The Science of Environmental Water Requirements in South Australia

24 September 2002
Adelaide, South Australia

Seminar Proceedings

The Science of Environmental Water Requirements in South Australia

Seminar Proceedings

Adelaide, South Australia
24 September 2002

Published by:
The Hydrological Society of South Australia, Inc.

Sponsors

Seminar

The Hydrological Society of South Australia Inc.
Department of Water, Land and Biodiversity Conservation
Onkaparinga Catchment Water Management Board
Torrens Catchment Water Management Board
Patawalonga Catchment Water Management Board
Northern Adelaide and Barossa Catchment Water Management Board
River Murray Catchment Water Management Board
South East Catchment Water Management Board
Arid Areas Catchment Water Management Board
Eyre Peninsula Catchment Water Management Board

Environmental Water Requirements Research Scholarship

South Australian Water Corporation
Department for Environment and Heritage

Organising Committee

Diane Favier
Department of Water, Land and Biodiversity Conservation. SA

Steven Gatti
Onkaparinga Catchment Water Management Board

Michael Good
Department of Water, Land and Biodiversity Conservation. SA

Barry Johnson
Hydrological Society of South Australia

Sebastien Lamontagne
CSIRO Land and Water

Bill Lipp (Treasurer)
Hydrological Society of South Australia

Glen Scholz
Department of Water, Land and Biodiversity Conservation. SA

Frontcover photographs (from left to right):

Blue Lake, Mount Gambier (Willem van Aken), Mt Bold Reservoir, Onkaparinga River (Sinclair, Knight and Merz), Cooper Creek in flood (Roger Jaensch), Banrock Station Wetlands (Barry Johnson), Broughton River (Healthy Rivers Unit, EPA).

Foreword to the proceedings

All plants and animals biota on Earth depend on water in some way. But for some ecosystems, water is the dominant force shaping their ecology. These include rivers and streams, riparian zones, wetlands, floodplains, estuaries, and even cave and aquifer ecosystems. The water regime, including volume, timing, duration, water quality, groundwater levels and groundwater-surface water interactions, is crucial to shaping these diverse and complex systems and their associated communities of native plants and animals.

Water for the environment is an all-encompassing term that refers to the management of water to sustain water-dependent ecosystems. It encompasses both environmental water requirements (EWR) and environmental water provisions (EWP).

> ***Environmental water requirements*** *are the water regime needed to maintain water-dependent ecosystems, including their processes and biological diversity, at a low level of risk.*
>
> ***Environmental water provisions*** *are that part of the environmental water requirements that can be met at any given time. This recognises that providing water to the environment is a part of water allocation and management and that we must balance social, economic and environmental needs[1]. (ARMCANZ & ANZECC, 1996).*

Water for the environment is a key issue in water resource management in South Australia. The *Water Resources Act 1997* requires assessments of, and provision for, water for the environment through its planning processes. There have now been 14 water allocation plans prepared for prescribed water resources across the State. In developing these plans the environmental water requirements of dependent ecosystems have been assessed and provisions made within the plans for meeting their water requirements. There are also five catchment water management plans that deal with water for the environment.

It is now five years since the *Water Resources Act 1997* came into operation and in that time, a number of EWR assessments have been undertaken. While the science underlying the assessment of EWRs has developed considerably, there are still significant gaps in our knowledge. It is therefore timely to review the science of environmental water requirements in South Australia and to explore future directions.

The aim of this seminar has been to:

- to review the state of scientific knowledge and methods currently being used in South Australia and nationally to assess environmental water requirements;
- to share information and create links between practitioners, academics and managers; and
- to set directions for the development of the science needed to improve strategies for assessment of environmental water requirements.

These proceedings provide an overview of our progress towards addressing the complex issue of assessing environmental water requirements as well as outlining emerging issues and new approaches and assessment tools. Some of Australia's

---

[1] ARMCANZ/ANZECC, 1996, *National Principles for the Provision of Water for Ecosystems*

leading thinkers and practitioners in the discipline of EWR science have contributed papers for these proceedings.

The authors were asked to write a series of related papers that as a whole cover the underlying scientific understanding, the status of assessments in South Australia and provide a national and forward looking perspective. For surface water systems, the authors in sequential order are Walker, Gippel, Scholz, Brizga and Sheldon. For groundwater systems, they are Lamontagne, Cook and Evans. Each paper has been reviewed for relevance to the topic by members of the organising committee although no formal peer review process has been undertaken.

We trust that their efforts have provided a valuable reference that will assist you in understanding and addressing this complex and important issue.

The Organising Committee,
20 September 2002

Table Of Contents

Ecology and Hydrology

Keith Walker[1]

SUMMARY: The ecological and hydrological significance of flow is well understood, but the convergence of hydrologists and ecologists is comparatively recent and owes much to the emergence of issues in environmental flow management. The scientific basis for this technology is still weak, and the most popular models in river ecology are too general to be directly applicable in environmental flow management. Rather, the natural flow regime of individual rivers has become a *de facto* model for research, restoration and management. Even so, dryland rivers have shared features that require special consideration; this is particularly so in regard to the nature and consequences of inter-annual flow variability. Investigations of the River Murray suggest that cause-effect relationships cannot be understood by dissecting the hydrograph into separate components; rather, we need to develop an hierarchical perspective. Other desiderata for a more effective approach to environmental flow management are presented. For example, the implicit goal is to promote recruitment in populations of native flora and fauna; this is seldom recognized, but should directly influence planning and monitoring.

MAIN POINTS

- Although ecology and hydrology have much to offer environmental flow management, their collaboration is immature and the scientific basis for management is weak. In place of generally applicable models relating hydrology and ecology, the natural flow regimes of individual rivers provide a template for management.
- Dryland rivers as a group have distinctive features that need to be considered in science and management.
- Hydrological and ecological changes to the River Murray suggest that cause-effect relationships cannot be understood by dissecting the hydrograph into components. An integrated, hierarchical approach is needed.
- Although new methods for environmental flow assessment are being vigorously pursued, there may be too much emphasis on rapidity and simplicity at the expense of ecological realism.

1. INTRODUCTION

The disciplines of hydrology, geomorphology and ecology all recognize flow as a governing variable in river ecosystems. Patterns of flow govern the physical and biological environment (thus, the shape of the channel, the nature of sediments, hydraulic environments, exchanges between river and floodplain and storage, transport and transformation of nutrients). Flow also selects for organisms with particular attributes. Its effects are so pervasive that flow regulation inevitably has far-reaching consequences.

Virtually all of the world's major rivers are affected by regulation, from direct diversions to more subtle changes in the spatial and temporal patterns of flow. In highly altered rivers the effects may be devastating for native flora and fauna, and they may prejudice its utility as a resource for humans. A recent 'snapshot' assessment of the ecological condition of rivers in the Murray-Darling Basin (Norris et al. 2001) is a case in point. Strategic responses to these problems call for rapid developments in the still-emerging technology of environmental flow management, and present a major challenge to ecologists and hydrologists (e.g. Gurnell et al. 2000). For the Murray-Darling Basin a general protocol has been established (Whittington et al. 2001), but many problems remain (Jones et al. 2002).

This paper comments on the nascent relationship between ecology and surface-water hydrology, from a South Australian viewpoint. The commentary is highly selective and overlooks topics described in accompanying papers (e.g. groundwater hydrology, geomorphology). The focus is surface waters in the Lake Eyre and Murray-Darling basins and the Gulf drainage.

The paper outlines popular scientific models, and attempts to clarify goals and concepts related to environmental flow management. It highlights a number of distinctive features of dryland rivers that are not well-served by existing models. It also surveys the nature of hydrological changes to the River Murray and their ecological consequences, and suggests that underlying cause-effect relationships cannot be understood by dissecting the hydrograph. Finally, the paper considers a number of ways in which the quest for simple methods of assessment needs to be reconciled with certain ecological realities.

2. CONCEPTS IN ECO-HYDROLOGY

2.1 Preamble

Numerous concepts, hypotheses and models refer to the ecology and hydrology of rivers. Most are aligned with one of two key models, the River Continuum Concept (RCC) and the Flood Pulse Concept (FPC). Each concept has strengths and weaknesses, and arguably the RCC is a better model for upland streams and the FPC a better model for lowland rivers. There may be scope to combine selected features in a more comprehensive model of a river from source to sea (Walker et al. 1995).

2.2 River Continuum Concept

The RCC (Vannote et al. 1980; Minshall et al. 1985) supposes that there is a continuous gradient of physical conditions between the headwaters and mouth of a river, and that the composition and dynamics of biological communities along the river change in response to that gradient. The concept overlooks the significance of floods, other than as a "system reset" phenomenon (cf. Ward and Stanford 1995). It also overlooks the significance of river-floodplain interactions.

The RCC is not easily applied to big rivers like the Murray, where there is a well-developed floodplain, or where the continuity of longitudinal gradients is interrupted by floods and droughts. In these systems lateral linkages are likely to be more important than longitudinal linkages. In rivers like Cooper Creek, where flow is ephemeral, longitudinal linkages may be less significant than vertical linkages between surface and subsurface water.

A corollary of the RCC, the Riverine Productivity Model, has been described for large rivers where the floodplain is restricted (Thorp and Delong 2002). It may have application for the gorge tract of the Murray in SA.

2.3 Flood Pulse Concept

The FPC (Junk et al. 1989; Tockner et al. 2000) arose partly from perceived deficiencies of the RCC as a model for floodplain rivers. The FPC emphasises lateral exchanges between a river and its floodplain (ponderously called the "*Aquatic-Terrestrial Transition Zone*"). It suggests that regular (seasonal) pulsing of discharge is the key variable governing the biota. The flood pulse imposes alternate wet and dry phases on the floodplain, maintaining high levels of productivity. The channel receives most of its nutrients from the floodplain, and provides an avenue for dispersal of biota. Connectivity between river and floodplain is critical for maintenance of the ecosystem (e.g. Ward et al. 1999).

The FPC also emphasizes the distinctive nature of the floodplain biota. Thus, species of the "aquatic" phase have characteristics (abbreviated life cycles, capacity for aestivation and migration) unlike those of permanent lotic or lentic habitats. Similarly, the "terrestrial" floodplain biota differs from that of a terrestrial habitat not subject to flooding (e.g. water-borne seeds, strong colonizing capability, tolerant of submergence).

The FPC requires modification before it can be applied easily to dryland rivers. One problem is that it emphasizes the role of a *regular* pulse in the dynamics of ecosystems, whereas floods in dryland rivers are not regular (Walker et al. 1995; Puckridge et al. 1998).

2.4 An operational model

These concepts offer a framework for understanding river ecosystems, but their perspectives are theoretical rather than practical, and they do not lend themselves readily to prediction or measurement (but see Minshall et al. 1985). It is remarkable that the FPC in particular is not constructed to predict the effects of changes to flow regimes.

If neither concept has immediate application to environmental flow management, it might be presumed that ecologists do not have an adequate model to describe, understand and measure the effects of flow regulation. In fact, the aforementioned models could be said to emphasize *generality* and *reality* at the expense of *precision* (cf. Levins 1968). Where precision is paramount, a *de facto* model is provided by each river's natural flow regime. This approach stresses the individuality of rivers, and is appropriate for a discipline where generalizations are elusive and the literature is dominated by case studies. It also has utility for management.

The natural regime is widely regarded as a template for river conservation and management (e.g. Poff et al. 1997). It may be reconstructed through historical data, or by association with other rivers. Although restoration of the natural pattern may not be a panacea for degradation, it usually is part of the remedy. Part of the rationale is that we should aim to conserve native flora and fauna and thereby protect ecological integrity. The native biota is adapted to (and, in an evolutionary time frame, selected by) the natural flow regime.

The implementation of environmental flow programs may be limited as much by communication across cultural, social, political and economic boundaries as by the uncertainties inherent in ecological science. Thus, application of a natural flow template requires an appropriate cultural paradigm. A presumption implicit in this approach is that we should strive to limit our impact on the environment rather than seek to manipulate it for our own advantage (e.g. Regier et al. 1989). That view of conservation is by no means universal.

3. SOME FEATURES OF DRYLAND RIVERS

3.1 Generalities

An emphasis on the hydrological "signatures" of individual rivers does not undermine the value of generalizations. On the contrary, they provide essential perspectives for management and research. In South Australia especially, any practical or theoretical framework must have regard for the distinctive features of dryland rivers. These include disparate rivers like the Finke (Lake Eyre Basin), which lies entirely within the arid zone, and others like the Murray, which flows for most of its length through semi-arid country but has a well-watered headwaters catchment.

This section highlights a select few attributes of dryland rivers, although there are profound differences in the degree of expression. A more effective review, framed by an ecosystem-level analysis of the Paroo River in New South Wales, is provided by Kingsford (1999).

3.2 Flow variability

In arid and semi-arid regions, rainfall and runoff vary with diel and seasonal cycles and aseasonal influences like the El Niño Southern Oscillation (ENSO: Puckridge et al. 2000). Spatial variability also is prominent. In these respects Australian rivers are among the most variable in the world (McMahon et al. 1992).

Variability is an inherent part of dryland environments, and many native plants and animals are adapted to tolerate and exploit erratic flood and drought conditions. In seasonally more stable environments, including regulated rivers in dryland regions, resident species are likely to be displaced by other, often exotic species. This is clearly shown, for example, in the Murray in South Australia (Walker 2001).

An active area of collaboration between hydrologists and ecologists concerns measurements of the extent of flow modification in regulated rivers. A variety of indices has been developed and applied with some success. The variability of dryland rivers means that some of these indices suspect, particularly those that assume symmetrical distributions, need to be applied cautiously (see, for example, Richter et al. 1996, 1997).

3.3 Hydrological persistence

Wet and dry years are not statistically independent but tend to occur in blocs. This indicates hydrological *persistence*, evident in global patterns of rainfall and runoff (e.g. McMahon et al. 1992). Its ecological significance is that serial floods have cumulative effects. Floods enable organisms to augment recruitment, disperse more widely and gain security, especially in erratic environments like dryland rivers. In Cooper Creek, for example, Puckridge et al. (2000) demonstrated cumulative effects in wetland areas, water temperature and transparency and recruitment to fish populations during a cluster of five floods in 1987-91. Their analysis suggested cyclical persistence over about 6 years, near the upper limit of occurrences of La Niña (hence, a flood-dominated regime). In this region La Niña has a return period of about 5.3 years and that of El Niño (hence drought) is about 4.8 years. Serial droughts are no less significant than serial floods.

The ecological correlates of persistence are not well-understood, but there is sufficient evidence from fish, waterbirds and riparian trees to show that it should be considered in water resource management. In environmental flow management, the 'carry-over' effects associated with persistence clearly are relevant. For example, there is a correlation between recruitment of trees and serial floods in the Murray (see below).

3.4 Refugia

In streams where flow is episodic, the surface water may contract to pools, and may disappear for protracted periods. In these circumstances, the remnant waterholes provide *refugia* for plants and animals unable to disperse to new locations. The waterholes may also provide temporary refugia for waterfowl travelling between remote areas. In gravel-bed streams the sub-surface water may harbour crustaceans, insects and other organisms that normally are part of the surface water community, and it may also contain a distinctive resident community termed the *hyporheos* (e.g. Cooling and Boulton 1993).

For resident aquatic and terrestrial species, refugia are the primary source of colonists when there is renewed flooding. If the refugia are to remain viable, they need to be protected against adverse effects like increased abstraction for irrigation or intensified stock watering.

A wide-ranging investigation of refugia in Cooper Creek is presently being conducted by the CRC for Freshwater Ecology at Griffith University.

The notion of refugia applies more widely than to streams where the water periodically is localised. Even in perennial rivers, some plants and animals cease to reproduce during drought, and thereby become vulnerable. The demands of human water consumers tend to increase at such times and, as a consequence, populations of some species may be depleted to a degree where their capacity to respond to renewed flows is diminished. This may have contributed to the decline of native fish in the lower Murray (e.g. Walker 2001).

4. ECOLOGY OF THE HYDROGRAPH

At least in theory, a long-term river hydrograph may be dissected into structural components including pulse frequency, amplitude, duration, seasonal timing, rates of rise/fall and other characteristics that are likely to be ecologically significant. All such components, however, are inter-related and cannot be manipulated independently. From a management viewpoint, it may be more useful to consider suites of characteristics (that is, to manipulate the shape of the hydrograph rather than its individual elements). Spells analysis (Donald et al. 1999) is one form of hydrological analysis that has considerable potential for ecological investigations.

The term *flow regime* refers to a long-term, statistical generalization of the hydrograph (Walker et al. 1995; Puckridge et al. 1998). A *flow pulse* refers to a rise or fall in discharge, and *flow history* is the sequence of pulses before any point in time. The former term is preferable to 'flood pulse' because changes in flow, particularly within the channel, may be ecologically significant yet not qualify as a 'flood'. A rising pulse may also be called an *expansion* and a falling pulse a *contraction* (Tockner et al. 2000).

The following examples illustrate ways that particular kinds of hydrological variation may be ecologically significant (Puckridge et al. 1998):

Variability in pulse timing
In some rivers a flow pulse may reinforce seasonality, but where the timing of the pulse is highly variable, as in dryland rivers, seasonal and hydrological cycles will not coincide and the biota may receive conflicting cues.

Variability in pulse duration
Where the duration of the flood pulse is variable, the role of the floodplain as a spawning, nursery and feeding ground for some species of fish may risk high mortality, affecting recruitment (cf. Humphries et al. 1999).

Variability in pulse amplitude
Rivers with highly variable pulse amplitudes may experience periods of zero flow. The beginning or end of an in-channel flow may be more significant than overflow, especially after drought, and water-level changes within the channel also may be significant (again, in this respect the term *flow pulse* is preferable to *flood pulse*). The flora and fauna of rivers with extremely variable amplitudes may depend on waterhole refugia during drought.

Long-term variation
In rivers which are stable over long periods, the characteristics of individual pulses correspond with the generalised characteristics of the flow regime. In rivers which are highly variable, individual pulses are unlikely to represent the flow regime, and single events in flow history may drastically affect the structure of the community and the geomorphic environment. This is a key issue for dryland rivers.

Spatial variation
Spatial flow variability is a feature of all lotic systems, and is related to patchiness of rainfall, local topography, channel form and longitudinal summation or attenuation of the flood pulse. Where it is pronounced, community structure and other ecological features may reflect local rather than catchment-wide hydrological conditions. In highly variable systems like Cooper Creek, floods may be exhausted before reaching the drainage terminus.

Unpredictability
The FPC is not easily applied to rivers where the features of the flood pulse are unpredictable as, according to Junk et al. (1989, p122), "unpredictable pulses generally impede the adaptation of organisms". Arguably, this is too narrow a perception of biotic adaptation and ecological response, as life-history attributes like opportunism and flexibility may be seen as adaptations to unpredictable regimes.

5. LOWER RIVER MURRAY: A CASE STUDY

5.1 Hydrological changes

Flows to SA are governed by dams in upstream catchments. Within SA, the Murray's flow regime is further influenced by serial weirs, levees and barrages. There are 10 weirs on the Lower Murray (the Murray below the Murray-Darling junction), and six within SA.

The hydrologic effects of regulation broadly are as follows (e.g. Maheshwari et al. 1995, Walker 2001):

- Average annual and monthly flows are substantially lower than they were under natural conditions. Annual diversions from the Murray-Darling Basin nominally are "capped" at levels that prevailed in 1993-1994. The median annual natural flow now is exceeded only 8% of the time, and the discharge of the Murray at Blanchetown has been reduced by more than half. Flows at the river mouth are down by about 80%.

- The regulated regime is dominated by low flows and occasional high flows. Low flows (<5000 GL) occur 66% of the time under regulation, but would have occurred 7% of the time under natural conditions. Ninety five percent of annual regulated flows are 0-15 000 GL, whereas 95% of natural flows would have been 2500-20 000 GL. Big floods (recurrence 20+ years) are little affected.

- The seasonal extremes of monthly flows are little affected, despite intensive irrigation, so that the pattern still tends to a summer–autumn minimum and winter–spring maximum. The *magnitude* of the peak, however, has decreased markedly.

5.2 Ecological consequences

The following outline is derived from Walker (2001) and papers cited therein.

The Murray's floodplain flora and fauna depend on the river for dispersal and replenishment, and the riverine biota depend no less on the floodplain for food, nurseries and refuges. The effect of regulation, particularly weir operations, has been to isolate parts of the floodplain for longer periods than would have prevailed under natural conditions. Regulation has also extended the area of permanently-flooded wetlands, so that two thirds of Lower Murray wetlands are now connected to the river at pool level. Many of these wetlands formerly were subject to larger, more frequent water-level changes, and some would have dried periodically.

One consequence is that these wetlands may no longer exhibit the pulse of plant and animal growth associated with a flood following a dry period. Thus, disruption of the natural drying and wetting cycle is believed to affect the capacity of the ecosystem to benefit from floods. This idea has not been rigorously tested, but enjoys wide acceptance: reinstatement of wetting-drying cycles is seen as a priority in restoration of Lower Murray wetlands (e.g. Jensen 2002).

The drying-wetting sequence is no less significant for habitats *within* the river channel. Changes in the water level affect the growth of biofilms (algae, bacteria, fungi growing on sediments, rocks and wood) that provide food for some fish, and for snails and other grazing invertebrates. Rapid water-level changes associated with weir operations are thought to have promoted the growth of algae at the expense of bacteria, and thereby reduced the nutritional value of the biofilms for grazing snails (e.g. Sheldon and Waker 1997). Of about 18 species present before weir construction, only the native 'freshwater limpet' *Ferrissia* sp. and the introduced pond snail *Physa acuta* are now common. Irrigation pipelines are a refuge for some species, notably *Thiara balonnensis* and *Vivipara sublineata hanleyi*. Other likely factors in the decline include the alienation of wetlands and predation by introduced carp.

The river channel is characterised by strong currents, unstable sediments, a shallow photic zone and other conditions that represent a harsh environment for many organisms. The littoral zone, however, supports a narrow band of emergent and submerged plants that is a refuge for many animals. The distributions of the littoral plants are influenced with the frequency of flooding and exposure (e.g. Walker et al. 1994; Blanch et al. 2000). Littoral plant assemblages were less abundant and diverse in the drought year of 1988, when most water in the Lower Murray was diverted from the highly turbid Darling, than in 1994, when the middle Murray was again flowing strongly.

The littoral plants are not a natural feature, but an artefact of weir construction (Blanch et al. 2000). The banks of the unregulated river were largely bare, but the weir pools and seasonally more stable regime have allowed numerous wetland species to invade the channel. The littoral zone acts as a refuge for some species, and so warrants special consideration in management. The plants would be adversely affected if the natural flow regime were restored, so that any initiatives to restore instream habitats would need to be developed in parallel with restoration of wetlands.

Exotic willows (Salicaceae) now rival the native river red gum as a dominant riparian tree. Attempts to remove or control the willows have met with strong community opposition, but there is similar opposition to leaving them unchecked and the debate is poorly supported by scientific evidence. The problems may increase if certain hybrid forms become established in SA (Sue Gehrig, Univ. Adelaide, pers. comm.).

Most native fish have declined in favour of introduced species like the common carp (*Cyprinus carpio*). One of few native species to have increased in numbers is the bony herring (*Nematalosa erebi*), which thrives in weir pools and rivals the carp as a dominant species. The weirs are obstacles to migrations, and the Murray-Darling Basin Commission has a program underway to facilitate fish passage over weirs.

The range of the obligate riverine freshwater mussel *Alathyria jacksoni* has contracted since construction of weirs and *Velesunio ambiguus*, a species typical of floodplain wetlands, has invaded the weir pools and the margins of the channel. A similar re-distribution has occurred among the crayfish: the yabbie (*Cherax destructor*) is typical of floodplain habitats, but is now common along the river margins, and the Murray crayfish (*Euastacus armatus*) is near regional extinction. In effect, the weir pools have provided a new habitat for flora and fauna typical of wetlands, and the survival of true riverine species is prejudiced.

5.3 Cause and effect

The Murray weirs supplement the effects of upstream dams and diversions, but also have their own distinctive effects. Thus, regulation has stabilised the river at a seasonal scale by maintaining near-bankfull capacities, but also introduced daily level fluctuations in the weir tailwaters. Processes active in periods of stable water have been reinforced, and those active in the rising and falling phases of the flood pulse have been disrupted. The frequency, amplitude and duration of floods have decreased, and short-term water-level fluctuations have become more frequent. The overall effects have been to strengthen *longitudinal* linkages within the river channel and to weaken the *lateral* linkages between the river and its floodplain. The lateral linkages need to be restored, and maintained, to protect the integrity of the system.

The hydrological and ecological changes imposed on the Lower Murray are so extensive and pervasive that it is unrealistic to attempt to separate causes and effects. It is more practical to recognize 'pathologies' in ecological responses (e.g. Rapport et al. 1995), and to adopt an hierarchical approach, as in landscape ecology (e.g. Ward et al. 2002), as a framework for management.

6. DESIDERATA

6.1 Preamble

This commentary concludes with a series of points to demonstrate a perception that, amid the burgeoning technology in the emerging field of environmental flow management, too little attention is being paid to underlying goals. Our energetic search for simple, rapid, economical and politically palatable assessments may blind us to the real nature of the task.

The problem is not wholly ecological. In advice tendered by a group of ecologists to the Murray-Darling Basin Commission (Jones et al. 2001), panellists were asked to put aside political, economic and other considerations, and to specify "how much water is needed" to sustain the Murray ecosystem. When pressed, the Panel ventured that at least two thirds of the river's natural discharge would be needed to maintain an acceptably low risk of environmental damage. In fact any quantity of water diverted from the river will have some environmental impact, and a more pertinent question is "how much impact is the community willing to tolerate?". That question, however, is no longer an issue merely for ecologists.

The ecological integrity of a river and its floodplain should be axiomatic. The first lesson in the ecology of lowland rivers is that rivers and floodplains are interdependent. Yet floodplains are attractive for development, particularly in dry regions, and the lesson often is sacrificed to expediency. Environmental flow management should consider both river and floodplain, and specifically target connections between the two.

Simple cause and effect relationships rarely apply in ecological systems. It usually is unrealistic to seek connections between single hydrological inputs and particular ecological outcomes. Rather, the hydrograph should be seen as a suite of interdependent characteristics, and ecological responses also should be seen in terms of "pathologies".

An holistic approach is needed. It is folly to manipulate flows to benefit particular groups or species of flora and fauna. A more sophisticated conservation ethic leads us to maintain whole systems in the knowledge that this will benefit species of commercial value, or species that attract special empathy, as well as organisms, like invertebrates, that have no significant profile in public perceptions of conservation. It is salutary to remember that invertebrates are 99% of all known animal species.

The issues are scale dependent, and can only be understood and managed within such a framework. Concepts of spatial and temporal scale befuddle many arguments in ecology (and other disciplines concerned with complex systems). Unless the observer defines a frame of reference, it is easy to have fruitless arguments over whether, for example, flow variability in the Murray has increased or decreased as a result of regulation. In fact it has done both, depending upon the perspective (Walker et al. 1995). Even a simple logarithmic framework could assist (e.g. 0-1, 1-10, 10-100 years or square kilometres, as appropriate).

The requirement is for systematic programs of flow management. Although occasional, *ad hoc* releases of water do have beneficial effects, they should not obscure the need for systematic programs of flow management that allow for climatic cycles, like ENSO, and for cumulative effects like those associated with hydrological persistence.

The goal is to promote recruitment. An implied, but seldom articulated, goal of environmental flow management is to maintain populations of flora and fauna. Distinctions are rarely made, however, between the quantities of water needed for individual organisms to survive, or survive and grow, or survive and grow and reproduce. The reproduction of individuals is often wrongly equated with maintenance of populations, when the loop closes only when individuals are mature and potentially reproductive. The security of populations depends upon the frequency of recruitment and the numbers of 'recruits'.

