Australian Government

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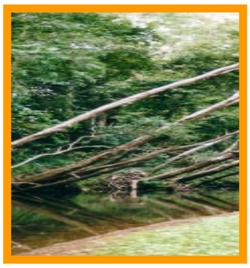
Australian groundwater-dependent ecosystems toolbox part 1: assessment framework

Sinclair Knight Merz

Waterlines Report Series No 69, December, 2011





Waterlines

A SERIES OF WORKS COMMISSIONED BY THE NATIONAL WATER COMMISSION ON KEY WATER ISSUES

Waterlines

This paper is part of a series of works commissioned by the National Water Commission on key water issues. This work has been undertaken by Sinclair Knight Merz on behalf of the National Water Commission.

Online/print: ISBN: 978-1-921853-52-4

Australian groundwater-dependent ecosystems toolbox part 1: assessment framework
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Published by the National Water Commission
95 Northbourne Avenue
Canberra ACT 2600
Tel: 02 6102 6000
Email: *enquiries@nwc.gov.au*

Date of publication: December, 2011

Cover design by: Angelink
Front cover image courtesy of (from left to right): South East NRM Board (Wetland, South Australia), Paul Howe (Coolabah Woodland, Mt. Bruce Flats, Pilbara) and Queensland Department of Natural Resources and Water (Sandy Creek, Pioneer Valley).

An appropriate citation for this report is:
Richardson S, et al 2011 *Australian groundwater-dependent ecosystem toolbox part 1: assessment framework*, Waterlines report, National Water Commission, Canberra

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Abbreviations and acronyms

ANAE	Australian National Aquatic Ecosystem
ANZECC	Australian and New Zealand Environment and Conservation Council
ARMCANZ	Agriculture and Resource Management Council of Australia and New Zealand
BACI	before/after and control/impact
BBN	Bayesian Belief Network
COAG	Council of Australian Governments
CSIRO	Commonwealth Scientific and Industrial Research Organisation
ECU	electrical conductivity units
EHZ	ecohydrogeological zone
ET	evapotranspiration
EWP	ecological water provisions
EWR	ecological water requirements
GDE	groundwater-dependent ecosystem
GDE toolbox	Australian groundwater-dependent ecosystems toolbox
GIS	geographical information system
LAI	leaf area index
LiDAR	Light Detection and Ranging
LWA	Land and Water Australia
NWC	National Water Commission
NWI	National Water Initiative
REM	Resource and Environmental Management
SKM	Sinclair Knight Merz
toolbox, the	Australian groundwater-dependent ecosystems toolbox

Acknowledgements

The information in this document, and the ideas on which the discussion is based, result from the experience of many people across Australia. The authors wish to thank all who contributed to this work.

In particular, the authors wish to acknowledge the input of Ian Overton, from CSIRO Land and Water, whose insight into the study of groundwater-dependent ecosystems was fundamental in the development of the Assessment Framework. Anthony O'Grady (CSIRO Ecosystem Services) and Sébastien Lamontange (CSIRO Land and Water) are also thanked for their inputs to tools T7, T10 and T11 (presented in *part 2: assessment tools* of the toolbox and review capacity). Lindsay Hutley (Charles Darwin University) and Des Yin Foo (Department of Natural Resources, Environment, the Arts and Sport, Northern Territory) are gratefully acknowledged for providing a 'tropics' perspective on groundwater-dependent ecosystems (GDEs).

The authors of the original (2007) version of the 'GDE toolbox' are also acknowledged: Paul Howe, Jodie Pritchard, Anthony O'Grady, Peter Cook, Tony Fas, Craig Clifton, Brendan Cossens and Chris McAuley, Rick Evans, Marcus Cooling and Andrew Boulton.

Three workshops were held during the development of this framework with the project's reference group. The members provided invaluable guidance and direction on the GDE toolbox content and structure. The reference group members are sincerely thanked for their enthusiasm and input to the project.

Laura Gow	Geoscience Australia
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Adam Sluggett	Murray–Darling Basin Authority
Jill McNamara, John Sharples	Department of Sustainability and Environment (Vic.)
Moya Tomlinson	Department of Environment and Resource Management (Qld)
Jason Vanlaarhoven, Clare Harding and David Deane	Department for Water (SA)
Mark Mitchell	Office of Water (NSW)
Adam Sincock, Shane Hogan	National Water Commission

1. Introduction

1.1. The GDE toolbox in context

All ecosystems (including ecological processes) are dependent upon a supply of water. Sometimes that supply comes from surface water—the water that occurs in rivers and lakes or falls directly on the surface of the land. In much of Australia, however, groundwater is a vital water source for ecosystems and for human use.

For the purpose of the GDE toolbox, groundwater is defined as subsurface water located in the zone of saturation in pores, fractures in rocks and cavities.

It is recognised that in some jurisdictions (such as Tasmania) groundwater means all water stored below the ground surface (including the unsaturated zone). Future versions of the GDE toolbox may better address the relationships between ecosystems and water in the unsaturated zone.

Australia has extensive groundwater reserves but they are being increasingly extracted for agriculture, industry and potable water supplies. Moreover, groundwater quality is often under threat due to the intrusion of saline water and nutrient enrichment and, in some cases, by contamination with pollutants.

Groundwater is especially important to ecosystems in arid and semi-arid parts of the country and when there are extended dry periods, during which evaporation markedly exceeds precipitation and surface water is scarce (Eamus et al. 2006).

Groundwater-dependent ecosystems (GDEs) are defined as ecosystems that require access to groundwater to meet all or some of their water requirements so as to maintain the communities of plants and animals, ecological processes they support, and ecosystem services they provide. (Modified from Clifton et al. 2007 and Tomlinson 2011.)

Just as terrestrial and aquatic ecosystems have a surface water requirement to maintain their structure and function, GDEs also have an ecological water requirement (see Section 1.2). The presence of groundwater drives the evolution, persistence and resilience of GDEs, and the state of these ecosystems is dependent on at least two aspects of the groundwater, including (modified from Brown et al. 2007):

- physical characteristics, such as the quantity, location, timing, frequency and duration of groundwater delivery (or supply)
- chemical characteristics, such as water quality (especially salinity and nutrient concentrations) and temperature.

Changes in any one of these variables can initiate changes in the structure and function of a GDE. Concern for the ongoing survival of GDEs and the need to manage the impacts from potential and expected changes to groundwater quantity and quality is integrated in current water management planning.

There are many management drivers for the assessment of GDEs, including the need to:

- account for the water requirements of GDEs within groundwater management plans. Tomlinson (2011) identifies national and state-based legislative policy drivers related to provision of environmental water within the water planning context

- support assessments of the potential impacts from resource development activities (such as mining and irrigation development) for environmental approvals (refer final reports from the 'Potential local and cumulative effects of mining on groundwater resources' project, available at *http://www.nwc.gov.au/www/html/2992-potential-effects-of-mining-reports.asp?intSiteID=1*)
- include the role of groundwater in surface water planning (e.g. Georges et al. 2003).

In 2007, Land & Water Australia (LWA) commissioned Sinclair Knight Merz (SKM), Resource & Environmental Management (REM) and CSIRO to develop a practical tool to assist in the identification of GDEs and the management of their environmental water requirements (Clifton et al. 2007). That tool, *A Framework for Assessing the Environmental Water Requirements of Groundwater Dependent Ecosystems*, became to be known as the 'GDE toolbox' or, more simply, the 'toolbox'. Significant advances have been made in our understanding of GDEs and their environmental water requirements since the toolbox was first developed. In recognition of those advances, the National Water Commission (NWC) undertook a revision of the toolbox to update the scientific tools it outlined, to facilitate its wider adoption, and to generally improve it in the light of feedback provided over the past four years by the practitioners at which it was targeted.

This revised version presents a suite of practical and technically robust tools and approaches that will allow water resource, catchment and ecosystem managers to identify GDEs, determine the reliance of those ecosystems on groundwater, and determine possible changes to ecosystem state or function due to changes in the groundwater environment.

The main revisions have been to update the scientific tools previously presented in the LWA (2007) version. More detail has also been provided on ecological water requirements (EWRs), the effect of climate variability, the problem of upscaling information across different spatial and temporal scales, monitoring and evaluation, and understanding the potential response of GDEs to changes in the groundwater environment. Note that this updated version of the toolbox replaces the term 'environmental water requirement' with 'ecological water requirement' to better reflect the focus on ecological assets rather than the broader environmental setting.

The most significant change to the original (2007) toolbox is the format in which all approaches, tools and case studies are presented; these are now presented in the context of a revised assessment framework which aligns the tool(s) and approach(es) with a series of key questions that require answers in order to identify GDEs and arrive at appropriate ecological water requirements.

1.2. Ecological Water Requirements of GDEs

In 1994 the Council of Australian Governments (COAG) endorsed reforms to move towards a sustainable water industry that included allocations for the environment and greater environmental accountability of water resource developments. In line with these reforms, the Agriculture and Resource Management Council of Australia and New Zealand (ARMCANZ) and the Australia and New Zealand Environment and Conservation Council (ANZECC) produced the National Principles for the Provision of Water for Ecosystems in 1996. These principles provide the basis for considering EWRs as part of water allocation decisions by water resource managers.

The ARMCANZ/ANZECC principles stipulate that environmental water requirements should be based on the best available scientific information and that estimates of the water regimes required by an ecosystem are to be developed through strategic scientific research. Determining ecological water requirements for an ecosystem involves identifying those

aspects of the natural water regime that are most important for the persistence of critical ecosystem features (i.e. ecological structure) and processes (i.e. ecological function).

EWRs include elements of quantity and quality which apply both spatially and temporally, and should define the intrinsic requirement an ecosystem (or ecological component) has for water. In the context of GDEs this is the requirement for groundwater, which may be considered in terms of flow, level (or depth to watertable), pressure and quality (Clifton & Evans 2001, Cook et al. 2004).

An EWR is related only to the ecosystem's intrinsic need and/or association with groundwater (including that of the ecological components and processes within it) and should not be constrained or defined by external factors, such as focused management objectives that incorporate assigned values or priorities unrelated to the water needs and ecological health of the ecosystem. EWR investigations should therefore focus on establishing clear study objectives and hypotheses related to the potential for ecosystems to be dependent on groundwater and the extent or nature of that dependence. The scientific knowledge of EWRs, determined through such targeted investigations, informs the identification of management objectives and the water required to meet those objectives. The ecological water requirements then contribute, along with socioeconomic evaluation and water consumption demands, to the development of Ecological Water Provisions (EWPs) within water management plans. Therefore EWRs are not subject to revision as management objectives, values and priorities change, but only as further scientific information regarding an ecosystem's requirements for water become available and the EWR can be defined more accurately.

The use of independent technical investigations to develop EWRs promotes confidence that the assessments are based on sound science. This is necessary in order to support those decisions made around the management of GDEs. This GDE toolbox seeks to provide updated information on the sound scientific approaches available in the identification of GDEs and EWR investigations.

1.3. Translating EWRs into provisions

The National Water Initiative requires water plans to recognise the environmental and ecological values of water, as does most state- and territory-based legislation (these requirements are summarised in detail in Tomlinson 2011).

Ecological water provisions (EWPs) are a management tool used to achieve agreed ecological objectives for key assets. Ecological objectives are statements of the desired future condition of particular ecological assets within a management area and are often expressed in terms of a target to maintain, restore or rehabilitate.

The setting of ecological objectives takes into account the current condition of particular ecological assets and the social and economic implications of resource use. EWPs are then the articulation of ecological objectives as 'operational rules' related to groundwater use.

The water regime (groundwater and other sources) required to meet these objectives can be tested via an understanding of how the ecosystem state may change under a range of scenarios for the delivery of groundwater (and other sources). In some cases this scenario testing will be based upon quantified relationships, but is more likely to be based on expert opinion—in all cases supported by a targeted and ongoing monitoring program. The EWP is ultimately a reflection of the balance between the provision of water for consumptive users and the environment. It is therefore critical that the process involves a robust discussion amongst all stakeholders during the determination of abstraction limits for aquifers that is

based on agreed resource conditions limits and agreed and acceptable impacts (Richardson et al. 2011). Each jurisdiction has an approach to how this balance is achieved (refer Tomlinson 2011).

EWPs for GDEs have been delivered through various policy mechanisms in groundwater management plans across Australia, including provision based on:

- share of the water balance (e.g. percentage of recharge or baseflow)
- resource condition limits (e.g. seasonal groundwater levels, groundwater salinity, or rates of groundwater discharge)
- buffer distances between abstraction and ecological assets
- monitoring and evaluation regime (linked to an adaptive management framework).

The EWP (as far as practicable) should be expressed with measureable indicators that are spatially and temporally explicit. In all cases it is necessary that the expression of EWPs within a management plan is supported by a targeted monitoring program which has been designed to test whether the EWPs will meet the agreed ecological objective.

The determination of an EWP will also depend partly on the level of risk that is accepted by stakeholders in setting EWPs. The level of risk taken is related to the uncertainty associated with the natural temporal variability in groundwater conditions due to factors such as variation in annual recharge, stream flow and rates of groundwater extraction.

More cautious decisions on how to protect a GDE may be taken where there is a larger degree of uncertainty (e.g. large range in temporal variability) and therefore greater risk that the environmental groundwater cannot be provided as required. If data is available, and if past function is considered to be representative of future behaviour, it is possible to use historic data to better understand temporal variability and present the EWP (or the key attributes such as flux or level) as a probability distribution (exceedence curve) to show how much risk may be generated from temporal variability.

1.4. Toolbox structure

This Australian GDE toolbox is structured to provide an intuitive reference document. It is presented in two parts, namely *part 1: assessment framework* and *part 2: assessment tool*s. Within part 1, individual sections of the document are structured to align with the revised assessment framework. The tools presented within part 2 are referenced throughout this assessment framework in numerical order; for example, tool 1 is referenced as ‘(T1)’.

Section 2 of part 1 provides a summary of the GDE typologies that are recommended for use in GDE assessments. Section 3 presents the GDE assessment framework, which outlines a hierarchical structure of GDE assessment with three stages. Each assessment stage is discussed in terms of answering ‘key questions’ and outlines the associated ‘approaches’ and ‘tools’ available to do so. A discussion on approaches for data-poor areas is also presented. Specific issues relating to GDE assessments and practical approaches to overcoming them are discussed.

The remaining sections (4 to 7) focus on the specific issues around implementation of the GDE assessment framework. The importance and the benefits of conceptualisation of GDEs is an important theme running through all levels of GDE assessment. Conceptualisation is discussed specifically in Section 4.

Section 5 gives a detailed, step-by-step guide to developing an effective monitoring program to answer the key questions outlined in the GDE assessment framework and provide long-term data for the adaptive management of GDEs.

Section 6 focuses on situations where it may be necessary to extrapolate from existing information on EWRs from better-studied areas.

Section 7 addresses the challenges presented by change. It discusses the implications of climatic impacts alongside those of anthropogenic actions, and provides information on their assessment and practical responses.

A series of additional case studies detailing the types of studies undertaken in GDEs across Australia is presented in Appendix A. These case studies provide further information relating to issues raised around the implementation of the framework including design of monitoring programs and understanding GDE responses to change. A glossary of terms used throughout the toolbox is provided in Appendix B.

2. Types of GDEs

2.1. GDE typology

Eamus et al. (2006) classify different types of GDE on a functional basis. The approach moves from viewing GDEs simply as an ecosystem expression (e.g. wetlands, baseflow or terrestrial vegetation) to providing a more holistic view of the ecohydrogeologic system and the ability to conceptualise the role of groundwater in maintaining biodiversity and ecological condition (Nevill et al. 2010). This typology provides the necessary flexibility to consider individual components (and the temporal and spatial variability of those components) within the broader ecosystem landscape.

On the basis of the typology developed by Eamus et al. (2006) and Eamus (2009), it is recommended that the following three classes of GDEs be adopted:

- **Aquifer and cave ecosystems (Type 1)** provide unique habitats for living organisms (e.g. stygofauna and troglofauna). These ecosystems typically include karst aquifer systems (such as Mole Creek in Tasmania, Figure 2-1), fractured rock and saturated (consolidated and unconsolidated) sedimentary environments. The hyporheic zones of rivers, floodplains and coastal environments are also included in Type 1.
  The deep subsurface groundwater environment provides relatively stable, lightless environmental conditions with restricted inputs of energy and low productivity which allows a particular suite of subsurface ecosystems to prosper. The ecological diversity is created from variable geology, oxygen, carbon and nutrient gradients (linked to the dynamics of water flow) and physico-chemical conditions. Subsurface ecosystems provide an important supporting service of bioremediation of contaminated groundwater, and provide an important role in carbon and nutrient cycling. The occurrence of stygofauna in Australia (and Europe) is described by Tomlinson and Boulton (2010).
- **Ecosystems dependent on the surface expression of groundwater (Type 2)** include wetlands, lakes, seeps, springs, river baseflow, coastal areas and estuaries that constitute brackish water and marine ecosystems. In these cases, the groundwater extends above the earth surface, as a visible expression. Examples include the mound springs of the Great Artesian Basin, Figure 2-2), and wetlands in the south-eastern part of South Australia (SKM 2010a; prepared for the Department for Water, SA). In these situations groundwater provides water to support aquatic biodiversity by providing access to habitat (especially when surface runoff is low) and regulation of water chemistry and temperature.
- **Ecosystems dependent on subsurface presence of groundwater (Type 3)** (via the capillary fringe) include terrestrial vegetation that depends on groundwater fully or on a seasonal or episodic basis in order to prevent water stress and generally avoid adverse impacts to their condition. In these cases, and unlike the situation with Type 2 systems, groundwater is not visible from the earth surface. These types of ecosystem can exist wherever the watertable is within the root zone of the plants, either permanently or episodically.

Best studied examples of this type of GDE in Australia are the Banksia woodlands on the Swan Coastal Plain of south-western Western Australia (Sommer & Froend 2010); Monsoon Vine Ticket vegetation in the Northern Territory (Liddle et al. 2008, Figure 2-3); and Pioneer Valley in Queensland (O'Grady et al. 2006). An example conceptual diagram for this type of GDE (Lexia wetland on the Swan Coastal Plain) is presented in

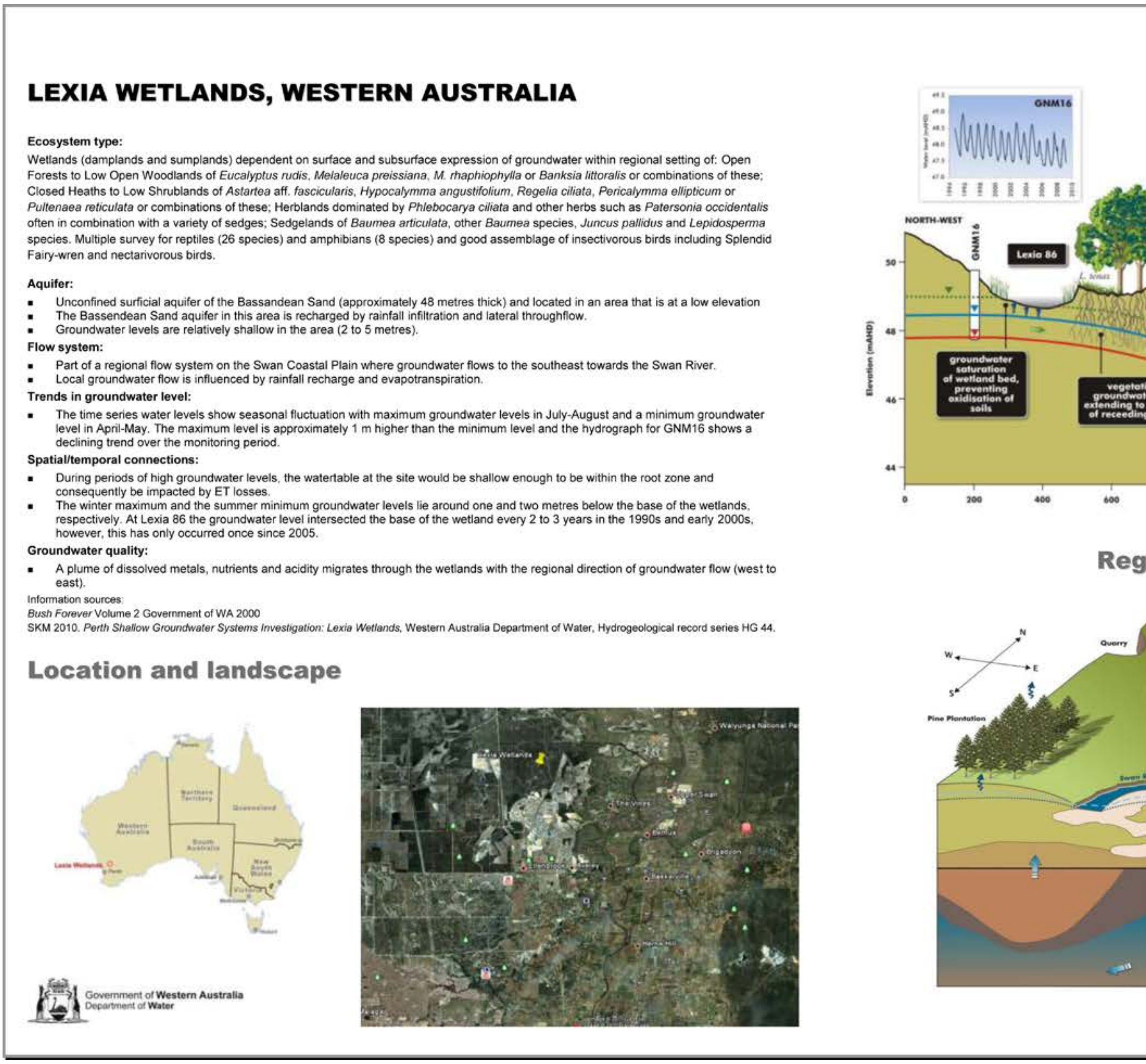


Figure 2-4.

MOLE CREEK KARST, TASMANIAN CAVE GDE

Ecosystem type:

Cave ecosystems supported by karst hydrological systems are composed of surface and underground streams, and percolation (seepage) water flowing through the soil and rock under gravity with natural processes control by flows of water and air, along with what is dissolved or transported into the cave system. Caves support a variety of fauna including flatworms and springtails, annclids, myriapods and molluscs, crustaceans, arachnids and insects. Most cave dwelling fauna is accidental but many species are troglobitic. Groundwater provides habitats within the cave system for aquatic species and can also provide a water source for any surface vegetation. Other ecosystem services provided by groundwater include moderation of the cave environment (humidity and temperature) to conditions suitable for many of the sensitive cave dwelling fauna.

Aquifer:

- In the Mole creek system shown, the cave is hosted within the karst aquifer of the Ordovician Gordon Limestone. Groundwater in this system is fracture and conduit hosted. Vadose and phreatic flow within the cave system can be up to 5 km per day.

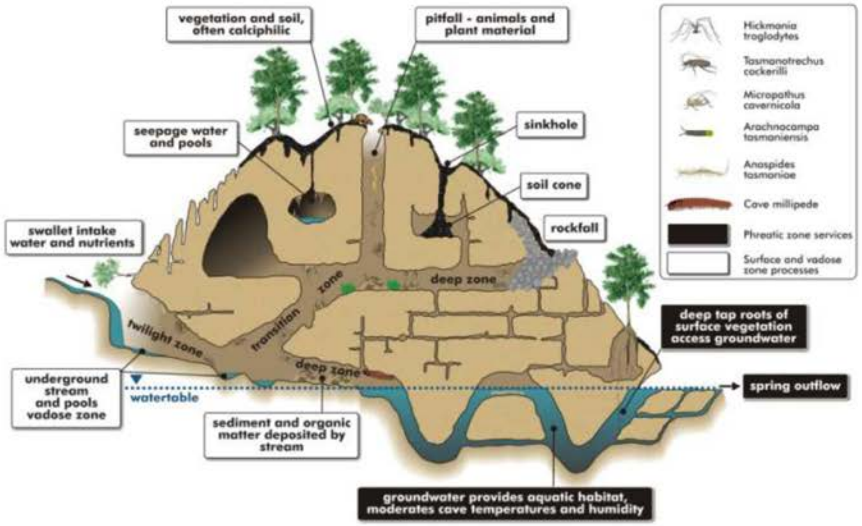
Trends in groundwater level:

- Groundwater levels within the Mole Creek cave system vary seasonally and in response to rainfall and surface water inflow events. The resulting *epiphreatic zone* (areas that are alternately saturated and non saturated within the cave) can be several meters thick.

Information sources:

- Burrett C and Geode A (1987) Mole Creek – A geological and geomorphological field guide. Geological Society of Australia (Tasmanian Division) Guidebook 1
- Kiernan K (1989) Karst, caves and management at Mole Creek, Tasmania. Department of Parks Wildlife and Heritage. Occasional Paper No. 22.
- Conceptual diagram figure adapted from Eberhard S and Huphreys WF (2003) The crawling, creeping and swimming life of caves in 'Beneath the surface: a natural history of Australian caves' (Eds. Finlayson B and Hamilton Smith). University of New South Wales Press Ltd
- Regional cross section adapted from Jennings, I.B. (1963). Geological Survey Explanatory Notes. Middlesex. Tasmanian Department of Mines. K 55-6-45.

Karst Ecohydrogeology



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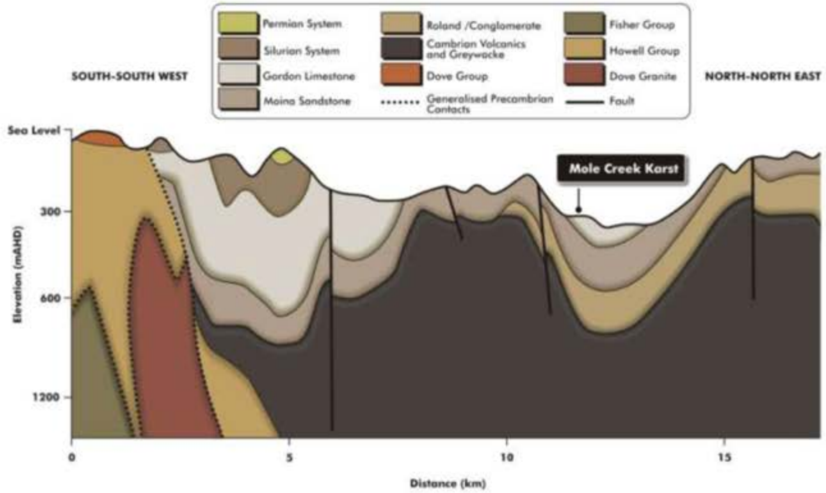


Location and landscape

Figure 2-1: Conceptual diagram of a karst GDE, an example of an ecosystem dependent on saturated aquifers or caves (Type 1)

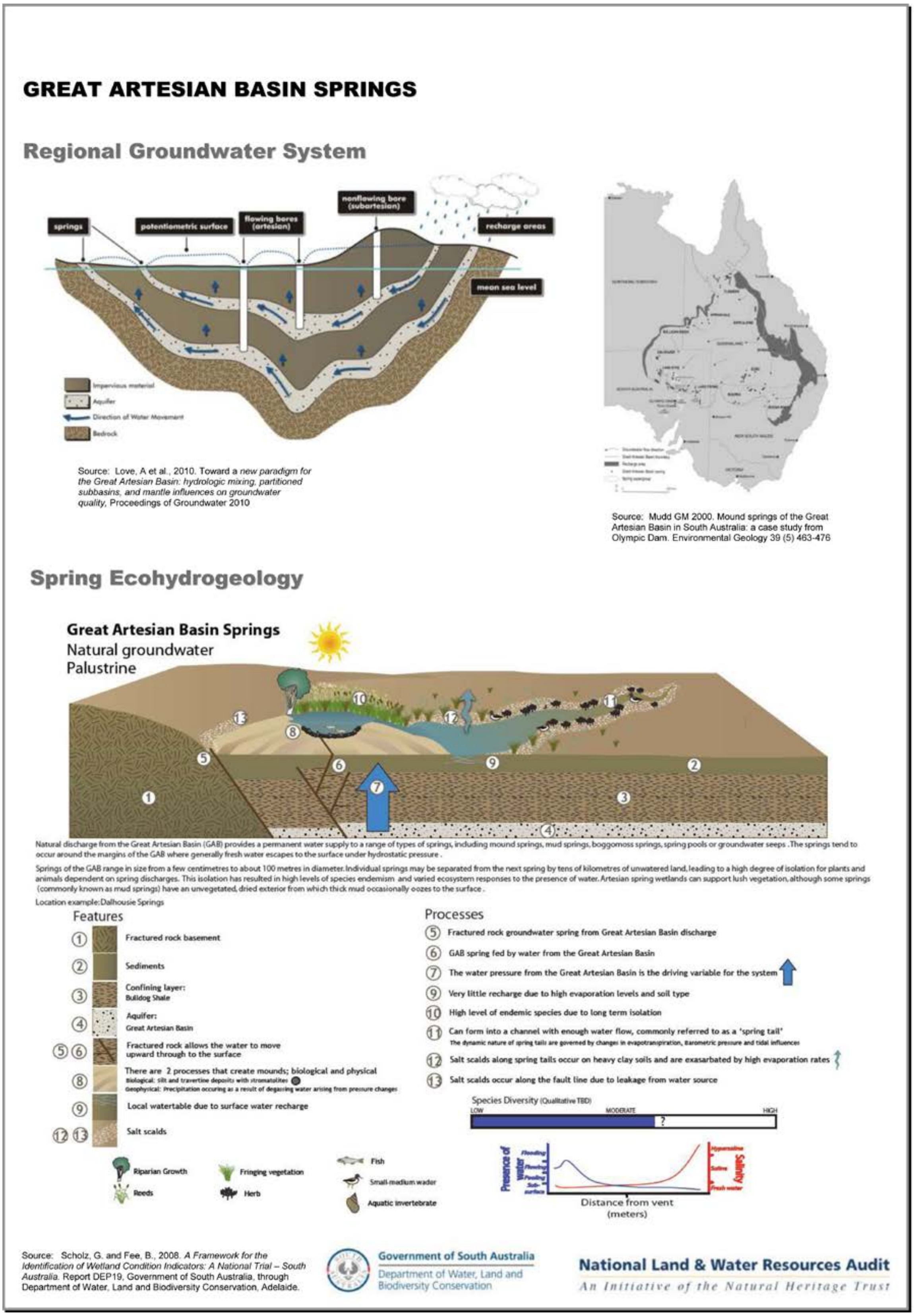


Figure 2-2: Conceptual diagram of mound spring GDEs of the Great Artesian Basin, an example of an ecosystem dependent on the surface expression of groundwater (Type 2)

MONSOON VINE THICKET

Ecosystem description

The Northern Territory wet monsoon vine thickets (MVTs) are spring fed rainforests dependent on both surface and subsurface expression of groundwater. MVTs typically occur in topographic lows in discrete 'patches' (usually less than 5 ha), quite often within Eucalypt savannah landscape. They contain a diverse range of plant life including numerous trees and shrubs (*Terminalia macrocarpa, Syzygium nervosum, Diospyros calycantha, Strychnos lucida, Barringtonia acutangula subsp. acutangula, Buchanania arborescens, Vavaea australiana, Ixora timorensis, Leea indica*), palms (*Livistona benthamii, Carpentaria acuminate*) and vines (*Flagellaria indica, Ichnocarpus frutescens*). VTs provide habitat for a range of mobile fauna, including frugivores such as the pied imperial pigeon, rose crowned fruit-dove, figbird, yellow oriole, common koel, great bowerbird and the black flying fox.

