ENVIRONMENTAL IMPACT OF MINING

Ecosystem Protection

Temporal variability of macroinvertebrate communities in Australian streams: Implications for the prediction and detection of environmental change[1]

CL Humphrey

Introduction

An accompanying paper by Humphrey et al in this volume ('AUSRIVAS operator sample processing errors') describes research carried out under the 'Monitoring River Health Initiative' (MRHI), a scheme developed and funded under the Commonwealth of Australia's 'National River Health Program'. The MRHI has involved government agencies from Australian States and Territories in developing a standardised and coordinated rapid assessment approach to biological monitoring of water quality in Australian rivers and streams. The basis of the Australian approach, AUSRIVAS (AUStralian RIVer Assessment Scheme), is the development and use of models that would predict macroinvertebrate community composition in the absence of human disturbance (Davies 2000).

It was recognised early in the development of the MRHI that temporal variability in macroinvertebrate communities in a country of climatic extremes such as Australia, could pose serious risks to development of sensitive, predictive models for biological monitoring. The issue of temporal variability of macroinvertebrate communities and possible implications to successful model development were the subject of this study.

An assumption of predictive modelling is that macroinvertebrate community composition is reasonably constant over time. This is an issue that has received only limited consideration during development of the British RIVPACS system (eg Wright 1995) on which the Australian AUSRIVAS system was based. The present broad-scale study was conducted to quantify the degree of temporal variability evident in long-term data sets from representative streams across Australia. Where lack of 'persistence' was observed, the implications of the result for model sensitivity were to be explored by assessing the degree of fidelity of long-term data in groups derived from current AUSRIVAS classifications. Some possible ways to account for temporal variability are also discussed.

[1] More detailed discussion of this research is provided in Humphrey CL, Storey AW & L Thurtell 2000. AUSRIVAS: Operator sample processing errors and temporal variability — implications for model sensitivity. In *Assessing the biological quality of fresh waters. RIVPACS and other techniques*, eds JF Wright, DW Sutcliffe & MT Furse, Freshwater Biological Association, Ambleside, 143–163.

Methods

Measure 'persistence' of macroinvertebrate communities from long-term data sets

Data from a number of researchers across Australia were analysed to quantify the degree of temporal variability of stream macroinvertebrate communities (Humphrey et al 1997b). Ten geographical regions, 15 catchments and 38 individual sites were represented. The average duration of the data sets was approximately 6 years, with some data sets extending to 10 years. Sites were located in riffle habitat of permanent and seasonally-flowing streams.

Temporal variability was expressed in terms of an index of 'Inconstancy', determined for each site and season as the proportion of interannual comparisons of community composition (presence/absence) and structure (rank abundance) for which Bray-Curtis dissimilarity measures (family level identifications) exceeded pre-determined thresholds. As dissimilarity measures are the basis of UPGMA (Unweighted Pair Group arithMetic Averaging) classification of MRHI data for model development, these were potentially best suited to quantifying the degree of temporal variability in a data set.

Implications of lack of persistence for classification

Where lack of persistence of macroinvertebrate communities was observed in a long-term data set, an objective of the current project was to explore the implications of the results for predictive modelling by assessing the degree of temporal variability in reference sites relative to classifications of related impacted sites. Using long-term data from the upper South Alligator River (SAR) (Humphrey et al 1995a, 1997a), Humphrey et al (1995b) explored the implications of a marked switch in structure of macroinvertebrate communities (rank abundances) that occurred between pre-1993 and post-1992 time periods, by assessing whether the post-1992 data classified near or together with those from both unpolluted/mine-polluted portions of the adjacent Rockhole Mine Creek.

The ultimate test of whether or not temporal variability presents problems for predictive modelling lies in running long-term community compositional data for particular sites, such as those from the SAR, through agency classifications and models. In this context the severity of any lack of community persistence can be fully measured. Misclassifications and poor predictions would indicate potential problems for model development. To this end, the same long-term SAR data (but this time using presence–absence data) were incorporated into an NT MRHI agency classification based upon riffle samples gathered throughout the NT.

Evaluate ways to account for temporal variability and make recommendations

It became evident through the course of this study that seeking environmental correlates that may account for temporal variability would be unlikely to be successful for many of the data sets for which 'high' temporal variability was found. This is reviewed further below.

Results

Persistence of macroinvertebrate communities from long-term data sets

Degree and extent of temporal variability: rank abundance data

For half of the catchments studied, over 30% of interannual comparisons exceeded a Bray-Curtis dissimilarity value of 0.5 (table 1). Only for a relatively small portion of southern

Australia, for which interannual variability of discharge is low — Tasmania, south-west WA and possibly parts of Victoria — would there appear to be potential for development of AUSRIVAS models based upon rank abundance (community structure) data. Given this restriction and the fact that current AUSRIVAS models use presence-absense (compositional) data, the rest of the discussion focuses on results using presence–absence data.

Table 1 Temporal variability of stream macroinvertebrate communities from riffle habitat across different regions of Australia, based upon family-level, presence–absence (PA) or relative abundance (RA) data. Inconstancy indices are averaged across seasons.

Region	Flow status (Permanent or Seasonal)	Inconstancy index, PA (%dissim >0.35)	Inconstancy index, RA (%dissim >0.5)	CV of annual flow	Latitude (degrees & minutes)
Temperate (VIC-Latrobe)	P	4.5	68.0[1]	0.32	38.0
Temperate (SW WA)	P	6.5	3.5	0.49-0.73	32.3
Temperate (TAS)	P	7.5	5.5	0.47	41.3
Wet-dry tropical (SAR, NT)	P	13.5	40.5	0.58	13.35
Wet tropical (NE QLD)	P	15.0	33.5	0.5	18.1
Wet-dry tropical (RMC, NT)	S	17.0	0	0.58	13.35
Subtropical (SE QLD)	P	19.0	52.5	1.04-1.07	26.3
Temperate-dry (VIC-Wimmera)	S	19.5	41.5	0.58-0.98	36.3
Wet-dry tropical (Magela, NT)	S	21.0	36.0	0.56	12.4
Temperate (SW WA)	S	24.0	15.0	0.49-0.73	32.3
Temp. semi-arid (Flinders, SA)	P (riffle)	25.0	6.0	1.25	31.1
Sub-alpine (NSW)	P	27.5	37.5	0.5-0.75	36.3
Temp. semi-arid (Flinders, SA)	P (MH[2], pool)	51.7	16.5	1.25	31.1
Dry tropics (NW, WA)	S	93.0	27.0	1.4	21.3

[1] Data gathered using a sample processing method inappropriate for recovering relative abundance data; [2]MH = macrophyte habitat.

Degree and extent of temporal variability: presence–absence data

A combined-seasons index was derived by averaging the Inconstancy index across seasons for presence–absence data (table 1). Regression analysis was used to seek relationships between dependant Inconstancy index and independent environmental variables. The best predictive relationship was one derived between the Inconstancy index variable and the independent variables, Coefficient of Variation of annual flow and flow status (permanent/seasonally-flowing) ($R^2 = 0.77$).

Three summary points may be made from the results of Humphrey et al (1997b) and from regression analysis. (The term 'persistence', the converse of 'inconstancy', is used to describe the degree of similarity in community composition over time.)

1a A high negative correlation is observed between persistence and interannual variation of stream discharge.

1b Persistence of macroinvertebrate communities is significantly higher in streams of permanent flow than in streams of seasonal flow.

2 Measures of temporal variability used in the study indicated relatively high persistence of macroinvertebrate communities for all but one or two regions represented. For regions exhibiting high Inconstancy index values, cyclonic disturbance and flooding were attributed as the cause (Humphrey et al 1997b). Nevertheless, the 'high temporal variability' regions represent a large portion of the continent. In particular, Humphrey et al (1997b) extrapolated the findings to suggest that the sensitivity of AUSRIVAS models developed for much of the drought-prone portion of eastern Australia, particularly NSW and QLD, could be compromised during (and possibly after) drought.

Implications of lack of persistence for classification

From ordinations that were conducted using SAR data, post-1992/pre-1993, and data from both unpolluted/mine-polluted portions of the adjacent Rockhole Mine Creek (RMC), Humphrey et al (1995b) showed that the magnitude of change occurring in the SAR post-1992 was even more severe than that occurring in polluted portions of RMC. Moreover, the direction of change occurring in the SAR data was in the same direction as the pollution gradient in RMC.

The limitations of this approach to MRHI modelling, however, are twofold: Firstly, the analysis for SAR-RMC was based upon family-level abundance data. Had the analysis been repeated using presence–absence data, it would probably indicate little change in SAR community composition between post-1992 and pre-1993 relative to that between the two RMC sites. Secondly and as described above, the better test of whether or not temporal variability presents problems for predictive modelling lies in running long-term data for particular sites, such as those from the SAR, through agency classifications and models.

Humphrey and Doig (1997) describe results of a classification incorporating long-term SAR data into an NT MRHI agency classification. Results showed misclassification of early (1988) SAR data in a UPGMA classification based upon late Dry season 1994 and 1995 NT riffle data, while for successive years of data (1994 and 1995), about 50% of the 15 comparable NT sites occurred in different classification groups. However, because of the low interannual pairwise dissimilarity, low inter-site dissimilarity generally, and the fact that the classification was based on few sites (less than 25), no firm conclusions could be drawn from the study. Consequently, the full implications of any lack of temporal variability present in other long-term data from elsewhere for agency model development, accuracy and precision, will require similar approaches to that used for NT data.

