Using linked surface-groundwater catchment modelling to assess protection options for environmental assets threatened by dryland salinity in southern-eastern Australia.

Craig Beverly[A], Anna Roberts[A,B,C], Mark Hocking[D], David Pannell[BE] and Phil Dyson[F]

[A]Department of Primary Industries, RMB 1145 Rutherglen, Victoria, Australia, 3685.

Ph: +61 2 60 304500, Fax: +61 2 60 304600, email craig.beverly@dpi.vic.gov.au

[B]Future Farm Industries Cooperative Research Centre

[C] Previous publications are under the name A.M. Ridley

[D]Hockingetal Pty Ltd, 55 Willis Street, Hampton, Victoria, Australia, 3188

[E] School of Agricultural and Resource Economics, University of Western Australia, 35 Stirling Highway, Crawley, WA, Australia 6009

[D]Dyson and Associates Pty Ltd, Arunga Drive, Strathfieldsaye, Victoria, 3554

Abstract

Dryland salinity threatens many high-value assets in Australia. Assessing the technical feasibility of intervention requires consideration of spatial and temporal dimensions of land management and associated surface-groundwater interactions. This paper presents results regarding the practicability of protecting assets in a 371,000-hectare agriculturally dominated catchment in south-eastern Australia using a physically-based catchment model, the Catchment Analysis Tool (CAT). The model links surface land management and hydrology at the paddock scale (daily time step growth and water use, crop rotations and management) with a fully distributed groundwater model enabling the assessment of agronomic versus engineering options for landscape asset protection. Scenarios assessing management options and costs of planting of perennial vegetation or engineering through groundwater pumping are outlined. The impacts of scenarios are reported under future climatic conditions as well as for the ‘steady state’ response. The study demonstrates a linked surface-groundwater modelling approach to estimate groundwater capture zones, response times to equilibrium under current and climate change conditions, as well as groundwater abstraction volumes required to protect landscape features. These factors are important considerations for evidence-based decision making about protecting assets from dryland salinity

Keywords: catchment modelling, CAT, groundwater capture zone, groundwater equilibrium

Introduction

Striking the balance between production and conservation goals in agricultural landscapes is a global challenge (Matson and Vitousek, 2006). Conservation in landscapes primarily used for agriculture is often costly and commonly the management changes required are beyond the resources of private farm owners. In addition, there are often long time lags to achieve positive ecological outcomes (House et al., 2008).

Dryland salinity is one of the major environmental problems faced in agricultural areas of Australia (Pannell, 2001). It threatens many biodiversity assets, particularly wetlands, riparian vegetation and rivers (Roberts and Pannell, 2009). As is the case with diffuse source nutrient pollution (Dowd et al., 2008), assessing the technical feasibility of dryland salinity management requires consideration of spatial and temporal dimensions. In particular, assessment of land management and groundwater interactions, responsiveness of aquifers and associated lag times are required (Beverly et al., 2005).

Barrett-Lennard et al. (2005) propose that, in the broad valley floors and sedimentary plains of Australia’s cropping regions, recharge is predominately 1-dimensional such that offsite recharge management may have little or no impact on protecting assets from high watertables. This outcome was based on a south-western Australian study where the groundwater gradient is negligible (of the order 2cm per km). In contrast, throughout much of south-eastern Australia significant areas of land salinity occurs at the interface between the Great Dividing Range and the Riverine Plains due to abrupt changes in groundwater gradients. Given the different groundwater processes that exist in the broad plains regions of south-western Australian, the effectiveness of offsite recharge management processes requires quantification in south-eastern Australia upland catchments.

A major national program for salinity management, the National Action Plan for Salinity and Water Quality, was completed in 2008, following expenditure of A$1.4 billion of public funding (and larger amounts of private funding contributed by landholders). An assessment of the program (Pannell and Roberts, 2010) concluded that the works undertaken due to the program will largely fail to deliver their intended outcomes ("prevent, stabilize and reverse trends in dryland salinity … improve water quality"). Reasons include: poor targeting of interventions; inadequate consideration of the practicability of management (including both technical and socio-economic considerations); and over-reliance on small, temporary incentive payments to landholders at a level insufficient to achieve land-use change at the scale required. As has also been suggested in the United States and Europe, there is major scope for improving the targeting (Dowd et al. 2008) and cost-effectiveness (Merckz et al. 2009; Wätzold and Schwerdtner 2005) of environmental programs. In Australia, increased cost-effectiveness can be achieved by stronger targeting based on threatened assets, based on appropriate consideration of technical and socio-economic issues (Roberts and Pannell 2009).

The Salinity Investment Framework (SIF3), described by Ridley and Pannell (2005), and implemented in north-central Victoria (Roberts and Pannell, 2009), provides a robust framework for making more evidence-based decisions about protecting high-value environmental assets under threat from dryland salinity. When the SIF3 study was conducted, it was acknowledged that more detailed feasibility studies based on catchment modelling (including land management-groundwater interactions) were needed.

In response to this need, this study was undertaken to assess the technical feasibility of protecting high-value assets threatened by dryland salinity using linked surface-groundwater catchment modelling. The available management options evaluated were planting of deep-rooted perennial plants or installation of groundwater extraction pumps. The modelling identified intensities of management required to maintain the groundwater at sufficient depth to minimise impacts on specific assets. It was also used to estimate response times and indicative costs of protecting each asset. Based on this information, the effectiveness of offsite recharge management processes in a south-eastern Australia upland catchment can also be assessed. Whilst this case study is specific to Australia, discussion about the large costs and trade-offs in achieving conservation outcomes in agricultural landscapes is relevant to many countries facing environmental problems resulting from agriculture.

Study area

The study area is the Avon-Richardson catchment in north central Victoria, Australia (Figure 1). The catchment is 371,000 ha in area, with over 85% currently under agricultural land use, dominantly annual plant species of pastures or winter crops, mostly cereals. There are also over 100 lakes and wetlands within the catchment (NCCMA, 2006). Average annual rainfall ranges from 350 to over 550 mm/year across the area.

Two primary ephemeral rivers drain the catchment, namely the Avon and Richardson Rivers. The Avon River has a mean annual flow of 117 GL/year and the Richardson River 189 GL/year, and flow into a terminal wetland, Lake Buloke. When full, the average depth of Lake Buloke is 4-5 metres and it only fills infrequently (approximately once per decade). Based on measured groundwater bore samples, the groundwater salinity of the upper aquifer is approximately 20,000 EC, or one third that of sea water. Under existing land uses, high groundwater recharge is resulting in discharge of this saline groundwater into waterways, affecting ecosystems and the aquatic environment in the lake.

The lower Avon-Richardson catchment is one of the most severely salt-affected parts of northern Victoria (Sinclair Knight Merz, 2002). Landholders began to express concern about salinity-related crop losses in the late 1960s. These concerns became widespread by the mid-1970s, with agricultural losses reported in the lower catchment area, including in the vicinity of Lake Buloke, particularly where large areas on the south-eastern side of the lake became saline. Poor water quality is considered to have a large impact on river and wetland health in the Avon-Richardson catchment (NCCMA, 2006).

Insert Figure 1 near here

Methodology

This study was divided into five steps as shown in Figure 2. High-value assets had previously been identified (Roberts and Pannell, 2009). The second step was the calibration of a linked surface water-groundwater model and the estimation of landuse change scenarios on the hydrological balance. Intervention or capture zones associated with each of the high-value assets were then calculated (step 3). Groundwater response times for varying levels of landuse change intervention through planting perennial vegetation or groundwater interception pumps were calculated (step 4). Finally preliminary costs of groundwater pumping and perennial vegetation were estimated (step 5).

Insert Figure 2 near here

Identification of high-value assets

The highest value environmental and infrastructure assets had been previously identified using SIF3 (Salinity Investment Framework version 3), as outlined by Roberts and Pannell (2009). These were (shown in Figure 1) Lake Buloke (nationally significant wetland), York Plains wetlands and river reach (considered to be regionally significant with very high community value), the township of Donald and Lake Batyo Catyo (both high economic and community value), Avon Plains Lakes and Cope Cope Lakes (all regionally significant), Box swamp and Jesse swamp (locally significant).

Modelling framework

The modelling approach used the Catchment Analysis Tool (CAT) framework (Beverly et al., 2005; DPI, 2009) to assess the on-site and off-site impacts of intervention strategies to protect assets. This model uses a combination of a suite of farming system models linked within a catchment framework with allowance for landscape connectivity and connection to a distributed, multi-layered groundwater model. The farm-scale models account for position in the landscape (topography, soil type, aspect and slope), climate, land use and land management and simulate water balance, nutrient transport and production on a daily time step. The CAT has also been developed to link with the fully distributed multi-layer groundwater model MODFLOW (McDonald and Harbaugh, 1988) to account for groundwater dynamics and provide a whole-of-catchment water balance whereby recharge estimates from the farming system models are explicitly incorporated into the groundwater model.

The CAT framework allows selection of farming systems models ranging in complexity from a simple crop factor approach to phenologically based crop, pasture and forest growth modules, as shown in Table 1. This framework provides the option of simulating the phenological development of a crop using either a user-defined crop-cover approach, a daily-heat-unit-accumulation approach or a vegetative-cycling approach depending on data availability and user capability. A common water balance, erosion, nutrient, carbon and soil evaporation module is used by all the available crop-growth models. The model estimates evaporation from soil and plants separately. Potential evapotranspiration can be calculated using either the Penman-Monteith (Monteith, 1965) or Priestley-Taylor (Priestley and Taylor, 1972) method depending on data availability. In this study the potential ET was computed using the Priestley-Taylor method based on observed daily air temperature, solar radiation and vapour pressure. Soil evaporation was estimated using the two-stage Ritchie approach (Ritchie, 1972) whereas potential soil water evaporation was estimated as a function of potential ET and leaf area index. In this study, a combination of models was used, specifically Williams et al. (1982) for crops, Johnson (2008) for pasture, Littleboy et al. (1992) for infrastructure and Feikema et al. (2007) for trees.

Insert Table 1 near here

Representation of lateral redistribution of surface water and sub-surface flows within the catchment was achieved through the development of a surface element network that disaggregated the catchment into a series of connected units, each unit representing the paddock/farm scale. Connection to adjacent up-slope and down-slope elements enabled the lateral redistribution of surface runoff and interflow (e.g. perched watertables) and facilitated the transport of water and nutrients from the top of the catchment to streams and ends of valleys. The partitioning of water excess into lateral flow components within any surface element was based on the approach of Rassam and Littleboy (2003).

Underlying the surface element network was a three-dimensional representation of the groundwater system modelled using MODFLOW (McDonald and Harbaugh, 1988). Deep drainage from each surface element was partitioned to lateral sub-surface flows and recharge, which was spatially assigned to the underlying unconfined aquifer. The groundwater model laterally redistributed water and simulated time varying groundwater discharge to stream and land surface. The effectiveness of engineering options, such as interception drains and groundwater pumping, could also be assessed. This approach overcame the need for the user to provide estimates of recharge *a priori* and enabled the simulation of the impact of rising water tables on plant performance, production and extent of waterlogging.

Input data description

The CAT model requires daily meteorological data including precipitation, maximum and minimum air temperature, solar radiation, evaporation and vapour pressure. Spatial data layers include a digital elevation model (DEM), land cover and soil type.

Meteorological daily data were obtained from the Bureau of Meteorology/Queensland Department of Natural Resources, Mines and Water SILO service (http://www.bom.gov.au/silo). Twenty-one climate stations exist within the catchment area, each with observed and interpolated long term data for the period 1900-2004. Climate station data combine original Bureau of Meteorology measurements for a given climate station with a process for infilling any gaps in the record using interpolation methods discussed in Jeffery et al. (2001). Because climate stations are located sparsely within the catchment, daily rainfall, temperature, evaporation and solar radiation data were scaled to each solution point within the catchment according to interpolated mean annual spatial layers created using the ANUClim software (Hutchinson 2001) which combines a DEM and temporal climatic data to generate a smoothed climate surface. As such, daily meteorological data assigned to each solution point within the catchment was a function of the twenty-one climate stations, landscape position and topography.