For example, there are 10 distinct stages in maintenance of floodplain eucalypt populations (Amy George, Univ. Adelaide, pers. comm.), from flowering though to germination, sinker root establishment and eventual maturation. Each stage has its own flow requirements, and regeneration could be easily disrupted by *ad hoc* allocations rather than systematic watering. It is likely that peak recruitment of river red gums over the past 100 y coincides with serial floods, where the germinants from one flood are maintained by successive floods.

Monitoring data are vital. Effective management requires good information. The present standard of environmental monitoring is inadequate to support for environmental flow management, and we are forced to guess at the answers to critical questions. New developments in remote sensing and other monitoring technology provide an opportunity to learn quickly and, most importantly, to ensure that future generations do not want for the kinds of data that our forebears neglected to gather on our behalf.

7. REFERENCES

Blanch SJ, KF Walker, GG Ganf 2000. Water regime preferences of plants in four weir pools of the River Murray, Australia. *Regulated Rivers: Research & Management* 16: 445-456.

Cooling MP, Boulton AJ 1993. Aspects of the hyporheic zone below the terminus of a South Australian arid-zone stream. *Australian Journal of Marine and Freshwater Research* 44: 411-426.

Donald A, R Nathan, J Reed 1999. Use of spell analysis as a practical tool to link ecological needs with hydrological characteristics. *Second Australian Stream Management Conference*, 8-11 February 1999, Adelaide, South Australia: 205-210.

Gurnell AM, CR Hupp, SV Gregory (eds) 2000. Special Issue: Linking Hydrology and Ecology. *Hydrological Processes* 14: 2813-3188.

Humphries P, King AJ, Koehn JD 1999. Fish, flows and flood plains: links between freshwater fishes and their environment in the Murray-Darling River system, Australia. *Environmental Biology of Fishes* 56: 129-151.

Jensen A 2002. Repairing wetlands of the Lower Murray: learning from restoration practice. *Ecological Management and Restoration* 3: 5-14.

Jones G, T Hillman, R Kingsford, T McMahon, K Walker, A Arthington, J Whittington, S Cartwright 2001. *Independent Report of the Expert Reference Panel on Environmental Flows and Water Quality Retirements for the River Murray System*. Report by CRC for Freshwater Ecology to Murray-Darling Basin Commission. 50p.

Junk WJ, Bayley PB, Sparks RE 1989. The flood pulse concept in river-floodplain systems. *Canadian Journal of Fisheries & Aquatic Sciences* 106: 110-127.

Kingsford RT (ed.) 1999. *A Free-Flowing River: The Ecology of the Paroo River*. National Parks & Wildlife Service, Hurstville, New South Wales. 320p.

Levins R 1968. *Evolution in Changing Environments*. Princeton UP.

Maheshwari BL, KF Walker, TA McMahon 1995. Effects of flow regulation on the flow regime of the River Murray. *Regulated Rivers: Research & Management* 10: 15-38.

McMahon TA, Finlayson BL, Haines AT, Srikanthan R 1992. *Global Runoff — Continental Comparisons of Annual Flows and Peak Discharges*. Catena Verlag: Cremlingen, Germany. 166p.

Minshall GW, Cummins KW, Petersen RC, Cushing CE, Bruns DA, Sedell JR, Vannote RL 1985. Developments in stream ecosystem theory. *Canadian Journal of Fisheries & Aquatic Sciences* 42: 1045-1055.

Norris R, P Liston, N Davies, J Coysh, F Dyer, S Linke, I Prosser, W Young 2001. *Snapshot of the Murray-Darling Basin River Condition*. Murray-Darling Basin Commission, Canberra. 48p.

Poff NL, Allan JD, Bain MB, Karr JR, Prestegaard KL, Richter BD, Sparks RE, Stromberg JC 1997. The natural flow regime. A paradigm for river conservation and restoration. *Bioscience* 47: 769-784.

Puckridge JT, KF Walker, JF Costelloe. 2000. Hydrological persistence and the ecology of dryland rivers. *Regulated Rivers: Research & Management* 16: 385-402.

Puckridge JT, F Sheldon, KF Walker, AJ Boulton 1998. Flow variability and the ecology of large rivers. *Marine & Freshwater Research* 49: 55-72.

Rapport DJ, Gaudet CL, Calow P 1995. *Evaluating and Monitoring the Health of Large-Scale Ecosystems*. Springer, Berlin: 450p.

Regier HA, Welcomme RL, Steedman RJ, Henderson HF 1989. Rehabilitation of degraded river ecosystems. *Canadian Journal of Fisheries & Aquatic Sciences* 106: 86-97.

Richter BD, Baumgartner JV, Powell J, Braun DP 1996. A method for assessing hydrologic alteration within ecosystems. *Conservation Biology* 10: 1163-1174.

Richter BD, JV Baumgartner, R Wigington, DP Braun 1997. How much water does a river need? *Freshwater Biology* 37: 231-249.

Sheldon F, KF Walker 1997. Changes in biofilms induced by flow regulation could explain extinctions of aquatic snails in the lower River Murray, Australia. *Hydrobiologia* 347: 97-108.

Thorp JH, Delong MD 2002. Dominance of autochthonous autotrophic carbon in food webs of heterotrophic rivers. *Oikos* 96: 543-550.

Tockner K, F Malard, JV Ward 2000. An extension of the flood pulse concept. *Hydrological Processes* 14: 2861-2883.

Vannote RL, Minshall GW, Cummins KW, Sedell JR, Cushing CE 1980. The river continuum concept. *Canadian Journal of Fisheries & Aquatic Sciences* 37: 130-137.

Walker KF 2001. A river transformed—effects of weirs on the lower River Murray. In SJ Blanch (ed.), *The Way Forward on Weirs*, Inland Rivers Network (Australian Conservation Foundation), Sydney: 7-22.

Walker KF, AJ Boulton, MC Thoms, F Sheldon 1994. Effects of water-level changes induced by weirs on the distribution of littoral plants along the River Murray, South Australia. *Australian Journal of Marine & Freshwater Research* 45: 1421-1438.

Walker KF, F Sheldon, JT Puckridge 1995. An ecological perspective on large dryland rivers. *Regulated Rivers: Research & Management* 11: 85-104.

Ward JV, Stanford JA 1995. The serial discontinuity concept: extending the model to floodplain rivers. *Regulated Rivers: Research & Management* 10: 159-168.

Ward JV, K Tockner, F Schiemer 1999. Biodiversity of floodplain river ecosystems: ecotones and connectivity. *Regulated Rivers: Research & Management* 15: 125-139.

Ward JV, F Malard, K Tockner 2002. Landscape ecology: a framework for integrating pattern and process in river corridors. *Landscape Ecology* 17 (suppl. 1): 35-45.

Whittington J, J Coysh, P Davies, F Dyer, B Gawne, I Lawrence, P Liston, R Norris, W Robinson, M Thoms 2001. *Final Report, Project R2004: Development of a framework for the Sustainable Rivers Audit, 9 August 2001*. Report by CRC for Freshwater Ecology to Murray-Darling Basin Commission. 422p.

The Links Between Fluvial Geomorphology and Hydrology

Christopher Gippel[1]

SUMMARY: This paper explores the link between hydrology and stream morphology in the context of the geomorphic component of environmental flow assessment. In disturbed streams, or streams subject to a highly variable flow regime, the classical geomorphic concepts of bankfull discharge, effective discharge and channel maintenance discharge may not apply in the same way as described for intact and less hydrologically variable North American and European streams. Sediment transport formulae may be used with caution where there is no limit to the supply of sediment to the river – an uncommon situation in the Australian setting. Regulated flow regimes will nearly always be simpler and less variable than natural flow regimes, so the resulting geomorphology will probably be less diverse. Stream classification and description tools are useful for defining reaches for environmental water assessment, but are generally less useful for explaining the process links between hydrology and geomorphology. Classification is an exercise in data organisation, which can be a useful tool in aiding decision making, but classification is not equivalent to decision making. Environmental flow assessment should first place the significance of flow regulation as an agent of geomorphic change within the wider spatial context and temporal history of the channel. This will provide a realistic perspective on the potential for environmental flows to either maintain or improve channel functioning.

THE MAIN POINTS OF THIS PAPER

- Understanding the creation and maintenance of physical structure (equivalent to ecological habitats) within a variable flow regime is the main geomorphological issue in environmental flow assessment.
- Classical models of channel forming discharge and sediment transport may not apply to many Australian rivers.
- Management should seek dynamic channel stability (instability within certain bounds), rather than absolute stability.
- Environmental (regulated) flows will likely produce stream forms that are less diverse than those in undisturbed streams.
- Many geomorphic problems in regulated rivers cannot be solved simply by flow intervention and should be considered in the wider context of human disturbance.
- Stream classification and description tools are useful for defining reaches for environmental water assessment, but less useful for explaining the process links between hydrology and geomorphology.

1. INTRODUCTION

Fluvial geomorphology has always relied heavily on hydrological knowledge and methods to provide explanations for the features observed in rivers and streams. Geomorphological features are built from sediment erosion or deposition, so river shape is the outcome of the interplay between hydrological and hydraulic processes and the sediment regime. Given the complex nature of these processes and the many interactions possible, it should come as no surprise that the links between geomorphology and hydrology are difficult to define. However, there has been considerable progress on several fronts, and there are geomorphological tools available that can help in the assessment of environmental water requirements.

This paper is one of a series of publications by the author on the topics of degradation and rehabilitation of Australian rivers, incorporating geomorphology into stream management, and development of environmental flow methodologies (Gippel, 1996; Gippel and Collier, 1998; Gippel, 1999, Gippel, 2001; Gippel et al., 2001; Rutherfurd and Gippel, 2001a; Gippel, 2001b; Gippel, 2002; Gippel et al., 2002). These papers are not meant to represent a comprehensive picture of the Australian literature on the topic of assessment of environmental water needs. There are many other important and relevant works published by other authors, with most of them being cited in these listed papers.

This paper is based on Gippel's (2002) recent discussion of the application of geomorphology in environmental flow studies, especially with respect to the link between geomorphology and hydrology. The original paper contains full citations to the literature, and provides worked data from Australian case studies. This paper provides some additional comments on the role of fluvial geomorphic classification in environmental flow assessment.

2. THE NATURAL FLOW PARADIGM

Given the currently limited understanding of the links between biological processes and aspects of flow variability (usually restricted to a few key species), and the improbability of ever being able to fully define the

[1] Director, Fluvial Systems Pty Ltd, PO Box 49, Stockton, NSW 2295, phone 02 49284128, fax 02 49284128, e-mail chris.gippel@bigpond.com

needs of the whole biological community, the conservative alternative is to assume that the natural flow regime is the best indicator of environmental needs. One set of principles currently popular in the environmental flow field is based around the *natural flow regime paradigm*, which states that discharge variability is central to sustaining and conserving biodiversity and ecological integrity (in environmental water assessments this manifests as a recommendation to mimic natural flow variability).

Hydrological processes alone do not sustain aquatic life. As well as factors such as adequate water quality, food supply, and colonisation sources, aquatic organisms require diverse and abundant in-stream, riparian and floodplain habitats. Understanding the creation and maintenance of these physical habitats within a variable flow regime is the main geomorphological issue in environmental flow assessment. When applying the flow variability paradigm, it is important to consider process thresholds as well as hydrological variability *per se*. The concept of thresholds is well known to fluvial geomorphologists, because the process of sediment mobilisation and deposition operates as a threshold phenomenon.

The geomorphological extension of the natural flow paradigm is the concept of *river attributes*. These are basic hydro-geomorphic characteristics of natural streams thought to be necessary for maintaining ecosystem integrity in rivers. McBain & Trush (1997) devised a river attribute system that can be applied to most alluvial rivers (rivers with adjustable bed and banks). Of their ten attributes, those most explicitly concerned with in-channel geomorphic processes are:

- Spatially complex channel morphology
- Frequently mobilised channelbed sediments
- Periodic channelbed scour and fill
- Balanced fine and coarse sediment budgets
- Periodic channel migration

The ecological significance of these geomorphic attributes has some support in the literature. It has been argued that the attributes can be applied to both regulated and unregulated rivers, with some attributes achievable without requiring the entire unregulated flow regime (McBain and Trush, 2001).

3. THE CASE FOR RETAINING CHANNEL FORMING FLOWS

Channel forming flows include any flows that have a role in shaping the physical form of the channel, maintaining habitat forms, prevention of vegetation encroachment, and removal of fine sediment and detritus from the surface of the substrate. Like many aspects of environmental flow assessment, the idea of providing special flows that maintain channel morphology has attracted some criticism. This in part reflects gaps in the knowledge of the way channels form, regional and site specific differences in the way channels form, and indeterminacy of channel form.

Geomorphic models tend to be site specific, or process specific, they rely on empirical observations, and they usually have a fairly high level of uncertainty attached to their predictions. Given the difficulty of making theoretical predictions, one way of establishing a case for retaining channel forming flows is to examine the geomorphic effects of dam regulation, where hydrological modifications have altered the nature of channel forming flows in the river downstream.

Most dam-regulation and water diversion projects decrease the capacity of a stream to transport sediment. The net effect is that sediment delivered to the stream by tributaries, channel banks or side slopes accumulates on the surface and can work its way into the subsurface, altering substrate quality, rather than being flushed away during flood events. The connections between substrate quality and fish and macroinvertebrate populations are well documented in the literature (Reiser et al, 1985; Brookes, 1995).

Channel bed degradation (scouring or bed lowering) below dams is a common phenomenon. This type of response would not be expected in rivers that naturally transport low sediment loads or have naturally armoured or bedrock beds. Channel enlargement can only occur if the post-impoundment flows have the capacity to mobilise the bed and/or bank sediments. This can be the case for rivers used to transfer water for irrigation supply, where high flows are maintained for long periods over spring and summer.

Regulation of rivers by large impoundments usually involves a profound reduction in the frequency of small to medium sized floods. Because these floods include channel forming flows, it is not surprising that channel contraction (narrowing) is commonly observed below dams. Reduction in channel migration rates has also been observed below dams. Channel contraction below a dam is normally limited in its downstream extent to the junction of a major unregulated tributary, which reinstates channel forming floods. In some cases, coarse sediment delivered by downstream tributaries forms large gravel bars at junctions, because the regulated main stream is does not provide the energy to transport it further downstream.

Floodplain maintenance flows are large floods that shape floodplain features through lateral erosion, meander cut-off, avulsion (rapid change of course), and overbank sediment deposition. Only very large dams interfere significantly with floodplain maintenance floods, so they are not usually considered in environmental flow assessments.

Where flows for geomorphic processes have been recommended, they invariably represent only a small part of the natural medium and high flow regime. Many of the important in-stream habitat features of rivers are medium-scale geomorphic features, such as bars, undercut banks, pools and riffles, rock bars, sand slugs and cascades. There is currently only limited understanding of how specific bed and bank features form and are maintained. Low flows can transport sand-sized sediment and cause in-fill of scour pools, so there is a need to also consider the impacts of the duration of low flow events on river geomorphology.

There is overwhelming evidence in the literature from Australia and elsewhere that marked alteration of channel forming flow processes is associated with declining ecological health, or degradation of the physical channel attributes required for normal ecological functioning (Gippel, 2002). Although models of channel formation contain uncertainty, this does not discount the importance of channel forming flows, nor does it prevent quantitative consideration of channel forming flows in environmental flow assessment.

4. THE NATURE OF CHANNEL FORMING FLOWS

Channel form is a complex function of flood frequency, flood duration, sediment transport and boundary conditions (resistance of bed and banks). The main concepts in this respect are bankfull flow, channel maintenance flow, dominant discharge and effective discharge. It has long been thought that the process of channel formation was fundamentally associated with such flows, which can be expressed in terms of a consistent frequency and magnitude, but this idea has been seriously challenged, especially with respect to the Australian context.

Bankfull flow is the flow which just fills the channel to the top of the banks (banktop). The flow which maintains the important ecological and small-scale morphological characteristics of a channel (channel maintenance discharge) corresponds to the level where plants show sensitivity to inundation or where rock surfaces are abraded by bedload. While bankfull and channel maintenance flows are defined principally in terms of channel geometry, the dominant, or effective, discharge is the flow that carries the majority of the sediment load over a long period of time.

Dominant discharge usually refers to suspended sediment transport, while effective discharge usually refers to bedload transport, although it can include the suspended sediment component. Effective discharge is higher than dominant discharge because bedload transport is nil until the critical threshold for bed particle movement is crossed. In any flow distribution, low flows are the most common, but they do not carry a high sediment load, whereas the highest flows carry a high sediment load, but they are infrequent. Therefore it is the medium-sized flows, which carry an intermediate amount of sediment but occur relatively frequently, that transport most of the sediment in the long term. The effective discharge could be said to be the cause, and the bankfull channel geometry is the effect.

Some studies have found that the dominant/effective discharge of Australian rivers crosses a broad range, suggesting that channels are naturally adjusted to a wide range of channel forming flows. This wide band of effective discharge possibly explains the common existence of complex channel morphologies in Australian rivers. Regulated regimes tend to have a narrower effective discharge band, and simpler channel morphologies would be expected under these conditions (Gippel, 2002).

Yu and Wolman (1987) argued that because any competent discharge will reshape the channel when it occurs, and because the current channel form is the result of antecedent channel forming flows, in naturally variable rivers it is impossible to associate the channel geometry with a single discharge of a certain frequency. Erskine and Warner (1988) found evidence that the geometry of N.S.W. coastal rivers varied cyclically in response to changes in flood regime from drought dominated periods to flood dominated periods (lasting 30-50 years). As these rivers are continually changing, they cannot be said to be adjusted to a fixed discharge level. Brooks and Brierley (2000) disputed the significance of secular climatic phases in controlling morphological change in Australian river channels, and preferred to explain the documented widespread metamorphosis of channels in terms of anthropogenic disturbance factors that altered hydrology (through land use changes) and weakened channel resistance (mainly through desnagging and riparian vegetation clearance). Over longer, pre-European settlement time-scales (17 thousand years to the present), Nanson et al. (1995) found evidence in Tasmania that riparian vegetation and climate change were inextricably linked in determination of channel form. It appears unlikely that the concepts of bankfull discharge, channel maintenance discharge and effective discharge, as described in the literature for North American and European rivers, would apply in the same way to highly modified Australian river systems.

Environmental flow regimes that focus on a single discharge, or limited range of discharges, to perform the channel forming role could result in simpler channel morphology. While short-tem processes are important, channel formation in Australian rivers appears to also operate over a long-term time scale. Environmental flows (which are generally designed for and managed over the relatively short time-scale) should be placed within this longer-term context of changing channel form.

5. METHODS FOR CHARACTERISING CHANNEL FORMING FLOWS

Methods for determining flows that maintain the physical form of rivers are of three major types: simple desktop approaches that use the unregulated discharge record; desktop approaches that use data from a reference or adjacent (similar) catchment; and field/desktop based approaches that predict the discharge required to mobilise bed sediments.

5.1 Using unregulated discharge records

Various desktop methods have been devised in the U.S. to recommend environmental flows on the basis of unregulated discharge records. The most well known are the Tennant (Montana) method, the Northern Great Plains Resource Program method, The Montana Department of Fish, Wildlife and Parks method, the Hoppe method and the the U.S. Forest Service method (for details, see Gippel, 2002). These desktop approaches are inexpensive and can be rapidly applied. However, application in another area, or for another purpose (most existing methods are for maintaining salmonid spawning gravels), would require a period of validation, and this would involve an extensive program of field observations, similar to that undertaken to originally develop the method.

5.1 Using data from a reference or similar catchment

In the same way that properly functioning streams are used as templates for creating channel morphology in stream rehabilitation projects, the characteristics of the channel maintenance flow in an adjacent well-studied, unregulated catchment can be used to recommend environmental flows in a study river (Gippel and Stewardson, 1995). Natural hydrographs can be used to recommend rates of rise and fall to avoid the possibility of bank collapse due to rapid draining of water from saturated banks. The main problem with using reference catchments is that suitable data are rarely available.

5.1 Predicting flows for sediment mobilisation

Gippel (2002) reviewed the methods available for predicting sediment movement in rivers. Sediment can move as fine suspended particles, or as coarse bedload, and different methods are used depending on the particle size or process of interest. For channel forming flows, it is the coarse bed material that is of principal interest. One general rule-of-thumb for channel maintenance that has been used in gravel-bed streams is that sediment begins to be mobilised at a flow depth just greater than 80% of the bankfull flow depth. Another simple relationship based on Lane's data equates tractive force (depth of flow in metre x slope x 10^3) (kg/m^2) to incipient particle diameter in centimetres. The Meyer-Peter and Müller equation and the Shield's entrainment function have been used to predict sediment transport in rivers. However, the review of sediment transport formulae by Buffington and Montgomery (1997) warned that great care was needed in choosing defendable critical shear stress values for application of these and similar methods.

Sediment load modelling is widely and routinely practiced throughout the world. Despite this, it is known to be an inexact procedure with error up to an order of magnitude. A review of the procedures by Nakato (1990) concluded that prediction of coarse sediment discharge in natural rivers is not a trivial task, and urged skepticism of the results from such modelling exercises.

Stream power is the ability of the stream to do work. For channelised sand and gravel bedded rivers in England, Wales and Denmark, Brookes (1990) found that channels with bankfull stream power below 10 W/m^2 were aggradational, between approximately 10 W/m^2 and 35 W/m^2 they are stable, between 35 W/m^2 and 100 W/m^2 they are actively meandering, and channels with bankfull stream power above 100 W/m^2 are usually braided (or eroding). Stream power modelling requires extensive and high quality cross-sectional data, and application of a hydraulic model.

In many situations in Australia there is no basis for assuming that sediment supply is abundant and available for transport, which is a fundamental condition required for the application of bed material transport formulae. Such modelling should probably be limited to rivers affected by sand slugs or excessive sedimentation in lower gradient depositional zones. Field measurement of bed material movement is also difficult, but large scale, properly designed, controlled release experiments may provide insights for channel forming flows in regulated rivers (for examples, see Gippel, 2002).

6. ENVIRONMENTAL FLOWS IN DEGRADED STREAM CHANNELS

Many of Australia's streams have suffered incision in response to inappropriate landuse (Gippel and Collier, 1998). While in many areas the cause of these problems has ceased, or reduced in severity, the geomorphic artefacts of the previous human impacts can continue to characterise channels for a long time.

While regulated stream channels are likely to show the effects of flow regulation (such as widening, narrowing, or deepening), unregulated streams, which probably make up the greatest proportion of stream channel length, are probably similarly impacted to some extent by other land use and channel management factors. In regulated rivers it should never be assumed that channel geomorphic condition is solely a function of flow regulation. In many regulated rivers, the channel condition is a product of myriad impacts. The first stage of any assessment should be to place the significance of flow regulation as an agent of geomorphic change within a wider context. This will provide a realistic perspective on the potential for environmental flows to either maintain or improve channel functioning.

7. THE USE OF STREAM CLASSIFICATION

Historically, most fields of science have undergone a phase of classification during their early stages of development. As advancements are made, classification gives way to development of empirical relations, and then to theoretical understanding of fundamental processes (Goodwin, 1999). Description and classification helps us to mentally organise, and thereby understand complex objects, systems and ideas, while the main limitation of this approach is its poor predictive power. The point of classification is not just to identify all of the distinctive features of rivers, but to clarify their differences. Classification schemes are of two main types: those that group like with like, and those that rate one system against another (perhaps relative to what would be expected in a reference river). Geomorphic classification schemes are usually based on physical criteria. They are used for river management with the assumption that the physical characteristics define the likely biological characteristics. Most classification schemes use structural rather than functional (process) characteristics as their criteria for similarity. Process-based schemes are less common, but their predictive capability allows river managers to plan for the impacts of intervention (with a modest level of confidence).

Environmental flow assessments utilize geomorphic classification and description in two main ways. The first is to select representative reaches. Environmental flow recommendations in rivers are usually made for certain "compliance points", which are located at the downstream ends of reaches that are assumed to be reasonably homogeneous in terms of flow requirements. The boundaries of these reaches may be selected partly on the basis ecological factors and/or hydrological factors, but they usually describe geomorphic zones with distinctive physical channel structure. The second way geomorphic classification and description is used is in identifying the physical stream "condition", or evaluating the level of disturbance to channel form due to regulated flows or other degrading factors (see discussion in previous sub-sections). Some models attempt to predict the likelihood of being able to restore a disturbed channel form to a previous state (Gippel, 2001).

The traditional engineering approach to stream management viewed the inherently dynamic nature of rivers as an annoyance that should be controlled. Regulated rivers often show signs of accelerated instability (unnatural rates of erosion, deposition or channel migration). Knowledge of a river's geomorphology potentially allows control over instability, and much of the geomorphologically-oriented stream management literature is concerned with how to attain stream stability as the ideal. Many classification schemes are based on stable streams as the ideal reference condition. For example, The Index of Stream Condition channel form assessment (Ladson and White, 1999) is couched in the simple notion that channels that have the appearance of being stable are desirable, while channels with bed material or banks that move are undesirable. However, this conventional paradigm is slowly falling out of favour. The alternative is to seek dynamic stability within acceptable limits. Quantifying this level of acceptable instability and determining how to manage for it in regulated rivers are important tasks for geomorphologists working in the environmental flows field (Gippel, 2001).

Geomorphic classification systems have traditionally been based on river shape. Rosgen (1994) used stream channel dimensions to define eight primary stream channel types in the U.S. This scheme provides detailed descriptions of the reach within the stream network, but there is no link to the hillslopes. Rosgen's (1994) method has also been criticised by for being subjective, and failing to properly identify terms such as "channel stability". The River Styles classification system developed by Brierley (1999) for Australian rivers is based on the hierarchical model of Frissel et al. (1986). Geomorphic units are the building blocks of River Styles. The scheme is strongly evolutionary, and it provides a common geomorphic language with which to describe the fluvial characteristics of rivers, and predict their recovery potential.

O'Keeffe et al. (1994) suggested that perhaps the greatest hurdle to overcome with classification schemes was the unrealistic expectations by potential users. Classification is an exercise in data organisation, which can be a useful tool in aiding decision making, but classification is not equivalent to decision making. Stream classification and description tools are useful for defining reaches for environmental water assessment, but less useful for explaining the process links between hydrology and geomorphology.

8. DISCUSSION AND CONCLUSION

Research suggests that the characteristics of the flows most important for controlling channel form vary depending on local and regional-scale physiographic features. There is evidence in the literature that a range of small (sub-bankfull), medium (near bank-full) and large flow events (overbank) have general significance for channel formation and maintenance. However, it appears that some Australian streams are in a continual state of adjustment to cyclical changes in flood regime, so they do not adjust to a fixed discharge. Also, most of the classical literature on channel forming processes does not consider the role of boundary roughness and strength in any detail, and it appears that extensive modification of this aspect of Australian rivers (through desnagging and riparian vegetation clearance) has caused a fundamental shift in their geomorphic character.

The literature contains adequate advice regarding geomorphic classification schemes suitable for environmental flow assessment. Also, there are many hydrological and geomorphological techniques and models available for application to the problem of designing a flow to achieve a particular geomorphic effect. While the models generally lack strong predictive power, they will produce an approximate (order of magnitude) solution. These models are used to make predictions about discrete geomorphic events, such as the flow required to mobilise sediment, the amount of scour during a given event, or load of sediment transported through a reach over a given time period. However, natural flow regimes are composed of numerous facets, or components, occurring as a complex time series, not discrete, predictable and independent events. Scaled-down flow regimes, or regimes with certain components culled from the regime, should not be expected to produce the same morphology as a natural flow regime. Environmental (regulated) flows will nearly always be simpler and less variable than natural flow regimes, so the resulting geomorphology will probably be less diverse, either through time or space or both (e.g. some features may be absent, or present less often), or be expressed at an altered scale (e.g. some features may be smaller, or overall channel size will be different).

