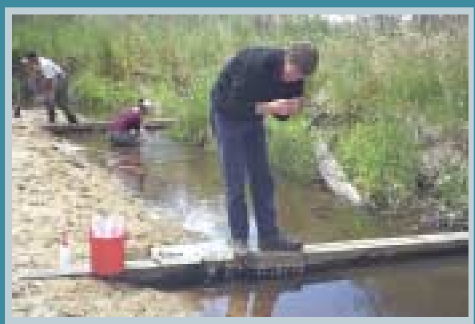
Recent lessons on river rehabilitation in eastern Australia



CRC FOR FRESHWATER ECOLOGY

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Cooperative Research Centre for Freshwater Ecology

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The Cooperative Research Centre for Freshwater Ecology (CRCFE) is a national research centre specialising in river and wetland ecology. The CRCFE provides ecological knowledge to help manage rivers in a sustainable way. The CRCFE was first established in 1993 as part of the Australian Government's Cooperative Research Centres Programme.

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Summary

It is widely recognised that many of Australia's rivers have been adversely affected by intensive human impacts. This has resulted in the implementation of hundreds of river rehabilitation projects over the last two decades, with millions of dollars spent annually. However, rehabilitation projects are rarely evaluated, making it difficult to determine if particular management or rehabilitation actions are successful. Unfortunately, experience both in Australia and overseas suggests that many rehabilitation projects fail to achieve their stated ecological objectives. Halting or reversing the decline in river condition will require long-term commitment and will also require the use of best available information on where and what type of rehabilitation activity is best suited to the circumstances. There is also a need to learn from recent rehabilitation efforts so that insights and lessons can be applied elsewhere and the best use made of available resources and knowledge.

This report describes some of the key findings gained from rehabilitation experiments and research projects conducted on rivers in temperate and arid regions in eastern Australia. The lessons learnt are then considered within the context of the design, implementation and/or evaluation of a river rehabilitation project. A number of valuable technical manuals are already available to help practitioners plan and implement their rehabilitation projects (these are cited in the body of this report). This report seeks to complement such volumes (rather than present another complete 'how-to' manual) by capturing recent lessons to have emerged from the Cooperative Research Centre (CRC) for Freshwater Ecology and associated research, and from the experience of research staff and associated practitioners engaged in river rehabilitation and management in eastern Australia.

The 'boom–bust' nature of many river systems in Australia is well known. Riverine communities of plants and animals have evolved to be dynamic, rather than stable, in space or time. Investigations of riverine community response to disturbance, including succession and dispersal patterns of biota and changes to ecosystem functions such as productivity and respiration, provided valuable insights that can assist river rehabilitation efforts in the future. A key attribute of healthy river systems is their resilience to disturbance. Ultimately, rehabilitation aims to increase resilience in degraded systems by contributing to their capacity to withstand natural and further human-induced disturbances and reorganise while undergoing change so as to retain essentially the same function, structure, and feedbacks as the pre-disturbed state.

A key consideration for those undertaking river rehabilitation is to consider carefully how, in space and time, their projects might be affected by large-scale factors, such as climatic extremes and catchment land-use. *How might these large-scale factors affect the outcomes anticipated from rehabilitation activities?* Experience in many river systems has also highlighted that the drivers of stream condition can be distant from the locality where ecosystem impacts are evident. Most rehabilitation projects conducted in Australia to date have focused on small-scale issues (e.g. reach, or even local site scale activities such as reinstating physical habitat features). A key question must

be '*does the scale of the proposed rehabilitation works match that of the drivers of ecosystem condition?'*.

Guiding principles

The following principles related to the ecology of river systems can help practitioners when planning and undertaking rehabilitation projects:

- Riverine ecosystems are structured hierarchically, with important processes operating at a range of spatial scales, from large regional and catchment scales, to sub-catchment and reach scales, and ultimately down to smaller site and micro-habitat scales.
- Riverine ecosystems can also be highly dynamic and variable in space and time. As such, stream ecosystems in good condition are resilient to periodic natural ecological disturbances (e.g. droughts, floods and fire), which can help drive important physical and biological processes.
- Hydrological connectivity provides strong spatial connections along river networks and between rivers and their floodplains, and plays a key role in ecological processes such as nutrient and energy cycling (spiralling), and the recovery of populations and communities following natural and human-induced disturbance.
- Stream rehabilitation activities are embedded in the hierarchical organisation mentioned above, and should begin with an examination of large-scale factors that might constrain processes acting at smaller spatial scales. Longitudinal processes should also be considered, as the source of degradation can be some distance from locations where ecosystem impacts are evident and rehabilitation is proposed, and loss of connectivity may constrain the biotic response to physical changes in the channel.
- The most effective form of rehabilitation is to prevent degradation of river ecosystems in the first place. Highest priority should go to protecting the remaining high quality river systems (or parts thereof), particularly those that serve as important refugia and are a potential source of colonising organisms.
- Rehabilitation should aim to increase the resilience of river ecosystems to natural (and further human-induced) disturbances so that ecosystems become self-sustaining and capable of responding to large-scale processes such as climate change and the condition of catchments.
- Where possible, rehabilitation efforts should aim to work with natural processes. This means considering rehabilitation at broader scales than is often practised (most rehabilitation work in Australia has been conducted at small scales) and choosing realistic rehabilitation targets. Given the nature of human impacts, it is unlikely that many degraded streams can be returned to their pre-disturbance condition. In such circumstances it can be inappropriate to adopt 'return to natural' as the target for rehabilitation but our best understanding of natural can serve as a benchmark to guide rehabilitation strategies.
- The fragmentation of populations, coupled with sometimes low levels of connectivity (whether because of human interventions such as barriers or naturally poor dispersal abilities), means that many plant and animal species will respond to habitat restoration only very slowly. Isolation may therefore be a major constraint to biotic recovery, and in some cases this

could take years or decades to become apparent. Further, local extinctions may preclude full population recovery.

For rehabilitation to be successful, proper planning and interaction with stakeholders must complement the ecosystem perspective outlined above. Socio-economic considerations will play a large role in determining where and when different rehabilitation measures are applied.

Planning rehabilitation projects

Reviews of rehabilitation project outcomes in Australia and overseas suggest that many projects are only partially successful in achieving their stated objectives, and that success rates in terms of ecological response can be low, even when habitat targets have been met. Some of the reasons rehabilitation projects fail are:

- a lack of clear and agreed rehabilitation objectives,
- a mismatch between the scale of the rehabilitation and the underlying drivers of degradation,
- the isolation of newly installed habitat from source populations,
- mismatch in the roles of different stakeholders (e.g. project implementation may be the responsibility of state or local stakeholders, but project funding may depend on other sources over a shorter timeframe),
- lack of monitoring, evaluation and review,
- failure to complete adaptive management cycles and adjust rehabilitation objectives and evaluation activites.

While this reports focuses mainly on rehabilitation from an ecosystem perspective, it is important to note that the management measures adopted will depend to a large degree on socio-economic considerations. Sources of funding, the role and responsibilities of various stakeholders, and a willingness by stakeholders to be involved (or not) are some of the factors that can affect the rehabilitation measures adopted, and where and when they are applied. Spending time with stakeholders who might be involved or affected by a project is a wise investment, as is a communication strategy that keeps stakeholders informed and committed. One of the biggest lessons learnt from recent rehabilitation projects is that *you are likely to be in this for the long haul, so proper planning is essential if you want to get the most from your efforts*.

The main steps in the design and implementation of rehabilitation projects are, generally:

- identify the processes leading to the degradation or decline in the condition of the river system;
- define opportunities and constraints to the reversal or amelioration of the drivers of degradation or decline;
- determine realistic goals for re-establishing species, assemblages of species, ecosystem processes and functional ecosystems;
- recognise potential ecological limitations and socio-economic and cultural barriers to implementation;
- consider the information needs of a monitoring and evaluation program;

- develop conceptual models and testable hypotheses so that rehabilitation projects can be used as management experiments that improve the knowledge base for the future;
- develop easily observable measures of ecological response that can be interpreted as 'success';
- develop practical techniques for implementing the rehabilitation goals at a scale commensurate with the drivers of the problem and the spatial scale at which the problem is apparent;
- document and communicate these techniques for inclusion in planning and management strategies;
- monitor key variables, and assess progress relative to the agreed rehabilitation goals; and adjust procedures if necessary.

Planning must account, as far as is possible, for past, present and future conditions. An understanding of the history of the catchment as well as the drivers of current condition is required if the planned rehabilitation measures are to match the nature and scale of the drivers of river condition. Experience has shown that time spent on problem definition, collecting pre-intervention data to allow performance evaluation, and securing the involvement of local stakeholders, is a very wise investment.

Identifying priorities for rehabilitation can be a difficult task. If rehabilitation is required to overcome the effect of a discrete disturbance (e.g. source of erosion or contamination, or an in-stream barrier to movement), then the priority for rehabilitation can be clear. If rehabilitation is to meet the needs of a single species or community assemblage, then their biological needs set the restoration agenda. However, most commonly, rehabilitation is attempted in river systems that are affected by multiple stressors, often acting at different spatial and temporal scales.

What should be the priorities for rehabilitation for a river system affected by multiple stressors? Priorities will often be a trade-off between ecological, social and economic considerations and there are a number of tools stakeholders can use (e.g. SMART (specific, measurable, achievable, realistic and time-bound) principles, multi-criteria analysis, matrices of stressors and environment, social and economic impacts, priorities from regional catchment strategies, adaptive management and assessment modelling and decision tools). It is generally agreed that the highest priority for management and rehabilitation should go the protection of high value, least-disturbed riverine and floodplain assets. These systems are often those that are representative of the best available ecological condition, can be hotspots of biodiversity or productivity, or serve as refugia that can supply colonising organisms that disperse to newly rehabilitated or available areas, and play a critical role in maintaining the resilience of river systems and their ability to recover from disturbance.

This report proposes a general framework that can be applied when considering rehabilitation priorities. The framework is consistent with the adaptive management approach for resource management, and includes steps related to condition assessment, identification of constraints on rehabilitation, goal and objective setting, implementing the planned rehabilitation activities, and evaluating outcomes. Management priorities will be influenced by social and economic considerations, in addition to the ecosystem considerations that are the basis of the framework. It is also important to consider whether it is feasible to address historical legacies and present disturbances that operate at large scales (e.g. catchment) with rehabilitation actions undertaken at smaller scales (sub-catchment, or reach).

Common rehabilitation techniques

The framework emphasises the need for a holistic view of river rehabilitation at large scales (e.g. catchment) and, in some instances, from river source to sea. This catchment-scale perspective helps us understand the key drivers of ecosystem condition, develop realistic rehabilitation objectives and set priorities for action. Often, a number (mix) of actions will be required, at a range of scales, if rivers are to be rehabilitated successfully. This report presents some of the insights gained on the effectiveness of rehabilitation measures such as:

- meeting the environmental-water requirements of rivers,
- reintroducing flow-related habitat for fish and invertebrates,
- reintroducing wood (e.g. resnagging) and other physical habitat features,
- rehabilitating riparian revegetation,
- introducing riffle habitat to streams in urban areas,
- managing sediment delivery and dynamics, both in-stream and in the riparian or catchment zone,
- restocking species and tracking recovery.

While all such rehabilitation measures can be successful in achieving ecosystem outcomes in appropriate circumstances, an important finding has been that large-scale factors (e.g. prevailing climatic or catchment conditions) can limit or reverse the gains achieved by short-term or localised rehabilitation activities.

Evaluating rehabilitation projects

The ecological outcomes of river rehabilitation in Australia are rarely evaluated; for example because of a lack of resources, because there are no clear rehabilitation objectives, or because roles and responsibilities of stakeholders are unclear. The end result is that we have forgone numerous opportunities to learn from experience and so inform future management within an adaptive management framework. Our investment in river rehabilitation will be maximised when projects include performance evaluation and the sharing of results, whether the projects achieve their stated objectives or not.

Limited resources and the large number and wide geographic spread of rehabilitation projects make it impossible to measure the physical or ecological outcomes of every project. A wise use of available resources will be to monitor and evaluate a few, well-designed and resourced experiments to generate learning that may be applied to other similar systems. Dedicated, large-scale rehabilitation experiments conducted within an adaptive management framework may offer the best combination of achieving rehabilitation objectives and learning, whether or not the stated objectives are achieved. Practitioners should always seek to implement monitoring and evaluation programs when:

- rehabilitation is to be attempted in a unique setting,
- a new rehabilitation technique is to be trialled,
- the objectives of the rehabilitation include the protection of endangered species,
- the project provides an opportunity to showcase river rehabilitation,
- there is a risk that the trajectory of recovery is different to that desired,
- major new targets have been set, for example ecosystem processes rather than population processes.

Given the rarity of large-scale ecosystem rehabilitation projects in Australia, it will be important to take advantage of opportunities for evaluating their outcomes wherever possible.

1. River rehabilitation as a management tool in Australia

Many of Australia's rivers have been adversely affected by intensive human impacts, such as catchment clearance, pollution and the physical modification of rivers and their floodplains (Commonwealth of Australia 2002a,b). Widespread recognition of these issues at all levels of society has seen an explosion in the number of river rehabilitation projects conducted in Australia over the last two decades, with millions of dollars spent annually on activities such as bed and bank stabilisation works, riparian revegetation, mitigation of pollution, and management of endangered aquatic species. However, most stream restoration projects are unmonitored and thus success is hard to judge (e.g. Bernhardt et al. 2005). Unfortunately, experience both in Australia and overseas suggests that many rehabilitation projects fail to achieve their stated or implicit ecological objectives (e.g. Smokorowksi et al. 1998; Lake 2001a; Bond and Lake 2003a; Pretty et al. 2003). Rehabilitation projects need to be implemented at appropriate scales or spatial extents and there must be an ongoing commitment to monitoring and evaluation if the decline in the condition of our rivers is to be halted and reversed (Lake 2005 in press). Such a commitment will also require the use of best available information and multiple lines of evidence on where and what type of rehabilitation activity is best suited to the circumstances. There is also a need to learn from recent rehabilitation efforts so that insights and lessons can be applied elsewhere and the best use made of available resources and knowledge.

A number of valuable technical manuals are available to help practitioners plan and implement their rehabilitation projects (e.g.; Lovett and Price 1999; Rutherfurd et al. 1999, 2000; Price and Lovett 1999; Koehn et al. 2001).

This report does not seek to provide another complete 'how-to' manual, but aims to capture recent lessons to have emerged from the Cooperative Research Centre for Freshwater Ecology research and the experience of research staff and associated practitioners engaged in river rehabilitation and management in eastern Australia.

This report describes some of the key findings gained from rehabilitation experiments and research projects conducted on rivers in temperate and arid regions in eastern Australia. The lessons learnt are then considered within the context of the design, implementation or evaluation of a river rehabilitation project. Some key areas of research that will assist practitioners in the future are also presented.

Note that this report uses the term 'rehabilitation' rather than the term 'restoration'. Rehabilitation refers to the reinstatement of features of the stream ecosystem (structural or functional) that may have been impaired or lost, rather than a complete return to 'natural' or pre-disturbance conditions, as could be implied by the term 'restoration' (e.g. Rutherfurd et al. 1999).

Flow manipulation is a key river management tool, and the delivery of environmental flows based on flow–ecology relationships represents a large investment in river rehabilitation in Australia. Examination of flow–ecology relationships has been a major activity of the CRC for Freshwater Ecology

(CRCFE) and findings are being reported in detail elsewhere. Our focus in this report is predominantly on the reintroduction or management of physical habitat to meet the needs of riverine biota and enhance important ecosystem processes. However, summaries of relevant flow–ecology relationships are also included, particularly as they relate to rehabilitation of flow-related habitat.

1.1. Showcase rehabilitation and research projects

A number of recent projects, conducted in both arid and temperate climates and in rural and urban settings, have contributed significantly to the body of scientific knowledge (restoration ecology) that underpins stream rehabilitation in Australia (Rae et al. 2004). Here we present a brief overview of the aims and major findings of each of these projects.

1.1.1. 'Dryland rivers — their refugia and biodiversity'

Rivers in arid areas (dryland rivers), such as those in the northern regions of the Murray-Darling Basin and in the Lake Eyre Basin, are renowned for their 'boom and bust' ecological response to episodic floods that can extend over large floodplains. However, for much of the time, dryland rivers exist as a network of ephemeral channels and turbid waterholes. The larger of these waterholes are often the only permanent habitat for aquatic biota in the landscape during the frequent and extended periods of low or no flow. Many of these river–floodplain systems are under increasing pressure from water resource development, land degradation from overgrazing and cropping, and invasions of alien plants and animals.

This project examined the importance of waterholes as refugia for aquatic organisms in anastomosing river catchments, such as those of Cooper Creek in the Lake Eyre Basin (e.g. Bunn et al. 2003; Arthington et al. 2005). The relationship between biodiversity and the physical attributes of individual waterholes was studied, as was the spatial and temporal pattern of connection in the landscape. Important considerations were how the freshwater organisms and species in a region are distributed relative to each other, at various scales ranging from waterhole to landscape, within and between catchments. What ecosystem and population level processes enable the organisms to survive in these waterbodies? The insights gained will help with predicting the likely impacts of water resource development, as well as changed floodplain and riparian management, on biodiversity and ecosystem function in dryland river refugia. This information can then be used to identify key environmental flow and land management criteria to restore dryland rivers where altered flow regimes and changed land management have affected connectivity and other key biophysical processes.

1.1.2. 'Connectivity and dispersal of aquatic biota and fragmentation of populations'

A key assumption of most river rehabilitation projects is that aquatic organisms will return to make use of any re-created or reinstated aquatic or riparian habitats. However, successful recolonisation of rehabilitated streams will depend not only on the availability of habitat suitable for aquatic organisms to survive and reproduce, but also (and among other things) on the ability of organisms to reach the new habitat via dispersal. This project examined the mechanisms and levels of dispersal across a range of taxa, habitats and rivers in eastern Australia to gain insights that can enhance recolonisation of disturbed or rehabilitated sites. For example, can we predict the effects of barriers (e.g. in-stream barriers such as weirs, or fragmentation of habitat) on dispersal of different aquatic taxa, and can we predict the suite of species likely to recolonise restored sites from their life histories and dispersal characteristics (e.g. number and type of propagules, migration propensity)?

While species of freshwater fish and invertebrates may occur widely across a region, each species may be subdivided into smaller populations by channel branching and natural barriers. Behavioural constraints and the time available for travel (dispersal) within a life-cycle may also be factors causing natural fragmentation of freshwater populations. This project assessed patterns of connectivity and fragmentation among populations of freshwater fish and invertebrates in different landscape settings and at different geographic scales in the short and long term. Two innovative techniques were employed in the project. First, genetic techniques were used to determine historical and contemporary patterns of connectivity between populations. Secondly, the chemistry of otoliths (a structure in fish ears) was used to examine patterns of movement within the life-time of individual fish with a view to assessing the immediate consequences of existing barriers to dispersal. This work showed that populations of aquatic species can be naturally fragmented across the landscape. This has implications for rehabilitation projects that rely on the dispersal of organisms to achieve recovery from disturbance (i.e. recovery may take much longer than is often anticipated).

1.1.3. ‘Rehabilitation in degraded rural streams — the Granite Creeks’

Slugs of sand occupy large sections in a number of ‘chain-of-ponds’ streams draining the Strathbogie Ranges in central Victoria (collectively known as the Granite Creeks) and they also occur in many streams throughout eastern Australia (from North Queensland to north-eastern Tasmania) and in south-western WA. Large-scale deposition of sand, forming sand-slugs, can occur when sediment from damaged catchments accumulates in the transition zone where streams with high gradient and stream power change to low gradient streams (Davis and Finlayson 2000). This form of degradation can adversely affect the geomorphology and ecology of streams (Rutherfurd 1996, 2000).

Investigations in the Granite Creeks evaluated the reintroduction of wood — a widely used restoration technique — as a means of increasing streambed complexity and providing structural habitat for stream biota including algae, invertebrates and fish. Hydraulic, geomorphological and ecological attributes were monitored to assess changes in habitat complexity, local species diversity and abundance, and ecosystem processes such as production, respiration, organic matter retention and nutrient cycling. This project was conducted within an extended period of drought, providing an opportunity to study factors that contribute to the resilience of stream biota to naturally occurring, large-scale perturbations (Bond and Lake 2005a,b). Drought was found to be a large-scale driver of ecosystem condition which can confound or reverse the gains expected with small-scale rehabilitation measures. In this project, the impact of drought curtailed the gains made with the reintroduction of wood (e.g. formation of scour pools, response of fish) as physical habitat for fish and invertebrates.

1.1.4. 'River rehabilitation through resnagging — resnagging the River Murray'

Recent investigations of fish biology and ecology have highlighted the dependence of native fish species such as Murray cod, trout cod and golden perch on wood (snags) as structural in-stream habitat. A pilot study of snag characteristics, distribution and abundance in the alluvial River Murray provided details of the preferred habitats used by Murray cod and trout cod.