DEM data were generated using a 10-m grid created from the 1:25000 topographic map provided by the Victorian Department of Primary Industries (http://new.dpi.vic.gov.au/vro). Slope and aspect data layers were derived from the DEM using ArcGIS Spatial Analyst based on a triangulated irregular network (TIN) and were important in determining the partitioning of excess water into vertical and lateral components as well as the solar azimuth which was used to estimate the daily solar radiation.

Land-use data were classified using the Australia Land Use Mapping (ALUM) classification Version 6 (BRS 2006) (http://adl.brs.gov.au). The ALUM taxa describe land cover against which management strategies need to be specified. In this study, crop rotations and cropping history was incorporated into the management scripts used in the catchment modelling.

Soil data were derived by merging broad scale 1:250000 Land Classification Survey data and 1:25000 soil attribute coverage (Smith 2002). The merged spatial soil layer was attributed using the Factual Key of Northcote (1979) at the Principal Profile Form (PPF) level to classify different soil types. Within the study area nineteen different soil types were identified and spatially assigned. The soil attribution through depth for each soil type was based on published data sources (McKenzie et al., 2000), field observations and pseudo-transfer functions (van Genuchten et al., 1991) and includes soil water characteristics, bulk densities, hydraulic properties and impedance properties specified for each soil layer modelled. Additionally each soil required specification of erodability, erosion and soil evaporation attributes.

Groundwater conceptualisation and model attribution

The Avon-Richardson Catchment has four dominant surface geological formations (King, 1977) namely the Cambrian-Ordovician bedrock (570-250 million years ago), Devonian granite (420-360 million years ago), Tertiary sediments (67-2 million years ago) and Quaternary sediments (2 million years ago-current). . The region can be segregated into five broad hydrogeological units that govern the movement of water within the landscape based upon geology, hydrogeology and the existing knowledge of Groundwater Flow Systems (GFS) as reported by Coram et al., (2001).

In order to reduce the complexity of the groundwater model, a three-layer MODFLOW groundwater model was developed based on available hydrostratigraphic delineation (Figure 1). The uppermost unconfined layer represented the amalgamation of the Pliocene Sand Aquifer (Parilla Sand) and Quaternary Alluvium (Shepparton, Coonambigdal, etc.) aquifers due to their similar characteristics and scarcity of data defining the exact properties of each geological unit. Underlying this aquifer, the confined/unconfined deep lead (river gravel aquifer) layer represented the Tertiary Sands and Calivil Formation, which extends from the upper parts of the catchment to beyond the north of the catchment. The third layer represented the Palaeozoic basement geology, which underlies the entire model domain. This aquifer represents the basement aquifer of the region.

The groundwater model adopted a uniform grid of 100 m resolution and weekly time-steps. Surface hydrology and drainage features were incorporated into the model. The following assumptions have been made in developing the Avon-Richardson groundwater model:

- with the exception of the upper aquifer, all aquifers are considered vertically uniform with varying transmissivities;
- groundwater concentration gradients have negligible impact on head pressures;
- evaporation depths and rates were spatially varied based on land use based on differing rooting depths of species used in the farming-systems models associated with the overlying land-use;
- calibration bores used are representative of all modelled aquifers in the region;
- the digital elevation model (DEM) has adequate vertical resolution to allow reasonable calibration of the model and represents the land surface topology.

Model performance evaluation

The CAT model was calibrated and validated using observed stream gauge information, groundwater hydrograph data and discharge data. The coefficient of determination (CD) and the Nash-Sutcliffe index (NSI) (Nash and Sutcliffe, 1970) were used to evaluate model performance. Each coefficient is defined by:

$$CD = \frac{\sum_{i=1}^{n}\left(V_{obs,i} - \overline{V}_{obs}\right)^2}{\sum_{i=1}^{n}\left(V_{cal,i} - \overline{V}_{obs}\right)^2} \tag{1}$$

$$NSI = 1 - \frac{\sum_{i=1}^{n}\left(V_{obs,i} - V_{cal,i}\right)^2}{\sum_{i=1}^{n}\left(V_{obs,i} - \overline{V}_{obs}\right)^2} \tag{2}$$

where n is the number of observations, $V_{obs,i}$ is the observed value at time i and $V_{cal,i}$ is the simulated value at time i. The CD value is a measure of the relationship between observed and simulated values. In contrast the NSI value indicates how well the plot of the observed versus simulated values fit the 1:1 line. If the CD and NSI values approach zero (or are negative), the model performance is considered to be unacceptable or poor. If the values are identical to one, the model prediction is considered perfect (MDBC, 2000). A NSI of 0.6 is viewed as "satisfactory" whereas a NSI of 0.8 or higher is considered "good" (Chiew et al., 1993).

Three stream gauges within the catchment were used to evaluate the surface hydrologic estimates derived using CAT. The period of record ranged from May 1969 to June 2009 with only 1974 to 1989 being concurrent amongst all gauges (Table 2). Monthly estimates of quick flow derived using CAT were compared to monthly stream gauge observations for each gauge period of record. Infilled monthly streamflow data were derived using a flow-stratified sampling analysis method. The streamflow calibration period was 1974 to 1989 with the validation period being all remaining measured streamflow data outside the calibration period.

The groundwater model calibration criteria were based on matching representative groundwater bore hydrograph levels, mapped discharge extent, depth-to-watertable estimates and regional baseflow volumes. Of the 412 groundwater observation bores within the catchment, 135 were selected based on the duration and frequency of monitoring, screen depth and location within the catchment. Of the 135 selected calibration bores, 66 were within the shallow aquifer, 24 in the deep lead and 45 in the basement aquifer.

In addition to comparing predicted catchment responses to observation stream gauge, groundwater hydrograph and discharge data, a comprehensive analysis was undertaken on the developed groundwater model to assess the sensitivity of key modelled outputs to variations in input data. Modelled outputs considered included baseflow volumes, saturated area and groundwater discharges (evapotranspiration, boundary fluxes and aquifer interflows). The sensitivity analysis procedure involved altering by three orders of magnitude the calibrated input data sets including lateral hydraulic conductivities, storativity, specific yield, vertical conductivities, recharge, river and boundary conductance and recording the groundwater response relative to the calibrated condition. Each input data set was systematically modified whilst maintaining all other input data. In total, approximately 120 simulations were undertaken to test the robustness of the groundwater model and the uniqueness of the combination of input data required to meet the calibration criteria.

Extent of intervention needed for planting perennial vegetation

The response of the shallow watertable beneath each asset due to planting native perennial vegetation within varying extents of the groundwater capture zone of each asset was determined in the following way:

a) Identify the groundwater capture zone influencing each high-value asset by calculating the inflows to each asset, based on the groundwater gradients derived using the calibrated groundwater model under current conditions.
b) Order all cells within the capture zone based on their proximity to the asset boundary. That is, the landuse was altered in bands radiating outward from the asset boundary resulting in the plains being revegetated first and the uplands revegetated last.
c) Systematically replace the recharge estimates under current landuse with native vegetation estimates in blocks defined by the user. In this case blocks of 1000 ha were incrementally replaced until the entire capture zone was modified. The groundwater model was re-run to derive the equilibrium shallow watertable response from which a response curve was generated.

The relationship between the shallow watertable beneath each asset and area of revegetation in the associated capture zone gives an indication of the level of groundwater connection to each asset. This information is useful in prioritising intervention strategies.

To evaluate the effectiveness of recharge management in the valley floors as proposed by Barrett-Lennard et al. (2005) and as adopted above, a comparative simulation was undertaken whereby step (b) was replaced by ordering the landuse change within the capture zone based on the magnitude of groundwater flux across the groundwater cell. An alternative approach also evaluated was to base the ranking on the expected recharge change under the modified landuse. However, results indicted that these alternative approaches generated equivalent response trajectories as each other due to the groundwater gradient being greatest in regions of high recharge. As such the comparative analysis will focus on results derived by ordering the landuse change within the capture zone based on the magnitude of groundwater flux across the groundwater cell.

Response time under future climate regime

The reduction in future groundwater connection to each asset was assessed based on the application of Intergovernmental Panel on Climate Change (IPCC, 2001) climate projections. To estimate future climate change, the IPCC (SRES, 2000) prepared 40 greenhouse gas and sulphate aerosol emission scenarios for the 21st century that combine a variety of assumptions about demographic, economic and technologic driving forces likely to influence such emissions in the future. In this analysis, the wetter low climate scenario (B2) of the IPCC (SRES, 2000) was generated using CSIRO's global atmosphere models (McGregor and Dix, 2001; Hennessy et al., 2006) integrated with annual global warming values. This scenario was chosen to estimate the most conservative condition in terms of greatest recharge and greatest likelihood of shallow watertable development beneath each asset. These monthly adjusted data were manipulated to derive future daily climate sequences as reported by Anwar et al. (2007) and Christy et al. (2009). The extended daily climate sequences for each climate station within the study area were used to derive recharge estimates for the period 1985-2050 under current land use and native vegetation conditions. The period 1982-1992 was used as the de-trended data sequence upon which the future predictions were based, as this period reported higher average annual rainfall (459 mm/yr average for all climate stations in the catchment) relative to the 1957-2006 average of 449 mm/yr. The resultant average annual rainfall for the period 2006-2050 was 429 mm/yr, which is marginally drier than the 1957-2006 historical climate sequence. Based on these recharge estimates, the transient groundwater model was used under various revegetation strategies for each asset to determine the frequency of shallow watertable development under future climate conditions.

Extraction volumes – area and drawdown

The calibrated groundwater model was used to estimate the daily extraction volume required to maintain the watertable beneath each asset to a depth of 1.5 metre and 2 metres under equilibrium conditions. These depths were chosen to firstly illustrate the non-linearity in extraction volumes required to modify watertable beneath each asset and secondly to conform to the widely held assumption that a depth of less than 2 metres causes land salinisation. The approach used to estimate extraction volume was to represent the extent of each asset as a surface drain within the groundwater model with a drain depth set at surface elevation less the required depth-to-watertable (in this case either 1.5 or 2 metres). Using this approach, the inflow volume to each drain within the asset represented the minimum extraction volume required to maintain the watertable beneath each asset to a nominated depth.

Given that the area of each asset varies significantly, an area-weighted extraction volume was used to enable a relative comparison between assets. Additionally, an extraction ratio defined as the ratio of the extraction volume required to maintain a 1.5 metre depth-to-watertable to the extraction volume required to maintain a 2 metre depth-to-watertable was used to illustrate the non-linearity of the required extraction volume with depth. The non-linearity with depth is a function of the aquifer and regolith characteristics, the groundwater gradients and connection with lower aquifers and position in landscape.

Indicative costs of asset protection

The costs of protecting assets were based on estimated establishment costs and recurring annual costs. These estimates are simplifications as they do not reflect the complexity of local practices and farming-systems adaptations. They do, however, give a reasonable indicative figure of the costs to protect environmental assets from dryland salinisation.

For revegetation, it was assumed that either lucerne (alfalfa – *Medicago sativa*) or native vegetation for biodiversity were used, requiring the model to be run under each vegetation type in addition to the current-practice simulation. Lucerne establishment was assumed to cost $250/ha (all costs are in Australian dollars) and native revegetation (primarily Australian native trees) as $650/ha (direct seeding), with this cost applied over the nominated capture zone, excluding the area of the associated existing wetland. The opportunity costs of changing land use were assumed to be $50/ha/yr for lucerne production and $200/ha/yr for native vegetation. The opportunity cost for lucerne is lower because it generates some income from livestock grazing (albeit at a lower level than the agricultural land-uses it replaces) whereas native trees were assumed to generate no income. Converting opportunity costs to present value figures over 20 years (5% real discount rate) resulted in figures for opportunity costs of $623/ha for lucerne and $2,492/ha for trees. For engineering, establishment costs used were installation of pumps (assumed as $10,000 for up to 3ML/day; $20,000 for 3-5; $50,000 for 5-15; $100,000 for 15-30) plus the construction of a lined evaporation basin ($100,000/ha for 7ML/ha/year disposal). Annual pumping costs were assumed as $10,000/year/ML plus salt disposal costs of $30/t. The assumed salinity concentrations ranged from 26,000 mg/L for Donald, Buloke, Wooroonook and Chirrup; 12,000 mg/L for York Plains, Avon Plains, Batyo Catyo; 10,000 mg/L for Cope Cope, Box and Jesse swamps based on sampled groundwater chemistry. Establishment and annual costs for vegetation and engineering, plus the annual opportunity costs associated with each option, were also converted to present value using a 5% real discount rate over 20 years. The political acceptability of disposal options was not considered.

