Impacts of Climate Change on Water Resources in South Australia

Phase 4, Volume 2

Predicting the impacts of climate change to groundwater dependent ecosystems

DEWNR Technical report 2015/01

Government of South Australia
Department of Environment, Water and Natural Resources

Impacts of Climate Change on Water Resources in South Australia

Phase 4, Volume 2

Predicting the impacts of climate change to groundwater dependent ecosystems: An application of a risk assessment framework to a case study site in the South East NRM region: Middlepoint Swamp

Claire Harding, David Deane, Graham Green and Peter Kretschmer

Department of Environment, Water and Natural Resources

January, 2015

DEWNR Technical Report 2015/01

Department of Environment, Water and Natural Resources

GPO Box 1047, Adelaide SA 5001

Telephone National (08) 8463 6946

International +61 8 8463 6946

Fax National (08) 8463 6999

International +61 8 8463 6999

Website www.environment.sa.gov.au

Disclaimer

The Department of Environment, Water and Natural Resources and its employees do not warrant or make any representation regarding the use, or results of the use, of the information contained herein as regards to its correctness, accuracy, reliability, currency or otherwise. The Department of Environment, Water and Natural Resources and its employees expressly disclaims all liability or responsibility to any person using the information or advice. Information contained in this document is correct at the time of writing.

ISBN 978-1-922255-23-5

Preferred way to cite this publication

Harding, C, Deane, D, Green, G, and Kretschmer, P. 2015, *Impacts of Climate Change on Water Resources in South Australia, Phase 4, Volume 2 – Predicting the impacts of climate change to groundwater dependent ecosystems: an application of a risk assessment framework to a case study site in the South East NRM region – Middlepoint Swamp*, DEWNR Technical Report 2015/01, Government of South Australia, through Department of Environment, Water and Natural Resources, Adelaide

Download this document at: http://www.waterconnect.sa.gov.au

Foreword

The Department of Environment, Water and Natural Resources (DEWNR) is responsible for the management of the State's natural resources, ranging from policy leadership to on-ground delivery in consultation with government, industry and communities.

High-quality science and effective monitoring provides the foundation for the successful management of our environment and natural resources. This is achieved through undertaking appropriate research, investigations, assessments, monitoring and evaluation.

DEWNR's strong partnerships with educational and research institutions, industries, government agencies, Natural Resources Management Boards and the community ensures that there is continual capacity building across the sector, and that the best skills and expertise are used to inform decision making.

Sandy Pitcher
CHIEF EXECUTIVE
DEPARTMENT OF ENVIRONMENT, WATER AND NATURAL RESOURCES

Acknowledgements

The authors wish to thank a number of people for their contribution to this work. We are particularly grateful to Saad Mustafa (Senior Hydrogeologist, DEWNR) and Jeff Lawson (Senior Technical Officer – Hydrogeology, DEWNR) who provided their technical expertise in the selection of representative observation wells for analysis, and general oversight and comment on the groundwater hydrological components of this report. We also acknowledge Jane Chambers (and team) (Murdoch University, WA) for encouraging the use of the risk assessment framework they developed (Chambers et al. 2013a,b) for managing groundwater dependent ecosystems with declining water levels.

Peer reviews of this report were provided by Jason Nicol (SARDI), Phil Pisanu (DEWNR), and Juliette Woods (DEWNR).

<u>Personal communications:</u>

Steve Clarke (Restoration Ecologist, SE NRM DEWNR) provided his knowledge of recent hydrological manipulations and infrastructure at Middlepoint Swamp.

Abigail Goodman (Bush Management Advisor, SE NRM DEWNR) provided her knowledge of drought mitigation measures for Mosquito Creek.

Contents

List of figures

List of tables

Summary

The South Australian Department of Environment, Water and Natural Resources (DEWNR) initiated the *Impacts of Climate Change on Water Resources* (ICCWR) project in 2010 to assist in informing future sustainable use of water resources. The project was delivered in a scaled approach with four major phases: a prioritisation of the state's water resources according to the potential risk posed by climate change; the selection of future climate change projections and downscaling methodology; and the detailed analysis of the impacts of climate change on water resources (groundwater, and surface water) for priority regions of the state. The final phase of the ICCWR project (Phase 4) was to assess the impacts of climate change to water dependent ecosystems (WDE) and was delivered in two parts: Volume 1 – a statewide prioritisation of ecosystems potentially at risk from climate change; and Volume 2 – a detailed assessment of risk of climate change to a priority WDE at the site-scale. This report presents Phase 4 Volume 2 of the ICCWR project, which details the application of a risk assessment framework for identifying the risk of climate change to Middlepoint Swamp, a coastal groundwater dependent ecosystem (GDE) in the Lower South East region of South Australia.

The aim of this study was to provide, for water planning and adaptation policy purposes, an understanding of the likely impact and responses of GDEs to projected climate scenarios, along with potential management/mitigation actions. This was achieved by applying a risk assessment framework at a priority case study GDE in the South East NRM region of South Australia to firstly, demonstrate the potential level of impacts on aquatic ecosystems from climate change, and secondly, to evaluate methods and techniques for assessing and communicating risk using available datasets. The risk assessment approach was also required to adequately account for uncertainties inherent in climate and ecological modelling, and be shown to be applicable to a wide range of groundwater dependent ecosystem types.

A risk assessment framework for assessing the impacts of declining groundwater levels on GDEs developed and tested in Western Australia by Chambers et al. (2013a,b) was adapted and applied to the Middlepoint Swamp case study site. The framework followed a step-wise risk assessment approach (Figure A). Central to the framework was the construction of a conceptual model and Bayesian Belief Network (BBN), which identified and examined the cause-and-effect interrelationships between climate, hydrology, water quality and biota. The outputs of the risk assessment framework use both the BBN and spatial modelling within a geographic information system (GIS) to illustrate the variation in level of risk from modelled climate change scenarios and model uncertainties.

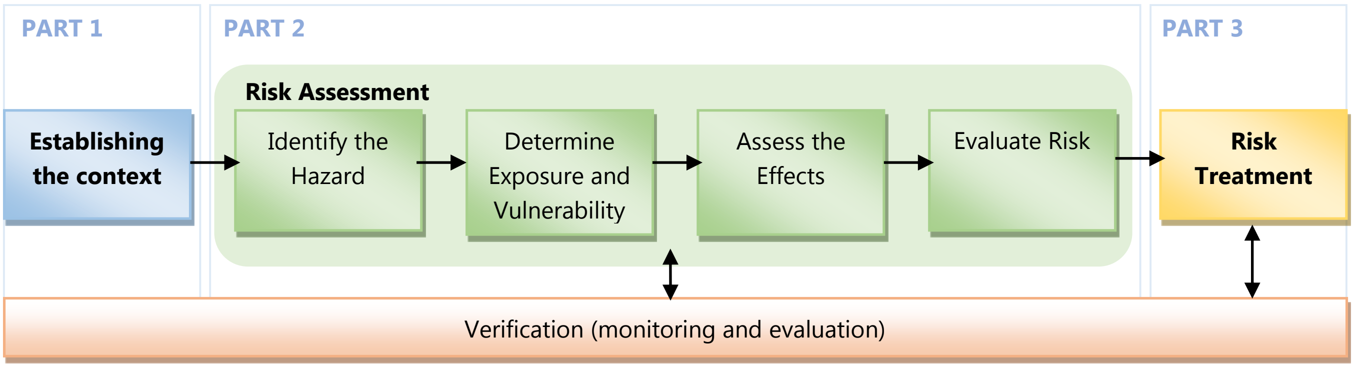


Figure A The risk assessment framework (adapted from Chambers et al. 2013a; Maxwell et al. 2012; Assante-Duah 1998)

The application of the risk assessment to the GDE case study documented in this report follows the flow diagram format presented in Figure A, where the methods and results are presented for each component of the assessment separately in a step-wise fashion. This structure allows a direct comparison between methods used by Chambers et al. (2013b) to assess case study sites, which were presented similarly, and also provides a clear description of how each component of the framework was addressed.

Application of the risk assessment framework to Middlepoint Swamp – Lower South East, South Australia

Climate change is likely to affect water quantity and quality in wetland ecosystems in multiple ways, and the future management of both groundwater and surface water resources to support dependent ecosystems requires predictions of plausible future conditions and ecosystem risk (Dyer et al. 2014). The Middlepoint Swamp case study investigated the observed effects of historic groundwater level decline and (through extension and modelling) the predicted impacts of additional climate induced groundwater decline on a groundwater dependent ecosystem. This case study examined the spatial vegetation response of aquatic plant communities, and the overall risk of the wetland transitioning to a terrestrial ecosystem (termed terrestrialisation herein), as a result of the hazard of further declines in groundwater levels, and hence declines in surface water depth and duration as a result of climate change predictions.

The exposure and vulnerability of Middlepoint Swamp GDE to a change in groundwater level as a result of reduced rainfall was determined by developing statistical relationships between groundwater levels, rainfall, and surface water levels and hydrograph characteristics (i.e. depth and duration) within the wetland from existing monitoring data. This was achieved by: modelling the projected change in groundwater levels into the future under selected climate change scenarios using an analysis tool called the Hydrograph and Rainfall Time Trend (HARTT) model; determining the relationships between groundwater and surface water dynamics within the wetland; and projecting the resulting change in surface water levels due to projected changes in groundwater spatially across the assessment area with use of a digital elevation model (DEM). Future climate scenarios modelled included four combinations of Global Climate Models (GCMs) and greenhouse gas concentration pathways representing a range of 'best case' and 'worst case' climate scenarios for the region to 2030.

The results of hydrological modelling and projecting the outcome of groundwater level decline on surface water levels (SWL) in Middlepoint Swamp indicated that under 2030 climate change scenarios, the maximum SWL in the wetland was predicted to be reduced by 0.46 to 0.98 m, the worst case essentially representing an almost total drying of the wetland. Changes in surface water area (area inundated) between the historic (1978) and current (2013) water levels of approximately -0.28m resulted in a relatively minimal net change in inundated area within the assessment area of -7.04%. Climate change scenarios to 2030 indicate more significant changes in net wetland area where reductions in predicted area inundated for the four scenarios modelled were between -26.63% (best case scenario) to -84.97% (worst case scenario).

To determine the effect of declining groundwater levels to vegetation characteristics (i.e. spatial distribution of vegetation communities) of Middlepoint Swamp, an understanding of the relationship between the wetland hydrology and the water requirements of vegetation communities and transition probabilities was developed and represented as a conceptual model of eco-hydrological function for the site (Figure B). Spatio-temporal trend analysis was conducted within a GIS to determine the historical observed effects of water level change on wetland vegetation between two years: 1978 to 2013. This trend analysis was used to determine vegetation community transition probabilities under an observed drying scenario. Vegetation mapping from the spatio-temporal analysis, along with a DEM and water level monitoring data were used to create hydrological niche models based on average annual hydro-period for each mapped community. These data and modelled outputs, along with the conceptual model, were then used to create and populate a BBN for which the probability of different vegetation communities being observed for hydroperiods corresponding to future projections in climate were determined and overall risk of terrestrialisation of the ecosystem was assessed (Figure B).

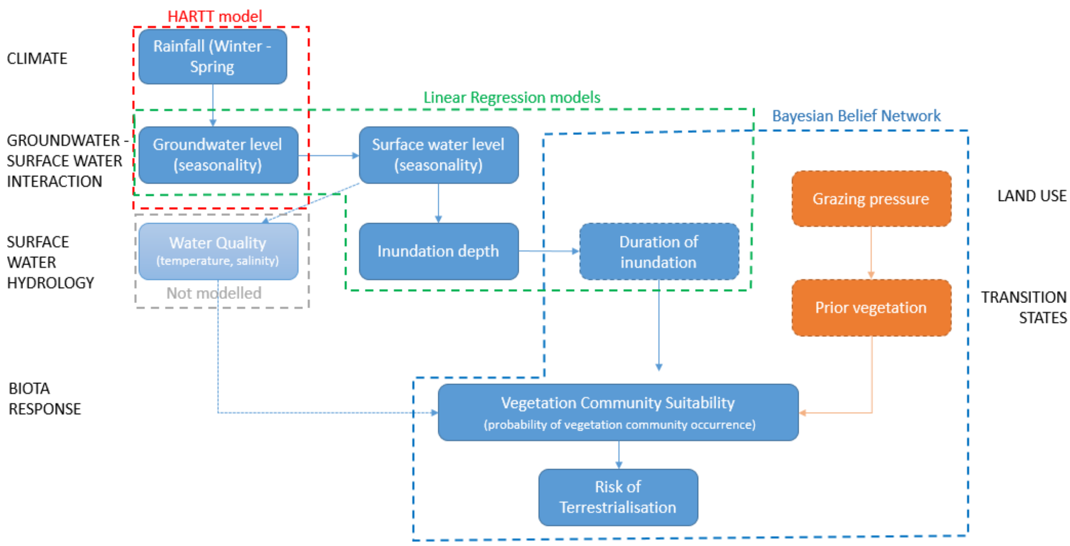


Figure B The conceptual model of hydrological function for Middlepoint Swamp, and modelling approach

The results of the application of the BBN modelling indicated that Middlepoint Swamp was largely (between 47 – 70% of the study extent) at moderate risk (25 – 50% probability) of terrestrialisation by 2030 under the modelled climate change scenarios, but indicated large scale vegetation community changes towards more dry adapted species in response to changed hydrology over the same time period. The highest water requirement vegetation communities were no longer predicted to occur in any of the 2030 scenarios, being replaced almost wholly by brackish herbland and exotic pasture grasses. The predicted vegetation communities for the worst case, high emissions climate change scenario are reduced almost entirely to terrestrial ecosystems. Uncertainties in the outputs of the BBN model are however generally high, with uncertainty in the identity of predicted vegetation communities for projected climate change scenarios particularly so - the majority of the assessment area for all three projections was identified as high uncertainty (<50% probability of the predicted vegetation community occurring). Given the compounding levels of uncertainty in climate modelling, determining groundwater and surface water relationships, and vegetation response, the generally high level of uncertainty of the modelling was expected.

Whilst the application of the risk assessment to the Middlepoint Swamp case study site identified significant negative ecological impacts for wetlands dependent on regional groundwater in the South East, a number of management intervention options were identified to adaptively manage for climate change impacts. These included improved provisions for environmental water requirements and adequately addressing the effects of climate change on water resources in future water allocation planning. Given that Middlepoint Swamp is a relatively permanent coastal discharge site, of greater than 1 m maximum water depth, it could be assumed that impacts on shallower GDEs with seasonal groundwater interactions could be far more severe. As a result, landscape scale management of some types of wetland ecosystems transitioning to terrestrial ecosystems may be inevitable despite land and water management interventions. The landscape scale implications for large-scale wetland loss could be significant under climate change scenarios for the South East. As a result, identification, prioritisation and protection of resilient ecosystems, as well as potential surface water augmentation from existing regional drainage infrastructure and restoration of water levels using weirs could prove to be an important adaptation strategy for protecting important wetlands in the region into the future.

Overall, the risk assessment framework developed by Chambers et al. (2013a) provided a robust and systematic method of assessing the impacts of climate change on GDEs as demonstrated by the application at Middlepoint Swamp, and other case study sites (Chambers 2013b). A major strength of the framework was its capacity to relate climate, hydrology and ecosystem response in a single tool (with use of a BBN) and the adaptability of the framework to suit a wide array of data availability and capacity, whilst clearly communicating model uncertainties and limitations. The application of any framework to predictively model and assess risk from climate change is, however, reliant on both sufficient monitoring data and ecological expert knowledge. Significantly both of these elements are becoming scarcer, contributing to the inability of managers to adequately consider and represent the environment in policy (Lindenmayer et al. 2014). By appropriately designing programs to include critical examination of the on-ground effectiveness of management and policy actions, together with long-term monitoring of key ecosystems, appropriate information to inform analysis (such as that presented in this study) can be produced to inform future decision-making and policy.

1 Introduction

Climate change is acknowledged as a potential threat to the future of South Australia's water security. The Water for Good plan identified climate change as a major challenge to water resources in most of South Australia's Natural Resources Management (NRM) regions.

The Commonwealth Scientific and Industrial Research Organisation (CSIRO) and Bureau of Meteorology (BoM) have previously undertaken investigations which project the likely impacts of climate change on South Australia (Suppiah et al. 2006; CSIRO 2007). Their projections indicate that through the 21st century, South Australia may be subject to:

- increased temperatures and reduced rainfall
- increased rainfall variability
- increased evaporation
- significantly increased frequency and severity of drought
- changes in the frequency of extreme weather events, including flooding.

In the southern agricultural area of South Australia, annual rainfall is projected to decrease by up to 10–15 % by 2030 and up to 25–30 % by 2070 (CSIRO 2007). Along with increased evaporation, these climatic variables will have significant impacts on the region's water resources and associated aquatic ecosystems, with subsequent consequences for sustainable water allocations (Harding 2012a).

Reduced rainfall can result in both direct and indirect impacts on groundwater resources, where 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 & Stephanopolous 2011; McFarlane et al. 2012). Declines in groundwater levels have long been associated with negative effects on groundwater dependent ecosystems (GDEs), resulting in changes to wetland water balance, leading to lower surface water level (depth, duration and frequency of inundation) and reduced groundwater inflow (Kløve et al. 2013). Where groundwater levels decline dramatically without recovery, potential disconnection from groundwater sources can occur (Cook et al. 2008). Increases in surface water temperatures, salinity and nutrients (Nielsen & Brock 2009) are also resultant of changed water regimes and other impacts of climate change.

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 even fewer on dependent ecosystems (Kløve et al. 2013). Relatively little is known about how climate change will affect GDEs and their biota, and this lack of knowledge hinders the adaptive management of GDEs in the face of climate change impacts. Any management decisions require consideration of potential conflicts between human resource use and GDE biodiversity. Significantly, GDEs tend to support high biodiversity and levels of endemism (Horwitz et al. 2009; Kløve et al. 2014), thus being of considerable conservation value.

In efforts to address the lack of knowledge of the impacts of climate change on surface water, groundwater, and GDEs, DEWNR initiated the *Impacts of Climate Change on Water Resources* (ICCWR) project in 2010 which has modelled the response of groundwater and surface water to climate change in a number of regions (Green et al. 2011; Green et al. 2012; Alcoe et al. 2012; Gibbs et al. 2012), and identified priority regions for assessment of impacts on water dependent ecosystems (Harding 2012). Additionally, a risk assessment framework has been developed and tested as a decision-making framework for managing GDEs with declining water levels due to climate change in Western Australia (Chambers et al. 2013a,b).

This report represents an application of the risk assessment framework developed by Chambers et al. (2013a,b) to a case study wetland GDE identified as being at high risk from the impacts of climate change (Denny et al. 2014) within a high priority region of South Australia (Lower South East, SA) identified by Harding (2012). This study is presented as Volume 2 of Phase 4 (assessing the impacts of climate change to water dependent ecosystems) of the South Australian ICCWR project and documents the development and evaluation of spatially explicit models of wetland vegetation response to historical, current, and projected future climate change scenario water levels at the GDE case study site.

1.1 Previous work of the ICCWR project

The DEWNR project *Impacts of Climate Change on Water Resources* (ICCWR) was established in 2010 under the *New Knowledge for the Future* component of DEWNR's Groundwater Program. This project was developed to address Target 75 of South Australia's Strategic Plan (2011) which requires that "South Australia's water resources are managed within sustainable limits by 2018" and also supports the achievement of Target 62 of the 2013 South Australian Strategic Plan "Climate change adaptation: Develop regional climate change adaptation plans in all State Government regions by 2016". The studies conducted by the ICCWR project will ultimately fulfil Action 43 of the Water for Good plan: "Commission, where required, regional scale studies on the Impacts of Climate Change on Water Resources".

The ICCWR project contributes towards the fulfilment of Object 3(a) of the *Natural Resources Management Act 2004* which states '*decision-making processes should effectively integrate both long term and short term economic, environmental, social and equity consideration*'. Climate change forecasting is also identified as a research priority under Goal One of the State NRM Plan (2006). The South East Natural Resources Management Plan (SENRMB 2010) identifies impacts of climate change as a key risk factor to surface water and groundwater resources and to the biodiversity of the region.

The ICCWR project was delivered in a scaled approach which included:

- Phase 1: first order assessment which prioritised groundwater and surface water resources according to the potential risk posed by climate change (Wood & Green 2011). This assessment was undertaken across the state using regions (i.e. NRM regions, prescribed water resource areas) as the base unit of assessment. It was weighted to values associated with the public, irrigation and industrial supply of water as well as environmental requirements.
- Phase 2: a selection of climate change projections and the downscaling methodology for climate models for South Australia was developed (Gibbs et al. 2011).
- Phase 3: applied the downscaled climate projection models to develop recharge and runoff models for climate change scenarios in priority regions identified in Phase 1 (Green et al. 2011; Green et al. 2012; Alcoe et al. 2012; Gibbs et al. 2012).
- Phase 4: the assessment of impacts of climate change on water dependent ecosystems (WDE). Similar to the Phase 1 prioritisation of water resources, a State-wide assessment of the relative risk of climate change to WDEs was applied using mapped WDEs (i.e. wetlands and watercourses) as the base unit of assessment (Harding 2012). This work identified priority regions and ecosystems within the State where WDEs were at most relative risk from the impacts of climate change. From this first order assessment, a priority region was chosen for this study (the South East NRM region), and a high priority/high risk ecosystem selected as a case study site for applying a detailed assessment framework of the impacts and risks of climate change.

A risk assessment framework for assessing the impacts of declining groundwater levels on GDEs was developed by Chambers et al. (2013a,b), based on a standard risk assessment protocol which was designed and intended to be broadly applicable elsewhere. The development of the framework was unrelated to the SA ICCWR project, although the timely completion of the framework, and rigorous testing already undertaken (Chambers et al. 2013b), made the use of the framework for the purposes of the ICCWR Phase 4 site scale assessment ideal.

The relationships between the various phases of the ICCWR project are shown in Figure 1.1.

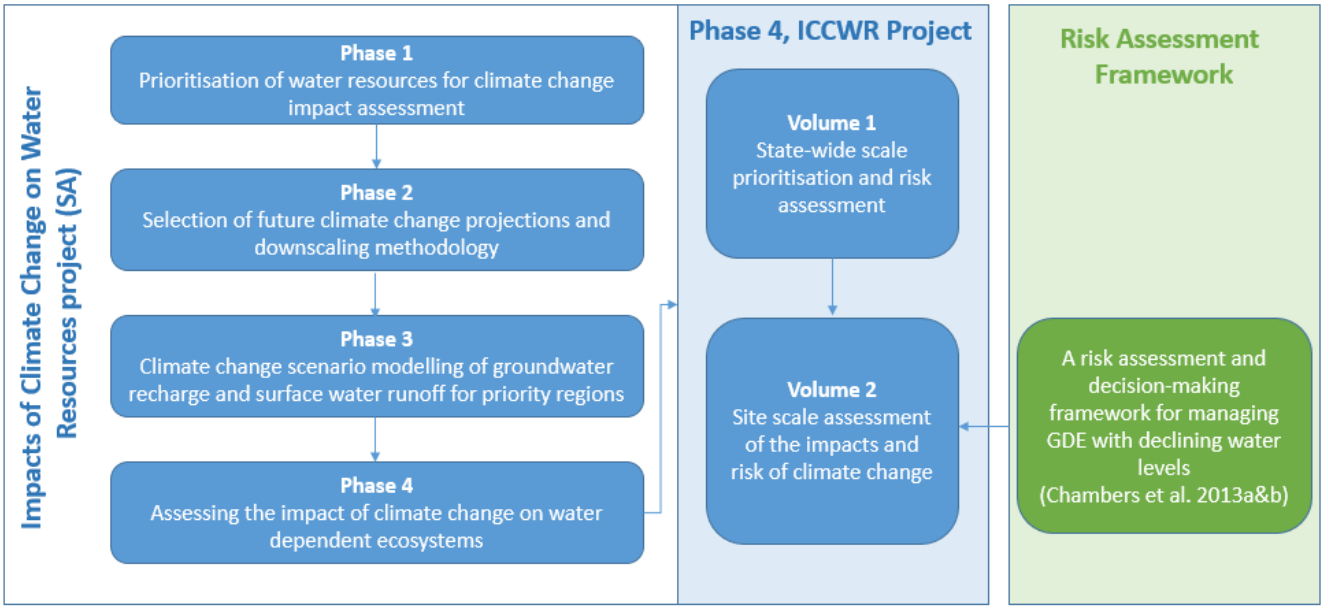


Figure 1.1 Flow diagram demonstrating the relationships between the various phases of the ICCWR project in the context of this study

1.2 Aims and objectives

The aim of this study was to provide, for water planning and adaptation policy purposes, an understanding of the likely impact and responses of GDEs to projected climate scenarios, including potential management/mitigation actions. This was achieved by applying a risk assessment framework at a priority case study GDE in the South East to firstly, demonstrate the potential level of impacts on aquatic ecosystems from climate change, and secondly, to evaluate methods and techniques for assessing and communicating risk using available datasets. The risk assessment approach was also required to adequately account for uncertainties inherent in climate and ecological modelling, and be shown to be widely applicable to a range of ecosystem types. Findings from the application of the risk assessment at the case study site were intended to be able to inform other likely impacts on similar GDEs within the region.

The specific objectives to meet the overall aims were to:

- trial the application of the risk assessment framework prepared by Chambers et al. (2013a) to a case study GDE in South Australia, and to evaluate methods and efficacy with limited available datasets
- depict spatially explicit models of ecosystem response to historical, current, and projected future climate change scenario water levels at a GDE case study site
- evaluate the overall risk of the case study GDE to climate change impacts
- identify management and risk mitigation options
- to develop methods within the risk assessment framework that could demonstrate the usefulness of the data outputs of the ongoing South East GDE Monitoring Program (SKM 2010).

1.3 Risk assessment framework and report structure

A standard risk assessment framework and guidelines for application was developed by Chambers et al. (2013a) specifically to address the hazard of declining groundwater levels to GDEs as a result of climate change. The framework was designed to be transferrable elsewhere, and workshops were held across Australia (Adelaide, Canberra and Brisbane) in March 2012 to refine the framework for wider use (Chambers et al. 2013a).

The framework was based on a standard risk assessment protocol (Assante-Duah 1998), and was tested on GDE case study sites in Western Australia at varying scales and with varying levels of data availability (Chambers et al. 2013b). Adaptations to this framework were made to align with the terminology of existing risk frameworks for water planning in South Australia (Maxwell et al. 2012), although the components and application of the framework in this study remain similar to that presented by Chambers et al. (2013a) (Figure 1.2).

The application of the risk assessment to the GDE case study documented in this report follows the flow diagram format presented in Figure 2.1, where the methods and results are presented for each component of the assessment separately in a step-wise fashion. This structure allows a direct comparison between methods used by Chambers et al. (2013b) to assess case study sites, which were presented similarly, and also provides a clear description of how each component of the framework was addressed.

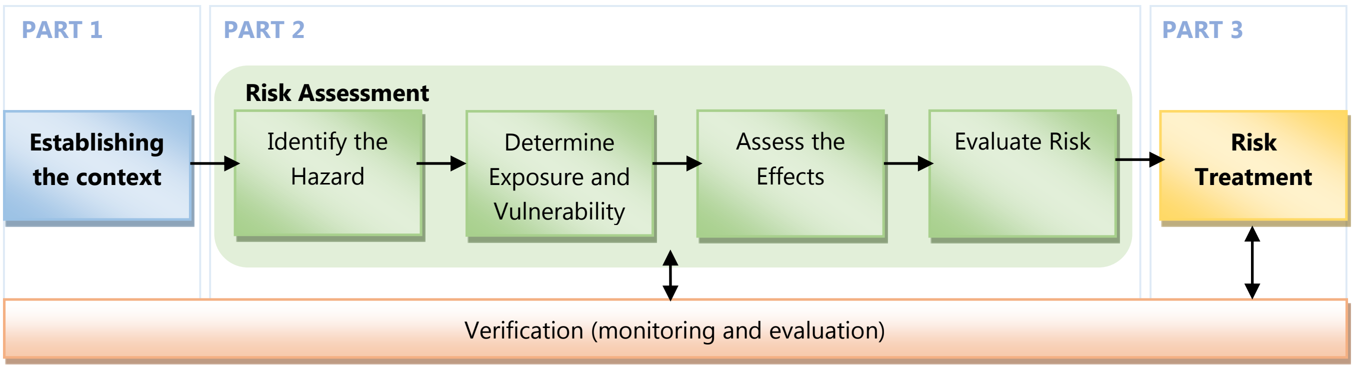


Figure 1.2 The risk assessment framework (adapted from Chambers et al. 2013a; Maxwell et al. 2012; Assante-Duah 1998)

Detailed guidelines and examples for application of the risk assessment framework are provided in Chambers et al. (2013a), and were used to guide the application presented in this report. A brief summary of the framework description is therefore provided here.

The risk assessment method was undertaken in three parts (Figure 1.2):

<u>Part 1</u>: Establishing the context – which involved identifying the objective for carrying out the risk assessment, management issues and the ecological and hydrogeological nature of the GDE being assessed. The spatial and temporal boundaries of the risk assessment are also defined.

<u>Part 2:</u> Application of the risk assessment and decision-making framework - This was applied in four steps: identify the hazard; determine exposure and vulnerability; assess the effects; and evaluate the risk. Specific details of the methodology applied to each of the four steps are dependent on data availability and are detailed within Section 4 of this report. Verification of modelling outputs occurred throughout the development of the risk assessment. Table 1.1 further details each of the steps within Part 2 of the risk assessment.

<u>Part 3:</u> Risk treatment – The framework provides a number of tools and outputs to support climate adaptation and management decisions at each step. Part 3 of the risk assessment articulates potential risk treatment and mitigation based on the findings. Risk treatment is linked to monitoring and evaluation, where any future application of mitigation responses should be monitored and evaluated for effectiveness. This requires an on-going commitment to monitoring and evaluation of high priority GDEs in order to be able to adaptively manage for ecosystem water requirements.

Table 1.1 Components of Part 2 of the Risk Assessment Framework (adapted from Chambers et al. 2013b)

Risk Assessment Component	Sub-tasks	Description
Step 1: **Identify the hazard**	• Identify primary and secondary hazards • Determine the cause of the primary hazard • Define spatial and temporal boundaries	The framework is designed to manage the hazard of declining groundwater levels. This step identifies the primary and secondary hazards and explores and identifies the cause of declining groundwater levels. The spatial and temporal boundaries of the assessment are largely determined by the management question and the available data.
Step 2: **Determine exposure and vulnerability**	• Determine the projected change in groundwater levels under climate change scenarios • Determine the relationships between groundwater and surface water dynamics within the wetland • Project the resulting change in surface water levels due to projected changes in groundwater levels	The magnitude and the rate of groundwater decline is determined spatially and temporally (historically, currently, and projected into 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 the effects**	• Collate existing data and field survey • Development of a conceptual model	A conceptual model of ecosystem function is developed based on available data and/or field survey, which describes 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: **Evaluate risk**	• Develop a predictive model using Bayesian Belief Networks (BBN) • Link predictive models to Geographic Information Systems (GIS) • Display spatial analysis of ecosystem response and risk	A number of techniques (expert opinion, statistical analysis) are used to determine the tolerance limits or thresholds of the key drivers to biotic change. The conceptual model, with key drivers now quantified by thresholds, are now incorporated into BBNs that can be easily modified to show changes in probability of risk resulting from these interactions. The outputs of the BBN can then be mapped spatially using GIS.

2 Case study site selection

Due to the time restrictions of the project, a single case study site was selected within a high priority region in South Australia identified by Harding (2012) to apply the risk assessment framework to. The Lower Limestone Coast Prescribed Wells Area was chosen given the large number of significant GDEs (over 12 000) and the availability of existing monitoring data and local expert knowledge for some key GDE systems (SKM 2009 & 2010) to support modelling applications. An initial regional scale wetland prioritisation was undertaken for the South East NRM region by Denny et al. (2014) based on wetland ecosystem value in addition to the potential risk from climate change that was presented by Harding (2012). This assessment identified 393 wetlands of very high priority (Denny et al. 2014) within the region.

From the high priority sites identified by Denny et al. (2014), a short-list of sites was developed for site-scale application of the risk assessment framework, which were considered to represent the range of GDE wetland types in the SE region, and met the following criteria:

- Existing groundwater and surface water monitoring, including a minimum of one gaugeboard and one suitable groundwater observation well, with a minimum of two years complete dataset;
- Established conceptual model of the type of groundwater dependency of the site;
- Existing vegetation monitoring and/or mapping data available;
- Nearby observation well of suitable construction with a long-term monitoring record (1970s – present) that can demonstrate a relationship with historical rainfall data;
- A good representative of a wetland type in the South East;
- A known historical change in groundwater conditions that was likely to have influenced the wetland biota within a time period (ideally around the late 1980s or early 1990s) which would allow for a discernable change in vegetation community with imagery and remote sensing limitations (i.e. over a time period where suitable quality imagery was available).

