An integrated eco-hydrological approach for assessing critical wetland habitats and conservation reserves in a changing climate

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Dedication

To my parents and my loving wife and daughter

Without their continuous inspiration and support
this thesis would never have been completed
Abstract

Wetlands are unique ecosystems and hotspots of biodiversity in many parts of the world. Globally, wetland biota are under threat due to anthropogenic pressures associated with land-use change and water abstraction. Climate change and increased climate variability are superimposed as an additional pressure that can amplify these existing stressors. Particularly in Mediterranean and semi-arid environments such as those in southern Australia, the degradation or loss of wetland habitats is already substantial and highly sensitive to reduced rainfall. For wetland-dependent species that are already endangered, changes to the quality or size of their present habitats can challenge their survival. Developing methods that are able to make the link between wetland ecohydrology and habitat suitability for biota is therefore essential to support wetland management and conservation of biodiversity. This thesis focuses on the relationship between ecohydrology and habitat suitability for the Critically Endangered Western Swamp Tortoise (WST, *Psedemydura umbrina*).

For a species such as the WST that is experiencing declining habitat suitability, one last-ditch conservation option is a type of translocation known as assisted colonization (AC). AC involves moving species outside of their historical range to sites where they are thought to have improved chances for population growth. However, the success rate of translocations outside of species ranges is often poor. Further, managers almost never have the information required to identify suitable wetland systems that can support the abiotic and biotic needs of the focal species both now and under future climate scenarios.

In order to improve the prospects for conserving the WST in the wild, the aim of this research was to develop a detailed understanding of the ecohydrology of two critical wetland sites that they currently inhabit and to explore how shifting ecohydrological dynamics under climate change will impact long-term habitat suitability. To achieve this, the research combines analysis of long-term data, collection of ecohydrological field data, detailed spatial analysis of habitat condition and development of complex wetland models. These models are ultimately applied to unravel the ecohydrological interaction of wetlands subject to shifting climate forces, and to identify potential regions for candidate sites within the south-west of Western Australia for assisted migration of the WST. Critically, these sites need to be suitable now, and under future scenarios of climate change.

This research was structured in four consecutive parts. The first part characterised the eco-hydrology of two adjacent protected wetlands (Ellen Brook Nature Reserve (EBNR) and Twin Swamps Nature Reserve (TSNR), located in the Swan Coastal Plain of Western Australia. These two wetlands are different in their hydrological characteristics, and the
population trajectories for the WST also differ markedly between the two reserves. The
analyses explored the regional context of groundwater-surface water connectivity relevant
to the two wetlands, as well as specific ecohydrological features of the sites including
inundation regimes, hydroperiods, water temperature, and soil moisture variability. Field
data on water levels, water temperatures, soil moisture, hydroperiods, and climate data
were also compared. The results of these analyses showed that climate change was clearly
evident in south-western part of Australia, and had affected the hydrological function of
the wetlands. There were also differences in the ecohydrological features of EBNR and
TSNR. The unique characteristics of each were discussed in relation to the role they may
be playing in shaping the population viability of WSTs at EBNR and TSNR.

In the second part of this research, a detailed spatial analysis of EBNR and TSNR
hydrology, morphometry and vegetation was conducted to demonstrate the unique natures
of the individual wetlands. A high resolution (1m) Digital Elevation Model (DEM) was
used for hydrological and morphometric analysis, and time-series of aerial imagery from
NearMap™ were used for assessment of vegetation cover. It was shown that both wetlands
had characteristic spatial heterogeneity and vegetation patterns, and the seasonal and
spatial variation and has implications for the water balance and thereby to the life-cycle
of the WST. The outcome of the data compilation and analysis was the development
of a conceptual model of the wetland ecohydrology upon which the development of an
eco-hydrological model could be based.

The third part of the research focused on the development of an eco-hydrological model
for the EBNR site where the most viable population of the WST population occurs. In or-
der to examine the capacity and accuracy of wetland models, two models were developed:
the first was a simple lumped water balance model, and the second was a detailed compart-
mentalized eco-hydrological model configured to represent the local sub-domains within
EBNR. Both models were tested and validated against field data collected from earlier
parts of this research, including hydroperiods, soil moisture and vegetation biomass. The
models were then forced with present and future climate scenarios. Whilst a good calib-
ration of the wetland volume was made with the lumped model, the more detailed model
had the additional capacity to represent the complexity of eco-hydrological interactions
and predict vegetation response, soil moisture and other hydrological processes that are
relevant to the physiological ecology of the WST. Outputs of the eco-hydrological model
were analysed to calculate and compare the water budget of EBNR, both under present
conditions and future climates and it was shown that the water budget is like to shift
considerably under projected drying trends for south western Australia.

In the final part of the research, an idealised implementation of the eco-hydrological
model validated in the previous research was applied as a screening tool to assess how a
wetland would respond to climate change at any location within the south-west of Western
Australia. Two extreme wetland types, in terms of their groundwater connectivity, were
assessed, and each simulated at 34 randomly-distributed locations in the southwestern
ecoregion of Australia. This was done for the present climate and under three future
climate scenarios for 2030. Additionally, the wetland predictions were incorporated with a
published WST reaction norm to calculate a suitability index that would infer physiological
performance across a spatial and temporal gradient. The index was constrained by known activity temperatures for WST and excluded outputs that did not meet the thermal criteria or that predicted a hydroperiods at a time of year when the WST goes into obligatory aestivation. The results were interpolated across a 1.5° grid and served to highlight the potential variability in the suitability of wetlands for WSTs. Wetlands that were disconnected from groundwater had shorter hydroperiods and were more sensitive to a drying climate, whereas groundwater-connected wetlands had overall better suitability under all four climate scenarios. The maximum suitability index values were found close to or near to the existing habitats as TSNR and EBNR, due to an ideal combination of hydroperiod and water temperature. However other locations were also suitable in well south of the current habitats, near the townships of Walpole and Albany. Further investigation of potential candidate wetlands for translocations are recommended, using high-resolution environmental datasets. In this way, ecohydrological variables can be further incorporated with other layers of decision support tools, such as land-use criteria and assessments of translocation risk.

This research showed that while having some common features, neighbouring wetland systems can be unique depending on their ecological and hydrological characteristics. Ecohydrology plays a key role in shaping the suitability of wetland habitats in terms of the services they provide to fauna and flora that rely on them. In order to implement any conservation interventions such as assisted migrations, it is necessary that conservation managers have detailed scientific understanding of these complex processes at their disposal. Modelling tools can be effectively used to thoroughly understanding these complex systems, and the predictive powers of such models can be harnessed to evaluate the impacts of climate change. The ecohydrological model developed in this thesis can be used as an effective tool to aid decision support for selecting potential assisted migration sites for the WST, and potentially for other endangered species.
Declaration

This thesis is my own composition, and contains no material which was been accepted for the award of any degree or diploma in any university. During the production of this thesis, a brief description of the validation of WET-R model was contributed to the following paper. I have the permission of the co-authors of this paper to include this work in my thesis.


I received scholarly inputs and suggestions in my thesis from my supervisors Matthew Hipsey and Nicola Mitchell and co-supervisor Gavan McGrath. I also received technical help from Dr. Jana Coletti in Chapter 4 for development of the model.

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## Acronyms

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<th>Description</th>
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<tr>
<td>AC</td>
<td>assisted colonization</td>
</tr>
<tr>
<td>DPaW</td>
<td>Department of Parks and Wildlife</td>
</tr>
<tr>
<td>BoM</td>
<td>Bureau of Meteorology</td>
</tr>
<tr>
<td>DoW</td>
<td>Department of Water</td>
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<tr>
<td>SWWA</td>
<td>South-west of Western Australia</td>
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<tr>
<td>EBNR</td>
<td>Ellen Brook Nature Reserve</td>
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<tr>
<td>TSNR</td>
<td>Twin Swamps Nature Reserve</td>
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<tr>
<td>SCP</td>
<td>Swan coastal plain</td>
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<td>WST</td>
<td>Western Swamp Tortoise (<em>Pseudemydura umbrina</em>)</td>
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<tr>
<td>EC</td>
<td>electric conductivity</td>
</tr>
<tr>
<td>DBH</td>
<td>diameter at breast height</td>
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<tr>
<td>IC</td>
<td>initial condition</td>
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<tr>
<td>ASL</td>
<td>above sea level</td>
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<td>OM</td>
<td>organic matter</td>
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<td>DI</td>
<td>dryness index</td>
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<td>GHG</td>
<td>greenhouse gas</td>
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<tr>
<td>GCM</td>
<td>general circulation model</td>
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<td>GDE</td>
<td>groundwater dependant ecosystem</td>
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<td>GIW</td>
<td>geographically isolated wetlands</td>
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<tr>
<td>CBE</td>
<td>clay-pan based ecosystem</td>
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<tr>
<td>AWAP</td>
<td>Australian Water Availibility Project</td>
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<td>LAI</td>
<td>leaf area index</td>
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<tr>
<td>LiDAR</td>
<td>Light Detection and Ranging</td>
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<tr>
<td>DEB</td>
<td>dynamic energy budget</td>
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<td>DEM</td>
<td>digital elevation model</td>
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<tr>
<td>AHD</td>
<td>Australian Height Datum</td>
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Chapter 1
Introduction

1.1 Background

Climate change is a major threat for many endangered species (Scheffers et al., 2016). Numerous studies have shown that climate change is likely to further stress many species, adding to the pressures already created by other threatening processes (e.g. Böhm et al., 2013). Impacts attributed to climate change include altering sex ratios, (e.g. Mitchell and Janzen, 2010), changing phenologies (e.g. White et al., 2005), or changing migration pathways (e.g. Vitt et al., 2010). These affects are predicted to alter the global distribution of species, with some studies already showing species shifting in range to track their preferred climates (generally polewards and towards higher elevations; Araujo et al. (2006). However, the dispersal ability of some species is limited, and habitat fragmentation can prevent even good dispersers from tracking a shifting habitat (Hewitt et al., 2011; Willis et al., 2009). This is a particular concern for aquatic systems as they demonstrate greater natural spatio-temporal variability (Tulbure et al., 2014). For this subset of species, unless they are capable of adapting in-situ, they may become stuck in an unsuitable habitat (Bishop-Taylor et al., 2015) and will in this case require intervention to be able to persist in the wild (Olden et al., 2011). Relocating animals within their range, or to parts of their former range (translocation), has become an increasingly necessary tool in wildlife management, both for conservation and other purposes (Griffith, 1989; Wolf et al., 1996). However, several studies suggest that many translocations are not successful (Dodd Jr. and Seigel, 1991; Griffith, 1989; Seddon et al., 2015, 2009, 2014; Wolf et al., 1996), and translocations can also be very expensive (e.g. Rahbek, 1993). Nonetheless, they are likely to be an increasingly important part of wildlife conservation, and recently there have been positive reports of species that have recovered thanks to this management action (Seddon et al., 2015; Seddon, 2010).

Although, the history of movement of species outside of their current or former range is not very short (Seddon et al., 2015), but still is generally regarded to be high risk (Ricciardi and Simberloff, 2009a,b) when compared to traditional translocations, mostly due to the risks of species becoming invasive when introduced to novel environments (Mueller and Hellmann, 2008). This relatively newly developed concept is termed assisted colonisation (AC) and it refers to any movement of species outside of their range that is motivated by a high extinction risk at the source location (Hulme, 2005); (n.b. the International Union for Conservation of Nature (I.U.C.N.) currently gives the definition of AC as deliberate outside of range movements for motivations that do not need to be climate change). In contrast to other relocations of species outside their known ranges, climate-motivated
movements have only been applied to a few species to date e.g. butterflies (Hunter, 2007; Parmesan, 2006) and some plants (e.g. McLane and Aitken, 2012) or even plant seeds (e.g. Chauvenet et al., 2013b; Stone, 2010).

While still a relatively new concept, there is now considerable literature on many aspects of AC, mostly emphasising decision making on whether or not to undertake AC (e.g. Hoegh-Guldberg et al., 2008; Richardson et al., 2009), as well as what (e.g. Kreyling et al., 2011), when (e.g. Burbidge et al., 2011) and how (e.g. Rout et al., 2013) to conduct ACs, in addition to ethical considerations (e.g. Sandler, 2010). To date, most of the literature has focused at the species level, with the notable exceptions of Lunt et al. (2013), who considers ecosystem functioning when species are added to, or taken from, AC sites, and Harris et al. (2013), who consider the general aspects of what makes good AC recipient sites. There is limited research attempting to link the processes of environmental change for any given habitat with the requirements of the species in question, as each environment is shaped by its own ecological processes.

Wetlands are specialized environment where soil, water, plants and animals exist in a complex ecosystem and share the ecological services in many ways (Dong and Zhang, 2011; Zhang et al., 2010). This is significant for the wetland-dependant species because the impact of increasing temperatures and altered hydrological regimes on populations is mediated by complex interactions between climatic variables, terrain and vegetation, that will be mediated by species-specific morphology, behaviour and physiological traits. Therefore, species-level assessment is only one part of the equation, and the interactions of the species with a potentially rapidly changing environment, and the degree to which the environment may persist or change under different climatic futures, needs to be more comprehensively considered (Kearney et al., 2010). In particular, in mid-latitude regions, climate change is increasingly threatening wetland dependent populations since wetland environments are especially sensitive to the effects of anthropogenic warming (e.g., increased drought periods and decreased rainfall), and they tend to amplify changes in rainfall patterns. With this context, the use of AC for wetland dependent species remains challenging, and the complexities of their current wetland habitats needs to be qualified, and the ways in which their habitat responds to shifting climates should be predicted in order to facilitate the selection of robust AC sites.

In general, modelling tools for identifying suitable sites for present and future sustainability of endangered species are necessary to support the design of AC interventions. One way to predict the future distribution of species is to use correlative climate-envelope models, which are based on occurrence data of species that are (or were) broadly distributed (Guisan and Thuiller, 2005). These approaches have an empirical basis whereby they linearly correlate presence-absence data of a species with the ecological features of its habitat. These models assume that a record of species abundance at a place, at some time, is an indication of suitability of that site (Kearney et al., 2009). Though widely used (e.g. Seddon et al., 2015), one of the challenges with this approach is that for many species that may immediately require AC they have such restricted distributions and small population sizes that climate-envelope models cannot be used to identify future habitats. This is for two key reasons. Firstly, because of the statistical nature of these models, strong correl-
1.2 The Future of an Endangered Wetland Dependant Species - the Case of Western Swamp Tortoise

Predictions (extrapolations) cannot be drawn from limited datasets (Dormann, 2007; Dormann et al., 2012). When predictions (extrapolations) are made for novel scenarios such as climate change, the performance of these models is limited to the strength of the underlying datasets. Secondly, the relationship between species’ health and habitat quality is not straightforward. Climate-envelope models are snapshots in time, and so they do not directly take into account any of the eco-physiological dynamics a species has, as driven by its environment (which will change through complex links between climate, soil, vegetation, and geomorphological attributes).

For species with limited distributions, mechanistic models offer an alternative that use a ‘first principles’ approach to understanding how a species relates to its environment. Several examples have now been published using these mechanistic models which work by considering local-scale drivers of micro-climate and links with organisms metabolic requirements (Kearney et al., 2013; Mitchell et al., 2013). This approach is used to define the fundamental niche of an organism (Kearney and Porter, 2016; Porter and Kearney, 2009) and is therefore suitable to operate independently of past distribution data, and, when adequately parameterised, is likely to perform better under future (novel) climates (Kearney et al., 2008). However, the mechanistic approach becomes more challenging for wetland dependent species as the prediction of the fundamental niche requires prediction of hydrologic processes within complex fluvial landscapes. To date, there has been no study linking the eco-hydrology of wetlands with an organisms eco-physiological requirements, and in particular for the purposes of supporting the AC of a Critically Endangered species within the context of climate change and land-use fragmentation. For a species with only a single and specialized habitat, an integrated approach (e.g. Sivapalan et al., 1996a,b) that links mechanistic, biophysical, and hydrological models is the best option for identifying potential AC sites. This research focuses on exploring how the wetland environments within the south-west of Western Australia (SWWA) are changing in the context of the requirements of a Critically Endangered wetland species.

1.2 The Future of an Endangered Wetland Dependant Species - the Case of Western Swamp Tortoise

The ephemeral wetlands of SWWA are amongst many wetlands that are under threat globally (Hill et al., 1996). Wetlands of all types support a great richness and abundance of wildlife, providing habitat, refuge, breeding grounds, and food sources for fauna, flora, and microbes (Costanza et al., 1997). Australia has one of the highest wetland diversities in the world, and includes a number of the worlds internationally recognized wetlands (Barron et al., 2012; Finlayson et al., 2011). However, a large proportion of these wetlands have been lost due to draining, filling, agricultural activities, alteration of natural water regimes, introduction of weeds and feral animals, and land-clearing for urban development (Davis and Froend, 1999).

AC has been proposed for Australia’s rarest reptile - the western swamp tortoise (*Pseudemydura umbrina*; WST). This tortoise is Critically Endangered (IUCN, 2016), and has previously benefited from translocations but now requires the identification of
new sites to improve its survival in the wild (Burbidge et al., 2010). This small, freshwater tortoise was considered extinct until it was rediscovered near Perth (Western Australia) in 1953, after which captive breeding was initiated to improve population numbers which have risen in recent years (the total population has grown from fewer than 20 individuals in the 1980s to over 500 individuals; Burbidge et al. 2010). However, as the only survivor of sub-family Pseumydurinae, the tortoise is thought to have limited ‘evolutionary potential due to the small population size and consequent low genetic diversity, as well as its long generation times and low reproductive rates (Burbidge, 1981; King et al., 1998). Its habitat is also surrounded by highly fragmented landscape, which hinders the tortoises’ dispersal ability. This is significant because the wetland environment which the tortoise inhabits is effected by climate change and is predicted to worsen (expanded upon in Chapter 2, but in short, rainfall is declining and temperatures are rising; (Ali et al., 2012a; Charles et al., 2010; Silberstein et al., 2012)).

The habitat for the WST is freshwater, ephemeral wetlands. When the wetlands are dry (over summer) the tortoises aestivate (undergo a period of dormancy) and when the swamps are full to approximately 20-30 cm of standing water (in winter, spring and early summer) they are able to feed, mate and grow (Burbidge, 1981; Burbidge et al., 2010). While the wetlands hold water, juveniles must reach a critical mass of 18g before their first summer aestivation, and females will reabsorb their eggs after mating or produce fewer or no eggs if their feeding opportunities are limited (Kuchling, 1993; Mitchell et al., 2012). During the aestivation period tortoises must not dessicate, and the nests which contain eggs that are laid annually in soil prior to aestivation (Kuchling, 1993) must also remain sufficiently moist (Mitchell et al., 2016). This species is therefore critically dependent on particular characteristics of its wetland habitat, including the swamp hydro-periods (the period of standing water), inundation area, soil moisture patterns, and temperature. Altered rainfall patterns and increased temperatures in this region are likely to lead to changes in wetland hydrology, which will not only impact the tortoise, but also the whole ecosystem as all fauna and flora species contained within the wetlands will be directly or indirectly dependent on hydrological regimes (Bates et al., 2008; Charles et al., 2010; IPCC 2007 and Barker, 2007). If the wetlands the tortoise inhabits continue to degrade, then AC may be the only option for this species’ long-term persistence in the wild (Dade et al., 2014; Mitchell et al., 2013, 2016).

There are uncertainties about how the current habitat is going to respond to climate change and how long into the future habitats will continue to be suitable. Already the two existing sites are experiencing different population trends (Burbidge et al., 2010). If AC is to take place for the WST, then the complexities of their current wetland habitats needs to be understood, and ways in which their habitat responds to shifting climates should be predicted to support decisions about any potential AC effort. Furthermore, whilst AC is potentially an effective tool, there is no science-based framework to decide on alternate candidate sites that has been tested from a real world translocation perspective. Therefore, in order to facilitate the selection of robust future candidate translocation sites, both for short- and long-term climate forecast horizons, modelling tools are needed that are able to adequately simulate wetland eco-hydrology and interface with WST eco-physiological
1.3 Modelling Wetland Eco-hydrology:

Models of wetland hydrology have been widespread for many decades. The basic mechanism behind these models involves balances of inputs and outputs in the wetland associated with precipitation, evaporation, seepage, solar radiation, wind speed, catchment-inflows, outflows (Colloff and Baldwin, 2010; Vervoort and van der Zee, 2012) and often capable of providing a water balance for a wetland system (e.g. Coletti, 2014; Coletti et al., 2011). Depending on the geomorphic context of the wetland, such models can range from simple lumped models to fully spatially-resolved distributed models, to event-based to continuous models. A lumped wetland hydrological model takes no account of spatial variability in parameters describing the ecosystem, hydrological process, the input data or the boundary conditions but offers advantages of less data requirements and low computational requirements. In contrast to lumped models, spatially-resolved models take into account of the heterogeneity in system attributes (e.g., soil, vegetation), but have a much greater data requirement and are computationally expensive. For this reason, the approach to development of wetland models can vary based on the site specific context such as wetland type, geographical area, degree of nonlinearity, parameter estimation approach, time scales and spatial resolution of input data and simulations (Costanza and Sklar, 1985; Haddeland et al., 2011). An important factor for the choice of model approach relates to the availability of data including the past, present and future time scales for input and validation purposes. In general, hydrologic modellers search for a parsimonious model, which potentially aimed to be a balance between rudimentary models and highly complex modelling systems all models need careful consideration and testing before applying on a real-world complex ecosystem (Park et al., 2014).

Modelling a wetland system undergoing change, such as is the case when they experience a non-stationary climate regime, is further challenging because climate can have important controls on how the interaction between water-soil and vegetation plays out and this must needs to be considered. This interaction can be time dependent (Hunt et al., 1999), and influenced by topographical, geological and climatic factors (Sophocleous, 2002; Winter, 2000). The magnitude of the flux can exert strong controls upon the hydrological regime, nutrient status and species composition (House et al., 2016; Wheeler et al., 2014). Depending on the geo-morphological context, spatial heterogeneity may also emerge (House et al., 2015; Hunt et al., 1996; Johnstone et al., 1973; Lowry et al., 2007), for example in inundation regimes and vegetation assemblage (Coletti et al., 2013a). Therefore, for modelling an ephemeral wetland, the model needs to be complex enough to include seasonal connectivity and its interaction with vegetation biomass, which shapes the soil-water-plant feedback mechanism (Baudena et al., 2008; Ridolfi et al., 2006; Vervoort and van der Zee, 2008). The total water balance of such systems can be mediated by connectivity or disconnectivity with groundwater and interaction with surface water dynamics (Bravo et al., 2002; Krause and Bronstert, 2005) which is common in many coastal plain wetlands.
Chapter 1. Introduction

In case of the Swan Coastal Plain of Western Australia (SCP), the wetlands manifest various types of interactions between the water bodies and groundwater aquifers. The uppermost aquifer on the SCP is an unconfined aquifer which has its upper boundary near 70 m AHD at the top of the regional ground water storage called the Gnangara Mound (Townley et al., 1993). The two study sites of this thesis Ellen Brook Nature Reserve (EBNR) and Twin Swamps Nature Reserve (TSNR) are located very nearby within the same aquifer at less than 4 km distance and at the elevation of 28 m AHD and 32 m AHD respectively (Smith and Allen, 1987; Townley et al., 1993). EBNR sits near the groundwater table at a lower elevation than TSNR. But even sitting within the integrated hydrological system of the SCP, wetlands can be hydrologically disconnected from other surface waters during part or all of the year. The EBNR is an example of such ephemeral wetlands underlain by a confining clay layer (Burbidge et al., 2010) and loses much of its standing water to evapotranspiration without generating overland flow which was found in different parts of the world (House et al., 2016).

Because of the complexity of process interactions in these wetlands, quantifying hydrodynamics (eg. flux movement and water balance) through field observations alone is often impractical (Ali et al., 2012a; Feikema et al., 2007). Comprehensive wetland studies have instead relied on simulation of hydrological processes within fully integrated or coupled groundwater (GW)/surface water (SW) models (Crowe et al., 2004; Frei et al., 2010; Krause and Bronstert, 2005; Refsgaard, 1997; Thompson et al., 2004, 2009). However, these modelling studies often contain simple interpretations of the saturated zone through single-layer lithology (Frei et al., 2010; Refsgaard, 1997; Thompson et al., 2004, 2009) or transfer functions (Krause et al., 2005). Where applied to wetlands with more complex subsurface hydrogeological structures, processes have been partially represented as boundary conditions (Crowe et al., 2004).

This complexity is further magnified by the effect of climate change over recent and future time horizons, that any chosen model needs to reasonably capture. Long-term water balance simulations require accurate prediction of changes in soil moisture (Alfieri et al., 2008), groundwater (Contreras et al., 2008) and vegetation (Bell, 1999) that may be assumed to be static in short-term simulations. Therefore, it is necessary to examine the detail of model parameterization based not only on actual site-specific data, but also considering its ability to capture long-term changes in water stores (Wagener et al., 2001) and vegetation biomass. Parameters such as vegetation and canopy cover, which controls factors, such as evapo-transpiration, shading and soil moisture dynamics needs to be included (Evans et al., 2013; Froend et al., 2013) as they are relevant not only to computation of the water balance, but also to the microhabitats of fauna. Furthermore, the model should be capable to capture the complex effect of temperature that plays a crucial role on the eco-hydrological processes such as vegetation dynamics and on animal growth, evaporation and food availability. In addition to this, engineering structures and management regimes aiming to improve wetland quality can bring further challenges to the development of a realistic model of a wetland system.

A thorough modelling approach to aid AC decision making for wetland dependant fauna must therefore not only assume a basic water balance but also consider ecosystem
processes like vegetation response and heterogeneity, wetland compartmentalisation, the
degree of groundwater connectivity, and shading and water temperature dynamics. At
the same time it must be flexible and able to be applicable with limited data. Currently
no integrated modelling packages exist that are suitable for integration with science-based
decision-making and policy formulation as in AC context. Further, models may need to be
applied at a range of scales, from local to regional, and to have a forecasting capability for
future changes and it also needs to interface with species-specific models for animal growth
to be used as a vital tool for AC purpose. Once such a calibrated and validated version
of the eco-hydrological model is developed, it would be very useful to apply it over the
whole SW Western Australia to pinpoint all the potential locations that meet the habitat
suitability requirements of WST. This screening process can then be repeated and checked
for available wetland types (eg. wetlands connected or disconnected with groundwater)
to ensure that the identified locations for AC will not only be sustainable and suitable at
present but also be suitable under future climates (such as extreme dry or very wet) to
provide viable habitats for the species.

1.4 Research Questions and Aims

This research aims to develop an integrated eco-hydrological model for the prediction of
suitable wetland habitats for the assisted colonisation of the WST in a changing climate. A
model that predict the species’ eco-hydrological requirements, will go a long way towards
strengthening the AC site selection process. The specific aim of this research was to
develop a detailed understanding of the eco-hydrology of critical wetland sites inhabited
by the WST, and to explore how shifting eco-hydrological dynamics under climate change
will impact upon habitat suitability. In so doing, this model can be applied across a grid
of sites in SW Western Australia to perform a screening of suitable wetlands where the
WST could best survive under future climates. This thesis explores some specific research
questions, such as:

1. What are the critical hydrological factors that influence habitat quality for the WST?

2. Are there ecohydrological processes that can explain why the population of WST at
two nearby wetlands systems are different?

