



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

Development and case studies

Jane Chambers, Gaia Nugent, Bea Sommer, Peter Speldewinde, Simon Neville, Stephen Beatty, Stacey Chilcott, Stefan Eberhard, Nicola Mitchell, Frances D'Souza, Olga Barron, Don McFarlane, Michael Braimbridge, Belinda Robson, Paul Close, David Morgan, Adrian Pinder, Ray Froend, Pierre Horwitz, Barbara Cook and Peter Davies



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

DEVELOPMENT AND CASE STUDIES

**Jane Chambers¹, Gaia Nugent¹, Bea Sommer², Peter Speldewinde³,
Simon Neville⁴, Stephen Beatty⁵, Stacey Chilcott¹, Stefan Eberhard⁶,
Nicola Mitchell⁷, Frances D'Souza⁸, Olga Barron⁹, Don McFarlane⁹,
Michael Braimbridge⁸, Belinda Robson¹, Paul Close³, David Morgan⁵,
Adrian Pinder¹⁰, Ray Froend², Pierre Horwitz², Barbara Cook³ and Peter Davies³**

¹ Murdoch University, Environmental Science,

² Edith Cowan University Centre for Ecosystem Management,

³ Centre for Excellence in Natural Resource Management, The University of Western
Australia,

⁴ Ecotones and Associates,

⁵ Murdoch University, Freshwater Fish group,

⁶ Subterranean Ecology,

⁷ The University of Western Australia, School of Animal Biology,

⁸ Department of Water, Government of Western Australia,

⁹ CSIRO,

¹⁰ Department of Environment and Conservation, Government of Western Australia



Published by the National Climate Change Adaptation Research Facility 2013

ISBN: 978-1-925039-22-1 NCCARF Publication 51/13

Australian copyright law applies. For permission to reproduce any part of this document, please approach the authors.

Please cite this report as:

Chambers, J, Nugent, G, Sommer, B, Speldewinde, P, Neville, S, Beatty, S, Chilcott, S, Eberhard, S, Mitchell, N, D'Souza, F, Barron, O, McFarlane, D, Braimbridge, M, Robson, B, Close, P, Morgan, D, Pinder, A, Froend, R, Horwitz, P, Cook, B & Davies, P 2013, *Adapting to climate change: A risk assessment and decision making framework for managing groundwater dependent ecosystems with declining water levels. Development and case studies*, National Climate Change Adaptation Research Facility, Gold Coast, 100 pp.

Acknowledgement

This work was carried out with financial support from the Australian Government (through the Department of Climate Change and Energy Efficiency and the National Water Commission) and the National Climate Change Adaptation Research Facility. 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.

The authors thank everyone who contributed to this work and acknowledge: Melita Penniford (Department of Environment and Conservation, WA - DEC) for providing data on macroinvertebrates and assisting with the framework development; Steve Appleyard (DEC) for providing the cumulative rainfall departure analysis of the Leeuwin Ridge Caves and contributing to the threshold workshop. Shireen McGuinness (Edith Cowan University) for data analysis of the Gngangara Mound macroinvertebrates, littoral and sub-littoral data. Geoff Hodgson (CSIRO Land and Water) for the CSIRO Sustainable Yields GIS Data. Mirela Tulbure (The University of New South Wales) and Richard Silberstein (CSIRO Land and Water) for help with the development of the proposal. Gill White, Lynette Galvin, Kelli O'Neill, Helen Nice, and Emma van Looij (Department of Water, Government of Western Australia) for data collection for the freshwater fish technological study. Mike Bamford (Bamford Consulting Ecologists) for data analysis of the Amphibians on the Gngangara Mound. Peter Muirden (WA DEC) for providing the figure on the baseflow discharge to the Blackwood River and Cahit Yesertener (Department of Water, Government of Western Australia) for providing the cumulative rainfall departure analysis of the Gngangara Mound. Greg Keighery, Melita Penniford, Steve Appleyard and Tom Ryan (WA DEC) for their contribution to the threshold workshops. The authors would also like to thank the National Advisory Panel (Appendix 2) for reviewing the transferability of the framework both at the beginning and final stages of development; the staff at NCCARF, for both their assistance and support throughout the year, and anonymous reviewers who made improvements to the draft document. Two workshops were held during the development of this framework with end-users. The participants provided invaluable guidance and direction on the GDE framework and structure. The reference group members are sincerely thanked for their enthusiasm and input.

Disclaimer

The views expressed herein are not necessarily the views of the Commonwealth or NCCARF, and neither the Commonwealth nor NCCARF accept responsibility for information or advice contained herein.

Cover image/s © Vasse Wonnerup Estuary, Western Australia (Krzysztof Wienczugow, Marine and Freshwater Research Laboratory, Murdoch University).

Table of contents

ABSTRACT	1
EXECUTIVE SUMMARY	2
1. INTRODUCTION AND OBJECTIVES OF THE RESEARCH	5
2. RESEARCH ACTIVITIES AND METHODS	8
2.1 Engagement of Expertise to Optimize Research Outcomes.....	8
2.1.1 <i>Multidisciplinary Research Team</i>	8
2.1.2 <i>End User Input</i>	9
2.1.3 <i>Transferability</i>	9
2.2 Methods.....	9
2.3 Rationale for GDEs Used in this Project.....	10
2.4 Reporting.....	13
3. RESULTS AND OUTPUTS	17
3.1 Gngangara Groundwater System.....	17
3.1.1 <i>Identifying Management Issues</i>	17
3.1.2 <i>The Nature of the Ecosystem</i>	18
3.1.3 <i>Step 1: Identify the Hazard</i>	23
3.1.4 <i>Step 2: Determine Exposure and Vulnerability</i>	25
3.1.5 <i>Step 3: Assess Effects</i>	31
3.1.6 <i>Step 4: Characterise Risk</i>	36
3.1.7 <i>Step 5: Manage Risk</i>	51
3.2 Blackwood River Case Study.....	55
3.2.1 <i>Identifying management issues</i>	56
3.2.2 <i>The nature of the ecosystem</i>	56
3.2.3 <i>Step 1: Identify the Hazard</i>	59
3.2.4 <i>Step 2: Exposure and Vulnerability</i>	60
3.2.5 <i>Step 3: Assess Effects</i>	65
3.2.6 <i>Step 4: Characterise Risk</i>	66
3.2.7 <i>Step 5: Risk Management</i>	71
3.3 Leeuwin Naturaliste Ridge Caves.....	72
3.3.1 <i>Identifying Management Issues</i>	72
3.3.2 <i>The Nature of the Ecosystem</i>	72
3.3.3 <i>Step 1: Identify the hazard</i>	74
3.3.4 <i>Step 2: Exposure and Vulnerability</i>	76

3.3.5 Step 3: Assess Effects.....	78
3.3.6 Step 4: Characterise Risk.....	81
3.3.7 Step 5: Risk Management.....	82
4. DISCUSSION	84
5. GAPS AND FUTURE RESEARCH DIRECTIONS.....	86
REFERENCES	87
APPENDIX 1.....	96
End-user Workshop Summaries	96
5.1.1 February 22 nd 2012.....	96
5.1.2 November 21 st 2012	96
APPENDIX 2.....	97
Members of National Advisory Panel.....	97
Synthesis of National Advisory Panel Feedback (March 2012).....	97
APPENDIX 3.....	99
5.1.3 Data availability for the Gngangara Mound and Blackwood River Case Studies.....	99

List of figures

<i>Figure A: An example of a conceptual model developed for the risk assessment and decision making framework, which can be used for top down and bottom up management approaches...</i>	2
<i>Figure 1: The steps involved in the risk assessment framework</i>	9
<i>Figure 2: The south west corner of Western Australia with the location of the three study areas: wetlands of the Gnangara Mound, river base-flow system in the Blackwood River and the Leeuwin Naturaliste Ridge Caves.....</i>	13
<i>Figure 3: The Gnangara groundwater system in south Western Australia, with the hydrogeology, land-use types and groundwater abstraction illustrated</i>	18
<i>Figure 4: Cross section of the Gnangara Mound showing the effect of declining groundwater levels on interdunal wetlands.....</i>	19
<i>Figure 5: The Gnangara Study Area, with the main types of geomorphic wetlands indicated. ...</i>	20
<i>Figure 6: The ‘wetland management category’ groups in the Gnangara study area.</i>	21
<i>Figure 7: The ‘indicative conservation values’ developed for the Gnangara study area wetlands</i>	22
<i>Figure 8: An example of Cumulative Rainfall Departure analysis for Bore PM6 on the Gnangara Groundwater System.....</i>	24
<i>Figure 9: The magnitude of groundwater decline has been projected into 2030 under a range of climate change and land use scenarios for the Gnangara Mound, Western Australia</i>	26
<i>Figure 10: Simple Impact Assessment maps combining groundwater decline and surface proximity for the Gnangara Mound Study Area</i>	29
<i>Figure 11: The rate of groundwater decline (m/year) was projected into 2030 under a range of climate change and land use scenarios for the Gnangara Mound, Western Australia</i>	30
<i>Figure 12: db-RDA ordination of plant community data and hydrological and physico-chemical factors on the Gnangara Mound.</i>	32
<i>Figure 13: db-RDA biplot for spring macroinvertebrate data and hydrological and physico-chemical factors on the Gnangara Mound.</i>	33
<i>Figure 14: Conceptual model of the Gnangara Mound wetlands and the key ecosystem drivers to macroinvertebrates.</i>	34
<i>Figure 15: Life cycle of an amphibian species - Pseudophryne guentheri - with terrestrial embryos and aquatic larval life stages.....</i>	35
<i>Figure 16: The conceptual model developed for the survival of the crawling frog (Pseudophryne guentheri), based on the species life cycle.</i>	35
<i>Figure 17: Regression tree showing the changes in the proportions of hydrotypes in relation to groundwater decline</i>	37
<i>Figure 18: Bayesian Belief Network for assessing the risk of change to the state of littoral and supra-littoral vegetation on the Gnangara Mound posed by the magnitude and rate of drawdown, and taking into account the groundwater depth at commencement of monitoring....</i>	40
<i>Figure 19: Bayesian Belief Network for predicting the relative dominance of macroinvertebrate functional groups in response to hydrological, chemical and landscape drivers</i>	41
<i>Figure 20: The Macroinvertebrate and Plant community models were combined to provide a Bayesian Belief Network to assess risk for the whole ecosystem.</i>	43
<i>Figure 21: The BBN for terrestrial-breeding amphibians with aquatic larvae, illustrating the impact of groundwater between 1-2 m from the surface, a hydroperiod less than three months, and a decline in the autumn rainfall trigger for breeding</i>	44

Figure 22: The relationship between groundwater levels and probability of population persistence for the Crawling frog, illustrating the influence the hydroperiods and seasonal rainfall triggers for key events in the species' lifecycle.	45
Figure 23 – Gngangara vegetation change model reporting points.....	46
Figure 24 – The risk of vegetation change in the Gngangara Mound, Western Australia overlaid on groundwater change projected by the South West Sustainable Yields Project, Scenario C Mid (2030 projections using median (mid) Global Climate Model climate change scenarios and current development) (CSIRO 2009a).	48
Figure 25: Spatial mapping of the 'Gngangara macroinvertebrate model' -probability that the risk of overall wetland change is low, moderate or high for the wetland survey start year (Table 4) on the Gngangara Mound.....	50
Figure 26: Spatial mapping of the 'Gngangara macroinvertebrate model' -probability that the risk of overall wetland change is low, moderate or high for the wetland survey finish year (Table 4) on the Gngangara Mound.....	50
Figure 27: Combination of the wetland risk and wetland conservation value for the 'Gngangara Macroinvertebrate Model'.	51
Figure 28: Conceptual model of the linkages between variables that were used in the construction of the BBN for the prediction of risk to wetland function based on changes in macroinvertebrate functional groups, hydrology and water quality.....	54
Figure 29: Discharge into Hut Pool in the Blackwood River groundwater intrusion zone showing contributions form the Leederville and Yarragadee aquifers	57
Figure 30: Consecutive annual baseflow longitudinal discharge traces in the main channel of the Blackwood River in March (2003-2006)	58
Figure 31: The spatial boundaries of the Blackwood River study area.....	59
Figure 32: The magnitude of groundwater decline was projected into 2030 under a range of climate change and land use scenarios for the Blackwood River study area	62
Figure 33: The rate of groundwater decline (m/year) was projected into 2030 under a range of climate change and land use scenarios for the Blackwood River study area, Western Australia	64
Figure 34: Conceptual model of the Blackwood River GDE.....	65
Figure 35: The mean, median, and 25 th , 75 th , 5 th and 95 th of the key water quality parameters within the habitats occupied by <i>Galaxias occidentalis</i> and <i>Nannatherina balstoni</i> in south-western Australia	67
Figure 36: Complete BBN for Blackwood River incorporating all fish species and index of fish health.....	69
Figure 37 - Blackwood River fish health model reporting points.....	70
Figure 38: Composite fish health index - probability that the risk to the freshwater fish species <i>Galaxias occidentalis</i> and <i>Nannatherina balstoni</i> is low, moderate or high for the Blackwood River study area.....	71
Figure 39: Conceptual hydrogeologic profile of the Leeuwin Naturaliste Ridge Caves indicating eco-hydrologic "end point states" (blue): high GW levels versus low GW levels	73
Figure 40: Leeuwin Naturaliste Ridge study sites showing surveyed cave outlines (yellow) of Lake, Jewel, Easter and Labyrinth Cave Karst System (Leeuwin Naturaliste Ridge Caves).....	74
Figure 41: Jewel Cave provisional hydro-climatic states 1958-2010 adapted from Eberhard (2004) with additional data AMRTA. SWL (mAHD) refers to the subterranean water level in the cave as metres AHD (Australian Height Datum).	75
Figure 42: Jewel Cave Cumulative Rainfall Departure (CRD) displays measured groundwater levels in Jewel Cave compared with simulated levels according to climatic data (Data source courtesy, Steve Appleyard). The measured surface water level is based on data collected from 1958 onwards although the trendline commences ca. 1975 due to averaging effects.....	76

<i>Figure 43: Graphical depiction of water level changes in Jewel Cave over three time periods – see SD 5: Chilcott 2013 and SD 7: Neville 2013 for greater detail and other caves. Jewel Cave map courtesy of Peter Bell; adapted from Eberhard 2004.</i>	<i>77</i>
<i>Figure 44: Jewel Cave groundwater level, measured between 1958-2012.</i>	<i>78</i>
<i>Figure 45: Number of invertebrate taxa plotted against water depth for those years where both variables were recorded in Jewel-Easter Cave.</i>	<i>79</i>
<i>Figure 46: Basic conceptual model of the Leeuwin Naturaliste Ridge Cave system.</i>	<i>80</i>
<i>Figure 47: Major contributing factors to the health of GDE's</i>	<i>80</i>
<i>Figure 48: BBN for stygofauna for Leeuwin Naturaliste Ridge Caves.</i>	<i>81</i>
<i>Figure 49: Graphical depiction of declining groundwater levels in Jewel Cave over three time periods.</i>	<i>82</i>

List of tables

<i>Table 1: The areas considered in this study, their location and the type of Groundwater Dependent Ecosystems (GDE) and biota considered in this framework.</i>	<i>12</i>
<i>Table 2: The title, reference and contribution to the risk assessment framework for each supporting document.</i>	<i>15</i>
<i>Table 3: Ranges of threshold values for individual hydrological and chemical variables determined from the mean values of the splits in the MRTs created for the years 1996, 1998, 2006 and 2010 for macroinvertebrate family composition and functional group composition.</i>	<i>38</i>
<i>Table 4: Wetland sites and data collection in the Gnangara study area.</i>	<i>49</i>

ABSTRACT

The objective of this research was to develop and test a risk assessment and decision-making framework for managing groundwater dependent ecosystems (GDEs) with declining water levels due to climate change, anthropogenic extraction, land use and land management. The framework was developed by a multidisciplinary team of ecologists, modellers and hydrogeologists in south-western Australia, a biodiversity hotspot that has already suffered three decades of below average rainfall and consequently declining groundwater levels due to increased groundwater abstraction and land use change. This has provided a 'living experiment' providing validation of the framework against observed changes (not just modelled projections). The combination of this research together with input from a suite of end-users, other scientists and experts from across Australia has provided a robust and adaptable framework.

The report outlines how the framework was developed and tested on three different types of GDEs: surface expression of groundwater in 1) wetlands on the Gngangara Groundwater System in Perth and 2) the Blackwood River, and 3) the subterranean expression of groundwater in the Leeuwin Naturaliste Ridge Cave System. However, the framework could be adapted to any type of GDE or surface water system.

The framework integrates a standard risk assessment protocol enabling the approach to be easily transferred to sites within Australia and internationally. The framework is based around the construction of a conceptual model which identifies the interrelationships between climate, hydrology, water quality and/or biotic resources and the biota in an ecosystem. Before the framework is undertaken, management issues are identified and the site is characterised in terms of the type of GDE, its spatial extent, hydrogeology and assets within the site location. The framework then proceeds through five steps: identify the hazard, determine the exposure and vulnerability of the GDE, assess the effects of the hazard, characterise risk and then manage the risk. A suite of tools are provided by this framework for managing risk and climate change adaptation including: the identification of hazards and their cause(s), exposure and vulnerability of GDEs to hydrological stress, key drivers that cause ecosystem change, thresholds of tolerance of the biota for these key drivers, conceptual models, and risk assessment and decision-making tools in the form of Bayesian Belief networks and spatial models of risk.

EXECUTIVE SUMMARY

One of the key gaps in climate change adaptation research is translating relevant science into tools useful for management. The objective of this research was to develop and test a risk assessment and decision-making framework for managing groundwater dependent ecosystems (GDEs) with declining water levels due to climate change, anthropogenic extraction, land use and land management.

The framework integrates a standard risk assessment protocol enabling the approach to be easily transferred to sites within Australia and internationally. The framework is based around the construction of a conceptual model which identifies the interrelationships between climate, hydrology, water quality and/or biotic resources and the biota in an ecosystem. It is a problem solving framework that provides a transparent outline of the cause and effects of change to an ecosystem, highlighting key drivers that provide the focus for management and climate change adaptation. The framework is designed for declining water levels within the ecosystem, so it can readily be adapted to address surface water ecosystems by substituting, or adding, surface water inputs into the conceptual model (Figure A).

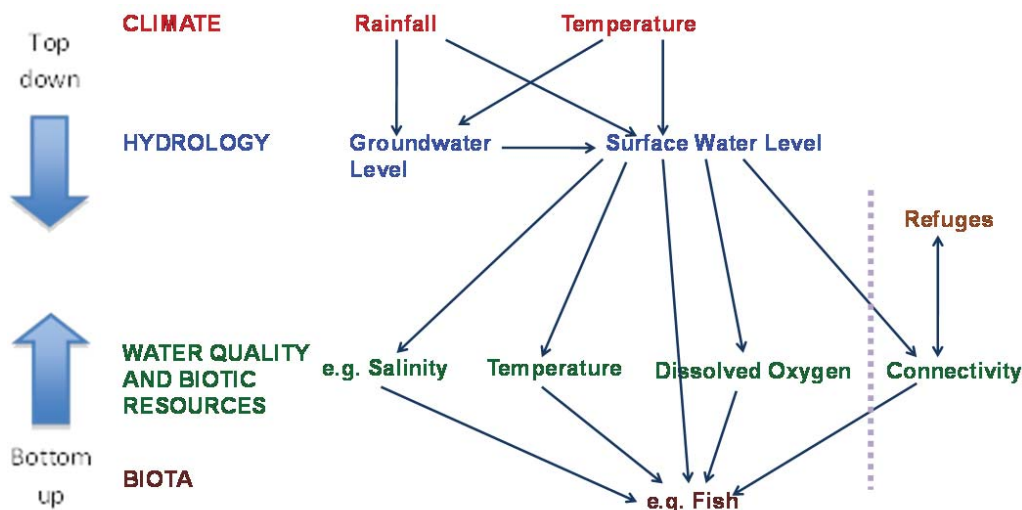


Figure A: An example of a conceptual model developed for the risk assessment and decision making framework, which can be used for top down and bottom up management approaches (see text).

Before the framework is undertaken, management issues are identified and the site is characterised in terms of the type of GDE, its spatial extent, hydrogeology and assets within the site location. The framework then proceeds through five steps: identify the hazard, determine the exposure and vulnerability of the GDE, assess the effects of the hazard, characterise risk and then manage the risk.

A suite of tools are provided by this framework for managing risk and climate change adaptation including: the identification of hazards and their cause(s), exposure and vulnerability of GDEs to hydrological stress, key drivers that cause ecosystem change, thresholds of tolerance of the biota for these key drivers, conceptual models, and risk assessment and decision-making tools in the form of Bayesian Belief networks and spatial models of risk.

To manage risk the use of a conceptual model provides the framework with a high degree of adaptability. The framework can be used to determine the effects of groundwater decline (whether through climate change, groundwater extraction, other cause, or a combination of these) on GDEs. This would be a top down approach using the conceptual model (Figure A) to test a number of scenarios of differing levels of groundwater decline. It can equally be used to determine the tolerance limits of GDEs or specific biota within them. This 'bottom up' approach would define the limits of unacceptable change that could inform management targets to conserve biota into the future.

The framework was developed by a multidisciplinary team of ecologists, modellers and hydrogeologists in south-western Australia, a biodiversity hotspot that has already suffered three decades of below average rainfall and consequently declining groundwater levels due to increased groundwater abstraction and land use change. This has provided a 'living experiment' providing validation of the framework against observed changes (not just modelled projections). The combination of this research together with input from a suite of end-users, other scientists and experts from across Australia has provided a robust and adaptable framework. The framework was developed in ecosystems that contain surface expression of groundwater in wetlands and rivers and the subterranean expression of groundwater in caves but could be adapted to any type of GDE or surface water system. This document showcases four case studies in these GDEs to provide first hand examples and variations of how the framework can be used.

The first two case studies investigated the risk to entire wetland ecosystems and to different guilds of amphibians on the Gngangara Groundwater System, a key water resource for the capital city of Perth, Western Australia. The large dataset and resources available for Gngangara Groundwater System enabled the exposure and vulnerability of the wetlands to be assessed through complex hydrological models and climate change projections. Long term data on hydrology, water quality and biota were analysed by a suite of multivariate statistical techniques to determine key drivers and thresholds for change in plant and macroinvertebrate communities. Incorporation of this data into conceptual models, Bayesian Belief Networks (BBNs) and spatial modelling illustrated the variation in level of risk to wetlands across the study area. The paucity of data available for amphibians led to a different approach where conceptual models and thresholds were developed through expert opinion and life cycle characteristics of the species. The key drivers identified can be used as management targets to determine where and when amphibians may be under stress under different climate change scenarios.

The third case study investigated the likelihood of survival of endemic freshwater fish species, two of which listed as threatened under the Federal EPBC Act (1999), in the groundwater intrusion zone of the Blackwood River.

These are threatened by salinity and lack of connectivity resulting from reduced groundwater intrusion into the river during the dry summer months in this region. A fish health model, using indicator species, was developed through the framework using relationships between a suite of hydrological and water quality parameters and presence/absence data at a large range of sites in south-western Australia. Spatial mapping of the fish model illustrated very high levels of risk to fish populations by 2030 but also key sites that could provide refuges with appropriate management.

The fourth case study investigated the potential extinction of Threatened Ecological Communities (EPBC Act 1999) of stygofauna in the Leeuwin Naturaliste Ridge Cave system due to declining groundwater levels. Cumulative rainfall departure analysis indicated that anthropogenic impacts rather than climate were likely to be the cause of the hazard, highlighting an intervention opportunity.

Exposure and vulnerability was determined through correlating groundwater depth to the extent and depth of flooding, and the distribution of communities in the cave. Multivariate statistics were used to identify key drivers of change, while expert opinion was used to develop conceptual models and thresholds that were incorporated into BBNs. Appropriate adaptation responses all focused on restoring groundwater input to the caves.

A major strength of the framework is its capacity to relate climate, hydrology and ecosystem response in a single tool. This innovative approach is presented in a user-friendly way for managers, enabling adaptation actions by one or more agencies, individually or in synergy to be assessed.

Guidelines for using the framework is outlined in detail in a companion document: *“Adapting to climate change: a risk assessment and decision making framework for managing groundwater dependent ecosystems with declining water levels: Guidelines for Use”*. Further detail can be obtained in a series of seven documents. To determine how these supporting documents fit into the overall study refer to Table 2 (page 26).

1. INTRODUCTION AND OBJECTIVES OF THE RESEARCH

One of the key gaps in climate change adaptation research is translating relevant science into tools useful for management. Communicating scientific knowledge so that management can encompass the complexity of ecological response to climate change without misinterpretation, and in a form appropriate to on-the-ground decision-making and action, is a challenge. The objective of this research was to develop and test a risk assessment and decision making framework for managing groundwater dependent ecosystems with declining water levels due to climate change, anthropogenic extraction, land use and land management. The framework, developed in south Western Australia, is transferable to locations both across Australia and internationally.

While reduced rainfall due to climate change is projected for many parts of the world, global climate change models consistently show high probabilities of declining rainfall in Mediterranean areas (south-western and South Australia, the south west United States of America, the Mediterranean, South Africa and southern South America) (Christensen *et al.* 2007). Climate change may also result in more frequent extreme events such as an increase in extreme drought (Jentsch and Beierkuhnlein 2008).

This can result in both direct and indirect impacts on groundwater availability. Reduced groundwater recharge occurs as a direct effect of reduced rainfall and runoff. Indirectly, reduced rainfall reduces surface water availability and, where groundwater is available, results in increased extraction (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).

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). 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 immediate threats such as land use changes, pollution and groundwater extraction. The Water Framework Directive (EU) has suggested that attention be given to how climate variability will affect GDEs (Kløve *et al.* 2011).

The international imperative to address how climate change will affect groundwater resources (for both human use and the ecosystems that depend on them) is the focus of this research. This project seeks to develop and test a risk assessment and decision-making framework in the Mediterranean climate of south-western Australia which has already undergone 30 years of drying that will be applicable internationally to Mediterranean semi-arid and arid climates where climate change projections indicate a very high probability of drying.

Australian climate change scenarios for 2030 project a warmer, drier climate with an increase in annual temperature from 0.7-1.2 °C, decreased precipitation by 2 to 5% in all regions of Australia (except the far north where precipitation is expected to increase) and higher potential evapotranspiration rates, particularly in winter (under the A1B scenario at the 50th percentile) (CSIRO 2007). Declining rainfall, together with increased temperature and evaporation, will result in a decline in runoff (Tomlinson and Boulton 2008). It is projected that the intensity and duration of hydrological drought in Australia is expected to increase by up to 20% (from the 1974 - 2003 baseline), with an increase of 40% in the south-west (Mpelasoka *et al.* 2008). This pattern is already evident in south-western Australia where the recently experienced dry period - partly as a consequence of anthropogenic-induced climate change (Australian State of the Environment Committee 2011) – has seen a decline in rainfall by 15% and a decline in runoff by 55% since 1975 (Bates *et al.* 2010).

The amount of groundwater recharge varies depending on rainfall, potential evapotranspiration, soil type, the amount and type of land cover, and the depth of the watertable (CSIRO 2009a). Reductions in runoff are disproportionately larger than the reduction in rainfall (Chiew and McMahon 2002, CSIRO 2009a). This is because runoff is more likely to occur when it falls on soil that is already wet (Roberts 2002). The corresponding reduction in recharge as a result of projected climate change is more complex (Hennessey *et al.*, 2007, Ali *et al.* 2012b) with a lag between surface water decline and groundwater decline (Bond *et al.* 2008; Tomlinson and Boulton 2008). This has highlighted the vulnerability of freshwater ecosystems to climate change and has impacted on GDEs by lowering the water table thus reducing groundwater surface expression and water levels.

In addition to the impacts of climate change, ecosystems are currently impacted by anthropogenic stressors including land use changes and land management. A symposium held in July 2010 by the NCCARF Freshwater Biodiversity Network identified the impacts, vulnerability and risk to different aquatic ecosystems in Western Australia. These findings, presented in “The Report Card of Climate Change and Western Australian Aquatic Ecosystems” (Kauhanen *et al.* 2011), highlighted groundwater dependent wetlands, cave and base-flow ecosystems to be the most vulnerable ecosystems in the South West due to climate change and anthropogenic stressors (land use changes, land management and groundwater abstraction). The severity of risk in this biodiversity hotspot necessitates action and highlighted the need for a risk assessment and decision support framework. This plan was strongly supported by State Government agencies responsible for water (and groundwater abstraction, Department of Water) and conservation (Department of Environment and Conservation) to guide their resource planning.

The framework is designed for ecosystems that contain surface expression of groundwater in wetlands and rivers and the subterranean expression of water in caves. However, because the framework is based on water levels in the ecosystem, this guide can be adapted for use in other freshwater ecosystems (such as ecosystems reliant on surface water) as well as GDE types not considered in this study (such as Mound Springs). It has been developed for sites in the south west of Western Australia, which have been under a drying climate for 30 years. These sites can be considered a ‘living experiment’, highlighting some of the future potential impacts that other regions in Australia and the world may incur. This enabled the framework to be validated against known change, rather than relying entirely on modelled future projections.

However, in common with other Mediterranean climates, global climate change models consistently agree that the future climate in south-western Australia will be drier and warmer (McFarlane *et al.* 2012), reducing the level of uncertainty of future risk.

This framework is designed to close the gap between research and decision-making. It follows the steps of a standard risk assessment framework (Assante-Duah 1998) to allow the level of risk to GDEs subject to declining water levels due to climate change or anthropogenic stressors, to be assessed.

2. RESEARCH ACTIVITIES AND METHODS

2.1 *Engagement of Expertise to Optimize Research Outcomes*

The multidisciplinary nature of the framework and the necessity to adapt it to a wide range of locations and uses required a diverse group of experts and end-users in the research team, together with advice from a diverse range of scientists and potential end users across Australia.

2.1.1 *Multidisciplinary Research Team*

The research team was created through the Western Australian NCCARF Freshwater Biodiversity network initiative to develop a “Report Card of Climate Change and Western Australian Aquatic Ecosystems” (Kauhanen *et al.* 2011).

This multidisciplinary team included experts in climate change, hydrology/hydrogeology, ecology, modelling and Geographical Information Systems (GIS), together with representatives of the two State government agencies wishing to use the framework. The research was contributed as a suite of coordinated tasks as follows:

Project coordination: Jane Chambers¹, Gaia Nugent¹

Literature review: Jane Chambers¹, Gaia Nugent¹, Peter Speldewinde²

Gnangara Mound ecological study on aquatic macroinvertebrates and littoral/supra-littoral vegetation: Bea Sommer³, Shireen McGuinness³, Ray Froend³ and Pierre Horwitz³

Gnangara Mound ecological study on Amphibians: Nicola Mitchell⁴, Bea Sommer³, Peter Speldewinde²

Blackwood River ecological study on freshwater fish: Stephen Beatty⁵, David Morgan⁵, James Keleher⁵, Alan Lymbery⁵, Paul Close², Peter Speldewinde², Timothy Storer⁶, Artemis Kitsios⁶

Leeuwin Naturaliste Ridge Caves ecological study on stygofauna: Stacey Chilcott¹, Stefan Eberhard⁷, Belinda Robson¹, Jane Chambers¹

Modeling risk using Bayesian Belief Networks: Peter Speldewinde²

Spatial risk and data analysis and presentation: Simon Neville⁸

Framework development: all of the authors and Frances D'Souza⁶, Olga Barron⁹, Mike Braimbridge⁶, Don McFarlane⁹, Adrian Pinder¹⁰, Melita Pennifold¹⁰, Barbara Cook², Peter Davies².

¹Environmental Science, Murdoch University, ²Centre for Excellence in Natural Resource Management, The University of Western Australia, ³Centre for Ecosystem Management, Edith Cowan University, ⁴School of Animal Biology, The University of Western Australia, ⁵ Freshwater Fish group, Murdoch University, ⁶Department of Water, Government of Western Australia, ⁷Subterranean Ecology, ⁸Ecotones and Associates, ⁹CSIRO, ¹⁰Department of Environment and Conservation, Government of Western Australia

2.1.2 End User Input

To ensure framework met the needs of end users, two of the key agencies (Department of Water and Department of Environment and Conservation) were on the research team. Seminars and end-user workshops were carried out in Western Australia during the course of the project and were attended by a wide variety of potential end users including various state government agencies, local councils, natural resource management groups, consultancies and universities (See Appendix 1: End-User report).

2.1.3 Transferability

To ensure transferability across Australia, a national advisory panel provided input at the outset of the project and will review this current draft framework before its final publication (See Appendix 2: NAP report). The principal investigator and research officer also attended a number of national and international conferences and workshops to discuss the work to obtain ideas from a wide variety of user groups. Workshops were held across Australia (Adelaide, Canberra and Brisbane) in March 2013 to refine the framework for wider use.

2.2 Methods

Three different groundwater dependent ecosystems (GDEs) (wetlands, a river and caves) were used to develop the case studies. The methodology for this was undertaken in two parts. In *Part 1) Identifying your Management Issue and the Nature of your Ecosystem* the management issues of concern were identified, the hydrogeology of the GDE was investigated, the spatial boundaries of the system were determined and the assets were prioritised within these boundaries.

In *Part 2) the Risk Assessment and Decision-Making Framework* (Figure 1) was developed by applying a standard risk assessment protocol (based on Assante-Duah 1998) to the hazard of declining groundwater levels. Each study area had differing amounts of data available, allowing the framework to be tested utilising a range of resources. For ease of understanding, specific details of the methodology used for each GDE test case is outlined as an independent case study in Section 3 – Results and Outputs.



Figure 1: The steps involved in the risk assessment framework (based on Assante-Duah 1998).

The steps involved in the risk assessment framework (based on Assante-Duah 1998) are:

Step 1: Identify the hazard - the framework is designed to manage the hazard of declining groundwater levels. This step identifies the primary and secondary hazards and the cause of declining groundwater levels.

Step 2: Determine exposure and vulnerability - the magnitude and rate of groundwater decline is determined spatially and temporally (historically, currently and projected to the future). The dynamics of hydrological change, such as changes in seasonality and/or the number and frequency of dry periods, are also considered.

Step 3: Assess effects - a conceptual model of ecosystem function is developed describing the cause and effect interrelationships between climate, hydrology, water quality, required biotic resources and biotic response. The key drivers that cause ecosystem change are identified.

Step 4: Characterise risk – A number of techniques (expert opinion, using presence/absence data and statistical analysis) are described to determine the tolerance limits or *thresholds* of the key drivers that drive change in the biota. The conceptual model, with key drivers now quantified by thresholds, are now incorporated into Bayesian Belief Networks (BBNs) that can be easily modified to show changes in probability of risk resulting from these interactions. The outputs of the BBNs or other risk models can then be mapped spatially using geographic information systems (GIS).

Step 5: Manage risk As described in the case studies, the framework provides a number of tools to support climate adaptation and management decisions at each step. This provides a high degree of adaptability depending on the resources and data availability of the user, from simple “what if?” scenarios to situations where there is significant data, access to modelling and GIS support. A major strength of the framework is its capacity to relate climate, hydrology and ecosystem response in a single tool.

From the experience gained in the three GDE examples a detailed, step-wise framework was developed through a series of workshops involving the whole research team. The framework incorporated input from the end-users on the team, from worksheets submitted at end-user workshops and from input from the national advisory panel to ensure steps were clear and that the output of the framework met end-user needs. This framework is outlined in a companion document “Guidelines for Use” (Chambers *et al.* 2013).

2.3 Rationale for GDEs Used in this Project

This framework is designed for groundwater dependent ecosystems (GDEs). In order to develop a robust, adaptable framework we wished to include a range of different GDE types in its development. The GDE Toolbox (Richardson *et al.* 2011) supported use of the typology created by Eamus *et al.* (2006). This includes three GDE types: aquifer and cave ecosystems (type 1), ecosystems dependent on the surface expression of groundwater (type 2) and ecosystems dependent on the subsurface presence of groundwater (type 3). A description of each of these types is provided below, taken directly from Richardson *et al.* (2011).

- **Aquifer and cave ecosystems (Type 1)** provide unique habitats for living organisms (e.g. stygofauna and troglofauna). These ecosystems typically include karst aquifer systems, fractured rock and saturated (consolidated and

unconsolidated) sedimentary environments. The hyporheic zones of rivers, floodplains and coastal environments are also included in Type 1.

- **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.
- **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 (phreatophytic vegetation). In these cases 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.

