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A Water Suitability Model for Riparian Vegetation in the Namoi Catchment

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SUMMARY

An integrated model of the socio-economic and environmental impacts of climate change, technology and water policy drivers has been developed for the Namoi Catchment. This technical report describes the ecological component of the integrated model.

The ecological model focuses on healthy river functions which consider:

- a sustained level of baseflow which provides refuges during drought;
- regular flushing at various levels of benches and anabranches in order to increase habitat areas and transport nutrients and organic carbon to the river system;
- regular flooding to sustain the growth of riverine vegetation and support regeneration;
- suitable groundwater and salinity levels to allow the access to water by riverine vegetation, particularly during drought.

The model estimates annual time series of hydrological and ecological indicators at nine ecological assets. Outputs include:

- hydrological indicators for channels: median daily baseflow, cease-to-flow days, and total flow;
- hydrological indicators for benches and anabranches: flood duration and frequency at three commence-to-flood levels;
- ecological indicators: maintenance and regeneration of four riverine vegetation species (river red gum, black box, lignum and water couch).

The model was tested against limited available data on river red gum dieback at the Gunnedah area, and descriptive observations on vegetation condition in the Namoi. The model outputs are consistent with the general patterns in the change of riparian trees and the distribution of vegetation species. To test the model more thoroughly, more river reach-scale monitoring data on vegetation condition across space and time is needed. We explored 23 years of NDVI data in the region. Preliminary results suggest that NDVI data is not yet suitable to test the suitability of hydrological and groundwater regimes due to the predominant climatic impacts on NDVI. Further study was suggested to fill this gap.

Two types of ecological knowledge are important for the model: the index curves which convert water attributes (e.g. flood duration) into water suitability index; and the weights which specify the relative importance of the water attributes. We undertook extensive uncertainty analysis to develop a better understanding of how the uncertainty in ecological knowledge contributes to the model outcomes. Without water management scenario inputs at the time of the study, we investigated the uncertainty by comparing the suitability of surface water and groundwater regimes for riparian vegetation at two 15-year historical periods (Pre90:1975 – 1989; Post90: 1990 – 2005).

The results suggest that with the uncertainties in index curves and weights that were framed by current ecological knowledge, we still have relatively high certainty in discriminating between the two periods at Upstream Mollee and Bugilbone to Walgett. However, much higher uncertainties in model outputs are found for other assets, suggesting that current state of knowledge is insufficient in these cases to conclude which period is better.

Note that these outcomes are case specific and thus only applies to the two water regimes (in this case, Pre90 and Post90) investigated. These results can be more useful when evaluating scenarios of different water management options. Identifying situations where knowledge is sufficient and conclusions are certain helps establish robust decision making; whereas identifying situations where current knowledge is insufficient helps direct monitoring and research efforts.

Relevant work in this report has been published in:

Fu, B. & Merritt, W. (2012). The Impact of Uncertain Ecological Knowledge on a Water Suitability Model of Riverine Vegetation. *In:* Seppelt, R., Voinov, A., Lange, S. & Bankamp, D. (eds.)

iEMSs 2012 International Congress on Environmental Modelling and Software: Managing Resources of a Limited Planet. , 1-5th July 2012, Leipzig. pp. 917-924.

Jakeman, A., Kelly, R., Ticehurst, J., Blakers, R., Croke, B., Curtis, A., Fu, B., Gardner, A., Guillaume, J., Hartley, M., Holley, C., Hutchings, P., Pannell, D., Powell, S., Ross, A., Sharp, E., Sinclair, D., Wilson, A., (2012). Modelling for the complex issue of groundwater management, in: Committee, C.O. (Ed.), *International Conference on Simulation and Modeling Methodologies, Technologies and Applications* (SIMULTECH 2012), Rome Italy, pp. IS25-IS35.

Additional uncertainty analysis work has been submitted for publication:

Fu, B. & Guillaumea, J.H.A. (Submitted). Assessing certainty and uncertainty in habitat suitability models by comparing extreme cases. *Environmental Modelling & Software*.

The work related to the above uncertainty analysis was also submitted to MODSIM 2013 conference:

Guillaumea J.H.A. and Fu, B. (Submitted). An interactive modelling tool to support knowledge elicitation using extreme case models. MODSIM 2013 conference, 1-6th December 2013.

The NDVI work has been submitted for publication:

Fu, B. & Burgher, I. (Submitted). Riparian vegetation NDVI dynamics and its relationship with climate, surface water and groundwater levels. *Journal of Arid Environments*.

1. INTRODUCTION

Riverine ecosystems rely on surface water and groundwater to maintain habitats, provide and transport food and energy, and support the growth of plants and animals. However, riverine ecosystems are increasingly under threat by drivers such as climate change and unsustainable development such as over grazing and excessive groundwater pumping. Ecological models are useful tools for assessing the ecological impacts of these drivers, and assisting the development of adaptive management plans for to the protection of riverine ecosystems and their ecological functions.

An integrated assessment of the socio-economic and environmental impacts of climate change, technology and water policy drivers is developed for the Namoi Catchment (Jakeman et al., 2012). A component of this integrated framework is an ecological model that can be used to examine the ecological impacts of climate change and management scenarios. The ecological model uses an index-based approach to estimate the suitability of the combined surface water and groundwater regimes for the riverine ecosystems. Index type approaches were initially developed and applied to hydro-ecological models in Australia as part of the Murray Flow Assessment Tool (Young et al., 2003), and later applied in other modelling tools such as the Eco Modeller (Little et al., 2011). The key to the index approach is to convert water regime to water suitability indexes, based on data, literature and/or expert opinions. The conversion is achieved through the use of index curves which define the relationships between water regime and water suitability.

This report provides a brief review on the types of hydrological and ecological indicators used for hydro-ecological models, as well as quantification of flow and ecological response relationships (Section 2). This is followed by an overview of the hydrology, water quality and ecology of the Namoi catchment (Section 3). Section 4 introduces the conceptual framework and modelling approaches for the Namoi case study. Section 5 reports methods and outcomes of model testing based on qualitative information on vegetation conditions. Uncertainty analyses on model outputs to hydrological and ecological parameters are described in Section 6. A preliminary assessment of NDVI on riparian vegetation is described in Section 7.

2. LITERATURE REVIEW

2.1 Hydrological indicators

The flood pulse concept has been widely accepted for river and floodplain ecosystems (Junk et al., 1989; Kingsford, 2000; Walker et al., 1995). Flow regime has long been used to indicate aquatic ecological health, based on the premise that natural (or reference) flow provides the ideal condition for aquatic ecosystems, and that flow alteration has an impact on the ecosystem health.

Flow magnitude remains one of the most important, and most widely used indicators for ecohydrology (Poff and Zimmerman, 2010). But it is increasingly accompanied by other types of indicators that describe duration, frequency, timing/seasonality and variation (e.g. Kennard et al., 2010b; Young et al., 2001). Some studies use hydrological indicators for different types of flow (e.g. high and low flows) due to the different ecological functions they serve (e.g. Davies et al., 2008; Richter et al., 1996; Sheldon et al., 2000). Olden and Poff (2003) identified 171 hydrological indicators for the rivers in USA. Kennard *et al.* (2010a; 2010b) analysed 120 hydrological indicators for Australia rivers, which were used for river classification in Australia. Tools such as the Indicators of Hydrologic Alteration (IHA) (Richter et al., 1996) and River Analysis Package (RAP) (Marsh, 2004) were developed to compute the hydrological indicators.

Although hydrological indicators can be comprehensive, they are not always informative, particularly when investigating long term changes and comparing scenarios. Aggregating selected hydrological indicators into a single index can be useful (e.g. Gehrke et al., 1995; Grows, 2008; Lloyd et al., 2003; Sheldon et al., 2000; Thoms, 1999). However, selecting relevant hydrological indicators is a difficult task given the large number of indicators available. A framework was developed to assist the selection of high information and non-redundant hydrological indicators (Olden and Poff, 2003). This approach should be used in conjunction with the research objectives when choosing a suitable combination of hydrological indicators (Olden and Poff, 2003).

2.2 Ecological indicators and flow-ecology relationships

Ecological indicators provide direct measurement on ecosystem health. These indicators should be sensitive to existing or proposed flow alterations, be able to be validated with monitoring data and valued by society (Kendy, 2009). Examples include aquatic invertebrate species richness, riparian vegetation recruitment and larval fish abundance. For groundwater related studies, potential groundwater dependent ecosystems such as indicators for terrestrial vegetation, river baseflow systems and wetlands should be considered as ecological indicators (Sinclair Knight Merz, 2001).

Ecological indicators are often generated through flow-ecology relationships. Two types of response functions have been identified to link water regime with ecosystem health: proportional and threshold response (Sinclair Knight Merz, 2001). These response functions can be quantified through benchmarking with similar type of ecosystems, interpretation from historical records and expert opinions. Sheldon *et al.*(2000) proposed a series of hypothetical response curves that describe the relationships between "ecological score" and hydrological change. They reported that the curves are preliminary supported by a case study in a pristine unregulated wetland system in central Australia.

The Ecological Limits Of Hydrologic Alteration (ELOHA) framework was suggested for developing holistic regional environmental flow management (Arthington et al., 2006; Poff et al., 2010). Central to the framework is the establishment of flow alteration–ecological response relationships. The framework assumes that the degree of ecological change is positively related to the degree of hydrological alteration: the greater the change in hydrological regime, the greater the change in ecological system.

However, quantitative relationships between hydrological changes and ecological response are yet to be supported by broader literature and case studies. For example, a comprehensive review on geomorphological and ecological responses to flow modification from pre 2003 literature (1976-2003) was undertaken by CRC for Freshwater Ecology (Lloyd et al., 2003). Approximately 45% of the references used are Australian studies. Of 657 studies reviewed, 70 studies matched the selection criteria and were used for qualitative analysis; 14 were used for quantitative analysis (i.e. flow-ecology relationships). No simple linear relationships were found between the extends of hydrological and ecological changes (Lloyd et al., 2003).

In a similar review by Poff and Zimmerman (2010), the ecological indicators were separated into three groups: macroinvertebrate, riparian vegetation and fish. The predictors are changes in flow magnitude and can be any of the following: peak flow, average discharge, baseflow and short-term variation. In total 165 papers were used for qualitative assessments and 55 for quantitative assessments, predominantly published during 1997 and 2010. No significant relationships were found between change of flow magnitude and change of macroinvertebrates or riparian vegetation (see example from Figure 1). However, reduced fish abundance was found associated with either increased or decreased flow. More recently, a comprehensive study was undertaken to test the concept of ELOHA framework in south-east Queensland (Arthington et al., 2012). They reported that for riparian vegetation "*no evidence was found to support the hypothesis that increasing hydrologic alteration will result in predictable patterns of increasing biotic change, as proposed in the ELOHA framework*" (Arthington et al., 2012).

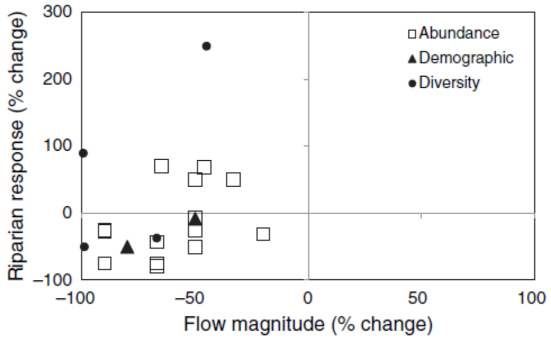


Figure 1. Percent change in riparian abundance, demographic parameters and species diversity (and/or richness) with respect to alteration of flow magnitude relative to a pre-impact or 'reference' condition (Poff and Zimmerman, 2010: Fig 4).

Although flow alteration has not been useful for generating flow-ecology relationships, empirical relationships between various flow indicators (e.g. flow variation) and ecological indicators (e.g. species richness) have been reported (Arthington et al., 2012; e.g. Overton et al., 2009). This is consistent with the foundation of stream ecology, that hydrology is the key driver for fluvial ecosystems (Allan, 1995). These empirical relationships are highly variable, and are often dependent on the location and species investigated.

The knowledge on water requirements of vegetation species in the Murray-Darling Basin is compiled in Rogers and Ralph (2010) and Roberts and Marston (2011). Both sources were based on extensive review of studies on species' water requirements in the Murray-Darling Basin. The variables considered in both sources are consistent: flood frequency, duration, timing, inter-flood dry period and flood depth. However, the thresholds proposed in these two sources differ due to differences in literature selection and interpretation of the literature by the authors. Rogers and Ralph (2010) also provides water requirements for waterbird, fish, frog, crustaceans and molluscs species.

2.3 Dependency of riverine vegetation to groundwater

Qualitative information on the dependency of riverine vegetation to groundwater is available through a limited number of studies. Eamus *et al.* (2006) suggest that compared to groundwater, flooding remains the most important factor for floodplain, wetland and baseflow river ecosystems. The dependency of riverine vegetation on groundwater is most likely to increase with the depletion of soil water, especially during drought (Eamus et al., 2006; O'Grady et al., 2002). Groundwater plays a greater role for vegetation maintenance than regeneration. Supporting this, Dawson and Ehleringer (1991) reported that both streamside and non-streamside trees use more groundwater once they are mature, compared to during establishment.

Factors influencing groundwater usage by trees include groundwater depth, the tree's maximum rooting depth and rooting distribution and soil water reserves (O'Grady et al., 2006). In general, depth to groundwater is recognised as the most critical for riverine ecosystems that rely on subsurface provision of groundwater (Eamus et al., 2006). O'Grady *et al.* (2006) reported the ranges of groundwater depths that can be accessed by most tree species are mostly from less than 4 m to 11 m. Juveniles woody species have a narrower range of depth-to-groundwater requirement than adult plants and generally grow where groundwater is shallower (Stromberg et al., 1996).

3. THE NAMOI

The Namoi Catchment is located in north western NSW, covering approximately 42,000 km^2. The lower Namoi River is categorized as an anabranch and distributary river zone where the condition of the floodplain is important to river function (Thoms et al., 1999). The lower Namoi River is also considered as a losing-connected system where stream water recharges the aquifer (NSW Office of Water, 2011b). Reduced physical habitat condition from downstream Carroll has been reported and attributed to river regulation (Thoms et al., 1999).

3.1 Hydrology and water quality

Annual flows generally increase with catchment area but in the Namoi catchment flows decrease downstream of Gunnedah due to increased evaporation, transmission losses and water use (Thoms et al., 1999). Flow variability is a feature of the Namoi River with most flows occurring in wet years and during major floods. (Thoms et al., 1999). Three typical periods are identified during the last 100 years: pre-1900 wet; 1900-1946 dry; and post 1946 wet (Riley, 1988). Sheldon *et al.* (2000) reviewed hydrological change (compared to natural condition) for six rivers in the Murray-Darling Basin, including the Namoi, and reported a decrease in median monthly flow, a decrease in large/moderate/zero flow frequency, and an increase in summer low flow. Compared to other river systems in the Murray-Darling Basin, changes in hydrological variables relative to reference condition[1] are minor in the Namoi (Davies et al., 2008).

The Namoi River has a long history of river regulation, with the first dam constructed in 1960 (Thoms et al., 1999). Table 1 summarises the capacity, location and timeline of major dams and weirs in the Namoi. The major impacts of river regulation in the Namoi include 1) altered seasonal flow; and 2) reduced flood frequency and flows, most pronounced on the small to medium (1 in 2 years to 1 in 5 years) flood events (Thoms et al., 1999). Larger flows and fluctuations in flow are of concern for the upper catchments communities, while little or no flow is an issue for people in the lower catchment. The river is considered an unreliable water source in the lower catchment.

Table 1. Dams and weirs in the Namoi Catchment

Name	Capacity (ML)	Constructed	Location	Notes
Keepit Dam	427000	1960	Namoi River upstream Peel River	
Chaffey Dam	62000	1979	Peel River upstream Tamworth	
Split Rock Dam	397000	1988	Manilla River	
Mollee Weir	3300		Downstream Keepit Dam	To improve precision of regulated flow
Gunidgera Weir	1900		Downstream Keepit Dam	To enable regulated flow to Gunidgera and Pian creeks.
Weeta Weir	280		Downstream Keepit Dam	To provide storage for downstream irrigators

Salinity is an issue of concern in the Namoi Catchment, which annually exports about 135,000 tonnes of salt to the Barwon-Darling River system (NSW Department of Water and Energy, 2008b). High salinity areas are mostly from the upper Namoi, including the Mooki River, Cox's River and Peel River (Mawhinney, 2011; Thoms et al., 1999). Turbidity increases downstream due to stock access and gully and streambank erosion (Mawhinney, 2011; Thoms et al., 1999). Nutrient levels are generally high, and mainly in particulate form and are strongly correlated with flow (Mawhinney, 2011; Thoms et al., 1999). Flooding and floodplain management is considered important to reduce the impact of adverse flooding and improve water quality (Thoms, 1999).

The Namoi Catchment has the highest groundwater use in the Murray-Darling Basin (CSIRO, 2007). In 2004/2005, groundwater extraction in the Namoi was estimated to be 255 GL, accounting for 15.2% of the total groundwater use in the Murray-Darling Basin; 35% of the groundwater extractions in the Namoi catchment was from the Lower Namoi Alluvium Groundwater

[1] Reference condition is a modelled hydrological sequence, assuming no direct human influence on water management (i.e. with storages, diversions and inter-valley transfers set to zero).

