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Adapting to climate change: A risk assessment and decision making framework for managing groundwater dependent ecosystems with declining water levels

Supporting Document 1: Literature Review

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SUPPORTING DOCUMENT 1:
Literature review

Murdoch University

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The role of NCCARF is to lead the research community in a national interdisciplinary effort to generate the information needed by decision-makers in government, business and in vulnerable sectors and communities to manage the risk of climate change impacts.

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Cover image: Pipidinny Swamp, Western Australia © The Department of Water, Government of Western Australia

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1. INTRODUCTION

The potential impacts of a drying climate on freshwater ecosystems in Australia are alarming. In many regions, competition for water resources between humans and the environment presents a challenge for environmental managers. This is because a complex array of factors, including climate change and human activity, interact to impact our freshwater ecosystems. The best way to address this challenge is to identify the level of risk to freshwater ecosystems associated, both individually and synergistically, with climate change projections and water resource and land use practices.

This literature review provides the basis for a project to develop and test a risk assessment and decision-making tool for managing groundwater dependent wetlands and caves affected by climate change and other stressors. The project: *Adapting to climate change: a risk assessment and decision framework for managing groundwater dependent ecosystems with declining water levels* (FW1108) is funded by the National Climate Change Adaptation Research Facility (NCCARF) (for further information refer to Chambers *et al*. 2013a,b).

A key knowledge gap in climate change adaptation research is the capacity for species, communities and ecosystems to adapt to changes in environmental variables, particularly water. In this project, habitat requirements (the hydrological and water quality conditions under which biota will persist) was identified from long-term datasets. This was based on the relationship between surface and groundwater levels and quality, rainfall recharge processes and biota requirements. Multivariate statistics was used to identify hydrological thresholds for functional groups of biota, which could be applied across Australia. Current distribution of these habitats was mapped and compared to predicted future distribution of these habitats based on groundwater levels modelled by different climate change or anthropogenic extraction scenarios. This will enable environmental managers to adapt to climate change at the local, landscape and catchment scales by identifying sites of high ecosystem value, including species and communities at risk. The framework was tested in south-western Australia, a global biodiversity hotspot and one of the earliest regions impacted by climate change, but the methodology is designed to be transferable to other types of GDEs and locations in Australia and internationally.

This literature review utilises a standard risk assessment framework (Asante-Duah 1998) to capture the state of our understanding of the scope of a drying climate (*1. Hazard*) and its likely distribution and level of impact on groundwater dependent ecosystems across Australia and internationally (*2. Exposure and vulnerability*). The effect this will have on the ecosystem function and biota is considered in *3. Effects assessment*. This will investigate how and which characteristics of water level decline might impact both wetland processes and the survival of wetland biota. Such information is critical to identifying the type of data needed and its analysis. Section 4 (*Thresholds and levels of acceptable change*) investigates the types of responses to perturbation as outlined by ecological models of disturbance, and seeks to evaluate appropriate models for this risk assessment and justify levels of unacceptable change. Appropriate tools for a risk assessment and decision-making framework of this type are outlined in section 5 (*Risk assessment tools*). The resources available on GDEs across Australia is considered in *6. Current GDE information/toolboxes.* Finally, appropriate ways of incorporating future climate change in management are dealt with in *7. Adaptation.*

2. HAZARD – DECLINING WATER LEVELS

2.1 Recent Drought in Southern Australia

The vulnerability of freshwater ecosystems to declining water levels has recently been highlighted as a result of widespread drought across southern Australia. Annual total rainfall has declined in the south-west, south-east and eastern regions of Australia over the last 40 years (1970-2011) by 5 to 50 mm/10 years (Figure 1). Increasing mean temperature by 0.05 to 0.6 ºC /10 years (Figure 2) and increased pan evaporation by 5-15 mm/year (Figure 3) has contributed to declining water levels in these regions. Bond *et al.* (2008) identified the recent (2001 – 2010) drought in south-eastern Australia as the most severe and longest drought experienced in the region in 200 years. The drought experienced in south-western Australia since 1975 is related to human-induced climate change and the natural fluctuations in the climate (CSIRO 2007). It has resulted in a decline in rainfall of 15-20% and consequently decreased runoff in Perth water catchments by 70% (Bond *et al.* 2008). The Australian State of the Environment Committee (2011) determined that drought recently experienced in the south-west has already impacted on GDEs by lowering the water table out of the reach of ecosystems.

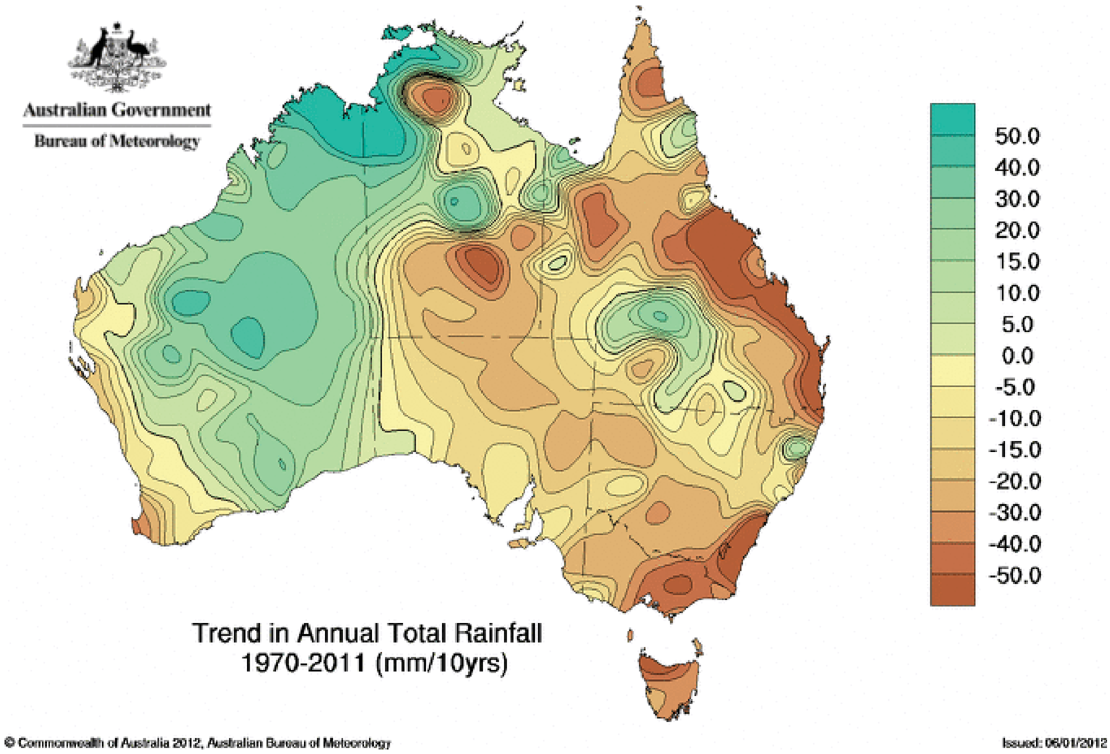


Figure 1: Trend in annual total rainfall in Australia from 1970 – 2011 (mm/10yrs) (Australian Bureau of Meteorology 2012).

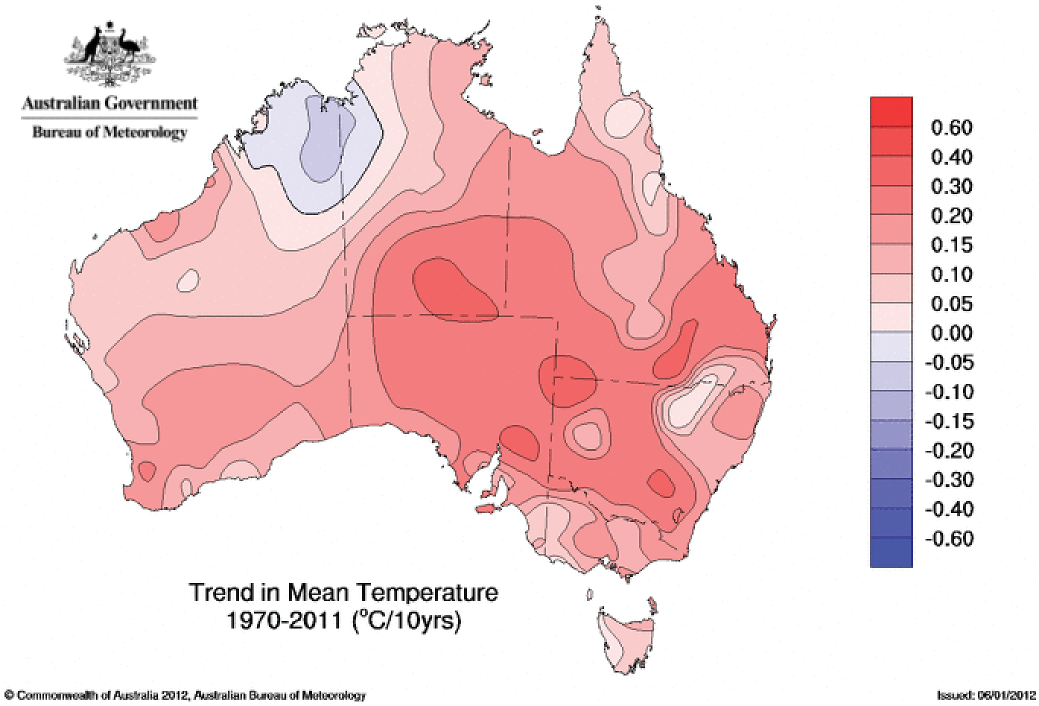


Figure 2: **Trend in mean temperature (ºC/10yrs) in Australia from 1970-2011 (Australian Bureau of Meteorolgy 2012).**

