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Salt, Nutrient, Sediment and Interactions:

Findings from the National River Contaminants Program



River Landscapes



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Anabaena circinalis.
Photo CSIRO.





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Chapter 1 — The National River Contaminants Program

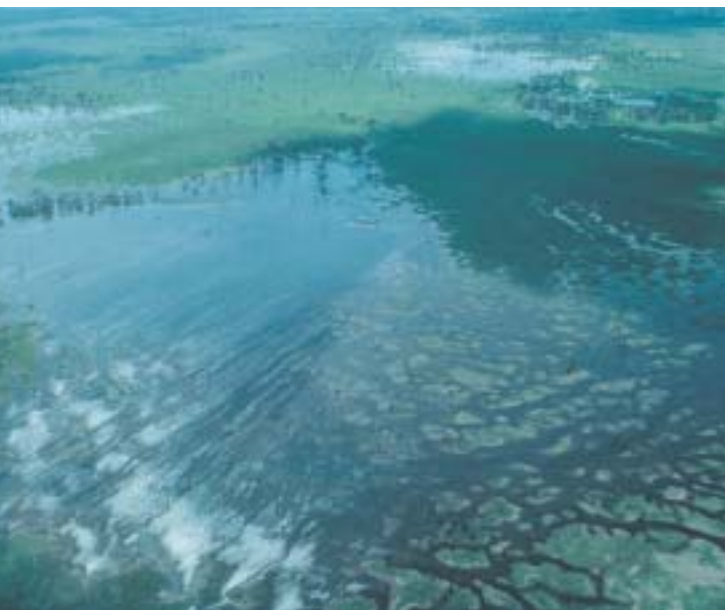
Brendan Edgar, Richard Davis and Phil Price

Land & Water Australia

Summary

The National Rivers Contaminants Program (NRCP) has had a range of important highlights including:

- new knowledge on how salinity affects animals in rivers, and guidance for managers on setting targets and environmental flows,
- improved management by the fertiliser industry in advice it provides to the dairy, sheep and beef producers, and
- new knowledge for managers on the role of riparian zones in influencing the passage and transformation of nitrogen into streams.



Macquarie Marshes, NSW. Photo Bill Johnson.



Gorgonian seafans off Whitsunday Island. Photo FRDC.

Background

The Australian community is increasingly aware of the importance of our water resources and riverine environments to the future sustainability of agriculture, rural and urban water supply, estuaries and in-shore fisheries, recreation, and conservation of our unique aquatic biodiversity. Consequently, river restoration has become a priority for many catchment and resource managers looking to repair damaged rivers. Contaminants in rivers are central to this issue because they determine both the quality of irrigation and drinking water, as well as the condition of in-stream habitats for river-dependent plants and animals.

River contaminants fall into two broad categories — firstly, substances that occur naturally, but for which significant increases in the amounts present contaminate the environment, and secondly, those that do not occur naturally, for which even small amounts may contaminate the environment. Examples of the first category are salt, nutrients, and sediments — about which we need to understand the sources of excess loads, their ecological effects, and options for improved management. Examples of the second category are agricultural chemicals and heavy metals, about which we need to understand their ecological effects and the extent to which we need to improve their management.

River contaminants are also a major threat to receiving waters (estuarine, coastal, wetland and reservoir). Some of these ecosystems are of enormous national value, e.g. the Great Barrier Reef, Gippsland Lakes, Macquarie Marshes, and Swan-Canning estuary.

To improve our understanding and management of river contamination issues and, ultimately, to help reduce the associated environmental, social and economic costs, the National River Contaminants Program (NRCP) was established in 2001 by Land & Water Australia (LWA) and the Murray-Darling Basin Commission (MDBC). This continued the partnership between LWA and the MDBC from the preceding National Eutrophication Management Program, that focused on the causes and management of algal blooms in waterways, including the role of phosphorus as a contaminant.

The National River Contaminants Program Strategic Plan (ATECH 2000) canvassed the views of catchment and river managers about the most important river contaminant issues. Using this data, outlined in Figure 1, it was agreed to focus the Program on developing strategies for better managing salt, nutrients and sediments as priority contaminant issues.

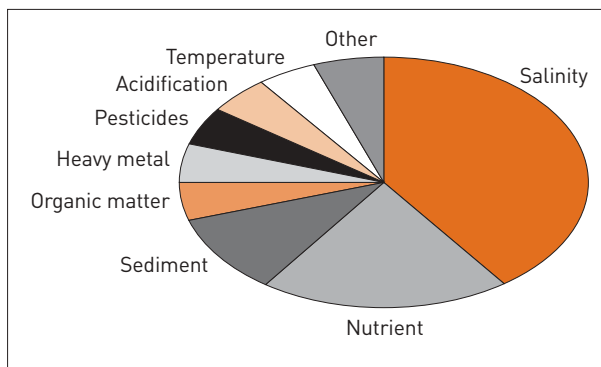


Figure 1. Indicative national significance of river contaminant issues (regional significance may be different).



Ninety-mile Beach and Gippsland Lakes, Vic. Photo © Reg Morrison/AUSCAPE.



The Channels, WA. Photo Jenny Davies.

The first activity of the NRCP was a workshop (June 2001) to scope the *sources, pathways and transformations* of each of these contaminants within river systems, and to consider the possible interactions between them. A Program Plan was prepared to guide investments, knowledge management and evaluation over the life of the Program (NRCP R&D Plan 2001, LWA).

The objective of the NRCP is to improve our understanding and management of river contamination issues, to help reduce the associated costs and to better manage the risk of river contamination. To do this we need to understand:

- where contaminants are coming from in the landscape,
- how they are transported to the river system,
- what transformations will occur as contaminants interact with the water column and bed or bank materials, as well as with each other, and
- what are the ultimate fates and impacts of river contaminants on water quality, aquatic life, and the riverine system overall.

The ultimate goal of the NRCP is improved water quality of Australian streams and rivers to meet the community's objectives of maintaining ecological integrity and biodiversity, and to underpin sustainable use of the water resources for current and future generations.

Areas of Program research

Salt as a contaminant

Salinisation of landscapes and rivers has been identified as one of Australia's most serious environmental issues, particularly for southern regions (MDBC 1999), but is also a high risk for some warm temperate and dry tropics regions (Council of Australian Governments 2000). In areas already affected, secondary (or dryland) salinity has severely impacted on ecosystems with large losses of habitat, biodiversity, native vegetation and water resource value.

Much is now known about the causes of dryland salinity, and a number of effective strategies have been demonstrated to reduce the problem. However, little research has been conducted on the specific environmental impacts of salinisation on streams and rivers (Bailey & James 2000). In particular, few investigations have examined the biological changes in salinised rivers or wetlands and, in general, knowledge concerning the effects of increasing salinity on aquatic ecosystems has been inadequate to guide decision making. Knowledge is required to understand how aquatic species respond to changing levels of salt in river and wetland systems, taking account of natural adaptation to salinity and the wide variation in primary and secondary salinised sites across Australia. The effects of salt as a contaminant are discussed in chapters 2 and 3.



Bega River, NSW. Photo Andrew Brooks.

Sediment as a contaminant

Worldwide, sediments are probably the most common river contaminant. Coarse sediments alter river habitats by infilling bed interstices, thus degrading benthic habitat. Widespread sediment deposition can bury entire riffle-pool reaches, creating sand slugs that replace diverse river habitats with uniform sand beds and wide shallow flow. Fine sediments that are carried in suspension interfere with the feeding of many river animals, for example, favouring fish (such as carp) that are not visual feeders. By increasing turbidity, and hence reducing light penetration, suspended sediments also reduce submerged plant photosynthesis and alter the light regime for phytoplankton. The reduced light penetration can favour algal species, such as toxic cyanobacteria, that are able to regulate their cell buoyancy and move into the narrow upper light zone. In highly turbid systems, such as the pools common along Australia's inland rivers and channel country, photosynthetic production may be restricted to a thin 'bath tub ring' around the periphery of the pool.

Many agrochemicals, heavy metals and nutrients chemically bind to sediments, and so are transported along with the sediment. Sediments form a substrate on which these contaminants can undergo chemical transformations. Any complete examination of river contaminants needs to consider both the direct contamination by sediment, as well as the role of sediment in transporting and transforming other contaminants. Sediment contamination is considered further in chapters 7 and 8.



Victoria Park Lake showing submerged macrophytes. Photo Paul Boon.

Nutrients as contaminants

Nutrients, such as nitrogen and phosphorus are essential chemicals for cellular growth. Under normal concentrations, river nutrient levels do not constitute a serious issue for irrigation or drinking water quality. However, when they are present in excess amounts, nutrients can cause severe ecological effects (eutrophication) and degrade water quality. Nutrient enrichment of rivers stimulates primary production — sometimes aquatic plant growth, but more commonly excessive algal growth. This is especially the case when nutrient enrichment is combined with lack of shade (e.g. through loss of riparian vegetation), as high light intensity and warmer water also stimulate primary production.

Excessive algal growth, often seen as algal blooms, is of concern to water supply authorities because both free floating algae (phytoplankton) and attached algae can block filters and extraction equipment, and the high organic load leads to increased water treatment costs. In addition, algal blooms can interfere with recreational uses of waterways and be unsightly. Blooms of many cyanobacteria species of algae are additionally problematic because of the highly potent toxins they can produce.

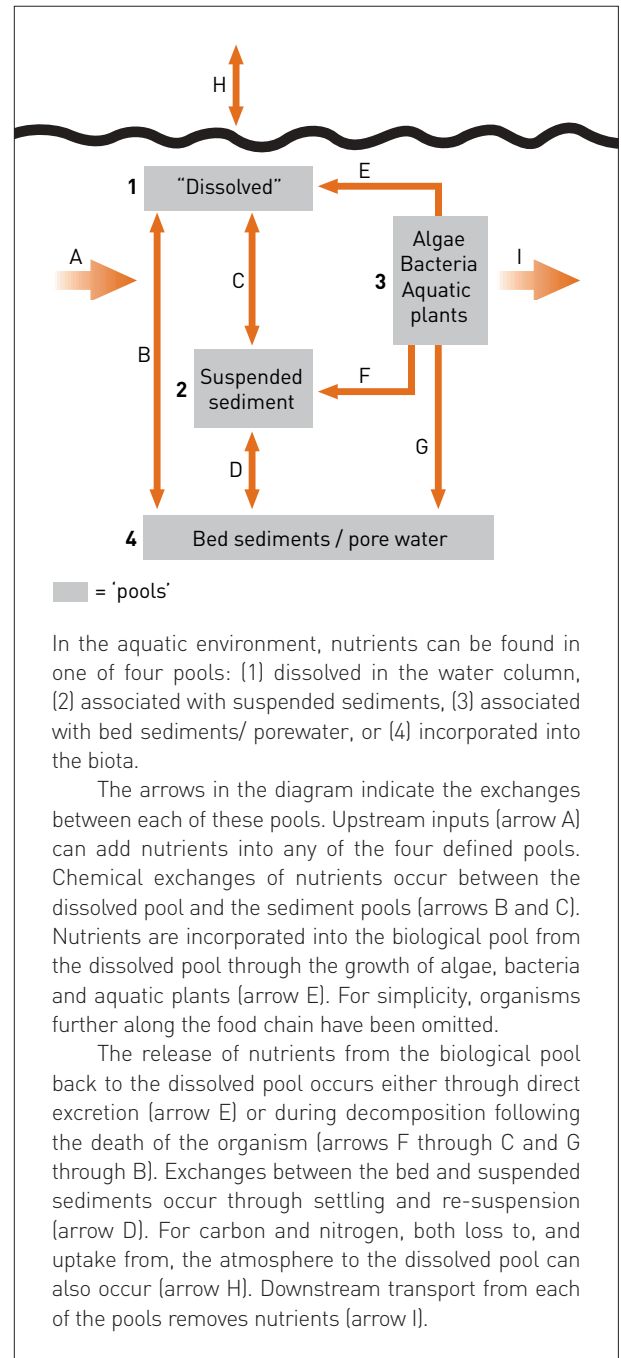
Much of the Australian eutrophication research on inland aquatic systems has focused on phytoplankton blooms, and understanding the roles of phosphorus supply and flow conditions. This has proven to be well founded, as in many rivers and reservoirs, the thermal stratification that occurs during low or no flow conditions provides the necessary environment



Photo Kay Morris.

for rapid algal population growth. The total biomass a bloom can achieve is controlled by the availability of nutrients.

As well as phosphorus, it has now also been demonstrated that nitrogen and the amount of light can limit phytoplankton growth in a number of inland waters, as well as influence the species of algae that constitute the bloom. Consequently, to control the size of algal blooms it is necessary to obtain a better understanding of the nitrogen cycle and its role in controlling algal biomass and species composition. This includes investigating and quantifying the links between surface and sub-surface nitrogen movement through riparian zones, as well as interactions with adjacent streams. There is also a need to understand the circumstances in which nitrogen is likely to be a significant management issue, and address these by developing guidelines for use by managers. Nutrients as contaminants are discussed in chapters 4, 5 and 6.



Interactions between contaminants

While understanding and managing river contamination by single substances might be relatively simple, very little is known about the synergistic or antagonistic effects of different contaminants. Different contaminants may chemically interact during transport or once deposited, and the ecological responses to 'cocktails' of chemicals are likely to be wide-ranging and complex. The interactions between contaminants, the net ecological responses, and the links back to catchment and river management options, are relatively unexplored in catchment-scale research.

The largest gaps in our understanding are related to the interactions between contaminants, both how they interact physically and chemically in transport or in storage, and the complex responses of aquatic biota to mixtures of contaminants. While relatively simple experiments can examine the tolerances and responses of individual organisms to particular contaminants or even combinations of contaminants, scaling these results up to predict ecosystem level response is extremely difficult. The combination of detailed experimental work with medium scale field testing and large-scale system modelling is likely to be the best way to advance our understanding. Considering overall contaminant risks and using models to help identify crucial risks and management responses, are discussed in chapters 9, 10 and 11.

Impact of the Program

A review of the NRCP was conducted by the consulting company Sinclair Knight Merz in 2004 (unpublished report to LWA). It found the science conducted in the Program to be of very high quality, with researchers uniformly enthusiastic about their projects. Projects were generally well connected to stakeholder groups through combinations of formal and informal networks, and linked with each other and with relevant work outside the Program. Several projects had excellent potential to influence river management practice or policy, and the research represented a well balanced portfolio that comprehensively addressed its objectives and research priorities.

This book summarises the research findings from the Program for water resources managers. These findings are presented in ways that will enhance the understanding on which managers have to base their decisions. While in many cases the findings reinforce existing understandings of the sources and effects of contaminants, in some cases — such as the importance of nitrogen and light in controlling riverine algal biomass — the findings are revolutionary and offer managers new opportunities for managing contaminants in Australian rivers.

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Salt as a contaminant

Salinisation of landscapes has been identified as one of Australia's most serious environmental issues in southern regions, and is also a high risk for some warm temperate and dry tropical parts of Australia. In many areas salinisation has already affected terrestrial ecosystems, leading to losses of habitat, biodiversity, and native vegetation, with increasing salinity predicted to cause deterioration in infrastructure, such as roads, buildings and bridges. Dryland salinity (secondary salinity) will also add to the salt loads in rivers to the point where it is estimated that by 2020, the Murray River's salinity will exceed drinking water standards for nearly 150 days a year (MDBC 1999). Increasing riverine salinity levels will also affect aquatic biodiversity in rivers and wetlands — which is the focus of this section.

Primary and secondary salinity: Primary salinity is where increases in salinity have occurred solely through natural processes. Secondary salinity is where increases have occurred due to land use changes made by human activity.

Much is now known about the causes of dryland salinity, and a number of effective strategies have been demonstrated to reduce the problem. However, little research has been conducted on the specific environmental impacts of salinisation on rivers (Bailey & James 2000). In particular, few investigations have examined the biological changes in salinised rivers or wetlands and, in general, knowledge concerning the effects of increasing salinity on aquatic ecosystems has been inadequate to guide decision making. We need to understand how aquatic species respond to changing levels of salt in river and wetland systems, and learn what is happening to ecosystems across the wide variation in primary and secondary salinised sites across Australia.

While there are some commonalities in understanding the effects of salinity across the fresh to hypersaline continuum, there are also many important differences in the characteristics of organisms and the communities inhabiting waters along this continuum. There may also be important differences in the major biological, physical and chemical processes that occur across this range of salinity values. Chapter 2 looks at this issue and addresses the effect of salinity increases in fresh, or only slightly saline, waters (≤ 3 mS/cm or 2.3 g/L) in eastern Australia. Chapter 3 considers further increases in salinity within saline waters (10–100 g/L or 13–130 mS/cm) and hypersaline water (>100 g/L or >130 mS/cm) in Western Australia. Both chapters examine how salinity targets can be set to trigger management intervention to protect biodiversity.

It is not enough to just understand the causes of salinity and the environmental consequences; preventative and remedial management actions also need to be developed. There is a range of management options that can be used to influence the salinity of rivers and wetlands in the short- to medium-term, including:

- release of environmental flows,

- altering the amount of water extraction,
- management of weir pool depth,
- interception of saline water inputs, and
- managing the disposal of saline water, including into freshwater systems.

Over the long term, salinity of waterways can in many places be influenced by altering landuse, vegetation (water use), drainage and hydrology in a catchment.

The detection and management of salinity impacts on aquatic ecosystems requires natural resource management standards that are based on scientific evidence. As these next two chapters illustrate, there is now sufficient information available for these standards to be set for both saline and hypersaline conditions. Establishing these standards is a task for managers.

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- Bailey, P.C.E. & James, K. 2000, *Riverine and wetland salinity impacts — assessment of R&D needs*, Report for Land and Water Resources Research and Development Corporation, Canberra.
- MDBC 1999, *Salinity Strategy — 10 years on*, Murray-Darling Basin Commission, Canberra.

Pink Lake, WA. Photo Jenny Davis.



Chapter 2 —

Understanding salinity thresholds in freshwater biodiversity: freshwater to saline transition

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Summary

- Changing salinity in freshwater systems can have detrimental impacts on biodiversity. While biodiversity is a management goal, it is also used as a surrogate for community structure/function and, therefore, ecosystem health/services. To prevent or minimise impacts, it is important to set maximum salinity targets. It is also important to identify taxa or other indicators of salinity impacts so that biomonitoring can identify impacts before they become severe or irreversible.
- Although there is limited data for some biological groups, the available evidence suggests that protecting salt sensitive freshwater macroinvertebrates from salinity changes will protect all biological groups found in freshwater.
- In general, the salinity sensitivity of related macroinvertebrate species is similar across eastern Australia. However, there is wide variation in the communities present, giving rise to differences in their sensitivity to salinity. Consequently, there is a need to derive regional salinity guidelines for freshwater systems.
- Based on results of laboratory experiments and field work, generalised salinity sensitivity scores have been assigned to families using a salinity index.
- We suggest an approach for developing regional guidelines for salt sensitivity, within a risk assessment framework. The major technical steps in this process involve:
 - a) making a regional macroinvertebrate species list and then assigning salinity sensitivity information to each taxa from all the available data sources.
 - b) applying a safety factor to transform the tolerances assigned to each listed species. Most available information only considers short-term salinity that is lethal to mature aquatic stages, however, it is known that lower concentrations will cause sub-lethal harm with longer exposure and that the young of some species are more salt sensitive. These new safety factors take this into account.
 - c) using the transformed list, the maximum permissible salinity level can be estimated that will protect a given percentage of the species present, e.g. 95%.
- Modification to the guidelines will be needed to account for varying factors. In general, sudden rises and falls in salinity will be more detrimental to freshwater organisms than gradual changes, and we recommend that rapid alterations be avoided. We also recommend it be acknowledged that the total effect of changes in salinity, and other factors, that co-vary with salinity, may be greater than the sum of each factor.

Background

It is now well accepted that there is a widespread dryland (or secondary) salinity problem in eastern and south-western Australia caused by historic clearing of native vegetation and, in some specific locations, by over-irrigation, leading to rising saline watertables. The impact of salinity on agriculture, infrastructure, water supply extractions and terrestrial biodiversity is well documented. Yet there remains considerable uncertainty about its implications for freshwater communities and associated biodiversity. The term 'freshwater biodiversity' suggests that the animals and plants that live in rivers, wetlands, lakes, etc. will be intolerant of saline water. However, all freshwater organisms can tolerate some level of salinity, and some organisms found in freshwater can even thrive in more saline environments (e.g. Kefford et al. 2006a). Some freshwater fish and macrocrustaceans are diadromous, moving between freshwater, estuarine and even marine environments.

All water bodies naturally have salts dissolved in them, and there is no such thing as natural, completely salt-free water. Indeed, the chemical elements that make up salts are essential for life, and without them life as we know it would not exist. At too high a concentration, however, they become lethal and can have non-lethal effects such as reduced growth and reproduction. The critical question therefore, is not whether salinity is

detrimental to freshwater biodiversity but rather, how much and at what rate can salinity rise above background concentrations in the freshwater environment before biodiversity is affected.

To answer this question, knowledge of the maximum salinity thresholds or sensitivity/tolerance of freshwater organisms is required. The protection of freshwater organisms or biodiversity is an important management goal in its own right. Protection will additionally ensure that community structure (the mix of organisms present), function (the ecological activities which they perform) and, therefore, ecosystem health and the ecosystem services that communities depend on, are also maintained.

This chapter addresses the effect of salinity increases in fresh, or only slightly saline, waters (≤ 3 mS/cm or 2.3 g/L total salts). The studies reported here were conducted in eastern Australia where the major concerns about the affects of secondary salinisation relate to the impacts of salt moving into waters that are naturally fresh or only slightly saline (a contrast to Western Australia where many water bodies are naturally brackish or have already become saline). While there will be regional differences that should be considered, the conclusions from the data presented in this chapter can be applied to predict the likely effects of salinity increases and their management in freshwater environments across much of eastern Australia.

Barney Creek (Logan-Albert River Catchment) south-east Queensland. Photo Ben Kefford.



Definitions

Lethal effects = result in mortality, when salinity kills individual organisms.

Sub-lethal effects = occur at salinity concentrations that do not kill an individual organism but affect its growth, reproductive potential, behaviour, physiology, etc.

Short term or acute effects (Acute toxicity) = effects, either lethal or sub-lethal that occur in the short-term relative to the life-time of the organism. Tests measuring the short term acute effects of salt on macroinvertebrates are usually conducted over ≤ 96 hours. While acute tests generally measure mortality, they can also measure sub-lethal effects.

Chronic effects = effects that occur over the longer-term relative to the life-time of the organisms; they can be either lethal or sub-lethal. While there is no one accepted timescale to define chronic effects for macroinvertebrates, we consider chronic effects to be those detected over a period of ≥ 7 days, but note that some may become apparent only over considerably longer periods.

Direct effects = when the actions of salinity affect an organism to elicit a response either lethally or sub-lethally.

Indirect effects = when salinity does not elicit a response via an organism's own physiological tolerance, but because more salt-sensitive species are its

predators, prey, competitors, etc. the organism is nevertheless affected via ecological interactions, either lethally or sub-lethally.

LCx value = the salinity concentration lethal to X% of individuals over some specified period. If, for example, X is 50% and the period of exposure is 72 hours, it would be expressed as the *72h LC₅₀ value*.

Safety factor = an approximate conversion factor to transform ecotoxicological data so that it includes unmeasured effects. For example, an acute LC₅₀ value may be divided by some number, and the resulting value be assumed to account for unmeasured sub-lethal and chronic effects. Safety factors are also known as extrapolation and application factors.

Species sensitivity distribution (SSD) = the statistical distribution of sensitivity in an aquatic community to a particular stressor (using a particular index to sensitivity e.g. LC₅₀ value).

Salinity = the total concentration of dissolved inorganic ions in water or soil. As measuring the concentration of all ions is time consuming and expensive, salinity is often inferred by measuring the electrical conductivity (EC) of a water or soil sample. EC is standardised to 25°C and is usually expressed in $\mu\text{S}/\text{cm}$ or mS/cm , with $1000 \mu\text{S}/\text{cm} = 1 \text{mS}/\text{cm}$.

Ionic proportions = the relative concentrations of individual major ions that make up salinity in a proportional basis.

Effects of increasing salinity on freshwater biodiversity

Why would we want to know the effects of increasing salinity on freshwater biodiversity? From a planning and policy-management perspective this information is needed to prevent or mitigate adverse impacts on the aquatic ecosystem and comply with Ecologically Sustainable Development (ESD) principles. Once impacts on biodiversity have occurred, it may be difficult to rehabilitate the system to its former condition. Therefore, it is essential that either the potential impacts and risks are known, or the impacts from salinity are detected at an early stage and managed appropriately to prevent decline in the ecological health of waterways. The ability to quantify subtle ecosystem changes and know acceptable concentration levels will help to provide

relevant and biologically meaningful targets for salinity. These would also complement salinity management measures on the broader (terrestrial) landscape.

As salinity is highly spatially and temporally variable, and is associated with a variety of direct and potentially indirect effects (see Rohr et al. 2006), predicting the consequences of salinity for freshwater biodiversity is not a simple task. There are several factors that make it difficult to determine what salinity levels are safe for freshwater biodiversity. Both field and laboratory approaches have been used to assess salinity impacts, and each has advantages and disadvantages.

Most salinity gradients are confounded with other changes in water quality, habitat, hydrology, etc. which also effect biodiversity, and may also alter the effect of salinity. It is relatively simple to compare biotic communities found in hypersaline and freshwater sites,



Jason Dunlop and Rajesh Prasad at one of the sample sites. Photo Ben Kefford.

and to conclude that the difference in biodiversity is caused by the salinity (Williams et al. 1990). However, across smaller changes in salinity it is more difficult to conclude that there is a causative relationship. Given these confounding effects, it is difficult to ascertain thresholds levels of salinity for freshwater biodiversity relying exclusively on field studies.

Laboratory and small scale field experiments are useful because they can establish a causal connection between salinity and measured responses. They may, however, over-simplify the complexity of nature and thus not accurately predict the effect of salinity on organisms in nature (Kefford et al. 2004a). Despite this, there is a range of experimental approaches that can be used to assess salinity thresholds (see box on previous page). Salinity thresholds for individual organisms can be defined as short-term (acute) or long-term (chronic) effects. Responses (or end-points) of organisms to salinity can include mortality, or a range of sub-lethal effects that can include changes in reproduction, growth, behaviour, physiology, etc. Both lethal and sub-lethal effects can be determined for organisms at different life-stages. Experimentally derived sensitivity values are available only for a small fraction of the total number of species that exist in Australian freshwaters. Different species will have different salinity thresholds, and for each species

when salinity sensitivity is assessed using different endpoints, exposure periods and life-stages, there will likely be different salinity thresholds. For each of these different endpoints, individuals of the same species collected from different locations can have different salinity threshold values. At different locations there will be a different mix of species present, so the distributions of salinity tolerances of the communities, or species sensitivity distributions (SSDs), differ between locations (Kefford et al. 2005).

The work described in this chapter improves predictions of the effects of increasing salinity on freshwater faunal biodiversity in eastern Australia, and identifies taxa sensitive and tolerant of high salinity levels for use in biomonitoring and bioassessment programs.

Key findings

Spatial variation in salinity tolerance

The relative salinity sensitivity (measured as 72h LC₅₀ values — see box page 11) of a wide range of macro-invertebrates was assessed in a six locations chosen to represent a wide biogeography range across eastern Australian regions likely to be affected by secondary salinisation (Figure 1). In Victoria, these were the

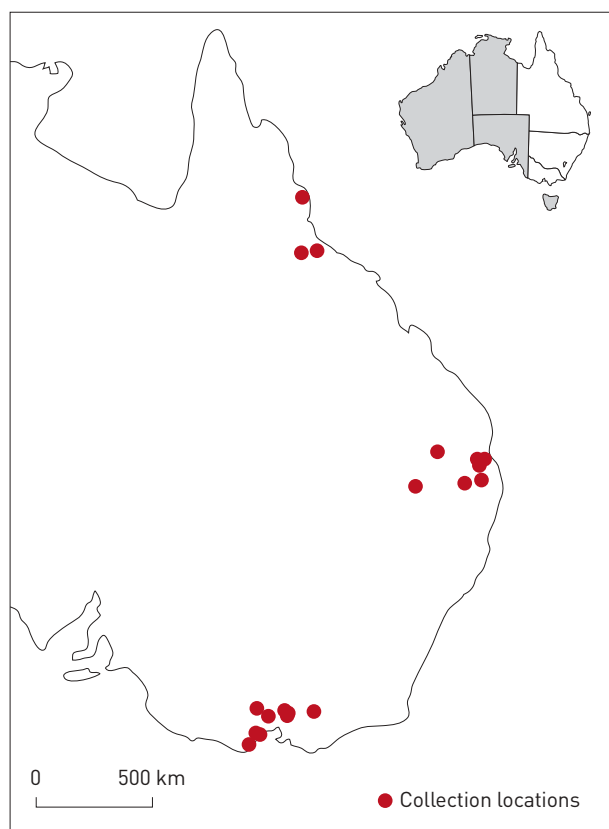


Figure 1. Locations of sites where invertebrates were collected for this study.

southern Murray-Darling Basin (MDB) (Goulburn, Broken, Loddon and Campaspe River Catchments in central Victoria) and south-west Victoria (Barwon River Catchment). In Queensland, the regions assessed were the south-east Queensland (Brisbane and Logan-Albert River Catchments), northern MDB (Condamine River Catchment), the dry tropics (Burdekin River Catchment) and the wet tropics (Mulgrave-Russell River Catchments).

When the same macroinvertebrate species was assessed from different sites — within a region, across regions, and even across Victoria and Queensland — there were no detectable differences in 72h LC_{50} values between locations (Kefford et al. 2003, 2006b; Dunlop et al. in press). The salinity tolerances of a given species appeared to be reasonably independent of the background salinity, other aspects of water chemistry and other environmental factors. This result can be extended to Tasmania — Allen (2006) measured 72h LC_{50} values of seven macroinvertebrates in Tasmania, and found that they had values similar to related species recorded by our studies in Victoria and Queensland (Kefford et al. 2003, 2006b; Dunlop et al. in press).

This means that in terms of acute salinity tolerance it would appear likely that in most cases a species will have little variation in salinity lethal tolerance across

eastern Australia and, while there are some differences and exceptions, related species (i.e. from the same genus, family and to some extent order) tend to have similar 72h LC_{50} values.

Although the SSDs were not significantly different between Victoria and Queensland, there were some minor differences in the SSDs between regions within Queensland (Dunlop et al. in press). Macroinvertebrates from the dry tropics and northern MDB were more tolerant to salt than those from south-east Queensland and the wet tropics. The differences in SSDs between these localities are consistent with regional differences observed in the values of a Salinity Index (SI) of Horrigan et al. (2005). The SI is based on the observed distribution of macroinvertebrate families in the field and indicates the average salinity sensitivity of the macroinvertebrate community (see Table 1). The slight differences in SSDs between regions were not caused by related taxa differing in their tolerance, but by differing proportions of taxa with differing salinity tolerance between regions. That is, although a species of a particular family or order tended to have a similar 72h LC_{50} value across eastern Australia and Tasmania (Allen 2006), the number of species found (or richness) of particular families and orders varied between different regions. As some families and orders tend to have difference salinity sensitivity, the community-level SSD from different regions can thus vary (Kefford et al. 2005). A possible explanation for these variations may be that communities having greater salt tolerance were associated with a greater ephemeral flows whilst those slightly more sensitive were associated with more regularly flowing streams (Dunlop et al. in press).

Taxa indicative of salinity levels

Most biomonitoring programmes in Australia routinely identify macroinvertebrates to the family rather than the species level. Within families, and even at the order level, there was limited variation in 72h LC_{50} values (Kefford et al. 2003, 2005, 2006b; Dunlop et al. in press), implying that these broad-scale monitoring programmes can be used to assess likely sensitivity to salinity. Based on their probability of occurrence at sites of different salinity across Queensland, Horrigan et al. (2005) assigned salinity sensitivity scores (SSS) to widely occurring macroinvertebrate families. The average SSS of all families observed at a given location can be calculated to produce a salinity index (SI) to work out whether the macroinvertebrate community at this site is experiencing salinity stress (Horrigan et al. 2005). The SSS of families have been shown to be correlated with the mean 72h LC_{50} value for that family (Horrigan et

al. 2007). Interestingly, the best correlation between laboratory and field tolerance results was not for the spot reading of salinity, but for the mean salinity determined over several visits (Horrigan et al. 2005, 2007). This is likely to be due to a lag effect of past salinity affecting the macroinvertebrates present.

Taking into account both laboratory and field results, a generalised sensitivity score has been proposed (Table 1). As with the original SSS, the generalised scores are from 1 to 10, with 1 = very tolerant (to salinity), 5 = tolerant and 10 = sensitive. The scores correspond to the widely used SIGNAL grades (Chessman 2003).

The salinity tolerance of different aquatic organisms

The variation in sensitivity within the groups that inhabit freshwater ecosystems — vertebrates (e.g. fish, amphibians, reptiles, birds, and mammals), invertebrates (e.g. insects, crustaceans, rotifers and snails), higher plants, algae, fungi and other microbes — is not widely known. However, it is possible that different groups of organisms will have different sensitivities to salinity. Unfortunately, there is limited salinity sensitivity information available and safety factors have not been developed for most groups. There are also ethical restrictions in most Australian states on the use of vertebrates (including fish) in experiments, and this complicates the task of collecting data from vertebrate species.

An important part of developing reliable ecosystem protection trigger values is the use of a safety factor. These are used to adjust laboratory derived toxicity data to provide greater certainty that the long term sustainability of populations will be protected when they are extrapolated for use in natural environment. Safety factors have not been developed for most groups.

As they are commonly used for biomonitoring in Australia (and overseas), most of the studies to date have focussed on macroinvertebrates. They also form an important component of aquatic food webs occupying all tropic levels occupied by animals (i.e. herbivores, detritivores and predators). There is now a substantial body of knowledge on the acute tolerance of macro-

Table 1. Suggested generalised salinity sensitivity scores for frequently occurring macroinvertebrates families in Queensland. Where a family is not listed but Horrigan et al. (2005) gives it a SSS, then these values should be used. Source Horrigan et al. (2007).

Family	Order	Suggested generalised score for sensitivity to salinity (1= tolerant, 10 = sensitive)
All families of Acarina	Acarina	5
Atyidae	Decapoda	5
Baetidae	Ephemeroptera	10
Belostomatidae	Hemiptera	1
Caenidae	Ephemeroptera	10
Calamoceratidae	Trichoptera	10
Cirolanidae	Isopoda	1
Coenagrionidae	Odonata	1
Corbiculidae	Bivalvia	5
Corixidae	Hemiptera	5
Culicidae	Diptera	5
Diphlebiidae	Odonata	5
Dytiscidae	Coleoptera	1
Gomphidae	Odonata	5
Helicopsychidae	Trichoptera	10
Hemicorduliidae	Odonata	5
Hydrobiidae	Gastropoda	5
Hydrochidae	Coleoptera	1
Hydrophilidae	Coleoptera	1
Hydropsychidae	Trichoptera	10
Leptoceridae	Trichoptera	5
Leptophlebiidae	Ephemeroptera	10
Libellulidae	Odonata	5
Naucoridae	Hemiptera	1
Notonectidae	Hemiptera	10
Palaemonidae	Decapoda	5
Parastacidae	Decapoda	1
Philopotamidae	Trichoptera	10
Physidae	Gastropoda	5
Planorbidae	Gastropoda	5
Pleidae	Hemiptera	5
Psephenidae	Coleoptera	5
Sciomyzidae	Diptera	10
Sphaeromatidae	Isopoda	1
Tabanidae	Diptera	1
Thiaridae	Gastropoda	1



Turtle (left). Algal toxicity experiment (right). Photos Jason Dunlop.

invertebrates to marine salts (e.g. Kefford et al. 2003, 2005a, 2006b; Dunlop et al. in press) and information on their occurrence in nature with respect to salinity has been compiled (Kay et al. 2001, Bailey et al. 2002, Kefford et al. 2004a, Horrigan et al. 2007) with further information available in databases held by various state and territory government agencies.

The acute salinity tolerance of macroinvertebrates is highly variable, ranging from taxa that are salt sensitive to those that are tolerant (e.g. Kefford et al. 2003, 2006b; Dunlop et al. in press). The data available demonstrates that sensitive macroinvertebrate species have similar or greater acute lethal sensitivity to those observed in the more sensitive tested species of other groups. This means that their protection should ensure the protection of other biotic groups and hence, overall biodiversity, community and ecosystem health and services. However, as many taxa within each biotic group have not been examined it is important to further test this hypothesis as more data becomes available.

Aquatic birds, mammals and reptiles are unlikely to be directly affected by increases in salinity (in the freshwater to slightly saline range) but they may be indirectly affected if their prey, habitat, etc. are affected (Hart et al. 1991) and thus the protection of other biotic groups from increased salinity should protect them. Less is known about the salinity tolerance of frogs and especially of their eggs and tadpoles, but what evidence there is suggests that the species studied to date are not particularly salt sensitive (e.g. Chinathamby et al. 2006). While further investigations on the effects of salinity on amphibians should continue, it is likely that protecting sensitive macroinvertebrates will also protect frogs.

The gametes and the eggs of freshwater fish can be considerably more salinity sensitive than adult fish. Given that the salinity of inland waters varies temporally, the implications for freshwater fish populations will depend on whether reproductive periods coincide with high salinity. In southern Australia many native fish breed in the warmer months of the year (see Allen et al. 2002) which means there is the potential for spawning to occur during periods of seasonal high salinity.

Our data also shows that adult microinvertebrates and sensitive macroinvertebrates have similar sensitivities to the early life-stages of sensitive freshwater fish (see James et al. 2003). Therefore, protecting freshwater invertebrates from salinity changes will also protect freshwater fish. In addition, changes in macroinvertebrate species richness across multiple samples (Kefford et al. 2006b, unpublished data) occur at even lower salinity concentrations than those that appear to affect any stage of freshwater fish.



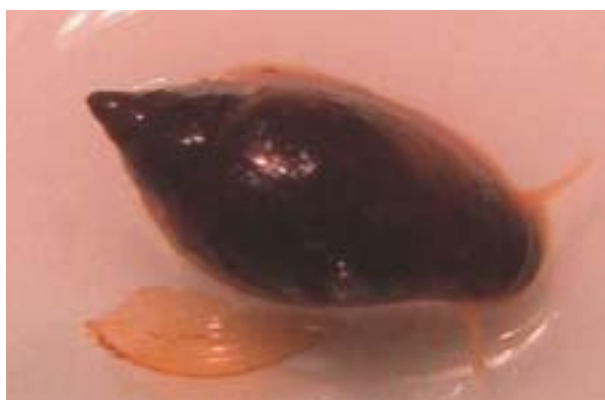
The salt tolerant bug *Austrochiltonia* sp. Photo Colin Clay.



Many soft-bodied non-arthropods like this flat-worm appear to be salt sensitive. Photo Colin Clay.



Physidae snail (left). Isopod (right). Photos Jason Dunlop.



The freshwater snail *Glyptophysa* sp. Photo Colin Clay.



The introduced freshwater snail *Physa acuta*. Photo Colin Clay.

The available evidence suggests that protecting macroinvertebrate communities from rising salinity will protect other freshwater fauna. Until such time that more data is available for other groups, we recommend that water quality guidelines and risk assessments in Australian freshwaters be based on protecting macroinvertebrate communities. This is consistent with national and state programmes that widely use macroinvertebrates as a surrogate indicator of aquatic ecosystem health.

Across all freshwater invertebrates found in rivers, the acute (≤ 96 h) LC_{50} values of mature older stages of the most salinity sensitive *microinvertebrates* are about the same as for the most sensitive *macroinvertebrate* tested (Kefford et al. unpublished data). However, the most tolerant *microinvertebrate* tested had an acute LC_{50} value less than half that of the most tolerant *macroinvertebrate* (i.e. they were 50% more sensitive). Therefore, toxicological results suggest that when risk assessments and salinity targets aim to protect some percentage of *macroinvertebrate* species, e.g. 95%, this percentage will somewhat overestimate the percentage of *microinvertebrates* protected, i.e. less than 95% of *microinvertebrates* will be protected.

Reductions in riverine *macroinvertebrate* species richness across multiple samples appear to occur at salinities greater than about 0.3–0.5 mS/cm (Kefford et al. 2006, unpublished). Ephemeroptera, Plecoptera and Trichoptera (EPT) species richness from multiple

samples is reduced at even lower salinity, above about 0.2 mS/cm, and changes in community structure also present at very low salinity levels (Kefford et al. unpublished data). The relationship between *microinvertebrate* species richness in multiple samples and salinity has not been investigated. Mesocosm experiments suggest some freshwater *microinvertebrate* (and aquatic plant) communities will be adversely affected by salinity > 1 g/L (about 1.6 mS/cm) (Nielsen et al. 2003, in press; Brock et al. 2005). While more studies on *microinvertebrates* are needed, especially at salinities < 1 g/L, the available evidence suggests that protecting salt sensitive *macroinvertebrate* community structure and richness across multiple samples should protect most *microinvertebrates*. With the information now available we suggest that, except where management action is aimed at protecting a particular iconic species, salinity risk assessment be based on protecting freshwater invertebrates.

From the limited data available, it appears that protecting *macroinvertebrate* community structure should also provide protection for fish, amphibians and other groups of freshwater organisms.

Early life-stages

Younger life-stages can be more salinity sensitive than their mature life-stages (Hart et al. 1991). This appears to be especially so for freshwater fish where, prior to

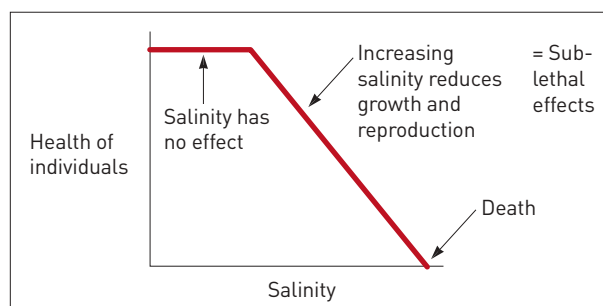
water hardening, their eggs can be much more salt sensitive than their adults (Chotipuntu 2003, James et al. 2003). Likewise, the sperm of Carp has been shown to be salt sensitive relative to adult fish (Whiterod 2001). However, the sensitivity of these stages of fish would appear to be no more sensitive than the comparable life-stages of sensitive invertebrate species.

While the eggs or hatchlings of some species of macroinvertebrates are more salinity sensitive than their mature aquatic life-stages, other species show little or no difference between life-stages (Kefford et al. 2004b in press). With the exception of the North American species, *Palaemonetes kadiakensis* (Palaemonidae) (Hubschman 1975), there appears to be little difference in the salinity tolerance of hatchlings and adults of decapod crustaceans (shrimps, crayfish, crabs, etc.) (Kefford et al. in press). It was found that eggs of “land-locked” freshwater decapods with relatively large eggs (e.g. *Caridina* and *Euastacus*) may be more sensitive than their adults. The tolerances of eggs and young of gastropods, other insects and mites were between 20%, about the same as their acute LC₅₀ value for the mature life-stage (Kefford et al. in prep). In some but not all species (two snails and one water bug) eggs took longer to hatch at sub-lethal high salinity (Kefford et al. in prep.). In conclusion, within macroinvertebrates there appears to be a diversity of relationships between the salinity tolerance of younger and older life-stages. As most sensitivity data is from acute exposure (and the endpoint is mortality), often acute to chronic ratios (ACRs) are used to convert acute data to account for long term (and sub-lethal) effects of contaminants. This work suggests that there are no consistent trends in the ratios of salinity sensitivity between acute tests on mature aquatic life stages and tests using early life stages. Further research is required to investigate these relationships to allow accurate assessments of salinity risk of the early life stages.

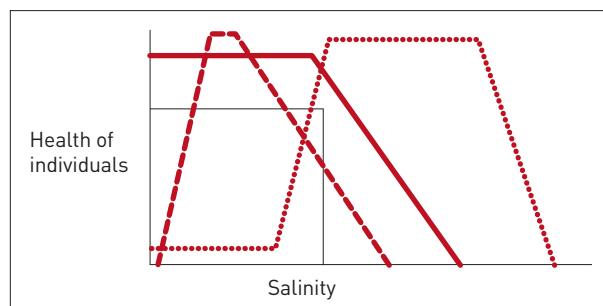
Low salinity and sub-lethal effects of salinity

The standard assumption is that as salinity increases, freshwater animals will at first be unaffected, but as salinity levels continue to rise a threshold will be crossed and they will begin to experience sub-lethal effects, such as reductions in growth and reproduction (Hart et al. 1991). As salinity increases further these sub-lethal effects will increase in magnitude and, if salinity is high enough, death will result. We call such a response a ‘threshold response’ (Figure 2a).

Our results have shown that this response to salinity is not universal. The freshwater snail *Physa acuta* had maximum growth and reproduction at intermediate salinity (0.5–1 mS/cm) both of which were reduced at



a) The usual assumption.



b) Revised model.

Figure 2. Hypothetical models of how salinity may affect individual freshwater animals. The different lines in b represent different species.

lower and higher salinities (Kefford & Nuggeoda 2005). Furthermore, similar inverted U-shaped responses have been observed in other freshwater snails (Duncan 1966, Jacobsen & Forbes 1997), a cladoceran (Yang & He 1997) fish (see review Boeuf & Payan 2001) and the hatching of some microinvertebrate from dormant eggs (Nielsen et al. 2003, Brock et al. 2005). The implications of an inverted U-shaped response is that if the salinity of very low salinity water increases, there will be biological consequences. These results appear to be in agreement with recent analysis of field data showing that as salinity increases from < 0.05 mS/cm to around 0.3 mS/cm there is an associated increased macroinvertebrate species richness in multiple samples across Victoria and South Australia (Kefford et al. unpublished data). Such an increase would not have been predicted if all species had a threshold response. However, it is also possible that the increase in species richness in multiple samples is caused by indirect effects of salinity (see Rohr et al. 2006).

We have, however, observed a variety of differing responses that vary according to the species investigated (Figure 2b). The damselfly *Ischnura heterosticta* had lower growth at salinity from 0.1–1 mS/cm than from 5–20 mS/cm with an apparent salinity threshold somewhere between 1 and 5 mS/cm (Kefford et al. 2006b). The baetid mayfly *Centroptilum* sp. had no detectable difference in growth and appeared to be unaffected sub-lethally by salinity (Hassell et al. 2006). There are

many sub-lethal end-points that may be used, and only growth was measured, it is not known whether the same conclusion would be reached for *Centroptilum* sp. if other sub-lethal responses had been measured. For some species, even different endpoints had different salinity response curves. For example, *Chironomus* sp. had maximum emergence as flying adults at intermediate salinity (2.5 mS/cm) i.e. an inverted 'U' response, while a threshold response was observed in terms of the number of days to emergence and their growth rate was maximum at the lowest salinity, subsequently decreasing without an apparent threshold (Hassell et al. 2006).

Inconsistent salinity concentration response relationships between different species (Figure 2b) and, for some species different relationships between responses (endpoints) observed, makes it unwise to calculate and then apply ACRs using the existing data to transform 72h LC₅₀ values so that they incorporate chronic sub-lethal effects. Likewise, the use of an arbitrary factor, e.g. 10, is also likely to be inaccurate. Instead, we (Kefford 2006b, unpublished) have calculated safety factors that are based on loss of species with increasing salinity in nature.

Reductions in salinity

We have shown that a sudden drop in salinity can be lethal to freshwater macroinvertebrates when it occurs after exposure to salinity concentrations just below their lethal salinity tolerance (Rutherford & Kefford, unpublished data). *Paratya australiensis* are affected by a sudden drop in salinity when the initial rise in salinity is rapid, whereas when salinity rises gradually over several days, individuals of this species showed no mortality. This suggests that, for this species at least, a rapid rise in salinity damages their ability to osmoregulate and extract ions from freshwater.

Is all salinity the same?

Salinity refers to the total concentrations of all dissolved inorganic ions. Different mixtures of different inorganic ions (ionic composition) with the same salinity may have different biological effects, and changes in salinity combined with other changes in water chemistry may alter the effect of salinity. Australian freshwaters (and freshwaters generally) tend to have different ionic compositions from those found in saline waters. Concentrations of individual ions in freshwater will be much lower than in saline waters. This means that when fresh and saline waters are mixed, the ionic composition of the mixed water will be approximately proportional with that found in the saline water. Some ionic



Rajesh Prasad picking bugs from sample sites. Photo Ben Kefford.

proportions of saline waters do affect acute lethal salinity tolerance (e.g. Kefford et al. 2004c, Mount et al. 1997, Dunlop et al. unpublished data).

The common ionic proportions in saline waters of south-east Australia (Radke et al. 2002) and south-west Australia (Pinder et al. 2005), which are sodium chloride dominated, would appear to have broadly similar effects on acute mortality to those observed using marine salts (Zalizniak et al. 2006, unpublished data). This similarity in toxicity was observed using acute mortality as an end-point. However, waters with low calcium concentrations were more detrimental (Zalizniak et al. 2006, unpublished data) in terms of sub-lethal effects.

There would appear to be greater variation in ionic composition of surface waters in Queensland than in southern Australia, with the variation associated with a particular soil type and geology (see McNeil & Cox 2000). Those having increased concentrations of magnesium were found to be more toxic than those with lower magnesium concentrations. Subsequent testing to isolate the effects of specific salts indicated that toxicity was reduced when magnesium sulphate (MgSO₄) was removed (Dunlop et al. unpublished data).

The pH of saline waters can vary widely, and it was thought that the combined effect of elevated salinity and acidic (low pH) water would create greater osmoregulatory stress for freshwater organisms. However, subsequent experiments showed that low pH had no



Rajesh Prasad and Nelli Horrigan in the Queensland Department of Natural Resources and Water's aquarium. Photo Ben Kefford.

effect on sub-lethal or acute lethal salinity tolerance using marine salts as test media (Zalizniak et al. unpublished data). High alkalinity (high pH) was observed to increase the sub-lethal effects of salinity on one of two species in which sub-lethal effects were studied.

We have also observed that water temperature had some relatively minor, but detectable, effects on the acute lethal salinity tolerance of three microinvertebrate species (Kefford et al. unpublished data). Along with the scientific literature (e.g. Dorgelo 1974, Aladin & Potts 1995), our results suggest that (within an organism's normal temperature range) temperature has a relatively minor effect on acute lethal salinity tolerance. The effect of temperature on sub-lethal salinity tolerance requires further study.

Validation of laboratory results and safety factors

As discussed in the background section, there are several reasons why it is difficult to establish salinity thresholds for the protection of freshwater biodiversity. While they have a number of advantages, laboratory experiments may not accurately predict the response of organisms in nature because the laboratory environment is a simplification of the natural world (Kefford et al. 2004a, 2006b; Horrigan et al. 2007). Furthermore, we know that salinity produces chronic and sub-lethal effects and can have greater effect on eggs and young life-stages

than mature life-stages (James et al. 2003, Kefford et al. 2004b, in press), it is also feasible that salinity has indirect effects (see Rohr et al. 2006). Therefore, 72h LC_{50} values are unlikely to predict the full effect of salinity in the natural world. This means there is a need to develop an appropriate safety factor to account for long term community level effects of salinity that cannot be predicted by 72h LC_{50} values. In our investigations we validated laboratory results against field data (Kefford et al. 2004a, 2006b, unpublished data; Horrigan et al. 2007), and developed safety factors so that 72h LC_{50} values could be transformed into salinity levels that protect species in nature (Kefford et al. 2006b, unpublished data).

