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Integrated estuary assessment framework

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1. Introduction

1.1. Purpose of report

In 2002, the Cooperative Research Centre for Coastal Zone, Estuary and Waterway Management (Coastal CRC) established a research area termed Natural Resource Governance and Partnerships, consisting of a suite of related research projects. One of these, ‘Assessing estuary condition and values: setting management priorities’, aimed to develop a framework that could be used to assess the biophysical health of estuarine and coastal systems but which would also link into the social and economic values of such systems. An important part of the research was to use the framework to identify appropriate indicators of both ecosystem and economic and social values.

This report describes the work undertaken to achieve these goals. For various reasons, work on the social and economic indicators was very limited, and application of the framework has therefore focussed on selection of appropriate biophysical indicators. However, the report does demonstrate how social values fit into the framework and how biophysical indicators link to social indicators.

2. Development of an integrated estuary assessment framework

2.1. Aims

The main aim of this work was to develop an integrated reporting framework for estuary and coastal systems that could be used to derive management actions and priorities. Specifically, the framework was to assess and report on the risk to and the condition of estuarine systems and to combine this with information on the values of systems to derive management priorities. These aims and an outline of how the framework would be developed are shown in Figure 1.

An important part of the work involved the development of indicators. However, indicators cannot be developed in isolation—they need to have a context and purpose. Many indicator documents can be criticised on the basis that they recommend long lists of indicators that have only very general contexts or purposes. The integrated estuary assessment framework (IEAF) developed here, on the other hand, provides a well defined context and purpose for development of indicators. Details of how the framework is used to derive indicators are given in Section 5 of this report and a full set of information sheets that demonstrate application of the framework is presented in the Appendix.

The reporting framework should ideally have the following characteristics:

- It should include a method for selection of locally relevant indicators, based on local issues (pressures) and local system and habitat types.
- It should be based on a pressure–state–impact–response (PSIR) model, where *pressures* are human impacts on the system, *state* is the current condition of the system, *impacts* are the effects of changes in the state (including impacts on the ecosystem and on the human population), and *response* describes the implemented management responses to local issues. Clear linkages between these indicators will make it easier to determine which management actions are appropriate for a given problem.
- There must be clear links between indicators at each stage of the PSIR model; a change in a pressure indicator should result in a change in a state indicator, etc. These links should be quantified wherever possible.

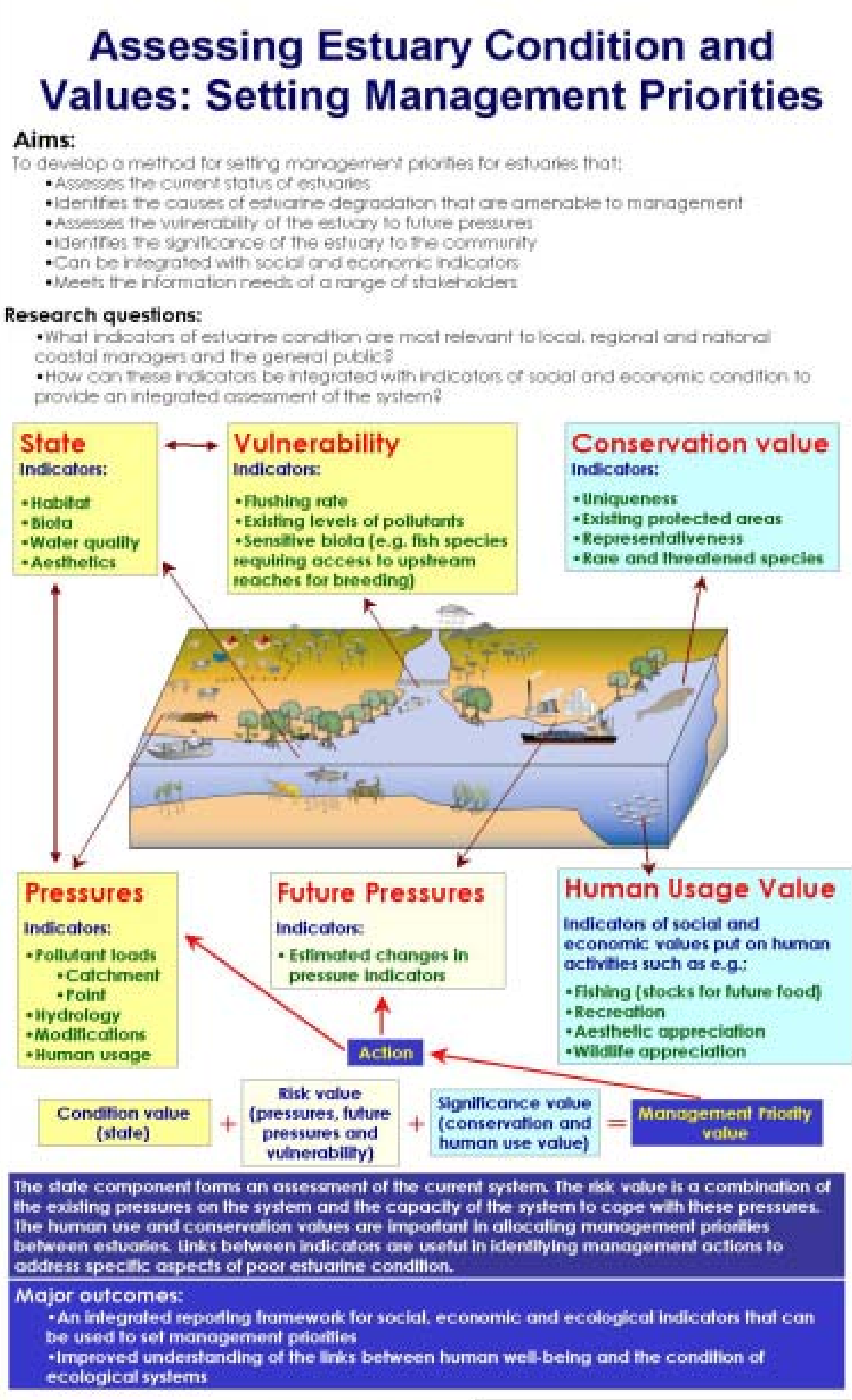

Figure 1. Aims and outline of integrated estuary assessment framework research activities

- The risk to the system based on pressures and the existing features of the system (such as the hydrodynamics, tidal range, existing habitat or animal populations) should be assessed, to assist in setting management priorities (both *within systems*—i.e. between responses to pressures—and *between systems*).
- Outputs of the reporting framework should include a condition assessment (which may be broken down into separate system components), pressure assessment (incorporating measures of risk or vulnerability to future pressures), recommended management actions, and management priorities (based on impacts, values, current condition and vulnerability).

Although not all the above characteristics were achieved, this list served as a guide throughout development of the framework.

2.2. Literature review of existing indicator frameworks

One of the initial models considered for this project was the Index of Stream Condition framework (Ladson *et al.* 1999). This was developed for, and has been applied to, freshwater streams in Victoria. It was designed to provide assessment of each of five key components of stream condition, namely water quality, habitat (streamside and instream), flow and biota. For each component a set of indicators (between one and five) was developed, based largely on expert opinion. These indicators are scored individually and then rolled up scores are used to rate each component. The relative scores for each component allow some prioritisation of management activity; for example, if flow is the most impacted then this would indicate a management priority. Adaptation of this type of index system to estuarine areas was considered but the absence of specific links between pressures and condition was seen as a disadvantage.

There have been a number of condition indexes developed specifically for estuaries. One of the early ones was developed for South African estuaries (Cooper *et al.* 1994) and involved consideration of three components (water quality, biota and aesthetics), each based on a set of indicators. In the United States, there have been a number of different condition indexes developed based on a range of different components (e.g. Kiddon *et al.* 2003; Paul *et al.* 1998). Many of the individual indicators were common to different systems but were sometimes grouped under different component headings. Both the South African and U.S. systems were focussed on condition and did not take into account the pressures on the systems. Similarly, selection of indicators was done on the basis of expert opinion rather than through an objective framework.

Ferreira (2000) developed a more sophisticated assessment system. This involved three condition components (water quality, benthic quality and trophodynamics) but also included a measure of vulnerability (defined as system buffering capacity) for each estuary. This is an important addition, allowing the condition of an estuary to be normalised against its natural features, particularly flushing rates. This concept of vulnerability was incorporated into the framework developed in the IEAF project.

While Ferreira’s system has some useful features, it is still basically a condition assessment system that takes little account of pressures on estuaries. Ferreira also clearly states that his methodology is not designed for detailed management of a particular system, which needs a completely different approach, focussing on specific problems and potential solutions (Ferreira 2000). The Ferreira system therefore has limited use in guiding management actions for individual estuaries. This comment could be equally applied to most of the other condition assessment systems in the literature. For the IEAF project, the aim was to have a framework that could be used for assessing individual estuaries and guiding local management actions.

Deeley and Paling (1999) discuss in detail a wide range of indicators for assessing estuary health in Australia but, again, do not put them into the context of an assessment framework.

The assessment of Australian estuaries conducted by the National Land and Water Resources Audit (NLWRA 2002) provides an invaluable information source for catchment and waterway management at a national or state level. However, the fact that it was developed specifically as a national assessment tool means that its application to smaller spatial scales is necessarily limited. The choice of indicators included in the audit process was limited to those that were available for a large number of estuaries. The indicators included may therefore not be those that are most relevant to specific waterways or managers at a local scale. The assessment was also hampered by a lack of data for many potential estuarine indicators. Although the assessment framework did include both pressure and state indicators, there was no explicit attempt to link the two. Explicit links between pressure and state indicators are essential, however, if changes in state indicators are to be interpreted in such a way as to improve management actions. The report indicates that further development of an ‘index of estuary condition’ building on the Audit’s assessment process, together with long-term monitoring of changes in this index, would be a useful management tool (NLWRA 2002).

Australia's National Strategy for Ecologically Sustainable Development, endorsed in 1992, calls for regular national state of environment (SoE) reporting. The aims of SoE reporting include to: provide accurate, timely and accessible information about the condition and prospects for the Australian environment; develop an agreed set of national environmental indicators; provide early warning of problems; report on the effectiveness of environmental policies; contribute to assessing progress towards ecological sustainability, protecting ecosystems and maintaining ecological processes and systems; integrate environmental information with social and economic information; identify knowledge gaps; and improve decision making through information (Ward *et al.*1998).

The reporting framework for SoE is based on the pressure–state–response model. Reporting is also based on eight groups of indicators: protected and cited species/taxa; habitat extent; habitat quality; renewable products; non-renewable products; water/sediment quality; integrated management; and ecosystem level processes. Within each of these groups, indicators are identified as relating to pressure, state or response. Although there is some information given on the linkages between the indicators, there is no guidance as to how to relate changes in individual indicators to other indicators in order to facilitate appropriate management action.

The SoE reporting framework recognises that knowledge of Australia's estuaries and marine environments is limited, particularly knowledge of structure and function. This lack of knowledge means that decisions about matching indicators to issues are risky, and that risk-management procedures need to be adopted in implementing the reporting program to confirm that indicators are related to sustainability issues. There is also recognition that credible cause-effect models need to be used as the basis for the design of monitoring programs.

The IEAF project builds on the pressure–state–response (PSR) model used in SoE but attempts to better quantify the linkages between pressure and state.

Bidone and Lacerda (2004) developed a driver–pressure–state–impact–response (DPSIR) framework to evaluate sustainability in a Brazilian coastal bay. This is essentially a more detailed version of the PSR model. The DPSIR framework includes measures of socioeconomic and physical drivers, physico-chemical pressures in the catchment, physico-chemical and biological state in the coastal zone, socioeconomic impacts, and management responses. This is an excellent example of incorporating biophysical and socioeconomic indicators into one assessment. There is some information provided on how to link these indicators and derive appropriate management actions, but no attempt to provide overall indices of the state of the system. Elements of this system, the division of pressures into drivers and material fluxes and the linkages to social impacts are applied in the IEAF project framework.

The above review illustrates some of the approaches that have been used to assess estuary condition. None of these answers in entirety the requirements set out in Section 2.1 but there are useful ideas in many of them. Based on these ideas and our own requirements, a proposed framework is described in the following section.

3. Proposed framework

The recognised framework for managing water quality in Australia is the National Water Quality Management Strategy (ANZECC 2000; see <www.deh.gov.au/water/quality/nwqms/index.html>).
It is therefore important that any other framework dealing with water quality has a clear relationship to the NWQMS. A schematic of the NWQMS, taken from the ANZECC 2000 Guidelines is shown in Figure 2.

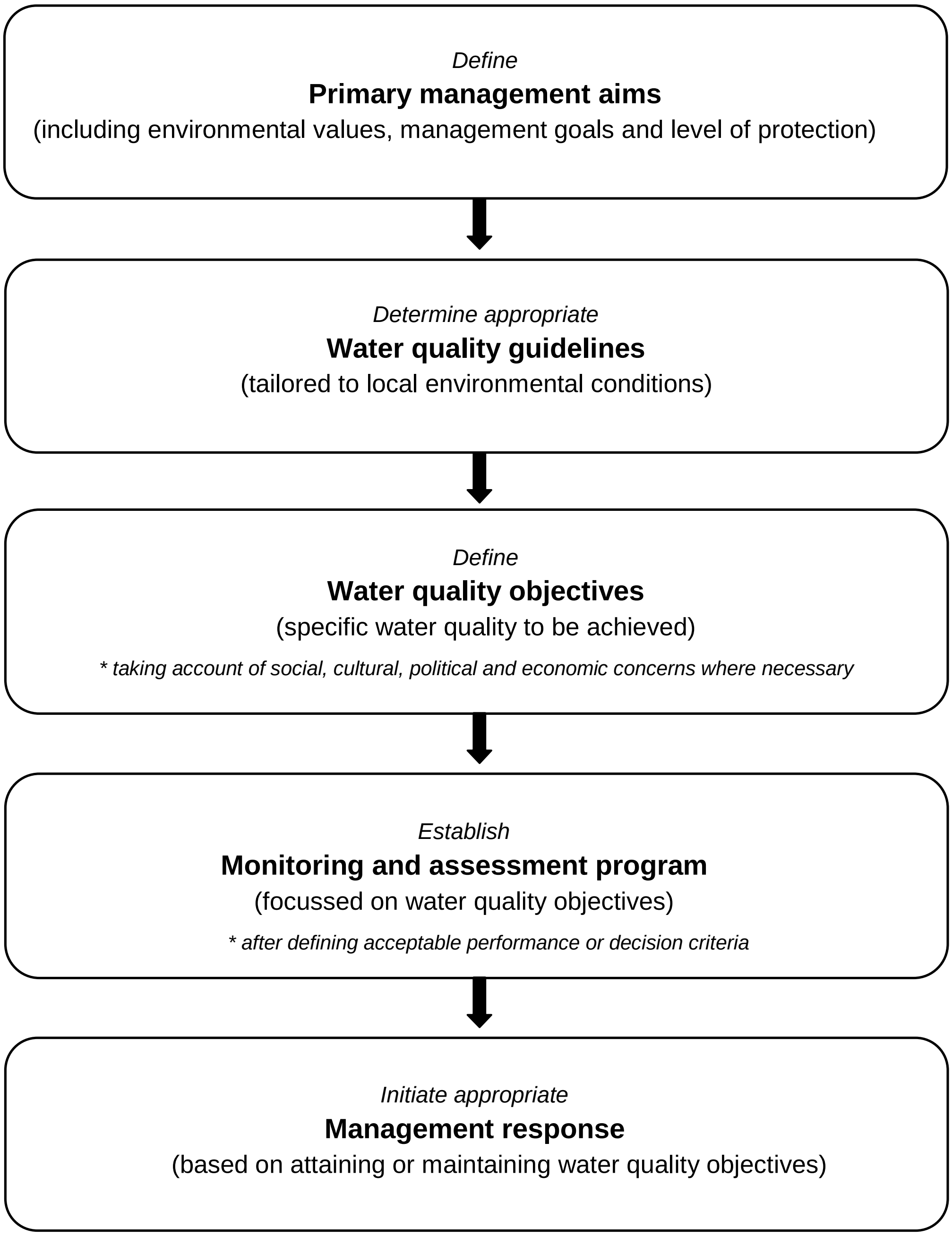

Figure 2. National Water Quality Management Strategy process outline
(adapted from ANZECC 2000)

Within this schematic, the first step—definition of primary management aims—is the most relevant to the work in this project. There are two main components in this. The first, determination of environmental values, is essentially about obtaining the views of the community on what uses and values of the system they wish to maintain or enhance. The second component, management goals, is about the identification of more specific aims that the strategy should address. The ANZECC 2000 Guidelines suggest that, in particular, management goals should reflect the specific problems and/or threats to the established values (ANZECC 2000). Given that resources for management strategies are always going to be limited, management goals need to reflect the highest priority threats and this in turn will guide which actions get the highest priority in the final management strategy.

The question that arises is how the main threats to the system should be identified in the first place. The framework developed in this project is aimed at addressing this question. It assesses a wide range of possible threats (termed stressors) to a system and then ascribes a priority to each one. Prioritisation is based on both the biophysical condition of the system and the values attributed to the system by the community. The framework additionally provides a logical means of selecting indicators relating to each threat. These indicators are used both to set targets and to monitor progress towards targets.

Thus, in summary, the framework developed in this study is about achieving the first step in the NWQMS, defining the primary management aims. Subsequent steps in the NWQMS—setting specific objectives, devising on-ground management strategies and monitoring the outcomes—are largely outside the scope of the framework.

A basic outline of the proposed framework is shown in Figure 3. The starting point of the framework is a set of defined pressures, termed stressors. With respect to each stressor, the framework considers both the risks to the system (a combination of the intensity of the stressor and the vulnerability of the system to the particular stressor) and the actual measured condition of the system. These two separate assessments of the system are then compared with each other. Where the two assessments do not agree, this is a trigger to re-examine the data for both assessments in order to determine the reason for the disparity and to try to resolve it. Where there is general agreement between the two (or once any disparities have been resolved), these are compared with desired condition. Desired condition is itself based on both community values and technical input in the form of guidelines. The comparison of desired condition with actual condition then provides an indication of which threats (stressors) are having the greatest impact on the system, and therefore what are the management priorities.

