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Recovery pathways after flow restoration in rivers

B J Robson, B D Mitchell & E T Chester

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Waterlines

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Waterlines

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Water-dependent ecosystems in Australia

Water-dependent ecosystems include wetlands, floodplains, riparian areas, estuaries and springs. They provide many important services, including provision of good quality water for irrigation and domestic use, habitat for fish and other aquatic fauna and flora, removal of wastes and contaminants, and aesthetic, cultural and recreational benefits. Without adequate and timely water, these ecosystems lose their capacity to provide such services. In some cases, the losses may be irreversible; in others, they may be difficult and costly to reverse. Under current conditions, many significant water-dependent ecosystems are under threat.

Commitments under the National Water Initiative to water-dependent ecosystems

Striking a balance between water for consumptive uses and water for ecosystem health—so that environmental, social and economic outcomes are optimised—is integral to the National Water Initiative Agreement. Water planning is the fundamental means for achieving this balance. Overallocated water systems need to be restored to environmentally sustainable levels of extraction; in other systems, crucial environmental assets and ecosystem services need to be protected.

The National Water Initiative calls for:

- environmental water to enjoy the same security as water for consumptive uses
- environmental water managers to be established and equipped with the necessary authority and resources
- water-market and trading arrangements to protect the needs of the environment
- environmental water to be included in water accounts and audited
- periodic assessments of river and wetland health to be conducted so that adaptive management can be undertaken on an evidence basis.

Progress on water-dependent ecosystems

The National Water Commission's 2007 First Biennial Assessment of Progress in the Implementation of the National Water Initiative found that all states had made statutory provision for water to meet environmental and public benefit outcomes within water plans, however:

- overallocated systems were not always adequately identified
- environmentally sustainable levels of extraction were poorly defined
- there was considerable variability in the quality of the science underpinning water plans
- in many cases, the trade-offs between environmental and consumptive uses were not transparent
- there was often a lack of specificity in the environmental outcomes.

The Commission considers that the protection of threatened water-dependent ecosystems, including the recovery of overallocated systems, continues to be a major challenge in implementing the National Water Initiative Agreement.

The Commission's water-dependent ecosystems activities

Over the past three years, the focus of Commission activities has been on filling knowledge gaps and promoting science to support good decisions about environmental water. These activities have included:

- commissioning the synthesis of existing knowledge about specific aspects of water-dependent ecosystems and their management
- commissioning scoping studies to identify critical knowledge gaps and provide guidance on research priorities
- providing grants to research programs addressing issues such as the formation of acid sulfate sediments, water requirements for native fish populations and the use of aerial surveys of waterbirds as indications for wetland health
- supporting environmental water managers by establishing a 'community of practice' where they can share experiences
- undertaking trials of a national framework for assessing river and wetland health (FARWH), with the intention that an agreed framework will be delivered in 2011.

Future directions for water-dependent ecosystems

The Commission will continue to build on these activities. However, improved knowledge alone will not ensure that environmental outcomes are achieved. The Commission has therefore adopted the following six priorities to guide future work involving the management of water-dependent ecosystems:

1. *Help develop and implement national guidelines and procedures for determining environmentally sustainable levels of extraction of water*. A nationally agreed method will expedite the formulation of water plans that protect water-dependent ecosystems and include a pathway to recover overallocated systems. The methods will include guidelines for establishing clear environmental outcomes.
2. *Pursue an agreed national inventory of overallocated water systems together with commitments by governments to return them to sustainable levels of extraction*. Identifying overallocated systems and recording agreed actions to recover the water needed to restore sustainability is central to achieving environmental outcomes contained in the NWI.

3. *Improve the security of environmental water.* In spite of the legislation now passed in all jurisdictions, environmental water allocations often lack specificity and there is uncertainty around the status and security of environmental water holdings.
4. *Support more effective management of environmental water.* There are many shortcomings in the governance and operations of environmental water managers. Statutory empowerment, funding, skills and access to science, data and best practice guidelines all require urgent attention. The development of a national community of practice in environmental water management is an important initiative that will support these water managers.
5 *Strengthen the role of adaptive management of environmental water.* Recent work commissioned by the Commission[1] showed there is a deficiency in monitoring and reporting on plan implementation. This is a significant weakness when coupled with gaps in ecological knowledge and the occurrence of climatic conditions outside the planned-for circumstances. More systematic monitoring and reporting is essential to enable the water management regime to be adapted intelligently in the light of experience.
6. *Implement the Framework for the Assessment of River and Wetland Health.* While the Commission will continue to support the implementation of the Framework for the Assessment of River and Wetland Health, its successful adoption rests with the parties to the National Water Initiative Agreement.

By pursuing these priorities, the Commission will play its part in promoting the enduring objective of the National Water Initiative to manage water-dependent ecosystems to best effect. We urge the parties to the National Water Initiative Agreement to do likewise.

National Water Commission
1 September 2008

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[1] Hamstead M, Baldwin, C and O'Keefe, V 2008, 'Water allocation planning in Australia – current practices and lessons learned', Waterline Occasional Paper No. 6, April 2008, National Water Commission.

Executive Summary

This Waterlines report is part of a series of papers commissioned on issues relating to Australian aquatic ecosystems. These Waterlines reports will contribute to improved environmental water management by stimulating discussion, synthesising current thinking, identifying knowledge gaps, and highlighting areas that warrant further investigation.

Aspects of river flow regimes, termed flow components, may be restored with an overall aim of rehabilitating some aspects of river ecology. Flow regime restoration can involve releases from storages, or the regulation of direct water extraction, or even the reinstatement of cease-to-flow periods in seasonally flowing rivers. Flow restoration is guided by environmental flow objectives, and a variety of these are cited in the literature. Some are highly specific ecological objectives, such as triggering fish spawning or waterbird breeding. Some are geomorphological objectives, such as restoring particular channel forms, where ecological effects are expected to cascade from this restoration of physical structure. Still others are large-scale, long-term declarations of whole-river objectives, such as the 'Healthy Working River' objective for the River Murray. Clearly, as these objectives vary, so will the number and types of strategies used, the spatial and temporal scaling of expected ecological outcomes, and the recovery pathways that produced these outcomes. The aim of this document was to identify and describe current work on recovery pathways after flow restoration in rivers so that knowledge gaps and directions for future investment could be identified.

We reviewed the scientific literature on recovery pathways after flow restoration in rivers and found many examples of the restoration of different types of flow components that had led to measurable ecological improvements. However, the biophysical pathways that led to these improvements had rarely been explicitly identified. That is, post-release monitoring could identify ecological outcomes, but it is often not designed to measure the ecological and physical processes that lead to these outcomes. There is a need for further research that focuses on these processes and how they are initiated and supported by flow restoration. It is also important that more of the outcomes of flow restoration studies (both monitoring and process-based studies) are published in the scientific literature where they are peer-reviewed and can be broadly accessed, in perpetuity, to enable restoration science to progress. In particular, this would assist in determining whether restoration of particular flow components results in the same recovery pathways in different places and therefore in developing generalisable flow release strategies.

The literature indicated that recovery pathways after flow restoration in rivers occur in two ways: by triggering direct biological responses in plants and animals and by initiating a sequence of ecological processes. Importantly, while flow releases directly trigger responses such as germination or spawning, the ultimate success of these ecological processes depends on other slower processes being initiated and sustained. That is, processes involving food webs and nutrient cycles, for example, are needed to support newly recruited plants and animals so that they develop into reproductive adults. It is also important to note that, when a particular flow component is restored, not all of the ecological processes depending on it will necessarily be restored because of the interacting effects of other disturbances (for example, presence of exotic species).

Different kinds of flow releases may initiate or support recovery pathways, and some options require more consideration and research in the Australian context. In particular, dam removal and the reinstatement of cease-to-flow periods have received little attention. Further innovation may also be possible, such as the strategic use of hypolimnetic releases to limit the impact of increased water temperatures resulting from climate change. Several case studies now show that recovery pathways in rivers will not be initiated or sustained if releases are too small in volume or duration, or if they are not repeated often enough. Environmental risks resulting from flow restoration are generally dealt with fairly briefly in the literature. They include: increased waterlogging, interactions with saline or sulfidic soils, blackwater events (in floodplain rivers), and reduced water quality from hypolimnetic releases.

We present an outcomes-based conceptual model, the Recovery Cascade Model, for predicting recovery pathways after restoration of components of flow regimes. The model is adaptable for restoration of different or multiple components. It aims to clarify the outcomes delivered at each stage of the process and how these outcomes can be blocked by barriers, and so fail to accurately represent the rates of ecological processes. The model presents the recovery processes as sequential and stepwise, although in reality, ecological processes occur partially in sequence and partially in parallel. A sequential model makes it easiest to evaluate outcomes and barriers to restoration. In all cases, more observations are needed of real recovery pathways after all types of flow restoration in rivers, and the model will need to be adapted to include this new information. The model presented here can be adapted for existing situations by individual managers for their own systems. Specific environmental risks, such as those related to water quality or salinisation should be incorporated on a case-by-case basis.

In conclusion, recovery pathways are described for some types of flow restoration in a few cases, but we are far from evidence-based documentation of recovery pathways. Further research investigating the biophysical processes following flow restoration is required. In planning recovery trajectories for environmental flow, there needs to be an audit and full evaluation of the interacting disturbances that could strengthen or dissipate the effects of the flow. Designs for monitoring responses to environmental flow releases are improving rapidly, but process-based studies of recovery pathways need further development. In particular, recovery pathways after the restoration of multiple flow components need to be studied and potential barriers to restoration need to be identified and consideration given to how they can be overcome.

The purpose of this document

The purpose of this review is to identify and describe current work on recovery pathways after flow restoration in rivers so that knowledge gaps and directions for future investment can be identified.

After briefly defining some terms, we delineate various flow components that are amenable to restoration. We then review international literature on flow restoration, grouped according to which flow components were restored and focusing on studies that contribute knowledge about recovery pathways after flow restoration. Following the review, we consider the potential environmental risks of flow restoration. We then present a model that can be used as a tool for conceptualising flow restoration strategies based on the outcomes desired and the barriers to recovery that might exist. Various versions of the model are presented and described to illustrate its use. Note that this model assumes that environmental flow releases will be of sufficient magnitude to cause a response in the river system. We then conclude the document by identifying knowledge gaps and directions for further research.

It is not the purpose of this review to establish the need for environmental flow, to describe the impact of flow regulation on rivers, or to detail the range of environmental flow objectives or types of flow releases (or release rules) that are used or possible. It is also not the purpose of this document to present a comprehensive review of the ecological or physical responses recorded by monitoring environmental flow releases. Post-release monitoring results vary in their capacity to provide information about the pathways by which river systems recover. Depending on the circumstances of the river system and the monitoring program, the results could provide evidence of a response to a flow release without indicating the pathway by which a particular response developed. The literature review presented here focuses on empirical studies that do indicate recovery pathways, as well as some conceptual and related studies that focus on pathways, rather than on environmental water allocation *per se*. Therefore, this document should not be regarded as presenting a summary of the outcomes of environmental flow releases, nor is its purpose to justify their use. This document assumes that environmental flow releases can be an effective restoration tool, and it focuses instead on the pathways by which restoration can occur.

Terminology and definitions

'Restoration ecology' and 'environmental water' are both minefields of terminology. We will define our use of these terms in this review, rather than discussing the various terms and the ways they are used.

Restoration is presently used as a general term for rehabilitation or restoration of ecological or other environmental components and we will use it as such. Hamstead (2007) discusses the terminology around environmental water and environmental flow and concludes that a more specific terminology, that incorporates environmental objectives, is required. He suggests the terms 'environmental flow objective' be used for when targeting a specific environmental outcome, which would be be achieved by one or more 'environmental flow strategies' (Hamstead 2007).

We will use his terminology in this report because the idea of targeting flow restoration activities for specific outcomes meshes with the idea of particular pathways for river recovery.

1. Introduction

1.1 Impact of flow regulation on rivers

Before considering how types of flow releases relate to recovery pathways, we will briefly review the impact of flow regulation on rivers. Worldwide, these impacts have been extensively studied and the results of these studies reviewed (Arthington and Pusey 2003; Gippel and Collier 1998; Kingsford and Auld 2005; Lloyd et al. 2004; Pedroli et al. 2002; Poff et al. 1997; Poff and Hart 2002; Richter et al. 1997; Richter et al. 2003). The main impacts are: loss of native species and increased spread of exotic species (Poff et al. 2007), loss of lateral connectivity (to floodplains), loss of longitudinal connectivity (between upstream and downstream reaches) (Poff and Hart 2002), declining water quality (Poff and Hart 2002), homogenisation of flow regimes among rivers that were once distinct (Poff et al. 2007), aseasonal flooding, aseasonal drought, anti-drought (*sensu* McMahon and Finlayson 2003), altered geomorphological processes (Lloyd et al. 2004), and even the loss of entire ecosystems (Arthington and Pusey 2003). In essence, alterations to natural flow regimes are bound to have pervasive effects on stream biota and processes because flow is the primary structuring force in rivers (Hart and Finelli 1999; Pedroli et al. 2002). Historically, management approaches to protecting rivers have focused on water quality and minimum flow requirements, ignoring the importance of flow variability (especially seasonal flow variability) in rivers (Poff et al. 1997; Richteret al. 1997; Souchon et al. 2008). Such an approach is clearly inadequate for a continent with some of the world's most naturally variable river flow regimes (Gippel and Collier 1998; Puckridge et al. 1998) .