Results

Model calibration

The correlation between observed and simulated 1995 potentiometric head for observation bores in each modelled aquifer is shown in Figure 3. The root mean square errors between observed and simulated potentiometric head at representative observation bores were 0.96, 0.94 and 0.97 for modelled aquifer layers 1, 2 and 3 respectively. The variation in accuracy is a function of the number of data points within each layer and the assignment of bores to modelled aquifers. The calibrated model had a scaled root-mean-square error of 3.85% and an absolute residual mean of 3.50 metres which was considered acceptable with respect to the scale, the accuracy of the digital elevation data and the hydrogeological conceptualisation assumptions of the catchment.

Insert Figure 3 near here

Table 2 summarises the simulated versus observed monthly streamflow statistics for the calibration and validation periods for each stream gauge within the study catchment. The best fit for monthly stream flow during calibration was R^2=0.85 and *NSI*=0.82 with an average across all gauges of R^2=0.80 and *NSI*=0.78. During validation the average across all gauges was R^2=0.71 and *NSI*=0.68.

Figure 4 shows the simulated 1995 depth-to-watertable. Also shown are mapped discharge sites, which indicate good alignment with the model predictions. Of all groundwater cells within the mapped discharge zones, 79% predict depth-to-watertable of 4 metres or less, which is within the resolution of the digital elevation data. Results generally showed that the greatest residual errors occur in locations where model layer three is unconfined.

Insert Table 2

Insert Figure 4 – Simulated depth-to-watertable

Sensitivity analysis

A sensitivity analysis was performed to assess the response of the groundwater model simulation to changes in various input parameter values. Although 120 simulations were examined in detail, only the sensitivity of the predicted catchment depth-to-watertable to variations in aquifer hydraulic conductivities is presented in Figure 5. This figure shows a typical output assessing the effect of varying lateral conductivities of each aquifer on the shallow watertable extent. The trajectory of each response curve reflects the stability of the model. The flatter the curve, the less sensitive the output to the associated model attribute. Figure 5 shows that layer 2 lateral conductivity has negligible impact on simulated watertable depth. This is consistent with the conceptualisation as layer 2 represented the deep lead which has limited outcropping, thereby having limited direct connection to an upper aquifer (and by inference to the shallow watertable). Results show model parameters generally fit within type I or II classification as described by MDBC (2000).

Insert Figure 5: Sensitivity analysis results

Intervention zones and groundwater response times around key assets

Figure 6 shows the extent of groundwater capture zones that affect the eight high-value assets. The capture zones incorporate all groundwater up-gradient of the asset as calculated using the calibrated groundwater model assuming current landuse and land management conditions.

The areas of the eight high-value assets are shown in Table 3. They range in size from approximately 843 ha (Box Swamp) to over 23,000 ha (Lake Buloke). The capture zones areas under current landuse are estimated to range from 46,805 to 230,677 ha (Table 3) depending upon landscape position, which influences both the groundwater flow path length and the degree of connection to drainage features (such as rivers). Internal zones of no-influence (shown in Figure 6 as holes within the capture zone extent) are due to local groundwater depressions or negligible groundwater gradient. The groundwater response time and dominant aquifer in connection with each asset is also shown in Table 3.

Insert Table 3 near here

Insert Figure 6 near here

Insert Figure 7near here

Within each capture zone, the prioritisation of which zones to revegetate in preference to others is based on the proximity to the asset boundary. Figure 6 shows the ranking in order of intervention whereby a value of one (blue) indicates the highest priority and should be the first cell to be revegetated based on the magnitude of the distance to the asset boundary. Conversely those cells assigned a high index are considered lower priority (red) and represent zones with the greatest distance from the asset. Figure 7 shows the comparative intervention strategy based on the magnitude of groundwater flux across a cell. In this alternative case, a cell with a ranking of 1000 (based on a groundwater gradient of 2) will be revegetated before a cell ranked 2000 (with a groundwater gradient of 0.5). In broad terms this ranking scheme is shown to prioritise the upland catchment in which the groundwater potentiometric gradient is greatest, although zones of equivalent gradient are also identified elsewhere within the capture zone. Both ranking schemes are used in this study as methods to spatially assign revegetation blocks to a specified area for intervention.

The change in the median depth-to-watertable under equilibrium or steady-state conditions beneath each asset as a function of re-establishment/replanting of native vegetation within the capture zone of the asset is shown in Figures 8a and 8b (Figure 8b is an expanded plot of Figure 8a). The order of landuse change within the capture zone was based on the proximity of a cell to the asset boundary such that targeted locations within the catchment were selected. Results suggest that the median watertable depth for most assets can be lowered by 2 metres or greater through the reintroduction of native vegetation. The exceptions are Lake Batyo Catyo, Lake Buloke, Cope Cope Lakes and Jesse Swamp. The impact of increasing areas of intervention is also shown to vary significantly between assets with some assets having a rapid response (for example, Box Swamp, Avon Plains and York Plains lakes having 2 metre groundwater falls after 6,000, 7,000 and 12,000 ha intervention respectively) in contrast to a more gradual monotonic increase (such as Donald which reached the 2 metre watertable depth reduction after 132,000 ha intervention). Results show significant variation in the area of intervention required to cause a similar change in median depth-to-watertable beneath assets. To reduce the median depth-to-watertable by 1 metre beneath an asset, the required capture area to be restored to native vegetation is 97% for Lake Buloke, 97% for Jesse Swamp, 10% for Cope Cope Lakes, 5% for Lake Batyo Catyo, 3% for Donald, 2% for York Plains and Box Swamp and 1% for Avon Plains Lakes.

Insert Figure 8a and 8b (trajectories)

Figure 9 shows the change in shallow watertable area (saturated area) within the catchment as a function of re-establishing native vegetation within the capture zones of each asset. For example, results show that a 200,000 ha revegetation strategy in the Donald capture zone reduces the saturated area to 15,306 ha (or 18% of the current shallow watertable area). Alternatively, revegetation of the entire Box Swamp capture zone reduces the saturated area to 77,636 ha (or 89% of the current shallow watertable area). This response is a function of landscape position as evidenced by the impact on the catchment shallow watertable area due to restoration of native vegetation required to reduce the median depth-to-watertable by 1 metre beneath each asset (Figure 8b).

Information regarding the median depth-to-watertable impacts beneath each asset due to the restoration of native vegetation is shown in Figure 8. This information can be combined with the associated off-site saturated area impacts shown in Figure 9 to estimate the change in saturated area within the catchment arising from intervention aimed at reducing the depth-to-watertable beneath each asset. For example, a 1-metre reduction in median depth-to-watertable under Lake Batyo Catyo results in a catchment shallow watertable area reduction of 6,338 ha (a reduction by 7% of current shallow watertable area). A similar 1-metre reduction in median depth-to-watertable beneath Lake Buloke is estimated to reduce shallow watertable area within the catchment by 72,051 ha (by 83% of current shallow watertable area); 73,687 ha (85% of current shallow watertable area) for Jesse Swamp; 76,323 ha (88% of current shallow watertable area) for Avon Plains Lakes; 79,133 ha (91% of current shallow watertable area) for Cope Cope; 81,392 ha (93% of current shallow watertable area) for Donald; 85,255 ha (98% of current shallow watertable area) for York Plains and 86,178 ha (99% of current shallow watertable area) for Box Swamp.

Insert Figure 9 (change in saturated area)

Extraction volumes and area of groundwater drawdown by pumping to protect individual high-value assets

Extraction volumes required to maintain a watertable depth to either 1.5 or 2 metres beneath each asset are reported in Table 4. The extraction volumes reflect the size of the asset and the watertable depth before pumping. The highest extraction volume was 7,185 ML/year to protect Lake Buloke to a depth-to-groundwater of 2 metres, it being the largest asset and already affected by salinity. Alternatively, the smallest extraction volume required to maintain a depth-to-groundwater of 2 metres was 657 ML/year for Box Swamp (small asset). By comparison, the extraction volumes required to maintain a depth-to-groundwater of 1.5 metres for Lake Buloke and Box Swamp are 4,932 and 465 ML/year respectively, corresponding to a reduction of 31% and 29% relative to the corresponding 2 metre depth-to-groundwater target. On a weighted-area basis, the York Plains was estimated to require 47% less extraction volume per ha of capture zone than Avon Plains lakes to maintain the depth-to-watertable at 1.5 metres.

Also reported in Table 4 are the associated area-weighted extraction volumes required to maintain a 2 metre depth-to-watertable and the ratio of these volumes. For instance, the Lake Batyo Catyo asset requires a 100% increase in extraction volume to lower the watertable from 1.5 to 2 metres, whereas the Jesse Swamp requires only a 30% increase in extraction volume.

Insert Table 4: Groundwater extraction volumes

Area of intervention to maintain the watertable depth at 1.5 and 2.0 metres

The areas of intervention required to maintain the watertable depth at 1.5 metres and 2.0 metres beneath each asset are summarised in Tables 5 and 6 respectively. The columns indicate different percentiles of asset areas satisfying the target groundwater depth. Each table reports the area of native vegetation required to meet the target watertable depth in zones either (a) ordered based on the proximity to the asset boundary (top section of table) or (b) ordered based on groundwater flux (bottom section of table). Intervention adjacent to each asset (referred to as the proximity approach and shown in Figure 6) represents a discharge mitigation strategy whereas intervention targeted on groundwater flux (Figure 7) broadly represents a recharge mitigation strategy within the study catchment. The shaded entries (referenced by the letter E) in Tables 5 and 6 indicate that the target watertable depth cannot be achieved by revegetation in these cases and would require the installation of a groundwater interception scheme whereas the zero entries imply that the target watertable depth requires no intervention. In the case of York Plains, results suggest that 381 ha of revegetation is required to maintain the watertable depth at 1.5m across 90% of the asset, assuming revegetation is adjacent to the asset boundary. In contrast 55,367 ha of revegetation is required in upland areas to meet the same target. Were the target watertable depth to be increased to 2 metres then the required area of revegetation adjacent to the asset boundary would increase to 3,467 ha (approximately a nine-fold increase) or 102,650 ha in upland areas. It is noteworthy that, in all cases, adoption of the discharge strategy requires substantially less area of intervention to meet a specific asset protection target than the corresponding recharge strategy.

Insert Table 5: Area of intervention required to maintain the watertable at 1.5 m by percentile

Insert Table 6: Area of intervention required to maintain the watertable at 2.0 m by percentile

Response times under future climate regime

The percentage change in the number of times the median shallow watertable depth beneath each asset was estimated to be less than 2 metres under the future climate scenario relative to the current landuse results is shown in the fourth column of Table 7. These results assume 1,000 ha revegetation to native species. The larger the percentage change the less often the asset is in connection with the shallow groundwater. This information is useful to identify those assets which are likely not to be impacted by shallow watertables under a future climate regime and therefore may not require any intervention. The range of change from current condition in the number of times the median shallow watertable depth beneath each asset was estimated to be less than 2 metres varies from approximately 1.6% to 4.9% depending upon landscape position and size of asset.

A typical response is shown in Figure 10 which shows the median depth-to-watertable less than 0.5 metre for York Plains under various revegetation options for the period 1985-2050. The 0.5 metre depth-to-watertable threshold was selected as this depth impacts on the root zone of most vegetation and therefore was considered a minimum watertable control target. Results suggest that 16% of the time the depth-to-watertable is greater than 0.5 metre (that is, disconnected from the asset) under current practice. This increases to 52% of the time under the 1,000 ha revegetation scenario and 72% of the time under the 5,000 ha revegetation scenario. Similar trends are estimated for the other assets.