Evaluate ways to account for temporal variability and make recommendations

Where 'significant' temporal variability is found after running long-term data for particular sites through agency classifications and models (viz misclassifications and poor predictions), possible approaches to dealing with this variability for predictive modelling include:

1 Risk-based assessment using AUSRIVAS models

Predictive regression relationships between temporal variability and environmental variables as described above, may be used to quantify degrees of 'risk' of model failure for a particular location. This would give managers some indication of how useful and accurate models might be that are developed for a particular location, ie what degree of error could be associated with predictions if temporal variability was the sole factor of concern.

2 Accounting for temporal variability

As an improvement upon (1), can temporal variability be accounted for?

(i) Modelling temporal variability

Humphrey et al (1997b) concluded that seeking environmental correlates that may account for temporal variability would be unlikely to be successful for a number of situations: (i) seasonally-flowing streams where shifts in community composition over time may be associated with stochastic recolonisation processes (see also Wright 1995); (ii) longer-term (several years) recovery and recolonisation of streams following massive disturbance (eg Robe R, north-west WA); and (iii) switches between different community 'steady states' where triggers for the switch may be clearly identified, but the trajectory of community composition thereafter is either lagged, or unknown and unpredictable (eg SAR and Yuccabine Ck, north-east QLD). Associated with these difficulties is the possibility of inter-catchment differences in community responses, as described for the South Alligator R and nearby Magela Ck (NT) in Humphrey and Doig (1997).

Modelling of drought-related changes to macroinvertebrate communities would be particularly useful for AUSRIVAS model development in eastern Australia. However, there is presently little understanding of the responses of macroinvertebrate communities to drought. Moreover, Humphrey et al (1997) reported very different responses to drought across Australia at regional and inter- and intra-catchment scales. Examination of existing agency data sets, some of which span periods of major drought (eg QLD, 1994–1995) would assist in redressing these information deficiencies.

(ii) Adjusting and updating model output

This would entail the re-sampling of suitable reference sites simultaneously with monitoring sites in order to adjust model output by some factor. A problem with this approach is that it assumes the 'correction' factor is similar across classification groups and between reference and disturbed sites. This assumption is unlikely to hold because, as described in (i) above, macroinvertebrate response to a similar disturbance may differ at different spatial scales. In addition, the degree of change to natural disturbance is likely to be greater for reference sites than for human-disturbed sites. Hence sufficient reference sites would have to be included that were representative of each of the classification groups, as well as re-sampling of selected disturbed sites, in order to derive appropriate scaling factors.

(iii) Models for different climatic conditions (especially drought vs non-drought)

Models empirically derived for different climatic conditions, such as drought vs non-drought, would have the advantage that fewer assumptions are made about the responses of macroinvertebrates in different habitats, between different parts of a catchment, amongst catchments, or across a disturbance gradient. However, this approach would be expensive and there is also the untested assumption that responses to one drought will be the same as the next, even though droughts differ in their intensity. At best, interpolation and extrapolation between different models may enable some allowance to be made for different climatic conditions. Nevertheless, some of the current agency data sets span a period of 'drought' and 'non-drought'; processing of all these data and derivation of different models for different climatic conditions may be valuable.

(iv) Combined-seasons/years models

Temporal variability can be reduced substantially where data for different seasons of the year or consecutive years for the same season are combined. One disadvantage with this approach is the need to accumulate two seasons/years of data before an assessment of water quality based upon macroinvertebrate communities can be made. This may provide some indication of longer-term severity of a water quality problem but it is contrary to the ethos of rapid biological assessment and rapid turn-around of results. Moreover, it may result in construction of a model so robust and overly-inured to natural environmental change that only impacts of a particularly severe nature are detected while impacts isolated to only one of the seasons may pass undetected.

Related to approaches (iii) and (iv), some agencies have constructed models by adding new reference sites gathered for a given season and from consecutive years of sampling, to an existing model (eg UK RIVPACS, MRHI ACT agency). Without simultaneous sampling of some common reference sites to account for possible temporal variation, this approach runs the risk of deriving models that are temporally confounded.

Some combination of approaches (ii) and (iii) may provide adequate solutions to developing AUSRIVAS models that account for temporal variability. For some geographical regions, temporal variability may be too large for useful predictive models to be developed. For these situations, it may be necessary to resort to more traditional hypothesis-testing approaches involving BACI designs and derivatives (eg Underwood 1991, Faith et al 1995).

Ongoing research

While some preliminary data simulations have been undertaken to determine the consequences to model development and sensitivity of temporal variability, a more complete sensitivity analysis is currently underway under Phase II of the National River Health Program to determine the full implications of variability (at various spatial scales) for model sensitivity. This analysis includes quantifying variation, as well as other error sources, and their effects on the rates of misclassification to quality bands (*sensu* Clarke et al 1996).

It would be prudent to be cautious in the promotion of AUSRIVAS for site-specific assessments until the sensitivity of the AUSRIVAS methodology has been fully assessed and data quantity increased and quality improved.

Acknowledgments

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An overview of the new water quality guidelines for Australian and New Zealand aquatic ecosystems[1]

CL Humphrey & KW McAlpine[2]

Background

The Australian and New Zealand Environment and Conservation Council (ANZECC) and the Agriculture and Resource Management Council of Australia and New Zealand (ARMCANZ) developed the National Water Quality Management Strategy (NWQMS) to provide a consistent framework for the sustainable use of water resources across Australia and to assist all levels of government and the community to manage these resources together (ANZECC & ARMCANZ 1994). A major focus of the NWQMS is to protect and maintain different (and sometimes competing) water resource values. The approaches for achieving these objectives are an important component of the *Australian Water Quality Guidelines for Fresh and Marine Waters* released in 1992 (ANZECC 1992).

In 1993, the ANZECC Standing Committee on Environmental Protection agreed that the Guidelines should be revised in due course to incorporate current scientific national and international information. Increasing scientific understanding of the complexity of ecosystems and food production systems has meant that new ways of managing water quality are replacing the more traditional scientific and management approaches. These more holistic, best practice approaches are designed to ensure that water resources are managed sustainably.

The Environmental Research Institute of the Supervising Scientist began the task of coordinating the revision of the Guidelines in 1996. The scope of the revision was extended so that the new Guidelines would also relate to water resources in New Zealand. The review process involved public consultation, establishment of technical groups to cover different subjects and phases of the revision process and ANZECC/ARMCANZ agency involvement to ensure the approaches and methodologies for the different sections were up-to-date, consistent and compatible with current policy at all levels of Government.

Philosophical basis

The new Guidelines have an expanded or new focus in important areas:

Holistic management. It is now well recognised that all aspects of the environment are interdependent and that influences on the environment can not be considered in isolation. For example, clearing of river bank vegetation combined with elevated nutrients from rural or urban catchments can give rise to algal blooms. The different environmental values (or uses) are also interdependent and must be considered together. Thus, types of ecosystems, food

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1 This paper is summarised from McAlpine K & Humphrey CL 2001. An overview of the guidelines for Australian and New Zealand aquatic ecosystems. *Australasian Journal of Ecotoxicology* 7, 41–50.

2 Department of Environmental Protection, PO Box K822, Perth WA, 6842.

production systems, interactions, cumulative effects and modifying factors must all be examined in studying water quality.

Cooperative best management. Environmental management in Australia and New Zealand is moving towards a 'cooperative best management' approach where the emphasis is on prevention, environmental outcomes and cooperation rather than control, prescriptive regulation and direction. This contemporary approach involves industry, Government and community working together to meet agreed environmental outcomes.

Water quality vs environmental quality. For many aquatic ecosystems, factors other than water (or sediment) quality may limit or prevent the achievement of management goals. It may be more appropriate to spend the majority of resources on programs such as habitat restoration or control of exotic pest species, ahead of water quality issues. The Guidelines should be integrated into, and implemented as part of, collective environmental guidelines (eg water quality, environmental flows and riparian condition).

Focus on managing issues. To protect designated environmental values the new Guidelines promote the use of management goals based on the environmental issues of concern, rather than the concentration of individual contaminants. By identifying the issues (eg algal blooms, deoxygenation, toxicity) the appropriate water quality indicators can be selected together with the environmental processes that can directly or indirectly affect the indicator (eg light, flow and algal grazing can affect chlorophyll *a* concentration). It is then possible to identify and take into account secondary, site-specific factors that can alter the effect of the threatening contaminant, or the severity of its effect. A guideline can then be applied according to the environmental/biological conditions prevailing at a specific site. This is an iterative risk-based process that leads to an improved estimate of the risk of an impact occurring. One of the key advances made in the revised Guidelines is the provision of risk-based decision frameworks to assist the user to make these site-specific assessments.

Continual improvement. Continual improvement is a fundamental principle embraced in water quality management strategies in Australia and New Zealand. Managers are encouraged to promote 'best practice' and improve the quality of water resources — using intermediate quality targets if necessary — rather than allowing them to degrade.

