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New generation water quality guidelines for ecosystem protection

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SUMMARY

1. Water quality guidelines have been used over the years as an important component of the management of water resources. Initially, these were focused on quality for domestic drinking water and water for agricultural, recreational and industrial purposes. More recently, however, the emphasis has been more on ecosystem protection.
2. This paper discusses the key elements of new risk-based water quality guidelines being developed in Australia and New Zealand, that should lead to more effective management and protection of aquatic ecosystems.
3. The essential elements of the new approach, shown diagrammatically in Figure 1, are:
- *ecosystem-based* - ideally the guidelines should be as far as possible ecosystem-specific.
- *issue-based* - the guidelines should focus on the actual issues or problems caused by physical, chemical and biological stressors rather than as present on the individual indicators.
- a *risk-based approach* - there is great difficulty in deciding whether adverse biological effects will result from various stressors added to an ecosystem. The new approach develops guideline "packages" for each issue and, where possible, for each ecosystem type. Each "package" consists of specified key performance indicators, trigger levels for these indicators (i.w. levels which indicate the risk that adverse biological effects may occur), and for high risk situations (where trigger levels are exceeded) a protocol for considering the effect of ecosystem-specific factors in reducing (or enhancing) the biological effects.

4. A case study related to a highly relevant aquatic ecosystem issue in Australia - the excessive growth of cyanobacteria (algal blooms) - is presented to illustrate how the new risk-based guidelines might be applied.

Introduction

Water quality guidelines have been used over the years as an important component of the management of water resources. Initially, these were focused on quality for domestic drinking water, and agricultural, recreational and industrial waters. More recently, however, the emphasis has been more towards ecosystem protection (Hart *et al.*, 1993). This paper reports a new risk-based approach towards the development of water quality guidelines for the protection of aquatic ecosystems.

Until very recently, the available water quality guidelines for ecosystem protection emphasised particularly the effects due to toxicants, with considerably less said about other factors (e.g. nutrients, organic matter, temperature, environmental flows, sediments) that can have equally important effects on ecosystem health. In many countries, including Australia and New Zealand, problems due to toxicants and particularly heavy metals, are of local significance. Often, there is much greater concern about problems caused by nutrients and other stressors [1] (Hart *et al.* 1993).

Hart *et al.* (1993) proposed a new approach for protecting ecosystem health, much of which was incorporated into the 1992 Australian Water Quality Guidelines for Marine and Freshwaters (ANZECC, 1992). These guidelines introduced two important new features. First, single "magic" numbers were not provided for key indicators of quality. Instead ranges of reference values were provided for guidance, together with a strong recommendation that site-specific investigations be conducted on each particular ecosystems to define the specific levels to be adopted. Second, for the first time, specific and quantitative biological indicators were introduced, these being species richness, species composition, primary production and ecosystem function. The introduction of these biological indicators into the national water quality guidelines played an important part in the establishment of the AUSRIVAS program - a national river bioassessment program based on macroinvertebrates (NRHP, 1994; Norris *et al.*, 1995; Schofield & Davies, 1996).

Despite these changes, the present Australian water quality guidelines, together with most water quality guidelines in use around the world, still have a number of limitations. First, the water quality indicators used are almost exclusively physico-chemical with few biological indicators recommended. Second, for toxicants, it is assumed that if certain key biological species (those used for bioassays) exhibit no toxic effects, this will be sufficient to protect the community as a whole from toxic effects. Third, these guidelines are based largely on the assumption that if the quality of the water is adequately protected so also will be the whole ecosystem. However, this

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1 Stressors - the physical, chemical or biological factors that can cause an adverse effect on an aquatic ecosystem. Toxic stressors include heavy metals and toxic organic compounds, salinity & pH. Non-toxic stressors include nutrients, turbidity and suspended particulate matter, organic matter, flow & habitat.

latter assumption neglects to consider that other stressors, such as seriously polluted sediments, reduction in stream flow (from impoundments or diversions) or removal of habitat (de-snagging, draining wetlands), could equally cause significant deterioration in the ecosystem.

There is a strong case for a new approach to aquatic ecosystem management if countries are to successfully tackle the task of ecologically sustainable development and the maintenance and improvement of their water resources (inland and marine) through the 1990's and beyond (Hart *et al.*, 1993). The main focus of this new approach must be towards ecologically-based management (Sparks, 1995). In the present context, this would involve extending the concept of water quality guidelines to *ecosystem or environmental guidelines*, where the maintenance of adequate water quality is seen as only one (albeit important) component of protecting the resource. This concept is seen as central to any water quality (environmental) management policy developed for the 1990's, and must be reflected in the water quality guidelines established as a backup to such a policy. Certainly, the present thrust in Australia and New Zealand is towards the development of broader, more ecosystem-related guidelines (ANZECC/AWRC, 1992; Hart *et al.*, 1993; Pyle, 1997; ANZECC, 1998)

In this paper we discuss the key elements of a new risk-based approach for establishing water quality guidelines being developed in Australia and New Zealand, that should lead to more effective management and protection of aquatic ecosystems. The approach has evolved from work being undertaken to update the 1992 ANZECC water quality guidelines (ANZECC, 1998). The essential elements of the new approach, shown diagrammatically in Figure 1, are:

- *ecosystem-based* - ideally the guidelines should be as far as possible ecosystem-specific.
- *issue-based* - the guidelines should focus on the actual issues or problems caused by physical, chemical and biological stressors rather than as present on the individual indicators.
- a *risk-based approach* - there is great difficulty in deciding whether adverse biological effects will result from various stressors added to an ecosystem. The new approach develops guideline "packages" for each issue, and where possible for each ecosystem type. Each "package" consists of specified key performance indicators, trigger levels for these indicators (i.e. levels which indicate the risk that adverse biological effects may occur), and for high risk situations (where trigger levels are exceeded) a protocol for considering the effect of ecosystem-specific factors in reducing (or enhancing) the biological effects.

