the system is based. Before discussing RIVPACS, we consider the range of environmental stresses encountered in flowing-water systems and some of the ways in which stresses may affect macroinvertebrate communities. The wide application and relevance of the RIVPACS approach was recognised when it was chosen as the biological method for use throughout the UK in the 1990 River Quality Survey (RQS). In the concluding section we list some lessons learnt both from the 1990 survey and from our own testing exercise, and we outline current developments which will lead to a new version of RIVPACS for use in the 1995 RQS.

Introduction

At the outset, it is worth restating the case for the biological approach within water quality management and the value of the community approach within the context of the wide range of available biological tools. There is now widespread recognition that chemical monitoring alone is not enough and that pollution is essentially a biological phenomenon because its impact is on living organisms. Hence chemical and biological approaches are complementary and it is appropriate to detect and assess impacts through an examination of the biota. The high level of interest and debate over methodologies used within the countries of the EEC (Newman *et al.* 1992) and in North America (Rosenberg & Resh 1993) suggests that the need for biological methods has been accepted. Nevertheless, when results from chemical and biological monitoring programmes are used for water quality classification, attempts are sometimes made to combine them for a given site without first considering the individual results. This is not a helpful procedure and, as pointed out in the recent report by the Royal Commission on Environmental Pollution (1992), it tends to obscure issues by trying to combine assessments of the causes of biological deterioration and their effects.

In considering the biological approach, methods based on individuals, populations and communities all have their place and a comprehensive monitoring programme for a river system might well incorporate aspects of each component. Some scientists are proposing that ecosystems respond predictably to environmental stress and that a synthetic view of ecosystems will lead to the development of new tools to assess and monitor ecosystem health (Perry 1994). Odum (1985) offered a table of expected ecosystem-level responses to stress and some recent studies suggest that freshwater ecosystems do respond predictably to stress (Schindler 1990, Perry *et al*. 1987). However, others believe that current theories do not account for the complexities exhibited by ecosystems in their response to and recovery from stress (Cairns 1990). Hence there is not yet agreement amongst aquatic ecologists about the feasibility of management at the ecosystem level. Perry (1994) suggests that some of the differences in viewpoints are a matter of scale and that with a larger-scale perspective, future scientists and managers will be able to collaborate to implement ecologically sound management policies.

At the community level, use of the biological approach is already well established and accepted. To quote Cairns & Pratt (1993): "Biological surveillance of communities – with special emphasis on characterizing taxonomic richness and composition – is perhaps the most sensitive tool now available for quickly and accurately detecting alterations in aquatic ecosystems". And later: "Faunistic changes in streams are always very meaningful, although it is not always clear if altered water quality is the cause".

Of the various major taxonomic groups which can be used for community level assessments, a literature search by Hellawell (1977) indicated that algae and macroinvertebrates were most often recommended for assessing water quality. In practice, it is very apparent that macroinvertebrates are now the most commonly used group for surveillance and monitoring in fresh water (Hellawell 1986; Rosenberg & Resh 1993). The advantages of using macroinvertebrates have been listed and discussed in a number of publications (see Rosenberg & Resh 1993 for further references) and will not be repeated here. More generally, Reice & Wohlenberg (1993) point out that the state of an aquatic ecosystem cannot be fully understood without knowledge of the zoobenthos because it plays an essential role in the food chain, productivity, nutrient cycling and decomposition.

Environmental stresses in running waters

Hellawell (1986) devotes a chapter to reviewing environmental stresses before considering, in later chapters, the impact of a wide range of stresses on the aquatic flora and fauna. He defines three major categories of environmental stress – natural, imposed and environmental manipulation – before reminding the reader that two or more categories of stress may be acting on a running-water system at one and the same time.

Natural environmental stresses are easily forgotten in the face of major stresses imposed by man, but the possibility of a significant natural stress should always be borne in mind when undertaking biological assessments. Flood events and droughts are obvious examples of natural environmental stresses which may have an impact on community structure, albeit of a temporary nature. A major flood can result in the removal of submerged macrophytes and some fractions of bed material, the redistribution of larger particles and, in consequence, heavy losses to the benthic community. Giller *et al.* (1991) documented the impact of a one in 50-year summer flood event on the macroinvertebrate fauna of a stream in Ireland, and showed that recovery was not complete after 3 years.

Drought conditions bring a different set of problems for the various members of the fauna, including modifications to the temperature regime, together with changes in habitat and food availability. Initially these environmental changes are likely to manifest themselves as changes in the abundance of some elements of the macroinvertebrate fauna (Wright 1992). Whether more extreme changes occur, including the loss or addition of taxa, will depend on a number of factors including the length of the drought, the location of the river-reach affected and the extent to which refugia are present nearby from which recolonisation can occur.

Between an extreme flood event and a major drought, there are years when the discharge regime exceeds and others when it is lower than the long-term mean. This is normal, and the macroinvertebrate community to be expected at a site, otherwise unstressed, is that which is capable of existing within the broad envelope of typical environmental conditions recorded at the site over a number of years.

Hellawell (1986) offers detailed tables with background information and comments on a number of imposed stresses, including sewage and organic wastes, toxic wastes, inert solids and heat. He also points out that some stresses imposed by man are similar to natural ones in kind, although not necessarily in degree. Others, and in particular some toxic wastes from industrial processes, together with pesticides and herbicides, are completely alien to the natural environment. Of those imposed stresses which have parallels with natural phenomena, he suggests that sewage effluents are an extreme form of the enrichment which occurs in streams after deciduous leaf fall in autumn. Also, industrial effluent containing heavy metals may affect invertebrate communities in similar ways to those in "natural" streams which gather their waters from strata bearing heavy metals.

This line of argument, in which natural phenomena and some imposed stresses or "pollution" can be similar in nature if not in intensity, brings into sharp focus the need for a definition of pollution. Hellawell (1986) favours the definition offered by Edwards (1972), that pollution is "the release of substances or energy into the environment by man in quantities that damage either his health or resources". Hellawell sees such an anthropocentric view as inevitable and more recent authors agree. Perry (1994) refers to "stress" as a force that results in a change in ecosystem behaviour such that behaviour is outside the bounds of "normal". In an ecosystem context, normal behaviour means the ways in which a set of structural and functional relationships interact and maintain ecosystem integrity. In the context of water quality management, normal behaviour means the continued provision of benefits which are of importance to society (e.g. fish production and drinking water quality). His definition of stress would therefore take account of both natural and imposed environmental stresses which lead to ecosystem behaviour falling outside management expectations.