Integrated assessment. Currently, scientific understanding of the environment is insufficient to allow confident predictions about how a particular concentration of a contaminant will affect an ecosystem. Therefore, biological as well as physical and chemical aspects must be measured to confidently assess whether contaminants have significantly affected ecosystem health. This integrated approach also acknowledges assessment of other key environmental indicators apart from water quality (see above). Integration also needs to be across the whole catchment to ensure management considers cumulative impacts and impacts on environmental values in downstream ecosystems, such as estuaries or coastal waters.

Implementation

Water quality management involves the consistent, and preferably integrated, use of the range of legislative and regulatory tools at the national, state/territory and regional or catchment levels, as well as community action. However, society must have a collective vision of what it wants for each water resource and there needs to be a good technical understanding of human impacts and their control. After the available technical information for a specific water resource has been collated, the steps described below in the water quality management framework could provide a consistent national approach to managing water quality.

The management framework

The management framework is based on the premise that overall responsibility for water resource management rests with the community. The tools, strategies and policies developed to manage and protect environmental values are applied in this wider context. With this in mind, a water quality management framework has been developed for managing water quality consistently across Australia and New Zealand (fig 1). At each step in the framework the community, government and industry are encouraged to work together cooperatively so that management strategies can be developed and implemented effectively.

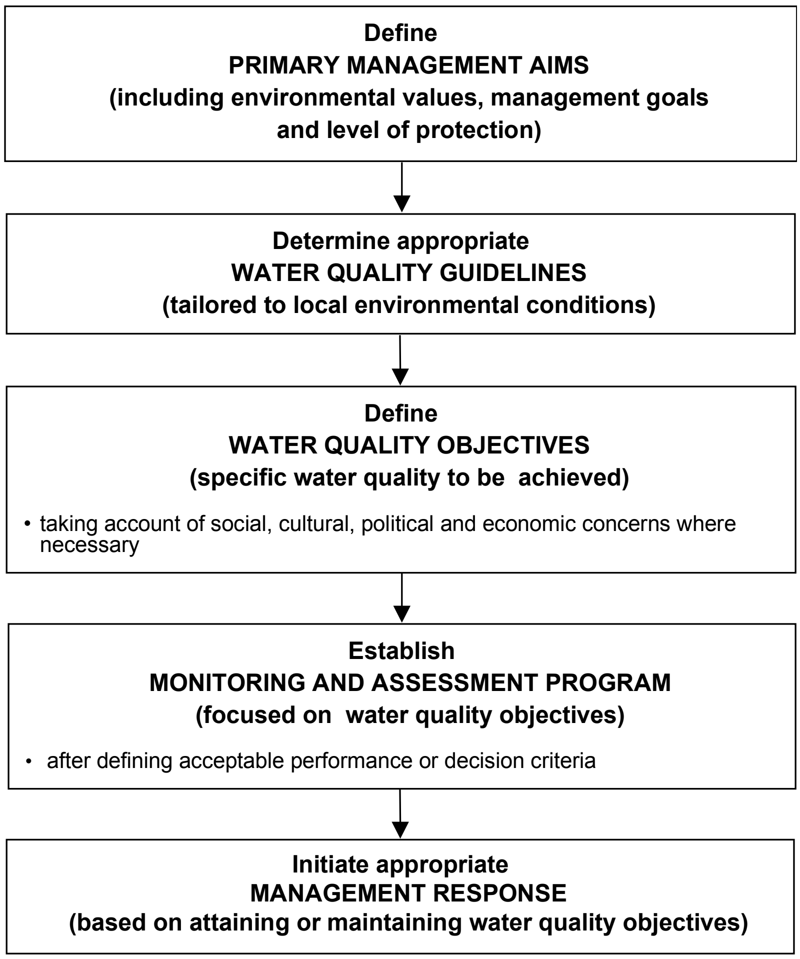

Figure 1 Management framework for applying the water quality guidelines (from ANZECC & ARMCANZ 2000a)

The first step is to identify the *environmental values* for a water resource through community involvement. Environmental values are particular values or uses of the environment that are important for a healthy ecosystem or for public benefit, welfare, safety or health and which require protection from the effects of pollution and waste discharges (ANZECC & ARMCANZ 2000a). Six environmental values are recognised in the Guidelines:

- aquatic ecosystems,
- primary industries (irrigation and general water uses, stock drinking water, aquaculture and human consumers of aquatic foods),
- recreation and aesthetics,

- drinking water,
- industrial water (no water quality guidelines provided), and
- cultural and spiritual values (no water quality guidelines provided).

In most cases more than one environmental value would apply to a water resource and managers would need to achieve the water quality of the most conservative of the values.

A primary focus for the new Water Quality Guidelines, and indeed the greatest interest and use of past guidelines, has been on the management of water resources to protect *aquatic ecosystems*. An outline of the important advances in the Guidelines for protection of aquatic ecosystems is the primary focus of the following discussion. McAlpine and Humphrey (2001) provide more detailed information about the Guidelines.[1]

For each environmental value *management goals* need to be formulated, in consultation with stakeholders, describing more precisely what is to be protected. Management goals need to be achievable, measurable and reflect community needs and desires. Examples could be to reduce the occurrence of algal blooms, or provide water quality safe for swimming.

A *water quality guideline* is a numerical concentration limit or descriptive statement recommended to support and maintain a designated environmental value. To protect aquatic ecosystems, guidelines are provided for four broad indicator types, ie:

1. *Biological indicators*. For example algae, macrophytes, macroinvertebrates and fish;
2. *Physical and chemical stressors.* These are natural water quality parameters: nutrients; biodegradable organic matter; dissolved oxygen; turbidity; suspended particulate matter; temperature; salinity; pH and changes in flow regime;
3. *Toxicants in water*. 'Toxicants', the term given to chemical contaminants such as metals, hydrocarbons and pesticides that can potentially have toxic effects at concentrations that might be encountered in the environment; and
4. *Sediment toxicants.* Sediments are a sink for many contaminants that can adversely affect benthic and other aquatic organisms.

A summary of the development, rationale and application of guidelines for each of these broad indicator types is provided in two special issues of the *Australasian Journal of Ecotoxicology*, volume 7.

For the physical and chemical (non-biological) indicators, the guidelines are termed 'trigger values' because if monitoring (or test site) data exceed the value, a management response is triggered — either conduct further site specific investigations to assess whether or not a problem really exists (using the risk-based decision frameworks) or instigate remedial action.

The preferred approach to deriving the guideline trigger values is using local effects-based data or, for some indicators and circumstances, using data from local reference sites. In the absence of these data, default trigger values are provided using regional reference data (physical and chemical stressors) or global biological effects (toxicological) databases (toxicants and sediments). In some situations, guideline trigger values may be modified to site-specific guidelines using the decision frameworks described later in this paper.

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[1] The web site for Australian water quality guidelines for fresh and marine waters is www.ea.gov.au/water/quality/nwqms.

Water quality objectives are the specific water quality targets agreed between stakeholders, or set by local jurisdictions, that are used to report on the performance of management strategies in meeting the management goals. They are based on water quality guidelines but may be modified by other inputs such as social, cultural economic or political constraints.

The established water quality objectives would generally be incorporated into water quality management plans, programs and strategies (including the use of regulatory instruments) that aim to protect the designated environmental values.

Monitoring and assessment programs are an essential component for measuring environmental performance and checking that the management goals are being achieved and the environmental values protected. The Water Quality Guidelines and their companion document, the *Australian Guidelines for Water Quality Monitoring and Reporting* (the Monitoring Guidelines) (ANZECC & ARMCANZ 2000b) contain information on the practicalities of designing monitoring programs and collecting and analysing data for the measurement of water quality. The Monitoring Guidelines contain a framework that sets out basic, general steps and details of how to plan a monitoring program, while the Water Quality Guidelines contain more detailed information specific to issues raised in its chapters.

New approaches to the protection of aquatic ecosystems

Although natural variations in the physical and chemical attributes of ecosystems can have important consequences for aquatic ecosystems, human-induced changes can be far more profound, and in many cases they can be effectively irreversible. The objective adopted for the protection of aquatic ecosystems is therefore: '*to maintain and enhance the 'ecological integrity' of freshwater and marine ecosystems, including biological diversity, relative abundance and ecological processes*' (ANZECC & ARMCANZ 2000a).

Depending on whether the ecosystem is non-degraded or has a history of degradation, the management focus can vary from simple maintenance of present water quality to improvement in water quality so that the condition of the ecosystem is more natural and ecological integrity is enhanced.

A more holistic set of environmental quality guidelines

Aquatic ecosystems cannot be considered as static environments — their biota, physical structure and chemistry all fluctuate according to seasonal and climatic influences and according to catchment vegetation and landuses. Water, sediment and biota are all in intimate contact with each other and the partitioning of chemical contaminants between these three media is under constant flux. Measurement of the biological components indicates whether in fact the ecosystem has been adversely affected by human activities and measurement of the physical and chemical indicators gives some insight into the cause of an observed change in the biota, or can be used to give early warning of potential impacts on the biota. To this end, the Guidelines provide guidelines for, and promote monitoring of, both biological and chemical components of surface waters and sediments.