A total of 14 wetland GDE sites were identified as meeting the minimum hydrological monitoring data requirements, of these four met all the above selection criteria, and included Middlepoint Swamp, Bool and Hacks Lagoon, Trail Waterhole and Topperwein, and Deadmans Swamp (Table 2.1). Deadmans Swamp was identified as already significantly impacted by declining groundwater levels, having remained dry since approximately 2006, and therefore further analysis of a drying climate was unlikely to produce results not already realised at this site. Middlepoint Swamp was selected from the three remaining sites after consideration of the completeness of the monitoring records of all three sites, and the complexity of the ecosystems. However, the framework and methods developed and presented in this report could conceivably be applied to any of the 14 sites, and any further work could focus on the short-listed wetlands in Table 2.1.

Table 2.1 GDE case study site selection – short-list of priority sites

Priority GDE name	Wetland type	Minimum groundwater and surface water monitoring	GDE conceptual model	Vegetation monitoring data available	Nearby long-term groundwater monitoring with seasonal rainfall response	Historical groundwater level change (>0.8 m, between 1980 -95)
Middlepoint Swamp	Coastal Peat Brackish Swamp	✓	✓	✓	✓	✓
Bool and Hacks Lagoon	Grass Sedge Wetland	✓	✓	✓	✓	✓
Trail Waterhole and Topperwein	Grass Sedge Wetland	✓	✓	✓	✓	✓
Deadmans Swamp	Grass Sedge Wetland	✓	✓	✓	✓	✓
Lake Hawdon South	Inland interdunal wetland	✓	✓	✓	✓	
Honans Complex	Grass Sedge Wetlands	✓	✓	✓		✓
Lake Robe	Saline Lake	✓	✓	✓	✓	
Big Dip Lake	Coastal Dune Lake	✓	✓	✓	✓	
Ewens Ponds	Karst	✓	✓		✓	✓
Lake Frome	Coastal Dune Lake	✓	✓		✓	
The Marshes	Grass Sedge Wetlands	✓	✓			✓
Butchers Gap	Saline Swamp	✓	✓	✓		
Kangaroo Flat	Grass Sedge Wetland	✓	✓			
Freshwater Lake	Coastal Dune Lake	✓	✓			

Case Study Site: Middlepoint Swamp GDE, Lower South East, SA.

3 PART 1: Establishing the context

3.1 Management issues

Groundwater is the predominant water resource in the South East region of South Australia, with the majority of water for economic and domestic activity allocated from the regional unconfined aquifer (herein referred to as the Tertiary Limestone Aquifer – TLA). The TLA consists mainly of calcareous sandstone and limestone and varies in thickness from~10 m north-west of Mount Gambier, increasing to more than 300 m thick south of Mount Gambier (Brown et al. 2001). Depth to groundwater in the TLA is generally shallow with the regions wetland ecosystems largely supported by the interaction of groundwater and surface water. The majority of the 16 000+ wetlands in the region (77% by number and 96% by area) have been identified as having a high likelihood of interaction with the TLA (SKM 2009) – referred to as groundwater dependent ecosystems (GDEs).

Groundwater levels in the TLA respond rapidly to contemporary rainfall. This is reflected in most groundwater hydrographs and was highlighted by Brown et al. (2001) who demonstrated a strong relationship between decreases in average annual rainfall and declining water levels measured in observation wells in the TLA. Increased extraction from the TLA, combined with declines in rainfall since the 1970s has resulted in widespread drawdown of groundwater across the region (DFW 2010). Increases in the reliance on groundwater extraction for anthropogenic use as a result of reduced rainfall is a trend not unique to the South East region, and has been witnessed in other parts of Australia where the reliability of groundwater and the reduced availability of surface water have led to increased extraction (McFarlane et al. 2012). The close relationship between climate and groundwater levels in the TLA will (with reduced rainfall as a result of climate change), continue to have a negative impact on the groundwater resources, and hence GDEs, of the South East region (Brown et al. 2006).

There is clear and growing evidence that GDEs are at risk in the South East as a result of falling groundwater levels (Cook et al. 2008; DFW 2010). Preliminary assessment of arbitrarily assigned 0.3 m decrease intervals in groundwater levels in relation to total wetland depth at 63 focus GDEs across the South East region indicated that a permanent drop of 0.6 m would lead to the complete loss (or loss of groundwater connectivity contributing to surface water inundation) of over one third of all the wetlands examined (DFW 2010). A reduction in water levels by 1.5 m was shown to have potential to cause the loss of all but the deepest wetlands. However, the DFW (2010) analysis did not attempt to realistically represent the effects of any particular future groundwater scenario, and crudely interprets the relationship between surface water and groundwater. Despite the limitations, this preliminary assessment surmised that even relatively small drops in groundwater level would result in the loss or degradation of wetland ecosystems (DFW 2010). If this is the case, it will become increasingly important into the future to be able to identify and predict the likely impact of reduced rainfall on groundwater resources, and therefore risks to GDEs. Knowledge of likely impacts and risk can then be used to manage resources adaptively to provide and plan for the provision of environmental water requirements for key GDEs, and/or to predict the scale of likely impacts, prioritise risk remediation, and manage for potential terrestrialisation (transition to terrestrial ecosystems) of wetland ecosystems where no remediation options are available.

The risk assessment framework was trialed at Middlepoint Swamp GDE located in the Lower Limestone Coast Prescribed Wells Area (Figure 3.1), where groundwater levels have shown long-term declines in response to reduced rainfall, and increased extraction for irrigation (Harding 2012b). Figure 3.1 shows the direction of groundwater flow in the TLA, as well as areas of centre pivot irrigation development observable from the 2013 aerial photo imagery. This was an area which was identified as a groundwater extraction 'hotspot' by Harding (2012b) and Harding & O'Connor (2012).

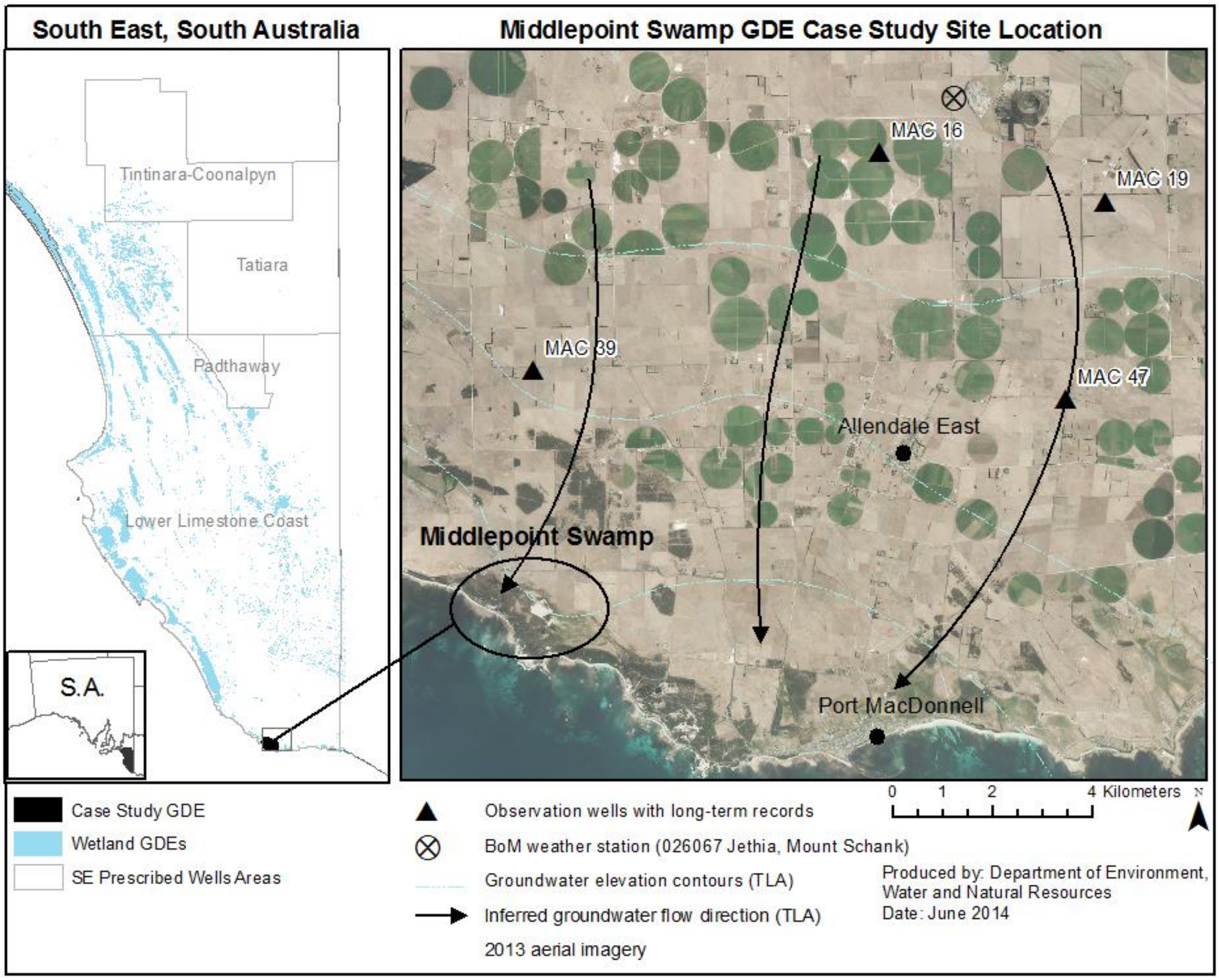
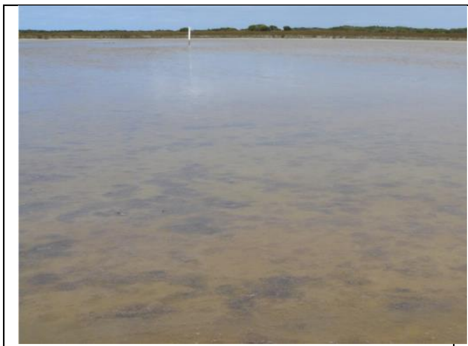


Figure 3.1 Location of Middlepoint Swamp GDE case study site, South East, SA

3.2 The nature of the ecosystem

Middlepoint Swamp is a seasonal fresh – brackish (~3500 – 5000 EC) coastal rising spring wetland located approximately 6.5 kilometres west of Port MacDonnell and lies directly behind the fore-dunes. The wetland covers an area of ~171 ha, and is dominated by brackish sedges (*Juncus kraussii*, *Gahnia filum*, *Baumea juncea* over *Distichlis distichophylla*), brackish herblands (*Sarcocornia quinqueflora*, *Selliera radicans*, *Samolus repens*, *Mimulus repens, Triglochin striata*), with an open water clay basin dominated by charophytes and *Ruppia polycarpa* (Figure 3.2). Localised freshwater springs within the wetland are dominated by *Leptospermum lanigerum*. The site is grazed restrictively by cattle from the landward side of the wetland (which is fringed by pasture grasses), and largely ungrazed from the dune-ward side, where the wetland is fringed by dense *Ficinia nodosa*, and coastal shrubland (*Leucopogon parviflorus, Ozothamnus ferrugineus*) (Figure 3.2). The wetland is of high ecological value as a wader and waterfowl refugia, supports several aquatic dependent threatened species (Southern Bell Frog (v, V), Golden-headed Cisticola (r), Southern Emu wren (r), Brolga (v), Swamp Antechinus (e), Australasian Shoveler (r), *Brachyscome graminea* (r)[1]), and is considered representative of brackish coastal dune wetlands of the Lower South East (SAWID; Ecological Associates 2010; Beacon Ecological 2010). The near-permanent nature of the deepest areas of the wetland basin are important habitat and refugia for waders, and up to four Brolgas (likely a pair and the last seasons juveniles) are known to use the site intermittently during the summer months.



Charophytes / *Ruppia polycarpa* open water

Sarcocornia quinqueflora, *Samolus repens*, *Selliera radicans, Triglochin striata* brackish herbland

Dense *Juncus kraussii*

Gahnia Filum fringing sedgeland

Dense *Ficina nodosa* with *Leucopogon parviflorus* / *Ozothamnus ferrugineus* fringing shrubland (ungrazed wetland margin)

Fringing pasture grasses (grazed wetland margin)

Figure 3.2 Major vegetation communities present at Middlepoint Swamp (March 2014)

The wetland reaches a seasonal maximum depth of approximately 1 m, and maintains inundation for approximately 8 to10 months annually. The depth of inundation has been reduced from historical levels by two artificial drains that have been cut through the coastal dunes to the sea (exact date unknown, pre-1960s). A regulator has been installed in 2011 in the main outlet drain to increase the depth and duration of inundation at the site.

Middlepoint Swamp is monitored as part of the GDE monitoring project established by the former Department for Water (SKM 2010; Beacon Ecological 2010). As such, the site is equipped with groundwater and surface water monitoring infrastructure and vegetation monitoring transects with data available from approximately 2009.

SKM (2010) produced a preliminary conceptual model of groundwater interactions of Middlepoint Swamp with the TLA in the 2009–10 monitoring period (Figure 2.3). Depth to groundwater was shown to be less than zero in the deepest part of the wetland during both periods of maximum and minimum groundwater levels during 2009–10. When the groundwater level was at its highest, levels appear to be over one metre above the wetland base, strongly suggesting the wetland to be gaining from the TLA during this period (SKM 2010). During periods of minimum groundwater levels the wetland was recorded as being damp, which may indicate that groundwater is likely to support wetland ecology during drier periods (SKM 2010).

[1] V = EPBC Act listed vulnerable; SA state threatened status: v = vulnerable; r = rare; e = endangered

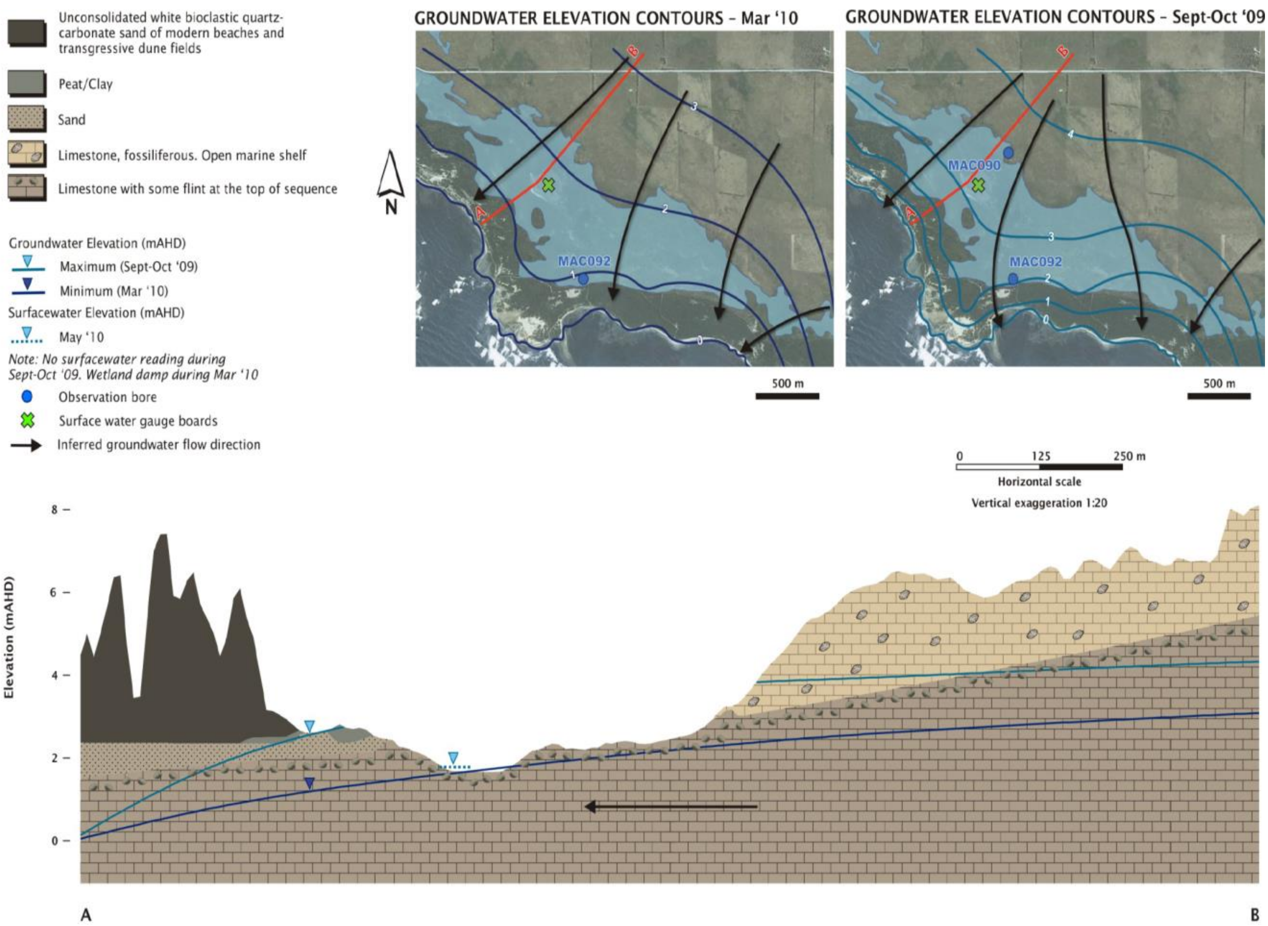


Figure 3.3 Middlepoint Swamp conceptual diagram of groundwater interaction with the TLA for the 2009-10 monitoring period (sourced from SKM 2010)

Groundwater discharge from the TLA is thought to control the quality, depth and duration of inundation of water in the wetland (Ecological Associates 2010; SKM 2010), and is therefore vulnerable to declining groundwater levels.

3.3 Aims of the risk assessment for Middlepoint Swamp

The aim of applying the risk assessment framework at the Middlepoint Swamp GDE case study site was to trial methods of predicting the likely ecological response to future groundwater levels based on projections of future rainfall scenarios, and with use of existing datasets. Total risk to the ecosystem was assessed as an overall risk of terrestrialisation (i.e. the risk that the wetland would no longer support aquatic vegetation communities, resulting in a complete conversion to a terrestrial ecosystem). Risks and predicted hydrological and ecological responses identified through this project can then be used to inform the likely risk of other GDEs interacting with the TLA, and potential implications for NRM in the region.

Application of the risk assessment framework at one of the monitored GDE case study sites in the South East region (SKM 2010) also provided an example of how existing monitoring datasets for GDEs can be used in such predictive modelling, reinforcing the value of ongoing, and targeted monitoring of high value ecosystems vulnerable to the impacts of climate change.

3.4 Define temporal and spatial boundaries of the assessment

3.4.1 Temporal boundaries

The risk assessment was constrained to three discrete epochs:

1) Historical (1978)
2) Current (2013)
3) Selected climate change projections to 2030.

The historic and current epochs were chosen as being reflective of groundwater levels within the TLA either side of a pronounced step-change (abrupt change in groundwater level attributed to declining rainfall and increasing extraction) from ~1992. Due to the lack of ecological data for the historic time period, the specific years (1978 and 2013) were chosen due to availability of aerial photography which was used to determine vegetation community distributions prior to the pronounced change in groundwater level (historic), and with the use of recent vegetation monitoring and groundtruthing, for the 2013 (current) epoch.

3.4.2 Spatial boundaries

The spatial area of assessment within Middlepoint Swamp was limited to a sample cross-section area which allowed extrapolation of groundwater, surface water, and vegetation community data to a higher level of accuracy and certainty. This also avoided complex freshwater spring communities for which no data were available, and limited the complexity of vegetation communities to be modelled.

The cross-section consisted of three 918 m transects, crossing the wetland basin perpendicular to topographic contours, bisecting the deepest part of the wetland basin, and extending into terrestrial vegetation associations at both extremities (Table 3.1 and Figure 3.4). The middle transect (Transect 2), is aligned with the existing GDE monitoring project infrastructure, groundwater well (MAC090) and surface water gaugeboard. Transects 1 and 3 were placed 50 m either side of Transect 2 (Figure 3.4). A 2 x 2 m raster grid within the cross-section area was used as the basis for model development and spatial outputs, and matches the existing 2 m LiDAR derived DEM (digital elevation model) available for the South East region.

Table 3.1 Spatial coordinates of the transects

	Start		Finish	
Transect	**Easting**	**Northing**	**Easting**	**Northing**
1	467442	5791063	466780	5790438
2	467476	5791028	466815	5790405
3	467510	5790994	466848	5790369

GDA 1994 MGA 54

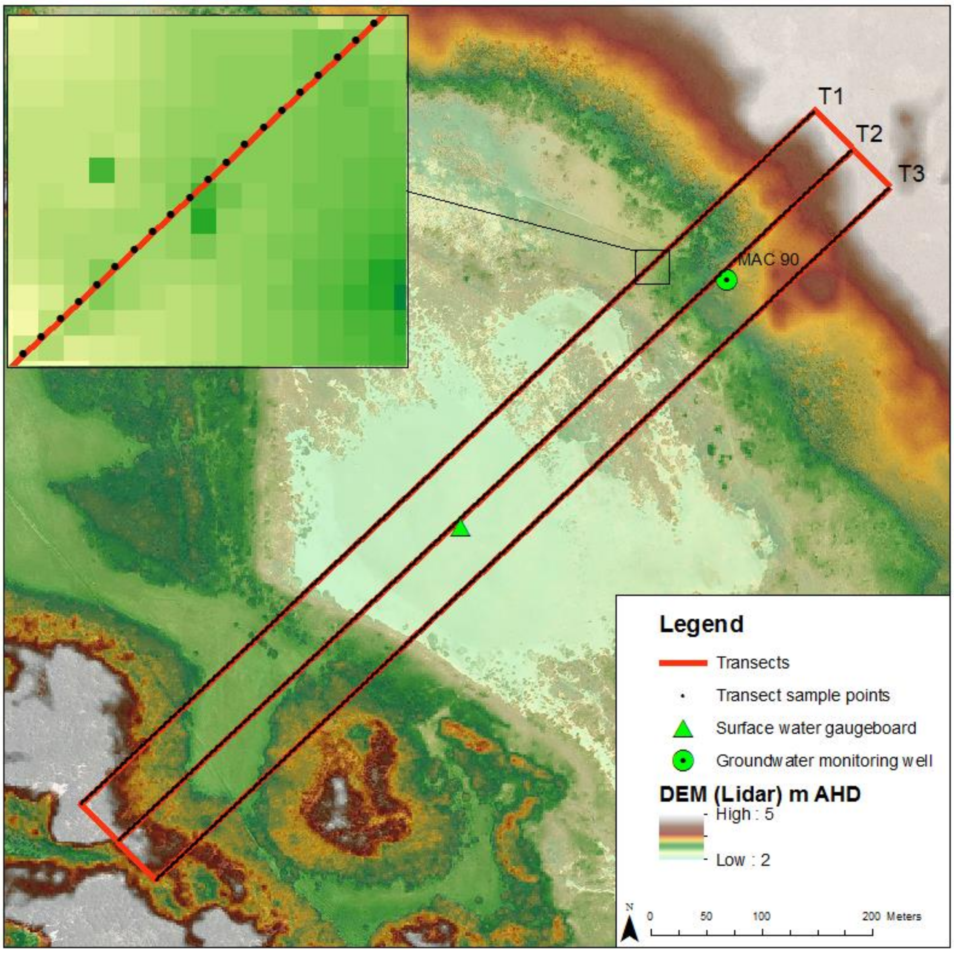


Figure 3.4 Spatial boundaries of the cross-section assessment area – Middlepoint Swamp

4 PART 2: Application of the risk assessment framework to the Middlepoint Swamp case study site

4.1 Step 1: Identify the hazard

4.1.1 Primary and secondary hazards

Primary hazard: The primary hazard to Middlepoint Swamp as a result of climate change is groundwater level decline in the TLA due to a reduction in rainfall, and increased groundwater extraction for human use. Given the strong interaction between surface water and groundwater inferred at the site from conceptualisation of the 2009-10 monitoring data (refer to Figure 3.3), a decline in groundwater level is likely to result in reduced seasonal groundwater discharge, and 'drier' dry periods if the wetland becomes disconnected from the TLA seasonally.

Middlepoint Swamp is also a coastal GDE and due to close proximity (~300 m) to the coastline and low elevation (below 2m AHD), has been identified by Mustafa et al. (2012) as being at risk from the effects of climate change induced sea level rise. A preliminary assessment of the potential hazard posed by sea level rise by applying the Bruun Rule analytical dune recession model (Bruun 1962) at Middlepoint Swamp was completed, and is presented as Appendix A. The results of the analysis suggested that sea level rise would not pose a direct hazard (i.e. a breach of dunes) to Middlepoint Swamp by 2030 (the time period of projected climate change scenarios produced for this risk assessment). For this reason, no further assessment of the hazards posed by sea level rise have been included.

Secondary hazard: A reduction in groundwater discharge from the TLA secondarily results in reduced depth and duration of inundation in groundwater dependent wetlands, and increased surface water salinity as a result of reduced groundwater inflow and evapo-concentration. Water regime (depth, duration, frequency, and timing of inundation) is a major determinant of plant community development and patterns of plant zonation in wetlands (Casanova & Brock 2000). Additionally, biodiversity and habitat complexity has been shown to reduce as salinity increases (Brock et al. 2005; Smith et al. 2009). As a consequence, and dependent on the magnitude of change to water regime, changes in the composition and patterns of vegetation communities can be expected, including loss of biodiversity and permanent water habitat/refugia (Nielsen & Brock 2009) and encroachment of terrestrial and exotic plant species (Casanova & Brock 2000). If groundwater levels decline below a point, GDEs can become disconnected from the regional unconfined aquifer (Cook et al. 2008) and with declining rainfall are at a high risk of terrestrialisation (i.e. no longer hold water long enough to support hydrophyte communities). The process of terrestrialisation within a fragmented and agricultural landscape favours the establishment of terrestrial pasture grasses and weeds (exotic species), and exposes wetland ecosystems to increased stock grazing pressure.

For the Middlepoint Swamp case study, the hazard of increased salinity was not assessed or modelled due to the lack of reliable data for determining species or vegetation community response curves. Middlepoint Swamp is however a brackish wetland, with inputs of fresh water from springs. It is likely that increased salinity will result as a consequence of reduced inundation depth and decreased spring discharge, potentially resulting in declining species richness (Smith et al. 2009; Brock et al. 2005).

4.1.2 Determine the cause of the primary hazard

To determine the main causes of groundwater decline, groundwater hydrographs from long-term monitoring wells were compared with cumulative Annual Residual Rainfall (ARR) (Figure 4.1) measured at the Bureau of Meteorology (BoM) weather station at Mount Schank (Station No. 026027). Four monitoring wells (MAC039, MAC047, MAC016 and MAC019) were identified as having monitoring records beginning in the 1970s (refer to Figure 3.1 for location map).

A pronounced step-change in groundwater levels in the TLA was exhibited from 1992 to approximately 2000, resulting in an approximately 2 m difference in groundwater levels from the 1970–92 baseline to current (2000–14). The sharp decline was reflected in a decline in rainfall over the same period. Additionally the development of groundwater resources for centre pivot irrigation coincided with the decline in rainfall from approximately 1992.

Whilst it appears that the primary cause of groundwater level decline can be attributed to reduced rainfall, markedly increased magnitudes in groundwater level fluctuation post 2000 may be indicative of the impact of seasonal pumping (extraction for irrigation). Increased extraction is likely as a result of predicted reduced rainfall (McFarlane et al. 2012), as such it could be expected that extraction may have a greater impact on groundwater levels into the future.

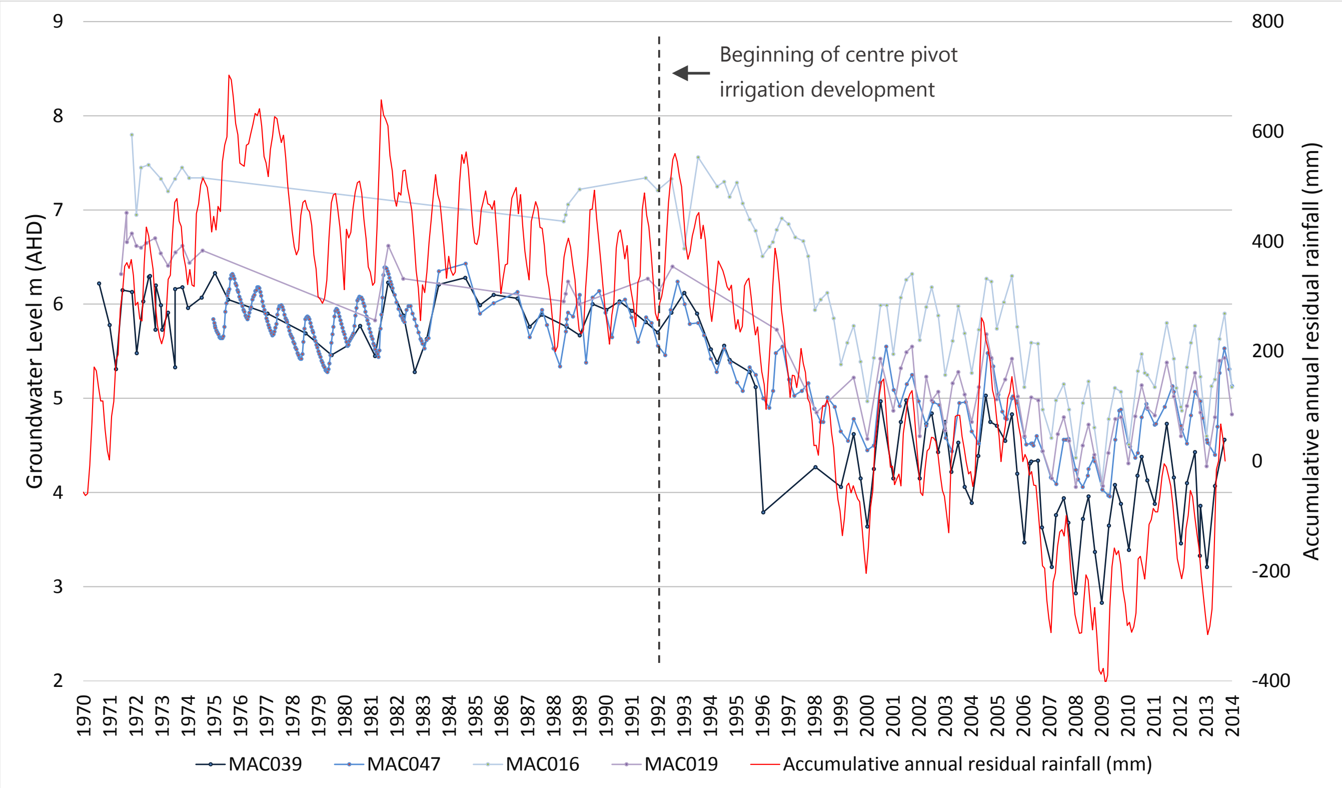


Figure 4.1 **Groundwater levels in long-term monitoring wells up-gradient of the Middlepoint Swamp GDE case study site, compared with accumulative annual residual rainfall**

4.2 Step 2: Exposure and vulnerability

The exposure and vulnerability of Middlepoint Swamp GDE to a change in groundwater level as a result of reduced rainfall was based on determining statistical relationships between groundwater levels, rainfall, and surface water levels and hydrograph characteristics (i.e. depth and duration) within the wetland from existing data. These relationships were then used to model how surface inundation will respond to changes in rainfall and groundwater conditions under climate change projections.

The approach taken was to examine trends and establish relationships between groundwater and rainfall in the closest observation well (MAC039) with a long-term groundwater monitoring record. Projected changes in groundwater level in response to predicted changes in rainfall from climate change scenarios were then modelled. The relationship between this well, and the local groundwater level being monitored at the wetland (MAC090) was then established. A relationship between local groundwater level and the surface water level within the wetland was then determined from site based groundwater and surface water monitoring. The impact of changes in rainfall on the surface water dynamics of the wetland (depth and duration) were then able to be modelled with use of a LiDAR DEM.

4.2.1 Determine the projected change in groundwater levels under climate change scenarios

A high quality historic record of groundwater level variations in the TLA of the Lower South East, coupled with a long-term record of daily rainfall from the Bureau of Meteorology (BoM) weather monitoring stations enabled a comparison of variations over time in groundwater level with variations in local rainfall. A close correlation may be observed between the cumulative deviation from mean (CDFM) rainfall and water levels in unconfined water table aquifers, particularly in aquifers that respond rapidly to rainfall recharge and have high transmissivity, such as in the limestone karst aquifer hydrogeological setting of the Lower South East (Brown et al. 2001).