Hydrogeological setting:

The geology of the greater Darwin area comprises Proterozoic metasedimentary basement unconformably overlain by Cretaceous sandstones and claystones (and intermediate variants) of the Bathurst Formation. Cainzoic cover is extensive and can be divided into two main groups including Tertiary weathering products or regolith and Quaternary sediments.

There are three main aquifers namely the karstic Proterozoic Koolpinyah Dolomite, the basal sandstone unit of the Bathurst Formation and upper lateritised profile (top 30 m) of the Bathurst Island Formation (Darwin Member). There is a high degree of interconnection between the aquifers with similar seasonal fluctuations in groundwater level observed. Wet season recharge causes rapid, large magnitude groundwater level rises (up to 15 m), followed by dry season recession due to natural discharge and groundwater pumping. The amplitude of the water level response to recharge can however vary at individual sites. At some points, groundwater pressures in the dolomite exceed those of the overlying laterite, reflecting semi-confined conditions at these locations.

Groundwater use and dependency:

In the greater Darwin area, MVTs are associated with topographic lows where watertable is close to, or intersects, the ground surface. Surface expressions of groundwater as springs and seeps are common and spring-fed baseflow contributes to drainage lines that extend downstream of these features. Often, springflow (localised discharge) emerges from semi-confined Koolpinyah Dolomite aquifer via sinkholes where collapsed rock provides a pathway for the deep groundwater to flow through the units to the surface.

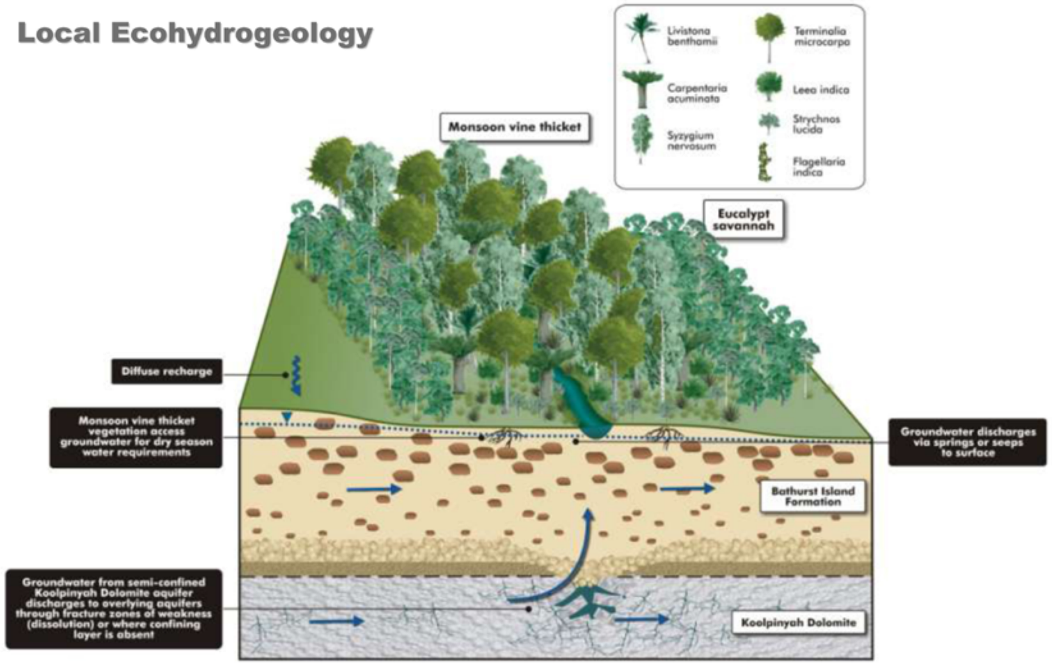
Groundwater contributions to the MVT environment are most critical during the dry season. Discharge from both the laterite and dolomite aquifers contribute to baseflow, providing a water source for MVT flora and fauna. Groundwater supplies up to up to 50 % of vegetation water use during the dry season.

Information sources:

Liddle, D.T., Boggs, D., Hutley, L., Yin Foo, D., Boggs, G., Pearson, D., Cook, P.G. and Elliott, L.P. (2008) *Biophysical modelling of water quality in a Darwin rural area groundwater dependent ecosystem.* Report to Natural Resource Management Board (NT), NHT Project 2005/133. Northern Territory Government Department of Natural Resources, Environment, The Arts and Sport, Palmerston

CSIRO (2009) *Water in the Van Diemen region*, pp 363-452 in CSIRO (2009) Water in the Timor Sea Drainage Division. A report to the Australian Government from the CSIRO Northern Australia Sustainable Yields Project. CSIRO Water for a Healthy Country Flagship, Australia. xl + 508pp

Local Ecohydrogeology



Regional Geology

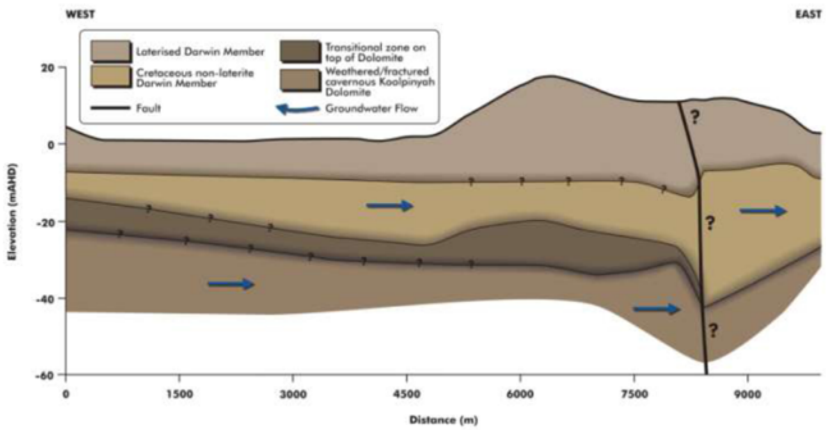


Figure 2-3: Conceptual understanding of Monsoon Vine Thicket ecosystems found in the Northern Territory. These ecosystems are an example of both Type 2 and Type 3 GDEs

LEXIA WETLANDS, WESTERN AUSTRALIA

Ecosystem type:

Wetlands (damplands and sumplands) dependent on surface and subsurface expression of groundwater within regional setting of: Open Forests to Low Open Woodlands of *Eucalyptus rudis*, *Melaleuca preissiana*, *M. rhaphiophylla* or *Banksia littoralis* or combinations of these; Closed Heaths to Low Shrublands of *Astartea* aff. *fascicularis*, *Hypocalymma angustifolium*, *Regelia ciliata*, *Pericalymma ellipticum* or *Pultenaea reticulata* or combinations of these; Herblands dominated by *Phlebocarya ciliata* and other herbs such as *Patersonia occidentalis* often in combination with a variety of sedges; Sedgelands of *Baumea articulata*, other *Baumea* species, *Juncus pallidus* and *Lepidosperma* species. Multiple survey for reptiles (26 species) and amphibians (8 species) and good assemblage of insectivorous birds including Splendid Fairy-wren and nectarivorous birds.

Aquifer:

- Unconfined surficial aquifer of the Bassandean Sand (approximately 48 metres thick) and located in an area that is at a low elevation
- The Bassendean Sand aquifer in this area is recharged by rainfall infiltration and lateral throughflow.
- Groundwater levels are relatively shallow in the area (2 to 5 metres).

Flow system:

- Part of a regional flow system on the Swan Coastal Plain where groundwater flows to the southeast towards the Swan River.
- Local groundwater flow is influenced by rainfall recharge and evapotranspiration.

Trends in groundwater level:

- The time series water levels show seasonal fluctuation with maximum groundwater levels in July-August and a minimum groundwater level in April-May. The maximum level is approximately 1 m higher than the minimum level and the hydrograph for GNM16 shows a declining trend over the monitoring period.

Spatial/temporal connections:

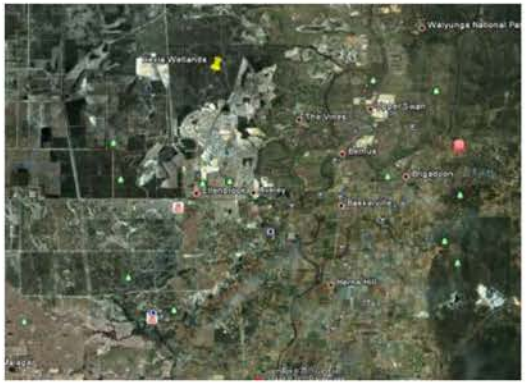
- During periods of high groundwater levels, the watertable at the site would be shallow enough to be within the root zone and consequently be impacted by ET losses.
- The winter maximum and the summer minimum groundwater levels lie around one and two metres below the base of the wetlands, respectively. At Lexia 86 the groundwater level intersected the base of the wetland every 2 to 3 years in the 1990s and early 2000s, however, this has only occurred once since 2005.

Groundwater quality:

- A plume of dissolved metals, nutrients and acidity migrates through the wetlands with the regional direction of groundwater flow (west to east).

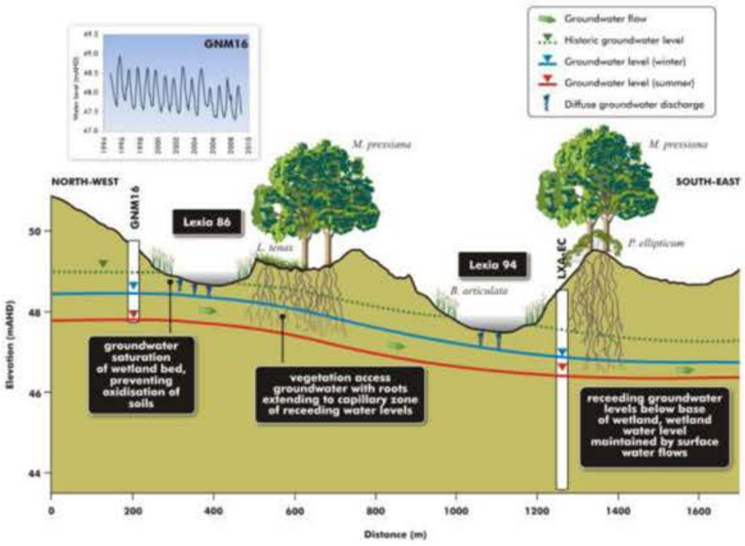
Information sources:
Bush Forever Volume 2 Government of WA 2000
SKM 2010. *Perth Shallow Groundwater Systems Investigation: Lexia Wetlands*, Western Australia Department of Water, Hydrogeological record series HG 44.

Location and landscape



Government of **Western Australia**
Department of **Water**

Local Ecohydrogeology



Regional Groundwater System

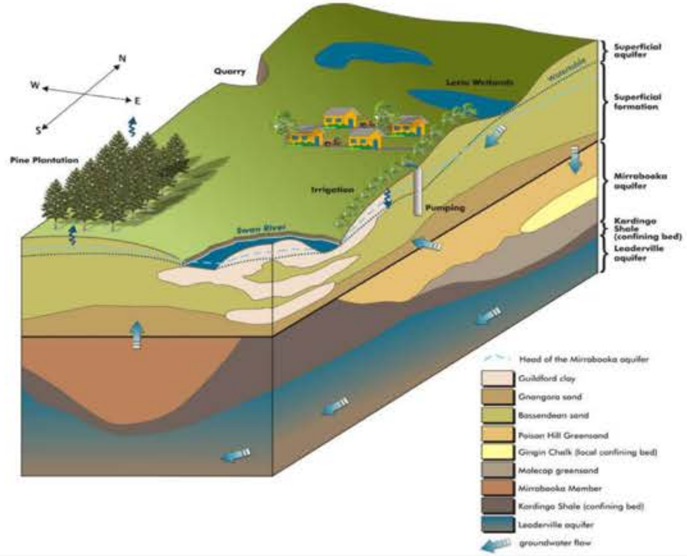


Figure 2-4: Relationship of vegetation to the groundwater system, Lexia Wetlands, Western Australia, an example of ecosystem dependent on the subsurface presence of groundwater (Type 3)

A summary overview of these GDE types (with supporting references) is provided by Nevill et al. (2010).

An assessment of GDEs should consider the broader ecohydrogeological setting and the connections that can potentially occur between different components of the system. The connections between the three GDE types outlined above have been represented schematically by Eamus et al. (2006, as referenced by Nevill et al. 2010, Figure 2-5).

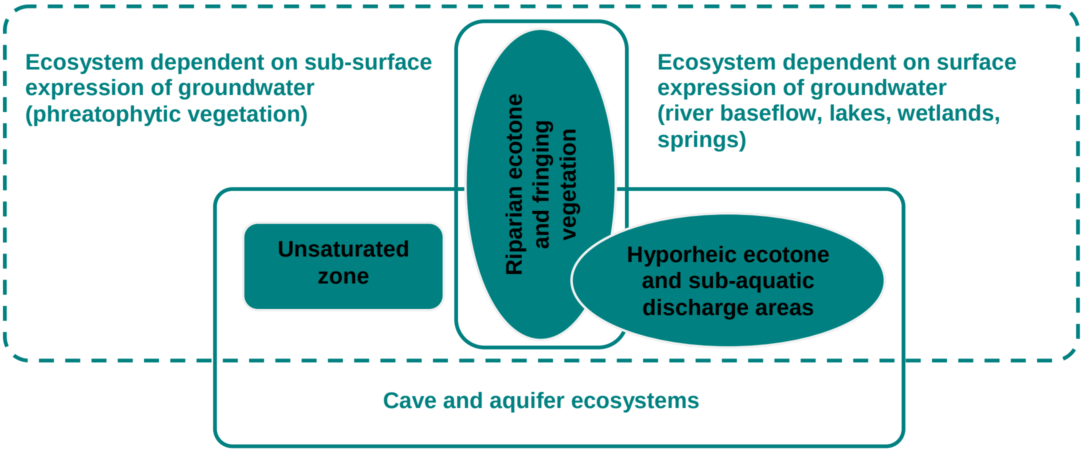


Figure 2-5: Links between the three types of groundwater-dependent ecosystems (modified after Nevill et al. 2010)

2.2. Perched aquifer GDEs

2.2.1. What are perched aquifers?

Ecosystems are sometimes dependent on perched aquifers, which can provide a source of water as a surface or subsurface expression of groundwater. Recognition that a groundwater body is potentially perched is necessary for appropriate groundwater management as groundwater flow pathways in a perched system may differ significantly from those in the underlying flow systems (Carter et al. 2011).

The conceptualisation of a perched aquifer setting requires careful consideration (and investigation) of the potential hydraulic connection between a perched aquifer and an underlying aquifer, especially when the underlying aquifer may be affected by groundwater abstraction. The implication of a GDE being potentially dependent on a perched aquifer is that it may be buffered from impacts of groundwater abstraction from underlying aquifers. It is also important to recognise, and correctly conceptualise, situations where abstraction occurs from a perched aquifer. In these cases, impacts to any associated GDE(s) may be significant, even if the amount of abstraction is relatively small.

Meinzer (1923, cited by Carter et al. 2011) defined a perched aquifer as:

> *water that is separated from an underlying body of groundwater by unsaturated porous material.*

The National Water Commission defines a perched aquifer as:

> *a region in the unsaturated zone where the soil or rock may be locally saturated because it overlies a low-permeability unit.*[1]

Perched aquifers tend to be quite localised in extent and can exist as seasonal or permanent features. A perched aquifer is typically unconfined in nature and can form at quite shallow depths in the unsaturated zone. Perched water can move laterally over the low permeability layer and discharge at the surface in the form of seeps.

Water percolating downward (recharge) though the unsaturated zone can be intercepted and accumulate on top of the low permeability layer, when the rate of recharge is greater than the infiltration rate through the low permeability layer. The low permeability layer responsible for perching can form within alluvial or lacustrine environments associated with relic stream channels. The complex layering in these environments means that it can be difficult to identify discrete perching layers during drilling investigations.

The material above a low permeability layer becomes saturated but separated from an underlying more extensive aquifer system. Such perched systems can be 'disconnected' from an underlying more extensive aquifer (although downward leakage will occur), and as such are often considered to be separate.

Based on these definitions a shallow aquifer sitting on a low permeability layer that is hydraulically connected to an underlying more extensive aquifer by a saturated zone is not considered a perched aquifer (Figure 2-6).

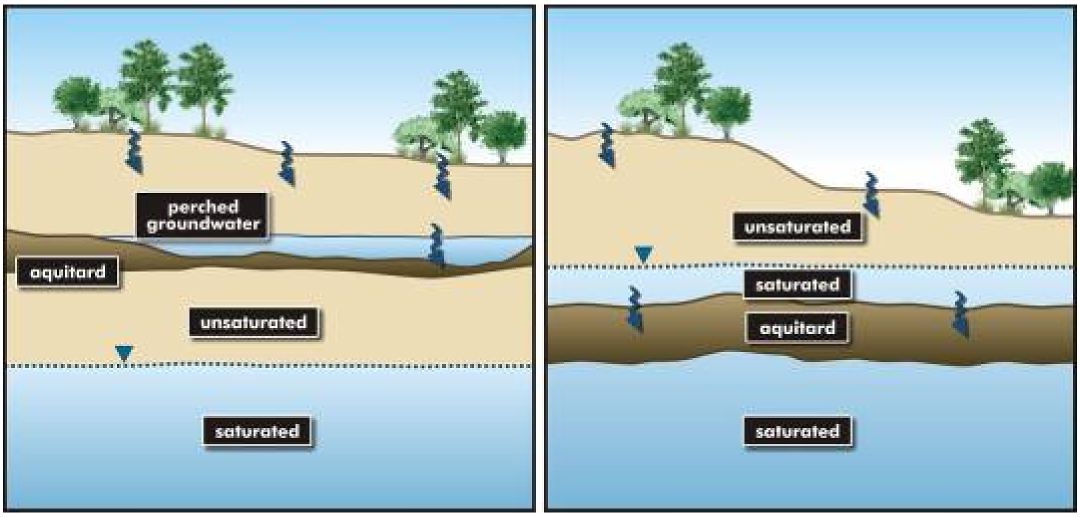


Figure 2-6 (left) perched groundwater system, (right) non-perched groundwater system

For the purpose of investigation of potential GDEs, the critical issues are whether there is hydraulic connection between a perched aquifer and the ecosystem (such as a wetland), and the degree of hydraulic connectivity between a perched aquifer and an underlying more extensive aquifer.

Examples of perched aquifers include those identified at Tangletoe and Lake Muckenburra on the Swan Coastal Plain in Western Australia.[2]

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1 *http://dictionary.nwc.gov.au/water_dictionary/item.cfm?id=395&cRefer=1&sRefer=40*
2 *http://www.water.wa.gov.au/PublicationStore/first/98975.pdf*

2.2.2. Investigation

Investigation programs designed to determine the likely hydraulic connection between a perched aquifer and underlying aquifer require careful planning and implementation. The objective of an investigation program is to demonstrate that:

- saturated conditions exist above a low permeability contrast
- a significant unsaturated zone exists at some point between the perched layer and the underlying aquifer. (Note that thin or narrow unsaturated zones may still provide hydraulic connection.)

Core recovery techniques can provide well defined lithology necessary for the identification of perched layers or layers that can potentially host a perched aquifer. Recovery methods must allow for characterisation of substrate moisture thus defining whether any underlying low permeability unit is saturated or unsaturated and allows hydraulic connections between the potentially perched and underlying aquifer. Drilling methods such as mud-rotary may not clearly identify the presence of a perched aquifer; however, air rotary, rotasonic, pushtubing and cable-tool methods are likely to be more successful. Collection of soil samples for hydraulic conductivity testing, soil moisture, soil suction and soil chloride analyses will be necessary for conceptualisation.

Observation wells should be constructed to target both the potentially (discreet) perched aquifers and underlying regional aquifer. Separate wells should be placed in each aquifer and not one well with multiple screens. Screens or slotted intervals should be placed so that the location of the phreatic – unsaturated zone interface (in the case of a potentially perched aquifer) and the watertable (in the case of underlying aquifers) can be measured within each well. An observation well completed immediately below the perched layer will also determine whether saturated or unsaturated conditions exist. Wells with faulty construction or improper design can potentially establish hydraulic connection between perched and underlying aquifers by short-circuiting low-permeability layers. This can result in the collection of ambiguous data.

Downhole geophysical techniques (such as downhole radar and electromagnetics) may also be useful in looking at variations in clay content and moisture within the profile.

Perched aquifers often occur only seasonally and so the collection of time-series groundwater-level data is required on (at least) at weekly to monthly intervals, depending on the timing of recharge events.

The differentiation of a perched water source from the underlying aquifer can be supported with the use of isotopic and geochemical methods (T11).

2.3. Karst and psuedokarst features

Karst landscapes are produced primarily through dissolution of rocks in natural waters (Ford & Williams 2007). Karst features occur in a wide range of landscapes in Australia, including the arid Nullarbor Plain, temperate Tasmania (e.g. Figure 2-1) and the seasonally wet tropics (e.g. Chillagoe, Figure 2-7). They are found in a wide range of geologies with soluble rocks, generally carbonates (although extensive karst systems in sulfates and halites are found elsewhere). Silicate karst (generally, but not exclusively found in sandstones and quartzites of the wet tropics) has recently been mapped in Australia, although the proportion of solution to mechanical erosion has yet to be established for many systems.

Pseudokarst landscapes possess similar landforms to karst (e.g. sinkholes, streamsinks, springs and cave systems) but are not primarily developed through the dissolution of rock. Australian examples are found in lava, duricrusts (e.g. silcretes, ferricretes), unconsolidated sediments or volcanic ash, talus and ice. They are particularly well developed in basalt flows (e.g. the western Victorian lava plains and Undara in Northern Queensland). Many extensive sea-caves and associated surface landforms are found around the coast (Finlayson & Hamilton-Smith 2003).

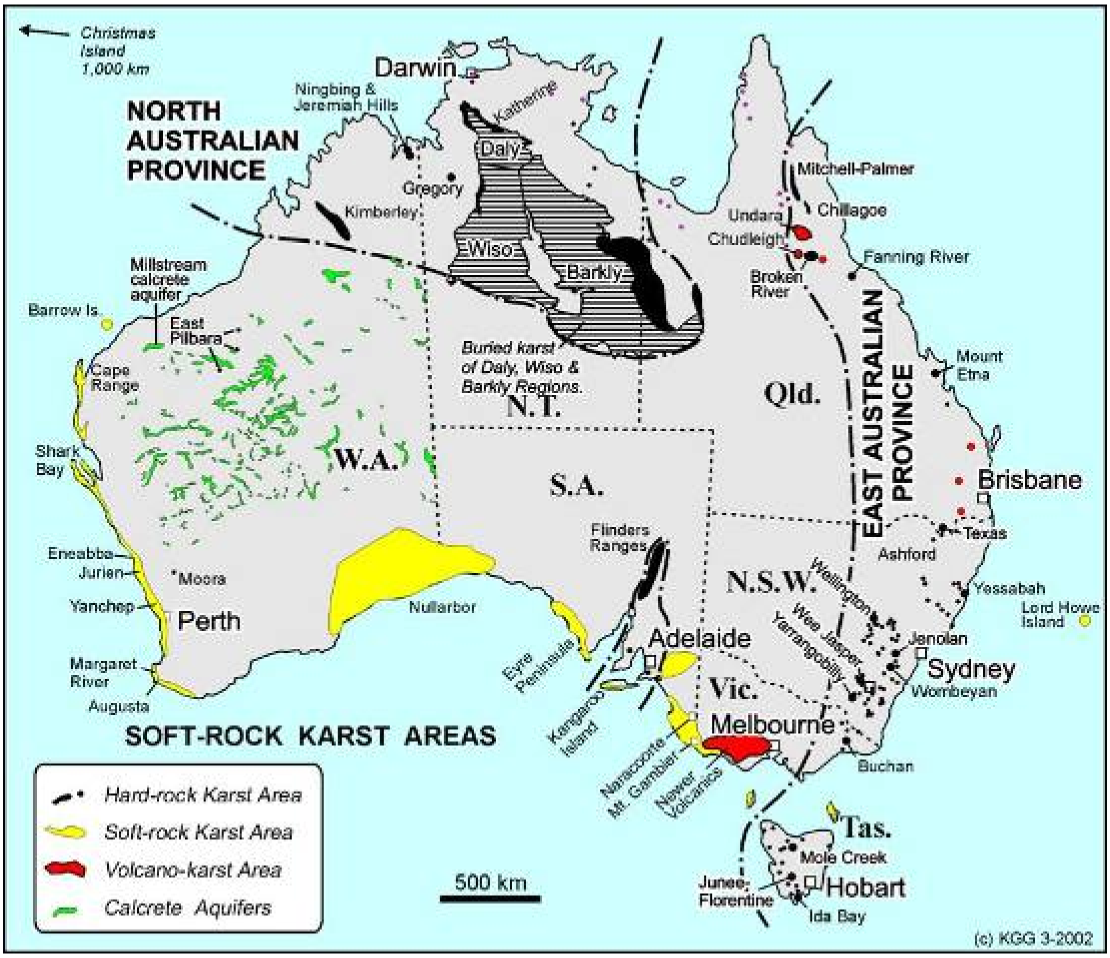


Figure 2-7 Karst locations any types across Australia (from Hamilton-Smith 2003)

The large variation in the types of karst and pseudokarst means that there are also many ways to investigate their occurrence and interaction with subsurface water. Karst and pseudokarst environments can be characterised using remote sensing (particularly LiDAR where available) and aerial photography, as well as various surface (e.g. ground-penetrating radar, resistivity and seismic refraction) and subsurface (e.g. borehole geophysics) techniques for regional scale assessments. Use of natural or introduced tracers (including rhodamine, fluorescein, lycopodium spores or various isotopes) enables determination of flowpaths, transmission times and connectivity. On-site observation and mapping (particularly of accessible cave systems) remains the most effective method for local-site scale assessments. This may be aided by core drilling; however, unintentional transfer of water between conduit systems should be avoided (for more detail refer *http://www.technos-inc.com/pdf/karst-technotes.pdf*).

Karst and pseudokarst ecosystems contain highly connected surface, vadose and phreatic components, each requiring specialist skills to document ecological communities, geomorphic values and their water needs. In surface karst, dolines often act as refugia from

fires or extreme climate and may contain relict vegetation communities. Underground, ecosystems are strongly zonal, comprising the cave entrance, twilight, transition and deep zones (Eberhard & Humphries 2003 in Finlayson & Hamilton-Smith 2003). Caves are commonly used by various bat species, as well as other vertebrates (mammals, birds and fish) and invertebrates (both terrestrial and aquatic). In contrast to other systems, each of these broad zones is often humanly accessible for assessment and sampling of species and water quality (although a significant proportion of cave life is now known to inhabit inaccessible meso- and micro-caverns).

Whilst maintenance of connected surface habitats is generally critical for management of underground ecosystems (through maintenance of food supplies and water quality and quantity), many systems rely on food production through subsurface chemical processes to fuel biological processes (chemautotrophic systems). Understanding the balance between surface and underground inputs, and the need for water to support key processes underpins effective management of karst ecosystems.

Preliminary surveys should document major biological communities, important geomorphic values and biological species, nutrient and sediment fluxes, and the presence of pollutants or erosion/sedimentation issues. These should be undertaken in each major surface and subsurface zone. Whilst most standard methods for assessment of surface aquatic ecosystems may be used in surface karst, it is critical to understand the special characteristics of the karst hydrological systems supporting them. This will differ regionally; however, many surface karst streams and wetlands are strongly seasonal or ephemeral and react to large changes in watertable depth (e.g. drowned dolines, springs etc.). It is important to assess water requirements in both the vadose and phreatic zones in subsurface karst, as the unsaturated zone can be up to hundreds of metres thick in some areas, with important ecosystems present in subsurface vadose streams and pools.

Ecological water requirements for subsurface karst ecosystems are not well established; however, some general principles are likely to apply in most areas. Due to rapid transmission times, karst and pseudokarst ecosystems are poorly buffered against changes to water quantity and quality. Due to rapid connectivity, pollutants in surface water sources will be rapidly transferred to karst groundwater, and polluted groundwater subsequently transferred rapidly to surface ecosystems below major springs (e.g. Kiernan 1988). As subsurface environments (particularly in the deep zone) are highly stable, with almost constant humidity, temperature and water quality parameters, ecosystems have evolved with relatively restricted tolerance to change (Culver 1982). Establishment of EWRs for subsurface karst and pseudokarst systems should be undertaken in this light.

Further information regarding karst systems in Australia can be found within *Beneath the Surface – A Natural History of Australian Caves*, by Brian Finlayson and Elery Hamilton-Smith, 2003.

3. The GDE Assessment Framework

3.1. Overview

This assessment framework provides a pathway for technical investigations which seek to determine where GDEs exist within the landscape and provide the necessary technical information to establish the EWRs. The framework provides connections between stages of assessment, key questions, general approaches, and the tools (as provided in *Part 2 Assessment Tools*) that can assist in answering key questions (

Figure 3-1).

