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Report

Ecological risk management framework for the irrigation industry

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Disclaimer

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EXECUTIVE SUMMARY

This document outlines an ecological risk assessment framework for the Australian irrigation industry. The objective of the framework is to provide a robust process that will assist the irrigation industry to incorporate a transparent, scientific, precautionary and ecologically sustainable approach to its management of environmental risks.

The framework is catchment-based and focuses on the difficult task of assessing the risks to multiple ecological assets from multiple hazards. This catchment-wide approach is needed since irrigation enterprises are normally only one of a number of human activities in the catchment (e.g. dryland grazing, forestry, urban, tourism) that can contribute to the degradation of environmental assets. The framework synthesises the methods required to achieve successful adaptive management of natural resources.

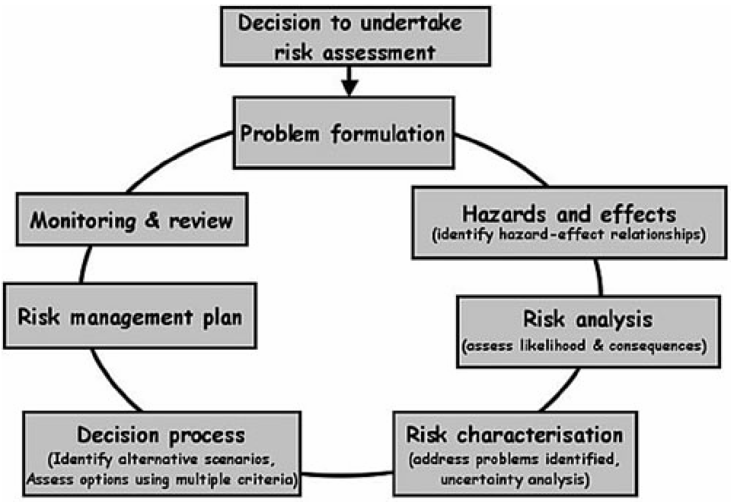
The framework was developed by a team using experiences from case studies undertaken in three irrigation districts, the Goulburn-Broken (Victoria), Fitzroy (Queensland) and Ord (Western Australia). Although the focus of this framework is primarily on the risks to aquatic ecosystems (e.g. rivers, wetlands, estuaries), it should be robust enough to also be used to assess the ecological risks to other natural resource assets in catchments (e.g. land, soil, vegetation, biodiversity).

The ecological risk assessment framework involves a number of key steps, including:

Defining the problem – this involves careful scoping of the problem, agreement on how it is to be assessed, and how the acceptability of actions will be judged.

Deciding on the important ecological values and hazards and threats to these values – hazards are evaluated and priorities set by evaluating effects on valued elements of ecosystems and ecosystem services.

Analysing the risks to the ecological values – the analysis process used needs to be appropriate for the situation in order to provide adequate information for decision-making. Guidance is provided on both qualitative and quantitative methods.



Characterise the risks - the technical details of risk analyses needs to be made accessible to decision-makers and broader stakeholders. In particular, the uncertainties and assumptions associated with analyses require careful and transparent documentation.

Making decisions – selection of the best management option or strategy will be the one that results in the effective minimisation of the ecological risks, while also being cost-effective and acceptable to the stakeholders. Guidance is provided on a number of multi-criteria methods for assisting this process.

Managing the risks – a risk management plan provides recommendations on managing or mitigating all high or unacceptable risks. The risk management plan should include a robust program to *monitor progress* to ensure the strategies are working, and a *review and feedback processes* for making changes if needed.

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1. INTRODUCTION

Irrigated agriculture is a major contributor to the economic and social well being of Australia. Each year these activities produce around $10 billion of agricultural output or 50% of the country's net agricultural revenue.

However, the sustainability of this important activity is currently under serious challenge. In response to these challenges, the National Program for Sustainable Irrigation (NPSI) funded an *Ecological Risk Assessment Project* with the overall objective of developing a generic framework for assessing the ecological risks associated with Australian irrigation systems (known as the *Irrigation ERA Framework*).

Three case studies were undertaken as part of the NPSI *Ecological Risk Assessment Project* in Goulburn-Broken, Ord and Fitzroy river irrigation regions, and the lessons learned from these have been used in developing the framework. The partners involved in these case studies and the availability of the final report are given below:

- Goulburn-Broken case study – Partners: Water Studies Centre & CRC for Freshwater Ecology, Monash University, Goulburn Murray Water (Feehan et al., 2004);
- Ord case study – Partners: Edith Cowan University, WA Rivers & Water Commission (Lund & McCrea, 2004a,b); and
- Fitzroy case study – Partners: Central Queensland University, Qld Dept Natural Resources & Mines (Duivenvoorden et al., 2004).

1.1 How will risk-based management assist the Australian irrigation industry?

Irrigation uses 75% of the water extracted from Australia’s rivers and groundwater systems. Environmental impacts from irrigation areas can include: water logging, salinisation, soil acidification, erosion, polluted runoff, draining of wetlands, and changes to the flow regimes in rivers.bo

The adoption of risk-based management approaches should assist the Australian irrigation industry in quantifying and prioritising the ecological risks arising from its activities, and in better focusing the management actions required to minimise these risks. Additionally, this information should also provide an opportunity for regulatory authorities (such as State Environment Protection Authorities) to better determine what needs to be protected and to use this information as the basis for licensing and monitoring requirements. This tool will help the Australian irrigation industry’s thrust towards more proactive sustainable management.

In most catchments, irrigation is only one of a number of land use activities (e.g. grazing, cropping, tourism, urbanisation/townships) that can adversely affect aquatic ecosystems. The catchment-based ecological risk assessment framework outlined in this document will assist the irrigation industry in putting their activities into the wider catchment context.

1.2 Purpose of this document

This document is intended to provide the information and steps required to undertake a catchment-wide ecological risk assessment, with a focus on irrigation enterprises.

The document is arranged in three sections:

- a discussion of why ecological risk assessment is a useful tool, and broadly what is involved in conducting one;
- a framework for assessing ecological risks with a focus on irrigation enterprises; and
- a framework for developing and implementing a risk management plan utilising the information obtained from the ecological risk assessment.

2. RISK MANAGEMENT APPROACHES FOR THE IRRIGATION INDUSTRY

2.1 Risks and perceptions of risk

Risk is the chance of an adverse event with specific consequences occurring within a certain timeframe. Risk assessment is a tool to facilitate informed decision-making. The process of risk assessment is based on quantifying the probability of an uncertain, undesirable event occurring, either now or in the future (Suter, 1993).

Perception of risk is very much an individual matter, and is affected by a variety of factors. Our personal experience and beliefs have a strong influence on our perception of risk (Pidgeon et al., 1992). Cultural differences also contribute substantially to perceptions and acceptance of risk, so that different social groups react differently when confronted by the same hazards (Rohrmann, 1994). Furthermore, individuals do not necessarily feel the same way about any given risk from day to day (Bernstein, 1996). Other factors affecting our perception of risk include how much control we have over the hazard in question, how equitably the risk is distributed, and how well we understand the technical details (Morgan, 1993).

In general, the less control and understanding we have, the greater we perceive the risk to be. Additionally, people are generally poor judges of probabilistic events, tending to substantially underestimate the risk of events that have a high probability of occurrence, and overestimate the risk of low probability events (Fischhoff et al., 1982). This framework emphasizes the benefits of *quantitative* risk assessment over *qualitative* risk assessment for the reason that the latter are subjective, and are influenced by the biases (mostly unacknowledged) of the risk assessor. Psychological biases that are often overlooked in subjective risk assessment (Burgman, 2001) include:

- the way an issue is presented may influence our behaviour even when the underlying outcome is unchanged;
- people are generally overconfident in assessing the quality and reliability of their own judgements;
- people have a tendency to be influenced by initial estimates, either their own or those of others; and
- most people, including experienced scientists, draw inferences from data to an extent that can only be justified with much larger samples.

2.2 Who uses risk assessment and risk management?

Risk assessments are now used by a wide range of disciplines including the nuclear industry, engineering, mining, finance, economics, public health and medicine to assist them in better managing their activities (AS/NZS, 2004a,b; Burgman, 2005). Recently, a number of government departments have also developed organisational risk management strategies (NRE, 1999).

In Australia the most widely used risk management approach is that outlined in Risk Management AS/NZS 4360 prepared jointly by Standards Australia and Standards New Zealand (AS/NZS, 2004a,b). The framework provided in this document builds on the AS/NZS 4360 approach in advocating a rigorous framework for assessing and managing the risks associated with Australian irrigation enterprises. The framework places considerable emphasis on the importance of embedding a risk management practice in the culture, processes and operations of the irrigation industry and key stakeholder organisations.

2.3 Why ecological risk assessments?

Ecological risk assessment is the process of estimating the *likelihood* (or probability) and *consequences* (or magnitude) of the effects of human actions or natural events on plants, animals and ecosystems of ecological value (Suter, 1993; AS/NZS, 2000). At its best it is a rigorous, transparent and quantitative method for assessing ecological risks that supports an environmental decision-making process. Hayes (2004) provides a useful assessment of best practice in ERA. ERA should be seen as a component in the overall risk-based approach to the management of natural resources. It provides a basis for comparing and ranking risks, so that natural resource managers can focus attention on the most severe risks first.

The application of formal risk assessment in natural resource management is relatively recent, perhaps 2-3 decades old (Burgman, 2005). This is surprising given that many (perhaps most) human activities have the potential to cause undesirable ecological effects. For example, the construction of water impoundments, clearing of land for urban development, harvesting of natural populations of fish and other animals, all have substantial potential environmental costs in addition to their better known economic and social benefits.

The US EPA (1998) list the following features of ecological risk assessments that contribute to effective decision making:

- they seek to quantify the changes in ecological effects as a function of exposure to various stressors, and as such will be particularly useful to decision makers who must invariably trade-off various alternatives aimed at reducing the level of the stressors;
- it is an iterative process in which new information can be incorporated into risk assessments to improve decision-making – an element of adaptive management;
- they should explicitly evaluate uncertainty, providing a measure of the degree of confidence in predictions (although many risk assessments still do not address uncertainty well);
- they provide a basis for comparing, ranking and prioritising risks; and
- they focus on management goals and objectives as well as scientific issues. In this way they should be more useful to managers.

Thus, risk assessment is a tool that can assist environmental and natural resource managers in their challenging task of minimising or eliminating environmental risks associated with human activities. They must balance the consequences of uncertain detrimental outcomes against the uncertain benefits of management actions aimed at minimising these environmental impacts.

Two additional features of risk assessments, discussed in detail later in this document, are that they can assist in:

- identifying the best management actions that might be implemented to minimise the risks of adverse effects occurring; and
- identifying key knowledge gaps that if addressed will add to the management knowledge base, and may lead to a better estimate of ecological risk.

Over the past few years, there has been a major increase in both Australia and overseas in the use of risk-based approaches in natural resource management, (USEPA, 1998; AS/NZS, 2000; Hart et al., 2001, 2003, 2005; Hart, 2004; Burgman, 2005).

Australian examples where risk-based approaches have been used include the new Australian National Water Quality Guidelines **(**ANZECC/ARMCANZ, 2000a), the National Environment Protection Measure (NEPM) for ambient air quality (Beer, 2003), and Victoria's new State Environment Protection Policy-Waters of Victoria (SEPP-WoV). EPA Victoria has recently provided guidelines on how to undertake risk-based assessments with respect to implementation of SEPP-WoV (VicEPA, 2004).

Perhaps the major advance made in this area has been the integration of ecological risk assessment approaches with catchment (or watershed) management approaches, to focus on the difficult task of assessing and managing the risks to multiple ecological assets from multiple hazards (Cormier et al. 2000; Hart et al. 2001, 2005; Diamond & Serveiss, 2001; Landis and Wiegers, 2001; Leuvin and Poudevigne 2002; Serveiss, 2002; Serveiss et al., 2004; Norton et al., 2002; USEPA, 2003; Lamon & Stow, 2004)[1].

This catchment-wide approach is certainly relevant to irrigation enterprises, since these are normally only one of a number of human activities in the catchment (e.g. dryland grazing, forestry, urban, tourism) that can contribute to the degradation of environmental assets.

2.4 Triggering a risk assessment

Decisions to trigger an ecological risk assessment may be based on any of the following reasons:

- to comply with regulatory guidelines. For example, where the State Environment Protection Policy-Waters of Victoria (SEPP-WoV) environmental quality objectives are not met ('triggered'), a risk-based assessment should be conducted to ascertain if there is a risk to the ecosystem (VicEPA, 2004);
- to better understand and manage the environmental risks associated with the expansion of existing, or the introduction of new, business or management activities;
- to demonstrate due diligence as part of corporate responsibility to make a business more environmentally sustainable;
- to engage stakeholders in socially sensitive issues; and
- to comply with the requirements of environmental impact assessment legislation.

1. See also web sites: http://cfpub2.epa.gov/ncea/cfm/recordisplay.cfm?deid=23734; http://www.epa.gov/owow/watershed/wacademy/acad2000/ecorisk

2.5 What is involved in undertaking an ERA?

ERA is particularly useful for assessing the effects of multiple hazards (or stressors) to a range of ecosystem components, including native flora and fauna, processes and services, while also taking account of the inherent variability and complexity of these systems.

Hazards are the sources of potential harm that threaten assets or values of the environment (AS/NZS, 2000). Hazards to natural values may be physical, chemical or biological entities (stressors) or threatening processes such as clearing of land or discharge of waste (USEPA, 1998).

Ecological risk is defined as the product of the *likelihood* (or *probability*) of a detrimental ecological event occurring and the *consequences* that arise if that event occurs.

Thus, a risk assessment requires that the important environmental/ecological values be clearly identified along with all the hazards that could potentially adversely affect these values.

There are four key steps involved in a risk assessment:

- *planning* the assessment;
- *formulating the problem* (identifying the scope of the assessment, the important ecological values and the relevant hazards);
- *analysing the risks* (consequences and likelihood) for each of the ecological issues; and
- *characterising the risks* where the risks are compared, ranked, and prioritised in terms of their seriousness with respect to the management objectives identified in the initial problem formulation.

The information from the risk assessment then feeds into a *decision-making process* that includes economic, social and political inputs, to develop a *risk management plan*. The management plan must include a robust *monitoring program* to provide information on the success (or otherwise) of management actions and a *review process* to ensure that the management plan is upgraded as knowledge improves and priorities change.

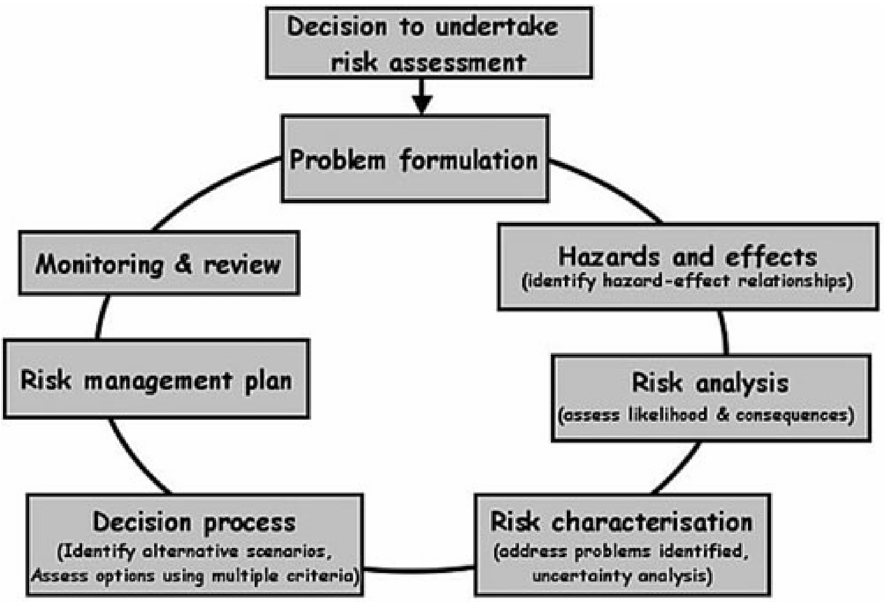


Figure 1: Overall risk assessment and management framework

Many organizations place great emphasis on separating the risk assessment from the risk management plan or strategy; the reason being that the former is supposedly based entirely on science, while risk management includes economic, social and political inputs as well as the risk assessment to help decide what is achievable. Because environmental values have an implicit social basis, we do not make this distinction. Our approach is based on an operational understanding of sustainability that involves industry demonstrating compliance with socially mandated endpoints. In this context, we recognise economic, social and political inputs early in the process because science has little to say about what is to be valued and what are acceptable thresholds for change.

Figure 1 is a flow diagram of the risk assessment and management framework. The key point of this framework is that the process should be *iterative*, allowing new information to be incorporated into the risk management plan as it becomes available. This is the essence of an adaptive management process. More detail on the various steps involved in the risk assessment and risk management processes is provided in Sections 3 and 4 respectively.

2.6 Adoption of risk management approaches by irrigation industry

The ecological risk management framework, guidance and tools provided in this report should assist the Australian irrigation industry to achieve its goal of long-term environmental sustainability. However, as discussed below, for there to be wide-spread adoption of risk management approaches by the industry, a number of other activities and changes will need to occur.

Risk management ethos – it will be vital that risk-based approaches are adopted at all levels (individual farmers, Board members and CEO's of irrigation companies, regulatory and NRM agencies). There must be ownership at all levels.

Culture change – the process required to establish effective risk management as part of the day-to-day business at both organisational and operational levels is likely to require a change of culture within many organisations. This is unlikely to occur without the active participation of national organisations such as ANCID, NPSI, CRC for Irrigation Futures and the eWater CRC.

Adequate skills training – the current (2004/2005) program of ERA training workshops sponsored by NPSI will increase the awareness of risk based approaches and produce a range of training products. For wide-spread adoption of risk management these training products will need to be delivered in a much more systematic way. It is possible that the CRC for Irrigation Futures could become the accredited trainer in risk-based approaches for the industry.

On-going technical support – risk-based approaches in natural resource management, including the irrigation industry, are being developed at a rapid rate. It can be confidently predicted that both the researcher providers (e.g. Universities, CRC for Irrigation Futures, eWater CRC, CSIRO) and consulting companies will continue to develop new approaches over the next few years. It will be desirable for the irrigation industry to form close links with these organisation to ensure the new knowledge and approaches is captured and used by the industry.

3. FRAMEWORK FOR ASSESSING ECOLOGICAL RISKS ASSOCIATED WITH IRRIGATION ENTERPRISES

3.1 General

Irrigation enterprises generally co-exist with other agricultural activities (e.g. dryland grazing, forestry) and with townships and tourism. All these activities can contribute to the degradation of ecosystems in the catchment.

Thus, risk assessments focused mainly on irrigation enterprises, but within a catchment context, will be influenced by multiple hazards (stressors from multiple sources), a range of ecological values, and a variety of political and economic factors.

Inevitably, this means that such ERA's need to be iterative, since it will rarely be possible to adequately address all issues the first time around. *Catchment-based ecological risk assessments are a key component of the toolkit of methods required to achieve successful adaptive management of natural resources.*

In this section, we build upon the generic information on risk assessments provided in Section 2, to develop a risk management framework for assessing the ecological risk associated with irrigation enterprises. Guidance on managing these risks is contained in Section 4. Almost all the examples used to illustrate various aspects of the Irrigation ERA Framework are focused on aquatic ecosystems (e.g. rivers, wetlands, estuaries). However, the framework is robust enough to be used to assess the risks to other natural resource assets in the catchment (e.g. land, soil, vegetation, biodiversity).

In developing this framework we have drawn heavily on the experiences gained in undertaking three Cases Studies. In particular, these Case Studies have provided valuable information on problem formulation, risk analysis, identification of knowledge gaps and how to address these, and communicating risk information.

The steps involved in the irrigation risk management framework are shown diagrammatically in Figure 1 and involve:

- formulating the problem;
- identifying the ecological values/assets perceived to be at risk;
- identifying the hazards likely to adversely impact upon these values;
- analysing the likelihood and consequences;
- characterising and ranking the risks;
- developing a risk management plan to minimize these risks;
- implementing this plan; and
- monitoring the system to ensure the management plan is indeed reducing the impact of the priority risks.

The first five of these are covered in this section and the development and implementation of the risk management plan is outlined in Section 4.

3.2 Getting started

3.2.1 Planning the risk assessment

The focus of a risk assessment, who conducts it, who participates, and what happens to the outputs are all largely determined by the context or setting of the problem(s) being investigated (Burgman, 2005). The context is largely controlled by those paying for the work to be done. This may be a government regulatory agency, a catchment management authority or an industry such as an irrigation enterprise.

The often-stated purpose of undertaking an ERA is to enable managers to make more informed environmental decisions. Ideally, an ERA should be able to transform scientific data and information into meaningful information about the risks to specific parts of the environment from human activities.

However, it is important to recognise that rather than being a totally objective scientific processes, a risk assessment can often be significantly influenced by the context and the way the risk assessment procedure is set up.

It is wise to address a number of high-level decisions before the ERA commences, as these will set the context for the project. These include (US EPA, 1998):

- the management goals in terms of the ecological values to be maintained or protected;
- the range of management options that the risk assessment is to support (this is often iterative with

a range of scenarios developed and tested during the risk assessment process);

- the objectives of the risk assessment (including criteria for success);
- the focus and scope of the assessment;
- the technical and financial resources available; and
- the extent to which stakeholders are to be involved and the degree of control over scope and outcomes that is to be afforded to the participants.

These are discussed more fully in the sections below.

3.2.2 Risk assessment team

The first step is to assemble an appropriate team to carry out the risk assessment. For ERAs focused on irrigation enterprises, the team ideally should be multi-disciplinary with the range of skills necessary to conduct the assessment. The team members should be trained in the ERA process, and in their respective roles and inputs.

The composition of the team should be determined, in large part, by the nature of the risks and the social context of the problems. It is common for the technical team to include a range of biophysical expertise, such as ecology, biology, hydrology, water quality, environmental chemistry, ecotoxicology, statistics and modelling.

Additionally, the involvement of a social scientist in the risk assessment team will be useful in most situations. Social scientists can assist the technical team particularly in the problem formulation stage, in planning how the key stakeholders will be involved in the particular ERA process, and in maintaining effective communication between technicians and stakeholders

In summary, members of the risk assessment team need to be trained in the ERA process, be skilled in their area of expertise (e.g. ecology, chemistry, engineering, hydrology, sociology, etc), have the skills and knowledge to select the right tool for the risk analysis step, and to have the skills to communicate the process and the outcome of the risk assessment to key stakeholders.

3.2.3 Stakeholder involvement

The involvement of key stakeholders is essential in undertaking most ecological risk assessments (Borsuk et al., 2001). In the past, risk assessments have been criticised for alienating stakeholders, particularly by placing social decisions in the hands of the technical people who conduct the assessments (O'Brien, 2000; Fisher, 2000). While the key stakeholders should be kept involved throughout the entire risk assessment process, it is particularly important that they are actively involved in the problem formulation step.

It is most important that the environmental and social values relevant to the stakeholders are obtained through a participatory process. In the absence of meaningful stakeholder input, a risk assessment will tend to be coloured by the personal and professional biases of scientists. Plans for stakeholder involvement will usually need to anticipate resistance from technical experts who often discount the opinions of non-technical participants (Freudenburg, 1999), and from planners who need to forego their sense of ownership of the problem and the responsibility for deciding on preferred options.