Many geomorphic problems in regulated rivers cannot be solved simply by flow intervention, and in unregulated rivers, increased water diversions will not necessarily be the main cause of future degradation. In heavily modified rivers, it may be preferable (or necessary) to use environmental flows to enhance the current geomorphic attributes, rather than seeking a return to some previous, or ideal state. It is important then for geomorphologists to take an exploratory approach towards assessing environmental flow needs, and be prepared to innovate and expand the scope of the assignment as necessary.

9. REFERENCES

Brierley, G. (1999) "River styles: an integrative biophysical template for river management." In Rutherfurd, I.D. and Bartley, R. (Eds) Proceedings of the Second Australian Stream Management Conference, 8-11 Feb, Adelaide, Cooperative Research Centre for Catchment Hydrology, Monash University, Clayton, pp. 93-100.

Brookes A. (1990) "Restoration and enhancement of engineered river channels: some European experiences." Reg Riv: Res Manage **5**:45-56.

Brookes A. (1995) "The importance of high flows for riverine environments." In Harper D.M. and Ferguson A.J.D. (Eds) The Ecological Basis for River Management. John Wiley & Sons, Chichester, pp. 33-49.

Brooks, A.P. and Brierley, G.J. (2000) "The role of European disturbance in the metamorphosis of the Lower Bega River." In Brizga, S. and Finlayson, B. (Eds) River Management; the Australian Experience. John Wiley & Sons, Chichester, pp. 221-246.

Buffington, J.M. and Montgomery, R. (1997) "A systematic analysis of eight decades of incipient motion studies, with special reference to gravel-bedded rivers." Wat Res Research **33**:1993-2029.

Erskine W.D. and Warner R.F. (1988) "Geomorphic effects of alternating flood- and drought-dominated regimes on NSW coastal rivers." In: Warner R.F. (Ed.) Fluvial geomorphology of Australia. Sydney: Academic Press, pp. 223-244.

Frissel, C.A., Liss, W.J., Warren, C.E. and Hurley, M.D. (1986) "A hierarchical framework for stream classification: viewing streams in a watershed context." Environmental Management **10**:199-214.

Gippel C.J., Jacobs T and McLeod T. (2002) "Determining environmental flow needs and scenarios for the River Murray System, Australia." Australian Journal of Water Resources **5**(1):61-74.

Gippel, C.J. (1996) "Water management." In Manual of Wetlands Management, Wetlands Conservation Report Series No. 4, National Parks Service, Department of Natural Resources and Environment, East Melbourne, Victoria, pp. 101-152.

Gippel, C.J. (1999) "Managing regulated rivers for environmental values: selected case studies from southeastern Australia." In Brizga, S.O. and Finlayson, B.L. (Eds) River Management: The Australasian Experience, John Wiley & Sons. Chichester, pp. 97-122.

Gippel, C.J. (2001a) "Australia's Environmental Flow Initiative: filling some knowledge gaps and exposing others." Water Science and Technology **43**(9):73-88.

Gippel, C.J. (2001b) "Hydrological analyses for environmental flow assessment." In Ghassemi, F. and Whetton, P. (eds) Proc. MODSIM 2001, International Congress on Modelling and Simulation, Modelling & Simulation Society of Australia & New Zealand, The Australian National University, Canberra, pp. 873-880.

Gippel, C.J. (2002) "Geomorphic issues associated with environmental flow assessment in alluvial non-tidal rivers." Australian Journal of Water Resources **5**(1):3-19.

Gippel, C.J. and Collier, K.J. (1998) "Degradation and rehabilitation of waterways in Australia and New Zealand." In DeWaal, L.C., Large, A.R.G. and Wade, P.M. (Eds) Rehabilitation of Rivers: Principles and Implementation, John Wiley & Sons, Chichester, pp. 269-300.

Gippel, C.J. and Stewardson, M. J. (1995) "Development of an environmental flow management strategy for the Thomson River, Victoria, Australia." Regulated Rivers: Research and Management **10**:121-135.

Gippel, C.J. Seymour, S., Brizga, S. and Craigie, N. (2001) "The application of fluvial geomorphology to stream management in the Melbourne Water area." In Rutherfurd, I., Sheldon, F., Brierley, G. and Kenyon, C. (Eds) Proc. Third Australian Stream Management Conference, Brisbane, 27-28 August, CRC Catchment Hydrology, Monash University, Clayton, pp. 239-244.

Goodwin, C.N. (1999) "Fluvial classification: Neanderthal necessity or needless normalcy." In Olson, D.S. and Potyondy, J.P. (Eds) Wildland Hydrology, American Water Resources Association, TPS-99-3, Herndon, Virginia, pp. 229-236, (www.stream.fs.fed.us).

Ladson, A.R. and White, L.J. (1999) An Index of Stream Condition: Reference Manual. Prepared for Waterways Unit, Department of Natural Resources and Environment, 2nd Ed. Melbourne, Vic.

McBain & Trush (1997) "Trinity River maintenance flow study final report." Hoopa Valley Tribe Fisheries Department, Hoopa.

McBain S. and Trush B. (2001) "Thresholds for managing regulated river ecosystems." In Sommarstrom S. (Ed.) Proc. Sixth Biennial Watershed Management Conference, Water Resources Center Report No. 92, Davis: University of California, pp. 11-13. http:/watershed.org/wmc/ (accessed November, 2001).

Nakato T. (1990) "Tests of selected sediment transport formulas." J Hydraulic Eng **116**:362-379.

Nanson, G.C., Barbetti, M. and Taylor G. (1995). "River stabilisation due to changing climate and vegetation during the late Quaternary in western Tasmania, Australia." Geomorphology **13**:145-158.

O'Keeffe, J., King, J. and Eekhout, S. (1994) "The characteristics and purposes of river classification." In Uys, M. (Ed.) Classification of Rivers, and Environmental Health Indicators. Proc. Joint South African Australian Workshop, Cape Town Feb 7-11, pp. 9-17.

Reiser D.W., Ramey, M.P., and Lambert, T.R. (1985) "Review of flushing flow requirements in regulated streams." San Remon:Pacific Gas and Electric Co. Department of Engineering Research.

Rosgen, D.L. (1994) "A classification of natural rivers." Catena **22**:169-199.

Rutherfurd I.D. and Gippel C.J. (2001) "Australia versus the World: do we face special opportunities and challenges in restoring Australian streams?" Water Science and Technology **43**(9):165-174.

Yu, B.,Wolman, M.G. (1987) "Some dynamic aspects of river geometry." Wat Res Res **23**(3):501-509.

Review of EWR Assessments in South Australia

Glen Scholz
Department of Water Land and Biodiversity Conservation, GPO Box 2834, Adelaide5001

SUMMARY: Over the last five years environmental water requirement (EWR) assessments have been conducted for a number of South Australia's significant river systems. The majority of methods used involved a scientific panel approach and were conducted over a period of one to two years. The methods used have included:

- Hydrological desktop studies
- Holistic methodologies involving multi-disciplinary scientific panel assessments
- Quantitative assessment methods

In developing EWR assessment programs there are a range of methods that can be applied depending on the outcomes required, the information available, the appropriate spatial and temporal scales and the resources available.

There are also challenges that are relevant across all methods. EWR assessment programs are particularly challenging in ephemeral river systems due to the spatial and temporal variability of flow and ecosystem responses. Our current knowledge of the ecological and geomorphological responses to flow regime is limited and our estimates of the environments water requirements may need refining. Therefore the methods we use to develop EWR assessments need to be scientifically defensible, repeatable, transparent, provide a solid base of knowledge and be adaptive to new information.

THE MAIN POINTS OF THIS PAPER

- The majority of EWR projects in South Australia involve a multi-disciplinary scientific panel approach.
- EWR assessment programs are particularly challenging in ephemeral river systems due to the spatial and temporal variability of flow and ecosystem responses.
- Need to recognise the limitations of the methodologies and the current information / knowledge gaps.
- The outputs of the assessment need to be scientifically defensible and provide a foundation for future work.

1. INTRODUCTION

Environmental water requirements (EWR's) are defined as '….descriptions of the water regimes needed to sustain the ecological values of water dependent ecosystems at a low level of risk' (ARMCANZ and ANZECC 1996).

The determination of environmental water requirements is based on identifying features of the natural flow regime that are considered essential for the maintenance of riverine ecosystems through linking hydrological features to ecological and geomorphological processes.

Most assessments of environment water requirements are based on the natural flow paradigm. That is, to ensure the functional integrity of a river ecosystem the natural flow regime needs to be maintained or key elements of the natural flow regime need to be conserved or restored

Features of natural flow regimes that are considered to be important for maintenance of river features are:

- the natural variability of flows in volume, duration, frequency, and seasonality;
- longitudinal, lateral and vertical connectivity.

The identification of environmental water requirements requires the assessment of biotic components of the river system (e.g. fish, macroinvertebrates and vegetation), the physical components, (e.g. pools, runs, riffles, floodrunners and flood plains), and the determination of the hydrological links to ecological processes (e.g. species life-cycles and migration requirements) and geomorphic processes (e.g. sediment transport, channel maintenance, pool scouring and structural resetting).

River systems in South Australia range from large permanently flowing systems such as the Murray River, to smaller seasonal systems that flow in winter and dry out in summer such as the Wakefield River, to highly variable systems such as the Cooper Creek. Most of the river systems in South Australia are seasonal or ephemeral. EWR assessment programs are particularly challenging in these environments due to the spatial and temporal variability of flow and ecosystem responses.

2. Assessments of EWR's

EWR assessments have been conducted for a number of South Australia's significant river systems. The methods used have generally been short-term assessment conducted over a period of one to two years. These have varied from:

- Rapid desktop studies using existing information to make scientific inferences about environmental water requirements. This method has relied mainly on hydrological modeling e.g. the 'Sustainable Diversion Limits' method.
- Short term assessments using a combination of a multi-disciplinary scientific panel approach (expertise in ecology, geomorphology and hydrology), with some level of field investigations. This approach has been used for most surface water assessments conducted to date, e.g. within the Mid North catchments, the Eastern Mount Lofty catchments, the Onkaparinga River and the Murray River.
- Longer term detailed assessments using quantitative methods and models to predict hydrological and biological relationships. This approach has been used for ARIDFLO project in the Lake Eyre Basin.

The methods vary and are dependent on the available information, attributes of the river system, the specified outcomes of the study and the time and resources involved.

A review of the following EWR programs provides an analysis of the range of methods used in South Australia. It discusses the advantages and limitations of the methods used, but is not a critical analysis of the projects within themselves. An overview of the methods used for assessing environmental water requirements for surface water systems in South Australia is presented in Table 1.

2.1 Hydrological Methods

Hydrological methods are defined as using historical flow to establish a percentage of mean annual flow or a specific hydrologic index that infers an environmental objective. (SKM 2002a)

Initial methods to assess allocations of water to the environment were based on providing minimum flows determined by the flow level that is exceeded 80% of the time (Knights and Fitzgerald, 1994). The selection of the 80th percentile has no scientific validation (SKM 2002a) and can be considered as a simple provision of water and not an EWR assessment. Some planning studies employed this method within a preliminary assessment process but it has never been used to support a formal allocation plan.

2.11 Sustainable Diversion Limits

The Sustainable Diversion Limits method is a more detailed hydrological assessment than using a flow percentiles method and is based on the broad ecological principle of maintaining key elements of the natural flow regime. This method aims to define an upper limit on diversions beyond which there is an unacceptable risk of degradation to the environment (Nathan, Doeg and Voorwinde 2002). The process uses a number of flow indices as a substitute for ecologically significant flow conditions. It is used to set a precautionary level of water resource development until more detailed investigations are completed. This method was trialed on the Marne River.

The advantages of this method are that it is a simple, rapid and low cost desktop assessment. It determines diversionary limits based on a precautionary level of low environmental risk. This method is most applicable for setting preliminary limits in catchments with a low level of water resource development.

The limitations of this method are that the diversion limits are based on broad assumptions of hydrological and ecological links and responses. The output produces an extraction limit but it does not provide an understanding of the functional links between ecology and hydrology within the system. This methodology was developed for Victorian catchments which in comparison to South Australian catchments are characterised by higher levels of rainfall and increased rainfall reliability.

Further refinement of the methodology is required for greater application to more ephemeral river systems.

2.2 Holistic methods

Holistic methodologies are distinguished from single purpose methods in that they aim to assess the flow requirements of the entire 'riverine ecosystem' (Arthington 1998). Holistic methods in South Australia have used scientific panels as a part of the process to integrate information from a wide range of disciplines. The scientific panel draws on experts from a range of ecological and physical process fields to undertake an assessment of the condition of a river system and provide recommendations for the management of the river system.

The scientific panel approach is widely used and has a number of advantages. It is a rapid assessment method and is both a systematic and scientific approach. It can draw on the combined strength of the scientific panel member's experience in situations where little scientific data is available. It is also a flexible approach which can be adapted to

Table 1: Environmental Water Requirement Methods Used for Surface Water Assessment in South Australia

Method Used	Catchments	Key elements	Advantages	Disadvantages
Sustainable Diversion Limits	Eastern Mount Lofty Ranges – Marne River	- Hydrological assessment based on the ecological principle of maintaining key elements of the natural flow regime	- Simple, rapid, low cost desktop method. - Determines water availability based on precautionary level of low environmental risk. - Appropriate for setting precautionary limits in catchments of low current impact.	- Based on assumptions of broad hydrological and ecological links and responses. -Produces an extraction limit but does not provide an understanding of the functional links between ecology and hydrology within the system. - Methodology developed for Victorian catchments based on greater rainfall and rainfall reliability. Further development is required to have greater application to more ephemeral river systems.
Scientific Panel Process	Murray River Lower Lakes and Coorong.	- Integrated approach using specialists to assess ecological, geomorphological and hydrological parameters of the river system. - Focus is providing recommendations for management actions to achieve environmental outcomes.	- Relatively rapid assessment - Sources information from a review of a large number of existing studies. - Integrates information across a range of disciplines. - A detailed assessment is not required to determine broad management actions.	- Strongly reliant on the skills and knowledge of the members on the scientific panel. - Does not identify detailed environmental links to flow regimes. - Not a repeatable assessment due to a reliance on a qualitative panel approach.
Scientific Panel Habitat Assessment Method (SPHAM)	Mid-North Region - Gawler, Light, Wakefield and Broughton Rivers Eastern Mount Lofty Ranges – Marne River	- This approach combines the expertise of specialists to assess ecological, hydrological and geomorphological parameters with field investigations, habitat transects and site assessments to relate habitats, biota and processes to critical flow levels.	- Cost effective method using a scientific panel combined with a minimal field survey assessment process. - Applicable to river systems with little available data. - Integrates information across a range of disciplines. - It provides a transparent method of developing flow benchmarks where future changes can be modeled and subsequent potential impacts identified.	- The habitat requirements of most species are based on untested hypothesis. - This method requires further monitoring and evaluation to ensure that the environmental assumptions are correct. - Reliant on the skills and knowledge of the members on the scientific panel - Assessments of the environmental water requirements are based on uniform geomorphic zones. Assessments of each zone are done at a single representative site, which may not represent all of the features and processes present within that zone.

Method Used	Catchments	Key elements	Advantages	Disadvantages
Flows Method	Western Mount Lofty Ranges – Onkaparinga	-This methodology follows a similar approach to SPHAM with a more detailed hydrological and hydraulic assessment process. -More detailed hydrological and hydraulic assessment process. -Sets flow and biodiversity objectives for watercourses and -Recommends flow scenarios incorporates testing of some ecological and geomorphological hypothesis -The development of a monitoring and evaluation program.	- Provides detailed hydrology and hydraulic assessment to improve spatial and temporal links to ecology and geomorphology. - Used multiple sites within a geomorphic zone which increases scientific rigour and confidence in developing recommendations. - It defines flow components and quantifies the hydrological links to geomorphological and ecological processes that vary spatially through the river system. -Tests some major hypotheses of hydrological influence on geomorphology and ecology. -Provides a transparent framework that allows repeatability.	- Has similar issues to the SPHAM approach in relation to reliance on environmental hypothesis and the scientific panel. - The application of this methodology requires good spatial and temporal hydrological data. - Hydrological tools used have been developed for a range of purposes i.e. operational requirements and therefore can be difficult to apply to ecological systems.
ARIDFLO	Lake Eyre Basin - Cooper Creek / Thomson, Diamantina / Warburton , Neales and Peake Rivers	-A multi disciplinary research project that combines scientific field studies with the development of rainfall run off and flow pattern models to predict hydrological and biological relationships for Australian arid zone rivers. - has developed grid based hydrological models to generate flow data - collected detailed biological data for a range of biotic groups - Statistical modeling of hydrological and biological relationships with the aim of developing predictive tools.	- Hydrological method is designed specifically for the purpose of generating ecologically significant hydrological data. - A greater understanding of the physical and ecological processes of the river system is possible due to: - Detailed biological assessment of the aquatic food chain, algae, zooplankton, macro-invertebrates, fish and birds. - A large number of survey sites. As the assessment of sites is repeated at regular time intervals, variations in hydrological regimes can be linked to biological responses. - Ecological assessment is based on bio-community analysis rather than reliance on specific species response. - Is highly quantitative and provides a comprehensive database which can inform subsequent monitoring and more specialised studies	- Requires a large amount of resources for field work - Requires high level of scientific expertise - The huge spatial and temporal variability of arid river systems means that there: - are unique challenges for statistical analyses - is a need for longer term data sets - Due to the spatial extent and variability accurate hydrological modeling is difficult.

the methods, techniques and information available (Cottingham, Thoms and Quinn, 2002). If properly applied the process is transparent and documents assumptions, hypotheses, knowledge gaps, conclusions and recommendations.

This method can provide a sound basis for developing water management rules, future investigations and monitoring programs.

2.21 River Murray Environmental Flows (Scientific Panel Assessment)

The vision of the MDBC River Murray Environmental Flows and Water Quality Objectives project is:
".... a healthy River Murray system, sustaining communities and preserving unique values." (MDBC 2002)

The Murray-Darling Basin Commission environmental flows program has conducted a number of projects to address water for the environment issues. Initially scientific panel assessments were conducted throughout the Murray and Lower Darling River systems. Following this an independent expert reference panel (ERP) was convened to develop a risk-based approach for identifying appropriate flow scenarios, based on probabilities of restoring a healthy river system.

At the state level, the River Murray Catchment Water Management Board and DWLBC have developed a Decision Support System (DSS) to facilitate the management of future flow events within South Australia to maximise environmental benefits (SKM 2002b). The development of the DSS has included input from stakeholders in prioritising environmental outcomes in the development of flow scenarios.

This paper will focus on the methodology used for the initial EWR assessments, as the latter two are not concerned with assessing EWR's but in the development of environmental water provisions.

The initial process of undertaking EWR assessments was based on a scientific panel approach. This involved two studies:

- 'Report of the River Murray Scientific Panel on Environmental Flows - River Murray - Dartmouth to Wellington and the Lower Darling River' (Thoms et. al 2000)
- 'River Murray Barrages Environmental Flows – an evaluation of environmental flow needs in the Lower Lakes and Coorong' (Jensen et. al 2000).

These studies essentially followed the same methodological framework as outlined below.

The scientific panel was comprised of a number of experts assessing the following parameters: Hydrology; geomorphology, vegetation, macroinvertebrates, fish, birds (in the barrages study) and river operations.

The assessments were conducted through a rapid field assessment supported by information provided from previous studies and the analysis of natural and current flow regimes. Within this program there was no formal scientific investigation process conducted to gather additional physical and ecological data.

The outcomes of the studies included the provision of broad management recommendations based on ecosystem health principles for river reaches and important ecological areas. The report identified both short-term actions and longer-term recommendations (involving a range of options) to improve the environmental flow regime of the River Murray. These studies focused mainly on the hydrological manipulation of flows within this system.

The advantages of this methodology are that it uses an integrated approach drawing upon a range of specialist fields to assess ecological, geomorphological and hydrological parameters of the river system. This method was a relatively rapid assessment as it sourced information from a review of a large number of existing studies and did not require detailed field assessments for the development of broad management actions.

The limitations of this methodology are that due to a lack of quantitative field data the project outcomes are strongly reliant on the skills and knowledge of the members on the scientific panel. Concurrently the reliance on the panel members also means that the assessment is more qualitative and not a scientifically repeatable process. This method is designed as a broad scale assessment approach and as such does not make detailed quantitative links between the hydrology and ecological responses.

2.22 Mid-North Environmental Flows (SPHAM)

The Mid-North Rivers Management Planning Project focused on the assessment of three ephemeral river systems, the Light, Wakefield and Broughton Rivers in the Mid-North Region of South Australia. The environmental flows component of this project aimed to determine the water requirements necessary to maintain the ecological health of the river and its major tributaries (EPA, unpublished). In doing so the project sought to develop flow benchmarks that could be used for future monitoring programs.

The method used incorporated key aspects of the Scientific Panel Assessment Method (Arthington et. al, 1996); the Habitat Assessment Method (Arthington, Brizga and Kennard, 1998); and the Ecosystems Approach (Burgess and Thoms, 1998).

The method has been named the Scientific Panel Habitat Assessment Method (SPHAM) (Favier, Rixon and Scholz 2000). The approach combines the expertise of specialists to assess ecological, hydrological and geomorphological parameters with habitat assessments at sites representative of each geomorphic zone.

The scientific panel assessing the environmental water requirements of the catchments comprised experts in the fields of hydrology, geomorphology, macroinvertebrates and fish ecology.

As part of the process, river systems were classified into uniform geomorphic process zones, and for zones where sufficient data was available, hydrological models were developed.

To provide information on the current state of the catchment, investigations of geomorphology, fish and macroinvertebrate populations were conducted for the study area.

Habitat transects at representative sites in each geomorphic zone were used to relate habitats, biota and processes to critical flow levels. In addition, the scientific panel undertook an in-field assessment of flow requirements at these representative sites.

A scientific panel workshop process was conducted to integrate information collated from desktop studies and field surveys across all disciplines. The panel developed hypotheses regarding ecology-flow and geomorphology-flow relationships that were used as the basis for identifying environmental flow requirements for each geomorphic zone. This determination of environmental water requirements involved:

- Identifying natural and current flow levels.
- Determining the environmental water requirements for specific ecological species and geomorphic processes for each zone.
- An assessment as to whether these flow requirements are still being met under the current flow regime.
- Identifying assets, threats and management issues.

The advantages of this method are that it is a cost-effective approach that combines a scientific panel with a process of gathering specific environmental data at key sites that are representative of the catchment. The method clearly outlines the hypotheses and assumptions used to determine the final environmental water requirements. This method is flexible and adaptable and can be applied to river systems that have little available data.

This method (in the latter stages of its development, (EPA, unpublished) provides a logical, transparent process of building and linking baseline environmental information to determine key flow components of the river system and their links to environmental responses. This method sets preliminary flow benchmarks where future changes can be monitored.

The limitations of this methodology are that it is reliant on the skills and knowledge of the members on the scientific panel. Although data collection provides site specific information the flow requirements of most species are based on untested hypotheses. Therefore the flow recommendations require a validation and refinement process through research, monitoring and evaluation to ensure that the physical and ecological assumptions used to define the key flow requirements are correct.

In addition, assessments of the environmental water requirements of the catchment are based on uniform geomorphic zones. Assessments of each zone are done at a single representative site, which may not represent all of the features and processes present within that zone.

2.23 Onkaparinga EWR Assessment (FLOWS)

The Onkaparinga Catchment Water Management Board (OCWMB) conducted an EWR assessment program as a requirement under the State Water Plan 2000. The aim of this project is to provide a scientific basis for the OCWMB to implement provisions for water dependent ecosystems, and in doing so, meet the object of the *Water Resources Act 1997* and the objectives contained in the Onkaparinga Catchment Water Management Plan (SKM 2002a).

To assess the environmental water requirements for the Onkaparinga River catchment the consultants Sinclair Knight Mertz (SKM) employed the FLOWS method (SKM 2002c). This methodology follows a similar approach to SPHAM and addresses the same components as discussed in the SPHAM methodology. The FLOWS method is also a flexible and adaptable methodology and in the case of the Onkaparinga project progressed further than the SPHAM method in four areas,

- it has a more detailed hydrological and hydraulic assessment process,
- it sets flow and biodiversity objectives for watercourses and recommends flow scenarios to achieve this
- incorporates testing of some ecological and geomorphological hypothesis to support management recommendations.
- the development of a monitoring and evaluation program.

The strengths of this method are through a detailed hydrologic and hydraulic assessment that defines flow components and quantifies the hydrological links to geomorphological and ecological processes that vary spatially and temporally through the river system.

This methodology provides a transparent framework and combined with a strong level of supporting data, is repeatable.

This process uses multiple assessment sites within a geomorphic zone that increases scientific rigour and confidence in developing recommendations.

The limitations of this method are similar to the SPHAM approach in relation to reliance on environmental hypotheses and the knowledge and experience of the scientific panel. However this method employs a clearer scientific process.

The application of this methodology is most appropriate in areas with good spatial and temporal hydrological data. The hydrological tools used in this method have been developed for a range of purposes. The HEC-RAS hydrological model has been developed for operational requirements and therefore can be difficult to apply to ecological assessment.

2.3 Quantitative Methods

Lake Eyre Basin EWR Assessment (ARIDFLO)

The Environmental Flow Requirements for Australian Arid Zone Rivers Project (ARIDFLO) is a multi-disciplinary research project on selected rivers in the Lake Eyre Basin. It aims to develop models of hydrology-biology relationships for Australian arid zone rivers (Costelloe 2002). This method is a more quantitative approach to assessment than those previously discussed.

The project involves field studies of fish, waterbirds, macroinvertebrates, zooplankton, riparian plants and water quality at 50 waterbodies in five reaches of three catchments of the Lake Eyre Basin.

Grid-based hydrological models (modeling rainfall-runoff and flow routing) were developed for the river reaches in the study area to determine spatial flow patterns. These models use interpolated rainfall data, gauging data, flow gaugings, satellite imagery and landholder observations.

The models were used to generate daily-time step flow data for each of the waterbodies using satellite imagery combined with weather and flow stations and geomorphological data. Statistical modeling was then used to relate 60 hydrological parameters (extracted from the modeled flow data) and 28 environmental/water quality parameters to the biological data including parameters such as species richness, abundance's, disease incidence, breeding status, migration directions etc.

The outcomes of this program will be the development of the ARIDFLO model as a predictive model in the assessment, management and monitoring of arid zone river systems.

The strengths of this method are that the hydrological method has been designed specifically for the purpose of generating ecologically relevant hydrological data.

This method allows greater understanding of the hydrological linkages to the physical and ecological functions, and processes of the river system, due to:

- a detailed biological assessment of the aquatic food chain, algae, zooplankton, macro-invertebrates, fish and birds.
- a large number of survey sites.
- the repeat assessment of sites at regular time intervals, which allows variations in hydrological regimes to be linked to biological responses.
- basing the ecological assessment on bio-community analysis (through nesting of data sets) rather than reliance on specific species response.

This method is highly quantitative and provides good scientific information to test ecosystem response hypothesis. It also provides a comprehensive database which can inform subsequent monitoring and more specialised studies.

The limitations of this method are that it requires high level of scientific expertise and a large amount of resources and time to conduct the field assessments.

The huge spatial and temporal variability of arid river systems presents unique challenges for statistical analyses. Longer-term data sets are needed to support the findings for programs in arid environments.

3 Lessons learnt from environmental water requirement assessments programs in South Australia

Over the last 5 years much has been learnt about the application of EWR assessment projects in South Australia. EWR assessments are conducted for a range of purposes requiring various outcomes. Firstly we need to clearly scope the project and tailor outputs to achieve specified goals. Secondly we need to recognise the limitations of the various methods and the current information/knowledge gaps. Finally the outputs of the assessment need to be scientifically defensible and provide a foundation for future work.

