Water Quality for Maintenance of Aquatic Ecosystems: Appropriate Indicators and Analysis

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Preface

Australia: State of the Environment 1996 (the first ever independent and comprehensive assessment of the state of Australia's environment) was presented to the Commonwealth Environment Minister in 1996. This landmark report, which draws upon the expertise of a broad section of the Australian scientific and technical community, was prepared by seven expert reference groups working under the broad direction of an independent State of the Environment Advisory Council. While preparing the report DEST, on behalf of the reference groups, commissioned a number of specialist technical papers. These have been refereed and are now being published as the State of the Environment Technical Paper Series. Reflecting the theme chapters of the report, the papers relate to human settlements, biodiversity, the atmosphere, land resources, inland waters, estuaries and the sea, and natural and cultural heritage. The topics covered range from air and water quality to sea grasses and historic shipwrecks.

Abstract

The purpose of this paper is to identify aspects of water quality that will give an indication of the suitability of a water body (lake, river, wetland) for the maintenance of aquatic ecosystems. These indicators could then serve as an appropriate basis for reporting on the national state of the environment. The approach followed was first to identify activities that have the potential to cause an impact on aquatic ecosystems, and second to identify indicators of the discharges associated with such activities. In this process, it was necessary to recognise that natural differences in water quality exist and that ecosystems are adapted to these differences.

1 Importance of appropriate indicators of water quality

Selection of indicators is a crucial element in the development of a State of the Environment (SoE) report. Indicators selected should be relevant to the important issues, and to the decision-making process surrounding these issues. If this focus is not present, the usefulness of the SoE report will be reduced.

1.1 Rationale behind choice of indicators

1.1.1 Anticipated problems or impacts on inland waters

The process of identifying appropriate indicators depends firstly on identifying problems and issues that relate to water quality in Australia. Two ways of identifying problems are suggested in CEPA (1992): current water quality issues identified by community or political processes, and impacts expected from pressures or activities known to occur in catchments. The process of using current issues tends not to result in a comprehensive identification of issues; major issues of current importance are highlighted but other issues can be overlooked. However, it is an important process as it flags issues in which the public are keenly interested. Identification of issues from a review of potential pressures on aquatic ecosystems can result in a more comprehensive review. Extensive overlap would be expected between the issues identified by these two processes.

This paper relied on a range of sources to identify current water quality issues: principally issues identified in the media and recent Government, industry and peak agency literature. When water quality is discussed in the media, three issues predominate: cyanobacterial blooms, salinisation of water bodies, and toxic chemicals in water bodies. Current literature largely reflects these concerns. A recent workshop on national SoE reporting (DEST 1994, unpubl.) highlighted the issues of channel erosion and sedimentation and the impact of land use and land management practices on such processes. A review of the status of Australian rivers in 1992 on behalf of the Commonwealth Government identified the issues of salinity, nutrients, toxicants and sediment as important issues for Australian river water (DWR 1992). The Murray–Darling Basin Commission has recently identified the issue of eutrophication to be of importance (MDBC 1994).

CEPA (1992) emphasises the importance of identifying issues rigorously and comprehensively; otherwise the following process of identifying relevant indicators and establishing guideline levels is flawed. In order to ensure a rigorous identification of impacts on aquatic ecosystems, pressures expected from particular activities have been identified from a range of sources and are listed in Table 1. Also listed are indicators that would provide a measure of discharges from such activities.

Table 1: Land uses and activities, potential discharges and indicators of discharge

Potentially impacting land use or activity	Potential discharge	Indicators
Forestry	Export of sediment, nutrients Saline water Decaying vegetation	Turbidity, suspended solids, nitrogen, phosphorus Conductivity Dissolved oxygen, organic carbon, biochemical oxygen demand
Rural use (grazing, crop production)	Export of nutrients Export of sediment Saline water Decaying vegetation Animal faeces Export of pesticides	Nitrogen, phosphorus Turbidity, suspended solids Conductivity Dissolved oxygen, organic carbon, biochemical oxygen demand Nitrogen, phosphorus Specific pesticides
Urban area	Export of nutrients Export of sediment Export of hydrocarbons Export of trace metals Domestic household chemicals Park maintenance (herbicides)	Nitrogen, phosphorus Turbidity, suspended solids Oil, petrol, polycyclic aromatic hydrocarbons Cadmium, lead, zinc, copper Pesticides, surfactants Glyphosate, simazine
Sewage treatment plants	Export of nutrients Export of sediment Export of trace metals	Nitrogen, phosphorus Turbidity, suspended solids, biochemical oxygen demand Cadmium, lead, zinc, copper
Mining (active and dormant)	Export of sediment Export of trace metals	Turbidity, suspended solids Cadmium, lead, zinc, arsenic, acidity (pH), copper
Water storage	Release of stored (hypolimnetic) water	Acidity (pH), dissolved oxygen, nitrogen, phosphorus, iron, manganese, turbidity, temperature
Land fills	Export of sediment Decaying vegetation Export of pesticides Export of hydrocarbons Export of nutrients Export of trace metals	Turbidity, suspended solids Dissolved oxygen, organic carbon, biological oxygen demand Specific pesticides Oil and grease, petrol, polycyclic aromatic hydrocarbons Nitrogen, phosphorus Cadmium, lead, zinc, copper, mercury, chromium, nickel, vanadium etc.
Industrial activity	Export of hydrocarbons Export of trace metals Export of acids	Oil, petrol, polycyclic aromatic hydrocarbons Cadmium, lead, zinc, arsenic, copper, mercury, chromium, nickel, vanadium pH
Sand mining	Export of sediment	Turbidity, suspended solids
Energy generation	Export of heat Export of trace metals	Temperature Trace metals

1.1.2 What makes a good indicator?

A number of agencies in Australia and overseas have been faced with identification of water quality indicators to assess ecosystem health. It is important to note that indicators provide a measure of the input of pollutants to a system, and from this potential impacts are implied. A useful approach taken has been to establish a set of criteria that describe good indicators, and use these criteria to assess indicators objectively. In reviewing criteria used to select indicators elsewhere, a set of common features can be identified. Indicators should be:

(a) Easily measured. It must be possible to monitor an indicator, to collect data of an acceptable quality.

(b) Scientifically credible. The indicator must be unambiguously related to environmental impact, and must be measurable at a level of accuracy that permits identification of significant change.