River flow regimes can be characterised by flow components, and various classifications have been described. For example, Poff et al. (1997) listed flow magnitude, duration, frequency, timing and rate of change in flow. Alternatively, other classification schemes identify different sections of the hydrograph. One example comes from the Victorian River Health Strategy, which defines cease-to-flow, low flow, freshes, high flow, bankfull flow and overbank flow (DNRE 2002). River regulation could affect some or all of these components, and similarly, flow restoration might restore all or only some components of the natural flow regime. These impacts of both regulation and flow restoration can be expected to have specific ecological impacts. For example, increased magnitude and frequency of flooding could cause 'wash-out' of individuals, disrupt species' lifecycles and thereby cause the loss of sensitive species (Poff et al. 1997). Artificially stable flow, perhaps the most common effect of river regulation, removes both flow peaks and cease-to-flow periods. This removal disrupts the life cycles of species whose life history stages (such as breeding) are triggered by or require high flow, inundated floodplains or periods of drying. While these types of responses to flow regime alteration have been documented extensively, far less often has the response to restoration of flow been monitored and the pathways by which recovery occurred identified. In general, the assumption is made that, if a particular flow component is restored, all the ecological processes depending on it will also be restored (Poff et al. 1997). However, impacts such as the loss of species means that this is not necessarily a sound assumption (see Lake et al. 2007).

1.2 Environmental flow objectives

A variety of environmental flow objectives are cited in the literature (not reviewed here). Some are highly specific ecological objectives, such as triggering fish spawning or waterbird breeding. Some are geomorphological objectives such as restoring particular channel forms, with ecological effects being expected to cascade from this restoration of physical structure. Still others are large-scale, long-term declarations of whole-river objectives, such as the 'Healthy Working River' objective for the River Murray. Clearly, as these objectives vary, so will the number and types of strategies used, as will the spatial and temporal scaling of expected ecological outcomes and the pathways by which recovery occurs.

1.3 Types of flow releases

Before considering the pathways for recovery after flow restoration in rivers, it is necessary to consider what types of flow restoration there might be. Only then can we to predict potential environmental consequences. As an example to form the basis for discussion, we use the classification scheme for flow regime from the Victorian River Health Strategy (DNRE 2002). It identifies three spatio-temporal scales in flow variability: daily (river height), seasonal (patterns of freshes, floods and cease-to-flow) and interannual (supraseasonal droughts and flooding events with a greater than annual average return time). It also identifies six components of flow regimes that are important for river ecology (DNRE 2002): cease-to-flow, low (or base) flow, freshes (small spates or flushing flow), base wet-season flow, bankfull flow, and overbank flow. Environmental flow release rules have been based on the restoration or maintenance of one or more of these components and some examples are presented in Table 1. We will use these components of river flow regimes as a basis for discussion because they were designed with ecological processes in mind. However, it is important to note that there are numerous other classification schemes for biologically-relevant flow components used in Australia and overseas that could also be applied.

Some examples of the restoration of particular flow components and their environmental flow objectives

Cease-to-flow events dry out shallow habitats and substrata and can create chains of pools, isolated pools or completely dry riverbeds, depending on riverbed morphology. While many Australian rivers naturally have cease-to-flow events annually, some are now drier for longer due to drought and water extraction, and this could exacerbate the effects of other disturbances such as salinisation (Lind et al. 2006). Environmental flow, termed sustaining flow, has been used in this situation to maintain a layer of low-salinity, oxygenated water in the upper levels of river runs and pools (Lind et al. 2007), to moderate water temperatures for fish migration (Tiffan et al. 2003) and to maintain depth (SCA 2006) (Table 1). So-called 'passing flows' have also been proposed to top up refuge pools and maintain acceptable water quality. They can be achieved by making weirs translucent, so that a proportion of streamflow continues downstream (passes the weir). In contrast, other rivers that once dried out annually now do not dry out because irrigation water runs through them during the dry season (McMahon and Finlayson 2003). This reduces seasonal fluctuations in biodiversity and might contribute to the storage of sulphides in formerly ephemeral floodplain wetlands (Lamontagne et al. 2006). In theory, cessation of flow can be a planned part of flow management (SKM 2002). This has been planned for some Australian rivers, for example the Shoalhaven River in NSW (SCA 2006). Average dry-season low flow occur in perennial rivers and these could also be a planned part of flow management for rivers used as dry-season irrigation channels.

Freshes are defined as flow greater than the median for that time of the year. They have a duration of days and act as triggers to ecological processes (DNRE 2002; SCA 2006) or refresh water quality by topping up or mixing river pools. Base wet-season flow is elevated above dry-season flow, inundating more of the riverbed while still being lower than bankfull flow and lasting for days to weeks. This flow can mobilise sediment, inundate larger areas of potential habitat, and connect in-channel habitats—thereby permitting migration of aquatic fauna (DNRE 2002). Bankfull flow is those larger flood events that occur at least annually; they are channel forming, moving bed sediments (Jeffres et al. 2007). Overbank flow achieves this, and also connect rivers to floodplains (DNRE 2002), stimulating a wide range of ecological processes and exchanges of materials, filling wetland and maintaining floodplain and riparian vegetation (for example, Hou et al. 2007; Lake et al. 2006). These are the largest flooding events and occur annually or less often. For example, environmental water allocations for the Barmah-Millewa Forest on Australia's Murray River have been stored without release for a few years and then used to prolong naturally-occurring overbank flow (AJ King et al. 2008; Ward 2005).

As well as the temporal duration of these flow components, their effects are expected to scale spatially. For example, passing flow could be targeted at maintaining specific refuge pools over a few hundred metres below a weir. In contrast, overbank flow could aim to influence an entire river system and its floodplain. Therefore, there is an implied scaling of the ecological responses expected from different environmental flow strategies.

Table 1: Natural flow components, types of environmental flow allocations and example rivers, with references (examples only, not a comprehensive list)

Flow component to be restored	Flow regime prior to EWA	Environmental flow strategy	Environmental objective	Rivers where applied	Example References
Cease-to-flow	Prolonged cease-to-flow	Sustaining flow	Maintain water quality and inundated habitat including pools	Glenelg and Wimmera rivers, Victoria, Australia	(Lind et al. 2006; Lind et al. 2007)
Cease-to-flow	Cease-to-flow absent	Cessation of releases	Restore natural seasonal variation	Broken-Bosey system, Vic	P.S. Lake (pers. comm.)
Dry-season low flow	No low flow in dry-season	Reduce releases	Restore natural seasonal variation	Rhone R. France; Tallapoosa R Alabama	(Lamouroux, et al. 2006; Travnichek, et al. 1995)
Freshes	Freshes absent	Short-term flow releases, passing flow	Maintain refuge pools Trigger specific ecological processes, flush sediment	Victoria Range streams, Grampians NP., Hunter R Aus.	(Hancock and Boulton 2005)
Base wet-season flow	No increase from dry-season low flow in wet season	Flow releases	Inundate specific habitat (e.g. woody debris for fish spawning)	Rhone R. France; Tallapoosa R Alabama	(Lamouroux, et al. 2006; Travnichek, et al. 1995)
Bankfull flow	Bankfull flow absent	Flow releases	Increase fish diversity, restore bed morphology	Colorado R, San Juan R USA, Switzerland	(Propst and Gido 2004; Robinson, et al. 2003; Shannon, et al. 2001)
Overbank flow	Overbank flow absent	Flow releases	Reconnect floodplain Fill wetlands Trigger waterbird breeding Improve health of floodplain trees	Murray R Aus., Midwest USA	(Hillman and Quinn 2002; Zhang and Mitsch 2007)

EWA = environmental water allocation

1.4 Design of studies monitoring responses to environmental flow releases and describing recovery pathways

Generally, published studies of restoration practices are rare (Souchon et al. 2008), and there is an urgent need for a more structured approach to evaluating restoration activities (Brooks and Lake 2007). Methods for assessing the ecological response to environmental flow are discussed by Chessman and Jones (2001) and Downes et al. (2002) and were reviewed by King et al. (2003). There is a variety of suitable methods. For example, multiple before–after control–impact designs are ideal for demonstrating that ecological responses arise from flow releases, but they require data from before flow restoration and comparable, unrestored sites to use as controls. The latter are often difficult to find. Alternatively, a 'multiple lines of evidence' approach can be used to draw together evidence from a variety of sources (see Downes et al. 2002; Lind et al. 2007). Sound design for both assessing ecological responses to flow regulation and quantifying recovery pathways (ecological processes) require data from before flow restoration occurred and afterwards. Importantly, data from multiple years enables the impacts of flow releases to be disentangled from annual climate variability and also allows the progress of recovery pathways to be tracked. Too few studies have observed recovery processes over long time periods (decades) or to their end point.

More commonly, there have been studies of the effects of river regulation by comparing unregulated and regulated streams and rivers or by comparing conditions before and after regulation (Lloyd et al. 2004). The former use space as a surrogate for time, that is, they compare places with different regulatory histories rather than the same rivers before and after regulation (or flow restoration) and the latter document the impact of regulation. While common, such studies show the effects of regulation but do not indicate what to expect from flow restoration: restoring a lost flow component does not necessarily mean that the associated lost ecological functions will also be restored. This is because the consequences of flow regulation or restoration are unlikely to be additive. For example, local extinction of species or the presence of exotic species could well prevent the river returning to its original pre-regulation condition (see Lake et al. 2007 for a conceptual overview of this). Therefore, we have not cited such studies here. Studies that purely describe ecological processes can contribute to understanding how recovery occurs. Some such studies were included here, where relevant to recovery after flow restoration.

Environmental flow studies fall into three categories:

- examination of the impacts of actual environmental flow release manipulations
- studies of the effects of components of the flow regime that occur naturally (such as the impact of floods arising naturally and not from environmental releases)
- studies proposing models for estimating environmental flow requirements or predicting their outcomes.

In the review below, we select from the former group of studies as containing the greatest empirical evidence of responses to environmental flow and, therefore, probable recovery pathways. We have also used some of the literature in the second group, where relevant to recovery pathways. We have deliberately avoided papers where models are proposed or tested on simulated data—that is, where there are no empirical data. There is a plethora of such studies (Arthington and Pusey 2003), but they do not inform an examination of recovery pathways after flow restoration. In particular, many of these studies do not go beyond methods for estimating likely flow requirements. For similar reasons, we have avoided review articles or papers solely devoted to conceptual models, unless those models focus on recovery pathways.

Studies of the impacts of environmental flow release manipulations generally take the form of environmental sampling pre-and post-flow release, but effects can be difficult to detect because releases are designed to mimic natural flow events (Lind et al. 2007). Background fluctuations in flow owing to droughts or higher-than-average rainfall can mask the effects of environmental flow releases; and downriver from release points it can be difficult to determine how much of the flow arises from the environmental release, especially if tributaries are ungauged. Pre-restoration data are often limited, making it difficult to evaluate the effects of restoration strategies (AJ King et al. 2003).

One important limitation to progress in environmental restoration is that the monitoring of many restoration attempts is not reported in peer-reviewed, perpetually accessible formats, such as books or scientific journals. Instead, the outcomes of many restoration efforts are reported only in brochures, fact sheets or other forms of report that are not widely available and do not contain sufficient details for the degree of scientific rigour of the data collection and analysis to be determined. If, in addition to this style of report, the design, results and conclusions of these studies were also presented in peer-reviewed, perpetually accessible formats, they would be readily available to scientists and practitioners and would contribute to improvements in both restoration science and practice. Evidence for the advantages of particular methods over others would accumulate and unnecessary duplication would be avoided. In addition, peer-reviewed scientific journals provide a quality-control process that ensures a minimum standard of scientific rigour and reliability.

The limited range of material that meets the criteria of either publication in a peer-reviewed journal or a published report where the study design, data analysis and results are fully presented (and therefore assessable for scientific rigour) has imposed constraints on the literature review. In particular, identifying and pursuing unpublished reports is extremely time-consuming. We suggest that a nationwide review of the outcomes of the many flow-release experiments that have occurred, both published and unpublished, would be both useful and timely and should be undertaken. However, this is not the purpose of the review presented below.