This information provided the basis for reintroducing snags to test whether the target native fish would use the newly available habitat. The experiment was undertaken on the River Murray between Lake Hume and Tocumwal (Nicol et al. 2002), in areas where Murray cod, trout cod and other snag-dependent native fish are present. This information helped to:

- determine the effectiveness of differing snag positions in reconstructing habitats for particular species (especially the nationally threatened trout cod),
- determine the longevity of occupancy at reconstructed habitat,
- determine the cost/benefit of reconstructing habitat through resnagging, and
- assist the recovery process for the species examined.

1.1.5. 'In-stream ecological issues and riparian revegetation in south-east Queensland'

Riparian vegetation shades stream channels and can buffer aquatic ecosystems from temperature extremes. In this project, the relationship between shading and temperature was examined in order to predict how the extent of cover and length of rehabilitation would reduce temperature impacts on fish and other stream organisms. This information could contribute to cost-effective guidelines for the design and monitoring of these landscape 'experiments', and identify some of the factors that may limit the re-colonisation of organisms and the initiation of basic ecological processes.

The effectiveness of reintroducing in-stream or riparian habitat for aquatic animals can be constrained if other factors, such as the ability of organisms to move within and between streams, prevent recolonisation. Our current ability to manage large river systems is hampered by a limited understanding of basic ecological processes. The applicability of ecological models of river systems, all of which have been developed overseas, to large rivers in Australia is largely untested and has major implications for management (especially of riparian and floodplain regions). The project found that in-stream primary production was the main driver of aquatic food webs in many arid river systems. This project also examined the extent to which riparian vegetation directly influences ecosystem processes (and the overall 'health') of large rivers, and produced a detailed conceptual model of riparian influences on streams, especially their fish faunas.

1.1.6. *'Riparian revegetation in the Murray-Darling Basin'*

The revegetation of riparian areas is one of the most common stream rehabilitation measures conducted in Australia (Rutherfurd et al. 2000). However, the effectiveness of this rehabilitation method in achieving ecological outcomes, especially in streams, has been little studied. This project (Stewardson et al. 2004; Anderson et al. 2005) is a dedicated experiment designed to improve our understanding of riparian revegetation as a form of habitat reconstruction and its role in ecosystem processes such as sediment and nutrient interception, shading and organic matter inputs to streams. While the full outcomes of this long-term trial will take some time to emerge, implementing the trials highlighted some of the common problems related to experimental and monitoring design, and emphasised the importance of considering the scale of degradation and of the response required within the planning process (Anderson et al. 2005).

1.1.7. *'Physical habitat assessment in urban streams'*

A common practice in river rehabilitation is to increase habitat diversity by re-introducing structural elements. This project in urban areas of Melbourne examined the response of aquatic macroinvertebrates to the introduction of pool–riffle sequences in channelised streams. In 1996, riffles were added to six degraded, channelised streams in the eastern suburbs of Melbourne. Invertebrates in those streams (and three other control streams) were monitored before and after the addition of riffles.

There was little change to invertebrate communities in the year following construction, suggesting that habitat availability may not be limiting recolonisation. The streams were revisited up to five years after the riffle addition to assess any changes in macroinvertebrate community composition. A continuing lack of change in community composition was considered to be strong evidence that catchment-scale factors, such as stormwater pollution, limit in-stream community development.

1.1.8. *'Stillwater and flow patches in the Broken River'*

Riverine habitat can be considered as an array of patches that are largely formed through spatially and temporally variable geomorphic and hydrological processes, with flow perhaps the overriding force in the structure and function of these patches (Bunn and Arthington 2002). Biota vary widely in their use of different habitat patch 'types', and there may also be ontogenetic shifts in patch use depending on the life history stage (e.g. larvae vs adults). In this study, physical structures were placed in low gradient sections of the alluvial Broken River in northern Victoria, to alter the distribution of 'slackwater' habitats, which are thought to be areas of high secondary production and important rearing habitats for larval fish. Flow was directed into 'slackwater' environments to create 'flowing' patches, and was also directed away from edge areas to create slackwater patches. A range of biotic and abiotic variables was measured to examine the response of aquatic organisms to altered hydraulic conditions (B. Gawne pers.comm.; Humphries et al. 2005). The project demonstrated that flow-related habitat patches can be reintroduced to streams where activities such as flow regulation have reduced or simplified the habitat available to aquatic organisms.

2. Insights on the ecological principles underpinning ecosystem rehabilitation

The 'boom–bust' nature of river systems in many parts of Australia highlights that riverine communities of plants and animals are likely to be dynamic, rather than stable in space or time (Lake 1995). Investigations of riverine community response to disturbance, including succession and dispersal patterns of biota and changes to ecosystem functions such as productivity and respiration, can provide knowledge to assist river rehabilitation efforts. Palmer et al. (1997) identified a number of ways in which investigations of restoration ecology can assist with river rehabilitation efforts, among them to answer questions such as:

- what is the role of natural disturbance in maintaining, enhancing or reducing species diversity or ecosystem processes?
- at what scale do we need to focus on species diversity and how does this relate to restoration of ecosystem function?
- is reintroduction of habitat sufficient to re-establish species and functions?
- what are appropriate end-points for rehabilitation?
- what are the benefits and limitations of using species composition or biodiversity measures as end-points in rehabilitation? Is it better to focus on measures of community function such as trophic structure?

The CRCFE set up a research program to explore some of these questions and build a knowledge base on which practitioners can draw when designing rehabilitation projects (Figure 1).

This chapter summarises some key ecological insights of relevance to rehabilitation practitioners, which have emerged from recent research on rivers in eastern Australia. The projects include investigations of

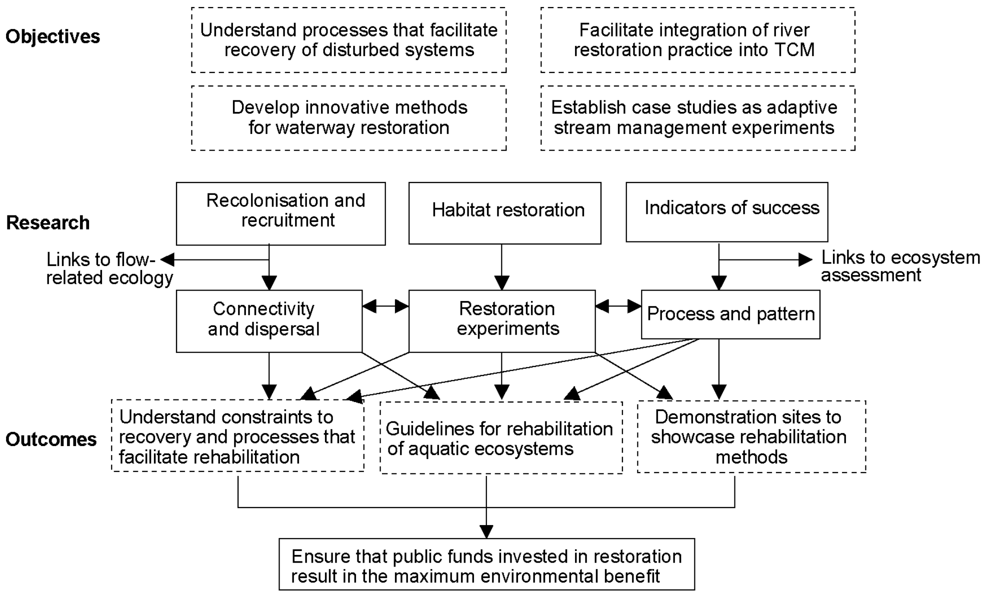

Figure 1. Summary of CRCFE river rehabilitation research. TCM = total catchment management.

fundamental river–floodplain ecological processes at a range of scales, as well as dedicated investigations of some common rehabilitation measures.

2.1. Insights on the scale of environmental processes and implications for ecosystem rehabilitation

A major lesson to have emerged from recent research on river systems in Australia is the importance of scale (in space and time) when considering ecosystem structure, function and condition. For example, Davis and Finlayson (2000) identified three issues that had to be addressed when considering the rehabilitation of the Granite Creeks in Victoria: (i) control of further delivery of sand from upper catchment areas; (ii) whether sand slugs are mobile in lower reaches; and (iii) how to improve in-stream habitat in reaches with sand slugs. The fact that the sand originated (decades ago) from upper catchment areas (first and second order streams) while rehabilitation was proposed for lower reaches (third and fourth order streams) highlighted the potential for a mismatch in the scale and location of rehabilitation if the broader catchment setting were not considered. If the delivery of sand from the upper reaches persisted, then there would be an increased risk that rehabilitation efforts in sand slug areas would fail. It may be very difficult to rehabilitate river systems where degrading processes are ongoing. Davis and Finlayson (2000) contended that the sand slugs, specifically the one in Creightons Creek, were moving very slowly, if at all, and suggested that the addition of large wood to the sand slugs would aid habitat restoration. Thus, the addition of structures made from river red gum railway sleepers to the sand slugs of two creeks, Castle and Creightons, was the restoration intervention. The intervention was set up as an experimental trial to assess the efficacy of using such structures at the large scale of an entire sand slug.

The following sections describe some of the major lessons learnt about large-scale ecosystem processes and the implications for river rehabilitation practitioners.

2.1.1. Some insights on the ecological effects of drought

Much of eastern Australia has experienced an extended period of drought in recent years. While this has posed great difficulty for those managing rivers and conducting river rehabilitation experiments in affected areas, it has also provided a timely reminder of and new insights into the effects of large-scale ecosystem processes that must be considered by rehabilitation practitioners.

Drought is a natural extreme of the hydrograph, with flooding at the other extreme. The nature of a drought can vary. For example, droughts may be a regular, seasonal occurrence, or they may be less frequent and less predictable supra-seasonal events such as those occurring in El Niño years (Lake 2003). In ecological terms, supra-seasonal drought can be considered as a ‘ramp’ disturbance (Box 1) (Lake 2000), where the severity of the disturbance increases as the drought progresses. Droughts can affect stream hydrology in many ways, for example by causing sequential drying in downstream, headwater or middle reaches and thus affecting hydraulic connectivity (longitudinal, lateral and vertical) (Lake 2003).

Box 1. Ecological disturbance

Ecological disturbance is the application of a force, agent or process that affects the biota of a river system. Disturbance has three main forms (Lake 2000) (Figure 2): (a) *pulses* are short-term events with a peak in intensity, (b) *presses* are long-term events of constant intensity and (c) *ramps* are long-term events that change in intensity over time. Ramp disturbances such as drought can result in press responses in biota and ecosystem processes when hydrological, geomorphic and water quality thresholds are reached (see Box 3).

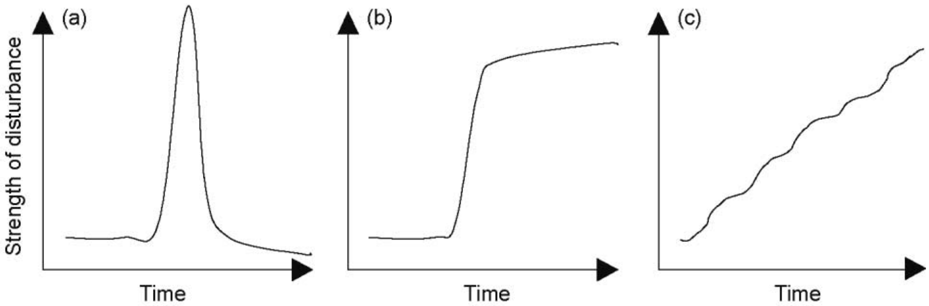

Figure 2. Three types of disturbance: (a) pulse, (b) press, (c) ramp (from Lake 2000; Downes *et al*. 2002)

In addition to the effects of drought discussed in this section, other natural ecological disturbances of note include floods, fire and sediment deposition. For example, floods can greatly reduce biotic abundances and species richness by mechanisms such as dislodgement and drift, poor water quality (e.g. hypoxia), and disturbance of habitat.

The 2003 bushfires that affected south-eastern Australia were a reminder of the potential impact of fire and associated processes such as sediment deposition in waterways. Many streams in fire-affected areas of north-eastern Victoria (EPA Victoria 2004) were adversely affected by the fires (results were based on rapid biological assessment protocols that use macroinvertebrates as river health indicators). The effects were predominantly due to sediment deposition following the fires, which resulted in poor water quality (e.g. elevated nutrient concentrations and turbidity, lowered dissolved oxygen concentration) and the smothering of in-stream habitat. Similar effects were also recorded in fire-affected streams of the Australian Capitol Territory (OCE 2004). For some streams, these effects are expected to be short-lived (i.e. a pulse disturbance), but for many streams the legacy effects of sand-slugs are expected to continue for many years (i.e. serve as a press disturbance).

The impact of natural disturbances can be exacerbated by human activities. For example, water resource management can prolong the effect of drought or flooding. Refugia from floods, such as behind logs or underbanks, may be greatly depleted by channelisation, levee construction, rockwalling and snag removal.

Our management of water resources and regulated rivers can exacerbate the effects of drought (e.g. decreased flow below large dams) and can also result in 'anti-drought' conditions (the elimination of low-flow conditions by the continued release of water for irrigation and other consumptive uses), which can also adversely affect stream biota (McMahon and Finlayson 2003). While often thought of as rare events, drought and anti-drought are now more common in rivers subjected to flow regulation and water extraction. A comparison of conditions in the Murray and Ovens rivers over the summer of 2003 (Gawne et al. 2004) highlighted the potential effects of drought and anti-drought in the southern Murray-Darling Basin. The unregulated Ovens River experienced a period of zero flow, increased temperature and salinity, and very low dissolved oxygen concentration (DO) due to high rates of decomposition. Conditions in the regulated River Murray were far more benign. The delivery of water for irrigation and other uses meant that flow in the Murray was maintained and changes to water quality were relatively minor when compared with that of the Ovens River. The long-term implications of drought and anti-drought for attributes such as river function and species interactions were not clear from this short-term study.

A drought can cause marked changes to biological community structure and ecological processes in response to severe changes to hydrological and geomorphological conditions (Humphries and Baldwin 2003; Lake 2003). Droughts can have many direct and indirect effects on stream ecosystems. Direct effects include loss of water, loss of habitat and disconnection of river channel and floodplain habitats along with the closure of dispersal routes used by aquatic organisms. Indirect effects can include a reduction in water quality, alterations to the supply and availability of food resources and the cycling of nutrients, and changes in the strength and structure of interspecific interactions between aquatic plants and animals (Boulton 2003; Dahm et al. 2003; Lake 2003).

Water quality decreases due to increased temperature, accumulation of organic matter and sediments and decreased DO. Decreased dissolved organic carbon (DOC), nitrogen and phosphorus inputs can result in limitation of microbial processes. This favours autotrophic production over heterotrophic production and increases the risk of algal blooms if there is little shading of the river. Water quality is likely to decline naturally, so management should aim not to exacerbate this problem further (e.g. control livestock access to waterholes). While some native fish species can tolerate low DO conditions (D. McMaster, Monash University, pers. comm.), rapid fluctuations in DO and other water quality attributes can add to the stress already faced by aquatic organisms.

As droughts proceed, mobile animals can become concentrated in features such as pools and billabongs as streams dry and habitat availability is reduced. This can have marked effects on the density and size- or age-structure of populations, on community composition and diversity, and on ecosystem processes. For example, work in the Cooper Creek system (Arthington et al. 2005) highlighted the rapid decline in fish abundance that often occurs during the drying season of rivers in arid areas. River–floodplain connectivity decreased, waterholes became isolated, water volumes declined due to evaporation, and small-scale habitat structures such as scour pools and anabranches became exposed. Fish abundance declined by 93% overall as the system dried and there was a loss of available habitat. The remaining waterholes (acting as refugia) then become hotspots of production and aquatic biodiversity in the landscape. Such hotspots are important sources of recolonising organisms and the loss of these hotspots can therefore have profound and lasting effects on the biota inhabiting the river system.

Implications for river rehabilitation

Drought is a large-scale disturbance that can affect river condition and the effectiveness of rehabilitation in a number of ways. For example, droughts can exacerbate ecosystem problems associated with water stress (e.g. the magnitude, duration and extent of ecological disturbance due to water diversions may all be increased during drought). The gains from rehabilitation measures that rely on a permanent supply of water may be lost when drought causes flow to decrease or stop and the streambed dries. For example, Bond and Lake (2005b) found that the introduction of wood structures promoted the formation of pools by scouring in the sand-affected sections of the Granite Creeks (as hypothesised). Fish quickly responded by inhabiting this new pool habitat. However, prolonged drought resulted in drying of the sand-bed sections. The fish persisted longer than at other ‘control’ sites (where there were no scour pools), but were ultimately lost as the stream dried and pools filled in with sand. This highlighted the need to consider the role and condition of existing refugia (both for drought and flood conditions) and their associated biota in rehabilitation experiments (Bond and Lake 2005a,b).

Many organisms can resist drought by seeking refuge in features such as pools, waterholes and billabongs. These refugia contribute to ecosystem resilience by maintaining the capacity of biota to recover from drought via dispersal and recolonisation. For river systems affected by large-scale disturbance such as drought, maintaining or rehabilitating refugia can be as important as rehabilitating residential habitat, as this will play a large part in maintaining or improving the resilience of river systems and their ability to recover from disturbance (Lake 2005; Lake et al. 2005).

Recovery by biota from seasonal or supra-seasonal drought varies. Recovery from seasonal droughts can follow predictable sequences (resistance, recovery and therefore strong resilience). Recovery from supra-seasonal drought varies from case to case (less predictability means less chance of life history adaptations) and may be marked by dense populations of transient species and the depletion of biota that normally occur. Fish, invertebrate and plant populations appear to recover rapidly from seasonal drought (Humphries and Baldwin 2003). However, innovative genetic approaches (see discussion below on genetic distribution of freshwater biota) suggest that droughts can have prolonged effects, resulting in population bottlenecks and altered courses of evolution (e.g. Douglas et al. 2003). Drought can therefore alter patterns of habitat availability, and dispersal and recolonisation by riverine organisms. This may result in a rehabilitation project failing to achieve its stated ecological objectives, as the response to the intervention follows an unexpected or undesirable path.

Long-term climate trends suggest that droughts in eastern Australia are likely to be more frequent in the future (Pittock 2003). The probable occurrence of drought should therefore be included in future river management plans — it is important to remember that large-scale disturbances such as drought may not always be evident during the planning stages of a rehabilitation project. Rehabilitation strategies for streams must include identification of important refugia and provision for drought refugia with the aim of maintaining or improving ecosystem resilience to drought. Some streams are now locked into permanent drought due to the presence of dams and diversions — this needs to be acknowledged and addressed as part of environmental flow allocations and stream rehabilitation activities.

2.1.2. Flow-related habitat connection

Implementing environmental flow regimes as a river protection or enhancement method is an important form of river rehabilitation in Australia (e.g. Boon et al. 2003; MDBC 2004). Ideas such as the natural flow paradigm (Poff et al. 1997) emphasise the critical role of the natural flow regime in providing or maintaining the energy sources, water quality, physical habitat and biotic interactions that sustain river systems. Bunn and Arthington (2002) proposed four principles to describe how the flow regime governs aquatic biodiversity, which can be used to guide river management:

1. Flow is a major determinant of physical habitat in streams, which in turn is a major determinant of biotic composition.
2. Aquatic species have developed life history strategies in direct response to natural flow regimes.
3. Maintenance of natural patterns of longitudinal and lateral connectivity is essential to the viability of populations of many riverine species.
4. The invasion and success of exotic and introduced species in rivers is facilitated by the alteration of flow regimes.

The flow regime plays a critical role in shaping the availability and quality of habitat for riverine plants and animals both in the river channel and on the floodplain. Attributes of the natural flow regime, such as flooding, bankfull discharge and flow pulses, control geomorphic processes and material transport contributing to habitat diversity in the river channel. Attributes such as flooding, flow pulses and low flows control factors such as habitat availability and quality, and can provide life-cycle cues for riverine biota (e.g. Poff et al. 1997; Arthington and Zalucki 1998; Bunn and Arthington 2002; Arthington and Pusey 2003).

