Assessment methods and strategies

Introduction

While technical quality is a vital component of an overall environmental sampling program the core issue should always be quality of decisions. Assessment methods need to be considered in the context of the specific program.

We believe that:

- *A priori* there are no core indicators
- Indicators must be chosen on the basis of the hypotheses of the program
- The "best" method is not necessarily the appropriate method (ie. The "gold standard" is a point of comparison not a given)
- Field and analytical methods must be comparable in terms of precision
- Analysis cannot compensate for poor experimental design
- Design, sampling, analysis, and reporting need to be integrated
- Quality control is a necessary component of any sampling program.

The important thing is that methods are appropriate to the outcomes, both in terms of approach and level of sensitivity required.

Methods

There are a number of relevant sources of information on sampling methods:

The ANZECC Monitoring Guidelines Chapter 3.5 discusses selection of measurement parameters

The Monitoring Guidelines Chapter 4 discusses Field sampling program including sampling and QA/QC

Physical and Chemical

Traditional physical and chemical methods of assessing water quality provide an indirect estimate of ecological detriment. They derive legitimacy mainly by recourse to data derived from single-stressor toxicity tests conducted under controlled laboratory conditions. By repeating this process for a number of biological taxa, an estimate is made of the 'population' response to individual chemicals, and numerical guidelines set accordingly. The derivation of 'global' guideline values, though conceptually simple, faces formidable logistical and statistical challenges. These include the reconciliation of data derived under experimental conditions that are not readily comparable and the relevance of simple guideline in complex *real world* ecosystems. Nevertheless, direct measurement of water quality parameters as a surrogate for ecological health has the advantages of:

- conceptual simplicity
- established technology
- explicit numerical objectives
- the ability to acquire meaningful quantities of data relatively quickly
- comparatively low costs

Types of physical and chemical stressors (ANZECC 3.3.2.1)

Physical and chemical stressors can be classified broadly into two types depending on whether they have *direct* or *indirect* effects on the ecosystem.

Direct effects

Two types of physical and chemical stressors that directly affect aquatic ecosystems can be distinguished: those that are directly toxic to biota and those that while not directly toxic can result in adverse changes to the ecosystem (e.g. by stimulating excess growth of cyanobacteria). While the focus here is mainly on problems caused by excessive amounts of direct-effect stressors, some of the elements and compounds covered here as stressors, are also essential at low concentrations for the effective functioning of the biota. Nutrients such as phosphorus and nitrogen are essential for plant growth, but at higher concentrations can stimulate excessive growth leading to problems. Similarly, heavy metals such as copper and zinc are essential elements for the growth of all biota at small concentrations, but can become toxicants at higher concentrations.

Two types of direct-effect stressors are considered:

- *Toxic, direct-effect stressors* – salinity, pH and temperature are considered in this section. All other toxic stressors are covered in Sections 3.4 and 3.5. The trigger levels of toxic stressors are generally established from laboratory-based biological effects (ecotoxicity) data obtained from the testing of a range of sensitive aquatic plant and animal species (section 3.4). However, these three stressors are naturally very variable between and within ecosystem types and seasonally, with the natural biological communities adapted to the site-specific conditions. This suggests that trigger levels for these three stressors may need to be based on site-specific biological effects data.
- *Non-toxic, direct-effect stressors* - these are not directly toxic, but can change the ecosystem in ways that can either cause changes in biological species diversity or abundance, or result in problems for some human use. Examples include:
    - nutrients that can result in excessive algal growth and cyanobacterial blooms;

- suspended particulate matter that can reduce light penetration into a waterbody and result in reduced primary production, and possible deleterious effects on phytoplankton, macrophytes and seagrasses, or smother benthic organisms and their habitats;
- organic matter that can significantly reduce the dissolved oxygen concentration and cause death of aquatic organisms, particularly fish;
- flow which can significantly affect the amount and type of habitats present in a river or stream.

Indirect effects

These include stressors (or factors) that while not directly affecting the biota affect other stressors making them more or less toxic. For example, dissolved oxygen can influence redox conditions and influence the uptake or release of nutrients by sediments. Equally, pH, dissolved organic carbon (DOC) and suspended particulate matter (SPM) can have a major effect on the bioavailable concentrations of most heavy metals.