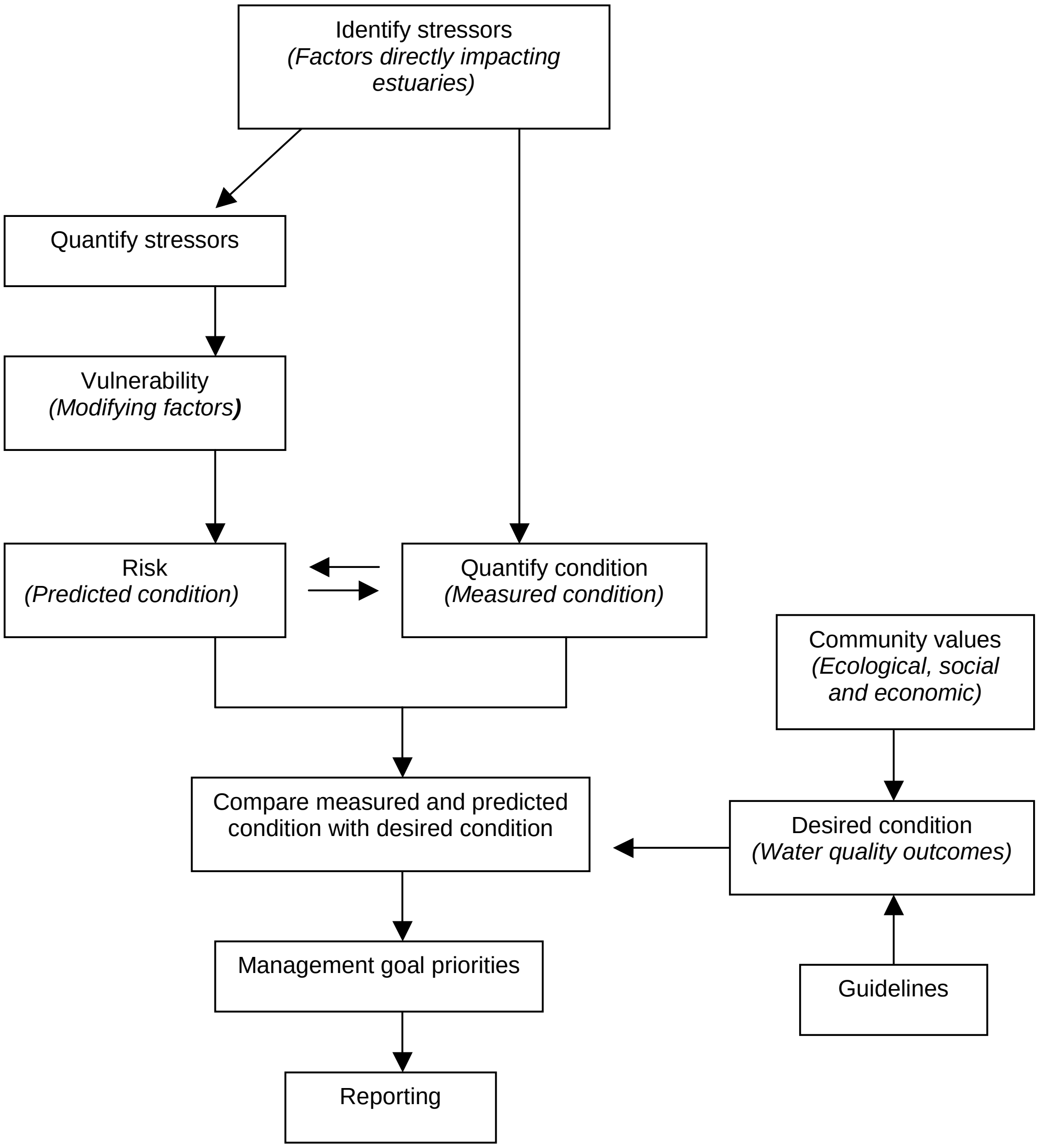

Figure 3. Outline of proposed integrated estuary assessment framework

While this framework is primarily an assessment framework, it also provides a logical basis for selecting indicators. Indicators are selected in a stepwise progression through the framework. The starting point is stressors, with indicators being selected that relate to stressors of interest. Next, indicators of any relevant vulnerability factors are determined for each stressor. Condition indicators are then selected that relate to the impacts of each stressor. Lastly, if required, indicators of values that relate to each stressor can be determined. The indicator selection process is described in more detail in Section 5.

The framework and indicator selection process developed in this project were adopted in a related Coastal CRC project (Scheltinga *et al.* 2004). This investigation was aimed at developing a set of national indicators for estuaries. The project applied the process described here to develop indicators for a set of 13 defined stressors. These are described in detail in Scheltinga *et al. (*2004).

The remainder of Section 3 describes the various components of the proposed framework in more detail and discusses how they link together.

3.1. Pressures and stressors

The term 'pressure' has been used fairly indiscriminately to describe a whole range of factors or activities at different levels (from population density to fertiliser application rates to changes in pollutant concentrations) that cause impacts on natural systems. Pressures can be natural factors that have been altered by human activity (e.g. increases in nutrient loads) or they can be entirely anthropogenic factors (e.g. fishing). Water quality management strategies have traditionally been focussed on physico-chemical pressures. In this framework, the scope of pressures has been broadened to include the whole range of factors that impact on aquatic ecosystems and includes pollutants, changes to habitat, changes to flows, pest species and direct human impacts such as fishing.

An initial step in developing the framework was to determine the division between pressures and condition. This is not always straightforward. Some issues can be separated into a whole series of causal links where A affects B which then affects C and so on. Thus, fertiliser use affects nutrient loads which affect instream water quality which then causes increased plant growth which then affects other biota. In this situation, the question arises as to which elements are pressures and which are condition; for example is instream water quality a measure of pressure or condition? Other issues have far fewer links; for example, fishing simply reduces fish populations so the division between pressure and condition is much clearer. For habitat issues, the division varies with different situations. The effect of boat anchors on seagrass and corals has a clear division between pressure (number of boats anchoring) and condition (damage to corals or seagrass). However, in the case of the direct removal of, say, 30% of mangroves from a system, while this is clearly a pressure, the 30% reduction in mangrove cover equally represents a change in condition and is effectively the same measure.

Just where a division between the two is drawn is probably not too important as long as it is clearly defined. For the purposes of this framework, condition is taken to be condition of any component of the system itself and includes water quality, habitat, biota and aesthetics. Pressures are those things that impact directly or indirectly on these components and include pollutant loads, gross habitat destruction or alteration, biota removal and alterations to freshwater inflows. Where there is overlap—for example, with respect to habitat removal—priority is given to expressing the measure as condition although it could also be placed in both pressure and condition.

For the purposes of the related national indicator project (Scheltinga *et al.* 2004), the concept of 'stressors' was developed. The reason for doing this was to get away from the very broad concept of pressures which, as mentioned earlier, can encompass a range of different levels. Stressors, in contrast, are aimed at identifying specific factors that directly impact on a system. Thus, for example, urban development within a catchment could be described as a pressure but the specific stressors caused by urban development would include, among others, increased sediment and nutrient loads and changes to freshwater inflow.

Stressors were defined in Scheltinga *et al. (*2004) as follows: Physical, chemical and biological stressors are major components of the environment that, when changed by human or other activities, can result in degradation to natural resources. Stressors can be:

- a component of the environment that transfers the impact of a pressure (e.g. human activity) to other parts of the environment by being changed from its natural state (e.g. nutrient concentrations changed from natural, habitat coverage less than natural or excess salt). These components of the environment are usually present in natural (healthy) ecosystems and are only considered stressors when they are different from natural; and
- a component of the environment that, when present, causes stress on the ecosystems (e.g. litter or pest species). These components of the environment are not usually present in natural (healthy) ecosystems and are considered potential stressors when they are present in any amount (Scheltinga *et al.* 2004).

The definition of stressors given above is still rather broad and, in practice, identification of stressors within the bounds of this definition is largely based on expert opinion. A set of 13 defined stressors were identified within the Scheltinga *et al.* (2004) project and were subject to expert review at two national workshops. Most, although not all, of these have been adopted for this project but there are also some additions. A list of proposed stressors for estuarine systems is given in Table 1. This list is intended to cover the broad range of possible stressors but is not necessarily exhaustive and, for particular systems, additional stressors might be identified.

For most stressors, Scheltinga *et al.* (2004) identified a range of both causes and symptoms (i.e. impacts on the system) and these are useful in identifying which stressors are likely to be most relevant to a particular system.

Table 1. List of proposed estuary stressors

Stressor	Examples of causes
Pollutants:	
Organic matter	Abattoir discharge
Fine sediments	Urban development
Acid runoff	Drainage of acid soils
Nutrients	Sewage discharge
Heavy metals	Mine waste
Pesticides and organics	Agricultural use
Oil	Marina operations
Pathogenic micro-organisms	Sewage discharge
Litter	Urbanisation
Riparian habitat removal or disturbance	Loss of mangroves
Direct removal of biota	Fishing, bait collection
Freshwater inflow alteration	Reductions in inflows due to dams
Alteration to hydrodynamics	Dredging of entrances
Pest species	Introduced dinoflagellates
Shoreline development	Urbanisation

Stressors are the real starting point in the framework. A first step in using the framework is therefore to identify the stressors (based on the list in Table 1) that are important to the system under investigation. The next step is to quantify the identified stressors. This involves identifying appropriate indicators (see Section 5 of this report) and then using the most appropriate method to quantify these. Pollutant stressors such as nutrients would ideally be quantified as measured loads entering a system. They could also potentially be quantified as concentrations within the system itself but, as defined earlier, instream water quality has been defined as a measure of condition rather than as a pressure. As another example, quantifying the stressor ‘direct removal of biota’ might involve assessing fish catch.

If direct measures of stressors are not available, it may be necessary to estimate them through indirect or proxy measures. In the case of the examples above, nutrient loads might be estimated through models or even simply from catchment land use while fish catch might be estimated through numbers of professional and/or amateur fishermen using a particular estuary.

Stressors are initially quantified in terms of some continuous measure (e.g. tonnes of nitrogen entering an estuary per year). For the purposes of this framework, this information is then transformed into categories. It was decided to use five categories (1–5) for expression of both stressors and the associated factors of vulnerability, risk and condition. The convention used in this framework is that category 1 expresses low stress, low vulnerability, low risk or good condition while category 5 denotes high stress, high vulnerability, high risk or poor condition.

Using categories that cover a range of values has the advantage that imprecise data (which is the norm) can be more readily accommodated. Also, where available input data is very imprecise, the framework allows input of just three categories (i.e. 1, 3 and 5) which are equivalent to low, moderate and high.

Determining the ranges of values that should be used for each stressor category clearly requires some prior knowledge of how pressures affect condition and is usually done by means of some form of predictive model. For example does a loading of ‘X’ kg of nitrogen represent a high or low stress to an estuary? Thus, it is necessary to be able to quantify the pressure/condition relationship. This is discussed further in Section 3.3.

3.2. Vulnerability

This denotes the vulnerability or sensitivity of the system to a stressor. Another term that has been used to express this idea is ‘modifying factor’, that is, factors that modify the impacts of a particular stressor (Paul *et al.* 2002). Vulnerability is specific to individual stressors but the most common factor involved here is flushing or exchange rate, which affects the sensitivity of systems to a number of pollutants. Other vulnerability factors include sediment type or ability of different target species (such as fish, shellfish or bait worms) to withstand different levels of exploitation. For some stressors, there may be no appropriate vulnerability factors or alternatively in our current state of knowledge we may be unable to identify such factors.

As with stressors, there is a need to quantify vulnerability where possible. Information for quantifying factors such as flushing rates may be readily available but for other factors (e.g. the resilience of a fish species to fishing), such information may be very difficult to obtain. It may be possible here too, however, to use proxy measures. Even where vulnerability factors are known, some stressors may be virtually impossible to assess. For example, the effect of flushing rates on capture of toxicants in estuaries is very hard to quantify.

Vulnerability values, as for stressors, are transformed into five categories. Determining the ranges of values for each category will naturally require some prior knowledge of how vulnerability affects condition.

The framework allows vulnerability to be omitted if no sensible estimate can be made or if it is not seen as relevant. In this situation, risk is assessed entirely from the stressor score.

3.3. Risk

Risk is commonly defined as the product of likelihood and consequence. In the context of this framework it is something slightly different: a prediction of the impact on condition of a given level of pressure and vulnerability. Quantification of this type of risk has to be based on prior knowledge of the relationship between risk and system condition, and again is determined through some sort of predictive relationship or model.

Such models can be empirical or deterministic. Deterministic models that link pressures to condition via a set of explicit and quantified processes usually require extensive inputs of data that in many cases is not available. This project therefore focussed on empirical models, that is, models that depend on statistically derived relationships between pressure and condition. These are sometimes termed ‘black box’ models because although the inputs and outputs are known, unlike deterministic models the detailed processes linking the two are not fully understood. An early example of this approach was Vollenweider’s model for freshwater lakes that related annual phosphorus (P) loading to chlorophyll *a* levels (Vollenweider 1971). This was empirically based on data from a large number of lakes. Vollenweider also included a vulnerability factor—depth, with deeper lakes showing less impact than shallow ones for a given P loading.

More recently there have been a number of studies that have attempted to link landscape metrics (causal pressures) to instream condition. Mallin *et al.* (2001) were able to derive relationships between demographics, landscape and rainfall and the microbiological pollution of coastal waters. Hale *et al.* (2004) derived statistical relationships between landscape indicators and estuarine benthic condition in U.S. estuaries. Paul *et al.* (2002) developed quite detailed statistical relationships between landscape metrics and a range of sediment contaminants. They also noted sediment characteristics and hydrology as important modifying or vulnerability factors.

In Australia, the Water Services Australia publication *Catchments for recreational water: conducting and assessing sanitary inspections* (WSAA 2003) similarly advocates the approach of assessing risk to systems through quantification of pressures on the system. Table 2, reproduced from that publication, illustrates the relatively simple approach used. The column 'Significance' represents risk or predicted condition, and 'Dilution' and 'Effect of origin of micro-organisms' are included as vulnerability or modifying factors.

Table 2. Risk assessment protocol (illustration of the approach to estimating the significance of faecal contamination in recreational waters, from WSAA 2003)

Source	Concentration (faecal streptococci)	Effect of dilution and dispersion		Effect of time	Resulting concentration (for comparison with monitoring results)	Effect of origin of micro-organisms	Resulting concentration (for determining significance)	Significance
		Nature of discharge and receiving water situation	Dilution factor					
Wastewater discharge A (secondary treatment, no disinfection)	10^5	Short outfall close to shoreline and beach	0.04	1	~4 000	1.0	~4 000	Very high
Wastewater discharge B (primary treatment with disinfection)	10^5	High flow rate discharge via long outfall	0.01	1	~1 000	1.0	~1 000	High
Stormwater A (urban, no sewage overflows)	10^4	Drain direct to beach	0.20	1	~2 000	0.5	~1 000	High
Stormwater B (rural, no sewage contribution)	10^4	Drain 500 m up-stream (prevailing current), direct discharge at beach	0.05	1	~500	0.1	~50	Moderate
Bathers (fewer than 20 per 150 m^3	10^1	No dilution; volume of swimming area ~150 m^3	1.00	1	~10	1.0	~10	Low
Rural stream	10^3	Discharge downstream (prevailing current)	0.02	1	~2	0.1	~0.2	Very low
Total					>7 500		>6 000	Very high

Notes:
1. Estimations taken during wet weather.
2. Number or organisms estimated by multiplication across the table, i.e. 10^5 x 0.04 x 1 = 4 000.

For the purposes of this framework, the measures of pressure and vulnerability are similarly combined to provide a measure of risk to the system from the particular stressor being assessed. This is undertaken through a simple-two way plot of vulnerability versus pressure. The method from Cox *et al.* 2004 has been applied and is illustrated in Table 3.

Table 3. Risk to a system, derived from pressure and vulnerability scores
(adapted from Cox *et al.* 2004)

Risk		Pressure				
		1	**2**	**3**	**4**	**5**
Vulnerability	**1**	1	1	1	2	3
	2	1	1	2	3	4
	3	1	2	3	4	5
	4	2	3	4	5	5
	5	3	4	5	5	5

Where: 1 is low pressure, vulnerability and risk, and 5 is high pressure, vulnerability and risk.
A system that has low pressure and low vulnerability is at low risk, while a system with high pressure and high vulnerability is at high risk.

The categories 1–5 *on the axes* (bolded numbers) are taken from the pressure and vulnerability assessments. The boundaries between categories are based on expert interpretation of the underlying relationships and are open to adaptation by other users of this framework.

The scores *within the table* (non-bolded numbers) which represent risk are to some extent arbitrary. Clearly a combination of low pressure and low vulnerability equates to low risk (1) and conversely, high pressure and high vulnerability equates to high risk (5). However, the scores in between have been interpolated assuming linear relationships between stressor, vulnerability and risk. The extent to which this assumption is true will depend on the particular pressure and vulnerability factors. For the purposes of the framework we have assumed that it is true in all cases but, as noted above, it is possible to alter the category boundaries for both pressure and vulnerability so that the distribution of risk numbers is better suited to particular systems.

In this study, for most of the identified stressors, black-box models that relate the pressure, vulnerability and risk have been developed based around the simple format described above. These are preliminary in nature and based on the best information currently available, although information of the type required for this sort of approach is limited. However, users of the framework can readily modify these or tailor them to particular systems where information is available. The black-box models for individual stressors are detailed in the Appendix.

While the proposed format of five categories is relatively simplified, it is worth noting that from a management perspective, five categories are generally a quite sufficient basis for determining management priorities.

3.4. Condition

This section describes the actual measured condition of the system as opposed to the predicted condition or risk described in Section 3.3.

