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What is river health?

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SUMMARY

1. Traditionally the assessment of river water quality has been based solely on the measurement of physical, chemical and some biological characteristics. While these measurements may be efficient for regulating effluent discharges and protecting humans, they are not very useful for large-scale management of catchments or for assessing whether river ecosystems are being protected.
2. Measurements of aquatic biota, to identify structural or functional integrity of ecosystems, have recently gained acceptance for river assessment. Empirical evidence from studies of river ecosystems under stress suggests that a small group of biological ecosystem-level indicators can assess river condition. However, physical and chemical features of the environment affect these indicators, the structure and function of which may be changed by human activities.
3. The term 'river health', applied to the assessment of river condition, is often seen as being analogous with human health, giving many a sense of understanding. Unfortunately, the meaning of 'river health' remains obscure. It is not clear what aspects of river health sets of ecosystem-level indicators actually identify, nor how physical, chemical and biological characteristics may be integrated into measures rather than just observations of cause and effect.
4. Increased examination of relationships between environmental variables that affect aquatic biota, such as habitat structure, flow regime, energy sources, water quality and biotic interactions and biological condition, are required in the study of river health.

Keywords: river health, ecosystems, river assessment

Introduction

Protecting ecosystems

The restoration and maintenance of 'healthy' river ecosystems have become important objectives of river management (Gore, 1985; Karr, 1991; Rapport, 1991). To ensure a reasonable probability of success these activities must be multifunctional (Brookes & Shields, 1996) because rivers are dynamic physical, chemical and biological entities. Aspects of the various biological, chemical, engineering and geomorphological approaches are reviewed in key river management volumes (Gore, 1985; Boon, Calow & Petts, 1992; Brookes & Shields, 1996). Recently there has been a trend towards the adoption of biological methods to assess river condition (Karr, 1991; Norris & Norris, 1995; Wright, 1995; Resh, Myers & Hannaford, 1996). This is because effects on biota are usually the final point of environmental degradation and pollution of rivers. Several legislative and practical developments (e.g. Karr, 1991; Norris & Norris, 1995) have also served to heighten the importance of biota and ecological values and have provided the methods for assessment, e.g. RIVPACS (Wright, 1995) and IBI (Karr, 1991). Consequently, guidelines for protection of rivers have shifted their focus from mainly physical and chemical measures (on the assumption that acceptable river condition would be achieved if these were met), to the inclusion of more biological measures.

The new emphasis on biota and ecosystems has led

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many people to embrace the term 'river health' which is often seen as being analogous to human health, giving a general sense of understanding (e.g. Rapport, 1989; Resh, Norris & Barbour, 1995). Unfortunately, the meaning of 'river health' remains obscure. Clearly, it is not the same as human health because ecosystems have a life of their own without humans (Rapport, 1989). It is not clear what aspects of river health sets of ecosystem-level indicators actually identify, nor how physical, chemical and biological characteristics may be integrated into measures rather than just observations of cause and effect.

Healthy or sick?

There is debate over the meaning of 'health' when applied to ecosystems (e.g. Rapport, 1989; Chapman, 1992; Scrimgeour & Wicklum, 1996). Rapport (1989) suggested three approaches that might differentiate 'healthy' from 'sick' ecosystems: (i) the absence of distress defined by measured characteristics or indicators; (ii) the ability of an ecosystem to handle stress, or bounce back — its resilience (Holling, 1973); and (iii) the identification of risk factors such as industrial or sewage effluents.

Of these the first has probably received most attention with a wide range of physical, chemical and biological indicators in use. Earlier attempts to define aquatic ecosystem health have failed to reach a consensus (Chapman, 1992). According to Karr *et al.* (1986), ' a biological system can be considered healthy when its inherent potential is realized, its condition is stable, its capacity for self-repair when perturbed is preserved, and minimal external support for management is needed'. This definition of ecosystem health emphasizes the biota, but ignores the use of non-biological parts of the ecosystem that may operate independently from biological ones but on which the biota may be dependent. The papers in this issue of *Freshwater Biology* extend the health debate specifically in relation to rivers.

Physical health?

