An overview of river health assessment: philosophies, practice, problems and prognosis

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SUMMARY

1. Philosophically, the term ‘river health’ is useful because it is readily interpreted by the general public and evokes societal concern about human impacts on rivers. The common goal of achieving healthy rivers unites ecologists and the general public because the value of the ecologists’ contributions is clear (and, hence, funded). The difficulty arises in the choice of relevant symptoms because there is a wide variety that can be measured with varying accuracy at a broad range of spatial scales. These indicators may respond to impacts at different time scales, and no single indicator is a ‘silver bullet’ that reveals river health unequivocally.

2. In practice, choice of indicator often shows personal bias, technical considerations, and constraints of knowledge. Selection of appropriate spatial and temporal scales for these measures is crucial. Although most measurements are spot samples (e.g. concentration, abundance, species richness), assessment of river health based on changes in ecological processes such as post-disturbance recovery rate or nutrient spiralling lengths may be more suitable in some cases.

3. Problems include validation of the indicator, its response time at a range of scales, and the reliability of its measurement. Assessment of river health should be accurate, timely (warning of deterioration instead of waiting until the patient is terminal), rapid (so that the response is swift), and inexpensive. The connectedness of running waters with their floodplains and catchments must be explicitly recognized. Hydrological and geomorphological modifications of rivers usually affect their health by severing or impairing the linkages, and the ‘cure’ may lie in addressing these causes. Often, we need landscape-level data for management because this is the scale where cumulative effects of impacts are evident.

4. The prognosis is uncertain. We need to explore further the use of integrative measures of river health, and focus on establishing a link between the measure and impaired ecological integrity. Ecosystem-level variables (e.g. estimates of production or respiration) show promise and recent technological advances make these more accessible. Data analytical approaches (e.g. multimetric vs. predictive models) need further debate but must not overlook the importance of high quality and relevant input data. Appropriate choice of indicators, rigorous sampling and analysis, and careful data interpretation must be matched with effective communication to policy-makers and the public. When this occurs, the concept of ‘river health’ becomes more than just a rhetorical tool.

Keywords: river health, assessment, models, scale, ecological integrity, indicators

Introduction

The papers in this special issue range from those
Defining river health

Some ecologists wince at the metaphor ‘river health’, feeling that the analogy is often taken too far and that it oversimplifies a complex issue. Karr (1999) reviews
the development of the concept of stream health, and, like Meyer (1997), makes a convincing argument for the incorporation of ecological integrity (maintaining ecosystem structure and function) and human values (what society values in the ecosystem) into the definition of river health (Fig. 1). This closely parallels the relationship between science and public opinion in the protocol for ecosystem management proposed by Stanford & Poole (1996).

In contrast to definitions of healthy ecosystems based on solely ecological criteria (e.g. Haskell, Norton & Costanza, 1992), judgements of river health must include human values, uses and amenities derived from the system (Rapport, 1989, 1995; Steedman, 1994; Meyer, 1997; Fairweather, 1999; Karr, 1999). Ecological criteria include sustainability, resilience to stress, and ecological integrity — the capacity to support and maintain a balanced, integrated, adaptive biologic system having the full range of elements and processes expected in the natural habitat of a region (Karr, 1996; p. 101). There are strong parallels between Karr’s (1996) definition of ecological integrity and the strategy of programs such as AusRivAS that measure river health by comparison with a nearby natural reference site (e.g. Smith et al., 1999; Turak et al., 1999).

Inclusion of the human dimension gives the concept of stream health part of its novelty and may provide some impetus for advances in river ecology (Meyer, 1997). Most humans value rivers as sources of clean water for drinking and washing, industrial and agricultural purposes, conduits for pollutants, places for recreation and aesthetic pleasure, fisheries, and other uses (Fig. 1). An ‘unhealthy’ river may satisfy only a few of these uses. Often human uses of rivers conflict with each other, damaging the health as assessed by some criteria (see examples in Karr, 1999). Although river ecologists may help identify where conflicts will arise, resolution of these issues needs societal debate mediated by well informed managers and policy-makers (Meyer, 1997). Increasingly, river ecologists are being brought into the debate, evidenced by special sections devoted to river management issues in journals such as Freshwater Biology and Journal of the North American Benthological Society.