(c) Easy to understand. The indicator and the potential impact on the environment that it reflects must be comprehensible to lay persons.

Despite meeting the above criteria, an indicator may fail to be useful because at present data have not been collected for this indicator. It has been suggested that an additional criterion should be that there are sufficient data for the indicator to be useful immediately. This criterion imposes severe constraints on indicators that may be used. The role of this document was seen to define the most appropriate set of indicators regardless of data availability. It was felt that the incentive to comply with national SoE reporting may provide an impetus to fill gaps in the data. The gap between what is recommended in this document in terms of indicators and data requirements and what is currently available nationally can provide a basis for recommendations on upgrading Australia’s water quality monitoring.

1.2 Recommended indicators

The issues and impacts listed in Table 1 can be grouped into two broad categories: those that apply to many catchments across Australia (e.g. rural use), and those that are linked to particular sites and geographically restricted (e.g. mining). For these two categories of impact, two sorts of indicators are required, general and point-source indicators.

Table 2: General indicators

Indicator	Input	Potential impact
Conductivity	Salt	Salinisation
Total phosphorus	Phosphorus	Eutrophication
Total phosphorus to total nitrogen ratio	Phosphorus and nitrogen	Cyanobacterial blooms
Biochemical oxygen demand	Carbon in organic material	Eutrophication or fish kills
Turbidity	Sediment	Change in ecosystem (changed habitat, loss of sensitive species, changed light climate)
Suspended solids	Sediment	Change in ecosystem (changed habitat, loss of sensitive species)
Chlorophyll	Nutrients	Eutrophication
Pesticides in sediment (aldrin, chlordane, endosulfan, endrin, heptachlor, lindane, methoxychlor and acrolein)	Pesticides	Change in ecosystem (loss of sensitive species)

General indicators provide a measure of discharges that cause broad-scale impacts—for example, catchment degradation, eutrophication and changes in aquatic ecosystems. Point-source indicators provide a measure of discharges that are more geographically localised—for example, mine discharges and other point-source discharges leading to localised changes in aquatic ecosystems.

1.2.1 General indicators

The following are recommended as general indicators:

- conductivity
- total phosphorus
- biochemical oxygen demand
- turbidity
- suspended solids
- chlorophyll
- pesticide concentrations in stream sediments.

Conductivity

The effect of changes in total dissolved substances on aquatic ecosystems will depend on the dissolved constituents, and their combined effects on the total osmolality. Hart et al. (1991) reviewed the effects of total dissolved substances on aquatic organisms, with special reference to native species. They identified a dearth of information on the responses of Australian species to overall changes in the concentration of total dissolved substances, and even less information on the effects of individual constituents. Nevertheless, given the extent to which human activity has changed concentrations of total dissolved substances, they are an appropriate indicator of water column change and potential impact on ecosystems.

Measures of the total dissolved substances in a water body include total dissolved salts, salinity, filterable residue and conductivity. Conductivity is recommended as the indicator to be used as a measure of the concentration of total dissolved substances in waters. It meets most of the criteria for a good indicator: it can be measured reliably and quickly, it is understandable, and there are obvious effects on biological processes.

Total phosphorus

Nutrients are among the factors that lead to eutrophication in aquatic ecosystems. The relationship between nutrients and eutrophication is not a simple one as there are a number of other factors which also affect aquatic plant growth, including light, mixing regime and temperature. The aquatic plant community occurring in a water body will be determined by these factors, in conjunction with conditions in the water body over the previous months and years.

Although nutrients are a natural component of all water bodies, increases in the nutrient supply tend to change the community structure. Phosphorus is the plant macro-nutrient most commonly associated with eutrophication. Two measures of phosphorus are commonly used: total phosphorus (TP) and orthophosphate. A measurement of total phosphorus includes all types of phosphorus present in the water column: dissolved forms, insoluble particle forms and phosphorus already incorporated in phytoplankton. While TP is a robust, reliable, comprehensive measurement, at times it may not accurately reflect the proportion of phosphorus that is immediately available for uptake by phytoplankton.

Conversely, orthophosphate is the common dissolved form of phosphorus, and it provides a more accurate indication of the phosphorus immediately available for uptake by phytoplankton. However, significant doubts have been cast on the reliability of many of the orthophosphate measurements taken in Australia because of the lability of the compound (Lambert et al. 1992).

Total phosphorus concentration is recommended for use as an indicator of aquatic ecosystem health in preference to orthophosphate for the reasons outlined above. Despite its general nature, TP provides a robust, reliable measure of the extent to which a number of land uses may be contributing phosphorus to aquatic ecosystems. TP is relatively easy to analyse (Lambert & Maher 1995), and its relationship to eutrophication, while complex, is fairly well understood (Harris 1994).

Total nitrogen to total phosphorus ratio

An aspect of the current concern with eutrophication is the apparent increase in the number of algal blooms, particularly of cyanobacterial (blue–green) blooms. An algal bloom refers to a situation in a water body when phytoplankton abundances are so high that the water becomes distinctly green in colour. The Organization for Economic Cooperation and Development (OECD) has formally defined a bloom as when chlorophyll *a* concentration reaches 20 µg/L. Cyanobacterial blooms may be even more marked than those of other phytoplankton as at times these phytoplankton float to the surface forming scums. Cyanobacteria are a natural component of aquatic ecosystems; however, for many streams, the frequency that populations reach bloom proportions is not. The conditions that lead to a cyanobacterial bloom are a combination of the primary factors that not only lead to an increase in phytoplankton biomass, but also allow cyanobacteria to numerically dominate over other phytoplankton. One feature which distinguishes many cyanobacteria from other phytoplankton is their ability to fix atmospheric nitrogen into plant tissue. Under conditions in which nitrogen is the limiting nutrient, some cyanobacteria have a competitive advantage and increase in abundance at the expense of other phytoplankton (Paerl 1988). When nitrogen is not limiting, the reverse appears to be the case.

This understanding has led to the suggestion that, under conditions conducive to the formation of an algal bloom, the total nitrogen to total phosphorus (TN:TP) ratio could be used as an indicator of the propensity for a cyanobacterial bloom (DWR 1992; MDBC 1993). The application of this approach as a management tool is not straightforward. The ratio of phosphorus to nitrogen in different phytoplankton species differs, suggesting that the TN:TP ratio at which nitrogen becomes limiting will also differ between species. There is also uncertainty as to whether total nutrient concentrations or dissolved nutrient concentrations should be used in calculating the ratio. Despite these concerns, the TN:TP ratio does offer a simple measure of the propensity of a water body to support a cyanobacterial bloom.