In summary, the criteria used to select reference material for the review below were peer-reviewed journal articles where:

- empirical data were collected using multiple before-after control-impact (mBACI) or similarly rigorous designs to assess the response to environmental flow and these designs consisted of more than single occasion (snapshot) post-release monitoring
- recovery pathways were documented or discussed
- conceptual models for recovery pathways were described
- the relationship between naturally occurring flow components and ecological pathways was described.

Information in other formats, such as published and unpublished reports, was also used where there was evidence that empirical data were collected using mBACI or similarly rigorous designs to assess the response to environmental flow, and these designs consisted of more than single occasion (snapshot) post-release monitoring.

2. Recovery pathways after flow restoration—a review

2.1 Cease-to-flow

Relatively few environmental flow studies consider this aspect of the flow regime, which is common in Mediterranean, semi-arid and arid climate regions (Gasith and Resh 1999). In the Western Victoria's Wimmera and Glenelg Rivers (Australia), regulation has limited bankfull and overbank flow but also increases the duration of cease-to-flow events, often far beyond natural levels (Lind et al. 2007). Sustaining flow has been used in these rivers to top up river pools and maintain an upper layer of low-salinity, oxygenated water and to provide some flow in runs (Lind et al. 2007). These are largely effective in supporting invertebrate assemblages in the Glenelg River, but they are less effective in the Wimmera River, which has higher inputs of saline groundwater and a longer section of dry channel (Lind et al. 2007; Westbury et al. 2007). A more spatially-intensive study of the lower Wimmera River during and after a single summer-autumn sustaining flow showed positive benefits of reduced salinity and improved macroinvertebrate indices at the majority of sites (Westbury et al. 2007). However, in the absence of a release the following summer-autumn, these improvements were not sustained at most sites. This suggests that while recovery may have occurred rapidly, the processes involved were relatively short-term and required annual flow releases.

Recovery in the Glenelg River consisted mainly of maintaining biodiversity and arresting its decline, rather than positive improvements. For this reason these releases were termed 'sustaining flow' because they were aimed at maintaining present condition and preventing further degradation rather than initiating recovery. However, this work was guided by a conceptual model (Mitchell et al. 1999) and examined the relationship between flow and ecological responses across a nested hierarchy of spatial scales (Lind et al. 2006), so it does provide information regarding processes sustaining recovery that has been incorporated into the Recovery Cascade Model described below.

Dryland rivers fluctuate between long periods as a sequence of perennial pools separated by stretches of dry river bed and shorter periods of flooding with extensive connectivity among pools and with the floodplain (Bunn et al. 2006). During each of these states, quite different ecological dynamics occur. For example, individual pool food webs are supported by algal productivity during prolonged dry periods (Bunn et al. 2003), but directly after floodwaters recede, carbon arising from the floodplain (via fish mortality) fuels food webs (Burford et al. 2008). During large floods, a variety of floodplain-derived food sources are used, including benthic algae, detritus and plankton (Bunn et al. 2006). For dryland rivers experiencing few cease-to-flow conditions due to water resource development, these temporal changes in food web structure would be disrupted because the channel tends to be continuously connected but with few overbank floods or cessation in flow (Bunn et al. 2006). Reinstatement of cease-to-flow conditions would lead to a recovery pathway initiated by the re-establishment of benthic algal mats in pools. This carbon would then support pool food webs, which would reassemble and, over time, support a diverse but relatively low biomass of fish and other higher-order consumers (Burford et al. 2008).

2.2 Dry season or wet season low flow

Few experiments with either wet or dry season baseflow exist, although several rivers have fixed minimum flow releases year-round, giving a relatively constant daily discharge regardless of season. One example, in the Tallapoosa River, Alabama showed that increased year-round baseflow doubled the diversity of fish species, mainly by increasing the number of species preferring faster flow (Travnichek et al. 1995). A similar study, in the Rhone River in France, saw increases to baseflow throughout the year for four years with a concomitant increase in fish species that preferred faster and deeper flow (Lamourouxet al. 2006). The pathways for this recovery are not shown in these studies, but clearly included migration of the new (fast-flow dependent) fish species into these sections of the rivers. This means that there were few barriers to fish movement and that there were source populations of these fish species within the river networks.

Fish recruitment may also have been improved by modifications to hydropower generation flow releases in the Roanoke River, North Carolina, USA (Richter et al. 1997). These fish recruit in spring, when river flow prior to regulation were high but relatively constant, whereas hydropeaking created a rapidly fluctuating discharge. Reducing the frequency of low flow and rises and falls in the hydrograph restored more normal springflow conditions that were associated with increases in the size of fish populations (Richter et al. 1997). This form of recovery suggests a very direct (short) recovery pathway, where flow requirements for breeding are met leading to increased breeding success.

Releases from numerous dams on the Tennessee River (USA) have been increased to elevate year-round baseflow along with oxygenation of hypolimnetic release water (Bednarek and Hart 2005). These releases both improved water quality and invertebrate species richness and the proportion of less-environmentally tolerant species, thereby achieving their release objective (Bednarek and Hart 2005). However, hypolimnetic releases can also reduce downstream water quality (Marshall et al. 2006). Required minimum flow releases can also restore vertical connectivity between the hyporheos and channel flow, at least in terms of water quality (Calles et al. 2007). A section of the Kissimmee River in Florida has had baseflow restored for around eight years. Prior to this, the channel was usually stagnant (Colangelo 2007). Restoration of flow also restored normal (oxygenated) levels of respiration and primary productivity from the very low levels present during stagnation (Colangelo 2007). However, this is not always be the case: large floods may be needed to de-silt gravel beds (Meidl and Schonbon 2004). These types of improvements to water quality are likely to initiate much longer recovery trajectories than situations where, for example, fish spawning is cued by increased flow speeds. Improved water quality is likely to trigger a cascade of responses such as increased primary productivity (benthic algal growth) due to decreases in turbidity and increased current, and successful colonisation by non-air breathing invertebrates due to oxygenation but also to the presence of a larger biomass of (algal) food.

In Australia, summer baseflow has been restored using year-round constant releases from the Parangana Dam to the Mersey River in Tasmania. After four years of releases and monitoring, increased invertebrate abundance and diversity and also increased trout recruitment were observed (AJ King et al. 2003), although again, this type of monitoring study does not indicate the recovery pathways that occurred to produce these improvements. Year-round inflow to the Snowy River was increased by closing a diversion aqueduct, resulting in a 50 per cent increase in inflow; but only about 30 per cent of the flow at downstream control and reference sites came from this source (Brooks et al. 2007). Five years of post-release invertebrate sampling showed consistent differences between Snowy River sites and reference and control sites and it appeared that the impact of Jindabyne Dam was still too large for the flow release to have affected the invertebrate assemblage (Brooks et al. 2007). That is, the increase in baseflow was insufficient to influence invertebrate assemblages. A similar result was obtained for fish, with strong spatial patterns among Snowy River sites and reference sites that were unrelated to the flow releases (Gilligan and Williams 2008). These studies occurred during a drought, when flow from unregulated tributaries was strongly reduced (Gilligan and Williams 2008), and this could have exacerbated the weak effect of the environmental releases. This is an important result because it is one of the few to document that environmental releases can fail to achieve environmental objectives because they are too small. Therefore, the magnitude of environmental releases can determine whether recovery pathways are initiated in the first place.

2.3 Freshes

Freshes have been used to attempt to remove fine sediment accumulated as a result of river regulation. This sediment is thought to weaken fluxes of materials between surface and subsurface environments (Hancock and Boulton 2005). In their experiment using a flow release of 5000 megalitres over three days, Hancock and Boulton (2005) showed that nitrate was flushed from sediments, but the effect was short-lived (days to weeks) and therefore unlikely to initiate recovery. However, some short recovery pathways could be initiated by freshes: short, dry season flow pulses released down the Olifants River in South Africa were associated with increased spawning activity of a native fish and appeared to have increased subsequent recruitment (Cambray et al. 1997).

Short (two-day) freshes released in association with continuous releases to increase baseflow have been shown to stimulate algal production in the Cotter River near Canberra. However, these spikes in flow were not large enough to alter the physiognomy of the algal assemblage, which remained dissimilar to nearby unregulated sites (Chester and Norris 2006). This release pattern was associated with increased small-scale spatial diversity of hydraulic conditions (Dyer and Thoms 2006), and there is evidence that these patterns are associated with the diversity of benthic macroinvertebrates (Brooks et al. 2005). Therefore, it has been suggested that a pattern of releases that maintains small-scale spatial diversity of hydraulic conditions could be a more effective use of small amounts of release-water than mimicking larger-scale flood events (Dyer and Thoms 2006). The recovery pathway initiated by this type of release pattern was to simultaneously stimulate algal production and create small-scale hydraulic habitat diversity to provide habitat and additional food for invertebrates, thereby improving invertebrate diversity. Worsening drought resulted in the implementation of even lower baseflow and less frequent release of freshes, but these releases appeared to have been sufficient to prevent any serious decline in river health as measured by a rapid assessment of macroinvertebrates (Peat and Norris 2007). However, preventing decline is not equivalent to initiating or sustaining ecosystem recovery.

2.4 Bankfull flow

A series of three experimental bankfull floods during a single year were released into the Swiss River Spöl to see whether they could remove fine sediment and sediment fans from the riverbed, restoring the pre-regulation bed morphology (Robinson, Uehlinger and Monaghan 2004). These floods successfully mobilised sediment, altered riverbed morphology and also reduced algal biomass and macroinvertebrate abundance for a short time (months). They were also associated with a threefold increase in the number of spawning trout (Ortlepp and Murle 2003), successfully meeting the short-term (annual) objectives of the project. Changes to riverbed morphology and sediment particle size, along with flow speed, possibly initiated the spawning response by trout. However, these changes could also have been sufficient to initiate longer-term changes in primary productivity and invertebrate species composition, but these were not investigated.

The 1996 flow release in the Colorado River lasted for seven days and was of shorter duration and size than pre-regulation annual floods (Schmidt et al. 2001). The flood successfully moved sand-sized particles and scoured out backwaters important for fish assemblages, but some of these effects were short lived. A few geomorphic alterations lasted several years (Schmidt et al. 2001). Macroinvertebrates and algae recovered from this release within three months and macrophytes within seven months (Shannon et al. 2001). Fish were hardly affected at all (Valdez et al. 2001). There appeared to be few lasting benefits from the release for plants and animals (Valdez et al. 2001), indicating that this release did not initiate pathways for longer-term recovery.

Similarly, in Californian rivers, bankfull flow releases of short (hours to days) duration are used to alter channel morphology, improve riparian vegetation and fish passage and enhance fish spawning (Jeffres et al. 2007). Releases could be related to the production of hydroelectricity, although generally in the literature such rapid fluctuations appear detrimental to river biota (for example, Blinn et al. 1995)). The effects on fish vary with species, location in the river and individual behaviour: these releases cannot be viewed as benefiting all fish species (Jeffres et al. 2007). However, in the San Juan River, near-bankfull spring releases, designed to mimic snowmelt events of a few days duration, were associated with positive responses by native fish (Propst and Gido 2004). So, bankfull flow releases, at the right time of year, could initiate recovery through direct effects on fish behaviour and indirectly through habitat modifications to improve fish movement and provide substrata for spawning. In contrast, hypolimnetic releases, if colder than normal, could exclude species of fish from rivers; for example, Murray Cod in the Mitta Mitta River (Todd et al. 2005). However, it is also possible that releases of hypolimnetic water could be used to reduce water temperatures to counteract the effects of climate change-induced increases, especially where shading from riparian vegetation is absent. This might require the release of smaller amounts of water for a longer period than is used to provide a fresh, but it might sustain populations of temperature sensitive species, or allow them to breed. Such an approach would require some experimental research to determine appropriate temperatures and dilutions.

Natural floods that pass reservoirs can also be used to assess the probable effects of environmental flow of at or near bankfull levels. One study on the Colorado River that looked at both short-term and long-term effects (up to 10 years) showed that two such floods over this period had no observable persistent effects on invertebrates, but that longer-term temperature changes, towards pre-regulation temperatures, did (Rader et al. 2007). They concluded that manipulating water temperature would be more effective than sporadic flooding events in rehabilitating macroinvertebrate diversity. However, these results are contradicted by another US study at Green River (Utah), which showed that partial restoration of pre-regulation temperatures led to a decline in invertebrate diversity because the change in temperature was insufficient on its own to ameliorate all the effects of the dam (Vinson 2001). Further research separating the effects of temperature from those of flow releases is required, especially under Australian conditions.