Stakeholders can be defined as people who share the burden of the risks associated with decisions. For risk assessments focused on irrigation enterprises, relevant stakeholders will include individual irrigators, industry representatives, regional catchment management authorities, state resource management agencies, regulators, other government organizations, traditional owners, environmental groups, local government, scientists, residents, NGOs, unions and consumer groups. All these individuals and groups may have an interest in improving the management of irrigation enterprises to ensure they are sustainable.

Most commonly, stakeholder involvement is elicited through stakeholder workshops. Essentially, stakeholder workshops provide a formal opportunity for stakeholders to state what they want and why, and for them to gain some ownership of the process. At such workshops, stakeholders can also directly assist in identifying the ecological values of these systems, the ecological values or assets potentially at risk, the hazards that threaten these values, the means by which the various risks will be quantified (assessment endpoints), and the cause-effect conceptual models linking the key hazards with the assessment endpoints (Glicken, 2000).

Formal workshops are not the only way to elicit input from stakeholders, and are probably best suited for the more 'technical' stakeholder groups.

Box 1: Stakeholder involvement

Stakeholders are those people or organisations who may affect, be affected by, or perceive themselves to be affected by a decision, activity or risk. Broad stakeholder involvement is essential for a successful outcome in the types of ecological risk assessments considered in this framework, i.e. catchment-based and involving multiple hazards and multiple issues. Ideally, the stakeholder involvement should occur throughout the assessment process and involve both technical and non-technical people and organisations.

Although most ecological risk assessments now involve key stakeholders to some extent, we believe this is often poorly done. Some of the problems we have observed include:

- failure to involve important, but often 'difficult', individuals or groups of stakeholders. This leads to these groups questioning the credibility of the process (often through the press);
- failure to include non-technical stakeholders (e.g. farmers and other landholders);
- failure to take notice of legitimate issues brought forward by stakeholders;
- failure to account for long-standing disagreements between 'competing' stakeholder groups;
- failure to put the necessary effort into informing and running the stakeholder workshops, resulting in key groups being disillusioned with the process and feeling alienated; and
- failure to allow the stakeholders to drive the identification of the issues or hazards that need to be considered (e.g. where the issues have been largely decided prior to the involvement of stakeholders).

If the nature of a problem is such that risks are technically driven, there are ample data, and scientific understanding of the processes is good, risks may be assumed to be largely technical. In this case, technical risk analysis, without substantial sociological input, may be sufficient. Public participation may be treated with post hoc devices such as a 'public comment' phase. However, when data are poor and there are substantial social pressures around the issues, greater public participation is warranted.

In these circumstances, the solution is to involve those affected by the outcomes of risk assessments closely and continuously in the risk assessment process, making a marriage of the technical and social dimensions of risk. Non-technical information should enter at the planning stage, and non-technical participants should contribute to all elements of the risk assessment including the final outcomes of the study.

Stakeholder map

We have found that the preparation of a *stakeholder map* is a very useful way to ensure that all key stakeholders are included (or at least identified) and for understanding the socio-political settings (Freudenburg, 1999; Glicken, 1999, 2000). Stakeholder maps identify the key stakeholders (e.g. government organizations, scientists, residents, NGOs, industry representatives, traditional owners, unions and consumer groups), and map the linkages between them.

There are a number of methods for acquiring stakeholder information including using existing data (archival and agency information), first-hand data collection (surveys, targeted interviews), and methods to identify gaps and oversights (sensitivity analyses, interdisciplinary double-checks and public involvement techniques). It is important to recognise that this process can take some considerable time.

Elicitation of stakeholder information

There are many methods available for eliciting information from stakeholder and community groups (Borsuk et al., 2001). They include preparation of stakeholder maps (Glicken, 1999, 2000), surveys, targeted interviews through one-on-one contact, use of focus groups and stakeholder workshops, adaptive management (Walters, 1986), building of Bayesian networks (BN; Bacon et al., 2002a,b), multi-criteria decision-analysis (MCDA; Roy, 1999) and management strategy evaluation (MSE; Smith, 1994).

The use of stakeholder workshops, particularly for involving key stakeholders in the early stages of the risk assessment and management process is covered in Box 2.

Newer methods (e.g. BNs, MCDA and MSE) are being effectively used to more rigorously incorporate the social, economic and ecological considerations that underpin the decision-making process, particularly when there is a range of stakeholder views. These are covered in Section 4.2.2.

We believe that the use of quantitative approaches, particularly emerging ones based on Bayesian methods, that explicitly include stakeholder inputs into risk-based decision-making would significantly strengthen the process.

Box 2: Stakeholder workshops

A common method for eliciting information from stakeholders is through *stakeholder workshops*, the success of which depends upon the knowledge base of the stakeholders, the mix of stakeholders attending, and the way in which the workshop is facilitated.

Successful stakeholder workshops require good planning. Here we summarise some of the important findings from the workshops that were run to establish the scope and issues for each of the irrigation Case Studies.
Stakeholder workshops should be used to assist the technical team in determining the following:

- *scope* of the risk assessment (spatial boundaries, time scale);
- *ecological assets* potentially at risk;
- *values* of these ecological assets that need to be maintained or protected;
- *activities* in the catchment (e.g. irrigation, dryland agriculture, forestry, urban, tourism, natural, pests) that are likely to threaten these ecological values;
- *specific hazards and threats* that could result from these activities; and
- *relationships* between the hazards and possible adverse ecological effects.

Stakeholders

Relevant stakeholders for irrigation risk assessments include individual irrigators, representatives of irrigator groups, industry representatives, regional catchment management authorities, state resource management agencies, regulators, other government organizations, traditional owners, environmental groups, local government, scientists, residents, NGOs, unions and consumer groups. We have found it useful to develop a *stakeholder map* for each catchment to help make the identification of all interested groups more apparent. This best done through one-to-one contact with individuals (via visits, telephone or email) where the risk assessment team member can introduce themselves and the project and work through a set of predetermined questions. This can be done both before or after background information is sent to the stakeholder.

Workshop objective

The purpose of the stakeholder workshop must be clearly identified before the workshop begins. These workshops obviously have a primary aim to inform the ERA process, but they are also very useful for informing the stakeholders on the range of issues associated with their operations.

Background information

It is important to have good technical/scientific information available for the workshop. We have found it vital to acknowledge and document all prior information and decisions relevant to the issue. In most cases, considerable work will have been done on the issue prior to the workshop, and if this work is not acknowledged and used it is likely some (many) stakeholders will be offended.

The primary purpose of the workshop is to obtain the views of the stakeholders. It is perhaps best to think of the stakeholders as being a source of information on the *ecological values* that they wish to achieve, and of the scientists as providing information on the *ecological consequences* of particular actions.
We have found that the preparation of such big picture conceptual model (or ‘mud map’) of the catchment is a useful way to summarise the activities occurring in the catchment (and beyond), and thereby helping the stakeholders to develop their lists of ecosystems, values, hazards and effects (see Section 3.3.2).

Running the workshop

It is important for the credibility of the process to consider ALL the issues. For example, issues associated with the fact that reservoirs have been built in the past to service irrigations schemes need to be included in the discussions.

Stakeholder workshops are most successful if there has been prior interaction between stakeholders (particularly irrigators) and the risk assessment technical team – the building of TRUST between the players is extremely important.

Independent facilitators have the advantage of transparent impartiality, but they carry the cost of not always being intimately familiar with the technical and social details of a problem. The ideal facilitator is one who in addition to skills in facilitation, has direct knowledge of the technical and social aspects of the project and is trusted by all participants.

Many ‘non-technical’ stakeholders either do not wish to attend workshops or if they do are reluctant to become fully involved in the discussions. We have found that one-on-one ‘interviews’, either face-to-face or by phone, can be particularly informative. Moreover, such interviews may be the only alternative for stakeholders that are pressed for time. For instance, many environmental groups and industry groups are run voluntarily, and members cannot take a day away from their regular employment to attend workshops.

Box 1 provides further information on how best to involve stakeholders in the risk assessment process and how to provide a mechanism for them to input their knowledge and wishes into the process. Guidance on how to set up and run a stakeholder workshop is provided in Box 2.

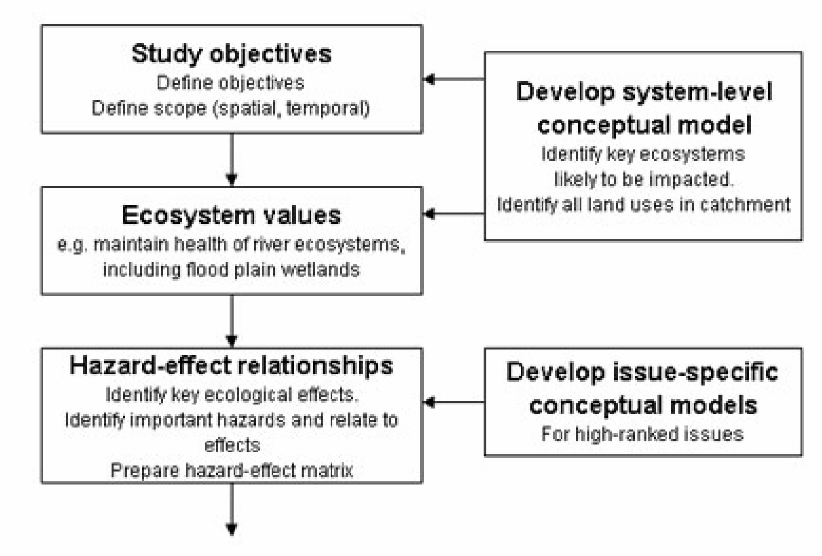


Figure 2: Flow chart of the problem formulation steps

3.3 Formulating the problem

Formulating the problem is probably the most important step in an ERA, as it sets up the scope of the problem and how it will be tackled.

This step involves (see Figure 2):

- determining the *scope* of the study, both the spatial extent and the time frame over which risks will be assessed. The model does this by illustrating the spatial relationship between the ecosystem (s) and hazards, and how they may be connected by, for instance, downstream flow in a river;
- developing a *systems-level conceptual model* of the catchment (sometimes referred to as a ‘mud map’) to assist in scoping the study. This ‘big picture’ conceptual model will help to identify the *ecosystem(s) at risk* and the various *human activities* in the catchment that may contribute hazards potentially adversely affecting the ecosystem(s);
- identifying the *ecological values* of the ecosystem(s) at risk. The ecological values of a particular natural asset are those aspects the managers are charged with maintaining or protecting;
- identifying the specific *ecological effects* that may arise because of the various human activities in the catchment, and identify the *assessment endpoint* that will be used to judge whether the ecological value is being adversely affected;
- identifying the specific *hazards* or threats that could potentially adversely affect each ecological value;
- developing *specific conceptual models* showing the relationships between hazard(s) and potential ecological effects for each of the values; and
- developing a *hazard-effect matrix* for the system being assessed, in which all the major hazards potentially causing adverse effects on the ecological values of the system are shown.

In many cases, a considerable amount of this information can be obtained during the *stakeholder workshop* (see Box 2)..

However, it is often difficult to obtain all this information through a single stakeholder workshop, and the risk assessment team should be prepared to continue the stakeholder interaction (not necessarily always through workshops) to obtain all the information needed. Additionally, a key decision that must be made at this early stage is the most appropriate *form and type of risk assessment* needed, a decision that is normally made by the risk assessment technical team. The different qualitative and quantitative methods available require different levels of data, expertise and knowledge (Section 3.4). They also produce different quality outputs. The time and funds available will influence this decision, as will the type of output that is required.

The sections below cover the important aspects of formulating the problem, and broadly follow the flow chart shown in Figure 2.

3.3.1 Study objectives and scope

Clearly stated objectives are essential for a successful risk assessment as these enable the study to be properly focused. At the highest level there should be a statement of the overarching management goals for the system (e.g. catchment) in terms of the ecological values to be maintained or protected. For natural assets, these goals are often articulated in government strategies (e.g. regional catchment strategies, water quality management plans, land & water management plans).

Also, individual industries often provide statements on their environmental management objectives[2].

We recommend that the objective of the ERA be clearly defined before the study commences, with the statement of objectives identifying:

- who the risk assessment is being done for (e.g. government agency, industry, catchment management authority);
- the ecological asset(s) being considered (e.g. river basin, wetland, estuary, terrestrial environment, riparian areas, catchment);
- the ecological values to be maintained or protected for each asset (e.g. ecologically healthy rivers, wetlands or estuaries);
- the spatial extent of the study region and the timeframe over which the assessment is targeted; and
- the major human activities and/or hazards that will be considered in the risk assessment.

As an example, the objective of an ERA being undertaken for the Mekong River Commission **(**Hart, 2004b**)** is *'to determine the potential risks to the ecological health of the Mekong River in the study region (Vientiane/Nongkhai) from urban wastewater, agricultural runoff and changed flow regimes over the next 10 years.'* Note that this objective links the environmental value (ecological health of the Mekong River) with the main threats or stressors (urban wastewater, agricultural runoff, changed flow regimes), and also specifies the temporal and spatial scale of the assessment.

3.3.2 Systems-level conceptual model

It is often very useful to develop a 'big picture' conceptual model (or 'mud map') of the catchment. This should clearly identify the environmental assets (ecosystems) potentially at risk and the various human activities in the catchment that may contribute hazards may result in adverse effects on these assets.

Preparation of such a big picture conceptual model is often a useful way to summarise the activities occurring in the catchment (and beyond), and thereby helping the stakeholders to develop their lists of ecosystems, values, hazards and effects. Often stakeholders from one part of a catchment have little appreciation of the activities and issues in other parts of the catchment. Preparation of a catchment conceptual model can assist in making all stakeholders aware of all the issues.

Figure 3 is the catchment-level conceptual model of the Goulburn-Broken catchment in Victoria, developed for an assessment of the ecological risks to rivers and wetlands in this catchment due to salinity increases (Hart et al., 2003).

We have found it best to develop a mud map of the catchment as one of the first exercises in the stakeholder workshop. Generally, this requires that the facilitator has some knowledge of the catchment, so that the key information – the environmental assets (ecosystems) at risk and the various human activities and hazards causing these risks – can be elicited from the group.

2. For example, Murray Irrigation Ltd (MIL) states that their overall policy is to work toward achieving a sustainable balance between environmental protection and agricultural production. Further, MIL is committed to addressing the direct environmental impact of providing irrigation water and drainage, through improvements in risk assessment and management systems, and will also strive to reduce the indirect impact of on-farm

It is also possible to provide an initial mud map as part of the background material for the stakeholder workshop. When developing the mud map, we do not place great emphasis on also developing cause-effect conceptual models linking particular hazards with particular endpoints (this comes later in the process), but often such linkages can be discussed by the group during the mud map development.

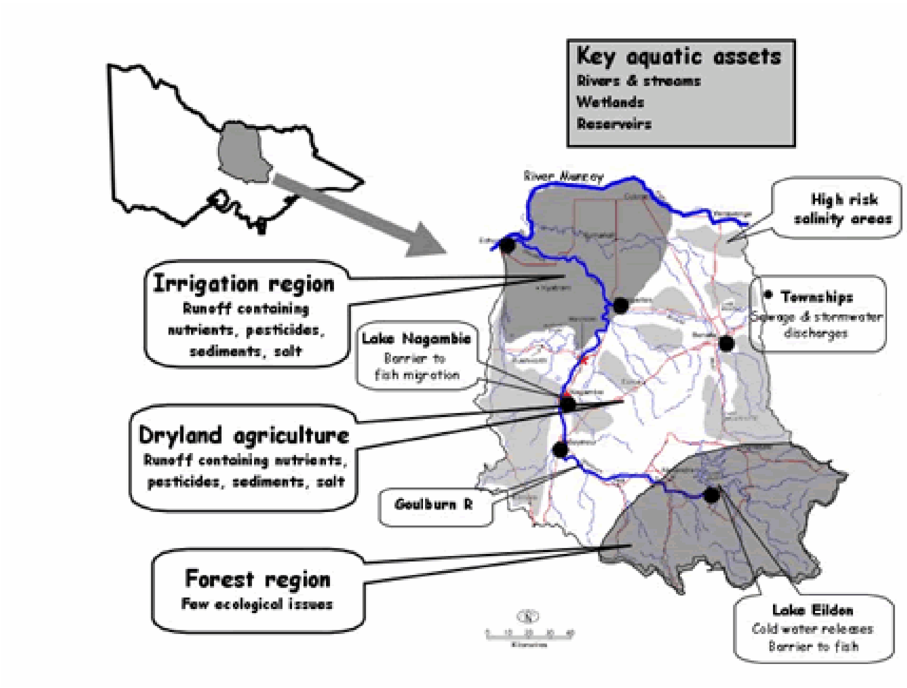


Figure 3: Example of a system-level conceptual model for the Goulburn-Broken catchment developed for an assessment of the ecological risks to rivers and wetlands in this catchment due to salinity increases

3.3.3 Ecological values

The ecological (or environmental) values of the potentially affected natural resource assets need to be clearly stated to provide a focus for the risk assessment. These are the values the community place on the region's natural resources and are what must be protected. Since these are largely community values, all key stakeholders must play a role in deciding these.

Ecological/environmental values are often articulated in Regional Catchment Strategies (Victoria, South Australia), River Basin Management Strategies (Queensland, Western Australia), Land & Water Management Plans (NSW) or State Environment Protection Policies (Vic).

For example, the Glenelg-Hopkins CMA (2003) in their Regional Catchment Strategy (RCS) lists five important natural resource assets in their region, including biodiversity, waterways and wetlands, soils, forests and costal areas, the value of which their RCS seeks to protect. For waterway health and water quality, their management objective is *'to maintain and enhance the ecological health of the region's water resources and waterways while maintaining economic and social development'.* Further, their aspirational target is *'a net increase in water quality across the region and an equitable allocation of water between environmental, social and economic uses by 2050 as measured by key indicators'.*

It would be rare that all the environmental assets listed in regional catchment strategies would be impacted by an irrigation enterprise. However, it is useful at least in these initial stages to be aware of all the natural resource assets in the catchment.

It is common in stakeholder workshops and individual interviews for there to be much discussion in defining more precisely what these broad ecological values entail and how they are influenced by human intervention.

3.3.4 Hazard-effect relationships

Once the environmental values are defined, the threats and hazards to these values need to be

Box 3: Criteria for assessing an ecologically healthy river

The vision for Victoria's rivers is based on ecological sustainability (VRHS, 2002) and is given below.

"Our rivers that are of the greatest value to the community will be protected as part of our natural heritage.

Our rivers will be ecologically healthy and managed within healthy catchments:

- *supporting a diverse array of indigenous plants and animals within their waters and across their flood-plains;*
- *flanked by a mostly continuous and broad band of native riparian vegetation;*
- *with flows that rise and fall with the seasons, inundating floodplains, filling billabongs and providing a flush of growth and return of essential nutrients back to the river;*
- *with water quality that sustains crucial ecological functions;*
- *major natural habitat features are represented and are maintained over time;*
- *with native fish and other species moving freely along the river and out onto the floodplains and billabongs to feed and breed during inundation;*
- *replenishing productive estuaries or terminal lakes;*

whilst

- *providing the essential basis for efficient, high value sustainable agriculture and other resource-based industries;*
- *supplying clean and safe drinking water;*
- *providing pleasurable environments for those enjoying a range of leisure pursuits;*
- *preserving the values that are fundamental to our indigenous cultures; and*
- *maintaining the rivers' place in our collective history.*

Our communities will be confident and capable, appreciating the values of their rivers, understanding their dependency on healthy rivers and actively participating in decision-making.

identified, and linked to the potential ecological effects, so that the stakeholders can make an initial qualitative assessment of the risks to the key values.

There are a number of ways to go about this task, and this section outlines a three-step process we have found effective. This is best done in collaboration with the key stakeholders (often via a stakeholder workshop).

The steps shown in Figure 2 are:

- identify the most serious *ecological effect(s)* that could be realised for each ecological value;
- identify the *hazards (threats, stressors)* likely to cause each of the adverse ecological effects. This process is normally iterative and can be assisted by developing a *conceptual model* that links the hazards with the ecological effect upon the asset; and
- *rank* the ecological effects in importance to the stakeholders (covered in Section 3.4.1).

Ecological effects

In most cases, the broad management objectives to protect or enhance important environmental values or assets are not specific enough for an ecological risk assessment. For example, a management objective often stated in catchment or river management strategies is 'to maintain a healthy ecosystem'. Unfortunately, the definition of 'ecological health' is contingent upon the stakeholders involved and will change from case to case (Fairweather, 1999), and before the risk assessment can proceed, this seemingly simple statement must be defined and agreed upon. A useful vision for healthy rivers that is based on ecological sustainability is provided in the Victorian River Health Strategy (VRHS, 2002) and is given in Box 3. Similar sets of attributes are being developed to define 'healthy' wetlands, estuaries and coastal zones. These attributes will provide guidance on the features of aquatic systems potentially at risk from irrigation developments, and that need to be assessed.

In addition to maintaining ecosystems in a healthy state, it often important that other 'environmental or ecosystem services' provided by these natural assets are not diminished **(**Binning et al., 2001; VCMC/ DSE, 2003). *Ecosystem services* come from natural assets, such as soil, water, plants, animals, other , pest control, maintenance and regeneration of habitat, provision of shade and shelter, genetic resources, filtration and erosion control, maintenance of soil health, regulation of river flows and groundwater levels and waste treatment.

For example, Binning et al. (2001) identified twelve ecosystem services in the Goulburn Broken catchment in Victoria, including pollination, aesthetic benefits, pest control, maintenance and regeneration of habitat, provision of shade and shelter, genetic resources, filtration and erosion control, maintenance of soil health, regulation of river flows and groundwater levels and waste treatment.

Thus, an important task for the risk assessment team (in collaboration with the key stakeholders) is to better define the *ecological effects* that would result in an unacceptable reduction in the value of a particular asset. As shown below, the situation can quickly become quite complicated because one or more hazards can cause an ecological effect and a number of hazards may often adversely affect a particular ecological value.

Table 1: Checklist of potential off-site ecological issues associated with irrigation systems. * Acceptance criteria would need to be developed to determine what level of reduction in biodiversity is acceptable.

Activity	Hazard/Threat/Stressor	Potential ecological effect
Flow-related	• Changed flow regimes & reduced flows • Barrier (weirs, dams) • Poor water quality (low temperature, low dissolved oxygen)	• Reduced biodiversity* - interferes with breeding cycles, loss of habitat • Reduced biodiversity - interferes with fish migration • Toxicity to fish, alien species take over (e.g. trout)
Contaminants	• Increased nutrients (P, N) • Increased toxicants (biocides, heavy metals) • Increased turbidity and suspended particulate matter	• Increased frequency of algal blooms • Reduced biodiversity - due to toxic effects • Reduced primary production, smothering of benthic habitat
Salinity	• Increased salinity	• Reduced biodiversity due to toxic effects on aquatic biota and terrestrial plants

In addition to defining the type of ecological effect (e.g. reduced fish biodiversity), there is also a need to define the *acceptance criteria* for each effect. For example, what reduction in fish biodiversity is regarded as an acceptable risk? Defining acceptance criteria is generally done through a mix of expert opinion and stakeholder input.

Table 1 provides a checklist of potential ecological effects that could be caused by activities specifically associated with irrigation systems. This list was largely derived from the Case Study projects, and focuses largely on the adverse effects on off-site aquatic ecosystems. However, the ERA process should not be restricted only to these ecosystems. For example, it would be possible to also include adverse effects on wetlands, terrestrial vegetation and soil quality within the irrigation system.