Ecosystem-specific Guidelines

Ecosystem type

In this section we develop two key themes essential in managing ecosystems; first, that the effective management of ecosystems requires a good understanding of how that system works, and second, that there are many different types of aquatic ecosystems which can each function quite differently making it desirable that ecosystem-specific management guidelines are developed where this is possible.

Aquatic ecosystems are characterised by great variability and complexity, and the fact that they are now increasingly impacted by human activities occurring within the catchment. Aquatic ecosystems, are largely structured by the climatic regime and physical (e.g. light, temperature, mixing, flow, habitat) and chemical (e.g. organic and inorganic carbon, oxygen, nutrients) environment with which they interact, and the biological interactions (e.g. grazing & predation) that occur within them. Variations in these physical and chemical factors can occur naturally due to droughts and floods, climatic conditions and erosion events, and these variables can have important consequences for the numbers and types of biota present at any one time. Climate variations, and consequent variations in rainfall, runoff and river flow, are particular marked in Australia (Finlayson & McMahon, 1988; Harris & Baxter, 1996; Harris, 1996), these being strongly linked to interannual climate variability through mechanisms such as the El Nino Southern Oscillation or ENSO (Simpson *et al.*, 1993). ENSO events influence fluctuations in sea surface temperature in the Pacific Ocean and large scale changes in the position of the continental high over both Australia and New Zealand (Harris & Baxter, 1996). Therefore, the natural variability in Australian and New Zealand aquatic ecosystems, which may be greater than elsewhere in the world due to the huge influence of ENSO effects, must be taken into account when developing ecosystem protection management plans.

Aquatic ecosystems are also characterised by a large number of quite complex interactions (Harris, 1994a, b). As indicated above, many physical, chemical and biological processes can influence the functioning of aquatic ecosystems. At the broad scale, the relationships between changes in the physico-chemical factors and the consequent changes in the biota are reasonably well known. For example, it is well known that increased nutrient concentrations can result in increased phytoplankton numbers, while increased turbidity can reduce phytoplankton numbers because of reduced light penetration into the waterbody. However, at the more detailed scale of the individual ecosystem, such "simple" relationships are rarely found because of the number and complexity of the interactions occurring at the ecosystem level, all of which may have an influence on the dominant processes at any one time.

Additionally, a wide range of human-related stressors can impact upon aquatic ecosystems, and modify their "health". These include: pollution from industrial, urban, agricultural and mining sources; regulation of rivers through the construction of dams and weirs; salinisation; siltation and sedimentation from land clearance, forestry and road building; clearance of stream bank vegetation; over-exploitation of fisheries resources; introduction of alien plant and animal species; removal and destruction of habitat, to name but a few.

All too often, the presently available guidelines simply lump ecosystems into two categories - "freshwater" and "marine" (ANZECC, 1992; USEPA, 1986). This is obviously insufficient to discriminate the differences between the ecosystem types that exist in many countries. Australia, for example, lies between latitudes 10^o 41' S and 43^o 39' S, which results in a wide range of climatic and geographic variations and a consequent diverse range of ecosystem types, including tropical, temperate, arid, alpine and lowland aquatic ecosystems. Further, within these broad geographic types there exist waterbodies that are static, flowing or ephemeral, deep or shallow, and fresh, brackish or saline.

Obviously, the biotic communities and the ecosystem functioning within this wide range of ecosystem types will differ, sometimes markedly. Thus, it is difficult to see how effective management can occur without some further discrimination between ecosystem types than the simple "freshwater" and "marine". The new Australian and New Zealand Guidelines recommend seven ecosystem types: upland rivers, lowland rivers, lakes and reservoirs, wetlands, estuaries, coastal and marine (ANZECC, 1998). However, even with this admittedly limited number of ecosystem types there is often a lack of knowledge on what lives in them and particularly on how they function. This lack of knowledge has precluded a further segmentation of these ecosystem-types on the basis of geography (e.g. tropical vs temperate, coastal vs inland), although, hopefully, this will come in the near future.

A good scientific knowledge of the ecosystem and its catchment is essential, not only to permit the differentiation between ecosystem-types, but also as a prerequisite to their effective management. For example, assessment of the biological effects of particular stressors involves having information on their sources, how they are transported through the catchment to the water body, how they behave within the aquatic system, and how they interact with the biota.

Over the past 10 years in Australia, and in many other developed countries, there has been increased focus on aquatic ecosystem research, with consequent advances in our knowledge of these systems (Cullen *et al.*, 1996). However, there still remains much to know, particularly about tropical ecosystems, many of which exist in less well developed parts of the world (Lewis, 1987).