Insert Figure 10 (York Plains response)

Figure 11 shows the percentage reduction (relative to current practice) in the proportion of time that median depth-to-watertable is less than 2 metres beneath each asset for various areas of intervention. This reduction is calculated based on the average median depth-to-watertable less than 2 metres for the simulation period 1985-2050. Results show the effectiveness of increasing areas of native revegetation in reducing the occurrence of shallow watertables (within 2 metres of the land surface). They indicate that 5,000 ha intervention will reduce the frequency of shallow watertables beneath most assets by between 1.8 to 5.3 percent only (with the exception of Donald at 28%) as reported in the fourth column of Table 8.

Insert Figure 11 (% reduction from current practice)

Costs of asset protection

The cost of asset protection (present value totalled over 20 years) required to maintain a depth-to-watertable of 1.5 metres below each asset is summarised in Table 7. The areas of revegetation intervention are based on maintaining 90 percent of the asset area to a watertable depth of 1.5 metres or greater. In all cases where discharge mitigation was feasible as indicated in Table 5, with the exception of Donald, lucerne was the most cost-effective option. Native vegetation was always 2-4 times more expensive than lucerne, due both to it being more expensive to establish, and it having larger ongoing opportunity costs. Engineering options were very expensive relative to the planting of perennials in most cases. For example, to protect the York Plains from salinity, the cost of engineering works (present value over 20 years) was estimated to be $131 million, compared with $0.5 million for lucerne and $1 million for native vegetation. Only Box Swamp had engineering costs of less than $100 million. Those with intermediate engineering costs ($100-160 million) were Lake Batyo Catyo, Avon Plains Lakes, York Plains and Donald. Cope Cope Lakes, Jesse Swamp and Lake Buloke were estimated to have costs over $200 million (Table 7).

Comparative costs of asset protection required to maintain a depth-to-watertable of 0.8 metres below each asset is summarised in Table 8. As above, the areas of revegetation intervention are based on maintaining 90 percent of the asset area to a watertable depth of 0.8 metres or greater. In all cases the establishment of lucerne was the most cost effective option. In this case, to maintain a 0.8 metre depth-to-watertable beneath 90 percent of the asset area the native vegetation option was between 2-40 times more expensive than lucerne, whereas engineering was 18-18,000 times more expensive than lucerne.

Insert Table 7: Costs of asset protection to 1.5 metres

Insert Table 8: Costs of asset protection to 0.8 metres

Effectiveness of recharge management

The change in the median depth-to-watertable under equilibrium or steady-state conditions beneath each asset as a function of re-establishment/replanting of native vegetation within the capture zone of the asset is shown in Figures 12a and 12b (Figure 12b is an expanded plot of Figure 12a). Figure 12 adopts an ordering of landuse change within the capture zone based on the magnitude of groundwater flux such that landuse was altered in zones of highest groundwater gradient resulting in the uplands being revegetated first and the plains revegetated last. This approach approximates a recharge intervention strategy as the upland regions in the study catchment are subject to higher recharge. These results are in contrast to Figure 8 which adopts an ordering of landuse change within the capture zone based on the distance from the asset boundary and consequently adopts a groundwater discharge intervention strategy. The impact of increasing areas of intervention is shown to vary between assets. However, unlike Figure 8, the response trajectories are less steep until a significant percentage of the capture zone has been modified – that is, the extent of altered landuse encroaches the discharge zones. These results indicate that it is not until the groundwater gradient in the upland region of the catchment has been significantly impacted by land-use change that changes in mean watertable elevation beneath each asset are measurable.

Insert Figure 12 (trajectories) near here

Discussion

Responsiveness of catchment to intervention

Examination of the future climate scenario responses highlight that the equilibrium steady-state responses shown in Figure 8 are believed to overestimate the impact on the median depth-to-watertable beneath each asset for the period 1985-2050. This is due to the episodic recharge and seasonal influences not accounted for under steady-state assumptions. However the steady-state response trajectories shown in Figure 8 delineate the equilibrium response between each asset and provide valuable information about the relative areas of intervention required to reduce the long-term threat of groundwater.

This study has also demonstrated the impact of the spatial patterning of intervention. Barrett-Lennard et al. (2005) proposed discharge management in close proximity to the asset (predominantly in valley floors of low surface relief) as a relatively effective strategy to ameliorate the impact of shallow groundwater on surface features. When this approach was applied to this study, the impact was not as pronounced as reported by Barrett-Lennard et al. (2005) but nonetheless was shown to be much more effective and much cheaper than upland recharge mitigation. Whereas the groundwater gradient in the south-western Australian environment in which the Barrett-Lennard et al. (2005) study focused is small, the groundwater gradient in this study is pronounced and characterised by break-of-slope processes where uplands connect with lowland plains. Nevertheless, this study confirmed the Barrett-Lennard et al. (2005) proposition and identified that the most effective intervention spatial patterning was based on proximity to the asset (Figure 6) rather than targeting zones of highest groundwater flux (Figure 7). The superiority of establishing perennial vegetation close to the assets rather than in upland recharge areas was evidenced by the steeper trajectories shown in Figure 8 compared to Figure 12. Only when the upland recharge intervention areas shown in Figure 12 extended beyond the break of slope and impact on the discharge zones did the medium watertable elevation beneath each asset start to fall.

Response times of planting deep rooted perennials are estimated to range from less than 1 year to 94 years, which is consistent with that estimated by groundwater flow system mapping (Coram et al., 2001). The presence of groundwater divides and low inflows within the relatively closed catchment results in short groundwater response times and makes solving the problem more tractable, even within regions that are connected with regional aquifers considered to have response times ranging from 50-150 years (Gilfedder et al., 2003). It is noteworthy that the longest groundwater response time was Box Swamp, the only asset wholly in direct connection with the bedrock aquifer (Figure 1). This longer response time is believed to be associated with the large contributing upper-catchment area and the steeper gradient of the bedrock aquifer potentiometric surface relative to the overlying aquifers resulting in the longer groundwater flow response time. The variability in response time for assets in connection with the alluvial aquifer is caused mainly by the relative size of intervention, aquifer transmissivity and upslope groundwater gradients. The response time of any intervention to protect assets is an important consideration influencing the cost-effectiveness of intervention (Graham et al., 2010) and varies depending upon landscape position.

Based on a notional index calculated as the multiplication of the normalised response times, normalised steady-state trajectories to reduce watertable depth by 1.5 metres and the normalised percentage change in groundwater connection assuming 1,000 ha native revegetation under future climate conditions, the ranking of interventions in the catchment based solely on groundwater responsiveness to management is as follows: Cope Cope (most responsive), Avon Plains Lakes, York Plains, Jesse Swamp, Lake Batyo Catyo, Donald and

Lake Buloke (least responsive). Of course, priorities for investment also depend on cost and asset values (see below).

These results highlight the value of adopting a multiple aquifer conceptualisation as opposed to relying on groundwater response times based on Groundwater Flow System (GFS) mapping alone. Previous studies (Beverly et al, 2005; Beverly and Hocking, 2009) have shown that the broad scale GFS mapping poorly describes complex aquifer interactions and response times. While long groundwater response times may negatively influence investment, the detailed groundwater modelling provides more specific information and understanding which may influence prioritisation in the catchment.

Extraction volumes

The groundwater extraction volumes reported in Table 4 are consistent with potential well yields in the area. Field observations have shown that the installation of groundwater pumps in the lower part of the catchment (for example adjacent to Donald) will lower the watertable to a new equilibrium within approximately one year.

Feasibility of protecting high value assets

It is clear that the cost of protecting assets from dryland salinity varies enormously from asset to asset. In some cases (for example Donald or Cope Cope Lakes) it is very expensive, due to the large areas of intervention for perennial establishment or large costs associated with engineering. In other cases (for example York Plains or Box Swamp) it can be relatively cheap. The least-cost options for the eight assets in the catchment to maintain a watertable depth of 1.5 metres across 90% of each asset were estimated to cost $1,612 million (total for all eight assets, present value over 20 years). This is similar in magnitude to the costs of protecting 30 of the most highly-valued biodiversity assets in Western Australia from dryland salinity ($854 million, Sparks et al., 2006).

However, results presented in this study clearly show that the least-cost option can significantly vary depending upon the target watertable depth beneath each asset and the degree of asset area to be protected. For instance, the least-cost options for the eight assets in the catchment to maintain a watertable depth of 0.8 metres across 90% of each asset was estimated to cost only $12 million (total for all eight assets, present value over 20 years). This variation between the costs to maintain a watertable depth of 0.8 and 1.5 metres across 90% of each asset is largely due to the transition from agronomic to engineering solutions, which are much more expensive in most cases, but which can meet more stringent groundwater targets than perennial vegetation is capable of in some cases.

It is noteworthy that results from this study are based on a wet future climate scenario (worst case scenario from a salinity cost perspective). If this scenario is inaccurate, the cost of intervention could be substantially less than suggested in this paper. Even so, costs of salinity mitigation under a drier climate regime are likely to remain far larger than budgets allocated to the problem by governments in Australia (Pannell and Roberts, 2010).

It is also important to note that the pumping assessments have high uncertainty. Further information about the effectiveness of engineering through pump testing and community reaction to salt disposal would need to be ascertained prior to deciding to use engineering to protect assets. Furthermore, social and political considerations are likely to be impediments to use of engineering, especially for the installation of 143 ha of evaporation basins estimated as needed to protect York Plains to dispose of the salt. Also of major concern are the ongoing maintenance and operation costs associated with pumping and disposal of the extracted saline groundwater.

Native revegetation, although by far the more expensive agronomic option, has low ongoing costs compared with lucerne (which will need sowing every 10 years or so). It also has potentially large unquantified biodiversity, amenity and carbon sequestration benefits. In the case of protecting very high-value assets such as the York Plains, these are important considerations.

Local investigations to assess watertable depths and groundwater responsiveness would be useful to increase the confidence in the modelling results. Trade-offs between available budget, response time, ongoing costs, biodiversity benefits, community preparedness for environmental stewardship and community resistance to groundwater disposal options need to be considered.

The most highly valued assets in the catchment are the York Plains, Avon Plains Lakes, Cope Cope Lakes, Lake Buloke and the township of Donald. Of these, the York Plains is the least costly to protect. The Avon Plains Lakes, Cope Cope Lakes and Donald were intermediate in cost, with Lake Buloke being very costly. Box Swamp is not costly to protect, but is of only local significance. Jesse Swamp is both expensive to protect and of only local significance. Table 8 includes a suggested ranking of investments, considering both asset values and intervention costs.

This study has demonstrated that the inclusion of a modelling analysis, including a biophysical catchment model (CAT) and a distributed groundwater model (MODFLOW), strengthens the capacity of an investment framework to identify worthwhile projects. Previously, analysis based on SIF3 recommended that engineering should be investigated for Donald (Roberts and Pannell 2009). It also suggested that York Plains and Avon Plains Lakes might be too large in area for engineering to be economic. In light of the intervention area required, this study has identified that lucerne is a feasible option for York Plains whereas engineering, while expensive, is assessed as the only feasible option for the Avon Plains Lakes, which has demonstrated the usefulness of detailed biophysical modelling.

Confidence in the modelling results

The groundwater conceptualisation was based on the work of Coram et al. (2001) and subsequent work by Hocking (2007) and Dyson (2008). There is still some uncertainty in the location of the regional deep lead aquifer north of Cope Cope which challenges the veracity of the extraction volumes required to protect assets in the lower catchment (Lake Buloke, Donald and Cope Cope). Notwithstanding this uncertainty, the results appear realistic and consistent with existing knowledge. Simulated versus observed bore hydrograph comparisons are reasonable given the resolution of the digital elevation data. That is, an inherent elevation error of approximately 2 metres potentially exists between solution points as a function of data sampling resolution. Modelled response times also appear reasonable, being within the limits of expected response for the identified groundwater flow systems (0-100 years) as reported by Coram et al. (2001).

This work has collated and interrogated spatial and temporal data sets and has conceptualised and modelled the groundwater system in a more comprehensive and integrated way than previously. Results presented identify that estimates of groundwater capture zone extents, response times under equilibrium and climate change conditions, and groundwater interception volumes required to protect assets are important to develop improved protection strategies and make more cost-effective investment decisions than currently.

