8.1 Biological indicators

In this section, some elaboration upon the desired and essential attributes of generalised indicator types (Section 8.1.1) and the merits and potential of different taxonomic and functional groups for monitoring (Section 8.1.2) are provided. This is followed by a list of indicators and methods recommended for assessment of water quality in aquatic ecosystems of Australia and New Zealand (Section 8.1.3). Though not an exhaustive review of the state of knowledge of biological indicators in Australia and New Zealand, the material presented in this section is deemed sufficient to substantiate and justify the approaches described in Section 3.2.2 (Volume 1) for biological assessment of water quality. Much of the terminology for classes of indicator used in this section has been defined in Section 3.2.2 and hence the two sections should not be read in isolation of one another.

8.1.1 Broad classes of indicators and desired attributes

In Section 3.2.2.1, desired or essential attributes of generalised indicator types (or methods) required to meet the three identified biological assessment objectives were briefly described. Some elaboration upon these attributes for the different assessment objectives is provided below.

8.1.1.1 Broad-scale assessment of ecosystem 'health' (at catchment, regional or larger scales)

Desired or essential attributes of this indicator type include: (i) measured response is widely regarded as one adequately reflecting the ecological condition or integrity of a site, catchment or region (i.e. ecosystem surrogate, from 3.2.1.3/3); (ii) where community or assemblage data are gathered, these and associated environmental data are analysed using multivariate procedures; (iii) approaches to sampling and data analysis are highly standardised; (iv) response is measured rapidly, cheaply and with rapid turnaround of results; (v) results are readily understood by non-specialists; and (vi) response has some diagnostic value.

A multivariate system for analysis and assessment of rapid biological assessment (RBA) data with broad spatial coverage has been adopted for stream macroinvertebrate assessment in Australia as part of the Australian River Assessment Scheme (AUSRIVAS), rather than a single metric or multi-metric approach. Doubts were held about the general applicability in Australia of metrics in use overseas, as well as about their ecological relevance, the manner in which they have been analysed and reported and their interpretability (see also critique of Suter (1993)). Accordingly, a system based on regionally-relevant, multivariate reference site data was considered preferable in which:

- reference biological site groups are defined based on classifications derived from similarity matrices;
- relationships are developed between environmental variables and biological groupings;
- predictions are made of taxonomic composition at a new ('test') site derived from environmental variables;
- comparisons are made between predicted (expected) community composition and that found at the test site and reported as standard indices.

This is the basis of the multivariate approach in the UK-based RIVPACS (River Invertebrate Assessment Scheme, Wright 1995) which forms the central component of AUSRIVAS. Additional discussion of RBA approaches as applied to stream macroinvertebrate communities is provided in Resh and Jackson (1993), Lenat and Barbour (1994) and Resh et al. (1995).

8.1.1.2 Early detection of acute and chronic changes

An early warning indicator can be described as a measurable biological, physical or chemical response in relation to a particular stress, prior to significant adverse affects occurring on the system of interest. The underlying concept of early warning indicators is that effects can be detected, which are in effect, precursors to, or indicate the onset of, actual environmental impacts. Such 'early warning' then provides an opportunity to implement management decisions before serious environmental harm occurs.

Ideal attributes of early warning indicators have been extensively discussed elsewhere (Cairns & van der Schalie 1980, Cairns et al. 1993, McCormick & Cairns 1994), and have been summarised in a modified form by van Dam et al. (1998), as presented below.

To have potential as an early warning indicator, a particular response should be:

1 *anticipatory*: should occur at levels of organisation, either biological or physical, that provide an indication of degradation, or some form of adverse effect, before important 'serious' environmental harm has occurred;

2 *sensitive*: in detecting potential important impacts prior to them occurring, an early warning indicator should be sensitive to low levels, or early stages of exposure to the stressor;

3 *diagnostic*: should be sufficiently specific to a stressor, or group of stressors, to increase confidence in identifying the cause of an effect;

4 *broadly applicable*: alternatively, an early warning indicator should predict potential impacts from a broad range of stressors;

5 *correlated to actual environmental effects*: knowledge that continued exposure to the stressor, and hence continued manifestation of the response, would eventually lead to important environmental effects is important;

6 *timely and cost-effective*: should provide information quickly enough to initiate effective management action prior to important environmental impacts occurring, and be inexpensive to measure while providing the maximum amount of information per unit effort;

7 *regionally and socially relevant*: should be relevant to the ecosystem being assessed, and of obvious value to, and observable by stakeholders, or predictive of a measure that is;

8 *easy to measure*: should be able to be measured using a standard procedure with known reliability and low measurement error;

9 *constant in space and time*: should be capable of detecting small changes, and clearly distinguishing that a response is caused by some anthropogenic source, not by natural factors as part of the natural background (i.e. high signal : noise ratio);

10 *nondestructive*: measurement of the indicator should be non destructive to the ecosystem being assessed.

The importance of the above attributes cannot be over-emphasised, as any assessment of actual or potential environmental degradation will only be as effective as the indicators chosen to assess it (Cairns et al. 1993). However, it should be stressed that it is impossible for an early warning indicator to possess all the above attributes. In many cases some of them will conflict, or not be achievable. For example, a biochemical biomarker might provide an excellent indication of exposure to a particular pollutant, but might not be correlated to effects at higher levels of biological organisation (e.g. ecosystems). Moreover, the biomarker may not be applicable to other pollutants. Similarly, a long term monitoring program might provide excellent baseline data from which small perturbations will be obvious, but may be neither time- nor cost-efficient. Subsequently, not all the attributes will be achievable for each indicator. Therefore, decisions are required as to which attributes are more appropriate and achievable for a particular purpose, and indicators chosen based on those attributes. Particular attributes can be prioritised, and this is further discussed below.

In Section 3.2.1.3/2, both sub-lethal organism responses and rapid biological assessment (RBA) were promoted as the most useful and appropriate early detection indicators, while Section 3.2.2.1/2 listed the attributes that each of these broad indicator types were able to meet. Sub-lethal organism responses and RBA methods combine different needs of early detection indicators. Measurement of sub-lethal organism responses is best suited to timely detection of effects of particular substances at specific sites. In this case, important attributes 1–3, 6–8, and 9 from above may be met. RBA on the other hand, is appropriate for identifying problem locations occurring over large spatial scales and is able to meet attributes relevant to low-cost, broad application, responsiveness to a broad range of stressors and ecological, regional and social relevance, i.e. attributes 4–8 from above. In a balanced program in which both early detection and biodiversity indicators are measured, attributes 2, 6 and 9 from above are still regarded as the most important in governing selection of the former indicator type. Nevertheless, sublethal organism responses and RBA methods may play highly complementary roles when combined.

Finally, prior information on the type of chemical stressor entering, or potentially entering, an aquatic ecosystem is also of great use when selecting indicators for their assessment. Such knowledge of the potential/existing stressor, and its potential effects, can be utilised for the selection of indicators in order to maximise relevant information and minimise redundant information (Cairns et al. 1993). Appropriate early warning indicators can then be chosen based on this information, to form an adequate ‘suite’ of indicators, as part of an integrated early warning system.

8.1.1.3 Assessment of biodiversity

As noted in Section 3.2, it is insufficient for some management objectives to have detected change in an ‘early detection’ indicator because the management objectives are linked to detecting changes at the population, community or ecosystem level of organisation. Indicators used for this purpose are classified together under the catch-all term ‘biodiversity indicators’, and there are two uses for such indicators. The first is for detecting an ecosystem-level response to an impact, and the second is for assessing changes to biodiversity or conservation status. Often these two uses will overlap.

The indicator(s) selected for this form of monitoring and assessment should, therefore, be those that adequately reflect the ecological condition or integrity of the target area or site. There are three broad classes of indicators that are potentially useful for this objective: population parameters, community level measures and measures of ecosystem processes.

Which of these classes is chosen depends crucially on the management question being addressed, but there will be occasions when the indicator of primary interest is too difficult or costly to monitor. In such circumstances alternative, surrogate indicators will need to be measured instead. For example, management may be focused on the conservation status of a rare species of fish, but numbers of individuals may be too low to monitor the population reliably and quantitatively; therefore, the monitoring program should include some other indicator such as the structure of the aquatic vegetation that provides crucial habitat for the fish. Desirable attributes for an indicator of biodiversity, conservation status and/or ecosystem-level response are summarised in table 8.1.1, along with some examples and the classes of methods appropriate for each class of indicator.

Table 8.1.1 The broad classes of indicators and their desirable attributes for the purpose of the assessment of biodiversity, conservation status and/or ecosystem-level responses

Class of indicator	Desirable attributes for assessment of biodiversity/ conservation status/ ecosystem-level responses	Examples	Class of applicable methods
Population parameters	Population is the management target	Endangered or socio-economically important species	Quantitative[1] *OR* rapid assessment
	OR		
	Population is 'keystone species' or provides habitat for the target community	Coverage by a species of seagrass	
Community measures	Community is the management target	Structure of freshwater fish communities	Quantitative[1] *OR* rapid assessment
	OR		
	Community provides habitat or resources for rest of ecosystem	Vegetation structure in wetlands	
	OR		
	Community measure is a surrogate for (i.e. closely related to) diversity or ecosystem function	Macroinvertebrate community structure	
Measures of ecosystem processes	Process is essential to functioning of the system	Measures of gross primary production and respiration in streams	Quantitative[1]
	AND		
	Linkages to structural attributes well demonstrated		

1 A 'quantitative' method refers here to an indicator measurement program that permits rigorous and fair tests of the potential impacts under consideration; typically, conventional statistical tools would be employed to attach formal probability statements to the observations – see Section 3.2.3, Vol 1.

Population parameters are included for biodiversity and conservation assessment for two reasons. First, conservation status and ecological integrity may be linked to a particular species. The species may be rare or endangered, or it may be of socio-economic importance (e.g. recreational angling). Alternatively, the species itself may be structurally important in the ecosystem. For example, vascular aquatic plants and macroalgae form important components of the habitat for many other plants and animals in many freshwater and marine systems. Sometimes the predatory or competitive activities of a particular species of animal can mediate the coexistence of many other species. The advantages of using population parameters are that quantitative[1] methods are easily developed, conventional statistical

1 See table note to table 8.1.1.

procedures can be used easily and the results are readily explained to a lay audience. However, if the population measure is being used as a surrogate for a more complex indicator (e.g. biodiversity), then the linkage between this indicator and the more complex one needs to be firmly established.