3. How sensitive is the water balance of their wetland environments to a change in
climate?

4. How will this translate to habitat availability, and will it reduce significantly under
a drying climate scenario?

5. How can local scale heterogeneity in wetland properties be linked to long term habitat
suitability?

6. If the current preferred habitats are likely to become inadequate under future scen-
arios, are there alternatives sites within the ecoregion where habitat may be more
resilient?
1.5 Research Objectives and Thesis Organization

To answer these questions, this research combines the analysis of long-term data-sets, a collection of eco-hydrological field data in the relevant nature reserves, detailed spatial analysis of habitat condition, and the development of a wetland model suitable for long-term assessments. The model is then applied to unravel the complex eco-hydrological interaction of wetlands that are subject to shifting climate forces, in order to identify potential candidate areas within the SWWA that may be suitable for the AC of WSTs under future climate scenarios.

The thesis is structured around four specific objectives:

1: A comparison of eco-hydrological features of two nearby wetland nature reserves to investigate if eco-hydrological mechanisms could explain differences in WST population trajectories between the two locations.

2: A spatial analysis of hydrology, morphometry, and vegetation of two wetland nature reserves to examine micro-habitat and spatial heterogeneity relevant to the eco-hydrology and habitat of the sites.

3: The development and validation of a parsimonious eco-hydrological model for exploring wetland response to climate change at the system-scale.

4: The employment of an idealised eco-hydrological model as a screening tool to assess how a wetlands are likely to respond to climate change at any location within the SWWA, to identify the suitability of AC candidate sites for the WST.

These objectives map to four main research chapters (Chapters 2, 3, 4 and 5) that are written as self-contained work packages, which are ultimately intended for publication as separate journal articles.

Chapter 2 (Objective 1) focuses on the eco-hydrology of the two adjacent protected wetlands: Ellen Brook Nature Reserve (EBNR) and Twin Swamps Nature Reserve (TSNR), both of which are located within the Swan Coastal Plain of Western Australia. Whilst they are very nearby and experience similar climate and slightly different management policies, the degrees of recruitment success of WST population are different in the two wetlands. This analysis therefore seeks to better understand the eco-hydrological differences of the two systems, and explores the regional context of groundwater-surface water connectivity relevant to the two wetlands. Specific eco-hydrological features of the sites were analysed and compared, such as inundation regimes, hydroperiods, water temperatures, and soil moisture variability. This chapter presents the results of field data collection related to water level, water temperature, soil moisture, and hydroperiod and climate data.

Chapter 3 (Objective 2) extends the work of Chapter 2 by examining the eco-hydrology of the study sites in more detail, using a GIS/remote sensing approach. This chapter undertakes a spatial analysis of EBNR and TSNR data to demonstrate the unique nature of the systems from a hydrological (catchment analysis, and channel profile and structure), morphometry (terrain ruggedness, slope, and micro-relief analysis) and vegetation point
of view. A high resolution (1m) Digital Elevation Model was used for the hydrological and morphometry analyses, and time-series aerial imagery from NearMap™ was used for vegetation analysis. This analysis, combined with the results from Chapter 2, are used to describe an improved conceptual model of the ecohydrology of the two wetland systems.

Chapter 4 (Objective 3) uses the findings of Chapter 2 and Chapter 3, plus the conceptual model of the systems, to develop and validate two numerical eco-hydrological models of EBNR. In order to examine the capacity and accuracy of the model, two model are tested: one simple lumped hydrological model, and another detailed compartmentalized eco-hydrological model which is used to represent the sub-catchments of EBNR, and to link the vegetation and the hydrology of the reserve together. Both models are tested and validated against field data using hydroperiods, soil moisture, and vegetation biomass as collected in Chapter 2. The models are applied under present and future climate scenarios, and the outputs are used to calculate, compare and predict the water budgets of EBNR in present and future climate. The results demonstrate that EBNRs water budget has shifted over the past decade, and is expected to shift further under future climates.

Chapter 5 (Objective 4) outlines the application of the validated and calibrated eco-hydrological model developed in Chapter 4 across the eco-region of SWWA. Two extreme wetland model configurations were used in the screening assessment, comparing the sensitivity of predictions to the level of groundwater connectivity that characterise the systems. The model types were each run across 34 sites randomly distributed throughout the SWWA eco-region, with one present and three future climate scenarios for the year 2030. The results were combined with a thermal reaction norm available for the WST to calculate a habitat suitability index for potential AC candidate sites within the study region.

Finally, Chapter 6 encapsulates the overall findings of the research and discusses the scope and limitation of this study, along with recommendations for future work.
Chapter 2
A tale of two wetlands: the comparative assessment of Ellen Brook Nature Reserve and Twin Swamp Nature Reserve ecohydrology in the Swan Coastal Plain, Western Australia

2.1 Chapter Prologue

The western swamp tortoise (WST, *Pseudemydura umbrina*) is said to have highly specialised habitat requirements which are strongly influenced by hydrological change, and thus climate change (Mitchell et al., 2013). Assisted colonisation has been proposed for this species’ conservation, but for future assisted colonisation sites to be identified it is first necessary to document and understand the characteristics of their current existing wetland habitats.

This chapter aims to examine the fine-scale hydrological characteristics of the wetlands inhabited by the western swamp tortoise, providing a comparison between the two natural reserves which they currently inhabit, and in which there are different population responses. By describing each wetland reserve in context of historical climate, inundation regimes, hydroperiods, water temperature, and soil moisture variability it is possible to create a reference ‘typical’ wetland to be modelled under future climates (in later chapters). This chapter presents the results of field data collection related to water level, water temperature, soil moisture, hydroperiod and climate data, and forms the direct foundation for the construction of the conceptual wetland models provided in Chapters 3 and 4.

2.2 Introduction

Australia has among the highest wetland diversity in the world and a large number of the world’s internationally recognized wetlands, with many of them being declared as RAMSAR sites (Barron et al., 2012; Finlayson et al., 2011). Those in the south-west region of Australia are a key component of Australia’s only recognized biodiversity hotspot (those areas where exceptionally high numbers of endemic species are undergoing exceptional losses of habitat; Meyers et al., 2000, CEPF, 2014) and thus support a high level of species endemism. At the beginning of this century, more than 25% of south-west Western Australia was covered by wetlands but the widespread loss and degradation, particularly on the Swan Coastal Plain (SCP), has continued unabated this century. It is estimated that around 70% of wetlands have disappeared since British settlement in 1829 (Davis and Froend, 1999), due to land drainage practices, filling, agricultural activities, alteration of natural water regimes, introduction of weeds and feral animals, and land-clearing for urban development impacting hydrological behaviour and water quality. Such developments have
directly affected wetland species through the loss of habitat, feeding and breeding grounds, potentially contributing to localized extinction (Barron et al., 2012). Due to the rapid rate of hydro-climatological change predicted for the region (e.g. Silberstein et al., 2012), ecosystem functioning is likely going to be further degraded into the future (Davies, 2010; Horwitz et al., 2008). Thus a clear understanding of basic eco-hydrological function of these systems is required to underpin conservation efforts.

Within the SCP region, there are seven major types of wetland environments (Semeniuk and Semeniuk, 1995) that are each characterized by different geomorphological characteristics and hydrological regimes (Barron et al., 2014; Horwitz et al., 2008). The two that are the focus of this investigation are Groundwater Dependent Ecosystems (GDEs) and Clay-pan Based Ecosystems (CBEs) for TSNR and EBNR, respectively. Of these, GDEs are thought to be a uniquely adapted feature of the SCP landscape (Barron et al., 2012). Whilst both wetland types can occur in close proximity to each other, their characteristics differ greatly. For example, in GDEs, the seasonal oscillation of groundwater has a direct impact on vegetation productivity in addition to surface water extent (Barron et al., 2014), and most vegetation can access the groundwater when it is within approximately 10 m of the land surface (Klove et al., 2011) (i.e. vegetation and ground water levels are closely interrelated). In contrast, CBEs are characterised by a substantial layer of low permeability clay that has accumulated by geomorphological process. Rather than being dependent on the groundwater seepage contribution, the surface water extent and characteristic hydroperiods of CBEs are driven primarily by surface hydrological processes such as rain, and so patterns of inundation are less sensitive to vegetation interactions with the water balance (Hanna et al., 2011). These differences are important because the degree of connectivity of the surface water habitat with the groundwater influences the type, biomass and seasonal variation of the vegetation (Coletti et al., 2014; Semeniuk, 2007), which in turn drives habitat quality for wetland species.

One species that is strongly dependent on seasonal wetlands is the Critically Endangered Western Swamp Tortoise (WST, *Pseudemydura umbrina*). Thought to be extinct before 1953 (Burbidge, 1967; Burbidge and Kuchling, 2004), this freshwater tortoise now has a wild population estimated at less than 50 adults occurring naturally in two locations in the SCP. After its rediscovery in 1953, the Western Australian Government declared two nature reserves at the highest level of protection (category ‘Class A’), requiring permission from Parliament for any changes to its status or for the approval of nearby activities that may impact upon it. Ellen Brook Nature Reserve (EBNR; predominantly a CBE) and Twins Swamps Nature Reserve (TSNR; predominantly a GDE), which are only about four kilometers apart from each other, were chosen in order to protect the last remaining WST habitat where this species was naturally found (Burbidge, 1981). Population monitoring at these reserves has been ongoing since 1953, and has shown that by 1985, the WST population in TSNR was nearly extinct. On the contrary, the neighbouring EBNR population was increasing gradually (Burbidge and Kuchling, 2004; Burbidge et al., 2010; Kuchling, 1993). Given the close proximity of these sites, the same climatic history and exposure, and the same level of protection the reserves are afforded, the question remains why breeding and survival success has been noticeably different in EBNR
and TSNR over the last 50 years. Whilst there are numerous factors that can influence population viability (e.g. feral animals, disease, and wildfire), one unexplored aspect is the role that ecohydrology might play in shaping the habitat and population growth. Because the WST has a specialized biology that demands specific ecohydrological services at different stages of its life-cycle (Mitchell et al., 2013).

It is therefore the aim of this chapter to undertake an ecohydrological comparison of these wetlands systems in the context of the regional shift in climate, to gain insights into how environmental factors may be influencing the population dynamics of the WST. Given the patchy nature of historical WST data sets, it is not possible to definitively connect ecohydrological features with the ultimate success of WST populations. However, it is possible to a) characterise the regional context of groundwater-surface water connectivity and interaction relevant to the two wetlands, b) explore the historical climate change that the sites have been experiencing, and c) compare the ecohydrological features of the sites, including inundation regimes, hydroperiods, water temperature, and soil moisture variability, in order to better understand the local scale differences in WST habitats under historic and current conditions.

2.3 Comparisons of EBNR & TSNR

2.3.1 Situation in the Landscape

Both EBNR and TSNR are located within a three to five kilometer strip of the SCP, parallel with the Darling Scarp (Burbidge and Kuchling, 2004; Burbidge et al., 2010; DEC, 2011; Gorham, 1990).

Soils

One of the most important distinguishers between EBNR and TSNR is soil type.

The soils of EBNR were mapped by Pym (1955) and are composed of partly gilgai and partly claypan components. The gilgai component has microrelief and consists of puffs and sinkholes, and the soil is a dark brown loam over a brown or yellow brown clay which occurs on the surface of the sinkholes. The puffs contain tunnels with openings to the surface, usually at the edge of the sinkholes or at a cave-in (Gorham, 1990). The claypans have similar soils but are flat (Burbidge, 1981) and have a characteristic depth of 40 cm to 1 m. Below the surface clay layer the profile returns to a sandy material that characterises much of the region (Gorham, 1990).

The soils of TSNR were described by Bettanay et.al. (1960) as a pattern of sandy and solenetzic soils, poorly drained and high in soluble salts owing to the presence of an impervious pan at shallow depths. At TSNR low and stable hills of light grey sand surround swamps in the eastern and southern parts, while the north-western area is uniformly low-lying. The swamps and low-lying areas have a sandy soil but with an impervious clay layer at a depth varying from about 15 to 30 cm (Burbidge, 1981).
Chapter 2. A tale of two wetlands: a comparative assessment of EBNR and TSNR ecohydrology

Figure 2.1: Map of the south-western climate region (green shading) and the south-western region of the ocean (indicated by yellow lines) for sea-surface temperature data used in the historical climate analysis. The green inset focuses the regions of the Swan Coastal Plain that was used for regional groundwater analysis and the locations of the study sites (pink and yellow polygons represent TSNR and EBNR, respectively). The circles with cross-marks show the locations of groundwater bores and the black line represents the cross-section of the digital elevation model used to gain insight into the groundwater of the region (further illustrated in Figure 2.2).

Vegetation

The major vegetation assemblages of EBNR and TSNR are similar in terms of species composition (Burbidge 1967). However, there are comparative differences in the overall patterns of the vegetation within the reserves. This is caused by the differences in the slope and soil types of the two sites, as well as the local hydrological regimes. These differences are best exemplified by the presence of deep rooted vegetation that is dominant in TSNR (primarily due to the sandy soil structure), compared to the shallow-rooted bushes and shrubs that characterise EBNR (due to the heavier clay soils).

A list of the flora species common to the two sites can be found in Burbidge (1967), while the vegetation structure in both reserves is documented in Keighery and Trudgen (1992), and Gibson et al. (1994) also describes the floristic communities. In short, the following structural units can be found in EBNR and TSNR (after Keighery and Trudgen (1992)):

Uplands: *Eucalyptus calophylla* open forest to woodland

Wetlands: *E. calophylla* open forest to woodland, *E. rudis* open forest, *Viminaria juncea* and *Acacia saligna* tall open scrub, *Kunzea spp.* and *Verticordia densiflora* open low heath, *Melaleuca laterita* shrub-
land, herbland, Neurachne alopecuroidea open grassland, sedgelands dominated by *Meeboldina cana*, *Chorizandra enodis*, *Cyathocheata avenacea*.

A spatial analysis of vegetation is explored in more detail in Chapter 3.

**Catchment Information**

There are no direct catchment inflows to either of the nature reserves. TSNR is predominantly a groundwater recharge system, which is fed by rainwater in most years. However, when the groundwater table is high enough, the swamps start filling with water via groundwater seepage. In the case of EBNR, the clay layer is connected with the groundwater table (Dansie et al., 2008; DEC, 2011; Gorham, 1990). There is an inflow drain from the west side of the catchment which has been blocked to secure the water quality of the reserve itself from possible chemical or nutrient run-off from the adjacent farms and highways. The outlet is open with a fixed water level maintenance valve that connects it with the adjacent channel, and eventually the overflow of the wetland is carried to the Ellen Brook channel to the south which flows downstream to the Swan river.

The comparative recharge dynamics of EBNR and TSNR have not yet been documented. To examine the recharge dynamics of both reserves, groundwater data from bores within the region of the SCP surrounding these study sites were examined. Different properties of the bores were considered, including the distribution of observed data both in space and time, the nature and confidence in the data used, the availability of independent data to support parametrisation, the proximity to the study sites, and the location and bore depths in relation to the regional groundwater contours. A conceptual model was developed following the framework of Brassington and Younger (2010), using the following considerations:

1. the availability, connectivity and seasonal pattern of the regional groundwater
2. the topography and surface water drainage
3. the geology of the area: geological maps from Geoscience Australia (http://www.ga.gov.au/cedda/maps/1084) were used for the geological information of the area
4. the groundwater contours: groundwater contours were examined to determine the groundwater levels in the region

A high resolution DEM (source: DPaW, 1 m resolution) was used to conceptualise the groundwater status in relation to the topographical variations in the area. Five bores (DoW WIN ID: 5413, 5415, 5419, 5422, 5423 and 5425) were selected (Figures 2.1 and 2.4) to create a cross section along the line of the bore locations.

The seasonal variation of groundwater for each of these five bores was also examined by normalising the data in relation to the minimum water depth of each bore \(GW^* = GW_i - GW_i^{\text{min}}\) (Figure 2.3), in order to observe the similarity in the extent of seasonal water table excursions. This analysis demonstrated a degree of similarity in the bores and confirmed they reflected the regional water table with minimal local differences. This
Chapter 2. A tale of two wetlands: a comparative assessment of EBNR and TSNR ecohydrology

Figure 2.2: Cross section of topography and groundwater along DoW bore-holes. The blue line represents the height (m from Australia Height Datum (AHD)) of groundwater and the extent of the nature reserves are indicated with green lines. The black line is the actual representation of the topography along the cross-section and thick-black lines represents locations and heights of the bores in the same scale.

suggested that the recharge dynamics of the EBNR and TSNR had a relatively minor role in terms of contributing to local mounding.

The regional groundwater fluctuations in the studied bores maintained an overall similar pattern between 1983 and 2013 (Figure 2.3). Bore 5413, located 6 km north of TSNR, showed the maximum variation (about 1 metre; represented in yellow dashed line) between 1988 and 1989. Bore 5423 also had upward pulses in the years 2000, 2001, 2003 and 2007. Bore 2419 located in between TSNR and EBNR showed regular downward pulses and reached a minimum in 2012. However, all the bores showed a downward trend in overall fluctuations from 2006 to 2013, suggesting EBNR and TSNR have a shared groundwater history.

2.3.2 Management Initiatives

Once EBNR and TSNR were declared as nature reserves, a number of in-situ and ex-situ management practises were initiated to increase the numbers of WST in the wild. A highly successful captive breeding program has operated at the Perth Zoo since 1987, and has served to bolster population numbers and provide individuals for restocking and translocation (Kuchling and DeJose 1989). While this ex-situ management initiative has considerably improved the population numbers of WST, there are a number of in-situ management practises that have also contributed to the ongoing conservation of the species. For example, since being declared as Class A nature reserves in 1962, both of the nature reserves have been increased in size through the purchase of adjacent land (the present size of EBNR is 79.7 ha and TSNR is 141 ha (Burbidge et al., 2010)). Both nature reserves have electrified fences along the perimeter to prevent introduced predators (e.g. fox *Vulpes vulpes* and cat *Felis catus*) and humans from unauthorized access, as well as to prevent the tortoises leaving the reserves during spring when the wetlands start to
2.3. Comparisons of EBNR & TSNR

Figure 2.3: Monthly normalized groundwater ($GW^* = GW_i - GW_i^{min}$) frequencies from the five boreholes near the two nature reserves from 1984 to 2011.

dry inside the nature reserve boundaries. Baits containing the poisonous compound 1080 (strychnine; Sodium fluoroacetate; (Armstrong, 2004; Mawson, 2004)) are distributed in both reserves for the control of foxes, and similarly rat baiting stations are also present at both TSNR and EBNR. There are also constructed ponds with artificial lining in both nature reserves to make water available to tortoises towards the end of the wet season as swamps dry.

The above management initiatives operate at both EBNR and TSNR, and are detailed in Burbidge et al. (2014). Further to these, there are also site-specific management initiatives that differ between the two reserves. At EBNR, a weir has been built along the north-west swamp, serving to hold water for longer periods of time towards the end of the wet season. There is also a sluice gate valve built at the main outlet to control the water outflow to the Ellen Brook channel. At TSNR there is no such weir and outflow control structure, but bore water pumping has been in operation since 1994. Revegetation initiatives are undertaken in both nature reserves, but EBNR has more frequent weed control than TSNR. Die-back disease (by *Phytophthora cinnamomi*) severely affects *Banksia* woodlands in TSNR, however vegetation in EBNR is free of this disease. Finally, to maintain the population of tortoises at TSNR after the severe reduction in numbers in 1989, reintroduction of WSTs to TSNR has occurred since 1994; by comparison no reintroductions have occurred at EBNR (Table 2.1).
Figure 2.4: Groundwater contours of the selected area with cross-section of the topography and locations of DoW bores (in black) and locations of the nature reserves (in green). The brown plate connecting the bores shows the line of cross-section used to conceptualise the ground water status.
Table 2.1: Comparison of management and monitoring initiatives in EBNR and TSNR

<table>
<thead>
<tr>
<th>Management Initiatives</th>
<th>EBNR</th>
<th>TSNR</th>
</tr>
</thead>
<tbody>
<tr>
<td>Weir inside the reserves to hold the water for longer period</td>
<td>Constructed</td>
<td>No dams</td>
</tr>
<tr>
<td>Outflow control gate</td>
<td>Present</td>
<td>Absent</td>
</tr>
<tr>
<td>Bore-water pumping before and after the rainy season</td>
<td>No bores</td>
<td>In operation since 1994</td>
</tr>
<tr>
<td>Electrified fences and poison baits for foxes</td>
<td>In practice</td>
<td>In practice</td>
</tr>
<tr>
<td>Rat control measures</td>
<td>In practice since 2005</td>
<td>In practice since 1999</td>
</tr>
<tr>
<td>Construction of artificial ponds with synthetic lining along the bottom</td>
<td>Present</td>
<td>Present</td>
</tr>
<tr>
<td>Providing nesting materials for the tortoises</td>
<td>Ongoing</td>
<td>Ongoing</td>
</tr>
<tr>
<td>Revegetation initiatives</td>
<td>Practiced</td>
<td>Practiced</td>
</tr>
<tr>
<td>Weed control</td>
<td>In practice</td>
<td>Less practiced</td>
</tr>
<tr>
<td>Fire control measures</td>
<td>In practice</td>
<td>In practice</td>
</tr>
<tr>
<td>Reintroduction of WST</td>
<td>Not done</td>
<td>In practice since 1994</td>
</tr>
</tbody>
</table>

2.4 Long-term Climate Analysis

2.4.1 Data Sources and Methods

To determine if the differences in WST population responses at EBNR and TSNR might be due to different climatic histories, a long-term climate analysis was undertaken.

A temperature time-series was calculated from a homogeneous temperature dataset (known as the Australian Climate Observations Reference Network - Surface Air Temperature, or ACORN-SAT, dataset) developed for monitoring climate variability and change in Australia. Average temperature values are provided for southwestern Australia (southwest of the line joining 30°S, 115°E and 35°S, 120°E) (CSIRO and BoM, 2014; Donat et al., 2013a,a; Hope et al., 2006).

Temperature time-series are presented as anomalies or departures from the 1961-1990 average because temperature anomalies tend to be more consistent throughout wide areas than actual temperatures. Normal values (1961-1990 averages) are provided for the selected variable and region. The period 1961 to 1990 is the current international standard period for the calculation of climate normals (CSIRO and BoM, 2014; Donat et al., 2013b).

Sea surface temperature (SST) time-series are available from 1900 for a number of regions around Australia. Mean SST values are provided for a boxed region around Australia and for the southwest region (22°S - 46°S, 94°E - 116°E) used here. The SST time-series are calculated from the NOAA Extended Reconstructed Sea Surface Temperature Version 3 (NOAA_ERSST_V3) data provided by the NOAA/OAR/ESRL PSD, Boulder, Colorado, USA, from their web site at http://www.cdc.noaa.gov/. A full description of the NOAA_ERSST_V3 data can be found in Smith (2004) and Smith et al. (2008).

Evaporation time-series are available from 1975 to 2014. Mean evaporation time-
series are calculated using a high-quality dataset corrected for discontinuities created by changes in instrumentation and location of observation sites. Due to the relatively sparse observation network, spatial averages are calculated using the method of Thiessen polygons (Jovanovic et al., 2007).

Regional rainfall time-series are available from 1900 and the same southwest region used for the temperature dataset was used also for rainfall (Hope et al., 2010; Smith, 2004). Mean rainfall time-series are calculated using a high-resolution gridded dataset developed for the Australian Water Availability Project (AWAP) (Hennessy, 2007; Hope et al., 2006; Jones et al., 2009; Smith, 2004). Statistical trends in annual temperature, rainfall, sea-surface temperature, and pan evaporation were examined using the ARIMA model of ‘forecast’ package in the statistical software R.

To compare the regional average shifts in climate with local conditions experienced at the wetlands, local rainfall and temperature data were analysed from Perth airport (BoM station ID 009021) as it was the nearest station to the study sites that offered the most complete data set that has been continued to date. The annual total rainfall and normalized anomalies (from 1961-1990 average 14 °C) was calculated and checked for long-term trend analysis.

Table 2.2: Summary of trend and the ARIMA model used for the historical climate trend analysis of south-west Western Australia

<table>
<thead>
<tr>
<th>Climate Components</th>
<th>Trenda</th>
<th>ARIMA Model Fit</th>
<th>a²</th>
<th>ACF1b</th>
<th>RMSEc</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Regional Temperature (°C)</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Anomalies of Maximum</td>
<td>0.1</td>
<td>0.263</td>
<td>-0.003</td>
<td>-0.003</td>
<td>0.513</td>
</tr>
<tr>
<td>Anomalies of Minimum</td>
<td>0.1</td>
<td>0.182</td>
<td>-0.095</td>
<td>0.427</td>
<td></td>
</tr>
<tr>
<td>Sea-Surface</td>
<td>0.08</td>
<td>0.019</td>
<td>-0.029</td>
<td>0.139</td>
<td></td>
</tr>
<tr>
<td>Local Temperature (°C)</td>
<td>Max</td>
<td>0.16</td>
<td>0.012</td>
<td>0.032</td>
<td>0.112</td>
</tr>
<tr>
<td>Regional Rainfall (mm)</td>
<td></td>
<td>10351</td>
<td>0.016</td>
<td>101.740</td>
<td></td>
</tr>
<tr>
<td>Anomalies</td>
<td>-11.41</td>
<td>0.016</td>
<td>101.740</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Total</td>
<td>-11.41</td>
<td>0.016</td>
<td>101.740</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Local Rainfall (mm)</td>
<td>Total</td>
<td>11345</td>
<td>0.058</td>
<td>85.239</td>
<td></td>
</tr>
<tr>
<td>Evaporation (mm)</td>
<td>Pan</td>
<td>16614</td>
<td>0.179</td>
<td>128.895</td>
<td></td>
</tr>
</tbody>
</table>

Significant auto-correlation was found in annual temperature, SST, and annual rainfall time series, but not in the pan evaporation time series (Table 2.2). Accordingly, the temperature, SST and evaporation time series were pre-whitened using the trend-free procedure of [Yue et al., 2002] prior to trend and change point analysis. The magnitude of the trend was calculated as the slope of the linear correlation between time and annual rainfall or pre-whitened temperature, SST and evaporation time series (Petrone et al., 2010).

2.4.2 Long-term Climate Trends

The above analysis has shown that consistent with literature (Silberstein et al., 2012), south-western Western Australia is experiencing overall warming, with mean surface air
temperatures warming by 0.9°C since 1910. Daytime maximum and minimum temperatures anomalies have warmed by 0.1°C over every decade (Figure 2.5 a and b).

SSTs in the south-west region have also warmed by 0.9°C since 1900 with a trend of 0.08 °C/decade (Figure 2.5 c). In 2013, temperatures were 0.5°C above the 1961-1990 average of 22.3°C. SSTs around this part of Australia have mostly been well-above average since 2010 (Donat et al., 2013a; Stocker et al., 2013), with persistent regions of very warm to highest-on-record temperatures to the south and west of the continent throughout much of 2013.

In the region of both EBNR and TSNR, rainfall has declined since 1970, dominated by reduced winter rainfall. There has been a 17 per cent decline in average winter rainfall in the southwest of Western Australia. Declining rainfall in the southwest has been statistically significant over the recent period (Jones et al., 2009; Smith, 2004), and has occurred as a series of step changes. The decline in rainfall anomalies and annual rainfall in this region has a negative trend of 11.41 mm in every decade (Figure 2.5 e and f) and has also been characterised by a lack of very wet winters (Trewin, 2013).

Pan evaporation in south-western Australia has shown a steady increase since 1975 to date. It has increased with a trend of 39.96 mm/decade and is associated with increased land and SST (Figure 2.5 d).