Management objectives

Perhaps the most important part of any management plan is a clear statement of the objectives or targets to be achieved. This is equally true for water quality or ecosystem management plans. However, all too often such plans have the targets stated in very general terms.

Additionally, the targets or objectives specifically aimed at protecting an ecosystem should be set in terms of ecosystem-specific indicators. At present this is largely restricted to the biological components of the ecosystem (e.g. reduction in biodiversity and/or abundance caused by toxicants, or changes in species composition and/or abundance caused by excessive nutrients), but with time may also include measures of ecosystem functioning (e.g. gross primary production and community respiration).

The present Australian water quality guidelines recommended that the objective for the protection of aquatic ecosystems should be *".. to protect biological diversity (biodiversity) and maintain ecological processes and systems."* (ANZECC, 1992). However, in setting the management targets for an ecosystem, the first question the manager must answer is what type of ecosystem is desired, i.e. what level of protection needs to be provided for the ecosystem. Many levels of ecosystem protection could be defined, but we suggest three are relevant, these being:

- *High conservation/ecological value systems* - these are pristine or other highly valued ecosystems, typically occurring in national parks, conservation reserves or in remote and/or inaccessible locations.
- *Substantially natural systems* - those ecosystems in which water quality and aquatic biota, by every indication, have not been adversely affected to any significant degree by human activities.
- *Modified systems* - these are ecosystems which are measurably different from the equivalent natural ecosystem.

The provision of the highest level of protection for pristine or near pristine ecosystems in a national park is obvious. However, for a significantly modified urban creek it is unlikely even with the best will in the world (and an appropriate bank account), that this could be rehabilitated to a near pristine system. In particular, the flow regimes in most urban systems have been permanently changed. To ensure that there is some balance and practicality in the targets set, it is therefore essential that decisions on the level of protection and the targets to achieve this level are negotiated between the stakeholders, who may include the community, management agencies and dischargers. In Victoria, the EPA achieves this balance through preliminary community consultation followed by public display of the State Environmental Protection Policy for that waterbody (e.g. Vic EPA, 1995).

It is not yet possible to specify with any degree of certainty just what constitutes a "healthy" or "acceptable" ecosystem. Of course, this is partially a scientific question and partially one related to what the community considers is acceptable. Agreement on an acceptable level of change for a particular ecosystem, even if only defined statistically (see below), between the various stakeholders, is essential in developing a successful ecosystem management plan.

Measurement of "acceptable" ecological change is extremely difficult (Keogh & Mapstone, 1995; Mapstone, 1995). In very few situations is the scientific knowledge sufficient for us to gauge with any certainty what change from the target condition will cause an adverse ecological effect. Both the time or duration of the change and the absolute level of change can be important. For example, an increase in toxicant concentration over a very short time period can cause a significant reduction in the biological diversity, while the deposition of particulate matter or silt on the bottom of a small stream to levels that cause problems may occur over a considerable time period. For these reasons, there are very few examples where the level of change from some prescribed target condition have been specified.

However, given an adequate data set, it is certainly possible to define a particular level of change in statistical terms (e.g. probability & power (Keogh & Mapstone, 1995; Mapstone, 1995)), although it must be recognised that a defined statistical change does not necessarily equate with any particular ecological change. In defining the trigger levels required for the risk-based guidelines (see below), we have assumed (somewhat arbitrarily) that the range of concentrations for each stressor which are representative of a well functioning unmodified aquatic ecosystem is given by the range between the 20th and 80th percentiles of the data distribution. Others (e.g. Kilgour et al. (1998) and references therein) have also defined a "normal" range on the basis of reference sites, but suggest the range should include 95% of the population.

Ecosystem issues

Present water quality guidelines focus almost exclusively on the individual indicators (or stressors); for example, on nutrients, turbidity or particular toxicants such as copper. However, it is generally the issues or problems caused by physical, chemical and biological stressors that need to be tackled by management agencies, and these rarely arise because of only one stressor. Therefore, we believe it more appropriate to focus on the issues rather than on single indicators, and as will be discussed later in this paper, such an issue-based focus requires that the guidelines be organised in terms of information provided on "packages" of indicators which relate to each particular issue.

Ecosystem management issues for which guideline "packages" are being developed in the new Australia and New Zealand guidelines include (ANZECC, 1998):

- effects due to toxicants in the water column (heavy metals, toxic organic compounds)
- effects due to toxicants in sediments
- nuisance growths of aquatic plants
- maintenance of dissolved oxygen
- effects due to suspended particulate matter
- effects due to salinity changes
- effects due to temperature changes
- effects due to pH changes
- ehanges in optical properties of waterbodies
- effects due to changes in flow (for rivers and wetlands).

Later in this paper, a case study addressing the development of quality guidelines for nuisance growths of aquatic plants is presented.

Risk-based management

The effect of a particular stressor on the biological diversity and abundance[2] depends upon three major factors:

- the type of ecosystem, and hence the biological communities;
- the type of stressors and the issues (or problems) these cause;
- the influence of environmental factors (which may also be stressors) which may modify the effect of the stressor.