Floodplains can be considered as riparian ecotones (transition areas between aquatic and terrestrial systems) that form dynamic habitat patches (mosaics) in the landscape. The episodic nature of flooding and the persistence of droughts have meant that much of the ecological research on Australian rivers has been conducted during low flow conditions. However, recent floods (2000 and 2004) in the Lake Eyre Basin have provided opportunities to examine ecological responses to the inundation of floodplain areas. Arthington et al. (2005) found that 11 of the 12 native species known to exist in the upper Cooper catchment used the floodplain, in some instances travelling tens of kilometres away from the nearest major channel. Most species were represented by both adults and juveniles, and in some cases by larvae (S. Bunn and S. Balcombe, Griffith University, pers. comm.). Opportunistic use of the floodplain is likely to happen for a number of reasons, including feeding and growth, dispersal and spawning. In the dry season, when fish are restricted to isolated waterholes, they tend to have simple diets that include benthic algae and zooplankton (Bunn et al. 2003). When they move onto the inundated floodplain, the fish have an increased range of feeding opportunities, and most species are found to have a more varied diet (Balcombe et al. 2005). Fish diets are dominated by food that originates in the water, both in the waterholes and on the floodplain, and there is little consumption of terrestrial organisms. The latter is somewhat surprising during flood events, given the access to a vast array of terrestrial invertebrates that cannot escape the floodwaters. As the waters recede from the floodplain, fish abundance and diversity in the waterholes can be extremely high — the result of spawning on the floodplain and/or increased survivorship of juveniles associated with the high resource base on the floodplain (Arthington et al. 2005). Fish abundance and diversity decline as water levels fall and competition for food resources and predation intensifies (both from within the waterhole and from birds and other terrestrial predators).

Although the flood pulse concept (Junk et al. 1989) predicted that fish would use floodplains, it is not always the case. Sampling of fish larvae and juveniles in the unregulated Ovens River floodplain in 1999 (non-flood year) and 2000 (flood year) recorded a total of 11 species, five of them introduced (King et al. 2003a). The only species that increased in larval abundance on the floodplain in response to flooding was the introduced carp (*Cyprinus carpio*). Other species used floodplain habitat such as billabongs and anabranches, in both flood and non-flood years, suggesting that they were not reliant on flooding or temporary floodplain habitat. These results led King et al. (2003a) to propose a new model of floodplain fish recruitment that emphasised the coupling (coincidence) of temperature and flooding (timing and duration) with life-history traits (e.g. spawning time) (Figure 3).

Successful recruitment of target species will also depend on dispersal patterns and the availability of habitat appropriate to critical life-cycle stages. Many native fish species are known to spawn during warm weather and low-flow conditions to take advantage of food resources and benign hydraulic habitat (still waters such as those found in littoral areas, backwaters and anabranches) favoured by larval and juvenile fish (Humphries et al. 1999, 2002, 2005). Some native fish species also have particular hydraulic habitat preferences. For example, large bodied fish such as Murray cod and golden perch were found to prefer flowing creek habitat in the vicinity of the Lindsay River (an anabranch of the River Murray near Mildura), while smaller bodied species such as gudgeons and Australian smelt preferred shallow ponded and weir-pool habitat (Meredith et al. 2002). Many of the native fish species recorded in the Broken River, Victoria, were found to make use of different in-channel habitats during different life stages (King 2004a), taking refuge from high water velocities and predators.

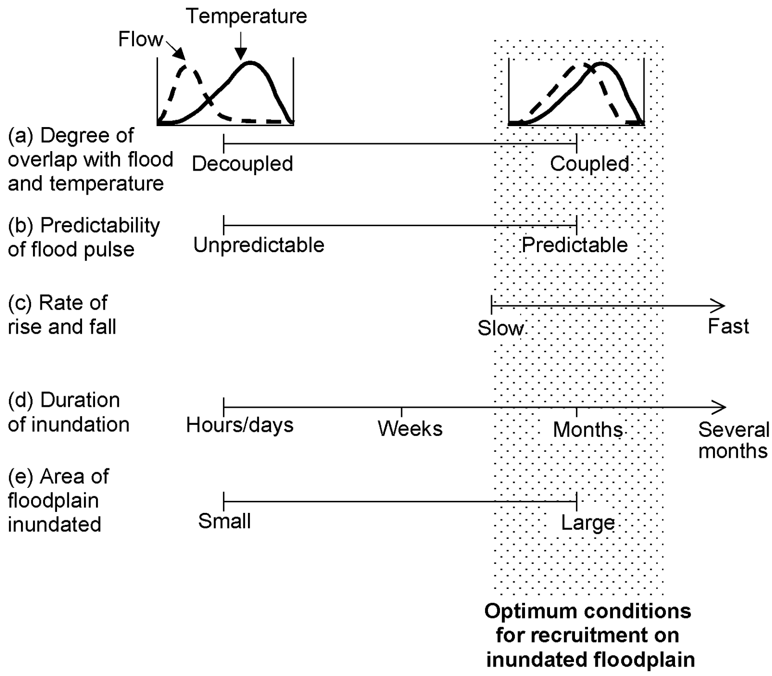

Figure 3. Proposed optimum environmental conditions for use of the inundated floodplain for fish recruitment (from King et al. 2003a)

Activities such as water resource and floodplain development can reduce the frequency and duration of lateral and longitudinal connection. They can alter the pattern of habitat availability, and important processes such as food and energy transfer between river channels and their floodplains which drive processes such as production and respiration and affect community structure (Robertson et al. 1999; Thoms 2003; Arthington et al. 2005). For example, Scholz and Gawne (2004) noted that alteration of the hydrological cycle of ephemeral deflation lakes in arid areas can be detrimental to ecosystem productivity and diversity of arid zone landscapes. As noted above, King (2004a) found that many native freshwater fish use different habitat at different life stages (e.g. as larvae and juveniles), taking advantage of epibenthic food resources that are evenly distributed across different still-water habitats such as backwaters, littoral zones, pools and runs (King 2004b). Thus, reducing the frequency and duration of connectivity has the potential to limit the availability of food resources at critical life stages of native fish.

The extent and duration of hydrological, geomorphological and biological interactions mean that an inter-disciplinary approach to research and management is required when developing floodplain management or rehabilitation strategies. For flow-stressed rivers, the best outcomes from improved river management across large spatial scales are only likely to be achieved when environmental flows are integrated with other complementary forms of river rehabilitation and management.

Implications for rehabilitation

Water resource development and associated river regulation and water diversions have contributed to a decline in the condition of many river systems across Australia (Arthington and Pusey 2003). Manipulation of the flow regime to achieve ecological outcomes (i.e. to meet environmental water requirements) has become a prominent rehabilitation measure in Australia.

Key attributes of the natural flow regime, such as low flows, flow pulses, bankfull discharge and floodplain flows, play critical roles in maintaining connection between river–floodplain habitats, in supporting various life-history stages of riverine plants and animals, and contributing to ecosystem processes (e.g. production, respiration and nutrient dynamics) essential to the functioning of rivers. Most environmental watering plans acknowledge the importance of the natural hydrological regime and seek to reinstate the volume, frequency or duration of ecologically important flow attributes that have been affected by water management (e.g. Arthington & Zalucki 1998; Arthington et al. 2000; Cottingham et al. 2002; DNRE 2002a; Stewardson and Cottingham 2001; Arthington et al. 2004).

The flow regime may be manipulated by returning water to flow-stressed rivers in the form of environmental flows, or by more ecologically sensitive water management practices and, potentially, by installing physical structures to increase the diversity of available hydraulic habitat (e.g. slack-water habitat favoured by juvenile fish and fish larvae — see section 4.2). Most environmental water plans also recommend complementary habitat protection or enhancement works, such as passage past barriers to migration (e.g. installation of fishways or the removal of barriers such as dams and weirs), water quality improvements, riparian and floodplain revegetation, control of stock access to waterways, and erosion control (e.g. Arthington et al. 2000; Jones et al. 2001; Cottingham et al. 2003b).

2.1.3. The influence of urbanization

Urbanization and its associated stormwater runoff is increasingly recognised as a threat to freshwater biodiversity (Grimm et al. 2000; Groffman et al. 2003; Walsh 2004) because it is associated with increased hydrological disturbance, habitat loss and an increased delivery of pollutants to streams. The effects of urbanization are not just a ‘big city’ problem, as many regional centres across Australia are located next to major waterways. Around the world, streams affected by urbanization have shown similar patterns of altered flow regime, altered channel form and declining water quality. Streams in urban areas typically are less able to process nutrients, and have greater in-stream plant growth and fewer animal species than streams with undeveloped catchments (Walsh et al. 2005b).

The nature of urbanization effects means that the location of river rehabilitation efforts can be quite distant from the stream location where impacts are evident[1]. Investigations of urban and peri-urban streams near Melbourne have highlighted the role of hydraulically efficient stormwater drainage as a key degrader of stream condition (Hatt et al. 2004; Walsh 2004). ‘Effective imperviousness’ (the proportion of a catchment covered by impervious surfaces that are connected directly to streams by efficient

[1] This can also be the case in rural streams, for example in situations where bank erosion in upper catchment areas delivers sediment that is deposited in downstream areas.

stormwater pipes) has been proposed as a major causal link between urbanization and poor stream condition in urban areas (Booth and Jackson 1997; Walsh et al. 2004). For example, urbanization is an important cause of eutrophication in waters draining urban areas. The direct connection of impervious surfaces to streams by stormwater pipes is hypothesised to be the main determinant of algal biomass in eutrophic urban streams through its effect on the supply of phosphorus (Taylor et al. 2004). Such findings suggest that stormwater management approaches that reduce drainage connection (e.g. via low-impact or water-sensitive urban design) are a necessary component of any rehabilitation project on streams affected by urbanization (Ladson 2004; Walsh 2004; Walsh et al. 2005a,b).

Implications for rehabilitation

The direct connection between impervious areas and streams via stormwater pipes poses great challenges for rehabilitation practitioners dealing with streams in urban areas. As land use change and altered catchment processes are effectively connected directly and efficiently to the stream, some of the more traditional approaches to river rehabilitation are unlikely to make significant and long-term contributions to improved stream condition. For example, riparian revegetation is a common rehabilitation measure, potentially providing benefits related to shading and temperature control (e.g. Davies et al. 2004), the processing of nutrients and input of energy sources that support foodwebs (e.g. Naiman and Décamps 1997), the supply of wood as structural habitat for biota such as fish and invertebrates (Nicol et al. 2002; Koehn et al. 2004a), and the interception of sediments and other contaminants (Naiman and Décamps 1997). However, the presence of stormwater drains means that runoff can bypass riparian areas and contribute to increased stream incision and widening, altered water tables and regular delivery of pollutants. These impacts can reduce or totally negate any benefits that might otherwise result from an improved condition and functioning of riparian zones.

Minimising the effects of drainage connection is best achieved by trying to increase infiltration 'at source' and reduce the amount of overland flow from small to medium-size rainfall events. This intervention will require the widespread adoption of new urban and drainage designs that manage stormwater runoff as near to source as possible, using a treatment train to identify the stormwater control measures best suited to the position in the catchment and the ultimate stream rehabilitation or protection objectives (Walsh et al. 2004). A review of existing objectives for stormwater management may also be required if they are based on target contaminant loads for downstream receiving waters (e.g. rivers, lakes or embayments). In some circumstances new objectives based on minimising the frequency of overland flow may be required for the protection or rehabilitation of streams in urban areas. Perhaps the most important lesson from this example is the fact that the most effective interventions for improving stream health will occur at some distance from the stream — targeting the source of the problem, rather than trying to mitigate the downstream impacts.

2.1.4. Spatial and temporal availability of physical habitat for stream biota

Recent research has emphasised the need to consider that habitat features are important for riverine biota, at particular scales. Potentially important habitat associations can be missed if assessments are conducted at inappropriate temporal or spatial scales (Crook et al. 2001; Lake 2001b; Lake et al. 2002; Bond and Lake 2003a,b; Arthington et al. 2005), and that in turn increases the risk that rehabilitation activities are misplaced.

In a study of the Cooper Creek system, Arthington et al. (2005) found that fish community structure and abundance was not fully revealed by investigations at small scales only (e.g. habitat structure within individual waterholes). They found that as well as the availability of structural elements such as scour holes, bars, boulders and wood within waterholes, understanding large-scale connection between the wider floodplain and features such as anabranches and waterholes was also important. Thus, a hierarchical, multi-scale approach that included the size of the floodplain around the waterhole, connectivity of waterholes, the size of the waterhole and the distribution and type of habitat patches within individual waterholes was necessary to explain patterns in fish assemblage structure. Additionally, in dryland river systems such as Cooper Creek, fish community structure changes as the floodplain dries after flooding and waterholes become isolated, adding a strong temporal component to the model. Arthington et al. (2005) found that changes to fish community structure were related primarily to habitat structure and habitat loss, rather than to factors such as water chemistry. The interaction between flow and habitat was considered to be crucial to the dynamics of fish populations during and after flooding and to the persistence of fish assemblages through the dry season.

Differences in the strength of associations between fish community structure and habitat at different scales have also been recorded in systems such as the Broken River and the Granite Creeks system in northern Victoria. For example, Crook et al. (2001) found that native fish such as golden perch were associated with wood habitat *only* in deeper river pools of the Broken River during the day. They did not take advantage of wood in shallower runs, so that failure to include both runs *and* pools would lead to incorrect conclusions regarding the role of wood as habitat. Small-bodied native fish in the Granite Creeks were also found to take advantage of habitat (vegetation or wood) in deeper pools. These findings are consistent with the fact that fish often prefer (or are more abundant within) deep-water habitats (i.e. greater than 1–2 m depth), possibly to avoid predation (Pusey et al. 2004). From a restoration perspective, they also suggest that augmenting habitat availability could lead to increased fish abundance, in the absence of other environmental constraints (Bond and Lake 2003a,b). Investigations of wood habitat in the River Murray also found associations between large-bodied native fish and wood habitat in the deeper water of meander bends (Nicol et al. 2002), where erosion processes lead to natural accumulations of structural wood habitat (Koehn et al. 2004a). Individual fish species also show preferences for particular flow velocities, substrates and cover (Crook et al. 2001; Meredith et al. 2002; Nicol et al. 2002; Pusey et al. 2004), and this should be considered in the design phase of timber-reintroduction projects.

Implications for rehabilitation

Patterns of habitat use by biota such as fish cannot be separated from issues of scale. Spatial scales of habitat selection vary between species and over time (e.g. dry–wet season, day–night). Smaller scale features (e.g. microhabitats such as wood) are nested within larger scale features (e.g. pools, waterholes, anabranches) and biota such as fish can respond to habitat patchiness over a range of scales. It is important to consider the scale-dependence of biota–habitat associations when developing rehabilitation projects, and ask *'what are the scales of habitat use relevant for the target species*?' Considering how habitat is arranged in the landscape can also help to identify where rehabilitation might seek to assist the dispersal of organisms, for example by providing 'stepping stones' of habitat that can be colonised (see also Box 4 on the factors to consider when reintroducing wood habitat).

2.2. Distribution of genetic diversity in freshwater biota — lessons on dispersal and recovery from disturbance

Successful rehabilitation requires not only suitable habitat for organisms to survive and reproduce, but also the ability for organisms to disperse and colonise new areas.

Molecular tools that explore genetic variation within populations have been used to examine dispersal patterns across broad geographic areas. Aquatic invertebrates with mobile adult life stages are often widespread (low genetic variation between populations), showing little limitation on dispersal (Baker et al. 2003; Hughes et al. 2003a,b; Hughes et al. 2004). However, fully aquatic animals often show marked differences in genetic patterns, suggesting restricted dispersal in many areas. For example, Hughes and Hillyer (2003) found that dispersal of the freshwater crayfish *Cherax destructor* was widespread within individual catchments of western Queensland but that contemporary dispersal across drainage boundaries did not occur. Another study (Hughes et al. 2004) found that there was limited dispersal of four cryptic species of a freshwater mussel genus (*Velesunio* spp.) between waterholes in individual drainages in western Queensland and little evidence of dispersal across drainage boundaries. Carini and Hughes (2004) expected to find the freshwater prawn *Macrobrachium australiense* to be widely distributed across western Queensland given the low topography, vast episodic flooding and dispersal capabilities. Not only was there no dispersal across catchment boundaries (Figure 4), indicating that episodic flooding does not result in connection or dispersal between catchments, but also dispersal among waterholes within catchments was limited. Baker et al. (2004) found significant structuring of the atyid shrimp *Paratya australiensis* in upland streams of catchments near Sydney, suggesting limited dispersal between and within catchments.

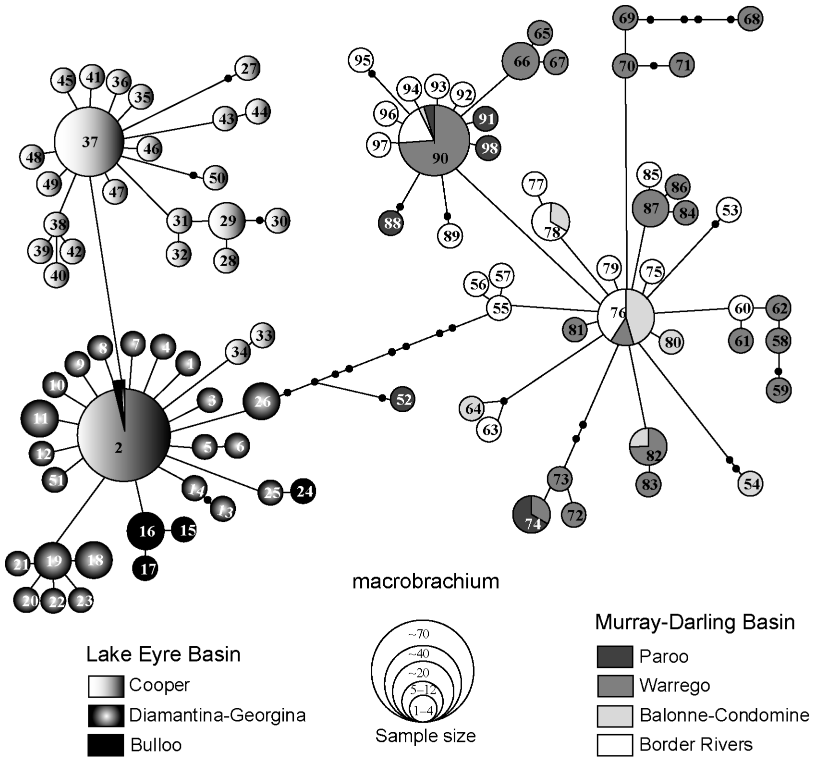

Figure 4. *M. australiense* genealogy network based on haplotype connections (from Carini and Hughes 2004). The results show a distinct separation between individuals collected from the Lake Eyre Basin (left) and Murray-Darling Basin (right). The numbers refer to different haplotypes recorded from individual prawns collected across the study area. The proportion of colours within each pie chart represents the relative number of a particular haplotype from each catchment. Haplotypes are different genotypes of the cytochrome oxidase I gene.

It is often assumed that human activities (e.g. building dams, weirs) block the migration pathways for riverine freshwater fish, thereby affecting the distribution and abundance of normally widely dispersing species. This assumption is being tested in the Ovens, Campaspe and Goulburn catchments in northern Victoria, using two fish species (Australian smelt and flathead gudgeon) that are thought to have very different capacities for dispersal. The chemistry of otoliths (fish ear bones) is being used to determine the scale of dispersal over the lifetime of individual fish, while three genetic techniques (analysis of allozymes, mitochondrial DNA and nuclear microsatellites) are being used to determine the level of contemporary and historical dispersal. Surprisingly, the otolith chemistry results suggest that both species had very limited dispersal among sites within the river systems studied in the first year of the study. Similarly, the genetic analysis suggested that there was either very limited contemporary dispersal between tributaries, or that there are regular localised population declines that result in distinct genetic frequencies between tributaries. The results from a second year of sampling are being analysed to further determine what the genetic and otolith chemistry data reveal about the dispersal patterns of the study species over multiple years (D. Crook, Arthur Rylah Inst., pers. comm.).

Implications for rehabilitation

Limited dispersal means that recolonisation by fully aquatic animals can be very slow, even if suitable habitat is available for survival and breeding. Reconstructing habitat may not be enough to achieve river rehabilitation, particularly in the short term, unless other measures that promote dispersal are also included. Thus, in terms of the ‘field of dreams’ hypothesis (Palmer et al. 1997), there is no guarantee that ‘if you build it, they will come’. While the isolation of populations may mean that the potential effect of barriers such as dams and weirs on recruitment may not be as great as sometimes thought, these findings also imply that restoration of connectivity alone may not be sufficient to ensure recolonisation. We would also hasten to add that there are many other reasons why in-stream barriers pose a problem to the proper functioning of aquatic ecosystems.

2.3. Invasive species

Invasive species have the potential to profoundly affect river ecosystems and restrict recovery from disturbance. Displaced or introduced fauna can have a number of impacts on aquatic ecosystems (Arthington and McKenzie 1997), including:

- physical and chemical impacts on inland waters, such as alteration or degradation of habitat and water quality;
- biological impacts on indigenous/endemic fauna, such as hybridisation or alterations to the genetic structure of populations, and disease transference associated with introduced and displaced fauna; and
- ecological impacts on indigenous/endemic fauna, such as effects on reproduction and survival, abundance, population structure and species distributions, as well as disruption of aquatic communities, and effects on ecosystem processes.