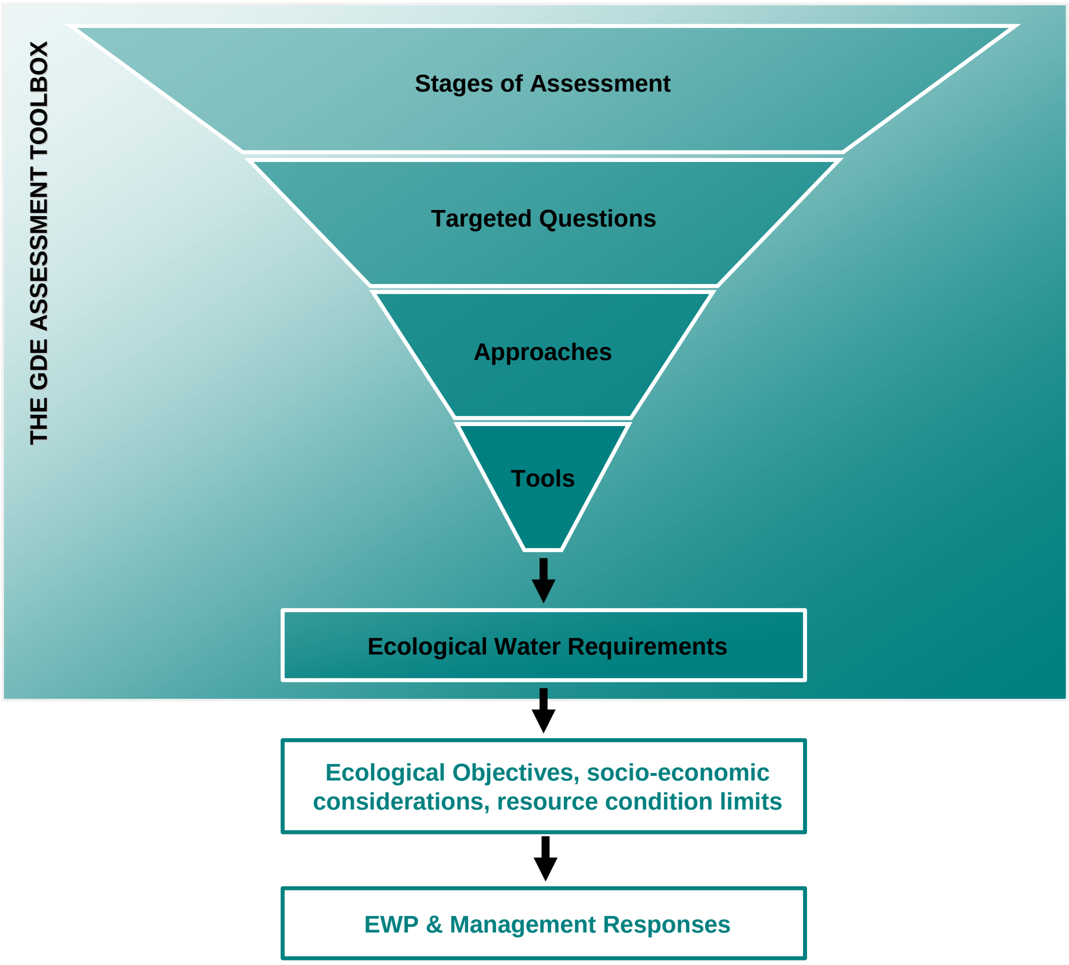


Figure 3-1 Hierarchical structure of the GDE Assessment Framework. The shaded area represents the components described within the GDE toolbox

3.2. Stages of assessment and the use of tools

The framework contains three stages of assessment which should be applied to determine EWRs for GDEs (Figure 3-2). The three stages are broadly consistent with the three levels of information contained within the Australian Atlas for groundwater-dependent ecosystems (GDE Atlas)[3]. There are a series of targeted questions for each stage of assessment. The

[3] Further information regarding this project is available on the National Water Commission website *http://www.nwc.gov.au/www/html/1054-atlas.asp?intSiteID=1*

nature of the key questions means that the complexity of approaches and data requirements needed to answer the questions increases through the assessment stages.

Key questions need to be considered at all stages of the framework in the process of deriving EWRs. The approach to answering the key questions is necessarily data-driven; however, it is recognised that a more hypothesis- and expert opinion-driven approach (supported by monitoring) is necessary in data-limited situations.

The level of confidence is higher where a data-driven approach is used. Before defining the scope of investigations of potential GDEs, effort and resources can be prioritised by conducting an initial risk assessment to identify the priority ecological sites or subregions. It may be that there will be some prior knowledge of the degree of groundwater development and the risk to the ecological and groundwater assets. An initial risk assessment can build on outcomes from Stage 1 to describe asset value, threats and uncertainty to create a relative ranking of priority of sites across the area of interest. In situations where it is determined that there is little risk to the assets or the resources being protected, use of comprehensive and detailed tools (e.g. T11 and T12) may not be warranted. Greater effort in investigation and analysis is justified where the combination of value, threat and uncertainty is greatest.

If this risk-based approach is taken it is critical that sufficient analysis and reporting is undertaken to justify the assignment of risk level. It is also important that the level of risk is revisited as additional information becomes available over time.

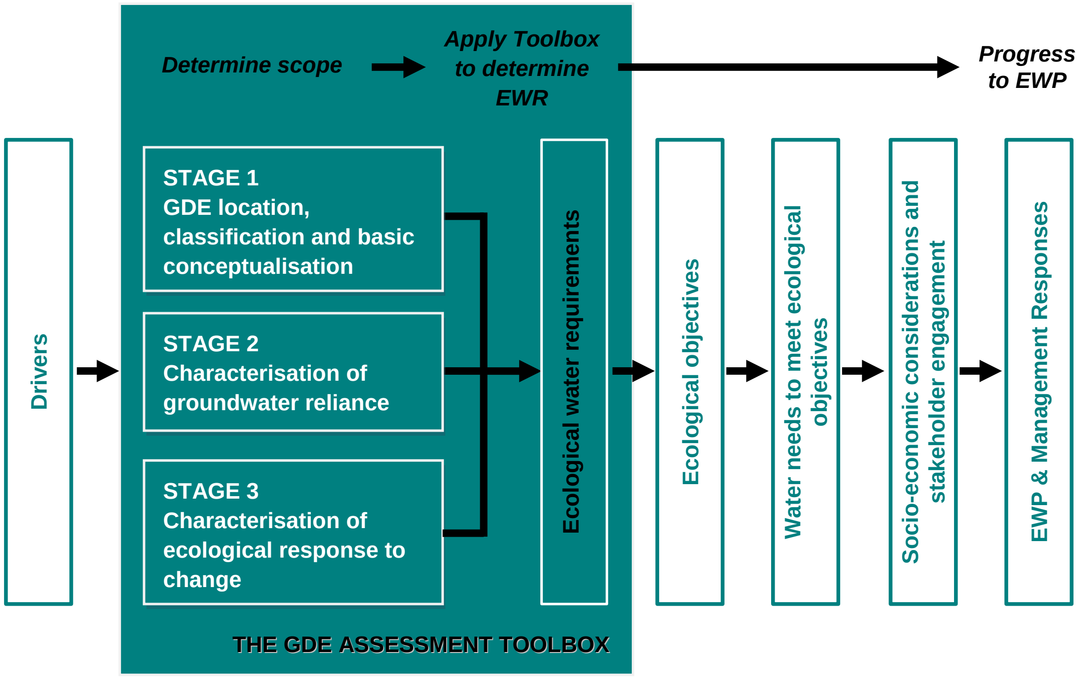


Figure 3-2: Three-stage assessment framework for determination of EWRs. The shaded area represents the steps in the process relevant to the GDE toolbox.

Assessments in Stage 1 focus on gaining a baseline understanding of where potential GDEs exist, classification of ecosystem type and conceptualisation of the ecohydrogeologic setting. Detail on approaches to conceptualisation can be found in Section 4 and also in T2. Stage 2 assessments build on this information to characterise the likely reliance of the ecological

asset on groundwater (e.g. describe timing of use of groundwater). Stage 3 involves creating a detailed and quantified understanding of how the biotic state of GDEs can change as abiotic (e.g. groundwater) conditions change. This Stage 3 assessment may not be achieved in a short time frame such as in the typical timeline for the preparation of a management plan or approvals process. It may be achieved only through analysis of monitoring data over the life of the plan or, more likely, over decades of research and monitoring.

Monitoring is the best approach to answering the key questions posed within Stages 2 and 3, and in the absence of monitoring data there is less confidence in the associated EWR estimates. Where available, monitoring data also allows more thorough treatment of Stage 1, but is arguably less crucial. Detail on the appropriate design of monitoring and evaluation programs is provided in Section 5.

Existing site-specific studies provide useful information on the location of GDEs; the ecohydrogeologic processes or components of a GDE; the ecosystem services they provide; and/or potential risks/threats that may be directly comparable to new investigations, particularly those undertaken *within* the area of interest. There may be a need to upscale this targeted information to a broader area such as a catchment, management area or groundwater flow system. Discussion and guidance on upscaling or transfer of EWR information is provided in Section 6.

3.3. Approach in data-limited regions

In many cases the required data and resources to answer key questions will be limited or not available in the decision-making time frame. Outputs are likely to be more qualitative but still provide the ‘best available science’ to inform management responses, albeit initially with a lower level of confidence.

Data-limited regions typically only have broad-scale information regarding the occurrence of ecosystems (e.g. vegetation mapping) and landscape datasets such as geology, soil type, climate and topography. There is likely to be a regional-scale view of the groundwater flow system but little groundwater data targeting potential GDEs. Often there is not enough available science to enable an understanding of the likely ecological consequences of the proposed management decision (Gardner & Bowmer 2007, as referenced by Ryder et al. 2010).

The approach in these cases will rely on a baseline assessment using regional datasets and existing information. The approach is likely to rely on suggestive and supplementary information (as described by Ryder et al. 2010) to answer the key questions. Suggestive information can be derived from observations of the relationships between the occurrence of ecosystems within the landscape and possible role of groundwater (e.g. occurrence of river pools in dry periods implies dependency on groundwater inflow). Supplementary information comes from engagement with a range of people with knowledge of the region or those that can extrapolate or transfer knowledge from other similar regions.

The hypotheses related to the key questions should be tested through targeted monitoring and evaluation programs with adjustment to the understanding of the EWR over time, as required.

In these cases it is important that the information and opinions used—along with the limitations, uncertainly and levels of confidence that result—are explicitly recognised by all decision-makers when determining management responses.

3.4. Using expert opinion

Investigations should be supplemented with knowledge from relevant practitioners/experts with experience in the region or in other similar regions. The approach can include both formal and less formal approaches. A more formal approach can require an 'expert panel' to be convened which provides critical comment in a review or audit capacity. Less formal approaches facilitate discussion amongst relevant practitioners and experts. In either case, experts take the best available science and create a conceptual model and a set of hypotheses related to the reliance of potential GDEs on groundwater. They can also provide opinion on a range of effects from changed groundwater conditions.

The composition of the forum should be considered in the context of the outputs required and the source of data; for example, local experts/naturalists who often know where groundwater discharges occur and where ecological condition is potentially linked to seasonal availability of groundwater. Indigenous knowledge is based on a long-standing connection with the environment and experience of this connection can provide unique information which links physical, spiritual and cultural values to identify the role of water (e.g. Zaar 2009). Ryder et al. (2010) and Ban et al. (2009) highlight the value in taking a multi-disciplinary and collaborative approach, especially where connection between science and policy is required.

The process should seek transparency regarding uncertainty, level of confidence and risks related to adopting expert panel outputs. Decision-makers should treat these outputs as the basis for investment in monitoring and evaluation.

3.5. Stage 1 assessment

Stage 1 investigations seek to answer questions on the extent and location of potential GDEs, building a conceptualisation of the ecohydrogeologic system and the critical processes related to the way groundwater potentially interacts with ecosystems. Key questions that need to be considered at this stage are:

- Where are the ecosystems that potentially use groundwater?
- What is the broad type of GDE and functional grouping?

The questions, approaches and associated tools available for Stage 1 assessment are given in Figure 3-3. Gaining a good understanding (through conceptualisation) of the potential interaction between ecosystems and groundwater provides the basis for the development of rule sets that can be applied to landscape-scale datasets to infer the location of potential GDEs at a broader scale. Information may be required on physical processes, such as surface water – groundwater interaction, and ecological processes, such as nutrient cycling, both of which allow the GDE to be allocated to one of the three typologies identified in Section 2.1.

A clear definition and description of the ecosystem (through conceptualisation) is required, along with a description of how groundwater may support ecosystem services. Potential GDEs within functional groups will occur in similar landforms, support similar plant and animal species, and occur in similar groundwater environments (e.g. 'river baseflow systems in upland fractured rock terrain' or 'stygofauna within an unconfined limestone aquifer').

STAGE 1
GDE location, classification and basic conceptualisation

Question	Approach	Tools
1.1 Where are the ecosystems that potentially use groundwater?	**Conceptualisation and landscape analysis, site specific information**	**T1 Landscape Mapping** **T2 Conceptual Modelling** **GDE Atlas**
1.2 What is the broad type of GDE and functional grouping?	**Comparison to standard guidelines and frameworks**	**ANAE Classification** **GDE Atlas** **GDE Typology**

Note: GDE Atlas refers to the *Australian Atlas for groundwater-dependent ecosystems*. ANAE Classification refers to the Australian National Aquatic Ecosystem classification scheme.

Figure 3-3: Targeted questions, approaches and tools for a Stage 1 assessment

Eamus et al. (2006) pose a series of questions to help determine the likelihood of whether an ecosystem is potentially dependent on groundwater and thus provide guidance on the identification of GDEs. They suggest that an affirmative answer to one or more of these questions means there is potentially a GDE present. The questions are reproduced in Table 3-1. Questions that can potentially be answered in a Stage 1 assessment are shaded in grey, but all are presented here as a guide.

GIS (geographic information systems) modelling, using datasets such as surface slope, soils, groundwater depth and ecological mapping (usually based on remote sensing, refer T1 and T2) is one approach that can provide an initial understanding of where GDEs may exist, especially where direct measurements are limited.

Most landscapes across Australia will have spatial coverage of data of some sort: several national datasets potentially useful for GDE assessments in data poor areas are listed in T1. These datasets provide broad-scale information on ecosystem type and landscape type. The GDE Atlas will provide an initial guide to the presence of known or predicted GDEs and the Australian National Aquatic Ecosystem (ANAE) classification scheme provides a framework for classifying GDEs that are aquatic.

Rule sets can be developed from these datasets to answer key questions that, when combined, can be used to map likely GDEs. Dresel et al. (2010) present an example of mapping potential GDEs employing regional-wide datasets. In this example the rules proposed were designed to identify:

- consistent as well as contrasting vegetation activity through interrogation of remote sensing data
- landscapes with shallow watertables and therefore where groundwater is potentially available to ecosystems.

Table 3-1 Questions posed by Eamus et al. (2006) to help determine the likelihood of an ecosystem being reliant on groundwater. Shaded questions can potentially be answered during assessment Stage 1.

Ecosystems reliant on surface expressions of groundwater	*Ecosystems reliant on the subsurface presence of groundwater*
• Does a stream/river continue to flow all year, or a floodplain waterhole remain wet all year in dry periods? • For estuarine systems, does the salinity drop below that of seawater in the absence of surface water inputs? • Does the volume of flow in a stream/river increase downstream in the absence of inflow from a tributary? • Is the level of water in a wetland maintained during extended dry periods? • Is groundwater discharged to the surface for significant periods of time each year at critical times during the lifetime of the dominant vegetation type?	• Is groundwater or the capillary fringe above the watertable present within the rooting depth of any vegetation? • Does a proportion of the vegetation remain green and physiologically active (principally, transpiring and fixing carbon, although stem-diameter growth or leaf growth are also good indicators) during extended dry periods? • Within a small region (and thus an area having the same rainfall and same temporal pattern of rainfall across its entirety), and in an area that does not receive overland flow and has no access to stream or river water, do some ecosystems show large seasonal changes in leaf area index while others do not? • Is the level of water in a wetland/swamp maintained during extended dry periods?
• Is the vegetation associated with surface discharge of groundwater different (in terms of species composition, phenological pattern, leaf area index or vegetation structure) to vegetation nearby that is not thought to access groundwater? • Is the annual rate of water use by the vegetation significantly larger than annual rainfall at the site and the site does not receive overland flow? • Are plant water relations (especially pre-dawn and midday water potentials and transpiration rates) indicative of lower water stress (potentials close to zero, transpiration rate larger) than for vegetation nearby not accessing groundwater? • Is occasional (or habitual) groundwater release at the surface associated with key developmental stages of vegetation (such as flowering, germination, seedling establishment)?	• Is the vegetation associated with surface discharge of groundwater different (in terms of species composition, phenological pattern, leaf area index or vegetation structure) to vegetation nearby that is not thought to access groundwater? • Are seasonal changes in groundwater depth larger than can be accounted for by the sum of lateral flows and percolation to depth (that is, is vegetation a significant discharge path for groundwater)? [If error terms in the estimation of lateral flow and percolation to depth are of similar of greater magnitude than the rate of vegetation use, this method may not be appropriate.]

Within landscapes where higher-resolution spatial data exist the process can be refined to incorporate specific ecosystem attributes. For example, ecological datasets which provide information on:

- common halophytic, phreatophytic and swamp species—may indicate permanent saturation of the soil zone
- the permanence of wetland inundation—may indicate an additional water source to surface water
- landscape setting; for example low-lying flood plains, dune and swale complex may indicate shallow groundwater.

Initial field assessments can support the desktop ecological and hydrological characterisation of a GDE and its biophysical setting. Information can be gathered relating to the structure and composition of ecosystems, landscape setting and presence of water.

The GDE Atlas uses GIS modelling with rule sets for large ecohydrogeological regions. These will be useful for guiding more detailed assessments of smaller areas.

3.6. Stage 2 assessment

Stage 2 assessments seek to characterise potential reliance of the GDE on groundwater. Key questions that need to be considered at this stage are:

- Is groundwater part of the ecosystem?
- How reliant is the system on groundwater?

This can be achieved through the use of physical measurements of variables such as water level, hydraulic gradients and fluxes (e.g. measurement of evapotranspiration (ET)). Geochemical and isotope analyses can provide a guide to the magnitude of exchange of water between ecosystems and groundwater. The questions, approaches and associated tools available for Stage 2 assessment are given in Figure 3-4.

The questions are difficult to answer in the absence of data. It must be stated where estimates are made from upscaling and transfer of information from other sites.

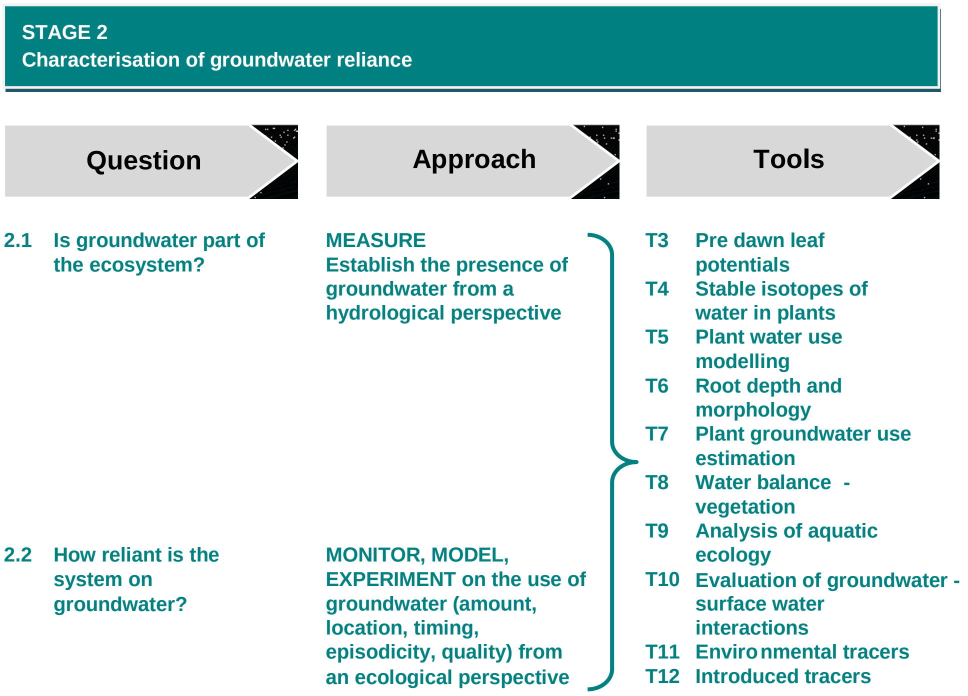


Figure 3-4: Questions, approaches and tools applicable to Stage 2 assessments

Ideally, measurements should be made across a range of timescales, in a range of landscapes and across ecohydrogeological gradients to establish a range of connections that may exist between potential GDEs and groundwater (e.g. Figure 3-5).

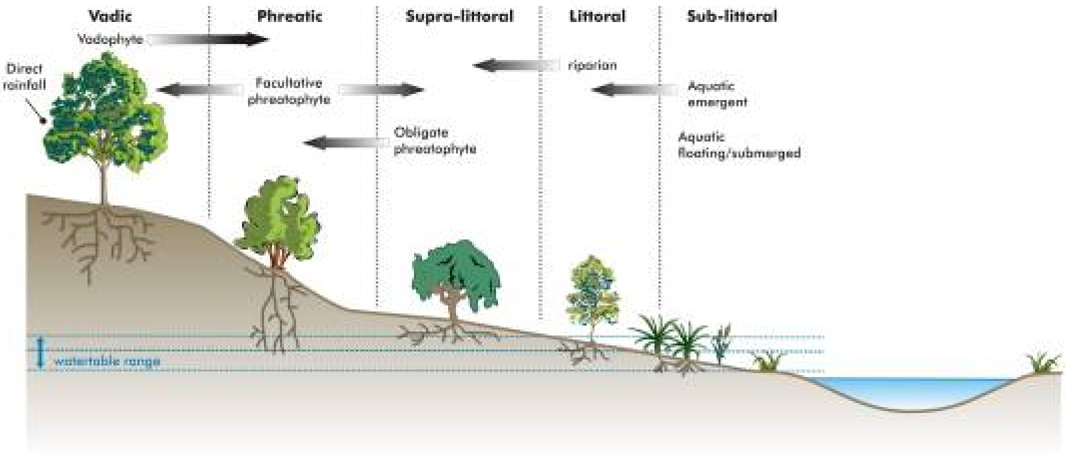


Figure 3-5: Schematic diagram (from Pettit et al. 2007) showing an example of the gradient of relationship between groundwater and ecosystem

Reliance of an ecosystem on groundwater is best assessed through collection of time-series data that are used to quantify the use of groundwater seasonally (and ideally through inter-annual wet and dry periods). The timing of groundwater use by the biota is an important consideration in the development of EWRs. In some situations, it is likely that regular contributions of groundwater are needed; for example, annual contributions of groundwater to ecosystems dependent on spring discharge or near-permanent groundwater contributions to river baseflow.

Water balance modelling can assist in the determination of whether groundwater is used by vegetation by providing an understanding between the balance of rainfall, ET and available soil moisture within the root zone (T8). The water balance approach can be supported with other evidence from pre-dawn leaf water potential measurements (T3; ideally over a number of seasons) and use of stable isotopes of water analysis to determine whether a groundwater 'signature' exists within the plant xylem (T4). Knowledge of the root depth and morphology can also contribute to the understanding of the potential reliance of the ecosystem on groundwater (T6).

The potential reliance of ecosystems (aquatic or terrestrial) where there is surface expression of groundwater can be understood by exploring whether groundwater maintains low flows or presence of water in dry periods (seasonally or during extended dry periods) and whether groundwater provides a particular temperature or water quality benefit (e.g. supply of essential nutrients).

There are a range of approaches to understanding the connection (spatially and temporally) between groundwater and ecosystems dependent on surface expression of groundwater. Some of these approaches involve physical measurements (e.g. hydraulic gradients) and flux calculations (e.g. T10) as well as geochemical approaches using environmental or introduced tracers (T11 and T12). Linked ecological and hydrogeological monitoring should be established that can be used to test hypotheses and confirm outcomes from these assessments. Where possible all techniques should be applied across a time frame that captures a representation of the prevailing climatic variability.

3.7. Stage 3 assessment

Key questions in Stage 3 of the assessment process are:

- What are the threats to the groundwater system and ecosystem?
- How might the current ecosystem change if the groundwater system changes?
- What is the long term ecosystem state due to the change? (Figure 3-6).

Monitoring programs are important in Stage 3 assessments; the data collected in Stage 3 monitoring programs can be used to create models of the relationships between the GDE and groundwater which can be used to quantify water requirements.

Threatening processes are those that have the potential to adversely impact the ecosystem state by changing groundwater conditions. Examples include abstraction of groundwater leading to a decline in groundwater level and reduced rates of groundwater discharge; land use change resulting in reduced recharge rates; and low rainfall or reduced stream flow.

STAGE 3
Characterisation of ecological response to change

Question	Approach	Tools
3.1 What are the threats to the groundwater system / ecosystem?	MONITOR, MODEL	T3-T12 (see Stage 2) T13 Long term observation of system response to change T14 Numerical groundwater modelling
3.2 How might the current ecosystem change if the groundwater system changes?	MONITOR, MODEL	
3.3 What is the predicted long term ecosystem state due to the change?	MONITOR, MODEL, EXPERIMENT	

Figure 3-6: Questions, approaches and tools applicable to Stage 3 assessments

Vegetation requiring access to groundwater for maintenance of normal ecological function may also be sensitive to changes in groundwater quality. Salinity is a particularly important determinant and Mensforth et al. (1994) and Holland et al. (2006) present discussions on the effects of groundwater salinity on the condition of terrestrial and riparian vegetation, with the latter presenting results of various floodplain inundation models of soil salinisation and vegetation health.

Quantification of the potential threats via water balance (T8) or groundwater modelling (T14) is needed to link changes in groundwater condition (e.g. drawdown of groundwater levels, saline water intrusion) with the driver of the threat (e.g. groundwater abstraction, drought, or land-use change). Modelling approaches should also take into account potential interaction between surface water features and groundwater.

It is critical that the scale of groundwater modelling is commensurate with the temporal and spatial scale of occurrence of the GDE as, often, the ecohydrogeologic analysis requires greater resolution than is available from many groundwater modelling approaches used to support regional management of groundwater systems. Multiple scales of models may be required to increase confidence.

The National Water Commission is (at time of publication of this Toolkit) preparing National Groundwater Modelling Guidelines that outline an approach to the development of numerical groundwater flow and solute transport models, including modelling surface water – groundwater interaction. <http://www.nwc.gov.au/national-groundwater-action-plan/national-groundwater-action-plan-projects/harmonisation-of-groundwater-definitions-and-standards,-and-improved-governance-and-management-practices/development-of-national-groundwater-modelling-guidelines>

Modelling and/or monitoring and evaluation programs (long-term observation of system response to change) can be designed to test threat scenarios.

Change in ecosystem state due to changing groundwater conditions has previously been captured under the concept of 'ecological response functions'. This has been a problematic concept in the sense there has been a perception that neat and predictable relationships exist. While there are some examples (see Appendix A, Case study 1), this is mostly not the case.

GDE responses to altered groundwater conditions are best described using demonstrated relationships based on long-term observational data of ecological function and altered water regimes (T13). The data collected from these programs can be used to derive conceptual and predictive analyses to further test hypotheses of how ecosystems may change if impacted by threatening groundwater-related processes.

Where data are not available to describe the response in detail it may be possible to transfer knowledge from other similar areas, and use expert opinion, hypothesis testing and commitment to an appropriate monitoring program (Tomlinson 2011).

3.8. Expression of ecological water requirements

Ecological water requirements can be defined and expressed in several different ways. Which method is adopted for use will depend on:

- **the detail of information available about the ecosystem's requirement for water**

  Does the information include quantitative associations or only inferred or qualified relationships? A simple qualitative statement such as 'have access to groundwater (of a certain quality) in a spring', for example, if verified by available scientific information, can define the EWR of an ecosystem. With increasing levels of assessment detail and application of techniques such as those outlined in this GDE toolbox, the more precise definitions of EWRs are possible.

- **the scale at which the requirement is determined**

  The requirement can be expressed at the ecosystem scale or component species/process scale. At the species scale, indicator species identified on the basis of their sensitivity to changes in groundwater regime may be selected as the basis for defining EWRs. At the ecosystem level, requirements may be expressed through an understanding of critical ecological processes and their requirement for the maintenance of a particular groundwater regime.

- **the level of understanding of the spatial and temporal variability in the requirement for water.**

  For many GDEs the requirement for groundwater is not constant and will vary in space and time. If the scientific information regarding an ecosystem's water requirement is sufficient to define the spatial and temporal variability in groundwater needs then it is critical to incorporate this in the EWR. This may involve stating the range of depths to watertable over a seasonal cycle or the short- versus long-term duration of groundwater contact and quality.

In the early phases of investigation (e.g. Stage 1) the EWR may be qualitative (and perhaps quantitative if data permit) and applied at a landscape or ecosystem scale. As more information becomes available about an ecosystem's requirement for water, the more likely the EWR of that system, or components of it, will be defined by a number of complementary statements (rule sets) of water requirement, in which case these can be tabulated as a set of ecology–hydrology linkages. Where detailed information is limited, the emphasis would be placed on identifying habitats, at a landscape scale, that potentially support GDEs. This is supported by analysis of where groundwater discharge can occur (possibly derived from a depth-to-watertable map). It is important that a statement of the level of confidence is also provided.

Where more detailed information is available (e.g. following Stage 2 and Stage 3 assessments) the EWRs may be quantitative. These are not inferred from principles, but measured *in situ* and would:

1. specify the requirement(s) in a hydrological context using measurable parameters (qualitative or quantitative) and may include, for example, the rate of discharge (or magnitude of connection) required, range in depth to watertable and/or pH

2. identify the ecological basis or bases for the requirement; for example, maintenance of anaerobic sediment conditions by continued groundwater discharge.

An example of the type of EWR information that can be established from this more detailed type of assessment is provided from Pirrabadoo Creek in Western Australia (Howe et al. 2009). At this location, stable isotope analyses of water, combined with sap-flow monitoring and measurements of leaf water potentials, were used to establish the groundwater requirements of riparian vegetation. The study found that groundwater meets some of the water requirements of coolibah (*Eucalyptus microtheca*) and paperbark (*Melaleuca* spp.) trees during the drier periods of the year or during prolonged dry seasons. Specifically, it is stated that vegetation requires access to groundwater and maintenance of watertable levels within an accessible range.

No matter whether the EWR is qualitative or quantitative, it must be expressed in a manner that is explicit spatially (e.g. where are the connections to a river?) and temporally (e.g. timing of groundwater level fluctuations) using measureable attributes of groundwater condition. Where surface water environmental flows are concerned, EWRs relate to capture volume, frequency, duration and timing of flow. For GDEs, EWRs may relate to volume of groundwater discharge, frequency of access of an ecosystem to groundwater, or duration and timing of groundwater access.