3.1 Adequate scoping of the Project

The selection and development of an appropriate assessment methodology is dependent upon clearly defining the scope and outcomes of a project.

Initially there is a need to define the objective or vision for the river system. Many of the streams in South Australia are ephemeral and alluvial and so can be subject to a continual state of adjustment due to cyclical changes in variability of flood regime and do not adjust to a regular flow regime (Gipple 2002). Also many regulated rivers lack the natural variations to maintain the diversity of communities they once supported. Therefore the goals may vary from maintenance of current flora and fauna populations, to restoration of pre European condition, to endangered species protection etc. To further scope the project the following points need to be addressed:

- the outputs needed to achieve the management objectives
- the size of the catchment area
- the scale of information you need to collect
- the level of data available
- the time limits
- the funds / resources available

3.2 Limitations of the scientific panel approach

Scientific panel approaches have played a cornerstone role in EWR assessment projects in South Australia. The key strength is the ability to draw together knowledge and experience from a range of environmental fields and conduct a relatively rapid, cost-effective assessment in comparison to undertaking empirical investigations. However, in lieu of good technical data this methodology can be heavily reliant upon the knowledge and experience of the members of the scientific panel. For this reason the results from EWR assessments can vary from simple qualitative 'Black box ' answers to transparent processes that build on empirical data and tested hypothesis.

Scientific panel assessment projects are generally developed as a snap shot of the catchment and have a limited ability to assess the temporal and spatial variability. EWR assessment programs need to develop a framework for longer term monitoring and evaluation.

3.3 Knowledge and Information Gaps

There are a number of significant knowledge and information gaps that limit the effectiveness of EWR assessment programs in South Australia and nationally.

The outcomes of these assessments are heavily dependent upon the level of meteorological and hydrological data available. The ability for analysis of the hydrology of South Australian catchments is limited by the shortage of available spatial and temporal hydrological data.

Many of the watercourses in South Australia can be described as ephemeral river systems. Further investigation is needed to determine the role of the wet and dry cycles and the succession and timing of flood events on the maintenance and recovery of water dependant ecosystems.

There is a heavy reliance on the ecological assumptions and hypothesis used as the basis for EWR assessments, however many of these have not been scientifically proven. There is a need to test and verify these hypotheses.

In developing EWR's there is difficulty in identifying the quantitative links between flow regimes and ecological and physical responses, and the threshold at which impacts will occur.

There is a lack of historic baseline information and research on the ecological and biological functions of species in South Australian environments. This has repercussions in applying ecological principles in their regional and local context.

Traditionally EWR assessments have focused on using species (eg. River Blackfish) as specific indicators of river health which can be misleading. Flow and ecological responses tend to be species specific. There is a need look towards developing EWR's for bio-community groups, to ensure a wider range of ecosystem responses are met.

The problem of identifying a specific species response to flow is that their response can be affected by a number of other environmental parameters and links e.g. predation, habitat condition and temperature. Further investigations are needed to address these links within a larger context.

Many of the hydrological models available and the sites selected for recording hydrological data are been developed for engineering and operational requirements and are difficult to apply to ecological assessments.

3.4 Scientific Outcomes

To produce scientifically defensible outcomes EWR methods that use scientific panels need to ensure quality control of the process i.e. protocols for conducting workshops, guidelines for the process of investigation, a rigorous process for assessment. Reliance on expert opinion requires that the process is transparent, that the logic supporting the assumptions is well documented and that the information provided can be built upon in future studies.

4. Conclusion

In developing EWR assessment programs there are a range of methods that can be applied depending on the outcomes required, the information available, the appropriate spatial and temporal scales and the resources available.

There are also challenges that are relevant across all methods. EWR assessment programs are particularly challenging in ephemeral river systems due to the spatial and temporal variability of flow and ecosystem responses. Our current knowledge of the ecological and geomorphological responses to flow regime is limited and our estimates of the environments water requirements may need refining. Therefore the methods we use to develop EWR assessments need to be scientifically defensible, repeatable, transparent, provide a solid base of knowledge and be adaptive to new information.

References

ARMCANZ and ANZECC. (1996). National principles for the provision of water for ecosystems. Occasional paper SWR No3, Sustainable Land and Water Resources Committee, Commonwealth of Australia, Canberra.

Arthington, B.A, Pusey, B.J. and Kennard, M. (1996) Ecological basis for river habitat and in-stream flow management. Milestone 2 Report, 1995, LWRRDC GRU-10. Unpub.

Arthington, A.H, (1998) Comparative evaluation of environmental flow assessment techniques: review of holistic methodologies. Occasional paper 26/98, Land and Water Resources Research and Development Corporation, Canberra.

Arthington, AH, Brizga, SO, and Kennard, M.J. (1998) Comparative evaluation of environmental flow assessment techniques: best practice framework. Occasional paper 25/98, Land and Water Resources Research and Development Corporation, Canberra.

Burgess, G.K., and Thoms, M. (1998) Environmental flow Management in Queensland Rivers. In CRCFE-AWWA Catchment Management Course.

Costelloe,J; Puckridge, J; Reid, J., Pritchard, J., Hudson, P., Bailey, V and Good, M. (2002) Environmental flow requirements in arid zone rivers – a case study from the Lake Eyre Basin, central Australia. (in Press)

Cottingham, P., Thoms M.C. and Quinn G.P. (2002) Scientific panels and their use in environmental flow assessment in Australia. In Australian Journal of Water Resources **5** (1): 49 – 59.

DWLBC (2002) National River Health Program Environmental Flows Initiative Environmental flow requirements for Australian arid zone rivers (ARIDFLO) Progress Report - Year 2. DWLBC, Adelaide

EPA (unpublished) A river management plan for the Light catchment. EPA, Adelaide

Favier, D., Rixon, S. and Scholz, G.D, (2000) A River Management Plan for the Wakefield River.” Department for Environment and Heritage. Adelaide.

Gippel, C.J. (2002) Geomorphic issues associated with environmental flow assessment in alluvial non-tidal rivers. In Australian Journal of Water Resources **5** (1): 49 – 59.

Jensen, A., Good, M., Harvey, P., Tucker, P., amd Long, M. (2000) River Murray Barrages Environmental Flows – an evaluation of environmental flow needs in the Lower Lakes and Coorong’. MDBC Canberra

Jones, G., Hillman, T., Kingsford, R., McMahon, J., Walker, K., Arthington, A., Wittington, J. and Cartwright, S. (2002) Independent report of the expert reference panel on Environmental flows and water quality requirements for the River Murray system. CRC Canberra.

Knights, P and Fitzgerald, B, (1994). A Pragmatic Approach to Environmental Flow Management in proceedings Australian Water Association Environmental Flow Seminar, Canberra p 131-141.

MDBC (2002) The living Murray - discussion paper on restoring the health of the River Murray. MDBC July 2002.

Nathan, R.J. Doeg, T and Voorwinde, B.E. (2002) Towards defining sustainable limits to winter diversions in Victorian catchments. In Australian Journal of Water Resources **5** (1): 49 – 59.

SKM (2002a) Determination of Environmental Water Requirements of the Onkaparinga River Catchment. Onkaparinga Catchment Water Management Board.

SKM (2002b) Decision Support System for Managing Flows in the River Murray, South Australia (draft). SKM

SKM (2002c) FLOWS – a method for determining environmental water requirements in Victoria. Dept.of Natural Resources and Environment. Melbourne

Thoms, M.C., Suiter, P., Roberts, J., Koehn, J., Jones, G., Hillman, T and Close, A. (2000) Report of the

River Murray Scientific Panel on Environmental Flows - River Murray - Dartmouth to Wellington and the Lower Darling River. MDBC Canberra

Review of SA Approaches in a National Context – Surface Water

Dr Sandra O. Brizga
S. Brizga & Associates Pty Ltd
PO Box 68, Clifton Hill Vic. 3068

SUMMARY: Many methodologies for determining environmental water requirements (EWRs) now exist, including a considerable number that have been developed in Australia. Apart from the hydrological index methods sometimes used for preliminary or rapid assessments of EWRs, most methodologies currently used in Australia are holistic in their scope, recognising that it is necessary to provide for ecosystems in their entirety. Variations in planning contexts have resulted in differences in approaches to EWR objective setting and the role of scientists in this task. All methodologies need to address two key questions: 1) which flows are important? and 2) how much of each flow is needed? The former question can be addressed on the basis of the flow requirements of specific organisms or processes and/or packages of flow indicators with broader ecological relevance. Hydrological or hydraulic indicators may be used in specifying significant flows. The latter question is usually addressed using best professional judgement, in some cases supported by quantitative predictive models and/or risk assessment models. Experimental data can also be used, although local experimental data is rarely available.

THE MAIN POINTS OF THIS PAPER

- The majority of EWR methodologies currently in use in Australia are holistic methodologies that aim to consider entire riverine ecosystems.
- Numerous EWR methodologies exist, but they have many common elements and all need to address two fundamental questions:
    - Which flows are important?
    - How much of each flow is needed?

1. INTRODUCTION

Environmental water requirements (EWRs) became widely recognised as an issue in Australia in the 1980s, when concerns about the impacts of flow regulation and water abstraction prompted efforts to identify and preserve key components of the flow regime for the survival of aquatic species. The Council of Australian Governments (COAG) made EWRs an important component of the national water reform agenda by recognizing the environment as a legitimate water user and stipulating that "EWRs to sustain the ecological values of aquatic ecosystems at a low level of risk" must be assessed and provided (ARMCANZ and ANZECC 1996). The State Governments have approached the water reform process in a variety of different ways, leading to the development of a range of methodologies for determining EWRs tailored to meet the requirements of their various planning frameworks. These differ amongst the states for a number of reasons, including the extent of natural surface water resources, current levels of development, scale (of the state and individual river basins) and levels of resourcing.

As a result of these historical factors, a number of Australian EWR methodologies now exist. They include the Holistic Approach (Arthington et al. 1992), Expert Panel Assessment Method (Swales and Harris 1995), Scientific Panel Assessment Method (Thoms et al. 1996), Flow Restoration Methodology (Arthington et al. 2000), Benchmarking Methodology (DNR 1998, Brizga 2000, Brizga et al. 2000a, 2000b, Brizga et al. 2001a, 2001b, Brizga et al. 2002), Scientific Panel and Habitat Assessment Method (SPHAM – EPA [2000]), Flow Events Method (Stewardson and Cottingham 2002), FLOWS (Shirley 2001, Howell and Doeg 2002) and ARIDFLO (Costelloe et al. 2002). A number of overseas methodologies have influenced Australian approaches, including the Instream Flow Incremental Method (IFIM) (Bovee 1982) and the associated PHABSIM and RHYHABSIM models, the South African Building Block Methodology (King and Louw 1998, Arthington and Long 1997, Arthington and Lloyd 1998) and Downstream Response to Intended Flow Transformations (DRIFT).

The scale of intended use of these methods covers all three categories identified by Arthington et al. (1998a), i.e. catchment-scale, subcatchment-scale and reach-scale, so different methodologies are appropriate in differing situations. The have also been designed within the context of a range of different planning frameworks, and for varying purposes ranging from the definition of bulk water entitlements to fine-tuning of regulated systems. A number of reviews of EWR methods have been published, including a major Australian review

(Arthington 1998, Arthington and Zalucki 1998) which identified a best practice framework (Arthington et al. 1998a) and R&D requirements (Arthington et al. 1998b). A collection of papers documenting some more recent developments has been published as a special issue of the Australian Journal of Water Resources (Volume 5, No 1. 2002).

This paper aims to demystify the process of determining EWRs. It does not provide descriptions of individual methods, but instead provides an overview of key elements common to most currently-used methodologies, which include:

- the use of a holistic approach;
- a process for determining environmental objectives for EWRs; and
- a focus on two fundamental questions:
    - Which flows are important?
    - How much of each flow is needed?

2. HOLISTIC APPROACH

Hydrological index methods are sometimes used for preliminary or rapid assessments of EWRs. Otherwise, most methodologies currently in use in Australia recognise that EWRs need to consider riverine ecosystems in their entirety, and recent South Australian studies are no exception (Scholz, this volume). However, the methods vary in the extent to which this is applied. The Benchmarking Methodology used in Queensland's Water Resource Planning (WRP) process has a broad scope. Ecosystem components that are considered include geomorphology, hydraulic habitat, water quality, riparian and aquatic vegetation, macroinvertebrates, fish and other vertebrates (amphibians, reptiles, birds, mammals). Other methods tend to focus on a smaller number of parameters, often macroinvertebrates and fish (Stewardson et al. 2002).

The terms EWR and "instream flow requirement" used to be used interchangeably. It is now generally recognised that, because riverine ecosystems can also encompass riparian zones, riverine wetlands, estuaries and nearshore marine ecosystems, EWRs may need to cover this broader area. Some of the EWR methodologies listed in the introduction can accommodate this wider spatial scope. Specialist methodologies for dealing EWRs in particular situations other than non-tidal river channels, such as estuaries and wetlands, are also under development.

Multidisciplinary teams are generally used in EWR studies so as to provide the necessary breadth and depth of knowledge. Workshops are commonly used to promote synergy between team members and integrate their various contributions.

3. OBJECTIVE SETTING

EWRs are ultimately determined by the desired outcome. Environmental objectives for EWRs can be developed in a number of different ways, depending on the planning context. For example, the purpose of an EWR study in a highly developed river system is likely to be to identify opportunities for making relatively minor changes to the flow regime to maximise ecological benefit without unduly compromising consumptive use commitments. Objectives in this context are likely to focus on specific improvements in geomorphological and/or ecological condition that would be sought from the EWR provision. On the other hand, in a river system with little existing development but high potential for future consumptive use development, the development of EWR objectives is likely to be linked to the determination of limits to acceptable development, which is a much more subjective and open-ended question, with potentially far-reaching social and economic ramifications. The approaches to objective setting used in the FLOWS methodology (as described by Howell and Doeg 2002) and Benchmarking Methodology are outlined below to provide examples of how objective-setting can be approached in these differing contexts.

In the FLOWS methodology, which has been designed for application in the development of Victorian Streamflow Management Plans, biodiversity objectives that specify desired future condition are developed by the scientific panel at an early stage in the process, taking into account existing condition and values, opportunities to maintain, restore or rehabilitate the riverine ecosystem, and other relevant planning frameworks. The biodiversity objectives are used as a basis for determining flow objectives, which are then tested in relation to their impacts on supply. If the flow objectives preclude supply objectives from being fully met, trade offs and risks of scaling back EWRs to meet supply objectives are examined. This information is then used by the parties involved in the development of a Streamflow Management Plan to negotiate an appropriate flow management strategy.

In the Benchmarking Methodology, which has been designed for application in the Queensland WRP process, EWR objectives are set by Government, as the elected representatives of the community, taking into consideration environmental, social and economic considerations. The role of the scientific panel is to provide advice regarding current environmental conditions and values, and the likely implications of a range of possible future water resource management scenarios for these conditions and values. The Government also receives advice on the likely social and economic implications of the scenarios, and extensive community consultation is carried out.

4. WHICH FLOWS ARE IMPORTANT?

Approaches to specifying important flows can be divided into two broad categories: 1) approaches that specify the flow requirements of particular organisms or processes and 2) packages of flow indicators with broader ecological relevance. Both approaches are sometimes used together so as to ensure coverage of the full breadth of relevant ecosystem functions while paying particular attention to the requirements of key species. Flows may be specified hydrologically or hydraulically.

4.1 Flow Requirements of Specific Organisms

Direct linkages have been identified between specific components of the flow regime and some particular species or processes – for example, flow requirements for the movement and reproduction of particular fish species. Such information can be used to examine the implications of flow regime modifications for particular species, and enables the development a flow regime that provides suitable or optimal conditions for target species. Detailed assessments along these lines are particularly appropriate in situations where EWR provisions need to be made for species of high conservation value.

There are several drawbacks to using the flow requirements of specific organisms as a basis for determining key flow parameters for EWRs more generally:

- a lack of suitable information for many species – for example, Roberts (2002) noted that existing information regarding the flow requirements of riverine and riparian plants was confined to a very limited number of species and that, given the large number of riverine and riparian plant species that exist, it would hardly be feasible to determine EWR requirements for each one;
- variations in flow requirements amongst different populations of the same species, in response to natural local or regional variations in flow regime; and
- flow conditions in a particular river system may provide naturally marginal habitat for its indigenous biota.

4.2 Packages of Ecologically-Relevant Flow Indicators

Many early attempts at defining EWRs focused exclusively on low flows. Some of the earliest studies defined EWRs in terms of a single low flow, but later studies define different flows for each season or month, so as to reflect natural seasonal variability. It is now widely accepted that riverine ecosystems are influenced by the entire flow regime that they experience. This includes flows of all magnitudes, from the smallest low flows to the largest of floods, as well as the incidence of no flows. A change in any aspect of the flow regime is likely to have some geomorphological and/or ecological effects. Geomorphological and ecological responses to flow regime change are interdependent – a change in one component of the ecosystem is likely to have flow-on effects for other components.

Six major categories of flow characteristics have been identified as being of particular ecological relevance, as well as being sensitive to the changes produced in flow regimes by impoundment, diversions, groundwater pumping, use of water for hydropower generation and catchment land-use changes (Brizga and Arthington 2001). Important flow characteristics are the magnitude of river flows at any given time, the timing of occurrence of particular flow conditions, the frequency of occurrence of particular flows such as flood flows, the duration of time over which specific flow conditions extend, the rate of change in flow conditions such as rise and fall of flood waters, and the seasonality and predictability of the overall flow regime.

Recent approaches tend to propose relatively comprehensive sets of flow indicators that address many of these flow characteristics. For example, in the environmental flow studies for the Barron, Pioneer and Burnett Basin WRPs, Brizga and her colleagues proposed a suite of key flow indicators falling into the following five categories: annual variability, seasonal variability, zero flow, low to medium flow, high flow and also outlined environmental assessment criteria for other aspects of the flow regime such as hydrograph rise and fall rates (Brizga 2000, Brizga et al. 2001a, 2001c). In South Australia, environmental flows for the Broughton and Light Rivers have been specified in relation to a range of flow bands, including groundwater, baseflow, low flow, mid flow, high flow, bankfull flow, overbank flows and catastrophic flows. A similar range of flow bands has been proposed for the Onkaparinga (SKM 2001).

It is important that the linkages between any flow indicators and their associated geomorphological and ecological functions are understood, as it must be possible to assess the implications of flow regime modification for the riverine ecosystem. Conceptual models provide a suitable framework for compiling and presenting this information. They should be underpinned by rigorous scientific information where available.

Instantaneous flows are the drivers of geomorphological and ecological functions, but for modelling to be feasible, a coarser time-step must usually be adopted. Mean daily flows generally provide a reasonable approximation of the instantaneous flow regime, except in rivers with flashy flood peaks, or in situations where there are human-induced within-day fluctuations in flow (e.g. due to synchronisation of pumping or hydropower releases). Monthly data are generally less appropriate for environmental flow studies than daily data as they mask geomorphologically and ecologically significant variations in flow, making it much more difficult to link flows with their geomorphological and ecological functions.

4.3 The Use of Hydraulic Information

Flows are linked to their geomorphological and ecological functions via their hydraulic properties, such as depth, wetted perimeter, velocity, pool flushing rates, shear stress or frequency of overbank flooding. Flow hydraulics are determined by hydrology, topography, and downstream controls (e.g. bed topography, channel shape, valley shape, main stream water levels in the case of a tributary, or tidal effects in the case of an estuary or reach just above it). Substantial variations in hydraulic conditions can occur within a single reach (Craigie et al. 2000). Flows of the same volume may have completely different geomorphological and ecological functions depending on hydraulic conditions.

Hydraulic information is used to varying degrees in environmental flow studies, depending on the scale of the study and the level of resourcing. Usually relatively simple one-dimensional techniques are used. Two- and three-dimensional models can provide much more comprehensive information about hydraulic conditions, but these advanced modelling tools have generally not been drawn upon in environmental flow studies due to cost constraints. Hydraulic assessments in environmental flow studies are often limited to a single “representative” site (e.g. one or two riffles or riffle-pool sequences) in each reach, again due to resourcing constraints.

5. HOW MUCH OF EACH FLOW IS NEEDED?

Professional judgement is almost invariably used to some degree in addressing the question of how much flow is needed. Quantitative models and risk assessment models are drawn upon in some methodologies, but such models seldom, if ever, provide a stand-alone basis for EWR determination – at the very least, professional judgement is required in model application. Experimental data have also been drawn upon in some EWR studies, but generally there are few opportunities for experimentation in the course of EWR determination.

5.1 Best Professional Judgement

Best professional judgement is used in the application and interpretation of quantitative models or risk assessment models in EWR studies or, in situations where no such models are available, as a stand-alone basis for determining EWR requirements. The latter approach is better suited to situations where opportunities for flow regime modification are confined within narrow parameters (e.g. fine-tuning of regulated systems). It is more problematic in more open-ended situations, such where sustainable development limits being sought. Scenario comparisons provide a means of focusing assessments in such situations.

The natural flow regime is widely regarded as being the optimum one from an environmental viewpoint in most instances. Exceptions may occur in situations where a riverine ecosystem is already highly modified due to other factors, or where it has been exposed to a modified flow regime for so long that it has fully adjusted to the new regime and reversion to the natural regime may threaten existing values. According to the natural flows paradigm, the purpose of EWRs is regarded as being to conserve key elements of the natural flow regime, or mimic or restore them.

5.2 Quantitative Models

Ideally, geomorphological and ecological implications of flow regime modification and their interactions would be tested using a fully integrated quantitative predictive whole-of-ecosystem model. However, such models do not currently exist. The level of resourcing that would be required to develop such models using currently available information and technologies would greatly exceed the resources usually provided for EWR studies in Australia.

Quantitative models that describe associations between flow and geomorphological or ecological parameters are available for some ecosystem components. For example, hydraulic geometry models can be used to provide an indication of likely net change in channel size resulting from flow regime change (Brizga et al. 2000b). Sediment transport models can provide an indication of likely implications of flow regime change for sediment transport. Statistical relationships between estuarine fisheries catches and total wet season flow in a given year have been established for a number of rivers in Queensland (e.g. Loneragan and Bunn 1998). The ARIDFLO project identified statistical associations between flows and a range of parameters describing selected abiotic and biotic ecosystem attributes in the Cooper Creek Basin (Costelloe et al. 2002).

The range of available quantitative models is generally too narrow to provide a comprehensive basis for EWR determination; therefore these types of models generally need to be used in conjunction with other methodologies. Many of the ecological models are black box models and the linking processes are not yet well understood. Quantitative models of secondary effects of flow regime change (e.g. impacts of channel contraction for vegetation and instream biota) are generally not available.

5.3 Risk Assessment Models

Risk assessment models that show levels of geomorphological and ecological risk associated with various degrees of flow regime change are an integral component of the Benchmarking Methodology. They are developed on the basis of assessed flow-related impacts on condition at benchmarking sites subject to a range of levels of flow regime change. Levels of flow regime change associated with only minor flow-related impacts on geomorphological and ecological condition at benchmarking sites are judged as being low-risk, while levels of flow regime change associated with major flow-related impacts are judged as being high risk. As there is a high degree of natural variability in terms of sensitivity to flow regime change, just because a level of flow regime change is judged as high risk, it does not necessarily mean that all sites exposed to this level of flow regime change will necessarily undergo the same degree of impact. Broad guidelines have been developed regarding factors that influence sensitivity (Brizga et al. 2001c).

5.4 Experiments

Experimental manipulation of flows can provide useful information for informing EWR assessments, and some experimental releases from dams have been made in this context. There are few, if any, opportunities for experimentation in unregulated systems and the high cost of water has precluded widespread experimentation in regulated systems. Infrastructural constraints also limit the scope of experimentation that is possible. A number of people have argued that EWRs can be regarded as experiments and have argued for rigorous and comprehensive monitoring.

6. REFERENCES

ARMCANZ and ANZECC (1996). National Principles for the Provision of Water for Ecosystems. Occasional Paper SWR No. 3. ARMCANZ and ANZECC, Canberra.

Arthington, A.H. (1998). Comparative Evaluation of Environmental Flow Assessment Techniques: Review of Holistic Methodologies. LWRRDC Occasional Paper 26/98.

Arthington, A.H. and Lloyd, R. (Eds.) (1998). Logan River Trial of the Building Block Methodology for Assessing Environmental Flow Requirements: Workshop Report. Centre for Catchment and In-stream Research, Griffith University, and Department of Natural Resources, Brisbane, Queensland.

Arthington, A.H. and Long, G.C. (Eds.) (1997) Logan River Trial of the Building Block Methodology for Assessing Environmental Flow Requirements: Background Papers. Centre for Catchment and In-stream Research, Griffith University, and Department of Natural Resources, Brisbane, Queensland.

Arthington, A.H., and Zalucki, J.M. (Eds.) (1998). Comparative Evaluation of Environmental Flow Assessment Techniques: Review of Methods. LWRRDC Occasional Paper 27/98.

Arthington, A.H., King, J.M., O'Keefe, J.H., Bunn, S.E., Day, J.A., Pusey, B.J., Bluhdorn, D.R. and Tharme, R. (1992). Development of an holistic approach for assessing environmental flow requirements of riverine ecosystems. In Pigram, J.J. and B.P. Hooper (eds) Proceedings of an International Seminar and Workshop on Water Allocation for the Environment. Centre for Water Policy Research, University of New England, Armidale, 69-76.

Arthington, A.H., Brizga, S.O. and Kennard, M.J. (1998a). Comparative Evaluation of Environmental Flow Assessment Techniques: Best Practice Framework. LWRRDC Occasional Paper 25/98.

Arthington, A.H., Pusey, B.J., Brizga, S.O., McCosker, R.O., Bunn, S.E. and Growns, I.O. (1998b). Comparative Evaluation of Environmental Flow Assessment Techniques: R&D Requirements. LWRRDC Occasional Paper 24/98.

Arthington, A.H., Brizga, S.O., Choy, S.C., Kennard, M.J., Mackay, S.J., McCosker, R.O., Ruffini, J.L. and Zalucki, J.M. (2000). Environmental Flow Requirements of the Brisbane River Downstream From Wivenhoe Dam. South East Queensland Water Corporation and Centre for Catchment and In-stream Research, Griffith University, Brisbane.

Bovee, K.D. (1982). A Guide to Stream Habitat Analysis Using the Instream Flow Incremental Methodology. Instream Flow Information Paper 12. US Department of Fisheries and Wildlife Service, Fort Collins.

Brizga, S.O. (2000). Burnett Basin Water Allocation and Management Plan: Proposed Environmental Flow Performance Measures. Department of Natural Resources, Brisbane, 2 volumes.

Brizga, S.O., Arthington, A.H., Choy, S., Duivenvoorden, L., Kennard, M., Maynard, R. and Poplawski, W. (2000a). Burnett Basin Water Allocation and Management Plan: Current Environmental Conditions and Impacts of Existing Water Resource Development. Department of Natural Resources, Brisbane, 2 volumes.

Brizga, S.O., Arthington, A.H., Choy, S., Duivenvoorden, L., Kennard, M., and Maynard, R. (2000b). Burnett Basin Burnett Basin Water Allocation and Management Plan (WAMP): Ecological Implications of Draft WAMP Scenarios. Department of Natural Resources, Brisbane.

Brizga, S.O., Davis, J.A., Hogan, A., O'Connor, R., Pearson, R., Pusey, B. and Werren, G. (2001a). Barron Water Resource Plan. Environmental Investigations Report. Department of Natural Resources and Mines, Brisbane.

Brizga, S.O., Hogan, A., Pearson, R., Pusey, B. and Werren, G. (2001b). Barron Water Resource Plan. Ecological Implications Report. Department of Natural Resources and Mines, Brisbane.