Management Unit (GMU) (CSIRO, 2007). Groundwater use in the Namoi represents three-quarters of total water use in years of minimum surface water diversion (CSIRO, 2007). The groundwater extraction in the lower Namoi is considered unsustainable, because the long term average extraction limit for the Lower Namoi Alluvium GMU exceeds total average discharge (CSIRO, 2007).

In terms of hydrological modelling for the Namoi, IQQM has been used for a few large projects including the CSIRO Sustainable Yield Project (CSIRO, 2007), the MDBA Sustainable Rivers Audit (Davies et al., 2008), the Basin Plan (MDBA, 2010), and the CSIRO Ecological Outcomes of Flow Regimes project (Overton et al., 2009). Surface water – groundwater interaction was not incorporated in IQQM for these projects, along with other variables such as farm dams (Davies et al., 2008). However, surface water – groundwater interaction is currently being incorporated into the IQQM. Salinity modelling for Namoi was undertaken using IQQM (NSW Department of Water and Energy, 2008b).

3.2 Ecology

Much of the Namoi Catchment has been cleared, except for habitat corridors and patches of riverine vegetation (Eco Logical, 2009b). The major streams and rivers are dominated by river oak (*Casuarina cunninghamiana*) and river red gum (*Eucalyptus camaldulensis*). Native floodplain vegetation communities include open grassy woodlands dominated by poplar box (*Eucalyptus populnea*), black box (*Eucalyptus largiflorens*) and coolibah (*Eucalyptus coolabah*), and native grasslands dominated by plains grass (*Austrostipa aristiglumis*). The remnant floodplain vegetation is in better condition than riparian vegetation, and lowland riparian vegetation is in better condition than upland riparian vegetation (Eco Logical, 2008).

A study on the potential causes of poplar box and river red gum dieback in the mid-upper Namoi was undertaken by the University of New England in 2001 (Reid et al., 2007). The causes of poplar box dieback are inconclusive, but the following causes are likely: reduced shallow water tables, insect damage and defoliant and herbicide drift. The authors are moderately confident that drought has caused river red gum dieback. The confidence in other causes such as reduced shallow water tables, livestock-induced soil compaction, water pollution and insecticide drift remains low (Reid et al., 2007).

The lower Namoi does not have large wetlands, but contains many small lagoons, wetlands, anabranches and flood runners (Green et al., 2011). Wetland mapping, classification and assessment in the Namoi Catchment were undertaken by Eco Logical (Eco Logical, 2008, 2009a; c). Although large in number (1829 natural and 937 artificial wetlands), most of the wetlands are small in size and scattered across the floodplain and major tributaries (Eco Logical, 2008). Foster (2004, cited in (CSIRO, 2007)) mapped the billabongs and wetlands along the Namoi River from Keepit Dam to Walgett and assessed their connection with flows in the Namoi River. It was estimated that flows in the range 4 – 5 GL/day at the Duncans Junction (gauge station 419094) flood nearly 1000 km of billabongs and wetlands along the Namoi River. Using 4GL/day flow threshold, it was estimated that under “best estimate 2030 climate” scenario, the hydrology of the Namoi River billabongs and wetlands does not change greatly from the current conditions (CSIRO, 2007).

Dissolved organic carbon (DOC) is identified as the limiting factor for riverine bacterioplankton production, and hence energy source for higher trophic levels at the lower Namoi (Westhorpe et al., 2010). DOC levels in the rivers are positively related to flooding. It is reported that the mean DOC concentrations increased by 35% at Bugilbone during a small flood event (peaked at 2500ML/day in January 2004) (NSW Department of Water and Energy, 2008c), and doubled during a bigger event (peaked at 5500 ML/day in December 2008) (NSW Office of Water, 2011a). At Walgett during a very large flood event (peaked at 46000ML/day in Dec 2004), the mean DOC concentrations increased from 10.3± 0.9 mg/L during routine monitoring to 20.4 ± 0.9 mg/L during the flood (NSW Department of Water and Energy, 2008c).

Macroinvertebrates play a vital role in stream ecology, contributing to the decomposition cycle and food chain. Leonard *et al.* (2000) reported that the endosulfan concentrations and river discharge are the main contributors to the macroinvertebrates population in the riffle-pool habitats in the Namoi River.

In terms of fish, the Namoi has a greater number of bony herring and less Australian smelt compared to other systems such as Macquarie, Lachlan and Murrumbidgee rivers (Growns, 2008). Common carp contributes to the decline of native fish abundances and are a primary concern for the local community (Thoms, 1999). Listed endangered fish species in the Namoi River include river snail, silver perch, purple spotted gudgeon and the olive perchlet (Green et al., 2011).

4. MODEL DESCRIPTION

4.1 Scope and ecological assets

The ecological model for the Namoi focuses on healthy river functions. This involves:

- a sustained level of baseflow, which provides refuges during drought;
- regular flushing at various levels of benches and anabranches, in order to increase habitat areas and transport nutrients and organic carbon to the river system;
- regular flooding to sustain the growth of riverine vegetation and support regeneration;
- suitable groundwater and salinity levels to allow the access of water by riverine vegetation, particularly during drought.

Nine ecological assets are modelled (Figure 2). Eight of these assets were selected based on work reported in Barma Water Resources *et al.* (2012). In addition, a river red gum corridor at Maules Creek was included; this asset has sustained little impact from groundwater extraction. All assets are important river red gum corridors in the region. Some assets such as Barbers Lagoon (Asset 2) and Duncans Warrambool (Asset 5) contain wetlands which are important waterbird and fish habitats. Areas and ecological values of the assets are listed in Table 2.

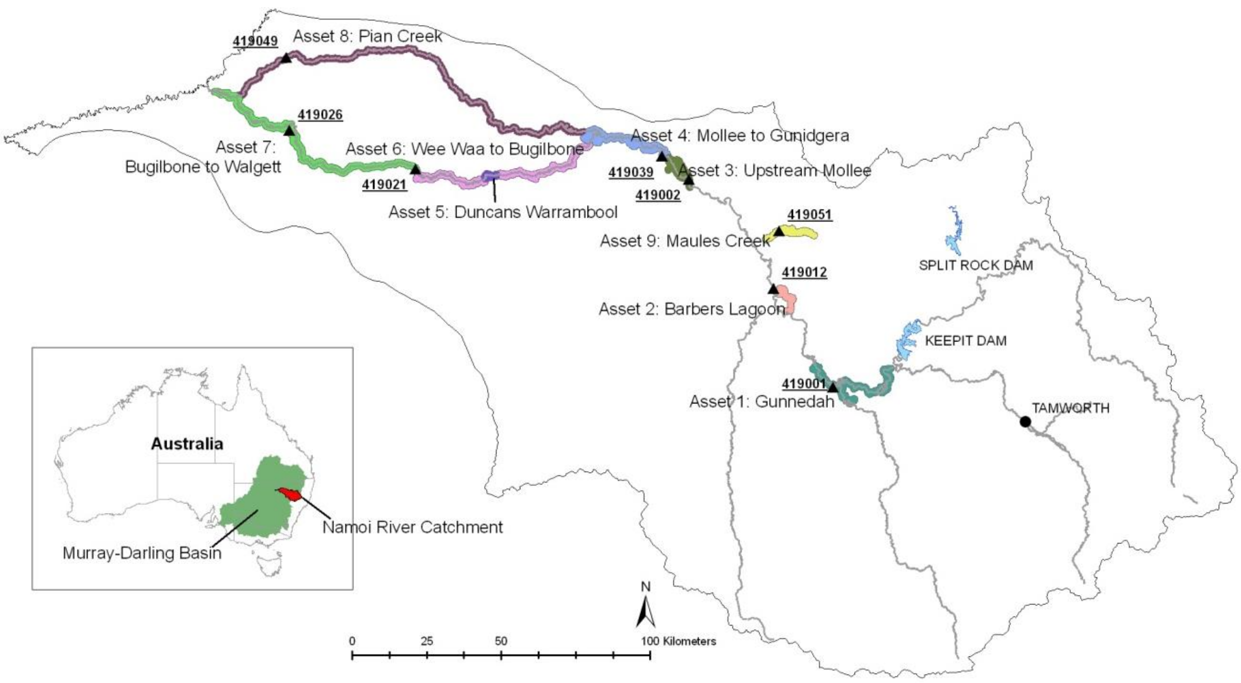


Figure 2. Locations of the nine modelled ecological assets, and related river gauges.

Table 2. Areas and ecological values of the modelled ecological assets in the Namoi River catchment

Asset ID	Name	Area (ha)	Ecological values
1	Gunnedah	2012	River red gum riverine, testing site
2	Barbers Lagoon	134	River red gum, wetlands (water couch)
3	Upstream Mollee	933	River red gum, wetlands
4	Mollee to Gunidgera	3195	River red gum, black box, coolabah, wetlands
5	Duncans Warrambool	301	River red gum, black box, coolabah, wetlands (lignum, billabong rush, nardoo, poison pratia)
6	Wee Waa to Bugilbone	3144	River red gum, coolabah, black box, wetlands (water couch, tall flat-sedge)
7	Bugilbone to Walgett	3570	Black box, coolabah, River red gum, wetlands (lignum, tussock rush, dirty dora, spiny sedge)
8	Pian creek	222	River red gum
9	Maules creek	425	River red gum

4.2 Conceptual framework

The conceptual framework for the ecological model is illustrated in Figure 3. Inputs of the model are daily time series of surface flow, groundwater level and groundwater salinity. These inputs can be derived from observations or models. Outputs of the ecological model are aggregated annual time series of hydrological and ecological indicators. Daily surface flow and baseflow data provides annual hydrological indicators for channels. The hydrological indicators for benches and anabranches are estimated through the identification of flooding or wetting events using commence-to-flood (CTF) levels. The ecological indicators include the maintenance and regeneration of four riverine vegetation species. Details of the methods are described below.

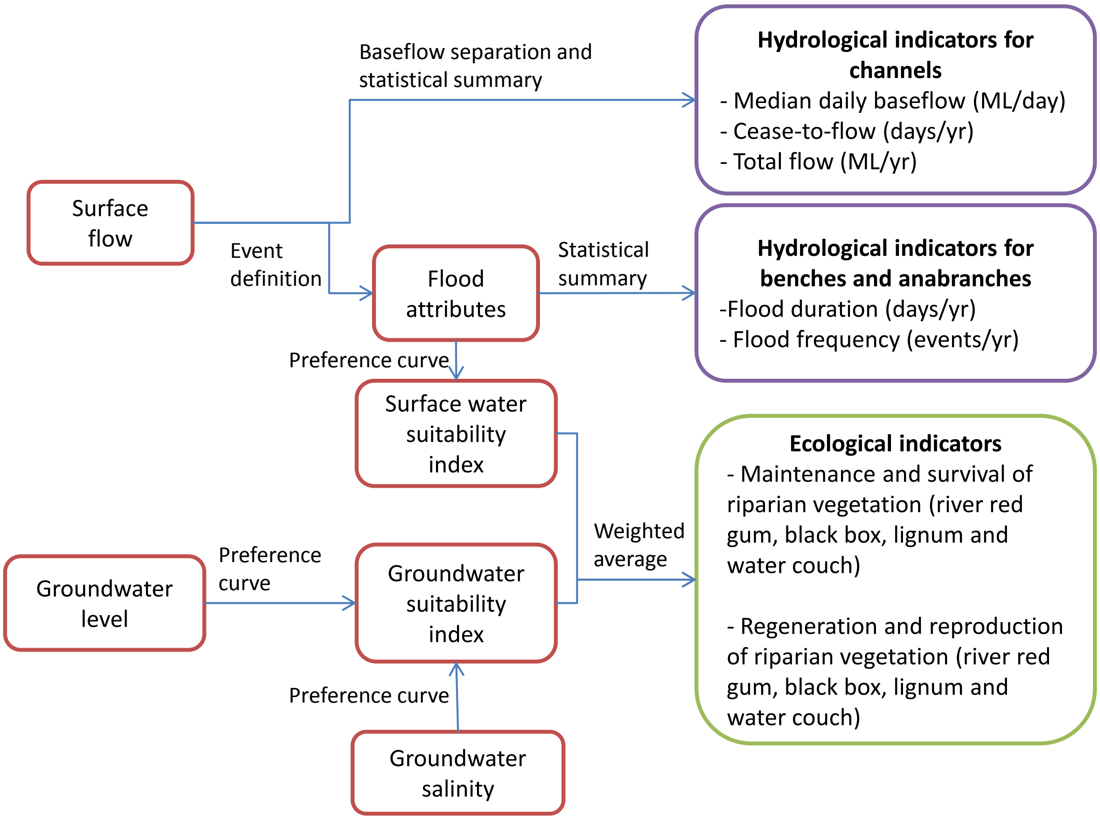


Figure 3. Conceptual framework for the ecological model.

4.3 Hydrological indicators for channels

Baseflow levels and cease-to-flow frequency were selected as the hydrological indicators because they can be related to groundwater changes. Annual total flow was also used as a hydrological indicator to reflect the overall surface water availability. Surface flow data was extracted for each ecological asset depending on their corresponding river gauge (Table 3 and Figure 2). Gauges for Assets 1-8 were identified from Barma Water Resources *et al.* (2012), whereas the gauge for Asset 9 was selected based on its proximity to the asset.

Table 3. Ecological assets in the model and the corresponding gauge and commence-to-flood (CTF) levels for benches and anabranches. Bold numbers are flood thresholds for ecological indicators.

Asset ID	Name	Gauge	CTF_low (ML/day)	CTF_med (ML/day)	CTF_high (ML/day)
1	Gunnedah	419001	**5000**	8000	15000
2	Barbers Lagoon	419012	**4600**	4600	5000
3	Upstream Mollee	419002	**1600**	2000	8200
4	Mollee to Gunidgera	419039	**4000**	7500	11000
5	Duncans Warrambool	419021	**4000**	4500	5500
6	Wee Waa to Bugilbone	419021	**1800**	4000	4500
7	Bugilbone to Walgett	419026	**1900**	6300	14000
8	Pian creek	419049	**2000**	2000	3000
9	Maules creek	419051	**110**	500	1000

Baseflow levels at each gauge were estimated through baseflow separation using a minimum filter, with a half width of 5 time steps (i.e. the minimum flow is calculated over 11 time steps: 5 before the current, the current and 5 after the current time step). Annual median baseflow level is then summarised from the daily baseflow time series. The number of cease-to-flow (Q=0 ML/day) days per year and the total flow volume (ML/year) were estimated from surface flow at the corresponding gauge for each ecological asset.

4.4 Hydrological indicators for benches and anabranches

Annual flood duration and the frequency of inundation of benches and anabranches were estimated by defining flood events at three different levels of benches/anabranches. Flood duration was defined by the number of days in a year that daily flow exceeds the corresponding CTF level (Figure 4). The CTF levels for Assets 1-8 (Table 3) were obtained from Barma Water Resources *et al.* (2012) and Powell (Sue Powell, pers. comm. July 2011), whereas the CTF levels for Asset 9 were derived from the cross section of Gauge 419051 and its rating curve (NSW Water Information, 2011). Although bench levels realistically vary at local scale along the channel, we assumed the change in bench level is not significant within each asset. This assumption is plausible as the river reaches within the same asset belong to the same River Styles® geomorphological classification (Lampert and Short, 2004). Conceptually, we considered the CTF value for each level of bench/anabranch useful in capturing the various extent of flooding within an asset.

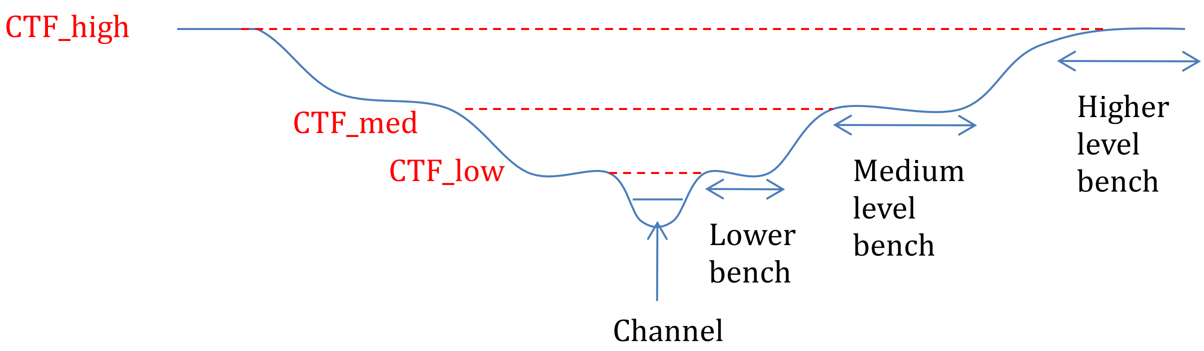


Figure 4: Schematic diagram showing a river cross section and the relationships between commence-to-flood (CTF) levels and flooding of different levels of benches.

The flood frequency is the number of flood events that occur in a year. By default, it was assumed that the minimum number of days in each event window is 1 day. The main ecological functions captured here are the flushing of nutrients and organic carbon to the river channel. Therefore it was assumed that even if the flow exceeds the CTF level for only one day, the flushing process occurs. We also assumed that when two events occur within no more than 5 days between each other, they are considered as one event. The number of days to separate events is arbitrary, but it is necessary to account for the fluctuation of river flow.

4.5 Ecological indicators

The ecological indicators were estimated using index curves. Water suitability index (WSI) was estimated for four riverine vegetation species: river red gum (*Eucalyptus camaldulensis*), black box (*Eucalyptus largiflorens*), lignum (*Muehlenbeckia florulenta*) and water couch (*Paspalum distichum*). These species were selected because they were found at the modelled areas, and knowledge of water requirements of these species is available (Roberts and Marston, 2011; Rogers and Ralph, 2010). The water suitability index is an aggregate of the surface water suitability index and the groundwater suitability index:

$I = w_g G + w_s S$ (Equation 1)

where I, G, S respectively denote the water suitability index, groundwater suitability index and surface water suitability index; and w_g and w_s are weights for groundwater and surface water indices respectively.