Acute lethal salinity tolerance appears to reflect the maximum salt concentrations at which taxa have been observed in the field. The 72h LC_{50} values of macroinvertebrate and fish species (Kefford et al. 2004a) and the mean value of macroinvertebrate families (Horrigan et al. 2007) is correlated with the maximum salinity at which they have been observed in nature. Furthermore, this correlation is stronger when the mean salinity at a site recorded over several visits is used rather than spot salinity readings (Horrigan et al. 2007). Likewise, there is a good correlation between the mean salinity at the site, the highest salinity which regularly supported a species, and the 72h LC_{50} value for salinity for that species (Kefford et al. 2004a). While 72h LC_{50} values are unlikely to be indicative of the salinity at which a

taxon can maintain its population (which will be influenced both by the maximum salinity but also the salinity history) our results show that 72h LC₅₀ values do give some indication of salinity distribution in the field.

A species sensitivity distribution (SSD) for salinity can be used to predict the salinity at which a given proportion of a community will be lost. Many factors other than salinity, can affect the number of species present (or species richness) in a sampling event (a sample). Consequently a SSD does not predict the species richness of a sample. Instead a SSD for salinity can predict the loss of the pool of species that may be present at a site with a particular salinity. We found that applying either of two simple mathematical safety factors to transform 72h LC₅₀ values into lower values to account for unmeasured sub-lethal, chronic and indirect effects, meant that an SSD could be derived which accounted for the loss of species in multiple samples as salinity increases across Victoria (Kefford et al. 2006b). Subsequent analysis using macroinvertebrate data from both Victoria and South Australia has confirmed this conclusion (Kefford et al. unpublished data).

We acknowledge that more research is needed on the appropriate safety factors required to transform 72h LC₅₀ values to account for unmeasured chronic, sub-lethal and indirect effects of salinity, and also to account for higher sensitivity of early life stages. The safety factors which we have developed are based on correlating the fall in species richness in multiple samples as salinity increases. It does not appear that a generalised reduction in water quality with increasing salinity is confounding this correlation (Kefford et al. unpublished data). While the available chronic and sub-lethal salinity tolerance data does not conflict with the safety factors proposed (Kefford et al. 2006b, unpublished data), this data is very limited and at present cannot provide a good test of the appropriateness of these safety factors. More chronic and sub-lethal data, as well as some data on indirect effects should be collected to test these safety factors. However, in the interim we recommend that these provisional safety factors be used.

Management implications

The studies referred to in this chapter have provided a significant advance in knowledge about salinity effects on aquatic organisms, particularly macroinvertebrates. In the past, default guidelines were derived using broad regional reference ranges (i.e. salinity guidelines derived based on background salinity concentrations) and/or

correlating biological changes in nature to salinity. For example, Hart et al. (2003) is based on effects data from the maximum salinity at which each species has been observed, this does not necessarily give a causal relationship between salinity and its occurrence. Muschal (2006) used effects data but included acute and chronic data from 14 and 10 species, respectively.

Sensitivity data for macroinvertebrate taxa collected in one location is transferable to other regions in eastern Australia. There is now sufficient information for managers to derive trigger/guideline values for salinity concentrations to protect salinity-sensitive species based on causal links between exposure and biological effects. Deriving this scientific basis for natural resource management target setting and management of salinity is an important step. In addition, the work reported here has a range of other outcomes that will affect the way salinity impacts are managed in a variety of ecosystem types, and highlights the need to manage factors co-occurring with salinisation to provide adequate protection of ecosystems exposed to impacts from salinisation.

While the focus of the investigations discussed here has been on the effects of salinity changes in fresh, or only slightly saline waters, it should not be forgotten that many inland water bodies are naturally saline (from primary salinisation) and contain important biodiversity values in their own right. The protection of these values is as important as those in freshwater systems (Williams 1993), as the biota endemic to these systems can tolerate high salinity but may not tolerate a wide range of anthropogenic disturbances.

It is not always feasible to prevent freshwater systems from becoming saline. Saline waterways can still contain important biodiversity values and, even when a waterway has become saline, needs to be protected (see following chapter). Changes in salinity often co-occur with changes in habitat (e.g. Lymbery et al. 2003), hydrology and/or other aspects of water quality (e.g. Kefford 1998) and these other changes may have their own effect on aquatic organisms or alter the effects of salinity. Managing for salinity in inland waters will also mean managing for these other changes (Kefford 2000).

Where should salinity sensitivity data come from?

A key requirement in conducting a risk assessment or developing water quality guidelines for the impacts of salinity on freshwater biodiversity is finding toxicity data that are relevant to local waterways (ANZECC/ARMCANZ 2000). Given the cost of collecting new data, it would be beneficial if data collected from one region can be used elsewhere.

Acute lethal sensitivity of many invertebrate species was observed to be similar across eastern Australia and so the sensitivity data reported here are likely to be valid for establishing ecosystem protection trigger values across much of eastern Australia. The trigger values should not be based on just one macroinvertebrate species, as there are differences in macroinvertebrate community compositions between regions and the communities in different regions can contain different numbers of species within specific salinity tolerance ranges. It is preferable that local species lists are used to determine which species are included in a local SSD. This study shows that the salinity sensitivity values for each species in a local list, based on data collected from different eastern Australian regions, can be used with some confidence (Dunlop et al. in press).

There is a much longer history of both primary (natural) and secondary (anthropogenic) salinisation in Western Australia than in eastern Australia (see Kay et al. 2001, Pinder et al. 2005) and some families of freshwater macroinvertebrates are found to inhabit much higher salinity waterbodies (e.g. Kay et al. 2001) than they do in eastern Australia. It is possible that the longer history of salinisation has resulted in genetic adaptation to higher salinity and/or the extinction of salt sensitive species in Western Australia. Consequently, salinity tolerance data from eastern Australia may overestimate the effect of salinity in Western Australia. Increased salinity of inland waters in the Western Australian wheat belt can co-occur with low pH and high trace metal concentrations. While pH would appear to have limited effects on salinity tolerance (Zalizniak et al. unpublished data), low pH and high metal concentrations can have severe impacts on biodiversity in their own right. Given the extent of the salinisation problem in Western Australia it is vital that salinity tolerance information be collected on freshwater macroinvertebrates from that state, and the combined effects of elevated salinity and metals and lowered pH be investigated. In the meantime, interim guidelines for Western Australia can be based on eastern Australian effects data, keeping in mind that they are likely to be conservative.

Biomonitoring and assessment

Although the electrical conductivity (EC) of water is relatively inexpensive to measure and can easily be continuously logged (unlike many other water quality parameters), EC data on its own is only a partial guide for river managers. There are still advantages to using taxa that are sensitive or tolerant of salinity to infer whether aquatic biodiversity (and hence community composition, ecological function and ecosystem

services) is being damaged. In particular, biological information provides a context and stronger basis for management intervention.

It is clear, however, that despite significant advances in our understanding of the biological effects associated with salinity impacts, the proposed salinity thresholds remain a work-in-progress and are likely to be modified as further data becomes available. Consequently, a monitoring programme based on the presence of species known to be sensitive, and the absence of species known to be tolerant of salinity, can provide direct evidence of salinity impacts. The salinity sensitivity scores (SSS, based on field distributions of macroinvertebrates), the generalised salinity scores (Table 1, based on both field distributions of macroinvertebrates and laboratory toxicity results) and a salinity index or SI (Horrigan et al. 2005, 2007) are useful screening tools to assess whether the macroinvertebrates of a water body are being impacted by high salinity concentrations. This information can be mapped to show localities where salinity is affecting biodiversity (Figure 3).

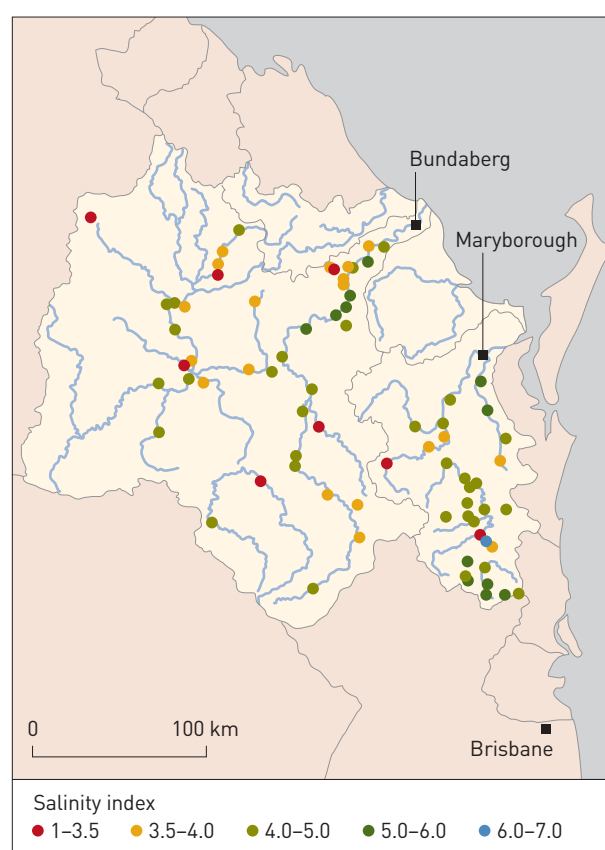


Figure 3. An example of using the salinity index (SI) to identify localities of potential salinity impact on biodiversity at the catchment scale. The higher the SI, the greater the proportion of the macroinvertebrate community that is represented by salinity sensitive families (i.e. the lower the risk of current salinity impact). Here the SI is calculated from SSS as per Horrigan et al. (2005).

Incorporating effects of sub-lethal low and high salinity

Low salinity can be more stressful to some animals than elevated salinity. Furthermore, there appears to be a diversity of sub-lethal effects associated with increasing concentrations of salinity. Few higher taxa (e.g. orders) have salinity tolerance data from more than one species, and it is not possible to identify whether there are any patterns in the types of responses between taxonomic and other groups or the prevalence of the different sub-lethal salinity responses. The fact that a species can have different salinity concentration response curves for different sub-lethal effects to salinity, contributes to the difficulty of determining the salinity at which their populations will be affected.

It was, therefore, not possible to confidently suggest a single safety factor that could be applied to 72h LC₅₀ values for all taxa to account for sub-lethal effects. Instead, we developed safety factors from a comparison of the loss of species predicted from a SSD (from 72h LC₅₀ values) to the actual loss of macroinvertebrate species observed (in multiple samples) as salinity increases.

Suggested stepwise protocol to derive thresholds for the protection of freshwater biodiversity

Before salinity thresholds for protection of freshwater biodiversity can be developed, it is necessary for stakeholders to decide what components of biodiversity they wish to protect. It would be possible, for example, to only protect certain iconic species or environmental assets and to then develop a risk assessment process around protecting only these species, their food chain and habitat. If, however, stakeholders wish to protect all freshwater biodiversity (and hence overall ecosystem health), we suggest that the following approach is suitable and updated as more data becomes available.

The steps described here are modified from the framework outlined in the national guidelines (ANZECC/ARMCANZ, 2000) and described in further detail in Dunlop et al. (2007).

1. **Consult stakeholders and determine environmental values and/or management aims and collaboratively determine environmental objectives.** If the management aim is to protect all freshwater biodiversity, then continue with the proposed approach.
2. **Define the water bodies or region of interest.**
3. **Classify the ecosystem and determine the level of protection required.** The ecosystem condition determines the effect size used in the development

and assessment of a water quality guideline. This determines what percentage of species are to be protected, e.g. 95%, and with what degree of confidence, e.g. 50% certainty.

4. **Determine the local natural or background salinity concentration.** Where a water body naturally has an elevated salinity, e.g. 2 mS/cm, it will support different macroinvertebrate communities to lower salinity waters, and consequently, the natural or initial salinity should be considered in guideline development and risk assessment. Decide on the 'natural' salinity in the region, or at specific localities prior to secondary salinisation, using a combination of historical records, paleolimnology, salinity values at reference sites, hydrological modelling and expert opinion.
5. **Make a species list of all macroinvertebrates known to occur in freshwater to slightly saline waters within the region of interest from existing sources and/or an additional study.** The assembled species list is based on current records and should be added to where more detailed local data is available. If the salinity of a region has already been elevated from secondary salinisation, this list should also include species that would likely be present if the salinity were at its 'natural' level. Wherever possible, it would be best to use a referential approach (i.e. compare with sites in reference or 'natural' condition). In such regions some degree of expert opinion and consideration of species lists from similar regions would likely be necessary.

In regions where macroinvertebrates are only identified to family level, a family list should be constructed and the relative species richness of each family estimated from expert opinion and consideration of similar regions. For example, if there were 100 species in the region, five would be in family A, two in family B, etc.

6. **Assign a 72h LC₅₀ value to each species on the list using sensitivity information from all available data from eastern Australia.** Where 72h LC₅₀ value of a species has been measured anywhere in eastern Australia, this value should be used. For species where no 72h LC₅₀ value is available, an estimated 72h LC₅₀ value should be used which should be based on the following (in decreasing order of preference).
 - a) 72h LC₅₀ values of related species.
 - b) For frequently occurring taxa and provided the presence/absence of the species is investigated at sites with elevated salinity (see Horrigan et al.

2007), the maximum salinity that a macro-invertebrate is recorded from in nature will be about the same as the 72h LC_{50} value (Kefford et al. 2004a).

- c) Salinity tolerance information from experiments other than 72h LC_{50} values.
- d) Expert opinion. We are currently (in conjunction with Samantha Low-Choy of the Department of Mathematics, Queensland University of Technology) developing a method in the Bayesian framework to facilitate estimating 72h LC_{50} values and their credibility values from a diverse range of input data.

For cases where there is no species list, but a family list and the relative richness of each family can be estimated, we suggest that 72h LC_{50} value for each family be estimated as the family mean 72h LC_{50} values in eastern Australia. Where there is no 72h LC_{50} value for a family, a value would be estimated using the weight of evidence approach described above.

7. **Apply a safety factor.** Acute LC_{50} values do not consider chronic, sub-lethal and indirect effects. We recommend that a safety factor be applied to transform the 72h LC_{50} value so that they account for these effects. The mathematical functions presented in Kefford et al. (2006b) can be used. As more data become available on: a) chronic, sub-lethal and indirect effects of salinity, and b) the relationships between these effects and the occurrence of species in the field with respect to salinity; the use of the above safety factors should be reviewed.
8. **Construct a SSD based on the 72h LC_{50} values after the application of the safety factor(s).** It is then mathematically simple to calculate the salinity at which any percentage of species will be lost with a given degree of confidence. The salinity for this region that would result in the pre-agreed (step 3) percentage loss of species, with the agreed confidence, can thus be calculated and become the target salinity. Any higher salinity is considered to produce unacceptable losses in freshwater biodiversity. Furthermore, if the salinity of a waterway is believed to be naturally elevated, e.g. 2 mS/cm, (step 4) it is a simple process to set this salinity as the base salinity and calculate how much salinity would have to increase (above this background level) before the pre-agreed percentage of species would be lost with the pre-agreed level of confidence.

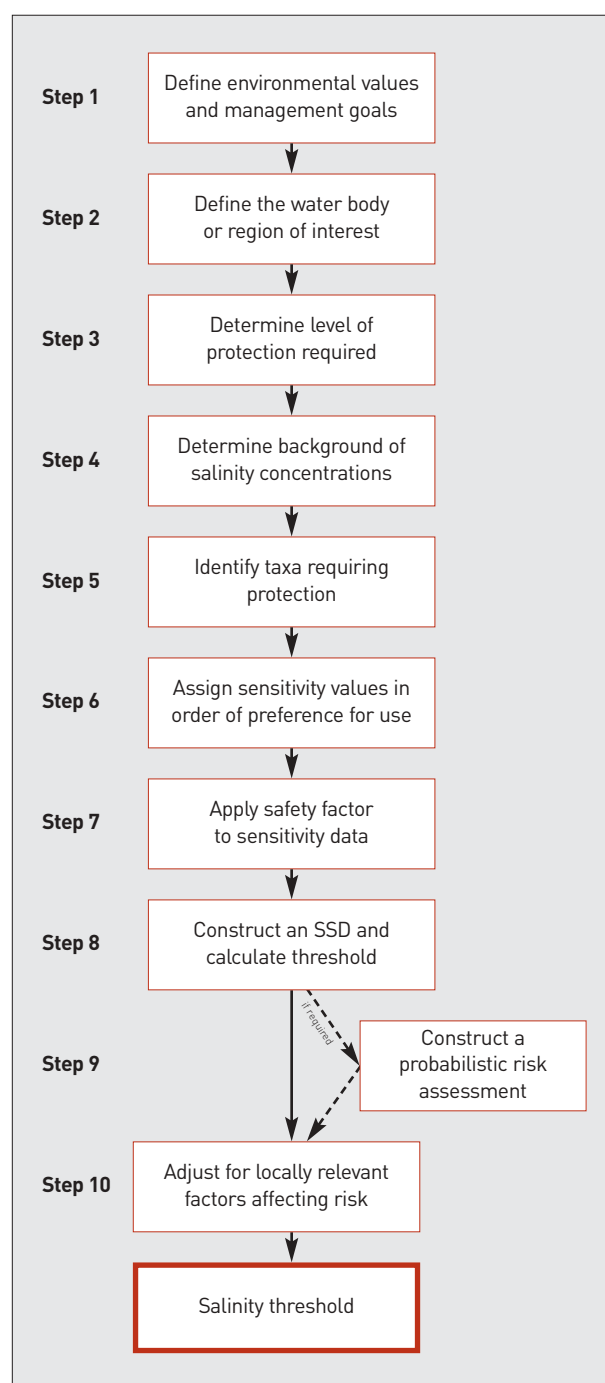


Figure 4. The 10 step protocol framework.

9. **Conduct a probabilistic risk assessment, if required.** For this step the above SSD is combined with a statistical distribution of the salinity concentration measured in the region (or the modelled distribution of salinity under various scenarios) and the risk of various percentage loss of species calculated.
10. **Adjust the threshold values, if required, to take account of local factors that may affect salinity risk (e.g. presence of Mg salts).**



The Barwon River in south-west Victoria. Photo Ben Kefford.

Where impacts of rising salinity have already occurred it does not necessarily follow that a water body has no biodiversity or other values. These values may still be retained even when salinity rises further. In these cases, the biodiversity values should be assessed within a regional context using the above protocol, together with the consequences of further diminishment of these values if salinity were to rise further.

Factors which modify the protocol

The effect of variable salinity

Most laboratory studies of salinity tolerance involve the exposure of organisms to salinity treatments that do not vary with time. However, in rivers, and to a lesser extent in wetlands, salinity is never constant. Although little is known about the effects of variable salinity exposure concentrations on freshwater organisms, it is known that a sudden rise in salinity imposes greater stress than a gradual rise to reach the same salinity concentration. Also, a sudden fall in salinity can be lethal to freshwater invertebrates that have been exposed to near lethal salinity concentrations.

Other things being equal, shorter durations of salinity exposure clearly have less effect than longer duration exposures at the same salinity concentration. For example, Nielsen et al. (in press) showed that constant salinity exposure reduced hatching from wetland plant seed and microinvertebrate egg banks, yet a single 14 day pulse of salinity of the same concentrations had little

or no negative effects. Based on the observations that sudden rises and falls in salinity are particularly stressful, it would appear reasonable that multiple pulses of elevated salinity with lower salinity 'rests' in-between, would be especially damaging. This is supported by an experiment by Marshall and Bailey (2004) who found that four pulses for durations from 4 to 13 hours of saline water over 5 days had more effect on macroinvertebrates than the same load of salt delivered continuously over 5 days.

On present knowledge, we suggest that rapid rises and falls in salinity, multiple pulses of salinity, and high peak in salinity concentrations will be damaging to freshwater biodiversity. It is therefore important that management actions should minimise these salinity transients by gradually disposing of saline water and gradually increasing or decreasing the start and cessation of environmental flows of freshwater. If sudden changes or pulses in salinity are unavoidable, then the calculated guideline or trigger values should be applied conservatively to provide greater protection of biodiversity.

Variation in the ionic proportions of saline waters

Most Australian experimental studies on the effect of salinity on freshwater organisms have considered the effect of salinity with an ionic proportion similar to sea water (which is sodium chloride dominated), as these proportions predominate in southern Australia. Ionic proportions of sodium chloride dominated saline waters vary with three common saline water types (Radke et al. 2002). Acute lethal effects of salinity are largely identical across these three saline water types (Zalizniak et al. 2006, unpublished data). However, sub-lethal effects of salinity in two of these three water types with reduced calcium concentrations will be more detrimental to invertebrates than from saline water with ionic proportions similar to sea water. Consequently water quality guidelines and risk assessments should be more conservative when dealing with changes in salinity with these ionic proportions with low calcium concentrations.

There are a few saline waters in southern Australia that are not sodium chloride dominated (see Williams 1981), and there seems to be more variation in ionic proportions in Queensland than in southern Australia (see McNeil & Cox 2000). Increasing magnesium concentration would appear to increase salinity toxicity in Queensland waters with variable ionic proportions (Dunlop et al. unpublished data). A factor can be used to adjust the toxicity values derived using marine salts to account for variation in ionic proportions according

to four commonly found water types in Queensland (Dunlop et al. 2007). However, as there are a number of possible variations in ionic proportions in Australia, it is difficult to know what the effects of salinity will be for other waters types that occur. This gap in knowledge means that we recommend site specific studies, such as Direct Toxicity Assessment, to determine the effects of non-standard ionic compositions.

Salinity in combination with changes in pH

Changes in pH can affect freshwater biodiversity and also alter the bioavailability of a number of contaminants, e.g. trace metals. Our results suggest low (acidic) pH will minimally alter the direct effect of salinity provided the ionic composition is similar to sea water. Although it has not been investigated, low pH may increase the sub-lethal effect of salinity when calcium concentrations are low, especially for animals that build shells of calcium carbonate, e.g. molluscs. Except in the case of saline waters with low calcium levels, the effect of salinity in combination with changes in lowered pH (more acidic) will likely be no worse than the effect of salinity plus the effect of pH. High (alkaline) pH may, however, result in an increase in the effect of salinity on some species. Consequently, a conservative approach should be adopted based on the assumption that alkalinity will magnify the effects of increases in salinity.

Salinity in combination with changes in other aspects of water chemistry

There is a wide range of water quality factors that may co-vary with changes in salinity and affect aquatic organisms, independent of any effect of salinity. While there is little information on the effect of most water quality factors on the toxicity of salinity, there is more information on the effect of salinity (within a species tolerance) on the toxicity of a range of chemicals. In a review of the effect of salinity on the toxicity of a range of inorganic and organic chemicals to aquatic organisms, Hall and Anderson (1995) concluded that the toxicity of most metals decreased with increasing salinity. This is most likely due to chemical interactions between the metals and salt ions, rendering the metals less biologically available to the organisms and thus decreasing their toxicity. In contrast, increasing salinity tended to make organophosphate insecticides more toxic (Hall & Anderson 1995).

Preliminary results from a study into the effect of salinity combined with one of three pesticides on a species of green algae, *Pseudokirchneriella subcapitata* suggest the combined effects of pesticides and salinity can be very complex (Dassanayake et al. 2005).



Sampling site in the Condamine River, Qld.

In managing the impacts of salinity in combination with other changes in water chemistry, the total impact should ideally be considered. Unfortunately, except in the case where salinity impacts are within the tolerance of the species of concern and the predominant impact is from the other chemicals, Hall and Anderson (1995) and most other studies in the ecotoxicological literature, do not provide a quantitative basis for the inclusion of combined effects in a salinity risk assessment. Given the current paucity of information we can only recommend that toxicity guidelines developed for the mixtures of chemicals other than salinity (Warne 2003) be followed. Furthermore, we advocate that a precautionary approach be adopted and that, in the absence of information to the contrary, it be assumed that combined effect of changes in salinity and other changes in water quality may be greater than the sum of their individual effects.

Known unknowns for which future research would improve management

While considerable progress has been made on understanding and predicting the impact of increasing salinity on freshwater biodiversity, the long term effects are uncertain. Further research into the following topics would improve the reliability and accuracy of predicting impacts from the salinisation of freshwaters.

Testing of interim safety factors

The recommended approach for setting water quality guidelines for salinity and assessing the risk of increasing salinity depends on interim safety factors. If these safety factors do not accurately account for effects of salinity that 72h LC₅₀ values do not measure (i.e. chronic, sub-lethal and indirect effects), the resultant guidelines and assessments risk will also be inaccurate. It should be a high priority to test the interim safety factors.

Use of data from eastern Australia in south-western Australia

Because of the rarity of freshwater bodies in south-western Australia it is especially important to protect their biodiversity from salinity (and other stressors). The long period over which primary (natural) and secondary (anthropogenic) salinisation has occurred in south-western Australia, may have led to the evolution of greater salinity tolerance in this region. This possibility is supported by the occurrence of specific macro-invertebrate families in south-western Australia (Kay et al. 2001) at salinities both considerably higher than they occur in eastern Australia (Kefford et al 2004a, Horrigan et al. 2007) and with higher than 72h LC₅₀ values for members of the same family in eastern Australia (Kefford et al. 2003, 2006; Dunlop et al. in press). There is an urgent need to assess the salinity tolerance of freshwater organisms from this region and compare their tolerance to related taxa in eastern Australia. This could be cost effectively done with macroinvertebrates using the same methods applied in eastern Australia to allow direct comparisons of salinity sensitivity between regions.

Effect of temporally variable salinity exposure

Effects on freshwater organisms would appear to be determined by not only the magnitude of increase in salinity but also the duration(s) of exposure and rates of change of both rises and falls in salinity. While we believe that gradual rises and falls in salinity will be less damaging than rapid salinity changes, we do not know if there are any thresholds for the rate at which salinity changes. Likewise, shorter durations of salinity exposure will be less damaging than longer durations of exposure at the same salinity, but the relative effects of a given load of salt are less clear as it travels through an aquatic system over different durations. Because rivers and wetlands experience all these variations, it is not possible to predict accurately the effect of salinity in nature. These same uncertainties also pose a problem for the many agricultural, mining and energy projects that produce

saline water that requires disposal into freshwater ecosystems, either under emergency or controlled release scenarios. Understanding the effect of temporally varying salinity on freshwater organisms is central to more accurately assessing the ecological risk to aquatic organisms in natural waterbodies, and hence, to preparing sound guidelines for management.

The combined effect of salinity in combination with other changes

The effects of salinity should be managed in conjunction with other changes in water quality, hydrology and habitat associated with degradation of land and water resources, because changes in salinity often co-occur with other environmental changes that may both have effects independent of salinity, and also alter the effect of salinity. The overall environmental assets and/or values to be protected, and their associated environmental objectives can be more effectively dealt with in this way, particularly when synergistic or antagonistic effects are suspected. As there is limited understanding of these combined effects, the national framework for determining guidelines for the toxicity of mixtures of chemicals should be followed (Warne 2003). It is prudent to assume that the combined effect of all changes may be greater than the sum of the individual effects of each change.

Measuring salinity stress in wild organisms

Current biomonitoring can only detect impacts of salinity after they have occurred, and is not amenable to preventing impacts on biodiversity. Financial, environmental and social costs of restoring the health of a freshwater system after it has been damaged are much greater than preventing an impact from occurring in the first place. Indeed, natural variation, statistical uncertainties and difficulties in defining reference (or 'natural') conditions, often mean that current biomonitoring systems can only detect relatively large changes and, therefore, provide a relatively poor early warning tool.

There is the potential to develop early warning tools to detect sub-lethal salinity stress by monitoring changes in the physiology, biochemistry, gene expression or histology of freshwater organisms. Such tools could be used (in conjunction with preventative management strategies based on risk assessments) to prevent damage to the health of freshwater systems from salinity stress before impacts occur at the population and community levels.

Conclusion

The effects of salinity on freshwater invertebrates has been investigated in a range of laboratory experiments, and these results have been compared to available information from other biological groups, including the occurrence of freshwater macroinvertebrates in nature as salinity increases. Compared to other biological groups, the effect of salinity on macroinvertebrates appears to be similar or greater. Therefore, the protection of sensitive macroinvertebrates, and use of the loss of macroinvertebrate species richness across multiple samples to assess salinity impacts, appears to also protect or demonstrate a lack of impact upon other biological groups. As a result, we have developed an approach for protecting all freshwater biodiversity based on our macroinvertebrates work, with a procedure for establishing target salinities that would trigger management intervention. In general, related macroinvertebrate species tend to have similar salinity tolerances across eastern Australia. Little comparable information is available for Western Australia where a longer period of salinisation may have resulted in greater salinity tolerance. Measurement of salinity (EC) value combined with biomonitoring for macroinvertebrates provides the underpinning data for river and wetland management. Issues of temporally varying salinity and changes in salinity co-occurring with other changes in water chemistry, hydrology and/or habitat will likely have their own effects, and modify the effect of salinity. At this stage it is not possible to quantify the combined effect of these complicating factors.

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Chapter 3 —

Understanding thresholds in the transition from saline to hypersaline aquatic ecosystems: south-west Western Australia

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Summary

1. Dryland salinisation in south-west Western Australia has caused major ecological changes in the aquatic ecosystems of this region, driven by a shift from fresh to saline and often hypersaline conditions. If salinisation continues to worsen, similar changes may also occur in the Murray-Darling Basin and other parts of the eastern states.
2. The change from saline to hypersaline conditions may cause a major alteration in ecological structure and function, resulting in the loss of the submerged macrophyte community that provides important habitat and food resources for invertebrate and vertebrate components of the aquatic food web.
3. Four main ecological regimes have been recognised in south-western Australian saline wetlands: i) clear, submerged macrophyte-dominated; ii) clear, benthic microbial-dominated; iii) turbid, phytoplankton-dominated; and iv) turbid, sediment-dominated.
4. The establishment of the submerged macrophyte regime appears to be controlled largely by salinity level, with the benthic microbial regime controlled by both salinity and water regime.
5. Research suggest that salt-tolerant macrophyte communities are unlikely to develop in seasonally-drying wetlands where the salinity is consistently greater than 45 ppt.
6. Although benthic microbial communities appear to be favoured by high salinities they are likely to be out-competed at low salinities by macrophytes, or by phytoplankton blooms if water column nutrient levels are high. However, the year-round dominance of benthic microbial communities at relatively low salinities in a permanent wetland indicates that physico-chemical stability driven by water regime may significantly alter ecological dynamics.
7. The ecological dynamics of saline wetlands appear to be driven by the combined effects of salinity and water regime acting on species life histories and competitive abilities, rather than by a single factor.
8. The alternative regimes conceptual model may not be appropriate to represent ecological regime shifts in seasonally-drying aquatic systems.

Background

Large areas of the Australian continent are currently affected by secondary (anthropogenic) salinisation. In some parts of Western Australia, particularly the ‘wheatbelt’ region which lies between the 600 and 350 mm rainfall isohyets, salinisation, primarily as a result of land clearing and the associated rise in saline watertables, has been occurring for over a century (Hatton et al. 2003, Figure 1). As a consequence, very few freshwater systems remain in this region, and in order to manage the changing landscape, a key question facing natural resource managers is which physico-chemical or ecological thresholds have most importance in the change from saline to hypersaline conditions? Knowing this will allow these systems to be managed so that further losses of ecological function and biodiversity can be prevented.

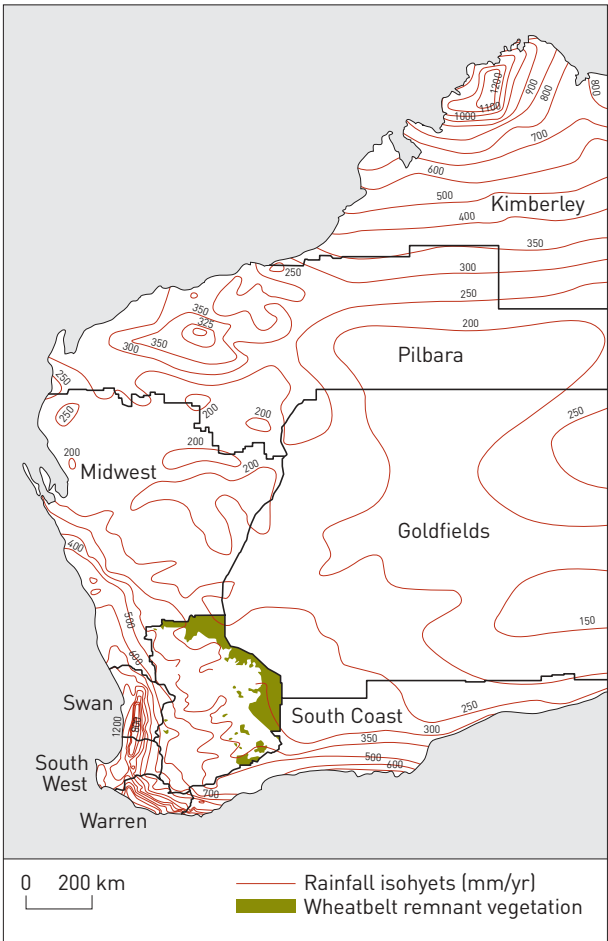


Figure 1. Map of Western Australia showing the location of the wheatbelt region and extent of vegetation clearing within this region.



Effects of salinisation at Lake Mears in the Western Australian wheatbelt.

This chapter considers the broad ecosystem changes that occur when salinity rises in waterbodies with salinities ranging from hyposaline to hypersaline (see Box 1). The research question explored in this chapter is ‘do well-defined thresholds exist that signal a change in ecosystem structure and function when moving from saline to hypersaline ecosystems?’

Box 1. Definitions of salinity levels for south-western Australian inland saline wetlands. Upper and lower salinities are likely to be found at the extremes of high and low water level.

Salinity level	Lower salinity (ppt)	Upper salinity (ppt)
Fresh	0	3 ^A
Hyposaline	3	10 ^B
Saline	5	50 ^C
Hypersaline	45 ^D	360 ^E

ppt = parts per thousand. 1 ppt ~ 1.6 mS/cm (although this relationship changes at high salinities).

A. 3 ppt is widely recognised as the upper limit for fresh and the lower limit for saline waters (e.g. Nielsen et al. 2003).

B. 10 ppt is a recognised threshold for effects of salinity on aquatic biota in Australia (Brock & Lane 1983, Nielsen et al. 2003).

C. 50 ppt is taken from Hammer (1986) and corresponds well with values observed in the field in south-west Western Australia (Sim et al. 2006d).

D. 45 ppt based on the upper threshold for submerged macrophyte establishment (Sim et al. 2006b).

E. NaCl saturation (Williams 1966). Reported as 270 ppt in Segal et al. (2005).



All photos throughout this chapter are taken by Lien Sim.

The research described here sought to investigate the effects of increasing salinity on plant, animal and microbial communities in salinising wetlands. It also examined the potential for transitions between 'ecological regimes' under different conditions. A further aim was to develop conceptual models to explain these 'regime shifts' in saline aquatic ecosystems.

The term 'ecological regime' is used to describe a persistent, characteristic assemblage of species groups and physico-chemical conditions in an ecosystem. Ecological regimes have also been widely referred to as ecosystem 'states' (e.g. Beklioglu & Moss 1996). Alternative states theory (sensu Moss 1990, Scheffer 1990) has been proposed as a possible conceptual framework for ecological regime shifts in salinising wetlands (Davis et al. 2003, Figure 2). As part of our

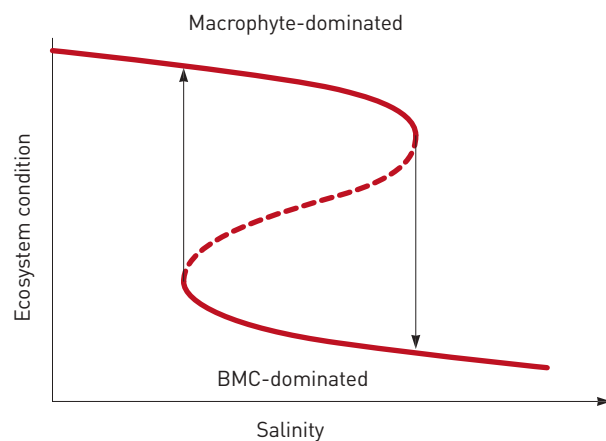


Figure 2. Conceptual diagram illustrating alternative regimes (or states). Adapted from Scheffer et al. 2001, Sim et al. 2006d.

wider research program, the application of this theory to saline wetlands was examined using observational data to track the occurrence and persistence of different ecological regimes (Strehlow et al. 2005). In addition, the mechanisms underlying these changes were investigated, in particular, those responsible for the formation and persistence of the macrophyte-dominated and benthic microbial community-dominated ecological regimes (Sim et al. 2006a, Sim et al. 2006b, Sim et al. 2006c, Sim et al. 2006d).

Although our research focused on secondary salinisation, primary (or naturally) saline and hypersaline systems also occur in Australia, particularly within arid and semi-arid regions and coastal areas. It is important that we understand the processes occurring within these systems and how these might change due to climatic variability and various human impacts, as this can also help us to better understand and manage secondary saline and hypersaline ecosystems.

Approach

Temporal changes in ecological regimes under a range of salinities

Water quality, submerged macrophytes and macro-invertebrates were monitored at four to six weekly intervals at six wetlands in south-west Western Australia over a period of 18 months, to investigate seasonal variation in ecological regimes in a range of primary and secondary saline aquatic ecosystems. The aim was to determine whether different ecological regimes could be recognised, and to identify the conditions that appeared to characterise these regimes. It should be noted that in south-western Australia, rivers are rarely perennial, and either form chains of pools or dry completely in the warmer months. Therefore, seasonally-drying wetlands are the dominant type of aquatic ecosystem throughout the region.

The study wetlands dried and filled at different times in response to local rainfall patterns, and salinities varied in accordance with evapoconcentration and dilution (Figure 3, overleaf). Two types of clear-water wetlands were recognised; those dominated by submerged aquatic macrophytes (*Ruppia* sp., *Lepilaena* sp. and *Lamprothamnium* sp.) and those dominated by benthic microbial communities. Two types of turbid wetlands were also recognised; those with high concentrations of phytoplankton, and those with high concentrations of suspended sediments. A primary saline lake (Lake Mount Brown) and two lakes that have only recently been affected by secondary salinisation (Meeking Lake and

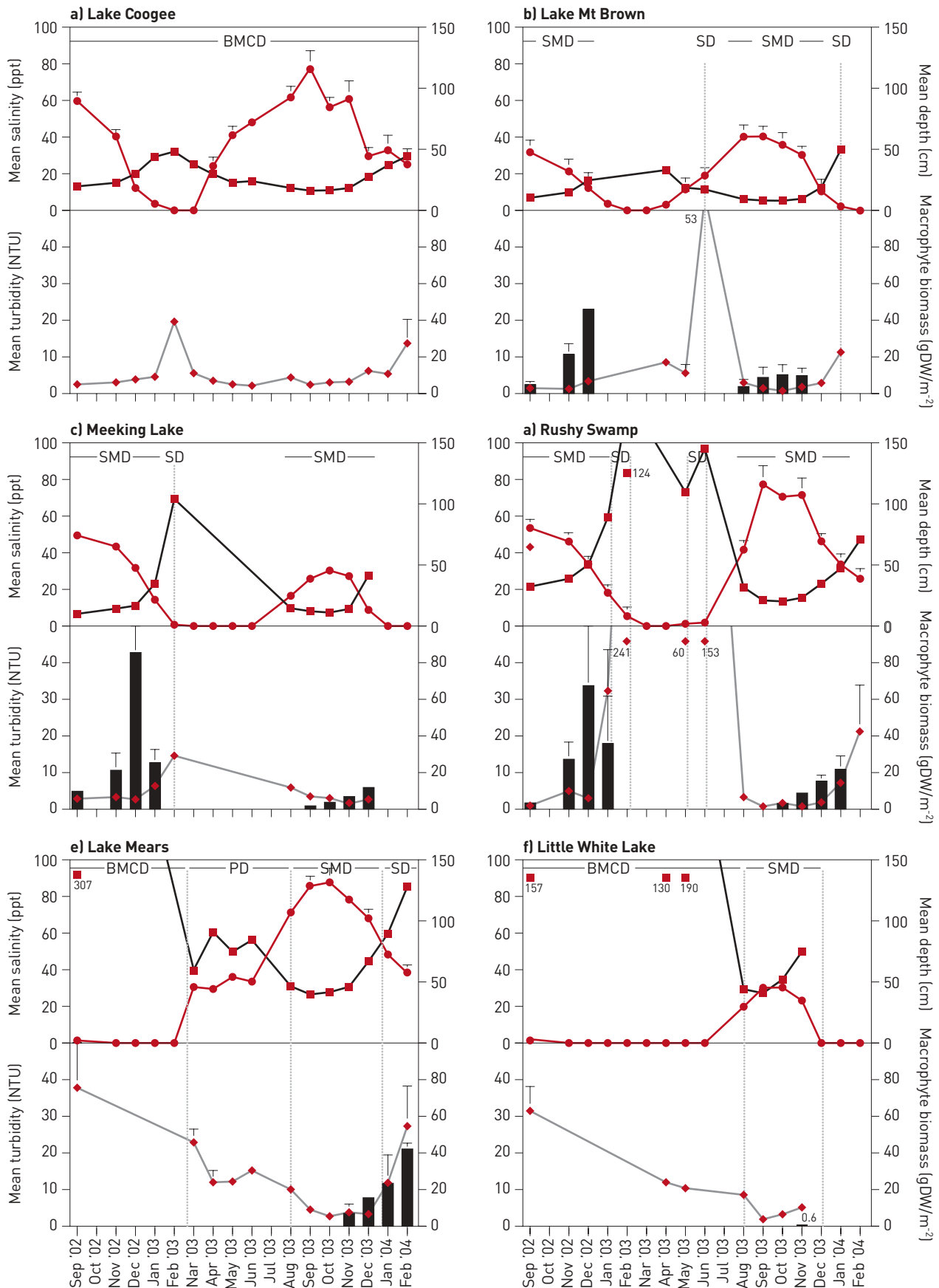
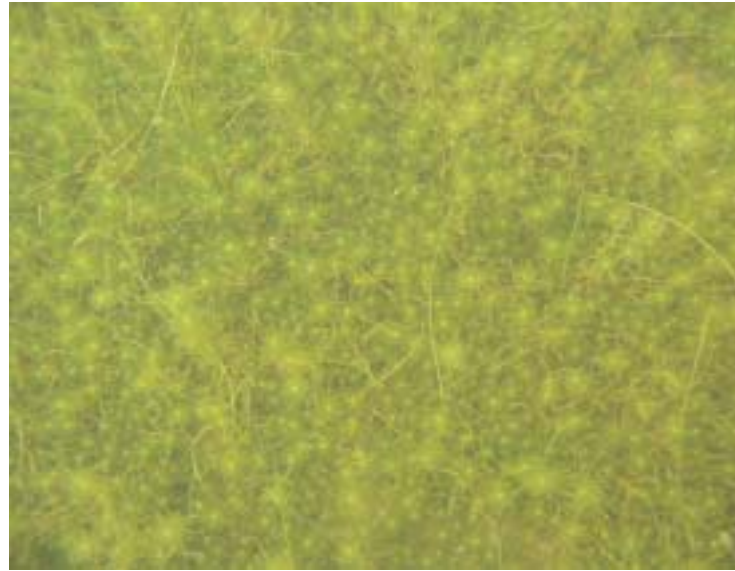


Figure 3. Seasonal changes in salinity level, water depth, turbidity and submerged macrophyte biomass at six saline wetlands (Strehlow et al. 2005).

Rushy Swamp) persisted as clear, macrophyte-dominated systems throughout most of the study period, except during drying and filling phases. Two lakes with a long history of secondary salinisation (70 years, Lake Mears and Little White Lake) moved between regimes during the study period. A clear, benthic microbial community-dominated regime only persisted at the (primary saline) wetland that contained permanent water throughout the study period (Lake Coogee). Both of the turbid regimes were only present during drying and refilling phases. A richer and more abundant macroinvertebrate fauna was associated with both the primary and secondary saline clear, macrophyte-dominated wetlands, indicating the importance of this community in ecosystem structure and function.

Common salt-tolerant submerged macrophyte species found in south-western Australian wetlands.



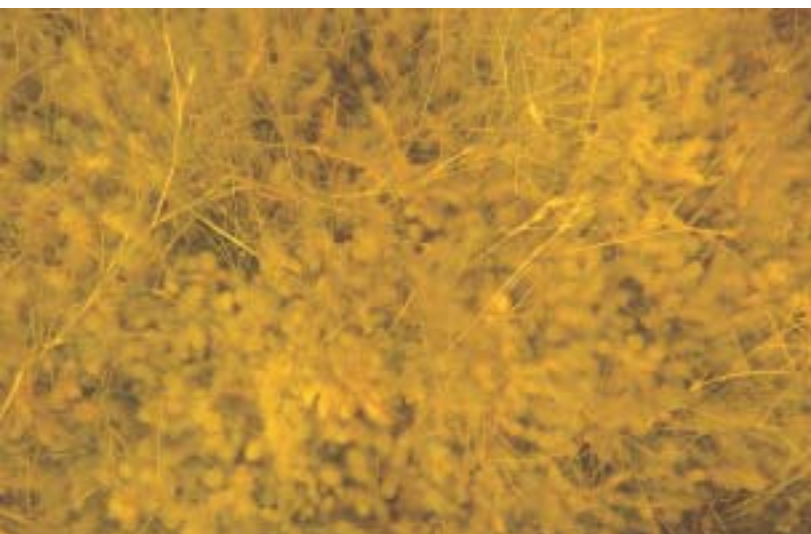
Lamprothamnium cf. succinctum.



Lepilaena preissii.



Ruppia polycarpa.



Ruppia megacarpa.



Lamprothamnium sp. (and Lepilaena preissii).



Above: Flocculent benthic microbial community at Lake Coogee.



Above and below: Salt crust with green benthic microbial layer underneath at Lake Mears.



This page and opposite: Examples of benthic microbial communities found in south-western Australian wetlands.



Clear macrophyte-dominated wetlands.
Above: Lake Mount Brown, below: Meeking Lake.





Below: Rushy Swamp.



Salinity thresholds for submerged macrophyte communities

After establishing the types of ecological regimes found in these systems, we focused on the most common two regimes (clear macrophyte-dominated and clear benthic microbial-dominated) to determine some of the mechanisms responsible for their formation and maintenance. Salinity tolerances are likely to be central to the ability of submerged macrophyte communities to persist in salinising aquatic ecosystems.

We studied the germination and flowering of four submerged macrophyte species common in saline Western Australian systems; *Ruppia polycarpa*, *Ruppia megacarpa*, *Lamprothamnium macropogon* and *L. cf. succinctum*, and tracked the survival of adult *R. polycarpa* as salinities were increased to a range of endpoints (6, 15, 45, 70 and 100 ppt). We found that increased salinity led to a decrease in the number of germinating plants, an increase in the time to their emergence from the sediment, and a decrease in the number of plants becoming fertile (*R. polycarpa*, *L. macropogon* and *L. cf. succinctum*)¹. The survival of adult *R. polycarpa* also decreased as salinity increased, and was negatively affected by faster rates of salinity increase. The experimental upper salinity limits for germination were 40–50 ppt for *R. polycarpa* and *L. cf. succinctum*, and 30–40 ppt for *L. macropogon*. Survival of adult *R. polycarpa* also declined markedly at above 45 ppt. Optimum salinities for germination and growth for this suite of species appear to be in the range 0–6 ppt, with dominance likely to drop off above 45 ppt (Sim et al., 2006b).

Salinity thresholds for benthic microbial communities

The dominant biological community in many salinising wetlands commonly comprises a cohesive layer of benthic microbes (often associated with very high salinities). However, the replacement of submerged macrophytes by these benthic microbial communities may not be due to increases in salinity alone. We investigated some of the environmental conditions required for initiation and dominance of benthic microbial communities using a combination of experimental and observational data. One experiment investigated the importance of prior establishment of benthic microbial communities on their ability to resist macrophyte colonisation ('persistence' experiment), while the other investigated hydrology and its effect on sediment perturbation, potential nutrient release and subsequent benthic microbial community establishment ('flooding' experiment).

¹ Germination of *R. megacarpa* was low, with only two seeds germinating under any conditions; providing limited information with regard to salinity response.



Lakes with a long history of salinisation, (left) Lake Mears and (right) Little White Lake.

The ‘persistence’ experiment measured the biomass of benthic microbial communities and emergence of macrophytes from sediments kept either wet or dry for four weeks, then flooded at a range of salinities. Benthic microbial biomass was similar across all of the salinities tested (15, 45 and 70 ppt), with a slight increase at higher salinities, suggesting that none of the salinity levels tested limited benthic microbial community development. Pre-wetting of sediments usually increased benthic microbial community biomass and reduced macrophyte germination, but the latter was attributed to the development of anoxic sediments rather than increased benthic microbial community biomass. Germinating macrophytes were able to emerge through both benthic microbial communities and dense heterotrophic bacterial blooms, demonstrating that they could become dominant even when another community was already established. Field data supported these results, suggesting that the development of benthic microbial communities is not limited by salinity alone, but includes other factors, such as water regime.

In the ‘flooding’ experiment, the largest differences in nutrient concentrations ultimately lay between the pre-wet and pre-dry treatments (due to the greater release of nutrients and development of anoxia in the latter) rather than those subjected to fast versus slow flooding. In response to this, highest benthic microbial community biomass occurred in treatments with pre-wet sediment, corresponding with lower phytoplankton biomass. This suggests that pre-establishment does play a role in benthic microbial dominance, even though macrophytes may still be able to subsequently colonise these areas (Sim et al. 2006c).

Testing of conceptual models for regime shifts

Observational data from seven saline wetlands over an 18-month period were used to evaluate which of three generalised patterns of change in community dominance could be caused by salinity increases in aquatic ecosystems. The three models for ecosystem behaviour were the continuum (approximately linear)

Persistent benthic microbial regime and permanent water at Lake Coogee.