Brizga, S.O., Arthington, A.H., Choy, S., Craigie, N.M., Mackay, S., Poplawski, W., Pusey, B. and Werren, G. (2001c). Pioneer Valley Water Resource Plan: Proposed Environmental Flow Assessment Framework. Department of Natural Resources and Mines, Brisbane. 2 Volumes.

Brizga, S.O., Arthington, A.H., Choy, S., Craigie, N.M., Kennard, M.J., Mackay, S.J., Pusey, B.J. and Werren, G.L. (2002). Benchmarking, a 'top-down' methodology for assessing environmental flows in Australian River. River Research and Management (in press).

Craigie, N.M., Brizga, S.O., Clark, S.Q. and Candy, R. (2000). 'Integrated hydraulic and geomorphological investigations of the Tambo River'. Xth World Water Congress (CD).

Costelloe, J.F., Puckridge, J.T., Reid J.R.W., Pritchard J., Hudson, P., Bailey, V. and Good, M. (2002). Environmental flow requirements in arid zone rivers – a case study from the Lake Eyre Basin, central Australia.

Department of Natural Resources. (1998) Fitzroy Basin Water Allocation and Management Plan, Technical Reports. Department of Natural Resources, Brisbane.

EPA. (2000). A River Management Plan for the Wakefield Catchment. Department of Environment and Heritage, Adelaide.

Howell, G. and Doeg, T. (2002). The new FLOWS method: how it really works! Paper presented at Environmental Flows Seminar, Richmond, May 2002.

King, J.M. and Louw, D. (1998). Instream flow assessments for regulated rivers in South Africa using the Building Block Methodology. Aquatic Ecosystem Health and Restoration.

Loneragan, N.R. and Bunn, S.E. (1999) River flows and estuarine ecosystems: implications for coastal fisheries from a review and a case study of the Logan River, southeast Queensland. Australian Journal of Ecology 24(4): 431-40.

Roberts, J. (2002). Species-level knowledge of riverine and riparian plants: a constraint for determining flow requirements in the future. Australian Journal of Water Resources 5, 21-31.

SKM (2001). Determination of the Environmental Water Requirements of the Onkaparinga River Catchment. Proposed Approach for Stages 3 and 4 of the Project. Report for Onkaparinga Catchment Water Management Board.

Shirley. M. (2001). Environmental Flows for the Wimmera, Avoca and Glenelg Rivers. Paper presented at seminar "Environmental Flows – Theory, Practice and Management", Parkville, November 2001.

Stewardson, M.J. and Cottingham, P. (2002). A demonstration of the flow events method: environmental flow requirements of the Broken River. Australian Journal of Water Resources 5, 33–47.

Stewardson, M.J., Lloyd, L., McCowan, A. and Rutherfurd, I. (2002). Environmental flows: theory, practice and management. Australian Journal of Water Resources 5, 1–2.

Swales, S. and Harris, J. (1995). The expert panel assessment method (EPAM): a new tool for determining environmental flows in regulated rivers. In: Harper, D. and Ferguson, A. (eds) The Ecological Basis for River Management, London, John Wiley and Sons, 125–34.

Thoms, M.C., Sheldon, F., Roberts, J., Harris, J. and Hillman, T.J. (1996). Scientific Panel Assessment of Environmental Flows for the Barwon–Darling River. Report for Department of Land and Water Conservation, New South Wales, Parramatta.

SCIENCE BASED TRADE-OFF TOOLS

FRAN SHELDON
Centre for Catchment and Instream Research, Griffith University, Nathan QLD 4111
F.Sheldon@mailbox.gu.edu.au

Introduction

Rivers and their biota depend on flow. Most of the world's rivers suffer not only from impacts relating to changes in flow but also from the pressures of urbanisation, vegetation clearance and often intensive riparian grazing. Thus, it is often difficult to isolate those impacts that relate directly to changes in flow. However, most river ecologists recognise 'flow' as the driving force in riverine ecology (e.g. Vanotte *et al.,* 1980; Junk *et al.,* 1989; Walker *et al.,* 1995; Poff *et al.,* 1997; Richter *et al.* 1997, 1998; Ward *et al.,* 1999). Flow controls habitat structure and availability, and acts to mediate exchanges of organisms, nutrients and energy along the four dimensions (longitudinal, lateral, vertical and temporal) of river systems (Vanotte *et al.,* 1980; Junk *et al.,* 1989; Ward, 1989; Richter *et al.,* 1998).

There is, therefore, overwhelming evidence that altering a rivers natural flow regime is likely to modify the distribution and availability of habitats with adverse consequences for the native biota (Dynesius and Nilsson, 1994; Poff *et al.*, 1997; Richter *et al.,* 1998; Ward *et al.,* 1999). In Australia flow changes induced by water resource development have been associated with declines in native fish abundance and diversity (Geddes and Puckridge, 1989; Gehrke *et al.,* 1995), reduced populations of crayfish (Geddes, 1990) and the extinction of native aquatic snails (Sheldon and Walker, 1993; Sheldon and Walker, 1997). Flow regulation in Australia has also had a drastic impact on floodplain wetlands, a review of which is provided by Kingsford (2000).

There is increasing recognition that Australian rivers, particularly those in drier regions, are markedly different hydrologically and ecologically from the North American rivers for which the most long-established environmental flows approaches (such as the Instream Flow Incremental Methodology) were developed. Approaches better adapted to Australian and South African dryland rivers have been evolving over the last decade, but have not yet been subjected to the same extensive trials, refinement and peer review in mainstream journals as their North American counterparts. They have also not been evaluated specifically against current concepts of river ecosystem structure and function.

Predicting the environmental impacts of altering flow regimes in large rivers is difficult. Knowledge of the natural (or pre-development) hydrology and the associated links between hydrology, morphology and ecology are required. Whilst long-term hydrological data may be available for many river systems there are often limited corresponding physical and biological data-sets that can indicate the natural cause and effect processes associated with flow changes. In some instances the modelling of relevant processes in the presence and absence of human activities has proved successful (Maheshwari *et al.,* 1995) but this also relies on the existence of quality flow and ecosystem data.

Environmental flows must be designed for both the flow regime and whole river scales. This is particularly the case for dryland rivers which, because of their long-term variability, require extremely long-term hydrological and biological data. However, the available information is often limited to the flow history and river reach scales. This makes ongoing monitoring and adaptive management an essential prerequisite for any environmental flows design.

This paper summarises two approaches to assessing the flow requirements of dryland rivers, the "benchmarking approach" and the "ecosystem approach" and discusses the "tradeoffs" made in determining environmental flows for dryland rivers.

"Trade Off" Tools for Environmental Flow Management

Trade-Off Tools and Environmental Flows

Most environmental flows methods rely to a greater or less extent on the availability of refereed published information on the relations between flow and ecology, for each river system under study. For many specific river systems this information is non existent and even if rivers are classified by type tested models for the relations between flow and ecology are sparse (Cullen and Lake, 1995). This lack of information is the main reason why so many environmental flows methods employ "expert" opinion. Environmental flows methodologies range from simplistic explorations of the hydrological record which may document base flow and flood flow conditions to sophisticated modeling approaches where different flow scenarios can be 'tested' at different scales and positions within the river system (Arthington, 1998).

The term "Trade Off Tools" implies that there exists in the science of environmental flows an approach that allows specific 'trade-off' decisions to be made between the ecosystem and the economic and or social benefits of flow development. There are a number of significant reviews of the range of environmental flows techniques available with respect to their applicability to Australian rivers. The most comprehensive of which can be found in the edited report by Arthington and Zalucki (1998). The science of environmental flows, however, does not provide the framework for making 'trade-offs" between the environment and social and economic needs (the cost-benefit analysis). The basis for all environmental flows methodologies is that the environment (the river or floodplain) needs water in a manner that 'mimics' its natural water regime. The methodology provides the means for determining how much water is needed, the timing and duration of that water and the expected environmental outcome of the environmental flow. Environmental flows methodologies, therefore, also provide information as to the impact on the environment if it does not receive the water. The "trade-off" decision must therefore be a social and economic one, one driven by anthropomorphic visions, to determine if the 'cost' of providing water for the environment results in a social and/or economic costs or a social and/or economic benefit.

Resilience of Rivers to Hydrological Change

What we do know is that river ecosystems are likely to respond to an impact, such as anthropogenic hydrological change, in a number of ways (Figure 1). The shape of the ecological response in relation to degree of hydrological change will vary depending on the flow parameter in question and the response being examined. The shape of the response curve is a reflection of the resistance or resilience of the system to the change

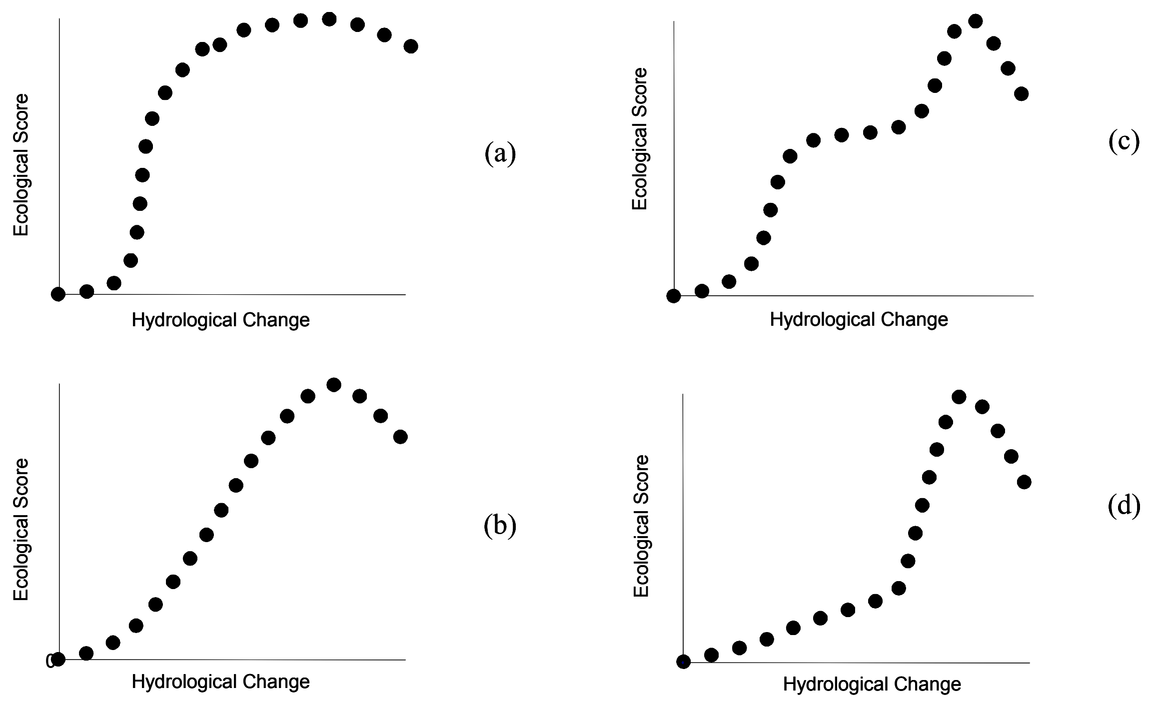


Figure 1. Shapes of ecological response curves to hydrological change (a) initially very little ecological change, followed by a rapid decline at a critical point of flow change; (b) a general sigmoid shape suggesting general ecological decline with increasing flow change (c) a period of initial rapid ecological change followed by a period of relative resilience, and thus little change, followed again by a period of rapid ecological decline at extreme levels of flow change; (d) initial rapid decline leading to major ecological change. From Sheldon et al. (2000)

Resistance is the term used to describe the ability of an ecosystem 'resist' or absorb change when exposed to a disturbance. Resilience is the term used to describe the ability of an ecosystem to return to its original state after disturbance (Fox and Fox, 1986). In measuring 'resilience' we assume there is a 'path' (such as succession) for the return of the system to some pre-disturbance state. It is also assumed that there is stability in the ecosystem – a pre-disturbance state which can be returned to, as compared with a completely new state. Even if we accept that natural systems are stable through time with natural fluctuations around this stable level in response to disturbances, we must also accept that in all systems there is a threshold to this stability and once passed the system is stressed beyond its amplitude for resilience.

Tools used in 'trade-off' decisions must surely be focused on preventing the stress on an ecosystem that would push it beyond its amplitude for resilience.

Measuring Resistance and Resilience

The measurement of ecosystem resistance to disturbance, and resilience following disturbance, may provide an early warning sign of the risk of degeneration of the ecosystem owing to external stresses or the transition of the ecosystem into an undesirable state (Whitford, et al. 1999). Most ecosystems, particularly those in drier regions are subject to episodic disturbances such as floods, droughts and fire which can provide information on their natural resistance and resilience (Whitford, et al., 1999). Whitford et al. (1999) suggest that measures of resistance could include parameters such as survivorship of specific species, maintenance of key ecosystem processes such as nutrient cycling and productivity and the maintenance of biodiversity. Resilience measures focus on the rate of recovery of the above parameters.

In Queensland the South East Queensland Regional Water Quality Management Strategy (SEQRWQMS or The Strategy) incorporated the use of various indicators in a survey approach to measure stream ecosystem vigor, organization and resilience to human disturbance (Smith and Storey, 2001). The chosen indicators included: physico-chemical indicators of ecosystem health, primary production, nutrient processes, microbial productivity, aspects of macroinvertebrate diversity, and aspects of fish assemblage diversity. These indicators were tested across a broad range of streams covering a large disturbance gradient and were seem to best reflect the overall health of the stream ecosystem (Smith and Storey, 2001). It is anticipated that ongoing monitoring of streams in South East Queensland will provide an indication as to the resilience and response of many streams to changes in the disturbance regime.

Approaches for Making Trade-Off Decisions

Below are summarized two approaches for assessing the flow requirements of both developed and undeveloped rivers, the "benchmarking approach" and the "ecosystem approach". As the Benchmarking Approach outlines the direction of ecological change in response to specific degrees of hydrological change it can easily be used in a "trade-off" scenario.

Ecosystem Approach

The Ecosystem Approach is a modification of the Holistic Approach (Arthington et al. 1992) with a greater emphasis on the relationship between hydrology and fluvial geomorphology of the river system. It is outlined in detail in Thoms and Sheldon (2002). The ecosystem approach emphasises the need to consider the entire river system but at different geomorphic scales. The approach has four stages, (1) determine the physical nature (habitat) of the riverine ecosystem; (2) identify the main ecological processes associated with the main physical habitats, (3) identify the key hydrological drivers and the implications of hydrological change on physical habitat and processes, and (4) derive key flow management options.

The "ecosystem approach" recommends that the flood pulse be a focus for environmental flow management in dryland river systems. It recognizes that a change in hydrological behaviour at the scale of a flood pulse will, with time, extend through to changes in flow history and the flow regime (Puckridge et al 1998; Thoms and Sheldon, 2000). The river system is classified into 'management zones' corresponding to particular geomorphic zones. Starting with downstream river zones first, and working progressively upstream (Figure 2) the first management question is *'does a particular flow reach the end of system'?* if not then the flood is reviewed for the next upstream zone. If the flood pulse does reach the end of a zone then the next question is *'does it reach a priority flow level'?* If it does then there can be no water extraction for consumptive use. Alternatively if it does not then water extraction can only take place down to the next priority flow level. These water management decisions are then made at each consecutive upstream station. This produces a water management decision tree (Figure 2) also allows for seasonal and antecedent features. Use of this decision tree allows managers to determine when, where and how much water can be extracted from a flood event without impinging upon important ecological functions along the river system.

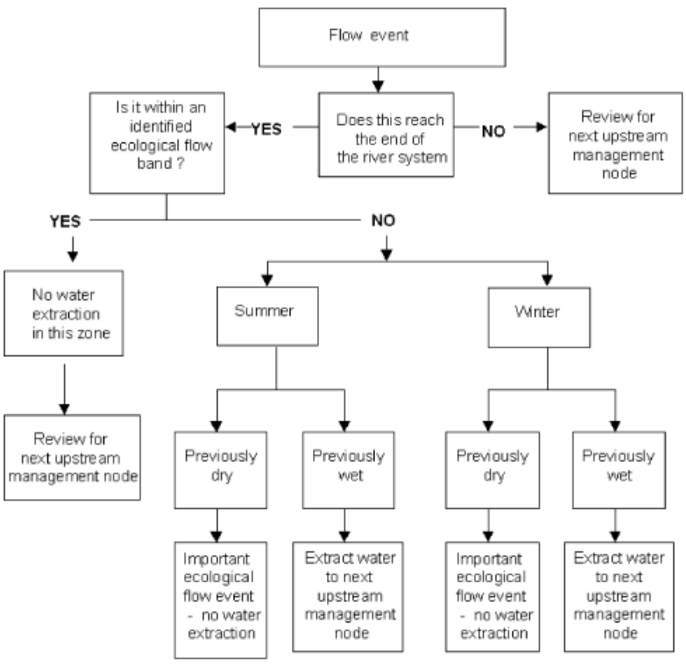


Figure 2. A management decision tree for determining environmental water allocations based on the sharing of individual flood pulses (from Thoms and Sheldon, 2002).

This approach provides managers with a number of “Trade-off Tools”. It identifies the critical flow related points within each catchment, these may be terminal wetlands, or wetlands or importance channel features along the length of the river. In upstream reaches the flow related points may reflect the height at which flows rise within the channel and this inundate various parts of the floodplain. Such a node based approach is useful for intermittent rivers where flows may not penetrate to the end point in the system but still provide significance environmental benefit by inundated wetlands or part of the channel along only part of their length.

The Benchmarking Approach

The “Benchmarking Approach” as outlined in Sheldon et al (2000) compares hydrological change with associated physical, water quality and biological changes (ecological indicators). This information is used to construct a series of hypothetical curves summarising ecological response to hydrological change ('benchmark' curves). These benchmark curves can then be used as a methodology for exploring the possible impacts on river systems of increased hydrological change. In the example provided in Sheldon et al. (2000) hydrological change was assessed using relatively simple flow statistics including Annual Proportional Flow Deviation (APFD: Gehrke *et al*. 1995), frequency of “Medium” and “High” flow events, and the duration of “Low” and “No Flow” events. However, the flow statistics chosen to assess hydrological change would need to reflect the hydrological nature of the river and/or rivers in question.

The degree of physical and biological change for each river can be assessed quantitatively by undertaking direct measurement (useful if a reference condition is known) or qualitatively, employing expert judgment informed by published papers and reports. As an information base for assessments, indicators of both ecological *structure* (community composition) and ecological *process* (both ecosystem level processes such as productivity and population level processes such as recruitment) are desirable (Bunn, 1995; Fairweather, 1999), and reliable indicators should ideally have the backing of a *process understanding* of the ecosystem in which they are to be used (Fairweather, 1999; Townsend and Riley, 1999).

The calculated flow statistic results for the rivers in question are then compared with information on the ecological condition of each river to generate a series of hypothetical relationships (‘benchmarking’) between change in a given flow statistic and ecological condition (Figure 3).

Theoretically, this approach provides a manager with a wide range of potential “trade-off tools”. The degree of ecological change can be quantified for a known amount of hydrological change. Decisions can then be made on whether this degree of ecological change is ‘acceptable’. There are problems, however, when considering ecosystem resistance and resilience. It often takes time (lag-time) before ecosystem changes in response to hydrological changes become measurable. Recent hydrological changes can take decades to be transferred into detectable environmental impact. Thus the construction of the above curves using quantitative data needs to be rigorous, such that the measured ecological response is indeed matched to the correct degree of hydrological change ensuring that the correct ecological change is not grossly underestimated.

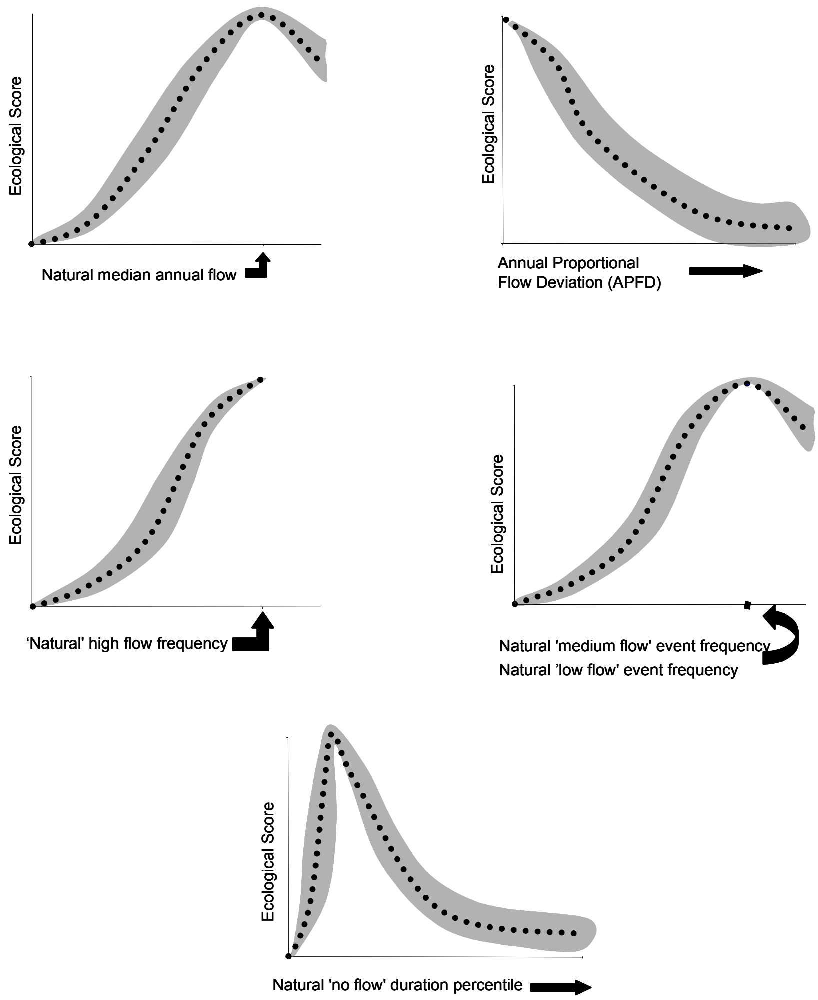


Figure 3. Scoring approach for different kinds of flow statistics and the predicted resulting ecological change (modified from Sheldon et al. 2000). Shaded region represents the range of expected ecological response.

Conclusions

In exploring scientific based "trade-off tools" it must be made clear that there is no one defensible approach that can work 'recipe' style for all river systems. From a purely environmental perspective no trade-offs decisions can be made, all the water in a river system is required for environmental maintenance whether that be within the river channel, sub-surface sediments, surrounding floodplain or further into the estuary and ocean. Any water removed from the system will have a corresponding impact. What needs to be measured is the degree of that impact and the social and economic benefits/costs of the impact.

The science of environmental flows provides a number of approaches and underlying theoretical constraints for making management based 'trade-off' decisions. In all trade-off decisions it is vital to understand the resistance and resilience of the system in question, how it will respond to hydrological change and its ability to 'absorb' hydrological change with very little impact. It is important to understand the concept of lag times and that a system that appears resilient or resistant to change may just have a very long lag time response and given time may be equally as ecologically devastated as a less resilient system to the same degree of change.

This paper has outlined two approaches for determining or understanding the environmental flow needs of river systems. These are merely examples and there are many more, a complete review of environmental flows methodologies is provided by Arthington and Zalucki (1998). Creating ecological response curves for rivers will provide an understanding of the direction of ecological change for given hydrological change which is a first step in being able to make management based trade-off decisions.

References

Arthington, A.H. (1998) Brisbane River Trial of a Flow Restoration Methodology (FLOWRESM). In *Water for the Environment: Recent Approaches to Assessing and Providing Environmental Flows*. (Eds A.H. Arthington and J.M. Zalucki) Proceedings of AWWA Forum. Pp. 35-50. (AWWA: Brisbane).

Arthington A.H., J.M. King, J.H. O'Keefe, S.E. Bunn, J.A. Day, B.J. Pusey, D.R. Bluhdorn and R. Tharme (1992) Develpoment of an holistic approach for assessing environmental flow requirements of riverine ecosystems. In *Proceedings of an international seminar and workshop on water allocation for the environment* (Eds J.J. Pigram and B.P. Hooper) The Centre for Water Policy Research, University of New England, Armidale, pp. 282.

Arthington, A.H. and J.M. Zalucki (Editors) (1998) *Comparative Evaluation of Environmental Flow Assessment Techniques: Review of Methods*. LWRRDC Occassional Paper 27/98.

Bunn, S. E. (1995) Biological monitoring of water quality in Australia: Workshop summary and future directions. *Ausralian Journal of Ecology* **20**, 220-227.

Cullen, P. and P.S. Lake (1995) Water resources and biodiversity: past, present and future problems and solutions. In *Conserving biodiversity: threats and solutions* (Eds R.A. Bradstock et al. Editors),. Beatty and Sons, Surrey, pp. 115-125.

Dynesius, M. and C. Nilsson (1994). Fragmentation and flow regulation of river systems in the northern third of the world." *Science* **266**: 753-762.

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

Fox, B. J. and M. D. Fox (1986). Resilience of animal and plant communities to human disturbance. In *Resilience in Mediteranean-type Ecosystems*. (Eds: B. Dell, A. J. M. Hopkins and B. B. Lamont).

Geddes, M. C. and J. T. Puckridge (1989). Survival and growth of larval and juvenile native fish: the importance of the floodplain. In *Proceedings of the Workshop on Native Fish Management*. Canberra, Murray-Darling Basin Commission**:** 101-115.

Gerhke, P. C., P. Brown, C.B. Schiller, D.B. Moffatt and A.M. Bruce (1995). River regulation and fish communities in the Murray-Darling river system, Australia. *Regulated Rivers: Research and Management* **11**: 363-375.

Junk, W. J., P. B. Bayley and R.E. Sparks (1989). The flood pulse concept in river-floodplain systems. *Special Publications of Fisheries and Aquatic Sciences* **106:** 110-127.

Maheshwari, B. L., K. F. Walker and T.A. McMahon (1995) Effects of regulation on the flow regime of the River Murray, Australia. *Regulated Rivers: Research and Management* **10**: 15-38.

Poff, N. L., J. D. Allan, M.B. Bain, J.R. Karr, K.L. Prestegaard, B.D. Richter, R.E. Sparks, and J.C. Stromberg (1997). The natural flow regime: A paradigm for river conservation and restoration. *Bioscience* **47**(11): 769-784.

Puckridge, J., F. Sheldon, K.F. Walker and A.J. Boulton (1998) Flow variability and the ecology of large rivers. *Marine and Freshwater Research* **49**: 55-72.

Richter, B. D., J. V. Baumgartner, R. Wigington and D.P. Braun (1997) How much water does a river need?" *Freshwater Biology* **37**: 231-249.

Richter, B. D., J. V. Baumgartner, D.P. Braun and J. Powell. (1998) A spatial assessment of hydrologic alteration within a river network. *Regulated Rivers: Research and Management* **14**: 329-340.

Sheldon, F. and K. F. Walker (1993) Pipelines as a refuge for freshwater snails. *Regulated Rivers: Research and Management* **8**: 295-299.

Sheldon, F. and K. Walker (1997) Changes in biofilms induced by flow regulation could explain extinctions of aquatic snails in the lower River Murray, Australia. *Hydrobiologia* **347**: 97-108.

Sheldon, F., M. C. Thoms, O. Berry and J.T. Puckridge (2000). Using disaster to prevent catastrophe: referencing the impacts of flow changes in large dryland rivers. *Regulated Rivers: Research and Management* **16**: 403-420.