Can physical features of rivers (channel geomorphology and hydrology) be 'healthy'? Certainly it has been argued that if the physical habitat is in poor condition we would expect the biological health of the stream to be affected adversely (Plafkin *et al.*, 1989; Brookes & Shields, 1996). The riparian zone and floodplain, through their connection with the main channel, will also have an influence on aquatic biota through organic matter inputs, shade and nutrients (Sweeney, 1992; Osborne & Kovacic, 1993). Earlier, Hynes (1975) argued that 'in every respect the valley rules the stream' where catchment character influences a river by large-scale controls on hydrology, sediment delivery and chemistry (Allan & Johnson, 1997). It follows that if we have an unhealthy catchment or valley, we will have an unhealthy stream. Usually, assessment is done in reverse: a stream may be assessed as being unhealthy and then it is concluded that the catchment is unhealthy.

Flow regime is a major component of the physical river environment. Flow regulation through dam and weir construction and water abstraction has led to severe stress being placed on many river ecosystems (e.g. Walker & Thoms, 1993; Thoms & Sheldon, 1997). In turn there has been a recognition of the need to allocate water to fulfill the needs of riverine environments, to protect these systems (e.g. Richter *et al.*, 1997). Unfortunately the connections between the disciplines of hydrology and biology have been poor (Statzner & Higler, 1986; Statzner, Gore & Resh, 1988; Newbury & Gaboury, 1993), especially at larger scales (Frissell *et al.*, 1986).

Fluvial geomorphology considers the processes controlling the structure and dynamics of river channels. Changes to flow and sediment regimes following catchment modification can markedly alter the physical nature of the channel and consequently the habitats that support organisms. River channels can function in unnatural ways (Schumm, 1988; Gurnell & Petts, 1995). Is it possible to have 'healthy' assemblages of biota associated with an 'unhealthy' channel? The scale at which river geomorphological changes operate may be much larger than that at which most biologists work and the absence of long-term, large-scale perspectives on the biota and the geomorphology is a bottleneck to understanding how river ecosystems interact with their catchments (Hart & Fonseca, 1996). However, Thoms, Ogden & Reid (1999) suggest that the use of palaeo-ecological records preserved in sediment deposits, may address some of these issues.

Society, economics and politics

The orientation of this issue of *Freshwater Biology* is predominantly scientific and relatively little is presented on social, economic and political views of 'river health'. However, Fairweather (1999) and Rogers & Biggs (1999) note that these views cannot be ignored and are essential in the definition of river health. Indeed, Rogers and Biggs go further, arguing that setting management targets for river health must be based on society's desires. Society may view a highly productive river as healthy, and a stable river as important for control and dependability (Rapport, 1989). Clearly, conflicts may arise if river ecosystems are found not to operate this way. Chapman (1992) makes the distinction between 'hard science' (ecologists with dirty hands) and 'soft science' (politicians determining scientific outcomes). Bringing together 'hard' and 'soft' science for protection of ecosystem health has been fraught with difficulties (Chapman, 1992). Nevertheless, Karr (1991) points out that advances in river assessment came about because of recognition that water resource problems involve biological as well as physical/chemical, social and economic issues.

Looking forward

Measurements of rivers made within the metaphorical 'health' framework share many of the objectives and practices of ecological research, such as the identification of spatial and temporal patterns and their underlying processes, and the use of indicators for detection and identification of patterns and processes (Scrimgeour & Wicklum, 1996). It may not be necessary to define river health to gain scientific and management value from the term (Rapport, 1989; Chapman, 1992; Scrimgeour & Wicklum, 1996). Rapport (1989) suggests that health is commonly taken to be the absence of detectable symptoms of pathology. From this viewpoint it may only be necessary to define the symptoms and their indicators. The term 'river health' will continue to be used; the papers in this issue are designed to clarify what is meant by it, and to suggest indicators, the links between them and how they might be measured.