The surface water suitability index was estimated based on weighted average of suitability of flood duration, flood timing and inter-flood dry period (Equation 2).

$S = w_d D + w_t T + w_f F$ (Equation 2)

where S, D, T, F are respectively the water suitability index, flood duration index, flood timing index and inter-flood dry period index; and w_d, w_t and w_f are weights for duration, timing and inter-flood dry period respectively.

The surface water suitability index is modelled using the following steps:

1. Critical flood attributes were generated from surface flow time series based on the definition of a flood event. The attributes are flood duration, flood timing and inter-flood dry period. Flood duration is the number of days that daily flow exceeds a specified flood threshold (bold CTF levels in Table 3). Flood timing is the month that a flood event commences (Jane Roberts, pers. comm., January 2012). Inter-flood dry period is the number of days between the end of the previous flood event and the beginning of the current event. By default, it was assumed that the minimum number of days in each flood event window is 3, and the minimum number of days that can separate events is 2. The parameters used to define a flood event are arbitrary; the sensitivity of the model outputs to these parameters is described in Section 6.1.

2. For each species, the suitability index of each flood attribute was estimated using index curves. The index curves were generated based on the requirements of each flood attribute reported in Roberts and Marston (2011). Additional information such as groundwater salinity thresholds was sourced from Rogers and Ralph (2010). More details in the generation of index curves are described in Section 4.6.
3. For each species, the surface water suitability index was estimated using the weighted average of the suitability index of each flood attribute. Weights were assigned based on Roberts and Marston (2011). More details in the generation of weights are described in Section 4.7.

The groundwater suitability index was based on the suitability of the groundwater level (groundwater level index) adjusted by groundwater salinity. Groundwater level is the depth (in meters) to the water table in the shallow aquifer. Groundwater salinity is the salinity level (in ppm) of the shallow aquifer. The groundwater level index was generated from index curves generated from Roberts and Marston (2011). If the groundwater salinity is higher than the salt tolerance threshold for a given species, the groundwater suitability index is reduced to 0; otherwise, the groundwater suitability index is equal to the groundwater level index. Note that without a groundwater salinity model and with recorded salinity levels at the ecological assets consistently below the salt tolerance thresholds of all modelled species, we assumed the groundwater salinity level under any scenario does not exceed the minimum salt tolerance threshold (10000 ppm). Therefore, effectively the current model does not consider the impact of salinity. However, it is conceptually more sensible to include salinity as one of the key drivers for estimating the groundwater suitability index.

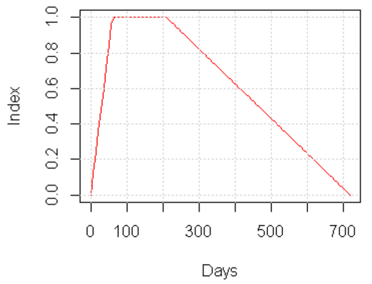
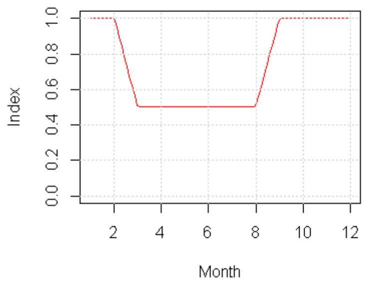
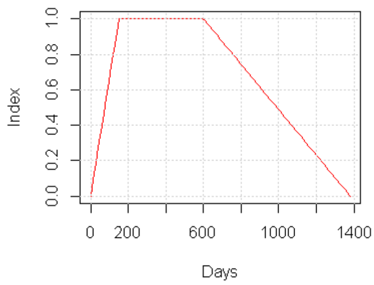
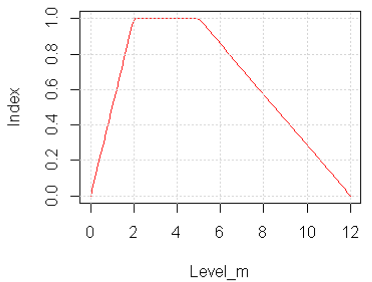
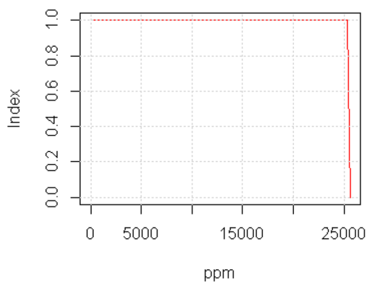
Finally the water suitability index was estimated using the weighted average of the surface water and groundwater suitability indexes. This is consistent with the observation reported by Bacon et al. (1993) that the effect of water supply for river red gum is additive. We assumed that the weights for groundwater and surface water indexes are 0.2 and 0.8, respectively. The annual water suitability index is the sum of the daily water suitability index in a calendar year.

4.6 Index curves

Index curves are the key parameters for converting water regime to suitability index. Two sources are available on the water requirements of vegetation species in the Murray-Darling Basin, Australia: Rogers and Ralph (2010) and Roberts and Marston (2011). The default index curves in the ecological model were generated from Roberts and Marston (2011), because it relies more on field observations (in contrast to laboratory experiments), and was derived from studies undertaken in areas more similar to the Namoi conditions. The way in which ecological knowledge uncertainty affects the outcomes of the Namoi ecological model is explored in Section 6.

Index curves were generated relating selected hydrological indicators to i) maintenance and survival and ii) regeneration and reproduction of each modelled species. The hydrological indicators related to riparian ecosystem in the Namoi are flood duration, flood timing, inter-flood dry period, groundwater level and groundwater salinity. Examples of the index curves for the maintenance and survival of river red gum is given in Table 4. The index curves for all species are listed in Appendix 1.

Table 4. Water requirements and index curves for the maintenance and survival of river red gum.

Attributes	Information from Roberts and Marston (2011)	Index curves
Flood duration	5-7 months for forests, 2-4 months for woodlands. Continuous Inundation of 2-4 years has been tolerated at a diverse site (Barmah Forest).	
Flood timing	Start of flooding is not critical, but more growth is achieved if flooded during spring-summer.	
Inter-flood dry period	NA (Defined based on information on flood frequency and flood duration[2].)	
Groundwater level	River red gum has dual root system with lateral roots close to surface and taproot penetrates to the ground to around 9m or above. But taproot can be damaged by waterlogging.	
Groundwater salinity	NA (Defined based on Rogers and Ralph (2010): groundwater should be less than 40dS/m (~25600ppm)).	

[2] The Namoi River catchment has river red gum forest and woodland. Flood frequency requirements are 1-3 yrs and 2-4 years for forest and woodland, respectively. Flood duration requirements are 5-7 months and 2-4 months for forest and woodland, respectively. Thus, the lower limit of the ideal inter-flood dry period is when it floods is 7 months per year (forest), i.e. 5 months (150days) of dry period. The upper limit of the ideal dry period is when it floods is 4 months every 2 years (woodland), i.e. 20 months (600days) dry period. The upper threshold of the dry period is when it floods 2 months every 4 years, i.e. 48 months (1380 days) dry period.

4.7 Weighting

A weighting approach was used to combine the flood attributes into a single measure of surface water suitability for the vegetation. Weights were assigned to each flood attribute based on information provided in Roberts and Marston (2011). For example, for the maintenance of river red gum, Roberts and Marston (2011) suggest that flood timing is not critical for the growth of river red gum. As a result, more weight was given to flood duration, and the weights for duration, timing and inter-flood dry period are 0.5, 0.2 and 0.3, respectively. The weights for all modelled species are listed in Table 5.

Table 5. Weights assigned for each flood attribute.

Modelled function	Modelled species	Flood timing	Flood duration	Inter-flood dry period
Maintenance and survival	River red gum	0.2	0.5	0.3
	Black box	0	0.9	0.1
	Lignum	0.1	0.6	0.3
	Water couch	0.5	0.25	0.25
Regeneration and reproduction	River red gum	0.3	0.7	0
	Black box	0.5	0.5	0
	Lignum	0.8	0.2	0
	Water couch	0.5	0.5	0

5. MODEL TESTING

5.1 Methods

Quantitative data is not available for testing the model. Therefore, we evaluated the model based on information on historical vegetation conditions. The information includes:

1. Study on river red gum dieback (Reid et al., 2007). Reid *et al.* (2007) reported that the crown health of many floodplain and riparian eucalyptus, including the river red gum, has declined significantly since early 1990s in the Gunnedah area, although fluctuations occur due to periodic recovery after rain. Analysis on aerial photographs suggested that crown shadow density of trees at site R11 (north-west of Gunnedah on the outskirts of town) are lowest in 1986, compared to 1991 and 1975 (Figure 5).

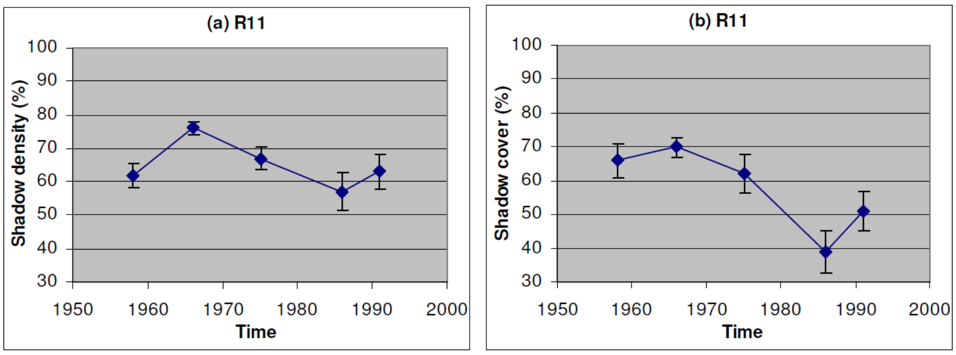


Figure 5. (a) Crown shadow density and (b) crown shadow cover of trees at site R11 (north-west of Gunnedah on the outskirts of town) in 1958, 1966, 1975, 1986 and 1991, as estimated from aerial photography. Data are the mean ± s.e.m. of the same ten trees in each photo run (Reid et al., 2007: Fig. 3.18).

2. Riparian vegetation survey and mapping (Eco Logical, 2009b). The Namoi riverine is mostly dominated by river red gum forest and woodland, and black box woodland on floodplain. There are patches of wetland species such as lignum and water couch. These species are found at Barbers Lagoon (Asset 2), Duncans Warrambool (Asset 5), and the riparian zones between

Wee Waa and Walgett (Assets 6 and 7) (unpublished Integrated Monitoring of Environmental Flow (IMEF) data at Namoi, Sue Powell, pers. comm. 2011).

From the information from Reid *et al.* (2007) and Eco Logical (2009b), we generated the following hypotheses for model sanity testing:

1. Water suitability indexes for the maintenance and survival of river red gum and black box at Gunnedah (Asset 1) are significantly lower since early 1990s (based on Reid *et al.* (2007)).
2. Water suitability indexes for the maintenance and regeneration of river red gum and black box at Gunnedah (Asset 1) fluctuate over time (based on Reid *et al.* (2007)).
3. Water suitability indexes for the maintenance of river red gum and black box at Gunnedah (Asset 1): 1975 > 1991 > 1986. Note that water suitability for pre 1970 cannot be tested due to lack of historical data (based on Reid *et al.* (2007)).
4. Water suitability indexes for river red gum and black box are significantly higher than lignum and water couch (based on Eco Logical (2009b)).

Historical river flow and groundwater levels were used as input data. Pre 2008 historical river flow data was obtained from PINNEENA 9.2 (Department of Water and Energy, 2008), with more recent flow records (2008-2010) obtained from the waterinfo website (waterinfo.nsw.gov.au).

Historical groundwater bore data was obtained from Groundwater PINNEENA 3.2 (NSW Office of Water, 2010). The groundwater bore data was corrected and modified, and then interpolated into daily time series using a linear regression (Blakers, 2011a). For each ecological asset, bores were selected based on their proximity to the asset and the number of records (Table 6), and the value of the daily groundwater levels at Pipe 1 (the pipe with the most shallow opening) of selected bores was used to represent the groundwater level for that asset.

Table 6. Groundwater bores used for each ecological asset.

Asset ID	Name	Groundwater bores	Distance to river (m)	Period
1	Gunnedah	GW030299	90	23/6/1973 – 4/4/2011
2	Barbers Lagoon	GW030050	44	11/9/1970 – 30/3/2011
3	Upstream Mollee	GW030122	454	5/3/1971 – 16/5/2011
4	Mollee to Gunidgera	GW025107	31	6/5/1968 – 18/5/2011
5	Duncans Warrambool	GW036157	110	28/4/1978 – 4/4/2011
6	Wee Waa to Bugilbone	GW021479	989	2/12/1969 – 20/3/2011
7	Bugilbone to Walgett	GW036540	1577	3/5/1985 – 4/4/2011
8	Pian creek	GW025137	610	10/4/1968 – 5/4/2011
9	Maules creek	GW030129	276	6/7/1971 – 12/1/2011

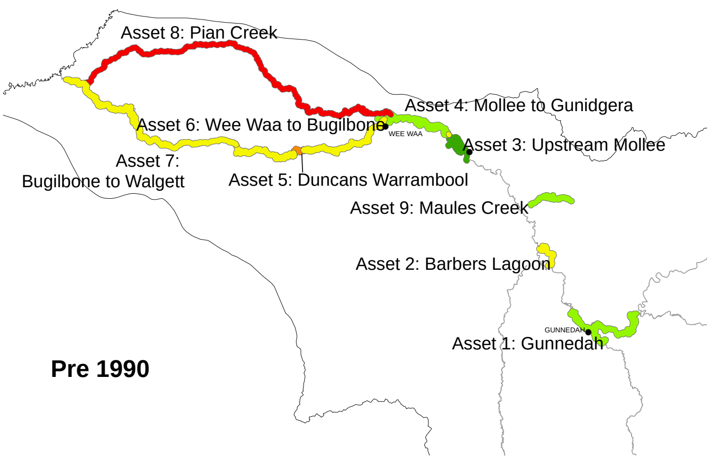
The available data periods vary depending on the location, with most of the sites having data from 1970s to 2010 (Table 7), mainly restricted by available groundwater data.

Table 7. Data periods available for each ecological assets, and their mean groundwater level, annual total flow and cease to flow days.

Asset ID	Name	Available data	Mean groundwater level in m below ground (SD in bracket)	Mean annual total flow (ML/year)	Mean annual cease to flow days (days/year)
1	Gunnedah	1974-2010	9.0 (1.9)	609,885	11
2	Barbers Lagoon	1971-2010	9.8 (1.5)	774,837	4
3	Upstream Mollee	1972-1994	6.1 (0.6)	63,620	172
4	Mollee to Gunidgera	1968-2009	9.3 (1.5)	747,032	0
5	Duncans Warrambool	1979-2010	12.7 (1.7)	443,578	21
6	Wee Waa to Bugilbone	1970-2010	16.1 (1.8)	548,694	16
7	Bugilbone to Walgett	1986-2010	16.4 (0.1)	470,442	43
8	Pian creek	1973-2010	22.0 (1.8)	89,754	116
9	Maules creek	1973-2010	7.7 (0.5)	19,480	24

5.2 Results

Spatially, the average annual water suitability indexes for the maintenance of river red gum are consistently low downstream of Wee Waa since 1970 (Figure 6), because of a lack of both surface water and groundwater. The riparian zones at Asset 3 (between Narrabri and Mollee) and Asset 9 (Maules Creek) consistently obtain reasonable water resources for the vegetation, even during the drought in 2000s. The vegetation at Asset 9 (Maules Creek) is mainly sustained by shallow groundwater (consistently less than 9 m below ground since 1970). The water suitability at Asset 1 (Gunnedah) and Asset 4 (between Mollee and Gunidgera) reduced gradually from high levels of suitability during the 70s and 80s, to moderate levels in the 90s, to low levels in the 2000s. This is the result of both reduced flooding and groundwater levels near these assets. Water suitability at Asset 2 (Barbers Lagoon) declined significantly since 1990, mainly due to reduced flooding.