Despite the apparent value of the health metaphor, it has its critics. Many argue that the term encourages a simplistic view of ecosystems as ‘superorganisms’ (Suter, 1993; Callicott, 1995; Jamieson, 1995; Simberloff, 1998). Furthermore, it is claimed that attempts to define ecosystem health operationally have resulted in creation of indices of heterogeneous variables that have no ecological meaning and no diagnostic power (Suter, 1993). However, this may indicate poor choice of metrics rather than being the fault of the metaphor (Karr, 1999). Perhaps the greatest challenge lies in the application of the health metaphor in developing policy (Jamieson, 1995; Ross et al., 1997) because of the difficulty in compromising human values about what constitutes a healthy river or ecosystem (Lackey, 1997) and the fact that policy usually controls funding priorities. Traditionally, scientists have proved to be poor at effectively communicating their knowledge in such a way as to affect policy and management of wetlands, but this trend is changing slowly (Cullen, 1998).

Choosing indicators

Perhaps the most important role for river ecologists in this field is to identify and measure indicators of river health (given some general societal consensus about what constitutes a healthy river (Meyer, 1997)). We cannot wait until this consensus is finally reached before we start seeking effective indicators. In the same way that the debate on human health continues, so too will there be uncertainty in the resolution of conflicting opinions about what constitutes a healthy river. Meanwhile, most ecologists seem to have explicitly or implicitly adopted the general idea that

![Ecological values](image-url)
A healthy river and its ecological integrity are as ‘natural’ and intact as possible – the dilemma arises when we seek variables that reveal departures from this natural state that exceed background variability. This is especially acute when no unmodified reference state exists (e.g. Richardson & Healey, 1996).

Such departures often prove to be caused by humans, but as most rivers are inherently highly variable, the departures cannot be detected unless they are quite extreme. The 1000 km cyanobacterial bloom in the Darling River, Australia, in summer 1991–92 was a spectacular example of a natural process exacerbated by human use of the river, that dramatically changed public perception about the health of this river (Alexandra & Eyre, 1993) and also increased the funding for research into causes of cyanobacterial blooms in Australian rivers. Ideally, indicators of river health should alert managers before extreme events occur, and in sufficient time to permit preventative action. Societal and political inertia, that delays preventative steps being taken in spite of river ecologists’ alerts, frequently results from communication problems. These should be partly resolved by a common definition of river health, and wider use of the term (Fairweather, 1999; Karr, 1999).

Cairns & McCormick (1992) recognize three types of indicators: early warning indicators that signify impending decline in health; compliance indicators that reveal deviations from acceptable limits; and diagnostic indicators that show the causes of the deviations. Most of the indicators reviewed in this conference are compliance indicators, and therefore depend heavily on definition of ‘acceptable limits’. However, it is possible to incorporate all three into an assessment of river health. For example, Hart et al. (1999) advocate a risk-based approach employing indicators that are ecosystem-specific and that are guided by the impact caused by the issue. In this context, some indicators reviewed by Hart et al. (1999) are diagnostic indicators, such as the nutrient concentrations in their decision tree for assessing cyanobacterial blooms (Fig. 2 in Hart et al., 1999). The development of growing conditions over periods of time exceeding 6 days would provide early warning of an impending bloom while changes in cell counts and toxicity levels might exceed those set for that particular river system.

Ideally, a good indicator should fill all three of the above categories. It should also be relatively cheap and quick to measure, amenable to sampling and assessment even by untrained field workers, readily repeatable over time, and there should be a good knowledge of its natural variation in the river where it is used (Cairns, McCormick & Niederlehner, 1993). As far as possible, it should be unambiguous in response to threats to river health, but as most variables interact in complex ways this may be unattainable (Suter, 1993). Further, a robust indicator is unlikely to also be sensitive (Fairweather, 1999) and some trade-offs are needed. Occasionally, an indicator may be chosen because it is ‘charismatic’ – it may be a species that has a high public profile (e.g. platypus) or is readily associated with a sensitive issue (e.g. cyanobacteria, heavy metals). However, an indicator needs to be credible both to scientists and to nonscientists in case the charismatic species turns out to be a poor indicator by all other criteria. Finally, the indicator should be one that can be validated: the reliability of the data and what they indicate should be clear (Cairns et al., 1993; Walker et al., 1996; Fairweather & Napier, 1997).