Through the risk-based hierarchical decision frameworks adopted for the application of the guidelines (section 3.1.6) these indirect stressors are considered through protocols requiring that ecosystem-specific modifying factors be considered in the assessment of each issue when a potentially high risk situation has been identified. Although many effects of these modifying factors are reasonably well known from a theoretical viewpoint, there are few quantitative relationships (or models) that allow these factors to be used to develop more ecosystem-specific guidelines (Schnoor 1996). Recommendations made in section 9.5.2 (Volume 2) cover the type of research and development needed to develop these relationships.

Issues affecting aquatic ecosystems that are controlled by the physical and chemical stressors (ANZECC 3.3.2.2)

Many aquatic ecosystems experience a range of problems that affect biodiversity or ecological *healt*h. These problems mostly result from anthropogenic activities. This section focuses on the development of guideline 'packages' to address eight specific issues likely to result from physical and chemical stressors:

- nuisance growths of aquatic plants
- maintenance of dissolved oxygen
- effects of suspended particulate matter
- effects of salinity changes
- effects of temperature changes
- effects of pH changes
- changes in optical properties of waterbodies
- effects due to changes in flow

8.2 Physical and chemical stressors (ANZECC Vol 2, Chapter 8)

8.2.1 Fact sheets

8.2.1.1 Nutrients

8.2.1.2 Dissolved oxygen

8.2.1.3 Turbidity and suspended particulate matter

8.2.1.4 Salinity

8.2.1.5 Temperature

8.2.1.6 pH

8.2.1.7 Optical properties

8.2.1.8 Environmental flows

8.2.1.9 Hydrodynamics

Additional information is also available in:

DECC *Approved methods for the Sampling and Analysis of Water Pollutants in New South Wales*

How do I test water salinity?

Biological

Biological assessment can measure the desired management end-points for an ecosystem (e.g. maintenance of a certain diversity of fish species or certain level of nuisance algae) as might be described in the management goals. Biological assessment provides information on biological or ecological outcomes that may result from changes in water quality but may also result from changes in the physical habitat (e.g. increased fine sediment deposition, or changes in hydrology) or of changes in biological interactions (e.g. the introduction of exotic species or diseases).

Thus, while bioassessment should be seen as a vital part of assessing changes in aquatic ecosystems, and as a tool in assessing achievement of environmental values and attainment of the associated water quality objectives, the resulting biological *message* provides an insight into a complex system which:

- integrates multiple natural and human changes in physico-chemical conditions;
- integrates impacts over time;
- absorbs human impacts into complex interacting biological communities and processes;
- can give a signal from more than one component (e.g. multiple species or community similarities or ecological processes). The guidelines for biological assessment are based on the notion of assessing a *significant* departure from a relatively natural, unpolluted or unimpacted state – the reference condition (see section 3.1.5 for discussion on reference condition). A significant departure is deemed to be one in which significant impacts occur on the biological diversity of ecosystems, including:

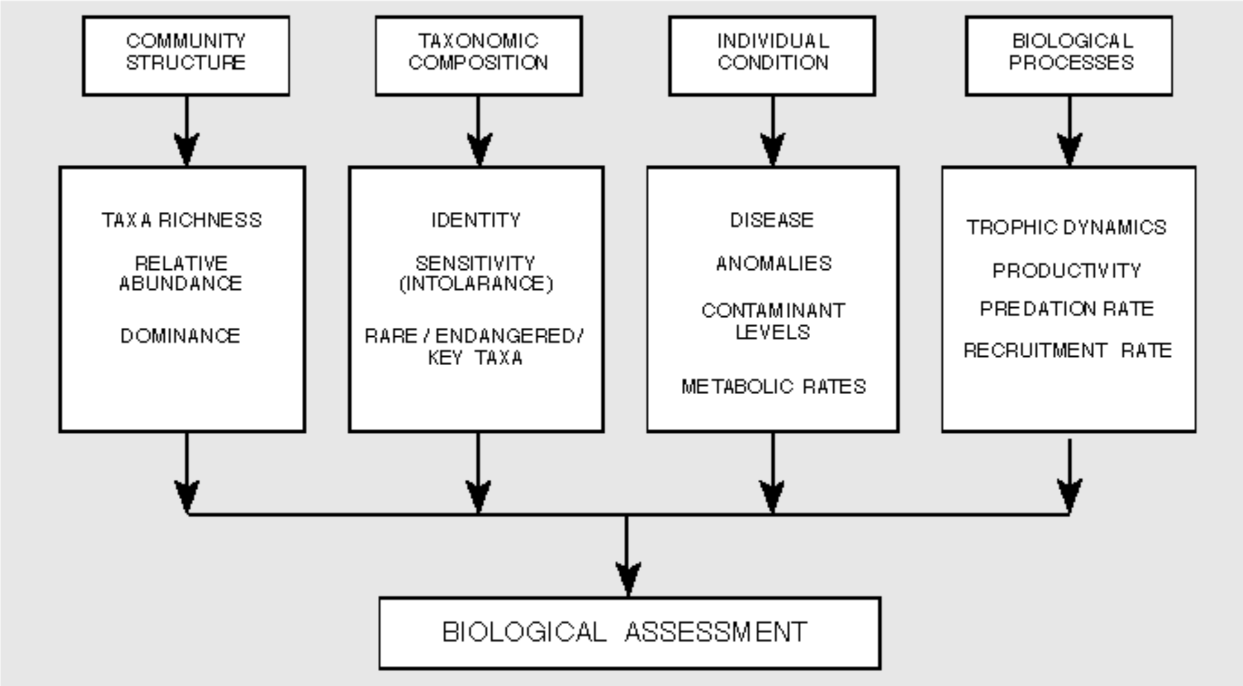
- changes to species richness, community composition and/or structure;
- changes in abundance and distribution of species of high conservation value or species important to the integrity of ecosystems;
- changes to ecosystem processes of a physical, chemical or biological nature.

Significant in this context does *not* mean mere statistical significance, which is only a tool in the context of a specific monitoring design. Rather it means a change or departure deemed practically significant in relation to previously agreed performance criteria for failing to achieve a water quality objective. The results of bioassessment may require interpretation using additional supporting information on water quality and physical conditions at site, catchment or regional scales.

Bioassessment provides a *window* onto the condition of the ecosystem being managed. Implicit in the use of biological indicators is the recognition that *single value* or *single state* threshold guidelines cannot be derived in a meaningful way for biological communities or processes. Strong variation in ecosystem processes and biological community composition in time and space is characteristic of many surface water environments, particularly in Australia. Given the variability in the state of biological systems it is important to understand the limitations of sampling designs and their ability to detect and quantify change relative to an undisturbed or reference state. For any given sample size or number of sample units taken during a monitoring or assessment program, there are quantifiable constraints on its ability to detect a change of a given magnitude. There is a strong relationship between the power (in statistical terms) of a monitoring program design, the magnitude of the effect that is detectable and the sample sizes involved. There is also a trade-off between an ability to detect change, the sample size and the willingness to risk non-detection of that change (or to risk detection of a change when it has not occurred). This trade-off is often negotiated on the basis of financial resources for monitoring programs, since increasing sample sizes or numbers of sample units is the most common way of increasing the power to detect a change.

It is vital to recognise the need for high quality, comprehensive designs in bioassessment and

biological monitoring. Significant efforts are being made to develop protocols for bioassessment with improved designs and rigour in both site selection, sampling approaches and analysis.



Choice of the appropriate indicator for biological assessment

Titles of the protocols relevant to biological indicators are listed in table 3.2.2 of Part 1 of the ANZECC Guidelines, with further discussion in Chapter 8.1, and 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 tables 3.2.2 and 8.1.3. 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 8.1.1 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 Chapter 8.1, 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”.

Macroinvertebrates and Habitat

Biological monitoring, the study of biological organisms and their responses, is used to determine environmental conditions. In wadable streams (streams that can be easily walked across, with water no deeper than about thigh high), the three most common biological organisms studied are fish, algae, and macroinvertebrates.

Macroinvertebrates are organisms that are large (macro) enough to be seen with the naked eye and lack a backbone (invertebrate). They inhabit all types of running waters, from fast flowing mountain streams to slow moving muddy rivers. Examples of aquatic macroinvertebrates include insects in their larval or nymph form, crayfish, clams, snails, and worms. Most live part or most of their life cycle attached to submerged rocks, logs, and vegetation.