Of these our study sites included one cave system (Type 1). This was a subsurface expression of groundwater in caves in a karst formation in the Tamala limestone of the Leeuwin Naturaliste Ridge. There were also two examples of surface expression of groundwater (Type 2). These included groundwater dependent wetlands, which were surface expressions of an unconfined superficial aquifer of the Gngangara Mound. These were primarily found in the depressions of the dunal swales of a sandy Swan Coastal Plain. The second example was a small section of the Blackwood River, which receives summer baseflow due to groundwater intrusion from the Yarragadee Aquifer. While this study includes a small subset the GDE types listed by Richardson *et al.* 2011, the framework should be adaptable to all GDEs.

The case studies were also chosen to include a range of freshwater biota (Table 1, Figure 2), various types of anthropogenic stressors resulting in groundwater decline and differing availability of data. The case studies incorporated:

- **Anthropogenic stressors:** including areas currently used for groundwater extraction (the Gngangara Groundwater System) or potential locations for more extensive future groundwater extraction (Yarragadee Aquifer in the Blackwood River catchment). It was likely the groundwater levels in the Leeuwin Naturaliste Ridge cave system were affected by land use change.
- **Ecosystems affected by groundwater decline:** ecosystems at all locations showed evidence of significant change in response to reduced groundwater levels and one was a listed threatened ecological community (root mat communities of the Leeuwin Naturaliste Ridge Caves are listed under the EPBC Act 1999).
- **GDEs in hydrologically complex groundwater systems,** making them a useful area for testing and developing the framework.
- **Differing availability of data on hydrology, hydrogeology, water quality and biota.** The data available ranged from data-rich sites (macroinvertebrates and vegetation on the Gngangara Mound) to sites where some aspects had good data (Blackwood – hydrology, fish; Leeuwin Naturaliste Ridge Caves – stygofauna; Amphibians – hydrological, physiochemical parameters) but were mostly data poor. The data available to this project is listed in Appendix 3.

In addition, all sites were in a region where climate change has been experienced for the previous 30 years. The area can be considered a 'living experiment', highlighting some of the future potential impacts that other regions in Australia or internationally, may incur. This enabled the framework to be validated against known change, rather than relying entirely on modelled future projections.

Table 1: The areas considered in this study, their location and the type of Groundwater Dependent Ecosystems (GDE) and biota considered in this framework.

Study Area	GDE	Biota	Location
Gnangara Mound, Perth Metropolitan region	Wetlands	Macroinvertebrates, littoral and sub-littoral vegetation	south-west corner: 32.056149°S, 115.722548°E; to north-east corner: 31.299071°S, 115.8122°E
Gnangara Mound, Perth Metropolitan region	Wetlands	Amphibians	south-west corner: 32.056149°S, 115.722548°E; to north-east corner: 31.299071°S, 115.8122°E
Blackwood River that receives baseflow from the Yarragadee Aquifer	Base-flow river	Freshwater fish	South west corner: 34.1081°S, 115.5661°E; north-east corner: 34.0421°S, 115.6025°E
Leeuwin Naturaliste Ridge Caves	Cave	Stygofauna	33°31'S, 114°59'E; 34°23'S, 115°15'E

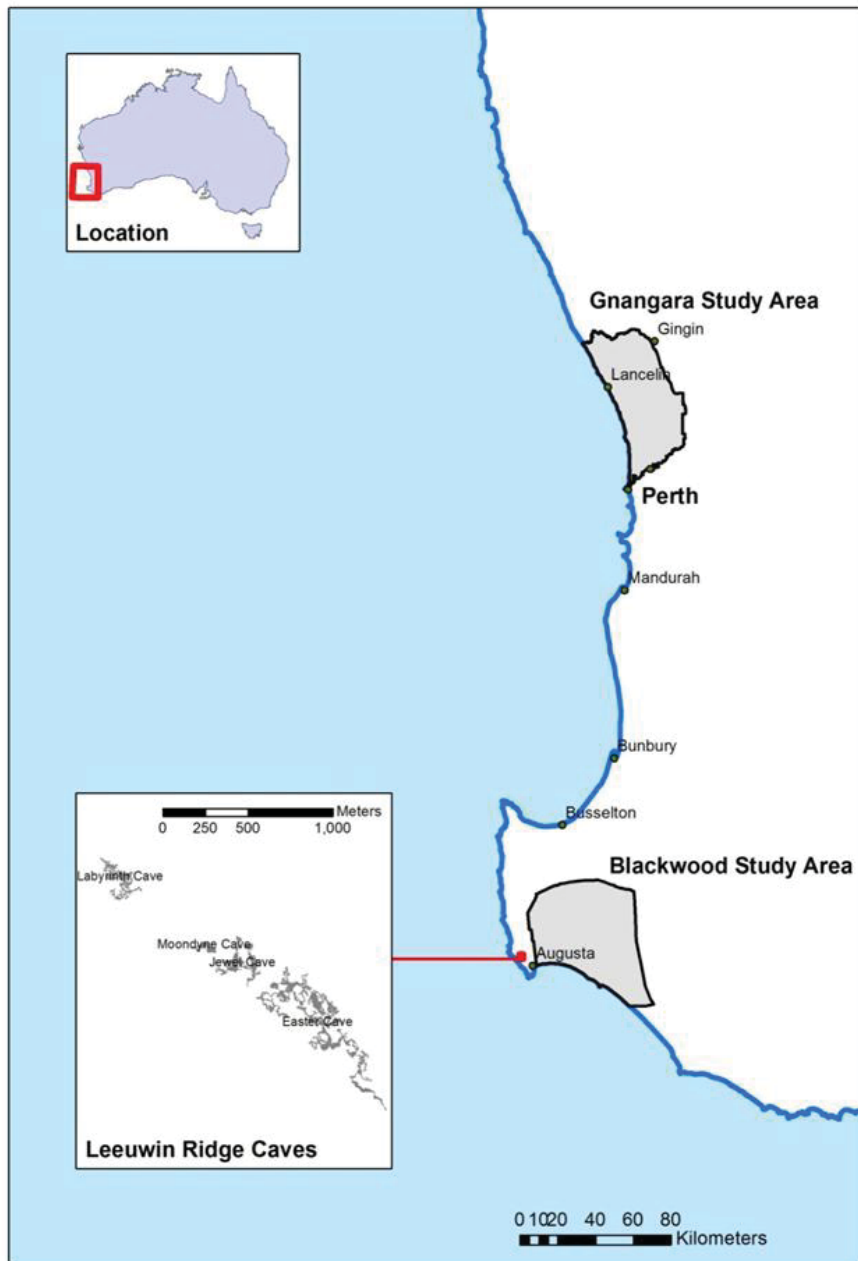


Figure 2: The south west corner of Western Australia with the location of the three study areas: wetlands of the Gnangara Mound, river base-flow system in the Blackwood River and the Leeuwin Naturaliste Ridge Caves.

2.4 Reporting

The framework and its development are captured in two companion documents: this document “*Adapting to climate change: a risk assessment and decision making framework for managing groundwater dependent ecosystems with declining water levels - Development and Case Studies*” and a user manual “*Adapting to climate change: a risk assessment and decision making framework for managing groundwater dependent ecosystems with declining water levels - Guidelines for Use*” (Chambers *et al.* 2013).

The 'Guidelines for Use' document explains how to use the framework, showcasing the three GDE case studies throughout to provide first hand examples and variations of how the framework can be used. It is supported by this document outlining how the framework was developed and tested in these three types of GDEs. Further detail can be obtained in a series of seven technical reports outlining the research behind this Development and Case Studies report.

Reference to the technical reports is made throughout this document and is cited by the relevant Supporting Document (SD) number and author date throughout. The title, authors, study area, biota considered and how each component contributed to this risk assessment framework are outlined in Table 2.

Table 2: The title, reference and contribution to the risk assessment framework for each supporting document.

SUPPORTING DOCUMENT (SD)	TITLE	STUDY AREA	BIOTA	CONTRIBUTION TO THE RISK ASSESSMENT FRAMEWORK	FULL CITATION
SD1	Literature Review	Australia	Project overview	Background information for Part 1 & Part 2.	Nugent, Chambers and Speldewinde 2013
SD2	Assessing risks to groundwater dependent wetland ecosystems in a drying climate: an approach to facilitate adaptation to climate change	Gnangara Mound	Macroinvertebrates and littoral and sub-littoral vegetation	Part 2: Steps 3 (Assess Effects), 4.1 (Identify Thresholds) and 5 (Manage Risk) for the Gnangara Mound macroinvertebrates and vegetation case study.	Sommer, McGuinness, Froend, and Horwitz 2013
SD3	Identifying thresholds for responses of amphibians to groundwater and rainfall decline	Gnangara Mound	Amphibians	Part 2: Steps 3 (Assess Effects) and 4.1 (Identify Thresholds) for the Gnangara Mound Amphibian case study.	Mitchell, Sommer and Speldewinde 2013
SD4	Environmental variables in the habitats of south-western Australian freshwater fishes: an approach for setting threshold indicator values	Yarragadee intrusion of the Blackwood River	Freshwater Fish	Part 2: Steps 3 (Assess Effects) and 4.1 (Identify Thresholds) for the Blackwood River freshwater fish case study.	Beatty, Morgan, Keleher, Lymbery, Close, Speldewinde, Storer, and Kitsios 2013
SD5	The impact of declining groundwater levels on Stygofauna communities in the Leeuwin Naturaliste Ridge Cave systems, Western Australia.	Leeuwin Naturaliste Ridge Caves	Stygofauna	Part 2: Steps 1 (Identify the Hazard), 2, 3 (Assess Effects), 4.1 (Characterise Risk) and 5 (Manage Risk) for the Leeuwin Naturaliste Ridge Caves case study.	Chilcott 2013
SD6	Development of Bayesian Belief Networks for modelling the impacts of falling groundwater due to climate	Gnangara Mound, Yarragadee intrusion of the	Macroinvertebrates and littoral and sub-littoral vegetation,	Part 2: Step 4.2 (Determine probability of risk) for each case study.	Speldewinde 2013

SUPPORTING DOCUMENT (SD)	TITLE	STUDY AREA	BIOTA	CONTRIBUTION TO THE RISK ASSESSMENT FRAMEWORK	FULL CITATION
	change on groundwater dependent ecosystems	Blackwood River and the Leeuwin Naturaliste Ridge Caves	Amphibians, freshwater fish and Stygofauna		
SD7	Spatially representing the impacts of falling groundwater due to climate change on groundwater dependent ecosystems	Gnangara Mound, Yarragadee intrusion of the Blackwood River and the Leeuwin Naturaliste Ridge Caves	Macroinvertebrates and littoral and sub-littoral vegetation, Amphibians, freshwater fish and Stygofauna	Part 1: Identifying assets, Part 2: Step 2 (Determine Exposure and Vulnerability), Step 4.3 (Spatially mapping risk) and Step 5 Manage Risk - Spatial mapping of consequence.	Neville 2013

3. RESULTS AND OUTPUTS

This section runs through Part 1) Identifying your Management Issue and the Nature of your Ecosystem and Part 2) the Risk Assessment and Decision-Making Framework for each case study. Reference to supporting documents is made where applicable.

3.1 Gngangara Groundwater System

(Key supporting documents: Case Study 1 - SD2: Sommer *et al.* (2013), Case Study 2 - SD3: Mitchell *et al.* (2013), Determine probability of risk - SD6: Speldewinde (2013), and Spatial modelling - SD7: Neville (2013)).

Due to the importance of the Gngangara Groundwater System for human use and conservation, it has been studied intensively and there is a large, long-term dataset available on the hydrogeology, hydrology (1975-present), water quality and biota (1996-present) (Appendix 3). Regional groundwater models 'PRAMS' (Perth Regional Aquifer Modeling System) were available and evaluate groundwater level declines (Vogwill, 2004; Xu, 2008), while climate change predictions were available from the South West Sustainable Yields project (CSIRO 2009a,b). The availability of data and the need for this information made the Gngangara an ideal site for this project.

PART 1: IDENTIFYING MANAGEMENT ISSUES AND THE NATURE OF THE ECOSYSTEM

3.1.1 Identifying Management Issues

Two separate management issues were trialled on the Gngangara Mound:

CASE STUDY 1: *Identifying the effect of declining groundwater levels due to climate change and extraction for human use on groundwater dependent wetlands in an urban landscape.*

The Gngangara Groundwater System is an important water resource for the Perth Metropolitan region. There has been a progressive decline in annual rainfall in the Perth region since the mid 1970s, and most notably since the early 1990s (Australian Bureau of Meteorology 2012). As a consequence, there has been a greater reliance on groundwater abstraction for anthropogenic use. This trend is not unique to the region and has been witnessed in other parts of the world, where the reliability of groundwater and the reduced availability of surface water have lead to increased extraction (Gemitzi and Stefanopolous 2011, McFarlane *et al.* 2012). As a result of reduced recharge and groundwater abstraction, the groundwater levels in the superficial aquifer have been gradually declining over the past 40 years. Consequently, the water levels in the GDEs have also declined (Yesertener 2008). Alongside declining rainfall, yearly average temperatures in the Perth region have steadily increased from 23.1°C (1900 to 1950) to 24.5°C (2000 to 2010) (Australian Bureau of Meteorology 2012). Yearly average temperatures during the last decade were thus 1.0°C above the long-term average (1900-2010) of 23.5°C. These rising temperatures exacerbated the effects of recent extreme dry years on the GDEs on the Swan Coastal Plain. The framework was developed and trialled in this location to predict the likely outcome of groundwater decline on the groundwater dependent wetlands into the future, based on future projections of climate change and groundwater extraction scenarios.

CASE STUDY 2: Predicting the response of nine amphibian species, with different life histories, to declining groundwater and rainfall.

Amphibians are key indicators of wetland health and under conditions of hydrological change it is important to be able to predict changes in their diversity and abundance (Bradford 2002). The framework was trialled to assess the effects of groundwater decline on three reproductive guilds: amphibians that breed in water, amphibians that breed in terrestrial nests that are later flooded, and one entirely terrestrial-breeding species. Seasonal rainfall events play an important part in triggering reproduction (Walls *et al.* 2013), so this case study sought to identify the combined effects of declining rainfall and groundwater on the different amphibian life histories and project the continued survival of nine species with climate change.

While there is significant hydrological and physiochemical data available for the Gngangara Groundwater System, there was relatively little data available for amphibians so this case study provided a low-data example with which to develop and test the framework.

3.1.2 The Nature of the Ecosystem

The Gngangara Groundwater System is a complex, multi-layered system, which consists of three aquifers (Figure 3). The superficial aquifer is the unconfined surface layer and is an important resource to groundwater dependent wetlands and their associated biota. Underlying the superficial aquifer are two confined aquifers - the Leederville and Yarragadee. The northern region of the Gngangara Groundwater System is the main source of groundwater recharge to these confined aquifers because there is no confining bed between the superficial aquifer and the underlying aquifers (Gngangara Sustainability Strategy 2007).

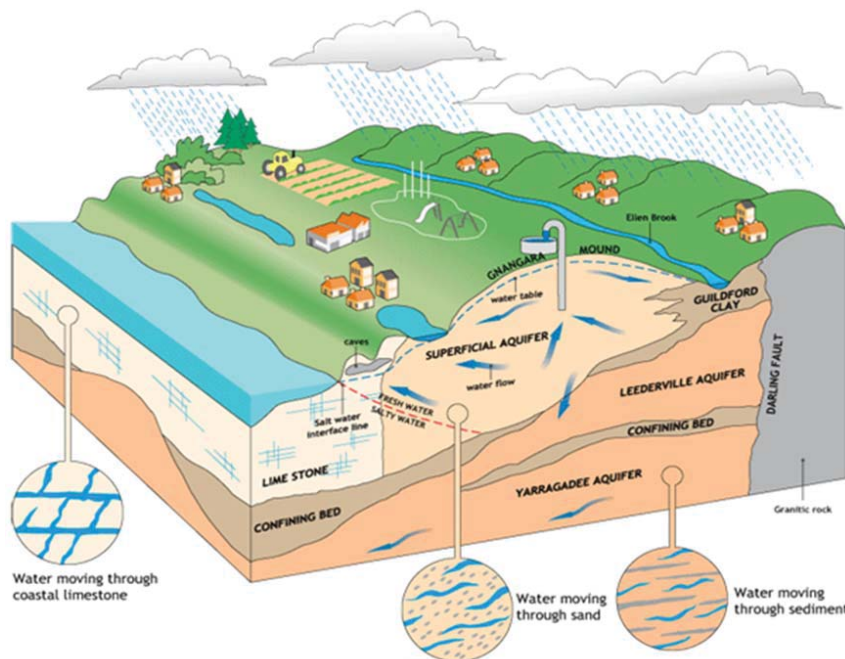


Figure 3: The Gngangara groundwater system in south Western Australia, with the hydrogeology, land-use types and groundwater abstraction illustrated. Source: Gngangara Sustainability Strategy managing land and groundwater for the future (2007).

The groundwater dependent wetlands in this region are surface expressions of the unconfined superficial aquifer - the Gngangara Mound – and are primarily found in the depressions of the dunal swales. With declining groundwater levels, the inter-dunal wetlands progressively dry over time (Figure 4). Surface water expression in the wetland is determined by how far above the groundwater table the GDE is positioned.

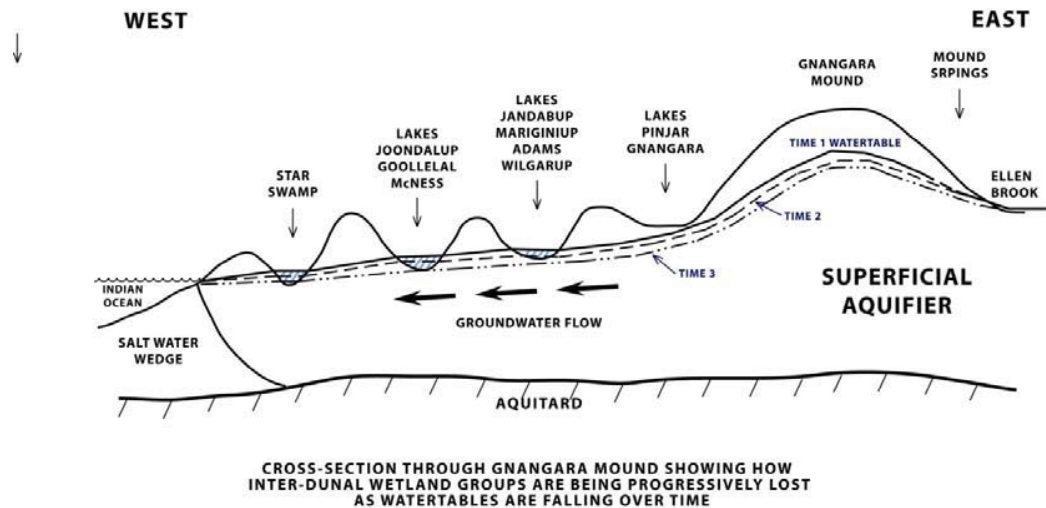


Figure 4: Cross section of the Gngangara Mound showing the effect of declining groundwater levels on interdunal wetlands (Source: Don Macfarlane CSIRO). Times 1, 2 and 3 are arbitrary time intervals over a period of drying climate.

Spatial Boundaries

The Gngangara Groundwater System (GGS) lies on the Swan Coastal Plain (SCP) and forms 2200 km² of the northern region of the Perth metropolitan area (south-west corner: 32.056149°S, 115.722548°E; to north-east corner: 31.299071°S, 115.8122°E) (Figure 5).

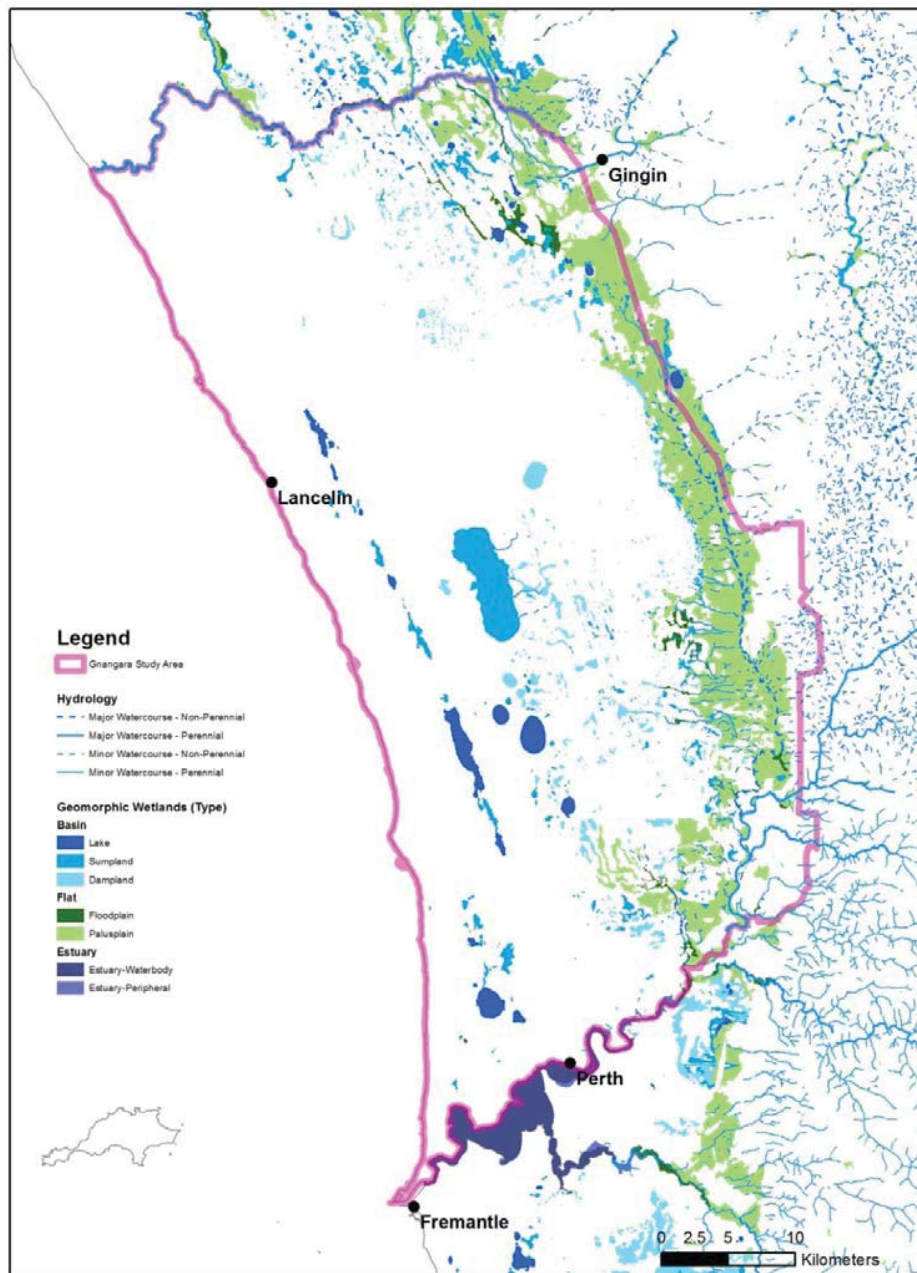


Figure 5: The Gngara Study Area, with the main types of geomorphic wetlands indicated.

Assets

In the Gngara study area, wetland asset identification is available in the Geomorphic Wetlands dataset (Hill *et al.* 1996), which has a 'wetland management category' - a form of valuation aimed at management. It identifies three management priorities:

- Conservation (to preserve wetland natural attributes and functions)

- Resource enhancement (to restore wetlands through maintenance and enhancement of natural attributes and functions); and
- Multiple use (to use, develop and manage in the context of water, town and environmental planning).

These wetland management categories are illustrated for the Gngangara study area in Figure 6.

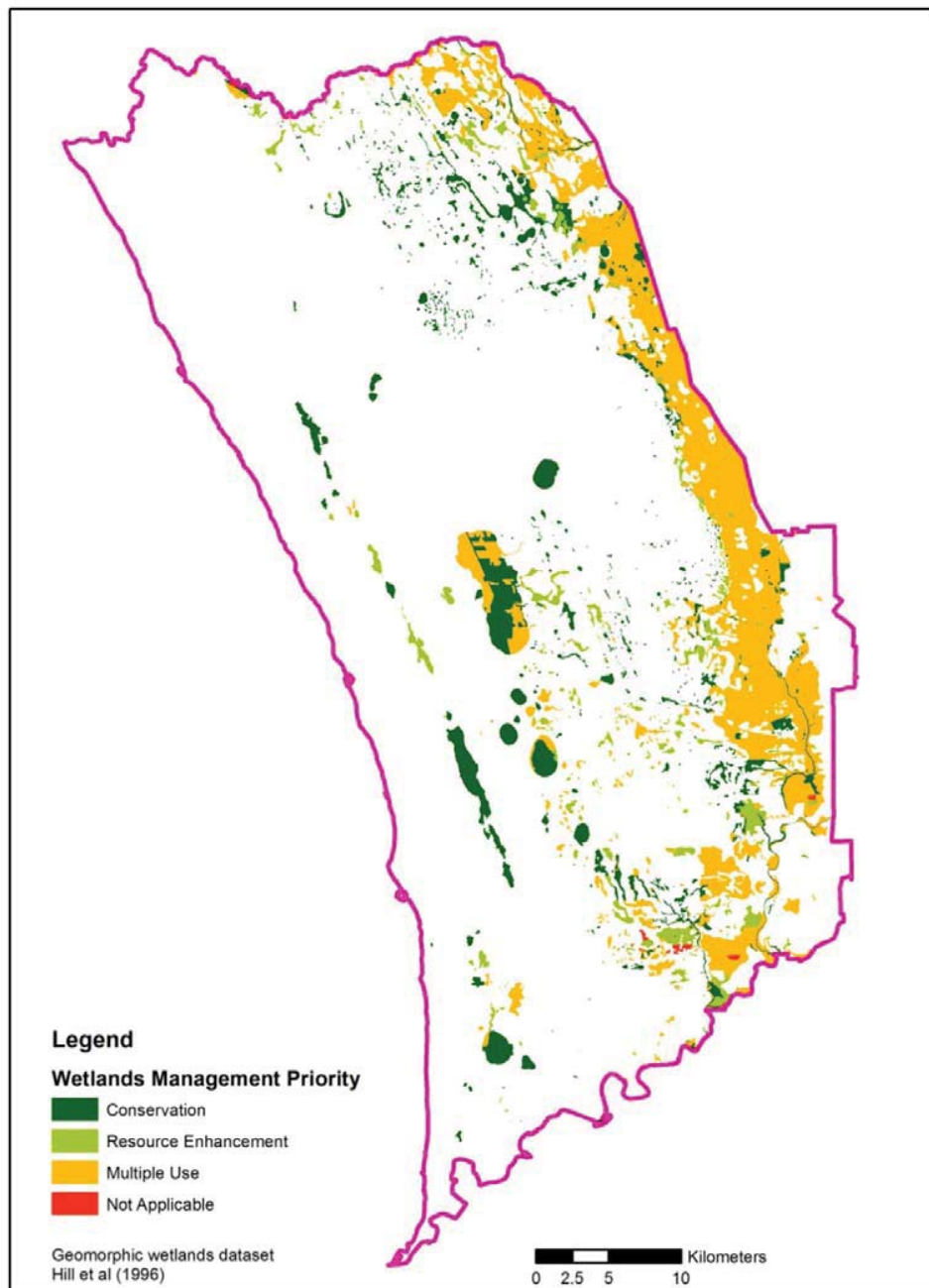


Figure 6: The 'wetland management category' groups (based on Hills *et al.* (1996)) in the Gngangara study area.

This data was further refined using wetland characteristic criteria weighted as follows:

- RARITY – Wetland Proportion of Consanguineous Suite – 1 (15%)
- RARITY - Consanguineous Suite Area (total) – 0.5 (8%)
- RARITY - Wetland - Proportion of Class – 0.5 (8%)
- DIVERSITY - Wetland Class – 1.5 (23%)
- NATURALNESS/AREA - Wetland Area (by class) – 1 (15%)
- NATURALNESS - Evaluation (Wetland Zoning) – 2 (31%)

The final conservation values, referred to as 'indicative conservation values', of wetlands on the Gngangara Mound are demonstrated in Figure 7.

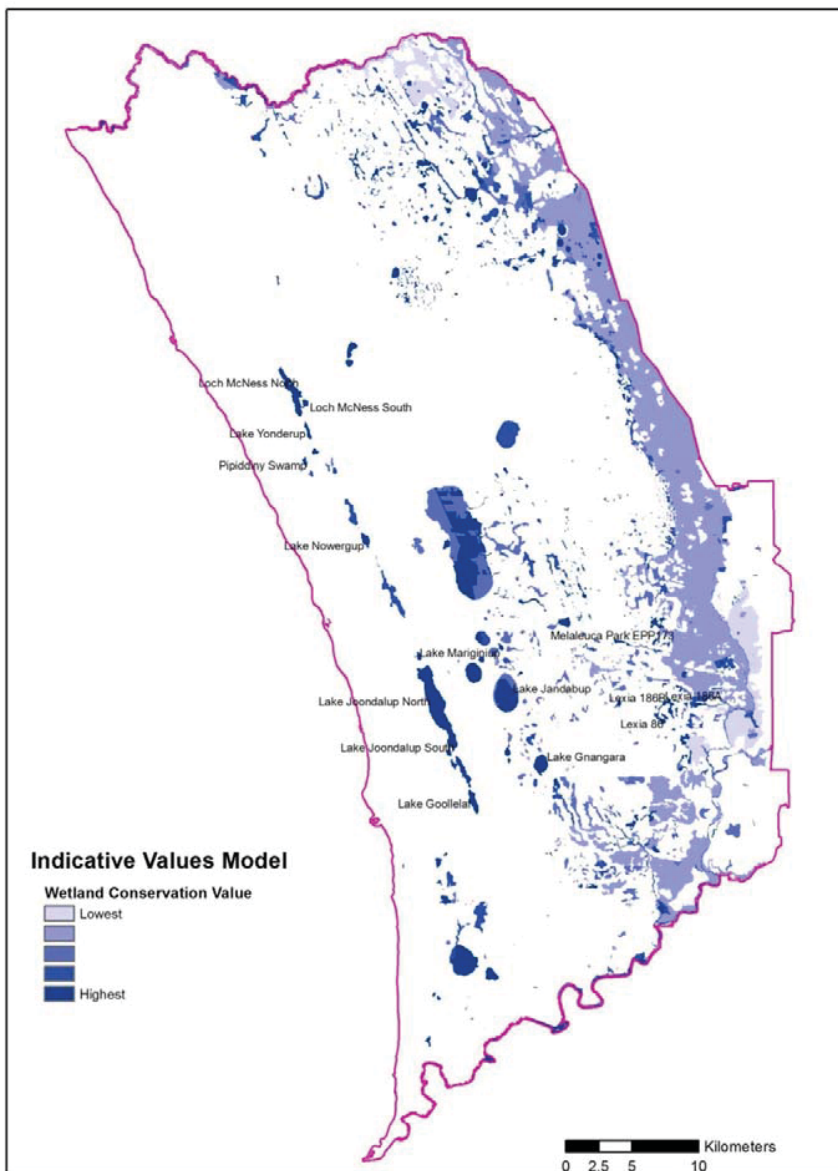


Figure 7: The 'indicative conservation values' developed for the Gngangara study area wetlands.

PART 2: THE RISK ASSESSMENT AND DECISION-MAKING FRAMEWORK

3.1.3 Step 1: Identify the Hazard

Step 1.1: Identify the hazard

Primary hazard: The primary hazard on the Gngangara Mound is groundwater decline due to a reduction in rainfall, groundwater extraction for human use and land use changes such as pine plantations (Yesertener 2008).

Secondary hazards: Declining water levels within wetlands are likely to result in decreased flushing and evapoconcentration of nutrients, salts and pollutants. Increased temperature (both average and diurnal maxima and ranges), resulting from shallower water depths, can lead to death of aquatic organisms (Davies 2010). Reduced groundwater input, 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 puts fauna such as amphibians at particular risk, because they need moist soil in which to lay their eggs (Eads *et al.* 2012). This can also result in compacting of clays and formation of hydrophobic sands that do not regain original soil structure on rewetting. Complete drying of the wetlands can expose pyritic sediments to oxygen, resulting in the production of actual acid sulfate soils. This has already occurred in wetlands in the Gngangara study area (eg: Lake Jandabup) where a reduction in pH from 6-7 to 4-5 was associated with a shift in community structure due to a loss of biota sensitive to acidic conditions (Sommer and Horwitz 2001).

Step 1.2: Define temporal boundaries

The study was constrained to three discrete time periods due to the availability of data (Appendix 3, Tables 1, 2 and 3) and the projections by the South West Sustainable Yields Project (CSIRO 2009a). Dates for hydrological data (below) were slightly different for ecological data (where the earliest data was from 1996).

1. Historical (1975-2007)
2. Recent climate (1997-2007)
3. Projections to 2030 - a range of scenarios

The projections to 2030 included a number of climate change and development scenarios:

- Scenario A - Historical climate: assumed the climate of 1975-2007 continued until 2030
- Scenario B - Recent climate: assumed the climate of 1997-2007 continued until 2030
- Scenario C: Future 1 - 2030 climate change and current development (Mid, Wet and Dry Scenarios)
- Scenario D: Future 2 - 2030 climate change and future development

The Future Climate (Scenario C, Scenario D) used 15 Global Climate Models (GCMs) to modify the Historical Climate to produce wet, median, dry and future development 2030 climate scenarios. More details on the above five climate scenarios and the impact on surface water resources, groundwater dependent ecosystems and divertible yields can be found in CSIRO (2009a and b), Silberstein *et al.* (2012), Ali *et al.* (2012a), Barron *et al.* (2012) and McFarlane *et al.* (2012).

Step 1.3: Determine the cause of the hazard

To determine the main causes of groundwater decline, groundwater hydrographs were compared with Cumulative Deviation From Mean rainfall (CDFM). An example of this analysis for one bore on the Gngangara Groundwater System (PM6) suggests that 2.7 metres of groundwater decline is due to reduced rainfall and 1.8 metres is due to groundwater abstraction (Figure 8). Through CDFM analysis three main causes of groundwater decline were identified: reduced rainfall, groundwater extraction and pine plantations (Yesertener 2008).

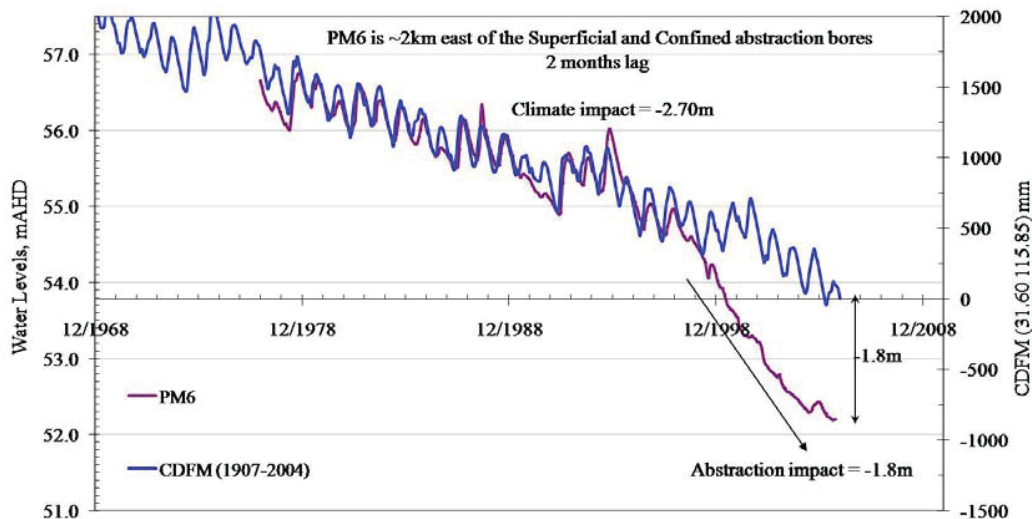


Figure 8: An example of Cumulative Rainfall Departure analysis for Bore PM6 on the Gngangara Groundwater System (Yesertener 2008) (Image courtesy of Cahit Yesertener, Government of Western Australia Department of Water).