In some assessment systems, condition assessment is designed to provide only a *general* assessment of condition: for example, the Sustainable Rivers Audit aims to provide a general assessment of the ecological health of Australian rivers by measuring macroinvertebrate and fish diversity. However, if condition is found to be impaired, these types of general indicators usually do not provide specific information about which particular pressure or pressures are causing the impact. It is a key feature of this framework that condition assessment is directly linked to specified stressors. Thus, condition indicators are selected on the basis that they provide information specific to the impacts of a particular stressor. This in turn provides better information about which stressors need to be addressed in management plans.

Selection of condition indicators should be based on:

- The nature of the impacts of the stressor on the system
- The values of the system on which the stressor is likely to impact.

With regard to the nature of the impact, these can be complex, which adds to the complexity of selecting appropriate indicators. Many stressors have both primary impacts on water quality and consequent impacts on the ecosystem values or human values. For example, organic matter impacts on dissolved oxygen which in turn impacts on the biota. The impacts on values (in this example the ecosystem) are of primary interest. However, direct indicators of ecosystem impacts are usually difficult to measure and are often confounded by the fact that a change in the indicator value can be caused by a number or combination of different stressors. It is therefore often much easier and more informative to measure water quality indicators and use these to infer likely impacts on the ecosystem based on known relationships.

Another example is the removal of habitat which directly affects habitat (obviously) but which will also have some follow-on consequence for the ecosystem. However, these consequences are usually hard to assess and directly measuring habitat loss and inferring impacts on the ecosystem is a more practical approach. Other stressors such as fishing have more direct impacts on the biota and therefore indicators need to reflect this impact.

Indicators also need to have some known relationship to one of the values of the system that is to be protected. Thus, changes in dissolved oxygen are relevant to ecosystem protection but largely irrelevant (except in extreme cases) to suitability of waters for swimming. In contrast, shoreline litter density is relevant to visual recreation but of limited relevance to ecosystems (except in the case of some specific types of litter).

Scheltinga *et al.* (2004) suggested that three major categories of indicators be considered—water quality, habitat and biota. For each stressor, indicators would be selected from one or more of these categories. In some cases, a stressor may not impact on a particular component (e.g. effects of litter on water quality) and so no indicators would be selected. In other cases, we may not have sufficient knowledge of how the stressor affects a particular component. This is especially true for the effects of stressors on the biota of Australian estuaries. These effects are very poorly understood.

Cooper (1994) derived relationships between overall estuary health and fish diversity in South African estuaries but no similar relationships have been documented in Australia. Various attempts have been made to relate the diversity of macroinvertebrates to estuary health in the same way as this has been done for fresh waters (Skilleter & Stowar 2001; Moverly & Hirst 1999). However, these have concluded that estuarine macroinvertebrates are too naturally variable to allow the effects of anthropogenic impacts to be reliably discerned. Thus, proven biological indicators for estuaries are scarce. The most reliable are biological indicators of nutrient stress such as increases in primary production manifested as increased biomass of phytoplankton, periphyton or macroalgae.

The grouping of indicators into the water quality, habitat and biota categories has some practical advantages. Ultimately, however, the main criteria for a condition indicator relating to a particular stressor are firstly that it reflects the impacts of that stressor (and not those of some other stressor) and secondly that it is related (either directly or indirectly) to one of the values of the system.

The Appendix of this report provides suggested condition indicators for all the proposed stressors. These should be seen as defaults and users can substitute other indicators that may be more relevant to their own particular systems.

Having selected condition indicators, these need to be rated, like other framework components, into five categories from 1 (very good) to 5 (very poor). Guideline information (e.g. ANZECC 2000 Guidelines) is a starting point in achieving this. However, guidelines are usually only useful in delineating category 1, that is, very good condition. There are some exceptions—for example, heavy metals—where the guidelines give different values for different levels of risk. However, for most stressors, boundaries for the other four rating categories have to be determined through expert opinion or through published methodologies (e.g. litter assessment methodologies often have associated rating categories). Some stressors such as habitat extent can be assessed through comparison with habitat extent prior to European settlement but it is still necessary to assign five categories to the extent of habitat loss.

An important point to note is that condition ratings, like guidelines, are specific to a particular value of the system. Thus, for example, thin surface films of oil affect the aesthetic value of a system but may be unimportant in terms of ecosystem protection. Most of the stressors considered in this framework impact mainly on one particular value and most commonly this is the value of the ecosystem itself. However, where a stressor does impact on more than one value, it is important to keep in mind that it may be necessary to apply different ratings categories to different values.

The Appendix provides default category boundaries for relevant condition indicators for all stressors. As described above, the numerical category boundaries are based on a mixture of guidelines values and expert opinion. The background information for each stressor indicates which value is being considered for each of the selected indicators. As with the risk assessments, the category boundaries can be modified by framework users to suit their own systems.

3.5. Comparing risk and condition

'Risk' is a measure of *predicted* condition while 'condition' is a direct measure of actual or current system condition. Having two separate measures is quite powerful as it gives two 'fixes' on the system. Provided that the scoring categories for each were selected based on good information, one would expect that the relationship between risk and condition would be consistent, that is, a system with high risk would have poor condition or vice versa. If this relationship holds up, then we can be reasonably sure that our diagnosis of the system is correct. Conversely, where this relationship does not hold up, this in itself provides useful information. Initially, it might flag that either our risk or our condition assessments are incorrect. Assuming this is not the case, a scenario of low risk combined with poor condition might indicate that a system has naturally poor condition and that this is not actually a cause for concern. Alternatively, where risk is high but condition is good, this might indicate potential for future problems with any increased level of anthropogenic activity.

This comparison is incorporated into the framework in the form of a two-way table (Table 4), similar to the stressor/vulnerability table (Table 3). The generic form, from Cox *et al.* (2004), is given below. In theory the risk should equate to the expected condition, and comparison of the risk and the observed condition provides a crosscheck on the pressure and condition assessments.

Table 4. Assessment comparison of observed condition and expected condition (risk) (from Cox *et al.* 2004)

		Observed condition				
		1	**2**	**3**	**4**	**5**
Risk	**1**	A	B	C	C	C
	2	B	A	B	C	C
	3	C	B	A	B	C
	4	C	C	B	A	B
	5	C	C	C	B	A

Where:
1 = Good condition or low risk, 5 = poor condition or high risk
A = Observed condition matches expected condition (i.e. risk)
B = Observed condition differs slightly from expected condition
C = Observed condition does not match the expected condition. These situations need to be examined in more detail.

For each stressor, categorised scores for risk and observed condition can be plotted in this way. The A/B/C result provides an assessment of the match between the two.

This type of approach, that is, comparison of risk with measured condition, is similar to that recommended in WSAA (2003). Table 5 illustrates the approach taken there. The vertical axis shows 'risk' as we have defined it from 'very low' to 'very high' (i.e. 1–5). Across the top are four 'measured condition' categories (<40, 40–200 etc.). The outcomes within the table are given verbal descriptions rather than alphabetic or numeric ratings. Thus our 'C' category is equivalent to their 'follow up' category. The table also shows an additional risk assessment based on number of organisms. However, this is done to demonstrate justification for the selected condition categories.

Table 5. Example of comparison of risk and condition (from WSAA 2003, based on results of monitoring and sanitary inspection of recreational water)

Risk ↓ / Ranking of significance of contamination[2]	Number of faecal streptococci determined from monitoring[1]			
	<40	**40–200**	**201–500**	**>500**
	Risk level inferred from numbers of organisms[3]			
	GI < 1 in 100 exposures AFRI <1 in 300 exposures	**GI < 1 in 20 exposures AFRI <1 in 40 exposures**	**GI < 1 in 10 exposures AFRI <1 in 25 exposures**	**GI > 1 in 10 exposures AFRI >1 in 25 exposures**
Very low	Very good	Very good	Follow up[4]	Follow up[4]
Low	Very good	Good	Fair	Follow up[4]
Moderate	Follow up[4]	Good	Fair	Poor
High	Follow up[4]	Follow up[4]	Poor	Very poor
Very high	Follow up[4]	Follow up[4]	Poor	Very poor

Condition →

Notes:

[1] 95th percentile values, as nominated by WHO. Monitoring results with a high degree of variability will need to be reviewed to determine the cause of the variability and the appropriate method for estimating the upper confidence limit. It may be appropriate to distinguish and separately classify different conditions such as wet weather.

[2] Ranking based on numbers of faecal streptococci present estimated from sanitary inspection with particular emphasis on human faecal contamination. Order of magnitude estimates only, based on available data.

[3] AFRI = acute febrile respiratory illness; GI = gastrointestinal illness.

[4] Unexpected result requiring reconciliation as far as is practicable. Generally, monitoring data for the bathing water should take precedence over estimates from the sanitary inspection unless the monitoring data is uncertain or limited (e.g. does not include the range of conditions such as wet weather).

3.6. Values

Values are the perceived values of the system. These values are analogous to the ‘environmental values’ identified in the National Water Quality Management Strategy (ANZECC 2000). For the purposes of this framework we have used the National Water Quality Management Strategy values as a starting point but have (a) broadened the scope to include some economic values such as fishing and (b) provided a more detailed break-up of the various values. In the same way that we have produced a set of *stressors* relevant to estuaries, the aim has been to provide a fairly extensive set of suggested estuary *values*. These are shown in Table 6.

Table 6. Proposed list of values of estuaries

Value categories	Values
Ecosystem values	
Ecosystem health	Water quality Habitat In-stream biota e.g. fish, shellfish, benthic invertebrates Water-associated wildlife e.g. wading birds Ecosystem processes e.g. denitrification
Conservation values	Rare and threatened species Representativeness Special habitats
Human-use values	
Recreational values	Recreational fishing Recreational bait collection Water skiing, jet skiing Sailing, windsurfing, canoeing General boating Swimming, diving Passive visual recreation (aesthetics) Consumption of shellfish and fish
Cultural values	Maintenance of cultural assets – Aboriginal or European
Economic values	
Commercial fishing	Netting Trawling Crabbing Bait collection
Aquaculture	Cultured shellfish Caged fish Prawn farms
Tourism	Scenic tours Fishing General recreation Diving
Other	

This default list can be used to match against risk and condition in order to better capture the potential impacts of changes in these characteristics on system values. Framework users can readily add other values to this list that may be relevant to their own particular systems.

Having identified a default list of values the next task is quantify the significance of values for a particular system.

3.6.1. Quantifying the significance of values

The National Water Quality Management Strategy process recommends a broad community consultation process for identifying the important values of a system. However, there are no specific recommendations on quantifying the significance of different values. For the purposes of this framework it is recommended that the default values in Table 6 should be rated in the same way as other framework components, that is, given a score of 1–5. However, in this case ‘1’ would represent a very low value and ‘5’ a very high value.

An example of where this type of value assessment has been done is shown in Table 7 (from Lockie & Jennings 2002), where respondents in Central Queensland were asked to rank the values of their waterways on a scale of 1–10.

Table 7. Example of community assessment of values of a waterway system on a scale of 1–10 (from Lockie & Jennings 2002)

Value	Mean score	Standard deviation
Ecological/environmental significance	8.96	1.577
Town water supply	8.86	2.225
Scenery and landscape	8.25	1.942
Symbol or landmark	8.11	2.045
Agriculture/farming	7.64	2.726
Tourism	7.54	2.434
Industrial water supply	7.51	2.828
Stormwater disposal	7.34	2.905
Land-based recreation	7.18	2.414
Cultural and festival activities	7.17	2.395
Heritage	6.98	2.748
Water-based recreation	6.64	2.874
Wastewater disposal	6.55	3.619
Entertainment and meeting	6.39	2.637
Commercial fishing	5.86	3.121
Sand and gravel extraction	5.73	3.324
Other commercial use	5.71	3.338
Residential development	5.26	2.926
Passenger transportation	5.25	3.026

3.6.2. Relating condition to values

For each stressor in the IEAF, the selected condition indicators are usually targetted at one particular environmental value, which in most cases is ecosystem protection. Thus, the condition ratings (1–5) attributed to each indicator are based on the likely impacts on that particular value. For example, the boundary for condition rating 1 is commonly based on the ANZECC guideline number that relates to the value in question. Boundaries for the other condition ratings are mostly determined by expert opinion but the numbers selected are based on the likely level of impact on the nominated environmental value.

Thus, condition ratings function as direct measures of the impact of the stressor on the value in question. To give an example, for the stressor organic matter, the main proposed condition indicator for the ecosystem value is dissolved oxygen. Condition rating 1 for dissolved oxygen thus indicates no impact on the ecosystem while condition rating 5 indicates a very major impact.

Provided the environmental value at which the indicator is directed is specified, the condition ratings can then be directly read off as measures of impact on that value. This means that in the assessment of management priorities, the significance of the environmental value (as defined by the community) can be directly compared with the condition rating (i.e. the impact on that value). Based on the level of impact and the significance assigned to that value, logical judgements can be made on the priority that should be given to the stressor in question.

4. Reporting and management priorities

As indicated in the framework outline, management priorities should be guided by consideration of risk, condition and values. Reporting therefore needs to cover these elements. The framework is based around the key stressors identified in Table 1 and reporting is likewise based on these stressors.

Essentially, the reporting framework seeks to address the following questions:

1. What level of risk does a particular stressor represent to the system?
2. To what degree is it currently impacting on condition?
3. How important are the values that are affected by this particular stressor?

Based on these questions, an example of the proposed reporting output format is given below in Table 8. Each element is scored from 1–5, with 1 being the best or lowest, through to 5 being the worst or highest. The exception to this is the score for significance of related values where 5 denotes the highest value.

Table 8. Example of reporting on a specific system

Stressor	Risk	Condition	Values affected	Significance of values as rated by community
Acid runoff	5	4	Ecosystem	5
Nutrients	2	1	Ecosystem	5
Habitat loss	4	4	Ecosystem	5
Pathogens	5	5	Recreation	2

In this particular example, acid runoff, habitat loss and pathogens are significant in terms of both risk and condition. Nutrients, however, are much less of a risk to the system. Acid runoff, nutrients and habitat loss all impact on the ecosystem value which has been given a high significance, while pathogens impact on recreation which in this example has a lower significance. In this example, therefore, acid runoff and habitat loss would be rated as the highest management priority as they have a high risk and are impacting on a highly rated value. Pathogens have a high risk but are impacting on a lower rated value and therefore are of lower priority. Nutrients impact on a high significance value (ecosystem) but have a low risk and are therefore similarly of lower priority.

5. Indicator development

Selection of indicators has to be done in a specific context and with some purpose in mind. A criticism that can be made of many SoE documents and other related literature is that they present long lists of indicators but with only loosely defined purposes.

The framework described in Section 3 provides both context and purpose for the development of indicators. Using the defined stressors as a starting point, indicators are selected based on a logical progression through the framework. This process is illustrated in Figure 4 below and a worked example is given in Table 9.

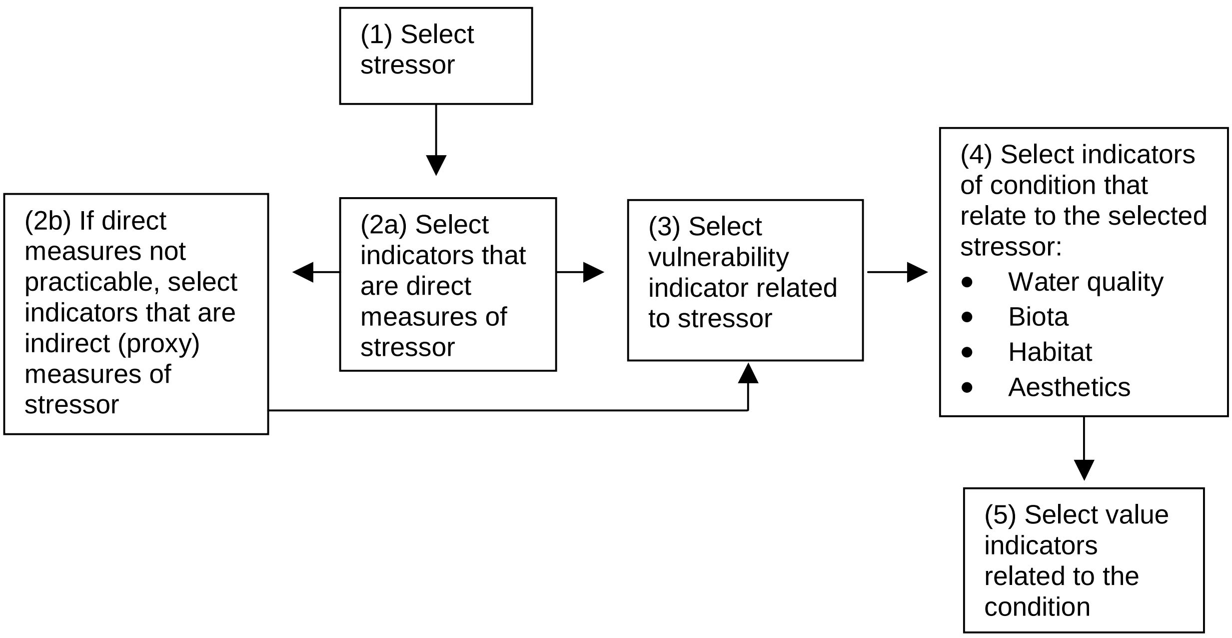

Figure 4. Process for selecting indicators

Table 9. Example of the application of the IEAF framework for selection of indicators, using nutrients as the stressor example (adapted from Cox *et al.* 2004)

Stressor: Nutrients (nitrogen and phosphorus)		
Pressure indicators	**Indirect pressure indicators**	Catchment land use % of length of stream with healthy riparian zone % of sewage treatment plants with tertiary treatment Volume/number of sewage overflow events % of area under aquaculture % area using septic systems
	Direct pressure indicators	Total diffuse nutrient load entering the system (monitored or modelled) Total point-source nutrient load entering the system (monitored or modelled)
Vulnerability indicators		Flushing rate of system
Condition indicators	**Water quality**	Total and dissolved nutrients in the water column OR Total and dissolved nutrients in the sediments
	Biota	Chlorophyll *a* OR Seagrass biomass of epiphytes Intertidal sand/mudflat: Macroalgal biomass, benthic microalgal biomass Rocky shores, rocky reef and coral reef: Algal biomass per unit area
	Habitat	Extent/distribution of seagrass
	Aesthetics	Algal blooms, macrophyte blooms
Value indicators	**Aesthetics**	Number/frequency of algal blooms
	Recreation	Number/frequency of recreational area closures Number of visitors to recreational areas (mangroves, reefs etc.)
	Aquaculture	Closure of aquaculture areas due to noxious blooms
	Fisheries	Reduction in fish catch, change in species
Response		% of farming area using best management practices (BMP) Upgrades to point sources Length of riparian zone rehabilitated % of septic systems maintained Incentives provided for converting to BMP Amount of fertiliser applied per unit area % of urban area under stormwater management plan

It is implicit in this process that the linkages between pressure and condition indicators are understood. This understanding has to be based on technical expertise but the use of pictorial conceptual models (such as that in Figure 1) is a good way to share this understanding and to make it explicit to all users.

