Groundwater-dependent ecosystems and the dangers of groundwater overdraft: a review and an Australian perspective

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In many parts of the world, access to groundwater is needed for domestic, agricultural and industrial uses, and global groundwater exploitation continues to increase. The significance of groundwater in maintaining the health of rivers, streams, wetlands and associated vegetation is often underestimated or ignored, resulting in a lack of scrutiny of groundwater policy and management. It is essential that management of groundwater resources considers the needs of natural ecosystems, including subterranean. We review the limited Australian literature on the ecological impacts of groundwater overdraft and place Australian information within an international context, focusing on lentic, lotic, stygobitic and hyporheic communities as well as riparian and phreatophytic vegetation, and some coastal marine ecosystems. Groundwater overdraft, defined as abstracting groundwater at a rate which prejudices ecosystem or anthropocentric values, can substantially impact natural communities which depend, exclusively or seasonally, on groundwater. Overdraft damage is often underestimated, is sometimes irreversible, and may occur over time scales at variance to those used by water management agencies in modelling, planning and regulation. Given the dangers of groundwater overdraft, we discuss policy implications in the light of the precautionary principle, and make recommendations aimed at promoting the conservation of groundwater-dependent ecosystems within a sustainable use context.

Key words: groundwater-dependent ecosystems, precautionary principle, policy, environmental impacts, stygofauna, hyporheic zone, phreatophytes, aquifers, artesian springs, Great Artesian Basin, conjunctive management, ground water, groundwater overdraught[1].

INTRODUCTION

GROUNDWATER dependent ecosystems (GDEs) are ecosystems that are totally, partially or seasonally dependent on groundwater. They fall into three broad groups: surface terrestrial, surface aquatic and subterranean, and include distinct ecotones where different ecosystems meet. Water moves constantly between groundwaters and surface waters. Most rivers, lakes and wetlands are fed by, and feed groundwater to varying degrees at varying times. Groundwater feeds soil moisture through capillary action and percolation, and many terrestrial vegetation communities depend directly on either groundwater or the percolated soil moisture above the groundwater, for at least part of each year (Eamus *et al.* 2006a). Hyporheic zones (the mixing zone of stream-water and groundwater) and riparian zones are examples of ecotones largely or totally dependent on groundwater. Subterranean aquatic ecosystems exist within aquifers, and often contain unique biota (Humphreys 2008a; Humphreys 2001; Humphreys 1999). These systems are mostly completely unstudied and, in spite of recent advances, are not well understood, appreciated or protected in Australia (Tomlinson and Boulton 2008; Humphreys 2006b). Often, the role in nutrient mediation played by ecosystems within aquifers (denitrification) can have important consequences for surface water ecosystems.

Because groundwater and surface river flows are interconnected, extraction of groundwater linked to a river system will impact on that river. This is a simple but fundamental fact which water management agencies around the world still struggle with (Sophocleous 2010) and Australia is no exception (Nevill 2009; Evans 2007; Evans *et al.* 2006). Extracting more groundwater than is recharged is referred to here as *groundwater mining*. Extracting groundwater at a rate which prejudices important values (ecological or anthropocentric) is referred to here as *groundwater overdraft* (following Zektser *et al.* 2005). *Over-allocation* is defined here as the administrative permission of water extraction levels which cannot be sustained indefinitely without damaging other beneficial uses. Note that these definitions differ slightly from those used in Australia's *National Water Initiative* (Government of Australia 2004:32).

It is critical that overdraft of water resources (both ground and surface) be addressed, and, where possible, avoided by prudent foresight. The precautionary principle also has an important role to play, both in broad planning

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approaches and in the details of techniques used to estimate groundwater stocks and flows.

In warm climates, surface water storage results in major water loss through evaporation — particularly in wide, shallow dams. Good groundwater management which fosters aquifer use for water storage can result in more efficient storage than damming surface water, as evaporation is eliminated — although transpiration and drainage within aquifers must be taken into account. Integrated and comprehensive management of connected ground/surface water resources is needed, including careful provision of adequate environmental flows for groundwater-dependent ecosystems. Environmental flows are managed flows dedicated for ecosystem support (see Poff *et al.* 1997, Bunn and Arthington 2002). In many cases, better water management would see increased use of groundwater (in some cases accompanied by artificial recharge) combined with substantial reductions in surface water storage and use. In some areas, such as the arid parts of the Great Artesian Basin there is little or no surface water most of the time, and aquifer recharge is negligible in comparison to extraction, so good management of groundwater is critical.

The surface water environmental flow concept, applied to groundwater, becomes more complex, given that groundwater flow is not always unidirectional, the time-scales over which change occurs may be extended, and information on the ecological (and economic) relevance of the flows is often subject to a high degree of uncertainty. While an environmental flow in a river may be described by a flow and quality regime varying over time (weeks to decades), the water requirements of GDEs are likely to be specific to particular GDE types and locations. Water levels, pressures, flows, chemical fluxes, and quality will vary in importance for different GDEs (SKM 2001). For example, a subterranean cave ecosystem may be highly dependent on *fluxes* (nutrients, oxygen, detritus) as well as stable *temperature*. Groundwater-dependent (phreatophytic) vegetation is likely to be highly dependent on the prevailing water *level*. Fauna in a near-surface aquifer may need a *stable* water table. River base-flow ecosystems will depend on a *flow* regime above a certain minimum. The health of a hyporheic ecotone may depend on seasonal *flow reversal* of a certain magnitude. These complications, as well as their policy implications, are discussed in more detail by SKM (2001) and Howe *et al.* (2007).

The purpose of this paper is to highlight the importance of groundwater to a variety of different ecosystem types, to discuss the effects of groundwater overdraft (using international examples where Australian information is scarce or not available) and to briefly present policy implications and recommendations.

Ecological importance of groundwater

The dependence of an ecosystem on groundwater varies with the particular structure and function of that ecosystem, which in turn are likely to vary over time. Such requirements include the water essential for cell growth, as well as dissolved nutrients important for maintaining healthy tissue. Groundwater provides more than water and nutrients. A key role is supporting biodiversity and ecological processes through the organizing principle of hydrological connectivity, defined by Ward *et al.* (1999:129) as "the ease with which organisms, matter or energy traverse the ecotones between adjacent ecological units".

Australian ecosystems dependent on groundwater can be broadly categorized into three different types (Fig. 1). The first of these are subterranean ecosystems, such as aquifer and wet cave ecosystems. These are entirely dependent on groundwater and are often considered the ultimate groundwater-dependent ecosystems (Humphreys 2006b). Next are the surface aquatic ecosystems which depend on their connection to aquifers to maintain water supply. Examples of these include permanent springs, streams, rivers and flow-through lakes and other wetlands. Finally, there are terrestrial ecosystems where plant roots extend into the phreatic zone to extract groundwater.

The vadose zone is the unsaturated zone above the watertable in which the spaces between particles are only partially filled with water. Vadose water can comprise either percolating rainwater or discharging groundwater. In the arid zone, where rainfall is low but groundwater tables are close enough to the surface to support terrestrial vegetation (O'Grady *et al.* 2006a), biota using the vadose zone can be considered groundwater dependent because vadose zone moisture is discharging groundwater.

Subterranean aquatic ecosystems

Subterranean aquatic habitats consist of submerged caves, cave streams, wet passages and aquifers in karst, pseudokarst, calcrete and fractured rock, and the water-filled interstitial spaces between sediments in alluvial aquifers (Hancock *et al.* 2005, Tomlinson and Boulton 2008). Ecosystems exist within these habitats under a variety of salinities and alkalinities (Humphreys 2008a). Often, conditions are characterized by an absence of light, low nutrients, and restrictions in available space (Coineau 2000). The biotic component of these ecosystems consists largely of diverse com-

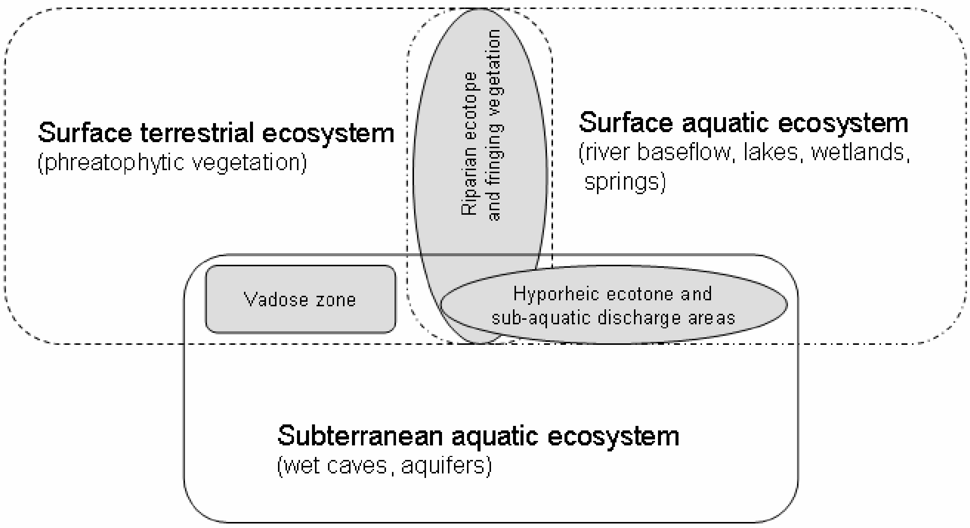

Fig. 1. A diagram showing subterranean, surface aquatic, and phreatophytic groundwater-dependent ecosystems with their ecotones. Redrawn after Eamus *et al.* (2006b).

munities of micro-organisms and invertebrate fauna — mainly crustaceans, gastropods and insects (Humphreys 2008a). Vertebrate fauna (fish and, outside Australia, salamanders) are also important in some localities. Most of the food webs in aquifers are carbon-based, although chemoautotrophic ecosystems are known (Movile and Frasassi caves, in Romania and Italy, Forti *et al.* 2002). In the absence of photosynthesis in aquifers, carbon must be sourced externally. Often, this is in the form of dissolved or particulate organic carbon, entering the aquifer in recharge water passing through the vadose zone and the beds of surface aquatic ecosystems, and its supply is a primary factor in determining biodiversity patterns (Datry *et al.* 2005).

Globally, it is only in the last decade that the magnitude of biodiversity present in subterranean waters has been recognized. It is apparent, despite the almost complete lack of study over much of the continent, that Australia contains a stygofauna of global significance (Humphreys 2008a). At a conservative estimate, at least 750 stygobitic species have been recorded from Australia, mostly in the last 10 years (Humphreys 2008a). This figure may be seen in context by comparing it with the 3410 described species in 13 higher taxa of freshwater stygobites that were enumerated from a 1986 global review (Humphreys 2008a). Much of these data come from extensive studies in the mineraliferous Pilbara and Yilgarn regions of Western Australia, where dewatering operations associated with mining is an issue of concern (more below) and where more than 500 Australian species occur. While these figures are now dated, Australia, which comprises ~7% of the Earth's land area, would appear to exceed the world average density (species per unit area) of stygal species (Humphreys 2008a) just based on this one small part of the continent.

These Western Australian calcrete aquifers, isolated for millennia, provide archipelagos of habitat "islands" containing distinctive and endemic fauna (Cooper *et al.* 2008). Identified stygobitic species from South Australia and Western Australia alone currently number 637, with an estimate of undiscovered species at over 2000 (R. Leijs, pers. comm. 22/3/10). Studies in other parts of Australia may reveal similar surprises in the future. Stygobitic animals are often efficient bio-accumulators, and slow to recover from reductions in their populations (Humphreys 2008b). By virtue of these biological characteristics and because many are small range endemics, the species inhabiting subterranean ecosystems are often considered intrinsically vulnerable to anthropogenic effects, despite evidence of exceptionally long persistence through geological eras in subterranean habitats subjected to massive geological and climatic change (Humphreys 2008a).

Declines in groundwater level usually increase the distance between the aquifer ecosystem and its source of carbon, reducing the amount of organic matter available to aquifer food webs. Water level decline may also have more immediate effects on faunal communities, causing stranding if decline occurs at a rate greater than that with which the animals are able to move downwards. This is of particular concern in those aquifers where there are limited vertical pathways to facilitate migration. In coarse alluvium stygofauna may be able to track water level decline, especially if the animals are small and the gaps between the particles are relatively large. However lenses of finer material may prevent this migration, and concurrent changes to water quality could also impose limitations (Dillon *et al.* 2009:28). When groundwater extraction bores are located too near to surface water bodies, the resultant influx of surface water may be detrimental to groundwater communities, and this can sometimes have effectively irreversible consequences for not only the ecological communities, but for human users as well. Surveys of hyporheic fauna suggest that pollutants (especially iron, chromium, and nickel) can enter the sediments further impacting on invertebrate communities (Plénet *et al.* 1996; Plénet and Gibert 1994).