Reduced rainfall: Two periods of rainfall were identified, a wet period from 1915-1968 and a dry period from 1969-2001. A 10%-16% decline in rainfall and consequently recharge was observed during the dry period. This was identified as the main cause of groundwater decline in the Gngangara Mound and resulted in reduced groundwater levels of up to 4m from 1979 – 2005 (Yesertener 2008).

Anthropogenic abstraction: The Gngangara groundwater system contributes a significant water resource to the Perth Metropolitan water supply. Declining groundwater levels within the Gngangara study area were due to extraction of groundwater from the Gngangara Mound superficial aquifer, as well as abstraction from the underlying Leederville and Yarragadee aquifers. The cumulative impact of extraction from the superficial aquifer and the confined aquifer was determined to be a groundwater decline of up to 3m from 1997-2005 and in some areas contributed to ~60% of the overall groundwater decline. The consequence of withdrawing groundwater can result in declining groundwater levels as far as 6kms from the abstraction point (Yesertener 2008).

Land-use and management changes: The primary land use change that impacted the Gngangara Mound was the planting of pines for forestry products. Initially pine plantations resulted in a rise in groundwater levels due to the clearing of native vegetation. Once the plantations began maturing they resulted in reduced groundwater recharge through evapotranspiration and interception of rainfall. The impact that they had on groundwater levels depended on the density of the stands. In areas managed at high density there was a moderate to high impact on groundwater levels, resulting in

declines of 3.5m from 1979-2005. In areas where there was low density there was no impact on groundwater levels when compared with the native vegetation control area. Removal of pine plantations was shown to cause up to a 2m rise in groundwater levels in some areas (Yesertener 2008).

3.1.4 Step 2: Determine Exposure and Vulnerability

Step 2.1: Determine the extent of spatial and temporal change

Information on groundwater level changes was obtained from the PRAMS model (Department of Water) and the South West Sustainable Yields project (CSIRO 2009a). It was mapped using Geographical Information Systems (GIS) to give a spatial distribution. The scenarios mapped were based on discrete time periods from the temporal boundaries identified in Step 1.3 above. Due to inherent errors involved in modelling of groundwater levels using PRAMS and its combination with climate change projections, this resulted in errors in projected groundwater levels of up to 3m. So, while these projected groundwater levels were used, they were taken to represent possible scenarios rather than accurate estimations of future levels. Decision-making would best be served by capturing the direction (decline or rise) of groundwater level change, the magnitude and rate of groundwater level change and/or the likelihood of groundwater being above a level where interaction with the GDE was possible, as the most useful indicators of hydrological change. Examples and use of different techniques to assess the extent of spatial and temporal change are described below.

Spatial and temporal change example 1: Magnitude of groundwater decline

The magnitude of groundwater change for 2030 in the Gngangara Mound is shown for each of the six climate change and land-use scenarios (Figure 9). Whilst these maps were produced using the raw output from the PRAMS models (rather than classified), they are identical to the South West Sustainable Yields mapping (CSIRO 2009a). Projected groundwater declines vary greatly across the Gngangara Mound under all scenarios. Clearly the CDry Scenario indicates the greatest groundwater decline as a reduced rainfall results in reduced recharge, but reductions are projected under A, B, CMid and D Scenarios. Note that many areas with projected groundwater decline do not contain GDEs due to the groundwater occurring at depths beyond the reach of surface ecosystems.

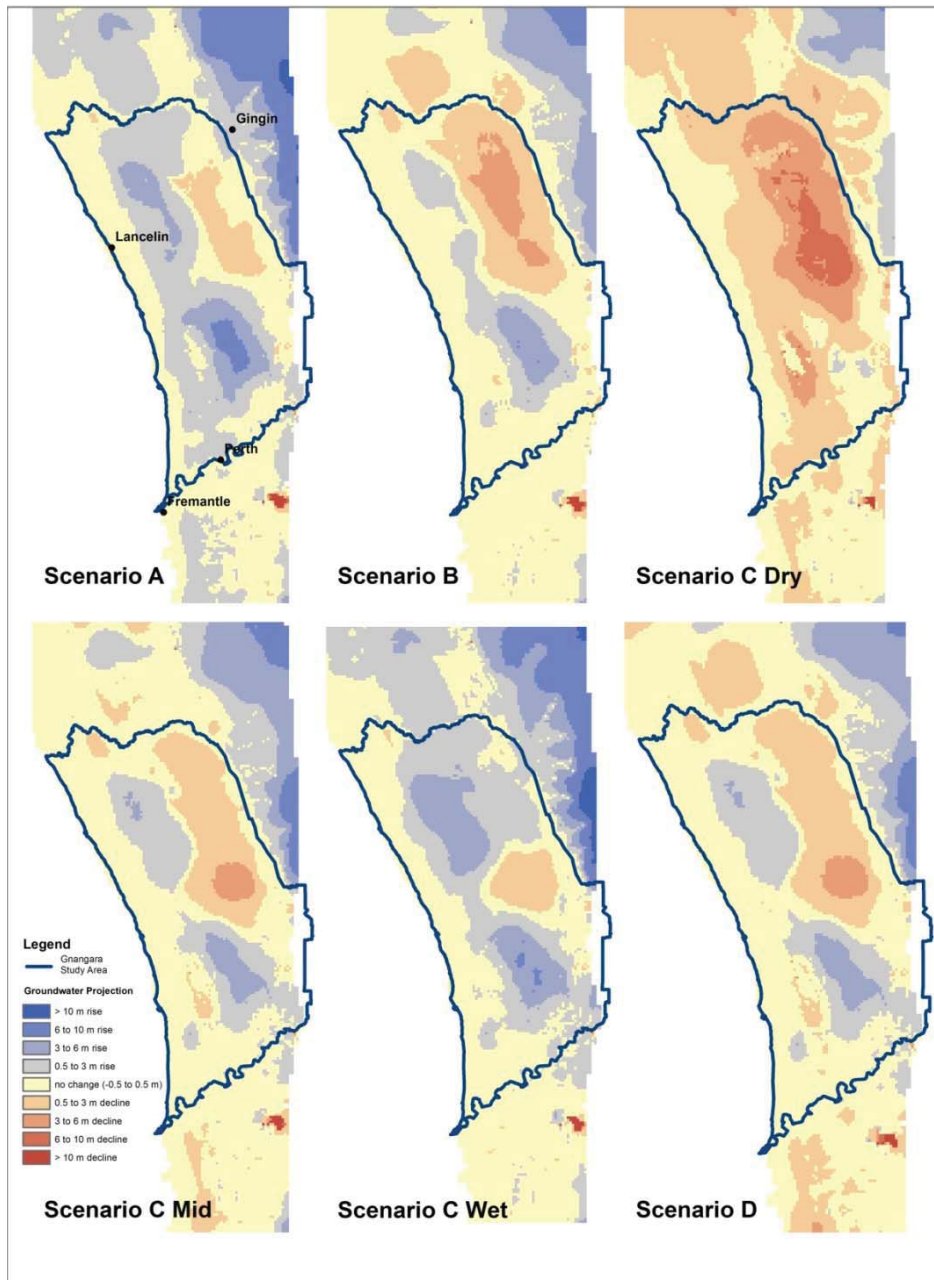


Figure 9: The magnitude of groundwater decline has been projected into 2030 under a range of climate change and land use scenarios for the Gnamptara Mound, Western Australia. These projections were made in the South West Sustainable Yields Project and are based on the PRAMS groundwater model. Scenario A: 2030 projections based on extension of the 1975-2007 period and current development (i.e. abstraction and land use unless known to change). Scenario B: 2030 projections based on extension of the 1997-2007 record and current development. Scenario C: 2030 projections using wet, median (mid) and dry GCM climate change scenarios and current development. Scenario D: Projections into 2030 under the CMid climate change scenario and abstraction taken out to the maximum limit (CSIRO 2009a).

Spatial and temporal change example 2: Combine groundwater decline and surface proximity – Simple Impact Assessment Maps

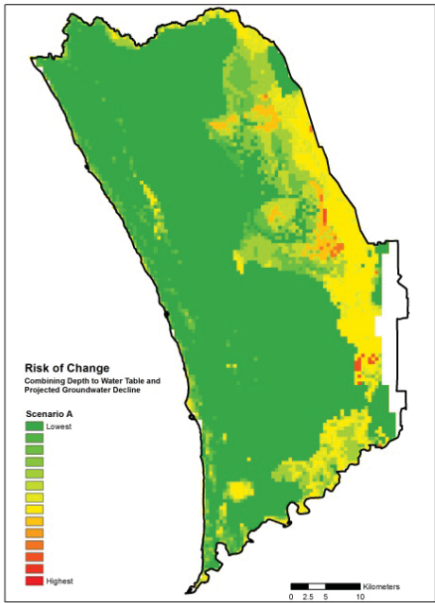
An alternative technique to assess risk is to undertake an assessment of the groundwater depths (Table 3) in conjunction with groundwater change projections (Table 4) to determine a 'simple impact assessment map' (Figure 10). Previous studies in the Gnamangara Mound (Froend and Loomes, 2004, Sommer and Froend 2010) as well as the analysis carried out in the current project (SD2 Sommer *et al.* 2013) have identified the importance of groundwater surface proximity in determining the sensitivity of GDEs to change in groundwater levels. The same studies identified that the amount (and by definition the rate) of groundwater change over time is also important.

Neither groundwater change nor depth to groundwater by itself will be a good indicator of risk: high levels of change at depth are unlikely to impact on GDEs, as the groundwater is beyond any rooting depth or surface hydrological impact. Alternatively, shallow water tables are not an issue if there is no projected decline into the future.

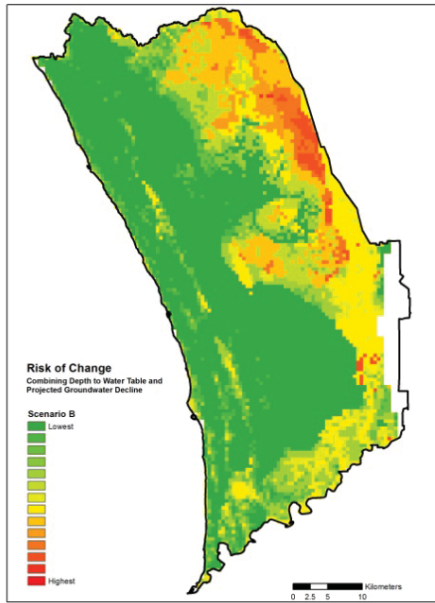
It is possible to combine these two factors – depth to groundwater and projected groundwater decline – in a simple weighted assessment, for example using the Modelbuilder Extension of ArcGIS 10. The 'simple impact assessment map' (see SD7 Neville (2013) for more information) is an adaptation of a technique developed by Froend and Loomes (2004) to estimate risk of groundwater decline on vegetation.

The technique requires that existing values for depth to water table or projected change are classified, and each class rated in terms of its perceived contribution to risk of change. The combination of factors ensures that areas are not falsely identified on the basis of large relative declines when the original depth to water table was minimal.

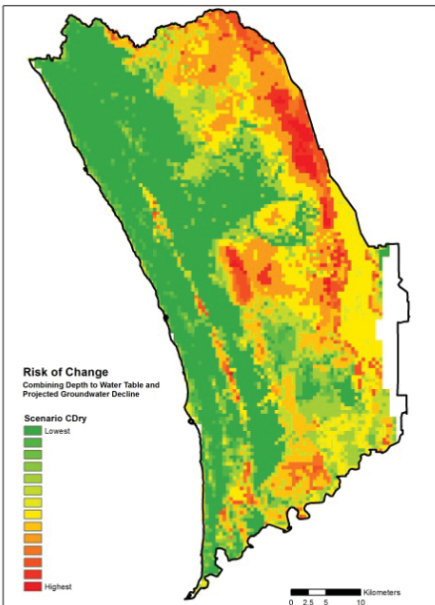
For the Gnamangara Mound, the groundwater depths less than 5m were rated as having the highest contribution (8-10) to impact, and depths below 20m rated with no contribution (Table 3). A projected groundwater decline of more than 0.5m is rated as having the highest contribution (8-10) and a rise of more than 0.5m rated as having no contribution (Table 4). This information was mapped to determine the areas that may suffer the greatest impact due to groundwater decline (Figure 10). This example shows ratings that are indicative and illustrative only.



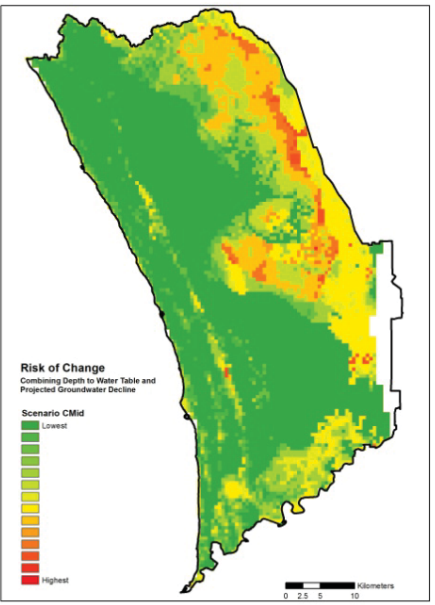
Scenario A



Scenario B



Scenario CDry



Scenario CMid

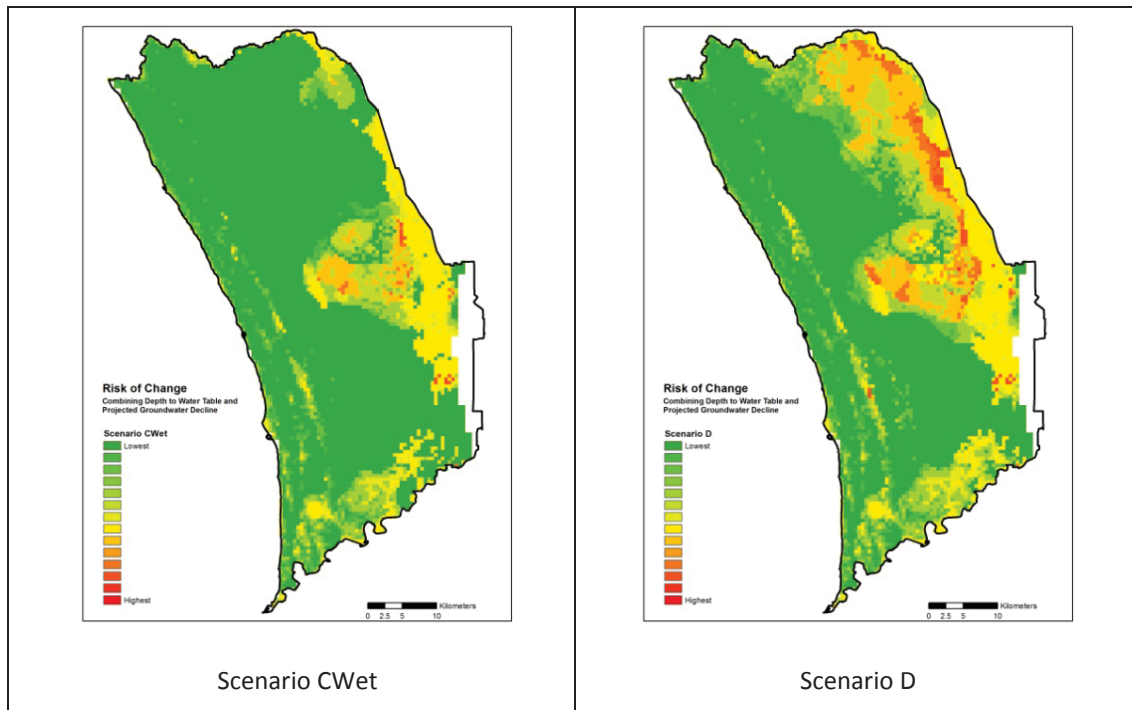


Figure 10: Simple Impact Assessment maps combining groundwater decline and surface proximity for the Gnamptara Mound Study Area. The maps identify areas that may be expected to suffer the greatest impacts from groundwater decline. These projections were made in the South West Sustainable Yields Project and are based on the PRAMS groundwater model. The Scenario A: 2030 projections based on extension of the 1975-2007 period and current development (i.e. abstraction and landuse unless known to change). Scenario B: 2030 projections based on extension of the 1997-2007 record and current development. Scenario C: 2030 projections using wet, median (mid) and dry GCM climate change scenarios and current development. Scenario D: Projections into 2030 under the CMid climate change scenario and abstraction taken out to the maximum limit (CSIRO 2009a).

Spatial and temporal change example 3: Determine the rate of groundwater decline

While the magnitude of groundwater decline is important, so too is the rate of decline (Figure 11). Slow changes provide time for the ecosystem to adapt, while rapid decline may exceed the plants' and animals' ability to respond and result in the death of organisms. Rapid water level decline can also result in rapid changes to water quality e.g. increasing temperature, concentration of salts, potentially resulting in the thresholds being exceeded for the extant biota.

Rate of decline can simplistically be measured by dividing the magnitude of decline by the time period it is observed or projected. This was undertaken for the Gnamptara Mound, where the magnitude of groundwater decline from the CSIRO South West Sustainable Yields Project, based on the PRAMS groundwater model (CSIRO 2009a), was divided by the number of years in each climate change and land use scenario to determine the rate (m/year). This is a simplification of change over time, as a decline in the water table level may actually increase during drier periods and slow or even reverse for periods of wet years – even though the long-term trend is one of decline.

High rates of decline, such as those that occur in severe droughts, may cause irreversible changes in the ecosystem. While it is not possible to predict these extreme events, rates of decline do provide some measure of the severity of drying.

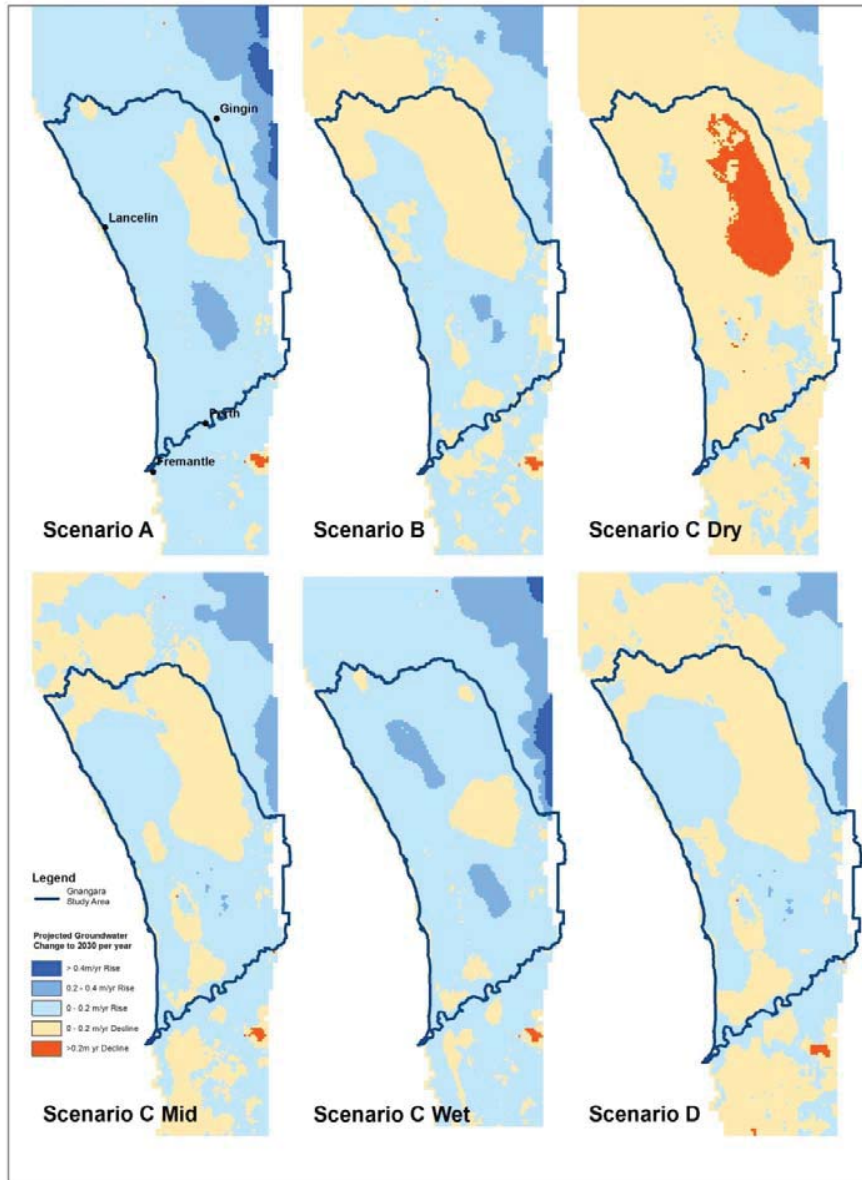


Figure 11: The rate of groundwater decline (m/year) was projected into 2030 under a range of climate change and land use scenarios for the Gnamagara Mound, Western Australia. These projections were made in the South West Sustainable Yields Project and are based on the PRAMS groundwater model (CSIRO 2009a). Scenario A: 2030 projections based on extension of the 1975-2007 period and current development (i.e. abstraction and landuse unless known to change). Scenario B: 2030 projections based on extension of the 1997-2007 record and current development. Scenario C: 2030 projections using wet, median (mid) and dry GCM climate change scenarios and current development. Scenario D: Projections into 2030 under the CMid climate change scenario and abstraction taken out to the maximum limit.

Spatial and temporal change – Projecting the resulting change in surface water levels due to projected changes in groundwater levels in 2030

An attempt was made to compare changes in groundwater level projected by the South West Sustainable Yields Project using the Perth Regional Aquifer Modelling System (PRAMS) (CSIRO 2009a) and changes in lake levels, using the existing relationship between the lake water depth and groundwater level in bores near each wetland. At time of writing this information had produced some valuable regressions but there was insufficient time to use it in water level projections for wetlands on the Gngangara Mound for 2030. This will be carried out in future studies.

Step 2.2: Accommodate dynamics of hydrological change

The South West Sustainable Yields projections of groundwater change to 2030 provide a single projected level at 2030 under the various scenarios (CSIRO 2009a). Although the models output monthly projected levels, these cannot be used to project hydroperiods (dry-wet periods) in wetlands. This is principally as the relationship between the groundwater level and the surface water level in each wetland is complex and individual, but also because the models are at a coarse scale relative to individual wetlands.

In the Gngangara Study area however an alternate dataset was available for 16 wetlands that were intensively studied and sampled between 1996 and 2010. The data collected included number of dry days per year over the duration of sampling. This data were compared to water quality and biotic variables in the effects assessment below, and was used in risk modelling.

3.1.5 Step 3: Assess Effects

Step 3.1: Collate available data

The data available varied for the separate biotic guilds and therefore the remainder of this section is separated into the two case studies: the data-rich Case Study 1, focusing on macroinvertebrates and littoral and sub-littoral vegetation to assess the impact to groundwater dependent wetlands from declining water levels, and the data-poor Case Study 2 determining the predicted response of amphibians to a drying climate. The data collected for each study area are available in Appendix 3, Table 3.

Amphibians had empirical data on species distribution available for a limited number of years for the Gngangara Mound, but this was insufficient for multivariate statistics to provide a conclusive result. Instead the data were interpreted in expert workshops (see SD3: Mitchell *et al.* 2013) and their combined expert opinion was used to characterise risk outlined in step 4.1.

Step 3.2: Develop a conceptual model

CASE STUDY 1:

Case Study 1 had a large, long-term data set on macroinvertebrates and littoral and sub-littoral vegetation in the groundwater dependent wetlands of the Gngangara Mound. To identify the key drivers that should be entered into the conceptual model, a suite of statistics were used to investigate the relationships between the biotic component (wetland plants or macroinvertebrates) and physiochemical parameters and/or needed biotic resources. Further statistical methods were used to identify which of these were key drivers in causing a change in species composition within the communities due to the decline in water levels that had already occurred on the Gngangara Mound since the onset of sampling. Examples are provided below of the methodologies used to identify the key drivers for the plant and macroinvertebrate communities.

Wetland plant communities

Methods

Canonical Analyses of Principal Coordinates (CAP) were used to investigate associations of vegetation with landscape factors. Plant species were classified into hydrotypes (or the plant requirements for water e.g. littoral, supra-littoral mesophytic, xerophytic plants) as this best reflects the functional relationship between the plants and the habitat requirements provided by site hydrology. Responses to declining water levels were expressed as changes in the proportion of hydrotypes between historical and the most recent data. Next, a distance-based (Bray-Curtis) redundancy analysis was performed in order to find out which hydrological and/or water quality variables best explain the spatio-temporal distribution of littoral vegetation on the Gngangara Mound.

Results

Littoral and supra-littoral vegetation on the Gngangara Mound was strongly related to lithology (and associated consanguineous suites and substrate types; Figure 12). These associations largely held true in the face of significant climate and hydrologic changes that have taken place in the recent past. Three-year mean annual rainfall, mean maximum summer temperature, groundwater depth, date on which the peak water level occurred and the number of days between recorded troughs and peaks were all significant drivers of floristic change that occurred between historical (i.e. the earliest) and recent monitoring years. However, of these, groundwater depth (including magnitude and rate of decline) was by far the most important. For the plant communities, the key drivers were determined through multiple regression tree analysis while identifying thresholds (Figure 16). For more detail on this analysis see SD2: Sommer (2013).

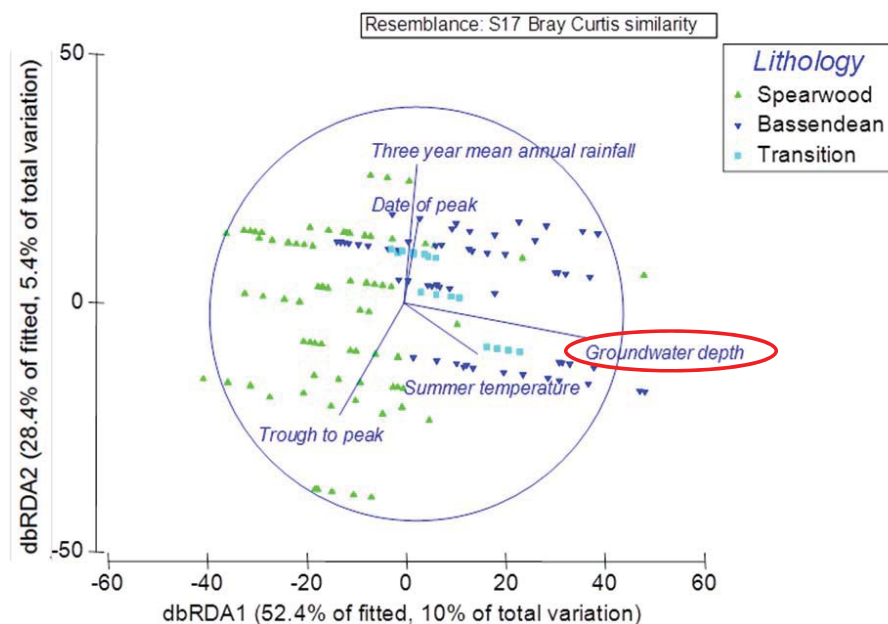


Figure 12: db-RDA ordination of plant community data and hydrological and physico-chemical factors on the Gngangara Mound.

Macroinvertebrate communities

Methods

Canonical Analyses of Principal Coordinates (CAP) were used to investigate if there were any associations of macroinvertebrates with landscape factors. A distance-based (Bray-Curtis) redundancy analysis was performed in order to find out which hydrological and/or water quality variables best explain the spatio-temporal distribution of aquatic macroinvertebrates on the Gngangara groundwater system.

Results

CAP analyses suggest a strong association of macroinvertebrate composition with geomorphology and associated habitat characteristics (substrate, vegetation association and consanguineous suite). The associations held true in spite of the considerable climatic and hydrological change that has occurred over the monitoring periods. These associations suggest that superficial geomorphic setting is an important factor and therefore should be taken into consideration when predicting responses to environmental change. The db-RDA analysis indicated that pH, annual maximum depth, ammonium and the number of dry days were the most important drivers of macroinvertebrate composition in Gngangara groundwater system wetlands (Figure 13). For more detail on this analysis see SD2: Sommer (2013).

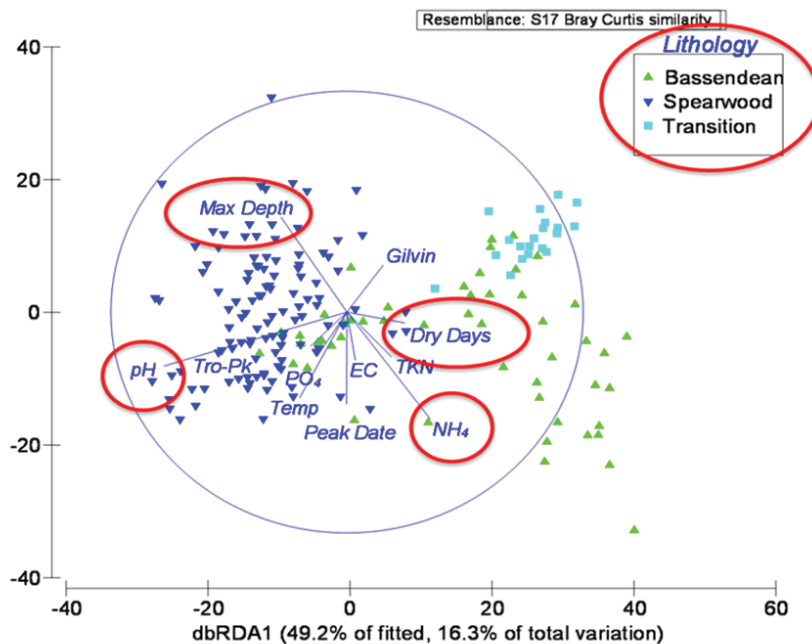


Figure 13: db-RDA biplot for spring macroinvertebrate data and hydrological and physico-chemical factors on the Gngangara Mound.

Conceptual model

The key ecosystem drivers identified by the statistical analysis in the methods section above were incorporated into the conceptual model (Figure 14). For more information on this case study see SD2: Sommer *et al.* (2013).

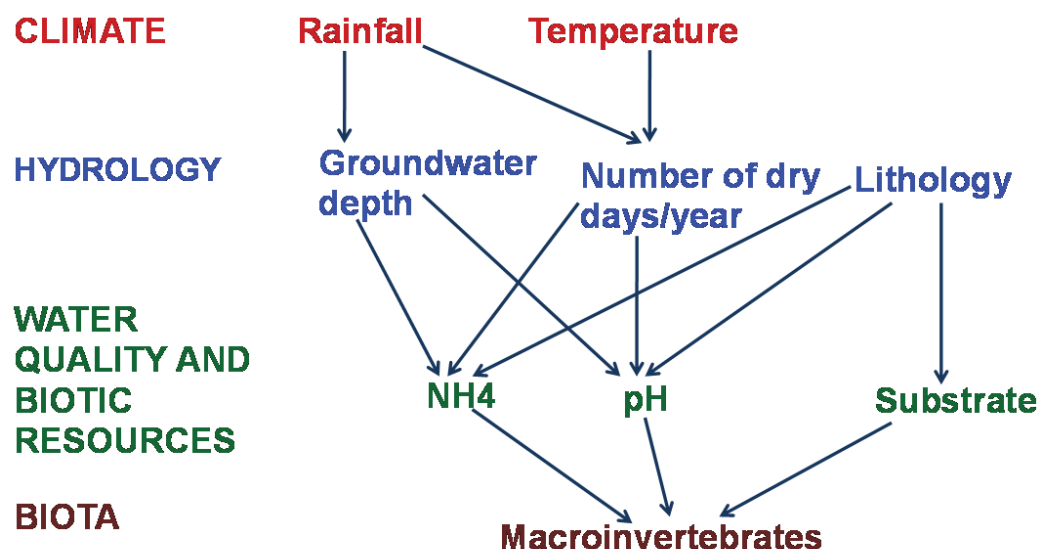


Figure 14: Conceptual model of the Gngangara Mound wetlands and the key ecosystem drivers to macroinvertebrates.

CASE STUDY 2:

Amphibians

Methods

While the distribution of nine amphibian species occurring within the Gngangara Mound had been systematically surveyed for previous projects, the dataset was not well suited to multivariate analysis of the environmental drivers that explained presence or absence at breeding sites. Similarly, there was an absence of empirical data on environmental thresholds of embryonic, larval and adult life stages for almost all the species. Instead, the opinions of four experts on amphibian biology were used to derive conceptual models and identify key thresholds for three reproductive guilds of amphibians: aquatic-breeding species (*Crinia glauerti*, *C. georgiana*, *C. insignifera*, *Limnodynastes dorsalis*, *Litoria adelaidensis*, *Litoria moorei*), species with terrestrial embryos and aquatic larva (*Heleioporus eyrei* and *Pseudophryne guentheri*), and an entirely terrestrial species that breeds underground (*Myobatrachus gouldii*) using life histories (e.g. Figure 15).

Results

In all cases, a reduction in seasonal rainfalls that trigger breeding activity (or other key events such as flooding of nest sites) were viewed as important drivers that would influence the probability that each species would either persist, or experience a slight or severe decline. Similarly, the wetland hydroperiod and salinity were two additional drivers considered to threaten recruitment to metamorphosis in the aquatic breeding guild, and the terrestrial-aquatic breeding guild of amphibians.

Conceptual model

As outlined in Case Study 1 above, the conceptual model shown in Figure 16 is a simple representation of the key drivers identified through the expert workshop using

life history traits shown in Figure 15. For more information on this case study see SD5: Mitchell *et al.* (2013).

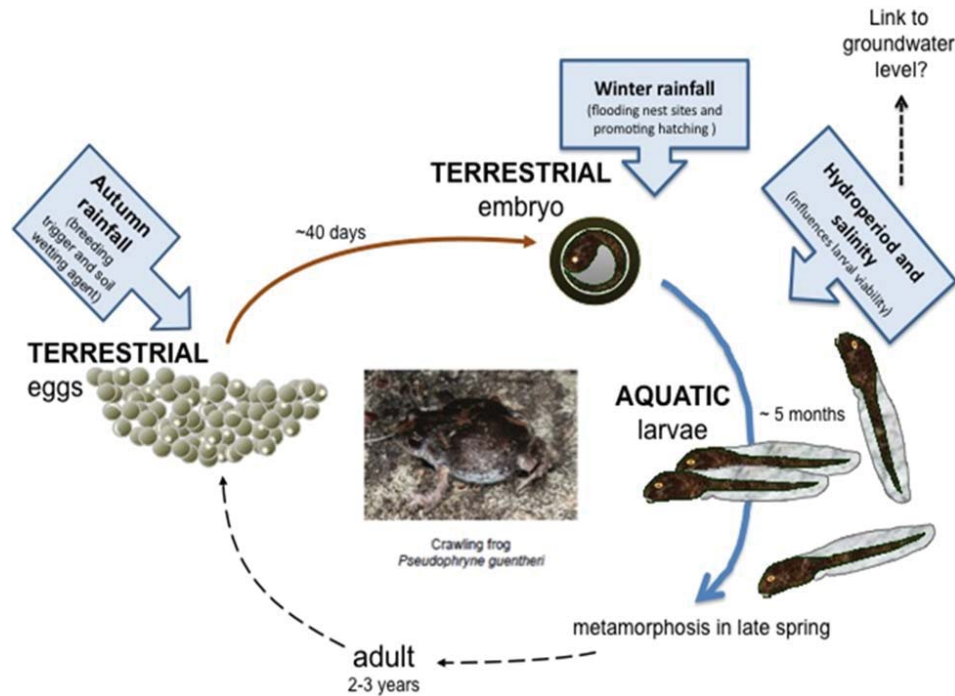


Figure 15: Life cycle of an amphibian species - *Pseudophryne guentheri* - with terrestrial embryos and aquatic larval life stages. The blue arrowed boxes indicate thresholds that were incorporated into Bayesian Belief Networks.