The third category of environmental stress defined by Hellawell (1986) is environmental manipulation, and once again he provides detailed tables giving the environmental implications of channel modification, reservoirs, river regulation and inter-river transfers. Although these operations are very different in character from the various forms of pollution considered earlier they can, nevertheless, have substantial consequences for macroinvertebrate communities. Changes in habitat and food availability, changes in temperature regime and water chemistry, and the more obvious changes in discharge regime, can all lead to modifications in macroinvertebrate community structure and function. The extent of channel modification in lowland Britain is very substantial (Brookes *et al.* 1983) and the impact on the invertebrate fauna of regulating reservoirs has been well documented (Armitage 1984). Although the effects of different forms of river regulation on invertebrate communities are complex and may be confounded by additional environmental stresses, the importance of maintaining a wide range of habitat types, and in particular emergent and submerged macrophytes, in order to maintain macroinvertebrate richness, is becoming apparent (Wright *et al.* 1993a).

Effects of environmental stress on macroinvertebrate communities

In an ideal world, river ecologists and managers would understand the processes which lead to the observed patterns of community structure in unstressed flowing-water systems. That is, the major factors influencing species richness and composition, including the relative abundance of the component species, would be known together with the way the observed patterns drive the functioning of the system. This would provide a firm foundation from which to investigate the processes taking place when environmental stresses lead to community change, both structural and functional.

Hildrew (1992) presents a valuable review of our current understanding of food webs and the biotic interactions which can influence community structure. Important insights are beginning to emerge, but much remains to be investigated and Hildrew concludes that currently we have a restricted ecological understanding of the basis of community structure in rivers.

Our current knowledge of the response of macroinvertebrate communities to stress is, therefore, based predominantly on direct observation, and our understanding of the processes involved is at an early stage. However, many of the structural changes in community composition following a wide range of environmental stresses are well documented.

Odum (1985) gives a table of the trends expected in stressed ecosystems in the longer term, including changes in energetics, nutrient cycling and community structure and function. Of the changes in community structure, he lists a decrease in species richness and an increase in the dominance of some species, although he also points out that if the original richness is low, the reverse can occur under stress. Concurrent with the familiar changes in richness and dominance, he suggests that the proportion of r-strategists increases, the size of organisms decreases, life spans decrease and food chains shorten. Odum (1985) also offers the hypothesis that functional properties at the ecosystem level may be more robust than structural properties. This idea appears to be gaining momentum as the results of large-scale experimental perturbations are published (Schindler 1990). Reice & Wohlenberg (1993) also present case histories of ecosystem studies, both lentic and lotic, involving assessment of the impact of known environmental stresses on benthic macroinvertebrates. They take the view that the distinction between community (structure orientated) and ecosystem (process orientated) is often obscure and they use changes in benthic community structure to make inferences about ecosystem level processes.

Perry (1994) discusses the ecosystem responses to stress which were first listed by Odum (1985) and explores their relevance in a water quality context. In particular, he considers the proposal that functional attributes (e.g. production, respiration, nutrient cycling) are more robust than structural qualities (e.g. species composition and species richness). In other words, at the ecosystem level, structure and function are not as closely linked as were once thought and therefore significant structural changes may be observed without concomitant observations of functional change. He suggests that redundancy and homeostasis explain the apparent anomaly. An important consequence of this in the field of biological monitoring is that structural attributes will change more readily in response to a given stress than will functional roles. Of course, if structural changes are sufficiently pronounced, they will affect ecosystem buffering capacity and functional changes will occur. However, it was the perceived sensitivity of community structure that led Cairns & Pratt (1993) to conclude that biological surveillance of communities, with special emphasis on characterising taxonomic richness and composition, is perhaps the most sensitive tool now available for quickly and accurately detecting alterations in aquatic ecosystems.

An approach to evaluating biological quality

If we are to use the macroinvertebrate fauna to detect the impact of one or more environmental stresses on a site, then ideally we need to know the fauna to be expected at the site in the absence of environmental stress. This presents a major challenge on two fronts. First, the macroinvertebrate community to be found at any given unstressed site will be influenced by many factors (geographical, geological and catchment related) which determine river type, but also by the location of the site along the length of the watercourse (Vannote *et al.* 1980). It is, therefore, apparent that in attempting to set site-specific "targets" for the fauna to be expected in the absence of environmental stress, access to good-quality information for a wide range of sites is essential. The second challenge is the selection of the sites to ensure that they are not already impacted by major environmental stress. In view of the extensive catalogue of environmental stresses listed in a previous section of this paper, one could question the feasibility of acquiring data suitable for use in developing site-specific predictions. This problem has been faced and, we believe, a realistic and practical stance is being taken. Great Britain has a high population density and rivers in some industrialised areas still have major problems. Nevertheless, a relatively small proportion of our rivers are badly polluted as judged by chemical criteria (National Water Council 1981; Department of the Environment and the Welsh Office 1986; National Rivers Authority 1991).

The research programme which eventually led to the development of RIVPACS (River Invertebrate Prediction and Classification System) started in 1977. It had the twin objectives of (1) developing a biological classification of unpolluted running-water sites based on the macroinvertebrate fauna and (2) determining whether the fauna to be expected at an unstressed site could be predicted from physical and chemical features only. When the first discussions with Water Authority (WA) and River Purification Board (RPB) biologists took place over site selection, strenuous efforts were made to focus on rivers and sites which the local biologists regarded as the best examples of their type. An effort was also made to avoid sites just downstream of reservoirs where substantial impacts to the invertebrate community would be anticipated. Sometimes samples were collected from a site, but after identification at species level, the fauna indicated some form of stress, leading to rejection of the site. The current version of RIVPACS has been developed in three stages, each one involving the addition of a wider range of sites. At each stage and as our knowledge widened with experience, stronger criteria were applied for the acceptance of sites for inclusion in the new system.