The correlation between water levels in a number of observation wells with long-term monitoring records in the vicinity of Middlepoint Swamp (MAC039, MAC019, MAC016, MAC047) and the CDFM rainfall at the nearby Mount Schank BoM monitoring station (Figure 3.1) was quantified in order to develop a simple predictive model of the groundwater level variation under varying rainfall patterns. Observation well MAC039 was selected as the most suitable to develop the predictive model as it had a relatively long and continuous record of observations, from 1971 to the present, and was less affected by direct impacts of groundwater extraction than other observation wells in the vicinity. The relationship was established between historic rainfall and groundwater levels using an analysis tool called Hydrograph and Rainfall Time Trend (HARTT) (Ferdowsian et al. 2001). The HARTT model can be applied in monthly or annual time steps to the accumulative monthly residual rainfall (AMRR), or the accumulative annual residual rainfall (AARR). In the application described here the model was applied in monthly time steps to the AARR, which is defined as (1):

$$AARR_t = \sum_{i=1}^{t} (M_i - = A/12) \qquad (1)$$

where M_i is rainfall in month i (a sequential index of time since the start of the dataset), A is the average annual rainfall, and t is months since the start of the dataset.

HARTT determines a regression model (2) to fit groundwater level as a function of AARR over time:

$$\text{Depth}_t = k_0 + (k_1 \ x \ AARR) + (k_2 \text{ x } t) \qquad (2)$$

where Depth$_t$ is depth (m) of groundwater below the ground surface at time t (months from start of regression), k_0 is approximately equal to the initial depth to groundwater (m) and k_1 (m/millimetre of rainfall residual) represents the impact of above or below average rainfall on groundwater level. The second part of equation (k_2 x t) represents a linear time trend component that accounts for any long-term groundwater level trend (m/month). Such a trend may be the result of changing groundwater extraction regimes or land-use change. The performance of the HARTT model's approximation of historic groundwater levels in response to rainfall variations is optimised to an objective function of a maximum R^2 for the regression between the modelled and observed groundwater levels over a historical baseline period of rainfall and water level observations.

A HARTT model rainfall/ groundwater level relationship was developed for the MAC039 observation well (the closest long-term monitoring well to Middlepoint Swamp). The correlation achieved between HARTT model groundwater levels and observed water levels at observation well MAC039 was optimised for the period of historical observations from March 1996 to December 2013 (Figure 4.2), during which observations of MAC039 water levels were at 4-month intervals. The optimised fit of the model to the observed water levels during this period had a correlation with an R^2 value of 0.6, using a constant (k_1) of -2.83×10^{-3} applied to AARR, and a time dependent constant (k_2) of -4.38×10^{-4}. Water level observations for the period prior to 1996 were not used in the analysis as the observations indicated that the aquifer may not have been in equilibrium with changing land-use (increased extraction for irrigation) in the early 1990s.

After optimising the correlation between modelled and historical observed water levels, the HARTT model for MAC039 was applied with projected rainfall from four future climate scenarios (two Global Climate Models (GCM) and two greenhouse gas representative concentration pathways (RCP)). One hundred statistically downscaled realisations of possible future rainfall from 2010 to 2100 at the Mount Schank BoM monitoring station from each GCM/RCP combination was drawn from the Goyder Institute Agreed Climate Change Projections for South Australia database (DEWNR 2014).

4.2.1.1 Model Assumptions

The application of the HARTT model here is dependent on some critical assumptions and limitations:

1. The HARTT model of water level response for MAC039 is calibrated over a period where water level depths varied between approximately 6.5 m and 11 m. It is not known whether groundwater levels will continue to respond to CDFM rainfall in the same way if they fall below the historic low water level depth of approximately 11.5 m. In reality, water levels falling lower that 11.5 m depth may start to exhibit a different response to CDFM rainfall compared to that observed during the historic calibration period, and is a limitation of the modelling approach.

2. The two-parameter HARTT model allows identification of a linear relationship with the cumulative deviation from mean (CDFM) rainfall over time and a second, undefined, time-varying influence on the water level. Thus, the HARTT model may identify a correlation between observed groundwater levels and CDFM rainfall, but only provide a good fit of its linear function of CDFM rainfall to observed water levels if it also adds a second parameter which is a constant. The latter parameter adds a linear trend to the HARTT water level simulation, which may be assumed to be, for example, the effect of a change in groundwater extraction rates over time, or the effect of a land use change impact, such as an increase in tree plantation transpiration of groundwater over time. In the HARTT simulation of water level of the MAC039 observation well, this second parameter as a constant was applied. We have not sought to identify what this time varying influence on water levels might be. However, to project the simulation of MAC039 water levels into the future using the HARTT model, we must assume that this time-varying influence will continue to behave the same way into the future as it has during the historic baseline period on which the HARTT model was based.

3. It is clear that there is a significant amount of groundwater extraction for irrigation occurring in the environs of observation well MAC039. The inter-annual variation in observed groundwater level is assumed to be due to a combination of variations in groundwater recharge from year to year in addition to variations in groundwater extraction. In projecting the MAC039 water level using the HARTT model under future climate scenarios, it is assumed that the amount of groundwater extracted in the vicinity of this well each year continues to vary in response to climate in the same way in the future as it has during the historic baseline period. This implies a continuation of the same land uses into the future, and that water users will continue to use the same groundwater resource and will continue to extract more groundwater in dry years than they do in wet years. It is acknowledged by the authors that this may well not occur in future, however the intention of the projected scenarios is to indicate what may happen to water levels under future climates if current land uses and irrigation practices continue into the future.

4.2.1.2 Future climate scenarios

The Intergovernmental Panel on Climate Change's Fifth Assessment Report (IPCC 2013) included new climate projections from updated and revised GCMs. These new climate projections have been utilised through the Goyder Institute for Water Research (GIWR) project, "An Agreed Set of Climate Change Projections for South Australia", which provides a comprehensive suite of downscaled climate data for locations throughout South Australian (DEWNR 2014).

The Goyder Institute project uses the non-homogeneous hidden Markov model (NHMM) to produce projections of rainfall and other hydrometeorological variables, such as temperature, solar radiation, pressure and humidity, which have been post-processed to give projections of potential evapotranspiration (PET). The NHMM has been developed over more than a decade (Bates et al. 1998; Charles et al. 1999a; Charles et al. 1999b; Hughes et al. 1999), and has been found to perform reasonably well in benchmark studies based on a range of average and extreme rainfall statistics (Frost et al. 2011). The NHMM simulations of future climate are available for numerous BoM weather monitoring stations throughout each of the eight NRM regions in SA. The BoM Mount Schank weather station (BoM Station No. 026067) was selected for this study, as it is the closest of all stations for which the NHMM simulations are available to the Middlepoint Swamp case study site and, in particular, to the selected observation well, MAC039 (Figure 3.1).

The GIWR climate change projections project identified 15 GCMs as being preferable for use in producing climate change simulations for South Australia, based on the performances of these GCMs in representing key climate drivers that affect rainfall in South Australia. Of these 15 GCMs, two were selected for this study that represent the low end and high end (among these 15) of rainfall and temperature change projections for the South East.

The ESM 2M GCM projects a drier and warmer climate than most of the 15 GCMs for the region under low and high greenhouse gas emissions scenarios, while the ACCESS 1.0 GCM projects a generally less warm and dry future.

The RCP 4.5 greenhouse gas concentration pathway represents a future scenario of coordinated mitigation of greenhouse gas emissions, while the RCP 8.5 greenhouse gas concentration pathway represents a 'business as usual' growth in greenhouse gas emissions without effective coordinated global emission mitigation efforts.

In the scenarios and simulations presented here, a combination of the ESM 2M GCM and the RCP 8.5 concentration pathway represents the 'worst case' 21st century climate projection for the region, while a combination of the ACCESS 1.0 GCM and the RCP4.5 concentration pathway represents the 'best case' climate projection.

4.2.1.3 Projecting future water levels in the MAC039 Observation Well

For each combination of the two selected GCMs and greenhouse gas concentration pathways (hereafter 'GCM/RCP combination') there are 100 NHMM downscaled simulations of future rainfall (termed realisations in this report) from 2013–50 at the Mount Schank weather station. Each of the realisations within these groups of 100 is based on the same GCM projection of rainfall change; hence they each have similar decadal-scale trends. However, within each GCM/RCP combination, there is a broad spread of future rainfall projections.

The four groups of future rainfall realisations were applied to the HARTT model relationship of groundwater levels to CDFM rainfall at the MAC039 observation well. From the 100 projections of possible future water levels resulting from each of the four scenarios, a median, 10th and 90th percentile projected groundwater level was identified, and selected to represent of the range of possible future groundwater levels for the TLA at MAC039 over the 37 year (2013–50) simulation period under each of the four future climate scenarios (Figure 4.2).

Where the HARTT model projected water levels fall to below the historic low of approximately 11.5 m, the projection can be considered for further analysis and application in two ways. Firstly, the projections of water level below 11.5 m can be applied to analyses of the possible impacts of these lower water levels under an untested assumption that when groundwater levels at the location of MAC039 fall below their historic minimum, they continue to respond to CDFM rainfall in the same way as during the calibration period. Alternatively, the impact of water levels falling to, and remaining at or below, the historic low of 11.5 m can be examined, with an assumption that the response of water levels below the historic minimum cannot yet be reliably projected by this method.

The HARTT model water level projections for MAC039 were applied in further projections of water levels for Middlepoint Swamp (Section 4.2.2) under the first of these assumptions, but only as far as the year 2030 in recognition of the limitations. At that point in time the median water level projections of two of the four GCM/RCP combination scenarios are at or very close to the historic minimum water level (Table 4.1). In the other two GCM/RCP combination scenarios, the water levels have fallen to approximately 1.5 m below the historic minimum level (Table 4.1). In the case of these two GCM/RCP combination scenarios, the corresponding projections of water levels at Middlepoint Swamp (Section 4.2.2) are subject to the assumption that groundwater levels in this range continue to respond to CDFM rainfall as they did during the historic calibration period.

Note that in all of the scenarios of projected changes in water level for Spring 2030 identified in Table 4.2, the unidentified time-varying influence, makes up approximately 0.09 m (-4.38×10^{-4} m/month x 17 years x 12 months) of the projected change. The remainder of the projected change in water level in each climate scenario is the HARTT model's projected response to the AARR within each projected climate's rainfall time series. Note, the unidentified time-varying influence is negligible compared to the water level response to the AARR.

Beyond 2030, groundwater levels in all the GCM/RCP combination scenarios are projected to decline, with a more marked decline after 2035 in most cases. However, these longer-term projections should be considered with considerable caution as, again, these are subject to the untested assumption that groundwater levels will continue to respond to CDFM rainfall in the same way even as groundwater levels fall markedly below their historic lowest levels.

Table 4.1 Projected change in groundwater level in MAC039 – Spring 2013 to Spring 2030

Climate Change Scenario (GCM/RCP)	Projected change in groundwater level (m) Spring 2013 to Spring 2030		
	10th percentile	Median	90th Percentile
ACCESS 1.0 RCP 4.5	0.39	-0.87	-1.88*
ACCESS 1.0 RCP 8.5	-0.4	-1.48	-3.05*
ESM 2M RCP 4.5	-1.4	-2.63*	-3.75*
ESM 2M RCP 8.5	-1.43	-2.45*	-3.67*
* projections below historic records			

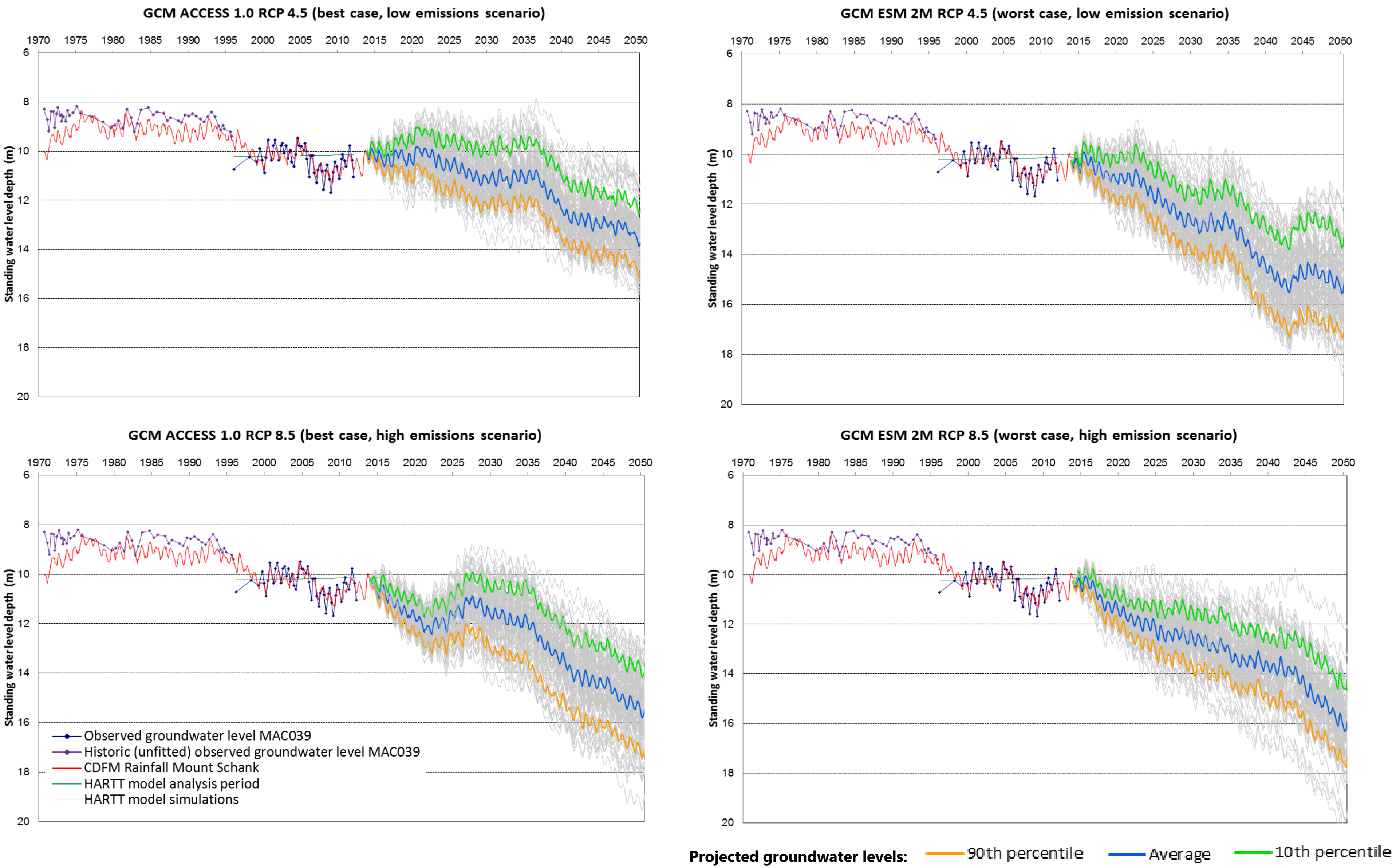


Figure 4.2 Projected future groundwater level scenarios for MAC039

4.2.2 Determine the relationship between groundwater and surface water dynamics within the wetland

Projections of groundwater levels can be used to assess the likely change in hydroperiod in a wetland based on establishing a relationship between surface and groundwater levels determined for a GDE (Chambers et al. 2013a). Relationships between surface water levels at Middlepoint Swamp, local groundwater levels at the wetland (MAC090 – refer to Figure 3.4), and the observation well (MAC039) used to project future groundwater levels were developed as a means to predict future surface water level and durations for the chosen climate change scenarios.

A two-stage modelling process was used to relate observation well MAC039 to MAC090 (both monitoring the top of the TLA), and then surface water levels due to the discrepancy in monitoring time-steps (4-monthly for MAC039 and 12-hourly from data logger in MAC090). The 4-monthly monitoring of MAC039 failed to capture the maximum groundwater levels indicated from the logger data at the wetland, and therefore failed to model maximum water levels within the wetland. A more reliable and robust relationship between groundwater and surface water was therefore possible using the two-step process, rather than relating MAC039 to surface water levels directly.

In addition to the model assumptions inherent in the applied GCMs to determine projected groundwater levels (refer Section 4.2.1), key assumptions of projecting groundwater changes on surface water dynamics included that:

- relationships between surface water and groundwater hydrology reflective of a relatively short period of monitoring data are representative of future relationships. Groundwater level drawdown may result in the wetland becoming disconnected from the TLA. The direct relationship between rainfall and surface water level in the wetland has not been assessed in relation to catchment runoff, however may assist in maintaining some seasonal level of saturation. Local surface water runoff to the wetland was assumed to be a far less significant component of the overall water balance than groundwater discharge. This was due to the relatively flat topography, small local catchment area where potential contributing water sources from the east have been diverted seaward by drainage infrastructure, absence of any contributing surface water drains and creeks, and porosity of soil types.
- available elevation data for the wetland was of an accuracy suitable for the modelling application. A 2 x 2 m pixel LiDAR (light detection and ranging) derived digital elevation model (DEM) was used to determine level, depth and duration of inundation across the wetland via spatial interpolation. Vertical accuracy of the DEM was typically ±0.15m (Wood & Way 2011). No further assessment of the accuracy of the DEM in this application was made.

4.2.2.1 Relationship between wetland surface water level and local groundwater level

Surface water level within Middlepoint Swamp had been recorded using an Odyssey™ water pressure logger (Dataflow Systems Pty Ltd) installed on a gauge board located at approximately the lowest elevation of the wetland basin (refer to Figure 4.1). Surface water records were logged six hourly, from 7/10/2011 and manual gauge board measurements taken infrequently as part of the GDE monitoring program (DEWNR; SKM 2010). Data were missing for approximately 7.5 months (between 24/08/2012 – 17/04/2013) due to the gauge board falling over.

Groundwater level local to the wetland was determined from the GDE Monitoring Program observation well MAC090 (refer to Figure 3.4) using an AquaTROLL® 200 (In-Situ Inc.) water level, temperature and salinity logger. Groundwater depth records were logged 12-hourly since 7/10/2011, and measured manually every four months.

All groundwater and surface water logger data was corrected to fit manual measurements, and converted to elevation (m AHD) and daily averages (Figure 4.3). The relationship between surface water (SWL) within the wetland and local groundwater level (GWL) was derived using daily time-step data from a linear regression (3), with an R^2 value of 0.7978, indicating a strong correlation between groundwater and surface water levels as expected from past conceptualisation of the groundwater dependency of the site:

$$SWL = (0.3268 \times MAC090) + 1.6544 \qquad (3)$$

The linear regression equation was used to interpolate missing surface water level values between 24/08/2012 and 17/04/2013 (Figure 4.3), and also to project the surface water level response to a projected change in groundwater level.

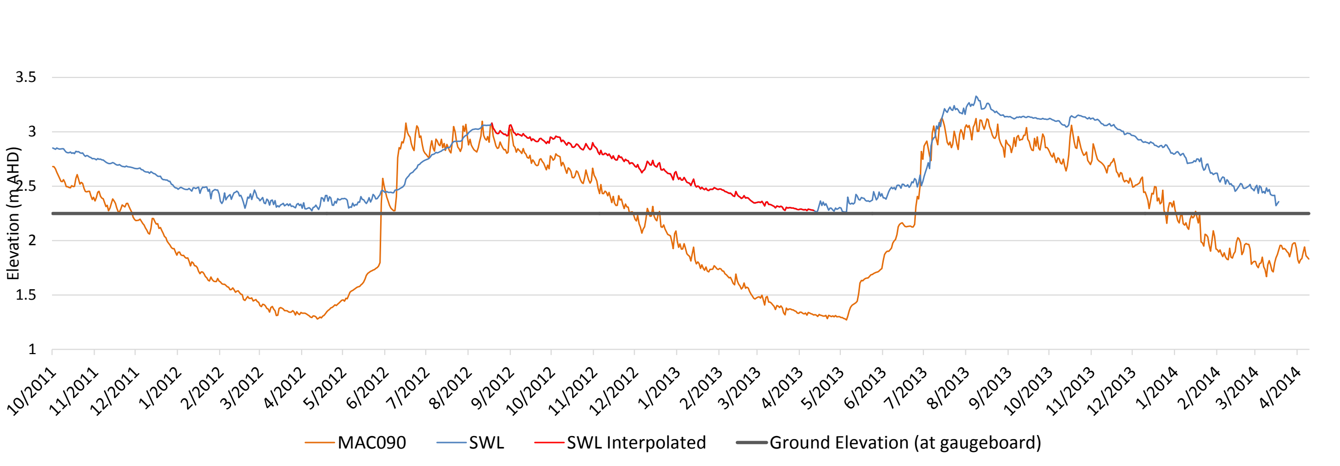


Figure 4.3 Surface water (SWL) and groundwater (MAC090) hydrograph for Middlepoint Swamp collected as part of the GDE monitoring project (DEWNR)

4.2.2.2 Relationship between local wetland groundwater level and MAC039

Due to the short time period monitored by the GDE observation well (MAC090), a nearby observation well with a long-term monitoring history (MAC039) was used to project changes in groundwater levels in response to climate change scenarios (see Section 4.2.1). Consequently, a relationship was required to be determined between observation well (MAC039) and the GDE observation well (MAC090). The relationship between local wetland groundwater level (MAC090) and groundwater level at MAC039 was derived from a linear regression (4), with an R^2 value of 0.8702 at a quarterly time-step (4-monthly):

$$\text{MAC090} = (0.9048 \times \text{MAC039}) - 1.3359 \qquad (4)$$

The equation was used to approximate groundwater levels at MAC090 back to 1978 (an historic level scenario) (Figure 4.4), and also to project groundwater levels at MAC090 (and ultimately surface water levels) for future groundwater level scenarios.

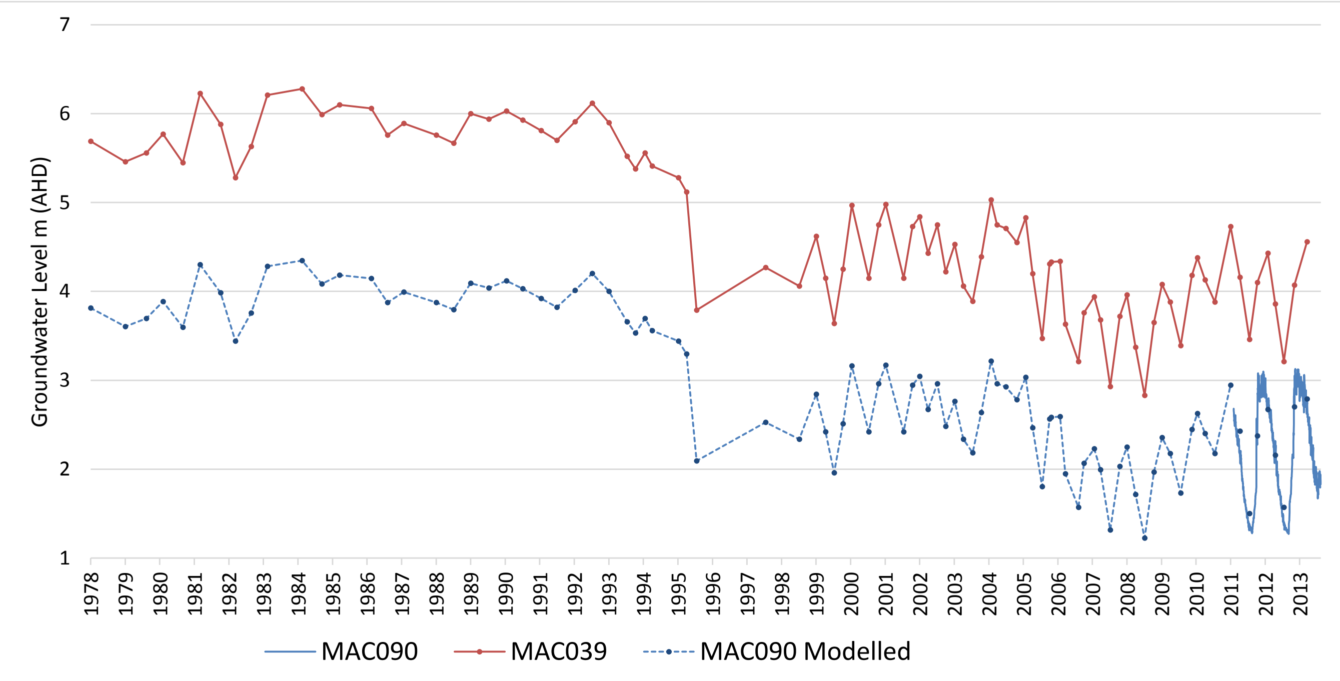


Figure 4.4 Modelled local wetland groundwater level (MAC090) based on the relationship between MAC039 and MAC090

4.2.2.3 Relationship between surface water level and depth and duration of inundation

The relationship between surface water level, depth and duration of inundation was modelled based on surface water monitoring data collected for the 2011–13 monitoring period. The 2011–13 surface water data was used in conjunction with the elevation data from the 2 m LiDAR DEM to calculate (by subtraction) the maximum and average water level depths at each sample point along the three transects and for each pixel within the cross-section assessment area (refer to Figure 4.1). The number of days where water depth exceeded zero was also calculated for each sample point for both time periods, providing a yearly duration of inundation (days per year).

There was a strong linear relationship demonstrated between maximum surface water depth (MaxWD) of inundation and duration of inundation per year (Dur). A linear regression was used to derive the relationship (5), with an R^2 value of 0.9916.

$$Dur = (339.35 \times MaxWD) - 4.2581 \qquad (5)$$

4.2.2.4 Summary of relationships between groundwater level and surface water level dynamics

A summary of all the equations used to relate groundwater and surface water levels, and depth and duration of inundation that were applied to projected future groundwater scenarios is provided as Table 4.2. The number of modelled relationships, and the scarcity and reliability of available data has the potential to introduce significant errors in the final model output. However, in the absence of ideal datasets (i.e. longer term at least monthly logged data for both groundwater and surface water levels at the wetland site), the relationships developed between the datasets was considered to be within acceptable limits of error for the purposes of the risk assessment, and the uncertainty of the hydrological modelling was reflected in the final assessment of risk to wetland vegetation communities.

Table 4.2 Summary of the equations used to relate groundwater and surface water dynamics for Middlepoint Swamp

Modelled relationship	Equation	R^2
Regional groundwater level (MAC039) to local wetland groundwater level (MAC090)	Linear Regression MAC090 = (0.9048 x MAC039) – 1.3359	0.8702
Local wetland groundwater level (MAC090) to wetland surface water level (SWL)	Linear Regression SWL = (0.3268 x MAC090) + 1.6544	0.7978
Surface water level (m AHD) to maximum water depth (MaxWD)	MaxWD = maximum SWL for scenario year – elevation (from 2 m LiDAR DEM)	N/A
Maximum water depth to annual surface water inundation duration in days per year (Dur)	Linear Regression Dur = (339.35 x MaxWD) – 4.2581	0.9916

4.2.3 Projecting the resulting change in surface water levels due to projected changes in groundwater levels

The changes in groundwater levels predicted by the HARTT modelling of climate change scenarios (Section 4.2.1), with the use of the relationships established in Table 4.2 were used to project the resulting change in surface water levels in Middlepoint Swamp to 2030. The median projected groundwater level (Figure 4.2) for each of the four GCMs tested were used in the final surface water projections (Figure 4.5).

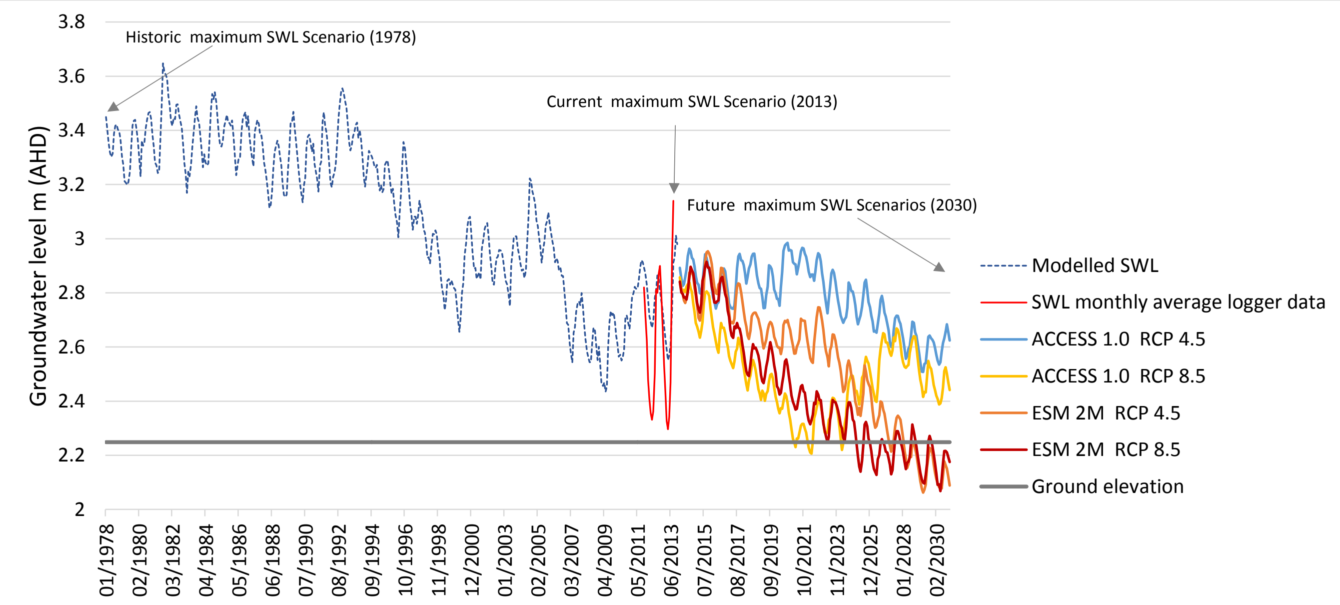


Figure 4.5 Projected surface water levels for Middlepoint Swamp for selected climate change scenarios to 2030

The results of modelling and projecting the outcome of groundwater level decline on surface water levels in Middlepoint Swamp showed that the applied relationships reasonably (within 0.15 m) predicted maximum surface water levels for 2011–13 monitoring period (Figure 4.5). The model however failed to predict the lower half of the hydrograph (wetland drying) largely due to limitations of establishing a reliable relationship between groundwater and surface water levels once the wetland is dry (i.e. surface water levels are not recorded in negative values below ground level). A more robust relationship between groundwater and surface water levels may be able to be achieved with further years of monitoring data, allowing the construction of a coupled surface water / groundwater model.

Considering the limitations of the projected surface water levels, only maximum predicted surface water levels for the three selected time epochs of the risk assessment were analysed further. The surface water level (elevation m AHD) for each time epoch and scenario selected was then transformed into wetland water level depth for each transect point and cross-section pixel using the 2 m LiDAR DEM. The established relationship between surface water depth and inundation duration (Table 4.2) was then used to convert the modelled water depth into duration values for the 1978 and 2030 scenarios. Actual monitoring data (both maximum SWL and duration of inundation) were used for the 2013 epoch.

The GCMs identified as worst case scenarios (ESM 2M), produced very similar water levels by 2030, and the GCM ESM 2M RCP8.5 (worst case, high emissions) scenario predicted very slightly higher water levels than the low emissions scenario of the same GCM (Figure 4.5). Due to the similarity of the two GCM ESM 2M outputs, RCP 8.5 (worst case scenario, high emissions) was selected for further analysis, along with the two ACCESS 1.0 models.

Figure 4.6 provides the spatial representation of surface-water level scenarios for 1978 (modelled), 2013 (actual) and the three selected 2030 (projected) climate change scenarios, developed with the use of the regional LiDAR 2 m DEM. Under all climate change scenarios, maximum water depths in the wetland are reduced to less than 0.6 m, and in the worst case scenario, water levels are shown not to exceed 0.1 m (essentially representing an almost total drying of the wetland). Changes in surface water area (area inundated) between the historic (1978) scenario and current (2013) water levels of approximately -0.28 m resulted in a relatively minimal net change in inundated area within the assessment area of -7.04% (Table 4.3). Climate change scenarios to 2030 indicate more significant changes in net wetland area where an additional -0.46 m SWL decline from 2013 resulted a projected 26.63% reduction in wetland area, a -0.62 m SWL decline resulted in a projected 44.17% reduction in wetland area, and a -0.98 m SWL decline resulted in a projected 84.97% reduction in wetland area (Table 4.3).

Changes in projected groundwater and surface water levels indicated that surface water levels would reduce at approximately half the level of groundwater decline (e.g. where groundwater level is reduced by 0.85 m, the corresponding decline in surface water level was 0.46 m) (Table 4.3).