Issues and developments

There have been recent advances both in the requirements for biological assessment of water resources and the methods used to make them (Rosenberg & Resh, 1993). For example, the Australian and New Zealand Environment and Conservation Council water quality guidelines (ANZECC, 1992) now call for biological assessment. Federal agencies and states in the USA have also called for direct biological assessment (Karr, 1991). The US Environmental Protection Agency (EPA) has included biological criteria in its water quality standards program and a range of other requirements that need biological assessment; see Karr (1991) for a summary. Much emphasis is being placed on rapid biological assessment in Australia, the United States and the United Kingdom, particularly using indices such as the Index of Biotic Integrity (IBI; Karr, 1981) and the benthic-IBI (B-IBI; Plafkin *et al*., 1989; Kerans & Karr, 1994), AusRivAS (Australian River Assessment System; Simpson *et al*., 1996), and RIVPACS (River Invertebrate Prediction And Classification Scheme; Wright, 1995). These approaches offer reductions in cost compared to previous methods, rapid turn-around of results, and the summary of results of site surveys into a score that can be understood by non-specialists. Using the human health metaphor such measures have been considered analogous to thermometers (Resh *et al*., 1995) because they give scores that can be compared to a threshold considered to be 'normal'. All of these methods have shifted the emphasis in biological measurements from intensive site assessments to multisite regional assessments.

The papers in this issue of *Freshwater Biology* demonstrate the current trend towards rapid biological assessment and in particular towards the use of invertebrates as indicators of river health. Other means of assessment are currently limited, although the use of palaeo-ecological methods (e.g. Thoms *et al*., 1999) provides an approach from which to understand longer term ecosystem processing and some impacts of natural and human-related activities on physical, chemical and biological variables.

In Australia, water allocations for the protection of river ecosystems (i.e. 'environmental flows') are now accepted as legitimate (in contrast to the former attitude that all river water was available for human use), and ways are being developed for determining how to make these allocations. For example, a national water resources policy which requires, *inter alia*, definition of water rights for the environment is a key part of the Council of Australian Governments (COAG) national agenda for microeconomic reform

(Water Resource Policy, 1994). This general ethic is similar in many countries. As a consequence environmental flow management strategies are now being developed. Richter *et al*. (1996) present a method by which to statistically characterize the temporal variability in hydrological regimes using biologically relevant statistical attributes. Thoms & Swirepik (1998) outline a method, derived from aquatic ecological theory, for determining water allocations to protect ecosystems. Their method accounts for the critical role of hydrological variability, and associated characteristics of timing, frequency, duration and rates of change, in sustaining aquatic ecosystems. They claim the method is intended for application to rivers where the conservation of native aquatic biodiversity and protection of natural ecosystem functions ('river health') are primary river management objectives.

Brooks & Lake (1998) have investigated how flow-generated disturbance affects benthic communities in 10 rivers with contrasting flow regimes. From a classification of southern Australian rivers, they identified rivers with 'constant' flow regimes and rivers with 'highly variable' flows, and they used portable weirs to assess the effects on invertebrates of altered flows. Their study is one of the few that make direct links between flow conditions and biota.

Australia, like many other countries, has seen an expansion of the activities of land-care groups and catchment management committees (Cullen, 1990). Their emphasis has been on rehabilitation of land, often with the objective of improving the condition of rivers, but with little direct measurement of the water, let alone river ecosystems (e.g. Walker & Reuter, 1996).

Assessments and comparisons

Assessment of river health involves comparisons. Indicators thought to represent river health are generally compared between sites that are thought to be similar in the absence of degradation. A recent development in river assessment has been the use of *reference conditions*, rather than reliance on single sites as controls. These reference conditions then serve as the control against which test site conditions are compared. The notion of a reference condition is really one of best available condition that could be expected at similar sites, and it is represented by several sites (Reynoldson *et al*., 1997). The reference condition is central to currently accepted ideas of 'biocriteria' being developed by the US EPA (Davis & Simon, 1995). A similar approach is being used in Canada to develop sediment guidelines for the Great Lakes (Reynoldson *et al*., 1995), and in the UK (Wright, 1995) and Australia (Parsons & Norris, 1996; Marchant *et al*., 1997) for river assessment.