Even when rivers are not noticeably polluted or subject to regulation, it would be naïve to regard them as being close to a pristine condition as man's activities, both instream and within the catchment over hundreds if not thousands of years, will have influenced community structure and function. This is likely to be more evident in lowland systems in Britain, and Hildrew (1992) suggests that most human impacts on rivers have led to simplified food webs and increased the influence of disturbance.

The implication then, is that targets set by analogy with sites in a database are, at best, high quality and realistically achievable targets and not theoretical targets which might be feasible in pristine catchments unaffected by man. Whilst we share with others a curiosity about the structure of invertebrate communities prior to the influence of man, we aim to offer a workable system rather than theoretical and unachievable goals. It is possible that targets for lowland river types heavily influenced by man may be of a lower standard. This is difficult to refute, but our assertion that the targets are realistic and achievable through positive river management is relevant to those concerned with the improvement of Britain's rivers.

Before moving on to the detail of the RIVPACS approach, two more general points are relevant here. First, the number of species at a site is not static and in fact some have argued that this limits the concept of "indicator species" since perpetual residence of a particular taxon is probabilistic (Cairns & Pratt 1993). The role of disturbance as a major force influencing community structure is now receiving close scrutiny (Hildrew & Townsend 1987; Resh *et al.* 1988) and implicit in this is the idea of species turnover at a given location. Second, faunal predictions offered by RIVPACS take the form of a list of probabilities of capture of taxa after a standard length (time) of sampling effort. They are based on information from a number of sites with environmental attributes which are similar but not identical to the site for which a prediction is being made. All the sites contributing to this and all other predictions will have met the most recent criteria laid down for site acceptability. Nevertheless, the sites used for a given procedure will differ in both species composition and richness and the predictions will be of the long-term average fauna to be expected at a site with those environmental features. Hence, in appraising sites by comparing the observed fauna with the expected fauna (O/E), it is possible for an exceptional site to have an O/E ratio above unity, just as other sites will have

O/E ratios below unity. In future we may come to understand those site features which result in O/E ratios above unity. An important question for a later section of this paper is the point below unity at which an O/E value indicates an environmental stress which requires action.

Development of RIVPACS

Although the essential techniques used to develop RIVPACS have been in the scientific literature for some time (Wright *et al.* 1984; Moss *et al.* 1987) a number of important practical and operational advances have been made in the last few years. A detailed account of the development of the system up to and including its use in the 1990 River Quality Survey has recently appeared (Wright *et al.* 1993b) and will not be repeated here. Nevertheless, some guiding principles and important milestones should be mentioned.

As previously indicated, rivers were chosen with great care after detailed consultations. Sampling locations along each river were selected to reflect the fact that changes in species composition are greater near the source (Verneaux 1976). Each site was sampled in spring, summer and autumn to ensure adequate representation of the macroinvertebrate fauna. On each occasion, all habitat types were sampled, approximately in proportion to their occurrence. The preferred method of sampling was a 3-minute timed pond-net collection using a standard FBA pond-net. Where this was ineffective at large deep sites, a light-weight dredge coupled with pond-netting of marginal areas was the preferred method.

Initially, field sampling was undertaken by WA/RPB biologists using guidance provided by the Institute of Freshwater Ecology (IFE) but in later phases of the work some field sampling was carried out by IFE staff. All sample sorting and identification of the fauna was undertaken by the IFE and identifications were at species level where adequate keys were available.

By 1981, 268 sites had been examined on 41 river systems. This was later extended to 370 sites (61 rivers) in 1984 and then 438 sites on almost 80 river systems in 1988. Environmental data for each site were collected at the time of the biological sampling. These data, along with further information collated from maps and chemical data provided by Water Industry staff, were essential for the development of the prediction system.

TWINSPAN (Hill 1979) was chosen for the classification of sites based on their faunal composition before multiple discriminant analysis (Klecka 1975) was employed to discriminate between the classification groups using environmental attributes. The use of these techniques on the 268 site data-set is demonstrated in Wright *et al.* (1984). Once the match between the biological features of good quality sites and their environmental features had been demonstrated, there was a basis for predicting the fauna of unsampled sites from their environmental features. This approach was extended by taking a novel step. Instead of simply using the environmental attributes of a site to predict to a particular classification group, whose sites have relatively similar invertebrate communities, a prediction was made in terms of the probability of capture at the site of named taxa after the standard sampling effort (further details in Moss *et al.* 1987). A non-mathematical treatment of the steps involved in the technique is presented in Furse *et al.* (1987).

By 1986, the 370-site version of the classification and prediction system (RIVPACS I) had been implemented on BBC B microcomputer and was then made available to the Water Authorities in England and Wales and the River Purification Boards in Scotland for testing. RIVPACS I offered two separate procedures: (i) the option of classifying a new (unstressed) site on the basis of the macroinvertebrate fauna recorded after sampling for three seasons, using the standard procedures, and (ii) a prediction of the fauna to be expected at an unstressed site after sampling for three seasons, based on environmental features for the site. The prediction system was of particular interest within the Water Industry, because it gave a target community against which the fauna observed at a site could be assessed.

Although early development work on the prediction system had used 28 environmental variables, it was demonstrated (Moss *et al.* 1987) that sub-sets of easily measured environmental features could be used with little loss of predictive capability. Hence RIVPACS I included four sets of environmental variables for appraisal. Predictions could be made at "species level" but in addition, a prediction capability at Biological Monitoring Working Party (BMWP) family level was added following the adoption of the BMWP system for routine use within the Water Industry (National Water Council 1981; Armitage *et al.* 1983). RIVPACS I also had the facility to attempt predictions of the number of BMWP taxa, BMWP score and the Average Score per Taxon (ASPT). The response to RIVPACS I, as determined by a standard questionnaire completed by WA and RPB personnel, together with our own tests, provided the basis for the next stage in development.