A number of workers have presented practical and theoretical arguments as to why biological monitoring programs should include, as indicators, more or less discrete assemblages of organisms or ‘communities’ — as opposed to sole reliance on population studies of single species (e.g. Smith et al. 1988, Faith et al. 1991, 1995, Cairns et al. 1993, Warwick 1993, Humphrey et al. 1995). Community measures initially seem to be direct measures of biodiversity. However, it will rarely be possible to measure all of the species which make up the complete assemblage of organisms as a site (i.e. all the bacteria, fungi, algae, vascular plants, invertebrates, and vertebrates), so often a particular assemblage (e.g. benthic invertebrates) is measured as a surrogate for the entire community. Such an assumption, that particular assemblages are appropriate ecosystem surrogates, has not yet been verified in Australia.

There are two methodological issues that need to be considered when using community measures: taxonomic resolution and whether to use rapid biological assessment (RBA) methods. The issue of taxonomic resolution depends both on the management question and the resources available to carry out the monitoring. If the management objective is couched in terms of species diversity, then species-level identifications would be required. However, for some plant and animal assemblages such as algae and macroinvertebrates this is much more costly than sampling for, and identifying to, higher taxonomic levels (e.g. family or order), requiring additional resources and a considerable skill base. Consequently, programs focusing on species diversity *per se* for such assemblages (biodiversity, conservation status) will often necessarily be of narrower geographic scope compared with programs using coarser taxonomic resolution. A variety of proposals have been made for using coarser levels of identification for rapid biological assessment — where such assessment includes information on biodiversity or conservation status. The motivation for these proposals is that diversity at, say, the family level of identification, may be highly correlated with species diversity. This topic of the suitability of RBA methods for biodiversity assessment using family-level identifications has received attention in the UK (Wright et al. 1998) but is still being actively researched in Australia and New Zealand (see for example Vanderklift et al. 1996). Further comments on the choice of level of taxonomic resolution are provided in Lenat and Barbour (1994).

Allied with the decision about taxonomic resolution, is whether to use quantitative[2] or rapid assessment methods to collect and analyse the data. The first consideration should be the management question or assessment objective, examples of which are summarised in table 8.1.2. In general terms, rapid assessment methods are cheaper but less sensitive than quantitative methods (see Section 7.3.3.3 for summary of some factors contributing to reduced sensitivity of rapid assessment methods). Thus a question involving detecting changes in biodiversity around a site-specific, point-source discharge requires quantitative procedures employing inferentially strong designs (as described in Section 7.2). While quantitative methods would also be desirable for regional or broad scale assessment objectives, the costs of implementation are prohibitive; accordingly rapid assessment methods employing properly-researched protocols will be the more appropriate. Some consideration of these and other issues

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2 See table note to table 8.1.1

as they would affect the decision of whether to employ quantitative or rapid assessment (AUSRIVAS) community-based stream monitoring, is provided in table 8.1.2.

Ecosystem-level indicators may include direct measures of ecosystem processes such as gross primary production or community respiration. As such, these indicators may provide a direct and interpretable statement of ecosystem function or 'services', though as with all biological indicators, changes in such indicators can only be interpreted within the context of appropriate designs (e.g. spatial and temporal controls). For such monitoring and assessment, these measures lend themselves readily to quantitative procedures using conventional statistical methods. These indicators are at an early stage of development in Australia and New Zealand. An important consideration when using such indicators as summary measures of ecosystem status is whether underlying changes in community structure can occur without changing the size of the ecosystem indicator. For example, it may be possible for an impact to alter species diversity without changing the amount of gross primary production. Whether such structural changes 'matter' depends, again, on the management question.

Table 8.1.2 Possible relative merits of AUSRIVAS rapid biological assessment vs quantitative community-based stream monitoring

Management objective	Rapid biological assessment	Quantitative
1. Monitoring broad-scale impacts (e.g. diffuse-source effluents) or recovery (e.g. in remediation programs)	Suitable	Limited application (costly to extend sampling over broad geographical range)
2. Monitoring impacts at specific sites (e.g. point-source effluents)	Suitable only for detection of moderate to large impacts	Suitable
3. Reporting for State of Environment (or equivalent)	Suitable	Limited application (as for 1.)
4. Detection of impacts in regions/ habitats of naturally low diversity	Probably limited application	Suitable
5. Detection of impacts in regions/ habitats exhibiting high temporal variability of macroinvertebrate communities	Probably limited application unless an adequate range of reference sites are re-sampled to 'update' and 'correct' models	Suitable (providing control sites behave similarly to exposure sites in the absence of disturbance)
6. Detection of subtle impacts and/or monitoring of situations where potential costs of Type I & II errors are very high.	Probably limited application	Suitable
7. Provision of information about ecosystem-level responses and/or ecological importance of impact	Suitable (limited to moderate to large impacts)	Suitable (limited if little regional 'contextual' data available)
8. Provision of information about biodiversity and/or conservation status of sites	Suitable (if family-level, pres/abs data shown to usefully serve this purpose or species data are collected)	Suitable (assuming level of taxonomic resolution used serves this purpose and regional 'contextual' data available)
9. Provision of diagnostic information (to determine possible cause of impact)	Suitable (family-level)	Possibly more sensitive than RBA if samples identified to genus or species level *and* confamilials/ congenerics shown to differ in their responses to key water quality stressors

8.1.2 Evaluation of biological indicators for water and sediment quality

In Section 8.1.1, the attributes of broad indicator types used for each of the three assessment objectives were defined. Thereafter, specific biological indicators (viz taxonomic grouping, functional or trophic organisation and/or life habit) must be selected to meet these attributes. In general and for well-known classes of pollutants, the choice of indicator can be determined from precedence elsewhere for similar types of impact (e.g. through a literature search). Where specific indicators (such as species) are required or where there is no basis *a priori* for selecting particular indicator types, selection might be determined experimentally, for example, from hazard assessment using toxicity testing data, or alternatively, using an empirical 'top-down' approach in which ecosystems disturbed by similar types of impact are surveyed and note taken of elements of the biota missing as a consequence of the disturbance (e.g. Underwood 1991, Cairns et al. 1993).

The following sections, together with table 8.1.5, provide a rationale for the use of general and specific indicator groups, many of which are recommended in the Guidelines for monitoring and assessment of water quality in the various aquatic ecosystems of Australia and New Zealand. As far as possible, these groups of indicators are evaluated in relation to their potential for early detection of effects and measurement of biodiversity effects, of broad classes of stressors (e.g. metals, excess nutrients) in the water column and in sediment (see in particular table 8.1.5).

8.1.2.1 Streams, lakes and wetlands

Historically and on a worldwide basis, the development of biological indicators of water quality of freshwater ecosystems is more advanced than that of marine and estuarine ecosystems. This is mainly attributable to human dependence on reliable, secure and clean fresh waters, the greater vulnerability of freshwater ecosystems to disturbance (e.g. reduced capacity for both dilution of wastes and chemical buffering) and the greater anthropogenic disturbance of inland waters and their catchments, compared to marine ecosystems. Freshwater ecosystems are also generally easier to access and study, and hence for all the aforementioned reasons it would be expected that the development of standardised methods for biological monitoring in these ecosystems would be further advanced. Other factors responsible for the lag in progress in developing indicators of water quality for marine and estuarine systems are listed below (Section 8.1.2.2).

Not surprisingly, all plant and animal populations and communities for which viable protocols have been developed for Australian streams, lakes and wetlands — i.e. algae, macro-invertebrates and fishes — are those that are truly aquatic and hence at most risk from water-borne contaminants (see discussion on 'Other taxa' below).

Algae

Algae satisfy many of the criteria required of effective indicator taxa (Hellawell 1986) and since they occupy a fundamental role in food chains and ecosystem functioning they warrant serious consideration for inclusion in biomonitoring programs. They are particularly suitable for investigations involving organic and inorganic nutrients, and should, in theory, display changes far more readily and at an earlier stage of contamination than more popular invertebrate indicators.

Phytoplankton biomass is routinely used in biomonitoring of lakes (e.g. Davis et al. 1993), estuaries (e.g. John 1987) and slow moving or impounded rivers (e.g. Whitton & Kelly 1995) to assess the degree of eutrophication. In shallower rivers, measurements of the biomass of attached algae (periphyton), and their growth rates (increase in biomass on artificial substrata with time), are widely used to measure/monitor enrichment (e.g. Biggs 1990, 1995). The ratio of total organic matter (measured as ash-free dry mass) to autotrophic biomass (measured as chlorophyll *a*) in periphyton, called the Autotrophic Index (Collins & Weber 1978), is a useful measure of organic pollution impacts in streams (Biggs 1989, Lowe & Pan 1996). These techniques have been researched in New Zealand, but the necessary ecological research required to underpin development of monitoring programs in Australia has not been conducted or is not readily available, especially for seasonal and temporary running waters.

Biotic indices based on species abundances in algal communities have been extensively used for biomonitoring in the northern hemisphere. The first of these, the saprobic index (Kolkowitz & Marsson 1908), has been widely used in Europe to gauge the degree of organic pollution. Although, some countries still apply the saprobic index, its categorisation of waterbodies into broad zones of pollution is generally viewed as insensitive and hence indices which reflect gradients in pollution are preferred (Whitton & Kelly 1995). There are several biotic and diversity indices based on diatom communities. Many habitats are dominated by cosmopolitan algal taxa (particularly periphyton in streams) so use can be made of extensive autecological information in the literature for such habitats (e.g. Lowe 1974, Christie & Smol 1993, Kelly & Whitton 1995). Where a flora is dominated by local taxa, then it may be more appropriate to use diversity indices which do not rely on known habitat ranges for data interpretation. However, the responses of such indices need to be evaluated for local conditions because there are no ‘magic bullets’ amongst such indices which can be unequivocally related to pollution (Washington 1984). A common problem with many of these indices is that they have little physiological base to them, being derived instead on the basis of distribution (and hence correlation).