Existing water quality guidelines do not adequately address either the *variability and complexity* known to characterise all aquatic ecosystems, or the influence of *environmental factors* in modifying the ecological effects of key stressors. In fact, variability and complexity are the factors that make it extremely difficult to effectively manage aquatic ecosystems.

Here we outline a new risk-based approach to the development of water quality guidelines, which by implicitly accounting for the variability and complexity, should provide a more realistic and effective means of protecting the biodiversity or ecological integrity of aquatic ecosystems. This risk-based approach is based on the ecological risk assessment (ERA) methodology, a process for determining the level of risk posed by stressors (e.g. chemicals, nutrients) to the survival and health of aquatic ecosystems. The ERA process is intended to take explicit account of the uncertainty provided by the inherent variability in natural aquatic systems and the large number of different species that make up an ecosystem (Suter, 1993; USEPA, 1995). Ideally, ERA's should be based on complete data sets for the intensity, frequency and duration of stressor effects. More usually, however, lack of information makes necessary a balance between available scientific data and professional knowledge and judgement.

The ecological risk assessment process has evolved because of difficulties in assessing the impact of multiple stressors on complex ecosystems. Initially, the focus was on the risks associated with the effects of multiple chemicals, but there is a growing realisation that degradation of catchments and waterways is also related to physical and biological stressors, in addition to the chemical stressors. Risk assessment involving ecosystems is particularly challenging because of the large number of different species involved and the difficulties in

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2 Broadly, the effects on the biological diversity and abundance are:

- *reduction in biodiversity and/or abundance* - due to toxicants such as heavy metals, pesticides or salinity;
- *changes in species composition and/or abundance* - particularly towards nuisance populations caused by excess nutrients or a lack of light (e.g. caused by increased turbidity).

deciding what "end points" or targets are to be used to assess whether adverse effects have occurred. Endpoints such as "ecosystem integrity" or "maintenance of ecological health" must be further quantified in terms of biological indicators (e.g. macroinvertebrate O/E ratios (Barmuta *et al.*, 1997), algal biomass) if they are to be useful.

Ecological risk assessment includes a consideration of both the severity (or hazard) and frequency (or exposure) of the problem. For example, a situation where an extremely toxic chemical (e.g. mercury) is effectively contained so there is no exposure to the ecosystem, represents a low-risk. On the other hand, however, a less hazardous material (e.g. orthophosphate) which is released in large quantities into the environment can result in a high-risk situation if toxic cyanobacterial blooms occur.

Although ERA's are now being used extensively in Europe (Calow, 1995) and North America (Renner, 1996; USEPA, 1995), it is only very recently that the method has been used in Australia, despite the fact that the broader area of risk assessment has been actively promoted by the Federal EPA (Beer & Ziolkowski, 1995). Recent examples of ERA's undertaken in Australia include an ecological and human health risk assessment of chemicals in sewage treatment plant discharges to the Hawkesbury-Nepean river system (Sydney Water, 1996), and the development of an ERA protocol for determining the risk of cyanobacterial blooms in lowland rivers in northern Victoria due to irrigation return drainage (Hart *et al.*, 1998a; SKM, 1997).

Trigger levels for key performance indicators

To help decide the likely severity of a stressor on an ecosystem, we introduce the concept of a low-risk "trigger" level for each issue. These trigger levels are defined as the concentrations (or loads) of the key performance indicators for the ecosystem type being managed, below which there exists a low risk that adverse biological (ecological) effects will occur. They are the levels that "trigger" the need for continued monitoring in the case of low risk situations, or further ecosystem-specific investigations in the case of high risk situations. We expect that trigger levels will be both ecosystem-specific and issue-specific.

Ideally, low risk trigger levels would be established in terms of the “bioavailable” concentration of each stressor. Unfortunately, the bioavailable concentration is rarely available, and the “total” stressor concentration is used as a conservative default (ANZECC, 1992, 1998). However, there is considerable effort around the world to develop analytical methods that will provide a better estimate of the bioavailable concentration. This work is probably most advanced for measurement of phosphorus (Hart et al. 1998b; Shalders et al., 1998).

Four methods are available for obtaining low-risk trigger levels for the designated performance indicators. These are:

(a) *biological effects data* - obtained either from ecotoxicity testing using local biota and local waters, or from the scientific literature. This method is most appropriate for toxicants (e.g. heavy metals, toxic organic compounds), but less appropriate for naturally-occurring stressors such as nutrients (Chapman, 1995; ANZECC, 1998).

(b) *reference system data* - obtained either from the same (undisturbed) ecosystem (i.e. from upstream of possible impacts) or from local (but different system) or regional reference ecosystems. This approach is particularly useful for aquatic ecosystems where the management target is to maintain (or restore) the system in an essentially natural or unmodified condition, and where there are sufficient resources to obtain the required information on the reference ecosystem (Reynoldson et al., 1997). This method takes account of the natural variability of the key indicators in the reference system.

We have assumed (somewhat arbitrarily) that the range of concentrations for each stressor representative of a well functioning unmodified ecosystem is given by the range between the 20th and 80th percentile of the data distribution. For stressors where high concentrations cause problems (e.g. nutrients, turbidity, BOD, salinity), the low-risk trigger level is taken as the 80th percentile of the reference distribution. For stressors where low concentrations cause problems (e.g. low temperature water releases from reservoirs, low salinity in estuaries, low dissolved oxygen in waterbodies), the 20th percentile of the reference distribution is taken as the low-risk trigger level. For stressors where both high and low levels can result in problems (e.g. temperature, salinity, pH), the desired range is defined by the 20th and 80th percentile of the reference distribution.