Selecting the minimum subset

It is clear from the papers in this issue that a large number of indicators is available for assessing river health. Not all of them meet the criteria above. The aim is to identify a small manageable subset and, if possible, to link specific indicators with ecosystem components and processes that are valued by society (Meyer, 1997). Where possible, we must also be able to identify the links between the levels of these indicators so that we can assess the correct scale at which to measure the direct response to an impact (Attrill & Depledge, 1997).

Fairweather (1999) posits three contemporary approaches for selecting among indicators of river health: first, a haphazard selection from divergent perspectives (e.g. chemistry vs. biology) based on personal biases of managers and politicians; second, adoption of a single perspective that is either better developed, favoured by circumstance or seen as an umbrella for protecting other sets of values (e.g. AusRivAS; Smith et al., 1999; Turak et al., 1999); and third, the synthetic approach that integrates quite distinct perspectives (e.g. the Index of Stream Condition; Ladson et al., 1999).

The first seems to typify debate about water quality guidelines in Australia and elsewhere but thankfully
seems to be used less frequently (Hart et al., 1999). The second, exemplified by the current emphasis on macroinvertebrate assemblage composition as an indicator of river health, has a number of strong advocates globally. The advantages of macroinvertebrates as indicators are well reviewed (Rosenberg & Resh, 1993; Chessman, 1995) and have led to the development of comprehensive sampling programs in the US (Plafkin et al., 1989), UK (Wright, 1995) and Australia (Norris, 1995; Norris & Thoms, 1999). There appears to be a general consensus that if macroinvertebrate assemblages indicate a healthy river, other values are likely to be protected as well although the umbrella approach may have its drawbacks (Simberloff, 1998).

The synthetic approach perhaps comes closest to satisfying the philosophical requirements of a pluralistic definition of river health. However, it is also likely to require a larger suite of variables, be heavily scale-dependent (Townsend & Riley, 1999) and practitioners must be alert to the dangers of data simplification that may sacrifice crucial information in an effort to summarize a huge data set (see Norris & Norris, 1995). Karr (1999) suggests that multimetric approaches would be fruitful, provided that selection of the individual metrics is made carefully and the data are collected properly, and this is exemplified by the use of the Index of Biotic Integrity for Australian fish (Harris & Silveira, 1999).

The single perspective vs. the synthetic approach

The relative merits of Fairweather’s last two approaches (Fairweather, 1999) can be contrasted using results from the papers in this conference. This also demonstrates common ground between these approaches, such as the importance of high quality input data, appropriate choice of scale, and the way in which indicators can be selected from a number of single perspectives to facilitate interpretation of the results of a synthetic approach.

The pluralism of the synthetic approach has intuitive appeal. There are two levels at which these synthetic studies can be viewed: a collection of variables from a number of perspectives or a process-oriented approach where focus is on an ecological process that may involve multiple elements (e.g. post-disturbance recolonization and the relationship between physical habitat and its inhabitants). As an example of the first type, Ladson et al. (1999) developed an Index of Stream Condition (ISC) that assesses features of hydrology, physical form, riparian vegetation, water quality and macroinvertebrate assemblage composition in streams in the state of Victoria, Australia. In this context, the outcome is a multimetric that has proved valuable as a tool for measuring and reporting on waterway management performance. It also has roles in setting priorities in the state, encouraging an integrated approach to stream management and providing feedback to the general public about the success of stream management (Ladson et al., 1999).

In a similar fashion, Townsend & Riley (1999) measured a large number of physical, chemical and biological features in a New Zealand river but extended their interpretation of the results to include the role of perturbations at several scales of time and space. This approach is not only synthetic but attempts to measure ecological variables that illustrate the importance of disturbance regime and refugia to river health. More importantly, it explicitly recognizes the value of linkages between wetlands and their catchments – protecting river health includes protecting the integrity of these linkages.