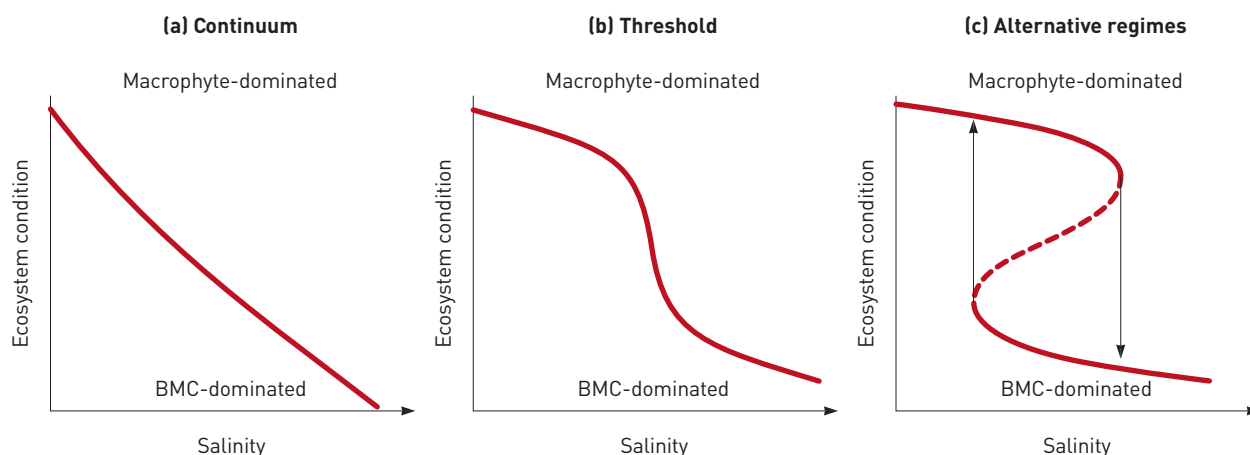


Figure 4. Three possible pathways along which the shift from a macrophyte-dominated to a benthic microbial-dominated regime might occur in a secondarily salinised wetland system. Adapted from Scheffer et al. 2001, Scheffer & Carpenter 2003, Sim et al. 2006a.

response, simple threshold (small changes in ecosystem structure up to a threshold, then a rapid change) or alternative regimes conceptual model (also known as the ‘alternative states’ model) in which both regimes are possible over a range of intermediate salinities, and each regime is maintained over this range by various self-stabilising mechanisms (Figure 4).

We also aimed to identify whether factors other than salinity played a major role in defining the ecological regimes of saline wetlands or in causing shifts between regimes.

Key findings

Ordination of biological variables revealed two groups of wetlands — those dominated by benthic microbial communities and those dominated by submerged macrophytes (Figure 5, overleaf). The mean salinities of these two groups were very similar, suggesting that a salinity threshold was not responsible for benthic microbial versus macrophyte-dominance. No other environmental variable was found to have a strong, direct influence on the groupings.

Data from the seven wetlands indicated that the continuum, simple threshold and alternative regimes conceptual models did not appropriately represent transitions between ecological regimes in seasonally-drying wetlands. Macrophyte and benthic microbial regimes occurred at overlapping salinity levels, excluding both the continuum and threshold models, and the regular occurrence of drying appeared to preclude the alternative regimes model. Drying prevented the development of strong positive feedback mechanisms, which might otherwise have maintained the benthic

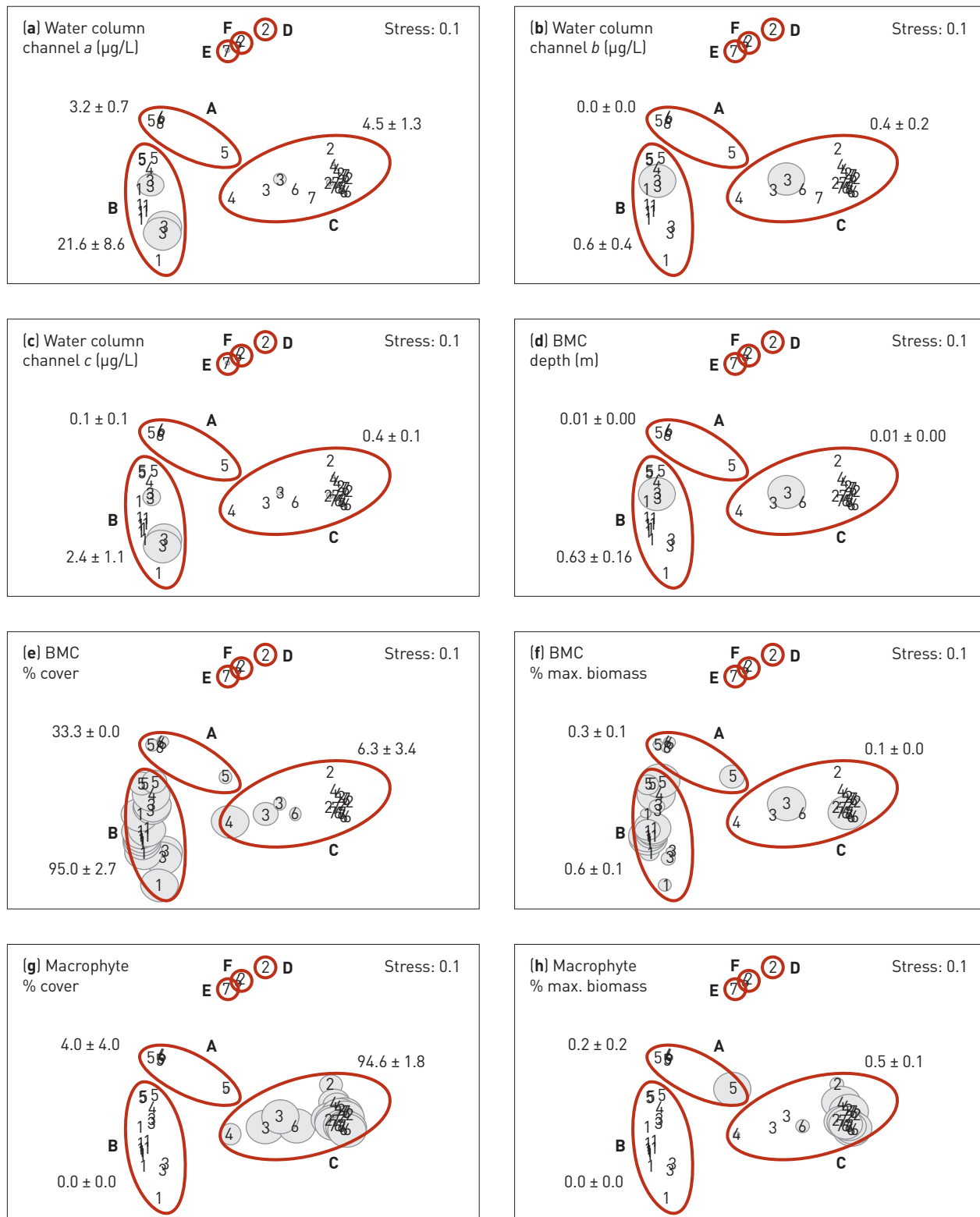
microbial community-dominated regime. We hypothesise that an alternative regimes model might still be valid for salinising ecosystems holding permanent water.

Management implications

In a landscape where there is little prospect of restoring freshwater ecosystems due to the scale and severity of salinisation (Hatton et al. 2003), saline macrophyte-dominated wetlands have structural and functional importance, and their replacement by benthic microbial communities is likely to lead to a reduction in these ecological values. Our results suggest that salt-tolerant macrophyte communities are unlikely to develop in seasonally-drying wetlands where the salinity is consistently greater than 45 ppt, and that salinity should not exceed 30 ppt until propagules have been produced if the macrophyte-dominated ecological regime is to persist.

Although benthic microbial communities appear to be favoured by high salinities, they are likely to be out-competed at low salinities in the field by macrophytes or by phytoplankton blooms if water column nutrient levels are high. However, the year-round dominance of benthic microbial communities at relatively low salinities in a permanent wetland indicated that physico-chemical stability driven by water regime may significantly alter ecological dynamics.

The dynamics of regime change in saline wetlands appear not to be driven by any single variable, but by the combined effects of salinity and water regime on species life histories and competitive abilities. Consequently, the development of management guidelines that recognise the presence of different ecological regimes and that



1. Lake Coogee. 2. Lake Mount Brown. 3. Lake Mears. 4. Little White Lake. 5. Arthur River Flats. 6. Rushy Swamp. 7. Meeking Lake.

Figure 5. Bubble plots showing the association of biological variables with the biological dataset. Larger bubbles indicate higher values of each variable. Plots depict: (a) water column chlorophyll a; (b) water column chlorophyll b; (c) water column chlorophyll c; (d) depth of benthic microbial community; (e) % cover of benthic microbial community; (f) % of maximum benthic microbial community biomass [calculated for each wetland and wetting–drying cycle]; (g) % cover of submerged macrophytes; and (h) % of maximum submerged macrophyte biomass [calculated for each wetland and wetting–drying cycle]. Biological cluster groups are circled. Numbers on each plot are mean values $\pm$ SE (Sim et al. 2006d).

consider the interactions between water regime, salinity, and primary and secondary production will be more useful in protecting biodiversity and ecological function in these systems than managing salinity as a single factor.

The findings from this research are currently being used in applied research and management planning for the disposal of saline groundwater from deep drains. Wetlands and rivers are often viewed as convenient conduits or disposal points for hypersaline groundwater, and in some cases this has taken place without a clear understanding of the implications of these changes in salinity and hydrology for aquatic ecosystems. Given the enormous pressure to approve drainage schemes in catchments in south-western Australia, the identification of important physico-chemical thresholds and an ability to predict ecological outcomes is more important than ever.

To date, the results of this research have been incorporated into at least four major applied research projects operating within the Western Australian wheatbelt, two of which are ongoing. These are:

- 'Downstream Ecological Impacts of Engineering Interventions for Salinity Control in the Wheatbelt of Western Australia' (CSIRO, Murdoch University, University of Western Australia, Department of Water),
- 'Yenyenning Catchment Engineering Salinity and Water Management Feasibility Project' (GHD Pty Ltd, Murdoch University, CSIRO, Department of Water),
- 'Wheatbelt Wetlands Assessment' as part of the 'Wheatbelt Drainage Evaluation' (Department of Environment and Conservation, Department of Water) — ongoing, and
- 'Avon Baseline Project' (Department of Environment and Conservation, Avon Catchment Council) — ongoing.

In all of these projects, knowledge about the key thresholds for transition between ecological regimes, and the interaction between salinity and other factors such as water regime, have shaped the development of management actions to address the release of saline groundwater into natural rivers and wetlands.

The knowledge generated by this research is likely to have great relevance to management planning for salinising wetland systems elsewhere in southern Australia, particularly due to its focus on hydrologically-dynamic wetlands that are subject to regular drying. Many northern hemisphere models of wetland function are developed for systems that experience much greater stability in conditions than the majority of shallow waterbodies in southern Australia.

Conclusions

This research has generated a greater understanding of the ecological function of salinised wetlands, as well as establishing thresholds that signal the transition from macrophyte-dominated to benthic microbial dominated ecological regimes. This transition between regimes results in a considerable loss of biodiversity and a simplification of ecological processes. The salinity threshold of 45 ppt, above which macrophyte germination is unlikely to occur, provides a tangible target for management that was not previously available.

Our testing of a range of conceptual models has provided a basis for predicting the likely ecological outcomes of increasing salinisation, and the consequences of discharging saline groundwater into natural waterbodies. This research has negated, to some extent, the prevailing paradigm that once a wetland undergoes secondary salinisation it loses all of its ecological values. Although considerable freshwater biodiversity is lost in the transition from fresh to saline conditions, macrophyte-dominated saline systems still support valuable ecosystem processes by providing habitat for aquatic invertebrates and vertebrates such as turtles and waterbirds.

Strategies for managing salinising wetlands and rivers, including those receiving groundwater discharge created by deep drainage or pumping, must not only recognise the importance of thresholds for aquatic macrophyte germination and adult macrophyte death, but also need to consider the interaction of salinity and other factors such as water regime.

Since the project was completed, we have become aware that saline groundwater discharge in some areas of the Western Australian wheatbelt is not only saline, but also highly acidic. As a consequence, further research is now needed to determine the combined, and possibly synergistic impacts of high salinity, low pH water drainage waters on aquatic systems.

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Nutrients as contaminants

At the levels typically found in Australia, nutrients in rivers do not generally constitute a serious issue for irrigation or drinking water quality. Rather, it is the ecological effects of nutrient enrichment (eutrophication) and the associated water quality degradation that present problems. Nutrient enrichment of rivers stimulates primary production resulting in aquatic plant growth, and sometimes excessive algal growth. This risk is exacerbated by the loss (or non-regeneration) of riparian vegetation and consequent loss of shade over the stream, leading to increased light intensity and higher water temperatures during periods of low flow. These conditions favour the development of problematic algal blooms.

The key nutrients studied to date are nitrogen and phosphorus. Both can influence in-stream production. Both have multiple potential pathways into streams attached to sediment or in dissolved or colloidal forms, in surface or sub-surface flows, and in readily bio-available or sequestered forms. Both have become increasingly available and more mobile following catchment development for agriculture or urban land uses. Improving the management of these nutrients has become a priority in many catchment plans, with targets established for their loads and/or concentrations in rivers and receiving waters.

Excessive algal growth, or algal blooms, is of concern to water supply authorities because both phytoplankton and attached algae can block filters and delivery equipment, and the high organic load leads to increased water treatment costs. Algal blooms also impose costs on recreation and tourism operations. Blooms of many cyanobacteria species are especially problematic because of the toxins they produce. Death and decomposition of excessive in-stream growth can reduce oxygen levels to the detriment of aquatic biota.



Traditionally, freshwater algal blooms were believed to be triggered by high levels of phosphorus, because that was the nutrient that was believed to limit their growth. Research during the 1990s into inland Australian rivers showed that low river flow was the primary trigger for causing algal blooms, although the amount of phosphorus present in the waterbody could still control the size of the bloom that developed. This was because the damming of these inland rivers, and low but continuous water releases to meet the needs of irrigators over summer, had effectively turned them into a series of shallow lakes where thermal stratification occurred. This provided the necessary conditions for rapid algal population growth.

Although phosphorus can limit the size of the blooms, the research also demonstrated that, in contrast to the conventional view, nitrogen can sometimes limit phytoplankton growth. Consequently, a better understanding was required of the nitrogen cycle and its role in controlling algal biomass and species composition. The research described in this section builds on these findings.

Nitrogen is a more mobile element than phosphorus, with multiple pathways by which it can enter and leave a

waterbody. It can enter by fixation from the atmosphere, through groundwater, or through surface water inflows. Once present in a water body it can be recycled through the sediments and organic matter. It can be removed through denitrification (being turned into nitrogen gas that escapes to the atmosphere), by being transported in river flow to oceans, and by being incorporated into plants and animals that are then harvested.

The NRCP research that has been conducted has contributed to understanding the nitrogen cycle by investigating how it enters waterways from adjacent farmland (chapter 4) — probably one of the sources that are most easily controlled by land managers. This work investigated the surface and sub-surface nitrogen movement through riparian zones and riverine sediments. It also looked at the potential of these zones to denitrify the dissolved nitrogen and thus remove it before it entered waterways. Chapter 5 discusses the options available to manage algal blooms, while chapter 6 considers how management of fertilisers can be improved to reduce the amount of nutrients reaching waterways in agricultural areas.

Photo Roger Charlton.



Chapter 4 — Managing diffuse nitrogen loads: in-stream and riparian zone nitrate removal

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Summary

- Riparian soils and in-stream sediments have the potential to reduce nitrogen loads reaching downstream environments, particularly through the process of denitrification (which converts nitrate to inert nitrogen gas). The microbes that carry out denitrification require organic carbon as a source of fuel and an environment with low or no oxygen. These conditions are often met in riparian zones and stream sediments.
- Riparian environments favour denitrification when nitrate-containing groundwater passes through the carbon-rich root zone of riparian vegetation. Many factors influence the amount of nitrate removed, including the flow rate, nitrate concentration, soil properties and riparian setting.
- Comparisons of 16 sites from contrasting environments in south-east Queensland, Western Australia and Victoria showed similar rates of denitrification potential across the three regions for some soil types, although there were several distinct regional differences.
- At all sites, rates of denitrification potential were highest at the surface of riparian soils, with rates decreasing down the soil profile. Rates were relatively high for in-stream sediments, indicating their potential to remove nitrate within the water body itself.
- Combined measurement of nitrate and organic carbon concentrations may provide a useful rapid assessment of the denitrification potential of riparian soils and in-stream sediments.
- New guidelines for riparian zone management recommend maintaining or increasing soil organic carbon levels to increase denitrification potential and reduce the delivery of nitrogen to streams.
- Riparian lands most conducive to denitrification are typically relatively flat, low-lying areas with the potential for slow groundwater seepage of nitrate.
- Restoration of riparian vegetation will have multiple benefits, including enhanced nitrogen removal through denitrification, enhanced habitat for biodiversity, and stream and bankside shading to avoid temperature extremes.

Background

Nitrogen plays a critical role in the functioning of Australian aquatic ecosystems. Aquatic organisms require nitrogen for their metabolism, growth and reproduction, but when present in excess, nitrogen can have adverse impacts that impair the health of freshwater, estuarine and coastal ecosystems. Problems may include excessive growth of algae and other plants, blooms of toxic algae, and more subtle changes to the species composition and food web structure of aquatic communities (Boulton & Brock 1999, Schindler 2006). Recent studies have shown nitrogen management to be critical for maintaining or improving ecosystem health in several parts of Australia, including coastal systems like Moreton Bay (Dennison & Abal 1999) and Port Phillip Bay (Murray & Parslow 1999), and freshwater streams in south-east Queensland (Mosisch et al. 1999, 2001).

Human activities can greatly increase the quantities of nutrients reaching aquatic ecosystems, for example, through the use of fertilisers, the management of human and animal wastes, and practices that increase rates of soil erosion. Amounts of nitrogen in precipitation may also be elevated via nitrogen-containing emissions from burning fossil fuels. There is increasing evidence that excessive nitrogen inputs are occurring in Australia, to the detriment of our rivers, reservoirs, and coastal environments (e.g. Hart & Grace 2001, Australian State of the Environment Committee 2001).

Over the last decade, it has been found that many freshwater systems are ‘nitrogen-limited’ — that is, it is the amount of bioavailable nitrogen in the water that controls algal or other plant growth, provided that factors such as light penetration into the water and temperature are also conducive to growth. Australian riverine systems, including the Darling River (Oliver et al. 1999) and waterways in the south-east Queensland study region (Dennison & Abal 1999), have been shown to be nitrogen limited, as have many other streams and lakes worldwide. This finding has dramatically changed attitudes and strategies for nutrient management in freshwaters, as the focus used to be solely on decreasing the amount of phosphorus. As highlighted by Boulton and Brock (1999), eutrophication management now requires careful consideration of both nitrogen and phosphorus, and the relative amounts of each.

An important first step for effective management of nitrogen in catchments is to manage source areas to minimise its off-site movement. However, some forms of nitrogen (especially nitrate) are very mobile in the environment as they are only poorly bound to soil particles and require multiple approaches to minimise



Photo Carol Conway.



Photo Mick Smith.

Riparian buffer strips are a “line of defence” to reduce nutrient loadings into streams.

their transport downstream. Riparian buffer zones can trap sediment and associated nitrogen from surface runoff and so reduce loadings to streams (Prosser et al. 1999). In addition, riparian buffers support a variety of sub-surface processes that have the potential to transform and remove nitrogen (for example, Cirimo & McDonnell 1997). Similarly, a potential further “line of defence” is provided by sediments within aquatic systems themselves, which can also remove nitrogen and so provide an additional buffer against excessive downstream loadings (Bartkow & Udy 2004).

Scientists in Europe, North America and New Zealand have explored the nitrogen removal capacity of in-stream environments and riparian zones with the aim of managing diffuse nitrogen inputs. These found that riparian zones can serve as buffers between land-based activities and downstream ecosystems by removing excess nitrogen. However, little is known about the extent to which this nitrogen-buffering effect may occur in Australia, given the large variation in climate, geology and surface water-groundwater interactions. This lack of data hampers our ability to successfully manage freshwater resources in Australia, and was a major issue raised at the 2000 Land & Water Australia “Nitrogen workshop” (Hart & Grace 2001). Consequently, the overall aim of our work was



Uncontrolled stock access is a major contributor of nutrient into streams. Photo Wade Hadwen.

to increase our understanding of nitrogen cycling processes in streams and riparian zones to improve water quality and ecosystem health. This work complemented two other Australian studies conducted over the last six years that have investigated nitrogen processes, modelling and management in riparian zones.

This chapter describes the research undertaken for this project and key project findings. It concludes with a discussion of management implications and guidelines.

Research approach

This project investigated broad patterns of nitrogen cycling in freshwater streams and their associated riparian zones, with an emphasis on the potential for reduction of nitrogen inputs to surface waters. The focus of past studies has typically been on either the riparian zone or the stream, but not both. This project took a unique perspective in examining both ecosystem components, and did so in multiple sites across three distinct biogeographic regions: south-east Queensland (SEQ), southern Victoria (VIC) and south-western Australia (WA). These regions were chosen for their contrasting climates and soil types, and the fact that development of conceptual models could draw on data from past and ongoing research conducted in these regions.

Conceptual model development and knowledge gap identification

The initial conceptual models developed for this project focused primarily on small streams (orders 1–3), as the dynamics of these low-order streams are known to be particularly critical to nutrient cycling and transport observed at a catchment scale. The models drew on a general understanding of interactions between hydrology, riparian zone vegetation, soil organic carbon and nitrogen cycling processes from both the international literature (e.g. Hill et al. 2000) and previous work in south-east Queensland (Rassam et al. 2006b, Hunter et al. 2006) to depict expected regional and seasonal differences. These initial conceptual models, as well as information generated from searching the literature and existing data for the regions, were used to identify knowledge gaps and guide the project research design.

Two main research questions were developed based on knowledge gaps identified: *1) What influences rates of nitrogen cycling processes in streams and their adjacent riparian zones?* A particular focus was placed on denitrification (see definition in text box overleaf), as this process is of great importance from a management perspective for removal of nitrogen from riverine ecosystems. The factors hypothesised to influence rates of denitrification were:

- a) soil and sediment organic carbon content,
- b) riparian zone vegetation type, and
- c) changes in soil moisture, the extent of saturation and stream flow related to season (wet versus dry season).

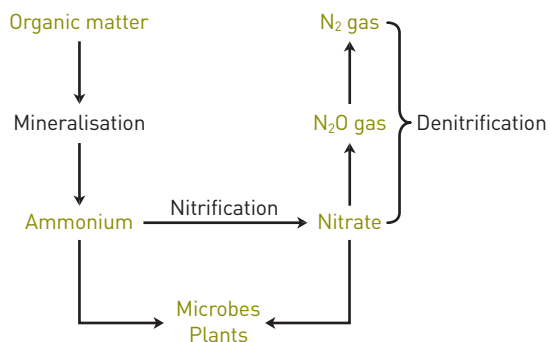
In addition to nitrate, the microbes that carry out denitrification require a source of organic carbon for energy and a low or no oxygen environment (often present under saturated conditions). As saturated conditions and nitrate concentrations influence the actual denitrification that occurs, the second research question was 2) *How do groundwater and surface water hydrology and chemistry vary across sites?*

Denitrification...

is the conversion of nitrate to nitrogen gas (N_2). Denitrification is carried out by microbes in conditions of low or no oxygen, and these microbes require organic carbon to fuel their metabolism. This process can effectively remove nitrate, as it is converted to N_2 that can diffuse out of the ecosystem. N_2 constitutes 78% of our atmosphere and is inert. Denitrification sometimes stops before the final step, resulting in the production of nitrous oxide (N_2O), which is a greenhouse gas. The conditions under which this occurs are not well known.

Other essential nitrogen cycling processes convert nitrogen from one form to another, such as assimilation (biological uptake), mineralisation (recycling organic nitrogen from plant and animal detritus to ammonium), and nitrification (ammonium to nitrate), but the nitrogen generally remains in the ecosystem. Ammonium and nitrate are biologically available forms, and in high concentrations are, as a result, particularly problematic for receiving waters.

Nitrogen cycling — forms and processes



Study sites

Four to six sites were selected in each region to span a gradient of riparian zone vegetation (grass versus trees). Sub-catchment land use was generally dominated by agriculture, including grazing and horticulture, with some forest and residential areas. Of the six sites in southern Victoria, five were situated on first and second order streams in the Woori Yallock Creek catchment in the Dandenong Ranges to the east of Melbourne. The sixth site was in Gippsland, also to the east/south-east of Melbourne. Four sites were chosen within a 100 km radius of Albany, south-western Australia, three in the King River catchment and one in the Marbellup catchment (surface water and groundwater were also assessed at an additional 16 sites in this region). In south-east Queensland, six sites were established, two in each of the Logan, Stanley, and Maroochy River catchments.



Victorian site, well treed (above), sparsely treed (below).
Photos Darryl Holland.



Measuring denitrification potential

Measurements of denitrification potential and soil/sediment properties were completed for the different zones identified during conceptual model development (Figure 1). Soil cores were taken from both within and outside of the riparian zone to a depth of 50–100 cm below the water table, with each core then separated into three layers: surface, mid-profile, and deep. Stream sediments were sampled over two depth intervals, 0–2 cm (benthic) and 2–10 cm (hyporheic). Denitrification potential was determined with and without added nitrate, using the acetylene inhibition method on soil-water slurries (Tiedje et al. 1989, Smith et al. 1991, Hill et al. 2000). These measurements of denitrification potential provide ideal conditions for denitrification to occur, including no-oxygen conditions and, in the case of the samples with nitrate added, abundant nitrate. Rates of

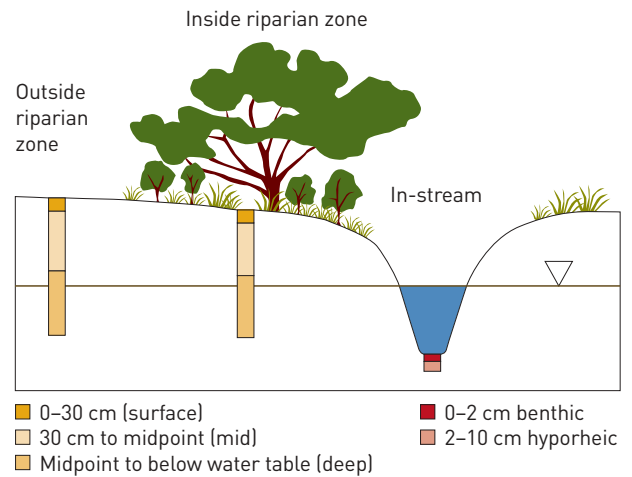


Figure 1. Sampling scheme for soils and in-stream sediment. 'Inside riparian zone' was located within 5 m of the stream channel and 'outside riparian zone' was located > 5 m from the stream channel, regardless of vegetation type present.



Western Australian site, well treed (above), sparsely treed (below).
Photos Craig Russell.



South-east Queensland site, well treed (above), sparsely treed (below).
Photos Carol Conway.



denitrification potential obtained in the laboratory are therefore considered to be a measure of the capacity of the soil and its existing microbial community to carry out denitrification, and are useful for comparison across soil types and sites. Actual rates of denitrification occurring in the field vary depending on the environmental conditions present (see ‘Assessing the relative importance of riparian and in-stream nitrogen removal’, page 54). The soils were also analysed for nitrate, ammonium, and dissolved organic carbon concentrations at the start of the incubations and for total organic carbon content (% by weight). The influence of season was investigated in WA and SEQ by measuring denitrification potential during both the dry and wet seasons.

Site hydrology and chemistry

Surface water and groundwater hydrology and chemistry were obtained from a combination of existing data and further sampling and analysis. Water samples were analysed for nitrate, ammonium, total nitrogen, and dissolved organic carbon concentrations. Groundwater wells and piezometers installed at the WA and SEQ sites were used to examine patterns of groundwater nitrogen inside and outside riparian zones. Stream stage and discharge (and groundwater levels, where available) were used to characterise durations of stream flow and riparian zone saturation.



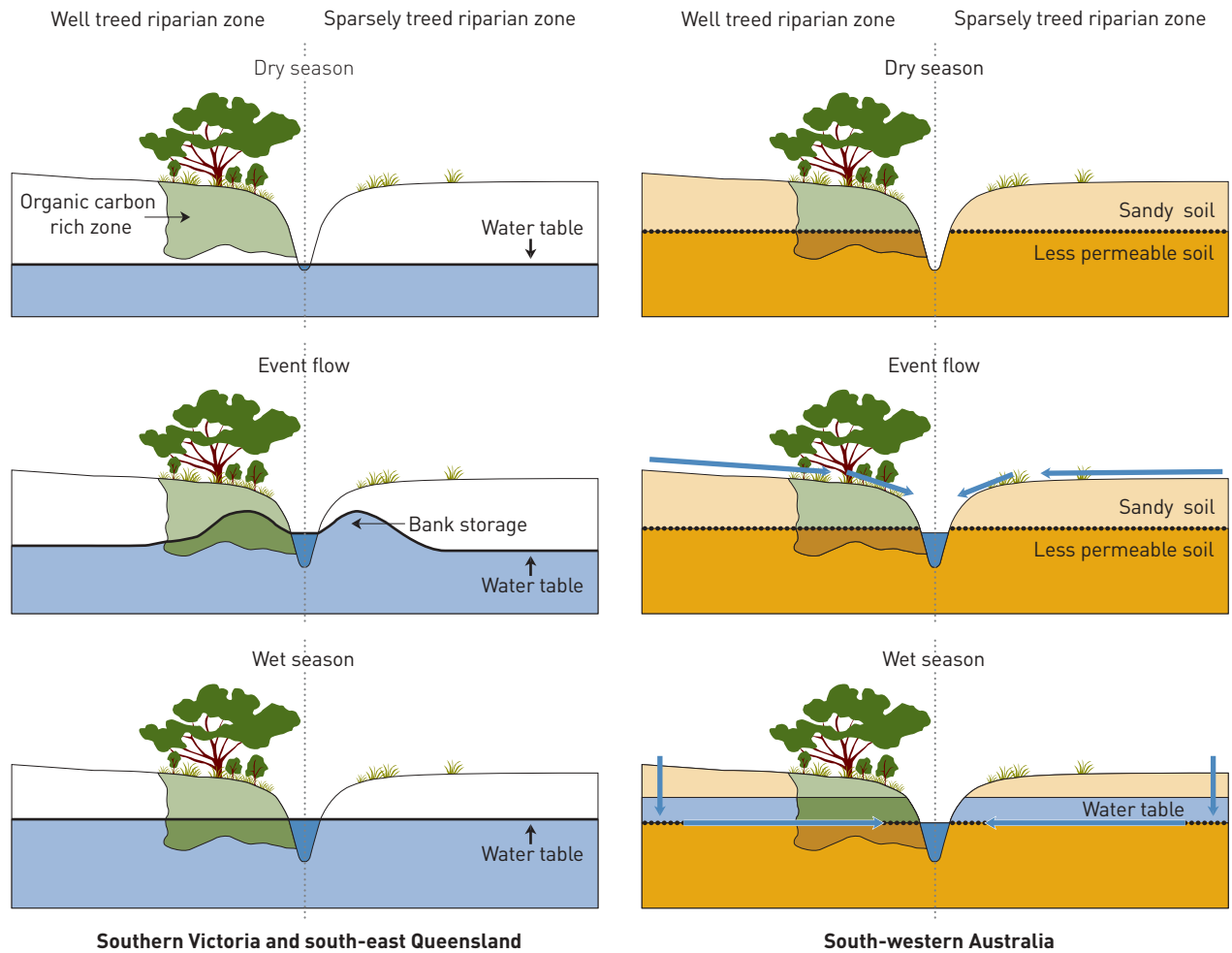
Collecting soil samples for denitrification potential measurements. Photo Carol Conway.



Collecting groundwater from a well for water chemistry analysis. Photo Carol Conway.



Soil samples prepared for laboratory measurement of denitrification potential. Photo Rob DeHayr.



Research findings

Conceptual models of nitrogen cycling

Initial conceptual models were developed for the three geographic regions and included four zones: stream channel, riparian, hyporheic (region below the streambed where groundwater and surface water mix), and hillslope/terrestrial environment. Three basic hydrological conditions were considered: dry conditions (low baseflow or no flow), wet conditions (high baseflow), and event flow (high stream stage/flood) (Figure 2). Although similar hydrological conditions are shown for all regions, the timing of wet season and event-based flow varies, with most rainfall occurring from April to October in WA, winter to spring in VIC, and summer in SEQ. The duplex soils found around Albany, WA, had very different sub-surface structure from soils in the other two regions, so WA sites were portrayed with a separate set of conceptual models. In all three regions well-treed riparian zones were predicted to have higher soil organic carbon contents (particularly deeper in the soil profile), and therefore to support higher rates of nitrogen cycling.

Figure 2. Initial conceptual models portraying the interactions between hydrology, riparian zone vegetation, and soil organic carbon. Shaded areas associated with riparian zone vegetation represent soil high in organic carbon. The position of the water table (shown by thick black line) influences the volume of organic-rich soil that is saturated and, therefore, the extent of denitrification. The presence of duplex soils at the field sites in WA (permeable soil layer above a less permeable layer, boundary shown with thick dotted line) controls riparian zone hydrology.

Below. Duplex soil visible along a road cut, WA. Photo Craig Russell.



Testing the conceptual models

Denitrification potential

Effects of nitrate and organic carbon

In all three regions, rates of denitrification potential were greatest in the surface soils and decreased considerably with depth, both inside and outside the riparian zone (Figure 3). In-stream sediment rates were intermediate. Surface soils also consistently had the greatest concentrations of nitrate and organic carbon, while in-stream sediments also had high organic carbon concentrations. These results suggest that microbial activity in general, and the potential for nitrate removal specifically, were much greater in surface soils and in-stream sediments than in deeper soils. While measured rates of denitrification potential were similar in shallow soils outside and inside the riparian zone, riparian zones are more likely to make significant contributions to nitrate removal via denitrification compared to upslope areas due to environmental conditions present. In particular, the saturated soils needed to support active denitrification are typically present over a greater spatial extent and saturated for a longer duration in riparian zones. See the section 'Assessing the relative importance of riparian and in-stream nitrogen removal' (page 54) for additional discussion on how potential denitrification relates to actual denitrification that occurs in the field.

While organic carbon content was predicted to strongly influence denitrification potential, soil nitrate concentration at the start of the incubation was the best single predictor of denitrification potential. Although there is some variability in the results, the

relationship between denitrification potential and nitrate concentration was significant for incubations with added nitrate, and also those with just the nitrate from the sample ('controls'), however the relationship was stronger for control incubations (Figure 4). This is consistent with denitrification potential being limited by nitrate at low concentrations; that is, there is adequate carbon available to support the denitrification process at these low nitrate concentrations. *A combination of measuring soluble nitrate and some indicator of soil organic carbon (either soluble or total % organic C) may be a good rapid assessment tool for estimating denitrification potential for use in catchment scale models.*

These results suggest that soil/sediment microbial communities that experience high supplies of nitrate through in-situ nitrification, or influxes of nitrate in surface water or groundwater, are 'primed' for high rates of denitrification when nitrate is present and saturated, low oxygen conditions occur. While denitrification potential showed a strong relationship with background soil nitrate concentration, high rates of denitrification can only be maintained over time if sufficient organic carbon is present.

Regional differences

Detailed comparisons of dry season denitrification potential among regions for surface soils and benthic sediments showed that rates were similar among regions for surface soils outside the riparian zone, but differed for benthic and surface soils that were inside the riparian zone. Rates at VIC sites were significantly greater than those from WA sites, and SEQ rates were also generally greater than WA (see Figure 5 for inside riparian zone

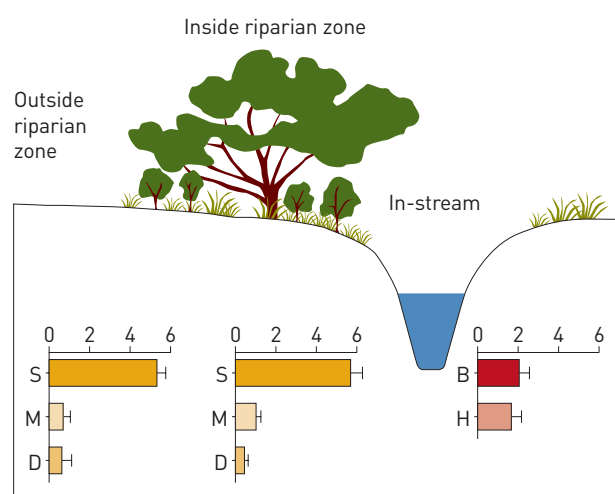


Figure 3. Denitrification potential for all soil layers. Bars represent mean ($\pm$ standard error) across all sites for incubations with added nitrate. S = surface, M = mid, D = deep, B = benthic and H = hyporheic (see Figure 1).

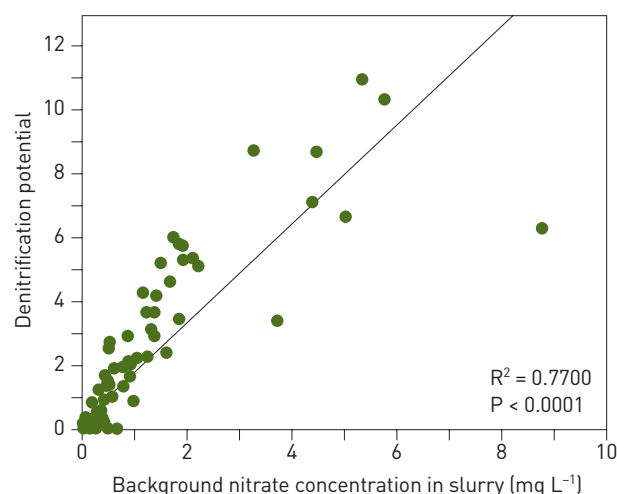


Figure 4. Denitrification potential and background nitrate concentration for all soil layers, all sites, for control incubations (no additional nitrate added to slurries). The relationship is significant (best fit line and linear regression analysis results shown).

surface soil results). The higher rates for VIC sites may be due in part to higher concentrations of soil nitrate, while lower rates for WA may be associated with the very sandy soils at the sites. Particularly low rates were found for dry stream bed sediments and some surface soils for WA, despite moderate levels of organic carbon. This suggests that levels of microbial activity may be lower overall due to the less favourable moisture regime.

Comparison of trees and grass

There were no consistent trends between denitrification potential and the composition of riparian zone vegetation (trees versus grass) (Figure 5), suggesting that both forms of vegetation can supply organic carbon to fuel denitrification (Figure 6). Alternatively, other factors may have a stronger influence on denitrification potential so that differences between vegetation types are less important. However, note that soil organic

carbon stores also reflect *past* vegetative cover, so current vegetation is not the only indicator of soil carbon levels. The measured soil organic carbon levels within or adjacent to the riparian zones, around 4% in the surface layer, are much higher than those recorded in soils that have been tilled and cropped for several years — generally around 1% or less. Attempts have been made overseas to increase rates of denitrification in riparian areas that are frequently saturated by incorporating high-carbon ‘wastes’ such as bedding straw into the surface soil.

Seasonal effects

There were also no significant differences in rates of denitrification potential between dry and wet seasons for SEQ or WA, the two locations where seasonal measurements were made. This suggests that for the coarse resolution required for catchment-scale nutrient models, measured rates of denitrification potential may be applicable across different seasons, but this assumption needs further testing. This result was contrary to expectations, since the actual in situ nitrate removal would be expected to vary seasonally in these regions, depending on factors such as temperature, the extent of saturated conditions and the flux of nitrate through the ecosystems.

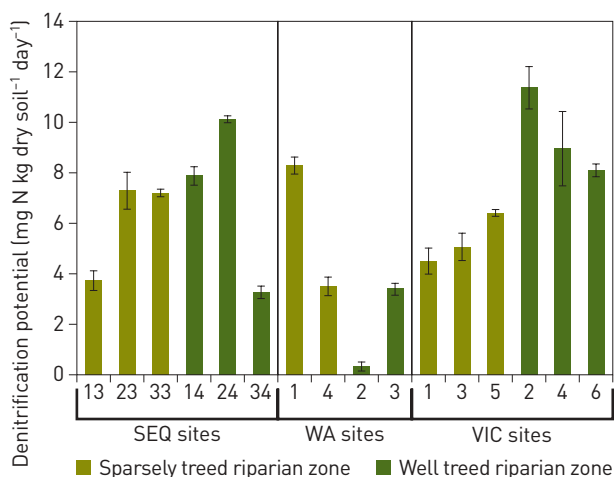


Figure 5. Denitrification potential of surface riparian zone soils from three different regions (dry season). Sites are categorised by riparian zone vegetation (sparsely treed = grass, few shrubs or trees; or densely treed). Values are the mean and standard error of three replicates. Soils received nitrate to increase the concentration by 3 mg N L^{-1} .

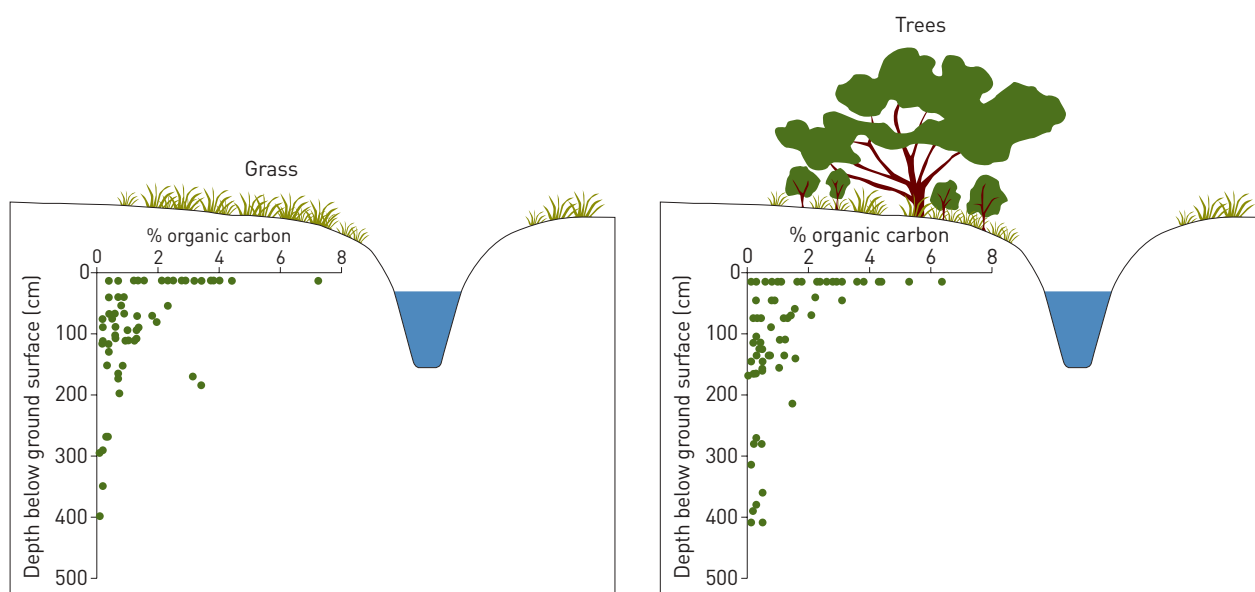


Figure 6. Soil organic carbon concentrations (per cent by weight) with depth for riparian zones with vegetation that is predominately trees or grass. Note that the concentrations of organic carbon and distribution with depth are very similar between the two vegetation types.



Site hydrology and chemistry

Depth to groundwater differed across the regions, in part due to differences in soil type and stream bank morphology. The groundwater table was quite close to the surface of the duplex soils in WA, due to the presence of a relatively impermeable layer (mean depth to groundwater = 40 cm (in riparian zone) and 50 cm (outside riparian zone)). The depth to groundwater in SEQ was greater than 1 m for most of the year (1–5 m across all sites), in part due to high stream banks associated with deeply incised streams. Although groundwater depths were not monitored in wells in VIC, the low bank heights (vertical distance from the stream bank to the water surface of around 0.2–1.6 m) at these sites suggests that the groundwater was closer to surface than at the SEQ sites.

At all the sites, the potential denitrification rate was highest in the 0–30 cm soil layer both inside and outside the riparian zone, but this soil layer is rarely saturated in SEQ due to the deep water table. This area of highest denitrification potential is more frequently saturated in WA, and presumably VIC, where the water tables are closer to the soil surface. The majority of sites in SEQ and WA had no stream flow during the dry season, with several sites having completely dry stream beds.

Top: Deeply incised stream with high banks, SEQ.

Photo Susie Green.

Left: Low stream banks, groundwater close to surface, VIC.

Photo Darryl Holland.

Below left: Low stream banks, groundwater close to surface, VIC. Photo Darryl Holland.

Below right: Dry stream bed in SEQ. Photo Carol Conway.



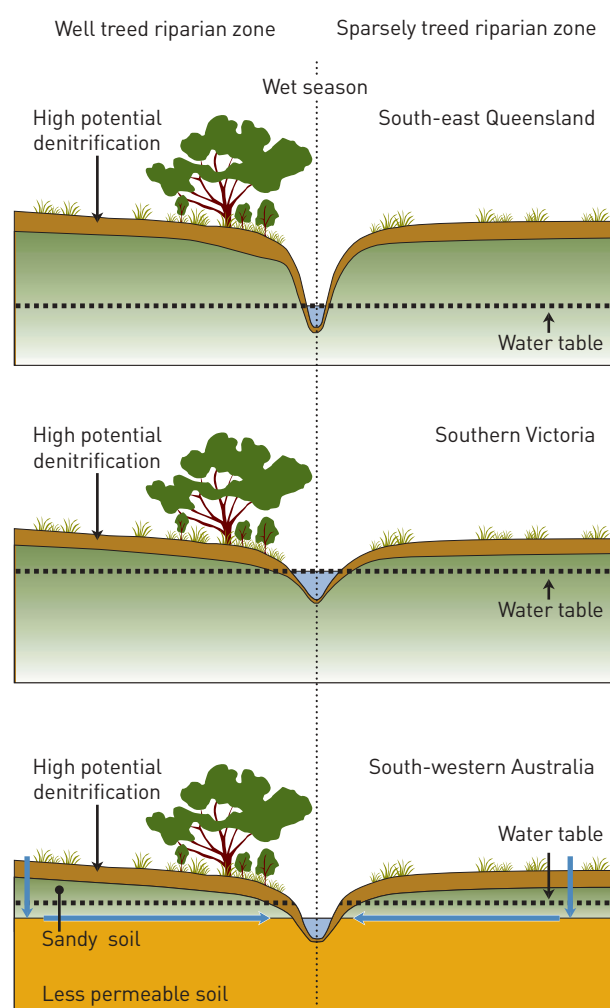
		Surface water			Groundwater			
					Field (outside riparian zone)		Riparian zone	
	Region	VIC	SEQ	WA	SEQ	WA	SEQ	WA
NH ₃	Median	0.018	0.020	0.140	0.023	0.300	0.061	0.410
(mg N/L)	Range	0.007–0.037	< 0.007–0.331	0.010–1.080	< 0.007–1.832	0–1.690	< 0.007–1.400	0.060–1.470
	# samples	112	24	112	24	88	95	111
	Period of record	Jan '04–Mar '06	Aug '04–Jun '05	May '04–Sept '05	Aug '04–Jun '05	May '04–Sept '05	Aug '04–Jun '05	May '04–Sept '05
NO ₃	Median	0.680	0.003	0.050	0.036	0.190	0.005	0.090
(mg N/L)	Range	0.230–1.310	< 0.002–0.057	0–0.270	0.002–0.877	0–2.790	< 0.002–0.356	0.010–0.760
	# samples	112	24	110	24	88	95	112
	Period of record	Jan '04–Mar '06	Aug '04–Jun '05	May '04–Sept '05	Aug '04–Jun '05	May '04–Sept '05	Aug '04–Jun '05	May '04–Sept '05

Table 1. Surface and groundwater chemistry at the study sites (# samples = the number of samples on which observations are based).

Surface water concentrations of nitrate were much greater at the VIC sites than the other two regions, with a median value more than 10 times higher (Table 1). Groundwater concentrations of ammonium were generally greater in WA than in SEQ, especially for groundwater outside the riparian zone. This may be due in part to the groundwater table being perched above the relatively impermeable layer of the duplex soils.

Combining the findings of nitrogen cycling process rate measurements and insights gained into site hydrology and chemistry, the original conceptual models were revised to include patterns of observed denitrification potential, soil organic carbon content, and depth to groundwater (Figure 7).

Figure 7. Revised conceptual models portraying the interactions between hydrology, riparian zone vegetation, and soil organic carbon for the three study regions. Soil organic carbon is highest in surface soils and decreases with depth (shown by intensity of green shading). In all three regions, the surface soils had the highest rates of denitrification potential (shown by the brown band at top), followed by in-stream sediments. Deep groundwater tables in SEQ result in the surface soils being saturated infrequently, and in-stream denitrification most likely dominates (except when streams are dry). Shallower groundwater in VIC and WA results in more frequent saturation of the surface layer, and a potentially larger role for riparian zone denitrification. However, soil and in-stream rates of denitrification potential are lower in WA than in VIC.



Assessing the relative importance of riparian and in-stream nitrogen removal

At any site, the actual removal of nitrate by denitrification reflects the denitrification potential, moderated by factors such as the extent of saturated, low oxygen conditions, and the flux of water and nitrate through the site. While actual removal will vary from site to site, and over time within a site, some generalisations can be drawn about the relative importance of riparian versus in-stream nitrate removal.

The relative contribution of a particular soil/sediment zone within a site, to total nitrate removal, depends on the spatial extent of exposure to the nitrate flux, the denitrification rate and the period of time over which denitrification occurs. At all sites, the potential denitrification was highest in surface soils, but these zones were the least likely to become saturated. Deep and mid-profile soils were more likely to be saturated, but generally had very low rates of denitrification. Benthic and hyporheic sediments were also likely to be saturated for extended periods of time, and had intermediate rates of denitrification potential, suggesting that they had the ability to make significant contributions to nitrate removal at most sites. Depending on groundwater flow at the site and the sources of nitrate, even the deep soil and mid-profile depths could contribute substantially to nitrate removal due to their large volume, compared to in-stream sediments.

In the Victorian sites, high rates of soil denitrification potential, coupled with groundwater relatively close to the surface, should result in the riparian zone dominating nitrate removal. However, despite a high potential for nitrate removal, in-stream concentrations of nitrate were high at these sites. These high concentrations are attributed not only to the intensive agriculture and horticulture, as well as the number of septic tanks in the Woori Yallock catchment, but also to the presence of acacias, which can fix nitrogen in the root zones, and then undergo nitrification to form nitrate. For example, nearby Lyrebird Creek, which is almost pristine, has a median nitrate concentration six times higher than the south-east Queensland and Western Australian sites in this study due to the large number of wattle trees in the riparian zone. While it was beyond the scope of this project to determine catchment sources of nitrate and transport pathways, in-stream concentrations of nitrate presumably would have been even higher without the influence of in-stream and riparian zone denitrification.

Alternatively, nitrate may be transported to the streams along pathways that by-pass the influence of the riparian zone. High to intermediate rates of denitrification

potential in riparian soils and relatively deep groundwater tables in south-east Queensland highlight the importance of in-stream processes for removing nitrate, with the exception of high stream stage, when soils closer to the surface become saturated. Selected sites in Western Australia generally have groundwater tables close to the surface, but typically have low rates of denitrification potential in surface soils and stream sediments. Soil/sediment zones which provide more favourable moisture regimes during the dry season may support a more active microbial community, resulting in higher denitrification potential when stream flow and saturation do occur.