RIVPACS II, the current version, is based on 438 sites within 25 groups and was released to the National Rivers Authority (NRA), the RPBs and the Department of the Environment (Northern Ireland) in 1990. The software runs on IBM and IBM-compatible personal computers, but a mainframe version is also available. It differs from RIVPACS I in a number of ways, the most important being the following. First, although individual predictions are still possible through the interactive input of environmental data for a site, a "batch mode" procedure enables the user to store environmental data for many sites and then run a series of predictions without further intervention. If biological data for a series of sites is also held on computer file, the program can be instructed to compare the observed and predicted fauna at the BMWP family level. Second, a menu of six environmental options is offered for prediction (Table 1) although Option 1 is recommended for general use. Third, predictions are available not only for three seasons combined, but also for any of the three single seasons or for combinations of pairs of seasons. Fourth, RIVPACS I taxonomic levels are available for predictions but an additional feature is "customization" of the taxonomic level at which the prediction is offered (Table 2). This enables the user to downgrade from "species" level to the taxonomic level used for a particular type of survey (e.g. all identifications to genus, or a mixture of levels for different macroinvertebrates as specified by the user). Finally, a procedure first proposed by Rushton (1987) is used for the classification of new sites which have been sampled in three seasons. It involves use of the full taxonomic list rather than a smaller number of "indicator species" as employed in the TWINSPAN key in RIVPACS I.

A RIVPACS prediction

The format of a "species level" prediction using RIVPACS II is given in Figure 1. This site is in the upper reaches of the River Brue in Somerset and was sampled in 1988 by IFE staff on behalf of the Nature Conservancy Council. Standard 3-minute pond-net samples were taken in spring, summer and autumn, along with measurements of water width, water depth and substratum composition. Further data on altitude, distance downstream, slope and discharge category was acquired from maps, an annual mean value for total alkalinity was obtained from the Water Authority, and information on the air temperature regime, the latitude and longitude was calculated by the RIVPACS software from the National Grid Reference.

These environmental variables were used in MDA (multiple discriminant analysis) equations to generate the probabilities of classification group membership. The frequency of occurrence of taxa in these classification groups, and the probability with which the new site belongs to each group, was then used to generate a list of the taxa to be expected at a site with the particular environmental features in the absence of environmental stress (Furse *et al.* 1987)

Table 1. *Six environmental options available for prediction in RIVPACS II.*

The following eight variables are used in all predictions:-
Distance from source (km)
Mean substratum (phi)
Altitude (m)
Discharge category (9 groups, in cumecs)
Mean water width (m)
Mean water depth (cm)
Latitude (°N)
Longitude (°W)

A further five variables are included, according to the option selected below.

Variable	Options 1	2	3	4	5	6
Alkalinity (mg $CaCO_3$ l^{-1})	+	+		+		+
Slope (m km^{-1})	+		+	+		+
Mean air temperature (°C)	+	+	+		+	
Air temperature range (°C)	+	+	+		+	
Chloride (mg Cl l^{-1})						+

Table 2. *The output options in the RIVPACS II system with respect to taxonomic level, season and level of detail in the prediction.*

In the seasonal option column,
3 = three seasons combined,
2 = any two seasons combined, and
1 = any one of spring, summer or autumn.

Taxonomic level	Seasonal option	Notes on prediction
Species	3, 2 or 1	Presence/absence only
All families	3, 2 or 1	Presence/absence only
'Customized'	3, 2 or 1	Presence/absence only
BMWP families	3, 2 or 1	Presence/absence + associated biological indices
All families	1 only	Families listed in descending order of predicted abundance. (Based on Log categories of abundance of families in 438-site data-set.)

The full taxonomic list runs from the highest to the lowest probability of capture but in Figure 1 it is terminated at the 50% probability level. The taxa captured at the site after sampling for three seasons are shown with an asterisk. Taxa which are predicted with almost 100% probability of capture should, with few exceptions, be present if the site is unstressed, but approximately 3 in 4 and only about 1 in 2 taxa should be present at the 75% and 50% probability levels respectively. It is apparent that the River Brue at Wyke is of high biological quality because the ratio of observed taxa to expected number of taxa is around unity at both the 75% and 50% probability levels.

The RIVPACS data-set

This section provides information on the biological data used in the construction of RIVPACS II before we examine the utility of the system for detecting environmental stress. The species richness of the macroinvertebrate fauna varied considerably across the 438 sites used in RIVPACS II (Fig. 2), although 83% of sites had between 50 and 99 taxa. Low taxonomic

River: Brue Site: Wyke

Environmental data used:

Water width (m)	5.2	Altitude (m)	45
Mean depth (cm)	28.3	Dist. from source (km)	12.5
Substratum composition		Slope (m km^{-1})	2.5
Boulders and cobbles (%)	63	Discharge category	3
Pebbles and gravel (%)	22	Mean air temp (°C)	10.37
Sand (%)	4	Air temp. range (°C)	12.23
Silt and clay (%)	12	Latitude (°North)	51.06
Mean substratum (phi)	-4.51	Longitude (°West)	2.29
		Alkalinity (mg l^{-1} $CaCO_3$)	227.7

Classification groups predicted from MDA with the above data.

5b 44.5% 7a 18.3% 7c 18.1% 8a 13.3% 4c 3.7%

Predicted taxa, in decreasing order of probability of capture:

*	97.9%	Micropsectra gp
*	96.7%	Gammarus pulex
*	95.4%	Glossiphonia complanata
*	95.0%	Hydracarina
*	94.5%	Elmis aenea
*	92.5%	Thienemannimyia gp
*	90.8%	Ephemerella ignita
*	90.7%	Cricotopus gp
*	90.1%	Ceratopogonidae
*	89.9%	Eukiefferiella gp
*	89.7%	Simulium (S.) ornatum gp
*	89.4%	Baetis rhodani
*	89.0%	Potamopyrgus jenkinsi
*	87.3%	Erpobdella octoculata
*	87.1%	Limnodrilus hoffmeisteri
*	85.9%	Pisidium subtruncatum
*	81.9%	Pisidium nitidum
*	79.7%	Baetis vernus
	79.5%	Polypedilum sp.
*	79.2%	Ancylus fluviatilis
*	79.1%	Psammoryctides barbatus
	76.9%	Hydroptila sp.
*	74.4%	Asellus aquaticus
*	73.4%	Stylodrilus heringianus
*	72.6%	Paratanytarsus gp
*	72.0%	Dicranota sp.
*	71.6%	Limnius volckmari
	71.4%	Helobdella stagnalis
	71.4%	Baetis scambus gp
*	70.7%	Lymnaea peregra
*	70.4%	Hydropsyche siltalai
*	70.3%	Oulimnius tuberculatus
*	70.0%	Rhyacodrilus coccineus
	69.2%	Lumbriculus variegatus
	69.2%	Hydropsyche pellucidula
	68.5%	Prodiamesa olivacea
	68.1%	Aulodrilus pluriseta
*	63.1%	Stylaria lacustris
	60.3%	Polycentropus flavomaculatus
	60.2%	Sphaerium corneum
*	59.6%	Rhyacophila dorsalis
	58.5%	Halesus sp.
*	57.7%	Microtendipes sp.
*	57.2%	Lumbricidae
*	56.7%	Potamophylax gp
*	56.3%	Rheocriotopus sp.
	56.0%	Macropelopia sp.
*	56.0%	Limnephilus lunatus
*	55.8%	Sialis lutaria
	54.3%	Ephemera danica
*	53.2%	Potthastia longimana gp
	52.9%	Caenis luctuosa gp
*	52.1%	Sericostoma personatum
	52.0%	Polycelis nigra gp
	51.8%	Potamonectes depressus
*	50.9%	Brillia modesta
*	50.5%	Brychius elevatus

Probability level	75%	50%
Observed no. taxa / Expected no. taxa	20 / 19.38 = 1.03	41 / 41.16 = 1.00

Figure 1. Demonstration of part of a "species" level prediction from RIVPACS II using the environmental variables in Option 1. Taxa observed at the site are indicated with an asterisk. See the text for further explanation.

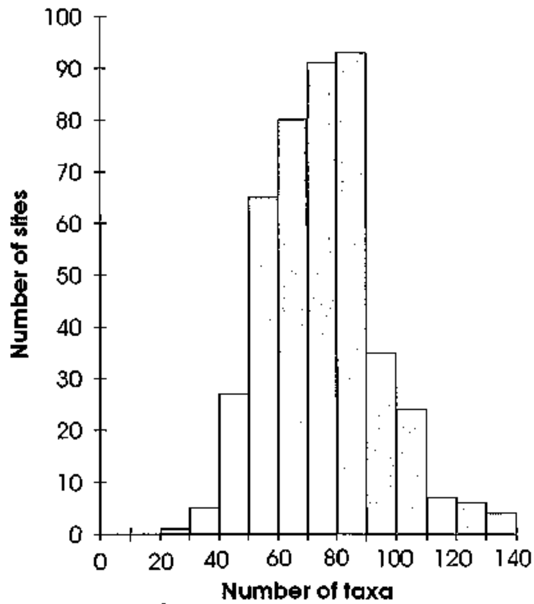

Figure 2. Macroinvertebrate taxon richness of the 438 sites in RIVPACS II.

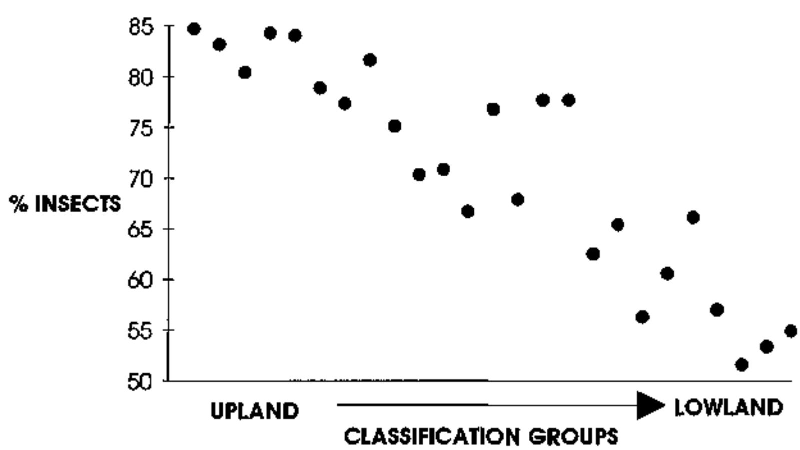

Figure 3. Percentages of insect taxa at the sites in each of the 25 classification groups in RIVPACS II. Each circle is the mean value for the sites within that group.

richness was a characteristic feature of some sites in upland parts of Scotland, Wales and England where the physical environment was harsh. However, high taxonomic richness was found at a wide range of sites, including rivers in mid and south Wales, streams in the south-west of England, chalk streams, and lowland rivers in East Anglia. These sites all shared high taxonomic richness, but they had distinctly different macroinvertebrate communities.

Table 3. *Number of different taxa found at the 438 sites in RIVPACS II.*

Asterisks denote taxa not identified beyond a major group.

Non-insect taxa		Insect taxa	
*Spongillidae	1	Ephemeroptera	38
*Hydridae	1	Plecoptera	26
Tricladida	9	Odonata	14
*Chordodidae	1	Hemiptera	24
*Bryozoa	1	Coleoptera	93
Gastropoda	32	Megaloptera	3
Bivalvia	21	Neuroptera	2
Oligochaeta	53	Trichoptera	90
Hirudinea	14	Diptera	169
*Hydracarina	1		
Crustacea	10		
TOTAL	144	TOTAL	459

Table 4. *Correlations between species richness and selected environmental variables, based on the 438 sites in RIVPACS II.*

Asterisks denote variables transformed into $\log_{10}$ values.

Environmental variable	Pearson r	Significance
Latitude	−0.476	0.0001
Longitude	0.196	0.0001
*Slope	−0.338	0.0001
*Altitude	−0.315	0.0001
*Distance from source	0.218	0.0001
*Mean width	0.119	0.0127
*Mean depth	0.180	0.0002
Mean substratum	0.258	0.0001
Alkalinity	0.263	0.0001

Across the 438-site data-set there were 144 non-insect taxa and 459 insect taxa, giving a total of 603 (Table 3). For certain members of the non-insect fauna, including some sessile organisms and also some small forms that are not routinely identified by freshwater biologists, no attempt was made to identify at the species level.