Surface aquatic ecosystems

Groundwater dependent surface aquatic ecosystems consist of those surface water bodies that are hydrologically linked to aquifers, and whose functioning depends on this connectivity. Although water is the principal component of this connection, the transfer of dissolved and suspended materials, as well as organisms can also be critical to the functioning of some communities (Dent *et al.* 2000). Groundwater/surface water exchange can also play a critical part in moderating physico-chemical conditions of the surface water body. As rivers, lakes, and wetlands often occur at the lowest points in the landscape, they regularly intercept the water table of unconfined aquifers. Lakes forming in depressions below the water table are sometimes called "window lakes". In parts of Australia there are many flow-through lakes (those of the Swan Coastal Plain in Western Australia) where groundwater flows through highly permeable landforms.

With the exception of regulated rivers, the ecologies of most Australian rivers, most of the time, depend on their connections with groundwater (Boulton and Hancock 2006). In unregulated rivers, other than minor snowmelt exceptions, all river flow when runoff is not occurring (base flow) comes from aquifers or shallow groundwater. This generalization does not extend to the proportion of flow, as many rivers receive most of their annual flow from runoff during periods of high rainfall (as in monsoonal Australia) though they rely on groundwater base flow over the majority of the year. The ephemeral rivers of Australia's arid interior are an exception to the generalisation: with little baseflow, they are almost entirely driven by runoff from intermittent high-rainfall events (Ladson 2008:145).

The exchange of nutrients between terrestrial aquatic ecosystems and the underlying aquifers is sometimes bidirectional and changes through the year. Pressure gradients caused by differences in head potential drive the direction of hydrological exchange. Thus where the water table of an alluvial aquifer is higher than the water level of an adjacent stream, groundwater flows to the stream and *vice versa* (Gordon *et al.* 2004; Packman and Bencala 2000). Pumping from bores that are near rivers can create a unidirectional flow of river water into the stream bed with consequent loss of surface water volume (and thus flow). For hyporheic ecotones, both downwelling (where surface water enters the stream bed) and upwelling (where aquifer water exits the bed) areas are necessary to sustain the function of the ecosystem, for example by maintenance of bed filtration. Fish nursery areas may depend on the movement of water into, and then out of, the aquifer (Hancock 2002). However, localised pumping from near-stream bores can change exchange patterns to downwelling only, meaning that the filtration effect of water passing through the hyporheic zone and back into the stream is reduced or locally eliminated (Mauclaire and Gibert 1998).

In artesian aquifers, water under pressure can be pushed to the surface through fractures in the confining layer, and create seepages or flowing springs. The spring wetlands of the Great Artesian Basin are critical habitats for invertebrates (Ponder 2004), fishes (Wager and Unmack 2000) and plants (Fensham and Fairfax 2003), but these communities are threatened by both pressure loss caused by high numbers of free-flowing bores, direct habitat modification (usually excavation) and stock damage (Fensham *et al.* 2007). While the bores provide watering-points for cattle and wildlife (Noble *et al.* 1998) in many parts of the basin numerous artesian springs now have severely reduced flows or have become extinct because of pressure loss caused by bores drilled to support agriculture and, to a lesser extent, mining as well as urban use. A programme of capping free-flowing agricultural bores to control water and pressure loss was initiated some years ago, but many bores remain uncontrolled (Fensham *et al.* 2007; Molsher and Coote 2003; Fensham and Fairfax 2003). More information on the spring ecosystems and the government responses to their decline are outlined below.

Dependent ecosystems also occur where groundwater flows into estuaries or onto the open coast. Diffuse discharges to aeolian landforms will create terrestrial wetlands, and submarine discharge may deliver significant nutrients to marine ecosystems (Rutkowski *et al.* 1999; Johannes 1980). Groundwater discharge may influence the diversity and density of seagrass beds, although this relationship is not straightforward or well understood (Kamermans *et al.* 2002; Rutkowski *et al.* 1999). The level and salinity of groundwater at turtle nesting beaches may be important, even critical, to nesting success: however there is little scientific agreement on the subject (Johannes and Rimmer 1984; Foley *et al.* 2006).

Phreatophytic ecosystems

Particularly in arid areas, many land-based vegetation communities depend on groundwater. These in turn support terrestrial faunal communities, and these two components make up groundwater-dependent terrestrial ecosystems. Many plants, mostly trees and large shrubs, have roots that penetrate the water table and use groundwater, in some instances from up to ~70 m below the ground surface (Lubczynski 2007; Canadell *et al.* 1996,). In soil, subsoil, or permeable material immediately above the watertable (the vadose zone) moisture moves upwards through the hydraulic lift created by capillary action. This water is utilized by plants, provided their roots can reach this zone.

The critical issue for phreatophytic vegetation is the depth of the water table (and its associated vadose zone) below the ground surface, and its accessibility by roots. If abstraction lowers the water table beyond the depth from which roots can obtain water, those elements of the vegetation community with full dependence on groundwater will die. This occurred in the Banksia woodland community of the Swan Coastal Plain north of Perth, Western Australia, after the underlying water table was lowered by 2.2 m (Groom *et al.* 2000a). However, some *Banksia* species (*B. attenuata*, *B. menziesii*), while able to use groundwater to a maximum depth of 9 m, switched over to soil moisture when the water table retreated beyond that depth (Groom 2004; Zencich *et al.* 2002) suggesting that they have a seasonal dependence on groundwater, using it during winter when rainfall recharges the aquifer, and then relying on soil moisture at other times of the year. Species that are restricted to low-lying, shallow-groundwater depth habitats (*Banksia littoralis*) are entirely dependent on groundwater all year round, and can be classified as obligate phreatophytes (Canham *et al.* 2009).

Most riparian plant communities use groundwater to degrees which vary spatially and temporally (Baird *et al.* 2005; Naiman *et al.* 2005). Along the Daly River in the Northern Territory, trees nearer the river used more groundwater than those further away and higher above the water (O'Grady *et al.* 2006b). In arid areas, the growth of riparian vegetation is influenced by depth to the water table, which fluctuates seasonally (Martí *et al.* 2000). Groundwater beneath riparian zones is often enriched in some nutrients compared to the stream water (Martí *et al.* 2000) and sediment microbial activity together with riparian adsorption potentially intercept nutrient-rich groundwater and moderate the speed with which nutrients enter the river (Trémoliéres *et al.* 1997). In artesian areas, localized groundwater is able to support shallow-rooted vegetation communities where pressure brings the water to the surface at seepages or springs. Nine native plant species are endemic to the spring environments of the Great Artesian Basin (Fensham *et al.* 2007) with most of them threatened by loss of aquifer pressure, grazing and trampling, excavation, and exotic grasses (Fensham and Price 2004).

For all phreatophytic ecosystems, there is a risk of isolation from the aquifer if significant groundwater extraction occurs. In artesian areas, this isolation could result from loss of aquifer pressure, while for unconfined aquifers it can simply be a result of the water table being lowered to a point where it is no longer in reach of the ecosystem. The wetland and lakes overlying the Gnangara Groundwater Mound in south Western Australia are suffering the results of water level decline due to groundwater abstraction combined with the expansion of pine plantations, burning of native forests, and naturally warm dry weather (Department of Water 2007; Groom *et al.* 2000a; Groom *et al.* 2000b). Climate change may also be playing a role.

Groundwater managers need to be aware that groundwater levels become depressed or elevated in the region surrounding discharge or recharge bores respectively. The magnitude of the change in elevation of the groundwater surface reflects the balance between the pumping rate at the bore and the hydraulic conductivity of the aquifer.

In bore fields the groundwater surface may become dimpled, both spatially and temporally, by drawdown cones of adjacent wells. In coastal areas, a serious issue is that of seawater intrusion. In some locations fresh water can extend for tens of kilometres offshore (Post 2005). The extraction of coastal groundwater at a rate which reverses the seaward hydraulic gradient will threaten not only the subterranean aquatic ecosystems (Hancock 2004), but also groundwater-dependent coastal terrestrial and aquatic vegetation, and reduce the value of the resource for human use. The same effect may occur wherever saline and fresh waters occupy the same aquifer, with overdraft of freshwater sometimes causing similar damage (Humphreys 2008b).

Any interface of freshwater with underlying sea (or saline) water will be dimpled in mirror image to that of the groundwater surface (Humphreys 2002) but amplified about fortyfold owing to the Ghyben–Herzberg effect (Ford and Williams 1989) which reflects the density differences between the fresh and saline waters. Consequently, minor drawdown of the water surface may result in a major intrusion of sea water into an aquifer where it may damage the aquifer and impinge on stygofauna as well as on overlying or downstream GDEs. Thus, even where overdraft is not an issue, the pumping rate itself may have major impacts on GDEs. This issue is described further by Humphreys (2006b) and Humphreys (2009).

Groundwater overdraft case studies

This section briefly discusses some examples of the effects of groundwater overdraft, sometimes combined with other factors, on both surface and subterranean ecosystems. International examples are used in the absence of Australian case studies.

In Australia, utilization of groundwater for human consumption has increased exponentially in the past 150 years (Eamus *et al.* 2006a). Extraction of groundwater to support irrigation, mining, and drinking water rose by 90% between 1985 and 1997 to c. 5000 GL per year (NLWRA 2001). In fact, in many regions of Australia, the extraction of groundwater now exceeds recharge to the point where natural groundwater regimes have been dramatically altered (NLWRA 2001). Considering that (a) such levels of extraction alter the groundwater regimes of groundwater-dependent ecosystems (GDEs) that have evolved over millennia and (b) there is evidence that such alterations have deleterious impacts on the structure and function of a range of GDEs, there are clear imperatives to understand the impacts of groundwater extraction on GDEs and their ecosystem services (Boulton *et al.* 2008; Tomlinson and Boulton 2008; Eamus *et al.* 2005) in order to provide sound management of valuable groundwater resources and their associated biodiversity (Howe and Pritchard 2007; Murray *et al.* 2006).

Phreatophytic and riparian vegetation

Groundwater overdraft has a range of environmental impacts including land subsidence, seawater intrusion into freshwater aquifers, and degradation of natural plant communities (Murray *et al.* 2006; Naumburg *et al.* 2005; Zektser *et al.* 2005). Groundwater overdraft typically produces falling groundwater levels, which in turn are associated with changes in vegetation structure and composition in many different terrestrial and riparian plant communities around the world, including alkali meadows (Elmore *et al.* 2006; Elmore *et al.* 2003), semi-arid woodlands (Groom *et al.* 2000a; Groom *et al.* 2000b), shrublands (Muñoz-Reinoso 2001) and woody forests (Xu *et al.* 2007; Chen *et al.* 2006; Webb and Leake 2006; Lite and Stromberg 2005; Liu *et al.* 2005; Liu and Chen 2002; Horton *et al.* 2001a; Horton *et al.* 2001b; Shafroth *et al.* 2000; Scott *et al.* 1999). Overwhelmingly, these studies indicate that groundwater declines, as a result of over-extraction, lead to serious degradation of natural ecosystems.

For alkali meadows in Owens Valley (California) Elmore *et al.* (2006) found that vegetation cover was negatively correlated with increasing depth to groundwater. When groundwater declined below natural fluctuation levels, the first observable response was a reduction in total cover of shallow-rooted plant species. At locations where there was a high cover of such plants, the decrease in total vegetation cover was more rapid than at locations where there was higher cover of deeper-rooted species. Of concern is that declines in cover of native plant species, in response to groundwater decline, are often followed by an increase in the cover of exotic plant species (Elmore *et al.* 2003). If groundwater-dependent plant communities are altered by the invasion of non-phreatophytic species then the ecosystem may change to an alternative state (Elmore *et al.* 2006). These changes can ultimately lead to degradation of ecosystem services and costly procedures to restore the former condition (van de Koppel *et al.* 2002; Scheffer *et al.* 2001).

In the lower reaches of the Tarim River (China), damming (altering recharge) and direct exploitation of water resources for farmlands have led to dramatic declines in groundwater levels in the last 30 years (Chen *et al.* 2006). As a consequence, plant species diversity has decreased and the overall structure of the plant communities has become much simpler (Chen *et al.* 2006), demonstrating the impacts of both indirect and direct effects of water extraction on groundwater-dependent ecosystems. There appears to be a pattern of "converse succession" (*sensu* Liu and Chen 2002) of plant communities with a progression starting with the disappearance of herbaceous taxa, followed by shrubs and then trees (Liu *et al.* 2005). A simple plant community is left that is characterized by the dominance of drought-tolerant species (Liu *et al.* 2005).

Recent studies in the south-western regions of Australia have explored the effects of declining groundwater levels on the native sclerophyllous vegetation of the Swan Coastal Plain (Froend and Drake 2006; Zencich *et al.* 2002; Groom *et al.* 2000a; Groom *et al.* 2000b). This coastal plain overlies several shallow superficial aquifers, the Gnangara groundwater mound being the largest. The Gnangara Mound supplies a large proportion of the water supply for Perth, the largest city in Western Australia. Comparison between *Banksia* woodland vegetation near a groundwater extraction bore before and after abstraction revealed that a major impact of lowered groundwater levels was extensive death and stress of the three *Banksia* overstorey species (Groom *et al.* 2000a). Further comparisons between the abstraction site and a site not influenced by groundwater extraction revealed that lowered groundwater levels, combined with below average annual rainfall, led to a loss of between 20% and 80% of adults of overstorey species and up to 64% of adults of understorey species at the abstraction site (Groom *et al.* 2000a). Groom *et al.* (2000b) examined the response of 10 myrtaceous shrub species to declining groundwater levels in the Swan Coastal Plain. The most clear-cut biological pattern to emerge was that species that displayed the greatest reductions in population size, in relation to lowered groundwater levels, were comparatively shallow-rooted species (rooting depth < 1 m). Further work has suggested that obligate phreatophytes, such as *Banksia ilicifolia*, are likely to be the most susceptible to groundwater decline, and could be used as possible indicators of artificial groundwater drawdown impacts in this area (Zencich *et al.* 2002).