It is recommended that the TN:TP ratio be used as an indicator. The ratio is straightforward to calculate, is understandable, and the processes linking this measure to the condition of water bodies are reasonably well understood.

Biochemical oxygen demand

Carbon loading of water bodies results from a range of land uses or catchment activities and can result in two deleterious effects. If the loading is severe, oxygen depletion of the water body can occur with concomitant impacts on the aquatic ecosystem, (e.g. fish kills). Where this situation arises, it is commonly the result of discharges of agricultural or industrial effluent, (e.g. carbon-rich effluent from pulp mills, sugar cane waste and feed lots).

The second effect of carbon loading tends to be more subtle and its effect is principally on nutrient cycling, not water column oxygen levels. When fixed carbon sinks to the substrate of a water body and is degraded by sediment microbiota, it can result in oxygen depletion at the sediment–water interface and release of sediment-bound nutrients, particularly phosphorus. Far smaller carbon concentrations are necessary to produce this effect than those necessary to produce oxygen depletion of the water column. Carbon loading at a sufficient level to modify sediment chemistry can result from more widespread land use activities (e.g. forestry and urban land uses).

Measurement of the propensity for a carbon load to produce either of these effects is complicated by the range of carbon species that may be present. Although almost all carbon compounds may ultimately be available for microbiological degradation, the most important characteristic of a carbon load reaching a stream is the rapidity with which it may be degraded: its ‘biological accessibility’. If carbon is not rapidly degradable, the consumption of oxygen in its degradation will be slow and readily replaced by diffusion and oxygen depletion will not occur in the water column or sediment–water interface. Conversely, if a carbon load can be degraded quickly, oxygen depletion may occur.

A practical measure for the amount of carbon that is readily available for microbial degradation is the biochemical oxygen demand (BOD) of a water sample. This measure provides an empirical way of assessing the overall biological accessibility of a whole range of carbon compounds in the water column. It is recommended that the BOD be used as an indicator of the loading of readily accessible carbon to aquatic ecosystems.

Turbidity

The optical clarity of a water body is one of its fundamental physical characteristics. Turbidity is determined by two groups of factors: particulate components which includes suspended particles and algae, and dissolved components which affect water colour, particularly humic and fulvic acids. Both these groups of factors can be modified by a range of land use activities.

Different components of ecosystems are adapted to different levels of light. The plant community structure and biomass in an ecosystem are determined by the light regime, in conjunction with nutrient concentrations, temperature and mixing regime (Reynolds 1984). Change in water clarity may produce changes in the dominant phytoplankton, and in the seasonal succession of phytoplankton groups (Reynolds 1984). In addition, the common anthropogenic impact of a reduction of optical clarity tends to reduce the primary productivity of systems, and make them more suitable for phytoplankton growth than that of macrophytes or periphyton (Davies-Colley 1991; Vant et al. 1987). Changes to the primary producers will have profound effects on other trophic components of the ecosystem. There may also be direct effects of changes in water clarity at higher trophic levels—for example, effects on the hunting ability of visual predators such as carnivorous fish and birds. The importance of this effect is largely unknown (Davies-Colley 1991).

The optical clarity of water can be quantified in a range of ways, including the Secchi disk technique as a measure of visual clarity, Photosynthetically Available Radiation (PAR) measured by a sensor, and light penetration as measured by a Nephelometer. It is recommended that turbidity (measured as Nephelometric Turbidity Units (NTU) be used as an indicator of optical clarity. The use of this measure as the indicator of turbidity has advantages over some other techniques; it is an objective measurement compared to measures such as the Secchi disk depth, and it is a straightforward and easy variable to measure. It is acknowledged that this measure of turbidity will not quantify changes to the photosynthetic light regime as well as would a measurement of the PAR; but conversely it is a cheaper, simpler, more widely applied measurement.

Suspended solids

All streams and rivers naturally carry some suspended material: organic and inorganic particles over a range of sizes. Most land uses and activities have the potential to alter suspended solids concentrations in streams. The principal change in Australian streams appears to have been an increase in the concentration of suspended particles, with two major impacts. Firstly, higher concentrations of suspended solids reduce the optical clarity of water, thus reducing primary production and changing the phytoplankton species composition (Reynolds 1984). Secondly, increases in suspended solids concentrations ultimately result in increased deposition of material to the substrate where communities may be smothered or otherwise impacted (Campbell & Doeg 1989). As with other water column variables, aquatic ecosystems may be presumed to be adapted to a particular regime of suspended solids, and a variation from that regime, by a change in type or concentration, would be expected to impact on the ecosystem.

A difficulty arises with establishing guidelines for suspended solids concentrations given that flow-related variability may occur. Additionally, the link between suspended solids concentrations and their effect on benthic communities is not clearly quantified. Yet, despite these shortcomings, this variable provides one of the most direct indications of potential impact on aquatic ecosystems resulting from land use change or poor land management practices. A common corollary of many land use changes and poor catchment management is erosion, resulting in soil loss to streams. The best indicator of these impacts is the suspended solids concentration. It is recommended that the suspended solids concentration be adopted as an indicator.

Chlorophyll a

Chlorophyll *a* concentration is commonly used as an analog for phytoplankton biomass. Phytoplankton are an integral component of aquatic ecosystems and all water bodies, even impacted ones, will have a phytoplankton community. The biomass of a phytoplankton community is determined by a range of factors both natural and anthropogenic.

In many cases high phytoplankton biomass is the result of elevated nutrient inputs. A measure of phytoplankton biomass, using the chlorophyll *a* concentration, can serve as a useful indicator of the extent to which an ecosystem has been affected by nutrient inputs. It is recommended that the chlorophyll *a* concentration be adopted as an indicator.