Table 4.3 Projected changes in groundwater and surface water levels, and surface water extent under climate change scenarios to 2030, Middlepoint Swamp

Time period	Change in groundwater level (m)	Change in surface water level in wetland (m)	Change in surface water extent (% cross-section area)
Historic to current	-0.81	-0.28	-7.04%
Current to 2030 (1)	-0.85	-0.46	-26.63%
Current to 2030 (2)	-1.34	-0.62	-44.17%
Current to 2030 (3)	-2.28	-0.98	-84.97%

(1) GCM ACCESS1.0 RCP4.5 (Best case, low emissions)
(2) GCM ACCESS1.0 RCP8.5 (Best case, high emissions)
(3) GCM ESM 2M RCP8.5 (Worst case, high emissions)

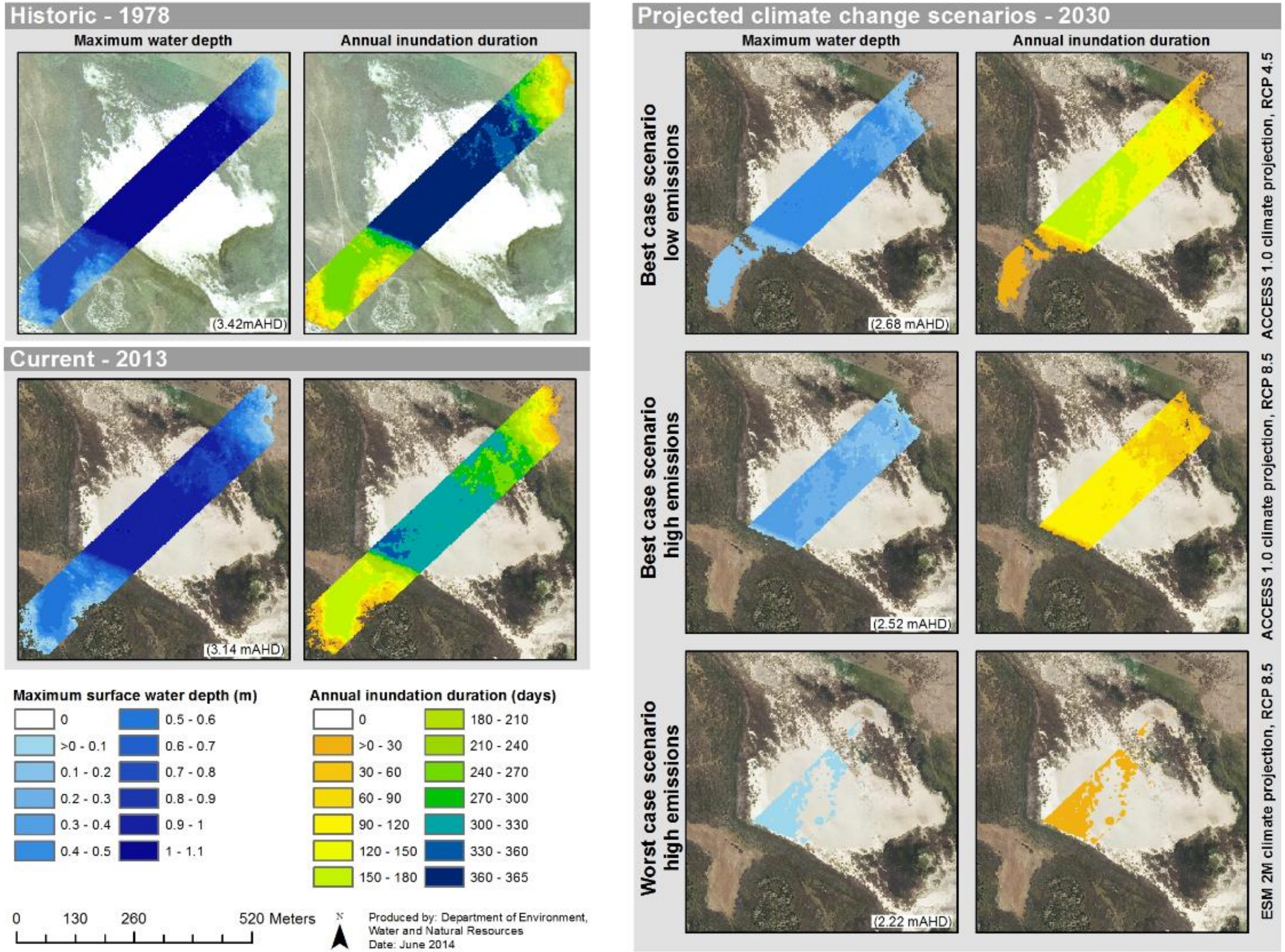


Figure 4.6 Spatial representation of predicted surface water level extent, depth and duration for 2030 climate change scenarios at Middlepoint Swamp. Surface water levels in m AHD indicated for each scenario

4.3 Step 3: Assess effects

In order to most effectively depict spatially (and therefore communicate) the risks of climate change on GDEs, the assessment of ecological effects of the projected hydrological changes at Middlepoint Swamp was determined based on the likely response of vegetation communities. This level of assessment was chosen, rather than focusing on the overall effects of habitat suitability for an individual key species (refer to Section 3.2), both due to data availability, and also the ability to make qualitative deductions about the likely effects on individual aquatic dependent species present based on the changes in habitat. Therefore, effects on individual species were not specifically assessed or addressed within the scope of this report.

To determine the effect of declining groundwater levels on vegetation characteristics of the Middlepoint Swamp case study site, an understanding of the relationship between the wetland hydrology (established in step 2 of the risk assessment process (Section 4.2)), and the water requirements of vegetation communities present was required (Hebb 2003; Casanova & Brock 2000). For Middlepoint Swamp, this was accomplished by collating and extrapolating available data generated through the GDE monitoring program and data available in the South Australian Wetland Inventory Database (SAWID), in conjunction with use of remotely sourced spatial datasets (aerial photography and existing DEM). These data were then used in the development of a conceptual model of eco-hydrological function based on a number of identified significant parameters, and for modelling of hydrological niches of vegetation via the deterministic groundwater/ surface water and elevation relationship known to be present at the site.

4.3.1 Collate existing data and field survey

Existing hydrological, ecological and spatial data for Middlepoint Swamp was collated and reviewed (see Table 4.4), as well as a limited review of literature pertaining to hydrological thresholds of tolerance for dominant vegetation types present at the site.

4.3.1.1 Data review

Existing hydrological monitoring data was collected as part of the GDE Monitoring Program (DEWNR; SKM 2010). Middlepoint Swamp was equipped with groundwater and surface water monitoring infrastructure, instrumented with loggers installed between 2009 and 2011. These data were used to create spatial depth and duration of inundation data for the three modelling epochs identified for this risk assessment (1978, 2013 and 2030, refer to Section 4.2).

Ecological data were sourced from various monitoring programs via the SAWID database (Table 4.4), with the majority of reliable quadrat data from the GDE Monitoring Program (Beacon Ecological 2010), and data collected as part of the Goyder Institute project - *Developing ecological response models and determining water requirements for wetlands in the South-East of South Australia* (Goyder Institute for Water Research 2014). Whilst the number of quadrats already sampled within the wetland was reasonable, there was inadequate coverage within the defined spatial boundaries of the risk assessment (refer to Figure 4.2), and spatial distribution was not consistent over all remotely (aerial photography) visually discernible vegetation communities. As such, a limited field survey and ground-truthing program was implemented specifically for this assessment (described in Section 4.3.1.3).

Available spatial datasets including LiDAR (light detection and ranging) derived digital elevation model (DEM) and temporal aerial photographs were sourced and used to extrapolate existing datasets.

Table 4.4 Summary of available datasets for developing an eco-hydrological conceptual model

Data source	Description	Data	Quantity	Source / Custodian
Vegetation monitoring				
GDE Monitoring Program – vegetation monitoring (Beacon Ecological 2010)	1 x 1 m vegetation monitoring quadrats sampled in Autumn 2010.	GPS coordinates, Species presence, cover abundance, water depth, photopoints	48 quadrats	SAWID (DEWNR)
Monitoring Ecological Response to High Value Wetlands to Changes in Groundwater (Ecological Associates 2010)	Vegetation zones recorded at waypoints, Autumn 2010.	Species presence, vegetation association, photopoints	10 waypoints	SAWID (DEWNR)
Goyder Institute - Developing ecological response models and determining water requirements for wetlands in the South-East of South Australia (Goyder Institute for Water Research 2014)	1 x 1 m vegetation monitoring quadrats sampled in Spring 2013.	GPS coordinates, Species presence, cover abundance, water depth	67 quadrats	SAWID (DEWNR)
Field survey (this project)	1 x 1 m vegetation monitoring quadrats sampled along three transects (Table 4.1 and Figure 4.1) in Autumn 2014. Data entered into SAWID. GPS locations of vegetation community change along transects.	GPS coordinates, Species presence, cover abundance, water depth, quadrat photo	30 quadrats	SAWID (DEWNR)
Hydrology				
Outputs from Step 2 of this risk assessment (Section 4.2)	Modelled depth and duration data for 1978, 2013 and 2030 time epochs.	Depth and duration raster datasets for the transect area	All assessment time epochs	DEWNR
GDE Monitoring Program – surface water gaugeboard	Surface water gaugeboard equipped with Odyssey™ pressure water level logger	Pressure, Water Level	Logged data, 6 hourly since 7/10/2011	Hydstra (DEWNR)
Spatial datasets				
LiDAR DEM	2 m grid DEM derived from LiDAR for the whole SE NRM region. Vertical accuracy ±0.15 m.	Elevation	Full coverage of case study site	DEWNR
Rectified aerial photographs	February 2013 February 2008	Colour air photo	Full coverage of case study site	DEWNR
Historic aerial photographs	17/02/1978 Survey: 2205 Frame: 109 06/02/1987 Survey: 3592 Frame: 211 30/01/1992 Survey: 4472 Frame: 144	Colour air photo	Full coverage of case study site	DEWNR

4.3.1.2 Review of hydrological threshold information

A review of information on the hydrological (depth and duration) preferences of dominant plant species and vegetation communities identified at Middlepoint Swamp revealed limited specific information was available. Ecological Associates (2009) investigated the use of Wetland Vegetation Components (WVC) to determine and characterise the required hydrology and salinity thresholds for particular vegetation types, although many of the thresholds were determined through expert opinion and the WVC reflective of habitat components useful at a large scale (e.g. regional scale). Plant Functional Groups (PFG) have also been developed to compare water plant responses to different depth, durations and frequencies of flooding (Casanova & Brock 2000; Casanova 2011), which classifies plants into three broad categories: terrestrial (intolerant of flooding); amphibious (tolerates or responds to flooding and drying) and submergent (intolerant of desiccation), with further splits between categories. The depth and duration hydrological preferences for identified WVCs and PFGs for plant species present in Middlepoint Swamp is presented in Tables 4.5 and 4.6 respectively.

Table 4.5 Wetland Vegetation Components (WVC) present at Middlepoint Swamp (source: Ecological Associates 2009)

Wetland Vegetation Component	Maximum depth (m)	Annual duration (months)
Gahnia filum tussock sedgeland	0.5	4
Seasonal saline low aquatic bed	0.8	6
Phragmites australis grassland*	2	6
Leptospermum lanigerum shrubland*	0.3	8
* WVC outside of risk assessment spatial boundaries		

Table 4.6 Plant Functional Groups (PFG) identified from literature for species present at Middlepoint Swamp

Plant Functional Group	Example species	Source
Amphibious fluctuation tolerators: Fluctuating water levels, plants do not respond morphologically to flooding and drying and will tolerate short-term complete submergence (<2 weeks)	*Baumea juncea* *Juncus kraussii*	(Casanova 2011)
Submergent r-selected: Temporary wetlands that hold water for longer than 4 months	*Ruppia polycarpa* *Chara sp.*	(Casanova 2011; Casanova & Brock 2000)

It was apparent that the scale of existing hydrological threshold and tolerance data relevant to the Middlepoint Swamp assessment area (consisting of 2 WVCs and four species with known functional groups assigned) was insufficient for modelling applications as part of this risk assessment. Therefore hydrological niche modelling of vegetation communities at a scale suitable for the case study site model boundaries was developed specifically from the available hydrological and ecological data (see Section 4.4.2.1).

4.3.1.3 Field survey

Additional vegetation surveys were undertaken in autumn 2014 to supplement the existing monitoring data. A series of 1 x 1 m quadrats were sampled and were positioned at different points along the elevation gradient and within the mapped vegetation zones distinguishable from the 2013 aerial photography (see Section 4.3.2) and within the extent of the modelling area. A GPS position was recorded between changing vegetation zones along the three transects, and for each quadrat. Data collected for the vegetation surveys were used to assist in ground-truthing remotely sensed data used for predictive modelling.

4.3.2 Development of the conceptual model

4.3.2.1 Data interpretation and analysis

Three epochs were chosen as the temporal boundaries of this risk assessment: 1978 (historic); 2013 (current) and 2030 (future climate change). Vegetation quadrat data available for Middlepoint Swamp were collected (sporadically, and non-replicated) between 2010 and 2014 (Table 4.4). There were no historic (pre-1992) vegetation monitoring data available. Vegetation community mapping within the assessment transect area was therefore used as a method of extrapolating existing datasets across an assessment area and as a surrogate for long-term time-series vegetation monitoring. Vegetation mapping was completed for the current period from 2013 aerial photography, and ground-truthed as part of this project. These data were then used to determine hydrological characteristics for each of the mapped vegetation associations, using surface water monitoring data for the same time period in association with the LiDAR DEM.

The 2013 vegetation community mapping data were then used to map vegetation communities thought to be reliably distinguishable from a 1978 colour aerial photograph. Given that there was a significant groundwater level decline between the 1978 and 2013 epochs (approximately 0.81 m decline in groundwater, resulting in approximately 0.28 m decline in maximum surface water levels (Table 4.3) in the wetland), the two epochs were used to evaluate the vegetation community scale response to a known direction change in hydrology (drying). The use of the historic and current time epochs enabled an initial assessment of the scale and direction of vegetation change that has occurred at the site as a result of the history of groundwater level decline.

Both the hydrological characteristics and the vegetation change analysis was used as the basis for the development of a conceptual model for Middlepoint Swamp.

Mapping of vegetation communities

Five historic aerial photographs taken between 1978 and 2013 (as listed in Table 4.4) were rectified within ESRI ArcGIS® to the 2013 aerial imagery and visually analysed for changes in extent of discernable vegetation communities. The images showed a gradual increase and retraction of vegetation communities along the elevation gradient as a result of changing hydrology. The most obvious of these was the gradual encroachment of *Juncus kraussii* into the charophyte/open water basin. Two aerial photo epochs, which represented the greatest groundwater level change, and the greatest visual change in vegetation communities (1978 and 2013) were selected for vegetation mapping and further analysis. Vegetation communities were manually digitised in ESRI ArcGIS®, with the use of vegetation monitoring data (overlayed spatial locations of vegetation monitoring quadrats) to identify likely species association within each visually homogeneous patch.

The vegetation mapping produced for the 2013 aerial photography was verified through ground-truthing in autumn 2014. Each point where a mapped transition in vegetation community crossed each of the three transects (Figure 4.7) was ground-truthed and a field GPS coordinate recorded. Adjustment to the mapping was made as a result of the ground-truthing, and the distribution of the same vegetation communities was then mapped for the 1978 epoch. A total of twelve individual vegetation communities were mapped in both epochs (Figure 4.7).

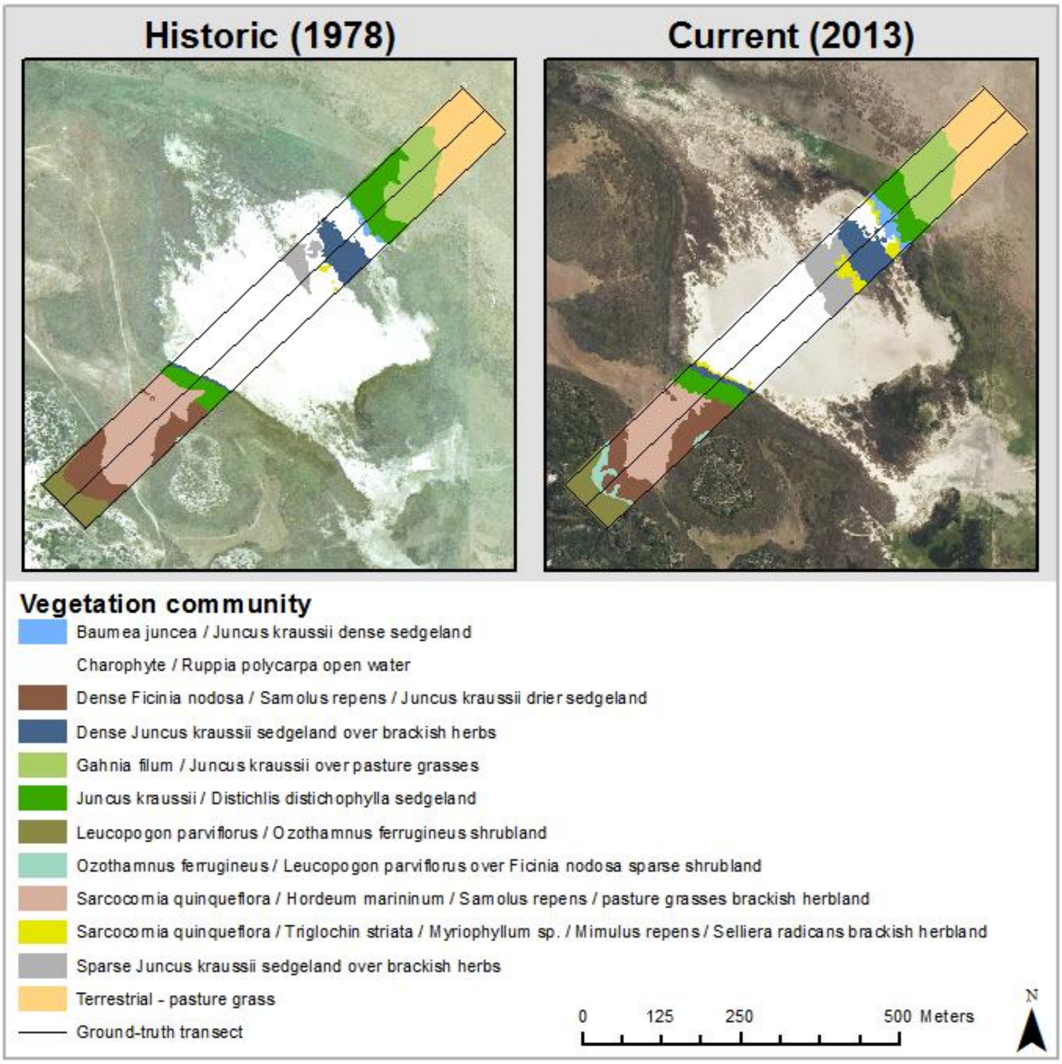


Figure 4.7 Vegetation community mapping for historic (1978) and current (2013) time epochs for Middlepoint Swamp

<u>Hydrological characteristics of vegetation communities</u>

The vegetation community polygon layers were converted to 2 x 2 m pixel rasters for further analysis. An overlay analysis was performed with elevation data obtained from the 2 m LiDAR DEM, and surface water level data was then used in conjunction with the DEM values (by subtraction) to calculate the maximum and average water level depths at each pixel for the 2012-2013 and 2013-2014 inundation periods. The number of days where water depth exceeded zero was also calculated for each pixel for both time periods, providing a yearly duration of inundation (days per year). Data for both years was then averaged to provide average maximum depth and duration over the 2012-2014 monitoring period for each pixel in the assessment area.

These data were used to initially identify the separation of hydrological characteristics (depth and duration) between the vegetation communities mapped, and in order to rank communities from wettest to driest hydrological niche preference. Figure 4.8 provides an initial box-plot assessment of the hydrological and elevation segregation between the mapped vegetation communities. The box-plots (Figure 4.8) showed good segregation between occurrence and hydrological parameters (water depth and duration) for most vegetation communities, with the greatest separation observed for the inundation duration parameter. This agrees with Casanova & Brock (2000) who also found that the duration of inundation was more important than depth of inundation in segregating wetland plant communities. Communities tended to exhibit distinct domains of preferred duration within the overall range of these data, with minimal overlap between them. Further modelling of hydrological niches was performed in order to determine vegetation community hydrological preferences (see Section 4.4.1).

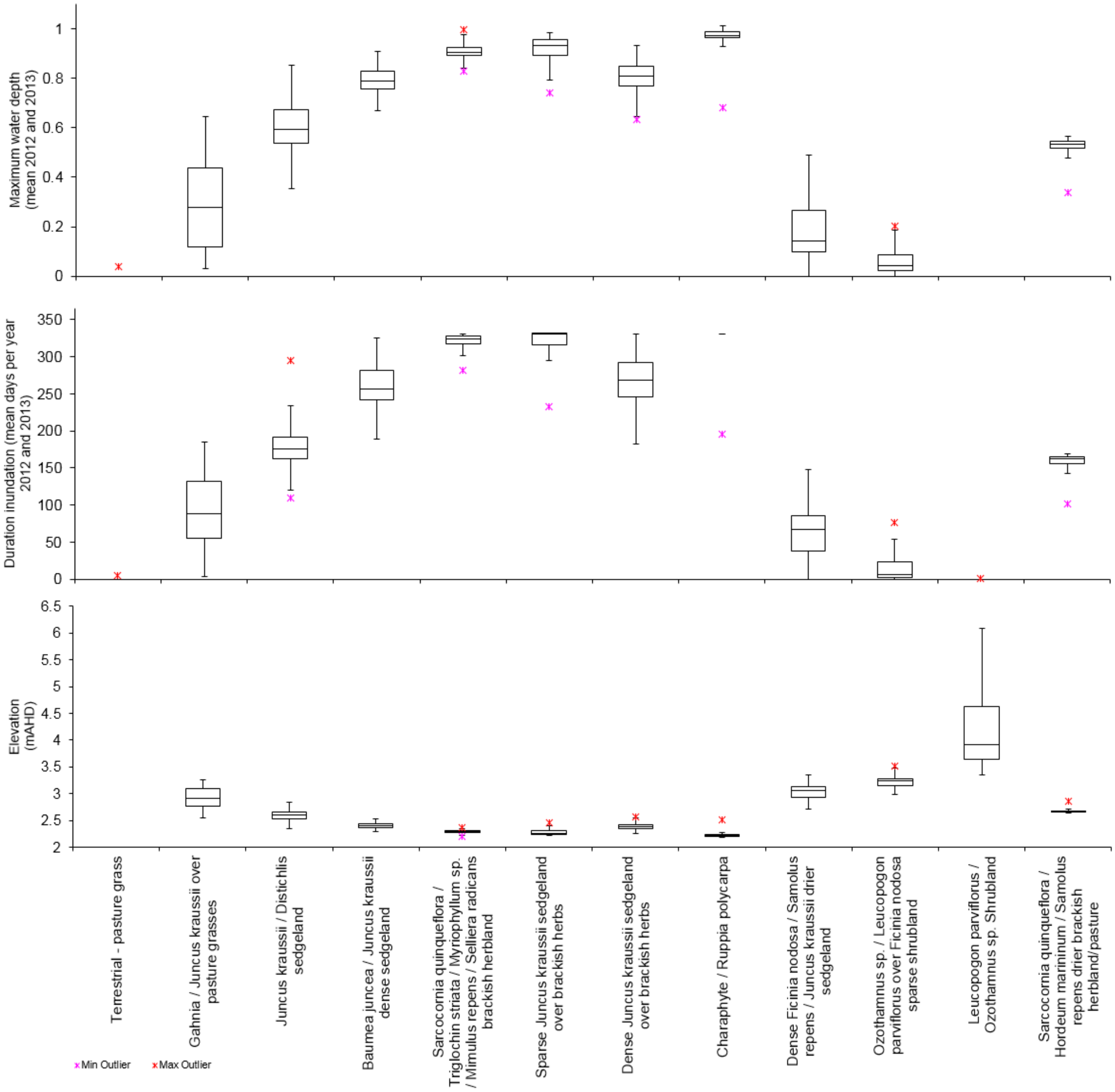


Figure 4.8 The median (solid line), and 25th, 75th (boxes) percentiles, and minimum and maximum (whiskers) of key hydrological and elevation parameters for mapped vegetation communities in Middlepoint Swamp assessment area

<u>Spatio-temporal analysis of vegetation change</u>

Changes within the wetland vegetation communities at Middlepoint Swamp were explored by a spatial analysis that involved overlaying the historic and current vegetation mapping to determine the type and location of change that occurred between the two epochs using ESRI ArcGIS® Spatial Analyst Combine tool. This analysis identified the major inter-community changes that occurred within the wetland vegetation under a drying scenario.

Table 4.7 provides a vegetation community change matrix between the 1978 and 2013 rasters, with values representing 2 x 2 m cells (pixels). A total of 4270 cells (18.1% of all raster cells in the assessment area) recorded a change in vegetation community, with the majority of cells (81.9%) remaining unchanged. Changes from charophytes / *Ruppia polycarpa* open water habitat (Char8) accounted for 34.9% of the total change, of which 858 cells (57.5%) changed to Sparse *Juncus kraussii* sedgeland over brackish herbs (SpJu6), and 440 cells (29.5%) changed to *Sarcocornia quinqueflora* / *Triglochin striata* / *Myriophyllum* sp. /

Mimulus repens / Selliera radicans brackish herbland (Sarc5). *Juncus kraussii / Distichlis* sedgeland (JunK3) accounted for 21.1% of the total change, of which a total of 415 cells (46.1%) changed to *Gahnia filum/ Juncus kraussii* over pasture grasses (Gahn2), 225 cells (25%) changed to *Baumea juncea / Juncus kraussii* dense sedgeland (Baum4), and 211 cells (23.4%) changed to Dense *Juncus kraussii* sedgeland over brackish herbs (DenJ7). Modest increases in drier and terrestrial vegetation communities were shown, with 460 cells changing from *Sarcocornia quinqueflora / *Hordeum marininum / Samolus repens* drier brackish herbland/pasture (SarcP12) to Dense *Ficinia nodosa / Samolus repens / Juncus kraussii* drier sedgeland (DenF9), 422 cells changing from DenF9 to *Ozothamnus ferrugineus / Leucopogon parviflorus* over *Ficinia nodosa* sparse shrubland (Ozoth10), 309 cells changing from Gahn2 to Terrestrial pasture grass (TerrP1), and 102 cells changing from DenF9 to *Leucopogon parviflorus / Ozothamnus ferrugineus* Shrubland (Leuc11).

The change matrix also identifies the direction of change, where vegetation communities were sorted from wettest to driest hydro-period (as determined in Figure 4.8), indicating that of the cells that changed, the majority (3260 cells, 76.4%) became drier vegetation communities, 591 cells (13.8%) became vegetation communities with similar water requirements, and 419 cells (9.8%) became wetter vegetation communities. It was difficult to reconcile a shift in zonation that favoured a community with higher water requirements given the observed decline in groundwater levels and overall drying trend. This may be indicative of an underlying uncertainty in vegetation mapping for the 1978 image which was not able to be assessed. The matrix also provides an indication of the likelihood of a vegetation community transitioning to another community based on the historical changes. It was evident that vegetation communities respond to water-level changes by shifting, expanding and contracting along a continuum, whilst the wetland vegetation community zonations (relative position and sequence of vegetation communities along an elevation gradient) remained similar. This was further explored by interpreting vegetation change along elevation gradients for the three assessment area transects (Figure 4.9).

Table 4.7 Wetland vegetation transition matrix for Middlepoint Swamp transects, 1978–2013 (number of cells*)

			Change to 2013												
	Wetland Community	Total 1978	Char8	Sarc5	SpJu6	DenJ7	Baum4	JunK3	SarcP12	Gahn2	DenF9	Ozoth10^	Leuc11^	TerrP1#	Decrease	% of Total Change
Change from 1978	Char8	7252	-	440	858	108	86	0	0	0	0	0	0	0	**1492**	**34.9**
	Sarc5	48	0	-	0	0	0	0	0	0	0	0	0	0	**0**	**0**
	SpJu6	586	1	0	-	0	0	0	0	0	0	0	0	0	**1**	**0.02**
	DenJ7	1225	14	74	36	-	0	0	0	0	0	0	0	0	**124**	**2.9**
	Baum4	149	0	12	0	0	-	16	0	0	0	0	0	0	**28**	**0.6**
	JunK3	2406	0	5	0	211	225	-	0	415	44	0	0	0	**900**	**21.1**
	SarcP12	3429	0	0	0	0	0	330	-	0	460	0	0	0	**790**	**18.5**
	Gahn2	2239	0	0	0	0	0	80	0	-	0	0	0	309	**389**	**9.1**
	DenF9	2240	0	0	0	0	0	0	12	0	-	422	102	0	**536**	**12.5**
	Ozoth10^	28	0	0	0	0	0	0	0	0	0	-	0	0	**0**	**0**
	Leuc11^	1142	0	0	0	0	0	0	0	0	0	10	-	0	**10**	**0.2**
	TerrP1#	2855	0	0	0	0	0	0	0	0	0	0	0	-	**0**	**0**
	Total	23599													**Total**	
	Increase		**15**	**531**	**894**	**319**	**311**	**426**	**12**	**415**	**504**	**432**	**102**	**309**	**4270**	
	% of Total Change		**0.3**	**12.4**	**20.9**	**7.5**	**7.3**	**9.9**	**0.3**	**9.7**	**11.8**	**10.1**	**2.4**	**7.2**		
	*****Net Change***		*-1477*	*531*	*893*	*195*	*283*	*-474*	*-778*	*26*	*-32*	*432*	*92*	*309*		

* raster cells representing 2 x 2 m pixel in the DEM. ** Net change (increase – decrease)
Orange: indicates transition to a drier vegetation community; Green: no change; Yellow: indicates transition to a vegetation community with similar water requirements; Blue: indicates transition to a wetter vegetation community.
Wetland vegetation communities: **Char8** = Charophytes / Ruppia polycarpa open water; **Sarc5** = Sarcocornia quinqueflora / Triglochin striata / Myriophyllum sp. / Mimulus repens / Selliera radicans brackish herbland; **SpJu6** = Sparse Juncus kraussii sedgeland over brackish herbs; **DenJ7** = Dense Juncus kraussii sedgeland over brackish herbs; **Baum4** = Baumea juncea / Juncus kraussii dense sedgeland; **JunK3** = Juncus kraussii / Distichlis sedgeland; **SarcP12**= Sarcocornia quinqueflora / Hordeum marininum / Samolus repens drier brackish herbland/pasture; **Gahn2** = Gahnia sp./ Juncus kraussii over pasture grasses; **DenF9** = Dense Ficinia nodosa / Samolus repens / Juncus kraussii drier sedgeland; **Ozoth10** = Ozothamnus ferrugineus / Leucopogon parviflorus over Ficinia nodosa sparse shrubland; **Leuc11** = Leucopogon parviflorus / Ozothamnus ferrugineus Shrubland; **TerrP1** = Terrestrial - pasture grass (Appendix B)
\# vegetation community only occurs in areas grazed by stock ^ vegetation community only occurs in un-grazed (conservation) areas.