Thus, three major activities associated with irrigation enterprises have been identified that can result in adverse ecological effects (Table 1). These are changes to rivers, tributaries and wetlands required to supply irrigation water, release of contaminants (nutrients, pesticides, herbicides), and increases in salinised runoff and seepage.

Table 2 contains some examples of ecological effects related to the protection of both wetland and river ecological health. The description of each ecological effect should contain both the ecological effect and the hazard(s) causing this effect. Note that at this stage only the possible adverse ecological effects have been identified and not their risk – the latter involves also assessing the likelihood that the effect will occur and the magnitude of the effect.

Assessment endpoints

Even though these ecological effects appear to be relatively explicit, they are often not explicit enough for an ecological risk assessment. It is therefore necessary to identify more specific and measurable attributes of the ecosystem or sub-components of the ecosystem – these are called *assessment endpoints*. Table 2 contains some examples.

Assessment endpoints are explicit expressions of the actual ecological value that is to be protected, and are the means by which the various risks will be quantified. They should be ecologically, socially and politically relevant, sensitive to the known or potential stressors, amenable to measurement, and relevant to the management goals (Suter, 1993).

They are vitally important in that they:

- provide an operational interpretation of the community agreed ecological values, and also provide the basis for demonstrating compliance with these socially mandated goals;
- help to focus the assessment specifically on the management concerns;
- are central to developing sensible conceptual models identifying the relevant stressor-response relationships (see p17); and
- assist in reducing linguistic or language uncertainty through the clear definition of endpoints.

Stakeholder involvement can often help to clarify the assessment endpoints that are best to use.

In addition to identifying the assessment endpoints, it is also necessary to determine the way in which possible changes will be measured (i.e. what metrics will be used). For example, the reduction in abundance and diversity of wetland macroinvertebrates (Table 2) could be measured by changes in the macroinvertebrate SIGNAL index. These more specific measures are often referred to as *measurement endpoints*.

Table 2: Examples of different ecological effects and assessment endpoints used to assess the risks to ecosystem values. * Desirable at this stage to also nominate how the particular change will be measured (see Section 4.3 and Box 12).

Ecosystem values	Ecological effects	Assessment endpoints*
Protection of wetland health	• Loss of biodiversity due to increased salinity	• Reduction in the abundance and diversity of macroinvertebrates • Reduction in the abundance and diversity of fringing macrophyte vegetation
	• Loss of aquatic macrophytes due to changes to the flow	• Reduction in the abundance and diversity of macrophytes
Protection of river health	• Reduction in native fish numbers due to cold water releases from a reservoir	• Reduction in the abundance and diversity of native fish
	• Toxicity to biota due to pesticide runoff	• Reduction in the abundance and diversity of native fish • Reduction in the abundance and diversity of macroinvertebrates
	• Blue-green algal blooms due to increased nutrient release from an irrigation area	• Increased frequency of cyanobacterial blooms, measured as the number of days the cyanobacterial cell numbers were >15,000 cells/mL
	• Loss of a threatened species due to loss of in-stream and riparian habitat	• Decline in the population size of the threatened species

Hazard identification

After establishing the ecological values of the ecosystem (or catchment) in question, and the assessment endpoints that best represent these values, it is then necessary to identify the hazards (threats or stressors) to these values in detail. Again the key stakeholders should be involved in this process so that their aspirations and motivations are clearly described at the start of the process.

It should be noted that although the processes of identifying ecological effects and the hazards causing these effects are separated in the text, in reality they are normally done together.

Hazards are sources of potential harm, or situations with the potential to cause loss or adverse effect (AS/NZS, 2000). Thus, hazards can be potentially harmful *stressors*, such as salinity, pesticides, heavy metals, nutrients, or they may be *threatening processes*, such as clearing of land or discharge of waste (AS/NZS, 2000).

We have found that focusing on hazards only, rather than the ecological values, can cause difficulties. Such a focus on hazards is the basis of the Hazard Analysis and Critical Control Point (HACCP) process (USFDA, 1997). There is a difference in philosophy between hazard analysis and risk assessment. Hazard analysis seeks to identify and prioritise those hazards likely to have an adverse effect on a particular asset. This is important for the manager who can then put in place actions to minimise the potential for adverse effects. On the other hand risk assessment seeks to quantify the probability that these hazards will have an adverse effect on the asset. Thus, hazard analysis focuses on the hazard, while risk assessment focuses on the asset or values potentially being affected.

For catchment risk assessments, it is possible that a number of hazards may compromise a particular ecological value. For example, in the native fish abundance ERA undertaken in the Goulburn-Broken system hazards considered included: water quality, flow changes, physical habitat changes, predation and barriers to migration (Box 7).

Table 3: Hazard-effect matrix showing the environmental values for Hattah-Kulkyne National Park and the potential threats (hazards) to those values (Carey et al., 2004). Cross denotes possible interaction between value and threat, with 'X' indicating strong interaction and 'x' weak. Highlighted cell denotes interaction considered in the next stage of the assessment.

Environmental Values	Potential Threats											
	Trespass or re-introduction of cattle	Overgrazing by kangaroos	Grazing by rabbits and goats	Inappropriate environmental flows	Irrigation impacts on groundwater	Predation by feral cats	Predation by foxes	Feral bees	Presence of weeds	Fragmentation of landscape	Inapropriate fire regime	Drought severity/frequency
Riverine woodlands	X	X	X	X	X		x	X	X		X	X
Four listed communities in Pine/Buloke/Belah complex	X	X	X		X		x	X	X		X	X
Mallee fowl	x	X	X			X	X			X	X	X
Regent parrot	X	x	X	X	X	X	X	X		X	X	X
Lizards	X	X	X			X	X			X	X	X
Mallee bird community	X	X	X	X	x	X	X	X	X	X	X	X
Wilderness values	X	X	X		X	X	X	X	X		X	X
Wetlands	X	X	x	X	X	X	X	X	X		X	X
Old growth/young mallee mosiac	x	X	X		x			X	x		X	
Availability of hollow-bearing trees	X	X	X	X	X			X		X	X	X

We recommend that the risk assessment team work with the stakeholders to prepare an initial hazard-effect matrix. Here the stakeholders list all relevant hazards against each of the ecological effects (or assessment endpoints). An example of a hazard-effect matrix is given in Table 3. We have found it useful after this initial hazard identification step, to refine the list of hazards around the development of specific cause-effect conceptual models that directly link the hazards to each ecological effect (assessment endpoint). This is covered more fully in the next section.

Specific cause-effect conceptual models

Conceptual models are diagrammatic representations of the current understanding of the overall ecosystem or of parts of the system.

We recommend that a *cause-effect conceptual model* be developed for each of the priority ecological effects (Figure 2). These help to focus the risk assessment process and to provide a basis for discussions about particular ecosystems and hazards with stakeholders. Additionally, it may be possible to capture alternative opinions of stakeholders in alternative models.

These issue-specific (or meso-scale) conceptual models are an ideal way to link the key hazards (threats/stressors) to the ecological effect or more specifically to the assessment endpoints. We have found it best to build at least some of the issue-specific conceptual models at the stakeholder workshop, although they can take considerable time. In almost all cases we have found it necessary for the risk assessment team to do further work on the various conceptual models after the stakeholder workshop.

Figure 4 shows two issue-specific conceptual models developed by stakeholders at a workshop focused on ecological risks in the Woori Yallock Creek catchment near Melbourne. Figure 4a shows the relationships between hazards likely to affect blackfish populations in Woori Yallock Creek, and Figure 4b the hazards affecting the intactness of the riparian zone along this creek. Note that in both cases, the means by which the assessment endpoints would be measured also needs to be defined.

Because of the time taken to develop these conceptual models, a judgement often needs to be made on whether additional workshops should be run to develop the remaining issue-specific models, or whether these additional conceptual models can be developed by the risk assessment technical team.

The involvement of stakeholders in the process of developing conceptual models has three positive outcomes:

- it provides the stakeholders with some ownership of the process;
- it often brings out other knowledge that is not formally documented; and
- it is a very useful means for increasing participants' basic knowledge of the ecosystems being assessed.

These issue-specific conceptual models also have two other important functions:

- they form the basis of the predictive cause-effect models that the risk assessment team will need to build to assist in assessing the identified ecological effects; and
- they are useful in clearly identifying the available knowledge and the key knowledge gaps for each issue.

In both cases the assessment endpoint is representative of one of the ecological issues for which the risk is being assessed. Note that the means by which the assessment endpoints would be measured also needs to be defined - (a) measured by abundance of Blackfish recruits, (b) measured by the condition of the trees (presence of dieback) and evidence of revegetation

3.4 Risk analysis

Conversion of the information gathered in the previous step on hazards and their potential ecological effects (the hazard-effect relationships) to a risk assessment involves gathering information on:

- the likelihood of the hazard(s) having an effect; and
- the size or magnitude of the effect if it does occur.

There are three levels at which this analysis of risks can be undertaken (Figure 5):

- tier 1 - Qualitative risk analysis – where words are used to describe the magnitude of potential consequences and the likelihood that those consequences will occur;
- tier 2 - Semi-quantitative risk analysis – where values are given to the qualitative scales used above to produce a more expanded ranking scale; and
- tier 3 – Quantitative risk analysis – where numerical values (not descriptive scales) are provided for both the consequences and likelihood using data from a variety of sources.

These are outlined in the following sections.

3.4.1 Initial screening using qualitative or semi-quantitative risk assessment

It is normal for the stakeholders to identify a large number of ecological issues and associated hazards. Since not all of these will be high priority, there is a need for a process by which these can be quickly ranked so that the risk assessment team can work on more quantitative methods for predicting the risk for the most important hazards. This is normally done using either a qualitative or semi-quantitative risk analysis (Figure 5).

A *qualitative risk analysis* is one that uses descriptions rather than numerical means to define risk (AS/ NZS, 2004b). Thus a typical qualitative risk matrix might describe the consequence as minor, moderate or major, and the likelihood as unlikely, possible or likely. Obviously, these simple terms are quite vague and may be interpreted quite differently by different people, and the resultant subjectivity is not transparent. Often, a more detailed description is provided in an attempt to make them more meaningful.

However, despite these attempts, qualitative risk ranking has a number of serious problems (Burgman, 2001). The ranking is entirely subjective and is dependent upon the mix of stakeholder skills and

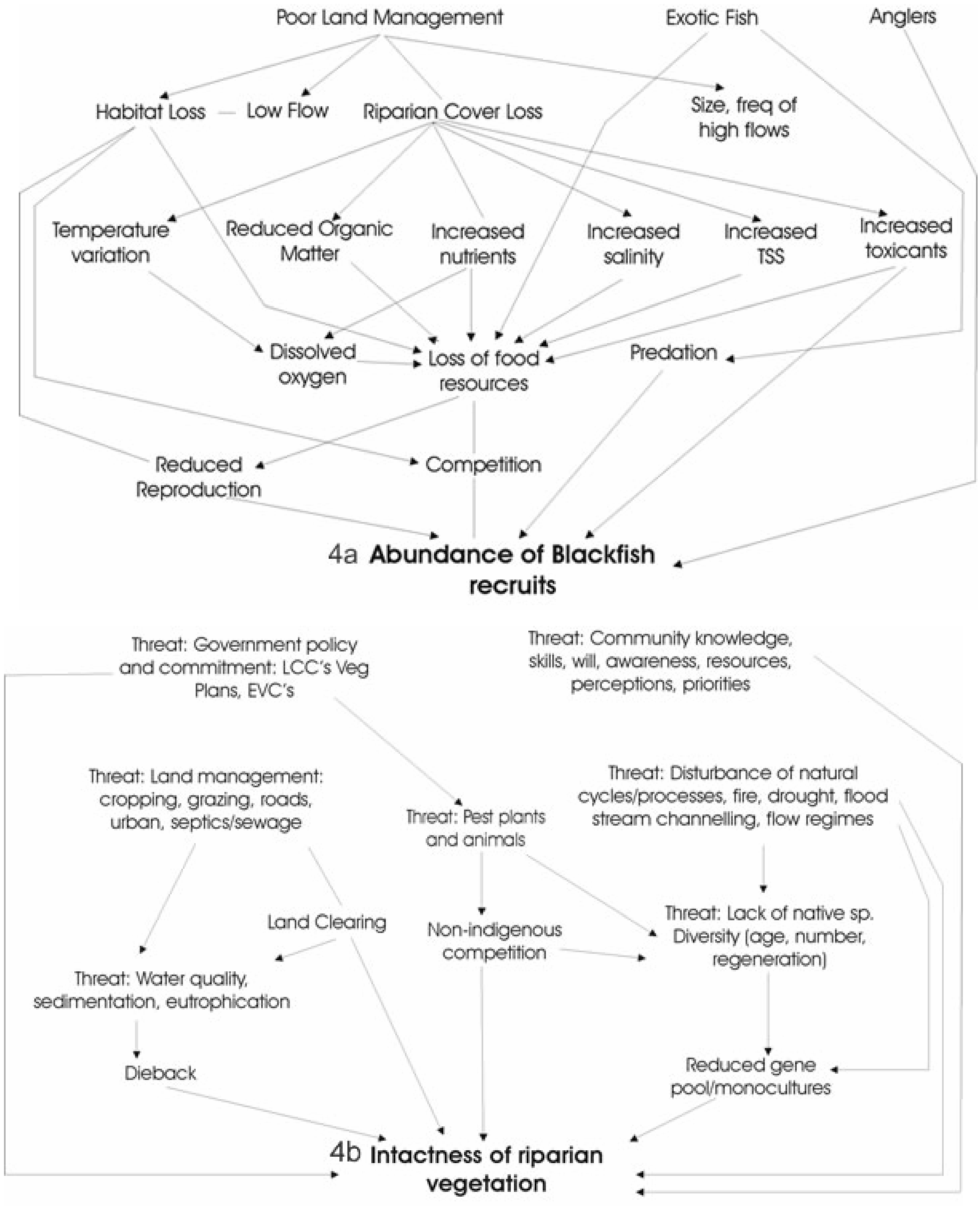


Figure 4: Cause-effect conceptual models for (a) sustainable native fish populations, and (b) intactness of riparian vegetation in the Woori Yallock Creek catchment. In both cases the assessment endpoint is representative of one of the ecological issues for which the risk is being assessed. Note that the means by which the assessment endpoints would be measured also needs to be defined – (a) measured by abundance of Blackfish recruits, (b) measured by the condition of the trees (presence of dieback) and evidence of revegetation.

experience. Additionally, any assumptions and biases inherent in the stakeholder choices are not made transparent.

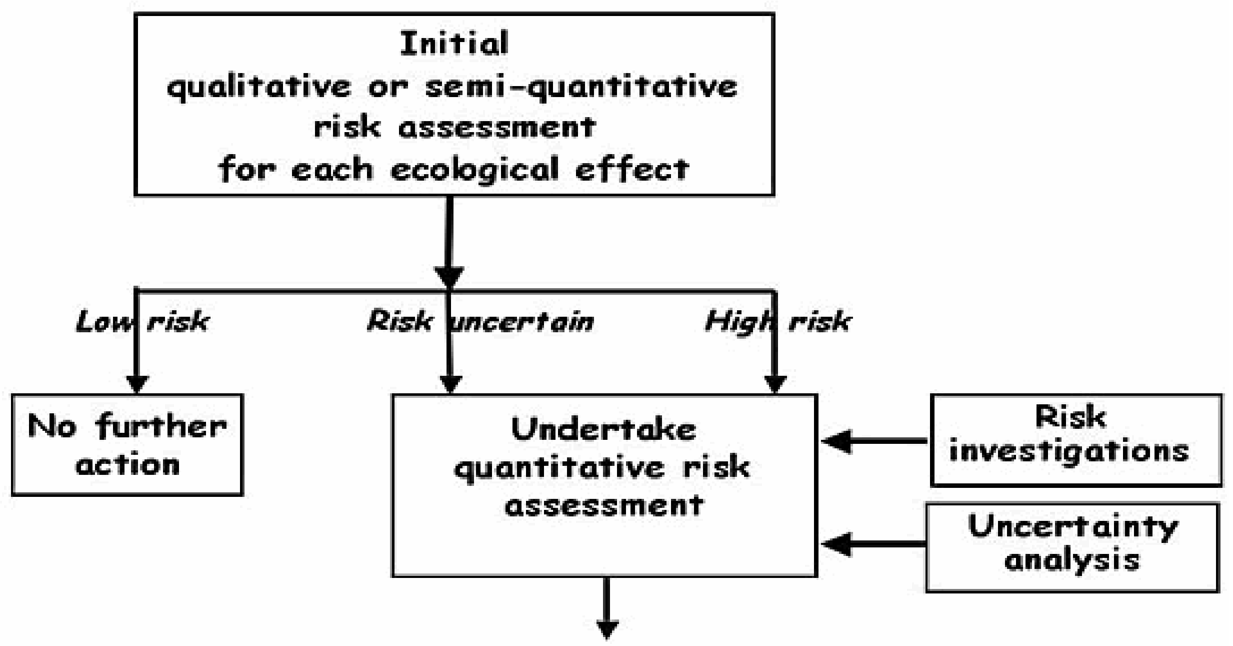


Figure 5: Flow chart showing the three tiers of risk analysis (qualitative, semi-quantitative and quantitative)

Semi-quantitative risk analysis was introduced in an attempt to reduce some of the inherent problems with qualitative analyses and to offset the often strong resistance by stakeholders to progress to fully quantitative methods (AS/NZS, 2000, 2004b).

Our experience, and that of Carey et al. (2004a,b) and Burgman (2004), is that the five point scoring system outlined in the Australian Standard is very effective (AS/NZS, 2000, 2004b). Table 4 provides an example of such a semi-quantitative risk matrix. Both the consequences and likelihood for each ecological effect is rated from 1 to 5.

In the workshop, we ask each stakeholder to independently (and subjectively) assign scores on the scale of 1 to 5 for:

- the likelihood of the ecological effect occurring (ranging from highly unlikely to almost certain); and
- the magnitude (severity) of its consequences (ranging from insignificant to catastrophic).

The stakeholder risk scores (likelihood x consequences) are then ranked with possible score ranging from 1 to 25. As shown by the numbers in Table 4, the higher the score the higher the risk. This type of assessment is not designed to produce quantitative estimates of risk, simply to produce scores for ranking of the risks.

Often there are obvious differences between the rankings derived from different stakeholders. These can be discussed with a view to identifying, and possibly resolving, these differences. In many cases, these differences are caused by linguistic uncertainties or vague definitions (Regan et al., 2002). Different stakeholders may attach different meanings to words and phrases used to describe hazards and how they potentially affect the ecological value. In these cases, it is important that the stakeholders discuss any differences in interpretation to see if there is common ground. The issue-specific conceptual models can also help in this respect, as they can more clearly show the linkages between particular hazards and how they might affect the ecosystem.

Following the discussions, the assignment of likelihood and consequence to each ecological issue by each individual stakeholder may be repeated, leading to a second set of risk rankings. An important outcome of this process is that the low ranked issues that all stakeholders agree upon can then be omitted from the risk assessment.

In those cases where the differences are genuine, an important role of the subsequent quantitative risk assessment is to assist in deciding which alternative view is most probable. Thus, all plausible options should be retained in alternative model formulations.

Carey et al. (2004b) discuss the use of a software package (Subjective Risk Assessment) for calculating and displaying the Spearman's rank correlations between participants. This information can assist the discussions to resolve any linguistic uncertainties.

Table 4: Categories of risk for a qualitative assessment (refer AS/NZS, 2000, 2004b; OGTR, 2004). Likelihood and consequence are each scored subjectively on a scale of 1 to 5. Risk ratings, as the product of likelihood and consequence, are shown in the body of the table. In this particular scheme, risk ratings have been categorized as follows: 15-25 High (dark shading); 5-12 Moderate (light shading); and 1-4 Low (unshaded).

Likelihood		**Consequence**				
		Insignificant (1)	Minor (2)	Moderate (3)	Major (4)	Catastrophic (5)
Almost Certain	(5)	5	10	15	20	25
Likely	(4)	4	8	12	16	20
Moderately likely	(3)	3	6	9	12	15
Unlikely	(2)	2	4	6	8	10
Rare	(1)	1	2	3	4	5

The output from an initial qualitative or semi-quantitative risk assessment allows the risks to the key ecological values to be arranged into three broad categories (Figure 5):

- low risk;
- high (or moderate) risk; and
- uncertain risk.

Sometimes this is sufficient to allow a risk management plan to be prepared. However, in controversial situations or where large sums of money will be required to minimise the risks, a quantitative assessment is normally required for the cases where the risk is moderate, high or uncertain.

3.4.2 Quantitative risk assessment

Quantitative risk analyses use numerical values (not descriptive scales) for both the consequences and likelihoods using data from a variety of sources (AS/NZS, 2004a).

A number of quantitative approaches are available to assess risk to the identified ecological values, ranging from simple desktop studies to full predictive modelling (Burgman, 2005). The issue-specific cause-effect conceptual models developed previously can be very useful in developing ratings of likelihood and consequence that are more quantitative than the vague value judgements used in qualitative and semi-quantitative risk analysis.

Selection of an appropriate analysis method from the wide range available will be dictated by the importance of the issue, the time, expertise and financial resources available, the availability of data and other information, and knowledge about the system and the cause-effect relationships.

A number of available methods are reviewed below. It should be noted that tracking sources of data, getting access and then synthesising these data into a form that can be used in a semi-quantitative or quantitative risk assessment is often a very time consuming and frustrating exercise, and it is common for the time to achieve these tasks to be significantly underestimated.

Environmental risk assessment calculator

This is a very simple semi-quantitative method for assessing risk. The method appears to have been first developed to assess environmental risk to humans (Keeney & Raiffa, 1976; QldDETIR, 1999). Several companies (e.g. AMCOR) have employed this method to increase the awareness of staff to common workplace risks.

A version of the method has also been used in an environmental impact assessment associated with a potential new dam in Queensland (Burnett Water, 2001).

Risk calculators are nomographs relating probability and duration to a hazard through a tie line to possible consequences and then to a risk score.

We have adapted the method to allow risks to natural resources to be assessed. An example is provided in Box 4 where an environmental risk calculator was developed and used to assess the risks associated with irrigation return drains (MIL, 2005).

Decision or logic trees

Logic trees are diagrams that link the processes and events that could lead to or develop from a hazard (Stewart & Melchers, 1997; Hayes, 2002b; Burgman, 2005). Two types of logic trees are used: *fault trees* work from the top down to link a chain of events with the outcomes, and *event trees* work from the bottom up and link possible outcomes to an initiating event(s). Logic trees are most used in engineering and are a very useful way to formalise a conceptual model (Stewart & Melchers, 1997). A full discussion of the development and use of logic trees, with many useful examples, is provided by Hayes (2002) and Burgman (2005). Box 5 provides an example of a simple decision tree to assess the risk of an algal bloom occurring in a lowland river.

Ecotoxicological & probabilistic methods

Ecotoxicological risk assessments have been used for many years to assess the risks to ecosystems from chemicals, existing and new (Suter, 1993; USEPA, 1998). The objective is to protect ecosystems, usually by protecting individual species and avoiding irreversible ecological change (Calow & Forbes, 2003).

Probabilistic risk assessment allows the effect of a single chemical (stressor) on relevant biota for which ecotoxicological data is available. The method involves comparing the cumulative distributions of the concentration of the chemical (likelihood data) with the ecotoxicological data of the relevant taxa (consequence data) (Solomon et al., 1996, 2000; Van Dam et al., 2004). Risk is quantified as the proportion of readings for which stressor concentrations exceed the tolerance value of a given percentage of the assemblage (see Box 7 for details).