Management of rivers in semiarid regions poses special problems when synthetic approaches are being used, because of the over-riding influence of the varying river-flow regime (e.g. Walker, Sheldon & Puckridge, 1995). In Kruger National Park, South Africa, indicators of river health are chosen to take into account the inherent flow variability and are used to generate hypotheses about the limits of acceptable change in ecosystem structure (‘Thresholds of Probable Concern’ or TPCs; Rogers & Biggs, 1999). Careful choice of TPCs provides indicators of threats to river health in the park. For example, sedimentation threatens the biodiversity of bedrock-controlled rivers so a TPC reflecting permissible rates of change in bedrock in specific reaches is generated (Rogers & Biggs, 1999). This approach extends to other aspects of the rivers in the park and yields a subset of the possible criteria that indicate threats to river health, resulting in a manageable number of indicators with which to assess ecosystem condition relative to stated goals. Choice of indicator must always be based on specific hypotheses of changes in response to threats to river health.

Meyer (1997) suggested that rates of some ecologi-
cal processes may be good indicators of river health and observed that impacted streams recover from disturbance more slowly than healthy rivers. However, there seems little empirical evidence for some of these claims (Townsend & Riley, 1999). Nonetheless, use of models to predict impacts of catchment land use upon ecological processes (e.g. productivity, respiration, decomposition and nutrient spiralling; Fig. 3 in Townsend & Riley, 1999) point to possible indicators that may prove to be integrative and relevant.

There have been recent technological advances in rapid measurements of these process variables, and this seems to be a productive research direction. A good example is the study by Bunn, Davies & Mosisch (1999). They suggest that measurements of ecological patterns provide little information about stream ecosystem processes. Instead, they assess river health using technologically elegant measurements of gross primary production (GPP) and respiration coupled with stable isotope analysis, and find a link with riparian canopy cover and, to a lesser extent, catchment clearance. However, integrative ratios of GPP and respiration alone are not considered reliable as indicators of river health (Bunn et al., 1999) suggesting that while separate metrics may be useful, their combinations into multi-metrics are not always successful.

Comparing single perspective studies

Although physical habitat is the fundamental template for life and ecosystem processes in rivers (Allan, 1995), Maddock (1999) considers that methods to assess physical habitat in terms of river health have lagged behind those for water quality and ecological assessment. River restoration procedures such as the reintroduction of physical habitat at a range of scales (artificial riffles to whole meanders; Boon, Calow & Petts, 1993) demonstrate scientific and general public acceptance of the relevance of physical habitat to river health, and the link between physical habitat and the living inhabitants. But this single perspective of physical habitat is assumed to be the fundamental variable, and must be established for each system before it is applied uncritically. Nonetheless, inclusion of the physical habitat in most indices of stream condition seems mandatory (Ladson et al., 1999), and there is need for more collaboration between ecologists, hydrologists and fluvial geomorphologists to identify the most suitable physical variables for assessing river health.

A logical extension of the importance of the physical habitat at a finer scale is discussed by Maher et al. (1999) who focus on using chemical variables to assess the health of the sediment ecosystem. They assume that a ‘healthy’ sediment ecosystem is a critical part of a healthy river, and suggest that measurements of concentrations of selected chemical contaminants provide a rapid and relatively inexpensive surrogate indicator of health. These concentrations are compared with guidelines based on demonstrations of adverse effects upon the biota, but because these data are statistically derived values from a large effects database (Maher et al., 1999), the best indication possible is ‘potentially unhealthy’. The issues here are complex – are the guidelines too crude, are there too few data, are the biotic responses too variable and prone to modification by complex mixtures of contaminants, are the wrong contaminants being measured, or is the problem a combination of these issues? It is important not to let these issues nullify the advantages of the chemical perspective but they illustrate the benefits of synthetic approaches to assessing river health (cf. Fairweather, 1999; Karr, 1999).