The choice of the 80^{th} percentile and 20th percentile cutoffs to represent a well functioning, unmodified ecosystem is arbitrary. There is currently no consensus on how best to define the influence of variations of physical and chemical stressors on the ecological functioning of an aquatic system (Keogh & Mapstone, 1995; Mapstone, 1995). For example, Kilgour et al. (1998) suggests the “normal” range as the region enclosing 95% of the reference site data.

(c) *professional judgement* - in some cases it will not be possible to obtain appropriate data for a reference ecosystem, either because no appropriate reference system exists[3] or insufficient study has been undertaken to provide an adequate data base. In these cases professional judgement may need to be used. This should be backed by appropriate scientific information, such as that available in existing water quality guideline documents (Hart, 1974; Hart, 1982; USEPA, 1986; CCREM, 1991; ANZECC, 1992).

(d) *predictive modelling* - is also available to obtain the trigger levels for certain physical and chemical stressors, particularly those whose impact occurs through transformations in the environment (e.g. nutrients, biodegradable organic matter). In these cases, because of the number of other factors also involved, there does not appear to be a direct relationship

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3 For example, there are few cases of lowland river systems in Australia that are sufficiently unmodified to be considered as a reference. Equally, the selection of an appropriate reference for most urban river systems is difficult.

between the ambient concentration of the stressor (e.g. total P concentration) and the biological response (e.g. algal biomass). However, there is often a plausible relationship between loading (or flux) and the biological response. The use of load-based predictive modelling can also be very useful in defining trigger levels for ecosystems where no relevant (unmodified) reference systems exist.

Low-risk trigger levels may be derived for each issue using the following hierarchy, where both the level of protection and the availability of ecosystems information are taken into account.

1. Substantially natural ecosystems

Here the management target is to maintain (or restore) the ecosystem in a substantially natural condition, and depending upon the significance and present condition of the ecosystem, two approaches may be taken to derive the most appropriate trigger levels.

(a) Special studies, reference ecosystem - High priority should be given to identifying an appropriate reference system (or systems), undertaking the necessary scientific studies to obtain the required information, and then using this information to derive the low-risk trigger levels. This approach will be relevant in those cases where an appropriate reference system(s) is available and there are sufficient resources to collect the necessary information for the reference system.

(b) Interim trigger levels - interim trigger levels may be used for those situations where either an appropriate reference system is not available, or the scale of the operation makes it difficult to justify the allocation of resources to collect the necessary information on a reference system. For the new Australian and New Zealand guidelines, interim trigger levels have been derived using data collected on unmodified ecosystems in the two countries (ANZECC, 1998). However, for a number of ecosystem types, it was either very difficult or impossible to obtain sufficient high quality data on unmodified systems to be able to use this data to derive interim trigger levels. In these cases professional judgement was used to derive the interim trigger level.

2. Modified ecosystems

For many aquatic ecosystems the management target will be to maintain (or restore) a modified ecosystem, the possibility of achieving a substantially natural system not being a realistic option. Urban aquatic systems (rivers, streams, wetlands, estuaries) are a case in point. For most of these, the hydrology in particular has been so markedly changed that at best a somewhat modified ecosystem can be achieved. The actual biological targets for each modified ecosystem will need to be decided by the management agency in consultation with the various stakeholders (e.g. Vic EPA, 1995). Again, depending upon the significance and present condition of the ecosystem, two approaches may be taken to derive the most appropriate trigger levels:

(a) Special studies and professional judgement - For significant waterbodies (e.g. Yarra River-Port Phillip Bay, Hawkesbury-Nepean system, Brisbane River-Moreton Bay), and those in

very poor condition, we recommend that appropriate site-specific scientific studies be undertaken, and the information from these studies be used together with professional judgement and other relevant information, to derive the trigger levels.

(b)

(b) Modified interim trigger levels and professional judgement - where there is either a lack of information or insufficient resources to undertake the necessary site-specific studies, we recommend that the unmodified reference ecosystem data set for the particular ecosystem type be used, but with a less stringent cutoff than the 80 percentile value, together with professional judgement to derive the appropriate trigger levels.

"Packages" of information

Guideline "packages" have been developed for each ecosystem type and each ecosystem issue. Each guideline package consists of two components (see Figure 1):

- *key performance indicators* - (e.g. total phosphorus concentration) are used to make an initial decision on the risk (high or low) that an adverse biological effect will occur in the particular ecosystem type. The low-risk trigger levels for these key performance indicators (mostly also key stressors) were established as outlined above. At present the trigger levels are all concentration-based, but we are developing a protocol that should result in load-based guidelines being developed for those cases where these are more relevant.
- *A protocol for further investigating the risk in those cases where the trigger level is exceeded* - for potentially high-risk situations, ecosystem-specific modifying factors that may alter the biological effect of the key stressor need to be considered before the final risk can be decided. Depending upon the situation, this protocol may involve the following steps:
  - (a) undertake a simple assessment of the possible effect of key ecosystem-specific modifiers. A simple "decision tree" model for undertaking this type of first-level assessment is provided in the case study below.
  - (b) if this simple assessment still suggests a high risk of adverse biological effects occurring, then further more sophisticated site-specific investigations and perhaps associated modelling may need to be undertaken. For example, a recent study of eutrophication of Port Phillip Bay, Australia, lead to the development of a comprehensive ecosystem-based model of the system (see Harris *et al.*, 1996).