2.5 Overbank flow

Confined channels

A series of bankfull and overbank floods released over a three year period into a Swiss River (the Spöl) resulted in adaptation of the fauna to a regime of more frequent disturbance, more similar to that of unregulated streams, and successfully met the release objectives (Robinson, Aebischer and Uehlinger 2004; Robinson et al. 2003; Scheurer and Molinari 2003). The invertebrate fauna recovered over a period of years as it adapted to the new flow regime (Robinson et al. 2003), suggesting that the recovery pathway involved the interaction of species life histories with a sequence of flood disturbance events. The repeated floods also progressively reduced benthic algal biomass (Robinson, Uehlinger and Monoghan 2004), which might have also contributed to the changes in invertebrate assemblages. These and other studies suggest that repeated allocations of environmental water could be needed to induce longer-term recovery in rivers.

In the Swiss experiment, clear objectives were set, with the aim of determining whether a few releases of bankfull to overbank size, each with a few days duration, would sufficiently mimic the pre-regulation flow regime (Scheurer and Molinari 2003). The temperature of hypolimnetic releases was not a factor as it did not differ from natural floods. However, in warmer climates, such as Australia (Lyon et al. 2008; Todd et al. 2005) and South Africa (J King et al. 1998), releases of cold water of this type could affect the hunting or spawning ability of native fish and give introduced fish a competitive advantage.

Floodplain systems

Overbank flow releases are also be used to flood floodplain forests and wetlands (Ellis et al. 1999; Hillman and Quinn 2002; Robertson et al. 2001; Siebentritt et al. 2004; Zhang and Mitsch 2007), restore connectivity to groundwater (Hou et al. 2007), restore riparian vegetation (Zamora-Arroyo et al. 2001), and trigger fish and waterbird breeding (AJ King et al. 2008; Kingsford and Auld 2005; Leslie and Ward 2002; Stewart and Harper 2002; Ward 2005). An experimental one-month inundation of floodplain forest on the Rio Grande for three successive years began to push organic matter biomass on the floodplain towards pre-regulation levels (Ellis et al. 1999). Repeated intra-annual flooding of a few days duration (each event) restored fluxes of nutrients and carbon between a Midwestern (USA) river and its floodplain, showing that even relatively small flood-pulses could restore some functional connectivity (Zhang and Mitsch 2007). A single experimental flooding of temporary and perennial floodplain wetlands of the River Murray did not show convergence between invertebrate assemblages of the two types of wetland. It was concluded that flow releases could be managed to recharge wetlands while preserving the differences in biota between seasonal and perennial waters (Hillman and Quinn 2002).

Occasional (every few years), large overbank flow releases from storages on the Colorado River are restoring native riparian vegetation and reducing cover of exotic species in the Mexican section of the river (Zamora-Arroyo et al. 2001). Such floods ensure both channel movement and regeneration opportunities for plants (Hughes et al. 2001). In contrast, a single experimental flooding of a floodplain on the lower River Murray, South Australia encouraged growth of riparian and floodplain plants but did not cause germination (Siebentritt et al. 2004). It was not clear whether the absence of germination was due to the short-term nature of the flooding (a few weeks) or the lack of a seed bank (Jenkins and Boulton 2007; Siebentritt et al. 2004). In other words, was the flood duration was too short to initiate that type of recovery or was there a barrier in the form of the loss of the seed bank?

A six-year controlled flooding experiment on the Murray River, Australia, showed that timing of flooding had little effect on floodplain tree growth, but that increased duration of flooding did promote tree growth. In contrast, wetland macrophytes responded strongly to restoration of spring floods, with consequent positive effect on algal biofilm and macroinvertebrates (Robertson et al. 2001). Therefore, both the presence of flooding and its timing affected floodplain plants, but in different ways for different groups. Studies on European rivers showed that unseasonal flooding was actually detrimental to tree survival, especially for young trees (Hughes et al. 2001). In addition, the rate at which water levels declined after flooding determined the survival of tree seedlings (Hughes et al. 2001; Rood et al. 2005). Restoration of riparian vegetation and macrophytes is likely to be an important step in recovery of river food webs and animal assemblages (by providing habitat). The different responses by vegetation to environmental releases could initiate different recovery pathways and this requires further investigation.

In dryland rivers, extended periods of overbank flow initiate high levels of productivity on floodplains—from benthic algal growth as well as detrital material (Burford et al. 2008). As the floods recede, fish that have fed on the floodplains retreat to pools, but there is insufficient productivity to support them and mortality ensues (Burford et al. 2008). This floodplain-derived carbon is then used by the pool food webs. The loss of large overbank floods in dryland rivers is therefore predicted to reduce the supply of floodplain-derived carbon to pool food webs (Burford et al. 2008), thereby potentially reducing total carbon supply and the biomass that can be supported by pools. Recovery after the reinstatement of large overbank floods would be initiated by high levels of floodplain productivity, which delivers benefits such as waterbird breeding in its own right. The recovery pathway would then be likely to continue in the main river channel, as the floodwaters recede, by the movement of the increased fish biomass into pools. This would be followed by increased biomass supported by this floodplain-derived carbon. In this sequence of events, the duration of drying experienced by the floodplain would limit productivity, at least initially, due to aging and non-replenishment of spore, seed and egg banks (Brock et al. 2003; Jenkins and Boulton 2007). There is also a risk of blackwater events when forested floodplains are inundated, especially after long periods of drying (Howitt et al. 2007).

Waterbird breeding in the Macquarie Marshes (NSW) is triggered by flooding and there is a relationship between the area flooded and the number of nests (Kingsford and Auld 2005). Duration of flooding is also important: if it does not last long enough, birds might abandon their nests before chicks are fledged (Kingsford and Auld 2005; Stewart and Harper 2002). A comparison of different proposed release strategies for the Macquarie Marshes concluded that waterbird breeding would best be supported by accumulating water in dams and releasing it less often but in larger amounts (Kingsford and Auld 2005). On the Murray River, stored water has been used to prolong inundation of floodplain wetlands to allow waterbird breeding triggered by natural flooding to reach a successful conclusion (Arthington and Pusey 2003; Stewart and Harper 2002; Ward 2005). These breeding events are supported by two recovery pathways: a short pathway that involves the direct biological breeding response by birds to the presence of nest sites in suitably inundated wetlands; and a longer pathway by which floodplain food webs reassemble and produce enough food to support breeding birds. While breeding can be triggered relatively rapidly, it is the second pathway that enables successful fledging of chicks and supports waterbird populations.

More recently, prolonged overbank flow has also been used to facilitate fish breeding in the Murray River. AJ King et al. (2008) compared counts of larval fish and fish eggs among three years, only one of which had an overbank flow that inundated the Barmah-Millewa floodplain forest (flooding was prolonged using an environmental water allocation). They found that two native fish species, Golden Perch and Silver Perch, increased spawning activity during the flooding, but Murray Cod and Trout Cod did not. There were also more young-of-year fish after the flooding than there were in a previous year with no flooding, a difference that showed increased recruitment by the two native cod species. As AJ King et al. (2008) point out, while this is strong evidence for the positive effects of overbank flow and forest inundation for fish populations, it did not show clearly the direct causes for the increased numbers of eggs, larvae or fish. That is, the pathways for these particular aspects of ecological recovery were not identified by the research. To better identify pathways, or mechanisms, by which these positive responses occur, AJ King et al. (2008) suggest that the cues for spawning need to be identified and whether or not fish actively forage on the floodplains or received additional food in the river channel that is transported from the floodplain. Studies of natural overbank floods on Cooper Creek indicate that fish do move onto the floodplain and forage there (Balcombe et al. 2005), but AJ King et al. (2008) did not capture larvae in the Barmah-Millewa forest itself, suggesting that recovery processes could vary among floodplain rivers.

2.6 Dam removal

Dam removal is described separately as it is a relatively radical approach to flow restoration that could, in theory, enable complete restoration of the original flow regime. It is becoming more frequent in North America, but there are few studies documenting its use in Australia (but see Dwyer 2007). Poff and Hart (2002) point out that dams vary greatly in size, position in the catchment, and in their operational purpose and release capacity. Such differences will affect their impact and also the risks and consequences for their removal, both for the river ecosystem and human communities. Hart et al. (2002) present a conceptual model for examining the potential ecological responses to dam removal, including the time-scale for changes, which is important in setting ecological and community expectations of post-removal impacts. They also discuss risk assessment and integration of dam removal with broader catchment management.

Generally, the North American studies show that immediately after dam removal, large volumes of fine sediment that have accumulated behind dam walls are washed downstream, infiltrating coarser sediments (Doyle et al. 2005). There is considerable variation between rivers in the rate and pattern of sediment transport (Hart et al. 2002). Sediment can take years to move through systems, and recovery to pre-regulation conditions can be so slow (decadal) as to be perceived as non-recovery (Doyle et al. 2005; Hart et al. 2002). In other cases, recovery to pre-regulation conditions is not possible because of other disturbances to the river system. The release of contaminants from dam sediments into downstream river reaches is also a potential risk in dam removal (Hart et al. 2002).

Different components of the ecosystem respond differently to dam removal. Most macroinvertebrates respond quickly and positively, but long-lived mussels did not respond at all in one study (Doyle et al. 2005). Riparian vegetation also responded very slowly. Fish took a few years to respond but they did experience an increase in population density in association with increases in cover from the more variable riverbed that developed after dam removal (references cited in Doyle et al. 2005). Changes in species composition from lake to river-dwelling fish and invertebrate species have also been recorded (Hart et al. 2002). For fish, dams are a migration barrier as well as a flow and riverbed modifier, so part of their response to dam removal involves migration rather than flow restoration (Doyle et al. 2005; Stanley et al. 2007). Improvements in fish passage are frequently observed following dam removal (Hart et al. 2002).

Most channel readjustments occurred within a year of the dam removal and these changes were closely tracked by changes in algal composition and production (Doyle et al. 2005). After initial declines, macroinvertebrate assemblages began to increase and adapt to the new conditions (weeks to months), followed by changes to fish assemblages (a few years) (Stanley et al. 2007). In contrast, suspension-feeding species downstream, such as freshwater mussels, have been negatively affected by sediment transport for years after dam removal (Hart et al. 2002). Riparian vegetation can take decades to respond to dam removal, but macrophytes colonise the newly available areas upstream of the dam wall location relatively quickly (Hart et al. 2002).

Doyle et al. (2005) concluded (after reviewing several studies from the northern USA) that the removal of small dams was a powerful and effective restoration tool, leading to rapid restoration of flow and ecological processes. In the Australian landscape, therefore, removing small dams warrants experimentation and consideration, and also post-removal monitoring where it has occurred (for example, see Dwyer 2007).

2.7 Synthesis—what does this research tell us about recovery pathways?

Several themes emerge from these studies.

First, while the timing of flow releases is important for some species and processes, it is not always of primary importance for initiating recovery. Some species and processes will respond regardless of when the flow is released. A more important aspect of timing is likely to be repeated flow releases, whether they be a form of pulsed flood (for example, Robinson et al. 2003) or a summer-autumn sustaining flow (for example, Westbury et al. 2007). For recovery to occur, restoration of some types of flow components will require repeated releases. In the Swiss example, repeated releases mimicked natural flood peaking and stimulated recovery in several response variables. In the Australian example, the consequences of being unable to repeat the dry-season sustaining flow were evident in a decline in invertebrate assemblages in the Wimmera River. That is, recovery was initiated but without a release the following year, recovery at many sites ceased and the river condition declined again. It appears likely that repeated releases will be required to sustain recovery where the releases are directed at remediating water quality (Lind et al. 2007; Westbury et al. 2007) or where a natural pattern of spatial (Dyer and Thoms 2006) or temporal (Robinson et al. 2003) variability needs to be maintained by flow.

Second, a related aspect to repeated releases is releases of a sufficient duration to sustain recovery pathways. The various experimental floods of Murray River floodplains (Hillman and Quinn 2002; AJ King et al. 2008; Leslie and Ward 2002; Stewart and Harper 2002; Ward 2005) show clearly that prolonging floods to allow ecological processes time to run their course (for example, fish and waterbird breeding) is more important than inundating the entire area of floodplain. In particular, the flow components for which duration is likely to be critical are cease-to-flow periods and overbank flow, especially for floodplains.

Third, the research described above indicates clearly that different recovery pathways are likely for different parts of river ecosystems. Some animals and plants show direct responses to flow releases without any intermediary processes. For example, breeding by many vertebrate species and germination or flowering of some plants is a direct response to increases or decreases in flow. However, other responses appear likely to depend on a sequence of events over a longer period of time. For example, while vertebrate breeding might be triggered by flooding, reassembly of a food web supporting prey species might be necessary for a successful conclusion to breeding and for new individuals to recruit successfully to vertebrate populations. In other cases, changes to species composition can take months or years while the impact of restoration cascades through biotic assemblages. Over time, individuals progressing through their life histories will be more or less successful due to changes in food quantity or quality, habitat quantity, quality or connectivity or physicochemical conditions initiated by flow restoration.