Classification of stream types is essential for establishing characteristics of reference sites. Stream classification schemes include those based on mainly geomorphological features of the catchments and the creation of 'ecoregions' (e.g. Hughes & Larsen, 1988; Plafkin *et al*., 1989; Hughes, 1995; Omernik, 1995) and those using multivariate analysis (or predictive models) of biological features (e.g. Wright *et al*., 1984; Moss *et al*., 1987; Parsons & Norris, 1996).

Hughes, Larsen & Omernik (1986) argue that the logical basis for developing regional reference sites lies in the ability to group watersheds (i.e. catchments) and common stream types into regions by integrating available maps of terrestrial variables that influence streams. These reference sites are useful for estimating attainable conditions for evaluating temporal and spatial changes in ecological integrity, and for setting biological and environmental criteria. A possible flaw in this argument is that usually it is not whole streams but sites within streams that are the units being compared. Most indicators, especially biota, from a site at the top of one stream will be unlikely to match those from a site at the bottom of the same stream, or another similar stream.

Another approach for classification of rivers that has not been linked well with biological assessment is that of river geomorphology. Rosgen (1994, 1996a,b) presents a classification system for natural rivers based on geomorphology. This system includes slope, channel patterns, cross-sectional character, dominant particle size of channel materials, and entrenchment (the vertical containment of the river that determines the presence and/or extent of a flood-prone area adjacent to the bankfull channel). An approach such as this may be useful biologically but it has been criticized because it does not provide a classification that predicts all geomorphological processes (Miller & Ritter, 1996). Whatever scheme is used, habitats must be distinguished at appropriate spatial scales, and evidence must be provided that organisms recognize and respond to the habitat classification (Knight & Morris, 1996).

The reason for establishing reference conditions is to compare like with like. Many definitions of ecosystem health include words such as 'reduced' (e.g. Rapport, 1989); therefore comparisons are needed to determine if a reduction has occurred. The approach most commonly used has been to select reference sites that are 'minimally disturbed' (e.g. Wright, 1995; Parsons & Norris, 1996; Reynoldson *et al*., 1997). Often, pre-European disturbance conditions are set as restoration goals (e.g. Chapman, 1992; Scrimgeour & Wicklum, 1996), but those conditions may not be attainable. This target denies the place of humans in the landscape. Britain, China and India have been settled for thousands of years: can rivers in these countries never be regarded as 'healthy'? The majority of human land uses are not going to be removed from the landscape. Therefore, perhaps management targets should initially be set so they can be achievable. Morris (1995) suggests we should relinquish the notion of restoration to prehistoric conditions, although others (e.g. Karr, 1999) argue this point.

While the current approaches are acceptable for use in upland areas they are difficult to apply to larger lowland rivers (Thoms *et al*., 1999). Lowland rivers frequently have a long history of intense flow regulation, agricultural development and pollution (Petts, 1989). Hence, reference conditions may not be obtained from a study of the present day river system. Moreover, there are relatively few lowland rivers, and that may preclude adequate replication to establish reference sites. Thoms *et al*. (1999) argue that the use of palaeo-ecological methods would be of benefit in the establishment of reference conditions and the investigation of disturbance in these systems.

Clear definitions of desired conditions, given surrounding land uses, are required for effective management and assessment (Rogers & Biggs, 1999). The primary needs for a healthy ecosystem are biotic integrity (Karr, 1991, 1999) and sustainability, neither of which, especially sustainability, is well defined (Rapport, 1989; Karr, 1991, 1999). Ecosystems need not be pristine (few are, now, because of large-scale changes such as the ozone hole, acid rain, global air pollution), but still can be judged healthy (Rapport, 1989; Chapman, 1992). The final conclusion on health may be dependent on social issues. For example, a river may be judged healthy if a single species commercial fishery is sustainable, but not healthy if a varied recreational fishery is lost. Judgments of ecosystem health take into account more than strictly ecological functions — uses or human amenities derived from the system, for example (Rapport, 1989).

Rivers can be restored (Gore, 1985; Brookes & Shields, 1996) and also enhanced (Rapport, 1989). A more useful target for management may be the best possible condition, given acceptable land or water use. Many balk at this suggestion, afraid that it will lead to a downward spiral in environmental quality. However, the reverse may be true. For example, if conditions in the upper part of a catchment are improved so that fewer problems are exported downstream, acceptable targets downstream may be raised rather than lowered.