Management implications

Incorporating denitrification rates into catchment water quality models

Information on the denitrification potential of riparian soils at the Victorian, south-east Queensland and Western Australian sites has been included as part of a “look-up table” for use in the Riparian Nitrogen Model (RNM), a filter (plug-in) module within the catchment-scale water quality model, E2 (<http://www.toolkit.net.au/>), that allows users to estimate the amount of nitrate removed by denitrification in riparian buffers. Model users can choose values from the table for sites most similar to the conditions they are modelling, in terms of soil texture, organic carbon content, vegetation, and region. The RNM is also available as a stand-alone model, and includes the Riparian Mapping Tool that allows a finer scale analysis to target specific stream reaches for rehabilitation (Rassam et al. 2005b, Rassam & Pagendam 2006a). For specific model applications, site-specific data may be needed from laboratory measurement of denitrification potential. An alternative, simpler approach may be to assess nitrate concentration and a measure of soil organic carbon as a more rapid and less expensive estimate of potential.

In addition to rates of denitrification potential for a wide range of sites, this project has provided evidence of relationships between denitrification potential and influencing factors that will be useful in modelling nutrient cycling and water quality at a range of scales. These relationships include decreasing denitrification potential and soil organic carbon concentration with depth (assumptions used in the RNM), and increasing denitrification potential with increasing background nitrate concentrations.

Table 2. Guidelines for management of riparian lands to optimise their denitrification potential (adapted from Hunter et al. 2006).

Focus	Management approach
Protect and/or increase levels of bio-available organic carbon in riparian soils, including those at depth	• Maintain a mix of vegetation types (trees, shrubs and grasses), species and ages to provide a range of: – rooting depths and rooting densities – litter types – decomposition rates
	• Minimise soil disturbance, e.g., due to: – livestock – vehicles – weed removal – revegetation
Identify riparian areas to target for rehabilitation	• Identify areas with optimal duration and extent of saturation: – low lying – relatively flat – low stream banks – soils of moderate hydraulic conductivity
	• Assess the potential diffuse sources of nitrate in the catchment. For example, higher loads can be expected from areas with the following land management practices, compared with less developed parts of a catchment: – extensive use of nitrogen fertilisers – intensive livestock production – use of septic systems in residential areas
	• Assess the type and condition of existing vegetation in the areas that meet the above criteria and determine the relative gains in denitrification potential likely to be achieved by their rehabilitation.

Enhanced guidelines for riparian and stream rehabilitation

Findings from this and other recent research have been used to propose guidelines for the management of riparian lands (Table 2), with the focus on increasing the potential for denitrification and thereby reducing the loads of nitrogen entering surface water bodies (Hunter et al. 2006). While these guidelines can be used to enhance nitrogen removal in riparian zones, it should be emphasised that overall management strategies for nutrients should aim to minimise nutrients at their source.

The first guideline deals with increasing or maintaining soil organic carbon so that a good supply is present to support denitrification when soils become saturated and nitrate is present. Organic carbon should be present throughout the soil profile, especially at the soil depths likely to be saturated, or most frequently, for extended periods. Management approaches are suggested to build up soil carbon reserves and minimise their breakdown and loss (Table 2). The required vegetation buffer width and depth of rooting differs in each situation and depends on factors such as the landscape setting, hydrology and soil type.

The RNM can assist in defining the optimal buffer width and depth for specific sub-catchments. The model was tested in the Maroochy River catchment, SEQ. The suggested widths of ≥ 5 –10 m and rooting depths of ≥ 2 –3 m (Rassam et al. 2005a, Rassam et al. 2005b, Rassam & Pagendam 2006) are consistent with buffer widths proposed by existing guidelines (Table 3). Increasing vegetation and organic matter in the riparian zone also increases the supply of organic matter to the stream sediments.

The second guideline recognises that landscape setting and hydrology are critical in determining the extent of riparian denitrification, even though they are factors that cannot be easily changed. Thus, identifying areas where conditions are most likely to be conducive to denitrification can help focus management efforts on the most suitable riparian lands for protection or rehabilitation. The first steps are:

- identify those areas where groundwater discharge is likely to occur,
- assess the likelihood that groundwater nitrate levels at these locations are elevated (often related to land use), and

- combine these with an assessment of the current condition of riparian vegetation to indicate those riparian zones in poor condition where revegetation is likely to provide the greatest benefits for denitrification (Table 2).

Modelling is one way to assess the above factors holistically, and the RNM (Rassam et al. 2005b) can indicate the sub-catchments where riparian rehabilitation is likely to yield the greatest reductions in stream nitrate loads. It can also highlight stream reaches within these sub-catchments where groundwater discharge of nitrate is most likely to occur (Rassam & Pagendam 2006). Follow-up field inspections are advised to support the model outputs. In addition, further assessment is recommended to consider the implications for other riparian rehabilitation objectives (Table 3) as well as any social or economic concerns and practicalities that may also influence the priorities for management.

The primary focus of existing riparian guidelines (Riparian Land Management Technical Guidelines [Australia, Lovett & Price 1999a, 1999b], Managing Riparian Zones [NZ, Collier et al. 1995], and Principles for Riparian Lands Management [Australia, Lovett & Price 2007]) has been the management of surface processes, including control of stream bank erosion, trapping of nutrients and sediment; and provision of habitat (both in-stream and terrestrial). In general, current recommendations for enhancing surface processes are broadly consistent with the aims of the two guidelines for optimising denitrification (Table 3). For example, recommendations on vegetation density, location and species for surface filtration and wildlife are also likely to maintain or enhance soil organic carbon, and enhancing the riparian inputs of leaves, wood and other detritus to streams as habitat and sources of organic carbon to support food webs, also would support increased in-stream denitrification. Riparian-derived organic carbon can be used by in-stream denitrifying microbes, and modification of flow through the action of large wood and debris dams creates quiescent zones, where silt and organic matter can accumulate and provide “hotspots” for denitrification activity.

There are often multiple objectives that must be considered when making decisions for riparian zone and stream management. For example, in this study, both trees and grass seemed to provide organic carbon that supported denitrification, but these vegetation types have very different characteristics in terms of habitat, provision of shade, etc. Additionally, tree cover can also influence microclimate, moderating extremes of temperature and increasing humidity, which may create a more favourable environment for microbial communities in surface soils.

Conclusions

This research has confirmed the broad applicability of concepts of nitrogen transport and transformation in contrasting riparian and riverine settings across Australia. It has provided new insights about the processes that underlie nitrogen removal through denitrification. As a result, conceptual models of nitrogen processing have been refined and new data made available to augment existing catchment-scale models that simulate the mobilisation, transport and transformation of nitrogen and the effects of management change.

Practical guidelines for riparian zone management of nitrogen have been developed from the findings of this research and other contemporary studies into nitrogen processes. The guidelines focus on enhancing the removal of nitrate through denitrification in riparian lands where sub-surface processes are likely to be important. The guidelines contain two main recommendations:

1. maintain and/or increase organic carbon levels in riparian soils, and
2. identify areas where conditions are optimal for denitrification to occur.

Supporting information in the guidelines provides advice on approaches for achieving these objectives. While the focus of these guidelines is on nitrogen management, they are also supportive of the aims and recommendations of many existing riparian guidelines, particularly those that seek to enhance riparian vegetation — for example, to improve stream and bank stability, stream-shading and temperature control, and terrestrial habitat.

The in-stream and riparian nitrogen dynamics project team are:

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Table 3. Current guidelines for riparian management and their associated sub-surface benefits related to increasing denitrification potential (from Hunter et al. 2006). Note 1: Adapted from Collier et al. (1995) and Lovett & Price (1999b).

Focus	Guideline ¹	Benefits for increasing denitrification potential
Stream and bank stability	• Stagger planting along the top bank as well as on the bank face and near-stream • Diverse root systems are needed to cover a range of erosion processes: – deep and extensive root systems – dense network of medium to small roots to reinforce upper soil • Use a range of native plants	• Provides a source of organic carbon in all of these areas • Differing root depths can provide a source of organic carbon throughout the soil profile • Different plants have different decay rates and provide a range of sources of organic carbon
Reduce contaminants in overland flow	• Riparian width should be 10 m or more from the top of the bank • If riparian land has a steep gradient, a 5 m dense grass buffer zone should be established at the outer edge of the riparian zone	• Provides organic carbon in this area • Slows down surface flow and increases infiltration into the soil and groundwater
Light and temperature	• 75% cover is needed for control of light and temperature • Although target cover can be achieved with a single line of trees, width should be over 10 m for other factors (micro-climate etc.) • Use native trees that are wide compared to their height, have high shade indices and can grow out over the stream	• Provides increased organic carbon in the soil as a result of leaf litter breakdown and roots
Managing inputs of terrestrial carbon	• Plant low, overhanging vegetation (provides terrestrial invertebrates and leaf litter) • To ensure a regular and diverse supply of terrestrial carbon plant a range of native vegetation with: – differing decay rates – differing sizes – differing growth rates	• Provides stream organic matter • Provides an organic matter source (leaf litter, roots) throughout the soil profiles and over time. Young, actively growing vegetation can take up and store nitrate, while older trees produce more abundant organic carbon from litter (root, leaf decay)
Terrestrial habitat	• Plant a range of native species at mixed densities and combinations • Plant native species that provide differing food and habitat sources • Plant native species with a variety of different life forms (shrubs and groundcover as well as trees) • Plant both long and short-lived trees (aim to have a mosaic of plant communities at different stages of development) • Maximise riparian area (50–300 m wide) as well as links to other riparian lands and bushland • Undertake pest control and control stock access	• Provides a mixture of organic carbon types in different areas • Provides a range of organic carbon types, well-distributed through the soil profile (from different rooting depths) • Provides a continuous supply of organic carbon over time (also see above about younger versus older vegetation) • Greater potential for organic carbon to accumulate • Minimise disturbance of soil
Reduce groundwater flow of contaminants	• Areas of concern (probable groundwater and nitrate input) should be planted with trees or deep-rooted perennial grasses • Plant riparian vegetation in areas of low relief and low gradients (slow groundwater flow) • Plant riparian vegetation in areas which experience seasonal saturation • Width should be 10 m from the top of the bank (buffers up to 50–100 m wide may be required in areas of fast flowing groundwater)	• See Table 1

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Chapter 5 — Managing algal blooms in Australia

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Summary

- Algal growth depends on the availability and supply of the nutrients nitrogen (N) and phosphorus (P), light and warm water temperature. Most inland rivers in Australia are slow flowing and have weirs placed along them for water storage, this slows the flow even further. Rivers often have high levels of turbidity (muddy water) that limits light penetration, and can become stratified with a warm surface layer of water over a colder bottom layer. This combination of low flows, stratification and turbidity favours blue-green algal growth.
- Most of the phosphorus and nitrogen found in rivers, storages and estuaries is located in the bottom sediments that have been eroded from the surrounding landscape over decades since catchments were cleared for agriculture. These nutrients are released into the water column, particularly when the water column becomes stratified (not mixed) and the bottom waters turn anoxic (lacking oxygen), and can be an important factor in the on-set of major algal bloom outbreaks.
- The biggest contributor of phosphorus to rivers in Australian catchments is naturally derived, and strongly associated with soil erosion. While managing point source inputs such as sewerage treatment plants is important, management practices developed to minimise or intercept eroding soil are also likely to minimise phosphorus transport.
- Managing flows in storages can minimise the extent of algal blooms, with options including; increasing base flows, using pulsing flows, using water off-take points near the bottom of reservoirs, limiting organic material entering reservoirs and managing stormwater.
- Different processes can operate in different parts of Australia. For example, nitrogen can play as important a role as phosphorus in controlling the biomass of freshwater algal blooms. Many of the processes involved in eutrophication (algal blooms) are now sufficiently understood for computer models to be developed for such processes as sediment-nutrient release, stratification, turbidity and algal growth in both freshwater and estuarine systems.

Background

Australian science has made rapid advances in the last decade in understanding algal bloom (eutrophication) processes in inland waters and estuaries. The freshwater research upon which these advances are based was triggered by well-publicised blooms of blue-green algae (*cyanobacteria*) during the 1980s and early 1990s, particularly a 1000 km long bloom on the Darling River. In estuaries, the Port Phillip Bay Study greatly enhanced our understanding, and served to stimulate further research into estuarine eutrophication. This study was initially designed to address perceived problems of toxicants in the Bay, but provided profound insights into drivers for, and ecosystem responses to, eutrophication.

Subsequent research on estuarine algal blooms has largely been stimulated by management questions arising from Australia's increasing residential coastal development. The research has shown that some of the beliefs extant at the time of the blooms were incorrect. For example, it is now clear that stratification and light penetration, not nutrient availability, are the major triggers for blooms in the impounded rivers of south-eastern Australia. Nutrient exhaustion does, however, limit the biomass of blooms.

Other findings include work that shows nitrogen plays as important a role as phosphorus does in controlling the biomass of freshwater blooms. The research has also shown that aspects of eutrophication,

such as nutrient transport, are dominated by different processes in different parts of Australia. Many of the biophysical processes involved in eutrophication have now been quantified sufficiently for models to be developed of processes such as sediment-nutrient release, stratification, turbidity and algal growth in both freshwater and estuarine systems. In some cases, the models are reliable enough for the knowledge gained in particular waterbodies to be applied elsewhere.

There is now a firm scientific foundation for managers to rely upon when managing algal blooms. One of the Programs that contributed a great deal of knowledge about eutrophication was the National Eutrophication Management Program and a short summary of the key findings from this Program follows.

National Eutrophication Management Program

The National Eutrophication Management Program (NEMP) 1995–2001 was established following community concern about outbreaks of algal blooms in rivers and lakes across Australia. The Program invested in research and development into ways to reduce the frequency and intensity of harmful algal blooms in Australian fresh and estuarine waters, as well as raising awareness and understanding about the processes that cause these blooms to develop. It was funded by the Land & Water Resources Research and Development Corporation and the Murray-Darling Basin Commission.



Photo Phillip Ford.



Photo above Phil Price. Photo at right Phillip Ford.

Over a five-year period, the Program identified and funded key research and development gaps that covered over-arching eutrophication issues. This research was 'grounded' in four focus catchments: Wilson Inlet (WA), Fitzroy (Qld), Namoi (NSW) and Goulburn-Broken (Vic). The Program concluded in 2001, with a review finding that it had increased understanding about the causes of algal blooms, and provided management techniques that could be practically applied. Some of the management techniques developed through the Program included:

- managing flows to reduce the stratification in the water column that promotes blue-green algal blooms,
- managing light penetration within waterbodies to control blue-green algal growth,
- using bio-manipulation to directly control concentrations and growth of blue-green algae,
- managing sediments in rivers, storages and estuaries so that the anoxic conditions favouring nutrient release and blue-green algae growth are avoided,
- managing nutrients so that they are not entering river systems in 'pulses' and promoting algal growth,
- controlling nitrogen to better manage algal blooms, and
- using tests to determine whether a particular waterbody is nitrogen or phosphorus limited, and developing management strategies accordingly.

Some of the key findings upon which these management techniques were developed are discussed in the following sections.

River flow and blue-green algae

Blue-green algae (*cyanobacteria*) are a natural part of Australia's river systems. When in balance they are not a problem, but increased nutrients and low flows have



contributed to severe algal bloom outbreaks in many of our river systems and water storage areas (see photo above). The conditions that favour algal blooms are now well understood. Algal growth depends on the availability and supply of the nutrients nitrogen (N) and phosphorus (P), light and warm water temperature. These conditions result in blooms of blue-green algae often coinciding with long periods of warm, sunny weather, high nutrient levels, and still water.

Research suggests that another factor that affects algal bloom formation is the management of flows in both rivers and water storage areas. Rivers rarely experience algal bloom outbreaks during periods of high flow. This means that new approaches to manipulating the flow of rivers and water reservoirs may hold the key to preventing algal blooms, saving millions of dollars in water treatment costs and environmental damage caused by algal bloom outbreaks.



Weirs slow flow encouraging algal growth. Photo David Morgan.

Australian rivers generally have low flows and are controlled or regulated for water supplies at different times of the year. During storms and flood events, large amounts of nutrients (nitrogen and phosphorus) are delivered to rivers and reservoirs from surface and subsurface erosion of soils and gully networks. These nutrients are either recycled within the water, or released from the sediments when bottom waters become anoxic (lacking oxygen) as a result of poor mixing between water temperature layers (stratification) and the decomposition of organic matter by bacteria. Algal growth is sustained by nutrients, and once a large flood event is over, the combination of increased nutrients and low flows create ideal conditions for algal bloom outbreaks to occur.

In a national study of 24 rivers in Australia, the links between river flow and blue-green algae abundance were researched and two dominant trends emerged. The first trend, identified in the temperate rivers of New South Wales and Victoria, showed that as flows

decreased blue-green algae abundance increased. The second trend, found mainly in tropical rivers in Queensland, showed that prolonged low flow conditions led to more blue-green algae being present. Notably, there were no instances where blue-green algae were recorded during high flow periods.

Why do low flows favour blue-green algae?

Most inland rivers in Australia are slow flowing due to the very low slope of the landscape. Weirs placed along the rivers to provide water storage slow the flow even further. This creates an environment that encourages blue-green algal growth. During low flows, stratification develops and the water forms layers with a warm surface layer on top of a cold bottom layer. Stratification often develops during the summer months due to high solar radiation during the day. Australia's inland rivers are also quite turbid due to high concentrations of suspended clay. This results in low light penetration through the water, limiting algal growth to the region near the surface.

High nutrient loads favour algal blooms. Photo Guy Roth.





Flood events input large amounts of nutrients into the stream. Photo Ian Rutherford.

The combination of low flows, stratification and turbidity favour blue-green algal growth. When rivers are stratified, a population of buoyant blue-green algae will float into the well-lit water layer close to the surface where they will receive the light necessary for growth. This is in contrast to the non-buoyant algae (e.g. diatoms and green algae) that are distributed throughout the water column, often in the dark where they will not survive. The ability of blue-green algae to stay at the surface means that in still warm waters they can grow to sufficient numbers to develop surface scums and cause management problems. For these reasons, blue-green algae tend to bloom when flow is reduced and stratification occurs.

During high flows, the turbulence caused by the flow over the river bottom is often strong enough to mix the entire water column from top to bottom. Other non-buoyant algae are dominant in these higher flows, as they are heavier than water and require well-mixed conditions to stay in suspension. They are also adapted

to low-light conditions, can grow better and successfully compete against blue-green algae. Once high flows recede, and the water column stratifies, they slowly sink to the bottom away from the light and cease to grow.

Algal growth and flow management

Algae grow rapidly, and under favourable conditions it takes very little time for a population to reach nuisance levels. A typical blue-green algae population can start at 100 cells/mL and reach 10,000 cells/mL in around 10 days. This rapid growth over a short time period means that the length of low flow period associated with persistent stratification is critical to monitor.

Computer models can now be used to assess whether a river section is likely to be stratified or mixed under different flow and weather conditions. These models have been applied to the Murray and Murrumbidgee rivers of south-eastern Australia, and have been extended to coastal Queensland rivers such as the

High flow mixes the water column decreasing algal growth. Photo Roger Charlton.





Water column testing equipment. Photos Myriam Bormans.

Fitzroy River. One of the main outcomes of the projects on the Murrumbidgee and Murray Rivers was the development of a 'mixing criterion' for turbid rivers. This criterion is a numerical value which can determine the time for the on-set of stratified and mixed conditions. It is based on estimates of river depth, flow, solar radiation, depth of light penetration and wind speed. Whether a river is stratified or not is determined by the relative balance between solar radiation (which has a tendency to stratify the system), wind, evaporation and flow (which have a tendency to mix the system).

The application of this model in the weir pools of the Fitzroy River Basin confirmed that the management of flows has very important implications for the control of blue-green algal blooms. In the Dawson River, a



Phragmites along edge of wetlands. Photo Roger Charlton.

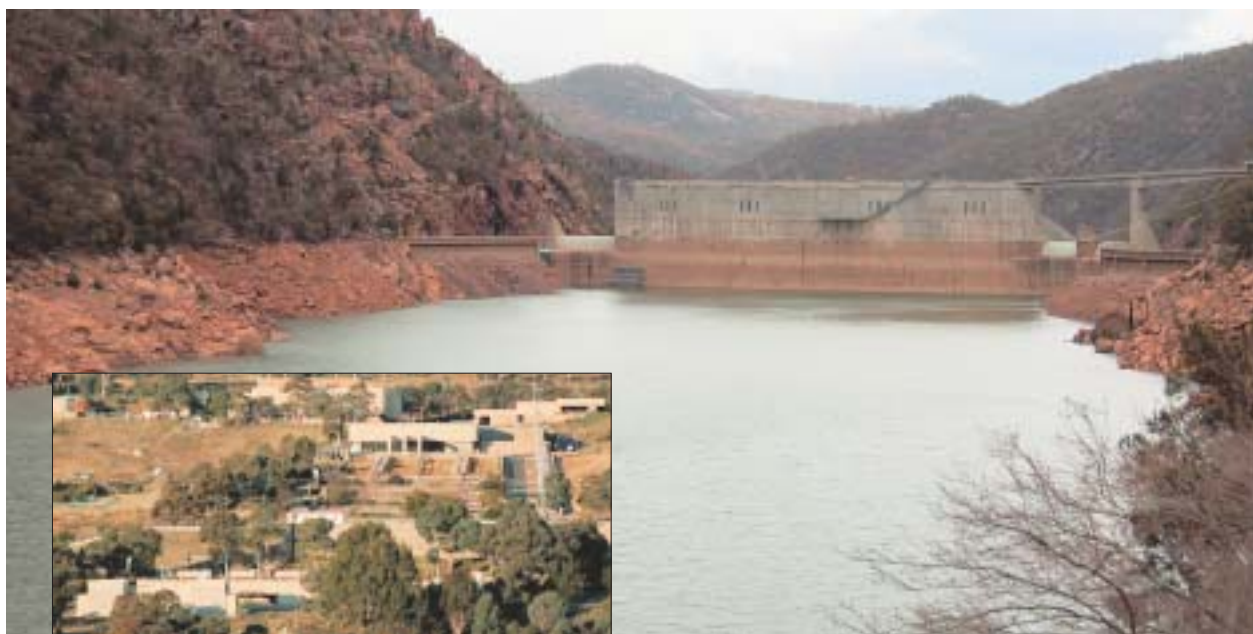


tributary of the Fitzroy, managed flow releases increased the turbidity of the river for sufficiently long periods to decrease the light available for blue-green algal growth, even when the flow releases contained high nutrients. It was found that daily flows greater than the capacity of the weir pool were required to ensure mixing of the entire water column, thereby removing the stratified conditions required for blue-green algal growth. In contrast, ad hoc flow releases had a short term impact (days) on light conditions and algal growth as they failed to mix the entire water column.

Flow management in reservoirs

Blue-green algae outbreaks also commonly occur in water storage areas and reservoirs. Stream flow plays a key role in determining the pattern and form of nutrients discharged into reservoirs. Under high flow conditions, nutrients will be discharged into the reservoir as organic material. The mixing of nutrient-rich (nitrogen and phosphorus) bottom waters into the warmer surface layer may occur as a result of strong winds, autumn cooling of surface waters, or rapid drawdown of reservoir water levels. Nutrients can also build up in the surface layer when wastewater effluent is discharged directly into surface waters of reservoirs.

The relationship between nutrient availability and blue-green algal growth means that it is important to limit the amount of organic material from catchments entering reservoirs. Reservoir inlets need to be managed so that they either prevent organic matter entering the reservoir or, if it does, disperses it over as wide an area as possible to prevent the water turning anoxic (no oxygen). Oxygen levels can also be improved by allowing the growth of plants (e.g. *Phragmites* spp.) along the edge of the reservoir, or mechanically mixing the water. If the level of organic material loading cannot be immediately reduced, it may be necessary to use chemicals containing nitrate to prevent decomposing organic material creating anoxic conditions in the bottom water sediments.



Burrinjuck Dam, with insert photo of Canberra Sewerage Treatment Plant. Photos Roger Charlton.

Burrinjuck storage in New South Wales near Canberra has a unique record of water quality data stretching over 18 years, including a period of nitrogen and phosphorus removal from in-flowing waters due to upgrades of the Canberra Sewerage Treatment Plant. A detailed study of this data has shown that the nutrients that fuel the blooms are most likely to come from bottom sediments, rather than directly from in-flowing waters. Research found that it is the low availability of nutrients that limits the biomass of the algae, and not other factors such as light. Organic carbon coming from the upstream catchment appears to be driving the release of nutrients from the bottom sediment within the storage. Measurements showed that overall algal biomass reduced after phosphorus was removed from Canberra effluent. In addition, there was a switch from harmful blue-green algae to more acceptable green algae. Based on this work, scientists and storage managers have developed guidelines for implementing different management options.

Recommended management practices

Whilst the findings from the NEMP have already been presented to managers and communities throughout Australia, there is still a considerable way to go before they are absorbed into their daily operational procedures (Davis & Koop 2006). However, there are a number of options for managing flows to minimise the extent of algal blooms, including:

1. **Increase base flows.** Maintain sufficient base flows through weir pools to prevent thermal stratification from occurring. Increased base flows

will also ensure the water in the weir pool is mixed and turbid, thereby eliminating the light and temperature conditions favourable for blue-green algal growth.

2. **Use pulsing flows.** Release pulses of flow into weir pools that are of sufficient size and duration to cause mixing of the water from the surface to the bottom.
3. **Use water off-take points near the bottom of reservoirs.** Water supply off-take points near the bottom of reservoirs are where blue-green algal concentrations are likely to be much lower than near the surface. Sometimes this management practice has to be balanced against possible water quality problems (e.g. high colour due to manganese, bad odours due to hydrogen sulphide, cold water).
4. **Limit organic material entering reservoirs.** In reservoirs, it is important to limit the organic material discharge (direct and indirect) from catchments into reservoirs. Priority should be given to reducing wastewater effluents, organic fertilisers, and leaves from deciduous trees that promote blue-green algal growth. Managers should also ensure that when well-nitrified wastewater effluent or drainage is discharged, it is low in ammonia and organic material.
5. **Stormwater management.** High stormwater discharges from urban areas increase the amount of organic material deposited in the reservoir inlet. Various stormwater management techniques such as infiltration and buffer zones can be used to reduce this problem.

Managing phosphorus in catchments

Phosphorus is an essential component of all plants and animals, and is a natural part of the rocks that comprise the earth's crust. While phosphorus is a natural and vital nutrient in our ecosystems, changes in land use (e.g. intensive agricultural development) have radically altered the amounts of phosphorus being delivered to our waterways, particularly river courses, reservoirs and lakes. Excessive nutrient loads in these water bodies can cause eutrophication, a process leading to deteriorating water quality and the increased occurrence of toxic and unsightly algal blooms such as blue-green algae or cyanobacteria.

There is a close relationship between how land is managed and the impact phosphorus may have on in-stream health. In order to manage Australian waterways effectively, we need to determine the relative importance of the various sources of phosphorus, as well as understand the processes by which phosphorus is delivered into our rivers. Knowledge about how and why phosphorus gets into waterways can help land and water managers make better management decisions.

Changes in land use have altered diffuse nutrient loads. Photos this page Peter Hairsine.

Sources of phosphorus in Australian catchments

Phosphorus enters our rivers and estuaries from a number of different sources. The relative significance of each source varies from place to place, depending upon such factors as land use, geology, population density, rainfall intensity and erosion. Sources of phosphorus include:

- 'point' sources such as sewage treatment plants, intensive animal industries and irrigation and storm-water drains,
- 'diffuse' sources such as soil and fertiliser runoff, and phosphorus-rich soils from eroding gullies.

As a general rule, point sources are often believed to be the major contributor of phosphorus to waterways in urban environments. Diffuse sources dominate in rural environments, with agricultural fertilisers and animal effluent previously considered the major sources of phosphorus entering rivers in these areas.

However, research in Australia suggests that only where there are high population densities and intensive agriculture do we see strong evidence that phosphorus comes directly from sewage, fertilisers and animal wastes.





A dramatic example of gully erosion, NSW. Photo Gary Caitcheon.

It has now been shown that the biggest contributor of phosphorus in Australian catchments is from diffuse sources, and this is strongly associated with soil erosion. This is because changes in land use have altered diffuse nutrient loads by increasing soil erosion rates. Increased soil erosion results in significant increases in the transport of naturally derived nutrients, such as phosphorus, into our waterways.

The amount of phosphorus available for transport to our waterways will depend upon:

- the geology of the catchment,
- the overlying soils and their natural phosphorus concentrations,
- landuse type and intensity, and
- the nature and magnitude of the erosion process.

The weathering and break down of different rock types results in varying amounts of natural phosphorus being present in the overlying soils. Some rocks, such as basalts have naturally high amounts of phosphorus. Other rock and soil types are more susceptible to erosion and gully development. When these soil types are cultivated, increased gully erosion can occur, with this interaction responsible for delivering large amounts of phosphorus into our river systems.

Research has used radio-isotopic tracers that act as sensitive markers for nutrients from fertilisers to enable their movement and ultimate fate to be recorded. The results showed that for a typical catchment in northern New South Wales with dryland agriculture, fertilisers were a negligible contributor to the phosphorus attached to sediment particles found in the river system. Most of the phosphorus comes from natural stores in the basaltic soils of this area and is liberated by sub-surface and surface soil erosion.

In contrast, however, in irrigated pasture areas such as in the Shepparton district of northern Victoria, or on the sandy soils of Western Australia, researchers discovered a significant contribution to the river phosphorus load from applied fertilisers. This work has highlighted the need for management strategies to take account of local conditions when developing approaches to phosphorus management.

Bio-availability of different forms of phosphorus

For many years there has been a belief that phosphorus from sewage is more readily taken up by algae than phosphorus attached to soil particles originating from erosion in catchments. If true, then this would support



A mass failure event resulting in large scale streambank erosion, NSW. Photo Gary Caitcheon.

arguments for upgrading sewage treatment plants as the highest priority since, kilogram for kilogram, this source of phosphorus provides more fuel for algae. However, a project in the Namoi area of New South Wales showed that there was very little trace of phosphorus from the Narrabri Sewerage Treatment Plant within 20 kms downstream of the outfall. This meant that the discharged phosphorus may have had a local effect, but it was a very small contributor to the overall downstream phosphorus load in the river.

Laboratories have long used the total phosphorus and total nitrogen concentrations in rivers as the standard measure for nutrient levels. However, research has now shown that most of the phosphorus is bound to sediment particles and, depending on the sediment characteristics, the phosphorus can be more or less available to fuel algal growth. Total phosphorus and total nitrogen are, therefore, poor measures of the potential for algal blooms; with bio-available phosphorus and nitrogen providing much better measures. A technique has been developed to determine whether nitrogen or phosphorus is controlling the growth of blue-green algae in a particular waterway. This is important, because it indicates whether managers should be trying to reduce phosphorus or nitrogen levels.

Phosphorus and the erosion process

Phosphorus moves through the landscape either dissolved in water, or attached to soil particles (particularly fine clays) that are carried along by the water. Soil eroded from the surface of hillslopes, or from the beds and banks of gullies and streams, can also carry phosphorus to streams.

Surface ‘sheet’ or rill erosion, can produce clay-rich sediments that are often high in phosphorus either naturally, due to increased biological activity and organic matter, or through fertiliser applications on cultivated soils. Surface erosion is especially important in areas with high rainfall intensities (such as the tropics in northern Australia) and where soils are intensively tilled. Over 85% of the sediment-bound phosphorus in far north Queensland is derived from hillslope erosion of surface soils (ANRA 2000).

In other areas subsurface erosion may dominate and phosphorus will be released primarily through gully erosion and the erosion of stream banks. Approximately 50% of the sediment-bound phosphorus in the Murray-Darling Basin and other catchments in New South Wales and Victoria, is derived from a combination of gully and channel erosion (ANRA 2000). Subsoil phosphorus concentrations are generally lower than those of surface soils, but the phosphorus from this source often dominates because of the greater mass of sediment eroded from gully erosion and channel collapse on floodplains.

In parts of the Murray-Darling Basin where gullies are widespread, most of the soil-bound phosphorus is washed from the gullies during large storms. The phosphorus in these gullies is naturally occurring and normally not influenced by fertiliser application. Although the worst of the gully erosion occurred decades ago, active gullies still continue to add phosphorus to our river systems.

Broadly speaking, catchments with a high drainage density (length of stream per unit area of catchment that can be increased through extensive gully development) and high natural phosphorus concentrations (due to

geology and soil type), will generate high sediment yields and total phosphorus. A significant amount of the phosphorus will be derived from erosion within the channels. In contrast, catchments with low drainage density and low natural phosphorus will have low overall sediment and phosphorus entering rivers. Under these conditions, a greater proportion of this material will be delivered from surface erosion.

Role of bottom sediments and their need for management

Most of the phosphorus and nitrogen found in rivers, storages and estuaries, is located in the bottom sediments that have been eroded from the surrounding landscape over decades since the catchments were cleared for agriculture. These nutrients are released into the water column, particularly when the water column becomes stratified (i.e. not mixed) and the bottom waters turn anoxic. A project in Wilson Inlet, Western Australia, measured the flux of nutrients from the sediments of that estuary and showed that about seven times as much nitrogen comes from these sediments as from fresh river inflows.

The project also showed that the sediments act as a trap for the phosphorus entering the estuary from river inflows i.e. there is a steady build-up of phosphorus in the sediments each year. This research shows that if there is an extended period of anoxia in the estuary, there could be a large release of nutrients from the bottom sediments that would fuel a major algal bloom.

Previous episodes of erosion and transport have delivered phosphorus to our waterways, with the phosphorus-rich sediment being stored at the bottom of riverbeds and in shallow lakes. Research has shown that phosphorus release from low-oxygen sediments in riverbeds is an important factor in the on-set of some major algal bloom outbreaks.

Depending on the conditions, the sediments of shallow aquatic systems are important sources of nutrient regeneration, as well as acting as temporary or near-permanent sites of nutrient storage (sinks). Sediments can act as phosphorus sinks under aerobic (oxygen rich) conditions because oxygen is freely available to the microbes living in the sediment. However, when bottom waters and sediment becomes anoxic (lacking oxygen), microbes release phosphorus into the water column. Microbes affect the release of phosphorus through their respiration, and this reduces oxygen concentrations in bottom waters during periods of temperature stratification (layers in the water column — warm on top and cooling with depth). This process also occurs when there are high organic loadings in the water.

Management practices to control phosphorus transfer to streams

Most phosphorus is transported attached to particles of soil. This means that management practices developed to minimise or intercept erosion (whether surface or subsurface) are also likely to minimise phosphorus transport. The following management practices can help reduce the generation and delivery of phosphorus in our catchments.

1. **Focus on controlling diffuse sources of phosphorus** (e.g. such as stream bank and gully erosion) that contribute substantial proportions of sediment and phosphorus.
2. **Stabilise stream banks and control stock access** to reduce the risk of bank collapse.



Photo Michael Askey-Doran.

3. **Develop engineering structures** (e.g. contour banks, gully sediment traps, artificial wetlands, farm dams) to reduce on-site erosion and sediment delivery.
4. **Use riparian grass filter strips to trap sediment and attached nutrients** before they can reach the stream.



Photo Roger Charlton.

Photo Michael Askey-Doran.



5. **Manage erosion in high flow events** to control the transport of phosphorus to downstream reaches and receiving waters.
6. **Develop practical methods to reduce flood peaks and volumes** by building appropriate conservation structures such as surface retention basins to store upland rainfall in the landscape/soils and by managing groundcover during these wet seasons.

Photo Roger Charlton.



7. **Time pasture management** (e.g. top dressing, pasture improvement) to limit the transfer of phosphorus-rich soils to streams.
8. **Manage grazing** (e.g. stocking rate and intensity) to limit erosion of soils due to stock tracks, groundcover destruction and excess surface erosion.
9. **Provide stock watering and shade away from drainage lines** to limit destabilisation and erosion of stream.

Conclusion

This chapter has highlighted the key findings from the NEMP, further research has now been undertaken and is outlined in the accompanying chapters of this book.

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Chapter 6 — Making better fertiliser decisions: grazed pastures

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Summary

- Fertiliser is a key input for most of Australia's pasture-based grazing enterprises (e.g. beef, dairy and sheep), with its strong influence on pasture production and profitability.
- Grazing enterprises can be a significant contributor to nutrient pollution of land, water and air. Producers and fertiliser advisors need the best possible information and tools to make better fertiliser decisions if they are to satisfy the goals of profitability, and of sustainability.
- The Better Fertiliser Decisions project was conducted to provide comprehensive information to improve fertiliser decisions for grazing industries across Australia. National in scope, the project compiled and interpreted results from pasture-fertiliser experiments and information on nutrient loss processes from all relevant regions.
- The project results are based on a large amount of data collated from an extensive national review of fertiliser-pasture response experiments conducted in the past 50 years. Sources of this information included peer-reviewed scientific publications, government and industry reports and unpublished data. All experimental data used in the development of the response relationships were standardised and met rigorous quality assurance criteria.
- The project has delivered soil test-pasture response relationships and critical soil test values for phosphorus, potassium and sulphur differentiated at regional, state and national scales, and also by soil characteristics such as soil texture and phosphorus buffering index.
- The project developed an interactive database containing all the data submitted on pasture response to nitrogen, phosphorus, potassium and sulphur fertilisers. The database serves as a comprehensive resource for information about pasture-fertiliser response experiments and provides the capacity to accommodate new data in the future.
- A Farm Nutrient Loss Index (FNLI) was developed, which is a decision support tool to assess the risk of nutrient loss from the paddock to the off-farm environment in the format of a user-friendly computer program.
- The FNLI was developed by collating regionally specific information on nutrient loss processes from scientific publications and existing data, and over 90 nutrient management researchers, extension experts and fertiliser company staff. The FNLI uses easily quantifiable inputs such as landscape features, climatic conditions, and pasture and stock management practices to calculate the risk of nutrient loss at the paddock scale and evaluate the effects of different management practices.

Background

Most Australian soils are old and weathered. In fact many constitute the oldest soils in the world. As a result, most of our soils have an inherently low nutrient status, particularly phosphorus (P), sulphur (S), nitrogen (N), and in the coastal regions, potassium (K). Not surprisingly, fertiliser applications to pasture land have been a routine practice since as early as the 1920s. The application of fertiliser is still considered to be necessary by many farmers to replace nutrients removed, fixed or lost in pasture soils.

However, increased community concerns about excess nutrients in water and the atmosphere, means that farmers and fertiliser service providers need to have access to, and use, the best possible information regarding optimum nutrient management practices for environmental as well as productivity benefits. A more tailored approach to nutrient management, based on the best available information for soil test targets and a greater understanding of nutrient loss processes and pathways, will lead to more efficient nutrition conversion to pasture on farm and reduce excess nutrients in the environment.

This chapter presents soil test–pasture response relationships and interpretations for the major P, K and S soil tests used in Australia, as well as a brief description of the Farm Nutrient Loss Index (FNLI). These results are endorsed by the Fertilizer Industry Federation of Australia and major fertiliser companies in Australia, and will be used in Fertcare, the environmental stewardship national accreditation initiative for the fertiliser industry.

Approach

Pasture response relationships

Soil test–pasture response calibrations define the relationship between pasture production and soil test value. The relationship allows users to predict the pasture production response if the soil nutrient level is altered by the addition of fertiliser. Agronomists from the pasture-based grazing industries of Australia provided experimental results to develop the newly defined pasture response relationships. Over 3000 experimental

Photos throughout this chapter are courtesy of the project team unless credited otherwise.



Definitions

Phosphorus (P) — is one of the three macronutrients required by plants. It has a role in photosynthesis, respiration, energy storage and transfer, cell division and enlargement, genetic coding and many other plant processes. Commonly used fertilisers to supply P are superphosphate and triple super.

Olsen and Colwell P — are laboratory tests to assess the level of P in the soil. Both tests are commonly used in Australia, though regions often use just one of the tests. The tests are poorly correlated.

Nitrogen (N) — is another of the three macronutrients required by plants. It has a role in photosynthesis, and is a constituent of amino acids, proteins and many other compounds. Commonly used fertilisers to supply N are urea and di-ammonium phosphate (DAP).

Potassium (K) — is the last of the three macronutrients required by plants. It has a role in activation of enzyme systems, photosynthesis, respiration, energy storage, and maintenance of protein structure, water use efficiency and many other plant processes. The most common fertiliser used to supply K is muriate of potash, also called potash.

Colwell, Skene and Exchangeable K — are laboratory tests to assess the level of K in the soil. All tests are commonly used in Australia. The tests are highly correlated.

Sulphur (S) — is an essential plant nutrient. It has a role in forming plant proteins and chlorophyll, enzyme activity, and nodule formation and nitrogen fixation in legumes. Commonly used fertilisers to supply S are superphosphate and gypsum.

CPC and KCl-40 S — are the two most commonly used tests to assess soil S levels in Australia. The CPC test uses the extractant calcium hydrogen phosphate with charcoal. The KCl-40 uses the extractant potassium chloride heated to 40°C for 3 hours. The two S tests are not correlated.

PBI — Phosphorus Buffering Index is a test that estimates the P fixing capacity of the soil. PBI is a relatively new test, and is now the national standard for estimating soil P fixing capacity.

Cations are an atom, group of atoms, or compounds that are positively charged as a result of the loss of electrons.

The soil test-pasture response relationships, and the Farm Nutrient Loss Index, collated or developed by this project, are available from the CSIRO Australian Soil Resource Information System (ASRIS) Internet site: www.asris.csiro.au



Figure 1. Pasture-based grazing regions of Australia based on climate, pasture type and irrigation.

years of research results were compiled, some dating back 50 years, including in excess of 250 experiments involving approximately 1600 field sites, with more than 48,000 individual pasture yield measures.

Experiments had to meet certain design, data collection and quality criteria to be included in the analysis. This included a zero application (control) and high application treatment of either P, K or S. Only experiments that used the following Australian soil tests: Olsen and Colwell P; Colwell, Skene and exchangeable K; and CPC and KCl-40 S, were analysed, as there were insufficient data to analyse less commonly used tests. It was not possible to develop soil test–pasture response relationships for N as there is no reliable soil test for N. Soil test sample depth was standardised to 10 cm.

A variety of experiments were conducted.



Soil test–pasture response relationships were prepared where possible, for P, K and S, nationally and differentiated by state, region, soil texture, phosphorus buffering index (PBI) and cation exchange capacity categories. The lack of quality data regarding pasture species, pasture composition, and grazing enterprise meant that soil test–pasture response relationships could not be differentiated by these factors.

The response relationships were statistically compared and significant differences identified. Where no statistical differences occurred, data were pooled to increase the precision of the final response relationship. The pooled national data set provides superior soil test–pasture response relationships for each nutrient. These response relationships are relevant across all grazing regions and livestock enterprises.

How the response relationships were developed

Pasture production data (kg dry matter/ha) were standardised to percentage yield to allow comparison of differences in pasture productivity between locations, seasons and climatic conditions. For each field experiment, the ‘percentage of maximum pasture yield’ was calculated from the zero and high nutrient treatments based on the following equation:

Percentage of maximum pasture yield	=	$\frac{\text{Pasture yield with no nutrient applied} \times 100}{\text{Maximum pasture yield when non-limiting nutrient is applied}}$
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Percentage maximum pasture yield and initial soil test value for each experiment were then used to define soil test–pasture response relationships. These response relationships can be used to determine the likely pasture response at any particular soil test value. Response relationships were specified to have a zero yield at zero soil test level, and to reach maximum potential yield (100%) at a very high soil test level.

Critical soil test value

A ‘critical soil test value’ is the soil test value where 95% of maximum pasture production occurs. These values were established from the soil test–pasture response relationships. The 95% critical soil test value is a simple, commonly used reference point to define where further applications of nutrients are unlikely to provide worthwhile increases in pasture production. The confidence interval around the critical value indicates the reliability of the estimate.

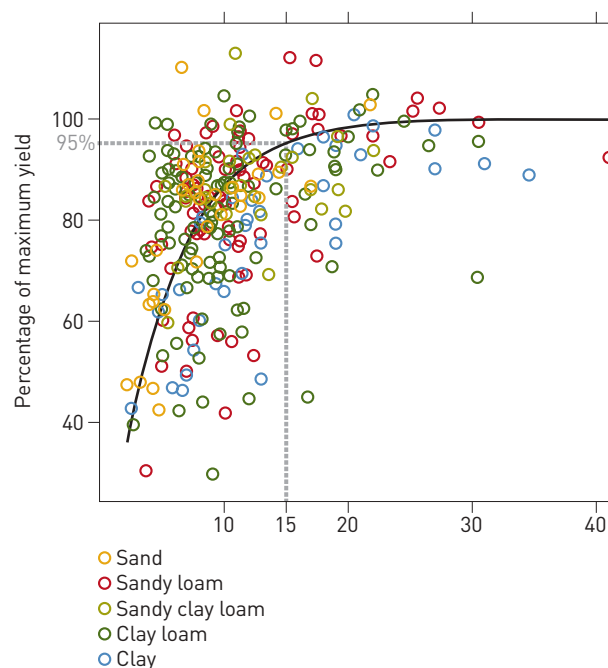


Figure 2. The relationship between percentage of maximum pasture yield and Olsen P soil test value from nationally collated experiments. The critical Olsen P soil test value at 95% of pasture production is indicated by the grey dashed line.

Key findings

Phosphorus

The bicarbonate extraction procedure of Olsen (Olsen P test), and the further modification by Colwell (Colwell P test) are the two most commonly used P soil tests in Australia. These tests differ in the ratio of soil and extracting solution, and duration of agitation, which affects the release of soil-bound ‘fixed’ P. The Colwell P test, with a larger soil extractant ratio and longer agitation time, extracts more fixed P than the Olsen P test. As a result it is well recognised that Colwell P tests need to be interpreted in association with an estimate of the soil’s

P fixing capacity. While soil texture or other measures have long been used as surrogates for soil P fixing capacity, the recently developed phosphorus buffering index (PBI) is now the national standard for estimating soil P fixing capacity.

Olsen Phosphorus

There were no significant differences between the Olsen P soil test–pasture yield relationships when they were differentiated according to states, regions, soil texture and PBI category. Therefore, the relationship based on the entire national Olsen P dataset (Figure 2) is recommended to guide fertiliser decisions (Table 1).

Table 1. The national critical Olsen P soil test value and the equation describing the relationship between Olsen P soil test value and percentage of maximum pasture yield.

Critical value ¹ (mg/kg)	Confidence interval ²	Number of experiments	Equation ³ % maximum yield =
15	14–17	303	$100 \times (1 - e^{0.202 \times \text{Olsen P}})$

1. Soil test value at 95% of predicted maximum pasture yield.
2. 95% chance that this range covers the critical soil test value.
3. e = Euler’s constant [approx. 2.71828].

Olsen P soil test interpretation should be based on the national collation of experiments, rather than soil texture, PBI, statewide or regional relationships.

The critical Olsen P soil test value to achieve 95% of maximum pasture production is 15 mg P/kg.

Colwell Phosphorus

The Colwell P soil test–pasture response relationship showed significant dependence on PBI, but there were no significant differences in the response relationships when differentiated by states, regions, or soil texture.

Twelve PBI classes with equal numbers of experimental data were used to derive soil test value–pasture response relationships for Colwell P. The resultant critical Colwell P values (to achieve 95% of maximum pasture production) and corresponding PBI values were plotted and an equation was derived (Figure 3). This equation enables the critical Colwell P value to be calculated when the PBI of a soil is known. The equation has been used to calculate critical Colwell P values for commonly used PBI categories (Table 2).

Figure 3. The relationship between critical Colwell P soil test value and soil PBI value.

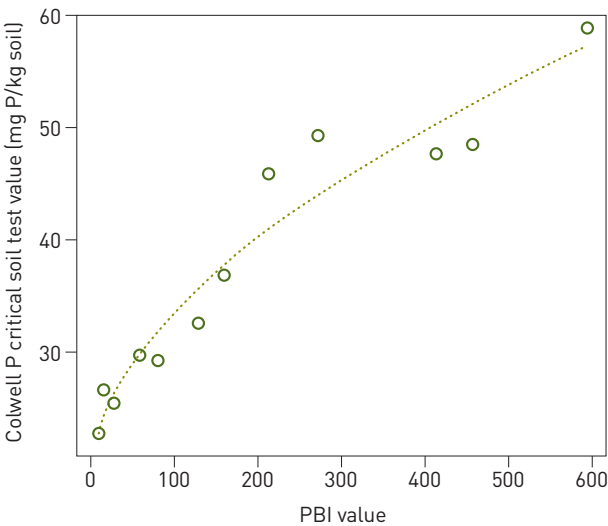


Photo Roger Charlton.

Table 2. Predicted critical Colwell P soil test value for standard PBI categories.

PBI category		Critical Colwell value ¹ for mid point of PBI category (range)
< 15	Extremely low	23 (20–24)
15–35	Very, very low	26 (24–27)
36–70	Very low	29 (27–31)
71–140	Low	34 (31–36)
141–280	Moderate	40 (36–44)
281–840	High	55 (44–64)
> 840	Very high	n/a ²

1. Critical Colwell P value at mid-point of PBI class. Values in parenthesis are critical Colwell P values at the lowest and highest PBI values within the range.
2. Insufficient data to derive a response relationship.

Colwell P critical soil test value = 19.6 + 1.1 x PBI value^{0.5515}

Colwell P soil test interpretation should be based on the soil PBI value, as the critical value increases with increasing PBI.

The critical Colwell P value to achieve 95% of maximum pasture production can be estimated from the PBI categories (Table 2) or the equation (Figure 3).

Potassium

The commonly used Colwell, Skene and exchangeable K soil tests are strongly correlated to one another and, therefore, the national K soil test data were standardised to Colwell K values. There were no statistical differences in the Colwell K–pasture response relationships when the data were differentiated according to state, region and cation exchange capacity class. However, the Colwell K–pasture response relationship did show significant dependence on soil texture class.

The national data were differentiated into five soil texture classes based on clay percentage to derive Colwell K–pasture response relationships and critical Colwell K values. Figure 4 shows the Colwell K–pasture response relationships for four soil texture classes. There were insufficient data to define a response relationship for the clay texture class. The critical Colwell K values and the equations which describe these relationships are provided in Table 3.

Figure 4. The relationship between percentage of maximum pasture yield and Colwell K soil test value for different soil textures. The critical Colwell K soil test value at 95% of pasture production is indicated by the grey dashed line.