Aquatic macroinvertebrates are good indicators of stream quality because:

- They are affected by the physical, chemical, and biological conditions of the stream.
- They can't escape pollution and show the effects of short- and long-term pollution events, although you should be aware of life stages and habit before making decisions based on particular species and results.
- They may show the cumulative impacts of pollution.
- They may show the impacts from habitat loss not detected by traditional water quality assessments.
- They are a critical part of the stream's food web.
- They varying degree of tolerance to pollution.
- They are relatively easy to sample and identify.

The basic principle behind the study of macroinvertebrates is that some are more sensitive to pollution than others. Therefore, if organisms that can tolerate pollution inhabit a stream site and the more pollution-sensitive organisms are missing a pollution problem is likely.

For example, stonefly nymphs (aquatic insects that are very sensitive to most pollutants) cannot survive if a stream's dissolved oxygen falls below a certain level. If a biosurvey shows that no stoneflies are present in a stream that used to support them, a hypothesis might be that dissolved oxygen has fallen to a point that keeps stoneflies from reproducing or has killed them outright. This brings up both the advantage and disadvantage of the biosurvey. The advantage of the biosurvey is that it tells us very clearly when the stream ecosystem is impaired, or "sick," due to pollution or habitat loss. It is not difficult to realize that a stream full of many kinds of crawling and swimming "critters" is healthier than one without much life. The disadvantage of the biosurvey, on the other hand, is that it cannot definitively tell us why certain types of creatures are present or absent.

In this case, the absence of stoneflies might indeed be due to low dissolved oxygen. But is the stream underoxygenated because it flows too sluggishly or because pollutants in the stream are damaging water quality by using up the oxygen? The absence of stoneflies might also be due to other pollutants discharged by factories or running off farmland, water temperatures that are too high, habitat degradation such as excess sand or silt on the stream bottom that has ruined stonefly sheltering areas, or other conditions. Thus a biosurvey should be accompanied by an assessment of *habitat* and *water quality* conditions in order to help explain biosurvey results.

Habitat, as it relates to the biosurvey, is defined as the space occupied by living organisms. In a stream, habitat for macroinvertebrates includes the rocks and sediments of the stream bottom, the plants in and around the stream, leaf litter and other decomposing organic material that falls into the stream, and submerged logs, sticks, and woody debris. Macroinvertebrates need the shelter and food these habitats provide and tend to congregate in areas that provide the best shelter, the most food, and the most dissolved oxygen. A habitat survey examines these aspects and rates the stream according to their quality. This chapter includes both simple and intensive habitat surveys volunteers can conduct.

Monitoring for water quality conditions such as low dissolved oxygen, temperature, nutrients, and pH helps identify which pollutants are responsible for impacts to a stream.

Uses of the Biosurvey and Habitat Assessment

The information provided by biosurveys and habitat assessments can be used for many purposes.

- Biosurveys can be used to identify problem sites along a stream. A habitat assessment can help determine whether the problem is due, at least in part, to a habitat limitation such as poor bank conditions.
- *To identify the impact of pollution and of pollution control activities.* Because macroinvertebrates are stationary and are sensitive to different degrees of pollution, changes in their abundance and variety vividly illustrate the impact pollution is having on the stream. Losses of macroinvertebrates in the stream, or of trees along the stream bank, are environmental impacts that a wide segment of society can relate to. Similarly, when a pollution control activity takes place say, a fence is built to keep cows out of the stream a biosurvey may show that the sensitive macroinvertebrates have returned and a habitat assessment might find that the formerly eroded stream banks have recovered.
- *To determine the severity of the pollution problem and to rank stream sites.* To use biological data properly, water resource analysts generally compare the results from the stream sites under study to those of sites in ideal or nearly ideal condition (called a reference condition). Individual stream sites can then be ranked from best to worst, and priorities can be set for their improvement.
- *To determine support of aquatic life uses.* All states designate their waters for certain specific uses, such as swimming or as cold water fishery. States establish specific standards (limits on pollutants) identifying what concentrations of chemical pollutants are allowable if designated stream uses are to be maintained. Increasingly, states are also developing biological criteria essentially; statements of what biological conditions should be in various types of streams throughout the state. States are required by the Clean Water Act to report on those waters that do not support their designated uses. Biological surveys directly examine the aquatic organisms in streams and the stressors that affect them. Therefore,

these surveys are ideal tools to use in determining whether a stream's designated aquatic life uses are supported.

- *To identify water quality trends.* In any given site, biological data can be used to identify water quality trends (increasing or decreasing) over several years.