Some recent research has highlighted the way in which invasive species can affect river ecology and increase the hysteresis[2] associated with ecosystem recovery (see Box 2).

Introduced plants and trees can alter stream geomorphology and affect riparian and in-stream food webs. The invasion of the riparian zone by the hybrid white-crack willow (*Salix x rubens*) along the Tarago River in Gippsland, Victoria, resulted in low densities of terrestrial arthropods compared with native riparian plants (Greenwood et al. 2004). This can indirectly alter in-stream food webs by affecting prey subsidies for higher-order consumers such as stream fish (Baxter et al. 2005). Native fish such as galaxiids feed extensively on terrestrial prey in the warm months of the year. Willows, like other introduced deciduous trees, have a massive leaf fall in autumn. This pattern of leaf drop is foreign to native biota and can result in decreased macroinvertebrate diversity because of changed food availability and habitat quality (Read and Barmuta 1999). Invasive grasses such as reed sweetgrass (*Glyceria maxima*) can invaded waterways and greatly alter channel capacity, hydraulics, water quality and habitat quality (Clarke et al.

[2] The influence of the previous history or treatment of a body on its subsequent response to a given force or changed condition.

Box 2. Ecological thresholds and their influence on river rehabilitation (alternate stable states and hysteresis effects)

Ecological thresholds can be considered as turning points, beyond which resource or ecosystem constraints affect river ecosystem attributes such as community structure and ecological processes (DSE 2002). Forcing ecosystems past thresholds can result in rapid regime shifts (cf. alternate stable states), whereby ecosystems with desired attributes are reduced to less desired states, for example with a reduced capacity to generate ecosystem services (e.g. Folke et al. 2004; Walker and Meyers 2004). A regime shift can occur when a threshold level of a controlling variable in a system is passed (e.g. rates of fecundity, growth, mortality, predation, competition, etc.), which in turn alters the nature and extent of feedback mechanisms. The result can then be a change in the nature of the system itself; the shift of a lake from a low turbidity, macrophyte dominated system to a turbid, algae dominated system is an oft-cited example of threshold change between alternative stable states.

Ecological thresholds are important to consider as they are often associated with hysteresis: that is, the influence of the previous history or treatment of a system on its subsequent response to a given force or changed condition. The recovery of river systems from stress or disturbance may or may not proceed in a predictable manner. For example, after rehabilitation, attributes of the system may recover with little delay (Figure 5a) or there may be a lagged response (i.e. hysteresis effect), whereby recovery takes a long time (Figure 5b) — generally much longer than the process of degradation. It is also possible that recovery leads to a new ecosystem condition, either directly (Figure 5c) or via a series of stable and unstable conditions (the 'humpty-dumpty' effect, Figure 5d) (Sarr 2002; Lake et al. 2005).

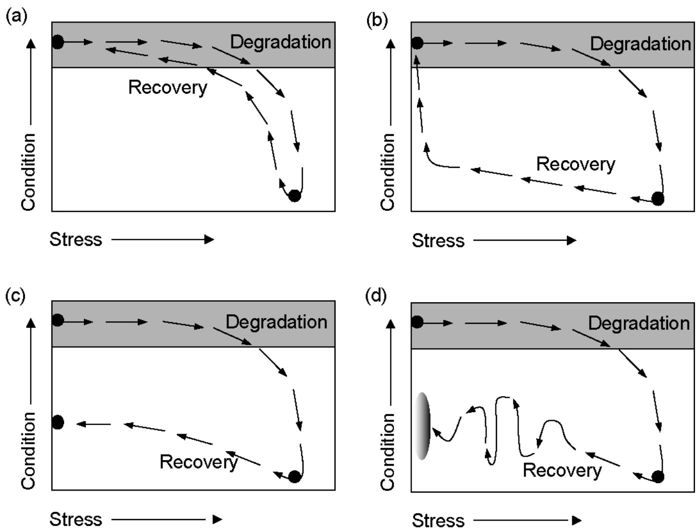

Figure 5. Examples of disturbance and recovery pathways following river rehabilitation (from Sarr 2002; Lake et al. 2005)

The presence of dams and weirs that restrict movement by biota is one example of a threshold or bottleneck that can constrain the recovery of target organisms in response to rehabilitation measures such as habitat rehabilitation. The impact of dams, for example on the recruitment of native fish, can be exacerbated when combined with cold water releases that maintain temperatures below that required by some fish to spawn (e.g. Ryan et al. 2001; Todd et al. 2005).

Box 2: Ecological thresholds and their influence on river rehabilitation (continued)

Salinity is another example where exceeding a threshold results in a disproportionate impact on riverine and floodplain ecosystems. Increases in salinity beyond 1500 EC units (µS/cm) can decrease the abundance, species richness and diversity of plants and animals in wetlands across the Murray-Darling Basin (Figure 6a, Nielsen et al. 2003a,b). Salt can also accumulate in the sediments of wetlands with increasing inflow salinity (Figure 6b). This means that recovery can be constrained by persistently high salinity, both in the water column and the sediments.

The success of proposed river rehabilitation actions can be greatly influenced by ecological thresholds and hysteresis effects. River systems that are in a degraded state can be resistant to change and so large interventions may be required for rehabilitation to succeed. Some examples of thresholds (bottlenecks) that might be considered in rehabilitation plans include:

- allowing a river to flood, so providing connection with its floodplain (i.e. hydrological connectivity threshold),
- passage past in-stream barriers so that biota can return to their previous range (i.e. connectivity threshold),
- salt interception schemes (e.g. catchment replanting or engineered schemes) to reduce stream salinity and remove constraints on species richness and abundance (i.e. water quality and biological threshold),
- replanting a complete reach of the riparian zone to provide sufficient shade to reduce stream temperature (i.e. physical threshold),
- removal of invasive species (plants and animals) and control of livestock grazing in riparian zones to provide a competitive advantage to native species (i.e. biological threshold).

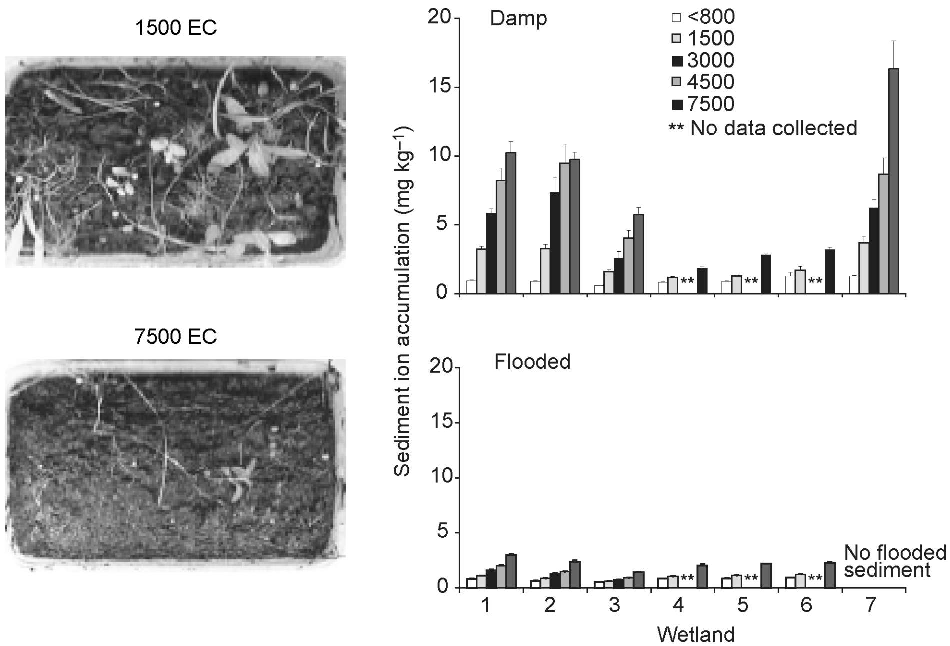

Figure 6. Impact of increasing salinity on (a) wetland plant diversity and abundance and (b) soil salt accumulation in seven wetlands (from Nielsen and Brock 2003a,b)

2004). Thus, invasive plants have the capacity to interfere with in-channel and riparian rehabilitation (see also Pusey and Arthington 2003).

Carp are recognised as a powerful invader of freshwaters in Australia (Koehn 2004; Koehn and MacKenzie 2004). Carp now dominate the abundance and biomass of fishes in many south-eastern Australian waterways, particularly in the Murray-Darling Basin, and may inhibit the recovery of native fish stocks due to competition for, or degradation of, habitat. Once invasive species are established they are usually very difficult, if not impossible, to eradicate. However, there are examples where the control of invasive species has been successful, enabling the recovery of threatened native species (Lintermans and Raadik 2003). Between 1992 and 1995, trout were removed from montane streams in the catchments of the ACT and the Goulburn River in Victoria, to promote the recovery of the mountain galaxias (*Galaxias olidus*) and the endangered barred galaxias (*G. fuscus*). Rainbow and brown trout were removed from the study streams by the use of an icthyocide in places where in-stream barriers (natural and artificial) prevented recolonisation. These actions have led to the establishment of thriving populations of *G. olidus* in the ACT streams, and of *G. olidus* and *G. fuscus* in the upper Goulburn streams.

Implications for rehabilitation

Invasive species can affect the physical, biological and ecological condition of rivers, and thus as well as acting as a biotic disturbance in themselves they can also limit the recovery of native flora and fauna from other forms of physical disturbance. Invasive species can therefore impose ‘threshold’ or ‘bottleneck’ effects that must be overcome if rehabilitation is to be successful (see Box 2).

2.4. In summary — principles for river ecosystem rehabilitation

2.4.1. Key lessons

The previous discussion highlights the influence of scale (both temporal and spatial) on riverine ecosystem structure and function. River and floodplain ecosystems have evolved in response to, and continue to be influenced by, the hydrological cycle, which includes naturally occurring extremes of flood and drought that serve as ecological disturbances.

Habitat for plants and animals exists within a hierarchical arrangement of spatial scales in the landscape, connected by the hydrology of the river. This connection can be longitudinal (along the river), lateral (between the river and its floodplain) and vertical (between surface water and groundwater), and varies both in space (e.g. position in catchment) and time. Projects undertaking river rehabilitation should consider carefully how, in space and time, their projects might be affected by large-scale factors, such as climatic extremes and catchment land-use. Experience in many river systems has also highlighted that the drivers of stream condition can be distant from the locality where ecosystem impacts are evident. Most rehabilitation projects conducted in Australia to date have focused on small-scale issues (e.g. reach or even local site scale activities such as reinstating physical habitat features and riparian vegetation). A key question must be: ‘*does the scale of* *the proposed rehabilitation works match that of the drivers of ecosystem condition?*'.

When considering the conservation or restoration of target species it is important to understand enough about their life cycle and dispersal patterns to effectively manage habitat availability and the species' response to it. The aim is not simply to get the species to aggregate at the restored habitat, but also to enable the species to successfully complete their life cycle. Genetic data suggest that high levels of population fragmentation may be natural in many Australian rivers. Thus we cannot assume that stocks of a local population will be replenished by recolonisation within a river system. In some circumstances, successful rehabilitation may be very slow.

A key attribute of healthy river systems is their resilience to disturbance. Ultimately, rehabilitation aims to increase resilience in degraded systems by contributing to their capacity to withstand natural (and future human-induced) disturbances and reorganise while undergoing change so as to retain essentially the same function, structure and feedbacks as the pre-disturbed state (Walker et al. 2004).

2.4.2. Guiding principles

The following principles related to the ecology of river systems can help practitioners when planning and undertaking rehabilitation projects:

- Riverine ecosystems are structured hierarchically, with important processes operating at a range of spatial scales, from large regional and catchment scales, to sub-catchment and reach scales, and ultimately smaller site and micro-habitat scales.
- Riverine ecosystems can also be highly dynamic and variable in space and time. As such, stream ecosystems in good condition are resilient to periodic natural ecological disturbances (e.g. droughts, floods, fire), which can help drive important physical and biological processes.
- Hydrological connectivity provides strong spatial connections along river networks and between rivers and their floodplains, and plays a key role in ecological processes such as nutrient and energy cycling (spiralling), and the recovery of populations and communities following natural and human-induced disturbance.
- Stream rehabilitation activities are embedded in the hierarchical organisation mentioned above, and should begin with an examination of large-scale factors that might constrain processes acting at smaller spatial scales. Longitudinal processes should also be considered, as the source of degradation can be some distance from locations where ecosystem impacts are evident and rehabilitation is proposed, and loss of connectivity may constrain the biotic response to physical changes in the channel.
- The most effective form of rehabilitation is to prevent degradation of river ecosystems in the first place. Highest priority should go to protecting the remaining high quality river systems (or parts thereof), particularly those that serve as important refugia and are a potential source of colonising organisms.
- Rehabilitation should aim to increase the resilience of river ecosystems to natural (and further human-induced) disturbances so that ecosystems

become self-sustaining and capable of responding to large-scale processes such as climate change and the condition of catchments.

- Where possible, rehabilitation efforts should aim to work with natural processes. This means considering rehabilitation at broader scales than is often practised (most rehabilitation work in Australia has been conducted at small scales) and choosing realistic rehabilitation targets. Given the nature of human impacts, it is unlikely that degraded streams can be returned to their pre-disturbance condition. In such circumstances it can be inappropriate to adopt 'return to natural' as the target for rehabilitation.
- The fragmentation of populations, coupled with sometimes low levels of connectivity (whether because of human interventions such as barriers or naturally poor dispersal abilities), means that many plant and animal species will respond to habitat restoration only very slowly. Isolation may therefore be a major constraint to biotic recovery, and in some cases this could take years or decades. Further, local extinctions may preclude full population recovery.

For rehabilitation to be successful, proper planning and interaction with stakeholders must complement the ecosystem perspective listed above. Socio-economic considerations will play a large role in determining where and when different rehabilitation measures are applied. Planning issues are considered further in Chapter 3.

3. Lessons on planning river rehabilitation projects

One of the biggest lessons learnt from recent rehabilitation projects is that *you are likely to be in this for the long haul, so proper planning is essential if you want to get the most from your efforts*. Often projects are conceived and implemented hastily to take advantage of funding opportunities that may quickly disappear if not used. However, hastily conceived projects will almost invariably fail if factors such as the underlying sources of degradation have not been considered properly, or if there are mismatches between the scale of rehabilitation, the scale of degradation, and the scope of the biota or ecosystem processes that are being targeted. Finally, while ecosystems can be damaged relatively quickly, recovery may be very slow and follow convoluted pathways (Lake et al. 2005).

This chapter emphasises lessons learnt about the importance of proper planning and setting priorities for rehabilitation. Practitioners should refer to Rutherfurd et al. (1999, 2000) and Koehn et al. (2001) for detailed information on how to plan and implement a rehabilitation project. Another resource to consider is the River Styles framework (Brierley et al. 2002; Brierley and Fryirs 2005), which uses the geomorphology templates of river reaches to assess river condition and identify potential rehabilitation options (see Box 3). These references contain considerable information on technical, scientific and engineering aspects of river rehabilitation, as well as tools for processes such as stakeholder interactions, team building and communication.

Reviews of rehabilitation project success (e.g. Smokorowski et al. 1998; Lockwood and Pimm 1999; Lake 2001a; Bond and Lake 2003a; Pretty et al. 2003) suggest that many rehabilitation projects are only partially successful in achieving their stated objectives; success rates in terms of ecological response can be low, even when habitat targets have been met. Some of the reasons rehabilitation projects fail are:

- a lack of clear and agreed rehabilitation objectives,
- a mismatch between the scale of the rehabilitation and the underlying sources of degradation,
- the isolation of newly installed habitat from source populations,
- mismatch between the roles of different stakeholders (e.g. project implementation may be the responsibility of state or local stakeholders, but project funding may depend on other sources over a shorter timeframe),
- a lack of monitoring, evaluation and review.

While this reports focuses mainly on rehabilitation from an ecosystem perspective, it is important to note that the management measures adopted will depend to a large degree on socio-economic considerations. Sources of funding, the role and responsibilities of various stakeholders, and a willingness by stakeholders to be involved (or not) are some of the factors that can affect the rehabilitation measures adopted, and where and when they are applied. In recognising the need for a more effective and sustainable balance between human and ecological needs for freshwater, Poff et al. (2003) identified four key elements they considered critical to

Box 3. Using River Styles for river rehabilitation

It is very important for a river manager to know the geomorphology of rivers. The geomorphology reveals all the physical processes operating in the river channel and on the floodplain. The physical aspects of a river provide the ‘container’ for the river’s ecosystem. For example, if you want an aquarium in your lounge room, you need first a glass tank, substrate and filter system to contain the water and fish. So, the condition of a river depends ultimately on its physical or geomorphic condition.

The River Styles® framework for the geomorphology of rivers (Brierley and Fryirs 2005) was developed in direct collaboration with river managers, with support from Land and Water Australia (LWA) and the NSW Department of Infrastructure, Planning and Natural Resources (DIPNR). To date, the River Styles framework has been applied to 25 catchments in NSW. Forty specially trained DIPNR and Catchment Management Authority (CMA) staff are now applying the framework across most other catchments in the State. The framework is also being used in Queensland, South Australia and Tasmania. At present, 50 River Styles have been identified in Australia. Given the open-ended nature of the procedure, the range of River Styles can be added to as new ones are found, provided they can be identified on air photos.

The River Styles framework uses standardised geomorphology techniques to produce consistent labels for the categories in each of four stages:

1. identification of geomorphic type (Style) using air photograph interpretation and tree diagrams or a dichotomous key for the typology;
2. assessment of present geomorphic condition relative to the evolution of a reference;
3. determination of geomorphic recovery potential;
4. determination of conservation or rehabilitation priority for geomorphic factors.

If a report does not include all four stages (usually due to funding and/or time constraints) then the report is called a ‘geomorphic categorisation’ or ‘geomorphic assessment’ report.

The framework uses a nested hierarchy of scale and all four stages relate to the ‘reach’ scale (a minimum of one channel bend and one transition to the next bend). This scale ‘filters’ or integrates the geomorphic and hydraulic unit information coming from the next lower scales. (e.g. pools, riffles, bars, etc.). The framework also considers the scales higher in the hierarchy: the valley setting, landscape units and catchment. The variability, appearance and behaviour factors of the geomorphic and hydraulic units are the ‘skid marks’ left behind by the geomorphic processes in the previous high flows on the river. This includes the floodplains, if present, and the geomorphic units on them. Each reach-scale Style has a unique appearance and behaviour. If we know the Style then we know its geomorphic behaviour and what it is likely to do in future flow events.

Homogeneous reaches can vary in length from a few hundred metres to tens of kilometres. Due to funding and time constraints, some reports just cover the trunk stream, but most cover the whole catchment including all named streams and the longest arms of those streams to the top of the catchment.

(continued on next page)

Box 3 (continued)

Knowledge of Style behaviour can then be used for river management and rehabilitation decisions (Brierley et al. 2002); for example, to:

1. determine a catchment-wide biophysical ‘vision’ for river management planning;
2. design appropriate ‘target conditions’ for each reach, based on the character, behaviour, condition and recovery potential of the reach, ensuring that they fit within the catchment ‘vision’;
3. prioritise conservation and rehabilitation strategies within-catchment to achieve appropriate target conditions for different types of river based on their particular character, behaviour and present condition;
4. choose a consistent biophysical baseline or ‘template’, upon which additional layers of management information can be added, such as measures of biophysical stress, habitat assessment, riparian vegetation surveys and benchmarking/biomonitoring.

Practitioners use classification systems such as River Styles in conjunction with available physical, biological and ecological information, as the application of River Styles alone may not account for some key drivers (e.g. temperature, stream size, hydrology) of ecosystem condition (Thomson et al. 2004). When it gets down to detail, CMAs and Landcare groups can use their knowledge of the river systems they manage and tools such as River Styles to select appropriate reach-rehabilitation options (Appendix 1).

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developing a shared vision and partnerships for river management and rehabilitation:

1. Implement more large-scale river experiments on existing and planned water management projects.
2. Engage the problem through a collaborative process involving scientists, managers, and other stakeholders.
3. Integrate case-specific contextual knowledge into broader scientific understanding.
4. Forge new and innovative funding partnerships.

A conceptual framework for linking these four elements is presented in Figure 7.

The best laid rehabilitation plans and associated priorities for action may mean little if there is no support, or if there is antagonism, from stakeholders. Time spent with stakeholders that might be involved or affected by a project is a wise investment, as is a communication strategy that keeps stakeholders informed and committed. The use of demonstration reaches and employment of ‘extension’ staff to promote rehabilitation activities, as proposed by the MDBC Native Fish Strategy (MDBC 2003), should be considered where circumstances permit (i.e. there are resources available to support such activities).