We contend that acceptable reference conditions for river health should be based on ecological understanding (e.g. biotic integrity). Features should be set *a priori* and sites to represent the reference condition selected and classified to provide site-specific comparisons of indicators of river health.

Indicators and scales

There are many possible indicators of river health, including measures of structure and function both of the biotic and of the physical components. Indicators of river health may represent spatial scales ranging from local to catchment, temporal scales from instantaneous to long-term, and may be direct or explanatory measures. Chapman (1992) suggests that holistic, 'top-down' approaches that assess structural ecosystem components are likely to be most useful, while 'bottom-up', reductionist approaches (Scrimgeour & Wicklum, 1996) are more likely to be useful for elucidating reasons for impacts. Karr (1991) argues that the reductionist viewpoint of several disciplines in state and federal water management agencies in the US has been a major impediment to the development of an understanding of 'biological integrity'. This view is supported by Schindler (1987) who concludes that the widespread use of single-species bioassays, complicated models, and impact-statement studies have been unsuccessful at predicting the effects of human-induced stress on biological systems. Studies of population dynamics, food-web organization, and taxonomic structure of communities have been more successful. For these reasons larger scale indicator approaches may be more appropriate (Fairweather, 1999).

Biota

There are many biological indicators from which to choose (Norris & Norris, 1995; Cranston, Fairweather & Clarke, 1996; several papers in this issue), but the most commonly used have been benthic macroinvertebrates (Resh & Jackson, 1993). Other commonly used groups are fish (Harris, 1995) and plants (Whitton & Kelly, 1995). Cranston *et al*. (1996) provide a summary of the types of biota that may be used to assess rivers and their attributes. Biological indicators may be grouped into several categories. Karr (1991), using fish, selected these categories: species richness and composition (six indices), trophic composition (three indices), and fish abundance and condition (two indices). Harris (1995) used the same categories but modified the indices in each to suit Australian conditions. Similar categories have been proposed for invertebrates (e.g. Barbour *et al*., 1992), structure, community balance, and functional feeding groups.

Thus, many biological indicators are available for assessing river health and biotic integrity. Regardless of the taxonomic group used, taxonomic richness, or a subset of it (e.g. richness of Ephemeroptera, Plecoptera, or Trichoptera), has been frequently selected as a robust indicator. Poorer conditions are usually indicated by a loss of taxa. Taxonomic richness of invertebrates is central to the British RIVPACS (Wright, 1995) and the Australian AusRivAS (Simpson *et al*., 1997) methods for assessment of rivers. Some of the more complicated indices have been found wanting (e.g. Reynoldson *et al*., 1997). In keeping with the arguments presented by Schindler (1987), Karr (1991) and Chapman (1992), and the demonstrated usefulness of broad scale structural measures such as taxonomic richness, we feel that such measures should be considered for use before other more complicated ones.

Habitat: physical and chemical indicators

Physical and chemical indicators (mostly of water quality) are the most commonly used and largest variety available (e.g. ANZECC, 1992; Hart, Maher & Lawrence, 1999; Maher, Batley & Lawrence, 1999). Most are highly specific measurements of single chemicals and offer little integration. Interpretation comes largely from experimental tests on the effects that they have on biota and results from these are used to set guidelines to protect rivers. Such measures are distinctly 'bottom-up' and may explain causes of damage to river health and biotic integrity rather than ecosystem condition (Karr, 1991; Chapman, 1992). Application of standardized criteria for chemical values fails to recognize natural geographic variation in water chemistry and resulting impacts, e.g. antagonistic interaction of heavy metals with major cations and effects of pH on solubility. Developments are now proposed for Australia's national water quality guidelines that aim to provide measures that are more ecologically meaningful and integrative (Hart *et al*., 1999; Maher *et al*., 1999).