With increasing stages of assessment, the accuracy of information on water requirements and ecosystem response to altered hydrology becomes available. The need for EWRs to include information on response to change in water availability increases when long-term

resource development and climate variability are considered. Further discussion around accounting for impacts of climate variability is provided in Section 7.

Regardless of the level of detail used to define an EWR, it is imperative that EWRs are developed specifically for the ecosystem (or ecosystem components) in question. Whilst similar ecosystems may share some similarities in groundwater requirements, the exact nature of these requirements may vary within and between locations. EWRs relating to groundwater accessibility can vary between species comprising an ecosystem (Table 3-2) and it is important to consider, and where possible account for, this variation.

Table 3-2: Basic water requirements of phreatophytes of the Swan Coastal Plain, Western Australia (Sommer & Froend 2010)

	Watertable depth (m below ground surface)		
Species	*High extreme*	*Low extreme*	*Range*
Astartea fascicularis	-0.20	6.43	6.63
Banksia littoralis	-0.04	9.27	9.23
Baumea articulata	-0.80	1.62	2.42
Eucalyptus rudis	-0.80	11.7	12.5
Melaleuca preissiana	0.18	13.9	13.7
Melaleuca rhapiophylla	-0.80	9.27	10.1
Typha orientalis	-0.50	2.40	2.90

3.9. Ecological response to change in groundwater conditions

The ideal form of expression of an EWR for GDEs is one that captures the change in ecosystem state due to changing groundwater conditions, representing the pathway of ecological change (of a dependent ecological attribute) along a scale of hydrological condition (i.e. a continuous scale of water availability). This can be referred to as an ‘ecological response function’ describing the relationship between groundwater and the ecosystem (Eamus et al. 2006, Murray et al. 2003).

Such a response function is usually represented diagrammatically if the understanding is conceptual or qualitative and/or as a mathematical relationship or model if the ecosystem (or component) response is quantifiable (see Appendix A, Case Studies 1 and 2). Irrespective of how this is represented, it should identify the known ecological response to variation in groundwater availability (or quality). For example, reduction in groundwater level or recharge due to reduced rainfall, increased evaporation or abstraction will result in a shift of that component or ecosystem along the function (pathway).

As discussed in Section 3.7, ecological responses are best described through long-term observational data; however, even this may not represent a neat and predictable relationship to define an ecological response function. Regardless, an EWR that incorporates an understanding of the ecosystem response to change should represent the known response of the system or system component (e.g. a plant species or just vegetation) over time and across a scale of groundwater availability (level etc.). In other words, an EWR is the known, intrinsic ecological requirement for groundwater. It therefore represents what is understood about:

- how an ecosystem (or ecosystem component) is likely to respond to a change in groundwater quantity and/or quality

- the range of hydrological conditions within which the ecosystem will persist
- the hydrological thresholds that represent the limits of ecosystem persistence and resilience.

Where the available data allow a semi-quantitative or quantitative response function to be developed for a particular GDE, the understanding of GDE relationships with groundwater is further developed, thus providing a sound basis for deriving defensible EWRs.

GDE responses to altered groundwater conditions are best described using demonstrated relationships based on long-term observational data of ecological function and altered water regimes. This would ideally require a number of field sites at which assessments of changes in ecosystem composition/function following reductions in groundwater availability and/or quality have been made. Unfortunately there are very few studies in Australia where sufficient temporal data exist to allow a subsequent assessment of the impact of altered groundwater regimes and, consequently, the development of a response function in biophysical terms. As indicated in Tomlinson (2011), time frames of studies for understanding the response of GDEs to changed groundwater conditions are in the order of years to decades (Table 3-3).

Perhaps the most documented group of studies of the response of GDEs in Australia are those relating to phreatophytic vegetation and watertable decline as a result of groundwater pumping and rainfall regime change on the Gnangara (groundwater) Mound, on the Swan Coastal Plain, Western Australia. For example:

- Groom et al. (2000) found that watertable decline was accompanied by changes in banksia community composition, floristic structure and, in some cases, increased mortality of canopy trees where watertable drawdown was substantial
- Sommer and Horwitz (2001) explored the response of macroinvertebrates to acidification following intensified summer droughts in a Western Australian wetland
- Zencich et al. (2002) related banksia transpiration response to watertable depth on the Swan Coastal Plain
- Froend and Drake (2006) provide details of ecological response functions developed for banksia spp. on the Gnangara Mound
- Sommer and Horwitz (2009) explored cycles of macroinvertebrate decline and recovery on the Swan Coastal Plain wetlands affected by drought-induced acidification
- Froend and Sommer (2010) presented two case studies of alternate pathways of phreatophytic vegetation change in response to groundwater decline
- Sommer and Froend (2011) identified the capacity for resilience in phreatophytic vegetation diminished with progressive, long-term drawdown.

More often than not it is not possible to demonstrate the ecosystem response to change with observational data. Where data from direct measurement of ecosystem response are not available other approaches are required to predict the impact of altered groundwater regime on GDE function. From a conservation perspective, it will not always be desirable or possible to allow GDEs to become stressed so that observational data can be collected.

In these cases, predictive modelling approaches can be crucial in defining responses for GDEs that may be impacted, or are being impacted, by threatening activities. Depending on the type of GDE being modelled, certain input data will be required to ensure the model predictions are representative and not based on conjecture.

Cook and O'Grady (2006) describe one such approach for terrestrial vegetation known to be groundwater dependent to some degree. Their approach allows field measurements of depth to watertable, plant water use, leaf water potential and soil matric potential to be used in developing a calibrated predictive model of plant water conductance. Reducing soil matric potential values in the model are akin to what would occur in conjunction with watertable decline, resulting in a reduced hydraulic gradient between the soil and leaf, and a subsequent decrease in transpiration. The method has been employed by Howe et al. (2006) in relating plant transpiration response and leaf water potential to watertable drawdown (see Appendix A, Case study 2).

Another difficulty often faced when trying to represent ecosystem response to changes in groundwater condition change is that we do not know the temporal context of the hydrological change:

- How long will it last? Is it an episodic (temporary) event that the GDE can recover from and, if so, will the path of recovery be different and confounded by other impacts associated with drying? In cases where the main cause of groundwater change is abstraction for consumption, the short- and long-term duration and rate of change can be determined. However, in a climate-change scenario, including interaction with groundwater abstraction, it can be difficult to identify the duration, rate of change and the likely response of ecosystems (a discussion on separating impacts of climate variability and groundwater abstraction is provided in Section 7.3). Again, examples of long-term monitoring can assist with understanding the possible pathways of ecosystem response. In the absence of direct measurement, scenario modelling of aquifer response coupled with an understanding of ecological water requirements of the system in question can be used.
- How long has the change been occurring? Does our measured dataset represent only a 'window' to a much longer process of hydrological change? In many cases the change we observe during a short monitoring program represents only a small part of a much longer change sequence, be it either progressive (directional) or fluctuating.
- What is the lag in ecosystem response to hydrological change? How long before the ecosystem responds?
- In considering these issues it becomes apparent that the notion of being able to quantify a relationship between groundwater condition and an indicator of ecological value is 'simplistic' (Boulton 2009, also highlighted by Tomlinson 2011.). The consequence of this position is that initial estimates of EWRs using expert knowledge and conceptual models need to be supported by a longer-term approach to understanding of ecological response to change through targeted monitoring programs designed to test specific hypothesis about the ecosystem response to hydrological change.

Table 3-3: Data type and time frames of GDE studies (adapted from Tomlinson 2011)

GDE	*Hydrological and ecological metrics*	*Application*	*Date record*
Banksia woodland, Northern Swan Coastal Plain, Western Australia (Froend & Sommer, 2010)	Groundwater level Floristic composition and abundance	Indication of species' drought vulnerability and identification of threshold change in floristic composition	Thirty years of data: vegetation composition monitored biennially or triennially
Riparian meadow, Northern California, USA (Loheide II & Gorelick, 2007)	Maximum groundwater level (spring), and duration of groundwater levels within 1 m of surface (summer) Xeric and mesic vegetation communities	Development of predictive vegetation model based on groundwater level thresholds for vegetation type—used to explain observed vegetation shifts in response to stream incision.	Two years of groundwater-level data
Alkali meadows, Intermontane valley, California (Elmore et al. 2006)	Depth to watertable in April (assumed shallowest) Vegetation cover (remotely sensed data)	Estimation of threshold depth to watertable for maintenance of alkali meadows	Sixteen years of groundwater-level data measured at least twice annually; Landsat imagery for each year
Woody riparian vegetation, Highlands of central Arizona, USA (Shafroth et al. 2000)	Change in annual lowest groundwater depth Change in stem density and basal area of stands of saplings	Development of conceptual model of woody vegetation response to watertable decline	Two and a half years of monthly groundwater levels; sapling quadrants sampled at three sites for two years
Riparian vegetation and aquatic flora and fauna, Daly River, Northern Territory (Erskine et al. 2003, DRMAC, 2009)	Groundwater level and spring discharge Habitat zones of preference for water dependent flora and fauna Water use of riparian vegetation	Development of ecohydrogeological conceptual models to determine groundwater depths and discharges for ecosystem maintenance and conservation	Three years of seasonal sampling; ongoing (10+ years) further monitoring, analysis and evaluation

4. Conceptualisation

4.1. Importance of conceptual models

Conceptual models are tools that formalise an understanding of the major components of a given system, their interactions, and how external changes can modify the system. They are a cohesive, often simplified, way of expressing what we know and can help us to define (and/or test hypotheses regarding) the critical components that make up the area or ecosystem in question, the relationships of 'cause and effect', and more generally 'how the system works'. Conceptual models can be developed to illustrate a range of ecological structures, processes, responses and interactions, and are highly effective in capturing the current scientific knowledge of an ecosystem and showing likely ecological responses to natural and anthropogenic stresses (Gentile et al. 2001, Ogden et al. 2005). Generally, a conceptual model provides four functions. It:

- clarifies the problem, and ensures that the critically important components and the ecological interactions between them have been identified. It thus ensures that all the participants managing a given system are working from the same basic understanding as to how that system is structured and functions
- identifies the 'knowns' and the 'unknowns', and thus the critical gaps in knowledge and where research investment needs to be focussed
- allows predictions to be made about the likely impacts of different management interventions, and thus which ones can be eliminated as unlikely to be useful
- assists with the design of monitoring programs and selection of appropriate indicators.

Further guidance on the construction of conceptual models and resources available to assist in the process are provided in T2 and in the following discussion.

Ultimately a conceptual model should reflect the information known about the study area (relevant to the hypotheses being tested/questions being asked) and wherever possible, be developed collaboratively between a project team with a knowledge base spanning different disciplines. Conceptual models should help form a shared understanding of the ecosystem that is the basis for a study (ANZECC & ARMCANZ 2000).

4.2. Conceptual model format

Conceptual models vary in complexity and can take several forms. In ecological and natural resource investigations they are often represented as a stylised oblique aerial view of the landscape or landform or as a combination of pictorial representations combined with process diagrams. Alternative formats of expression include written narratives, tables, mathematic formulations and schematic diagrams such as box-and-arrow models. Plumb (2003) provides an overview of the different ways conceptual models can be expressed.

It is often useful to have stylised, whole-of-ecosystem conceptual models to provide an easily understandable, visual representation and focus attention on linkages between smaller subunits. However, the limitation with such spatially extensive models is that, by attempting to be all-encompassing, they can be too broad to be useful. The topic of unnecessary complexity in conceptual models has been reviewed by Gross (2003) and Plumb (2003).

A distinction is often drawn between control and stressor types of conceptual models (e.g. Gross 2003). The former—control models—include all the main components of the system

under study, as well as the various drivers and feedback mechanisms. They are intended to give an accurate portrayal of the system at a given level of complexity or spatial scale. Most of the simplified, oblique-view model types seen in the natural resource literature are control models. In contrast, stressor models are an even greater abstraction of a particular system, and focus explicitly on the links between stressors, components, responses and effects. Accordingly, stressor models do not incorporate all the system components as do control models, but instead limit themselves to those elements that are important in showing the main links between a given (usually single) stressor and the ecological responses it creates. The models developed as part of Bayesian analyses of ecological responses to given disturbances are often stressor models.

4.3. What makes a good conceptual model?

There is no such thing as a single, best, all-purpose conceptual model. Different types of models are best suited to different purposes (e.g. the difference between control and stressor models) and for different audiences; a model developed for a technical analysis of the likely impacts of groundwater drawdown on biogeochemical processes is likely to be quite different from one designed to inform about the general need for EWRs to maintain GDEs.

When developing conceptual models, there is always a trade-off between realism, generality, and precision; it is not possible to maximise all three simultaneously (Levins 1968). It is important to remember that a conceptual model is a simplified representation of a complex natural system; it can never be as complex or as complete as the original. The critical issue is only whether the model suffices for the task it is expected to address.

What makes a given conceptual model useful or not often depends on its internal complexity. For certain purposes, a conceptual model need only be a simple box diagram that illustrates the components and linkages in the system to be monitored. Models can quickly become too complex if they incorporate too many stressors, components, feedbacks or processes. In such cases the models are undeniably comprehensive, but are usually too complex or detailed to be understood or useful. Thus a good conceptual model does not attempt to explain all possible relationships or contain all possible factors that influence the target condition, but instead tries to simplify reality by containing only the information most relevant to the model builder (Gross 2003).

For GDE assessments, a good conceptual model will describe the ecosystem's hydrology and hydrogeology, and biotic components and processes. It is important to understand what specific processes need to be considered when building a conceptual model—in most cases, hydrogeological processes including aquifer-to-aquifer interactions and surface water-to-groundwater interactions should be considered. Further guidance on the conceptualisation of interaction between surface water and groundwater systems can be found within the Commission's *Groundwater / Surface Water Connectivity Guidelines* (NWC 2011). Aquifer-to-aquifer interactions can be more difficult to describe, particularly in data-limited environments. Of particular concern in conceptualisation of GDEs is the nature of any potentially perched aquifers (Section 2.2).

Other abiotic components to consider include recharge, discharge and storage processes, and mixing and groundwater flow. It is important to understand the temporal patterns of groundwater levels (seasonal and inter-annual).

Biotic components to consider include any flora or fauna that forms part of the ecosystem that is potentially reliant on groundwater for habitat, and a water source, food supply or other service. As outlined in T2, processes such as primary production, herbivory, predation and competition could also be considered.

Understanding the water regime that supports the GDE and the links between the abiotic and biotic components are also important. The key linkages to describe include the water regime (and characteristics thereof) that support the GDE and, most importantly, any critical groundwater services. A critical groundwater service may include water provision for habitat or use, artesian (or other) pressure, thermal water supply, nutrient supply or some other modifier of water quality critical to ecosystem function. How these vary (or are hypothesised to vary) with seasonal or other long-term variations could also be described.

In any conceptual model, any key assumptions made in the development must be made explicit, along with an indication of the degree of uncertainty around these assumptions.

An example of a how an ecohydrogeological conceptualisation can be derived from limited information is provided in Figure 4-1 from Collie River in south-western Western Australia (SKM 2010b). In this case, information describing the hydrology and hydrogeology was compiled to formulate the hydrogeological conceptual model to support the hypothesis that both permanent pool and riparian vegetation ecosystems are groundwater dependent.

CHINAMANS POOL – SOUTH COLLIE RIVER

Ecosystem type:

Permanent river pool dependent on the surface expression of groundwater with narrow strip of open woodland of *Eucalyptus rudis* and *Melaleuca rhaphiophylla* over *Astartea scorparia* over *Watsonia meriana* var. *bulbilifera* and pasture grass. Supports aquatic species of macroinvertebrates, crayfish, fish and amphibians.

Aquifer:

- Chinaman's Pool is incised into the unconfined shallow aquifers of Quaternary cover over the Nakina Sand and the upper 10 m of the Muja Coal Measures.

Flow system:

- The pool is located in a catchment scale groundwater discharge zone on the South Branch of the Collie River.
- Recharge to the groundwater system is from rainfall with groundwater levels indicating a downward gradient between the shallow aquifers and the deeper aquifers.

Trends in groundwater level:

- The time series water levels show current, historical maximum and historical minimum levels are all more than 3 m above the cease-to-flow level of the pool and appear stable with the 2009 level similar to the 2006 level at around 179 m AHD.

Spatial/temporal connections:

- Groundwater levels are above the base of the pool and above the cease-to-flow level – connected and gaining when pool levels are at cease-to-flow level.
- There is the potential for groundwater to perch within the Nakina sand. Perched water could would re-enter the pool on the recession of higher flow events

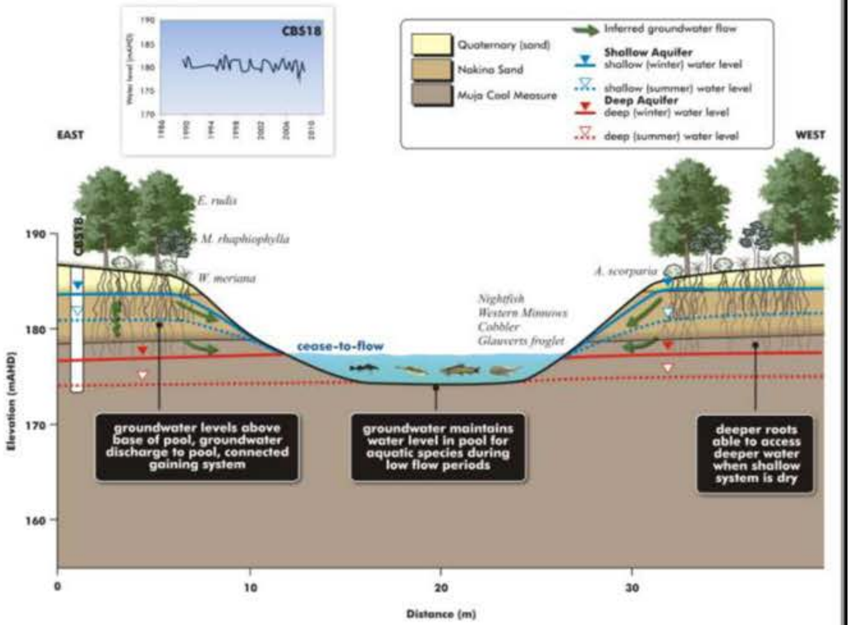
Information sources:

Department of Water (WA) 2009, *Upper Collie water allocation plan*, Water resources allocation and planning series, Report no. 20, August 2009.

SKM 2010. *Identification and Mapping of Groundwater Dependent Ecosystems Associated with the Collie River.* A report to the WA Department of Water.

Varma, S., 2002. *Hydrogeology and Groundwater Resources of the Collie Basin, Western Australia*, Water and Rivers Commission, Hydrogeological Record Series, Report HG 5, 80 p.

Local Ecohydrogeology



Location and landscape

Regional Groundwater System

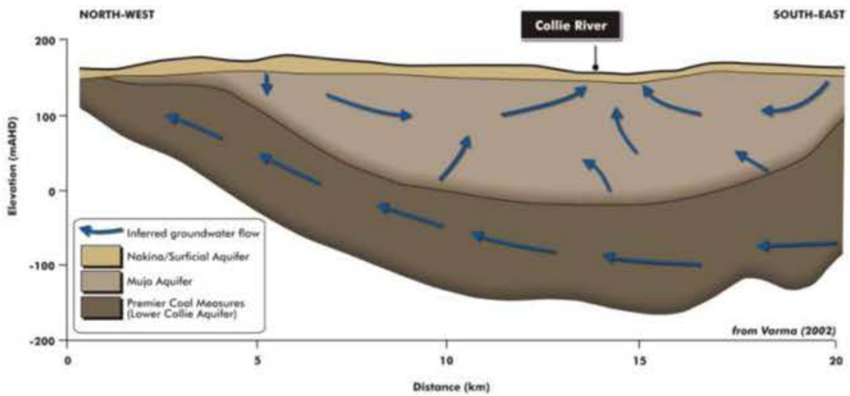


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Figure 4-1: Conceptual diagram of a groundwater dependent pool in the South Collie River, Western Australia, an example of an ecosystem dependent on the surface expression of groundwater (Type 2)

5. Monitoring and evaluation

5.1. Introduction

Monitoring and evaluation should form an integral part of any detailed GDE assessment and management plan. Not only does effective monitoring and evaluation assist in meeting requirements of the National Water Initiative[4], programs can be used to provide a sound scientific basis for developing and implementing environmental water provisions (EWPs) or to support allocations made and/or proposed in water management plans.

The aim of the GDE toolbox is to aid in the development of appropriate EWRs and, as discussed previously, the ideal EWR is one that is based on quantified and validated relationships between the ecosystem and groundwater source, derived from observational data. Monitoring and evaluation plays an essential role in developing, implementing and evaluating appropriate requirements for the management of GDEs. The absence of long-term monitoring data, particularly ecological monitoring, is considered to be one of the greatest impediments to accurate and widespread EWR definition (Parsons et al. 2011).

Within this assessment framework, monitoring is recommended under both Stage 2 and Stage 3 assessments—for the purposes of assessing the level of dependency of an ecosystem on groundwater and how the ecosystem responds to changes in the groundwater system, respectively. Monitoring data also allows more thorough treatment of Stage 1.

Following EWR definition, a robust program of monitoring is also required to determine whether or not an EWR derived for a given GDE, and any management programs that follow, are appropriate and effective. Well-designed monitoring programs and hypothesis testing also add to the broader scientific knowledge base, which not only assists in the management of GDEs at specific locations but can be used as case-studies for other GDEs at other locations (i.e. as part of an extrapolation exercise, see Section 6).

The following discussion provides some guiding principles for developing and implementing appropriate and effective monitoring programs to aid in the development, implementation and evaluation of EWRs of GDEs. This includes critical aspects for consideration when determining what needs to be monitored and why, and what actions may need to be implemented if the results are not as expected. The discussion is developed with reference to the Australian Guidelines for Water Quality Monitoring and Reporting (ANZECC & ARMCANZ 2000).

To complement this discussion, three contrasting case studies to highlight the diversity, challenges and choices that need to be made when developing monitoring programs for GDEs are presented (Appendix A). In ‘Designing a monitoring program: The Hyporheic Zone of Streams in the Flinders Ranges, South Australia’ (Case study 3), interactions between surface water and groundwater control not only the size and functioning of the hyporheic zone under and around the stream but also the productivity of algal mats—the main food source for stream invertebrates in an otherwise harsh and dry ecosystem. ‘Designing a monitoring program: Coastal Vegetation – Saltmarshes and other Estuarine Wetlands in South-eastern Australia’ (Case study 4), investigates the way that groundwater discharge influences salinity and nutrient regimes in estuarine wetlands, and thus the patterning and productivity of saltmarshes, mangroves and reedbeds. ‘Designing a monitoring program: Terrestrial and Aquatic Vegetation of the Tomago Sandbeds, New South Wales’ (Case study 5), builds upon prior analyses of the Tomago Sandbeds, New South Wales. The sandbeds

[4] More information regarding the National Water Initiative is available at
http://www.nwc.gov.au/www/html/117-national-water-initiative.asp

have provided groundwater for potable, agricultural and industrial uses for over 70 years, but lie under (and support to various degrees) a diverse mosaic of terrestrial and wetland vegetation, which includes a large number of rare, threatened or endangered plant species.

5.2. Principles of monitoring

Monitoring is the systematic collection of data over time to test a hypothesis or gauge compliance with a standard or model (Hellawell 1991). It differs from other environmental data-collecting activities, such as survey and surveillance, in two ways (Finlayson & Mitchell 1999) in that:

- it is underpinned by a specific reason for collecting the data
- the results are compared with a prediction, model or standard, which is used iteratively to interpret, check or test the data. Actions then arise according to whether or not the prediction or standard is supported or refuted.

As with any natural system, there is no off-the-shelf monitoring program that can be applied to all GDEs, and the key difference in studies will relate to monitoring objectives and the components and processes of interest at each site. A common basis for all GDE-related monitoring programs, however, is the need to understand the relationship between different components (physical, biological, and physico-chemical) of an ecosystem and the processes that underpin them. A typical program would include the collection of data (hydrological, hydrogeological, chemical and/or biological) over a period of time (to determine temporal trends), which would be evaluated regularly in line with the ongoing management of the resource.

A well-designed and implemented monitoring plan can greatly increase the understanding of GDEs and how they interact with and respond to their environment.

5.3. Setting monitoring objectives

The first and arguably the most important step for planning an effective monitoring program is to set clear objectives. Monitoring objectives should be specific, measurable, realistic, attainable and meaningful. These objectives should capture the overall aims of the program, address gaps in knowledge or answer key questions that will inform management decisions. In the case of GDEs, these objectives should, in the first instance, relate to unknowns or uncertainties in the links between groundwater and an ecosystem or EWR knowledge. Following development of EWRs and implementation of a subsequent management plan, monitoring objectives may be more focused toward management objectives. Regardless of the primary driver, three components are outlined below to guide the setting of monitoring objectives (Figure 5-1).

Figure 5-1: Component tasks to setting monitoring objectives

5.3.1. Define the issue

The first important question to ask is 'What is the purpose of the monitoring program?' In GDE assessments, monitoring programs will most likely be either 'investigative' and relate to informing the conceptual understanding of the ecosystem or be related to management actions such as evaluating whether or not an EWR (or EWP or other management initiative) defined for a given GDE is appropriate and effectively supports the ecosystem.

Some issues that may drive GDE monitoring with regards to EWRs include:

- developing an EWR to manage, protect and/or rehabilitate a GDE so that it can support its ecological values
- identifying any responses (e.g. condition of a GDE) to hydrological changes in line with a management plan implemented on the basis of the derived EWRs
- describing the impacts of an anthropogenic or climatic stressor on a GDE (e.g. groundwater abstraction, drought, or contamination) and the impacts these may have on a defined EWR.

5.3.2. Develop a conceptual understanding

Conceptualising how the target ecosystem works is imperative to undertaking any monitoring program. Maddox et al. (1999) suggest that it is generally impossible to design an effective environmental monitoring program unless a conceptual model has been developed first to describe the system. This is primarily because (as discussed in Section 4) a conceptual model should explicitly show all relevant information known about a GDE; they ensure an understanding of the critical biological, physical and physico-chemical components of a GDE, the interactions between them, and the way the system may respond to external disturbances such as groundwater abstraction (based on available data). The designer of a monitoring program can then use the model as a tool for choosing what indicators (e.g. components or attributes of a system), and/or variables, should be monitored.

When undertaking conceptualisation for the purposes of informing a monitoring and evaluation program, the scope of any such program should be clearly defined. Does a physical (biotic or abiotic) characteristic of the ecosystem or a management boundary define the spatial limits of the GDE for the purposes of EWR definition and management? And what are the implications of this? Any threats or drivers external to the GDE boundaries that could influence the outcomes of the monitoring program should be identified at this point and recognised throughout the program development and implementation. Particular consideration should be given to the scale of the groundwater flow system and proximity of the GDE to any development (i.e. abstraction and any associated lag effects).

5.3.3. Set monitoring objectives

The objectives of the monitoring program should be based on a conceptual model and aim to answer key questions that will inform management decisions and/or assess the impact or effectiveness of management decisions taken. Hypotheses are a common and effective format for framing monitoring objectives. They usually take the form of statements or theories that can be subjected to statistical evaluation when monitoring data has been obtained to determine whether they can be accepted (or rejected) (ANZECC & ARMCANZ 2000).

Examples of hypotheses that relate to informing management objectives include 'We hypothesise that increasing depth to watertable will lead to a change in the structure, condition and vigour of vegetation, resulting in the decline of habitat values' or 'We

hypothesise that groundwater is maintaining water levels in refuge pools during drought conditions'. Associated objectives may be 'to determine if reductions in watertable depth result in a decline in vegetation indexes and habitat values' or 'to confirm that groundwater does maintain levels in refuge pools during drought conditions', respectively.

In the case of assigning the impact of effectiveness of management decisions undertaken, an example hypothesis could include 'We hypothesise that reduced groundwater extraction will result in an increase of habitat values'. The associated objective would be 'to determine if reductions in groundwater extraction result in the expected change'.

Clearly articulated objectives form the basis of a monitoring program and shape the study design, field methods, data analysis and reporting. At all stages of a monitoring program, the monitoring team should be checking back to the objectives (and the conceptual model if necessary) in the light of the results and vice versa. If necessary, the program (and/or conceptual model) should be reassessed (ANZECC & ARMCANZ 2000).

5.4. Program design

Effective monitoring programs are carefully developed to ensure the objectives are achieved (ANZECC & ARMCANZ 2000). Monitoring programs can be expensive and few organisations have the resources to monitor over a large geographical area or a long period of time. Ideally, each monitoring program is set up in awareness of and in cooperation with other past and present monitoring programs. Forming linkages with other monitoring and evaluation programs allows returns from available resources to be maximised and collaboration between water and land managers results in more successful GDE management. The data generated from all programs can then be compared, integrated or collated to report trends (ANZECC & ARMCANZ 2000). There are three crucial components to developing a monitoring program, which are outlined in Figure 5-2.

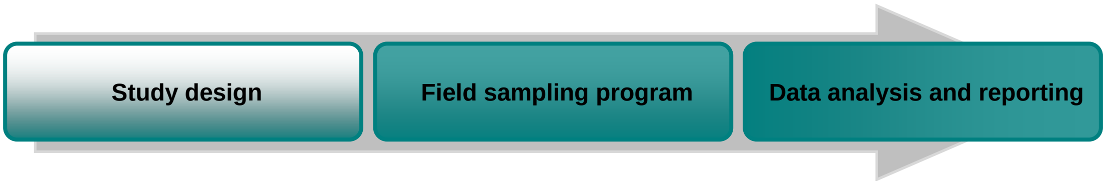

Figure 5-2: Components in developing a monitoring program

5.4.1. Study design

Experimental design is a critical aspect of devising an effective monitoring program and, unless it is right from commencement, the data it generates will be at least compromised and, at worst, useless for the intended task.

Green (1979) outlines the fundamentals of good monitoring design, and his ideas were developed further by Underwood (1991, 1993). Despite the age of these reports, the information they hold is still valid and useful to those who design or interpret monitoring programs. Updated information on more complex monitoring programs is provided by Downes et al. (2002).

There are many ways to design a monitoring program either to detect changes or test hypotheses in a given environmental system. Any strong inferences (i.e. those with the least uncertainty) about environmental changes, however, require observations to be made from

areas that are affected by the perturbation ('impact' sites) and areas that are not ('control' sites), as well as before and after the perturbation in both areas (Downes et al. 2002). These requirements are met most clearly by BACI (Before/After and Control/Impact) or Beyond-BACI-type designs.