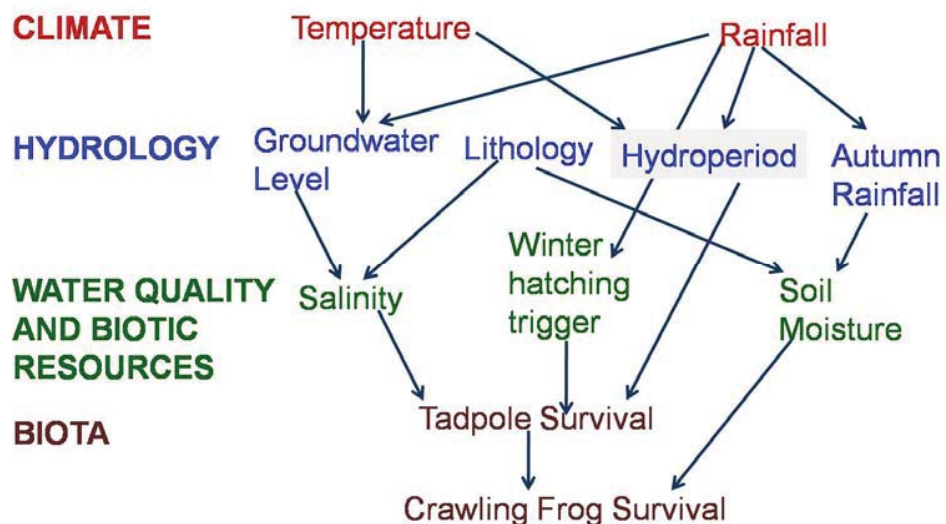


Figure 16: The conceptual model developed for the survival of the crawling frog (*Pseudophryne guentheri*), based on the species life cycle.

3.1.6 Step 4: Characterise Risk

Step 4.1: Determine Thresholds

CASE STUDY 1

Wetland plant communities

Methods

Multivariate Regression Trees (MRT) were used to identify hydrological habitats as characterised by ranges of groundwater depths and corresponding hydrotype community composition. Further MRTs were performed to determine the expected change in hydrotype composition in response to the magnitude and rate of groundwater drawdown, given the initial groundwater depth (i.e. historical depth before reduced rainfall occurred). Finally, BBN analyses were performed in order to build predictive models based on future climate, water allocation, and landuse scenarios.

Results

The MRT analyses also identified groundwater depth as being the most important driver of hydrotype composition, the main split criterion (i.e. the most important threshold value) being at 2.0 m. This criterion essentially divided the observations into those dominated by hydrophytes associated with a <2 m groundwater depth, and those not dominated by hydrophytes found at groundwater depths >2 m. The MRTs that were run to directly assess changes in hydrotype composition (proportion and percentage change in abundance) as a function of the magnitude of drawdown, whilst also taking into consideration the initial water depth, are shown in Figure 17. The thresholds derived from these MRTs were used in the BBN shown in Figure 18.

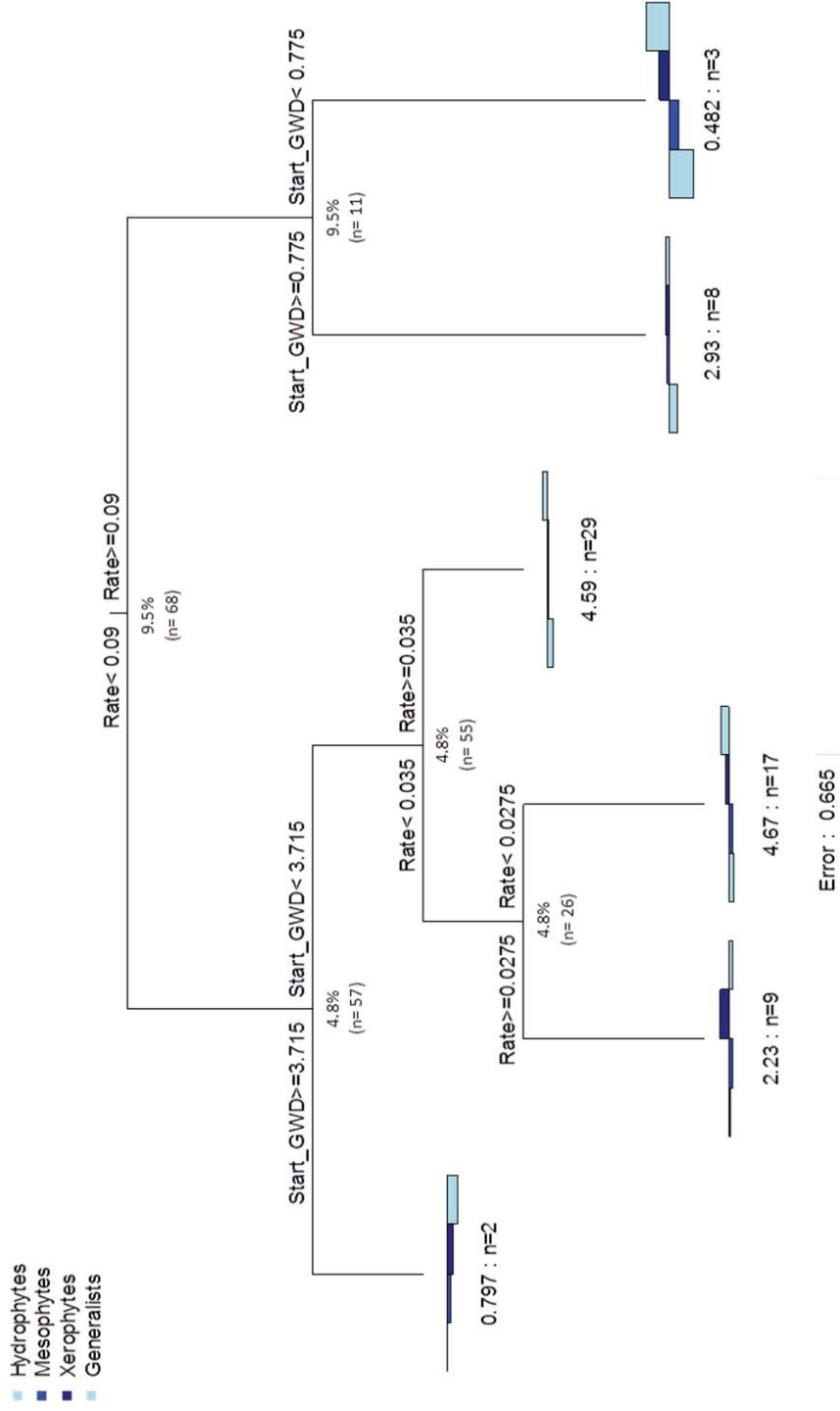


Figure 17: Regression tree showing the changes in the proportions of hydrotypes in relation to groundwater decline ('magnitude' in meters). Bar plots are hydrotypes, from left to right, hydrophytes, mesophytes, xerophytes, and generalists. 'Start-GWD' = groundwater depth (in metres) during historical wet period.

Macroinvertebrate communities

Methods

Multivariate Regression Trees (MRT) were used to determine hydrological and water quality thresholds for suites of macroinvertebrates (families and functional groups). These thresholds were used in Bayesian Belief Networks (BBN) that were run using the same data as for the MRT analyses.

Results

The MRT analyses identified pH as being the most important driver of both macroinvertebrate family composition and composition of functional group categories. Other factors included gilvin (dissolved organic carbon), ammonium (NH₄⁺), number of dry days and maximum depth of the wetland (Table 3).

Table 3: Ranges of threshold values for individual hydrological and chemical variables determined from the mean values of the splits in the MRTs created for the years 1996, 1998, 2006 and 2010 for macroinvertebrate family composition and functional group composition.

Variable	threshold ranges	Description
pH	>7.5	alkaline
	6.1 - 7.5	slightly acidic to slightly alkaline
	5.5 - 6.1	acidic
	4.1 - 5.5	very acidic
	<4.1	highly acidic
gilvin	<7	not coloured
	7 - 26	coloured
	>26	very coloured
no dry days	>55	strongly seasonal
	<55	seasonal
max depth	<0.65	shallow
	0.65 - 0.95	medium
	>0.95	deep
Ammonium (inconclusive)	< 20 ug/L	low
	>20 ug/L	high

CASE STUDY 2

Amphibians

Thresholds for each of the key drivers outlined in the conceptual model (Figure 16) were determined through expert opinion in the workshops mentioned above. In the example of an amphibian species with terrestrial embryos and aquatic larval life stages these were defined as a hydroperiod of at least three months to allow tadpoles to complete metamorphosis, a salinity of less than 8ppt and the presence of autumn and winter rainfall triggers.

Step 4.2 Determining probabilities of risk: Bayesian Belief Networks

CASE STUDY 1

Wetland plant communities

The development of Bayesian Belief Networks under data-rich conditions is complex and the reader is directed to SD2: Sommer *et al.* (2013) for detail on the methodology. You may like to skip down to Case Study 2 (pg 44) to work your way through a simpler example before interpreting the BBNs for wetland plant and macroinvertebrate communities.

The Bayesian Belief Network in Figure 18 shows a 'bad-case scenario' for a 100% 'groundwater decline' of 0.45 m to 1 metre, 100% 'rate of decline' of 0.03 to 0.10m/year, and 100% 'starting groundwater depth' of 0.50 m to 2 m. Under such a scenario, there would be a 68.3% probability that there would be a large adverse change in the proportion of functional groups (or hydrotypes e.g. littoral, supra-littoral mesophytic, xerophytic plants), an 89.5% probability of a change in vegetation state and a 93.7% probability of a large adverse change in plant abundance.

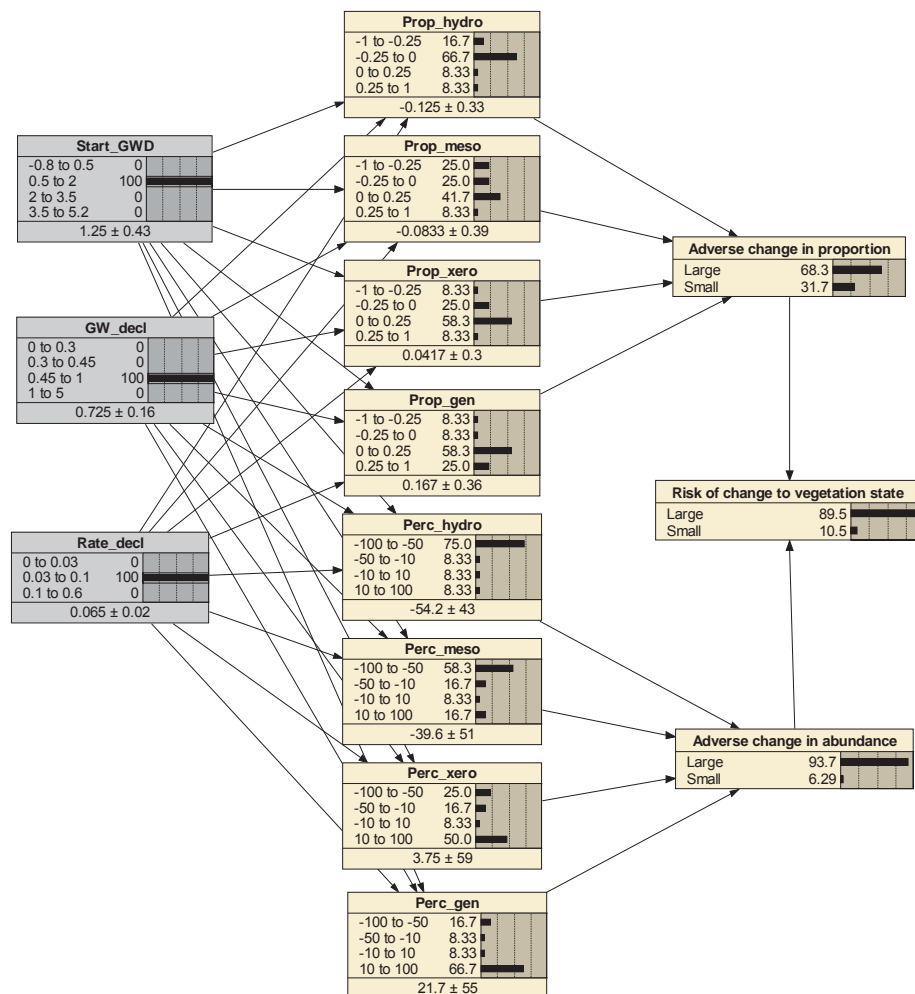


Figure 18: Bayesian Belief Network for assessing the risk of change to the state of littoral and supra-littoral vegetation on the Gngara Mound posed by the magnitude (meters) and rate (meters/year) of drawdown, and taking into account the groundwater depth (meters) at commencement of monitoring.. The change in vegetation state is characterised by the change in the proportion (top 4 boxes) and the percentage change in abundance (bottom 4 boxes) of hydrotypes. The belief bars show the conditional probabilities of a particular change occurring. Portrayed are results for a 100% 'groundwater decline' of 0.45 m to 1 meter, 100% 'rate of decline' of 0.03 to 0.10 m/year, and 100% 'starting groundwater depth' of 0.50 m to 2 m (i.e. a 'bad-case' scenario).

Macroinvertebrate communities

The Bayesian Belief Network in Figure 19 shows a 'worst-case' scenario in which 'number of dry days'= 55 to 330, 'groundwater depth'= 0 to -0.65 m, and 'lithology'= 'Bassendean', where the probability of 'overall wetland risk' being 'high' is 73.5%. Under such a scenario, there would be a 50% probability that pH would be between 3 and 4.1, a 52.9% probability that ammonium concentrations would be very high, and macroinvertebrates would be strongly dominated by insects, predators and acid-tolerant taxa.

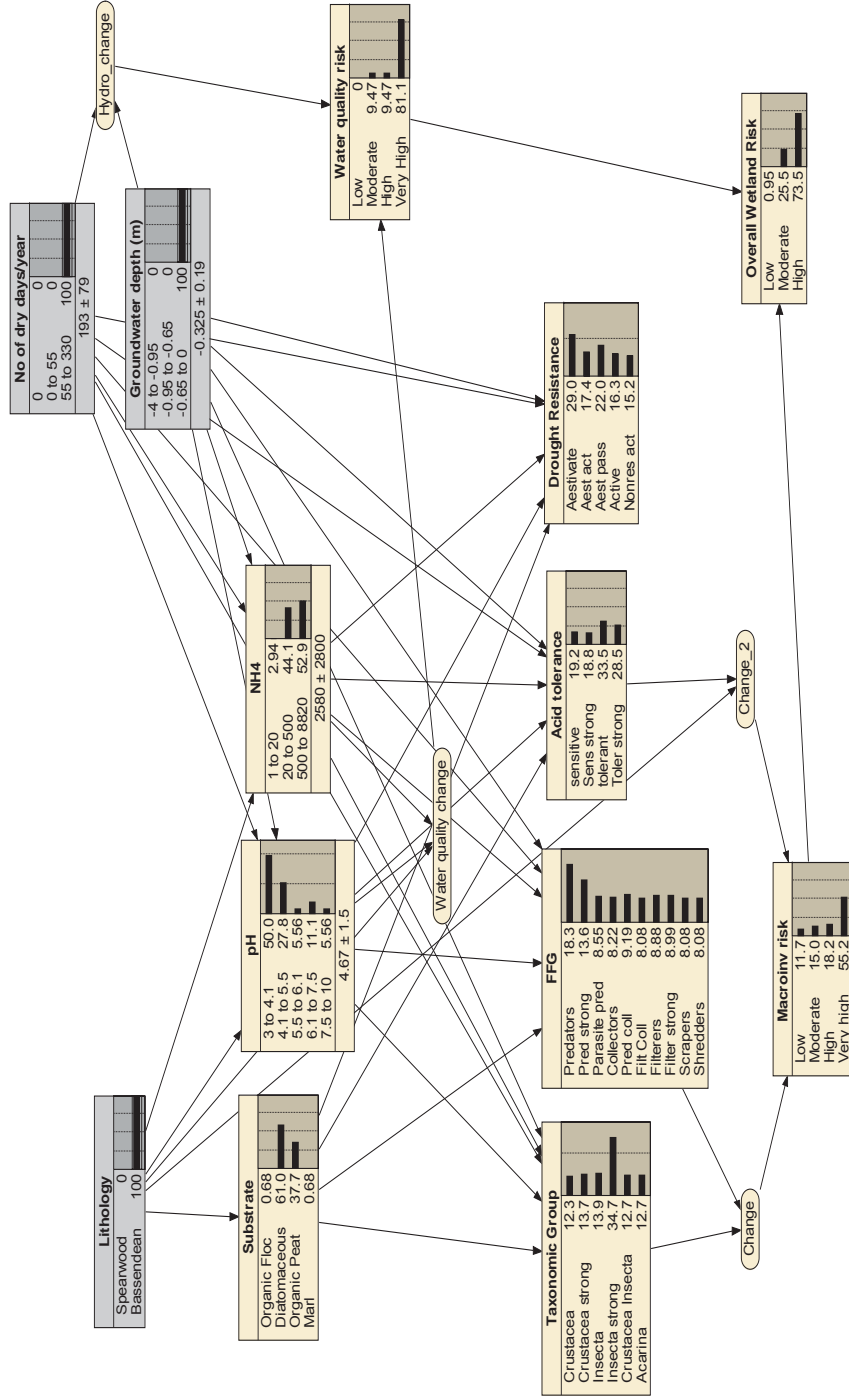


Figure 19: Bayesian Belief Network for predicting the relative dominance of macroinvertebrate functional groups in response to hydrological, chemical and landscape drivers. The belief bars show the percentage of cases that fall within respective thresholds (derived from MRT analyses) or within categories (functional classes and risk boxes). The drought resistance class was not linked to risk because this did not influence risk (see text). This BBN shows results for 100% '55 to 330' dry days, 100% '0-0.65 to 0 m' groundwater depth, and 100% Bassendeen lithology (i.e. a 'worst-case' scenario).

A combination of Water Quality, Plant and Macroinvertebrate community outcomes to provide a whole ecosystem risk assessment

To develop an overall risk of wetland health based on both plant and macroinvertebrate communities the two models were joined, with their final outputs being used to populate a wetland health index conditional probability table (Figure 20). The two outputs were combined into a wetland health conditional probability table where if both inputs were 100% low risk then wetland health was rated 100% low risk, if both inputs were 100% high risk then wetland health was rated high risk (Figure 17). Risk between the two extremes was determined by expert opinion.

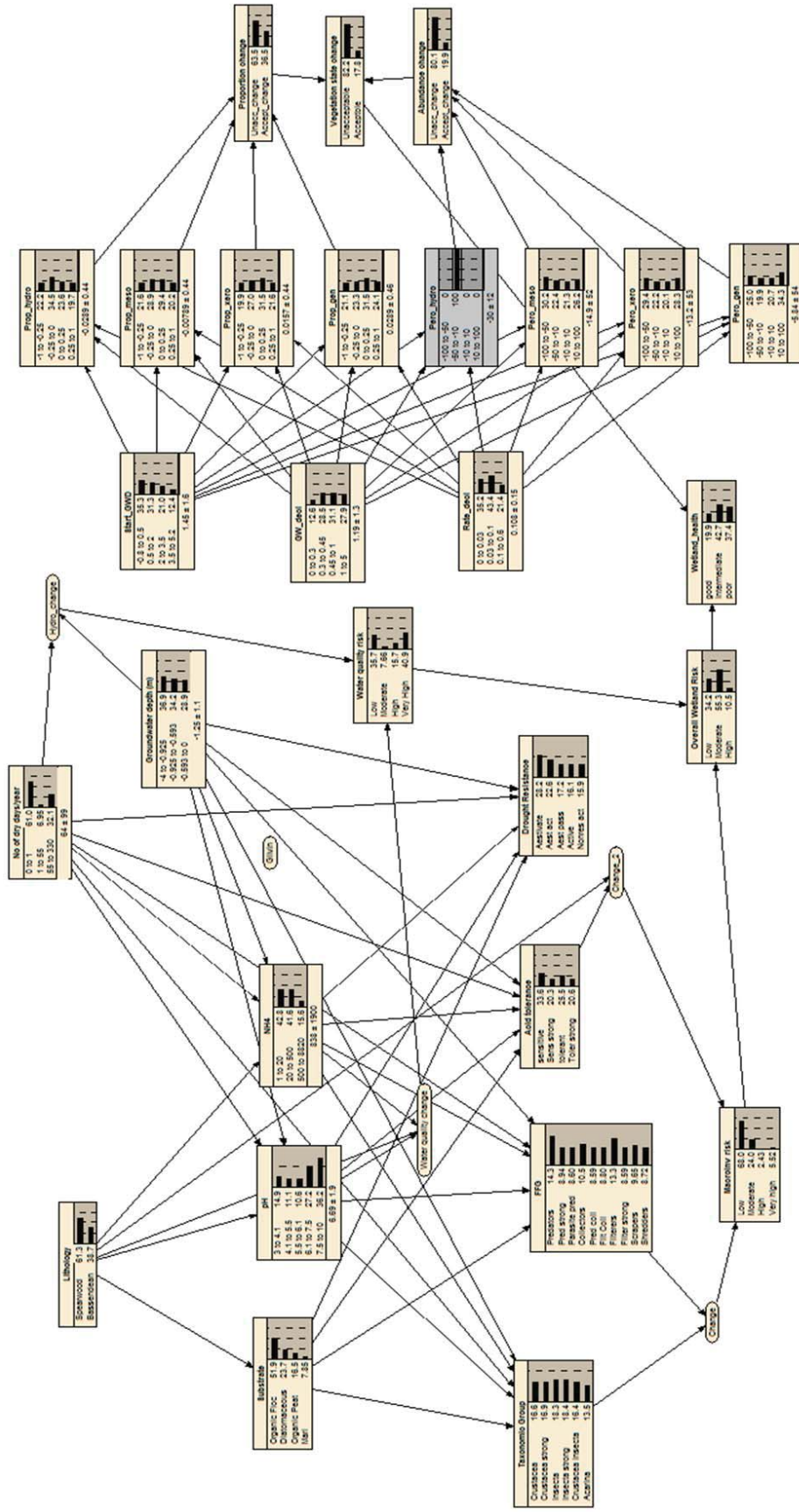


Figure 20: The Macroinvertebrate and Plant community models were combined to provide a Bayesian Belief Network to assess risk for the whole ecosystem.

CASE STUDY 2

Amphibians

After consideration of the relationships between hydrology, water quality and resource requirements of amphibians, separate Bayesian Belief Networks were constructed for the three reproductive guilds. The same panel of experts used to identify thresholds (Step 4.1: Determine Threshold for Amphibians) were presented with a preliminary structure for each BBN, based on conceptual models devised by the lead author of the amphibian study (Mitchell). The experts were given the opportunity to modify the models, but were advised to prevent the models from becoming too complex. Once agreement on model structure was reached, experts then populated the conditional probability tables of the BBN by a process of consensus.

Once conditional probability tables for each node were completed, experts were given the opportunity to alter the tables if required and simple scenarios were run through the model to check if model outcomes matched with the expected outcome predicted by the panel. In general, experts relied on a combination of their knowledge of relevant publications on amphibian life histories, unpublished environmental and physiological data, and personal experience to decide on the structure of the conceptual model and the probability values. An example of the BBN devised for terrestrial breeding amphibians with aquatic larvae is shown in Figure 21.

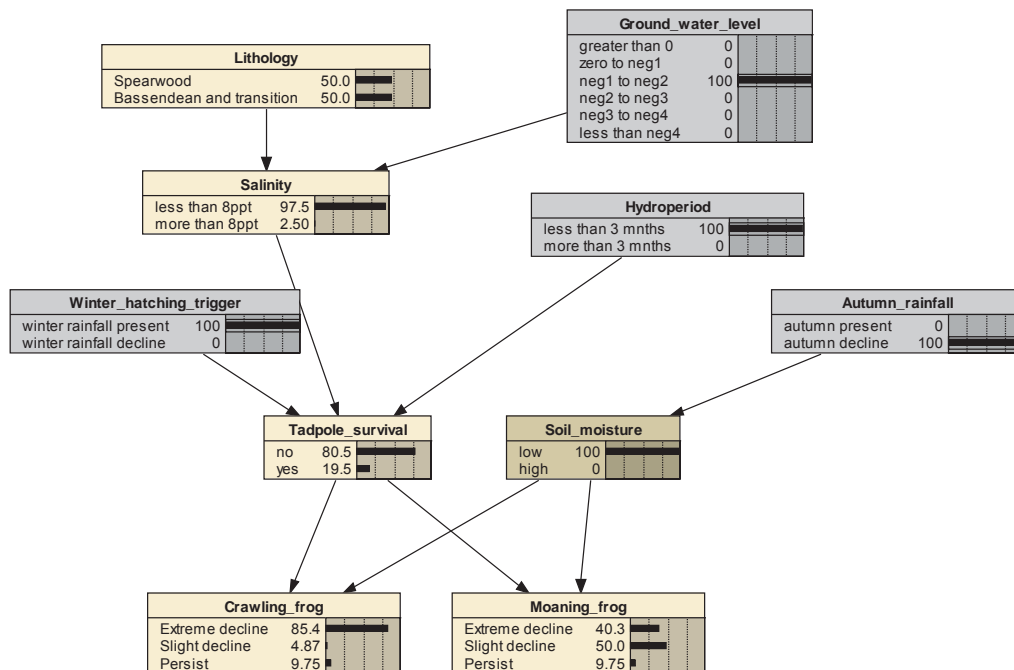


Figure 21: The BBN for terrestrial-breeding amphibians with aquatic larvae, illustrating the impact of groundwater between 1-2 m from the surface, a hydroperiod less than three months, and a decline in the autumn rainfall trigger for breeding. The difference in dependence on the autumn rainfall trigger is primarily driving the different responses between the two species.

The relationship between groundwater levels and wetland hydroperiod could not be determined with empirical data or a simple wetland model, due to the differing dependence of wetlands on groundwater. The hydroperiod of perched wetlands, with no groundwater connectivity, would be primarily influenced by rainfall, vegetation and catchment topography, hence hydroperiods were incorporated into the BBNs as a stand-alone node that influenced the probability of tadpole survival. The influence of groundwater levels were instead expressed as the probability that wetlands would become saline (exceed 8 ppt), based on empirical data for the Gngangara Mound across two lithology categories (Spearwood and Bassendean sands). Hence the outputs of the models indicated a negligible influence of groundwater decline on amphibian communities, chiefly because high salinity values were rarely expressed. In contrast, factors such as the wetland hydroperiod, and declines in seasonal rainfall that acts as a breeding trigger, soil wetting agent, and source of runoff for flooding of nest sites had a greater influence on the probabilities of population decline (e.g. Figure 21).

For the crawling frog example show in Figure 22, the probability of extreme population decline exceeded 80% under hydroperiods less than 3 months and a decline in autumn and winter rainfall. This was one of three species (*C. glauerti*, *C. insignifera* and *P. gunetheri*) that were viewed as being most sensitive to hydrological change due to their relatively short life cycles and specific breeding requirements. Notably, the entirely terrestrial species (the turtle frog *Myobatrachus gouldii*) modelled in this project could potentially benefit from hydrological change if wetlands transition to terrestrial woodlands. However, turtle frogs depend on functioning Banksia woodlands, which are vulnerable to over abstraction of groundwater (Groom *et al.* 2000).

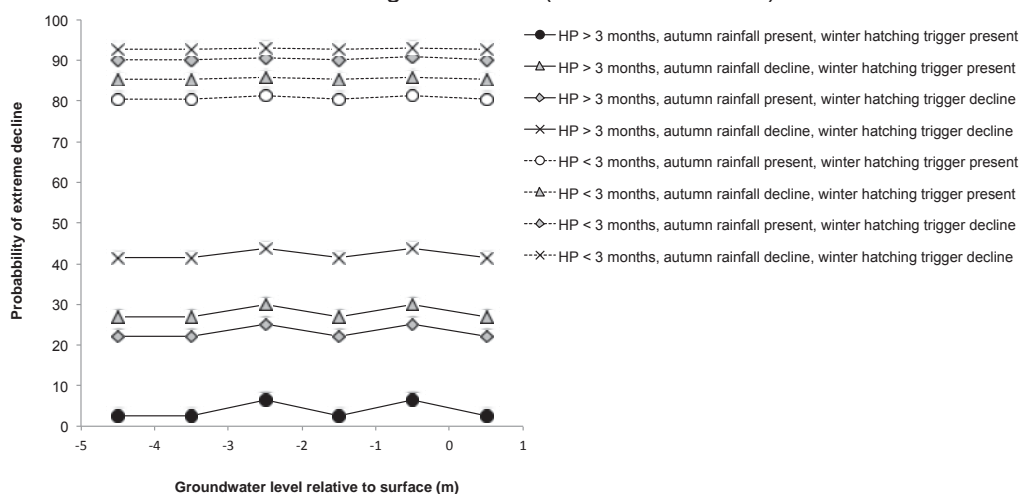


Figure 22: The relationship between groundwater levels and probability of population persistence for the Crawling frog, illustrating the influence the hydroperiods and seasonal rainfall triggers for key events in the species' lifecycle.

Step 4.3: Spatial mapping of risk

Wetland plant communities

The wetland vegetation BBN developed in Step 4.2 above can be applied spatially to predict risk probability over the Gngangara Study Area, and in this form is referred to as the 'Gngangara vegetation change model'. This is a broad-scale model that predicts vegetation change in areas where groundwater is less than 5.2m deep. The Gngangara Study area is extensive (228,000 ha), having 8870 PRAMS reporting cells on a 500m

grid. However the limitation on groundwater depth means that results are reported from only 1503 cells, or just fewer than 17% of all cells, as shown below (Figure 23).

The model uses three data inputs from the BBN: starting groundwater depth, groundwater decline and rate of groundwater decline. All of these are sourced from the PRAMS model monthly results tables, which report projected water table heights in meters for each month of the year at 2030. Data was extracted for each of the 6 CSIRO Scenarios (CSIRO 2009a) at the year 2030.

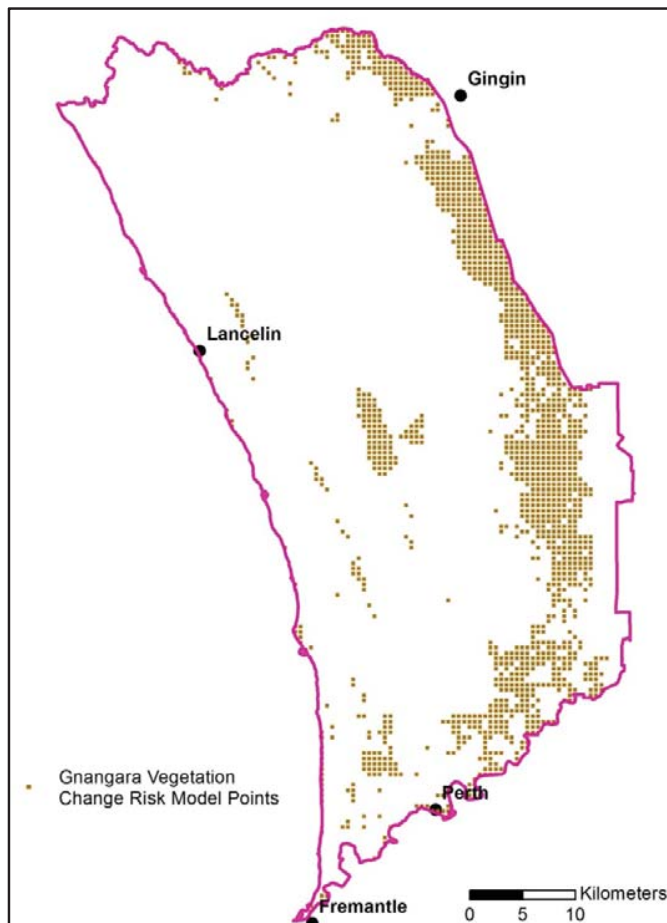


Figure 23 – Gnamptu vegetation change model reporting points.

The required data values were calculated from the files of monthly projected heights (AHD) for each model point as exported from PRAMS. Each PRAMS Scenario output was joined to the X,Y coordinates for each point, added into the ArcGIS ArcMap GIS software, and a point file values for starting groundwater depth, groundwater decline and rate of groundwater decline were created for each scenario.

The ‘Vegetation Change model’ is able to use continuous values, but it was necessary to truncate input values to match the extents used in the BBN. Starting groundwater depth was left unchanged, but values greater than 5.2m were excluded from final reporting, while values beyond -0.8 (i.e. above ground) were treated as -0.8. Groundwater decline values were truncated to remain between 0 and 5, while rate of decline values were truncated to remain 0 and 0.6.

The results for each scenario with returned to ArcMap and mapped. The output representation is in the form of single quantified legend for the probability that the risk of change to vegetation state is large. Colours were chosen to make visual identification of this easy. The example provided in Figure 24 is for Scenario CMid but in fact the probability for a large vegetation change does not drop below 65% for all scenarios (see SD7: Neville, 2013).

This spatial risk assessment suggests that irrespective of scenario the outlook for wetland vegetation on the Gngangara Mound is bleak. This is despite the rising watertable areas shown in Figure 24. However, it is likely that the combination of scale issues (i.e. resolution differences in the PRAMS modelling (500m) with the size of the wetland plant community composition (often only tens of metres)) and holes in the range of scenarios projected (i.e. not all interactions in the BBN are supported by actual events) has impaired the predictive capacity of the model. Further work is needed to rectify these difficulties. This highlights the necessity to track the validity of model outcomes before use in decision-making.

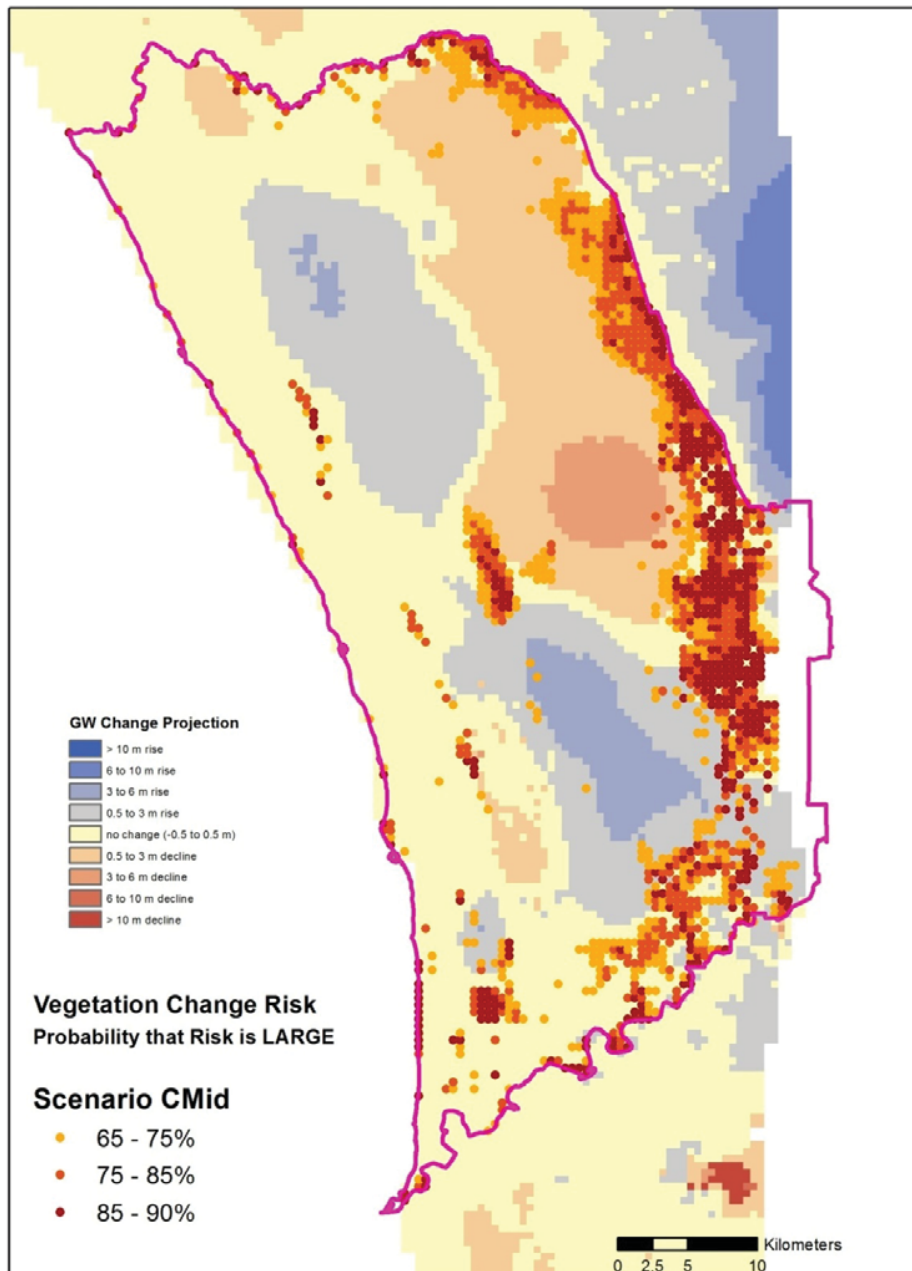


Figure 24 – The risk of vegetation change in the Gnamptu Mound, Western Australia overlaid on groundwater change projected by the South West Sustainable Yields Project, Scenario C Mid (2030 projections using median (mid) Global Climate Model climate change scenarios and current development) (CSIRO 2009a).