Application of this approach to derive relevant indicators for a wide range of stressors is detailed in Scheltinga *et al.* (2004). Again, this process needs to be tailored to individual types of systems.

6. Application of the framework

Having developed a framework, an important part of the IEAF project was to populate it with information so that it would become a useful tool for management. For each identified stressor, this involves developing indicators for the various components of the framework, as outlined above, and then quantifying the links between them. This is not a trivial task. It is made more complex by the fact that both the physical and biological characteristics of estuaries vary significantly around Australia.

What has been done here, therefore, is to develop some default information for each of the major stressors. This provides sufficient information on indicators and linkages to enable the framework to be used and at the same time to provide guidance on how the framework is applied. However, other users can tailor the information to their own systems. The information is provided in a series information sheets for each stressor and these sheets are contained in the Appendix. Each sheet provides information under the following headings.

- Stressor name
- Background information
- Pressure indicators and scoring categories
- Vulnerability indicators and scoring categories
- Condition indicators and scoring categories.

The information provided will then allow users to progress through the framework using a spreadsheet.

6.1. A worked example of using the framework

Essential steps in using the framework are as follows:

6.1.1. Select stressors

The default list of stressors is as shown previously in Table 1, namely:

- Pollutants:
  - Organic matter
  - Fine sediments
  - Acid runoff
  - Nutrients
  - Heavy metals
  - Pesticides and organics
  - Oil
  - Pathogenic micro-organisms
  - Litter
- Riparian habitat removal or disturbance
- Instream habitat removal or alteration
- Direct removal of biota
- Freshwater inflow alteration
- Alteration to hydrodynamics
- Pest species
- Shoreline development.

One alternative is for users to select only those stressors relevant to their estuary. However, it is recommended that all stressors be considered rather than just a selection. If this approach is used then the final report will provide an assessment of all stressors. Even though some are not significant, the fact that they would score lowly as management priorities is in itself useful information.

6.1.2. Quantify risk for identified stressors

As defined in Section 3.3, risk is equivalent to predicted condition, and is quantified through assessment of stressors and vulnerability.

The stressor of nutrient pollution is used here as an example. The recommended default pressure indicators for nutrients (see Appendix) are $mg/m^2/d$ of nitrogen or phosphorus. The values for these should be calculated from local information and the resulting values categorised from 1–5 according to the scoring categories given in the Appendix.

The default vulnerability indicator for nutrients is flushing rate. This should be calculated for the estuary under study and the value categorised according to values in the Appendix.

The pressure and vulnerability scores can then be combined (see Table 10) using the two-way table shown in Section 3.3.

Table 10. Assessing risk from nutrient pollution, using pressure and vulnerability

Risk		Pressure (mg P or N per m^2 per day)				
		1	**2**	**3**	**4**	**5**
Vulnerability (flushing rate)	**1**	1	1	1	2	3
	2	1	1	2	3	4
	3	1	2	3	4	5
	4	2	3	4	5	5
	5	3	4	5	5	5

Where: 1 is low pressure, vulnerability and risk, and 5 is high pressure, vulnerability and risk.

The risk value (1–5) is then read off from the matrix of the table. A high risk (expected condition) value equates to a poor expected condition. The matrix values in this table are a default for all indicators. However, users can adapt these values to their own systems if relevant.

Since we have considered both N and P we therefore have two risk values for nutrients. This situation of having more than one indicator and therefore more than one risk value arises with a number of stressors. It is suggested that the highest risk value be used in the final reporting on the estuary.

6.1.3. Assess condition

Again using nutrients as the example, the default condition indicators are chlorophyll *a* and macroalgae. Users should select the indicator most appropriate to their system or use both if this is appropriate. Condition indicator values are derived from local information and then categorised (from 1–5) according to the scores in the Appendix. If both indicators are used then the highest (i.e. worst) value should be used in the final report.

6.1.4. Compare risk with condition

The purpose of this comparison is to highlight any inconsistencies between risk (predicted condition) and actual condition values. The presence of inconsistencies would lead to a re-evaluation of the data before proceeding further in the framework. The comparison is done using the two-way table below (Table 11). If risk and condition are highly consistent then the matrix will give a score of A. If they are slightly inconsistent then the score will be B. If they are highly inconsistent then the score will be C which would be a trigger for significant further investigation of the data and assessments.

Table 11. Comparing a system's risk and condition for the nutrient pollution stressor

		Observed condition				
		1	2	3	4	5
Risk	1	A	B	C	C	C
	2	B	A	B	C	C
	3	C	B	A	B	C
	4	C	C	B	A	B
	5	C	C	C	B	A

Where:
1 = Good condition or low risk, 5 = poor condition or high risk
A = Observed condition matches expected condition (i.e. risk)
B = Observed condition differs slightly from expected condition
C = Observed condition does not match the expected condition. These situations need to be examined in more detail.

6.1.5. Report on biophysical status of estuary

Assuming risk and condition are reasonably consistent (i.e. A or B), the scores can then be used to generate a final biophysical report. (C ratings would require further examination.) Assuming also that all stressors have been addressed, then a final report would look like the example shown in Table 12.

Table 12. Example of a final report on the biophysical condition of an estuary

Report on status of Six Mile Creek estuary		
Stressor	**Risk***	**Condition***
Organic matter	2	3
Fine sediments	4	4
Acid runoff	1	1
Nutrients	3	3
Heavy metals	1	2
Pesticides and organics	2	2
Oil	3	3
Pathogenic micro-organisms	2	3
Litter	4	3
Riparian habitat removal or disturbance	3	3
Direct removal of biota	4	4
Freshwater inflow alteration	2	1
Alteration to hydrodynamics	2	2
Pest species	5	5
Shoreline development	4	4

*where 1 = Low risk or good condition, through to 5 = high risk or poor condition

6.1.6. Determine management priorities

The example above would suggest that the management priority in this particular estuary may be control of pest species (because both the predicted condition and actual condition are very poor). Fine sediments, shoreline development and fishing are also important, while a range of other stressors are having more limited impact. However, to determine final management priorities, the biophysical assessment then needs to be linked in with the perceived values of the estuary. In most situations, the biophysical priorities will end up being the final priorities. Nevertheless, it is important to go through the values assessment process to ensure that priorities are acceptable to the community.

7. Future development of the framework

This project has to a large extent completed development of the basic framework outline. Future development should focus largely on adding to or refining the information within the framework. This would include:

1. Improving empirical relationships in the black boxes
2. Developing quantitative relationships between risk/condition and values
3. Reviewing and developing indicators for each of the stressors
4. Tailoring all the above to different types of systems within Australia.

This represents a significant body of work and there are no mechanisms in place to undertake it at this stage. It seems most likely that such development would be carried out by agencies or other groups that are interested in tailoring the system to their local estuaries. There is sufficient information in this report and its appendix to allow this to happen.

In addition, the research team undertook development of a software package designed to take the user through the framework process. This package, termed the vulnerability–pressure–state–impact–risk–response framework (VPSIRR) is complete in terms of the software but is still under development in terms of populating the package with data. In principle it follows the same process outlined in the example in Section 6. However, the software makes the process much simpler for the user as it prompts for entry of the required data and then undertakes all the comparisons in the various tables.

Acronyms and abbreviations

ANZECC	Australian and New Zealand Environment and Conservation Council
ASS	Acid sulfate soils
BMP	Best management practice
BOD	Biochemical (or biological) oxygen demand
CRC	Cooperative Research Centre
DAFF	(Commonwealth) Department of Agriculture, Fisheries and Forestry
DO	Dissolved oxygen
DPSIR	Driver–pressure–state–impact–response (model)
GIS	Geographic information system
IEAF	Integrated estuary assessment framework
N	Nitrogen
NTU	Nephelometric turbidity unit
NWQMS	National Water Quality Management Strategy
P	Phosphorus
PSIR	Pressure–state–impact–response (model)
RUSLE	Revised universal soil loss equation
SoE	State of Environment
VPSIRR	Vulnerability–pressure–state–impact–risk–response (software)
WHO	World Health Organisation
WSAA	Water Services Association of Australia

References

ANZECC (2000) *Australian and New Zealand guidelines for fresh and marine water quality.* Australian and New Zealand Environment and Conservation Council/Agriculture and Resource Management Council of Australia and New Zealand. <www.deh.gov.au/water/quality/nwqms/index.html> (Accessed 16 June 2006).

Bartram, J. & Rees, G. (eds) (2000) *Monitoring bathing waters: a practical guide to the design and implementation of assessments and monitoring programs*. E & FN Spon, London.

Bidone, E.D. & Lacerda L.D. (2004) The use of DPSIR framework to evaluate sustainability in coastal areas. Case study: Guanabara Bay basin, Rio de Janeiro, Brazil. *Reg. Environ. Change* 4: 5–16.

Cooper, J.A.G., Ramm, A.E.L. *et al*. (1994) The estuarine health index—a new approach to scientific-information transfer. *Ocean & Coastal Management* 25(2): 103–141.

Cox, M., Scheltinga, D., Rissik, D., Moss, A., Counihan, R., & Rose, D. (2004) Assessing condition and management priorities for coastal waters in Australia. *Proceedings of the Coastal Zone Asia Pacific Conference*, 5–9 September 2004, Brisbane.

CSIRO (Commonwealth Scientific and Industrial Research Organisation) (2002) *SedNet: Assessing catchment water quality.* <http://www.clw.csiro.au/publications/general2002/managing_regional_water_quality.pdf> (Accessed 16 June 2006).

DAFF (Department of Agriculture, Fisheries and Forestry) (2006) Catchment condition online maps. <http://www.brs.gov.au/mapserv/catchment/> (Accessed 16 June 2006).

Deeley, D.M. & Paling, E.I. (1999) *Assessing the ecological health of estuaries in Australia*. Marine and Freshwater Research Laboratory, Institute for Environmental Science, Murdoch University. LWRRDC Occasional Paper 17/99 (Urban Subprogram, Report No. 10) December 1999.

Environment Agency (UK) (2002) *Aesthetic assessment protocol (beach survey).* R&D Technical Summary, E1-117/TR. Environment Agency, Bristol, UK.

Ferreira, J.G. (2000) Development of an estuarine quality index based on key physical and biogeochemical features. *Ocean & Coastal Management* 43: 99–122.

Hale, S.H., Paul, J.F. & Heltshe, J.F. (2004) Watershed landscape indicators of estuarine benthic condition. *Estuaries* 27(2): 283–295.

Kiddon, J.A., Paul, J.F., Buum, H., Strobel, C.S., Hale, S.S., Cobb, D. & Brown, B.S. (2003) Ecological condition of US mid-Atlantic estuaries, 1997–1998. *Marine Pollution Bulletin* 46: 1224–1244.

Ladson, A.R., White, L.J., Doolan, J.A., Finlayson, B.L., Hart, B.T., Lake, P.S. & Tilleard, J.W. (1999) Development and testing of an index of stream condition for waterway management in Australia. *Freshwater Biology* 41(2): 453–468.

Lockie, S. & Jennings, S. (2002) *Central Queensland healthy waterways survey*. Cooperative Research Centre for Coastal Zone, Estuary and Waterway Management, Brisbane.

Mallin, M.A., Ensign, S.H., McIver, M.R., Shank, G.C. & Fowler, P.K. (2001) Demographic, landscape, and meteorological factors controlling the microbial pollution of coastal waters. *Hydrobiologia* 460: 185–193.

MBWCP (Moreton Bay Waterways and Catchments Partnership) (2006) *Ecological Health Monitoring Program*. <http://www.ehmp.org/ehmp/> (Accessed 16 June 2006).

Moverly, J., & Hirst, A. (1999) *Estuarine health assessment using benthic macrofauna*. LWRRDC Occasional Paper 18/99. Urban Sub-program Report No 11.

NLWRA (National Land and Water Resources Audit) (2002) *Australian catchment, river and estuary assessment 2002.* Volume 2, pp. 193–386. National Land and Water Resources Audit, Commonwealth of Australia, Canberra.

Paul, J.F., Stroebel, C.J., Melzian, B.D., Kiddon, J.A., Latimer, J.S., Campbell, D.E. & Cobb, D.J. (1998) State of the estuaries in the mid-Atlantic region of the United States. *Environmental Monitoring and Assessment* 51: 269–284.

Paul, J.F., Comeleo, R.L. & Copeland, J. (2002) Landscape and watershed processes; landscape metrics and estuarine sediment contamination in the mid-Atlantic and Southern New England Regions. *J. Environ. Qual*. 31: 836–845.

Peirson, W.L., Bishop, K., van Senden, D., Horton, P.R. & Adamantidis, C.A. (2002) *Environmental water requirements to maintain estuarine processes.* Environmental Flows Initiative Technical Report No. 3, Commonwealth of Australia, Canberra.

Pillans, S., Pillans, R.D., Johnstone, R.W., Kraft, P.G., Haywood, M.D.E. & Possingham, H.P. (2005) Effects of marine reserve protection on the mud crab *Scylla serrata* in a sex-biased fishery in subtropical Australia. *Mar. Ecol. Prog. Ser.* 295: 201–213.

Queensland Department of Natural Resources, Mines and Water (2006) *Presence/extent of litter (Indicator status: for advice).* <http://www.nrm.gov.au/monitoring/indicators/estuarine/presence-of-litter.html#analysis> (Accessed 16 June 2006).

Scheltinga, D.M., Counihan, R., Moss, A., Cox, M. & Bennett, J. (2004) *Users' guide for estuarine, coastal and marine indicators for regional NRM monitoring*. Cooperative Research Centre for Coastal Zone, Estuary and Waterway Management, Brisbane.

Skilleter G.A. & Stowar M. (2001) *Development of an assessment scheme for estuarine health in SE Queensland.* Final report to Coast and Clean Seas. Prepared by Marine and Estuarine Ecology Unit, Department of Zoology and Entomology, University of Queensland, Brisbane.

Uncles, R.J., Stephens, J.A. & Smith R.E. (2002) The dependence of estuarine turbidity on tidal intrusion length, tidal range and residence time. *Continental Shelf Research* 22: 1835–1856.

US Ocean Conservancy (2002) National Marine Debris Monitoring Program <www.oceanconservancy.org/ site/PageServer?pagename=mdm_debris> (Accessed 16 June 2006).

Vollenweider, R.A. (1971) *Scientific fundamentals of the eutrophication of lakes and flowing waters, with particular reference to nitrogen and phosphorous as factors in eutrophication*. OECD, Paris.

Ward, T., Butler, E. & Hill, B. (1998) *Environmental indicators for national State of the Environment reporting – Estuaries and the sea*. Australia: State of the Environment (Environmental Indicator Reports), 81 pp. Department of the Environment, Canberra.

Waugh, P.S. & Padovan, A.V. (2004). *Review of pesticide monitoring, use and risk to water resources in the Darwin region.* Northern Territory Government Department of Infrastructure, Planning and Environment, Darwin.

Webb, McKeown and Associates, Pty Ltd (1999) Wallis Lake estuary processes study. *Journal of Soil Research* 10: 127–42.

WSAA (Water Services Association of Australia) (2003) *Catchments for recreational water: sanitary inspections*. Occasional Paper No. 8, WSAA, Melbourne.

Appendix. Set of detailed information on stressors and related pressure, vulnerability and condition indicators for practical application of the IEAF framework

Contents

Presented in this appendix are:

- Detailed information on each of the stressors selected for use in an example of application of the IEAF framework
- Proposed pressure, vulnerability and condition indicators for each stressor
- Proposed categories for scoring indicators.

This information allows users to apply the framework for assessing estuarine condition that is described in the main report.

Stressors considered in this document

The stressors considered here are those proposed in Section 3 of the report. They are reproduced below in Table A1, and addressed in sequence throughout the remainder of this document.

Table A1. List of estuary stressors considered in this example

Stressor	Examples of causes
Pollutants:	
Organic matter	Abattoir discharge
Fine sediments	Urban development
Acid runoff	Drainage of acid soils
Nutrients	Sewage discharge
Heavy metals	Mine waste
Pesticides and organics	Agricultural use
Oil	Marina operations
Pathogenic micro-organisms	Sewage discharge
Litter	Urbanisation
Riparian habitat removal or disturbance	Loss of mangroves
Direct removal of biota	Fishing, bait collection
Freshwater inflow alteration	Reductions in inflows due to dams
Alteration to hydrodynamics	Dredging of entrances
Pest species	Introduced dinoflagellates
Shoreline development	Urbanisation

A.1. Organic matter pollution

A.2.1. Background information

Organic matter is any matter derived from a biological source (e.g. plants, animals and bacteria). Organic matter can enter aquatic systems from external sources (catchment runoff or point-source discharges) or may be generated within the system via photosynthesis. From either source, the organic matter is a primary driver of aquatic food webs. Living organic matter is consumed by secondary producers while dead organic matter is metabolised by bacteria and the breakdown products fuel further production.