Changes in the contribution of insect and non-insect taxa in relation to community composition is readily apparent by examining the percentage of insect taxa across the 25 classification groups in RIVPACS II. The percentage of insect taxa within each group, derived as the mean of the sites in the group, is shown in Figure 3, where the 25 groups are given in the sequence in which they are listed after the TWINSPAN analysis. This is essentially from upland to lowland groups and it is apparent that insects dominate the upland groups whilst in the extreme lowland groups non-insects make a much greater contribution to community composition.

Species richness of the macroinvertebrate fauna at the 438 sites was examined in relation to a number of the environmental features used for prediction (Table 4). Many of these attributes are interrelated, but nevertheless some of the correlations with taxon richness are instructive. On the geographical scale, species richness decreases from south to north in Britain. Topographical features such as high altitude and steep slopes are correlated with lower richness, whilst features which relate to location downstream (distance downstream, together with width and depth) imply a marginal increase along the length of the watercourse. The positive correlation with mean substratum indicates higher richness on fine rather than coarse substrata and the positive relation with alkalinity is not unexpected.

Detection of environmental stress using RIVPACS

In this section we offer demonstrations of the use of RIVPACS at three levels of detail to illustrate the flexibility of the approach. A "species" level prediction is followed by a prediction of BMWP families, and finally we give examples of the use of indices based on the BMWP system. The decision on which option is appropriate will be influenced by whether RIVPACS is being used for the assessment of many sites in a national survey, for routine monitoring purposes, or for the investigation of a specific pollution incident. In choosing an option the constraints of time, the cost, the availability of expertise in identification and the potential consequences of a given incident for a site will also have been taken into account.

Predictions at the level of species

We present here a species level prediction to detect the impact of man-made stresses on a lowland river in southern England. Within the IFE, most RIVPACS-compatible survey work is undertaken at what are believed to be high quality sites in order to extend the database. In the mid 1980s an extensive survey (Wright *et al.* 1988) was undertaken on the macroinvertebrate fauna of the Moors River in Dorset, some sections of which are a grade 1 Site of Special Scientific Interest (SSSI) (Ratcliffe 1977). The good quality sites have since been incorporated into RIVPACS II, but at the same time we examined a number of stressed sites in the lower part of the catchment. Wright *et al.* (1993c) give an account of the value of RIVPACS I (i.e. prior to the incorporation of sites on the Moors catchment) to identify sites of high conservation value and those under stress. In this paper we investigate the mismatch between the observed and the predicted fauna (based on a RIVPACS I prediction using eleven environmental features) in the lower reaches of the Moors River at Hurn, just before the confluence with the Dorset Stour. This section of river was known to be suffering from organic pollution due to progressive urbanisation within the catchment and the heavy demands placed on the sewage treatment works at Palmersford some 5 km upstream (Nature Conservancy Council & Wessex Water 1982). In addition, a serious pollution event involving timber preservative which impacted directly on the Uddens Water, a major tributary, but which also affected the Moors River, occurred during the course of the survey.

The layout of the species level prediction closely follows that for the River Brue at Wyke (Fig. 1) and will not be shown in this form. Instead, Table 5 presents a condensed version of the site-specific information relating to the Moors River at Hurn, incorporating all taxa with a predicted probability of over 50%. All major taxonomic groups and the full number of taxa listed on the printout between 100 and 50% probability are detailed in Table 5. Of the 71 taxa listed, 30 were observed at the site after sampling for three seasons; the table indicates the major taxonomic groups to which they belonged. The number of taxa expected in each major taxonomic group is the sum of the probabilities of capture of each component member of the group as given in the RIVPACS prediction. In this case the overall ratio of Observed (O) to Expected (E) number of taxa (30/52.8 = 0.57) is indicative of loss of taxon richness. It is also possible to calculate O/E ratios for each major taxonomic group, but the ratios will be more reliable with high individual O and E values.

Considering first the non-insect taxa, the range of oligochaetes present is well below expectation for an unstressed site. Perhaps more striking is the under-representation of insects and in particular Plecoptera (absent), Trichoptera (O/E = 0.11) and Ephemeroptera (O/E = 0.51). The absence of so many taxa in these three groups is clear evidence of a substantial problem, and the fact that the individual taxa expected with a high probability can be named provides a firm basis for investigating the cause(s) of the loss of taxon richness.

Table 5. *The mismatch between the observed and predicted fauna for a stressed site (Moors River at Hurn in Dorset).*

Species level prediction obtained from RIVPACS I based on eleven environmental variables and taken to the 50% probability level.

Taxonomic group	No. taxa listed to 50% level	No. taxa observed (O)	No. taxa expected (E)	O/E
Gastropoda	3	2	2.48	0.81
Bivalvia	2	1	1.51	0.66
Oligochaeta	9	4	6.88	0.58
Hirudinea	4	3	3.00	1.00
Hydracarina	1	0	0.98	0
Crustacea	2	2	1.77	1.13
Ephemeroptera	11	4	7.90	0.51
Plecoptera	3	0	2.02	0
Coleoptera	5	2	3.48	0.57
Megaloptera	1	0	0.55	0
Trichoptera	12	1	8.81	0.11
Diptera	18	11	13.42	0.82
TOTALS	71	30	52.80	0.57

Predictions at the level of families

The second example is a prediction at BMWP family level to determine the status of a site at Sanduck on the Wray Brook in the south-west of England, which is 1 km downstream from a sewage treatment works. Extensive observations were made at the site in the mid 1980s, and the RIVPACS II prediction (Option 1, three seasons combined) in Figure 4 shows the environmental data used and the BMWP families predicted as far as the 50% probability level. The 21 BMWP families observed at the site as a result of standard RIVPACS sampling in April, July and October 1986 are indicated with an asterisk. It is apparent that there is an under-representation in the number of families of Trichoptera, Plecoptera and Ephemeroptera

River: Wray Brook Site: Sanduck

Environmental data used:

Water width (m)	2.1	Altitude (m)	145
Mean depth (cm)	28.2	Distance from source (km)	4.6
Substratum composition		Slope (m km^{-1})	8.3
Boulders and cobbles (%)	10	Discharge category	2
Pebbles and gravel (%)	53	Mean air temperature (^{0}C)	10.59
Sand (%)	22	Air temperature range (^{0}C)	11.31
Silt and clay (%)	15	Latitude (^{0}North)	50.38
Mean substratum (phi)	-0.86	Longitude (^{0}West)	3.44
		Alkalinity (mg l^{-1} $CaCO_3$)	28.7

Classification groups predicted from MDA with the above data.