Combination of water quality and macroinvertebrate outcomes to provide a whole ecosystem risk assessment

The macroinvertebrate BBN developed in Step 4.2 can be applied spatially to predict the probability, and in this form is referred to as the 'Gnamptu macroinvertebrate model'. It is a site-specific model that predicts risk of change to macroinvertebrate

communities and water quality. The data required for the macroinvertebrate BBN are lithology, groundwater depth and number of dry days/year (or hydro-period).

The model is applicable to any wetland area within the PRAMS SWSY area; however for the spatial mapping the necessary datasets are only currently available from 16 wetland sites. The sites and the years of data collection are shown below (Table 4).

Table 4: Wetland sites and data collection in the Gngangara study area.

Name	Start Year	Finish Year
Lake Gngangara	1996	2010
Lake Goollelal	1996	2010
Lake Jandabup	1996	2010
Lake Joondalup North	1996	2010
Lake Joondalup South	1998	2010
Lexia 186A	2000	2007
Lexia 186B	2003	2005
Lexia 86	2000	2007
Lake Mariginiup	1996	2009
Loch McNess North	1998	2010
Loch McNess South	1996	2010
Melaleuca Park EPP173	2000	2010
Lake Nowergup	1996	2010
Pipiddiny Swamp	1996	2008
Lake Wilgarup	1996	1998
Lake Yonderup	1996	2010

Some of these sites (eg Lake Wilgarup, Lexia 186B) have earlier finish dates due to the wetland drying. Each site provided a single case for the BBN.

Data was provided from each wetland for two times in the year, effectively a dry and wet point. For the purpose of modelling we selected just the first and last data point for each site – the ‘start’ and ‘finish’ point. This gave the opportunity to illustrate change over time.

Each wetland site was saved in a table; supplied with AWRC reference number to geo-reference it, and added into the GIS. An output file listed lithology, groundwater depth and the number of dry days/year for each site.

Other than lithology (which was already categorical) the model was set up to use continuous values, so categorisation was not required. All input cases were within the model category bounds so no truncation was required. Results were saved as a case file for Netica.

The results were added into ArcMap and represented in the form of a single pie chart for each wetland, showing three probabilities of ‘overall wetland risk’ - low, moderate and high for the wetland survey start year (Figure 25) and the wetland survey finish year (Figure 26). Colours were chosen to make visual identification of this easy.

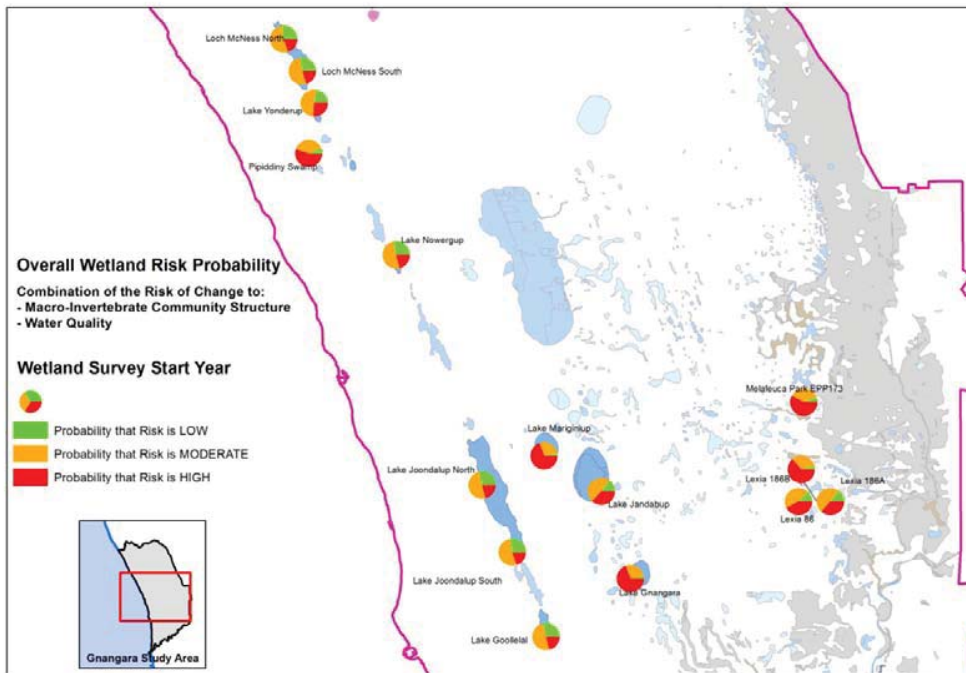


Figure 25: Spatial mapping of the 'Gnangara macroinvertebrate model' - probability that the risk of overall wetland change is low, moderate or high for the wetland survey start year (Table 4) on the Gnangara Mound.

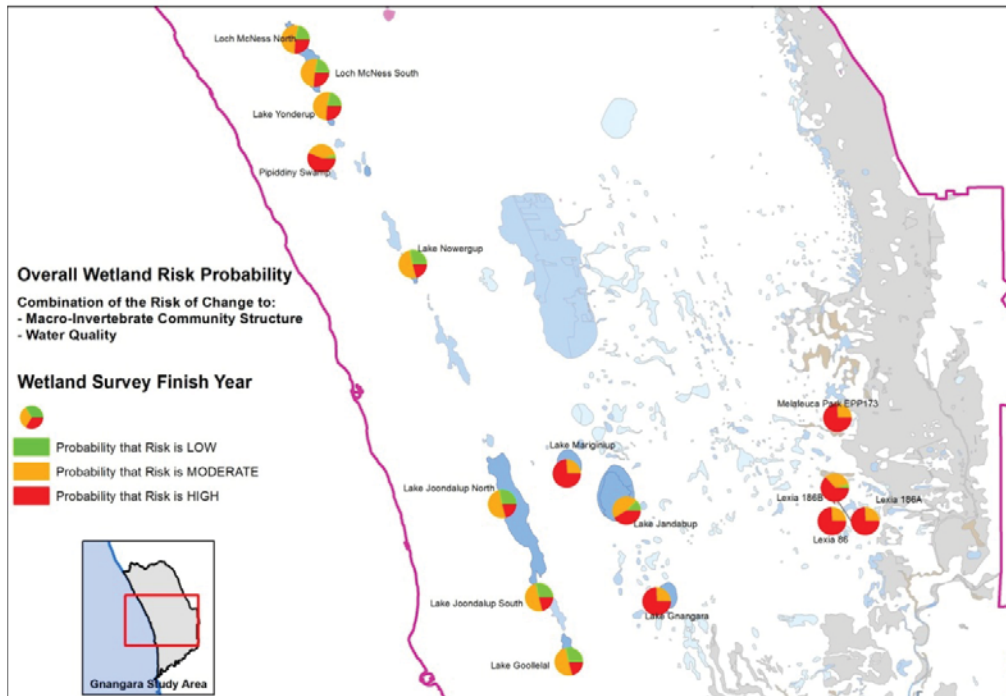


Figure 26: Spatial mapping of the 'Gnangara macroinvertebrate model' - probability that the risk of overall wetland change is low, moderate or high for the wetland survey finish year (Table 4) on the Gnangara Mound.

This spatial risk assessment shows a high degree of accuracy based on actual events because it tracks known changes. Wetlands indicated by a high degree of risk have dried in recent times. To project to 2030 requires knowledge of how change in groundwater levels relate to the input variables of surface water level change and dynamics. An attempt was made to compare changes in groundwater level projected by SWSY PRAMS and changes in lake levels, using the existing relationship between the lake water depth and groundwater level in bores near each wetland. At time of writing this information had produced some valuable regressions but there was insufficient time to use it in water level projections for wetlands on the Gngangara Mound for 2030. This highlights the necessity to understand groundwater-surface water interactions of GDEs for predictive capacity.

Implications of risk: combination of the ‘wetland risk’ and ‘wetland conservation value’ for the ‘Gngangara macroinvertebrate model’

Combining the results of the ‘Gngangara macroinvertebrate model’ with the ‘indicative wetland value’ mapping illustrates where the wetland survey sites coincide with high-value wetlands (Figure 27). The probability that wetland risk is moderate to high is significant in most of the high value wetlands, indicating severe consequences of change over the survey period.

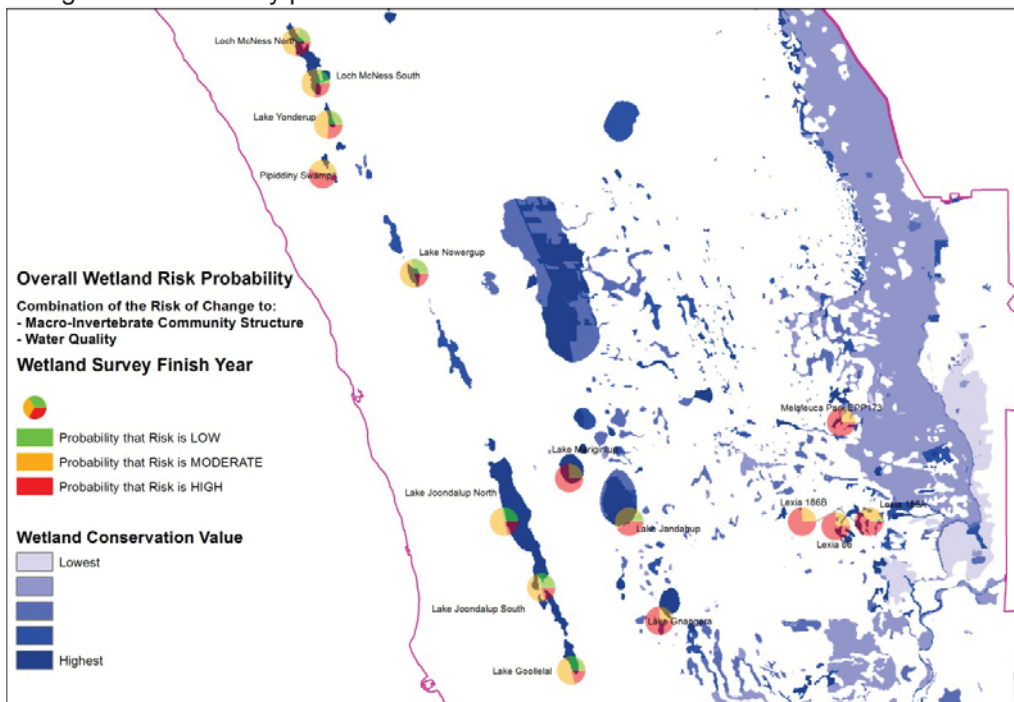


Figure 27: Combination of the wetland risk and wetland conservation value for the ‘Gngangara Macroinvertebrate Model’.

3.1.7 Step 5: Manage Risk

CASE STUDY 1

Development of the framework

GDEs on the Gngangara Groundwater System have been under threat from a drying climate, and compounding stressors such as groundwater abstraction and other

landuses, for over three decades. The management of these ecosystems depends on the ability to predict responses and assess risks posed by projected climate and landuse scenarios. Prediction for climate change adaptation and adaptive management requires an approach that enables resource managers to adapt their conservation strategies to minimise the impacts from other - controllable - stressors to these ecosystems.

The aim of this research was to develop a methodology for predicting risks to groundwater dependent wetland ecosystems in a drying climate. The methodology applies specifically to data-rich situations where the objectives are to (1) facilitate adaptation to climate change and climate change related factors, and (2) to define risk in terms of ecosystem function. We have demonstrated the approach on a case study from the Gngangara Groundwater System, where aquatic macroinvertebrates and littoral/supra-littoral vegetation were used as surrogates for wetland ecosystem function. Because the focus of this research was prediction and risk assessment, the analytical steps of the methodology are presented within the context of a risk assessment framework.

Key features of the methodology are:

1. Applies to data rich situations (e.g. long-term monitoring data are available),
2. Incorporates expert knowledge to estimate the complexities that could not be derived from the quantitative data;
3. Uses a combination of multivariate statistical methods (therefore requires some statistical expertise) in a sequence of steps that fit into the context of a risk assessment framework;
4. Designed to facilitate adaptation to climate change by setting targeted objectives;
5. Uses the concept of species functional groups, therefore:
 - the methodology becomes geographically transferable
 - functional responses can be directly related to adaptation
 - functional characteristics can be directly linked with the hazard
 - simplifies and contextualises biotic response to reflect the functional relationship with the hazard that is driving the change, which facilitates the prediction of future impacts.
6. Can be used to spatially represent risk by mapping different climate and landuse scenarios and associated risk.

Managing Risk on the Gngangara Groundwater System

Decision support has been provided throughout this framework to enable the appropriate type and spatial extent of interventions to be decided upon to strengthen the resistance and resilience of wetlands to the ongoing threat of climate change.

These are outlined below:

Step 1: Identify the hazard

Identification of the causes of the hazard enables management to address human use issues that may improve the condition of GDEs and their resilience to the drying effects of climate change. This has already been taken into consideration by State Government authorities in view of groundwater extraction for human use (e.g. decisions on the extraction limits for different bore locations - Department of Water, Western Australian Government), investigation for different sources of water (e.g. desalination plants – Department of Water, Western Australian Government and Water Corporation) and land use and management (e.g. thinning and removal of pine

plantations - Department of Environment and Conservation, Western Australian Government).

Step 2: Identify hydrological risk

Magnitude of groundwater decline

Comparison of the different scenarios (Figure 9) indicates that all except Scenario CDry will have rising groundwater tables in some areas of the Gnamptara Mound. All scenarios agree on the location of the areas that are likely to provide the best refuges or sites for wetland conservation or restoration into the future. This can inform future conservation estate, water extraction or land use planning. Areas where GDEs are at risk (through declining groundwater levels), are consistently in the north-east corner of the mound, with areas also in the south-west under some scenarios. This information needs to be interpreted together with the Asset Identification, outlined in Part 1, to inform decision-making.

Spatial impact assessment maps

The spatial impact assessment maps (Figure 10) provide the most detailed spatial indication of hydrological risk to GDEs because they address the zone in which groundwater interacts with surface water expression in GDEs. The maps identify specific areas in which GDEs may be expected to suffer the greatest impacts from groundwater decline. This provides a higher resolution than that provided by magnitude of groundwater decline but can be used in the same way to better inform future conservation estate, water extraction or land use planning. While a simple indicator, care must be taken that the assumptions on which it is based are valid.

Rate of groundwater decline

Spatial distribution of rate (Figure 11) illustrates the greatest differences between the scenarios, with a high rate (and inherent risk to GDEs) in the worst-case scenario (C dry). This provides hope that rates of decline in the Gnamptara Groundwater System may be within the scope of adaptation both for managers and biota, and that management measures put in place may have a good chance of success. The rate is not extreme under any scenario, although this is a simplification of how actual rates of decline may occur. It does not account for extreme events.

Dynamics of hydrological change

Availability of data meant that only historical and recent changes in hydrological variability could be included in the assessment, it was not possible to project future scenarios. However, monitoring the frequency and duration of dry periods has indicated the level of stress already evident in the GDEs and provides a baseline. It is known that future projections will result in greater severity of the current levels of hydrological stress.

Step 3: Assess effects

Detailed information and development of conceptual models for assessing the effects of drying on wetland water quality, vegetation and macroinvertebrate communities is provided in SD2: Sommer *et al.* (2013). Identification of the key parameters driving water quality, vegetation and macroinvertebrate response to drying provides valuable information when designing monitoring regimes and developing management plans. The conceptual models describe the linkages between the ecosystem and the landscape and provides an excellent communication tool for explaining the focus of management plans. Conceptual models can be integrated with information on the causes of the key drivers to provide a map of potential management interventions (e.g. Figure 28).

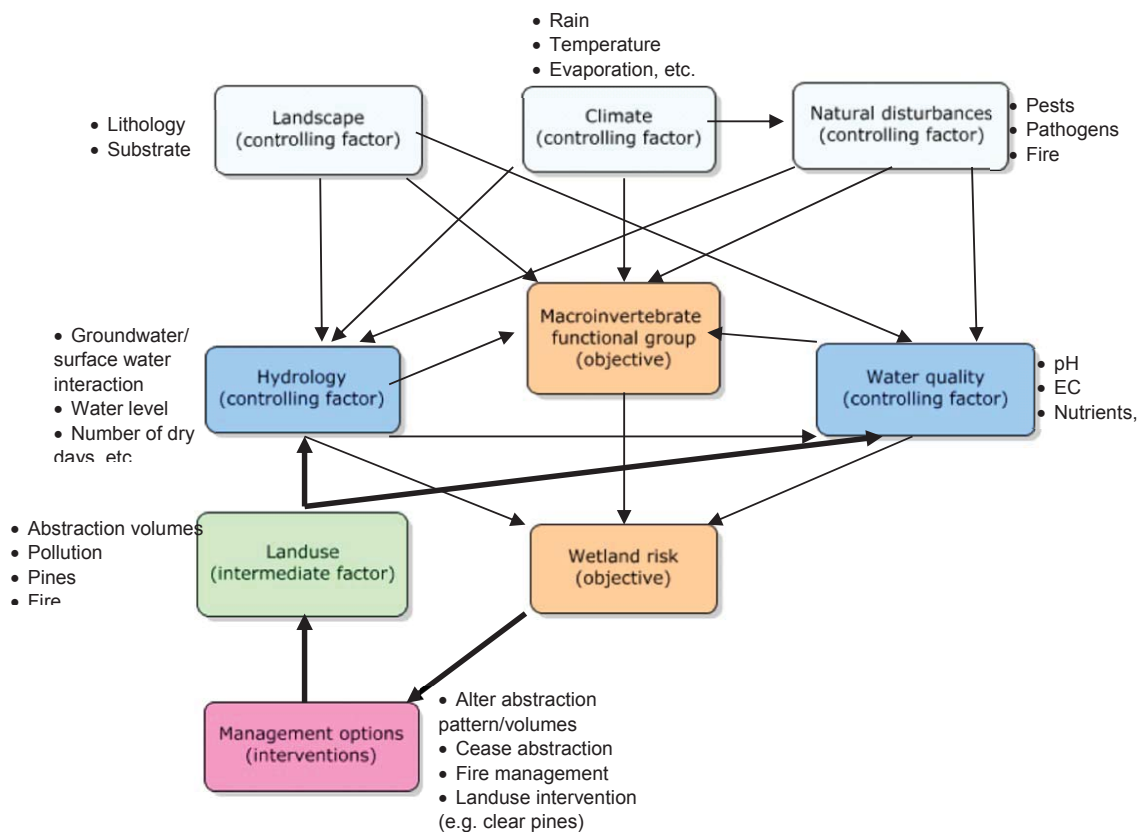


Figure 28: Conceptual model of the linkages between variables that were used in the construction of the BBN for the prediction of risk to wetland function based on changes in macroinvertebrate functional groups, hydrology and water quality. The bold arrows show the main connections between management options (interventions) and the resultant wetland risk. Controlling factors in the light blue boxes are beyond the control of management agencies, but do control the environmental system. Controlling factors in the dark blue boxes are largely beyond the control of management agencies, however, they can be manipulated to a certain degree by adjusting landuses such as groundwater abstraction (in which case the variables become objectives). Landuse is an intermediate factor as it links the objectives and interventions.

Step 4: Characterise risk

Thresholds

The thresholds identified for the plant and macroinvertebrate communities define the boundaries in which the biota of the GDE is likely to survive or where community composition will change if the thresholds are exceeded. This information is valuable as targets for monitoring programs, management plans, to define acceptable limits of change and for use in conservation and restoration initiatives.

Bayesian Belief Networks

As outlined in the Guidelines for Use document Bayesian Belief Networks have a wide range of applications relevant to management of the Gngara Groundwater System.

They can be easily modified to show changes in probability of risk resulting from the interaction between climate, hydrology, water quality, biotic resources requirements and biotic response. This provides a transparent and interactive template for decision-making at a range of levels. It equally shows the impact of extracting water on biota as it does the capacity for biota to survive under different climate and/or land use scenarios.

Due to the short time frame (one year) to develop the framework, it was not possible to run through specific scenarios to illustrate how the BBNs will be used for decision support on the Gnangara Groundwater System. However with the plant, macroinvertebrate and whole ecosystem BBNs now created it will be possible with the modelling capacity of the regional groundwater models 'PRAMS' (Perth Regional Aquifer Modeling System (Vogwill, 2004; Xu, 2008), and the South West Sustainable Yields (SWSY) project (CSIRO 2009a,b) to develop a range of scenarios to test various groundwater extraction and climate change scenarios. For example, using the PRAMS model, the groundwater decline resulting from extracting 45 GL (or other volume) from the Gnangara Groundwater System can be estimated. This decline in groundwater can be plugged into the BBN and the probability of risk to the GDEs ascertained. Similar scenarios can be developed for reduced rainfall and groundwater recharge due to climate change.

Spatial modelling

Spatially overlapping risk and asset maps can be an excellent tool to identify the risk to sites of high conservation or other values (Figure 27). This can inform a range of potential adaption outcomes including passive management (such as using zoning to indicate values in land use planning) or active management (e.g. choosing which wetlands to restore through pumping water to maintain water levels). Water resource agencies could use such mapping to develop protocols to identify which bores should be used when, to ensure groundwater extraction retains appropriate water levels in areas of high value conservation or other assets. Agencies may use consequence mapping to make decisions on triage – where some wetlands may be abandoned and management resources focussed on others that have a greater probability of maintenance of function and biota.

CASE STUDY 2

The framework has informed a number of climate adaptation options to retain the diversity of amphibian reproductive types and species into the future. The key drivers identified can be used as management targets during monitoring to determine where and when certain reproductive guilds may be under stress. For example, if winter rainfall does not provide a hatching trigger, flooding of conservation wetlands may enable terrestrial eggs to hatch. As salinity was identified as an issue, the threshold of 8ppt can be used to monitor wetland and undertake reduction of salinisation through land use changes. Thresholds would provide valuable information for conservation of any threatened species into the future.

3.2 Blackwood River Case Study

(Key supporting documents: Blackwood River Case Study - SD4: Beatty *et al.* (2013), SD6: Bayesian Belief Networks - Speldewinde (2013), and Spatial Modelling - SD7: Neville (2013))

The Blackwood River had limited data available on the hydrogeology, hydrology, water quality (1998-present) and biota (2005-present) (Appendix 3). Regional groundwater

models 'SWAMS' (South West Aquifer Modeling System) were available to evaluate groundwater level declines (Vogwill, 2004; Xu, 2008), while climate change projections were available from the South West Sustainable Yields project (CSIRO 2009a,b).

PART 1: IDENTIFYING MANAGEMENT ISSUES AND THE NATURE OF THE ECOSYSTEM

3.2.1 Identifying management issues

South-western Australia has the highest rate of endemism in freshwater fishes of any Australian Drainage Division (82%) but they have undergone major range reductions and are imperilled through a number of stressors such as salinisation, riparian degradation, in-stream barriers, introduced fishes, flow and groundwater reductions, and climate change (e.g. Morgan *et al.* 1998, 2003, 2004; Morgan and Beatty 2006; Beatty *et al.* 2010, 2011; Morrongiello *et al.* 2011). Of the eleven freshwater fishes, two are listed as threatened under the Federal EPBC Act 1999 and those plus an additional two species are listed under the State Wildlife Conservation Act 1950.

The Blackwood River is the largest by discharge in south-western Australia and is secondarily salinised. As a result much of the fish fauna of its main channel and upper tributaries now being dominated by euryhaline species typically associated with estuarine environments in south-western Australia and the introduced species *Gambusia holbrooki* (Morgan *et al.* 2003; Beatty *et al.* 2011). However, a major zone of fresh groundwater intrusion from the Yarragadee Aquifer maintains permanent flow in the lower reaches of the main channel of the river. This has enabled continued occupancy by stenohaline, endemic freshwater fishes that no longer occupy the upper catchment. The groundwater inflow has been demonstrated to reduce salinity in the main channel of the river during baseflow when potamodromous freshwater fishes utilise it as a refuge when most tributaries contract or dry (Beatty *et al.* 2009, 2010). Furthermore, by maintaining riffle zones, the groundwater has been demonstrated to facilitate longitudinal movements of native freshwater fishes and also maintain permanent refuge habitat within two tributaries that house threatened freshwater fishes (Beatty *et al.* 2009, 2010).

For example, groundwater discharge during baseflow in the NCCARF project study area in the Blackwood River maintains habitat connectivity for the largest freshwater fish of the region, *Tandanus bostocki* and provides refuge habitat for the nationally endangered *Nannatherina balstoni*. Therefore, the groundwater in the study area in the Blackwood River plays a critical role in supporting remnant freshwater fish populations and there are clear links between hydrology, water quality and resource requirements of fish communities.

The importance of fresh groundwater in maintaining lentic and lotic refuge habitats during the naturally dry summer and autumn in the region has recently been recognised. This crucial input is now threatened by declining groundwater levels.

3.2.2 The nature of the ecosystem

The Blackwood River floodplain and tributaries contain a range of groundwater dependent ecosystems within the river catchment, particularly in an area about 25 km west of Nannup where the Yarragadee Formation is close to or at the land surface. Within the study area, groundwater intrusion maintains permanent flow in two tributaries (Milyeannup Brook and Poison Gully) whereas others in the catchment either annually cease to flow or dry completely during baseflow (Strategen 2006; Del

Borrello 2008) (Figure 29). The contribution of freshwater into the tributaries reduces salinities in the river, that are elevated in the estuarine part of the river downstream by salt water intrusion from the ocean and upstream by dryland salinisation in the upper agricultural catchment. This small zone of permanent freshwater provides a crucial habitat for freshwater fish in the area.

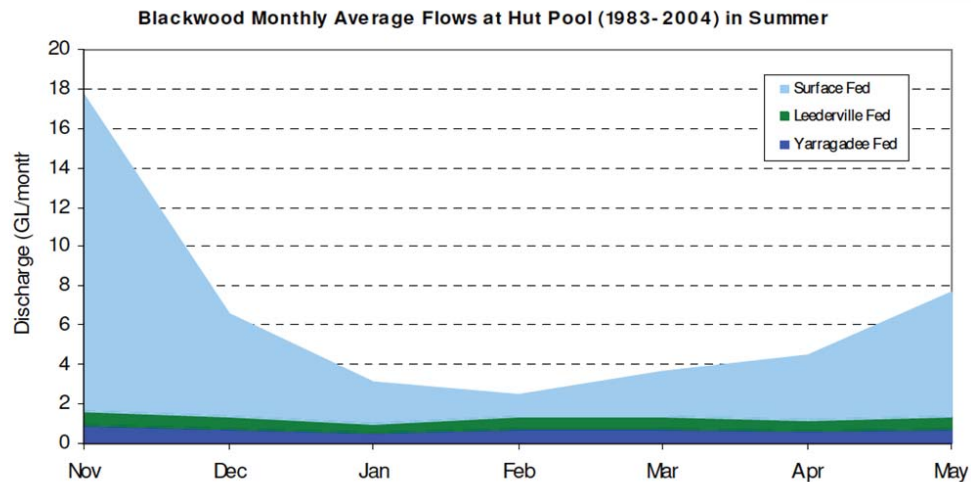


Figure 29: Discharge into Hut Pool in the Blackwood River groundwater intrusion zone showing contributions from the Leederville and Yarragadee aquifers (Strategen 2005). Figure is courtesy of Strategen Environmental Consultants and the Water Corporation.

Although the Yarragadee Aquifer contributes only ~3% of the ~650 GLyr⁻¹ of the annual discharge at Hut Pool on the Blackwood River (Department of Water database, long term average (1955- 2005) (Peter Muirden pers comm. March 27, 2013), during dry months its groundwater along with that of the Leederville Aquifer can contribute to between 30-100% of the discharge (Strategen 2005). This significantly reduces the salinity in the river (Beatty *et al.* 2010). Figure 30 shows the results of single measurements of flows at many locations along the lower reach of the Blackwood River. The measurements were taken once per year at each site over a period of three years. It shows a zone 40-120km upstream of Molloy Island where input from groundwater from both the Yarragadee and Leederville Aquifers increases the rate of flow of the Blackwood River. This zone was the study area of this project.

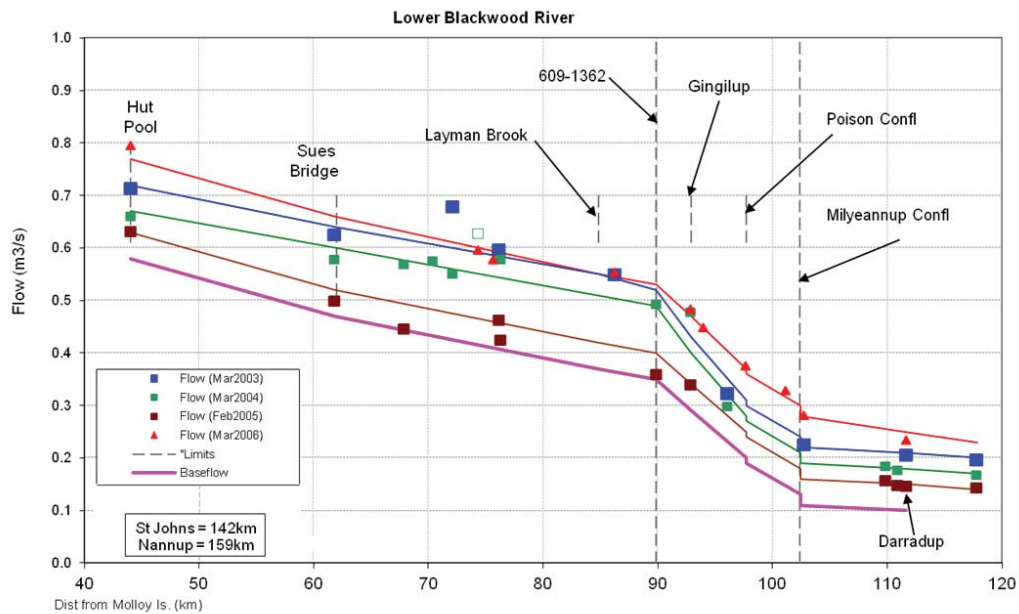


Figure 30: Consecutive annual baseflow longitudinal discharge traces in the main channel of the Blackwood River in March (2003-2006). N.B. The locations of several key tributaries and riffle sites are indicated, Limits are the key breaks where different aquifers contribute. Of particular note the major vertical limit break at Milyeannup confluence which delineates the upstream limit of Yarragadee Aquifer contribution to baseflow. Note: River flows from right to left in this diagram. Figure is courtesy of the Department of Water, Government of Western Australia.

Spatial boundaries

The Yarragadee intrusion zone of the Blackwood River forms an area of 400 km² in the extreme south-west corner of Western Australia (34.1081°S, 115.5661°E; 34.0421°S, 115.6025°E) (Figure 31).

Assets

The whole study area is a high conservation asset.

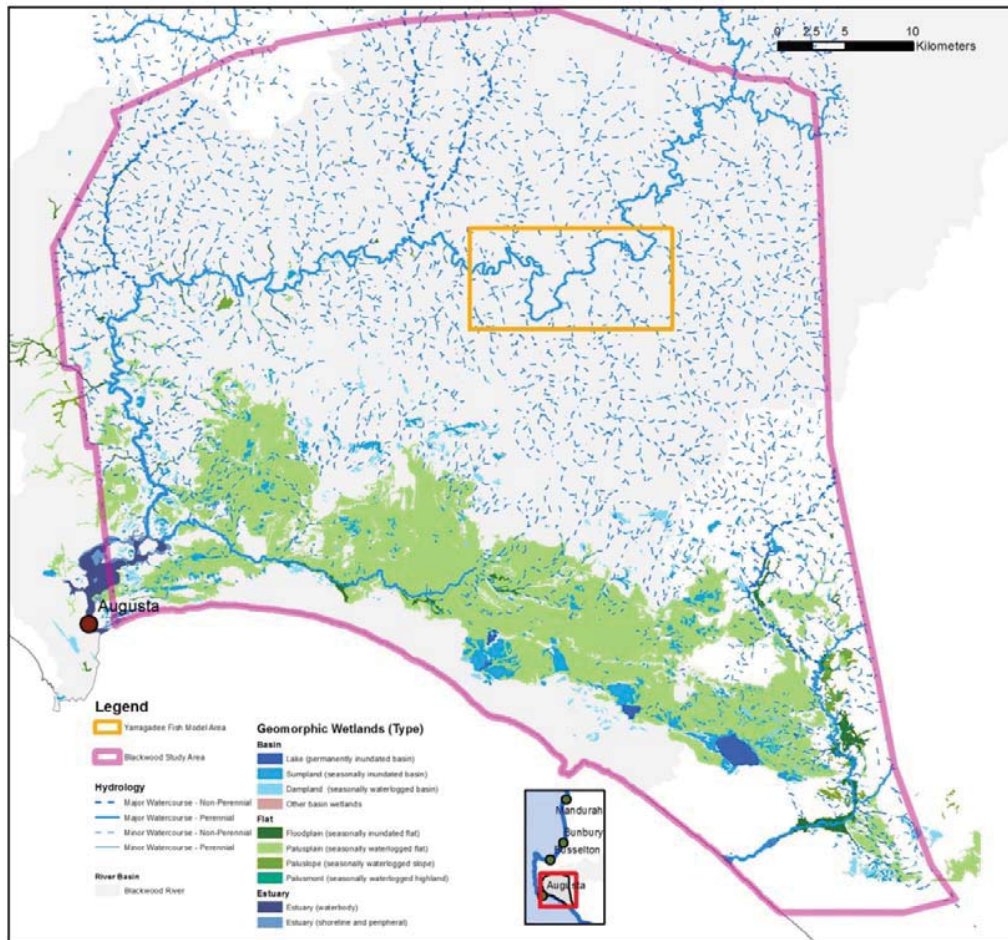


Figure 31: The spatial boundaries of the Blackwood River study area.

PART 2: THE RISK ASSESSMENT AND DECISION-MAKING FRAMEWORK

3.2.3 Step 1: Identify the Hazard

Step 1.1: Identify the hazard

Primary hazard: The primary hazard is groundwater decline due to reduction in rainfall, land use and private groundwater extraction. The reason this is a hazard is because the main channel of the Blackwood River is salinised due to clearing for agriculture in the headwaters of the catchment. The freshwater intrusion provided during the dry summer month maintains freshwater tributaries and reduction of salinity in the main channel. Without this freshwater input a suite of endemic freshwater fish would be unable to survive.

Secondary hazards: Reduction in groundwater intrusion secondarily results in reduced flow, drying of the river to pools resulting in a lack of connectivity, increased temperature, evapoconcentration of nutrients and salts, decline in dissolved oxygen concentration and exposure to exotic fish species.

Step 1.2: Define temporal boundaries

The study was constrained to three discrete time periods due to the availability of data (Appendix 3, Tables 1, 2 and 3) and the projections by the South West Sustainable Yields Project (CSIRO 2009a). Dates for hydrological data (below) were slightly different for ecological data – earliest time frame was from 1996 (Appendix 3).

1. Historical (1975-2007)
2. Recent climate (1997-2007)
3. Projections to 2030 - a range of scenarios

The projections to 2030 included a number of climate change and development scenarios:

- Scenario A - Historical climate: assumed the climate of 1975-2007 continued until 2030
- Scenario B - Recent climate: assumed the climate of 1997-2007 continued until 2030
- Scenario C: Future 1 - 2030 climate change and current development (Mid, Wet and Dry Scenarios)
- Scenario D: Future 2 - 2030 climate change and future development

The Future Climate (Scenario C, Scenario D) used 15 Global Climate Models (GCMs) to modify the Historical Climate to produce wet, median, dry and future development 2030 climate scenarios. More details on the above five climate scenarios and the impact on surface water resources, groundwater dependent ecosystems and divertible yields can be found in CSIRO (2009a, b), Silberstein *et al.* (2012), Ali *et al.* (2012a), Barron *et al.* (2012) and McFarlane *et al.* (2012).

Step 1.3: Determine the cause of the hazard

Clearing for agriculture in the headwaters of the Blackwood River has caused dryland salinisation (rising groundwater due to removal of deep rooted trees bringing salts stored in the soil profile to the surface). This has resulted in increased salinity and increased surface water contribution into the Yarragadee intrusion zone of the Blackwood River, particularly in the main channel.