The next steps needed to engender confidence in modelled results are investment in field programmes to ground-truth modelled predictions. It is recommended that field research should be undertaken for one of the assets that appear to be the best prospects for investment, including pre-treatment pump testing and the establishment of appropriate monitoring networks to evaluate the robustness of modelled predictions.

Conclusion

The CAT modelling has identified and differentiated the impacts of intervention options aimed at reducing the threat from rising groundwater and dryland salinity on key environmental assets in Victoria, Australia. The linked surface-groundwater modelling approach can estimate groundwater capture zones, response times under equilibrium and climate change conditions and groundwater interception volumes required to protect landscape features. Results presented show that the appropriate intervention strategy and associated costs are highly dependent upon the target watertable depth beneath each asset and the degree of asset area to be protected. Such information is necessary to inform more rigorous economic analysis and provide a basis for evidence-based decision making about protecting assets from dryland salinity.

The study shows that, in this Victorian catchment, establishing perennial vegetation in regions adjacent to groundwater discharge zones is a far more effective strategy for dryland salinity mitigation than establishing perennial vegetation on upland recharge areas. This is consistent with earlier findings from Western Australia, despite large differences in physical circumstances. The scale of perennial vegetation required to protect particular assets varies enormously. In some cases, perennials can provide salinity mitigation relatively cheaply, while in others, the areas required are so large that expensive engineering responses are likely to be more cost-effective.

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Table 1: Available crop models by crop class in CAT

	Vegetation model class			
Pasture growth models	Fixed crop factor cover	Heat unit	Phenological	Composite
	Littleboy et al., 1992	Williams et al., 1982, Neitsch et al., 2001; Ritchie, 1985	Moore et al. 1997, Freer et al. 1997	Johnson et al. ,2003
			Southwell, 2009	Johnson, 2008
				Jones et al., 2006
Crop growth models	Littleboy et al., 1992	Williams et al., 1982, Neitsch et al., 2001; Ritchie, 1985	Dynamic wheat model (Jones and Kiniry, 1986; Littleboy et al., 1992)	
			Dynamic sunflower model (Ritchie, 1985)	
Tree growth models	Littleboy et al., 1992	Williams et al., 1982, Neitsch et al., 2001; Ritchie, 1985	Feikema et. al., 2007; Landsberg and Waring, 1997	

Table 2: Simulated versus observed monthly statistics for sub-basin calibration and validation

Basin	Gauge	Measurement dates	Calibration period		Validation period			
			1974-1987		1988-1995		1995-2005	
			CD[A]	NSI[B]	CD	NSI	CD	NSI
Wimmera Hwy	415220	Aug 1974 – Mar 2009	0.85	0.82	0.76	0.76	0.74	0.72
Carrs Plains	415226	Apr-1971 – May 2009	0.71	0.72	0.66	0.58	0.68	0.67
Beazleys Bridge	415224	May 1969 – Oct 1995	0.84	0.81	0.72	0.67	nr	nr

[A] Coefficient of determination

[B] Nash-Sutcliffe Index

nr = no records

Table 3: High value asset value, response time and dominant aquifer.

ID	Location	Area of asset (ha)	Value of asset as defined using SIF3	Intervention area under current conditions (ha)	Response time (years)	Dominant aquifer in connection with asset
1	York Plains	3,868	Very high	110,436	50	Quaternary
2	Avon Plains Lakes	2,069	Very high	126,472	40	Quaternary
3	Lake Batyo Catyo	2,946	High	181,892	55	Quaternary
4	Lake Buloke	23,048	Very high	230,677	67	Quaternary
5	Donald	1,263	Very high	219,902	< 1	Quaternary
6	Cope Cope Lakes	4,274	Very high	98,877	40	Quaternary
7	Box Swamp	843	Moderate	46,805	94	Palaeozoic
8	Jesse Swamp	1,456	Moderate	61,235	43	Tertiary

Table 4: Groundwater volumes required to be extracted to maintain various depth-to-groundwater (dwt) beneath each asset.

ID	Location	Asset area	Extraction volume		Area-weighted extraction volumes		Response Ratio
			dwt=1.5 metre	dwt=2.0 metre	dwt=1.5 metre	dwt=2.0 metre	
		ha	ML/yr	ML/yr	ML/yr/100ha	ML/yr/100ha	
1	York Plains	3,868	984	1,845	25.4	47.7	1.9
2	Avon Plains Lakes	2,069	983	1,452	47.5	70.2	1.5
3	Lake Batyo Catyo	2,946	877	1,770	29.8	60.1	2.0
4	Lake Buloke	23,048	4,932	7,185	21.4	31.2	1.5
5	Donald	1,263	1,161	1,763	92.0	139.6	1.5
6	Cope Cope Lakes	4,274	2,220	3,349	51.9	78.4	1.5
7	Box Swamp	843	465	657	55.1	78.0	1.4
8	Jesse Swamp	1,456	2,566	3,222	176.2	221.3	1.3

Table 5: Area of intervention (ha) by percentile required to maintain a depth-to-watertable of 1.5 metres.

		Percentile of asset area						
Id	Asset name	100	95	90	80	70	60	50
Intervention area (ha) required based on locating vegetation using the proximity approach (discharge strategy)								
1	York Plains	E	868	381	0	0	0	0
2	Avon Plains Lakes	E	E	E	4,068	286	0	0
3	Lake Batyo Catyo	E	E	6,195	864	141	0	0
4	Lake Buloke	E	E	E	7,922	0	0	0
5	Donald	E	E	192,793	7,114	1,984	854	419
6	Cope Cope Lakes	E	E	E	6,915	1,295	140	0
7	Box Swamp	731	363	145	0	0	0	0
8	Jesse Swamp	E	E	E	E	4,359	381	0
Intervention area (ha) required based on locating vegetation using the flux approach (recharge strategy)								
1	York Plains	E	74,267	55,367	0	0	0	0
2	Avon Plains Lakes	E	E	E	105,000	34,833	0	0
3	Lake Batyo Catyo	E	E	165,000	114,409	23,063	0	0
4	Lake Buloke	E	E	E	120,465	0	0	0
5	Donald	E	E	218,115	185,667	159,091	137,275	115,296
6	Cope Cope Lakes	E	E	E	82,939	65,071	19,667	0
7	Box Swamp	39,374	34,735	28,457	0	0	0	0
8	Jesse Swamp	E	E	E	E	45,769	27,323	0

E = target can only be met by engineering option.

Table 6: Area of intervention (ha) by percentile required to maintain a depth-to-watertable of 2.0 metres.

		Percentile of asset area						
Id	Asset name	100	95	90	80	70	60	50
Intervention area (ha) required based on locating vegetation using the proximity approach (discharge strategy)								
1	York Plains	E	5,752	3,467	2,268	1,132	396	0
2	Avon Plains Lakes	E	E	E	E	E	682	0
3	Lake Batyo Catyo	E	E	E	E	71,786	2,866	890
4	Lake Buloke	E	E	E	E	9,156	0	0
5	Donald	E	E	E	E	8,734	4,617	2,926
6	Cope Cope Lakes	E	E	E	E	31,111	5,509	1,300
7	Box Swamp	1,238	716	521	243	0	0	0
8	Jesse Swamp	E	E	E	E	E	8,469	942
Intervention area (ha) required based on locating vegetation using the flux approach (recharge strategy)								
1	York Plains	E	104,430	102,650	94,245	84,913	63,452	0
2	Avon Plains Lakes	E	E	E	E	E	71,517	0
3	Lake Batyo Catyo	E	E	E	E	180,841	162,485	150,503
4	Lake Buloke	E	E	E	E	117,829	0	0
5	Donald	E	E	E	E	193,182	172,753	162,990
6	Cope Cope Lakes	E	E	E	E	96,102	81,985	61,653
7	Box Swamp	41,150	40,056	38,364	30,877	0	0	0
8	Jesse Swamp	E	E	E	E	E	52,730	35,386

E = target can only be met by engineering option.

Table 7 – Area of intervention needed under perennial vegetation to maintain the depth-to-watertable of 1.5 metres beneath each high value asset and the comparative cost (present value over 20 years) of intervention options (native vegetation, lucerne, groundwater interception).

ID	Location	Asset Area (ha)	Percent change in the number of events the depth-to-watertable was less than 2m with 1,000 ha native revegetation under climate change	Native vegetation intervention area (ha) to maintain watertable at 1.5 metres across 90% of asset area	Lucerne intervention area (ha) to maintain watertable at 1.5 metres across 90% of asset area	Native vegetation cost ($M present value) to maintain watertable at 1.5 metres across 90% assets	Lucerne cost ($M present value) to maintain watertable at 1.5 metres across 90% assets	Engineering cost ($M present value) to maintain watertable at 1.5 metres	Cheapest Intervention	Asset value	Ranking of intervention based on asset value and cost
1	York Plains	3,868	3.1	381	592	1	0.5	131	Lucerne	VH	High
2	Avon Plains Lakes	2,069	2.1	Engineering	Engineering	-	-	131	Engineering	VH	Mod-High
3	Lake Batyo Catyo	2,946	1.9	6,195	6,116	19	5	117	Lucerne	H	Mod-High
4	Lake Buloke	23,048	3.1	Engineering	Engineering	-	-	681	Engineering	VH	Low
5	Donald	1,263	4.9	192,793	189,226	606	165	160	Engineering	VH	Mod-High
6	Cope Cope Lakes	4,274	1.6	Engineering	Engineering	-	-	294	Engineering	VH	Mod-High
7	Box Swamp	843	1.6	145	324	0.5	0.3	62	Lucerne	M	Mod
8	Jesse Swamp	1,456	2.0	Engineering	Engineering	-	-	340	Engineering	M	Low

Table 8 – Area of intervention needed under perennial vegetation to maintain the depth-to-watertable of 0.8 metres beneath each high value asset and the comparative cost (present value over 20 years) of intervention options (native vegetation, lucerne, groundwater interception).

ID	Location	Asset Area (ha)	Percent change in the number of events the depth-to-watertable was less than 2m with 5,000 ha native revegetation under climate change	Native vegetation intervention area (ha) to maintain watertable at 0.8 metres across 90% of asset area	Lucerne intervention area (ha) to maintain watertable at 0.8 metres across 90% of asset area	Native vegetation cost ($M present value) to maintain watertable at 0.8 metres across 90% of asset area	Lucerne cost ($M present value) to maintain watertable at 0.8 metres across 90% of asset area	Engineering cost ($M present value) to maintain watertable at 0.8 metres	Cheapest Intervention	Asset value	Ranking of intervention based on asset value and cost
1	York Plains	3,868	5.3	8	11	0.3	0.01	23	Lucerne	VH	High
2	Avon Plains Lakes	2,069	4.8	24	23	0.8	0.02	51	Lucerne	VH	Mod-High
3	Lake Batyo Catyo	2,946	3.3	9	7	0.3	0.01	24	Lucerne	H	Mod-High
4	Lake Buloke	23,048	3.8	13	15	0.4	0.01	178	Lucerne	VH	Low
5	Donald	1,263	28.2	1,053	2,321	3.3	2.0	49	Lucerne	VH	Mod-High
6	Cope Cope Lakes	4,274	2.7	34	33	0.1	0.03	76	Lucerne	VH	Mod-High
7	Box Swamp	843	2.3	51	53	0.2	0.05	14	Lucerne	M	Mod
8	Jesse Swamp	1,456	1.8	10,746	11,037	33.8	9.6	176	Lucerne	M	Low