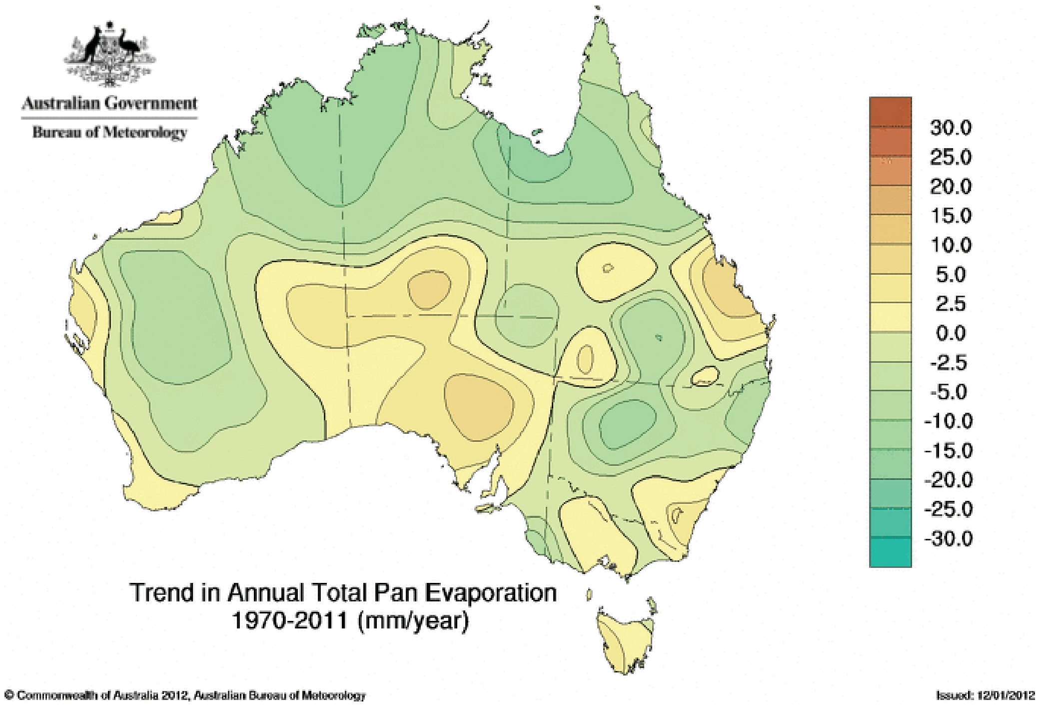


Figure 3: **Trend in annual total pan evaporation (mm/yr) in Australia from 1970-2011 (Australian Bureau of Meteorolgy 2012).**

2.2 Climate Change Projections

Climate change precipitation, temperature and evapotranspiration projections for Australia in 2030 were taken from a technical report published by CSIRO (2007). This report focuses on the A1B emissions scenario, which is a mid-range emission scenario for temperature. The 10th, 50th (median) and 90th percentile projections were taken from the range of projections. The median projections determine the best estimates of change, while the 10th and 90th percentile projections deal with the levels of uncertainty surrounding climate change projections. To determine the accuracy of the model, simulations of present (1961 -1990) precipitation, surface air temperature and sea level were compared with experienced climate.

The median projections for 2030 of average annual temperature (Figure 4) suggest that there will be an increase of 0.7 - 0.9ºC in coastal areas and 1- 1.2 ºC in inland regions. Winter warming is expected to be smaller than summer, as low as 0.5 ºC in the south (Figure 4). The 10th and 90th percentile projections suggest a level of uncertainty ranging from 0.6 to 1.5 ºC (Figure 5) (CSIRO 2007).

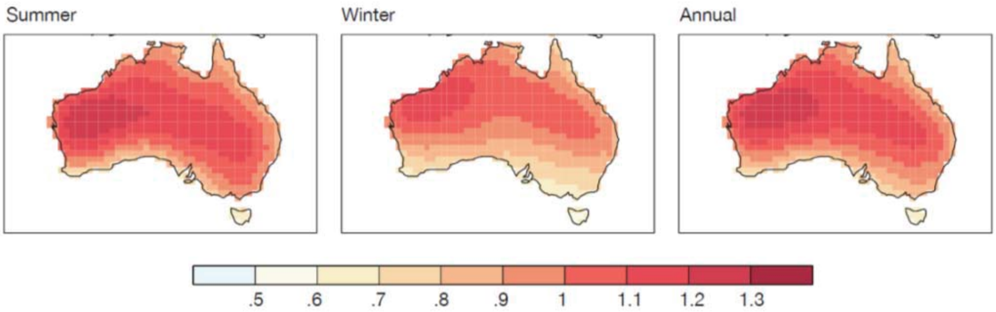


Figure 4: Range of warming (ºC) over land by 2030 under the A1B scenario for the median (best estimate) value for summer, winter and annual (CSIRO 2007).

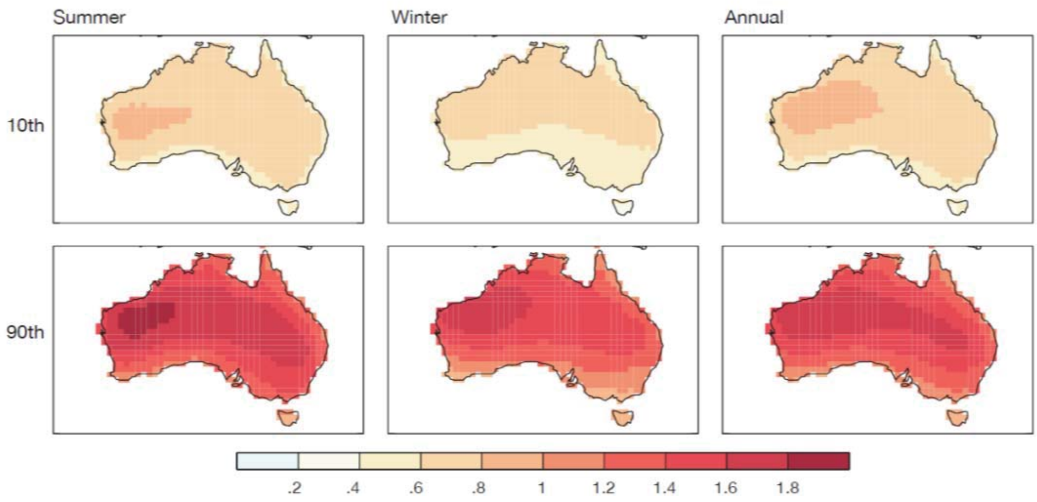


Figure 5: Range of warming (ºC) over land by 2030 under the A1B scenario for the 10th and 90th percentile value for summer, winter and annual (CSIRO 2007).

Annual precipitation is projected to decrease by a minimum of 2 to 5% in all regions of Australia, except the far north where precipitation is expected to increase (Figure 6). The major decrease in precipitation is projected to occur in winter and spring with a 5% decrease nation-wide and a decrease of 10% in the south-west (Figure 7). The 10th and 90th percentile models suggest a level of uncertainty ranging from -10% to very little change in southern Australia and -10% to +5% in northern Australia for the annual projections (Figure 8) (CSIRO 2007). Seasonal projections show a level of uncertainty in winter and spring typically ranging from -15% to little change in southern Australia and -15% to +5% in eastern areas. Summer and autumn the range is -15% to +10% (Figure 9).

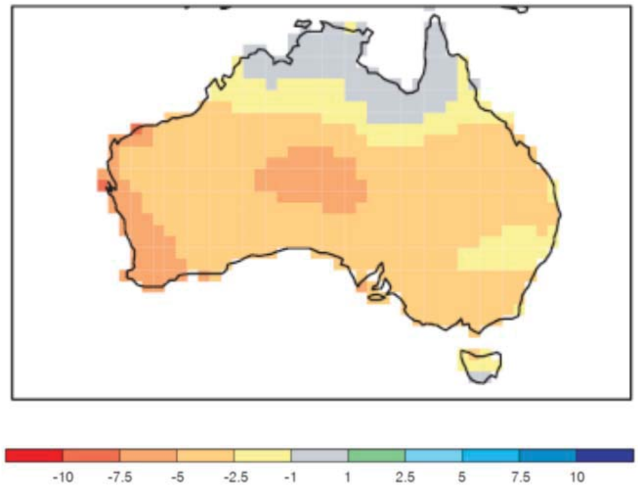


Figure 6: Best estimate of the projected change of Australian annual precipitation over land by 2030 for the A1B emission scenario, as a percentage of 1961 -1990 values (CSIRO 2007).

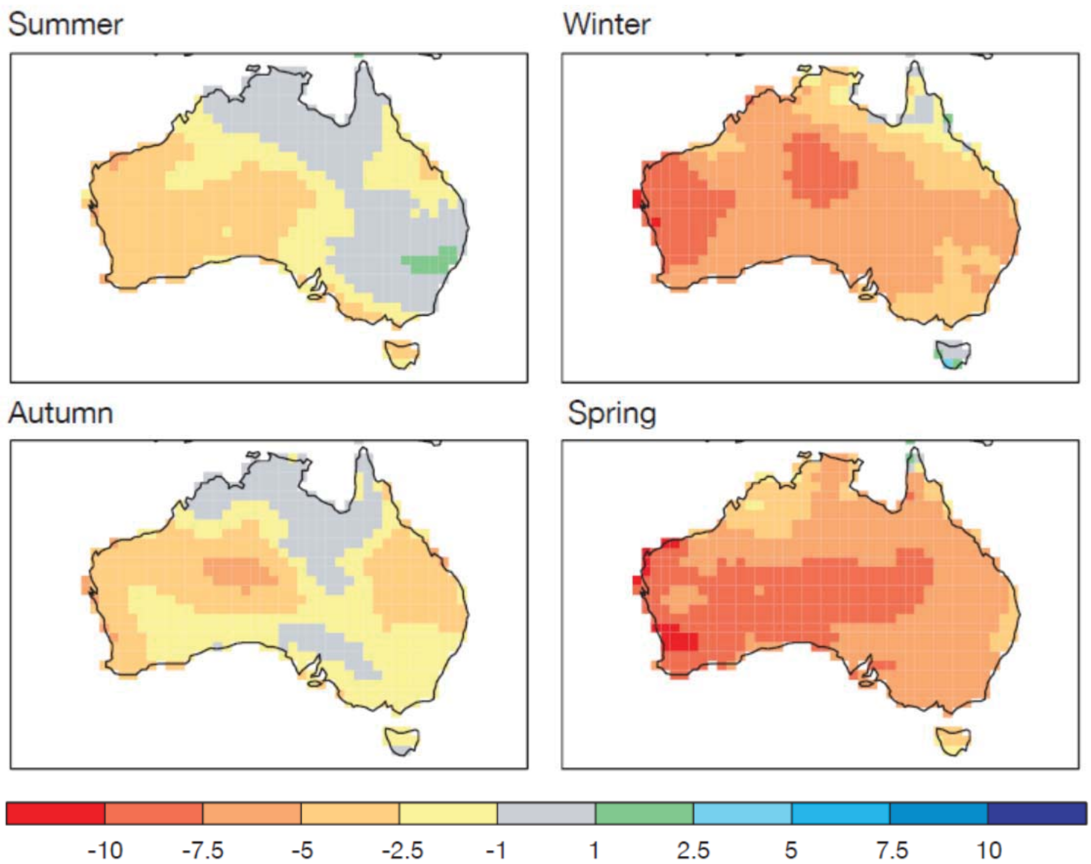


Figure 7: Best estimate of the projected change of Australian seasonal precipitation over land by 2030 for the A1B emission scenario, as a percentage of 1961 -1990 values (CSIRO 2007).