5a 40.8% 6a 27.0% 2a 15.2% 3a 11.7% 5b 3.7% 2d 1.1%

Predicted taxa, in decreasing order of probability of capture:

*	100.0%	Oligochaeta
*	100.0%	Chironomidae
*	99.9%	Elmidae
*	99.9%	Baetidae
*	99.6%	Tipulidae
*	97.5%	Limnephilidae
	96.2%	Hydropsychidae
*	95.3%	Simuliidae
	94.6%	Leuctridae
*	94.3%	Gammaridae
	93.6%	Heptageniidae
*	93.6%	Nemouridae
	93.2%	Polycentropodidae
*	93.1 %	Ephemerellidae
	92.0%	Sericostomatidae
*	90.9%	Dytiscidae
*	90.4%	Rhyacophilidae
*	88.8%	Sphaeriidae
	86.5%	Perlodidae
*	86.4%	Hydrophilidae
*	85.4%	Ancylidae
	83.4%	Leptophlebiidae
*	82.6%	Hydrobiidae
	71.4%	Ephemeridae
*	70.6%	Glossiphoniidae
	68.2%	Gyrinidae
	67.4%	Caenidae
*	67.3%	Erpobdellidae
	66.5%	Leptoceridae
*	63.6%	Lymnaeidae
	63.2%	Sialidae
	59.0%	Chloroperlidae
	54.5%	Lepidostomatidae
	53.5%	Hydroptilidae
	52.7%	Planariidae
[*	44.6%	Asellidae
*	12.2%	Planorbidae]

Figure 4. BMWP family level prediction using RIVPACS II (Option 1, three seasons combined). Listing of predicted families to 50% probability level only, but all families observed at the site are indicated with an asterisk.

that are predicted with a probability in excess of 50%. Thus, routine BMWP family level samples, coupled with RIVPACS predictions, offer the biologist information on which to make an assessment of the problem at a site, after a consideration of those components of the fauna which are unexpectedly absent.

Using biological indices

Where many sites are undergoing rapid appraisal and there is a need for effective communication with non-biologists within an organisation or with the general public, the use of biological indices may become necessary. At Sanduck, the observed number of BMWP families was 21 but the total expected number was 34.7, giving an O/E ratio (Observed/Expected) of 0.61. As part of the RIVPACS prediction, expected values of BMWP score and ASPT (Average Score per Taxon) can also be calculated and these, when appraised against the observed values, gave O/E ratios of 96/219.7 = 0.44 for BMWP score and 4.57/6.3 = 0.73 for ASPT. Since all three O/E ratios are well below unity, there is strong evidence of organic pollution at this site.

A further step in the simplification of the initial output, in order to satisfy the requirements of a national survey, is to band O/E ratios into a small number of classes in order to represent change from good to poor biological quality. Banding divides up a continuum of sites and in the process results in the further loss of some information. However, it does allow broad comparisons to be made between sites in space and time. RIVPACS II was chosen for the biological appraisal of sites during the 1990 River Quality Survey (RQS) and prior to this the NRA asked the IFE to develop a banding system for reporting the results. The banding system had to be scientifically credible, but just as important it had to band sites in a manner which was relevant in terms of river management. For example, if it failed to highlight those sites most in need of remedial action, it would be of limited value.

Using the 438-site (RIVPACS II) database, a protocol was formulated for defining the lowest value of the O/E ratio for each of the BMWP score, the number of taxa and ASPT which might be acceptable as the definition of an unstressed site (Wright *et al.* 1991). Sites with higher O/E ratios would then be regarded as unstressed and allotted to band A. Band widths for three further classes (B-D) indicative of progressive loss of biological quality were based on the difference between an O/E of unity and the lowest acceptable O/E for band A.

The banding widths for each index as used in the 1990 RQS for sites sampled in three seasons are given in Table 6. The biological class given to a site was then derived as the median of the three individual bands, except when the band for ASPT was lower, in which case the ASPT band took precedence. This rule was devised because of the greater reliability in predicting ASPT but also because it would be possible to generate elevated O/E ratios for BMWP score and taxa if the sampling protocol was not followed and over-sampling took place at a site. Thus, in the example from the Wray Brook, where O/E ratios for BMWP score, number of taxa and ASPT were 0.44 (= C), 0.61 (= B) and 0.73 (= C) respectively, the site classification was band C.

The 1990 RQS took place throughout the UK and a total of 8,796 sites were examined, of which over 7000 were sampled in each of three seasons (Sweeting *et al.* 1992). For each site, the environmental data for predictions and the biological data, as BMWP family occurrences, were entered into computer files and validated. RIVPACS II was then used to obtain predictions of the expected fauna and to generate O/E ratios for each of BMWP score, number of taxa and ASPT before each site was allocated to a biological band using the criteria described above.

A further example, this time using a 1990 RQS result for a site which suffers from heavy-metal pollution, is given in Table 7. The biological class for this site is B. The individual O/E values demonstrate that the site is not suffering from organic pollution but is, nevertheless, taxon poor. Previous studies (Armitage 1980) suggest that this is due to the loss of a number of families which are sensitive to the presence of heavy metals.

Table 6. *Biological banding of the BMWP scores, number of taxa and ASPT based on samples from 438 sites for three seasons.*

BMWP = Biological Monitoring Working Party
ASPT = Average score per taxon
O/E = Observed/Expected from RIVPACS II prediction (Option 1)
Biological Bands A–D are explained in the text.

Band	O/E BMWP	O/E No. taxa	O/E ASPT
A	≥0.75	≥0.79	≥0.89
B	0.50-0.74	0.58-0.78	0.77-0.88
C	0.25-0.49	0.37-0.57	0.66-0.76
D	<0.25	<0.37	<0.66

Table 7. *BMWP index values and Biological Bands for the River Nent at Alston, Cumbria, based on samples from three seasons.*

For further explanation, see Table 6.