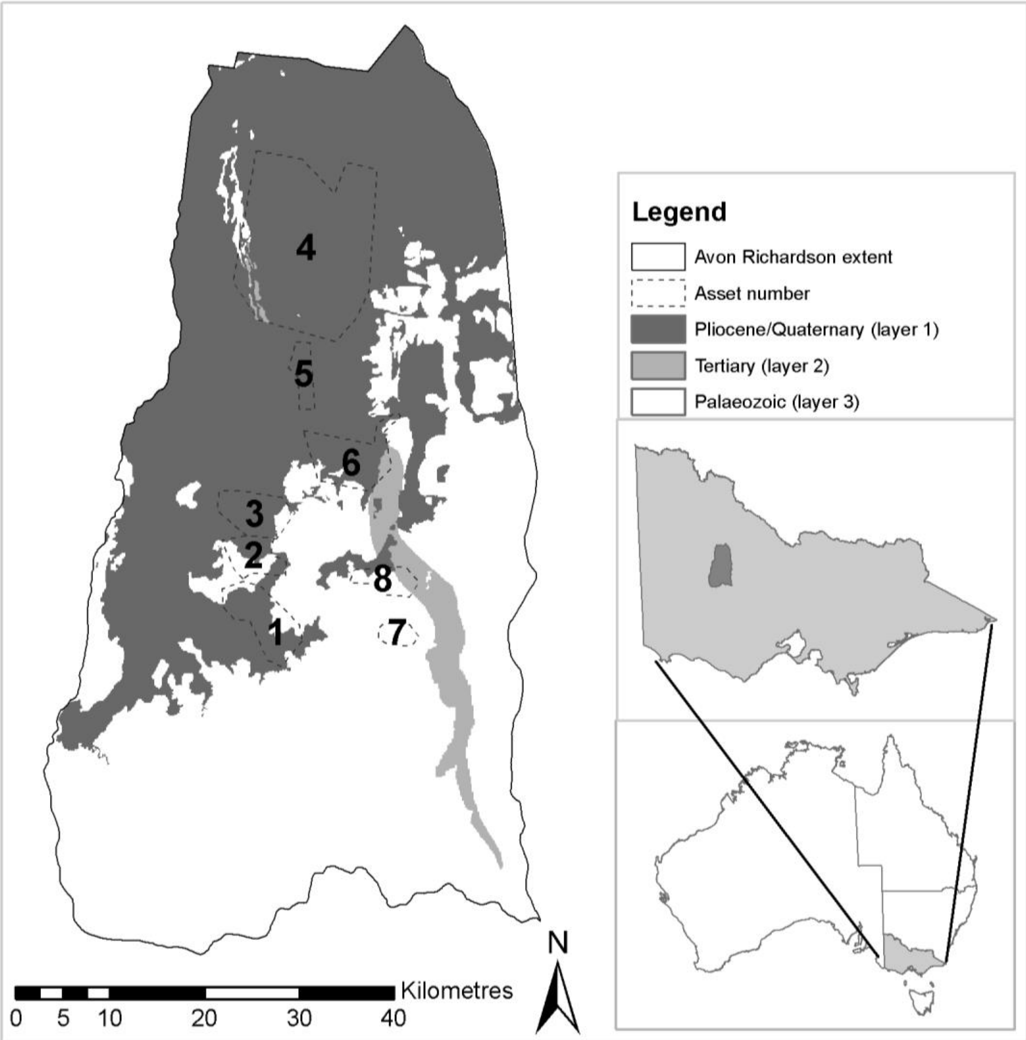


Figure 1: Location of the Avon Richardson catchment in Victoria, Australia. Also shown are the locations of high-value assets in relation to groundwater extent and conceptualisation. Assets 1-8 are respectively York Plains, Avon Plains Lakes, Lake Batyo Catyo, Lake Buloke, Donald, Cope Cope Lakes, Box Swamp and Jesse Swamp

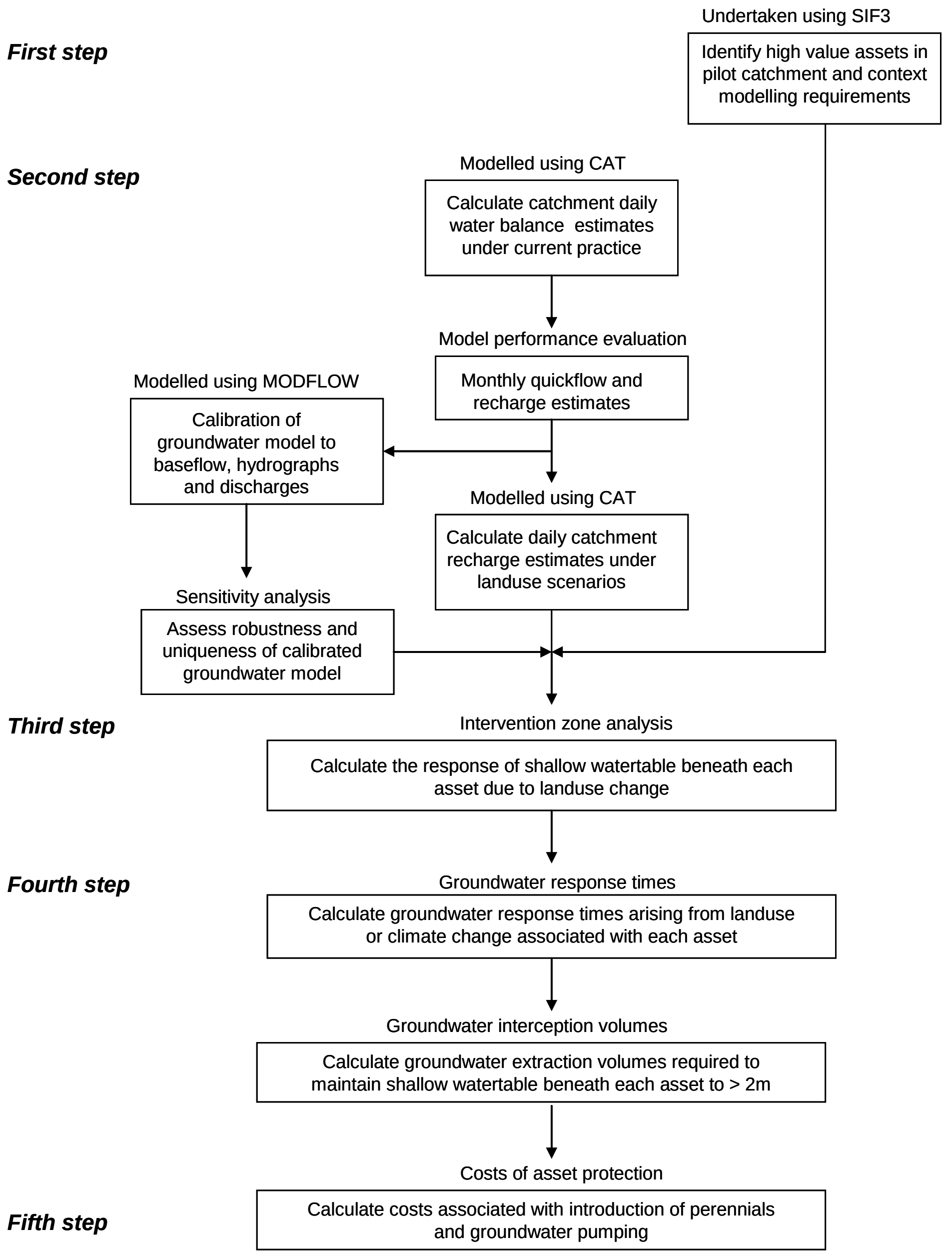


Figure 2: Steps undertaken to assess whether high-value assets can be protected from dryland salinity in Australia.

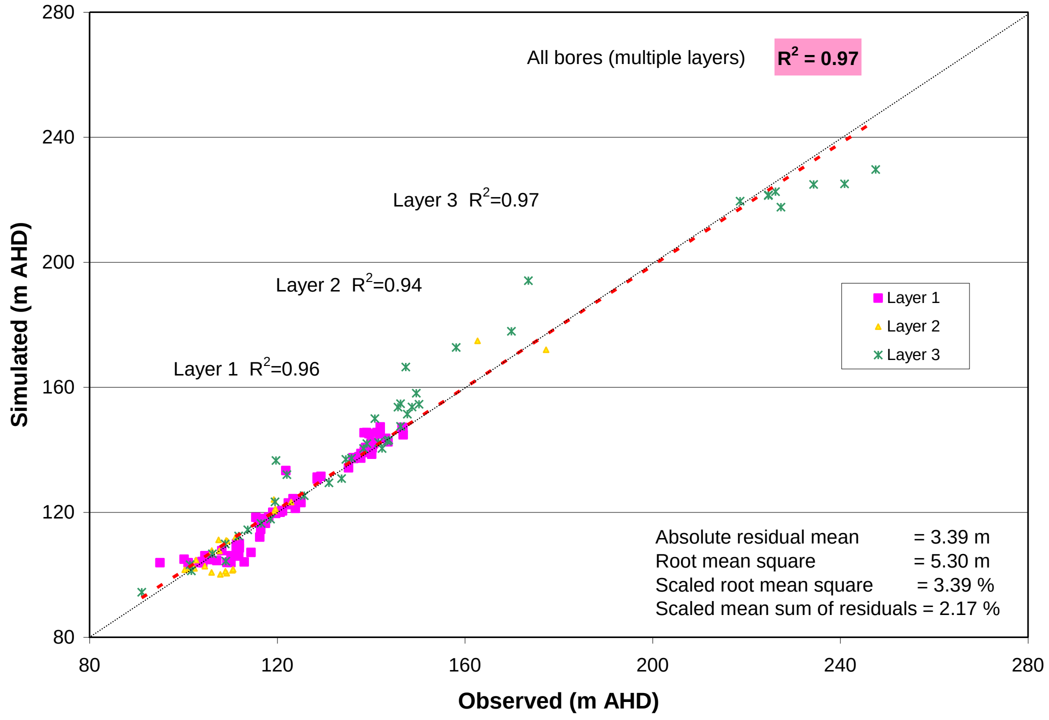


Figure 3: Correlation between observed and simulated 1995 potentiometric head for observation bores in each modelled aquifer.

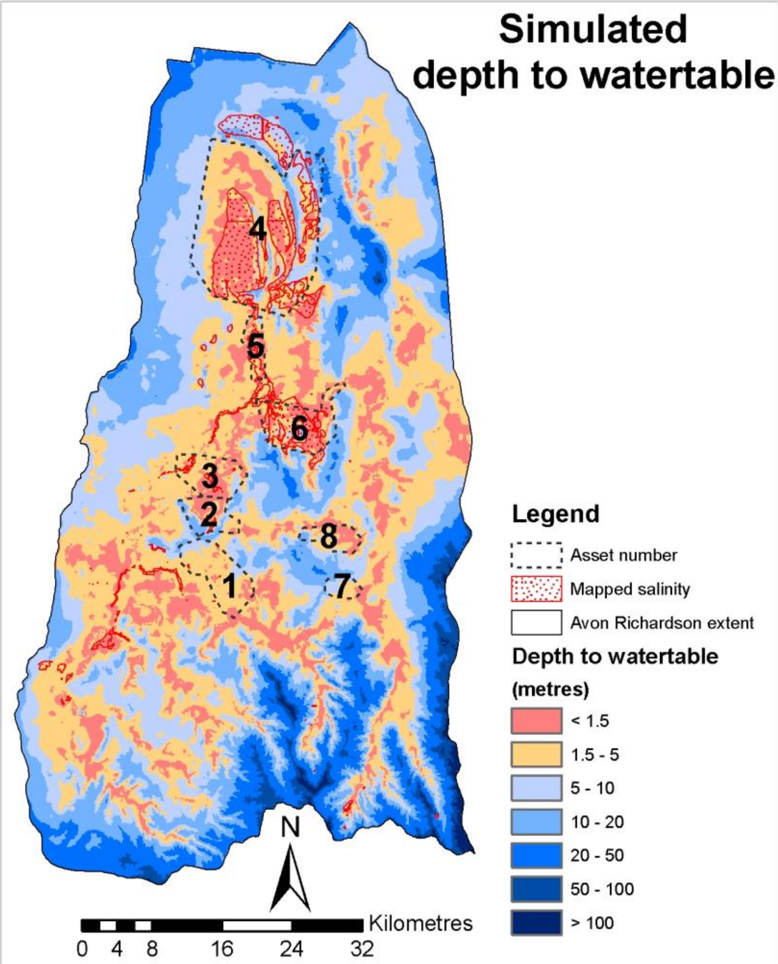


Figure 4: Simulated 1995 depth-to-watertable for the Avon Richardson catchment (Australia). Also shown are the mapped salinity area (hatched) and asset sites (dashed boundaries and numbers).