Unfortunately, at present, there are only a small number of cases where sufficient scientific information is available to enable the required quantitative relationships between the key stressor and the environmental factors controlling the bioavailability of this stressor to be developed.

Case Study - Nuisance aquatic plant growth

Issue

The ecosystem issue considered in this case study is the excessive growth of nuisance algal species, an increasingly important problem in Australia (SoE, 1996). High concentrations of nutrients, particularly phosphorus and nitrogen (sometimes silica), can result in excessive growth of aquatic plants, such as phytoplankton, cyanobacteria, macrophytes, seagrasses, and filamentous and attached algae, in most ecosystems. These excessive growths can lead to a number of problems including: toxic effects, particularly due to cyanobacteria in fresh and brackish waters, and dinoflagellates in marine waters; reduction in dissolved oxygen concentrations when the plants die and are decomposed; reduction in recreational amenity (phytoplankton blooms & macrophytes in wetlands and lakes, seagrasses in estuaries and coastal lagoons); blocking of waterways and standing waterbodies (macrophytes); and changes in biodiversity as the species composition is changed.

High concentrations and loads of nutrients are necessary for excessive growth of aquatic plants. But, many other factors can also play a part in limiting the growth of nuisance species, in particular toxic cyanobacteria, including hydraulic retention time, mixing conditions, light availability (turbidity), temperature, suspended solids (nutrient sorption), grazing pressure and type of substrate (Harris, 1994a).

Targets

For this issue, the targets would be set in terms of either chlorophyll-a concentration, cell numbers (particularly of cyanobacteria) or species composition. The key stressors are assumed to be the nutrients phosphorus and nitrogen, and the potential ecosystem factors that could modify the biological effects of these nutrients would include hydraulic retention time (flows & waterbody volume), mixing regimes, light regime, turbidity, temperature, suspended solids (nutrient sorption), estimates of grazing rates, and type of substrate.

Nutrients may also become available as a result of remobilisation from sediments, where nutrient release is influenced by the composition of the sediments (particularly bioavailable organic matter, Fe, S, N, P), temperature, mixing regime of the water body and oxygen transfer rates (Bostrom *et al.*, 1988). At present, it is not possible to recommend quantitative relationships to estimate these releases, although such relationships should become available in the next few years (Lijklema, 1993).

Low-risk trigger levels

If sufficient information is available, the low-risk trigger concentrations for the three key performance indicators (TP, TN and Chl a) should be determined from the reference system. The reference ecosystem may be: the same system, but upstream of a major effluent discharge; nearby, similar but unmodified ecosystems; and an ecosystem(s) from the region that is the desired target.

Table 1 provides interim trigger levels for six ecosystem types obtained using data from within Australia. The low-risk trigger levels were taken as the 80th percentile of the data distributions for unmodified ecosystems. Where these data were not available, professional judgement was used to derive an appropriate trigger level.

Use of the guideline package

The recommended approach to determining the risk of nuisance aquatic plant growth occurring in a particular ecosystem is shown in Figure 1. The approach involves three steps:

- Test the three performance indicators (Chl a, TP, TN concentrations) for the particular ecosystem against the appropriate low-risk trigger level for that ecosytem type (Table 1).
- If the test values are less than the trigger levels, there is low risk that adverse biological effects will occur and no further action is required (except for regular monitoring).
- However, if the test values are higher than the trigger levels, there is an increased risk that adverse biological effects will occur, and further ecosystem-specific investigation is required.

For situations were the problem is so serious or the potential is high, it is possible to develop complex (and quite expensive) models; for example, Port Phillip Bay (Harris *et al.*, 1996), Hawkesbury-Nepean river (Sydney Water, 1995), and the coastal waters off Perth (DEP, 1996).

Decision tree

However, for the majority of situations where this type of funding is not available, less expensive options are needed. One such option is a simple "decision tree" approach. Below we provide an example of a decision tree developed as part of an environmental audit protocol for Goulburn-Murray Water to determine the risk of cyanobacterial blooms occurring in lowland rivers due to nutrients added by irrigation drains (Figure 2; Hart *et al.*, 1998a; SKM, 1997).

The conceptual model for this case study assumed that cyanobacterial growth in lowland rivers is controlled by three major factors:

- the concentrations of the nutrients P and N;
- the light climate (turbidity used as a surrogate because of a lack of data on light);
- the flow conditions in the river when cyanobacterial growth can occur.

The "guideline package" in this case includes the nutrient concentrations as the key stressors, and turbidity and flow flow as the modifiers. The numbers provided in the decision boxes for TP, TN and turbidity should be taken as indicative only as they still require some refinement, and may be a little different depending upon the particular ecosystem being considered.