Following the regional climate data, local climate data from Perth airport showed similar trends in temperature and rainfall. Local rainfall (Figure 2.5 g) demonstrated a steady decline of total annual rainfall since 1944 to date, with a decrease of 20 mm/decade, and local temperature (Figure 2.5 h) showed an upward trend of 0.16 °C/decade which was more pronounced than the regional temperature trend (by 0.06 °C/decade).
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Figure 2.5: Analysis of long term climate data for south-western Australia. Panel a to f represent the south-western climate region of BoM (shown in Figure 2.1) and panel g and h represents local (Perth airport) data. Panel a, b, c, e and g are normalized anomalies from the standard averages based on a 30 year climatology (1961-90). Panel a and b are maximum and minimum temperature anomalies (°C) for south-western Australia. Panel c is sea-surface temperature anomalies (°C), panel d pan evaporation (mm), panel e rainfall anomalies (mm) and panel f total rainfall (mm) of south-western Australia. Panel g is maximum temperature anomalies (°C) and panel h is total annual rainfall (mm) recorded in local weather station (Perth airport).
2.5 Wetland Echohydrology

2.5.1 Data Availability and Methods

To compare the wetland ecohydrology of the two nature reserves, field data including soil moisture, soil conductivity, water temperature, and hydroperiods were collected at EBNR from June 2010 to March 2013, and from January 2012 to March 2013 at TSNR. A summary of the length of data availability and the frequency of data collection is presented in Table 2.3, while detailed methods of data collection are given in the relevant sections below.

<table>
<thead>
<tr>
<th>Data</th>
<th>Time/ Duration of Data</th>
</tr>
</thead>
<tbody>
<tr>
<td>Water level (manual observation)</td>
<td>1972 to ongoing (monthly)</td>
</tr>
<tr>
<td></td>
<td>1971 to ongoing (monthly)</td>
</tr>
<tr>
<td>Water level (CTD logger)</td>
<td>15/05/2012 to 21/11/2012 (15 min interval)</td>
</tr>
<tr>
<td></td>
<td>15/05/2012 to 15/03/2013 (15 min interval)</td>
</tr>
<tr>
<td>Water Temperature (CTD logger)</td>
<td>15/05/2012 to 21/11/2012 (15 min interval)</td>
</tr>
<tr>
<td></td>
<td>15/05/2012 to 15/03/2013 (15 min interval)</td>
</tr>
<tr>
<td>Soil conductivity (EM 38)</td>
<td>02/06/12 to 29/09/12 (Every 2 m)</td>
</tr>
<tr>
<td></td>
<td>04/06/12 to 30/09/12 (Every 2 m)</td>
</tr>
<tr>
<td>Soil Moisture (MEA t-Bug)</td>
<td>June, 2010 to March, 2013 (15 min interval)</td>
</tr>
<tr>
<td></td>
<td>—</td>
</tr>
<tr>
<td>Weather station</td>
<td>2009 to ongoing (15 min interval)</td>
</tr>
<tr>
<td></td>
<td>—</td>
</tr>
<tr>
<td>Groundwater monitoring</td>
<td>not in operation</td>
</tr>
<tr>
<td></td>
<td>maintained by DoW</td>
</tr>
<tr>
<td>Water quality</td>
<td>1971 ongoing (annually)</td>
</tr>
<tr>
<td></td>
<td>1971 ongoing (annually)</td>
</tr>
</tbody>
</table>

**Table 2.3:** Duration and timing of different field data collected from EBNR and TSNR

*Area-Volume Calculations*

Digital elevation models (DEMs; 1 m resolution, source: Department of Environment and Conservation, 2011) of EBNR and TSNR were cut according to the active catchment areas of the two nature reserves using ArcGIS 10. The XYZ data (length, width and height) of both DEMs were exported to MATLAB and the respective volume of the different heights of the wetlands were calculated as a function of the area of that particular height. This area-volume function was vital to define the shape of the wetlands for the model used for this experiment. This function expressed the slope of the wetland, and provided the
volume of the water and areas of saturated and unsaturated zones as the water levels changed seasonally inside the wetland.

**Catchment Boundary Delineation**

Periphery and the nearby areas of the study sites were carefully inspected to determine the physical boundary of the catchments surrounding the sites. The DEM was then delineated according to the catchment boundary inside ArcGIS 10. These catchments were used to calculate the volumes for the individual sites.

**Soil Moisture Records (MEA)**

Soil moisture data was collected using a MEA t-Bug logger with theta probe (SM20 and SM40). The theta probe is a type of volumetric soil moisture content sensor consisting of a waterproof housing containing electronics with an array of stainless steel electrodes at one end. Two t-Bug moisture loggers were deployed at two spots at EBNR at depths of 10, 20, 30 and 40 cms. The spots chosen were places highly likely to be nesting and aestivation areas within the nature reserve (G. Kuchling, unpublished data). Data was recorded at 15 min intervals and was collected every three weeks using a hand-held MEA retriever (with some breaks in seasons) from July 2010 to present.

**Soil Moisture Records (EM38)**

The EM38 is a widely used example of an electromagnetic instrument for soil sensing developed by Geonics Ltd. (Ontario, Canada). It comprises two electrical coils, one a transmitter (Tx) and the other a receiver (Rx), placed 1 metre apart in a wooden frame (Figure 2.7). An EM pulse is sent from the Tx end and received back at the Rx end. The consecutive reading is displayed in digital meter.

A schematic diagram of the EM-38 is presented in Figure 2.7 showing the location of induced magnetic fields during operation. A transmitting coil (Tx) in one end of the instrument creates a primary magnetic field (Hp). This field creates current loops in the ground below and the current loops induce their own magnetic field (Hi). The induced field is superimposed on the primary field and both Hp and Hi are measured in a receiving coil (Rx) at the other end of the instrument (Hossain et al., 2010; McNeil, 1980). The measured response is a function of ground conductivity, which is linear in the range of soil conductivity of 0 to 10 dS m$^{-1}$. Measurements of ground conductivity can be made with the instrument in either the vertical or horizontal orientation. In the horizontal orientation the instrument measures to a depth of about 0.75 m with the greatest sensitivity just under the instrument. With the instrument in the vertical orientation it measures to a depth of about 1.5 m with the greatest sensitivity at about 0.4 m (Hossain et al., 2010; Misra and Padhi, 2014; Padhi and Misra, 2011; Quinn et al., 2010; Robinson et al., 2004; Rodríguez-Pérez et al., 2011; Tromp-van Meerveld and McDonnell, 2009).

In addition to the soil moisture recordings made by the MEA loggers, vertical ($EM_H$) and horizontal ($EM_V$) conductivity data was recorded with the EM38 over the spots where the MEA moisture sensors were located at stations 1 and 2 (Figure 2.6). Soil
moistures at 10 cm and 40 cm were considered to be the closest to the surface (to use with $EM_H$) and furtherest from the surface (to use with $EM_V$) respectively. There was a significant linear relationship between the soil moisture reading (%) from the theta probes and conductivity readings (mS/m), as given in Figure 2.8. The green line with green diamond-dots represents the relationship of the 40 cm soil moisture logger at station 1 against the $EM_V$ reading. Table 2.4 summarises the model fit results of this relationship showing the most significant relationship ($St_{140cm} - EM_V$) with $R^2 = 0.999$ and p value 0.014. This linear equation was used to convert EM38 conductivity data into soil moisture.

The EM38 was used at both EBNR and TSNR from June 2012 to December 2012, and was connected with a high precision (1 cm horizontal accuracy) differential GPS (EPOCH 35 GNSS System from Spectra Precision, USA). The EM38 surveys were undertaken at three distinct times: when the swamps were completely dry, when the soils were moderately wet (after a few rainfalls during the beginning of the wet season), and when the soils were almost fully saturated (just before the swamps started filling up). Similar paths were followed on each survey event in each nature reserve. The full area inside the perimeter fence at EBNR was surveyed, while partial areas (covering two functional ponding areas) of TSNR were surveyed. Conductivity values from EM38 were first converted to soil moisture.
using equation \( \text{Soil Moisture} = 0.0430 \times EM38Value + 35.056 \) from Table 2.4, and then interpolated inside ArcMap 10.2.1 using the ordinary Kriging interpolation algorithm.

**Water level and Temperatures**

Two conductivity-temperature-depth loggers (CTD-Diver DI271, Schlumberger Water Services, Netherlands) were deployed in two TSNR swamps and one EBNR swamp, with a data logging interval of 15 mins. One of the CTD-Diver data loggers in TSNR (south-west swamp) was associated with a barometric pressure logger (Baro-Diver DI500, Schlumberger Water Services, Netherlands) and was set to log at 15 min intervals. This was fixed 1 metre from ground level and was used to calculate the water level in relation to the vertical reference datum of the Baro-Diver. Data was collected at the end of the 2012 wet season when the swamps had completely dried.

**Calculation of Hydroperiods**

Hydroperiods were calculated using historical data measured by eight depth gauges in EBNR and four depth gauges in TSNR (Data courtesy G. Kuchling, DPaw). A probability density function was calculated using the presence and height of the visible surface water in different pools.

### 2.5.2 Hypsographic Curves

The hypsographic curves of EBNR (Figure 2.9 black line) and TSNR (Figure 2.9 blue line) both increased with height but the rate of increase in volume was considerably different. At EBNR, volume increased at a constant rate until a height of 3.5 m where it reached a volume of \( 2 \times 10^{15} \text{ mm}^3 \) and then the volume stayed unchanged for the remainder of the height of EBNR. This indicates there were few areas included between the 3.5 m and 7 m heights. On the other hand, in TSNR, volume increased with height at a much lower rate than at EBNR. The volume reached \( 5 \times 10^{15} \text{ mm}^3 \) by a height of 3 m, and then increased...
2.5. Wetland Echohydrology

Figure 2.8: Correlation between soil conductivity (from EM38) and soil moisture (from Theta probe) from three dates of EM38 surveys at EBNR. A line was fitted for EM38 horizontal readings with 10 cm theta moisture probe values at station 1 and station 2 (St1rcm - EMH and St2rcm - EMH) and EM38 vertical readings with 40 cm theta probe readings at station 1 and station 2 (St1dcm - EMV and St2dcm - EMV).

at a much slower rate until around 4 m, after which the volume became constant to 7.3 m height. This constant volume between 5 m and 7.3 m height relates to a very small area (about 5%) of the total nature reserve.

Figure 2.9: Height-volume (panel a.) and height-area (panel b.) curves of EBNR (black line) and TSNR (blue line).

2.5.3 Water Level and Ponding Extent

EBNR became saturated faster than TSNR (Figure 2.10 top row). Once saturated, the water level at EBNR rose very quickly (to above 35 cm within around 15 days, and increased again to 38-40 cm after dropping in August to 23 cm). The water level stayed
above 30 cm until the first week of October, after which water levels began receding.

At TSNR there was no steady, long-term standing water. Instead water levels at TSNR fluctuated greatly over the wet season, with water frequently pooling and then quickly disappearing. Twice (mid-September and early October 2012) the water level reached above 30 cms.

The overall ponding extent lasted nine months in EBNR and less than six months in TSNR.

![Figure 2.10: Water level and water temperature from EBNR and TSNR from CTD Diver sensor](image)

### 2.5.4 Hydroperiod

The normalised frequency of hydroperiods at EBNR and TSNR were considerably different from each other. The red solid line in Figure 2.11 demonstrates that around 42 per cent of hydroperiods at EBNR had a duration of six months. The maximum hydroperiod of EBNR was up to nine months, and there were very few events where hydroperiods only lasted two or three months. Most of the hydroperiods in EBNR were between four to seven months, and there was a sudden decline in the hydroperiod distribution after six months.

In contrast, at TSNR (represented by black dashed line in Figure 2.11), most hydroperiods (around 40 per cent) were just over four months. The maximum hydroperiod length was around seven months, and the minimum was less than one month. The majority of hydroperiods fell between three and six months.

Additionally, the long term trends in hydroperiod were explored in context of the drying climate trend summarised in section 2.3.2. Figure 2.12 a) represents rainfall (mm) from a local weather station in cumulative deviation from mean (CDFM) which is an effective way of representing rainfall trends in order to examine long term patterns (Emelyanova
2.5. Wetland Echohydrology

et al., 2013; Yesertener, 2005). The CDFM clearly shows a peak wet period from 1944 to 1968, immediately followed by a drying period with a downwards trend that extends until recently (2011).

2.5.5 Water Temperature

Both reserves showed similar overall seasonal trends, with water temperatures mostly encompassing the preferred activity temperature range of the WST (14 to 28°C; (Mitchell et al., 2013) ), particularly from August when temperatures started warming. Fluctuations in water temperature were greater for EBNR than TSNR, but the reverse was seen for surface water temperature. The differences in water temperature were likely due to the pumping of bore water at TSNR versus the surface water ponding of EBNR, whereby the pumping of bore water moderated water temperatures (at EBNR water temperatures shifted more frequently and there was a wide range in short-term water temperature fluctuations, while in TSNR, the temperature fluctuations were less sharp, with temperature buffering best coinciding with the June/July bore water pumping). For surface water temperatures, EBNR had far less fluctuations than TSNR, likely due to the volume of standing water available.

Water temperatures of both EBNR and TSNR showed a linear relationship with air temperature (Figure 2.13). A stronger linear relationship was seen at TSNR ($R^2=0.763$, $p < 2.2e^{-16}$) than at EBNR ($R^2=0.647$, $p < 2e^{-16}$). In EBNR, the variance (in square root) was larger towards higher temperatures (from 15°C to 25°C) and the variance became
Chapter 2. A tale of two wetlands: a comparative assessment of EBNR and TSNR ecohydrology

Figure 2.12: Panel a) illustrates rainfall BoM weather station (Perth airport) represented as cumulative deviation from mean (CDFM (mm)) and panel b) illustrates the long-term (1972 to 2012) hydroperiods of EBNR (black solid line) with its trend in blue solid line and TSNR (red dashed line) with its trend in red solid line

smaller below 15°C. In contrast, the variance in TSNR was similar between 20° to 30°C but tended to be smaller below 20 °C. The distribution of water and air temperature data in EBNR had outliers, with more outliers in air temperature than water temperature which may have caused the larger variances seen in EBNR. By comparison, water temperature had no outliers and air temperature had few outliers at TSNR.

2.5.6 Soil Moisture

The interpolated soil moisture maps from EBNR and TSNR showed clear variation in terms of the pattern and percentage of soil moisture (Figure 2.15). Soil moisture at EBNR was overall higher than TSNR throughout the season. EBNR sustained more moisture (about 51%) after the second big rainfall event around the middle of June 2012, whereas TSNR reacted more slowly in terms of moisture holding capacity (not exceeding 41%) in response to the same rainfall event. Most of the moisture was retained near the swamps in both EBNR and TSNR, and the volume of moisture compounded as swamps started filling up with consecutive rainfall events during the wet season.

The height frequency data (every 15 min) in EBNR did not exceed 60% at any time. The 30 cm logger showed the maximal amount of soil moisture (60%) through the wet season, whilst the other three loggers showed similar types of seasonal variation (30% to 50%) between 2010 to 2014, at both stations 1 and 2.
2.5. Wetland Echohydrology

Figure 2.13: Scatterplots showing relationship between air temperature and water temperatures of EBNR (panel a.) and TSNR (panel b.). The red lines represent the regression line and box plots next to x and y axis represent the distribution of temperatures of air and water respectively. The grey dashed line on both sides of the regression lines shows the variance in square root.

Figure 2.14: High resolution (15 minutes interval) soil moisture data from EBNR (station 1 indicated by a) and station 2 by b); location of station 1 and station 2 are shown in Figure 2.6
Chapter 2. A tale of two wetlands: a comparative assessment of EBNR and TSNR ecohydrology

Figure 2.15: Soil moisture maps of EBNR (panel b.) and TSNR (panel c.) on three different dates from EM38 surveys. Panel a. shows rainfall (in black solid lines), temperature (in green solid line) and relative humidity (in blue dashed line) from EBNR weather station during the period of EM38 moisture surveys. The blue arrows represents the survey dates in EBNR and the green arrows represents the survey events in TSNR.
2.6 Discussion

2.6.1 Site-level Differences

While many aspects of EBNR and TSNR are similar, this study has shown that the ecohydrological features of the two reserves are different. One of the most important habitat factors thought to influence WST population success is the ponding (hydroperiod) capacity of the reserves (Dade et al. 2014). Comparisons of the two reserves examined here show that TSNR has typically shorter and less thermally stable hydroperiods than EBNR. Thermal stability is linked to vegetation type (shade provision) and the volume of water held in the swamps, which in turn are both influenced by soil type. TSNRs sandy soils are less likely to retain moisture, and so the extent of local ponding fluctuates greatly. In contrast, the clay pan system of EBNR has a much greater capacity for holding water. Once the clay layer is saturated (generally after a period of heavy rainfall; Burbidge 1981), water accumulates and remains pooled, resulting in longer hydroperiods (around 6 months during the wet season) with less seepage. The presence of a thicker layer (30 to 60 cm) of sand on topsoil in TSNR provides more hydraulic conductivity (5-20 m/day) than in EBNR with its clay or shallow sand over clay soils (0.01-0.2 m/day and 0.1-1 m/day respectively (Moore, 2001)), explaining why the water retention capacity differs between each site.

The vegetation dynamics of EBNR and TSNR also vary, mainly because of differences in the surface water to groundwater interactions, and soil structure. The maximum soil moisture was measured at 30 cm depth at EBNR by the soil moisture logger most likely because of the abundance of shallow rooted vegetation like Banksia around those locations. Dodd and Bell (1993a,b); Zencich et al. (2002) found similar soil moisture patterns in Swan Coastal Plain in Banksia woodlands. Rooting depth can be a deterministic factor for the amount of soil moisture in semi-arid landscapes (Groom, 2004). TSNR has more groundwater-dependent rooted vegetation than EBNR because the groundwater is within reach of the vegetation (<1 to 2 meter depth). Conversely, in EBNR, the average depth of groundwater is 5 to 6 m. This results more shallow rooted shrubs and bushy vegetation, which provides greater shade coverage. This also explains why the soil moisture content at TSNR is lower than that of EBNR. TSNR has more areas exposed to direct sunlight with less shading from the vegetation, in turn causing greater evaporation and lower soil moisture content. This is most apparent at the end of the wet season. The soil moisture patterns recorded using EM38 in EBNR and TSNR clearly show that soil moisture in TSNR is much lower than that at EBNR during the wet season (maximum of 43%, compared to 55% at EBNR; measured during June and September). Although the EM38 is a indirect measurement of soil moisture, by combining it with the direct measurements of soil moisture probes (MEA t-Bug) it was possible to create spatial snapshots by calibrating the EM38 field measurements against the actual soil moistures recorded by the soil moisture probes (meaning soil moisture estimates could be generated for the entire EBNR and part of TSNR without directly needing to place soil moisture loggers across entire reserves). Future work should use this approach to examine the seasonal variation of soil moisture, and should also aim to describe how this is influenced by soil types and depths.
Additionally, a detailed high resolution analysis of vegetation should be undertaken to determine how vegetation acts as a factor for surface water temperatures.

### 2.6.2 Site-level Similarities

The most significant similarity between EBNR and TSNR was their shared history for a worsening climate. The local analyses provided here demonstrate a clear declining trend for rainfall, plus an increasing trend for temperature, SST, and pan evaporation in the south-west Western Australia where both reserves occur. For this region, maximum and minimum temperature had a trend of +0.1 °C/decade and the trend was +0.8 °C/decade for SST. In conjunction with these increasing temperatures, rainfall steadily decreased across this century. Pan-evaporation also increased with increasing temperature, and this is accompanied by even greater declines in annual stream-flow for the region (Kinal and Stoneman, 2012). This combination of increased temperature, declining rainfall, and high evaporation rates affected the wetlands by shortening the hydroperiods in both reserves, and also reduced the water available for vegetation dynamics. These results are consistent with a large and growing body of literature (for example, Kinal and Stoneman (2012); McFarlane et al. (2012); Petrone et al. (2010); Raisin et al. (1999) and Hughes et al. (2012)) which have shown that groundwater and stream-flow in south-west Western Australia has steadily declined, affecting many aspects of wetlands, such as groundwater storage, rainfall-runoff relationships, vegetation dynamics, and habitat quality. Building on this history are simulations predicting that the climate for south-western Western Australia is likely to continue to worsen, with a hotter and drier climatic future anticipated (Blois et al., 2013; Klove et al., 2013; Silberstein et al., 2012) which will continue to affect both GDEs and CBEs such as TSNR and EBNR.

### 2.6.3 Implications for the WST

Myers et al. (2000) defined the south-western corner of Australia as an area where "exceptional concentrations of endemic species are undergoing exceptional loss of habitat" and Horwitz et al. (2008) further argued that hydrological changes can escalate ecosystem stress, placing threatened species at further risk. Specifically, human-driven and/or climatically-driven hydrological change directly or indirectly manifests the threats of complete extinction of at-risk species, such a the WST which is Critically Endangered (IUCN, 2016).

Both EBNR and TSNR have shown hydrological change over time. Changes to wetland water temperature and hydroperiod length directly impact WSTs by influencing their capacity to feed and reproduce. For example, the WST needs a minimum water depth of 40 cm in the wet season (around six months) to actively feed and reproduce. Additionally, even if 40 cm of standing water is present, if the water temperature is too cold or too hot to support WST activity (e.g. the WST can not maintain body temperatures between 14-28 °C) then the tortoise will stop feeding (Lucas et al., 1963). This is significant because WST juveniles must feed actively and gain enough body mass (18 g) during the wet season to be able to survive their first summer aestivation (Mitchell et al. 2013). Further, there
is evidence that females do not produce eggs in low-rainfall conditions, and two successive years of good rainfall is required for recruitment to take place (Burbidge, 1967, Kuchling unpublished). Both the presence and temperature of water also drives when WSTs will commence their aestivation, as when the swamps are nearly dry and water temperatures rise above 28°C, the tortoises will leave the water to aestivate (Burbidge 1981). Finally, the success of hatching in wild nests can be affected by excessive summer heat or localised flooding. All these responses of the WST depend on the small, fine scale parameters of the wetlands such as hydroperiod length, water temperature, vegetation composition, and soil moisture.

The site-specific data collected here suggests that EBNR is a more stable wetland for WSTs than TSNR. At TSNR, the water level rarely reaches or maintains a level of 40 cm, despite the pumping of bore water. The hydroperiods at TSNR are shorter and earlier than those at EBNR, and the soil moisture in TSNR is less than that of EBNR, due to soil type and structure. This can affect breeding and aestivation success because tortoises must dig deeper into the sandy soils of TSNR to access sufficient soil moisture. Because soil moisture is higher at EBNR, and the capacity of clay to retain moisture is high, tortoises at EBNR are better protected from dessication. Nesting in loose sandy soil also makes the tortoise more vulnerable to predation (Burbidge, 1967, 1981; Burbidge and Kuchling, 2004; Burbidge et al., 2010).

From a temperature perspective, because of the bore water pumping, water temperatures in TSNR have less diurnal fluctuations and so provide more stable water temperatures during the day. There was a linear relationship between water temperature and air temperature at both sites, but there was a lower significance of this relationship in EBNR. This lower significance can be explained by presence of shrubby vegetation which provides more shade, reducing the influence of sunlight on water temperature, which is likely to be important under increased climate change. Site-specific linear relationships of air-water temperature will be useful for calculating water temperature as function of air temperature data which are more readily available (obtainable from BoM and DoW) than direct water temperature measurements.

This study illustrates how neighbouring wetland ecosystems can respond quite differently despite sharing the same broad climate histories and locations in space. Both EBNR and TSNR have experienced the same climate history, and also the same pattern of rainfall decline and increasing temperature that has been reported in the region (Dade et al., 2014; McFarlane et al., 2012; Silberstein et al., 2012). But the differences in their geomorphology and vegetation has led to the different hydroperiod and temperature responses recorded at these sites. For the WST, this has meant that populations at these reserves have experienced different local conditions over time, despite the two reserves occurring in close proximity to each other. Because threatening processes (e.g. predation pressures) have been controlled in much the same way in both reserves, it is possible that the two different population responses (historical declines at TSNR and stability at EBNR) could be explained by ecohydological factors over the last 25 years, particularly as critically both hydroperiod length and water temperatures have differed between both reserves. The next challenge will be to predict how these wetlands might respond to changes in fu-
ture climate, and how these changes might influence WST population responses. Because the local differences in soil and vegetation at TSNR and EBN have such an influence on wetland hydroperiods and temperatures (and thus, WSTs), a detailed spatial analysis of the micro-habitats with terrain complexity, within-system hydrological flow patterns, and vegetation dynamics should form the next step to achieve this.
Chapter 3
Geospatial analyses of terrain and vegetation in the Ellen Brook and Twin Swamps nature reserves

3.1 Chapter Prologue

Chapter 2 demonstrated that despite being in close proximity to each other, the two wetlands critically important for western swamp tortoise (WST, *Pseudemydura umbrina*) survival (Ellen Brook Nature Reserve, EBNR and Twin Swamps Nature Reserve, TSNR) had many differences in terms of their vegetation, soil, and hydrological characteristics. This chapter expands on this by examining the fine-scale structure of each wetland in order to predict how these wetlands might respond to novel future climates.

The mechanistic underpinnings that are important to quantify here are primarily vegetation and water-flow dynamics. Therefore, this chapter focuses on the unique hydrology, morphometry and vegetation characteristics of EBNR and TSNR. Specifically, the objectives of this chapter are to:

1. Demonstrate the potential hydrological features of EBNR and TSNR through spatial analysis, and to highlight implications for ecological and hydrological services.

2. Analyse the morphometry of EBNR and TSNR through application of spatial techniques, and describe its implication for the ecohydrology of the systems.

3. Quantify vegetation coverage and variability of EBNR and TSNR using remote sensing techniques, and to identify its potential for supporting WST management and as a data source for model validation.

Overall, the outcomes of the above analyses are closely related with the field data collection in the previous chapter, and together they are used to develop an improved conceptual model of the wetland systems which is presented at the end of this chapter. The data is also critical for the validation of the hydrological model presented in Chapter 4.

3.2 Introduction

One of the important hypotheses of ecology is that structurally complex habitats may provide more niches and diverse opportunities for biota to exploit environmental resources in than homogenous habitats, thus increasing overall species diversity (Bazzaz, 1975; Tews et al., 2004) and ecosystem resilience (van Nes and Scheffer, 2005). Species richness and abundance have often been considered functions of variation in local resource availability,
vegetation structure, and the size of a given habitat patch (Saab, 1999). Habitat variation exists on many scales, and ecologists have become increasingly aware of the importance of examining ecological processes at spatial and temporal scales relevant to both the organisms and processes under study (Wiens, 1989). As wetlands can offer highly diverse habitat assemblages with strong gradients in environmental conditions, the importance of studying them becomes increasingly important for understanding how wetland spatial heterogeneity might help to buffer the impacts of threatening processes such as climate change.

Measuring spatial heterogeneity in wetlands is challenging, but can be aided by Geographical Information Systems (GIS) which has become increasingly popular for the integrated development and application of environmental management planning and ecosystem assessments. GIS has a unique capability of integrating wide arrays of information (such as spatial analysis and remote sensing) and is able to provide useful baseline information on numerous factors such as soils, land use, vegetation, surface and groundwater, geology, landforms, topography, and settlements, in both micro and macro scale perspectives (Barron et al., 2012; Dade et al., 2014; Hopkinson et al., 2009; Parker et al., 2010; Uniyal and Gupta, 2013). With the growing availability of high-resolution terrain and land cover image data that can be used to provide hydrological and water resource information (Pietroniro and Leconte, 2005), GIS modelling approaches are becoming increasingly popular for hydrological delineation tasks.

One of the most important data sources for hydrological delineation tasks is the digital elevation model (DEM). DEMs enable the direct extraction of watershed attributes, such as channel network topology, channel storages and watershed extent, while also providing critical data on terrain attributes that control the distribution of hydro-meteorological inputs (such as radiant energy, precipitation, and sensible heat)(Zhang et al., 2014). DEM data allows the user to model hydrological inputs, transfers, and outputs both vertically and horizontally. Digital morphometry also allows the quantification of topographic features, thus enabling the objective comparison of different segments of the Earth’s surface (eg. Pike and Thomas, 1998).