Predictive models offer a relatively new approach to water quality assessment. The development and application of these methods have been discussed at length in a number of papers (e.g. Wright 1995, Reynoldson et al. 1995) and have generally been applied to macroinvertebrates in rivers. This approach may also be applicable to algae and need not be restricted to flowing waters. A model for periphytic diatoms is being developed for streams and rivers in the United Kingdom (Whitton & Kelly 1995). Similar modelling approaches are also being trialed in Australia as part of the National River Health Program (Schofield & Davies 1996) though Research and Development to underpin this development is urgently required.

One of the major difficulties which arises when algae communities are used for biological assessment is that taxonomic keys are not readily available for local environments. This necessitates that monitoring outside of simple biomass measures will require skilled operators — at least for situations in which species or generic-level of identification is required.

Charles et al. (1994) suggested that palaeolimnological approaches should be incorporated into biological monitoring programs as a means of ascertaining whether a trend, for example, to poorer water quality, is due to anthropogenic causes or simply lies within the range of natural variation for that system. The simplest form of palaeolimnological monitoring involves the collection of a single core from the deepest part of a waterbody. Lake sediments contain a suite of indicators that could be used in a biological monitoring program. Amongst aquatic organisms, frustules of diatoms, calcite bivalve shells of Ostracoda and chitinous parts of the exoskeletons of Cladocera and Chironomidae preserve well in sedimentary profiles. Pollen grains and plant spores from the terrestrial environment are also suitable for this purpose.

Charles et al. (1994) suggest that historical sediment data are important in assessing surface water trends because they provide information on the nature of conditions existing before various forms of human disturbance occurred, as well as the nature and magnitude of natural variability and trends that were present before the monitoring program began. Eutrophication and acidification are two water quality issues where palaeolimnological studies are particularly useful. It must be noted that palaeolimnological studies cannot be undertaken in erosive environments and thus are not suited to riverine biomonitoring. Such studies, however, are applicable to estuaries, reservoirs and wetlands.

Three generic early detection-type protocols have been developed for streams and wetlands using algae; two of these protocols are also applicable to measurement of biodiversity responses (Vol. 1 table 3.2.2 and Section 8.1.3).

Macroinvertebrates

Benthic invertebrates cover a diverse assemblage of organisms that live on, or in, the solid substrates at the bottom of rivers, wetlands and lakes. Suitable habitat includes wood, aquatic plants, fine organic sediments and inorganic substrata such as sand, gravel and cobbles. These invertebrates cover a large range of sizes, but most are less than 2 cm in length and need to be identified with the aid of magnification. The major taxonomic groups are the insects, crustaceans, molluscs, flatworms and annelids. Most invertebrates are important components of ecosystems. They graze periphyton (and may prevent blooms in some areas), assist in the breakdown of organic matter and cycling of nutrients and, in turn, may become food for predators (e.g. fish).

Benthic invertebrates are the organisms most commonly used for biological monitoring of freshwater ecosystems worldwide, including Australia. This is because they are found in most habitats, they have generally limited mobility, they are quite easy to collect by way of well established sampling techniques and there is a diversity of forms that ensures a wide range of sensitivities to changes in both water quality (of virtually any nature) and habitats (Hellawell 1986, Abel 1989). Many forms live in sediment and because of this, the resident community, or representative taxa therein, are the most common choice in assessment of sediment toxicity. A number of invertebrate species also live for a sufficiently long time (e.g. molluscs and crustaceans) to be of value as bioaccumulating indicators. Information about an organism's preferred habitat and tolerances to certain types of pollution are used to interpret habitat quality and degree of water pollution. (The responses of different taxa to specific types of chemical contamination is reasonably well documented for Northern Hemispheric waters (e.g. Hellawell 1986). While this information is also available for large parts of Australia and New Zealand, there has been no useful synthesis and review of the scattered reports.)

Reflecting their popularity and inherent virtues for biological monitoring of water quality, macroinvertebrates have been selected as the key indicator group being developed for bioassessment of the health of Australia's streams and rivers under the National River Health Program (Schofield & Davies 1996). Active state/territory agency programs using the AUSRIVAS–RIVPACS type approach, underpinned by extensive Research and Development, are well established across the country. A large number of studies have been published on the use of benthic invertebrates in the assessment of water quality in Australia (e.g. see articles in special issue of *Australian Journal of Ecology*, Vol. 20, Issue 1 (1995)).

Analysis of invertebrate data is often assisted by calculating one or more measures or indices such as diversity indices or the percentage of the community composed of more pollution sensitive groups (e.g. % Ephemeroptera + Plecoptera + Trichoptera). Such measures have also been developed in some areas based on the known pollution tolerances of common taxa. For example, the Macroinvertebrate Community Index (MCI) (Stark 1985, 1993) has been developed in New Zealand and is now widely used by Regional Councils to detect and monitor water quality degradation. Similarly Chessman (1995) has developed the SIGNAL index (Stream Invertebrate Grade Number — Average Level) for invertebrates identified to family level in south-eastern Australia. Such biotic indices are based on the premise that pollution tolerance varies between species or higher taxa.

Two problems arise, however, in developing and applying pollution tolerance scores in Australia and New Zealand. Firstly, most tolerance information relates to organic pollution, although knowledge of tolerances to acidification and heavy metals is growing (see comments above); a further complication in Australia is that most information on pollution tolerances comes from the wetter and better-studied, temperate south east and south west portions of the country. Secondly, a number of groups of invertebrates are poorly known taxonomically at genus or species level; while many tolerance indices are developed for family or coarser-level identifications, it is acknowledged that for some groups the constituent taxa may vary widely in their tolerances.

Apart from analysis of the composition and structure of macroinvertebrate communities, functional feeding group measures are other community measures sometimes used for summarising community response to water quality. Functional groups reflect trophic levels (herbivores, detritivores and carnivores) and hence available food resources. Dominance of particular functional feeding groups at a site (e.g. scrapers, collector-filterers) is purported to indicate particular types of chemical contamination. Although the method is reasonably well established in North America (e.g. Resh & Jackson 1993), it is less commonly employed in Australia at least. Choy et al. (1997) and Choy and Marshall (1999) demonstrated changes in functional feeding groups — together with changes in community composition — in sections of streams affected by altered flow regimes (downstream of impoundments) in south-east Queensland. The approach depends on accurate assignment of taxa to feeding guilds, information for which is not yet comprehensively documented for Australian stream invertebrates.

Four generic biodiversity-type protocols and three early detection-type protocols have been developed for streams and wetlands using macroinvertebrate species or communities (Vol. 1 table 3.2.2 and Section 8.1.3).

Freshwater fish

Fish have considerable potential for use in bioassessment of water quality in some locations (Harris 1995). Australia has a freshwater fish fauna that is highly diverse in the northern part of the continent, but of low diversity in southern and inland regions (Bishop & Forbes 1991). In addition, much of the southern inland water fish fauna comprises exotic species, now known to dominate fish fauna in both abundance and biomass in many areas and to have had significant impacts on native fish populations (e.g. Lloyd & Walker 1986, Arthington & Bluhdorn 1995). Most notable among these are European carp, trout and mosquitofish (Wasson et al. 1996). The fish fauna also has a relatively high proportion of species with marine or estuarine migratory life stages. In New Zealand, there are strong idiosyncrasies in the distributions of fish species. Indeed, at any latitude about 75% of the native fish fauna is diadromous. Thus, use of natural freshwater fish communities for field bioassessment of water and habitat quality is not recommended in New Zealand (McDowall 1996).

Bioassessment using fish in freshwaters has been performed in several ways in Australia:

- assessments of change in abundance, population structure, recruitment or distribution of single species (e.g. Davies 1989, Davies et al. 1996);
- assessments of change in community composition (e.g. Humphrey et al. 1990, Harris 1995);
- assessments of physiological or biochemical changes in fish tissues (e.g. Ahokas et al. 1994);
- assessments of contaminant loads in fish tissues (e.g. Nowak 1990, Noller et al. 1993);
- toxicological assessments of ambient waters or effluents (e.g. Humphrey & Brown 1991).

Few of these methods have been used actively in assessment of water quality, or of human impacts due to changes in water quality, for management purposes. Examples where this has occurred include federally-funded bioassessment programs at Ranger uranium mine and Rum Jungle in the Northern Territory.

Fish populations and communities can respond actively to changes in water quality, but are also strongly influenced by changes in hydrology (affecting recruitment, habitat and food availability, e.g. Gerhke 1992) and physical habitat structure (such as snags and pools etc., e.g. Hortle 1983, Davies & Nelson 1994, and barriers to migration e.g. Harris 1985).

Current attempts to develop standardised bioassessment approaches using fish are in their infancy in Australia. The most notable examples include:

- the development of fish bioassessment using the AUSRIVAS–RIVPACS type approach (currently under development by the National River Health Program, Arthington pers. comm.) for Queensland streams;
- the use of the 'index of biological integrity' (IBI) approach (Karr et al. 1986, Miller et al. 1988) in a fish survey in NSW (Harris & Silveira 1997);
- the use of single species habitat-abundance models for blackfish and brown trout in Tasmania (Davies 1989).

These methods have not yet been sufficiently tested to determine their applicability at a broad scale. There are fundamental conceptual problems with the IBI approach (Suter 1993), exacerbated in any direct application to Australian systems by a lack of understanding of fish population dynamics and ecology. There are also problems with the use of simple relationships between ambient habitat data and fish abundance for fish species for which fundamental biological knowledge is lacking, restricting the application of the approach adopted by Davies (1989) to individual species. Davies (1989, 1992) also observed that, even for the well understood brown trout, interannual variation in recruitment was high within its established range, requiring quantification of relationships between hydrology and abundance at least at a catchment scale, in any bioassessment application.