Three levels of protection

It is unrealistic to impose uniform management goals, guidelines and regulatory frameworks across the spectrum of ecosystem conditions. To assist users, different sets of guidelines are recommended for each of three levels of protection, based on ecosystem condition:

- high conservation/ecological value systems;
- slightly to moderately disturbed systems (where the guidelines will mostly apply); and

- highly disturbed systems.

A management goal of 'no change' to biological diversity is recommended for systems of high conservation or ecological value or quality, and local biological effects and monitoring data are of overriding importance in guideline trigger value derivation and assessment of test data. Until such biological data are available, the recommended starting point for all indicators is generally no change beyond natural variability for this level of protection.

The default trigger values provided in the Guidelines are generally intended for use in slightly to moderately disturbed systems. If these values are unsuitable for highly disturbed systems, more relaxed (less protective) defaults are available for toxicants in water. Nevertheless, for both of these ecosystem conditions, local reference data (see following section) or preferably, local biological effects and monitoring data, will assume greater importance in trigger value derivation and test site assessment.

Accounting for variability between and within different ecosystem types

It is unrealistic to expect a uniform set of guidelines to apply across all ecosystem types and all regions and, therefore, effective management must incorporate site-specific information.

Firstly, it involves classifying the water resource according to *ecosystem type*. Guidelines have been tailored as far as possible to different broad ecosystem types.

Secondly, a greater emphasis has been placed on the use of reference sites to define a *reference condition* that can be used to provide the quality targets for management to achieve, as well as meaningful comparisons to use in monitoring and assessment programs.

Thirdly, an appropriate *level of protection* must be selected based on the target condition for the ecosystem.

Finally, *risk-based decision frameworks* are provided to help address the issues of ecosystem variability and complexity, giving a more realistic estimate of the ecological risk arising from contamination of the environment. They help the user to refine the measured test value so that it is appropriately compared with the conservative single-number guideline trigger value, for local application. In addition, the frameworks may be used to refine the trigger values themselves. In either case, they allow the user to take into account local environmental factors that may potentially affect the action of a particular contaminant, bringing the values closer to those that may elicit a biological response. For biological indicators, a different decision framework is incorporated since the measured biological responses are actually the management end-points in the decision trees for the physical and chemical indicators.

A generalised example of the risk-based decision tree is shown in figure 2. In most cases use of the decision trees will only be triggered once the level of a guideline trigger value has been exceeded. Through the decision frameworks, additional site-specific information is obtained on a step-by-step basis to modify the test site values and re-assess whether the guideline is exceeded or not. While some simple trigger value refinements can be achieved upfront, for most indicators and issues, trigger values are refined using the decision trees only after continuous and extensive monitoring shows that test site data exceedances are consistently assessed as posing no risk to the ecosystem. For whichever application the decision trees are used, the initial steps are relatively simple adjustments, but the ultimate steps are more resource intensive, usually requiring local toxicological data to be gathered.

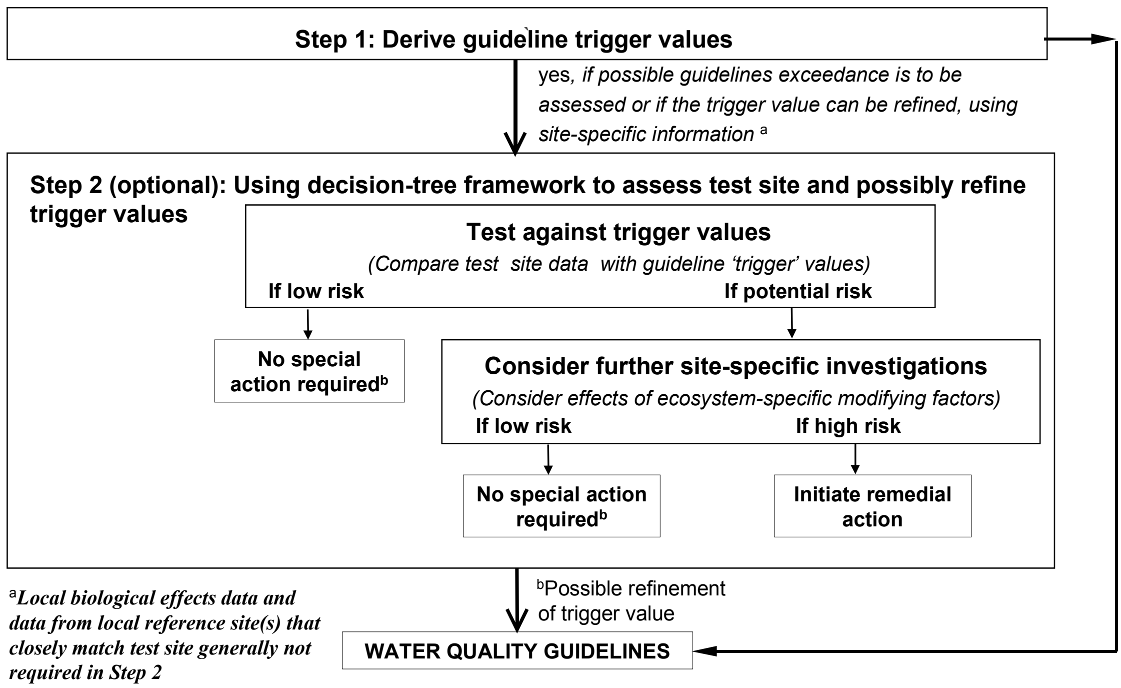

Figure 2 Use of decision trees for assessing test sites and refining trigger values, for physical and chemical indicators in water and sediment. Adapted from ANZECC and ARMCANZ (2000a).

Use of the decision frameworks is not mandatory, but they can reduce the amount of conservatism built into the guideline trigger values allowing more realistic targets to be achieved. If managers do not use the decision frameworks then their alternative option is to apply the trigger values as default guidelines. Obviously there is a cost associated with using the frameworks just as there is a potential (remedial) cost associated with not using them for cases where a test value exceeds the trigger value. All stakeholders will need to be involved in balancing the costs and the benefits of each option. Ultimately, the availability of data, expertise, resources and time will determine which, if any, steps in the framework are used.

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AUSRIVAS operator sample processing errors — implications for model sensitivity[1]

CL Humphrey, AW Storey[2] & L Thurtell[3]

Introduction

In 1993 the Commonwealth of Australia funded the 'National River Health Program' (NRHP) to monitor and assess the health of the nation's rivers and streams (Schofield & Davies 1996). Part of this program was the 'Monitoring River Health Initiative' (MRHI), involving government agencies from all Australian States and Territories to develop a standardised and coordinated rapid assessment approach to biological monitoring of water quality in Australian rivers and streams.

The River InVertebrate Prediction And Classification System (RIVPACS; Wright 1995) was adopted as a national framework for the Australian program. Models were to be based on family-level identifications of macroinvertebrates collected by habitat-specific kick-sweep sampling at an initial 1400 reference sites sampled across Australia in two seasons (Schofield & Davies 1996). A full description of the program as developed to 1997, AUSRIVAS (AUStralian RIVer Assessment Scheme), is provided by Davies (2000).

eriss was contracted by the NRHP to develop and implement programs for Quality Assurance/Quality Control (QA/QC) of key aspects of the MRHI protocol, including agency sample processing procedures. While AUSRIVAS has adopted a standardised approach for sample collection, agencies opted for one of two approaches for sample processing in Phase I of the program (1993–96): 30 min live-sort of each sample in the field (QLD, NSW, VIC, TAS & south-west WA) or field preservation of samples and subsequent laboratory-based subsampling and sorting (NT, SA, ACT and north-west WA). It was recognised early in the development of the MRHI that sample processing error due to live-sorting of samples in the field could pose serious risks to development of sensitive, predictive models for biological monitoring. To this end, the nature and degree of error arising in agency sample processing procedures was quantified and the implications of high error rates for model sensitivity explored by way of preliminary simulations. The results of this investigation are summarised in this paper.

Methods

QA/QC procedures for assessment of agency sample processing performance

Sample residues left after agency sample processing (field live sorting and laboratory subsampling and sorting) were selected at random and transported to ***eriss*** for external

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1 More detailed discussion of this research is provided in Humphrey CL, Storey AW & L Thurtell 2000. AUSRIVAS: Operator sample processing errors and temporal variability — implications for model sensitivity. In *Assessing the biological quality of fresh waters. RIVPACS and other techniques.* eds JF Wright, DW Sutcliffe & MT Furse, Freshwater Biological Association, Ambleside, 143–163.

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QA/QC processing. Processing entailed subsampling and sorting of residues and comparison of macroinvertebrate community composition and structure data present in an estimate of the 'whole sample' (live-sort + residue) with those present in the agency component. Agency data were assessed against the degree of departure in taxa number and community composition from whole sample estimates (WSE). Complete descriptions of this work are contained in Humphrey and Thurtell (1997).

Implications of errors associated with live sorting for modelling

As reported below, errors arising from live sorting were associated with biases in taxa recovery such that small and cryptic taxa commonly occurring in samples and across sites were often missed. A simulation study was undertaken to address the consequences of missed common taxa to model construction and performance. Sample processing errors were introduced into an agency UPGMA (Unweighted Pair Group arithMetic Averaging) classification and model that was relatively 'error-free' (ie derived from a laboratory subsampled and sorted data set).