Pesticides in stream sediments

Pesticides can have a major impact on aquatic ecosystems through both lethal effects and sub-lethal mutagenic or teratogenic effects on organisms. Contamination may also persist for months or years, long after the source of contamination has been removed (ANZECC 1992). However, because of the discrete nature of contamination events, contaminants may not be detected in the water column by a routine monitoring program. Pesticides tend to be insoluble in water and tend to bind to particulates and precipitate out of the water column (ANZECC 1992). They may spend only a short time in the water column before being partitioned to sediments. Measurement of pesticides in sediments provides a more realistic measure of the historical presence of these substances in the water column over the preceding weeks, months or years. The principal focus here is not the impact of pesticides upon sediment fauna, but the use of sediment concentrations to indicate what has passed through the water column.

The major source of pesticides in Australian water bodies is believed to be from agricultural application, though there may be some point-sources associated with urban development (e.g. landfill sites, termite treatment of house blocks). As a consequence, impact from pesticides is potentially widespread, and sediment pesticide concentrations should be monitored in a range of catchments. Representative catchments should identified on the basis of the type of agricultural activity, associated pesticide application and geographical region. Few data are currently available on stream sediment pesticide levels, and the expense of analysis will possibly limit the use of this indicator.

It is recommended that this indicator be adopted for national SoE reporting, and measurement be encouraged at a number of representative sites. Specific pesticides to be used as indicators have been selected on the basis of their usage and toxicity, and are: aldrin, chlordane, endosulfan, endrin, heptachlor, lindane, methoxychlor and acrolein. It is further recommended that the list of pesticide indicators be regularly reviewed given the changes that can occur in this industry.

1.2.2 Point-source indicators

The following are recommended as indicators of localised impacts.

Trace metals in sediment (As, Cd, Cu, Hg, Pb, Zn)

This group of contaminants includes compounds that are potentially very damaging to aquatic ecosystems through both lethal and sub-lethal effects on organisms. Sources of these contaminants tend to be spatially limited and linked to particular activities (e.g. mining, oil refining, smelting). As with pesticides, these contaminants tend to be hydrophobic, become bound to particles and removed from the water column relatively quickly. Routine water column sampling may not detect these chemicals, and the monitoring of sediment concentrations is advocated.

Trace metal pollution of waterways is caused principally by point-source discharges, particularly from the mining and manufacturing industry, and from urban runoff. Trace metal concentrations in sediment will provide an indication of the impact on ecosystems in areas under threat from such discharges.

It is recommended that trace metal concentrations in sediment be used as indicators of impact in areas identified as under threat from specific localised activities. Specific trace metals to be used as indicators are: cadmium, lead, zinc, copper, mercury and arsenic. These elements are those most commonly implicated in impacts on aquatic ecosystems. Given the changes in metal use that may occur, it is also recommended that the list of trace metals to be used as indicators be regularly reviewed.

Polycyclic aromatic hydrocarbons

Another group of substances that have caused impacts to aquatic ecosystems elsewhere are the polycyclic aromatic hydrocarbons (PAHs). Many PAHs have been demonstrated to be extremely toxic to aquatic biota (ANZECC 1992). PAHs are produced by a range of manufacturing processes, but overseas problems have principally been traced to oil-refining activities. As this industry in Australia is almost exclusively located on the coast or estuaries, PAHs are not believed to pose a significant water quality threat in freshwater waterways in Australia. PAHs are not necessary as freshwater indicators.

2 Analysis of data required to assess impact on and risk to aquatic ecosystems

2.1 Indicators as measures of water quality status and trend

2.1.1 Water quality status

The concentrations of many water quality variables (e.g. suspended sediments, salinity, turbidity and nutrient concentrations) vary both spatially and temporally. There may be marked changes along a river or between rivers, and over time at a particular site. Much of this variability is natural and is associated with seasonal changes or changes in flow. Superimposed on this natural condition, anthropogenic activities may exacerbate or modify the concentrations of limnological variables in water bodies. As a consequence, a water quality variable such as turbidity may vary by several orders of magnitude over the course of a year.

The challenge faced in national SoE reporting is to characterise the status of a stream in terms of a guideline value, taking account of this system variability. Given the flow dependence of many of the indicators recommended for the Australian SoE report, one possibility is to incorporate flow information into the assessment of water quality variables. This could involve discarding samples that were taken during very high or very low flow regimes. Alternatively, if the relationship between flow and the water quality variable were well understood, water quality measurements could be adjusted on the basis of the flow regime. There are currently insufficient data on a national basis to identify flow–variable relationships, and to either adjust or discard outlying values. Consequently neither of these approaches is recommended.

The approach recommended for the SoE report is to use all the sample data available, and to derive a statistic that provides a measure of status. This approach has the advantage that it makes use of all available data. It is also robust, and does not rely on an imperfect understanding of water quality dynamics. Although anthropogenic impact on water quality will be disguised to some extent by naturally high and low values, such changes will still be detectable.

Statistical analysis of water quality data reveals that in many cases variables are not normally distributed. Typically there is a skewing of the distribution towards low values. With this type of statistical distribution, the median is the most appropriate measure of status. This is known as a ‘resistant’ statistic as it is less affected by extremely high or low values than is the mean. The common alternative to the median, the mean, is more strongly influenced by the infrequent extreme concentrations that may occur in water quality monitoring. The use of the median (or any other measure of central tendency) tacitly accepts that, while at times there may be individual water quality measurements higher than a guideline value, the median value for a period falls within the guideline value. This is considered to be appropriate as most of water quality indicators vary naturally, and the guidelines are formulated on that basis (ANZECC 1992).

If data are available, the median and the 10th and 90th percentiles for each indicator could be plotted to give an indication of variability in the system.

The period over which water quality status is calculated needs to be long enough to incorporate the two main sources of temporal variability of water quality: seasonal and episodic variability. Conversely, the period used should not be so long that a measure of status is confounded by trend. It is recommended that status be calculated for the period between SoE reports, but that at least 20 measurements are required. This figure of 20 measurements is a compromise between the need for statistical robustness, and the desire not to impose data requirements so stringent that most sampling sites in Australia are excluded. Even so, much of the sampling carried out on Australian streams and rivers will have been too infrequent or over too short a period to be used to provide a measure of the status of a water body. It is recognised that at times sampling may occur episodically in response to a flow event or a suspected input. To ensure that a status measurement reflects water quality in the period between SoE reports, measurements contributing to a status calculation should span at least half of the reporting period.