Lastly, recovery pathways are likely to be strongly affected by the capacity for species to move along rivers and recolonise habitat and therefore by barriers to movement. Species vary strongly in their inherent capacity to disperse and colonise newly restored habitat. In addition, their dispersal is often blocked by physical barriers (such as dam walls or stretches of dry stream bed), or local extinctions mean that there are no populations remaining nearby to supply colonists. Information regarding the movement capacity of many freshwater species is limited (Robson et al. 2008). This aspect of recovery pathways requires considerable further research. In the conceptual model below, we include examples of potential barriers and limitations to recovery under different restoration scenarios.

3. Current work and knowledge gaps

Presently, the focus is on post-release monitoring to quantify responses to different environmental releases. This is important work that needs to be continued, using the latest improvements in monitoring designs, and the results should be published (as discussed above). However, as the review above shows, post-release monitoring often finds positive biological responses but does not usually identify the steps in the biophysical pathways that led to each response. The identification of the physical and ecological processes and mechanisms that lead to ecological responses is essential because it will allow greater predictability in river restoration and will enable the use of environmental water to be fine-tuned to deliver the desired outcomes. The general study of river restoration has produced several conceptual papers for guidance (Jansson, Backx et al. 2005; Jansson et al. 2007; Lake et al. 2007; Palmer et al. 2005), and there are also some conceptual papers specific to flow restoration (Arthington and Pusey 2003; Mitchell et al. 1999); but there are relatively few published studies of recovery pathways after flow restoration either from Australia or internationally (Souchon et al. 2008).

Regionally, existing studies are biased in their distribution. For example, there are no studies of recovery from Australian tropical rivers, although there is some predictive work (Leigh and Sheldon 2008). Similarly, there are few data for arid zone rivers in Australia, although there is a good understanding of the ecology of some systems that confers considerable predictive ability for the outcome of any flow modifications (for example, Cooper Creek and the Macquarie Marshes). Most Australian studies have been carried out in semi-arid, Mediterranean or temperate climate areas. They have focused on large, lowland or flood plain rivers (Hillman and Quinn 2002; AJ King et al. 2008; Lind et al. 2007; Siebentritt et al. 2004; Stewart and Harper 2002; Todd et al. 2005), although there are a few studies of more upland sections (Brooks et al. 2007; Chester and Norris 2006; Dyer and Thoms 2006; Gilligan and Williams 2008). These studies are still too few; and recovery in some river types, such as ephemeral and headwater streams, is not covered at all.

Worldwide, most studies focus on restoring baseflow and floods (from freshes to overbank floodplain flow). There have been relatively few studies of the restoration of other flow components. In particular, we found no studies reporting recovery pathways after restoration of cease-to-flow events, although a Monash University study is currently examining fish responses to cease-to-flow in the Broken River – Boosey Creek system of Victoria. More studies are required at this more arid end of the flow regime spectrum.

At present, studying recovery after flow releases is not occurring in a coordinated way (Souchon et al. 2008). Richter et al. (2003) recommended that experiments be used to adapt environmental flow release regimes in an adaptive management framework. Arthington and Pusey (2003) recommended that a coordinated set of flow release and recovery experiments be established around Australia. They acknowledged that there are significant design issues, but suggested that this be attempted anyway; and we agree that a national, coordinated approach is more likely to obtain the necessary information. In addition, Souchon et al. (2008) have suggested a more coordinated approach to this work occur worldwide, to improve knowledge and also methodology for flow restoration. We suggest that a further discussion paper could examine the options for design and implementation of a large-scale research program in Australia. Furthermore, such a program should focus on recovery trajectories. That is, they should document the sequence of physical, chemical, biological and ecological changes that occur over a long time period (years) in response to flow releases. This work should include measuring processes such as benthic metabolism, algal productivity, carbon and nitrogen flow through food webs and detritus processing as well as responses such as breeding events and recruitment to populations. It is important that these studies are sufficiently long-term to show full recovery trajectories. This could well require a commitment to studies of ten years duration, or more. As discussed above, repeated flow releases over a number of years could be required to initiate recovery in some rivers. The effect of such activities should be incorporated into research projects.

Finally, it is often the case that multiple flow components will need to be restored to initiate river recovery, so future work will need to consider following recovery pathways after the implementation of more than one type of environmental flow release.

4. Environmental risks of flow restoration

This section addresses the environmental risks consequent upon actual release of environmental flow. It does not address the risk of not releasing environmental water (see the discussion by Ladson and Finlayson 2002). Note that this discussion of risks is by no means comprehensive and there are likely to be other, unforseen risks of environmental releases. More experience and monitoring of environmental flow releases will greatly assist in future risk assessments.

There are few explicit studies of the risks of flow restoration in the literature: potential risks are usually mentioned in passing. Environmental risks generally arise from two sources: the quality of release water, and the impact of flooding on species unused to it for some years. Damage to infrastructure can also constitute an environmental risk. For example, flooding that damages sewage infrastructure or connects waste treatment ponds to a river system can cause pollution. Physical damage such as riverbank slumping might occur (Leslie and Ward 2002). It is also possible that some environmental release patterns favour exotic species such as trout or willows (Ladson and Finlayson 2002). Dam removal will not be considered further here, but note that it carries other sources of risk, such as the movement of sediment downstream.

Release waters from many dams are hypolimnetic (from the bottom of the water column) and this water is often colder than is natural and it tends to be low in oxygen. Unnaturally cool water is potentially detrimental to fish behaviour and survival in warm climate regions (J King et al. 1998; Ladson and Finlayson 2002; Lyon, et al. 2008; Todd et al. 2005), so release infrastructure could need to be modified to protect fish. Similarly, low levels of dissolved oxygen can be a problem, although the turbulent mixing created by release pipes could sufficiently oxygenate release water. The quality of release waters therefore needs careful consideration, especially if this water is a by-product of agricultural (for example, deep drainage) or industrial activity (for example, cooling water). Bushfires can also lead to poor water quality in reservoirs, especially high turbidity (Peat and Norris 2007). Modified release strategies have been shown to be capable of minimising the risk to macroinvertebrate assemblages posed by turbid release water (Peat and Norris 2007).

The impacts of floodwaters are sometimes of concern because floods can lead to lost habitat, reduced population densities or blackwater events. For example, the 1996 flood release into the Colorado River was of concern because some endangered fish and invertebrate species had increased in abundance after the river was regulated, and it was thought they would be threatened by high flow (Collier, Webb et al. 1997). In contrast, most studies indicate that biota are either adapted to and survive flooding well, or if their populations decline initially, they recover relatively quickly post-flood (Lytle and Poff 2004). Depending on the rapidity of river rise and fall, there is a risk of fish being stranded on floodplains (Leslie and Ward 2002).

Blackwater events are a natural occurrence in floodplain ecosystems. These events result from large concentrations of dissolved organic carbon leaching from leaf litter on the floodplain floor (Howitt et al. 2007). Microorganisms that use this carbon also use dissolved oxygen as they grow, and this can cause oxygen levels to fall low enough to cause problems for other biota (Howitt et al. 2007). The less frequently flooding occurs, the more leaf litter will accumulate on the floodplain and the more likely blackwater events become. This means that the risk of these events is likely to be increased in regulated rivers where floodplains are rarely inundated. The model of Howitt et al. (2007) can be used to predict the likelihood of these blackwater events under different conditions, and therefore this risk can be evaluated.

Saline and sulfidic (acidic) soils are present on river floodplains in Australia and the former are associated with salinisation of river landscapes (Lamontagne et al. 2006). The real level of risk is unclear at present, but sulfidic sediments produce toxic hydrogen sulphide and sulphuric acid and could increase chemical oxygen demand to the extent that fish kills occur (Lamontagne et al. 2006). Impacts could occur either when these sediments dry out during the implementation of a cease-to-flow event or once they are reflooded. In addition, floods that connect saline and sulfidic wetlands to the main channel could transport sediment or toxic compounds more widely. It is not yet known whether dilution will be sufficient to ameliorate these effects, and further study and experimentation is required to assess the level of risk.

A similar risk arises if flow releases increase waterlogging in areas where saline groundwater occurs. Salts accumulate in sediment when waterlogging occurs, but not when sediment is completely inundated (Brock et al. 2005). The germination of aquatic plants is therefore more successful under total inundation than when sediment is merely waterlogged. Flow releases that create large areas of waterlogged floodplain soils, but little inundation, are therefore likely to exacerbate both sediment salinity and its impact on the biota. Effects are likely to be greater in ephemeral wetlands than in perennial wetlands (Brock et al. 2005). Consideration should therefore be given to preferentially inundating areas of floodplains during overbank flow rather than causing waterlogging, even if this means restricting a given volume of water to a smaller area so as to maximise inundation.

One increasingly apparent but rarely discussed environmental risk is that the environmental release volume or duration is too small to have the desired benefit. In Australia, this has now been observed in the Snowy River (Brooks et al. 2007; Gilligan and Williams 2008), Wimmera River (Lind et al. 2007), Goulburn River (Ladson and Finlayson 2002), and on the Murray River floodplains (Ward 2005). The amount of inflowing water from less-regulated tributaries at least partly determines whether releases achieve their objectives (for example, see AJ King et al. 2008) or do not (Brooks et al. 2007; Gilligan and Williams 2008). Therefore, prevailing weather and climate conditions, independently of the release volume itself, are an important factor. As more environmental releases are studied, the level of knowledge about the necessary volumes to initiate recovery pathways is increasing, but it is expected to vary among rivers and flow components.

5. Conceptual models for recovery

5.1 Endpoint models

Having a clearly defined objective, goal or endpoint is one of the five criteria for ecologically successful river restoration (Palmer et al. 2005), but there are few detailed models for endpoints after flow restoration that are articulated in the literature. One Australian example of an endpoint model is the 'Healthy Working River' model for the Murray-Darling system. A 'Healthy Working River' is defined as 'a managed river in which there is a sustainable compromise, agreed to by the community, between the condition of the natural ecosystem and the level of human use' (Whittington 2002). This definition allows for different levels of river health to be defined for different rivers, depending on how they are used. Some rivers that are not be used at all might be preserved in a near-pristine state. For others, the community might accept a certain level of degradation for the services provided and the river could then be regarded as 'unhealthy'. The objective is for the river to be managed to be in as good an ecological condition as possible while still providing the services desired by the community. Importantly, both the economic activity and the ecological condition must be sustainable in the long term (Whittington 2002).

One exceptionally well-articulated endpoint model is the 'Living River' model for the River Meuse in France (Pedroli et al. 2002). This model was selected from three alternatives that each comprised different engineering works and flow control strategies. Small-scale trials of habitat restoration proved successful in supporting threatened species, and the model emphasised improved connectivity along the river, which was expected to improve biodiversity.

Other endpoint models are not so ambitious. For example, the large experimental flood in the Colorado River, USA in 1996 had endpoint goals of restoring beaches for recreational activity, controlling exotic vegetation, deepening the channel and restoring backwater habitats (Collier et al. 1997; Schmidt et al. 2001). Beach restoration, channel deepening and physical restoration of backwaters were successful with the effects lasting months to years, but the effects on vegetation were limited (Collier et al. 1997). There were also few lasting effects on fish, invertebrates or algae (Shannon et al. 2001; Valdez et al. 2001), although these were not specifically part of the endpoint model. From the point of view of geomorphic change to the riverbed, the experimental flood was deemed a success and likely to be repeated in order to maintain the restored geomorphic conditions (Collier et al. 1997).

Endpoint models, therefore, can vary widely in their objectives for flow releases—from relatively simple, easily measurable geomorphic changes to long-term ecosystem sustainability. While scientific information can indicate what conditions might comprise sustainability and what type of environmental flow might produce certain outcomes, the determination of what the endpoint should be for river restoration involves the whole community and its political processes (Jones 2002; Whittington 2002). Consequently, endpoints should be defined before restoration works commence to provide a clear goal (Palmer et al. 2005). Further consideration of these models is therefore beyond the scope of this review.