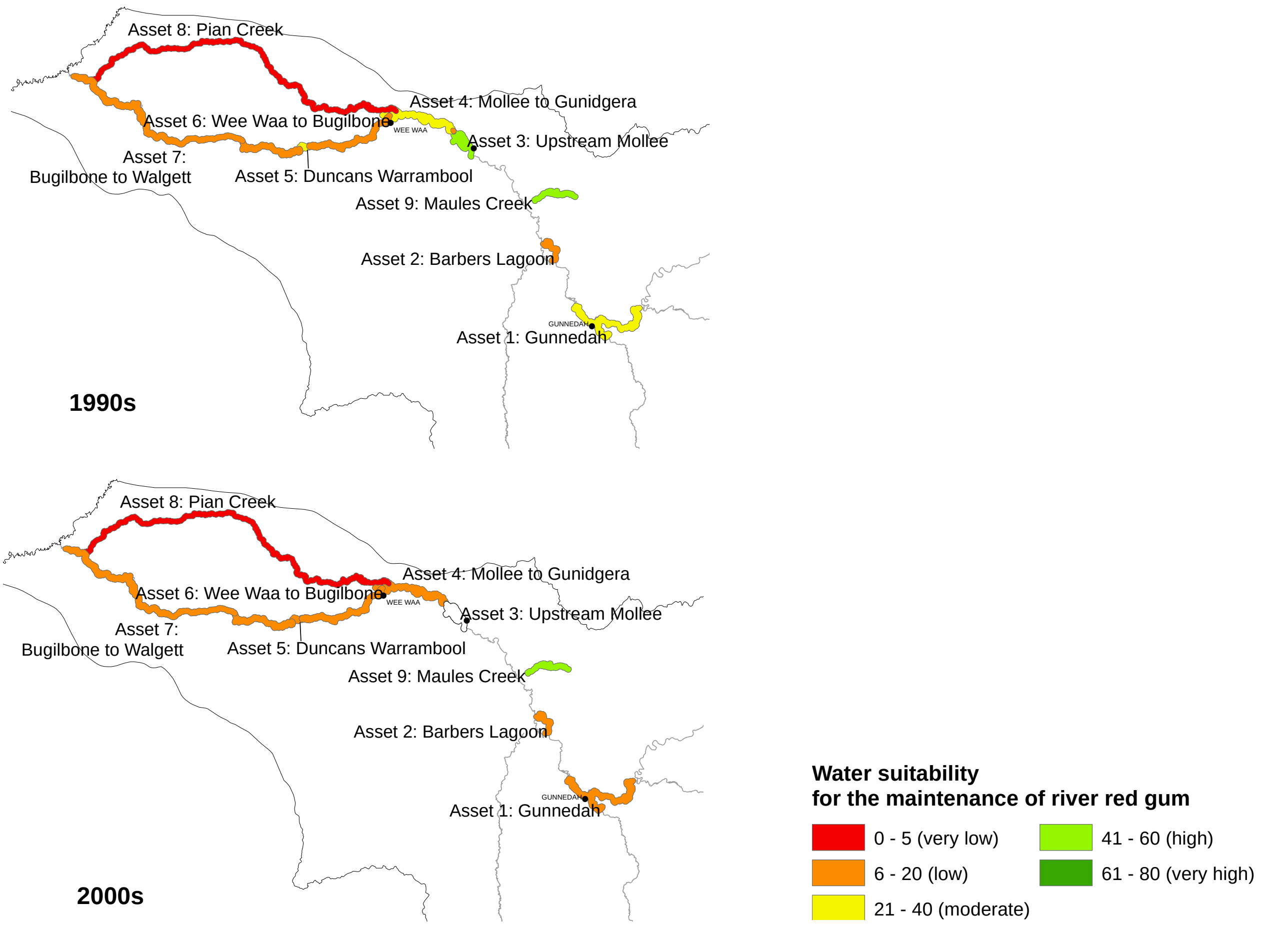
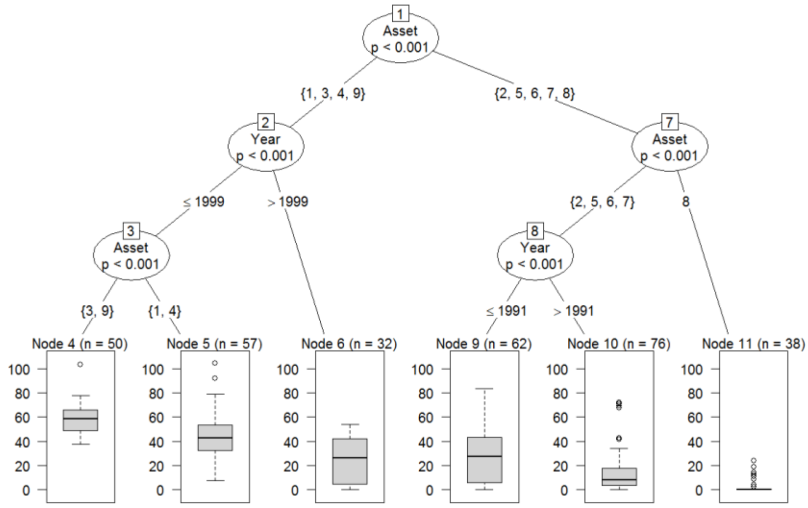


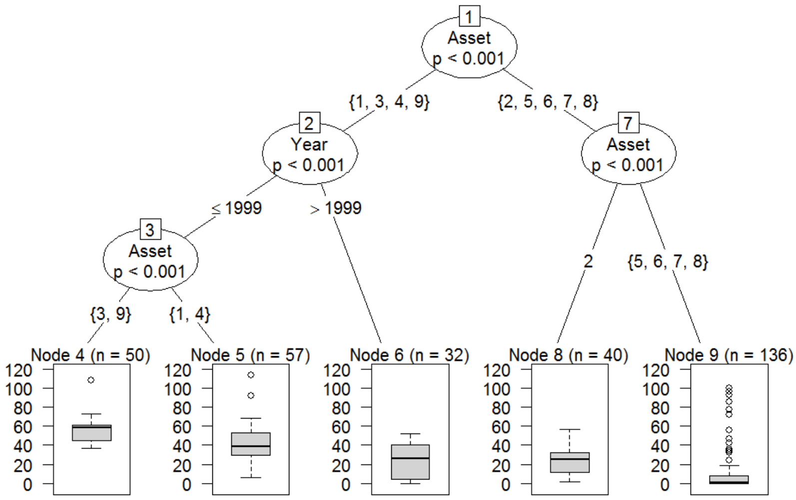
Figure 6. Water suitability for the maintenance of river red gum during 1970-1989, 1990-1999, and 2000-2010. Note there is no data for Asset 3 during 2000-2010.

Decision tree analysis was used to investigate the patterns of annual water suitability for the maintenance of river red gum and black box across space and time. The space element was represented by model outputs for different assets, and the time element was represented by year. The *ctree()* function in **party** package in R was used.

It is found that water suitability for river red gum is consistently the lowest at Assets 2, 5, 6, 7 and 8, i.e. Barbers Lagoon and the areas downstream Wee Waa, in the last three decades (Figure 7). Pian Creek has the lowest water suitability (mean WSI = 2.87), while the other assets have lower water suitability after 1991 (i.e. since 1992) (mean WSI = 13.79) compared to the period before 1992 (mean WSI = 28.18). For the maintenance of river red gum at Assets 1, 3, 4 and 9, the water suitability indices are higher in 1999 and before (mean WSI = 51.52) compared to those after 1999 (mean WSI = 24.73). Before 2000, Assets 3 and 9 have higher water suitability than assets 1 and 4, with mean WSI 58.46 and 45.43 respectively. A similar pattern is found for the maintenance of black box. The test statistics (i.e. p-values) for all splits are below 0.001, suggesting the distributions of each split groups are significantly different. These results are broadly consistent with the first hypothesis, that the water suitability indexes for the maintenance and survival of river red gum and black box at Asset 1 (Gunnedah) are significantly lower after early 1990s.



(a)



(b)

Figure 7. Decision tree analysis showing the patterns of annual water suitability index (WSI) for the maintenance of (a) river red gum and (b) black box across space (i.e. Assets) and time (i.e. Year).

Temporal dynamics of the water suitability index were explored by examining the time series of the model outputs. Assets 1 and 5 are used as examples (Figure 8). At Asset 1 (Gunnedah), the water suitability indexes for the maintenance of river red gum and blackbox gradually decline since 1990. In contrast, at Asset 5 (Duncans Warrambool), there is a slight increase in water suitability for the maintenance of river red gum and black box around 2000. This is due to both larger surface flows and shallower groundwater levels. Fluctuations are observed for the maintenance and regeneration of both species at both locations, and mostly correspond to the change in annual total surface flow. For example, water suitability indexes for the regeneration of river red gum are high in 1977, 1984, 1990 and 1998, which broadly corresponds to higher flow in these years. This analysis confirms the second hypothesis that water suitability indexes for the maintenance and regeneration of river red gum and black box at Asset 1 (Gunnedah) fluctuate over time.

Fluctuations are greater when groundwater plays a limited role (i.e. when groundwater is too deep to sustain the vegetation during dry time). Groundwater levels at Asset 5 are generally deeper than the threshold to maintain vegetation (12 m). Therefore, the fluctuation of water suitability index for vegetation maintenance is greater at Asset 5 than Asset 1. Similarly, groundwater levels at Asset 1 are generally within the threshold for vegetation maintenance but outside the threshold for vegetation regeneration. As a result, the fluctuation of water suitability index for vegetation regeneration is greater than vegetation maintenance at Asset 1.

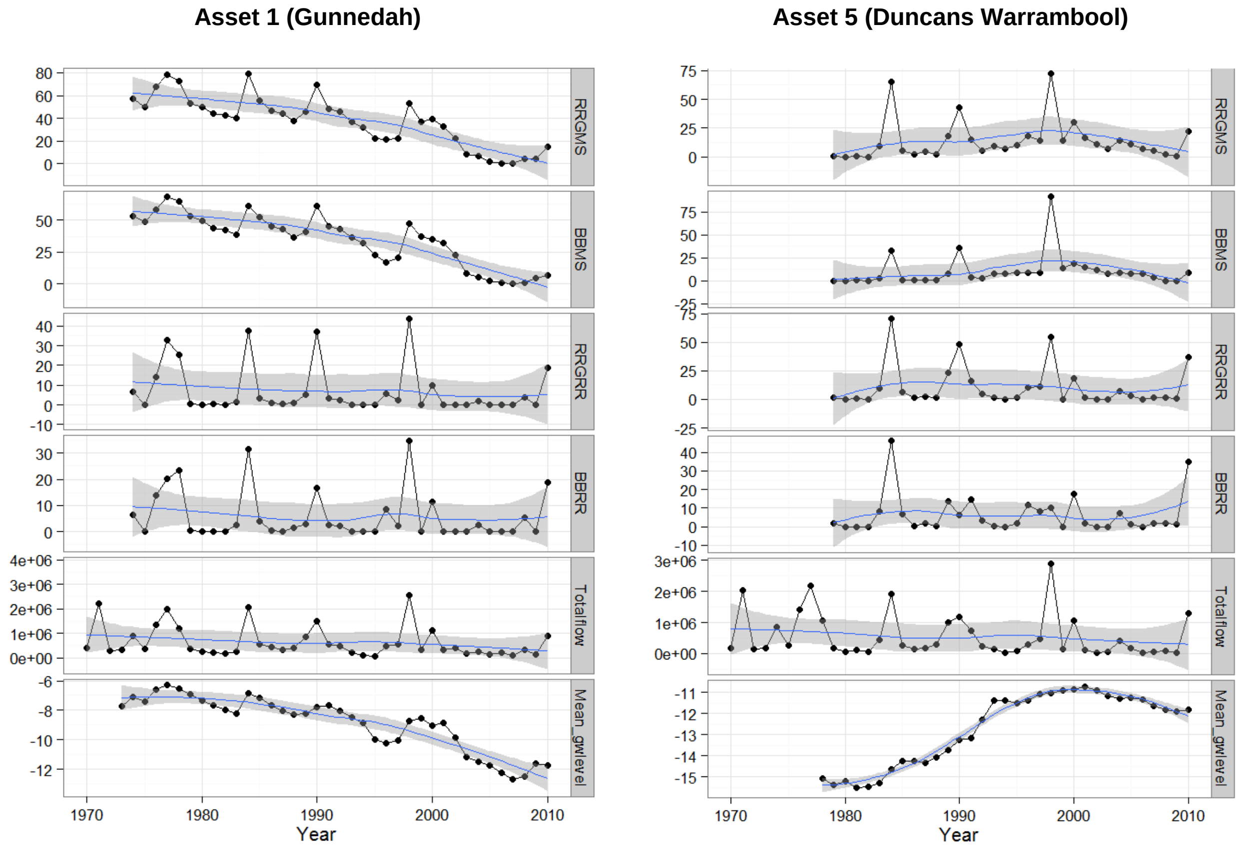


Figure 8. Time series of water suitability for the maintenance of river red gum (RRGMS), water suitability for the maintenance of black box (BBMS), water suitability for the regeneration of river red gum (RRGRR), water suitability for the regeneration of black box (BBRR), annual total flow (Totalflow, in ML/year), and mean annual groundwater level (Mean_gwlevel, in meter), for Asset 1 (Gunnedah, left) and Asset 5 (Duncans Warrambool, right). Each time series is accompanied by a smooth line (blue line) and its 95% confidence interval (grey shade), to identify the trend.

Note that the input data for the model is daily time series. Therefore, the same annual total flow and mean groundwater level do not necessary result in the same water suitability for a species. For example, at Asset 1 (Gunnedah) similar annual total flow (about 2000 GL) and mean groundwater level (about 6 m below ground) were reported in 1977 and 1984, but the water suitability for the regeneration of river red gum or black box in 1984 is higher than that in 1977. This is because the timing of the flooding was very different in both years (Figure 9). In 1977 the Gunnedah area was dominated by autumn and winter floods, which contribute little to the regeneration of riparian vegetation. In contrast, in 1984 floods occurred mostly in late winter and summer, which are beneficial to vegetation regeneration.

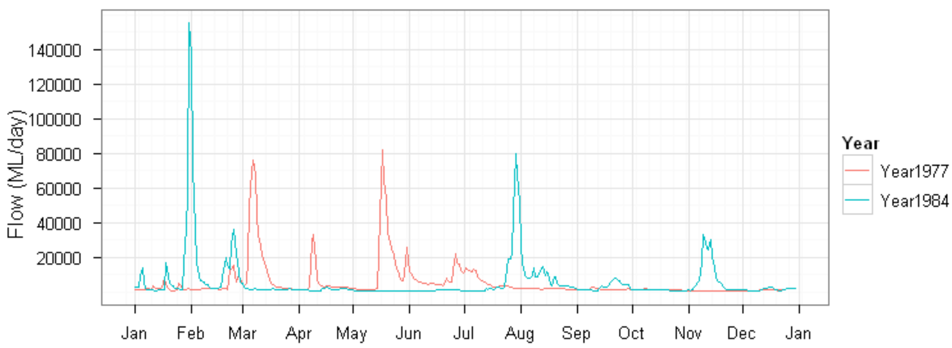


Figure 9. Time series of daily flow in 1977 and 1984 at Asset 1 (Gunnedah).

Water suitability for the maintenance of river red gum and black box at Asset 1 (Gunnedah) in 1975, 1986 and 1991 were extracted to test the third hypothesis. As shown in Figure 10, for both species, the water suitability indexes are highest in 1975, followed by 1990 and then 1986. This is consistent with the relative crown shadow density and crown shadow cover of trees in 1975, 1986 and 1991, as reported in Reid *et al.* (2007) and shown in Figure 5.

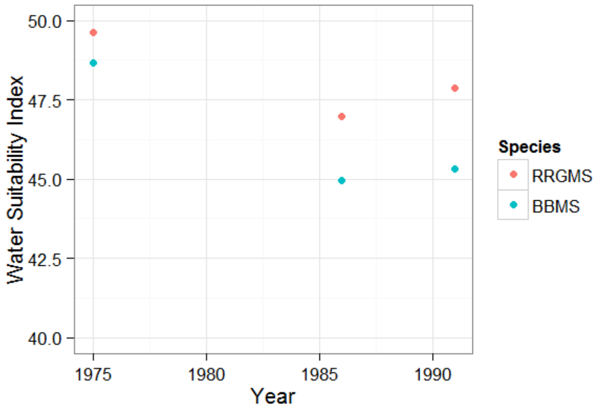


Figure 10. Water suitability for the maintenance and survival of river red gum (RRGMS) and black box (BBMS) at Asset 1 (Gunnedah) in 1975, 1986 and 1991.

Annual water suitability indexes at all assets were summarised for each species (Figure 11). The Kruskal-Wallis test confirms that the distributions of water suitability index for river red gum and black box are significantly different ($p < 0.001$) with those for lignum and water couch. This confirms the fourth hypothesis, that water suitability indexes for river red gum and black box are significantly higher than lignum and water couch.

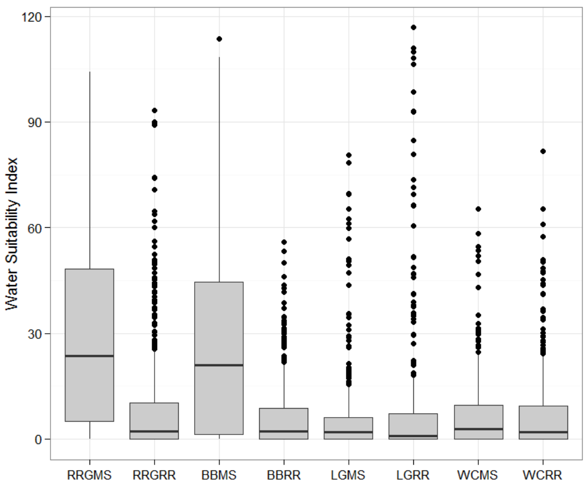


Figure 11. Water suitability index for the maintenance and survival (MS) and regeneration and reproduction (RR) of river red gum (RRG), black box (BB), lignum (LG) and water couch (WC).

In summary, all four hypotheses were confirmed. However, due to the extremely small number of testing data points and the qualitative nature of the hypotheses, the testing outcome should be used with caution. For example, only three data points are available for the testing of the third hypothesis; randomly there is 17% chance that the model outputs match the order of the observed data. Therefore, each test alone is insufficient to evaluate the model. However, it can be useful when used in conjunction with other evidence (i.e. the confirmation of other hypotheses). In this case, the model outputs were consistent with the general patterns in the change of riparian trees and the distribution of vegetation species in the study area. To test the model more thoroughly, more monitoring data on vegetation condition across space and time is needed. This work has started and a preliminary assessment of NDVI for the riparian vegetation is presented in Section 7.