Groundwater fauna may survive through long periods of geological time relative to surface faunas (Longley 1986; Humphreys 2000b) as they can be buffered from major changes in climate. For example, stygobitic animals in Iceland survived beneath the continental ice sheet during the Pleistocene in water kept liquid by geothermal heat (Bjarni *et al.* 2007). However, human-induced threats to these unique ecosystems threaten their rapid demise though overdraft, pollution or flooding.

There are relatively few scientific studies concerning the impacts of water abstraction on ecosystems within aquifers (Rouch *et al.* 1993), especially deep aquifers (Longley 1992). The diverse groundwater fauna of the Edwards Aquifer, Texas, which includes both vertebrate and invertebrate stygobites with both marine and freshwater affinities, is under threat from over-extraction which removes almost all of the natural recharge. The adverse effects are caused by loss of spring flow, dewatering of parts of the karst, and saltwater intrusion (Longley 1992). The Texas blind salamander, *Typhlomolge rathbuni*, inhabits the artesian part of the aquifer and is threatened by over-pumping (Elliot 2000). The use of Valdina Farms Sinkhole, Texas, as a recharge well for the Edwards Underground Water District appears to have extirpated the only known population of the salamander *Eurycea troglodytes* (Elliot 2000).

Rouch *et al.* (1993) conducted a high discharge pumping test in a sinkhole to investigate its effect on the movement of stygofauna out of the saturated zone of the Baget Karst (Ariège, France). The water in the sinkhole was lowered by 21 m on three occasions over four days, and as a result the microcrustacean migration (mainly harpacticoid copepods) from the karst increased. The community had not returned to its original state when surveyed one year later. Pollution can result from increased energy input which may permit the successful invasion of surface forms into a previously oligotrophic environment (Notenboom *et al.* 1994; Malard *et al.* 1994) and fauna may be lost rapidly following contamination (Iliffe *et al* 1984). Elliot (2000) documents many areas in the US and Yucatan, Mexico, where groundwater contamination, especially sewage from humans and stock, has been reported to have, or was expected to have an adverse affect on groundwater fauna, either by loss, displacement or changing composition.

Knowledge of the stygofauna in Australia is recent and still very incomplete (Tomlinson and Boulton 2008). Despite extensive areas of drawdown (Froend *et al.* 2004), changed hydrology and salinization (Leaney *et al.* 2003; Sattler and Creighton 2002:Fig. 4.2; Gerritse 1998) and contamination (Carroll and Goonetilleke 2004; Barber *et al.* 1998) of groundwater, the lack of prior data precludes evidence of change to stygofauna communities (Humphreys 2008b; Humphreys 2006a) and from most areas there are no data at all. There are areas in the Pilbara and Kimberley where the water table had been lowered to levels below the karst but where no stygofauna were reported despite conditions being otherwise suitable. However, the recognition that changing groundwater conditions may threaten groundwater ecology is now evident in Australia, both in research (Humphreys 2008b; Humphreys 2006b; Hancock *et al.* 2005; Boulton *et al.* 2003; Humphreys 1999) and management/regulatory fields (EPA 2007; EPA 2003; Playford 2001).

The best documented cases of Australian stygofauna being threatened by changed groundwater conditions refer to cave streams. Exit Cave in south-east Tasmania contains endangered hydrobiid snail populations, which became further restricted because of increased siltation from upstream quarrying activities. Rehabilitation of the quarry resulted in a decreased silt load and populations subsequently increased (Hamilton-Smith and Eberhard 2000; Eberhard 1999; Eberhard 1995). Illustrating the diversity of snail fauna, the Precipitous Bluff cave ecosystem in south-west Tasmania contains eight species of hydrobiid snails and about 30 others from other Tasmanian caves remain unnamed (Ponder *et al.* 2005).

A second example relates to threatened communities (listed under Australia's *Environment Protection and Biodiversity Conservation Act 1999)* associated with root mats in phreatic pools, and root mat communities in cave streams in Western Australia. At Yanchep, north of Perth, drawdown of a large freshwater mound for city water supply has reduced water flow through cave streams and stranded root mat communities that are now maintained by artificial flow (Jasinska and Knott 2000; Jasinska *et al.* 1996). Similar root mat communities in phreatic waters of the southern Augusta-Margaret River region, also in syngenetic karst (Grimes 2006), are threatened by a long-term fall in water level, the cause of which is unclear. The working hypothesis is that a changed fire regime with less frequent fires (from average 2.4 fires/decade to < 0.5 fires/decade) in the overlying karri forest (*Eucalyptus diversicolor*) have increased scrub cover and reduced recharge (Eberhard 2004) leading to the long-term decline in water level which is threatening groundwater fauna. The region has also experienced a drier climate over the last 2 decades (compared to century-long statistics) which complicates arguments on cause and effect. While overdraft may not be the main problem in this example, it does illustrate the type of effects which may be expected to occur in areas affected by overdraft.

A third example relates to the extensive (2000 km^2) shallow aquifer of the Western Fortescue Plain, Pilbara WA. The aquifer's drainage was captured by the Fortescue River in the late Pleistocene or Holocene, and in consequence the water table within the Millstream Dolomite fell from about 310 m to c. 294 m AHD (Humphreys 1999; Barnett and Commander 1985). The aquifer discharges near Millstream and is recharged there by the Fortescue River during episodic floods typical of this arid region. This 16 m reduction in level stranded previously phreatic cavities (Barnett 1980) but was insufficient to entirely eliminate the rich groundwater fauna of the area (Poore and Humphreys 1998). However, many species are likely to have become more restricted in their distribution, and are now present mostly in the region of interaction with the river (Humphreys 2001). Prior to these discoveries of stygofauna, abstraction from a bore field to supply the mining towns of the Pilbara led to reduced discharge at Millstream, which threatened groundwater-dependent vegetation and aquatic communities. As a result, infrastructure was established to allow artificial recharge at these locations.

Australia's Great Artesian Basin

This sedimentary basin of aquifers and aquitards (impervious layers) underlies approximately one-fifth of Australia (1.76 million km^2) and extends beneath arid and semi-arid regions of Queensland, northern New South Wales, north-eastern South Australia and the eastern areas of the Northern Territory (Habremehl 2006; Fensham 2006; Habermehl 1980) (see Fig. 2). Where water from the basin reaches the surface, water temperatures vary between 30-100^{0}C. Areas of recharge occur mainly along the eastern margins of the Basin, with natural discharge occurring in the south and west (Department of Natural Resources and Mines Queensland 2009). The oldest artesian waters (aged at nearly 2 million years) occur in the southwest of the Basin (Habermehl 1996).

The Great Artesian Basin (GAB) has an estimated water storage capacity of 65,000 million megalitres. This groundwater is the only reliable source of water in parts of the GAB, and water has been mined since the late 1800s. Initially, development of artesian bores was unregulated, and remained so for many decades. While GAB water continues to underpin considerable agricultural production (particularly through supply of stock water) (Mues and Hardcastle 1998) pastoralists have had a history of profligate use, with about 3100 artesian bores (Habermehl 2006) allowed to run freely into open drains.

Prior to European settlement, natural discharge occurred as springs or seepages, which are commonly referred to as mound springs, mud springs, boggomoss springs, spring pools, or groundwater seeps (Fensham 2006; Habermehl *et al.* 2006; Fensham and Fairfax 2003; Ponder 1986). Some springs form a distinctive mound, formed from the sediments and salts that are deposited by spring water as it evaporates. These "mound springs" can reach as much as 10 metres in height but are usually much lower, and some have been shown to be 10–15,000 years old (Habermehl 2006) and, as attested by the many endemic species they contain (see below), provide long-term stable habitats. The discharge rates of free-flowing artesian springs (as well as bores) have declined considerably since the early 1900s, causing the extinction of many springs (Fensham *et al.* 2007; Fensham and Price 2004; Fairfax and Fensham 2003; Fairfax and Fensham 2002; Ponder 1986). For example, only 36% of the 300 spring groups present in the 1800s are still active in western Queensland (Fensham and Fairfax 2003) with the resultant loss of their associated communities, and in the NSW part of the basin only one very small group of what was initially two supergroups comprising more than 20 spring complexes remains in a semi-viable state (Pickard 1992).

An estimated two-thirds of this extraction is currently wasted, mostly to evaporation and seepage (Noble *et al.* 1998). The Queensland

State government was the first to introduce regulations regarding capping and piping new bores at the landowner's expense in the 1950s. The South Australian government later paid to have this done. A combined Commonwealth-State Great Artesian Basin Sustainability Initiative (GABSI) began in 1999 with a programme of bore capping and piping with funding extended in 2004 to 2009 (Government of Australia 2009a). Progress has been slow, a reflection of funding levels.

The protection of springs and their flows is now written into the Water Resource (Great Artesian Basin) Plan 2006 and the Great Artesian Resource Operations Plan (Department of Natural Resources 2007) (ROP). The ROP requires monitoring (by relevant State agencies) of artesian water pressure, subartesian water levels and the flow of water to springs (as well as the baseflow to water courses). It also requires the monitoring of water use. The plan also requires that, for the first time, a register of the

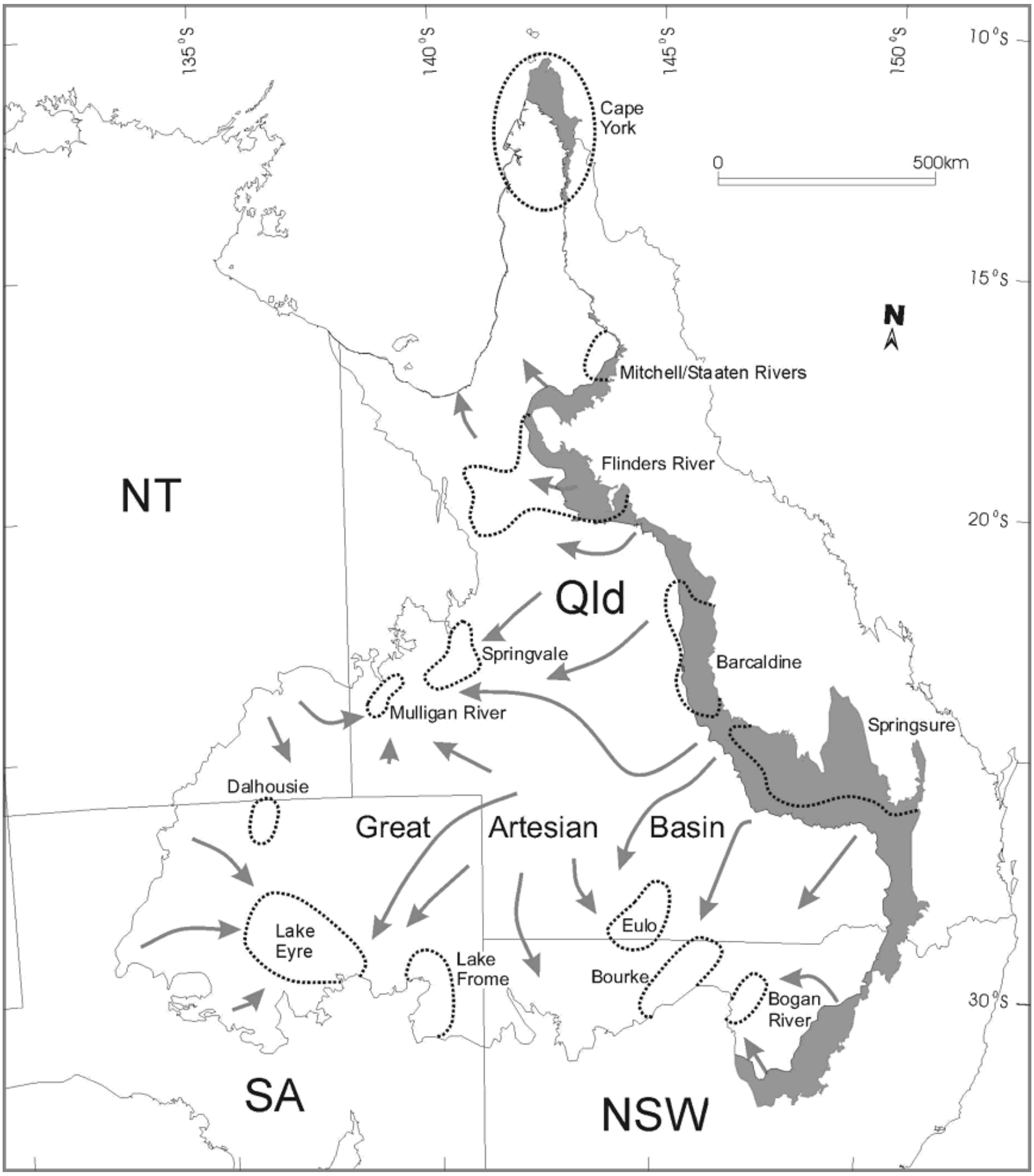

Fig. 2. The Great Artesian Basin. Shaded areas denote recharge, arrows denote modelled flowlines, and dotted lines identify areas of past spring occurrence.

springs "that support significant cultural and environmental values" be maintained, and the establishment of a spring monitoring network. Activities that involve water extraction from or near springs are also tightly controlled under the ROP. The plan also requires the monitoring and assessment of artesian water levels and pressure by monitoring a large number of artesian and subartesian bores, but does not address filling knowledge gaps in aquifer structure that may be relevant to management outcomes. The plan has had some successful outcomes, particularly in a much increased knowledge of the location and status of springs (Department of Environment and Resource Management 2010; Fensham and Fairfax 2009). It is to be hoped that it will be fully implemented.