The stressor concentration distribution is generally obtained from monitoring data, although when this is unavailable computer modelled data may need to be used (Hart et al., 2003). The consequence distribution is obtained by combining the ecotoxicity data for relevant species to produce a species sensitivity distribution (SSD). It is also necessary to decide upon the proportion of the species to be protected. The Australian water quality guidelines provide threshold concentrations for a range of toxicants relevant to the protection of 90%, 95% and 99% of species, depending upon the level of protection required for the particular environment (ANZECC/ARMCANZ, 2000a).

Boxes 6 and 7 provide examples of ecotoxicological risk assessments of relevance to the management of aquatic ecosystems. Box 6 provides an analysis of the possible toxic effects on fish caused by low dissolved oxygen concentrations, while Box 7 provides an example of a probabilistic risk analysis of the effects of increased salinity on wetlands.

Predictive models

A fully quantitative risk assessment normally requires the development of a predictive quantitative model for each ecological issue, and may include the evaluation of the consequences of various management options. It may also be necessary to develop and compare more than one model if the stakeholders have provided sensible alternative models.

Invariably, the predictive model will be based on converting the conceptual model into a set of more formal mathematical relationships. This is relatively simple for the few cases where the problem being considered is reasonably well understood. However, in many cases the lack of knowledge about the links between the stressor(s) and the ecological effect makes it difficult to quantify the likelihood. In fact, the most challenging part of most quantitative risk analyses is the estimation or prediction of the likelihood of an adverse effect occurring.

Below the various types of predictive models are reviewed. The choice of the most appropriate analysis method from the wide range available will be dictated by the importance of the issue, the time, expertise and financial resources available, the availability of data and other information, and knowledge about the system and the cause-effect relationships, and also the nature of the predictive information required (i.e. how precise does it need to be?).

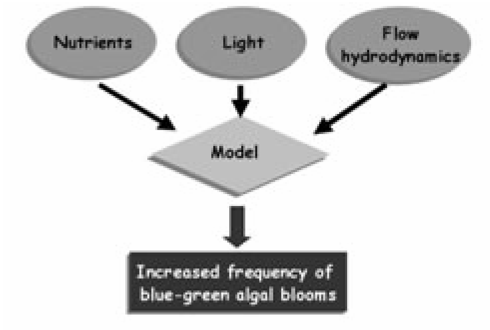
Because of the additional technical skills, time and resources needed, there is often considerable resistance to undertaking a quantitative risk analysis. However, they have the advantage that they make explicit the absence of data, simplifying assumptions and uncertainties, rather than leaving these largely unsaid as in most qualitative risk assessments.

Box 4: Risk analysis: Decision tree to assess the risk of cyanobacterial bloom in a lowland river

Achieving a reduction in the frequency and severity of blue-green algal blooms in many lowland rivers in Australia has been the concern of natural resource managers and the local community for many years. The Figure below shows a simple decision tree for predicting the risk from algal blooms in a lowland river (Hart et al., 1998; Cottingham et al., 2002).

This is based on a conceptual model in which three factors are considered to control blue-green algal growth:

- Adequate concentrations of nutrients (surrogate Total Phosphorus (TP),
- Adequate light for photosynthesis (surrogate Turbidity), and
- Low flow conditions that will allow the algal populations to achieve bloom numbers before they are washed out (see Hart et al., 1998 for discussion on this factor).

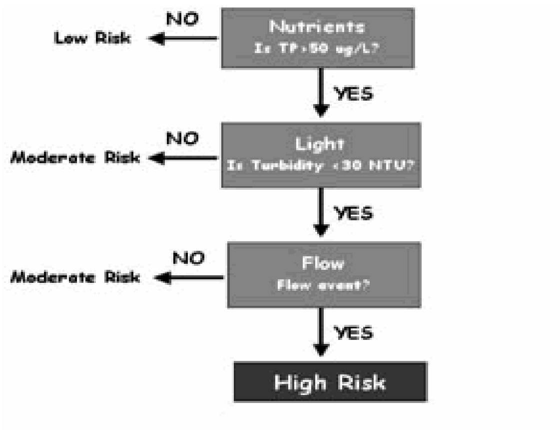


Simple conceptual model showing the main factors controlling blue-green algal growth.

Oliver & Ganf (2000) have developed a more detailed decision tree for blue-green algae.

Information on the risks of an algal bloom forming is likely to be used in two main ways:

- To identify sites across a catchment that have the highest risk of a bloom occurring (regional scale),
- To assess the risk of a bloom developing at any one site (local scale).



Decision tree for predicting risk of algal blooms

The simplest use of this model would involve comparison of a point estimate (e.g. mean, median) for each of the three variables relevant to the study system with a threshold to obtain a Yes/No decision. The cyanobacteria threshold values in the example decision tree were obtained from published water quality guidelines (e.g. ANZECC, 2001).

In the example shown, if the nutrient concentration measured by TP is >50 g/L, and there is sufficient light (assumed to occur if the turbidity is <30 NTU) and the flow conditions are right (Hart et al., 1998), then there is a high risk that an algal bloom will occur.
This decision tree method is semi-quantitative, in that while quantitative trigger values are provided for TP, turbidity and flow, the combination of these drivers is still qualitative.

In this form, with single thresholds for TP concentration, turbidity and flow conditions, this simple decision tool does not include any explicit consideration of the known variability in these drivers. Cottingham et al. (2002) and Hart et al. (2002) discuss modifications to the simple decision tree to make it more quantitative by using the known data distributions for each of the three drivers.

Dynamic simulation models

The most common form of quantitative risk assessment is to build an explicit dynamic mathematical model, including stochastic elements to represent natural variability, and to solve the model to explore assumptions and evaluate knowledge gaps with sensitivity analysis (Vose, 1996, 2000). Monte Carlo simulation models for ecological systems have been used since the 1950s (Burgman, 2005).

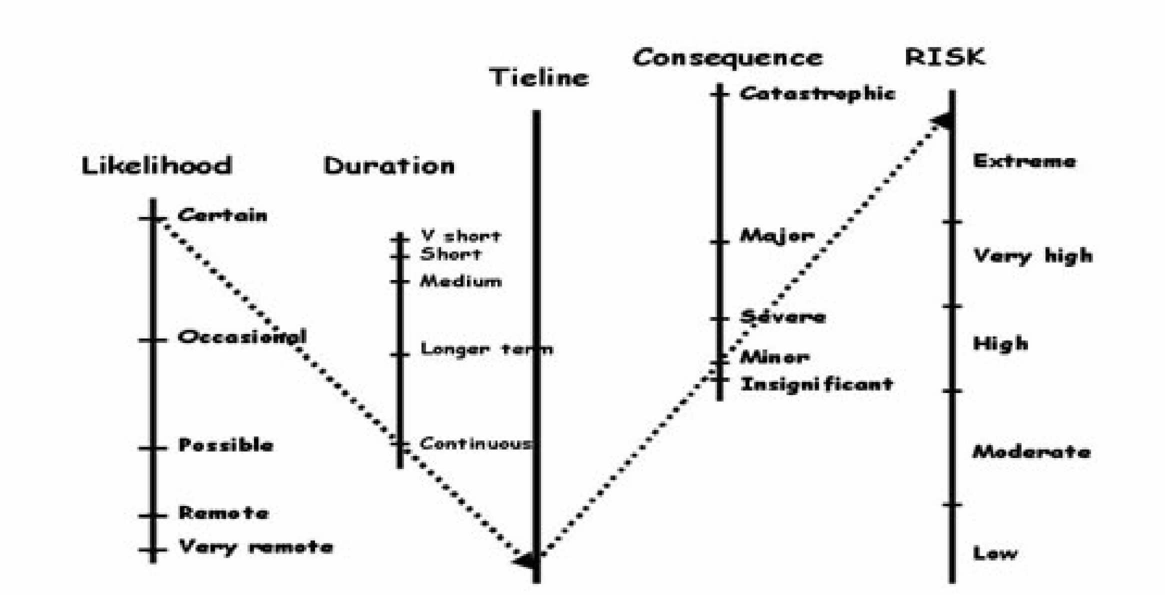
Box 5: Environmental risk assessment calculator

This simple semi-quantitative method for assessing environmental risk to humans has been adapted by MIL (2005) to assess the risks associated with irrigation drains and to form the basis for a drains risk management plan. The risk calculator is a nomograph (see below) relating *probability* and *duration* to a hazard through a tie line to possible *consequences* and then to a *risk score*. The MIL risk calculator was developed at a workshop attended by 15 MIL technical staff.
The various ratings agreed at the workshop are listed in the Table below:

Likelihood	No	Duration	No	Consequence	No	Risk	No
Almost certain (daily)	1.0	Very short (1 d)	1	Catastrophic	20	Extreme	>20
Occasional (weekly)	0.6	Short (2 d)	2	Major	10	Very high	15-20
Possible (monthly	0.3	Medium (5-7 d)	3	Severe	5	High	10-15
Remote (yearly)	0.1	Longer term (30 d)	6	Minor	2	Moderate	3-10
Very remote (>1 in 10 years)	0.05	Continuous	10	Insignificant	1	Low	<3
						Very low	0

A total of 10 issues were assessed, these being: chemicals in drains, salt load in drains, nutrients and turbidity, drain maintenance (e.g. weeds), flooding, water harvesting from the drain (e.g. reuse), infrastructure failure, litter in drains, channel escape flows and pest animals in drains.

The figure shows an example of the risk due to ingress of salty groundwater into a drain and the subsequent impact on a receiving water. For this issue the risk assessors rated the likelihood as a1most certain (1.0), the duration as continuous (10), the consequences as between severe and minor (3), resulting in a risk rating of extreme (1.0x10x3 = 30).



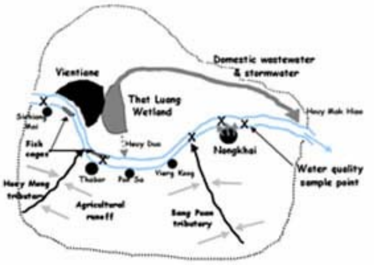
Box 6: Risk analysis: Toxic effects on fish of low DO concentrations

A method for predicting the risk of toxic effects to fish from low dissolved oxygen concentrations in the Mekong River is provided. More detail is available in Hart (2004b).

Problem formulation

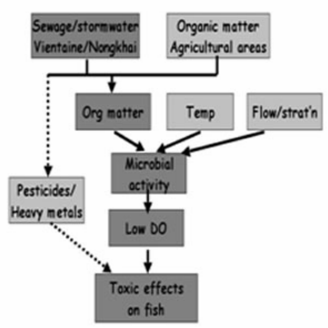
Objective - to determine the potential risks to the ecological health of the Mekong River in the study region (Vientiane, Nongkhai) from urban wastewater, agricultural runoff and changed flow regimes over the next 10 years.

Ecological issue – the adverse effects on the biological communities (particularly fish) in the Mekong River due to toxicity caused by low dissolved oxygen concentrations and high concentrations of pesticides and herbicides.



'Big picture' conceptual model - shown in Figure below. The assumptions underlying this model are:

- The main inputs of wastewater to the Mekong River will occur throughout the year from domestic & industrial discharges and stormwater from the two urban centers, and wastes from fish cages.
- The wastewater & stormwater discharges will contribute organic matter and nutrients to the Mekong.
- Agricultural runoff will enter the Mekong River predominantly in the wet season via a number of tributaries, and will contribute organic matter, nutrients and pesticides to the Mekong.



Cause-effect conceptual model - is shown in the figure below. The assessment endpoint is toxicity to fish in the study region. The assumptions underpinning this model are:

- The main toxic effect is due to low dissolved oxygen concentrations during low flows in the dry season,
- Dissolved oxygen concentrations will be mainly affected by the inputs of organic matter from sewage discharge, stormwater runoff and discharges from fish cages,
- Agricultural inputs (pesticides, herbicides) will occur during wet season from runoff when dilution is high. For this reason, and the fact that there is no pesticide monitoring information available, we have assumed pesticide toxicity is very low.
- The main problems are expected in the Mekong River (a) in the region of Nongkhai, and (b) further downstreamwhere Vientiane's wastewater enters the Mekong (see Figure above).

Semi-quantitative risk analysis

Consequence analysis **-** Information on the sensitivity of (tropical) fish to low DO was obtained from the literature. While there is a considerable amount of data relating to the toxicity of low DO to a wide range of organisms, making it possible to construct a species sensitivity distribution (SSD), there is very little information relevant to tropical fish (and even less for larval fish).
The thresholds selected are:

Consequence	Description	DO concentration
Major	Major impact, contributing to long-term degradation that cannot be reversed, e.g. fish death)	<2mg/L <20% saturation
Moderate	Transient impact that does not contribute to long-term degradation	2-6 mg/L 20-80% saturation
Minor	Negligible or transient impact, does not contribute to long-term degradation	>6mg/L >80% saturation

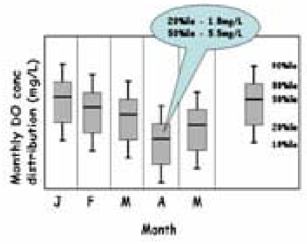
Likelihood analysis - Because there was very little DO monitoring data available for the Mekong River at any sites in the study region, DO concentrations were simulated using a water quality model (WASP6) of the system. The time step of the WASP6 model was 6-hourly (i.e. 4 per day) over a 12-month period, and DO profiles were calculated for each of 5 segments. The lowest flows in the Mekong at Vientiane were used for these calculations

(Box 6 continued on page 25)

(Continued from page 24)

(estimated from a 40 year flow record).

It was assumed that if the risks due to low DO were not high at this 'worst case' situation, then they would be low at all other flow conditions. However, if the risks were assessed as high at these low flow conditions it would be necessary to undertake further modelling to estimate the DO concentrations at other river flows.



The 6-hourly DO concentrations were then plotted on a frequency curve for each month – this curve was available as a cumulative frequency curve or a box plot (see Figure).

The likelihood thresholds were assumed to be:

Likelihood	Description	DO concentration
Unlikely	Low DO could occur at some time but is not expected	<20 percentile (%ile)
Possible	Low DO will probably occur at some time	20-50%ile
Likely	Low DO will probably occur in most circumstances	>50%ile

Putting these together

Analysis of the data showed that the DO concentrations are lowest during April in Segment 5. For April, the DO concentration rankings were: likely - >5.8mg/L, possible - 1.8-5.8mg/L, unlikely - 1.8mg/L (see figure).

Using the risk matrix for these data show that the risks range from high to moderate to high.

H - high risk: M - moderate risk: L - low risk

Likelihood	Consequences		
	Minor (>6mg/L)	Moderate (2-6mg/L)	Major ((<2mg/L)
Unlikely (<20%ile)	**L**	**M**	**H**
Possible (20-50%ile)	**M**	**M-H**	**H**
Likely (>50%ile)	**M**	**H**	**H**

Deterministic models

There are now an increasing number of (bio)physical models available that do a reasonable job in predicting the generation and transport of contaminants, such as salt, sediment and nutrients, within catchments. Examples include:

- EMSS (Environmental Management Support System) – is a catchment-scale model used to estimate daily runoff and pollutant loads. It has two components – a hydrologic model and a pollutant export load model. The input climate data into the model are daily rainfall and mean monthly area potential evapotranspiration, and the main model outputs are total daily runoff and total daily pollutant loads (total suspended solids, total phosphorus and total nitrogen) (Vertessy et al., 2001)[3];
- SedNET - constructs sediment budgets for river networks to identify patterns in the material fluxes. This can assist effective targeting of catchment and river management actions to improve water quality and riverine habitat[4]; and

3. Web site: www.healthywaterways.org/ PAGE170624PM2P324V
4. Web site: www.toolkit.net.au/cgi-bin/WebObjects/toolkit.woa/wa/productDetails?productID=1000013

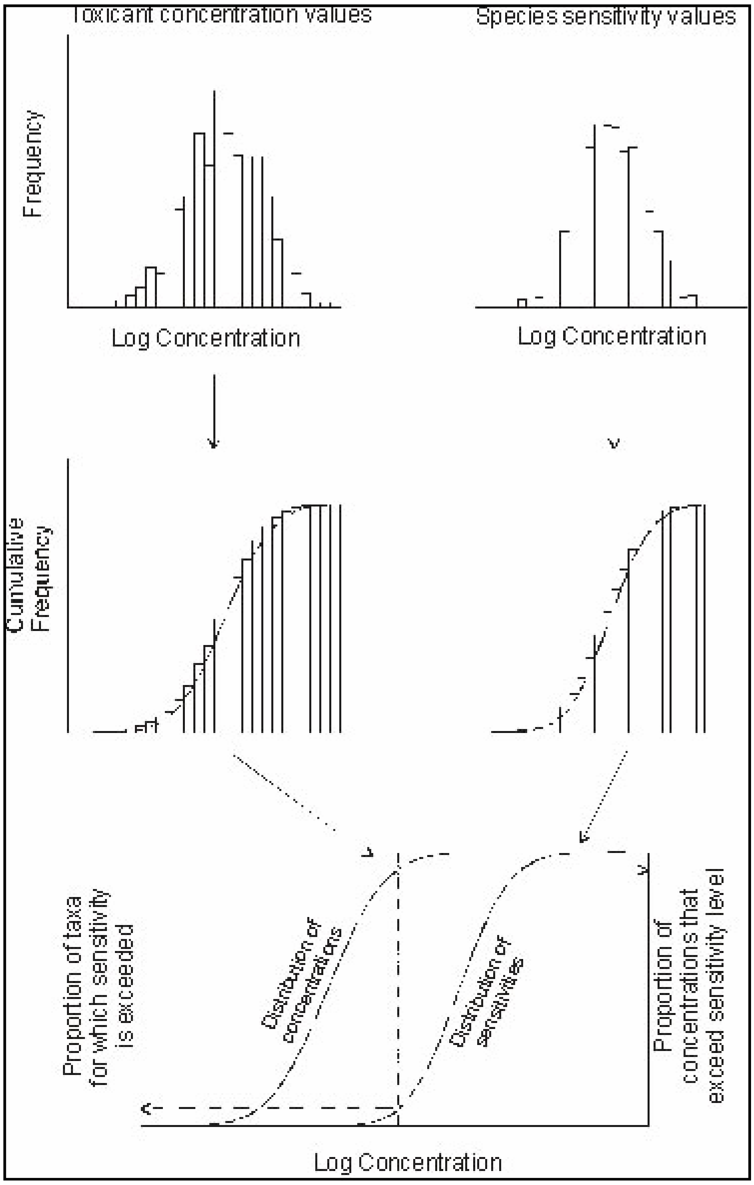
Box 7: Risk analysis: Salinity case study

A risk assessment of the effects of salinity on aquatic biodiversity in the Goulburn-Broken catchment is briefly described here. Further details are available in Webb & Hart (2004).

The focus of this risk assessment was determined by the funding body and not by the stakeholder workshop. However, the stakeholder workshop did determine the scope of the study. The risk assessment examined biodiversity as the principal endpoint, and considered the entire catchment both at the present, and also in the future. For practical purposes, these objectives were refined to:

- 5 case study sites that were located throughout the catchment,
- biodiversity was expressed in terms of expected loss of species richness at the case study sites due to salinity,
- the risks were assessed for the present day and for future scenarios at 20, 50 and 100 years hence, under an assumption of 'no further intervention' in salinity management.

The probabilistic risk assessment method of Solomon et al. (2000) was used for this study. It is suited to situations where there exist good data on both occurrence of the stressor (likelihood), and toxic effects of the stressor (consequence), but for which understanding of the precise links between cause and effect is poor. It also is ideal for examining many taxa at once, and hence for assessing the biodiversity effect of the stressor.



Probabilistic Risk Assessment works by comparing the cumulative distributions of the level of stressor and of the tolerance values of the resident taxa. Risk is quantified as the proportion of readings for which stressor concentrations exceed the tolerance value of a given percentage of the assemblage (see Figure).
Likelihood data for salinity were drawn from a REALM model of the Goulburn Broken catchment. For the future scenarios, data from the Ultimate Salt Load project (SKM, 2004) were used with the REALM model to synthesize daily salinity values. Consequence data were synthesised by combining data on salt tolerance of taxa in the field (Bailey & Boon, 2002) with species lists compiled for each of the case study sites (Webb & Hart, 2004).

To account for uncertainty caused by the fact that the risk assessment used samples of both likelihood and consequence data to represent the entire populations of data points, 95% confidence limits around both data distributions were used to provide best-case and worst-case scenarios, in addition to the more common median "best" estimates described above.

Originally the risk assessment aimed to examine risks in both running water (rivers, streams) and standing water (lakes, wetlands) sites.

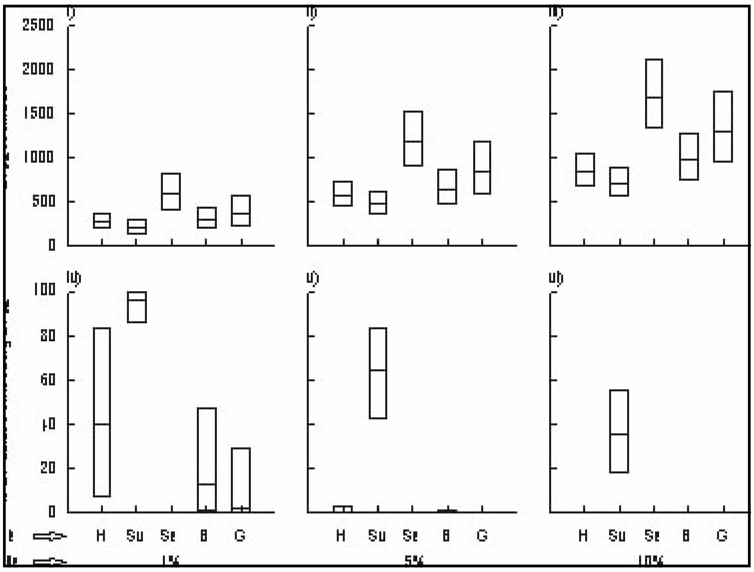
However, there were insufficient data for the standing water sites, both in terms of salinity regimes, and also of the biotas that inhabit those sites. This was recorded as a major knowledge gap, and a priority for future research.

A site situated on Sunday Creek was found to be currently at greater relative risk than the other sites considered.

(Box 7 continued on page 27)

(Continued from page 26)

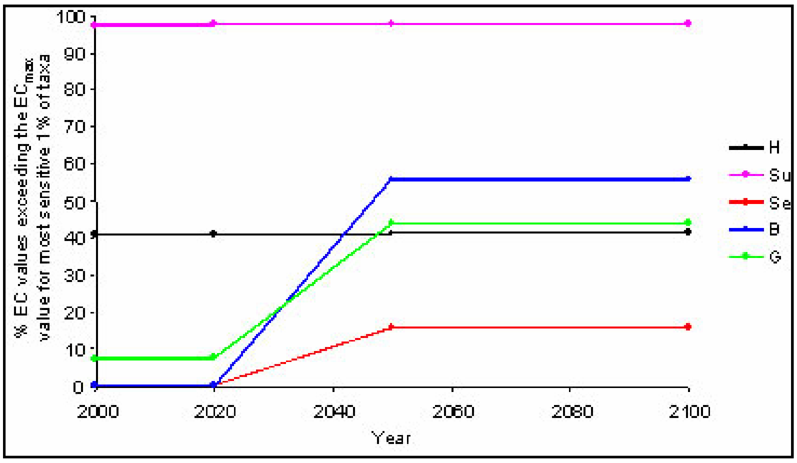
Risk ranking was due primarily to the salinity regimes being experienced at the case-study sites, rather than the assemblage at any one site being more sensitive than at others. The most sensitive assemblage was found at the site with the most saline flows. This was taken as evidence that salinity has not affected assemblage composition at any of the case-study sites to this point.