6. UNCERTAINTY ANALYSES

There are two key pieces of ecological knowledge that are central to the ecological model: the index curves which convert water attributes (e.g. flood duration) into water suitability index; and the weights which specify the relative importance of the water attributes. The objective of the uncertainty analysis is to develop a better understanding of how the uncertainties in the ecological knowledge contribute to the model outcomes. A preliminary study was undertaken to investigate model uncertainty from two different sources of index curves and weights; this work is published in Fu and Merritt (2012). This was followed by a more comprehensive uncertainty analysis which assessed uncertainty from all possible index curves and weights defined by current state of ecological knowledge. This work has been submitted for publication (Fu and Guillaumea, Submitted).

In the preliminary study, we investigated how index curves and weights generated from two different sources (Roberts and Marston, 2011; Rogers and Ralph, 2010) contribute to the difference in model outputs. Both of these sources are based on extensive review of studies on species' response to flooding in the Murray-Darling Basin. The water requirements proposed in these two sources differ due to differences in literature selection and interpretation of the literature by the authors. We examined how these interpretations affect the outcomes of the Namoi ecological model. Two sets of index curves were produced based on (Rogers and Ralph, 2010) and (Roberts and Marston, 2011), respectively, and the model outputs were compared. The goal is to understand the robustness of the model and how the model behaves in face of imperfect knowledge and information in ecosystem's response to water regime. Details of this study is published in Fu and Merritt (2012), with the results reproduced below.

The results for Asset 2, Barbers Lagoon, are illustrated in Figure 12. In general, water regime (including surface and groundwater) is in favour of the growth of river red gum more than for water couch. This is consistent with the site condition where river red gum is much more widespread than water couch which only exists in patches. Unsurprisingly, significantly higher water suitability indices are reported during the wet period for both species, especially for the river red gum. This is because apart from longer and more frequent flooding events during the wet period, shallower groundwater levels (average 8.6m below ground, SD=0.4) also sustains the growth of river red gum. In contrast, the average groundwater level dropped to 11.9m (SD=1.0) below ground during the dry period.

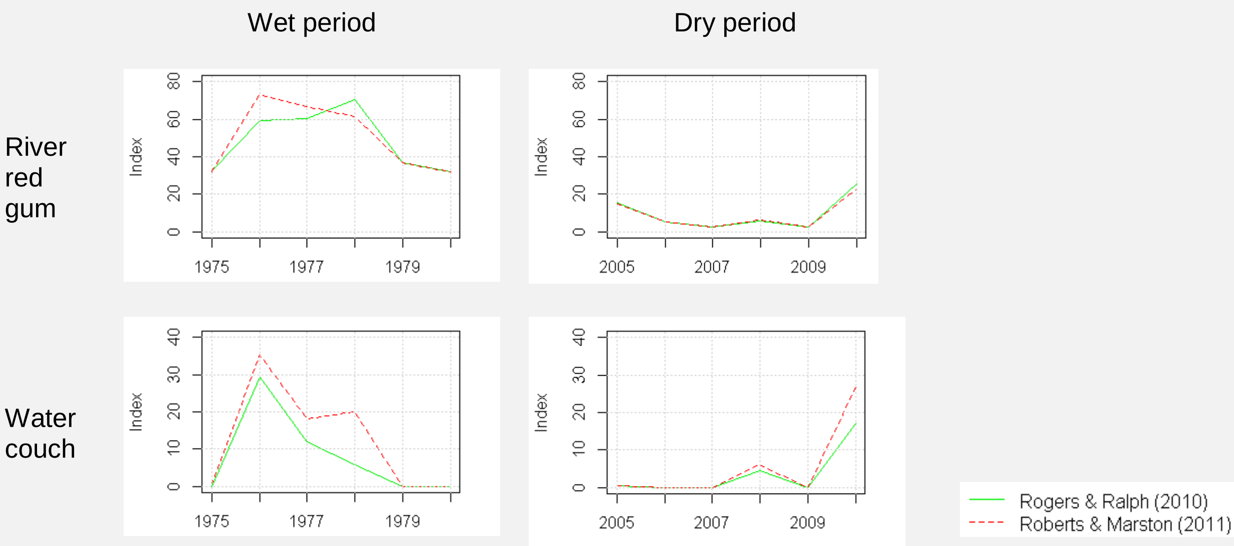


Figure 12. Annual water suitability index for river red gum and water couch at wet (1975-1980) and dry (2005-2010) periods at Barbers Lagoon, Namoi.

In general, greater differences are found for the wet period when using different sets of preference curves. This is because the wetter climate generates large and more frequent events, accumulatively resulting in greater differences in model outcomes. The differences in model outputs for river red gum are smaller than those for water couch, primarily because the preference curves generated for river red gum are more consistent between the two sources. In Australia, the ecology of river red gum has been studied more than water couch, and hence the water requirements of river red gum are better understood and well documented. In contrast, there is a greater uncertainty in the water requirements of water couch, leading to greater disagreement in preference curves and model outputs.

For river red gum, highest water suitability index is estimated in 1976 using the preference curves from Roberts and Marston (2011), while 1978 is estimated to be the best year when using preference curves from Rogers and Ralph (2010). This is mainly caused by differences in flood timing. The total flow in 1976 and 1978 is 1875 and 1310 GL, respectively. In 1976 most floods occurred from late January to early March, while in 1978 most floods occurred in July and then in September. The different opinions in preferred flood timing for the growth of river red gum contribute to the differences in water suitability outcomes.

In terms of water couch, the model outputs using preference curves from Roberts and Marston (2011) are consistently higher than from Rogers and Ralph (2010). In 1978, the estimated annual water suitability index using Roberts and Marston (2011) is more than three times as much as that from Rogers and Ralph (2010). This may be because the preference curves from Roberts and Marston (2011) cover greater ranges in all flood attributes (duration, timing inter-flood dry period) than those from Rogers and Ralph (2010), particularly in the preference of winter-spring floods. Interestingly, the references used in Rogers and Ralph (2010) to inform water requirements of water couch are mostly taken directly from Blanch et al. (1999), which is based on an once-off survey at the lower Murray, the southern end of the Murray-Darling Basin. In contrast, Roberts and Marston (2011) attempted to generalise water requirements based on longer term (several years) studies and observations, mostly at the Gwydir Wetlands, which are located at the north of the Namoi River Catchment.c

Source: Fu, B., Merritt, W., 2012. The Impact of Uncertain Ecological Knowledge on a Water Suitability Model of Riverine Vegetation, in: Seppelt, R., Voinov, A., Lange, S., Bankamp, D. (Eds*.), iEMSs 2012 International Congress on Environmental Modelling and Software: Managing Resources of a Limited Planet*, Leipzig, pp. 917-924.

The limitation of the above preliminary uncertainty assessment is that it only considers two sources, i.e. two sets of index curves and weights. Therefore, we undertook a more comprehensive uncertainty analysis which compares the suitability of surface water and groundwater regimes, albeit in two 15-year periods (Pre90:1975 – 1989; Post90: 1990 – 2005) but the approach could be easily extended to compare different environmental flow options. The uncertainty analysis consisted of defining constraints on index curves, weights of attributes and the direction of the relationship between these parameters. Within these constraints, we used linear programming to select index curves and weights that alternately maximise or minimise the difference between suitability indices, favouring each period in turn. The results reveal for a specific asset whether or not one period is clearly better than the other despite the uncertainties. The uncertainty analysis makes minimal assumptions about what is plausible, and evaluates whether extreme cases within these assumptions agree and are indeed plausible in terms of evaluating scenarios. This prompts learning about the boundaries of our knowledge, with the focus on what is certain rather than on quantifying uncertainty in a model output. Details of this uncertainty analysis has been submitted for publication (Fu and Guillaumea, Submitted), with selected results reproduced below.

Hydrology and groundwater

Average annual flows in the two test periods vary depending on the assets. At most assets average annual flows in the Pre90 period are higher than Post90. For example, in Gunnedah the average annual flow in Pre90 and Post90 are 716GL and 602GL respectively. Exceptions are Maules Creek, where the average annual flows are similar in the two periods (22GL and 21GL in Pre90 and Post90, respectively), and at Duncans Warrambool and Bugilbone to Walgett, where average annual flows in Post90 are higher than Pre90. For assets 1 to 8, respectively the ratios between Pre90 and Post90 average annual flow are 1.19, 1.24, 1.03, 4.35, 1.19, 0.71, 1.21, and 0.20. Wet periods are recorded in the late 1970s, mid 1980s, early 1990s and late 1990s. In most cases the Pre90 period has more low to medium flows (Figure 13). However, the exceedance probabilities of high flows are less distinguishable between the two periods. At Bugilbone to Walgett, however, more frequent high flows are recorded for the Post90 period.

During the Pre90 period, groundwater levels range from 6 m below ground at Upstream Mollee to 16 m below ground at Bugilbone to Walgett. Most assets have deeper groundwater levels in the Post90 period (Figure 14). Notable assets are Gunnedah and Wee Waa to Bugilbone, where average groundwater levels are 1.8 m deeper in Post90 than Pre90. Groundwater levels at Maules Creek and Bugilbone to Walgett are similar in the two periods. Groundwater at Duncans Warrambool is significantly shallower in the Post90 period than that in Pre90, with mean groundwater levels rising from 14.8 m below ground to 11.5 m below ground.

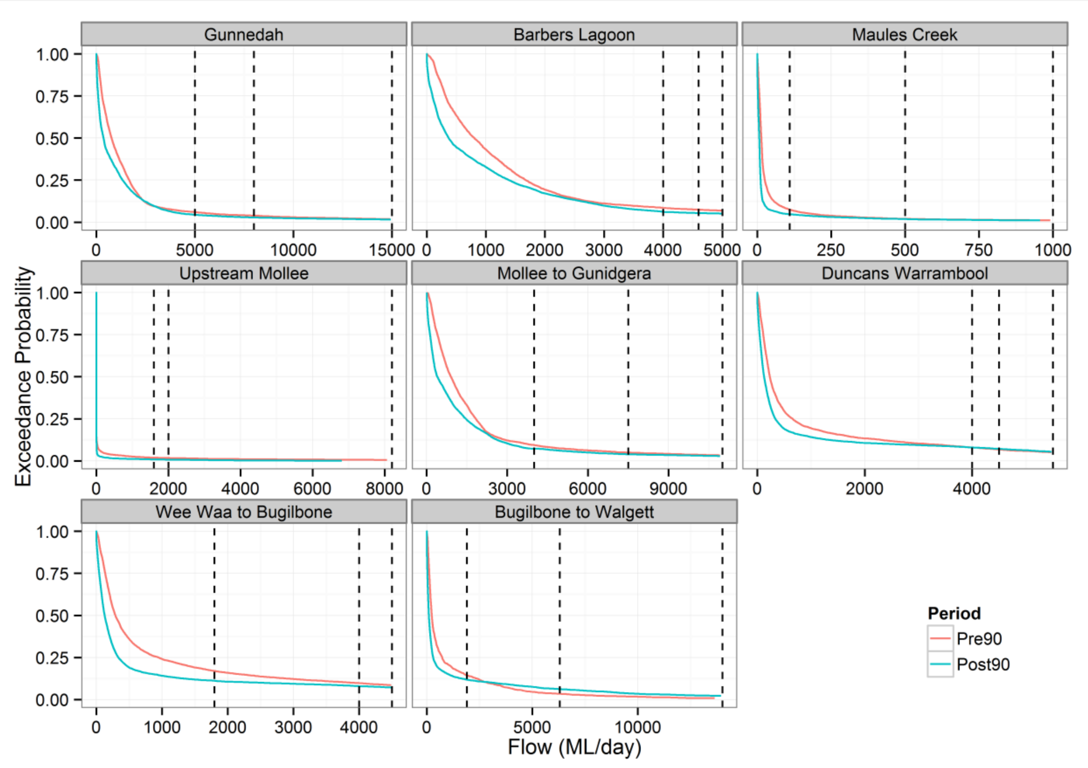


Figure 13: Exceedance probability curves of daily flow at modelled assets during Pre90 and Post90 periods. Dashed lines are identified low, middle, and high commence-to-flood levels corresponding to flooding of different areas.

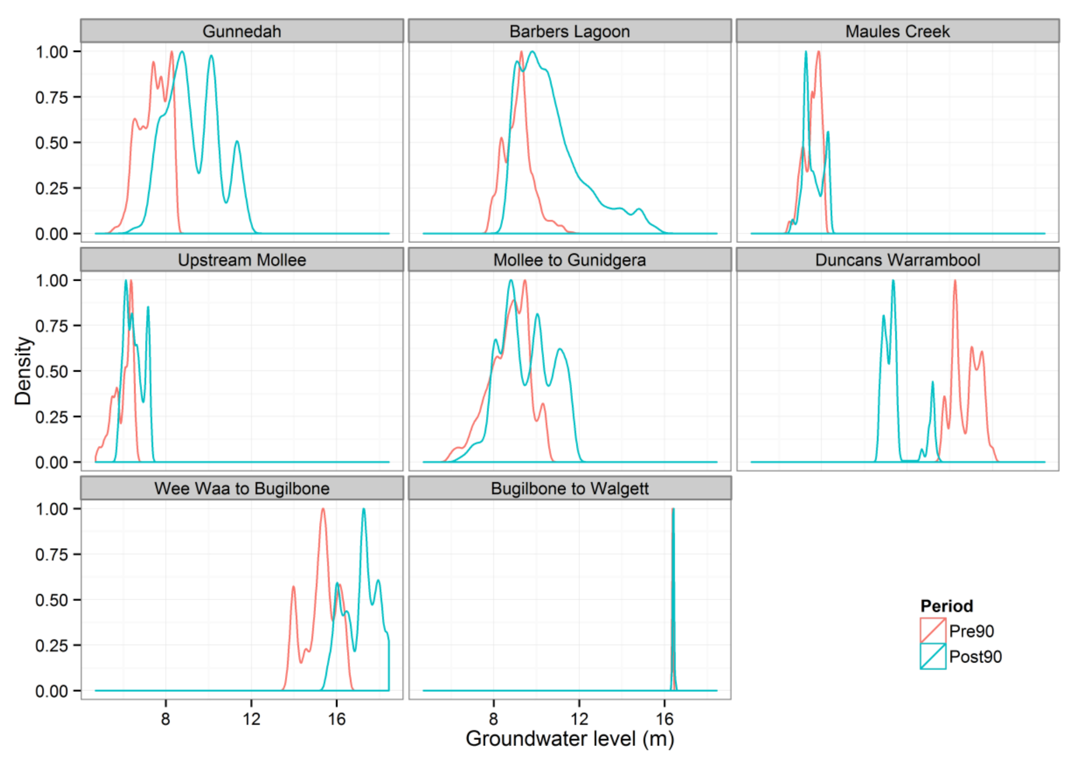


Figure 14: Groundwater level distributions in Pre90 and Post90 periods.

Surface water suitability

The results of uncertainty in surface water suitability are shown in Figure 15, with no constraints on the relative importance of flood attributes (i.e. any flood attribute can have a weight between 0 and 1, but all weights for a species must sum to 1). The results suggest that given any possible weights and any possible index curves within the defined boundaries, we can say for certain that, at Upstream Mollee for example, the Pre90 period always has a higher mean water suitability index than the Post90 period at all commence-to-flood levels for the regeneration of river red gum, water couch and lignum. In terms of maintenance of these species, we can be certain that suitability is better in Pre90s for areas further away from the banks, where the commence-to-flood level is above 4000 ML/day. Assuming current knowledge is correct, there is little benefit to further reduce uncertainty in weights and index curve for this asset if areas that require higher commence-to-flood levels are of concern. However, for the areas closer to the river banks, there is still uncertainty in evaluating which period is better and further knowledge is needed. Similarly for Bugilbone to Walgett, given uncertainties in weights and index curves, at high commence-to-flood levels we can still be certain that the Post90 period has a better surface water regime for the maintenance and regeneration of river red gum, water couch and lignum.

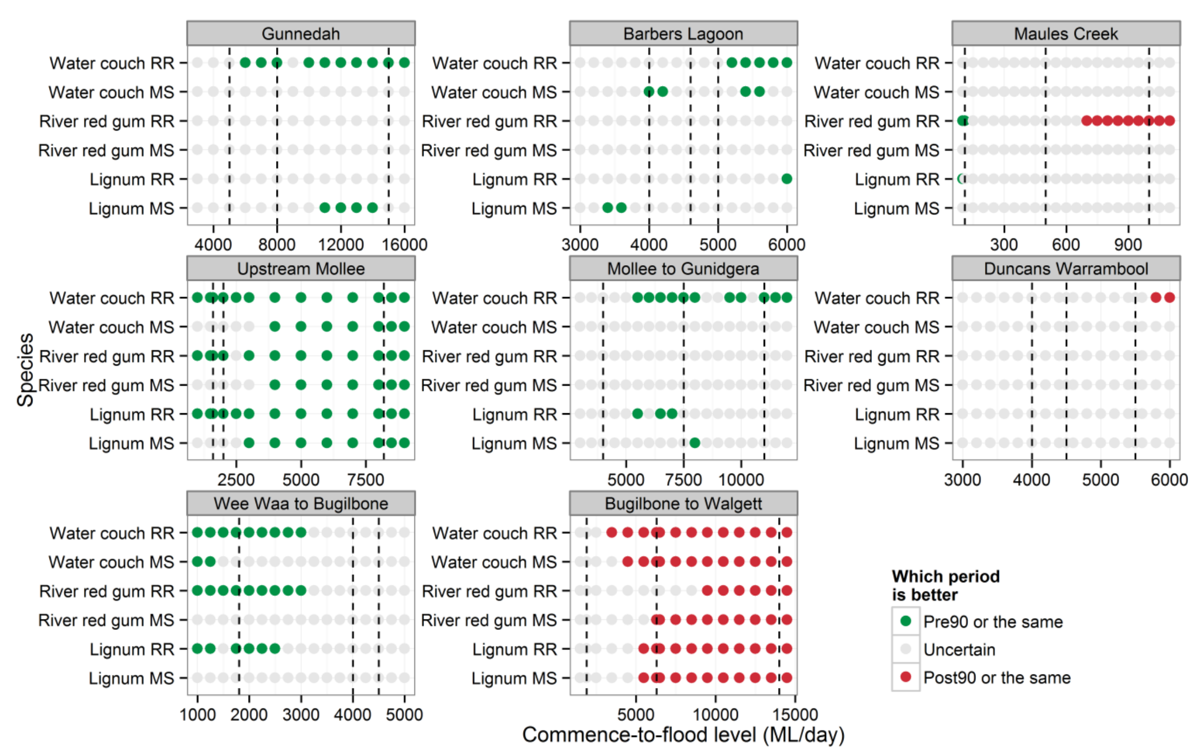


Figure 15: Uncertainty in surface water suitability index at various commence-to-flood levels, given constraints in index curves. Weight constraints are not specified. Dashed lines identify low, medium and high CTF thresholds for that ecological asset. MS: denotes Maintenance and Survival, RR: Regeneration and Reproduction.