Prior to bore developments, the artesian springs were critical water sources for humans, stock and native animals (Harris 1981). As mentioned above, GAB springs are home to a diverse array of unique and unusual plants and animals, including a number of rare species of invertebrates, plants and fishes, some of which are endemic to a single group of springs (Fensham *et al.* 2007; Fensham 2006; Wilson 1995; Ponder 1986). Detailed studies of endemic snails (Ponder and Clark 1990; Ponder *et al.*1989), including some genetic studies (Perez *et al.* 2005; Ponder *et al.* 1995) and of crustaceans (Murphy *et al.* 2009; Wilson and Keable 2004) provide evidence of high levels of endemism, some of which is cryptic, in aquatic animals in GAB springs. While spring endemic plants are relatively few (see above) recent studies on *Eriocaulon* have shown that their diversity is greater than previously suspected (Davies *et al.* 2007).

In Queensland, only 44 per cent of the original 171 discharge spring complexes currently have at least some active springs. and overall about 20 per cent are severely damaged as a result of excavation or other modification (Fensham 2006). The biological community dependent on the discharge springs were listed as endangered under the Commonwealth's *Environment Protection and Biodiversity Conservation Act* in 2001 (Government of Australia 2009b) and there is now a draft recovery plan in place (Fensham *et al.* 2007). How effective this will be remains to be seen.

Mining, water quality, and groundwater management

Groundwater is not only extracted for direct use. Where mineral exploitation occurs below the water table, groundwater needs to be removed so that mining can proceed. An example of this is the regional dewatering of the dolomite karst overlying the goldfields in Gauteng, South Africa (Buttrick 2005). In Australia, extensive water extraction from deep palaeochannel aquifers on the Western Shield is a potential threat to groundwater biodiversity, as the palaeochannel aquifers are connected to shallow calcrete aquifers that contain unique and diverse communities of stygofauna (Humphreys 2008a; Humphreys 2006b).

Profligate deep water mining from palaeochannel aquifers, with a residence time in the order of 10^5 years (Johnson *et al.* 1999), is widespread, and increasingly common in the mineraliferous regions of arid Australia. The potential for connectivity between deep palaeochannel aquifers and the overlying unconfined calcrete aquifers, which support diverse and locally endemic stygal communities (Humphreys 2008a) and other groundwater-dependent ecosystems, has been confirmed, although the manner of the connection is not well understood (S. Johnson, Water and Rivers Commission, pers. comm., 3 May, 2000; R. Martin, Anaconda Nickel Ltd., pers. comm. 21 August 2000).

While drawdown of groundwater affects surface GDEs, contamination of groundwater with nutrients or toxins will directly impact groundwater fauna, and will persist until it is remediated or naturally dissipates (which may be a very slow process). It has been suggested that the restoration of groundwater ecology should be the ultimate measure of successful groundwater remediation (Humphreys 2000a).

Pumping should be considered excessive if it results in a significant deterioration in groundwater quality, irrespective of whether or not there is considerable drawdown (Margat 1994). Whether this is classified as overexploitation depends on the criteria established, but adverse effects on groundwater fauna would typically not be amongst the criteria in common use by managers, in Australia and elsewhere. "In practice, overexploitation diagnoses are made *a posteriori* on the basis of observed symptoms of prolonged imbalance — continued drawdown, possible effects on boundary flows, and water quality" (Margat 1994:515).

Groundwater-dependent ecosystems have adapted to natural variation in four main attributes of groundwater over thousands or millions of years (Hatton and Evans 1998). These attributes are groundwater level, pressure, flux and quality. Groundwater extraction for mining will affect a combination of these attributes, with resulting impacts depending on the particular ecosystems involved. More specifically, the effects will be determined by whether the ecosystem is a cave/aquifer ecosystem (stygofaunal assemblages within groundwater), an ecosystem dependent on the surface expression of groundwater (wetlands and mound springs), or an ecosystem dependent on the subsurface expression of groundwater (terrestrial ecosystems such as river red gum forests) (Eamus *et al.* 2006a).

It is thus vital that the effects of groundwater extraction for the purposes of mining are managed accordingly. That is, an understanding of the effects of extraction on these groundwater attributes is essential for management of the various ecological effects on the groundwater-dependent ecosystems. Of particular importance in the mining context is the effect of groundwater extraction on groundwater quality. Deterioration of groundwater quality has the potential to reduce stygofaunal biodiversity (Eamus *et al.* 2005). As a consequence, the ecosystem services provided by groundwater ecosystems, such as water and nutrient regulation, are likely to decline (Tomlinson and Boulton 2008; Murray *et al.* 2006).

Two key issues that need to be considered are the parameters of groundwater that must be monitored to detect changes in groundwater quality, and the levels of acceptable change in these parameters to ensure ecosystem health (for biodiversity) and the continued provision of ecosystem services (for humans). Thus, a critical component that needs to be taken into account is the fact that invertebrate species and assemblages of groundwater systems are unique and display a very different suite of life-history strategies to invertebrates of surface waters (Gibert and Deharveng 2002; Danielopol *et al.* 2000; Danielopol *et al.* 1994). Simply basing groundwater quality guidelines on guidelines aimed at surface water species and assemblages is inadequate (Eberhard and Hamilton-Smith 2000). Furthermore, it should be stressed here that groundwater quality must not be managed just using "acceptable" values that have been determined independently of stygofaunal species (Hahn 2006). Indeed, it is advisable to go beyond an approach based on the response of one or two "indicator" species (a common flaw of many surface-based ecotoxicological investigations) to one based on groundwater ecosystems (the assemblages of species).

The precautionary principle in groundwater policy and management

It is usually the case that groundwaters are accessed for human consumption through many relatively small, poorly controlled and poorly monitored wells and bores, and the users seldom (if ever) pay the external costs of the water they extract. In these circumstances the environmental costs caused by a single user will fall on many, thus encouraging over-exploitation of the resource — the tragedy of the commons (Sophocleous 2010; Hardin 1968). Moreover, groundwater managers know that the damage caused by adding just one more user will not be discernable for several years, if at all, creating a situation where incremental increases in permitted water use can easily undermine well-intended strategic conservation rules — the tyranny of small decisions (Odum 1982).

Referring to natural resources generally, these two mechanisms have contributed to (if not driven) a long history of resource over-use and degradation (Ludwig *et al.* 1993). Australia's Murray-Darling Basin provides an Australian example (Nevill 2009). In this context, comprehensive application of the precautionary principle appears essential in order to protect the long term values of natural resources — including water and associated aquatic ecosystems.

At the level of policy, the precautionary principle has been widely accepted by governments around the world, at least on paper. Australia is no exception. The purpose of this section is to discuss the implications of the precautionary principle for groundwater management. While this section does refer briefly to Australian groundwater policy, it is not a review. A full discussion of national and State groundwater policies, and the implementation of those policies (or lack thereof) deserves discussion in a separate paper. The focus of this section is the general consequences of a precautionary approach in groundwater policy formulation. We list some of the most important consequences in "Recommendations" below.

In regard to natural resource management, the Australian (Commonwealth) Government and all State and Territory Governments espoused a strong commitment to the precautionary principle in the early 1990s (through, for example, the National Strategy for Ecologically Sustainable Development (Government of Australia 1992)). According to the precautionary principle, *where there is the possibility of significant harm, lack of scientific certainty should not prevent prudent action to avoid or mitigate such harm.* For the principle to apply, two factors must be present: uncertainty regarding the outcome of an action, and the possibility of the action resulting in harm.

Given this strong commitment by governments, which includes statutory requirements in some cases (eg: the Commonwealth *Fisheries Management Act 1991*) the wide application of precaution both in policy and programme might be expected, at least by optimists (Stein 1999). The most comprehensive consideration of the principle in Australian law is that of Justice Preston (Preston 2006) summarized by Mohr (2006). Preston made a number of important points, including:

- application of the principle results in a reversal of the onus of proof, requiring the proposer of an action to demonstrate in advance the safety of the proposed action; and
- the greater the uncertainty, or the greater the possible harm, then the greater should be the caution applied in decision-making.

These two points are particularly relevant to groundwater management (below).

Possibility of harm

Certain problems have beset the use of groundwater around the world. Just as rivers have been over-used and polluted, so too have aquifers. The big differences between rivers and aquifers are that aquifers are out of sight (and often out of mind), and the effects of groundwater extraction (and pollution) are often invisible for extended periods (below). As discussed above, groundwater drawdown from over-allocated aquifers has the potential to cause severe damage to both terrestrial and aquatic ecosystems (see also Clifton and Evans 2001; Hatton and Evans 1998) — in some cases very conspicuously but in others quite imperceptibly due to the extended period over which the damage occurs (Sophocleous 2002).

Apart from the ecosystem effects discussed above, four general hazards should be noted. First, flood mitigation schemes, intended to protect infrastructure built on floodplains, have in some cases had the unintended consequence of reducing aquifer recharge by restricting the extent of natural flooding. Australia is a generally flat continent, with extensive floodplains associated with the Murray-Darling Basin and Cooper/Lake Eyre Basin, for example. Such floodplains contain palaeochannel and other aquifers which recharge from floodplain flooding. Second, prolonged depletion of groundwater in extensive aquifers can result in land subsidence, with associated infrastructure damage — as well as (third) saline intrusion (Zektser *et al.* 2005). Fourth, draining acid sulphate soils, often found in low-lying coastal plains, can result in acidification and pollution of freshwater and estuarine streams (Sommer and Horwitz 2001). Damage to ecosystems can be effectively irreversible.

To illustrate the magnitude of potential effects, groundwater extractions in the lower Murrumbidgee Valley (central Murray Darling Basin) increased by around 50% over the two years to mid-2003 because drought reduced the availability of surface water. While this groundwater system provided irrigators with a significant buffer against reduced surface water availability, this increase in use led to a 10–20 metre drop in hydraulic head in most parts of the deeper aquifer (Earth Tech Engineering 2003, quoted in Goesch *et al.* 2007:9). A study of the Dumaresq River Aquifer by the Queensland Government (Chen 2003, quoted in Hafi *et al.* 2006:11) indicated "the temporary sale of surface water at $100 a megalitre is estimated to result [through surface water/groundwater substitution] in additional aquifer drawdown . . . leading to the groundwater table falling a further 34 metres." Movement of the groundwater table on scales considerably smaller than these drops has the potential to cause the death of terrestrial vegetation communities over considerable areas, especially where climate change (through reduced rainfall) is placing vegetation under stress. Similarly, such changes can not only cut off natural groundwater flows to rivers, but reverse them, draining water away for river ecosystems already in stress.

At best, such changes place groundwater-dependent ecosystems under some physiological stress; at worst, they can result in irreversible loss of significant species and/or ecological communities (Fensham *et al.* 2007; Zektser *et al.* 2005; Danielopol *et al.* 2003; Pringle 2001).

Uncertainty

Groundwaters move below the landscape largely unseen. Uncertainties in predicting the response of aquifers to drawdown stem from several factors:

- details on the size, position and permeability of aquifers and aquitards are often poorly understood: assumptions or projections must be made on the basis of limited information from bores and other geological surveys;
- information on water flows and quality may similarly be scarce — in particular the direction, timing and volume of interchange between groundwaters and surface waters is usually poorly quantified; and
- the dynamic response of aquifers to extraction, due in large part to their size, occurs over extended periods of time.

This last point is extremely important. The long time lags inherent in the dynamic response of groundwater to development have often been ignored or sidelined by water management agencies, long after scientific understanding of the issue was consolidated. In brief, the effects of groundwater overdraft (although undeniably real) may take decades or centuries to manifest themselves. In a classic study in 1982, Bredehoeft and colleagues (discussed in Sophocleous 2002) modelled a situation where groundwater extraction in a intermontane basin withdrew the entire annual recharge, leaving nothing for natural groundwater-dependent vegetation communities. Even when the borefield was situated relatively close to the vegetation, 30% of the original vegetation demand could still be met by the lag inherent in the system after 100 years. By year 500 this had reduced to 0%, signalling death of the groundwater-dependent vegetation. The science has been available to make these calculations for decades; however water management agencies have often ignored effects which appear outside the rough timeframe of political elections (3 to 5 years). Sophocleous (2002) argued strongly that management agencies must define and use appropriate timeframes in groundwater planning. This is essential, and will mean calculating groundwater withdrawal permits based on predicted effects decades, sometimes centuries in the future.