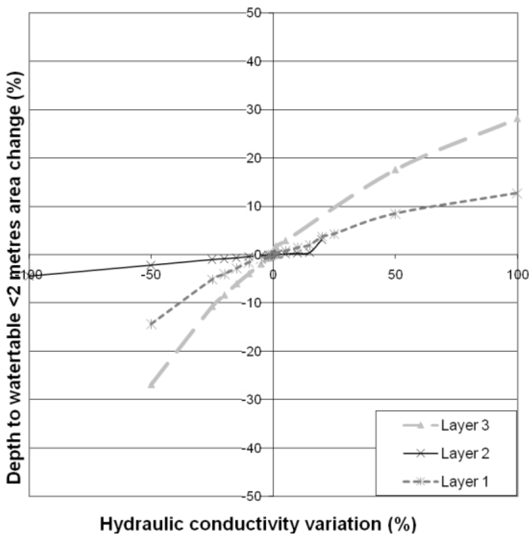


Figure 5: Sensitivity analysis of the impact of varying lateral conductivity of each aquifer on watertable depth for the groundwater model developed for the Avon Richardson catchment.

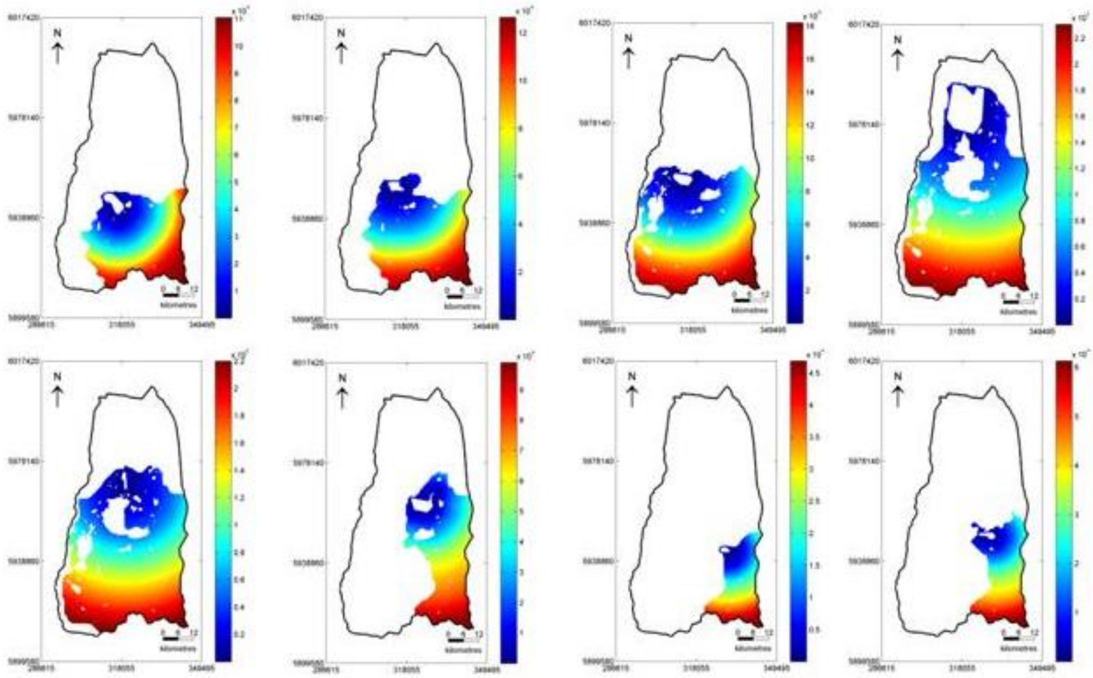


Figure 6: Up-gradient intervention area for the eight high value assets in the Avon Richardson catchment. From top left to right and top of page to bottom of page the assets are York Plains, Avon Plains Lakes, Lake Batyo Catyo, Lake Buloke, Donald, Cope Cope Lakes, Box Swamp and Jesse Swamp. The colour coding is based on zones closest to asset boundary (blue) to zones furthest away from asset boundary (red).

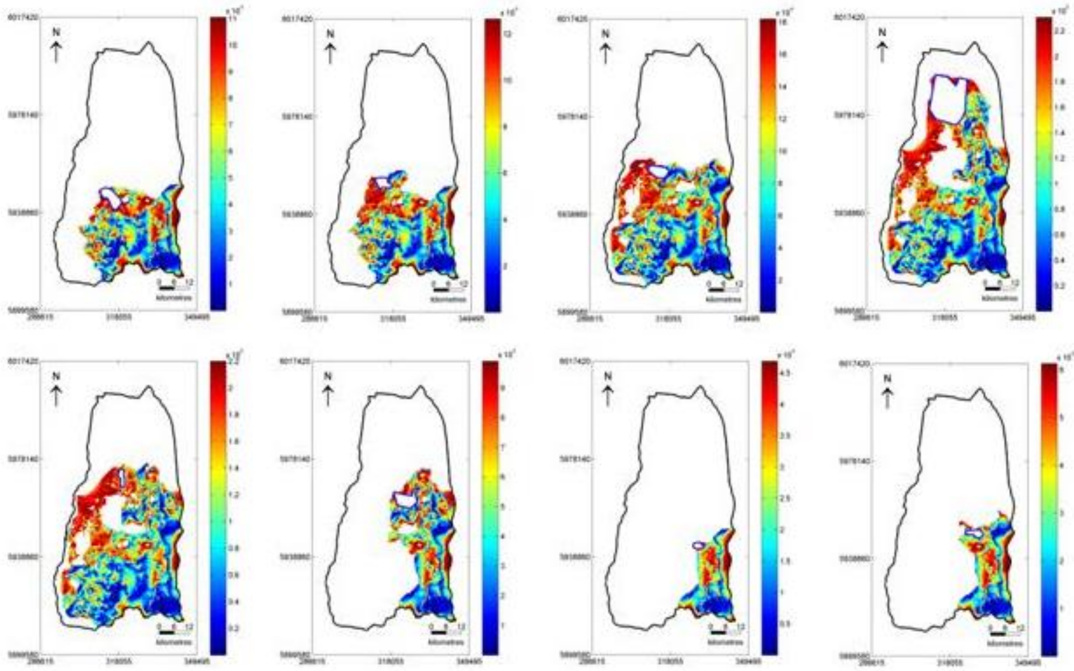
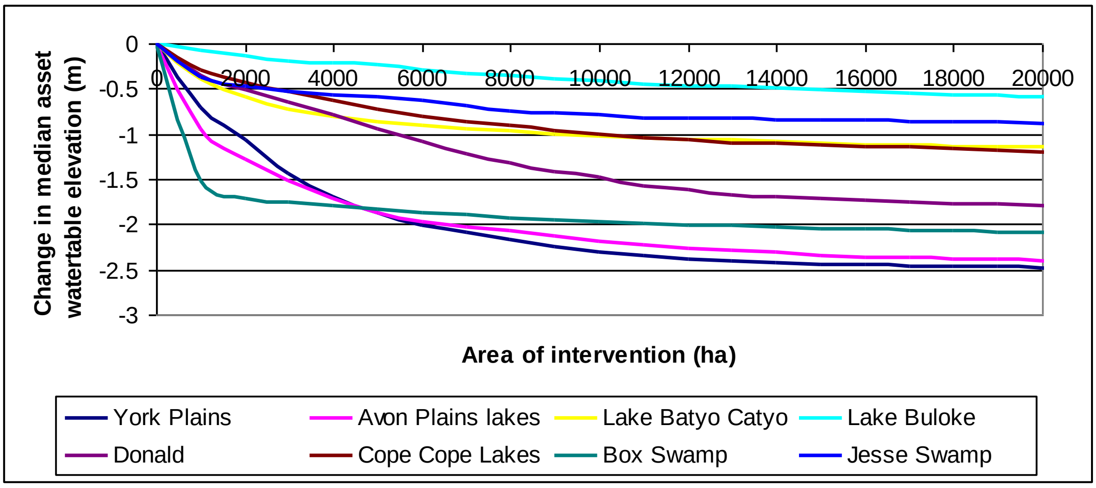


Figure 7: Up-gradient intervention area for the eight high value assets in the Avon Richardson catchment. From top left to right and top of page to bottom of page the assets are York Plains, Avon Plains Lakes, Lake Batyo Catyo, Lake Buloke, Donald, Cope Cope Lakes, Box Swamp and Jesse Swamp. The colour coding is based on zones of highest groundwater flux (blue) to zones of low groundwater flux (red).

a)



b)

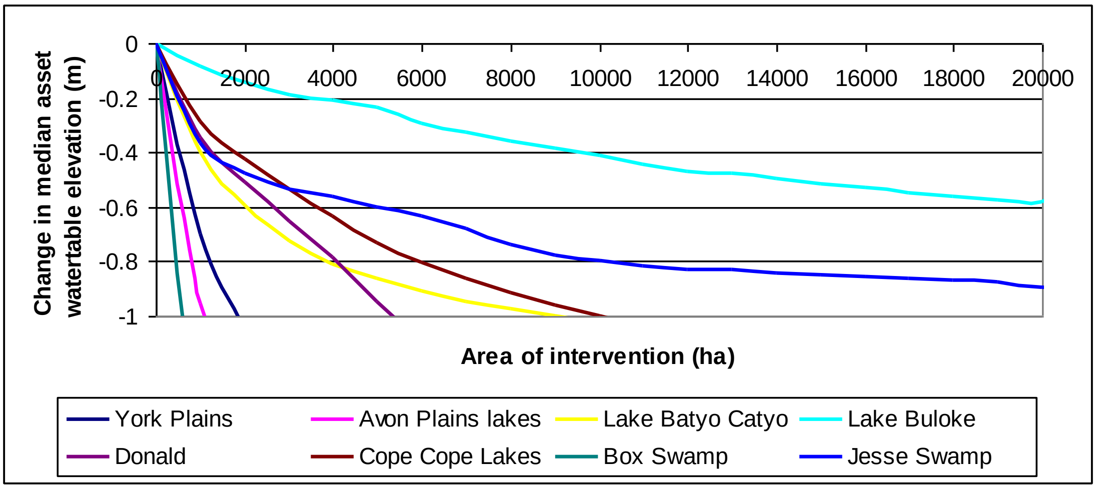


Figure 8: Change in median watertable elevation within each high value asset as a function of restoration of native vegetation in order of closest proximity to each asset. Figure 8(a) shows the median water table change to -3 metre whereas Figure 8(b) is an expanded plot shown to a water table change of -1 metre.