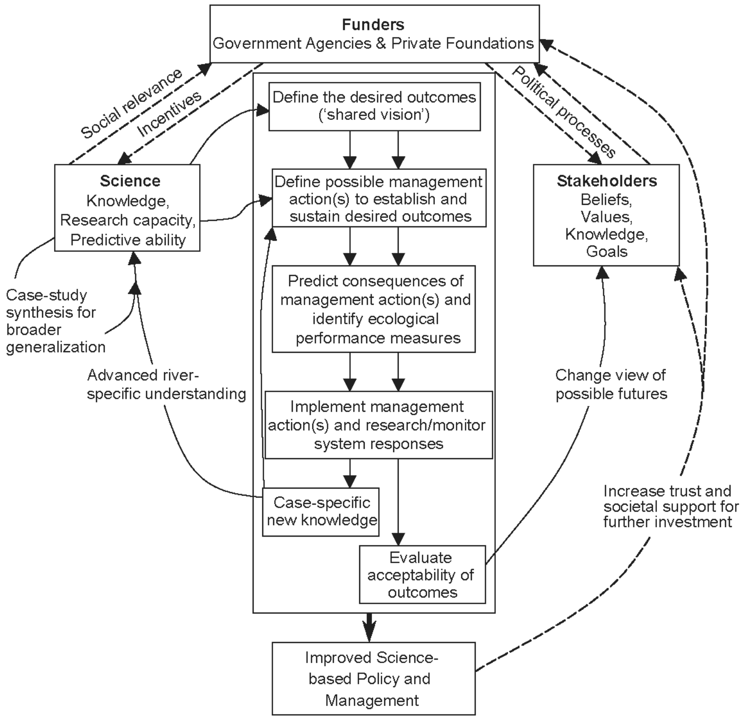

Figure 7. Conceptual flow diagram illustrating interactions and feedback loops between science, stakeholders and funders in the pursuit of improved science-based policy and management of river ecosystems (from Poff et al. 2003).

Adaptive management (AM) is often advocated as a way to improve environmental management and decision making through a process of 'learning by doing'. It is considered to be particularly useful when decisions have to be made in circumstances of incomplete knowledge. However, the application of AM has been criticised (e.g. Schreiber et al. 2004), as it has led to ad hoc changes in managing environmental resources in the absence of adequate planning and monitoring. The best AM outcomes require rigorous and formalised approaches to planning, collaboration, monitoring, modelling and evaluation (Walters 1986; Schreiber et al. 2004).

Rehabilitation projects stand a greater chance of success if the proposed activities operate in concert with natural processes (but recognise that this may not always be enough), and are coordinated with broader land use and management strategies of partners and stakeholders (Hargrove et al. 2002). Obtaining support from the human community at local, regional, and national levels is important to ensure long-term commitment and funding for implementation, monitoring, and the sharing of insights and outcomes.

The main steps in the design and implementation of rehabilitation projects (e.g. Hobbs & Norton 1996; Lake 2001a,b; Suding et al. 2004) are, generally:

- identify the processes leading to the degradation or decline in the condition of the river system;
- define opportunities for and constraints to the reversal or amelioration of the drivers of degradation or decline;

- determine realistic goals for re-establishing species, assemblages of species, ecosystem processes and functional ecosystems;
- recognise potential ecological limitations and socio-economic and cultural barriers to implementation;
- consider the information needs of a monitoring and evaluation program;
- develop conceptual models and testable hypotheses so that rehabilitation projects can be used as management experiments that improve the knowledge base for the future;
- develop easily observable measures of ecological response that can be interpreted as 'success';
- develop practical techniques for implementing the rehabilitation goals at a scale commensurate with the drivers of the problem and the spatial scale at which the problem is apparent;
- document and communicate these techniques for inclusion in planning and management strategies;
- monitor key variables, and assess progress relative to the agreed rehabilitation goals; and adjust procedures if necessary[3].

Planning must account, as far as is possible, for past, present and future conditions. An understanding of the history of the catchment as well as the drivers of current condition is required if the planned rehabilitation measures are to match the nature and scale of the drivers of river condition. The practitioner should expect that gaining such a picture of the rehabilitation location can take some time — spending 6–12 months on the scoping and planning stages is common. Experience has shown the value of care in problem definition, collecting pre-intervention data to allow performance evaluation and securing the involvement of local stakeholders. A pilot study is also very valuable, to add understanding of existing processes at multiple scales, confirm or refine the rehabilitation approach, and identify the variables to be monitored for project evaluation. For example, a pilot study in the early stages of the Granite Creeks project indicated that macroinvertebrate assemblage structure was not likely to be a good indicator of ecosystem response to potential rehabilitation measures, but that individual taxa could be useful indicators (Downes et al. 2005).

3.1. Setting ecosystem rehabilitation objectives

Freshwater ecosystems are naturally dynamic in both space and time. Natural systems are not static and rehabilitation goals should account for this. The stochastic (i.e. random or chance) nature of environmental drivers such as flow may mean that the same rehabilitation measure can have different outcomes when applied at different locations or different times (Hobbs and Norton 1996). It is therefore important to first focus on landscape-scale processes when planning a rehabilitation project, before considering location specific actions (Lake 2005; Lake et al. 2005).

[3] It should be remembered that there will not be the opportunity or resources available to monitor every rehabilitation project that is implemented. The criteria by which practitioners can determine whether or not monitoring and evaluation should be an essential part of a rehabilitation project are considered further in Chapter 5.

The following key steps (Kershner 1997) should be considered when setting rehabilitation objectives:

1. Characterisation of the system where rehabilitation activities are to occur.
2. Identification of key issues and questions to be answered as part of a rehabilitation (experimental) project.
3. Documentation of current condition.
4. Description of reference condition.
5. Identification of potential rehabilitation objectives.
6. Summary of conditions and determination of causes.
7. Recommendations on specific and measurable objectives.

Koehn et al. (2001) also recommend that rehabilitation objectives consider four areas: (i) the restoration activities, (ii) the monitoring activities, (iii) evaluation activities, and (iv) maintenance activities. There should be well-defined objectives for each river attribute to be addressed, including measurable goals to be achieved within a given timeframe. A key question when developing rehabilitation objectives is 'how much rehabilitation is enough' given the circumstances?

3.1.1. Priority setting

Identifying priorities for rehabilitation can be a difficult task. If rehabilitation is required to overcome the effect of a discrete disturbance (e.g. source of erosion or contamination, or an in-stream barrier to movement), then the priority for rehabilitation can be clear. If rehabilitation is to meet the needs of a single species or community assemblage, then their biological needs set the agenda. However, most commonly, rehabilitation is attempted in river systems that are affected by multiple stressors, often acting at different scales.

What are the priorities for rehabilitation for a river system affected by multiple stressors such as catchment land clearing, draining of wetlands, a highly modified flow regime, high nutrient levels, excessive erosion and sedimentation, loss of in-channel habitat, and/or damaged riparian zones?

Koehn et al. (2001) identify a number of tools that can be used by project teams as they identify objectives and set rehabilitation priorities, such as:

- SMART principles (specific, measurable, achievable, realistic and time-bound),
- multi-criteria analysis,
- matrices of stressors and environment, social and economic impacts,
- strategic priorities (e.g. from regional catchment strategies),
- adaptive management and assessment modelling and decision tools.

A two-tiered approach for identifying priority areas for conservation and rehabilitation was trialled on the Sydney catchments (Linke and Norris 2003). The first step used the AUSRIVAS methods of assessing river health (e.g. Turak et al. 2004) to identify areas with significant macroinvertebrate biodiversity loss (Figure 8). The second step involved separating out those areas affected by biodiversity loss and identifying areas with higher than expected taxon (species) richness. This pilot study identified three groupings: (i) areas with a high conservation value, (ii) areas with significant biodiversity loss but with potential for recovery to become biodiversity hotspots, and (iii) areas that had suffered severe loss of biodiversity and required extensive rehabilitation. The approach used by Linke and Norris (2003) was data-driven and repeatable, and has potential as a management tool as it integrates the management of condition and biodiversity conservation.

It is generally agreed that the highest priority for management and rehabilitation should go the protection of high value, least-disturbed riverine and floodplain assets (e.g. Rutherfurd et al. 1999, 2000; Koehn et al. 2001; Roni et al. 2002). These systems are often those that are representative of the best available condition, that can be hotspots of biodiversity or productivity, or serve as refugia to supply colonising organisms that can disperse to newly rehabilitated or available areas. These assets play a critical role in maintaining river systems' resilience and ability to recover from disturbance (Lake 2001b; Roni et al. 2002; Bond and Lake 2004).

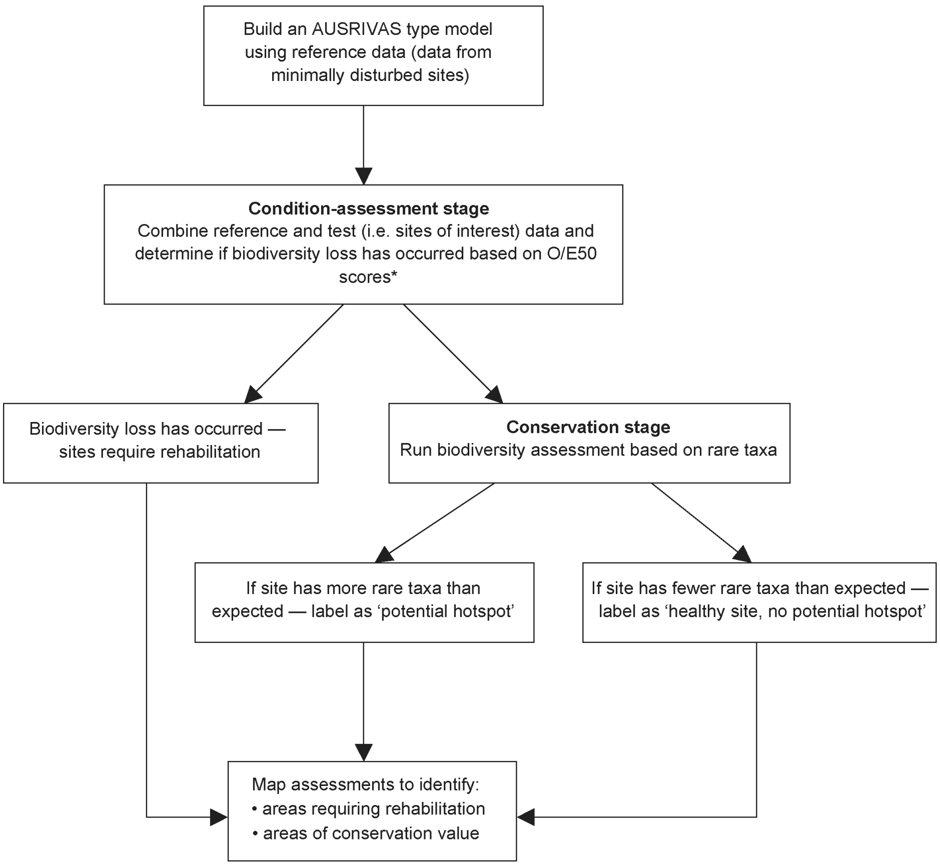

Figure 8. Flowchart for a two-tiered approach, integrating assessment of condition and conservation value (adapted from Linke and Norris 2003). *An O/E50 score is the ratio of the number of taxa recorded (Observed) at a test site relative to the taxa with a probability of occurrence >50% (Expected) at reference sites (see Linke and Norris 2003 for more details).

Table 1. Likelihood of success in achieving five rehabilitation goals (columns) in urban areas by means of five types of management action, alone or in combination (rows) (from Walsh et al. 2005b). Allied stressors include sanitary sewer overflows or leaks, and point source or legacy pollutants.

Rehabilitation measure	Aesthetics/ amenity	Channel, stability	Enhanced N, processing	Improved ecological condition	
				Riparian	In-stream
1. Riparian revegetation	s			s	
2. In-stream habitat enhancement	s	s	s		
3. End of pipe stormwater treatment	*?		*		
4. Elimination of allied stressors	*?		*?		
5. Dispersed stormwater treatment		*	**		
3 + 4	*?		*		
5 + 4	*?	*	**		*
5 + 4 + 2	*	*	***		**
5 + 4 + 2 + 1	*	*	***	*	***

Dispersed stormwater treatment is assumed to be extensive enough to reduce frequency of runoff from the catchment to near the pre-urban state. s = some improvement likely but long-term sustainability unlikely, *? = improvement likely in some cases, *, **, *** = increasing degrees of improvement likely.

Many rivers are affected by multiple stressors, particularly in urban areas. This can make priority setting very difficult. For example, Walsh et al. (2005b) suggest that a number of measures would be required to ensure the rehabilitation of streams affected by stormwater runoff and point-source or legacy pollutants (Table 1). A similar situation probably applies in many rural landscapes.

Figure 9 presents a general framework that can be applied when considering rehabilitation priorities. Management priorities will be influenced by social and economic considerations, in addition to the ecosystem considerations that are the basis of the framework. It is also important to consider whether it is feasible to address historical legacies and present disturbances that operate at large scales (e.g. catchment) with rehabilitation actions at smaller scales (sub-catchment, reach).

3.2. In summary — lessons on planning rehabilitation experiments

The previous discussion highlighted the following points:

- Australian and international experience suggests that most rehabilitation projects will fail if the scale of stressor(s) and/or the scale of the response required are not considered carefully. It is important to take the time to examine and understand the history of land-use change and management in your catchment.
- It is important to establish working relationships with the stakeholders who will be involved in decisions on the rehabilitation techniques that can be applied and the ongoing conduct of a project. This will take time and energy, but is a wise investment.
- Planning for rehabilitation projects should include an assessment of the scale of past and present factors that have affected current conditions and resulted in the need for rehabilitation. Examine the history of your

catchment. Have the stressors that have caused degradation been identified? Are they still active? At what scale do the stressors apply? Is the river system still responding to these or other stressors?

- The highest priority should go to the protection of high value riverine and floodplain assets (systems); for example, those in best condition, that are representative, that are hotspots of biodiversity or productivity, or serve as refugia and can supply colonising organisms that may disperse to newly rehabilitated or available areas. These assets play a critical role in maintaining the resilience of river systems and their ability to recover from disturbance.
- River systems are often subject to multiple impacts and legacy effects from past land and water management practices. While some riverine attributes may respond quickly to rehabilitation, full recovery is likely to take decades and will require ongoing commitment and patience.

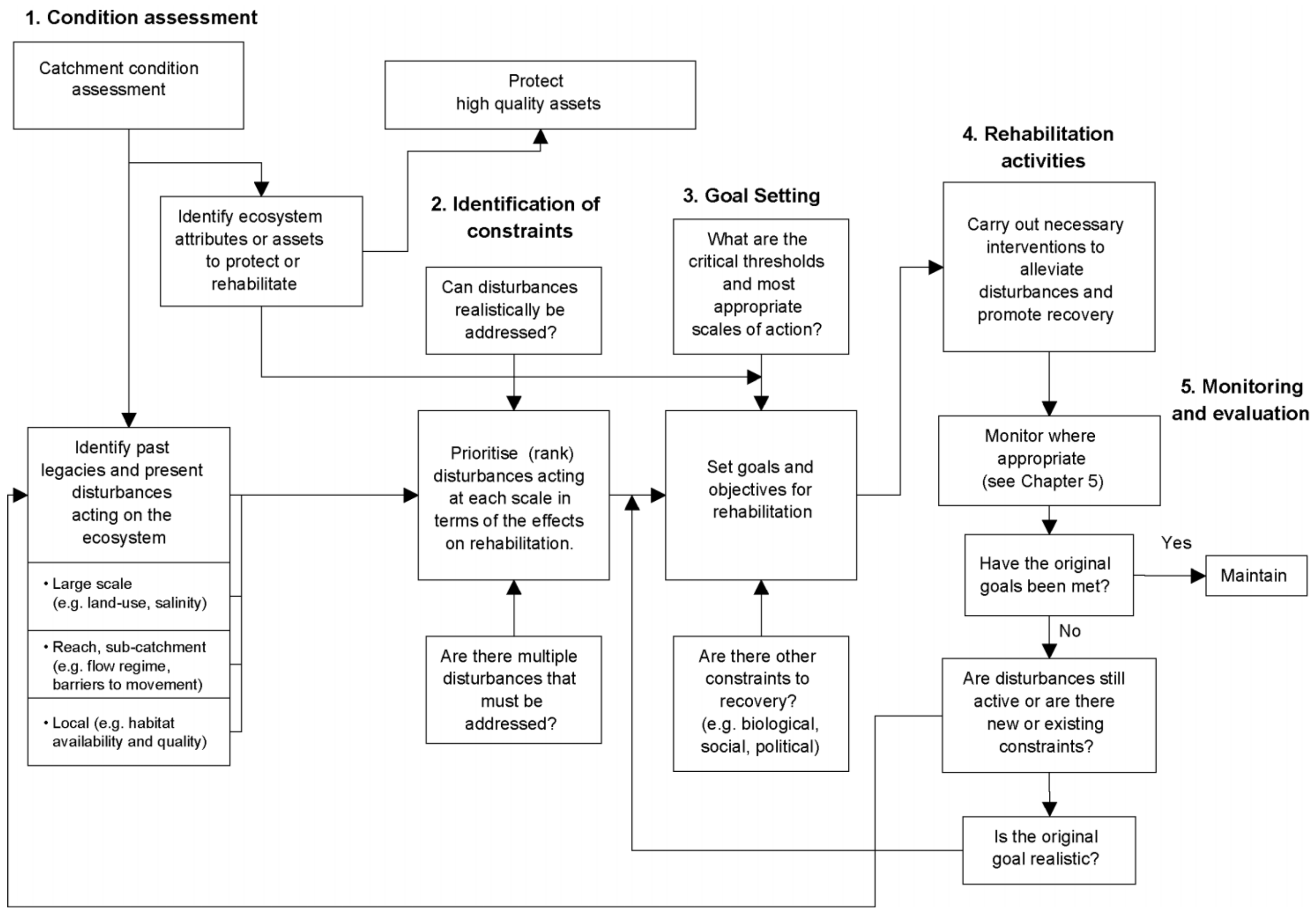

Figure 9. General approach to identifying rehabilitation constraints and priorities (adapted from Suding et al. 2004)

4. Rehabilitation tools — where and when do they work?

The previous discussion has emphasised the need for a holistic view of river rehabilitation at large scales (e.g. catchment). This helps us understand the key drivers of ecosystem condition, develop rehabilitation objectives and set priorities for action. Often, a number (mix) of actions will be required, at a range of scales, if rivers are to be rehabilitated successfully. This chapter now describes some of the lessons learnt about individual rehabilitation techniques and their application.

Actions such as (i) reintroducing wood and riffle habitat for fish and macroinvertebrates; (ii) riparian revegetation to enhance in-stream and riparian ecosystem processes and trap sediment and contaminants; and (iii) the use of wood and rocks to stabilise streams, are commonly considered by rehabilitation practitioners in Australia. However, until recently, it was assumed that biota would automatically respond to the reintroduction of habitat — and the truth of this premise, in the short and the long term, was not evaluated (see Lake 2001). Recent research and rehabilitation projects have provided some useful insights on the effectiveness of rehabilitation measures such as:

- meeting the environmental water requirements of rivers,
- reintroducing flow-related habitat for fish and invertebrates,
- reintroducing wood (e.g. resnagging),
- rehabilitating riparian revegetation,
- introducing riffle habitat to streams in urban areas,
- managing sediment, both in-stream and in the riparian zone,
- adding artificial substrate.

4.1. Protecting or reinstating components of the flow regime (environmental flows)

Protecting or reinstating components of the flow regime of a river system is an important rehabilitation tool for flow-stressed rivers. Numerous environmental flow methods have been proposed over the last decade (Arthington and Zalucki 1998), based on aspects of stream hydrology and hydraulics, the needs of individual species or community assemblages, and water quality. Most recent environmental flow studies have applied ‘holistic’ methods (e.g. Building Blocks, Victorian FLOWS method, Flow Events Method) that recognise the *natural flow paradigm* (Poff et al. 1997) and consider the river system at the widest possible scale (i.e. river channel and its associated floodplain). Holistic methods consider the role of the flow regime from the perspective of hydrology, geomorphology, biological and ecological disciplines (Arthington et al. 2000 a; Jones et al. 2001; Stewardson and Cottingham 2001; DNRE 2002; Cottingham et al. 2003b). A key aim in flow-stressed rivers is to reintroduce to the flow regime the variability which has been modified with flow regulation and diversion, particularly when this affects habitat availability or ecosystem functions and community structure (e.g. King 2004a,b; Maddock et al. 2004; Scholz and Gawne 2004; Arthington et al. 2005).