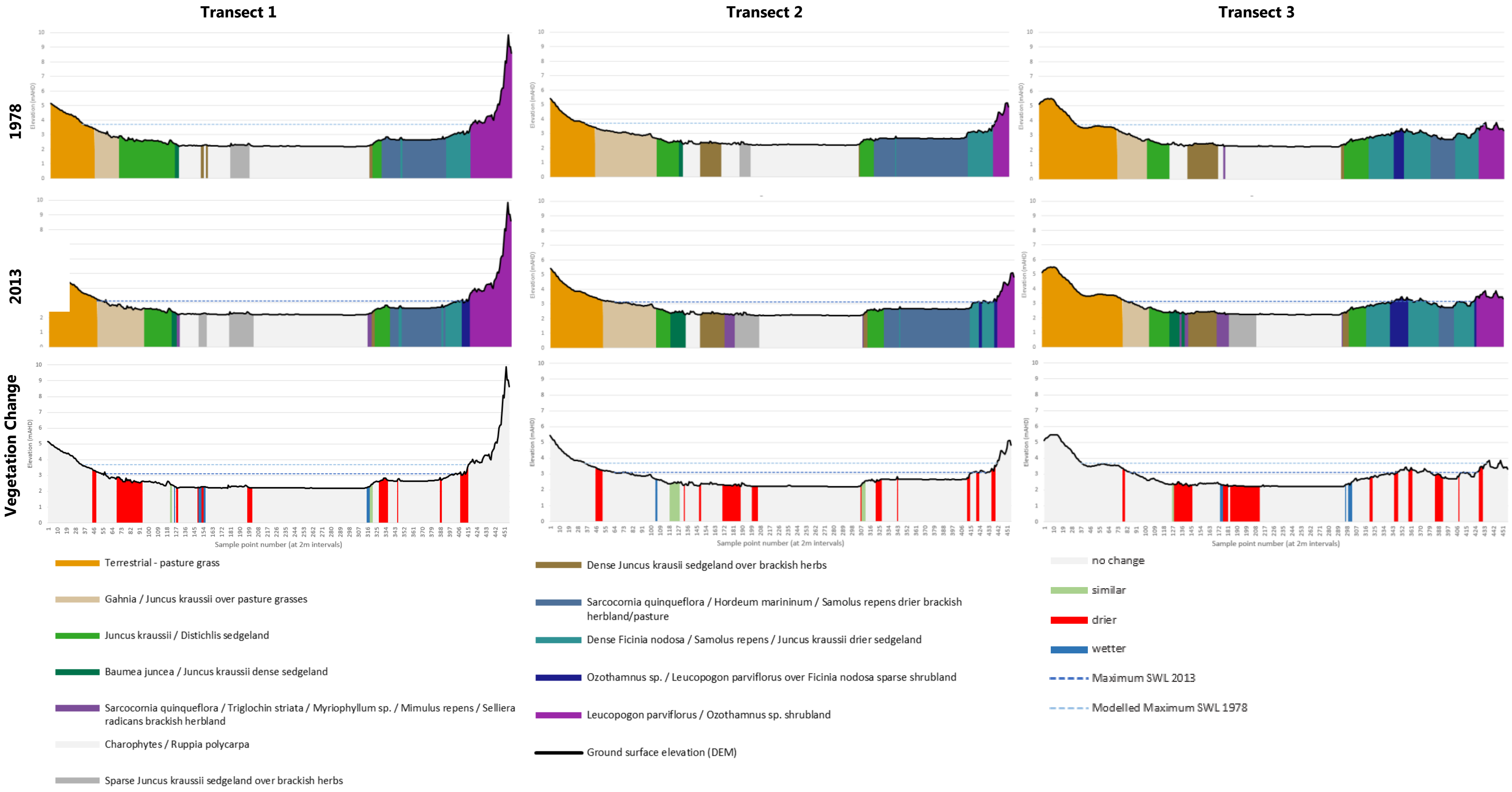


Figure 4.9 **Vegetation community change between 1978 and 2013 along three transects within the Middlepoint Swamp assessment area, indicating direction and type of change**

Transects illustrated in Figure 4.9 indicate the general maintenance of vegetation community zonation between the two epochs, suggesting that changes in vegetation communities are potentially dependent on the original vegetation present prior to a change in hydrological conditions. This may also be as a result of a greater ability of some species to persist in sub-optimal conditions or differential competitive ability under drying conditions such as higher propagule density or superior seed energy reserves, favouring establishment. The overall change along the elevation gradient is shown (Figure 4.9) to be one of a gradual drying, and shifting towards vegetation communities more tolerant of drier conditions. Figure 4.9 also shows the differences between the grazed (left-hand side of the cross-section), and un-grazed (right-hand side), where terrestrial pasture grasses are encroaching in the grazed area, and native terrestrial and drier vegetation communities (*Leucopogon parviflorus* / *Ozothamnus ferrugineus* shrubland and *Ozothamnus ferrugineus* / *Leucopogon parviflorus* over *Ficinia nodosa* sparse shrubland) gradually encroach from the un-grazed dunes. The preferential invasion of exotic species as a consequence of a decline in the occupancy and cover of native species that are adapted to the historical hydrological regime has been identified by Catford et al. (2014), attributed to drying conditions.

<u>Key Findings</u>

The hydro-niche and spatio-temporal analyses of the mapped vegetation communities provide an indication of how wetland vegetation in Middlepoint Swamp may respond to alterations in water levels due to projected climate change of a comparable magnitude that occurs over a similar timeframe. This understanding of historical changes was used to develop a conceptual model for Middlepoint Swamp. Key findings of the analysis indicate that hydro-period, prior vegetation and transition states, and land use were important variables in determining the magnitude and direction of vegetation community change in Middlepoint Swamp between 1978 and 2013 and are summarised in Table 4.8. Similar patterns of wetland vegetation response to water level fluctuations, particularly water level decline, have been documented (Keddy & Reznicek 1985; Casanova & Brock 2000; Hebb 2003; Hudon et al. 2006; Hebb et al. 2013; Catford et al. 2014).

Table 4.8 Key findings of the hydro-niche and spatio-temporal analyses of mapped vegetation communities

Hydro-period	Prior vegetation and transition states	Land use
• Vegetation communities at Middlepoint Swamp generally occur within hydro-niches determined by depth and duration • Duration of inundation performs slightly better than maximum inundation as a determinant for vegetation community	• The greatest change in vegetation communities in a drying period was a reduction in the wettest vegetation community (the vegetation community with the highest depth and duration of inundation preferences) • Modest encroachment of terrestrial and near-terrestrial vegetation communities has occurred • The majority of change in vegetation communities was towards a drier ecosystem • Zonation (relative position and sequence) of vegetation communities along an elevation remain similar • Changes in vegetation communities are influenced by the original vegetation prior to a change in hydrological conditions • Preferential invasion by exotic species as a result of altered hydrological conditions has occurred	• The land use of the site (grazed or un-grazed) will influence the type of terrestrial vegetation encroachment

4.3.2.2 Conceptulisation of the ecosystem function

The frequency, timing, duration and magnitude of groundwater level fluctuations influence surface water depth and duration of flooding, key factors that shape the composition and extent of wetlands, and more specifically influence the abundance and spatial distribution of vegetation communities within a wetland complex (Casanova & Brock 2000; Hebb et al. 2013). Persistent, long-term surface water level changes as a response to declining groundwater leads to a compression of the natural water level, depth and duration range, and results in changes in the spatial distribution of vegetation communities and other dependent biota. From analysing the historic changes in spatial vegetation community distribution between 1978 to 2013 (a period of declining groundwater levels), it was also evident that vegetation communities respond to water-level changes by shifting, expanding and contracting along a continuum, however generally the wetland vegetation community zonation (relative position and sequence of vegetation communities along an elevation gradient) were maintained. As such, there was not a total re-assortment of vegetation communities as a result of a change in water depth and duration, and the likelihood of vegetation community change was not only dependent on hydrological properties (depth and duration), but also the existing vegetation community prior to change.

A conceptual diagram (Figure 4.10) was produced to illustrate the conceptual understanding of the impact of declining groundwater levels on wetland vegetation as a result of climate change at Middlepoint Swamp. The upper cross-section (Figure 4.10) depicts an approximation of the current scenario, where groundwater seasonally discharges into the wetland basin, and remains shallow throughout the year supporting wetland vegetation communities requiring near permanent inundation durations at the lowest elevations. The open water basin is dominated by charophytes and submergent aquatic plants. Emergent fringing sedges more tolerant of seasonal inundation fluctuations fringe the littoral zones, and drier emergents dominate the outer damp areas.

The middle cross-section (Figure 4.10) illustrates the expected impact of a further arbitrary decline in groundwater levels (based on previously observed impacts). Groundwater no longer discharges into the wetland basin, although remains shallow supporting seasonal surface water from rainfall. Reduced depth and duration of inundation result in the gradual establishment emergent sedges (amphibious fluctuations tolerant) into the wetland basin, and the loss of open water habitat. Terrestrial vegetation communities begin to encroach the margins of the wetland. Under grazed land use conditions, this results in the invasion of exotic pasture grasses, and increased grazing pressure from stock with associated pugging, increases in nutrients and turbidity. Where intact native terrestrial vegetation buffers the wetland, terrestrial shrubs begin to encroach. With ongoing declines in groundwater, and reduced rainfall, GDEs are at risk of gradual terrestrialisation (represented by the lower cross-section in Figure 4.10), where aquatic vegetation communities are replaced by terrestrial species. The resultant ecosystem that develops under a severe change in hydrology will be reflective of the existing buffering vegetation.

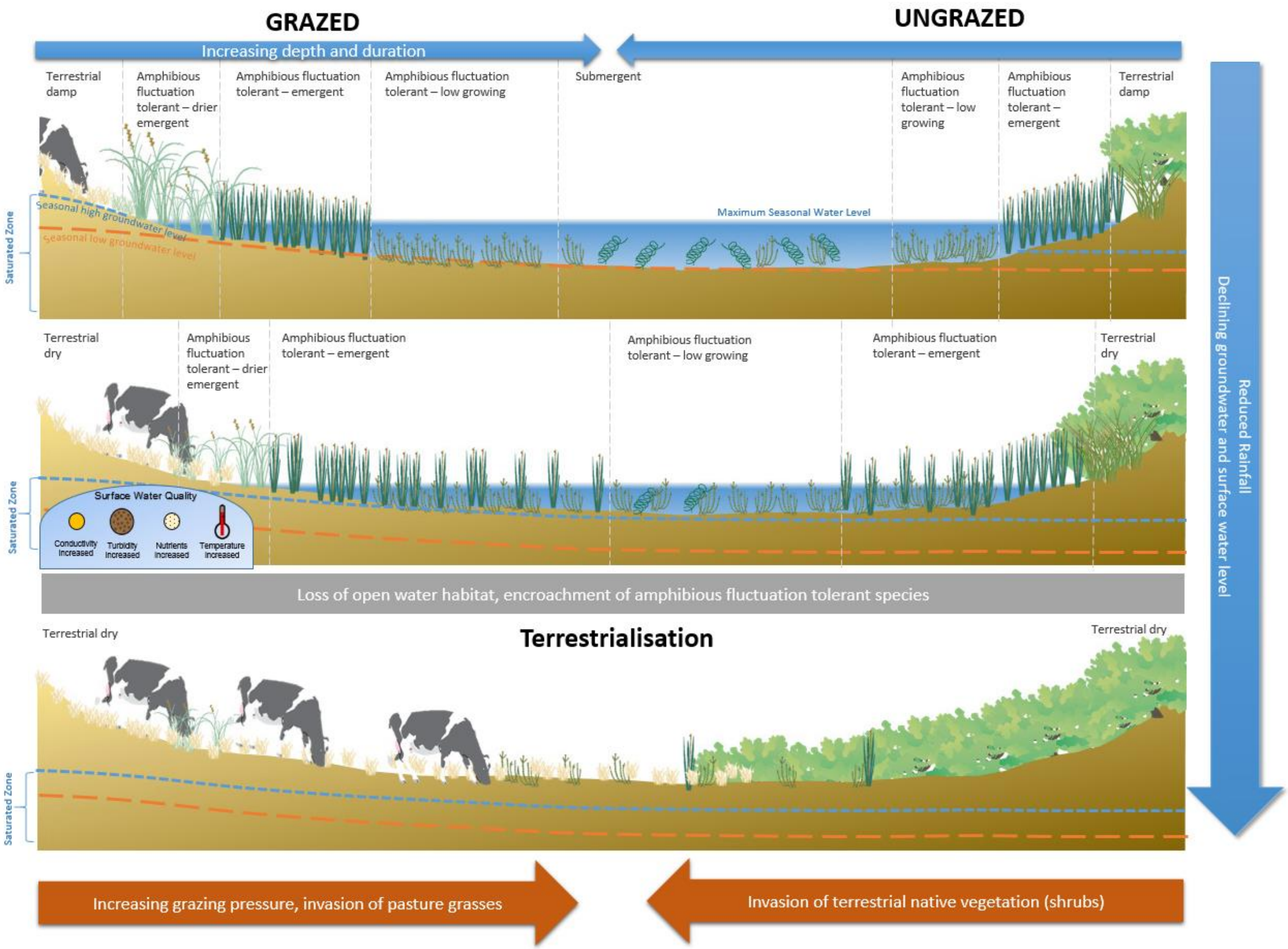


Figure 4.10 Conceptual diagram of the impact of hydrological change on vegetation communities along a cross-section of Middlepoint Swamp, depicting current and future groundwater level scenarios

Based on the key findings of the analysis of vegetation community and hydrology data for Middlepoint Swamp (Table 4.8; Figure 4.10), a simplified conceptual model (Figure 4.11) was developed to support the implementation of a Bayesian Belief Network (BBN) for analysis of risk from climate change scenarios.

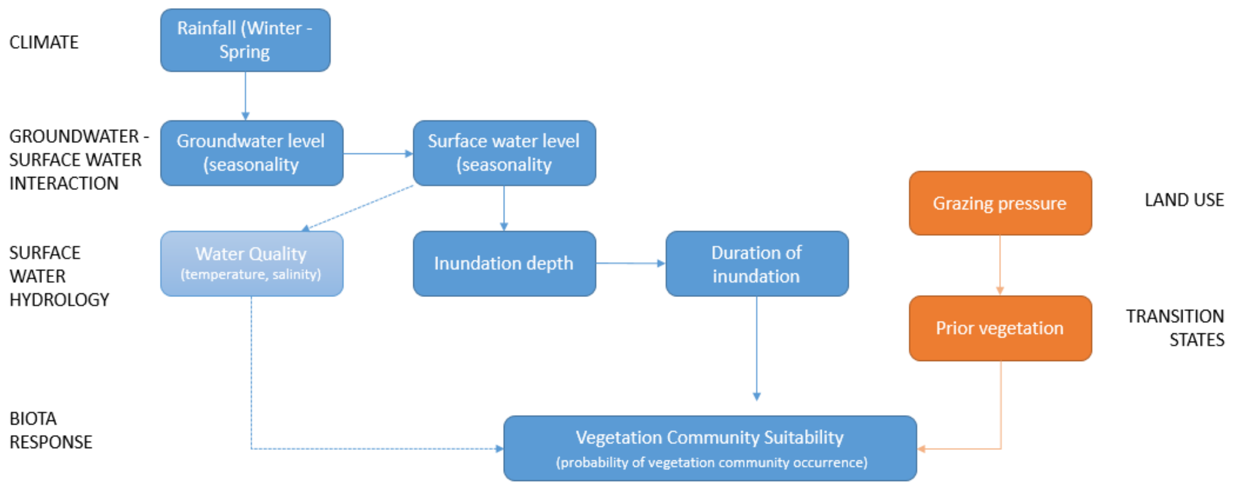


Figure 4.11 Conceptual model of vegetation community response from climate change at Middlepoint Swamp GDE

The conceptual model draws upon the basic model frameworks developed for case study GDE sites in Chambers et al. (2013b), and incorporates key drivers and influencers (hydro-period: depth and duration of inundation; land use; and probable vegetation transition states) identified in this report. The model also reflects relationships between climate, groundwater and surface water level and surface water hydrology developed and modelled in Section 4.2. Water quality, was included in the conceptual model (in recognition of the likely rise in salinity due to increased evapo-concentration (Nielsen & Brock 2009) and temperature), however was not further modelled or assessed. It is recognised that a number of additional parameters not identified within the conceptual model (e.g. other climate variables, soil characteristics, pugging from stock, competitive advantages of atmospheric CO_2 enrichment of C_3 over C_4 plants) may also have an effect on vegetation community response, however reliable relationships were unable to be established with the data available and within the scope of this study.

4.4 Step 4: Evaluate risk

The risk assessment framework developed by Chambers et al. (2013a) specifically identifies Bayesian Belief Networks (BBN) as a tool for determining the probabilities of risk to GDEs as a consequence of declining groundwater levels. BBNs are models that represent the correlative and causal relationships between variables graphically and probabilistically, and have been widely adopted in applications where causality plays a role, but our understanding of relationships is incomplete (Speldewinde 2013). For this reason, BBN's can provide effective decision support tools for problems involving uncertainty and probabilistic reasoning (Cain 2001). Several studies have demonstrated the utility of BBNs in capturing and integrating expert knowledge and empirical data to model the effects of climate change on wetland ecosystems (Chambers et al. 2013b; Speldewinde 2013; Mitchell et al. 2013; Dyer 2014), and were also identified as a useful tool for modelling the biotic response in Middlepoint Swamp.

Two components of risk to the Middlepoint Swamp ecosystem were evaluated spatially for each raster cell in the assessment area, and for each identified scenario, based on modelling and available data:

1. The impact on vegetation communities was predictively modelled for climate change scenarios, illustrating the most probable changes to vegetation communities as a result of predicted hydrological changes
2. The overall risk of the ecosystem becoming terrestrial (i.e. no longer supporting hydric vegetation communities)

A combination of modelling approaches were used to evaluate risk to vegetation communities at Middlepoint Swamp based on (and constrained to) the relationships determined in the conceptual model (Figure 4.12). The outputs of both the modelling of climate and groundwater level (Section 4.2.1), and groundwater levels and surface water characteristics (Section 4.2.2), resulted in the final hydrological output of duration of inundation for each chosen scenario. A BBN was then developed for a sub-set of the entire conceptual model in order to determine the probabilities of vegetation response with use of GIS raster dataset inputs for duration of inundation, grazing pressure, and prior vegetation for each scenario (Figure 4.12). The decision to simplify the BBN to a sub-set of the conceptual model was due to the trial nature of the study, although additional nodes for climate, groundwater and surface water could be integrated using relationships developed in this report.

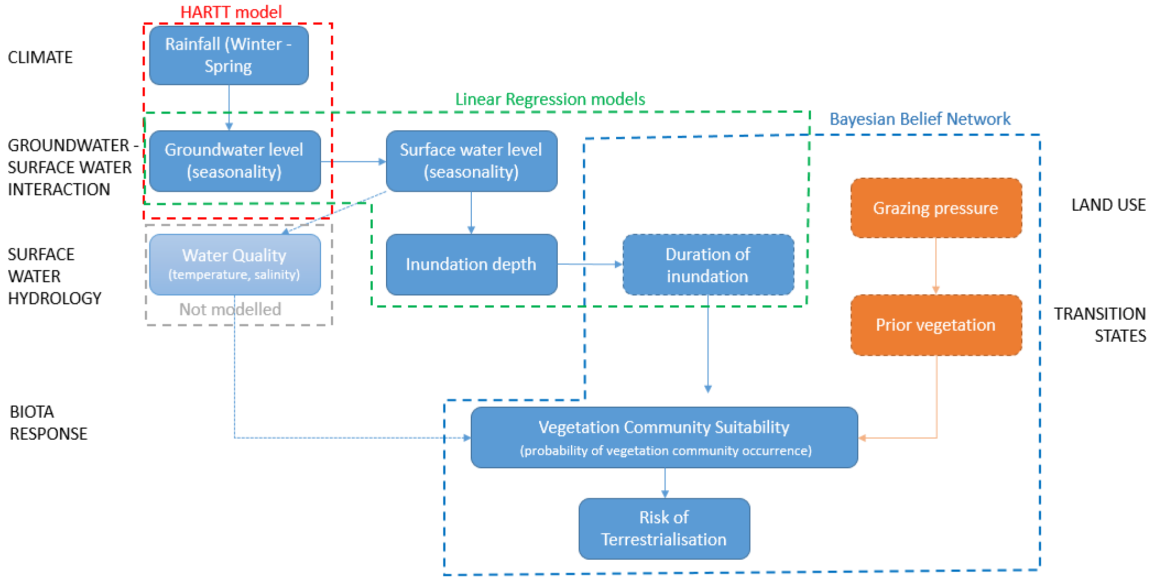


Figure 4.12 Modelling methods applied to the conceptual model in order to evaluate risk to vegetation communities at Middlepoint Swamp (dotted-border nodes represent GIS inputs)

Once the conceptual model had been developed, the vegetation response and risk modelling process using a BBN consisted of two main steps: (a) development of the predictive model and determination of thresholds; and (b) linking the predictive model to GIS scenario case files in order to spatially illustrate vegetation community suitability and risk of terrestrialisation (Figure 4.13).

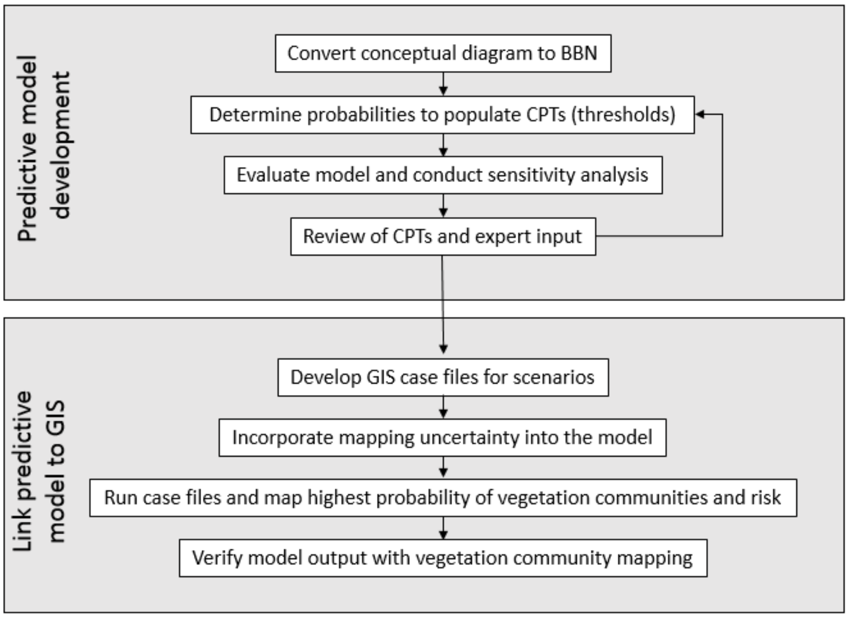


Figure 4.13 Schematic showing the sequential steps used to model and map vegetation response to climate change at Middlepoint Swamp

4.4.1 Vegetation community response modelling assumptions

Key assumptions of the modelling of vegetation community response from climate change at Middlepoint Swamp were that:

- hydrological preferences of vegetation communities can be adequately described using the annual duration of inundation.
- changes in vegetation community zonation can be predicted on the basis of observed hydrological niche, grazing pressure, and transition states. Any other variables affecting vegetation community condition (such as soil properties or salinity) are assumed uniform across the site and invariant over the modelling period.
- transition states between vegetation communities observed between 1978 and 2013 (a period of groundwater level drawdown) were predictive of likely future changes to further groundwater level drawdown. Competitive and other interactions that produce extant zonation are assumed to persist under future climatic conditions.
- other effects of a changing climate (such as increased temperature, carbon dioxide concentration, seawater intrusion and sea level rise) are not considered, nor are the potential effects of such changes on competitive hierarchies (e.g. through changes in C_3 and C_4 plants relative carbon uptake efficiencies).
- grazing regime, stock type, intensity and spatial exposure of vegetation communities to stock damage has remained the same throughout the modelling period, and for future scenarios.
- no new anthropogenic disturbance is introduced including no change in matrix land use or hydrology (e.g. either further drainage or increased weir heights or manipulation; any slashing or burning regime imposed at the site).
- no new plant species (exotic or native) are introduced to the site that may change zonation or displace existing vegetation communities.
- assumptions inherent in the modelling of projected hydrological characteristics (Sections 4.2.1 and 4.2.2) also apply to the use of these data in modelling vegetation response. Uncertainties in modelled input data were therefore required to be recognised and accounted for in the development of the vegetation response BBN.

4.4.2 BBN Predictive model development

4.4.2.1 Convert conceptual diagram to a BBN

The sub-set of the conceptual diagram identified in Figure 4.12 was converted into a BBN using Netica™ software (Norsys Software Corporation 1998) (Figure 4.14). The model was composed of key predictive variables identified in the conceptual model and available as GIS raster files for the assessment area, an intermediate node reflecting uncertainty in the GIS input of mapped duration of inundation, hydrological niche thresholds for each vegetation community, final probability of suitable conditions for the establishment of each vegetation community, and the overall risk of terrestrialisation. Intermediate nodes for land use and prior vegetation were not required as these input data were assumed to have 100% certainty within the constraints of this model. Appendix C provides a description of all nodes and their output states.

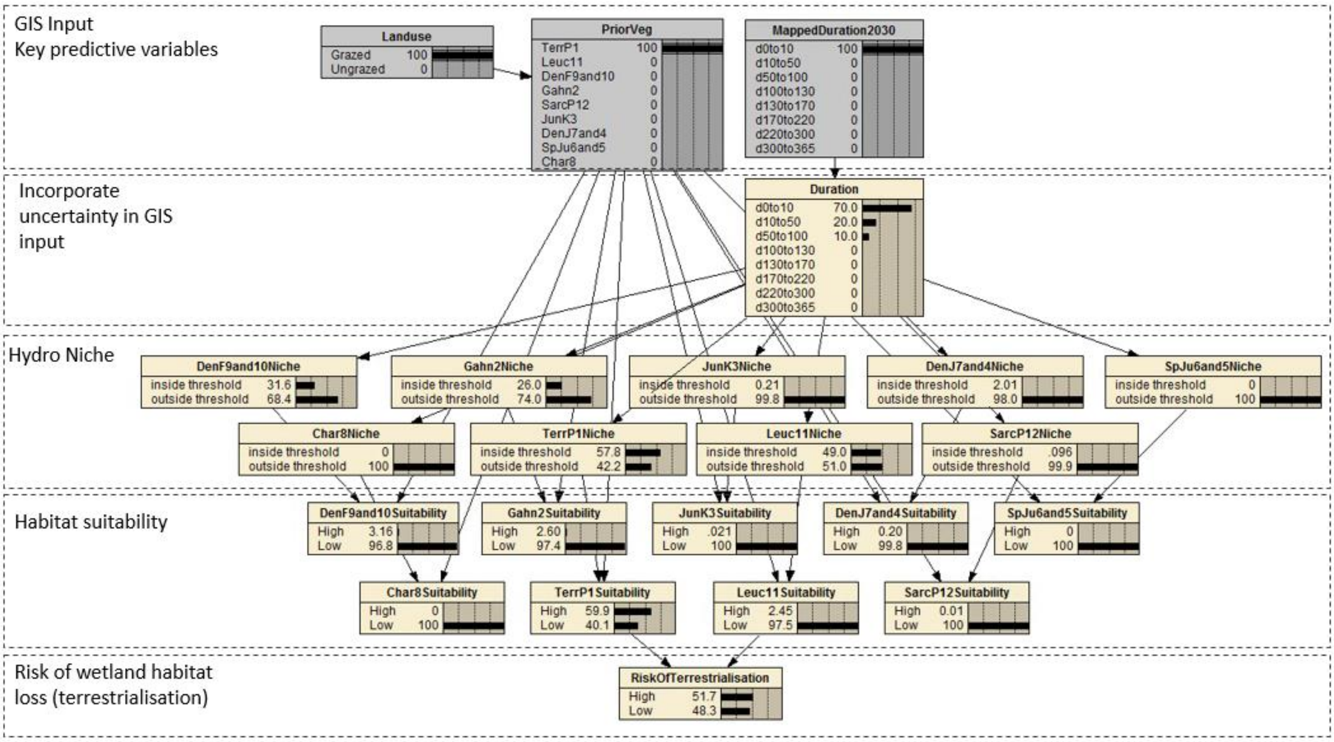


Figure 4.14 Populated BBN for Middlepoint Swamp vegetation communities habitat suitability and overall risk of terrestrialisation

This conversion also required quantifying the relationships between the GIS inputs (duration of inundation, and prior vegetation) and vegetation community suitability and also discretising continuous datasets (e.g. inundation duration). BBNs use conditional probability tables (CPT) to quantify relationships between nodes (boxes) in a conceptual model. In the case of the Middlepoint Swamp example, a combination of empirical data, and expert ecologist input was used to determine thresholds and subsequently populate CPTs.

4.4.2.2 Determine probabilities to populate CPTs

<u>Hydrological niche model construction</u>

Hydrological niche modelling was used to determine duration of inundation thresholds (defined as either inside threshold, outside threshold in the BBN) for mapped vegetation communities at Middlepoint Swamp. It was found that duration of inundation was the strongest hydrological determinant of vegetation community segregation (Figure 4.8), a finding which was also supported by Casanova & Brock (2000).

Hydrological niches of mapped vegetation communities were modelled using multinomial logistic regression using package 'nnet' (Venables & Ripley 2002) for R (R Core Team 2014). This form of generalised linear modelling (GLM) is an extension of binary logistic regression, modelling the log odds of one category (the baseline) being present compared with all other modelled categories as a linear function of the explanatory variables (Zuur et al., 2007). Model coefficients (Appendix D) represent the amount by which the log odds of the vegetation community being present, compared with Terrestrial pasture grass (TerrP1) being present, changes with the explanatory variable. Vegetation community TerrP1 was chosen as the baseline to allow interpretation of model coefficients against the value of a terrestrial vegetation community.

Vegetation communities tended to prefer distinctive ranges of elevation, indicating clear preferences along the hydrological gradient (Figure 4.8). Initial model fits indicated broadly unimodal shaped probability distributions as a function of inundation durations. Mapping indicated that distinctive vegetation communities were present at opposite ends of transects (Figure 4.7), despite there being considerable overlap in hydrological niche preference. As indicated in the conceptual model, land use was considered the likely cause differentiating one community from the other with more northerly areas of the Swamp subject to higher grazing pressure. This predictor was incorporated (along with prior vegetation transitions) within the BBN structure.

Hydrological preference was modelled using duration, a continuous variable representing the average period of time per year (in days) that water tables were at or above wetland sediments for the observed data point. The optimal model was determined based on the maximum amount of deviance explained and included both linear and quadratic terms for duration. The final model explained 72% of null model deviance and all model terms were significant (Appendix D - coef ref; duration - Wald test all $p<0.01$).

An attempt was made to model all 12 mapped vegetation communities but results indicated combination of some groups was warranted. Two groups of low prevalence and high overlap in niche preference (vegetation communities Sarc5 and SpJu6, samphire and wet herb associations) were combined. Vegetation communities Baum4 and DenJ7 (dominated by *Baumea juncea* and *Juncus kraussii*), were also pooled, being of similarly low prevalence and occupying adjacent, strongly overlapping hydrological niches.

<u>Hydrological niche model validation</u>

Data were randomly split 70:30 into training and testing datasets, with the latter held back from model construction and used to assess predictive capability. With the model successfully verified, final models were constructed based on the complete dataset. Validation proceeded by use of a confusion matrix and calculation of a range of statistical measures presented in Fielding & Bell (1997). Sensitivity and specificity reflect conditional probabilities: whether the model predicts an association should be present given it was observed and the inverse of this respectively. In both cases a 1 indicates perfect performance. Sensitivity was consistently lower than specificity. This was also observed in the false negative (predicting absence when it was present), which was worse than false positive, with vegetation communities JunK3, SpJu6and5 and Ozoth10 particularly poor. From this it can be concluded that the model tends to predict absences with more reliability than presence. The standard normal statistic (z-scores) provides one measure as to whether the prediction is better than could be expected at random and is calculated according to a null hypothesis of no difference between the observed and expected correct classification rates (Fielding and Bell, 1997):

(observed – expected)/(expected(Number- expected)/N)

The kappa statistic was also included as it provides a robust measure of model performance in terms of its improvement over chance. This is usually interpreted as: <0.4 = poor; 0.4 - 0.75 = good; >0.75 = excellent. This statistic suggests most associations are predicted well (Table 4.9) and only two associations were poorly predicted. The reason for the two poor predictions are clear when the hydrological niche is viewed graphically (Figure 4.14).

Table 4.9 **Selected statistics summarising model predictive performance (from Fielding and Bell 1997).**

Vegetation Community:	**TerrP1**	**Gahn2**	**JunK3**	**DenJ7and 4**	**SpJu6and5**	**Char8**	**DenF9**	**Ozoth10**	**Leuc11**	**SarcP12**
Sensitivity	0.87	0.86	0.41	0.70	0.50	0.72	0.79	0.33	0.83	0.95
Specificity	1.00	0.97	0.97	0.97	0.90	0.99	0.99	0.98	1.00	0.96
False positive rate	0%	3%	3%	3%	10%	1%	1%	2%	0%	4%
False negative rate	13%	14%	**59%**	30%	**50%**	28%	21%	**67%**	17%	5%
kappa	0.92	0.72	0.44	0.69	**0.17**	0.76	0.81	**0.16**	0.90	0.78
z =	9.90	6.17	3.84	6.36	1.33	13.06	7.31	0.55	6.15	7.89
probability (Z > z)	0.00	0.00	0.00	0.00	**0.09**	0.00	0.00	**0.29**	0.00	0.00

Wetland vegetation communities: Refer to Appendix B for descriptions.
Marginal values are shown in bold. Sensitivity - conditional probability of correct classification pr(model T|observed); specificity - inverse of sensitivity pr(model F|not observed); false positive rate – proportion of samples predicted present when not observed; false negative – proportion of samples predicted absent when present; kappa – is the proportion of specific agreement

Vegetation community Ozoth10 was poorly distinguished from DenF9 because of the low prevalence of the *Ozothamnus* community samples leading to low maximum probability (Figure 4.15). This cannot be rectified with existing data and suggests pooling of the two (Ozoth10 and DenF9) could be warranted. The charophytes / *Ruppia polycarpa* open water community (Char8) was very well predicted, but the drier end of the gradient showed minimal differentiation with the pooled samphire-wet herbland group (SpJu6and5), which was among the poorest. Vegetation community JunK3 was of marginal predictive power and appears to be poorly distinguished because of uncertainties around the spatial extent of grazing; JunK3 and SarcP12 occupy very similar areas of hydrological niche space and only grazing differentiates between the probability of one being present over the other.