Smith, M.J. and A.W. Storey (Editors) (2001) Design and Implementation of Baseline Monitoring (DIBM3): Developing and Ecosystem Health Monitoring Program for Rivers and Streams in Southeast Queensland. 416pp.

Thoms, M. C. and F. Sheldon (2000). Water resource development and hydrological change in a large dryland river: The Barwon-Darling River, Australia. *Journal of Hydrology* **228**: 10-21.

Thoms, M.C. and Sheldon, F. (2002) An ecosystem approach for determining environmental water allocations in Australian dryland river systems: the role of geomorphology. *Geomorphology* **47** (2-4): 153-168.

Townsend, C.R. and R.H. Riley (1999). Assessment of river health: accounting for perturbation pathways in physical and ecological space. *Freshwater Biology* **41**, 393-405.

Vannote, R. L., G. W. Minshall, K.W. Cummins, J.R. Sedell and C.E. Cushing (1980). The river continuum concept. *Canadian Journal of Fisheries and Aquatic Sciences* **37**: 130-137.

Walker, K. F., F. Sheldon and J.T. Puckridge (1995). A perspective on dryland river ecosystems. *Regulated Rivers: Research and Management* **11**: 85-104.

Ward, J. V. (1989). The four dimensional nature of lotic ecosystems. *Journal of the North American Benthological Society* **8**: 2-8.

Ward, J. V., K. Tockner and F. Schiemer (1999). Biodiversity of floodplain river ecosystems: ecotones and connectivity. *Regulated Rivers: Research and Management* **15**: 125-139.

Whitford, W. G., D. J. Rapport and A.G. deSoyza (1999). Using resistance and resilience measurements for 'fitness' tests in ecosystem health. *Journal of Environmental Management* **57**: 21-29.

Groundwater-Dependent Ecosystems in South Australia

Sébastien Lamontagne
CSIRO Land & Water, PMB 2, Glen Osmond SA 5064

SUMMARY: South Australia has a diversity of groundwater-dependent ecosystems (GDEs), including wetlands, lakes, mound springs, streams, phreatophyte communities, aquifer and cave ecosystems, and possibly some estuarine and near-shore marine habitats. The needs of groundwater-dependent ecosystems are recognized in the most recent water allocation plans for prescribed water resources areas. However, the inventory for some types of GDE is incomplete (especially for aquifer, cave, and marine near-shore ecosystems). For most GDEs, groundwater-dependency could only be established using the precautionary principle (i.e., the ecosystem was assumed groundwater-dependent because some use of groundwater was suspected). This approach is suitable for ecosystems where the relationship to groundwater is obvious (for example, aquifer and caves) but problematic for other ecosystems where the relationship is more difficult to establish (for example, phreatophyte communities). Where needed, more advanced techniques could be used to determine groundwater-dependency in the future. The groundwater regime required to maintain GDEs in a healthy state is usually not known. While modern groundwater management practices implicitly recognize that ecosystems dependent on groundwater can be damaged or lost by modifications to the groundwater regime, the lack of understanding of the nature of the relationship between groundwater and most GDEs in SA hamper efforts to properly preserve them.

- The *Water Resources Act 1997* recognizes that water provisions must be made for groundwater-dependent ecosystems.
- Not all groundwater-dependent ecosystems have been identified in SA.
- The Environmental Water Requirements for GDEs are usually not well known.

1. INTRODUCTION

In the past, the management of groundwater resources was made using the now defunct concept of the "safe yield", which was defined as "the attainment and maintenance of a long-term balance between the amount of ground water withdrawn annually and the annual amount of recharge" (Sophocleous 1997). However, this approach to the management of groundwater resources failed to recognize that many ecosystems rely on groundwater to properly function. Many of these ecosystems are unique, are among the most productive and biologically diverse in the Australian landscape, and may cease to exist when their groundwater regime is modified (Hatton and Evans 1998). This dependency of ecosystems on groundwater is now included in the definition of the "sustainable yield" of groundwater resources in South Australia and elsewhere (Australian Water Resources Council 1992; ARMCANZ/ANZECC 1996; COAG 1996; Government of South Australia 2000).

How important are GDEs for South Australia? While a complete inventory has not been made, South Australia has a wide diversity of groundwater-dependent ecosystems, including some of national and international significance. In addition to aesthetic and cultural values, GDEs play many roles in the landscape, including preservation of biodiversity, flood control, and the maintenance of water quality for consumptive use. Hatton and Evans (1998) and Clifton and Evans (2001) have summarized the information available on GDEs nationally. Additional information on South Australian GDEs is also summarized in the recently developed water allocation plans for the State's prescribed water resource areas (see list in Government of South Australia 2000 and Cook and Lamontagne 2002). Despite ongoing effort to preserve them, many South Australian GDEs have been lost or have declined since European settlement (Government of South Australia 2000).

In the following, a summary will be made of the different types of groundwater-dependent ecosystems, with examples from South Australia. A brief summary will be made of methods used to assess groundwater-dependency. Finally, the nature of the dependency of these ecosystems relative to groundwater will be discussed.

1.1 Definition of groundwater and groundwater-dependent ecosystems.

Groundwater here includes water that is found in saturated pores, fractures and karstic features forming extensive and persistent aquifers systems as opposed to soil water, seasonal perched water tables, and transient subsurface stormflow on hillslopes. This definition may not be ideal under all circumstances but should be applicable under most conditions. While relatively small in extent, freshwater lenses (such as the ones in the Musgrave and Southern Basin Prescribed Wells Area on the Eyre Peninsula) are permanent features and fit within this definition of "groundwater".

The definition of groundwater-dependent ecosystems, and the extent of their dependency towards groundwater, is more difficult because a wide variety of relationships exist between ecosystems, individual species, and groundwater. As a precautionary rule, ecosystems that derive a part of their water budget from groundwater must be assumed to have some degree of groundwater dependency (Hatton and Evans 1998). In other words, if part of this groundwater was made unavailable (for example, through pumping or lowered water quality), the health of these ecosystems would diminish or they would disappear altogether. While it is usually considered that a decline in the availability or quality of groundwater may adversely impact the health of ecosystems, too much groundwater can also be detrimental (Hatton and Evans 1998).

1.2 Types of groundwater-dependent ecosystems

Following an update of the initial classification proposed by Hatton and Evans (1998), Clifton and Evans (2001) defined six types of groundwater-dependent ecosystems based on distinctive fauna or flora. These were:

- Wetlands
- Terrestrial vegetation
- Aquifer and cave ecosystems
- Baseflow ecosystems
- Terrestrial fauna
- Estuarine and near-shore marine habitats

Groundwater-dependent ecosystems have also been classified in function of the type of aquifer system they are usually associated with. For example, in NSW five broad types of aquifer and associated ecosystems are recognized (deep alluvial, shallow alluvial, fractured rock, coastal sand and sedimentary rock aquifers; NSW Department of Land and Water Conservation 2002). In the lower South-East, ecosystems associated with karst have also been recognized (URS 2000).

In general, SA's water allocation plans classify GDEs following the categories proposed by Hatton and Evans (1998) and Clifton and Evans (2001). These will be further described below.

Wetlands

A wetland is defined in the Water Resources Act 1997 as a swamp or marsh and includes any land that is seasonally inundated. More recently, the State Water Plan 2000 extended the definition of wetlands to include the broader concept used in the Ramsar convention (Government of South Australia 2000). Wetlands encompass the greatest variety of groundwater-dependent ecosystems in SA. While not all wetlands are groundwater-dependent, most will rely to some extent on groundwater. The wetlands of SA are too numerous to list here but include:

- Permanent lakes and ponds (Blue Lake and Piccaninnie Ponds in the South-East)
- *Eucalyptus camaldulensis* and *E. largiflorens* woodlands along the River Murray
- Swamp forests and woodlands (various species) in the Mount Lofty Ranges
- Peat swamps and freshwater swamps on the Fleurieu Peninsula
- Saline swamps, and coastal heath ecosystems on the southern and western Eyre Peninsula.
- *Melaleuca* swamps in internal drainages on the Yorke Peninsula and in the upper South-East
- Permanent swamps and lakes in solution hollows on Kangaroo Island
- Emergent herblands (fens) on Eight Mile Creek (lower South-East)

Probably the most unusual groundwater-dependent wetlands in SA are the mound springs of the Great Artesian Basin (GAB). Mound springs occur throughout the Great Artesian Basin but active ones are most common in the far North of South Australia (Harris 1992). Mound springs occur when artesian water from the GAB discharges to the surface through fault lines or other structural weaknesses. The hydrogeology, fauna, and flora of mound springs are relatively well understood in comparison to other South Australian GDEs (Harris 1992; Hatton and Evans 1998) but knowledge gaps still remain. Several management programs to protect mound springs are in place, including the establishment of national parks, fencing to exclude cattle grazing, and bore capping to maintain pressure in the GAB (Harris 1992).

Wetlands are probably the GDEs most impacted by land-use change, with up to 66% of wetland area lost since European settlement in SA (Government of South Australia 2000). Because of their shallow depth, many wetlands are sensitive to changes in the water table because small variations can result in large differences in the surface area inundated (Hatton and Evans 1998). Threats to wetlands in SA include decline in water tables, drainage for agriculture, decreased flooding frequency from river regulation, invasion by exotic species, and salinisation.

Terrestrial vegetation

Terrestrial plant communities dependent on groundwater (or phreatophytes) are often similar to wetland plant communities, with the exception that they occur in areas where the water table does not reach the surface. Some phreatophytes appear to rely heavily on groundwater and mostly occur in areas with shallow water tables (i.e., *Melaleuca* spp.). However, other phreatophytes, such as the *Banksia* community on the Gnangara mound in Western Australia, can use deeper water tables (Water Authority of Western Australia 1992). Many phreatophytes use groundwater depending on the availability of other sources. For example, *E. camaldulensis* in the Chowilla floodplain (SA) were found to use rain-derived soilwater and groundwater in winter and groundwater only during summer (Mensforth et al. 1994). Example of terrestrial vegetation dependent on groundwater in SA include:

- *Eucalyptus camaldulensis* and *E. largiflorens* woodlands on the River Murray floodplain.
- *Eucalyptus camaldulensis* woodlands on southern and western Eyre Peninsula
- *Melaleuca halmaturorum* shrublands and *Eucalyptus* spp. woodlands in the South-East

Threats to terrestrial vegetation dependent on groundwater include declining water tables, groundwater salinisation, and drowning caused by rising water tables. Evidence from tree plantations suggests that many terrestrial plant communities may rely more heavily on groundwater than previously recognized (Hatton and Evans 1998).

Baseflow systems

In streams and rivers, baseflow is the maintenance of flowing water or permanent pools over prolonged periods with no rainfall. Several sources of water can contribute to baseflow (including delayed drainage, bank discharge and unsaturated flow) but groundwater discharge is usually a significant component. Droughts or low flows are a common feature of Australian streams and many organisms are well adapted to cope with the absence of surface water for prolonged periods (see also *aquifer and cave ecosystems* below). However, other species (such as many fishes, amphibians, and benthic invertebrates) cannot persist unless significant surface water habitat remains during drought periods. Examples of baseflow systems in SA include:

- Eight-Mile Creek and the Glenelg River in the South-East
- Streams in the Mount Lofty Ranges and Kangaroo Island with permanent flow or permanent groundwater-fed pools

Threats to baseflow systems in SA include water table decline, salinisation, pumping of surface water from permanent pools, and decreased recharge of alluvial aquifers from the interception of stormwater by farm dams and reservoirs.

Cave and aquifer ecosystems

A continuum of life exists at the interface between surface water and groundwater. At one extreme, some surface water organisms will occasionally seek refuge in groundwater during drought periods, while others are obligate cave or aquifer dwellers. Groundwater-dwelling animals (or *stygofauna*) can be classified in function of their size. The *macrofauna* is restricted to aquifers with large pore sizes (such as karst, fractured rock, and coarse alluvium) and is composed of invertebrates (especially crustaceans) and occasionally vertebrates such as fishes (Humphreys 2002). The *meiofauna* consists of smaller invertebrates and protists (protozoans, etc) and can also be found in aquifers with smaller pore sizes. Finally, the *microfauna* (bacteria, fungi, and small protists) may be ubiquitous to all aquifers. The stygofauna has also been categorized according to its use of sub-surface habitats (Table 1). In this respect, the interface between streams and aquifers (or *hyporheic* zone) is especially rich in stygofauna (Boulton 2000; Humphreys 2002).

Table 1: Categorization of the stygofauna based on the use of sub-surface habitats during their life cycle (adapted from Gibert et al. 1994 and Boulton 2000).

Category	Sub-Category	Characteristics
Stygoxenes		Species with no affinities for groundwater but accidentally occurring in caves or stream sediments.
Stygophiles		Actively exploit subsurface resources, especially seeking protection from unfavorable surface conditions. Mostly associated with the hyporheic zone of streams and shallow alluvial aquifers.
	Occasional hyporheos	Can complete their life cycle without entering the subsurface but occasionally found there.
	Amphibites	Require both the utilization of surface and subsurface habitats to complete their life cycle.
	Permanent hyporheos	Spend their complete life cycle in the subsurface but could survive in surface environments.
Stygobites		Specialized subterranean organisms that cannot survive in surface environments.

Because of long periods of isolation and the presence of barriers to migration, Australia has a very diverse cave and aquifer fauna (Humphreys 2002). For example, at least two stygobytes (*Koonunga crenarum* and *Uronyctus longicaudus*) are known to be endemic to the karst ecosystems of eastern SA and western Victoria (Zeidler 1985; URS 2000). Rare freshwater stromatolites are also found in lakes and other karst features of the lower South-East (Thurgate 1996; URS 2000).

The functional significance of groundwater ecosystems is not completely known at present. However, it is now increasingly recognized that aquifer ecosystems can be used as biomonitoring tools (Humphreys 2002), contribute to nutrient recycling and biological productivity in streams (Boulton et al. 1998), and are actively involved in maintaining water quality in aquifers, for examples through the elimination of introduced pathogens (Toze and Hanna 2002) and contaminants (Anderson and Lovley 1997). Examples of cave and aquifer ecosystems in South Australia include:

- The hyporheic communities in terminal rivers from the Flinders Ranges (Cooling and Boulton 1993)
- Karst ecosystems in the South-East and the Nullarbor Plains
- Stygofauna in fractured bedrock aquifers (Mount Lofty Ranges)
- Meiofauna and microorganisms in alluvial aquifers (Murray-Darling Basin)

Threats to cave and aquifer ecosystems include water table decline and groundwater pollution (salinity, anoxia, etc).

Estuarine and marine near-shore ecosystems

There is increasing evidence that some estuarine and near-shore marine ecosystems rely to some extent on submarine groundwater discharge. These include some coastal swamps, mangroves, lagoons, and marshes (Johannes and Hearn 1985; Hatton and Evans 1998) and offshore fresh groundwater springs (Stieglitz and Ridd 2000). The input from groundwater of the nutrients responsible for estuarine and coastal eutrophication is an important topic of research in Australia (Linderfelt and Turner 2001) and overseas (Burnett et al. 2002). Submarine groundwater discharges appear to have received little attention to date in SA.

Terrestrial fauna

Although not an ecosystem in itself, migratory fauna and some terrestrial animals can be highly dependent upon the availability of groundwater seasonally or during droughts (Hatton and Evans 1998). These would include some terrestrial fauna relying on waterholes during droughts (emus, kangaroos, koalas) and migratory birds using wetlands as feeding grounds, nesting habitats and refuges.

1.3 Methods to assess groundwater dependency

The dependency of ecosystems towards groundwater is most commonly assessed using the precautionary principle. In other words, an ecosystem is assumed to have some level of groundwater-dependency if a change in the availability or quality of groundwater would result in a decline in ecosystem health. In some cases, groundwater dependency is obvious and does not require an elaborate analysis (for example, aquifer ecosystems). In other cases, the links can be more difficult to establish (for example, phreatophytes). A desktop methodology to identify groundwater-dependent ecosystems has been developed (PPK 1999). The precautionary principle has been the principal tool used to evaluate groundwater-dependency in SA.

A range of other techniques can also be used to assess groundwater-dependency when required. In general, these involve either directly measuring that groundwater is used or deducing it through some other means. For phreatophytes, clues that groundwater is used include:

- Greater leaf area indexes relative to surrounding areas
- Tree vs. shrub growth forms
- Diurnal fluctuations in the water table

More direct assessments include:

- Pre-dawn leaf water potential
- Water balance analysis
- Identification of the sources of water used for transpiration using the stable isotopes of the water molecule.

Stable isotopes have been used quite extensively to identify groundwater use by phreatophytes in the River Murray floodplain and in *Melaleuca* stands in the South-East (see review in Walker et al. 2001).

1.4 Level of ecosystem dependency

Hatton and Evans (1998) have proposed a classification system for the level of dependency of GDEs. This classification is based on the proportion of groundwater in the ecosystem water budget and the expected sensitivity of the ecosystem to a change in the groundwater regime (Table 2). At one extreme, ecosystems that rely on groundwater at all times and that would be impacted by even small changes in the groundwater regime are classified as entirely dependent (for example, mound springs, saline discharge lakes and aquifer ecosystems). At the other extreme are ecosystems that do not rely on groundwater significantly and that will not be impacted by a change in the groundwater regime (for example some rainforests, episodic floodplain lakes, rockholes, and many desert ecosystems).

1.5 The groundwater regime

The dependency of ecosystems on groundwater is usually more complicated than simply a fixed amount of groundwater used during a given year. Clifton and Evans (2001) propose that the pattern of water usage of four key groundwater attributes (or *groundwater regime*) will determine the requirements of GDEs. These attributes are:

- *Flow* or *flux*: the rate and volume of supply of groundwater.
- *Level*: for unconfined aquifer, the depth below the surface of the water table.
- *Pressure*: for confined aquifers, the potentiometric head of the aquifer and its expression in groundwater discharge areas.
- *Quality*: The physico-chemical properties of groundwater, including oxygen, temperature, pH, salinity, nutrients, contaminants, etc.

The pattern in water usage of the groundwater attributes also includes:

- *Thresholds*: Boundaries within which a given attribute must be maintained to prevent ecosystem collapse.
- *Rates of use*: How much water is used and at which rates.
- *Temporal distribution in use*: The temporal dimension of usage, including timing, frequency, duration, and episodicity.

The *natural groundwater regime* will be the one under unimpacted conditions (usually, pre-European settlement). Under managed conditions, the *environmental water requirement* (EWR) will represent the different possible groundwater regimes that will ensure that the key ecological values of a GDE will remain at a low level of risk when the groundwater resource is used or modified.

Which attribute(s) will be significant will vary from ecosystem to ecosystem and how ecosystems will respond to a change in a particular attribute may be different. In some cases, the response of the ecosystem could be proportional to the change in the attribute. For example, the health of a riparian community may progressively decline as groundwater salinity increases. For other ecosystems, some threshold or boundary in an attribute will need to be maintained otherwise the ecosystem will collapse. For example, a key attribute for mound springs is the pressure or potentiometric head in the GAB. If aquifer pressure decreases below some threshold value, the spring will cease to discharge and the associated ecosystem will perish.

The natural groundwater regime of most groundwater-dependent ecosystems in SA is not known. In addition, many aquifers are presently readjusting to new landscape use and rates of extraction, which further complicates the assessment of the EWR of GDEs (Cook and Lamontagne 2002).

2. CONCLUSION

South Australia has a diversity of groundwater-dependent ecosystems and many, such as the mound springs and other wetlands, have unique biological and cultural values. However, there is no comprehensive survey of GDEs in South Australia, especially for more poorly understood ecosystems such as caves and aquifers. Better management of GDEs will involve a better assessment of their EWR. To date, these have been mostly established by assuming that current conditions are the EWR (Cook and Lamontagne 2002). While reasonable as an initial step, this approach cannot always guarantee the long-term health of the ecosystem (Cook and Lamontagne 2002). Determining the EWR more quantitatively can be challenging but is not impossible. A range of techniques can be used to define the EWR of groundwater-dependent

Table 2: Levels of ecosystem dependency on groundwater (modified from Hatton and Evans 1998)

Type of groundwater dependency	Description
Entirely dependent	Ecosystems undoubtedly dependent on groundwater and where slight changes in the groundwater regime, or a change below some threshold, would result in the system ceasing to exist.
Highly dependent	Ecosystems where moderate changes in the groundwater regime would substantially decrease the health of the ecosystem. Entire ecosystem collapse is possible.
Proportional dependence	Where the health of the ecosystem will be proportional to the change in the water regime. Ecosystem will disappear if the groundwater resource is eliminated.
Opportunistic or limited use	Groundwater may only play a significant role in the water balance in times of droughts. Groundwater still important in the long-term, but the immediate impact of a substantial reduction in groundwater may be muted.
No apparent dependency	No changes expected in the health of the ecosystem from a change in the groundwater regime.

ecosystems.

It is unlikely that resources will be available to define the EWR for all the GDEs of South Australia using all the techniques and tools that are currently available. Management schemes have been proposed to classify GDEs by relative level of importance based on international and national significance, biodiversity, cultural value, and other criteria (Clifton and Evans 2001). Such a classification should be undertaken in SA to help prioritize resource allocation for future EWR assessments.

3. ACKNOWLEDGEMENTS

Comments and suggestions were provided by T. Hatton, I. Jolly, K. Holland, G. Walker, P. Cook, R. Evans, D. Favier, M. Good, S. Gatti, and S. Richardson.

4. REFERENCES

Anderson, R.T. and Lovley, D.R. (1997). "Ecology and biogeochemistry of *in situ* groundwater bioremediation". Advances in Microbial Ecology **15**: 289-350.

ARMCANZ/ANZECC (1996). "National principles for the provision of water for ecosystems." ARMCANZ and ANZECC Occasional Paper No. 3.

Australian Water Resources Council. (1992). "Draft guidelines for groundwater protection. National Water Quality Management Strategy". ISBN 0 642 18302 3.

Boulton, A.J. (2000). "The subsurface macrofauna". *In* Jones, J.B. and Mulholland, P.J. [Eds.] Steams and ground water. pp: 337-361. Academic P., San Diego, CA

Boulton, A.J., Findlay, S., Marmonier, P., Stanley, E.H., and Valett, H.M. (1998). "The functional significance of the hyporheic zone in streams and rivers." Annual Review of Ecology and Systematics **29**: 59-81.

Burnett, B. and 11 others (2002). "Assessing methodologies for measuring groundwater discharge to the oceans." Eos **83**: 117-123.

Clifton, C. and Evans, R. (2001). "Environmental water requirements to maintain groundwater-dependent ecosystems". Environmental Flows Initiative Technical Report No. 2. Commonwealth. of Australia, Canberra).

Cook, P.G. and Lamontagne, S. (2002). "Assessing and protecting water requirements for groundwater dependent ecosystems". Proc. of the Science of Environmental Water Requirements in South Australia seminar. 24 Sep 2002, Adelaide, SA.

Cooling, M.P. and Boulton, A.J. (1993). "Aspects of the hyporheic terminus of a South Australian arid-zone stream." Australian Journal of Marine and Freshwater Research **44**: 411-26.

Council of Australian Governments [COAG]. (1996). "Allocation and Use of groundwater: A national framework for improved groundwater management in Australia." Occasional Paper Number 2. 9 p.

Gibert, J., Stanford, J.A., Dole-Olivier, M.-J., Ward, J.V. (1994). "Basic attributes of groundwater ecosystems and prospects for research". *In* Gibert, J., Danielopol, D.L., and Stanford, J.A. [Eds.] Groundwater Ecology. pp: 7-40. Academic Press, San Diego, CA.

Government of South Australia. (2000). "State Water Plan 2000". Adelaide, SA.

Harris, C.R. (1992). "Mound springs: South Australian initiatives". Rangeland Journal **14**: 157-173.

Hatton, T. and Evans, R. (1998). "Dependence of Ecosystems on groundwater and its significance to Australia". LWRRDC Occasional Paper No. 12/98.

Humphries, W.F. (2002) "Groundwater ecosystems in Australia: An emerging understanding." Proceedings of the IAH groundwater conference: "Balancing the groundwater budget", 12-17 May 2002, Darwin, NT.

Johannes, R.E. and Hearn, C.J. (1985). "The effect of submarine groundwater discharge on nutrient and salinity regimes in a coastal lagoon off Perth, Western Australia." Estuarine, Coastal and Shelf Science **21**: 789-800.

Linderfelt, W.R. and Turner, J.V. (2001). "Interaction between shallow groundwater, saline surface water and nutrient discharge in a seasonal estuary: the Swan-Canning system". Hydrological Processes **15**: 2631 – 2653.

Mensforth, L.J., Thorburn, P.J., Tyerman, S.D. and Walker, G.R. (1994). "Sources of water used by riparian *Eucalyptus camaldulensis* overlying highly saline groundwater." Oecologia **100**: 21-28.

NSW Department of Land and Water Conservation. (2002). "The NSW Sate groundwater dependent ecosystem policy." NSW Government, Sydney, Australia.

PPK (1999). "Desktop methodology to identify groundwater dependent ecosystems." Nature Conservation Council of NSW, Sydney.

Sophocleous, M. (1997). "Managing water resources systems: why "safe yield" is not sustainable." Ground Water **35**: 561.

Stieglitz, T. and Ridd, P. (2000). "Submarine groundwater discharge from paleochannels? "Wonky Holes" on the inner shelf of the Great Barrier Reef, Australia." Proceedings of the HYDRO 2000 Conference, Perth, 20 – 23 November 2000.

Thurgate, M.E. (1996). "The stromatolites of the cenote lakes of the lower South East of South Australia." Helictite **34**: 17-25.

Toze, S. and Hanna, J. (2002). "The survival potential of enteric microbial pathogens in a reclaimed water ASR project." Proceedings of the ISAR-4 Symposium. 22-26 September 2002, Adelaide, SA.

URS (2000). "Assessments of the environmental requirements of groundwater dependent ecosystems in the South East prescribed wells areas". Report to the South East CWMB, Adelaide, SA.

Walker, G. and 8 others (2001). "The use of stable isotopes of water for determining sources of water for plant transpiration". CSIRO Land and Water Technical Report 05/01. Adelaide, SA.

Water Authority of Western Australia. (1992). "Gnangara Mound vegetation stress study." Report No. WG 127, Water Authority of WA, Perth, WA.

Zeidler, W. (1985). "A new species of crustacean (Syncarida: Anaspidacea: Koonungidae), from sinkholes and caves in the South-East of South Australia". Transactions of the Royal Society of South Australia, **109**: 63-75.

Assessing and protecting water requirements for groundwater dependent ecosystems

Peter Cook and Sebastien Lamontagne
CSIRO Land and Water, PMB 2, Glen Osmond, South Australia

SUMMARY:

Under equilibrium conditions, aquifer recharge will be exactly balanced by aquifer discharge. Thus any groundwater extraction will ultimately result in a decrease in natural discharge, which will impact on the environment. However, in most groundwater systems, there will be a timelag between groundwater extraction and ecosystem impact, which may range from a few years to many centuries. While methods for assessing groundwater dependency of ecosystems are available, water requirements are more difficult to determine. Perhaps because of this difficulty, most Water Allocation Plans in South Australia do not included adequate assessments of the water needs of groundwater dependent ecosystems, or adequate measures for their protection. Rather, they aim to maintain current ecosystem status, and appear to assume that the current extraction regime will achieve this. Because of the timelags involved in these processes, this is not necessarily the case. There is a need for a closer link between water allocation mechanisms and environmental water provisions. In particular, catchment-scale limits on volumetric extractions will usually not ensure that water provisions for ecosystems are met.