In addition, approaches based on comparative measures of community composition are compromised in most of southern and inland Australia where species diversity is low, fluctuations in species abundance and occurrence are extreme (driven by unpredictable flow events), and the relative dominance of exotic species is high. Better understanding of population dynamics of fish species is required in these regions (most of the Australian continent!). Fundamental knowledge is required on fish larval ecology (e.g. Gerhke 1992), habitat requirements (e.g. Anderson & Morrison 1988, Humphries 1995) and community dynamics (e.g. Humphrey et al. 1990, Bishop et al. 1990, Pusey et al. 1995).

Finally, any bioassessment approach using fish species must take into account the scale at which fish populations and communities interact with the physico-chemical environment. Many riverine and wetland fish species are wholly or partially migratory, with migrations within or between water bodies, including many with marine life stages. Thus, populations may be 'driven' by factors operating at catchment or regional scales (such as flood flows, ENSO events etc.) and are often affected by natural and man-made barriers to fish passage (Harris 1985). This must be taken into account when assessing changes in fish populations in freshwaters, making the use of control sites with appropriate spacings and sizes mandatory. This also applies to the temporal scale of fish population and community dynamics, which is fundamentally different to the scale at which other aquatic biota (bacteria, macroinvertebrate etc.) respond to environmental factors and cues.

Bioassessment with fish is, however, practicable, using advanced quantitative designs as described in Section 7.2, Volume 1 (e.g. MBACI), provided the above issues are adequately addressed in the monitoring or assessment design (e.g. Boyden & Pidgeon 1994, Davies & Nelson 1994). Sampling methods are well established (including trapping, netting, electrofishing, poisoning, recapture after marking, counting of migrating fish etc.) for both running and still waters. Nevertheless, it will be some time before a credible, tested, standardised bioassessment approach using fish populations or communities is adopted at a national level or applied routinely for bioassessment of water quality impacts.

Guidance on the use of freshwater fish assemblages for measurement of biodiversity responses has been prepared and one early detection-type protocol developed using a freshwater fish species, for streams and wetlands of Australia (Vol. 1 table 3.2.2 and Section 8.1.3).

Other taxa

For microorganisms (other than algae and zooplankton), macrophytes, zooplankton, frogs and aquatic and semi-aquatic reptiles and waterbirds, very few viable protocols have been developed for their use as indicators of water quality in streams, rivers, wetlands and lakes of Australia and New Zealand. The potential of these other taxa for such a role in biological monitoring is briefly discussed below. Many of the generalisations have been drawn from Hellawell (1986) and Humphrey and Dostine (1994).

Bacteria, protozoa and fungi

Many microorganisms are decomposers and so have potential as indicators of early warning of ecosystem change. Bacteria are well studied as a result of their impacts on human health and so techniques for isolation and examination are well developed. Artificial substrates may provide a suitable means of obtaining fungal and protozoan samples that are standardised to some degree amongst sites. The flora and fauna that colonise artificial substrates are often unrepresentative of natural communities in the same water body, though if the study aims are focused on biological response and detection of change *per se*, this is not necessarily a disadvantage of the approach. Nevertheless, there are significant problems with the use of these groups for biological monitoring:

- generation times are rapid and so they may recover from episodic pollution events before a sample is taken;
- little is known of the responses of these groups to pollutants;
- there appears to be considerable variation in community structure over time and across microhabitats, confounding their use as indicators of pollution;

- taxonomic knowledge of these groups is poor hampering identification and interpretation of results;
- there is little ongoing research on most of these groups in Australia and New Zealand.

Macrophytes

Like algae, macrophytes are regarded as potentially useful indicators of water and/or sediment quality, but this is very much dependent on life form. Emergent monocots, for example, predominantly utilise nutrients in the sediments and so respond indirectly to water quality changes. On the other hand, many submersed species can take up nutrients via their leaves, as well as through their root systems, so their response is more immediate. The relative importance of either compartment, water or sediment, as a source of nutrients can vary even within a species, depending on which is the richest source.

Apart from nutrient enrichment, other water quality stressors that macrophytes may be particularly sensitive to include suspended solids (through smothering or light inhibition) and herbicides (Hellawell 1986) and, indirectly, acidification through associated changes in the carbon (CO_2) and nitrogen budgets of some freshwater ecosystems (Roelofs et al. 1984). Few forms are directly sensitive to metals.

Submergent and emergent plants can often absorb or adsorb high concentrations of metals without apparent toxicity and hence the group holds much promise as bioaccumulating monitors of these substances. For this reason, macrophytes can serve useful roles in wetland filtration systems, designed to scavenge metals from industrial effluent. Outridge and Noller (1991) concluded that rooted macrophytes were of little use for biomonitoring of sedimentary metals but that free-floating species could be potentially useful for biomonitoring of metals in water.

A major limitation on use of macrophytes as indicators is the lack of scientific knowledge about their population dynamics, and of the circumstances in which factors other than water quality affect their growth (J Roberts, pers. comm.). As with other taxa, an abrupt elimination of macrophytes from a given area is easily detectable but the causes are not so easily diagnosed. Careful interpretation and special, rather than routine, sampling may be needed for diagnosis, and factors other than water quality must be considered.

The potential for the development of macrophyte bioassessment procedures for Australian streams is currently being evaluated under the National River Health Program (Schofield & Davies 1996). There is other promising developmental work being conducted in Australia at present such as on the Hawkesbury-Nepean river system but at this stage few universally applicable protocols are available (J Roberts, pers. comm.).

One generic biodiversity-type protocol has been developed for wetlands using vegetation structure (Vol. 1 table 3.2.2 and Section 8.1.3).

Zooplankton

The zooplankton are noted to be very sensitive to a wide range of pollutants — as evidenced by the use of microcrustacea as standard test organisms in toxicity testing programs worldwide, including Australia (see articles in special issue of *Australian Journal of Ecology*, Vol. 20, Issue 1 (1995)).

The potential for use of zooplankton as biological indicators is best met in lentic waters and slow-flowing rivers and streams where they may occur in abundance. In fast-flowing streams and rivers, densities may either be greatly reduced (as the result of dilution), or animals may be absent from sites because flow velocities are too high for populations to establish.

Zooplankton populations are often highly dynamic and variable, a factor limiting the use of community structure in biological monitoring programs. However, taxonomic aggregation may reduce natural variability substantially without reducing sensitivity to impact (Frost et al. 1990).

Frogs

Frogs represent the highest form of life to lay naked eggs in fresh water. External fertilisation exhibited by frogs, moreover, exposes gametes of both sexes to ambient water. Thus gametes and fertilised eggs may be directly exposed to any form of contamination present in ambient waters. Toxicological studies have demonstrated the high sensitivity of frogs to a wide range of environmental insults and the semi-permeable nature of the skin places all life-stages at risk from uptake of contaminants present in the ambient environment. At exposure to low concentrations of contaminants, normal patterns of growth can be altered, so producing abnormalities of the limbs. In some cases, high incidence of abnormalities occurring amongst frogs is believed to indicate exposure during embryonic and/or larval stages. There have been several compilations of the thresholds of effects in frogs to toxic and teratogenic compounds (Tyler 1989, 1994, Harfenist et al. 1989).

Over the past 20 years considerable use has been made of frogs as bioindicators of environmental change though in Australia their use in biological monitoring of freshwater pollution has been limited. In Australia, the incidence of soft tissue and skeletal limb abnormalities in frogs arising from exposure to metals and radionuclides at mine sites has been reported by Tyler (1989, 1994) and Read and Tyler (1994).

Factors limiting the utility of frogs as indicators of water quality include:

- the semi-aquatic nature of the life cycle of most species (i.e. metamorphosed adults are not continuously exposed to any contaminants that may be present in a water);
- the highly seasonal and transient nature of the fully aquatic (larval) phase of the life cycle;
- the frequent opportunistic selection of breeding sites by adults. Lentic waters are more often preferred as breeding sites and recruitment may be poor wherever significant populations of fish predators are present;
- the mobility of adults and high inter-annual variability of breeding adults, spawn and recruitment (Freda 1991);
- the causes of global declines in populations of many frog species are not yet fully understood, complicating the interpretation of changes in populations as a consequence of specific water quality issues.

Aquatic and semi-aquatic reptiles and waterbirds

In general, the respiratory surfaces of animals have poor discrimination against chemicals compared with the gastrointestinal tract. It is for this reason that gill-breathing, aquatic organisms are at most risk from water-borne contaminants.

Direct poisoning of wildlife (and hence the need to instigate adequate monitoring programs in the possible event) through dietary uptake is well documented for waterfowl in the following circumstances:

i. Metal poisoning arising from ingestion of lead pellets. Considerable amounts of lead shot accumulate in the sediment of many wetlands both in Australia and worldwide as the result of hunting activities. Many waterfowl species actively turn over the soil/sediment to obtain plant material for food. Lead shot may be ingested either deliberately or incidentally with this food and associated sediment and actively retained thereafter in the gizzard as a grit supplement.

ii. Cyanide, involved in the processing of gold ore and which, if improperly stored and managed in mine tailings ponds, can present a serious hazard to waterbirds.

iii. Waterbirds in some wetlands (e.g. south-west WA) suffer high mortality from blue-green algal toxins (Balla 1994).

Otherwise, air-breathing animals linked to aquatic food chains are at risk from only a relatively small and specific suite of water-borne contaminants encountered in the diet; these substances include certain organic forms of metals (e.g. methyl mercury) and particular non-metallic organic compounds (e.g. some pesticides) that can biomagnify through food chains to levels of high toxicity. A discussion of such hazards that have bioaccumulating and biomagnifying potential is provided in Sections 3.4.2 and 8.3.5.7.

Populations of aquatic and semi-aquatic reptiles and waterbirds might be expected to be indirectly and adversely affected, in cases of extreme pollution, by possible extinction of aquatic organisms from their diet. For example, a drastic decrease in the availability of dietary calcium, due to the loss of such aquatic invertebrate taxa as molluscs and crustaceans in acidified environments, could lead to adverse effects on egg laying and eggshell integrity in waterbirds (Scheuhammer 1991). Obviously, monitoring of invertebrate organisms in this case would provide advance warning of potential effects on higher vertebrates.

The inclusion of waterbirds in a monitoring program may be fraught with problems of interpretation given that these animals can readily move amongst wetlands. Rather than population or community structure, breeding success might be a more suitable response to measure in monitoring programs.