The decision box for flow was based on the need for a sufficient period of low flow to allow cyanobacterial numbers to increase to an alert level of 5,000 cells/mL. A period of 6-10 days was estimated based on a cyanobacterial doubling time of 2 days and an initial cyanobacterial concentration of 10-100 cells/mL. A "growth event" was then defined as a period consisting of at least 6 consecutive days when the flow was less than the 25th percentile flow obtained from the long term flow record for the system.

For the system described in Figure 2, a high risk situation is indicated if the TP concentration is >15 ug/L, the turbidity less than 30 NTU and there is more than one "growth event" of >6 days duration per year. In this case, further investigation and appropriate management actions would be warranted.

Further refinement of this simple model could include:

- determining a more quantitative relationship between turbidity and the light climate for algal growth;
- validation of the assumption that the <25th percentile flows are the most appropriate low flow conditions to use. The present simple protocol does not take any consideration of stratification that is now known to have a significant influence on cyanobacterial growth in lowland rivers (Webster *et al.*, 1996);
- introduction of measures of the "bioavailable" fractions of the nutrients rather than TP and TN (Hart *et al.*, 1998b);
- including the possibility that sediment release of nutrients (particularly phosphorus) may occur under low flow conditions;
- incorporation of the various decision "rules" into a user-friendly computer program for ease of use by managers.

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Table 1: Interim trigger levels for assessing risk of adverse effects due to nutrients in different ecosystem types. Level of protection to maintain a substantially natural ecosystem. Trigger levels are 80 percentiles of distributions obtained using data from unmodified ecosystems.

Ecosystem type	**Chl a** (ug/L)	**TP** (ug/L)	**TN** (ug/L)
Lowland river	2 *	37	1600
Upland river	**ND ****	35	340
Freshwater lakes & reservoirs	9	50	440
Wetlands	**ND**	**ND**	**ND**
Estuaries	2	45**	80***
Coastal & marine	0.3	55**	350***

ND = no data
* Professional judgement used to obtain values
** FRP (estuaries) - 4 ugP/L; FRP (coastal & marine) - 6 ugP/L
*** NO_X (estuaries) - 5 ugN/L; NO_X (coastal & marine) - 60 ugN/L
NH_4 (estuaries) - 20 ugN/L; NH_4 (coastal & marine) - 40 ugN/L