DEMs are widely applied for the recognition and genetic classification of landforms (Scott and Pinter, 2003), surface process modelling, and geomorphology (eg. Ganas et al., 2005; Jordán et al., 2005; Molin et al., 2004; Zuchiewicz, 1991), but are limited in their applications towards vegetation. For this reason, remote sensing is often coupled with DEM use(Samanta et al., 2012). Spatial sampling techniques can be integrated with remote sensing to effectively quantify the type and amount of vegetation within an area, and this approach is widely used for land resource surveys (particularly in the quantification of forest cover)(Samanta et al., 2010). Manual monitoring is costly, time consuming and sometimes not possible at all; hence, it is important that more efficient means of data collection be developed and utilised. In this regard, the utility of remote sensing technology in obtaining reliable, timely vegetation related information quickly, is well recognised for land character assessments.

Spatial analyses and remote sensing has many advantages for inventoring and monitoring wetlands (Ozesmi and Bauer, 2002). Uses of this technology for wetlands included
but not limited to wetland crop management (Metternicht, 2003), mapping riparian (Congalton et al., 2002; Harvey and Hill, 2010) and coastal vegetation (Caccetta et al., 2012), predicting vernal pool locations (Cormier et al., 2013), quantitative assessment of wetland vegetation such as for structural attributes (Fensham et al., 2002), stand biomass estimation (Suganuma et al., 2006), estimation of leaf area index (LAI) (Cohen et al., 2006), gross and net primary production (GPP and NPP) (Turner et al., 2006, 2003, 2005) and more recently for mapping groundwater-dependent ecosystem (GDE) through vegetation and moisture dynamics (Barron et al., 2014). Vegetation structure, assemblage and functional group distribution in wetlands can play important roles in eco-hydrological feedback mechanisms (Humphries et al., 2011) and both satellite (with higher spectral bands) or aerial imageries can be used for this purpose. However, satellite imagery also has limitations compared to aerial photography. Because of the maximum spatial resolution of most satellite imagery (20–30m), it is difficult to identify small or long, narrow wetlands (Ozesmi and Bauer, 2002), which is the case of this study for Ellen Brook Nature Reserve (EBNR) and Twin Swamps Nature Reserves (TSNR). Aerial photograph can be available in much higher temporal (24 to 48 hours) and spatial (> 5 cm) resolution (e.g. NearMap™ in Australia) and are often cheaper compared to satellite imagery. In case of small study sites, it is often difficult to separate different vegetation types from one another in satellite imagery because of the overlap in their spectral signatures.

Here these spatial and remote-sensing techniques are applied to two important wetlands in Western Australia; the wetlands of EBNR and TSNR, which are both threatened ecological communities and which house the Critically Endangered Western Swamp Tortoise (*Pseudemydura umbrina*; Burbidge et al. 2010). It is thought that these wetlands will change with escalating climate change (Burbidge et al. 2010) but how they will shift is unknown. Therefore, this study aims to derive the watershed, flow routing and hydrogeological parameters of the wetlands (by applying the use of DEMs), as well as describe the vegetation (through the use of remote sensing of high resolution aerial photograph), to be able to devise a simple conceptual model for these wetlands, and to support future hydrological model simulations under climate change.
Figure 3.1: 3D-view of Digital Elevation Model (DEM) in m AHD of the catchment areas of EBNR and TSNR
3.3 Methods

3.3.1 Hydrological Analysis Using High Resolution DEMs

LiDAR data is the most popular source for DEM generation due to its precision and availability in different resolutions (1 m, 2 m, 10 m and 100 m) relevant to the purpose and extent of a study. One benefit of LiDAR over other more traditional terrain data sources that are used for the derivation of channel networks is that it can be used to generate DEMs at resolutions approaching or better than 1 m point spacing. LiDAR therefore offers the potential to identify zero- (Tsukamoto et al., 1982) and first-order channel features within wetland micro-topography that control runoff generation and flow routing processes in relatively flat environments. Further, it has also been used to map high resolution landscape depressions of hydrological importance beneath tree canopies (e.g., (Lindsay, 2005), where traditional stereo photogrammetric methods tend to be inaccurate due to canopy shadowing (Hopkinson et al., 2009). This was particularly necessary for accurate definition of the DEMs for EBNR and TSNR since they are densely covered with woodland tree and small bush/shrub canopies.

A LiDAR-derived high resolution (1 m) DEM was therefore developed for hydrological feature delineation, based on data obtained through the Department of Parks and Wildlife (DPaW), Western Australia. The DEMs were clipped with the catchment boundaries for both EBNR and TSNR (Figure 3.1a). Ground-truthing was performed using observed elevation readings from a RTK differential geographical positioning system (EPOCH 35 GNSS System from Spectra Precision, USA). To ensure the hydrological analyses were as widely applicable and transferable as possible, all hydrological operations were carried out in Arc Hydro (Maidment, 2002) Tools 2.0 (latest version October, 2012) within the Arc GIS suite (ESRI, 2014).

The following hydrological features were assessed and compared:

(a) watershed extent and areas,

(b) cross-sections of the DEMs and its comparison based on the cross-section data, and

(c) stream channel network (zero-order and first-order) delineation and main channel profile analysis.

A flow-chart of the main workflow of the Arc Hydro tools applied to the DEMs are illustrated in Figure 3.2. At first, presence of possible sinks (a faulty grid cell that is lower than all the grid cells around it in the DEM that prevents the flow direction in the middle of nowhere; usually created during the interpolation of original contours from the LiDAR image) were checked and filled using the 'flow direction', 'sink' and 'fill' tools in Arc Hydro to create a depressionless DEM. A flow accumulation raster was created from the depressionless DEM and this was used for creating stream order (only zero and first order streams were selected) and stream features using the 'flow accumulation' and 'stream to feature' tools. Pour points were created based on the field survey points of actual drainage outlets of the catchments of EBNR and TSNR. The 'snap pour point' tool was applied on the flow accumulation raster created in the previous step to create
watersheds that contribute to the channels. For cross-section and stream profile analysis, the '3D analysis' tool was used in Arc Map and Arc Scene.

Figure 3.2: Flowchart showing the hydrological modelling steps taken in ArcGIS (ESRI, 2014)

For analysis of inundation patterns or ponding extent, DEMs of EBNR and TSNR were examined in ArcMap v 10.2.2 using the 'raster calculator' to calculate the extent of inundation at different depths. Ponding extents at 0 cm, 10 cm, 30 cm, 50 cm, 70 cm and 90 cm depths were calculated for EBNR. TSNR ponding extents were calculated for 0 cm, 30 cm, 58 cm, 72 cm, 84 cm and 115 cm depths, and the extent areas (in log units) were plotted as a probability distribution function in R to compare the ponding pattern at different depths.

3.3.2 Morphometric Analysis

In case of similar and closely located study sites which might have similar landscape characteristics, it is often reasonable to try few spatial algorithms for morphometric analysis. This way any misleading error originated from calculation methods can be identified and rectified. Here, six of the most commonly used indices was used for calculating and comparing terrain ruggedness indices of EBNR and TSNR.

Relative Topographic Position (RTP)

Relative topographic position (RTP; also called the topographic position index) is a terrain ruggedness metric and a local elevation index (Jenness, 2004). The topographic position of each pixel is identified with respect to its local neighbourhood, and thus in respect of its relative position. A neighbourhood roving window considers all the cells specified by the window size (e.g. a $3 \times 3$ window is comprised of the surrounding eight cells) and calculates the statistics (e.g. mean, max, range, etc.) as specified by the user for the cell being processed, as illustrated in Figure 3.3. This index is useful for identifying landscape patterns and boundaries that may correspond with dominant geomorphic processes, soil characteristics, vegetation, or drainage features. The final output raster was classified into high, medium, and low based on natural breaks or breaks that are meaningful for the output raster. Natural break classes are based on natural groupings inherent in the data. Class breaks are identified using best group similar values that maximize the differences
between classes. The features are divided into classes whose boundaries are set where there are relatively big differences in the data values. One of the most commonly used natural breaks are 'Jenks natural breaks' which are data-specific classifications and can be useful for comparing, describing and displaying multiple maps built from similar or nearly-similar underlying information.

In the following subsections, six different types of RTP analyses were conducted which are described in detail under the subsequent headings.

**Figure 3.3:** Illustration of how focal neighbourhood statistics operate with a roving window

### Standard Deviation of Elevation

Standard deviation of elevation is a measure of topographic roughness (Ascione et al., 2008). A mean elevation raster and a range elevation raster were created from the main catchment DEMs of EBNR and TSNR using the 'focal statistics' tool in ArcMap 10.2.1 with a neighbourhood of a $3 \times 3$ cell rectangular roving window using mean and range statistics. The final output raster was created using the 'raster calculator' in ArcMap 10.2.1 applying equation 3.1. Jenks natural breaks with three classes were used to display the resulting raster.

\[
SD \text{ of Elevation} = \frac{MeanDEM - DEM}{RangeDEM} \tag{3.1}
\]

where,

- $meanDEM$ = mean elevation raster,
- $rangeDEM$ = raster containing the range of elevation values and
- $DEM$ = the original elevation raster

### Slope Variability

Slope variability is a measure of the "relief of slope" of a landscape. It is best calculated using a slope raster and a roving window relative to the size of the full DEM (Ruszkiczay-Rüdiger et al., 2009). A slope raster was created in ArcMap using the 'slope' tool, and this slope raster was used to calculate the maximum and minimum rasters using the 'focal statistics' tool with a $5 \times 5$ cell rectangular roving window with max and min
statistics selected respectively. The final slope variability raster was created using the 'raster calculator' in ArcMap, applying equation 3.2 with three classes of Jenks natural breaks used to display the resulting raster.

\[ SV = S_{max} - S_{min} \]  

(3.2)

where,

* SV = slope variability output raster to be created,
* S\(_{\text{max}}\) = maximum slope value of the raster in the focal neighbourhood and
* S\(_{\text{min}}\) = minimum slope value raster in the focal neighbourhood

**Riley Terrain Ruggedness Index (TRI)**

Terrain Ruggedness Index (TRI) is the difference between the value of a cell and the mean of the specified cell neighbourhood of the surrounding cells. Firstly, the two input neighbourhood rasters were created from the original catchment DEMs using a 3 \(\times\) 3 cell neighbourhood roving rectangular window with the statistics min and max respectively. The resulting minimum and maximum rasters were then used for calculating the Riley TRI raster, applying equation 3.3 in the raster calculator. The resulting ruggedness index values were then classified using the categories of Riley et al. (1999) and Pareta and Pareta (2011). Riley et al. (1999) noted that a particular DEM may not yield the full range of values possible and as a result, fewer ruggedness categories may result. For EBNR and TSNR three categories were created considering the range of minimum and maximum index values and the distribution of the pixels using Jenks natural breaks.

\[ \text{Riley TRI} = \sqrt{\text{Abs}\{ (\text{MaxDEM})^2 - (\text{MinDEM})^2 \}} \]  

(3.3)

where,

* Riley TRI = resulting raster with terrain ruggedness index categories
* MaxDEM = maximum elevation value of the raster in the focal neighbourhood
* MinDEM = minimum elevation value of the raster in the focal neighbourhood

**Surface Area Ruggedness Index (SARI)**

Slope is one of the most important features of the earth’s surface form (Pareta and Pareta, 2011). The area of a sloping pixel is not the same size as one that lies flat, i.e. a sloped pixel occupies more area than a flat one as illustrated in Figure 3.4 (Jenness, 2004). The slope of the catchment DEMs of EBNR and TSNR were calculated using the ‘slope tool’ in ArcMap 10.2.1 in degrees. The resulting rasters were processed in the raster calculator to calculate SARI following equation 3.4 for each of the respective DEMs. The final SARI rasters were categorized with Jenks natural breaks to display the SARI ranges for each of the nature reserves.
3.3. Methods

Figure 3.4: a) shows how the 9 cells create a continuous surface to form Triangulated Irregular Networks (TIN), and Triangle pair VI-VIII (darker grey in b) comprise more surface area than other pairs in IV-II, I-III and V-VII (After Jenness, 2004).

Surface Area Ruggedness Index, $SARI = \frac{\text{Non-slope Area}}{\cos(Slope^\circ)}$ (3.4)

where,

$\text{Non-slope Area} = $ areas not contributing in slope areas and

$Slope^\circ = $ slope areas in degrees

Melton Ruggedness Number (MRN)

The Melton Ruggedness Number (MRN) is the basin relief divided by the square root of basin area described by Melton (1965) and used by Malik et al. (2011); Pareta and Pareta (2011); Parker et al. (2010); Voarintsoa et al. (2012) and Ruszkiczy-Rüdiger et al. (2009).

Firstly, maximum and minimum zonal elevation rasters were created using the 'focal statistics' tool (with 3 by 3 rectangular roving neighbourhood window) from the catchment DEM of EBNR and TSNR. Then using the same DEMs, drainage basins were delineated for both reserves. ArcHydro toolset 2.0 was used in ArcMap10.2.1 to delineate the drainage basins (detail description in subsection 3.3.1). The 'zonal statistics' tool was then used to calculate the basin areas and respective maximum and minimum elevation of the basins. This was followed by calculating the MRN using equation 3.5 in the attribute table of the basin layer with the field calculator in Python mode.

$$\text{Melton Ruggedness Number, } MRN = \frac{H}{\sqrt{A}} = \frac{(Z_{max} - Z_{min})}{\sqrt{A}}$$ (3.5)

where,
Chapter 3. Geospatial analyses of terrain and vegetation

\( H \) = vertical relief
\( Z_{\text{max}} \) = maximum elevation of the basin
\( Z_{\text{min}} \) = minimum elevation of the basin
\( A \) = basin area in \( m^2 \)

3.3.3 Quantifying the Seasonal Vegetation Pattern

To determine how vegetation might change seasonally, aerial imagery was obtained from NearMap™ (www.nearmap.com) on multiple dates (see Table 3.2). Images were downloaded through HiperTiles™ under license from the University of Western Australia.

NearMap™ use their own engineered camera capture, named a 'HyperPod', which is attached to Cessna 210 light aircraft that flies at a high altitude, enabling images of entire cities to be captured over 24-48 hours. The captured images are processed through a super-computer(s) running the software 'HyperVision', which automatically processes and stitches together the many individual photographs captured into a single virtually seamless 'PhotoMap' (www.nearmap.com).

Both EBNR and TSNR have been covered by NearMap™ imagery since 2007 but at different intervals and dates. The best possible images (those with the least cloud cover) were selected from the server and saved in the highest available resolution (4.2 cm for both nature reserves) which was significantly higher than the resolution of available LandSAT imagery. Two images were analysed per year and the images were selected to cover both the wet and dry seasons, from December 2007 to February 2014 (except for 2007 at TSNR which had a single photo, 2008 at TSNR which had no photos, and 2009 at TSNR which had one photo).

NearMap™ aerial imagery is only available in three bands (red, green and blue) and combinations of these bands were used to analyse the images covering TSNR and EBNR. Figure 3.5 illustrates the model which was used to calculate the areas of two types of vegetation and the rest of the areas as one other category. The vegetation was broadly classified into the following three categories:

1. Vegetation 1 (Veg 1) with canopies from larger trees (over 2 m height),
2. Vegetation 2 (Veg 2) with smaller bush and shrubs (less than 2 m), and
3. Other as the bare soil, visible grass and water visible from the sky).

Images were pre-processed and clipped in ArcGIS 10.2.1 with the catchment boundaries of each nature reserve to allow the model to calculate only the areas inside the catchment. Each of the clipped images was then used to create individual training samples based on visible trees and shrubs, with ground truthing based on personal study site familiarity and field photographs. Each of the training samples were used as an input along with the respective clipped images, and were classified using the 'maximum likelihood supervised classification' algorithm inside ArcMap’s model builder (using the bands detailed above). The pixel counts of each class of the resulting classified images were then processed (using the 'table select' tool and the 'table to excel' script in ArcMap) and exported to a Microsoft excel spreadsheet. These spreadsheets were then processed in R for calculating the vegetation area coverage and plotting the results. Each of the classified images were
3.4. Results

3.4.1 Hydrological Analysis

Analysis of Digital Elevation Model (DEM)

From analysis of the 1 m DEM, TSNR showed a higher elevation (from 25.5 to 32.9 m AHD) than EBNR (from 13.9 to 21.2 m AHD) of approximately 11 m. Topographically, the centre of EBNR had a lower elevation than its surroundings (see Figure 3.6). This directs most of the surface run-off towards the main drainage channel connected with Ellen Brook river channel. This channel runs through EBNR (outside the fenced area) and connects downstream with one of the Swan River tributaries to the west of the nature reserve. This allows any overflow from EBNR in the wet season to drain directly to the Swan River with limited time-lag. While the Ellen Brook channel bed has an elevation of only 4 to 6 m AHD, no upstream inflow enters EBNR from the channel.

Conversely, the drainage channels of TSNR had no direct connection to any outflow channels. The catchment showed irregular topography forming some isolated swamps (Figure 3.7. The north-west and north-east swamps receive most of the surface runoff.

Cross Section Profiles

Cross section profiles were taken along straight lines with an effort to include the catchment boundaries, swamps and channels. Figure 3.8 illustrates the variation of the catchment along the cross section in m AHD.

In EBNR, the north-west to south-east (NW-SE) cross section showed the NW boundary had an elevation of about 19.7 m AHD with a comparatively lower elevation (18 m) at the SE boundary. The main swamp elevation was around 16.7 m AHD while the main drainage channel had a depth of 1 to 1.5 m from the swamp bed. For the south-west to north-east (SW-NE) cross section, the SW end had an elevation of about 20 m AHD at the SW boundary, with a gradual slope reaching an elevation of 18.5 m towards NE boundary. The main swamp elevation was around 16.5 m AHD while the main drainage channel had a depth of 1.5 m from the swamp bed at this cross section (Figure 3.8a).
Figure 3.6: Map showing the location of cross-sections, zero-order and first-order channels and channel profile of EBNR

In the case of the south-west to north-east (SW-NE) cross section of TSNR, the SW boundary had an elevation of over 31 m AHD which dropped to 28.6 m AHD at 700 m along the cross section and again peaked at 31.2 m AHD at 1200 m along the cross section. It sharply declined to 28 m AHD at 1300 m and then had a steep rise to 30.8 m AHD at 1500 m along the cross section towards the NE boundary. The main swamp had an average elevation of 28.6 m AHD without the presence of a distinct drainage channel at this cross section. For the north-west to south-east (NW-SE) cross section, the NW boundary had an elevation of over 30.6 m AHD which dropped to 29.8 m AHD at 1550 m along the cross section and again peaked to 30.8 m AHD at 1450 m along the cross section. It sharply declined to 27.4 m AHD at 1300 m and had a vertical rise to 29.5 m AHD at 1100 m along the cross section. Towards the SE boundary, the elevation reached its lowest height of 27.4 m AHD. The main swamp had an average elevation of 28.4 m AHD with a drainage channel at this cross section at a depth of 1 m from the swamp bed (Figure 3.8b).

Channel Profiles of EBNR and TSNR

The profile of the main drainage channels were analysed within both EBNR and TSNR (Figure 3.9). The main drainage channel of EBNR commenced at the eastern side of the reserve and ran through the main swamp in the middle of the reserve, and ending by connecting with the Ellen Brook channel. The main drainage channel in TSNR started at the north-west side of the reserve and ended at the boundary of the northern side (see Figure 3.6 and 3.7).

The EBNR drainage channel started at an elevation of 16.4 m AHD with an overall downward trend, falling to approximately 14 m AHD. The continuous oscillation along the channel represents the micro structure and gilgai clay nature of the reserve. The flow-
3.4. Results

Figure 3.7: Map showing the location of cross-sections, zero-order and first-order channels and channel profile of TSNR

control valve to control outflow in the wet season is represented by the sharp peak at the end of the channel, measured at 15.4 m AHD of elevation (Figure 3.9a). At EBNR, there are seven first-order channels which all run into the single second-order channel. This second-order channel functions as the main drainage channel in EBNR (Figure 3.6).

In TSNR, the main drainage channel started at an elevation of 29.4 m AHD which then sharply declined to 28.2 m AHD at 200m along the length of the channel. The channel elevation remained at an average of 28 m AHD until 800 m along the channel length and then dropped down to 27.2 m AHD at the end. The TSNR channel profile had a lot less variation in elevation than the EBNR channel profile which reflects the smoother topography of TSNR (Figure 3.9b). There was a total of seven first-order channels and three second-order channels in TSNR; three of the first order channels ran into one second-order channel in the north-east swamp, another three first-order channels ran in the second second-order channel of the north west swamp, and the last first-order channel merged with the third second-order channel near south-west swamp of TSNR (Figure 3.7).

Figure 3.8: Cross-sections of EBNR and TSNR showing variations in profiles for South-West (SW) to North-East (NE) (red line) and North-West (NW) to South-East (SE) (black solid line) directions
Ponding Extent Analysis of EBNR and TSNR

EBNR and TSNR varied considerably in regards to their extent and pattern of inundation. At EBNR, only the main drainage channel was filled at 0 cm depth (the base of the main swamp was taken as 0 cm). Ponding extent was constrained to the area of the main swamp until 30 cm of depth at the middle was reached. At 50 cm depth, the inundation started to cover the eastern swamps and continued to create small isolated pools between 50 to 70 cm of water depth. At 90 cm depth, about 80% of EBNR was under water, suggesting that the point of swamp overflow was likely between 70 to 90 cm of water height (see left hand panels in Figure 3.10).

In TSNR, ponding did not become visible until 30 cm of depth (taking the lowest swamp bed as 0 cm) at the north-eastern corner of the catchment, and 58 cm at the south-east and south-west corners of the swamp. At this same depth, the north-east swamp continued to cover more area and became connected with the north-west swamp. At 72 cm of depth, inundation of the east swamp became visible, and all the swamps continued to get larger in extent until 84 cm of depth. At a depth of 115 cm, most of the swamps were connected and inundated (see right hand panels in Figure 3.10).

The probability distribution function of the ponding area extent for EBNR and TSNR were plotted against a log Area scale for comparison. From this it was clearly evident that in EBNR the expansion of ponding extent occurred at a symmetric rate, resulting in very similar distribution curves at different depths (Figure 3.11a). For TSNR, there were differences for each distribution curve, indicating that the ponding extent was not expanding in a symmetric way.

3.4.2 Morphometric Analysis

A total of five types of morphometric analyses were conducted for EBNR and TSNR, including calculations for different types of topographic indices, variability of slope, and standard deviation of elevation. Table 3.1 summarises the classifications of different topographic categories with respective domain coverage, and the area covered by the results are described below with sub-headings.
3.4. Results

Figure 3.10: Ponding extent of differing depths at EBNR and TSNR. Panels in the left column (panels EB 1 to EB 6) represent the ponding extent in EBNR, and panels in the right column (panels TS 1 to TS 6) represent the ponding extent in TSNR.
Chapter 3. Geospatial analyses of terrain and vegetation

Figure 3.11: Ponding extent at different depths at EBNR and TSNR

Relative Topographic Position (RTP)

The RTP index for EBNR was from 0.12 to 0.84, and for TSNR RTP was from 0.14 to 0.86 (Figure 3.12a (a) and (b)). The value of RTP did not vary much between the reserves but the dominant class (Class 2, with values ranging from 0.44 to 0.54) was occupying 59.74% of EBNR, and 65.14% of TSNR. In TSNR, RTP classes were distributed homogeneously, except where there were some Class 2 patches near the south-west side. A large number of Class 2 values occurred in EBNR to the north-west corner.

Figure 3.12: Relative Topographic Position of the two nature reserves

Standard Deviation of Elevation

The standard deviation of elevation ranged from -0.84 m to 0.76 m, and -0.77 m to 0.79 m for EBNR and TSNR, respectively (Figure 3.13 (a) and (b)). TSNR had a slightly higher standard deviation in elevation than EBNR. Class 2 (white class), with a deviation value of 0.1 to 0.094, occupied 81.1, and 87.5% of the total catchment in EBNR and TSNR, respectively. Standard deviation was highest for the middle-northern part of EBNR, whereas in TSNR high standard deviations were only found in the north-west part and in the middle-north side of the catchment.
3.4. Results

Figure 3.13: Standard deviation of elevation of the two nature reserves

**Slope Variability**

Slope variability ranged from 0 to 37 degrees, and 0 to 44 degrees for EBNR and TSNR, respectively (Figure 3.14 (a) and (b)). TSNR had more variation in slope than EBNR, which suggests topography was more variable in TSNR than EBNR. Class 1, with values of 0 to 3.9 degree slope variability (EBNR) and 0 to 2.6 degree slope variability (TSNR), occupied 58.05 and 68.81 % of the catchments for EBNR and TSNR, respectively. The greatest slope variation was on the south-eastern part of EBNR and in the north-western part of TSNR.

Figure 3.14: Slope variability of the two nature reserves

**Riley Terrain Ruggedness Index (RTRI)**

The RTRI was calculated for both EBNR and TSNR with ruggedness values classified into three classes following the criteria of Riley et al. (1999). Both EBNR and TSNR had 'level', 'nearly level' and 'slightly rugged' classes. Values were different in the two sites. For EBNR, 0 to 1.6, 1.6 to 2.9 and 2.9 to 8.2 RTRI values were assigned to each class type respectively. For TSNR, 0 to 1.7, 1.8 to 3.0 and 3.1 to 11 RTRI values were assigned to the three classes. Around half of EBNR and TSNR had the RTRI classed as
'level’ (represented in black). TSNR had more RTRI values for the slightly rugged class, and these occupied 11.45% of the catchment. EBNR had less RTRI values in the slightly rugged class which occupied 13.38% of the domain.

![Riley Terrain Ruggedness Index of the two nature reserves](image)

**Figure 3.15:** The Riley Terrain Ruggedness Index of the two nature reserves

**Surface Area Ruggedness Index (SARI)**

The SARI value in EBNR ranged from 0.01 to 0.39, and in TSNR the range was 0.01 to 0.44 (Figure 3.16 (a) and (b)). Class 1 (representing lower SARI values) occupied 62.18, and 58.01% of the domain in EBNR and TSNR, respectively. Class 3 (representing higher SARI values) ranged from just 0.44 in TSNR to 0.39 in EBNR, representing 1.98% and 2.57% of the catchment areas respectively.

![Surface Area Ruggedness Index of the two nature reserves](image)

**Figure 3.16:** Surface Area Ruggedness Index of the two nature reserves

**Melton Ruggedness Number (MRN)**

The Melton Ruggedness Number (MRN), which is a ratio of basin relief and square root of basin area, was calculated for both reserves. This number is assigned to sub-catchment or watershed, and therefore is dependent on the nature and size of a catchment. The MRN of EBNR ranged from 0.0066 to 0.0248, compared to 0.0060 to 0.018 for TSNR. A
3.4. Results

MRN value of 0.0094 applied to the largest, central sub-catchment of the EBNR reserve. A small sub-catchment on the western side of EBNR had the highest MRN (0.0248). In TSNR, the highest MRN was found for a small sub-catchment on the south-east corner of the catchment. The lowest MRN was for the biggest sub-catchment located at the middle of the TSNR catchment.

Figure 3.17: Melton Ruggedness Number of the two nature reserves

Table 3.1: Summary results of the ruggedness analyses of EBNR and TSNR

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<th>Morphometric Features</th>
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<th>Break Value</th>
<th>Pixel Count</th>
<th>Class Percent</th>
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</table>

3.4.3 Vegetation Analysis

Vegetation in both EBNR and TSNR had prominent seasonal patterns and variations. The vegetation data plotted with monthly average rainfall from the weather station located at EBNR, and monthly average groundwater data from the nearby DoW bore (no. 5423), is presented in Figure 3.18.