Bacterial metabolism requires oxygen and therefore the breakdown of organic matter places an oxygen demand on the system. In undisturbed systems the demand is relatively small and has minimal impact on the system. However, unnaturally large organic loads entering systems can stimulate bacterial activity (and thus increase oxygen demand) to the point where oxygen levels in the system may be reduced to very low levels. This in turn may cause the death of some biota. *The increased oxygen demand and subsequent impact on dissolved oxygen levels caused by organic matter is the main stress factor considered here.*

Different types of organic matter vary greatly in the rate at which they can be broken down. Some types of organic matter break down very quickly and therefore create a strong short-term demand on oxygen, resulting in rapid reductions in oxygen levels. Other types of matter (e.g. cellulose) break down very slowly and therefore have much less marked effects on oxygen levels.

Causes

The main sources of organic matter are:

- **Point sources:** Most commonly discharges of treated sewage or effluent from abattoirs or sugar mills. Discharges from point sources are generally consistent over time and thus create a constant oxygen demand on a system.
- **Catchment sources:** Organic matter from catchment sources enters estuaries mainly during rainfall events. This can result in large loads entering in a short period, which in turn can cause very significant oxygen depletion for periods of a few days following the event.

- **Internal sources:** Excessive nutrient loading to a system can lead to algal or macrophyte blooms. Breakdown of the organic matter created by blooms can of itself lead to low dissolved oxygen levels.

Symptoms

The main symptom of excess organic matter loads is reduced dissolved oxygen levels. At very low levels this can result in the death of biota—generally fish and crustacea.

A.1.2. Pressure indicators and scoring categories

The oxygen demand of organically rich waste is commonly measured as the biochemical (or biological) oxygen demand over 5 days (BOD_5). This measures the amount of oxygen consumed by bacteria over that time period.

For external sources of organic matter, the recommended pressure indicator is the BOD_5 load entering the estuary each day:

BOD_5 load (mg/d) = *BOD_5 of effluent/runoff (mg/L) * volume of effluent (L) /day*

This should then be normalised to the volume of the whole estuary or a segment of the estuary:

Areal BOD_5 load (mg/m^2/d) = *BOD_5 of effluent/runoff (mg/L) * volume of effluent (L)/ volume of estuary/day*

Assessing BOD_5 loads in discharges is straightforward since flows and quality are relatively consistent. Measuring BOD_5 in catchment runoff is much more demanding. It requires taking measurements of both flow and BOD_5 at frequent intervals during a runoff event. The alternative to direct measurements is to estimate loads using coefficients from the literature. These coefficients provide estimates of BOD_5 loads that can be expected to be contributed per unit area of a particular land use. However, such estimates are very approximate and it is preferable to take direct measurements.

Determining the surface area of estuary to be considered in calculation of the areal loading rate requires consideration of the likely rate of dispersion of the effluent. For point sources, the worst affected area is likely to be the area within which the effluent is dispersed within the first 24 hours. This area can be estimated from mixing models or if necessary from simple approximations. The BOD load should then be normalised to that area. Note that in tidal estuaries the dispersion area includes reaches upstream of the discharge as well as downstream.

For catchment sources, the area to be considered will be that part of the estuary affected by freshwater inflows. The extent of this will vary depending on the size of the event and therefore needs to be determined on a case by case basis.

Some preliminary proposed categories for BOD_5 loading are shown in Table A2.

Table A2. Scoring categories and indicator values for BOD_5, as an indicator of stress from organic matter pollution

Stressor: *Organic matter pollution* Pressure indicator: *BOD_5 loading*	
Scoring category	**Indicator value (g BOD_5 / m^3 /day)**
1	< 0.2
2	> 0.2 and < 0.5
3	> 0.5 and < 2 g
4	> 2 and < 5
5	> 5

A.1.3. Vulnerability indicators and scoring categories

The vulnerability of systems to organic loads is largely related to flushing rates, depth and vertical mixing rates. Flushing is mostly determined by tidal range, although the status of the estuary entrance is also important. In estuaries with small tidal ranges, the organic matter tends to be concentrated in a smaller area and therefore has a proportionately larger effect in that zone. Slow exchange rates also limit the rate at which low oxygen water can be replaced with cleaner oxygen rich water. Estuaries with small tidal ranges also experience low rates of vertical mixing which reduces the rates of re-aeration of the water.

Shallow systems are less vulnerable because re-oxygenation from the surface more rapidly penetrates the whole water column. Conversely, deeper systems and particularly those with poor vertical mixing are much more vulnerable and are likely to develop very low oxygen or even anoxic conditions in their deeper parts. This effect is greatly accelerated if there is thermal or salinity stratification present.

Table A3. Scoring categories and indicator values for flushing rate as an indicator of vulnerability to stress from organic matter pollution

Stressor: *Organic matter pollution* Vulnerability indicator: *Flushing rate*	
Scoring category	**Indicator value**
1	Receiving reach very well flushed—tidal range > 2.0 m and close to mouth of estuary; macro-tidal estuary
2	
3	Receiving reach moderately well flushed—tidal range > 2.0 m; extensive tidal movement; not an upstream reach of estuary
4	
5	Receiving reach of estuary very poorly flushed—micro-tidal estuary; coastal lagoon estuary or upstream reach of estuary

A.1.4. Condition indicators and scoring categories

The recommended condition indicator is dissolved oxygen (DO), measured as percent saturation. Where consistent point sources are the main concern, measurements can be taken at any time. The location of monitoring sites will be dictated by the likely spread of influence of the organic matter and this may need to be assessed through pilot studies. In poorly mixed systems, measurements should be taken at a range of depths. In systems with high levels of primary production, it is desirable to take DO readings in the early morning which is when minimum values occur.

Where a catchment source is the main threat, then measurements need to be undertaken at least daily following a significant runoff event. Such events cause only short-term oxygen depletion but the effects may be very large. Therefore it is important in order to capture the DO minimums which commonly occur days or even a week or two after the main inflow has ceased. The exact time period depends on the size of the system and the size of the event.

Table A4. Scoring categories and indicator values for minimum values of dissolved oxygen as an indicator of stress from organic matter pollution

Stressor: *Organic matter pollution* Condition indicator: *Dissolved oxygen*	
Scoring category	**Indicator value (DO % saturation)**
1	DO > 70% saturation at all times
2	DO > 50% saturation at all times
3	DO falls below 50% saturation for periods of > 24 hours
4	DO falls below 35% saturation for periods of > 24 hours
5	DO falls below 20% saturation for periods of > 24 hours

A.2. Fine sediment pollution

A.2.1. Background information

Fine sediments are defined as sediment particles having a diameter less than 63 µm. Fine sediments enter estuaries as a result of natural landscape erosion processes. Within the estuary they either become deposited in low energy areas (e.g. mangroves), forming muddy deposits, or are advected out of the estuary via tidal currents and are deposited in nearshore coastal areas.

Human disturbance of catchments (mainly the clearing of vegetation) has resulted in large increases in the loads of fine sediments entering estuaries. Such increases lead to a general increase in the “muddiness” of estuarine and nearshore coastal areas. Specific impacts include:

- increased turbidity and consequent reduced light availability which affects species such as seagrasses
- smothering of some benthic habitats
- changes in channel morphology
- changed water depth.

There is anecdotal evidence that seagrass areas in and around estuary mouths have been lost in many parts of Queensland over the past hundred years and this is thought to be due to increased turbidity caused by fine particulates.

Other human disturbances such as dredging, alteration of hydrodynamics and boat wash cause increased turbidity and also affect the distribution of fine sediments within estuaries. The presence of large impoundments in catchments may trap some fine sediments and thus reduce loads. However, the associated reduction in freshwater inflows reduces flushing of the estuary which often results in increased siltation, particularly in upper estuary reaches.

Factors that may work to modify these impacts include the tidal exchange rate; estuaries with higher tidal exchange are less likely to retain additional sediment due to higher flushing rates than those with smaller velocities. The natural turbidity of estuaries also varies widely depending on the estuary type. Macrotidal estuaries tend to be highly turbid due to high tidal velocities. At the other end of the scale, coastal lagoons tend to have much lower turbidities because of their negligible tidal flows.

Causes

The main sources of fine sediment pollution are:

- Altered sediment loads from catchment (urban and rural)
- Loss of vegetation on banks or dunes
- Point-source changes (e.g. sand washing, primary treated sewage)
- Reduction in sediment input (e.g. from impoundments in catchment)
- Estuary bank erosion (e.g. resulting from boat wash or other disturbance).

Symptoms

Symptoms include:

- Turbid water, changed light penetration
- Changed water depth
- Bottom vegetation (e.g. seagrass) lost by smothering or lower light availability
- Changed sediment grain size (e.g. muddier or sandier)
- Erosion or sediment deposition patterns changed
- Benthic animals lost due to smothering or suspended sediments.

A.2.2. Pressure indicators and scoring categories

Total diffuse and point source fine sediment load entering the estuary

There are three main sources of sediment to the estuary:

- Diffuse catchment load
- Diffuse estuarine load (from estuarine erosion)
- Point-source load.

Of these, catchment loads are usually strongly dominant although estuarine erosion may be significant in local areas.

There are various ways to calculate the total load from these sources. In general, if it is possible to measure the total load, this will be the best possible estimate. Where this is not possible, modelling may be undertaken to estimate the loads. In the absence of sufficient information to conduct rigorous modelling, the indirect pressure indicators may be used. Pressure indicators are listed below in order of preference (and complexity of data requirements).

Total load (measured)

The total load is best measured as the point source load plus the diffuse load. In most instances, the diffuse load will be greater. In most areas, point source loads should be licensed. The licence conditions should set out the maximum sediment load allowed for discharge; in some cases, the licensee will monitor the actual load discharged. This information may be obtained from the licensing authority or from the licensee directly. Note that this point source includes for example sediment extraction processes with discharges back to the estuary (gravel-wash processes).

The diffuse estuarine load is difficult to measure or model, but may be measured directly using sediment traps. Sediment loads from the catchment may also be measured using flow weighted automatic samplers; however, the load entering from the estuarine flood plains will be much harder to estimate.

OR ***Total load (modelled)***

Where loads cannot be measured directly, models such as SedNet (<www.toolkit.net.au/sednet>) may be used to estimate the load based on land-use and soil information. Equations such as the revised universal soil loss equation (RUSLE) may be used to determine erosion rate based on current land use.

The RUSLE calculates mean annual soil loss (tonnes/hectare/year) as:

Annual soil loss = R x K x L x S x C x P

Where: R is rainfall erosivity factor
K is soil erodibility factor
L is hillslope length factor
S is hillslope gradient factor
C is ground cover factor
P is land-use practice factor.

Table A5. Scoring categories and indicator values for sediment load entering the estuary as an indicator of stress from fine sediment pollution

Stressor: *Fine sediment pollution* Pressure indicator 1: *Fine sediment load entering the estuary*	
Scoring category	**Indicator value* (kg fine sediment/year/m^3 estuary volume)**
1	< 5
2	
3	5–10
4	
5	> 10

*Values derived from Webb *et al.* (1999) but with converted units

Indirect pressure indicators

Indirect pressure indicators for fine sediment loads include:

- % cleared area in catchment
- length shoreline eroded.

Percent cleared area

Where sediment load information is not available, the percentage of the catchment that has been cleared of vegetation may be used as a surrogate. This is based on the assumption that if bare areas expand as the result of human activities, then accelerated erosion pressure increases. Percent cleared area is best estimated using land-cover maps and geographic information systems (GIS). Maps showing native vegetation cover can be found at the Catchment Condition Online Maps website at the Commonwealth Department of Agriculture, Fisheries and Forestry (DAFF 2006). The percentage of native vegetation can be estimated as a proportion of catchment area. The area of cleared land can also be determined using topographic maps.

Eroded shoreline length as a proportion of total shoreline length

The eroded shoreline length can be determined by viewing aerial photographs and site surveys of the estuary banks.

Table A6. Scoring categories and indicator values for percent surface area cleared as an indicator of stress from fine sediment pollution

Stressor: ***Fine sediment pollution*** **Pressure indicator 2:** ***Percent surface area cleared* in catchment***	
Scoring category	**Indicator value (% cleared surface area in catchment)**
1	< 30
2	30–49
3	50–65
4	66–80
5	> 80

* ‘Cleared’ is defined as every land use except water, nature conservation, minimal use, production forestry (i.e. all cropping, grazing, horticulture, manufacturing and urban). However, grazing is something of an in-between category. Low intensity grazing with minimal clearance of native vegetation might be categorised as uncleared while more intense grazing with loss of significant vegetation could be classified as cleared. This would need to be assessed on a catchment by catchment basis.

A.2.3. Vulnerability indicators and scoring categories

The vulnerability of estuaries to increased fine sediment load is related in part to their natural levels of clarity. Natural values of turbidity in estuaries vary (Uncles *et al.* 2002) depending on:

- The tidal range: Estuaries with larger tidal ranges tend to have higher tidal velocities and these act to keep particles in suspension thus causing higher turbidity levels
- The estuary length: Longer estuaries are flushed more slowly than short estuaries and therefore tend to retain suspended particulate loads for much longer.

Thus, at one extreme, long macrotidal estuaries have naturally very high levels of turbidity while at the other extreme, coastal lagoons have naturally low levels of turbidity. Increases in fine particulate loads to already very turbid estuaries are likely to have much less impact than increased loads to systems that have high natural clarity and that have biota adapted to such conditions. Vulnerability is therefore estimated from tidal range and tidal length.

The tidal range is defined as the difference between the mean high water tide and the mean low water tide at the mouth of the estuary. This information is usually available from state transport departments. The tidal length is defined as the length of the estuary that experiences tidal water flow. In unregulated rivers this varies with type of tide and amount of rainfall and an average value is required. State departments responsible for water resources may have defined tidal reaches for extraction purposes.

Tidal ranges (mean spring) have been classified as follows:

1. < 1 ultra-microtidal
2. 1–2 microtidal
3. 2–4 mesotidal
4. 4–6 macrotidal
5. > 6 hypertidal.

The following table (Table A7) is a two-way matrix of tidal range and tidal length and the values within the table represent estimated vulnerability scores.

Table A7. Matrix of tidal length and mean spring tidal range, with estimated scores indicating an estuary's vulnerability to fine sediment pollution

Stressor: ***Fine sediment pollution*** **Vulnerability indicators:** ***Tidal length and mean spring tidal range***					
Tidal length	**Mean spring tidal range**				
	> 6	**4–6**	**2–4**	**1–2**	**< 1**
> 75	1	3	4	5	5
26–75	1	2	2	4	5
10–25	1	2	2	3	4
< 10	1	1	1	2	3

A.2.4. Condition indicators and scoring categories

Turbidity (or Secchi depth or suspended solids)

The aim of these indicators is to assess the amount of suspended fine particulates and at the same time to provide an estimate of their effect on light penetration. Suspended solids is a direct measure of suspended particulates, while Secchi depth is an indicator of light penetration. Turbidity is an artificial measure that combines attributes of both. It is possible to derive relationships between these indicators but these tend to be specific to individual waterways. Unless the actual mass of suspended solids needs to be assessed, the use of either turbidity or Secchi depth is recommended, and preferably both.

Specifically, turbidity is a measure of the light scatter as a result of particles in the water (including particles other than sediment, such as algae or organic matter). Turbidity is measured most commonly using a nephelometer. Measures should be taken in the field, at various depths. The optimal frequency for turbidity measurements depends on local conditions: in tidal systems, turbidity varies markedly during a single tidal cycle; in some coastal or oceanic systems, however, wind speed may produce the most significant variation. Secchi depth is a measure of water clarity and is measured using a Secchi disc, as described in Australian Standard AS 3550.7–1993. Total suspended solids is a measure of the total weight of particles in the water column, and is analysed in the laboratory from field samples.

As a result of the natural variation in turbidity caused by tidal range and estuary length, it is not possible to provide a single set of scoring categories for all estuaries. The categories below are applicable to estuaries with tidal ranges less than 1 m and to most estuaries with a length of 25 km or less. For macrotidal estuaries and especially those longer than 25 km, site-specific categories need to be determined.

Table A8. Scoring categories and indicator values for turbidity as an indicator of fine sediment pollution

Stressor: *Fine sediment pollution* Condition indicator 1: *Turbidity*	
Scoring category	**Indicator value (Nephelometric turbidity units, NTU)**
1	< 5
2	5–10
3	11–20
4	21–40
5	> 40

Seagrass depth range

Seagrass depth range is defined as the difference between the shallow and deep distributional limits of the dominant seagrass species, measured by an autoset level. The assumption is that the deep distribution limit is constrained by light availability. Seasonal patterns in distribution should be taken into account in interpretation.

These will be dependent on local conditions—see Ecological Health Monitoring Project technical reports for examples of application of this technique (MBWCP 2006).

Loss of light-dependent or sessile biota

Sessile biota may be smothered by excessive sedimentation. Biota species that are dependent on light (e.g. corals) may also be negatively affected by reduced clarity as a result of increased sediments. Changes in population abundance or distribution of these species may be a useful indicator of the effects of sediments in the water column.

Scoring categories are:

1. Abundant light-dependent or sessile species
3. Rare light-dependent or sessile species
5. No light-dependent or sessile species, although other conditions are appropriate.

Table A9. Scoring categories and indicator values for presence of light-dependent species as an indicator of fine sediment pollution

Stressor: ***Fine sediment pollution*** **Condition indicator 2:** ***Presence of light-dependent species***	
Scoring category	**Indicator value (Presence of light-dependent species)**
1	Light-dependent species abundant and in good condition
2	
3	Light-dependent species present but with patchy distribution and/or poor condition
4	
5	Light-dependent species absent, although other conditions are appropriate

A.3. Acid runoff pollution

A.3.1. Background information

Acid runoff is caused by exposure of sulphides to oxygen and their subsequent oxidation and conversion to sulphuric acid in the presence of water. Sulphide-rich rocks or soils occur naturally in subsurface areas where oxygen does not normally penetrate. Exposure of sulphide-rich deposits may occur due to mining activity or to the disturbance of acid sulfate soils (ASS) by agricultural activity or construction (e.g. tidal canals). Acid sulfate soils are by far the most common cause of acid runoff entering estuaries because many coastal flood plains contain large deposits of subsurface sulphide-rich soils.