Process geomorphology is concerned with factors operating at various scales that affect the function and form of rivers (e.g. de Boer, 1992). Changes to catchment conditions and flow regimes can markedly alter the functioning of river channels and thus the habitat available for organisms. Biota aside, changes to catchments and flows modify river channels via changes in erosion rates after catchment and riparian clearing, separation from floodplains resulting from drainage and flow reduction. Therefore, it is sensible to consider indicators of the geomorphological condition of rivers in their own right. It may be quite possible to have degraded channels that have a quite healthy biota associated with them.

The International Union of Geological Sciences (IUGS) recently constructed a list of 27 geoindicators, within 16 fields, that assist with the assessment of the condition of abiotic components of the environment. Seven of the 27 geoindicators are directly related to the condition of river systems (Table 1). It is suggested that they indicate local, regional and global change during observational periods of up to 100 years (Osterkamp & Schumm, 1996). To use the physical characteristics of rivers as indicators of environmental conditions, it must be understood that alluvial rivers, in particular, are inherently unstable. Natural changes, as well as those imposed by human-induced causes, can all be interpreted as the result of environmental change. The challenge is to separate the two. Moreover, change can occur over a variety of time scales so there must be assessment not only of the main stimulus of change but also of the

Table 1 Physical indicators of river system condition

Indicator	Components
Sediment sequence and composition	Rate of accumulation
	Sediment calibre
	Mineralogy
	Geochemistry
Soil and sediment erosion	Rate of erosion
	Source of sediment
	Mode of transport
Stream flow	Total annual flow
	Variability
Stream channel morphology	Slope
	Pattern
	Cross-sectional dimensions
Stream sediment storage and load	Sediment flux
	Mode of transport
Surface water quality	Turbidity
	Total suspended solids
Floodplains/wetlands structure and hydrology	Wetting and drying regimes
	Connectivity with the river
	Area

time scale over which this change is occurring. Thoms *et al*. (1999) suggest that the indicators contained in sediment deposits can be useful for the interpretation of longer time scales. Additionally, within river systems there is great variation, and each river changes throughout its length. Osterkamp & Schumm (1996) propose that, overall, the most valuable observations from rivers for use as indicators of physical change are continuing measurements of water and sediment discharge.

Integration

Human activities such as land use change and water resource development can alter physical, chemical and biological processes of river ecosystems thus modifying their biological communities (Karr, 1991). While there is no doubt that biological criteria can assess these alterations there has been little detailed examination of the relationships between ecosystem degradation and biological response. Toxicologically such relationships are referred to as dose–response curves. Karr (1999) cites Karr & Chu (1998) who found a linear decline in the richness of mayfly taxa (Ephemeroptera) with increases in the percentage of impervious surface area surrounding the lowland streams of Puget Sound. Thoms (1987) shows a more complex relationship between percentage urbanization and habitat character, in this case the sediment texture of gravel bed rivers, for 12 urban streams in the UK. There was a sigmoid relationship between habitat character and the age of the urban area, its position within the catchment, the nature of urban activities (industrial vs. residential) and the size of the urban area. Marchant *et al*. (1997) demonstrated linear relationships between the observed/expected ratio for number of taxa, from a predictive model, and the log-transformed 14-week means of faecal coliforms, total organic carbon, ammonia and total zinc. This also demonstrated the difficulty of matching spot water quality measures with biological measures that integrate effects of long periods. Better understanding of the relationships between the environmental variables that affect biological condition (Fig. 1) will advance river assessment and our understanding of river health.

Stream biotic composition is strongly influenced by physical habitat. Stream habitats provide the template upon which the ecological organization and dynamics of lotic ecosystems are observed (Minshall, 1988; Resh *et al*., 1988; Poff & Ward, 1989; Townsend & Hildrew, 1994; Richards, Johnson & Host, 1996). Elton (1966) and Southwood (1977) advocated a habitat-centred view of ecological systems and there is considerable evidence to support this view for streams (e.g. Hynes, 1970, 1975; Vannote *et al*., 1980; Hawkins, 1984). Many assessment programs incorporate habitat assessment (e.g. Plafkin *et al*., 1989; Wright, 1995; Parsons & Norris, 1996).