Graphs (i) to (iii) show estimates of salinity tolerance (ECmax) for the indicated percentage of most sensitive species recorded at each site. Graphs (iv) to (vi) show the proportion of EC values from each site that will exceed the ECmax estimate. Boxes span the range of upper and lower 95th percentile estimates. Centre line within boxes shows the median or 'best' estimate.

In terms of future relative risks, greater increases in salinity between 2020 and 2050 for the lower Goulburn River and lower Broken River than for the other sites will lead to a greater change in relative risk for these sites. Risks at two sites will not change greatly, and risks at the Seven Creeks site will change by a small amount.
Abbreviations: H = Hughes Creek, Su = Sunday Creek, Se = Seven Creeks, B = Broken River, G = Goulburn River.

The figure below shows the proportion of EC readings for which the ECmax value for the most sensitive 1% of species found at each case-study site will be exceeded in 2020, 2050 and 2100. X-axis shows years. Plotted points indicate the years for which the calculations have been made.



Co-occurring stressors other than salinity were not considered, and these could be expected to affect the tolerance of a species in the field. However, the technique chosen allowed an assessment of the risks to a broad range of taxa at several study sites, without having to develop an understanding of the mechanisms by which salinity affects various taxa. These mechanisms will be many, and will operate differently on different species. In its current form, the salinity risk assessment can be used to assist management decisions on salinity mitigation schemes and rehabilitation of sites. Its major strength is the use of pre-existing salinity data and salinity tolerances without the need for further investigations. The study identified a substantial knowledge gap with regards to data existing for wetlands. These data are of primary importance, as it is almost certain that wetlands will be at greater risk from salinity than will running water sites.

- Salt transport models – There is a myriad of salinity models developed or under development across Australia (Littleboy et al., 2003). A recent stocktake of models for the National Action Plan for Salinity and Water Quality (URS Australia, 2002) contained details of approximately 90 models that can be used to assess salinity management options. The major focus of salinity management in eastern Australia has been the concept of achieving a stream salinity target, and consequently, salinity modelling in Eastern Australia has a stream electrical conductivity (EC) focus to assess stream targets. Models have been developed that predict future increases in stream EC as well as the impacts of intervention strategies on stream EC (Littleboy et al., 2003). Complex groundwater modelling in Eastern Australia has typically focussed on groundwater allocation or groundwater contamination rather than salinity.

However, these models all suffer from the following deficiencies:

- they do not address multiple stressors in any systematic way;
- they rarely couple the contaminant with its ecological effect, particularly in downstream waterways, wetlands and estuaries (i.e. are not good quantitative cause-effect models); and
- they do not explicitly treat uncertainty.

Of course, the task of assessing the ecological risks associated with contaminants generated from multiple sources within a catchment is a difficult one, since many of the processes being predicted are both complex (many interrelationships) and inherently variable (mostly driven by climate) (USEPA, 2003).

Algal bloom models

Many different models are available that seek to predict the occurrence of cyanobacterial blooms based on other biophysical information (e.g. flow, nutrient levels, light climate, grazers). These models take many forms and include explicitly process-based models (e.g. Everbecq et al., 2001), models that incorporate empirical relationships as well as processes (e.g. Reynolds et al., 2001), purely empirical models (e.g. Dillon and Rigler, 1974), and neural network models (e.g. Recknagel, 2002). Some of these models have been taken up for use in management of waterbodies (Reynolds et al., 2001).

Box 8 provides information on a modelling study to determine the risk of algal blooms in the Goulburn-Broken river (Feehan et al., 2004).

Aquatox

AQUATOX[5] was developed by the US EPA (Office of Science and Technology) and is a freshwater ecosystem simulation model that predicts the fate of various pollutants, such as nutrients and organic chemicals, and their effects on the ecosystem, including fish, invertebrates, and aquatic plants. It is composed of a complex simulation model that attempts to model complex ecosystems, with seasonal and annual variations and multiple interactions among species. AQUATOX is used by the US EPA as a tool for performing ecological risk assessments for aquatic ecosystems, and can be used for improving ecological understanding and for testing environmental management scenarios. Model relationships are largely deterministic and issues of uncertainty are addressed via sensitivity analysis. Components of the model have been validated on three sites.

Bayesian networks

Given the inherent complexity and lack of knowledge about many of the basic processes and relationships between contaminants and biota, other types of models may offer more promise for progress. Perhaps the most promising of the alternative modelling approaches available is the technique of Bayesian networks (Reckhow, 2003; Clark, 2005).

Recent years have seen major advances in the availability of empirical and Bayesian methods to quantify aspects of ecological risks (Stow, 1999; Stow & Borsuk 2003; Stow et al., 1997, 2001; Borsuk et al., 2001, 2002, 2004; Pollino et al., 2004a; Bromley et al., 2004; Wooldridge & Done, 2004; Robertson & Wang, 2004; Burgman, 2005; Prato, 2005). General introductions to Bayesian methods may be found in Ellison (1996), Cain (2001), Wooldridge (2003) and Korb & Nicholson (2004).

Bayesian networks can assist with multiple hazard (stressor) problems, in that they are able to incorporate information with high uncertainty, including poor or incomplete understanding of the system, and can combine both empirical data and expert opinion. Most importantly, prior beliefs can be updated as more information becomes available.

Bayesian networks are graphical models that are able to maintain clarity by making cause-effect assumptions explicit. This modeling framework supports analysis of probabilistic relationships and facilitates probability calculations. This information can then be incorporated into a framework that permits decision-making, inference and 'learning' with the availability of additional information.

Bayesian networks have been successfully used for a wide range of risk predictions, including relating catchment nutrient management actions to algal bloom formation in the Neuse River estuary, North Carolina (Borsuk et al., 2003), assessing native fish abundance in the Goulburn River (Pollino et al., 2004a,b), survival of an estuarine clam in anoxic bottom waters (Borsuk et al., 2002), assessing New Zealand abalone stocks (Breen et al., 2003) and to account for the combined variability and uncertainty of model parameters in a crayfish bioaccumulation model (Lin et al., 2004).

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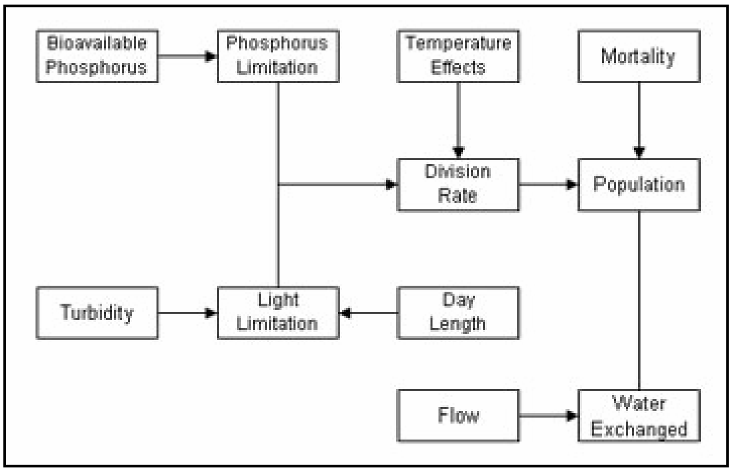
5. Web site: www.epa.gov/waterscience/models/aquatox/

Box 8: Risk analysis: Algal case study

The stakeholders in the Goulburn Broken catchment identified the occurrence of cyanobacterial (blue-green) algal blooms as a priority issue (Cottingham et al., 2001). Webb & Chan (2004) investigated the risks at two case-study sites that had also been chosen by stakeholders: Lake Nagambie and the lower Goulburn River, both located in Victoria. Neither site had a documented history of algal blooms, which ruled out the possibility of using empirical modelling approaches to assess the risks.

A model developed for algal blooms in Bourke Weir, NSW (Webb et al, submitted) was applied to the two case-study sites. The numerical model was based upon the conceptual model shown.



In the model, light, nutrients, temperature and flow regulate the cyanobacterial population. The links between model components were formed by equations that were either based on existing research on algal population dynamics, or on system-specific knowledge generated during the risk assessment (Webb & Chan 2004).

The model did not 'predict' any blooms when driven by data from the two case-study systems. This was in line with the observation of generally low cyanobacterial abundances at the case-study sites.

A series of scenarios were then tested (see Table below), in which one or more of the drivers were adjusted to make conditions more conducive to algal blooms. In these scenario tests, a substantial increase in phosphorus concentrations at the Lake Nagambie site was the only perturbation that caused the model to forecast substantially increased cyanobacterial population densities.

Lake Nagambie/Goulburn Weir

Parameter	Bloom Likelihood	Uncertainty
Increased FRP (x 4)	Medium	High
Increased temperature (+ 4°C)	Low	High
Decreased turbidity (0 NTU)	Low	Low
Decreased flow (halved)	Low	High

Lower Goulburn River - Rodney Main Drain to the Murray

Parameter	Bloom Likelihood	Uncertainty
Increased FRP (x 5)	Low	High
Increased temperature (+4°C)	Low	High
Decreased turbidity (halved)	Low	High
Increased flow (x 4)	Low	Low
Decreased flow (not modelled)	Low	High
Increased salinity (not modelled)	Low	High

High uncertainty was associated with most of these forecasts, due to the fact that it was not possible to validate the model due to the lack of blooms at the case-study sites.

This risk assessment represents an example where a quantitative model was driven by monitoring data to investigate risks. However, the risk estimates generated are still qualitative due to the high degree of non-quantifiable uncertainty associated with the use of an unvalidated model. This approach, however, still represents an advance over the subjectively derived qualitative risk assessment, due to the increased objectivity of the risk estimates.

Interestingly, the work by Borsuk et al. (2004) illustrated well the need to consider models at a range of scales in order to account for the predictable patterns that can emerge at a variety of scales. They used a Bayesian network to integrate a number of models of the various processes involved in eutrophication of an estuary. They integrated process-based models statistically fitted to long-term monitoring data, Bayesian hierarchical modelling of cross-system data gathered from the literature, multivariate regression modelling of mesocosm experiments and judgements elicited from scientific experts. The Bayesian network approach allowed the outputs from these disparate models to be integrated into a cohesive overall model that related nitrogen inputs to the river to endpoints that were identified as important by stakeholders.

Box 9 summarises a Bayesian network approach to predict native fish abundance in the Goulburn River subject to a number of threats (e.g. barriers, flow changes, lower water temperature, competition with introduced fish species).

3.4.3 Risk investigations

Very often when the risk assessment technical team bring together all the relevant information and data on both components of risk, it is necessary to undertake further studies either to synthesise existing data into a more useful form (e.g. a predictive model), or to undertake further field or laboratory studies to generate more site-specific data.

This section provides guidance on how to identify knowledge gaps and then how best to determine what types of investigations are needed to fill these gaps. This guidance has drawn on our experiences in defining and undertaking the further investigations as part of the three case studies. We have found it useful to separate the process of identifying the ecological issues from that of identifying knowledge gaps. The former is best done with stakeholder involvement and the latter by the technical team. The best outcomes will result if the technical team is multidisciplinary and able to adequately assess all aspects of each issue. This minimises the potential for scientific team ‘bias’ reflecting particular interests. Additionally, the process of identifying knowledge gaps needs to be transparent, with all data and information relevant to each issue clearly identified, collated, analysed and documented. It is also desirable that the final recommendations on the main knowledge gaps, and what needs to be done to address these gaps, are peer reviewed. In many cases, enough information is already available to undertake at least a preliminary risk assessment, perhaps with the addition of some site-specific information. Most often, however, it is clear that the risk assessments produced would be improved by the addition of further information from dedicated studies. Both these situations are covered below.

Synthesis of existing data

All relevant existing data and information should be used before additional studies are contemplated. However, tracking sources of data, getting access and then synthesising these data into a form that can be used in the risk assessment is a very time consuming exercise, and the time to achieve these tasks is commonly significantly underestimated. Usually experts are used as a way of assessing broad-based, dispersed and complex information. Expert judgements may themselves be biased by context and personal interests, but are usually better than uninformed opinion. Experts can also participate in workshops to provide information in addition to any background material provided by the ERA technical team.

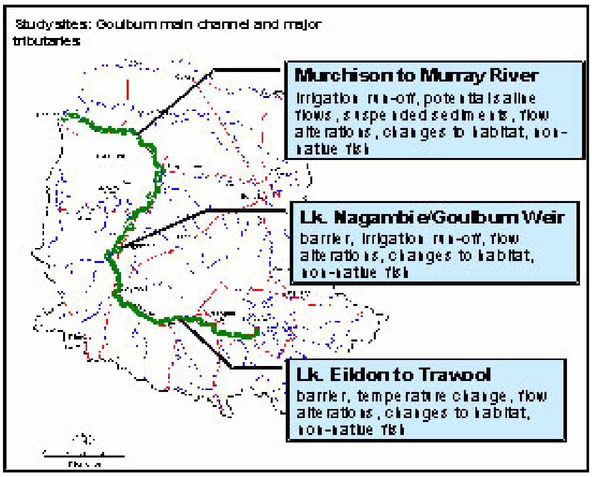
Field investigations

Often even after all available data have been captured and synthesised, it is still necessary to undertake further field or laboratory studies to generate more site-specific data for the risk assessment. Examples of two field investigations designed to provide specific information to assess ecological risks in the Fitzroy River and the Ord River are provided in Box 10 and 11 respectively. In designing such additional investigations, it is important to recognise the difference between investigations required for risk assessments and ‘normal’ (hypothesis based) scientific investigations. Most scientists are trained to consider only the latter type of investigation.

Risk investigations are focused on quantifying the relationships between cause and effect, or more often, between indicators of cause and effect, rather than in determining whether statistically significant effects have occurred due to the imposition of a stressor on the ecosystem (Reckhow, 2003; Burgman, 2005). Given the complexity and variability of most natural systems, and the high level of uncertainty, a statistically significant finding (with sufficient power) generally requires the collection of many samples over a long period of time. This often results in prohibitively expensive investigations that are either done poorly or not at all. Studies based on hypothesis testing also rarely concentrate on quantifying the effect size of a certain level of stressor. This information is essential to develop the cause-effect relationships described above.

Box 9: Risk analysis: fish case study

Over the past 100 years, the Goulburn catchment has been highly modified, primarily for irrigation. As a result of the changed environmental conditions, native fish populations have declined. The cause for this decline is attributed to a combination of factors, shown in the conceptual model below.



System Description

The headwaters of the Goulburn River flow into Lake Eildon. Water released from Lake Eildon is delivered 218 km downstream to Goulburn Weir. From Goulburn Weir, outflows are to the lower Goulburn River and three irrigation channels.

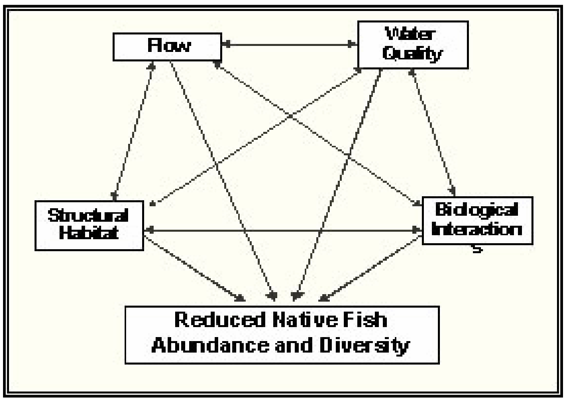
Although the major factors contributing to the decline in fish abundance and diversity have been broadly identified, they have not been fully characterised or measured over time.

Our simple conceptual model (Figure 2) of the system recognises four main factors influencing native fish abundance and diversity in the catchment.

Study Objectives

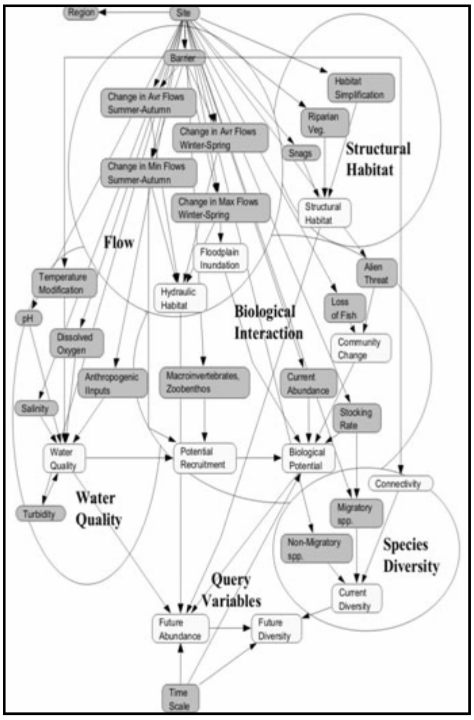
To support future decision-making in the catchment, a model with the following requirements was desired:

- quantify the linkages between multiple stressors and native fish communities,
- incorporate data with high uncertainties,
- integrate information from multiple sources,
- be specific for sites and reaches,
- assist in prioritising management actions,
- predict changes to fish communities post-management interventions



To suit these criteria, a Bayesian network (BN) was constructed. BNs are probabilistic graphical models that can utilise and represent both expert knowledge and empirical data in specifying model structure and relationships.

Causal relationships expressed in the model structure are explicit and the probabilistic relationships between variables are updated based on Bayes' theorem.



Causal Structure

The BN structure (Figure opposite) is based on a comprehensive conceptual model developed in collaboration with fisheries experts.

Probabilistic Relationships

The probabilistic relationships between variables in Figure were quantified (parameterised) based on expert elicitation, information from the literature, and monitoring data. Parameterisation was conducted using an iterative process.

Final Model

The final network can be used to

represent the spatial scale of interest to describe the current state of the system, and

test alternative scenarios, representing system changes or management interventions.

How accurate is the model?

Model validation/verification tests conducted include plotting fisheries data vs. model predictions, sensitivity analyses, predictive accuracy measurements, and expert assessments.

Implications for fisheries management in the Goulburn Catchment

(Box 9 continued on page 32)

(Continued from page 31)

Downstream of Lake Eildon, water quality (temperature) and changed hydrology appear to be the variables primarily influencing native fish abundance. At and below Lake Nagambie, fish abundance is primarily under the influence of the biological potential (potential recruitment and current abundance), water quality (turbidity, dissolved oxygen and pH), and flow.

Final Product

In the field of ecological risk assessment, there is increasing demand for ecological models that consider multiple stressors, incorporate data with high uncertainty, and integrate scientific information gained from multiple sources into a single predictive framework. The BN described is ideal for use in a Decision Support Tool in environmental risk management. The model construction process promotes stakeholder involvement, and thus the final product and modelling approach is more likely to be widely accepted and adopted.

References

3.5 Risk characterisation

3.5.1 General

This phase seeks to integrate and interpret the results of the risk analysis phase and to address the problems formulated in the planning and scoping phase. The output from this final stage should:

- describe the qualitative and/or quantitative analysis results, with particular focus on providing a ranking of the risks assessed;
- provide all the information and knowledge obtained from the risk assessment that will be useful in managing the ecosystemobtained from conducting the risk assessment;
- list the important assumptions, limitations and uncertainties associated with these results; and
- discuss the ultimate use of the outcomes.

It is important that the key stakeholders are involved in this stage. They should understand the outcome of the risk assessment, ask questions about how best to frame the interpretation and confirm that the risk assessment has met the goals set in the problem formulation stage, and if not, why not.

Risk assessments are usually undertaken for one of the following reasons: to support an action (act/don't act) or decision (e.g. allow a proposal/disallow the proposal), to provide information in developing risk management plans, or to identify where resources should best be targeted to minimise high priority risks.

3.5.2 Uncertainties & assumptions

The outputs from all risk analyses are based on a set of assumptions and simplifications, and it is important that the decision-maker know how sensitive the predictions are to the assumptions. When quantitative mathematical models area used, it is common for a sensitivity analysis to be undertaken to evaluate the changes in the predictions resulting from manipulations in the model. This can also be done for qualitative conceptual models (Burgman, 2005).

If nothing important changes as a result of the sensitivity analysis, then the decision may be considered robust to the uncertainties and assumptions in the model, or the judgements of experts.

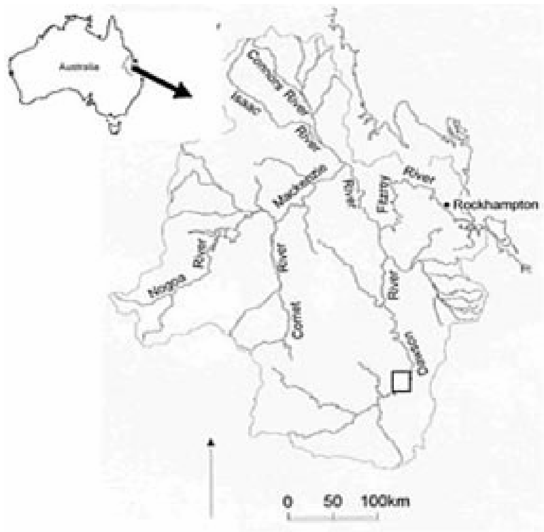
However, it is still not common for these uncertainties to be adequately estimated, even in situations where quantitative modelling has been used (Burgman, 2005).

Risk assessments must deal with three types of uncertainty (Regan et al., 2002):

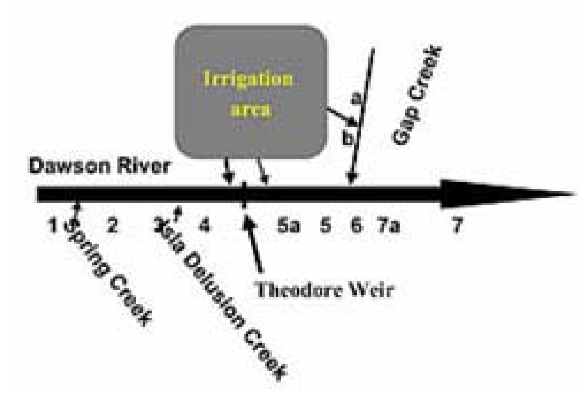
- v*ariability* – arises from natural variation and is common in all parameter (e.g. nutrient concentrations, rainfall) measurements. This type of uncertainty is the most commonly considered, and cannot be reduced by more effort;
- *incertitude* – arises from incomplete knowledge. This can be manifest either as uncertainty in measured parameters due to small sample size, or to an incorrect or partially correct model for the system being studied. This form of uncertainty can be reduced by more empirical effort; and
- *language uncertainties* – arise because of vague, ambiguous or confused definitions (e.g. concepts that permit borderline cases). Vague statements or definitions are common in ecological risk assessments. For example, it is common to read statements that a certain hazard will pose a 'low risk'. Such vague statements can be strengthened if the definitions of low, moderate and high are quantified.

Box 10: Fitzroy River case study

Here we provide a brief summary of an ERA undertaken for the Dawson Valley Irrigation Area (DVIA), located within the Fitzroy catchment in Central Queensland (Figure 1). This irrigation region does not have the tailwater recycling dams, though irrigators spend considerable effort recycling water from the drains that enter the Dawson or its tributaries. The study focused on Gap Creek, a major drain entering the Dawson River (Figure 2). Further details are available in Duivenvoorden et al. (2004).