The main effect of acid runoff is to reduce pH levels but it can also mobilise heavy metals which have additional toxic effects. Because of the high buffering capacity of sea water, small amounts of acid runoff may not have noticeable effects. However, large acid runoff events can cause major mortality of both fish and benthic organisms.

A.3.2. Pressure indicators and scoring categories

Acid runoff occurs largely during storm events. An ideal pressure indicator would be some measure of the acid load entering the estuary. Although this type of data is rarely collected, the development of better *in situ* pH meters means that this is now quite a practical undertaking. The indictors would be pH and a measure of flow. This data would allow a rough measure of acid load to be calculated.

Alternatively, pressure on a system could be estimated using catchment characteristics. It is suggested that only three of the possible five categories (as shown in Table A10) be used.

Table A10. Scoring categories and indicator values for presence or disturbance of acid sulfate soils as an indicator of acid runoff pollution

Stressor: ***Acid runoff pollution*** **Pressure indicator:** ***Presence/disturbance of acid sulfate soils***	
Scoring category	**Indicator value (Presence/disturbance of ASS)**
1	ASS not present or if present not disturbed in any way
2	
3	ASS present and subject to limited disturbance e.g. through construction of canal estates
4	
5	ASS widespread and subject to broadscale disturbance e.g. through agricultural activity

A.3.3. Vulnerability indicators and scoring categories

Vulnerability to acid runoff is largely a function of the size of the estuary and its flushing rate. Suggested categories are shown in Table A11.

Table A11. Scoring categories and indicator values for flushing rate as an indicator of vulnerability to acid runoff pollution

Stressor: ***Acid runoff pollution*** **Vulnerability indicator:** ***Flushing rate***	
Scoring category	**Indicator value**
1	Well flushed estuary with no barrier or bar at its mouth
2	
3	Estuary moderately well flushed or if less well flushed then a large volume relative to the likely volume of runoff
4	
5	Small, poorly flushed estuary – a coastal lagoon or tidal range < 1 m

A.3.4. Condition indicators and scoring categories

The recommended condition indicator is pH. Note however that due to the transient nature of acid runoff, pH measurements need to be concentrated in post-runoff event periods and be fairly frequent. Ideally, measurements would be taken using an *in situ* recording pH meter. Alternatively, measurements can be taken using hand-held instruments if someone living locally is able to do this at frequent intervals. Note that even short-lived episodes of low pH can be lethal to fish and benthic organisms. Thus the minimum sustained pH values over a period of a few hours should be used to categorise condition.

Table A12. Scoring categories and indicator values for pH as an indicator of acid runoff pollution

Stressor: ***Acid runoff pollution*** **Condition indicator:** ***Minimum sustained pH values***	
Scoring category	**Indicator value (Minimum sustained pH values during the days following an inflow event)**
1	> 7.0
2	6.1 – 7.0 (based on sustained minimum values over a few hours)
3	5.1 – 6.0 (based on sustained minimum values over a few hours)
4	4.1 – 5.0 (based on sustained minimum values over a few hours)
5	< 4.0 (based on sustained minimum values over a few hours)

A.4. Nutrient pollution

A.4.1. Background information

The nutrients nitrogen and phosphorus are essential for the growth of plants. Sources of nutrients include point-source discharges (particularly sewage discharges) and runoff from urban and rural areas. Catchment areas that have been cleared of vegetation typically provide more nutrients than areas in their natural state. Potential consequences of increased nutrient loads to estuaries include eutrophication, algal blooms or excessive macrophyte growth, anoxic events due to decay of plant matter, and fish or animal kills from lack of oxygen.

A.4.2. Pressure indicators and scoring categories

The most direct pressure indicator of nutrients is the load entering a system. Nutrient loads entering estuaries from the catchment can be measured directly by monitoring the nutrient concentrations and volumes of inflow waters and using this information to calculate the load. Since most nutrient from the catchment enters during large flow events, this requires intensive sampling during these events, which usually last only a few days. Undertaking this type of program requires significant resources and expertise. The alternative is to estimate loads from the output of catchment runoff models such as SedNet (CSIRO 2002). This will give less precise load estimates but is much cheaper and easier to undertake.

Nutrient loads entering from point-source discharges can be measured more readily since the discharges generally have relatively constant flows and nutrient concentrations.

Nutrient loads are commonly expressed as loads per annum. However, because catchment loads mainly enter during occasional storm events, these loads are highly variable. It is suggested therefore that loads should be expressed as loads per week. For catchment loads this means that loads for most weeks will be close to zero while for two or three weeks a year they will be very high. It is these high loads that are of interest since these are what cause the main impacts. Point-source loads should remain consistent over the year. However, expressing loads on a weekly basis allows a better comparison of point and diffuse loads.

A given nutrient load will obviously affect a small estuary more than a large one. Therefore, to allow comparison between estuaries, loads need to be normalised to the surface area of the estuary.

Proposed scoring categories for nutrient loads are given in Tables A13 (phosphorus) and A14 (nitrogen).

Table A13. Scoring categories and indicator values for total phosphorus load as an indicator of stress from nutrient pollution

Stressor: *Nutrient pollution* Pressure indicator 1: *Total phosphorus load*	
Scoring category	**Indicator value (P loading kg/ha/yr)**
1	0 – 0.6
2	0.7 – 1.1
3	1.2 – 1.65
4	1.66 – 2.2
5	> 2.2

Table A14. Scoring categories and indicator values for total nitrogen load as an indicator of stress from nutrient pollution

Stressor: *Nutrient pollution* Pressure indicator 2: *Total nitrogen load*	
Scoring category	**Indicator value (N loading kg/ha/yr)**
1	0 – 5.5
2	5.6 – 11
3	11.1 – 16.5
4	16.6 – 22
5	> 22

A.4.3. Vulnerability indicators and scoring categories

Two measures used to determine the vulnerability of an estuary to excessive nutrient loadings are its flushing rate and dilution efficiency (used for event-based rainfall runoff as a proportion of estuary volume).

Flushing rate

An estuary's flushing rate is the most important factor affecting its vulnerability to excessive nutrient loads. Macrotidal estuaries quickly disperse nutrient loads over large areas while, at the other end of the spectrum, dispersion in coastal lagoons is very slow. Flushing time for a particular segment of an estuary is defined by convention as the time (in days) it takes for the concentration of a conservative tracer to be reduced to a concentration of 1/e of its original concentration (where 'e' is the base for natural logarithms). This convention is used because the flushing of a substance follows an asymptotic function and therefore complete flushing theoretically takes an infinite amount of time. This is of no practical use and therefore the value 1/e is used as a default convention.

The time taken to achieve 1/e (i.e. the flushing time) can be determined from hydrodynamic models if these are available. If these are not available, then flushing rate will have to be estimated by a technique that is appropriate to the particular system. In estuaries with very low base inflows, the rate of re-establishment of salinity (which acts as a conservative tracer) following a flood event which completely flushes out an estuary with fresh water can be used to estimate flushing rates. In estuaries with significant base freshwater inflows, the rate at which the estuary water is flushed out by fresh water will be the most significant factor in calculating flushing times.

Proposed scoring categories for flushing rates are given in Table A15.

Table A15. Scoring categories and indicator values for flushing rate as an indicator of vulnerability to stress from nutrient pollution

Stressor: *Nutrient pollution* Vulnerability indicator 1: *Flushing rate*	
Scoring category	**Indicator value (Flushing rate in days)**
1	0 – 10
2	11– 20
3	21 – 30
4	31 – 40
5	> 40

Dilution efficiency (event-based runoff/estuary volume)

In the even of heavy rainfall, the smallest dilution ratios have the greatest dilution capacity, that is, the influence of fresh water would be minimal.

Proposed scoring categories for dilution efficiency are given in Table A16.

Table A16. Scoring categories and indicator values for dilution efficiency as an indicator of vulnerability to stress from nutrient pollution

Stressor: ***Nutrient pollution*** **Vulnerability indicator 1:** ***Dilution efficiency***	
Scoring category	**Indicator value** **(Event-based runoff/estuary volume)**
1	0 – 20
2	21 – 45
3	46 – 79
4	80 – 230
5	> 230

A.4.4. Condition indicators and scoring categories

Chlorophyll a

Concentrations of chlorophyll *a* are a useful indicator of algal biomass in the water column. All algae contain chlorophyll *a* and changes to the concentrations of chlorophyll *a* imply increased biomass, which has been shown to respond to increases in nutrients.

Table A17. Scoring categories and indicator values for chlorophyll *a* as an indicator of stress from nutrient pollution

Stressor: ***Nutrient pollution*** **Condition indicator 1:** ***Chlorophyll*** **a** ***in the water column***	
Scoring category	**Indicator value** **(Chl *a* µg/L)**
1	0 – 3
2	4 – 6
3	7 – 9
4	10 – 12
5	> 12

Macroalgal extent

Macroalgae are filamentous algae which have rapid growth rates and which respond rapidly to increased nutrients. They generally occur in shallow regions of estuaries.

Macroalgal biomass or extent (% of mud flats less than 2 metres deep covered with filamentous algae) is usually measured by visual estimate.

Proposed scoring categories for macroalgal extent are given in Table A18.

Table A18. Scoring categories and indicator values for macroalgal extent as an indicator of stress from nutrient pollution

Stressor: *Nutrient pollution* Condition indicator 2: *Macroalgal extent*	
Scoring category	**Indicator value (% of biomass in shallow mudflats)**
1	Filamentous algae (0–2% of estuary shallows < 2 m depth)
2	Filamentous algae (3–10% of estuary shallows < 2 m depth)
3	Filamentous algae (11–20% of estuary shallows < 2 m depth)
4	Filamentous algae (21–30% of estuary shallows < 2 m depth)
5	Filamentous algae (>30% of estuary shallows < 2 m depth)

A.5. Heavy metal pollution

A.5.1. Background information

Heavy metals, metalloids and organometallics are used in many common items and may enter coastal waterways through point-source discharges (e.g. industrial or sewage discharges) or through diffuse runoff. Most point-source discharges are controlled by licence. Concentrations in diffuse runoff are usually higher in runoff originating from urban areas; for example, copper is used in car brake linings and is therefore higher in areas with high road density; zinc concentrations in runoff may be related to roof density. High concentrations of heavy metals in coastal waters can lead to health problems in aquatic biota, including diseases and fish kills. Human health problems can also result from consumption of contaminated seafood.

Causes

The main causes of high concentrations of heavy metals are:

- **Point sources:** industrial discharge, sewage treatment plant discharge, dumping of toxicants or wastewater
- **Diffuse sources:** catchment runoff (rural and urban pesticide use).

Symptoms

Symptoms include:

- Poor water quality: toxicant concentrations
- Animal (fish/macrobenthos) kills
- Animal (fish) disease, lesions, mutations, aberrant growth and reproductive, neurological or respiratory dysfunction
- Change in animal or plant population or loss of species (particularly toxicant-sensitive species)
- Shellfish/fisheries closures.

A.5.2. Pressure indicators and scoring categories

The best pressure indicator is the total (measured) load of each metal entering the estuary. Ideally, this would be calculated by measuring the metal loads in all point sources (this is often required for discharge licences), and by measuring the total diffuse load entering the estuary (e.g. using an event sampler at the head of the estuary). In practice, however, this will almost always be impossible. In

Australia, cities or towns are often located along much of the estuary, making it almost impossible to measure the urban diffuse load, and in any case there are rarely sufficient resources to conduct regular event monitoring for metals. In virtually all cases, therefore, the diffuse load will have to be estimated. The most common method of estimation is by using coefficients from the literature that give estimates of the loadings of metals per unit area of catchment. These coefficients depend on land use and, for metals, by far the highest coefficients are for urban areas. It is therefore possible to obtain an estimate of the relative loading of metals from the urban area within the catchment.

Where point sources are present, loads from these will have to be estimated from licence discharge data or, in the case of old mine sites, from the best information available.

More detailed information on potential indicators is given below.

Total metal load entering the estuary (point source + diffuse)

This is measured at point of entry to the estuary, for example, at end of pipe for point sources, head of estuary for catchment diffuse sources or at entry point (e.g. stormwater pipe) for other diffuse sources. Given the large number of metals and the variation in geology of catchments, it is not possible to give precise values for loadings. Instead, it is proposed that loads be expressed as a proportion of loads under natural conditions.

Table A19. Scoring categories and indicator values for metal load levels as an indicator of stress from heavy metal contamination

Stressor: ***Heavy metals pollution*** **Pressure indicator:** ***Metals load to estuary***	
Scoring category	**Indicator value** **(Metal loads to estuary)**
1	Metals loads close to natural
2	
3	Metals loads measurably greater than natural e.g. due to presence of normal urban areas but with no major metals industries
4	
5	Metals loads significantly higher than natural, due to industrial activities or presence of metalliferous mine sites

A.5.3. Vulnerability

The distance between the estuary and the impervious areas affects the loads of metals that actually reach the estuary. If this indicator is used, it should be used to weight the percentage impervious area such that areas close to estuaries have more influence than those further away.

Levels of flushing in the estuary may also be important, particularly for water-soluble toxicants. Patterns of sediment erosion and deposition within the waterway may be more important for sediment-bound toxicants. There is currently no information available to quantify the link between flushing and deposition and toxicant concentrations and impacts.

A.5.4. Condition indicators and scoring categories

Metal concentrations in sediment

Sediment samples should be collected in 375 mL acid-washed glass jars. No preservative or refrigeration is required; samples should be stored for a maximum of 7 days before analysis.

Table A20. Scoring categories and indicator values for metal concentrations in sediment as an indicator of stress from heavy metal contamination

Stressor: ***Heavy metals pollution*** **Condition indicator 1:** ***Concentration of metals in sediment***									
Scoring category	**Indicator values Metals concentration in sediment (mg/kg)**								
	Sb	**Cd**	**Cr**	**Cu**	**Pb**	**Hg**	**Ni**	**Zn**	**As**
1	<2	<1.5	<80	<65	<50	<0.15	<21	<200	<20
2									
3	2–25	1.5–10	80–370	65–270	50–220	0.15–1.0	21–52	200–410	20–70
4									
5	>25	>10	>370	>270	>220	>1.0	>52	>410	>70

Metal concentrations in biota

The effects of metals in biota differ according to species, with some species more sensitive than others. This is particularly true for shellfish, as some species may accumulate some metals but depurate (purify) others. Guidelines for metal concentrations in biota are supplied by Food Standards Australia New Zealand for the protection of human consumers of seafood. Guidelines for the protection of the health of aquatic organisms are not available; however, regional guidelines based on concentrations in biota in relatively unimpaired areas can provide an indication of natural background levels. At least three individuals, or at least 100 g of sample, should be collected for analysis. Samples should be placed in polyethylene bags or wrapping and frozen as soon as possible. The guidelines provided here are relevant to seafood for human consumption only and not to other species.

Proposed scoring categories for metals in biota (condition indicator 2)

Metals in biota (concentrations are sample means unless otherwise stated).

Inorganic arsenic in crustacea or fish (mg/kg)

1. <1
3. 1–2
5. >2

Inorganic arsenic in molluscs (mg/kg)

1. <0.5
3. 0.5–1
5. >1

Cadmium in molluscs (mg/kg)

1. <1
3. 1–2
5. >2

Lead in fish (mg/kg)

1. <0.25
3. 0.25–0.5
5. >0.5

Inorganic arsenic in molluscs (mg/kg)

1. <1
3. 1–2
5. >2

Mercury in crustacea, fish and molluscs (mg/kg) (maximum value for any sample)

1. <0.25
3. 0.25–0.5
5. >0.5

Metal concentrations in water

Water samples should be collected in 250 mL acid-washed plastic containers and nitric acid added as a preservative. No refrigeration is required; samples should be stored for a maximum of one month.

Table A21. Scoring categories and indicator values for metal concentrations in water as an indicator of stress from heavy metal contamination

Stressor: *Heavy metals pollution* Condition indicator 3: *Concentration of metals in water*								
Scoring category	**Indicator values Metals concentration in water (µg/L)**							
	Cd	**Cr III**	**Cr IV**	**Pb**	**Hg**	**Ni**	**Ag**	**Zn**
1	<0.7	<7.7	<0.14	<2.2	<0.1	<7	<0.8	<7
2	0.7–5.5	7.7–27.4	0.14–4.4	2.2–4.4	0.1–0.4	7–70	0.8–1.4	7–15
3	5.6–14	27.5–48.6	4.5–20	4.5–6.6	0.5–0.7	71–200	1.5–1.8	16–23
4	15–36	48.7–90.6	21–85	6.7–12	0.8–1.4	201–560	1.9–2.6	24–43
5	>36	>90.6	>85	>12	>1.4	>560	>2.6	>43

A.6. Pesticides pollution

A.6.1. Background information

Pesticides, herbicides and insecticides are used in both rural and urban areas to control pest plants, insects and other animals. The term pesticides can be used as a general term to encompass any chemical product used to kill a pest, whether the pest is animal or plant. In rural areas, pesticides are generally used to control weeds and insect pests on crops. In urban areas, uses can include termite and pest control in residences, weed control in private and public areas and mosquito control. Pesticides are of concern as they commonly have some effect on non-target organisms, particularly in aquatic systems. Pesticides can enter waterways via a number of pathways, including leaching through the groundwater, surface runoff, soil erosion, aerial drift or spills. Pesticides vary in the time they take to break down (their half-life), the toxicity of breakdown products, their tendency to adsorb to sediment or be taken up by organisms and their toxicity to non-target organisms, and may present problems even after their use has been discontinued. Pesticide residues have been found to be present in many aquatic systems in Australia.

Causes

The main sources of pesticide pollution are:

- Diffuse runoff from catchment (rural and urban)
- Insect control chemicals
- Point sources (industrial, dumping).