2.1.2 Water quality trends

As with determination of water quality status, the natural variability of water quality variables associated with flow, seasonality or other factors can complicate identification of any trend. For example, despite the recognition that increasing salinity in the Murray River has been a major problem for many years, statistical demonstration of this trend has been made difficult by the large natural variability of salinity. A range of sophisticated techniques are available to remove or allow for some of the factors that contribute to data variability. These techniques generally rely on a better understanding of the factors involved than exist for most Australian streams and rivers. Their use is not yet advocated as a general reporting procedure for the national SoE reports. Trend evaluation for the SoE reports should be based on both a statistical and a graphical evaluation of the change in a variable over time. It is recommended that simple linear regression be used to investigate the statistical significance of a trend detected graphically.

Data used to identify a trend should span a period greater than that between national SoE reports. The longer the span of record the greater will be the power of data analysis to detect existing trends. It is not possible to define completely the span of record required as it will depend in part on the trend present, and the noise in the data. The minimum number of data points required statistically to confirm an apparent trend will depend on the data variability, but should not be fewer than 40. This figure represents a compromise between the statistical problems encountered in attempting to identify a trend with too few data points, and the desire to use the limited data that exists for most sites.

2.1.3 Concentration or load

Concentrations of pollutants are normally taken as the quantitative measure of water quality. This approach is based on the assumption that concentration of pollutants directly determines organism responses, and hence the impact of pollutants on an ecosystem. This assumption is valid in the case of toxicants. However, ecosystem response to physical (suspended solids, temperature) and common chemical variables (nutrients, dissolved oxygen and salinity) is far more complex. Many pollutants accumulate in sediments and are released under certain streamflow and seasonal conditions with detrimental impacts on aquatic ecosystems. The calculation of contaminant loads requires ‘event sampling’, the measurement of water quality and flow at a number of times during each rainfall event, and for a number of events. Where event sampling has occurred, its importance can be assessed by the high proportion of the total pollutant load that may pass along a waterway during floods. Measurement of total phosphorus exports from an urban catchment over two and a half months revealed that approximately half of the load passed in three short duration events (Cullen et al. 1978).

While this sort of finding indicates the importance of event sampling, only rarely is this type of sampling carried out. The inclusion of load-based indicators is not practical at this time due to the lack of event-based sampling. Alternatively, concentrations can serve as a measure of the impact of loads. Consequently, for the SoE report, pollutant concentrations rather than pollutant loads are recommended as the most appropriate measurement.

2.1.4 Data presentation

Status data

Reporting on water quality status should be at two levels: individual indicators and composite indicators. The purpose of this recommendation is to provide data at two levels of detail for different readers. Data for an individual indicator should be presented in relation to its particular guideline. Data for the different water quality indicators should also be amalgamated to provide a composite picture of water quality at a site or in an area.

At the scale of a national SoE report there are too many sampling sites to provide status information on a site-by-site basis. An alternative approach, and that recommended for the national SoE report, is to indicate where the sampling sites used are, but to provide information on a regional basis.

With individual indicators, the status of a region would be reported on the basis of the percentage of sites that were of higher or lower quality than that designated as the guideline. Section 2.4 discusses guideline values in more detail. Similarly, with composite indicators the status of a region would be reported as the percentage of sites of different overall quality (Section 2.4). The value of this approach is that a large quantity of information can be summarised concisely and in a form that can be easily understood.

Trend data

It is unlikely that there will be a large number of sampling sites at which sufficient information has been collected to enable trends to be determined. For appropriate sites two approaches are recommended. Several representative sites should be selected for which there are good data sets. For these sites data should be displayed as time series plots to demonstrate both trends in water quality and the underlying variability that can confound trend identification. Again, the median and the 10th and 90th percentiles for each indicator could be plotted to discern trends and indicate variability in systems. Trend information for remaining sites should be displayed symbolically, using symbols to indicate positive, negative trends, or an absence of trend, at different sites. It is anticipated that with the limited number of sites for which suitable data exist, it will be possible to present trend information for each individual site.

The suggestion may arise of combining trend data from different indicators into a composite view of trend, as is recommended with status data. Over a particular period, some indicators may demonstrate a positive trend and others negative in response to different catchment activities or climatic events. Combination of such trends would confuse understanding of any one relationship, and is not recommended as a useful exercise.

2.2 Spatial scale on which indicators of impact on aquatic ecosystems should be reported

Water quality data are available for a number of sites across Australia. In some areas, particularly the East Coast, sites are too numerous and closely spaced to be reported on individually. Consequently, it is advised that reporting for the SoE report be on the basis of a set of regions across Australia. Within a region, data for each indicator should be reported in a regional context, not on a site-by-site basis.

Regions used for reporting should be appropriately sized, ecologically significant, relatively homogeneous areas. They should be appropriately sized because the resolution chosen will be a compromise between a fine scale, at which it may be hard to appreciate much of the detail presented, and a coarse scale, at which over-generalised data may become meaningless. They should be ecologically significant, relatively homogeneous areas because if they are not, data from areas in which the natural ecosystems are different will be combined. Again the report will lose meaning.

If possible it would be useful to use regional divisions established for other reporting or administrative purposes. This would allow information in the national SoE report to be related to that from other sources. If, in addition, regions echoed administrative boundaries, there would be clearer delineation of management and political responsibilities for the state of water quality.

It is recommended that catchments form the regional basis for national SoE reporting in Australia. A catchment basis for reporting is appropriate ecologically. The natural water condition in each catchment, to which aquatic ecosystems will be adapted, will be determined by geological and geomorphological conditions unique to the catchment. Polluting activities and the effects of land use changes will tend to be restricted to the catchment in which they occur.

The Australian Bureau of Statistics (ABS) has divided Australia into 12 principal catchments, termed drainage divisions, each of which is further divided into a series of catchments termed basins. This catchment breakdown forms the geographical basis for some ABS reporting, and is recommended for the national SoE report.

Neither the drainage division scale nor the basin scale is an appropriate reporting scale for the entire country. In the north, some divisions are large as they will contain few sampling sites. Conversely in the south-east of Australia, there are sufficient sampling sites to allow meaningful reporting at the basin scale. It is recommended that a mixture of divisions and basins be used for reporting, to reflect the more intensive sampling in some parts of the country, and permit higher resolution of some threatened or impacted systems. The regions for which reporting should be made at the basin level are indicated in Table 3.

These regional divisions are used for other reporting purposes, most importantly by the ABS, providing ancillary information for the same area. As a final point, some administrative boundaries follow catchment boundaries, simplifying responsibility and management.