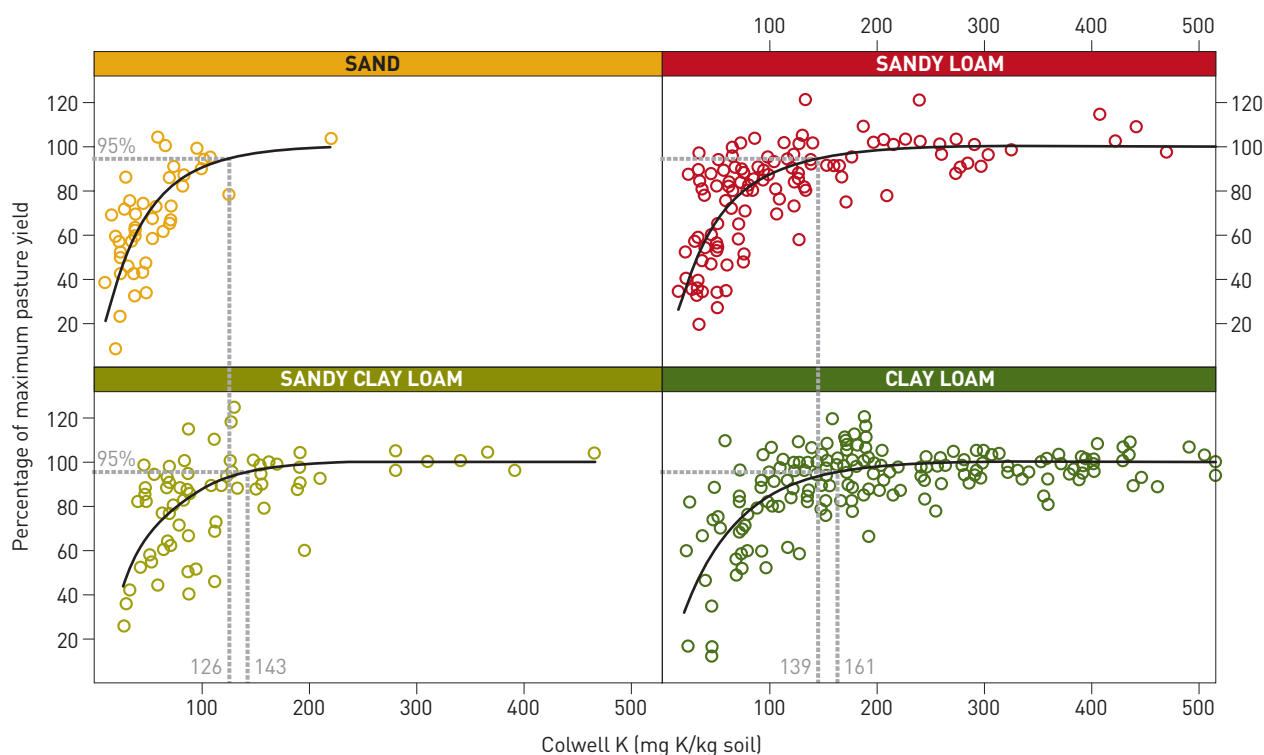


Table 3. The critical Colwell K soil test values for soil texture classes and the equations describing the relationship between Colwell K soil test value and percentage of maximum pasture yield.

Soil texture	Critical value ¹	Confidence interval ²	Number of experiments	Equation ³ % maximum yield =
Sand	126	111–142	194	$100 \times (1 - e^{-0.024 \times \text{Colwell K}})$
Sandy loam	139	125–157	50	$100 \times (1 - e^{-0.022 \times \text{Colwell K}})$
Sandy clay loam	143	127–172	75	$100 \times (1 - e^{-0.021 \times \text{Colwell K}})$
Clay loam	161	150–181	122	$100 \times (1 - e^{-0.019 \times \text{Colwell K}})$

1. Soil test value (mg/kg) at 95% of predicted maximum pasture yield.
2. 95% chance that this range covers the critical soil test value.
3. e = Euler's constant (approx. 2.71828).

Colwell K soil test interpretation should be based on soil texture, as the critical value increases with increasing clay content. The critical Colwell K value to achieve 95% of maximum pasture production for each soil texture class is indicated in Table 3.

Sulphur

Due to the historical widespread use of superphosphate, which has generally provided adequate S for plant growth, there have been fewer S experiments conducted compared with P or K. The two main soil S tests used in Australia are CPC (calcium phosphate plus charcoal) and KCl-40 (potassium chloride heated to 40°C for 3 hours). The two S tests are not correlated and therefore experimental data could not be pooled.

The use of each S soil test tended to be regionally specific, and most S experiments were conducted on clay loam or sandy loam soils. Therefore, there were insufficient data available to investigate whether soil S test–pasture production response relationships differed between soil texture, states or regions.

The S soil test–pasture response relationships for CPC S and KCl-40 S, derived from the national data set, are presented separately (Figure 5 and Table 4).

Figure 5. The relationship between percentage of maximum pasture yield and soil test value for CPC S and KCl-40 S tests. The critical S soil test value at 95% of pasture production is indicated by the grey dashed line.

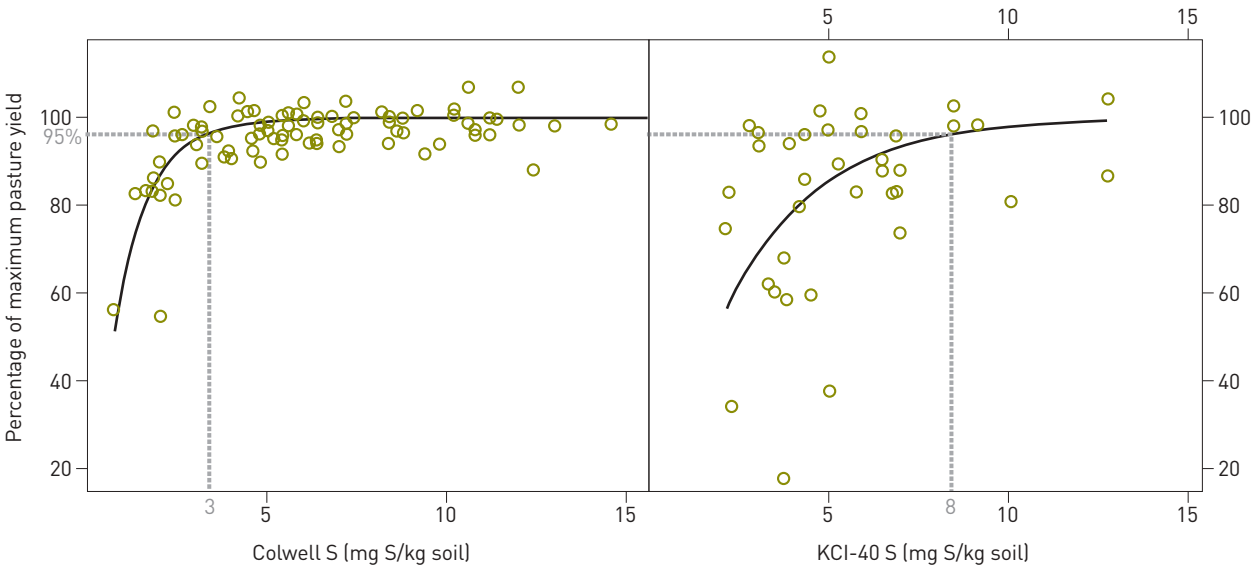


Table 4. The national critical CPC S and KCl-40 S soil test values and equations describing the relationship between CPC S and KCl-40 S soil test value and percentage of maximum pasture yield.

Sulphur test	Critical value ¹	Confidence interval ²	Number of experiments	State ³	Equation ⁴ % maximum yield =
CPC	3	2–4	94	Vic, NSW, Qld	$100 \times (1 - e^{-1.014 \times \text{CPC S}})$
KCl-40	8	6–10	37	NSW, SA	$100 \times (1 - e^{-0.388 \times \text{KCl-40 S}})$



1. Soil test value (mg/kg) at 95% of predicted maximum pasture yield.
2. 95% chance that this range covers the critical soil test value.
3. The two soil S tests were calibrated in different states.
4. e = Euler's constant (approx. 2.71828).

Sulphur soil test interpretation should be based on the national relationships developed for the CPC S and KCl-40 S tests. The critical value to achieve 95% of maximum pasture production for the CPC S soil test is 3 mg/kg, and for the KCl-40 S soil test is 8 mg/kg.

Photo at left: By making better decisions about fertiliser use, creeks like this will be less likely to be contaminated by excess nutrient running off farm paddocks.

The Farm Nutrient Loss Index

Nutrient loss from farms to the environment

Nutrient loss from farms to the off-farm environment can be costly and cause degradation of waterways, groundwater and add to greenhouse gases. The grazing and fertiliser industries in Australia identified a need for a simple and practical tool to help farm advisors identify nutrient loss issues within individual farms.

Understanding the principles of nutrient loss is an important component of integrated nutrient management. The Farm Nutrient Loss Index (FNLI) computer program was developed for nutrient management advisors to use in conjunction with soil fertility testing and nutrient budgeting, so that they can make informed decisions about how to maximise nutrient use efficiency, and therefore minimise negative environmental impacts. The FNLI can also be used to demonstrate the principles of N and P loss from pasture-based grazing systems to the wider environment.

Over 90 nutrient management researchers, extension staff and fertiliser company representatives were consulted in the development of the FNLI. A participatory workshop approach was used to harness regionally-specific scientific knowledge of nutrient loss processes. Focus group meetings and field assessments were conducted to provide technical review and to develop the utility of the FNLI for existing nutrient management advisory services. The FNLI risk outcomes were also validated against measured nutrient loss data from 17 field experiments across Australia.

A User Manual that provides background information on how to navigate through the FNLI software, how the FNLI calculates risks, and the scientific principles of nutrient loss that underpin the Index, is also available from the ASRIS website.



Developing the FNLI was a collaborative process between researchers and practitioners.

How the FNLI works

The FNLI identifies the risk of nitrogen and phosphorus loss from individual paddocks to the wider environment via four nutrient loss pathways: runoff across the soil surface (runoff), drainage past the root zone (deep drainage), lateral flow within the root zone of the soil profile (subsurface lateral flow), and emission of nitrous oxide, which is a powerful greenhouse gas (gaseous emission) (Figure 6). The FNLI is not designed to estimate actual loads of nutrients lost from farms.

Risk of nutrient loss is the combination of the likelihood and magnitude of nutrient loss occurring from a paddock on an average yearly basis. The risk of nutrient loss is influenced by climate, features of the landscape and management of the land. The FNLI identifies the key factors that influence the availability of nutrients ('source' factors), and the transport and delivery of nutrients ('transport' factors). If source and transport factors occur together, nutrient loss will also occur. The important nutrient loss factors for the grazing regions of Australia are shown in Figure 7.

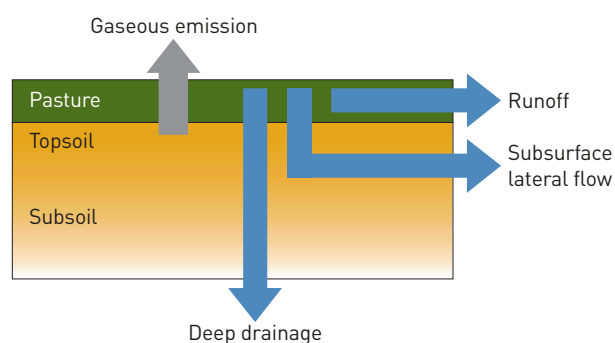


Figure 6. Pathways of nutrient loss.

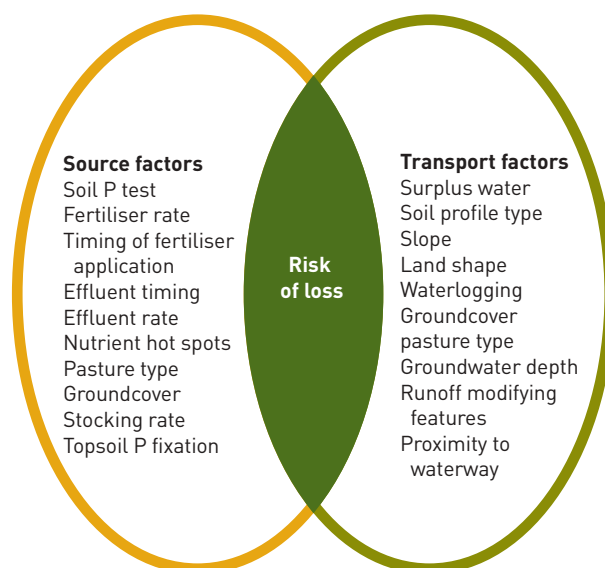


Figure 7. When there is both a source of nutrients and the potential for nutrient transport, there is a risk of nutrient loss.

To use the FNLI computer program, a series of questions about the paddock of interest need to be answered. Users select the options that best match their paddock characteristics and management. The questions can be readily answered from farm records and observation. For each paddock assessed, the FNLI identifies factors that pose a significant risk of nutrient loss and calculates a risk ranking of N and P loss (low, medium, high or very high) for each loss pathway. A paddock report containing the risk results and inputs can be generated (Figure 8).



Farm Nutrient Loss Index Report					
Farm information					
Farm name	Jones	Paddock	South 5	Enterprise type	Dairy
Region	West Gippsland	State	Victoria		
Nutrient loss pathway		Risk ranking	Reasons for high or very high risks		
P in runoff		High	Fertiliser timing, Land shape, Surplus water		
P in subsurface lateral flow		High	Fertiliser timing, Surplus water		
P in deep drainage		Medium			
N in runoff		High	Fertiliser timing, Land shape, Surplus water		
N in subsurface lateral flow		Low			
N in deep drainage		High	Fertiliser timing, Surplus water, Watertable		
N gaseous emission		Medium			
Land characteristics					
Slope		Hilly 6–15%			
Land shape		Converging hillslope			
Waterlogged area		1–10%			
Runoff modifying features		No features present			
Proximity to nearest waterway (m)		40			
Soil profile type		Moderate infiltration but poor drainage			
Groundwater		< 1.5 m			
Topsoil P fixation (PBI)		> 280			
Surplus water score (1, 2, 4 or 8)		8			
Nutrient management					
P test		Olsen			
Soil P (mg P/kg soil)		16– 25			
Fertiliser P rate (kg P/ha)		25–59 annually			
Fertiliser N rate (kg N/ha)		30–60 per application, 100–250 total per year			
Hotspots		Low < 5%			
Timing of fertiliser application		Apply when high runoff or drainage risk			
Effluent applied		No			
Effluent rate		–			
Effluent timing		–			
Pasture management					
Stocking rate		Medium			
Pasture type		Shallow rooted perennials			
Groundcover (%)		90			
Irrigation		No			

Figure 8. The FNLI prepares a report for each paddock showing the input factors, risk rankings and the factors contributing to high or very high risk outcomes.

Interpreting FNLI results

High or very high risk rankings indicate that aspects of the grazing system may need to be modified to minimise potential nutrient loss. Where a high or very high risk ranking is indicated, the main contributing factors are listed. These factors are intrinsic features of the landscape, such as surplus water and soil type, or imposed by management, such as stocking rate. Alternative management practices can be trialled to check strategies aimed at lowering the risk of nutrient loss.

Since the potential for nutrient loss depends on a combination of characteristics specific to each paddock or land management unit, the appropriate management for each paddock can vary. For example, Paddock A and B both have a very high soil fertility (Figure 9), but have a different risk of nutrient loss because paddock B has a drainage line running through it. The FNLI can help land managers identify the risks of nutrient loss on different parts of their farms, and explore nutrient management options which can minimise nutrient losses.

Conclusion

Current information on soil fertility and fertiliser use available to farm advisors and producers is often based on one or two local experiments, and sometimes the basic characteristics of the soils tested are not recorded. Farm advisors and producers require validated information on soil fertility and fertiliser rates that are based on a wider range of data and related to measurable soil attributes; they must also be environmentally sound and profitable. Providing such information for the pasture-based industries was one aim of this NRCP project.

Advisors and producers also need user-friendly information packages and tools that enable them to put this information to effective use by tailoring it to specific situations. The Australian fertiliser and grazing industries want information about fertilisers that is relevant, credible, impartial and scientifically sound. Governments and the community want grazing industries that are competitive in national and international markets, yet operate using environmentally acceptable farming systems.

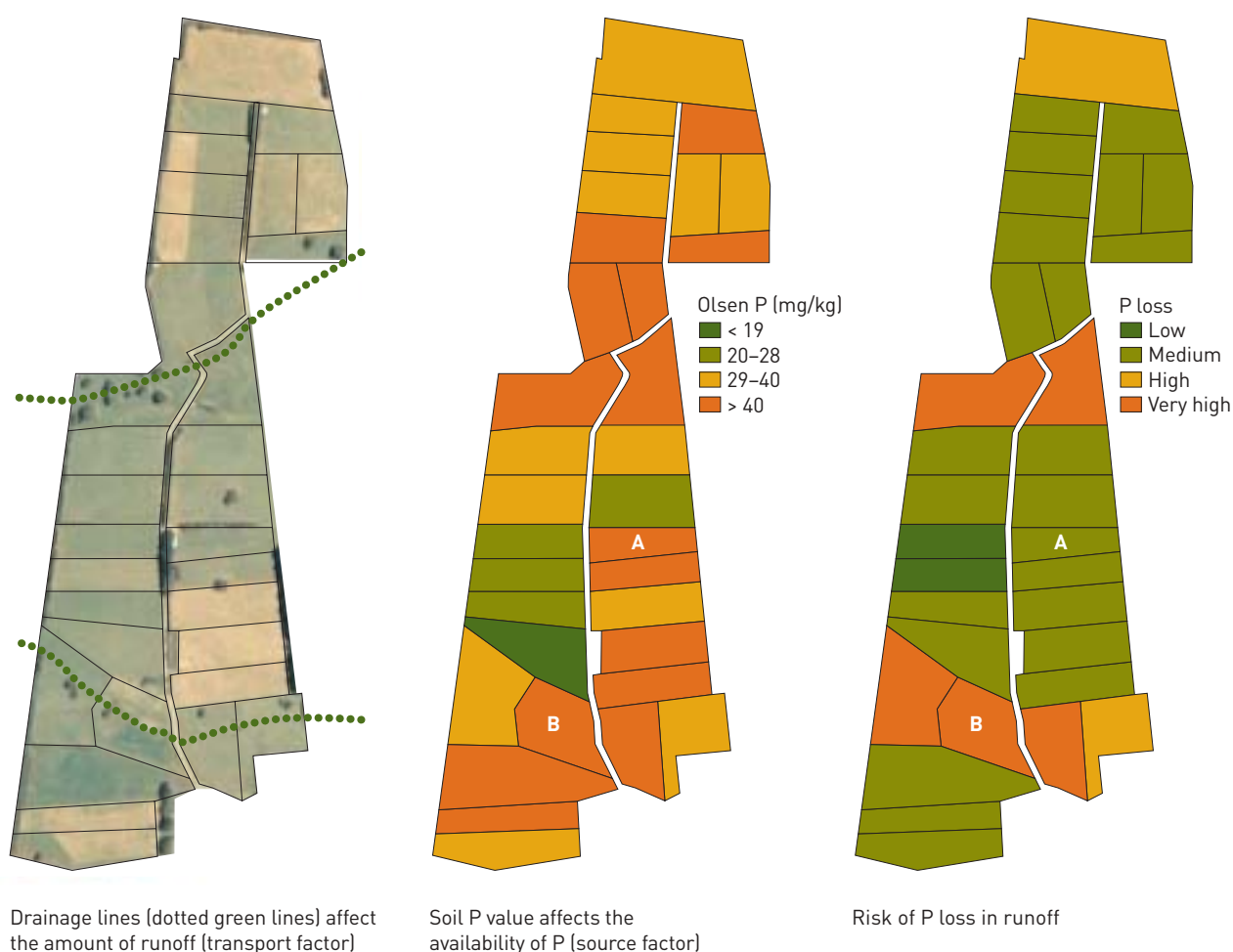


Figure 9. The combinations of transport and source factors across a farm influence the nutrient loss risk.



By using the Farm Nutrient Loss index farmers can work to apply the right rate of fertiliser to improve productivity on-farm and reduce negative water quality impacts on surrounding creeks.

The Better Fertiliser Decisions project has equipped the Australian fertiliser and grazing industries with the information to improve their practices for multi-nutrient management, and to become more profitable and responsible. Providing a sound and justifiable scientific basis for making fertiliser decisions will reduce nutrient losses and improve water quality. There will be greater collaboration between fertiliser companies and the pasture-based industries as well as improved uptake of scientifically sound soil test–pasture response functions.

These direct management implications will increase efficiency of pasture production through optimal and targeted multi-nutrient use by Australian grazing industries, with farmers and advisers more informed about the constraints that landscape characteristics impose on nutrient management. A reduced nutrient surplus in soils and decreased nutrient losses from pastures to groundwater, waterways and the atmosphere is a significant outcome. Other outcomes include potentially reducing the risk of economic and environmental trade barriers to export markets, and providing greater consistency in fertiliser recommendations by Australian fertiliser companies.

Sediment as a contaminant

Globally, sediment is probably the most common river contaminant. While sediments play a beneficial role in the functioning of river systems by providing a substrate for biological and chemical processes, excess quantities of sediments cause a range of problems. The balance between sediment and river flow is also important, and both can change with catchment development and changes in land use. In a sediment-starved river the banks and bed may erode, while excessive amounts of sediment may remain in the river as sand or gravel 'slugs'.

Coarse sediments alter river habitats by infilling pools and destroying these drought refuges, while finer particles can clog bed interstices thus degrading benthic habitat. Large-scale sediment deposition buries entire riffle-pool reaches, replaces diverse river habitats with uniform sand beds, and creates zones of wide shallow flow subject to greater temperature extremes and at risk of invasion by aquatic weeds. Fine sediments that are carried in suspension interfere with the breathing and feeding of many river animals, for example, favouring fish (such as carp) that are not visual feeders. By increasing turbidity, and hence, reducing light penetration, suspended sediments also reduce submerged plant photosynthesis and alter the light regime for phytoplankton. This can favour toxic cyanobacterial species that are able to regulate their cell buoyancy and move into the narrow upper light zone.

Finally, many agrochemicals, heavy metals and nutrients chemically bind to sediments. Consequently, sediments provide a transport mechanism for these contaminants as well as a substrate where they can react. Thus, any complete examination of river contaminants needs to consider both the direct effects of sediment, as well as the role of sediment in transporting and transforming other contaminants. Chapters 7 and 8 address difference aspects of managing sediment within a catchment context.



Photo Phillip Ford.

Chapter 7 —

Identifying sources of sediment in river basins to help develop revegetation priorities

Scott Wilkinson and Cris Kennedy

CSIRO Land & Water

Summary

- SedNet is a modelling tool developed to assess spatial patterns in the sediment sources and sediment transport at the river basin regional scale.
- SedNet has enabled catchment management agencies to target areas for riparian restoration, develop measures to reduce bank erosion, and reduce soil erosion.
- The SedNet tool can be used to identify the primary sources of sediment that is carried by rivers, and to model the relative costs of different management interventions to achieve catchment sediment targets.
- The SedNet software is available as a free download that uses GIS data layers to predict spatial patterns in the sediment and nutrient fluxes, and to identify the upstream sources of impacts on downstream water quality and sedimentation.
- The SedNet software is suitable for use by environmental consultants and natural resource management agencies with GIS, catchment hydrology and erosion assessment expertise.
- SedNet has been applied to develop strategies to reduce sediment and nutrient export in the Great Barrier Reef catchment, Sydney, Gippsland Lakes and Moreton Bay.

Background

The changes man has made to the Australian landscape since European settlement have had a significant impact on our river systems. Sediment eroded from gullies, hillslopes and river banks is transported in faster-moving river reaches, and accumulates in slower ones. In some situations this delivers much-needed nutrients, but in other cases it leads to increasing turbidity that reduces production of plankton and algae, and hence, the amount of food and oxygen available to aquatic life (Davies-Colley et al. 1992, Newcombe & MacDonald 1991, Quinn et al. 1992). Transport of suspended fine sediments brings nutrients as well; in Australian rivers around 75% of phosphorus is transported attached to sediment particles (Young et al. 2001a). Sediment has a similarly important role in the transport of agricultural chemicals and heavy metals (Thoms et al. 2000). Coarser sediment can settle along the stream bed, replacing diverse and stable riverine habitats “with flat sheets of coarse sand and gravel extending for kilometres” (Nicholas et al. 1995, Rutherford 2000).

Trying to redress these changes to the landscape in order to improve water quality and the health of aquatic ecosystems, is a significant challenge for our

land and water managers. The National Land and Water Resources Audit found that suspended sediment loads were typically 10–50 times pre-European levels in many river systems, and that tens of thousands of kilometres of rivers were affected by sand and gravel accumulation resulting from upstream erosion (NLWRA 2000). Organisations across Australia are now trying to improve management of these impacts and in some regions water quality targets have been legislated. SedNet is one tool that can assist the development of strategies to most effectively reduce sediment loads and meet those targets.

The Commonwealth Scientific and Industrial Research Organisation (CSIRO) team that developed SedNet (the sediment river network model) in the 1990s sought to identify the sources and sinks of eroded material in order to identify the dominant erosion processes and source areas within the landscape. SedNet was used to deliver a continent-scale assessment of sediment sources and impacts. Land & Water Australia recognised the potential benefits the modelling tool would have if it was applied at a regional-scale, enabling groups to target riparian revegetation and other restoration activities conducted as part of the Natural Heritage Trust, the National Action Plan for Salinity and Water



A range of erosion processes can cause water quality impacts at landscape scales, with the degree of connectivity between the sources and the river system also important. This photograph of Mirrool Creek in NSW shows cultivated paddocks in close proximity to a river. Bank erosion along the meandering river channel can deliver large amounts of sediment directly into the river during flood events. Buffer strips can provide opportunity to trap sediment and nutrients from paddocks if well managed. Photos throughout this chapter courtesy CSIRO Land and Water.

Quality, and other regional planning activities. This would allow funding decisions for river management to increasingly be made at a regional or catchment level. The Catchment Assessment Techniques project (see 'At work in the catchment', page 89) was designed to develop the SedNet model to make best use of the higher-resolution data available at the river basin scale, and to improve and test the process representations in the model to provide better spatial resolution in model predictions. This NRCP project developed and tested the research in three focus catchments.

How SedNet works

SedNet is available as free software through the Catchment Modelling Toolkit, at www.toolkit.net.au/sednet. Due to the quantitative nature of the modelling, a certain level of GIS, catchment hydrology and erosion assessment expertise is essential to understand the complexity of modelling sediment and nutrient budgets. An online modelling community maintain a constantly evolving repository of software intended to improve the efficiency and standard of catchment modelling. This has seen the software adopted for use by a number of agencies across the country.

SedNet constructs sediment and nutrient (phosphorus and nitrogen) budgets for regional scale river networks (3000 to 1,000,000 km²) to identify patterns in the way materials move through the landscape. A budget is an account of the major sources, stores and fluxes of material. Separate budgets are constructed for each river link, or reach, allowing the spatial patterns of erosion to be assessed, and also permitting the connectivity between upstream sources and downstream reaches to be examined. This analysis then enables the areas contributing most to sediment and associated nutrient problems to be identified. It is important to recognise that the parts of a catchment where sediment is generated by hillslope or gully erosion may not be the main contributors to river sediment loads. This is because the sediment may be stored on the floodplain or in a reservoir before it can reach the river. SedNet identifies the source areas that are closely connected to the river system, that is, those areas where sediment is both generated and likely to move into the river.

In the model, material budgets are constructed using Configurations and Scenarios. A *Configuration* describes the stream network and several catchment



The impact of intensive rural landuses on water quality depends on the potential erosion rates and also on the connectivity to the river system. The impact can be minimised if land management practices are well managed.



Software is now available to enable management agencies and consultants to calculate sediment and nutrient budgets across large catchments using local datasets. The results of modelling can be tested using water quality monitoring data, sediment tracing techniques and geomorphology studies.



Some of the outcomes of the project have been made available in the SedNet software model available online with user documentation.

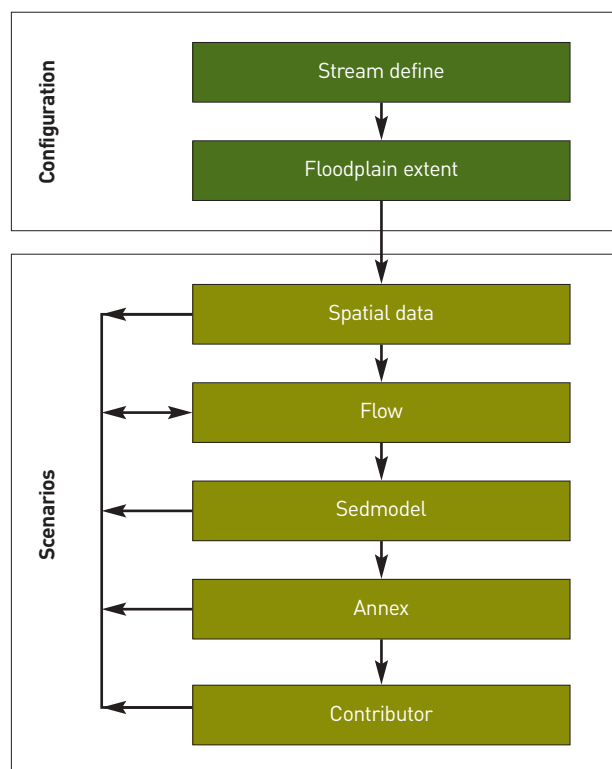


Figure 1. Configuration and scenarios with components.

attributes that do not change, this is the framework on which the budgets are constructed. Each *Configuration* can contain several *Scenarios*, each containing the datasets, parameters and results associated with a particular catchment condition, whether historical, present-day or a simulation of possible future condition (Figure 1).

SedNet requires information on the spatial features of the region to be modelled including the terrain, reservoirs and floodplains. Other data includes stream



Decades of gully erosion has resulted in large accumulations of sand and gravel sediment in this section of the Murrumbidgee River, NSW. The growth of many gullies in the area has now naturally slowed down, but the sediment has filled in river pools and may take decades or longer to be transported downstream.

Photo Louise Gallagher.

flow records and maps of hillslope and gully erosion, Using this data, SedNet predicts in tonnes per year the sediment supplied from surface or hillslope erosion, gully erosion, bank erosion and nutrients from overland flow. The supplied material is routed through the river network, giving predictions of the rates of transport of bedload sediment, suspended sediment transport, particulate and dissolved nitrogen and phosphorus, floodplain and reservoir deposition and in-channel bedload deposition (Figure 2).

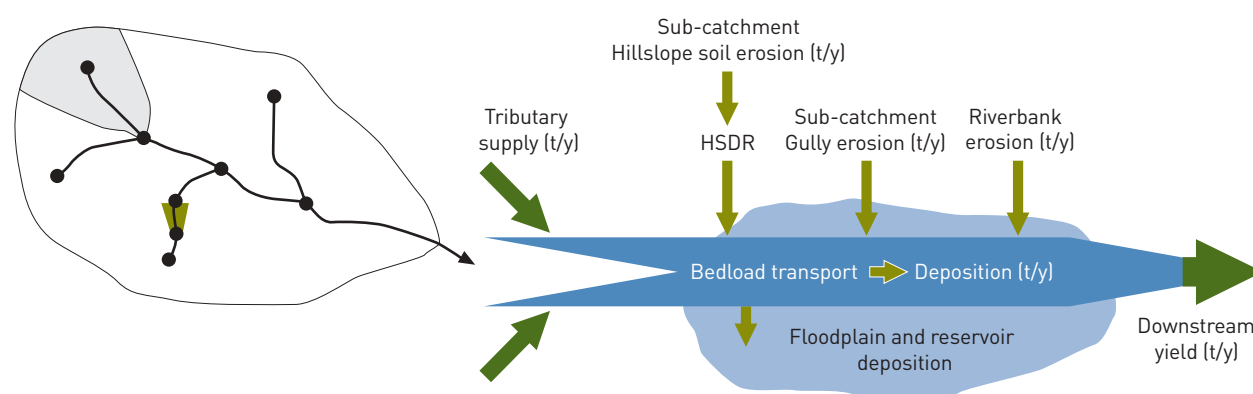


Figure 2. Geomorphologists have long used sediment budgets to determine sediment sources within catchments. SedNet gives sediment budgets a spatial dimension by constructing separate sediment budgets for each river link in a network. The approach accounts for the major sources and sinks of sediment, including hillslope, gully and riverbank erosion which is important for targeting specific management actions. What is exported from river outlets is only a small part of what is supplied to the river network through erosion; accounting for floodplain and reservoir deposition helps to understand what parts of the catchment have greatest impact on downstream reaches.

SedNet differs from other models of regional water quality by identifying the contribution of individual source and sink processes (e.g. hillslope, gully, riverbank), to assist with planning of specific source control measures (Figure 3). Sediment yield over time is ultimately limited by the rates of erosion upstream rather than the stream flow during a particular event. Erosion rates are determined by terrain, the soil type and history of vegetation cover. SedNet represents the integrated effects of hydrologic variability including floods and droughts over time to simulate the long-term effects of past or possible future management over several decades. This is ideal for catchment planning, since management actions, such as rehabilitating riparian vegetation or controlling gully erosion, can take decades to reach their full effect (Figure 4).

At work in the catchment

SedNet techniques were applied to three catchments in south-eastern Australia to determine restoration priorities for improving water quality and riparian condition. The upper Murrumbidgee (NSW), Goulburn-Broken (Vic) and Mt Lofty Ranges (SA) catchments

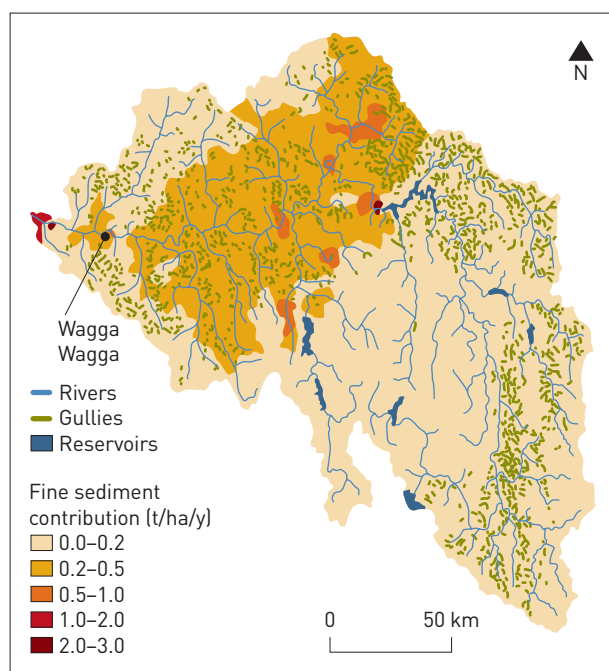


Figure 3. In the Murrumbidgee River catchment, there is a catchment target to reduce the suspended sediment load at Wagga Wagga near the downstream end by 30%. Research has shown that gully and stream bank erosion contribute most of the sediment load. This is reflected in the regional catchment plan. The map shows the huge length of gullies in the region (green lines). However, 80% of the suspended load at Wagga Wagga is derived from just 20% of those sites (the darker orange colours). For the rest of the catchment, there is either little erosion or the sediment is deposited and does not reach Wagga Wagga.



Photo above: River bank erosion along the Murrumbidgee River, near Gogeldrie Weir, NSW. River gum roots are exposed by fast flowing water, released from weir at times of flood, eventually toppling them into the river, 1991.

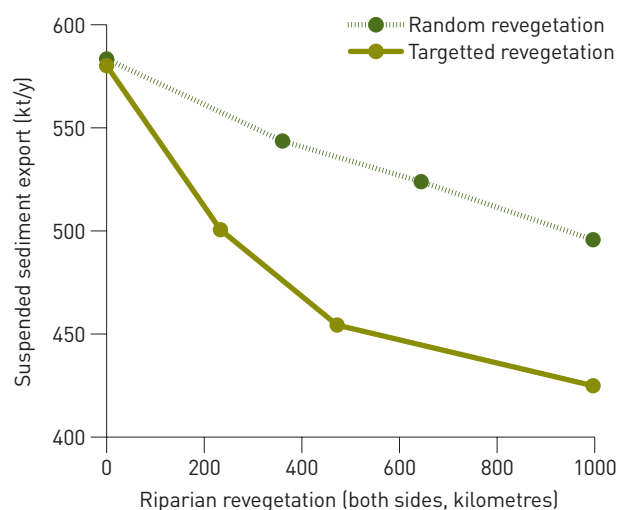


Figure 4. Comparison of investment strategies in the Murrumbidgee River catchment. The CMA plans to prevent streambank erosion by revegetating riverbanks. The graph, which uses sediment modelling to relate revegetation to sediment loads at Wagga Wagga, shows that if correctly located, such revegetation has the potential to significantly reduce sediment load. If revegetation is randomly allocated, or allocated only considering other criteria such as willingness of farmers to co-invest, the revegetation will have a much smaller impact on river sediment loads (Wilkinson et al. 2005).



Soil around fenceline being eroded by stream, near Young, NSW.

were chosen for their proactive management agencies interested in prioritising stream rehabilitation at a regional scale, and for the ready availability of data to develop the assessment techniques. This work has seen a number of improvements to the SedNet software and application techniques, including improvements to several components of the model to enable better use of catchment-scale data and representation of erosion and deposition processes in sediment budgets (hydrology, bank erosion, gully erosion, hillslope erosion).

In the Murrumbidgee catchment, an improved prediction of the location of bed material accumulation and its effect on river habitat in river basins, has led to ongoing work to understand the biological impact of bed material accumulation. This work helped the Murrumbidgee Catchment Management Authority (MCMA) to define priorities for a \$1 million river restoration program (MCMA 2004) that is intended to contribute towards addressing water quality management targets as specified in the Murrumbidgee Catchment Blueprint (MCMB 2003).

Improved capacity for the prediction of the sources and transport of suspended sediment loads through river catchments allowed the Goulburn-Broken Catchment Management Authority to define a number of reaches where protection or improvement of water quality had

been identified as a high or very high priority. Priority sites for erosion control were identified based on SedNet outputs, and ranked in descending order of their contribution to reduction in suspended sediment load (Wilkinson et al. 2005).

Results and advice from these projects have been implemented across the three focus catchments, with members of the Goulburn-Broken Catchment Management Authority reporting they have:

“enabled us to target areas for riparian protection, bank erosion, catchment erosion activities, to address the source of erosion and sediment problems, rather than using a random approach throughout the catchment. This targeted approach has been accepted by the CMA’s partners in land management. The results are also being used to target grant proposals.” (Tennant, pers. comm. 2005).

Vast improvements in the understanding of the uncertainties in sediment budget modelling have allowed for a smaller margin of model error, although models and their outputs must continue to be compared with on-ground, measured data to maximise their effectiveness. These model improvements to the SedNet software have now been used to assess the sources and transport of sediment in many catchments including:

- the Great Barrier Reef catchments to help identify strategies to reduce sediment and nutrient export to the Great Barrier Reef,
- Sydney's main water supply catchment to assist management of sediment sources,
- Gippsland Lakes catchments to reduce nutrient supply associated with blue-green algae blooms, and
- the Brisbane River and adjacent catchments to reduce sediment delivery to Moreton Bay.

Putting SedNet to work

Protocols for use of SedNet should be followed to ensure they are used in the context of a broader catchment planning process that considers a range of other condition assessments (e.g. Rutherford et al. 2000). Integrating the modelling process with other prioritisation techniques for salinity, biodiversity and the many other factors impacting on water quality and river health will lead to the most informed decisions being made. Furthermore, while SedNet assesses total load and identifies the processes responsible, there remains a need to assess ambient water quality for river health assessment, which can also be affected by local issues such as carp, livestock access and road crossings. SedNet could find future applications in other industries, such as targeted planning for the agroforestry and plantation industry.

CSIRO Land and Water continues to undertake research to improve the degree of reliability around the sediment budgets, by testing them against independent data and investigating sediment transport processes. We need to better understand the spatial patterns in each

erosion process in different environments, for example the tropical rangelands and temperate gully networks. SedNet can help to answer some of the questions facing catchment managers, such as what is the likely trajectory for sediments yields in those environments now that the initial phase of gully expansion has stabilised in some areas but appears to be continuing in others, what sediment movement can we expect in future, and what might we expect to happen under climate change?

Conclusion

SedNet is a tool that can help a wide range of land and water managers to make the most informed decision when targeting land rehabilitation to reduce erosion into the river network, allowing proposed management strategies to be simulated in the model, so that the predicted outcomes of alternative management actions can be considered and costed.

It is being used to develop water quality improvement strategies in a number of focus catchments, and managers report the approach has enabled them to target areas for riparian protection, bank erosion, catchment erosion activities. The tool enables them to address the source of erosion and sediment problems, rather than using a random approach throughout the catchment. This targeted approach has been accepted by CMAs and their partners in land management, as well as being used to target grant proposals. Continuing refinements to the application make SedNet an important collaborative agent for environmental improvement.

Suspended sediment visible in the Murrumbidgee River near the weir at Maude, NSW, 1994. Upstream erosion can transport large amounts of sediment and nutrients to downstream water bodies during floods.





In many areas riverbank and gully erosion are important sediment sources. Fencing out and revegetating riparian areas can help to stabilise these areas and reduce erosion rates. These plantings can also have provide benefits to biodiversity, particularly when they are close to existing native vegetation.

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Chapter 8 — Budgeting and monitoring for sediment and nutrients at the catchment scale

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Summary

- While the ecological effects of sediment and nutrient inputs in Australian rivers are relatively well understood, there is limited quantitative information about the effects of human induced, land-based processes on delivery of nutrients and sediments to rivers. Environmental researchers and catchment managers alike, need to understand the pathways of nutrient movement (source, transport and transformation) to be able to successfully manipulate the nutrient cycle and set priorities aimed at reducing sediment and nutrient inputs.
- Nutrient and sediment budgets at the sub-catchment scale, which account for inputs, outputs and changes in materials stored in the river, have the potential to generate the knowledge required to underpin management decisions. However, the construction of such budgets is constrained by the need for extensive data sets derived from long-term monitoring. This project developed budgets derived from both observations and modelling. Use of information from modelling has the potential to reduce monitoring costs and widen the application of nutrient and sediment budgeting.
- Regional nutrient (N and P) and sediment budgets for four catchments with different land uses, soils, and hydrological regimes showed wide variations in the dominant sediment source (hillslope, bank and gully erosion) between catchments.
- The computer models SedNet/Annual Network Nutrient Export (ANNEX) and Hydrological Simulation Program–Fortran (HSP–F) were applied in the Johnstone River (Queensland) catchment to predict annual dissolved nutrient and sediment loads. There was good agreement between the models and direct observations, increasing confidence that either of the models could provide realistic end of catchment loads.
- Several key knowledge gaps were identified when establishing parameters for the sediment generation processes: the hillslope delivery ratio, gully sediment generation, and stream bank erosion rates. More realistic estimates of soil nutrient concentrations would lead to improved estimates of nutrient end of system deliveries.
- The effectiveness of current monitoring strategies for the four river systems varied widely. Broadly, as the episodic character of the flow increased, the effectiveness of strategies based on routine monitoring at fixed intervals diminished. In the worst cases, four well-timed observations would generate as good an estimate of end of system discharges as 22 regular observations.

Background

Sediment and nutrient inputs into aquatic systems have considerably increased since European settlement of Australia, however, the ecological effect of these changes is still relatively unknown. Carbon (C), nitrogen (N) and phosphorus (P) exert a bottom-up control on aquatic ecosystems. In many situations the processing of these nutrients has changed dramatically as a result of catchment land use and modifications to our waterways. This, in turn, can alter the composition and biomass of primary production with follow on effects up the food chain. The cycling of C, N and P are intimately linked because N and P cycles include a significant organic component, and because the response of an ecosystem is dependent on the ratios between these elements and their forms, not just the concentration of an individual element.

There is general agreement that N is a limiting nutrient for estuarine ecosystems. Recent research by Wood and Oliver (1995), Baker et al. (2000) and Bormans et al. (2004) has shown that N (as well as P) can be a limiting nutrient to primary production in freshwater systems, and play a role in changing the dominant phytoplankton species, or in affecting the physiological state of nuisance species of cyanobacteria (Brookes & Ganf 2001). The dynamics of P in Australia's river systems has been the subject of intensive investigation, and while significant knowledge gaps remain in that area, there is even less understanding about N and C dynamics and their interactions with microbial, physical processes, and ecological effects. (Chapter 4 by Fellows et al. discusses C and N interactions in the riparian zone.)

It is only through understanding of the pathways of nutrients (source, transport and transformation), that we can determine how to successfully manipulate the nutrient cycle to minimise nutrient driven problems. There is also a growing appreciation that with intensification of agriculture and the shift to higher value crops, terrestrial nutrient applications are increasing with serious implications for nutrient loads into streams.

Land resource managers face a difficult task in devising appropriate strategies to reduce and ameliorate the impacts of land use on freshwater aquatic systems, and to reduce the terrestrial nutrient loads delivered to coastal and estuarine waters (see Chapter 6 for one example of an industry response to this growing problem). The difficulties arise from the scale of the problem, which reflects more than 200 years of landscape transformation, and the limited resources available for on-ground works. Managers have to make hard choices in prioritising sites for remedial measures against a background of limited information. The in-stream impacts are often remote from the sources of sediments and nutrients; there are often widely dispersed multiple sources; and the available in-stream monitoring data is often not at the right spot or of the right frequency to reliably identify the sources at a sub-catchment scale. While reliable experimental methods capable of addressing these issues are available, they are too costly to be widely applied. As well as being unable to quickly quantify rates of delivery of materials to streams and rivers, we also lack reliable and simple ways to determine how rapidly sediments and their attached nutrients are either moving through the system or being stored in bed and bank deposits.

Brisbane River. Photos throughout this chapter are courtesy of the project team.





One of the research catchments, the Johnstone River at Crawford's lookout. Photo R. Walton.

From the perspective of setting priorities aimed at reducing sediment and nutrient inputs, the way ahead lies in a better understanding of the processes responsible for mobilising and transporting materials. It is important to know how these various processes vary at a sub-catchment scale (a realistic scale for amelioration measures), how they vary spatially, and how land use and other factors affect the relative significance of the various processes. This knowledge needs to be part of an integrated framework which brings together the various contributions and characteristics from different parts of the catchment to determine the overall delivery of sediments and nutrients at the end of the catchment. Mathematical models have been developed for dealing with the complex integrative task but they often require many parameters to be known or measured, and this is very resource demanding. There are also concerns about their transferability from one catchment to another when they rely heavily on catchment-specific parameters.

This research set out to tackle some of these problems with the ultimate goal of improving the prioritising tools available to land resource managers. Using a combination of existing sediment and nutrient measurements together with spatial modelling of erosion and transport processes, we constructed regional sediment and nutrient budgets for four catchments (Latrobe, Murrumbidgee, West Brisbane and Johnstone) with different characteristics of climate, soil type, land use

and level of flow regulation. They were chosen to assess how the processes of sediment and nutrient generation and loss change with environmental conditions.

Research approach

Research commenced by reviewing the existing data on nutrients and sediment movement at catchment scale. Material budgets were constructed for the four catchments by identifying all the sources, sinks and changes of contaminants (sediments and nutrients) from their source to the catchment outlet. Several different methods were used to construct the budgets. They ranged from wholly empirical approaches using just the available concentration and discharge data, to modelling relying on SedNet/Annual Network Nutrient Export (ANNEX). These approaches, in combination, allowed the dominant processes to be quantified, revealing the major variations between the catchments. Understanding the relative contributions of the different processes is necessary before more-detailed process studies can be performed. The initial research focused on nutrients containing organic carbon and nitrogen, which, while critical to the ecological character of streams and rivers, have been much less thoroughly investigated than nutrients containing phosphorus. We were constrained, however, by the lack of extended time series of data on organic carbon concentrations, and the sparse spatial

coverage of this data. We used the available carbon data in the Murrumbidgee and Latrobe catchments, but the emphasis on nitrogen and phosphorus budgets here reflects the data imbalance. This data gap points to a deficiency in past monitoring strategies which should be redressed in updating the monitoring networks — a theme we return to later.

The empirically based budgets were constructed by two methods using the Brisbane and Johnstone Rivers. Regression relationships were constructed relating contaminant loads to daily discharge. This relationship was then used to construct inferred daily nutrient loads, and these were then integrated over a year to produce the annual loads. This method was also used in the Latrobe catchment and was supplemented by methods which interpolated from the more frequent concentration measurements to generate daily concentrations. This value was then multiplied by the corresponding daily discharge, and the resulting daily loads summed to produce the annual load. In the Murrumbidgee, an approach was used measuring the changes in concentration as a “parcel” of water moves through the system. This approach was necessary to take account of the large irrigation abstractions.

Denitrification data (Table 1) is derived from spot measurements in the Johnstone using the acetylene-block technique and scaled up to the river length based on stream width and mean depth. Vegetation cover assessment was derived from maps of land use and the same data was used in the SedNet/ANNEX modelling. Similar details were used with Hydrological Simulation

Program–Fortran (HSP–F). P-adsorption is a physico-chemical process and dependent on fine particles, and was inferred to occur in all systems. Macrophyte (“plant”) uptake will occur when the water depth is relatively shallow and was inferred from the mean depth of the various reaches. Only observations from the Johnstone River provided reliable evidence of groundwater inputs.

Budgets for N, P and sediment were constructed for each of the four catchments using the SedNet/ANNEX programs also (HSP-F generated data was used for the Johnstone). The programs were modified to meet the demands of regional applications. The budgets integrate the diverse data and knowledge related to material transport in each catchment. They use local data, including stream monitoring, erosion measurements, fertiliser inputs and GIS data on controlling factors such as rainfall, soil erodibility, terrain and land use.

The SedNet model generates a mean annual sediment budget (an algebraic sum of the inputs and outputs in each river segment) moving downstream through a river network. Using estimates of the amount of sediment originating from hillslope, gully and streambank erosion, the model calculates the amount of sediment that is delivered downstream. The ANNEX model predicts the average annual loads of phosphorus and nitrogen in each link in a river network in a similar way to SedNet. The main sediment and nutrient particulate sources are hillslope, gully and riverbank erosion, while the dissolved nutrient loads are delivered by the overland runoff and groundwater and point sources.

Hydrological Simulation Program–Fortran (HSP–F) simulates for extended periods of time the hydrologic, and associated water quality, processes on pervious and impervious land surfaces and in streams and well-mixed impoundments.



Bank erosion in a turbid tributary.

Table 1. Characteristics, sediment and nutrient loads and dominant processes in the four studied catchments (TSS = total suspended sediment, DP:TP = dissolved to total P ratio, FRP = filterable reactive phosphate).

	Murrumbidgee	Johnstone	Latrobe	Brisbane
Location	NSW	Far North QLD	VIC	South-east QLD
Size (sq km)	39,600	1634	4681	12,600
Rainfall (mm)	300–1600	1600–3550	600–1900	700–2000
Land uses	Grazing, cropping, forest, irrigation	Rainforest, pasture, dairy, banana, sugar cane, unsewered residences	Improved pasture, horticulture	Forest, grazing, cropping, urban
Vegetation cover	Low, except in upper catchment	> 80%	Low, except in upper catchment	> 60% headwaters > 40% Lockyer, Bremer, upstream of Wivenhoe Dam
Annual discharge (ML)	1.1×10^6	2.8×10^6	0.93×10^6	1.2×10^6
% of contaminants exported in floods	89% TSS, 91% TN, 92% TP	97% TSS, 75% TN, 87% TP	67% TSS, 67% TN, 67% TP	> 90% TSSs
Main erosion types	Bank, gully, hillslope	Hillslope	Bank	Bank, gully, hillslope
TSS loads in (kt/y) TSS loads out (% retained) (note 1)	1480 60 (96)	237 231 (3)	62 37 (40)	690 280 (60)
TP loads in (t/y) TP loads out (% retained)	1308 157 (88)	326 319 (2)	97 87 (10)	783 351 (55)
TN loads in (t/y) TN loads out (% retained)	11,270 5522 (51)	2243 2228 (< 1)	673 615 (9)	3040 1561 (49)
DP:TP loads in DP:TP loads out	24%	21% 19%–5DIP, 14DOP	10–20% (note 2)	25% 50%
DN:TN loads in DN:TN loads out	66%	80% 50–70%	40% (note 2)	25% 50%
Floodplain deposition	35%	No	44%	11%
Reservoir deposition	28%	No		38%
Denitrification	Yes	No	Low	Yes
Plant uptake	High	No	High	?
P-adsorption	Yes	Yes	Yes	Yes
Groundwater interaction	No	Yes	No	Probably
Extent of flow regulation	High, unseasonal	None	Moderate, seasonal	High, unseasonal

1. % retained is the percentage of the incoming material retained within the budgeted section of the river.