BACI and Beyond-BACI

A BACI design is, in principle, the most powerful way to unequivocally test for environmental change in response to some external disturbance (Green 1979). This method takes samples from a single 'impact' site and a single control site (as similar as possible to the 'impact' site in its characteristics), both before and after the impact or management intervention. If the usual precautions are taken to ensure that the samples are representative, unbiased and independent, the analysis is relatively straightforward. A statistical analysis is applied to the data and the existence of an impact (or some response to the management intervention) is indicated by a statistically significant result (Underwood 1991). The rationale of the approach is summarised in

Figure 5-3.

The simple BACI design however suffers from two main theoretical and a number of practical drawbacks. The first is that a single control site may not be statistically valid and multiple control sites are needed. Second, the use of single before and after sampling does not allow temporal changes to be detected in response to the disturbance or intervention. BACI designs that take into account multiple control sites and/or repeated sampling before and after the 'event' are often called 'Beyond-BACI' designs. Although complex, the same general type of interpretation is used for them as for simpler BACI designs. Downes et al. (2002) describe a number of Beyond-BACI experimental designs suitable for use in environmental monitoring programs.

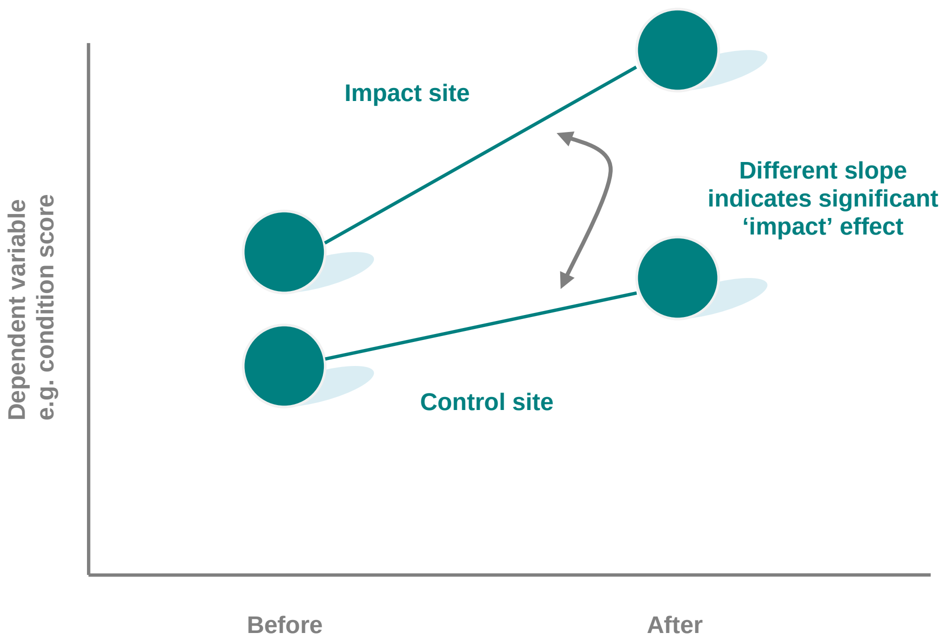

Figure 5-3: Example of a simple BACI type approach to detecting an environmental impact

In addition to being used as the basis of a monitoring program in their own right, BACI and Beyond-BACI designs can be useful in answering questions in Stage 2, and particularly

Stage 3, of the EWR assessment framework. Comparison of the behaviour in a control site that is known to be neither reliant on groundwater nor subject to groundwater extraction (or other threat) to the behaviour of the target site can go some way to determining whether the target site is or is not reliant on groundwater and the response of the ecosystem (if any) to the threats.

The main practical drawback with any BACI-type design is the availability of suitable control sites. It is often not easy to find a GDE site which can act as a control. Downes et al. (2002, p. 219) provides some simples rules for deciding whether or not a site would be a good control area in BACI-type designs.

Other practical drawbacks with the BACI approach are often matters of simple pragmatism. For example, the same types of sampling need to be undertaken in all the control and impact sites, which can place a large demand on human and financial resources. Finally, the type of statistical analysis required for Beyond-BACI type designs is inevitably more complex than that used for BACI designs (Underwood 1993). Quinn and Keough (2002) provide a detailed description of the types of statistical analyses that might be required for Beyond-BACI designs.

Other approaches

As discussed above, the theoretical and practical limitations of complex BACI-type experimental designs often rules them out for monitoring GDEs and EWRs. Other approaches to environmental assessments include before/after contrasts at a single site, repeated before/after sampling at a single site, and gradient analysis. Results from what can be considered simpler approaches are however limited.

Before/after contrasts at a single site

The simplest approach to detecting an environmental change (or response to a management intervention, such as the effectiveness of an EWR) is to take a single sample before and a single sample after the disturbance or intervention. However, the main limitation of this approach is that there is no way that any change in the intervening period could be attributed solely to the putative disturbance; any changes that are detected may be entirely coincidental. A single ‘after’ snapshot also may not adequately capture any lag effects.

Repeated before/after sampling at a single site

This design takes repeated samples before and after the management intervention, but only at a single site and usually at fixed time intervals. As with before/after contrasts at a single site it ‘cannot indicate that there is some relationship between the change in numbers and the putative disturbance’ (Underwood 1991, p. 571) because there is no control site to provide a comparison of what happened in sites that were not subject to the disturbance. For these reasons, this approach is not highly recommended for inferring the effectiveness of management interventions such as gauging the effectiveness of EWRs.

Gradient analysis

Gradient analysis is an empirical analysis methodology to explore trends and changes in environmental variables over spatial or temporal gradients. Gradient analysis requires fewer assumptions than BACI and Beyond-BACI designs and, in real-world terms, is perhaps often a more robust approach to detect the effect of environmental perturbations in long-term, field-based studies.

Although gradient analysis generally permits weaker inference than BACI-type designs, it is powerful when the impact and control sites themselves are in the gradient space (Downes et al. 2002). This is likely to be the case with many GDE and EWR monitoring programs where extraction (or other driver of change) is in effect prior to the implementation of a monitoring program (i.e. no 'before' data available). Where extensive field work is undertaken, gradient analysis has the additional benefit of allowing the detection of gradual changes across the perturbation, which is often the more interesting ecological response (Downes et al. 2002) and more relevant for informing EWR development. Gradient analyses can include simple classification approaches (e.g. see Raulings et al. in press for an example related to environmental water and aquatic systems in Australia) or more complex multivariate and ordination analyses.

Importance of levels of evidence and environmental monitoring

It is rare in environmental studies to have a single, unequivocal answer to a given question. It is more common that a practitioner has a suite of reports and data to analyse, all either supporting or opposing a given interpretation/hypothesis, and being seemingly robust or of more dubious value. The traditional approach under these circumstances is to invoke a 'levels-of-evidence' analysis and this becomes particularly important when elements of BACI-type designs are missing or alternative study design methodologies as detailed above are used (Downes et al. 2002). The issue here is not so much one of whether the appropriate experimental design was used or even whether the correct variables were monitored in a single study, but what can be done about the increased level of inferential uncertainty that arises from such cases. The approach uses causal criteria (effectively a set of circumstantial arguments) to improve the inferential strength of monitoring design (Downes et al. 2002).

The levels-of-evidence approach may include criteria such as:

- Strength of association: how strong is the response relationship?
- Consistency of association: has the association been observed repeatedly in different studies?
- Specificity of association: is the association generalised, or specific to a single driver?
- Temporality: does the response occur after the accepted or supposed cause?
- Gradient response: is there a gradient in the ecological response to different levels of the putative impact or stressor?
- Plausibility: is there a biologically plausible link between the response and the putative cause?
- Coherence: is the proposed cause-and-effect relationship in conflict with other known associations?
- Experimental evidence: is there any experimental evidence to support the association?
- Analogy: are there other cases where analogy could be used to interpret the data?

Downes et al. (2002) provide a comprehensive overview of the levels-of-evidence approach. It is likely to be useful in many cases when interpreting monitoring data from diverse sources on the condition of GDEs and the effectiveness of EWRs designed to protect them.

5.4.2. Field sampling program

Designing a field-sampling program involves selecting the indicators to monitor, the data collection methods to use, and the frequency of monitoring.

Choice of indicators and variables for GDEs

The choice of indicators and variables should be identified based on the conceptual model developed to describe the system (as discussed in Section 4). There are two types of indicators that can be used to investigate environmental changes in natural systems:

- analysis of ecological structure (including abiotic indicators)
- quantification of ecological processes.

Note that for the purposes of this discussion, a clear distinction is made between the terms indicators and variables (see 'Some notes on terminology' on page 44).

A monitoring program based on analyses of ecological structure would generally quantify the physical, chemical and biological characteristics that define the GDE. This approach has provided the basis of almost all environmental monitoring programs in Australia in the past, and has mostly been framed in terms of quantifying physico-chemical variables or biological characteristics. Many of the applicable tools presented in *Part 2 Assessment Tools* (including T4, T6, T10-T12 and T14) can be applied to assess elements of ecosystem structure.

Physico-chemical monitoring typically quantifies water-quality variables such as salinity, pH, or concentrations of dissolved oxygen and nutrients (see Jolly et al. 1996). Biological monitoring attempts to infer ecological condition by assessing the abundance of a particular species, sometimes termed an 'indicator species', or by examining the community composition of aquatic biota, especially macro-invertebrates and less frequently plants, fish or birds (Cranston et al.1996). Biological monitoring is common for surface waters (Bunn 1995) but has been used less commonly for GDEs until very recently. For GDEs, microbes and invertebrates are likely to be the most relevant biota monitored (Tomlinson & Boulton 2010).

Although both the physico-chemical and biological monitoring approaches are useful, neither has generated a consistent and comprehensive assessment of the condition of a water body, or provided great insight into the ecological state of aquatic systems (Schofield & Davies 1996; Finlayson & Mitchell 1999). Indeed, the recent review of estuarine ecosystems by Elliot and Quintino (2007) concluded that monitoring programs that have been developed to detect human impacts on aquatic systems have almost always placed too great a reliance on structural indicators, which means they have generally been unable to differentiate between anthropogenic and natural changes.

The quantification of critical ecological processes underpins the alternative 'process-based' approach to developing a monitoring program. In this approach, the monitoring program measures the rates of critical ecological processes (Bunn 1995; Fairweather 1999). The term 'function' is often used interchangeably with 'process' to describe this approach. The advantage of monitoring ecological processes over merely describing changes in ecological structure is that, provided the processes are well understood, they are more likely to give a direct and holistic measure of the condition of the ecosystem. Process-based monitoring is a new area of development and only recently have practical advances been made in surface water monitoring programs. The main difficulties have been the cost of repeatedly measuring ecological processes, and then having a sufficiently robust underlying knowledge to know what changes in rates of the critical processes mean for ecosystem structure and function.

There is also the problem of knowing which of the diverse range of ecological processes should be monitored.

Examples of key indicators and variables that can be used for GDE assessments from an ecological structure or ecological process perspective are provided in Table 5-1. Regardless of whether measures of ecosystem structure or ecosystem process are being made, indicators used need to clearly connect the threat (or other identified monitoring driver) and the impact on the GDE (Parsons et al. 2011). Identifying the links between change driver and potential impact in the conceptualisation stage will aid in determining the correct type of monitoring approach, indicator and supporting tools that can be used. Thorough treatment of the conceptualisation process will also identify any data gaps in understanding the change driver and/or the potential impact and help to inform the types of indicators and variables required.

Some notes on terminology

The monitoring literature is filled with terms that mean different things to different people. It is essential that the different terms are defined and their meanings and linkages with other terms are made very clear, otherwise the diverse parties involved in a monitoring program will not be able to communicate without ambiguity. The term 'assets', for example, can have a number of different meanings, depending on the spatial scale of the definition. A clear distinction also needs to be made among the terms 'indicators', 'variables', 'units' and 'parameters'.

In State of the Environment reporting, an indicator is a physical, chemical, biological or socioeconomic measure that best represents the important elements of a complex ecosystem or environmental issue (Ward et al. 1998). In many cases it is appropriate to limit the term 'indicator' to mean a high-level element of an ecosystem that relates specifically to a given ecosystem value. This means there is a conceptual distinction between the high-level 'indicator' and the lower-level 'variable'. The term 'variable' should be used in its statistical sense, to mean the something that varies; that is, the property that is actually measured in the monitoring program and which illustrates an important and quantitative aspect of the higher-level indicator. Under this schema, water quality is an indicator and the individual components of water quality (such as salinity, dissolved oxygen concentration, nitrate concentration) are variables.

Units should be self-explanatory. Nevertheless, it is essential that units are stated; salinity, for example, can be given in units of conductivity (e.g. ECU and $\mu S\ cm^{-1}$) or in total dissolved solids (e.g. $mg\ L^{-1}$). In the case of salinity, incorrect conductivity units are sometimes used especially for hypersaline groundwater (see Williams & Sherwood 1994). Parameters also are taken to have their statistical meaning: the various ways that data can be summarised to show some measure of central tendency and variability such as mean, median, standard variation.

What makes for good environmental indicators and variables?

There is a large literature on factors to be considered in the selection of monitoring indicators and variables. There is general agreement (e.g. Cranston et al. 1996; Jolly et al. 1996) that the best indicator variables have the following characteristics:

- Ecologically relevant: the variable reflects a process that is important in maintaining the condition of the system under study.
- Socially relevant: it is obvious and important to stakeholders.

- Ease of capture: the measurements are easy to make.
- Cost: the measurements are cheap to make.
- Measurable: is the variable capable of being operationally defined?
- Standard method: is there a standard method for making the measurements?
- Interpretation criteria: can the data be interpreted easily?
- Scale-appropriate: is the variable measurable at a scale consistent with biological processes or management interventions?
- Error rate: is the variable likely to be associated with a high error in measurement?
- Response time: is the variable sensitive enough for measurement to respond adequately to the changes expected in environmental condition?
- Stability: is the variable relatively stable over time, or does it fluctuate?
- Non-redundant: does the variable provide unique information that is not supplied by other indicator variables?
- Context: is the variable relevant within the broader context of the monitoring program and regional natural-resource management?

Frequency and scale of monitoring

The frequency of measurement is dependent on the issue being examined and the variation in the indicators in both a temporal and spatial sense; for example, changes in quality and quantity of groundwater are usually much more gradual than those in surface waters (MDBC 2006). The level of variations should be recognised before a long-term sampling program is formalised. This step can occur through a pilot study, but for many abiotic variables, monthly sampling is normally adequate (MDBC 2006). Continuous monitoring equipment can also be a cost-effective method of obtaining abiotic data in highly variable systems. Further advice on specific groundwater monitoring methods is provided in Table 5-2.

The frequency and scale of monitoring biotic indicators can however be more case dependent. In very large groundwater management areas in particular, a sufficient number of representative GDE sites need to be chosen to cover the full spectrum of variation within the region, or at least to allow extrapolation of results with a high degree of confidence (Parsons et al. 2011). The frequency at which monitoring is undertaken also needs to be commensurate with the rate of change experienced and any lag effects. Some indicators and tools lend themselves to understanding rapid change (e.g. T3, T4 and T10) whereas others are more suited to understanding change over longer time frames (e.g. T13).

Regardless of the monitoring methods or indicators chosen, in many GDE assessments the ability to implement a fully comprehensive monitoring and evaluation program may be limited by resources (budgetary, time) or the scale of the GDE relative to the scale of the groundwater flow system (e.g. relatively small-scale GDEs associated with a large-scale groundwater flow system, such as the Great Artesian Basin mound springs). In such cases, the decision may be taken to focus monitoring activities on higher-value or higher-risk assets. Information obtained from these targeted locations can then be used—through upscaling or transfer of information (see Section 6)—to inform decision making for similar, but lower-value and lower-risk, locations. An important consideration in implementing this type of approach is however the resulting uncertainty.

Table 5-1: Recommended key indicators and variables for GDE assessments (adapted from Froend & Zencich 2002)

Indicators	*Variables*
Ecosystem structure	
Hydrology and hydrogeology	Groundwater levels Fluctuating water regimes (duration of wet/dry phases, seasonality) Groundwater pressures, abstraction amounts Surface water flow Surface water levels
Water quality	Electrical conductivity Total dissolved solids pH Nutrients Temperature Metals Sulfate Stable isotopes Dissolved oxygen Turbidity
Vegetation	Species diversity of plant and algal communities Cover and abundance of indicator plant and algal species Species evenness over time Weed index over time Regeneration index over time Canopy fullness/density of indicator species Community distribution and/or zonation change or distribution of indicator plant species along a gradient Size (height) and age structure of a local population Canopy health
Fauna	Presence/absence of indicator species Community composition Stygofauna species, abundance and/or community composition Microbial species, abundance and/or community composition Macroinvertebrate species, abundance and/or community composition Fish species, abundance and/or community composition
Climate	Rainfall Maximum temperatures Evaporation
Ecosystem Processes	
Biogeochemistry and ecosystem function	Rates of community respiration Rates of primary production Rates of nutrient cycling Rates of acid generation

Table 5-2: Useful references for groundwater monitoring methods

Guidelines for Groundwater Monitoring – Department of Agriculture and Food, Government of Western Australia

http://spatial.agric.wa.gov.au/combores08/docs/GW%20monitoring%20guidelines.pdf

These guidelines are for landholders, natural resource management officers and the general community, and are a guide to planning, implementing, and managing a bore network at a local scale, and include instructions on how to monitor a bore.

Murray–Darling Basin Groundwater Quality Sampling Guidelines

http://catalogue.nla.gov.au/Record/108013

This document provides a set of guidelines for groundwater quality sampling with an emphasis on regional monitoring networks. It provides a general overview for practical purposes and covers the elements of effective groundwater sampling and the basic capabilities for routine applications. The procedures for sampling from the bore site to delivery at the laboratory are provided; however, bore construction and development or laboratory analyses are not discussed. The document is a general field manual including sampling for physical parameters, major ions, metals, nutrients, pesticides and microbiology. It is not aimed for use by researchers requiring specialised sampling methods for specific studies. There is an emphasis on trying to include explanations for the various procedures.

SKM (2000) GSC Groundwater 1.1 *State Groundwater Network Monitoring. Procedure for the Measurement of Water Levels in Groundwater Observation Bores.* A report prepared by SKM for Department of Natural Resources and Environment, Victoria.

This report summarises an appropriate methodology that can be applied to the manual collection of groundwater pressure information, which ensures consistency, quality control and data integrity, and safety of operators. The procedure is based on common monitoring practise adopted for the State Groundwater Bore Observation Network, but has been made generic so that its application may be widespread.

ANZECC and ARMCANZ (2000) *Australian guidelines for water quality monitoring and reporting*. Summary. Australian and New Zealand Environment and Conservation Council, Agriculture and Resource Management Council of Australia and New Zealand.

http://www.mincos.gov.au/publications/australian_guidelines_for_water_quality_monitoring_and_reporting

The *Australian Guidelines for Water Quality Monitoring and Reporting* (the Monitoring Guidelines) provides a comprehensive framework and guidance for the monitoring and reporting of the quality of fresh and marine waters and groundwater.

Victorian EPA Groundwater Sampling Guidelines

http://epanote2.epa.vic.gov.au/EPA/publications.nsf/PubDocsLU/669?OpenDocument

The objective of this document is to foster practices that will assist with accurate and consistent determination of chemical and biological indicators of groundwater. Such practices will ensure that groundwater samples are representative of groundwater in the aquifer and will remain representative until analytical determinations or measurements are made.

5.4.3. Data analysis and reporting

The focus of data analysis is on discovering and understanding the dynamics of an environmental effect. It seeks to build evidence to accept or reject the hypotheses set in the initial stages of the program (ANZECC & ARMCANZ 2000). Effective data analysis is important to increase the understanding of the GDE under investigation.

One of the main challenges for the data analyst is to extract a ‘signal’ from an inherently noisy background environment. A number of statistical methods are available to assist with trend analysis. These range from simple descriptive tools, such as time series plots, to more sophisticated modelling techniques. The variability of a constituent or process is equally as important as, and sometimes more important than, real changes in level (ANZECC & ARMCANZ 2000). Data storage and sharing can also be challenging, particularly in an environment where multiple organisations are involved.

Another challenge for GDE monitoring and assessment is to determine the most appropriate way in which to analyse and present data. For each of the relevant variables or indicators used to define an EWR it is important to analyse and present the data in a way that is immediately relatable. For example, some measure of groundwater level is often included in EWR definitions, possibly relating to the maintenance of groundwater above or below a certain depth, maintaining an historical or seasonal variability, long-term increase or decrease in levels above or below a certain rate. In these cases, presentation of groundwater data as a simple hydrograph may not suffice. Frequency diagrams, exceedence curves, trend analysis or other statistical measures may be more informative. Ideally, reporting needs are incorporated into the initial study design.

5.5. Links between conceptualisation, monitoring and adaptive management

Monitoring and the subsequent evaluation of data collected follows logically from the construction of a conceptual model of any GDE (T2) and the management of such systems in an adaptive management framework. Four broad approaches to environmental management can be distinguished on the basis of relative uncertainty and the potential for control (Peterson 2005; Figure 5-4). In situations where there is a good degree of controllability (at least in principle) but high uncertainty, adaptive management is the favoured approach. In situations that lack either control or certainty, emerging approaches—such as scenario planning and resilience building—are more appropriate (Peterson 2005). Where there is good control and low uncertainty then continuation of current approaches may be warranted.

In most cases, monitoring and evaluation programs would normally include the collection of data over some period of time (to determine temporal trends) and be linked closely with ongoing management of the asset. In many cases, the management will be adaptive.

From the other perspective, monitoring should be a core component of any adaptive management framework so that management actions can be specific, tailored and responsive to changes in the environment. Using monitoring data, managers can identify trajectories of change in condition (positive or negative), or when critical thresholds are breached or at the point of being breached, all within a time frame where management interventions can be put in place to prevent the further decline in condition of the ecosystem.

Adaptive management is based on the precept of 'learning by doing'. For over 25 years it has been adopted as a highly effective approach to unified natural resource management and, in particular, the integration of monitoring, environmental management and research. Walters (1986) provided one of the earliest descriptions of the adaptive management approach, which was updated by Allan and Stankey (2009).

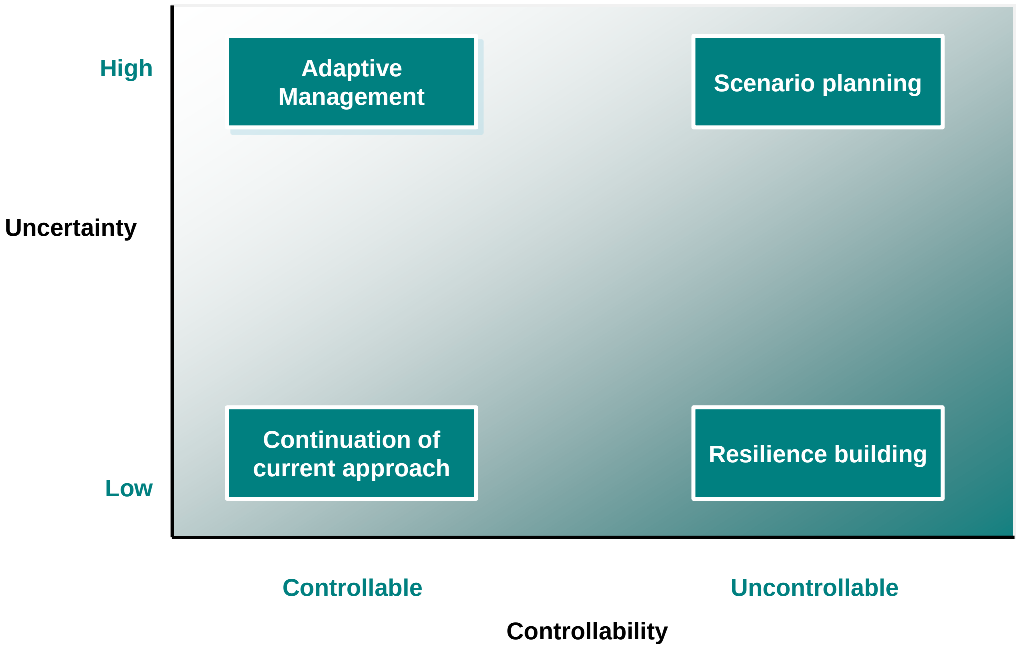

Figure 5-4: Four approaches to environmental management, based on the degree of controllability and certainty (modified from Peterson 2005)

The adaptive-management approach is shown diagrammatically in Figure 5-5, which shows the process to have a number of inter-related steps. Central to the approach is the formulation of a conceptual model. As discussed in Section 4, a conceptual model provides four functions:

1. It clarifies the problem, and ensures that the critically important components and the ecological interactions between them have been identified. It thus ensures that all the participants managing a given system are working from the same basic understanding as to how that system is structured and functions.
2. It identifies the 'knowns' and the 'unknowns', and thus the critical gaps in knowledge and where research investment needs to be focused.
3. It allows predictions to be made about the likely impacts of different management interventions, and thus which ones can be eliminated as unlikely to be useful.
4. It informs the design of monitoring programs as critical links between the abiotic and biotic components of the GDE are identified and data gaps become apparent.

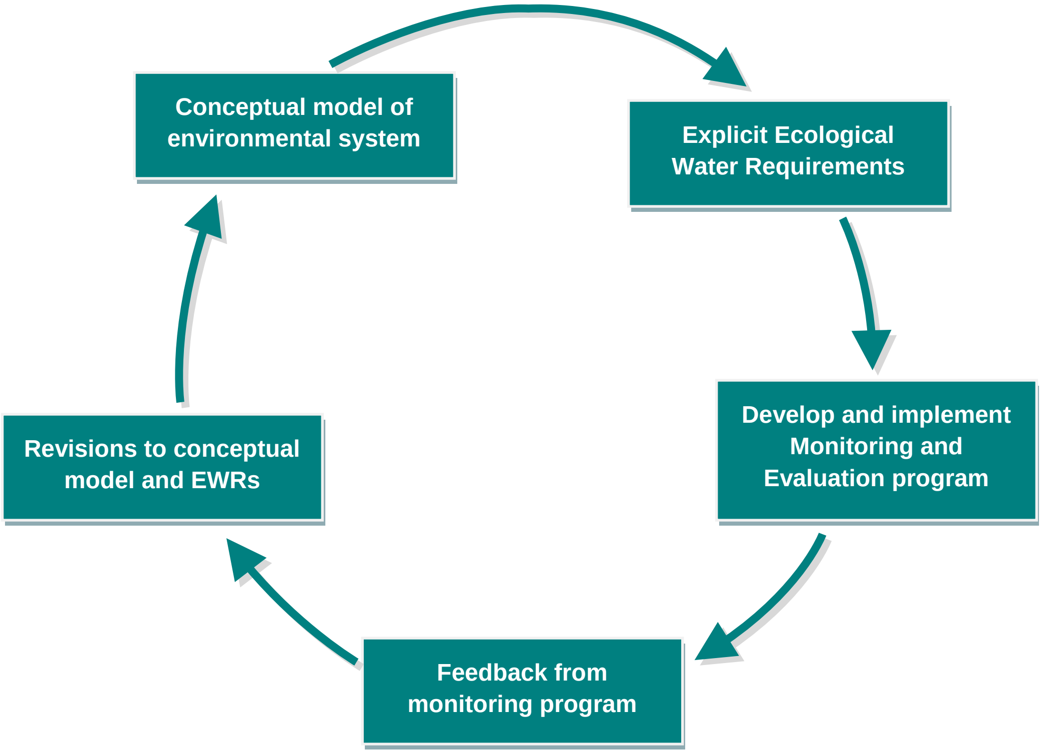

Figure 5-5: Summary of the adaptive management feedback loop for defining the ecological requirements for water of a GDE using a conceptual model and monitoring and evaluation approach

In combination with a set of explicit environmental objectives, the conceptual model allows the range of potential management interventions to be sifted and those most likely to result in the desired outcomes to be identified. These outcomes can, and should, be phrased as a set of predictions or hypotheses. After the management actions have been undertaken, the monitoring program is used to determine whether the predictions were substantiated. If they were, the conceptual model gains credibility. If they were not, the conceptual model has to be refined, and a second set of predictions made and tested in the next iteration of management interventions. In this way, environmental managers ‘learn by doing’.

6. Extrapolating EWRs over space and time

6.1. Introduction

Scientifically based, on-the-ground or *in situ* investigation is the traditional and most robust approach to determine the ecological water requirements (EWRs) of a GDE, and the tools presented in *Part: 2 Assessment Tools* are largely provided with this in mind. However, the water requirements of ecological assets within a water management area (or 'unit') are typically managed at a much larger spatial and temporal scale than that at which many of the tools can be applied. Resourcing constraints (i.e. capacity, time and budget) may also prohibit ground investigations, or monitoring and evaluation activities, at the required scale. In these circumstances, it may be appropriate and/or necessary to extrapolate existing EWR knowledge to broader areal or temporal extents.

Extrapolating EWR knowledge in Australia is common and usually occurs as a result of limited on-the-ground knowledge about the GDE that requires management (Parsons et al. 2011). The process generally involves taking knowledge based on robust evidence from one site or time and applying that knowledge elsewhere under the assumption that the same concepts apply. However, the approach is not without issues; extrapolating scientifically based information for management purposes is often challenging and can be characterised by large levels of uncertainty (Smith et al. 2008).

This discussion aims to capture some of the concepts of upscaling and transfer as they apply to EWR knowledge for GDEs, and provide some advice on limitations and uncertainties in the approaches available.

6.2. Upscaling and transfer for GDE assessments

In the process of upscaling there is an inherent assumption of homogeneity and belonging—that the area over which the scaling is undertaken bears similar characteristics to the original site, and that the site is nested within the broader area (Loveland & Merchant 2004). In many instances, upscaling is undertaken intuitively and without any formalised processes; for example, physical properties derived from a drill point or points are considered to be representative of the aquifer at that location. Upscaling differs from the concept of transfer; in transfer, information from one site or area is taken and applied in another site spatially removed from the original. The two sites between which information is transferred may share some physical characteristics but are likely to have known differences in at least some fundamental aspects.

In transferring or upscaling EWR information, a combination of approaches is typically used. In the first instance, remote sensing or GIS-based broad-scale approaches may be used as a first pass to identify and highlight known and potential GDEs at a landscape and management scale (see T1). It is common practice to use GIS to identify or define GDEs through the assemblage of spatially mapped datasets describing a variety of ecosystem components (e.g. Ransely et al. 2007), as is using remotely sensed data, typically in combination with other spatial datasets to achieve the same end (e.g. Dresel et al. 2010). There is no evidence that one approach is uniformly better than the other. However, results may be quite different within a single case, and the differences may reflect the constraints, biases and desired outcomes of the practitioner (Loveland & Merchant, 2004).