5.2 Process-based models

At present, two forms of models are apparent: (1) those based on the literature, and (2) a more nebulous set of models that are implicit in the decision to allocate and supply environmental flow. The implicit models underlying recovery objectives such as bird and fish breeding and tree recovery appear to be very simple. For example, adding the required amount of water to inundate a given area of floodplain or create the right hydraulic habitat for fish spawning in channels will result in breeding activity or healthier trees. However, in each case, there are events that must occur between a flow release and the desired outcome. For example, waterbird breeding requires inundation of the floodplain, followed by the reconstruction of an aquatic floodplain food web that can support the food requirements of breeding birds. That is, an entire food web must be reassembled in order to support this high energy activity. In addition, the birds must be able to locate the flooded area—that is, they must exist in the nearby landscape so that they can respond to the flooding trigger. The same is true for fish: the fish must be present in nearby river reaches to be able to respond to hydraulic cues for breeding, and they must also be supported by a sufficient abundance of the right prey animals to support this high energy-cost activity.

Lake et al. (2007) described four trajectories for river recovery following restoration (restoration in general not just flow restoration): recovery that follows the path of degradation, but in reverse order; recovery that shows hysteresis; recovery but to an endpoint that differs from the pre-degraded state (Humpty Dumpty model); and recovery where the endpoint is dependent on stochasticity in the trajectory itself. Of these, the recovery patterns documented in the literature after flow restoration match the hysteresis model and the 'Humpty Dumpty' model, although so few studies have followed recovery to an endpoint that it is difficult to assess the models proposed by Lake et al. (2007). As mentioned above, more long-term recovery studies are needed.

Many conceptual models for recovery after flow restoration focus on a particular component of the biota. For example, one international study of the regeneration of floodplain trees on several European rivers showed that channel movement was necessary to maintain habitat diversity. Overbank flow could achieve this and also recharge floodplain water tables to support tree growth (Hughes et al. 2001). A slow rate of decline of the flood peak was also important for seedling recruitment, and unseasonal floods also caused high seedling mortality. These findings describe a sequence of processes associated with overbank flooding to support recovery of floodplain trees, and together they allow the development of a conceptual model for effective use of overbank flow. A Chinese study on the same processes shows also that the effect of watertable recharge is uneven and declines with increasing distance from the source of the release (Hou et al. 2007). That study demonstrates that there are multiple factors determining the effectiveness of recharge and that these limit the response by floodplain vegetation, thereby providing evidence of the stepwise or threshold nature of recovery after flow restoration.

Perhaps the best-articulated model for recovery after flow restoration comes from the synthesis of dam removal studies by Doyle et al. (2005). They described two models: one for full ecosystem recovery and one for partial recovery. In both models different parts of the ecosystem recover at different rates. Which of the models applies in a particular situation would depend on, for example, whether other dams are still present on the same river. Both models have the geomorphic changes following dam removal as their initial driver—the first threshold crossed by the recovering system is the adjustment of the channel. The pathways followed then depend on factors such as water quality, habitat quality, sources of colonists and the presence or absence of other disturbances.

5.3 An outcomes-based model for recovery pathways after flow restoration

There have been several conceptual papers published that highlight the importance of the stages of re-connection, recolonisation, resumption of ecological processes, and increased biodiversity and recruitment following restoration (for example, Jansson et al. 2007), but there has been little specific recognition of the sequential nature of these stages. Acknowledgement of these steps in recovery is important for realistic expectations to be developed (Doyle et al. 2005). Figure 1 shows an example of the steps involved in restoration of baseflow. For example, it is important for recovery of native fish stocks that there is sufficient invertebrate food: invertebrates must be themselves increasing in abundance to support increased fish biomass (Figure 1, step 8). However, fish biomass will not increase if fish cannot get to the restored river section (Figure 1, step 5) and remain there (Figure 1, step 6). Similarly, invertebrate biomass and diversity will not increase unless they can reach the restored area and remain there (Figure 1, steps 5 and 6), and they also require the re-establishment of the food web (Figure 1, step 8) and the ecological processes that support it, such as nutrient spiralling, algal growth and the breakdown of detritus (Figure 1, step 7). Partial recovery of ecosystem processes (Figure 1, step 7) can occur when steps 5 and 6 are incomplete and will lead to some improvements in the endpoint (step 8), where environmental objectives are likely to be located. However, the rate of recovery is likely to be controlled by these successional processes regardless of whether the endpoint is reached (Jansson et al. 2007). Species recolonisation and succession are in turn controlled by the regional species pool as well as dispersal and habitat constraints (Lake et al. 2007). Therefore, steps 5 to 8 in this model will occur simultaneously and contain feedback loops, controlling the type of endpoint reached.

The Recovery Cascade Model presented here (Figures 1 to 6) is drawn as though these steps are sequential because they are, in part, successional in nature, but also because the model is focused on identifying outcomes and barriers to success in meeting the objectives of flow restoration. The model is aimed at clarifying what outcomes are delivered at which stage of the process and how these outcomes could be blocked by barriers, rather providing than an accurate representation of the ecological parts of the process. We acknowledge that different parts of the ecosystem will recover at different rates (Doyle et al. 2005), but we have not included time explicitly in our model. It is designed therefore, as a tool for management of recovery pathways and expectations from recovery, rather than as a conceptual model for the rates and levels of ecological recovery. In constructing this model, we have used our own experience with flow restoration and models either implicit or explicit in several publications (Doyle et al. 2005; Gore and Shields 1995; Jansson 2007; Lake et al. 2007; Leigh and Sheldon 2008; Mitchell 1999) in addition to those used in the literature review.

The Recovery Cascade Model can be adapted for the restoration of different flow components. For example, Figure 1 shows the restoration of baseflow. By sufficiently increasing baseflow, some benefits arise immediately in terms of human amenity, but other benefits, including such things as increased fish production, do not arise until towards the end of the recovery process. Note that the resumption of ecological processes does not deliver any new amenity from the human perspective, although it is an essential stage in ecological recovery that supports outcomes at step 8. Note also that re-connecting rivers and allowing organisms to emerge from refuges (steps 5 and 6) does increase biodiversity at the restored location, but that this effect will be temporary if these species cannot successfully establish at the site. In effect, biodiversity does increase by individuals passing through the restored site, but the site might not actually provide resources such as food, habitat or mates unless step 7 and 8 occur.

The top of Figure 1 lists the barriers that could prevent each step in the sequence from occurring. For example, release water that is very cold or low in dissolved oxygen might not permit step 3 to occur, and even if habitat areas are inundated (step 4), the habitat cannot be used by species unless the conditions for step 3 (necessary water quality) are met.

Even though baseflow is restored, there could be other barriers for dispersal and recolonisation, such as alienation of the floodplain and its wetlands (Gore and Shields 1995), the presence of dam walls, or loss of refuges in tributaries. If flow restoration has objectives related to the ecological parts of the system, then these barriers will also require amelioration (Gordos et al. 2007; Jansson et al. 2007). This relates to a commonly cited phenomenon in restoration ecology, the so-called ‘Field of Dreams’ hypothesis (Palmer et al. 1997). Barriers before step 5 can mean that, despite inundated habitat with suitable water quality, fish or other biota do not (cannot) recolonise the restored river.

Even if these dispersal and colonisation processes occur, continuous or multiple simultaneous disturbances of the river can hinder effective restoration. This phenomenon has been observed in the Wimmera and Glenelg rivers (Lind et al. 2007), which are subject to the effects of saline groundwater intrusion, stock access, sediment infilling and loss of riparian vegetation as well as flow regulation. The effects of these other disturbances can limit what can be achieved by flow restoration and need consideration when selecting which flow components to restore. In these two rivers, environmental water could be allocated to spring freshes or winter bankfull flow. But in these rivers, low baseflow in summer and autumn, combined with these other disturbances, can lead to prolonged cease-to-flow events with very low water quality (including high salinities, high temperatures and low levels of dissolved oxygen) and limited inundated habitat, potentially resulting in the loss of many taxa (Lind et al. 2007; Westbury et al. 2007). Therefore, summer–autumn sustaining flow is used in these rivers, instead of winter or spring releases, to try to maintain some areas of inundated habitat with acceptable water quality, thereby successfully maintaining biodiversity (Lind et al. 2007;

Westbury et al. 2007). In other rivers, unpredictable water level fluctuations from hydropeaking or irrigation releases continuously disturb the river and prevent establishment of some ecological processes. Therefore, in planning recovery trajectories for environmental flow, there needs to be an audit and full evaluation of the interacting disturbances that could strengthen or dissipate the effects of the flow. Potential barriers need to be identified and consideration given to how they can be overcome. Other restoration methods, such as the control of exotic species, could be needed to maximise the benefit of flow restoration (Williams et al. 2008).

The presence of invasive species or the loss of key species could mean that, although ecological processes re-establish, they are carried out by different taxa and this places limits on the level of ecological restoration that occurs. One example would be the re-establishment of aquatic vegetation in river pools, but with non-native rather than native species. The assemblage of invertebrates and fish supported by the exotic macrophytes will probably differ from that supported by native plants, but depending on the objectives, the outcome could be a sufficient improvement in biodiversity to be deemed a success.

This baseflow model has been adapted to show the recovery pathway, outcomes and barriers for each of the flow components identified in Table 1, but it could be readily adapted to use other schemes for describing flow regime. The release of freshes is a commonly proposed flow restoration strategy, but empirical studies show that sometimes the effects are of too short a duration. With a relatively small amount of water, released for a few days, physical barriers such as fords and culverts can cause water to bank up and flood land, resulting in the water never reaching the desired river sections (Dyer et al. 2007). Apart from this, the recovery pathway for freshes (Figure 2) and the outcomes of and barriers to, recovery are similar to those for restored baseflow.

The recovery pathway for restoring bankfull flow (Figure 3) resembles that for freshes (Figure 2) except that bankfull flow can cause an initial reduction in biodiversity as biota are swept away or damaged by scouring flow. This can lead to a 'resetting' of the biotic assemblage, which can be useful in restoration, such as by removing exotic or invasive species from the riverbed. This flow can be used to remodel the riverbed and we have assumed that potential physical barriers to dispersal are also likely to be washed away or otherwise remodelled (Figure 3).

Overbank flow in confined channels (Figure 4) will probably follow a recovery pathway similar to that for bankfull discharge (Figure 3) but with greater changes to bedform and greater loss of plants and animals. 'Resetting' will be a strong component of this flow and recovery will be slower than for bankfull releases. In contrast, where there is a floodplain (Figure 4), the recovery pathway will differ strongly because of the reconnection to the floodplain and associated dynamics. Outcomes will be associated with this reconnection and the initiation of ecological processes on the floodplain, as much as or more than in the river channel. In this case, the loss of the seed or egg bank in floodplain sediments could be a barrier to recovery (Figure 4) (Jenkins and Boulton 2007). As the floodwaters recede, the river channel is likely to receive additional carbon produced on the floodplain, for example by fish migration back into river pools (Burford et al. 2008).