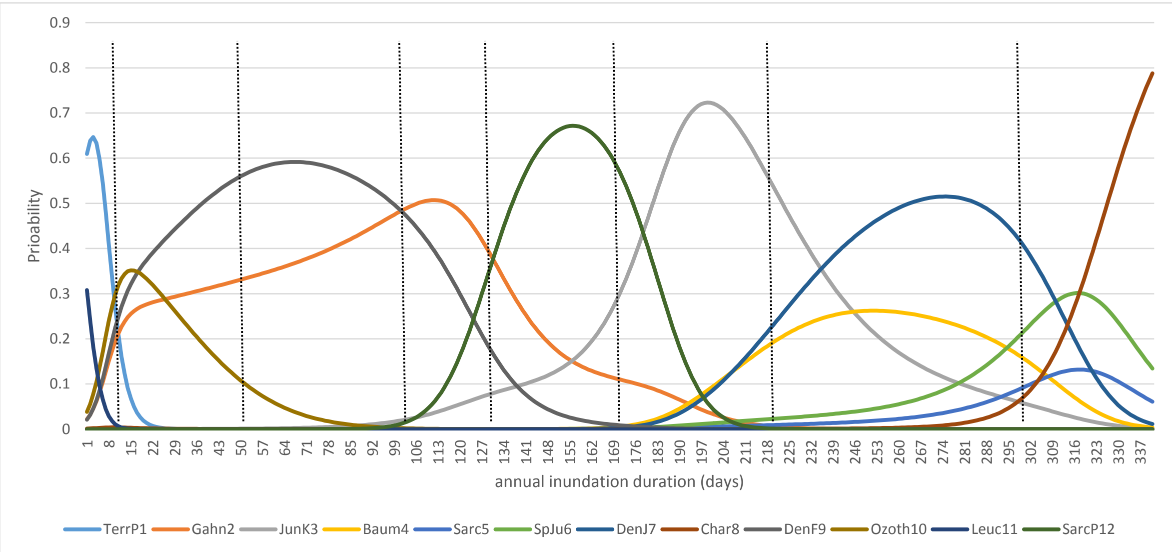


Figure 4.15 Hydrological niche space for the modelled vegetation communities at Middlepoint Swamp, with duration bin thresholds (vertical lines) used in the BBN (see Appendix B for vegetation community descriptions)

The hydrological niche space modelling was used to determine eight discreet ranges for duration of inundation (Figure 4.15) for the BBN and the highest probability of each vegetation community within each duration range was used to populate CPTs as the 'inside threshold' value. Groupings of vegetation communities with similar hydrological niches (vegetation communities Baum4 and DenJ7, Sarc5 and SpJu6, and DenF9 and Ozoth10) were maintained as groups in the final BBN, resulting a total of nine vegetation communities (or groups of similar communities) modelled.

Prior vegetation probabilities

The probability of a prior vegetation community transitioning to a new vegetation community under a drying scenario was calculated based on the changes already observed in a process similar to that detailed in Hebb et al. (2013). The transition probabilities were derived from the overlay of the wetland vegetation community rasters for 1978 and 2013 in the spatio-temporal change analysis (Section 4.3.2.1). Using the change matrix analysis table (Table 4.7) derived from the spatial analysis, transition probabilities were calculated in MS Excel® by dividing the number of cells that represented each unique transition within a specific wetland community by the total number of cells of that wetland community for the base year (1978). The transition probabilities were sorted from greatest to least likelihood of occurring for a particular wetland community and used to populate the CPT for vegetation community habitat suitability.

4.4.2.3 Populating the CPTs

The thresholds for the hydrological niche and transition states for each vegetation community were based on data and modelled relationships (as described in Section 4.4.2.2), with additional iterative expert input and adjustment as required. Two possible outcomes were defined for each vegetation community: High suitability, and Low suitability based on the duration of inundation and the prior vegetation community probabilities (e.g. Table 4.10 and 4.11).

The prior vegetation, or likelihood of one vegetation community transitioning to another, was based on previous observed changes in Middlepoint Swamp between 1978 to 2013. However, the scale of the observed hydrological change was small in comparison to the final projected changes for climate change scenarios to 2030 (refer to Table 4.2 and Figure 4.6). For this reason, vegetation transition probabilities were not limited to those observed. A transition probability of 10% was applied to any potential change provided it was within the hydrological niche threshold. Similarly, in recognition of some vegetation communities (e.g. *Gahnia filum*) being able to withstand prolonged periods of less than optimal inundation conditions (Ecological Associates 2009), a likelihood of the vegetation community being maintained although outside of the hydrological niche threshold was also applied (e.g. Table 4.11).

Table 4.10 An example conditional probability table for *Gahnia filum / Juncus kraussii* over pasture grasses (Gahn2) hydrological niche

Duration (of inundation, days per year)	Gahn2 Hydrological Niche (% probability)	
	*Inside threshold	Outside threshold
d0to10	19.4	80.6
d10to50	33.1	66.9
d50to100	47.9	52.1
d100to130	50.7	49.3
d130to170	37.1	62.9
d170to220	11.2	88.8
d220to300	0	100
d300to360	0	100
*populated from modelled hydrological niche space where the highest probability value for each duration category was populated as 'inside threshold' (refer to Figure 4.14)		

Table 4.11 An example conditional probability table for *Gahnia filum / Juncus kraussii* over pasture grasses (Gahn2) vegetation community suitability based on hydrological niche and prior vegetation probability data

Gahn2Niche	Prior vegetation (2013 veg mapping)	Gahn2 suitability (% probability)	
		High	Low
Inside threshold	TerrP1	10	90
Inside threshold	Leuc11	10	90
Inside threshold	DenF9and10	10	90
Inside threshold	Gahn2	82.6	17.4
Inside threshold	SarcP12	10	90
Inside threshold	JunK3	18.5	81.5
Inside threshold	DenJ7and4	10	90
Inside threshold	SpJu6and5	10	90
Inside threshold	Char8	10	90
Outside threshold	TerrP1	0	100
Outside threshold	Leuc11	0	100
Outside threshold	DenF9and10	0	100
Outside threshold	Gahn2	10	90
Outside threshold	SarcP12	0	100
Outside threshold	JunK3	0	100
Outside threshold	DenJ7and4	0	100
Outside threshold	SpJu6and5	0	100
Outside threshold	Char8	0	100

Grey figures represent probabilities outside of observed transitions arbitrarily determined by the project team.
Prior vegetation: see Appendix B for descriptions.

The final node in the BBN was an overall risk of terrestrialisation. This node combines the two truly terrestrial vegetation communities (Terrestrial pasture grass (TerrP1), and *Leucopogon parviflorus* dominated shrubland (Leuc11), and therefore provides an overall probability value for each cell within the assessment area raster becoming of high suitability for either terrestrial vegetation community (Table 4.12).

Table 4.12 Conditional probability table showing the final assessment of risk of terrestrialisation

Terrestrial pasture grass (TerrP1) vegetation community suitability	*Leucopogon parviflorus* dominated shrubland (Leuc11) suitability	Risk of terrestrialisation (% probability)	
		High	Low
High	High	100	0
High	Low	95	5
Low	High	95	5
Low	Low	0	100

4.4.2.4 Evaluate model and conduct sensitivity analysis

Evaluation of the model was undertaken in two forms, expert opinion and sensitivity analysis. Expert opinion in the form of professional judgement from project team ecologists reviewed CPTs and outputs of the vegetation community suitability and terrestrialisation risk probabilities in relation to specific site knowledge of how the wetland had responded to water level declines in the past, and responses of other similar ecosystems in the region.

Sensitivity analysis was then performed on the populated BBN using Netica's entropy reduction sensitivity analysis (Norsys Software Corporation 1998). Sensitivity analysis identifies how sensitive a conclusion is to the evidence provided, which allows key nodes in the model which play a significant role in the outcome to be identified (Speldewinde 2013). The higher the value for a node, the more that node influences the outcome node. In an iterative process, the CPTs were revised for vegetation community suitability and hydrological niches. The final sensitivity analysis of the vegetation community suitability generally indicated of the key predictive variables, hydro-period (duration of inundation and hydro-niche) as the most influential factor, followed by the prior vegetation and land use. Land use most strongly influenced the suitability for Terrestrial pasture grass (TerrP1). Vegetation community hydro-niche ([VegCom]Niche) was consistently more important than duration of inundation (Duration), however relative importance varied considerably between vegetation communities. This reflected the relative probabilities of the hydro-niche thresholds determined for the vegetation communities. Also, for some communities, the prior vegetation (PriorVeg) is a greater indicator of the vegetation community being present than either niche or duration (e.g. JunK3, DenF9and10, and Gahn2). These communities had very small probabilities of transition to other vegetation communities (i.e. were most likely not to change) based on the transition state analysis. DenF9and10 (dense *Ficinia nodosa* community) and Gahn2 (*Gahnia filum* community) also have similar hydro-niche preferences, making the prior vegetation the more important variable (Table 4.13).

Table 4.13 Sensitivity analysis of vegetation community suitability to key predictive variables calculated using entropy reduction

Node	Entropy reduction (%)								
	TerrP1	Gahn2	JunK3	SpJu6and5	DenJ7and4	Char8	DenF9and10	Leuc11	SarcP12
[VegCom]Niche	45.4	15.8	7.69	53.6	23.3	43.2	10	48.1	41.2
Duration	33.2	5	3.5	23.9	7.9	31.6	4.01	39.3	26
PriorVeg	12.3	23	27.6	13.7	17.9	14.5	31.1	14.7	11.4
Landuse	10.76	8.64	2.15	5.26	2.16	3.46	2.39	2.24	1.34
Vegetation communities: see Appendix B for descriptions.									

The sensitivity analysis for the overall risk of terrestrialisation was as expected most influenced by hydrological drivers (duration of inundation and terrestrial vegetation community niche and suitability) (Table 4.14).

Table 4.14 Sensitivity analysis of risk of terrestrialisation to key predictive variables calculated using entropy reduction

Node	Entropy reduction (%)
TerrP1Suitability	60.7
TerrP1Niche	43.1
Duration	40.7
Leuc11Niche	28.1
Leuc11Suitability	21.8
PriorVeg	6.88
Landuse	2.64

4.4.3 Link predictive models to GIS

In this modelling study, the BBN was developed to model the risk of vegetation change and terrestrialisation at a scale corresponding to individual raster cells within the assessment area. In order to infer the level of risk over the entire study extent, the BBN was coupled to a GIS model of the site, from which was obtained the input values (variables of BBN nodes) for each cell. Raster surfaces comprising BBN model predictions for each modelled scenario were then linked back to GIS to create a spatial model output.

A number of studies have demonstrated the value of linking BBNs and GIS for mapping risk (Neville 2013) and habitat suitability (Smith et al. 2007). The general process adopted to link the outputs of the Middlepoint Swamp BBN, and map both vegetation community suitability and risk was adapted from the process outlined in Smith et al. (2007).

Develop case files for scenarios

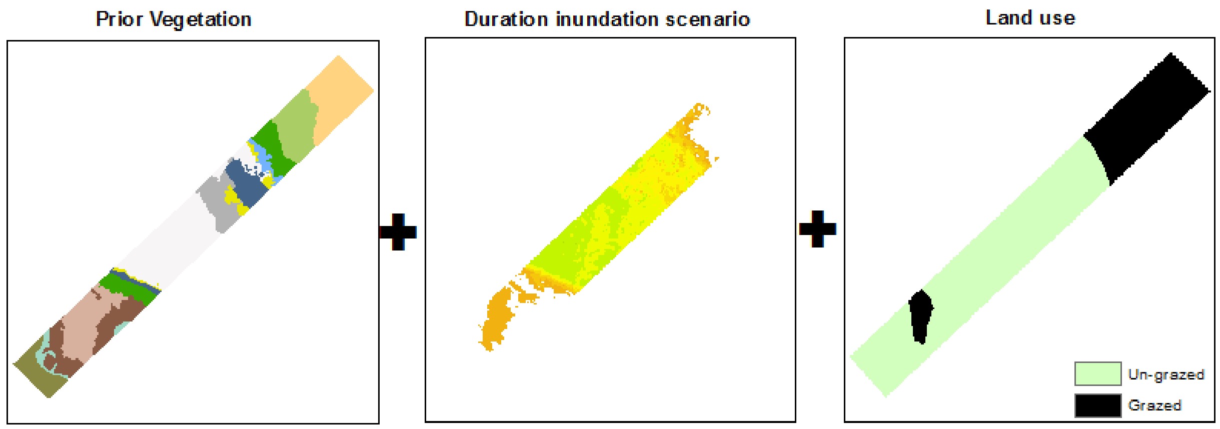
To produce the vegetation suitability and risk of terrestrialisation maps for all climate change scenarios, case files were required to be developed from the GIS layers forming the key predictive variable inputs (Figure 4.14). This involved overlaying prepared raster layers for the assessment area in ESRI ArcGIS® to produce four scenario case files: a 2013 case file, and three climate change scenario case files for 2030 (Table 4.15).

The ArcGIS Spatial Analyst raster Combine tool was used for this process. The resulting combined raster layers were then converted to point files and exported as MS Excel® files for use as case files in the BBN. Each case file contained a total of 23 596 rows (or points within the point layer), representing each 2 x 2 m pixel in the input raster datasets. Each row (point) contained the intersecting GIS variables from each of the input rasters in the assessment area, and an identification number that could be used to link the case files to spatial point data. The attribution of each variable was then edited to exactly match the names of nodes and states within the BBN, and saved as .csv files.

Table 4.15 Case files developed for the Middlepoint Swamp BBN, climate change scenarios based on GIS raster inputs

Case Files	GIS raster files		
	Prior vegetation community	**Mapped Duration of inundation**	**Land use***
2013 case file	1978 vegetation mapping	2013 inundation duration	Land use 2013
2030 (Scenario 1)^	2013 vegetation mapping	2030 (Scenario 1) inundation duration	Land use 2013
2030 (Scenario 2)^	2013 vegetation mapping	2030 (Scenario 2) inundation duration	Land use 2013
2030 (Scenario 3)^	2013 vegetation mapping	2030 (Scenario 3) inundation duration	Land use 2013

e.g.



* no apparent changes in land use were discernable from aerial photography between 1978 and 2013. No attempt was made to predict any change in land use for 2030.
^(Scenario 1) GCM ACCESS1.0 RCP4.5 (Best case, low emissions)
(Scenario 2) GCM ACCESS1.0 RCP8.5 (Best case, high emissions)
(Scenario 3) GCM ESM 2M RCP8.5 (Worst case, high emissions)

Incorporate mapping uncertainty into the model

An intermediate node to account for the uncertainty in the modelled hydrological outputs for the three climate change scenarios was factored into the BBN (refer to Figure 4.14). The CPTs were populated with an arbitrary 70% likelihood of the mapped 2030 projected inundation durations being correct was applied, with a 15% likelihood that the actual inundation duration could fall either side of the specified duration category.

For the 2013 case file, the duration of inundation GIS input was assumed to be 100% correct within the bounds of the duration categories, as the layer was produced from logged surface water data (not modelled) and LiDAR DEM data. Similarly, no uncertainty was applied to either the land use or vegetation mapping inputs.

Run case files and map highest probability of vegetation communities and risk

The prepared case files were then processed through the BBN with the relevant mapping uncertainty applied using the 'Incorp Case File', and then 'Process Cases' functions in Netica. The probability distribution for each vegetation community (High) suitability and the risk of terrestrialisation nodes were identified as the outputs for each case (point) and Netica control files were written to export the required findings.

The vegetation community with the highest probability value for 'High' suitability became the vegetation community for that cell within each of the case file scenarios, enabling the predicted vegetation responses for each of the climate change scenarios to be spatially mapped. The output files were linked back to GIS point files using the unique identifier numbers, and converted back into singular raster layers for each scenario. The maximum probability value was also spatially mapped for each predictive scenario, providing a mapped indication of the uncertainty of the modelled outputs.

Similarly, the overall risk of terrestrialisation probability of being 'High' was mapped for each of the three climate change scenarios. This spatially displayed the probability of the wetland ecosystem becoming a terrestrial ecosystem under the projected climate change scenarios.

Figure 4.12 depicts an example of the populated BBN, with posterior probabilities for key predictive variables set to a grazed land use, with a prior vegetation community of Terrestrial pasture grass (TerrP1), and a duration of inundation of 0 to 10 days (annually). As can be seen, the highest probability vegetation community remains TerrP1 and these conditions represent a 52% probability of raster cells meeting the key predictive variables being at high risk of terrestrialisation.

4.4.3.1 Verify model output with vegetation community mapping

Predictions from the BBN were validated using the same statistics as the GLM for hydrological niche space (Section 4.4.2.2 above) for 2013 (mapped versus predicted), with generally similar results. As with the GLM for hydrological niche, false negative rates were higher than false positives, indicating the model was unlikely to predict any vegetation community that was present when it was in fact absent, but could be prone to predicting absence, when a given vegetation community was present. Four vegetation communities however had excellent overall predictive performance, and as these included the highest and lowest positions on the hydrological gradient, providing some confidence in the model outputs. The overall model predictions for groups TerrP1, JunK3, Char8, Leuc11 and SarcP12 were of highest reliability (Table 4.16).

Overall validation statistics provided good confidence in the predictive ability of the model and identified the associations that were of lower reliability (e.g. SpJu6and5, DenF9and10) (Table 4.16). The dense *Ficinia nodosa* and *Ozothamnus ferrugineus* community (DenF9and10) was influenced in the BBN by the prior vegetation variable, which was highly uncertain, contributing to the lower reliability of prediction. The predominantly herbaceous vegetation communities of SpJu6and5 are likely to be highly responsive to change, are difficult to discern from aerial photography, and had a relatively low prevalence in the monitoring dataset. Hydro-periods for SpJu6and5 also overlapped strongly both up and down the hydro-period gradient, therefore providing a narrow predictive niche contributing to the low ability to model this community.

Table 4.16 Validation results for Middlepoint Swamp BBN. Selected statistics summarising BBN model predictive performance (from Fielding and Bell 1997)

Vegetation Community:	TerrP1	Gahn2	JunK3	DenJ7and4	SpJu6and5	Char8	DenF9and10	Leuc11	SarcP12
Sensitivity	1.00	0.60	0.84	0.59	0.24	0.99	0.50	1.00	0.99
Specificity	0.98	1.00	0.96	0.99	1.00	0.91	1.00	0.95	0.98
False positive rate	2%	0%	4%	1%	0%	9%	0%	5%	2%
False negative rate	0%	40%	16%	41%	**76%**	1%	**50%**	0%	1%
kappa	0.924	0.721	0.687	0.675	**0.368**	0.832	0.633	0.710	0.897
z =	19.32	10.58	11.53	9.00	4.66	25.90	10.05	11.30	16.91
probability (Z > z)	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00

Marginal values are shown in bold. Sensitivity - conditional probability of correct classification pr(model T|observed); specificity - inverse of sensitivity pr(model F|not observed); false positive rate – proportion of samples predicted present when not observed; false negative – proportion of samples predicted absent when present; kappa – is the proportion of specific agreement
Vegetation communities: see Appendix B for descriptions.

4.4.4 BBN model output - Vegetation response and risk

The predicted vegetation community response to current (2013) and the three climate change projections (to 2030) produced by the BBN model are presented in Figure 4.16, including probability expressed as model uncertainty. Figure 4.16 also provides a spatial comparison between actual (mapped) and predicted (BBN model) vegetation communities for 2013. A total of 80.35% of all raster cells in the assessment area were predicted correctly for 2013, with verification analysis indicating that the highest and lowest water requirement vegetation communities were well predicted, improving confidence in the modelled interpretation (Table 4.16).

Uncertainties in the outputs of the BBN model are however generally high, with uncertainty in the identity of predicted vegetation communities for projected climate change scenarios particularly so - the majority of the assessment area for all three projections was identified as high uncertainty (<50% probability of the predicted vegetation community occurring) (Figure 4.17). Two major factors contribute to output uncertainty: firstly a conservative approach was adopted in recognition of the known high levels of uncertainty in species-distribution models, which can exceed that attributable to global circulation models (Buisson et al. 2010). Moreover, the magnitude of change in surface water level (and therefore inundation duration) predicted under climate change scenarios was significantly higher than observed historic changes. Hence climatic projections represent a significant extrapolation beyond known conditions and are naturally associated with very high levels of uncertainty. The resulting response probabilities are low because no comparable precedent was observed in the data and time-frame used to identify niche thresholds and transition states which were used to populate the CPTs.

A spatio-temporal analysis similar to that presented in Section 4.3.2.1, in which the three predicted climate change scenarios were compared with the mapped current (2013) vegetation community extents was performed. Table 4.14 provides the resultant change matrix for each scenario, where values represent 2 x 2 m cells (pixels) in the assessment area. Extensive changes in vegetation communities are shown for each of the 2030 climate change scenarios, with a 67.6% total change for the best case (low emissions) scenario, and an 81.73% total change in the worst case (high emissions) scenario (Table 4.17).

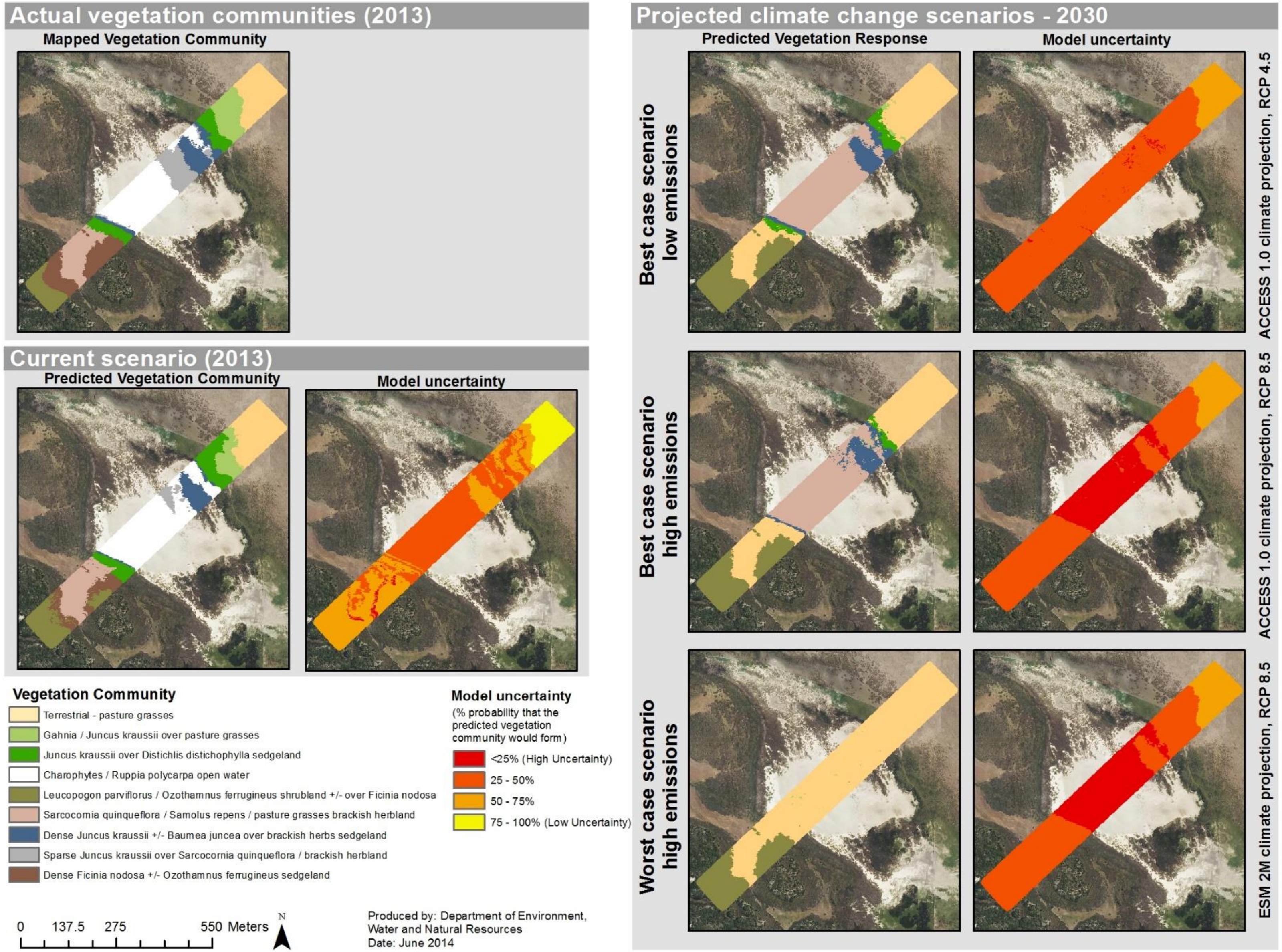


Figure 4.16 Predicted vegetation community response at Middlepoint Swamp to projected climate change scenarios

Table 4.17 Wetland vegetation transition matrix for Middlepoint Swamp transects, 2013 to 2030 predictions (number of cells*)

Scenario 1 ACCESS 1.0 RCP 4.5 Wetland Community	Total 2013	Change to 2030 (predicted)										
Change from 2013 (mapped)		Char8	SpJu6and5	DenJ7and4	JunK3	SarcP12	Gahn2	DenF9and10	Leuc11	TerrP1	Decrease	% of Total Change
Char8	5775	-	0	0	0	5774	1	0	0	0	5775	36.34
SpJu6and5	2058	0	-	1	0	2057	0	0	0	0	2058	12.95
DenJ7and4	1852	0	0	-	0	0	0	0	0	0	0	0
JunK3	1932	0	0	0	-	0	0	0	0	623	623	3.92
SarcP12	2651	0	0	0	0	-	0	0	0	2651	2651	16.68
Gahn2	2265	0	0	0	0	0	-	0	0	2115	2115	13.31
DenF9and10	2668	0	0	0	0	0	0	-	2668	0	2668	16.79
Leuc11	1188	0	0	0	0	0	0	0	-	0	0	0
TerrP1	3103	0	0	0	0	0	0	0	0	-	0	0
Total	**23492**										**15890**	**67.6**
Increase		0	0	1	0	7831	1	0	2668	5389		
% Change		0	0	0.006	0	49.28	0.006	0	16.79	33.91		
Net Change		-5775	-2058	1	-623	5180	-2114	-2668	2668	5389		

Scenario 2 ACCESS 1.0 RCP 8.5 Wetland Community	Total 2013	Change to 2030 (predicted)										
Change from 2013 (mapped)		Char8	SpJu6and5	DenJ7and4	JunK3	SarcP12	Gahn2	DenF9and10	Leuc11	TerrP1	Decrease	% of Total Change
Char8	5775	-	0	0	0	5721	50	0	0	4	5775	33.85
SpJu6and5	2058	0	-	200	0	1854	0	0	0	4	2058	12.06
DenJ7and4	1852	0	0	-	0	0	0	0	0	142	142	0.83
JunK3	1932	0	0	0	-	0	0	0	0	1498	1498	8.78
SarcP12	2651	0	0	0	0	-	0	0	0	2651	2651	15.54
Gahn2	2265	0	0	0	0	0	-	0	0	2265	2265	13.28
DenF9and10	2668	0	0	0	0	0	0	-	2668	0	2668	15.64
Leuc11	1188	0	0	0	0	0	0	0	-	0	0	0
TerrP1	3103	0	0	0	0	0	0	0	0	-	0	0
Total	**23492**										**17057**	**72.6**
Increase		0	0	200	0	7575	50	0	2668	6560		
% Change		0	0	1.17	0	44.41	0.29	0	15.64	38.46		
Net Change		-5775	-2058	58	-1498	4924	-2215	-2668	2668	6560		

Scenario 3 ESM 2M RCP 8.5 Wetland Community	Total 2013	Change to 2030 (predicted)										
Change from 2013 (mapped)		Char8	SpJu6and5	DenJ7and4	JunK3	SarcP12	Gahn2	DenF9and10	Leuc11	TerrP1	Decrease	% of Total Change
Char8	5775	-	0	0	0	0	30	0	0	5745	5775	30.07
SpJu6and5	2058	0	-	4	0	0	0	0	0	2054	2058	10.71
DenJ7and4	1852	0	0	-	0	0	0	0	0	1852	1852	9.64
JunK3	1932	0	0	0	-	0	0	0	0	1932	1932	10.06
SarcP12	2651	0	0	0	0	-	0	0	0	2651	2651	13.80
Gahn2	2265	0	0	0	0	0	-	0	0	2265	2265	11.79
DenF9and10	2668	0	0	0	0	0	0	-	2668	0	2668	13.89
Leuc11	1188	0	0	0	0	0	0	0	-	0	0	0
TerrP1	3103	0	0	0	0	0	0	0	0	-	0	0
Total	**23492**										**19201**	**81.73**
Increase		0	0	4	0	0	30	0	2668	10754		
% Change		0	0	0.02	0	0	0.16	0	13.89	56.00		
Net Change		-5775	-2058	-1848	-1932	-2651	-2235	-2668	2668	10754		

* raster cells representing 2 x 2 m pixel in the DEM. **Net change** = (increase – decrease). **Vegetation Communities:** see Appendix B for descriptions. **Orange:** indicates transition to a drier vegetation community; **Green:** no change; **Blue:** indicates transition to a wetter vegetation community; **Red:** indicates transition to a significantly drier vegetation community (transition state outside of previous observations for Middlepoint Swamp).

In all three scenarios, the majority of change involved vegetation communities transitioning to much drier communities, outside the range of previously observed vegetation transition (indicated as red shading in Table 4.15). The highest water requirement vegetation communities (charophytes / *Ruppia polycarpa* open water, and sparse *Juncus kraussii* over *Sarcocornia quinqueflora* dominated herbland (Char8 and SpJu6and5)) are no longer predicted to occur in any of the 2030 scenarios, being replaced almost wholly by *Sarcocornia quinqueflora* dominated brackish herbland with pasture grasses (SarcP12) in Scenarios 1 and 2, and terrestrial pasture grasses (TerrP1) in Scenario 3. This change accounted for 36.34% and 33.85% of the total change for climate change Scenarios 1 and 2, and 56% of Scenario 3 (Table 4.17). Terrestrial pasture grasses (TerrP1) was predicted to significantly encroach, particularly in areas grazed by stock, in both Scenarios 1 and 2, with an increase of 33.91% and 38.46% respectively. The predicted vegetation communities for Scenario 3 (worst case, high emissions) are reduced almost entirely to terrestrial ecosystems (*Leucopogon parviflorus* dominated shrubland (Leuc11) and terrestrial pastures grasses (TerrP1)). The existence of grazing appears to restrict the encroachment of Leuc11, where terrestrialisation favours the encroachment of pasture grasses instead.

However, given the unprecedented magnitude of water level change, the model potentially provides an indication of the ideal niche suitability for each vegetation community, and significant time lags may apply for vegetation to respond to the rapid change in water levels projected and transitions to occur. Some vegetation communities present may persist for significant periods of time outside of their optimum hydrological threshold as has been observed elsewhere in the South East region that has been subject to drainage induced hydrological change (e.g. *Ficinia nodosa, Gahnia filum, Juncus kraussii*). Therefore the level of change indicated by the BBN modelling outputs may be indicative of the direction of change (or the eventual result of steady state conditions), although overestimates the actual change expected within the modelling time period.

The final risk of terrestrialisation combines both the probabilities of the two terrestrial indicator communities: pasture grass (TerrP1) and *Lecuopogon parviflorus* dominated shrubland (Leuc11) (Figure 4.17). Whist the predicted vegetation community responses depict large scale habitat changes (Figure 4.16), the overall risk of the ecosystem becoming an entirely terrestrial ecosystem by 2030 is mostly moderate, and low within the deeper areas of the wetland basin. Existing terrestrial vegetation community extents were the only areas to be shown as high risk (ie. high probability of remaining terrestrial ecosystems). Of the modelled extent, if all areas identified as moderate risk that were not already terrestrial ecosystems in 2013 were to transition to terrestrial ecosystems, a proportional loss of wetland ecosystem of 47%, 55%, and 70% respectively for each of the 2030 climate change scenarios would result.