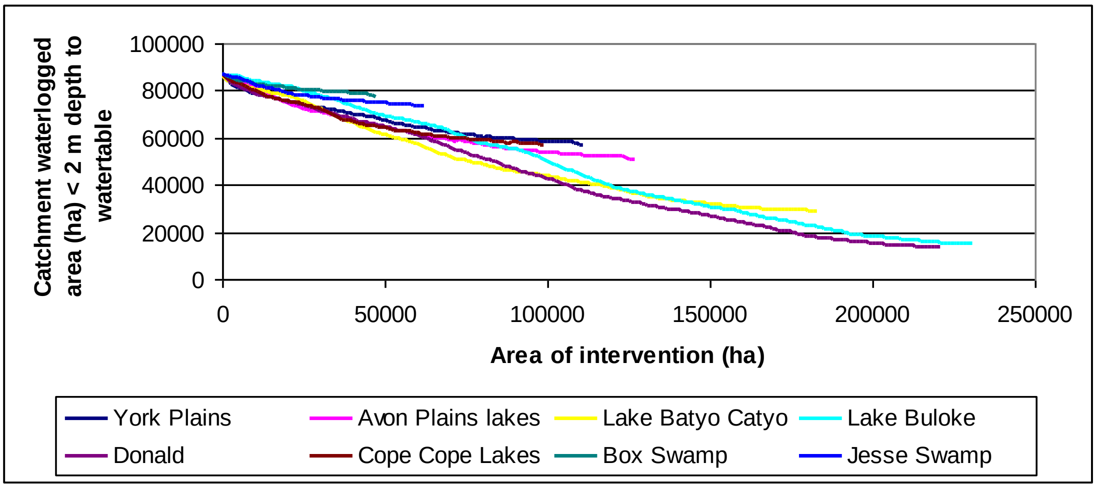
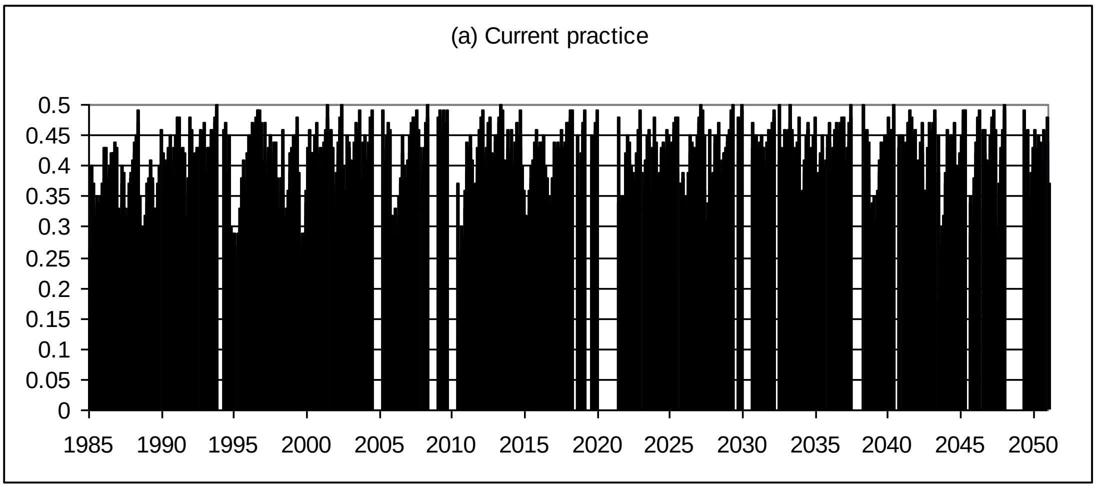
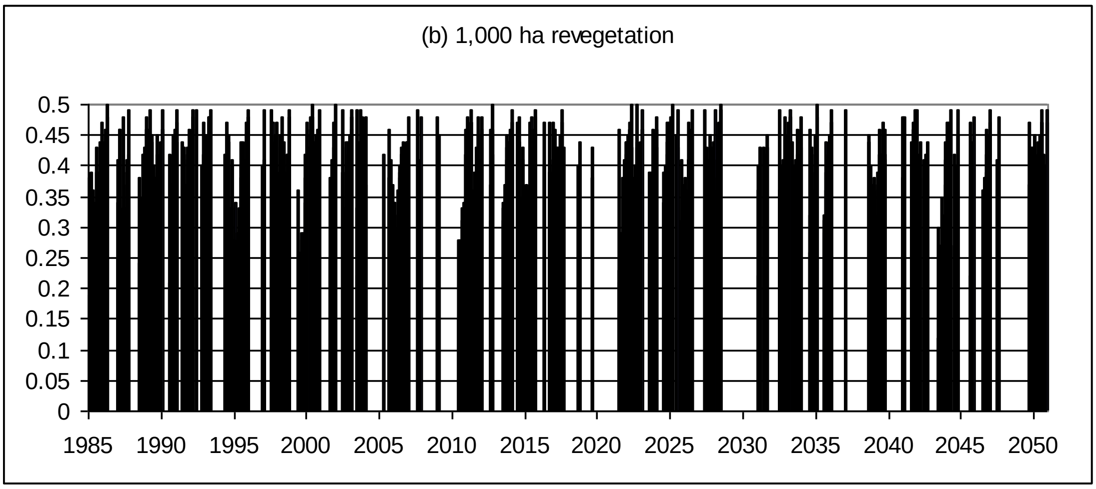


Figure 9: Change in saturated area (< 2 metre watertable depth) within the Avon-Richardson catchment as a function of restoration of native vegetation within the up-gradient contributing area of each high value asset.





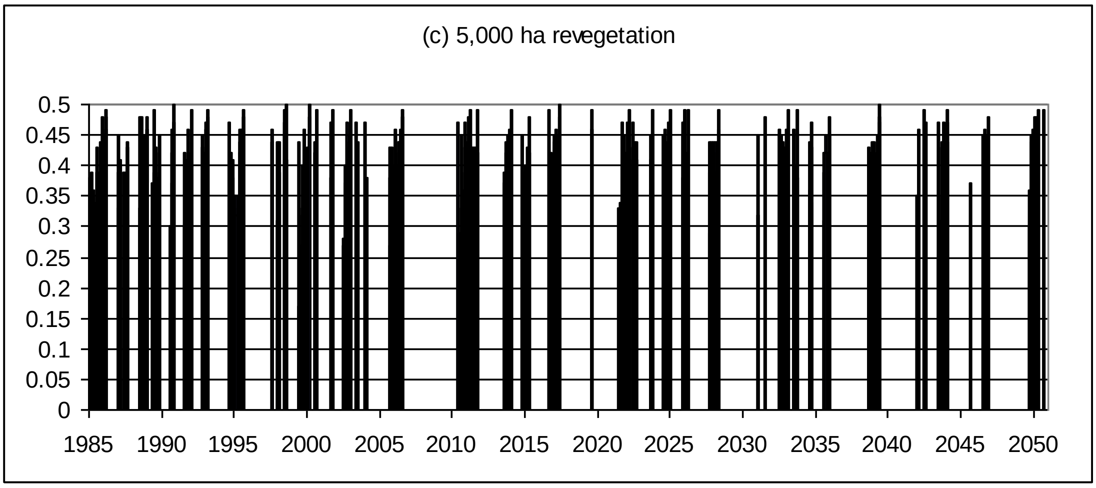


Figure 10: Median depth to shallow watertable less than 0.5 metre for York Plains under (a) current practice, (b) 1,000 ha native revegetation and (c) 5,000 ha native revegetation for the period 1985-2050.

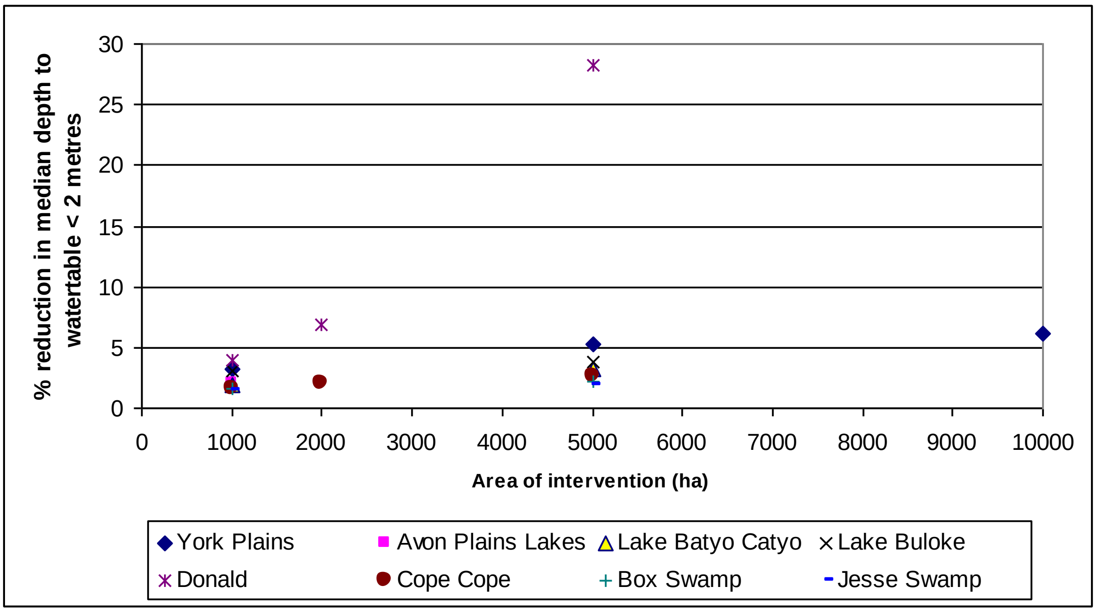
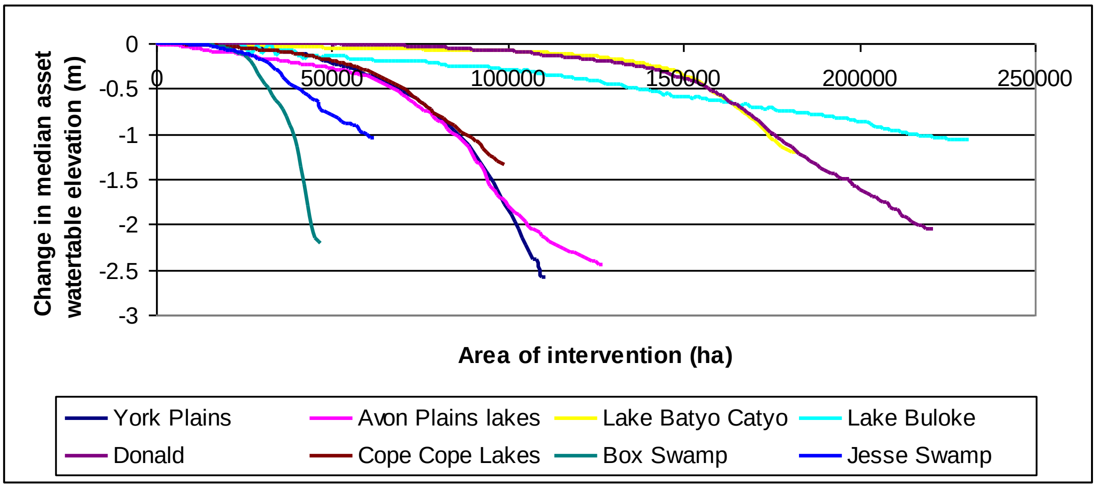


Figure 11: Percent reduction in the proportion of time that the median depth-to-watertable is less than 2 metre beneath each asset under climate change conditions for the period 1985-2050.

a)



b)

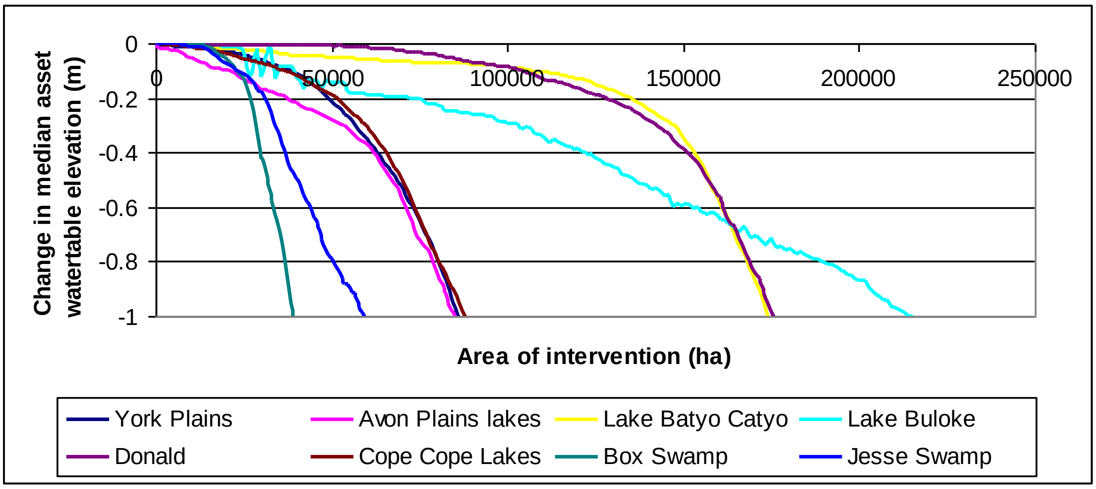


Figure 12: Change in median watertable elevation within each high value asset as a function of restoration of native vegetation in the associated up-gradient contributing area of that asset assuming the order of intervention in the landscape is based on the magnitude of groundwater flux across a groundwater cell. Figure 12(a) shows the change in elevation to 3 metres whereas Figure 12(b) is an expanded plot to a depth of 1 metre.