Specifically, the taxa in an AUSRIVAS data set for which data on taxa commonly-occurring across sites were well represented (ie an ACT subsampled and sorted data set) were altered to match the bias observed in live-sort data. Two sets of live-sort data were used in the simulations: NSW, one of the poorer performing agencies, and for the average bias observed across eastern states, QLD, NSW, VIC and TAS. The average bias was not as severe as that for the single agency. Deletion of taxa was performed at random from actual occurrences in the original ACT data set, until the occurrences matched that of the bias represented in the two data sets. The deletions involved 16 out of a total of 39 taxa. For each of the single agency and average agency data, 10 separate simulations and classifications were run.

Results

Agency sample processing performance

Humphrey and Thurtell (1997) describe results of the external QA/QC audits. For agencies using a live-sort method for sample processing, two main sources of error were identified from assessment of 95 agency samples, ie (i) under-representation of taxa; and (ii) different taxa recovery rates depending upon the efficiency of the operator. Factors contributing to poor taxa recovery in live-sorted samples included (i) low live-sort sample size, (ii) operator inexperience (see figure 1), and (iii) taxa commonly occurring in samples and across sites being missed (Humphrey & Thurtell 1997).

For agencies using a lab subsampling and sorting method for sample processing, the main errors associated with the 40 samples examined were poor taxa recovery at low sample size, a consequence mainly of proportional subsampling (Humphrey & Thurtell 1997).

For MRHI, recovery of taxa that have a frequency of occurrence in a group of >50% is particularly important as these taxa are used in modelling and represent taxa expected at a site. To quantify the extent to which these key taxa were being missed from live-sort samples, taxa occurring in more than 50% of samples from any of the agencies, for either the live-sort or corresponding WSE component were listed. For each of these taxa and for each agency, the percentage occurrence amongst all samples for which the taxon was found in both LS and corresponding WSE components was recorded (figure 2).

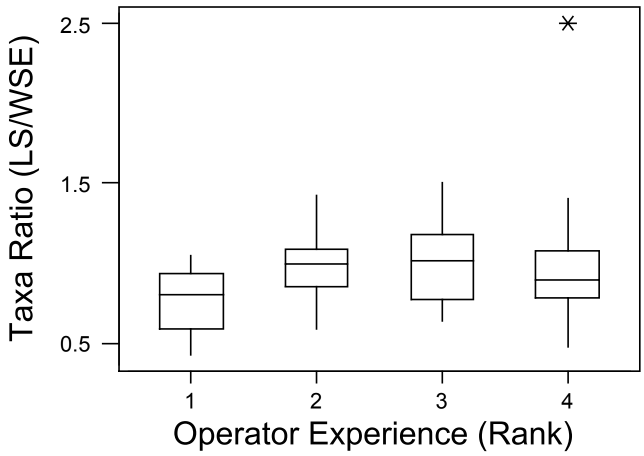

Figure 1 Boxplot showing comparison of LS-WSE (presence–absence data) for operators of different levels of live-sorting experience using the LS/WSE ratios of taxa number. Boxplot defines lower and upper quartiles divided at the median, with vertical lines showing the range of values that fall within 1.5 times the interquartile range. Outlier (point outside these limits) plotted with asterisk. Levels of operator experience for Ranks: 1 = < 2 years, 2= 2–3 years, 3 = 3–4 years, 4 = 5 years or more.

In figure 2, taxa have been ranked, from left to right, from greatest deficit to greatest surplus in occurrence in live-sort component compared to occurrence in corresponding WSE, when data were averaged across all live-sort agencies. Comparisons between occurrences of taxa present in the live-sort and corresponding WSE components show that similar taxa were either missed or better represented (in comparison to WSE occurrence) across all agencies and operators. Thus, there is consistency amongst all agencies in the biases in taxa recovery.

The results show that small and/or cryptic taxa are often overlooked during the live-sort process, regardless of agency. Thus, chironomid pupae and other small cryptic Diptera such as ceratopogonids and empidids were frequently missed during live-sorting (figure 2). This is also the case with the cryptic elmid larvae, hydroptilids (micro-caddis) and oligochaetes.

Figure 2 also demonstrates that some taxa are better represented in live-sort data than WSE (= laboratory subsampled and sorted) data. Thus the large but less abundant taxa, such as the odonates, shrimps and adult beetles, are often missed during the more objective subsampling process. These results show the extent to which laboratory subsampling and sorting are biased in taxa recovery. In practice, this is minimised for most MRHI agencies which process samples in the laboratory because an additional search of the entire sample for large taxa missed during the subsampling process is usually carried out.

Implications of errors associated with live sorting for modelling

The rate of bias against common taxa due to live sorting (single agency and average of 4 agencies) was superimposed upon an AUSRIVAS dataset in which common taxa were well represented (ACT laboratory subsample agency). The consequences of missed common taxa for UPGMA classification were examined.

In the original ACT classification, six clearly-defined groups were identified and a predictive model was successfully constructed. However, in the classifications derived after error rates were applied to the ACT data, there was little evidence of preservation of group structure. All classifications exhibited dilation and breakdown or 'chaining' in classification structure.

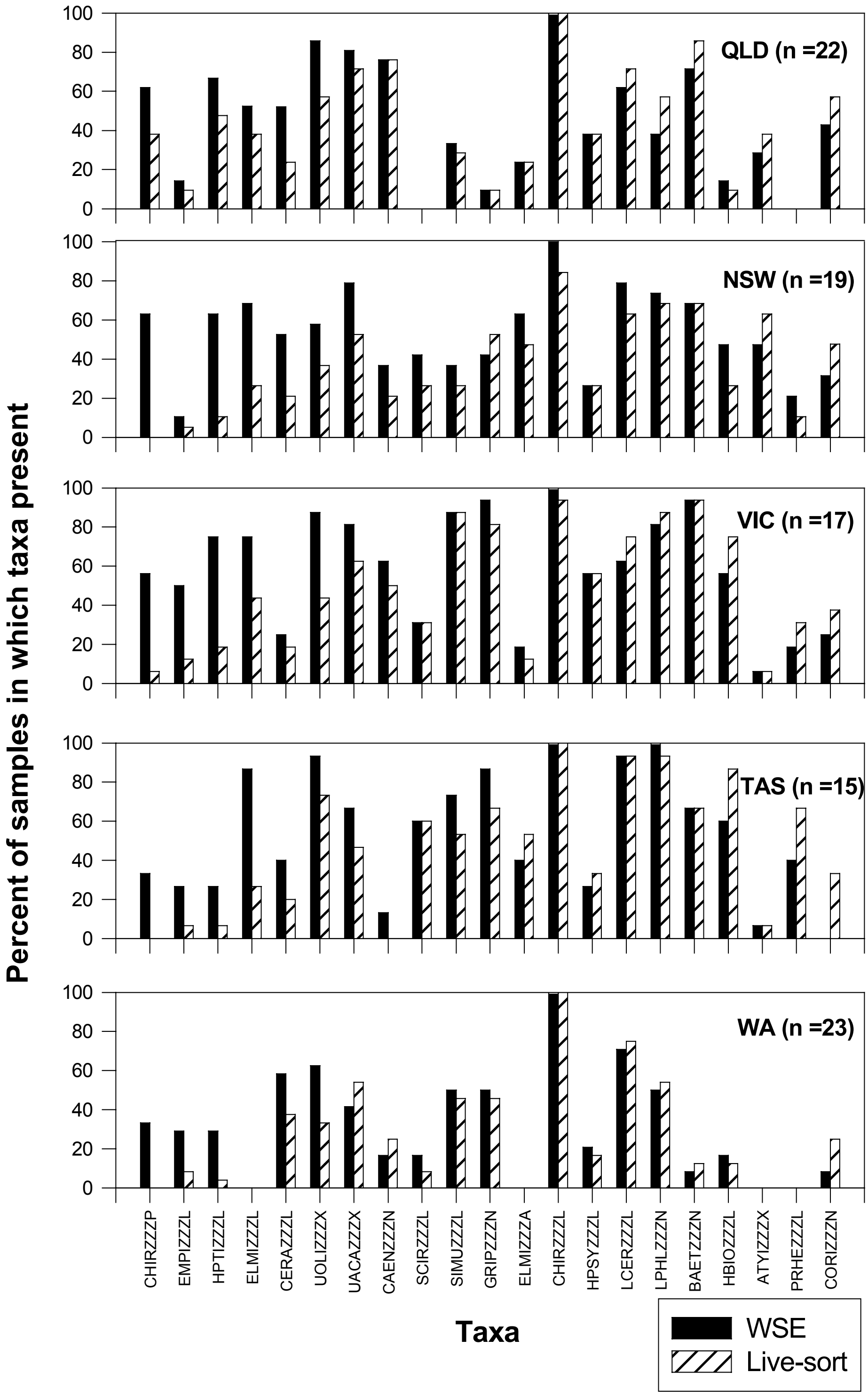

Figure 2 Taxa occurring commonly across samples for different MRHI agencies and their percentage occurrence (presence–absence data) in both LS and corresponding WSE components of agency samples. N = number of samples examined per state.