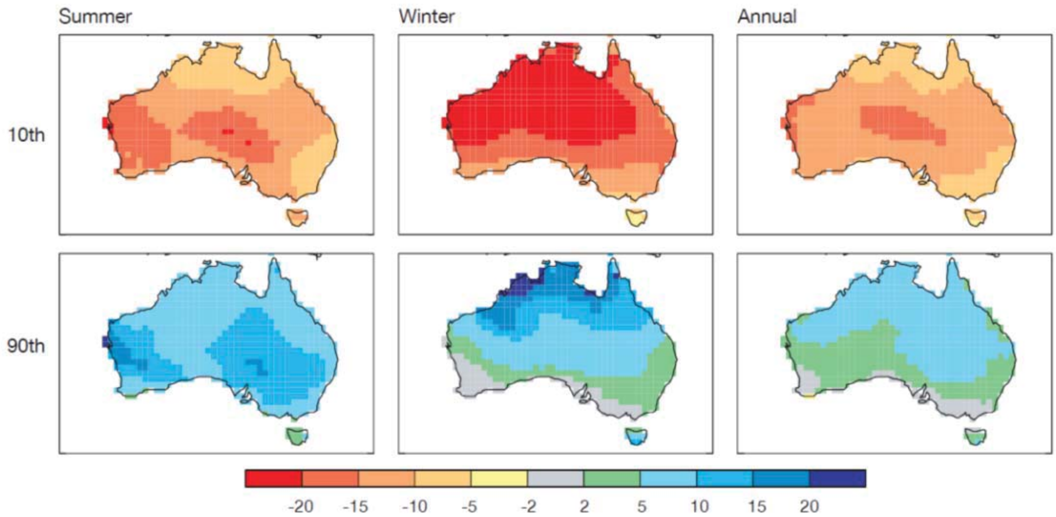


Figure 8: Projected Australian precipitation range (%) by 2030 for scenario A1B from the 10^{th} to 90^{th} percentile value for summer, winter and annual (CSIRO 2007).

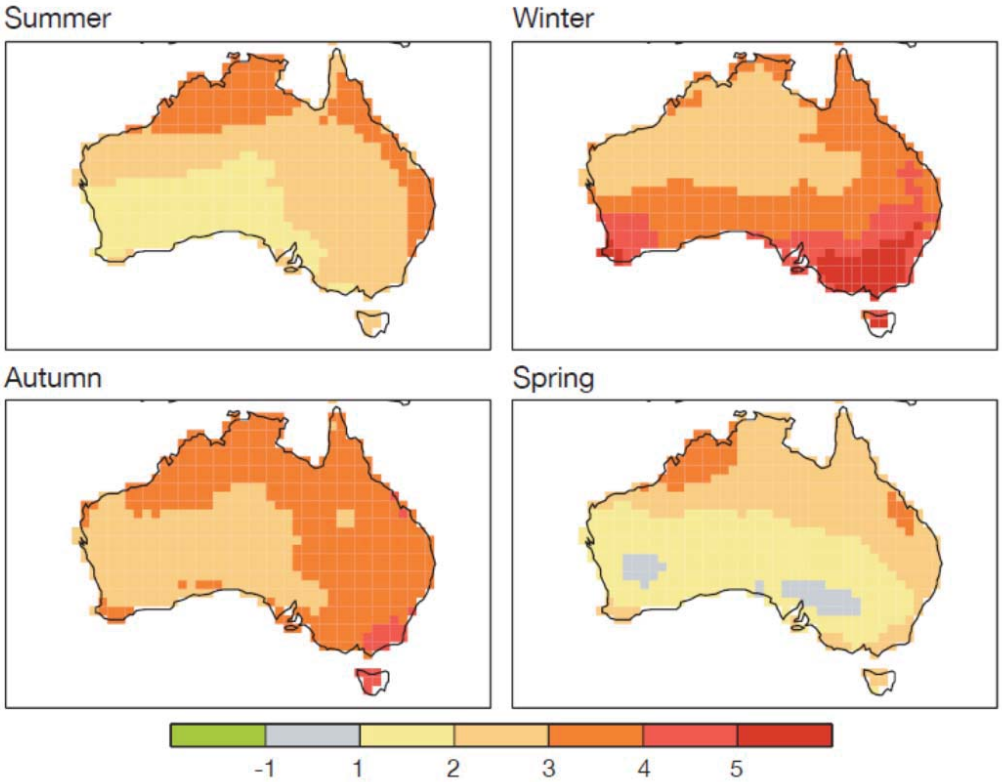


Figure 9: Changes in median (50th percentile) potential evaporation (%) by 2030, relative to 1990 under the A1B emission scenario, based on 14 climate models in summer, autumn, winter and spring (CSIRO 2007).

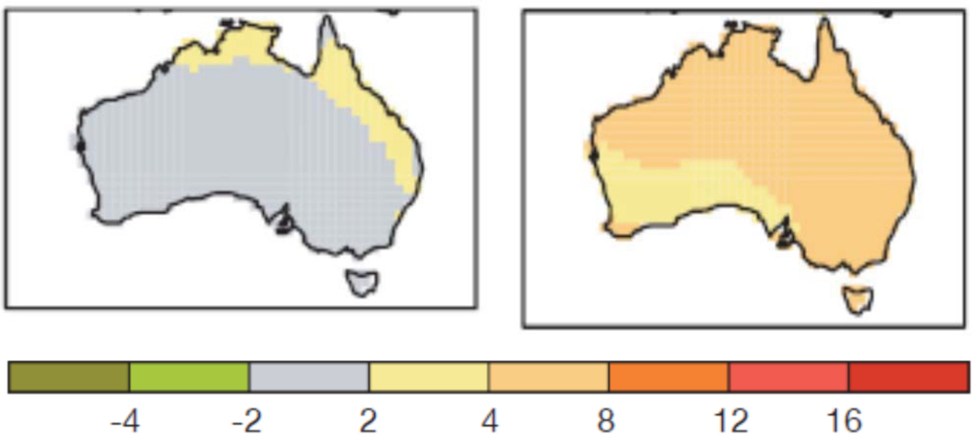


Figure 10: 10th (left) and 90th (right) percentile changes in annual potential evapotranspiration (%) in 2030 relative to 1990 under the A1B emission scenario, based on 14 climate models (CSIRO 2007).

Seasonal potential evapotransporation is expected to be the highest in winter under the A1B emission scenario for 2030, with an expected increase of 5% occurring in the south-west and south-east regions (Fig 9). 10th and 90th percentile projections show a potential range in annual evapotranspiration from +2% to +8% (Figure 10). Expected decline in precipitation, along with increased temperature and evapotranspiration in Australia, particularly in the southern regions, will result in continued decline of water levels in freshwater ecosystems and a reduction in groundwater recharge (Hennessey *et al.* 2007). It is expected that limited water availability for anthropogenic purposes will lead to increased groundwater extraction, further contributing to the stress of groundwater dependent ecosystems (Hennessey *et al.*2007). The high level of variability shown by the 10th and 90th percentile projections highlights the uncertainty involved with climate change projections. However, despite this uncertainty all climate change models show a decline in precipitation in the south-west region (Kauhanen *et al.* 2011, McFarlane *et al.* 2012). Global climate change models also consistently show high probabilities of declining rainfall in other Mediterranean areas (the south west United States of America, the Mediterranean, South Africa and southern South America) (Christensen *et al.* 2007).

2.3 Drought

Drought is a natural component of climate variability and is an abnormal period of dry weather and a lack of precipitation (meteorological drought) (Mpelasoka *et al.* 2008). This causes a hydrological imbalance (hydrological drought) and results in moisture deficiencies which can seriously impact surface water, ground water, reductions in water level, declined water quality and disturb riparian and wetland habitats (Riebsame *et al*. 1991 in Mpelasoka *et al.* 2008). The Australian climate experiences seasonally occurring drought in short periods of “frequent and predictable” drying (Bond *et al.* 2008). The intensity and duration of hydrological droughts in Australia is expected to increase by up to 20% (from the baseline 1974-2003), with an increase of 40% in the south-west (Mpelasoka *et al.*). This will see a shift towards “aberrant and unpredictable” ‘supra-seasonal drought’ (Bates *et al.* 2008, Bond *et al.* 2008). The lack of seasonal rainfall leads to a loss of soil-moisture, surface runoff, surface water inundation and groundwater recharge (Bond *et al.* 2008, Dunlop and Brown 2008, Nielsen and Brock 2009). At the first stages of supra-seasonal drought marginal habitats will become isolated due to the contraction of standing and flowing waters. If it persists, running water may be diminished to small isolated pools which may also eventually disappear (Boulton 2003, Bond *et al.* 2008). Due to the prevalence of drought in the Australian climate biota have become well adapted to survive in freshwater ecosystems that experience seasonal drought. However, they are not adapted to supra-seasonal drought and freshwater biota may experience losses and local extinctions (Boulton 2003, Bond *et al.* 2008, Morrongiello *et al.* 2011).

3. EXPOSURE AND VULNERABILITY

3.1 How does declining rainfall affect groundwater levels?

Reduced rainfall can result in both direct and indirect impacts on groundwater availability. Reduced groundwater recharge occurs as a direct effect of reduced rainfall and runoff. Declining rainfall, together with increased temperature and evaporation, will result in a decline in runoff and groundwater recharge (Tomlinson and Boulton 2008), with a lag between surface water decline and groundwater decline (Bond *et al.* 2008).