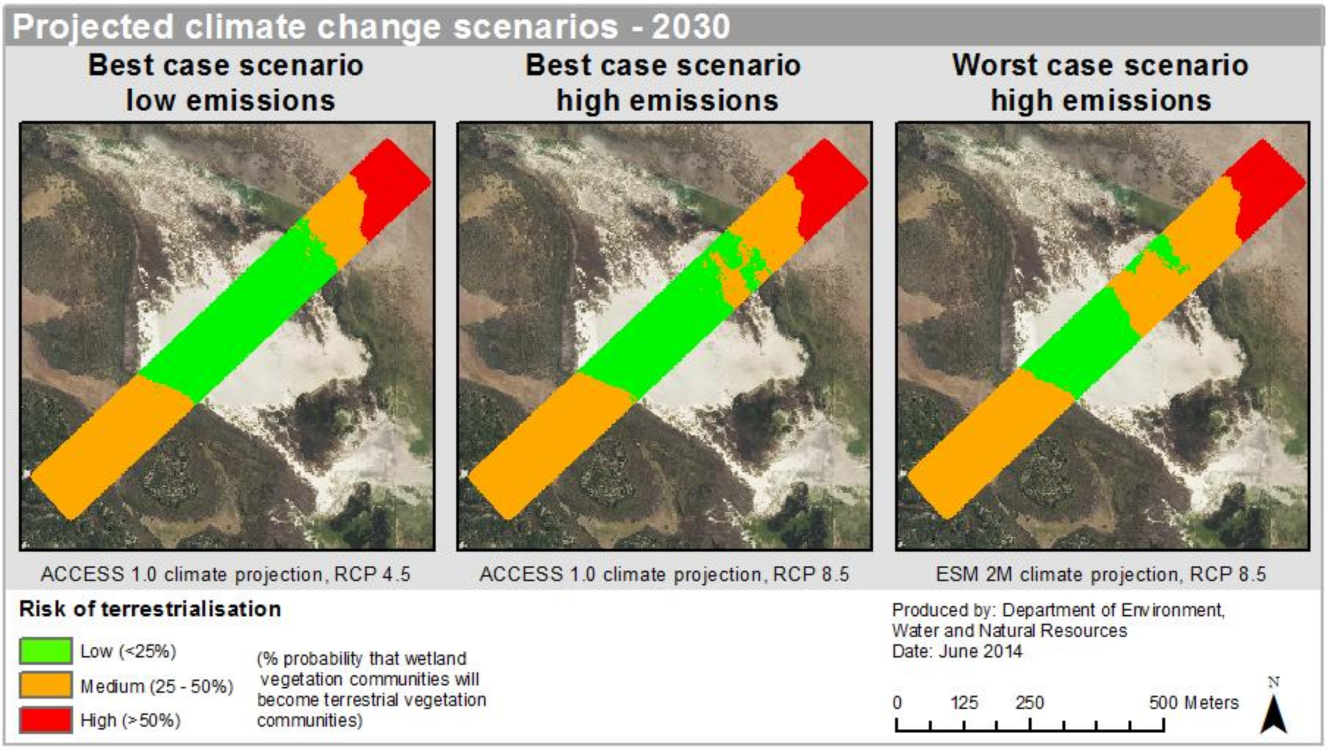


Figure 4.17 Predicted risk of terrestrialisation of Middlepoint Swamp under projected climate change scenarios

5 PART 3: Risk treatment

The Middlepoint Swamp case study illustrates the observed effects of groundwater level decline, and by extension and modelled scenarios, predicted impacts of additional climate induced groundwater decline on a coastal wetland GDE in the Lower Limestone Coast WAP area. By determining relationships and translating data through a conceptual model into a BBN model, and displaying the outputs spatially, the potential impacts of groundwater level decline caused by climate change on wetland GDEs can be better envisioned and communicated.

There are a number of measures already in place in the South East NRM region to manage the regional unconfined Tertiary Limestone Aquifer (TLA), principally Water Allocation Plans (WAP) developed pursuant of the *Natural Resources Management Act* (2004). The objectives of the Lower Limestone Coast (LLC) WAP is to protect the underground water resources to ensure its ongoing availability to sustain ongoing economic, social and environmental systems, while providing flexibility and equity of access to water. WAPs set out the principles for allocation, use and transfer of groundwater in each Prescribed Wells Area (PWA). As part of the planning process, an assessment was made of the needs of water dependent ecosystems which informed specific management policies in the WAP for high value ecosystems. The LLC WAP (SENRMB 2013) PWA, in which Middlepoint Swamp is located, provides principles for groundwater management that aim to maintain the current quantity and quality of groundwater available to GDEs by:

- managing groundwater salinity and levels within limits of acceptable risk (less than -0.05 m groundwater level decline per year measured over the previous five years for high value GDEs, and 0.1m/year elsewhere) to attempt to retain GDE connection to the TLA. This is achieved broadly by managing licensing of further groundwater extraction from the TLA based on existing groundwater conditions and by ensuring the total volume of extraction does not exceed 1.25 times the amount of annual average vertical recharge. The WAP also sets out principles for the reduction of allocations so as to not exceed a Target Management Level (TML) determined for each groundwater management area.
- applying minimum set-back distances from identified high value GDEs for all new groundwater extraction and forestry. Set-back distances are determined by an equation (known as the Dependent Ecosystem equation, developed by REM 2006) which takes into account the distance between the proposed point of taking and the GDE, the volume of water proposed to be extracted, and the local aquifer characteristics.

The modelling of groundwater response to reduced rainfall as a result of projected climate change scenarios, and response of surface water and ecosystems (i.e. this project and report) has identified a number of risks not explicitly incorporated into the current management policies for the unconfined groundwater resources (TLA) of the SE. These risks are further discussed in the following sections (Section 5.1 to 5.3).

5.1 Groundwater management

The declines in groundwater level projected as a result of climate change in this case study are subject to numerous limitations and assumptions (refer to Section 4.2.1), however the level of impact projected at a single observation well for this study provides an indication of the potentially significant effect reduced rainfall may have on the unconfined groundwater of the TLA. Further modelling of future impacts across the South East would therefore be warranted in order to inform adaptive management options. Examples of managing declining groundwater levels as a result of climate change and increased extraction has been well documented and demonstrated in Western Australia (Gnangara Sustainability Strategy (GSS) Taskforce 2009; McFarlane et al. 2012). Many of the recommendations of the GSS for managing the environment under a drying climate are potentially applicable in the South East NRM region. Some of the key proposed management options identified through the application of the risk assessment framework and with reference to the GSS Taskforce (2009) included:

- The requirement for the effects of climate change to be adequately addressed within future Water Allocation Plans. The National Water Initiative (NWC 2011) recommends that water laws define entitlements as a share of a variable pool that is determined by rainfall and groundwater recharge. By ensuring recharge rates defined by the WAPs for the purpose of determining TMLs are reflective of potential impacts of climate change, and resource allocations are adaptive and responsive to a changing climate, groundwater could potentially continue to be managed within sustainable (and ecologically relevant) levels.

- An adaptive management approach to the monitoring of the environmental impacts of groundwater decline be developed, that can separate the role of climate, and anthropogenic extraction/land use. This would include long-term monitoring of indicator species and/or vegetation community change and site hydrology (both groundwater and surface water) to detect ecosystem change, confirm conceptual understandings of the impacts of groundwater decline, and inform frequent review of management actions.
- Maintaining hydrogeologically and ecologically relevant set-back distances for new extraction and forestry to high value GDEs, and monitoring the effectiveness of current dependent ecosystem protection WAP policy.
- Improved knowledge of specific groundwater and surface water interactions and dynamics for GDEs in the South East region could provide ecologically significant groundwater management levels (environmental water requirements) for high priority/value ecosystems.

In the case of Middlepoint Swamp, thresholds of depth and duration of existing vegetation communities could be translated into maximum groundwater drawdown level management thresholds. Using the information presented in this case study, the EWRs for Middlepoint Swamp could be maintained by the current hydrological regime (the period between approximately 1992–2013), where groundwater is required to reach a seasonal maximum of approximately 2.5 – 3.2 m AHD, with a seasonal minimum of approximately 1.2 – 2 m AHD (measured at MAC090 – refer to Figure 4.4). The relationships developed for modelling the impact of groundwater levels on surface water levels could then be used to determine the acceptable limits of change, and the BBN used to model the likely ecosystem response. Similar approaches to managing EWRs for GDEs have been applied in Western Australia (Department of Water 2008; 2012; Hyde 2006).

Under the current policy (LLC WAP), groundwater can decline 0.05 m per year (measured over the previous five years) before triggering buffer protection policy, unless the wetland is listed as a 'priority wetland complex' (SENRMB 2013). Whilst a drop in groundwater level of a maximum of 0.25 m over five years would likely have a minor impact on the Middlepoint Swamp ecosystem, a shallower wetland, with less permanent groundwater discharge, could be significantly impacted. Additionally, sustained and unabated declining groundwater levels will ultimately result in significant impact to dependent ecosystems. Note that Middlepoint Swamp is part of the Lower South East Rising Springs West Complex, identified as a 'priority wetland complex' in the LLC WAP and is therefore protected from additional groundwater extraction by the buffer policy regardless of groundwater level decline triggers.

5.2 Landscape scale ecosystem responses

The Middlepoint Swamp case study identified terrestrialisation (transition of wetlands to terrestrial ecosystems) as a significant risk of climate change. The modelling indicated that Middlepoint Swamp was largely (between 47–70% of the study extent) at moderate risk (25–50% probability) of terrestrialisation by 2030 under the three climate change scenarios assessed (refer to Figure 4.17), but indicated large scale vegetation community changes towards more dry adapted species in response to changed hydrology (Figure 4.16) over the same time period. Given that Middlepoint Swamp is a relatively permanent coastal discharge site, of greater than 1 m maximum water depth, it could be assumed that impacts on shallower GDEs with seasonal TLA interactions could be far more severe. As a result, landscape scale management of some types of wetland ecosystems transitioning to terrestrial ecosystems may be inevitable despite land and water management interventions. In managing this risk, options included:

- Determining the relative risk of a range of wetlands to terrestrialisation (using similar approaches applied in this case study) as a result of climate change projections. Where wetlands are predicted to dry out (indefinitely) despite land and water management interventions, management could focus on transition to a terrestrial ecosystem. The Middlepoint Swamp example indicated the likelihood of terrestrial pasture grasses (exotics) preferentially transitioning into drying wetlands as hydrology changes. This could have a significant impact on the biodiversity values of the South East region (a large percentage of which is associated with aquatic ecosystems) and also on the future prioritisation of biodiversity management investment (restoration and threatened species management / re-introduction / pest control activities).

- Prioritising the retention of blocks of extant remnant vegetation (with wetland matrices intact). This would include protection from clearing, further fragmentation and multiple threats (agricultural pollutants, feral animals, weed invasion, fire, groundwater level decline) to strengthen the ecological resilience of remaining relatively intact systems. Enhancing ecological linkages between existing priority blocks of remnant vegetation, may also increase the resilience of ecosystems to climate change impacts.
- Actively managing surface water hydrology via the existing South East drainage network has been (and is being) investigated as a measure to mitigate the impacts of climate change for priority wetlands (Denny et al. 2014). The use of the South East drainage network and infrastructure for delivering/augmenting environmental water (where technically feasible) to priority ecosystems may become a significant mitigation measure which would require site based assessment in order to consider the impacts of diverting water (on downstream wetlands, and on the receiving ecology), and landscape scale prioritisation. The placement of weirs in drainage infrastructure to seasonally raise local groundwater levels has been applied in an adaptive management framework in the Upper South East (DFW 2011). Also, a weir structure has been installed in the outlet of Middlepoint Swamp, which functions to raise the water level within the wetland, increasing the depth and duration of inundation, and also locally raising groundwater levels in the TLA.

5.3 Site-based remediation

Site specific remediation of GDEs to the risks posed by climate change, outside of management of the regional groundwater resources (discussed above), include changes in both on-site land use practices, and technical solutions to reduce impacts or emergency drought response where feasible.

5.3.1 Land use practices

On-site land use practices to mitigate impacts on GDEs include establishment of buffer zones (vegetation buffers) between actively used agricultural land and vulnerable GDEs in order to reduce grazing pressure and associated exotic pasture grass invasion, and pollution from runoff/spray drift/stock (Kløve et al. 2014), which compounds the direct hydrological threats posed by climate change. Wetlands with no buffering native vegetation within an agricultural landscape, are vulnerable to weed invasion as a result of drying conditions, and associated increases in stock grazing. Management of encroachment of native terrestrial shrubs and trees into high value aquatic ecosystems has been suggested by Dickson et al. (2014) for maintaining the characteristics of Seasonal Herbaceous Wetlands (an EPBC Act listed threatened community). The encroachment of terrestrial vegetation (exotic or native) is a symptom of drying conditions, and the efficacy of management actions to remediate encroachment will depend on the relative risk posed by climate change, and the ability to re-instate or manage the hydrology of wetland systems in the long-term. Additionally, Catford et al. (2014) proposes that rather than allowing communities to self-assemble following hydrological change, managers could augment the propagule supply of native species that possess characteristics suitable under new hydrological conditions.

The Middlepoint Swamp case study indicated the preferential encroachment of terrestrial (exotic) pasture grasses under drying conditions in the presence of grazing. Restricted grazing, and the establishment of native vegetation buffers and potential seeding of native vegetation more suited to drier conditions, may lessen the impacts of climate change on Middlepoint Swamp existing vegetation communities and mitigate the potential for transition of the wetland into a largely exotic pasture dominated community.

5.3.2 Technical solutions

On-site technical solutions generally include local placement of drainage infrastructure (weirs installed in outlet drains), to both increase surface water depth and duration, and raise groundwater levels, or directly increasing inflow by pumping. Pumping groundwater into GDEs as remediation is generally not feasible, given that the pumping itself may further groundwater level decline, and depending on the presence and properties of confining clay layers at the base of the GDE, may be unsuccessful (where pumped water returns directly as recharge to the unconfined aquifer). Additionally, it is largely impractical and inefficient to pump significant amounts of groundwater into large wetland ecosystems such as Middlepoint Swamp. The use of pumping has however been implemented to sustain GDE permanent pool habitat in Mosquito Creek (South East, SA) to support source populations of threatened native fish (Yarra Pygmy Perch) during drought conditions (A. Goodman (DEWNR)

2014 pers. comm., 10 June). Similar emergency augmentation, or relocation of populations to more secure habitat, to support the source populations of threatened aquatic species may be required to be considered elsewhere in the South East region as an adaptation measure to climate change.

In the case of Middlepoint Swamp, continued management of water levels within the wetland with use of the weir structure installed in the outlet drain in 2011 (S. Clarke (DEWNR) 2014, pers. comm., 27 March), and potential investigation into other hydrological manipulation of existing drainage infrastructure are likely the most significant management actions that can be applied on-site as remediation.

6 Discussion and conclusion

The application of the risk assessment framework developed by Chambers et al. (2013a) to the hazard of declining groundwater levels at a wetland GDE case study site in the South East NRM region of South Australia, in conjunction with the testing of the framework by Chambers et al. (2013b), provided a robust testing field of various GDE ecosystem types, scales of assessment and data availability. A major strength of the framework was its capacity to relate climate, hydrology and ecosystem response in a single tool (a Bayesian Belief Network). The Middlepoint Swamp BBN used modelling techniques outside of the BBN to relate climate and hydrology, and the BBN was limited to modelling ecosystem response from hydrological change. However, the Middlepoint Swamp model could be adapted to include climate and groundwater nodes, as well as water quality, reflective of the conceptual model developed (with further work and analysis).

The use of the BBN approach to assessing risks to GDEs from climate change also allowed simple modification and the ability to run a variety of scenarios through a single model. This enabled changes to be shown in terms of probability of risk resulting from the interaction between hydrology and biotic response. The BBN structure also allows for a wide range of data sources (from expert opinion to empirical data) to be integrated in a relatively transparent and interactive way. The outputs of which are valuable decision-support tools, being able to assess the impacts of a range of actual, predicted, or theoretical scenarios. As such, it is possible for the BBN to be used to test hypotheses and conceptualisations of relationships between drivers and biotic responses and also in identifying the critical factors influencing ecosystem response based on available knowledge and data (Smith et al. 2007). Similarly, the development of BBNs could be used as tools for determining thresholds of acceptable change (response) to ecosystems, and therefore optimal EWRs to inform water management policy.

In regard to the overall risk assessment framework (refer to Figure 1.2 and Table 1.1), the initial development of conceptual models and diagrams of relationships between identified climate and hydrological drivers at a case study site scale was valuable in itself as a useful communication tool for management and policy development purposes. The conceptual models assist in simplifying potentially complex (and unknown) interactions between drivers that are not well understood. In converting the conceptual model to a BBN, the conceptual models can be tested, and spatial representations of the outputs provide powerful imagery reflecting modelled response and risk, in conjunction with modelling uncertainty. The overall success of utilising the framework for the Middlepoint Swamp case study site, and particularly the BBN approach, may in turn provide impetus for further site-scale risk assessment at a range of GDEs in the South East.

As with all modelling approaches, and climate change modelling in particular, one of the major limitations was the accuracy of the model predictions, with low output probability values and limitations of the model as determined in the verification process. The accuracy and robustness of the outputs are reflective of the credence of the input data. While the properties of BBNs allow a reasonably robust delivery of probability of risk, whether using expert opinion or detailed verified models supported by extensive datasets, the inherent limitation of the inputs must always be considered (Chambers et al. 2013b). Primarily, projecting surface water levels from groundwater levels into the future with predicted rainfall from Global Climate Models (GCM) incorporates accumulated errors inherent in modelling approaches and limitations of these data. For example, downscaling GCMs to local areas, determining groundwater response to changes in rainfall outside of historic ranges, and determining groundwater – surface water interactions all involve significant levels of uncertainty. Additionally, each dataset may have insufficient resolution to provide meaningful estimations of projected groundwater levels at a site-scale, and most projections deal with a mean of conditions, when in reality it is the extreme events that are likely to have the greatest impact (Chambers et al. 2013). As a result, the outputs of the hydrological and ecological response modelling presented in this study should be used with caution (as represented in output uncertainties), indicating a likely direction of change rather than providing exact groundwater/surface water levels and ecological response into the future.

The output of the risk assessment framework provides a probability of risk, not an actual outcome. Outputs, particularly spatial mapping of response and risk, should therefore be appropriately interpreted so as not to be misconstrued and inadvertently used in applications (policy or otherwise) outside of their limitations. A strength of the use of BBNs in this framework was the transparency of the networks and the ability to also spatially map uncertainty to assist in interpretation. Potentially the largest limitation of implementing the framework, however, was the resources required (principally time and expertise, but also data) to develop a model for a single case study site. It is likely that to apply similar methods to other case study sites, many of the methods and outputs of the Middlepoint Swamp example could be re-utilised, making future application potentially less resource intensive. A restriction of this study was a very limited time-frame for development and testing, and a small project team. The application of the framework in this study was therefore a trial of methods, and more specifically a trial of the use of existing monitoring datasets for predictive modelling in a way that could be used to inform future policy and planning in relation to climate change. Overall, the framework has been demonstrated to be adaptable and flexible enough to be applied to other types of GDEs, and importantly, other types of datasets and scales of assessment, as intended by Chambers et al. (2013a).

6.1 Gaps and future research directions

The application of any framework to predictively model and assess risk from climate change is reliant on both sufficient monitoring data and ecological expert knowledge. Significantly both of these elements are becoming scarcer (Lindenmayer et al. 2014), contributing to the inability of managers to adequately consider and represent the environment in policy (Lindenmayer et al. 2014). Lindenmayer et al. (2014) identifies that the reasons for ongoing failure of policy to reduce the rate and scale of environmental degradation are:

1. a lack of appropriately designed programs that allow critical examination of the on-ground effectiveness of management and policy actions
2. a lack of monitoring to gather the necessary data to inform management and policy adaptation.

By improving these two aspects, and maintaining ongoing long-term monitoring of key ecosystems, such as that implemented in the GDE monitoring program (SKM 2010; Beacon Ecological 2010) in the South East, appropriate information to inform analysis (such as that presented in this study) can be produced to support future decision-making and policy, and to improve confidence in modelled outputs.

Ongoing monitoring including combined groundwater, surface water and biota (vegetation and key indicator species) at representative GDEs would therefore appear critical to:

- Minimising uncertainty regarding groundwater and surface water interactions at GDEs
- Understanding time-specific ecological responses to known changes in hydrology
- Understanding relationships with climatic variables.

All of which form the basis for producing conceptual understandings of GDEs, and predictive models.

Ongoing resourcing of the existing GDE monitoring network in the South East was therefore a key recommendation of this study, as is the application of the risk assessment framework (or similar predictive modelling) to a range of GDE types in the South East region, in order to support decisions on future prioritisation of resources, adaptive management and planning. This study identified 14 short-listed GDEs in the South East that were potential candidates for future climate change modelling (refer to Table 2.1), however a similar approach could be used to assess risks to individual threatened aquatic species and habitat security under climate change scenarios.

Additionally, as shown by this study, the impacts of reduced rainfall on groundwater levels in the South East may be significant. Simplistic analytical modelling was used at a single observation well for the Middlepoint Swamp case study site, with significant uncertainty. Given the importance of the groundwater resources, both economically and environmentally, to the South East region, further modelling of the impacts of climate change on groundwater resources appears warranted.

7 Appendices

A. A preliminary assessment of the hazard of sea level rise to Middlepoint Swamp – an example application of the Bruun Rule

Background

By 2100 sea level is projected to rise 0.56–2.0 m or ~0.8 m (updated IPCC 4 projections). Rignot (2011) projected a rise of 32 cm by 2050. In addition to higher projected storm surge and oceanic inundation levels, a rise in the mean sea level will also result in landward recession of unconsolidated (sandy) shorelines. Saltwater intrusion and landward advance of tidal limits within estuaries will have significant implications for freshwater and saltwater ecosystems and development margins. Sea level rise will also influence entrance opening regimes for intermittently closed and open lakes and lagoons.

Mustafa et al. (2012) completed a preliminary analysis into the impact of sea level rise in South East South Australia to identify ecosystems potentially vulnerable to the combined effects of sea level rise (SLR) and salt water intrusion (SWI). An elevation of 3.5 m AHD was selected as a "cut off height" for potential SLR/SWI impacts to WDEs along the Lower Limestone Coast. This height reflected the Climate Commission's (2010) maximum projected sea level rise of 1 m by 2100 and the maximum high tide observation of 1.535 m (based on 1981 record for Victor Harbour taken from the National Tidal Facility). A conservative safety factor of 1 m was added to account for storm surges/waves and erroneously high elevations in the 10m Digital Elevation Model (DEM) resulting from vegetation interference with the LiDAR (accuracy ± 15 cm). The analysis of spatial data to inform SLR/SWI potential was performed using GIS. All areas in the Lower Limestone Coast Prescribed Wells Area (PWA) with 3.5 m AHD elevation or less were extracted from the 10 m DEM to create a raster layer. The raster layer was converted to polygons and intersected with the WDE Polygon Layer. This resulted in the identification of 487 wetland polygons within the PWA that have basin elevations of 3.5 m AHD or less. Fotheringham & Rutherford (2013) applied similar methods using the regional DEM to project future sea level rise scenarios for the South East coast.

These studies (Mustafa et al. 2012; Fotheringham & Rutherford 2013) represent an initial attempt to incorporate SLR/SWI into climate change risk assessments on at a regional scale. However sea level rise will also result in the recession of unconsolidated shorelines. Coastal recession is also likely to be an important factor for consideration in risk assessment for the South East region due to the flat terrain and number of near – coast wetlands, and artificial estuarine outlet drains. The Bruun Rule (Bruun 1962) was identified as a non-data intense analytical modelling method by which coastal recession may be estimated and applied at a case study site scale. Cooper and Pilkey (2004), however dispute the usefulness of the Bruun Rule, detailing that several assumptions behind the Bruun Rule are known to be false and nowhere has the Bruun Rule been adequately proven. Despite this, no universally applicable model of shoreline retreat under sea-level rise has yet been developed, and the Bruun Rule remains widespread (perhaps erroneously) in its use at a global scale both as a management tool and as a scientific concept. It is acknowledged that the outcomes of applying the Bruun Rule analytical model are likely to be inherently flawed, and therefore would serve as low confidence assessment only.

The Middlepoint Swamp GDE case study site, identified as a wetland at risk of sea level rise by Mustafa et al. (2012), was used as a case study in the use of the Bruun Rule to assess potential risks of sea level rise for coastal wetlands in the South East region.

Bruun Rule overview

The 'Bruun Rule' is a simple two dimensional analytical model which can be used as a broad approximation for determining coastal dune recession. Recession due to sea level rise can be estimated simply as the product of the sea level rise (over the planning timeframe of interest) multiplied by the inverse of the active profile slope (DECCW 2010).

This simple model states that the beach profile is a parabolic function whose parameters are entirely determined by the mean water level and the sand grain size. Bruun (1962) states that within the closure zone of the beach (typically the limit of significant wave-driven sediment transport), the beach will adjust to maintain its equilibrium profile relative to the still water level. This is achieved by translating the profile landwards and upwards, with eroded sediments at the landward end of the profile being deposited in the lower portion of the profile, and raising the bed, maintaining a net sediment balance (Figure 1).

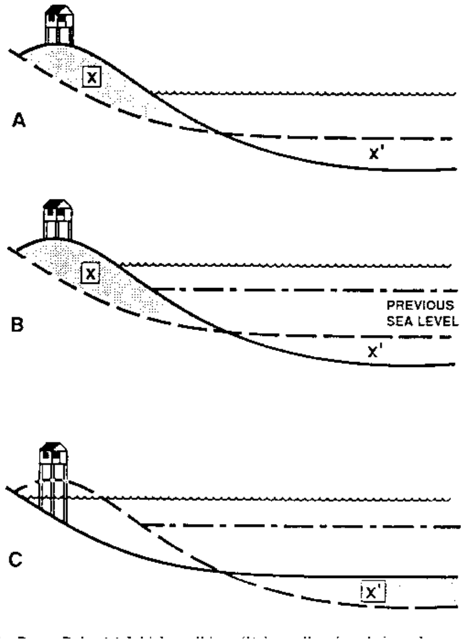


Figure 1. Diagram showing sea level rise and coastal recession impacts (Source: http://papers.risingsea.net/Holding/fig4.html)

Shoreline recession (*R*) predicted by the Bruun Rule is given by: $R = S\left(\frac{L}{B+h}\right)$ (Eq. 1)

where *S* (metres) is the predicted sea level rise scenario, *L* is the length of the profile (distance from the top of the dune berm to the distance at which the sea bed reaches equilibrium), *B* is the berm height, and *h* is the closure depth.

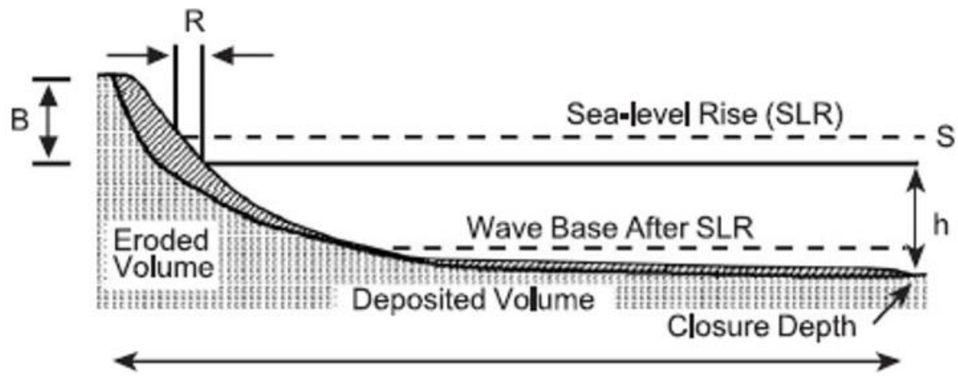


Figure 2. Bruun Rule diagram for shoreline response to sea level rise

Limitations and assumptions

There are many limitations in using the *Bruun Rule* for determining foreshore recession due to sea level rise. More complex three-dimensional models enable consideration of a broader suite of natural processes and physical attributes on a site-specific basis. The rule does not account for long shore interactions, and secondly, the rule assumes the wave climate is steady and hence the equilibrium profile remains the same - simply translated landwards and upwards with the rise in mean sea level. Such limitations should be considered when the Bruun rule is applied (ACE CRC 2008).

The extent of recession calculated with the rule has not been successfully validated (Cooper & Pilkey 2004), and is considered at best an 'order of magnitude' estimate.

The limitations of the Bruun Rule as summarised from Ranasinghe *et al.* (2007) are outlined below.

- The Bruun Rule does not include three-dimensional variability, as it assumes two-dimensional (cross-shore) sediment movement only, therefore, the rule does not include alongshore gradients in longshore transport (such as a regional transport rate); alongshore features or structures such as headlands, engineering structures and nearshore reefs that control the shoreline shape due to their impact upon sediment transport; or estuaries/inlets which may act as both source and sink for sediments in the nearshore zone.
- The Bruun Rule is only applicable on 'equilibrium' beach profiles, that is, it is not applicable at beaches where there is ongoing profile change (for example, the profile is still evolving to the most recent rise/fall in sea level, or change in sediment supply)
- The Bruun Rule assumes there is no sediment movement (such as offshore sediment loss) seaward of the depth of closure
- The Bruun Rule does not allow for a majority of fine sediments in the dune, which when eroded would be too fine to deposit and remain in the nearshore, and it does not allow for variations in sediment between the nearshore, beach berm and dune.

A fundamental assumption of this rule is that over time the cross-shore shape of the beach, or beach profile, assumes an equilibrium shape that translates upward and landward as sea level rises. Four additional assumptions of this model are that:

1) The upper beach is eroded due to landward translation of the profile.
2) The material eroded from the upper beach is transported offshore and deposited such that the volume eroded from the upper beach equals the volume deposited seaward of the shoreline.
3) The rise in the nearshore seabed as a result of deposition is equal to the rise in sea level, maintaining a constant water depth.
4) Gradients in longshore transport are negligible.
5) The rule has also only been tested on flat coastal areas.

Application of the Bruun Rule at Middlepoint Swamp case study site

Method

The Bruun Rule equation was applied at three cross-sections of the shore/dune interface along the coastline of Middlepoint Swamp. The transects were chosen to reflect the varying dune berm heights. LiDAR 2 m DEM was used to obtain the berm-height data, and profile length and closure depth were estimated from visual changes in the sea floor from aerial photography and the use of local knowledge of depth off-shore from local crayboat operators and recreational divers.

The scenarios tested incorporate projected sea level rise estimates for 2050 and 2100. The estimates that are used are 0.3 m sea level rise by 2050 and 1 m rise by 2100, which are the projected estimates adopted by the SA government for the purposes of planning (Fotheringham & Rutherford 2013). Each of the two projected sea level rise estimates were combined with a normal tidal water height (Mean High Water Spring (MHWS)), and also an estimated level reached by a 1 in 100 year average recurrence interval (ARI) storm. Tide data were sourced from Fotheringham & Rutherford (2013), which was derived from the Tide Tables for South Australian Ports maintained by DEWNR coastal staff. The estimates were prepared based on analysis of tidal records at a nearby location (Port MacDonnell) where sufficient historical tidal data existed. As per Fotheringham & Rutherford (2013), tides for MHWS and 1 in 100 year ARI storm values for Port MacDonnell were determined as 0.3 m and 1.3 m respectively.

Five sea level rise scenarios were tested (Table 1). Scenarios 1 and 2 represent sea levels that would occur at high tides for the two sea level rise projections for 2050 and 2100 respectively. Scenarios 3 and 4 represent extreme 1 in 100 year ARI event water levels with the two sea level rise projections. Scenario 5 adds an additional 1m buffer to the worst case scenario (1 in 100 year ARI for 2100) to account for potential inaccuracies in projections and data used, although is not indicative of any projection currently available. A similar approach was taken by Mustafa et al. (2012).

Table 1. Sea level rise scenarios tested with the Bruun Rule using tide data from Port MacDonnell

Scenario no.	Description	Resulting sea level rise scenario (m AHD)	Indicative time period
1	MHWS(0.3) + 0.3 m sea level rise	0.6	2050
2	MHWS(0.3) + 1 m sea level rise (2100)	1.3	2100
3	1 in 100 ARI storm(1.3) + 0.3 m sea level rise	1.7	2050
4	1 in 100 ARI storm (1.3) + 1 m sea level rise	2.4	2100
5	1 in 100 ARI storm(1.3) + 1 m sea level rise + 1m buffer	3.4	N/A

Three cross-sections were chosen along the Middlepoint Swamp coastline, at points representative of varying dune heights to take into account variable coastline conditions. The length of the cross-sections were determined by the profile length as stated by the Bruun Rule (distance from the top of the dune berm to the distance at which the sea bed reaches equilibrium) with use of the 2 m LiDAR DEM and local knowledge of coastal conditions.

The resulting shoreline retreat estimates for each cross-section and scenario were plotted using ArcGIS to show potential dune retreat estimates for the entire stretch of coastline adjacent to Middlepoint Swamp.

The 2 m LiDAR DEM was also used to map the extent of potential inundation based solely on the height of the predicted sea level rise scenarios similar to that produced by Fotheringham and Rutherford (2013).