- All groundwater use will have an environmental impact
- The current health of ecosystems does not imply that the current extraction regime is sustainable
- New tools need to be developed for determining ecosystem water requirements
- Catchment-scale volumetric limits on groundwater extraction are not an adequate tool to ensure that ecosystem water provisions are met

1 INTRODUCTION

Sustainable use of groundwater must ensure not only that future exploitation of the resource is not threatened by current overuse, but also that natural environments that depend on the resource are protected. Hatton and Evans (1998) identified four major types of ecosystems that may be dependent on groundwater. They are: (1) terrestrial vegetation, (2) river base flow systems, (3) aquifer and cave ecosystems, and (4) wetlands. Nearshore marine ecosystems represent a possible fifth category (Figure 1). Lamontagne (2002) summarises those major groundwater dependent ecosystems (GDEs) that have been identified in South Australia. This paper firstly describes groundwater flow processes as they relate to GDEs, and the theoretical basis for assessing environmental water requirements and environmental water provisions. With this theoretical framework established. it then discusses the methods that have been used for assessing ecosystem water requirements and water provisions. The final section discusses water allocation mechanisms that have been applied in South Australia, and the ability of these allocation mechanisms to achieve the specified water provisions.

2 THEORY

2.1 The Groundwater Balance

Under natural conditions, the recharge to an aquifer will be exactly balanced by the discharge. Groundwater will flow from the recharge areas to the discharge areas, and the position of the water table will reflect the distribution of the recharge and discharge areas, as well as the aquifer geometry and transmissivity (Figure 2a). Any groundwater extraction will upset this equilibrium by (temporarily) increasing aquifer discharge, and producing a loss from aquifer storage. The position of the water table will change, as the system adjusts to the change in discharge (Figure 2b). If the rate of extraction is less than the rate of recharge, then a new equilibrium will eventually become established, in which discharge is again equal to recharge (Figure 2c). If the recharge rate remains unchanged, then the natural discharge will be reduced by an amount equal to the rate of groundwater extraction. Natural groundwater discharges that may be affected include flow to streams, lakes and springs, water use by phreatophytic vegetation, leakage to adjacent aquifers and flow to the sea.

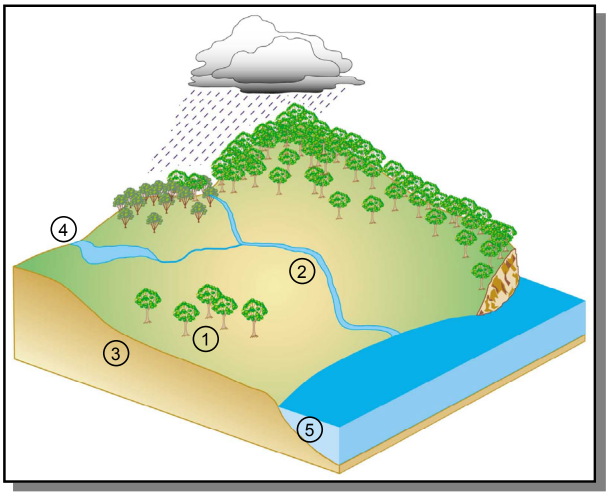


Figure 1. Potential groundwater dependent ecosystems within a catchment: (1) terrestrial vegetation, (2) stream receiving groundwater as baseflow; (3) aquifer ecosystems, (4) wetlands and (5) marine nearshore marine environment receiving submarine groundwater discharge.

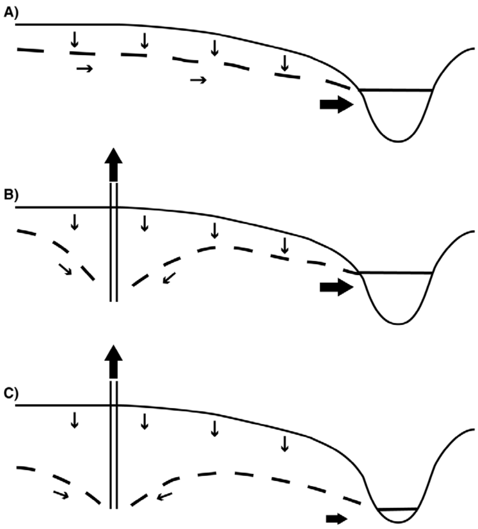


Figure 2. Changes in aquifer hydraulics in response to groundwater extraction. A) Under natural conditions, groundwater recharge is equal to groundwater discharge to the river. B) After commencement of groundwater extraction, a cone of depression develops around the pumping well. Groundwater discharge to the river is unchanged. Total groundwater discharge is greater than groundwater recharge. C) Under the new equilibrium, groundwater discharge to the river is reduced by an amount equal to the groundwater extraction rate.

The time lag between groundwater extraction and reduction in natural groundwater discharge will depend on the groundwater extraction rate relative to the natural recharge and discharge rates. It will also depend on the proximity of the groundwater extraction bores to the natural recharge and discharge zones of the aquifer. Knight et al. (2002) and Cook et al. (2002) have developed simple models to predict changes in flow to rivers following changes in aquifer recharge or discharge. For example, if an aquifer has only one source of natural discharge, then the relative rate of natural groundwater discharge following groundwater extraction from a single bore can be calculated as:

$$\Delta q(t) = 1 - \frac{Q}{RA}\, erfc\left[\frac{a}{2}\sqrt{\frac{S}{Tt}}\right] \qquad (1)$$

where T is the aquifer transmissivity, S is the specific yield, a is the distance of the groundwater extraction from the river, Q is the pumping rate (ML/yr), t is time, R is the aquifer recharge rate (mm/yr), A is the total area of the basin, and RA is the total aquifer recharge (ML/yr) . Figure 3 shows the reduction in discharge over time due to groundwater extraction, where the total extraction rate is equal to half of the aquifer recharge rate. In all cases, discharge is eventually reduced to half of its natural level (although the figure only shows the first 100 years). The reduction is most rapid, however, where the distance of extraction from the river is small. Locating bores large distances from the river will therefore delay the eventual impact on the natural ecosystem. Equation 1 and Figure 3 are for the situation where groundwater extraction occurs over a small area located a fixed distance (a) from the river. Similar equations can be derived the case where recharge is distributed over a larger area, and these can be found in Knight et al. (2002) and Cook et al. (2002).

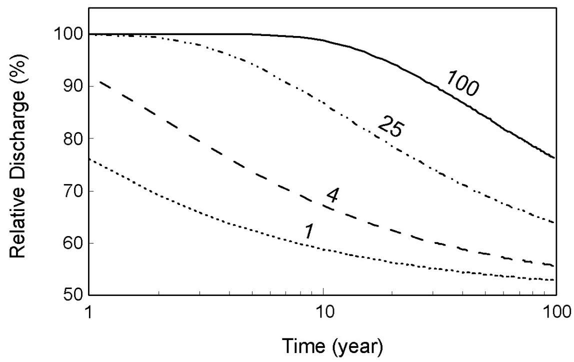


Figure 3. Reduction in discharge over time due to groundwater extraction. Curves are for Q/RA = 0.5, and denote various values of a^2S/T (units of years). In all cases, discharge is eventually reduced to half of its natural level, although this is most rapid where a and S are small, and T is large.

Figure 4 shows the effect of controlling the distribution of extraction within a region. While buffer zones can provide some protection for ecosystems, the protection is reduced if groundwater extraction is concentrated at the edge of the buffer zone. In particular, it is preferable to distribute extraction evenly throughout a catchment, than to specify buffer zones around ecosystems, but then allow extraction to be concentrated at the edge of the buffer zones.

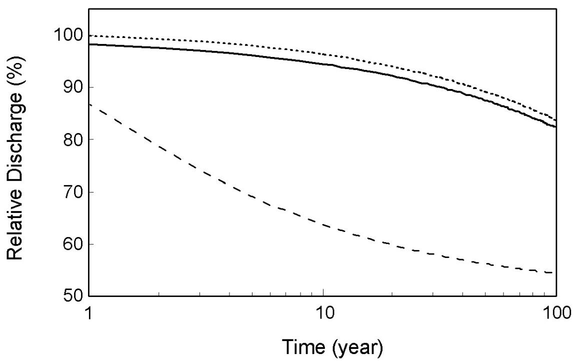


Figure 4. Effect of distributing extraction over a region. All simulations are for the situation where the total extraction rate is equal to half of the total aquifer recharge rate (Q/RA = 0.5) and for $T/S = 10^5$ m^2/yr. The solid line shows the effect of distributing groundwater extraction evenly over the entire catchment (which is 10 km in length). The broken lines both contain a 500 m buffer zone adjacent to the river. The dotted line has groundwater extraction evenly distributed in the rest of the catchment, whereas the dashed lines has all groundwater extraction located on the edge of the buffer zone.

To illustrate the time-delay concept, the above discussion has considered that an aquifer has only one natural discharge point (here represented as a stream), and that this discharge supports groundwater dependent ecosystems. Where this is the case, the long-term impact of groundwater extraction on the ecosystem will depend only on the total extraction rate relative to the total aquifer recharge rate (Q/RA). Over shorter timescales, however, the impact will depend to a much greater extent on location of groundwater extraction relative to GDEs. Where there is more than one natural discharge point (Figure 1), the long-term combined impact on the ecosystems (expressed in terms of discharge reduction) will depend only on the total extraction rate relative to the total recharge rate. However, the decrease in flow to any particular ecosystem over both short and long terms will depend on the location of extraction relative to the ecosystems. In this case, a detailed groundwater investigation needs to take place to determine which natural discharges will be impacted by the proposed groundwater extraction, and to what extent. The relative impacts of different groundwater extraction options can then be compared.

2.2 Requirements, Provisions and Allocations

It is important to note that there is no level of groundwater extraction that will not, in the long run, result in declines of natural discharges, with consequent environmental impacts. Of course, sometimes such impacts will be small, and not readily identifiable. In other cases, they may be much more dramatic, such as in the drying up of mound springs of the Great Artesian Basin. The task of groundwater managers is to determine what level of environmental impact is acceptable, and also to manage extraction to maintain the impacts to within these acceptable limits.

We can identify a five-step process for assessing and protecting water requirements for groundwater dependent ecosystems. This involves: (1) establishing water dependence; (2) establishing the natural water regime; (3) determining the water requirements; (4) balancing the ecosystem requirements with social and economic needs for water to decide on a ‘water provision’; and (5) developing water allocation policies that protect this water provision.

In some cases, such as for surface water systems receiving groundwater baseflows, establishing water dependence is an easy matter. In other cases, such as for terrestrial vegetation, it is less obvious. As discussed by Hatton and Evans (1988), simple measures of vegetation vigour, such as Leaf Area Index, can be strong indicators of water availability in semi-arid to arid environments, which can indicate possible access of vegetation to sources of water other than rainfall. A more quantitative approach might involve measurement of plant water use (transpiration) using sap flow techniques or methods such as ventilated chambers. For example, transpiration by some bluegum plantations over shallow watertables in the southeast of South Australia exceeds mean annual rainfall, indicating that the vegetation are using groundwater, and allowing quantification of the volume of use (Dillon et al., 2001). In contrast, in the Howard Basin, Northern Territory, measurements of tree water use and soil properties determined that the unsaturated zone soil storage was more than sufficient to sustain transpiration through the dry season, and so the vegetation was unlikely to be dependent on groundwater (Cook et al., 1998). Although not universally applicable, isotope techniques can potentially determine what sources of water are being accessed by vegetation at any particular time. Provided that soil water and groundwater water have distinct isotopic compositions, comparison of plant water chemistry with soil water and groundwater chemistry can determine relative proportions of water obtained from these different sources (Walker et al., 2001). Importantly, however, these methods only provide a snapshot in time. Some ecosystems might only depend on groundwater at certain times (e.g., during drought periods), and so measurements would need to be made at these critical times.

Figure 5 shows the relationship between ecosystem health and water availability for a hypothetical GDE. Ideally, the task of assessing the natural water regime and the water needs of the ecosystem should involve derivation of this relationship. For surface water systems, ecological ranges of various faunal and floral assemblages have been determined from collation of results from numerous studies. This information forms a basis for derivation of relationships such as those shown in Figure 5 for surface water systems. However, this information has not been widely collected for groundwater dependent terrestrial vegetation, for cave and aquifer ecosystems, or for groundwater dependent nearshore marine ecosystems. One of the more detailed studies relating to terrestrial vegetation is that of Stromberg et al. (1996) who derived relationships between frequency of occurrence and depth-to-groundwater for the San Pedro River area of Arizona (Figure 6). In Western Australia, Groom et al. (2000) recorded declines in Banksia woodland ecosystems following a decline in the watertable of 2.2 m between 1990 and 1991.

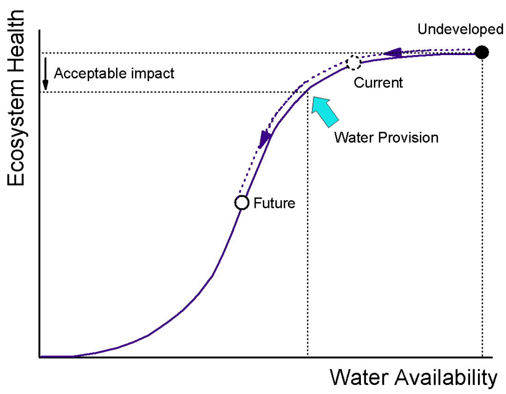


Figure 5. Ecosystem health – water availability relationship for a hypothetical ecosystem.

The scarcity of such studies makes determination of water requirements for marine and terrestrial GDEs and for aquifer and cave ecosystems extremely difficult. For riparian and wetland ecosystems that are dependent on groundwater, it is probably sufficient to develop relationships between ecosystem health and surface water flow availability. Baseflow separation methods (e.g., Ellins et al., 1990) can then be used to quantify the groundwater contribution to the surface water flow, and calculations can be performed to determine the likely change in surface water availability due to changes in groundwater availability (see Section 2.1).

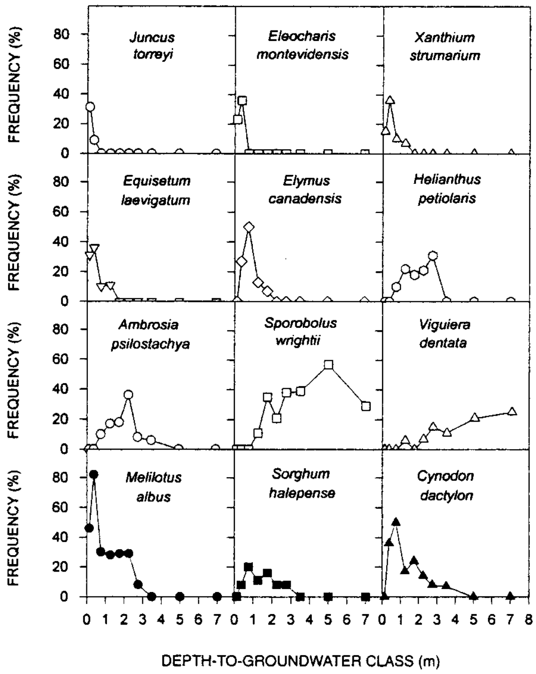


Figure 6. Distribution of herbaceous plants of the San Pedro River floodplain in relation to mean depth to groundwater. From Stromberg et al. (1990).

Once the relationship between ecosystem health and water availability has been established, determination of the water provisions is then a subjective exercise that essentially involves choosing a point along the curve shown in Figure 5 that will equitably balance ecosystem health with the social and economic demands for water. The water allocation policy then needs to be designed to ensure that this water provision is achieved. Any groundwater extraction will move the ecosystem status towards the left along this curve, from the initial 'undeveloped' position in the top right hand corner. It may be that the provision is lower on the curve than the current ecosystem status. Even so, if the catchment is not currently in balance, then the ecosystem status may ultimately move further to the left than is desired (and further than specified by the water provision). The broken line shows the result of a particular groundwater extraction scenario. In this example, groundwater extraction commenced some time in the past, and has already resulted in a small decline in ecosystem health (position marked 'current'). Continued groundwater use at this rate will ultimately produce the ecosystem status denoted 'future'. The rate at which the ecosystem status moves along this line, and its ultimate position will depend upon the volume and location of groundwater extraction.

An important part of groundwater management is the link between the water provision for GDEs, and the water allocation mechanisms that are put in place to achieve these. Technically, this is a relatively straightforward matter, although it does not seem to be a strength of current policies. Appropriate water allocations will ensure that water availability for GDEs does not become less than the defined water provisions, with unacceptable impacts on ecosystem health.

3 GDE WATER REQUIREMENTS AND WATER PROVISIONS IN SOUTH AUSTRALIA

In South Australia, the requirements and provisions of GDEs are described in Water Allocation Plans (WAPs). These provisions include specifications for baseflows of river reaches (peak flows, daily flows and flow frequencies), seasonal fluctuations of groundwater levels and groundwater salinity. However, because of the difficulty of describing ecosystem health – water availability relationships, these specifications generally are not based on assessments of ecosystem needs, but rather they are descriptions of the current water regime. In most cases, it is assumed that the current groundwater conditions reflect the needs of the ecosystem. Where water provisions are specified they are also usually the same as the current water regime. However, while construction of water availability – ecosystem health relationships is difficult, it is essential if decisions on water allocation require evaluation of trade-offs between environmental and social uses. Certainly, the inherent assumption that both the current ecosystem water status and the current level of groundwater use can be maintained ignores the timelag between development and impact. It will only be valid if the groundwater system is in balance, which is unlikely to be the case for the groundwater systems in South Australia.

Furthermore, most of the WAPs in South Australia do not specifically link the water provision for GDEs with the water allocation mechanisms that are put in place to achieve them. For example, WAPs for the South East region specify mean water table elevation targets, seasonal water table ranges and timing, and salinity targets as environmental water provisions, but it is not clear from the available documentation whether the monitoring systems are sufficient to enforce this policy and protect specific sites with GDEs. The mechanism for water allocation is to allocate 10% of the aquifer recharge to GDEs, and the remaining 90% is available for groundwater extraction. This mechanism may not in itself achieve these GDE provisions. Similarly, the WAP for the Barossa Prescribed Wells Area (PWA) specifies baseflow ranges for rivers to protect these ecosystems. Water allocation for the Barossa PWA also involves volumetric limits of water use, and the plan does not discuss how this policy will lead to preservation of the specified flows. In fact, in most cases, the water allocation mechanisms will not ensure that the water needs of the ecosystems are met.

When considering environmental flows for surface water systems, once environmental water provisions have been decided it is a relatively straightforward matter to regulate surface water storages and surface water extraction to achieve the desired flows. For groundwater systems, the situation is much more complex. Even after water provisions have been decided, it is not always immediately clear how changes to groundwater management will affect these provisions. This is the fifth step of the five step process (see Section 2.2). In the following section, the abilities of different water allocation mechanisms to sustain flows to groundwater dependent ecosystems is briefly discussed.

4 WATER ALLOCATION MECHANISMS IN SOUTH AUSTRALIA

Water allocation mechanisms that have been used in South Australia include:

- Allocation of a percentage of groundwater recharge
- Use of groundwater level benchmarks
- As a separation distance between extraction wells and the GDE (buffer zones)

4.1 As a percentage of groundwater recharge

This is the approach that has been most widely adopted, probably largely due to its ease of implementation. Groundwater allocations for GDEs range from 10% of aquifer recharge in the South East to 90% in the case of the Tertiary Sands aquifer in the Musgrave and Southern Basin PWA.

The approach represents a modification to the now rejected concept of aquifer 'safe yield'. The safe yield of an aquifer has been defined as: "the attainment and maintenance of a long-term balance between the amount of groundwater withdrawn annually and the annual amount of recharge" (Sophocleous, 1997). It is now widely understood that this approach ignores natural groundwater discharges, and so does not include a water allocation for the environment (Bredehoeft, 1997; Sophocleous, 2000; Cook et al., 2001). The obvious modification is to allow a fraction of the recharge for the environment, and to allocate the remainder.

However, nominal allocation of a fixed percentage of groundwater recharge to GDEs does not ensure their protection. As discussed in Section 2.1, the environmental impact of groundwater extraction depends not only upon the volume extracted, but also on the location of pumping bores relative to recharge and discharge areas, and sometimes also to the timing of the extraction. It is clear that even for a given basin, it is not possible to define a volume of extraction that would ensure protection of GDEs. The environmental consequences of groundwater exploitation will depend on the characteristics of the particular extraction scheme being considered, not just on the volume of water to be taken. Management of total volume is thus unlikely to achieve environmental provisions.

4.2 Groundwater level benchmarks

The most notable example of the use of groundwater level benchmarks to protect ecosystems is for the mounds springs of the Great Artesian Basin. Under the Roxby Downs (Indenture Ratification) Act 1982, the Olympic Dam mining project is allowed to take underground water from the GAB, provided that potentiometric pressures at the boundaries of the designated area are not reduced by more than 5 m, so that stress on the mound springs is controlled.

Maintenance of groundwater levels in the vicinity of GDEs will result in maintenance of groundwater flows to those ecosystems. This is usually the required result, and so this allocation mechanisms is most likely to achieve the desired purpose of protecting the ecosystems in question. While maintenance of groundwater levels is more practical than maintenance of groundwater flows (which can be difficult to measure), it is still difficult to apply where there are many groundwater users (due to monitoring requirements). It is most easily applied in areas where there is one (or few) major groundwater users, such as for town water supplies or large mines.

4.3 Use of buffer zones

A few WAPs combine volumetric allocations with the use of buffer zones. However, as discussed in Section 2.1, buffer zones will be ineffective if extraction is allowed to concentrate on the edge of these zones. In the Clare Valley, zones of influence of 200 m radius have been defined around groundwater dependent ecosystems. Groundwater extraction is limited to a maximum usage of 24 ML/yr, a minimum separation between wells of 200 m, and rules banning overlap of zones of influence. The zone of influence surrounding a well is calculated as a circular area centred around the licensed well, given by

$$Area\ of\ Influence = \frac{Q}{R} \tag{2}$$

where Q.is the pumping rate and R is the aquifer recharge rate

The regulation concerning areas of influence for adjacent bores effectively limits total groundwater extraction in the basin to be less than total aquifer recharge. (In practice, it will probably mean that extraction will be less than 80% of total aquifer recharge, because small areas between adjacent zones of influence are likely to be too small for future development.) The consequence of the rule preventing overlap with zones of influence of GDEs is such as to enforce a minimum distance from GDEs equal to

$$a = 200m + \sqrt{\frac{Q}{\pi R}} \tag{3}$$

However, the regulation for separation of adjacent bores acts to further spread extraction away from groundwater dependent ecosystems (and prevents concentration of extraction on the edge of these zones). As discussed above, buffer zones are of reduced value if bores are concentrated on the edge of the buffer zones, but the bore separation rule prevents this from occurring.

5 DISCUSSION

Traditionally, groundwater management has sought to evaluate viable long-term extraction rates by monitoring trends in groundwater levels. Aquifers were considered to be overexploited wherever water levels were declining, or where adverse environmental impacts were noted. However, this approach represents a misunderstanding of groundwater flow processes. Declines in water level will always occur after commencement of groundwater extraction, and reflect a temporary decrease in aquifer storage, which occurs before a new equilibrium is established. Furthermore, the absence of measurable environmental impact does not mean that there will be no significant environmental impact in the future. There will always be a time lag between groundwater extraction and reduction in natural discharge, and so the current apparent health of an exploited aquifer and the ecosystems that depend upon it does not necessarily indicate that the situation will be sustainable in the longer term. This approach also assumes that groundwater allocations can be reduced if they are found to be too high. In practice, political and social constraints have meant that this is very difficult to achieve.

Assessment of water requirements for many GDEs is hampered by the scarcity of detailed scientific studies, and there is a need for the development of additional tools. There is also a need for water allocation mechanisms to be more closely linked to the water provisions that it is desired to protect. The impact of groundwater extraction on ecosystems will depend on aquifer characteristics, and on the relative locations of extraction wells and GDEs. Regulation of groundwater use using simple basin-wide volumetric allocations will usually not be sufficient to ensure protection of critical ecosystems. Use of groundwater level benchmarks is a much more robust method, but is not easy to implement where there are multiple groundwater users. Buffer zones may represent the best compromise between ease of implementation and likelihood of achieving the desired result, but these need to be carefully designed.

6 REFERENCES

Bredehoeft J. (1997) Safe yield and the water budget myth. Ground Water, 35(6):929.

Cook P.G., Herczeg A.L. and Harrington G.A. (2001) Is groundwater sustainability possible? In 2001 Water Odyssey. AWA 19th Federal Convention, Canberra, 1-5 April 2001.

Cook P.G., Hatton T J., Pidsley D., Herczeg A.L., Held A., O'Grady A.O. and Eamus D. (1998) Water balance of a tropical woodland ecosystem, Northern Australia: a combination of micro-meteorological, soil physical and groundwater chemical approaches. J. Hydrology, 210:161-177.

Cook P.G., Jolly I.D., Walker G.R. and Robinson N.I. (2002) From drainage to recharge to discharge: some timelags in subsurface hydrology. Dubai International Conference on Integrated Management of Water Resources in the Third Millenium. 2-6 February 2002, Dubai.

Dillon P., Benyon R., Cook, P., Hatton T., Marvanek S. and Gillooly J. (2001) Review of research on plantation forest water requirements in relation to groundwater resources in the southeast of South Australia. Centre for Groundwater Studies, Report No. 99.

Ellins K.K., Roman-Mas A. and Lee R. (1990) Using ^{222}Rn to examine groundwater/surface discharge interaction in the Rio Grande De Manati, Puerto Rico. J. Hydrology, 115:319-341.

Groom P.K., Froend R.H. and Mattiske E.M. (2000) Impact of groundwater abstraction on a Banksia woodland, Swan Coastal Plain, Western Australia. Ecological Management and Restoration, 1(2):117-124.

Hatton T.J. and Evans R. (1998) Dependence of Ecosystems on Groundwater and its Significance to Australia. Occasional Paper No 12/98. Land and Water Resources Research and Development Corporation, 77p.

Knight J.H., Gilfedder M. and Walker G.R. (2002) Impacts of irrigation and dryland development on groundwater discharge to rivers - a unit response approach to cumulative impacts analysis, CSIRO Land and Water, Technical Report 03/02, 16p.

Lamontagne S. (2002) Groundwater-dependent eco-systems in South Australia. Proc. of the Science of Environmental Water Requirements in South Australia seminar, 24 Sep. 2002, Adelaide.

Sophocleous M. (1997) Managing water resources systems: why "safe yield" is not sustainable. Ground Water, 35(7):561.

Sophocleous M. (2000) From safe yield to sustainable development of water resources - the Kansas experience. J. Hydrology, 235:27-43.

Stromberg J.C., Tiller R. and Richter B. (1996) Effects of groundwater decline on riparian vegetation of semiarid regions: the San Pedro, Arizona. Ecological Applications, 6(1):113-131.

Walker G., Brunel J.P., Dighton J., Holland K., Leaney F., McEwan K., Mensforth L., Thorburn P. and Walker C. (2001) The use of stable isotopes for determining sources of water for plant transpiration. CSIRO Land and Water, Technical Report 05/01, 36p.

7 ACKNOWLEDGEMENTS

The authors would like to thank Stuart Richardson (REM) for background information, and helpful discussions regarding current practices in water allocation.

Environmental Water Requirements of Groundwater Dependent Ecosystems – The South Australian Approach in the National Context

Richard Evans [1]

SUMMARY

The technical understanding and management approaches to dealing with groundwater dependent ecosystems (GDE) in Australia are reviewed. A modest increase in understanding and appreciation of GDE's is occurring. The understanding of the temporal distribution of the groundwater regime and its impact on GDE health is poor. Nationally, all States are making efforts to include consideration of GDE's in their water allocation and planning process. In many practical cases this translates to setting minimum groundwater levels. There does not appear to be any clear linkage between allocation volumes and groundwater level targets in some parts of South Australia. Also the relationship between any target levels and GDE health is unclear. A conceptual framework is proposed for assessing environmental water requirements and then translating them into environmental water provisions for GDE's.

