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Groundwater-dependent ecosystems: the where, what and why of GDEs

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Introduction

Principles for allocation of groundwater to the environment

Until the early 1970s, the management of water resources in Australia was predominantly concerned with the assessment, development and harnessing of new water resources for irrigation, urban and industrial, stock and domestic water supply. The consequences of excessive and unsympathetic groundwater abstraction on groundwater-dependent (phreatophytic) vegetation, such as tree decline and mortality, have been observed throughout Australia (Arrowsmith 1996; Hatton and Evans 1998; Clifton and Evans 2001). With increasing demand for water and a changing climate regime, the need to mitigate the environmental impacts of groundwater development is increasing. Current borefield operation in Australia is largely responsive to consumption demand and often in conflict with environmental needs for groundwater, resulting in drought stress and sometimes death of phreatophytic vegetation and other impacts on GDEs.

Groundwater resource managers commonly ask how much water can be taken from the aquifer while still maintaining a low level of risk to GDEs. This requires quantified information on the relationship between the health of a GDE and groundwater depth (or other parameter; see Eamus *et al.* 2006*a*). Recommendations are generally made by defining the acceptable level to which groundwater can be allowed to fall, while maintaining important environmental values (see Murray *et al*. 2006).

The Council of Australian Governments (COAG) endorsed reforms in 1994 to achieve a sustainable water industry that included allocations for the environment and greater environmental accountability of water-resource developments. The National Principles for the Provision of Water for Ecosystems (ARMCANZ/ANZECC 1996) produced by the Agriculture and Resource Management Council of Australia and New Zealand (ARMCANZ) and the Australia and New Zealand Environment and Conservation Council (ANZECC) provide the basis for considering ecological water requirements (EWR) as part of water allocation decisions by water resource managers throughout Australia (see Mackay 2006 for discussion of this in relation to South African regimes).

These principles propose EWRs[1] as the scientific information necessary to inform water resource managers and decision makers of the water required to sustain, and where necessary restore, ecological processes and biodiversity of recognised associated ecosystems. The allocation of water to the environment, or the environmental water provision (EWP), is the water that will be allocated after consideration of social, economic as well as ecological water requirements and may involve trade-offs between these requirements. Clearly, it is desirable that the EWR and EWP are the same. However, they may not be equal because of conflicts over the use of water. In such cases the issue of whether the EWP should be equal to or less than the EWR will largely depend on the relative importance placed on the protection of ecological values by the community concerned (see Murray *et al*. 2006).

Estimations of the water regimes required by an ecosystem are developed through strategic scientific research or through the application of local knowledge based on many years of observation. Determination of EWRs for an ecosystem involves identifying those aspects of the natural water regime

[1]It is worthy to note the term 'EWR', in essence, refers to the intrinsic water requirements of ecological attributes of water-dependent ecosystems. In the context of this report, however, 'EWR' refers to what may be more accurately described as 'Regulatory EWRs'. The determination of regulatory EWRs is framed by socially constructed ecological values and ecological management objectives (differentiated from intrinsic EWRs, which exist irrespective of social values).

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that are most important for maintaining key ecosystem features and processes. EWRs include elements of quantity and duration and apply both spatially and temporally and are used to inform water-resource management and decision makers in the determination of EWPs.

Before water provisions to the environment can be set, identification of environmental, social and economic values need to be considered and trade-offs may need to be proposed to protect values. It is generally recognised that ecosystems are dynamic in nature and are continually changing in response to natural processes, although a greater degree of change and/or an accelerated rate of change may be induced by altered water regimes. Depending on the ecological values attributed to an ecosystem there may be a degree to which an ecosystem may be altered from its 'natural' state and still remain acceptable within the community. The degree to which such changes are allowed is known as the 'limit of acceptable change'. The accepted level of change is determined from a balance of existing and perceived future uses and maintenance of inter-generational equity (Department of Environmental Protection 1996), generally described as the desired future state. This approach is consistent with State conservation strategies in Australia (e.g. Department of Conservation and Environment 1987) and with National and World Conservation Strategies (Department of Environmental Protection 1996). The desired future state represents the level of change between a 'natural' state and the complete breakdown of ecological integrity of an ecosystem that society is willing to accept (Department of Environmental Protection 1996).

Even though the principles of provision of groundwater to the environment are generally agreed on, in practice there are some significant issues yet to be overcome.

Groundwater-dependent ecosystem management: current issues

Management agencies and industries responsible for mitigating impacts on GDEs deal with many specific problems associated with planning and operation of groundwater-resource developments. Even though there have been significant advances in defining GDEs (Eamus *et al.* 2006*b*), the need for approaches to defining groundwater requirements of GDEs (Froend and Loomes 2004) still remains. Poorly defined EWRs, often the result of insufficient data and time, lead to 'technical' breaches of environmental conditions (without obvious ecological impact) or understated water requirements, resulting in unexpected environmental impacts. Below are several recommendations by the authors for future research and management frameworks, some of which are dealt with in this issue.