Much higher uncertainties are found in the results for other assets (Figure 15). In these cases, the current knowledge defined by our constraints is often insufficient to evaluate which period has a better surface water regime for different species. Variation between species becomes more visible. For example, in Gunnedah the uncertainty in evaluating periods is much lower for the regeneration of water couch (where the Pre90 period is better at most commence-to-flood levels). But for other species things are much more uncertain in Gunnedah. There are some cases where we can be certain for a given commence-to-flood level, but uncertain for higher and lower commence-to-flood levels. This means that the uncertainty becomes sensitive to the commence-to-flood level. For example, we can be certain that the Pre90 period is better for the maintenance of water couch at Barbers Lagoon at commence-to-flood levels of 4000 and 4200 ML/day (assuming current knowledge in index curves is correct). However, we do not know which period is better for areas closer to the river bank or further away from it.

Overall, species within an asset exhibit similar direction of change. For instance, for Upstream Mollee the water regime in the Pre90 period generally produces better surface water suitability for all species; while for Bugilbone to Walgett the Post90 water regime is better for all species (Figure 15). However, between locations in the catchment, different direction of change can be detected. Bugilbone to Walgett has a noticeably favourable water regime in the Post90 period in comparison to other assets which favour the Pre90 period.

Adding additional constraints on the relative weights of each hydrologic attribute based on our assumed knowledge adds additional certainty in discriminating between the two periods (Figure 16). Significant improvement in certainty is found at Wee Waa to Bugilbone where the Pre90 water regime is better than Post90 at any commence-to-flood levels for the maintenance of water couch and regeneration of river red gum, and at lower commence-to-flood levels for the maintenance of lignum. There are scattered improvements in certainty across other assets, mostly in the maintenance of lignum and water couch. Additional constraints in the weights for river red gum have resulted in little gain in certainty when evaluating the surface water regime of the two periods.

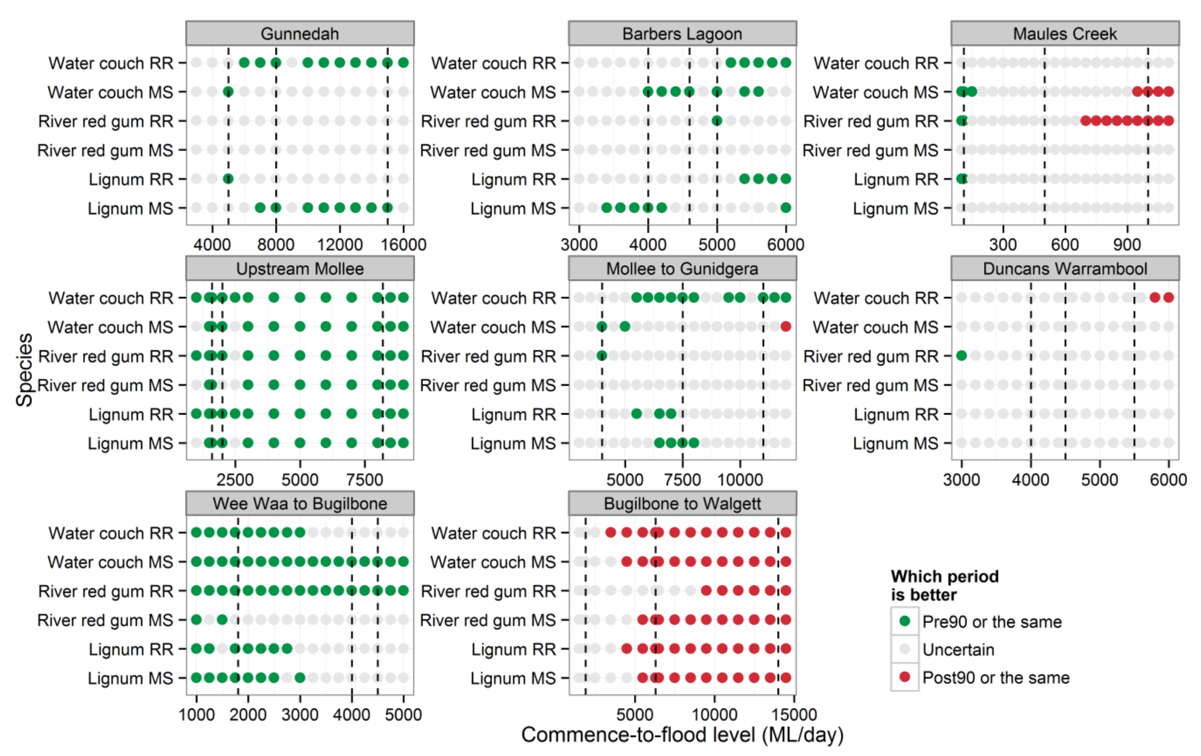


Figure 16: Uncertainty in surface water suitability index at various commence-to-flood levels, given constraints in index curves and weights. Dashed lines identify low, medium and high CTF thresholds for that ecological asset. MS: denotes Maintenance and Survival, RR: Regeneration and Reproduction.

Groundwater suitability

In terms of the uncertainty in the groundwater suitability index, it is found that in most cases the Pre90 and Post90 periods yield the same results as for the surface water case. This holds especially for the regeneration of all species, the maintenance of water couch for all assets (except Upstream Mollee), maintenance of lignum for the western assets and maintenance of river red gum for Bugilbone to Walgett (Figure 17). This occurs because the required groundwater levels are often shallower than the minimum observed groundwater levels. Groundwater levels in these cases are effectively all unsuitable. For the remainder of the species, given the current knowledge in groundwater requirements, we can be certain that in most cases Pre90 period is better, with the exception of the maintenance of river red gum for Duncans Warrambool. Only three cases were found uncertain: maintenance of lignum and river red gum at Maules Creek, and maintenance of river red gum at Upstream Mollee.

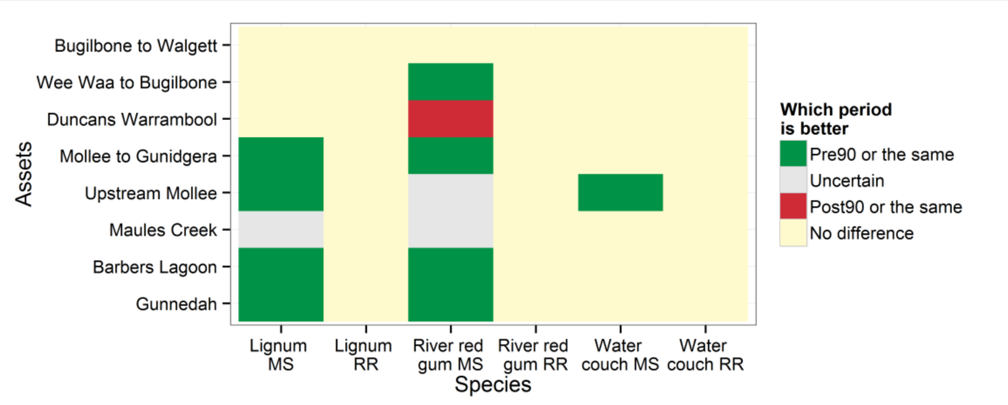


Figure 17: Uncertainty in groundwater suitability index, given constraints in groundwater level index curves. MS: denotes Maintenance and Survival, RR: Regeneration and Reproduction.

Combined surface water and groundwater suitability

The uncertainty in combined surface water and groundwater suitability index is similar to that in surface water suitability with both weight and index curve constraints (Figure 18). The only difference is that for Upstream Mollee the outcomes for the maintenance of river red gum are no longer as certain as for surface water suitability. This is because of the uncertainty introduced by groundwater suitability. The effect of groundwater uncertainty at Maules Creek is not seen for the maintenance of river red gum and lignum because these species are already uncertain according to surface water suitability. This result suggests that for the two periods evaluated uncertainty in groundwater levels has a small impact on the overall uncertainty of the water suitability index.

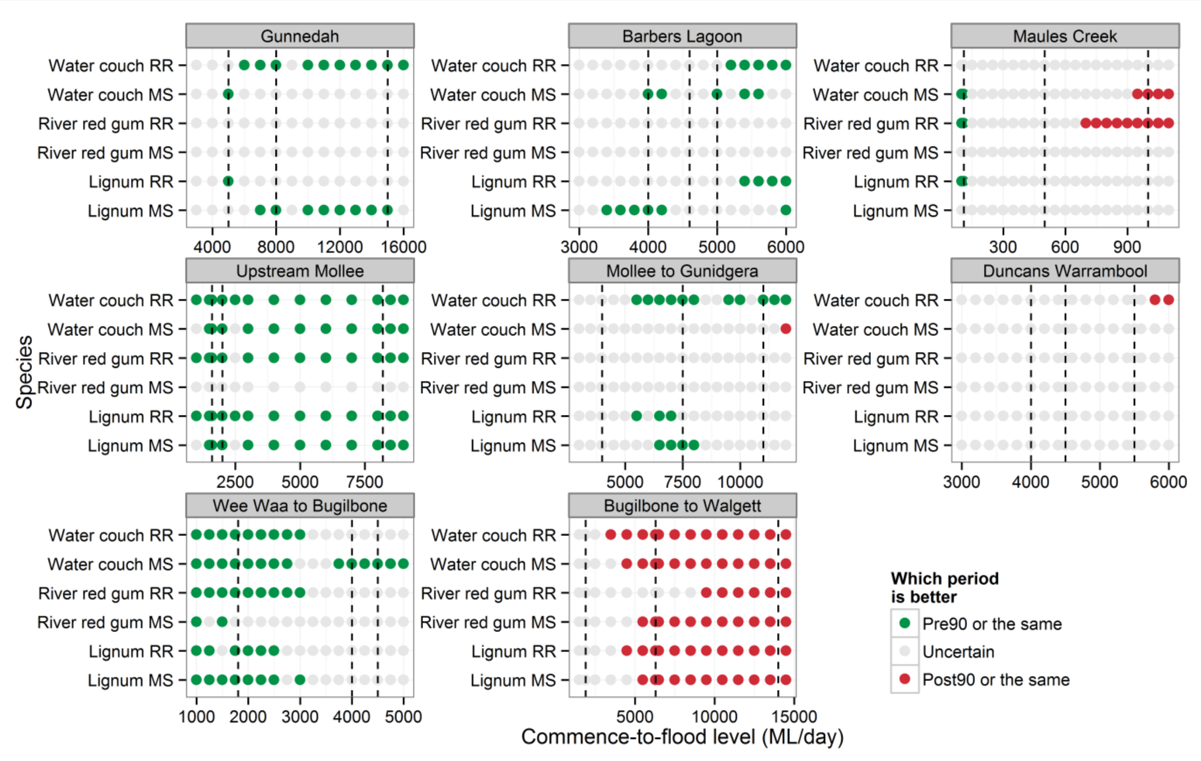


Figure 18: Uncertainty in combined surface water and groundwater suitability index at various commence-to-flood levels, given constraints in index curves and weights. Dashed lines identify low, medium and high CTF thresholds for that ecological asset. MS: denotes Maintenance and Survival, RR: Regeneration and Reproduction.

Evaluation of current knowledge

Broadly we could make conjectures about water suitability for riparian vegetation based purely on the volume of water, by assuming that more water is generally better. Under this assumption, we could reach conclusions about water suitability at the studied assets by simply comparing average annual flow in the Pre90 and Post90 periods The Pre90 period would be significantly better than the Post 90 at Upstream Mollee (ratio between Pre90 and Post90 average annual flow is 4.35); Post90 would be significantly better than Pre90 at Bugilbone to Walgett (Pre90/Post90 ratio is 0.20); while the differences between the two periods in other assets are probably marginal (Pre90/Post90 ratios range from 0.71 to 1.24).

However, there are two major limitations in adopting such a flow volume-based method. Firstly, the ecological knowledge is much more advanced, and in many cases it is known that volume of water is not the only factor contributing to the maintenance and regeneration of riparian vegetation. Too much water can have undesirable outcomes like tree die back due to prolonged flooding (Roberts and Marston, 2011). Other factors such as flood timing can also be important, particularly for regeneration (Roberts and Marston, 2011). Secondly, the flow volume-based method does not differentiate between different vegetation species, whereas many ecological studies have demonstrated different water requirements by different vegetation species (Rogers and Ralph, 2010). From a water management point of view, differentiating between species can help identify water management targets and monitoring strategies. Therefore, this valuable ecological knowledge should be used to help us better understand and predict water suitability for riparian vegetation. The question is, “Is the current state of ecological knowledge sufficient to help us make a robust decision about preferred water regimes” - in this illustrative case being able to evaluate comparative water suitability between two periods?

Our results suggest that while the results are broadly consistent with the expected outcomes based on flow volume, they show variability and uncertainty due to the particulars of species, assets and commence-to-flood thresholds. By considering this variation, we can identify useful differential impacts of a hydrological regime, as well as propositions as to where trade-offs may need to be made between protecting particular species or locations. With the hydrological inputs specifically used in this paper and the uncertainties in index curves and weights that were framed by current ecological knowledge, we still have relatively high certainty in discriminating between the two periods at Upstream Mollee and Bugilbone to Walgett. As a result, little additional knowledge (i.e. having tighter constraints) is needed. However, much higher uncertainties in model outputs are found for other assets, suggesting that current state of knowledge is insufficient in these cases to conclude which period is better. In these cases, better ecological knowledge in vegetation water requirements is needed to reduce uncertainty.

Spatially, certainty in model outputs varies depending on the commence-to-flood levels. In general, for the commence-to-flood levels investigated there are more certainties in water suitability for the mid to higher benches compared to the areas closer to the river. Identifying where changing commence-to-flood levels result in change in outcome suggests possible sensitive tipping-point thresholds if commence-to-flood levels are considered to be uncertain. If commence-to-flood levels are considered to correspond to spatial locations, this transition identifies marginal areas that are more likely to be impacted by any change. Overall, the ability to consider patterns across species, assets and commence-to-flood levels helps draw attention to whether trends are widespread, and explain why a particular outcome is observed.

Note that the outcome of knowledge evaluation is case specific and thus only applies to the two water regimes (in this case, Pre90 and Post90) investigated. These results can be more useful when evaluating scenarios of different water management options. Identifying situations where knowledge is sufficient and conclusions are certain helps establish robust decision making; whereas identifying situations where current knowledge is insufficient helps direct monitoring and research efforts.

In this paper, we also demonstrate that a progression of additional knowledge can reduce or increase uncertainty in the model outputs, the extents of which vary on a case-by-case basis. For example, incorporating additional knowledge on the relative importance of flood attributes (i.e. weights) adds additional certainty in discriminating between the two periods. However, this added certainty is not distributed equally across assets and species. We found significant improvement in certainty in evaluating water suitability for some species at a specific asset (e.g. maintenance of water couch at Wee Waa to Bugilbone), but little improvement in some other species or assets. In contrast, adding additional knowledge with respect to vegetation requirements on groundwater levels led to increased uncertainty in overall water suitability for the maintenance of river red gum at Upstream Mollee. However, in other cases the effect of uncertainty in groundwater level is trivial because the surface water suitability is already highly uncertain. In breaking down the uncertainty analysis, we are able to more specifically identify which component of the model is contributing to the uncertainty in the model outputs, and therefore not only identifying knowledge gaps but also recognising the different roles of these knowledge gaps for different species and assets. Consequently, research into closing knowledge gaps can be better targeted depending on desired outcomes in model certainty.

Source: Fu, B., Guillaumea, J.H.A., Submitted. Assessing certainty and uncertainty in habitat suitability models by comparing extreme cases. *Environmental Modelling & Software*.

7. PRELIMINARY ASSESSMENT OF NDVI

We explored 23 years of NDVI data for the modelled assets in order to gain better understanding of riparian vegetation responses to water availability and potentially test the ecological model. Preliminary results suggested that NDVI data in its current form is not suitable to test the suitability of hydrological and groundwater regimes due to the predominant climatic impacts on NDVI. Although NDVI data has not been useful yet within this study, it shows promises. Firstly, the data is relatively comprehensive. We were able to generate 228 usable datasets at a resolution of 30m between 1987 and 2010. This resolution is especially suitable for Namoi which has fairly narrow corridors of riparian vegetation. Secondly, inter-flood dry period and groundwater levels were found important in classifying NDVI, although in most assets their influences were outweighed by maximum temperature and local rainfall. Further study is needed to analyse and interpret NDVI and soil moisture to understand vegetation response in order to update and calibrate the existing ecological model.