- Significant technical knowledge gaps exist in our understanding of the groundwater regime affecting GDE health
- The linkage between allocation volumes and a specified groundwater regime is generally poor
- A conceptual framework for translating the EWR into the EWP is proposed

1. INTRODUCTION

The appreciation and understanding of the ecosystems which have a dependence on groundwater is rapidly increasing. Nonetheless the general level of Australian scientific understanding of groundwater dependent ecosystems (GDE) is at a relatively basic level, with a few significant exceptions. Even more rudimentary, however, is the general level of understanding of appropriate methods to be used to determine the environmental water provisions (EWP) for GDE's. The difference between EWR and EWP is summarised in Figure 1. This distinction is important and often not appreciated.

This paper follows the two papers by Lamontagne (2002) and Cook and Lamontagne (2002) and is designed to provide a technical and management perspective of the South Australian approach to dealing with the environmental water requirement (EWR) assessment for GDE's in the National context. The basic technical understanding for GDE's is not presented here and it is assumed that the reader has a good knowledge of GDE processes. The debate currently underway on methods to determine EWR for GDE's is considered.

2. EVOLUTION AND ADVANCES IN TECHNICAL UNDERSTANDING

2.1 Classification

Hatton and Evans (1998) classified groundwater dependent ecosystems into four major types. The scope of their brief specifically excluded marine processes and faunal linkages where included in the four types. Subsequent thinking (Clifton and Evans, 2000) has resulted in a suggested classification of six major types:

- terrestrial vegetation – vegetation communities and dependent fauna that have seasonal or episodic dependence on groundwater;
- river base flow systems – aquatic and riparian ecosystems that exist in or adjacent to streams that are fed by groundwater base flow;
- aquifer and cave ecosystems – aquatic ecosystems that occupy caves or aquifers;
- wetlands – aquatic communities and fringing vegetation dependent on groundwater fed lakes and wetlands;
- terrestrial fauna – native animals that directly use groundwater rather than rely on it for habitat;
- estuarine and near-shore marine ecosystems – coastal, estuarine and near shore marine plant and animal communities whose ecological function has some dependence on discharge of groundwater.

The last two additions are considered to be significant in many parts of Australia.

[1] Sinclair Knight Merz, 590 Orrong Road, Armadale, Victoria, 3143
Telephone: 03 9248 3369, Facsimile: 03 9248 3364, Email: revans@skm.com.au

Figure 1: Allocating water to meet the environmental needs of groundwater dependent ecosystems: the key stages

Groundwater dependency	Determine the important groundwater dependent ecosystems and the nature of threats to key ecological processes
Environmental water requirement (EWR)	Develop a process by which the water regimes needed to sustain key ecological values of groundwater dependent ecosystems at a low level of risk are determined
Environmental water provision (EWP)	Develop a process for groundwater allocation that balances water requirements to sustain key ecological values of dependent ecosystems and broader social and economic objectives for the resource

Adapted from Water and Rivers Commission (1999).

2.2 Areal Coverage

The conclusion by Hatton and Evans (1998) that only about 2% of the Australian continent is covered by entirely or highly dependent ecosystems is now considered to be somewhat incorrect and even misleading. Much work by Humphreys (2000 and 2001) and Thurgate et al. (2001) and others has demonstrated a wider distribution of stygofauna than previously thought. In addition other researchers (not yet published) have demonstrated significant floral dependencies on groundwater which were previously thought to be minor. An example is forestry plantations in the South East of South Australia. Furthermore, quoting any percentage is misleading, as even though the area might be small the significance of groundwater processes is paramount. This especially applies to groundwater sustaining life in arid areas. Another example is the base flow to streams and rivers. Recent work by Boulton (2000) illustrates this.

2.3 Definition of Groundwater

This is currently an issue of debate. Lamontagne (2002) defined groundwater essentially the same way as Hatton and Evans (1998). They both exclude unsaturated zone water, seasonal perched water tables and bank storage. This may be viewed by some as a pedantic matter, however it has some significance in groundwater and surface water licensing in being able to precisely define what is being licensed and hence traded. In addition, often surface water managers tend to somewhat simplistically view groundwater as being all water below the land surface. The danger in not defining groundwater carefully is that groundwater managers may be held responsible for the health of ecosystems which are dependent on water in the unsaturated zone.

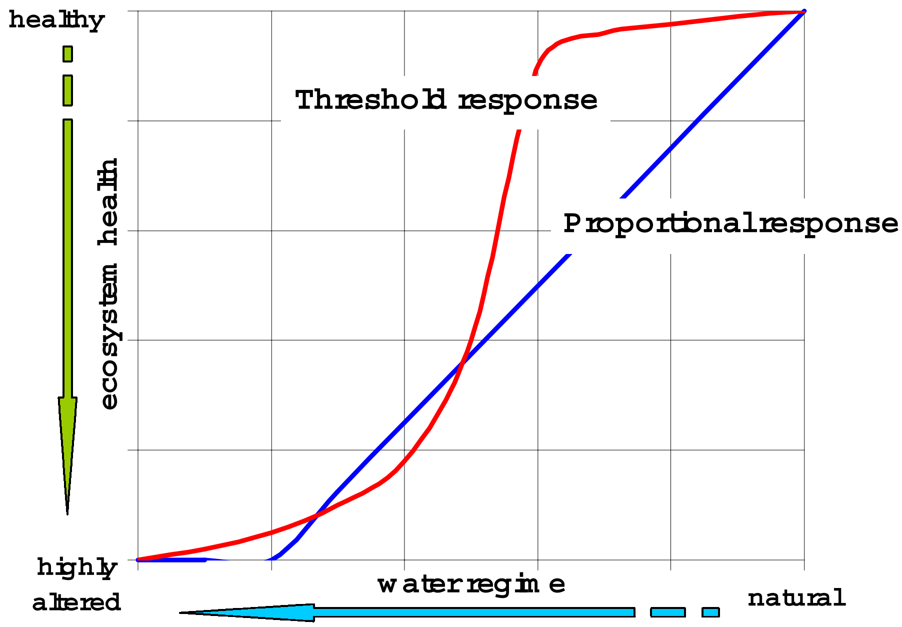
2.4 Groundwater Regime

Clifton and Evans (2000) emphasise that defining the groundwater regime and hence the nature of the GDE dependency is critical to understanding the environmental water requirements (EWR) for GDE's. This is also emphasised by Lamontagne (2002) and Cook and Lamontagne (2002). This includes two key aspects:

- Key Attributes. These are flux, level, pressure and quality. Cook and Lamontagne (2002) make the valid point, when considering quantity aspects, flux, level and pressure are essentially the same thing. This is true from a technical perspective, however from a management perspective, it is necessary to be able to define these different attributes separately.

- Temporal Distribution. including timing, frequency, duration and episodicity for which groundwater is used. This aspect remains perhaps the key knowledge gap and major research is required to address this. Boundary or threshold values, as distinct from proportional responses, are an especially key issue. This is illustrated in Figure 2. The importance of understanding the temporal water balance has been shown in several projects (eg Cook et al., 1998) where an annual water balance would be quite misleading, and separate dry season and wet season balances are required to understand the flow regime.

Figure 2: Illustrations of the broad types of response function between ecosystem health and water regime (from Clifton and Evans, 2000)



2.5 Groundwater Response and Time Lag

Lamontagne (2002) and Cook and Lamontagne (2002) emphasise that there can be a significant lag time in groundwater responses. It is important to appreciate that there are two components:

- Firstly, there is usually a lag time (often years, and sometimes decades or longer) between groundwater extraction (or change in recharge) and the decrease in discharge.
- Secondly, there is a lag time between decrease in discharge (eg groundwater flow into a wetland, or fall in groundwater level beneath a forest) and consequent change in ecosystem health. This poses a significant problem for both research and management.

2.6 Groundwater Quality

The significance of changes in groundwater quality on ecosystem health is especially poorly known. An exception is the impact of saline groundwater on floral health. The effects of pollution due to hydrocarbons, nitrate, heavy metals, bacteria and other organic contaminants on ecosystem health is generally not well known. Even relatively ordinary parameters, such as temperature of groundwater, could have a significant impact, albeit mostly unknown. In most practical management cases, quantity and quality are interrelated and hence both need to be considered.

3. SUMMARY OF CURRENT APPROACHES TO DEFINING EWR FOR GDE'S

3.1 General

To the author's knowledge there have been few technically comprehensive studies undertaken to understand the hydrogeological regime influencing ecosystem health. A clear distinction is drawn between studies undertaken to describe the various groundwater dependent ecosystems present in a region, as distinct from studies which have investigated the groundwater processes which influence ecosystem health. The more significant process studies which have been undertaken in Australia are:

- Howard East, Northern Territory
- Perth coastal wetlands, Western Australia
- Gnangara Mound, Western Australia
- Chowilla, South Australia
- Mound Springs, parts of the Great Artesian Basin

3.2 Terrestrial Vegetation

No generally accepted method or approach exists for specifying the EWR for terrestrial vegetation. In a few cases (see Hatton and Evans, 1998) where a clear dependence has been established or suspected then a minimum groundwater level is specified and the groundwater extraction theoretically managed to this minimum level. The linkage between volume used and the desired minimum groundwater level is often poorly established, although methods are available to define the linkage (eg. modelling). Of greater importance, the relationship between the temporal variation of the groundwater regime and the vegetation health is generally unknown. It is possible that specifying a fixed (albeit minimum) level could actually be detrimental to health, although it is recognised that in practice this would rarely occur as there is usually considerable groundwater level variation.

Methods are available to determine if terrestrial vegetation is or is not dependent on groundwater. In many cases it is not self evident. Isotope methods (see, for example, Cook et al., 1998) provide a practical tool to establish the linkage. However, as with any method, careful interpretation is required because at the time of sampling the vegetation may not be using groundwater.

3.3 River Base Flow Systems

The study of surface water / groundwater interaction is receiving much attention at this moment and advances are being made, as illustrated by Cook and Lamontagne (2002). SKM (2001) reviewed the current understanding of surface water / groundwater interaction in the Murray Darling Basin and concluded that about 60% of all groundwater extracted would have ended up as stream flow. SKM (2002) assessed the base flow component of 178 unregulated streams in the Murray Darling Basin and derived a mean annual base flow index of 25%. (The base flow index is a statistically derived measure of the relative proportion of base flow to total stream flow).

River base flow is made up of several components:

- groundwater discharge
- bank storage (water held temporarily in aquifers adjacent to streams)
- unsaturated flow
- surface water delayed drainage, for example from lakes, wetlands and certain tributaries
- groundwater delayed drainage, for example from shallow seasonal perched aquifers

It is important to be able to separate out these base flow components so that the true role of groundwater can be established and hence managed. It is recommended that this is a key research need.

Currently the fundamental significance of groundwater in influencing river ecosystem health is poorly recognised. There are, however, a few exceptions. For example, the Gellibrand Groundwater Management Area in southern Victoria was assigned a zero Permissible Annual Volume (ie zero allocations) specifically because of concern over a reduction in the base flow to the Gellibrand River.

3.4 Aquifer and Cave Ecosystems

Several States are currently grappling with how to deal with this GDE. For example, a Marble mine in Queensland has recently been closed down because of concern about dewatering lowering groundwater levels in an adjacent cave which contained important GDE's. The method being applied now is to normally consider a minimum groundwater level for these systems. The technical basis for the level is often unknown.

3.5 Wetlands

Pioneering work on wetlands in the Perth coastal plain (see, for example, Arrowsmith, 1996) has provided the benchmark for wetland assessments. Substantial monitoring and modelling was undertaken to define the hydraulic relationship between wetland water level and the groundwater regime. In turn the relationship with groundwater pumping and other factors (eg. climate, urban hydrology) was established. In addition, the ecological health response to varying wetland water levels was deduced. This led to specifying a range of desired wetland water levels and the associated permitted groundwater extraction regime. In spite of this excellent work it has generally not been repeated elsewhere, largely because of the cost of the investigations which would be required. The Great Artesian Basin mound springs are another obvious exception. Nonetheless groundwater interactions with wetlands are being often studied. Applying the derived understanding to practical groundwater management often does not occur.

3.6 Terrestrial Fauna

The author is not aware of any cases where fauna have received a specific GDE allocation.

3.7 Estuarine and Marine Ecosystems

This potentially represents a huge field of research that is currently almost absent. For example, the dependence of many ecosystems (eg turtles, crocodiles and smaller vertebrates and invertebrates) on relatively fresh groundwater discharge in estuarine environments controlling reproduction cycles is suspected but poorly understood. Some work is planned in the South East of South Australia.

4. THE NATIONAL AGENDA

ARMCANZ (1996) proposed the bold initiative to apply the National Principles for the Provision of Water for Ecosystems to protect GDE's. At that time the only State that was effectively considering GDE's at all was Western Australia. Hatton and Evans (1998) produced the first national assessment of GDE's. Clifton and Evans (2000) followed this national review by proposing a methodology for determining the EWR of GDE's. This approach is summarised in Figure 3. They went on to propose a methodology for translating the EWR into EWP. This is summarised in Figure 4. It is strongly emphasised that the effort to assess the EWR can vary depending upon resource and information availability. Obviously, there is a corresponding variation in the confidence in the EWR determination.

There is much debate underway in Australia on the definition of Sustainable Yield – see, for example, Russel (2002). The debate rarely involves any thorough consideration of GDE's. In many situations there is no absolute sustainable yield. In practice there is a complete continuum from zero sustainable yield up to very large numbers, depending upon how much impact on GDE's is considered appropriate. Defining what groundwater regime causes various impacts on GDE health is a key input to the debate on sustainable yield values.

The growing realisation throughout Australia of the significance of GDE's has prompted most States to enact legislation to require consideration of GDE's. Clifton and Evans (2000) have described the various approaches adopted by various States. Cook and Lamontagne (2002) discusses three common approaches undertaken to protect GDE"s:

- allocation of a % of groundwater recharge
- use of groundwater level benchmarks
- use of buffer zones

Even though some States have adopted a % of recharge to protect GDE's (eg. NSW has nominally allocated 30% of recharge to GDE's; DLWC, 2002), in most environments there is little to no practical link to how this actually protects GDE's. The use of groundwater level benchmarks is undoubtedly a technically defensible position for some GDE's, although as discussed earlier in many cases it is far more complex than just a single level. Defining the groundwater regime is required.

It is recommended that the GDE's should be viewed as an asset to be protected. The GDE assets need to be clearly defined in water allocation plans (WAP) for prescribed water resources and protection strategies developed. Ultimately through the community input process to the WAP process, the community will need to decide if it really wants to maintain the asset relative to the economic benefit of the use of the groundwater.

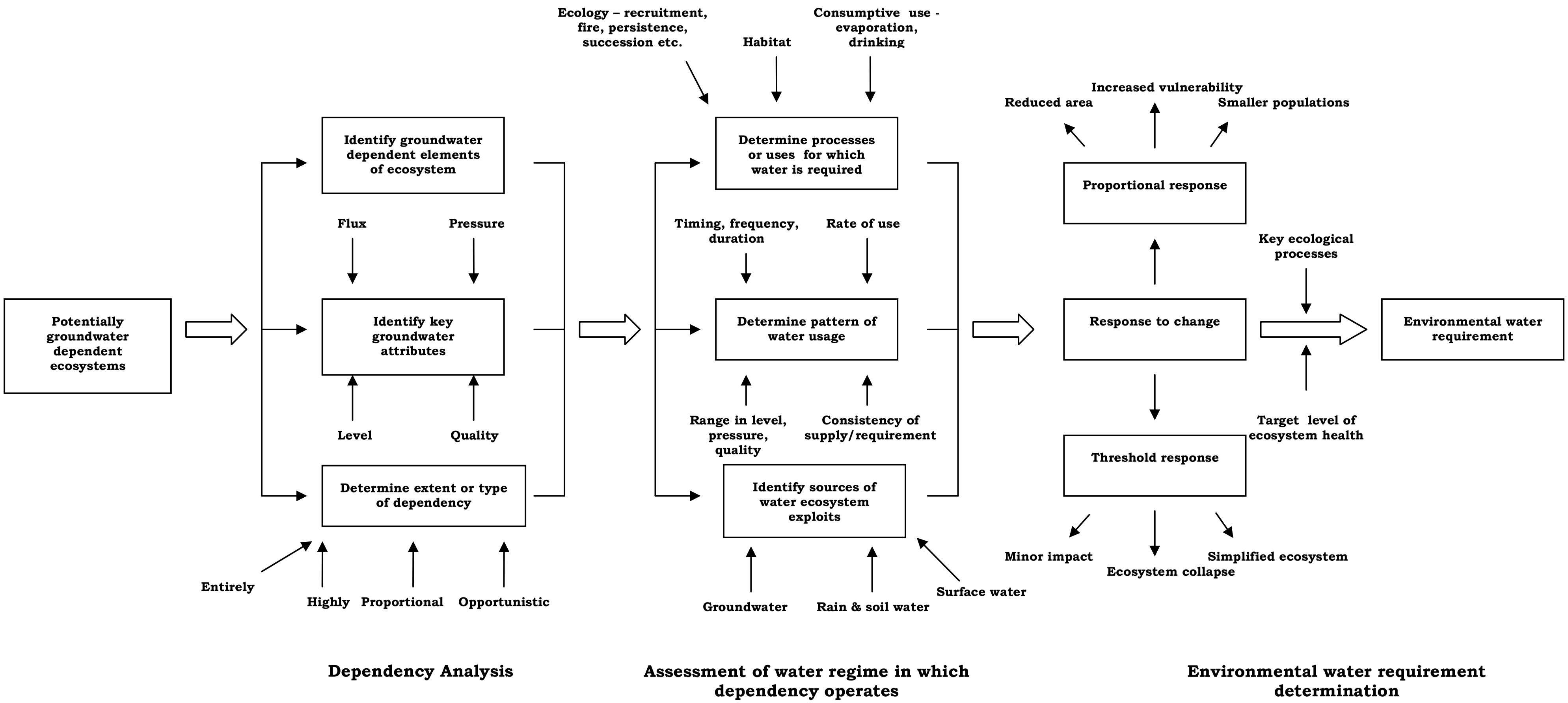
A very recently completed program (Merrick et al., 2002) allows for the specification of groundwater levels to meet GDE requirements for defined allocations. Conventional numerical modelling is routinely used for this purpose. This new analytical program provides a simpler (and hence less expensive) tool.

In view of the high cost of research, a partial way forward is a well-targeted monitoring program, which over time will assist in understanding the relationship between GDE health and the groundwater regime. It is considered that little of the current monitoring program will be useful for this purpose as, for example, most bores are monitoring deeper aquifers.

5. THE SOUTH AUSTRALIAN APPROACH

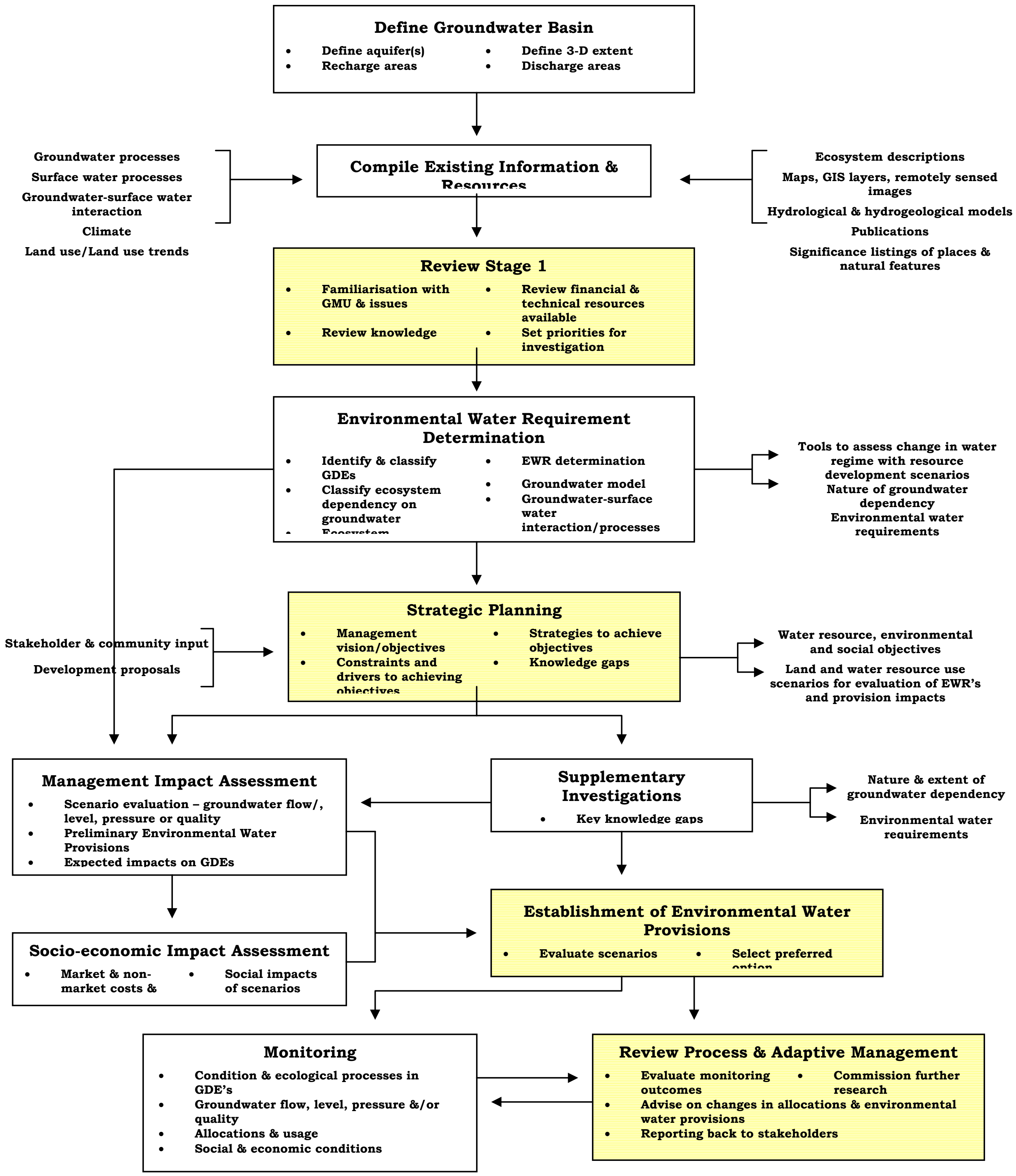
Cook and Lamontagne (2002) describe the requirements of the Water Resources Act (1997). Even though a significant effort has been made to describe the nature of GDE's in South Australia through the WAP process, the relationship between the volume licensed for extraction and the water level elevations specified to protect GDE's is either unclear or non-existent. Furthermore the basis for the water level targets is generally not based on science. Hence the ability to actually protect the GDE is generally also unknown. Considering the Australia wide dearth of understanding of GDE water requirements, the SA approach is understandable. Even though Western Australia is clearly leading the way, the SA approach is not too far behind. The high level of groundwater allocation in the south east of SA (and even over-allocation) means that the nominal 10% recharge to be assigned to GDE's is a bold step. However it is likely that it is inadequate and a greater provision for GDE's will be required. Nonetheless the notional 10% (or whatever other notional % is assigned in the future) in practice bears little relationship to specific GDE requirements. To move forward it is necessary to identify specific GDE's at risk – assets – and systematically specify appropriate groundwater regime requirements, as in Figure 4. For the moment they might have to be based on "expert opinion" – ie. little real science, but nonetheless this will be a step forward.

Figure 3: Conceptual framework for determination of environmental water requirements of groundwater dependent ecosystems



From Clifton and Evans (2000)

Figure 4: Framework for assessing environmental water provisions for groundwater dependent ecosystems (participatory steps shaded)



From Clifton and Evans (2000)

6. CONCLUSIONS

The recognition of the importance of GDE's is slowly increasing. There are major scientific knowledge gaps present which will only be filled by significant research programs. The relatively few studies undertaken to date have generally produced unexpected results (for example, the vegetation at Howard East was shown to be not groundwater dependent). The major scientific uncertainty is in defining the groundwater regime which controls GDE health. The temporal aspects of this regime is especially unknown.

The South Australian approach is rudimentary, but nonetheless it is a first step. The community will have to decide the value to be placed on GDE's. Within this context GDE assets will need to be identified and by an ongoing program of research, investigation and monitoring, the controlling groundwater regime attributes will need to be defined.

A possible process for determining EWP (Fig. 4) is included because the lack of priority for GDE's is partly driven by a lack of any accepted method to define the EWP. If such a method was accepted then it is believed that this would drive the science forward.

7. REFERENCES

ARMCANZ (1996) "Allocation and Use of Groundwater – A National Framework for Improved Groundwater Management in Australia". Policy Position Paper for Advice to States and Territories.

Arrowsmith, A (1996) "Developing Criteria for Wetland and Vegetation Management Within Groundwater Abstraction Areas". Groundwater and Land Use Planning, Centre for Groundwater Studies.

Boulton, AJ (2000) :"River Ecosystem Health Down Under: Assessing Ecological Condition in Riverine Groundwater Zones in Australia". Ecosystem Management, University of New England, New South Wales.

Clifton, C and Evans R (2000) "Environmental Water Requirements of Groundwater Dependent Ecosystems" Environment Australia.

Cook, PG, Hatton, TJ, Herczeg, AL, Held A, O'Grady, AO and Eamus, D (1998) Water Balance of a Tropical Woodland Ecosystem, Northern Australia: A Combination of Micro-meteorological, Soil Physical and Groundwater Chemical Approaches. J. Hydrology. 210:161-177.

Cook, P. and Lamontagne, S. (2002) "Assessing and protecting water requirements for groundwater dependent ecosystems" Proceedings of the Science of Environmental Water Requirements in SA.

Department of Land and Water Conservation (2002). "The NSW State Groundwater Dependent Ecosystems Policy". A Component Policy of the NSW State Groundwater Policy Framework Document.

Hatton, T and Evans, R (1998) "Dependence of Ecosystems on Groundwater and its Significance to Australia". Land and Water Resources, Research and Development Corporation.

Humphreys, WF (2000) "The hypogean fauna of the Cape Range Peninsula and Barrow Island, Northwestern Australia." In Wilkens, H, Culver, DC and Humphreys, WF (eds) Ecosystems of the world, volume 30: Subterranean ecosystems: 581-601. Elsevier, Amsterdam.

Humphreys, WF (2001) "Groundwater calcrete aquifers in the Australian arid zone: The context to an unfolding plethora of stygal biodiversity". Records of the Western Australian Museum. Supplement No 64, 63-83.

Lamontagne, S (2002) "Groundwater-Dependent Ecosystems in South Australia". Proceedings of the Science of Environmental Water Requirements in SA.

Merrick, NP, Evans, R and Merrick, DP (2001) "Groundwater Hot Spot Management: Principles, Methodologies and Software". Report on behalf of Insearch Limited.

Russell, L (2002) Development of an Approach and Definition of Sustainable Yields. Balancing the Groundwater Budget. International Groundwater Conference. Darwin. May 2002.

Sinclair Knight Merz (2001) "Projections of Groundwater Extraction Rates and Implications for Future Demand and Competition for Surface Water". Murray Darling Basin Commission.

Sinclair Knight Merz (2002) "Survey of Baseflows in Unregulated Catchments in the Murray-Darling Basin" Murray Darling Basin Commission.

Thurgate, ME, Gough, JS, Clarke, AK, Serov P, and Spate A (2001) "Stygofauna diversity and distribution in Eastern Australian cave and karst areas." . Records of the Western Australian Museum. Supplement No 64, 49-62.

Water and Rivers Commission (1999) "Draft environmental water provisions policy for Western Australia. A draft policy statement to support implementation of the COAG Water Reform Framework Agreement (1994)." Water and Rivers Commission, Water Reform Series, WR4.

I:\wcms110.1\papers\SA GDE4doc