- Consider the groundwater requirements for as many components of a GDE for which necessary data are available. For example, this would require the determination of wetland GDE water requirements to be an integration of vegetation, vertebrate, macro-invertebrate and physicochemical water requirements. Single components may dominate the EWR assessment of particular GDEs if insufficient data exist to incorporate the other components of the ecology, or if the requirements of one component (e.g. 'umbrella' species) can be demonstrated to cater for all other key components.
- Acknowledge variability in groundwater requirements within ecological components of a GDE. Not all phreatophytic vegetation, for example, has the same degree of dependency on groundwater and therefore the same response to drawdown. This variability in dependence has a significant effect on the risk of impact from groundwater drawdown. The expression of EWRs should therefore incorporate the range in water requirements (not absolute 'threshold' values only) and/or categories of differing requirements/dependency.
- Recognise other groundwater variables important to the ecology of the GDE (e.g. duration, timing and rate of seasonal flooding/drying and the episodicity of extreme flooding/drying events), where data permit.
- Consider the cumulative effects of reduced groundwater availability by assessing historical changes in groundwater and determine the net change in availability over key periods of time. This historical change should then be considered in addition to any proposed impacts from future developments or increased allocations. A lag-response in a GDE may occur some time after initial alteration to groundwater availability. Identification of EWRs should consider the rate at which GDEs are likely to respond to changes in groundwater availability.
- Acknowledge the resilience of GDEs to altered groundwater availability. Ecological values may be able to be restored/maintained if remedial/mitigation practices are put in place. Therefore, a longer-term perspective in water requirements necessary to maintain ecological values should be adopted.
- Consider system/catchment-level groundwater requirements as well as single GDE requirements. Important landscape level ecological processes should be considered (e.g. acid sulfate soils).
- Define the uncertainty surrounding water requirements of GDEs and the groundwater models used to predict hydrological changes caused by future borefields, catchment land use and climate.

Other issues reflect how EWRs are used in the determination of environmental water provisions (EWPs) or probable impacts. These include

- absence of a risk (of impact) assessment that incorporates variability in current vulnerabilities (water requirements and drought stress) and potential degree of change/impact,
- management (environmental compliance) criteria based on simplified minimum 'threshold' water-table levels

without consideration of acceptable changes to ecological values, and
- direct translation of EWRs to EWPs or management criteria without sufficient consideration of social and economic requirements.

Previous research on phreatophytic vegetation has revealed seasonal variability in both the quantity of groundwater used and the relative importance of groundwater as a water source (Zencich *et al*. 2002). Use of groundwater by phreatophytes is highest during the driest season of the year when alternative water sources become depleted and transpirational demand is highest. It is also suggested that phreatophytes are susceptible to the rate as well as season of drawdown (Mahoney and Rood 1992; Stromberg and Patten 1992; Tyree *et al*. 1994; Scott *et al*. 1999, 2000; Groom *et al*. 2000; Shatfroth *et al*. 2000; Horton *et al*. 2001; Eamus *et al*. 2006*a*) by having a higher rate of water-table decline than fine-root elongation rate, and/or lowering the water table during a time other than the root growth season (Sorenson *et al*. 1991). Low magnitude and rates of change in groundwater levels as opposed to rapid drawdown, may allow intra- and inter-generational adaptation and persistence of phreatophytes (Scott *et al*. 1999; Shatfroth *et al*. 2000). Application of this knowledge in the planning and operation of borefield would minimise impacts on phreatophytic vegetation at least.

In areas of highest environmental risk, modification of groundwater pumping to be sympathetic to, rather than in competition with, environmental water demand offers benefits for sustainable operation of existing assets as well as environmental protection. The operation of borefields to be environmentally sympathetic, as opposed to risking GDE decline by competing for the resource when ecosystems are most vulnerable, is a novel concept that is not generally practiced in Australia or elsewhere (Clifton and Evans 2001; Eamus *et al*. 2006*b*). Use of ecosystem groundwater requirements and adaptability to formulate sustainable borefield planning and operations is arguably the ultimate goal of groundwater resource managers. Modification of the timing of abstraction and the magnitude and rate of drawdown may significantly reduce the risk to GDEs by avoiding times of peak environmental demand and allowing adaptation of dependent biota to a lower water table. However, there is little information currently available on the process of adaptation to altered groundwater availability. Some research has suggested that populations of phreatophytes may be able to adapt to low rates/magnitudes of groundwater drawdown through a combination of modification of fine-root distribution and corresponding altered water-source partitioning (Scott *et al*. 1999; Shatfroth *et al*. 2000). However, the magnitude, rate and seasonality of water-table reductions that permit (if at all) adaptation are not quantified, despite previous attempts to classify *Banksia* response by using predominately empirical relationships (Froend and Loomes 2004).