Index or Band	BMWP score	No. Taxa	ASPT
Observed (O)	101	15	6.7
Expected (E)	173.5	25.9	6.7
O/E	0.58	0.58	1.0
Band	B	B	A

Current and future developments

Before any system undergoes further development, it is essential to make a thorough appraisal of its current performance and also to take account of future needs. RIVPACS II was subjected to appraisal on two major fronts. First, it was used by biologists throughout the UK for the biological assessment in the 1990 RQS and second, the IFE was asked to undertake a thorough testing of the system, funded by the NRA.

Operationally, RIVPACS was applied to 8,796 RQS sites, not only in England, Wales and Scotland where RIVPACS was developed, but also in Northern Ireland. It was anticipated that use of the system in Northern Ireland would demonstrate some inadequacies, predominantly due to the fact that the fauna is less species-rich than it is in Great Britain.

The collection, logging and verification of the biological and environmental data for these sites, the majority of which were sampled in three seasons, was an enormous task, coordinated by NRA Thames region, and reflects great credit on all those who took part. Valuable lessons were learnt for use in future surveys. Examination of the RIVPACS II predictions suggested that, whereas the system performed well on some rivers, the database was inadequate for certain river types and in some geographical areas. In addition there was evidence that the system was under-predicting the fauna to be expected on chalk streams. Despite this, the overall response to RIVPACS II was favourable and the decision was taken to attempt to rectify current inadequacies in time for the 1995 RQS.

The testing exercises by the IFE involved two main approaches. First, a series of analyses was made on the 438-site database used to construct RIVPACS II. Here the objectives were to look for any inadequacies in the database, and also to discover any weaknesses in the classification system and the prediction mechanism. Predictions were made at BMWP family and species level before results were assessed using O/E ratios and comparison of the listings of taxa expected with those observed at a given site. Second, RIVPACS II predictions were undertaken on an independent set of 65 high quality sites from a variety of locations throughout Great Britain for which both BMWP family and species level identifications were available. Results from these predictions confirmed the performance of the system for some river types but demonstrated that it was inadequate in other river types or geographical areas.

Taken together, these tests provided valuable insights into the strengths and weaknesses of RIVPACS II and were instrumental in defining the work programme required for the development of a new system for the 1995 RQS. The new project includes increasing the database, exploring new methods of classification and prediction and, finally, enhancing the robustness of the system to increase its reliability for river management.

A very few of the 438 sites in RIVPACS II appear to be stressed or under-sampled and these will be screened out before the new database is expanded to about 700 sites, to include river types and geographical areas currently under-represented. The new database will also include sites from Northern Ireland.

The current procedures used for classification and prediction (TWINSPAN and Multiple Discriminant Analysis) have been in place for some years. Analyses undertaken by the IFE indicated that under-prediction of the fauna in chalk streams stemmed from the original TWINSPAN classification. This example and other findings of the study suggest that an exploration of some new procedures for site classification may offer improvements in predictive capability. This study is now underway, along with a further consideration of the prediction technique itself. Procedures developed during the IFE testing exercise will be useful in comparing the performance of RIVPACS II with the new system required for the 1995 survey.

Use of RIVPACS II for the 1990 RQS helped in the continuing dialogue between regional biologists and IFE staff concerning such subjects as banding, the relative merits of sampling in three, two or one season only and factors which may influence the reliability of the predictions. A number of these and associated topics concerned with the robustness of the system are now receiving attention in order to ensure that the new system meets customer requirements.

In an earlier section we reviewed the effects of environmental stress on macroinvertebrate communities. At present, the major focus for RIVPACS is on changes in taxonomic composition at a site (i.e. species losses and gains) and the consequent changes in taxonomic richness due to environmental stress. Currently, differences between the observed and expected fauna (and also BMWP score and ASPT) are being expressed in the form of O/E ratios. However, there are more sophisticated ways of looking at the relationship between the observed and expected taxa. Alternative approaches may offer relevant information for identifying the form of environmental stress and will be explored in due course. The method of sampling precludes a quantitative assessment of differences between the densities of the observed and predicted fauna. Nevertheless, at family level, use of logarithmic categories of abundance means that predictions can be offered with families listed in descending order of predicted abundance, for comparison with observations made at the site. Attempts will be made to develop procedures for measuring the differences between observed and expected abundances in order to use these early signs of stress.

In conclusion, developments currently underway should lead to a more reliable system for the assessment of biological quality based on macroinvertebrates, for use in the 1995 RQS. A similar approach was recently advocated by the Royal Commission on Environmental Pollution (1992), involving the use of environmental features to predict other elements of the flora and fauna (e.g. macrophytes and fish).

In the future, it should also be possible to streamline the acquisition of environmental data for predictions. A Geographic Information System holding a range of geological and catchment-based features, together with land characteristics, could help, particularly if a reliable suite of environmental factors could be acquired directly without the need to make site visits to record environmental data.

Finally, RIVPACS gives a prediction of the fauna to be expected at an unstressed site after sampling all the available habitat types for a specified period of time. However, it offers no information on whether a particular taxon is ubiquitous or whether, for example, it has a strong preference for emergent macrophytes compared to submerged macrophyte or non-macrophyte substrata. These considerations are important, both in river management and for the conservation of the fauna, and attempts have recently been made to acquire and present habitat-based information from lowland river sites to address this issue (Wright *et al.* 1993a).

This research has been possible as a result of the long-term commitment of a number of different organisations. In the early stages of the programme, major funding came from the Natural Environment Research Council and the Department of the Environment, with further support from the Welsh Office and the Scottish Development Department. More recently, new developments have been possible as a result of funding by the National Rivers Authority, the Scottish Office Environment Department and the Department of the Environment (Northern Ireland). The NERC funded the writing of this manuscript.

We would like to thank all the members of the team who have undertaken sample processing, data collation, data analysis and software development, and also our colleagues within what are now the NRA regions, the RPBs and the DoE(NI) for their help, interest and advice in numerous ways over a number of years.

Finally, our thanks to Diana Morton for typing the manuscript.

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