Following identification of potential ‘target’ GDEs, it then becomes a management decision to prioritise areas for ground-truthing and detailed assessment. Prioritisation may be based on ecosystem ‘value’ or resource pressure, or a combination of the two within a risk assessment framework. Following prioritisation, field-based techniques (e.g. ground-truthing, site-specific field assessments) can be undertaken to develop EWRs for prioritised sites utilising the assessment framework outlined in Section 3.

When considering extrapolation as an approach to defining EWRs for GDEs, understanding the nature of the available knowledge, and the inherent uncertainties, is critical. There are no clearly defined rules or hard and fast approaches for applying what has been learned from site-specific assessments to other locations as each situation will present different challenges and requirements. The overall objective of the process is however to take what is understood about EWRs at known locations and apply this knowledge where appropriate—through logic and reasoning—as a series of untested hypotheses to the target locations. When undertaking this process, reasonable limits to the extent of upscaling or transfer should be observed. Additionally, recognition that the best available science has been applied (as opposed to the best possible science) should be reflected in the level of certainty ascribed to the EWRs of target GDEs. To inform the extrapolation process, conceptualisation and/or models of the GDEs, (both the original and the ‘target’ sites) are particularly useful, irrespective of the quality and quantity of the data. Such models allow definition of the key components or descriptors of the ecosystems and allow identification of the similarities or differences (in both abiotic and biotic components) between the sites, thus indicating the level of confidence in the upscaling or transfer process.

General guidance around the limitations of upscaling or transfer relates to recognising the degree of similarity between any ground-truthed sites and target sites. In determining similarity any number of key physical components may be invoked. These may relate to physical attributes (such as climate, geology, landform, community composition) or general principles or ecosystem ‘behaviours’ such as mode of groundwater flow (e.g. local, intermediate or regional scale) or conceptualisation (e.g. emergent macrophytes present in low landscape positions). The more alike the studied and target GDEs are, the greater the transferability, the greater the ability to extrapolate and the more confidence in the approach. Conversely, the greater the degree of dissimilarity between GDEs, the higher the level of uncertainty.

6.3. Key considerations

Some key questions to consider when deciding whether upscaling is appropriate in a particular instance or which upscaling process or methodology should be applied are discussed below.

- **What is the spatial and temporal extent of the information available?**

  Ecosystems can be defined at any level of spatial scale (Forman & Godron 1981). When taking information from one GDE and applying it to a larger scale (upscaling) or to another GDE with similar characteristics (transfer) it is important to first define the scale of the existing information. Understanding the spatial scale extent will help to place limits on how far, and to what other locations, the existing knowledge may be applied. In terms of temporal scale, GDE information is often available only for a snapshot in time or a limited temporal extent. Effects of seasonality, drought cycles or longer-term drivers such as climate change may be unknown and this should be recognised in the upscaling or transfer process. Where the effects of these factors are unknown, the uncertainty in the process is increased.

- **Are the effects of scaling on the available EWR knowledge (data, relationships) understood?**

Ecosystems encompass both biotic and abiotic components, and understanding how any characteristics, as well as the relationship between them, may change with scale is essential to understanding any introduced uncertainty with upscaling or transfer (Joschko et al. 2008).

With regards to biotic components, ecological relationships in particular are often dependent on the scale at which they are defined and simple extrapolations that assume a linear-type scaling relationship will often not provide good predictions (Underwood et al. 2005). Ecological processes and patterns are better understood and described by cross-scale observations and modelling (Rietkerk et al. 2002).

Scale issues are also experienced with abiotic components, with commonly cited groundwater examples being hydraulic conductivity and determinations of recharge and discharge, all of which can vary by orders of magnitude depending on the scale and the method of assessment (Crosbie et al. 2009, Narsilio et al. 2009). Similarly, soil hydraulic conductivities at the landscape scale may differ from small-scale or point definitions of hydraulic conductivity (Raupach et al. 2009).

When upscaling or transferring EWR knowledge, it is imperative that the information available from the 'source' GDE is appropriate for upscaling (i.e. it is scale independent, or a relationship between scale and the process known). At a minimum, how this information on EWRs may change at larger scales should be understood.

- **Is the existing knowledge in a form that lends itself to upscale and transfer?**

EWR information may be specific to a particular spatial point and to a particular point in time, and for transfer or upscaling to be appropriate the EWR information must have a generic component. For example, an influx (defined as a volume or volume over time) of groundwater to a wetland or spring discharge most likely will not be directly transferrable to other groundwater-dependent wetlands or springs on the assumption of a linear or otherwise relationship between GDE size and flux, but will more likely be related to a particular feature of the ecosystem such as water levels being maintained within a certain distance of fringing vegetation.

- **What is the degree of similarity between the original site and the upscale or transfer area?**

Confidence in the process of transferring or upscaling knowledge is increased with increasing similarity between the original site and the upscale or transfer area.

In determining similarity between sites, both biotic and abiotic characteristics are important, and inputs from several knowledge bases (including geological, ecological, geographical, botanical, hydrological, hydrogeological, soil science and climatological) may be needed (Loveland & Merchant 2004). National or other ecosystem classification schemes or datasets may also be useful in identifying similar landscape properties and on the basis that ecosystems are closely related to climate, geology and geomorphology, spatial datasets along with floral/faunal coverages often provide further support for the process. At the national scale, these types of datasets include:

  - bioregions (source: Geoscience Australia): brings climate as well as ecology into the ecohydrogeological zones (EHZs)
  - hydro-ecological zones (source: Pusey et al. 2009): ecohydrology zonation

- groundwater flow systems (source: Australian Bureau of Agricultural and Resources Economics and Sciences): delineates regional, local, intermediate flow systems
- groundwater provinces (source: Geoscience Australia): defines hydrogeological extents such as the Great Artesian Basin and Murray–Darling Basin
- draft Watertable Aquifer map (source: Bureau of Meteorology): identifies watertable aquifers across Australia
- river basins and groundwater systems (source: MapConnect, Geoscience Australia).

- **What levels of confidence and uncertainty will result?**

An inherent assumption in any upscaling or transfer process is that the behaviour of the single system predicts the behaviour in a broader or different area (Loveland & Merchant 2004). Only through testing or validation of upscaled or transferred knowledge can the assumption of sufficient similarity or homogeneity be validated. This type of testing may involve integrated modelling, GIS, remote sensing and/or field-based observation. However, limitations of remotely sensed data in this capacity, particularly in verifying homogeneity, should be recognised (T1). Particular caution should be exercised where GDEs are subpixel sized or influence only subpixel extents.

When undertaking any extrapolation exercise, it is important to acknowledge (i) the increasing heterogeneity of systems at larger scales, (ii) the negative impact the lack of ground-truthing has on confidence in the results, and (iii) the complications that can arise from upscaling in strongly heterogeneous systems (Raupach et al. 2009). Each of these factors will impact on the confidence level in any results of the extrapolation exercise. Important limitations to upscaling are associated with an increase in spatial heterogeneity with increasing spatial scale; changes to species composition with changes in spatial scale; and species' behaviour that emerge at larger scales (e.g. the population, community or ecosystem scale) but are absent in smaller (and relatively homogenous) experimental settings (Underwood et al. 2005).

The extent of knowledge (and associated confidence) in particular disciplines also requires consideration. Using river corridor management and groundwater – surface water interactions as an example, abiotic components (geomorphology, hydrology and geochemistry) are generally better understood than biotic components (aquatic ecology, microbial ecology) and may lend themselves more easily to transferability (Smith et al. 2008).

In circumstances where extrapolation is undertaken, either through upscaling, transfer or a combination of both, it is important to recognise that extrapolated EWRs remain hypothetical to the target area until some field validation (i.e. ground-truthing) is undertaken. In the first instance, validation should be undertaken to assess the degree of similarity between 'original' and 'target' sites and secondly, to assess the appropriateness of the EWR. If this final step is not undertaken, extrapolated EWRs remain to a large degree hypothetical for a site with a large degree of uncertainty. It is important to note that following field validation, extrapolated EWRs still remain a 'best estimate' and will remain as such with respect to individual ecosystems until resources permit site or ecosystem-specific EWR investigations. Consideration of whether the resulting uncertainty is acceptable will require an assessment of uncertainty versus risk to the ecological asset, and consideration of uncertainty in the context of trade-offs when ecological provisions are determined. In the absence of any ground-truthing, the relevancy, accuracy, and therefore the effectiveness of any extrapolated knowledge remains unknown.

7. Responding to change

7.1. Introduction

A significant challenge in the management of GDEs is the determination of EWRs for systems that are in a state of change, where this change is rapid, different in nature to that historically experienced and/or otherwise substantial. Stage 3 of the EWR assessment framework details the key questions that need to be asked with respect to ecological response in the face of a driver of change; however, these questions are most easily answered in a system where the 'baseline' state of the GDE is known and the drivers of change can be clearly identified. Often this is not the case.

Common, and often difficult to separate, drivers of change to GDEs include climate variability (considered here to be intra- or inter-annual, or inter-decadal fluctuation), longer-term climatic change (anthropogenic or otherwise), and other threats such as over-abstraction of groundwater. These change drivers have the potential to alter the hydrogeological state of a GDE through changes to water availability and/or quality (Froend & Sommer 2010).

In any case, EWRs represent the intrinsic requirements of an ecosystem (or an attribute of it) for water and this remains unchanged so long as the availability of groundwater does not change beyond that historically experienced. EWRs therefore define groundwater requirements for an ecosystem under any hydrological impact scenario. Climate change 'impacts' (e.g. reduced or increased rainfall recharge of aquifers and increased ET) or other threats represent another process by which groundwater availability may be altered. The rate and permanency of hydrogeological change may however be significantly different (e.g. reductions in groundwater recharge rates experienced over a longer period of time compared to drawdown induced by abstraction experienced over the short term). With a change in the hydrogeology of an ecosystem beyond that historically experienced, the intrinsic ecological requirements for water also change as the ecosystem adapts, and the EWR may require redefinition (

Figure 7-1).

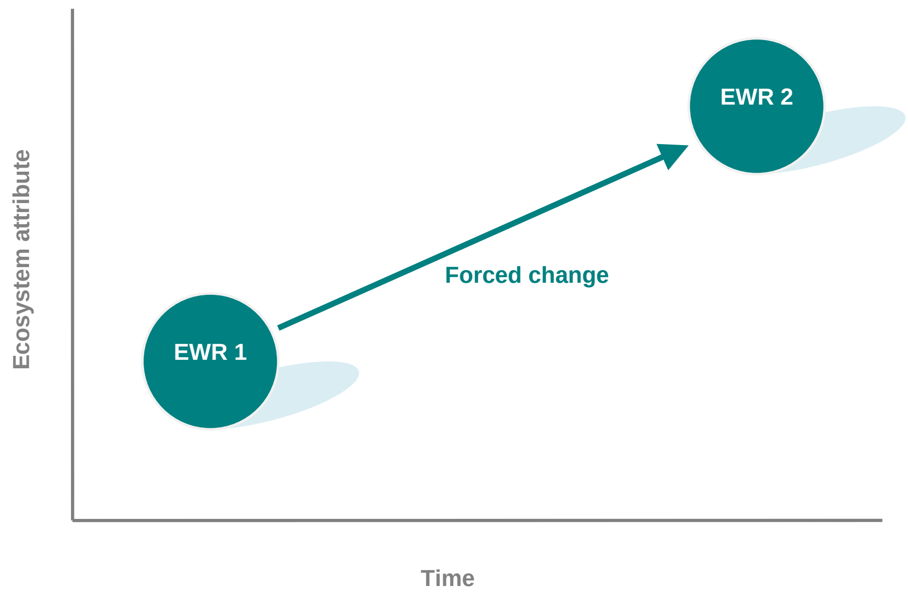

Figure 7-1: Effect of change on ecosystems and EWRs where EWR 1 represents the ecosystem's requirements for water under historically experienced conditions and EWR 2 represents that under altered hydrogeological regimes and ecosystem adaptation

7.2. Management considerations

In situations where groundwater availability is predicted to vary over the long term (e.g. rainfall reduction due to climate change), the following principles should be considered:

- The EWR remains unchanged provided the state of the ecosystem does not change. What do we know about the water requirements of the GDE at the time of assessment?
- Known EWRs need to be considered in the context of the predicted change. Will the hydrogeological change be beyond what the ecosystem has historically experienced (i.e. beyond assumed tolerance limits of the GDE) and therefore are the groundwater requirements of the GDE unlikely to be met in the short or long term?
- If the EWRs are unlikely to be met, what are the likely ecological responses to reduced or increased groundwater availability over the time frame in question? (This is where Stage 3 assessments can provide valuable information.)

What follows is essentially a management response. Are the likely ecological responses to altered groundwater availability acceptable given management objectives? If not, what are the possible (and predetermined) actionable management responses?

Some possible management responses include:

- set ecological water provisions (EWPs) that account for variability and change in climate—communicate consequences of variability and change to stakeholders
- monitor change in soil/groundwater-ecosystem-atmosphere continuum as part of a targeted monitoring and evaluation program to inform adaptive management
- review ecological and management objectives to determine if they are realistic under the altered groundwater regime

- investigate feasibility of altering the way groundwater is managed if there are unacceptable impacts
- consider feasibility of supplementing groundwater with an alternative source, subject to appropriate analysis of impacts of using alternate sources of water.

These responses will be informed by the individual GDE and the time frames over which a likely response to change is expected. Inherently, some GDEs are more susceptible to climatic variability and change than others and will require different management responses.

As a general rule, a less robust aquifer (a low storage-to-recharge ratio, local groundwater flow system) is more hydrogeologically susceptible to climatic variability. This does not necessarily imply that the associated ecosystem is more likely to be adversely affected; rather, it is more susceptible to variations in recharge processes. Surface water aquatic ecosystems located within environments where extremes are experienced (e.g. arid/semi-arid areas) are often more resilient and have the ability to manage variability in climate, water availability and water quality (Hudson et al. 2003; Kingsford et al. 1999). Resilience strategies can include acclimatisation to new or changing conditions, adaptations for feeding, water use or reproduction, or avoidance (Capon 2003; Costelloe et al. 2004; Nielsen 2003).

Ecosystems associated with more robust aquifers are less likely to have adaptations to deal with stress or alterations in water availability. This makes them more vulnerable to hydrogeological change over long periods even though the hydrogeologic system supporting them is less likely to be affected by climatic variability.

Low storage-to-recharge ratio, local groundwater flow systems are typically less robust and more susceptible to the effects of climate variability or longer-term climatic shifts than high-storage, regional groundwater systems. Biotic components of the hydrogeologically less robust systems however will more likely have adaptive strategies to deal with the naturally variable conditions (Shafroth et al. 2000). Understanding the likely response of the aquifer and the ecosystem will help inform any management response to climatic change.

Understanding the response of the particular biotic components of a GDE will help to inform management during climatic or other changes. For example, some groundwater-dependent vegetation can adapt to changes in groundwater level by extending their root networks to greater depths in the soil (maximum rooting depths range from 0.3 m to 68 m for different plant species; Canadell et al. 1996). Individual species may however be sensitive to the rate of watertable decline. Banksias growing over the Gnangara (groundwater) Mound in Western Australia were found to tolerate declines in groundwater level so long as they occurred at a rate of less than 0.5 m per year (Groom et al. 2000).

7.3. Understanding drivers of change

Examples of Australian studies that contribute to our understanding of GDE response to clearly defined but multiple drivers of change are few.

One of the best known examples of ecosystem response to both climatic change and groundwater abstraction is provided from the Gnangara Mound in Western Australia (see Case study 6 in Appendix A to this report) where change in the ecohydrological state of Gnangara phreatophytic vegetation has over time been driven by both drought and groundwater abstraction (Sommer & Froend 2010).

Approaches taken to understand the effects of climate change, and how these differ to extraction-related drawdown include monitoring and evaluation (long-term monitoring of several ecosystem components allowed relatively comprehensive conceptualisation and

understanding of the impacts of climate change and groundwater abstraction on phreatophytic vegetation) and spatial risk or threat assessment (to complement the spatial risk-based approach to GDE management).

Details regarding the specific methods used in the Gnangara Mound study are provided in detail in the case study. Some further background to these and other approaches to assist in the separation of climatic and groundwater extraction effects follows.

7.3.1. Monitoring

Monitoring methods allow a retrospective assessment of impacts of change on a dependent ecosystem. Targeted observation of a system subject to a single or multiple change driver(s) (e.g. climate change, drought and groundwater extraction) can help to identify triggers or patterns that discern the effects of each change driver and hence what can be expected from future predictions. Further discussion on monitoring and evaluation in the GDE context, including setting up monitoring programs and effective design, is presented in Section 5.

As an example of the types of outputs that can be derived from long-term monitoring, the following generalisations are offered (based on the observed responses of phreatophytic ecosystems in Western Australia subject to climate change, drought and groundwater extraction, Froend & Sommer 2010; Sommer & Froend 2011):

- gradual changes (in this case climatic changes) result in progressive transitions of ecological state
- rapid changes (in this case groundwater abstractions) can result in system threshold responses or breaches and transitions to alternative stable ecohydrogeological states.

For these types of GDEs, at the community or ecosystem scale, the rate, magnitude and duration of watertable drawdown primarily determines the short- and long-term impacts on dependent vegetation (Scott et al. 1999; Shafroth et al. 2000). Rapid declines in groundwater levels and separation from the watertable result in the acceleration of impacts and likely threshold responses (Segelquist et al. 1993; Lite & Stromberg 2005), whereas gradual reductions provide a greater opportunity for recovery via recharge to the system and/or mitigation of the effects of water stress and allow time for plants to adapt (i.e. root growth, physiological adjustments) in the shorter term, with progressive vegetation change occurring over the long term.

7.3.2. Risk assessment

Risk assessments provide a mechanism to make an indicative assessment, via a threat analysis, of the likely impacts of changes, such as climate variability and/or groundwater abstraction, on a GDE. An advantage of risk-based approaches is that they are predictive, can be used over large spatial extents, can be integrated with GIS information and allow assessment of multiple scenarios. Outputs may be qualitative or quantitative depending on assessment structure and inputs. More qualitative risk analysis approaches are particularly useful in data-poor areas and have been undertaken throughout South Australia to identify risks to GDEs from groundwater abstraction (SKM 2010c; prepared for the Adelaide and Mount Lofty Ranges NRM Board SA, and REM 2005; prepared for the South East Catchment Water Management Board SA).

Typically, risk analyses define and develop relationships between (i) the risk to the system as a function of the severity of the threat (in this case either climate variability or groundwater abstraction); (ii) the likelihood of the threat impacting an asset (which may be represented

qualitatively or quantitatively); and (iii) the consequence, which would be a measure of conservation value associated with the asset (REM 2005), or a likely response. This type of simple analysis can be adapted to suit the purpose; for example, by determining the risks posed by groundwater extraction and climatic change to a GDE relative to each other.

In more complex situations (e.g. where there may be multiple change drivers) risk analysis approaches can be developed within a Bayesian Belief Network (BBN). BBNs allow multiple relationships between causes and effects to be defined on the basis of probability and can be coupled with spatial data. An advantage of the BBN approach is the clarity in which relationships are established and expressed within the model, and the ability to transparently incorporate uncertainty. Although multiple 'threats' can be included in a network, feedback loops are generally not well managed (Hart & Pollino 2009). The Land Use Impact Model, which includes a wetland module, allows for rapid assessment of risks to ecosystems from multiple drivers (e.g. climate variability and/or groundwater abstraction) using Bayesian networks, exemplifies the type of approach appropriate for these types of assessments (DPI 2011).

A BBN or other risk assessment method may provide qualitative or quantitative outcomes, depending on the nature of input data. Regardless of data type, to adequately assess the risk of a GDE to change(s), it is paramount to understand the likely response of the abiotic and biotic components of a system. Observation (through monitoring and evaluation) of groundwater system response to historical changes in climate (seasonal or longer term) can give an indication of the aquifer(s), and by extension, the associated ecosystem's susceptibility to climate variability.

7.3.3. Modelling approaches

Modelling approaches use available data to create an empirically or physically-based model that describes system response to drivers. The information derived may be used in a stand-alone manner or incorporated into a risk assessment or other framework. An advantage of modelling approaches is that temporal effects—such as changes over time and lags in response—can be included. Additionally, in the case of differentiating the effects of climate variability and groundwater abstraction on GDE function, either one or both of these drivers can be included in a model and used in a predictive sense. A model designed to meet these ends can vary in complexity, depending on the level of understanding of GDE function, and amount and quality of data available.

One of the simpler modelling approaches to separate climate from abstraction effects is to use correlative statistics to define a basic relationship between groundwater levels (which is assumed to be related to GDE function) and rainfall (which is used as a proxy for climate). **Error! Reference source not found.** shows where a relationship could be established between the cumulative deviation from average annual rainfall and depth to watertable (SKM, 2010d; prepared for Eyre Peninsula Natural Resource Management Board, SA). This relationship may then be used to infer future groundwater trends from climate predictions and, from there, determine if groundwater abstraction or climate is the main driver changing groundwater levels.

At the other end of the complexity scale are numerical groundwater models. Such models can be interrogated to predict, or retrospectively be used to determine, the effects of multiple drivers on groundwater levels that support a particular GDE. More information on these types of models is presented in the tool T14.

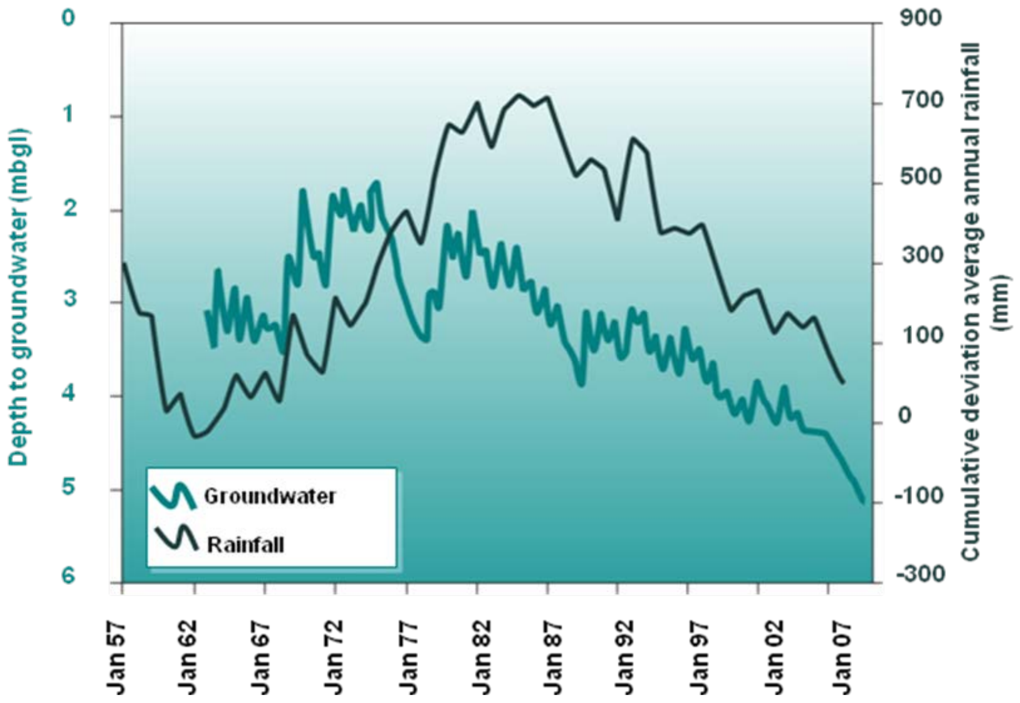

Figure 7-2: Depth to watertable compared to cumulative deviation from average annual rainfall (adapted from SKM 2010d; prepared for Eyre Peninsula Natural Resource Management Board, SA)

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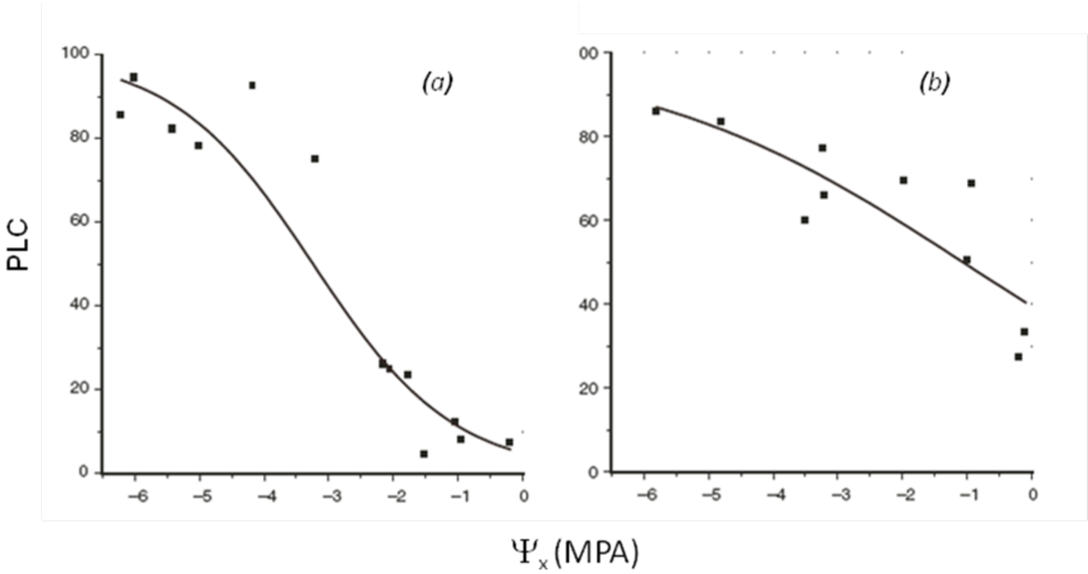
Appendix A: Case studies

Case study 1
Phreatophyte response to reduced water availability

For some GDEs dependent on the subsurface presence of groundwater, groundwater often forms only part of the overall ecosystem water requirements. This will especially be the case where rainfall is seasonal and the soil water store has the potential to be regularly replenished. Where there is insufficient soil water to meet plant water requirements (either seasonally or annually), plants that can access groundwater will become increasingly dependent on that water source as the soil water store is depleted. In these situations, there is a risk of plant water stress occurring if for some reason (e.g. groundwater pumping) the watertable is in decline and dependent vegetation cannot extend its root systems to maintain contact with the capillary fringe.

Froend and Drake (2006) presented the results of a study to assess the response of banksia woodland species to groundwater drawdown. The researchers developed a response function that related percentage loss of plant conductance (shown as PLC on the vertical, or y, axis on Figure A1), the ability of a plant to move water along water conductance pathways (principally the xylem vessels), and risk of tree mortality to drawdown, as represented by altered xylem water potentials (Ψ_x) (Figure A1). The research outcomes are a crucial step in quantifying the critical ecophysiological response to *Banksia* woodland to reduced water availability.



Note: The plots show that for *B. menziessi* there is an approximate PLC of 20% for a Ψ_x of -2 MPa, and for *M. preissiana* there is an approximate PLC of 60% for a similar Ψ_x, indicating *M. preissiana* is less drought tolerant than *B. menziessi*, consistent with their relative dependencies on groundwater.

Figure A1: Ecological response function for *Banksia menziessi* (a) and *Melaleuca preissiana* (b), relating xylem function (maintenance of plant water conductance pathways) to water availability (adapted from Froend & Drake 2006)

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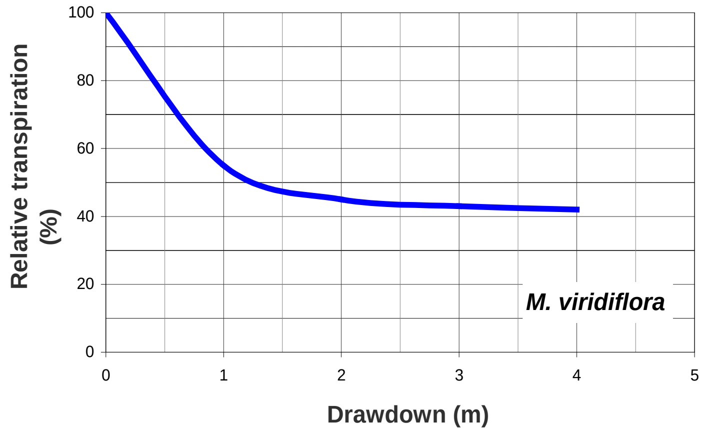
Case study 2
Plant water use model for developing an understanding of the response of phreatophytic terrestrial vegetation to change

Water extraction by vegetation is often assumed to be proportional to the difference between leaf water potential and soil water potential (T3). Leaf and soil water potential are both negative, but where leaf water potential is more negative than soil water potential there is an upward gradient that can drive water movement through the plant. As the watertable declines, soil water potentials also decline (i.e. they become more negative) and the pressure differential driving plant water uptake reduces.

Howe et al. (2006) reported the results of plant water use modelling that simulated plant transpiration response to reduced water availability (watertable drawdown) (Figure A2). Model input parameters included soil matric potential profiles, root distribution and leaf water potentials. Stable isotopes were used to determine vegetation rooting depth. The model conservatively assumed that leaf water potentials and root distribution do not change in response to reduced water availability, and so the reduction in transpiration is proportional to the reduction in water potential difference between the leaf and the soil (see Cook & O'Grady 2006 for model details).



Note: The model demonstrates that *M. viridiflora* has a much-reduced water availability, with transpiration rates predicted to reduce by 50% if the water table is allowed to decline by 1 m or more.

Figure A2: Ecological response of *Melaleuca viridiflora* transpiration in response to a declining watertable

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Case study 3
Designing a monitoring program: hyporheic zone of streams in the Flinders Ranges, South Australia

Define the issue

The hyporheic zone is the saturated interstitial space below the streambed and adjacent stream bank where surface water and groundwater mix (Tomlinson & Boulton 2010). The hyporheic zone is a dynamic ecotone in that its boundaries fluctuate according to variations in the rate of exchange between surface water and groundwater. This exchange is regulated by two interacting physical factors: i) sediment structure, especially permeability; and ii) vertical hydraulic gradient (VHG), which in turn are controlled chiefly by two geomorphological factors: i) the slope of the streambed; and ii) stream meandering (Winter et al. 1999; Malard et al. 2002; Kasahara et al. 2009). Vertical hydraulic gradient is the difference in pressure head between surface and subsurface water at a given depth (Kasahara et al. 2009). It has a direction (downwards into saturated sediments in downwelling zones, and upwards in upwelling zones) and magnitude (the pressure gradient).