The drying climate over the last 30 years has resulted in a 15% decline in rainfall and a 55% decline in runoff (Kauhanen *et al.* 2011). The resultant decline in ground water tables has reduced the freshwater input crucial to the last remaining refuges for freshwater fish in this region. In summer, reduced flow and groundwater intrusion results in the river drying to pools where increased salinity, lack of connectivity, heating and reduction in dissolved oxygen concentration are detrimental to the survival of fish populations.

3.2.4 Step 2: Exposure and Vulnerability

Step 2.1 Determine the spatial and temporal change

Change in groundwater levels was mapped using Geographical Information Systems to give a spatial distribution. The scenarios mapped were based on discrete time periods from the temporal boundaries identified in Step 1.3 above. Due to inherent errors involved in modelling of groundwater levels using the South West Aquifer Modelling System (SWAMS) and its combination with climate change projections, this resulted in errors in projected groundwater levels of up to 3m. So these projected groundwater levels were taken to represent possible scenarios rather than accurate estimations of future levels.

Decision-making would best be served by capturing the direction (decline or rise) of groundwater level change, the magnitude and rate of groundwater level change, as the most useful indicators of hydrological change. Examples and use of different techniques to assess the extent of spatial and temporal change are described below.

Spatial and temporal change example 1: Magnitude of groundwater decline

The magnitude of groundwater change for 2030 for each of the climate change and land use scenarios is shown for the Blackwood River study area (Figure 32). Whilst these maps are produced using the raw output from the SWAMS models (rather than classified), they are identical to the SWSY mapping (CSIRO 2009a). Comparison of the different scenarios indicates that while all of them show regions of rising water table in the catchment, it does not occur in the zone of interest: the Yarragadee intrusion zone of the Blackwood River (c.f. Figure 32).

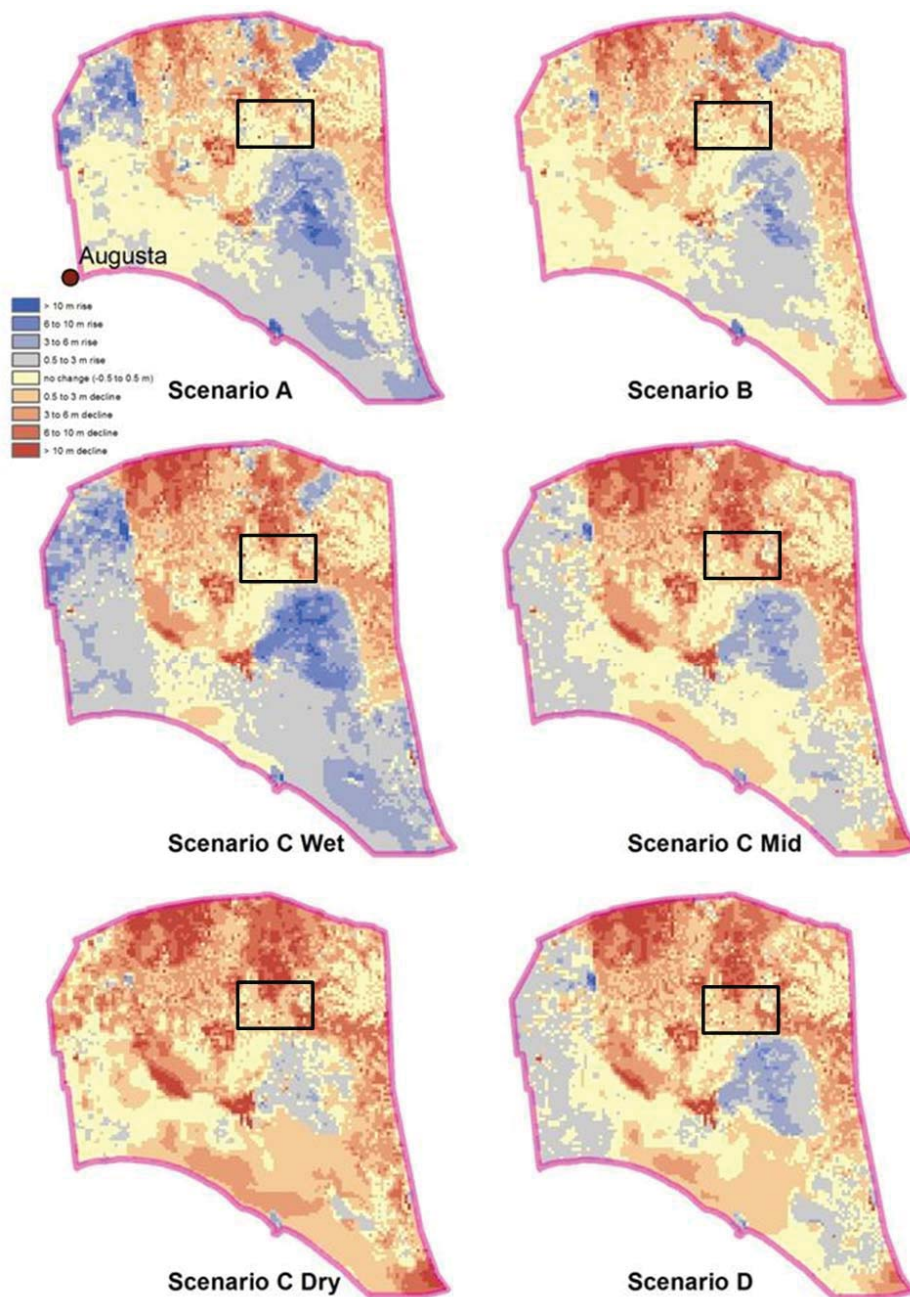


Figure 32: The magnitude of groundwater decline was projected into 2030 under a range of climate change and land use scenarios for the Blackwood River study area. These projections were made in the South West Sustainable Yields Project and are based on the SWAMS groundwater model. Scenario A: 2030 projections based on extension of the 1975-2007 period and current development (i.e. abstraction and landuse unless known to change). Scenario B: 2030 projections based on extension of the 1997-2007 record and current development. Scenario C: 2030 projections using wet, median (mid) and dry GCM climate change scenarios and current development. Scenario D: Projections into 2030 under the CMid climate change scenario and abstraction taken out to the maximum limit (CSIRO 2009a). The black squares show approximate location of the study area.

Spatial and temporal change example 2: Rate of groundwater decline

While the magnitude of groundwater decline is important, so too is the rate of decline. Slow changes provide time for the ecosystem to adapt, while rapid decline may exceed the plants' and animals' ability to respond and result in the death of organisms. Rapid water level decline can also result in rapid changes to water quality e.g. increasing temperature, concentration of salts.

Rate of decline can simplistically be measured by dividing the magnitude of decline by the time period it is observed or projected. This was undertaken for the Blackwood River study area where the magnitude of groundwater decline from the CSIRO South West Sustainable Yields Project, based on the SWAMS groundwater model (CSIRO 2009a), was divided by the number of years in each climate change and land use scenario to determine the rate (m/year) (Figure 33). This is a simplification of change over time, as a decline in the water table level may actually increase during drier periods and slow or even reverse for periods of wet years – even though the long-term trend is one of decline.

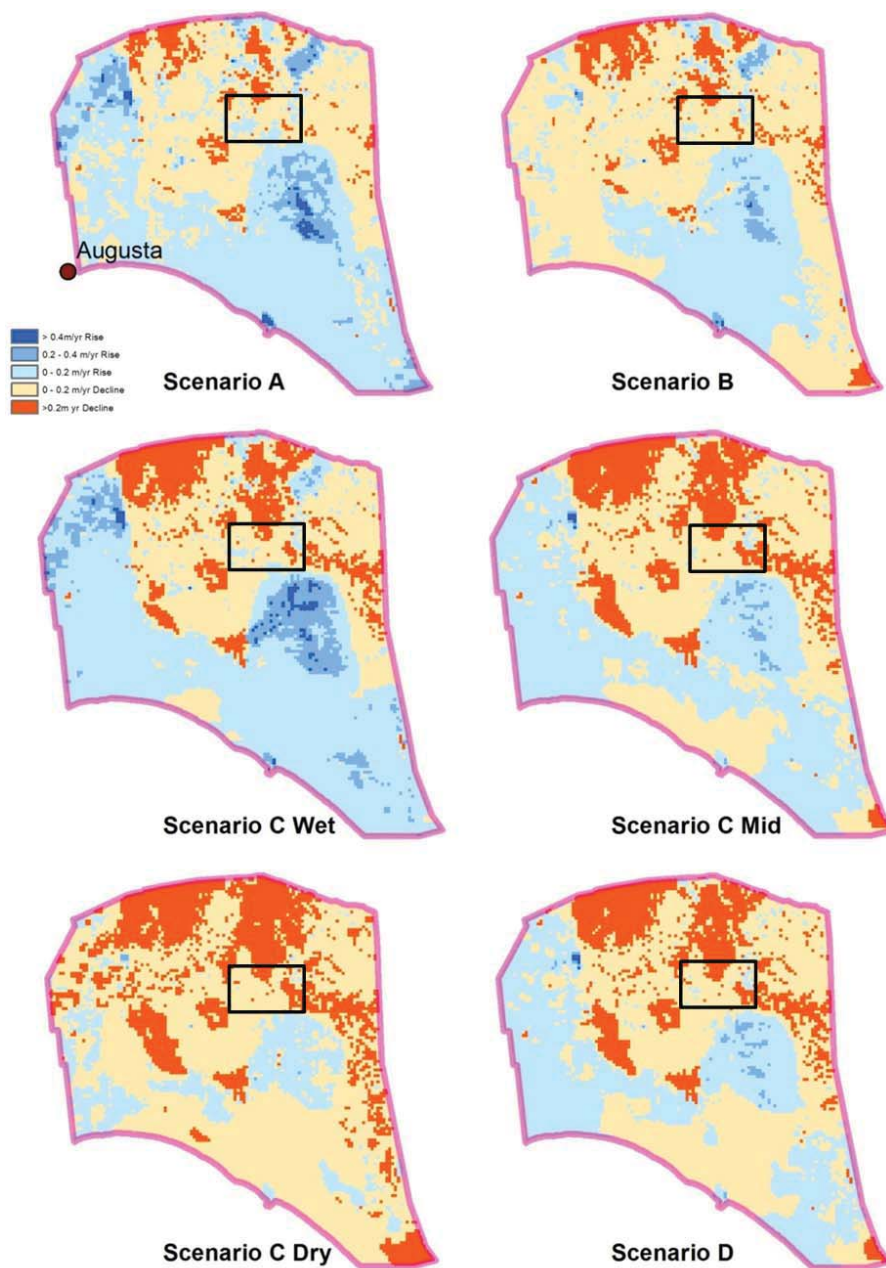


Figure 33: The rate of groundwater decline (m/year) was projected into 2030 under a range of climate change and land use scenarios for the Blackwood River study area, Western Australia. These projections were made in the South West Sustainable Yields Project and are based on the SWAMS groundwater model. Scenario A: 2030 projections based on extension of the 1975-2007 period and current development (i.e. abstraction and land use unless known to change). Scenario B: 2030 projections based on extension of the 1997-2007 record and current development. Scenario C: 2030 projections using wet, median (mid) and dry GCM climate change scenarios and current development. Scenario D: Projections into 2030 under the CMid climate change scenario and abstraction taken out to the maximum limit (CSIRO 2009a). The black squares show approximate location of the study area.

Step 2.2: Accommodate dynamics of hydrological change

This case study focused on the period of groundwater input that was key to fish survival: the baseflow input during summer. In this way the dynamics of hydrological change were already accounted for, as this was the low flow period in the hydrograph.

3.2.5 Step 3: Assess Effects

Step 3.1 Collate the data

The study reviewed all available literature on the distribution and water quality at presence sites of six native freshwater fish species known to be present in the Blackwood River study area; i.e. *Galaxias occidentalis*, *Galaxiella munda*, *Nannoperca vittata*, *Nannatherina balstoni*, *Bostockia porosa* and *Tandanus bostocki*, and the introduced *Gambusia holbrooki*. This review included published papers, unpublished reports, and the unpublished raw data of the authors. From this review, sampling points (n = 1098) were extracted that had recorded both the presence of a relevant species and water quality variables that included temperature, pH, conductivity, dissolved oxygen, turbidity, TN and TP. A compilation of the distribution of each species with hydrological and physiochemical variables enabled experts to identify the key drivers to be incorporated into the conceptual model and Bayesian Belief Networks.

Step 3.2: Develop the conceptual model

Based on information from the literature and expert opinion within the research team, a conceptual model was developed to support a Bayesian Belief Network (Figure 34). River connectivity and water quality variables (salinity, temperature and dissolved oxygen) were identified as the key drivers of survival for fish populations. These drivers were found to be negatively influenced by declining groundwater levels. See SD 4 Beatty *et al.* 2013 for details of the methodology.

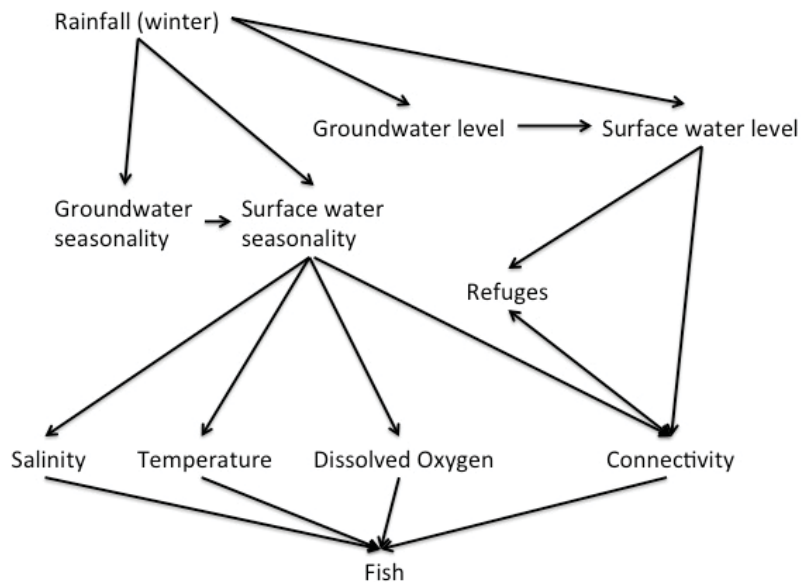


Figure 34: Conceptual model of the Blackwood River GDE.

3.2.6 Step 4: Characterise Risk

Step 4.1: Determine thresholds

Whilst some information on minimum pool and riffle passage depth requirements of several species existed, opinion of the authors was also used to set thresholds for water quality parameters for each species. For each species, the proportion of sites of occupancy, and the median, mean, maximum, minimum, 5th, 25th, 75th and 95th percentiles were calculated across their sites in south-western Australia. An example of this is shown in Figure 35 for two species *Galaxias occidentalis* and *Nannatherina balstoni*.

The analyses revealed significant differences in several environmental variables between the sites occupied across the range of the seven freshwater fishes. Temperature and conductivity varied significantly among species with the introduced *G. holbrooki* occupying significantly warmer habitats than all other species. The more common species *G. occidentalis*, *N. vittata* and *B. porosa* occupied sites with similar conductivities, temperatures, and pH. The two threatened endemic fishes, i.e. *G. munda* and *N. balstoni* occupied the coolest habitats with the former species also occupying the freshest sites. The difference between the two galaxiids were notable with *G. occidentalis* occupying sites that were on average 2.7°C warmer and ~1784 $\mu\text{S}\cdot\text{cm}^{-1}$ higher in conductivity than *G. munda*. Difference also existed amongst the percichthyids with the temperature at sites occupied by *N. balstoni* >~1.1 °C cooler than *N. vittata* or *B. porosa*. The largest native species of the region *T. bostocki* that prefers larger river systems occupied sites that were of relatively high conductivity compared to other native species. Probably due to a paucity of data points, the dissolved oxygen, turbidity, total nitrogen and total phosphorus did not reveal significant differences between species.

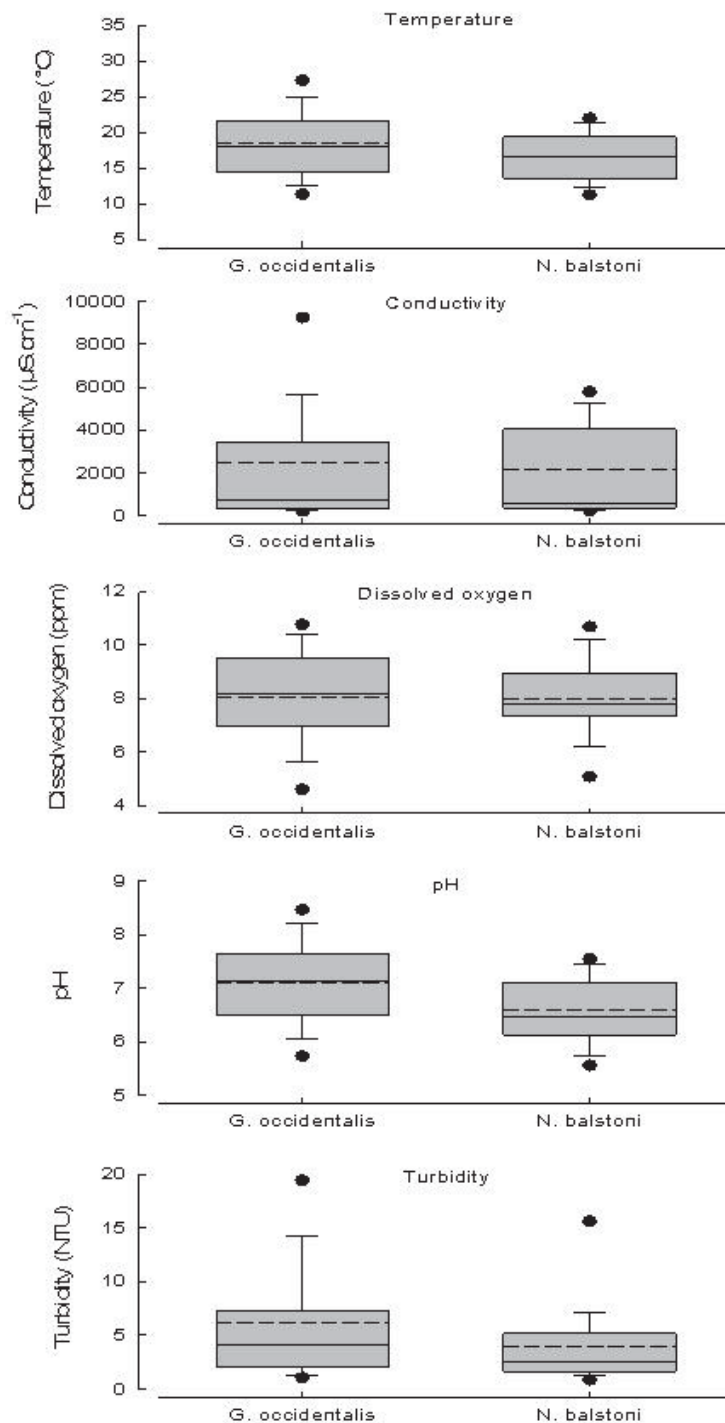


Figure 35: The mean (dotted lines), median (solid line), and 25th, 75th (boxes) 5th and 95th (dots) of the key water quality parameters within the habitats occupied by *Galaxias occidentalis* and *Nannatherina balstoni* in south-western Australia. See SD4 Beatty *et al.* 2013 for more details.

Step 4.2 Determining Probabilities of Risk: Bayesian Belief Networks

The Blackwood study site was not as data rich as the Gngangara Mound study site, therefore a mixture of data and expert opinion was utilized to develop the BBN for fish in the Blackwood River.

The relationship between groundwater levels (GWL), surface water levels (SWL) and water quality was determined only for the summer months when groundwater inflow was the main contributor to surface water levels. A regression was derived between groundwater levels (GWL) and surface water levels (SWL) using surface water level data from Department of Water gauging station on the Blackwood River and the SWAMS groundwater levels. The relationship between surface water levels and water quality variables (temperature, salinity, dissolved oxygen and pH) were determined using water quality and surface water level data from a Department of Water gauging station.

The thresholds for the environmental variables were based on data but were derived by expert opinion. Three possible outcomes were defined for the threshold, population persist, population likely decline and population extreme decline. For an outcome to fall into the population extreme decline category, one or more of the environmental thresholds had to fall outside of the known range for that species. The exceptions being if the salinity fell below recorded values or if Dissolved Oxygen (DO) was above recorded values it was considered to be within the species thresholds. Population likely decline was defined as three or more of the environmental variables being recorded in the 0-25 percentile or 75-100 percentile. If all of the variables fell in the 25-75 percentile the outcome was defined as population persist (the exceptions being salinity and DO where salinity was <75th percentile and DO being >25th percentile, it was considered that salinity below recorded values or DO levels above recorded levels were within the species thresholds).

A complete model of all fish species is shown in Figure 36. However when developing an overall index of fish health in the Blackwood study area, two indicator species were chosen to contribute to the index (*Nannatherina balstoni* and *Galaxias occidentalis*) (see Figure 9, SD 6 Speldewinde 2013). *N.balstoni* only occurs over a narrow range of environmental conditions, while *G.occidentalis* occurs over a wide range of environmental conditions. A measure of fish community health node was therefore constructed in the model based on the characteristics of these two species (the thresholds of the remaining species lie between the two extremes of *N.balstoni* and *G.occidentalis*). If both species were found to persist in the system, it was defined as 100% healthy, if both species were classified as being in severe decline then the system was defined as 100% unhealthy. Various combinations in-between these two extremes were given probabilities by the expert panel. For more detail on this methodology see SD4 Beatty *et al.* (2013) and SD6 Speldewinde (2013).

This technique provides a valuable way to reduce the complexity of outcomes by carefully choosing indicator species (in this case an index based on the combined characteristics of a robust and a vulnerable species). The selection of appropriate indicators can serve to summarise numerous responses of different taxa and hence simplify and strengthen the development of models for decision-making.

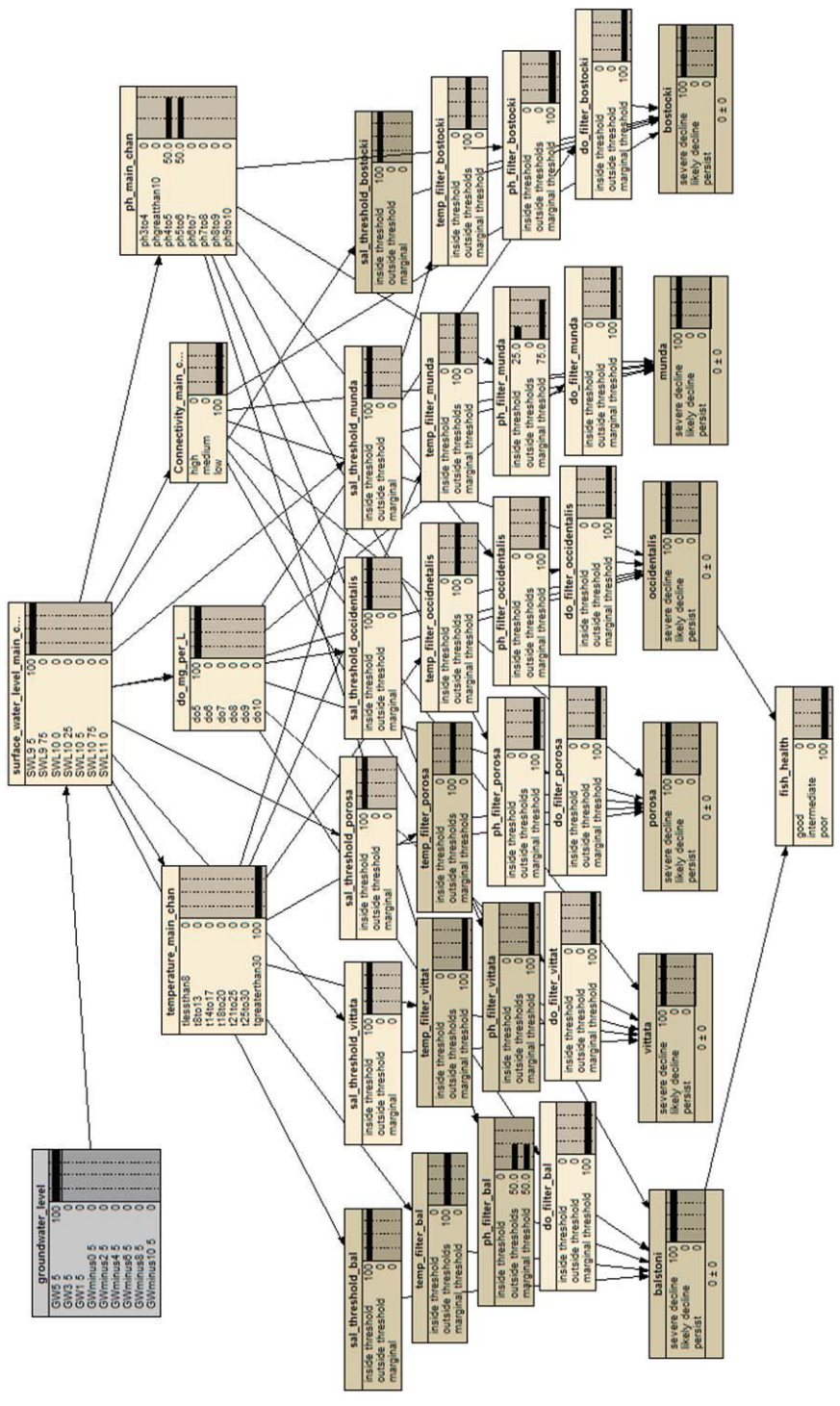


Figure 36: Complete BBN for Blackwood River incorporating all fish species and index of fish health. Note the model consists of six basic water parameter units repeated for each species specific threshold.

Step 4.3: Spatial mapping of risk using GIS

In its spatial form, the BBN for fish in the Blackwood River is referred to as the 'Blackwood River fish health model'. The model uses a single data input – depth to water table (m) in March, as this is the critical time when groundwater height influences water quality in the river. For the spatial modelling, the data is sourced from the SWAMS model, which reports projected water table heights in meters for each month of the year at 2030. Data was extracted for each of the 6 CSIRO Scenarios at the year 2030. Data extraction was carried out for each model point in the fish model area – a total of 191 points (Figure 37).

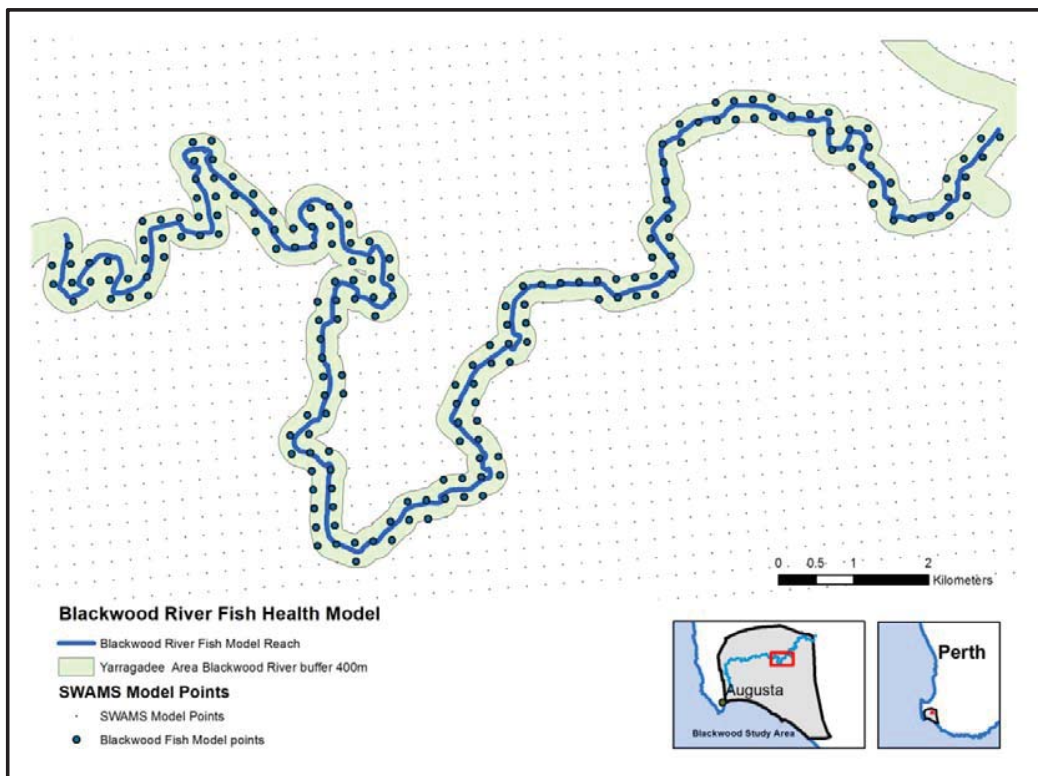


Figure 37 - Blackwood River fish health model reporting points

These values for each South West Sustainable Yields Scenario were sourced directly from the point files as exported from SWAMS, and joined to X,Y coordinates as a new point file. The values were extracted from the GIS, classified according to the BBN model categories, and comprised as case files for Netica.

We ran each case file through Netica using the function cases - process cases. A Netica control file was written to export the required findings: a finding for fish health (good, intermediate or poor), and findings for each of the 6 species involved (persist, likely decline or severe decline). The results were returned to ArcMap and displayed. For more detail on this methodology refer to SD 7: Neville (2013).

The final fish health index combines the results from two indicator fish, *Galaxias occidentalis* and *Nannatherina balstoni* (Figure 38). The 'composite fish health index' was created as follows:

- GOOD: both species persist
(all parameters within 75% range for both species)
- INTERMEDIATE: combinations of species persistence and decline
(1 or 2 parameters out of 75% range for one species)
- POOR: both species experience severe decline
(3 parameters out of acceptable range or 1 out of range)

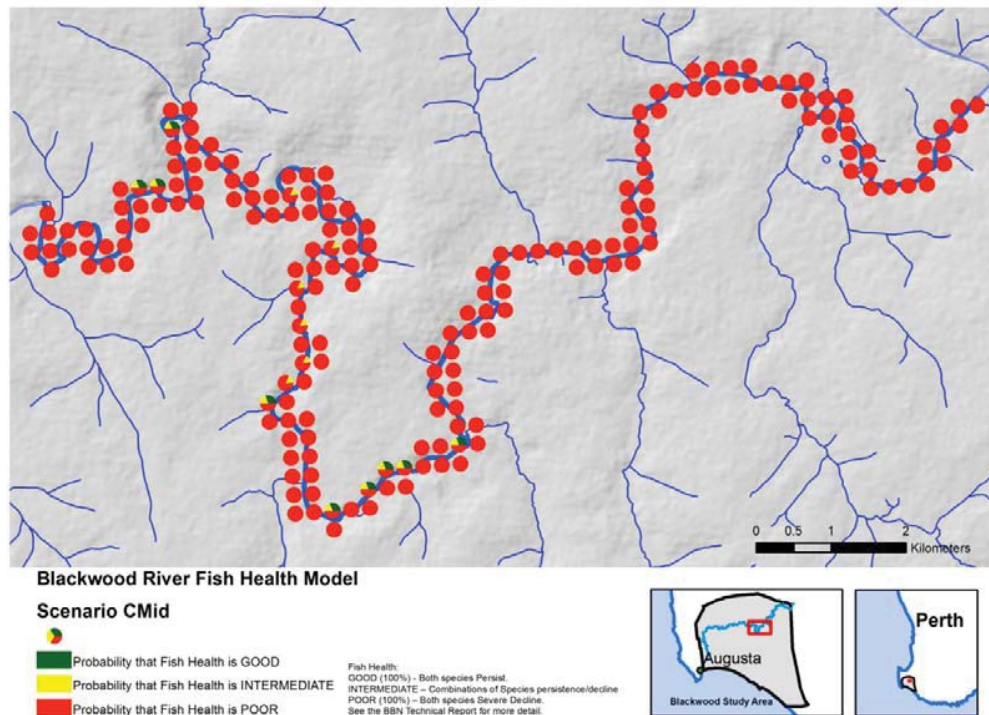


Figure 38: Composite fish health index - probability that the risk to the freshwater fish species *Galaxias occidentalis* and *Nannatherina balstoni* is low, moderate or high for the Blackwood River study area.

3.2.7 Step 5: Risk Management

The maps for exposure and vulnerability indicated a high risk of groundwater decline in the region. Development of conceptual models, thresholds and BBNs has provided a closer examination of the interaction between the variables to provide a spatial risk map of the composite fish index. Once again only a single example is provided for Scenario CMid (other scenarios can be found in SD7: Neville (2013)). While this paints a bleak picture, it also provides key locations where freshwater fish may find refuge. Interpretation of these maps by the fish research team, indicate that these points largely coincide with sites where freshwater tributaries are currently sustained by groundwater baseflow. Indications are that if these sites can remain in good environmental condition through appropriate management, the species may be able to survive. These provide priority areas for conservation, the potential to improve the sites through reducing other environmental stressors (e.g. salinity) and perhaps increasing connectivity through restoration or other management initiatives.

3.3 Leeuwin Naturaliste Ridge Caves

(Key supporting documents: SD5: Chilcott (2013), SD6: Speldewinde (2013), and SD7: Neville (2013))

The Leeuwin Naturaliste Ridge Caves had data available on the groundwater quantity, water quality and stygofaunal assemblage from 1997-2012. This case study was chosen due to the presence of several Threatened Ecological Communities (TEC's) under the *Environmental Protection and Biodiversity Act (1999)* within the study area.

PART 1: IDENTIFYING MANAGEMENT ISSUES AND THE NATURE OF THE ECOSYSTEM

3.3.1 Identifying Management Issues

Southwest Western Australia is known as a hotspot for stygofauna in groundwater dependent ecosystems (Barron *et al.*, 2012), but these are imperilled by the projected effects of climate change through diminishing rainfall, and/or land use and management (Eberhard 2002, Eberhard 2004, Jasinska 1997). Since 1975, a climate change induced rainfall decline in southwest Western Australia has limited the available groundwater supply (Danielopol *et al.*, 2003, Skurray *et al.*, 2011) to cave catchments. The situation in Leeuwin Naturaliste Ridge is so dire that after research efforts into the cave stygofauna began in 1993, several stygofauna communities became listed under the *Environmental Protection and Biodiversity Act (1999)* as Threatened Ecological Communities (TEC's) (Eberhard 2002). Water levels in cave pools on which these threatened communities depend have declined by up to 2m, severely decreasing the spatial extent and depth of the pools, many of which are now dry.

3.3.2 The Nature of the Ecosystem

The Leeuwin Naturaliste Ridge is an aeolian (windblown) limestone palaeodune created in the Plio-Pleistocene and Holocene (Eberhard 2002). Jewel, Easter and Labyrinth caves are subsystems of one hydrologically connected system of the Augusta Water Table Caves in the Leeuwin Naturaliste Ridge (Eberhard 2002). Caves are underlain by anoxic clays and granite-gneiss basement rocks, so once the water table reaches the base of the cave there is no habitat remaining for stygofauna (Figure 39).

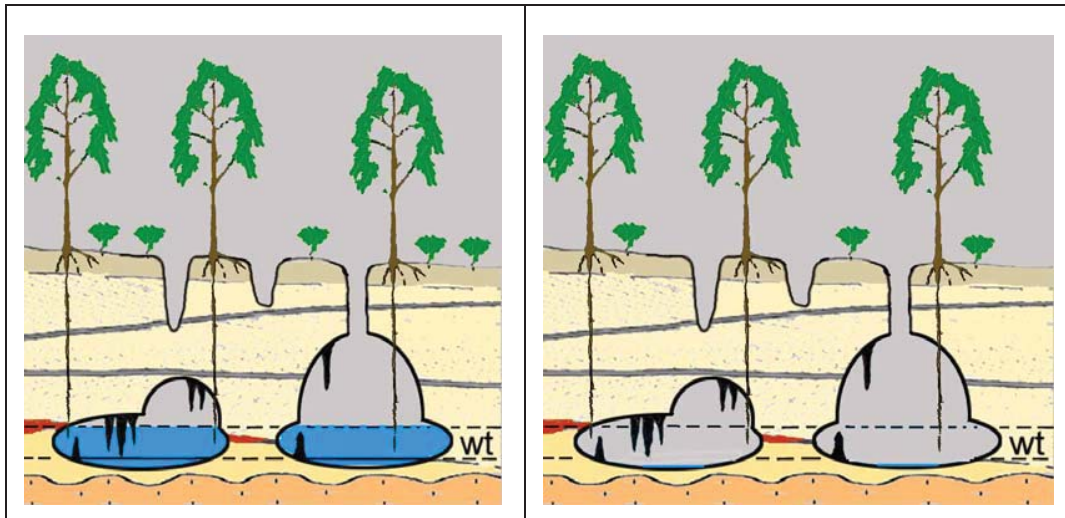


Figure 39: Conceptual hydrogeologic profile of the Leeuwin Naturaliste Ridge Caves indicating eco-hydrologic “end point states” (blue): high GW levels versus low GW levels. The caves are developed in aeolian dune calcarenites with interbedded palaeosols (grey – red bands) overlying relatively impermeable granit-gneiss basement rocks. Adapted from Eberhard 2004.