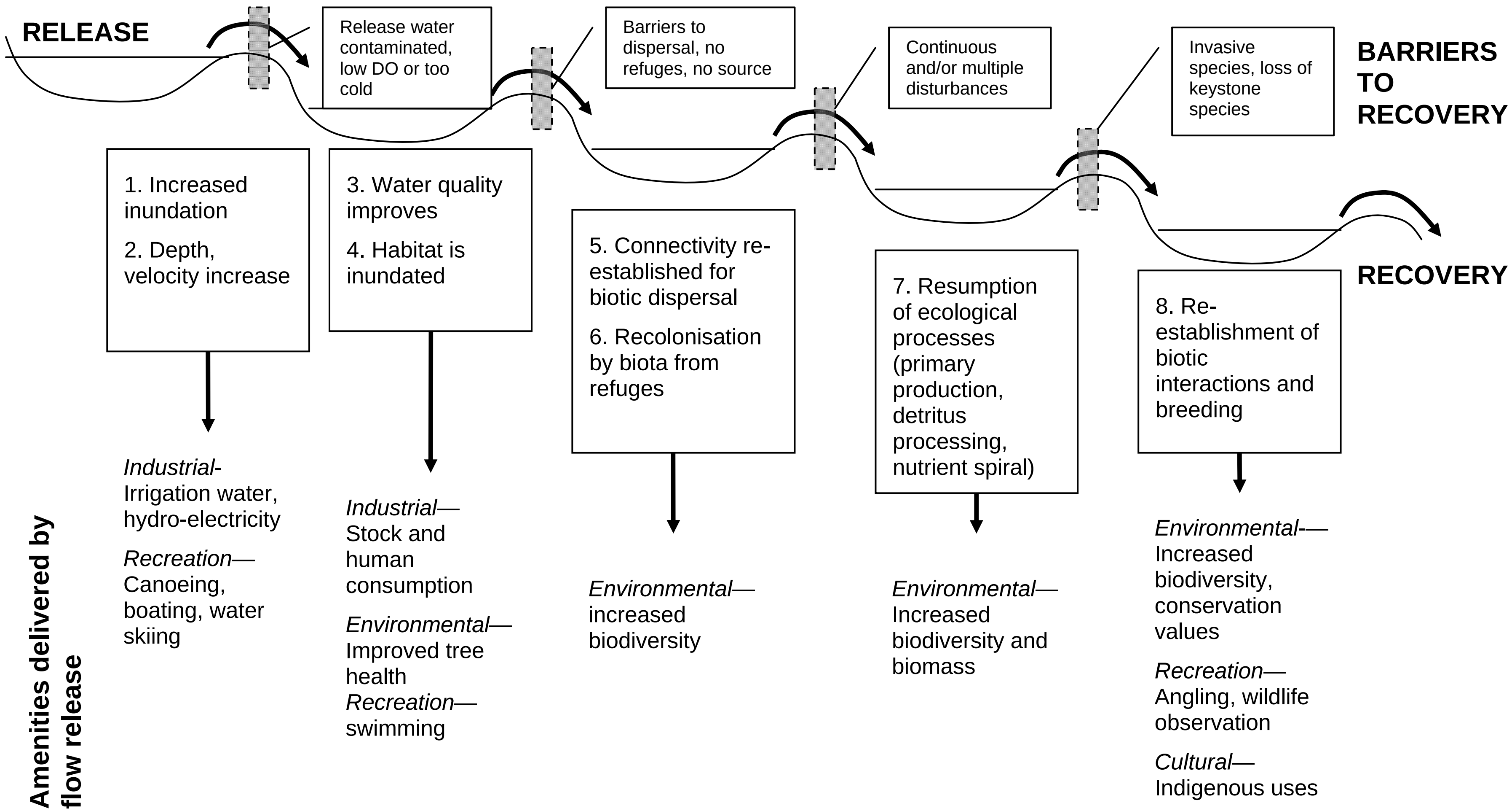
Recovery after the reinstatement of cease-to-flow events in a river that has become artificially perennial differs markedly from the other models (Figure 5). As with the floodplain, the loss of egg or seed banks could be a barrier to recovery of the pool or lentic conditions that will form in the river channel during cease-to-flow. Excessive drying and loss of pool refuges becomes a risk, as does poor water quality in these pools (Figure 5). During dry periods, river pools are often used as a water supply for livestock, so damage to the channel habitat and to the riparian zone may damage refuges and limit recovery. Isolated river pools are also potentially more vulnerable to repeated or multiple disturbances than a flowing river, because aquatic animals may be unable to escape disturbance.

Lastly, the pathway for recovery after dam removal (Figure 6) has some similarities to restored baseflow (Figure 1). In this case, most of the benefits of restoration occur at the endpoint (step 8) which is potentially problematic for local communities trying to see the benefits of dam removal (Doyle et al. 2005). As with some other restoration methods, a lack of refuges or the presence of other barriers (particularly other dams) on the river may limit what can be achieved.

5.4 Application of models for recovery pathways

Conceptual models for river recovery after restoration (all restoration, not just flow) have been given considerable status as providing guidance for both designing restoration projects and assessing their success (Jansson et al. 2005). Greater thought about how ecosystems recover is required (Jansson et al. 2005), and we would also argue that more data are required. In particular, observations of recovery after reinstatement of cease-to-flow periods is limited. But in all cases, we need more observations of real recovery pathways after all types of flow restoration in rivers, as discussed above. As more data become available, models will need to be adapted in the same way as flow management will be adaptive depending on the ecological response. These models could also be set in the framework of adaptive environmental assessment and management (Grayson and Doolan 1995) and be part of the review and adaptation cycle: see Pearsall et al. (2005) for an example in the environmental flow context. The models presented here can be adapted for existing situations by individual managers for their own systems. Specific risks, such as those related to water quality or salinisation, should be incorporated on a case-by-case basis.

Figure 1: The recovery cascade—the sequence of river recovery after restoration of baseflow after artificial cease-to-flow or after artificially low baseflows



Synthesized from Mitchell et al. (1999), Jansson *et al.* (2007), Lake *et al*. (2007) and others

Figure 2: The recovery cascade—the sequence of river recovery after restoration freshes

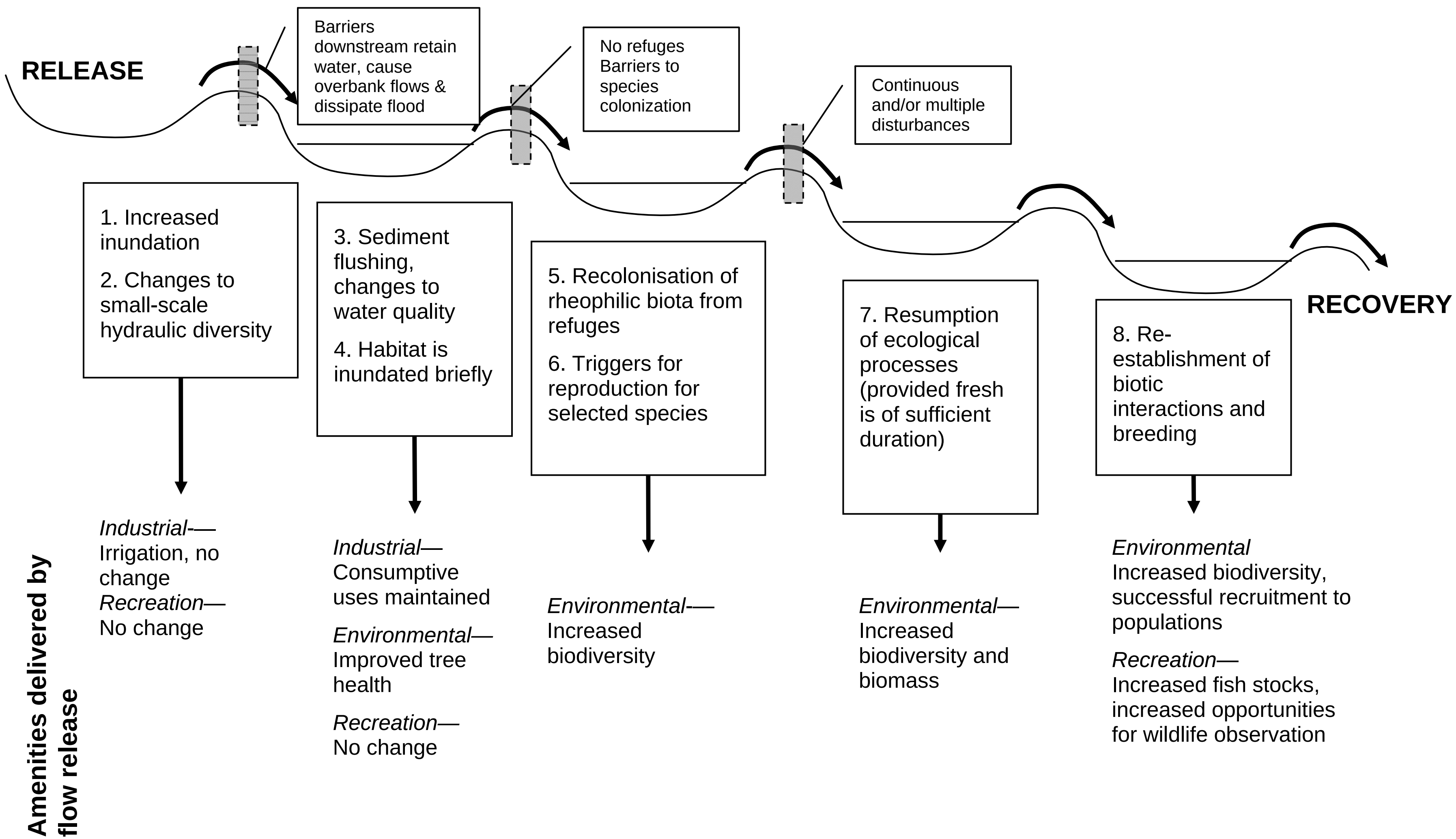


Figure 3: The recovery cascade—the sequence of river recovery after restoration of bankfull flows after loss of bankfull flows from hydrograph

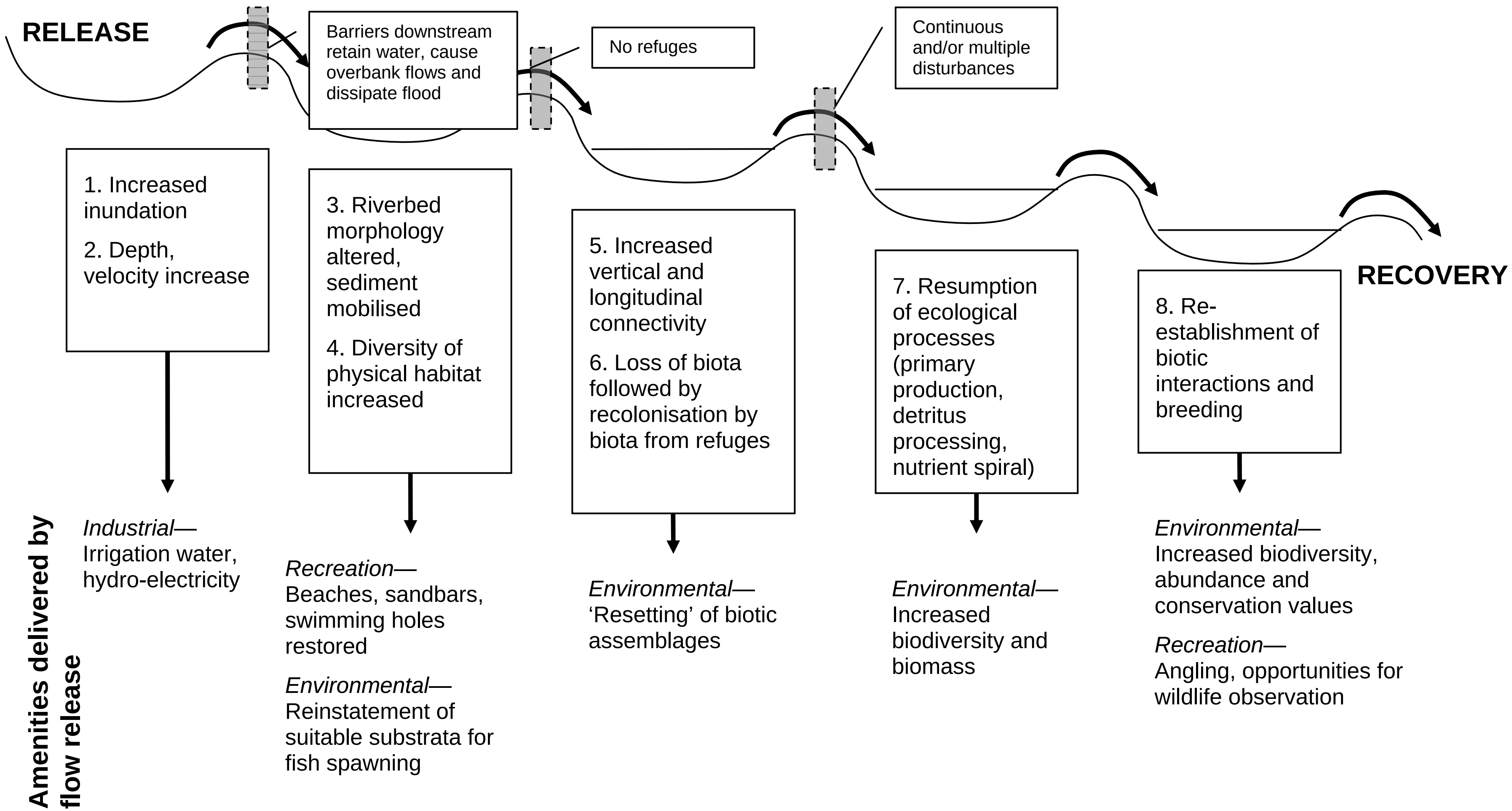


Figure 4: The recovery cascade—the sequence of river recovery after restoration of overbank flows onto floodplains (in the correct season, for a flooding duration resembling pre-regulation conditions)