Extracts from Chessman, B.C. 1995. Rapid assessment of rivers using macroinvertebrates: A procedure based on habitat-specific sampling, family level identification and a biotic index. Australian J. Ecology. 20:122-129.)

> When whole communities are considered, discrimination between sites using classification and ordination can be as effective, or nearly as effective, at the family level as at the species level. Family-level studies of river macroinvertebrates have been used successfully for such purposes as describing biogeographical patterns across large areas and assessing responses to regulation.
>
> It should be emphasized that the rapid techniques are not advocated as a substitute for detailed quantitative studies at the species level. Such studies will always be needed, for example where the precise quantification of the impact of a major development is required, or where the conservation of rare species is of concern.
>
> Rapid techniques are particularly appropriate in two circumstances:
>
> 1. The first of these is when a preliminary survey is required over a broad area and at a large number of sites. In such cases rapid assessment can be used to describe general patterns and to pick out those sites which appear to be most degraded. such sites can then be subjected to more detailed studies, in order to confirm or modify the preliminary findings, determine cause and effect relationships, and provide a firmer basis for remedial management.
> 2. The second main use of rapid techniques is when a preliminary assessment of the condition of a waterway is needed immediately for management purposes.

Extracts from: Reynoldson, T.B., Norris, R.H., Resh, V.H., Day, K.E., and Rosenberg, D.M. 1997. The reference condition: a comparison of multimetric and multivariate approaches to assess water-quality impairment using benthic macroinvertebrates. J. N. Am. Benthol. Soc. 16(4):833-852

The reference condition is the condition that is representative of a group of minimally disturbed sites organized by selected physical, chemical, and biological characteristics.

> The reference condition is used by comparing the biological attributes of individual test sites with a group of reference sites expected to be similar.
>
> The reference-condition approach differs fundamentally from other approaches commonly used for water quality assessments (e.g., traditional studies using Before After Control Impact designs and ANOVA) in that sites, rather than multiple collections within sites, serve as replicates.
>
> An advantage of using this definition of the reference condition is that, after reference sites have been grouped by some method (e.g., classification using biota), independent data (e.g., physical and chemical) can be used to match test sites to the most appropriate group of reference sites for bioassessment. Specimens were identified to the family level.

Conclusions of the authors: "Multimetrics are attractive because they produce a single score that is comparable to a target value and they include ecological information. However, not all information collected is used, metrics are often redundant in a combination index, errors can be compounded, and it is difficult to acquire current procedures. Multivariate methods are attractive because they require no prior assumptions either in creating groups out of reference sites or in comparing test sites with reference groups. However, potential users may be discouraged by the complexity of initial model construction. The complementary emphases in the multivariate methods examined (presence/absence in AusRivAS cf. abundance in BEAST) lead us to recommend that they be used together, and in conjunction with, multimetric studies."

Additional Information Is Available In:

New South Wales (NSW) Australian River Assessment System (AUSRIVAS) Sampling and Processing Manual 2004

AUSRIVAS Macroinvertebrate bioassessment Predictive modelling manual

ANZECC Guidelines

NSW EPA QA-QC Guidelines

Visual indicators

The issue of scale remains a key consideration in any assessment. Visual indicators, such as geomorphology, vegetation communities, physical impacts can demonstrate medium- to long-term processes and as such provide information that short-term chemical indicators may not. Although it is easy to consider visual indicators as "unscientific" and "good for volunteers" there has been considerable work done on these and you should always consider their inclusion in any surveys. Ther is also evidence that properly undertaken visual surveys correlate well with more comprehensive biological surveys.

Useful information is available in:

Stream Visual Assessment Protocol USDA National Water and Climate Centre Technical Note 99–1

Rangeland Monitoring Series: Visual Assessment of Riparian Health UNIVERSITY OF CALIFORNIA Division Of Agriculture And Natural Resources

Devising a sampling protocol

The sampling program in support of virtually any monitoring project will be subject to resource constraints. These will often be financial but may also be logistical, for example, the number of samples that can be processed and analysed using the laboratory staff and facilities available within the time frame of the project. While this may seem self-evident, it must be explicitly addressed in the project plan, and the number of samples that can realistically be processed specified in advance. It is important to remember that quality assurance samples, such as replicates and field blanks, will increase the total number of analyses which in turn restricts the number of unique samples that can be determined. Quality assurance requirements increase the need to adopt a critically thoughtful approach to sampling to ensure that maximum information is extracted from the analytical data. Vital for consideration in developing the sampling protocol are:

(i) site selection,

(ii) ensuring the representative nature of samples for each site, and

(iii) management of variance.