The level of recharge varies depending on the type of soil, and the amount and type of land cover (CSIRO 2009a). However, reduction in runoff is disproportionately larger than the reduction in rainfall (Chiew and McMahon 2002). This is because runoff is more likely to occur when rain falls on soil that is completely saturated (Roberts 2002). As a result the amount of runoff experienced due to climate change will vary depending on the catchment. Chiew and McMahon (2002) simulated hydrological models under various climate change emission scenarios and determined that ephemeral catchments with low runoff coefficients (due to unsaturated soils) may experience four times the percentage decline in runoff than the percentage decline in rainfall. Wet and temperate catchments are likely to experience twice the percentage decline in runoff than the percentage decline in rainfall. In temperate catchments change in rainfall has a small effect on the soil moisture level while in drier catchments the percentage change in soil wetness can be greater than the percentage change in rainfall.

Runoff simulations undertaken in eight catchments in different regions across Australia (Chiew and McMahon) indicated that the largest declines in runoff were projected for south-eastern Australia (-20%) and the South Australian Gulf (-25%). The south-west coast was projected to experience a varied decline in runoff (- 25 to +10). As changes in precipitation patterns are difficult to project, these simulations involve uncertainty (Chiew and McMahon 2002), however, they highlight the potential impacts climate change might have on freshwater ecosystems. This discusses the direct effect of reduced rainfall on groundwater recharge.

Reduced rainfall also indirectly effects reduces groundwater levels. As a consequence of reduced surface water availability due to declining rainfall, increased extraction occurs where groundwater is available (Gemitzi and Stefanopolous 2011). For example, Taylor *et al*. (2012) described increased groundwater extraction during a drought (2006 – 2009) in the Californian Central Valley and severe degradation of groundwater resources in the coastal Chaouia aquifer in Morocco has been shown to be primarily due to intensive extraction during intense drought (Moustadraf *et al*. 2008). Groundwater is often exploited (extraction exceeds recharge) in arid and semi-arid regions (Margrat *et al*. 2006), with groundwater depletion noted in support of irrigated agriculture in the arid and semi-arid regions of North China Plain, Northwest India and the USA high plains (Taylor *et al.* 2012). Increased groundwater extraction due to reduced availability of surface water has also been experienced in the south-west. With a further decline in rainfall due to anthropogenic climate change, this trend is expected to increase into the future (McFarlane *et al.* 2012).

While significant research has evaluated the effects of climate change on water resources in general, fewer studies have been undertaken on the effects of climate change on groundwater and the consequences are uncertain (UNESCO 2008, Treidel *et al*. 2012).

In south-western Australia extensive research has been carried out by the CSIRO sustainable yields projects to determine the effect climate change (projected to 2030) is likely to have on surface and groundwater yields. From the CSIRO sustainable yields projects in the south – west division it was determined that groundwater yields would decrease significantly in the Gnangara Mound, which is a source of drinking water to the Perth Metropolitan region (Australian State of the Environment Committee 2011; CSIRO 2007). Ecological river functions that require high river flows and wetlands with a surface expression of groundwater were identified as the ecosystems most vulnerable to climate change (CSIRO 2009a, CSIRO 2009b, Australian State of the Environment Committee 2011).

The CSIRO sustainable yields project have also carried assessments on Tasmania and the Murray-Darling Basin and are currently working on the Northern Australia and the Great Artesian Basin. From these studies it was determined that the Murray-Darling Basin and Tasmanian divisions would also receive reduced rainfall, however, the level of impact this would have on groundwater dependent ecosystems was uncertain (Murray-Darling Division) or minor (Tasmanian Division) (Australian State of the Environment Committee 2011).

3.2 Types of GDEs

Hatton and Evans (1998) extensively researched the literature on GDEs in Australia. They identified four types of GDEs: terrestrial vegetation, river base flow systems, aquifer and cave ecosystems and wetlands (including mound springs). In their review Hatton and Evans (1998) gave examples of where each GDE occurs in Australia. SKM (2001) built on this review by identifying two additional GDES: terrestrial fauna and estuarine and near-shore marine ecosystems. Eamus *et al*. (2006) determined a methodology for determining the water requirements of GDEs. Three classes of GDEs were suggested to simplify the categories: aquifer and cave ecosystems (type 1), ecosystems dependent on the surface expression of groundwater and ecosystems dependent on the subsurface presence of groundwater (type 3) (Eamus and Froend 2006, Eamus *et al.* 2006). GDE type 3 is outside the scope of this review.

The following GDEs will be considered in this review: river base flow, wetlands and aquifer and cave ecosystems (Hatton and Evans, 1998). Terrestrial vegetation, terrestrial fauna and estuarine and near shore environments are outside the scope of this review.

River base flow systems include the ecosystems dependent on base flow in streams and rivers supplied by groundwater. The base flow is the portion of the stream flow comprised of groundwater discharge and bank storage. Examples of river base flow systems include the dry season flow in permanent and semi-permanent streams in northern Australia, dry season flow in coastal rivers from the north-west to the northeast and annual flow in coastal rivers of south-eastern Australia. The riparian and aquatic biota in base flow dependent ecosystems would be directly or indirectly groundwater dependent themselves (Hatton and Evans 1998, SKM 2001).

Groundwater dependent wetland ecosystems: the minimal requirement for these ecosystems is seasonal waterlogging or inundation. They are highly diverse and include mesophyll palm vine forests, paperbark swamp forests and woodlands, swamp sclerophyll forests and woodlands, swamp scrubs and heaths, swamp shrublands, sedgelands, swamp grasslands, swamp herblands and mound spring ecosystems (entirely dependent on groundwater).

Cave and aquifer ecosystems: these include the ecosystems found within the free water of aquifers and cave systems. Cave ecosystems are found in karst landscapes which are scattered in the outcrops in the highlands from northern Queensland to Tasmania with particular prevalence in south-eastern Australia. They are also found in the south-westerly facing dunal limestones from Cape Range in Western Australia to Victoria; marine limestones in the Nullarbor; in the Kimberley and across northern Australia (Finlayson and Hamilton-Smith 2003 in Tomlinson and Boulton 2008). Aquifer ecosystems are found in the following habitats: unconsolidated aquifers, fractured rock aquifers, calcrete aquifers and pisolite (Tomlinson and Boulton 2008). The location of Karst Rock systems has been studied and mapped by Hamilton-Smith (2003 in Richardson *et al.* 2011). Stygofauna are the highly specialised invertebrates found in subterranean ecosystems and are well adapted to darkness, low energy, low oxygen availability and consistent undisturbed habitats (SKM 2001).

3.3 Physical changes to GDEs related to groundwater decline

Ecosystems are dependent on one or more of the following groundwater attributes to maintain the ecosystem (SKM 2001):

Flow or flux is the rate and volume of groundwater flow and is important for determining the amount of water available to the ecosystem. This applies to ecosystems which rely on groundwater discharge to maintain the water levels (e.g. cave systems, river base flow systems, groundwater-fed wetlands) and ecosystems that rely solely on groundwater discharge (eg: mound springs). Flow or flux maintains an appropriate supply of water to these GDEs and stops the ecosystems from completely drying out in summer, providing a refuge to aquatic biota (Murray *et al.* 2003) and ensuring the survival of species which depend on water both directly and indirectly. It is also important that the quantity of water received during the wet season is enough to support the biota until the end of the dry season (Murray *et al.* 2003).

Ecosystem usage of groundwater in unconfined aquifers is determined by the level or depth of the water table (SKM 2001). These ecosystems include wetlands fed by unconfined aquifers, some cave and aquifer systems and base-flow dependent ecosystems.

For confined aquifers the pressure or the potentiometric head of the aquifer will determine the level of groundwater discharged (Eamus *et al.* 2006; SKM 2001). This is of particular concern to mound springs of the Great Artesian Basin, for example.

All of the groundwater dependent ecosystems considered in this review require a sufficient groundwater level and groundwater flux to ensure adequate water supply (SKM 2001; Hatton and Evans 1998). Cave ecosystems are particularly sensitive to alterations in groundwater level because species composition changes with depth. Reduction in water level could result in a loss or change in ecosystem structure and function and the potential for species loss (Froend and Loomes 2004; SKM 2001). The timing of the discharge is also important because this can determine whether there is enough water available to biota at the end of the dry season when they are most vulnerable (Murray *et al.*2003). A consistent annual input of groundwater is important to most GDEs as it will maintain ecosystem processes and structure (Murray *et al.* 2003).

The quality or physiochemical properties of the groundwater (pH, salinity, nutrients, contaminants) (SKM 2001) are important because they determine which species can survive in the ecosystem. Stygofauna are particularly sensitive to subtle changes in physiochemical properties due to the highly stable environment they have developed in (SKM 2001).

3.4 Level of GDE dependence on groundwater

Hatton and Evans (1998) determined the level of dependence Australian GDEs have on groundwater:

Ecosystems entirely dependent on groundwater would disappear if the groundwater were to be depleted or altered slightly, either below a threshold (eg: the water level fell below the surface) or a surface system ceases to flow. Some of the examples Hatton and Evans (1998) gave are listed below:

- Mound spring ecosystems of the Great Artesian Basin
- Karstic groundwater ecosystems in the Exmouth Cape, Yanchep Caves and the Nullarbor
- Permanent lakes and associated ecosystems on the Swan Coastal Plain
- Saline discharge lakes of the Murray Darling Basin

The health and expanse of ecosystems highly dependent on groundwater would substantially decline with considerable changes to the groundwater discharge or water level. There is the potential for the ecosystem to collapse, however this is not certain. Examples include (Hatton and Evans 1998):

- Karst ecosystems on the Nullarbor
- Permanent water holes on the rivers of the central lowlands of South Australia
- Damplands on the Swan Coastal Plain
- Base-flow ecosystems of the south-eastern uplands
- RAMSAR listed wetlands in the Basalt plains of the western district of Victoria

Ecosystems with proportional dependence: a change in the amount of groundwater will result in a proportional change in ecosystem health or expanse. This category contains the largest diversity of ecosystems and includes base-flow systems and a large diversity of GDE wetlands (Hatton and Evans 1998).