Monitoring such effects against a background of natural variability presents significant challenges (especially given changing priorities and budgets within water management agencies). Clearly commitments must be made to long-term monitoring programmes, within a management framework which relies heavily on mathematic models to predict the effects of water extraction. Over time monitoring results must feed back into calibrating and validating the models used on a day-to-day basis for licensing purposes.

The other critical factor increasing uncertainty is the lack of accurate metering of groundwater use. Goesch *et al.* (2007:1) estimated that 60% to 80% of major users (across Australia) have not been required to meter their usage. Unmetered flows make accurate catchment/aquifer water planning impossible and foster a culture where compliance with licence conditions is seen as unimportant. It should also be noted that present assessments of groundwater sustainable yield are substantially inaccurate: Evans *et al.* (2003) suggested regular inaccuracies of around 25%, and this estimate may itself err on the conservative side, given the complications in making such estimations (IAHA 2004). It should be noted that the approaches referred to by Evans and IAHA have yet to account in any accurate way for the needs of groundwater-dependent ecosystems.

Precautionary groundwater policies: the CoAG and NWI reform frameworks

In February 1994 the Council of Australian Governments (CoAG) adopted "a strategic framework for the reform of the Australian water industry" — to become known as the Water Reform Framework. The Framework had two central elements: *economic reform* to increase competition and efficiency within the industry, and *environmental reform* to increase emphasis on sustainable use of natural resources, and protection of environmental (especially biodiversity) values.

The Framework was to evolve over the following decade. In 1996 CoAG agreed to specific additions, including agreements focused on groundwater:

- to integrate groundwater and surface water resource management;
- to develop a nationally consistent definition and approach to calculating sustainable groundwater yield;
- to prepare groundwater management plans, policies and strategies;
- to base groundwater allocations on groundwater management plans;
- to ensure that such plans included environmental water provisions in accordance with agreed principles; and
- to address and retrieve over-allocation issues on a plan-by-plan basis.

These policies, particularly the first (often referred to as "conjunctive water management") support a precautionary approach to groundwater management. The Framework included an important policy document which set out principles for the provision of environmental flows (ARMCANZ 1996). Although the language used in this document focused largely on surface water, SKM (2001) showed that, with slight changes, the principles could be re-written to include groundwater. ARMCANZ (1996) took a precautionary approach to environmental flows, emphasizing that flows to support ecosystems should take precedence over irrigation demands (although not over domestic and stock watering).

CoAG's commitment to integrated groundwater and surface water management was re-stated again in the development of the National Water Initiative (NWI) 2004. The *InterGovernmental Agreement on a National Water Initiative*, signed by the Commonwealth and participating States on 25 June 2004, listed "recognition of the connectivity between surface and groundwater resources, and connected systems managed as a single resource" as one of ten core NWI objectives. While the general thrust of the NWI supported precautionary management, *limiting* the application of conjunctive management to situations where "close interaction between groundwater . . . and streamflow exists" is *not* a precautionary approach, given generally high levels of uncertainty regarding groundwater/surface-water interaction. A precautionary approach should *assume* full connectivity until proven otherwise.

Policy recommendations from groundwater professionals

While many of the policies set out above clearly lie in a precautionary framework, implementation of these policies within government programmes has generally been weak and slow, with some key elements entirely without implementation in some Australian States after the passage of ten years (Nevill 2009; Nevill 2008).

In 2006 a group of Australian hydrogeologists released a short statement titled *National Groundwater Reform* (Evans *et al.* 2006). Their call for better groundwater management reiterated many of the policy initiatives dormant over the previous decade. Moving past policy, the group's statement of concern drew attention to funding shortfalls related to water infrastructure and data collection, technical and public education, and compliance programmes. The group's main points were:

- Planning must identify sustainable levels of groundwater extraction and Governments must return over-allocated systems to sustainable levels;
- All groundwater use, except low-yielding domestic or stock bores, must be licensed and large users metered;
- Australian governments must develop compliance programmes to stop unauthorized use of groundwater;
- All groundwater must be properly priced to pay for the ongoing resource assessment, monitoring and management, and compliance programme;
- There are opportunities for surface water to be stored in aquifers rather than surface storages which have such high evaporation losses.
- Effective management of groundwater cannot be achieved with the current organizational arrangements within Government.
- Environmental water allocations must be managed by agencies that are not the same agencies who allocate water.

The late Professor Peter Cullen (2006:5) was also concerned:

"To avoid making further costly mistakes with groundwater I believe we need to reverse the burden of proof. We should assume aquifers are connected to surface water unless proven otherwise, and we should assume any further extraction of groundwater is not sustainable unless demonstrated otherwise."

An essential precautionary policy regarding conjunctive management is to assume, in the absence of a validated local model, a 1:1 ratio between groundwater extraction and surface baseflow reduction.

Some groundwater managers argue that this precautionary "default" is unduly conservative. However, within a risk assessment framework (the *Australian and New Zealand Guidelines for Fresh and Marine Water Quality,* ANZECC 2000, provide a good example of a risk-based approach to setting management targets) the function of a stage-one precautionary default is to consistently err on the conservative side. Where there is good evidence which can justify a less conservative default through a connectivity risk assessment, or better still a validated model that can be used to determine a realistic catchment/aquifer water balance, then less conservative parameters can be determined and used with confidence.

According to Evans (2007:66) "A draft national policy to address the impacts of surface water/groundwater interaction in Australia (SKM 2006) proposes ten policy principles to be adopted at the national level. Perhaps the most significant principle is that the jurisdictions need to assess the impacts of groundwater abstraction on streams, and if no assessment is undertaken then a 1:1 hydraulic relationship is to be adopted. The adoption of the 1:1 impact is based on the precautionary principle and although, as discussed earlier, this is unlikely to be the norm, there are many situations in Australia where this is the case."

RECOMMENDATIONS

Recommendation 1

There have been calls by Australian groundwater experts for the States to develop uniform (or at least compatible) methods for determining necessary environmental flows for groundwater-dependent ecosystems and for determining aquifer sustainable yields (taking GDE needs into account). These are aspects of the same problem, and need to be addressed within the agreed CoAG/NWI framework of catchment/aquifer water balance planning. Several studies have paved the way for intergovernmental agreement (Howe *et al.* 2007; SKM 2006; SKM 2001). What is needed is vision and co-operation on the part of politicians and senior government officials.

The need for national compatibility and co-operation in fact applies to all aspects of water management policy, not just to resources overlapping administrative boundaries.

Recommendation 2

Groundwater policies should provide guidance on determining the relative priority of water users (including GDEs) in receiving allocations in a scenario of "permanent" reductions in rainfall — which climate change appears to be bringing to southern and eastern Australia. Guidance should be provided, for example, on how to compare the water needs of a high value agricultural enterprise with a stygofaunal community which may support fauna found nowhere else. Among the principles currently supported by CoAG is Principle 4 of ARMCANZ (1996) which states:

"In systems where there are existing users, provision of water for ecosystems should go as far as *possible* to meet the water regime necessary to sustain the ecological values of aquatic ecosystems, whilst recognizing the existing rights of other water users" [emphasis added].

Important ecosystem services provided by natural GDEs need protection. In any policy revisions undertaken under the NWI, the existing policy on priorities should be maintained — the needs of natural GDEs should be placed above irrigation, industrial and urban demands, but below rural domestic consumption and stock watering.

Recommendation 3

Two of the most pressing practical issues, in implementing CoAG / NWI groundwater policy commitments, are to:

(a) develop catchment/aquifer management plans (or water allocation plans) which clearly demonstrate effective integration between ground and surface water management. Such plans must use a realistic catchment/aquifer water balance to produce a water balance account, and use this account in determining allocations in a precautionary way. Plans must use principles of sustainability which acknowledge the long time-frames involved in aquifer response. If groundwater is to be used as a buffer against drought, it is vital that a reserve be left for this purpose, for example by aiming to allocate no more than 50% of the annual sustainable yield (here defined as including the needs of GDEs) in "average" years.

(b) develop integrated surface/groundwater plans which include specific allocations for environmental flows (to protect identified values of affected groundwater-dependent ecosystems) calculated and delivered in a way which meet agreed Commonwealth/State environment flow principles (ARMCANZ 1996). While these principles remain official CoAG policy, they should be revised to better cater for GDEs (SKM 2001). In addition (in line with recommendations in Evans *et al.* (2006)) environmental allocations should be determined by an agency separate from the agency immediately responsible for determining water allocations for human use. The former agency should have a clear statutory responsibility to ensure the maintenance of agreed GDE values or ecosystem services over time-scales of centuries.

Recommendation 4

A recommendation made by Goesch *et al.* (2007:14) is particularly important: it relates to the *prior* determination of management responses: the use of decision rules formulated in advance. The authors, bearing in mind the difficulty of making decisions on water allocation which place farmers' livelihoods at risk, recommend that:

". . . groundwater managers [should] formalize a set of management actions that would be activated in the event of groundwater stocks falling below some predetermined thresholds.

To implement this type of strategy, it would be necessary to specify the relevant "reference" points needed to guide management decisions. For example, "target" reference points would be needed that specified the desired status of stocks and desired extractions. "Limit" reference points that identify points beyond which the risk to the aquifer and related ecosystems is regarded as unacceptably high would also be required. A set of operational rules would then be required to regulate extractions, so that stocks remained at [or above] target levels. These rules would also specify the action to be taken if the limit reference point was breached."

Rules should reduce groundwater allocations as a target point (which might be groundwater table level near a connected stream, for example) was approached. For example, additional restrictions might apply on pumping volumes or times. Once a target point had been exceeded, heavier restrictions would apply; for example irrigation extractions might be prohibited while still allowing extractions for limited town water and rural stock and domestic uses. Once a limit point was reached, all groundwater extraction in that aquifer should cease. It is important to stress again that such actions *must be discussed and agreed in advance*. This approach also reduces the likelihood of political "interference" in agreed decision-making processes — see comments by Tan (2000) quoted by Nevill (2009).

Groundwater professionals are beginning to promote the use of target and limit reference levels (Ray Evans, pers. comm. 3/4/2007) however at this stage no agreed guidelines for the use of this approach exist. Effort (perhaps on the part of the NWC) to develop and promote such guidelines is needed.

Recommendation 5

It is also important that integrated management should be applied to *all* surface/aquifer systems, not just highly connected systems. As IAHA (2004:14) pointed out: "It has been shown that even in disconnected systems, the use of one resource can affect the other". Precaution should increase as uncertainty increases.

Recommendation 6

A critical aspect affecting implementation of all of these recommendations is the routine use of independent peer review prior to finalizing plans and associated allocations. It is essential that the plans, their supporting information, as well as the peer reviews be readily available to all stakeholders and interested parties.

Recommendation 7

Concluding with a final general point, aquifers and their connected rivers should be managed within a landscape framework. In this context the recommendations made by Finlayson *et al.* (2008) concerning the control of the cumulative impacts of incremental water development within a catchment management framework need progressive adoption by State water management agencies. Their five core points, including the need to determine an upper limit on the yield of a catchment *prior to additional water being allocated*, fall within a precautionary framework.

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REFERENCES

Acworth R. I. and Timms W. A., 2003. Hydrogeological investigation of mud-mound springs developed over a weathered basalt aquifer on the Liverpool Plains, New South Wales, Australia. *Hydrogeology Journal* **11:** 659–672.

ANZECC and ARMCANZ, 2000. Australian and New Zealand Water Quality Guidelines for Fresh and Marine Waters. Australian and New Zealand Environment and Conservation Council, and Agricultural and Resource Management Council of Australia and New Zealand, Canberra.

ARMCANZ Agriculture and Resource Management Council of Australia and New Zealand, 1996. National principles for the provision of water for ecosystems. Standing Committee on Agriculture and Resource Management, Canberra.

Baird, K. J., Stromberg J. C. and Maddock T., 2005. Linking riparian dynamics and groundwater: An ecohydrologic approach to modeling groundwater and riparian vegetation. *Environmental Management* **36:** 551–564.

Barber, C., Bates L. E., Barron R., Allison H., 1998. Comparison of standardised and region-specific methods for assessment of the vulnerability of groundwater to pollution: a case study in an agricultural catchment. *In* Shallow Groundwater Systems, Volume 18, International Contributions to Hydrogeology. Verlag Heinz Heise, Hannover.

Barnett, J. C., 1980. Mesozoic and Cainozoic sediments in the Western Fortescue Plain. Western Australian Geological Survey Annual Report **1980**: 3542.

Barnett, J. C., Commander D. P., 1985. Hydrogeology of the western Fortescue Valley, Pilbara Region, Western Australia. Geological Survey Record 1986/8. Western Australia Geological Survey, Perth.

Bjarni, K. Kristjánsson B. K., Svavarsson J., 2007. Subglacial refugia in Iceland enabled groundwater amphipods to survive glaciations. *American Naturalist* **170**: 292–296.