Chapter 3 emphasised that protection of riverine systems in good condition is a high priority for river rehabilitation. In Victoria, the Sustainable Diversion Limits (SDL) approach was developed to protect ungauged and unregulated streams from over-extraction during winter when water is commonly diverted to fill farm dams. SDLs ensure that enough water is left in the stream to meet its environmental requirements (DNRE 2002b). They are based on the following four rules when streamflow data are available:

- diversions should only occur over the months of July to October, inclusive;
- diversions should cease when flows drop below the larger of:
  - 30% of the mean daily flow (computed over the whole year),
  - the median winterfill flow that is exceeded in 95% of years (computed over the months of July to October, inclusive);
- the maximum daily rate of diversions should be set to the difference between the median winterfill flow exceeded in 50% and 80% of years (computed over the months of July to October, inclusive); and,
- a fixed annual limit should be placed on diverted volumes that ensures that the reliability of winterfill supply is *at least* 80%.

The above rules need to be applied in the context of the overall health of the ecosystem. In particular, in-stream habitat conditions, water quality and riparian condition need to be part of the decision making process.

Most flow-stressed river systems are subject to multiple stressors (i.e. stressors in addition to a modified flow regime). Successful rehabilitation will require not only a suitable environmental watering regime, but also complementary actions such as the:

- amelioration of cold-water release from large dams;
- control of sediment erosion and movement;
- review and removal of unnecessary levees and block banks;
- provision of fish passage around barriers to movement;
- implementation of invasive-species (e.g. carp) control strategies;
- control of livestock access to the riparian zone and wetlands;
- implementation of rehabilitation strategies for water quality, revegetation and stream physical habitat.

4.2. Increasing hydraulic habitat diversity

The amount and condition of physical or hydraulic habitat has declined in many river systems across Australia (Commonwealth of Australia 2002a,b) through actions and processes such as alterations to flow regimes, wood removal (desnagging), channel realignment and stabilisation works, stormwater runoff from rural and urban areas, and access by livestock. A number of rehabilitation measures that aim to improve habitat condition and availability have been trialled in south-east Australia in recent years. The following sections describe some of the lessons learnt from these trials.

4.2.1. Reintroducing flow-dependent habitat

Slackwater patches within rivers are often small shallow areas of still water formed by sand bars, woody debris and bank morphology. Slackwaters provide protection from fast currents, have abundant biofilm, and contain high densities of zooplankton and microbenthos — prey for the young stages of fish and shrimp. These areas can be highly productive and are important rearing habitats for many species of fish and shrimp (e.g. Humphries et al. 1999, 2002; King 2002, 2004a). However, slackwater patches are also very vulnerable, and may be lost with altered hydrology (e.g. increased discharge that raises water velocity), or physical works such as the simplification of channels and removal of structural features such as logs, or the deposition of sediments from erosional areas.

Physical structures such as groynes, piles and revetments have often been used as stream stabilisation measures. Such physical structures can also be used to create or reintroduce hydraulic habitat (slackwater areas) that may have been lost due to past or current river management or operations. In an experimental trial of slackwater habitat, sandbag walls were used to alter the availability of hydraulic habitat for macro- and micro-crustaceans and fish in the Broken River in northern Victoria (Figure 10). The sandbags were used to direct streamflow into 'slackwater' environments to create 'flowing' patches, and also to direct flow away from edge areas to create slackwater patches (B. Gawne pers.comm.; Humphries et al. 2005).

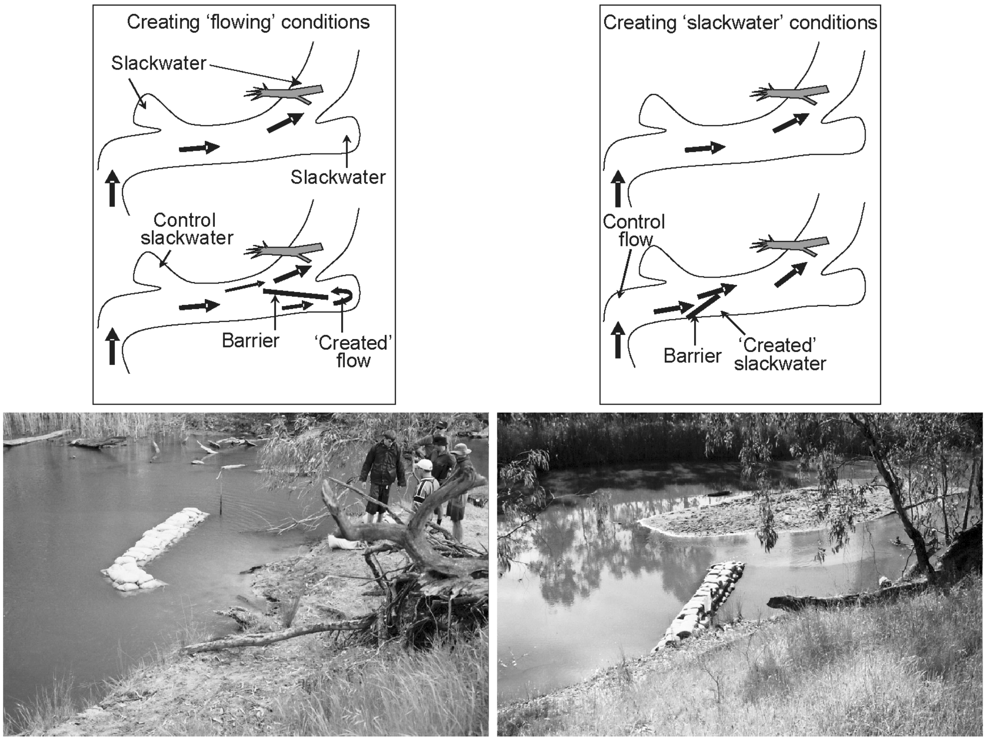

Figure 10. 'Created flow' habitat and 'Created slackwater' habitat in the Broken River, using sandbag walls (P. Humphries, Charles Sturt University, pers. comm.)

The biotic and abiotic characteristics of the newly created flowing and slackwater patches were found to be equivalent to those occurring naturally. Although there was no difference in either the abiotic character or primary productivity of the flowing and slackwater patches, each formed distinct biotic communities. Microinvertebrate, fish and shrimp abundance was greatest in the slackwater habitats, whereas macroinvertebrate abundance was greatest in the flowing patches. These results suggest that hydraulic conditions influence the stream communities that may be present, and ultimately the structure of food-webs. The project was an initial trial of a technique and was not set up to answer questions such as how much hydraulic habitat may be necessary and where it should be located along a river. One way to gain this information is to develop a hydraulic model to examine flow-habitat relationships for the area of interest. For example, such a model could be used to examine the slackwater habitat required by target organisms at critical stages in their life cycle. A comparison of current habitat availability with that available under some reference condition (e.g. with the influence of flow regulation removed) can show if the availability of slackwater habitat has decreased, and help to identify the locations and sizes of habitat patches that might be reintroduced (B. Gawne, MDFRC, pers. comm.).

Another experiment (commissioned by the MDBC) to increase hydraulic habitat diversity was conducted in the Murrumbidgee River in the ACT. The aim was to increase hydraulic habitat diversity and promote the dispersal of native fish, considered to be impeded by shallow water depths and a lack of cover. Rock groyne flow deflectors were installed to create scour pools in a 1.5 km reach of the river that had been affected by sand deposition (Lintermans 2005). A habitat pool (15 m long, 5 m wide, 3–5 m deep) was also excavated from the riverbed below the series of groynes to increase hydraulic habitat diversity in an otherwise featureless sandy bottom (Figure 11a). The groynes were placed approximately 60 m apart (approximately 5 times the low channel width), extending to approximately 60% of the channel width from the bank closest to the thalweg. Hardwood logs (6–8 m long) were placed immediately downstream from alternate groynes, and in the habitat pool, to increase structural complexity. Flow around the groynes created scour pools and therefore increased hydraulic habitat diversity (Figure 11b). The scour pool results suggested that creating a continuous scour pool would require a deflector spacing between 1 and 1.5 times the width of the low flow channel. The project also found that changes in thalweg position can result in deflectors becoming isolated from the low flow channel (therefore reducing effectiveness).

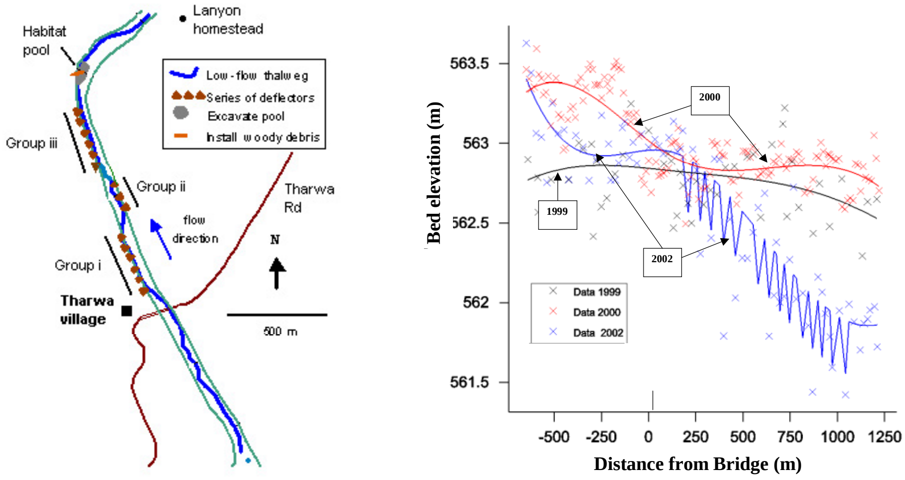

Figure 11. Diagrams showing (a) location of the low-flow thalweg, deflectors and habitat pool in in the Murrumbidgee River, ACT, and (b) plots of bed elevation with distance in the Murrumbidgee River in 1999 and 2000 (pre in-stream works) and 2002 (post in-stream works) (from Lintermans 2005). The lines represent a smoothed function fitted to the raw data, and allowing it to change from year to year. The statistical analysis takes account of the very strong correlation of one measurement to another. An average effect of the deflectors has been incorporated into the 2002 fitted function. The bridge is at distance 0, with negative distances indicating upstream of the bridge and positive distances downstream of the bridge.

Our ability to create or reinstate hydraulic habitat diversity could be very useful in streams where the flow regime has been altered. For example, many regulated rivers are used to deliver water for irrigation, often with flows approaching bankfull discharge. This can lead to a seasonal ‘inversion’ of the flow regime (e.g. Thoms et al. 2000; Jones et al. 2001; McMahon and Finlayson 2003) where high discharge and fast water velocity occurs at times when we would normally expect low-flow, low-velocity conditions. Altered hydrological patterns can affect hydraulic habitat availability and disrupt biological patterns of aquatic organisms that have life-cycle stages adapted to natural flow patterns (e.g. Humphries et al. 1999). Being able to reintroduce some of the hydraulic habitat lost due to the imposition of a new flow regime would mean that it may be possible to meet the needs of aquatic organisms, while still being able to deliver water for consumptive use.

4.2.2. Reintroducing wood to create habitat for fish

A number of projects have noted a positive response, particularly by fish rather than benthic invertebrates, to the addition of wood as a habitat feature. However, the positioning of wood in a river reach is an important consideration. Recent investigations have noted particular small-scale habitat associations between native fish and the positioning of structural habitat such as wood (Crook et al. 2001; Nicol et al. 2002; Bond and Lake 2003b).

Early investigations in the Granite Creeks project suggested that the fish species present were likely to respond positively to the reinstatement of small-scale habitat patches. The small-bodied species present were found to be 'elastic' in their habitat preference, and likely to take advantage of any suitable habitat present (deep, slow flowing, close to vegetation cover or wood) in an otherwise homogeneous environment (Bond and Lake 2003b). Augmenting small-scale habitat availability was therefore considered likely to result in increased fish abundance. Wood structures (railway sleepers, singly or in groups of four) were placed in sand-affected sections of the Granite Creeks to test their effectiveness as a rehabilitation tool with a view to scaling up their installation to cover the lengths of the sand slugs.

The wood structures (Figure 12) increased habitat heterogeneity (additional wood, formation of scour holes, accumulation of coarse particulate organic matter (Figure 13) (Bond and Lake 2005a,b). Native fish such as mountain galaxias responded positively (Figure 14) by increasing in abundance in newly formed scour pools (i.e. taking advantage of deep water habitat), particularly at sites with the greater number of wood pieces (i.e. four railway sleepers). However, the relationship between the wood structures and the resulting scour pools was not simple, as initially assumed (Borg et al. 2005). The scour pools that formed were smaller, more variable and did not extend downstream as far as predicted. The scour pools subsequently filled in during periods of low flows, and eventually the initial habitat gains were lost.

The Granite Creeks project coincided with the recent period of extended drought in eastern Australia. While the initial response of fish to newly formed habitat in sand-affected areas was positive, the response did not persist when the drought took hold, the streams dried and the scour holes filled in. This highlighted the need to consider seasonal and supra-seasonal factors and how they may affect recovery. Where residential and refuge habitats do not coincide, extra effort may be required to enhance the ability of organisms to recover from disturbances such as drought (i.e. it is important to protect existing high quality habitat, especially if it serves as refuge for recolonising organisms).

Similar associations between native fish and small-scale habitat availability have been recorded for other lowland rivers. Radio-tracking of fish in the Broken River (Crook et al. 2001) showed that golden perch were mostly associated with wood in deep water (pools), particularly during daylight hours (Figure 15), often ignoring wood habitat in run sections of the river that did not provide the same level of protection from predation. These findings emphasise the need to identify habitat features that are important for rehabilitation and the scales at which they are available in the landscape (Bond and Lake 2003a; Lake et al. 2005). For example, what residential, breeding, rearing or refuge habitats might a target species require, where are they located and are there barriers that may prevent access to these habitats at critical life stages (Box 4)?

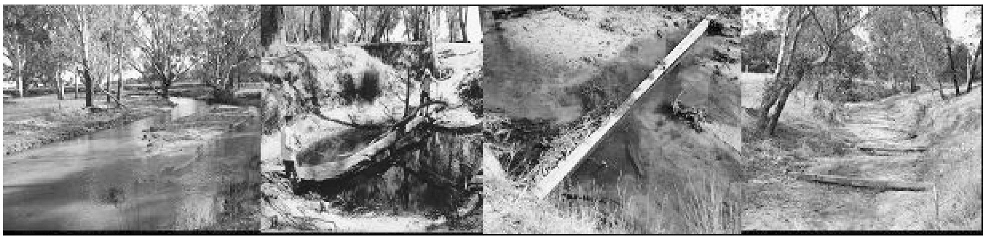

Figure 12. Features of the Granite Creeks, (i) sand-slug section with little in-stream habitat diversity, (ii) a natural scour pool in the 'chain-of-ponds' sections, (iii) accumulation of organic matter around a wood structure, and (iv) wood structures after the stream had dried (all photos by N. Bond)

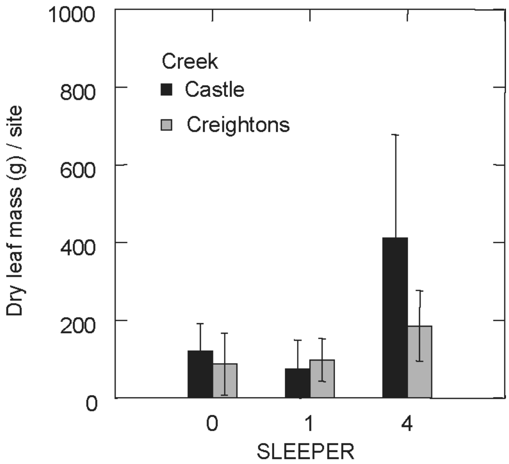

Figure 13. Coarse particulate organic matter response to the wood introduction in sand-affected sections of the Granite Creeks (N. Bond, Monash University, pers. comm.)

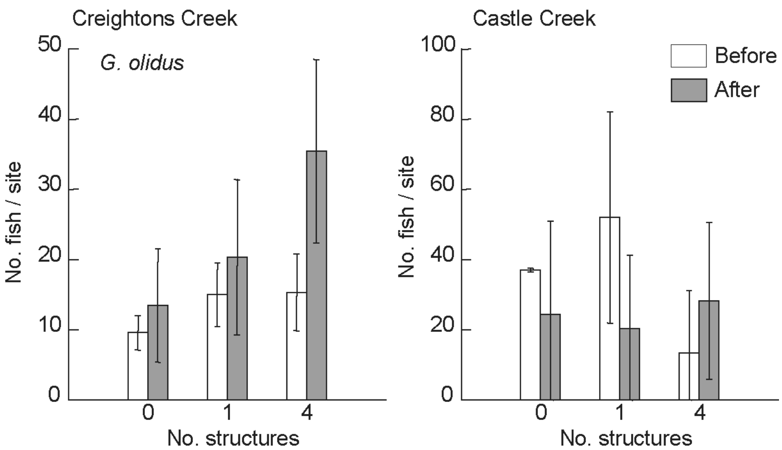

Figure 14. Mountain galaxias (*Galaxias olidus*) response to the introduction of wood structures in the Granite Creeks (from Bond and Lake 2005a)

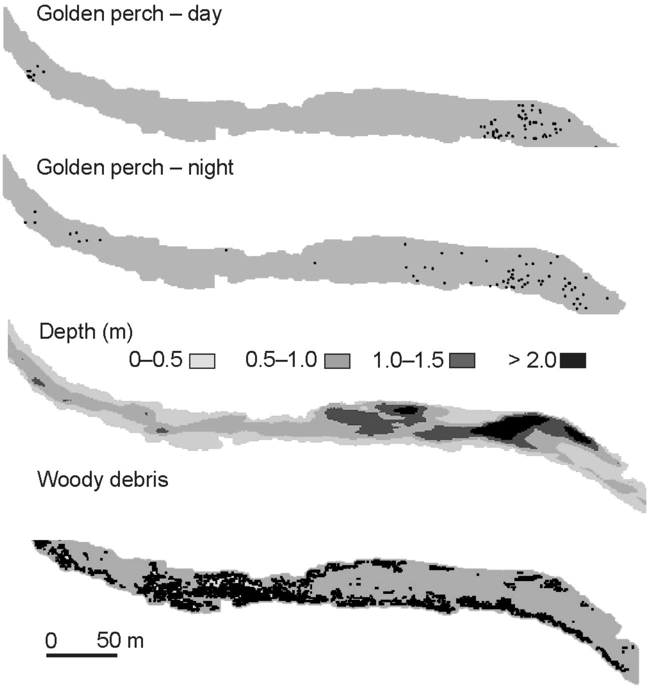

Figure 15. Distribution of golden perch and spatial arrangement of measured habitat variables within the 450 m study reach of the Broken River, Victoria (from Crook et al. 2001). The direction of flow is from right to left.

Resnagging works in the River Murray between Yarrawonga and Cobram provided a number of insights on the natural distribution of wood and its importance as structural habitat for native fish in large rivers (Hughes and Thoms 2002; Nicol et al. 2002; Koehn et al. 2004a,b). Large logs were found to be more concentrated (i.e. more densely arranged) in eroding zones of river bends (e.g. second half of outside bends and first half of inside bends). The distribution of logs along straight river sections was found to be uniform. The orientation of logs was related to patterns of tree fall, which was mostly perpendicular to the bank. Wood was mostly present as trunks and large branches, with smaller material either degrading faster or being broken off and transported downstream. However, there was found to be little movement of large logs following a 1 in 20 year flood, so log movement was considered to contribute little to the observed distribution of timber in the reach of the River Murray that was studied.

Native fish such as Murray cod, golden perch and trout cod were more common in moderately curving meanders and eroding banks that were coincident with snag piles. Nicol et al. (2002) examined fish distribution within four quadrants of the meander bends of the River Murray and found that each species studied preferred different quadrants (Figure 16). Murray cod were found to prefer shallower water in quarters 1 and 4, trout cod preferred deeper water in quarter 1, golden perch preferred deeper water in quarters 1, 3 and 4. All were more abundant near snag piles, suggesting that native fish responded to an increased density of snags, rather than uniform distribution (Figure 17).

Box 4: Check-list when planning to reintroduce wood

Based on their experience in the Granite Creeks and elsewhere, Bond and Lake (2003a) posed a number of questions to be considered when contemplating the reintroduction of wood as habitat for fish and macroinvertebrates. These questions can also apply to other rehabilitation methods and target organisms.

1. Are there barriers to colonisation?
   - What and where are the source populations?
   - How can potential barriers be overcome?
2. Do the target species have particular habitat requirements at different life stages?
   - What are these requirements?
   - How should these habitats be arranged relative to each other?
3. Are there introduced species that may benefit disproportionately to native species from habitat restoration?
   - Can colonisation of these organisms be restricted?
4. How are long-term and large-scale phenomena (e.g. drought, flood, urbanization, climate change) likely to influence the likelihood or timeframe of response?
   - Will these affect the endpoints or just the timeframe of responses?
   - How will this affect monitoring strategies, and can monitoring strategies be adjusted to deal with this?
5. What size habitat patches must be created for populations, communities and ecosystem functions to be restored?
   - Is there a minimum area required?
   - Will the spatial arrangement of habitats affect this (e.g. through the outcomes of competition and predation)?