Key to taxon codes

ATYIZZZX, Atyidae; BAETZZZN, Baetidae; CAENZZZN, Caenidae; CERAZZZL, Ceratopogonidae; CHIRZZZL, Chironomidae; CHIRZZZP, Chironomidae; CORIZZZN, Corixidae; ELMIZZZL, Elmidae; ELMIZZZA, Elmidae; EMPIZZZL, Empididae; GRIPZZZN, Gripopterygidae; HBIOZZZL, Hydrobiosidae; HPSYZZZL, Hydropsychidae; HPTIZZZL, Hydroptilidae; LCERZZZL, Leptoceridae; LPHLZZZN, Leptophlebiidae; PRHEZZZL, Philorheithridae; SCIRZZZL, Scirtidae; SIMUZZZL, Simuliidae; UACAZZZX, Unidentified Acarina; UOLIZZZX, Unidentified Oligochaeta.

Suffixes L, N, X, A, P on codes refer to life stages Larvae, Nymph, Life stage not identified, Adult and Pupae respectively.

There was no evidence that classifications based on average agency (with lower error rates) were an improvement on single agency classifications (higher error rates). Even for one or two (single agency) classifications for which chaining in classification was least evident, there was a loss of two or three groups, whilst dissimilarity cut-offs for the groups were found to be higher in the altered data, indicating introduction of errors (ie higher mean within-group dissimilarities).

Placement of the simulated data site by site through an ACT model constructed using the original ACT data resulted in a high incidence of predictive failure (Humphrey et al 2000). When original data were run through the model, the majority of sites were recorded as Reference (Band A) — as expected. However, approximately 80% of sites from simulated data were recorded as impacted when run through the model, compared with 10% from the original data.

Part of the general failure to derive well-defined classifications in both sets of simulations may be related to the level of taxonomic resolution used for MRHI. Family-level presence–absence data may be so coarse that any structure present in the classifications may be easily lost. The effect of errors due to differences in sorting efficacy will depend to a large extent on the magnitude of the real differences in the data set being analysed; error will have a greater effect in a data set with small differences (short gradients). This is likely the case for the ACT, where group definition based upon family-level presence–absence data could be expected to be quite subtle because of minimal biogeographical signal. All other agency classifications are derived for the entire state or territory, minimising this problem.

Consequences of this study for ongoing AUSRIVAS programs

As a consequence of the findings reported above, aspects of the live-sort protocol were revised for implementation during the First National Assessment of River Health in Australia (1996–97). The 30 minute time limit for sorting was replaced by a target sample size of 200 animals or sorting to one hour, whichever was reached first, while agency staff were made aware of the taxa commonly missed in samples so that training programs could be implemented to redress deficiencies. In addition, poorly-sampled sites (identified by low sample size) were re-sampled to replace reference site data of dubious quality. Additional changes to the protocol will follow as the results of further R&D appear. For laboratory subsampling agencies, some recommended changes leading to standardisation of protocols have been made. These include (i) an emphasis on maximising taxa recovery (including 'large rares') through a coarse-screen search of the entire sample, and (ii) fixed-count subsampling.

Internal and external QA/QC must accompany all future sampling and sample processing by MRHI agencies so that unacceptable quality can be detected promptly to allow remedial action.

A more complete sensitivity analysis is currently underway under Phase II of the National River Health Program to determine the full implications of sampling and operator error for model sensitivity. This analysis includes quantifying error sources and their effects on the rates of misclassification to quality bands (*sensu* Clarke et al 1996).

Acknowledgments

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Chemical characteristics of stream waters in the Jabiluka region

C leGras, D Moliere & D Norton

Introduction

The physiography of the Jabiluka minesite embodies a significant paradox. This is that a world-class uranium (U) orebody is overlain by streams that contain the element at concentrations of only a few nanograms per litre, which is near the practical detection limit. Indeed, uranium concentrations ([U]) in Ngarradj Creek (Ngarradj) and its tributaries are in the bottom percentile of freshwater [U] worldwide (fig 1).

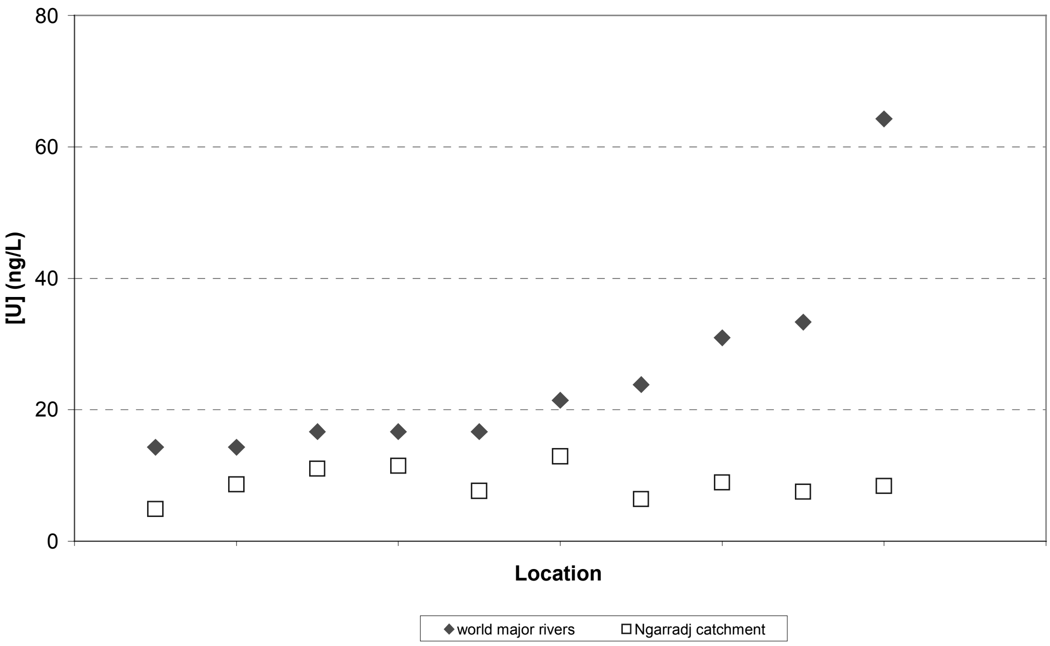


Figure 1 A comparison of mean [U] of various streams of the Ngarradj Creek catchment and the ten major world rivers with the lowest [U] (world data from Palmer & Edmond 1993)

The Jabiluka orebody also contains copper and lead values much greater than average crustal abundance (though below ore grade), but these metals also occur in stream water at concentrations that are frequently too low to measure.

The explanation for this situation lies in the local geology, where the orebody is hosted mostly by Lower Proterozoic graphitic schists, which are overlain unconformably by Middle Proterozoic sandstone and orthoquartzite (ERA 1996). The surface stratum is heavily leached, and even unweathered rock contains very low heavy metal contents (ERA 1996).

Therefore, the baseline and near-baseline data contained in this report provide a data set which will allow very small deviations from unaffected concentrations to be easily detected. This is an ideal position from a regulatory and monitoring viewpoint.

This report details critical indicator values in streams of the Ngarradj Creek catchment, and demonstrates the sensitivity of the sampling strategy that has been implemented. Two small streams in the immediate vicinity of the mine portal have been subjected to minor perturbations, probably with minor environmental consequences. Nevertheless, these perturbations are observed clearly and consistently, which increases confidence in the efficacy of the program. The main stream near the mine, Ngarradj Creek, shows no measurable effects from mining at present. This is despite the data being sufficiently sensitive and precise to demonstrate a high degree of intra-year and inter-year consistency for the indicators measured. This data set should therefore provide an adequate basis from which to observe small mine-related excursions from baseline, should they occur. For many indicators, notably U and sulphate (SO_4^{2-}), small spatial and temporal variations have been measured at concentrations that were below commercially accessible quantitation limits until recently.

A description of the sampling program and data set

The sampling approach

The sampling program commenced during the 1997–98 Wet season, and concluded at the end of the 2000–01 Wet season, yielding four years of data. Only data from the first three years are included in this report, except for qualitative reference to 2000–01 data where these are particularly relevant. A total of 31 sites have been sampled in Ngarradj Creek and its tributaries, and in a number of small creeks that flow westward from the escarpment outlier west of the minesite to the Oenpelli Road. These west-flowing stream sites constitute control samples. In addition, a further six sites were sampled in three adjacent catchments, also for comparison purposes.

A total of 21 physical and chemical indicators were measured at these sites, including general water parameters (pH, electrical conductivity-EC, alkalinity, organic carbon and turbidity), nutrients (total phosphorus and orthophosphate), major ions (chloride-Cl^-, SO_4^{2-}, magnesium-Mg and calcium-Ca) and heavy metals (aluminium-Al, cadmium-Cd, chromium-Cr, copper-Cu, iron-Fe, manganese-Mn, nickel-Ni, lead-Pb, U and zinc-Zn).

Most of these indicators are present in very low concentrations, at or near practical detection limits in some cases. These indicators are unable to provide meaningful spatial or temporal information, though the data are still useful as a basis for assessing mine-related deviations. Notable in this group is Pb, with a detection limit of 0.02 µg/L and with few measurements unequivocally above this. For this reason, Pb has not been discussed in this report. Many other indicators did not vary in a readily interpretable way, or else are not expected to be mine-related contaminants, and so have also been omitted from this report.