For EBNR vegetation, type 1 (trees < 2 m height) showed prominent seasonal variation and occupied larger areas of the reserve ranging from 0.18 square km to 0.21 square km. Conversely, vegetation type 2 varied regularly with seasonal change over a period of seven years (2007 to 2014). Bare soil or ground classified as ‘other’ showed a subtle seasonal pattern. For TSNR, both vegetation types 1 and 2 showed seasonal variations ranging
from 0.45 square km to 0.7 square km of canopy cover. The maximum canopy coverage was found in 2012. For vegetation type 2, the canopy cover started at as low as 0.54 square km in 2007 and increased to 0.75 square km in 2011. The ‘other” area showed very few variations.

Compared with the seasonal pulses of rainfall and groundwater, the vegetation in both EBNR and TSNR responded in different ways. Vegetation type 1 of EBNR followed the hydrological pulses, and there was a clear corresponding seasonal increase and decrease to canopy cover. This pattern was not as obvious for vegetation type 2. In TSNR, there was not any clear trend for either vegetation type.

Figures 3.19 a to h and Figure 3.20 i to m represent the canopy coverage after supervised classification of the aerial imagery of EBNR. TSNR vegetation maps are shown in Figure 3.21 a to f and Figure 3.22 g to j. Both sets of vegetation maps show how the two groups of vegetation, and the bare soil cover, change seasonally.

Table 3.2: Dates of NearMap™ aerial photos used for quantification of vegetation in EBNR and TSNR

<table>
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</tr>
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<td>22-Dec-07</td>
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<tr>
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<td>26-Sep-09</td>
</tr>
<tr>
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<td>12-Jul-11</td>
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<td>08-Sep-11</td>
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</table>
3.4. Results

Figure 3.18: Vegetation of EBNR and TSNR, plotted with rainfall and groundwater of the area. Panel a), b) and c) represent EBNR, TSNR and without vegetation. Panel d) is monthly average groundwater and panel e) is monthly average rainfall.
Figure 3.19: Vegetation maps of EBNR showing seasonal variation in canopy cover
Figure 3.20: Vegetation maps of EBNR showing seasonal variation in canopy cover
Figure 3.21: Vegetation maps of TSNR showing seasonal variation in canopy cover (Contd..)
Figure 3.22: Vegetation maps of TSNR showing seasonal variation in canopy cover
3.5 Discussion

The work here has highlighted that EBNR and TSNR, while both being classed as equally important core habitats for the WST (Burbidge et al. 2010), have very different internal workings. By quantifying the flow networks, drainage, terrain, and vegetation of the two reserves, it was possible to create a conceptual model of both wetlands (Figure 3.23), which helps to explain why pool formation and ponding extent differ greatly between the two reserves.

In terms of terrain, TSNR had more internal ruggedness than EBNR as shown by the RTP index, RTR index, and SARI values measured. The slopes of the two study sites showed that EBNR has 33.73 per cent of its total area within 3.9° and 12° slope, while TSNR had only 20.28 per cent in the same slope group. This influences the shading and water flow within the reserves, which in turn influences both water availability and water temperature, which are key habitat aspects that influence the behaviour of the WST (tortoises can only forage when water temperatures are warm enough, and the presence and absence of water links to aestivation and nesting behaviours; (Mitchell et al., 2013, 2016)). If temperatures rise (as they are predicted to do so; IPCC 2007 and Barker (2007); Silberstein et al. (2012)), then shading is likely to become increasingly important, and along with slope, this shading is created by the vegetation of the reserves. At TSNR, shrubs and large tree canopies were very responsive to groundwater oscillations and large rainfall events (Figure 3.18), which can be explained by their close association with the groundwater table. Conversely, shrubs in EBNR had less seasonal dynamics than that of large tree canopies (which had distinct seasonal variations), and shrubs showed less responses with groundwater oscillations and rainfall events when compared to large trees. This can be inferred as the disconnection with groundwater in EBNR, which is caused by the gilgai clay that characterises this wetland. These differences in vegetation dynamics for each reserve show that EBNR has a more stable annual vegetation coverage when compared to TSNR, meaning the degree of shading over water bodies is more persistent and more independent of rainfall events. Therefore, EBNR is likely to better buffer the effects of a drier future climate.

When considering the availability of pooled water for the WST, perhaps the most significant difference between the two reserves is in the internal flow paths and potential ponding areas. Generally, the difference in flow pathways and terrain conditions manifest in the distribution of pools within the domains. For example, Nath et al. (2013) undertook a pool distribution analysis as a function of hydrological connectivity with groundwater for a groundwater dependent wetland site in southern WA and highlighted it was a useful metric related to assessment of acid sulphate soil risk and hydrological processes (e.g. seepage potential). The clay-pan wetland here was considerably different to that of Nath et al. (2013), due to the unique geomorphological conditions and lack of groundwater connectivity, but the metric is potentially relevant to tortoise habitat in terms of impacts on the soil moisture associated with aestivation sites, or access to pools for feeding. For similar heights of inundation, the number and extent of the pool distribution was significantly different for EBNR and TSNR. In EBNR, one major outflow carries water to the Ellen
Brook channel in the wet season. Conversely, water flows through five different outflow paths in TSNR, none of those being dominant over the others. Despite the flatness of the reserve, the connectivity of the flow networks within EBNR are more robust than those of TSNR, since TSNR has less connected networks and more internally draining micro-catchments with less defined channels. Internal redistribution of water is also different since the internal catchment boundary delineation varies with each of the nature reserves. This translates to numerous pools forming at EBNR which are capable of connecting with each other under very high water levels. Conversely, the morphology of TSNR prevents this, and instead there are distinct, independent pools. These differences are likely to be emphasised under a hotter, drier future (Charles et al., 2010) where good seasons are going to be more important for WSTs and the ability of the wetlands to buffer the effects of climate will be more pronounced.

The classic view of wetland vegetation zonation depicts strong gradients in vegetation functional groups moving from a central lake region to the upland areas within the domain (Coletti et al., 2013b; Mitsch et al., 2012; Sommer and Froend, 2014). However, this generic view of gradients in vegetation morphology was inadequate for the two wetland systems studied here. In the case of EBNR, the gradient of vegetation types did not clearly emerge as in the classic conceptual model of vegetation zonation because of the high degree of patchiness in the overstorey and understorey. The low groundwater level and presence of gilgai clays at EBNR means that groundwater seepage is slow and unidirectional from the lakes. As the clays crack in summer following desiccation, the seepage losses from the surface water pools will be much more rapid during the initial filling period of the wetland...
before the clays have time to swell. This is somewhat different to TSNR, which also shows patchy vegetation, but also has more distinct pools, and a different vegetation assemblage to EBNR. The more sandy soils of TSNR lead to pools draining more rapidly, and as a result, management actions such as lake supplementation with bore water pumping (in effect since 1994; Burbidge et al. 2010 and chapter 2) have been necessary.

Characterising microhabitat structure is important for improving our understanding of how species interact with the environment, and may help guide conservation solutions. However, studies that consider multiple spatial scales of habitat condition and identify how these connect to population viability are lacking. Furthermore, linking local scale habitat heterogeneity with long-term forecasts of ecosystem function remains a critical challenge. Where suitable proxies can be identified, tools such as those described here can enable detection of key habitat structures that may be crucial for maintaining species viability. Examples from temporary wetlands in agricultural fields and solitary trees in South African savannas (Tews et al., 2004) have demonstrated that detailed knowledge of key habitat structures may simplify biodiversity conservation by further protecting a wide array of species and functional mechanisms at the same time. However, these critical habitat structures have not yet been well defined for the WST, though those included in the conceptual model here would provide a good foundation. The indicators discussed herein (specifically hydroperiod, inundation pattern affected by terrain variability and shading and canopy cover as a function of vegetation dynamics) are relevant ecologically and physiologically for the WST, and were chosen since they are predictable using contemporary ecohydrological model approaches (e.g. Mitchell et al., 2013; Coletti et al., 2013). In the next chapter, we aim to test an ecohydrological model able to explore, albeit simply, how such micro-habitat structure can influence key hydrological processes relevant to the WST populations.
Chapter 4
A model assessment of the ecohydrology of Ellen Brook Nature Reserve

4.1 Chapter Prologue

The previous two chapters led to the development of a conceptual wetland of both Ellen Brook Nature Reserve (EBNR) and Twin Swamps Nature Reserve (TSNR). This conceptual model is formalised for EBNR in this chapter, by generating two types of wetland model (a simple lumped bucket model, and a more complex compartmentalised model) and testing the performance of these against the field data and observations assimilated through Chapters 2 and 3. Importantly, the wetland models provided here present the first predictions of how EBNR, the wetland that is critically important for the survival of the western swamp tortoise (WST, *Pseudemydura umbrina*) in the wild, will perform under future climates. By demonstrating that these models are able to adequately capture the dynamics of EBNR, they can be extrapolated across the surface of South-western Western Australia (SWWA) to help inform the selection of wetlands suitable for assisted colonisation, as is show in Chapter 5.

4.2 Introduction

Hydrological regimes are regarded to be the key driver of wetland ecosystems (Bunn and Arthington, 2002; Lamontagne and Herczeg, 2009). They ultimately control vegetation productivity, biogeochemistry, water extent and the feeding, breeding, and survival of the fauna living within it. Given the rate of loss and degradation of wetlands (Davis and Froend, 1999) and the threat of climate change (Finlayson et al., 2011), in terms of altering the hydrological regimes, it is important to have a detailed understanding of the hydrological processes that control the water balance. Predictive tools able to simulate wetland function and condition are also essential to support the management and restoration of these systems. The application of the wetland water-balance approach to hydro-ecological studies provides valuable insights into the relative importance of processes acting within wetlands (Gasca and Ross, 2009), and can guide our assessment of how they will respond to future climate scenarios. But a detailed wetland water-balance can be surprisingly difficult to simulate because of the spatio-temporal heterogeneity and complex dynamic nature of these ecosystems (see Chapters 2 and 3). Furthermore, extending wetland simulations to predict critical habitat for key species is also challenging.

Modelling wetland ecosystems to gain scientific insights has become increasingly popular in recent years (Chui et al., 2011; Karvonen et al., 1999; Munro et al., 2000; Park et al., 2014). Generally, one of the biggest problems for managing ecosystems is lack of
enough information as data collection can involve significant cost and effort, and sometimes it is impractical to cover all aspects. Modelling of wetland ecosystems is a powerful tool that can complement the practical drawbacks involved in monitoring and managing such systems (Cormier et al., 2013; Palanisamy and Chui, 2013; Park et al., 2014; Zhang and Mitsch, 2005) by allowing exploration of the system interactions (e.g. feedbacks) and changes in system states over long-time frames.

There has been a increasing trend of using lumped bucket models for simulating wetland hydrology, which captures basic hydrological processes of the system (Granier et al., 1999; Karpouzos et al., 2011), but which lacks the capacity to capture spatial heterogeneity. Beyond water balance modelling, a parallel stream of simulation for wetland assessment is the use of GIS based spatial tools to look at water balance processes using spatial and remote sensing data (Brunner et al., 2009; Luzio et al., 2004; Zhang et al., 2014), but these approaches lack the capacity to capture dynamic processes within the ecosystem and how the ecosystem may respond to non-stationary climatic conditions. So the question becomes, how can these approaches be applied to simulate the hydrological dynamics and essential feedbacks between climate and vegetation that impact upon wetland hydrological regimes over long-term time-frames? There is therefore a need for a parsimonious model that is able to resolve key aspects of heterogeneity relevant to system function, whilst also supporting a level of process complexity suitable for capturing how the system will change over time.

The development of eco-hydrological models has advanced considerably (Nordbotten et al., 2007) and they are now capable of producing simulations that account for the feedbacks between vegetation and changes in water budget over a range of temporal scales (Eamus et al., 2006). To date, there are relatively simple applications of ecohydrological models to wetlands (Coletti et al., 2013b; Park et al., 2014). An eco-hydrological model that is capable of encapsulating the ecology-hydrology feedback mechanisms in sufficient detail can be used to generate wide array of relevant information on habitat condition (e.g. hydroperiods, vegetation shading etc.), and can be particularly useful for threatened species management, where guidelines and local management initiatives place particular emphasis on undertaking feasibility and risk analyses of the suitability of existing (and future) conservation sites.

Here I develop a detailed ecohydrological wetland model that aims to predict habitat suitability for the Critically Endangered WST, which occurs exclusively in ephemeral swamplands and the surrounding terrestrial habitat. The objectives of this chapter are to firstly compare the performances of a simple pan model against a more comprehensive compartmentalized eco-hydrological model, and secondly, to use the validated ecohydrological model to compute a detailed water balance and assess how it responds under future climate projections. The detailed hydrological information simulated, when used in conjunction with the data analysis of the previous chapters, helps to bridge the gap between understanding wetland hydrology and the provision of critical habitat for the WST.
4.3 Model Approach

Two types of wetland model were generated (a simple pan, lumped bucket model, and a more complex compartmentalized model) in order to determine if a) a simple bucket model would be able to adequately describe the wetland habitat of the WST, and b) if not, what extra does the compartmentalised version bring in terms of model accuracy and performance?

The lumped bucket model adopts a generic shape based on cylindrical geometry with a paraboloid depression embedded at the centre of the cylinder (Figure 4.1 a). With this simple model it is possible to simulate some hydrological features like lake height \( h_L \), hydro-period, and water temperature, but it lacks the capacity to simulate the feedback mechanism between vegetation and water, and other detailed ecohydrological properties like infiltration, percolation, surface water-groundwater interaction (seepage), run-off and vegetation dynamics. It also assumes no compartmentalisation in the wetland system and treats the disconnected pool as one generic storage.

Where a catchment has two or more sub-catchments, and each sub-catchment operates in relative isolation from a water-balance point of view, then a lumped model will overly simplify the actual hydrology. Being able to customise wetland models to site-specific situations makes the application less general but allows the local variability of hydrological regimes to be simulated more realistically. In the case of EBNR, because of construction of weirs, the wetland is in fact divided into three sub-catchments all of which run into a larger fourth sub-catchment. The last sub-catchment received the outflows of the three adjacent catchments as its inflow while acting as a sub-catchment itself and any outflow from this fourth sub-catchment flows through the drainage channel to outside the wetland (Figure 4.1 b). The second model applied is therefore a semi-distributed (compartmentalized) application, with each sub-basin simulating a unique water balance.

4.3.1 Lumped Water Balance Model

For the lumped hydrological model, the lake bed has a conic shape and is embedded in the centre of a cylinder, with a height \( h \) meters above the deepest point of the wetland. The maximum radius of the cone is limited to the domain boundary, \( r_W \). Equation 4.1 was used to relate between lake volume and lake height:

\[
LV = \frac{\pi r^2 h}{3} \tag{4.1}
\]

The water budget of the lumped model was computed using the following derivation of standard water balance equation:

\[
\frac{dLV}{dt} = PL A + Q_{gw} - Q_S - EL A - Q_{out} \tag{4.2}
\]

where, \( LV \) is the lake height (mm), \( PL \) is precipitation (mm), \( Q_{gw} \) is groundwater inflow (mm), \( Q_S \) is seepage (mm), \( EL \) is total evaporation, \( A \) is the area of the wetland (mm\(^2\)) and \( Q_{out} \) is the outflow from the wetland (mm).
Figure 4.1: Conceptual diagram of lumped model (panel a.) and panel b. shows the conceptual diagram of the compartmental model and how it connects more than one partitions of small watersheds running into another as inflow and producing cumulative outflow while functioning as individual hydrological systems by each of its own. Inset c. illustrates the digital elevation model (DEM) of EBNR divided into sub-catchments 1, 2, 3 and 4.
4.3. Model Approach

Heat Balance of Wetland Lakes

Water temperature is computed based on a heat budget according to (Mitchell et al., 2013) considering solar heating, longwave radiation and standard bulk aerodynamic flux parameterisations for sensible and latent heat. The budget is driven by the prevailing meteorological conditions and the heat balance is calculated as:

\[
(L\rho_w c_p)\frac{dT_w}{dt} = \rho_w c_p [IT_i - OT_w + RA_L T_{sky}] + [H - E - S + \phi_{LW} + \phi_{SW}]A_L
\]  

(4.3)

where \(T_w\) is the lake temperature, \(T_i\) is the inflow temperature and \(T_o\) is the outflow temperature, \(T_{sky}\) is the air temperature (all in °K), \(\rho_w\) is the density of water (kg/m³), and \(c_p\) is the specific heat capacity of water (J/kg/K). \(L\) is the lake volume (m³) and \(I, O\) and \((A_L R)\) are the inflow, outflow and rainfall volume fluxes respectively (m³/d), where \(A_L\) is the lake surface area (m²) used to multiply the rainfall depth (m/d). Sensible heat, evaporation and soil heat conduction are in heat fluxes (W/m²) and are denoted with \(H, E\) and \(S\), respectively, and calculated as:

\[
H = C_H u_2 (T_{sky} - T_w)
\]

(4.4)

\[
E = \lambda C_E u_2 (e_s - e_a)
\]

(4.5)

\[
S = C_S (T_s - T_w)
\]

(4.6)

where \(C_H\), and \(C_E\) are bulk transfer coefficients for sensible and latent heat respectively, and \(C_S\) is the diffusivity of heat into the soil below the water (W/m²K). \(\phi_{LW}\) is the net longwave radiation (W/m²) and \(\phi_{SW}\) is the solar insolation (W/m²), calculated from:

\[
\phi_{LW} = \varepsilon_w \sigma (T_{sky} + 273.15)^4 - \varepsilon_w \sigma (T_w + 273.15)^4
\]

(4.7)

\[
\phi_{SW} = (1 - \alpha) \bar{\phi}_{SW} \left( \frac{A_L}{A_{MAX}} k_s \right)
\]

(4.8)

where \(\varepsilon\) is the emmisivity of air or water, \(\sigma\) is the Stefan-Boltzman constant, \(\alpha\) is the shortwave radiation albedo, \(\phi_{SW}\) is the incident solar radiation above the vegetation (W/m²). The last term of Eq. 4.8 accounts for the effect of wetland vegetation shading on the incident solar intensity, with \(k_s\) defined as a constant to increase the shading effect, and \(A_{MAX}\) is the maximum area of inundation extent (m²).

4.3.2 Compartmentalized Ecohydrological Model

The wetland model applied here is based on Coletti et al. (2013b) which divides the wetland into different zones of hydrological function and simulates partitioning of water between the zones as mediated by climate, soil, and vegetation controls. As a result, the model predicts the changes in the extent of surface water inundation, soil moisture and vegetation, and captures the various eco-hydrological feedbacks that mediate wetland response to changes in rainfall and air temperature. In this analysis, the above model
was adapted to operate on a sub-daily (hourly) time-step and to simulate soil and water thermodynamics. The shape of the wetland inside the model was adapted to suit the shape of the study sites, and the thermodynamics were also modified by including an additional shading factor.

**Compartment Ecohydrology**

The compartment specific model can be used as a tool to model the major hydrological fluxes through linking three conceptual water storages relevant to wetland systems typical of those in southern Australia - the open water/lake (L), and the water contained in the unsaturated (U) and saturated (S) soil zones (Figure 4.2). The water balance of each store within the wetland then can be simulated using a mass balance approach. Each of the compartments of the model used the shape from the DEMs of the individual sub-catchments using a hypsographic curve similarly calculated for nature reserves (see Chapter 2, section 2.4.2).

![Figure 4.2: Conceptual ecohydrological model showing different parameters used in the model. Items in blue letters denotes climate forcing data. Items in black letters are different components of the model.](image)

### Standing Water Storage and Connectivity Between Compartments

The standing water storage of each sub-catchment is calculated as:

\[
\frac{dL_V}{dt} = P_{L_i}A_{L_i} + Q_{gw_i} - Q_{Si} - E_{bL_i} - Q_{out_i}
\]  

(4.9)

where, \(L_V\) is lake height (mm) for sub-catchment i, \(P_{L_i}\) is precipitation for sub-catchment i (mm), \(Q_{gw_i}\) is groundwater inflow for sub-catchment i (mm), \(Q_{Si}\) is seepage (mm) for sub-catchment i, \(E_{bL_i}\) is total evaporation, \(A_{L_i}\) is the area of the lake for sub-catchment i (mm²) and \(Q_{out_i}\) is the outflow from the wetland for sub-catchment i (mm). The sub-catchments are denoted by i, where i = 1, 2, 3, and 4.
The connectivity between compartments are calculated by:

$$\frac{dL_i}{dt} = Q_{in}^i - Q_{out}$$  \hspace{1cm} (4.10)

For $i = 1$, 2 and 3, $Q_{in} = 0$

For, $i = 4$, $Q_{in} = \sum_{i=3}^{i=4} 0$

where, $i$ denotes the number of the sub-catchments.

**Groundwater Saturated Zone**

The maximum soil storage, $S_{max}$, is the difference between the maximum volume of the wetland, $W_{max}$, and the maximum lake volume, $L_{max}$, multiplied by soil porosity $\phi$. $S_t$ is the total effective volume present in the soil which changes in time as function of the volume that infiltrates the soil, $I$, the seepage from/to the lake, $Q_S$, the volume lost as baseflow, $Q_{ss}$, the total evapotranspiration from the ground, $E_{soil}$:

$$\frac{dS_t}{dt} = I \pm Q_S - Q_{ss} - E_{soil}$$ \hspace{1cm} (4.11)

In the above equation, $E_{soil}$ is the sum of transpiration (E) and bare-soil evaporation ($E_b$) from both unsaturated and saturated environments, such that $E_{soil} = E_{bU} + E_{bS} + E_U + E_S$ (terms defined further below). The volume lost as baseflow, $Q_{ss}$, is defined as:

$$Q_{ss} = (\alpha G h_S) A_W$$ \hspace{1cm} (4.12)

The maximum net capacity of water storage in the unsaturated zone, $U_c$, is the difference $S_{max} - S_{sat}$, where $S_{sat}$ is the volume of water ($m^3$) that is stored in the saturated pool below the water table level, $h_S$, such that $S_{sat} = S$. When the soil is totally saturated, $S_t = S_{sat} = S_{max}$ and $S_{us} = U_c = 0$, where $S_{us}$ is the effective volume of water ($m^3$) present in the U zone. Under such a condition, the infiltration rate, $I$, is equal to zero. For all other times, the total water store is a combination of the saturated and unsaturated region, such that:

$$S_t = S_{sat} + S_{us}$$ \hspace{1cm} (4.13)

The water balance of sub-region U and S is respectively defined as:

$$\frac{dS_{us}}{dt} = I - E_U - E_{bU} + Q_A - Q_p$$ \hspace{1cm} (4.14)$$

and

$$\frac{dS_{sat}}{dt} = Q_p - Q_s - E_S - E_{bS} - Q_A \pm Q_{ss}$$ \hspace{1cm} (4.15)
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$Q_A \ (m^3 d^{-1})$ represents the volume transferred from S to U via capillarity. It is linearly related to the water table level via a constant based on the soil type, $k_A$. The infiltration rate depends on the unsaturated zone soil moisture content prior to rainfall (Choudhury and Blanchard, 1983; Dall’O’ et al., 2001), $\theta(-)$, defined as $S_{us}/U_c$. Soil type affects the infiltration rate through the saturated hydraulic conductivity, $K_s \ (m d^{-1})$, and the arbitrary recession coefficient for infiltration, $k_I(-)$:

$$I = \begin{cases} 
-K_S(\theta - 1)^{k_I} A_U & \text{if } I < P A_U \\
PA_U & \text{if } I \geq P A_U \\
U_c & \text{if } I > U_c 
\end{cases} \quad (4.16)$$

The evaporation from bare soil, $E_b \ (m^3 d^{-1})$, is calculated based on the potential evaporation, $E_0$. If the soil is not saturated, the ratio of $\theta$ to water content at field capacity, $f_c$, is also considered as a scaling factor (Aydin et al., 2005). Further, evaporation from bare soil is adjusted based on total LAI (sum of all vegetation types) to reflect the cover that vegetation causes due to shading of the exposed soil surface. Therefore, the evaporative rate applied over the relevant areas, $A_S$ and $A_U$, respectively, are:

$$E_{bS} = E_0 \left(1 - 0.9 \frac{LAI_S}{LAI_{max}} \right) A_S \quad (4.17)$$

and

$$E_{bU} = E_0 \left(1 - 0.9 \frac{LAI_U}{LAI_{max}} \right) \left(\frac{\theta}{f_c} \right) A_U \quad (4.18)$$

Transpiration, $E$, is modelled as a function of the plant water uptake rate, $W \ (m d^{-1})$, integrated over the relevant environment, $n$, such that:

$$E_n = W_n A_n \quad (4.19)$$

where $W$ is a function of the normalized potential water uptake, $\Psi (-)$ and potential evapotranspiration (Skaggs et al., 2006), depends on the plant functional type, $i$, and their associated water uptake strategy (defined later in this Section), and normalized depending on the wetland zone maximum $LAI_{max_i, n}$ ($n = L, U, S$), and associated soil moisture conditions experienced by the roots at a given time. As a result, the total water uptake rate is the sum of all vegetation groups, $i$, coexisting in a particular environment, $n$, such that:

$$W_{i,n} = \sum_{n}^{i} \Psi_{i,n} E_0 \frac{LAI_{i,n}}{LAI_{max,n}} \quad (4.20)$$

For mass balance purposes, all the water that is obtained by plants from below the water table level is added to $E_S$, independently if the plants are standing over the saturated
or unsaturated area. Note that the ability of vegetation to regulate their water storage is not taken into account, meaning that there is no change in water use efficiency of the vegetation. Thus, vegetation water usage is linearly proportional to $LAI_n$ ($m^2$ leaf $m^{-2}$ land). Further, the vertical root profile is assumed as constant in time and aside from competition for water and light, no other specific competition factors are considered (Coletti et al., 2012, 2014).

Percolation of water from the vadose to the saturated zone, $Q_p$ ($m^3d^{-1}$), takes place when the volume of water in the unsaturated zone at the end of time-step surpasses the volume that can be held at field capacity, $U_{c,fc}$. Since the water table is shallow the distance the water needs to percolate is short, so we assumed the soil can reach field capacity within one day (Coletti et al., 2012, 2014). The percolation is calculated following losses from evapotranspiration and surface runoff are computed, such that:

$$S_{US}^* = S_{US}^{t-1} + I - E_{US} - E_{bU}$$

and

$$Q_p = \begin{cases} 
S_{US}^* - U_{c,fc} & \text{if } S_{US}^* > U_{c,fc} \\
0 & \text{if } S_{US}^* \leq U_{c,fc}
\end{cases}$$

Note that from a computational point of view, the infiltration rate is calculated based on the soil moisture from the previous day, which means that at the first day the water table level reaches ground level, infiltration can be different from zero. In this case, the exceeding volume is diverted to the lake by adding to $Q_{se}$.

Generally, after a drought period if the soil is rich in clay, there are formation of fissures. When the dry period is over, the fissured clay creates macropores which markedly helps raising the water velocity and infiltration rate in the vadose zone (Coletti et al., 2014). The model accounts for this phenomenon by simply increasing the hydraulic conductivity, $ks$. Schematically, $ks$ is multiplied by a constant, $ks_M$, when the area of the unsaturated zone, $A_U$, is above a defined threshold, $k_{AU}$. $k_{AU}$ is a fraction of the total wetland domain, $A_W$, and represents a proxy for dry conditions.

**Vegetation Biomass Calculation**

The wetland model considers how variables such as rainfall, temperature, vegetation cover and evapotranspiration influence the wetland ecosystem. The vegetation is measured as coverage of leaf area per unit area, namely Leaf Area Index, LAI (Huang and Pretzsch, 2010), evapotranspiration (ET) is measured as a function of solar radiation and the model has been setup as in Coletti et al. (2013b).