2.3 Guidelines for indicators

2.3.1 What are appropriate guidelines?

Water quality guidelines establish benchmarks for assessing the acceptability of water for a particular use (Lawrence et al. 1994). Guidelines represent the desirable condition of water bodies, and should be distinguished from water quality criteria and water quality standards.

Table 3: Australian drainage divisions and basins

Drainage division	Number of basins	Scale for reporting
North-east coast	46	Groups of basins: Cape York to Herbert R. basins Black R. to O'Connell R. basins Pioneer R. to Kolan R. basins Burnett R. to South coast basins
South-east coast	39	Groups of basins: Tweed R. to Hunter R. basins Macquarie Lakes to Towamba R. basins East Gippsland to Yarra R. basins Maribyrnong R. to Millicent coast basins
Tasmanian	19	Groups of basins: East coast basins West coast basins
Murray–Darling	26	Groups of basins: Murray R. to Mildura and Victorian tributaries Murrumbidgee and Darling tributaries from the east Darling R. and northern tributaries Mallee and Lower Murray R. basins
South Australian gulf	13	Groups of basins: Fleurieu Peninsula to Broughton R. basins Mambray coast to Eyre Peninsula basins
Western plateau	9	Entire Division
South-west coast	19	Groups of basins: Esperance coast to Busselton coast basins Preston R. to Ninghan basins
Indian ocean	10	Entire Division
Timor sea	26	Groups of basins: Cape Leveque coast to Daly R. basins Finniss R. to Buckingham R. basins
Gulf of Carpentaria	28	Entire Division
Lake Eyre	7	Entire Division
Bulloo–Bancannia	2	Entire Division

Table 4: Australian Water Quality Guidelines (1992) for the indicators recommended in this document

Indicator	Guideline
Conductivity	1500 µS/cm
Total phosphorus*	Varies with ecoregion Range for lakes: 5–50 µg/L Range for rivers: 10–100 µg/L
TN:TP ratio	No guideline
Biochemical oxygen demand*	No guideline
Turbidity*	< 10% change from seasonal mean
Suspended solids*	< 10% change from seasonal mean
Chlorophyll*	Range for lakes: 2–10 µg/L
Pesticides in sediments	No guideline
Trace metals in sediments	No guideline

* Denotes indicators for which guidelines should differ regionally

Water quality guidelines are measures of water quality that protect a particular environmental value, and are based solely on technical and scientific information (ANZECC 1992). Water quality guidelines are developed for management purposes from criteria that take into account value judgements on acceptable risk to humans or aquatic ecosystems. When the step is taken to enforce compliance with water quality guidelines by law, they are known as water quality standards. In the context of national SoE reporting we are dealing with guidelines; we require the value judgements on acceptable risk to ecosystems, and there are no national water quality standards for protection of aquatic ecosystems.

The guideline values for the indicators recommended in this document have been derived, where possible, from the *Australian water quality guidelines for fresh and marine waters* (AWQG)(ANZECC 1992). Where the AWQG provided no guideline, other State and Territory reporting frameworks were relied on. It was not felt that it was the role of this document to redefine guideline values, particularly nationally accepted guidelines.

For some of the indicators recommended in this document, the AWQG provide a guideline value to be applied to all water bodies across Australia (e.g. conductivity). However, for most of the recommended indicators, the AWQG recognise that differences in ecosystems will necessitate setting different guidelines for different types of water body. This is done in the AWQG either explicitly by recommending a range of guideline values (e.g. for total phosphorus), or implicitly by recommending that future change not exceed a certain percentage of the long term mean (e.g. for turbidity). A disadvantage inherent in the latter approach is that the future condition of a water body will be judged from its present state which may be degraded. Consequently, a degraded system may be judged as satisfactory because it has not deviated significantly from the previous (degraded) state. The guideline values from the AWQG for each of the recommended indicators are given in Table 4.

2.3.2 Different guidelines for indicators in different ecoregions

In Australian water bodies, naturally occurring concentrations of many common variables range widely. In particular, levels of suspended solids, turbidity and nutrients exhibit marked regional differences in concentration (e.g. coastal versus inland streams). For such variables an assumption commonly made (e.g. ANZECC 1992) is that aquatic ecosystems are adapted to site-specific water quality, and any change from such conditions will result in impact on the ecosystem. This assumption is based on experimental work on the tolerance of individual species, and the common–sense observation that there are stable aquatic ecosystems in pristine waters of very different water chemistry. The assumption will be adopted here as the basis for establishing different guidelines for indicators in different regions. Indicators for which different guidelines are required in different regions are indicated by an asterisk in Table 4.

The process of establishing different guidelines in different regions requires rigorous definition of those regions, 'ecoregions', based on the ecosystems they support. In fact this process is hampered by the poor definition of aquatic ecosystems in most regions, and the rudimentary understanding of the spatial scales and limnological gradients over which they change. A practical solution is to assume that aquatic ecosystems will change in synchrony with more easily quantifiable geographical features such as topography, river altitude or river slope. Although such an approach is based on broad generalisations, it does allow some recognition of the very real spatial differences that occur in ecosystems. This approach is recommended for the SoE report given the spatial scale over which this report is to apply, and the need to evolve a robust yet feasible scheme.

The most useful single analog for differences in aquatic ecosystems is altitude. In most places in Australia there are altitudinal gradients in temperature, stream morphology, and hydrology. These have given rise to less well-defined altitudinal differences in land use and associated water quality gradients in salinity and turbidity. As a secondary consideration, altitude data is readily available for all parts of Australia.

It is accepted that this approach is coarse and has flaws. For instance, obvious latitudinal gradients have been ignored at this stage to ensure practicability. Numerous local examples are available of ecosystem differences that do not follow altitudinal differences, and vice versa. It is possible to suggest a more complicated and perhaps realistic determination of ecoregions (e.g. altitude × latitude × a climate factor). However, the limnological information needed to establish guidelines for this complexity of ecoregions does not yet exist, and the authors would have no confidence in assigning guideline values to such ecoregions. Conversely, at the scale of a national report, a simple altitude-based approach is practicable, and will reflect true ecosystem differences to a useful extent.

Formation of categories within the altitudinal gradient is based on apparent changes, but will always contain an element of subjectivity.