Location of Dawson Valley Irrigation Area in the Fitzroy catchment in north-eastern Australia.

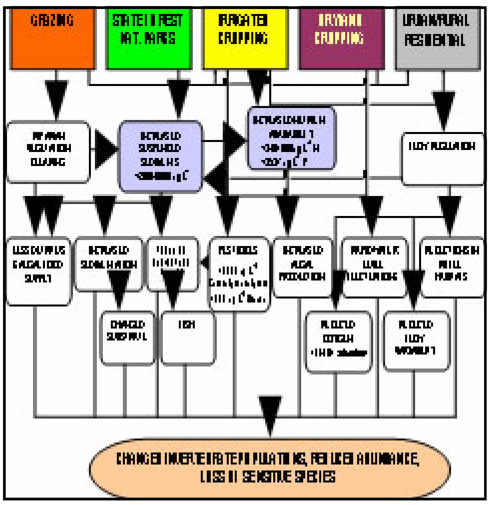


Dawson River Irrigation area in relation to study sites. The drain between sites a and b enters Gap Creek 3.5 km above its junction with the Dawson River. Theodore weir is located approximately 11 km upstream of the Gap Creek entry to the Dawson.

Stakeholder (irrigators, community members and relevant technical experts) identified the main ecological effects of irrigation in the study region as: (a) decline in water quality, (b) soil degradation, increase in salinity, (c) changes in composition and decrease in abundance of aquatic macroinvertebrates, (d) changes in nutrient cycles and (e) decreases in desirable fish populations. The highest priority effects were (a) decline in water quality, since there is wide acknowledgement that nutrient and pesticide concentrations often exceeded water quality guidelines, and the potential effects of these contaminants on the aquatic communities (macroinvertebrates and fish). Conceptual models of the effect of irrigation and other factors on macroinvertebrate and fish populations were developed (Figure 3 overleaf).

At the time the risk analysts determined that there was insufficient information on the system to assess this risk. For this reason a field study was undertaken during 2002/2003, the objective being:

- what is the magnitude of the effect that the water from the drain has on the macroinvertebrate assemblage in the river;
- what is the relative significance of pesticides compared to other environmental parameters (such as reduced oxygen levels and river discharge) on changes to macroinvertebrate communities;
- how does the relationship between pesticides and effect on the macroinvertebrate assemblage change with distance down the river; and
- what is the rate of recovery of the macroinvertebrate assemblage from the effects of disturbance?



Cause-effect conceptual model linking key threats and hazards with effect on macroinvertebrate populations

Studies were made of a site upstream and another downstream of where Gap Creek entered the river, as well as at control sites upstream of the irrigation area and at other sites downstream of Gap Creek to assess the magnitude and extent of the potential impact. The main experimental period extended over 9 months, including the main irrigation season over January-February, when pesticide applications peak.

(Box 10 continued on page 34)

(Continued from page 33)

A range of measures were undertaken, including physical-chemical parameters (including a focus on particular pesticides), macroinvertebrate parameters including taxa richness, abundance and community composition and direct toxicity tests using caged snails. Pesticides were monitored using passive sampler devices that integrate pesticide concentrations at the sites over each three week sampling period.
Risk analysis involved the following:

- *calculation of risk quotients* for various pesticides using the pesticide concentrations in the water estimated from the passive samplers as a function of the trigger values in the ANZECC (2000) guidelines. This analysis suggested that the risk of impact was likely to be low for the Dawson River over the 40 km length below Theodore weir, but high from December to February at one site on Gap Creek immediately downstream of where the drain from the irrigation area entered the creek. This prediction was then checked against the biological data;
- *regression analyses* of the data failed to detect a gradient of impact along the river. ANOSIM pair-wise tests found many significant differences between sites (including control sites), but because of this high background variability between sites along the river, it was difficult to detect an effect of water from Gap creek on the macroinvertebrates in the river. Cluster analyses and MDS generally support these results. Hence, the magnitude of the effect of runoff from the irrigation area into Gap Creek on macroinvertebrates in the river was essentially too small to be detected.;
- *effects (absence of sensitive taxa)* were observed at the site in Gap Creek immediately downstream of the drain. These results provided confidence in the assessment that risk to macroinvertebrate populations was likely to be low in the section of the Dawson River within 40km below Theodore weir and that effects, where present, were highly localised. (Results of direct toxicity tests were inconclusive due to high mortality of snails at control sites);
- *correlation and multiple linear regression analyses* were initially used to determine the relative extent to which factors such as flows and pesticides affect variation in macroinvertebrate populations, but the predictive power of the environmental variables was poor; and
- *comparison of the biological differences* between sites (based on MSD) with environmental data (from PCA analysis) was undertaken using the BIOENV procedure in PRIMER. The analyses suggested that the most important factors explaining differences in the macroinvertebrate structure were discharge and levels of the pesticide endosulfan sulphate.

Conclusions
The risk assessment showed that the magnitude of the effect of water from the drain on macroinvertebrates in the river was essentially too small to be detected, based on the family level of identification end points used. However, effects (loss of sensitive taxa) were recorded at a site on Gap Creek, where risks from endosulfan exposure were determined to be high from December to February. The most important factors explaining differences in the macroinvertebrate structure appear to be discharge and levels of the pesticide endosulfan sulphate. Discharge variables identified as being of significance were discharge divided by time since that discharge, maximum discharge and discharge >200 ML/day.
The risks to macroinvertebrate communities from runoff from irrigated land were highly localised. Rate of recovery from effects was difficult to assess given the single impacted site where measurable effects were recorded. It is postulated that measured pesticide contamination of stream sediments may hinder such recovery and this may be a useful avenue of future research.

For *qualitative risk assessments*, it is not possible to assess uncertainty in any quantitative way. However, it is possible to clearly document all assumptions so that the decision-maker can make an assessment of the importance or otherwise of the outputs. Such declarations of assumptions are also required for quantitative risk assessments, where model output will be partly determined by the assumptions made.

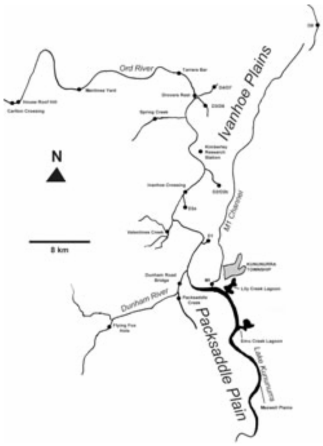
As noted above, with *quantitative risk assessments*, it is often possible to undertake a sensitivity analysis, where the model parameters or assumptions are changed in a logical way and the resultant change in the prediction is observed. In addition to sensitivity information, the better predictive models also provide an estimate of the uncertainty in the predicted variable in the form of a confidence interval or distribution of results generated by multiple trials (i.e. Monte Carlo analysis).

After seeing the results of the first iteration, the decision-maker may wish to consider some different scenarios, which could involve running the predictive models with a different set of assumptions or perhaps even constructing a different model. The important point is that this process is iterative, allowing a range of options or scenarios to be tested. The decision-maker is then in a better position to make the best choice regarding the project or management action to be implemented.

Box 11: Risk of algal blooms in the lower Ord River

The Ord Irrigation Area releases over 400 GL per year of drainage and unused irrigation water back into Ord River, contributing large loads of nitrate/nitrite and filterable reactive phosphorus. In a preliminary Ecological Risk Assessment, stakeholders identified algal blooms as a low priority risk. Despite this there was a high interest in the possibility of algal blooms in the Ord River and how Ord Stage 2 developments may impact on these. It was therefore decided to investigate the risks further. It was beyond the scope of this study to conduct any fieldwork and so the focus was on evaluating existing published and unpublished data on the river.

The purpose of the study was to assess the risk of algal blooms in the lower Ord River from irrigation return flows. The key environmental value to be protected was the health of the lower Ord River.



Algal blooms were divided into three levels of severity:

- level 1 - >500 cells mL-1 of cyanobacteria, management interest;
- level 2 - >5000 cells mL-1 of algae, management action required; and
- level 3 - >15000 cells mL-1 of algae, visible algal bloom

The lower Ord River was divided into four areas where algal blooms were likely:

- the lower Ord River, downstream of the Kununurra Diversion Dam to the start of the estuary;
- the pool at the Dunham River confluence with the Ord River;
- Lake Kununurra, formed by the Diversion Dam; and
- Lily Creek Lagoon a backwater of Lake Kununurra.

Four parameters identified by Reynolds and Descy (1996) as regulating algal growth in rivers were investigated in detail:

- Bormans et al. (2004) suggest that water temperatures >25oC were responsible for the dominance of typical tropical cyanobacteria. The onset of stratification was seen as a major trigger for algal blooms;
- turbidities of <15 NTU are unlikely to hinder algal blooms;
- DIN:FRP weight to weight ratios indicated that N was probably limiting algae in the river. Irrigation drainage return was shown to be a significant source of nutrients into the river; and
- Parslow et al. (2003) found that hydraulic residence times in the lower Ord River were higher than previously thought at >15 days during normal dry season flows.

The limited algal data was examined and high temperatures >30oC, low turbidites <15 NTU, and long residence times were associated with the two Level 1 algal blooms recorded at Lily Creek Lagoon and the Dunham River Pool. Nutrient concentrations did not appear to be a significant contributor to the creation of the bloom. It is suspected that nutrients are more likely to influence the severity of the bloom.

Likelihood of an algal bloom

The four factors were then assessed over the months of the year to determine the months where algal blooms are most likely. The likelihood of each Level of algal bloom was estimated for each area.

Consequences of algal blooms

Each algal bloom level was assigned a severity value from low through to high, these values were then modified by locality factors. For example, in Lily Creek Lagoon, its proximity to Kununurra Township, increases the consequences of algal blooms, by including a range of social values.

The risks

When likelihood is multiplied by the consequence this determines the risk, which is assigned to categories (L =6, 7<

Site	High risk times	Severity	Likelihood	Consequence	Risk
Durham River	Aug - Oct	Level 1 Level 2 Level 3	6 4 2	1 4 7	6 (L) 16 (H) 14 (H)
Lake Kununurra	Jan – Feb	Level 1 Level 2 Level 3	2 1 1	2 5 9	4 (L) 5 (L) 9 (M)
Lily Creek Lagoon	Jan - Feb	Level 1 Level 2 Level 3	5 2 1	2 5 8	10 (M) 10 (M) 8 (M)
Lower Ord River	Jul - Oct	Level 1 Level 2 Level 3	2 1 1	1 3 4	2 (L) 3 (L) 4 (L)

(Box 11 continued on page 36)

(Continued from page 35)
M =12, H =13). It can be seen that in the Dunham River Pools, Level 1 blooms are very likely but of little consequence and so have a low overall risk. However as the blooms get more severe, the consequences increase substantially, but the likelihood reduces, resulting in high overall risks. There is relatively low risk of algal blooms in the lower Ord River and Lake Kununurra. Lily Creek Lagoon however has medium risks for all Levels and so along with the Dunham River pool should remain a focus for management attention.

The results of this assessment shows that the initial assessment of the low risk in the lower Ord River was correct, however it did identify areas where the risk was much higher than had previously been believed.

Ord Stage 2 developments, potentially will result in a 4 fold increase in irrigated lands and a 1000 GL per annum increase in demand for water, reducing flows in the lower Ord River. Lower flows would substantially increase the risk of algal blooms by increasing the chances of stratification, residence times, and increasing nutrient concentrations. This study did not speculate on the likely impacts of Ord Stage 2 as more research is required into algal growth in the

A number of researchers have argued that the specific analysis of uncertainty can produce a better understanding of risk, promote transparency, enhance credibility and generally improve decision-making (Pate-Cornell, 1996; Burgman, 2001). Characterisation of uncertainty helps answer the question: 'How likely is a predicted outcome?' and needs to be an integral part of any ecological risk assessment.

A range of mathematical techniques is now available to assist with uncertainty calculations, including worst-case analysis, interval arithmetic (Burgman, 2005), probability theory (Monte Carlo analysis) and Bayesian analysis (Nayak & Kunda, 2001; Reckhow, 2003). There are also a number of specific computer packages available for this purpose (e.g. RAMAS RiskCalc (web site: www.ramas.com), NETICA (web site: www.norsys.com), WinBUGS (web site: www.mrc-bsu.cam.uk/bugs)).

It should be noted that in the main these methods aim to quantify the uncertainty associated with variability. Situations where there is a lack of knowledge or ignorance about how an ecosystem functions, cannot be assisted using these methods (with the possible exception of interval analysis and related techniques implemented in RiskCalc). Such uncertainty must be documented when reporting the results of the risk assessment.

A summary of the assumptions, uncertainties and any limitations of the analyses should accompany all risk assessments.

3.5.3 Risk communication

No risk assessment is complete until the results are communicated in a clear and honest way. The communication of risk should be an interactive process involving the exchange of information and opinions between individuals and groups (and in some cases institutions) (Burgman, 2005).

There are no definitive guides on the development of risk communication strategies (Bier, 2001). A range of factors, including age, gender, socio-economic status, language, religion, cultural background, knowledge base and attitude towards risk, can all influence the ability of an individual to interpret the message communicated. The best approach to risk communication can also be influenced by the purpose of the communication (e.g. raise awareness, educate people, motivate people to take action, reach agreement on a controversial issue or obtain people's trust) (Burgman, 2005).

A particular challenge is to communicate the uncertainty and incomplete knowledge associated with the risk assessment in a way that is helpful and does not undermine the usefulness of the results. Burgman (2005) argues that when people are presented with a point estimate for a risk (i.e. a single number with no confidence interval), they are forced to be 'risk neutral', and are denied the ability to make better decisions that come with knowing the bounds on the judgements. Questions such as 'which risk is larger?' should be replaced with 'how confident can we be as to which risk is larger, and by how much, and with what consequences if I am wrong?'

4. ECOLOGICAL RISK MANAGEMENT IN THE IRRIGATION INDUSTRY

4.1 General

Risk management uses the results and insights from risk assessment to determine what actions should be taken to minimise the high priority risks, and therefore result in more sustainable management of the environment. It involves forecasting, setting priorities, formal decision-making, and reconciling different viewpoints (Beer & Ziolkowski, 1995). Typically, these risk management actions are judged against environmental, social, economic and political factors.

An important reason for undertaking ecological risk assessments is that by following a logical, transparent and credible process, the conclusions reached from the ecological information should be more robust, both technically and socially.

There are important differences between the assessment of risks and the management of those risks. *Risk assessment is focused on the ecological values* of the system being investigated, in particular on prioritising, as quantitatively as possible, those values that are most at risk from a range of hazards. On the other hand, *risk management is focused on the hazards*, in particular in removing or minimising the impacts of the most important of these hazards for the ecological values assessed, and done in ways with which the key stakeholders agree.

The development of a risk management plan inevitably involves deciding which scheme is 'best' (or at least 'acceptable') amongst a number of possible options to address (minimise or remove) the risks. However, because the outcomes of the management actions are uncertain, the choices between options often are often not self-evident. Burgman (2005) covers some of the grounds for making choices, and has also reviewed a number of the techniques available to assist in decision-making.

Methods available for integrating the ecological risk information with the social and economic requirements include multi-criteria decision analysis (MCDA) and management strategy evaluation (MSE). Both are discussed in detail in Section 4.2.2.

This section covers the development and implementation of risk management plans. The process is summarised in the flow chart in Figure 6.

4.2 Developing a risk management plan

As noted above, when managing natural resources, it is essential that information from the risk assessment is combined with economic, social, political and cultural information to develop a full risk management plan. The objective of the plan is to address the risk to the particular ecosystem with the most effective use of available resources, and in a manner that is consistent (as far as possible) with the wishes of stakeholders.

In effectively managing catchments, the resource manager is often faced with a large number of environmental assets that the community wants protected, managed or rehabilitated, an equally large number of threats and hazards to these assets, and a limited budget to fund management actions. How then to decide on the most cost-effective actions?

In developing a risk management plan, four key questions need to be addressed to satisfy the 'triple bottom line'[6] requirement being demanded these days. These are:

- what ecological/environmental assets are most at risk;
- what hazards most threaten these assets, and are there management actions available to effectively minimise the risks;
- are these actions acceptable to the stakeholders/community and
- are these the most cost-effective actions?

The ecological risk assessment process, in providing (quantitative) information on the ecological values most at risk and the hazards that most threaten these values, effectively addresses the first two of these questions. The other two questions must be answered through stakeholders/community interactions and economic analysis.

6: Where environmental, economic and social factors are considered in the final decision

Thus, the development of a risk management plan involves four steps shown in Figure 6:

1. identification of potential actions to reduce the effects of the key hazards associated with each high-risk ecological issue;
2. a decision-making process that involves:
   - identification of possible options for implementing the above actions; and
   - selection of the 'best' option, based on ecological, social and economic criteria. Note - the best option must result in the effective minimisation of the ecological risks, while also being cost-effective and acceptable to the stakeholders or community.
3. preparation of the risk management action plan; and
4. implementation of the plan.

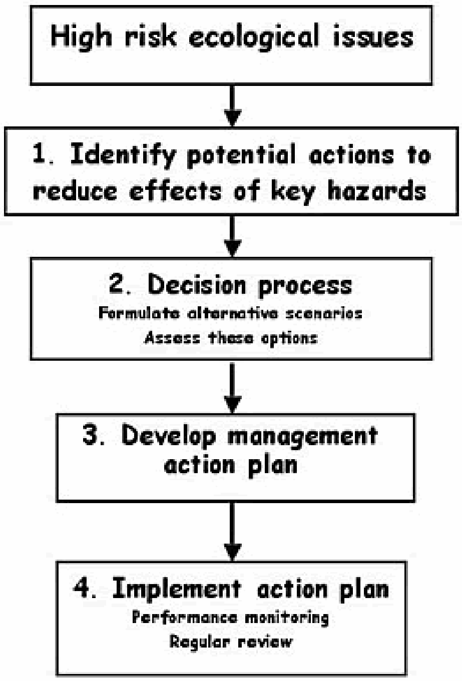


Figure 6: Steps involved in developing a risk management plan

4.2.1 Potential actions to address the priority hazards

The priority hazards associated with each of the high-risk ecological values will have been identified during the risk assessment process (issue-specific conceptual models and the hazard-effect matrix - Table 3). In those cases where a quantitative model is available, it will also often be possible to test the sensitivity of the assessment endpoint to changes in each of the hazards, and in this way help to prioritise which hazards should be targeted for action.

Once the hazards have been prioritised, possible actions to address each of these should be identified. These actions might include: policy changes, regulation, capital works, improved management, and education.

A possible hierarchy of actions could be:

Extreme to high risk situations

- elimination of the hazard should be attempted wherever feasible;
- often it is possible to replace the hazard by one of lower risk as has occurred with a number of toxic chemicals; and
- engineering solutions have also been used effectively, for example, to provide fish passage over dams and weirs, and to provide multi-level off-takes in reservoirs to avoid the release of cold water to downstream river and wetlands.

Moderate to low risk situations

- administrative and policy changes can be used to reduce risks by altering planning controls and requiring improved management; and
- education is also a vital element for achieving better management.

4.2.2 Decision-making process

Decision-making and risk management are obviously very closely linked, although in this framework they have been separated (Figures 1 & 5). The reason for separating them is to highlight the fact that before a risk management plan can be developed, a number of social and economic factors must be considered together with the ecological assessment, to set the priorities for action. Rarely are there sufficient resources available to do everything needed, and some tough trade-offs will need to be made.

The key stakeholders need to be involved in this step, as they will play an important part of any trade-off process.

Identification of plausible options

The development of plausible options for addressing the high priority hazards is not a trivial exercise. Obviously, the range of possible actions identified above will be important input, as will the effectiveness of these actions, their cost and their acceptability by the stakeholders.

One of the most difficult tasks in the management of ecological risk is to adequately predict the magnitude of the risks in the future, i.e. in 20-30 years time. Decision-makers must try to imagine what the (uncertain) future will be like and use this to decide on the best course of action. However, uncertainty will remain, and the further into the future we need to project, the more uncertain the predictions will be.

A logical approach to coping with uncertainty is to try to make decisions that are robust under a range of possible futures. However, most strategic planning processes focus on a single, most plausible future. Van der Heijden (1996) introduced the use of scenarios (or sets of stories about the future) as an alternative method. Scenario planning involves thinking about a wide range of plausible futures that bound the range of possible alternatives, factoring in well-known trends and uncertainties, and using this information to establish several stories to guide the decision-making process.

We believe concepts from *scenario planning* can assist in improving risk management in an uncertain future (Hart, 2004a).

In the framework of quantitative risk assessment, quantitative risk models (hazard-effect models) provide the template upon which alternative, plausible scenarios can be developed and trialled. The predictions that are produced by these models can be compared to real data and can be evaluated with critical thinking. They form a kind of qualitative sensitivity analysis in which alternative assumptions about a model are explored. Unimportant assumptions may be discarded. Important ones may be identified and targeted for further data collection and evaluation.

A particular advantage of scenario planning is that it trains one to expect the unexpected and to be on the lookout for the resulting opportunities (Van der Heijden, 1996; Bennett et al., 2003; Peterson et al., 2003).

We see great potential in using scenario planning, in conjunction with risk-based approaches, to scope some innovative and environmentally sustainable management approaches for irrigation enterprises (Hart, 2004a). This will involve the development of a number of scenarios, each of which could be assessed for the environmental risks. The benefit would be that the proposed irrigation scheme could be assessed against a number of possible futures, which would allow decisions to be made on the scheme that stood up under a number of these possibilities.

Assessing the options

After the range of plausible options has been developed, the 'best' of these alternatives needs to be selected. The best option will be the one that most effectively addresses the ecological risks, while also being cost-effective and acceptable to the stakeholders or community.

There are now a number of techniques available to assist with this process. Three are discussed here. For further information the reader should consult Burgman (2005).

Multi-criteria decision analysis (MCDA) is increasingly being used with risk assessments to assist the final decision-making step (Brown & Joubert, 2003; Khadam & Kaluarachchi, 2003; Kangas & Kangas, 2004). Multi-criteria analysis is a particularly useful aid for narrowing down large numbers of options in cases where data may be incomplete and/or where some subjectivity is involved in deciding on the relative importance of the evaluation criteria. Generally, multi-criteria analysis involves four tasks:

1. identifying a range of options consistent with the objectives of the assessment (i.e. to minimise risk to ecological values);
2. identifying a set of evaluation criteria that highlight key environmental and community impacts (these criteria may be both qualitative or quantitative);
3. weighting the evaluation criteria in terms of their relative importance; and
4. assessing the performance of each option against the weighted evaluation criteria so that the differences between options can be better appreciated.

Management strategy evaluation (MSE) is another method that has been used to identify and assess competing management options, particularly in the fishing industry (Smith, 1993, 1994; Mapstone et al., 2004). Importantly, the method presents the results in a way that reveals clearly the tradeoffs in performance across a range of management objectives. The main advantages of MSE are that the method (Smith, 1993, 1994; Mapstone et al., 2004):

- is focused on evaluating the medium to long-term performance of management strategies;
- is comparative rather than prescriptive, seeking to compare likely outcomes of a range of management scenarios rather than prescribed action required under a regulatory framework;
- seeks to compare the future status of a range of performance indicators with their desired values (objectives) under a range of management strategies;
- seeks to compare the performance of a range of candidate management strategies in terms of multiple performance indicators that reflect the diversity of stakeholder objectives, including social, economic and ecological; and
- seeks to provide a system for comparing the performance of alternative management strategies against different stakeholder's objectives based on a common currency across all or most objectives.