Stream community metabolism

Community metabolism refers here to the biological movement of carbon and is an ecosystem attribute involving two processes, production (*via* photosynthesis) and respiration. Community metabolism is a process which is sensitive to small changes in water quality (particularly input of labile organic pollution and sedimentation) and riparian conditions, including light inputs. As such, it is a useful technique for impact assessment. Metabolism, as a measure of basic ecological processes, may enable early detection of an impact before it is manifest in changes in organism assemblages (e.g. macroinvertebrate community composition, see below).

Metabolism is best measured by monitoring oxygen concentration. In systems of high or rapid metabolism, whole-river measurements can be made using the two station (e.g. Odum 1956, Young & Huryn 1996) or single station technique over 24 hours (e.g. Bunn et al. 1997). In systems of low metabolism or high re-aeration due to turbulence (e.g. forested upland streams), closed system procedures are recommended (e.g. Davies 1994). These can be conducted over 24 hrs (e.g. Davies 1997) or over short time periods in full sunlight then without light (Hickey 1988). Usually a single habitat type is selected to measure stream metabolism. Measuring closed-system metabolism on cobbles can be used to maximise the amount of variation explained in catchment characteristics (Bunn et al. 1999).

The P/R (Gross Primary Production: Respiration) ratio is considered a key biological indicator of a system. Unimpacted forest stream sites are typically heterotrophic (e.g. P/R<1) and therefore are a net consumer of carbon. Davies (1997) showed that autotrophy or a P/R ratio >1 typically characterises an impacted site, indicating a fundamental shift in the energy base of the ecosystem. A shift from heterotrophy to autotrophy was indicative of catchment clearing and/or nutrient enrichment. The P/R ratio is, therefore, considered an index with ecological meaning.

Indicative of the sensitivity of the method, in streams of the nutrient-impoverished, northern Jarrah forest of south-west WA, a four-fold increase in nutrient levels resulted in a large increase in primary production that was not 'detected' by analysis of macroinvertebrate community composition (Davies 1997). (Macroinvertebrate data were derived from the AUSRIVAS method, using family-level, presence-absence data.) In these streams, measurements of metabolism in 'unmodified' sites were used to derive bandings (see table 8.1.3).

Table 8.1.3 A range of metabolism rates to signify different river health in streams of the Jarrah forest, south-west WA. This model is based on data from the winter of 1996. R_{24} refers to the daily respiration rate or total carbon consumed by an ecosystem over 24 hours (from Davies 1997).

Parameter	Unmodified range	Moderately-impacted	Degraded
P/R	<1	1–1.5	>1.5
GPP ($mgC.m^{-2}.d^{-1}$)	20–200	10–20 or 200–500	<10 or >500
R_{24} ($mgC.m^{-2}.d^{-1}$)	50–600	30–50 or 600–1000	<30 or >1000

Departures from these bandings could be used to confer a level of impact where typically, increased gross primary productivity (GPP) indicated nutrient enrichment and catchment clearing, and depressed GPP indicated sedimentation or degradation in water quality (see table 8.1.4). These results illustrate the potential for development of guidelines for stream metabolism — and hence early warning of impact arising from a number of key stressors — for other regions of Australia.

A protocol employing stream community metabolism, and applicable to both early detection and biodiversity measurement, has been developed for streams (Vol. 1 table 3.2.2 and Section 8.1.3).

Natural disturbances in streams

In fresh water systems, natural events such as floods and drought can disturb the biota in profound ways. The design of biological monitoring programs conducted in these ecosystem types needs to take such natural perturbations into account so that the effects of human-related impacts are not confounded in their interpretation by the effects of natural events. As a consequence of the temporary disturbance to the biota after floods and drought, it may be necessary to delay sampling for some period after these events. For example, sampling of macroinvertebrate communities in streams for AUSRIVAS is avoided until two weeks after major spates (Davies 1994).

Table 8.1.4 The categories of river health based on metabolism with possible causes for the departure in the metabolism from reference conditions. R_{24} refers to the daily respiration rate or total carbon consumed by an ecosystem over 24 hours. Values of GPP and R_{24} in mgC.m^{-2}.d^{-1} (from Davies 1997).

Unmodified	
GPP 20–200	Typical range of in-stream primary production
R_{24}	Typical range of in-stream respiration
P/R<1	Stream is a net consumer of carbon (heterotrophy)
Moderately impacted	
Parameter	*Possible cause*
GPP 200–500	Small increases in primary production, possibly due to small-scale catchment clearing and nutrient input
GPP 10–20	Secondary salinisation in an otherwise forested catchment. Low-level degradation in water quality
P/R 1–1.5	Small shift into autotrophy
Degraded	
Parameter	*Possible cause*
GPP<10	Substantial depression in primary production. Possibly the function of herbicides, sediment
GPP>500	Significant nutrient enrichment, catchment clearing; increased light inputs
R_{24} >1000	Nutrient enrichment (photo-respiration), sedimentation (bacterial respiration)
P/R>1.5	Fundamental shift to autotrophy. Algal material possibly remains ungrazed in channels causing secondary water quality degradation and problems with flood conveyance

8.1.2.2 Marine and estuarine systems

For marine and estuarine ecosystems, approaches to the development of biological indicators are not as well advanced as are those for freshwater ecosystems. While this is primarily because ecological understanding of the processes and the structure of marine and estuarine ecosystems is not as well advanced, other factors are also responsible for this lag in progress. Thus, in estuaries and coastal marine systems, spatial and temporal variability is complex and poorly understood. Further, the pressures on these systems have been more diffuse and less well defined, leading to a plethora of cause-effect problems few of which have been studied in detail. Although some specific local habitats have been well studied, such as temperate rocky intertidal systems, most are difficult and expensive to sample, are remote from research facilities, and have a vastly greater range of spatial scales than do freshwater ecosystems.

In marine and estuarine ecosystems, there is a large range of important structural and functional components. These include the water column flora and fauna, and its processes and dynamics, and the benthic system, including the sediments and rocky substrates, and the flora and fauna that lives in or on them. In shallow waters of estuaries and coastal continental shelves these two systems are often tightly coupled, but in deeper offshore waters they are decoupled. Selecting bioindicators from this large possible suite of structural and functional aspects is difficult in the absence of specific cause-effect understanding. In marine and estuarine ecosystems, there are no species or groups of species that can be universally identified as the central ecosystem component (the ‘keystone’ species) and so there is no simple way of choosing a representative taxon to use as a bioindicator. For example, benthic plants (algae and seagrasses) may have considerably narrower environmental tolerances than much of the fauna for some key pressures in Australia (nutrients, sediments) and are probably a good overall choice as a single class of bioindicator for these specific problems in shallow coastal waters and estuaries. Nonetheless, species that are dependent on plants may be also adversely affected because of an ecological linkage (such as sites for recruitment, food or shelter) and so a better (more cautious) approach for choosing bioindicators would be to include a number of taxa from the key components such as fish, plants, benthic infauna, and benthic epifauna; together these cover a number of the major biotic classes of relevance, and could be considered to contribute to a 'weight of evidence' approach. So, while it is plausible that overall classes of bioindicators could be developed in response to classes of marine and estuarine water quality issues, there are too few documented examples of successful case studies to permit an extensive number of generalisations about bioindicators to be assembled.

In this context of a highly variable environment, and one of large uncertainty about controlling processes and the nature of the biological diversity, most approaches to the development and use of bioindicators have focussed on local scales. It is here, at scales of a few km, where intensive sampling and analysis of changes in a range of bioindicators determined on-site, can be used to assess the effects of water quality issues. For example, to assess the effects of metal pollution on seagrass infauna in upper Spencer Gulf, SA, because the metal pollution did not appear to be taxonomically selective, bioindicators from a range of the locally occurring taxa have been suggested to detect and monitor the ecological effects of the metals (Ward & Hutchings 1996). However, these locally applicable bioindicators (3 species of polychaete, 2 molluscs and 2 crustaceans) would not necessarily apply in other circumstances. In other places the fauna may be different, and have different ecological functional roles, but most critically, the physico-chemical conditions may be very different from those in upper Spencer Gulf, and so metals may have a different range of interactions and effect on the fauna. So the particular taxa suggested as bioindicators in upper Spencer Gulf for metal pollution may not exist or may not be sensitive to metals in other places.

The local approach to establishing bioindicators has also been adopted to deal with discharge of treated sewage, or the inadvertent overflows of sewerage systems, to estuaries and coastal waters. A combination of monitoring of locally abundant taxa (offshore discharges) and the deployment of artificial settlement panels and monitoring of rocky shores has been developed to monitor the effects of sewage and sewerage overflows in Sydney (Sydney Water 1995).

The difficulties of developing robust and broadly applicable bioindicators for marine and estuarine ecosystems are well acknowledged. A recent review of this area by the UN Joint Group of Experts on the Scientific Aspects of Marine Environmental Protection (GESAMP) found that, rather than defining specific bioindicators, all that could be recommended was a process by which such indicators could be defined for each particular problem (GESAMP 1995). The types of bioindicators considered to be useful, depending on the problem at hand, covering both detection of exposure and effects, included biomarkers, histopathology, physiology, and ecological variables. Each of these were considered to be appropriate for use in specified problems, and preferably in a tiered approach to assessing environmental quality. Even for eutrophication issues, where the primary effects are likely to be detected in plants, there are few generalisations that can be applied to marine and estuarine ecosystems. For eutrophication problems, while plants may be severely affected, the most useful bioindicators for early warning of effects may be fauna, if they are more sensitive and more readily monitored than are plants, or the nitrogen cycling capacity of soft sediment systems which, in places, may be the key factor controlling denitrification in shallow marine systems and thus the process controlling progression towards eutrophication.

The following protocols have been developed for marine and estuarine systems of Australia.

Biomarkers

Biomarkers have been equally applied worldwide in both freshwater and marine ecosystems. At this stage in Australia, they have been developed only for estuarine and marine systems.