Modelling of GDE and groundwater interactions is perhaps a key research area that would benefit management by allowing the projection of GDE response to different magnitudes, rates and season of groundwater drawdown, as well as different climatic scenarios. Current groundwater models, such as the Perth Regional Aquifer Model System currently being developed and employed by the Water Corporation and Department of Environment in Western Australia, are far more advanced in application than models of GDE response, which are essentially conceptual only (with some exceptions specific to vegetation, e.g. soil–plant–atmosphere model) and require additional research on GDE response functions to be complete.

Lost groundwater production from existing borefield infrastructure because of environmental risk and regulation represents a significant economic loss to industry. For example, operation of existing borefields on the Gnangara Mound north of Perth, Western Australia, has been significantly constrained, with up to 40% of superficial bores for public water supply turned off in 'high risk' areas of shallow groundwater (<5 m) on the Gnangara Mound, in response to statutory water-level criteria (environmental approval criteria) imposed by State environmental regulators. Although this precautionary approach is valid, given the absence of information on phreatophyte adaptability and tolerance to drawdown, further research is required to determine the potential for operating 'high risk' borefields while mitigating impacts on GDEs of high conservation value. With a better understanding of ecosystem response to groundwater abstraction, reduced environmental impacts will benefit the broader community through reduced community expenditure on rehabilitation, maintenance of green open spaces for passive recreation, and maintenance of other ecosystem services provided by healthy urban and rural ecosystems. A significant benefit will also be reduced customer cost of water services as a result of cost recovery from existing infrastructure and improved certainty for future groundwater-development schemes.

This special issue

This special edition of the *Australian Journal of Botany* includes 11 research and review papers that can be divided loosely into four groups. The first consists of a single paper, by Eamus *et al*. This paper presents a practical and detailed description of how ecologists and resource managers can identify, in the field, the location of groundwater-dependent ecosystems and how groundwater abstraction (the timing, duration and amount of abstraction) can be managed in order to maintain a desired level of ecosystem function. Although Hatton and Evans (1998) provided a review of the nature and distribution of groundwater-dependent ecosystems, there appears to be no widely available synthesis of methods available to identify the location and nature (e.g. obligate *v.* facultative) of groundwater dependency nor any explicit consideration of the three fundamental questions that challenge managers of water and GDEs, i.e. (i) which attributes of the groundwater regime are important to the GDE (e.g. pressure, flow rate, depth), (ii) what are the safe limits to changes in groundwater regime and (iii) which features of vegetation can be measured to monitor ecosystem function? It isn't until we have identified the location of GDEs and the groundwater regime required to maintain them, that protection of these important ecosystems can be realistically attempted. This first paper sets the scene for the remainder of this special edition and provides consideration of these fundamental questions.

Having established a classification scheme for GDEs in the first paper, we move to the second group, consisting of five papers that deal with five distinct groundwater-dependent ecosystems. The first of these, by Bill Humphreys in Western Australia, deals with the diverse and highly endemic fauna found in Australian aquifers. Few people understand the complexity and diversity of the ecology of groundwater stores (aquifers) and this paper presents a review of the location, biodiversity, energetics, interactions with plants and conservation requirements of stygofauna of Australia. This paper shows that, far from being semi-deserts, groundwaters are dynamic systems with a complexity of structure and function approaching that of some terrestrial systems.

Groundwater does not, of course, remain in splendid isolation of surface waters. Recharge and discharge of groundwater can occur from and to rivers and the ecology of many rivers is therefore groundwater-dependent for at least part of the year. In the absence of groundwater input during extended periods of zero rain (e.g. the dry season in monsoonal Australia or during a drought), many dryland rivers draining arid and semi-arid Australia and even some rivers in monsoonal (tropical) Australia would cease to flow. The functional significance of groundwater input to the ecology of such rivers is only now being explored (Dent *et al*. 2000). Andrew Boulton and Peter Hancock review the hyporheic zone of rivers and review the functional dependency of river base-flow systems on groundwater at three spatial scales, channel reach, catchment and landscape, and assess four features of groundwater dependency, namely hydrological, physical, chemical and biological. In particular, they focus on the groundwater dependency of the hyporheic zone and the ecological significance of flow paths and groundwater residence time.