High rates of streamwater–groundwater exchange in upwelling zones can maintain a high sediment permeability by flushing fine sediments from pore spaces. Conversely, downwelling streamwater can draw fine particles into the sediment and clog the streambed. The exchange of water can be influenced also by the growth of algal mats on the sediment surface, or fungal mats/biofilms deeper in the sediment according to the availability of light, dissolved organic carbon, and nutrients. Excessive abstraction of groundwater or streamwater, especially during low-flow periods, can have extreme effects on sediment structure and VHG, and thus on the interchange of water from the stream surface into the hyporheic zone (Kasahara et al. 2009).

The hyporheic zone—with its associated mosaic of upwelling/downwelling zones and patterning of surface-water expressions—plays a crucial role in nutrient cycling and primary production in arid-zone streams (Boulton 1993). It does this mostly by mediating changes in physico-chemical variables, especially dissolved oxygen concentrations (Tomlinson & Boulton 2010). Rapid rates of nitrification (the bacterial conversion of ammonium to nitrate under oxic conditions) in the hyporheic zone cause nitrate-enriched waters to upwell into surface waters at the end of hyporheic flowpaths, and this nutrient enrichment results in thick mats of epibenthic algae growing at the sediment surface. Thus, streams with many active hyporheic zones are predicted to cycle nitrogen more rapidly than those with few zones (Boulton et al. 2010) and are also likely to show higher rates of gross primary production, via rapid algal photosynthesis. The increased rate of algal primary production would promote higher rates of secondary production by stream invertebrates, and thus strongly affect the structure and function of streams in arid and semi-arid zones of the country.

Given that the size and functioning of hyporheic zones in ephemeral streams is controlled so closely by surface water and groundwater dynamics, the issue is that abstraction of either, especially in water-sensitive arid or semi-arid environments, can have drastic effects on stream structure, function and value.

Develop a conceptual model

Exchange between stream waters and groundwater in the hyporheic zone is controlled at the landscape scale by changes in the vertical and horizontal direction of the stream; in other words, by changes in streambed slope and stream meandering. Figure A3 shows a conceptual model illustrating these processes. At the reach scale—the scale most often studied by stream ecologists—the hyporheic zone is delimited primarily by variations in sediment characteristics and VHG. The conceptual model in Figure A4 shows that changes in the slope of the streambed result in alternating upwelling and downwelling zones, with the conversion of ammonium to nitrate within the hyporheic zone, and the emission of nitrate-enriched waters at the end of hyporheic flowpaths. The growth of benthic algae is promoted when these nutrient-enriched waters re-enter the stream, resulting in dense accumulations of benthic algae on the sediment surface.

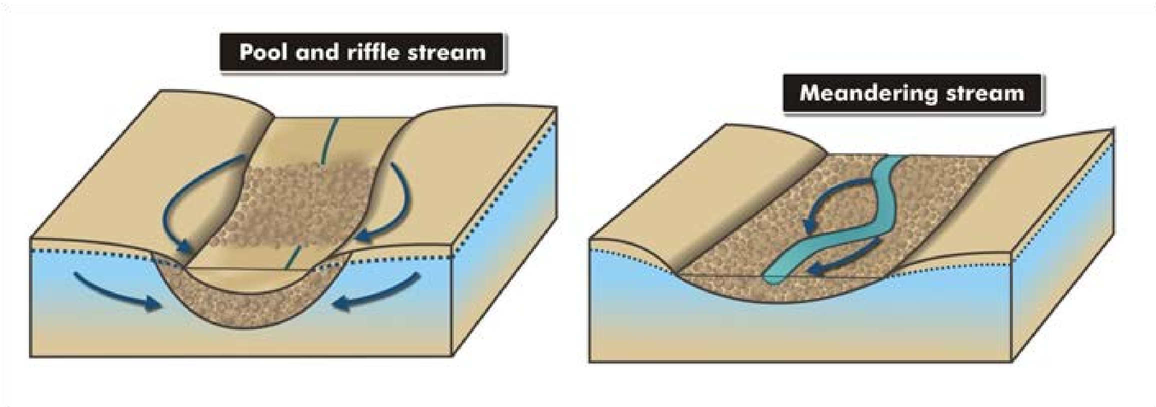

Figure A3: Conceptual model of hyporheic zone in arid zone streams: importance of geomorphological factors at the landscape scale (from Winter et al. 1999)

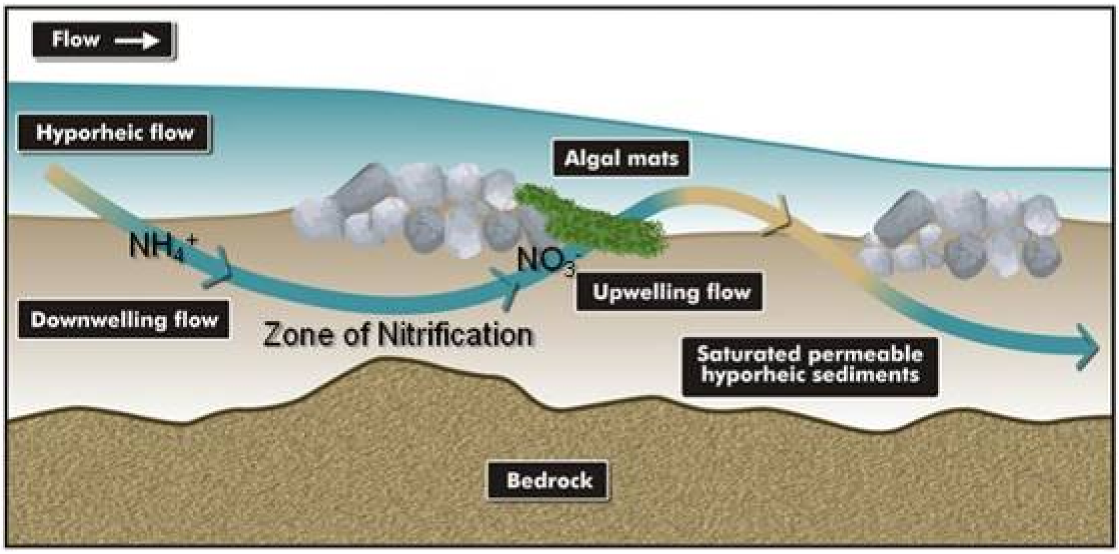

Figure A4: Interactions between streambed topography, sediment permeability and surface flow characteristics (modified from Boulton et al. 2010)

Set objectives

The monitoring program would aim to detect changes in the location, size and ecological functioning of hyporheic zones at the reach scale of arid-zone rivers. It may be postulated that excessive abstraction of groundwater, say for a tourism development, would lead to altered patterns of exchange between surface and groundwater in nearby ephemeral streams. The specific objectives of the monitoring program therefore would be to:

- determine the location and relative extent of upwelling and downwelling zones, including rates of hydrological exchange
- quantify physico-chemical characteristics of surface and hyporheic waters, allowing some inference of biogeochemical activity
- quantify abundance and taxonomic composition of hyporheic fauna
- determine the location, biomass and primary production of benthic algal mats
- select a study design and field methods.

Boulton (1993) outlined the suite of techniques that are available to study the hyporheic zone, and updated that review in 2010 (Boulton et al. 2010) with a summary of technological advances over the intervening two decades, including the use of tracer dyes (T12), estimation of nutrient-retention factors, and advances in numerical modelling approaches (T14). Practicable ways in which the ecological condition of the hyporheic zone could be assessed were also collated and assessed by Boulton (2000).

Mini-piezometers and hydraulic potentiomanometers can be used to measure the extent of upwelling and downwelling zones and VHG. Hand-pumping hyporheic water from mini-piezometers is a way of collecting samples for physico-chemical analyses (e.g. concentrations of dissolved oxygen, ammonium and nitrate, temperature and redox) and of hyporheic organisms for subsequent sorting and analysis. Seepage flux between surface and subsurface waters can be measured with seepage meters (T10, Boulton 1993). Analysis of breakthrough curves following the injection of conservative (e.g. bromine) and non-conservative (e.g. nitrate) tracers (T12) can be used to make reach-scale estimates of the size of the hyporheic storage zone and nutrient-retention factors (Triska et al. 1989). Invertebrate fauna, biomass and productivity of algal mats, and stream respiration/productivity can be quantified with traditional limnological methods (Wetzel & Likens, 1991).

Design a field-sampling program

A field-sampling program must consider the range of spatial and temporal scales at which surface waters and groundwater interact in arid-zone streams; scaling issues are discussed in Malard et al. (2002), Karsahara et al. (2009), and Tomlinson and Boulton (2010). For pragmatic and interpretative reasons, most monitoring would be undertaken at the reach scale. The field-sampling program would include the establishment of mini-piezometers and hydraulic potentiomanometers at various distances along a range of streams likely to be affected by the development (‘impact’ sites), as well as along streams not likely to be affected (i.e. ‘control’ sites). In this scenario, there would be no reason why a classic BACI-type design could not be implemented as there should be adequate warning of the development taking place and thus ample opportunity to establish and monitor before-impact field sites.

The distance between consecutive devices would be determined according to the predicted frequency of upwelling and downwelling zones (which can be inferred in the first instance by visual inspection, on the basis of the patchiness of algal growth and/or water temperature).

As devices might need to be placed every ~50-100 m (e.g. see Grimm et al. 1991), large numbers of mini-piezometers may need to be used and monitored, especially if both impact and control streams are assessed.

The timing of routine sampling of the devices would be contingent largely upon practical considerations given that arid-zone streams are likely to be in remote areas not easy to access. Sampling every two months may be warranted initially, supplemented by event-based sampling according to need. Variables such as VHG and the range of physico-chemical indicators should be monitored at every field visit; other variables, such as invertebrate assemblages, benthic algal mats, and tracer-injection trials would need to be undertaken less frequently, perhaps twice yearly but with due acknowledgement given to the temporal scale at which the various indicators are likely to vary in the field.

A large number of samples for chemical and biological analysis would be generated during the field sampling, and resources must be available to process them upon return from the field site.

Data analysis and reporting

Data analysis would be directed initially to answer the four specific objectives outlined above. The BACI-type design necessitates relatively complex statistical procedures (e.g. see Quinn & Keough 2002), and a skilled statistician would be required for data analysis. Once sufficient data have been obtained, trend analysis would be undertaken to determine whether temporal patterns (e.g. seasonal, event-based, or inter-annual) exist between different indicators, any causal relationships can be identified and how these can be used to (e.g. determine an EWR for impact streams or inform management actions for the future etc.). The monitoring program should be re-examined at adequate junctures to determine whether it has sufficient resolution to answer the stated objectives or whether it could be reduced in scope in the light of knowledge gathered to date.

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Case study 4
Designing a monitoring program: coastal vegetation – salt marshes and other estuarine wetlands in south-eastern Australia

Define the Issue

Coastal wetlands usually represent the lowest point in regional and local systems of groundwater flow; accordingly, groundwater often discharges into them, sometimes in quantities large enough to affect the salinity of the estuary (Carter 1996).

Recently it has been realised that groundwater can alter not only the salinity of coastal waters into which it discharges, but also plays a critical role in supplying nutrients to plants in coastal wetlands, including reedbeds, salt marshes and mangroves. Such coastal wetlands play critical roles in providing habitat and food to higher consumers in the estuary, including recreationally and commercially important species, such as fin fish and shellfish (e.g. see Crinall & Hindell 2004; Hindell & Jenkins 2004; Spencer et al. 2009).

There are a number of cases where the discharge of fresh groundwater is likely to have, or indeed does have, demonstrable effects on estuarine salinities and patterns of vegetation. In brackish lagoons and the upper reaches of estuaries along the New South Wales coast, for example, halophytic salt marsh can merge into reed swamps dominated by glycophytic (salt-intolerant) taxa such as *Phragmites australis* and *Schoenoplectus*, *Bolboschoenus* and *Typha* spp., with the patterning controlled by the balance between freshwater and saltwater inputs (Adam 1994). Similarly, coastal salt marsh in Victoria is often found in mosaics with a diverse range of other estuarine wetland types according to the interplay of tidal and freshwater (surface water and groundwater) influence. The seaward reaches of these systems are frequently vegetated by dense stands of rhizomatous perennials such as *Juncus kraussii* and *Phragmites australis*, and *Bolboschoenus caldwellii* and *Schoenoplectus pungens* in fresher areas (Sinclair & Sutter 2008). The relative ratio of inundation with tidally forced seawater and with fresh water is thought to be the factor that controls many aspects of salt marsh ecology (Adam, 1990; Boorman, 2009). Some of the interactions among tidal inundation, climate, groundwater levels, and soil salinity and waterlogging have been described for coastal wetlands in the Sydney region by Clarke and Hannon (1969, 1970).

Groundwater discharges can also import nutrients into estuaries and coastal wetlands. Nitrogen and phosphorus can be the limiting factor alone or even simultaneously for plant growth in estuaries, and temporal shifts can occur across seasons or years in a single estuary (Howarth 1988; Davis & Koop, 2006). Some studies have examined the role played by groundwater in estuarine nutrient dynamics (Valiela et al. 1990) and at least one (Krest et al. 2000) has shown that saline aquifers could contribute significantly to both the import and export of nutrients from a coastal salt marsh.

For the purposes of the toolbox, the overarching issue for this case study is the lack of knowledge around whether coastal vegetation of salt marshes and other estuarine wetlands in south-eastern Australia are influenced by groundwater discharge.

Develop a conceptual model

Because the importance of groundwater discharge into coastal wetlands is a newly developing and poorly studied field, there are no existing conceptual models that can be used to help underpin the monitoring program for this specific area (as with case studies 3 and 5 of this appendix). Figure A5 shows the conceptual model devised specifically for the toolbox in order to show the ways that groundwater discharge can influence salinity regimes and nutrient fluxes in estuarine wetlands in south-eastern Australia.

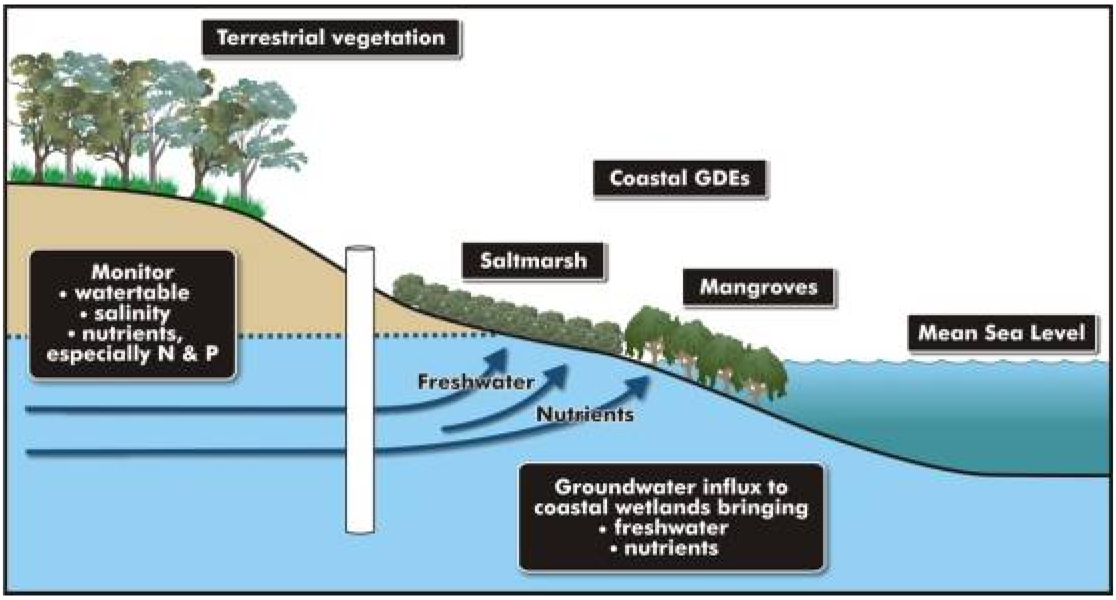

Figure A5: Groundwater discharge to estuarine wetlands

Set objectives

In light of the issue defined above, the monitoring objectives would be to determine whether changes to groundwater levels or water quality have an influence on the supply of fresh water or nutrients to coastal wetlands. The prediction would be that lowered watertables would lead to decreases in the supply of fresh groundwater to the coastal GDEs, with impacts on nutrient supply and salinity regimes in salt marshes, mangroves and other types of estuarine wetlands.

Select a study design and field methods

In this example, the monitoring design would primarily test the hypothesis that groundwater levels and/or nutrient concentrations had altered in response to an anthropogenic development inland. Although ideally it would require 'before' and 'after' sampling, unlike the case with arid-zone streams discussed in Case study 3 it is unlikely that a suitable control site could be found for coastal wetlands. Thus the experimental design would not have the full inferential power of a complete BACI design. The monitoring program would need to include measurements of depth to watertable, a small suite of groundwater physico-chemical variables (focusing on nutrients), and vegetation type, extent and condition. The program would monitor groundwater discharge along a defined front in a coastal wetland rather than across a large expanse of a borefield, as would be necessary in Case study 5. There would also likely need aerial photographic interpretation (API) of vegetation, combined with extensive ground-truthing (Fensham & Fairfax 2002). Fieldwork would also be required to determine vegetation condition, especially the incidence and success of plant recruitment.

Design a field-sampling program

An array of piezometers traversing the wetlands of interest would be used to monitor depth to watertable. They would be sampled monthly, with care taken to ensure that tidal influences were taken into account, either around the time of sampling or via post-collection data treatment. Water samples from the piezometers would be taken for salinity measurements and to determine nutrient concentrations. Because of budgetary constraints, it is likely that only the total form (total nitrogen, total phosphorus) of nutrients would be routinely measured. If funds permitted, a complete analysis of dissolved inorganic species (ammonium, nitrate, nitrite, soluble reactive phosphorus) should be undertaken, at least at quarterly intervals.

Vegetation surveys would include the use of aerial photographs (probably at five-year intervals) with extensive ground-truthing to confirm the accuracy of API. Fieldwork would not only confirm vegetation type and boundaries, but would also allow an assessment of vegetation condition and of vegetation characteristics that cannot readily be undertaken with API, such as the presence of diagnostic, cryptic or rare, threatened or endangered species. Estuarine wetlands contain a large number of annual plant species (Boon et al. 2011), so vegetation assessments would have to be timed to capture data on these plants.

Data analysis and reporting

In line with the objective above, data generated from this monitoring program would be used primarily to infer the discharge of freshwater and flux of nutrients to coastal wetlands via groundwater, and only secondarily in terms of the direct effect of altered watertables on vegetation. Initially, flux estimates would be determined and then compared across years to assess whether the development in the hinterland has altered the pattern of nutrient delivery to the wetland. Data collected from API and vegetation surveys could then be assessed in conjunction with the flux data analysis to identify any potential impacts of altered waterables and nutrient supply reduction.

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Case study 5
Designing a monitoring program: terrestrial and aquatic vegetation of the Tomago Sandbeds, New South Wales

Define the Issue

The Tomago Sandbeds cover an area of ~10 000 ha in a coastal barrier setting just north of Newcastle, New South Wales. They support a wide range of terrestrial and aquatic vegetation, including swamp and swamp woodlands, sedgelands and other wetlands, dry open forests, and wallum heath and woodlands. Over 20 plant taxa are listed as regionally important under state legislation, and seven as nationally threatened under the E*nvironment Protection and Biodiversity Conservation Act 1999*. The sandbeds are part of the urban water supply for the lower Hunter Valley, managed by Hunter Water Corporation (HWC). They currently supply 20% of the total supply, and are one of the longest continuous borefield operations in Australia. Although the HWC is the main user of groundwater, abstraction also occurs for stock and domestic use, industry and commercial tanners, and the Williamstown RAAF base. The sandbeds were used as a case study in the recent review of the management of GDEs by Parsons et al. (2011).

The management issues are summarised by Parsons et al. (2011, p. 92) as follows:

> *there is some evidence that historical abstraction practices have driven a gradual change in vegetation, but because of the pulsing nature of the abstraction, the decreased demand for further abstraction and the relatively constant inter-annual rainfall patterns, these vegetation changes have been small and are considered acceptable.*

In future, however:

> *planning must account for possible periods of severe future drought and the sandbeds being one of HWC's emergency supplies. The strategy should include a contingency for modifying abstraction if recharge is reduced (i.e. due to climate change) and the vegetation/groundwater equilibrium cannot be maintained by mimicking historical abstraction regimes.*

In addition to lowering of groundwater levels and possible effects on vegetation, an issue exists with pyrites (iron sulfide) in the lower depths of the sandbed aquifer. When the watertable falls after the abstraction of groundwater, pyrites in these layers are oxidised and groundwater may become contaminated by sulfuric acid and heavy metals. Also, the proximity of the aquifer to the ocean, to tidal creeks around Port Stephens to the east, and the Hunter River to the west means that intrusions of seawater are also a major risk should groundwater levels fall excessively.

Develop a conceptual model

Parsons et al. (2011) discuss the conceptual model of surface and groundwater flows in relation to freshwater and saltwater interfaces and GDEs in the Tomago Sandbeds region (Figure A6).

Set objectives

Detailed vegetation surveys of the study area had earlier demonstrated that 'all vegetation across the sandbeds was considered to depend on groundwater to some degree' (references within Parson et al. 2011, p. 103). The degree of groundwater dependency was calculated on the basis of depth to watertable, which was used to define four classification:

- Obligate wetland or seasonal inundation GDEs: 0 m to 1 m median depth to watertable
- Obligate GDEs: 1 m to 2 m median depth to watertable
- Obligate/facultative mixed GDEs: 2 m to 3 m median depth to watertable
- Facultative: greater than 3 m median depth to watertable.

Accordingly, it can be predicted that when groundwater levels decline greater than 1 m below the surface the first vegetation group will be threatened, whereas obligate/facultative mixed GDEs will be threatened with drops greater than 3 m below the surface. Similar predictions can be made for individual plant species (e.g. see Figure C3 of Parsons et al. 2011 for the example of a Callistemon-Lepidosperma shrub swamp).

Although the effect of groundwater levels on vegetation type and condition is a critical issue in the study area, pyrites oxidation and seawater intrusions are important too, not only for vegetation condition but also for the quality of the groundwater to extractive users. In addition to groundwater levels, water-quality objectives therefore need to be set for pH and metal concentrations (a function of pyrites oxidation) and for salinity (a function of saline intrusions).

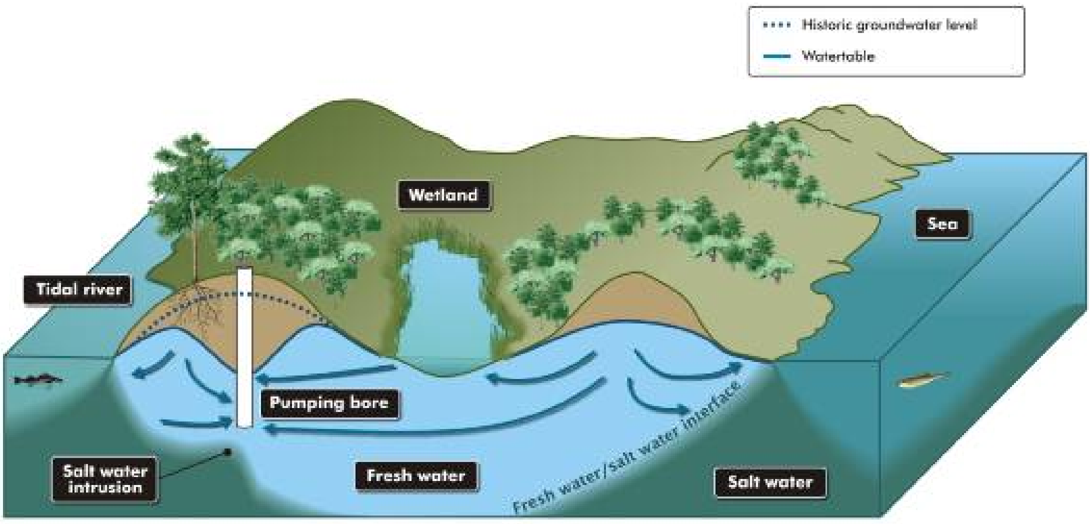

Figure A6: Conceptual model of the Tomago Sandbeds, New South Wales (modified from Department of Land and Water Conservation (NSW) 2002).

Select a study design and field methods

The study design would need to include measurements of depth to watertable, a small suite of groundwater physico-chemical variables, and vegetation type, extent and condition. As groundwater has been extracted from the sandbeds since 1939, there are few opportunities to establish a BACI-type design with 'unimpacted' sites. Instead, a gradient-type analysis is likely to be a better option for long-term monitoring. Given the large size of the study area (100 km^2), vegetation assessments will require aerial photographic interpretation (API), with extensive ground truthing to confirm vegetation type in a supervised classification (Fensham & Fairfax 2002). Fieldwork would also be required to determine vegetation condition, and the presence of species not easily seen on aerial photographs, such as understorey species or annuals.

Design a field-sampling program

An array of piezometers across the study site would be used to indicate depth to watertable. These should be sampled monthly. As salinity and pH measurements are easy, inexpensive and readily undertaken by non-specialist staff using field-portable methods, both water-quality criteria would be measured also at monthly intervals (alongside watertable depths measurements). Heavy-metal concentrations are more expensive to measure, and would be undertaken less frequently (perhaps twice yearly, according to abstraction regime) or when pH or salinity values indicated otherwise.

Vegetation surveys would include the use of aerial photographs (probably at around five-year intervals) with extensive ground-truthing to confirm the accuracy of API. Groundwork would not only confirm vegetation type and boundaries but would also allow an assessment of vegetation condition and of vegetation characteristics that cannot readily be undertaken with API, such as the presence of diagnostic or rare, threatened or endangered species. These analyses are probably required only annually, although particular care would need to be taken to ensure that seasonality was considered and that annual or cryptic species were not missed.

Data analysis and reporting

A monitoring program for the Tomago Sandbeds would require a long-term commitment for fieldwork, data analysis and data reporting. At the shortest temporal scale, watertable depth and physico-chemical data would need to be analysed at monthly intervals. A watching brief should be kept on the data, so more expensive analyses (e.g. heavy metals) could be undertaken when indicated. Vegetation outputs would include maps of vegetation type and extent, and of condition. Analysis of trend in floristic composition or vegetation condition would require long-term datasets, which in turn necessitates a long-term commitment from the bodies charged with undertaking the monitoring program. The relationship between vegetation attributes and groundwater attributes is likely to be complex; recent developments in data analysis (e.g. Laidig et al. 2010) will need to be acknowledged and incorporated as required.

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Case study 6
How to recognise and manage GDE response to climate change:Gnangara Mound, Western Australia

This Department of Water (WA) and Edith Cowan University project arose from a need to manage and monitor the ecological health of phreatophytic vegetation on the Gnangara Mound under a drying climate. (The Gnangara Mound is a shallow aquifer resource near Perth that is developed for public and private groundwater abstraction. It forms Perth's most important freshwater resource, supplying up to 60% of its drinking water.) Key areas of concern were the remnant banksia woodlands, the diverse wet heath vegetation, and groundwater-dependent wetland vegetation. The study aimed to:

- improve understanding of key ecohydrological processes on the Mound and how they influence vegetation
- improve understanding of the risks to groundwater dependent vegetation under a drying climate
- provide the basis for implementing management actions that are able to effectively manage identified risks under a drying climate.

Long-term monitoring datasets formed the basis for the investigations carried out in this study, including terrestrial vegetation monitoring conducted on 16 permanent transects over more than 30 years, and wetland vegetation monitoring undertaken on an annual basis since 1996, which included 12 permanent transects.

The studies showed that at locations typified by lower rates of drawdown (greater distances from production bores), the corresponding change in floristics has been gradual and controlled primarily by reduced rainfall. In contrast, at locations of higher rates of drawdown induced by close proximity to a production bore, rates of vegetation change are elevated and greater dissimilarity in composition relative to the pre-abstraction condition is observed.

A description of the key tasks and methodologies of the study follows.

Defining vegetation ecohydrological states

The first task was to describe the ecohydrological (EH) character of different vegetation states on the Gnangara Mound and how these have changed over time in relation to the ongoing decline in groundwater levels and other environmental drivers. State and transition models, which draw on the theory of alternative states, are particularly useful for the representation of complex natural communities.

Classification and regression analyses were then used to identify vegetation ecohydrological states in relation to hydrological habitats at the landscape scale. 'Hydrological habitat' is broadly defined as the habitat provided to groundwater-dependent vegetation by the vertical position of the watertable. The terrestrial dataset showed clearly identifiable vegetation assemblages that could be related to hydrological habitat (Table A1). The ecohydrological character was defined by allocating various ecological character traits (hydrotype, rooting class, lifeform, fire response and biodiversity) to the species present within each state/transition.

Table A1: Ecohydrological vegetation states and transitions derived for the Gnangara Mound

Ecohydrological vegetation state or transition	*Hydrological habitat (depth to watertable)*
aquatic	<- 0.75 m
sublittoral / littoral	-0.75 m to -0.23 m
littoral/supra-littoral	-0.23 m to 0.81 m
supra-littora/obligate phreatophytic	0.81 m to 2.1 m
obligate phreatophytic’	2.1 m to 4.6 m
obligate/facultative phreatophytic	4.6 m to 7.7 m
facultative phreatophytic	7.7 m to 10.5 m
facultative phreatophytic/vadophytic	> 10.5 m

Quantifying floristic change

Floristic change between the earliest and latest monitoring year (2008), expressed as a measure of dissimilarities, ranged from 6.3% to 100% over wetland and terrestrial transects. The large floristic dissimilarities over time were characterised by marked decreases in plants adapted to wet and moist habitats, but also in marked decreases in overall plant abundances. In the majority of cases these changes were unidirectional, reflecting slow and progressive floristic change over the monitoring period.

Determining environmental drivers

Redundancy analyses were used to ascertain which environmental drivers (including climate, hydrological and fire regime) were responsible for the observed floristic changes. These revealed two important aspects of the floristic temporal and spatial patterns. First, floristic dissimilarities were always greater along topographical gradients than over time. This is because the spatial variation in groundwater depth was much larger than the temporal one. Therefore, despite the temporal shift in the hydrological gradient, the majority of the compositional and structural attributes that define each community has remained at most sites. However, at the longest term monitoring sites, temporal variation was approaching the spatial variation. Secondly, of the environmental drivers tested, depth to watertable was the single strongest predictor for species composition (both spatially and temporally) at all of the transects.