Molecular biomarkers are characteristic signatures of pollution expressed in enzymes, cell constituents, or metabolism products within organs of animals and plants. Organisms respond to stress by invoking molecular responses, and these can be then expressed as physiological or other changes. The molecular responses to pollution stress are likely to be the earliest form of organism response, and potentially should be capable of being used as an early warning indicator of changes induced by pollution. Of course, changes at the molecular level in an organism may not necessarily reduce its ecological fitness — its ability to function normally — because of various compensatory factors. Nonetheless, molecular responses to various pollutants have been sought in a very broad range of species, and many site and host-specific responses have been detected in biomarkers based on the activity of specific enzymes in liver, kidney and blood. This field of science is rapidly expanding, and the new scientific journal *Biomarkers* was created in 1996 in response. The most intensively studied group of organisms is fish, both freshwater and marine. Here, a number of studies have examined the utility of sub-cellular biomarkers to respond effectively to, and generally indicate, the effects of a number of pollutants. Their most promising use is as screening tools for detection of pollution by the expression of unusual patterns in a suite of biomarkers (Adams et al. 1989, Viarengo et al. 1997, Gunther et al. 1997). In Australia, biomarkers in flathead have been used to detect pollution (Holdway et al. 1994, 1995).

Frequency of algal blooms

Algal blooms are undesirably high densities of naturally-occurring algae. They may comprise micro or macro-algal species, and they may be toxic or non-toxic to humans or livestock. Large, persistent or suddenly collapsing algal blooms can have undesirable environmental consequences, even if they are non-toxic to humans, because they produce large amounts of biomass that eventually (directly or indirectly) dies, and proceeds to decay consuming oxygen and releasing large amounts of waste products. This process of decay may lead to extended periods of deoxygenation in bottom waters, and to the elimination of all but the most robust of benthic organisms, and affect for long periods the species composition of the benthic fauna. It also may inhibit the passage of mobile species like fish and prawns, and affect their larvae. Decaying algal blooms often emit noxious and offensive odours, affecting waterfront properties and the local recreational amenity. Some species of algae are toxic, or produce toxic materials, and can affect humans and wildlife.

In the natural (unmodified) ecosystems of coastal estuaries, bays and near-shore waters the biomass of both macro and micro-algae fluctuate substantially, typically related to seasonal factors, like light availability, temperature, nutrients, river runoff, weather conditions, stratification, or ocean currents. Around Australia, in the open ocean, and in some near-shore regions, blooms of *Trichodesmium* (a microscopic single-celled alga) occur regularly in spring and summer, and these are likely to be natural occurrences, even though they may affect beaches and coastal islands in tropical and subtropical areas. However, increasingly, in many coastal waters and estuaries, large algal blooms are considered to be the result of pollution of waterways from both point and non-point sources, with nitrogen, phosphorus, and other trace elements needed for plant growth. Because of the broad range of human actions that can lead to input of nutrients to estuaries and coastal waters, and the difficulty of measuring and controlling them, and because algal blooms are an integrated biological response to various forms of nutrient input, the frequency and intensity of algal blooms can be used as a measure of the quality of a water body. Algal blooms are particularly useful as an indicator in estuaries or coastal lagoons that are suspected of being influenced by nutrients derived from urban, agricultural or industrial sources. Lack of a bloom does not mean that there is no nutrient pollution, and like all natural systems, natural levels of algal growth vary from place to place. However, detecting and identifying blooms and the factors that control them is a complex process, but is of critical importance given the high value placed on the resources and biodiversity of Australia’s coastal ecosystems (McComb 1995).

Seagrass depth distribution

Seagrasses are aquatic angiosperms: they are flowering plants that spend most, or all, of their life submerged in marine or brackish waters. Australia has 30 species of seagrasses, the largest number of seagrass species in the world, widely distributed in both tropical and temperate coastal waters. While some species can grow in very low light conditions, light is a central limiting factor for the deep-water distribution of all subtidal species. Where other conditions (like sediment type) are suitable, seagrasses can grow only to a depth of water where there is sufficient light. If light is reduced, then the depth at which a species can grow is also reduced. Available light for seagrass growth may be influenced by sediment particles in the water column, by colour from natural or industrial processes, by high concentrations of plankton, and by the growth of fouling algae on the seagrass leaves. These may, in turn, be related to various land-based sources of sediments and nutrients. This means that for seagrasses, as for freshwater angiosperms, the zone of light-availability, given the prevailing water quality, can be measured to assess the potential for broad-scale depth distribution of seagrasses because the plants act as a time-integrated sensor of light availability (Chambers & Kalff 1985, Duarte 1991, Abal & Dennison 1996, WADEP 1996). The converse is also true in many situations: the depth-distribution of seagrasses is a useful integrated indicator for long term water quality (light) conditions (Giesen et al. 1990, Abal & Dennison 1996). The depth distribution of seagrasses is an important water quality indicator because it can integrate changes in aquatic light climate caused by various factors, and because seagrasses themselves are important and highly-valued elements of marine and estuarine environments.

Imposex in marine gastropods

Imposex is the term given to the development of male genitalia, or other form of physical abnormality, in female marine gastropod molluscs. Although the presence of a penis in female gastropod molluscs is thought to be a naturally occurring abnormality, it usually has a very low incidence of occurrence (Blaber 1970). Increased frequencies of imposex are caused by organotin compounds, particularly varieties of butyl and phenyl tins used in antifouling compounds (Bryan et al. 1986). Imposex been documented in 100% of females in seriously affected populations (Ellis & Pattisina 1990). This unique cause-effect relationship between imposex and organotin pollution means that the distribution of imposex can be strongly inferred to be directly related to the distribution and availability of organotins in the environment. This, together with the undoubted detrimental effects of imposex on gastropod populations, means that imposex is an important biological indicator.

The unique cause-effect relationship between organotins and imposex has prompted a number of proposals for the use of imposex as a universal indicator of organotin pollution, and standard protocols have been proposed and trialed (Ellis & Pattisina 1990). Numerous studies in Australia and New Zealand have used imposex in gastropods to detect the magnitude and distribution of biological effects of organotins near slipways, marinas and shipyards (see, for example, Smith & McVeagh 1991, Foale 1993, Nias et al. 1993, Wilson et al. 1993).

Although imposex is diagnostic for the presence and availability of organotins, it does not define when the exposure occurred. The abnormality is apparently irreversible, so once induced, a pattern of imposex may be detected in a population even though the organotins are no longer present. The length of this ‘memory’ is related to the life cycle of the gastropod species concerned, but is typically at least several years. Many species of gastropod have been used for analysis of imposex frequency, and the precise methodology for determining imposex occurrence in any individual will depend on the species concerned because of the slight variation in morphology amongst the various gastropod species. Nonetheless, a global protocol has been developed and trialled that is applicable to a number of taxa (Ellis & Pattisina 1990). Although organotins are now restricted in use, they are still widely used in commercial fleets of ships, and there may be large quantities remaining in sediments from past uses.

For practical monitoring and assessment and where levels of imposex are considered to be elevated, confirmation and the degree and extent of contamination by organotins can be sought by complementary residue analysis of sediments, and possibly biological tissues.

Density of capitellid worms

In Australia there are 36 known species of marine polychaete worms belonging to the Capitellidae family. Some of the common genera found in Australia and/or New Zealand include: *Capitella*, *Heteromastus*, *Barantolla*, *Mediomastus*, *Scyphoproctus*, *Notomastus* and *Leiochrides*. They live, primarily, in marine and estuarine sediments that range from soft mud to muddy sand. Their wide distribution, their important ecological role in sediment processes and food webs, their easy identification (to family and genus level), a considerable history of research on their biology and ecology, and their known responses to various forms of pollution means that they are suitable taxa for use as biological indicators of water quality.

Polychaete worms in general have been recognised in many studies as useful indicators of environmental quality, and are widely recommended for this purpose (see for example Pocklington & Wells 1992). There is an extensive history of research relating polychaetes to polluted conditions, especially to nutrient enriched environments (Pearson & Rosenberg 1976, Gray & Pearson 1982). In particular, capitellids have been identified as responding to organic enrichment of sediments, typically, although not exclusively, in response to inputs of sewage (Reish 1957, Tsutsumi 1990, Weston 1990). In Australia and New Zealand, although there are few published studies, the general trend of greatly increased abundances of Capitellid worms in response to nutrient or other organic enrichment, as observed in other countries, has also been documented (Dorsey 1982, Roper et al. 1989, Ward & Hutchings 1996).

8.1.3 Choice of the appropriate indicator for biological assessment

Titles of the protocols relevant to biological indicators (listed in table 3.2.2 of Volume 1) are provided below while summary descriptions of these protocols, with references to important source documents, are provided in Appendix 3. These represent the available protocols for biological assessment in Australia, and in many cases New Zealand. The assessment objectives and water quality stressors that particular indicators may usefully be applied to are provided in Volume 1 tables 3.2.2 and in the current section. Selection of indicators should not be decided upon in isolation of the situation in which an environmental monitoring and assessment program is being developed. To this end, managers should also consider the advice provided in Section 7.2.1 that may assist them in deciding upon the type and number of indicators for their particular situation.

Only a limited number of general remarks are made at this stage about the choice of indicator organisms to select for in water quality assessment programs and these pertain to freshwaters. On balance and where it may not be immediately obvious as to the choice of biodiversity indicator to apply to streams, wetlands and lakes, macroinvertebrate communities probably represent the most broadly applicable group. Apart from the inherent virtues of the group for monitoring that were raised in Section 8.1.2, it is worth noting that there are very few water quality stressors to which macroinvertebrate community structure is unlikely to respond. A factor further enhancing their appeal for biological monitoring in Australia at least, is the enormous skill base that has developed across the country over the past several years largely as a consequence of the National River Health Program (NRHP) (Schofield & Davies 1996). Both as part of the NRHP and as a consequence of independent research, a substantial amount of work on taxonomy, ecology and technique development has also been conducted to underpin development of monitoring techniques using macroinvertebrate communities.