Ecosystems which may only use groundwater opportunistically or to a small extent are important in supplying water to these ecosystems during extreme drought or at the end of the dry season. There may not be an immediate result in groundwater water decline due to the long term dependency required by these ecosystems (Hatton and Evans 1998). The persistence of these systems in to the future may entirely depend on groundwater availability during extreme climatic events (SKM 2001). Examples include the ecosystems of the Coorong, ecosystems of permanent lakes and swamps at the end of inland rivers in the Central Lowlands and South Australian Ranges and intermittent floodplain lakes of the Central Lowlands (Hatton and Evans 1998).

4. EFFECTS ASSESSMENT

4.1 Physiochemical effects of declining water levels

With declining water levels, freshwater ecosystems are likely to experience decreased flushing and evapoconcentration of nutrients, salts and pollutants. This can lead to shifts in faunal assemblages, a reduction in biodiversity and productivity of GDEs. Reduced groundwater, relative to surface water input can cause poor water quality as surface water can often have greater loads of nutrients and pollutants than groundwater. Decreased soil moisture due to drying and increased evaporation will affect biogeochemical reactions such as oxidisation, decomposition and nutrient dynamics as well as and physical processes such as wind erosion (Bond *et al*. 2008). Eventually a reduction in flooding of temporary wetlands and river base flow systems may mean that they do not hold water for long enough to support hydric communities (Nielsen and Brock 2009). Complete drying of the wetlands can expose pyritic sediments to oxygen, resulting in the production of acid sulfate soils.

Specific changes in water quality resulting from reduce groundwater input and water levels include changes in:

Temperature: Increased temperature (both average and diurnal maxima and ranges) can lead to death of aquatic organisms (Davies 2010). Earlier and longer lasting stratification through increased surface heating can lead to a dominance of cyanobacteria (especially if high temperature is combined with high nutrient loads) and consequently a loss of phytoplankton diversity (Bates *et al.* 2011). Stratification will also lead to increased stagnant conditions resulting in decreased dissolved oxygen and increased sediment nutrient release (Bond *et al.* 2008).

Salinity: Salinities can increase due to decreased flushing and evapoconcentration. This can lead to shifts in faunal assemblages to more salt tolerant biota, a reduction in biodiversity and productivity of wetland ecosystems (Smith *et al.* 2007, Tomlinson and Boulton 2008).

Nutrients: Loss of surface water due to drawdown and evaporation will result in a build-up of nutrients, especially nitrogen and phosphorus (Bond *et al.* 2008). This is a function of reduced flushing, increased evapoconcentration and relatively increased inflow from surface water. Drying of sediments can result in pulse of phosphorus release from the sediment at rewetting (Bond *et al.* 2008).

pH: Sediment containing AASS or PASS must remain permanently saturated to prevent acidic surface water and an acidic groundwater plume (Froend *et al.* 2004). Complete drying of the wetlands can expose pyritic sediments to oxygen, resulting in the production of acid sulfate soils. This has already occurred in wetlands in the south-west of Western Australia (eg: Lake Jandabup) due to groundwater extraction. A reduction in pH from 6-7 to 4-5 saw a shift in community structure due to a loss of biota sensitive to acidic conditions (Sommer and Horwitz 2001).

Soil characteristics: Decreased soil moisture due to drying and increased evaporation will affect biogeochemical reactions such as oxidisation, decomposition and nutrient dynamics as well as and physical processes such as wind erosion (Bond *et al*. 2008). This can result in compacting of clays and formation of hydrophobic sands, that do not regain original soil structure on rewetting. Organic matter can be lost though oxidisation modifying the capacity for microbial recolonisation, further affecting soil processes.

4.2 Biotic effects of declining water levels

Australia is a land of climatic variability where drought and floods are a natural part of the environment. A large number of our wetlands and rivers are seasonally, episodically or ephemerally flooded and the biota has adaptations to survive these natural perturbations (Humphries and Baldwin 2003). Native biota possess adaptations to survive drought (resistance traits e.g. desiccation life history stages: eggs, seeds, cysts) or make use of refuges in the landscape to survive in an otherwise harsh environment (resilience traits – Bond *et al.* 2008). Populations in these refuges act as source populations for consequent recolonisation of the drought affected area and have the capacity for widespread and rapid dispersal (Refs in Bond *et al.* 2008). One of the main concerns about climate change is the long-term exceedance of these resilience and resistance capacities leading to the reduction and possible extinction of species. For adaptation to climate change the identification of naturally-occurring refuges or the capacity to create refuges in the landscape is critical to ensuring biodiversity into the future (Robson *et al.* 2008).

4.3 Reduction in habitat

Freshwater flora are well adapted to various wetting and drying regimes, providing heterogeneity of habitats at the landscape level. A large range of ecosystems are capable of providing different types of ecosystems which will support a diverse range of biota, therefore providing biota with a greater level of resilience to change. Heterogeneity at the landscape level provides biota with a greater level of resilience (Davis and Froend 1999, Brock and Jarman 2000). Surface water drawdown due to climate change will result in a reduction in spatial and temporal variability of hydrological regimes, reducing the heterogeneity of wetland types and habitats and thereby biological diversity both directly and indirectly (Davis and Froend 1999). For example, surface water drawdown will isolate the productive littoral zone, resulting in a loss of sessile fauna and flora such as snails and fringing vegetation (Davis and Froend 1999). In groundwater dependent lotic systems, discharge during baseflow can maintain perennial riffle habitats (Beatty *et al.* 2010). The important components of the wetland regime are the quantity of water and the timing, duration and frequency of inundation (Davis and Froend 1999). Seasonality results in varying water quantity over time, providing habitats of different depths throughout the year. Varying depths are characterised by particular plant and waterbird species (Davis *et al.* 2001). In general the water quantity available can dramatically change the available habitat area, with a small decrease in water volume resulting in a large reduction of inundated area (Taylor *et al*., 1996 in Davis *et al.* 2001).

4.4 Elements of water regime vital to biodiversity

The distribution of species is determined by the level of water availability they require (Roberts *et al.* 1999 in Froend and Loomes 2004). This includes:

Depth: Different species are able to tolerate different water depths. Some species require free water (eg fish) while others are able to tolerate flooding or waterlogged soils (eg sedges). For example, groundwater discharge during the baseflow period has been demonstrated to maintain access to riffle habitats for fish in south-western Australia's largest river system (Beatty *et al.* 2010).

Duration (time surface water is present) of inundation: defines the potential growing period for adults and the interval in which the life cycle can be completed to reproduction for future generations. The length of time soils are wet, moist or dry is a key determinant of species type and of opportunities for recruitment (e.g. terrestrial-breeding amphibians). It also determines the availability of oxygen/carbon dioxide n the sediment and affects nutrient dynamics and other microbial processes. A minimum period of 4 months/yr is required for macroinvertebrates to complete their lifecycle with max drying rate of 0.02m/day to enable invertebrates that lay eggs in flooded fringing vegetation to complete life cycle in open water before the wetland dried (WAMA 1995 in Davis *et al.* 2001). In south-western Australia, aestivating freshwater fishes in seasonal lentic systems require surface waters in order to complete their lifecycles, although it is unclear what the minimum inundation period is required for population viability (Morgan *et al.*1998).

Season/timing of flooding: impacts on climatic factors that affect plants: temperature (germination temperature thresholds), day-length (energy available for photosynthesis) and whether day length is increasing/decreasing. For fish and other fauna day length and temperature can be important as reproductive cues (Morgan *et al.* 1998, Pusey and Arthington 2003).

Rate of rise or drawdown: Particularly for plants rapid rates of rise can drown plants (Brock and Casanova 1997) while rapid drawdown can prevent the plants from growing roots to the water table.

Frequency of flooding events: Flooding at a critical frequency is essential for species dependent on the seedbank to maintain their presence. In the absence of flooding, a seedbank ages and propagules lose viability. Frequent flooding and drying can cause false starts causing eggs and seeds to hatch/germinate but prevent organisms from completing their life cycle resulting in depletion of the egg/seedbank (Bishop 1967).

Inter-flood interval: length and recurrence of this is most relevant to those plants that maintain a low level of growth in the absence of flooding.

4.5 Seasonality/flushing life cycle

Altered water regime (e.g. delayed onset of winter filling, premature drying, extended dry spells, unseasonal rainfall due to cyclonic activity) has the potential to disrupt reproductive cycles of biota. Changes to seasonal migration triggers. Depletion of seed and egg banks.

4.6 Connectivity

With decreased availability of surface water due to climate change and anthropogenic stressors, water levels will reduce and temporary wetlands may disappear altogether (Nielsen and Brock). This will increase the distance between available surface water and will have an impact on biota by reducing the available habitat (Davis and Froend 1999) and potentially reducing gene flow (Berry 2001). In rivers loss of flow can prevent upstream migrations (Robson *et al.* 2008) or strand organisms in pools or channels where conditions may become deleterious to survival. Natural and man made barriers to dispersal and migration prevent biota from finding aquatic habitats (Davis *et al.* 2001). This could result in potential extinctions of endemic species unable to cope with the rate of change, especially those with poor dispersal mechanisms or in habitats affected by human activity (Brinson and Malvarez 2002). Drought occurs at the landscape level and therefore has the potential to affect the populations of entire species rather than individuals.

4.7 Biota Dependence

In a study on ecological water requirements of wetlands of environmental significance on the Swan Coastal Plain (Gnangara and Jandakot mound), Froend *et al*. (2004) put fauna into four categories based on their dependency on groundwater: low, moderate, high and very high. With the exception of invertebrates, fish, amphibians and turtles, most fauna are indirectly dependent on groundwater.

Low dependence on groundwater included species that occur in upland habitats or do not have a preference towards groundwater dependent vegetation. *Moderate* dependence was characterised by seasonal use of groundwater dependent vegetation or species that make some use of habitats that are dependent on groundwater. *High* dependence were species that make use of phreatophytic vegetation for habitat. Because phreatophytic vegetation occurs low in the landscape a reduction in groundwater would result in a loss of vegetation rather than a shift in its position, resulting in a loss of dependent fauna. *Very high* dependence were those species that rely on aquatic habitats. These species would not survive if a loss of surface water (e.g.macroinvertebrates, fish or frogs that require free water for all or part of their life cycle). Most turtles (e.g. *Chelodina oblonga*) are highly dependent and require permanent or near-permanent surface water, although they can tolerate drying for up to six months (Froend *et al.* 2004).