A relevant and very useful application of MSE is provided by Mapstone et al. (2004) who used the method to assess a number of management options aimed at reducing the adverse effects of reef line fishing on the productivity of target species and its impact on other reef fish species on the Great Barrier Reef.

Bayesian belief networks approach this issue somewhat differently by explicitly including the stakeholder ideas in a model and then use the model to test various options (Borsuk et al., 2001). For example, site-specific and sub-basin Bayesian belief network models were used by Marcot et al. (2001) to evaluate population viability outcomes of wildlife species for the Columbia Basin. In another example, Ames et al. (2004) reported a case study to determine the most beneficial actions to manage coldwater biota within a watershed. They ran a simulated Bayesian analysis of the costs and benefits likely to be realised by different stakeholders (ranchers, factory owners and anglers) under different regimes for the management of impacts of increased biological oxygen demand within the watershed. The modelling suggested that the optimum management scenario for risk reduction would not necessarily be selected if the objectives were expanded to also consider the acceptability to stakeholders of associated implementation costs (monetary and non-monetary). Bacon et al. (2002a, b) used Bayesian belief networks to model stakeholder preferences for land use change (e.g. intensive farming, conservation, organic farming etc) given the local land types (e.g. good soil, poor drainage). The network was used to estimate whether the costs of changing from present land use to a potentially better one were outweighed by the anticipated benefits. The study showed that stakeholders used different criteria to make their decisions, and that introduction of economic incentives affected the probability that some farmers would adopt more 'environmentally friendly' land uses.

We believe that the use of quantitative approaches, particularly emerging ones based on Bayesian methods, that explicitly include stakeholder inputs into risk-based decision-making would significantly strengthen the process.

4.2.3 Risk management plan

Ecosystem values at risk generally require one (or more) of the following broad management actions:

- *protection* - for ecosystems that are assessed as being at low risk of adverse effects in the medium to long-term. Previous research has estimated that it is at least 10 times cheaper to protect and maintain existing ecosystems than to restore and repair lost ecosystem function (Balmford, 2002);
- *active management* - for ecosystems that are assessed at high risk now or will be in the near future;

- *rehabilitatation* - systems that are currently in poor to reasonable condition, where the hazards causing this poor condition are clearly identified, and actions have been identified to minimise these hazards. The aim is to restore condition in the system; and
- *low priority for any action* - systems that are currently in poor condition and at high risk of further adverse effects. Exceptions may be systems that are highly valued by the community for which the costs of effective risk management are clearly communicated and accepted.

Table 5 illustrates these possible management actions as a function of the assessed risk of adverse effects and the present condition of the ecosystem. Thus, for a system currently in excellent condition and where the risk is low, the appropriate management action is to protect this asset. On the other hand, for a system currently in reasonable condition and where the risk of further adverse effects is high, the appropriate management action would be to actively manage this asset, probably by putting resources into addressing the hazards and rehabilitating the system.

Table 5: Possible management actions based on both the assessed risk to an ecological asset and the condition of that asset

Risk of adverse effect	**Present ecological condition**		
	Good	***Reasonable***	***Low***
Low	P	P+R	R
Moderate	AM	AM+R	AM+R
High	AM	AM+R	R*

We do not believe it advisable to be too prescriptive about the possible management actions. These need to be decided after stakeholder consideration of the case-specific trade-offs of values, risks and the costs of the effective risk management.

Two other situations should also be considered:

- for new irrigation schemes, it is possible that the cost to make the identified ecological risks acceptable through management actions are prohibitive. In this case, the outcome of the risk assessment may be a recommendation that the scheme not proceed; and
- for existing irrigation schemes, it is also possible that it could be prohibitively costly to make the ecological risks acceptable through management actions. In this case, the outcome of the risk assessment may be a recommendation that that the on-site ecological assets are compromised, but that investment is made in protecting or restoring comparable off-site ecological assets.

4.3 Implementing the risk management plan

Implementation of the risk management plan involves a range of actions, generally focused on removing or reducing the influence of key hazards. These actions will vary depending upon the particular risk, and may include enhanced regulation/compliance requirements, capital investment to improve performance, on-ground actions, and education.

In this section we focus on two important aspects of implementing the risk management plan that are often either poorly done or not done at all - performance monitoring and review. The reason for focusing on these two aspects is to highlight the importance of reviewing the performance of the management plan, and modifying the actions if the plan is not as effective as expected. This is the basis of adaptive management (Walters, 1986).

For many ecosystems, there is a poor level of knowledge about the ecological functioning of the system and the way human activities may impact upon the system. This is reflected in a high level of uncertainty in the ecological risk predictions. However, despite this lack of knowledge, decisions still need to be made, and risk management action plans are developed on the basis of the best information available at the time.

In view of the inherent uncertainty in almost all risk management plans, it should be mandatory that a robust monitoring program be initiated, and that the information from this program is regularly reviewed and fed back into the management process. The section below contains guidance on establishing effective monitoring and review programs.

4.3.1 Monitoring program

Although rarely stated, natural resource management plan monitoring programs have multiple objectives. The main objectives of such a monitoring program are to:

- evaluate the state and trends of key environmental assets, i.e. is the system improving, getting worse or being maintained;
- provide information on the effectiveness of the management strategies in place; and
- improve the knowledge-base and assist in refining the risk management plan (particularly by evaluating assumptions and uncertainties in the models used to make the decision and modifying the predictive models as knowledge improves).

Unfortunately, all too often this essential part of the management of natural resources is done poorly or omitted altogether. Lake (2001) recently commented on the widespread the poor design of restoration projects in Australia. He identified the following factors that were frequently missing:

- knowledge of the conditions prior to the management intervention ('before' data);
- the need for replication (if possible);
- the provision of control or reference sites;
- the establishment of feasible goals and targets;
- flaws in the design of sampling programs that result in little capacity to test even basic hypotheses; and
- failure to consider the spatial and temporal scales of the restoration project.

Box 12 outlines the major steps in development of a robust monitoring program, which will then enable interrogation of the collected data to assess the effectiveness of the management action.

Key features of a sensible monitoring program include:

- identifying clear objectives for the monitoring program before it starts;
- using quantitative methods to monitor the key assets or values;
- deciding up front what methods will be used to analyse the monitoring data; and
- determine the rules for action if and when the monitoring results identify significant adverse effects (or trends).

Further useful information on the design and implementation of water quality monitoring and reporting programs may be found in ANZECC/ARMCANZ (2000b).

4.3.2 Review and feedback processes

Monitoring programs that clearly differentiate circumstances in which management strategies are working or failing over reasonable time frames are commonly cost-prohibitive (Mapstone 1995). An important issue for decision-makers and their broader stakeholders is the burden of proof associated with shifting investment from one strategy (where there is vague evidence of failure) to an alternative strategy (that might deliver better outcomes).

Currently, there are few effective methods for capturing and using new information from monitoring, research and expert opinion to update management plans. We believe the application of Bayesian techniques to this task is worthy of attention (Bacon et al., 2002a,b; Clark, 2005). The advantage of Bayesian models is that they can be explicitly updated and improved as more information becomes available. Additionally, Bayesian models are very useful in identifying the particular areas where better information can have the greatest effect in improving the predictive model.

Box 12: Key steps in developing a robust monitoring program

The key steps are:

- agreement on purpose, questions and hypotheses after consideration of resources & timeframe (this is derived directly from the ERA problem formulation phase);
- consideration of spatial and temporal scales;
- selection of sites, indicators and analytical methods (including QA/QC protocols to be adopted);
- determining appropriate methods to analyse the data;
- ongoing review and feedback; and
- reporting.

The first step in the monitoring program should be straightforward, as these matters are addressed in the problem formulation phase of the ERA. However, it must be stressed that inadequate attention to this step can jeopardize or invalidate all that follows.

The spatial and temporal parameters of the monitoring may be the same as for the complete risk assessment, or more commonly may be at a smaller scale, given finite resources. For example, a subset of all possible monitoring sites in a catchment may be selected so that adequate temporal sampling can be performed within the limits of the project budget.

Table: Common physico-chemical and biological indicators in aquatic ecosystems

Indicator type	Indicator	Used for/Comments
Physical	Temperature Electrical Conductivity Turbidity Dissolved Oxygen pH	Use to assess cold water pollution Salinity of potable & irrigation water Determines light penetration Viability of water for biological life Impact on toxicity, metal speciation, physiology
Chemical	Nutrients Toxicants Bioavailable nutrients Total Organic Carbon	Phosphorus & nitrogen concentrations measured as indicator of eutrophication Examples included : heavy metals (Pb, Cd, Cu, Hg), pesticides and herbicides, PCBs & dioxins FRP, NO_x & NH_3 are better indicators of potential biological impact Used as a broad indicator of organic pollution
Biological	Macroinvertebrates Fish Algae Ecosystem processes	Basis of National Bioassessment Program. Rapid methods also available (e.g. AUSRIVAS) Operate at large scales, Expertise in agencies Especially diatoms, good potential – methods still being developed Measures of metabolism (photosynthesis, respiration) show good potential – methods still being developed

Sampling sites will be determined after consideration of the following issues:

- will the site provide relevant information about the major question being asked;
- is it logistically possible and safe to access the site whenever needed? (e.g. difficult to access remote sites in poor weather); and
- are there pre-existing data available for this site, which may assist placing the current results in a historical context? In addition, is there other information available that may help explain the collected data? (e.g. land use data) .

Directly assessing the 'health' of a lake or river (or any ecosystem) is an extremely difficult task, as there are no direct measures of ecosystem health. In human health assessment, indicators which measure components of 'health' (e.g. temperature, blood pressure) are used to provide information about the total well-being of the body. Similarly, in ecosystems, a relevant indicator is a measure of a biotic or abiotic variable that can provide uantitative information on the state or functioning of that ecosystem. Traditionally, physico-chemical indicators (e.g. pH, turbidity, nutrient concentrations) have been used, as these are relatively easy to measure.
However, these measures are difficult to relate to the health of the ecosystem.

Another advantage of some physico-chemical indicators is that they are direct measures of the stressors identified

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in the model(s) developed for the ERA (e.g. electrical conductivity is a very simple measure of salinisation). Biological indicators are a better indication of ecosystem health (e.g. the absence of living entities in a system are a direct indication of the poor condition of that system), but they are generally more time consuming (& perhaps expensive) to measure. Biological indicators integrate ecosystem conditions over longer periods, whereas physico-chemical measurements are an instantaneous 'snapshot'.A judicious combination of physico-chemical ('cause') and biological ('effect') indicators offers the best prospect of obtaining information about the condition of an ecosystem and the causes of any degradation (stressors) (see Table).

In monitoring the selected indicators, it is assumed that the replicate measurements taken are a tiny fraction of the infinite number of results that could be obtained given an infinite amount of time and money. As the laws of statistics apply only to populations, we then need to assume that the sample we have measured is truly representative of the population. Consideration of this assumption is critical when interpreting the data.

The timing and frequency of sampling depends entirely on the purpose of the monitoring program:

- for pollutant loads - then must sample high flow events;
- for compliance with guidelines - must sample under base flow conditions;
- for monitoring algal blooms - sample mostly in summer time - sample interval dictated by algal growth rates e.g. every 2 weeks will detect a growth event provided a representative sample is taken; and
- long term (multi year) monitoring is required to separate out annual trends and seasonal effects from random fluctuations.

An inherent assumption in almost all aquatic monitoring programs is that the system is spatially homogeneous and that the analysis of a single sample will provide a representative measure of concentration. A study of phosphorus concentrations in the Latrobe River examined sources of variability (Lovell et al., 2001). Analysis showed that the largest variability occurred between spots within a 100m site rather than sites several kilometers apart. Variation at this "reach" scale is rarely examined. Hence taking a few replicate samples at a specific spot may significantly underestimate the "real" variability.

Statistical power analysis of the Latrobe River data showed that, given the observed variability, in order to detect a difference of 25 µg P/L between locations, 10 replicates should be collected within the location. This analysis was based on a statistical power of 80% - i.e. there is an 80% chance of detecting a difference when there really is one (a 20% chance of a Type II error), and uses the almost unquestioned a value of 0.05 (i.e. there's a 5% chance of making a Type I error - predicting an effect when there isn't one). Taking more samples will increase the power but this becomes cost prohibitive.

For a set number of samples, Type I & Type II errors must be traded off. Which is the most important to avoid? Unthinkingly, people usually set a low a (avoid Type I) without considering power - in many environmental situations, it may actually be preferable to have false alarms rather than false security (see Figure), especially if the consequences of the event are severe.

		Measured response (outcome of test): Positive (impact)	Measured response (outcome of test): Negative (no impact)
True response	Positive (impact)	True positive	False negative (Type II error) “False Security”
True response	Negative (no impact)	False positive (Type I error) “False Alarm”	True negative

Figure: Error type matrix

A Receiver Operating Characteristic (ROC) curve is a diagrammatic representation of the trade-off between Type I and II errors. Several computer packages are dedicated to calculation and presentation of ROC curves (mainly as a basis for diagnostic decisions in medicine), and the reader is encouraged to examine these in the case where such trade-offs are environmentally, socially or economically critical.

The importance of careful consideration of the design of the sampling strategy cannot be overemphasized. Textbooks have been written on this subject and it is clearly not feasible to encapsulate all of the key issues here. However, a brief listing of a few fundamental points is certainly warranted.

As a case study, consider the common task of assessing the impact of a point source on a receiving stream. A difference between upstream and downstream measurements cannot be taken as evidence of the effect of the point source, as there may be natural attenuation (this is already assuming that the point source discharge is

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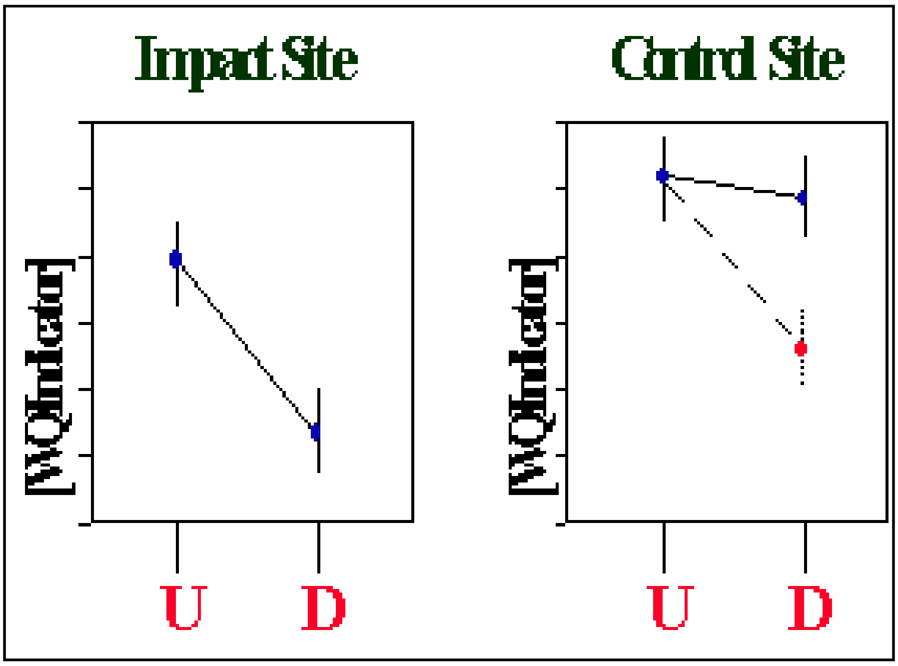
completely mixed with the receiving water at the downstream site). It is far better to include another 'control' upstream-downstream pair.

The Figure highlights the interpretation of possible outcomes:

Scenario 1: There is a significant difference between Upstream and Downstream at the impact site. There is no significant difference between U & D at the control. Hence, we can infer (weakly) that the U-D difference at the Impact Site is due to a change in WQ between the U and D sampling spots at this site.
Scenario 2: There is a significant difference between U & D at the impact site but there is also a significant difference between U & D at the control. Hence, we cannot infer any effect due to the point source at the Impact site.

Figure - Use of control and impact sites



Ideally, sampling is performed before and after the impact has commenced, with at least one, but preferably multiple, control sites. This design is known as "BACI" (Before and After at Control and Impact sites). This requires planning! Even with sampling before and after impact, it is vital that control sites are retained as these may detect other catchment changes, i.e. there may be other factors such as changed catchment management practices causing differences between the 'before' and 'after' data at the impact site.

5. GLOSSARY

Based on definitions provided by Suter (1993), Hayes (1997), VRHS (2002), AS/NZS (2004a) and OGTR (2004).

Term	Definition
Adverse effect	A change to the ecosystem or a component of the ecosystem that is judged to be detrimental.
Catchment	The region that drains all the rainfall, other than that removed by evaporation or evapotranspiration, into a stream, which then caries the water to the sea or a lake.
Conceptual model (Cause-effect model)	A written description and/or visual representation of actual or predicted relationships between humans or an ecological asset and the hazards, stressors or threats to which they may be exposed.
Consequence	The outcome or impact of an adverse event. More than one consequence can be associated with an event.
Context	Parameters within which risk must be managed, including the scope and boundaries for the risk assessment and risk management process.
Cumulative risk	The combined risks from aggregate exposures to multiple hazards or stressors.
Ecological asset	A natural resource that is valued by the community, e.g. river, wetland, forest, coastal zone, catchment.
Ecological effect	A change in the state or condition of an organism, population or ecosystem due to exposure to hazards or stressors.
Ecological risk assessment	The process of defining and quantifying risks to valued ecological assets, and determining the acceptability of those risks.
Ecological endpoints	An expression of the ecological values that are to be protected. *Assessment endpoint* is the quantitative or quantifiable expression of the ecological value considered at risk, e.g. a reduction in the abundance of fish in the river X. *Measurement endpoint* is the particular aspect of an assessment endpoint that can be measured.
Ecological value	The value of an ecological asset that the community wishes to protect, e.g. an ecologically healthy wetland, an ecologically sustainable catchment.
Estuary	The zone where a river meets the sea, is influenced by the river flows and tides, and is characterised by a gradient from fresh to salt water.
Event	The occurrence of a particular set of circumstances, which may be certain or uncertain, and may be a single occurrence or a series of occurrences.
Exposure	The process by which the temporal and spatial distribution of a chemical in the environment is converted to a dose.
Floodplain	The relatively smooth valley floors adjacent to and formed by alluviating rivers that are subject to overflow during flood events.
Frequency	The rate of recurrence of a particular event or action.
Hazard	A source of potential harm, or a situation with a potential to cause loss or an adverse effect.
Hazard identification	The process of analysing hazards and the events that give rise to harm.

Indicator	A characteristic of the environment that provides evidence of the occurrence or magnitude of exposure or effects. Formal expressions of the results of measuring an indicator are referred to as measurement endpoints.
Likelihood	The chance that something happening (can be expressed qualitatively or quantitatively).
Probability	The likelihood of a specific outcome. Probability is expressed as a number between 0 and 1, with 0 indicating an impossible outcome and 1 indicating that an event or outcome is certain.
Restoration	Improvement or enhancement of the environmental condition of the river in the direction of 'ecologically healthy'.
Risk	The chance of something happening that will have an undesirable impact on the objectives. Risk is measured in terms of a combination of the likelihood that a hazard gives rise to an undesirable outcome and the seriousness (consequences) of that undesirable outcome.
Risk acceptability	The acceptability of the risk by the stakeholders and those who will bear the consequences.
Risk analysis	The actual determination of the likelihood and consequences of undesirable effects on the ecosystem.
Risk assessment	The overall process of hazard identification, risk estimation and risk evaluation (may be qualitative, semi-quantitative or quantitative).
Risk management	The processes and structures to manage potential adverse impacts.
Stakeholder	Those people or organisations who may affect, be affected by, or perceive themselves to be affected by a decision, activity or risk.
Stressor	Any physical, chemical or biological entity that can induce an adverse effect. A stressor may also be the lack of an essential entity, such as a habitat.
Threat	An action or activity that has the capacity to adversely affect an ecological value or asset.
Uncertainty	A lack of knowledge arising from changes that are difficult to predict or events whose likelihood and consequences cannot be accurately predicted.
Wetlands	Inland, standing, shallow bodies of water, which may be permanent or temporary, fresh or saline.

6. REFERENCES

Ames, D. P., Neilson, B. T., Stevers, D. K. and Lall, U. (2004). Using Bayesian networks to model watershed management decisions: an East Canyon Creek case study. *J. Hydroinformatics* (in press).

ANZECC/ARMCANZ (2000a). *Australian and New Zealand Water Quality Guidelines*, Australia and New Zealand Environment and Conservation Council & Agriculture and Resource Management Council of Australia and New Zealand, Canberra.

ANZECC/ARMCANZ (2000b*). Australian guidelines for water quality monitoring and reporting*, Australia and New Zealand Environment and Conservation Council & Agriculture and Resource Management Council of Australia and New Zealand, Canberra.

AS/NZS (2000). *Environmental Risk Management: Principles and Practice*, Standards Australia & Standards New Zealand, HB203, Australian Standards International Ltd, Sydney.

AS/NZS (2004a). *Risk Management* (AS/NZS 4360: 2004), Standards Australia International, Sydney.

AS/NZS (2004b). *Risk Management Guidelines* (HB 436:2004), Standards Australia International, Sydney.

Bacon, P. J., Cain, J. D. and Howard, D. C. (2002). Belief network models of land manager decisions and land use change. *Journal of Environmental Management* 65(1): 1-23.

Bacon, P. J., Cain, J. D., Kozakiewicz, M. B. and Liro, A. (2002). Promoting more sustainable rural land use and development: a case study in eastern Europe using Bayesian network models. *J. Environ. Assessment Policy & Mangement* 4(2): 199-240.

Bailey, P. C. E. and Boon, P. I. (2002). *Contaminants fact sheet and salt sensitivity database*, Land and Water Resources Research and Development Corporation, Canberra.

Balmford, A., Bruner, A., Cooper, P., Costanza, R., Farber, S., Green, R. E., Jenkins, M., Jefferiss, P., Jessamy, V., Madden, J., Munro, K., Myers, N., Naeem, S., Paavola, J., Rayment, M., Rosendo, S., Roughgarden, J., Trumper, K. and R.K., T. (2002). Economic reasons for conserving wild nature. *Science* 297: 950-953.

Beer, T. (2003). Environmental risk and sustainability. In: *Risk Science and Sustainability*. T. Beer and A. Ismail-Zadeh (Eds.). Kluwer Academic Publishers, Dordrecht, 39-61.

Beer, T. and Ziolkowski, F. (1995). *Environmental Risk Assessment: An Australian Perspective*, Report No. 102, Supervising Scientist, Barton, Canberra.

Bennett, E. M., Carpenter, S. R., Peterson, G. D., Cumming, G. S., Zurek, M. and Pingali, P. (2003). Why global scenarios need ecology. *Frontiers in Ecology and the Environment* 1(6): 322-329.

Bernstein, P. L. (1996). *Against the Gods: The Remarkable Story of Risk*, John Wiley & Sons, New York.

Bier, V. M. (2001). On the state of the art: risk communication to the public. *Reliability Engineering and System Safety* 71: 139-150.

Binning, C., Cork, S., Parry, R. and Shelton, D., Eds. (2001). *Ecosystem Services - Natural Assets: An Inventory of Ecosystem Goods and Services in the Goulburn Broken Catchment*. CSIRO Sustainable Ecosystems, Canberra.