It was clear from the vegetation change that has already occurred on the Gnangara Mound that if regional groundwater declines and reduced rainfall continue into the future, further losses of wet- and moist-adapted species, as well as further ‘thinning’ of the remnant vegetation can be expected on the Mound. Critical here is that if the trajectory of environmental change is faster than vegetation adaptability (e.g. by extending deep roots to follow a receding capillary zone), a threshold change on a regional scale may be unavoidable.

Mapping ecohydrological habitats and scenario modelling

Using the groundwater threshold ranges determined for the ecohydrological state and transition model, a map of vegetation ecohydrological habitats was produced for the reference year 2008 (Figure A7). The map shows very large areas covered by facultative phreatic/vadic habitats, and compares well with the inferred water requirements of the vegetation complexes. It can be used as a risk assessment tool at the landscape scale by considering areas covered by 'aquatic' to 'obligate phreatic' habitats as high risk areas in terms of further groundwater decline.

Several scenarios (based primarily on groundwater levels) were considered in this study, including a base case, current state (year 2008) and future state (year 2031) under different water use scenarios. 'Base case' shows the distribution of hydrological habitats based on where groundwater levels would be in 2031 if abstraction volumes (public and licensed and unlicensed private) remained at 2007 levels (i.e. 135 GL/a, 200 GL/a and 60 GL/a respectively), and the pine plantations were replaced by 'grassland'; and climate remains as the average values for the period 1997–2006.

In Figure A7, 'GSS composite' shows the distribution of hydrological habitats based on where groundwater levels would be in 2031 if abstraction volumes (public and licensed and unlicensed private) are reduced to 120 GL/a, 160 GL/a and 60 GL/a respectively, and the pine plantations were replaced by 'grassland'; and climate remains as the average values for the period 1997–2006. 'No abstraction' shows the distribution of hydrological habitats based on where groundwater levels would be in 2031 if abstraction volumes (public and licensed and unlicensed private) are reduced to 0 GL/a, 0 GL/a and 0 GL/a respectively, and the pine plantations were replaced by 'grassland'; and climate remains as the average values for the period 1997–2006. Modelling of scenario cases suggests that between 1980 and 2008 large areas (~50%) of littoral and obligate phreatic habitats were lost in favour of the drier habitats. A '2031 no abstraction' scenario would result in expansions of these wetter habitats by ~44% when compared with 2008. The '2031 GSS composite' and the '2031 base case' scenarios showed similar distributions of ecohydrological habitats. Under these scenarios, the maps suggest that the wetter habitats (aquatic to obligate phreatic) would decline in favour of the drier habitats when compared with 2008. It is apparent that, particularly for the '2031 base case' scenario, these pumping regimes would pose a significant quantifiable risk to the maintenance of the wetter hydrological habitats.

Interpretation of the vegetation ecohydrological habitat maps should be based on the premise that they show the potential distribution of vegetation ecohydrological states based on the spring peak groundwater levels. They do not show the actual distribution of vegetation.

Risk assessment framework

Two complementary explanatory and predictive analysis techniques—multivariate regression trees and Bayesian Belief Networks—were used to model the risk of impact of declining groundwater levels to vegetation character. Again, the groundwater threshold ranges determined for the ecohydrological state and transition model were used to define hydrological habitats. In the context of a risk assessment process, the framework development was focused on the 'risk characterisation' step.

The end products of the framework were two graphical tools suitable for rapid risk assessment and an ecohydrological risk matrix that requires the incorporation of additional controlling factors (e.g. site-specific factors such as soil characteristics, etc.) designed to lower uncertainty. Response patterns were clearly detectable, and in contrast to risk assessments lacking historical data, both the expected levels of risk and the associated levels of uncertainty were able to be quantified in a more or less objective manner. Predictive

accuracy can be improved by directing future studies towards improving our understanding of the factors that influence the large spatial and temporal variability in the response of groundwater dependent vegetation to fluctuating watertables.

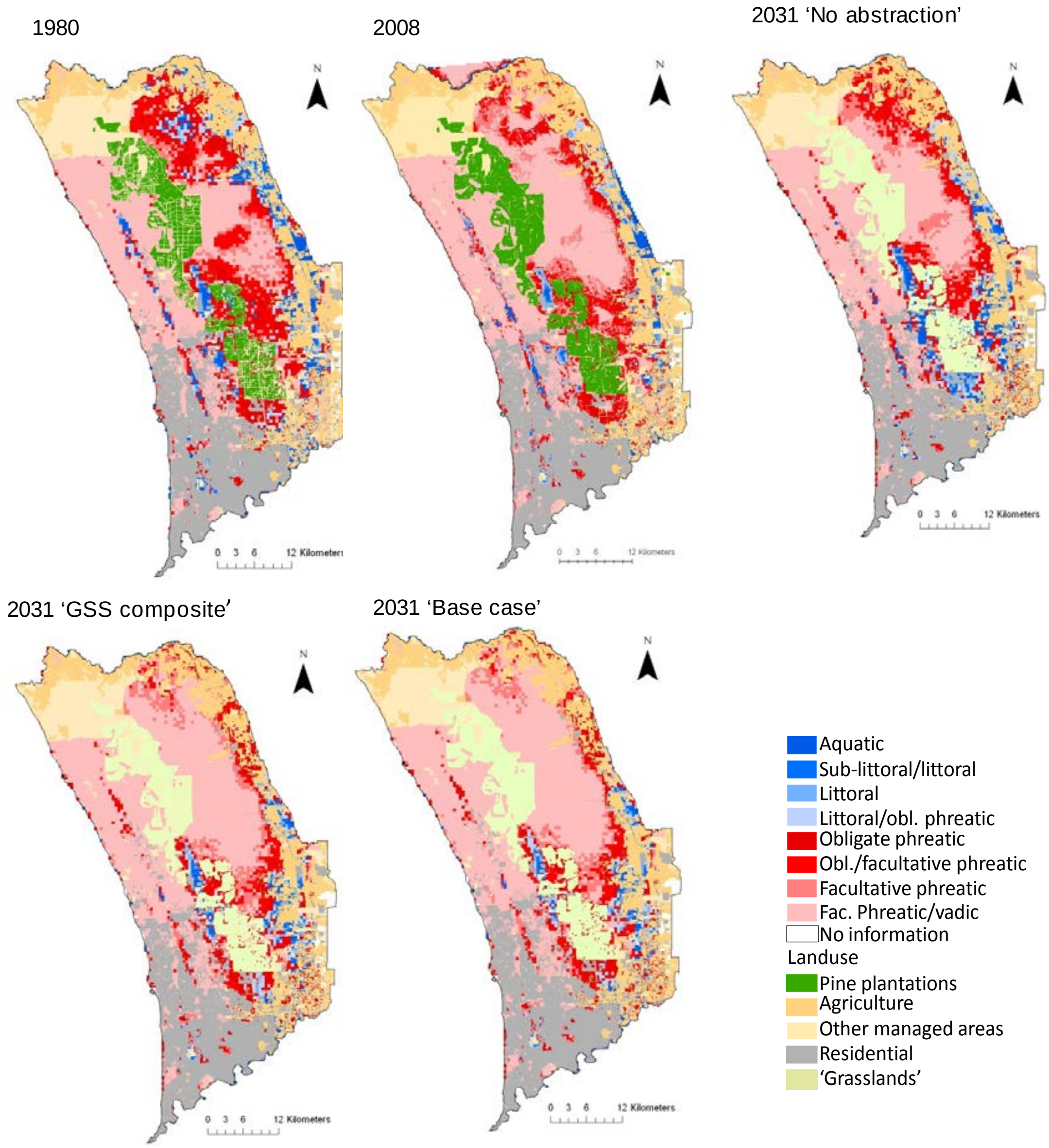

Figure A7: Distribution of vegetation ecohydrological (EH) states on the Gnangara Mound. Based on numerical ecological analyses of the floristic and hydrological historical datasets, blue and dark-red areas are potential 'high risk' areas in terms of vegetation response to further groundwater decline.

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Appendix B: Glossary

abiotic	Non-living chemical and physical factors in the environment.
anion	Negatively charged ion. Source: Australian and New Zealand Guidelines for Fresh and Marine Water Quality (2000) *http://www.mincos.gov.au/__data/assets/pdf_file/0016/316123/wqg-apps.pdf.*
applied tracer	Non-natural constituent that is intentionally introduced to a hydrologic system to characterise groundwater flowpaths and estimate velocities.
aquifer	An underground layer of permeable rock, sand or gravel that absorbs water and allows it free passage through pore spaces. Source: Australian and New Zealand Guidelines for Fresh and Marine Water Quality (2000) *http://www.mincos.gov.au/__data/assets/pdf_file/0016/316123/wqg-apps.pdf.*
aquifer, confined	An aquifer that is overlain by a confining bed. The confining bed has a significantly lower hydraulic conductivity than the aquifer. Source: National Water Commission Water Dictionary *http://dictionary.nwc.gov.au/water_dictionary/pdf/WaterDictionary.pdf*
aquifer, perched	A region in the unsaturated zone where the soil or rock may be locally saturated because it overlies a low-permeability unit. Source: National Water Commission Water Dictionary *http://dictionary.nwc.gov.au/water_dictionary/pdf/WaterDictionary.pdf*
aquifer, unconfined	An aquifer in which there are no confining beds between the saturated zone and the surface. There will be a watertable in an unconfined aquifer. Source: National Water Commission Water Dictionary *http://dictionary.nwc.gov.au/water_dictionary/pdf/WaterDictionary.pdf*

aquitard	A confining unit that retards but does not prevent the flow of water to or from an adjacent aquifer. An aquitard is the less-permeable bed in the stratigraphic sequence. An aquitard does not readily yield water to wells or springs but may serve as a storage unit for groundwater and can transmit water slowly from one aquifer to another. Source: Poehls DJ and Smith GJ 2009, *Encyclopedic dictionary of hydrogeology*, Elsevier Inc., 517 pp.
baseflow	The component of streamflow supplied by groundwater discharge. Source: National Water Commission Water Dictionary *http://dictionary.nwc.gov.au/water_dictionary/pdf/WaterDictionary.pdf*
Bayesian Belief Network (BBN)	Formalised description of causal relationships between key factors and final outcomes or responses (e.g. of an ecosystem).
biotic	A living component of a community, such as plants and animals.
capillary zone	The zone of soil moisture above the watertable where water is drawn upwards by capillary tension in which the water is at less than atmospheric pressure.
conservative ion	A non-reactive ion; speciation does not change over a given process.
Darcy's law	A law that relates the rate of fluid flow to the flow path and hydraulic head gradient, assuming that the flow is laminar and that inertia can be neglected. Source: Poehls DJ and Smith GJ 2009, *Encyclopedic dictionary of hydrogeology*, Elsevier Inc., 517 pp.
digital elevation model (DEM)	A depiction of relief using points and lines, which contain the elevation of each point or the elevation of each point in a line. The data may be in a regular grid or have an irregular spacing. Source: Geoscience Australia. Glossary to NTDB and NTMS Specifications (250K and 100K) *http://www.ga.gov.au/mapspecs/250k100k/appendix_l.jsp*.
dissolved oxygen (DO)	The measure of the amount of gaseous oxygen dissolved in an aqueous solution.
doline	Natural depression or hole in the Earth's surface caused by karst processes, commonly referred to as a 'sinkhole'.
drawdown	The distance between the static water level and the surface of the cone of depression.

ecohydrogeological zone (EHZ)	Regions where similar processes are likely to determine the interaction between groundwater and ecology, due to similar ecology, geology, climate, groundwater/surface water connections.
ecosystem	A system that includes all living organisms (biotic factors) in an area as well as its physical environment (abiotic factors) functioning together as a unit.
ecosystem services	Fundamental characteristic of ecosystems related to conditions and processes necessary for maintaining ecosystem integrity, which implies intact abiotic components (e.g. soils and water), biodiversity and resilience to natural successional cycles (e.g. fire, flooding, predation). Ecosystem function will include such processes as decomposition, nutrient cycling and production. It is generally considered that maintenance of biodiversity is integral to ecosystem function.
ecological values	The natural ecological processes occurring within ecosystems and the biodiversity of these systems. Source: adapted from ARMCANZ/ANZECC 1996, *National principles for the provision of water for ecosystems*, Sustainable Land and Water Resources Management Committee Subcommittee on Water Resources Occasional Paper SWR No 3 July 1996. *http://www.environment.gov.au/water/publications/environmental/ecosystems/pubs/water-provision.pdf*
ecological water provision (EWP)	Part of the environmental water requirement (or ecological water requirement) that can be met. Source: ARMCANZ/ANZECC 1996, *National principles for the provision of water for ecosystems*, Sustainable Land and Water Resources Management Committee Subcommittee on Water Resources Occasional Paper SWR No 3 July 1996. *http://www.environment.gov.au/water/publications/environmental/ecosystems/pubs/water-provision.pdf.*
ecological water requirement (EWR)	Descriptions of the water regimes needed to sustain the ecological values of water-dependent ecosystems at a low level of risk. Source: adapted from definition for *Environmental Water Requirements* in ARMCANZ & ANZECC 1996, *National principles for the provision of water for ecosystems*, Sustainable Land and Water Resources Management Committee Subcommittee on Water Resources Occasional Paper SWR No 3 July 1996 *http://www.environment.gov.au/water/publications/environmental/ecosystems/pubs/water-provision.pdf.*
eddy covariance	Technique that measures fine-timescale carbon and water fluxes between vegetation and the atmosphere.

electrical conductivity (EC)	Electrical conductivity (EC) measures dissolved salt in water. The standard EC unit is microSiemens per centimetre (µS/cm) at 25 °C. Source: National Water Commission Water Dictionary *http://dictionary.nwc.gov.au/water_dictionary/pdf/WaterDictionary.pdf*
epibenthic	Living on the surface of bottom sediments in a water body.
environmental flow	A water regime provided within a river, wetland or estuary to improve or maintain ecosystems and their benefits where there are competing water uses and where flows are regulated. Source: National Water Commission Water Dictionary *http://dictionary.nwc.gov.au/water_dictionary/pdf/WaterDictionary.pdf*
environmental tracer	Naturally occurring indicator, usually physico-chemical, used to identify and constrain specific biotic or abiotic processes.
evapotranspiration (ET)	The combined loss of water from a given area during a specified period of time by evaporation from the soil or water surface and by transpiration from plants. Source: Australian and New Zealand Guidelines for Fresh and Marine Water Quality (2000) *http://www.mincos.gov.au/__data/assets/pdf_file/0016/316123/wqg-apps.pdf.*
geophysical survey	A process of searching and mapping the subsurface structure of the earth's crust using geophysical methods such as seismic, magnetic, electromagnetic, gravity and induced polarisation techniques.
gaining stream	A stream where groundwater discharge contributes to stream flow.
groundwater	Subsurface water located in the zone of saturation in pores, fractures and cavities in rocks.
groundwater-dependent ecosystem (GDE)	Natural ecosystems that require access to groundwater to meet all or some of their water requirements on a permanent or intermittent basis so as to maintain their communities of plants and animals, ecological processes and ecosystem services.
groundwater flow system	The total system which describes the movement of water in the subsurface from the point where it enters the ground to where it leaves.
ground-truthing	The use of a ground survey to calibrate and/or confirm hypotheses relating to the interpretation of aerial survey, satellite imagery or other remote method.

gypsum blocks	Gypsum blocks consist of two electrodes embedded in a block of gypsum used to measure soil water tension, a reflection of the force that a plant must overcome to extract water from the soil. The resistance between the two electrodes varies with the water content in the gypsum block, which will depend directly on the soil water tension.
hydraulic conductivity	A coefficient of proportionality describing the rate at which water can move through a permeable medium. Horizontal hydraulic conductivity (K_h) refers to the coefficient of proportionality in the horizontal direction, whereas vertical hydraulic conductivity (K_v) refers to the coefficient of proportionality in the vertical direction.
hydraulic gradient	The rate of change in total head per unit distance in a given direction. The direction of gradient is that yielding the maximum rate of decrease in head.
hydrograph	Graphical representation of river or stream discharge or of groundwater-level fluctuations in a well. Source: Australian and New Zealand Guidelines for Fresh and Marine Water Quality (2000) *http://www.mincos.gov.au/__data/assets/pdf_file/0016/316123/wqg-apps.pdf.*
hyporheic zone	The saturated interstitial areas beneath the streambed and into the stream banks that contain some proportion of channel water or that have been altered by channel water infiltration. Source: White DS 1993, 'Perspectives on defining and delineating hyporheic zones', *Journal of the North American Benthological Society* 12:61–69.
hypothesis	Statements or theories that can be subjected to statistical evaluation when monitoring data has been obtained to determine whether they can be accepted (or rejected). Source: Australian and New Zealand Guidelines for Fresh and Marine Water Quality (2000) *http://www.mincos.gov.au/__data/assets/pdf_file/0016/316123/wqg-apps.pdf.*
interflow	The runoff infiltrating into the surface soil and moving toward streams as shallow, perched ground water above the main groundwater level. Source: USGS Definition of Terms *http://pubs.usgs.gov/ha/ha747/pdf/definition.pdf.*

introduced tracer	Dyes or conservative tracer that may be added to streams or aquifers to identify the occurrence and flux of discharging groundwater to river and wetland systems.
Isotope	A particular atom of an element that has the same number of electrons and protons as the other atoms of that element, but a different number of neutrons, i.e. the atomic numbers are the same but the atomic weights differ. Isotopes have essentially the same chemical properties as other atoms of the same element. Source: National Water Commission Water Dictionary *http://dictionary.nwc.gov.au/water_dictionary/pdf/WaterDictionary.pdf*
karst	A terrain characterised by sinkholes, caves and springs developed most commonly in carbonate rocks where significant dissolution of the rock has occurred due to flowing water. Source: Jennings 1985; Culver et al. 1995; Fetter 2001 as referenced in M Tomlinson and A Boulton 2008, *Subsurface groundwater dependent ecosystems: a review of biodiversity, ecological processes and ecosystem services*, National Water Commission Waterlines Occasional Paper No. 8, October 2008 *http://www.nwc.gov.au/resources/documents/Waterlines__subsurface_full_version.pdf*
LiDAR	Light Detection and Ranging is a remote sensing technique that uses ultraviolet, visible or near infrared light to image objects.
laser scintillometer	Measurement system for the determination of the turbulent fluxes of heat and momentum based on optical scintillation measurements.
leaf area index (LAI)	The ratio between the total upper leaf surface area of vegetation and the surface area of ground over which the vegetation grows.
leaf water potential (LWP)	Measure of the water pressure of a leaf and hence the plant. A plant that is fully hydrated may exhibit a water potential close to zero.
losing stream	A stream from which water is lost to the surrounding and underlying substrate via infiltration through the streambed.
macrophyte	A member of the macroscopic plant life of an area, especially of a body of water; large aquatic plant. Source: Australian and New Zealand Guidelines for Fresh and Marine Water Quality (2000) *http://www.mincos.gov.au/__data/assets/pdf_file/0016/316123/wqg-apps.pdf*

major ions	Constituents commonly present in concentrations exceeding 1.0 milligram per litre. For dissolved cations this includes calcium, magnesium, sodium, and potassium; the most prevalent anions include sulfate, chloride, fluoride, nitrate, and those contributing to alkalinity, most generally assumed to be bicarbonate and carbonate.
matric potential	A variable describing how strongly the water within a soil matrix is bound to the soil by capillary and other forces. Source: National Water Commission Water Dictionary *http://dictionary.nwc.gov.au/water_dictionary/pdf/WaterDictionary.pdf*
model, conceptual	Documentation of a conceptual understanding of the location of GDEs and interaction between ecosystems and groundwater.
model, analytical / numerical	Simulates groundwater flow indirectly by means of governing equations considered representative of the physical process occurring in the system, in addition to equations describing heads or flow along the model boundaries. Mathematical models can be solved analytically or numerically.
multiple lines of evidence	Weight of the evidence based on different types of information from a variety of different sources and studies. Source: Australian and New Zealand Guidelines for Fresh and Marine Water Quality (2000) *http://www.mincos.gov.au/__data/assets/pdf_file/0016/316123/wqg-apps.pdf*
neutron probes	Neutron probes enable a rapid measurement of soil moisture to be made. The neutron probe has a radioactive source which releases neutrons. The neutrons are emitted into the soil when the probe is lowered into an aluminium tube which has been installed in the ground. Whenever a neutron collides with a hydrogen atom (part of a water molecule) it is slowed down. A detector counts the slow neutrons that have been deflected back to the instrument. A calibration equation is used to convert this number into the soil moisture content.
osmotic potential	Osmotic potential is the potential of water to move into a region by the process of osmosis, the potential of the water to travel from a hypotonic (low concentration) solution to a hypertonic (high-concentration) solution.
parameter	Statistical constants that can be summarised to show some measure of central tendency and variability such as mean, median, standard variation.

piezometer	A non-pumping well, generally of small diameter, that is used to measure the elevation of the watertable or potentiometric surface. A piezometer generally has a short well screen through which water can enter. Source: National Water Commission Water Dictionary *http://dictionary.nwc.gov.au/water_dictionary/pdf/WaterDictionary.pdf*
pH	Value that represents the acidity or alkalinity of an aqueous solution. It is defined as the negative logarithm of the hydrogen ion concentration of the solution. Source: Australian and New Zealand Guidelines for Fresh and Marine Water Quality (2000) *http://www.mincos.gov.au/__data/assets/pdf_file/0016/316123/wqg-apps.pdf*
phreatophyte	Plant that draws water from groundwater or the capillary zone to maintain vigour and function.
potentiometric surface	A surface representing the hydraulic head of ground water; represented by the watertable altitude in an unconfined aquifer or by the altitude to which water will rise in a properly constructed well in a confined aquifer. Source: USGS Definition of Terms *http://pubs.usgs.gov/ha/ha747/pdf/definition.pdf*
redox (reduction – oxidation)	Chemical reactions in which the oxidation states of atoms are changed.
remote sensing	Any kind of data recording by a sensor which measures energy emitted or reflected by objects located at some distance from the sensor (i.e. no direct ground contact). Can include aerial photographs, airborne digital sensors and satellite imagery.
resilience, ecosystem	The persistence of relationships within an ecosystem and a measure of the ability of the ecosystem to absorb changes in external and internal conditions and still persist. Source: modified from Holling CS 1973, 'Resilience and stability of ecological systems' *Annual Review of Ecology & Systematics* 4:1–23.

resilience building	Increasing the ability of an ecosystem to cope with stress or unexpected events, usually by manipulating the system to increase the likelihood of it persisting despite disturbance. Modified from Peterson GD 2005, 'Ecological management: control, uncertainty, and understanding' in K Cuddington and BE Beisner, *Ecological Paradigms Lost*, pp. 371–396. Elsevier, Amsterdam.
resistance, ecosystem	The ability of an ecosystem to return to an equilibrium state after a temporary disturbance. It can involve factors such as constancy (lack of change), persistence (survival times), inertia (ability to resist external perturbations), elasticity (speed with which a system returns to its original state following a perturbation), and amplitude (range over which a system is stable). Source: modified from Holling CS 1973, 'Resilience and stability of ecological systems' *Annual Review of Ecology & Systematics* 4:1–23 and Orians GH 1974, 'Diversity, stability and maturity in natural ecosystems' in WH van Dobben and RH Lowe-McConnell (eds) *Unifying Concepts in Ecology*, De W. Junk, The Hague, pp. 139–149.
riparian	An area or zone within or along the banks of a stream or adjacent to a watercourse or wetland; relating to a riverbank and its environment, particularly to the vegetation. Source: eWater Toolkit Glossary *http://www.toolkit.net.au/support/Glossary.aspx*
salinity	The concentration of soluble salts in a solution, soil or other medium. Source: eWater Toolkit Glossary *http://www.toolkit.net.au/support/Glossary.aspx*
sap flow techniques	A direct measurement of plant water use using heat balance, heat pulse and thermal diffusion techniques. The heat-balance sensor encloses the stem, while the heat pulse and thermal diffusion sensors require probes to be inserted into the plant stems.
saturated zone	The part of the lithosphere where each void space in subsurface material is filled with water, or is saturated, under greater pressure than that of the atmosphere. Source: Poehls DJ and Smith GJ 2009, *Encyclopedic dictionary of hydrogeology*, Elsevier Inc., 517 pp.
scenario planning	A systematic method for thinking creatively about possible futures in which uncertainty is high and controllability is low. Source: Peterson GD 2005, 'Ecological management: control, uncertainty, and understanding' in K Cuddington and BE Beisner (eds) *Ecological Paradigms Lost*, Elsevier, Amsterdam pp. 371–396.

sinkhole	See ‘doline’.
slug test	A single-well test to determine the hydraulic conductivity of a formation, by which a known volume (either of water or solid) is inserted into a well and the water level response to insertion of the slug (falling head test) and removal of the slug (rising head test) is monitored. Source: Poehls DJ and Smith GJ 2009, *Encyclopedic dictionary of hydrogeology*, Elsevier Inc., 517 pp.
stratification	The formation of layers in a water body, showing differences in temperature, turbidity, pH, nutrients, salinity, dissolved oxygen and light penetration at various depths; lack of mixing within a water storage. Source: eWater Toolkit Glossary *http://www.toolkit.net.au/support/Glossary.aspx*
stressors	The physical, chemical or biological factors that can cause an adverse effect in an aquatic ecosystem as measured by the condition indicators. Source: Australian and New Zealand Guidelines for Fresh and Marine Water Quality (2000) *http://www.mincos.gov.au/__data/assets/pdf_file/0016/316123/wqg-apps.pdf*
specific yield	The amount of water that a unit volume of saturated permeable rock would yield if drained by gravity.
stable isotope	An isotope that does not undergo radioactive decay.
stygobite	Aquatic animal that completes its life cycle in groundwater. Source: Tomlinson M and Boulton A 2008, *Subsurface groundwater dependent ecosystems: a review of biodiversity, ecological processes and ecosystem services*, National Water Commission Waterlines Occasional Paper No. 8, October 2008 *http://www.nwc.gov.au/resources/documents/Waterlines__subsurface_full_version.pdf*

stygofauna	Aquatic animals found in groundwater; sometimes used as a synonym of stygobite. Source: Tomlinson M and Boulton A 2008, *Subsurface groundwater dependent ecosystems: a review of biodiversity, ecological processes and ecosystem services*, National Water Commission Waterlines Occasional Paper No. 8, October 2008 *http://www.nwc.gov.au/resources/documents/Waterlines__subsurface_full_version.pdf*
stygophile	Animals which spend part of their life cycle in groundwater. Source: Tomlinson M and Boulton A 2008, *Subsurface groundwater dependent ecosystems: a review of biodiversity, ecological processes and ecosystem services*, National Water Commission Waterlines Occasional Paper No. 8, October 2008 *http://www.nwc.gov.au/resources/documents/Waterlines__subsurface_full_version.pdf*
stygoxene	Animals which occur accidentally in groundwater but have no affinity with groundwater habitats. Source: Tomlinson M and Boulton A 2008, *Subsurface groundwater dependent ecosystems: a review of biodiversity, ecological processes and ecosystem services*, National Water Commission Waterlines Occasional Paper No. 8, October 2008 *http://www.nwc.gov.au/resources/documents/Waterlines__subsurface_full_version.pdf*
taxon (taxa)	Any group of organisms considered to be sufficiently distinct from other such groups to be treated as a separate unit (e.g. species, genera, families). Source: Australian and New Zealand Guidelines for Fresh and Marine Water Quality (2000) *http://www.mincos.gov.au/__data/assets/pdf_file/0016/316123/wqg-apps.pdf*
taxonomic (group, resolution)	An organism's location in the biological classification system used to identify and group organisms with similar physical, chemical and/or structural composition. Source: Australian and New Zealand Guidelines for Fresh and Marine Water Quality (2000) *http://www.mincos.gov.au/__data/assets/pdf_file/0016/316123/wqg-apps.pdf*

tensiometer	An instrument designed to measure the tension or suction that plants' roots must exert to extract water from the soil. This tension is a direct measure of the availability of water to a plant. A tensiometer consists of an air tight, water filled tube with a porous ceramic tip at the bottom and either a vacuum gauge at the top or a resealable rubber bung for a portable vacuum meter.
throughfall	Precipitation that falls directly through a vegetative canopy or is intercepted by vegetation and then drips to the ground. Source: The Dictionary of Forestry, Society of American Foresters *http://dictionaryofforestry.org/dict/*
total dissolved solids (TDS)	A measure of the inorganic salts (and organic compounds) dissolved in water. Source: Australian and New Zealand Guidelines for Fresh and Marine Water Quality (2000) *http://www.mincos.gov.au/__data/assets/pdf_file/0016/316123/wqg-apps.pdf*
transmissivity	The rate at which water moves through a unit width of aquifer or aquitard under a unit hydraulic gradient. It is the product of aquifer thickness and hydraulic conductivity.
transpiration	Evaporation loss of water from the leaves of plants through the stomata; the flow of water through plants from soil to atmosphere. Source: eWater Toolkit Glossary *http://www.toolkit.net.au/support/Glossary.aspx*
troglofauna	Terrestrial animals living in caves and other air-filled subterranean spaces. Source: Tomlinson M and Boulton A 2008, *Subsurface Groundwater Dependent Ecosystems: a review of biodiversity, ecological processes and ecosystem services*, National Water Commission Waterlines Occasional Paper No 8, October 2008 *http://www.nwc.gov.au/resources/documents/Waterlines__subsurface_full_version.pdf*
typology	Classification of habitats into types defined by ecological descriptors. Source: Tomlinson M and Boulton A 2008, *Subsurface groundwater dependent ecosystems: a review of biodiversity, ecological processes and ecosystem services*, National Water Commission Waterlines Occasional Paper No. 8, October 2008 *http://www.nwc.gov.au/resources/documents/Waterlines__subsurface_full_version.pdf*

unsaturated zone	The areas below the ground where void spaces are filled with a mixture of water under pressure less than atmospheric which includes water held by capillarity and air (gases) under atmospheric pressure. Source: Poehls DJ and Smith GJ 2009, *Encyclopedic dictionary of hydrogeology*, Elsevier Inc., 517 pp.
vadose zone	See ‘unsaturated zone’.
variable	A measurable or quantifiable characteristic or feature.
ventilated chamber	Measures transpiration by sampling the vapour pressure of air entering and leaving a plastic chamber enclosing a tree. The difference in vapour pressure between the two samples in used to calculate the transpiration rate.
water balance	Balance of the water resources of a region, comparing precipitation and inflow with outflow, evaporation, and changes in storage.
watertable	The top of the water surface in the saturated zone of an unconfined aquifer. Source: USGS Definition of Terms *http://pubs.usgs.gov/ha/ha747/pdf/definition.pdf*