Most of the protocols described in Appendix 3 are generic and are broadly applicable to most regions of Australia and possibly New Zealand. Other protocols, however, have been developed for specific taxa. Implicit in applying a non-generic protocol of the latter type, is the availability or presence of these taxa in the region being investigated. Developmental work will normally be required in transferring protocols developed for one species to another (even congeneric) species, and in finding suitable species to monitor and test (e.g. Humphrey et al. 1995).

8.1.3.1 Protocols for streams, wetlands and lakes

Direct toxicity assessment

Suitable laboratory tests that may be used to predict the toxicity of a waste water prior to its release to the environment are listed in Section 8.3.6.

Method 1A(i), (ii): Instream/riverside assays measuring sublethal 'whole-body' responses of invertebrate and/or fish species

Suitable protocols which may be modified for local conditions are described in Appendix 3, Methods: 1A(i) 'Riverside monitoring: freshwater snail reproduction and survival', and 1A(ii) 'Riverside monitoring: larval fish survival test'.

Method 1B(i), (ii): Measurement of chemical/biochemical markers in aquatic organisms

Suitable protocols which may be modified for local conditions are described in Appendix 3, Methods: 1B(i) 'Bioaccumulators of metals and radionuclides', and 1B(ii) 'Molecular biomarkers in fish'.

Method 2A: 'Whole-sediment' laboratory toxicity assessment (where sediment tests are available).

Suitable protocols which may be modified for local conditions are described in Appendix 3, Methods: 2A 'Chironomid sediment test'. Additional sediment toxicity tests that have been developed for Australian conditions are listed in Section 8.3.6 of this volume.

Method 2B: Bioaccumulation/biomarkers (for organisms that feed through ingestion of sediment); other sublethal responses (incl. behavioural) where protocols developed.

Suitable protocols which may be modified for local conditions are described in Appendix 3, Methods: 2B 'Morphological deformities in chironomid mouthparts'.

Method 3A(i), (ii): Monitoring and assessment of streams using macroinvertebrate communities.

Suitable standardised protocols or those which may be modified for local conditions in Australia are described in Appendix 3, Methods: 3A(i), 'AUSRIVAS, a Rapid Biological Assessment method using stream macroinvertebrate communities', and Method 3A(ii) 'Changes in structure of stream macroinvertebrate communities for detecting and assessing impact'. Standard sampling methods for New Zealand streams are described in detail by Biggs (1983). The use of a macroinvertebrate community index for assessing organic contamination of streams in New Zealand is described by Stark (1985, 1993).

Method 3A(iii), (iv): Monitoring and assessment of wetlands and lakes using macroinvertebrate communities.

Suitable protocols which may be modified for local conditions in Australia are described in Appendix 3, Methods: 3A(iii), 'Rapid Biological Assessment of wetlands using macroinvertebrate communities', and 3A(iv) 'Changes in structure of lentic macroinvertebrate communities for detecting and assessing impact'. Standard sampling methods for New Zealand lakes are described in detail by Biggs (1983).

Method 3A(v): Structure of freshwater fish communities

Guidance on use of freshwater fish in monitoring programs in Australia is described in Appendix 3, Method: 3A(v), 'Guidance on the use of freshwater fish communities for detecting and assessing impact'. McDowall (1996) has concluded that it is inappropriate to attempt to use freshwater fish in New Zealand for bio-assessment because so many of the species are strongly diadromous. Further, there is no general, quantified relationship between either individual species' population characteristics, or community characteristics, and habitat quality.

Method 3B: Stream metabolism

A suitable protocol which may be modified for local conditions is described in Appendix 3, Method: 3B, 'Changes in community metabolism (e.g. GPP, R, P/R and NDM) as a means of detecting and assessing impact'.

The algal protocols of Method 4, for freshwaters, are divided into three categories, periphyton, phytoplankton and macroalgae. The first two categories are divided further into protocols for simple biomass measurements and protocols for community data where taxonomic identifications are required.

Method 4(i): Periphytic algae

Suitable protocols which may be modified for local conditions are described in Appendix 3, Methods: 4(i)A, 'Biomass of periphytic algae', and 4(i)B, 'Diatom community structure in rivers and streams'.

Method 4(ii): Phytoplankton

Suitable protocols which may be modified for local conditions are described in Appendix 3, Methods: 4(ii)A, 'Biomass of phytoplankton', and 4(ii)B, 'Phytoplankton cell density'.

Method 4(iii): *Macroalgae*

Suitable protocols which may be modified for local conditions are described in Appendix 3, Methods: 4(iii)A, 'Biomass of macroalgae', and 4(iii)B, 'Species composition of macroalgae'.

Method 5: Changes to wetland vegetation structure as measured through remote sensing

A suitable protocol which may be modified for local conditions is described in Appendix 3, Method: 5, ‘Changes to wetland vegetation structure as measured through remote sensing’.

8.1.3.2 Protocols for marine and estuarine ecosystems

Method 6: Seagrass depth distribution

Suitable protocols which may be modified for local conditions are described in Appendix 3, Method: 6, ‘Seagrass depth distribution, A: Seagrass Light Climate’, and ‘Seagrass depth distribution, B: Seagrass (Deep-water Edge) Distribution’.

Method 7: Frequency of algal blooms

A suitable protocol which may be modified for local conditions is described in Appendix 3, Method: 7, ‘Frequency of algal blooms’.

Method 8: Density of capitellids

A suitable protocol which may be modified for local conditions is described in Appendix 3, Method: 8, ‘Density of capitellid worms’.

Method 9: Imposex in marine gastropods

A suitable protocol which may be modified for local conditions is described in Appendix 3, Method: 9, ‘Imposex in marine gastropods’.

Table 8.1.5 Matching indicators/ variables to the problem. (Biodiversity indicators are denoted 'Q' for quantitative studies or 'RBA' for rapid biological assessment.) For Columns 3 and 5, letters S = streams, W = wetlands, L = lakes and M = estuarine/marine, denote the ecosystem type the indicator is relevant to. '?' indicates that confirmation of utility is required for ecosystem type in Australia and New Zealand.

Problem	Potentially suitable indicators and measured responses	Ecosystem type	Advantages (+) & disadvantages (-)	Ecosystem type
Nutrients, herbicides: (Early detection/changes to biodiversity)	Periphytic (benthic or epiphytic) algae		+ Direct and rapid response to nutrients.	All
	– Biomass (chlorophyll *a*)	S, W, L, M?	+ Overseas techniques well developed; embryonic use in Australia but techniques extensively developed for flowing waters in New Zealand. Generally applicable to most inland waters.	S, W, L
	– Community structure (diatom RBA)	S, W?, L?		
	– Community structure (Q)	S, W?, L?	+ Diatoms preserved in sediments may be useful for palaeolimnological monitoring	W, L, M?
	– Presence / absence or abundance categories (macroalgae)	All?	- May not be useful in deep, high turbidity waters or in coloured (high gilvin content) wetlands	All
			- Spatial and temporal variations in community structure may be very high (low power to detect effects)	All
	Phytoplankton		+ Direct and rapid response to nutrients	S, W, L, M?
	– Biomass (chlorophyll *a*)	All	+ Includes simple methodologies	All
	– Frequency of algal blooms	All	+ Overseas techniques well developed and documented	All
	– Community structure (Q)	All	+ Some freshwater taxa sensitive indicators of trophic state	S, W, L
			+/- Algal blooms may be localised (enabling identification of source of effects); this advantage may be negated in marine environment by ready movement by currents	All
			- Little expertise available in Australia, esp. taxonomic (community structure)	All
			- Not applicable to swift-flowing upland streams	S
			- Poor background knowledge of ecology	All
			- May not be useful in deep, high turbidity waters or in coloured (high gilvin content) wetlands	All
			- Algal blooms difficult to quantify and may not always be obvious at surface	M, all?
			- Algal blooms not always directly linked to nutrients; are also a complex function of dissolved form of nutrient, light and temperature (and hence season)	All
			- Spatial and temporal variations in community structure may be very high	All
	Macrophytes		+ Potentially useful indicators of water and/or sediment quality, depending upon life form	W, L, M
	– Emergent or submersed vegetation	W, L	+ Ground survey techniques easily applied; GIS approaches established	W, L, M?
	– Seagrass depth limits	M	+ Many species are habitat-forming (changes may cascade through system)	W, L, M
			- Lack of knowledge about population dynamics, and of how factors other than water quality affect distribution and growth	W, L, M

Table 8.1.5 (continued) Matching indicators/variables to the problem

Problem	Potentially suitable indicators and measured responses	Ecosystem type	Advantages (+) & disadvantages (-)	Ecosystem type
Nutrients: (Early detection/ changes to biodiversity)	Stream metabolism – GPP, R, P/R and NDM	S	+ In some nutrient-poor forest streams, may provide inexpensive, advanced warning of nutrient enrichment - Unless correlated with changes in structure of aquatic ecosystem components, may lack information about ecological relevance and importance	S (all attributes)
Nutrients: (Changes to biodiversity)	Macroinvertebrates – Community structure (Q, RBA)	S, W, M	+ Ubiquitous and found in most habitats + Large no. of taxa offers a wide range of responses (diagnostic value) + Limited mobility allowing effective spatial analyses of disturbance effects + Larval stages often extend up to and beyond one month, hence integrate effects of prolonged or intermittent exposure +/- Identification relatively easy at family level, less so at species level; methods well established and improving ecological knowledge base +/- RBA methods quicker, but some loss of information incurred; quantitative methods slower, more expensive but more sensitive - Sample processing and identification of samples labour intensive	S, W, M S, W, M S, W, M S, W, M S, W, M? S, W, M? S, W, M
General organic and inorganic contaminants (including metals, pesticides): (Early detection of changes (water column & sediments))	General	S, W, M	Following advantages/disadvantages apply to early detection indicators listed in the general group, 'Early detection of acute and chronic changes, water column and sediment': + High sensitivity; timely detection of effects of particular substances at specific sites, i.e. prior to, or indicating the onset of, actual environmental impacts +/- Response may be highly specific for particular contaminants or be very general; this must be known. Specific choice of indicator will usually depend on stressor and on system in question - Timeliness may be compromised by need for both adequate baseline and post-disturbance data from the field, and (usually) dose/exposure-response relationships from laboratory studies, to interpret results and strengthen inferences - Site-specific assessments difficult with mobile species (e.g. fish, frogs) - May lack ecological relevance: very few responses have been linked to effects at higher levels of biological organisation (e.g. ecosystems); sensitivity of the selected test species in single-species tests/studies may be unrepresentative of the wider assemblage of organisms in the field. - Biology of organism measured must be understood	S, W, M