In the same way, the number of sites discussed has been restricted to those that would be most affected by mining activities, together with the corresponding control sites.

Mean values for important indicators

The broad overview of mean indicator values (table 1) shows that variation is relatively small throughout the whole suite. The main excursions are between the upstream and downstream sites of North and Central Tributaries. These small creeks define the northern and southern (respectively) boundaries of significant disturbance due to the mine. However, even though mine-related impacts can be inferred from these numbers, there exist substantial inter-year differences in the magnitude and temporal patterns of these indicator variations. These will be discussed in detail below.

Table 1 Mean indicator values at selected sites for the years 1997–98 to 1999–2000

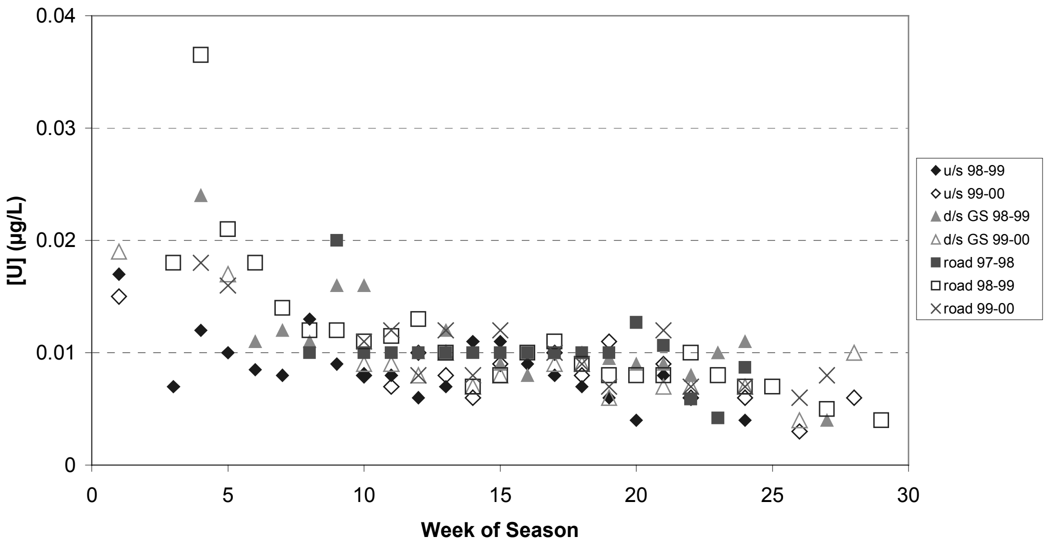
Site	GPS location	pH	EC uS/cm	Turb. NTU	[Mg] mg/L	$[SO_4^{2-}]$ mg/L	[Cu] µg/L	[Mn] µg/L	[U] µg/L
Ngarradj Creek upstream 1	132.931444 ºS 12.504000 ºE	4.74± 0.36	11.0± 2.1	1.1± 0.4	0.25± 0.07	0.32± 0.10	0.16± 0.09	3.2± 1.6	0.008± 0.003
Ngarradj Creek upstream 2	132.933940 ºS 12.503911 ºE	4.91± 0.22	11.4± 2.5	1.4± 2.1	0.24± 0.06	0.31± 0.20	0.18± 0.12	3.5± 1.6	0.008± 0.003
Ngarradj Creek downstream 1	132.921528 ºS 12.494194 ºE	5.37± 0.69	9.2± 4.3	6.2± 12.9	0.36± 0.09	0.23± 0.07	0.18± 0.06	3.5± 1.6	0.010± 0.002
Ngarradj Creek gauging station	132.922438 ºS 12.491447 ºE	5.33± 0.27	10.5± 3.0	2.1± 1.6	0.34± 0.12	0.24± 0.21	0.18± 0.11	3.4± 2.0	0.010± 0.004
Ngarradj Creek downstream 2	132.916667 ºS 12.484111 ºE	5.02± 0.58	13.0± 12.5	2.1± 0.8	0.39± 0.12	0.25± 0.13	0.18± 0.07	3.9± 2.3	0.011± 0.004
Ngarradj Creek (Oenpelli Road)	132.913628 ºS 12.467907 ºE	5.40± 0.26	11.2± 2.7	2.6± 2.2	0.36± 0.10	0.22± 0.14	0.20± 0.16	3.5± 1.3	0.011± 0.005
Ngarradj Creek west branch	132.927417 ºS 12.505722 ºE	5.83± 0.28	11.3± 7.8	5.9± 7.4	0.69± 0.49	0.16± 0.12	0.16± 0.07	3.2± 2.2	0.020± 0.011
East Tributary	132.932810 ºS 12.495093 ºE	4.97± 0.26	9.9± 3.1	1.3± 1.2	0.20± 0.06	0.22± 0.20	0.18± 0.15	2.6± 1.5	0.007± 0.002
Central Tributary causeway	132.915750 ºS 12.499653 ºE	6.05± 0.31	15.6± 4.0	2.0± 4.2	0.99± 0.27	0.08± 0.03	0.17± 0.17	3.5± 2.7	0.008± 0.004
Central Tributary downstream	132.911444 ºS 12.499361 ºE	6.10± 0.26	21.1± 6.3	3.0± 5.0	1.35± 0.50	0.11± 0.06	0.09± 0.09	6.2± 3.8	0.009± 0.007
North Tributary upstream	132.913712 ºS 12.498266 ºE	5.90± 0.28	8.5± 2.0	1.0± 0.6	0.47± 0.10	0.10± 0.05	0.16± 0.12	0.79± 0.40	0.007± 0.003
North Tributary downstream	132.915972 ºS 12.498556 ºE	6.12± 0.19	19.0± 15.6	4.3± 1.7	0.94± 0.78	0.27± 0.10	0.11± 0.12	2.8± 2.9	0.016± 0.011

Variation in indicator concentrations in Ngarradj Creek, North Tributary and Central Tributary

The physical and chemical character of Ngarradj Creek

Only small differences are evident between the six Ngarradj Creek sites sampled for the critical indicators Mg, Mn, U and SO_4^{2-}. The differences between the two sites upstream from the mine, and the four downstream sites are due mainly to the significantly different water chemistry of the West Branch of Ngarradj Creek. This is the largest tributary of the main channel and its confluence is between the two groups of sites. West Branch has higher [U] and [Mg], but lower $[SO_4^{2-}]$ than Ngarradj Creek. Mn concentrations are almost identical, hence little difference in [Mn] is observed between the groups of sites. East, Central and North Tributaries, though with measurably different water chemistry from Ngarradj Creek, are either not sufficiently different or have too small a discharge to make an observable difference. Intra-year differences are more important than inter-year variations for SO_4^{2-} and

U (fig 2), with a pronounced 'washoff' effect consistently observed through the years. Except for the ordinate value, the figures are almost superposable. There is no evidence for any mine-related influence on water chemistry at any site in Ngarradj Creek.



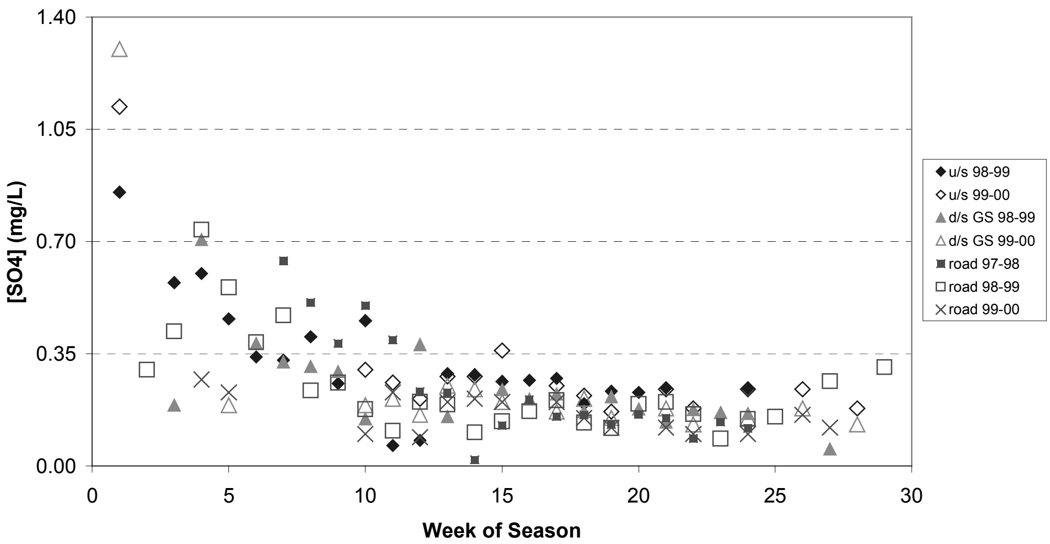


Figure 2 Spatial and temporal variation in [U] (top) and [SO_4^{2-}] (bottom) at three sites in Ngarradj Creek from 1998–2000

An interesting observation is the greatly divergent behaviour of U and Mn in Ngarradj Creek as a function of turbidity (fig 3). In this case, turbidity is used as a surrogate for discharge, for which detailed data are not available.