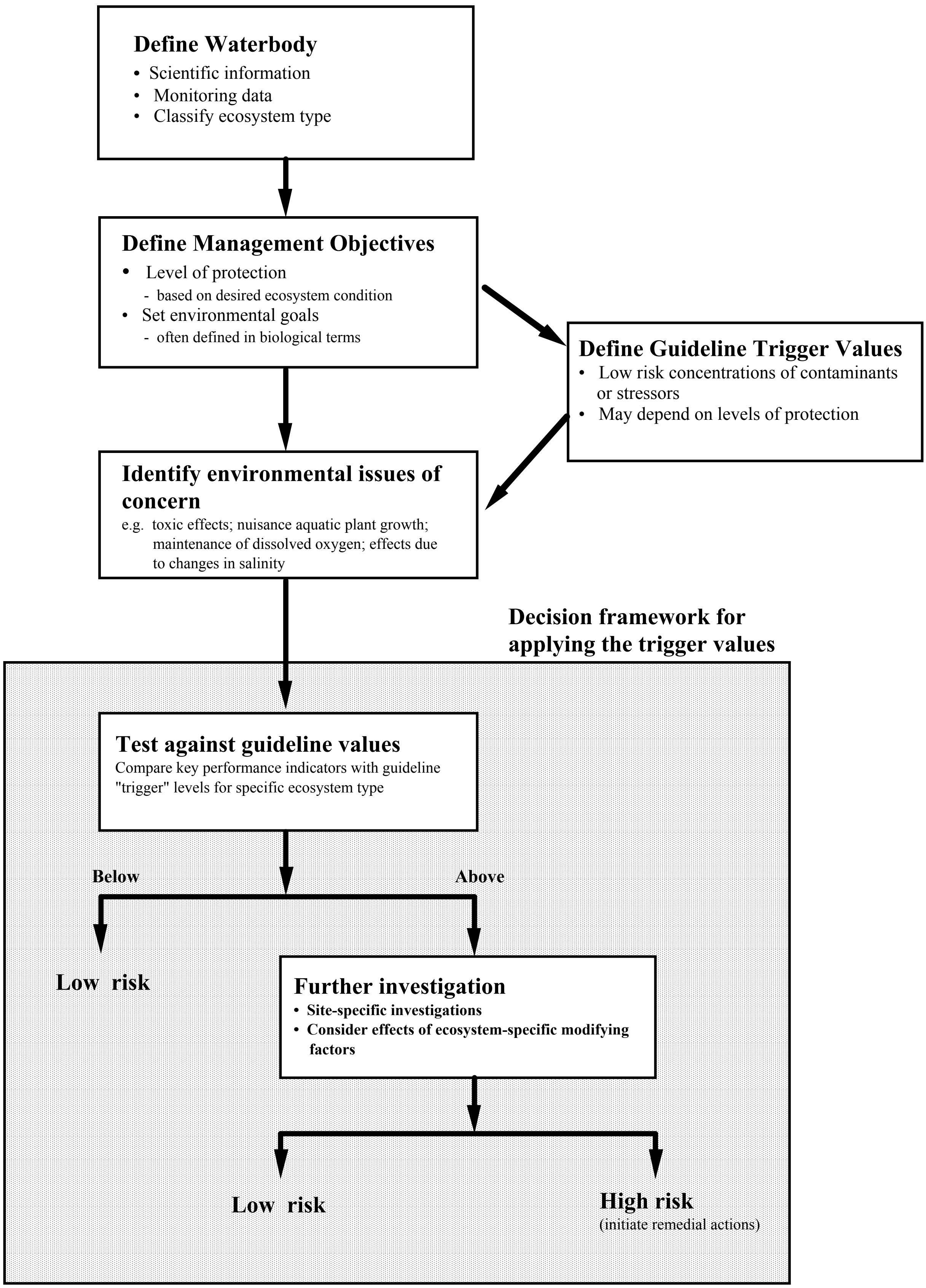

Figure 1: Flow chart showing the steps involved in using the new guidelines for aquatic ecosystem protection

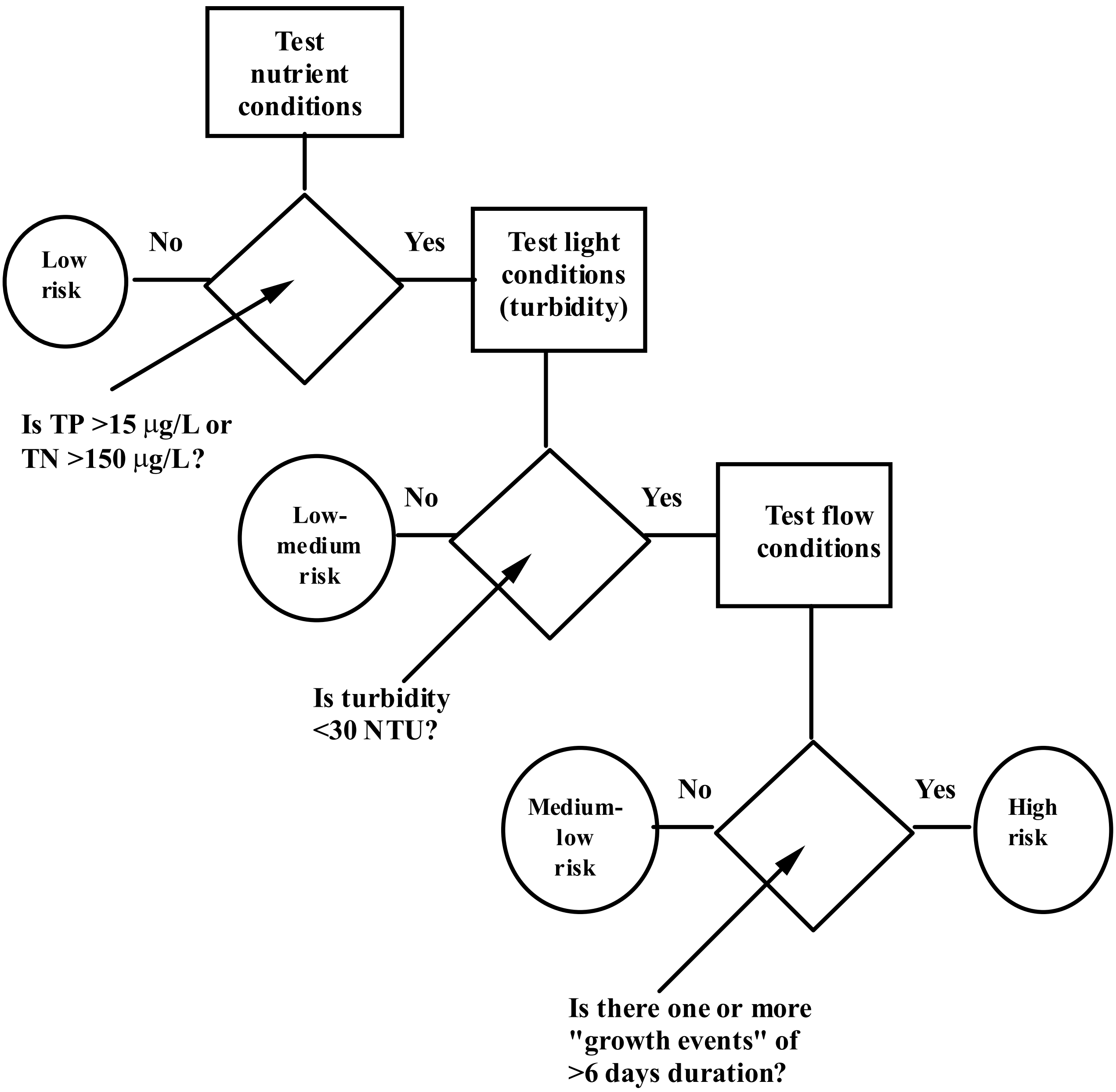

Figure 2: Decision tree for assessing the risk of cyanobacterial blooms in lowland rivers caused by irrigation return drainage.

The conceptual model used assumes cyanobacterial growth is controlled by: nutrients (TP - total phosphorus; TN - total nitrogen), light climate (turbidity used as a surrogate) and flow conditions in the river.