In a study to determine the environmental drivers which could explain the floristic changes to phreatophytic vegetation on the Gnangara Mound, Western Australia, it was determined that groundwater was the main environmental driver for predicting the species composition. It was concluded that a continued decline in groundwater levels and reduced rainfall would contribute to a further thinning of remnant vegetation, with losses of wet-adapted and moist-adapted species (Froend *et al.* 2004; Sommer and Froend 2010).

5. THRESHOLD / LEVELS OF ACCEPTABLE CHANGE

The response to an environmental perturbation can take different forms. Proportional response describes a progressive decline in ecosystem processes with a change in the water regime, while threshold responses occur when little change is evident until a particular threshold is reached and rapid and extensive changes follow (Froend and Loomes 2004). In reality, ecosystem responses to water regime changes are often a combination of these. Methods that can be used to describe response functions include benchmarking with similar types of ecosystems, interpretation from historical records and expert opinion.

Proportional or linear responses are of less concern since the trajectory of recovery can be the same as that of degradation i.e. reducing the stressor will revive the ecosystem health. However for vital requirements, thresholds can be evident eg sufficient oxygen to allow organisms to respire, lethal tolerance limits of pH or salinity, removal of connectivity to spawning grounds, absence of surface water as a habitat for fish. Restoration after crossing thresholds can be more problematic. The identification of thresholds however can be a valuable management tool as they define the boundaries in which ecosystem health can be maintained and provides targets for water quality and quantity (SKM 2001).

Ecological regime models conceptualize ecosystem change and have been used in aquatic ecosystems to examine the impact of multiple stressors on an ecosystem (Gordon *et al.* 2008, Davis *et al.* 2010). They provide ecologists with a tool for anticipating abrupt changes or regime shifts to an ecosystem, for example ‘ecological surprises’ or ‘catastrophic ecological regime shifts’ (Gordon *et al.* 2008, Davis *et al.* 2010). Regime shifts have occurred in water bodies in southern Australia where changes in land use due to urbanisation and agriculture due to multiple external stressors (such as nutrient loadings, secondary salinisation, acidification, increased water depth and altered hydrological regimes) and gradual internal changes (such as loss of ecosystem processes and services). This has resulted in regime shifts over different scales of space and time.

5.1 Models of ecosystem response

Ecosystems have four response types to environmental stressors when changing from a desired (A) to a degraded (B) regime (Figure 11). Desirable states can be defined as those providing the system with optimal ecosystem function and resilience. Gradual ecological change occurs when an ecosystem responds to an external stressor by changing from a desirable to a degraded state in a linear relationship (Sim *et al.* 2009, Davis *et al.*2010). If the external stressor were removed, then the ecosystem would return to a desirable state. Similarly ecosystem restoration can be accomplished by reducing the severity of the external stressor (Davis *et al.*2010).

The threshold model is a non-linear response model, with an abrupt change or regime shift from a desirable to a degraded state occurring with a small change in the external stressor. Restoration will occur if the external stressor is decreased below the threshold level (Davis *et al*. 2010).

The hysteresis model is a non-linear response model, with an abrupt change or regime shift occurring in response to a small change in the external stressor. The pathway the ecosystem takes during a regime shift is different to that of recovery. To achieve restoration the external stressor must be reduced below the threshold that initially caused the collapse (Davis *et al*. 2010).

Irreversible change describes a non-linear response model, with an abrupt shift or regime change occurring over a small change in the external stressor. There is no pathway for recovery, leading to a catastrophic change in ecosystem services (Davis *et al*. 2010). The non-linear, abrupt regime shifts (threshold model, hysteresis model and model of irreversible change) all have the possibility for an 'ecological surprise' or catastrophic regime change' (Gordon *et al.* 2008, Davis *et al.* 2010).

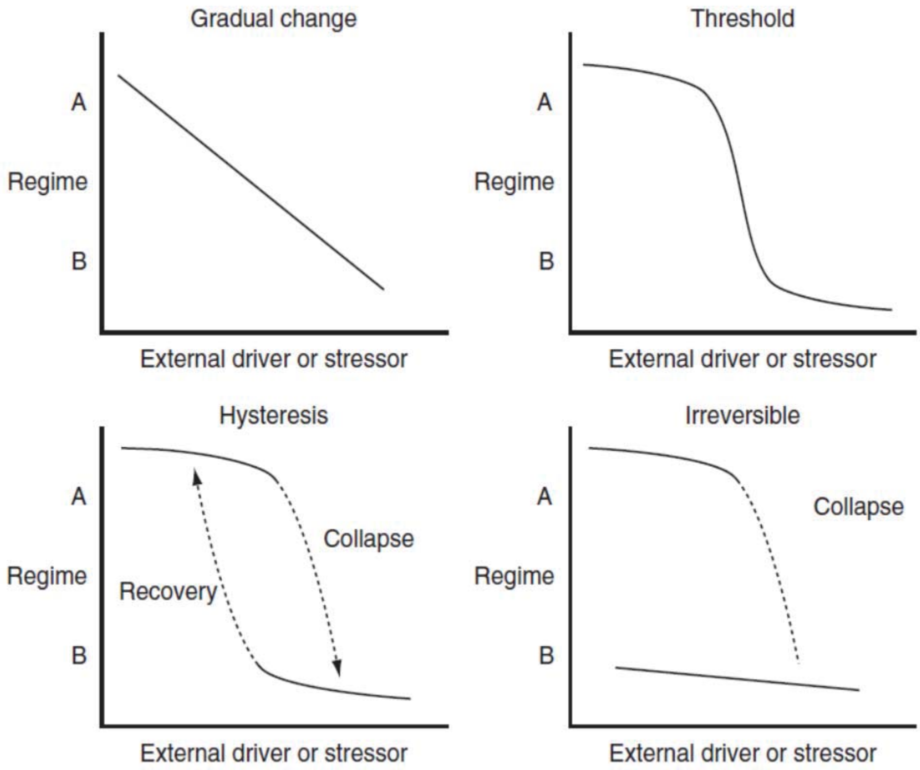


Figure 11: The model of gradual ecological shows a linear response from a desired (A) to a degraded (B) regime due to an external driver or stressor. A gradual change in the ecosystem stressors results in a gradual shift from regime A to regime B. Degradation can be reversed by reducing the external driver or stressor. The threshold model is a nonlinear response to an external driver or stressor. An abrupt change from A to B can occur with a relatively small change in the external driver or stressor. A decrease in the external stressor below the threshold value can restore the ecosystem to regime A. The hysteresis model shows a nonlinear response, where the threshold of change differs between the pathway of collapse from regime A to B and the pathway of recovery from B to A. The final model describes an irreversible change from A to B (Davis *et al.* 2010) (Figure courtesy of Wiley).

A linear response might be evident where more gradual change in the health, composition and/or ecological function of communities is expected as, for example, may occur with increasing groundwater salinity or contaminant concentration. However this might also be defined as a series of thresholds for individual types of fauna or species. An example of a threshold response might be individual mound spring communities supported by groundwaters of the Great Artesian Basin (Bekesi *et al.* 2009). These would cease to exist if pressures in the GAB fell to the point where there was no further surface discharge (SKM 2001).

In urban wetlands in Perth groundwater extraction from the unconfined aquifers has resulted in drying and terrestrialisation of some wetlands. Drying of these wetlands has exposed pyritic sediments resulting in acidification. For example, Lake Gnangara is now permanently acidic (pH < 3) and is dominated by a benthic microbial community. This is an example of a catastrophic regime shift and is an irreversible change (Sim *et al.* 2009).

5.2 Identifying levels of unacceptable change

Identifying unacceptable change can be difficult and requires an understanding of wetland ecosystem components, processes and services. Davis and Brock (2008) suggest development of conceptual models to identify components and key processes and drivers in the ecosystem. These models provide a description of elements of the ecosystem and a baseline against which unacceptable changes in biophysical character can be assessed. They propose that the ecosystem under consideration is assessed (i) at a landscape context identifying relevant spatial scales, (ii) through development of a conceptual ecological model of relevant biophysical data; and (iii) uses these as the basis for recognizing a unique set of wetland identifiers. Construction of a driver/stressor model facilitates the recognition of external drivers which create ecological stressors and lead to adverse ecological effects and loss of ecosystem function and resilience. Unacceptable changes in ecological character are those that result in a loss of identifiers, disrupt critical processes and reduce function, services or benefits (Davis and Brock 2008).

5.3 Methods to identify thresholds in wetland ecosystems

The focus of species-environment modelling has shifted to a predictive approach due to factors such as climate change and the requirement for determining the likely impact changing environmental variables will have on biota. Classification and regression tree (CART) analysis are a useful tool for determining the relationship between species or species assemblages and their environment. Univariate regression trees (URT; eg: CART) explain the variation of a single response variable (eg: species) using environemntal characteristics. Multivariate regression tree analysis (MRT) replace the univariate response with a multivariate response such as species assemblages (De'ath and Fabricius 2000, De'ath 2002). They are well suited for 'predictive ecology' and the analysis of complex ecological systems because they are able to deal with missing data, non-linear relationships and interactions between environmental variables (De'ath and Fabricius 2000, De'ath 2002, Sommer and Froend 2010). They have been used in risk assessment because they are able to reveal environmental thresholds that assessors can use to determine the level of risk species or species assemblage are exposed to. For example, the relationship between River Red Gum health and the flooding regime of the Murray Darling Basin was examined in a study by Wen *et al.* (2009). CART analysis was used to quantitatively determine the inundation thresholds required by the species, making it possible to categorise the species condition in a way that can be included in wetland management targets (Wen *et al.* 2009). One of the main advantages of this analysis is that they highlight the most critical environmental variables that determine thresholds (Wen *et al.* 2009, Sommer and Froend 2010). This is an advantage to risk assessment because it enables researchers to highlight which of the multiple hazards is likely to have the largest implications on the ecosystem (Sommer and Froend 2010).