It is also well to remember that in systems affected by large-scale disturbance such as drought, maintaining or rehabilitating refugia can be as important as rehabilitating residential habitat.

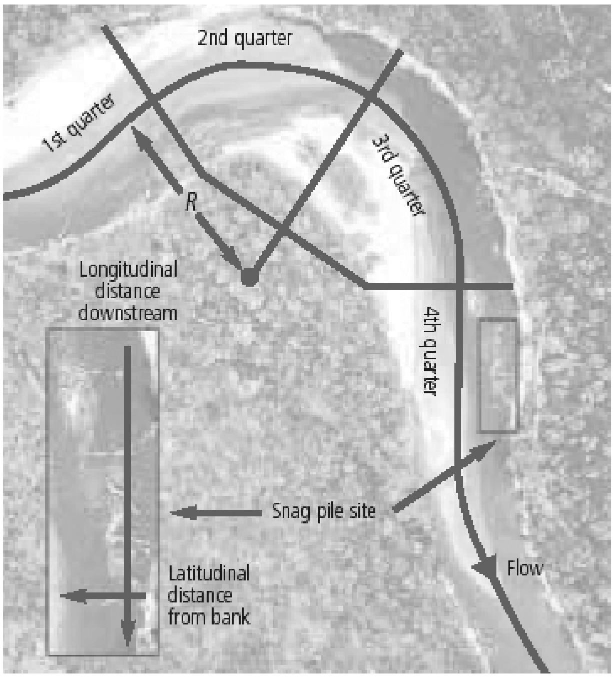

Figure 16. Quadrants of a meander bend in the River Murray (from Nicol et al. 2002)

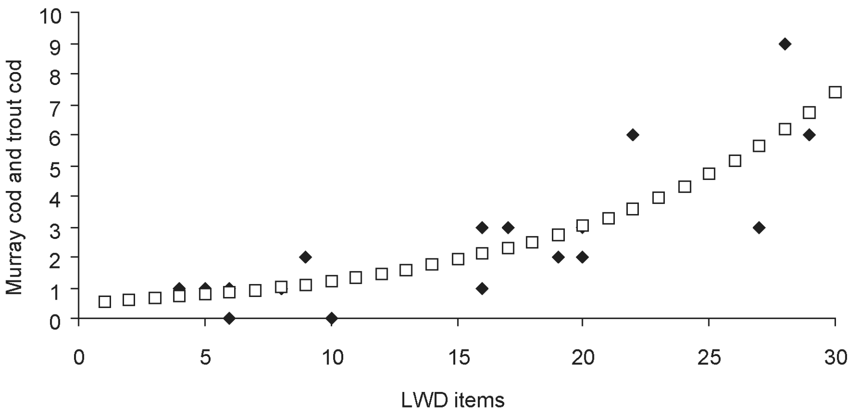

Figure 17. Plot of the relationship between the number of Murray cod and trout cod in a site and the abundance of wood items in the site (filled black diamonds). The clear squares are the line of best fit for this relationship (from Nicol et al. 2002).

The distribution of logs and the pattern of habitat use by native fish provided a template that could be used for reintroducing wood to nearby reaches of the Murray. The template indicated that logs are best placed on eroding banks on the outside of meander bends. The protection of this zone by revegetation was also recommended, to ensure natural rates of bank erosion and to provide a future source of snags.

Native fish responded to the reintroduction of wood at sites on the River Murray that had been previously cleared of logs (Nicol et al. 2002), with native fish abundance increasing to levels comparable with reference sites (Figure18). Macroinvertebrates were found to colonise conditioned wood (previously submerged and saturated) more rapidly than green or dry wood. Nicol et al. (2002) used their findings to develop landscape and local-scale cost–benefit models (landscape and local site scales) to help identify where the reintroduction of logs could be used to the best possible effect. The

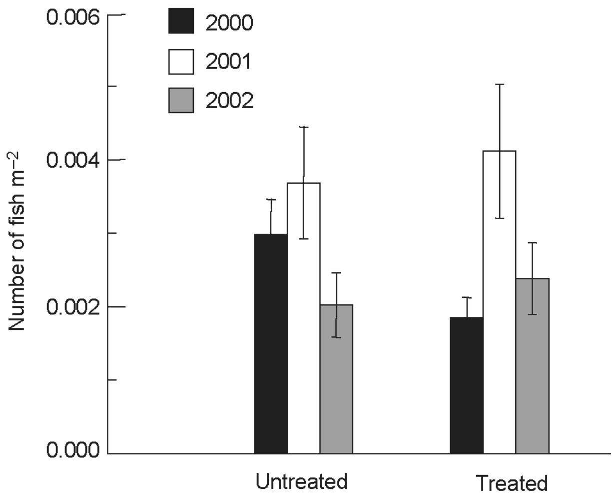

Figure 18. Total numbers of native fish caught at untreated (no added wood) and treated (wood added) sites before and after the addition of wood (resnagging) (from Nicol et al. 2002). Note untreated and treated sites were significantly different before resnagging in 2000 but were not significantly different after resnagging (2001 and 2002).

landscape scale model identified that resnagging sites to establish spatially independent sub-populations (metapopulations) within the colonisation range of a source population was a critical consideration. For the section of the River Murray studied, the modelling results suggested that resnagging in a tributary downstream of existing native fish populations, rather than solely in the main channel, should be considered to increase the resilience of target populations (e.g. trout cod) to disturbance.

The results of the wood reintroduction projects described above have so far confirmed that wood is a vital component of riverine ecosystems. They underpin moves to return wood to rivers where it can contribute to increased species richness and abundance and enhance ecosystem processes, provided there are no other ecological or physical constraints. In a recent review of findings related to the ecological and geomorphic role of wood in streams, Cottingham et al. (2003a) detailed five steps that should be taken when seeking to reintroduce logs:

(i) identify geomorphic and ecological objectives,
(ii) establish rehabilitation design and targets,
(iii) consider the likely response of the stream to the rehabilitation measures,
(iv) locate an appropriate source of wood,
(v) monitor and evaluate the success of the project.

These are consistent with the planning and assessment protocols advocated elsewhere in the document, and it is recommended that practitioners review these steps when planning to reinstall wood in rivers. Nicol et al. (2002) also provide useful tips for reintroducing logs, such as the use of reference reaches for estimating log density, how best to handle green (recently fallen) or dry timber (the latter may require ballast as they become buoyant after drying), the shape of timber preferred by fish (e.g. branched rather than uniform, or with hollows).

Artificial substrate is sometimes considered as an alternative when natural materials such as logs are unavailable. Artificial macrophyte beds were introduced to sections of the Broken River in Victoria to examine whether they could support the survival of newly hatched fish by increasing food availability and habitat structure. Beds (1 m x 1 m) of positively buoyant green 'Canvacon' plastic tarpaulin were cut ('leaf blades' 30–50 cm long and 1–2 cm wide) to mimic the aquatic macrophyte *Vallisneria.* Chlorophyll-a, algae, macroinvertebrates and fish associated with the artificial beds, natural *Vallisneria* beds, and in-channel controls were analysed, as were whole river metabolism and metabolism on the 'leaf blades' of the artificial macrophyte beds (Merrick et al. 2001).

Even within the relatively short period (1 year) of this study, the beds were found to be reasonable surrogates for natural macrophytes, in terms of epiphytic biomass, community composition and chlorophyll concentration. Both the artificial and the natural macrophytes enhanced the development of algal, zooplankton and macroinvertebrate communities. These results suggest that these substrates can help to support fish populations, provided that other habitat requirements are met. Surprisingly, larval and juvenile fish did not use the artificial macrophyte beds or the natural stands of *Vallisneria* for residential habitat, despite the additional food resources that were provided. This was unexpected and requires further investigation. A major problem with the deployment of artificial macrophyte beds in the Broken River was the high rate of siltation, and in some cases burial, that occurred. The beds were deployed in shallow regions of the river, but where natural macrophytes did not occur, even though light conditions appeared suitable to support submerged plants. While the absence of natural macrophytes can be related to a range of factors, the rapid siltation of the artificial beds suggests that abrasion due to suspended particles carried by the water, and smothering by sediment, may be critical factors in the loss of macrophytes from the Broken River.

4.3. Riparian revegetation

The presence and condition of riparian vegetation play an integral role in maintaining the integrity of riverine ecosystems (Naiman and Décamps 1997; Wiens 2002; Arthington and Pusey 2003; Groffman et al. 2003; Davies et al. 2004; Baxter et al. 2005). For example, it:

- provides shade that moderates water temperature and limits in-stream plant and algal growth,
- supplies organic matter (particularly fallen leaves) that can be used as a source of energy for food webs,
- contributes to the processing or transformation of nutrients, and may reduce the transfer of nitrogen to streams,
- supplies structural habitat (e.g. in the form of wood) for aquatic organisms,
- provides a physical buffer to trap sediments and sediment-bound contaminants carried by surface flow,
- provides reciprocal (two-way) prey subsidies between terrestrial and aquatic food webs.

The width of the riparian zone can vary greatly along a river. The riparian zone can be relatively narrow (metres to tens of metres) along constrained reaches, but may be very wide in unconstrained reaches (tens of metres to kilometres) as the riparian zone includes the floodplain. For example, Molyneux (2003) used vegetation composition as the indicator of the riparian zone along the Acheron River, Victoria. The results indicated that the riparian zone is wide for low-order montane reaches, narrows down in valley mid-reaches (e.g. 4th order streams, where manna gum forms the dominant riparian tree species) and then widens out again in floodplain reaches where redgum is the dominant overstorey species.

In a review of the importance, to fish, of maintaining functioning riparian zones, Pusey and Arthington (2003) emphasised the diverse and important linkages between riparian condition and the conservation of fish communities. An absence of riparian vegetation and increased irradiance can alter the thermal regime and its coupling with the flow regime of streams. This can affect fish populations in many ways, for example by disrupting breeding, decreasing disease resistance and increasing mortality rates. A decline in riparian condition can result in altered habitat and food-web structure, and can promote invasion by alien fish species. Baxter et al. (2005) highlight the important role of the riparian zone in providing prey subsidies that support food webs. For example, terrestrial invertebrates can be an important energy source for fish, and potentially affect foraging habits and local fish abundance. Conversely, emergent adult insects can represent a substantial export of aquatic production to riparian consumers such as spiders, reptiles, mammals and birds. Thus, there may be benefits to both the terrestrial and aquatic ecosystems in restoring the riparian zone.

Riparian revegetation can influence the temperature regime of smaller streams (first and second order) and riparian vegetation cover can affect the growth and distribution of in-stream algae and macrophytes. Mosisch et al. (2001) used artificial substrate in forested and open streams in south-east Queensland to examine relative importance of shading and nutrients on periphyton growth and whether N or P limited algal productivity. They found that shading was the over-riding factor controlling algal biomass, with nutrients playing a relatively minor role, and cited studies that suggested riparian canopy cover providing 60–90% shade was needed to limit filamentous green algae in disturbed streams.

Revegetation of the riparian zone is an important river rehabilitation tool. However, the recovery of streams in response to riparian revegetation may not occur in a linear fashion (i.e. as immediate or continued improvement following replanting). A collaborative project with the CRC for Catchment Hydrology found that revegetation of a section of Echidna Creek, a small upland stream in south-east Queensland, resulted initially in an increase in the maximum daily summer temperatures, and diurnal temperature fluctuations increased in the summer following revegetation (i.e. the temperature regime worsened), before decreasing continuously over subsequent summers towards the desired (and expected) temperature regime (lower and less variable temps; Marsh et al. 2005). The initial increase in stream temperatures following revegetation was due to the removal of woody weeds (blackberry, lantana) and tall grasses prior to revegetation, which temporarily reduced the shading of the stream. While summer temperatures continue to improve, equilibrium in the summer temperature regime is not expected for at least eight years after revegetation as the canopy cover matures and increases the shading of the stream. Similarly, Marsh et al. (2004) found that that suspended sediment yield delivered to Echidna Creek increased by around 100% immediately following revegetation due to disturbance of bank material and clearing of riparian weeds. These results illustrate the hysteresis that may be common in trajectories of change following restoration, and highlight the possibility that the anticipated improvements in stream condition can take some time to emerge. Stream managers need also to be mindful that stream conditions may deteriorate in the short-term in response to localised disturbance associated with the rehabilitation works.

Investigations funded by the National Riparian Lands Program (via Land and Water Australia) found that the amount of riparian vegetation required to alleviate temperature stress in smaller streams can be predicted using physical models that describe heat fluxes in and out of water (e.g. Davies et al. 2004). Approaches such as relating thermal tolerance limits of biota to temperature rehabilitation targets can then be used to identify the length of stream requiring revegetation. However small streams may require continuous riparian cover greater than 100 metres, and preferably greater than 300 metres to significantly dampen stream water temperature — a larger distance than the length of many riparian replantings. Davies et al. (2004) also identified a priority order for revegetation to alleviate temperature stress:

- lower order streams before higher order streams,
- streams with woody vegetation before those with lower density of degraded vegetation,
- streams with north-west aspects before those with south-east aspects,
- streams where soil properties are favourable for vegetation establishment.

Digital elevation maps overlaid with vegetation maps and solar radiation information can provide a powerful tool for identifying priority areas for riparian rehabilitation. For larger rivers, canopy cover provided by riparian vegetation can have less influence on shading and river temperature than for smaller streams. However, the riparian zone still plays an important role by providing material and energy subsidies (e.g. through leaf fall), trapping sediments and contributing to the cycling of nutrients (e.g. denitrification).

The ecological role of the riparian zone can be confounded by urbanization, particularly with the delivery of runoff from impervious areas via stormwater drains. For example, Taylor et al. (2004) recorded large increases in algal biomass in small shaded streams with increasing levels of urbanization. Catford et al. (2005, in review) found that the urbanization gradient still explained a large increase in algal biomass and a shift to filamentous green algae, even under controlled (low-level) light conditions. Such findings led Walsh et al. (2004) to conclude that the ecological benefits commonly associated with the retention or reinstatement of riparian vegetation (over- and under-storey) are likely to be substantially reduced by the impacts of conventional stormwater drainage. For example, increased flashiness in flows and associated channel incision and widening reduces the riparian zone's shading and temperature effects. Also, the delivery of runoff via hydraulically efficient stormwater pipes can greatly reduce the pollutant interception capacity of the riparian zone.

4.4. Rehabilitation in an urban setting — WSUD, the treatment train and habitat reintroduction

While reintroducing structural habitat can result in a positive response by biota, attempts are not always successful. For example, riffle habitat was introduced to a number of streams in urban areas of Melbourne to increase the diversity of macroinvertebrate communities (Walsh and Breen 2001). However, the additional habitat did not result in improvements to species richness, as stream macroinvertebrate communities one year after habitat-introduction remained dominated by a small number of pollution-tolerant species. The macroinvertebrate communities remained in a degraded condition even five years after the riffle habitat was introduced (C. Walsh, pers. comm.). There were occasional instances when pollution-sensitive taxa were recorded in the riffle habitat (and none in any of the control streams without introduced riffle habitat), but none persisted. Altered stream hydrology (e.g. more frequent and larger flow pulses), and more frequent delivery of pollution associated with storm events were considered to be major factors limiting the recovery of macroinvertebrates in the urban streams studied. These results emphasise the importance of identifying catchment-scale factors, such as altered drainage patterns, when considering potential rehabilitation activities.

There is now strong evidence indicating that the direct connection of impervious surfaces (effective imperviousness) to streams via the stormwater drainage system is a major driver of stream condition in urban areas (Hatt et al. 2004; Walsh 2004; Walsh et al. 2004). This means that many location specific rehabilitation actions focusing on in-channel and riparian features are unlikely to succeed in areas receiving water from traditional stormwater drainage.

Improving stream ecosystems in urban areas will require new stormwater drainage and water sensitive urban designs (WSUD) (Victorian Stormwater Committee 1999; Cottingham et al. 2004; Walsh et al. 2004). A main feature of WSUD is the inclusion of measures to reduce the degree of effective imperviousness at scales ranging from single houseblocks to local neighbourhoods and subcatchments (Victorian Stormwater Committee 1999; Lawrence 2001). Measures such as stormwater tanks for reuse, permeable pavements, vegetated swales and constructed wetlands can be included to build up a treatment train to achieve local and catchment objectives for the protection or rehabilitation of stream ecosystems. Models such as MUSIC (Model for Urban Stormwater Improvement Conceptualisation) (CRCCH 2003) are useful tools for evaluating the rainfall–runoff and pollutant patterns in catchments in urban and pre-urban conditions. This, in conjunction with estimates of effective imperviousness for traditional and WSUD drainage designs, can be used to identify the potential benefits of applying WSUD to new developments or retrofitting existing urban areas. However, it may be that ecological rehabilitation of streams in intensely urbanized catchments is unattainable. This may mean re-examining priorities for management, or a focus on other catchments and downstream water bodies (Walsh et al. 2004).

4.5. Species restocking and recovery

River rehabilitation can be a slow process if populations of target organisms are fragmented across the landscape and there are constraints on connectivity and dispersal that limit recolonisation. In some circumstances, the rate of rehabilitation can be accelerated by intervention to reintroduce target species, for example by direct restocking. Restocking is often applied in threatened species management, when trying to overcome bottlenecks to ecosystem recovery or when a rapid rehabilitation response is required. This approach often underpins other rehabilitation such as revegetation of degraded riparian zones.

The Department of Sustainability and Environment (DSE; Victoria) has developed a computer program entitled **E**ndangered **S**pecies **S**urvival **E**xti**n**c**ti**on **A**na**l**ysis (ESSENTIAL) that predicts what happens to a population of fish under various conditions. By providing a range of life history and environmental data, ESSENTIAL allows managers to construct predictive models for threatened species and evaluate scenarios before they are implemented (see www.nre.vic.gov.au/ari/software). The management of threatened fish species has included restocking of fingerlings. This approach may work for stable, non-threatened species where critical life stages are understood and where population growth is most threatened at the early life-history part of the lifecycle. But for species already in decline, populations may also be affected at other life stages. Determining what these impacts are, and how and when they act, is important if species are to recover. While specifically developed for Australian native fish, ESSENTIAL can be used to model any population, terrestrial or aquatic, animal or plant.

Revegetation and fish restocking are common rehabilitation tools, but recent studies provide a cautionary tale and highlight the need for careful consideration before translocating species. Hughes et al. (2003c) describe how translocation of the freshwater shrimp *Paratya australiensis* between subcatchments of the same drainage system in south-east Queensland resulted in the extinction of a local genotype (i.e. a sub-species with a distinct genetic make-up) over a short time period (seven generations). Translocations of species, for example the movement of fish associated with interbasin water transfers, can also promote the spread of diseases such as the EHN virus that affects native fish (DAFF 2004). Practitioners should consider the potential impact of restocking on resident populations, particularly if threatened species are at risk, prior to the translocation of target organisms in order to avoid the extinction of local genetic diversity, and the spread of disease and parasites.

Multiple stressors and impacts affect most river systems that require rehabilitation. This means that actions such as species restocking are unlikely to lead to successful rehabilitation if conducted in isolation. In most instances, restocking will be but one of a number of management actions required to improve ecosystem condition. For example, the Murray-Darling Basin Native Fish Strategy (MDBC 2003) recognises that a number of actions will be required if native fish are to be rehabilitated (Figure 19).

Recovery plans such as those developed for silver perch and freshwater catfish (Clunie and Koehn 2000a,b) also recognise that a multi-pronged approach would be required for the rehabilitation of native fish species.