The best explanation for this dramatically different behaviour is that Mn is derived mainly from groundwater intrusion, and hence is present in higher concentration when hyporheic water forms a greater proportion of total discharge, that is, at low flow. Uranium, conversely is much more directly related to turbidity, which suggests that it is more closely associated with runoff. This interpretation accords with the expected redox behaviour of the elements, where Mn oxides should be readily reduced to Mn^{2+} in organic-rich shallow groundwater.

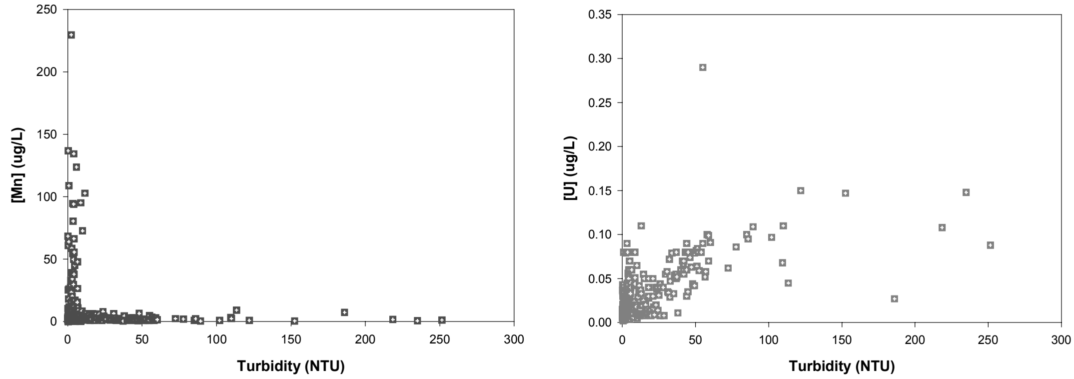


Figure 3 The relationship between [Mn] (left), [U] (right) and turbidity in Ngarradj Creek samples (all sites)

The physical and chemical character of Central Tributary

Two sites were sampled on Central Tributary. One site is immediately upstream from a vehicular causeway and is believed to be minimally affected by mine-related disturbance. The second site is about 500 m farther along, downstream from likely inputs from mine construction. Although there are no obvious sources of mine-related contaminants to Central Tributary, the water chemistry of the two sites is significantly different. This was not evident for the 1998–99 Wet season samples, the first after the commencement of mine workings. However, for the 1999–2000 samples, very evident differences were observed for [Mg], [Mn], [Ca] and pH, and to a lesser extent, [SO_4^{2-}]. Significantly, [U] does not change appreciably from year to year, averaging 0.008 µg/L at both sites during 1998–99, and being 0.009 and 0.010 µg/L respectively during 1999–2000. These trends have continued for the incomplete 2000–01 data set (not discussed further). The increase in pH suggests that the input source may be the dissolution of carbonate minerals (containing Ca, Mg and minor amounts of Mn). Increased SO_4^{2-} suggests that a small amount of sulphide mineralisation may have oxidised. The likely explanation is the partial weathering of waste rock used for construction purposes, although the mechanism of transport to Central Tributary is not immediately evident. Selected values are recorded in table 2. The generally higher values at the causeway during 1998–99 may be an artefact of the activity associated with construction.

Table 2 Selected indicator values for the causeway (upstream) and downstream sites on Central Tributary for the 1998–1999 and 1999–2000 Wet seasons

	[Ca] (mg/L)	[Mg] (mg/L)	[Mn] (µg/L)	[SO_4^{2-}] (mg/L)	pH
1998–1999	0.27 (causeway) 0.29 (d'stream)	1.1 (causeway) 1.1 (d'stream)	5.6 (causeway) 6.3 (d'stream)	0.08 (causeway) 0.08 (d'stream)	6.1 (causeway) 6.1 (d'stream)
1999–2000	0.04 (causeway) 0.47 (d'stream)	0.89 (causeway) 1.9 (d'stream)	1.8 (causeway) 6.0 (d'stream)	0.09 (causeway) 0.16 (d'stream)	5.7 (causeway) 6.0 (d'stream)

The physical and chemical character of North Tributary

In North Tributary, unlike Central Tributary, the reason for differences in indicator values between the upstream and downstream sites is evident. This is the large quantity of unmineralised orthoquartzite overburden that has been placed in the stream channel. The contents of target indicators in this rock are very low. This therefore allows a sensitive test of the ability of chemical testing to discern an impact on stream water quality from its presence.

A substantial difference was observed in the behaviour of common ions (Ca^{2+}, Mg^{2+} and SO_4^{2-}) and Mn as a group and U, as detailed in table 3.

Table 3 Selected indicator values for the upstream and downstream sites on North Tributary for the 1998–1999 and 1999–2000 Wet seasons

	[Ca] (mg/L)	[Mg] (mg/L)	[Mn] (µg/L)	[SO_4^{2-}] (mg/L)	[U] (µg/L
1998–1999	0.15 (upstream) 1.1 (d'stream)	0.48 (upstream) 1.3 (d'stream)	0.58 (upstream) 4.2 (d'stream)	0.11 (upstream) 0.31 (d'stream)	0.006 (upstream) 0.009 (d'stream)
1999–2000	0.04 (upstream) 0.25 (d'stream)	0.42 (upstream) 0.51 (d'stream)	0.74 (upstream) 1.0 (d'stream)	0.10 (upstream) 0.22 (d'stream)	0.006 (upstream) 0.024 (d'stream)

In 1998–1999 (the first Wet season after placement of the quartzite), electrolyte and Mn concentrations were much higher at the downstream site than in the succeeding year (1999–2000). This was particularly marked for Mn, where the concentration of this element progressively declined during the first year, as shown in figure 4, and did not return to previous, relatively high concentrations in 1999–2000. These observations suggest an initial washoff effect for this group of indicators.

Conversely, [U] is similar at both sites during 1998–99, but significantly higher at the downstream site in 1999–2000, as depicted in figure 5. This suggests that a period of initiation was necessary before measurable uranium values were released from the nominally unmineralised rock. The higher values are, however, only a factor of about five greater than the practical detection limit, and extremely low in world terms. These trends apparently continue in 2000–01, according to the incomplete data set for the current year.

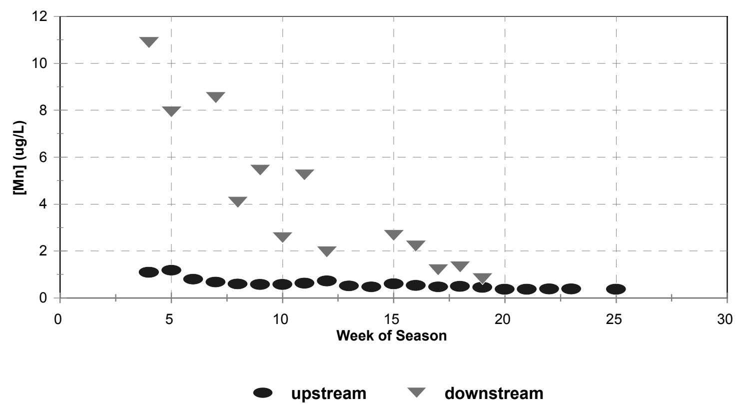


Figure 4 [Mn] in North Tributary during the 1998–1999 Wet season

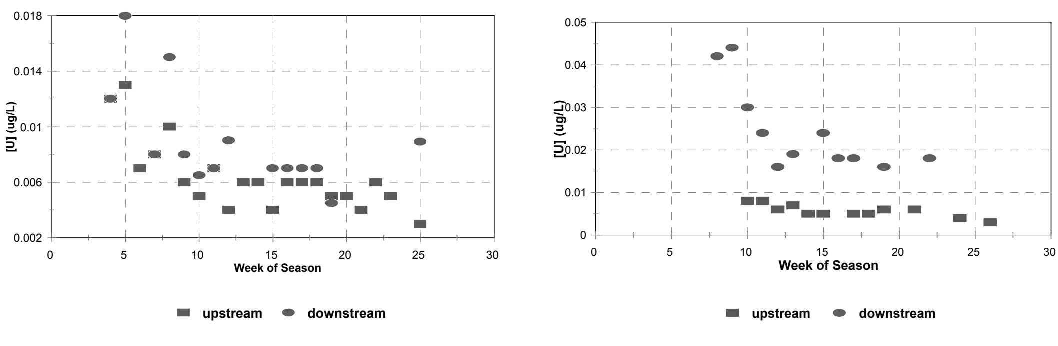


Figure 5 Comparison of [U] at the upstream and downstream sites of North Tributary in 1998–99 (left) and 1999–2000 (right)

Conclusions

The objective of the continuing project described here is to monitor a number of physical and chemical indicators in the streams draining the immediate vicinity of the Jabiluka mine. An original objective was to establish baseline indicator values at various sites. The main conclusion to date is that the concentrations of all measured indicators are very low, in some cases near or at the practical limits of detection. So far, the evidence from chemical monitoring suggest that the disturbance associated with mine construction has impacted minimally on proximate streams. Nevertheless, low-level impacts can be measured in creeks immediately downstream from sites of overt disturbance. These observations lend confidence to the expectation that physical and chemical monitoring will be able to detect significant changes in water quality that may be occasioned by further mine development.

References

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