Habitat variation through time and across the landscape provides dynamic patterns to which organisms, species, and communities must either adapt or perish (the 'habitat templet', *sensu* Southwood, 1977; Townsend & Hildrew, 1994; Townsend, Dolédec & Scarsbrook, 1997). This template can be seen as a moving mosaic of environmental conditions to which the biota adapt (Ebersole, Liss & Frissell, 1997). Biotic adaptation continues until drastic environmental change, often from human activities, occurs and results in extinction. The persistence and recovery of the habitat–biota system is dependent upon characteristics of the habitat as well as the biological communities living within it (Detenbeck *et al*., 1992). Fundamental to assessment of river health and biotic integrity is an understanding of the links between the habitat in which organisms live and the factors shaping it. Desired changes in a river's biotic integrity will often be set as targets for management, but the

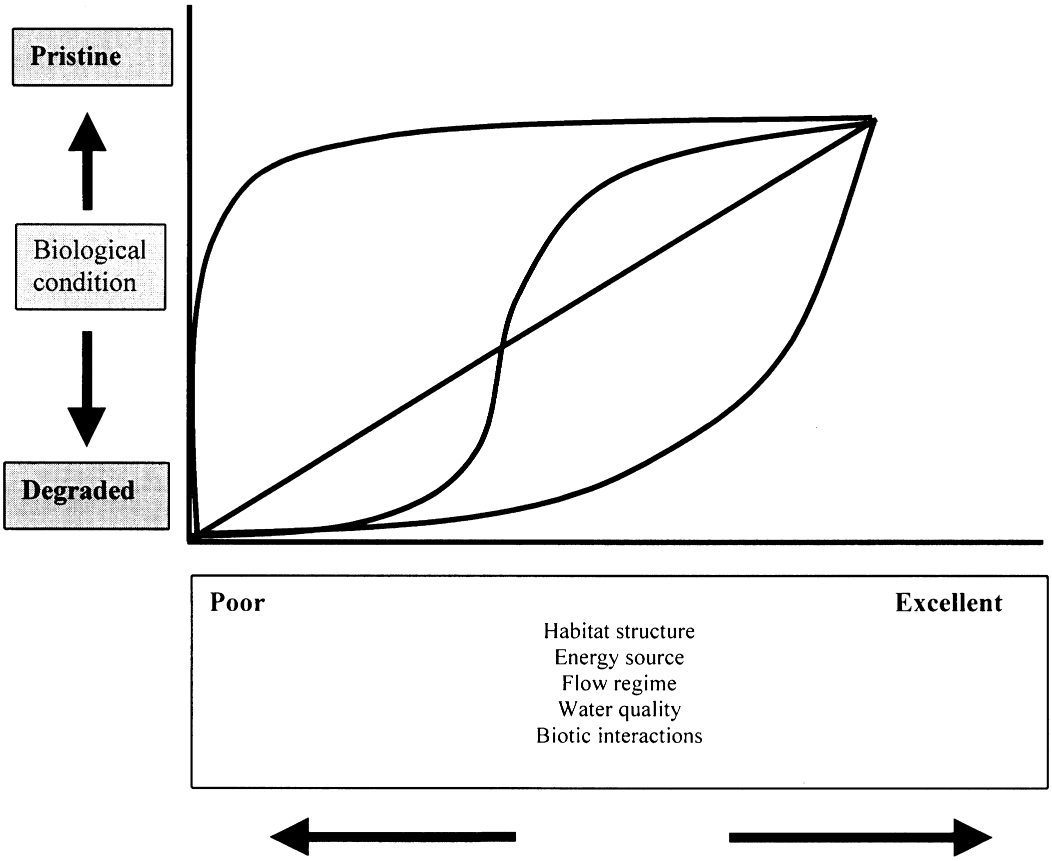

Fig. 1 Hypothetical relationships between environmental variables that affect aquatic biota and biological condition.

changes will usually be brought about by altering the physical and chemical environment. It will be essential to understand what features should be altered to achieve the desired changes.