Spatial boundaries

The Leeuwin Naturaliste Ridge is located between 33°31’S and 34°23’S latitude, and 114°59’E and 115°15’E longitude (Figure 40).

Assets

The whole study area is high conservation asset.

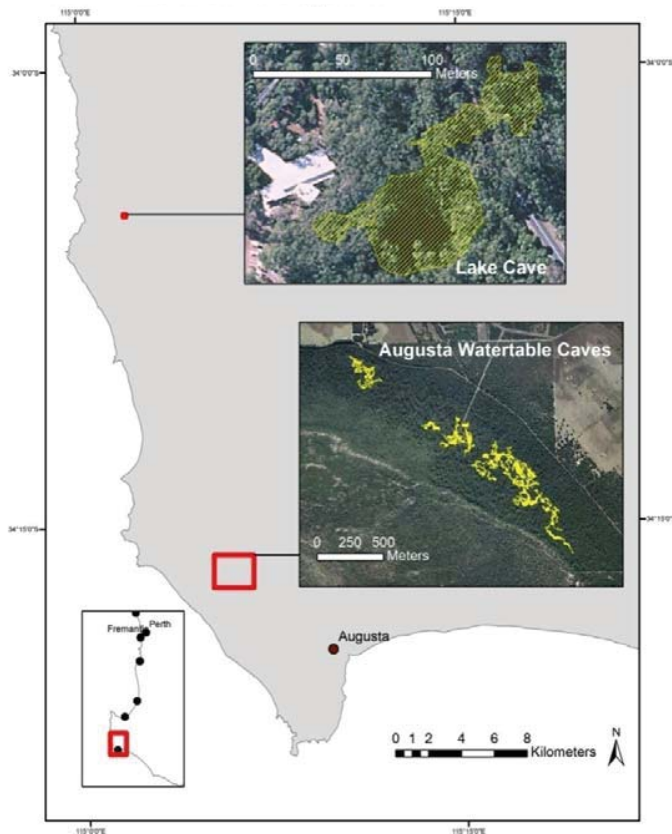


Figure 40: Leeuwin Naturaliste Ridge study sites showing surveyed cave outlines (yellow) of Lake, Jewel, Easter and Labyrinth Cave Karst System (Leeuwin Naturaliste Ridge Caves). Map of Lake Cave provided courtesy of Augusta Margaret River Tourism Association; Jewel Cave Karst System adapted from Eberhard (2004). Aerial photograph reproduced by permission of Western Australian Land Information Authority, C/L28 –2013

3.3.3 Step 1: Identify the hazard

Step 1.1 Identify the hazard

Primary hazard: Groundwater decline potentially due to climate change (declining rainfall and groundwater recharge), increasing temperature and evapotranspiration (Kauhanen et al. 2011), land use (tree plantations) and land management (fire regime).

Secondary hazard: Potential change in water quality due to changes in groundwater dynamics and pollution of groundwater with nutrients from agricultural land use in the catchment.

Step 1.2 Define the temporal boundaries

In the Leeuwin Naturaliste Ridge Caves, three broad groundwater level condition-states and time periods were identified as an initial basis for assessing groundwater ecosystem 'health'. The states were "wet", "drying", and "dry", occurring between 1958-1982, 1995-2004 and 2010-2012 respectively (Figure 41).

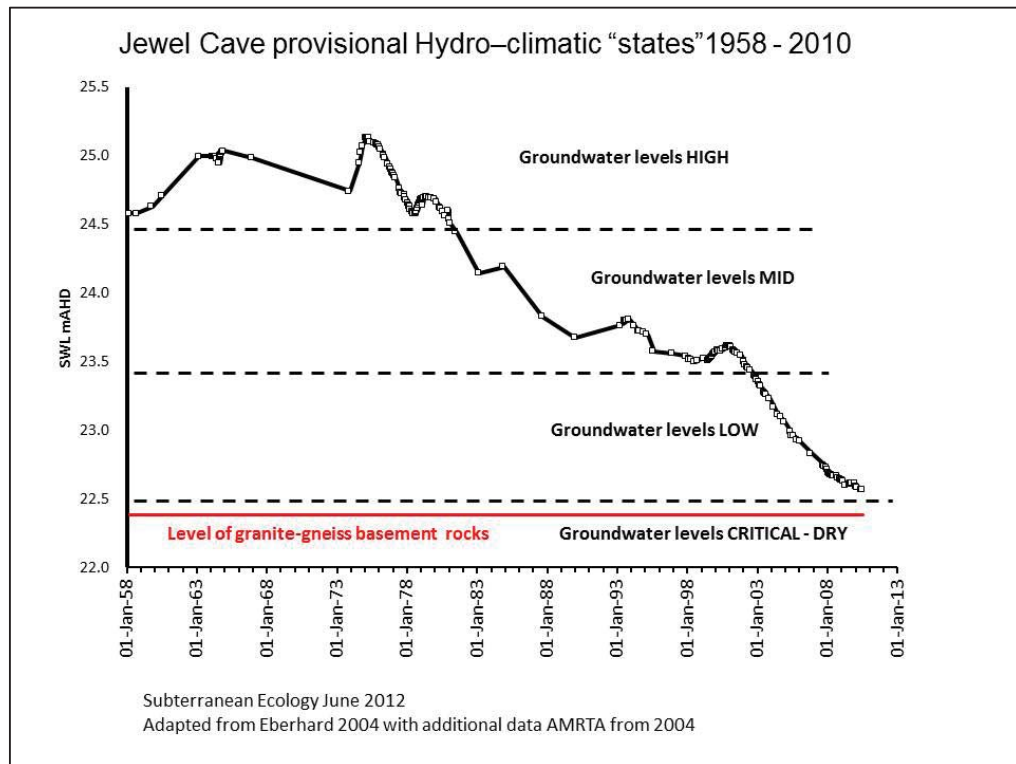


Figure 41: Jewel Cave provisional hydro-climatic states 1958-2010 adapted from Eberhard (2004) with additional data AMRTA. SWL (mAHd) refers to the subterranean water level in the cave as metres AHD (Australian Height Datum).

Step 1.3 Determine the cause of the hazard

The cause of the groundwater decline was investigated through Cumulative Rainfall Departure (CRD) analysis (Figure 42). CRD is widely used to untangle the effects of rainfall, and land and water management practices on groundwater levels. It assumes that rainfall is the only driver in changing groundwater levels and thus any deviation indicates that other factors influence water levels. From 1975 – 1990: simulated water level matches rainfall, suggesting rainfall is the main driver. From 1990 – 2000: simulated water level shows a muted response to rainfall suggesting changes in catchment intercepting groundwater recharge, or groundwater use by vegetation within catchment has changed (e.g. possible response to drought). After 2000, there is an overall drying trend in measured water level and rainfall, but the measured water level is approximately one metre lower than that projected by rainfall decline.

The CRD analysis of the Leeuwin Naturaliste Ridge Caves indicated changes in the relationship between rainfall and recharge and suggests a cumulative impact and/or contributory stressor in addition to the primary cause of climatic drying. Further research is needed to verify if the cause of these changes is due to changed groundwater use by native vegetation in the catchment and/or land use practices such as tree plantations or altered drainage.

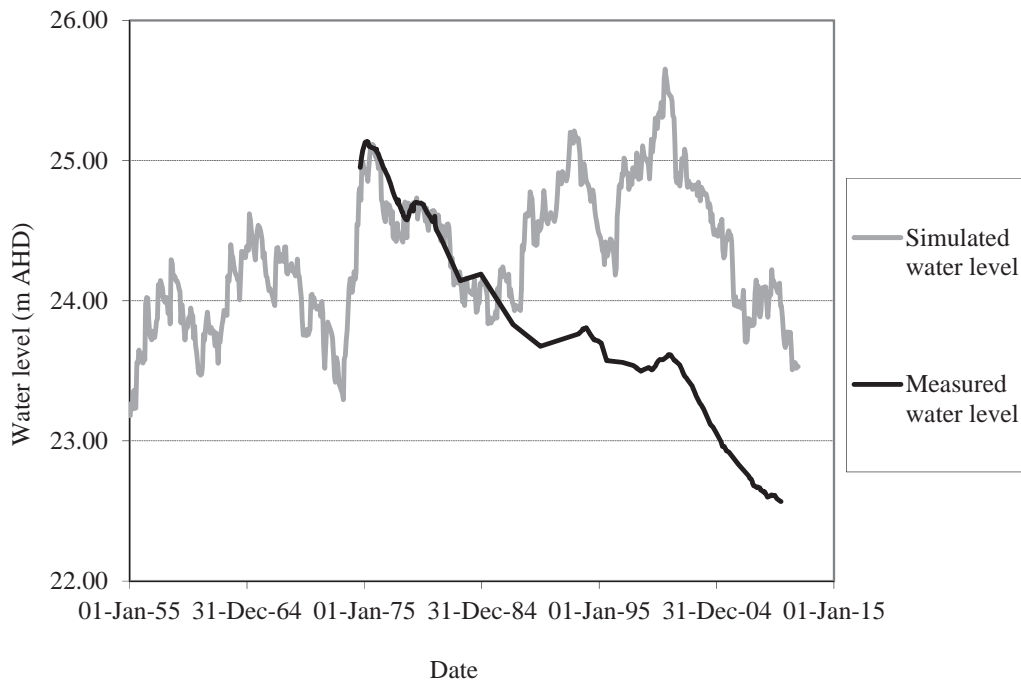


Figure 42: Jewel Cave Cumulative Rainfall Departure (CRD) displays measured groundwater levels in Jewel Cave compared with simulated levels according to climatic data (Data source courtesy, Steve Appleyard). The measured surface water level is based on data collected from 1958 onwards although the trendline commences ca. 1975 due to averaging effects.

3.3.4 Step 2: Exposure and Vulnerability

Step 2.1 Determine spatial and temporal change

Groundwater expression in the caves was mapped based on field measurement of water depths at different time intervals from 1958 to 2012 (Eberhard 2004). The area of water coverage was estimated based on historical photographs and personal observations (Figure 43).

Outlines of caves had been surveyed manually in the past, and were available as graphics with scale and north point (from Eberhard, 2004), although not as geo-referenced datasets.

We therefore created shapefiles from the original graphic cave outlines for Lake, Easter and Jewel Caves and the Labyrinth using a 3-stage geo-rectification process:

- Images were scanned and the scanned image was scaled;
- Cave entrance datum coordinates were obtained; and
- Each image was rotated to align north points with map north.

The resulting cave outlines and the entrance datums were checked against known features on the ground surface and positioned within the landscape.

Hand-drawn maps of estimated water levels were provided for each cave (S. Eberhard, P. Bell), identifying the approximate extent of water in the caves for two historical periods (1958-1982, 1995-2004) and the present (2010-2012). These outlines were converted to solid shapes and rasterised using the ArcScan extension of ArcGIS..

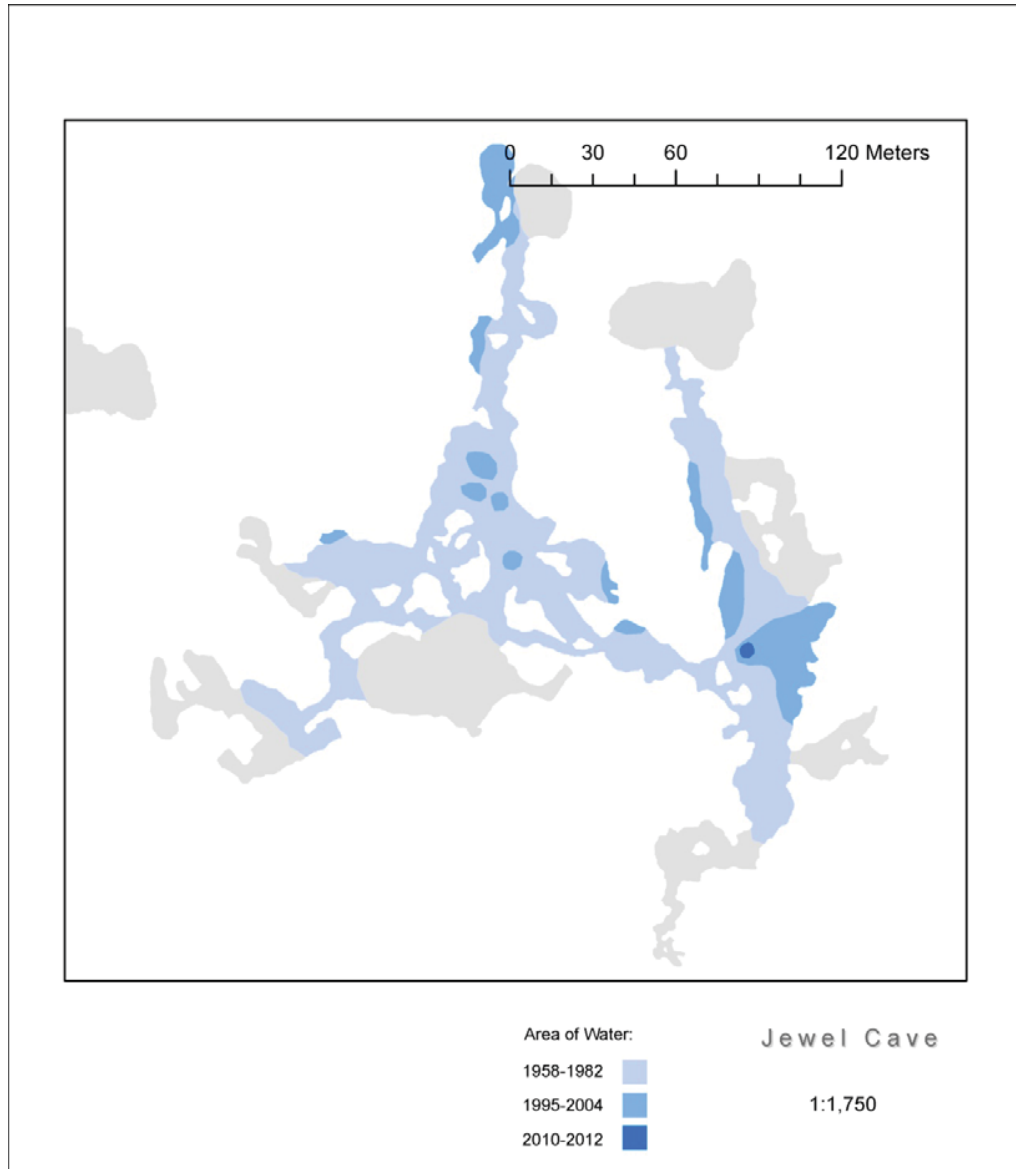


Figure 43: Graphical depiction of water level changes in Jewel Cave over three time periods – see SD 5: Chilcott 2013 and SD 7: Neville 2013 for greater detail and other caves. Jewel Cave map courtesy of Peter Bell; adapted from Eberhard 2004.

Step 2.2 Accommodate dynamics of hydrological change

Water level was measured directly over time so the inherent dynamics were captured in a hydrograph for each cave. An example hydrograph (Jewel Cave) is shown in Figure 44.

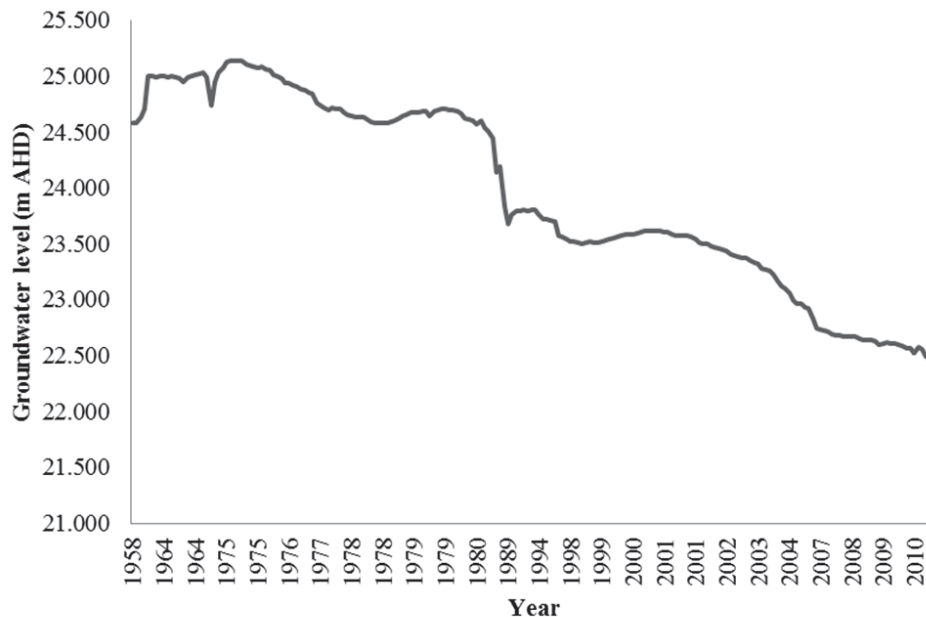


Figure 44: Jewel Cave groundwater level, measured between 1958-2012.

3.3.5 Step 3: Assess Effects

The steps involved in assessing the effects of groundwater decline to the Leeuwin Naturaliste Ridge Cave water quality and stygofauna are explained below.

Step 3.1 Collate available data

Collation and integration of groundwater quality, and faunal assemblage data from previous surveys (theses, published and unpublished reports) was undertaken for the Leeuwin Naturaliste Ridge Caves. Collection of new groundwater quantity, quality and faunal assemblage data from three caves within the Leeuwin Naturaliste Ridge (Jewel, Easter and Lake cave) was also carried out. These data were collated to assess the effects of changing water quality and depth parameters on the stygofauna. Refer to the methods section of Supporting Document 5 (Chilcott 2013) for more detail on how this was undertaken.

Water depth and water quality parameters were compared to changes in stygofauna community species composition using multivariate statistics (see SD5: Chilcott 2013 for methodology). There were no significant effects of changes in water quality but a clear reduction in species richness (number of species) with declining water level (Figure 45).

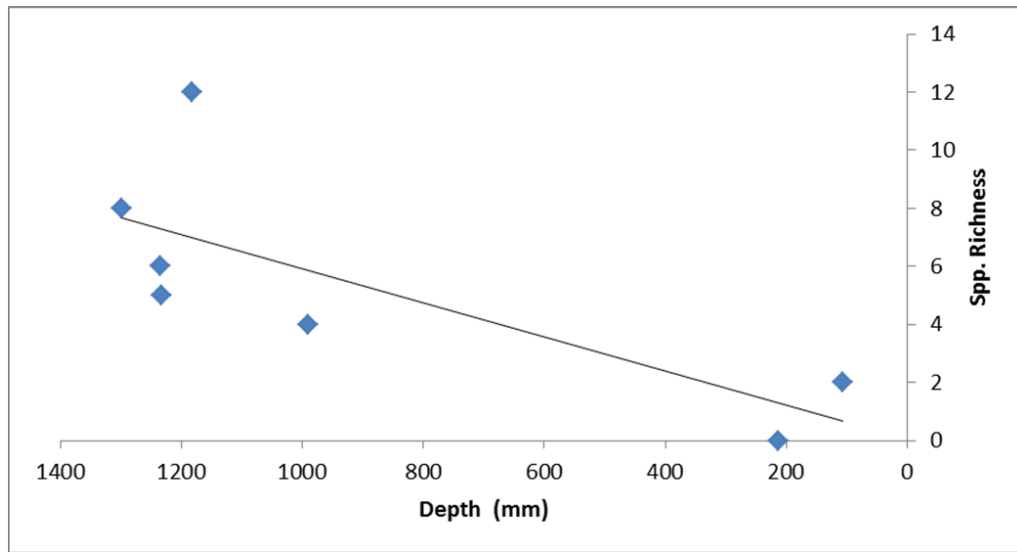


Figure 45: Number of invertebrate taxa plotted against water depth for those years where both variables were recorded in Jewel-Easter Cave. Three clusters of data points are evident: higher species counts and water levels in the 1990s, lower water levels and species richness 2000-03, and very low water levels with no taxa recorded in 2010 and only 2 species recorded in 2012.

Step 3.2 Develop a conceptual model

The first step in developing the detailed conceptual model described below was a group exercise to define the linkages between climate, hydrology, water quality and other physiochemical parameters and the biota (Figure 46). This provided a framework of understanding. Specific conceptual models for the Leeuwin Naturaliste Ridge cave ecosystems were derived from a detailed comparative study on stygofauna in both the Leeuwin Naturaliste Ridge and Yanchep cave systems (Figure 47) (see SD5: Chilcott, 2013).

Margaret River Caves Conceptual Model

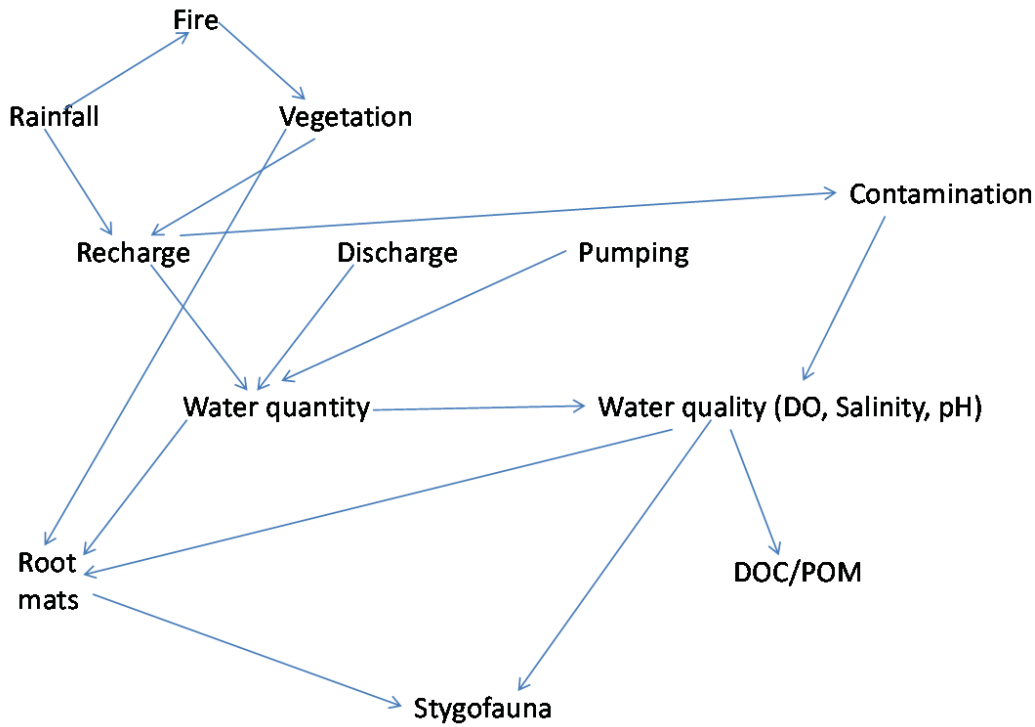


Figure 46: Basic conceptual model of the Leeuwin Naturaliste Ridge Cave system.

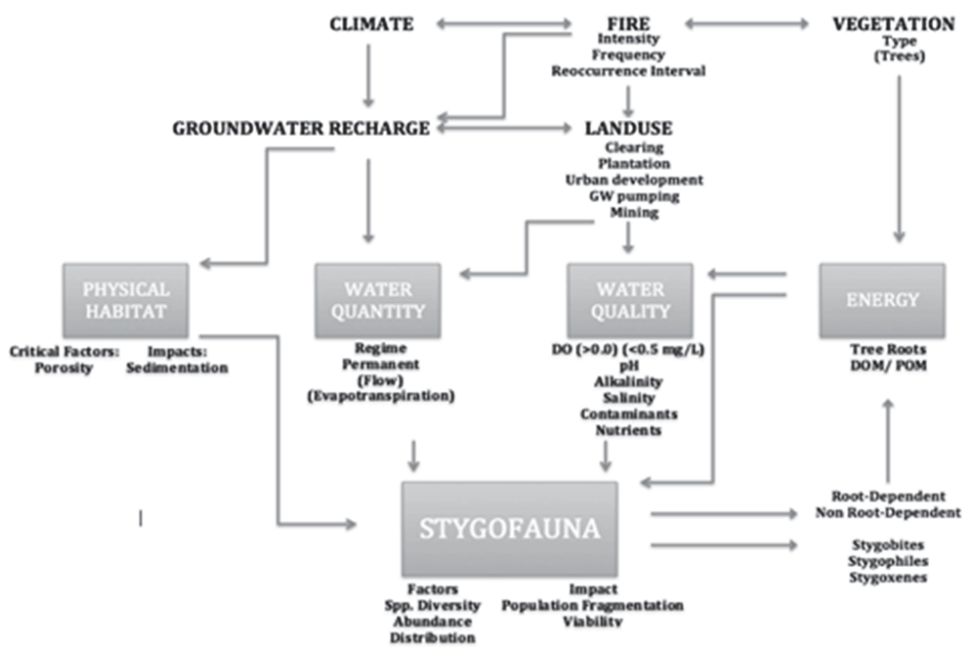


Figure 47: Major contributing factors to the health of GDE's. Climate drives the fire regime, groundwater recharge and presence of groundwater dependent vegetation which in turn creates a stable environment rich in energy and stygofauna (from Chilcott 2013).

3.3.6 Step 4: Characterise Risk

Step 4.1: Determine thresholds

The historical data available for the caves consisted of groundwater levels (measured from 1958 to present) water quality analysis and stygofauna community assessment ca. 1990 onwards (Jasinska 1996, Eberhard 2004). Practical limitations involved with monitoring underground ecosystems and the limited amount of biological and other environmental attribute data collected from the Leeuwin-Naturaliste Ridge cave systems meant that the BBN was constructed based on expert opinion. Two experts on the systems (Stefan Eberhard and Stacey Chilcott) working from the initial conceptual model derived a basic network structure based solely on groundwater level inputs and populated the conditional probability tables for each node based on their experience with the caves and field observed eco-hydrological condition state “thresholds” and “end-points”.

Step 4.2 Determining Probabilities of Risk: Bayesian Belief Networks

The initial conceptual model for the Leeuwin Naturaliste Ridge Caves was complex (Figure 48). As a number of variables could not be modelled in relation to climate change and groundwater decline (e.g. vegetation changes), the BBN was simplified to just model changes in overall species richness in relation to changes in groundwater level. Running this simple model showed that as groundwater levels declined so did species richness. Changes in the tree root mat dependent fauna node were the main influence because the roots were a food source and habitat for more than 50% of species and as the groundwater level declined and the root mats dried-out, this food source/ habitat was lost. Other species of stygofauna were not dependent on the tree roots and continued to persist after the tree roots dried-out, however, these species still remained vulnerable to further groundwater decline with most groundwater habitat and species presumed lost after the groundwater level (GWL) declined below 23 m AHD.

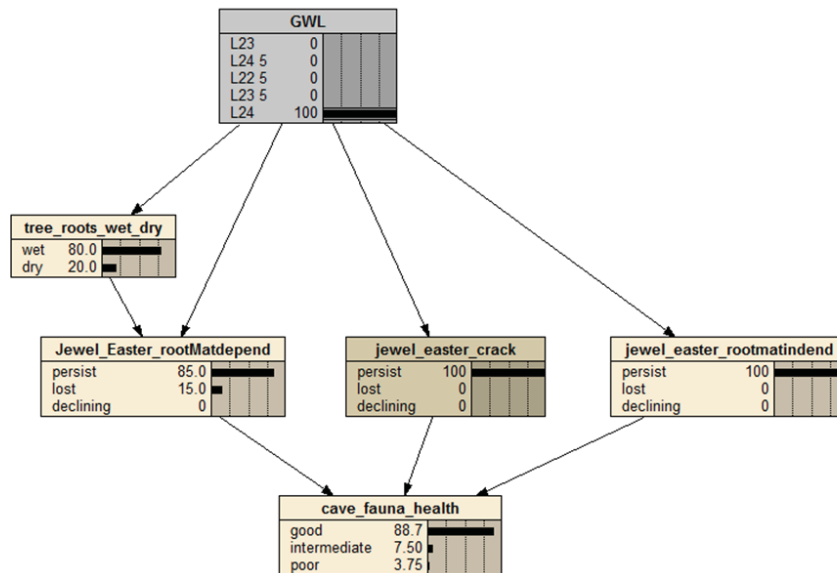


Figure 48: BBN for stygofauna for Leeuwin Naturaliste Ridge Caves.

Step 4.3: Spatial mapping of risk using GIS

The simplistic temporal nature of this study (three time periods) meant that risk was illustrated simply as the extent of water (habitat) available to stygofauna. The clear relationship between stygofauna species richness and water depth (Figure 49) meant this was a very simple effect to display.

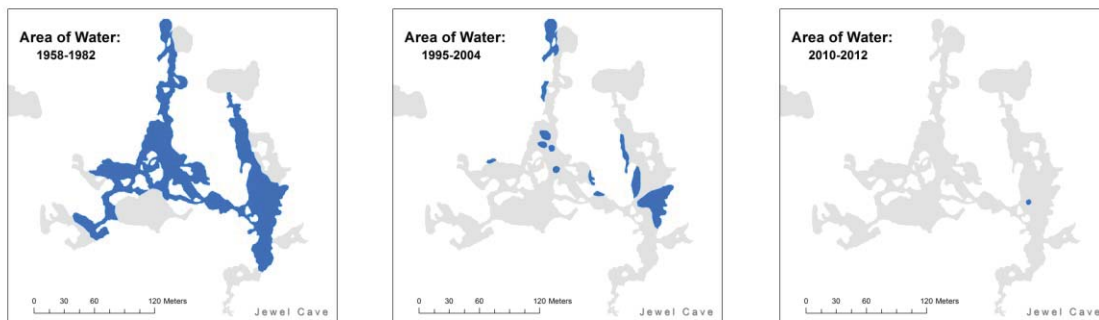


Figure 49 – Graphical depiction of declining groundwater levels in Jewel Cave over three time periods. The areal extent of free-standing water bodies (lakes) within the cave are shown in blue. Jewel Cave map courtesy of Peter Bell; adapted from Eberhard 2004.

3.3.7 Step 5: Risk Management

This case study illustrates the observed impacts of groundwater decline on endangered Subterranean Groundwater Dependent Ecosystems (SGDEs) in the Leeuwin Naturaliste Ridge Cave System. Spatially depicting this using GIS and translating the data through a conceptual model into a simple BBN model has facilitated communication of this case study, while positioning it within the broader context of climate change impacts to GDEs. The retrospective approach (from a healthy system containing high water levels to a practically dry cave) has helped to characterize and define the condition, “thresholds” and “end-points” due to declining groundwater level and associated changes in water quality, using a limited macroinvertebrate dataset. It is critical to appreciate that the condition state “thresholds” and “end-points” (in terms of ground water level and species richness) characterized herein are specific to the Leeuwin Naturaliste Ridge Cave System.

Other GDEs in other sites will have different and locally-specific characteristics and responses. For example, in deeper aquifers tree roots are less likely to be a food source/habitat for stygofauna. The framework successfully defined the key drivers and response of the GDE to declining groundwater levels and has been shown to be a useful tool for subterranean GDEs transferable to other locations. An adaptive management approach needs to take into consideration site-specific characteristics and assess the requirements of each SGDE on a case by case basis.

The management responses / options for the Leeuwin Naturaliste Ridge Cave System highlighted by this framework are:

1. Do nothing. This is the current situation and under the declining SW rainfall regime all known occurrences of this SGDE (including some locally endemic stygofauna species) will be lost within a few years.

2. Attempt to ameliorate the rate of groundwater decline by managing recharge / discharge within this small karst aquifer. This might be attempted by managing vegetation within the catchment, through the influence that vegetation type and cover, etc has on groundwater recharge / discharge responses. While (selective) thinning or clearing of native vegetation or applying enhanced fire regimes as a means to increase recharge is certainly not advocated in the Leeuwin Naturaliste Ridge Caves, further research into the potential contributory effects of nearby land-use practices (which include drainage enhancement in cleared pasture and tree plantations) is warranted. If these are found to be contributory stressors then ameliorative management action could be taken.
3. Contribute additional recharge by, for example, capturing and storing local rainfall (or other local water source) and delivering this at a measured sustainable rate into the cave. This system has been successfully trialled at Lake Cave in the previous two years. While careful attention needs to be paid to maintaining the hydrochemistry of recharge waters within appropriate natural ranges, as well as controlling the risk of inadvertent introduction of contaminants and/or foreign organisms, vigilant monitoring of the water chemistry and stygofauna has shown no adverse effects from this treatment (Subterranean Ecology 2012).

4. DISCUSSION

The application of a standard risk assessment protocol (Assante-Duah 1998 - Figure 1) to the hazard of declining groundwater levels, in three different groundwater dependent ecosystems (GDEs) (wetlands, a river and caves) with differing availability of data provided a robust testing arena. The combination of this testing with input from a suite of end-users, other scientists and experts from across Australia resulted in a framework that is straightforward to use, but has a high degree of adaptability both in terms of the type of GDE it can be applied to and the range of uses to which it can be put. The framework developed is outlined in detail in the companion document: *“Adapting to climate change: a risk assessment and decision making framework for managing groundwater dependent ecosystems with declining water levels: Guidelines for Use”*.

The guide explains how to use the framework, showcasing the three GDE case studies throughout to provide first hand examples and variations of how the framework can be used. The guide also has a detailed section (Step 5: Risk management) outlining the numerous ways it can be used to adapt to climate change and manage GDEs with declining water levels. Rather than repeat the scope of the framework here, the reader is directed to this section of the companion document: Guidelines for Use (Chambers *et al.* 2013)

A major strength of the framework is its capacity to relate climate, hydrology and ecosystem response in a single tool. Bayesian Belief Networks can be easily modified to show changes in probability of risk resulting from the interaction between climate, hydrology, water quality, biotic resources requirements and biotic response. This provides a transparent and interactive template for decision-making at a range of levels. It equally shows the impact of extracting water on biota as it does the capacity for biota to survive under different climate and/or land use scenarios. This innovative approach is presented in a user-friendly way for managers, enabling adaptation actions by one or more agencies, individually or in synergy to be assessed.

There are a number of tools for managing GDEs that have been developed across Australia. This risk assessment framework has taken into consideration these existing tools and adapted to use and augment them into the framework methodology. In this way the risk assessment framework has collated a larger toolbox available to managers. How to identify GDEs by Eamus (2009) provides seven tools useful for determining an ecosystem's dependence on groundwater. The GDE toolbox (Richardson *et al.* 2011) “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 National Water Commission (NWC) has funded The Atlas of Groundwater Dependent Ecosystems of Australia, which maps the position of GDEs in Australia and includes systems which are reliant on the surface, subsurface or subterranean supply of water (SKM 2012). Groundwater modelling guidelines have recently been developed by the National Water Commission, providing a reference of best practice for managers (Barnett *et al.* 2012). This provides a methodology for developing physical, mathematical, analytical or numerical groundwater models. The NWC document also discusses flow regimes and types of connectivity found in GDEs. Surface water/groundwater modelling guidelines have also been developed for river systems in Australia (Rassam *et al.* 2012). In developing the framework a number of key concepts that affect the accuracy or value of the tool became apparent. Some of these are considered further in the next section on Gaps and Future Research.

Primarily is the need for users of the framework to be aware of the accuracy and robustness of the outcomes based on the credence of the input data. While the properties of Bayesian Belief Networks allow a fairly robust delivery of probability of risk whether using expert opinion or detailed verified models supported by extensive datasets, the inherent limitations of the inputs must always be considered. Projecting groundwater levels into the future incorporates a number of steps each with inherent error. For example, downscaling global climate change models to local areas, incorporating changed rainfall regimes to the response by groundwater (or especially groundwater-surface water interactions) using hydrological models, using landscape scale data for projections for small localised areas. Each may have insufficient resolution to provide meaningful estimation of projected groundwater levels at a small scale. Instead, as outlined in the “Guidelines for Use”, the outputs of such scenarios should be used with caution perhaps indicating a first approximation, a likely direction of change rather than providing exact groundwater levels into the future.