Results

Shoreline retreat was calculated using the Bruun Rule (Eq. 1) for each of the three transects, and for all four scenarios (Table 2). Using a baseline of the current shoreline (determined as 0m AHD), the shoreline retreat measurements were displayed using buffer tools in ESRI ArcGIS® (Figure 3).

Based on the calculations, a 2050 projection of an increase in sea water level of 0.3m would result in coastal dune retreat between 18 to 28 m at MHWS, and 51 to 81 m in a 1 in 100 year ARI storm surge (Scenarios 1 and 3). Along with analysis of the LiDAR DEM (Figure 3), this level of rise is unlikely to directly impact (by way of sea water inundation) Middlepoint Swamp, however may have an influence on the outlet drain, potentially allowing storm surge sea waters into the wetland basin.

The 2100 sea level rise projections produce more concerning results, indicating an increase of 1.3 m in sea level would result in coastal dune retreat between 39 to 62 m at MHWS, and 73 to 114 m in a 1 in 100 year ARI storm surge (Scenarios 2 and 4). Under these scenarios, the outlet drain is potentially at risk of becoming a sea water inlet, and the wetland basin may be at threat from sea water inundation during storm surges.

The final scenario, which was essentially Scenario 4 plus an addition 1 m buffer to account for any major errors in sea level rise projections, indicates that significant landward intrusion of sea water is possible, where sea water may breach the dunes and inundate Middlepoint Swamp (Figure 3). Under this scenario, it is highly likely that the ecosystem would be effected by sea water entering the ecosystem.

Table 2. Bruun Rule calculations of shoreline retreat from selected sea level rise scenarios for cross-sections at Middlepoint Swamp

Cross-section 1						
		Scenario 1	**Scenario 2**	**Scenario 3**	**Scenario 4**	**Scenario 5**
S	Sea level rise (m)	0.6	1.3	1.7	2.4	3.4
L	Length of profile* (m)	250	250	250	250	250
B	Berm height (m)	4.2	4.2	4.2	4.2	4.2
h	Closure depth# (m)	4	4	4	4	4
R	**Shoreline retreat (m)**	**18.3**	**39.6**	**51.8**	**73.2**	**103.6**
Cross-section 2						
S	Sea level rise (m)	0.6	1.3	1.7	2.4	3.4
L	Length of profile** (m)	400	400	400	400	400
B	Berm height (m)	5.1	5.1	5.1	5.1	5.1
h	Closure depth# (m)	4	4	4	4	4
R	**Shoreline retreat (m)**	**26.4**	**57.1**	**74.7**	**105.5**	**149.5**
Cross-section 3						
S	Sea level rise (m)	0.6	1.3	1.7	2.4	3.4
L	Length of profile*** (m)	292	292	292	292	292
B	Berm height (m)	3.1	3.1	3.1	3.1	3.1
h	Closure depth# (m)	3	3	3	3	3
R	**Shoreline retreat (m)**	**28.7**	**62.2**	**81.4**	**114.9**	**162.8**
*	*based on ausobath + visual change in sea floor from aerial photo (distance from shore to top of berm (50 m) + 200 m offshore)*					
**	*based on visual change in sea floor from aerial photo (distance from shore to top of berm (250 m) + 100 m visual change in sea floor)*					
***	*based on visual change in sea floor from aerial photo (distance from shore to top of berm (22 m) + 270 m visual change in sea floor)*					
#	*based on local knowledge (crayboat operators / recreational divers)*					

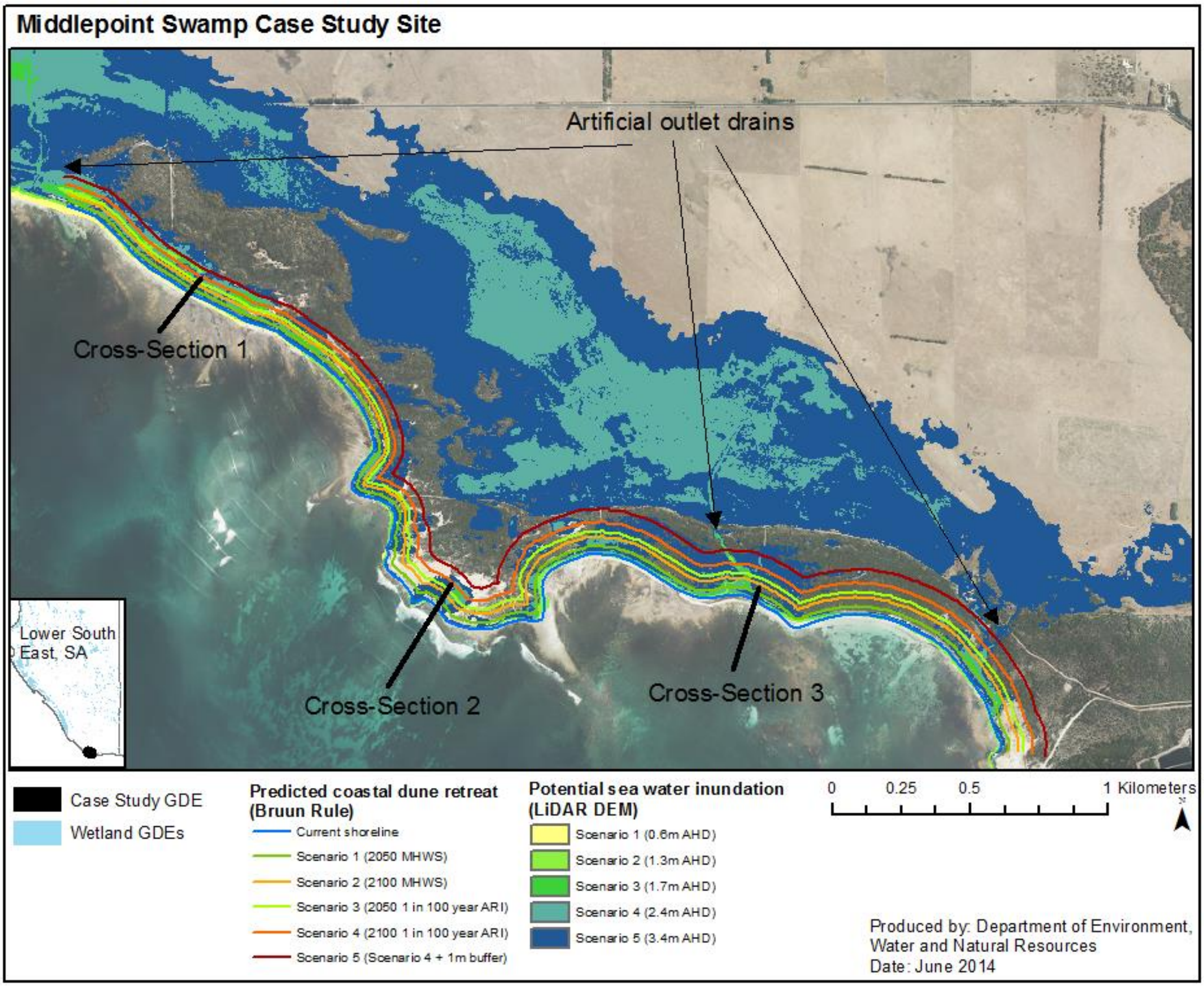


Figure 3. Results of Bruun Rule calculations of coastal dune retreat from sea level rise scenarios. Concentric coastal lines indicate dune retreat. Potential sea water inundation extent displayed using 2 m LiDAR DEM

Discussion

The outputs of this initial application of the Bruun Rule and LiDAR DEM modelling of the potential hazard of sea level rise to a coastal wetland ecosystem on the Lower South East coast (Middlepoint Swamp) indicate that potential impacts, particularly from sea water storm surges intruding via outlet drains could occur by 2050 (for storm surges), and more likely to be a significant issue by 2100.

This analysis however should be viewed in context of the limitations of the analysis methods, and input data. The resulting maps should be considered as potential hazards, and not actual. Given the low-lying topography of the South East region, and the likelihood that sea level rise may impact coastal areas, and therefore coastal ecosystems, it is suggested that further flood mapping with use of more robust modelling applications be considered. For the purposes of the risk assessment for Middlepoint Swamp as part of the ICCWR project (this report), this level of analysis was considered of an appropriate scale and accuracy to rule out sea level rise as a direct hazard to Middlepoint Swamp within the temporal boundaries of the risk assessment framework (2030 – refer to Section 3.4.1). It was considered unlikely that impacts from sea level rise would be realised by 2030, however consideration of mitigation activities (including management of outlet drains to prevent storm surge inflows from sea water), where possible, should be investigated for Middlepoint Swamp, and other coastal GDEs.

B. Vegetation community identifiers

Vegetation community ID	Vegetation community description
TerrP1	Terrestrial - pasture grass
Gahn2	*Gahnia filum* / *Juncus kraussii* over pasture grasses
JunK3	*Juncus kraussii* / *Distichlis distichophylla* sedgeland
Baum4	*Baumea juncea* / *Juncus kraussii* dense sedgeland
Sarc5	*Sarcocornia quinqueflora / Triglochin striata / Myriophyllum* sp. / *Mimulus repens / Selliera radicans* brackish herbland
SpJu6	Sparse *Juncus kraussii* sedgeland over brackish herbs
DenJ7	Dense *Juncus kraussii* sedgeland over brackish herbs
Char8	Charophytes / *Ruppia polycarpa* open water basin
DenF9	Dense *Ficinia nodosa / Samolus repens / Juncus kraussii* drier sedgeland
Ozoth10	*Ozothamnus ferrugineus / Leucopogon parviflorus* over *Ficinia nodosa* sparse shrubland
Leuc11	*Leucopogon parviflorus / Ozothamnus ferrugineus* Shrubland
SarcP12	*Sarcocornia quinqueflora / Hordeum marininum / Samolus repens* drier brackish herbland/pasture
DenF9and10	Vegetation community group: DenF9 and Ozoth10
DenJ7and4	Vegetation community group: DenJ7 and Baum4
SpJu6and5	Vegetation community group: SpJu6 and Sarc5

C. Description of nodes in the Middlepoint Swamp BBN and their output states

	Node	Description	Possible node states
GIS raster dataset inputs	Landuse	GIS raster input for assessment area, indicating grazed and un-grazed land use/	Grazed Ungrazed
	PriorVeg	Prior Vegetation – GIS raster input for assessment area of the vegetation community present in the base year. This was determined from vegetation mapping in 2013 (as the base year for projections to 2030), and vegetation mapping from a 1978 epoch (as the base year for predicting 2013)	Vegetation communities: TerrP1 Leuc11 DenF9and10 Gahn2 SarcP12 JunK3 DenJ7and4 SpJu6and5 Char8 (see Appendix B for descriptions)
	MappedDuration2030	Duration of inundation – GIS raster input for assessment area for climate change scenarios to 2030 in annual days of inundation. Discretised based on vegetation hydrological niche modelling.	d0to10: 0 to 10 days d10to50: 10 to 50 days d50to100: 50 to 100 days d100to130: 100 to 130 days d130to170: 130 to 170 days d170to220: 170 to 220 days d220to300: 220 to 300 days d300to365 300 to 365 days
GIS input uncertainty	Duration	Duration of inundation GIS input uncertainty. An arbitrary 70% certainty of the mapped projected durations being correct was applied, with a 15% certainty that the actual inundation duration could fall either side of the specified duration category.	d0to10: 0 to 10 days d10to50: 10 to 50 days d50to100: 50 to 100 days d100to130: 100 to 130 days d130to170: 130 to 170 days d170to220: 170 to 220 days d220to300: 220 to 300 days d300to365 300 to 365 days
Habitat suitability	[VegCom]Niche (e.g. TerrP1Niche)	The hydrological niche (duration of inundation) of each of the nine vegetation community modelled. Determines if the duration of inundation is within the thresholds identified for each vegetation community.	Inside threshold Outside threshold
	[VegCom]Suitability (e.g. TerrP1Suitability)	Suitability of the environmental conditions for the presence of each vegetation community (based on duration of inundation thresholds, and prior vegetation probabilities of transition).	High Low
Risk	RiskOfTerrestrialisation	Overall risk of the wetland no longer supporting aquatic vegetation communities (terrestrialisation) based upon the states of terrestrial vegetation communities TerrP1and Leuc11.	High Low

D. Hydrological niche model coefficients

Vegetation community	intercept	z-score	p Z < z	Duration	z-score	p(Z<z)	(Duration)^2	z-score	p(Z<z)
Gahn2	*-3.7*	-7.9	0.000	**126.4**	5.6	0.000	**-6.5**	-0.3	0.748
JunK3	*-18.6*	-4.6	0.000	**168.7**	6.1	0.000	**-27.7**	-2.0	0.044
SpJu6	*-59.0*	-3.2	0.001	**247.3**	5.1	0.000	**-62.2**	-2.6	0.009
DenJ7	*-56.6*	-7.8	0.000	**263.3**	8.3	0.000	**-83.1**	-5.3	0.000
Char8	*-8.5*	-2.9	0.004	**111.7**	4.4	0.000	**24.3**	1.7	0.088
DenF9	*-16.3*	-3.7	0.000	**184.7**	5.6	0.000	**-92.2**	-3.4	0.001
Ozoth10	*-52.9*	-4.3	0.000	**172.1**	5.0	0.000	**-104.3**	-2.5	0.014
Leuc11	*-26.6*	-2.2	0.031	**-225.5**	-2.0	0.049	**298.6**	296.0	0.000
SarcP12	-80.6	-4.5	0.000	**436.5**	5.5	0.000	**-317.1**	-4.5	0.000

Wetland vegetation communities: **Char8** = Charophytes / Ruppia polycarpa open water; **SpJu6** = Sparse Juncus kraussii sedgeland over brackish herbs; **DenJ7** = Dense Juncus kraussii sedgeland over brackish herbs; **JunK3** = Juncus kraussii / Distichlis sedgeland; **SarcP12**= Sarcocornia quinqueflora / Hordeum marininum / Samolus repens drier brackish herbland/pasture; **Gahn2** = Gahnia / Juncus kraussii over pasture grasses; **DenF9** = Dense Ficinia nodosa / Samolus repens / Juncus kraussii drier sedgeland; **Ozoth10** = Ozothamnus ferrugineus / Leucopogon parviflorus over Ficinia nodosa sparse shrubland; **Leuc11** = Leucopogon parviflorus / Ozothamnus ferrugineus Shrubland

8 Units of measurement

Name of unit	Symbol	Definition in terms of other metric units	Quantity
day	d	24 h	time interval
hectare	ha	10^4 m^2	area
kilometre	km	10^3 m	length
metre	m	base unit	length
year	y	365 or 366 days	time interval

~ approximately equal to

EC electrical conductivity (µS/cm)

9 List of acronyms

AARR — Average annual rainfall residual

AHD — Australian Height Datum

BBN — Bayesian Belief Network

BoM— Bureau of Meteorology

CDFM — Cumulative Deviation From Mean

CPT — Conditional Probability Table (Netica)

DEM — Digital Elevation Model

DEWNR — Department of Environment, Water and Natural Resources (South Australia)

EPBC (Act) — Environment Protection and Biodiversity Conservation Act 1999

ESRI — Environmental Systems Research Institute, Inc. (ArcGIS)

EWR — Environmental Water Requirement

GCM — Global Climate Model

GDE — Groundwater Dependent Ecosystem

GIS — Geographic Information Systems

GIWR — Goyder Institute for Water Research

GLM — Generalised Linear Modelling

HARTT — Hydrograph and Rainfall Time Trend (model)

ICCWR — Impacts of Climate Change on Water Resources (project – DEWNR)

LiDAR — Light Detection and Ranging. A remote sensing technology that measures distance by illuminating a target with an airborne laser and analysing the reflected light.

NHMM — Non-homogeneous hidden Markov model

NRM — Natural Resource Management

PET — Potential Evapotranspiration

PWA — Prescribed Wells Area

RCP — Representative Concentration Pathways (greenhouse gas)

SE NRM — South East Natural Resources Management (Board / Region)

SAWID — South Australian Wetland Inventory Database

SWL — Surface Water Level

WAP — Water Allocation Plan

10 References

ACE CRC (2008). Sea Level Rise. Understanding the past – improving projections for the future. URL: http://www.cmar.csiro.au/sealevel/sl_drives_short.html

Alcoe, D., Gibbs, M. & Green, G. (2012) Impacts of Climate Change on Water Resources Phase 3 Volume 3: Alinytjara Wilurara Natural Resources Management Region. DFW Technical Report 2012/05, Government of South Australia, through Department for Water, Adelaide

Bates, B. C., Charles, S.P., & Hughes, J.P (1998). Stochastic downscaling of numerical climate model simulations. Environmental Modelling & Software, 13(3-4), 325-331.

Beacon Ecological (2010). Groundwater Dependent Ecosystem Monitoring Program – Vegetation Monitoring. Report to Department for Environment and Heritage, South Australia.

Brock, M.A., Nielsen, D.L. and Crossle, K. (2005) Change in biotic communities developing from freshwater wetland sediments under experimental salinity and water regimes. *Freshwater Biology* 50: 1376-1390.

Brown, K., Harrington, G., & Lawson, J. (2006). Review of groundwater resource condition and management principles for the Tertiary Limestone Aquifer in the South East of South Australia. Department of Water, Land and Biodiversity Conservation. DWLBC Report 2006/2.

Brown, K.G., Love A.J., Harrington, G.A. (2001). Vertical groundwater recharge to the Tertiary confined sand aquifer, South East, South Australia. Dept. Water Resources South Australia. Report Book DWR 2001/002 (unpublished).

Bruun, P. (1962). Sea level rise as a cause of shore erosion. *Journal Waterways and Harbours Division*, 88(1-3), 117-130.

Buisson, L., Thuiller, W., Casajus, N., Lek, S. & Grenouillet, G. (2010). Uncertainty in ensemble forecasting of species distribution. *Global Change Biology* 16: 1145-1157.

Cain, J. (2001). Planning improvements in natural resources management: guidelines for using Bayesian networks to support the planning and management of development programs in the water sector and beyond. Crowmarsh Gifford, Wallington, Oxon, UK: http://nora.nerc.ac.uk/9461/1/N009461BO.pdf

Casanova, M.T. & Brock, M. How do depth, duration and frequency of flooding influence the establishment of wetland plant communities? *Plant Ecology* 147: 237-250.

Catford, J.A., Morris, W.K., Vesk, P.A., Gippel, C.J. & Downes, B. (2014). Species and environmental characteristics point to flow regulation and drought as drivers of riparian plant invasion. *Diversity and Distributions* 20: 1084-1096.

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. (2013a). Adapting to climate change: a risk assessment and decision making framework for managing groundwater dependent ecosystems with declining groundwater levels – Guidelines for use. National Climate Change Adaptation Research Facility, Gold Coast.

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. (2013b). Adapting to climate change: a risk assessment and decision making framework for managing groundwater dependent ecosystems with declining groundwater levels – Development and case studies. National Climate Change Adaptation Research Facility, Gold Coast.

Charles, S. P., Bates, B.C., & Hughes, J.P. (1999a). A spatio-temporal model for downscaling precipitation occurrence and amounts. Journal of Geophysical Research, 104(D24), 31657-31669.

Charles, S. P., Bates, B.C., Whetton, P.H. & Hughes, J.P. (1999b). Validation of downscaling models for changed climate conditions: case study of southwestern Australia. Climate Research, 12, 1-14.

Cook, P.G., Simmons, C.T. & Brunner, P. (2008). Regional groundwater dependent ecosystems – our undiscovered assets at risk. Report to the Centre for Natural Resource Management. CSIRO Land and Water, Flinders University.

Cooper, J.A.G & Pilkey, O.H. (2004). Sea-level rise and shoreline retreat: time to abandon the Bruun Rule. Global and Planetary Change 43(3-4): 157-171

CSIRO (2007). Climate Change in Australia. Pearce, K, Holper, P, Hopkins, M, Bouma, W, Whetton, P, Hennessy, K and Power, S, (Eds.), CSIRO Technical Report 2007, Commonwealth of Australia, through Commonwealth Scientific and Industrial Research Organisation and Bureau of Meteorology, Melbourne

DECCW (2010). *Coastal Risk Management Guide: Incorporating sea level rise benchmarks in coastal risk assessment*. Department of Environment, Climate Change and Water, Sydney South, NSW.

Denny, M., Herpich, D., Cetin, L., Harding, C. & Green, G. (2014). South East wetlands: climate change risks and opportunities for mitigation. DEWNR Technical Report 2014/13. Government of South Australia, through Department of Environment, Water and Natural Resources, Adelaide.

Department of Water (2008). Management triggers and responses for groundwater-dependent ecosystems in the South West groundwater areas. Department of Water, Water resource allocation planning series, Report no.31. Government of Western Australia.

Department of Water (2012). Ecological water requirements of the Lower Robe River. Environmental water report series, Report no. 22. Government of Western Australia.

DEWNR) (2014). Dataset of Agreed Climate Change Projections for South Australia. Government of South Australia, though Department of Environment, Water and Natural Resources, Adelaide.

DFW (2010). South East water science review. Lower Limestone Coast Water Allocation Plan Taskforce, Department for Water, Adelaide.

DFW (2011). Upper South East Drainage Network Management Strategy – prepared under Section 43A of the Upper South East Dryland Salinity and Flood Management Act 2002. Department for Water, Adelaide.

Dickson C.R., Farrington L., & Bachmann M. (2014). Survey and description of the Seasonal Herbaceous Wetlands (Freshwater) of the Temperate Lowland Plains in the South East of South Australia. Report to Department of Environment, Water and Natural Resources, Government of South Australia. Nature Glenelg Trust, Mount Gambier, South Australia.

Dyer, F., ElSawah, S., Croke, B., Griffiths, R., Harrison, E., Lucena-Moya, P., Jakeman, A. (2014). The effects of climate change on ecologically-relevant flow regime and water quality attributes. *Stoch Environ Res Risk Assess* 28: 67-82.

Ecological Associates (2009). Estimation of Water Requirements of Wetlands in the South East of South Australia. Department of Water, Land and Biodiversity Conservation, South Australia.

Ecological Associates (2010). Monitoring the Ecological Response of High Value Wetlands to Changes in Groundwater. Ecological Associates report CC017-2-C prepared for Department for Environment and Heritage, South Australia.

Ferdowsian, R., Pannell, D.J., McCarron, C., Ryder, A., Crossing, L. (2001). Explaining groundwater hydrographs: separating atypical rainfall events from time trends, *Australian Journal of Soil Research*, 39, pg 861 – 875.

Fotheringham, D. & Rutherford, H. (2013). Projected sea level rise flood risk mapping – SE coast of South Australia. Report prepared for the Coast Protection Board. Department of Environment Water and Natural Resources.

Fielding, A. H. & Bell, J. F. (1997). A review of methods for the assessment of prediction errors in conservation presence/absence models. *Environmental Conservation,* 24**,** 38-49.

Frost, A. J. et al. (2011). A comparison of multi-site daily rainfall downscaling techniques under Australian conditions. Journal of Hydrology 408, 1-18.

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.

Gibbs, M., Green, G., & Wood, C. (2011). Impacts of Climate Change on Water Resources Phase 2: Selection of Future Climate Projections and Downscaling Methodology. DFW Technical Report 2011/02, Government of South Australia, through Department for Water, Adelaide.

Gibbs, M., Alcoe, D., and Green, G. (2012). Impacts of Climate Change on Water Resources Phase 3 Volume 4: South Australian Arid Lands Natural Resources Management Region. DEWNR Technical Report 2013/06, Government of South Australia, through Department of Environment, Water and Natural Resources, Adelaide.

Gnangara Sustainability Strategy Taskforce (2009). Gnangara Sustainability Strategy – situation statement, Department of Water, Government of Western Australia.

Goyder Institute for Water Research (2014 in prep). Developing ecological response models and determining water requirements for wetlands in the South-East of South Australia, Goyder Institute for Water Research URL: http://goyderinstitute.org/index.php?id=50

Green, G., Gibbs, M., and Wood, C. (2011). Impacts of Climate Change on Water Resources, Phase 3 Volume 1: Northern and Yorke Natural Resources Management Region. DFW Technical Report 2011/03, Government of South Australia, through Department for Water, Adelaide.

Green, G., Gibbs, M., Alcoe, D. and Wood, C. (2012). Impacts of Climate Change on Water Resources, Phase 3 Volume 2: Eyre Peninsula Natural Resources Management Region. DFW Technical Report 2012/04, Government of South Australia, through Department for Water, Adelaide.

Hallermeier, R.J. (1981). A profile zonation for seasonal sand beaches from wave climate, *Coastal Engineering,* 4: 253–77.

Harding, C. (2012a). Impacts of Climate change on Water Resources in South Australia – Phase 4 Volume 1: First Order Risk Assessment and Prioritisation of Water Dependent Ecosystems. DFW Technical Report 2012/07, Government of South Australia, through Department of Water, Adelaide.

Harding, C. (2012b). Extension of the Water Dependent Ecosystem Risk Assessment Framework to the South East NRM Region. Department for Water, Technical report 2012/10.

Harding, C. & O'Connor, P. (2012). Delivering a strategic approach for identifying water dependent ecosystems at risk: a preliminary assessment of risk to water dependent ecosystems in South Australia from groundwater extraction. DFW Technical Report 2012/03, Government of South Australia, Department for Water, Adelaide.

Hebb, A.J., Mortsch, L.D., Deadman, P.J. and Cabrera, A.R. (2013). Modelling wetland vegetation community response to water-level change at Long Point, Ontario. *Journal of Great Lakes Research* 39:191-200.

Hebb, A.J. (2003). Implementation of a GIS to Assess the Effects of Water Level Fluctuations on the Wetland Complex at Long Point, Ontario. Thesis. University of Waterloo, Canada.

Horwitz, P., Sommer, B. & Froend, R. (2009). Chapter four: Wetlands and groundwater-dependent ecosystems. In: Gnangara Sustainability Strategy – Biodiversity Report (eds B.A. Wilson & L.E. Valentine). Department of Environmental and Conservation, Government of Western Australia.

Hudon, C., Wilcox, D., Ingram, J. (2006). Modelling wetland plant community response to assess water-level regulation scenarios in the Lake Ontario – St. Lawrence river basin. *Environmental Monitoring and Assessment* 113: 303-328.

Hughes, J. P., Guttorp, P. & Charles S. P. (1999). A non-homogeneous hidden Markov model for precipitation occurrence. Applied Statistics, 48(1): 15-30.

Hyde, N.L. (2006). A summary of investigations into ecological water requirments of groundwater-dependent ecosystems in the South West groundwater areas. Environmental Water Report 3, Department of Water, Government of Western Australia, Perth.

IPCC (2013). Climate Change 2013: The physical science basis. Contribution of Working Group I to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change [Stocker, T.F., D. Qin, G.K. Plattner, M. Tignor, S.K. Allen, J. Boschung, A. Nauels, Y. Xia, V. Bex and P.M. Midgley (eds)]. Cambridge University Press, Cambridge, United Kingdom and New Your, NY, USA.

Keddy, P.A. & Reznicek, A.A. (1986). Great Lakes vegetation dynamics: the role of fluctuating water levels and buried seeds. *Journal of Great Lakes Research* 12: 25-36.

Kløve, B., Ala-Aho, P., Bertrand, G., Gurdak, J., Kupfersberger, H., Kværner, J., Muotka, T., Mykrä, H., Preda, E. Rossi, P., Bertacchi Uvo, C., Velasco, E., Pulido-Velazquez, M. (2013). Climate change impacts on groundwater dependent ecosystems. *Journal of Hydrology,* article in press.

Kløve, B., Balderacchi, M., Gemitzi, A., Hendry, S., Kværner, J., Muotka, T. & Preda, E. (2014). Protection of groundwater dependent ecosystems: current poligies and future management options. Water Policy, uncorrected proof.

Lindenmayer, D. Burns, E., Thurgate, N., Lowe, A. (Eds) (2014). Biodiversity and Environmental Change: Monitoring, challenges and direction. CSIRO publishing, Melbourne.

McFarlane, D., Stone, R., Martens, S., Thomas, J.m 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.

Mitchell, N., Sommer, B., & Speldewinde (2013). Adapting to climate change: a risk assessment and decision making framework for managing groundwater dependent ecosystems with declining groundwater levels – Supporting Document 3: Identifying thresholds for responses of amphibians to groundwater and rainfall decline. National Climate Change Adaptation Research Facility, Gold Coast.

Mustafa S., Slater S., & Barnett S. (2012). Preliminary investigation of seawater intrusion into a freshwater coastal aquifer – Lower South East. DEWNR Technical Report 2012/01, Government of South Australia, through Department of Environment, Water and Natural Resources, Adelaide.

Neville, S. (2013). Adapting to climate change: a risk assessment and decision making framework for managing groundwater dependent ecosystems with declining groundwater 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, Gold Coast.

Nielsen, D.L. & Brock, M.A. (2009). Modified water regime and salinity as a consequence of climate change: prospects for wetlands of Southern Australia. *Climatic Change* 95: 523-533.

Norsys Software Corporation (1998). Netica™ Application for Belief Networks and Influence Diagrams: User's Guide. Norsys Software Corporation, Vancouver, Canada.

NWC (2011). The National Water Initiative – securing Australia's water future: 2011 assessment, National Water Commission, Canberra.

R Core Team (2014). R: A language and environment for statistical computing. R Foundation for Statistical Computing. Vienna, Austria.

Ranasinghe, R., Watson, P., Lord, D., Hanslow, D. & Cowell, P. (2007). Sea Level Rise, Coastal Recession and the Bruun Rule, *Proceedings of the 18th Australasian Coastal and Ocean Engineering Conference*, July.

REM (2006). A review of the environmental water requirements of the groundwater dependent ecosystems of the South East Prescribed Wells Areas Stage 2 report. Prepared for the South East Catchment water Management Board by Resource & Environmental Management Pty Ltd, and Ecological Associates Pty Ltd.

Rignot, E. I., Velicogna, M. R., van den Broeke, A., Monaghan, and J. Lenaerts (2011). "Acceleration of the contribution of the Greenland and Antarctic ice sheets to sea level rise". *Geophysical Research Letters* 38 (5).

Rollason, V. Patterson, D. and Huxley, C. (nd) Assessing shoreline response to sea level rise: an alternative to the Bruun rule. Unpublished paper. http://www.coastalconference.com/2010/papers2010/Verity%20Rollason%20full%20paper.pdf

SENRMB (2010). South East Regional Natural Resources Management Plan – February 2010. South East Natural Resources Management Board, Government of South Australia.

SENRMB (2013). Water Allocation Plan for the Lower Limestone Coast Prescribed Wells Area. Prepared by the South East Natural Resources Management Board, Department of Environment, Water and Natural Resources.

SKM (2009). Classification of groundwater-surface water interactions for water dependent ecosystems in the South East, South Australia. Report to Department of Water, Land and Biodiversity Conservation, Sinclair Knight Merz.

SKM (2010). Conceptual diagrams of wetland groundwater dependent ecosystems – GDE monitoring project South East, SA. Report to Department for Water, Sinclair Knight Merz.

Smith, C.S., Howes, A.L., Price, B. & McAlpine, C.A. (2007). Using a Bayesian belief network to predict suitable habitat of an endangered mammal – The Julia Creek dunnart (*Sminthopsis douglasi*). Biological Conservation 333-347.

Smith, M.J., Keely, M.O., Scroggie, M.P., Schreiber, E.S., Kohout, M. (2009). Asessing changes in macrophyte assemblages with salinity in non-riverine wetlands: A Bayesian approach. *Aquatic Botany* 90: 137-142.

Speldewinde, P. (2013). Adapting to climate change: a risk assessment and decision making framework for managing groundwater dependent ecosystems with declining groundwater levels – Supporting Document 6: Development of Bayesian Networks for modelling the impacts of falling groundwater due to climate change on groundwater dependent ecosystems. National Climate Change Adaptation Research Facility, Gold Coast.

Suppiah, R., Preston, B., Whetton, P.H., McInnes, K.L., Jones, R.N., Macadam, I, Bathols, J. & Kirono, D. (2006), Climate change under enhanced greenhouse conditions in South Australia, Commonwealth of Australia, through Commonwealth Scientific and Industrial Research Organisation Marine and Atmospheric Research, Melbourne

Venables, W. N. & Ripley, B. D. (2002). *Modern Applied Statistics with S*, Springer, New York. .

Wood, C. & Green, G. (2010). Impacts of Climate Change on Water Resources, Phase 1: First Order Risk Assessment and Prioritisation. DFW Technical Report 2010/01, Government of South Australia, through Department for Water, Adelaide.

Wood, G. & Way, D. (2011). Development of the Technical Basis for a Regional Flow Management Strategy for the South East of South Australia. DFW Report 2011/21, Government of South Australia, through Department for Water, Adelaide.

Zuur, A. F., Ieno, E. N. & Smith, G. M. (2007). *Analysing ecological data,* New York USA, Springer.

Government of South Australia
Department of Environment,
Water and Natural Resources