Bormans, M., Ford, P. W., Fabbro, L. and Hancock, G. (2004). Onset and persistance of cyanobacterial blooms in a large impounded tropical river, Australia. *Mar. Freshwater Res*. 55: 1-15.

Borsuk, M. E., Higdon, D., Stow, C. A. and Reckhow, K. H. (2001). A Bayesian hierarchical model to predict benthic oxygen demand from organic matter loading in estuaries and coastal zones. *Ecological Modeling* 143: 165 -181.

Borsuk, M. E., Stow, C. A. and Reckhow, K. H. (2002*). Integrative environmental prediction using Bayesian networks: A synthesis of models describing estuarine eutrophication.* Proceedings of IEMSS Conference, Lugano, Switzerland.

Borsuk, M. E., Stow, C. A. and Reckhow, K. H. (2004). A Bayesian network of eutrophication models for synthesis, prediction, and uncertainty analysis. *Ecological Modelling* 173(2-3): 219-239.

Breen, P. A., Kim, S. W. and Andrew, N. L. (2003). A length-based Bayesian stock assessment model for the New Zealand abalone Haliotis iris. *Marine and Freshwater Research* 54(5): 619-634.

Bromley, J., Jackson, N. A., Clymer, O. J., Giacomello, A. M. and Jensen, F. V. (2004). The use of Hugin to develop Bayesian networks as an aid to integrated water resource planning. Environ. *Modelling & Software*: (in press).

Brown, C. A. and Joubert, A. (2003). Using multicriteria analysis to develop environmental flow scenarios for rivers targeted for water resources management. *Water SA* 29(4): 365-374.

Burgman, M. A. (2001). Flaws in subjective assessments of ecological risks and means for correcting them. *Aust. J. Environmental Management* 8(4): 219-226.

Burgman, M. A. (2005). *Environmental Risk and Decision Analysis: For Conservation and Natural Resource Management*, Cambridge Univ Press, London.

Burnett Water (2001). *Burnett River Dam - Environmental Impact Statement*, Burnett Water Pty Ltd, Sept 2001, Brisbane.

Cain, J. (2001). *Planning improvements in natural resources management*, Centre for Ecology & Hydrology, Wallingford, UK.

Calow, P. and Forbes, V. E. (2003). Does ecotoxicology inform ecological risk assessment. *Environ. Sci. Technol*. 37: 146-151.

Carey, J. M., Burgman, M. A., Miller, C. and Yung, E. C. (2004b). An application of qualitative risk assessment in park management. *Aust. J. Environ. Management*: (submitted).

Carey, J. M., Burgman, M. A. and Yung, E. C. (2004a). *Risk assessment and the concept of ecosystem condition in park management*, Technical Series 13, Parks Victoria, Melbourne.

Clark, J. S. (2005). Why environmental scientists are becoming Bayesians. *Ecological Letters* 8: 2-14.

Cormier, S. M., Smith, M., Norton, S. and Neiheisel, T. (2000). Assessing ecological risk in watersheds: a case study of problem formulation in The Big Darby Creek watershed, Ohio, USA. *Environ. Toxicol. Chem*. 19: 1082-1096.

Cottingham, P., Beckett, R., Breen, P., Feehan, P., Grace, M. and Hart, B. T. (2001). *Assessment of Ecological Risk Associated with Irrigation Systems in the Goulburn Broken Catchment*, Technical Report No. 3/2001, CRC for Freshwater Ecology, Monash University, Melbourne.

Das, B. (2000). *Representing uncertainty using Bayesian Networks*, DSTO-TR-0918, Defence Science & Technology Organisation, Salisbury, South Australia.

Diamond, J. M. and Serveiss, V. B. (2001). Identifying sources of stress to native aquatic fauna using a watershed ecological risk assessment framework. *Environmental Science and Technology* 35: 4711-4718.

Dillon, P. J. and Rigler, F. H. (1974). The phosphorus:chlorophyll relationships in lakes. *Limnol. Oceangr*. 19: 767-773.

Duivenvoorden, L., Price, M., Noble, B. and Carroll, C. (2004). *Assessment of ecological risk associated with irrigation in the Fitzroy basin: Phase 2 - Analysis and characterisation of risk with emphasis on effects on macroinvertebrates*, National Program for Sustainable Irrigation Project UCQ3, Land & Water Australia, Canberra.

Ellison, A. M. (1996). An introduction to bayesian inference for ecological research and environmental decision-making. *Ecol. Applications* 6: 1036-1046.

Everbecq, E., Gosselain, V., Viroux, L. and Descy, J.-P. (2001). POTAMON: a dynamic model for predicting phytoplankton composition and biomass in lowland rivers. *Water Res*. 35: 901-912.

Fairweather, P. G. (1999). State of environment indicators of 'river health': exploring the metaphor. *Freshwater Biol*. 41: 211-220.

Feehan, P., Pollino, C., Webb, A., Chan, T., Grace , M., Hart, B. T. and Woodberry, O. (2004). *Ecological Risk Associated with Irrigation in the Goulburn Broken, Stage 2: Final Report*, Land & Water Australia, May 2004, Canberra.

Fischhoff, B., Slovic, P. and Lichtenstein, S. (1982). Lay foibles and expert fables in judgements about risk. *American Statistician* 36: 240-255.

Fisher, F. (2000). Citizens, experts, and the environment, Duke University Press, Durham.

Freudenburg, W. R. (1999). Tools for understanding the socioeconomic and political settings for environmental decision making. In: *Tools to aid environmental decision making*. V. H. Dale and M. R. English (Eds.). Springer, New York, 94-125.

Glenelg-Hopkins CMA (2003). *Glenelg-Hopkins Regional Catchment Strategy 2003-2007*, Glenelg-Hopkins Catchment Management Authority, Hamilton.

Glicken, J. (1999). Effective public involvement in public decisions. *Science Communication* 20(3): 298-327.

Glicken, J. (2000). Getting stakeholder participation 'right': a discussion of participatory processes and pitfalls. *Environmental Science & Policy* 3: 395-310.

Hart, B. T. (2004a). Environmental risks associated with new irrigation schemes in Northern Australia. *Ecological Management & Restoration* 5(2): 107-111.

Hart, B. T. (2004b). *Ecological Risk Assessment Training Program [MRC Environment Program Activity B-0009] - Final Report*, Mekong River Commission, Vientiane.

Hart, B. T., Breen, P. and Cullen, P. (1998). Ecological risk assessment for irrigation drainage discharges. In: *Proceedings of the Multi-objective Surface Drainage Design Workshop, Drainage Program, Tech. Report No. 7* (Eds.). Murray Darling Basin Commission, Canberra, 7-23.

Hart, B. T., Burgman, M., Grace, M., Pollino, C., Thomas, C. and Webb, J. A. (2005). Risk-based approaches to managing contaminants in catchments. *Human and Ecological Risk Assessment:* (in press).

Hart, B. T., Grace, M. R., Breen, P., Cottingham, P., Feehan, P. and Burgman, M. A. (2001). Application of ecological risk assessment in river management. In: *Proc. Third Australian Stream Management Conference - Value of Healthy Streams*. I. Rutherfurd, F. Sheldon, G. Brierley and C. Kenyon (Eds.). CRC for Catchment Hydrology, Melbourne, 289-295.

Hart, B. T., Lake, P. S., Webb, A. and Grace, M. (2003). Use of risk assessment to assess the ecological impacts of salinity on aquatic systems. *Aust. J. Botany* 51: 689-702.

Hart, B. T., Webb, A., Grace, M. and Feehan, P. (2002). *Ecological risk assessment from Australian irrigation enterprises. Proc. ANCID Conference*, Griffith, 1-4 September 2002.

Hayes, K. R. (1997). *Ecological risk assessment review*, CRIMP Technical report No. 13, CSIRO Marine Research, Hobart, Australia.

Hayes, K. R. (2002). Identifying hazards in complex ecological systems. Part 1: fault-tree analysis for biological invasions. *Biol. Invasions* 4: 235-249.

Hayes, K. R. (2004). *Ecological implications of GMOs: robust methodologies for ecological risk assessment. Best practice and current practice in ecological risk assessment for genetically modified organisms*, p73 [www.deh.gov.au/industry/biotechnology/ecological-risk/pubs/ecological-risk.pdf].

Kangas, A. S. and Kangas, J. (2004). Probability, possibility and evidence: approaches to consider risk and uncertainty in forest decision analysis. *Forest policy & Economics* 6: 169-188.

Keeney, R. L. and Raiffa, H. (1976). *Decisions with multiple objectives: preferences and value tradeoffs*, Wiley Publ., New York.

Khadam, I. M. and Kaluarachchi, J. J. (2003). Multi-criteria decision analysis with probabilistic risk assessment for the management of contaminated groundwater. *Environ. Impact Assessment Review* 23: 283-721.

Korb, K. B. and Nicholson, A. E. (2004). *Bayesian Artificial Intelligence*, Chapman & Hall, Boca Raton, Florida.

Lake, P. S. (2001). On the maturing of restoration: Linking ecological research and restoration. *Ecological Management & Restoration* 2: 100-115.

Lamon, E. C. and Stow, C. A. (2004). Bayesian methods for regional-scale eutrophication models. *Water Research* 38: 2764-2774.

Landis, W. G. and Wiegers, J. A. (1997). Design considerations and a suggested approach for regional and comparative ecological risk assessment. *Human and Ecological Risk Assessment* 3(3): 287-297.

Leuven, R. E. W. and Poudevigne, I. (2002). Riverine landscape dynamics and ecological risk assessment. *Freshwater Biology* 47: 845-865.

Lin, H.-I., Brerzins, D. W., Myers, L., George, W. J., Abdelghani, A. and Watanabe, K. H. (2004). A Bayesian approach to parameter estimation for a crayfish (Procambarus spp.) bioaccumulation model. *Environ. Toxicol. Chem*. 23(9): 2259-2266.

Littleboy, M., Vertessy, R. A. and Lawrence, P. (2003). *An overview of modelling techniques and decision support systems and their application for managing salinity in Australia*, 9th National Productive Use & Rehabilitation of Saline Land (PUR$L) Conference, September 2003 (CD ISBN 1 920920 31 5), Yeppon, Queensland.

Lovell, B., McKelvie, I. D. and Nash, D. (2001). Sampling design for total and filterable reactive phosphorus monitoring in a lowland stream: considerations of spatial variability, measurement uncertainty and statistical power. *J. Environmental Monitoring* 3: 463-468.

Lund, M. A. and McCrea, A. (2004a). *Ecological risk assessment associated with the impact of irrigation return on the risk of algal blooms in the Ord River*, Centre for Ecosystem Management, Edith Cowan University, Perth.

Lund, M. A. and McCrea, A. (2004b). *Ecological risk assessment associated with the impact of irrigation return on biodiversity in the Ord River*, Centre for Ecosystem Management, Edith Cowan University, Perth.

Mapstone, B. D. (1995). Scalable decision rules for environmental impact studies: effect size, type I and type II errors. *Ecol. Applications* 5: 401-410.

Mapstone, B. D., Davies, C. R., Little, L. R., Punt, A. E., Smith, A. D. M., Pantus, F., Lou, D. C., Williams, A. J., Jones, A., Ayling, A. M., Russ, G. R. and McDonald, A. D. (2004). *The effects of line fishing on the Great Barrier Reef and evaluations of alternative potential management strategies*, Tech. Report No. 52, CRC Reef Research Centre, Townsville.

Marcot, B. G., Holthausen, R. S., Raphael, M. G., Rowland, M. M. and Wisdom, M. J. (2001). Using Bayesian belief networks to evaluate fish and wildlife population viability under land management alternatives from an environmental impact statement. *Forest Ecology & Management* 153: 29-42.

MIL (2003). *Environmental Report 2003*, Murray Irrigation Ltd, Deniliquin.

MIL (2005). *Risk management plan for irrigation drains in MIL*, Murray Irrigation Ltd, Deniliquin.

Morgan, M. G. (1993). Risk management and analysis. *Scientific American* 269: 32-41.

Nayak, T. K. and Kunda, S. (2001). Calculating and describing uncertainty in risk assessment: the Bayesian approach. *Human and Ecological Risk Assessment* 7(2): 307-328.

Norton, S. B., Cormier, S. M., Suter, G. W., Subramanian, B., Lin, E., Altfater, D. and Counts, B. (2002). Determining probable causes of ecological impairment in the Little Scioto River, Ohio, USA: Part 1. Listing candidate causes and analyzing evidence. *Environmental Toxicology and Chemistry* 21(6): 1112-1124.

NRE (1999). *Risk Management: Strategic Framework and Process*, Dept Natural Resources and Environment, Melbourne.

O'Brien, M. (2000). *Making better environmental decisions: An althernative to risk assessment*, The MIT Press, Cambridge, Massachusetts.

OGTR (2004). *Risk Analysis Framework*, Consultation version, Office of Gene Technology Regulator (August 2004), Canberra.

Oliver, R. L. and Ganf, G. G. (2000). Freshwater blooms. In: *The Ecology of Cyanobacteria*. B. A. Whitton and M. Potts (Eds.). Kluwer Academic Publishers, Netherlands, 149-194.

Parslow, J., Margvelashvili, N., Palmer, D., Revill, A., Robson, B., Sakov, P., Volkman, J., Watson, R. and Webster, I. (2003). *The response of the lower Ord River and estuary to management of flows, sediment and nutrient loads*, OBP Project Report Final Science Report, CSIRO, Canberra.

Pate-Cornell, M. E. (1996). Uncertainty in risk analysis: Six levels of treatment. *Reliability Engineering & System Safety* 54: 95-111.

Peterson, G. D., Cumming, G. and Carpenter, S. R. (2003). Scenario planning: A tool for conservation in an uncertain world. *Conservation Biology* 17: 358-366.

Pidgeon, N., Hood, C., Jones, D. S., Turner, B. and Gibson, R. (1992). Risk perception. In: *Risk: Analysis, Perception and Management* (Eds.). Report of Royal Society Study Group, The Royal Society, London, Chapter 4.

Pollino, C. A., Feehan, P., Grace, G. and Hart, B. T. (2004b). Fish communities and habitat changes in the highly modified Goulburn catchment, Victoria, Australia. *Mar. Freshwater Res*. 55: 769-780.

Pollino, C. A., Woodberry, O., Feehan, P., Grace, M. R., Nicholson, A. E., Korb, K. B. and Hart, B. T. (2004a). Development of a Bayesian network to quantify the risks to fish in a highly modified catchment. *Ecological Modelling*: (in press).

Pollino, C. A., Woodberry, O., Nicholson, A. E. and Korb, K. B. (2005). *Parameterising Bayesian Networks: A Case Study in Ecological Risk Assessment*. Proc. 2005 Conference on Simulation and Modelling.

Prato, T. (2005). Bayesian adaptive management of ecosystems. *Ecol. Modeling*: in press.

QldDETIR (1999). *Workplace Health & Safety Risk Management Advisory Standard 2000*, Queensland Department of Employment, Training and Industrial Relations, Brisbane.

Reckhow, K. H. (2003). Bayesian approaches to ecological analysis and modelling. In: *Models in Ecosystem Science*. C. D. Canham, J. J. Cole and W. K. Lauenroth (Eds.). Princeton Univ Press, 168-183.

Recknagel, F., Ed. (2002). *Ecological Informatics: Understanding Ecology by Biologically- Inspired Computation*. Springer Verlag, Heidelberg.

Regan, H. M., Colyvan, M. and Burgman, M. A. (2002). A taxonomy and treatment of uncertainty for ecology and conservation biology. *Ecological Applications* 12: 618-628.

Reynolds, C. S. and Descy, J. P. (1996). The production, biomass and structure of phytoplankton in large rivers. *Arch. Hydrobiol. Supplement* 113: 161-187.

Reynolds, C. S., Irish, A. E. and Elliot, J. A. (2001). The ecological basis for simulating phytoplankton responses to environmental change (PROTECH). *Ecol. Modeling* 140: 271-291.

Robertson, D. and Wang, Q. J. (2004). Bayesian networks for decision analysis - an application to irrigation system selection. *Aust. J. Experimental Agriculture* 44: 145-150.

Rohrmann, B. (1994). Risk perceptions of different societal groups. Australian findings and cross-national comparisons. *Aust. J. Psychology* 46: 150-163.

Roy, B. (1999). Decision-aiding today: What should we expect. In: *Multicriteria Decision Making: Advances in MCDM Models, Algorithms, Theory and Applications*. T. Gal, T. J. Stewart and T. Hanne (Eds.). Kluwer Publishers, Boston.

Serveiss, V. B. (2002). Applying ecological risk principles to watershed assessment and management. *Environ. Management* 29(2): 145-154.

Serveiss, V. B., Bowen, J. L., Dow, D. and Valiela, I. (2004). Using ecological risk principles to to identify the major anthropogenic stressor in the Waquoit Bay Watershed, Cape Cod, Massachusetts. *Environ. Management* 33(5): 730-740.

SKM (2004). *No intervention future salt load projection: Murray Darling Basin salinity management strategy*, Sinclair Knight Merz, Melbourne.

Smith, A. D. M. (1993). *Risk assessment or management strategy evaluation: what do managers need and want?,* ICES CM D:18, Copenhagen.

Smith, A. D. M. (1994). Management strategy evaluation - the light on the hill. In: *Population Dynamics for Fisheries Management.* D. A. Hancock (Eds.). Proc. Australian Society for Fish Biology Workshop, Australian Society for Fish Biology, Perth, 249-253.

Solomon, K., Giesy, J. and Jones, P. (2000). Probabilistic risk assessment of agrochemicals in the environment. *Crop Protection* 19: 649-655.

Solomon, K. R., Baker, D. B., Richards, R. P., Dixon, K. R., Klaine, S. J., LaPoint, T. W., Kendall, R. J., Weisskopf, C. P., Giddings, J. M., Giesy, J. P., Hall, L. W. and Williams, W. M. (1996). Ecological risk assessment of atrizine in North American surface waters. *Environ. Toxicol. Chem*. 15(1): 31-76.

Stewart, M. G. and Melchers, R. E. (1997). *Probabilistic Risk Assessment of Engineering Systems,* Chapman & Hall, London.

Stow, C. A. (1999). Assessing the relationship between Pfiesteria and estuarine fishkills. *Ecosystems* 2: 237-241.

Stow, C. A. and Borsuk, M. E. (2003). Enhanced causal assessment of estuarine fishkills using graphical models. *Ecosystems* 6: 11-19.

Stow, C. A., Borsuk, M. E. and Stanley, D. W. (2001). Long-term changes in watershed nutrient inputs and riverine exports in the Neuse River, North Carolina. *Water Research* 35(6): 1489-1499.

Stow, C. A., Carpenter, S. R. and Lathrop, R. C. (1997). A Bayesion observation error model to predict cyanobacterial biovolume from spring total phosphorus in Lake Mendota, Wisconsin. *Can. J. Fish. Aquatic Sci*. 54: 464-473.

Suter, G. W. (1993). *Ecological Risk Assessment*, Lewis Publishers, Chelsea MI.

URS Australia (2002). *National Action Plan for Salinity and Water Quality - Evaluation of Models and Tools*, Milestone Report to Natural Resources Management Business Unit, Department of Agriculture, Fisheries and Forestry, Canberra.

USEPA (1998). *Guidelines for Ecological Risk Assessment*, EPA/630/R-95/002F, U.S. Environmental Protection Agency, April, 1998, Washington.

USEPA (2003). *Framework for Cumulative Risk Assessment*, EPA/630/P-02/001F, U.S. Environmental Protection Agency, May 2003, Washington.

USFDA (1997). *Hazard analysis and critical control point principles and application guidelines*, US Food and Drug Administration, Washington.

van Dam, R. A., Camilleri, C., Bayliss, P. and Markich, S. J. (2004). Ecological risk assessment of tebuthiuron following application on tropical Australian wetlands. Health & *Ecol. Risk Assessment* 10 (6): 1019-1048.

Van der Heijden, K. (1996). *Scenarios: the art of strategic conversation*, Wiley, Chichester.

VCMC/DSE (2003). *Ecosystem Services through Land Stewardship Practices: Issues and Options*, Victorian Catchment Management Council and Department of Sustainability & Environment, April 2003, Melbourne.

Vertessy, R. A., Watson, F. G. R., Rahman, J. M., Cuddy, S. M., Seaton, P. S., Chiew, F. H. S., Scanlon, P. J., Marston, F. M., Lymburner, L., Jeannelle, S. and Verbunt, M. (2001). *New software to aid water quality management in catchments and waterways of the south-east Queensland region*. Proceedings of the 3rd Australian Stream Management Conference, Brisbane.

VicEPA (2004). *Guidelines for Environmental Management: Risk-based Assessment of Ecosystem Protection in Ambient Waters*, Publication No 961, Victorian Environment Protection Authority, Melbourne.

Vose, D. (1996). *Quantitative Risk Analysis: A Guide to Monte Carlo Simulation Modeling*, John Wiley and Sons, New York.

Vose, D. (2000). *Risk Analysis: A Quantitative Guide*, John Wiley & Sons, Chichester.

VRHS (2002). *Victorian River Health Strategy - Healthy Rivers, Healthy Communities and Regional Growth*, Dept Natural Resources & Environment (August 2002), Melbourne.

Walters, C. J. (1986). *Adaptive Management of Renewable Resources*, Macmillan Publishing Company, New York.

Webb, J. A. and Chan, T. U. (2004). *Ecological risk associated with irrigation systems in the Goulburn-Broken Catchment - Phase 2: Priority risk - blue green algal blooms*, Water Studies Centre, Monash University, Melbourne.

Webb, J. A. and Hart, B. T. (2004). *Environmental Risks from Salinity Increases in the Goulburn-Broken Catchment*, Water Studies Centre, Monash University, Melbourne.

Webb, J. A., Linacre, N. A. and Grace, M. R. (2004). Management-oriented modelling of blue-green algal blooms: an example from Bourke Weir, NSW, Australia. *Ecological Modeling*: (submitted).

Woodberry, O., Nicholson, A. E., Korb, K. B. and Pollino, C. A. (2004). Parameterising Bayesian networks: *A case study in ecological risk assessment*, Proc. Pacific Rim International Conference on Artificial Intelligence, August 2004, New Zealand.

Wooldridge, S. (2003). *Bayesian Belief Networks*, Centre for Complex System Science, CSIRO, Canberra.

Wooldridge, S. and Done, T. (2004). *The use of Bayesian Belief networks to aid in the understanding and management of large-scale coral bleaching*, MODSIM 2003 Conference, Modelling & Simulation Society of Australia and New Zealand, 614-619 (http://mssanz.org.au/modsim03/modsim2003.html).

7. RELEVANT WEB SITES

Water Studies Centre, Monash University	http://www.wsc.monash.edu.au
USEPA, National Center for Ecological Risk Assessment	http://cfpub2.epa.gov/ncea/cfm/ecologic.cfm
MERIT Project (Management of the Environment & Resources using Integrated Techniques)	http://www.merit-eu.net
Watershed risk assessment training notes	http://www.epa.gov/owow/watershed/wacademy/acad2000/ecorisk
Society for risk analysis	http://www.sra.org/
NPSI, National Program for Sustainable Irrigation	http://www.lwa.gov.au/irrigation

The National Program for Sustainable Irrigation focuses irrigation research on critical emerging environmental issues while also aiming to improve the productivity of irrigated agriculture and maximise community benefits.
http://www.lwa.gov.au/irrigation/

The Water Studies Centre at Monash University aims to generate and exchange scientific knowledge to underpin the effective management of Australia's waterways.

http://www.wsc.monash.edu.au/

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