Altitudinal categories suggested for use are:

Mountains—areas above 500 metres

> Areas above this height are largely forested and contain some of the most pristine water bodies in Australia.

Hills—areas between 100 and 500 metres

> This altitudinal region contains most of the inland rivers in the eastern States, and the middle reaches of rivers in the North, West and Tasmania.

Lowlands—areas below 100 metres

> This region comprises the coastal lowlands with slow-flowing rivers, coastal lakes and the water quality conditions associated with them.

A further distinction should be made between ecosystems in lakes, rivers and wetlands, although such distinctions can be blurred, especially in inland water bodies.

2.3.3 Guidelines

One of the purposes of this document is to establish operational guidelines for indicators. For most indicators this means establishing different guidelines for different ecoregions. From the AWQG in Table 4 it can be seen that for most indicators further definition of guidelines is required if they are to be reported on. It is recognised that in most cases there is insufficient scientific evidence to make unequivocal recommendations for guideline levels of indicators. Despite this, it is felt that current knowledge is adequate to establish working guidelines. These guidelines should be accepted as preliminary and subject to revision with growing understanding of aquatic ecosystems.

Recommended guidelines for indicators for the SoE report are given in Table 5. These values are based on the AWQG, and on State and Territory water quality guidelines. They also rely on state of environment reports from different States, in particular the Victorian report (*Victorian Inland Waters* 1988) which explicitly recognised different ecoregions. In essence the values recommended in Table 5 reflect the ranges given for these indicators in the AWQG, with cutpoints for ecoregion determined by reference to the other works and based on experience.

Table 5: Recommended guidelines for national state of the environment indicators

Indicator	Ecoregion		
	Lowlands	Hills	Mountains
Total phosphorus (µg/L)	50	50	20
TN:TP ratio	15:1	15:1	15:1
BOD (mg/L)	10	5	5
Turbidity (NTU)	20	15	10
Suspended solids (mg/L)	40	30	20
Chlorophyll (µg/L)	20	20	10

Total phosphorus: The guidelines recommended here are based on the ANZECC (1992) guidelines for lakes which suggest a range of 5–50 µg/L dependent on local conditions. The lake values have been used as many of the inland rivers in which algal problems occur may more appropriately be considered linear lakes (ANZECC 1992). The highest value has been adopted here for hill and lowland ecoregions on the basis that many of the aquatic ecosystems in these regions are naturally quite turbid, are possibly light-limited, and the phosphorus may be bound to particles and inaccessible. Under these conditions higher concentration of nutrients may occur without creating problems. In the mountains ecoregion the concentrations of TP is naturally lower than 50 µg/L, but considering the diversity of ecosystems within the ecoregion it is not appropriate to set the guideline at the lowest end of the AWQG scale.

TN:TP ratio: The guideline recommended here reflects the recognition in the area of water quality management that, when TN:TP ratios reach between 15:1 and 20:1, problems with blue green algae become more common.

BOD: The national guidelines recommend no value for the BOD concentration. The guidelines recommended here are based on guidelines used in the Victorian SoE report on inland waters (1988), the only State SoE report to date that has recognised the importance of geographically independent ecoregions. This document recommends that to maintain moderate water quality, concentrations of 5, 5 and 10 mg/L BOD respectively should not be exceeded for their three ecoregions: mountains, valleys and plains. While these concentrations may seem quite low, it should be remembered that even at low concentrations BOD may effect nutrient release from the substrate.

Turbidity: The national guidelines recommend only a relative change for turbidity. The reasons for recommending absolute guidelines for the SoE report have been discussed in Section 2.3. The guidelines recommended here are based on guidelines used in the Victorian SoE report on inland waters (1988). This document recommends that to maintain moderate water quality, concentrations of 10, 15 and 20 mg/L respectively should not be exceeded for their three ecoregions: mountains, valleys and plains.

Suspended solids: As with turbidity, the national guidelines recommend only a relative change for suspended solids, and the rationale for deviating from this approach has been discussed in Section 2.3. The guidelines recommended here are based on guidelines used in the Victorian SoE report on inland waters (1988). This document recommends that to maintain moderate water quality, concentrations of 20, 30 and 40 mg/L respectively should not be exceeded for their three ecoregions: mountains, valleys and plains.

Chlorophyll a: The ANZECC (1992) guidelines recommend up to 10 µg/L as the guideline for chlorophyll *a*, and this upper value has been adopted as the guideline for the mountains ecoregion. There is evidence that chlorophyll a levels may be higher than this figure without causing algal problems in more turbid waters (Ganf 1980), and the guideline for hills and lowlands ecoregions is set to 20 µg/L.

For reporting purposes it will not be useful to give absolute values because the significance of a particular value will differ between ecoregions. A practical approach is to define water quality categories based on the guideline value. At each site the water quality for a particular indicator (e.g. turbidity) would be assessed as one of four qualitative categories: excellent, good, poor, unacceptable (Table 6). The value given to the turbidity indicator reported for a division or basin would be based on the proportion of sites which fell into each of the quality categories. With this approach, the water quality in different divisions and basins may be compared even though they may comprise different ecoregions with different guidelines.

Table 6: Water quality categories

Water quality categories (all variables except TN:TP ratio)	**Assessment**
Median less than 50% of guideline value	Excellent
Median between 50 and 100% of guideline value	Good
Median between 100 and 150% of guideline value	Poor
Median more than 150% of guideline value	Unacceptable
TN:TP ratio	**Assessment**
Median more than 150% of guideline value	Excellent
Median between 100 and 150% of guideline value	Good
Median between 50 and 100% of guideline value	Poor
Median less than 50% of guideline value	Unacceptable

2.3.4 Composite indicators

Information from all the primary indicators should also be combined to create a composite picture of ecosystem water quality within a reporting region. This will give an idea of the overall impacts on an ecosystem as measured by the chosen indicators.

It is recommended that a measure of overall water quality be determined objectively using the following process. Using just the core variables—TP, suspended solids, conductivity, chlorophyll and turbidity—each variable would be given a score at each site as follows:

Excellent = 3 points
Good = 2 points
Poor = 1 point
Unacceptable = 0 points

For each site the scores for the five indicators would be summed to arrive at a composite score. As with individual indicators, water quality within a region would be reported on the basis of the percentage of sites falling into different categories of overall water quality. The categories recommended for assessment of whole ecosystems are:

Pristine:	Water quality close to pristine conditions. Overall score 12–15
Near pristine:	Water quality high enough to maintain all biota. Overall score 8–11
Moderate:	Water quality that would result in slight degradation of ecosystem. Overall score 4–7
Degraded:	Water quality that would result in marked deterioration of a natural aquatic ecosystem. Overall score 0– 3

2.4 Trace metals and pesticides in sediments

Problems associated with these toxicants, particularly trace metals and PAHs, tend to be site-specific. It is worth considering briefly where such problems have occurred, and their cause. Details of some of the better documented case studies are listed in Table 7.