Most habitat assessment uses the US EPA guidelines (Plafkin *et al.*, 1989) and many assessments based on fish surveys largely measure local-scale indicators (e.g. Simonson, Lyons & Kanehl, 1994; White, 1996). However, variables operating at larger scale may be more important in controlling local-scale effects on fish (e.g. Roth, Allan & Erickson, 1996) and invertebrates (Parsons & Norris, 1996; Richards *et al.*, 1996) and geomorphological processes (e.g. Frissell *et al.*, 1986; de Boer, 1992). It is desirable that comparisons are site-specific so that differences can be clearly attributed, where necessary, to other than natural features of the site. Large-scale studies must take into account zoogeographic attributes of the biota including the regional pool of taxa with potential to colonize the site of interest (Frissell *et al.*, 1986; Caley & Schluter, 1997). To account for such effects, habitat variables (Table 2) are measured at several spatial scales as part of many rapid biological assessment programs — Australia's AusRivAS program, for example.

Large-scale landscape attributes, namely land use, surficial geology, elevation, and hydrography related to physical habitat characteristics, were found by Richards *et al.* (1996) to have the greatest influence on macroinvertebrate assemblages. Stream buffers (100 m) were more important than whole catchment data for predicting sediment-related habitat variables; however, channel morphology was more strongly related to whole catchments (Richards *et al.*, 1996). It was concluded that catchment-wide geology and land use characteristics may be more important than stream buffers for maintaining or restoring stream ecosystems.

Ebersole *et al.* (1997) propose a framework for stream habitat restoration emphasizing stream habitat–biota development. They argue that restoration is fundamentally about allowing stream systems to re-

Table 2 Commonly used predictor variables for AusRivAS models

Variable	No. of models† using variable as a predictor	Variable	No. of models using variable as a predictor
Longitude	31	Macrophyte taxa*	6
Latitude	28	Flow pattern	5
Alkalinity	28	Macrophyte cover	5
Altitude	21	Shading	5
Distance from the source	15	Bedrock	5
Catchment area	12	Stream width	4
Conductivity	9	Riffle depth	4
Stream slope	8	Pebble	4
Riparian width	8	Edge bank vegetation	4
Cobble	8	Vegetation category*	4
Boulder	7	Range of mean annual air temperature*	4
Stream order	7	Gravel	3
Discharge	7	Silt	3
Sand	6	Clay	2

†No. of models = 40.
*Variables used only by six models in Victoria.

express their capacities. To achieve that, historical patterns of habitat development and developmental constraints should be identified; constraints should be relieved; sensitive, critical, or refuge habitats should be classified; the developmental diversity that remains should be protected; and the biotic responses to habitat development should be monitored. This approach attempts to match the potential of the environment to produce habitat (for fish) with that which could be realized given constraints of human use.

There are several strengths to the current rapid assessment methods. First, test sites are matched with groups of reference sites using data independent of human activities. Second, all available reference sites are used to calculate the probabilities of taxa occurring at a test site. Third, predicted taxa are compared with those observed at the site to provide a simple index. Fourth, comparisons account for the independent characteristics of the test site providing a site-specific assessment. Fifth, analyses are performed in a standardized computer program, and outputs are straightforward (e.g. observed/expected variable ratio), with bands of impairment (Wright, 1995) that are easy for non-specialists to understand. Similar, more integrated approaches could be applied to an overall assessment of river condition.

Conclusion

It may not be necessary to define the term 'river health' to gain scientific and management value from it. The symptoms and the indicators of poor health may be more easily defined and these should include physical, chemical, biological, social and economic variables. There have been marked advances, particularly in rapid biological assessment methods, that are now coming into widespread use, particularly at large scales. However, the biota often will be dependent on their physical and chemical environment for survival. The traditionally used physical and chemical indicators are also being revamped in Australia to render them more ecologically relevant.

Analysis of river geomorphological and hydrological characteristics in relation to river biota is less well developed, but research is gaining momentum. River rehabilitation activities and the recognition of the need for environmental flows have been catalysts for work in this area. This is, perhaps, the most pressing need — that is, the development of geomorphological, hydrological and chemical indicators of river health with an understanding of their relationships to aquatic biota.

The papers in this issue have been selected to bring together these areas of research that we believe are too often undertaken in isolation.

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