Selection of sampling sites

The selection of sites within the defined sampling area may be either random or constrained. Random site selection may be appropriate under the following circumstances:

- a formal (especially legal) requirement to sample randomly;
- equal importance being placed on all locations within the sampling area, for example, a need to monitor an entire catchment for nutrient loads;
- a need to identify a source of contamination where no information exists about its origin.

The advantages of random sampling include the elimination of observer bias in the selection, for example choosing sites (such as boat ramps or bridges) that are easily accessible. For this reason alone such locations may be inappropriate choices because of increased risk of contamination from visitors to the site. Random site allocation is also a formal requirement for the use of inferential statistics in data interpretation. However, the extent to which the validity of conclusions is undermined by a more constrained approach to site selection may be small, assuming there are no major methodological deficiencies. This point is elaborated later.

A random selection of sites is readily obtained using graphical and random number generating techniques, for example, the superimposition of a fine grid on the sampling area, and allocation of sequential numbers to the intersection points of grid lines.

Where random site selection is not imperative, but no data exist on which to make an informed judgment on site selection, an alternative is to use an optimisation technique. A number of ‘spatially optimum water sampling plans’ exist, the merits of which have been recently reviewed (Dixon & Chiswell 1996). A common feature of these is the minimization of a mathematical function. One promising technique owing to its ability to escape from local minima and find the true function minimum is *simulated annealing*. This method mathematically simulates a process analogous to annealing of metals. Annealing causes a realignment of the crystalline metal structure into a state of minimum potential energy, and is facilitated by heating the solid, followed by slow cooling. For simulated annealing, the ‘cost function’ to be minimised is defined in terms of the requirements of the sampling problem. For example, if sampling sites are to be located at points of equal upstream drainage area, the cost function would be the standard deviation of subcatchment areas. Similarly, the standard deviation of discharge volume or cumulative stream length could be the value designated for minimisation. The technique can be constrained to select only from allocated sites, such as gauging stations or public access areas. For simulated annealing a digital elevation model of the catchment, obtained from a geographic information system is required.

The design of most sampling programs for physical and chemical indicators have at least an element of judgment in the selection of sampling sites because the experimenters usually have a specific issue or problem to monitor, and therefore have a reasonable idea of where to confine their sampling activities. If for example a point source of effluent is being investigated, in the absence of geographical or arbitrary boundaries, the sampling area will probably be defined by comparison with reference or control values of the indicators being determined. Ideally the criterion in this case would be that indicator concentrations measured at the sampling point most distant from the source of effluent would be statistically indistinguishable from those at control sites. However, the practicalities of sample collection and analysis may require that sampling resources be deployed in places of higher effluent concentrations. Decisions such as these are matters for judgment in specific circumstances, but would be decided, apart from considerations of resources and logistics, by factors such as:

- concentration gradients in the water system,
- consistent observations (from historical measurements or pilot studies) that all indicators at a certain site were a small proportion of the guideline values (for example 20%).

The issues of historical databases and pilot studies are discussed below.

Because most guideline values are absolute concentrations, there is usually no *formal* requirement to include control or reference sites in a sampling program. The exceptions to this are the small number of guideline values expressed in terms of percentage change from a control or reference value. However, data from control sites are almost always useful in defining a sampling area, and often sampling intensity. Any sampling study that purports to be complete must include at least one ‘unaffected’ site in a sampling program, and depending on circumstances, several may be required. Control values are vital for the comparative calculation of *loads* at control and impact sites, and this information may be necessary for ameliorative strategies.

Once a sampling area has been decided, it is probably best to determine unique sites using a random selection procedure. Apart from eliminating observer bias, this probably validates the use of inferential statistics, on the basis that the ‘population’ of affected sites is randomly sampled. This assumes the spatial and temporal basis by which the ‘population’ boundary is defined includes a high proportion of all possibly relevant observations.