Table 8.1.5 (continued) Matching indicators/variables to the problem

Problem	Potentially suitable indicators and measured responses	Ecosystem type	Advantages (+) & disadvantages (-)	Ecosystem type
General organic and inorganic contaminants (including metals, pesticides): (Early detection of changes (water column))	Physiological or biochemical (suborganismal) changes: includes invertebrates & fish	All	+/- Advantages & disadvantages listed above ('General') + Provides indication of bioavailability of contaminant + Overseas techniques well developed and documented; limited use in Australia & New Zealand - Can be expensive	S, W, M
	Whole-body responses of organisms (field and laboratory toxicological assessments of ambient waters or effluents): includes lethality, growth, reproduction	All	+/- Advantages & disadvantages listed above ('General') + End-points regarded as being of greater ecological relevance than many other 'early detection' methods - Field application can be expensive (non *in situ* methods) - Techniques being developed overseas; limited use in Australia & New Zealand +/- Laboratory direct toxicity assessment (DTA): See sect 8.3.6; DTA is carried out in conditions that rarely match environmental conditions in the field.	S, W?, M? (all attributes)
	Whole-body responses of organisms (field surveys): abnormalities	All	+/- Advantages & disadvantages listed above ('General') *Frogs* + Skeletal abnormalities purported to indicate exposure to low concentrations of contaminants, including metals and radionuclides +/- Other advantages/disadvantages listed under 'Frogs' below (General contaminants: changes to biodiversity) *Birds (egg shell thinning)* +/- Very specific in terms of contaminants - Sublethal but slow/lagged response - Mobility of birds (site-specific assessments may not be possible) *Imposex in gastropods* + Direct relationship with exposure to organotins + Limited mobility allowing effective spatial analyses of disturbance effects + Robust overseas models/concepts – Observed effects requires laboratory verification	All S, W, L (all attributes) W (all attributes) M (all attributes)
	Sentinel organisms (bioaccumulation, body burdens): includes macrophytes, long-lived invertebrates (molluscs, crustaceans), fish	S, W, L, M	+ Species selected can absorb or adsorb quite high concentrations of contaminants without direct toxicity + Provides indication of bioavailability of contaminants + Methodology well established. - Some lakes / wetlands may lack indicator of sufficiently large size or age.	S, W, M

Table 8.1.5 (continued) Matching indicators/variables to the problem

Problem	Potentially suitable indicators and measured responses	Ecosystem type	Advantages (+) & disadvantages (-)	Ecosystem type
General organic and inorganic contaminants (including metals, pesticides): Early detection of changes (sediments)	'Whole-sediment' laboratory toxicity assessment	All?	+/- Advantages & disadvantages listed above ('General') + End-points of lethality, growth, reproduction regarded as being of ecological importance + Overseas tests commonly based upon *Chironomus*, a genus common in wetlands/ lakes of Australia and New Zealand. - Few sediment tests available in Australia (see sect 8.3.6) - Laboratory results might not accurately reflect effects that can occur at the ecosystem level (i.e. test conditions may not simulate actual environmental conditions)	W, S, L, M W, S, L, M W, S, L W, S, L, M
	Morphological deformities of sediment-dwelling organisms	S, W	+/- Advantages & disadvantages listed above ('General') + Methodology well established for specific groups (e.g. chironomids) - Few laboratory studies undertaken to validate/verify effects observed in the field - Need to ascertain degree of applicability to specific stressors	W, S (all attributes)
General organic and inorganic contaminants (including metals, pesticides): Changes to biodiversity	Algae	S, W?, L?	- Amongst metals, generally only sensitive to copper - Planktonic forms may be too 'dilute' and transported readily in upland portions of streams - Little expertise available in Australia and New Zealand - Spatial and temporal variations in community structure may be very high	S, W, L (all attributes)
	Macrophytes	S, W, L	+ Many species are habitat-forming (changes may cascade through system) + Ground survey techniques easily applied; GIS approaches established (emergents) -/+ Few forms directly sensitive to metals; useful for herbicides, suspended solids, acidification and as bioaccumulators - Lack of knowledge about population dynamics, and of how factors other than water quality affect distribution and growth	S, W, L (all attributes)
	Zooplankton	S, W, L	+ Sensitive to wide range of contaminants - Spatial and temporal variations in community structure may be very high - Little expertise available in Australia and New Zealand - May be too 'dilute' and transported readily in upland portions of streams	S, W, L (all attributes)

Table 8.1.5 (continued) Matching indicators/variables to the problem

Problem	Potentially suitable indicators and measured responses	Ecosystem type	Advantages (+) & disadvantages (-)	Ecosystem type
General organic and inorganic contaminants (including metals, pesticides): Changes to biodiversity	Macroinvertebrates – Community structure (Q, RBA)	 S, W	As for nutrients	S, W
	Fish	S, W, L	+ High public profile + Sensitive to wide range of contaminants + Taxonomy usually simple; sample processing costs generally 'small' + Sampling methods are well established - Assemblage-based approaches compromised in most of southern and inland Australia where species diversity is low, fluctuations in species abundance and occurrence are extreme, and relative dominance of exotic species is high. - High mobility: interpretation of data must take into account factors affecting entire catchment/region; studies at small spatial scales generally not possible without experimental designs that incorporate truly independent controls (outside catchment/region). - Diadromous nature of most of native fish fauna in New Zealand precludes use of natural freshwater fish communities for bioassessment of water quality	S, W, L, M S, W, L, M S, W, L, M? S, W, L, M? S, W, L S, W, L, M S
	Frogs	S, W, L	+ High public profile and concern + Sensitivity to a wide range of contaminants + Skeletal abnormalities purported to indicate exposure to low concentrations of contaminants - No standard techniques developed in Australia and New Zealand - Semi-aquatic nature of the life cycle of most species - Fully aquatic (larval) phase often highly seasonal and transient - Mobility of adults - High inter-annual variability of breeding adults, spawn and recruitment - Little known about response to changing environmental conditions	S, W, L (all attributes)
	Waterbirds	W	+ Good baseline info – RAOU - Indirectly affected by most contaminants - High mobility/migratory nature [as for fish above]	W

Table 8.1.5 (continued) Matching indicators/variables to the problem

Problem	Potentially suitable indicators and measured responses	Ecosystem type	Advantages (+) & disadvantages (-)	Ecosystem type
RBA for early detection	Macroinvertebrates	W, S	+ Rapid techniques recently established in Australia + In their broad coverage may identify and detect problem locations and stressors that would otherwise pass unnoticed + Can be measured at relatively low cost at a large number of sites or over large geographical areas + Have ecological, regional and social relevance - Low power to detect changes - For particular sites, not sufficiently sensitive to detect subtle impacts at an early stage - RBA methods for wetlands not developed beyond south-west WA - Wetland Biotic Index (WBI) approach developed for south-west WA may be restricted to effects of eutrophication (J Davis, Murdoch University, pers. comm.)	S, W? (all attributes)
RBA for biodiversity/ broad-scale ecosystem health	Macroinvertebrates	W, S	+ Same advantages as listed above for macroinvertebrates ('Nutrients: changes to biodiversity') + Rapid techniques recently established in Australia + Method is widely regarded as one adequately reflecting the ecological condition or integrity of a site, catchment or region + Approaches to sampling and data analysis are highly standardised + Response is measured rapidly, cheaply and with rapid turnaround of results + Results are readily understood by non-specialists + Response has some diagnostic value - Method is not designed to detect minor or subtle impacts (negating their sole use in regions of higher conservation value) - For site-specific assessments, the AUSRIVAS method may have limitations - see Sections 3.2.2.1/3 & 8.2.5.2/1, and table 8.1.2 - RBA methods for wetlands not developed beyond south-west WA - Wetland Biotic Index (WBI) approach developed for south-west WA may be restricted to effects of eutrophication (J Davis, Murdoch University, pers. comm.)	S, W? (all attributes)

Table 8.1.5 (continued) Matching indicators/variables to the problem

Problem	Potentially suitable indicators and measured responses	Ecosystem type	Advantages (+) & disadvantages (-)	Ecosystem type
RBA for biodiversity/ broad-scale ecosystem health	Fish – Presence/absence and/or indices	S?, W?	+/- Same advantages/disadvantages as listed in 'Fish: General contaminants, changes to biodiversity', above + Simple developing approaches in Australia (streams), cheaply applied over large spatial scales - No method currently applicable at a broad scale and none extensively tested to date; use of overseas indices problematic (see Section 8.1.2) - Lack of understanding of fish population dynamics and ecology in Australian systems. - Not appropriate in New Zealand	S?, W? (all attributes)
	Frogs – Presence/absence	W?	+/- Simple approach (frog calls) but highly dependant upon seasons, environmental conditions +/- Some background data available (FROGWATCH) but not yet developed +/- Same advantages/disadvantages as listed in 'Frogs: General contaminants, changes to biodiversity', above	W? W? W?
	Birds – Presence/absence	W?	+ Good baseline information - RAOU - Indirectly affected by most contaminants - High mobility/migratory nature [as for fish above]	W W
Suspended solids; Sedimentation; Dredging	Macrophytes – Emergent or submersed vegetation – Seagrass depth limits	 W, L M	+/- Same advantages/disadvantages as listed on 'Nutrients: Early detection/ changes to biodiversity', above	W, L, M
	Macroinvertebrates – Community structure (Q, RBA)	 S, W	+/- Same advantages/disadvantages as listed on 'Nutrients: Early detection/ changes to biodiversity', above	S, W
	Stream metabolism – GPP, R, P/R and NDM	S	+ In some nutrient-poor forest streams, depressed GPP may indicate sedimentation or degradation in water quality - Unless correlated with changes in structure of aquatic ecosystem components, may lack information about ecological relevance and importance	S (all attributes)

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