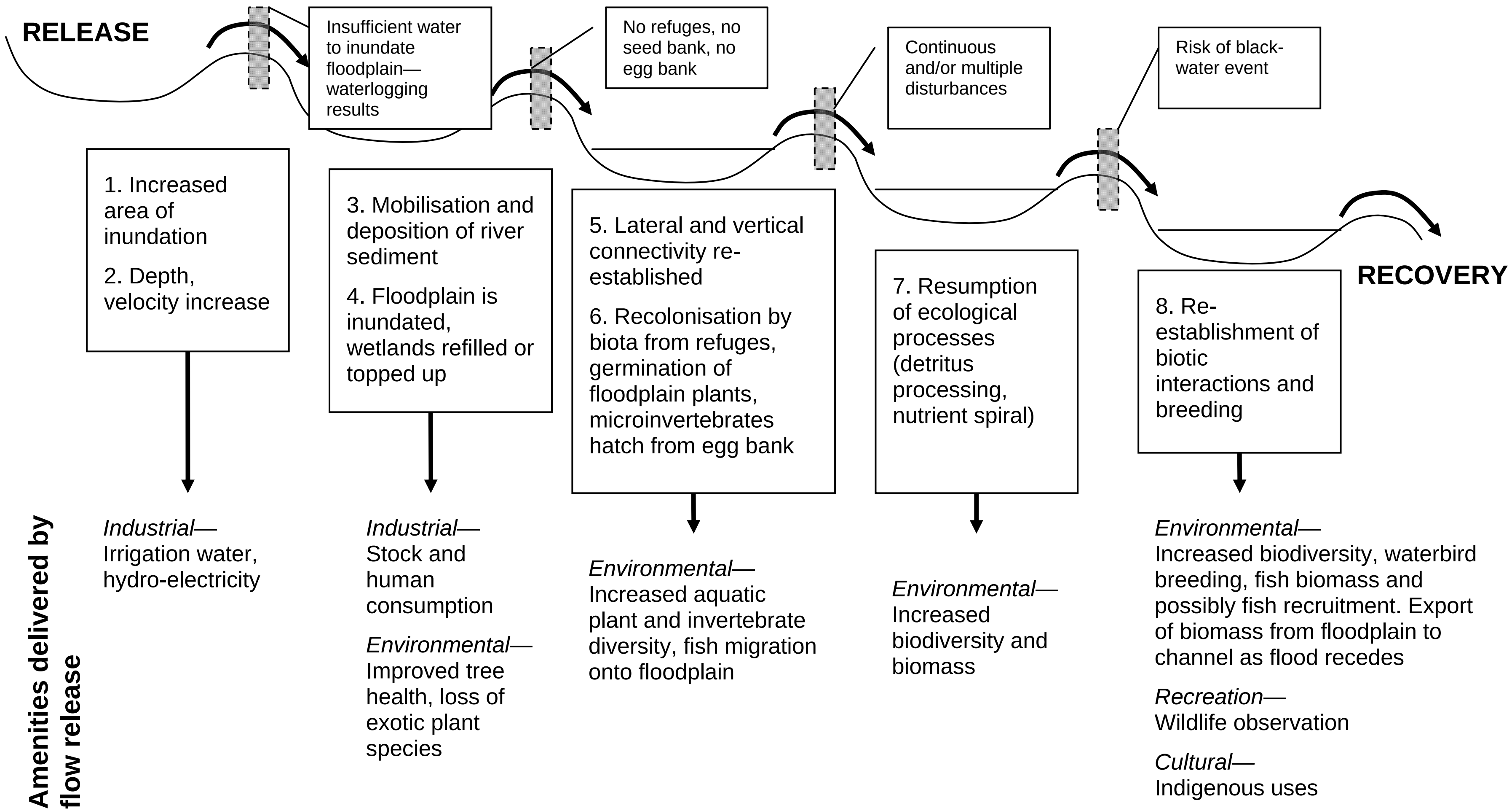


Figure 5: The recovery cascade—the sequence of river recovery after restoration of cease-to-flow after artificial perennial flows

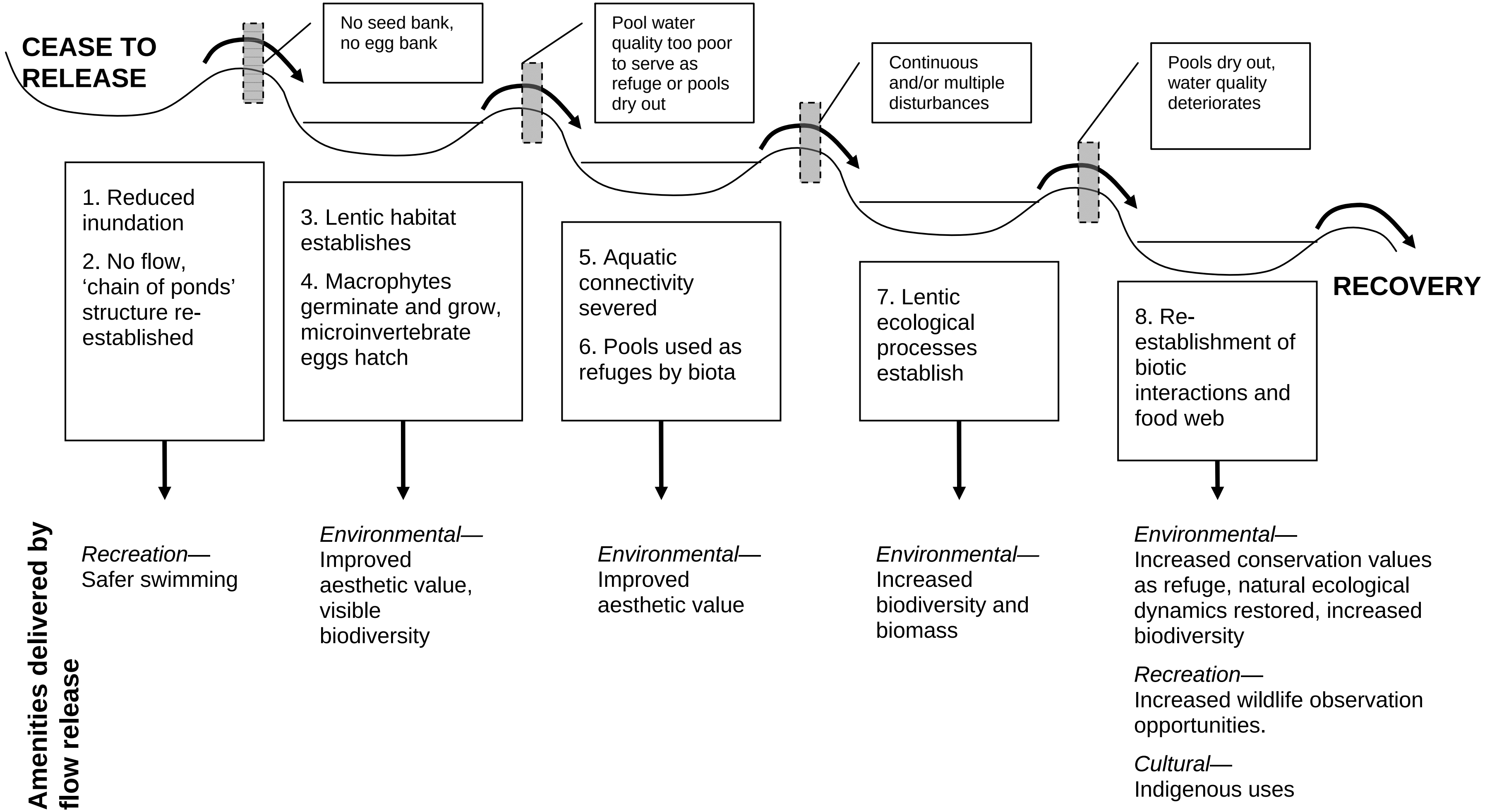
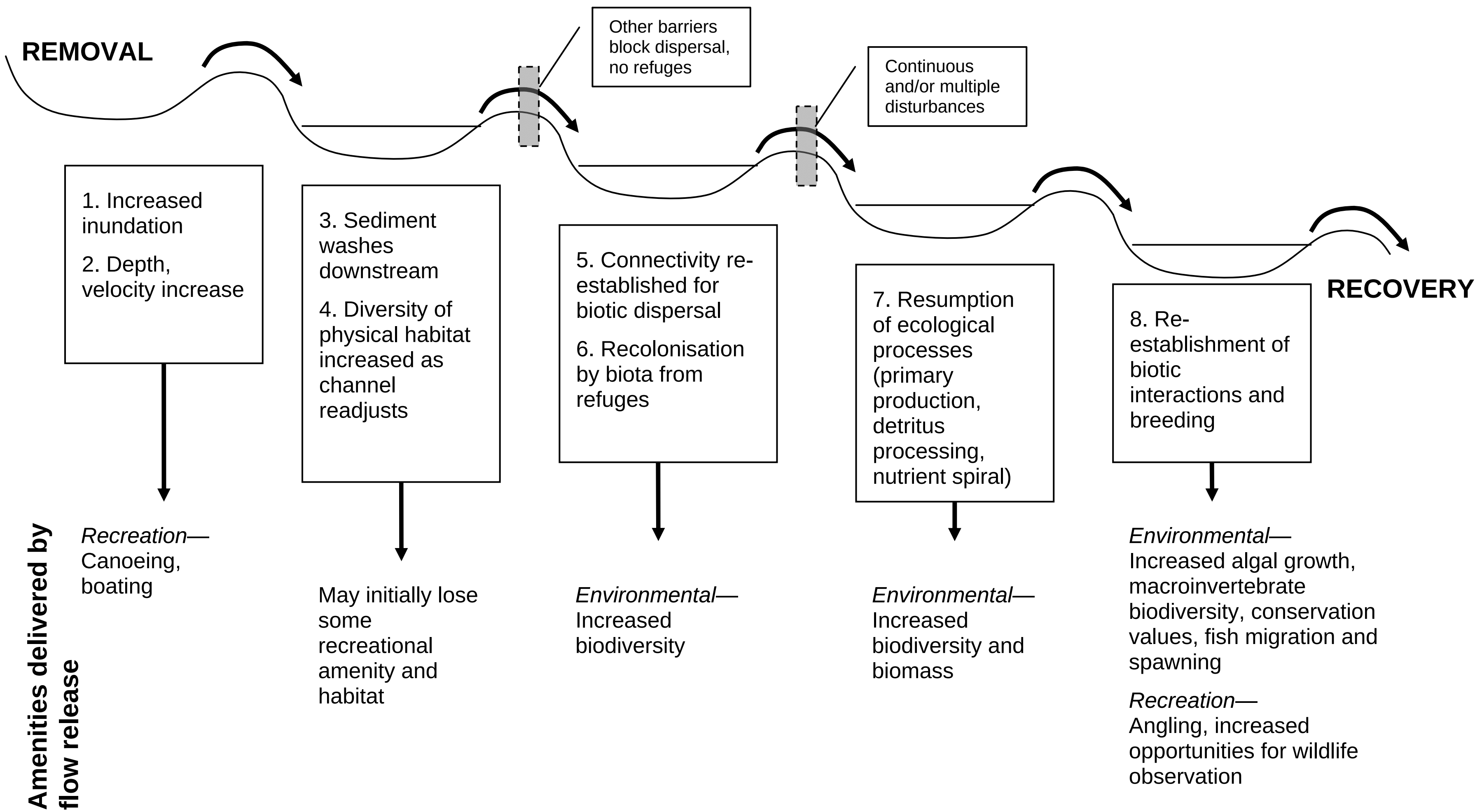


Figure 6: The recovery cascade – the sequence of river recovery after dam removal on streams and rivers with no floodplain



6. Conclusion

Recovery pathways for some types of flow restoration have been documented in a few case studies, but we are far from evidence-based documentation of recovery pathways. In planning recovery trajectories for environmental flow, there needs to be an audit and full evaluation of the interacting disturbances that may strengthen or dissipate the effects of the flow.

Designs for monitoring responses to environmental flow releases are improving rapidly, but process-based studies of recovery pathways need further development. In particular, recovery pathways after the restoration of multiple flow components need to be studied and potential barriers to restoration need to be identified and consideration given to how they can be overcome.

Here, we have presented six versions of the Recovery Cascade Model for recovery pathways based on the literature as well as theories of ecosystem recovery more generally. These models can be adapted by users for specific circumstances and should be regarded as works-in-progress, or hypotheses to be tested as data are collected.

Recovery pathways after flow restoration in rivers occur in two ways:

- by triggering direct responses in plants and animals
- by initiating the sequence of ecological processes shown in the Recovery Cascade Model.

Importantly, while flow releases may directly trigger responses such as germination or spawning, the ultimate success of these ecological processes depends on other, slower processes being initiated and sustained. That is, processes involving food webs and nutrient cycles, for example, are needed to support newly recruited plants and animals so that they develop into reproductive adults.

Different kinds of flow releases can initiate or support recovery pathways, and some options require more consideration and research in the Australian context. In particular, dam removal and the reinstatement of cease-to-flow periods require further study. Further innovation might be possible, such as the strategic use of hypolimnetic releases to fight the effects of increased temperatures from climate change.

Lastly, several case studies now show that releases that are too small in volume or duration, or that are not repeated sufficiently often will not initiate and sustain recovery pathways in rivers. Such outcomes need to be reported in the scientific literature to lead to improvements in predicting the duration, volume and timing of flow releases to increase the likelihood of initiating and sustaining recovery in rivers.

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