6. APPROPRIATE TOOLS FOR RISK MANAGEMENT.

6.1 Conceptual Models

Conceptual models are a useful tool for simplifying how an ecosystem works and provide a synthesised diagram of the major ecosystem components and their interactions. They are useful for showing likely ecological responses to multiple anthropogenic and natural stressors and can be used to predict the consequences of future change such as management intervention or climate change. They ensure that all parties managing a system have the same level of understanding on ecosystem structure and function. Through the construction of conceptual models managers and researchers can identify knowledge gaps which need to be addressed (Richardson *et al.* 2011).

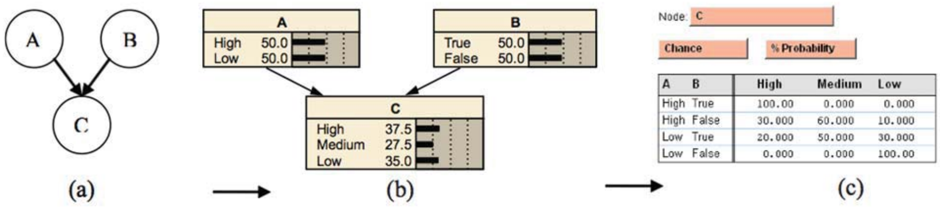
For GDE assessments the following components should be included in a good conceptual model (Richardson *et al.* 2011):

- the ecosystem's hydrogeological processes (such as surface water-to-groundwater interactions)
- biotic components such as groundwater dependent flora or fauna
- biotic processes such as primary production, herbivory, predation and competition
- other abiotic components such as recharge, discharge and storage processes, and mixing and groundwater flow
- the temporal patterns of groundwater levels (seasonal and inter-annual variability)
- the water regime that supports the GDE and the links between the abiotic and biotic components
- groundwater services critical to ecosystem function, such as artesian (or other) pressure, thermal water supply and nutrient supply.

6.2 Bayesian networks

A Bayesian Belief Network (BBN) is a graphical model which can be used to establish the causal relationships between key factors and final outcomes (Hart and Pollino 2006). BBN's can provide effective decision support tools for problems involving uncertainty and probabilistic reasoning (Cain 2001). The networks are models that represent the correlative and causal relationships between variables graphically and probabilistically (Cain 2001). BBNs can model a situation where causality plays a role but our understanding of what is going on is incomplete.

Bayesian Belief Networks are composed of a series of nodes, which represent a variable in the model. Each node has a number of states with an associated probability distribution. Where there is a casual link between nodes the nodes are linked, the relationship between the nodes is defined by a conditional probability table (Moe 2010). The conditional probability table represents likelihoods based on prior information or past experience (Anon 2008). The outcome of the BBN is a probability for the hypothesis, given the data or other evidence (Moe 2010). For example in Figure 1 an example of a simple network structure can be seen where nodes A and B represent causal factors influencing the probability of C (Figure 12a). The values of the nodes are defined in terms of states (Figure 12b). A conditional probability table (Figure 12c) defines the causal relationship between A, B and C. This results in the probability of the three outcomes of C (high, medium, low) occurring.



A	B	High	Medium	Low
High	True	100.00	0.000	0.000
High	False	30.000	60.000	10.000
Low	True	20.000	50.000	30.000
Low	False	0.000	0.000	100.00

Figure 12: Example of a simple Bayesian Network structure (Kragt 2009).

Bayesian Belief Networks are based Bayesian probability theory. Bayes rule states that for any two events, A and B, the probability of event B occurring given that event A has happen ($p(B|A)$) can be determined using the formula

$$p(B|A) = p(A|B) \times p(B)/p(A)$$

where $p(A|B)$ is the probability of event A occurring given B, $p(B)$ is the probability of event B and $p(A)$ is the probability of event A (Jensen and Nielsen 2007).

Bayesian probability theory allows the modelling of uncertainty and outcomes by combining expert knowledge and observational evidence. The probability is based on expert knowledge and data. When there is very little data the model will rely heavily on expert knowledge, where there is more data the model relies less on expert knowledge. One of the important features of BBNs is that the probabilities do not need to be exact to be useful. BBN's are generally robust to imperfect knowledge and approximate probabilities (even educated guesses) very often give very good results.

BBNs have a number of advantages (Jakeman *et al.* 2009):

- Easily updated with new submodels and new information
- Spatial and landscape components can be included as separate nodes
- Easily used as a tool for communicating complex environmental problems among experts, managers and stakeholders
- Can integrate models of different types
- Can be used as a decision making tool
- Transparent

Disadvantages include (Jakeman *et al.* 2009):

- Cannot be used as dynamic models (*e.g.* time step models)
- Cannot use feedback loops
- Variables must be discreet
- Not optimal for statistical inference
- Sometimes difficulty can be experienced in obtaining agreement on network structure

Bayesian belief networks have been used in a variety of fields including medicine, engineering, finance and ecology. BBNs have been used in a number of ecological and natural resource management contexts (Aguilera *et al.* 2011). An example of the use of Bayesian networks in natural resource management can be seen in (Chan *et al.* 2012) where BBNs were used to assist decision-making on the environmental flow requirements for the Daly River in the Northern Territory. In this case BBNs were used to determine the impacts of altered flows on the abundance of two fish species. Due to the lack of data the majority of the relationships between flow and fish abundance were defined by expert opinion, with data being used where available. When the model was validated with field data prediction errors were between 20 and 30%. The models indicated that an increase in water extraction would deleteriously impact on the fish populations.

Marcot *et al.* (2001) used BBNs to evaluate fish and wildlife population viability under a number of land management alternatives. The BBN modelled the ecological causal web of a number of key environmental variables that influenced habitat capability, potential population response for each species and the influence of habitat planning alternatives. The probabilities within the model were obtained through a mixture of empirical data and expert opinion. The modelling allowed identification of planning decisions and key environmental variables that most impacted on species viability and therefore helped to prioritise management activities.

For the modelling of the impacts of climate change on groundwater dependent ecosystems BBN’s are an appropriate tool because:

- Knowledge of the interactions involved in groundwater dependent ecosystems is incomplete therefore some of the processes have to be modeled using expert opinion on top of the available data, BBNs are very robust to the use of imperfect knowledge
- Much of the data on groundwater dependent ecosystems has spatial components, BBNs are composed of nodes which can incorporate separate spatial components
- This project aims to develop a tool for use in assessing the impact of climate change on groundwater dependent ecosystems. Due to its visual nature BBNs present an excellent tool not only for communicating the environmental issues and processes but also a means of gathering additional information to feed into models or develop new models
- BBNs are composed of nodes which allow the manipulation of starting conditions for the model, they therefore present a useful management tool to test different scenarios.

6.3 GIS/spatial assessment

Ecosystem management can be carried out at different scales, depending on the management objectives. Maintaining resources such as biological diversity and water quality require a regional or landscape scale, however, all management decisions lead to activities which alter landscape patterns. Consequently, managers require a tool for visualising ecosystem dynamics of the entire landscape to determine the impact of local and regional management decisions (Turner *et al.* 1995, Plant and Vayssiares

2000). Spatially explicit ecosystem models are useful for determining effective management by spatially representing species-habitat relationships, fundamental to understanding species ecology (Turner *et al.* 1995, Plant and Vayssiares 2000). By considering the ecosystem at a landscape scale questions of fragmentation, isolation, habitat shape, and patch size can be addressed. This can provide management with a means for determining the type of habitats and the spatial arrangement of habitats required at a landscape level (Turner *et al.* 1995). Appropriate management practices can be determined at the landscape level by testing 'what if' scenarios (Turner *et al.* 1995, Plant and Vayssiares 2000) such as the effect of no management, alternative management strategies and natural events (Turner *et al*. 1995).
For information on examples of methods for integration of GIS datasets with BBNs, refer to section 2.4.1 of SD7, Neville 2013.

6.4 Risk Assessment

Environmental risk assessment closes the gap between science and decision makers by identifying potential problems and prioritising management. Risk assessment can be either *retrospective* – minimising impact after the fact or *predictive* – assesses the potential for future damage. The basic elements of risk assessment are defined by Asante-Duah (1998):

<u>Hazard identification</u> involves identifying the level of impact that stressors may have on the ecosystem site. Potential stressors include exotic species, excessive nutrients, water diversion, the release of toxic chemicals, fire, and alteration to the frequency and severity of natural hazards. These have the potential to alter the landscape and ecosystem processes (Carpenter 1995).

<u>Effects assessment</u> estimates the level of exposure likely to be received and the level of severity this impact may have. It considers the types of adverse effects associated with the hazard, the relationship between the degree of exposure and detrimental effects, and the uncertainties involved.

An <u>exposure and vulnerability</u> assessment is conducted to estimate the actual and/or potential ecosystems exposed to the hazards, the frequency and intensity of exposure, the biological traits and size of the risk group, and the means by which the risk group may be exposed.

<u>Risk management</u> determines the probability that an ecosystem will be exposed to adverse effects under a set of exposure conditions. This will highlight which conditions will result in the greatest level of risk to ecosystems and prioritise management response (Asante-Duah 1998).

Comparative ecological risk assessment incorporates these basic elements of risk assessment highlighted by Asante-Duah (1998). This objective of this form of risk assessment is to broadly rank risk to ecosystem habitats by determining the impact of environmental issues to target management to deal with those sites most at risk of degradation. A practical method for comparative ecological risk assessment should qualitatively and methodically determine risk by combining best-judgement from ecologists and natural resource managers with on-site experience. The use of maps, easily interpreted scoring systems, well explained evaluative criteria and a manageable set of ecological stressors can be used to effectively communicate scientific knowledge to decision makers. During the first stage of this process it is important to define the specific problem areas and categorise the ecosystems of the study region (Carpenter 1995).

6.5 Current GDE Information/Toolboxes

Current information is reviewed to determine the resources available on GDEs in Australia that might complement this project. This information is also useful to identify GDE regions to which the framework could potentially be transferred.