For many reasons (see above), composition of macroinvertebrate communities has been widely used to assess river health; in Australia, this has culminated in a national program (Australian River Assessment Scheme (AusRivAS)) based on the RIVPACS model used in Britain (see Wright, 1995). One common theme that emerges is that substantial inherent variability may cloud detection of impacts that unequivocally result from human activity. This is especially true for large-scale assessments (Marchant et al., 1999). Sampling protocol issues are also important and Smith et al. (1999) emphasize the importance of quality control in faunal identification and sorting methods. The present models used in AusRivAS seem capable of detecting severe ecological impairment but may not identify subtle impacts such as erosion or slight nutrient enrichment, e.g. in samples from a number of rivers in Western Australia, perhaps because of the low numbers of families in that region. In north-western Australia, macroinvertebrate data indicated marked differences between habitats, but a lack of clear biogeographical patterns (Kay et al., 1999). Conversely, in eastern Australia, many sites
could be grouped into broad geographical regions (lowland, western-flowing rivers vs. coastal fringe streams vs. highland streams) although habitat differences were still marked (Turak et al., 1999). Results from these studies reiterate Karr’s (1999) recommendation that environments be classified into homogenous sets at the finer scale of habitat. Failure of the broader-scale approach in some areas, even when habitats are separated, may result from specific features of the regional biota (e.g. the low taxonomic diversity and ubiquity of common, mobile taxa such as Odonata and Hemiptera in north-western Australia (Kay et al., 1999)). The same problem occurs when diatoms are used to assess river health in south-eastern Australia (Chessman et al., 1999) where the wide distributions of many diatoms hampers precise site classification. In these situations, the fault may not be in the indicator but in the underlying model. Not only might the models need adjustment, but also there is a need for better insights into how these variables relate to ecological processes (Chessman et al., 1999).

Scaling considerations in management and indicator selection

Discussions about spatial scale in stream studies often invoke modifications of the hierarchical classification of physical habitat by Frissell et al. (1986) (e.g. Fisher et al., 1998), and Maddock (1999) reviews applications of this scheme to river restoration and physical habitat assessment. In this sense, the importance of assessing scale-dependence of these variables (see MacNally & Quinn, 1998), whether for ecological or management purposes, applies to single perspective and synthetic approaches. Marchant et al. (1999) demonstrate convincingly the scale-dependence of predictive models of macroinvertebrate distribution, and the futility of the ecoregion approach of habitat classification in Victoria because community similarity only weakly matches geographic proximity (see also Karr, 1999). This has serious implications for selection of the right model for river management – if environmental disturbances occur at large spatial scales, appropriate management must also focus at these scales (Johnson & Gage, 1997).

Increasing attention is being paid to river conservation and lotic ecology at the landscape scale (reviewed in Ward, 1998). Our understanding of the significance of connectivity between wetlands and the main channel in many floodplain river systems stems from this view (e.g. Walker et al., 1995). Kingsford’s (1999) use of aerial survey of temporal changes in waterbirds in floodplain wetlands is an outstanding case of a sampling method and an indicator that can be used at a landscape scale. Not only does this approach explicitly recognize the link between ‘wetland health’ and the health of its associated river, but waterbirds are good examples of ‘charismatic’ indicators that have public appeal and legislative relevance (Kingsford, 1999). However, further work is needed to establish whether waterbirds are a reliable measure of river health in systems other than large floodplain rivers (cf. Kingsford & Thomas, 1995). Perhaps they will prove to be ‘keystone species’ that are appropriate targets in themselves for ecosystem management (cf. Simberloff, 1998).

Choice of appropriate temporal scales also is vital for assessing river health (as well as all other ecological research). Often, the inherent natural variability of rivers means that long time scales are needed to identify trends or departures from ‘normality’ (cf. MacNally & Quinn, 1998). As hydrologists have often noted, data sets are seldom long enough, especially for rivers with highly variable flows (e.g. Puckridge et al., 1998), although Thoms et al. (1999) explore the use of palaeo-geomorphological indicators to enhance our understanding of longer term trends. This issue of temporal scale has been a prime concern of the Monitoring River Health Program in Australia (C. Humphrey, ERISS, personal communication) because of the need to isolate seasonal and other natural temporal trends in macroinvertebrate data from changes due to human impacts. However, long-term data sets on macroinvertebrates are rare in Australia.

Predictive model vs. multimetric approaches for assessing river health

Few ecologists rigidly adhere to a single-perspective approach for assessing river health. While they may focus on a particular taxonomic group, a number of other variables (e.g. water quality, flow regime, catchment characteristics) are usually measured, and subsequent data analysis involves relating data about the particular group to patterns in these other variables. Most commonly, a predictive model approach is adopted and this has several advantages.
(see Wright, 1995; Reynoldson et al., 1997). Critics of the predictive model approach confuse it with the use of multivariate statistics and imply there is excessive dependence upon statistics (e.g. Karr, 1999), but the methods provide simple outputs from a model (e.g. RIVPACS, Wright, 1995; AusRivAS, Simpson et al., 1997) that is easily used.