Rivers, of course, are surrounded by, and interact with, the riparian vegetation adjacent to them. Riparian forests can be structurally, floristically and topographically complex and determining the groundwater dependency of all or some of the components of such forests is a difficult task. By comparing the stable isotope composition of groundwater, soil water and xylem sap, it is sometimes possible to determine the sources of water being transpired (Zencich *et al*. 2002). O'Grady *et al*. examine the spatial and temporal patterns of transpiration by a complex riparian forest in tropical Australia and establish that different species access groundwater to differing extents and that the utilisation of groundwater varies seasonally. This theme is continued in a companion paper by O'Grady *et al*. who compare the rates of water use and groundwater utilisation of riparian communities, woodlands and open forests of a valley in Queensland. In addition to using stable isotope analyses, they use leaf water potentials to establish groundwater dependency of the dominant tree species and demonstrate that all communities showed some degree of dependency on groundwater supplies.

The final paper in this second group, by Ray Froend and Paul Drake, examines the vexed question of how to quantify the response of vegetation to a decline in the availability of groundwater. Whereas many authors have discussed the theoretical basis for predicting the form of the response of vegetation to changes in groundwater availability (Clifton and Evans 2001; Murray *et al*. 2003), this paper represents one of the first to establish a response curve from field data by using vulnerability of xylem to cavitation (Tyree and Sperry 1989) as a measure of the species response to reduced water availability. They identify a critical range of the percentage loss of conductance for four *Banksia* woodland tree species.

There is considerable interest in Australia in the use of trees to ameliorate dryland salinity (Hatton and Nulsen 1999; Hatton *et al*. 2003). Changes in catchment hydrology as a result of deforestation, with resultant movement of the water table towards the ground surface have been extensively documented (Hatton *et al*. 2003). Although not previously understood to be groundwater-dependent ecosystems, the direct interaction of woodlands, both plantation and native remnant, with groundwater through the processes of recharge and discharge of groundwater (Jolly and Cook 2002; Stirzaker *et al*. 2002), justifies their inclusion in this special edition and makes this third group of papers particularly important. In the first of these three papers, Richard Benyon *et al*. examine 21 plantations in south-eastern Australia and compare rates of water use of vegetation with changes in soil water content over periods of 25 years. Groundwater depth was between 1.5 and 22 m for the 21 sites and these authors show that groundwater was a significant resource for many sites, with between 108 and 670 mm of groundwater being used per year across all sites. They also establish that groundwater of low salinity can be used by some plantation species.

Groundwater has, in the past century, risen towards the ground surface in many parts of Australia (Dunin *et al*. 1999). As the water table has moved upwards, salt has been brought towards to the soil surface and groundwater, and indeed stream salinity has increased across much of temperate Australia (McFarlane and Williamson 2002). The ability of trees to reduce recharge of the water table, and the ability of trees to use groundwater of low-to-moderate salinity, is subject to an on-going investigation. In the second paper of this group of three, Kate Holland *et al*. investigate the interactions between saline groundwater, water uptake by trees and recharge. They show that most trees used water from the capillary fringe lying above the water table and that this capillary fringe was derived from a thin layer of relatively fresh water sitting on top of the saline groundwater. Furthermore, they show that recharge of groundwater along riparian forests occurred through two processes, namely direct infiltration of rainfall and bank recharge from floods. However, it is not possible to measure the interaction of saline groundwater and trees at every possible site and an ability to model the potential impacts on vegetation health of the utilisation of saline groundwater is of particular interest to landscape managers. Chowilla floodplains in the lower River Murray are undergoing significant salinisation (Akeroyd *et al*. 1998) and native riparian health, including the iconic river red gum and black box trees, are dying. Ian Overton *et al*. provide a model of the salinisation of the floodplain by using a spatial and temporal model of salt accumulation and this is used to infer vegetation health. However, most importantly, the inferences were calibrated against current vegetation health, which in turn was assessed with a combination of satellite and field data. They estimate that almost half of the 8000 ha of floodplain trees are affected by salinisation, compared with ~40% in 1994. The also conclude that the best management option for Chowilla is a combination of increased depth to groundwater and increased flooding frequency.

The final group of papers in this special edition deals with management issues. In the first of the two papers, Brad Murray *et al*. provide a practical methodology for prioritisation of the most valuable GDEs. When faced with several threats to several diverse GDEs, resource managers must allocate scarce resources (money, time, people) to the most valuable GDE. But how managers can establish the relative value of a GDE has not been established. Murray *et al*. propose an 8-step method for the valuation and prioritisation process and employ both economic and ecological values. Finally, Heather Mackay, from South Africa, provides a global perspective on the management and policy requirements for the protection of GDEs. She also provides a discussion of the newly emerging approaches to this problem.

In conclusion, this special edition tackles the major issues confronting managers of groundwater resources, GDEs and policy makers. Theoretical, modelling and practical perspectives are presented, including specific examples of important studies of a range of diverse range of GDEs.

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