Boulton, A. J. and Hancock P. J., 2006. Rivers as groundwater-dependent ecosystems: a review of degrees of dependency, riverine processes and management implications. *Australian Journal of Botany* **54**: 133–144.

Boulton, A. J., Fenwick, G. D., Hancock, P. J. and Harvey, M., 2008. Biodiversity, functional roles and ecosystem services of groundwater invertebrates. *Invertebrate Systematics* **22**: 103–16.

Boulton, A. J., Humphreys, W. F. and Eberhard, S. M., 2003. Imperiled subsurface waters in Australia: biodiversity, threatening processes and conservation, *Aquatic Ecosystem Health and Management* **6**(1): 41–54.

Bredehoeft, J. D., Papadopulos, S. S. and Cooper, H. H., 1982. Groundwater: the water budget myth. *In* Scientific Basis of Water Resource Management ed by National Research Council. National Academy Press, studies in geophysics, Washington DC.

Bunn, S. E., and Arthington, A. H., 2002. Basic principles and ecological consequences of altered flow regimes for aquatic biodiversity. *Environmental Management* **30**: 492–507.

Buttrick, D., 2005. Case study #15. hazard assessment on dolomite at Simunye, South Africa. Pp. 341–346 *in* Sinkholes and Subsidence: karst and cavernous rocks in engineering and construction ed by T. Waltham, F. Bell, M. G. Culshaw. Springer Praxis Books, Berlin.

Canadell, J., Jackson, R. B., Ehleringer, J. R., Mooney, H. A., Sala, Q. E. and Schulze, E-D., 1996. Maximum rooting depth of vegetation types at the global scale. *Oecologia* **108**: 583–95.

Canham, C. A., Froend, R. H. and Stock, W. D., 2009. Water stress vulnerability of four *Banksia* species in contrasting ecohydrological habitats on the Gnangara Mound, Western Australia. *Plant, Cell and Environment* **32:** 64–72.

Carroll, S., and Goonetilleke A., 2004. Assessment of high density of onsite wastewater treatment systems on a shallow groundwater coastal aquifer using PCA. *Environmetrics* **16**: 257–274.

Chen, D., 2003. Report on Dumaresq River groundwater model: model development, Department of Natural Resources and Mines Queensland, Brisbane.

Chen, Y. N., Zilliacus H., Li W. H., Zhang H. F. and Chen Y. P., 2006. Ground-water level affects plant species diversity along the lower reaches of the Tarim river, Western China. *Journal of Arid Environments* **66**: 231–246.

Clifton, C. and Evans R., 2001. A framework to assess the environmental water requirements of groundwater-dependent ecosystems. Pp 149–156 *in* Proceedings of the 3rd Australian Stream Management Conference' ed by I. Rutherford, F. Sheldon, G. Brierley and C. Kenyon. CSIRO Sustainable Ecosystems, Brisbane.

Coineau, N., 2000. Adaptations to interstitial groundwater life. Pp. 189–210 *in* Subterranean Ecosystems ed by H. Wilkens, D. C. Culver and W. F. Humphreys. Elsevier, Amsterdam.

Cooper, S. J. B., Saint, K. M., Taiti, S., Austin, A. D. and Humphreys, W. F., 2008. Subterranean archipelago: mitochondrial DNA phylogeography of stygobitic isopods (Oniscidea: *Haloniscus*) from the Yilgarn region of Western Australia. *Invertebrate Systematics* **22**: 195–203.

Cullen, P., 2006. Flying blind - the disconnect between groundwater and policy. Paper presented to the Murray-Darling Basin Groundwater Workshop, Canberra 19 September 2006.

Danielopol, D.L., Creuzé des Châtteliers, M., Mösslacher, F., Pospisil, P. and Popa, R., 1994. Adaption of Crustacea to interstitial habitats: a practical agenda for ecological studies. Pp. 218–244 *in* Groundwater Ecology ed by J. Gibert, D. L. Danielopol and J. A. Stanford. Academic Press, San Diego California.

Danielopol, D.L., Griebler, C., Gunatilaka, A. and Notenboom, J., 2003. Present state and future prospects for groundwater ecosystems. *Environmental Conservation* **30**: 104–30.

Danielopol, D.L., Pospisil, P. and R. Rouch, R., 2000. Biodiversity in groundwater: a large scale view. *Trends in Ecology and Evolution* **15:** 223–224.

Datry, T., Malard F. and Gibert J., 2005. Response of invertebrate assemblages to increased groundwater recharge rates in a phreatic aquifer. *Journal of the North American Benthological Society* **24:** 461–477.

Davies, R. J.-P., Craigie, A.I., Mackay, D. A. Whalen M. A., Cheong J. P.-E. and Leach G. J., 2007. Resolution of the taxonomy of *Eriocaulon* Eriocaulaceae taxa endemic to Australian mound springs, using morphometrics and AFLP markers. *Australian Systematic Botany* **20:** 428–447.

Dent, L. C., Schade, J. J., Grimm, N. B., and Fisher, S. G., 2000. Subsurface influences on surface biology. Pp 381–404 *in* Streams and Ground Waters ed by J. B. Jones and P. J. Mulholland. Academic Press, San Diego.

Department of Environment and Resource Managment Queensland, 2010. Annual report 2008–09. Queensland's water resource plans. http://www.derm.qld.gov.au/wrp/annual_reports.html

Department of Natural Resources and Mines Queensland, 2009. The Great Artesian Basin: information site. Http://www.nrm.qld.gov.au/water/gab/, accessed 6 Feb. 2009.

Department of Natural Resources, 2007. Great Artesian Basin resource operations plan. Department of Natural Resources and Water, Queensland; http://www.nrw.qld.gov.au/wrp/pdf/gab/gab_rop.pdf.

Department of Water, 2007. Environmental Management of Groundwater Abstraction from the Gnangara Groundwater Mound 2004–05: Annual compliance report to the Environmental Protection Authority, July 2004 to June 2005. Department of Water, Perth.

Dillon, P., Kumar, A., Kookana, R., Leijs, R., Reed, D., Parsons, S. G. and Ingleton, G., 2009. Managed Aquifer Recharge: Risks to groundwater dependent ecosystems: A review, Land and Water Australia, Canberra.

Eamus, D., Froend R., Loomes R., Hose G. and Murray B. R., 2006b. A functional methodology for determining the groundwater regime needed to maintain the health of groundwater-dependent vegetation. *Australian Journal of Botany* **54:** 97–114.

Eamus, D., Hatton, T., Cook, P. and Colvin, C., 2006a. Ecohydrology: Vegetation function, water and resource management. CSIRO Publishing, Melbourne.

Eamus, D., Macinnis-Ng C. M. O., Hose G. C., Zeppel M. J. B., Taylor D. T. and Murray B. R., 2005. Ecosystem services: an ecophysiological examination. *Australian Journal of Botany* **53:** 1–19.

Earth Tech Engineering, 2003. Review of selected factors that may change future flow patterns in the River Murray system. Murray Darling Basin Commission MDBC, Canberra.

Eberhard S. and Hamilton-Smith E., 2000. Conservation of cave communities in Australia. Pp. 647–664 *in* Ecosystems of the World, Volume 30, Subterranean Biota ed by H. Wilkens, D. C. Culver and W. F. Humphreys. Elsevier, Amsterdam.

Eberhard, S. M., 1995. Impact of a limestone quarry on aquatic cave fauna at Ida Bay in Tasmania. Pp. 125–137 *in* Proceedings of the 11th Australasian Cave and Karst Management Association Conference. Tasmania May 1995.

Eberhard, S. M., 1999. Cave fauna management and monitoring at Ida Bay Tasmania. Nature Conservation Report 99/1. Parks and Wildlife Service: Tasmania.

Eberhard, S. M., 2004. Ecology and hydrogeology of a threatened groundwater-dependent ecosystem: The Jewel Cave karst system in Western Australia. Ph.D. thesis. Murdoch University, Perth, Australia.

Elliot, W. R., 2000. Conservation of the North American cave and karst biota. Pp 665–689 *in* Ecosystems of the World: Subterranean Ecosystems ed by H. Wilkens, D.C. Culver and W.F. Humphreys. Elsevier, Amsterdam.

Elmore, A. J., Manning S. J., Mustard J. F. and Craine J. M., 2006. Decline in alkali meadow vegetation cover in California: the effects of groundwater extraction and drought. *Journal of Applied Ecology* **43:** 770–779.

Elmore, A. J., Mustard J. F. and Manning S. J., 2003. Regional patterns of plant community response to changes in water: Owens valley, California. *Ecological Applications* **13:** 443–460.

EPA, 2003. Consideration of subterranean fauna in groundwater and caves during Environmental Impact Assessment in Western Australia. Guidance Statement No 54. Environmental Protection Authority, Perth.

EPA, 2007. Sampling methods and survey considerations for subterranean fauna in Western Australia. Draft Guidance Statement No 54a (Technical Appendix to Guidance Statement No 54). Environmental Protection Authority, Perth.

Evans, R., 2007. The impact of groundwater use on Australia's rivers: technical report. Land and Water Australia, Canberra.

Evans, R., Evans, R., Jolly, P., Barnett, S., Hatton, T., Merrick, N. and Simmons, C., 2006. National Groundwater Reform. Sydney, Accessed 12 November 2008 (www.ncgm.uts.edu.au/media/National_GW_Reform.pdf).

Evans, R., Parsons, S. G., O'Grady, A. and Boulton, A. J., 2007. An assessment of groundwater-dependent ecosystems in the Burnett River Groundwater area, Queensland Australia. Paper presented to the International Association of Hydrogeologists: XXXV Congress, Lisbon Portugal, 17–21 September 2007.

Evans, R., Richardson, S., Hillier, J., Bonte, M., Dyson, P., Ross, J. and Middlemis, H., 2003. Watermark: sustainable groundwater use within irrigation regions. Murray-Darling Basin Commission MDBC, Canberra.

Fairfax, R. J. and Fensham, R. J., 2002. In the footsteps of J. Alfred Griffiths: a cataclysmic history of the Great Artesian Basin Springs in Queensland, Australia. *Australian Geographical Studies* **40**(2): 210–30.

Fensham R., Ponder, W. and Fairfax, R., 2007. Recovery plan for the community of native species dependent on natural discharge of groundwater from the Great Artesian Basin. Report to Department of the Environment, Water, Heritage and the Arts , Canberra. Queensland Parks and Wildlife Service, Brisbane. Pp. 44. (http://www.gabcc.org.au/tools/getFile.aspx?tbl=tblContentItemandid=172).

Fensham, R. J. and Fairfax R. J., 2003. Spring wetlands of the Great Artesian Basin, Queensland, Australia. *Wetlands Ecology and Management* **11:** 343–362.

Fensham, R. J. and Price R. J., 2004. Ranking spring wetlands in the Great Artesian Basin of Australia using endemicity and isolation of plant species. *Biological Conservation* **119:** 41–50.

Fensham, R. J., 2006. Spring Wetlands of the Great Artesian Basin. Department of the Environment and Heritage, Canberra.

Fensham, R. J. and Fairfax, R. J., 2009. Development and trial of a spring wetland monitoring methodology in the Great Artesian Basin,Queensland. Report to the Department of Environment and Resource Management Queensland, Brisbane.

Finlayson, B., Nevill, J. and Ladson, A., 2008. Cumulative impacts in water resource development. Paper presented to the Water Down Under Conference, Adelaide, April 14–18, 2008.

Foley, A. M., Peck, S. A. and Harman, G. R., 2006. Effects of sand characteristics and inundation on the hatching success of loggerhead sea turtle (*Caretta caretta*) clutches on low-relief mangrove islands in southwest Florida. *Chelonian Conservation and Biology* **5**(1): 32–41.

Ford, D. C., and Williams, P. W., 1989. Karst Geomorphology and Hydrology. Unwin Hyman, London.

Forti, P., Galdenzi S., and Sarbu S. M., 2002. The hypogenic caves: a powerful tool for the study of seeps and their environmental effects. *Continental Shelf Research* **22**, 2373–2386.

Froend, R. H. and Drake P. L., 2006. Defining phreatophyte response to reduced water availability: preliminary investigations on the use of xylem cavitation vulnerability in Banksia woodland species. *Australian Journal of Botany* **54**: 173–179.

Froend, R., Loomes R., Horwitz P., Rogan R., Lavery P., How J., Storey A., Bamford M., Metcalf B., 2004. Study of EWRs on the Gnangara and Jandakot Mounds under Section 46 of the Environmental Protection Act. Prepared for the Waters and Rivers Commission, Perth. Centre for Ecosystem Management, Edith Cowan University Joondalup.

Gerritse, R. G., 1998. Biogeochemical changes in aquifers from injected waste water. CSIRO Land and Water Report No 16/98.

Gibert, J. and Deharveng, L., 2002. Subterranean ecosystems: a truncated functional biodiversity. *Bioscience* **52:** 473–481.

Goesch, T., Hone, S. and Gooday, P., 2007. Groundwater Management: Efficiency and Sustainability. ABARE Australian Bureau of Agricultural and Resource Economics, Canberra.