Symptoms

Symptoms include:

- Poor water quality: toxicant levels
- Habitat lost or disturbed
- Biota (plants and animals) lost or disturbed; community composition changed
- Animal (fish/macrobenthos) kills
- Animal (fish) disease
- Human health problems (skin irritations, disease, etc.)
- Animal and plant physiology changed
- Seafood catch or stock changed.

A.6.2. Pressure indicators and scoring categories

Total pesticide load entering the waterway

The total pesticide load is made up of the point-source and diffuse loads. In nearly all cases, these will be assessed separately.

The total point-source load can usually be estimated using information from the licences of point sources. In most cases, the licences should set a maximum concentration and total flow allowable, which can be used to calculate the total allowable load (although in many cases the actual load may be less). In some cases, licence conditions require the licensee to monitor concentrations in this discharge; in this case, reasonably exact quantities can be estimated. Information may be available from the licensing authority or direct from the licensee.

In theory, it is possible to measure the total diffuse load of pesticides entering the estuary, but in practice, this will be very difficult. One mechanism to measure diffuse inputs is to use flow-weighted sampling in the major freshwater inputs to the estuary (e.g. in the main river at the head of the estuary). This may involve wet-weather sampling using automated samplers that take samples for later analysis (along with a flow meter), or the use of passive samplers, which are used for time-averaged sampling. Although these methods can give a good estimation of the diffuse inputs to an estuary from one or two major freshwater sources, it will almost never be possible to measure all inputs—for example, if an urban area surrounds an estuary, there will be diffuse runoff through all stormwater outlets at a number of locations. Where the number of inputs is limited (and a suitable budget is available), however, this may be a good option.

An alternative is to estimate the inputs based on the total amount of pesticide used in the catchment, the transport rate of each pesticide, and the distance from the site of application to the estuary. Although this will be a very rough estimate, in many cases it will be the only available option. To estimate the amount of pesticide applied in the catchment, information on total sales from major retailers of pesticides may be sought. This method does not provide any information on the likely spatial distribution of pesticide application within the catchment, or on the timing of the application, and there would be difficulties with this method in areas with many retail outlets. Additional information may be sought from growers to fill these gaps. An alternative is to estimate the amount of pesticide applied based on land-use data, where the typical application rates for each crop are known. This information may also be sourced from local growers. See (Waugh, 2004 p. 126) for an example.

In many estuaries very little of this type of information will be available. Therefore the proposed primary pesticide pressure indicator is a semi-quantitative one based on extent of use of pesticide used in catchment.

Table A22. Scoring categories and indicator values for pesticide use in the catchment as an indicator of stress from pesticide pollution

Stressor: *Pesticide pollution* Pressure indicator: *Pesticide use in the catchment*	
Scoring category	**Indicator value (Extent of use of pesticide in catchment)**
1	Minimal use of pesticide in catchment
2	
3	Some use of pesticide in catchment, urban areas or limited cropping use
4	
5	Significant use of pesticide in catchment e.g. large irrigation areas

Accumulation of pesticides in a passive sampler

If technical support is available, the deployment of passive samplers at the lower end of the freshwater section of the catchment could be a good alternative pressure indicator. These would provide direct measures of pesticides moving from the catchment into the estuary. Assessing the results would rely on advice from the technical support group.

A.6.3. Vulnerability

No advice can be provided for vulnerability indicators at this stage.

A.6.4. Condition indicators and scoring categories

The most straightforward approach to assessing condition is to measure the amounts of pesticide present in the various ecosystem components. Measuring levels in the water column has the advantage that results can be compared directly with ANZECC guideline values, but has the considerable disadvantages that levels are in practice difficult to measure accurately and are highly variable over time. Levels in sediments are much more stable and are much higher than levels in water and therefore easier to measure. Guideline values are also available for sediment levels. Measuring levels in biota has the advantage that it provides a direct measure of the degree to which a pesticide is actually available to the biota. The disadvantage is that there are no generally accepted guidelines with which to compare the measured values. There are guidelines for pesticide concentrations in biota for the purpose of protecting human consumers but these guideline values are not necessarily relevant to protecting the biota themselves.

A recently developed technique involves the use of passive samplers. These are materials that adsorb pesticides from the water column. The amount adsorbed is proportional to (i.e. in equilibrium with) the concentrations in the water column and thus can be used to assess water-column concentrations. One advantage is that the concentrations in the passive sampler are much higher than in the water column and therefore easier to measure. More importantly, the passive sampler concentration (like the sediment and biota) reflects the recent history of concentrations in the water column rather than being an isolated sample and can therefore detect the presence of recent pulses of pesticides.

It is recommended that **sediment concentrations** be used as a primary indicator of condition for pesticides. If resources are available, at least one species of locally important biota should also be tested. If significant levels are detected in sediments and biota have not yet been tested, then testing biota should be the next step. Passive samplers are still in the development stage but they are cheap and easy to use and if technical support is available from a local scientific group then these should also be considered.

Many species (including invertebrates, fish, mangroves and seagrass) are sensitive to pesticides and will exhibit signs of distress (poor growth, low population sizes, low reproductive rates etc.) or will die off in the presence of pesticides. The presence and condition of these species or populations is therefore a useful indicator of the effects of pesticides. However, as the species likely to be affected will vary with region, estuary type and habitat type, it is not possible to provide detailed information here on assessment methods for this indicator. It is recommended that where species that are likely to be affected by pesticides are present, then populations of those species should be monitored for change.

Monitoring for pesticides, whether in sediment or biota, should be directed towards periods when the highest concentrations are likely to be present. This is most commonly during summer when most pesticides are applied and after a significant rainfall event which would have washed residues into the estuary.

Table A23. Scoring categories and indicator values for pesticide levels in sediment as an indicator of stress from pesticide pollution

Stressor: ***Pesticide pollution*** **Condition indicator 1:** ***Pesticide levels in sediment***	
Scoring category	**Indicator value (Pesticide levels in sediment after summer rainfall)**
1	No pesticide residues detected
2	
3	Trace levels of pesticide detected
4	
5	*Significant levels of pesticide detected

*The ANZECC 2000 Guidelines provide information on what are “significant” levels for a few organochlorine residues. For other pesticides, these values will need to be estimated based on their toxicity in water as indicated by the guidelines for water-column concentrations.

Table A24. Scoring categories and indicator values for pesticide levels in a key species of local biota as an indicator of stress from pesticide pollution

Stressor: ***Pesticide pollution*** **Condition indicator 2:** ***Pesticide levels in biota***	
Scoring category	**Indicator value (Pesticide levels in a key species of the local biota)**
1	No pesticide residues detected
2	
3	Trace levels of pesticide detected
4	
5	*Significant levels of pesticide detected

*The ANZECC 2000 Guidelines provide information on what are ‘significant’ levels for a few organochlorine residues. For other pesticides, these values will need to be estimated based on their toxicity in water as indicated by the guidelines for water-column concentrations.

A.7. Oil pollution

A.7.1. Background information

Oil is most commonly sourced from urban runoff or boats. Oil does not have a high toxicity but can cause taste problems in seafood in locations subject to consistent discharges, but this is now a rare occurrence. The incidence of unsightly slicks and the effects on aesthetic values are seen as the main issue with this stressor. Large oil spills from tankers or the like are specific cases and are not considered here.

A.7.2. Pressure indicators and scoring categories

The most likely source of oil slicks is moored boats. Therefore the suggested pressure indicator is the presence (preferably number and size) of boats moored in the estuary.

Table A25. Scoring categories and indicator values for presence of moored boats as an indicator of stress from oil pollution

Stressor: *Oil pollution* Pressure indicator: *Moored boats*	
Scoring category	**Indicator value (Presence of moored boats)**
1	Boats moored only on rare occasions
2	
3	Small numbers of boats permanently moored
4	
5	Large number of boats moored, presence of a port or marinas

A.7.3. Vulnerability

No vulnerability indicators are recommended for this stressor.

A.7.4. Condition indicators and scoring categories

It is suggested that a semi-quantitative visual assessment of the presence of oil slicks is the most useful indicator. Ideally this should be assessed through a routine visual appraisal in order to give objective results. If this is not possible, then the frequency of complaints about oil slicks could be used. Suggested condition categories are:

- Significant slicks never sighted, no complaints received (*category not used)*
- Small number of slicks sighted, low level of complaints (*category not used)*
- Slicks often sighted, consistent complaints received

Table A26. Scoring categories and indicator values for presence of oil slicks as an indicator of stress from oil pollution

Stressor: *Oil pollution* Condition indicator: *Oil slicks*	
Scoring category	**Indicator value (Oils slicks sighted and/or complaints received))**
1	Significant slicks never sighted, no complaints received
2	
3	Small number of slicks sighted, low level of complaints
4	
5	Slicks often sighted, consistent complaints received

A.8. Pathogenic micro-organism pollution

A.8.1. Background information

It is suggested that users refer to the document *Catchments for recreational water: sanitary inspections: Occasional Paper No. 8* (WSAA 2003) for detail on this stressor.

Recommendations for indicators and categories are sourced from the above document.

A.8.2. Pressure and vulnerability indicators and scoring categories

Table A27 is taken directly from the WSAA (2003) document and addresses both pressure and vulnerability indicators and categories. The right-hand column gives a measure of risk from ‘very low’ to ‘very high’.

A.8.3. Condition indicators and scoring categories

Condition indicators and categories are also sourced directly from the WSAA (2003) document. Table A28 shows what we term risk (in their terms ‘significance of contamination’) compared with condition (numbers of faecal streptococci). There are four condition categories applied. The table also derives a risk to human health based on the numbers of faecal streptococci measured.

Table A27. Illustration of the approach to estimating the risk (significance) of faecal contamination in recreational waters (from WSAA 2003)

Source	Concentration (faecal streptococci)	Effect of dilution and dispersion		Effect of time	Resulting concentration (for comparison with monitoring results)[1,2]	Effect of origin of micro-organisms	Resulting concentration (for determining significance)	Significance
		Nature of discharge and receiving water situation	Dilution factor					
Wastewater discharge A (secondary treatment, no disinfection)	10^5	Short outfall close to shoreline and beach	0.04	1	~4 000	1.0	~4 000	Very high
Wastewater discharge B (primary treatment with disinfection)	10^5	High flow rate discharge via long outfall	0.01	1	~1 000	1.0	~1 000	High
Stormwater A (urban, no sewage overflows)	10^4	Drain direct to beach	0.20	1	~2 000	0.5	~1 000	High
Stormwater B (rural, no sewage contribution)	10^4	Drain 500 m up-stream (prevailing current), direct discharge at beach	0.05	1	~500	0.1	~50	Moderate
Bathers (fewer than 20 per 150 m^3	10^1	No dilution; volume of swimming area ~150 m^3	1.00	1	~10	1.0	~10	Low
Rural stream	10^3	Discharge downstream (prevailing current)	0.02	1	~2	0.1	~0.2	Very low
Total					>7 500		>6 000	Very high

Notes:

1. Recordings taken during wet weather.
2. Number or organisms estimated by multiplication across the table, i.e. 10^5 x 0.04 x 1 = 4 000.

Table A28. Comparison of risk and condition of faecal contamination in recreational waters, based on results of monitoring and sanitary inspection (from WSAA 2003)

Risk	Ranking of significance of contamination[2]	Number of faecal streptococci determined from monitoring[1]			
		<40	**40–200**	**201–500**	**>500**
		Risk level inferred from numbers of organisms[3]			
		GI < 1 in 100 exposures AFRI <1 in 300 exposures	**GI < 1 in 20 exposures AFRI <1 in 40 exposures**	**GI < 1 in 10 exposures AFRI <1 in 25 exposures**	**GI > 1 in 10 exposures AFRI >1 in 25 exposures**
	Very low	Very good	Very good	Follow up[4]	Follow up[4]
	Low	Very good	Good	Fair	Follow up[4]
	Moderate	Follow up[4]	Good	Fair	Poor
	High	Follow up[4]	Follow up[4]	Poor	Very poor
	Very high	Follow up[4]	Follow up[4]	Poor	Very poor
	Condition				

Notes:

1. 95th percentile values, as nominated by WHO. Monitoring results with a high degree of variability will need to be reviewed to determine the cause of the variability and the appropriate method for estimating the upper confidence limit. It may be appropriate to distinguish and separately classify different conditions such as wet weather.
2. Ranking based on numbers of faecal streptococci present estimated from sanitary inspection with particular emphasis on human faecal contamination. Order of magnitude estimates only, based on available data.
3. AFRI = acute febrile respiratory illness; GI = gastrointestinal illness.
4. Unexpected result requiring reconciliation as far as is practicable. Generally, monitoring data for the bathing water should take precedence over estimates from the sanitary inspection unless the monitoring data is uncertain or limited (e.g. does not include the range of conditions such as wet weather).

A.9. Litter pollution

A.9.1. Background information

Litter in the coastal zone can be sourced from the shore (either directly or in runoff or wind transport), or from rubbish dumped at sea. The main impact of litter is on the visual amenity of a waterway. This can have quite significant effects on the attitude of local populations to where they recreate and what they feel about their local environment. Another important effect of some specific types of litter is that they can cause the death or injury of animals that become either trapped or entangled in litter or ingest litter.

Causes

The main sources of litter pollution are:

- Litter from human populations nearby (recreation in coastal zone, including fishing; general litter in stormwater from urban populations)
- Litter from shipping and boating.

Symptoms

Symptoms include:

- Visual amenity decreased
- Tangling of animals and plants in litter (e.g. plastic bags, fishing line) or ingestion of litter by animals.

A.9.2. Pressure indicators and scoring categories

Catchment population density and tourist visitor numbers

The number of people living in the catchment will be related to the amount of litter produced. Population estimates can be obtained at small spatial scales (approximately 200 households) from the five-yearly census information provided by the Australian Bureau of Statistics (ABS).

In some areas, the permanent population is small relative to the visitor population. Tourists therefore potentially constitute a major source of litter in some areas. Information on visitor numbers can be obtained from the ABS for tourism region; information on smaller spatial scales would be best sourced from local tourism operators or associations.

For ease of assessment, the tourist visitor numbers can be added to the population. The first step is to ensure that the tourist visitor numbers are estimated on a catchment basis, as the tourism regions are usually much bigger than individual catchments. The average number of additional persons per day can then be calculated from the ABS data (e.g. the average number of additional people for September is the Guest Nights divided by 30). This number can then be added to the total population as an estimate of the total residents plus temporary population. The tourist population is likely to be highly seasonal; temporary population should therefore be estimated as frequently as the litter assessment (e.g. if litter is assessed once a year, a yearly average for population is appropriate; if assessed four times a year, quarterly averages should be used). In areas with a high resident population and small tourist population, the resident population only may be used.

Table A29. Scoring categories and indicator values for catchment population density as an indicator of stress from litter pollution

Stressor: ***Litter pollution*** **Pressure indicator 1:** ***Catchment population density***	
Scoring category	**Indicator value (Catchment population density, 1000 people/km^2)**
1	< 2
2	2–10
3	11–25
4	26–50
5	>50

Boating activity

There are two main types of boating activity that may generate litter. The first is local boating activity, consisting primarily of small recreational boats, including houseboats. The number of boats registered within the catchment area can be obtained from the Department of Transport. The second is shipping, which may generate much larger amounts of litter, but further away from the coast.

Table A30. Scoring categories and indicator values for boating activity as an indicator of stress from litter pollution

Stressor: ***Litter pollution*** **Pressure indicator 2:** ***Boating activity***	
Scoring category	**Indicator value (Number of boat registrations per length of coastline, km)**
1	< 5
2	5–10
3	11–20
4	21–50
5	> 50

A.9.3 Vulnerability

Strength and direction of wind and currents, stream flow

Wind and current direction and strength play a major role in determining the amount of litter arriving on the coast from offshore. A coastline that is scoured by strong currents or wind may have litter regularly removed, while a more depositional environment may accumulate litter. Rain, stream flow and wind are also likely to affect the amount of litter entering the coast from the land; litter is more likely to be washed or blown into creeks or drains during rain or wind events. However, no generic scoring categories can be offered at this stage.

A.9.4. Condition indicators and scoring categories

Litter quantity and quality

There are a number of methods for assessing the litter status of beaches and waterways. They are all based on quantification of the various categories of litter present along a defined stretch of beach or waterway. The categories employed depend on the purpose of the survey. The categories may be grouped in such a way as to facilitate assessment of their source (i.e. floating and non-floating) or they may be grouped into categories according to their aesthetic impact and scored accordingly (e.g. syringes and faecal matter might be grouped as having high aesthetic impact). From a management perspective, it may be useful to have both types of information so that both the main sources can be identified and the highest impact wastes that are present can be targeted.

With respect to indicators and scoring, it is recommended that users select a method from the literature and use the litter categories and scoring approach provided in that method. A number of sources of information are listed below but these are by no means exhaustive.

Condition indicator 1: Litter quantity and quality – Select indicator and scoring from recognised method

(see selection of methods below)

Bartram, J. & Rees, G. (eds) (2000) *Monitoring bathing waters: a practical guide to the design and implementation of assessments and monitoring programs*. E & FN Spon, London.

Environment Agency (UK) (2002) *Aesthetic assessment protocol (beach survey).* R&D Technical Summary, E1-117/TR. Environment Agency, Bristol, UK.

Queensland Department of Natural Resources, Mines and Water (2006) *Presence/extent of litter (Indicator status: for advice).* See <http://www.nrm.gov.au/monitoring/indicators/estuarine/presence-of-litter.html#analysis>.

The US Ocean Conservancy's National Marine Debris Monitoring Program <www.oceanconservancy.org/ site/PageServer?pagename=mdm_debris>.

Litter-related deaths

Some marine animals ingest or become entangled in litter (particularly plastics) in the water. Litter may be general plastic litter from land or, more frequently, litter from recreational or commercial fishing, such as lines, bait bags or nets. Some animals may also become entangled or trapped in crab pots. Information on number and likely causes of deaths may be obtained from marine parks organisations. Note that care should be taken to ensure that deaths are counted only in this section **or** under accidental death of biota, and not in both. Which section deaths are counted in will depend on local conditions and the way deaths are classified.