The total carbon amount accumulated as vegetation biomass, $B$ (kg C), is governed by the rate of carbon uptake via photosynthesis, $\Pi_A (kgC d^{-1} m^{-2})$, and losses due to litterfall, $(L_l)$, root death $(R_d)$ and respiration $(R)$, with all the loss terms given in $kg C d^{-1} m^{-2}$. Although the biomass of any particular plant type $i$, $B_i$, changes in time, it is spatially stationary and equally distributed within any distinct hydrological environment, $n$. As the spatial extent of each area with distinct hydrological function (U, S and L) changes, the
amount of any vegetation type present in each environment changes accordingly. Therefore, the balance equation for any vegetation type is defined as:

\[
\frac{dB(i,n)}{dt} = \begin{cases} 
(\Pi_{A,i,n} - L_{i,n} - R_{i,n} - R_{dl,i,n})A_n + D_{i,n-1}\frac{dA_n}{dt} & \text{if } \frac{dA_n}{dt} > 0 \\
(\Pi_{A,i,n} - L_{i,n} - R_{i,n} - R_{dl,i,n})A_n + D_{i,n}\frac{dA_n}{dt} & \text{if } \frac{dA_n}{dt} \leq 0
\end{cases}
\] (4.23)

where \(A_n\) is the area of the \(n^{th}\) wetland zone (\(m^2\)), and \(D\) is the carbon density per unit area (= \(B/A\)). Vegetation biomass is constrained by \(D_{max}\), the maximum carrying capacity that the system can hold given a soil water-holding capacity and climate, when in hydrological equilibrium (Nemani and Running, 1989). Mass conservation is assured if a portion of biomass previously belonging to another area, \(B_{n-1}\), is incorporated into \(A_n\) when \(dA_n/dt > 0\) (i.e., \(A_n\) expanding) such that:

\[
D_{i,n-1} = \frac{B_{i,n-1}}{A_{i,n-1}}
\] (4.24)

However, when \(A_n\) is shrinking (\(dA_n/dt < 0\)), then the following mass is removed:

\[
D_{i,n} = \frac{B_{i,n}}{A_{i,n}}
\] (4.25)

In order for us to validate the model, we must be able to convert between predicted biomass density and observation of LAI. Given the veg cover is patchy in reality, then we have to account for \(\chi_{Vi}\), the weighted average biomass. This conversion is therefore done as:

\[
\chi_{Vi} = \frac{A_{Vi}LAI_{Vi}\phi_{Vi}}{A_T}
\] (4.26)

where \(LAI_{Vi}\) is volume of leaf area index in \(m^2\) foliage /\(m^2\)land used as a parameter, \(A_{Vi}\) is the extent of vegetation \(i\), \(A_T\) is the total area of the domain and \(\phi_{Vi}\) is LAI to biomass conversion factor (0.38 KG/LAI from Suganuma et al. (2006)) used as a parameter.

Here it is assumed that biomass is linearly related to LAI, according to \(\phi_{Vi}\) (e.g. see Suganuma et al. (2006) and Coletti (2014) demonstrated similar models for Western Australian vegetation including Eucalyptus, Melalula and Casuarina).

Litterfall \((L_l)\) and root turnover \((R_d)\) are linearly related to biomass according \(X_{Lt}\) and \(X_{Rd}\), respectively (Friend et al., 1997). Plant respiration is configured as a function of temperature, such that:

\[
R = k_R e^{\mu T}(K_r B_r + K_l B_l)B
\] (4.27)

where \(k_R\) (\(m^2\) \(d^{-1}\)) adjusts the respiration to the hydrological environment such that it is lowest when vegetation is exposed to its preferred hydrological conditions. \(K_r\), \(K_l\) and \(\mu T\) are scaling factors, whose values are given by Running and Coughlan and Running (1989).
4.3. Model Approach

$B_{ri}$ and $B_{li}$ are the fraction of the biomass that is allocated to roots and leaf respectively (Running and Gower, 1991). Respiration from plant stems is neglected.

The gross assimilation of carbon, $\Pi_A$ ($kg\ C\ m^{-2}\ d^{-1}$), is a function of the uptake efficiency $\eta\Pi(kg\ C\ kgCO_2^{-1})$, the potential uptake rate, $\Pi_0$ ($md^{-1}$), $\Delta CO_2$, the carbon dioxide air leaf diffusion gradient ($kgCO_2\ m^{-3}$) (Lohammar et al., 1980) and LAI:

$$\Pi_A = \eta\Pi_0 \Delta CO_2 LAI \tag{4.28}$$

The potential photosynthesis rate is a function of the canopy and mesophyll conductance, $CC$ and $CM\ (ms^{-1})$ respectively, integrated over the day length, $dl\ (sd^{-1})$ (Running and Coughlan, 1988):

$$\Pi_{0i} = \frac{CC\ CM}{CC + CM} dl \tag{4.29}$$

$CM$ is based on a maximum mesophyll conductance ($ms^{-1}$) (Running and Coughlan, 1988), modified by normalizations that account for temperature and solar radiation dependencies, summarized as:

$$CM = CM_{max} CM_{\Phi} CM_t \tag{4.30}$$

where:

$$CM_{\Phi} = \frac{\Phi_C - \Phi_0}{\Phi_C + \Phi_{0.5}} \tag{4.31}$$

and

$$CM_t = \frac{T_{max} - T}{T - T_{min}} \tag{4.32}$$

$\Phi_0$ and $\Phi_{0.5}$ are the photosynthesis light compensation point and the radiation level that causes $CM_{\Phi}$ to be equal to 50% of its maximum. $T_{min}$ and $T_{max}$ are the minimum and maximum temperature for photosynthesis and $T$ is the air temperature and $\Phi_C$ is the solar radiation at the understory or overstorey canopy level (denoted by sub-index u or o, respectively).

The canopy intercepts the incident radiation in a non-linear proportion to its LAI as given by the radiation extinction coefficient, $\phi$, which is set for each vegetation level, u or o. Overstorey vegetation is comprised of trees and understory vegetation comprises grasses and aquatic plants. The absorbed photosynthetic radiation by the overstorey vegetation is (Feikema et al., 2010a):
\[ \Phi_{Cu} = (\Phi_p - \Phi_C)(1 - e^{\phi_u \text{LAI}_u}) \] (4.34)

Similar to CM, CC is based on a maximum conductance $CC_{\text{max}}$, modified by the normalized potential water uptake, $\Psi$ such that:

\[ CC = \Psi CC_{\text{max}} \] (4.35)

where $\Psi$ is a function of the normalized water availability for plant uptake, $\alpha$ and the root length in contact with the water table, $\beta$ (Skaggs et al., 2006). Water availability for plant uptake depends on the soil water content and the water table depth, and follows the principles introduced by Ridolfi et al. (2006) and Muneepeerakul et al. (2008a), whereby different plant functional groups can uptake water from either above the vadose zone, below the water table level, or from both. Any compensation in root water uptake by enhanced water uptake from more moist regions of the soil profile is neglected. Thus:

\[ \Psi = c_o \alpha_U (1 - \beta) + c_o \alpha_S \beta \] (4.36)

Water availability under saturated conditions, $\alpha_S$, is always equal to 1 since there is no water limitation. Conversely, in the vadose zone, water availability for plant uptake, $\alpha_U$, depends on soil moisture ($\theta$) and also on plant characteristics such as the soil moisture at wilting point ($\theta_W$) and optimal soil moisture for plant uptake ($\theta_0$):

\[ \alpha_U = \begin{cases} 
\frac{\theta - \theta_w}{\theta_0 - \theta_w} + 1, & \text{if } \theta < \theta_0 \\
1, & \text{if } \theta_0 \leq \theta < 1 \\
0, & \text{if } \theta = 1 \text{ & if } \theta < \theta_w 
\end{cases} \] (4.37)

This approach follows that of (Feddes et al., 1976), using soil moisture instead of pressure head as thresholds, with example values for $\theta_0$ and $\theta_W$ available from Guswa (2005), who studied plant transpiration under water-limited environments.

Whether plants effectively take water from below or above the phreatic surface (or from both regions) depends on the uptake strategy of the relevant functional group. As a result, each plant group can only obtain water from the wetland environment that matches its hydrological requirements. The model is able to simulate as many plant types are required.
4.4. Model Setup and Parameterisation

By default, two plants, whose type is defined by parameters, are set-up. An example of two vegetation types and their compatibility parameters are described below:

1. Vegetation 1: Plants can take water from the unsaturated and saturated zones
2. Vegetation 2: Plants require standing water conditions

<table>
<thead>
<tr>
<th>Functional Vegetation type</th>
<th>Requirements</th>
<th>Compatible Parameters</th>
</tr>
</thead>
<tbody>
<tr>
<td>Deep rooted</td>
<td>Saturated or unsaturated soil</td>
<td>$c_o = 1; c_u = 1$</td>
</tr>
<tr>
<td>Shallow rooted</td>
<td>Unsaturated soil</td>
<td>$c_o = 0; c_u = 0$</td>
</tr>
</tbody>
</table>

4.4.1 Morphological Information

The wetland is a claypan system, with no significant inflows from surface waters or groundwater. It is therefore considered to be a bowl, which fills seasonally from direct rainfall and due to the low permeability of the sediments, it drains slowly to groundwater and more rapidly due to evaporation. Weirs installed at a height of 450 mm above the base of the wetland limited the depth of inundation.

The wetland morphology, soil, vegetation and initial conditions were based on application of the model to the wetland system at EBNR. A digital elevation model of the area at 1 m resolution was obtained from the Western Australian Department of Parks and Wildlife (DPaW) and was used to derive the area-depth-volume curve for the model.

For the compartmentalised version of the model, the DEM was cut according to the shape of the sub-catchments. The hypsographic curves were calculated from each of the compartment-wise DEMs.

4.4.2 Meteorological Forcing Data

For both models a 20 year period was simulated. To force the model, a time-series of daily weather data from the Australian Water Availability Project was used for the first period, but both of the models were also run using the locally measured meteorological data generated by the on-site weather station to provide the most accurate assessment of the performance of the model in predicting wetland water temperatures (see Figure 4.3).

4.4.3 Parameters

Meteorological parameters were assumed as in Table 4.2. For the compartmentalised model, the hydrological and vegetation parameters were applied similar to Coletti et al.
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Figure 4.3: Climate forcing data as the boundary condition of the model

(2013b), where the model was applied across the range of environments representative of the region, with some minor adjustments (Table 4.2).

4.4.4 Calibration Approach

For both model applications, water level observations taken at EBNR (observed water depth gauge values from DPaW) for the years 1999-2009 were used to calibrate the water balance sub-model (Chapter 2). Observations were derived from visual inspection of up to eight level gauges installed in the centre of deeper pools throughout the wetland, and were made at approximately monthly intervals.

The predicted water table depth was compared against depths recorded by DPaW in bores 5425 and 5423 (see Chapter 2), whose location is depicted in Figure 2.2. These bores were chosen primarily because they are located outside the EBNR.

Calibration of the simple lumped model involved adjustment of three hydraulic parameters and five thermal parameters. The three hydraulic parameters calibrated included the saturated hydraulic conductivity of the lake sediments \( h_c \) (0.1 mm/hour), the adjacent sandy sediments controlling internal water redistribution \( h_{cu} \) (0.11 mm/hour) and the exponent on infiltration rate relation to saturation \( k_i \) (3.0). A simulated annealing algorithm (Belisle, 1992; Byrd et al., 1995) was used in the R framework to obtain the best-fit of these hydraulic parameters by reducing the root mean square error (RMSE) between observed and modelled water levels. A Markov Chain Monet Carlo (MCMC)
approach was trialled for calibration of the complex model but convergence issues meant that only the default parameters applied (in Table 4.2) were able to be presented.

Modelled temperature was compared against water temperature that has been logged every 15 minutes at EBNR since early 2009 using an automated weather station (Unidata 6540V) with a linear temperature probe positioned in the deepest part of the wetland.
Table 4.2: Summary of the parameters and initial conditions used in the lumped and compartmental model (SC1, SC2, SC3 and SC4 for sub-catchment 1, 2, 3 and 4 respectively)

<table>
<thead>
<tr>
<th>Parameter description</th>
<th>Symbol</th>
<th>Units</th>
<th>Lumped Model Value</th>
<th>WET&amp;D Value</th>
<th>Reference/Remarks</th>
</tr>
</thead>
<tbody>
<tr>
<td>Capillary rise fraction</td>
<td>kA</td>
<td>m³m⁻³</td>
<td>10⁻⁶ hS 10⁻⁶ hS 10⁻⁶ hS 10⁻⁶ hS</td>
<td>-</td>
<td>DPaw, unpublished data.</td>
</tr>
<tr>
<td>Soil porosity</td>
<td>f</td>
<td>m³m⁻³</td>
<td>0.4 0.3 0.3 0.3 0.3</td>
<td>Drake et al. (2012); Taplin et al. (2010)</td>
<td></td>
</tr>
<tr>
<td>Soil moisture at field capacity</td>
<td>qf</td>
<td>m³m⁻³</td>
<td>0.45 0.15 0.15 0.15 0.15</td>
<td>Lowered to represent the lower soil moisture experienced by plants</td>
<td></td>
</tr>
<tr>
<td>Vertical hydraulic conductivity</td>
<td>kv</td>
<td>m⁻¹</td>
<td>0.1 0.36 0.28 0.5</td>
<td>Taplin et al. (2010)</td>
<td></td>
</tr>
<tr>
<td>Horizontal hydraulic conductivity</td>
<td>kh</td>
<td>m⁻¹</td>
<td>0.4 0.3 0.3 0.3</td>
<td>Merz (2000)</td>
<td></td>
</tr>
<tr>
<td>Vertical hydraulic conductivity of the deepest soil layer</td>
<td>qv</td>
<td>m⁻¹</td>
<td>4.4 10⁻⁴ 4.4 10⁻⁴ 4.4 10⁻⁴ 4.4 10⁻⁴</td>
<td>This study.</td>
<td></td>
</tr>
<tr>
<td>Recoission coefficient for infiltration</td>
<td>ki</td>
<td>-</td>
<td>3 3</td>
<td>-</td>
<td></td>
</tr>
<tr>
<td>Pan-to-lake evaporation factor</td>
<td>c</td>
<td>-</td>
<td>0.8</td>
<td>Dobramusi et al. (2003)</td>
<td></td>
</tr>
<tr>
<td>Maximum LMI for carbon uptake</td>
<td>LAM</td>
<td>m³m⁻³</td>
<td>2</td>
<td>Nemani and Running (1989)</td>
<td></td>
</tr>
<tr>
<td>before restrictions</td>
<td>Albico</td>
<td>a</td>
<td>0.25</td>
<td>Ward and Trimble (2004)</td>
<td></td>
</tr>
<tr>
<td>Specific heat of water</td>
<td>Cp</td>
<td>Jkg⁻¹ K⁻¹</td>
<td>0.007</td>
<td>Ward and Trimble (2004)</td>
<td></td>
</tr>
<tr>
<td>Ratio leaf to foliage biomass</td>
<td>XLI</td>
<td>kgCO₂kg⁻¹</td>
<td>0.003</td>
<td>Friend et al. (1997)</td>
<td></td>
</tr>
<tr>
<td>Ratio root litter to root biomass</td>
<td>XRL</td>
<td>kgCO₂kg⁻¹</td>
<td>0.01</td>
<td>Friend et al. (1997)</td>
<td></td>
</tr>
<tr>
<td>Scaling factor that relates temperature to respiration</td>
<td>Kr, Kl and mT</td>
<td>-</td>
<td>0.095, 0.00015, 0.000001</td>
<td>Running and Coughlan (1988)</td>
<td></td>
</tr>
<tr>
<td>Carbon dioxide air leaf diffusion gradient</td>
<td>DCO₂</td>
<td>kgCO₂m⁻³</td>
<td>0.0007</td>
<td>Lehmann et al. (1980)</td>
<td></td>
</tr>
<tr>
<td>Maximum mesophyll conductance</td>
<td>CMmax</td>
<td>m s⁻¹</td>
<td>-1</td>
<td>Running and Coughlan (1988)</td>
<td></td>
</tr>
<tr>
<td>Photosynthesis light compensation point and boundary conditions</td>
<td>F0 and kJ m⁻²</td>
<td>432</td>
<td>Running and Coughlan (1988)</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Radiation level to normalize solar radiation</td>
<td>F0.5</td>
<td>kJm⁻²</td>
<td>9730</td>
<td>Running and Coughlan (1988)</td>
<td></td>
</tr>
<tr>
<td>Radiation extinction coefficient (V2, V3 and V1)</td>
<td>j</td>
<td>-</td>
<td>0.5</td>
<td>Fekoms et al. (2016b)</td>
<td></td>
</tr>
<tr>
<td>Maximum and minimum photosynthesis temperature</td>
<td>Tmax, Tmin</td>
<td>°C</td>
<td>37, 0</td>
<td>Running and Coughlan (1988)</td>
<td></td>
</tr>
<tr>
<td>Photosynthetically active radiation</td>
<td>Fp</td>
<td>kJm⁻²</td>
<td>50% of solar radiation</td>
<td>Lundberg and Waring (1997)</td>
<td></td>
</tr>
<tr>
<td>Maximum canopy conductance</td>
<td>Cmax</td>
<td>m⁻¹</td>
<td>0.0036</td>
<td>Running and Coughlan (1998)</td>
<td></td>
</tr>
<tr>
<td>Maximum LMI for n environment (n=U, V and L)</td>
<td>LAMmax</td>
<td>m³m⁻²</td>
<td>2</td>
<td>Arbitrarily defined.</td>
<td></td>
</tr>
<tr>
<td>Maximum precipitation interception</td>
<td>Imax</td>
<td>m</td>
<td>0.002</td>
<td>Arbitrarily defined.</td>
<td></td>
</tr>
<tr>
<td>Vertical root depth, hr</td>
<td>V1: Upper</td>
<td>m</td>
<td>1.4</td>
<td>Drake et al. (2012)</td>
<td></td>
</tr>
<tr>
<td>Preferential uptake depth observed in the field</td>
<td>V1: Upper</td>
<td>m</td>
<td>10m above hS</td>
<td>DFC unpublished data</td>
<td></td>
</tr>
<tr>
<td>Soil potential that stops carbon assimilation</td>
<td>U: Lower</td>
<td>m⁻³</td>
<td>-1.8</td>
<td>Drake et al. (2012)</td>
<td></td>
</tr>
<tr>
<td>Wilting point, qW (without solutality restrictions)</td>
<td>m⁻²</td>
<td>0.97</td>
<td>0.1</td>
<td>Drake et al. (2012)</td>
<td></td>
</tr>
<tr>
<td>Optimum soil moisture for carbon uptake, q0</td>
<td>m⁻³</td>
<td>0.45 0.45</td>
<td>Drake et al. (2012)</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Maximum salt concentration acceptable in U, V max U</td>
<td>kgs⁻²</td>
<td>36.3 20.7</td>
<td>Drake et al. (2012)</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Maximum salt concentration acceptable in S, V max S</td>
<td>dSmm⁻¹</td>
<td>12 20</td>
<td>**</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Maximum salt concentration acceptable in L, V max L</td>
<td>dSmm⁻¹</td>
<td>1.8 3.7</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Rate of salt uptake in U, V up U</td>
<td>kgs⁻²</td>
<td>0.2 0.2</td>
<td>Drake et al. (2012)</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Rate of salt uptake in S, V up S</td>
<td>kgs⁻²</td>
<td>0.3 0.2</td>
<td>Drake et al. (2012)</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Carbon uptake efficiency to reach carbon assimilation rate hF U</td>
<td>kgs⁻²</td>
<td>1.05 1.1</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Carbon uptake efficiency to reach carbon assimilation rate hF S</td>
<td>kgs⁻²</td>
<td>0.2 0.3</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Carbon uptake efficiency to reach carbon assimilation rate hF L</td>
<td>kgs⁻²</td>
<td>0.2 0.9</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Respiration to biomass parameter, hF U</td>
<td>m³day⁻¹</td>
<td>10 50</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Respiration to biomass parameter, hF S</td>
<td>m³day⁻¹</td>
<td>10 10</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Respiration to biomass parameter, hF L</td>
<td>m³day⁻¹</td>
<td>20 20</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>lake level</td>
<td>hL</td>
<td>m</td>
<td>0.38</td>
<td>0</td>
<td>George et al. (2004)</td>
</tr>
<tr>
<td>Water table</td>
<td>hS</td>
<td>m</td>
<td>0.07 0.14 0.16 0.18 0.11</td>
<td>More (2000)</td>
<td></td>
</tr>
<tr>
<td>Wetland Temperature</td>
<td>°C</td>
<td>20</td>
<td>This study.</td>
<td>Harell-Lennard and Callow (2008)</td>
<td></td>
</tr>
<tr>
<td>Soil moisture</td>
<td>q</td>
<td>-</td>
<td>0.108 0.45</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>
4.5 Model Performance

The simulated water level over the 20 year period is plotted with the field data in Figure 4.4 to examine the performance and calibration of the simple lumped wetland model. The calibrated model had an Root Mean Squared Error (RMSE) of 69.9 mm between the observed and predicted hydroperiod values over the ten-year period for which we had relevant data from surface water gauging stations (1999 - 2009). In most years the simulated water levels follow the field data. When particularly wet (2005) and dry (2007) years are magnified (figure 4.4 b and c), similar high performance of the model is also seen. The model was further compared against the normalized frequency of hydro-periods, which also followed the field data closely.

Results from the compartmentalised model show more detailed predictions (Figure 4.5). These include the prediction of the different groundwater and lake levels for the different sub-catchments (where data was available), soil moisture from the in-situ soil moisture loggers, and water temperature predictions. Table 4.3 shows the error statistics. The calibrated compartmentalized model for sub-catchment 1 (SC1) had an Root Mean Squared Error (RMSE) of 12.82, whereas the weighted average of the 4 sub-catchment increased to 40.75 mm between the observed and predicted hydroperiod values over the 14-year period for which we had relevant data from surface water gauging stations (1999 - 2014). The other parameters (soil moisture, groundwater and water temperature) followed the field observations even more closely with lower RMSE and Nash-Satcliff efficiency presented in table 4.3. Overall, the model captured the dynamics reasonably well.

The outputs of the sub-catchment 1 (SC1), sub-catchment 2 (SC2), sub-catchment 3 (SC3) and sub-catchment 4 (SC4) (figures 4.6, 4.7, 4.8 and 4.9 respectively) showed the dynamics of eco-hydrology such as the water storages, hydrological processes and vegetation biomass predictions inside a single catchment and thereby demonstrates the usefulness of compartmentalization opposed to using the whole catchment as a lumped model. The ability of the model to hindcast, and therefore potentially forecast, the key microhabitat parameters at this site are clearly demonstrated.

Table 4.3: Error table of the compartmental model. SC 1 columns are for sub-catchment 1 and W. Ave columns are area-weighted averages of the 4 sub-catchments used by the compartmental model

<table>
<thead>
<tr>
<th>Validation item</th>
<th>MAE</th>
<th>NSE</th>
<th>R²</th>
<th>RMSE</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>SC 1</td>
<td>W. Ave.</td>
<td>SC 1</td>
<td>W. Ave.</td>
</tr>
<tr>
<td>Lake height (mm)</td>
<td>9.09</td>
<td>23.37</td>
<td>0.09</td>
<td>-8.18</td>
</tr>
<tr>
<td>Soil Moisture (m³/m³)</td>
<td>0.01</td>
<td>0.02</td>
<td>0.49</td>
<td>-2.95</td>
</tr>
<tr>
<td>Groundwater (m AHDd)</td>
<td>0.53</td>
<td>0.94</td>
<td>0.34</td>
<td>-0.88</td>
</tr>
<tr>
<td>Water temperature (°C)</td>
<td>1.83</td>
<td>1.87</td>
<td>0.41</td>
<td>0.40</td>
</tr>
</tbody>
</table>

a Mean Absolute Error;  b Nash-Sutcliff Efficiency;  c Root Mean Square Error;  d metre from Australian Height Datum
Chapter 4. A model assessment of EBNR ecohydrology

Figure 4.4: Time series of modeled (solid black line) and actual (red dotted line) hydroperiods and water temperatures at Ellen Brook Nature Reserve (EBNR). a) water levels and hydroperiods of EBNR from the WET-R simulation from 1999 to 2009; b and c) magnified views of water levels and hydroperiods for a wet (2005-06) and dry (2006-07) year; d) normalized frequency of actual and simulated hydro periods (months) over 1999-2009; e) actual and simulated water temperature during the 2009 hydroperiod (days 175-326). Adjusted from Mitchell et al., 2013.
Figure 4.5: Validation plots of the compartmental model with field data. Blue solid lines represents simulation from the model and red dashed line with red dots represents field observation.
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Figure 4.6: Simulation results for sub-catchment 1 (SC1)

Figure 4.7: Simulation results for sub-catchment 2 (SC2)
4.5. Model Performance

Figure 4.8: Simulation results for sub-catchment 3 (SC3)

Figure 4.9: Simulation results for sub-catchment 1 (SC4)
4.6 Water Balance Analysis

The model was applied with past, recent and future (median dry) climate to undertake a detailed water balance analysis of EBNR. The output of the eco-hydrological model was used to compute the yearly average of water balance components and decadal averages were calculated to get the decadal values. Two types of water balance were calculated for the reserve. Firstly, we took sub-catchment 1 (SC1) and calculated the water balance by its own as we considered this sub-catchment more representative to generic clay pan wetlands without the presence of the engineering interventions (e.g. weirs and artificial ponds in other sub-catchments). The water balance for SC1 was computed for three decades: past (1991 - 2000), recent (2001-2010), and future (2021 - 2030) (Table 4.4).

For SC1, the recent decade (2001-2010) experienced the driest years (2006 and 2010 being the driest in last 50 years (Silberstein et al., 2012)) which resulted in a 15% reduction in precipitation than the past decade (1991-2000). This reduction in precipitation for the recent decade reduced the outflow to about 28 % from the past decade. For the future decade (2021-2030), the water balance showed further reduction in rainfall (16%) from the past decade resulting in a 27% reduction in outflow.

For the second type of water balance, we computed the area-weighted average of the water balance components (using equation 4.38) for the full reserve and calculating it for past (1991-2000) and recent (2001) decades, as summarised in Table 4.5.

\[
AW_{Ave} = \frac{\sum_{i=1}^{4} A_i \times C_i}{\sum_{i=1}^{4} A_i} 
\]  

(4.38)

where, \(AW_{Ave}\) is the area-weighted component of the water balance, \(A_i\) is the area of the respective sub-catchment and \(C_i\) is the respective component of the water balance to be computed.