Gordon, N. D., McMahon, T. A., Finlayson, B. L., Gippel, C. J. and Nathan, R. J., 2004. Stream Hydrology: An introduction for ecologists. John Wiley and Sons, Ltd, West Sussex, England.

Government of Australia, 1992. National strategy for ecologically sustainable development Australian Government Publishing Service, Canberra.

Government of Australia, 2004. Intergovernmental Agreement on a National Water Initiative. CoAG Council of Australian Governments, Canberra.

Government of Australia, 2009a. Great Artesian Basin Strategic Initiative. Http://www.environment.gov.au/water/environmental/groundwater/gabsi.html, accessed 6 Feb. 2009.

Government of Australia, 2009b. List of endangered species and communities. http://www.environment.gov.au/cgi-bin/sprat/public/publicshowcommunity.pl?id=26andstatus=Endangered, accessed 6 Feb. 2009.

Grimes, K. G., 2006. Syngenetic karst in Australia: a review. *Helictite* **39**(2): 27–38.

Groom, P. K., 2004. Rooting depth and plant water relationships explain species distribution patterns within a sandplain landscape. *Functional Plant Biology* **31**, 425–428.

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

Groom, P. K., Froend R. H., Mattiske E. M. and Koch B., 2000b. Myrtaceous shrub species respond to long-term decreasing groundwater levels on the Gnangara Groundwater Mound, northern Swan Coastal Plain. *Journal of the Royal Society of Western Australia* **83**: 75–82.

Habermehl, M. A., 1980. The Great Artesian Basin, Australia. *Bureau of Mineral Resources, Journal of Australian Geology and Geophysics* **5:** 9–38.

Habermehl, M. A., 1996. Groundwater Movement and Hydrochemistry of the Great Artesian Basin,Australia. Pp. 228–236 *in* Mesozoic Geology of the Eastern Australia Plate Conference, Geological Society of Australia, Sydney.

Habermehl, M. A., 2006. Artesian springs of the Great Artesian Basin. Pp. 1–12 *In* T. B. Goth, and J. Defavari (compilers) *6th GAB Spring Researchers Forum, Adelaide, 23–24 February, 2006.* South Australian Arid Lands Natural Resources Management Board, Adelaide.

Habermehl, M. A., Ransley, T., McNee, A. , van der Voort, A., 2006. Report on the pilot project Improved management and protection of endangered groundwater dependent ecosystems within the Great Artesian Basin, Australia. Bureau of Rural Sciences, Department of Agriculture, Fisheries and Forestry, Australian Government, Canberra.

Hafi, A., Brownlowe, N. and Welsh, W., 2006. Groundwater and surface water use in the Dumaresq River Valley. Australian Bureau of Agricultural and Resource Economics ABARE, Canberra.

Hahn, H. J., 2006. The GW-Fauna-Index: A first approach to a quantitative ecological assessment of groundwater habitats. *Limnologia* **36**: 119–139.

Hamilton-Smith, E., and Eberhard S., 2000. Conservation of cave communities in Australia. Pp 647–664 *in* Ecosystems of the World: Subterranean Ecosystems ed by H. Wilkens, D. C. Culver and W. F. Humphreys. Elsevier, Amsterdam.

Hancock, P. J., Boulton A. J. and Humphreys W. F., 2005. Aquifers and hyporheic zones: towards an ecological understanding of groundwater. *Hydrogeology Journal* **13**: 98–111.

Hancock, P., 2002. Human impacts on the stream-groundwater exchange zone. *Environmental Management* **29:** 761–781.

Hancock, P., 2004. Groundwater-dependent aquatic biota survey of the Pioneer Valley, Queensland. Aquatic Ecosystem Health Technical Report, Queensland Department of Natural Resources, Mines and Energy, Brisbane.

Hardin, G., 1968. The tragedy of the commons. *Science* **162**: 1243–8.

Harris, C., 1981. Oases in the desert: the mound springs of Northern South Australia. *Proceedings of the Royal Geographers Society of South Australia* **81**: 26–39.

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

Horton, J. L., Kolb T. E. and Hart S. C., 2001a. Responses of riparian trees to interannual variation in ground water depth in a semi-arid river basin. *Plant, Cell and Environment* **24**: 293–304.

Horton, J. L., Kolb T. E. and Hart S. C., 2001b. Physiological response to groundwater depth varies among species and with river flow regulation. *Ecological Applications* **11**: 1046–1059.

Howe, P. and Pritchard J., 2007. A Framework for Assessing the Environmental Water Requirements of Groundwater-dependent Ecosystems: Report 3 Implementation. Land and Water Australia, Canberra.

Howe, P., Pritchard, J., Cook, P. G., Evans, R., Clifton, C. and Cooling, M., 2007. A framework for assessing the environmental water requirements of groundwater dependent ecosystems: Report No.3: Implementation., Land and Water Australia, Canberra.

Humphreys, W. F., 1999. Relict stygofaunas living in sea salt, karst and calcrete habitats in arid northwestern Australia contain many ancient lineages. Pp. 219–227 *in* The Other 99%: The Conservation and Biodiversity of Invertebrates ed by W. Ponder and D. Lunney. Royal Zoological Society of New South Wales, Mosman.

Humphreys, W. F., 2000a. First in, last out: should aquifer ecosystems be at the vanguard of remediation assessment? Pp. 275–282 *in* Contaminated Site Remediation: from source zones to ecosystems ed by C.D. Johnston. Centre for Groundwater Studies, Wembley, Western Australia.

Humphreys, W. F., 2000b. Karst wetlands biodiversity and continuity through major climatic change — an example from arid tropical Western Australia. Pp. 227–258 *in* Biodiversity in Wetlands: Assessment, function and conservation: Volume 1 ed by B. Gopal, W. J. Junk and J. A. Davis. Backhuys Publishers, Leiden.

Humphreys, W. F., 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.

Humphreys, W. F., 2002. Keynote address: groundwater ecosystems in Australia: an emerging understanding. *In* "Proceedings of the International Association of Hydrogeologists conference, Darwin, Australia, 12–17 May 2002". [CD-ROM].

Humphreys, W. F., 2006a. Groundwater fauna. Paper prepared for the 2006 Australian State of the Environment Committee, Department of the Environment and Heritage, Canberra, http://www.deh.gov.au/soe/2006/emerging/fauna/index.html accessed June 4, 2007.

Humphreys, W. F., 2006b. Aquifers: the ultimate groundwater-dependent ecosystems. In D. Eamus (ed.) Special edition on Groundwater-dependent Ecosystems. *Australian Journal of Botany* **54**: 115–132.

Humphreys, W. F., 2008a. Rising from down under: developments in subterranean biodiversity in Australia from a groundwater fauna perspective. *Invertebrate Systematics* **22**: 85–101.

Humphreys, W. F., 2008b. Hydrogeology and groundwater ecology: Does each inform the other? *Hydrogeology Journal* **17**(1): 5–21.

Humphreys, W. F., 2009. Hydrogeology and groundwater ecology: does each inform the other? *In* Special Issue, Hydrogeoecology, the interdisciplinary study of groundwater dependent ecosystems ed by P. J. Hancock, J. H. Randall and A. J. Boulton. *Hydrogeology Journal* 17: 5–21.

IAHA International Association of Hydrogeologists Australia, 2004. Guiding principles for sustainable groundwater management. Murray-Darling Basin Commission in collaboration with Resource and Environmental Management SA, Canberra.

Iliffe, T. M., Jickells, T. D. and Brewer, M. S., 1984. Organic pollution of an inland marine cave from Bermuda. *Marine Environment Research* **12**: 173–189.

Jasinska, E. J., Knott B. and McComb A. J., 1996. Root mats in ground water: a fauna-rich cave habitat. *Journal of the North American Benthological Society* **15**: 508–519.

Jasinska, E. J., Knott B., 2000. Root-driven faunas in cave waters. Pp 287–307 *in* Ecosystems of the World: Subterranean ecosystems ed by H. Wilkens, D. C. Culver and W. F. Humphreys. Elsevier, Amsterdam.

Johannes, R. E. and Rimmer, D. W., 1984. Some distinguishing characteristics of nesting beaches of the green turtle *Chelonia mydas* on North West Cape Peninsula, Western Australia. *Marine Biology* **83**(2): 149–54.

Johannes, R. E., 1980. The ecological significance of the submarine discharge of groundwater. *Marine Ecology Progress Series* **3**: 365–73.

Johnson, S. L., Commander, D. P. and O'Boy, C. A., 1999. Groundwater resources of the Northern Goldfields, Western Australia. Hydrological Records Series, Report HG 2, Water and Rivers Commission, Perth.

Kamermans, P., Hemminga, M. A., Tack, J. F., Mateo, M. A., Marba, N., Mtolera, M., Stapel, J., Verheyden, A. and Van Daele, T., 2002. Groundwater effects on diversity and abundance of lagoonal seagrasses in Kenya and on Zanzibar Island (East Africa), *Marine Ecology Progress Series* **231**: 75–83.

Ladson, A., 2008. Hydrology: an Australian introduction. Oxford University Press, Melbourne.

Leaney, F. W., Herczeg, A. L., Walker, G.R., 2003. Salinisation of a fresh palaeogroundwater resource by enhanced recharge. *Ground Water* **41:**, 84–93.

Lite, S. J. and Stromberg J. C., 2005. Surface water and ground-water thresholds for maintaining *Populus-Salix* forests, San Pedro River, Arizona. *Biological Conservation* **125**: 153–167.

Liu, J. Z. and Chen Y. N., 2002. Analysis on converse succession of plant communities at the lower reaches of Tarim River. *Arid Land Geography* **25**: 231–236.

Liu, J. Z., Chen Y. N., Chen Y. J., Zhang N. and Li W. H., 2005. Degradation of *Populus euphratica* community in the lower reaches of the Tarim River, Xinjiang, China. *Journal of Environmental Sciences* **17**: 740–747.

Longley, G,. 1992. The subterranean aquatic ecosystem of the Balcones Fault Zone Edwards Aquifer in Texas - threats from overpumping. Pp. 291–300 *in* Proceedings of the First International Conference on Groundwater Ecology ed by J. A. Stanford and J. J. Simons. American Water Resources Association, Bethesda Maryland.

Longley, G., 1986. The biota of the Edwards Aquifer and the implications for palaeozoogeography. Pp. 51–54 *in* The Balcones Escarpment Central Texas ed by P. L. Abbott and C. M. Woodruff Jr. Geological Society of America, Washington DC.

Lubczynski, M. W., 2007. Ecohydrology and groundwater dependency of water limited ecosystems. Pp. 16-26 *in* The XXXV IAH Congress of the International Association of Hydrogeologists: Groundwater and Ecosystems 17–21 September 2007 ed by L. Ribeiro, A. Chambel and A. Condeooa de Melo. International Association of Hydrogeologists London, Lisbon Protugal.

Ludwig, D., Hilborn, R., and Walters, C., 1993. Uncertainty, resource exploitation and conservation lessons from history. *Science* **260**(2):17.

Malard, F., Reygrobellet J-L., Mathieu J., Lafont M., 1994. The use of invertebrate communities to describe groundwater flow and contaminant transport in a fractured rock aquifer. *Archiv fur Hydrobiologie* **131**: 93–110.

Margat, J., 1994. Groundwater operations and management. Pp. 505–522 *in* Groundwater Ecology ed by J. Gibert, D. L. Danielopol, and J. A. Stanford. Academic Press, London).

Martí, E., Fisher S. G., Schade J. J. and Grimm N. B., 2000. Flood frequency and stream-riparian linkages in arid lands. Pp. 111–136 *in* Streams and Ground Waters ed by J. B. Jones and P. J. Mulholland. Academic Press, San Diego.

Mauclaire, L. and Gibert J., 1998. Effects of pumping and floods on groundwater quality: a case study of the Grand Gravier well field (Rhône, France). *Hydrobiologia* **389**: 141–151.

Mohr, R., 2006. Guidelines on the precautionary principle and development. *Deacons Legal Update* (April 2006): 2–3.

Molsher, R. and Coote, D., 2003. Environmental and social valuation of bore-fed wetlands in the NSW portion of the Great Artesian Basin: Stage 1 — Preliminary investigation. NSW National Parks and Wildlife Service — Western Directorate, Sydney.

Mues, C., and Hardcastle, S., 1998. Resource Management Issues in the Great Artesian Basin. *In* Proceedings of the National Agricultural and Resource Outlook Conference. Canberra 3–5 February 1998. ABARE, Canberra.

Muñoz-Reinoso, J. C., 2001. Vegetation changes and groundwater abstraction in SW Doñana, Spain. *Journal of Hydrology* **242**: 197–209.

Murphy, N. P., Adams, M. and Austin, A. D., 2009. Independent colonization and extensive cryptic speciation of freshwater amphipods in the isolated groundwater springs of Australia's Great Artesian Basin. *Molecular Ecology* **18**:109–22.

Murray, B. R., Hose G. C., Eamus D. and Licari D., 2006. Valuation of groundwater-dependent ecosystems: a functional methodology incorporating ecosystem services. *Australian Journal of Botany* **54**: 221–229.

Naiman, R. J., Décamps, H. and McClain, M. E., 2005. Riparia: Ecology, Conservation, and Management of Streamside Communities. Elsevier Academic Press, California.