Table A31. Scoring categories and indicator values for percent recorded deaths caused by litter as an indicator of stress from litter pollution

Stressor: ***Litter pollution*** **Condition indicator 2:** ***Litter-related deaths***	
Scoring category	**Indicator value (% of total recorded deaths caused by litter)**
1	< 2%
2	2–5%
3	6–10%
4	11–20%
5	> 20%

A.10. Habitat removal or disturbance

A.10.1. Background information

This stressor includes both direct removal of areas of riparian or shoreline habitat and activities that disturb or damage habitat areas. Removal may occur for several reasons, including construction work, foreshore development, marine facilities, aquaculture and urbanisation, or for other reasons. Disturbance of habitat includes things such as anchor damage, bank or beach erosion as a result of boat or vehicle use (causing instability in the underlying soils), oil spills, changes in sedimentation resulting in smothering, shading, etc. Habitat damage may also be caused by natural events, particularly storms, but these are not considered here. The potential impacts of negative changes in habitat include erosion, sediment deposition, poor water quality (particularly turbidity), loss of habitat-dependent species, and a loss in visual amenity.

For the purposes of this document, habitat is defined in terms of the categories outlined in Scheltinga *et al.* 2004. These include:

- Coastal floodplains
- Saltflats, salt scalds, saltpans
- Saltmarshes
- Mangroves
- Seagrasses
- Intertidal mud and sand substrates
- Subtidal mud and sand substrates
- Rocky shores and rock reefs
- Coral reefs
- Beaches and dunes
- Cliffs and cliff top communities
- Water column.

Causes

The main causes of disturbance are:

- Removal of habitat (e.g. for buildings, construction, foreshore development, roads and bridges, marine facilities and infrastructure, aquaculture or urbanisation)
- Bank and beach erosion caused by boat wash or off-road vehicle use

- Dredging and extractive operations (sand and gravel mining) and trawling
- Modification of natural drainage pathways.

Symptoms

Symptoms include:

- Loss of habitat or decrease in vegetation cover e.g. in wetlands, dunes, estuarine riparian zone or foreshore
- Beach and foreshore sediment erosion and accumulation
- Change in species population or loss of species (especially shorebirds, and seafood)
- Poor water quality: associated with habitat removal; turbidity
- Visual amenity decreased.

A.10.2. Pressure and condition indicators and scoring categories

Proportion of habitat lost

For this particular stressor, there is no clear differentiation between pressure and condition and these are therefore considered together. This stressor is essentially about the physical loss or disturbance of habitat and the recommended indicator is simply the proportion of habitat lost or disturbed relative to baseline (pristine) condition. If pristine condition is not known then the best approximation should be used. Habitat loss should be related back to the habitat types listed above. Thus, where possible, data should be collected on changes in each of the defined habitat types that are relevant to the particular estuary or coastal area.

In most cases, information on habitat extent is best collected using remote sensing data or aerial photographs, although some ground truthing may be required. State government agencies (e.g. State Herbarium or State Fisheries) may have this information available. The data may be limited in that it may not be updated frequently or regularly, or may be of low precision. As the use of remote sensing data becomes more widespread, these limitations should disappear, although problems may remain with sourcing data on historical condition.

Table A32. Scoring categories and indicator values for proportion of habitat lost as an indicator of stress from habitat removal or disturbance

Stressor: *Litter pollution* Pressure/condition indicator: *Proportion of habitat lost*	
Scoring category	**Indicator value (% habitat lost compared to natural condition)**
1	< 5%
2	6–10%
3	11–25%
4	26–50%
5	> 50%

A.11. Direct removal of biota

A.11.1. Background information

This includes direct removal of biota through commercial and recreational fishing, bait collection and aquarium fish collection. Fishing results in the removal of large numbers of individuals of the target species. The effects on these target species are undoubtedly significant and may be one of the more significant stressors. The improvement in target species stocks in areas that have been temporarily closed to fishing is evidence of this (Pillans *et al.* 2005).

A certain amount of information is available on the numbers of fish caught by professional fishermen and there is somewhat less comprehensive data on amateur catch, although this latter is at least as significant as professional take for some species. Unfortunately, the catch data is not always able to be tied back to specific estuaries. In Queensland, for example, professional fish catch is assessed on the basis of a set of grid squares that do not necessarily tie in with individual estuaries or coastal areas. With regard to the status of fish stocks, very little information is available except for some individual studies.

A.11.2. Pressure indicators

Pressure on target species can be estimated in terms of catch. However, as discussed above, catch data may be fairly imprecise and in some cases may be unavailable. Surrogates for catch include numbers of licensed commercial fishers and numbers of amateur fishers. Data for these types of indicators may be more easily gathered. Data for both catch and numbers of fishers would need to be normalised to the size of the estuary. Suggested indicators and scoring categories are as follows:

Target species catch

Catch is estimated as the number of target species caught per annum per m^2 of estuary. The numbers of fish caught must be normalised to the area of the estuary to allow for the size of the estuary and to allow comparison between estuaries. This information would need to be compiled for each significant target species. No generic scoring categories can be proposed at this stage.

Number of professional fishers

The measure is the number of licensed fishers per km length of estuary. Information on licensed fishers can be gathered fairly easily but the numbers working in an estuary at any one time is variable. Nevertheless, it should be possible to develop some assessment of this measure. No scores can be proposed at this stage.

Number of amateur fishers

The measure is the number of amateur fishers per km length of estuary. This information is harder to come by than for professional fishers but, again, it should be possible to undertake some estimate of this measure. No scores can be proposed at this stage.

A.11.3. Vulnerability

Vulnerability of target species can vary depending on their reproduction rates and growth rates relative to annual catches. However, given the complexity involved in assessing this, no vulnerability indicators are recommended at this stage.

A.11.4. Condition indicators and scoring categories

Condition could be assessed directly in terms of target species populations and size distributions compared with undisturbed conditions. However, this type of information is hardly ever available and therefore indirect indicators must be employed. In most cases it will only be possible to make semi-quantitative assessments. The number of species being targeted is also important as, clearly, the larger this number is, the greater the impact on the estuary. Therefore it is proposed that condition be assessed through a two-way table whose axes are number of target species and the approximate condition of the target species population. The number of target species includes all species that are caught in significant numbers. Importantly it should also include species that were caught in the past but are no longer present in significant numbers. Condition assessments should be done for each target species. The condition assessment will have three broad categories:

1. Species in near natural condition
2. Species exploited but maintaining viable population
3. Species overfished and has a declining population.

Condition assessments will need to be obtained from local fisheries experts.

It is probably necessary to do separate tables for professional and amateur fishermen.

Table A33. Condition and number of target species as an indicator of stress from biota removal

Stressor: *Biota removal* Condition indicator: *Condition and number of target species*				
Condition of target species	**No. of target species**			
	1	**2–3**	**4–5**	**> 5**
All target species in condition category 1	1	1	1	2
50% target spp. in category 1, 50% in category 2	1	2	3	3
100% target spp. in category 2	2	2	4	4
20–50% target spp. in category 3	2	3	4	5
> 50% target spp. in category 3	3	4	5	5

The scores in the matrix of the table represent the condition scores for comparing with pressure scores.

A.12. Freshwater inflow alteration

A.12.1. Background information

Construction of impoundments on many Australian rivers has resulted in altered (and in virtually all cases this means reduced) freshwater inflows to estuaries. This has impacts on estuarine productivity, on fish breeding cycles, on siltation in upper estuary areas, on salinity and on dispersion of pollutants. A detailed review of the effects of reduced flows on estuaries is provided in the document: *Environmental water requirements to maintain estuarine processes* (Peirson *et al.* 2002).

Alterations to flow characteristics can be assessed using a range of measures. It is important to consider these different measures as they have different effects on estuaries. Two of the most important characteristics to consider for estuaries are:

- Reductions in the duration of base inflows
- Reductions in the frequency and size of large inflows.

A.12.2. Pressure indicators and scoring categories

Ideally, pressure indicators should include measures of reduction in large flows and reductions in the duration of base flows. However, such detailed information is rarely available. A much simpler measure is to calculate the proportion of median annual flow that is contained in all the impoundments in the catchment. While this is a coarse measure, it does provide an indication of the likely overall level of flow reduction. Suggested categories are shown in Table A34.

Table A34. Scoring categories and indicator values for annual inflow impounded as an indicator of stress from freshwater flow alteration

Stressor: *Freshwater flow alteration* Pressure/condition indicator: *Proportion of median annual inflow impounded*	
Scoring category	**Indicator value (Proportion of median annual inflow impounded)**
1	No impoundments in catchment
2	Total impoundment volume < 20% median annual flow
3	Total impoundment volume 20–50% median annual flow
4	Total impoundment volume 51–100% median annual flow
5	Total impoundment volume > 100% median annual flow

A.12.3. Vulnerability

The vulnerability of estuaries to changes in freshwater inflow relates to many different factors and therefore no one simple vulnerability factor can be recommended here.

A.12.4. Condition indicators

Freshwater inflow affects the functioning of estuaries in a number of ways. Therefore it is not possible to recommend a single condition indicator. Also, our knowledge of the quantitative relationships between changes to inflows and impacts on estuaries is very limited so that even if some condition indicators were recommended it would be difficult to provide condition categories.

As a starting point, it is suggested that condition indicators should be derived that relate to the following areas of potential impact:

- Changes to salinity regime
- Increased silting in the estuary and particularly the upper estuary
- Impacts on estuary productivity
- Impacts on fish or crustacean reproduction
- Generally poorer water quality due to reduced flushing.

A.13. Alteration to hydrodynamics

A.13.1. Background information

This includes any modifications that impact on the hydrodynamic features (local patterns of waves, currents or tidal exchange) of the estuary. These could include artificial closing or opening of entrances, breakwaters, canals, marinas, retention walls, training walls, levees, sea walls, spits, water barriers, artificial islands or reefs, and dredging, extraction or aquaculture structures. Impacts from changed hydrodynamics include changes to water depth, coastal currents, wave patterns, entrance opening pattern, turbidity, salinity, erosion and deposition patterns, erosion, eutrophication, algal blooms and loss of biota.

The most common, and often the most significant, impact on hydrodynamics results from dredging at the mouth of an estuary. In estuaries that are permanently open, dredging gives rise to incremental increases in tidal velocities, salinity and exchange rates. In coastal lagoons that are naturally cut off from the ocean, artificial opening of the entrance can have very large effects on salinity, which in turn significantly impacts on the nature of the biota present.

At the other end of the scale, construction of barrages, canals or marinas can create areas of relatively poorly flushed water, and this reduction in exchange rates can have detrimental impacts on water quality and on biota, although this is by no means always the case.

Causes

The main causes of alteration to hydrodynamics are:

- Entrance and estuary modification (including artificial closing and opening, breakwaters, canals, marinas, retention walls, training walls, levees, sea walls, spits, water barriers, artificial islands or reefs and aquaculture structures)
- Dredging and extraction.

Symptoms

Symptoms include:

- Water depth changed
- Tidal range altered
- Current velocities changed
- Estuary mouth open/close pattern changed

- Turbidity
- Erosion and sedimentation or deposition
- Habitat loss through erosion
- Eutrophication
- Algal blooms
- Change or loss of biota.

A.13.2. Pressure indicators and scoring categories

Although a number of factors can cause alterations to estuary hydrodynamics, the most important, as described above, are (a) alterations to the entrance of the estuary and (b) creation of quiescent artificial waterways (canals or barrages). Therefore the recommended pressure indicators relate to these issues.

Alteration to estuary entrance

Table A35. Scoring categories and indicator values for alteration to estuary entrance as an indicator of stress from alteration to hydrodynamics

Stressor: ***Alteration to hydrodynamics*** **Pressure indicator 1:** ***Alteration to estuary entrance***	
Scoring category	**Indicator value (Alteration to estuary entrance)**
1	No modification to entrance
2	
3	Some alteration to entrance
4	
5	Major alteration to entrance

Creation of canals or barrage

Table A36. Scoring categories and indicator values for creation of canals or barrage as an indicator of stress from alteration to hydrodynamics

Stressor: ***Alteration to hydrodynamics*** **Pressure indicator 2:** ***Creation of canals or barrage***	
Scoring category	**Indicator value (Creation of canals or barrage)**
1	No canals or barrage present
2	
3	Some canals or a barrage present
4	
5	Extensive tidal canals present relative to the size of the estuary

These indicators are not likely to change over short time scales, but may change over longer periods as a result of changes to policy. Periodic updating of information only is required. Data may be obtained from state government departments responsible for marine transport, licensing of dredging, and/or construction of dams and barrages. Site inspection is also useful. Aerial photography may be helpful in measuring the length of modifications.

A.13.3. Vulnerability

A major factor in vulnerability to hydrodynamic alterations is the tide range. The relative effect of alterations in entrances or construction of canals on hydrodynamics is much larger in estuaries with small tides than it is in macrotidal estuaries. Therefore, vulnerability is scored according to tidal range as in Table A37.

Tidal ranges (mean spring/metres)

Table A37. Scoring categories and indicator values for tidal range as an indicator of vulnerability to stress from alteration to hydrodynamics

Stressor: ***Alteration to hydrodynamics*** **Vulnerability indicator:** ***Tidal range***		
Scoring category	**Indicator value [Tidal range m (mean spring)]**	
1	> 6	Hypertidal
2	5–6	Macrotidal
3	3–4	Mesotidal
4	1–2	Microtidal
5	< 1	Ultra-microtidal

A.13.4. Condition indicators and scoring categories

The main direct impacts of alterations in hydrodynamics are changes to exchange rates (i.e. the rate of exchange between the estuary and adjacent coastal waters) and changes to the salinity regime, and the proposed indicators relate to these effects. There are consequent impacts on biota but these are relatively site-specific. Indicators of biological change could be derived at a local level.

Exchange rates

Table A38. Scoring categories and indicator values for exchange rate change as an indicator of stress from alteration to hydrodynamics

Stressor: ***Alteration to hydrodynamics*** **Condition indicator 1:** ***Exchange rate changes***	
Scoring category	**Indicator value (Rate of exchange between estuary and adjacent coastal waters)**
1	No alteration to exchange rate
2	
3	Some alteration to exchange rate
4	
5	Extensive alteration to exchange rate

Salinity regime

Table A39. Scoring categories and indicator values for salinity regime change as an indicator of stress from alteration to hydrodynamics

Stressor: ***Alteration to hydrodynamics*** **Condition indicator 2:** ***Salinity rate changes***	
Scoring category	**Indicator value (Extent of alteration to salinity regime)**
1	No change in salinity regime
2	
3	Some change in salinity regime
4	
5	Extensive change in salinity regime

A.14. Pest species

A.14.1. Background information

Pest species are defined as invasive organisms that are detrimental to an ecosystem. They are usually exotic to the system, although in some situations local species may also become pests. Exotic species enter the system as escapees from aquaculture, aquaria or gardens, and during transport, attached to hulls, in ballast water or via dredge spoil. Pest species can have significant effects on a system, resulting in the loss of native species, reductions in biodiversity and alterations to habitat. Examples of pest species in Australian estuarine/coastal waters include the infestation of Port Philip Bay with the giant sabellid worm and the occurrence of a toxic dinoflagellate species in the Derwent estuary in Tasmania.

A.14.2. Pressure indicators and scoring categories

Vessels originating from overseas are by far the most likely cause of introduction of pest species. Therefore the suggested pressure indicator is the frequency of visits of such vessels to a location. Categories are based on semi-quantitative assessments.

Table A40. Scoring categories and indicator values for presence of overseas vessels as an indicator of stress from introduction of pest species

Stressor: ***Pest species*** **Pressure indicator:** ***Presence of overseas vessels***	
Scoring category	**Indicator value** **(Frequency of visits of overseas vessels)**
1	Vessels from overseas almost never visit the estuary
2	
3	Small vessels from overseas sometimes visit and spend time in the estuary
4	
5	The estuary is a significant port with frequent visits from both large and small overseas vessels

A.14.3. Vulnerability

No vulnerability indicators can be recommended at this stage.

A.14.4. Condition indicators and scoring categories

Condition with respect to pest species relates to the number of pest species present and the extent to which these species have impacted on the local ecosystem. The suggested indicators are simply the presence or otherwise of a pest species and the extent of its impact. Proposed categories are shown in Table A41.

Table A41. Scoring categories and indicator values for presence of pest species as an indicator of stress on the system

Stressor: *Pest species* Pressure indicator: *Presence of pest species*	
Scoring category	**Indicator value (Presence of pest species)**
1	No pest species known to be present
2	
3	One or more pest species present but there are no major impacts on the local ecosystem
4	
5	One or more pest species present and at least one is having a major impact on the local ecosystem

A.15. Shoreline development

A.15.1. Background information

Shoreline development may be defined as the replacement of natural shoreline vegetation with some form of agriculture or with buildings. In this context, shoreline is taken to mean the shoreline in the immediate vicinity of the estuary. Shoreline development in this zone is a stressor that principally affects the aesthetic condition of an estuary through its impact on visual amenity. Generally speaking, agricultural development—especially low-intensity agriculture such as grazing—would have a much lower impact on visual amenity than would urban development.

A.15.2. Pressure and condition indicators and scoring categories

For this stressor, pressure and condition are expressed in terms of the same indicator and so they are considered together. The proposed indicator is percentage of the estuary shoreline that is not in natural condition. A distinction is made between agricultural and urban development. Proposed categories are shown in Table A42.

Table A42. Scoring categories and indicator values for shoreline development as an indicator of stress on the system

Stressor: *Shoreline development* Pressure/condition indicator: *Extent of shoreline development*	
Scoring category	**Indicator value (Extent of shoreline development)**
1	< 5% of shoreline developed
2	< 50% of shoreline developed for agriculture, < 5% urban
3	> 50% agriculture or 5–20% urban
4	21–50% urban shoreline development
5	> 50% urban shoreline development