2. Ratio of NO_x / TN and FRP/TP.

Key findings

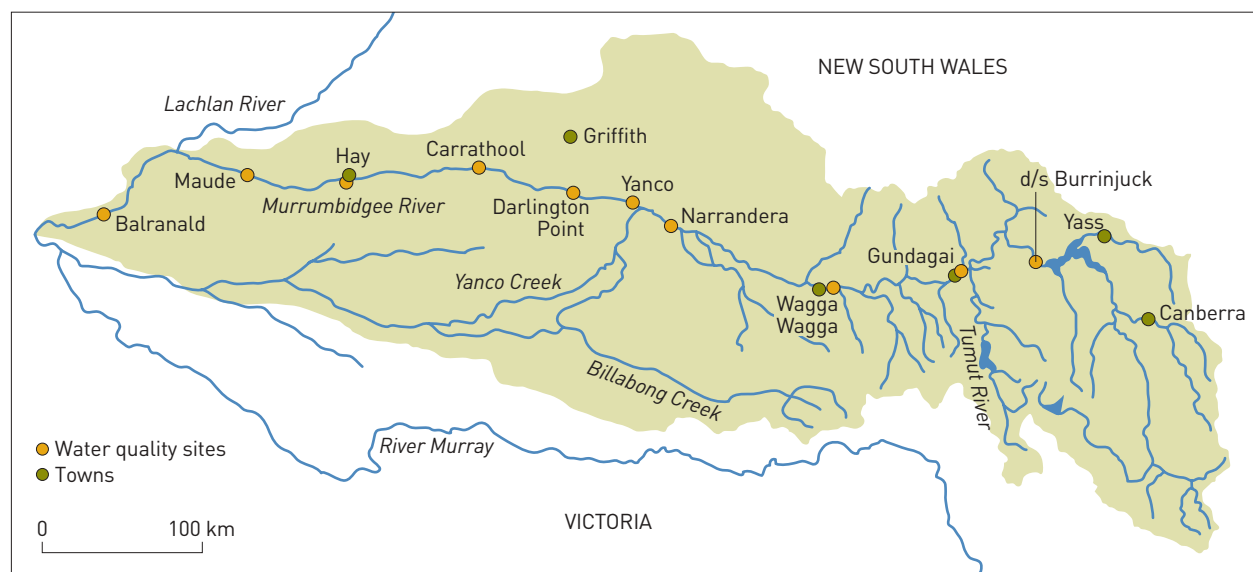
Summary of modelled and measured loads

Table 1 (on the previous page) shows the characteristics, the sediment and nutrient loads, and dominant processes in the four studied catchments.

Carbon movement

In the Murrumbidgee River, measurements were taken to infer a carbon budget between Gundagai and Darlington

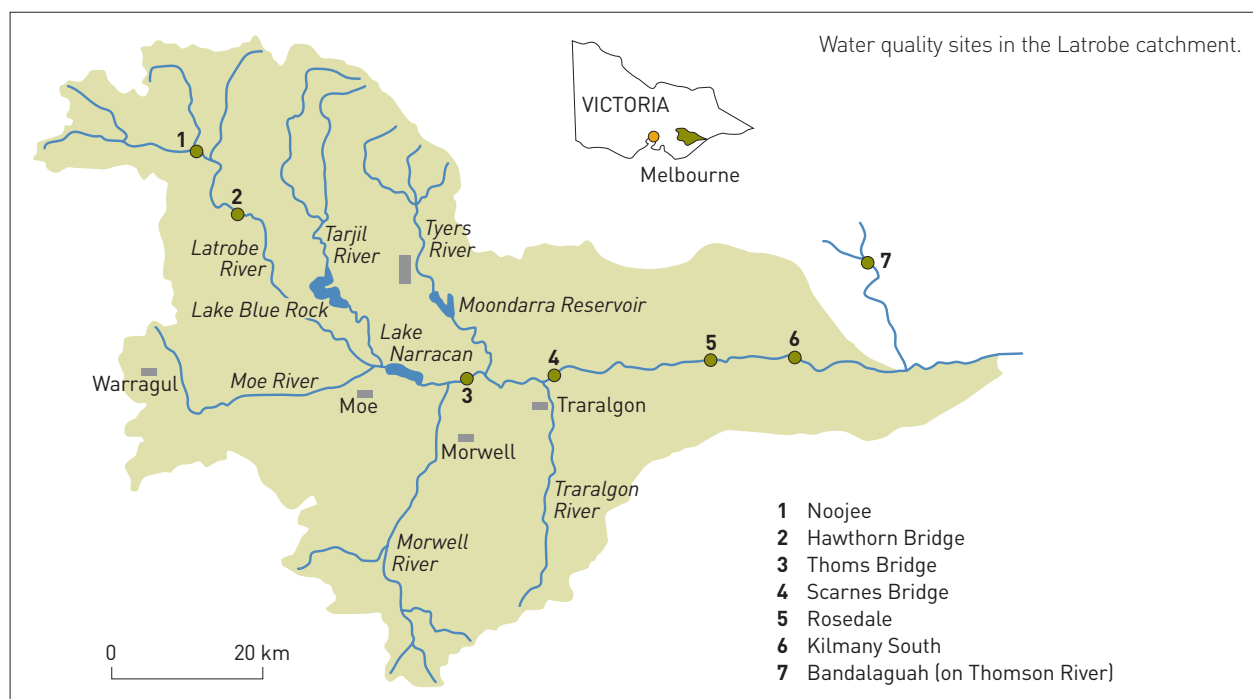
Point. The samples were analysed for dissolved organic carbon (DOC), total organic carbon (TOC), as well as particulate C. Previous research (Olley 2002) has shown that soil sources of C dominate during flood events. During non-flow periods, riparian vegetation and soil are important sources in the upper reaches of the main channel. During transport downstream, much of the carbon is metabolised and assimilated by primary producers who contribute more than 75% to the carbon source in the lower reaches.



Top: The Murrumbidgee catchment showing the location of the water quality sites.

Above: The stream network and gauging stations in the Johnstone catchment.

Left: The West Brisbane catchment.



In the Latrobe catchment, total organic carbon measurements have been taken at two sites over a period of about 10 years, but only four times a year. Higher TOC concentrations correspond to conditions of high flow (and increase in TSS). There is a general increase in concentration as we move downstream, which is associated with two sources, increased bank erosion and increased primary production. No organic carbon data was available for the other two catchments. In the two catchments examined, bank erosion is a significant contributor to particulate organic carbon and, therefore, measures to reduce bank erosion will similarly reduce inputs of organic carbon. This material is, however, likely to be relatively unreactive and its contribution to supporting higher trophic levels will be slight. In situ primary production will be of greater ecological

significance. This is higher in the Latrobe than the Murrumbidgee due to greater nutrient availability and a better light climate in the Latrobe.

Sediment movement

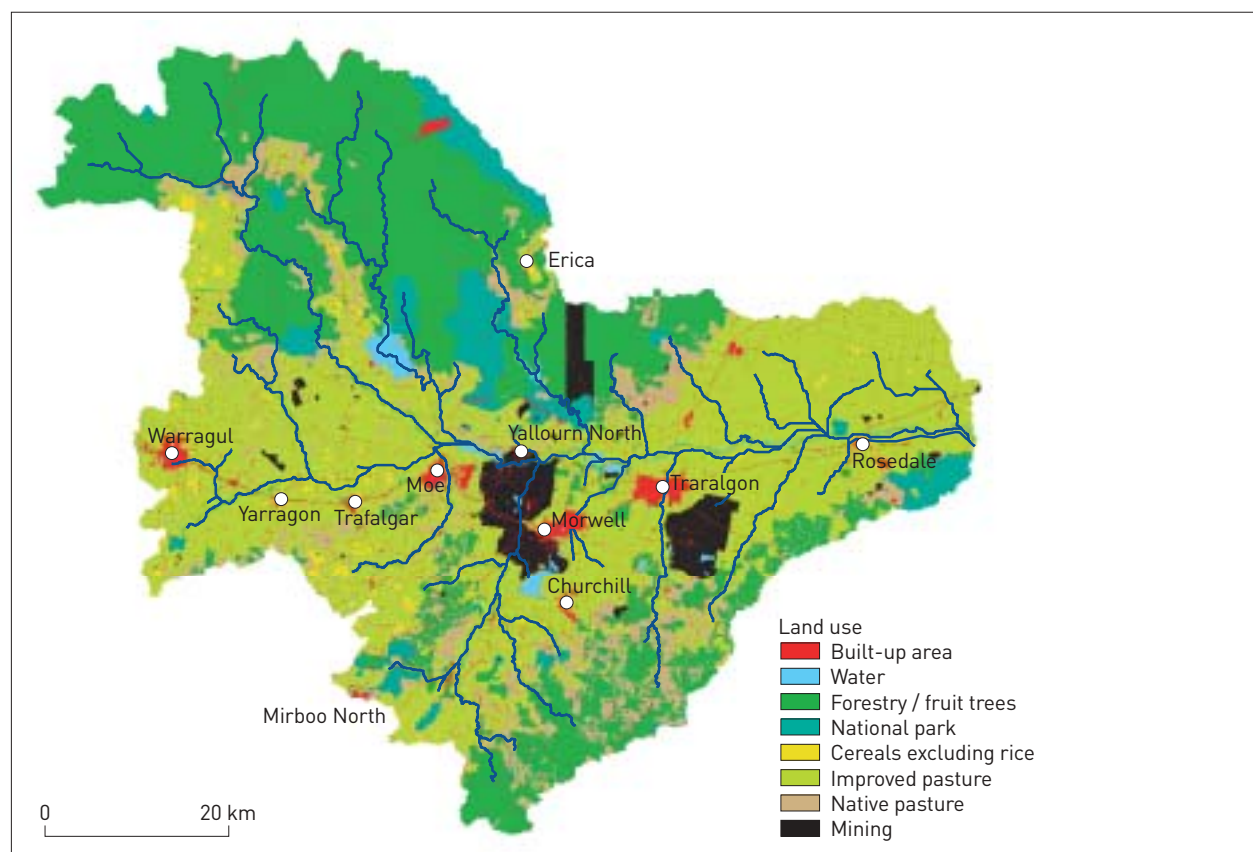
The main processes controlling sediment transport are:

- i) bank, gully and hillslope erosion, which generates sediments, and
- ii) deposition and resuspension of sediments in reservoirs, floodplains and channels.

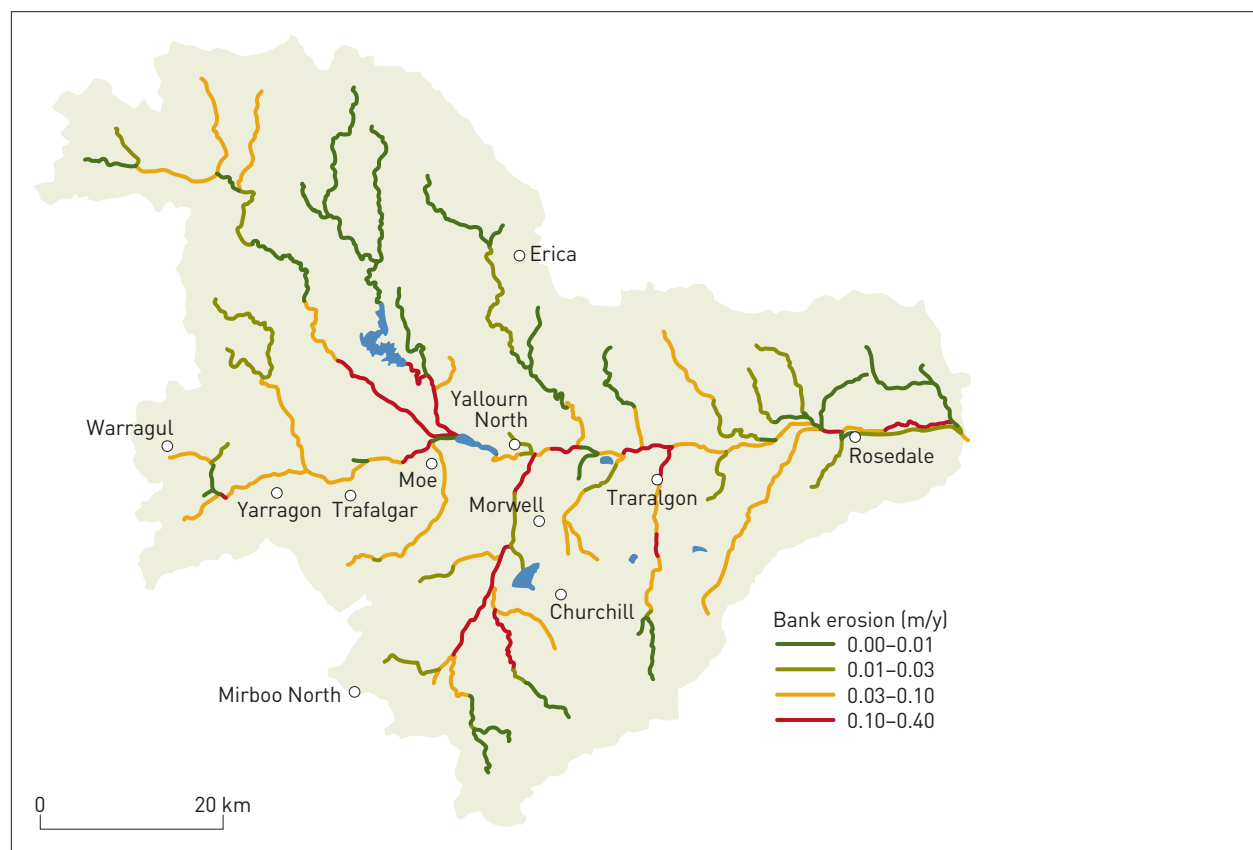
There was a large variation in the relative importance of these processes across the four catchments. This was expected due to the different environmental conditions and topography in each catchment. Table 2 shows the relationship between the different processes and the environmental variables controlling them.

Table 2. The relationship between the different sediment generation and movement processes and the environmental variables controlling them.

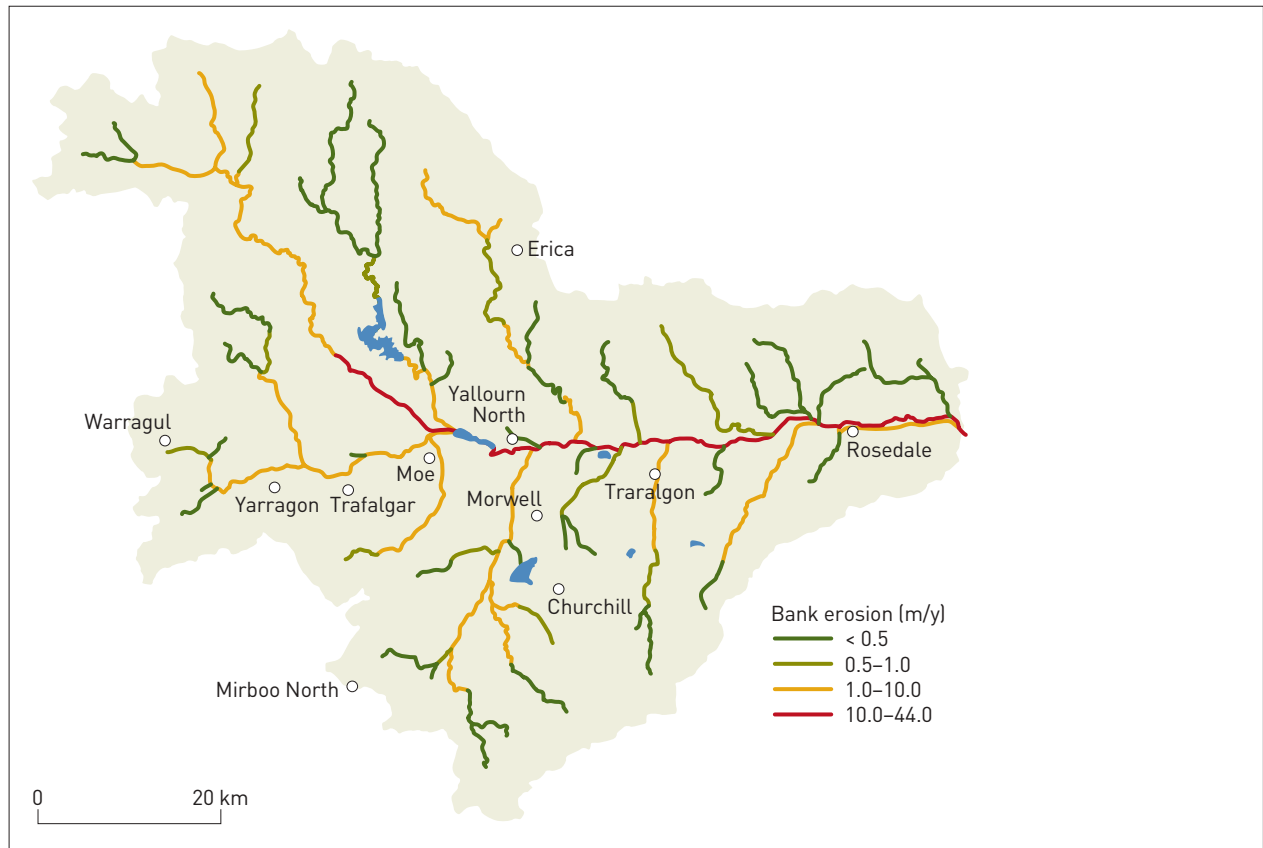
Process	Environmental variable
Hillslope erosion	Vegetation cover, rainfall, terrain, soil erodibility
Bank erosion	Discharge volume and frequency, riparian vegetation
Gully erosion	Gully density (a function of land use, hillslope inclination and length, geology, soil properties and rainfall), sub-catchment area, average gully cross section, soil density, time for gully to develop
Deposition – in floodplain – in reservoir – in channel	Overbank component of discharge, concentration of TSS in channel, floodplain area Mean annual inflow to reservoir, reservoir volume River regulation, residence time, depth, particle size



Land use (grazing and industry) in the Latrobe catchment.



Distribution of bank erosion predicted by the SedNet model in the Latrobe catchment.



Distribution of suspended sediment loads predicted by the SedNet model for the Latrobe catchment.

Nutrient movement

The main processes controlling the movement of particulate nitrogen and phosphorus are erosion (hillslope, bank and gully); deposition; and resuspension. Dissolved nutrients are transported by surface runoff from different land uses and point sources, and sometimes by sub-surface flows. The SedNet/ANNEX model calculates nutrient loads on an annual basis for all the critical nutrient sources and transport processes, although the 'inputs of dissolved nutrients' component needs to be refined for improved predictions. On a shorter time scale, adsorption/desorption of the inorganic form of dissolved phosphorus to sediments can account for about 10% of extra phosphorus source, on average, across a number of lowland rivers.

SedNet assumed that the bedload fraction, consisting of coarse material such as sand, does not contribute significantly to nutrient sources or sinks, and only

the suspended load of finer particles will transport nutrients. This simplification does not take account of hyporheic flows (see box), which can control or dominate the variability of inorganic N and P forms through microbiological processes in the bed sediments. Although absolute values are uncertain, SedNet can predict the relative importance of the different processes across the four catchments. The difference between modelled and measured loads varied between catchments.

The HSP-F model results used for comparison in the Johnstone catchment are described in Hunter and Walton (1997). In this catchment, the agreement between measurements and the model was very good along the entire river network for both sediment and nutrient loads, including dissolved forms. This is because intensive measurements had been taken at many sites over a number of years, enabling good inputs for initial model calibration.

Hyporheic flow is the percolating flow of water through the sand, gravel, sediments and other permeable soils under and beside the open streambed. Hyporheic flow has a significant effect on nitrogen cycling (including denitrification, see Chapter 4 Fellows et al.) in freshwater streams and thus on downstream deliveries of biologically available nitrogen.



The Murrumbidgee River at Gundagai (above) and at Darlington Point.



Sandbars and snags (fallen trees) on a bend of the Murrumbidgee River near Darlington Point, NSW. Reduced water flow in recent times has caused the river to silt up in many sections and made river navigation hazardous to all but small craft, 2002.

In the Murrumbidgee catchment, sediment loads were reasonably well predicted at the outlet, but there was no comparison with measured loads along the catchment. The model did not predict nutrient loads well due to a lack of measurements on dissolved nutrients above Wagga Wagga.

In the Brisbane catchment there was no monitoring suitable for calculating nutrient loads, either at the end of the catchment or along the river network. Measured sediment loads were well correlated with modelled loads despite an overall lack of sufficient measurements.

In the Latrobe catchment a large number of measurements have been taken at various sites along the river over the past 30 years. The fit between modelled and measured data could be improved by better inputs of data on dissolved nutrients per land use as well as by incorporating the distribution of active gullies.

There is a strong relationship between total suspended solids (TSS) and particulate phosphorus (PP) concentration in both the Johnstone and Murrumbidgee Rivers. Soil maps have confirmed the higher than average P concentrations in those two catchments.

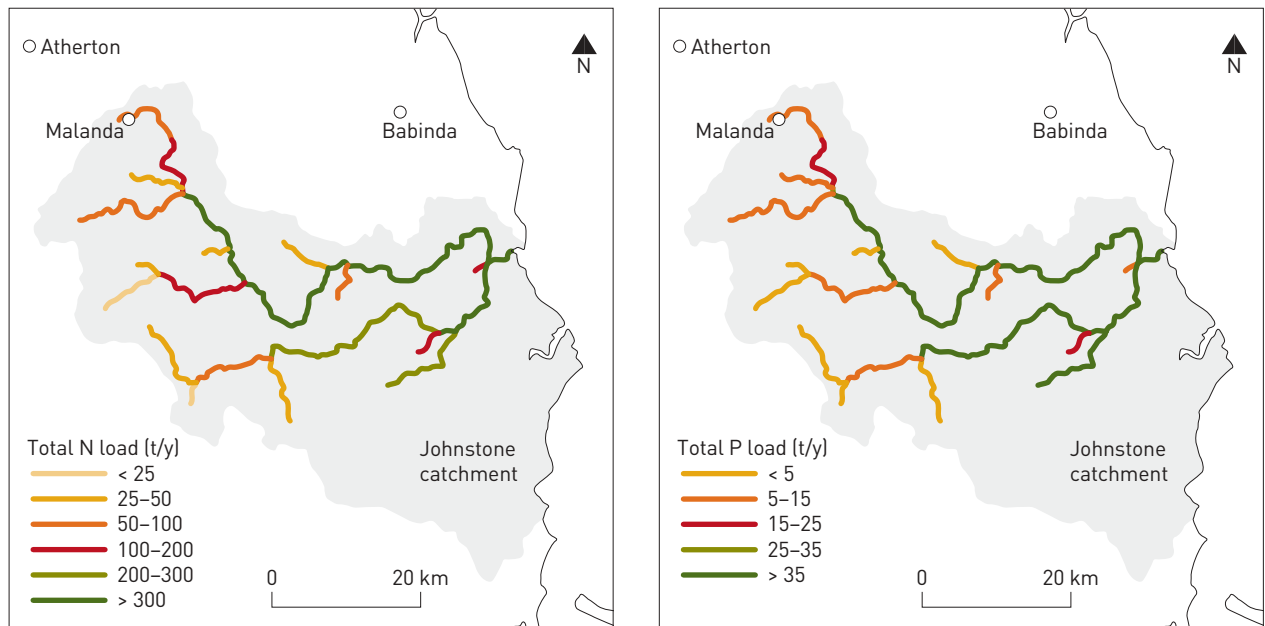


Figure 1. Total N and Total P load distributions predicted by SedNet for the Johnstone catchment.

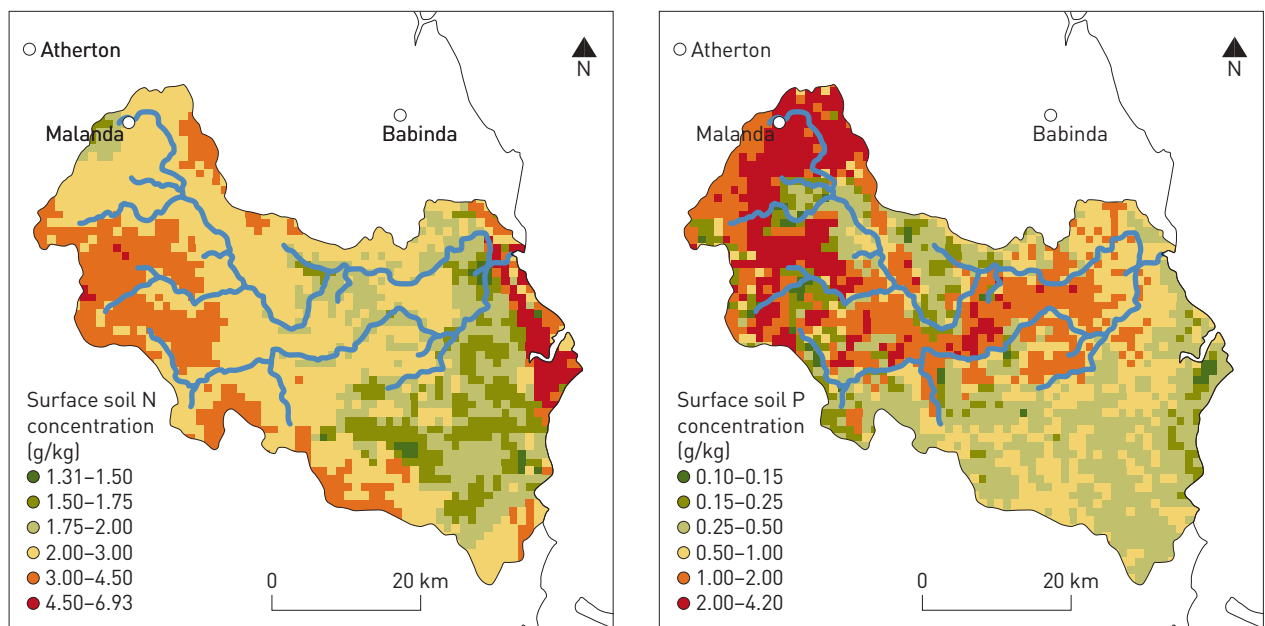


Figure 2. Surface soil concentration of nitrogen and phosphorus in the Johnstone catchment.

In all four catchments the distribution of total phosphorus sources and sinks was very similar to that of suspended sediment because 80% of P in Australian soils is associated with the particulate form or sediment bound. The only exception is the significant dissolved load at the bottom of the Brisbane catchment resulting from contributions of sewage treatment plants in the large urban areas.

Phytoplankton uptake of phosphorus was found to be important in the Murrumbidgee River under dam release flows and in the mid region of the Latrobe River under low flow. Significant algal problems have been

identified in the Brisbane catchment. Denitrification was higher in the Murrumbidgee after runoff flows from the catchment, rather than after dam release flows due to a higher organic input. In the Brisbane catchment, significant denitrification rates were found only at low flow.

In the Johnstone River, groundwater interaction was a major driver of the nitrate dynamics under low flow. This effect was less pronounced in the other larger catchments which were more arid with reduced infiltration.

Comparing the four systems it is clear that dissolved nutrient inputs (per unit area) are greater from the three wetter catchments (Latrobe, Johnstone and Brisbane). The mechanisms for dissolved nutrient delivery differ however. Groundwater is an important route for dissolved nutrient delivery into the Johnstone, while overland flow is of greater significance in the Brisbane and Latrobe. In the Murrumbidgee, dissolved nutrient inputs are dominated by episodic inputs when moderate to large rainfall generates significant overland flows. Upstream storages in the Murrumbidgee remove much of the dissolved nutrients when residence times are large. The heterogeneity of landuses across the different catchments precludes definitive conclusions regarding the role of land use and vegetation type in determining dissolved nutrient deliveries. The data is consistent, however, with the observations from north America and Europe, that wetter climates and intensification of agriculture lead to stronger land–river connection and increased nutrient deliveries. The dissolved nutrient concentrations in the Australian systems fall well short of the “pathological” northern hemisphere systems.

Monitoring implications

Much of the monitoring carried out by state agencies is based on regular collection of nutrient samples at fixed time intervals. This is not effective when the aim is to monitor catchment loads. This type of sampling often misses unregulated flows when most nutrient and sediment transport occurs.

Johnstone catchment: The frequency and extent of monitoring is not sufficient to calculate loads or to quantify transformation processes. It only indicates condition assessment. In the wet tropics, where more than 95% of annual loads can be delivered in one storm that only lasts a few days, accurate calculation of loads requires event-based sampling.

Murrumbidgee catchment: The emphasis on using riverine loads to assess the effectiveness of catchment management and the impacts of landuse change on river conditions is likely to increase in future. The results in this catchment showed that most (80–85%) of the regular observations characterised only about 30% of the load, and the remaining small percentage of observations characterised most of the load. These results indicate that efforts should be redirected away from routine sampling towards event-based sampling. This would not necessarily require an increase in effort. The analysis of the Murrumbidgee catchment data suggests that four well timed observations during an event would capture as much detail of the overall load being transported as 22 observations at regular intervals.

Latrobe catchment: This catchment has very efficient, coordinated and ready-to-use monitoring data. Although it is based on evenly spaced sampling, the seasonality of flows in Victoria is less sporadic than in Queensland and the monitoring is therefore considered adequate.

Brisbane catchment: The frequency and extent of monitoring is not sufficient to either calculate loads or to quantify transformation processes. It only indicates condition assessment.

There is great scope to improve the quantity and quality of information about contaminant movement through catchments by adopting a coordinated approach to monitoring, whereby information on all scales, ranging from state-wide to local, can be collected by the appropriate authorities to a common set of standards with stringent quality control and quality assurance criteria. There are wide divergences in the capacities of the various levels of government to meet their water quality and aquatic ecosystem health information needs, especially at the regional level. Many regions suffer from insufficient funding, too few institutional partners, limited access to sources of expert knowledge, and councils that are widely separated geographically.

The most important features of a coordinated monitoring and data gathering system are:

- coordination and leadership by one body (comprising representative stakeholders) which sets standards and quality control for data collection and storage in a central repository,
- a communication strategy — to ensure effective two-way communication of data and information, and
- sufficient resources to share among partners.

Modelling improvements

In order to develop techniques for catchment managers to effectively target remedial actions, we need further development and field testing of new quantitative descriptions of sediment and nutrient generation and transformation processes from improved monitoring. These can then be incorporated into the predictive models. A number of parts of the SedNet model need further development to improve local-scale predictions. For robust predictions of long-term annual sediment loads:

- predictions of channel bank erosion, bed sediment transport and floodplain deposition depend on knowing the size and shape and hydraulic condition of the channel,



The Brisbane River at Savages Crossing.

- hillslope delivery ratio — needs to be made spatially variable (at present a constant value of 5 or 10% of the mobilised sediment is treated as reaching the stream and is applied universally across all catchments),
- channel bank and gully erosion — often represent the largest sediment sources but are poorly represented in the model,
- measurements of floodplain deposition — although estimates seem to be in the right ballpark (0.5 to 2 mm/year) local measurements would help improve these estimates, and
- floodplain hydrology — none of the available models deal with the typical conditions in many of the river systems in the wet tropics of Queensland where extensive overbank flow has the potential to be both a source and a sink of nutrients. Groundwater inflows are a possibly significant contributor of dissolved nutrients but there is a paucity of observations and the methodology is not clearly defined.

For contaminant transport, the following information should be improved:

- dissolved nutrient inputs per land use,
- concentration of particulate of C, P and N in banks and gullies,
- event-based in-stream nutrient and DOC concentration measurements,
- for shorter time scales, the contribution of hyporheic flows and groundwater interaction are virtually unknown, and
- nutrient removal and releases processes such as plant uptake, adsorption/desorption, organic matter remineralisation and sediment release are not included in existing catchment models.

For contaminant impacts on ecological responses:

- temporal patterns of delivery — to be of greater use in predicting ecological responses to changes in flow, water quality and bed sediments, the model needs to be able to make predictions on a much shorter time step, and
- there is a need to establish a few key sites for long-term measurements to evaluate natural variability as well as the impact of potential implemented changes.

Conclusion

Catchment management authorities need a good understanding of contaminant sources and transport through catchments for assessing annual loads at catchment outlets and subsequent impacts on coastal water; predicting impacts on ecological responses, such as habitat changes, light availability and stream metabolism; scenario testing; and prioritising remedial action. Regional catchment management groups need a high level of technical information about sediment and nutrient transport in their catchments so they can plan and evaluate on-ground remedial action. This includes information about where sediments and nutrients are generated and transported.

Sediment and nutrient budgets provide a method for predicting the level of sediment and nutrient inputs to waterways over time. The four catchments studied in vastly different environments have given a picture of how sediment and nutrient generation and loss change with environmental conditions. The study also highlighted the importance of carefully designed monitoring programs which reflect the purpose for which data is being collected, and identified necessary improvements to the models being used for sediment and nutrient budgeting in these catchments.

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Interactions between contaminants

While understanding and managing river contamination by single substances might be relatively simple, very little is known about the synergistic or antagonistic effects of different contaminants. Different contaminants may chemically interact during transport or once deposited, and the ecological responses to ‘cocktails’ of chemicals are likely to be wide-ranging and complex. The interactions between contaminants, the net ecological responses, and the links back to catchment and river management options are relatively unexplored in catchment-scale research.

The largest gaps in our understanding are those related to the interactions between contaminants, both in terms of how they interact physically and chemically in transport or in storage, and in terms of the complex responses of aquatic biota to mixtures of contaminants. While relatively simple experiments can provide information about the tolerances and responses of individual organisms to particular contaminants, or even combinations of contaminants, scaling these results up to predict ecosystem level response is extremely difficult. The combination of detailed experimental work with medium scale field test and large-scale system modelling is likely to be the best way to advance our understanding.



The following three chapters examine this problem from different perspectives. Chapter 9 studies the interactions between flow and contaminants at a range of scales, and provides comments about implications for management of environmental flows. Chapter 10 considers the role of ecological risk assessments in helping managers and communities make difficult

decisions under conditions of uncertainty and incomplete data, with two practical examples; it emphasises the importance of monitoring to evaluate and improve decision-making. Chapter 11 discusses the role of models in helping to scale up from processes to catchment-scale targets, again with examples drawn from NRCP research.

Near to Wyangala Station homestead, Wyangala Dam, NSW, one of the sites used in NRCP research studying the interactions between different contaminants.



Chapter 9 — Managing regulated flows and contaminant cycles in floodplain rivers

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Summary

- Understanding ecosystem processes such as nutrient cycling, primary production and respiration (metabolism), and their integrated response to present day contaminant and flow regimes, is critical for the management of regulated rivers to improve river health and sustain industries and populations reliant on water resources.
- Identifying the sources and sinks of contaminants such as nutrients, salts and sediment at multiple spatial scales (e.g. catchment, sub-catchment, reach, habitat) is important for all river systems where environmental flow regimes are developed with the aim of improving river health. This information also allows for the identification and prioritisation of restoration initiatives such as riparian and corridor plantings in catchments identified as contributing disproportionately high loads of contaminants.
- Using the processes and protocols developed through this research, priority areas for river and landscape management can be identified. For example, unregulated tributaries in the upper reaches of catchments can hold significant stores of salts and nutrients within the stream channel that are readily mobilised during small rainfall events. This can have localised detrimental effects on river function.
- Research has shown that catchment run-off events can have a different chemical character and, consequently, a quite different ecological significance to releases from dams. Managers therefore need to consider if topping up small runoff events with low-nutrient water from large dams (without entraining material from the floodplain) will meet the goals for these environmental water releases.
- Understanding the cycling of contaminants throughout the year is an important component of managing the health of regulated river systems. The first irrigation flows for the season can have elevated contaminant loads from scouring of contaminants stored within the channel under preceding low flow conditions. The use of environmental flow releases that precede water used for irrigation will help dilute and transport this material out of the system rather than deliver it to irrigation areas.
- In large, regulated rivers of the southern Murray-Darling Basin, turbidity and temperature, and not contaminants, appear to be the drivers of riverine metabolism. This may be due to the relatively low concentrations of contaminants currently in the system. The reduced productivity of tributary creeks where contaminant loads are highest, provides evidence for the potential detrimental impacts of increased contaminant loads if stream and catchment restoration initiatives are not successful.

Background

Many river systems in south-eastern Australia originate in relatively wet upland ranges where they are usually highly regulated by large impoundments. They then flow for the majority of their length through semi-arid landscapes of very low relief (Thoms & Sheldon 2000). It is this low relief that drives the hydrology of these river systems, with continued losses of water through evaporation, evapotranspiration, groundwater recharge and a lack of tributaries along the length of the river (Thoms & Sheldon 2000). These rivers have spatial and temporal flow variations that are more extreme and less predictable than those in more humid regions of the world (Lake 2000, Puckridge et al. 1998). This flow variability underpins many ecosystem processes, and regulates the transport of nutrients, carbon and biota within river channels and onto the floodplain (Robertson et al. 1999). Changes to the natural disturbance regime provided by flow variation combined with altered land-use practices have resulted in many rivers now containing highly modified sources and concentrations of natural contaminants such as nutrients, salt, and suspended sediment (Robertson et al. 1999). However, the relationships between flow regime, and the sources, sinks and transport of contaminants are poorly understood, making it a difficult task to manage flow releases to best effect for river rehabilitation initiatives.

As well as a resource for critical ecological functions and sites of great biological diversity, these rivers play an important societal and economic role in Australia. In Australia's semi-arid landscape, many river systems and their floodplains provide the resources for agricultural development and urbanisation. As a result of this development, many rivers have been modified significantly since European settlement (Crabb 1997). The storage of high percentages of mean annual runoff in headwater dams, the regulation of flows by weirs, and the extraction of water to obtain reliable water supplies, have resulted in highly regulated river flows (Maheshwari et al. 1995). This complex situation has provided unique challenges for scientists and managers attempting to restore flow regimes to regulated rivers, whilst ensuring the viability of agricultural communities (Bunn & Arthington 2002). As well as large reductions in total flow due to abstraction, the timing and duration of flows have been radically altered to meet the needs of irrigators and urban populations.

For rivers in which ecosystem processes have been degraded by flow regulation and water extraction, rehabilitation efforts often depend on the return of at least part of the natural flow regime to ensure preservation of biogeochemical and life-history cycles (Poff et al. 1997). In Australia, environmental flows are assuming a central role in the sustainable management of Australian rivers. Such flows, however, are now

Large dams play an important role in regulating the transport of nutrients, carbon and biota in rivers.

Photos throughout this chapter are courtesy of the project team.





Environmental flow releases are being implemented in a highly modified landscape.

being implemented in a landscape which has been highly modified. The water of many rivers now has a biochemical composition altered from its original state (Harris 2001), and this is highly dependent on in-channel regulated flows. For example, using environmental flows as artificial floods in regulated rivers has the potential to mobilise and transport nutrients and salts stored within the channel during low flows (Ryder et al. 2006), or conversely, they may dilute beneficial natural floods with nutrient-poor water from dams. Identifying where these contaminants are stored within the river system, the flow regime necessary to mobilise them, and the effect of these contaminants on river health, is necessary for effective management of contaminant cycles in regulated rivers.

Biogeochemical cycles and river health

Assessing the impacts of managed flow releases on the ecological health of rivers and streams is an important issue for the management of water resources in Australia.

Traditionally, these assessments have been dominated by the measurement of patterns in species distribution and abundance, which contribute important information such as the status of threatened species and their habitat requirements (e.g., Wright 1995). However, many goals of river management refer to concepts of sustainability, viability and resilience, that require an implicit knowledge of ecosystem or landscape-level interactions and processes influencing organisms or populations (Ehrenfeld 2000). Such ecosystem-level processes are based on the transformations and flow of energy and matter, and are concerned with a heuristic approach comprising interactions among biological organisms and their abiotic environment (Bunn et al. 1999, Ryder & Miller 2005). This '*biogeochemical*' approach to river health is based on the idea that the ratio of elements available to organisms affects the production and transformation of organic matter; and therefore drives the trophic structure and food webs of the river system.

The biogeochemistry of streams and rivers can be an ideal measure of their ecological condition by providing an integrated response to a broad range of catchment disturbances. Nutrients such as nitrogen, phosphorus, and carbon can play an integral role in regulating rates of primary production in floodplain rivers. However, anthropogenic changes to catchment land-use have led to increased supply of nutrients and salts from diffuse or point sources, as well as altering light and turbidity regimes through increased suspended sediment loads (Harris 2001), and loss of riparian vegetation. The impoundment of runoff and regulation of downstream water supply have also impacted biogeochemical cycles in rivers, affecting the timing, magnitude and form of contaminant fluxes available to downstream environments and biological communities (Meyer et al. 1988). These landscape-level processes define the supply of contaminants to a stream and provide the framework within which other processes operate at smaller spatial scales and shorter temporal scales to regulate their supply and availability. Interactions among fluxes of water, transported components and organisms, occur between different geomorphic features and result in a mosaic of interdependent habitats, each one a potential source, sink or site for contaminant transformation.

Both contaminant transformation and primary production can take place through stationary organisms such as biofilms attached to hard surfaces, as well as by phytoplankton that are translocated with flow. Water column processes can regulate overall stream biogeochemistry through phytoplankton and microbial metabolism driving in-stream primary production and nutrient supply, with flow regime affecting these biogeochemical processes through its influence on residence time of biota, adsorption/ desorption of contaminants from suspended sediments, and availability of light through changes in turbidity (Vink et al. 2005). However, it is also likely that attached biofilms play a significant role in river biogeochemistry, as they are often 'hotspots' of primary production (Ryder 2004, Fellows et al. 2006), acting as both a source and sink for contaminants under different physico-chemical conditions. Biofilms are assemblages of algae, fungi, bacteria and unicellular animals in a matrix of polysaccharide exudates and detritus attached to submerged surfaces such as rocks, wood, and sediment particles (Burns & Ryder 2001). The diversity of organisms within biofilms and the range of biogeochemical processes they perform means that changes in the concentration or composition of contaminants can have profound impacts on stream biogeochemistry and the food webs that rely on them.

Project design

Regulated rivers occur throughout inland and coastal Australia. Regulation has the potential to alter the chemistry and productivity of the system through manipulating the timing, frequency and magnitude of managed releases. Identifying the sources and sinks of contaminants such as nutrients, salts and sediment at multiple spatial scales (catchment, sub-catchment, reach, habitat) is important for all river systems where environmental flow regimes are developed with the aim of improving river health. Our aim for this work was to develop sampling strategies and protocols that



Above: Contaminant cycles can occur during transport in the water column or by organisms in sediments and biofilms.



could be transferable to other river systems. While the data gathered in this study were from a single large river in south-eastern Australia, our aim is to develop a framework for understanding contaminant cycles in regulated rivers that can be tested in regulated systems throughout Australia and internationally.

With this background, the interplay between flow, contaminants and primary production, and how it changes spatially and temporally, requires us to think beyond the water column when evaluating the cycling of matter and energy in river systems. Understanding ecosystem level processes such as primary production and food web structure, and their integrated response to present day contaminant and flow regimes is critical for the management of regulated floodplain rivers to sustain processes vital to improve river health. This project sought to identify the role of different ecological compartments (water column, surface sediments of the wetted channel bed, and littoral biofilms) in contaminant cycles and investigate relationships between flow regime, contaminant sources, sinks and transformations and stream metabolism. We chose the Murrumbidgee River as the focus for our study as it is one of the most regulated rivers in the Murray-Darling Basin, has unregulated tributaries with varying degrees of salinisation, and has well defined flow periods.

Below: Biofilms on snags are often hotspots for primary production as well as a source and sink for riverine contaminants depending on flow conditions.



The Murrumbidgee Catchment — A case study in managing regulated flows and contaminant cycles

The Murrumbidgee River

The Murrumbidgee River in south-eastern Australia is a major tributary of the Murray-Darling River system and has a total catchment area of 84,000 km² (Figure 1, on the following page). It flows over 1500 km from its source in the Snowy Mountains to its confluence with the Murray River. Annual average rainfall varies across the catchment from 1500 mm in the headwaters to only 300 mm at Balranald. The eastern regions of the catchment have been extensively cleared for grazing and cropping agriculture, with the semi-arid western region having areas of intensive irrigated agriculture with high water demands.

This study focuses on the 657 km stretch of the Murrumbidgee River and its main tributaries from Burrinjuck Dam to Carrathool. The upper Murrumbidgee Catchment stretches from the headwaters of the Murrumbidgee River in the Snowy Mountains to just downstream of Gundagai. The upper catchment covers approximately 13,000 km² and contains numerous regulated and unregulated rivers with high gradients and narrow channels. The dominant substratum in the upper catchment is cobble, creating riffle, pool and run habitats in the Murrumbidgee River and its tributaries. The floodplain reaches occur from Gundagai to Hay with a widening of the alluvial floodplain and a reduction in gradient occurring from east to west, resulting in a meandering river channel and floodplain wetlands. Depositional banks and fallen River Red Gum (*Eucalyptus camaldulensis*) logs and associated organic debris are the dominant in-stream habitat types in these reaches.

The Murrumbidgee is a heavily regulated river with 26 dams and weirs, and over 10,000 km of irrigation canals (Kingsford 2003). The large capacity Burrinjuck (1,026,000 ML) and Blowering (1,600,000 ML) dams both provide irrigation releases, and stock and domestic water to the lower Murrumbidgee catchment. Maximum mean discharge of 12,700 ML d⁻¹ occurs at Wagga Wagga, and is reduced downstream to 4178 ML d⁻¹ at Balranald due to anabranches, distributaries, irrigation diversions and evaporation (Page et al. 2005). Prior to regulation, flows were highly variable, exhibiting winter/spring maximums. Regulation for downstream agricultural irrigation and domestic and stock uses has altered the flow to a summer dominated regime. Storage of the majority of upper catchment rainfall in Burrinjuck and Blowering dams has resulted in a decrease in the

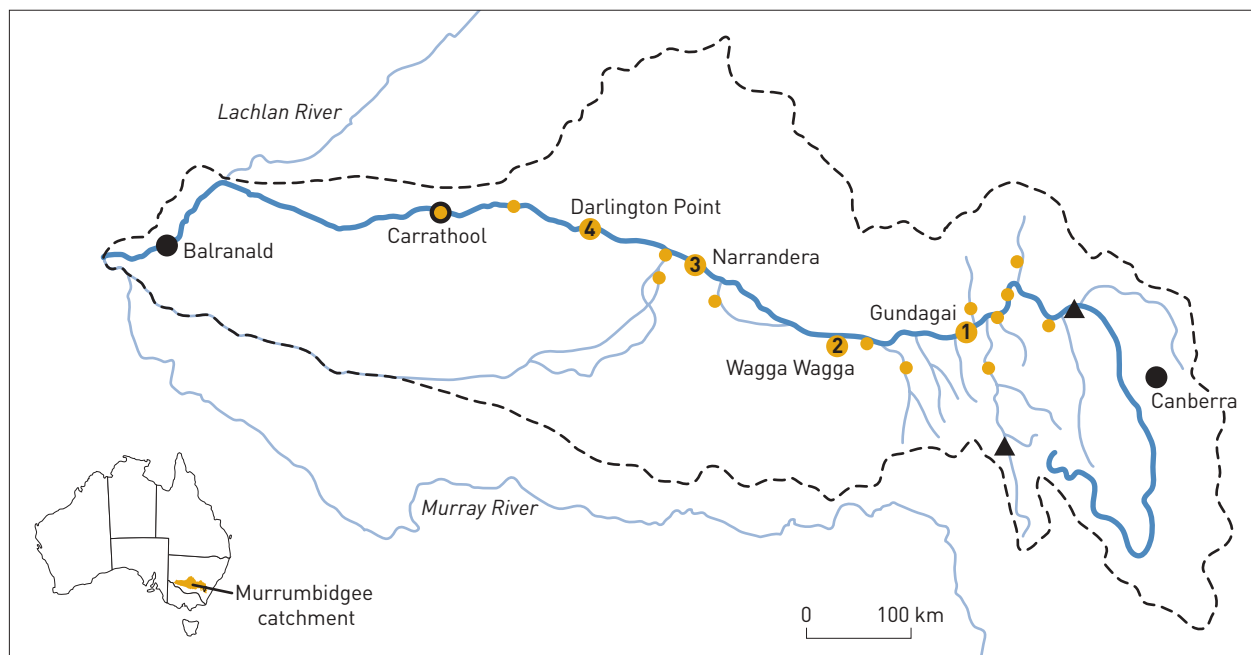


Figure 1. The location of the 17 sample sites in the Murrumbidgee catchment, and the four stations on the main stem at 1) Gundagai, 2) Wagga Wagga, 3) Narrandera, and 4) Darlington Point used for temporal changes in contaminant loads and river metabolism.

frequency and duration of small to moderate flows in floodplain reaches (Page et al. 2005). The river flows in the mid-catchment are regulated by the upstream dams, but are also influenced by several influent tributaries. Downstream of Wagga Wagga, numerous distributaries, regulatory structures and off-take channels for the consumptive use of water regulate hydrology.

Identifying the spatial distribution of contaminants

The first step in exploring contaminant cycles in rivers is locating sub-catchments that might provide disproportionate loads of contaminants to the main river channel, and discovering if biofilms and sediments are acting as biological traps for contaminants that might be released under certain flow conditions. The ‘Murrumbidgee Catchment Blueprint’ and subsequent ‘Murrumbidgee Catchment Action Plan’ both highlighted the need to develop maps that outlined priority sub-catchments for salinity management. Based on the ‘Murrumbidgee Catchment Management Authority Case Study — application of state-wide standards and targets’ (2004) and research data that were regarded as “isolated, sporadic or completely absent” (Draft MCAP 2005) 12 sub-catchments were priority listed. However, this project sought to provide a more detailed understanding of where salt occurs within and among streams, and how it might influence the biogeochemistry of the river.

The information that led to the listing of the priority sub-catchments was based on monitoring of water column ‘electrical conductivity’ (EC) as a surrogate for salinity. EC is measured as an electrical current that flows through the water and is proportional to the concentration of dissolved ions in the water — the more ions, the more conductive the water resulting in a higher electrical current. However, not all salts are the same, as high concentrations of sodium, potassium and magnesium can be detrimental to river health, while calcium can promote primary production in certain conditions (see also Chapter 2). We therefore wanted to quantify the concentration of total salt and the dominant cations (calcium, magnesium, potassium, sodium) in each of three major habitats i) water column, ii) sediments and iii) biofilms, under base flow conditions to identify catchments and habitats with the highest potential for supplying salts to the main stem of the Murrumbidgee River.