Naumburg, E., Mata-Gonzalez R., Hunter R. G., McLendon T. and Martin D. W., 2005. Phreatophytic vegetation and groundwater fluctuations: a review of current research and application of ecosystem response modeling with an emphasis on Great Basin vegetation. *Environmental Management* **35**: 726–740.

Nevill, J., 2008. Comment on recent progress in water-balance planning and the supply of environmental flows to ground water-dependent ecosystems. *Ecological Management and Restoration* **9**(2): 145–8.

Nevill, J., 2009. Managing cumulative impacts: groundwater reform in the Murray-Darling Basin, Australia. *Water Resources Management* **23**: 2605–31.

NLWRA, 2001. Australian Water Resources Assessment 2001. Land and Water Australia, National Land and Water Resources Audit, Commonwealth of Australia, Canberra.

Noble J. C., Habermehl M.A., James C.D., Landsberg J. and Langton A.C., 1998. Biodiversity implications of water management in the Great Artesian Basin. *Rangeland Journal* **20:** 275–300.

Notenboom J., Plénet S., Turquin M-J., 1994. Groundwater contamination and its impact on groundwater animals and ecosystems. Pp. 477-–504 *in* Groundwater Ecology ed by J. Gibert, D. L. Danielopol, and J. A. Stanford. Academic Press, London.

Odum, W., 1982. Environmental degradation and the tyranny of small decisions. *BioScience* **32**(9): 728–9.

O'Grady A., Cook P. G., Duguid A. and Knapton A., 2006a. Water use by *Corymbia opaca* trees over a depth to water table gradient, Ti Tree, Central Australia. *In* Groundwater Surfacewater Interaction in the Arid Zone, Seminar organised by the Northern Territory Branch of the International Association of Hydrogeologists, Alice Springs, 7–8 September 2006.

O'Grady, A. P., Cook P. G., Howe P. and Werren G., 2006b. Groundwater use by dominant tree species in tropical remnant vegetation communities. *Australian Journal of Botany* **54:** 155–171.

Packman A. I. and Bencala K. E., 2000. Modeling surface-subsurface hydrologic interactions. Pp. 46–80 *in* Streams and Ground Waters ed by J. B. Jones J. B. and P. J. Mulholland. Academic Press, San Diego.

Perez, K. E., Ponder, W. F., Colgan, D. J., Clark, S.A., and Lydeard, C., 2005. Molecular phylogeny and biogeography of spring-associated hydrobiid snails of the Great Artesian Basin, Australia. *Molecular Phylogenetics and Evolution* **34**: 545–556.

Pickard, J., 1992. Artesian Springs in the Western Division of New South Wales. Graduate School of the Environment, Working paper 9202, Macquarie University, Sydney, Australia.

Playford, P. E., 2001. Subterranean biotas in Western Australia. Report for the Environmental Protection Authority, Perth, Australia.

Plénet S. and Gibert J., 1994. Invertebrate community responses to physical and chemical factors at the river/aquifer interaction Zone I: upstream from the city of Lyon. *Archiv für Hydrobiologie* **132**: 165–189.

Plénet S., Hugueny H. and Gibert J., 1996. Invertebrate community responses to physical and chemical factors at the river/aquifer interaction Zone II: downstream of the city of Lyon. *Archiv für Hydrobiologie* **136**: 65–88.

Poff, N. L., Allan, J. D., Bain, M. B., Karr, J. R., Prestegaard, K. L., Richter, B. D., Sparks, R. E. and Stromberg, J. C., 1997. The natural flow regime, *BioScience* **47**(11): 769–84.

Ponder, W. F. and Clark, G. A., 1990. A radiation of hydrobiid snails in threatened artesian springs in western Queensland. *Records of the Australia Museum*. **42**(3): 301–363.

Ponder, W. F., 1986. Mound springs of the Great Artesian Basin. Pp. 403–420 *in* Limnology of Australia ed by P. DeDeckker, and W. D. Williams. CSIRO and W. Junk, The Hague.

Ponder, W. F., 2004. Endemic aquatic macroinvertebrates of artesian springs of the Great Artesian Basin — progress and future directions. *Records of the South Australian Museum Monograph Series* **7**: 101–110.

Ponder, W. F., Clark, S. A., Eberhard, S. and Studdert, J. B., 2005. A radiation of hydrobiid snails in the caves and streams at Precipitous Bluff, southwest Tasmania, Australia (*Mollusca: Caenogastropoda: Rissooidea:* Hydrobiidae s.l.). *Zootaxa* **1074**: 1–66.

Ponder, W. F., Eggler, P. and Colgan, D. J., 1995. Genetic differentiation of aquatic snails (*Gastropoda: Hydrobiidae*) from artesian springs in arid Australia. *Biological Journal of the Linnean Society* **56**: 553–96.

Ponder, W. F., Hershler, R. and Jenkins, B., 1989. An endemic radiation of *Hydrobiidae* from artesian springs in northern South Australia: their taxonomy, physiology, distribution and anatomy. *Malacologia* **31**: 1–140.

Poore G. C. B., Humphreys, W.F., 1998. First record of *Spelaeogriphacea* from Australasia: a new genus and species from an aquifer in the arid Pilbara of Western Australia. *Crustaceana* **71**: 721–742.

Post, V. E. A., 2005. Fresh and saline groundwater interaction in coastal aquifers: Is our technology ready for the problems ahead? *Hydrogeology Journal* **13**: 120–123.

Preston, C. J., 2006. Telstra Corporation Limited v Hornsby Shire Council, New South Wales Land and Environment Court, Justice CJ Preston, NSWLEC 133, 24 March 2006.

Pringle, C. M., 2001. Hydrologic connectivity and the management of biological reserves: a global perspective. *Ecological Applications* **11**: 981–98.

Rouch R., Pitzalis A., Descouens, A ., 1993. Effets d'un pompage à gros dèbit sur le peuplement des Crustacès d'un aquifére karstique. *Annales de Limnologie* **29**: 15–29.

Rutkowski, C., Burnett, W., Iverson, R. and Chanton, J., 1999. The effect of groundwater seepage on nutrient delivery and seagrass distribution in the northeastern Gulf of Mexico. *Estuaries and Coasts* **22**(4): 1033–40.

Sattler, P. and Creighton, C., 2002. Australian Terrestrial Biodiversity Assessment 2002. National Land and Water Resources Audit, Canberra.

Scheffer M., Carpenter S. R., Foley J.A., Folke C. and Walker B., 2001. Catastrophic shifts in ecosystems. *Nature* **413**: 591–596.

Scott M. L., Shafroth P. B. and Auble G. T., 1999. Responses of riparian cottonwoods to alluvial water table declines. *Environmental Management* **23**: 347–358.

Shafroth P. B., Stromberg J. C. and Patten D. T., 2000. Woody riparian vegetation response to different alluvial water table regimes. *Western North American Naturalist* **60**: 66–76.

SKM Sinclair Knight Metz, 2001. Environmental Water Requirements to Maintain Groundwater-dependent Ecosystems. Sinclair Knight Metz, Environmental Flows Initiative Technical Report No. 2, Commonwealth of Australia, Canberra.

SKM Sinclair Knight Metz, 2006. Towards a national framework for managing the impacts of groundwater and surface water interaction in Australia. Department of Agriculture, Fisheries and Forestry Australia, Armadale Melbourne.

Sommer, B. and Horwitz, P., 2001. Water quality and macroinvertebrate response to acidification following intensified summer droughts in a Western Australian wetland. *Marine and Freshwater Research* **52**: 1015–21.

Sophocleous, M., 2002. Interactions between groundwater and surface water: the state of the science. *Hydrogeology Journal* **10**: 52–67.

Sophocleous, M., 2010. Review: groundwater management practices, challenges, and innovations in the High Plains aquifer, USA — lessons and recommended actions. *Hydrogeology Journal* **18**: 559–75.

Stein, P. L., 1999. Are decision-makers too cautious with the precautionary principle? Paper presented to the Land and Environment Court of New South Wales Annual Conference Oct 14–15, Sydney.

Tan, P-L., 2000. Conflict over water resources in Queensland: all eyes on the Lower Balonne. *Environment and Planning Law Journal* **17**: 545–68.

Tomlinson, M. and Boulton, A. J., 2008. Subsurface groundwater dependent ecosystems: a review of their biodiversity, ecological processes and ecosystem services. National Water Commission, Canberra.

Trémolières, M., Correll D. & Olah J., 1997. Riparian vegetation and water quality improvement. In: Groundwater/ Surface Water Ecotones — Biological and Hydrological Interactions and Management Options (eds Gibert J., Mathieu J. & Fournier F.), pp. 227–230. Cambridge University Press, Cambridge.

Van de Koppel, J., Rietkerk M., van Langevelde F., Kumar L., Klausmeier C. A., Fryxell J. M., Hearne J. W., van Andel J., de Ridder N., Skidmore A., Stroosnijder L. and Prins H. H. T., 2002. Spatial heterogeneity and irreversible vegetation change in semiarid grazing systems. *American Naturalist* **159**: 209–218.

Wager, R. and Unmack, P. J., 2000. Fishes of the Lake Eyre Catchment of central Australia. Department of Primary Industries and Queensland Fisheries Service, Brisbane, Queensland.

Ward, J. V., Tockner, K. and Schiemer, F., 1999. Biodiversity of floodplain river ecosystems: ecotones and connectivity, *Regulated Rivers: Research and Management* **15**: 125–30.

Webb, R. H. and Leake S. A., 2006. Ground-water surface-water interactions and long-term change in riverine riparian vegetation in the southwestern United States. *Journal of Hydrology* **320**: 302–323.

Wilson, B. A., 1995. Artesian springs of the Great Artesian Basin in Queensland. Report to the Queensland Department of the Environment and Heritage and the Australian Nature Conservation Agency, Brisbane.

Wilson, G. D. F. and Keable, S. J., 2004. A new family and genus of *Phreatoicidea (*Crustacea: Isopoda*)* from artesian springs in southwestern Queensland, Australia. *Memoirs of the Queensland Museum* **49**: 741–759.

Xu, H., Ye M., Song Y. and Chen Y., 2007. The natural vegetation responses to the groundwater change resulting from ecological water conveyances to the lower Tarim river. *Environmental Monitoring and Assessment* **131**: 37–48.

Zektser, I. S., Loaiciga, H. A, and Wolf, J. T., 2005. Environmental impacts of groundwater overdraft: selected case studies in the southwestern United States. *Environmental Geology* **47**: 396–404.

Zencich, S. J., Froend R. H., Turner J. V. and Gailitis V., 2002. Influence of groundwater depth on the seasonal sources of water accessed by Banksia tree species on a shallow, sandy coastal aquifer. *Oecologia* **131**: 8–19.

BOOK REVIEW

Ecological Restoration: Principles, Values, and Structure of an Emerging Profession

Andre F. Clewell and James Aronson, 2008
Island Press, Washington, D.C.
ISBN: 9781597261692
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The field of ecological restoration is a rapidly growing discipline that encompasses a wide range of activities and brings together practitioners and theoreticians from a variety of backgrounds and perspectives. This textbook is part of the Science and Practice of Ecological Restoration Book Series which is intended to serve a broad audience of people who are active in the field of ecological restoration or have specialized interests in it. In the authors' own words, this book aims to move beyond the past twelve volumes in the series and create a unified vision of ecological restoration as a field of study, one that clearly states the discipline's precepts and emphasizes issues of importance to those involved at all levels. In doing this, the authors of book fundamentally aim to embrace a wider, holistic definition that takes into account both environmental and social components.

To achieve this, the authors outline four major aims: to identify fundamental concepts upon which restoration is based; to consider the principles of restoration practice; to explore the diverse values that are fulfilled with the restoration of ecosystems; and, to review the structure of restoration practice. The reviews of practice include various contexts for restoration work, the professional development of its practitioners, and the relationships of restoration with allied fields and activities.

I found the book goes some way in achieving these aims, but in an imbalanced manner. Due to the need to make the text relevant to all people involved with ecological restoration, the text is light in detail for achieving the first two aims (i.e., identifying and describing fundamental concepts and principles of the discipline) and heavy on the last two (i.e., examining values and structures that underpin good ecological restoration activities). There was no mention, for example, of methodologies behind conducting system prioritization plans for where to conduct restoration activities, but a lot of detail on practical steps, such as how to engage local communities to conduct activities. For this reason, I think this book is more useful for practitioners actually conducting ecological restoration than for students studying the discipline in a more theoretical manner, or for planners, wanting to know how to conduct restoration activities in the most effective manner.

A unique feature of the book is the inclusion of eight "virtual field trips": short photo essays of project sites around the world that illustrate various points made in the book and are "led" by those who were intimately involved with the project described. These field trips are case studies from all the inhabited continents, are different from each other, and cover a variety of different issues. The format of having these field trips entwined with the chapters of the book makes the book engaging and lively. However, the accompanying photographs are poor quality black-and-white reproductions, serving to break up the text rather than provide a more lucid visual representation of the discipline.

Despite these drawbacks, I would recommend the book to those working hands-on in the conception, planning and management of ecological restoration projects.

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