The National Water Commission (NWC) has funded numerous projects on GDEs – the GDE toolbox, groundwater modelling guidelines and the GDE Atlas (Richardson *et al.* 2011, NWC 2012). The GDE toolbox is a revised version to the original (2007) toolbox which “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” (Richardson *et al.* 2011). The project developed a three stage assessment framework for determining the Ecological Water Requirements (EWR) of GDEs (Figure 13). Stage 1 involves a basic assessment of the type of GDE and requires the development of a conceptual model of the GDE. Stage 2 determines how reliant the ecosystem is on groundwater and stage 3 characterises the ecological response of GDEs to change. An important component of the GDE toolbox involves developing conceptual models of GDEs and examples are given for wetlands (Lexia Wetlands, Western Australia), Caves (Mole Creek Karst, Tasmania), Mound Springs (Great Artesian Basin) and a Monsoon Vine Thicket. The toolbox also explores the development of a risk assessment framework which can be incorporated into a BBN and adaptive management (Richardson *et al.* 2011). All of these tools are pertinent to this project.

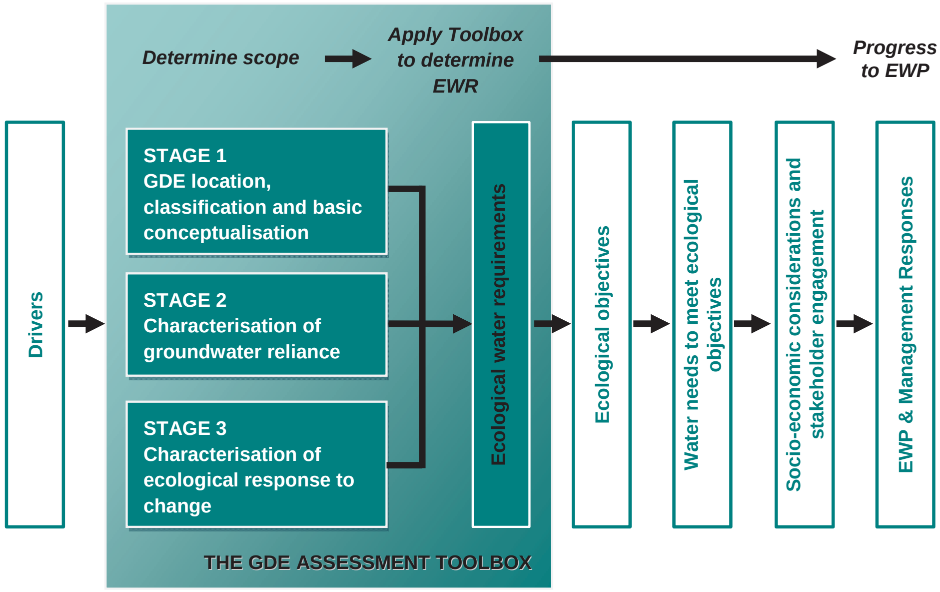


Figure 13: The three stage process for determining the EWRs of GDEs. The shaded section outlines the methodology outlined in the GDE toolbox (Richardson *et al.* 2011).

GDE modelling guidelines have recently been developed and provide an “overview of the way interaction between surface water and groundwater is conceptualised, and the approaches to design and construction of models that include surface water–groundwater interaction” (Barnett *et al.* 2012). Steps involved in determining surface water-groundwater connectivity include: conceptualisation, design and construction, calibration and sensitivity analysis, prediction, uncertainty analysis, and reporting (Barnett *et al.*2012). Once again this is a key element of our risk assessment framework.

A team funded by the National Water Commission consisting of Sinclair Knight Merz, CSIRO and Cogha, with input from state and territory agencies, have developed ‘The Atlas of Groundwater Dependent Ecosystems of Australia’. It is an interactive atlas which maps the position of GDEs in Australia and classifies them into three categories: ecosystems that rely on the surface expression of groundwater, ecosystems that rely on the subsurface expression of groundwater and subterranean ecosystems (SKM 2012). The GDE Atlas is hosted on the Bureau of Meteorology website.

A strategic approach for identifying water-dependent ecosystems at risk from groundwater extraction was developed for the South Australian Department of Water by Harding and O’Connor (2012). This study undertook a review of water dependent ecosystems and threatening processes. Spatial mapping of wetlands was produced to identify and display those ecosystems vulnerable to groundwater resource development due to loss of groundwater quantity and discharge. It was determined that freshwater ecosystems were at highest risk due to the suitable quality of the groundwater for anthropogenic uses, resulting in high intensity groundwater extraction. This assessment will be used to inform conservation, management and administration (Harding and O’Connor 2012).

In their review on Mound spring aquifers in the Great Artesian Basin, Fensham and Fairfax (2003) determined the hydrology and location of these GDEs through desktop and field-based studies. The water quality and biology of the ecosystems was determined through field studies and declining quality and disappearing systems highlighted key threatening processes. These include reduced pressure due to aquifer draw-down, land-use changes resulting in modification of the systems and exotic plants and animals. This study outlined the need for identifying high value sites and undertaking appropriate conservation actions (Fensham and Fairfax 2003).

The NSW state of the catchments reports undertook comprehensive data collection and analysis to determine the quality and availability of groundwater and threatening processes to the ecosystems. The state was separated in to 13 study regions and a technical report was developed for each region. The assessment determined GDEs with a high priority for management and these were mapped for each region (eg: the Central West Study Region - Figure 18). Management activities were suggested for each region depending on the major threatening processes (NSW Government, Department of Environment and Heritage 2010).

Serov *et al.* (2012) developed risk assessment guidelines for GDEs for the NSW Office of Water. This framework outlined how to identify GDEs, determine their ecological value and vulnerability. A risk matrix was developed to determine the management actions and level of priority depending on the ecological value and vulnerability of the system to short, mid and log-term risks. The guidelines suggested that the development of conceptual models and regular monitoring of appropriate indicators (set out by the Water Management Act 2000) would reduce the risk of thresholds being exceeded and ensure that GDEs do not fall in to a higher risk category (Serov *et al.* 2012).

An NCCARF funded project - predicting water quality and ecological responses to a changing climate: informing adaptation initiatives - has been developed in the Murrumbidgee Catchment. This project will combine hydrological modelling with climate change projections to determine the potential temperature and drought frequency. Conceptual models have been developed to model the response of the ecosystem to projected water quality and quantity changes. The thresholds suitable to biota have been determined in a separate study in the Gouldburn Broken region and Molonglo River (salinity thresholds only). This project will be used to test adaptation initiatives and identify management priorities (Dyer 2012).

Southern Rural Water (2011) developed a groundwater atlas which describes the information currently available on the hydrology and uses of groundwater in the south-west region. As part of this review vegetation and high value wetlands which interact with the upper aquifers were identified. Southern Rural Water is in the process of developing groundwater atlases for the Gippsland, Port Phillip and Westernport regions (Southern Rural Water 2011).

All of these studies provide pertinent information to inform the current risk assessment framework being developed.

7. ADAPTATION

In their paper on ecological restoration and climate change adaptation Harris *et al.* (2003) discuss appropriate research techniques and restoration targets for dealing with a rapid shift in climate. Appropriate restoration needs to build resilience in the ecosystem rather than focus on historical targets. One means of adapting to the impacts of climate change is to determine the environmental requirements of ecosystems. This can be achieved by combining the current distribution of species, communities and ecosystems with key environmental variables to determine their environmental tolerances. Conducting controlled bioassay trials can also provide valuable information on the thresholds of individual taxa. Together with climate modelling these ecosystems thresholds can be used to project the likely spatial extent of ecosystems in the future (Harris *et al*. 2003). This predictive response will be particularly useful to freshwater ecosystems as suggested declining water levels and altered physiochemical properties make them highly vulnerable to climate change. Combined with appropriate management responses this method could be beneficial in maintaining ecological integrity and ecosystem functioning.

Adaptation is a response to climate change involving a change in processes, practices or structures (Smit *et al.* 2000 in Jenkins *et al.* 2011). Adaptation can be either autonomous (ecological and human responses) or planned. Planned adaptation facilitates the protection or buffering capacity of the ecosystem. Before adaptation actions can be incorporated into policy and management, the vulnerability of the ecosystem to climate change must be determined (Jenkins *et al.* 2011). This process can be assisted through the use of a Risk Assessment Framework (RAF), which identifies and evaluates the risk to human health and ecosystems, and determines suitable actions to reduce this risk (Jones 2001). Jones (2001) developed a RAF to determine the potential impacts climate change may have on vulnerable exposure units. An example based on irrigation demand was used to illustrate how predicting the possibility that critical system thresholds will be exceeded with altered climatic variables under climate change projections can determine the need for adaptation. RAF provides managers with a window for adaptation action, by providing an early warning signal that the system is at risk of unacceptable change (Jones 2001). During the risk assessment process it is important to have a strong focus on stakeholder involvement and communication between decision makers and scientists.

There are a variety of ways that a spatial and temporal risk assessment of projected groundwater levels to GDEs might be used to allow freshwater ecosystems with declining groundwater levels to adapt to climate change. These include planning water extraction volumes and locations, locating species and ecosystems at risk to minimise other anthropogenic stressors in these locations, prioritising restoration activities, identifying potential locations where groundwater rise might provide the opportunity for new wetlands to emerge and identifying and conserving refuges and connectivity corridors in the landscape.

While a spatial and temporal risk assessment framework would be valuable to climate change adaptation of GDEs, there are a number of research gaps that will need to be filled beforehand. Groundwater systems lag a long way behind surface water systems in terms of the knowledge available on them (Boulton 2005). On an international level, many of the GDEs have not yet been mapped spatially and there is limited information on the available water within the aquifers (Vorosmarty *et al*. 2005). This is despite an estimated one third of the world's water resources being supplied by groundwater (Vorosmarty *et al*. 2005). While the impact of climate change on groundwater resources has received increasing attention over the last ten years (Green *et al.* 2011), there is limited information on impacts on groundwater dependent ecosystems (GDEs) derived from a combination of climate change and management scenarios (Risbey *et al*., 2007, Candela *et al*. 2009). The current focus on GDE conservation is on the more immediate threats of land use changes, pollution and groundwater extraction (UNESCO 2008).

While recent research in Australia is beginning to fill these knowledge gaps (the Groundwater Dependent Atlas of Australia, Barron *et al.* 2012, McFarlane *et al.* 2012), our knowledge on the location, hydrogeology and ecosystem function of GDEs is still limited. Current research is addressing the likely impacts of climate change and adaptation cannot occur without this vital knowledge.

8. REFERENCES

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