Potentially a key limitation to using this framework is that the high degree of uncertainty of future climate change projections (e.g. different global change models predicting both wetter and drier future climates such as for north-west Australia - Kauhanen *et al.* 2011)) will result in a broad spectrum of possible outcomes. For this reason, the framework is best suited for Mediterranean climates and locations where climate change projections have a lower level of uncertainty.

Secondly, as outlined in the framework, most projections deal with a mean of conditions, when actually it is the extreme events that are likely to have the greatest effect (Jentsch and Beierkuhnlein 2008). We have attempted to consider the impact of non-linearity in the framework but users need to keep in mind the possibilities. The framework can inform management of extreme events by indicating the thresholds at which biota will incur a high degree of risk. Running scenarios with a large number of dry days for example, could indicate when these thresholds are likely to be breached.

Thirdly, the framework provides a probability of risk not an actual outcome. Outputs should be appropriately interpreted. A spatial risk assessment map is easily misconstrued if not considered with appropriate knowledge of the restraints of in its construction. A strength of the framework is the transparency of the Bayesian networks. It is likely that these will provide more robust and intuitive use for practitioners with little data available. While considerable time and resources may be required to create BBNs and/or spatial risk assessment maps, once created they could be readily maintained and updated. Where fewer resources are available the problem-solving nature of developing conceptual models as defined by the framework can still provide a transparent and valuable tool for decision making.

As a consequence of its adaptability, its central tenet of developing a conceptual model to drive the outcomes of the framework, its capacity to deal with functional groups of biota, and the input provided by national advisors, we believe the methodology will be transferable to other types of GDEs and locations in Australia and internationally.

5. GAPS AND FUTURE RESEARCH DIRECTIONS

A crucial variable required for the framework is the relationship between groundwater level and surface expression of water in GDEs. This project highlighted the complexity of this relationship. Differences in whether the soil profile is saturated or unsaturated, rainfall, surface water inflows, evapotranspiration effects, relative permeability at depth are just some of the factors that contribute to this complexity. Lack of a good understanding can prevent projection of future scenarios and spatial assessment of risk, such as occurred for the 2030 projection of risk for wetlands on the Gngangara Mound. Current work by Barron investigating the relationship between historical surface water levels in wetlands and groundwater levels in adjacent bores may shed light on this area that can be incorporated into future use of the framework.

A key requirement for this framework is the necessity to test it under a variety of conditions including different locations across Australia and internationally, for different GDEs and for different purposes (eg biodiversity, water extraction). It is clear from input from the national advisory panel that the information on GDEs available in many parts of Australia is limited. While the current development of the framework on three diverse ecosystems with differing data availability has produced a valuable tool, only through further testing can we hone that tool to provide a robust, tested product.

During discussions throughout the framework's development there have been questions about the reversibility of the thresholds determined for biota and whether recovery is possible or whether hysteresis would prevent recovery of GDEs subject to drying. In the last year there have been a number of research projects investigating the scope of resilience, refuges and connectivity in recolonisation and recovery (eg Robson *et al.* 2013). A synthesis of this information could improve the capacity of this framework to predict the potential of rising groundwater in the recovery of GDEs previously subject to drying.

There are number of ways in which the framework could be improved. The current framework had only one year for development. During this time a good product was developed but that process also highlighted a range of possibilities that would benefit from further work. The work presented here on multiple criteria analysis for assessing values and condition of GDEs has had little time for development. End-users saw value in this approach for conservation estate planning. Using finer scale hydrological models (now available in Western Australia) to increase the resolution and accuracy of spatial risk assessment would be valuable. Investigating the impact of real climate scenarios (rather than mean trends), changes in seasonality, extreme events and non-linear effects would strengthen the framework.

While this framework considers the ecological implications of declining groundwater levels, managers are going to need to weigh the economic and social tradeoffs in using a tool of this nature. Addressing the socioeconomic and policy implications of scenarios presented by the framework would increase its usability and uptake.

REFERENCES

- Ali, R., McFarlane, D., Varma, S., Dawes, W., Emelyanova, I., Hodgson, G. and Charles, S. (2012a). Potential climate change impacts on groundwater resources of south-western Australia. *Journal of Hydrology* 475, 456–472. <http://dx.doi.org/10.1016/j.jhydrol.2012.04.043>
- Ali, R., McFarlane, D., Varma, S., Dawes, W., Emelyanova, I. and Hodgson, G. (2012b). Potential climate change impacts on the water balance of regional unconfined aquifer systems in South-Western Australia. *Hydrological and Earth Systems Science* 16, 4581–4601. www.hydrol-earth-syst-sci.net/16/4581/2012/doi:10.5194/hess-16-4581-2012
- Asante-Duah, D. K. (1998). Risk assessment in environmental management, John Wiley and Sons, Brisbane.
- Australian Bureau of Meteorology (2012). Australian climate variability and change - trend maps, Australian Bureau of Meteorology, viewed 23 January 2012, <http://www.bom.gov.au/cgi-bin/climate/change/trendmaps.cgi?map=evap&area=aus&season=0112&period=1970>.
- Australian State of the Environment Committee (2011). Australia: State of the Environment 2011, Australian State of the Environment Committee, viewed 11 April 2012, <http://www.environment.gov.au/soe/2011/report/inland-water/index.html>.
- Barnett, B., Townley, L. R., Post, V., Evans, R. E., Hunt, R. J., Peeters, L., Richardson, S., Werner, A. D., Knapton, A. & Boronkay, A. (2012). Australian groundwater modelling guidelines, Waterlines report, National Water Commission, Canberra, viewed 13 February 2012, http://www.nwc.gov.au/_data/assets/pdf_file/0016/22840/Waterlines-82-Australian-groundwater-modelling-guidelines.pdf.
- Barron, O., Silberstein, R., Ali, R., Donohue, R., McFarlane, D.J., Davies, P., Hodgson, G., Smart, N. and Donn, M. (2012). Climate change effects on water-dependent ecosystems in south-western Australia. *Journal of Hydrology* 475, 95-109. <http://dx.doi.org/10.1016/j.jhydrol.2012.02.028>
- Bates, B. C., Chandler, R. E., Charles, S. P. & Campbell, E. P. (2010). Assessment of apparent nonstationarity in time series of annual inflow, daily precipitation, and atmospheric circulation indices: A case study from southwest Western Australia. *Water Resources Research*, 46.
- Beatty, S. J., McAleer, F. J. & Morgan, D. L. (2009). Migration patterns of fishes of the Blackwood River and relationships to groundwater intrusion. Centre for Fish & Fisheries Research, Murdoch University, report to the Department of Water, Government of Western Australia, viewed 15 September 2012. <http://www.water.wa.gov.au/PublicationStore/first/97061.pdf>

- Beatty, S. J., Morgan, D. L., McAleer, F. J. & Ramsay, A. (2010). Groundwater contribution to baseflow maintains habitat connectivity for *Tandanus bostocki* (Teleostei: Plotosidae) in a south-western Australian river. *Ecology of Freshwater Fish* 19, 595-608.
<http://onlinelibrary.wiley.com/doi/10.1111/j.1600-0633.2010.00440.x/pdf>
- Beatty, S. J., Morgan, D. L., Rashnavidi, M. & Lymbery, A. J. (2011). Salinity tolerances of endemic freshwater fishes of south-western Australia: implications for conservation in a biodiversity hotspot. *Marine & Freshwater Research* 62, 91-100. <http://www.publish.csiro.au/?paper=MF10100>
- Beatty S., Morgan D., Keleher J., Lymbery A., Close P., Speldewinde P., Storer, T., & Kitsios, A. (2013). Adapting to climate change: a risk assessment and decision making framework for managing groundwater dependent ecosystems with declining water levels. Supporting document 4: Environmental variables in the habitats of south-western Australian freshwater fishes: an approach for setting threshold indicator value. National Climate Change Adaptation Research Facility, Canberra, Australia.
- Bond, N., Lake, P. & Arthington, A. (2008). The impacts of drought on freshwater ecosystems: an Australian perspective. *Hydrobiologia* 600, 3-16.
- Boteva, D., Griffith, G. and Dimopoulos, P. (2004). Evaluation and mapping of the conservation significance of habitats using GIS: an example from Crete, Greece. *Journal for Nature Conservation* 12, 237—250.
- Bradford, D.F. (2002). Amphibian Declines and Environmental Change in the Eastern Mojave Desert. Conference Proceedings. Spring-fed Wetlands: Important Scientific and Cultural Resources of the Intermountain Region, p 1-7.
<http://www.wetlands.dri.edu>
- Candela, L., von Igel, W., Javier Elorza, F. & Aronica, G. (2009). Impact assessment of combined climate and management scenarios on groundwater resources and associated wetland (Majorca, Spain). *Journal of Hydrology*, 376, 510-527.
<http://www.scopus.com/inward/record.url?eid=2-s2.0-70349173754&partnerID=40&md5=e3696d23a3120b0348c1ce1f97032761>
- Chambers, J., Nugent, G., Sommer, B., Speldewinde, P., Neville, S., Beatty, S., Chilcott, S., Eberhard, S., Mitchell, N., D'Souza, F., Barron, O., McFarlane, D., Braimbridge, M., Robson, B., Close, P., Morgan, D., Pinder, A., Froend, R., Horwitz, P., Cook, B. & Davies, P. (2013). Adapting to climate change: a risk assessment and decision making framework for managing groundwater dependent ecosystems with declining water levels. Guidelines for use. National Climate Change Adaptation Research Facility, Canberra, Australia.
- Chilcott, S. (2013). Adapting to climate change: a risk assessment and decision making framework for managing groundwater dependent ecosystems with declining water levels. Supporting document 5: The impact of declining groundwater levels on Stygofauna communities in the Leeuwin Naturaliste Ridge Cave

systems, Western Australia. National Climate Change Adaptation Research Facility, Canberra, Australia.

Chiew, F. H. S. & McMahon, T. A. (2002). Modelling the impacts of climate change on Australian streamflow. *Hydrological Processes* 16, 1235-1245.

Christensen, J. H., Hewitson, B., Busuioc, A., Chen, A., Gao, X., Held, I., Jones, R., Kolli, R. K., Kwon, W. T., Laprise, R., Rueda, V. M., Mearns, L., Menéndez, C. G., Räisänen, J., Rinke, A., Sarr, A. & Whetton, P. (2007). Regional Climate Projections, in S. Solomon, D., Qin, M. M., Chen, Z., Marquis, M., Averyt, K. B., Tignor, M. & Miller, H. L. (eds), *Climate Change 2007: The Physical Science Basis. Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change*, Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA.

CSIRO (2007). Climate change in Australia—technical report 2007, CSIRO, viewed 11 June 2012.

<http://www.climatechangeinaustralia.gov.au>.

CSIRO (2009a) Groundwater yields in south-west Western Australia. A report to the Australian Government from the CSIRO South-West Western Australia Sustainable Yields Project. CSIRO Water for a Healthy Country Flagship, Australia.

CSIRO (2009b). Water yields and demands in south-west Western Australia. A report to the Australian Government from the South-West Western Australia Sustainable Yields project; 276pp.

Danielopol, D. L., Griebler, C., Gunatilaka, A. & Notenboom, J. 2003. Present state and future prospects for groundwater ecosystems. *Environmental Conservation*. 30, 104-130.

Davies, P. M. (2010). Climate Change Implications for River Restoration in Global Biodiversity Hotspots. *Restoration Ecology*. 18, 261-268.

Del Borrello N. L. (2008). Management triggers and responses for groundwater dependent ecosystems in the South West groundwater areas, Water resource allocation planning report no. 31, Department of Water, Government of Western Australia, Perth. Viewed 14 December 2012

<http://www.water.wa.gov.au/PublicationStore/first/84212.pdf>

Eads, A. R., Mitchell, N. J. & Evans, J. P. (2012). Patterns of genetic variation in desiccation tolerance in embryos of the terrestrial-breeding frog, *Pseudophryne guentheri*. *Evolution* 66, 2865-2877

Eamus, D. (2009). Identifying groundwater dependent ecosystems. A guide for land and water managers, Land and Water Australia viewed 11 June 2012,

http://lwa.gov.au/files/products/innovation/pn30129/pn30129_1.pdf.

- Eberhard, S. M. 2002. Jewel Cave karst system, Western Australia: Environmental hydrogeology and groundwater ecosystem. Report to the Augusta-Margaret River Tourism Association.
- Eberhard, S. M. 2004. Ecology and hydrology of a threatened groundwater-dependent ecosystem: the Jewel Cave karst system in Western Australia. Murdoch University.
- Froend, R. & Loomes, R. (2004). Approach to determination of ecological water requirements of groundwater dependent ecosystems in Western Australia, A report to the Department of Environment.
- Gemitzi, A. & Stefanopoulos, K. (2011). Evaluation of the effects of climate and man intervention on ground waters and their dependent ecosystems using time series analysis. *Journal of Hydrology* 403, 130-140.
- Green, T. R., Taniguchi, M., Kooi, H., Gurdak, J. J., Allen, D. M., Hiscock, K. M., Treidel, H. & Aureli, A. (2011). Beneath the surface of global change: Impacts of climate change on groundwater. *Journal of Hydrology* 405, 532-560.
- Groom P. K., Froend R.H., Mattiske E. M. (2000). Impact of groundwater abstraction on a Banksia woodland, Swan Coastal Plain, Western Australia. *Ecological Management & Restoration* 1, 117–124.
- Hart, B. T. & Pollino, C. (2006). Ecological risk assessment and Bayesian modelling, Water Studies Centre, Monash University, viewed 19 March 2012, http://ns1.mrcmekong.org/download/programmes/ep/Bayesian_Network_Reading1.pdf.
- Hennessey, K., Fitzharris, B., Bates, B. C., Harvey, N., Howden, S. M., Hughes, L., Salinger, J. & Warrick, R. (2007). Australia and New Zealand, in ML Parry, Canziani, OF, Palutikof, JP, van der Linden, PJ & Hanson, CF (eds), *Climate Change 2007: Impacts, Adaptation and Vulnerability. Contribution of Working Group II to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change*, Cambridge University Press, Cambridge. http://www.ipcc.ch/publications_and_data/ar4/wg2/en/ch11s11-4.html#11-4-1
- Hill, A.J., Semenuik C.A., Semenuik V. & Del Marco, A. (1996). Wetlands of the Swan Coastal Plain Vol 2A: Wetland Mapping, Classification and Evaluation. Water Authority of Western Australia.
- Jasinska, E. J., Brenton, K. & McComb, A. J. 1996. Root Mats in Ground Water: A Fauna-Rich Cave Habitat. *Journal of the North American Benthological Society* 15, 508-519.
- Jasinska, E. J. 1997. *Faunae of aquatic root mats in caves of southwestern Australia: origins and ecology*. University of Western Australia.
- Jentsch, A. & Beierkuhnlein, C. (2008). Research frontiers in climate change: Effects of extreme meteorological events on ecosystems. *Comptes Rendus Geoscience*, 340, 621-628.

- Kauhanen K, Chambers J & D'Souza F (2011). Report Card of Climate Change and Western Australian Aquatic Ecosystems. Report to NCCARF Brisbane.
- Kløve, B., Ala-aho, P., Bertrand, G., Boukalova, Z., Ertürk, A., Goldscheider, N., Ilmonen, J., Karakaya, N., Kupfersberger, H., Kværner, J., Lundberg, A., Mileusnić, M., Moszczynska, A., Muotka, T., Preda, E., Rossi, P., Siergieiev, D., Šimek, J., Wachniew, P., Angheluta, V. & Widerlund, A. (2011). 'Groundwater dependent ecosystems. Part I: Hydroecological status and trends', *Environmental Science & Policy*, vol. 14, no. 7, pp. 770-781.
- Kundzewicz, Z. W. & Doll, P. (2009). Will groundwater ease freshwater stress under climate change?, *Hydrological sciences journal* 54, 665-675.
- Margat, J., Foster, S. & Droubi, A. (2006). Concept and importance of non-renewable resources. In: *Non-Renewable Groundwater Resources: A Guidebook on Socially-Sustainable Management for Water-Policy Makers* (ed. by S. Foster & D. P. Loucks), 13–24. IHP-VI, Series on Groundwater no. 10, UNESCO, Paris, France.
- Margules, C. & Usher, M.B. (1981) Criteria used in assessing wildlife conservation potential: A Review. *Biological Conservation* 21 79-109.
- Margules, C.R., Higgs, A.J. & Rafe, R.W. (1982). Modern Biogeographic Theory: Are there any Lessons for Nature reserve Design? *Biological Conservation*, 24 115-128
- Margules, C.R. & Nicholls, A.O. (1988) Selecting Networks of Reserves to Maximise Biological Diversity. *Biological Conservation* 43 ,63-76.
- McFarlane, D., Stone, R., Martens, S., Thomas, J., Silberstein, R., Ali, R. & Hodgson, G. (2012). Climate change impacts on water yields and demands in south-western Australia. *Journal of Hydrology* 475, 488-498
- McNeill J., MacEwan R. & Crawford D. (2006). Using GIS and a land use impact model to assess risk of soil erosion in West Gippsland. *Applied GIS* 2,19.1–19.16.
- Mitchell, N., Sommer, B. & Speldewinde, P. (2013). Adapting to climate change: a risk assessment and decision making framework for managing groundwater dependent ecosystems with declining water levels. Supporting document 3: Identifying thresholds for responses of amphibians to groundwater and rainfall decline. National Climate Change Adaptation Research Facility, Canberra, Australia.
- Morgan, D. L. & Beatty, S. J. (2006). Use of a vertical-slot fishway by galaxiids in Western Australia. *Ecology of Freshwater Fish* 15, 500-509.
<http://onlinelibrary.wiley.com/doi/10.1111/j.1600-0633.2006.00190.x/abstract>
- Morgan D. L., Beatty S. J., Klunzinger, M. W., Allen, M. G. & Burnham, Q. E. (2011). A Field Guide to Freshwater Fishes, Crayfishes & Mussels of South-Western Australia, SERCUL & Freshwater Fish Group & Fish Health Unit Murdoch University, Murdoch, Western Australia.

- Morgan, D. L., Gill, H. S. & Potter, I. C. (1998). Distribution, identification and biology of freshwater fishes in southwestern Australia. Records of the Western Australian Museum Supplement, No. 56, pp. 1-97.
<http://museum.wa.gov.au/store/museum-books/supplements/supplement-56-distribution-identification-and-biology-freshwater-fish>
- Morgan, D. L., Gill, H. S., Maddern, M. G. & Beatty, S. J. (2004). Distribution and impacts of introduced freshwater fishes in Western Australia. *New Zealand Journal of Marine and Freshwater Research* 38, 511-523.
<http://www.tandfonline.com/doi/abs/10.1080/00288330.2004.9517257>
- Morgan, D. L., Thorburn, D. C. & Gill, H. S. (2003). Salinization of south-western Western Australian rivers and the implications for the inland fish fauna – the Blackwood River, a case study. *Pacific Conservation Biology* 9, 161-171.
http://pcb.murdoch.edu.au/toc/pcb_contents_v9.html
- Morrongiello, J. R., Beatty, S. J., Bennett, J. C., Crook, D. A., Ikedife, D. N. E. N., Kennard, M. J., Kerezszy, A., Lintermans, M. L., McNeil, D. G., Pusey, B. J., & Rayner, T. (2011). Climate change and its implications for Australia's freshwater fish. *Marine and Freshwater Research* 62, 1082-1098. Viewed on 10 October 2012 <http://www.publish.csiro.au/paper/MF10308.htm>
- Moustadraf, J., Razack, M. & Sinan, M. (2008). Evaluation of the impacts of climate changes on the coastal Chaouia aquifer, Morocco, using numerical modeling. *Hydrogeology journal* 16, 1411-1426.
- Mpelasoka, F., Hennessy, K., Jones, R. & Bates, B. (2008). Comparison of suitable drought indices for climate change impacts assessment over Australia towards resource management. *International Journal of Climatology* 28, 1283-1292.
- Neville, S. (2013). Adapting to climate change: a risk assessment and decision making framework for managing groundwater dependent ecosystems with declining water levels. Supporting document 7: Spatially representing the impacts of falling groundwater due to climate change on groundwater dependent ecosystems. National Climate Change Adaptation Research Facility, Canberra, Australia.
- Nugent, G., Chambers, J. & Speldewinde, P. (2013). 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. National Climate Change Adaptation Research Facility, Canberra, Australia.
- Panitsa, M., Koutsias, N., Tsiripidis, I., Zotos A. & Dimopoulos, P. (2011). Species-based versus habitat-based evaluation for conservation status assessment of habitat types in the East Aegean islands (Greece). *Journal for Nature Conservation* 19, 269– 275
- Rassam, D., Jolly, I. & Pickett, T. (2012). Guidelines for modelling groundwater-surface water interactions in eWater source: Towards best practice model application, EWater Cooperative Research Centre, Canberra. Viewed 5 December 2012,

[http://www.ewater.com.au/uploads/files/eWater-Modelling-Guidelines-GSWI-Source-\(v1-Mar-2012\).pdf](http://www.ewater.com.au/uploads/files/eWater-Modelling-Guidelines-GSWI-Source-(v1-Mar-2012).pdf).

- Richardson, S., Irvine, E., Froend, R., Boon, P., Barber, S. & Bonneville, B. (2011). Australian groundwater-dependent ecosystem toolbox part 1: assessment framework, Waterlines Report, National Water Commission, Canberra. Viewed 23 February 2012, http://nwc.gov.au/_data/assets/pdf_file/0006/19905/GDE-toolbox-part-1.pdf.
- Risbey, J. S., Hamza, K. & Marsden, J. S. (2007). Use of climate scenarios to aid in decision analysis for interannual water supply planning, *Water Resources Management* 21, 919-932.
<http://www.scopus.com/inward/record.url?eid=2-s2.0-34248220475&partnerID=40&md5=c91a8bd200e87368937315d5ccd2edfd>
- Roberts, J. (2002). Hydrology, *Encyclopedia of Global Environmental Change*, Wiley, Chichester, United Kingdom.
<http://go.galegroup.com/ps/i.do?id=GALE%7CCX3438400378&v=2.1&u=murdoch&it=r&p=GVRL&sw=w>.
- Robson, B. J., Chester, E. T., Allen, M., Beatty, S., Close, P., Cook, B., Cummings, C. R., Davies, P. M., Lester, R., Lymbery, A., Matthews, T. G., Morgan, D., Stock, M. (2013). Novel methods for managing freshwater refuges against climate change in southern Australia. Final Report. NCCARF, Brisbane.
- Silberstein, R.P. Aryal S.K., Durrant J., Pearcey, M. Braccia M., Charles S.P., Boniecka, L., Hodgson, G.A. Bari, M.A. and McFarlane, D.J. (2012). Climate change and runoff in south-western Australia. *Journal of Hydrology* 475, 441-455.
- SKM (2012). SKM Achieve Magazine, Issue 4.
http://www.skmconsulting.com/SiteDocuments/Knowledge-andInsights/achieve/issue4_2012/SKMachieve_Issue4_2012_Online-groundwateratlas-safeguards-ecosystems.pdf.
- Skurray, J. H., Roberts, E. J. & Pannell, D. J. 2011. Hydrological challenges to groundwater trading: Lessons from south-west Western Australia. *Journal of hydrology*, 256-268.
- Sommer, B. & Froend, R. (2010). Gngangara mound ecohydrological study. Final Report to the Western Australian Government, Department of Water, Joondalup: Centre for Ecosystem Management, Edith Cowan University. Viewed 10 February 2012, <http://www.water.wa.gov.au/PublicationStore/first/100015.pdf>.
- Sommer, B. & Horwitz, P. (2001). Water quality and macroinvertebrate response to acidification following intensified summer droughts in a Western Australian wetland. *Marine and Freshwater Research* 52, 1015-1021.
- Sommer, B., McGuinness, S., Froend, R. & Horwitz, P. (2013). Adapting to climate change: a risk assessment and decision making framework for managing groundwater dependent ecosystems with declining water levels. Supporting

- document 2: Assessing risks to groundwater dependent wetland ecosystems in a drying climate: an approach to facilitate adaptation to climate change. National Climate Change Adaptation Research Facility, Canberra, Australia.
- Speldewinde, P. (2013). Adapting to climate change: a risk assessment and decision making framework for managing groundwater dependent ecosystems with declining water levels. Supporting document 6: Development of Bayesian Belief Networks for modeling the impacts of falling groundwater due to climate change on groundwater dependent ecosystems. National Climate Change Adaptation Research Facility, Canberra, Australia.
- Stratgen (2005). Hydrology and hydrogeology of the groundwater dependent ecosystems of the Blackwood River south west Yarragadee water supply development. Information series report no 6, the Water Corporation.
- Stratgen (2006). South West Yarragadee Water Supply Development: Sustainability Evaluation/Environmental Review and Management Programme Volume 2. Report for the Water Corporation, Perth, Western Australia. Viewed 14 December 2012 <http://www.water.wa.gov.au/PublicationStore/first/82411.pdf>
- Subterranean Ecology Pty Ltd & Association, A. M. R. T. 2011 Lake Cave Eco-Hydrology Recovery Project - Progress Report No. 3. Report to Government of Western Australia Natural Resource Management Grant Scheme (No. 09075), April 2012.
- Taylor, R. G., Scanlon, B., Doll, P., Rodell, M., van Beek, R., Wada, Y., Longuevergne, L., Leblanc, M., Famiglietti, J. S., Edmunds, M., Konikow, L., Green, T. R., Chen, J., Taniguchi, M., Bierkens, M. F. P., MacDonald, A., Fan, Y., Maxwell, R. M., Yecheili, Y., Gurdak, J. J., Allen, D. M., Shamsudduha, M., Hiscock, K., Yeh, P. J.F., Holman, I. & Treidel, H. (2012). Ground water and climate change. *Nature Climate Change*, 3, 322-329. <http://dx.doi.org/10.1038/nclimate1744>
- Tomlinson, M. & Boulton, B. (2008). Subsurface groundwater dependent ecosystems: a review of their biodiversity, ecological processes and ecosystem services, National Water Commission, Canberra. Viewed 14 March 2012, http://www.nwc.gov.au/__data/assets/pdf_file/0007/11014/Waterlines__subsurface_full_version.pdf.
- Treidel, H., Martin-Bordes, J. L. & Gurdack, J. J. (2012). Introduction, in H Treidel, Martin-Bordes, JL & Gurdack, JJ (eds), *Climate Change Effects on Groundwater Resources, A Global Synthesis of Findings and Recommendations*, CRC Press, London, UK. <http://dx.doi.org/10.1201/b11611-2>.
- UNESCO (2008). International Hydrological Programme of the United Nations Educational, Scientific and Cultural Organization (UNESCO). Groundwater Resource Assessment under the Pressure of Humanity and Increasing Climate Change (GRAPHIC) framework document. <http://unesdoc.unesco.org/images/0016/001631/163172e.pdf>

- Voinov, A. & F. Bousquet (2010). Modelling with stakeholders. *Environmental Modelling & Software* 25, 1268-1281.
- Vörösmarty, C. J., Lévêque, C. & Revenga, C. (2005). Fresh Water. Chapter 7, in R Hassan, Scholes, R & Ash, N (eds), *Ecosystems and Human Well-Being: Current State and Trends. A report to the Millennium Ecosystems Assessment.*, Island Press, Washington, DC.
- Walls, S.C., Barichivich, W.J. & Brown, M.E. (2013). Drought, deluge and declines: the impact of precipitation extremes on amphibians in a changing climate. *Biology*, 2, 399-418.
- Yesertener, C. (2008). Assessment of the declining groundwater levels in the Gnangara Groundwater Mound, Government of Western Australia, Department of Water, Perth, Western Australia, <http://www.water.wa.gov.au/PublicationStore/first/62920.pdf>.

APPENDIX 1

End-user Workshop Summaries

Feedback for each workshop was collated into detailed Excel spreadsheets. Below is a very brief summary of the pertinent findings.

5.1.1 February 22nd 2012

- The end user workshop was successful
- The participants were involved, engaged and gave helpful feedback.
- Participants indicated they had access to some threshold information, but were pleased that this information would be collated, readily accessible and cover a wide geographic range.
- There was interest in Bayesian networks and said they would use it, however, most preferred GIS.
- Concern was raised to the level of uncertainty and limitations of the tools and participants requested these would be highly visible to avoid misuse.

5.1.2 November 21st 2012

- End-users were interested in applying each stage of the RAF to management
- Local government and small consultancies don't have the resources to develop the methodology themselves – ideally it would be developed for them
- Trend towards finding the BBNs to be the most applicable section – useful to support decision making, liked the fact that it was robust and dealt with expert opinion
- Will need training for BBN or a detailed end-user guide/help manual
- Visual representation of risk very useful, however, wary of the limitations
- Quite a few people don't like the term 'tool'
- Wanted to know what an 'acceptable' level of ecosystem change is

APPENDIX 2

Members of National Advisory Panel

Name	Organisation	State
Alys Wall	BOM	ACT
David Deane	Department for Water, Land and Biodiversity	SA
Tanya Doody	CSIRO Land and Water	SA
Anthony O'Grady	CSIRO Ecosystem Sciences	Tas
Mark Mitchell	Office of Water	NSW
Roger James	Victoria University previous CSIRO	Vlc
Evan Dresel	Department of Primary Industries	Vic
Moya Tomlinson	DERM	Qld
Rebecca Lester	Deakin University	Qld

Synthesis of National Advisory Panel Feedback (March 2012)

The following synthesises the feedback from the National Advisory Panel and suggests potential solutions both suggested by the panel and from discussions by the FW1108 research team. Following the synthesis, the individual questions and answers from each of the panel are provided in full.

Concerns of transferability and potential solutions

1) **Data required and its availability:** The amount of data that might be necessary to use the framework in another location was one of the main concerns for the transferability of this approach. The panel outlined that data on hydrology and modelling, the different types of GDEs (even their location) and the biota (both types present and threshold information) was patchy and tailored to suit different purposes that may not render it valuable to enter into a framework such as this. The framework was seen as data intensive and comment was made that it may only work in areas where sustainable yields type projects had been carried out and that this did not encompass many ecological hotspots.

The case study approach, with development in one area and then testing to scope transferability, was supported. Comment was made that the identification of the data required to underpin such an approach was as important as the framework itself. This would alert end-users to what information was needed to create an appropriate risk assessment protocol. It was suggested that while one data-poor area was to be tested within this project, similar testing on other sites outside WA would be required after this project was complete.

2) **GIS and scale issues:** Most of the panel thought GIS was a good tool, others needed more information as to how it would work before commenting. The main concern, identified also by the FW1108 research team and being addressed, was the differential scales at which data was available and whether it would be possible to make predictions for asset-based systems (wetlands) based on landscape scale data. The FW1108 research team suggested a potential way around this was to use the landscape scale for projections, determine where on the landscape the asset was and use finer scale Bayesian modelling to address individual wetland risk.

3) **Groundwater/surface water linkage:** The capacity to predict the effect of surface water expression based on groundwater level for different types of GDEs was seen as a significant limitation to the project. This was identified also by the FW1108 research team and is being addressed.

4) **Transferability to other types of GDEs:** It was suggested that the framework would need to be tested for different GDEs. There was need for clarity as to what hydrogeological information was required and at what scale. Once again the case study approach was supported.

5) **Transferability of thresholds:** There was concern that the biotic thresholds may not be transferable. Broader trialling of the method across Australia was recommended. Another comment regarding thresholds was the necessity to be clear as to the nature and shape of the threshold being used. How the threshold was determined, whether it was a physiological absolute, acute or chronic impact etc. This type of information would be important to managers.

6) **Utilisation by End-users:** The tool was seen as valuable for a wide range of end-users, particularly government agencies. However the panel warned that uptake of new planning tools was poor and suggested:

- transparency of how the tool worked, building in uncertainty measures (e.g. use of biotic thresholds in other locations would have a low-med level of certainty at prediction in the new area) so users did not confuse Bayesian outcomes as facts and did not misuse the tool through lack of understanding of its limitations.

- the need to demonstrate applicability in data poor areas (provision of workshops)

- the necessity to link to other tools such as the GDE Atlas and other tools end-users were currently using

- having a robust version on a website so people could try it out.

APPENDIX 3

5.1.3 Data availability for the Gngangara Mound and Blackwood River Case Studies

Hydrological and physiochemical metrics

Long-term hydrological data (1975-2011) are available for ground- and surface water levels from the Western Australian Government Department of Water (DoW), rainfall from Bureau of Meteorology and climate change predictions from the South West Sustainable Yields (SWSY) project (CSIRO 2009a,b) (Table 1). Regional groundwater models 'PRAMS' (Perth Regional Aquifer Modelling System) and 'SWAMS' (South West Aquifer Modelling System) are capable of evaluating the impacts of various factors (including abstraction, climate and various landuse practices) that have contributed to water level declines (Vogwill, 2004; Xu, 2008). Scenarios used will include predictions for 2030 based on the SWSY report and water extraction scenarios already developed using the PRAMS and SWAMS models (eg 45GL extraction from the Yaragadee aquifer). Hydrological and ecological data for the caves is available through the "Lake Cave Eco-Hydrology Recovery Project" which is supported through grant funding to Augusta Margaret River Tourism Association (AMRTA) from the Government of Western Australia's Natural Resource Management Grant Scheme (State NRM). An outline of the hydrological and physiochemical data to be used, including the length of the data collection and an indication of availability and applicability of the data in a form appropriate to the analysis required for this project, can be found in Tables 2 and 3, Appendix 3.

Table 1: Hydrological and physiochemical metrics and source of data available in appropriate format.

Variable	Past and current data encompassing spatial variability		Future prediction for Climate Change scenarios		Prediction for other stressors eg GW abstraction	
	G	BW	G	BW	G	BW
Groundwater	✓ DoW	✓ DoW	✓SWSY (CSIRO)	✓SWSY (CSIRO)	✓PRAMS (DoW)	✓SWAMS (DoW)
Surface water	✓ DoW	✓DoW	✓SWSY (CSIRO)	✓SWSY (CSIRO)	✓PRAMS (DoW)	✓SWAMS (DoW)

DoW = WA Department of Water; SWSY = South-west Sustainable Yields Project CSIRO; PRAMS = Perth Region Aquifer Management System, SWAMS = South West Aquifer Management System G = Gngangara mound, BW = Blackwood River

Table 2: Physiochemical metrics and source of data available in appropriate format

Variable	Database available and holder of data		Length of dataset		Frequency	
	G	BW	G	BW	G	BW
<i>Temperature</i>	✓ECU DoW	✓DEC Murdoch DEC	1996-2011	2005-2011 (R) 2005-2011 (R) 1998-2006 (W)	Biennial and then annual	Annual 30 samples Annual
<i>Chemical</i> (Salinity, pH, nutrients)	✓ECU DoW	✓DEC* Murdoch UWA DEC	1996-2011	2005-2011 (R) 2005-2011 (R) 2005 (R) 1998-2006 (W)	Biennial and then annual	Annual 30 samples 1 survey Annual

DoW = WA Department of Water; DEC = WA Department of Environment and Conservation Murdoch= Murdoch University; UWA = The University of Western Australia. G = Gngangara mound, BW = Blackwood River

Biological and Ecological metrics

Table 3: Biotic metrics and source of data available in appropriate format

Variable	Database available and holder of data		Length of dataset		Frequency	
	G	BW	G	BW	G	BW
<i>Riparian and phreatophytic vegetation</i>	✓ECU DoW	✓	1996-2011 1975-2011		Annual Triennial	
<i>Macro-invertebrates (wetlands)</i>	✓ECU DoW	✓ECU DEC	1996-2011	1998-2004	Biennial and then annual	Annual spring
<i>Macro-invertebrates (rivers)</i>	NA	✓DEC UWA	NA	2005-2011 2005	NA	Annual One survey
<i>Fish</i>	NA	✓Murdoch UWA	NA	2005-2011		30 surveys
<i>Amphibians</i>	✓DoW DEC UWA		1994-2010		Annual and then biannual	

NA – not applicable; DoW = WA Department of Water; DEC = WA Department of Environment and Conservation Murdoch= Murdoch University; UWA = The University of Western Australia. G = Gngangara mound, BW = Blackwood River