Representative sampling and the management of variance

For the purpose of discussion, it is assumed the sampling area (and sites within it) have been selected in a way that permits the assumption that the sites are in principle representative. The term representative is taken here to mean *relevant to the purpose intende*d. Therefore, to take representative samples, there is no alternative to explicitly defining the purpose intended. A few examples have been provided in the following box to help clarify this point.

An issue closely related to the collection of representative samples is ‘control of variance’. This term is not meant to suggest that an attempt should be made to minimise the estimate of true variance in the sample population. Rather, the inclusion of samples that are not representative with those that are, will inevitably increase the component of variance that is irrelevant to the defined objectives of the project. Increased variance of this kind, in addition to impairing the integrity of data to an extent that may make them useless or misleading, will also make it more difficult to adhere to the recommended general attainment benchmark for these guidelines, as specified later. Often, ostensibly well-designed sampling programs will incorporate misleading variance, which can be traced to the inclusion of non-representative samples. One example can arise with the acquisition of depth samples. Depth samples, whether individual or integrated, are inconvenient and expensive to obtain and analyse. If depth gradients are observed, a decision is required on whether the whole water column represents the population of interest, or whether a subset, such as the surface or the bottom is more relevant to the objectives of the study. An extreme example of this is the existence of a thermocline, as in the case described above.

Another example of the inclusion of non-representative samples is the equal temporal spacing of sampling events in a system where relevant indicator concentrations are dominated by disruptive events. Where the sampling interval is much longer than the duration of the relevant event, a significant water quality incident may be missed altogether, or only the tail of the event observed.

Where there is a possibility that short-duration events may be critically relevant to the objectives of a project, a sampling strategy to account for this contingency must be included in the sampling protocol.

Hydrology and representative samples

Rapid changes in the discharge rate of streams can have profound implications for indicator concentrations, and therefore the representativeness of sampling. Even under relatively stable flow conditions, hydrological measurements must be made simultaneously with water sampling where indicator loads (rather than concentrations) are required. Several methods for determining stream discharge are available (see for example Standards Australia 1997). In general, initial calibration is made by directing stream flow through an aperture (weir) of known cross section. The velocity of flow is measured and used with the cross-sectional area to calculate a flow volume, which is then related to a gauge height. After the calibration has been made, only water height need be measured until a recalibration is performed. The measurement of a single parameter (gauge height) is convenient and is not subject to failure, mechanical impairment or occlusion, as may occur with other continuous measurement devices (such as flow meters).

Changes in stream hydrology can alter water quality parameters rapidly and sometimes unpredictably. Some reasons for this are:

- hydrological changes alter the relative proportion of discharge originating from runoff, baseflow and groundwater. Runoff water may be (Hart et al 1987), but is not invariably, of better quality than groundwater and baseflow. This can often be observed as a period of decreasing conductivity on the rising hydrograph, followed by rising conductivity on the falling hydrograph. Alternatively, runoff may contain increased concentrations of nutrients (from fertilised fields, urban areas or sewage treatment plants) or heavy metals and organics (from contaminated sites).
- differential rainfall within a catchment may give different patterns of water quality depending on the location of the rainfall;
- deliberate releases of contaminants may coincide with extreme hydrological conditions to take advantage of the large dilution factors available;
- extreme rainfall conditions may breach bunds and other containment devices used for the retention of contaminants;
- in the case of temporary streams, very large changes in water quality may occur during extreme recessional flow and during the ‘first flush’. In the latter case, accumulations of chemical species in catchment soils and near-surface groundwater (sometimes acidic from organic degradation or sulfide oxidation) may dramatically alter the concentration of some indicators.

Short-duration storm events are often unpredictable, and logistic considerations may make it impractical to initiate a sampling effort when such events occur. Where sampling during disruptive events is judged to be of critical importance, suitable strategies can be incorporated into the sampling protocol, mostly involving the use of automatic samplers. A variety of robust and reliable automatic sampling devices can now be obtained. These may be deployed continuously, with the samples being discarded during routine site visitation if a target hydrological event has not occurred. More sophisticated automatic sampling devices are available, which can be triggered to commence collection at specified conditions of flow. Very large numbers of samples can be acquired using ‘continuous samplers’ (Standards Australia 1997). The samples thus acquired can be bracketed, so that only a subset reflecting the conditions of particular interest need be analysed. It should always be borne in mind that the integrity of samples collected by automatic devices may be compromised by delayed preservation. This would normally require samples to be collected and processed as soon as possible after the relevant event.