From Table 7 it may be seen that the trace metal problem areas that have been identified are largely associated with mining activities. It is also indicated that there is a potential for many urban streams to be impacted by a range of activities. Although localised, these problems are important and have resulted in some of the most severe impacts on aquatic ecosystems in Australia. It is important to appreciate that, especially in urban areas, factors other than trace metals may impact on ecosystems (e.g. habitat modification). In such situations an attempt should be made to identify possible confounding factors, and isolate the effects of trace metals. Reporting on trace metals in identified problem areas and urban streams should comprise part of the national SoE report.

Some of the pesticide problems tabulated are a legacy of the period when the use of organochlorine pesticides was more widespread. With the restriction on their usage there has been a measurable change in water column concentrations of organochlorine pesticides (Hill & Nicholson 1992). Nevertheless, widespread use of pesticides and herbicides continues for a range of activities at a range of sites, including some of those above, and reporting on contamination should be included in the national SoE reporting.

Table 7: Examples of trace metal and pesticide problem areas in Australia

Location	Problem trace metals	Polluting activity
South Esk R., Tasmania	Cadmium, zinc	Mining
Molonglo R. NSW	Copper, zinc	Mining
King R., Tasmania	Copper, zinc	Mining
Finniss R. NT	Copper, manganese, zinc	Mining
Potentially many urban streams	Copper, lead, cadmium, zinc	Urban runoff, industrial and wastewater discharge
Location	**Problem pesticides**	**Polluting activity**
Namoi and Gwydir R. NSW	DDT, endosulfan	Cotton growing
Preston R. WA	Dieldrin	Intensive agriculture
Upper Yarra R. Vic	DDT, lindane, malathion	Intensive agriculture
Condamine/ Balonne R Qld	Dieldrin, DDT	Cotton growing
Ovens and King R. Vic	DDT, dieldrin, aldrin, lindane, heptachlor	Tobacco growing
Piccadilly Valley, SA	DDT, chlorpyrifos, lindane, endosulfan	Intensive agriculture

Although some problem areas have been identified (e.g. cotton areas, rice growing areas), the extent of the problem is unknown. The extent of the problem and impact may be much more widespread.

The AWQG provide guidelines for toxicant concentrations in the water column, but not for toxicant concentrations in sediments. It is not appropriate to apply water column guidelines to sediment concentrations for several reasons. First, sediment concentrations present a time integrated picture of the metal exposure in the water column. For example, a chronic but sub-guideline water column contamination by a toxicant could accumulate to a high concentration in the sediment. Secondly, a concentration process may be operating in that if all water column toxicants are sequestered to the sediment, they are removed from perhaps some metres of water column to several centimetres of sediment. Finally, toxicants in sediments may be essentially unavailable. Application of water column toxicant guidelines to the sediment will be misleading.

Given the uncertainty surrounding sediment levels of trace metals, PAHs and pesticides, it is not practical at this stage to attempt to establish guidelines for these substances. Consequently it will not be practicable to report on the status of these pollutants. It is recommended, however, that trends in these pollutants be reported following the approach proposed in section 2.1.4.

2.5 Suggested approach

1. Conductivity, total phosphorus, BOD, turbidity, suspended solids, chlorophyll *a* and TN:TP ratio should be used as general indicators of ecosystems' water quality.

   In addition, pesticides in stream sediments should be measured because of the broad scale use of these chemicals.
2. Trace metal concentrations in sediment should be used as indicators of impact in areas identified as under threat from specific localised activities.
3. Status should be calculated for the period between SoE reports, and at least 20 measurements are required. All water quality data available should be used to derive a statistic that provides a measure of status. If data are available medians and 10 and 90 percentiles should be used to give an idea of variability in systems.

   Data for individual indicators should also be presented in relationship to water quality guidelines. The status of a region should be reported as the percentage of sites that were of higher or lower quality than the designated guideline. Alternatively water quality categories based on guideline values (i.e. excellent, good, poor, unacceptable—Table 6) can be defined. The reporting on indicators for a chosen geographical scale (see Five) would be the proportion of sites which fell into each of the water quality categories.

4. Data used to identify trends should span a period greater than that between national SoE reports. The minimum number of data points required to confirm an apparent trend will depend on data variability but should not be fewer than 40.

   Several representative sites should be selected for which there are good data sets. Time series plots can be used to demonstrate both trends in water quality and variability that may confound trend identification. Again, the median and 10 and 90th percentiles for each indicator could be plotted to discern trends and indicate variability.

5. Reporting should be on the basis of a set of regions across Australia. Within a region, data for each indicator should be reported in a regional context, not on a site-by-site basis. The ABS's twelve drainage systems further subdivided into groups of basins (see Table 3) is recommended as the geographical basis of SoE reporting.

6. In general the guidelines used for reporting on water quality indicators should be those in the AWQG (ANZECC 1992). For total phosphorus, BOD, suspended solids and chlorophyll, different guidelines are required for different ecoregions reflecting the range of naturally occurring concentrations of these variables. Distinctions need to be made between natural waters in mountains (>500 m); hills (100–50 m) and lowlands (<100 m). Guidelines for these variables are proposed (Table 5) based on AWQG, State and Territory guidelines and SoE reports from different States.

7. In future SoE reporting, information from the indicators should also be combined to create a composite picture of ecosystem water quality within a reporting region. Using the core variables (TP, suspended solids, conductivity, chlorophyll, turbidity) each variable would be given a score—that is, excellent (3 points), good (2 points), poor (1 point), unacceptable (0 points).

   For each site the scores for the five indicators would be added to arrive at a composite score. Four water quality categories can be defined: pristine (score 12–15), near pristine (score 8–11), moderate (score 4–7) and degraded (score 0–3). Water quality within a region would be reported on the basis of the percentage of sites falling into the different categories.

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