Table 4.4: Decadal water budget calculated from the model output of EBNR sub-catchment 1 for three decades, past (1991-2000), recent (2001-2010) and future (2021-2030)

<table>
<thead>
<tr>
<th>Components (mm)</th>
<th>1991-2000</th>
<th>2001-2010</th>
<th>2021-2030</th>
</tr>
</thead>
<tbody>
<tr>
<td>Precipitation, (P_L)</td>
<td>809</td>
<td>692</td>
<td>688</td>
</tr>
<tr>
<td>Lake level (L_L)</td>
<td>160</td>
<td>135</td>
<td>140</td>
</tr>
<tr>
<td>Infiltration (Q_I)</td>
<td>398</td>
<td>391</td>
<td>372</td>
</tr>
<tr>
<td>Percolation (Q_P)</td>
<td>138</td>
<td>142</td>
<td>131</td>
</tr>
<tr>
<td>Runoff (Q_R)</td>
<td>42</td>
<td>39</td>
<td>34</td>
</tr>
<tr>
<td>Seepage (Q_S)</td>
<td>98</td>
<td>78</td>
<td>85</td>
</tr>
<tr>
<td>Lake Evaporation (E_L)</td>
<td>149</td>
<td>139</td>
<td>137</td>
</tr>
<tr>
<td>Total ET (ET_{Tot})</td>
<td>398</td>
<td>369</td>
<td>358</td>
</tr>
<tr>
<td>Net groundwater (GW_{Net})</td>
<td>-12</td>
<td>-11</td>
<td>-11</td>
</tr>
<tr>
<td>Outflow (Q_{out})</td>
<td>235</td>
<td>163</td>
<td>169</td>
</tr>
</tbody>
</table>

To compare the water balance in past and recent decades with SC1 and area-weighted average (\(AW_{Ave}\)), a water balance pathway in representative wetland is illustrated in Figure 4.10.
Table 4.5: Area-weighted water budget calculated from the model output of EBNR for the four sub-catchments for two decades, past (1991-2000) and recent (2001-2010)

<table>
<thead>
<tr>
<th>Components (mm)</th>
<th>1991-2000</th>
<th>2001-2010</th>
</tr>
</thead>
<tbody>
<tr>
<td>Precipitation, $P_L$</td>
<td>809</td>
<td>692</td>
</tr>
<tr>
<td>Lake level $L_L$</td>
<td>297</td>
<td>221</td>
</tr>
<tr>
<td>Infiltration $Q_I$</td>
<td>600</td>
<td>545</td>
</tr>
<tr>
<td>Percolation $Q_P$</td>
<td>198</td>
<td>165</td>
</tr>
<tr>
<td>Runoff $Q_R$</td>
<td>26</td>
<td>23</td>
</tr>
<tr>
<td>Seepage $Q_S$</td>
<td>58</td>
<td>46</td>
</tr>
<tr>
<td>Lake Evaporation $E_L$</td>
<td>177</td>
<td>145</td>
</tr>
<tr>
<td>Total ET $ET_{Tot}$</td>
<td>379</td>
<td>347</td>
</tr>
<tr>
<td>Net groundwater $GW_{Net}$</td>
<td>-157</td>
<td>-150</td>
</tr>
<tr>
<td>Outflow $Q_{out}$</td>
<td>303</td>
<td>216</td>
</tr>
</tbody>
</table>

The differences in rainfall seasonality in the different scenarios caused variations in the most active water pathways between U, S and L. Scenario SC1: 2001-2010 resulted in a flow regime (Figure 4.10) that emphasized infiltration (I), and percolation (Qp) but had low amounts of surface runoff ($Q_R$) and seepage (Qs) relative to scenario SC1:1991-2000. Infiltration (recharge) was 2% lower and the percolation was around 4% higher. On the other hand the infiltration and consequently percolation were minimized in scenario AW_Ave: 2001-2010 but surface runoff ($Q_R$) and a seepage (Qs) were respectively 7% and 5% higher than in scenario AW_Ave:1991-2000. As a consequence of diminished infiltration, leaching (Qp) was also reduced. In all simulations, temperature, wind speed, cloud cover and relative humidity were kept the same, so that the evaporation from the bare soil, Eb_U, responded to only water available at the vadose zone.

This demonstrated how climate change driver can shape the hydrological pathways and how the similar changes can be magnified with larger catchment scales.
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Figure 4.10: Summary of the water balance of EBNR. Panel a. and b. illustrates annual average water balance for sub-catchment 1 of EBNR for 1991-2000 and 2001-2010 respectively. Panel c. and d. represents the water balance of area-weighted annual average of the 4 sub-catchments used in the compartmental model of EBNR for 1991-2000 and 2001-2010 respectively.

4.7 Discussion

Both the simple lumped wetland model, and the compartmentalised wetland model performed well against historic and current hydrological and climate data for EBNR. Both models were able to predict water levels, hydroperiods and water temperature with reasonable accuracy, and both provide a good foundation for approximating how EBNR might change between years and potentially under future climates. However, for the lumped model, major parameters such as runoff generation, seepage and evaporation are based on empirical values and are not transferable to different wetland types and may not be suitable for other types of wetlands under altered climate scenarios. In particular, the lumped model does not account for important ecological processes that can shape wetland hydrology, such as the effect of vegetation, or the role of soil geomorphology. Vegetation can play a significant role in hydro-dynamics in ephemeral wetlands such as EBNR (Thompson et al., 2009) and can have variable roles to play depending on different functional groups of plants (Muneepeerakul et al., 2011; Stefanik and Mitsch, 2012), and how patchy the vegetation is within the domain. Since vegetation controls some main ecosystem processes, by interception loss, soil water uptake, bare soil evaporation, transpiration and shading, it can shape the runoff generation and basin hydrological cycle (Brannen et al., 2015) which is ignored in the lumped model.

On the other hand, the compartmentalised model, while more time and data demanding, affords many advantages over the simple lumped wetland model. By including dif-
ferent vegetation functional groups it can predict the dynamics of vegetation-soil-water feedback (Muneepeerakul et al., 2008b) in relation to rainwater availability and can be customized for specific types of wetland systems. The compartmentalised model was able to predict microclimate features such as soil moisture and water temperature at higher spatial resolution. This model has the flexibility to be used as a groundwater connected or disconnected type wetland and hence is able to quantify the effect of groundwater on wetland hydrodynamics. This generates a more realistic representation of EBNR, and takes into account the spatio-temporal variability of the distinguishable hydrological zones within the reserve.

Having demonstrated that the spatially-resolved wetland models generated here are capable of approximating the historical and current conditions at EBNR, it is now possible to use the models to predict how this wetland will change under future climates, both at the local (EBNR) and regional (SW WA) scales. This can be achieved by forcing the model with current and future rainfall and temperature conditions (from the Australian Water Availability Project, Raupach et al. 2011, and downscaled global climate models, Charles et al. 2010; respectively). Of particular interest are the dynamics and temperatures of standing ponded water. The WST has several key habitat requirements from a wetland perspective; these being a hydroperiod sufficiently long to allow tortoises to accumulate energy reserves capable of seeing them through summer aestivation periods and allowing reproduction (Arnall et al., 2015), an ephemeral type-wetland (to cue breeding behaviours such as nesting; (Kuchling and DeJose, 1989), and water temperatures that allow activity and foraging (Mitchell et al. 2013). As demonstrated by chapters 2 and 3, these aspects will be influenced by soil composition, wetland morphology, and vegetation type, and having captured these elements for EBNR and TSNR, the logical next step is to predict how these two wetland types (claypan and groundwater dependant) will respond to different climatic futures.

One of the limitations of the compartmentalized model was how the vegetation assemblage was defined in the model. In this case, although EBNR was divided into compartments, the vegetation inside each compartment were considered uniform. Further spatial heterogeneity can be adopted in terms of vegetation assemblage by field sampling and remote sensing of the spatial distribution of the vegetation assemblage and incorporating it into the model. This can improve model performance by more precisely defining the vegetation functional group and thereby better predicting the hydrodynamics of EBNR. Another limitation of the compartmentalized model was that it assumed similar infiltration rate for vegetated and bare soil, whereas ideally the latter should have lower infiltration rate. Furthermore, the model assumes a lumped vadose zone that could result in evaporating all the water under unsaturated zone. This could lead to overestimates of bare soil evaporation. One realistic approach to resolve these limitations is to use models with finer spatial resolution for a shorter time scale, and to use the results to determine site-specific parameter feedbacks in the model. An example of this approach is HYDRUS-1D (Simunek et al., 2005, 2008) which disaggregates the soil moisture pattern for a specific type of wetland. Short-term site-specific simulation of this type of 1D model with higher vertical resolution can be set-up to represent the actual geological layers or the lithology of the
site instead of defining a single confining or permeable layer like the compartmental model used here (Twarakavi et al., 2008). Although a detailed on-site study of soil structure and geology is required for this type of model setup, it can achieve more precise parameters (eg. soil hydraulic conductivity, infiltration and seepage rate) for soil-water interaction between unsaturated and saturated zones (Dafny and Simunek, 2016) and thus improve model performance.

These limitations notwithstanding, the compartmentalized model can be recommended for other potential uses with its present functionality. With the flexibility and convenience to customize the algorithm and the sub-routines of each compartment, it has the potential to be coupled with other models that require realistic eco-hydrological feedback to get conclusive insights into complex eco-hydrological processes. An earlier version of this model has already been used in studies for carbon biomass estimation of Lake Tulibin of Western Australia (Coletti et al., 2014) and coupled with an eco-physiological model (NicheMapper™) for a pilot study to screen AC candidate wetlands for WST (Mitchell et al., 2013).

The parsimonious nature of this compartmentalized model means that it is possible to use the outputs of more complex hydrodynamic models as catchment inflow to evaluate long-term hydrological trends (Gillan et al., 2005; Krupka et al., 2007), sediment analysis (Jenkins, 2009) and to predict future trends. An example of this is using TUFLOW 2D, which is a fully spatially resolve two-dimensional hydrodynamic model (WBM, 2008) to estimate the surface water dynamics and use the outputs of this model as catchment inflow in the compartmentalized model. This practice is not restricted to only commercial modelling application like TUFLOW 2D but also other open-source applications, such as HEC-RAS 2D (Brunner, 2001) has similar potential to be used as coupled model. HEC-RAS can bring capabilities to incorporate the power of 3D mesh from GIS layers into the model calculation(Ackerman, 2005; Yang et al., 2006).

Other possible applications of this eco-hydrological model can be as a tool to evaluate different types of wetlands under various management scenarios. Example of such uses can include evaluating the effect of short-term and long-term engineering interventions in wetlands such as structures to improve impoundment and groundwater extraction. The model can also be customized for different geomorphology which allows it to simulate other types of groundwater dependent ecosystems as well as to assess the impact of land use changes. Another advantage of the compartmentalized model is, different functional plant groups are possible to be defined inside the compartmentalized model. The plant groups can be parameterized following a specific vegetation assemblage and therefore can be applied to other ecosystems in addition to semi-arid system. The model can also be applied to evaluate the effect of seasonal hydrological connection between geographically isolated wetlands (GIW) during a flood or wet season (Cohen et al., 2016; Evenson et al., 2015; Golden et al., 2014).
Chapter 5

Using an ecohydrological model to identify optimum assisted colonization sites for the Western Swamp Tortoise under a changing climate

5.1 Chapter Prologue

The final data chapter of this thesis represents the amalgamation of the previous chapters into a screening model useful for aiding in the determination of suitable wetland habitats for the western swamp tortoise (WST, *Pseudemydura umbrina*) under future climate scenarios. Specifically, the validated eco-hydrological model developed in Chapter 4 is applied in a more idealised manner across the eco-region of South-West Western Australia (SWWA).

Two extreme wetland model configurations were used in the screening assessment, comparing the sensitivity of predictions to the level of groundwater connectivity that characterise the systems. The model types were each run with one present (1990-2009) and three future climate scenarios for 2030, and the results were combined with a tortoise thermal reaction norm to calculate habitat suitability index for potential candidate sites within the study region. The differences generated by the two different wetland model types are then explored to make predictions about WST habitat suitability in the future.

5.2 Introduction

Wetlands are governed by ecohydrological processes that control and ultimately shape habitat suitability for biota (Foti et al., 2012). As identified in Chapter 2 (and references therein), wetlands are under increasing degradation due to human-induced pressures such as agricultural and industrial development, as well as urban expansion. In addition to these anthropogenic pressures, climate change is a major threat to many wetland systems due to the ecological impacts of increasing temperatures and altering hydrological regimes (Cole et al., 2006; Dawson et al., 2003; Foti et al., 2012; Junk, 2012; Kennedy et al., 2006; Winter, 2000; Zhang et al., 2010). New and innovative approaches are needed to help managers identify the most effective options to conserve threatened wetland species, which may include improving local habitat, or identifying alternative sites.

This is particularly necessary in the south-west of Western Australia, where a drying trend has been observed over the past three decades, which is predicted to continue (Ali et al., 2012a,b; Barron et al., 2012; Charles et al., 2010; Davis and Froend, 1999; Silberstein et al., 2012) with non-stationarity in climate conditions having major impacts on wetland hydrology (Barron et al., 2012; Coletti et al., 2013b; Dade et al., 2014; Klove et al., 2013;
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Mitchell et al., 2013; Winter, 2000, see also Chapter 4). A drying climate is likely to diminish aquatic habitats within wetlands, and alter other important aspects of ecosystems (such as vegetation cover and salinity), which will negatively impact wetland-dependent communities (Dawson et al., 2003; McMenamin et al., 2008; Pearson and Dawson, 2003). The challenge is magnified for species that are already endangered through small population sizes and a limited ability to migrate to more suitable habitats, such as the WST; which is a Critically Endangered freshwater tortoise endemic to wetlands in the south-west of Western Australia (SWWA) (Burbidge et al., 2010).

The ecological habitat required for the WST is very specific, and certain needs must be satisfied for feeding and aestivation to occur (Burbidge, 1981; Kuchling, 1993, outlined in detail in Chapter 1). Since these requirements are now often marginally met at EBNR and TSNR (Burbidge et al., 2010), introduction to new sites is now considered to be a realistic conservation option (Dade et al., 2014; Mitchell et al., 2013). However, there remains much uncertainty as to where such an introduction should occur (Burbidge et al., 2010; Mitchell et al., 2013). Introducing animals within their range (‘translocation’), or to parts of their former range, have become an increasingly necessary tool in wildlife management both for conservation and other purposes (Chauvenet et al., 2013b; Griffith, 1989; Hancock and Gallagher, 2014; Harris et al., 2013; Loss et al., 2011; Rout et al., 2013; Seddon, 2010). In Australia, conservation introductions have been implemented for a wide range of vertebrate species with varying levels of success (Seddon et al. 2015). However, many introductions have not been successful (Chauvenet et al., 2013a; Dodd Jr. and Seigel, 1991; Griffith, 1989), and these movements can also be very expensive (e.g. Rahbek, 1993), or risky should the new species become invasive (Mueller and Hellmann, 2008; Ricciardi and Simberloff, 2009a,b). One early review of translocation for reptiles and amphibians reported that only 19% were successful (Dodd Jr. and Seigel, 1991) and Germano and Bishop (2009) found that many of the translocation efforts for reptiles between 1991-2006 had a high failure rate when compared to other taxa.

Despite the mixed results reported in the literature, once a habitat becomes too degraded or if the underlying hydro-climatological drivers shift substantially to no longer support the basic needs of a species, translocations as a management option become emphasised. The more controversial practice of assisted colonization (the deliberate movement of species to climatically suitable habitats outside of their historical range (Hulme, 2005), may be one of the few options for species at risk from climate change to survive in the wild. This decision must be cognisant of the threat of invasion, but where the risk of this is low then a strong case can be made to undertake assisted colonisation as a preferred conservation option (Hancock and Gallagher, 2014; Harris et al., 2013; Lunt et al., 2013; Rout et al., 2013). This is the situation being evaluated for the WST, and this has led to increased interest in identifying sites that may be suitable for supporting WSTs under different climatic futures (Dade et al., 2014; Mitchell et al., 2013).

To choose translocation sites, correlative climate-envelope models based on occurrence data are often used to predict the future range of species that currently or used to have broad distributions (Guisan and Thuiller, 2005). Correlative approaches linearly correlate presence-absence data of a species with the environmental features of its habitat. They
5.2. Introduction

assume that suitability of that site records of abundance at a particular point in time indicates the suitability of that site (Kearney and Porter, 2009). However, this requires 1) good historical data that spans a broad biogeographic gradient, and 2) that the future local environment conditions will respond in a linear way to regional and global drivers of change. A major challenge becomes finding suitable future habitat in the case where historical empirical data is limited and where strong non-stationarity in hydro-climatological conditions are being experienced. This draws into question the ability to build a robust statistical predictions under novel conditions (Dormann et al., 2008).

![Figure 5.1](image)

**Figure 5.1:** Wetlands acts as dynamic filter for climate variables resulting in varying hydroperiods and temperatures that mediates the typical metabolic rate of tortoises

The lack of suitability of climate-envelope models is particularly applicable for wetland dependent species as wetland environments are sensitive to climatic shifts and may amplify changes in climate signals. Wetlands act as a dynamic filter for rainfall and temperature, in terms of how stochastic forcing will ultimately manifest in habitat provision (e.g., Park et al., 2014). The wetland eco-hydrological system responds in a dynamic fashion to the duration of wet days (hydroperiods) which varies with location and depending on the type of wetland (for instance on the degree of groundwater connectivity). In addition, changes in water extent and vegetation productivity may result from altered temperatures, with water temperature subsequently change depending on the depth of water and shading by the vegetation canopy (Fig. 5.1). Eco-hydrological factors such as hydroperiods and water temperatures play a crucial role for feeding, breeding and ultimately survival of wetland species.

Wetland properties relevant to the WST will not change linearly in response to climate changes, and because the species is so restricted in its distribution, there are not enough occurrence data points with which to construct a correlative species distribution model (Mitchell et al., 2013). Mechanistic models offer an alternative way in which to model species distributions independent of occurrence data (Kearney 2009). For the WST, an integrated approach based on mechanistic models that are able to quantify the specific processes which drive the tortoises’ functional response to changes in resources/environment is required to identify potential assisted colonization sites (Mitchell et al., 2013). This ap-
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approach should account for the eco-physiological needs of the tortoise as linked to wetland habitat quality, which is anticipated to vary under future climate change scenarios and impact upon habitat stability and persistence. To date, there has been no work linking the eco-hydrology of wetlands and organism eco-physiological requirements for the purposes of supporting assisted colonization of Critically Endangered species (with the exception of preliminary models generated for this broader project; Mitchell et al., 2013).

It is the aim of this chapter to undertake a ‘screening assessment’ to identify how future changes in climate will affect wetland habitats across south-western Western Australia through a mechanistic approach. Specifically, the eco-hydrological model used to simulate critical natural habitat for the WST at Ellen Brook nature reserve (EBNR; a clay-pan wetland system), presented in Chapter 4, is run across a grid of sites across the south-western Western Australian region. Since many potential translocation sites are not clay-pan systems, we further explore how robust non-clay-pan systems would be, since they are surface expressions of regional groundwater systems and these might not respond to changes in rainfall in the same way as clay-pans would.

In order to assess habitat suitability, I used an idealised version of the Chapter 4 eco-hydrological model to represent more general wetland response in south-western Western Australia and compute the long-term responses of hydroperiod and water temperature in both present and future climates. To predict the future eco-hydrology of the wetlands in the screening process under climate change, I tested both wetland types with three future climate scenarios: a future wet climate, a future median climate, and a future dry climate (Charles et al., 2010). To capture the physiological response of the WST, a thermal reaction norm (resting metabolic rate: Arnall et al., 2015) was implemented using outputs from the eco-hydrological model predictions in order to calculate a general suitability index for tortoise activity. This considers both the ideal hydroperiod and the most suitable water temperature range to allow WSTs to be active and feed. Therefore, the outcome of this analysis is to firstly demonstrate how sensitive the predicted geographic range of suitable sites is to projected future climate scenarios, and secondly, to demonstrate how the degree of wetland groundwater connectivity influences the suitability of potential translocation sites across south-western Western Australia.

5.3 Screening Setup

5.3.1 Study Area and Grid Creation

The validated model of Chapter 4 was initially applied across the part of the South-West Eco-region of Australia (black shaded region in the inset of Fig. 5.2). There were 34 sites randomly distributed inside the South-West Eco-region (represented in grey triangles in Fig. 5.2) where hourly climate data was used from 1990 from the Australian Water Availability Project (Raupach et al., 2009, 2011). The study area was divided into 1.5 degree square grids equally dividing the region into 2400 grids. The grid was created in ArcMap 10.2 (ESRI, 2014) and later used for disaggregating the results of the 34 stations across the grids.
5.3. Screening Setup

Figure 5.2: Study area with the grey triangles representing the location of the AWAP sites and the black circle indicating the location of Ellen Brook Nature Reserve. Black area inside inset indicates the area covered by the grid within South-West Western Australian eco-region.

5.3.2 Model Setup and Input

Wetland Morphological Types

The eco-hydrological wetland model used here was described in Chapter 4 of this thesis. The model was developed and validated in Chapter 4 and is highly site specific, as the study site has engineering interventions such as constructed weirs and controlled outlets. The model therefore was developed as a compartmental model to distinguish the sub-catchments. To simulate the common types of natural wetlands in South-West Western Australia, we adapted the previously mentioned compartmentalized model with a more generic, parabolic, wetland shape without compartments and only two extremes in terms of groundwater connectivity. The first extreme is a clay-pan type without connection to groundwater, representing wetlands with a clay-pan or clay over sand with low infiltration rate and no seasonal groundwater feedback. The second is groundwater-fed type wetland...
which represents wetlands with a more sandy soil or sand over clay with high infiltration rates with seasonal feedbacks from groundwater.

The model in Chapter 4 used the actual shape from a Digital Elevation Model (DEM) of the wetland (EBNR) based on the calculated hyposgraphic curve. For the purposes of investigating the response of an idealised wetland, the wetland was adapted to adopt a simple cylindrical geometry. The lake-bed has a simple parabolic shape that is embedded in the centre of this cylinder, above the deepest point of the wetland. The maximum radius of the paraboloid is limited to the domain boundary, and the relationship between lake storage \( L_v \) and lake height \( h_L \) is given by the paraboloid equation:

\[
h_L = \sqrt{\frac{2L_v b}{\pi}}
\]  

(5.1)

As in Chapter 4, the model divides the wetland into functional zones including the saturated and unsaturated soil compartments and the lake. These vary in response to the hydrological processes that are controlled by climate, soil properties, groundwater level, and vegetation. The model predicts hydroperiods, water level, soil moisture, water temperature, and other eco-hydrological responses shaped by the climate forcing data for each station.

**Model Input**

The adapted wetland model takes the climate forcing data of rainfall, air temperature, relative humidity, wind speed, and solar radiation as inputs for daily time-steps frequencies. Each of the 34 sites used its own set of climate input data. The model parameters used in this study were the same used in Chapter 4, except for water table height \( h_S \) which was 0.4 m and 8.0 m for clay-pan type and groundwater-fed type wetlands, respectively.

**Calculation of Hydroperiod and Water Temperature**

Hydroperiods were calculated for each simulated year from the water level prediction in the wetland for each station. A minimum level of 5 cm of water was classed as ‘wet’ in the hydroperiod calculation, and the sum of all the ‘wet’ days in a year was taken as the hydroperiod of that year.

Water temperature was calculated from the following linear regression equation:

\[
W_T = 0.678 * A_T + 5.325
\]  

(5.2)

where, \( W_T \) is water temperature and \( A_T \) is air temperature with \( R^2 = 0.647 \) and \( p < < 0.00012 e^{-16} \) derived from Fig. 2.13 of Chapter 2.

**Incorporation of a Tortoise Thermal Reaction Norm \( f(T) \)**

A thermal tolerance function, \( f(T) \) was introduced into the model based on the initial work of Arnall et al. (2015). The function is adapted from an Arrhenius equation (equation 5.3) and uses the rate of carbon dioxide production \( \dot{V}CO_2 \) WST produced when measured
as a function of temperature (in Kelvin). This indicates the standard metabolic rates for WST over the typical range of temperatures at which they are active (14°- 28°C):

\[
 f(T) = \frac{e^{\left(\frac{T_{AH} - T_{ref}}{T}ight)}}{1 + e^{\left(\frac{T_{AL} - T_{L}}{T}ight)}} + e^{\left(\frac{T_{AH} - T_{H}}{T}ight)} \dot{VCO}_2 \\
\text{(5.3)}
\]

where, \(T_{AH}\) is the Arrhenius temperature in Kelvin, \(T_{AL}\) is the Arrhenius temperature for lower boundary (in Kelvin), \(T_{AH}\), is the Arrhenius temperature for upper boundary (in Kelvin), \(T_L\), lower boundary tolerance range (in Kelvin), \(T_H\), the upper boundary tolerance range (in Kelvin), \(T_{ref}\) is the temperature for which rate parameters are given (in °C), \(\dot{VCO}_2\) is an adjustment factor to be multiplied for adjusted \(\dot{VCO}_2\) taking 20°C as a reference point.

The critical thermal maximum for WST was set at 42.5°C (after Burbidge, 1981), and the critical minimum was set as 5°C after Lucas et al. (1963) and Leung (1963) who reported the WST withdrew into their shells and became torpid at approximately 10°C. Activity of WST were therefore only assumed to occur between 14°C and 28°C, after Mitchell et al. (2013).

\[\text{Figure 5.3: A thermal reaction norm for the WST, based on empirical data on rates of CO}_2\text{ production of resting adults (Arnall et al. 2015). Red dots (from left to right along the dashed curve line) indicates measurements at 5°, 14°, 28°and 42.5°C. The dashed ine indicates the fit of Eq 5.3}\\
\]

\textit{Calculation of a Suitability Index}

The suitability index was calculated through the tortoise thermal tolerance function, \(f(T)\) (described above) and was applied conditionally based on water inundation and water temperature. The index was calculated in a scale of 0 to 1, where 0 is the least suitable and 1 the most suitable value for the WST by summing over the suitable period of tortoise
activity:

\[ SI = \frac{1}{N-S} \sum_{i=S}^{N} f(T_i) \]  

(5.4)

The function was applied by setting the start (S) and end (N) dates according to the following conditions: a) where the hydroperiod was within the wet season (May to December) and b) where the water temperature was within the maximum activity range of the WST, i.e. 14°- 28°C (Burbidge, 1981; Burbidge et al., 2010; Mitchell et al., 2013). Any hydroperiod from January to April each year was excluded as WSTs often remain in aestivation during this time. Further, hydroperiods of less than five months were also excluded because juveniles WSTs require this time to attain a minimum body weight of 18g which is required for them to be able to survive their first aestivation period (Burbidge et al., 2010; Mitchell et al., 2013).

5.3.3 Climate Scenarios and Model Application

The two types of the model configuration (based on connectivity to the groundwater) were applied across the 34 sites for a total of four climate scenarios (one present climate, and three future climate scenarios). The present climate scenario encompassed the time period from 1990 - 2009, and the future scenarios were run from 2010 - 2030. Climate data for each scenario was sourced from the Australian Water Availability Project (AWAP), which is statistically downscaled from the Global Circulation Models (GCM) of CSIRO (Raupach et al., 2009, 2011).

To model and predict future climates, Charles et al. (2010) identified three possible future climate scenarios for 2030 based on green-house gas emission data. To compare how the spatial extent of the suitability index varied under future climate conditions, we ran the above analysis for the year 2030 by adjusting the AWAP data with projected future climates based on 16 GCMs, each driven by one of three future greenhouse gas emission levels (Charles et al., 2010). The three climate scenarios used were “wet future” (0.7°C increase in air temperature, -1% decrease in rainfall), “median future” (1°C increase in air temperature, -7% decrease in rainfall) and “dry future” (1.3°C increase in air temperature, -14% decrease in rainfall). Temperatures of the AWAP data were increased and rainfall was modified for all 34 sites according to these future climate scenarios, and these were applied to the meteorological boundary condition file for the wetland model.