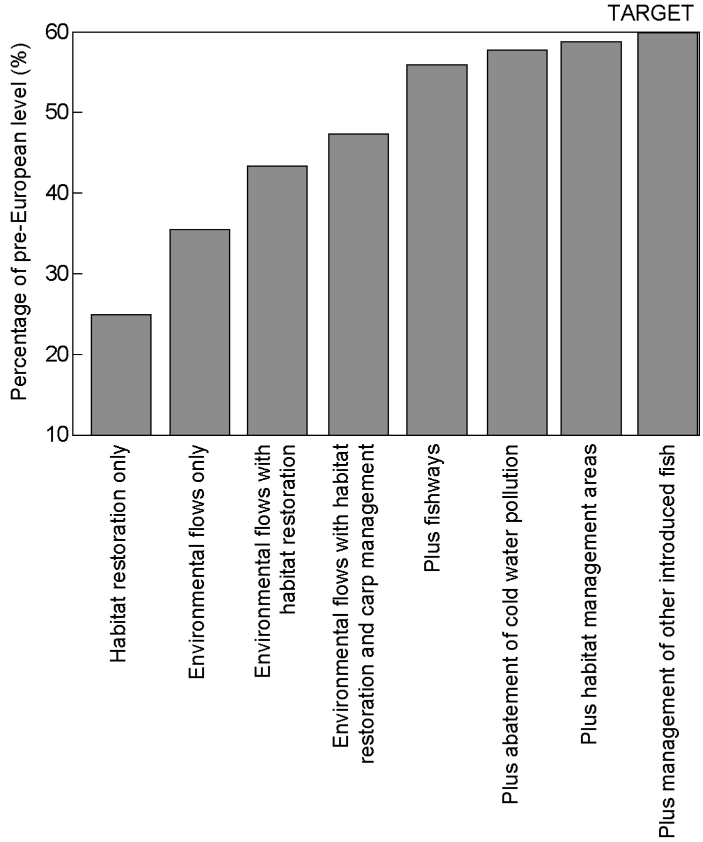

Figure 19. Rehabilitation of native fish communities — expected cumulative impact of all interventions (from MDBC 2003)

4.5.1. Invasive species control

As discussed in chapter 2.3, the recovery of native fish species can be facilitated by the control of invasive species (Lintermans 2000). Lintermans and Raadik (2003) reported on a successful recolonisation by native fish species once introduced trout were removed from sub-catchments of montane streams in the ACT and Victoria. Such eradication and recovery programs are very costly and difficult to implement, and only possible for small-scale interventions.

Carp have been a very successful invader of rivers across Australia, particularly in the Murray-Darling Basin. A national strategy has been prepared to help control the spread and impact of carp (MDBC 2000). The key principles adopted by the strategy include the following:

- carp control should be based on best practice management and underpinned by scientific evidence and pest management principles;
- any practice that makes it easier for carp to move around should be discouraged;
- the presence of carp is not conducive to the enhancement of biodiversity;
- eradication with current technologies is not feasible on a national scale;

- commercial use should not compromise the maintenance and restoration of biological diversity, nor result in the development of *de facto* property rights that may compromise the development of more efficient control methods;
- while recognising existing control measures, the strategy must seek a new vision to progress beyond *status quo*.

Programs such as the *Daughterless carp* project being conducted by the CSIRO may prove successful in controlling this species at large spatial scales (CSIRO 2002) but the implementation of this technology is still some time off.

The key principles outlined in the carp strategy could also be applied to other invasive species. However, experience has shown that once invasive species have established they are very difficult (sometimes impossible) to eradicate. A more cost-effective approach is to prevent establishment in the first place. Investment by state and federal jurisdictions will be required to establish a nationally coordinated program for managing invading species and the damage they cause to freshwater systems (Georges and Cottingham 2002). The program should include:

- an inventory of species already introduced, flagging those species that are or may potentially be invaders;
- compilation of information about potential new species likely to be imported, and ecological risk assessment to assess their potential as invading species;
- agreement with the aquarium fish industry on restricting or eliminating trade in those species of greatest concern;
- rapid response plans that streamline approvals and spell out responsibilities to deal with infestations before they get a secure foothold;
- implementation of the National Carp Management Strategy, including the listing of all strains of carp as noxious species and biodiversity threats in NSW (in line with other states);
- better coordination of government agencies to ensure proper coordination of risk assessments from the perspectives of potential economic damage, disease, and ecological impacts (including impacts on biodiversity);
- research to better understand the role of invaders in the degradation of biodiversity in lakes and streams.

5. Has river rehabilitation been effective?

In general, there has been a lack of monitoring and evaluation of rehabilitation projects in Australia (Curtis et al. 1998; Lake 2001a,b; Brooks et al. 2002; King et al. 2003b; Schreiber et al. 2004). The reasons for this are many (e.g. a lack of resources, no clear rehabilitation objectives, unclear roles and responsibilities of stakeholders), but the end result is that we have forgone numerous opportunities to learn from experience and so inform future management within an adaptive management framework. While we cannot expect that the outcomes of every rehabilitation project will be evaluated, we must still design and implement projects to the best possible standards. Our investment in river rehabilitation will have most value when projects include performance evaluation and the sharing of results, whether the projects achieve their stated objectives or not.

5.1. Criteria for measuring the success of rehabilitation

The characteristics constituting successful river rehabilitation can be relative, depending on social, economic and ecological perspectives. For example, improvements such as landscaping degraded river frontages, a common practice in urban areas, may result in aesthetically pleasing environs that increase social and economic activity but generate few, if any, ecological benefits. The most effective rehabilitation project from an ecological perspective is one that is successful from an ecological perspective, meets stakeholder expectations and provides a learning opportunity so that new insights can be applied elsewhere (Palmer et al. 2005).

Palmer et al. (2005) have identified five criteria by which rehabilitation experiments may be judged as successful or not, from an ecological perspective:

1. The design of a rehabilitation project should be based on a vision for an ecologically healthy river (i.e. a clear rehabilitation endpoint has been identified).
2. The river's ecological condition must be measurably improved.
3. The river system must be more self-sustaining and resilient to perturbations (such as drought or flood) so that only minimal follow-up maintenance is required.
4. No lasting harm should be inflicted on the river system during the construction (implementation) phase of the rehabilitation.
5. Both pre- and post-assessment must be completed and data made publicly available.

The adoption of the above criteria by state and federal jurisdictions and other funding providers will be required if we are to maximise the benefits gained from rehabilitation projects by applying new insights to future river management.

Two main types of evaluation are considered for rehabilitation projects: (i) confirming the works identified for the project are delivered as intended; (ii) assessing the effectiveness of the project (Rutherfurd et al. 2004). While all projects should be required to report on delivery, evaluating project effectiveness can be problematical. The large number and wide geographic spread of rehabilitation projects across catchments means that resources are not available to measure the physical or ecological outcomes of every project. For example, detecting statistically significant changes to parameters in systems with high natural variability may require very intensive sampling programs and considerable cost (e.g. Downes et al. 2002; Rutherfurd et al. 2004). A wise use of available resources will be to monitor and evaluate a few well-designed and resourced experiments to generate learning that can be applied to other similar systems. Dedicated large-scale rehabilitation experiments conducted within an adaptive management framework may offer the best combination of achieving rehabilitation objectives and learning, whether or not the stated objectives are achieved (Cottingham et al. 2001). Practitioners should always seek to implement monitoring and evaluation programs when:

- rehabilitation is to be attempted in a unique setting,
- a new rehabilitation technique is to be trialled,
- the objectives of the rehabilitation include the protection of endangered species,
- the project is an opportunity to showcase river rehabilitation,
- there is a risk that the trajectory of recovery is different to that desired,
- when major new targets have been set, for example ecosystem processes rather than population processes.

Given the rarity of large-scale ecosystem rehabilitation projects in Australia, it will be important to take advantage of opportunities for evaluating their outcomes wherever possible.

5.2. Design of monitoring and evaluation programs

There are many references available that describe the steps involved in designing, monitoring and evaluating river rehabilitation (e.g. ANZECC & ARMCANZ 2000; Downes et al. 2002; Quinn and Keough 2002). For example, Cottingham et al. (2005) proposed the following key steps when designing programs to evaluate the effectiveness of environmental flow regimes when implemented:

- Define the scope (spatial and temporal) of the rehabilitation project and its objectives.
- Define the conceptual understanding that underpins the rehabilitation method/s and the questions (hypotheses) to be tested by the monitoring and evaluation program.
- Select variables to be monitored.
- Determine study design, accounting for the specific activities and location.
- Optimise study design and identify how data are to be analysed,
- Implement the study design.
- Assess whether the specific rehabilitation objectives have been met, and review conceptual understanding and hypotheses.

The same steps may be applied to most rehabilitation projects.

Study design is likely to vary for different rehabilitation projects, depending on the rehabilitation methods being employed and factors such as the availability of control and reference locations that are important elements of before–after control–intervention (BACI) designs that are commonly used for impact assessment[4]. BACI designs include spatial and temporal controls that allow us to isolate the effects of rehabilitation measures from other natural or human-induced changes (Downes et al. 2002; Quinn and Keogh 2002). BACI designs that collect information before and after an intervention, at both control and intervention locations, can be very powerful for inferring causality between a management action and ecological response. However, BACI designs can be difficult to implement in practice. For example, there may not be suitable control locations (those similar to the rehabilitation (intervention) location but without the intervention) or there may not have been the opportunity to collect pre-intervention information. In such circumstances, monitoring designs may be restricted to simply assessing the rehabilitation responses at intervention locations (no controls), or may be restricted to post-intervention assessment at the intervention locations (no before-data). Cottingham et al. (2005) identified a number of potential study designs that may be employed, the level of causal inference, and possible data analyses that could be considered for each. Wherever possible, practitioners should endeavour to collect pre-intervention data to enable analysis before and after the rehabilitation measures are implemented (Downes et al. 2002; Palmer et al. 2005). Advice from experienced statisticians will be helpful when considering the inferences that may be drawn from the study designs available and how best to proceed with data analysis.

The need to consider rehabilitation projects as dedicated experiments was highlighted by Stewardson et al. (2004) when considering how best to evaluate the performance of riparian revegetation as a rehabilitation measure. Post-project evaluation (i.e. analysing results from past examples of a particular rehabilitation measure) was problematical, as historic rehabilitation projects used techniques that are no longer in common practice or were at sites unsuitable for evaluation. Adding monitoring and evaluation to current management activities was also considered. However, this approach ran the risk that changed management needs might see resources redeployed to other issues (i.e. risk to the long-term viability of the performance assessment) or that the effect of the rehabilitation measure could become intertwined and inseparable from other management actions. A well-designed and resourced project dedicated to improving our knowledge about river rehabilitation (habitat reconstruction) has the advantage of assigning a high level of causal inference between the rehabilitation measure and ecosystem response.

5.2.1. Variables to measure

There is an enormous amount of literature on potential variables of a wide range of different stressors in river systems (Downes et al. 2002). For example, ANZECC & ARMCANZ (2000) and Baldwin et al. (2005) provide details on water quality and biological variables and their measurement, along with recommendations for quality control that will ensure that collected data are of high quality. ANZECC & ARMCANZ (2000) also provide a useful

[4] Traditional impact assessment usually focuses on detecting deleterious changes due to disturbance; for example, damage to river biota resulting from the presence of toxicants. Assessing the effectiveness of river rehabilitation interventions has much in common with traditional impact assessment but with an emphasis on the recovery of river systems from disturbance.

checklist that can be used when developing a monitoring and evaluation program (Box 5). Factors to consider when selecting variables include:

- the specific rehabilitation objectives and hypotheses to be explored by the monitoring and assessment program,
- the degree of confidence that changes in a variable imply that there are causal links between the rehabilitation measure and ecosystem response,
- information that may be required to assess and manage risks to the system (e.g. if the rehabilitation activity is incomplete or if it results in some undesirable outcome),
- information to assist communication and foster community engagement (e.g. icon species).

Palmer et al. (2005) emphasise that variables should be easily measured, be sensitive to stresses on the systems, demonstrate predictable responses to stresses, and be integrative. They suggested a number of indicators that could be used to evaluate the five criteria they propose for assessing ecologically successful rehabilitation (Table 2). Some examples of variables used successfully for detecting ecosystem responses in rehabilitation experiments are listed in Table 3.

Table 2. Indicators proposed for evaluating ecologically successful rehabilitation projects (from Palmer et al. 2005)

Success criterion	Indicator
Guiding vision of an ecologically healthy river	• Presence of a design plan or description of desired goals that are not oriented around a single, fixed and invariant endpoint (e.g. static channel, invariant water quality)
Ecosystems are improved	• Water quality improved • Natural flow regime implemented • Increase in population viability of target species • Percentage of native versus alien species increased • Extent of riparian vegetation increased • Increased rates of ecosystem function • Bioassessment index improved • Improvements in limiting factors for a given species or life stage
Resilience is increased	• Few interventions needed to maintain a site • Scale of repair work required is small • Documentation that ecological indicators (see above) stay within a range consistent with reference condition over time
No lasting harm	• Little native vegetation removed or damaged during implementation • Vegetation that was removed has been replaced and shows signs of viability • Little deposition of sediments because of the implementation process
Ecological assessment is completed	• Available documentation of preconditions and post-assessment

Table 3. Example response variables used successfully in river rehabilitation monitoring and evaluation projects (adapted from sources such as Bunn et al. 1999; Bond and Lake 2003b, 2005; King et al. 2003b; Davies et al. 2004; Borg et al. 2005)

Ecosystem component	Response variables	Example river where used	Comments
Fish	Larval fish (occurrence, relative abundance, community composition)	Broken (Vic)	Results suggest good response to creation of slackwater habitat
	Abundance	Murray (NSW), Granite Creeks (Vic)	Results suggest good response to wood addition
Shrimp	Occurrence, relative abundance	Broken (Vic)	Results suggest good response to creation of slackwater habitat
Macro-invertebrates	Abundance, community composition	Broken (Vic)	Results suggest good response to creation of fast-water patches
		Granite Creeks (Vic)	Responses hard to detect due to high inherent variability even in controls
Macrophytes	Riparian vegetation		
Water quality	Temperature	SE Queensland streams	Results suggest decreased temperature with shading greater than 60% for 100 metres or longer
Metabolic activity	Gross primary productivity Respiration	Experimental riffles (USA) Granite Creeks (Vic)	
Functional measures	Amount of POM accumulation and leaf retention at site level	Granite Creeks (Vic)	Both POM accumulation and leaf retention much higher at restored sites. In this system POM is a valuable commodity as the system is severely carbon-limited
Geomorphology	Scour pool formation?	Granite Creeks (Vic) Snowy River (Vic) Williams River (NSW)	

POM = particulate organic matter

Box 5: Checklist for monitoring program design

The *Australian Guidelines for Water Quality Monitoring and Reporting* (ANZECC & ARMCANZ 2000) provide a useful checklist from which to assess the final monitoring study design:

1. Has the study type been made explicit and agreed upon?
2. Have the spatial boundaries of the study been defined?
3. Has the scale of the study been agreed to?
4. Has the duration of the study been defined?
5. Have the potential sources of variability been identified?
6. Are there sufficient sampling stations to accommodate variability?
7. Are the sites accessible and safe?
8. Can sites be accurately identified?
9. Has spatial variation in sites been considered, and have options to minimise this variation been considered?
10. On what basis is the frequency of sampling proposed?
11. Have decisions been made about the smallest differences or changes that need to be detected?
12. Is replication adequate to obtain the desired level of precision in the data?
13. Have the measurement parameters been chosen?
    (a) Are they relevant?
    (b) Do they have explanatory power?
    (c) Can they be used to detect changes and trends?
    (d) Can they be measured in a reliable, reproducible and cost-effective way?
    (e) Are the parameters appropriate for the time and spatial scales of the study?
14. Has the cost-effectiveness of the study design been examined?
15. Have the data requirements been summarised?

6. Some future directions

The science of restoration ecology is relatively new and evolving. This means that there is still much we have to learn about how river systems are affected by and respond to disturbance. How should we best make use of new insights when designing and implementing rehabilitation projects? Some of the major areas requiring further exploration relate to:

- the protection and/or rehabilitation of refugia, especially in the long term (i.e. building resistance and resilience to large-scale disturbances such as floods, droughts and fires into rehabilitation programs);
- the importance of stream metabolism and production, including material and energy subsidies from riparian zones, and their roles in rehabilitation;
- the search for key indicators that are simple and relatively inexpensive to measure, are closely linked to the rehabilitation goals and encompass a range of response times;
- how best to distribute localised restoration efforts at the catchment scale:
    - evidence from genetic studies?
    - better models linking local and landscape scale processes, including population models.

Perhaps the most pressing requirements for rehabilitation practitioners are bio-economic tools that help them with decisions on where and at what scale rehabilitation works should be applied within a catchment (i.e. where to get the best return for each rehabilitation dollar). This is an area of ongoing research in Australia.

For example, Linke et al. (2005) have explored how to separate the effects of habitat degradation from other impacts such as poor water quality, by using a method (Assessment by Nearest Neighbour Analysis — ANNA) that allows habitat condition to be predicted in the absence of human impact. Existing local habitat was described by 21 variables potentially associated with the occurrence of nine fish species found in south-east Queensland streams. The predicted condition could then be compared to actual habitat condition, and degradation could be assessed. Information on the unimpacted habitat state was also thought to be useful for predicting potential distribution of species based on their habitat requirements. The study showed that local habitat variables can be predicted from catchment-scale variables (easily derivable from maps), allowing habitat degradation to be assessed together with an estimation of the potential distribution of fish species in the absence of human disturbance. With additional species-specific research, prediction of pre-disturbance habitat would be useful in explaining the decline of species in certain locations and the rehabilitation measures required to reverse these declines.

Research on riparian zones in upper catchments of the Murrumbidgeee River has shown that riparian vegetation composition can be predicted successfully based on factors such as flooding, geomorphic landform, soil and climate (Evans 2002). The vegetation data collected can be used to develop predictive models using the mathematical methods developed for river health assessment based on macroinvertebrates (AUSRIVAS). The incorporation of this information into AUSRIVAS-type models allows the species likely to occur at a site to be predicted even if there is no native vegetation present, making it a valuable tool for stream managers to use when a stretch of river is to be rehabilitated. Similar information could also be sourced, albeit at a coarser resolution, using existing maps of ecological vegetation classes (EVCs).

Adaptive management will be well-served in the future by the consolidation of central repositories of learning and information on restoration ecology and the outcomes of rehabilitation efforts (including environmental flows). Such repositories exist for water quality information (e.g. http://www.vicwaterdata.net/vicwaterdata/home.aspx) and other aspects of river management (e.g. http://www.rivers.gov.au). It will be important that the lessons learnt from rehabilitation projects are captured in such repositories.

The development and application of new and innovative bio-economic models and tools that can be used to set river rehabilitation priorities will be an important feature of the new eWater CRC that succeeds the CRCs for Freshwater Ecology and Catchment Hydrology. It is expected that the new CRC will be a repository of information for interested rehabilitation practitioners in the future.

7. Further reading

In addition to the references provided below, information on river management and rehabilitation can also be obtained from the River landscapes web site: www.rivers.org.au/.

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8. APPENDIX 1 — example River Styles

Table 4. Example River Styles and associated rehabilitation activities (D. Outhet, DIPNR, pers. comm.)

Basic Style	Description/Behaviour	Responses if Disturbed	Options: Preserving Intact	Options: Rehab/Control
Confined by Valley				
Headwater	V-shaped lower order streams slowly eroding bedrock uplands.	Gullying if on colluvium or deep weathering.	No works usually needed.	No works usually needed.
Gorge	Deep bedrock channel with no floodplain. Locked in place.	Minimal changes. May have short term sediment slug.	Prevent excess sediment supply.	Reduce sediment source. Slug will flush when sources reduced.
Chain of Ponds (2 sub-Styles)	Swampy depressions joining ponds or pools. No channels.	Fill flushed out yielding large sediment volumes, becomes low sinuosity type.	As above.	As above.
Partly Confined by Valley				
Bedrock Controlled (3 sub-Styles)	Bends fixed in place by bedrock valley walls or terraces. Floodplain pockets on most inside bends.	Channel expands on inside bends. Floodplain stripping. Bed aggradation/degradation.	Preserve vegetation and LWD.	Revegetation and LWD installation.
Planform Controlled Anabranching	Anabranch abuts valley margins. New anabranches formed by avulsions.	Increased avulsing.	Preserve vegetation and LWD. Remove or realign major channel blockages.	Floodplain and channel revegetation and LWD installation.
Planform Controlled Anastomosing	Myriad anabranching channels separated by floodplain. Some abut valley margin. New anabranches formed by avulsions.	Increased avulsing.	Preserve vegetation and LWD. Remove or realign major channel blockages.	Floodplain and channel revegetation and LWD installation.
Unconfined by Valley				
Confluence Wetland	Long pool ending at a low natural levee across the stream where it joins a larger stream.	Pool drained by levee break or filled by sediment slug.	Preserve confluence levee. Control sediment sources.	Reconstruct levee. Extract sediment.
Anabranching (2 sub-Styles)	Parallel floodplain-separated multiple channel network with anabranch flow proportion variable.	Accelerated migration and expansion of anabranches, choking of main channel.	Preserve all vegetation and LWD. Control sediment sources.	Log/debris jams at upstream end of anabranches. Revegetate anabranches.
Anastomosing	Myriad anabranching channels separated by floodplain. New anabranches formed by avulsions.	Increased avulsion.	Preserve all vegetation and LWD.	Log/debris jams at upstream end of anabranches. Revegetate anabranches and floodplain.
Bank Confined (2 sub-Styles)	Banks fully locked in place by cohesion and/or vegetation and/or bank protection works.	Minimal response.	No works usually needed except preserve vegetation where confining.	No works usually needed except revegetation where confining.