Preliminary assessment of NDVI is described in Fu and Burgher (Submitted). Selected contents on methods and results are reproduced below.

Methods

Existing riverine vegetation in the Namoi River Catchment often occurs in a fairly narrow fashion along water courses (see Figure 19). We thus used remote sensed data acquired by the Landsat 5 TM and Landsat 7 ETM satellites as their 30 m resolution can target such a fine spatial configuration. This avoids the inclusion of potentially irrigated crops which would have been almost impossible with commonly used larger resolution imagery such as MODIS or AVHRR (with resolutions of 250 m or greater). Another important advantage of using Landsat imagery is the early start time of Landsat 5 which has been operating since 1984, allowing for analysis of a relatively long time-series of data. In contrast, 250 m resolution data from MODIS is only available from 2000.

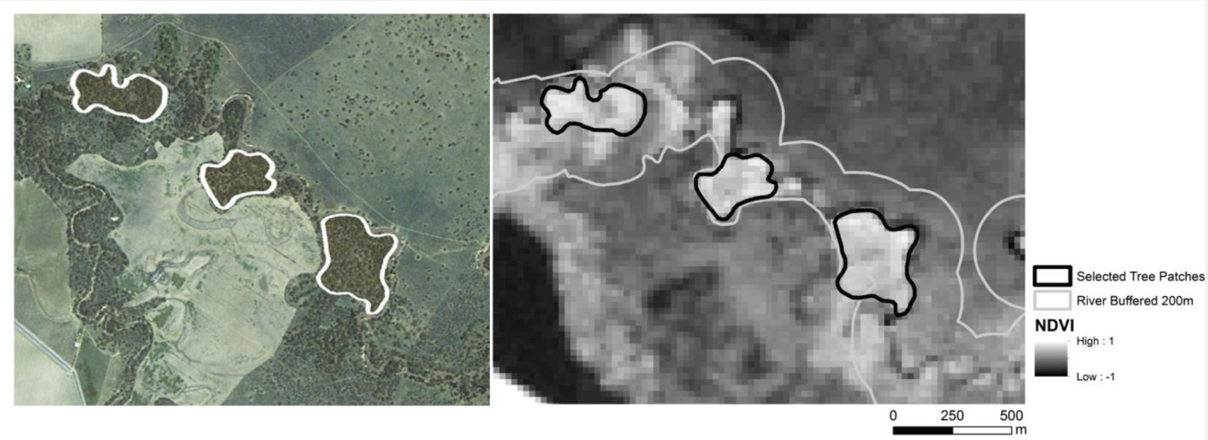


Figure 19: Section of Wee Waa to Bugilbone as viewed by Google Earth showing selected tree patch polygons (left) and LandSat derived NDVI values, showing tree patch and 200m river buffer with crops clipped out (right). NDVI values tend to increase near the river channel due to presence of riparian vegetation.

We acquired Landsat images which provided coverage of all assets (path 91/ row 81) from Geoscience Australia (1987 - 1998) and the United States Geological Survey (1999 – 2010) comprising a data time-series of 23 years. Ideally, Landsat offers images on a periodic 16-day basis. However, image availability from the data distribution agencies and obscuring of assets from periodic cloud cover means that the number of usable data points for the years examined is inconsistent. We extracted a total of 228 usable images, with most years having between 8 to 15 images, although the years 1987, 1995, 1998, 1999, 2005 and 2008 have 5 images or fewer.

Using ENVI 5.0 (Exelis Visual Information Solutions, 2012), images were corrected for sensor defects and sensor differences by converting to top of atmosphere reflectance using published post-launch gains and offsets (Chander et al., 2009). Dark object subtraction was then performed using the band minimum from each image. Dark object subtraction was chosen because this method is one of the simplest yet most widely used methods of atmospheric correction for land-use classification and change-detection purposes (Song et al., 2001). Following this correction, NDVI was calculated for all images as per Eq. 1 using bands 3 and 4 in Landsat which have been calibrated to sense radiation in the visible (*Red*) and near-infrared (*NIR*) regions of the spectrum respectively.

$$NDVI = NIR - Red / NIR + Red \qquad \text{(Eq. 1)}$$

NDVI values range between -1.0 and 1.0 with values nearing zero and below indicating features which are not vegetated such as water, snow, ice, clouds and barren surfaces.

We estimated NDVI values for two different types of riparian areas: tree patches which are predominantly river red gums, and riparian vegetation zones which consist of riparian trees and grasses. For tree patches, polygons were manually drawn around selected areas of fairly dense forest adjacent to river channels within assets using Google Earth (Figure 19). Mean NDVI across each asset was calculated for each date. For riparian vegetation zones, the river channels within assets were buffered by 200 m on each side and areas corresponding with crops clipped out (Figure 19). Then, in each asset the area of each NDVI classes: [–1, 0), [0, 0.2), [0.2, 0.4), [0.4, 0.6), [0.6, 0.8), [0.8, 1] was calculated. This area was then standardised for each asset by calculating the proportional area of each class in each asset. We used NDVI classes rather than mean NDVI values for riparian vegetation zones because the variations of NDVI are too high due to large areas and different riparian vegetation types. Mean NDVI values cannot fully reflect the NDVI values in the riparian vegetation zones in each asset. In contrast, variations of NDVI in tree patches are very small and mean NDVI values are good indicators of NDVI values in each asset.

Daily rainfall records were obtained from the Australian Bureau of Meteorology website, using the most proximate station to each asset which had complete or near complete data records (Australian Bureau of Meteorology, 2013). In the vicinity of Duncan's Junction, Wee Waa to Bugilbone and Bugilbone to Walgett, only one rainfall station had sufficient data, and this station is used for all three assets. Daily surface flow data before 2008 was extracted from PINNEENA 9.2 (NSW Department of Water and Energy, 2008a), with more recent data taken from the NSW government water information website (waterinfo.nsw.gov.au). River gauges were selected based on their proximity to the assets. Historical groundwater bore data was obtained from Groundwater PINNEENA 3.2 (NSW Office of Water, 2011c). The groundwater bore data was interpolated into daily time series using a linear regression (Blakers, 2011b). Bores were selected based on their proximity to the asset, and the mean value of the daily groundwater levels at Pipe 1 (the pipe with the shallowest opening) of these bores was used to represent the groundwater level for that asset.

Three types of analyses were undertaken to investigate NDVI dynamics of tree patches and riparian vegetation zones: 1) long term trend of NDVI in the past 23 years; 2) seasonal behaviour of NDVI; and 3) relationships between NDVI and climate and hydrological variables using classification. Annual average NDVI tree patch and class area percentage values for each asset were used to identify long-term trends and variability over the 23 year period. Average monthly NDVI values over the 23 year period were used to assess seasonality over the year. Regression tree analysis was used for classification.

Regression tree analysis is a commonly used statistical method for non-parametric regression and classification that has been widely applied for ecological data (De'Ath and Fabricius, 2000). It generates a regression tree that classifies a response variable based on explanatory variables; in doing so thresholds can be identified to best separate values of the response variable. We used the *ctree()* function in the **party** package in R (Hothorn et al., 2006) for regression tree analysis. The explanatory variables used for regression tree analysis include daily maximum temperature (Temp_max), antecedent rainfall (Rainfall_total), antecedent flow (Flow_total), groundwater level (Groundwater), inter-flood dry period (indicated by days since last flood, Days_lastflood), the proportion of flow relative to the overbank flood threshold (Perc_low), and the mean Perc_low over the past days (Perclow_mean). We used Perc_low to account for variation in channel capacity across assets, thus a Perc_low value of 1 indicates that the flow is at the overbank flood threshold regardless of the channel size at different assets. We used antecedent rainfall, antecedent flow and Perclow_mean to account for potential lag response of NDVI to rainfall and flow. The lag was identified by calculating 0 to 60 days (at a 1 day interval) of total rainfall, total flow and mean percentage low, then estimating correlations between NDVI and these antecedent values. The lags that correspond to the highest correlation were used to calculate Rainfall_total, Flow_total and Perclow_mean.

These explanatory variables were related to three sets of response variables for regression tree analysis: 1) mean NDVI values of the tree patches at all assets; 2) proportion of areas that has high NDVI (i.e. NDVI > 0.6) in riparian vegetation zones that have dense trees, which include assets 5-7 (Duncan's Junction, Wee Waa to Bugilbone, Bugilbone to Walgett); and 3) proportion of areas that has high NDVI (i.e. NDVI > 0.6) in riparian zones that have sparse trees, which include assets 3 and 4 (Upstream Mollee and Mollee to Gunidgera). Calculation of these NDVI was described in Section 3.1. We separated NDVI data for riparian zones with dense and sparse tree vegetation in order to reduce the impact of land clearing on areas of NDVI values.

Results

In terms of tree patches, maximum temperature is the primary variable classifying the NDVI data, with cooler maximum temperatures (<33.9) tending to relate to higher NDVI for tree patches than warmer maximum temperatures ($p<0.001$) (Figure 20). In either case, after temperature, antecedent rainfall (i.e. 28 days total rainfall) becomes the next variable which splits the data. When maximum temperatures are cooler (<= 33.9 °C), higher antecedent rainfall (> 23.8 mm) is related to observations with the highest NDVI ranges (mean = 0.60, sd = 0.07) in the tree patches (Node 6 in Figure 20). When antecedent rainfall is lower (< 23.8 mm) inter-flood dry period becomes an important split for the data ($p < 0.001$), with shorter dry periods associated with higher NDVI.

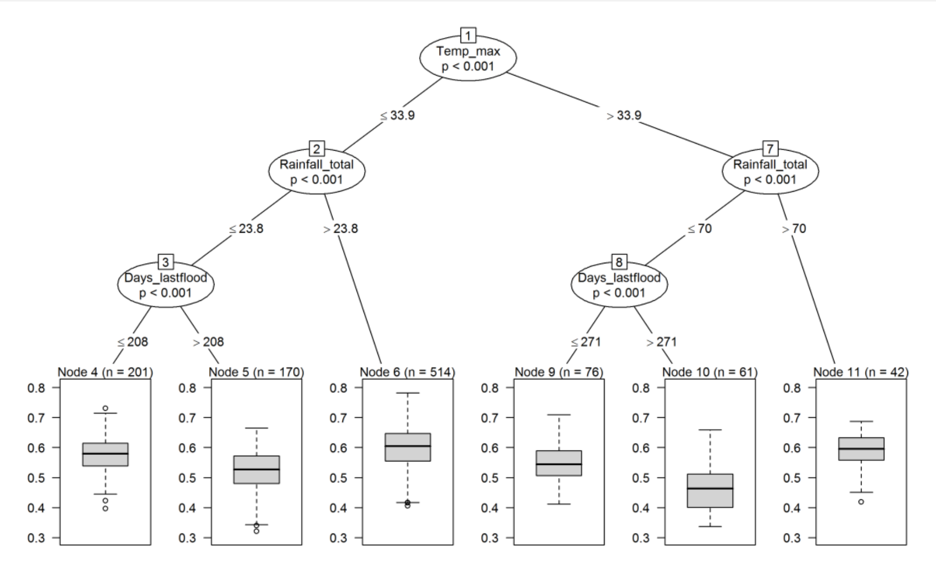


Figure 20: Regression tree showing classification of mean NDVI for the tree patches in all assets except Pian Ck.

When the weather is hotter (Temp_max > 33.9 °C) and when the antecedent rainfall in the past 28 days exceeds 70 mm, high NDVI values (mean = 0.59, sd = 0.07) were recorded for the tree patches (Node 11 in Figure 20). This is similar to cooler weather though with a lower rainfall threshold (23.8 mm) to achieve similarly high NDVI values (Node 6 in Figure 20). However, if the weather was hotter (Temp_max > 33.9 °C) and dryer (Rainfall_total <= 70mm), and with longer dry periods (Days_lastflood > 271 days), the lowest range of NDVI values was recorded for the tree patches in the assets with a mean of 0.46 (sd = 0.07) (Node 10 in Figure 20).

Compared to tree patches (Figure 20), a similar regression tree was generated when relating proportion of NDVI class areas with climatic and hydrological variables (Figure 21). It depicts maximum temperature above or below 25.5 °C as the primary determinant of the proportion of areas of high NDVI (i.e. NDVI > 0.6). Lower maximum temperatures are generally related to larger proportions of high NDVI area.

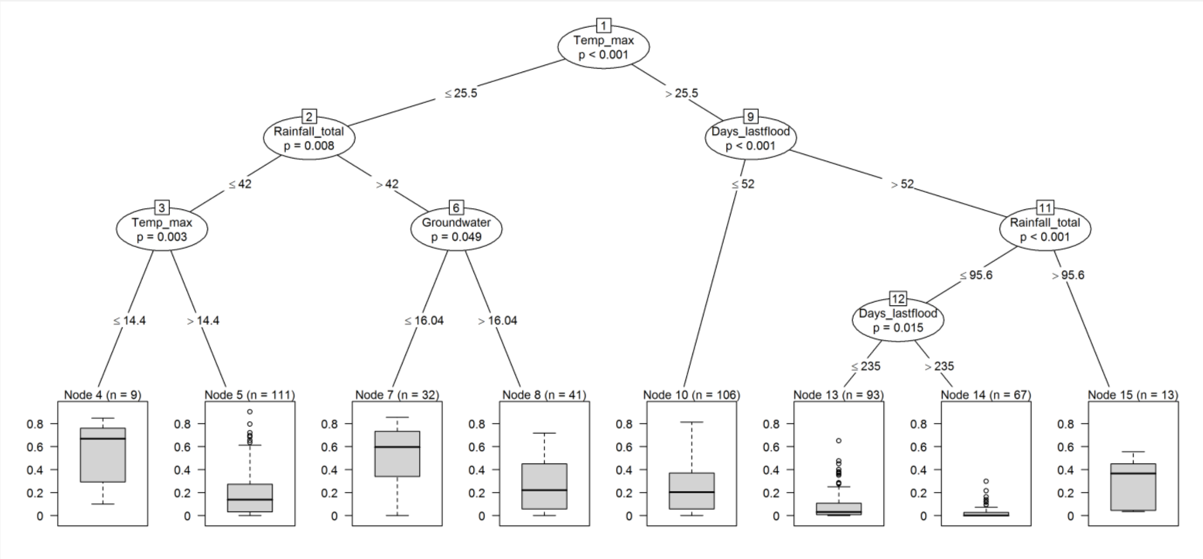


Figure 21: Regression tree showing classification of proportion of area with high NDVI (>0.6) for the assets with dense riparian vegetation (assets 5-7).

When maximum temperatures are lower than 25.5 °C, larger proportions of high NDVI area were found in situations when the antecedent rainfall is low (Rainfall_total <= 42 mm) but cooler (Temp_max <= 14.4 °C) (Node 4 in Figure 21), or when antecedent rainfall is high (Rainfall_total > 42 mm) and groundwater is shallower (Groundwater <= 16.04 m) (Node 7 in Figure 21). In contrast, smaller proportions of areas of high NDVI were found related to situations when it is warm and dry (Node 5 in Figure 21), or wet but with deeper groundwater (Node 7 in Figure 21). When maximum temperatures are cooler (Temp_max <= 25.5 °C) and antecedent rainfall is higher (Rainfall_total > 42 mm), area of high NDVI tends to comprise more than 40% of the assets if groundwater is shallower than 16.04 m (Node 7 in Figure 21), but less than 40% of asset area if groundwater is deeper (Node 8 in Figure 21).

When the weather is warmer (Temp_max > 25.5 °C), the next determinant of high NDVI area is the inter-flood dry period. Situations when the last flood was occurred more than 52 days ago are related to very low areas of high NDVI (in average 7%, Nodes 13 and 14 in Figure 21), unless antecedent rainfall has been very high (Rainfall_total > 95.5 mm) (Node 15 in Figure 21). When the weather is warmer but when flooding occurred in the past 52 days see greater NDVI, when in average 20% of the asset areas has high NDVI (Node 10 in Figure 21).

Similar to tree patches and dense riparian vegetation zones, the primary split in high NDVI area for riparian vegetation community with sparse trees is determined by maximum temperature, with 21.2 °C as the pivotal temperature (Figure 22). Below or equal to 21.2 °C, area of high NDVI has a large spread but has a median of about 40% of the assets. Above a maximum temperature of 21.2 °C, rainfall becomes the next most important variable. When antecedent rainfall is less than 54.8 mm and days since last flood exceed 36 days, areas of high NDVI are very low (Node 6 in Figure 22). Warmer temperature, lower rainfall but recently flooded dates (Node 5 in Figure 22) have higher NDVI areas which are within a similar range to cooler temperature (Node 2 in Figure 22). If antecedent rainfall is greater than 54.8 mm and maximum temperature is higher than 21.2 °C (Node 7 in Figure 22), this can produce a slightly lower range of high NDVI area as compared to when the weather is cooler (Node 2 in Figure 22).