Figure 5.4 illustrates how the ecohydrological model was applied for this screening study. For hydroperiod maps, the yearly mean of 20 years (both for present and future climates) was calculated as the mean hydroperiod of a site. The mean hydroperiods were then interpolated using a simple Kriging algorithm (Pebesma, 2004) to produce interpolated spatial extents of hydroperiods across the region. For suitability index maps, yearly mean index values were calculated for each of the stations and interpolated using the same simple kriging algorithm to delineate the spatial extents of suitability index across the grids. The interpolation maps of hydroperiods and suitability indices were further
examined for coverage by acceptable values. For the hydroperiod, cells with more than 180 days were identified, and for suitability index cells with more than an 80% suitability index value were classed as suitable. The area of these acceptable regions was computed as a percentage of the total domain (2400 grid cells).
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**Figure 5.4:** Flow diagram illustrating the application of the ecohydrological model to the screening process for identifying ideal assisted colonisation sites for the WST.
5.4 Wetland Scenario Analysis

5.4.1 Patterns of Hydroperiod

Here we explore the results of the idealised eco-hydrological model and assess how the two types of wetlands (based on groundwater connectivity) performed across the study domain under the different climate forcing conditions. The resulting hydroperiods varied significantly depending on the wetland type, with longer hydroperiods predicted for groundwater fed type (GFT) wetlands than clay-pan type (CPT) wetlands. Under current climates (Figure 5.5 i).a and ii).a), the GFT hydroperiod distribution varied from 240 days around Perth to east of Bunbury, down to 80 days around the north-eastern and eastern boundaries of the domain. The region with the longest hydroperiod was a narrow strip in the south-eastern end of the domain between Walpole and Albany. For CPT wetlands, scenarios for the current climates produced much shorter hydroperiods. The region with the longest hydroperiods occurred slightly to the south of the current habitat EBNR. Hydroperiods of 120 days occurred from were found around Perth to the east of Bunbury, and in the slightly northern parts of the southern coast from between Augusta to Albany. The remainder of the domain was drier (<120 days; Figure 5.5 i).a and ii).a).

Under a wet future there was less variation in hydroperiod lengths between the two wetland types. The overall distribution pattern of hydroperiods was similar for CPT wetlands and GFT wetlands, (Figure 5.5 i).b and ii).b), with the only notable difference being the distribution of wetlands with the highest hydroperiod lengths (320 days). For GFT wetlands, these areas were located to the east of Mandurah and near Walpole in the south, whereas in CPT wetlands, there was only one location (near Walpole) that showed a high (320 days) hydroperiod length.

For the median future climate scenario (Figure 5.5 i).c and ii).c), the highest hydroperiod lengths (240 days) were found across a large area for GFT wetlands, starting from south of EBNR and stretching to the east of Bunbury. By contrast, CPT wetlands that showed high hydroperiod lengths only spanned a short extent of the South-West eco-region, and were constrained to a small elliptical area around the south-western side of EBNR.

With a dry future scenario, GFT wetlands had higher values for hydroperiod extent (>160 days) than CPT wetlands (Figure 5.5 i).d and ii).d). There was some notable spatial variation in the hydroperiod pattern and distribution, with GFT wetlands showing two locations for >160 days of hydroperiod (one immediately next to EBNR, and another at the north-west side of Walpole), while by contrast, CPT wetlands did not exceed 160 days for the same locations. CTE wetlands generally had hydroperiod lengths of 40-120 days across the domain.

5.4.2 Patterns of Suitability Index

The location specific outputs such as hydroperiod and water temperature from the idealised eco-hydrological model were used in combination with the tortoise thermal reaction norm, \( f(T) \) to compute a Suitability Index (SI). This was interpolated across the 2400 grids of the study site, as shown in Figure 5.5 iii)a. to iv)d.
The SI values for GFT and CPT wetlands varied considerably, demonstrating the sensitivity of both wetland types to the individual climate scenarios. Under current climates (Figure 5.5 iii.a and iv.a), the variations were mainly in the mid-range values (0.6 to 0.7) for both wetland types. For GFT wetlands, this mid-range SI extended from near Geraldton in the north to Augusta in the south, covering all of the domain in between. There was a higher SI range (0.8) around EBNR, and the highest SI (0.9) corresponded directly to the EBNR site. By contrast, mid range SI values covered a smaller domain extent for CPT wetlands. This spanned from Geraldton to Bunbury, and had comparatively less coverage in the eastern part of the domain. Like GFT wetlands, CPT wetlands also showed high SI values (0.7) around EBNR, with the highest SI value (0.8) on the EBNR location.

Both GFT and CPT wetlands were predicted to have a very high SI value (<0.7) over most of the study domain under the future wet climate scenario (Figure 5.5 iii.b and iv.b). For CPT wetlands, SI values <0.7 covered most of the domain, with the highest value of 1.0 around the EBNR location. The GFT wetland showed a similar pattern for high SI values (<0.7), and differed slightly by producing lower SI values (0.6) from the east of Denmark to eastern end of the study domain.

For the median future climate scenario, higher SI values covered a smaller extent compared to the wet future climate in both GFT and CPT wetland types (Figure 5.5 iii.c and iv.c). In CPT wetlands, higher SI values (<0.8) corresponded to a narrow strip from the south of Geraldton to Mandurah, with the highest SI (0.8) occurring around EBNR. For GFT wetlands, higher SI values (0.8) covered more of the domain than CPT wetlands, and generally extended eastwards from Augusta.

For the dry future climate scenario, both wetland types showed the lowest SI values which did not exceed 0.6 in any part of the domain ((Figure 5.5 iii.d and iv.d). Similar to the above trends, CPT wetlands had a lower SI values than GFT wetlands, but in this case the difference between the wetland types was more pronounced. The highest SI value in the CPT wetland was 0.6 (occurring around EBNR) whereas the highest SI value (0.7) in the GFT wetland occurred just south of EBNR. Most of the domain for CPT wetlands had low SI values of 0.2 to 0.4, whereas GFT wetlands had much larger areas of higher SI values (0.6), generally covering the areas around EBNR, near Mandurah, Margaret River and Albany.

It is particularly noticeable that in all instances of wetland types and climates scenarios, the highest SI values were found around the EBNR region, or just south of EBNR. Compared to the hydroperiod plot, this reflects the fact that the warmer temperatures in the north of the domain were able to offset the shorter hydroperiods.
5.4. Wetland Scenario Analysis

Figure 5.5: Screening maps of study site showing hydroperiods (in blue hue) and suitability index (in red-orange hue) for WST for clay-pan and groundwater-fed type wetlands. Panels a, b, c, and d are climate scenarios representing the present climate, a future wet climate, a future median-wet climate, and a future dry climate, respectively. Panels in rows i) and ii) are hydroperiods of clay-pan and groundwater fed type wetlands, in a colour scale of dark blue to light blue representing 40 to 360 days of hydroperiods, respectively and panels in rows iii) and iv) are a suitability index for WSTs in a colour scale of yellow to red representing 0.2 to 1.0 suitability index for clay-pan and groundwater-fed type wetlands, respectively.
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5.5 Climate Scenario Analysis

The interpolated hydroperiod and suitability indices were checked for coverage by particular values. For cells with more than 180 days of hydroperiod and SI values of more than 0.8, the percentage of the total domain is plotted in Figure 5.6. The resulting figure highlights the large variations in hydroperiod and SI for the different climate scenarios.

For hydroperiods of more than six months (<180 days) (Figure 5.6 a.), the present climate (Pr) had 30% and 70% domain coverage for CPT and GFT wetlands respectively. For wet future climate (Fw) and median future climate (Fm), the domain coverage ranged from 80% to 90% for CPT wetland and 82% to 95% for GFT wetland. For a future dry climate (Fd), CPT wetlands covered 30% of the total domain whereas GFT wetlands had 62% coverage of the total domain.

For the WST suitability index (SI) values of more than 0.8 (Figure 5.6 b.), were considered to be highly suitable, under the present climate (Pr) only 3% and 6% of the screened area was highly suitable for CPT and GFT wetlands, respectively. The highest percentage of the domain was covered in the future wet climate (Fw), which was 18% and 31% for CPT and GFT wetland, respectively. In case of future median climate (Fm), the CFT and GFT wetland occupied 5% and 12% of the total domain respectively. The lowest amount of coverage was found under future dry climate (Fd) which was 0% and 2% for CPT and GFT wetland respectively.

In examining the spatial variability of the suitability index, the values varied between the four different climate scenarios such that SI values under a future dry climate were the lowest, with high spatial variation for both wetland types. The highest SI values occurred in the future wet climate scenario, with low spatial variation. The future median climate and the present climate had the second and third highest SI values, respectively. Spatial variation across the domain in the future median and present climate scenarios were lower in GFT wetlands than CPT wetlands.
5.6 Discussion

5.6.1 Projected Regional Changes in Wetland Ecohydrology

This study has served to quantify how sensitive wetland hydroperiods are to climate change across the South-West eco-region of Western Australia, and this is explored for wetlands that are isolated from the regional water table (CPT) and wetlands that are highly dependent on groundwater connectivity (GFT). The regional climate data came from the Australian Water Availability Project (AWAP) project that monitors the state and trend of the terrestrial water balance of the Australian continent, using model-data fusion methods to combine measurements and model predictions. The AWAP initiative determines the past history and present state of soil moisture and all water fluxes contributing to changes in soil moisture (rainfall, transpiration, soil evaporation, surface runoff, and deep drainage), across the entire Australian continent at a spatial resolution of 5 km (Raupach et al., 2009, 2011). In this assessment, 34 sites were used to capture the climate variability across the region. However, should super-computing resources become available in the future, then higher resolution sampling from the AWAP dataset could allow for the development of improved surfaces of the suitability index.

The future climate scenarios used apply a gradual increase in temperature and reduction in rainfall according to the mean trajectories predicted by an ensemble of GCM simulations (Charles et al., 2010). In reality, the forecast rainfall patterns would vary in space and time, and therefore the future projections may differ in a more complex way. However, for the purposes of the screening assessment, the use of the ensemble prediction gives an indication of future climates by which we can assess wetland sensitivity. Some shifts in seasonality and rainfall characteristics may lead to more complex wetland hydrological responses, e.g. if the intra-annual rainfall distribution is impacted differently.

Figure 5.6: Percentage of domain covered by Hydroperiods < 180 days (panel a.) and high Suitability Index (<0.8) in four climate scenarios (Pr = Present, Fw= Future wet, Fm = Future median and Fd = Future dry) for clay-pan and groundwater-fed (CP, in red and GW, in green) types of wetland.
Chapter 5. Using an eco-hydrological model for AC

across the region (eg. Coletti et al., 2013a). More recent ensemble climate projections are being developed, and it is recommended that the analysis can be repeated using higher resolution climate projections, for example, by using updated predictions from the latest Climate Model Inter Comparison Project (Karl et al., 2012; Taylor et al., 2012).

This screening process reduces some of the uncertainty surrounding of translocation site selection based solely on hydrological suitability. Due to the diversity of wetland systems that may be encountered in the field, it was not the aim to predict the response of individual sites situated within the landscape, but rather I assumed wetland characteristics were constant and so simulated two extreme wetland morphological types as a way to demonstrate the range of sensitivities that may be observed across the region and into the future. Through the application of the model in Chapter 4 I am reasonably confident in the ability of the model to capture the dominant hydrological processes for clay-pan wetland systems (the model has also been applied to groundwater dependent systems in the Swan Coastal Plain (Coletti, 2014; Coletti et al., 2014; Hanna et al., 2011). Specific sites will have a different balance of fluxes from the model configurations adopted here, e.g. if the water balance was dominated by tributary inflows then the impact of a drying climate would be amplified or dampened depending on how the broader catchment would respond to the changing rainfall trend. Similarly, the groundwater fed systems may be sensitive to regional changes in groundwater recharge and abstraction (Ali et al., 2012b; Barron et al., 2014) and this may also change the sensitivity of the signals reported here.

The screening process is therefore designed to identify (in general terms) regions that will provide potentially suitable introduction sites for the WST in order to mitigate the impacts of climate change, and attributes of wetland hydrology that conservation managers should be aware of. For example, the results of the screening process found that groundwater-fed wetlands offer a more persistent hydroperiod than clay-pan type wetlands, which was most noticeable in the southern region and around EBNR. This indicates that groundwater-fed type wetlands may be more resilient to a drying climate trend than shallow clay-pan systems which rely on rainfall alone. Future climate projections, assuming a median decrease in precipitation and increased temperature, restricted the high suitability index areas by around ten percent from current values, with large areas in the north of the current habitat range becoming less suitable. This includes the area of the EBNR wetlands system, suggesting it will become increasingly unviable. As highlighted in Chapter 2, the present climate conditions used are in fact representing a very dry period already. Before any conclusive decisions are made on site suitability, investigation of specific candidate sites are required in order to develop well-calibrated local models that are specifically tailored to local conditions.

5.6.2 Controls on Habitat Suitability

The spatial variability of the suitability index (SI) across the South-West ecoregion indicates that hydroperiod length, in conjunction with water temperature, plays a crucial role in determining the suitability of wetland habitat for WSTs. Inadequate water levels and/or hydroperiod lengths will directly impact WSTs by reducing their capacity to forage, and thereby grow. WSTs need a minimum five-month hydroperiod for the juveniles
to grow large enough (18g) to survive their first aestivation period (hatchlings can die of dessication during aestivation if they do not reach an adequate size; (Burbidge, 1967, 1981)). Short hydroperiods or very shallow water reduce the amount of time WSTs have to accumulate energy stores, which in turn influences survivorship and reproduction (Kuchling 1993). The SI takes this into account by giving higher values to those locations which produce long hydroperiods.

However, even if the duration and depth of wetland hydroperiods is adequate, locations can be unsuitable if the local water temperatures do not reach the levels required for adequate growth. WSTs are only active between water temperatures of 14°C to 28°C (Mitchell et al., 2016), and will either leave the water to bask or become inactive in the water if temperatures exceed this range (Leung, 1963; Lucas et al., 1963). I identified some cases where the SI value was low due to the colder climate in some southern parts of the ecoregion, despite the required hydroperiod being easily met under present, median future and dry future climate projections. An interesting outcome would be to determine if the increased temperatures predicted under climate change could compensate for shorter hydroperiods by increasing the metabolism and rate of growth (eg. Mitchell et al., 2011). This could be an important consideration that impact translocation decisions, but more research on this topic is required before this mechanism can be clearly understood.

Assessing broad indicators such as hydroperiod and water temperature is obviously a simplification of complex relationships between the WST and its environment. Variability in climate, and local scale heterogeneity in vegetation and geomorphological parameters can also determine the extent of habitat suitability. For example, if the air temperature increases or decreases unexpectedly, it may immediately effect the water temperature which can cause animals to leave or avoid entering the water for feeding. Alternatively, prolonged drought periods or floods could impact population viability in individual years, making the site unsuitable. Detailed micro-habitat features such as shading, vegetation, slope, and topography can be indirect or additional controls for habitat suitability (as introduced in Chapter 3). Shading is primarily related to the types and density of vegetation, and can control the water temperature as well as influencing nesting and aestivation behaviours of WSTs(Kuchling, 1993). The topography and slope can also influence factors such as vegetation type, ponding duration, direction of shading, and the water holding capacity of soils. Other hydrological factors such as soil conductivity, infiltration rate, and seepage can influence hydroperiod, and water salinity can further contribute to the suitability of a habitat. Developing methods able to better link WST physiological needs and life-cycle attributes to local-scale environmental properties is ultimately needed to improve confidence in predictions of future suitability (e.g. Mitchell et al. 2013).

5.6.3 Implications for Translocation Decision

The projected change in wetland hydrology is only one factor influencing decision-making around introductions of the WST outside their historical range. The early results of this hydrological study have already been used in a spatially-explicit multiple criteria decision analysis (MCDA) to assess how a combination of regional and local factors can influence suitability of candidate wetlands for assisted colonisations (Dade et al., 2014). Despite
uncertainties in the above predictions, the results do highlight the general trends that will be faced over the next two decades, and these can be factored into management plans for the WST. For example, the results clearly indicate the increasing likelihood that translocation sites to the north of EBNR will be less suitable, but that sites on the Swan Coastal Plain as far south as Bunbury will likely remain acceptable. This potentially may influence the current practice of releasing captive-bred WST to the two northern-most translocation sites at Mogumber Nature Reserve and Moore River National Park.

The connectivity of the wetland with groundwater appears to be an important factor governing the long-term suitability of wetland habitats. Wetlands that are buffered by groundwater seepage inputs showed higher hydroperiod lengths than clay-pan wetlands. Apart from the two examples of groundwater connectivity explored in this study, there are other wetlands with intermediate levels of connectivity that may show similar patterns when screened across the SW ecoregion. Assessing the specific nature of groundwater to surface water interactions in potential candidate sites is therefore essential, and the model presented here can then be adjusted appropriately to fit specific locations. Another factor to include is the amount of catchment inflow into a wetland, as the presence of an inflow can buffer the effect of climate change and may increase the chances of a habitat being suitable.

Therefore, in order to develop local models of candidate sites for assisted colonisation, it is essential that hydrological monitoring is in place before translocations are undertaken. There is scope to use hi-resolution data input, in addition to the generation of vegetation and soil moisture maps, and to compare these with the past and future scenarios from the CSIRO AWAP model. Future work demands more detailed analyses of the eco-hydrological parameters of the model to specify the unique characteristics of real wetlands, and to come to more robust conclusions about site suitability.

This study illustrates the potential of using simple ecohydrological models to substantially aid the complex decision making processes for threatened species, and has the potential to be applied for assessments of other species that are wetland-dependent. While this study aimed to identify assisted colonisation sites specifically for the WST, the process-explicit approach illustrated here has the potential to be applied to any wetland species. This work has provided a novel science-based approach that links both climate and eco-hydrological changes into the assisted colonization assessment process. It potentially helped to generate a more comprehensive understanding of how the wetland dynamics of WST habitats might change under future climates, and how ultimately, this may impact the availability of natural habitat of the WST as the regional climate becomes warmer and drier.
6.1 Summary and Key Findings

The research aimed to develop an improved understanding of wetland eco-hydrology within the context of the conservation of the Critically Endangered Western Swamp Tortoise (*Pseudemydura ambrina*; WST). The research progressed by firstly exploring long-term hydrological changes that have occurred in the two case study sites (EBNR and TSNR), and then focusing on local-scale variations in properties such as the inundation extent, soil moisture, vegetation, and topography. The field data analysis in Chapter 2 and Chapter 3 was used as the basis to develop an improved conceptual model of the site’s eco-hydrology, and this was subsequently used as the basis to develop a more detailed wetland model. In the final work task, an idealised version of the model was used to demonstrate how wetland hydroperiod and habitat suitability varies across the southwest eco-region and also under future climate scenarios.

This research has shown that the broader climate drying trend observed for Australia and the southwest has had a large impact on the two reserves critically important for the WST, with a drying trend notable from 1968 at the EBNR and TSNR wetlands. This drying trend has shortened the wetland hydroperiods in both reserves, with recent periods being particularly dry. These shortened hydroperiods directly affect the WST by reducing the amount of time they have each year to feed and gather energy stores for breeding and surviving their annual aestivation. Through investigating microhabitats and vegetation extent, it was found that the two wetlands are spatially heterogeneous and the local differences in wetland morphology, hydro-geology, soil types, and vegetation could generate different responses to climate. The site-specific data collected identifies that EBNR is a more stable wetland for WSTs than TSNR because at TSNR the water level remains low, hydroperiods are shorter, and the soil moisture is low. This research has shown that despite the hydrological management of TSNR (which includes the pumping of bore water), the wetland appears to be marginal and it is a weak surface expression of the regional water table that has limited GW connectivity to buffer it from reducing rainfall. The underlying eco-hydrology of TSNR suggests that it will remain so under future climate condition. The poor ability of TSNR to buffer the recent decline in rainfall that has occurred over the past two decades may explain why historically, WST populations at this reserve have declined while those at EBNR have remained stable.

EBNR is a clay-pan wetland, and claypan wetlands are likely to be more sensitive to a future drier climate than wetlands with high groundwater connectivity in terms of hydroperiods. This was shown in Chapters 4 and 5, where a simple pan model and a
Chapter 6. Conclusion

complex eco-hydrological model were compared and validated on the EBNR dataset. The final ecohydrological model was semi-distributed (compartmentalised based upon within wetland drainage regions) and reasonably predicted water ponding, soil moisture, groundwater levels and vegetation. This model was applied to the southwest eco-region and used to demonstrate that the region near the EBNR and TSNR was in fact an optimum location in terms of a balance between hydroperiod length and water temperature for growth. Whilst sites further in south have longer hydroperiods, there temperature is less conducive to high growth rates. This begins to change over the warming scenarios with sites in the southern most region improving in terms of both hydroperiod length and water temperature.

6.2 Limitations of the Study

In setting up the model, many generic parameters for vegetation and soil were used. This was largely due to the lack of comprehensive information on soil and vegetation specific to the study sites, which could only be identified after the field work was undertaken while developing the model. Extensive field work for soil and vegetation surveys and analysis would have been required to gain this site-specific information, which was beyond the possibilities of this study.

Soil moisture loggers were only deployed in two sites in EBNR and the moisture data was used to disaggregate the soil electrical conductivity collected by EM38 survey. It would be ideal to deploy similar sets of soil moisture sensors in TSNR and compare the field data to evaluate any differences. Moreover, moisture loggers in EBNR were placed very closely to capture the range of soil moisture of two nearby nests of WST. Although they collected valuable information related to WST hatching and aestivation, the spatial variation was not captured in the data set. Future studies should aim for more distributed moisture sensors.

One of the limitations of the compartmentalized eco-hydrological model we developed in Chapter 4 was, how the vegetation assemblage was defined in the model. In this case, although EBNR was divided into compartments, the vegetation inside each compartment were considered uniform. Further spatial heterogeneity can be adopted in terms of vegetation assemblage by field sampling and remote sensing of the spatial distribution of vegetation assemblage and incorporating it into the model. This can improve model performance by more precisely defining the vegetation functional group and thereby better predicting the hydrodynamics of EBNR.

Another limitation of the compartmentalized model was, it assumed similar infiltration rate for vegetated and bare soil, whereas ideally the later should have lower infiltration rate. Also, the model assumes a lumped vadose zone estimating all the water under unsaturated zone has potential to evaporate. This could result into overestimates of bare soil evaporation. These limitation can be overcome by short-term site-specific simulation of high resolution 1D model. The example of this type of study was discussed in Chapter 4 discussion.

Although a detailed on-site study of soil structure and geology is required for this
6.3. Recommendation and Future work

This study provided us with valuable insights on the capacity of hydrological models to capture complex processes that occur in wetlands. However, regular monitoring and measurements of critical parameters are required in addition to the ongoing monitoring initiatives.

High-resolution imagery using recent advances in AUVs (Aerial Unmanned Vehicles or Drones) fitted with hyper-spectral imaging devices has shown promise for assessing vegetation and habitat on EBNR. This kind of advanced aerial survey can provide a better quality DEM and vegetation distribution map, and similarly can be applied to any type of model setup, yet it can be helpful to achieve more precise parameters (e.g. soil hydraulic conductivity, infiltration and seepage rate) for assessing soil-water interaction between unsaturated and saturated zones (Dafny and Simunek, 2016) and thus improve model performance.

A detailed uncertainty analysis of the model (for example, by using Monte Carlo Markov Chain (MCMC) approach (e.g. Vrugt et al. 2009)) could greatly improve the performance of the model in terms of accuracy, and in terms of identifying more meaningful ranges of parameters. Incorporation of the model within an MCMC framework was trialled but not finalised or reported in the thesis. Further efforts to implement this is recommended before the model is used to supporting decision making.

For the screening assessment in Chapter 5, 34 stations randomly distributed within the study region were used. A higher-resolution input for the climate forcing data from AWAP, or other source, could improve the interpolated surfaces of hydroperiods and suitability index across the region. High-resolution input data would make the model computationally expensive, and without access to such computing facilities at present, it was not possible to work with a larger sample size.

A ‘control’ wetland could be useful for standardising some basic parameters and comparing results of the two types of wetlands in this study. The control wetland could also be used for evaluating the factors that were responsible for having two types of reproduction responses in EBNR and TSNR. However, at present, WST only have one naturally sustaining population (i.e. EBNR; TSNR is intervened to some extent) and captive-bred WST juveniles are far too precious for a control study. This is why, a control wetland could not be used in this study. In future such control wetlands should be considered depending on the availability of natural population of WST and wetlands.

For this assessment only two types of wetlands configurations were tested in respect to the degree of groundwater connectivity. This was to understand how sensitively the model responded relative to the groundwater inputs to the two extreme types of wetlands. We took the validated and calibrated model from the compartmentalized model in Chapter 4 and customised it to fit these two types of wetlands. In reality, there are many more types of wetland configurations which were not assessed mainly because of time constraints. Other types of wetlands that may be candidate sites for WST may have more substantial surface inflows and more work would be required on those sites to give insights to such ecosystems, relative to the wetland types assessed here.
future candidate sites for assisted colonization as a rapid means of gaining necessary data for model setup.

Use of distributed temperature sensing devices using fibre-optics can provide detailed information on wetland temperature which is very important for improving heat budget of such modelling practices as well as can be useful for managers of the reserve in relation to tortoise metabolism. Further assessment of spatial patterns in microclimate of candidate sites may also help understand and predict wetland water temperatures.

The compartmentalized model can be further recommended for other potential uses relative to its present functionality. With the flexibility and ability to customize the algorithm and the sub-routines of this compartmentalized model, it has potential to be coupled with other models that require realistic eco-hydrological feedback for policy making decisions. An earlier version of this model has already been used in studies for carbon biomass estimation of Lake Toolibin of Western Australia (Coletti et al., 2014) and already coupled with an eco-physiological model (NicheMapper™) for a pilot study for WST (Mitchell et al., 2013).

Furthermore, taking the advantage of the parsimonious nature of this model, it is possible to use the outputs of more complex hydrodynamic models as catchment inflow into the compartmentalized model (as a ‘coupled model’) to evaluate long-term hydrological trends (Gillan et al., 2005; Krupka et al., 2007) and predict future scenarios. An example of this can be using TUFLOW 2D which is a fully spatially resolve two-dimensional hydrodynamic model (WBM, 2008) to estimate the surface water dynamics and using the output of this model as catchment inflow in the compartmentalized model (e.g., see Coletti et al., 2014). This use is not restricted to only commercial modelling application like TUFLOW 2D but other open-source applications, for instance HEC-RAS 2D (Brunner, 2001) with capabilities to incorporate the power of 3D flexible mesh from GIS layers into the model calculation (Ackerman, 2005; Yang et al., 2006).

The outcome of the screening model from Chapter 5 needs to be further rectified before performing any AC activity on those site. The purpose of the screening study was to narrow-down the choice of the suitable habitat, which has been done in this study. The chances of these sites as successful AC spots can be maximized for present and future by applying one of the 2D model mentioned above coupled with this eco-hydrological model. This can minimize the risk of losing the captive-bred juveniles of WST at new AC sites as happened in some initial trials (Burbidge et al., 2010) without detailed study.

Other possible applications of this eco-hydrological model can be as a tool to evaluate different types of wetlands under various scenarios and assess the water requirements of wetlands (Evans et al., 2013). Example of such uses can include evaluating the effect of short-term and long-term engineering interventions in wetlands such as structures to improve impoundments and groundwater extraction. The model can also be customized for different geomorphology which implies its use for other types of groundwater dependent ecosystems and to assess the impact of land use changes.

The flexibility of defining plant groups inside the compartmentalized model can be parameterized following a specific vegetation assemblage and therefore can be applied to other ecosystems in addition to semi-arid system. The model can also be applied to
evaluate the effect of seasonal hydrological connection between geographically isolated wetlands (GIW) that already has generated a lot of research interests (Cohen et al., 2016; Evenson et al., 2015; Golden et al., 2014).

Similarly, future studies can include measuring extent of shading by functional vegetation group, which can greatly improve the energy budget inside the model. A detailed vegetation survey (e.g., rooting depth, measurement of leaf area and chest height and biomass estimation) would provide vital information for validating vegetation estimates by satellite data in future. An extended studies of wetland vegetation should consider comparing long-term satellite imagery with high temporal resolution aerial imagery. These two types of imagery can complement each other and provide valuable vegetation information. It is also possible to study the wetland groundwater and surface water through remote sensing (Barron et al., 2014).

Future efforts can be made for calibration of this eco-hydrological model with the above mentioned field data to improve the robustness of the model. The forecasting ability can be used to resolve future candidate sites to assess their viability and the approach in the overall study of applying an eco-hydrological model has the potential to be used for other Critically Endangered animals in other parts of the world.
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