Seventeen sites on tributaries, distributaries and the main stem of the Murrumbidgee River were sampled in February 2005 at the cessation of irrigation flows, and at base flow in tributary streams (Figure 1). Influent tributaries included the highly saline Muttama and Jugiong Creeks ($> 2000 \mu\text{S/cm}$), the moderately saline Tarcutta Creek ($400 \mu\text{S/cm}$), and the low salinity Tumut River ($30 \mu\text{S/cm}$). Main stem sites stretched over 650 river km from constrained reaches immediately below the Burrinjuck Dam to floodplain reaches at Carrathool. At each site, triplicate samples were collected

for: i) filtered water ($0.45\ \mu\text{m}$), ii) $3.9\ \text{cm}^3$ surface sediment cores ($19.6\ \text{cm}^2 \times 0.2\ \text{cm}$ deep) from permanently submerged in-stream sediments, and iii) $19.6\ \text{cm}^2$ cores of algal biofilms from permanently submerged cobble (in upland reaches) and wood (in lowland reaches) substrata. Surface waters were measured in situ for electrical conductivity and turbidity, and in vitro for total nitrogen (TN) and total phosphorus (TP). Dried sediment and biofilm samples and filtered water samples were analysed for potassium, magnesium, calcium and sodium using a Perkin Elmer ELAN 6000 inductively coupled plasma mass spectrometry (ICP-MS).

This column, top to bottom: Downstream Burrinjuck Dam, Tumut River, Gundagai.



Sources and sinks of contaminants in the Murrumbidgee catchment

Two main trends are immediately evident from the data shown in Figure 2; there is a trend of decreasing salinity in all habitats from headwater tributaries to floodplain reaches, and the chemical composition of the water column is exceptionally distinct from that in the biofilm and surface sediments. Water column salts were dominated by sodium (with very low calcium concentrations) and the water column was consistently higher in sodium concentration than other habitats. The total concentration of salts was highest in upper

This column, top to bottom: Wagga Wagga, Narrandera, Carrathool.



catchment tributaries and, importantly, contained very high relative concentrations of sodium. Water column concentrations of salts in the main stem of the Murrumbidgee River were grouped into reaches, with intermediate salinities upstream of Gundagai, and those from Gundagai to Carrathool with low salinities and total salt concentrations consistently less than 20 mg L^{-1} .

At the time of sampling, the Tumut River was the dominant source of water to the main stem of the Murrumbidgee, as there were minimal inflows from tributary streams. The Tumut River has the lowest concentration of water column salts at 4.5 mg L^{-1} , and highlights this river's role in diluting the salt concentrations contributed to the main stem of the Murrumbidgee by upstream tributaries. The lower concentration of salts in upstream constrained reaches suggests that the high salt concentrations in tributary streams (Jugiong 201 mg L^{-1} and Muttama Creeks 385 mg L^{-1}) play a minimal role in contributing salt loads to the Murrumbidgee River during base flows. Tarcutta Creek was also identified as a priority sub-catchment for salinity. Under base flow conditions, it too had water column concentrations more similar to those of the Murrumbidgee main stem at 35 mg L^{-1} , however, the salts within Tarcutta Creek are dominated by sodium, similar to the influent streams higher in the catchment.

The biofilms and surface sediments in all sites contained substantial concentrations of salts (up to $19,000 \text{ mg kg}^{-1}$ in Muttama Creek), with relative concentrations that were chemically distinct from the water

column, but displaying the same trend of a longitudinal decrease in salinity. The high concentrations of salts in biofilms and sediments highlight their importance as a store during low flows and a potential source during in-stream freshes. Importantly, these stores contain relatively small concentrations of sodium, and high concentrations of calcium, magnesium and potassium. Ryder et al. (2006) have demonstrated that flow velocities as low as 0.3 to 0.55 m sec^{-1} were sufficient to significantly reduce biofilm biomass in low gradient regulated rivers, releasing the salts and nutrients stored within these habitats to the water column. This suggests that environmental flow releases may mobilise these contaminants from biofilms (and also potentially from inundated surface sediments within the channel), and may result in shifts in main stem biogeochemistry and subsequent changes to in-stream productivity and food webs.

The longitudinal trend of decreasing downstream concentrations of salts and differences in chemical composition among habitats was mimicked by total phosphorus and total nitrogen concentrations. This highlights that knowledge of the chemical composition of stored and suspended salts is required for prioritising catchments for rehabilitation, or for the timing of releases from impoundments for environmental purposes. Our results indicate that catchment run-off events that mobilise contaminants stored in tributaries will have quite a different chemical character to artificial floods from dam releases and, consequently, a quite different ecological significance.

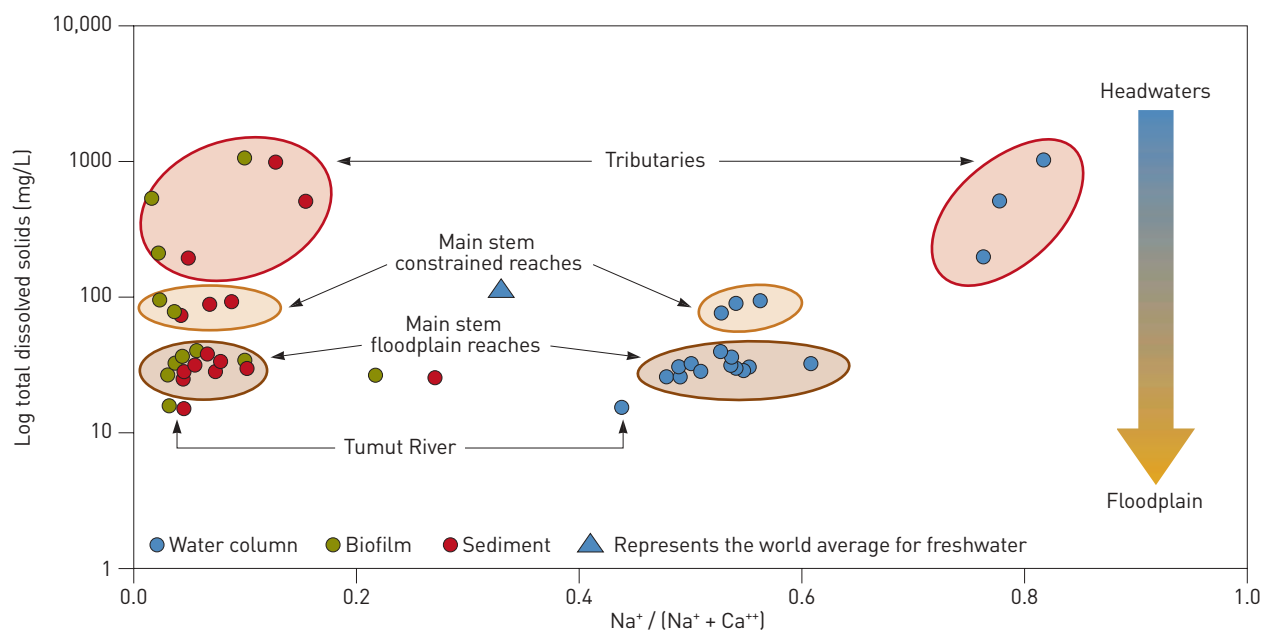


Figure 2. Relationship between salinity and cationic composition for tributaries and main stem reaches of the Murrumbidgee River. [Project data displayed as proposed by Gibbs 1970].

Linking hydrology, contaminants and productivity

Just as identifying the spatial distribution of contaminants is an important step in understanding contaminant cycles, effective river management also requires that temporal changes relative to flow regime and how these might influence river health are measured.

Current water quality monitoring practices involve sampling at a particular site at fixed, widely-spaced time intervals. In large regulated systems such as the Murrumbidgee, the relative timing and frequency of sampling at the various sites bears little relationship to the transit time of the water between sample sites, and takes no account of changes in discharge. In the Australian context of high flow variability, these practices are problematic as annual average concentrations, are strongly biased towards the low-flow contaminant concentrations and the loads derived from these data are very imprecise.

In this project we adopted an alternative ‘Lagrangian’ sampling strategy (Rutherford 1994) where we followed the evolution of the water column primary production capacity, respiration and contaminant load by measuring repeatedly the same ‘parcel’ of water as it progresses through the system. This requires a series of widely-spaced stations where the relevant ecosystem parameters, including contaminant concentrations and discharge, are sampled at a relatively high frequency, but only when that water parcel is passing a monitoring station. This approach allowed an understanding of the interplay between flow, contaminants and primary production and how it changes spatially and temporally.

Hydrology

The Lagrangian framework was based at four widely-spaced sites on the main stem of the Murrumbidgee River: 1) Gundagai, 2) Wagga Wagga, 3) Narrandera and 4) Darlington Point (as numbered in Figure 1). To encompass the hydrologic variability of the Murrumbidgee River below the two principal storages, with its mixture of regulated flows and episodic events due to rainfall, we adopted a sampling regime that represented each of the major flow categories that occurred in the river during the life of the project:

- regulated low flows (March to July), < 600 ML/day,
- in-channel regulated releases (July to October environmental flows), 1000–18,000 ML/day,
- in-channel fresh from catchment run-off,
- commencement of irrigation flows (November), < 5000 ML/day, and
- established irrigation flows (December to February), < 10,000 ML/day.

Irrespective of the magnitude of the discharge, the flow regime of the Murrumbidgee has a consistent downstream trend of discharge, increasing to a peak at Wagga Wagga, and then decreasing to Darlington Point. Although the pattern of discharge was similar throughout each flow period, the variability at each site was unique. During the low flow periods, discharge was most variable at Darlington Point, where extractive uses and the manipulation of upstream weirs contributed to highly variable flows. The combination of environmental releases and catchment rainfall resulted in high flow variability over the length of the river system during this period. Our sample collection in the environmental flow period coincided with a catchment rainfall event that peaked at 17,765 ML/day at Wagga Wagga, driven by upper catchment rainfall that had peak inputs from the highly saline Jugiong Creek (1600 ML/day @ 1500 $\mu\text{S}/\text{cm}$) and Muttama Creek (350 ML/day @ 1900 $\mu\text{S}/\text{cm}$), and Tarcutta Creek (2300 ML/day @ 430 $\mu\text{S}/\text{day}$).



In-stream benches and the riparian zone play an important role in regulating the input of contaminants in floodplain rivers.

Contaminants

The relative concentrations of the major cations (Na^+ , K^+ , Ca^{2+} , and Mg^{2+}) in each of the different compartments (water column, biofilm, and sediments) remained relatively constant irrespective of the site and flow conditions. The reach scale budgets are presented in Table 1 on a time and discharge normalised basis ($\text{kg d}^{-1} \text{GL}^{-1}$) to remove solely flow related changes for water column only, as no research has been conducted to quantify the aerial extent of biofilm and sediment surface habitats, and load estimates cannot be calculated for these environments. The results are thus flow weighted concentrations, and the differences (either increases or decreases) between adjacent stations reflect either net release or uptake from compartments within the reach between the two stations. In interpreting the contaminant budgets (Table 1), it should be kept in mind that the September 2004 data captured an event where rainfall throughout the catchment led to runoff that was observed at all stations. In November 2005, a relatively small inflow containing terrestrially derived material was observed only between Narrandera and Darlington Point.

Table 1 illustrates that much of the total contaminant load enters the system upstream of Wagga Wagga, and then declines as an increasing fraction of the discharge is

diverted from the main stem for irrigation. Sodium is the dominant cation in the environmental flow period from catchment runoff and the commencement of irrigation period. A paradoxical consequence of the high diversions is the delivery of additional salt to the irrigation areas, as only in unregulated environmental flows is this material transported out of the system. The daily cation loads during irrigation flow are approximately twice the loads during constant low flows, but both are considerably smaller than those at the commencement of irrigation flows. This is probably due to higher evaporation towards the end of the irrigation season, as well as greater groundwater inputs arising from higher water levels during the preceding full supply period.

Under the constant low flow period (May 2004), the amount of sediment entering the system is low and there is very limited generation of suspended sediment in the Gundagai to Wagga Wagga stretch of the Murrumbidgee. From Wagga Wagga to Narrandera there is deposition of sediment and the system is in approximate sediment equilibrium between Narrandera and Darlington Point. Under irrigation flows (February 2005) the concentration of sediment entering the system has increased and there is additional re-suspension between Gundagai and Wagga Wagga and limited deposition in the Wagga Wagga to

Table 1. Major cation and nutrient budgets [$\text{kg d}^{-1} \text{GL}^{-1}$] for the Murrumbidgee water column at main stem sampling stations under different flow regimes. N.D. = no data available.

Date and station	Na ⁺	K ⁺	Mg ²⁺	Ca ²⁺	TSS	TP-P	FRP-P	TN-N	NO ₃ N
Low flows									
Gundagai	4900	928	2179	6091	9493	27	4	309	90
Wagga Wagga	11430	2223	6332	15901	10489	29	4	308	83
Narrandera	3400	714	1823	4853	5346	23	4	270	100
Darlington Point	1874	384	951	2427	6849	26	3	298	105
Environmental flows									
Gundagai	20024	4789	7266	14546	11441	47	13	1067	650
Wagga Wagga	82108	11938	33310	55913	N.D.	65	13	1292	762
Narrandera	43883	6607	18186	34506	36863	50	9	1067	711
Darlington Point	33857	5181	13844	23314	51110	47	9	1295	786
Commence irrigation flows									
Gundagai	7939	1500	2858	5573	13466	24	3	345	156
Wagga Wagga	92247	9816	36184	34320	38286	33	4	348	82
Narrandera	15328	2443	6867	12847	N.D.	37	2	306	2
Darlington Point	11466	1914	4897	7210	75554	34	3	703	330
Irrigation flows									
Gundagai	7285	1504	2954	6551	15620	14	2	253	96
Wagga Wagga	25160	5522	12161	21210	29344	22	2	216	40
Narrandera	12589	2328	8366	10206	26075	19	2	181	1
Darlington Point	3222	893	1737	2777	21604	20	2	243	1

Darlington Point sector. These changes reflect only in-stream processes. The case of the environmental release (September 2004) coincided with significant runoff throughout the catchment and the concentration of suspended sediment increased downstream. We infer that this additional material arose from tributary and terrestrial inputs rather than from in-stream processes.

The most significant result is the consistent low concentrations of total nitrogen (TN) (ANZECC trigger value for south-east Australian lowland rivers is 500 kg GL⁻¹ and 50 kg GL⁻¹ for TN and TP respectively) and total phosphorus (TP) and the even smaller amounts of the dissolved inorganic species — NO₃ (nitrate) and FRP (filterable reactive phosphorus) — that are immediately bioavailable to organisms for uptake and growth. This suggests that there is a very tight coupling between the processes producing and consuming the dissolved nutrients. Given that light and temperature limitation affect phytoplankton growth throughout the year, these results underline the importance of bacterial processes within the system. The role of the terrestrial inputs as a source of NO₃ is especially noteworthy in summer at the downstream sites which are generally nutrient limited, when a small inflow produced a hundred-fold increase in NO₃ concentration. Our results

show that catchment run-off events that mobilise terrestrial and in-stream contaminant stores will have quite a different chemical character to managed dam releases.

Ecosystem metabolism

The production and transformation of organic carbon is a fundamental process that regulates the trophic structure of river systems and is reliant on the supply of macronutrients (TN and TP) and trace elements (Ca, Mg, K). Our data have highlighted the importance of catchment driven events in changing water chemistry and illustrated the potential constraints on ecosystem metabolism from prolonged regulated in-stream flows.

The four main stem sites and four flow periods within the Langrangian framework were also used to examine the trophic status of the Murrumbidgee River relative to hydrology and contaminant loading, and to partition the role of dominant habitats in regulating the river as an autotrophic or heterotrophic environment.

Ecosystem metabolism consists of respiration that represents the consumption of oxygen, and gross primary production (GPP) that represents the production of oxygen. Daily gross primary production (GPP) and respiration (R₂₄) were estimated using a

Low flow periods have reduced contaminant loads, but result in increased sedimentation that smothers highly productive biofilms attached to rocks, logs and sediment surfaces.





Colonised cobbles and blocks of river red gum were used for habitat specific metabolism measurements.

single station modification of the oxygen mass balance approach of Odum (1956). This approach quantifies the net contribution from all autotrophic and heterotrophic organisms in all habitats (i.e., biofilm, water column and sediment) and net gas exchange in a river reach (detailed methods are provided in Vink et al. 2005).

At the site scale, productivity within the three major habitats involved in riverine biogeochemical cycles (water column, sediment, and biofilms) were measured as the change in oxygen production or consumption within sealed chambers. Fully automated, recirculating chambers have been developed that alleviate many of the problems associated with chamber studies of metabolism in running waters. The chambers vent at regular intervals to prevent deoxygenation, provide variable flow rates across biofilm surfaces to mimic in situ velocities, and float at set depths to avoid problems associated with changing water levels. Blocks of River Red Gum (*Eucalyptus camaldulensis*) to simulate snag habitat in floodplain reaches, and cleaned cobbles in upland reaches were colonised at each site for 30 days prior to measurement, allowing the biofilm to develop in flow and water quality conditions representative of each



flow period. Sediment metabolism was measured from sediments permanently inundated during each flow period using dome chambers that push into the substrate to form a sealed environment. All chambers were run for 28 hours at each station to capture the diurnal cycle of production and respiration.

In general, the river was net heterotrophic in all reaches (i.e. respiration exceeded production), although some reaches tended towards balanced or slight net autotrophy during the Irrigation flow releases (Figure 3). Not surprisingly, lowest overall rates of GPP were observed during the autumn/winter low flow period and September environmental release when lower water temperatures reduce metabolic rates. Lowest R_{24} rates



Sealed chambers used for measurement of habitat specific metabolism.



were observed during the environmental release period. The measurement period during catchment runoff (September) is the only period in which the majority of the river was net autotrophic in the upstream reaches. These results contrast with the results from the beginning of the irrigation flow releases when the river was close to balanced upstream, but highly heterotrophic downstream. This suggests that in spite of lower water temperatures and increased sediment loads that would have decreased the light conditions (Table 1), reducing overall GPP rates during the catchment run-off event, increased input of organic matter and contaminants from catchment runoff during the spring fresh increased GPP relative to respiration. These results contrast with results from the beginning of the irrigation flow release

when higher water temperatures and better light conditions resulted in overall higher metabolic rates, but where respiration rates were greater than GPP. This markedly different response between a flow driven by catchment runoff and that driven by a dam release, is important when setting restoration targets for ecosystem processes such as metabolism.

With the exception of the September fresh, GPP generally increased downstream reflecting an increase in available nutrients (particularly nitrogen) and increased temperature. Total respiration also increased downstream during irrigation releases but tended to be more variable during other flow periods. During both the low flow period and September fresh, respiration was depressed in the middle (Wagga Wagga and Narrandera) reaches. The relatively high suspended sediment loads found in the Murrumbidgee throughout the study are likely to have the greatest impact on lowering GPP through limiting light for photosynthesis and smothering highly productive biofilm algae.

Habitat scale measurements of production and respiration reveal the water column is a net consumer of oxygen in all flow periods, with the exception of the Darlington Point site that has a positive oxygen production throughout the year irrespective of discharge. Sediment metabolism was also predominately heterotrophic, with the two most downstream sites of Narrandera and Darlington Point the only ones with positive daily oxygen production, and only during low flows. Biofilms were consistently the most productive habitat and were consistently the sole sites of net autotrophic production. They were most productive in reaches downstream of Wagga Wagga where improved water clarity and increased temperatures promoted primary production.

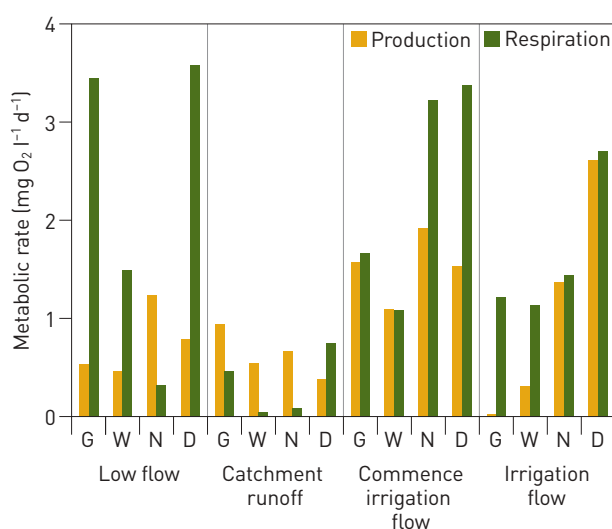


Figure 3. Whole-of-system scale rates of production and respiration recorded at each of the main stem sites. G = Gundagai, W = Wagga Wagga, N = Narrandera, D = Darlington Point.

A conceptual contaminant-flow-productivity interaction model

We have developed a conceptual model for the interactions among flow, contaminants and riverine productivity for the three flow scenarios commonly experienced in regulated floodplain rivers in south-eastern Australia; constant low flows, catchment run-off events, and releases from impoundments — either for environmental or irrigation demands (Figure 4).

Constant low flows

Riverine metabolism under low flow conditions is net heterotrophic in all reaches, however, a reduction in respiration rates and increase in gross production in the mid-reaches is associated with increased salt loads, decreased available nutrients, and a reduced suspended sediment load from the deposition of suspended sediments with low velocity flows. These changes in metabolism are seen in the water column and sediment compartments, with the concentration of contaminants in surface sediments increasing sharply only in these floodplain reaches. This supports the model that the deposition of sediments and associated contaminants in these reaches are promoting autotrophic water column and sediment production, as well as heterotrophic biofilm production through the smothering of algal assemblages with deposited silt. We hypothesise reduced loads of nitrogen in upstream reaches that occurs predominantly in low flow periods are through denitrification. This process occurs in the sub-surface (hyporheic) zones beneath the cobble bars, and the exchange of surface water with the sub-surface is maximised during periods of low flows. The loss of nitrogen from the system will also contribute to reduced metabolism in the floodplain

reaches. The increase in metabolism rates in the lower reaches reflects an increase in available nutrients (particularly N) and increased temperature.

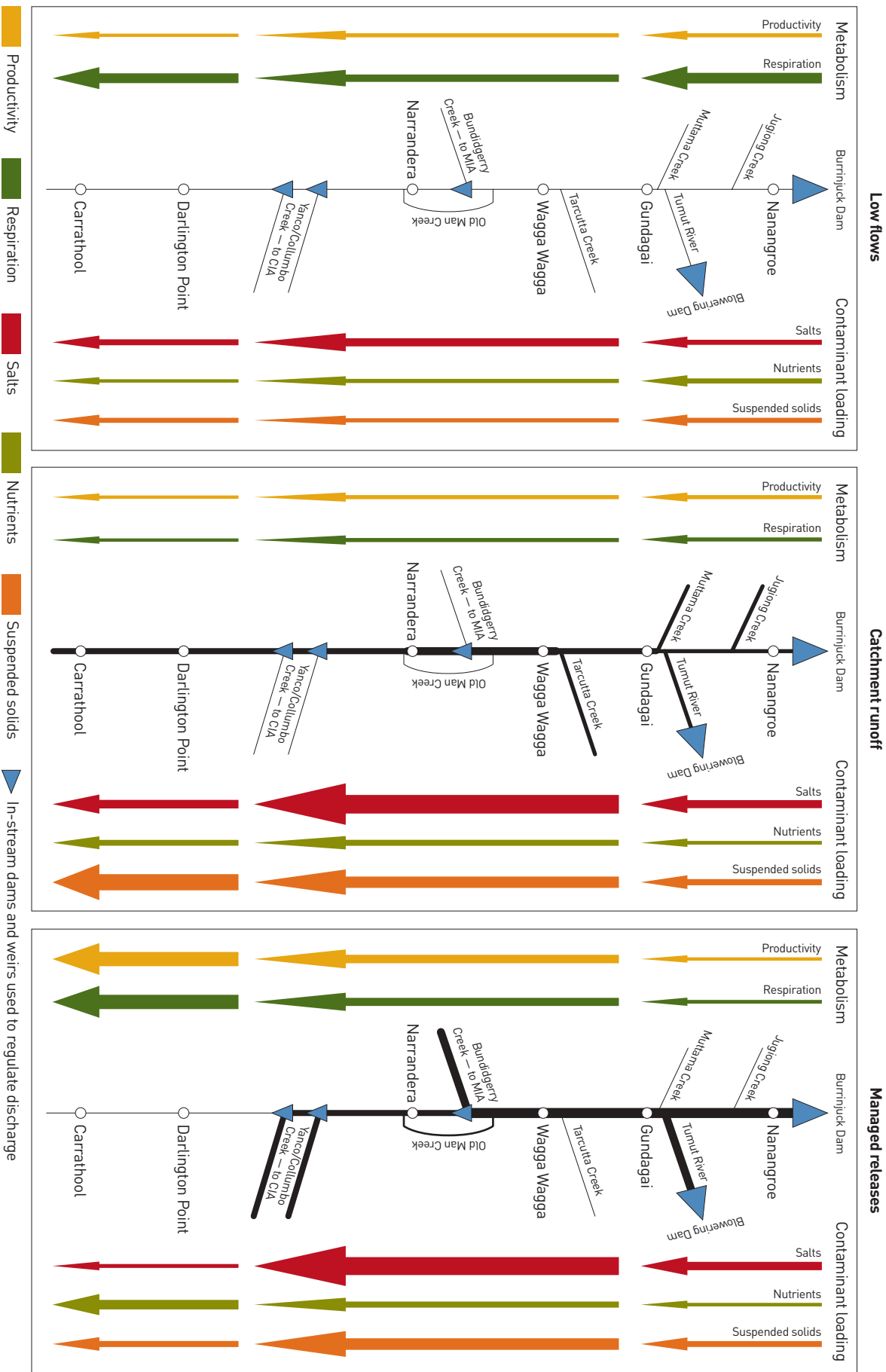
Catchment run-off events

The lowest rates of riverine metabolism were observed during the September “fresh” when lower water temperatures, increased suspended sediment load, and reduced light for autotrophic production suppressed metabolic rates, although the ratio of GPP to R_{24} increased markedly. Our model suggests that the increased input (approximately three times that of irrigation flows) of organic matter, salts and nutrients from catchment runoff (rather than from in-stream processes) during the spring fresh reduced ecosystem productivity when compared to similar discharge in November emanating from a dam release (more akin to an environmental flow release). Only during run-off generating events downstream of the storages does the water column chemistry change significantly. Sodium is the dominant cation in the environmental flows period and indicates that much of the sodium entering the system is from terrestrial run-off and not from in-stream (sediment and biofilm) and riparian sources. This markedly different response between a flow driven by catchment runoff, and one driven by a dam release, is important when setting restoration targets for ecosystem processes such as metabolism. Biofilm and sediment compartments in the floodplain reaches appear to play important roles in removing nutrients and salts (except sodium) from the water column during catchment runoff events. However, these compartments then become stores of contaminants for release into the water column during flows of sufficient discharge to scour contaminants, including irrigation flows.

Low flows expose much of the log surfaces and sediment where highly productive biofilms may colonise.



Figure 4. The interactions among flow, contaminants (nutrients and suspended sediments) and riverine productivity for the three flow scenarios commonly experienced in regulated floodplain rivers in south-eastern Australia: Constant low flows, Catchment run-off events, and Releases from impoundments — either for environmental or irrigation demands



Releases from impoundments — environmental or irrigation demands

The increase in riverine metabolism with distance downstream during discharge from impoundments reflects increased temperature, longer residence time and better light climate due to settling of suspended sediments and lower water levels along the river continuum. The role of the terrestrial inputs as a source of NO_3 is especially noteworthy in summer at the downstream sites which are generally nutrient limited, when a small inflow produced about a hundred-fold increase in NO_3 concentration. Sodium is the dominant cation at the commencement of irrigation periods and a paradoxical consequence of the high diversions are to deliver additional salt to the irrigation areas. Only in unregulated environmental flows is this material transported away from the agricultural regions and out of the system. The daily cation loads during irrigation flow are approximately twice the loads during constant low flows, but both are considerably smaller than the commencement of irrigation flows. We suggest this is due to higher evaporation towards the end of the irrigation season and to greater groundwater inputs arising from higher water levels during the preceding full supply period.

Implications for river management

Importance of identifying where contaminants are in the catchment

This project has developed a framework for identifying habitats that act as sources and sinks of contaminants at multiple spatial scales. This information also allows for the identification and prioritisation of restoration

initiatives such as riparian and corridor plantings in catchments identified as contributing disproportionately high loads of contaminants. Upper catchment tributaries are often cleared for agriculture and can be a significant source of contaminants during large rainfall events. However, we have demonstrated that the surface sediments and biofilms (attached algae and slime on rocks and logs) can hold significant stores of salts and nutrients (up to $19,000 \text{ mg kg}^{-1}$) within the stream. These stores of contaminants are readily mobilised from small catchment rainfall events and have localised detrimental effects on riverine metabolism and biodiversity. In contrast, loads of contaminants in lowland reaches are highest in catchment run-off events. Very low stores of contaminants within biofilms and surface sediments in lowland reaches suggests that the contaminants are of terrestrial or catchment origin. The focus of using managed flows to reduce contaminant loads should be based on knowledge of where in the landscape and the stream the contaminants reside, and how these interact with flow regime.

Piggy-backing flow releases with catchment floods

The piggy-backing of environmental flow releases onto catchment runoff events is a management option in many rivers. Our results indicate that catchment run-off events in lowland rivers have a different chemical character and metabolic response to releases from impoundments. Topping-up rainfall driven events with managed releases that do not connect with the floodplain and entrain organic material, may simply dilute potentially ecologically beneficial nutrients from reaching downstream foodwebs. This process was evident in this study, when a small inflow produced about a hundred-fold increase

Run-off from tributary streams can provide substantial inputs of contaminants to main stem reaches.





Initial irrigation flows have elevated contaminants which are diverted directly to irrigated crops via off-take channels.

in NO_3 concentration with subsequent increase in riverine metabolism. However, it is important to stress that managing flow releases for positive benefits such as diluting saline flows from upstream tributaries instead of diluting potentially beneficial transport of nutrients downstream, is reliant on an understanding of contaminants cycles in each system.

Which contaminants regulate ecosystem processes?

Turbidity and temperature, and not contaminants, appear to be the drivers of riverine metabolism in low nutrient floodplain river systems. Restoration efforts should be focused on maintaining or improving the current loads of suspended sediment and contaminants. This project has documented the reduced productivity and biodiversity in systems where contaminant loads are highest, providing evidence for the potential impact of increased contaminant loads if stream and catchment restoration are not successful.

Managing flows throughout the year

Understanding the cycling of contaminants throughout the year is an important component of managing the health of regulated river systems. The first irrigation flows for the season can have elevated contaminant loads (particularly sodium) from scouring of contaminants stored within the channel under preceding low flow conditions, elevated groundwater inputs from persistent saturation of riverbanks, and evaporation. Under these conditions, the initial irrigation flows have elevated

contaminant loads which are diverted directly to irrigation areas. Only in unregulated environmental flows are the contaminants transported out of the system. The reduction of discharge between environmental releases and commencing irrigation releases to dry in-stream habitats, or the use of environmental flow releases that precede water used for irrigation to help transport this material out of the system, should be considered to reduce inputs of contaminants to irrigation systems.

Conclusion

Regulated rivers occur throughout inland and coastal Australia. Regulation has the potential to alter the chemistry and productivity of the system through manipulating the timing, frequency and magnitude of managed releases. Understanding ecosystem processes such as nutrient cycling and river metabolism that fuel food webs for fish and birds, and how they respond to present day contaminant and flow regimes, is critical for the management of regulated rivers to improve river health, and sustain industries and populations reliant on water resources. We have developed a framework for understanding contaminant cycles in rivers that can be tested in regulated systems throughout Australia and the world. The framework relies on an understanding of where the contaminants are in the landscape (from catchment to habitat scales) and how each of these interact with flow regime. Armed with this knowledge, we can better tackle the sustainable use of water resources for industry, society and the environment.

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Chapter 10 — Risk-based approaches for managing contaminants in catchments

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Summary

- Natural resource managers currently have few quantitative tools to assist them in identifying which of their environmental assets are at greatest risk from degradation and then to decide upon the best options for managing these risks.
- Risk-based approaches are increasingly being used in natural resource management. An Ecological Risk Assessment (ERA) framework initially developed for the Australian irrigation industry (Hart et al. 2006), is available and applicable for assessing risks to all natural resources.
- A major difficulty with the ERA process is to obtain a quantitative analysis of the key risks when there is a large degree of uncertainty — the case for many situations. Bayesian Network modelling is a relatively new technique that shows great promise in this area.
- We provided examples of the development and use Bayesian Network models to quantify the ecological risks to a) a *Eucalyptus camphora* wetland, and b) freshwater catfish in the lower Wimmera River (environmental flows). Such Bayesian Network models will eventually be used to link catchment contaminant reduction targets (e.g. end-of-valley targets for nutrients, salinity and sediment) with the ecological benefits in receiving water bodies.
- New guidelines for monitoring and assessment programs associated with Ecological Risk Assessments are now available as a result of the work undertaken through this project.

Background

Natural resource managers currently have few quantitative tools to assist them in identifying which of their environmental assets are at greatest risk from ecological degradation and then to decide upon the best options for managing these risks.

Many catchment activities can adversely affect environmental assets, such as rivers, wetlands and estuaries. Irrigation, grazing, cropping, tourism and urbanisation all have the potential to cause undesirable ecological effects, and it is common that managers have to address the effects of multiple threats and hazards. Irrigation, for example, may be associated with threats such as salinity, toxicants, nutrients and altered flow regimes. These threats, in turn, may affect valued assets of the catchment such as aquatic ecosystems. The relevant benefits and costs of different management options having uncertain/unpredictable outcomes will need to be weighed against one another.

Risk-based approaches, such as Ecological Risk Assessment (ERA) are increasingly being adopted in Australia (Hart et al. 2005, Burgman 2005) and overseas (Borsuk et al. 2006, Uusitalo 2007) to assist in the management of natural resources. In fact, risk-based approaches are now required or recommended in many regulatory or management contexts, including Australia's National Water Quality Management Strategy and Victoria's State Environment Protection Policy 'Waters of Victoria'.

This chapter covers the main findings from research undertaken to develop risk-based approaches for managing contaminants in catchments using the ERA Framework developed by Hart et al. (2005). Two case studies were undertaken to trial the development and application of Bayesian Network (BN) models as a quantitative risk-based assessment tool to provide guidance to assist in improving the management of diffuse contaminants in catchments. These were: a) managing a *Eucalyptus camphora* wetland, and b) managing environmental flows in the lower Wimmera River. Such BN models will eventually be used to link catchment contaminant reduction targets (e.g. end-of-valley targets for nutrients, salinity and sediment) with the ecological benefits in receiving water bodies.

Ecological risk assessment

Ecological risk assessment is the process of estimating likelihood and consequence associated with the effects of human actions or natural events on plants, animals and ecosystems of ecological value, i.e. the study of risks to the natural environment. These risks may be biological

(e.g. predation, invasive species), physical (e.g. drought, flood) or chemical (e.g. toxicants). ERA also often needs to consider social, political or economic issues, which may be important in influencing ecological outcomes. The objective of ERA is to provide a robust process that incorporates a transparent, scientific, precautionary and ecologically sustainable approach to the management of environmental risks.

There are several frameworks for risk assessment and management available for different settings and disciplines, as well as some specifically aimed at ecological risk (e.g. Hart et al. 2005, Burgman 2005). These frameworks have many features in common, in that they outline a structured iterative process for the identification of threats or hazards, the analysis of risks to valued assets, the management of these risks, and the monitoring of outcomes to ensure the management plan is working.

The Water Studies Centre and partners have developed an ERA Framework that is catchment-based and focuses on the difficult task of assessing the risks to multiple ecological assets from multiple hazards (Hart et al. 2005). The framework synthesises the methods required to achieve successful adaptive management of natural resources. This framework is primarily focused on the risks to aquatic ecosystems (e.g. rivers, wetlands, estuaries), but is robust enough to be used to assess the ecological risks to other natural resource assets in catchments (e.g. land, soil, vegetation, biodiversity), as is illustrated in the case studies outlined on the following pages.

What is involved?

The ERA framework involves a number of key steps (Figure 1), 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 assets, and identifying hazards to these assets* — hazards are prioritised by evaluating their effects on valued elements of ecosystems and ecosystem services.
- *Analysing the risks to the ecological values* — risk analysis involves combining information on two aspects (AS/NZS, 2004): a) the *likelihood* of the hazards or threats having an effect, and b) the size or magnitude of the effect if it does occur (*consequences*). The analysis process used (qualitative or quantitative) needs to be appropriate for the situation in order to provide adequate information for decision-making. In this chapter we provide information on a new quantitative technique — Bayesian Network (BN) modelling.

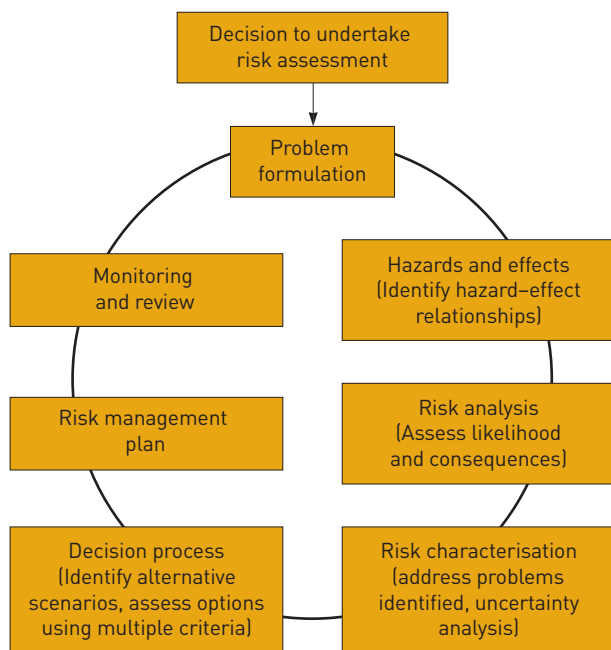


Figure 1. Overall risk assessment and management framework (Hart et al. 2005).

- *Characterising the risks* — the technical details of risk analyses need 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 process* for making changes if needed.

The key point of this framework is that it should be iterative and adaptive, allowing new information to be incorporated into the risk management plan as it becomes available.

The major **advantages** of modelling are (Uusitalo 2007):

- ability to handle missing data,
- excellent tool for expert elicitation,
- allow combination of different forms of knowledge, from expert opinion and intuition to quantitative data,

- facilitate learning about causal relationships between variables,
- show good prediction accuracy even with rather small sample sizes, and
- can be easily combined with decision analytic tools to aid management.

The major **limitations** of BN models are:

- the need to express conditional probabilities as discrete forms so that models can be solved analytically (unlike Bayesian hierarchical models which use Monte Carlo methods, where distributions are estimated by simulation),
- the inability to incorporate feedbacks or loops in models, and
- the difficulties associated with eliciting expert knowledge, and evaluating models built largely on expert opinion.

Bayesian network models

Why use them?

A major difficulty faced by most managers of aquatic and terrestrial resources is the need to make decisions for situations where there is considerable uncertainty in understanding how the system works and how particular management actions will influence the system. It is rare to have well understood cause-effect relationships between the threats and the ecosystem.

For these reasons, BN models are increasingly being used as decision support tools to aid in the management of ecological systems, and particularly in those situations where the risks are such that quantitative methods are warranted (Burgman 2005, Pollino & White 2005, Hamilton et al. 2005, Pollino et al. 2006a, b 2007).

A particular advantage of BN models is that they can incorporate both quantitative information (obtained from existing models, monitoring and from site-specific investigations) and qualitative information (obtained mostly from expert opinion), and can be updated as new information or data becomes available (Uusitalo 2007).

What are they?

Bayesian decision networks are graphical models used to establish the causal relationships between key factors and final outcomes (cause-effect relationships). They can readily incorporate uncertain information, with uncertainties being reflected in model outputs. They are particularly useful in modelling ecological processes because Bayesian inference provides a probability-based approach that can update scientific knowledge when new information becomes available.

How do they work?

A Bayesian decision network is made up of a collection of nodes that represent important environmental variables. Arrows represent the causal relationships between the nodes (variables). BNs use the network structure to calculate the probability certain events will occur, and how these probabilities will change given subsequent observations or a set of external (management) interventions:

- A prior probability represents the likelihood that an input parameter will be in a particular state.
- The conditional probability calculates the likelihood of the state of a variable given the states of input variables affecting it.
- The posterior probability (or predicted probability) is the likelihood that a variable will be in a particular state, given the input variables, the conditional probabilities, and the rules governing how the probabilities combine.

A number of commercially available modelling shells are now available. The software Netica (www.norsys.com) was used in our studies.

What is involved in building a Bayesian network model?

Model structure

The first step in constructing a BN is to develop a causal structure (often based on a conceptual model), with relevant variables (nodes) and dependencies. Important criteria for inclusion of variables in BNs are that the variable is either: a) manageable, b) predictable, or c) observable at the scale of the management problem. Any processes or factors not included become part of the uncertainty of the network, forming the predictive uncertainty described in probability distributions. (See Figure 2.)

One of the strengths of BNs is their ability to integrate existing models or processes and to integrate existing datasets. Figure 3 shows the conceptual

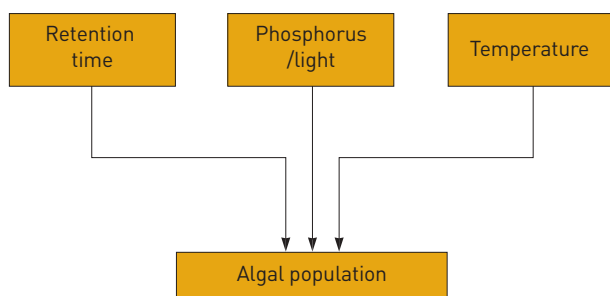


Figure 2. Simplified version of an algal network.

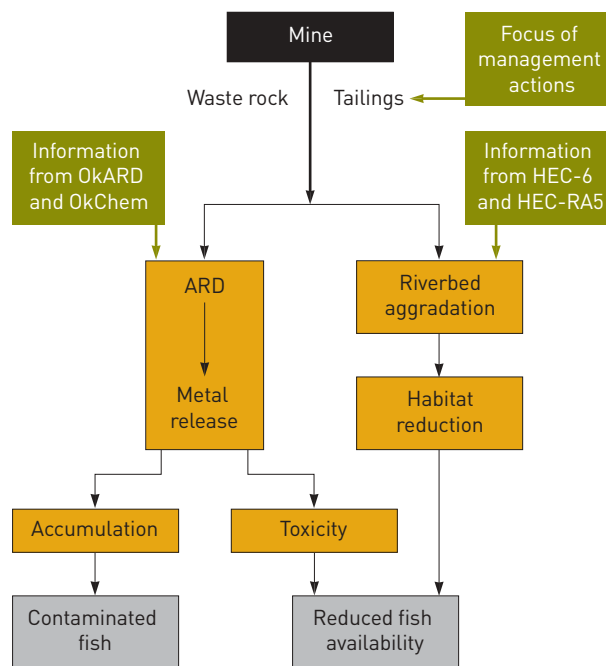


Figure 3. Schematic of a fish BN model developed to assess impacts from a mine.

structure of a model for predicting the impact of riverbed aggradation and water quality, particularly increased copper concentrations produced by acid rock drainage (ARD), on fish abundance and contamination. The BN also integrated information from two sub-models (HEC-6 and OkARD/OkChem) into a single predictive framework.

Building a BN model

A summary of the main steps follows. Further detail can be obtained from Carey et al. (2006), Borsuk et al. (2006) and Pollino et al. (2006a, b).

Structure of a Bayesian network

As discussed earlier, developing a causal structure, with relevant variables and dependencies, is the first step in constructing a BN model. In most cases the conceptual model(s) developed during the Problem Formulation stage will form the basis of the BN structure.

Discretisation of nodes (assigning states)

States or condition of the variables can be categorical, continuous or discrete. In order to represent continuous relationships in a Bayesian network, a continuous variable must be divided or discretised into states. The states of a variable can be numerical ranges (≤ 3 , > 3) or expressions (that can also represent data if appropriate, e.g. acceptable ≤ 3 , unacceptable > 3). If relevant, these states can represent targets, guidelines, existing classifications or percentiles of data.

Specification of prior probabilities

After defining node states, the linkages between nodes need to be described. Parent nodes lead into child nodes, the outcome of child nodes are conditional on how the parent variables combine. This relationship is defined using conditional probability tables (CPTs). In the networks, sub-networks describe physical or chemical processes relevant to the spatial scale specified. The impacts of these on the final outcome node, which often represent a biological/ecological process, are combined in the CPTs. For this reason, BNs are often described as being integrative models. CPTs can be derived via one or a combination of methods:

- direct elicitation of scenarios from experts,
- parameterisation from datasets, and
- equations that describe the relationships between variables.

It should be noted that for BNs, the more complex the interactions the more conditional probabilities there are to specify.

Calculating posterior (predicted) probabilities

Data or new knowledge can be incorporated into a BN and used to calculate posterior probabilities. Data sources can be entered into the network as a series of ‘cases’. Cases can represent data collected during a monitoring exercise or undertaken as part of a research study.

Model evaluation

A range of validation tools can be used for BNs. Evaluation can involve data or technical experts, or both. Quantitative evaluation with data is preferable. Such measures include predictive accuracy and sensitivity analyses.

Predictive accuracy tests are used to determine model error rates, which are quantified using data (although not the same data used for model parameterisation). This method measures the frequency with which the predicted node state (that with the highest probability) is observed, relative to the actual value. Outcomes can be used to identify weaknesses in the model, and where more effort can be targeted in order to improve model accuracy.

Sensitivity analysis is used to identify the key drivers in the model (or importance of the variables) and major knowledge gaps in our understanding. Sensitivity analysis of mathematical models can be used to investigate the uncertainties and inaccuracies in model structure, relationships and outputs, and subsequently identify where priority knowledge and data gaps exist. Thus, based on these results, recommendations for targeted monitoring and research studies can be made.

Knowledge gaps and priority risks

Having established the structure of the model, and the relationships used to drive the model, the key knowledge gaps in our understanding and priority risks can be identified. To do this, sensitivity analysis is used.

Testing management scenarios

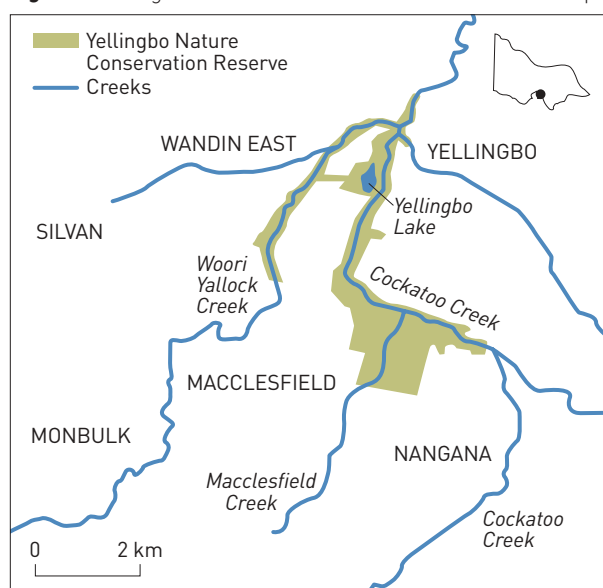
Management scenarios can be tested by entering new information into the network as evidence, directly changing the distribution of probabilities on the node itself.

Case study 1 — Managing *Eucalyptus camphora* wetlands in Woori Yallock catchment

Issue

The Woori Yallock Creek catchment is located in the Yarra River Catchment close to Melbourne, and supports a mix of forestry, intensive horticulture and urban development. The Yellingbo Nature Conservation Reserve, located 50 km east of Melbourne, is an area of significant botanical and zoological significance, is also located within the Woori Yallock catchment. This Reserve contains approximately 285 native flora species and 230 native vertebrate species. Within the Reserve, extensive dieback of the sedge-rich *Eucalyptus camphora* swamp (wetland) community has occurred, as have reductions in the populations of the Helmeted honeyeater (*Lichenostomus melanops cassidix*) and the Leadbeater's possum (*Gymnobelideus leadbeateri*).

Figure 4. Yellingbo Nature Conservation Reserve location map.





Leadbeater's possum (*Gymnobelideus leadbeateri*). Photo © Jean-Paul Ferrero/AUSCAPE.

A stakeholder workshop identified the sedge-rich *Eucalyptus camphora* community within the Yellingbo State Nature Reserve as a priority threatened environmental value (asset). This tree species is only found in the Yellingbo Nature Conservation Reserve, an isolated patch of forest in the Yarra Valley.

The swamp Eucalypt (*E. camphora*) has suffered from dieback since the 1970s, and this is ongoing. The tree community is restricted to the Reserve and the *E. camphora* swamp community is listed as endangered. The reserve has been actively managed since about the 1950s, and attempts to manage the dieback have been in place since the 1990s. However, to date, no management strategy has been effective in slowing dieback in the area.

Two hypotheses have been advanced to explain the dieback, namely that:

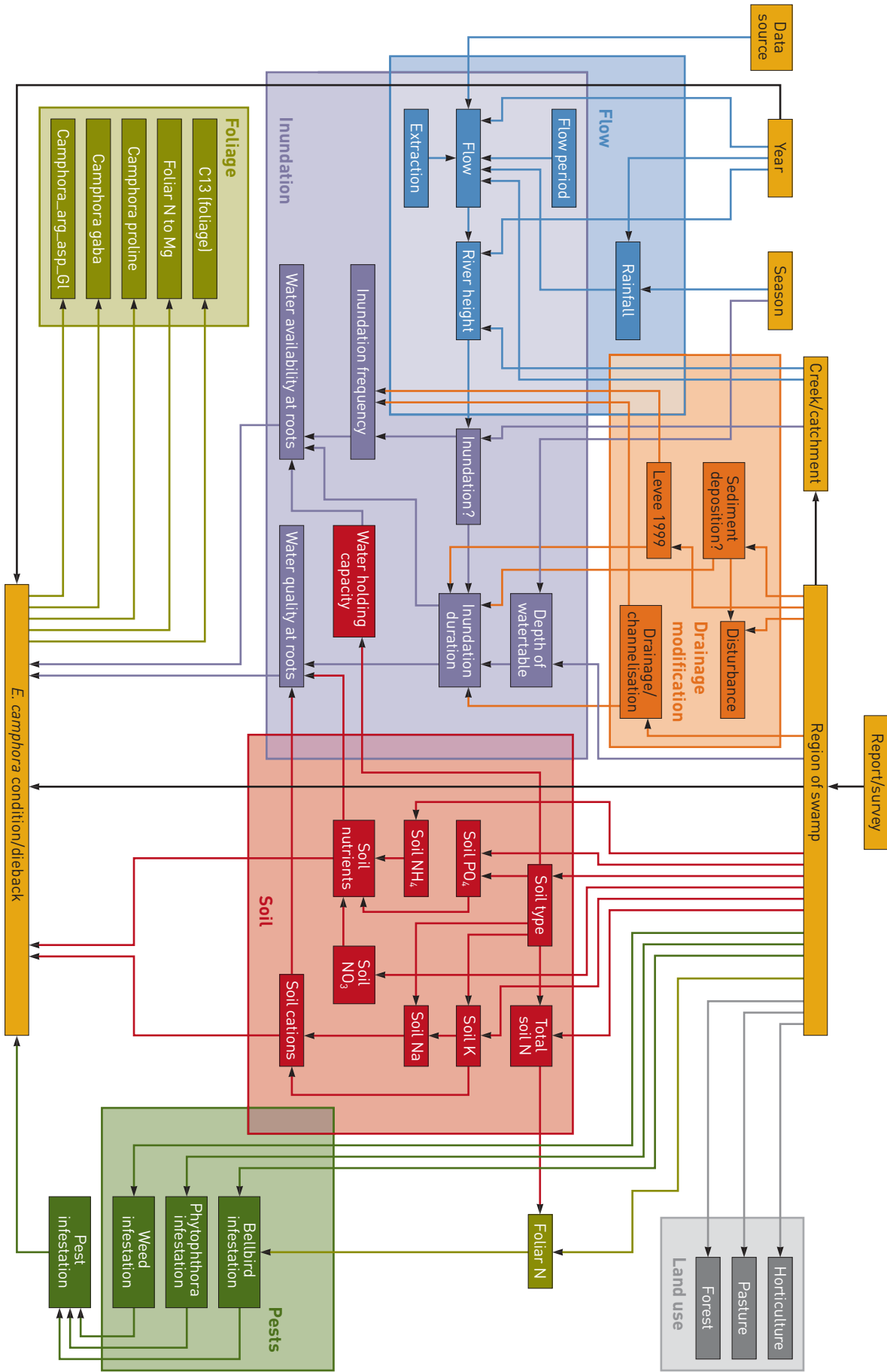
1. the hydrological regime has been altered within the catchment, causing erosion and changed sedimentation patterns,
2. nutrient enrichment (eutrophication) from the surrounding horticulture and livestock has occurred. To a lesser extent the presence of pests may also be having an impact on the health of the trees.