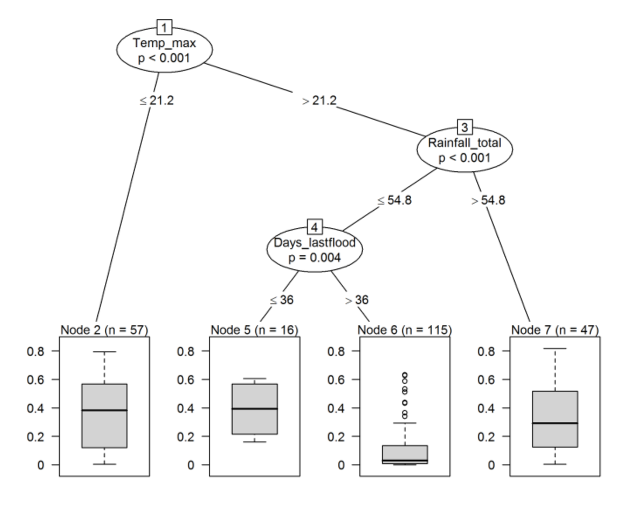


Figure 22: Regression tree showing classification of proportion of area with high NDVI (>0.6) for the assets with sparse riparian vegetation (assets 3 and 4).

Conclusion

Regression tree analysis was used to investigate how water availability affects vegetation vigour in riparian zone in nine ecological assets in the Namoi catchment, Australia. This was achieved by relating a range of climatic and hydrological variables to 23 years of NDVI data. Of these variables, the regression trees presented here consistently indicate that maximum temperature is the variable that primarily splits NDVI values, followed by antecedent 28-day rainfall and then the surface water variable (inter-flood dry period) and groundwater levels. Maximum temperature is the dominant variable and is negatively related to NDVI. This can be due to lower soil moisture caused by higher temperature, especially in semi-arid regions when rainfall is depleted. More rain is required in the warmer months compared to cooler months to achieve similar mean NDVI values in tree patches or areas of high NDVI in riparian zones, presumably because of higher evaporation. Inter-flood dry period was identified to be the only significant surface water variable in regression trees. Generally this variable becomes important when rainfall is limited. Our study also suggested that shallower groundwater levels sustain the NDVI and hence vegetation greenness when cooler and wetter.

Source: Fu, B. & Burgher, I. (Submitted). Riparian vegetation NDVI dynamics and its relationship with climate, surface water and groundwater levels. *Journal of Arid Environments*.

ACKNOWLEDGEMENT

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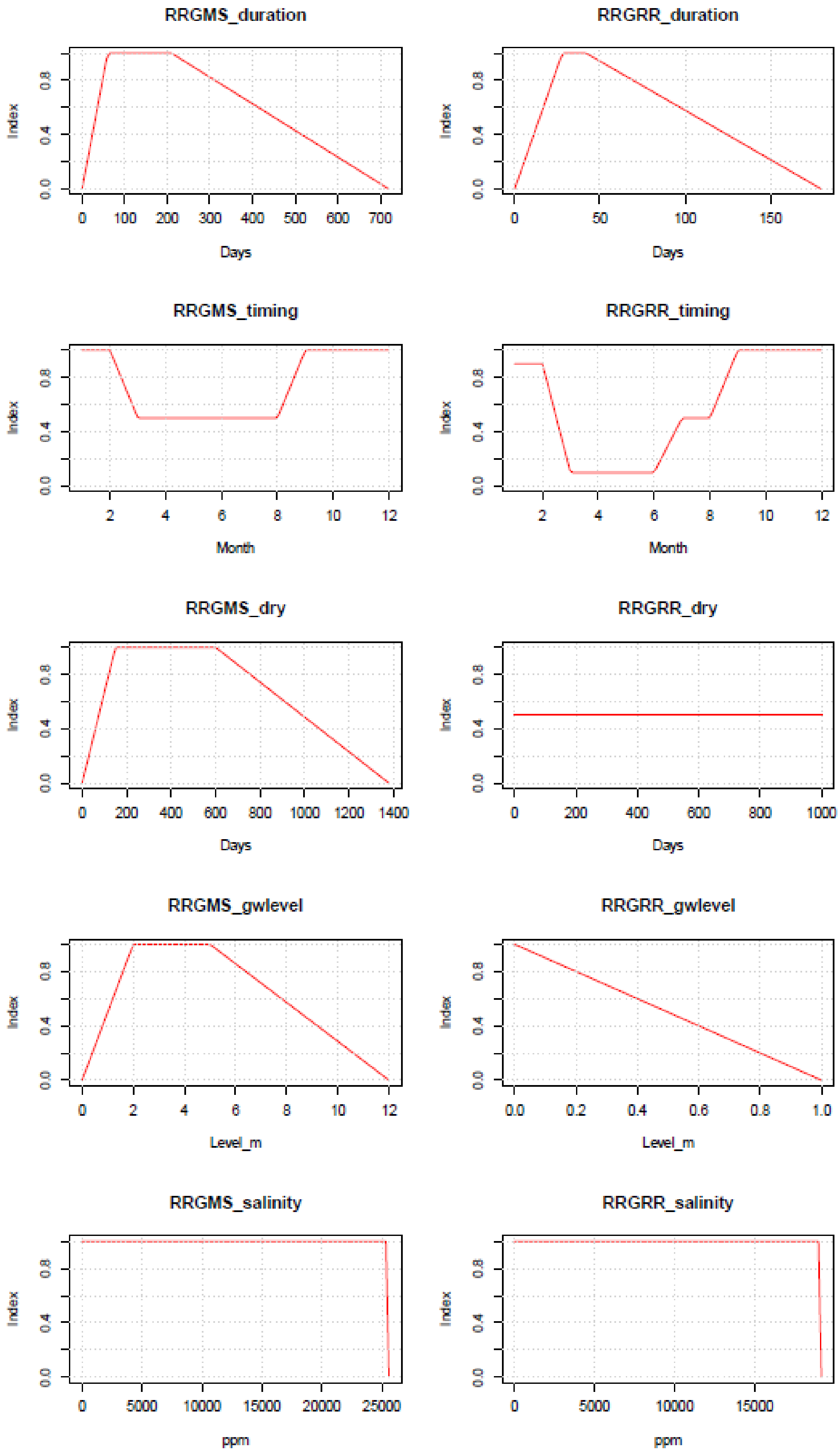
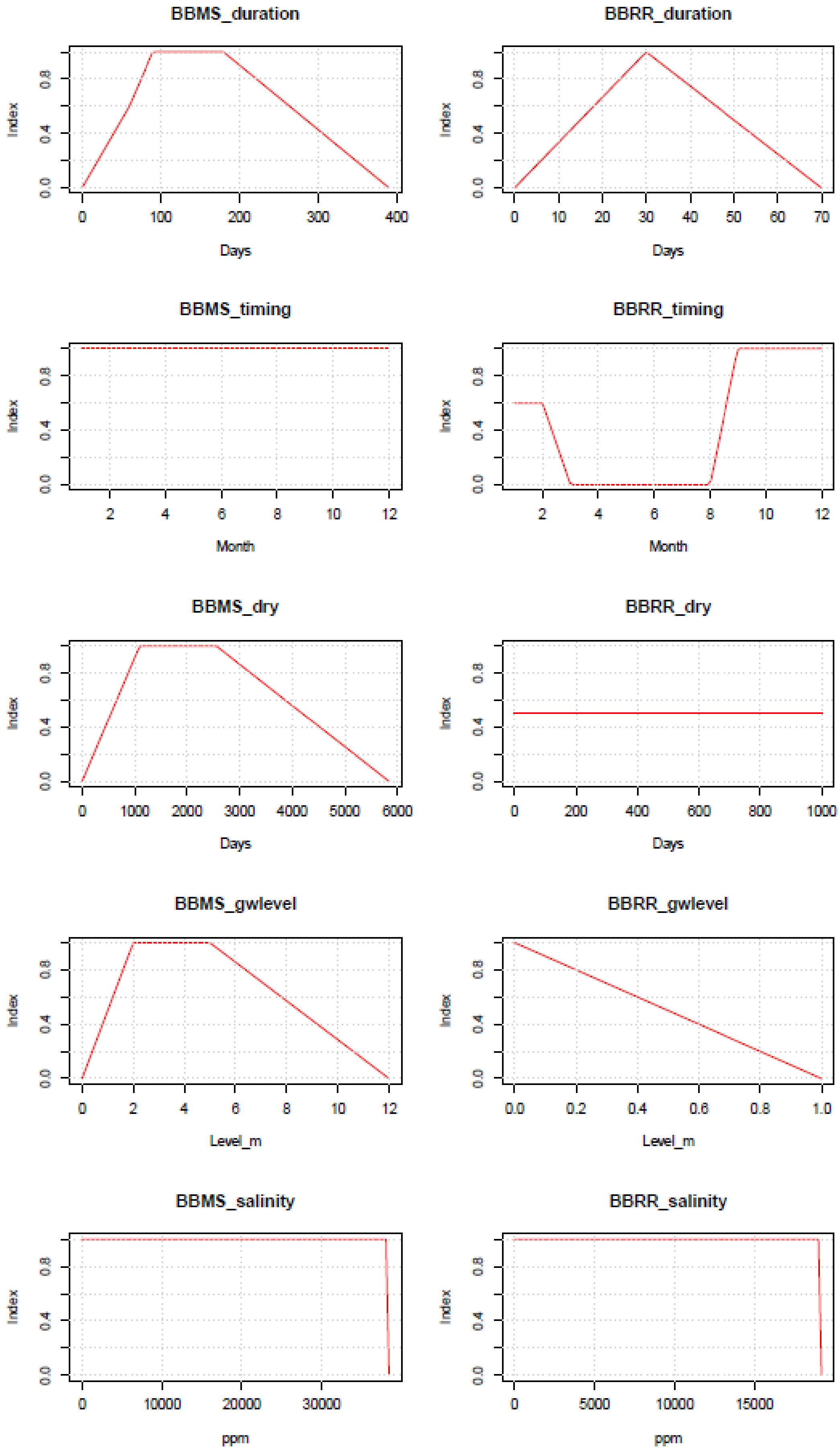
Appendix 1: Index curves for i) maintenance and survival (MS) and ii) regeneration and reproduction (RR) of river red gum (RRG), black box (BB), lignum (LG) and water couch (WC).

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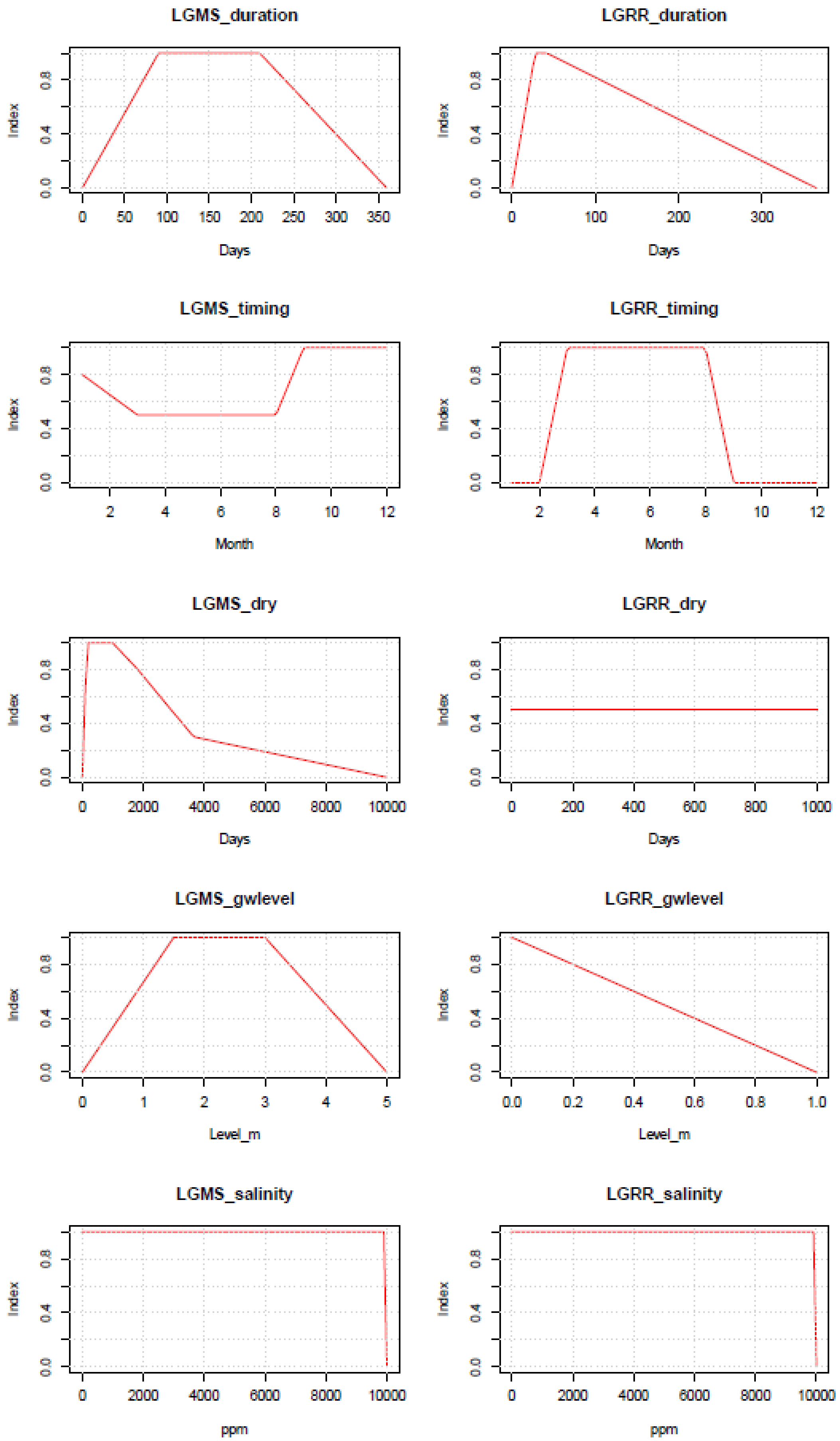
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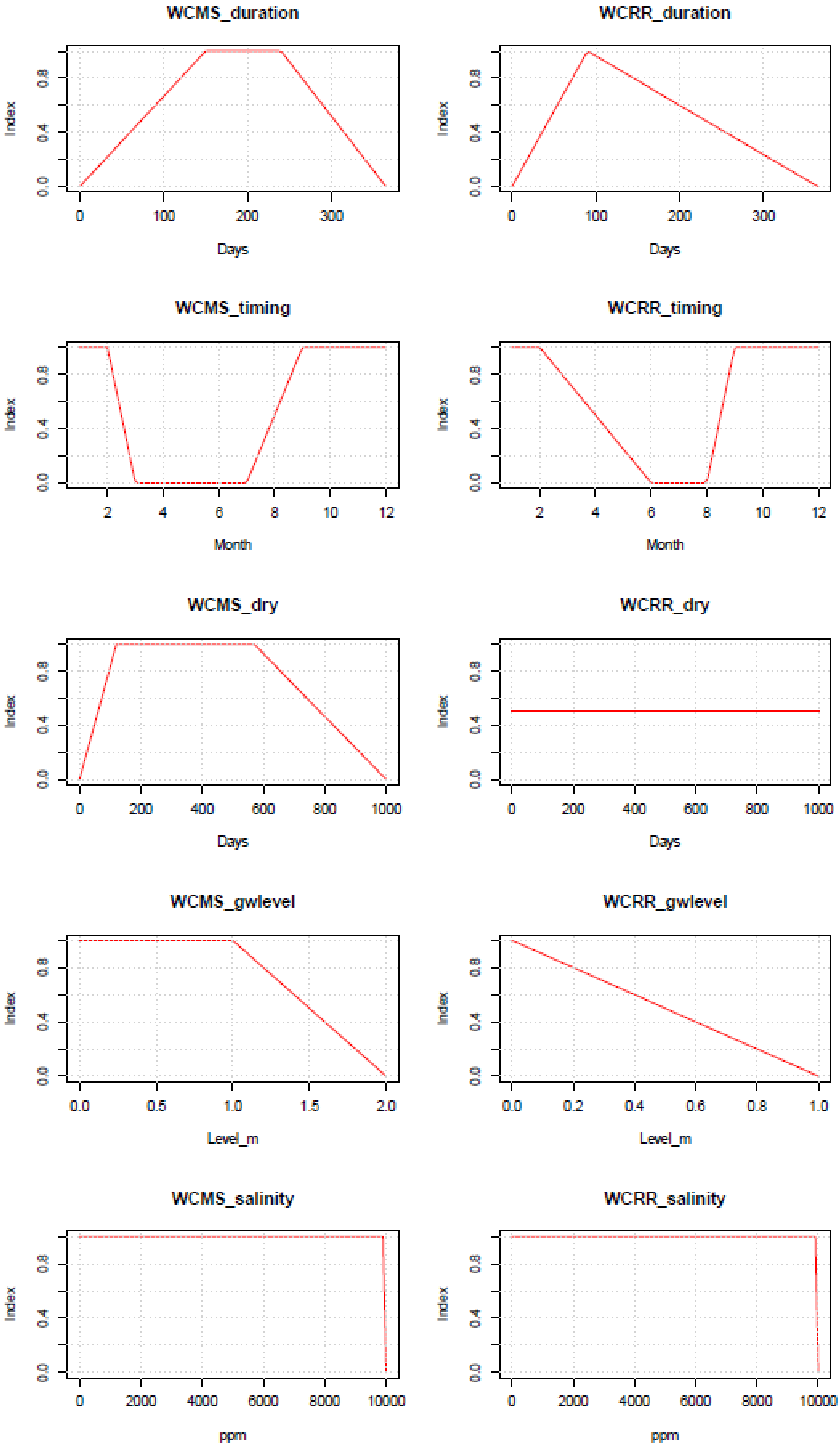
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