When data are analysed using a multimetric approach, choice of the metrics to be used is essential (Karr, 1999). In the same way that some multivariate analyses can be misleading if applied uncritically (Fore, Karr & Wisseman, 1996), the failure of some multimetric indices demonstrates inappropriate application. Multimetric approaches have proved successful in assessment of river health in a number of countries including Australia. Harris & Silveira (1999) have found that a modified version of the Index of Biotic Integrity (IBI) applied to fish communities is able to differentiate sites of different environmental condition across a diverse array of streams in southeastern Australia. The IBI has the added advantage that it does not rely on an ‘ecological standard’ – an undisturbed river – but is a relative measure (Harris & Silveira, 1999). This is useful because some types of undisturbed reference rivers (e.g. lowland rivers) are extremely rare (Thoms et al., 1999).

The choice and validation of appropriate metrics is a contentious issue that requires considerable effort (Karr & Chu, 1997; Reynoldson et al., 1997; Harris & Silveira, 1999; Karr, 1999) for each application. No widely available platform seems to have been developed for subsequent calculation of the metrics and the IBI, making their widespread use more difficult than it might otherwise be. In contrast, the RIVPACS (Wright, 1995) and AusRivAS (Simpson et al., 1997) software perform all necessary calculations and provide simplified outputs including bands of impairment. Some metrics (Average Score Per Taxon in RIVPACS and SIGNAL score in AusRivAS), that are probability weighted by the models to be site specific, are also calculated. The availability of AusRivAS via the Internet, with models that cover all of Australia, vastly enhances the utility of the approach. Both predictive models and multimetric approaches are designed to produce easily communicated outputs.

The conference did not deal extensively with the relative merits of the two analytical approaches (but see reviews by Norris, 1995; Fore et al., 1996; Reynoldson et al., 1997). However, it is clear that the analysis of the data is relevant to the interpretation of the results and assessment of river health. In deference to the societal component of the concept of river health, we must ensure that we communicate our results at the right level for full comprehension of their implications.

Conclusions

There is probably no ‘holy grail’ of indicators of river health common to all running waters but the search is not fruitless. If we accept that the concept of river health is useful for directing aspects of our ecological research (Meyer, 1997) and inspiring public interest in our efforts, we move on to selecting the best indicators (symptoms). Ideal indicator variables have features that include ease of measurement and relevance to river health, and can deliver early warning, or check compliance, or diagnose causes of poor health. Sampling to evaluate the indicators must be as stringent as any ecological study and it is in the choice, sampling and analyses of these variables that the concept of river health has its strong scientific grounding. As always, issues of temporal and spatial scale are crucial when collecting data and interpreting the results. Many river health issues are directly related to the connectivity between rivers and their catchments, floodplains and wetlands; therefore a landscape scale may often prove most suitable. Consequently, models should be broad-scale ones that are valid for use by river managers.

No single indicator alone is best and a synthetic approach that adopts a group of relevant metrics may prove most effective at measuring river health. These metrics may be physical, chemical and biological variables. Ecosystem processes such as carbon and nutrient dynamics or post-disturbance recovery trajectories may prove useful because patterns of species distributions alone contribute little to our understanding of how ecosystems function (Harris, 1994). The debates about sampling protocol and analytical approach (e.g. predictive model vs. multimetric) are useful and relevant to pure and applied stream ecology. But always the data must be of high quality and collected to answer the question, ‘Is this river healthy?’

It is easy to paint a grim prognosis for the health of the world’s rivers. But the increasing trend for stream ecologists to take an active part by applying their
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expertise to assessment of river health is encouraging. Effective communication with the general public, concerning the patient’s condition, is also improving and perhaps there is time to slow down and even halt the deterioration in health. To extrapolate the river health metaphor further, preventative medicine is always best, and the success of this approach relies on ecologists being able to describe the threats to river health in a sufficiently provocative way and recommend ways to keep the river healthy.

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