Bayesian network model

A BN model was developed to assist in the management of the endangered eucalypt wetland. The model quantifies the relationships between various threats and hazards and *E. camphora* condition, and was then used to examine the differences between the two hypotheses at left. The hazards and threats were identified in the stakeholder workshops and additional information was gathered from the literature and from further interviews of experts.

From this conceptual model, a BN model (Figure 5) was built and tested (Pollino & White 2005, Pollino et al. 2005). The full graphical Bayesian probability network model linked a number of threats to the probability of *E. camphora* condition or dieback in Yellingbo Nature Conservation Reserve. The main threats were:

Figure 5. Identification of main threats and hazards to the condition of the *Eucalyptus camphora* within the Yellingbo Nature Conservation Reserve. [Note: 'foliage' has a number of variables that provide information on the condition of *E. camphora*.]



- catchment land use (surrogate for contamination),
- flow regime in Woori Yallock Creek (which effects inundation of the wetland),
- modification of the water regime in the wetland due to drainage works,
- soil condition (particularly nutrient status), and
- pests including bellbirds, phytophthora fungus, and weeds.

Application of model to decision-making

The BN model was used in two ways (Pollino & White 2005, Pollino et al. 2005):

1. as a *hypothesis specific model* — to link the findings of each variable to the report/study where that data/information was obtained. The BN variables were explicitly linked to particular reports to determine whether the findings of each report are specific to *E. camphora* or if reports are limited or biased in their focus. Sensitivity analysis was used to test the strength of causal links between the endpoint variable (*E. camphora* dieback) and parent variables.
2. as an *integrated model* — where an integrated assessment of all study findings was undertaken, with information specific to particular reports or surveys removed as an explicit factor in the analyses. In the integrated model, the major variables important to maintaining *E. camphora* condition were again ranked using sensitivity analysis. Assessments looked at influential variables for all creeks combined, and individual creeks in the Reserve.

The most influential variables in determining *E. camphora* condition were found to be soil nutrients, soil cations, pests or disease, and inundation patterns (duration and frequency). The models also showed that *E. camphora* responses are region-specific, which is consistent with advice from experts stakeholders consulted in this study.

The data used to parameterise the BN model was very patchy and much of the data was qualitative. A well-structured monitoring program was recommended to investigate the main knowledge gaps and to iteratively update the BN model (Pollino & White 2005).

Despite its limitations, the *E. camphora* model enabled a complex system to be conceptualised, complex processes to be integrated into a single endpoint, and disparate studies and monitoring data to be aggregated. The model is currently being used by various management agencies to review and modify the management regime for Yellingbo Nature Conservation Reserve.

Case study 2 — Use of a Bayesian network decision tool to manage environmental flows in the Wimmera River

Issue

The Wimmera region is located in north-west Victoria, and covers almost 30,000 km². The dominant land use is agriculture, mainly dryland cropping of cereals, grain legumes and oilseeds. The Wimmera River flows from the Grampians and Pyrenees Ranges in a largely northerly direction, ending up in a set of terminal lakes (Lake Hindmarsh, Lake Albacutya and terminal lakes in Wyperfeld National Park) (see map below).

The lower Wimmera River is of high environmental value and still contains many sections with relatively intact riparian and in-stream vegetation. It includes the Heritage River section from Polkemmet Bridge down to the Wirrengren Plain, incorporating Lake Hindmarsh, a nationally important wetland, and Lake Albacutya, a RAMSAR wetland. However, the lower Wimmera is threatened by reduced environmental flows and poor water quality, in particular increased salinity.



Figure 6. Wimmera River location map.

Stakeholders decided that ‘sustainable populations of a native fish species’ was an important environmental value of the lower Wimmera, and that the viability of the Freshwater catfish (*Tandanus tandanus*) would be an appropriate assessment endpoint for this value. Freshwater catfish typically occupy bottom waters that are prone to water quality problems, which catchment managers hope to address through the appropriate management of environmental flows. So in addition to the importance of the viability of Freshwater catfish populations as a management objective in its own right, it was hoped that Freshwater catfish might also serve as a useful indicator of in-stream conditions, a surrogate for the broader community of aquatic biota.

Bayesian network model

The development and application of the Freshwater catfish BN model is reported in Chee et al. (2005). The spatial boundary of the model encompassed a 62 km reach of the lower Wimmera River, from the junction of the Wimmera and McKenzie Rivers to Big Bend. The model is focussed on environmental conditions

during summer and the model endpoint is the viability of Freshwater catfish populations within the specified river reach.

The full graphical Bayesian probability network model linking summer flow regime to Freshwater catfish populations in the Lower Wimmera River is shown in Figure 7, and was structured according to the main conceptual ideas as follows:

- The viability of Freshwater catfish populations in the study reach (Box A) is determined by the abundance of the breeding population and the recruitment of juvenile and larval catfish. These population characteristics are in turn dependent on the quantity and quality of physical hydraulic habitat for living and spawning, and on stochastic elimination via adverse ‘slug’ events, and in the case of larval catfish, via high flow velocities as well.
- The quantity and quality of physical hydraulic habitat is determined by a combination of flow cross-sectional area and bottom water quality, while the severity of ‘slug’ events depends on bottom water quality and the peak cross-sectional flow velocity experienced over the summer period.

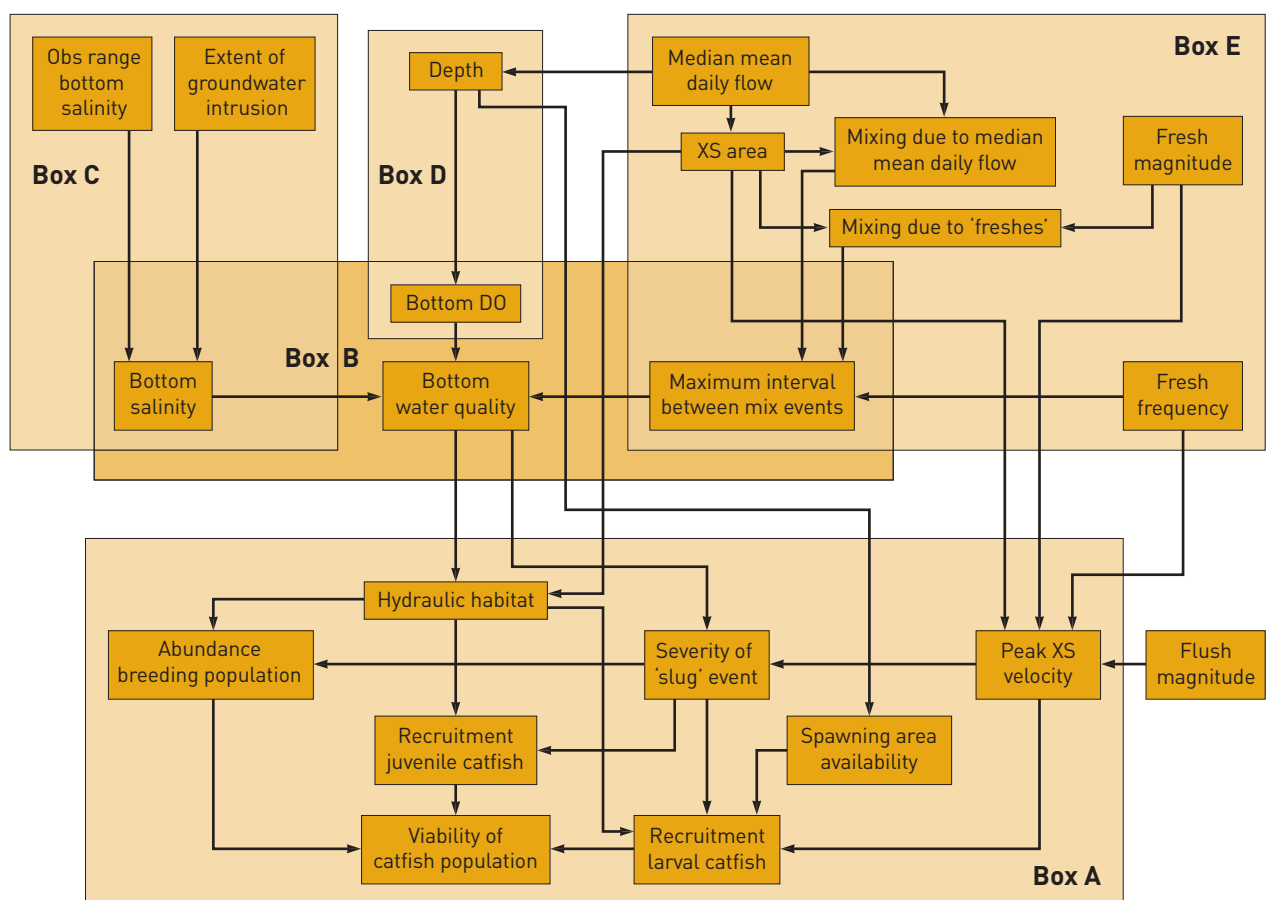


Figure 7. Bayesian network model for Freshwater catfish in the lower Wimmera River.



Freshwater catfish (*Tandanus tandanus*). Photo Gunther Schmida, courtesy MDBC.

- Bottom water quality (Box B) is determined by the interaction between bottom salinity and bottom dissolved oxygen (DO) status, and the maximum interval between mixing events, which are expected to have an ameliorating influence on bottom water quality.
- Bottom salinity (Box C) is predicted from the extent of groundwater intrusion and the observed range of bottom salinity within the focal reach. Bottom DO (Box D) is predicted on the basis of stream depth, which in turn depends on the median mean daily flow (MDF) over summer. The maximum interval between mixing events (Box E) is controlled by the frequency of occurrence of flow velocities sufficient to produce full mixing of unstratified or thermally stratified waters and depends on the pattern of summer flow distribution of median MDF and ‘freshes’.

The main flow components of the summer flow regime that are amenable to management are the cease-to-flow period, minimum flows and ‘freshes’. Minimum flows are represented in the model by the median mean daily flow over the summer period. ‘Freshes’ are defined as small peak flow events that exceed the summer median mean daily flow and several may occur over the summer period. Both ‘fresh’ frequency and magnitude have

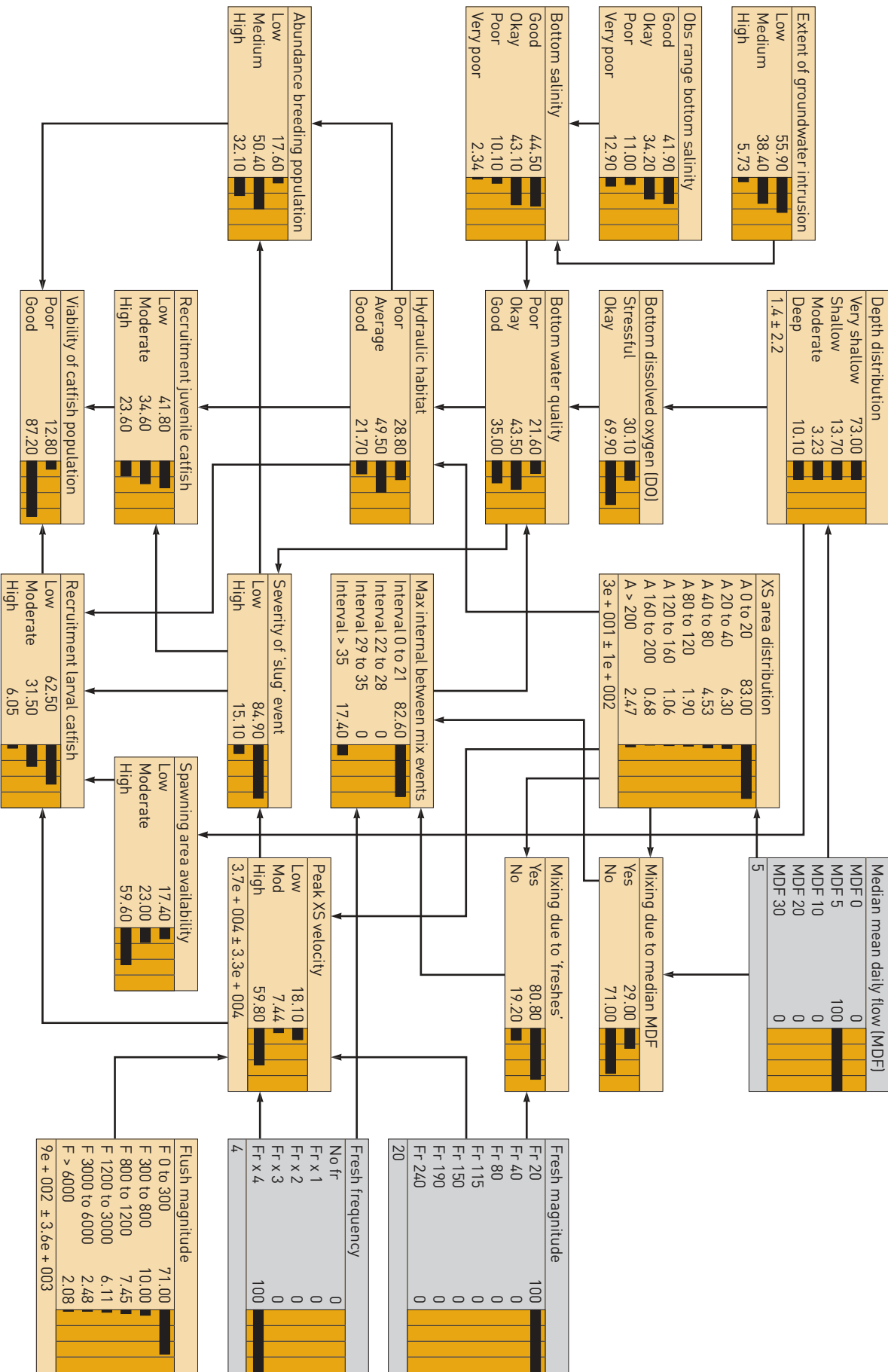
been included in the model. Flush flow refers to the largest flow event experienced over the summer and is considered to be a stochastic event with potentially detrimental impacts on Freshwater catfish.

Application to decision-making

The pattern of allocation of available water over the suite of flow components, comprising cease-to-flow periods, minimum flows and ‘freshes’, constitutes the environmental flow release strategy over summer. Exploration of flow scenarios or strategies using the BN model was carried out by entering findings at the appropriate nodes and examining the resultant changes in probability distributions of the variables of management interest. This allows the model-user to assess the sensitivity of outcomes to different management interventions. Two basic scenarios were used to provide baselines for comparative evaluation of environmental flow strategies:

- a *worst case scenario* (called ‘Dry’) in which the study reach receives no flow at all throughout summer, and
- the *current environmental flow recommendations*, which consisted of a cease-to-flow period of 21 days, median MDF = 5 ML/day and four evenly distributed ‘freshes’ over summer of 20 ML each. The implementation of this strategy in the full BN model is shown in Figure 8.

Figure 8. Full Bayesian network model for Freshwater catfish in the Lower Wimmera River showing the situation if the current summer flow release strategy is implemented (median mean daily flow = 5 ML/day; 'Fresh' frequency = 4 and 'Fresh' magnitude = 20 ML).



Under conditions of low median mean daily flow, the most likely state of cross-sectional area distribution is 0–20 m² with a probability of 0.83 and the most likely states of bottom water quality are ‘Okay’ and ‘Good’ with probabilities of 0.44 and 0.35 respectively (Figure 8). These conditions in turn result in Hydraulic habitat being most likely to be in an ‘Average’ state over summer, with a probability of 0.50. Under the current flow recommendations, the probability that the severity of a ‘slug’ event will be ‘Low’ is 0.85 (Figure 8).

Application as a decision network

The Wimmera BN model was also converted into a decision network by specifying decision nodes for variables under management control and specifying a utility model with which to measure the expected value of different choices.

Initially, the flow variables amenable to management control (i.e. median mean daily flow, ‘Fresh frequency’ and ‘Fresh magnitude’) were converted into decision variables, and the utility was based on the states for ‘Viability of catfish population’. However, this endpoint was not very sensitive, since the adult catfish seem to be able to persist at high densities under extremely poor physico-chemical conditions for extended periods of time. Indeed, the viability of populations in the study reach is likely to be an issue only if there is a combined sequence of adverse events such as a large number of consecutive years (e.g. 10 years) of poor conditions, during which little or no breeding occurs and recruitment over that period is insufficient to offset mortality in the breeding population.

As an alternative, ‘Hydraulic habitat’ and ‘Severity of ‘slug’ event’ were chosen as key indicators of environmental condition. The goal in managing environmental flows over summer was to maximise hydraulic habitat while minimising ‘slug’ events of ‘High’ severity. This goal was translated into a simple utility function formalising both the order and strength of hypothetical preferences for different consequences, and was incorporated into the decision network as a utility node representing quantities to be maximised.

The Netica software was then used to find values for the decision nodes that result in the largest possible expected value for the utility node. As an example, we assumed that operational constraints in a particular summer dictated that only a maximum of three ‘freshes’ and a median mean daily flow of 10 ML/day could be released. When these conditioning values were entered into the network as findings, the utility value was maximised when the ‘Fresh magnitude’ is 115 ML/day.

Monitoring and assessment for ERA

There are important differences between conventional environmental monitoring programs and those required for ecological risk assessments. The main differences are that an ERA monitoring program generally comes after the risk assessment, data collection is repetitive (either over a definite or indefinite time horizon), and there is a feedback loop that links predictions with an assessment of current conditions and trends. As a result of monitoring, the risk analyst and stakeholders should be in a better position to assess:

- the adequacy of the risk assessment and ensuing policy, management, and intervention strategies including any unanticipated consequences,
- the status of the environmental asset relative to baseline conditions and likely trends,
- any gaps, omissions, or errors in the risk models, and
- the likelihood of the emergence of new threats/stressors.

Additionally, ERA monitoring specifically seeks to answer the ‘how well did we do’ question.

Risk assessment is a risky business, and the chances of ‘getting it wrong’ are high. While type I and type II errors are familiar to many researchers and risk analysts, errors of the third type are less well known. ‘Type III errors’ have been used to describe the situation where a problem is so completely misunderstood that the right answer is given to the wrong problem. Type III errors are particularly likely to arise in the context of a risk assessment for previously unobserved events. The risk analyst is invariably required to pass judgment on previously unobserved or rarely observed events, thus plunging us even further into the realm of uncertainty, ambiguity, and indeterminacy — a place where the ‘law of averages’ and conventional statistical inference breaks down.

Nevertheless, a cornerstone of the ERA process is the identification and manipulation of *uncertainty*. Monitoring activities need to inform our models of uncertainty, which in turn allow us to refine our monitoring programs. Issues of spatial-temporal variability, autocorrelation, natural variation and stochasticity all need to be addressed. Finally, we need simple, yet effective tools and devices for designing monitoring systems, collecting and analysing the data and conveying our uncertainty to decision-makers and stakeholders.

Developing an ERA monitoring program

To be successful, ecological monitoring programs must be ecologically relevant, statistically credible, and cost-effective. Carey et al. (2006) provides a set of guidelines

that should assist in developing ERA monitoring program that are robust and fit-for-purpose. The level of complexity of ecological and statistical models needs to be assessed with due regard to the objectives of the task, the desired specificity of the outputs, and the ability to adequately estimate model parameters.

The development of any monitoring program must start with a clear statement of the monitoring objectives, which may include:

Ecological status and trends — Without a measure of ecological status and trends, there is no rational basis for judging the quality of the ERA, nor is it possible to assess the effectiveness of subsequent management actions or identify strengths and weaknesses of the ERA process. The ‘status’ component will typically gather data on an ecosystem unit(s) at a fixed point in time, possibly over a number of spatial scales. To assess trends, the ‘status’ component of the monitoring program will need to be replicated over a number of time periods. Decisions need to be made about the frequency of sampling, the intensity of sampling (do we need to monitoring everything over time or just key variables/indicators?), and the purpose of sampling (are we more interested in an ‘early warning’ system by making general inference about change — increasing/decreasing/ no change or do we need to accurately determine rates of change?). There are many statistical techniques likely to be useful for trend detection.

Assessment of the ERA — this objective is concerned with addressing the ‘how well did we do’ question. Two modes can be distinguished for this analysis: a) a hard assessment in which inference is based on an analysis of key ecological indicators in a typical Before-After-Control-Impact (BACI) type experimental design, and b) a soft assessment that draws inference from an analysis of ‘expert’ and stakeholder opinion sampled on a number of occasions following the completion of the ERA.

Assessment of management effectiveness — this provides a link between the ERA assessment and the management response, which are often not well coupled. This involves undertaking a critical appraisal of the management response with reference to the outcomes of the ERA assessment. At the simplest level, data could be collected and analysed in the form of a simple contingency table (Carey et al. 2006).

Process improvement — involves an analysis of the strengths and weaknesses of the decision-making process, coupling the outcomes of the other assessments to highlight aspects of the ERA-management that require refinement or modification. Clearly, there are limits (and possible sensitivities) associated with any ‘reworking’ of the ERA and/or decision-making process. The key idea

here is that the quality of environmental decision-making can only improve if there is a mechanism to identify gaps and flaws in the process by which decisions are made. While an individual ERA is generally a ‘one-shot’ process, which may result in binary decision-making (e.g. urban development or no urban development) the ERA process is not. It is founded on current best practice, best available models, data and information available at the time, and prevailing socio-political norms. Thus, in order to remain relevant, robust, and credible, the ERA process and tools need constant refining and updating.

Modes of analysis

Carey et al. (2006) provides a short description of a number of statistical modelling approaches and tools that are likely to be useful in analysing monitoring data, including data reduction techniques, statistical analyses, probability models, assessing the suitability of a candidate probability model, decision-making under uncertainty, assessing trends in time, and early-warning monitoring.

Conclusion

Risk-based approaches to management have been used for many years in finance, engineering and medicine. There is also a long history of such approaches being used in fisheries management. However, it is only recently that risk-based approaches have been applied more broadly in the management of natural resources.

There is now available a set of guidelines for applying ERA to the management of natural resources (Hart et al. 2005). This chapter reports on the outputs of research that aimed to extend the current ERA guidelines by further developing quantitative risk-based assessment guidelines that will assist in improving the management of diffuse contaminants in catchments.

In particular, additional information is provided on the development and application of BN models in the area of quantitative risk analysis. Two case studies are reported in which BN models were developed to assist the decision-making processes associated with 1. managing a *Eucalyptus camphora* swampland, and 2. managing environmental flows in the lower Wimmera River.

A section on ERA monitoring and assessment is provided to take account of recent advances in this area. A new framework for data analysis and presentation to assist management is provided. The increased application of ERA to a wide range of natural resource management areas will see innovation and new processes being developed. Thus, it will be important that the current ERA guidelines be regularly updated.

Five reports on the project are available (Hart et al. 2006, Carey et al. 2006, Henderson & Bui 2005, Pollino et al. 2005, Chee et al. 2005, all can be found at www.sci.monash.edu.au/wsc/).

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Chapter 11 — The role of modelling in catchment management

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Summary

- Models are increasingly being used to support catchment management, especially to set targets and to develop management strategies aimed at meeting those targets.
- Careful consideration is needed to identify the appropriate role of modelling in the context of target setting for catchment management.
- The process of model development and implementation needs to be well planned and carried out with consideration of guidelines of good modelling practice to ensure reliability in outputs, transparency in decision-making and continued use and development of modelling capacity.
- Modelling is a potentially key element in the adaptive management cycle and effort needs to be invested in incorporating improved feedback into the cycle, particularly through well targeted monitoring and assessment of the efficacy of remediation measures.

Background

The development of comprehensive contaminant cycle model that could be applied to large catchments, taking account of sediment, nutrients, salt and ecology, and the linking of land, groundwater and rivers is reported in this chapter. Work has focused on clarifying the management requirements of catchment-scale models and, where required, developing new models and providing data for model users.

A survey of model end users was used to identify priorities to be addressed by the project (Newham et al. 2004). The following conclusions were drawn from the survey:

1. catchment-scale models need the capacity to model sediment, nutrient and salt fluxes,
2. models need to be able to simulate the effects of common management interventions, especially for assessing the impacts of land-use change and riparian zone management,
3. contaminant cycle models should be able to simulate the effects of contaminant loadings on ecological functioning and habitat values,
4. contaminant cycle models need to be applicable across a wide range of spatial scales to meet the needs of a variety of end user groups, and
5. results from contaminant cycle models need to be effectively communicated to end users to increase their use as tools in the decision-making process.

To address these priorities, the researchers developed specifications for several new models. These include a riparian particulate model (see the box below); a gross

primary productivity model; a model of time-varying salinity effects on in-stream macroinvertebrates; and an algal biomass model. The riparian particulate model (RPM) was taken through to implementation and is available both as a stand-alone model and as a component model within E2 (a catchment modelling platform developed by the former CRC for Catchment Hydrology, see Argent et al. 2005).

Drawing on this work and experience in other projects which the authors have been involved in, this chapter examines the role of simulation modelling in catchment management and the target setting process.

Target setting

Institutional and policy context

The National Framework for Natural Resource Management Standards and Targets establishes the principles and requirements for natural resource management standards and targets to guide investment. The framework contains outcomes and guidelines for natural resource management on a national scale, as well as protocols for regional target setting, monitoring and reporting. The framework also covers national standards and defines best practice in natural resource management. These apply mainly to legislative, policy, process and institutional systems. The framework sets out consistent national directions and approaches to natural resource planning, target setting, implementation and performance measurement at all levels. (See www.nrm.gov.au/publications/standards for further details.)

Development of the riparian particulate model

Restoration of riparian buffers is widely promoted as a way of improving water quality in catchments. However, it is difficult to measure their performance and thus assess their value against other activities. The riparian particulate model (RPM) was developed to address this deficiency.

The RPM is a simple, conceptual model of particulate trapping in riparian buffer zones that quantifies the capacity of grassed riparian buffers to trap particulates through settling, infiltration and adsorption. The model is sensitive to the effects of consecutive inflows of pollutants to the buffer and variations in buffer characteristics, such as vegetation type, slope and width. It can be applied at both site and catchment scales. The RPM captures some important features of experimental findings, notably vegetation damage during high flows and the recovery of the buffer's trapping capacity between events.

At catchment scales, the RPM has been used to help catchment managers decide where to invest in the establishment and/or protection of riparian buffers. It has been applied in the North Pine catchment of south-east Queensland, where it is proving useful to calibrate particulate phosphorus loadings by allowing larger flow events to contribute higher loads to the stream and reducing the impact of smaller events through particulate phosphorus trapping in the riparian zone. More research is needed to better integrate field-scale experimental and modelling work into the RPM.



The upper reaches of the Campbells River in the Ben Chifley Dam catchment, NSW is one of the catchments used in this study. Photos throughout this chapter are courtesy of the project team.

Target setting by Australian natural resource managers

Regional catchment strategies (RCS) are an important component of catchment management throughout Australia. A RCS is a community led, strategic document which is intended to coordinate and focus a region's natural resource management effort towards priority projects, and sets the framework to guide regional investment (CRCCH 2004a). Regional catchment strategies need to include meaningful, measurable targets and time frames to achieve them, which may range from a few years to several decades. This can be a challenge, as appropriate, accessible data on basic resource condition is often scarce and setting realistic levels for targets is difficult.

Target setting, and reporting progress towards meeting targets is an integral part of virtually all natural resource management strategies nowadays, providing a mechanism for accountability and performance assessment. For example, the RCS may specify targets for reductions in pollutant loadings, e.g. nutrient loads will be reduced by 30% over 4 years. But, what benchmark is the reduction being measured against? ...an average of the last 5 years? or 10 years? How accurately can the

load be measured and so how do we know if the target is being met? If the next 5 years have well above average rainfall and streamflow or a major disturbance, such as a bushfire, which creates high loads regardless of local management, does this mean management has failed because the target was not met? Conversely, a drought may result in targets being met without any management intervention.

In practice, targets need to be set by taking into consideration the availability of existing data, the cost of collecting any new data, the notion of benchmarks, and methods for dealing with effects on the target other than the management actions that the target was meant to assess (CRCCH 2004a).

The target setting process

When resource managers set targets to achieve greater sustainability they must consider the level of uncertainty in the information on which decisions are to be based, potential conflicts and the particular nature of the catchment (or catchments) being considered. In considering all these factors, a process is required that must involve continuing choice for managers and the community (Jakeman et al. 2005).

Environmental management always involves balancing trade-offs between different interest groups, for example, diverting water for irrigation versus providing environmental flows in rivers, or moving to more-intensive agriculture versus the risk of increasing nutrient runoff. The selection of targets may initially be based on a relatively narrow vision but eventually it should be based on broad perceptions of benefits and costs. The selection should also be modified by the quality of existing knowledge and ability to meet those targets. This iterative process of assessment, target setting, implementation and monitoring means that targets may need to be adjusted over time. Importantly, the process of assessment and management must allow for improved knowledge and understanding as well as for new conflicts and issues which arise as old solutions cause new, unforeseen problems (Jakeman et al. 2005).

The role of models in target setting and assessment

Data alone, even if easily accessible, goes only part of the way to helping monitor against targets or better understanding environmental systems. Such systems are constantly changing, due to management actions, weather conditions, changes in population, land use etc. This is where modelling can play an important role.

Models provide a way of filling in the gaps between observations and addressing the likely effects of differences in catchment conditions between measurement periods. A model such as E2 (Argent et al. 2005) or CatchMODS (Newham et al. 2004) is designed to represent the hydrological and water quality response of a system for a given set of climatic, land-use and management conditions. Such models can be used to assess how different management actions or land-use changes would affect water quality and quantity by running a range of scenarios over a specified climatic period. In this way, models can help to set realistic targets by simulating the effects of likely management actions and adoption rates over certain time periods (CRCCH 2004a).

What if?

Models enable the likely effects of management actions and climate scenarios to be simulated so that comparisons can be made between different options 'virtually'. The broad objective of modelling is to increase understanding of the directions and level of change under different options, for example, investigating the water quality impacts of riparian revegetation versus broad-scale land use changes.

As a minimum, environmental models can clarify the potential consequences of decisions. At best, they may also indicate the relative likelihoods and extents of outcomes well enough to support decisions which yield much more sustainable outcomes than at present. Generally, the more complex the system being considered, the greater the role for models to keep account of the interactions (Jakeman et al. 2005), and to identify the dominant processes and likely consequences. Even when the detail and degree of confidence in a model are limited, the contrasts in modelled outcomes may focus attention on issues which might otherwise have been overlooked and may point to important areas that need closer attention.

Integrated scenario models allow users to simulate how impacts of changes in climate and human activities, for example, affect outputs. These models allow catchment communities and managers to answer the 'what if?' questions, such as "what happens to stream-flow if these trees are removed?" "what is the impact downstream if water is removed for irrigation?". This type of modelling has many benefits. It provides a way of investigating and explaining trade-offs; a focus for researchers and stakeholders to work together; a starting point which can be adapted and further developed by stakeholders; and a way of making the management analysis transparent (Jakeman et al. 2005).

Effects of salinity on invertebrates

Modelling has been used to investigate the effects of salinity on invertebrates. Salinity varies in streams due to natural rainfall variations and planned discharges. A new model was developed for predicting the mortality of invertebrates exposed to salinity (see Figure 1). Macroinvertebrates are an important component of stream ecosystems and provide an indicator of ecosystem health. Many freshwater macroinvertebrates are regarded as more salt sensitive than freshwater fish and many freshwater plants. Managers can use this information to

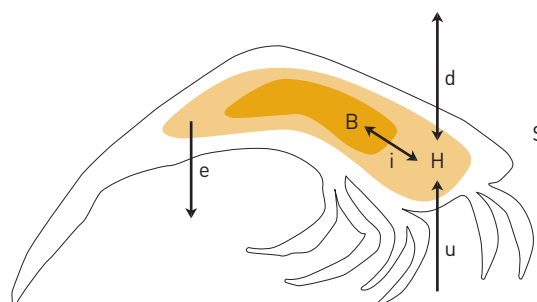


Figure 1. Schematic representation of an aquatic invertebrate. Salinity: S = external, H = free haemolymph and B = bound. Fluxes: d = diffusion, u = uptake, i = storage/release and e = excretion.



The algal biomass model is used to represent algal biomass during periods of steady low flow, such as in this stretch of the Gilmandyke Creek in the Ben Chifley Dam catchment, NSW.

set salinity targets which protect salt-sensitive macro-invertebrates, knowing that such levels will also be safe for other more salt-tolerant organisms (see Chapter 2 for more details). The model was calibrated for two common macroinvertebrates and is providing information about how animals respond to changing salinity levels over time.

Algal biomass model

In another activity, researchers developed a model to represent algal biomass during periods of steady, low flow. The aim was to develop a model that can be linked to existing catchment scale models of flow and contaminant concentration to improve the ability to predict ecosystem response to management (e.g. land use change, riparian restoration). The model predictions compare favourably with a limited dataset from cobble-bed rivers in New Zealand but it has not yet been tested using data from Australian rivers.

A vehicle for communication and participation

Public participation is the direct involvement of catchment stakeholders in the decision-making process (Mostert 2003). Participatory activities give stakeholders a sense of involvement in solving catchment problems solutions, and can also provide local and expert knowledge against which models can be tested and

updated. Involving stakeholders in model development and application can add to the validity of the final product and also create an opportunity for constructive interaction between stakeholders. In fact, one of the most useful outcomes may be the learning experience of researchers and stakeholder groups (Newham et al 2006).

Involving catchment stakeholders in policy decisions makes the decisions more legitimate to those who may be required to implement them, increases the likelihood of implementation, reveal information which would otherwise be unavailable to the policy maker, and allow alternative decisions to be explored. This is particularly important in the early steps in the process of model development and application that involves defining the model purpose, specifying the model context and conceptualising the catchment system, data and other prior knowledge.

Participation may also be useful to improve coordination between organisations and individuals involved in catchment management. Integrated catchment-based approaches to planning and management using collaborative partnerships are likely to be much more successful for water management than a series of local or remote decisions taken without consideration of their larger-scale impacts.



Involving model users and stakeholders in the process of model development leads to valuable learning experiences for all involved. This photo shows managers and researchers in the Ben Chifley Dam catchment discussing water quality models and their outputs.

Although participation can make negotiations more complex because more parties are involved, without public participation support for any decisions may be limited. Participation is essential in helping to address issues which have negative impacts on a section of the community, such as problems caused by upstream water diversion or by pollution. Collective action to improve the local environment, for example ecological restoration of a public wetland, including those which have positive impacts, will also benefit from effective stakeholder participation to help ensure support and implementation of agreed solutions.

For successful participatory activities in support of integrated catchment modelling, consultation should be included from the beginning of the project until the end. A number of different mechanisms need to be developed for groups and individuals to be involved, sufficiently long-time scales available to develop trust, and input invited from a broad range of organisations and individuals. In the work to develop a catchment-scale contaminant cycle model, stakeholders were involved in identifying priority contaminants and the scales of time and coverage important to managers.

The role of models in developing management strategy

The issues involved in catchment management tend to be interrelated, especially when a problem has offsite or downstream effects. Often managers are dealing with multiple, sometimes conflicting, issues, which need to be assessed. Consideration of a broad range of issues helps to clarify their effects, perhaps even changing the perception of the overall severity of the problem (Jakeman et al 2005).

Some of the issues which may be important to catchment stakeholders are:

- increasing the viability of towns and industries, including agriculture and tourism,
- reducing land and river degradation, including salinisation and erosion,
- allocating scarce surface and groundwater between competing uses, including allocation for environmental needs,
- protecting water quality,
- managing pests,
- maintaining or improving biodiversity, both terrestrial and aquatic,
- managing and distributing resources equitably,
- coping with and adapting to impacts of climate change and climate variability, including floods and droughts,
- adhering to government policies, and
- recognising the influence of market volatility including declining trade prices.

Integrated modelling of problems in catchments must cover a range of disciplines, such as hydrology, ecology, agriculture, forestry, economics and politics, and a range of people affected. When these disciplines are brought together, models can provide a powerful tool to help managers answer the 'what if' questions and examine scenarios over different time scales.



Erosion site in the Ben Chifley Dam catchment, NSW. An example of an environment where models may be able to help us to predict restoration trajectories.

Management implications

With the increasing reliance on models to inform and support natural resource management, users need to have some understanding of how good models are developed. From the perspective of a model user, the choice of model style must be made using a “horses for courses” approach. There is no particular style of model that is inherently better for all applications than another. The general maxim is to choose the simplest model that will do the job required.

The basic considerations in choosing the right model for a particular job (CRCCH 2004b) are:

- objectives of the exercise,
- access to data,
- access to expertise, and
- availability of resources (time and money).

When defining the objectives, some of the key questions (see Figure 2) that developers of the model need to consider are: How are the results of the modelling going

to be used? What specific output is needed? i.e. what does the model need to compute, e.g. “daily runoff from catchments ranging in size from 1–100 km²” or “average annual sediment load from streambank erosion”. Where will the model be applied? Who will be interpreting the results and what decisions will they be making?

Whatever the type of modelling problem, certain common steps will be used to develop the model. These steps are largely iterative, involving trial and error (see Figure 3). Best practice in model development involves identifying clearly the clients and objectives of the modelling exercise; documenting the nature (quantity, quality, limitations) of the data used to construct and test the model; providing a strong rationale for the choice of model family and features (encompassing review of alternative approaches); justifying the techniques used to calibrate the model; serious analysis, testing and discussion of model performance, and a statement about model assumptions, accuracy and limitations (Jakeman et al. 2006).

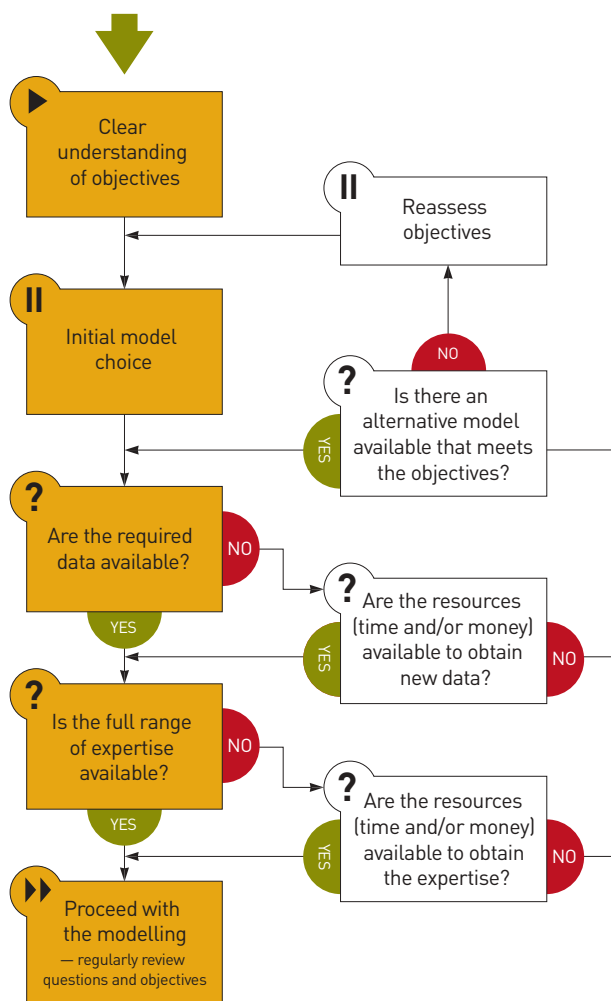


Figure 2. Key questions in choosing a model for practical application. Source: CRCCH 2004.

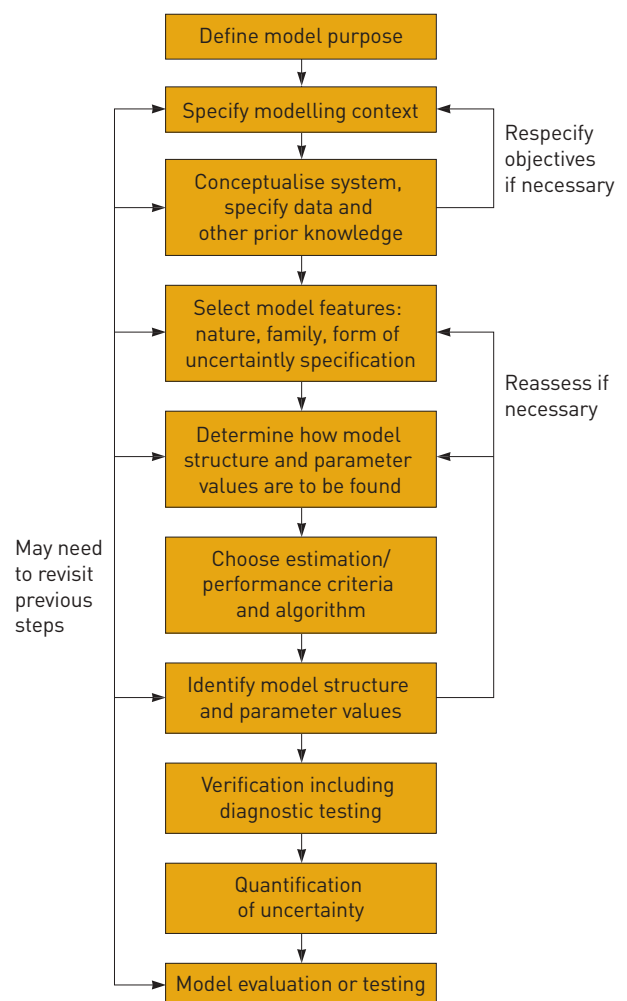


Figure 3. Iterative relationship between model building steps.



A model may be able to simulate a particular type of observation, such as streamflow at a catchment outlet but it does not mean that other predictions made by the model, such as streamflow at internal gauging stations are correct.

Iterative model development, implementation, monitoring and evaluation cycle

There are two basic approaches to building models, the ‘downward’ or top-down approach, and the ‘upward’ or bottom-up approach. The downward approach involves starting with the simplest model that is able to reproduce a set of observations (e.g. to convert rainfall to runoff) and introduce added complexity only when it consistently improves the fit to observations and aligns with processes that are known to occur (CRCCH 2004a). This approach places a high priority on field data and the ability to define model parameters by calibration (i.e. choosing model parameters that provide a “best fit” between observations and simulations). In practice, this approach may result in a model that is too simple to address some problems. For example, if we want to explore the effects of land-use change on runoff water quality, the model must be able to represent the effects of different types of land-use. If there is insufficient data available to derive such effects from observations, the data cannot be included in the model.

The second approach is to represent all the processes which are thought to be important and assume that because our understanding and representation of individual processes are right, the overall model is right. This approach has been more common in catchment modelling but it can lead to models that are too complex and cannot be properly tested. Even if a model is able to simulate a particular type of observation, it does not indicate that other predictions made by the model are correct. For example, a model may give good fits to streamflow data at a catchment outlet, but this does not guarantee that there will be accurate simulation of streamflow at internal gauging stations. In the end, the two approaches should converge to give a model that represents important processes and can be fully tested.

In practice, pragmatic choices have to be made about the appropriate level of model complexity and the consequences of those choices. The level of sophistication of a model should depend on the context — the questions and issues that modelling seeks to address and the knowledge base and resources available for development and application of the model. Complex models simulate more physical processes and so are likely to have more parameters.



This page and opposite: Streamflow gauging site on the Gudgenby River, ACT.

For given model complexities, increasing data availability leads to better predictive performance up to a certain point, after which adding more data does not improve performance. In this case, a more complex model might need to be considered to better exploit the information available. The more common case in catchment modelling is that the model is too complex for the limited data available which means you cannot define an optimum set of parameters with confidence.

Ultimately, the answer to “what model complexity is warranted” depends on the objectives of the modelling exercise and knowledge of the system being modelled. Every time a new (or more complex) process description is included in a model, more parameters are added, each of which must either be calibrated or have a value assigned.

Limitations of models

One of the biggest limitations of modelling is the availability (quantity and quality) of data to drive, calibrate and test the model. All models are a simplification of reality and the key to smart model use is knowing which model to use for a given application and so maximise its usefulness (CRCCH 2004b). Often

there is a trade-off between having a model that can satisfy the desire for detailed spatial and temporal resolution, against the development of an overly complex and unwieldy model with potentially lesser predictive performance.

It is easy for model users or managers using model outputs to be unaware of the limitations, uncertainties, omissions and subjective choices in models, with the risk that they read too much into the model’s outputs and/or predictions. There is also a danger of models being used for purposes different from those intended, making invalid conclusions very likely. The limitations of modelling are affected by:

- the way the problem is formulated,
- the knowledge (data, assumptions and information) at our disposal,
- the style of model,
- the performance criteria used to judge the success of the model, and
- the rigour used to apply the modelling process, including the degree to which we evaluate the model’s applicability and limitations (including considerations of the effects of uncertainty).

Conclusion

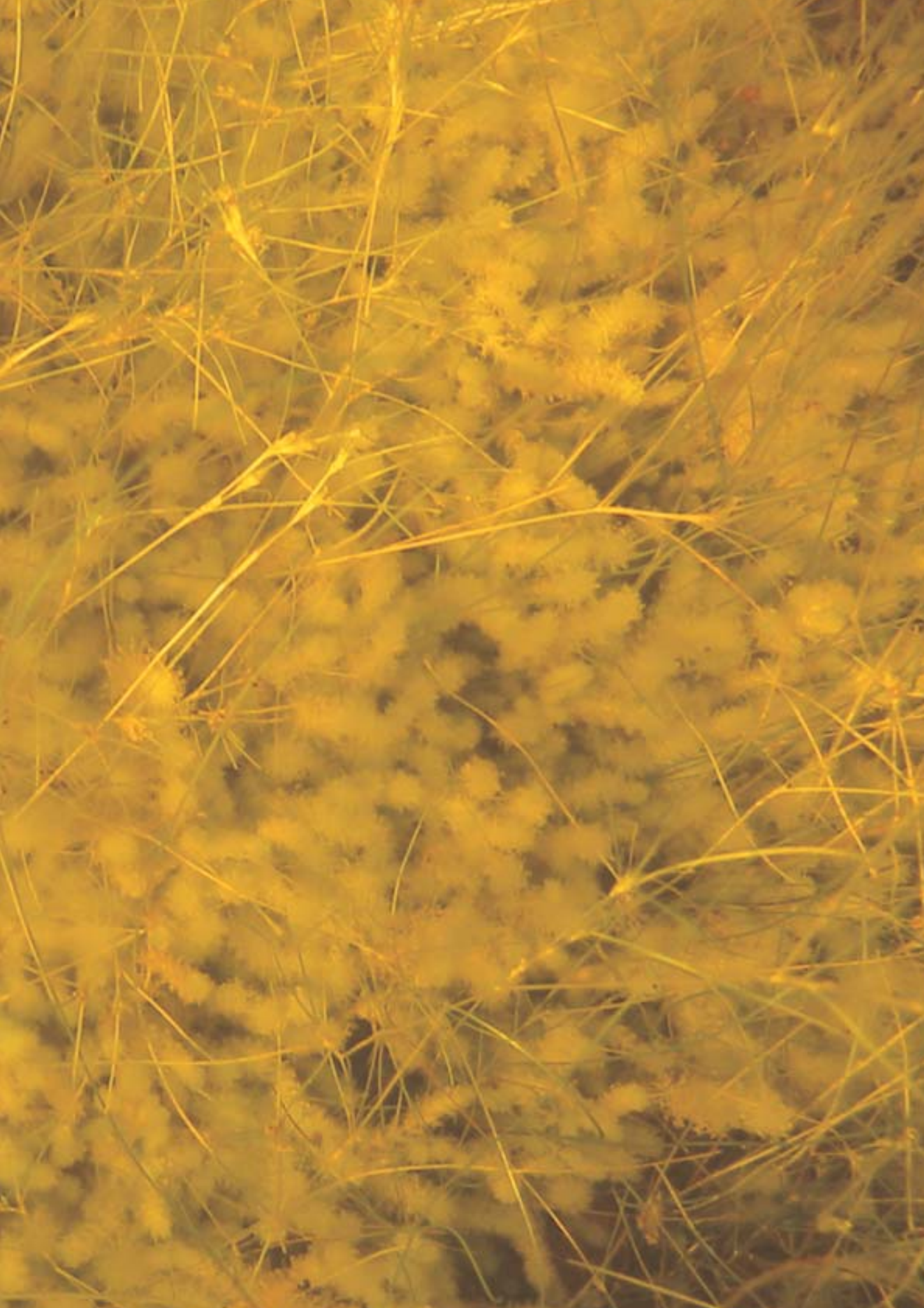
Models are increasingly being developed and applied to support catchment management, including for the process of setting targets and developing management strategies aimed at meeting those targets. Models are potentially useful tools for such purposes but careful consideration is needed to identify the appropriate role of modelling in management decision-making. The process of model development and implementation needs to be well planned and carried out with consideration of guidelines of good modelling practice to ensure reliability in outputs, transparency in decision-making and continued use and development of modelling capacity.

Modelling is a potentially key element in the adaptive management cycle and effort needs to be invested in incorporating improved feedback into the cycle, particularly through well targeted monitoring and research.

Two key areas were identified for future research: firstly, the need to better integrate field-scale experimental and modelling work with catchment-scale models in order to accurately quantify the effects of mitigation measures such as riparian buffers for trapping particulates; and secondly, the area is the need to more closely align measures of 'ecosystem health', which is what managers are often concerned about, with the capability of catchment-scale models.

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River Landscapes

